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Foreword

This proceeding presents papers that were given at the 4th International SWAT Conference, which convened at UNESCO-IHE, Delft, The Netherlands.

The focus of this conference was to allow an international community of researchers and scholars to discuss the latest advances in the use of the SWAT (Soil Water Assessment Tool) model to assess water quality trends.

The SWAT model was developed by researchers Jeff Arnold of the United States Department of Agriculture – Agricultural Research Service (USDA-ARS) in Temple, Texas and Raghavan Srinivasan, Director of the Texas A&M University Spatial Sciences Laboratory.

SWAT is a comprehensive computer simulation tool that can be used to simulate the effects of point and nonpoint source pollution from watersheds, in the streams, and rivers. SWAT is integrated with several readily available databases and Geographic Information Systems (GIS).

Because of the versatility of SWAT, the model has been utilized to study a wide range of phenomena throughout the world. At the same time, the research community is actively engaged in developing new improvement to SWAT for site-specific needs and linking SWAT results to other simulation models.

This conference provided an opportunity for the international research community to gather and share information about the latest innovations developed for SWAT to discuss challenges that still need to be resolved.

This proceedings includes papers covering a variety of themes, including agricultural management, sediment modeling, climate change, integrated modeling, hydrological processes, and ecological and water quality processes. In addition to papers presented at the conference, posters shown at the conference are also included in this proceeding.

The organizers of the conference- Ann van Griensven and Raghavan Srinivasan- want to express thanks to organizations and individuals who made this conference successful. Organizations that played a key role in this conference include USDA-ARS, Texas A&M University, and UNESCO-IHE. We also thank ESRI for their involvement and cooperation with the conference.

To learn more about SWAT visit: http://www.brc.tamus.edu/swat/ or contact Raghavan Srinivasan at r-srinivasan@tamu.edu

Conference Objective

Soil and Water Assessment

Natural watershed systems maintain a balance between precipitation, runoff, infiltration, and water which either evaporates from bare soil and open water surfaces or evapotranspires from vegetated surfaces, completing the natural cycle. The understanding of this hydrological cycle at a watershed scale, and the fate and transport of nutrients, pesticides and other chemicals affecting water quality is essential for development and implementation of appropriate watershed management policies and procedures.

In recent years, application of models has become an indispensable tool for the understanding of the natural processes occurring at the watershed scale. As the natural processes are more and more modified by human activities, application of integrated modeling to account for the interaction of practices such as agricultural management, water removals from surface bodies and groundwater, release of swage into surface and sub-surface, urbanization, etc., has become more and more essential.

The program SWAT (Soil and Water Assessment Tool) due to its continuous time scale, distributed spatial handling of parameters and integration of multiple processes such as climate, hydrology, nutrient and pesticide, erosion, land cover, management practices, channel processes, and processes in water bodies has become an important tool for watershed-scale studies.

The fourth international SWAT conference to be at UNESCO-IHE, Delft The Netherlands will devote itself to discussions around the application of SWAT to watershed problems world wide. The five-day program will include two days of hands on learning of the SWAT program at the introductory and advanced levels, followed by three days of conference covering a variety of topics related to watershed modeling such as hydrology, water quality, land use management, erosion, and system analytic topics in calibration, optimization, and uncertainty analysis techniques.

Scientists associated with research institutes and those associate with government agencies and center for policy making are encouraged to take part in this international conference in order to become familiar with the latest advances and developments in the area of watershed scale modeling and applications.

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Using SWAT for BMP Implementation in the Pike River Watershed, Canada

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ABSTRACT

Hydrological and erosion processes as well as phosphorus transport were modelled with the Soil and Water Assessment Tool (SWAT-2000) for the Pike River watershed (630 km^2), an important tributary of the Missisquoi Bay in the north-eastern part of trans-border Lake Champlain (Canada-USA).

The model's calibration and validation was based on data from four hydrometric and water quality stations at the outlet of watersheds of varying size (7 - 600 km^2). The model-derived sediment and phosphorus loads showed a clear spatial pattern: under present soil and crop management methods, over 50% of modelled phosphorus loads originated over roughly 10% of the watershed's area. Based on these results, agri-environmental scenarios were devised in order to reduce by 41% the phosphorus loads reaching the Missisquoi Bay. These are presented in this paper.

KEYWORDS: SWAT, sediment, phosphorus, BMP.

Introduction

In recent years, in spite of the efforts made for enhancing water quality in the Missisquoi Bay, recurrent algal blooms triggered by an excess of phosphorus still cause important economic losses to the region and represent a threat to human health. In 2002, the province of Quebec and the state of Vermont have therefore reached an agreement seeking a 57,6 t/yr (41%) reduction in phosphorus loads reaching the bay.

In order to understanding the global dynamics of phosphorus transport in the environment, the Research and Development Institute for the Agri-Environment (IRDA), in collaboration with universities and the government has participated in several multi-disciplinary studies in this region. Having now extensively characterized the Pike River watershed, at plot scale (Michaud and Laverdière, 2004), field-scale (Enright and Madramootoo, 2004), meso-scale (6-11 km^2) (Michaud et al., 2004a; 2005) and macro-scale (630 km^2) (Deslandes et al., 2004), answering questions pertaining to the potential efficacy of BMPs in reducing P loads can now be done quickly, cheaply and over a greater time-scale through modelling. Given its ability to simulate the long-term effects of different land use management scenarios on water, sediment and nutrient transport over large heterogeneous watersheds, SWAT-2000 was chosen to devise cropping systems and land development scenarios that could meet target P-loads set by the Quebec-Vermont agreement.

This paper presents a brief summary of the calibration and validation results for the Pike River watershed. The results of the agri-environmental management scenarios are also presented here. The relative effectiveness of selected best management practices was evaluated in comparison to a reference scenario, representing current agricultural production systems employed in the watershed. A complete description of the methods, materials and results can be found in Deslandes et al. (2007) and Michaud et al. (2007).

Materials and Methods

Site Description

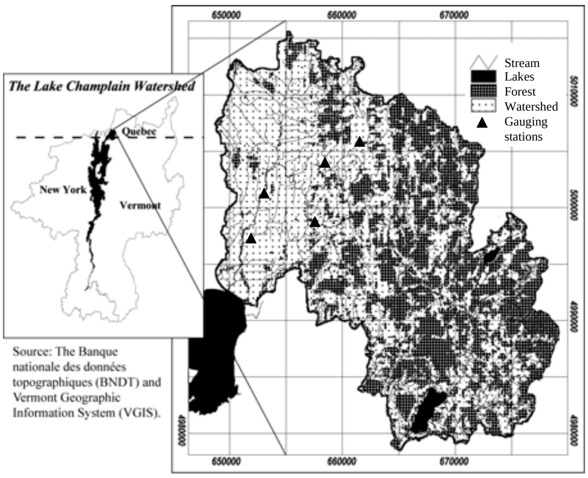
The Pike River is among the main watercourses discharging into the Missisquoi Bay, a spur of Lake Champlain extending in Canadian territory (*Figure 1*). The Pike River watershed (630 km^2) has been identified as one of the principal contributors of P to the bay (Hegman et al., 1999).

Spanning the Appalachian piedmont (elevation 50 to 710 m, mean slope 5°), the watershed's upstream region is ill suited for intensive agriculture. Only 35% of the region's area is devoted to agriculture, 22% to perennial forage crops and 13% to annual crops. The downstream portion of the basin includes the majority of the watershed's agricultural activities as well as its industrial core. Elevation ranges from 20 to 130 m, with flatter slopes (0.6°). Three quarters of the downstream region is cultivated, and of cultivated lands, roughly 20, 30 and 50% respectively, are devoted to hay crops, perennial forages, and field crops.

Data sources

Daily precipitation and temperature data for the 1997-2003 study period were drawn from three weather stations, located around the periphery of the watershed. Overall, annual precipitation during the study period remained within 11% of the norm, while mean annual temperatures matched long term means. A 30 m resolution digital elevation model (DEM) was developed from a multi-source database (Deslandes et al. 2004). A Landsat 7 ETM+ image served for land use mapping (Cattaï, 2004), while soil-type mapping drew from a number of sources (Talbot, 1943; Cann et al., 1946; USDA-NCRS, 1999). The exact position of subsurface drainage in the basin was unknown. Therefore, the area under annual field crops (60%) was taken as being drained. This is comparable to tile drained area inventoried in an experimental watershed in the study region (Michaud et al., 2004a, b).

Figure 1 – Lake Champlain and the Pike River Watershed.



Five hydrometric stations were used in the calibration and validation of the model (Table 1). Only stream flow was available at two stations located on the main channel of the Pike River (PR_{up}, PR_{down}). Within the Pike River watershed, three smaller (7-11 km^2) experimental watersheds of the Walbridge Creek and Beaver Brook were also monitored for suspended solids and P loads in addition to streamflow.

SWAT-2000 set-up

SWAT was set-up in order to keep the maximum information. As such, all soil types and land uses were kept in the process of Hydrological

Response Unit (HRU) definition. Overall, 3885 HRU, of which 2253 were cultivated, were defined within 99 subwatersheds.

Table 1. Hydrometric stations in the Pike River watershed.

Station ID.	Associated Watershed	Data availability and modelling time frame	Description	Measurements
PR_{up}	Pike River upstream (385 km^2)	1997-2003	Rolling landscape, mainly wooded	Stream flow
PR_{down}	Pike River downstream (561 km^2)	Nov. 2001-Dec. 2003	Rolling and forested lands of the watershed's headwaters and a part of the flat, agricultural lands	Stream flow
WC_{up}	Walbridge Creek (6.3 km^2)	Nov. 2001-Dec. 2003	Rolling and agricultural (61%), typical of the Appalachian piedmont	Stream flow, sediments, and P
WC_{down}	Walbridge Creek (7.9 km^2)	Nov. 2001-Dec. 2003	Flat and agricultural (63%), typical of St. Lawrence lowlands	Stream flow, sediments, and P
Beaver	Beaver Brook (11 km^2)	1997-2003	Flat and agricultural (97%) typical of St. Lawrence lowlands	Stream flow, sediments, and P

The model was ran from 1997 to 2003. The hydrology was calibrated daily using flow at two gauging stations: PR_{up} and PR_{down}. The year 2000 was used for the calibration of PR_{up} and 2002 for PR_{down}. The year 2003 was used as the validation period for both gauging stations. Daily flows and monthly sediment and phosphorus loads, for the period extending from November 2001 to December 2003, were also calibrated at WC_{up} and WC_{down}. The calibrated parameters governing sediment and phosphorus exports for the WC_{up} were applied to upstream portion of the Pike River watershed and those from the WC_{down} stations were applied to the downstream portion of the Pike River. To overcome the lack of validation data for these stations, a validation of the retained parameters was done with the data from the Beaver Brook from March 1997 to September 2002.

Calibration was done manually and a visual consideration of the hydrograph was used in conjunction with the Nash-Sutcliffe Coefficient (N-S), the Pearson correlation coefficient (R) and the Percent Deviation (D_v) for assessing the accuracy of the predictions of the model.

Cropping systems, nutrient management and agrienvironmental scenarios

A **reference management scenario** was built for the 2253 agricultural HRUs. The spatial distribution of crops was maintained during the modelling period. Sowing, tillage, and fertilizer application dates were adjusted according to crop type, probable management schedule, and the precipitation series. Based on standard practices in the region, fall ploughing and spring harrowing was maintained for annual crops. Broadcast manure application date in Spring was set after the first 72 h precipitation-free period starting last week of April and a 48 h precipitation-free period had to occur prior to secondary tillage, sowing and fertilizer application. Phosphorus inputs were based on annual spendings in inorganic fertilizers and on the types of livestock and crops produced. The spatial distribution of fertilizer inputs was established using management data collected at the field-scale (Michaud, 2004a,b). Farm manure inputs were allocated 45% to preplant, 36% to post-emergence, and 16% to fall applications. Inorganic phosphorus fertilizer was applied in a single operation at sowing.

The effect of different best management practices (BMPs) on the water balance, sediment and P exports were modelled for the study period (2000-2003) over the entire Pike River watershed. **Timely manure incorporation**, **cover cropping**, **conservation tillage**, **riparian buffers and structural runoff control** were selected since their feasibility has been documented in field studies in the North-East (Gangbazo et al. 1997; Kleinman et al. 2005; Angers et al. 1997; Duchemin and Majdoub 2004). SCS curve numbers and Manning’s surface roughness were the two parameters, notwithstanding the changes made in the management operations, that were modified in the scenarios described below. A common modelling period for the reference and agri-environmental scenarios permitted a comparison of the relative effect of selected BMPs on non-point source sediment and phosphorus exports. Individual BMP's effects on water, sediment and phosphorus yield were first investigated. Then, the influence of mixed scenarios, combining various levels of implementation of BMPs at HRU scale, was evaluated.

In order to quantify the effect of different **lag times in soil incorporation of farm manures** on P exports, spreading dates were modified so as to precede tillage by a single day. The simulated incorporation-lag in the BMP's scenario was set at ≤24 h for all manure P inputs. Cover cropping and no-till practices were also systematically subjected to a lag optimisation (<24 h). Comparing with the reference scenario (6 to 16 days lag; 4 to 87 mm rainfall), this first scenarios allowed to deduce the net effect of optimizing the incorporation.

The **cover crop scenarios** assumed the establishment of (i) a perennial legume or grass cover crop (CC_{per}), (ii) a small grain intercropped with red clover (IC_{sg+rc}), or (iii) a small grain followed by a late season cruciferous cover crop ($CC_{sg\rightarrow cr}$), on annual crops. Under the $CC_{sg\rightarrow cr}$ and IC_{sg+rc} scenarios, fertilizer applications were split 55% and 45% between pre-plant and mid-August post-harvest, respectively. The CC_{per} scenario followed the reference scenario's three dates of manure application, based on the date of hay cuttings.

Modelled **conservation tillage** scenarios were assigned according to the crop type and soil hydrological group. These include, for small grains, soya and corn: (i) no-till seeding (NT_{pe}) on soils of hydrological group A and B, (ii) reduced tillage with fall stubble ploughing (RT_f) on hydrological group C, and (iii) reduced tillage with spring stubble ploughing (RT_s) for group D. For no-till (NT_{pe}) seeding of corn, all manure applications were shifted to post-emergence burial while for small grains, they were limited to the spring, simulating a superficial burial. Fall manure spreading was maintained for RT_f, whereas for RT_s it was shifted to the spring, preceding stubble ploughing and post-emergence in summer.

Riparian buffers and catch-basin inlets were simulated empirically, by attributing pollutant trapping-efficiency coefficients (PTECs) to HRU's particulate P outputs. An overall PTEC of 25%, based on a study on the Beaver Brook watershed (Michaud et al. 2005) was used. In this watershed, catch basin inlets at the outlets of non-subsurface drained fields (50 structures per 10 km^2) were installed on the most hydrologically active parcels of land (42% of the total watershed area) and permanent buffer strips of roughly 3 m above the bank were established along 4 km of the waterway's main reaches. The relative trapping contributions of buffer strips and catch-basin inlets were set according to coefficients from literature and in proportion to lands draining to buffer strips (33%) or to catch-basin inlets (67%) (Duguet et al. 2002). The ensuing PTECs were 9% for buffer strips and 16% for and catch-basin inlets.

A combination of different BMPs were implemented on different targeted cropped HRUs. Three criteria were followed in the selection and spatial distribution of BMPs: (a) reaching a 41% decrease in annual mean total P loads, (b) prioritising BMP simulation to the HRUs with the greatest vulnerability to P exports, as drawn from the reference scenario, and (c) generating realistic scenarios while minimizing constraints and enhancing environmental benefits. A three-step cumulative approach was thus implemented: (i) the basic scenarios insure the protection of the watershed flood plains, the generalised implementation of a three meters wide buffer strips along waterways and reduced farm manures incorporation to less than 24 hr; (ii) The conservation practices scenarios implement conservation tillage and cover crops, without altering the crop rotation; and (iii) The crop substitution scenarios shift most P-loss vulnerable annual crops to small grains with a cover or catch crop. For all three types of mixed scenarios, the implementation of buffer strips and catch-basin inlets was incorporated in the simulations.

Results and Discussion

Calibration and validation results

Overall, the water balance appeared to be consistent with local agroclimatic conditions. The mean predicted annual water balance for 2000 to 2003 showed that of the 1154 mm yr^{-1} precipitation, roughly 560 mm was lost through evapotranspiration. Surface runoff, lateral flow, and tile flow accounted for 218, 39 and 180 mm yr^{-1}, respectively. Some 211 mm yr^{-1} contributed to shallow groundwater, while 60 mm were lost to deep aquifers. However, comparing tile flow predictions to measurements made on instrumented fields in the Pike River watershed, Enright and Madramootoo (2004) suggested that tile drainage depths might be slightly underestimated.

SWAT satisfactorily predicted streamflow at the four stations within the Pike River watershed (Table 3), with most daily and monthly Nash-Sutcliffe coefficient (NSC) being above 0.50 and with correlation coefficient (r) over 0.7. Predicted flows were generally within 20% of the expected volumes, except at PR_{down}, which showed a 20% and 30% underestimate over the calibration and validation periods despite *r* and NSC values above 0.5 and 0.7, respectively. This underestimate is essentially attributable to exceptionally unseasonable near-zero and above-zero temperatures during the winter periods. Under such conditions, SWAT has difficulty distinguishing between rainfall and snowfall. Given the important magnitude of peak flows associated with winter thaws, which represents more than 20% of the annual water yield, predictions of water depths were strongly affected.

Table 3. SWAT'S stream flow daily and monthly closeness of fit indicators for the calibration and validation period at four hydrometric stations.

Watershed	Period	D_v	Monthly streamflow		Daily streamflow	
			r	NSC	*r*	NSC
PR_{up}	Calib: 04/1998 to 12/2000 and 05/2001 to 12/2002	-3%	0.93	0.85	0.75	0.56
	Valid: 01/2003 to 12/2003	-8%	0.97	0.91	0.71	0.50
PR_{down}	Calib: 11/2001 to 12/2002	-20%	0.82	0.52	0.76	0.55
	Valid: 01/2003 to 12/2003	-33%	0.88	0.60	0.86	0.64
WC_{up}	Calib: 11/2001 to 12/2002	-9%	0.74	0.49	0.77	0.58
	Valid: 01/2003 to 12/2003	+19%	0.93	0.64	0.78	0.44
WC_{down}	Calib: 11/2001 to 12/2002	+3%	0.78	0.60	0.77	0.59
	Valid: 01/2003 to 12/2003	-15%	0.94	0.85	0.82	0.66

SWAT's ability to discriminate the hydrologic response between HRUs was verified at HRU scale by analysing the contribution of surface runoff, lateral, tile and groundwater flow to the water yield for two corn HRUs (not shown) having significantly different physical properties. As expected, the well drained sandy-gravely loam showed much greater surface runoff (143 mm vs. 50 mm) than the sandy loam. On both sites the main runoff events occurred in the spring or fall, when the soil was either frozen or saturated, as well as during the spring snowmelt, which accounted for 60 to 80% of the annual predicted runoff. With respect to subsurface flows, tile drainage represented 34% of total runoff for the HRU on the sandy loam and 63% on the well drained sandy-gravely loam. The scarcity of relevant measurements prevents a full validation of the model's relative allocation to lateral flow, shallow groundwater, and tile flow at the HRU or field scale. However, recent data based on the separation of the hydrograph using the electrical conductivity of water at the outlet of three subwatersheds (WC_{up}, WC_{down} and Beaver) shows that roughly 55 to 75 % of the flow transits below ground level. Although not shown here, results at the HRU scale over the entire Pike River watershed present a high variability, with tile flow alone representing from 2 to 78% (average 40%) of the total water yield. Moreover, studies on two fields located in the Pike River watershed highlighted the importance of tile flow. On these two fields, tile flow represented between 83% to 90% of the annual water yield, defined as the sum of surface runoff and tile flow (Simard, 2005; Gollamudi 2006).

The calibrated SWAT model was also able to satisfactorily predict (0.82<*r*<0.88; 0.55<NSC<0.76) monthly sediment and P loads at the outlet of the 6-8 km^2 Walbridge experimental watersheds (Table 4). The results show that exported sediments remained within 10% of the measured data and phosphorus loads within 13% at both WC_{up} and WC_{down} watersheds. Overall, the model tended to underestimate sediment and P exports during winter and autumn, while overestimating in early spring and summer. This reflects in large part the modelled watershed's hydrology.

Table 4. SWAT's monthly sediment and phosphorus exports closeness of fit indicators for the calibration and validation subwatersheds.

Watershed	Period	Sediment			Total P Exports		
		R	NS	D_v %	R	NS	D_v %
WC_{up}	Calibration: 11/2001 to 12/2002	0,86	0,70	10%	0,88	0,76	-13%
WC_{down}	Calibration: 11/2001 to 12/2002	0.82	0.55	-4%	0.87	0.73	7%
Beaver	Validation: 03/1997 to 09/2002	0.92	0.84	-8%	0.94	0.87	-20%

For validation purposes, even though the model was not calibrated on the Beaver Brook subwatershed, the statistical indices showed SWAT's efficiency to reproduce monthly sediment (*r*=0.92, NSC= 0.84, *Dv*=-8%) and P exports (*r*=0.94, NSC= 0.87, *Dv*=-20%) between March 1997 and September 2002. The complete validation of SWAT would have required more spatial and temporal water quality data, especially at the Pike River downstream station ($PR_{\mathbf{down}}$). Unfortunately, only sparse data was available at that station.

The analysis of daily sediments and P exports at the outlet of the Walbridge Creek experimental watersheds, highlights the very episodic nature of hydrologic events leading to erosion and P transport. For example, 80% of sediment loads predicted in 2002 arose during short periods totalizing 28 days or 8% of the year, while 80% of the P was exported over a total of 67 days (17% of the year), mainly during significant spring and fall hydrological events. Land use patterns and topography are also involved in creating a significant spatial gradient in field-scale-modelled erosion rates. Indeed, roughly 50% of the erosion modelled was associated with less than 10% of the watershed's total area. Similarly, around 13%, of the watershed contributed to 50% of the total P exports. This episodic nature of modelled P fluxes parallels observations made on a number of instrumented agricultural watersheds in the region (Michaud et al., 2004a, b; Meals, 2004).

Overall, SWAT indicated the prominence of particulate P exports (67% of total P) across the Pike River watershed, and particularly over erosion-prone zones (corn, soybean, cereals). This speciation is consistent with those observed on experimental watersheds (Michaud et al., 2004a, b; Sharpley et al., 1992) where particulate P forms account for roughly 60% to 90% of the exported loads. In less erodible lands (grassland/pastures, forest) however, soluble forms of P dominated.

Results of Agri-Environmental Scenarios

The **elimination of delays in soil incorporation** of farm manures, compared to the reference scenario, resulted in a 3% overall watershed-wide decline in P exports. However, at the HRU scale, the magnitude of the reduction in delay is rather variable (0 to 70%; 0 to 1,3 kg P ha^{-1}) reflecting variations in rainfall received between manure spreading and soil incorporation, tillage practice, HRU-specific soil permeability, and the quantity of manure applied.

The influence of **cover crops** on surface runoff depths strongly influenced watershed scale P and sediment exports: 81 to 86% lesser sediment and 63 to 68% lesser phosphorus loads at the watershed's outlet were simulated when annual crops were shifted to prairie (CC_{per}) or cereal-clover (IC_{sg+rc}) intercropping. A slightly lesser decrease (68% and 55% for sediment and P, respectively) occurred with a small grain crop followed by a cruciferous cover crop ($CC_{sg\rightarrow cr}$). Such trends in predicted sediment and P loads can be attributed to the differences in soil and manure management practices, and associated effects on the mobilisation of soluble P. Indeed, the model's algorithms simulate a stratification of topsoil P in prairie soils, which translates to an enrichment of surface runoff waters with soluble P.

Comparing reductions in P and sediment loads amongst rising rates of targeted or random cover crop or inter-crop implementation, highlights the effect of spatial targeting of BMPs on the overall export balance. While the targeted conversion of the most vulnerable 10% of annual crop lands would result in a 21-24% and 27-31% decreases in predicted P and sediment loads at the watershed outlet, respectively, a similar but randomly assigned land conversion would only result in a third of the decrease in predicted P and sediment loads. The merits of spatial targeting of BMP implementation reflect the strong field-scale spatial clustering in vulnerability within the study watershed, which is often reported in watershed studies done elsewhere in North America (Sharpley et al. 1994; Daniel et al. 1994).

Compared to the reference scenario's conventionally tilled HRUs planted to annual crops, **reduced tillage** (RT_s and RT_f) led to a mean decreases in runoff depth, sediment loads and P loads of 42%, 46%, and 53%, respectively. Such predicted effectiveness concurs with North American reports of natural or simulated rainfall-impacted plot studies (McDowell et al. 1984; McGregor et al. 1999; Meyer et al. 1999; Franti et al. 1999; Dabney et al. 2000). Reduction in predicted P load largely reflects the reduction in predicted sediment yield, which are not only sensitive to the type of conservation practice governing the protection of the soil but also to the crop type and the hydrologic group. Similar to the projected effects of cover crops, the conversion of 1726 ha (10%) of the watershed's most vulnerable lands under annual crops to no-till (NT_{pe}) results in reductions in sediment and P exports of 5000 t yr^{-1} and 4.6 t yr^{-1}, respectively, corresponding to 10 and 16% of their respective total exports.

Phosphorus trapping coefficients of 9% and 16%, respectively for **buffer strips** and **runoff control structures** implemented on the watershed's 1726 ha (10%) of most vulnerable lands planted to annual crops resulted in a 4% simulated decline in total P exports at the watershed outlet.

Several **mixed agrienvironmental scenarios** were subsequently created to simulate the 41% reduction sought by the Quebec-Vermont agreement. The effects of those scenarios on the hydrological balance, erosion and P mobility are shown in Table 5. The implementation of easily acceptable and feasible scenarios was first investigated. Overall, the conversion of flood plains to prairie, combined with the optimisation of delays in manure incorporation, and the installation of buffer strips and runoff control structures, would translate into a 21.4% drop in simulated P exports at the Pike River watershed's outlet. This is however insufficient to meet the reduced target loads.

The second step in creating agri-environmental scenarios involved a shift to conservation tillage, a practice that is widely gaining acceptance from farmers. Theoretically, shifting all crops to conservation practices would result in a 47 and 35 % drop in sediment and P exports, respectively. However, targeting 100% of the cropped area is not technically feasible nor socially acceptable. For this reason, a random or targeted implementation over various portions of agricultural lands (10 or 50%) of conservation practices together with timely manure incorporation, conversion of flood plains to prairie, and the installation of buffer strips and runoff control structures was tested. While using a random implementation over 50% of the cropped area resulted in a 43 and 34% drop in sediment and P, a targeted approach leads to similar results (38 and 29%). Moreover, the reductions obtained when targeting 50% of the area are very close (53 and 40% reduction in sediment and P respectively) to those sought, but do not quite reach the 41% mark.

Table 5 - Predicted reduction in runoff, sediment and total P exports at the Pike River watershed's outlet associated with the implementation of agri-environmental scenarios.

No. Scenario	Late season cover crop	Conservation tillage or Inter-cropping	Timely manure incorp.	Perennial cover crop	Buffer strip	Runoff control structure	Runoff		Sediments		Total P	
							mm	% red.*	t	% Red.	t	% Red.
Reference scenario							**181**		**30 500**		**46.1**	
Derivatives of the reference scenario												
1			**ALL**	**FP**	**ALL**	**ALL**	182	0%	22 500	26%	36.3	21%
Conservation agricultural practices												
2		**ALL**					171	6%	16 100	47%	30.1	35%
3		**T 10%**	**ALL**	**FP**	**ALL**	**ALL**	180	0%	18 900	38%	32.9	29%
4		**R 50%**	**ALL**	**FP**	**ALL**	**ALL**	176	3%	17 300	43%	30.5	34%
5		**T 50%**	**ALL**	**FP**	**ALL**	**ALL**	180	1%	14 200	53%	27.7	40%
Scenarios with shifts in crops												
6	**T 10%**		**ALL**	**FP**	**ALL**		180	1%	19 900	35%	32.9	29%
7	**T 10%**	**R 45%**	**ALL**	**FP**	**ALL**	**T 10%**	175	4%	15 000	51%	27.2	41%
8	**T 50%**		**ALL**	**FP**	**ALL**		173	4%	11 200	63%	22.4	51%

% reduction with respect to the reference scenario; **T** = targeted implementation of the BMPs on a certain percentage of annual cropped lands; **R** = random implementation of the BMPs on a certain percentage of annual cropped lands; **FP** = Flood plains; **ALL** = BMP applied to all cultivated lands.

The third step was therefore to change the crops cultivated. The influence of changing a targeted 10 and 50% of the crops to late season cover crops along with manure incorporation, flood plain cultivated as grassland and buffer strips was thus tested. These scenarios lead to a significant decrease in sediment (35 and 63%) and phosphorus (29 and 51%). Several other scenarios were tested and a number met both objectives of a 41% decrease in P inputs to the Missisquoi Bay while being technically feasible and acceptable for the farming community. As such, scenario no. 7 appears to be the most appropriate. A spatially random distribution (R45%) of conservation cropping practices indeed represents the voluntary response of agribusiness owners, whereas the shift of 10% of vulnerable lands to late season cover crops, as well as the implementation of runoff control structures on critical cropped lowlands are better suited to targeted interventions, supported by an incentive programme.

Conclusions

The modelling of different agri-environmental scenarios highlights the importance of spatial targeting when implementing BMPs. Results show that a targeted implementation of BMPs on the most vulnerable of annual cropping land can end in environmental gains four-fold greater than a random implementation over an equivalent area.

With respect to the different BMPs' efficacy, the model attributes the greatest decreases in P exports to cover crops, followed by no-till cultural practices, and runoff control structures. The large capacity to decrease diffuse source P exports attributed to cover crops is tied to their ability to limit erosion and to a field management protocol which simulate a rapid soil incorporation of farm manures. No-till practices also substantially reduce predicted P exports. Given the influence of soil properties on runoff depths and the accumulation of nutrients in the topsoil interacting with surface runoff, the effectiveness of these cultural practices shows a greater variability than that of cover crops. With respect to agricultural water conservation practices, it should be noted that the efficacy of trapping of sediments and P used reflect a particular set of interventions, combining in-ditch catch-basin inlets and shrubby buffer strips, whose efficacy was documented on an experimental watershed. One finding derived from the relative efficacy of the different BMPs modelled is the priority that should be given to the implementation of field-level cultural practices as a first line of agri-environmental defence, with runoff control structures providing a complementary role in diminishing exported flows.

From an operations perspective however, the foremost finding of the current study is the feasibility of attaining the objective of reducing P loads entering the Missisquoi Bay by 41%, as agreed in the Québec-Vermont agreement.

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POSSIBILITIES and Limitations of AVSWAT2000 for the Assessment of Environmental Impacts of Farming Practices

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Abstract

Farming practices (FP) affect the transport and fate of water, carbon, nitrogen, phosphorus, pesticide and other bio-geo-chemical substances in agricultural watersheds. We used the AVSWAT2000-modelling framework to assess environmental impacts of an array of FP and evaluated its possibilities and limitations for this purpose. We mainly considered FP which are important in the context of the EU's Common Agricultural Policy: diversification of crop rotations, introduction of cover crops, shift to zero-tillage, shift to organic fertilization, afforestation. We calibrated SWAT for total discharge in a medium-sized, almost flat, semi-rural watershed taking into account current FP. Next we simulated 11 land use and land management scenarios involving the selected FP. We finally compared the scenarios in terms of the modeled discharge at the outlet and the soil organic carbon stocks of HRU. Although the relative simulation results are largely according to expectation, we found that the plant growth and management modules of AVSWAT2000 are not fully operational to efficiently and realistically parameterize and evaluate the impacts of FP. We identified the simulation of biomass development over time, the computation of a true carbon balance, the handling of crop rotations and the definition of the till-operation to be among the major points of attention for improving AVSWAT2000's capabilities in dealing with FP and their environmental effects.

Keywords

Farming practices, environmental impact, river discharge, soil organic carbon

Introduction

Agricultural or farming practices (FP) deal with issues as basic as the choice of crops, crop varieties and crop succession, crop calendars, nature and timing of soil preparation, crop fertilization, protection, drainage, irrigation and harvest. From an agro-economical point of view, FP can be considered as driving forces which modify the bio-hydro-geo-chemical cycles to obtain maximum desired biomass output, in terms of both quantity and quality. The cycles affected are especially the ones related to water, carbon, nitrogen, phosphorus and pesticide components.

The EU's common agricultural policy (CAP) provides the legal framework for Member States to prescribe, prohibit and recommend FP that should result in Good Agricultural and Environmental Conditions (GAEC). GAEC must be achieved by farmers for being eligible for income support from the CAP. These conditional CAP-payments can be considered as a policy response which is based on the assumption that by controlling the driving forces, i.e. the FP, pressures on the agri-environment will decrease, states improve and negative impacts be reduced.

All the cycles are highly interrelated and extremely dynamic in space and time. In order to understand, assess and predict the environmental impacts of FP on policy relevant scales, an integrated modeling approach is required.

In this study the AVSWAT2000-modelling framework was examined for its possibilities and limitations to assess environmental impacts of various FP for an almost flat, medium-sized river basin in north-eastern Belgium. The specific objectives of the exercise were to:

- Examine the plant growth and management modules of SWAT in terms of the possibilities for parameterization of crop successions within and over individual years, tillage versus no-tillage, mineral versus organic fertilization practices and afforestation. Also loss of agricultural land through sealing was considered;
- Study the relative effects of the FP in terms of both river discharge at the outlet of the catchment and soil organic carbon (SOC) storage in HRU.

Materials and methods

SWAT 2000 + ArcView GIS 3.2 interface

The version of the Soil Water Assessment Tool (SWAT) used was the version 2000, with ArcView GIS 3.2 as pre- and postprocessor (Di Luzio et al., 2002). It is further termed AVSWAT2000 or SWAT.

Study area

Several research projects have shown that SWAT has a good potential to study the hydrology of small to medium sized river basins in Belgium, using available data sources (e.g. Abu El-Nasr et al., 2005; Heuvelmans et al., 2004, Van Griensven, 2002). For this study the Grote Nete River basin, situated in the northeast of Belgium, with an area of 383 km^2 was selected. Rouhani et al. (in press) provide a detailed physiographic description of the basin.

Model input and reference data

A spatial model was established for the river basin using a DEM and geodatasets encompassing river network, land use and soil associations, coming from the Flemish regional Spatial Data Infastructure (www.agiv.be). Based on the 50 meter resolution DEM and the river network, the catchment was automatically delineated and divided into 57 sub-basins. Whereas the 'official' area of the catchment is 383 km2, SWAT retains 348,5 km^2 as draining to the studied outlet. Soil associations were characterized by a dominant soil series for which a typical synthetic soil profile with horizon characteristics was retrieved from the Aardewerk-BIS Soil Information System (Van Orshoven et al., 1991). The soil profile and horizon characteristics were used to derive bulk density, available water capacity and saturated hydraulic conductivity by means of the pedo-transfer functions developed by Vereecken et al. (1989) and Vereecken et al. (1990). SWAT's multiple HRUs option was used to enable the creation of multiple HRU for each sub-basin based on the threshold values of 10% for land use type and 20% for soil type. In total 347 HRUs, varying in size, were derived. 126 km^2 or 36% of the modeled watershed, distributed over 104 HRU, are under arable land. 98 km^2 or 28%, distributed over 94 HRU, are under pasture. Based on a high resolution agricultural parcel geodataset, it was assumed that all arable land was under silage maize and all pasture land under winter pasture (CSIL and WPAS in SWAT's land use database respectively).

Daily rainfall data for seven stations, other daily data needed for PET-computation at one station and the monthly statistics for minimum and maximum temperature for eleven stations were retrieved from the Royal Meteorological Institute of Belgium (1985-1995).

Stream gauging records for the period 1985-1995 at the outlet of the basin (Flemish regional administration for waterways and maritime affaires) and SOC-stocks for landscape units (Lettens et al. 2005) were available for calibration and validation.

'Default' land management and farming practices

Land management and farming practices are of crucial importance for this study. From the 13 FP which can be parameterized by SWAT (Neitsch et al., 2002 – p.193), irrigation and auto-irrigation, pesticide application, street sweeping and impound and release of water for rice growing are not considered in this study.

Default practices and scheduling were accepted for all land use classes but the agricultural classes. Based on common agronomic knowledge and on the lessons learnt by trial and error, the 'baseline' FP for CSIL and WPAS were parameterized according to Tables 1 and 2.

Table 1: Adopted management practices and scheduling for CSIL

#	Timing	Practice	Remark
0	1-jan	Initialisation of SCS Curve Number (CN)	CN is initialized at the default CN provided by SWAT. This means that the soil cover effect (CN to increase since soil cover is not complete) is assumed to be compensated by the slope effect (CN to decrease since slopes are smaller than 5%).
1	15-april	Tillage	Mixing depth = 350 mm; Mixing efficiency = 0.70; Implement choosen from the SWAT-tillage database is the Subsoiler-Bedder Hip-Rip (SBEDHIPR; code 57). CN is increased to default CN +3.
2	11-may	Sowing	CN is set back to the default CN – 2.
3	12-may	Start of Autofertilisation	28-10-10 fertiliser is used as soon as the nitrogen stress factor drops below 0.95. Maximum allowed fertilization dose per intervention is 200 kg. Maximum allowed total fertilization amount = 800 kg. 20% of the fertilizer is applied to the upper 10 mm. The remaining 80 % is applied to the rest of the upper (= Ap) horizon.
4	15-oct	Harvest and Kill	Default harvest index (90% of aboveground dry biomass) and default efficiency (=100%, no losses at harvest). CN is set back to the initial value representing conditions where maize residue is present and slope is smaller than 5% (=default CN or previous CN + 2).

Table 2: Adopted management practices and scheduling for WPAS

#	Timing	Practice	Nature
1	1-jan	Start of growing season	CN = default CN provided by SWAT throughout the year
2	15-apr	Start of Autofertilisation	Using 28-10-10 fertiliser (code=10) as soon as the nitrogen stress factor drops below 0.95. Maximum allowed fertilization dose per intervention is 200 kg. Maximum allowed total fertilization amount = 800 kg. 20% of the fertilizer is applied to the upper 10 mm. The remaining 80 % is applied to the rest of the upper (= Ap) horizon.
3	10-may	Harvest; 1st cut	Default harvest index (90% of aboveground dry biomass) and default efficiency (=100%, no losses at harvest).
4	25-june	Harvest; 2nd cut	Default harvest index (90% of aboveground dry biomass) and default efficiency (=100%, no losses at harvest).
5	10-august	Harvest; 3d cut	Default harvest index (90% of aboveground dry biomass) and default efficiency (=100%, no losses at harvest).
6	10-october	Harvest, 4th cut	Default harvest index (90% of aboveground dry biomass) and default efficiency (=100%, no losses at harvest).
7	31-dec	Kill	SWAT requires a crop to be killed before the same or another crop can be 'planted' again. With this kill-operation, biomass produced between 10-oct and 31-dec is added to the upper 10 mm of soil as crop residue. Due to dormancy, growth stops mid November already and no additional biomass is produced between that date and 31-dec.

Calibration and validation

River discharge

The model was calibrated using meteorological data of 1986-1989 and validated for 1991 and 1992-1995. This period was chosen because of data availability and because it represents a combination of dry, average, and wet years (annual precipitation ranged from 646.5 to 988.7 mm according to the detailed records).

Values for the 10 most sensitive parameters, as identified by Rouhani et al. (in press) were adjusted according to Table 3. The value for RDHRG_DP is the only one different from Rouhani et al. (in press).

Table 3: Default and calibrated values for adjusted parameters

Flow component	Parameter	Default value	Adjusted value
Quick flow	CN2 (dimensionless)	Based on permeability, land use, management practices and antecedent soil moisture conditions	All CN increased by 2
	SOL_AWC (mm water/mm soil)	Based on soil characteristics	All AWC decreased by 0.01

	CH_N1 (dimensionless)	0.014	0.5
	SURLAG (days)	4	3
Slow flow	GW_REVAP (dimensionless)	0.02	0.14
	REVAPMN (mm)	1	1.1
	GWQMN (mm)	0	0.35
	ALFA_BF (days)	0.048	0.15
	GW_DELAY (days)	31	55
	RDHRG_DP (dimensionless)	0.05	0.25

The observed and simulated average daily discharge values (m^3/s) of the calibration and validation runs are given in Table 4. Also the RMSE for daily flow (m^3/s) are given.

Table 4: Observed versus simulated average daily discharge and RMSE

	Observed discharge (m^3/s)	**Simulated Discharge (m^3/s)**	**RMSE (m^3/s)**
Calibration (1986-1988)	4.70	4.53	1.29
Validation (1990, 1992-1995)	3.74	3.45	1.41

SOC

SWAT does not output the terms of the SOC-balance for HRU. Yearly values for SOC-content by HRU were assessed by first computing the SON-balance for the humus pool of the top soil compartment of 100 cm and next deriving the SOC-balance from it, using the default carbon-to-nitrogen- (C/N) ratio of 14 (Neitsch et al., 2002 – p.178). Below 100 cm, the amounts of SON and SOC present are considered to be irrelevant.

The SON-balance equation used for the humus pool was:

$SON_i = SON_{i-1} + N\_APP + N\_GRZ + 0.2*F\_MN – A\_MN – ORGN$ [eq.1]

in which SON_i is the soil organic nitrogen content (T/ha) in year i, SON_{i-1} is the soil organic nitrogen content (T/ha) in year i-1, N_APP is the amount of organic N added by fertilization, N_GRZ is the amount of organic N added by grazing animals (no grazing animals present in our simulations; N_GRZ = 0), F_MN is the amount of nitrogen coming from the fresh organic nitrogen pool (of which 80% is assumed to be mineralized and 20% humified), A_MN is the amount of nitrogen mineralized and ORGN is the amount of organic nitrogen removed by surface run off.

To convert SON to SOC, equation [2] is used:

SOCi = SONi * 14 [eq.2]

in which SOCi the soil organic carbon content of the humus pool (T/ha) in year i, SONi is the soil organic nitrogen content (T/ha) for the same year obtained using [eq.1] and 14 is the default C/N-ratio.

In Table 5, average annual SOC-changes obtained using SWAT are compared with changes assessed for 1990-2000 from independent reference data (Lettens et al., 2005).

Table 5: Simulated annual C-changes versus reference data

	SOC Changes 1995-1985 (T C/ha/yr), 100 cm, Grote Nete, simulated	**SOC Changes 2000-1990 (T C/ha/yr), 100 cm, Flanders**
Arable land – CSIL	-0.36	-0.73
Grassland – WPAS	-0.33	-0.82

The SOC-routine was not further calibrated since the nature of the reference data was too general and additional data for validation lacking.

Scenarios studied regarding farming practices

In order to study the outcome effect of changes in farming practices, the scenarios listed in Table 6 were established. Every scenario is run for 11 consecutive years (1985-1995). 1985 is considered a model warm up year. Its output is not considered where it regards river discharge. The output for 1990 and 1992 is not considered because of shortcomings in the meteorological input data.

Table 6: Scenarios studied regarding farming practices

Acronym	Description	HRU
Sc1_CSILaRYEG	Alternation of 2 seasonal crops, CSIL and the temporary grassland composed of Italian Ryegrass.	on all HRU initially under CSIL
Sc1_CSILaCLVR	Alternation of 2 seasonal crops, CSIL and the legume Red Clover	on all HRU initially under CSIL
Sc1_CSILaFLAX	Alternation of 2 seasonal crops, CSIL and the oil-rich Flax	on all HRU initially under CSIL
Sc2_CSILcover	Continuous CSIL with Italian Ryegrass as cover crop which is planted after harvest and killed next spring before the next CSIL-campaign	on all HRU initially under CSIL
Sc3_CSILnoTill	Continuous CSIL without any till operation (Compare to table C)	on all HRU initially under CSIL
Sc4_CSILorgfert	Continuous CSIL with autofertilisation using mineral 28-10-10 fertiliser replaced by animal manure	on all HRU initially under CSIL
Sc5_CSIL2cWPAS	Continuous conventional WPAS from the 2nd year of simulation	on all HRU initially under CSIL
Sc5_WPAS2cCSIL	Continuous conventional CSIL from the 2nd year of simulation	on all HRU initially under WPAS
Sc6_CSIL2cPINE	Continuous PINE from the 2nd year of simulation	on all HRU initially under CSIL
Sc7_CSIL2cSEAL	Continuous sealing for commercial estates from the 2nd year of simulation	on all HRU initially under CSIL
Sc8_CSILrandom	One of the 10 above sets of FP (baseline + 9 scenarios applicable to CSIL) is attributed at random to each HRU initially under CSIL	On all HRU initially under CSIL

Scenario 8 is studied since it provides a way of evaluating the impact of FP-related agri-environmental commitments as proposed in regional rural development plans. In a region or watershed, different farmers may adopt different agri-environmental measures. By spatially randomising these, possibly making use of a priori-knowledge or estimates of probability of (non-)take up, is a possible approach.

Results

Environmental performance of baseline and alternative FP scenarios

All scenarios listed in Table 6 were parameterized and run. Table 7 gives a concise overview of the simulation results.

Table 7: Simulation results for the 11 scenarios

#	Acronym	Average daily discharge 1986-1988, 1990, 1992-1995 (m^3/s) at outlet	Contribution of surface runoff to total discharge at outlet (1986-1988, 1990, 1992-1995, m^3/s)	HRU under CSIL SOC Changes 1995-1985 (T C/ha/yr), 100 cm, Grote Nete, simulated
0	Baseline	3.86	0.69	-0.36
1	Sc1_CSILaRYEG	3.74	0.48	-0.34
2	Sc1_CSILaCLVR	3.79	0.49	-0.35
3	Sc1_CSILaFLAX	3.86	0.64	-0.36
4	Sc2_CSILcover	3.80	0.49	-0.30
5	Sc3_CSILnoTill	3.85	0.69	-0.36
6	Sc4_CSILorgfert	3.86	0.69	+0.38
7	Sc5_CSIL2cWPAS	3.69	0.49	-0.34
8	Sc5_WPAS2cCSIL	3.98	0.84	-0.36
9	Sc6_CSIL2cPINE	3.77	0.35	-0.40
10	Sc7_CSIL2cSEAL	3.99	1.37	-0.38
11	Sc8_CSILrandom	3.79	0.54	-0.25

Discussion

Environmental effects of simulated farming practices

The simulations confirm a number of scientific and empiric expectations, at least in relative terms:

– There is an inverse relationship between increased length and density of soil cover (as reflected in CN-values) and the average surface runoff. Lower values for average surface runoff are mainly due to reduced peak discharges (not shown). When the absolute value of surface runoff is decreasing, also the total discharge at the outlet of the basin is decreasing and vice versa. This means probably that part of the extra infiltrated water is lost by extra evapo-transpiration or evacuated out of the basin via the deep aquifer;
– Sealing increases surface runoff considerably. Total discharge is affected to a lesser extent;
– Amendment of organic fertilizer in addition to input of crop residue turns the HRU under CSIL from a source of carbon into a sink;
– The more biomass is produced and the more residue is returned to the soil (e.g. by growing cover crops or shifting to pasture of pine plantation), the lower the SOC-losses. On an annual basis, differences are small though;
– When FP are attributed at random to HRU, an 'average' behaviour is found. SOC-stock changes are less negative because of the input of organic fertilizer to a fraction of the HRU. Discharge is slightly lower than the reference as more HRU are covered for longer periods within the year.

Expectations which are not confirmed are that:

– The no-till operation has a negligible effect on either discharge or SOC-stock although the detailed daily output discharge values reveal that peaks in spring are very slightly attenuated. Slightly less organic N (the equivalent of 0.5 kg/ha/yr of SOC) is lost through runoff so that the SOC-pool increases slightly.

Parameterisation of FP

Based on the experiences gained in parameterizing the FP, running the model for the various scenarios and examining the output, following observations can be made regarding the plant growth and management modules of AVSWAT2000:

Regarding biomass development and crop residue production:

N and C in crop residues entering the fresh crop residue pool are important inputs to the SON- and SOC-balances. Crop-residues equal the total biomass produced by the crop minus the biomass removed in the yield operation. The latter consists of the amount of product harvested minus the unintended losses which become part of the residue. However, we found that the development of biomass of silage maize and winter pasture is weakly simulated (in absolute terms) by SWAT, at least for the conditions of this watershed:

– When land management and FP are scheduled by means of the heat unit system, very many inconsistencies regarding biomass development were encountered. Biomass produced is by far too low, growing season is too short, some years biomass is not simulated, some years the harvest/kill operation is not executed. Most of this erratic model behaviour can be removed by scheduling using fixed dates, by reducing the temperature stress factor and by reducing nitrogen stress by auto-fertilisation. For WPAS and PINE, a kill operation is necessary at the end of each year although it are permanent crops. This affects severely the crop residue inputs to the soil.
– Biomass produced by WPAS and removed by grass cutting is confusingly reported in the yearly output files. Apparently, the biomass reported by SWAT is the sum of the biomass present at the days of cutting and at the day before the start of the dormancy period

(around mid November) without accounting for the fact that at harvest not all biomass is removed. As such a considerable percentage of the reported biomass is double counted !

Regarding crop rotations, cover crops and shift to permanent crops:

It was impossible to make a seasonal crop follow by a permanent crop and vice versa. As such the scenarios in which CSIL is followed by WPAS, WPAS followed by CSIL and CSIL followed by PINE from the 2nd year onwards, could not be run properly. WPAS, CSIL and PINE had to be simulated from the 1st year onwards.

Cover crops, sown after harvest of the main crop and removed or plowed before establishing the main crop of the next year cannot be maintained from one year to the next. It must be killed on 31-december and resown on 1-january. Again this affects the inputs of residue to the soil.

Regarding tillage:

For the long term water average discharge, the absence of change with respect to the reference simulation is in agreement with Chaplot et al. (2004). The slight attenuation of the discharge peaks in spring can be perfectly explained by the fact that by tilling, during the few weeks between plowing and sowing of CSIL, the CN-value of the HRU is increased to reflect runoff and erosion sensitivity. With no-till, this high CN-period is eliminated. Since SWAT defines tillage merely as “the mixing of the upper soil layers upto a specified depth resulting in the redistribution of the residue/nutrients/chemicals/etc. within this depth with a specified efficiency”, from the modeling point of view it is not surprising that the water balance of HRU is not affected. The fact that SOC-contents and –stocks are almost not affected by no-till can also be explained by this modeling concept. In our case, this means that residue is mixed up to a depth of 350 mm so that the residue pool in the upper 10 mm is reduced. It is the latter pool which provides organic and mineral N which is available for runoff. Slightly less organic N (the equivalent of 0.5 kg/ha/yr of SOC) is lost through runoff so that the SOC-losses are also slightly less. It can be recommended to redefine the tillage operation as to include the effect of tillage on soil hydraulic and humus mineralization properties.

Regarding SOC-balance computation:

SOC is a very important issue in current agri-environmental policy. A dedicated SOC-balance module is required for SWAT, dealing properly with first order kinetics and two or more SOC-pools (e.g. Andren and Katterer, 1997). The developed spreadsheet based approach to compute annual SOC-balances from the terms of the SON-balance should be considered as a temporary work around.

Conclusions

The capability to ex-ante assess the expected environmental impacts of FP for larger territories is a necessity for effective and efficient agri-environmental policy design and implementation. A similar capacity for ex-post assessment is useful for explanatory studies of obseved impacts. Given its (semi-)distributed character, its physical basis and its plant growth and management module, AVSWAT2000 has the potential to provide this capability. We identified the simulation and reporting of biomass development, the computation of a true carbon balance, the handling of multi-annual crop rotations and the definition of the till-operation to be among the major points of attention for improving SWAT’s capabilities in dealing with FP and their environmental effects.

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Assessment of Hydrology and Sediment Transport and Prospects of Simulating Agri-Environmental Measures with SWAT

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Abstract

Tools are needed to assess loading from agricultural sources to water bodies as well as the effect of alternative management options in varying environmental conditions. For this, mathematical models like SWAT offer an attractive option. In addition to loading estimates, SWAT also offers a possibility to include various agricultural management practices like fertilization, tillage practices, choice of cultivated plants, buffer strips, sedimentation ponds and constructed wetlands (CWs) in the modeling set-up. In this study, the parameterization of SWAT has been developed particularly in terms of discharge dynamics and sediment fluxes and a sensitivity analysis was made. Moreover, modeling strategies with dominant land uses and soil types vs. land uses and soil types exceeding certain thresholds within subcatchments were compared. In the thresholds-exploiting SWAT project agricultural land was divided into 5 classes whereas the "competing" project only had 2 classes. These SWAT modeling exercises were performed for a 2nd order catchment (Yläneenjoki, 233 km^2) of the Eurajoki river basin in southwestern Finland. The Yläneenjoki catchment has been intensively monitored during more than 10 years. Hence, there is abundant background information available for both parameter setup and calibration. Moreover, information on local agricultural practices and the implemented an planned protective measures are readily available thanks to aware farmers and active authorities. In future, our aim is to exploit this knowledge by modeling different management scenarios and assessing their effects on loading with SWAT.

KEYWORDS: SWAT, agriculture, hydrology, sediment loading, management actions.

Introduction

Due to the obligations set by the EU Water Framework Directive (WFD), the environmental authorities are in need of information on the effects of different management options aimed at improved water quality. In addition to the knowledge obtained from the Finnish field-scale experiments on the effects of agricultural practices and protection measures on loading (e.g. Turtola and Kemppainen 1998, Koskiaho et al. 2003, Puustinen et al. 2005, Uusi-Kämppä 2005), mathematical modelling tools are needed to generalise the effects in varying environmental conditions on catchment scale. Models like SOIL/SOILN (Johnsson et al. 1987), GLEAMS (Knisel 1993) and ICECREAM (Tattari et al. 2001) have been used to assess phosphorus (P) and nitrogen (N) losses from agricultural land in Finland (Granlund et al. 2000, Knisel and Turtola 2000, Tattari et al. 2001). These models have, however, limitations in terms of catchment-scale evaluations of loading and the effects of management actions. For these purposes, the SWAT model (**S**oil and **W**ater **A**ssessment **T**ool (Arnold et al. 1998, Neitsch et al. 2001)) offers an attractive alternative. In Finland, the

SWAT model has been previously applied to the rivers basins of Vantaanjoki (Grizzetti et al. 2003) and Yläneenjoki (Bärlund et al. 2007).

One aim of the ongoing CATCH_LAKE project is to test and evaluate the applicability of the SWAT model (version 2005) for the previously stated needs. Moreover, SWAT output will be used as input for a lake model (LakeState (Malve and Qian 2006)). Based on the load reduction objectives set by the LakeState model, SWAT will be employed to assess the possibilities to reach this goal with the implemented and intended management actions in the Yläneenjoki area. The modelling set-up and approach, as well as calibration made this far in the CATCH_LAKE project, are described in this study.

Material and methods

The SWAT model

The SWAT model is a continuous time model that operates on a daily time step at catchment scale. It can be used to simulate water and nutrient cycles in agriculturally dominated landscapes. The catchment is generally partitioned into a number of subbasins where the smallest unit of discretization is a unique combination of soil and land use overlay referred to as a hydrologic response unit (HRU). SWAT is a process based model, including also empirical relationships. The model has been widely used but also further developed in Europe (e.g. Eckhardt et al. 2002, Krysanova et al. 1999, van Griensven et al. 2002). SWAT was chosen for the CATCH_LAKE project for three main reasons: its ability to simulate suspended sediment –as well as P and N– loading on catchment scale, its European wide use and its potential to include agricultural management actions. SWAT has already been found to have potential with respect to the future requirements set by WFD in Scotland (Dilks et al. 2003) and in Finland (Bärlund et al. 2007).

The experimental catchment

The Yläneenjoki catchment, 233 km^2 in area, is located on the coastal plains of south-western Finland, thus the landscape ranges in altitude from 50 to 100 m a.s.l. The soils in the river valley are mainly clay and silt, whereas tills and organic soils dominate elsewhere in the catchment. Long-term (1961–1990) average annual precipitation is 630 mm of which approximately 11% falls as snow (Hyvärinen et al. 1995). The average monthly temperature for the period November to March ranges between –0.5 and –6.5 ºC. The warmest month is generally July when average temperature is 16.2 ºC (1980–2000). Average discharge measured in the Yläneenjoki main channel is 2.1 m^3s^{-1} (Mattila et al. 2001), which equates to an annual water yield of 242 mm (1980–1990). The highest discharges occur in the spring and late autumn months.

Agriculture in the Yläneenjoki catchment consists of mainly cereal production and poultry husbandry. According to surveys performed in 2000–2002 75% of the agricultural area is planted for spring cereals and 5–10% for winter cereals (Pyykkönen et al. 2004). Agriculture in the Yläneenjoki catchment is intensive for Finland. The Yläneenjoki catchment is responsible for over 60 % of the external nutrient loading into the Lake Pyhäjärvi (Ekholm et al. 1997).

The regular monitoring of water quality of the river Yläneenjoki started as early as in the 1970s. The nutrient load into the Lake Pyhäjärvi via the river has been calculated from the (generally) bi-weekly water sampling results and daily water flow records at one point (Vanhakartano, see Fig. 1). Furthermore, water quality was monitored on a monthly basis in three additional points in the main channel in the 1990s and in 13 open ditches running into the river Yläneenjoki since the 1990s.

Background data

For the SWAT simulations the available data on elevation, land use and soil types had to be aggregated. The resolution of the digital elevation model (DEM) proved to be rather poor (5 m) for successful set-up of the Yläneenjoki catchment. Hence, we used a modified DEM where the main channels of the catchment were somewhat deepened to emphasize the actual routes of water.

The standard GIS data available (CORINE 2000) recognizes three types of agricultural land; (i) actively cultivated fields, (ii) grasslands and pasture and (iii) fragmented agricultural land. Here, in the case of Yläneenjoki basin, one of the two SWAT projects was based on 6 land use classes (urban areas, wetlands (bogs), actively cultivated fields, grasslands and pasture, actively growing forest and old forest). In this project, the actively cultivated fields were parameterized to be spring barley since, as previously described, spring cereals are the most common crop type in the catchment. In this project, the forested areas were grouped into two classes according to their stage of growth. The distribution of the different land uses in the Yläneenjoki catchment is presented in Table 1.

The other project had otherwise identical land use, but (i) the agricultural areas were divided into 5 classes (autumn cereals, spring cereals, root crops, grasses and gardens, Fig. 1) and (ii) there was only one class of forested areas. Both projects had 4 soil types (clay, moraine, silt and peat), of which the distribution is presented in Table 1.

Table 1. Distribution of soil type and land use in the Yläneenjoki area.

Land use	Percentage of the catchment area	Soil type	Percentage of the catchment area
Grasslands	0.5	Silt	2
Wetlands	4	Peat	14
Urban areas	4	Clay	42
Old forest	15	Moraine	42
Agricultural areas	28		
Actively growing forest	49		

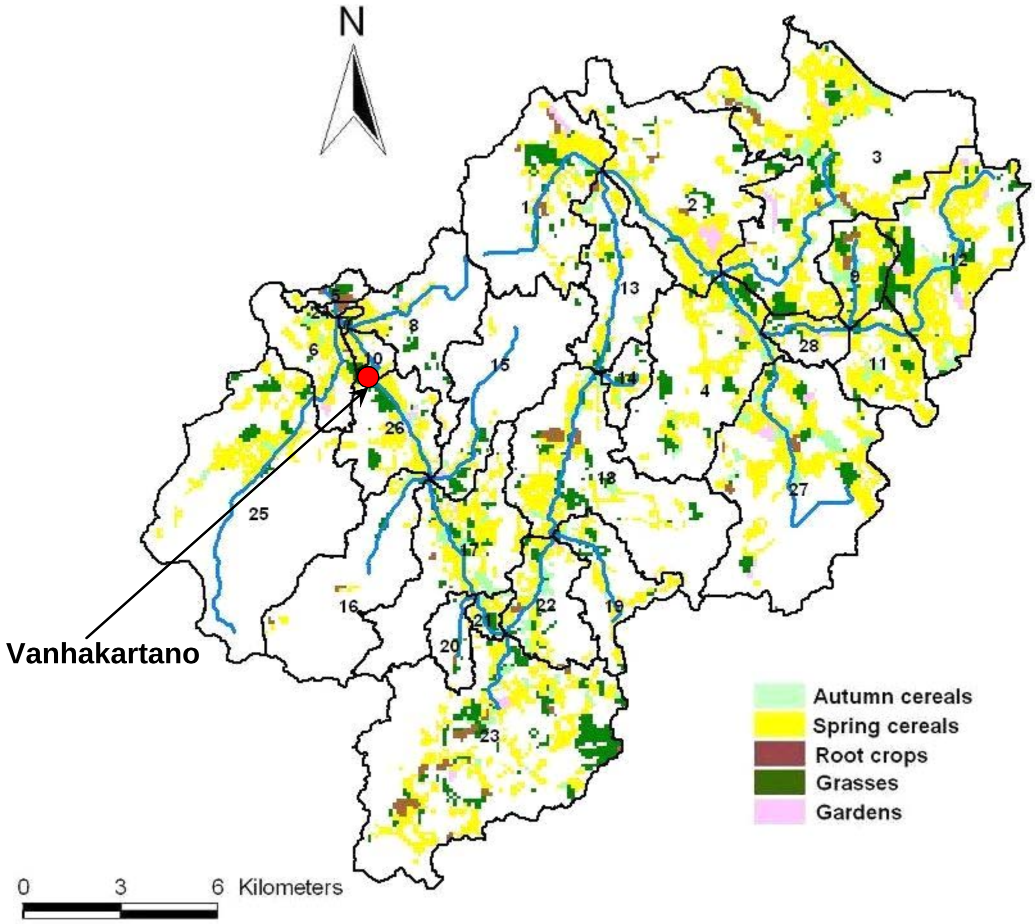

Fig. 1 Division of the Yläneenjoki subcatchments and agricultural land use data (situation in the year 2004) used in the project "threshold". The white areas mostly represent forest.

The modeling approach

The wider-scope modelling strategy in the Pyhäjärvi area is based on linking the Yläneenjoki catchment – from which most of the loading comes into the lake – with Lake Pyhäjärvi (Bärlund et al. 2007). The LakeState model (Malve and Qian 2006) will be utilised to set the load reduction target which is required to achieve improvements in water quality of Lake Pyhäjärvi. The task of the SWAT model is to assess the possibility to reach this target using a variety of management options such as buffer strips, artificial wetlands, as well as changes in fertilization or tillage practices.

Two different SWAT projects for the same Yläneenjoki catchment were set up. The differences were in the number of HRUs and in agricultural land use (Table 2). The projects titled "dominant" and "threshold" resulted in 27 and 28 subcatchments, respectively. In the project "dominant", the land use and soil types (Table 2) with the highest percentages in a subcatchment were applied for the whole subcatchment. For example, for the westernmost subcatchment, the model assumed that the area was entirely forest on moraine soil. In the other project, threshold values (Table 1) were used to distinguish different land use and soil types within each subcatchment. For example, if more than 1% of a subcatchment was under grass and these areas were divided on clay and silt soils both soil types representing more than 10% of the subcatchment area, this would make the HRUs "grass-clay" and "grass-silt" for this subcatchment. This approach resulted in 257 HRUs in the project "threshold". Due to the use of the available data with finer resolution, the use of threshold values should lead to more accurate results than the use of dominant land use and soil types.

Table 2. Characteristics of the two different SWAT projects.

Project	N of subcatchments	Threshold value for the HRU determination	N of HRUs	N of agricultural land use types
"dominant"	27	none [1)]	27	2 [2)]
"threshold"	28	soil: 10%, land use: 1%	257	5 [3)]

[1)] dominant land use and soil was used
[2)] row crops with autumnal tillage and grass
[3)] autumn cereals, spring cereals, root crops, grass and garden

Results and discussion

Sensitivity analysis

Sensitivity analysis can be seen as an instrument to analyse the impact of model input on model output as well as for model calibration, validation and reduction of uncertainty. The sensitivity analysis method implemented in SWAT is called **L**atin **H**ypercube **O**ne-factor-**A**t-a-**T**ime (LH-OAT) (Morris 1991). Sensitivity is expressed by a dimensionless index I, which is calculated as the ratio between the relative change in model output and the relative change of a parameter.

In this study, the sensitivity analysis was performed for the project "threshold" only. Thirty-three parameters were selected for the analysis. The range for each parameter was first estimated based on earlier SWAT sensitivity studies and expert knowledge. We ended up decreasing the default ranges in order make them more realistic for the conditions in the Yläneenjoki basin and to quicken the sensitivity analysis runs.

Table 3 shows the results of the sensitivity analyses. Every model run covered six years (1994-1999). The first year was used as "warm-up" period. In this study, the sensitivity analysis was performed on the daily average flow and on the daily average sediment concentration. The analysis was first performed without observed data and then using observed data. The general watershed parameters SURLAG and TIMP seemed to be the most sensitive parameters in the case without observed data. However, with observed data the importance of groundwater (GWQMN), management (CN2) and soil parameters (SOL_AWC, SOL_Z and SOL_K) was increased.

Table 3. Sensitivity of the average daily flow and sediment concentration to input parameters with and without observed data for the project "threshold".

FLOW, no observed data			FLOW, with observed data			SED.CONC., no observed data			SED.CONC., with obs. data		
Rank	Mean	Parname	Rank	Mean	Parname	Rank	Mean	Parname	Rank	Mean	Parname
1	0,41	SURLAG	1	0,71	GWQMN	1	8,23	SPCON	1	4,35	SPCON
2	0,30	TIMP	2	0,38	SOL_Z	2	2,49	CH_COV	2	2,61	CH_COV
3	0,16	SMFMX	3	0,37	TIMP	3	1,90	CH_EROD	3	1,80	CH_EROD
4	0,13	SMFMN	4	0,36	ESCO	4	1,15	SURLAG	4	1,04	SURLAG
5	0,10	SOL_AWC	5	0,33	SOL_AWC	5	1,06	CN2	5	0,77	GWQMN
6	0,09	GWQMN	6	0,30	CN2	6	0,98	CH_N	6	0,76	TIMP
7	0,08	CH_K2	7	0,20	SOL_K	7	0,70	SPEXP	7	0,53	CH_K2
8	0,07	SMTMP	8	0,19	REVAPMN	8	0,64	GWQMN	8	0,49	CH_N
9	0,06	ESCO	9	0,09	RCHRG_DP	9	0,63	ESCO	9	0,45	CN2
10	0,05	CN2	10	0,08	SMTMP	10	0,57	SOL_K	10	0,34	ESCO
11	0,04	SOL_K	11	0,06	SFTMP	11	0,52	SMFMN	11	0,24	SOL_AWC
12	0,04	SOL_Z	12	0,06	SMFMX	12	0,49	TIMP	12	0,17	REVAPMN
13	0,02	SFTMP	13	0,06	SMFMN	13	0,48	SOL_AWC	13	0,15	SPEXP
14	0,01	CH_N	14	0,06	SURLAG	14	0,43	SMFMX	14	0,14	SOL_Z
15	0,01	RCHRG_DP	15	0,01	SLOPE	15	0,42	SMTMP	15	0,13	SMTMP
16	0,01	ALPHA_BF	16	0,01	GW_REVAP	16	0,37	CH_K2	16	0,12	SMFMX
17	0,01	REVAPMN	17	0,01	GW_DELAY	17	0,30	SOL_Z	17	0,11	RCHRG_DP
18	0,00	SLSUBBSN	18	0,01	CH_K2	18	0,29	REVAPMN	18	0,08	SOL_K
19	0,00	SOL_ALB	19	0,01	CANMX	19	0,20	RCHRG_DP	19	0,07	SMFMN
20	0,00	SLOPE	20	0,01	SOL_ALB	20	0,17	ALPHA_BF	20	0,06	SLOPE
21	0,00	GW_REVAP	21	0,00	CH_N	21	0,12	GW_DELAY	21	0,05	SFTMP
22	0,00	GW_DELAY	22	0,00	ALPHA_BF	22	0,12	SLOPE	22	0,05	ALPHA_BF
23	0,00	CANMX	23	0,00	SLSUBBSN	23	0,06	SFTMP	23	0,05	GW_DELAY

24	0,00	EPCO	24	0,00	EPCO	24	0,05	USLE_C	24	0,04	USLE_C
			25	0,00	USLE_C	25	0,04	SLSUBBSN	25	0,02	SLSUBBSN
						26	0,02	GW_REVAP	26	0,02	CANMX
						27	0,02	CANMX	27	0,00	GW_REVAP
						28	0,01	SOL_ALB	28	0,00	SOL_ALB
						29	0,00	USLE_P	29	0,00	USLE_P
						30	0,00	EPCO	30	0,00	EPCO
						31	0,00	BIOMIX			

Very high sensitivity
High sensitivity
Medium sensitivity
Low sensitivity

In terms of sediment concentration, the parameter set was in general alike for both projects. As can be seen from Table 3, more parameters were in the sensitivity class of "very high" and "high" than in the case of flow. The most sensitive parameters were general watershed parameters SPCON and SURLAG and main channel characteristics (CH_COV and CH_EROD).

Calibration

The years 1995–1999 were chosen as the calibration period. In terms of hydrology, the years differed quite much from each other (Fig. 2). The year 1999, for example, represents a typical runoff pattern with dry summer and distinct spring and autumn peaks. Meanwhile in 1998 runoff peaks occurred throughout the year.

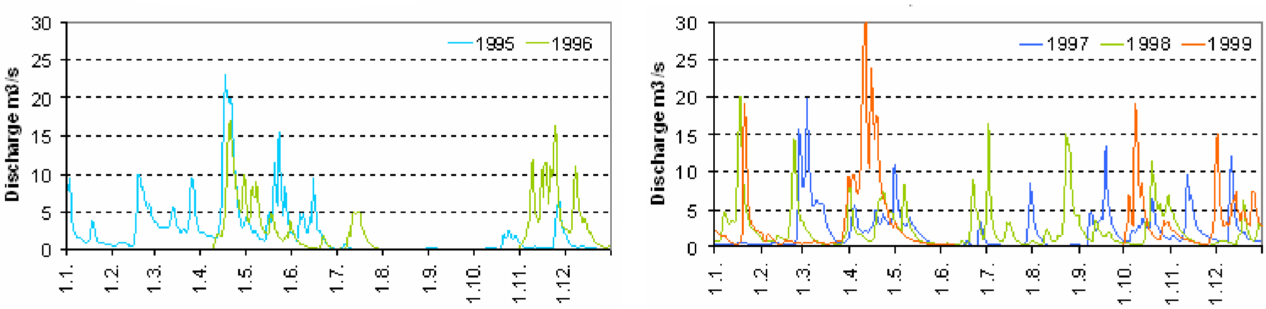

Fig. 2. Discharge from the Yläneenjoki catchment at the Vanhakartano measurement station during 1995–1999.

We utilized the autocalibration tool available in the 2005 version of SWAT (Neitsch et al. 2005). The autocalibration procedure was rather slow; in our cases, a laptop PC was typically left to work over a weekend to complete a SWAT autocalibration procedure.

As suggested by the sensitivity analysis, the parameters GWQMN, TIMP, ESCO, SOL_AWC, CN2 (agric. land), SMTMP, SFTMP and SURLAG were chosen for autocalibration runs made for daily discharge with observed data. The first autocalibration run was made with a simulation using the "initial" set of parameters obtained from previous modeling efforts in the Yläneenjoki area. After every run, the values from the output file bestpar.out were input to the model and a new simulation was run. In all, 6 simulation-autocalibration runs were made. The observed data was from the Vanhakartano measurement station, and hence the results for the outlet of subbasin 26 were examined (see Fig.1). Autocalibration results are presented in Table 4.

Figure 3 shows the improvement obtained by the calibration process in the fit between simulated and observed daily flow. Particularly in spring and autumn the amendment was clear: not only the simulated peaks were much closer to the observed, but also the autumnal low flow period appeared much more realistic. In mid-winter and summer

Parameter	Initial, best guess	Best parameter	Good range
SMTMP	-0,1	1,28	0,92-1,63
SMFMX	2,6	4,56	3,6-6
SMFMN	1,3	0,096	0-0,46
TIMP	0,9	0,983	0,8-1
ESCO	0,95	0,891	0,79-1
EPCO	1	0,889	0,79-1
SURLAG	4	0,424	0,25-0,52
GWQMN	0,4	158	0-206
SOL_AWC	0,22	0,94	0,78-1
CN2	82	81	77-90

the results remained, even after calibration, rather weak. As for annual average flow, the simulated values were generally lower than the measured.

In spite of the visual improvements in flow dynamics achieved by the autocalibration process, the fit between the simulated and observed daily flow remained poor when it was assessed by Nash-Sutcliffe (NS) coefficients. Hence, we calculated the NS coefficients for monthly average flows. Then, the best NS value for the project "threshold" was 0.7. When the same set of parameters was used with the project "dominant", the corresponding coefficient was only 0.14. On the one hand, this may be due to less accurate use of land use and soil data. On the other hand, it is possible that the parameter set calibrated for the project "threshold" was not optimal for the project "dominant".

Table 4. Autocalibration results of 6 simulations made for a selected set of parameters with SWAT project "threshold".

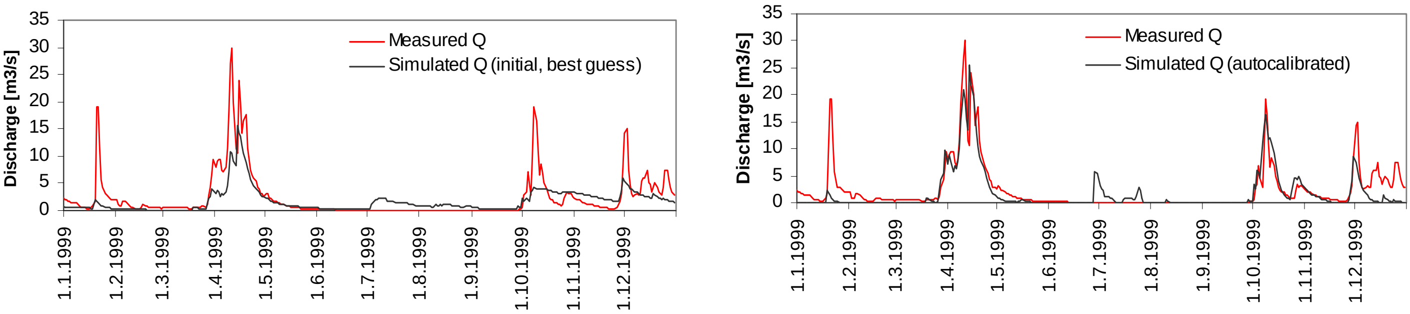

Fig. 3. Measured and modeled discharge at the Vanhakartano measurement station in 1999. The left graph with initial parameters and the right graph with autocalibrated parameters.

In terms of surface runoff and sediment loading from differently cultivated fields, the SWAT output (Fig. 4) was in line with the results reported in Finnish studies (e.g. Puustinen et al. 2005). Also the lower runoff and sediment loss, as well as the higher evapotranspiration, from forest than from agricultural land were results "in the right ballpark". The high spatial variation of sediment loss in the class "Spring cereals" indicates the high erosion risk from highly sloped fields with bare agricultural soil outside of the growing season. Moreover, the use of data from two precipitation gauges and differences in soil type may have increased the variation.

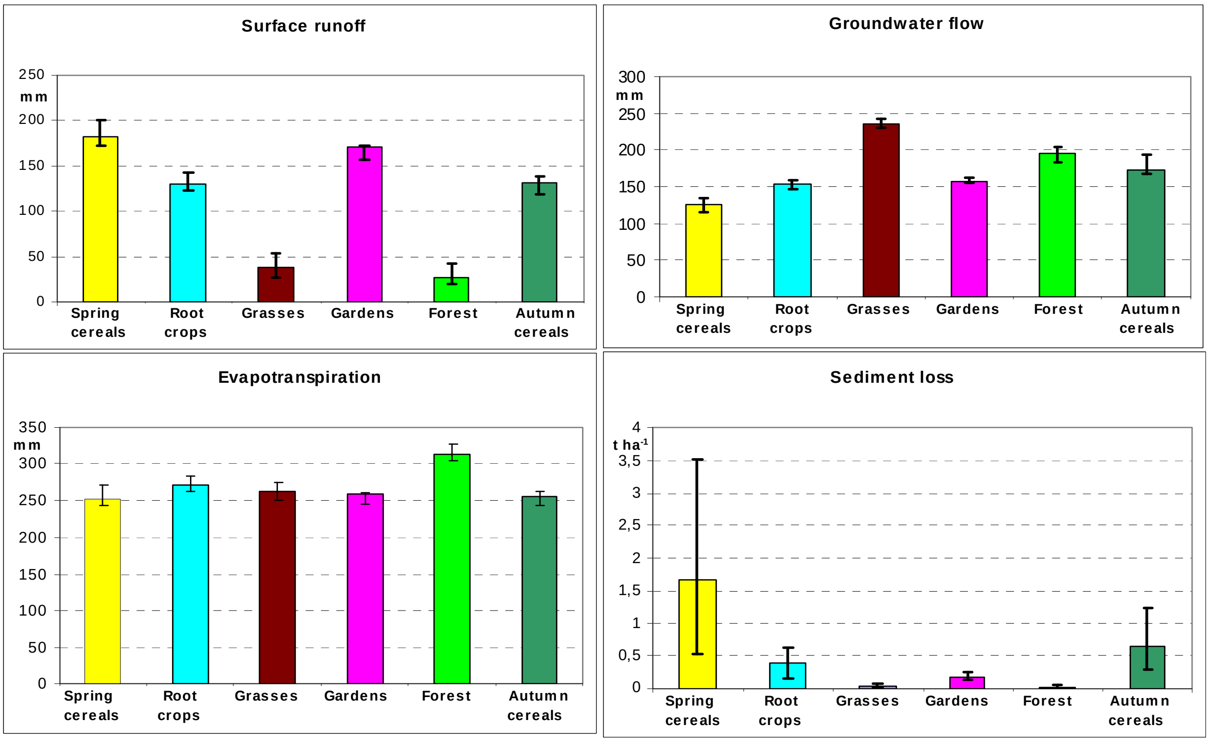

Fig. 4. Surface runoff, groundwater flow, evapotranspiration and sediment loss (5-yr means) in the Yläneenjoki basin according to the SWAT project "threshold". The error bars denote spatial variation in HRUs of the project.

Conclusions and the future work with SWAT

The sensitivity analysis proved to be useful by pointing out the most important parameters for calibration. Condensing the range of parameters probably improved the outcome. However, the condensations perhaps should have been made systematically for all parameters.

Autocalibration tool proved useful by not only producing reasonable parameters but also by saving the modeler's time and systematizing the calibration process. In our case, it seems that it would have been better to keep a uniform set of parameters throughout the process.

The parameterization of the model to achieve satisfactory calibration results in terms of flow and sediment dynamics proved to be a laborious task. Achieving satisfactory fit on daily basis is quite challenging for a long calibration period with varying flow patterns (see Fig. 2). As expected, using the available land use and soil data with finer resolution, as well as utilizing more accurate information on agricultural land use, led to improved calibration result.

Invaluable information about the agricultural management practices used and protective measures implemented in the Yläneenjoki catchment so far, as well as about the measures planned for the future to protect the Lake Pyhäjärvi, is available by cooperative local farmers and authorities. For example, there are 7 CWs already established and many more in planning phase in the catchment. The more detailed agricultural land use data will further improve the spatial accuracy and realism when simulating different cultivation practices. By exploiting the good background information and local knowledge, efforts will be made to make realistic scenarios of the effects of agricultural management actions by using the SWAT model.

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LAND Use Change Effects on River Sediment Yields in Western Greece

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Abstract

The purpose of this paper is to implement the "Soil and Water Assessment Tool" model in order to examine the effects that land use change scenarios have on river sediment yields of the Arachtos catchment in Western Greece. Prior to the scenarios, the particular hydrological and erosion regime of the catchment, representative of the Mediterranean climate, was fully demonstrated. The main anthropogenic intervention in the Arachtos river is a dam construction a few kilometres above the watershed outlet that traps sediment material causing the rapid decrease of the reservoir dead storage capacity and the dramatic reduction of sedimentation rates at the estuary.

SWAT successfully predicted soil losses from different HRUs that caused significant river sediment yields. Facing the problem of reservoir inadequacy in the near future, the study attempted to assess the impact of pre-specified land use change scenarios, in terms of quantifying the results from the application of crop rotations and special cultivation techniques on parts of the agricultural land that was the most susceptible landcover type to erosion. All scenarios resulted in a decrease in soil losses and sediment yields comparing to the current state, with winter wheat cultivation under strip-cropping system causing the highest annual reduction of sedimentation rates in the reservoir.

The model predicted explicitly the consequences of non-structural mitigation measures against erosion sustaining that the understanding of land use changes in relation to its driving factors provides essential information for land use planning and sustainable management of soil resources, under the special Mediterranean conditions of Greece.

KEYWORDS: SWAT, agricultural land, erosion, soil losses, sedimentation rates, sediment yields, land use change scenarios, reservoir.

Introduction

Findings arising by catchment experiments provide clear evidence of the strong relation of erosion rates, land use and human activities (Walling, 1999). Interventions along rivers, such as dams, clearly affect the balance of materials exchanged between the land and the sea that are considered to be a dynamic equilibrium. Changes of the flow and sediment discharges can become influential in the evolution history of the coast (Bonora et al, 2000) as well as the trap of sediments in the reservoirs can rapidly decrease their dead storage capacity. This currently happens in the Arachtos river basin in Western Greece, where the sedimentation patterns of the downstream part of the catchment have been significantly changed, mainly due to dam construction along the river, approximately 20 km upstream of the watershed outlet. Since 1981, when the Pournari I dam became operational, the reduction in sediment supply caused erosion in the coastal areas, mainly adjacent to river mouths, while in the following years the erosion phenomena became more widespread and involved progressively larger coastal segments with the lowest part of the river mouth being significantly retreated (Kapsimalis et al., 2002). In addition, seasonal extreme meteorological phenomena occurring under the Mediterranean climate caused significant soil losses and sediments transport with river flow to the reservoir, while their deposition increased the danger of constituting it operationally inadequate in the near future.

Facing the aforementioned problems, considerable efforts are needed in order to achieve the required improvements in the ecological status of surface waters, primarily focusing on those areas likely to present the greatest risk. Agricultural land that is usually very susceptible in soil losses is considered to be the most favorable landcover type where low-cost mitigation measures against erosion can be applied. Land use changes which are biophysically, or more commonly in the last years, artificially based (Skole and Tucker, 1993), often have significant effects on the surrounding environment and consequently on the hydrological cycle. Although the empirical knowledge of the consequences of a land use change is generally common (i.e. crops cultivation under rotations, strip-cropping, contours or terrace systems can decrease soil loss and sediments discharge), it is often very difficult to make an explicit quantification of these consequences.

Towards this end, this study presents a method for quantifying the impacts from specific landuse changes, on soil losses of basins, by examining the case study of the Arachtos river basin in Epirus, Greece. Based on our knowledge in using the AVSWAT2000 version of Soil and Water Assessment Tool model (Arnold et al., 1998) for different purposes (Mimikou, 2000 et al., Varanou et al., 2002, Panagopoulos et al., 2007), we modeled the catchment trying to come to reliable conclusions regarding the implementation of erosion restriction measures on catchment scale. The importance of land uses in the SWAT simulation of erosion and sediments transport lies mainly in the computation of surface runoff with the help of the SCS curve and of soil losses with USLE_C and USLE_P parameters of the MUSLE equation that refer to the landcover type and its support practice respectively (Neitsch et al., 2001).

Methodology

Study area

The Arachtos river basin (figure 1) is located in the western part of Greece. The river drains into the Amvrakikos gulf, a semi-closed marine area of 405 km^2, connected to the Ionian Sea to the West through a narrow natural channel. The climate is of Mediterranean type with temperate winters, high rainfall and sun exposure with the mean annual temperature and rainfall depth being 15^{o}C and 1500 mm respectively. The watershed has an area of 2000 km^2 and is predominantly agricultural land (arable and pasture). The elevation range is 0 to 2400 m with the mean elevation being 785 m. The length of the main stream is 110 km. High rainfall events in combination with the impermeable soil formations cause significant runoff and subsequently high soil losses and river sediment yields. Parts of the watercourse are influenced by man-made interventions, such as the Pournari I dam, which was constructed in the early 80’s, located in the homonymous region 3 km upstream of the Arta city (fig. 1). The production of hydropower in the peaks of electric energy demand, the storage of water for irrigation use in the Arta plain and the flood protection constitute the multi-scope of this construction that has a total storage capacity of 865*10^6 m^3 and became operational in 1981.

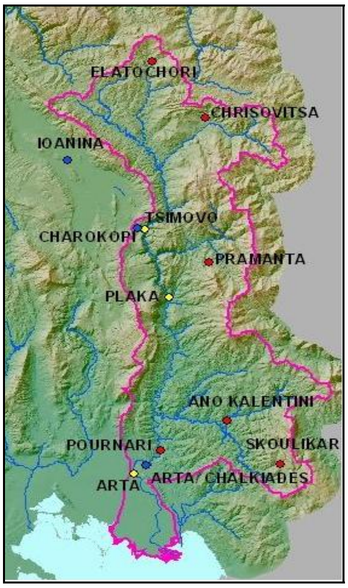

Figure 1. Main locations in the Arachtos river basin.

Model parameterization

Based on the DEM of 50 x 50 m resolution (fig. 2a), the model delineated the watershed and subdivided it in several smaller subbasins. For modeling purposes, ten subbasin outlets were finally selected to be active (fig. 2d), based on the spatial differentiation of meteorological information and on the location of monitoring stations with available flow and sediment measurements. The landcover map (fig. 2b) was obtained from the European CORINE project, mainly representing forest (FRST) occupying the 30% of the total area and agricultural land (60%) that was distinct into pastureland (WPAS) and arable land. The latter consisted of vine (VINE), rice fields (RICE), orchard trees (ORCH) and predominantly of agricultural land of row crops (AGRR), that mainly included corn cultivation. Geological formations (fig. 2c) consisted of flysch deposits that mainly covered the northern part of the catchment, karstic systems of limestones that were encountered in the central part, sandstones, which were impermeable geological formations and alluvial that have been transported and deposited in the Arta plain. Meteorological data of daily rainfall, air temperature, net solar radiation, wind speed and humidity, collected from various stations around the catchment (fig. 2d) were also inserted, covering the period from 1964 to 2003. For the estimation of potential evapotranspiration, we used the Penman-Monteith method.

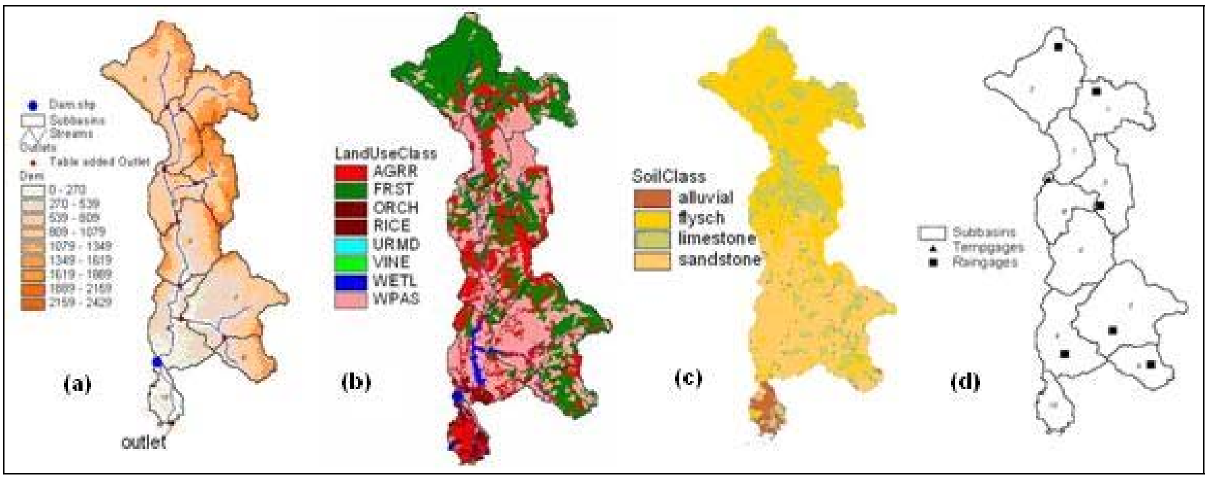

Figure 2. Representation of (a) topography, (b) landcover types, (c) geological formations and (d) watershed delineation and meteorological stations in the catchment.

Model calibration for flows and sediment yields

The available measurements of flows and sediments were used for comparison with the predicted results in order to test the SWAT simulation efficiency. Hydrological calibration was carried out by adjusting groundwater and soil properties. Sediments calibration was achieved by mainly adjusting the concentration of sediments in subsurface flow and the K_{USLE} factor of the MUSLE equation (Neitsch et al, 2001). Calibration took place in annual and monthly basis at three sites named Tsimovo, Plaka and Arta (fig.1), where measures of flow (1965-2003) and sediments (1965-1975) existed. Figure 3 represents the graphical comparison between predicted and observed flows and sediment yields in Arta.

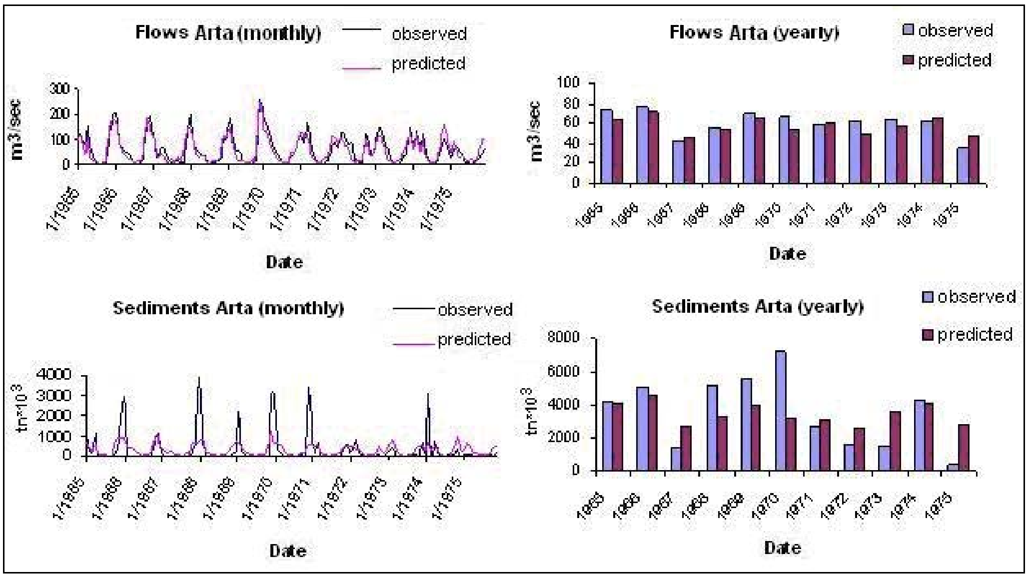

Figure 3. Comparison between observed and predicted flows and sediment yields in Arta.

River sediment yields were estimated primarily by quantifying soil losses from HRU's with the Modified Universal Soil Loss Equation (MUSLE) (Neitsch et al, 2001). The values of MUSLE factors for the main landcover and soil types of the Arachtos catchment that were finally determined after calibration are presented in table 1.

Table 1. Calibration values of MUSLE factors determining soil losses.

MUSLE coef / Landcover type	Forest (FRST)	Arable land (AGRR)	Pasture (WPAST)	Orchard (ORCH)
C_{USLE}	0.001	0.2	0.003	0.001
P_{USLE}	1.00	1.00	1.00	1.00
MUSLE coef / Soil type	Alluvial	Sandstones	Flysch	Limestones
K_{USLE}	0.10	0.20	0.12	0.15
CFRG	10%	10%	10%	10%

K_{USLE} is the USLE soil erodibility factor, C_{USLE} is the USLE cover and management factor, P_{USLE} is the USLE support practice factor and *CFRG* is the coarse fragment factor (Neitsch et al, 2001). The C_{USLE} coefficient is the initial value assigned by SWAT to each landcover type and is updated daily during the simulation according to the growth cycle of the plant and its canopy development. P_{USLE}, is defined as the ratio of soil loss with a specific support practice to the corresponding loss with up-and-down slope culture (Neitsch et al, 2001). Because no support practices existed in the Arachtos catchment, P_{USLE} coefficient took its default and maximum value (1.00). Finally, due to the lack of sufficient soil studies in the catchment, K_{USLE} was adjusted during calibration, while for *CFRG* factor the default percentage of soil content 10% was used for all geological formations.

Model results at the current state

The mean annual precipitated water of the simulation period (1964-2003) approached 1500mm in the Arachtos catchment, and was converted into 450 mm of actual evapotranspiration, 150 mm of percolated water to the deep aquifer and 900 mm of annual runoff. During the period 1964-1980 of no structural interventions along the river, when flow and sedimentation rates occurred physically in Arta, the average annual water flow and sediment yield was estimated 61 $m^3\ s^{-1}$ and 3,80 x 10^6 tn y^{-1} respectively with the highest peaks occurring mainly in December. These estimations could also be considered representative for the Pournari dam location, 3 km upstream of Arta (fig. 1). Although seasonal flow patterns in Arta after the reservoir operation starting (period 1981-2003) were not significantly changed (Panagopoulos and Mimikou, 2006) due to the regulated water amounts released from the dam for hydropower production, the mean annual sedimentation rate was dramatically reduced to 0.064 x 10^6 tonnes, a reduction of 98.5% (Panagopoulos and Mimikou, 2006), This percentage also represented the dam trap efficiency. These conclusions were strengthened by findings from Syvitski et al. (2002). We applied their methodology making an estimation of 3,73 Mtn y^{-1} of long-term flux of sediments in Arta and of 93% dam trap efficiency, very close to the predicted by SWAT respective results. As for SWAT mean monthly predictions regarding physically routed flows and sediment yields in Arta (1964-1980), these are presented in figure 4(a), in parallel with catchment precipitation. Runoff and sedimentsation rates were maximized during winter months when the maximum precipitation depth also occurred. Figure 4(b) summarises the non-uniform in time hydrological regime of the Arachtos catchment, governed by the special characteristics of the Mediterranean climate. During the wet period (October-March) 76% of the mean annual precipitation occured causing 80% and 82% of the mean annual water and sediment yields in Arta.

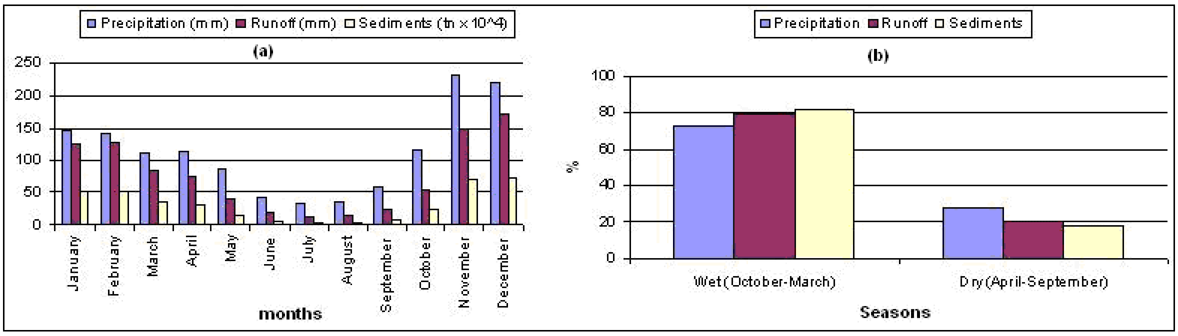

Figure 4. Representation of the (a) mean monthly precipitation, runoff and sediment yields in Arta and (b) their relative occurrence during the wet and dry period.

As indicated in table 1, C_{USLE} and K_{USLE} of the MUSLE equation were the major factors that governed the erosion susceptibility of the different landcover and soil types respectively. Their annual contribution to soil losses to the river in respect to precipitation received and runoff generated is presented in figure 5. Arable land of row crops (AGRR) generated the highest soil losses in the catchment, 3100 tn m^{-2}, while forest (FRST), orchard trees (ORCH) and pasture (WPAS) caused significant lower soil loss and subsequently contributed much less to the river sediment transport as lower values of C_{USLE} were assigned to them (table 1). On the other hand, sandstone was the most susceptible geological formation in soil loss processes. According to the results, the mean annual erosion rate of sandstones was 3400 tn km^{-2}, while the respective losses originating from the rest three formations ranged between 1000-1500 tn km^{-2}, something that was attributed to their K_{USLE} values adjusted for calibration purposes (table 1).

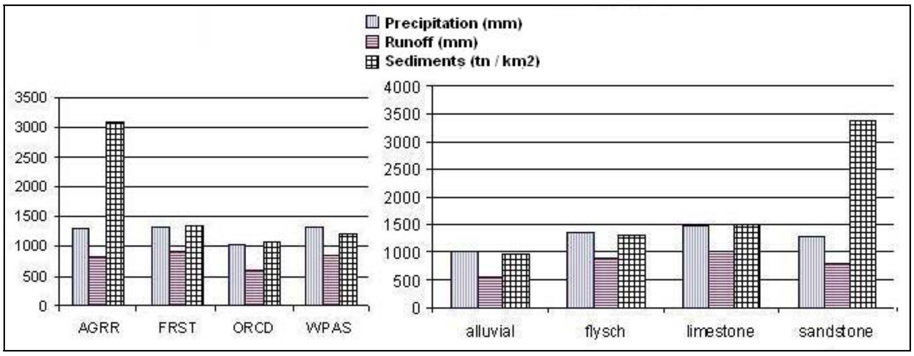

Figure 5. Mean annual soil losses originating from different land cover and geological types.

Land use change scenarios

High sedimentation rates and dam trap efficiency demonstrated the need for the application of mitigation measures against erosion in the catchment. Thus, we tried to assess the impact that land use changes had on sediments discharge, in terms of quantifying the results from pre-specified scenarios related to the application of special management practices on parts of the highly susceptible in erosion agricultural land where such scenarios were desirable and feasible because of their low cost and non-structural character.

The process of building an agricultural management scenario in SWAT is the alteration of the management (mgt) files related to the HRU's that have agricultural land as landcover type. In figure 2(b) AGRR represented the part of the arable land (almost the 30% of the agricultural land and 20% of the catchment area) that was subject of the scenario applications. As this part of the catchment was covered by row crops, especially corn, it was the most appropriate for the application of realistic scenarios, in contrast to the other arable land types of permanent cultivations (e.g orchard trees, vines) where no realistic scenarios of land use change could be easily applied. The re-creation of the mgt files was made according to the theoretical base of SWAT (Neitsch et al, 2001). The model was executed for the period 1964-1980, by keeping the same set of all other model parameters of the original calibration, giving river discharge outputs in Arta. These outputs were then compared to the respective ones of the base run with the purpose of estimating the decrease of sediments transport to the reservoir, thus estimating the percentages of discharge change for every scenario.

Land use changes referred to the application of crop rotations and special cultivated techniques. Three rotations consisting of corn-wheat, corn-hay and wheat-hay were firstly examined. More specifically, the two years rotation of corn-wheat cultivation started with corn sowing on 10 April, harvest on 10 September, a tillage at 10 mm depth of the soil profile in the middle September and was finalized by wheat sowing at the end of the same month with harvest at the end of August of the second year. The second four-year rotation scenario also consisted of corn cultivation with the difference that three years of hay cultivation followed it. The third one was similar to the second but with winter wheat being the crop in the four-year rotation. The next step was to examine the less susceptible pair in erosion under the support practices of contours, strip-cropping on the contour and terraces.

Results and discussion

The land cover and management factor C_{USLE} as well as the support practice factor P_{USLE} were the two components of the M_{USLE} equation that reflected the land use change information in the SWAT code in order to make the new calculations. Crop rotations and different cultivation techniques resulted respectively in the modification of the aforementioned indices that subsequently caused changes in soil losses generated by each HRU. C_{USLE} factor was automatically modified according to the assigned respective value in the landcover/plantgrowth database of SWAT, while P_{USLE} was modified according to the basis described in SWAT theoretical documentation for land slopes between 1% and 2%. Table 2 summarises the values that the two aforementioned factors received in each scenario.

Table 2. C_{USLE} and P_{USLE} values of agricultural mgt files under six scenarios and the base run.

	Base run	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5	Scenari
Crop rotation	AGRR - no rotation	corn - wwht	corn - hay	wwht - hay	wwht - hay	wwht	wwht - l
ivation -Technique	none	none	none	none	contours	terraces	strip-crop
C_{USLE}	0.2	0.2 - 0.03	0.2 - 0.003	0.03 - 0.003	0.03 - 0.003	0.03 - 0.003	0.03 - 0.
P_{USLE}	1.00	1.00	1.00	1.00	0.60	0.50	0.30

Mean annual sediment yields and the percentage of annual change in the river yield caused by each scenario are presented in figures 6(a) and 6(b) respectively.

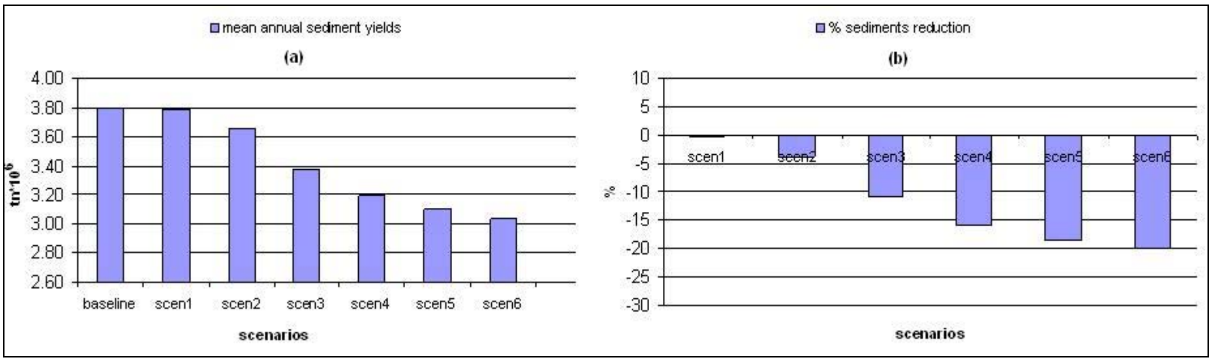

Figure 6. Mean annual sediment discharges in the reservoir under the existing situation and six different land use change scenarios (a) and the percentage of change in annual yield caused by each scenario (b).

From the above figure it can be firstly concluded that the effects from the rotated cultivation of corn and winter wheat (scen. 1) can be nearly neglected. Although winter wheat reduced the C_{USLE} coefficient, the presence of corn, even in only one of the two years rotation, was restrictive for sediments reduction. On the other hand, winter wheat and hay (scen. 3) was the most effective rotation in decreasing soil losses from the arable land as the annual sedimentation rate of 3.8 Mtn y^{-1} of the baseline was reduced to 3.38 Mtn y^{-1}, a reduction of 11%. This change can be firstly attributed to the reduced C_{USLE} factors of winter wheat and hay in relation to this of the agricultural land (AGRR) that existed during the baseline and secondly to the reduced surface runoff generated by hay, as for that type of landcover, lower CN values were also assigned by the model (Neitsch et al, 2001). Corn-hay cultivation (scen. 2) also resulted in an intermediate reduction of almost 4%. Further examination of the most efficient scenario (scen. 3) on sediments reduction, with the incorporation of specific support practices significantly improved the results. The rotation of winter wheat and hay on contour tillage and planting cultivation reduced annual sedimentation rates by 16% due to the decreased P_{USLE} factor that received the value of 0.6. When winter wheat was cultivated under strip-cropping on the contour (scen. 6), meaning that strips of winter wheat were alternated with equal-width contoured strips of sod, the P_{USLE} factor was minimized, receiving the value of 0.30 that resulted in the greatest reduction (20%) in annual river sediments yield (3.04 Mtn y^{-1}). When winter wheat was cultivated in terraced fields (scen. 5), a significant reduction of 18.5% in sediment yields also occurred as we reduced the P_{USLE} factor to the value of 0.50.

CONCLUSION AND OUTLOOK

Soil losses from different geographical units of the Arachtos catchment were sufficiently quantified resulting in a significant annual average sedimentation rate of 3.80 Mtn yr^{-1} at the Pournari I dam location. Land use change scenarios based on application of crop-rotations and support practices on parts of the agricultural land seemed to be efficient mitigation measures against erosion. Although financial and social consequences of these changes were not taken into consideration in this study, the results strongly suggested the incorporation of hay cultivation in the arable land of the catchment. Moreover, the cultivation of hay and winter wheat under the strip-cropping support practice resulted in the highest annual reduction in sediment yields in the reservoir, a percentage that in combination with other mitigation options related to reservoir management can prolong the project life with operational, economical and environmental benefits. The conclusion drawn from the paper also stimulates public institutions to define and develop guidelines for the defence of land degradation by preferably applying non-structural and low-cost measures instead of adopting other very hard policy choices like the opening of dams. Further monitoring, study and experimentation concerning erosion in the Mediterranean areas are of paramount importance.

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Sediment Yield Modelling Using SWAT model at Larger and Complex Catchment: Issues and Approaches. A Case of Pangani River Catchment, Tanzania

by

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ABSTRACT

A semi-distributed, physics based watershed model, Soil and Water Assessment Tool (SWAT) was used to model sediment yield in a larger and complex catchment where the dominant erosion process is sheet erosion. Based on adequate data of subdaily and daily sediment flow and streamflow at the study area an excellent suspended sediment-rating curve has been established and verified. Since, the ongoing study intends to predict long-term sedimentation rate and the remaining economic life of the reservoir downstream, the lumping nature and assumptions of stationarity and linearity of the rating relationship are not desired. Therefore, a more process based and distributed tool was required. The runoff component of the SWAT model was calibrated from six years of streamflow data, whereas the sediment component of the model was calibrated using daily sediment flow data from one hydrological year. A long-term period over 37 years simulation results of the SWAT model was validated to downstream Nyumba ya Mungu reservoir sediment accumulation information.

The SWAT model captured 56 percent of the variance of the observed daily sediment loads during calibration period. In the latter period the model underestimated the observed sediment load by 0.9 percent. Also, the model has identified erosion sources spatially and has replicated some erosion processes as determined from indirect methods, fingerprinting techniques and field observations. The predicted and measured long-term sediment yields are comparable with a relative error of 2.6 percent. This result suggests that for catchments where sheet erosion is dominant SWAT model is a better substitute of the sediment-rating curve and long-term prediction of sedimentation rate can be done with reasonable accuracy. It should be noted that the calibration was done during the normal wet year when most of hydrometeorological data required for SWAT model is available.

KEYWORDS: Sediment Yield Modelling, Sediment rating curve, SWAT, Pangani River Catchment, Tanzania.

INTRODUCTION

Sediment yield refers to the amount of sediment exported by a basin over a period of time, which is also the amount, which will enter a reservoir located at the downstream limit of the basin (Morris and Fan, 1998). The subject of sediment yield modelling has attracted the attention of many scientists but lack of resources and compelling methods to predict sediment yields are some of the deadlocks towards this direction (Ndomba, 2007). The sediment flow data over a decade and periodic reservoir survey information are some resources demanding methods for estimating sediment yield rates at a catchment level. Besides, other workers such as Wasson (2002) have noted the transferability problem of plot or micro scale studies results to larger catchment. Others have also cautioned that long term sediment monitoring of suspended sediment loads does not necessarily give better results (Summer *et al*., 1992). A number of workers have suggested that an excellent sediment-rating curve could be constructed from detailed sediment flow data of single flood events (Summer *et al*., 1992). However, Ferguson (1986) indicated that most of the sediment-rating curves underestimate the actual loads. Once load bias correction factor is applied the rating relationship has been reported to estimate the actual load reasonably (Ferguson, 1986). In the ongoing research in Pangani river basin, Ndomba(2007) has compared most of the available types of rating relationships and correction factors and he found that rating developed from all data points and corrected based on actual load is a better approach when mean daily streamflow data is used. Many workers such as Bogen and Bønsnes (2003) have cautioned that such relationships should be used on catchment where no significant landforms, landuse and sediment supply sources changes are expected. In this study the authors believe that the lumping nature, stationarity and linearity problems of the rating curve could be avoided by replacing it by distributed and process based sediment yield models. Physically based, spatially distributed modelling systems have particular advantages for the study of basin change impacts and applications to basins with limited records (Bathurst, 2002). Their parameters have a physical meaning and can be measured in the field and therefore model validation can be concluded on the basis of a short field survey and a short time series of meteorological and hydrological data (Bathurst, 2002). As noted by Ndomba(2007) that not all rainfalls in the Pangani basin yield sediments and that sediment sources are only localized areas in the catchment. This suggests that sediment yield varies temporally and spatially. It is envisaged that once the model is calibrated various landuse changes scenarios can be incorporated into the model and hence avoiding the stationarity assumption, for instance. It should be noted here that a number of distributed and physically based sediment yield models already developed by other workers are candidate tools for such studies.

The sediment yield model that is used in this study is Soil and Water Assessment Tool (SWAT). The SWAT model was originally developed by United States Department of Agriculture Research Service (USDA-ARS) to predict the impact of land management practices on water, sediment, and agricultural chemical yields in large ungauged basins (Arnold *et al*., 1995). SWAT model has a long time modelling experience since it incorporates features of several (ARS) models (Neitsch *et al*., 2002). Erosion and sediment yield are estimated for each Hydrologic Response Unit (HRU) with the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1975). The runoff component of the SWAT model supplies estimates of runoff volume and peak runoff rate, which, with the subbasin area, are used to calculate the runoff erosive energy variable. The crop management factor is recalculated every day that runoff occurs. It is a function of above ground biomass, residue on the soil surface, and the minimum C factor for the plant (Neitsch *et al*., 2002). Other factors of the erosion equation are evaluated as described by (Neitsch *et al*., 2002). The current version of SWAT model uses simplified stream power equation of Bagnold’s (1977) to route sediment in the channel. The maximum amount of sediment that can be transported from a reach segment is a function of the peak channel velocity. Sediment transport in the channel network is a function of two processes, degradation and aggradations(i.e. deposition), operating simultaneously in the reach (Neitsch *et al*., 2002).

Apart from the capability of the model as discussed above, some workers such as Ndomba *et al*., (2005) have successfully applied SWAT model for sediment yield modeling in ungauged catchments in Tanzania. This paper reports the application of SWAT model in a well-studied catchment (i.e. with intensive data on sediment flow in fluvial system and sediment accumulation information at the downstream reservoir, Nyumba ya Mungu, is available). However, it should be noted that in the ongoing study the suitability of short-term sediment flow data for calibrating parameter intensive sediment yield models is being investigated and its performance is validated with long-term (i.e. 37 years) reservoir sediment accumulation information at the outlet of the catchment. This approach is adopted in this study because other workers such as Morris and Fan (1998) have noted that reservoir survey data generally represent a more reliable measure of the long-term basin sediment yield. Based on the knowledge of the authors little has been done towards this direction. But the authors believe that comparison with long-term reservoir sediment accumulation will measure the robustness of the proposed approach. This study intends to reduce to some degree the uncertainty of parameters estimation of the watershed sediment yield model by calibrating the model in a normal hydrological wet year with most of hydro-climatic data required by the model available. As noted by Yapo, *et al*., (1996) that wet years when used for model calibration result into reliable and optimum parameters set.

DESCRIPTION OF THE STUDY AREA

The study area covers the upstream of Nyumba ya Mungu dam (NyM) in Pangani River basin (Figure 1). The catchment of NyM occupies a total land and water area of about 9000 km^2. It is located in the North Eastern part of Tanzania between Latitudes 3^{o}00'00" and 4^{o}3'50" South, and Longitudes 36^{o}20'00" and 38^{o}00'00" East.

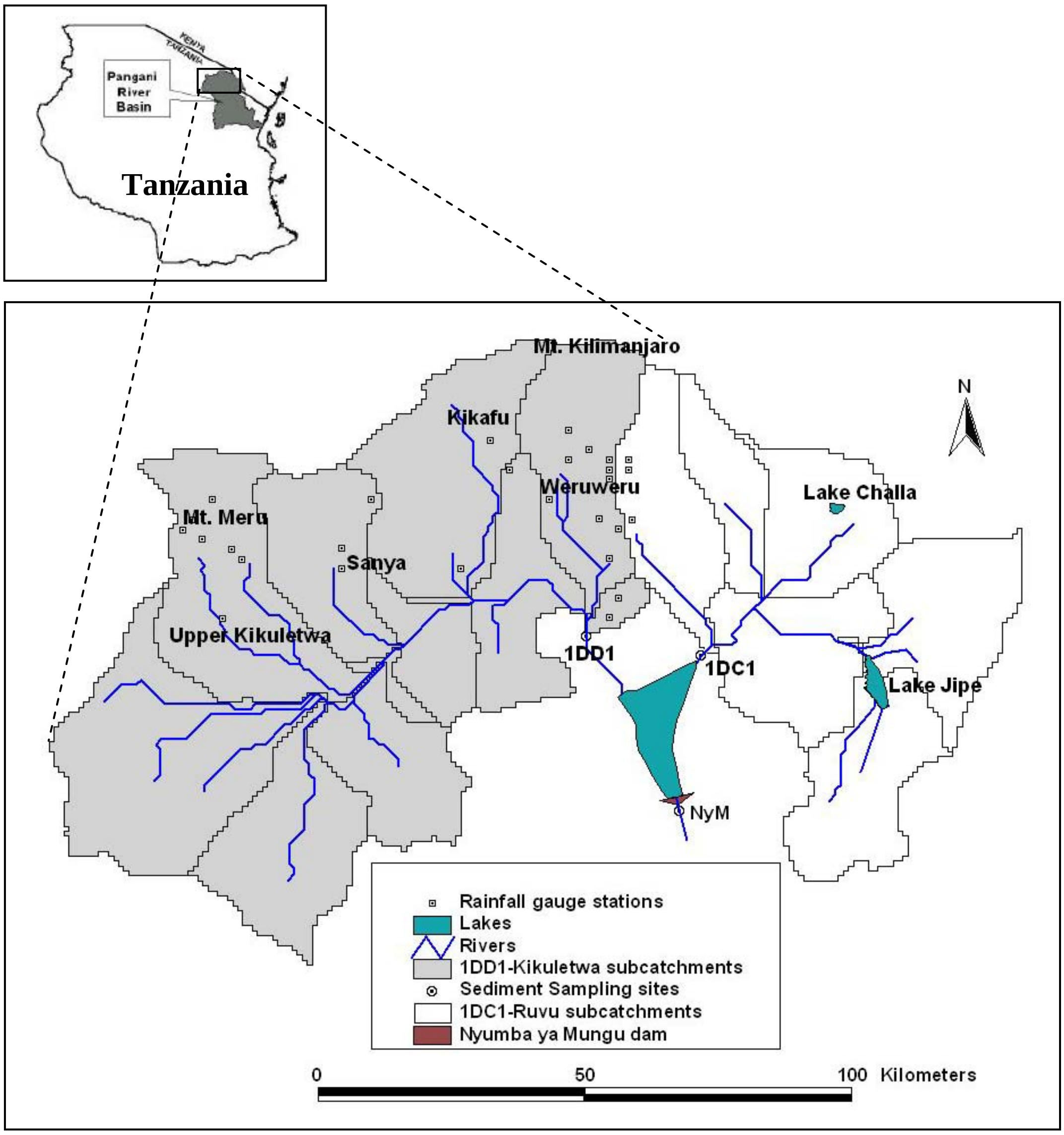


Figure 1. A location map of Pangani river basin, upstream of Nyumba ya Mungu dam.

The Pangani River has two main tributaries (Figure 1), the Kikuletwa (1DD1) and the Ruvu (1DC1), which join at Nyumba ya Mungu, a reservoir of some 140 km^2. These are perennial rivers with mean discharges of about $11 m^3/s$ (1DD1) and $3 m^3/s$ (1DC1) during low flows. The catchment comprises of complex geological formations such as North Pare Mountains, Mount Kilimanjaro, and Mount Meru. The geology of region is mainly of Neogene Volcanic and pre-Cambrian metamorphic rocks, which are extensively covered by superficial Neogene deposits including calcareous tuffaceosus materials, derived from the Kilimanjaro volcanic and the deposits around Lake Jipe. Based on the Soil Atlas of Tanzania, the main soil type in the upper Pangani is clay with good drainage. Actively induced vegetation, forest, bushland and thickets with some alpine desert chiefly characterizing the land cover of the catchment. The altitude in the study area ranges between 700m and 5825m a.m.s.l. The ice cap of Mount Kilimanjaro forms a highest ground in the catchment.

Rainfall is bimodal in these areas, peaking between March and May, with a smaller peak between October and November. Rainfall in the former season may exceed 600 mm a month, and 300 mm in the latter. The highland area is considered to be that land lying about 900 m, around the slopes of Mounts Meru and Kilimanjaro and Pare Mountains. High levels of precipitation can be found in the southern slopes of the mountain areas with annual precipitation of 1000-2000mm/year. Recent findings by Rohr and Killingtveit (2003) indicated that the maximum precipitation on the southern hillside of mount Kilimanjaro takes place at about 2,200 m.a.s.l., which is 400-500m higher than assumed previously. Low land areas towards Arusha Chini and Nyumba ya Mungu dam receive an average annual rainfall of 600-1200mm. Seasonal variation of temperature in the basin ranges from 14^{o}C to 25^{o}C. Maximum and minimum temperatures occur between March and July respectively.

The majority of the population in the basin depends on irrigated agriculture directly or indirectly. Agriculture is concentrated in the highlands, while the lowlands are better suited for pastoralism. The basin is also important for hydropower generation, which is connected to the national grid. Hydropower plants, which are downstream of Nyumba ya Mungu Reservoir (NyM) are Nyumba ya Mungu (8MW), Hale (21MW), and New Pangani falls (66MW).

METHODOLOGY

It should be noted that the sediment sampling network as initiated by Ndomba(2007) in the study area was limited in terms of spatial coverage. In order to estimate sediment yields and identify sediment sources from individual subbasin multi-sampling sites would have to be installed in each tributary river in the basin. Though the approach seems to be scientifically attractive, practically it is impossible to implement. A compromise lies then to complementing the sampling programme findings by applying distributed sediment yield or erosion model to simulate the erosion processes and yields spatially.

A SWAT model was applied in a bigger subcatchment called 1DD1-Kikuletwa (Figure 1). The subbasin is known to contribute 97.5 percent of total sediment inflow load into the NyM reservoir and the 1DC1-Ruvu-subcatchment-sediment load is only 2.6 percent of the Kikuletwa subcatchment one (Ndomba, 2007). The proportions of sediment delivery between the two subcatchments were established based on one year intensive sampling programme between March, 2005 and January, 2006. Since these River tributaries have the same stream order, therefore, this study assumes that they hydrologically respond in a similar manner seasonally and inter-annually.

It should be noted that Ruvu subcatchment is complex and the required data that are hydrometeorological and physical data of upstream natural lakes such as Jipe for SWAT model setup is missing. Therefore, long-term total sediment yield for Ruvu subcatchment was estimated as 2.6 percent of the long-term total simulated sediment yield of 1DD1 catchment. The runoff component of the SWAT model as calibrated by Ndomba (2007) was used for sediment yield modelling in the basin. Sensitivity analysis tools (Van Griensven *et al*., 2002) were used in order to identify important parameters that govern sediment yield and routing phases. Their ranking guided the calibration exercise. For example, the estimation method of the most sensitive parameters was critically reviewed. Both manual and Autocalibration approaches were used to train the sensitive model parameters. Additionally, some parameters that affect peak runoff rate and indirectly sediment yield and transport were calibrated. It should be noted here that all the channel sediment routing parameters were calibrated. A proposed equation for soil erodibility for tropical conditions by Mulengera (1999) was used to estimate USLE_K. Some sensitive parameters such as practice factor, USLE_P and cropping factor, USLE_C were not calibrated instead the default values were used. The default parameters as estimated by the model seem realistic and are comparable to field observations. For instance, the USLE_P value of 1 is comparable to the absence of soil conservation programmes in the study area (Mtalo and Ndomba, 2002).

The model was calibrated for one hydrological year that is between 1 November 1977 and 31 October 1978. The period falls within a normal wet year because other workers such as Yapo, *et al*., (1996) have indicated that the variability of parameter estimates increases with ‘dryness’ of the data. In this study wet year is defined as that year with total annual rainfall near or above the long-term mean annual rainfall. It should be noted here that a period between year 2005 and 2006, during sediment sampling programme, was not used for model calibration because the catchment experienced a critical hydrological condition (i.e. drought). Alternatively, an excellent sediment rating curve had to be developed, verified and used to generate sediment loads to other periods as reported by Ndomba(2007) for calibration and validation purposes. And the simulation of calibrated model was validated to long-term period (i.e. between 1 January 1969 and 31 December 2005) sediment accumulation in the downstream reservoir. This period corresponds to the age of the reservoir. Besides, the modeling exercise entailed estimating erosion rates spatially. The spatial soil loss pattern was validated with findings from others studies (Mtalo and Ndomba, 2002; Ndomba, 2007).

RESULTS AND DISCUSSIONS

Calibration results in daily and monthly time steps are presented in Figures 2 through 4 below. In Figure 2 one would note that the rising and falling of a big flood event are reasonably simulated. A few sporadic sediment spikes are also evident. The streamflow and sediment transport in the study area are characterized as highly variable within a day (Ndomba, 2007). It should be noted that the mean streamflows used to compute observed sediment loads are derived from two to three manual gauge heights measurements during day time while the SWAT model computes mean stream flow for each day. Sediment loads in the recession of the medium flood events such as that of December, are over-predicted.

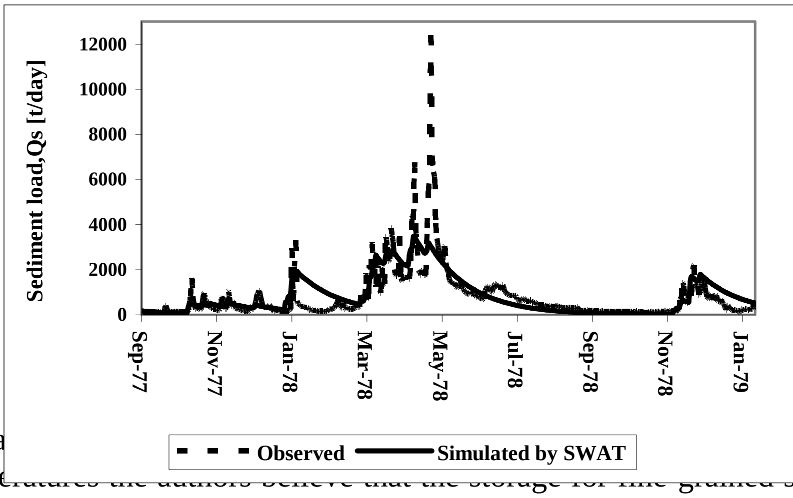


Figure 2. A comparison between Observed and simulated daily sediment loads at 1DD1 sampling station, in Pangani river basin during calibration period (R^2=56% and TMC=0.9%).

Ba ... ngle hydrological events and literatures the authors believe that the storage for fine grained sediment in the main tributary river channels isn’t much. Therefore, model deficiency here may be a better explanation.

Besides, other workers have successfully used this model at monthly time steps [Schmidt and Volk, 2005; Van Liew, *et al*., 2005]. The result from this study at monthly time step is also sound as demonstrated in Figures 3 and 4 below. However, the deficiency of the model to simulate the sediment load in the falling limb for the medium floods as noted above is also reflected in monthly time step in the same periods (Figure 3). However, the performance of the model in simulating monthly total sediment loads is excellent (R^2=86%) as Figure 4 illustrates. This result suggests that as time step of simulation increases the SWAT model performance improves and therefore, its application for long-term (i.e. about 37 years) simulation using annual time step is also justified. It should be noted also that the size of the smallest subbasin or HRU used for this study is greater than 1000km^2 (Table 3). As Schmidt and Volk (2005) indicated in their study in Western Germany that the SWAT model efficiency increases with the size of the catchments with a break point at a basin area size of approximately 300 to 500km^2. This is compounded by the insufficient capability of SWAT to simulate the process dynamics in small catchments. However, Schmidt and Volk (2005) recommended the use of the highest scale-adequate input data resolution available for predicting high-resolution dynamics and process quantification.

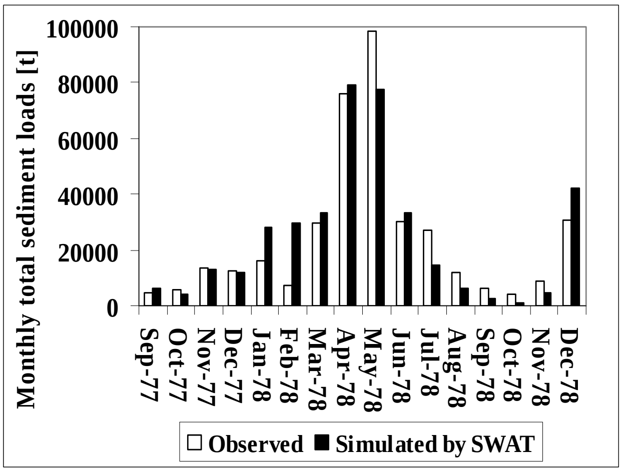


Figure 3. A comparison between Observed and simulated Monthly totals sediment loads at 1DD1 sampling station during calibration period (TMC=0.9%).

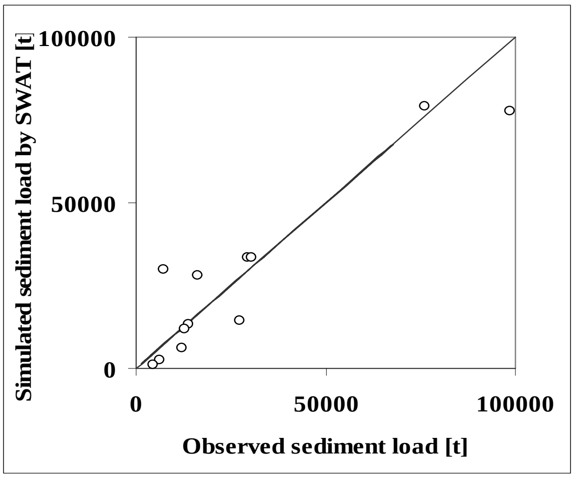
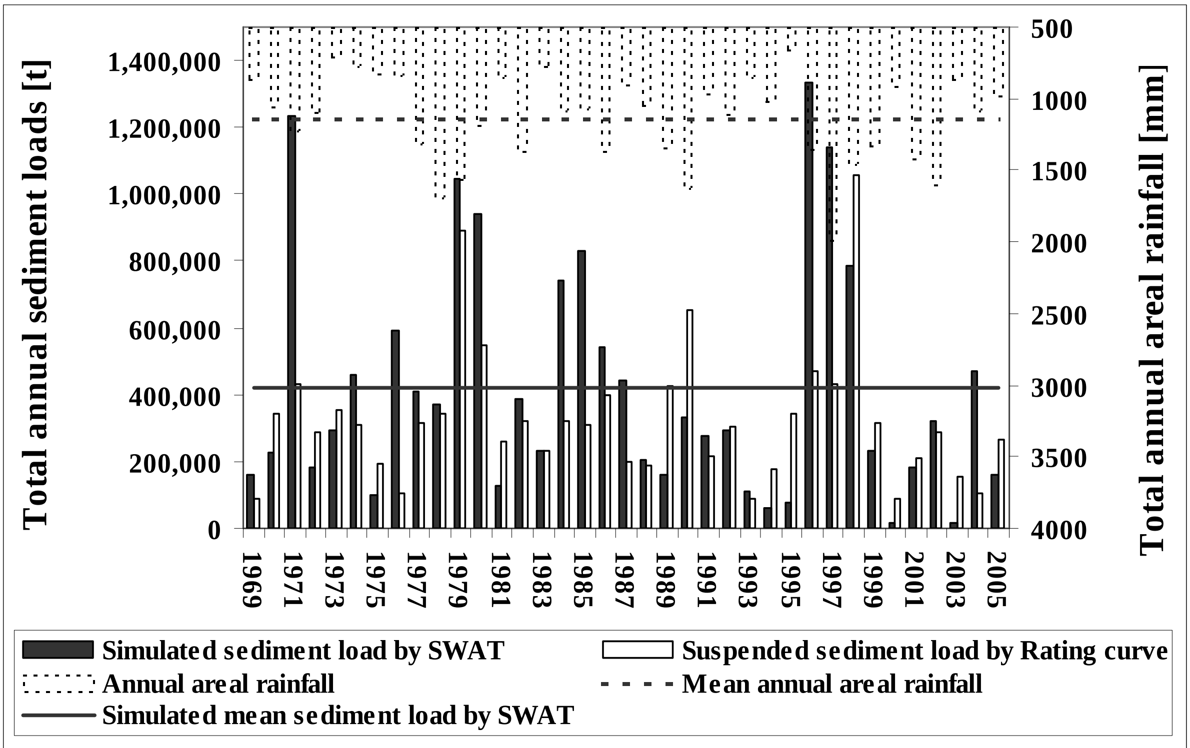


Figure 4: Scatter diagram of total monthly sediment loads for calibration period between November 1977 and October 1978 (R^2=86%).

Long-term simulation result in Figure 5 below generally indicates that estimated and observed (i.e. based on suspended sediment rating) annual total sediment loads are comparable. However, according to Total Mass balance controller (TMC) as objective function the simulated loads overestimates the observed by 28.7%. Similarly, the authors expected such a discrepancy because SWAT model simulates bed-material load (i.e. bed and suspended load) while the rating curve computes only suspended sediment load. It should be noted here that this rating curve was used by Ndomba (2007) to estimate the long-term sedimentation rate in the Nyumba ya Mungu reservoir and gave reasonably result with relative error of 20.3 percent. With the exception of a few cases Figure 5 below, indicates that wet years seem to transport much of sediment loads. One would note also that problems of



linearity as assumed in the rating curve concept are well demonstrated between 1979 and 1999 period, where sediment loads increase with rainfall and vice versa. On the other hand SWAT model results suggest that not all rainfalls yield the same rates of sediment. The latter observations are supported by Ndomba (2007) work's findings from using indirect method techniques of sediment sources and processes identification. The inter-annual variability of sediment loads as depicted in Figure 5 suggest that sediment sources from upland catchments are not exhausted. This is explained by the fact that fine-grained soils on the mountain slopes as the case for Pangani basin are always available for transportation. However, Ndomba(2007) found that sediments get exhausted seasonally because of high content of clay in the soils of the sediment source areas.

Figure 5. Comparison between simulated (i.e. by SWAT) and observed (i.e based on suspended sediment rating) total annual loads between 1969 and 2005 (TMC=28.7%).

The long term (i.e. 37 years) simulation by SWAT model was used to estimate total sediment yield and long term sediment yield rate for 1DD1-Kikuletwa catchment. The long term predicted total sediment yield and annual sediment yield rate for 1DD1-Kikuletwa by SWAT model is 15.50 Mt and 0.419 Mt/yr (i.e. 419,000 t/yr) respectively. A 2.6% of 1DD1-Kikuletwa catchment yield/rate as simulated by SWAT model derived sediment fluxes for

1DC1-Ruvu catchment. Therefore, long term sediment yield and sediment yield rate for 1DC1-Ruvu are 0.40 Mt and 10,890 or 11,000 t/yr respectively. Total sediment yield for NyM reservoir catchment (i.e. 15.90 Mt) was derived by summing up long term sediment yield from 1DD1-Kikuletwa and 1DC1-Ruvu catchments. Therefore, the predicted long term NyM reservoir catchment sediment yield rate (i.e. 15.90 Mt/37) is 0.430 Mt/yr or 430,000 t/yr.

It has been established in the sampling programme that 7,939 t/yr (about 8,000 t/yr) of sediment load is released from the reservoir annually (Ndomba, 2007). Therefore, sedimentation rate estimated based on modeling and sampling is 422,000 t/yr. (Table 1). Actual sedimentation rate based on reservoir survey is 411,000 t/yr. The comparison is based on relative error performance criterion (Table 1). A relative error in percent (i.e. 2.6%) is computed as the ratio of absolute error (11,000 t/yr) to reservoir survey sedimentation rate (411,000 t/yr) (Table 1).

Table 1 Comparison of reservoir sedimentation rates based on SWAT model simulations and sampling programme and reservoir survey.

Method	Sedimentation rate [t/yr.]
SWAT model prediction and sampling programme	422,000
Reservoir survey	411,000
Absolute error	11,000
Relative error in percent = 2.6 %	

Based on the relative error of estimate of 2.6 percent as presented in the last row of Table 1 above, one would note that the SWAT model has predicted the actual sedimentation rate with reasonable accuracy. However, according to TMC criterion, this approach overestimates the actual sedimentation rate by 2.6 percent. The accuracy achieved by using SWAT model was expected because one of the underlying hypotheses in this study stipulates that correct estimation of surface runoff will lead to better prediction of sediment yield. Other workers such Garde and Ranga Raju (2000) have similar opinion.

In the ongoing study not only the sediment fluxes but also erosion processes and sediment sources areas are explored. In Table 2 below major five representative subbasins as depicted in Figure 1 with their long-term simulated annual averages of sediment yields (SYLD_MUSLE), and Surface runoff (SURQ) are presented. Two main observations can be made on those catchments that experience high sediment yields. Firstly; the dominant landuse is agriculture and secondly; the subbasins are located in the slopes of Mount Kilimanjaro (Weruweru and Kikafu) and Meru. On the other hand low sediment supply subbasins (Upper Kikuletwa and Sanya) are located in rangeland and/or low-lying terrain. It should be noted from Table 2 that not all subbasins where agricultural activities is practiced such as Sanya yield high sediment load to Pangani river system. Probably, this can be explained by the fact that surface runoff generation from these subcatchments is low as supported by values in the last column of Table 2.

Table 2. Long-term simulated average of spatial sediment yields (SYLD_MUSLE), and Surface runoff (SURQ) as predicted by SWAT model.

Subbasin (HRU)	Area [Km^2]	Sediment yield (SYLD_MUSLE) [t/ha]	Landuse	Surface runoff (SURQ) [mm]

Weruweru	1,361	1.21	Agriculture	83.6
Kikafu	1,082	0.95	Agriculture	74.5
Mt. Meru	1,079	0.83	Agriculture	44.4
Sanya	1,039	0.26	Agriculture	20.6
Upper Kikuletwa	2,674	0.08	Rangeland	12.2

One would also note from Table 2 above that high sediment yields are occurring in headwater regions of Pangani river basin (i.e. Weruweru, Kikafu and Mt.Meru slopes). Based on literature the latter result is well supported. For instance Wasson (2002) noted that the headwater regions yield almost all of the sediment transported downstream, and channels and slopes in downstream areas contribute very little.

Although the result of this study is encouraging, it should be noted that stationarity assumption as the case for rating curve has not been well addressed. Only, a static landuse map developed in late 1990’s has been used for the entire period of 37 years. Nevertheless, a good result obtained from this study would probably, suggest that landuse from main sediment sources areas have not significantly changed. Besides, the authors are aware from literature of the number of disadvantages of physically based models. They include heavy computer requirements, the need to evaluate many parameters (i.e. uncertainty) and a complexity, which implies a lengthy training period for new users (Bathurst, 2002). As reported by Ndomba(2007) parameters identifiability and evaluation exercise for the runoff component of the SWAT model was limited to six years of daily streamflows data because of a huge computational resource requirement. Eight wet years of calibration data as recommended by Yapo, *et al*., (1996) resulted to longer computer simulation time. And some of a few insensitive parameters have been assumed spatially uniform in order to achieve a practical computation time. In this study the issue of parameter uncertainty has not been dealt rigorously as attempted by others (Beven and Binley, 1992). Alternatively, in this study the suspended sediment loads as computed from excellent rating curve and the long term reservoir sediment accumulation information are used as lower and upper bounds of the model outputs. Although, the SWAT model has overestimated the actual sedimentation rate by 2.6%, it should be remembered that the sampling stations are located more than 5km upstream. Probably, some sediment loads are deposited in the main tributaries river channels upstream of the NyM reservoir. Therefore, a comprehensive sediment transport channel network model is recommended to account for the discrepancy.

CONCLUSIONS

A semi-distributed, physics based watershed model (SWAT) has reasonably simulated sediment yields, and has replicated the erosion processes and sources in the Pangani basin using one year hydrological daily sediment loads. The SWAT model captured 56% of the variance of the observed daily sediment loads during calibration. The application of the model in longer period (i.e. 37 years) has predicted well the reservoir sediment accumulation with a relative error of estimate of 2.6 percent. Such estimation accuracy can be attributed to both sound sediment sampling programme design and well calibrated components of SWAT model. This result also suggests that for catchments where sheet erosion is dominant SWAT model is a better substitute of the sediment-rating curve and long-term prediction of sedimentation rate can be done with reasonable accuracy. It should be noted that the calibration was done during the normal wet year with most of hydrometeorological data required for SWAT model is available.

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Rapid Geomorphic Assessment of Watershed Sediment Budgets for Water Supply Reservoirs Using SWAT and Sub-Bottom Acoustical Profiling

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Reservoirs in North Texas supply over 99 percent of the water to the Dallas Fort Worth Metroplex or to over 5.8 million people. The rate of sedimentation to these reservoirs has been estimated to be in the range of 2 cm a year. This is equivalent to an annual loss of future water supply for up to 3000 people per reservoir (lost storage capacity). The costs of new reservoirs as well as the cost of dredging makes options other than watershed management (BMP's) unrealistic. In order to best manage sedimentation into the reservoir, the source of sediment into the reservoir must be quantified. This study represents the approach used by the Tarrant Regional Water District to study reservoir sedimentation. It consists of three integrated steps: (1) rapid geomorphic assessment by sub-watershed and physiographic province to ascertain the magnitude of active processes; sheet/rill; gully; stream erosion supplemented with historic air photographic inventory (2) profiling the reservoir with differential GPS and a 5 transducer sub-bottom acoustical system which gives sediment thickness within the reservoir which is verified with vibracore and Cesium 137 analysis, and (3) modeling the watershed with the SWAT model calibrated to the total volume of sediment as detailed in the survey, watershed erosion processes, any gaged data, climate and land use. Failure to follow each step is shown to produce large errors in the model and potential effectiveness of BMP's.

INTRODUCTION

A sediment budget is an accounting of the sources and disposition of sediment from its point of origin to its eventual exit from a watershed. Although erosion from upland sources is most often cited as a major source of sediment load; channel/streambank erosion can also be a major source in some watersheds contributing from 17% to 93% of the total sediment load (Trimble 1997 and Sekely et al. 2002). Watershed scale models such as SWAT, HSPF and GWLF are widely used to assess sediment and nutrient budgets within the watersheds. According to the EPA's recent National Water Quality Inventory report (USEPA 2002), 31% of the river and stream miles are impaired due to sediment; sediment/siltation is the second leading cause of water quality impairment, next only to pathogens. In this research, watershed scale model SWAT is used to quantify the sediment budget of the Cedar Creek reservoir watershed, located in North-Central Texas.
More specifically, the objectives of this study are:

1) To test SWAT channel sediment routing routine against field geomorphic data, and
2) To illustrate the use of rapid geomorphic assessment of watersheds for use in SWAT

SWAT sediment and water routing routines

In SWAT a simplified stream power approach developed by Bagnold (1977) and adopted by Williams (1980) is used for estimating channel degradation or deposition. The maximum amount of sediment that can be transported in a reach segment is calculated as:

$$conc_{sed,mx} = spcon \times v_{ch}^{sp\exp} \quad (1)$$

where $conc_{sed,mx}$ is the maximum concentration of sediment that can be transported by the water (ton/m^3), spcon is a coefficient defined by the user, v_{ch} is the peak channel velocity (m/s), and spexp is an exponent defined by the user.

If the maximum concentration of sediment calculated with Eq.1 is lesser than the concentration of the sediment entering the reach then deposition is the dominant process in the reach segment. Conversely, if the maximum concentration of the sediment calculated with Eq.1 is higher than the concentration of the sediment entering the reach then channel degradation is the dominant process in the reach segment. Sediment degradation is calculated as:

$$sed_{\deg} = (conc_{sed,mx} - conc_{sed,ch}) V_{ch} K_{ch} C_{ch} \qquad (2)$$

where sed_{deg} is the amount of sediment degraded/reentrained from the reach segment (metric tons), $conc_{sed,ch}$ is the initial sediment concentration in the reach (ton/m^3), V_{ch} is the volume of water in the reach sediment (m^3), K_{ch} is the channel erodibility factor and C_{ch} is the channel cover factor.

As can be seen from the above equations, flow velocity (v_{ch}) in Eq.1 is an important variable in determining the streambank erosion or deposition from a reach segment. In SWAT water can be routed from upstream to downstream using either the variable storage or Muskingum routing methods. In both these methods a bucket type approach is used for calculating the amount of water entering the reach, i.e. the water is moved from one reach to another by volume basis. In the bucket type approach the flow rate in each segment is also a function of stream length (Narasimhan et al. 2007). Hence, the flow velocity predicted is artificially higher at a smaller reach downstream of two big reaches (due to the nature of watershed delineation). Hence, an iterative approach developed by (Narasimhan et al. 2007) was used (Figure 1).

$$q_{ch} = \frac{V_{ch}}{(24 \times 60 \times 60)}$$

Do While ($q_{it} < q_{ch}$)
  $D_{ch} = D_{ch} + 0.01$
  $A_{ch} = (W_{btm} + z_{ch} * D_{ch}) * D_{ch}$
  $P_{ch} = W_{btm} + 2. * D_{ch} * \mathrm{Sqrt}(1. + z_{ch}^2)$
  $R_{ch} = A_{ch} / P_{ch}$
  $v_{ch} = \frac{1}{n} R_{ch}^{2/3} S_{ch}^{1/2}$
  $q_{it} = v_{ch} * A_{ch}$
end do

Figure 1. Modification to water routing algorithm

where q_{ch} is the flow rate of water entering the channel (m^3/s) and V_{ch} is the volume of water entering the channel during the day (m^3/day).

The flow velocity calculated by this modified approach is no longer a function of length and hence the flow velocity will vary gradually independent of length as the water moves downstream. In this research, the modifications discussed above were made to the variable storage routing method for hydrology simulation in the Cedar Creek watershed.

Study Area

Water quality is a growing concern in the Cedar Creek reservoir (Figure 2). In addition to nutrient enrichment and reservoir siltation from upland and streambank erosion affects the quality of drinking water and the life of this reservoir. On an average 1032 ac-ft of reservoir volume (679,200 ac-ft) is lost every year due to siltation. Cedar Creek is one among the five major reservoirs managed by Tarrant Regional Water District (TRWD) in the Trinity River Basin. Currently TRWD supplies water to about 1.6 million people across 11 counties from its network of five reservoirs and are expected to serve 2.66 million by 2050. Due to this projected population increase in North Central Texas over the next 50 years, TRWD is collaborating with the Texas Water Resources Institute and Texas A&M University to assess the watershed condition and develop watershed protection plans for these five major reservoir catchment. Cedar creek reservoir has a contributing drainage area of about 2611 sq. km. The average annual rainfall is about 991 mm. The western half of the watershed is underlain by shale and residual clay soils; the eastern half of the watershed is underlain by interbbedded sands and shales and mostly loam to sandy soils. The land cover is predominantly pasture (62%), followed by forest (16%), Urban (7%), cropland (6%) and rest occupied by water (6%) and wetlands (3%) types.

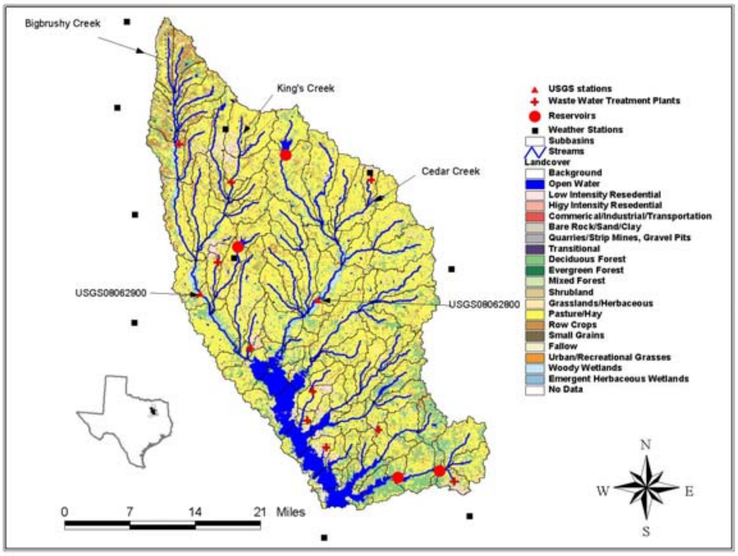

Figure 2. Land cover classes, sub-basin and stream network of Cedar Creek watershed.

WATERSHED AND LAKE SURVEY

In order to assess the accuracy of SWAT model output, various Rapid Geomorphic Methods can be utilized. The methods found to be useful in the Cedar Creek Watershed are shown in Table 1.

	Rapid Method	Time Frame	Comments
I	Point Monitoring with erosion pins/scour gages; sediment monitoring	Individual storm-years	Costly for large watersheds; gage sites very costly to install and maintain; cheaper *duratrac* (Figure 10.) can give quantitative cumulative data
II	Volumetric Monitoring of Downstream reservoirs	Repeat bathymetric surveys (decades) or Sub-bottom acoustical	Volumetric not real good for sedimentation amounts; acoustic methods are superior and sediment flux possible with Cesium-137 (for reservoirs older than 1964)
III	Photo Interpretation	decadal	Problems for small streams which make up to 70% of watersheds owing to riparian vegetative cover.
IV	Literature Rates and Equations	Not applicable.	Due to wide scatter of data, empirical equations require a lot skill to interpret results; land use impacts hard to discriminate; typically short term data.
V	Visual site geomorphic Assessment	Individual storm or trend	Gives qualitative estimate of recent trends and rates

Table 1. Rapid Geomorphic Assessment Methods

The first problems to assess are the basic elements of the sediment budget within the watershed. A more detailed list is given by Reid and Dunne (1996). For the Cedar Creek watershed, the major contributors found in the field were sheet, rill and ephemeral gully erosion, and concentrated flow erosion (gully erosion and channel erosion). In order to verify the output of the SWAT model, various methods were utilized to assess the magnitude of each contributing sources. It was found easiest to work backward from the most inclusive source of data or the reservoir sediment survey. The sediment survey was done to estimate the density and volume of the post-impoundment reservoir sediment. A Vibracoring technique was used for obtaining undisturbed cores of unconsolidated sediment at saturated or nearly saturated conditions. Five cores were collected within the reservoir. The core locations were determined with the aid of acoustic sub-bottom profiling, Figure 3.

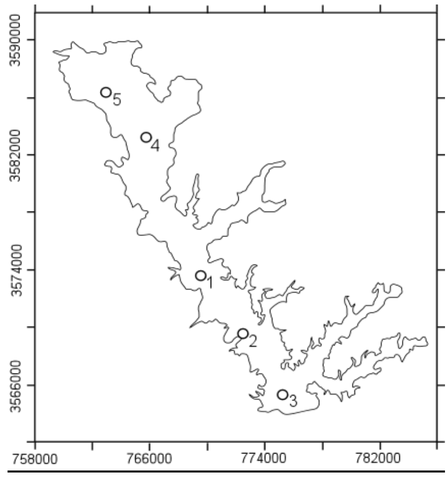

Figure 3. Core Locations Cedar Creek Reservoir

The cores were used to determine the thickness, density and dry weight of post-impoundment sediment in the reservoir, Figure 4. Acoustic sub-bottom profiling of the lake showed that the sediment is uniformly distributed along the length of the lake with an average thickness of 37.4 cm. The average dry density of the sediment was estimated to be 344.5 kg/m^3. A reservoir volumetric survey conducted by Texas Water Development Board (TWDB) in 2005 showed that the lake has lost a total of 41,276 acre-ft during the 40 year period since its construction in 1965. (Cesium 137 could not be used in this reservoir). The volume loss is attributed to the deposition of sediment from the supplying watershed. Multiplying this sediment volume and the average dry weight density determined in the current study and by assuming a reservoir sediment trap efficiency of 98%, indicates that the total dry weight of the post-impoundment sediment in the lake is 17,862,337 Metric Tons over the 40 year period or an average sedimentation rate of 446,558 Metric Tons/year. In order to quantify the individual contribution of sheet and rill versus concentrated flow erosion, supplemental methods were employed.

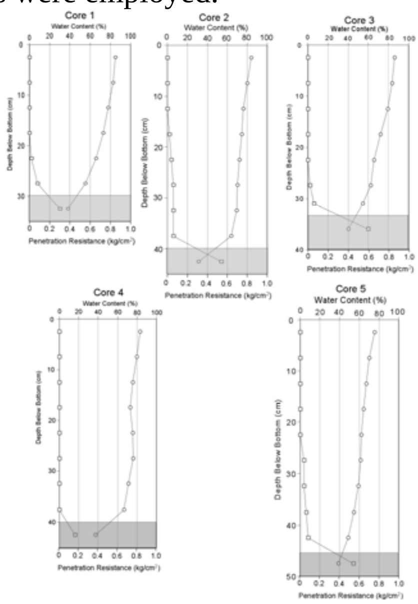

Figure 4. Cores for assessing sediment thickness and density

For sheet and rill erosion, regional estimates based on regression equations for the same physiographic provinces from sediment gage data, past USLE and sediment delivery calculations and visual assessment of soil loss by NRCS personnel were utilized. For channel and gully erosion, a comprehensive field survey was done at 56 sites across the watershed to identify eroding stream segments in the watershed. At each site a survey form was used to categorize the level of channel incision as well as degree of erosion in addition to several other parameters, Figure 5.

Streambank erosion from the watershed is estimated to be 152,572 Metric Tons per year. This is based on five different methods of channel erosion assessment: (1) erosion assessment made for the basin based on NRCS field evidence (Griener, 1982), (2) field assessment of channel erosion and SWAT generated channel lengths and dimensions using theRapid Assessment Point Method (RAP-M) used by NRCS (Windhorn, 2001), (3) using power functions utilized in SEDNET (Wilkinson, et. al. 2004), (4) comparison of erosion rates to gage data by Ecoregion after Simon, et. al. (2004) and (5) literature review of channel erosion rates. Based on these methods, channel erosion could be assigned an average annual loss rate. Volumetric loss was calculated by the product of channel length times the erosion rate and the soil density and eroded channel height. In addition to the field survey, historical aerial photographs were also analyzed to identify historical trends in erosion across the watershed. Based on land use, stream condition at surveyed locations, and using historical air photographs, the streambank erosion categories were extended between field locations to the entire watershed area (Figure 6). The streambank erosion estimate along with the streambank erosion category mapped from site visits and field photographs were used for both assessing original SWAT model output, and then in calibrating the SWAT Streambank erosion parameters erodibility factor (K_{ch}) and channel cover factor (C_{ch}).

Based on this reservoir sedimentation rate (446,558 Metric Tons/year) and the Streambank erosion rate (152,572 Metric Tons/year) from the field survey, the overland erosion rate was inferred as 293,986 Metric Tons/year. These erosion rates were used as target numbers for calibrating the SWAT model parameters.

Results

Flow calibration and validation

SWAT was calibrated for flow by adjusting appropriate inputs that affect surface runoff and base flow. Adjustments were made to runoff curve number, soil evaporation compensation factor, shallow aquifer storage, shallow aquifer re-evaporation, and channel transmission loss until the simulated total flow and fraction of base flow were approximately equal to the measured total flow and base flow, respectively. Flow calibration was performed from 1963 through 1987. For this period predicted flow matched measured flow very well at the two USGS stream gages in the watershed. With the same calibration inputs, flow was validated from 1980 through 2002 using the measured mass balance of Cedar Creek Reservoir for comparison to predicted inflow values (Figure 7). The predicted inflow match measured inflow very well (r^2 = 0.76 and Nash-Sutcliffe COE of 0.80).

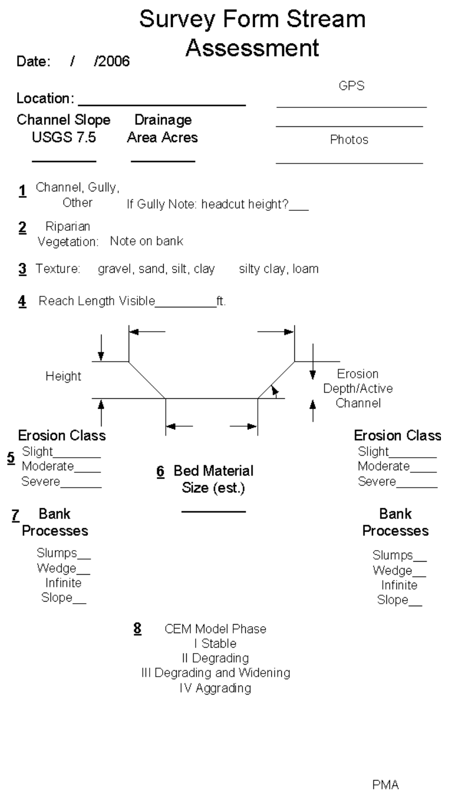

Figure 5. Field Survey Form

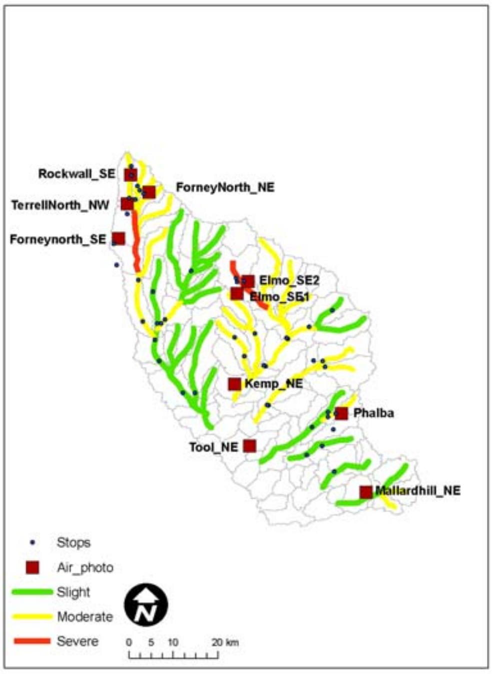

Figure 6. Erosion Ratings and Sites

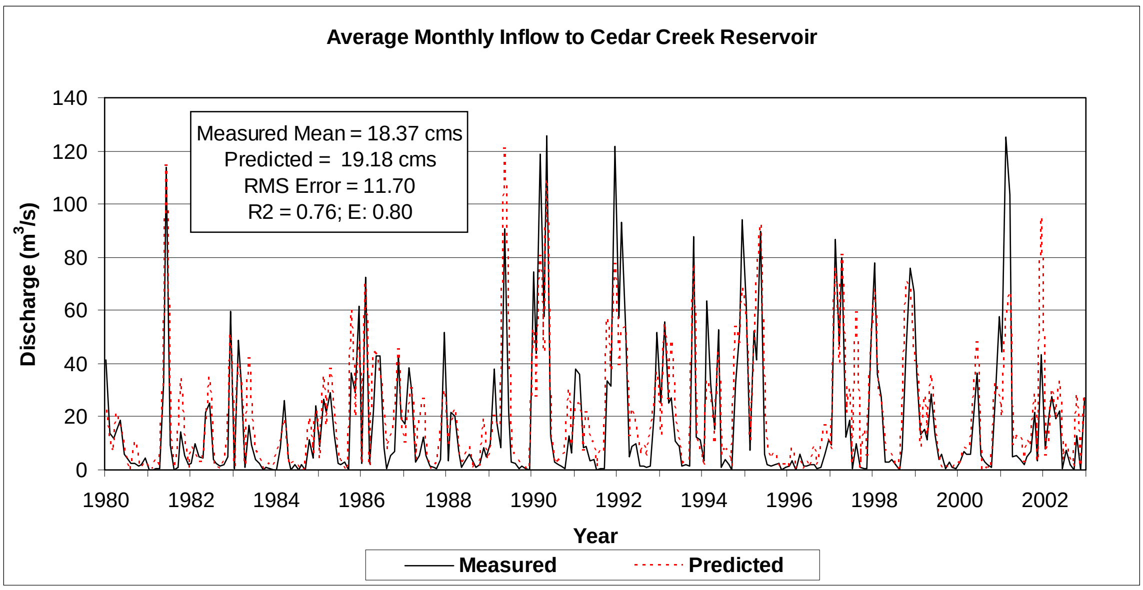

Figure 7. SWAT monthly streamflow validation with measured reservoir inflows.

Sediment Calibration

Annual overland and Streambank erosion rates determined from watershed survey and lake survey were used as target rates for adjusting model parameters. Streambank erosion, power function parameters, spcon (0.01) and spexp (1.5) in Eq.1 were adjusted based on limited storm flow total suspended solids (TSS) data available at various stream segments. The coefficients were chosen in such a way that the average simulated suspended sediment concentration is two to three times higher than the measured TSS. This was done so because SWAT does not simulate bed load transport and all the sediments are assumed to be in suspension. If these (spcon and spexp) coefficients were tightly calibrated with measured TSS without accounting for bed load transport, the model will considerably underestimate the sediment transport power of the water.

Channel physical properties such as channel vegetation cover factor (C_{ch}) (0.1 to 1.0) and channel erodibility factor (K_{ch}) (0.3 to 0.8) were adjusted for individual stream segments based on field assessment and geological data. The higher these values are, greater is the vulnerability of the channel for streambank erosion and vice versa (Eq.2).
Annual sediment contribution simulated from various sources is shown in Figure 8. Channel degradation is the second major contributor of sediment (33%) next to cropland (44%). It is important to note that SWAT predicted overland erosion was very close to the field estimation; hence no adjustments were made to the SWAT overland erosion parameters. The spatial distribution of Streambank erosion rates is shown in Figure 9. Overall there is a good agreement between the field assessment (Figure 6) and the model estimates (Figure 9). However, when comparing these maps it should be kept in mind that the field assessment technique is only a qualitative measure of erosion based on visual analysis.

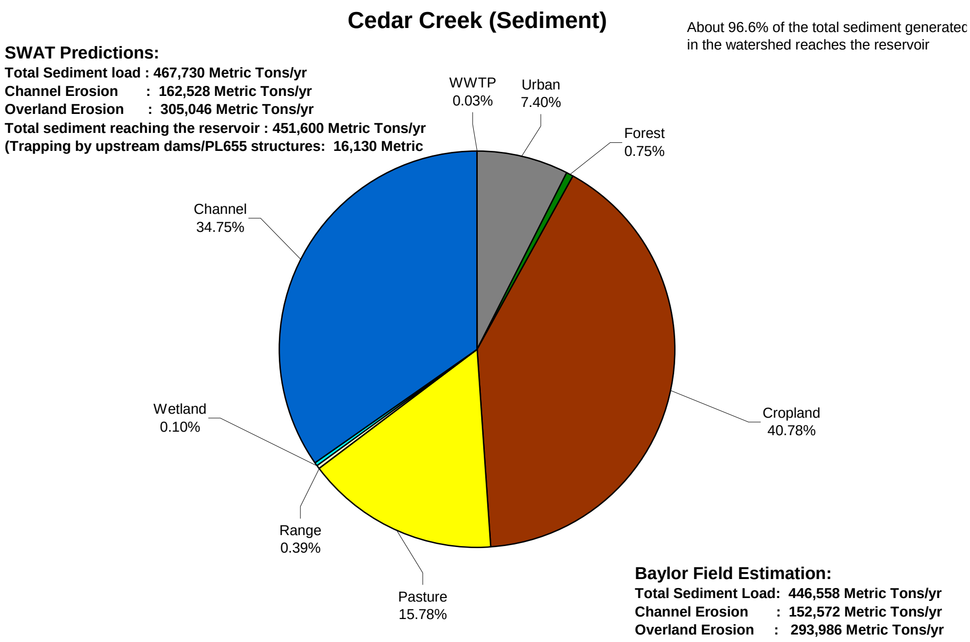

Figure 8. Sediment contribution from channel and overland erosion.

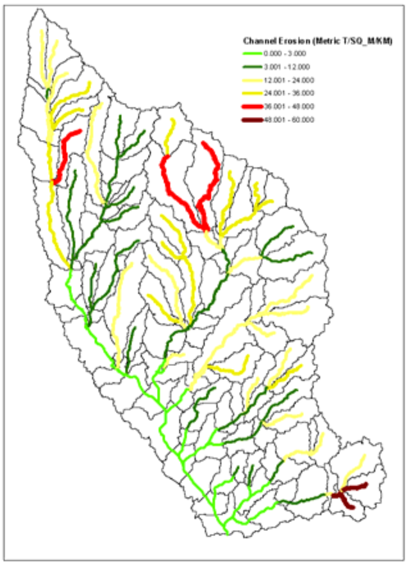

Figure 9. Streambank erosion rate predicted by SWAT in terms of Metric Tons per unit cross-section area and per unit length of the channel.

In terms of streambank and gully erosion, a new method was created for longer term assessment of erosion rates. In conjunction with erosion pins, a devise called the "duratrac" was formulated (Figure 10). Basically, the device is a tachometer which records with a quartz clock the water level at or above a selected height. In this way, the cumulative tractive force of a set of erosion pins can be known and the SWAT model erosion coefficients adjusted accordingly. The device costs about 35-50 US dollars and will run about six months or more under normal conditions. The device could also be used to assess the length of time the water was over bankfull stage. By placing these simple devices over the watershed, cheap accurate data can be collected for any chosen time interval; the only cost after installation is associated with a trip to the field to measure the erosion pins and visually download the cumulative time.

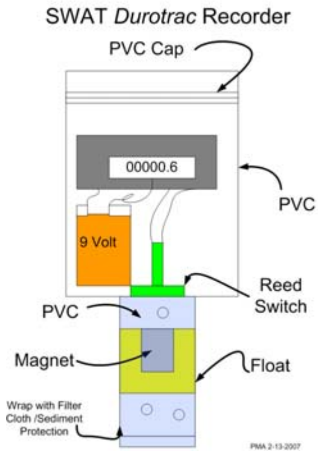

Figure 10. SWAT ***Duratrac*** recorder.

SUMMARY

A rapid geomorphic assessment in the field is useful to assess the SWAT model outputs given the paucity of real gage data. Methods employed can be accomplished at about 250 sq. km.a day. This would include field assessments of streams at road crossings and visual inspection of surrounding terrain for evidence of landuse and erosion processes. A modification to the velocity subroutine is proposed. Finally a simple device to be employed to assess cumulative tractive force for stream or gully erosion assessment is introduced.

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Modelling soil erosion in a sub-humid tropical environment at the regional scale considering land use and climate change

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Abstract

Soil degradation is a severe problem in Africa. The resulting decline of crop yields threatens food security and forces poverty, migration and land use conflicts. In this study, which is part of the German integrated water resource management project IMPETUS, the SWAT2005 model was applied to the Upper Ouémé catchment, Benin (~14500 km^2) in order to quantify water and sediment yield. Future climate and land use change were considered. Prior intervention areas for soil conservation measures should be identified.

The model was successfully calibrated and validated for the years 1998 to 2005 using daily measurements of discharge and suspended sediment concentration at various outlets in the catchment. Subsequently, the model was applied for different scenarios of climate and land use change until 2025 and 2050. Therefore, spatially distributed results from the regional climate model REMO (IPCC SRES scenarios A1B and B1) and the land use/land cover change model CLUE-S, produced by other IMPETUS members, have been processed. CLUE-S results were disaggregated from a 250 m to a 25 m grid for a better representation of agricultural land. Daily REMO output in a 0.5° grid resolution was attributed to the rain gauge sites considering the rainfall distributions of the rain gauges to obtain a correct frequency distribution of site-specific events.

The results of the scenario analysis for the period 2000-2025 indicate opposed impacts of land use change and climate change on water and sediment yield in the same order of magnitude. Land use change led to a strong increase in sediment yields in several subbasins, whereas climate change, in particular lower precipitation, reduced water and sediment yield in most parts of the catchment. Regions with actual and future high erosion risk were identified.

KEYWORDS: SWAT, soil erosion, climate change, land use change, Africa, Benin, tropics

Introduction

Soil degradation is a severe problem in Africa. Soil erosion, the most important process leading to soil degradation, contributes one fourth to the productivity loss in Africa (Oldeman, 1998). The resulting decline of crop yields threatens food security and forces poverty, migration and land use conflicts. Despite the flat relief soil erosion is a considerable problem in Benin caused by high precipitation intensities and unsustainable cultivation methods. Although the amounts of soil loss in Northern and Central Benin are rather moderate the effects of soil erosion cannot be neglected because soil depths are often low and farmers can rarely afford fertilizer to compensate the decline of soil fertility due to topsoil loss. In order to improve the effectiveness of soil management measures prior intervention areas with current and future high risks of soil degradation need to be identified.

This study is embedded in the IMPETUS framework, an integrated water resource management project, funded mainly by the German Ministry of Research and Education (Speth et al., 2005). The project design covers three 3-year periods with a focus on process understanding (2000-2003), modelling and scenario analysis (2003-2006) and the development of decision support systems (2006-2009). The research areas of the project are the Ouémé catchment in Benin and the Wadi Draa in Morocco.

Research area

The Upper Ouémé catchment (~14500 km^2) is situated in Central Benin (Figure 1). The research area is part of the sub-humid climate zone with one rainy season between May and October. The annual means for rainfall and total discharge are 1100 mm and 150 mm, respectively. The vegetation is dominated by wet savannah types, which are severely degraded in the North-Western part of the catchment. The area can be characterized as an undulating pediplain relief overlying a pre-cambrian crystalline basement. Acrisols, Lixisols and Ferralsols are the dominant soil types and show often gravely or plinthic horizons.

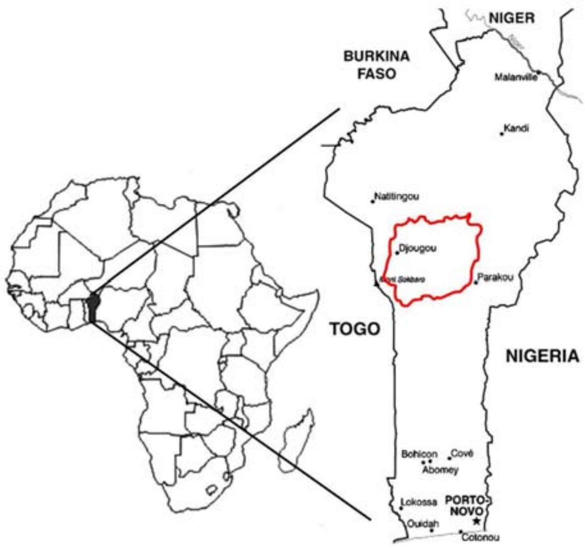


Figure 1. The Upper Ouémé catchment in Benin

Farmers depend mostly on subsistence farming based on crops like yam, cassava and maize. Cotton and cashew are cultivated as cash crops. The agricultural area is rapidly expanding due to population growth, migration and an improved accessibility. Field studies in the first phase of the IMPETUS project revealed that the amount of soil erosion depends significantly on the land use system and the rainfall intensity. Soil loss rates on agricultural land were 10 times higher than on savannah land, with a maximum on cotton and yam fields (Junge, 2004).

Methodology

The quantification of soil erosion at the regional scale until 2050 required a modelling approach. The free available, time continuous, semi-distributed model SWAT (soil water assessment tool, version AVSWAT 2005) was chosen due to its capability to simulate large catchments with a manageable demand of input parameters. For African conditions a comparatively good database was available from the work of IMPETUS and counterparts in Benin. Own field studies were required to obtain soil properties for the French soil map (Faure & Volkoff, 1998) and time-continuous suspended sediment concentrations at four river outlets.

Figure 2. The subcatchments Donga-Pont and Terou-Igbomakoro

Thresholds of 6000 ha for stream delineation and 10% for HRU delineation resulted in 121 subcatchments and 926 HRUs in the catchment. Hydrological model calibration and validation were performed for the years 1998-2001 and 2002-2005 using daily discharge data provided by IRD France and the *Diréction Générale de l'Hydraulique* (Benin). Sediment calibration was carried out for the years 2004 and 2005. Calibration was conducted simultaneously at two outlets: the Terou-Igbomakoro catchment (2323 km^2) and the intensively agriculturally used Donga-Pont catchment (586 km^2) (see Figure 2). The goodness of the model results was evaluated using the model efficiency (ME), the index of agreement (IoA) and the coefficient of determination (R^2).

Subsequently, the model was applied to compute scenarios of climate and land use change until 2025 and 2050 using spatially distributed results from other project members. The input parameters for the climate scenarios were provided by the regional climate model REMO (Paeth, 2004) driven by the IPCC SRES scenarios A1B and B1. Feedbacks of expected deforestation rates and land degradation on the regional climate were included in the model. Three ensemble runs were available for each climate scenario. For a correct representation of the amounts and the frequency distribution of daily rainfall in the SWAT model, daily REMO output in a 0.5° grid resolution was attributed to the rain gauge sites in the catchment.

As input for the land use scenario land use maps were generated for each year between 2000 and 2025 in a 250 m resolution by the land use/land cover change model CLUE-S (Thamm et al., 2006). This model considered several driving forces like population density, topography and distances from roads and rivers. In order to reduce the error in the representation of farmland due to the coarse resolution of the land use map a simple disaggregation algorithm was applied: The 250 m-grid cells were disaggregated into 25 m cells according to the mean fractions of land use types represented in a 250 m – grid cell of the original, not aggregated land use map derived from a satellite image of the year 2000.

Results

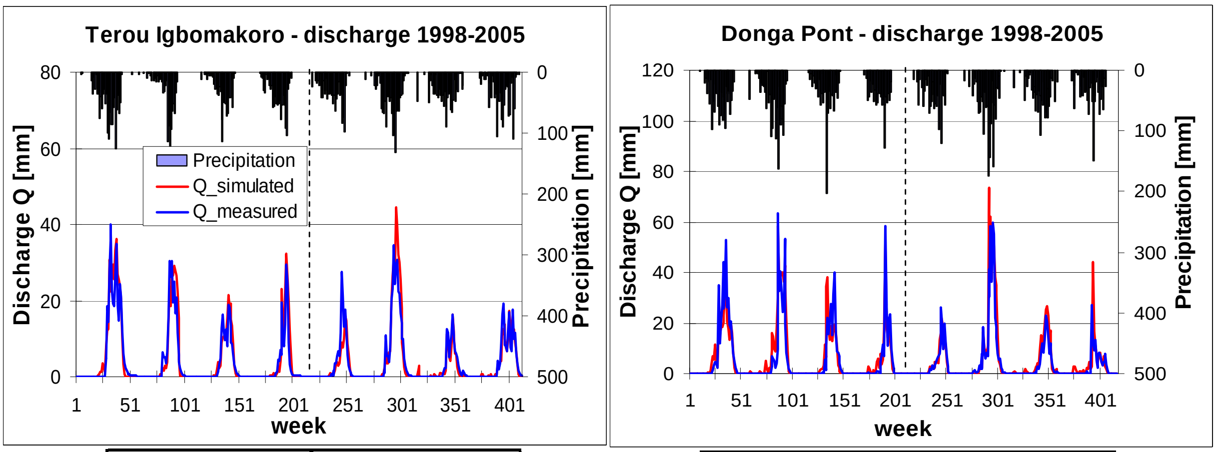
As a prerequisite for the scenario analysis the model was extensively validated after calibrating the model for the hydrology and sediment budget.

Model calibration and validation

The calibrated parameters included the SCS curve numbers, the surface runoff lag coefficient (SURLAG), the soil evaporation compensation factor (ESCO), several groundwater parameters (ALPHA_BF, GWQMN, GW_REVAP, REVAP_MN, RCHRG_DP) and the USLE C-factors. Furthermore, the available water capacity was increased by 30% for all soils because the estimated values according to the pedo-transfer function of Rawls and Brakensiek (1985) were too low. Finally, model calibration was successfully performed for the period 1998 to 2001. Weekly model efficiencies of 0.87 and 0.78 were obtained for the Terou-Igbomakoro and the Donga-Pont outlets, respectively. During the validation period (2002-2005) model efficiencies were similarly good (ME 0.85 and 0.88). In general, the discharge dynamics were well reproduced in both subcatchments (see Figure 3). However, some difficulties in capturing extreme events were observed. The fractions of slow and fast discharge components were as well correctly simulated (see Table 1). A spatial validation at several other outlets in the catchment led also to satisfactory results.

Table 1. Comparison of mean simulated and measured discharge components for the calibration period at the Terou-Igbomakoro and Donga-Pont outlets

	Simulated			Measured					
	Q_{tot} [mm]	Q_{surf} [mm]	Q_{base} [mm]	Q_{tot} [mm]	Q_{surf} [mm]	Q_{base} [mm]	Q_{tot}_sim/ Q_{tot}_meas [%]	Q_{surf}_sim/ Q_{surf}_meas [%]	Q_{base}_sim/ Q_{base}_meas [%]
Donga-Pont	318	185	133	311	176	135	102	105	99
Terou-Igbomakoro	242	106	136	229	103	126	106	103	108



ME	R^2	IoA	ME	R^2	IoA
0.87	0.87	0.72	0.85	0.85	0.72

ME	R^2	IoA	ME	R^2	IoA
0.78	0.81	0.75	0.88	0.85	0.69

Figure 3. Measured (blue) and simulated (red) discharge curves for the calibration period (1998-2001) and the validation period (2002-2005) for the Terou-Igbomakoro and the Donga-Pont subcatchments

In the two years of sediment calibration the simulated and the measured sediment amounts in the Donga-Pont catchment and the Terou-Igbomakoro agreed quite well. Only in the Terou-Igbomakoro catchment the sediment yield was significantly underestimated for the year 2005 because discharge peaks were considerably underestimated. However the temporal characteristics of the simulated sediment curve are well reproduced in both subcatchments (see Figure 4). The measurements of goodness for the weeks with valid measurements were satisfactory for the Terou-Igbomakoro catchment (ME 0.68, R^2 0.76, IoA 0.58) as well as for the Donga-Pont catchment (ME 0.61, R^2 0.64, IoA 0.71).

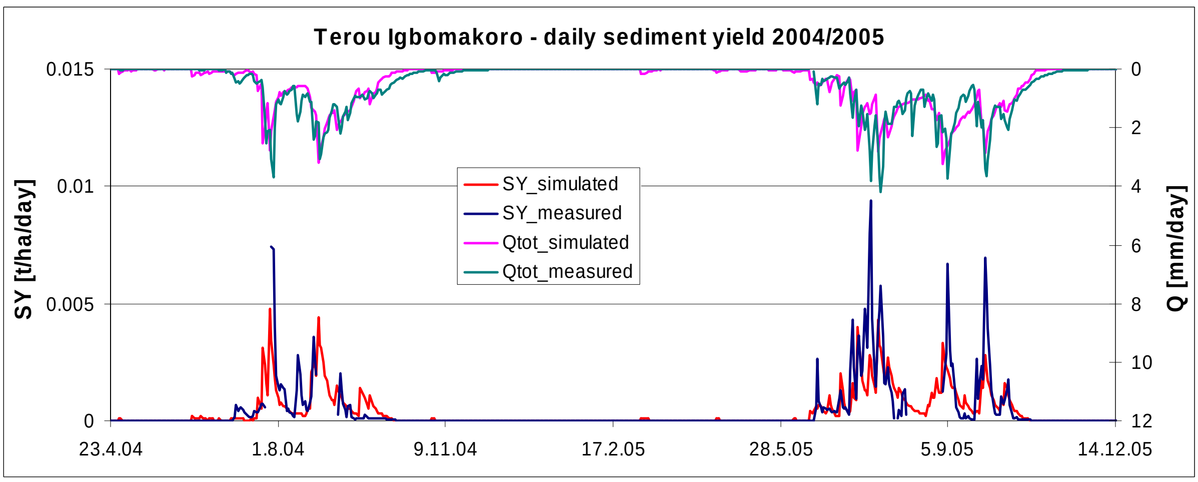


Figure 4. Comparison of measured and simulated daily discharge and sediment yield (SY) for 2004 and 2005 at the Terou-Igbomakoro outlet

Modelling results 1998-2005

As expected, absolute and relative surface runoff amounts were considerably higher in the Donga-Pont catchment than in the Terou-Igbomakoro catchment, due to higher land use intensity, higher rainfall amounts and rainfall intensities. In the whole Upper Ouémé catchment surface and groundwater flow account each on average 10% of annual rainfall. Mean annual sediment yields for all subcatchments ranged from 0.04 to 5.11 tons per hectare for the 121 subcatchments. Highest values were obtained around the cities (Parakou and Djougou) and along the roads. The mean annual sediment yield of 0.35 t/ha/yr in the Donga-Pont catchment was two times higher than for the Terou-Igbomakoro catchment. Cropland hat contributed with about 76% to the total sediment load, followed by brush savannah with about 18%. The average sediment yield on cropland was 4.5 t/ha/yr.

Climate scenarios

After post-processing the climate data from the regional climate model REMO, the mean simulated monthly rainfall amounts, the distribution of daily rainfall and the potential evapotranspiration for the period 1960 to 2000 were well reproduced by the model (see Figure 5). However, monthly rainfall values were slightly higher at the beginning und lower in the middle of the rainy season. For the Parakou station low intensity rainfalls (1-20 mm) were slightly overestimated and high intensity rainfalls (20-200 mm) slightly underestimated. The mean simulated rainfall for the globally economy-orientated scenario A1B for the period 2000-2025 and 2026-2050 was slightly higher than for the global sustainability-orientated scenario B1. Table 2 provides an overview of the results of the climate scenarios for the whole catchment and the two subcatchments to point out the regional variation. The results of each scenario are averaged from three runs with the SWAT model referring to the three ensemble runs from the climate model REMO. The reduction of mean rainfall amounts by 2% to 14% in the Upper Ouémé and the Donga-Pont catchment led to a decrease of mean water and sediment yield in the periods 2001-2025 and 2026-2050 in both catchments for at least five of six runs. One ensemble run of the A1B scenario even leads to a mean reduction of water and sediment yield in the period 2026-2050 of -46% and -53%, respectively. However, standard deviations of the mean values for the ensemble runs are considerably high for the period 2026-2050 reaching up to 22%. For the Terou subcatchment the effects of climate change are not so clear showing reductions and increases of water and sediment yields. For the B1 scenario an increase in the mean annual rainfall and significantly increased mean annual sediment and water yields was simulated in all three runs. Looking at the spatial maps, which are not presented here, it becomes obvious that the spatial pattern of sediment yield remained quite similar. However, a few subbasins experienced a pronounced decrease (Donga region) or increase (North-Eastern edge of the catchment, only for B1 scenario) of sediment yield.

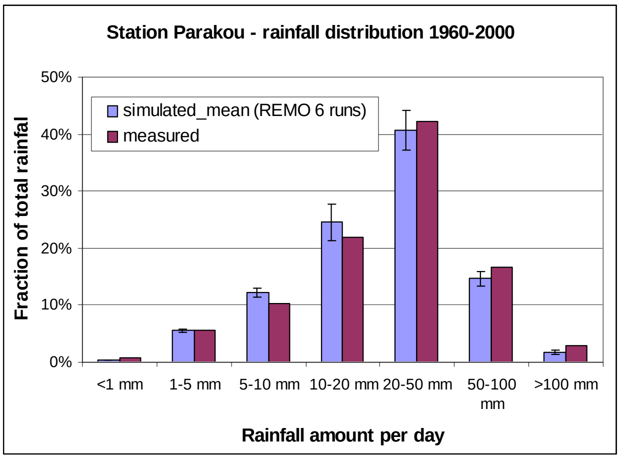


Figure 5: Parakou station –comparison of measured and simulated rainfall data for the period 1960 to 2000

Table 2. Mean simulated annual values of rainfall (PCP), sediment yield (SY) and water yield (WY) of the three ensemble runs for the climate scenarios A1B and B1, change in % from the baseline scenario (1998-2005)

	1998-2005			2001-2025 A1B			2026-2050 A1B			2001-2025 B1			2026-2050 B1		
	PCP [mm]	WY [mm]	SY [t/ha]	PCP [%]	WY [%]	SY [%]	PCP [%]	WY [%]	SY [%]	PCP [%]	WY [%]	SY [%]	PCP [%]	WY [%]	SY [%]
Upper Ouémé	1184	222	0.24	-4	-12	-10	-8	-23	-21	-3	-6	-1	-5	-12	-6
Donga	1294	302	0.35	-10	-23	-28	-14	-33	-37	-9	-17	-21	-12	-25	-30
Terou	1157	216	0.18	2	1	10	-2	-11	0	4	10	27	1	3	15

Land use scenarios

Although spatial land use maps are available for each year in the period 2000-2025 and for different scenarios the implementation of all maps is not feasible because the SWAT model requires for each map a new HRU delineation and the readjustment of all calibration parameters for each map. In a first approach the model has been run for the period 1998-2005 with the land use map for 2025. The results were compared to the baseline scenario with the disaggregated land use map for 2000. Figure 6 illustrates the calculated relative change of sediment yield. It can be seen that the highest relative increase took place in the South-Western and Eastern part of the catchment where land resources are still abundant. In the most degraded region of the catchment around Djougou sediment yield increases were moderate. As an assumption of the scenario the protected forest in the centre of the catchment remained stable. Highest absolute changes in sediment yield occurred in the South-Eastern part of the catchment around Parakou and North from Djougou. Overall, for the whole Upper Ouémé catchment the increase of 40% of cropland area led to a mean increase of 10% in surface runoff and sediment yield in the whole catchment reaching in individual subbasins up to 60% and 680% of increases in surface runoff and sediment yield. In some subbasins water and sediment yield were even slightly reduced due to changes in the HRU distribution.

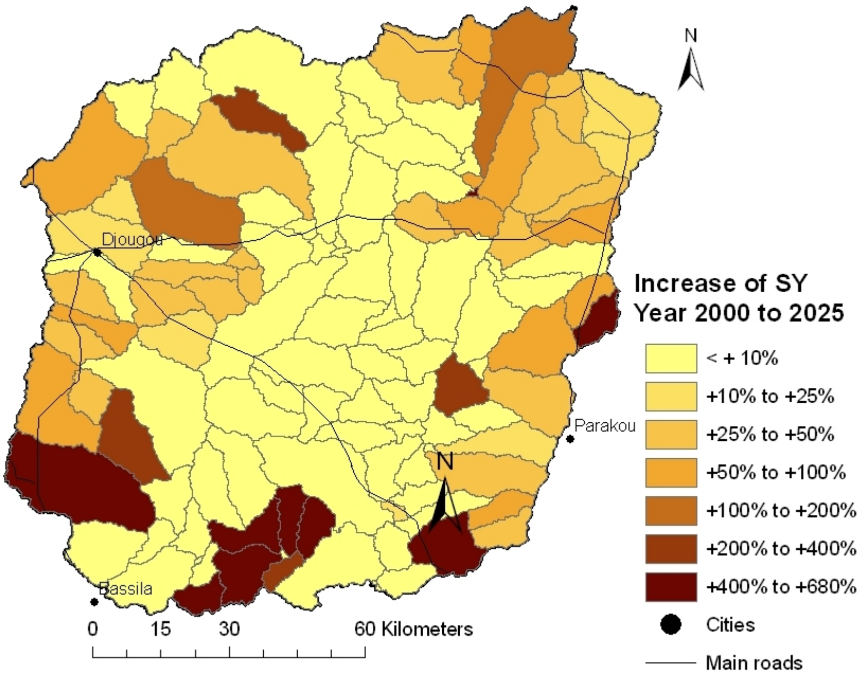


Figure 6: Land use scenario "Business as usual" - Relative increase of sediment yield (SY) per subcatchment; comparison of model runs (1998-2005) with land use maps for the years 2000 and 2025

Conclusions and outlook

This study has shown that the SWAT model is applicable to a sub-humid catchment and delivers reasonable results for current and future time periods. Current hotspots of soil erosion were identified in the North-Western and South-Eastern part of the catchment. Land use change caused higher surface runoff and sediment yield, especially around Parakou and North from Djougou. Climate scenarios led to a reduction of sediment and water yield in most parts of the catchment. Reductions were significantly higher in the period 2026-2050 than in the period 2001-2025. Currently, further land use scenarios and combined climate and land use change scenarios are calculated. Previous results from Busche (2005) and Sintondji (2005) indicated a stronger impact of land use change on the sediment yield than climate change. In future, the modelling results will be assessed by an uncertainty analysis. Furthermore, results from the models SWAT and EPIC will be integrated in a spatial decision support system for various stakeholders in Benin.

Acknowledgement

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. Assessment of impacts of climate change on runoff: River Nzoia catchment, Kenya

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Abstract

The SWAT model has been used to investigate the impact of climatic change on runoff of Nzoia river catchment in Kenya. River Nzoia is one of the several streams that drain into Lake Victoria. It has a catchment area of 12,709 km^2 and a length of about 334 km. The required model parameters were obtained from various sources while some were estimated from existing data. The model was calibrated against measured discharge data for the period 1980-1985. The model results showed relatively good fits between measured and modelled discharge with the Nash Sutcliffe efficiency and R^2 >0.7. Percentage changes in rainfall and temperature obtained from GCMs were imposed on the calibrated model. The results showed significant changes in the simulated hydrologic catchment response at the 0.05 significance level. The A2 2020 scenario GCM predicts increases in rainfall of 42% and a temperature change of about 0.8°C yielding an increase in surface runoff of 190%. The B2 2020 predicts rainfall increase of 12% and a temperature change similar to that of A2 2020 but increasing surface runoff by 108%. In the 2050s, A2 yields changes in rainfall, temperature and surface runoff of about 12%, 1.7°C and 72% respectively while B2 yields 11%, 1.4°C and 77%. Results show that the 2020s are likely to experience more flooding events as a result of increased rainfall than in the 2050s. For the same amount of increase in rainfall (12%), a temperature difference of about 0.32°C did not show any significant change in the amount of water yields while that of about 0.84°C did show significant change.

Introduction

The assessment of impacts of climate change on the water resource management systems is complicated by the course spatial resolution of climate change predictions. In Africa, relatively little work has been carried out with regard to potential impacts of climate change on water resource management systems. Even in the absence of climate change, water resources in Africa are predicted to come under extreme pressure during the next half century. Changes in water availability are due to global warming add further pressure on the adaptability of the water systems. The challenge for water managers is how water resource management systems will cope with increased variability and decreased availability over time, including the impacts of climate change.

Kenya is considered a water scarce country with about 647m^3/cap/year. This would stand at 150m^3/cap/year in 2050 without climate change, and between 210-250 m^3/cap/year under a range of climate change scenarios, (Watson et al., 1996). The World Bank bench mark for water scarcity is 1000m^3/cap/year and this indicates the extent of the future crisis in water availability in Kenya. According to one HADCM2 scenario that follows IPCC scenario IS95a which assumes little direct policy intervention globally to restrict greenhouse gas emissions and mid-range settings for all model parameters, simulations of climate change in Africa indicate that Kenya will be about 1.4^{o}C warmer by the year 2050 (about 0.2^{o}C/decade) (Lasse et. al., 1996). It also shows annual rainfall increases of 20% by 2050 throughout the country especially in the highlands. The north and west of the country however may experience smaller increases of about 5%. Potential evapotranspiration (PET) is expected to increase throughout the region by 10% due to the increase in temperature, and about 15% with the inclusion of other climatic changes and changes in the plant physiological characteristics in a CO_2 enriched environment. This scenario of climate change implies favourable changes in the water resources for Kenya. Largely because of the increases in rainfall, runoff would increase across the entire country, with much of it 50% above current levels. The extreme west of the country and around Lake Victoria increases may be quite small.

There are many effects and impacts of climate change on water resources some of which include:
i) Changes in variability, spatial patterns and seasonality of precipitation and changes in temperature will have the effect of changing the soil moisture, river runoff and groundwater recharge, peak runoff, basin hydrology and these will consequently cause changes in projected yield of reservoir systems, water quality, water supply infrastructure, requirement of storage in water supply systems; ii) Changes in sea level rise will cause loss of land due to saline intrusion into coastal aquifers and movement of salt-front estuaries affecting freshwater abstraction points. This implies reduced water quality and ground water abstractions; iii) Due to CO_2 enrichment, there would be increased photosynthesis and reduced transpiration leading to increased water use efficiency; iv) An increase in temperature could result in faster plant growth and increased transpiration. This would cause increased evaporation from lakes and reservoirs, reduced runoff and reduced groundwater recharge, higher demand for water for irrigation, bathing and cooling due to increased temperatures, leading to changes in water yields, with high stress on water delivery systems; v) Changes in drought and flood hazards will cause changes in seasonal water replenishment, risk in flood plains and the area affected. These will alter risks for water resources and reservoir operations.

In western Kenya the most catastrophic impact of floods is loss of human lives. People in the lower catchment are taken unaware when the floods occur because the upper catchments receive heavy rainfall while the plains lower down receive relatively minor amounts. There is also destruction of property and loss of livestock. The situation is aggravated by a number of other problems such as over-cropping of marginally productive land and other unsustainable farming practices, and deforestation in the watershed.

The study of climate change is becoming a major issue in hydrological studies since it impacts on the water yield of catchments, especially under the impacts of significant human-induced land use changes. In this study region, it has been hypothesised that i) land use change in the upper reaches of the Nzoia basin has led to frequent flooding in the lower Nzoia, ii) climate change has contributed to drying up of rivers in the catchment, iii) climate variability is increasing fast and future weather is likely to be more extreme more frequently, and iv) population dynamics and culture have played a role in the vulnerability of the local communities to floods. This paper addresses the changes in climate, if any, that have taken place, their impacts on the water resources of the region and the future potential impacts. This will be a very useful study to all those involved in the planning, use and management of water resources e.g. domestic, agriculture, industries, water managers etc.

Hydrologic models have been used to investigate the relationship between climate and water resources. Many studies have been carried out that deal with the application of hydrologic models to assess potential impacts of climate change on water resource issues (e.g. Dvorak et al., 1997, Pao-Shan Yu et al., 2002, Miller et al., 2003 among many others). General Circulation Models (GCMs) are important tools in the study of climate variability and change. At planetary scales, GCMs are able to simulate reliably mean features of the global climate, e.g. the intertropical convergence zones (ITCZ), the three-dimensional atmospheric circulation cells, the jet streams, etc. (Zorita and von Storch 1998). However, they are unable to simulate local subgrid-scale features and dynamics (Wigley et al., 1990), a requirement for most hydrologic models. To deal with these problems of scale, downscaling approaches have been used to relate large scale atmospheric predictor variables to local or station scale meteorological series. There are two broad classes of downscaling: dynamic methods that solve the process-based physical dynamics of the system and statistical methods that use system relationships derived from observed data, (Chong-yu and Xu 1999, Wigley et al., 1990).

In this study, the Soil and Water Assessment Tool - a river basin scale model developed by USDA Agricultural Research Service (Arnold et al., 1998) was used to determine the effects of climate change on the water yield of the basin. The GCM CGCM2 climate change predictions were downscaled using statistical downscaling tool, LARS_WG (Semenov & Barrow, 2002). This tool uses statistical methods that use relationships derived from observed data. It simulates time-series of a suite of climate variables at a single site, and can be used to generate long weather time-series suitable for the assessment of agricultural and hydrological risk and to produce high resolution climate change scenarios incorporating changes in climate variability. The simulated weather consists of daily values for minimum temperature, maximum temperature, precipitation and radiation and can be produced for any length of time. The generator utilises semi-empirical distributions for the lengths of wet and dry day series daily precipitation and solar radiation. The distribution is a histogram with a number of intervals that are chosen based on the expected properties of the weather variable. Each interval has a certain number of events from the observed data. The simulation of precipitation is modelled as alternate wet and dry day series. Daily minimum and maximum temperatures are considered as stochastic processes with daily means and daily standard deviations conditioned on the wet and dry status of the day.

Study Area

The study area (Figure 1) is Nzoia River catchment in western Kenya in the Lake Victoria basin and lies between latitudes 1º 30’N and 0º 05’S and longitudes 34º and 35º 45’E. River Nzoia has a catchment area of 12,709 km^2 with a length of 334 km up to its outfall into Lake Victoria. The mean annual rainfall varies from a minimum of 1076 mm to a maximum of 2235 mm with a catchment average of 1424 mm.

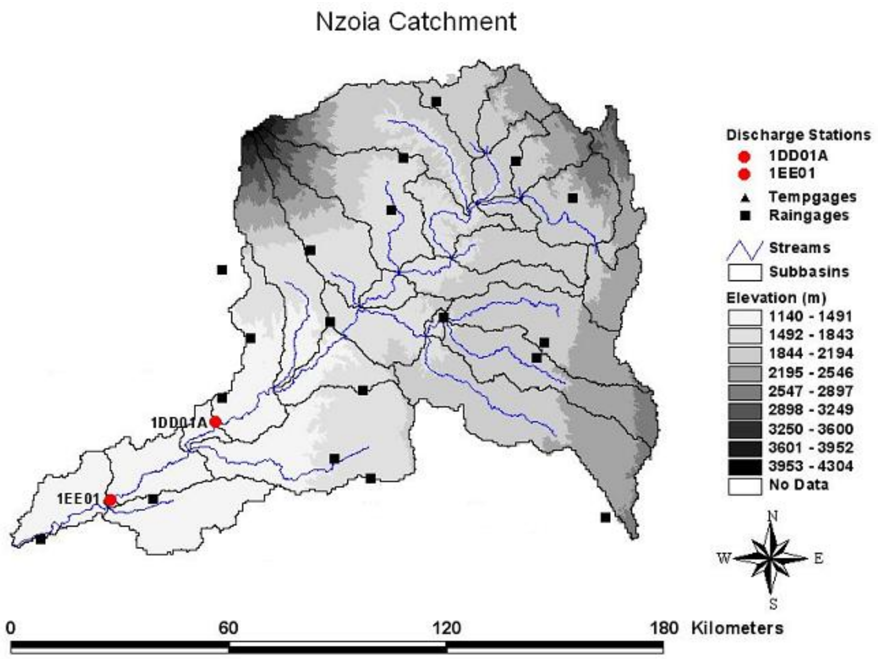


Figure 1: Nzoia Catchment (1.5°N -0.5°S and 34°E-35.75°E)

Methodology

SWAT Model description

SWAT is a continuous time model that operates on a daily/sub-daily time step. It is physically based and can operate on large basins for long periods of time. The sub-basin watershed components can be categorised into the following components – hydrology, weather, erosion and sedimentation, soil temperature, plant growth, nutrients, pesticides and land management. In the land phase of the hydrologic cycle, surface runoff is predicted separately for each hydrologic response unit (HRU) and routed to obtain the total runoff for the watershed. Once the loadings of water, sediment, nutrients and pesticides to the main channel are determined, they are routed through the stream network of the watershed.

SWAT model inputs

The Arcview-SWAT interface was used to generate input files for the watershed. The data used are given in Table 1 below.

Table 1: Data type and source

DATA TYPE	SOURCE
Rainfall, maximum and minimum temperatures, radiation, wind speed, relative humidity	Kenya Meteorological Department
River discharge	Ministry of Water and Irrigation
Land cover data	Food and Agricultural Organization, FAO-Africover project. (produced from LANDSAT images)
Soil data	International Soil Resource and Information Centre (ISRIC) in conjunction with Kenya Soil Survey
Digital Elevation Model (DEM)	Shuttle Radar Topography Mission (SRTM), USA.

Model calibration, validation and evaluation

A sensitivity analysis of the influence of the parameters on the model outputs was carried out. The model was then calibrated for the period 1980-1985 at the gauging stations 1DD01A and 1EE01 (see Figure 1) that drain catchment areas of 10100 and 11760 km^2 respectively. As an initial step, the base flow was separated from the surface flow using Water Engineering Time Series PROcessing Tool, WETSPRO, (Willems, 2000). This is a generalization of the Chapman filter which was based on the linear reservoir modelling concept. The generalization consists of time variability in the fraction of the cumulative values in the total time series that is related to the filtered component. The model was evaluated on the long term from 1968-1995 (1DD01A) and 1970-1990 (1EE01). The evaluation both on calibration and long term results was done based on the mean, standard deviation, coefficient of determination (R^2) and Nash-Sutcliffe Efficiency NSE (Nash and Sutcliffe, 1970).

Climate change scenarios

The GCM model CGCM2 from the Canadian Center for Climate Modelling and Analysis (CCCma) was used with only two scenarios considered, A2 and B2. The A2 scenario describes a very heterogeneous world with emphasis on self-reliance and preservation of local identities. Fertility patterns across regions converge very slowly, which results in continuously increasing global population. Economic development is primarily regionally oriented and per capita economic growth and technological changes are more fragmented and slower than in other storylines. The B2 scenario describes a world in which the emphasis is on local solutions to economic, social, and environmental sustainability. It is a world with continuously increasing global population at a rate lower than A2. It is oriented toward environmental protection and social equity, and focuses on local and regional levels (IPCC, 2001).

The daily data was obtained and used together with observed data in LARS-WG to generate monthly changes in rainfall and temperature. Three stations, Kakamega, Kitale and Eldoret (see Table 2) were chosen for generating climate change scenarios. This is because they are the only stations which had both temperature and rainfall data and are also well distributed in the catchment. After calibration, the simulations were carried out over a period of thirty years each and the mean value calculated. The changes obtained were then averaged over the catchment and these were used in SWAT. Table 2 shows the calibration and validation periods for the three stations. The output from Station 1EE01 was used in the impact assessment of climate change as it is the most downstream station and it is of great significance because flooding always occurs downstream of this station.

Table 2: Stations used in the generation of climate change scenarios

Station	Calibration period	Validation period
Kakamega	1981-1995	1996-1999
Kitale	1981-1995	1996-1999
Eldoret	1977-1990	1991-1999

Results and discussion

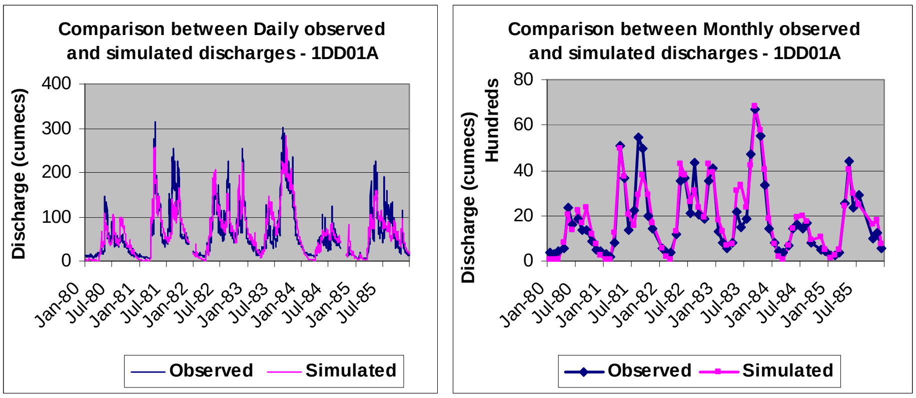
Sensitivity analysis

Results from the sensitivity analysis of the SWAT parameters indicated that the following parameters were most sensitive CN2 (Initial SCS CN II value), RCHRG_DP (Deep Aquifer percolation coefficient), GWQMN (Threshold water in the shallow aquifer for flow), GW_REVAP (Groundwater revap coefficient), CANMX (Maximum canopy storage) and SOL_AWC (Available water capacity). This helped to guide the process of calibration to ensure the differences between the observed and simulated values were minimised.

Calibration and validation

The results of the calibration are presented in Figures 2 (a) and (b). The results show that the model was adequately calibrated, as is evident from the evaluation statistics in Table 3. Observed and simulated daily and monthly flows matched well. Means and standard deviations of the observed and simulated flows for the calibration period at both stations (1EE01 and 1DD01A) were within less than 13% of each other except for the standard deviation for the monthly flows at 1EE01 which was about 17%. Further agreement between the daily observed and simulated flows are shown by values of R^2>0.74 and NSE>0.71. The estimated fractions of base flow from the observed flow at 1DD01A and 1EE01 were 72.4% and 80.8% while those simulated were 71.9% and 79.9%. These results show that the hydrologic processes were modelled realistically and that this model could be used for impact assessment.

(a)



(b)

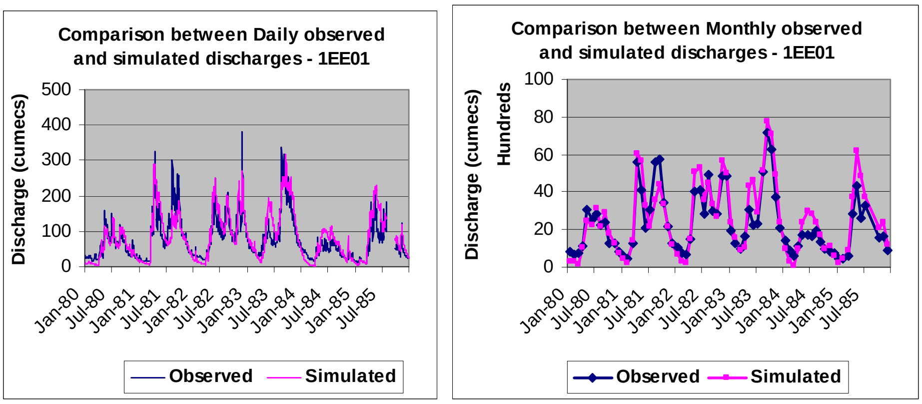


Figure 2: Results for the calibration period; a) daily and monthly discharge at 1DD01A b) daily and monthly discharge at 1EE01

Table 3: Evaluation statistics for the calibration period

		Observed		Simulated			
		Mean (cumecs)	Standard Deviation	Mean (cumecs)	Standard Deviation	NSE	R^2
Daily	1DD01A	61	57	63	53	0.77	0.74
	1EE01	76	59	84	66	0.71	0.78
Monthly	1DD01A	60	52	63	50	0.86	0.85
	1EE01	76	53	83	63	0.76	0.84

The model also performed well on the long term and the evaluation statistics for comparison of daily discharges between the observed and simulated are given in Table 4. Figure 3 shows the annual discharges which were well simulated, except for the year 1971 (at both stations) and 1977/78 (at 1DD01A). The evaluation statistics for the annual discharges are NSE=0.78, R^2=0.78 and NSE=0.69, R^2=0.71 for 1DD01A and 1EE01 respectively.

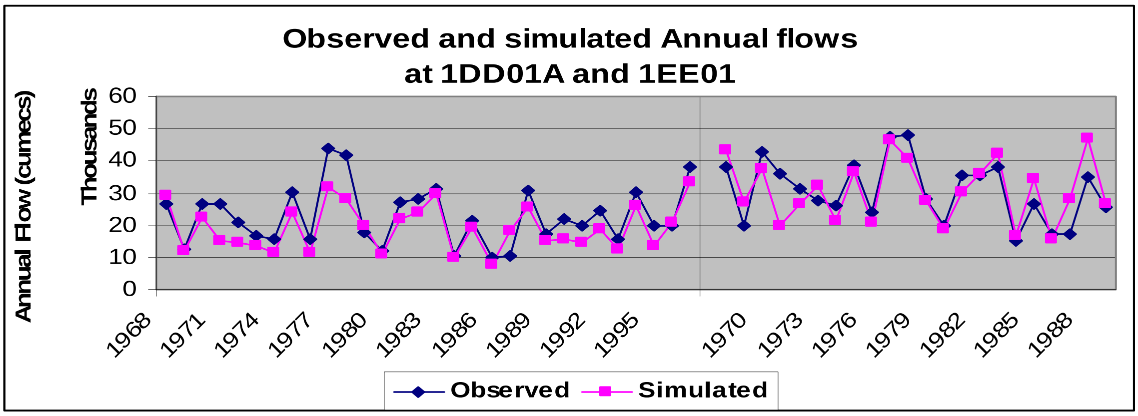


Figure 3: Long term simulation of flows at 1DD01A and 1EE01 (**NB**: The first and second half of the graph represents 1DD01A and 1EE01 respectively)

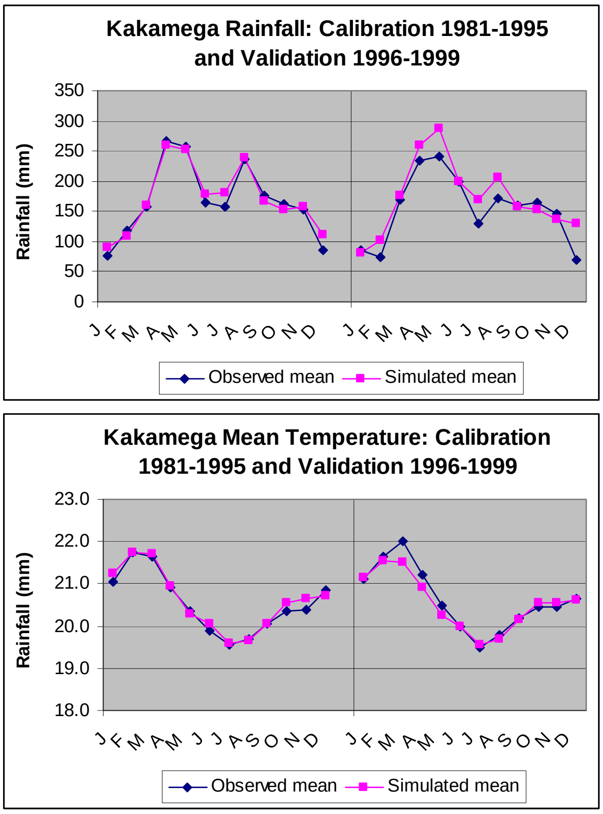
Table 4: Evaluation statistics for the long term period

	Observed		Simulated			
	Mean (cumecs)	Standard Deviation	Mean (cumecs)	Standard Deviation	NSE	R^2
1DD01A	62	57	63	51	0.65	0.66
1EE01	90	60	97	66	0.55	0.65

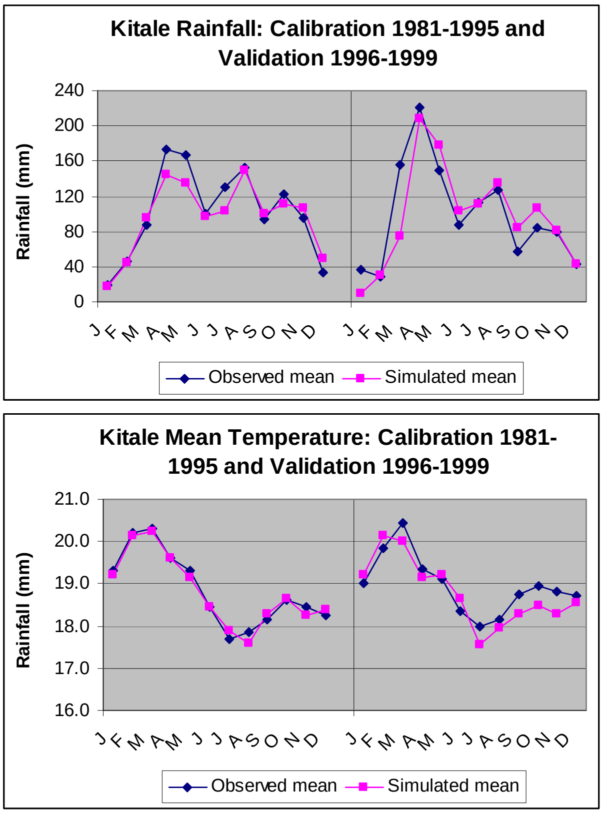
Climate change scenarios

The three stations that were used to generate climate change scenarios are shown in the figures below.

(a)


Kakamega Rainfall: Calibration 1981-1995 and Validation 1996-1999
Rainfall (mm)
350
300
250
200
150
100
50
0
J F M A M J J A S O N D J F M A M J J A S O N D
Observed mean
Simulated mean
Kakamega Mean Temperature: Calibration 1981-1995 and Validation 1996-1999
Rainfall (mm)
23.0
22.0
21.0
20.0
19.0
18.0
J F M A M J J A S O N D J F M A M J J A S O N D
Observed mean
Simulated mean

(b)


Kitale Rainfall: Calibration 1981-1995 and Validation 1996-1999
Rainfall (mm)
240
200
160
120
80
40
0
J F M A M J J A S O N D J F M A M J J A S O N D
Observed mean
Simulated mean
Kitale Mean Temperature: Calibration 1981-1995 and Validation 1996-1999
Rainfall (mm)
21.0
20.0
19.0
18.0
17.0
16.0
J F M A M J J A S O N D J F M A M J J A S O N D
Observed mean
Simulated mean

(c)

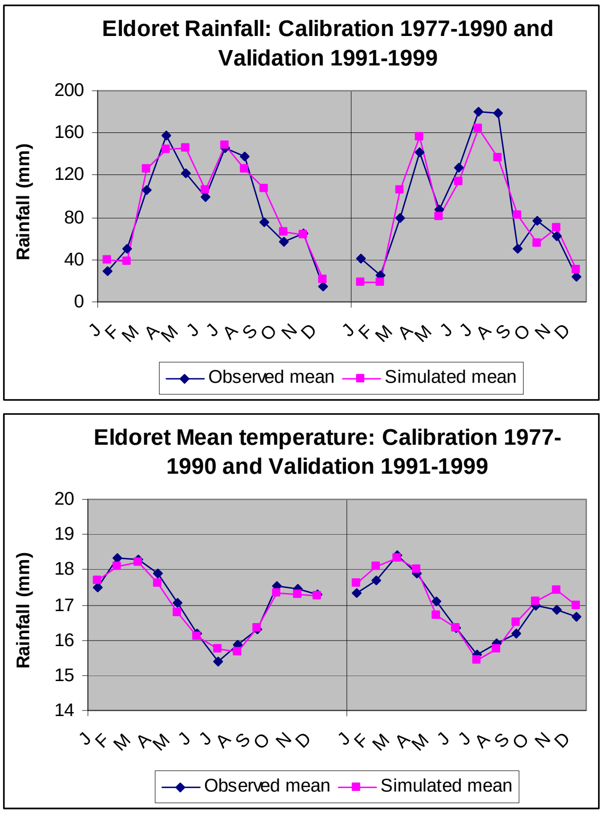


Figure 4: Rainfall and mean temperature for a) Kakamega , b) Kitale and c) Eldoret (**NB**: The first and second half of the graph represent the calibration and validation periods respectively)

Table 5: Values of Coefficient of determination R^2 for mean monthly rainfall and mean monthly temperature.

	Rainfall		Temperature	
	Calibration	Validation	Calibration	Validation
Kakamega	0.94	0.82	0.97	0.92
Kitale	0.88	0.74	0.97	0.83
Eldoret	0.89	0.83	0.95	0.91

Both calibration and validation statistics were quite high for temperature in all the stations (R^2>0.8). Although the R^2 values for rainfall were less than those for temperature, the model was able to adequately simulate the monthly rainfall. Figure 4 (a-c) shows the calibration and validation results for rainfall and temperature. Table 5 gives the corresponding R^2 values.

The CGCM2 data was processed in the same way as the observed data but for the period 1961-1990 as the baseline and the time slices 2010-2039 and 2040-2069 representing the 2020s and 2050s respectively. The monthly changes (% change for rainfall and absolute change for temperature) for each time slice were then imposed on the calibrated model and simulations of 30 years each generated. Monthly mean averages were then calculated for these times slices and superimposed in SWAT to represent changes in climate (*.sub files-these contain variables related to climate change among others). The changes for rainfall and temperature are given in Table 6. Rainfall changes are highly variable from one month to another with the highest changes in December which shows increases in both scenarios in the 2020s and 2050s. Increases in rainfall in May could translate to high surface runoff and this could aggravate the already existing problem of flooding in lower Nzoia.

Temperatures increases in both scenarios in the 2020s are about 0.85°C while in the 2050s, it is 1.69°C for the A2 scenario and 1.37°C for the B2 scenario. The highest temperature increases are observed in the months June and July which are normally the cold season. These changes in rainfall and temperature could imply a number of things ranging from changes in planting seasons, types of crops grown, emergence of diseases where none existed before, change in land management systems among others.

Table 6: (a) Average monthly changes in catchment rainfall (%) b) Average monthly changes in temperature (°C).

(a)

	Scenario A2		Scenario B2	
	2020s	2050s	2020s	2050s
Jan	69.6	-9.0	-6.0	2.2
Feb	14.1	-12.2	-19.6	-10.8
Mar	26.9	0.5	3.6	8.9
Apr	-0.4	-4.8	-1.6	7.7
May	25.6	27.4	39.6	28.8
Jun	19.5	16.9	18.8	33.0
Jul	16.7	9.6	-3.8	11.1
Aug	12.4	-7.1	5.3	-15.5
Sep	42.0	31.4	21.3	-2.2
Oct	21.8	29.2	15.7	17.7
Nov	36.9	-0.6	17.4	-5.9
Dec	226.7	66.9	57.5	59.3

(b)

	Scenario A2		Scenario B2	
	2020s	2050s	2020s	2050s
Jan	0.8	1.7	0.8	1.4
Feb	0.8	1.6	0.7	1.2
Mar	0.8	1.8	0.9	1.3
Apr	0.9	1.7	0.9	1.4
May	0.9	1.7	0.8	1.3
Jun	1.1	1.9	1.0	1.6
Jul	1.2	2.0	1.1	1.7
Aug	0.9	1.7	0.8	1.4
Sep	0.8	1.7	0.9	1.5
Oct	0.6	1.4	0.7	1.2
Nov	0.7	1.6	0.9	1.3
Dec	0.8	1.7	0.8	1.3

Changes in model output

From the model output, the imposed climate change A2 scenario has increased the surface runoff by an average 187% and base flow by 94% in 2020s. In the 2050s the percentages stand at 67% and 26% respectively. For the B2 scenario, these percentages are 100% and 41% in 2020s and 69% and 27% in 2050s (Table 6). In general the A2 scenario is seen to yield more than the B2 scenario for the 2020s whereas in the 2050s the percentage changes are similar for both scenarios. The A2 scenario in 2020s is only slightly warmer (0.002°C) than the B2 scenario but the changes in rainfall are quite large leading to high water yields. In 2050s, the A2 scenario is much warmer than B2 (by 0.32°C) but the changes in rainfall are similar and this leads to similar changes in runoff. This shows that the difference in temperature in this case does not yield significant changes in the catchment yields. Table 7 shows the number of times bankfull discharge (about 270m^3/s) is exceeded in each scenario. More floods are likely to be experienced in the 2020s than in the 2050s according to the two scenarios, where fewer events are observed in scenario B2 than in A2. This will create a need to put in place flood protection structures well designed to withstand discharges of up to about 900m^3/s.

Time series analysis of rainfall and temperature in this catchment has shown that over the last 40 years, rainfall amounts have increased by about 2.3mm/year especially in the upper part of the catchment and mean temperature by about 0.21°C to 0.79°C since 1990. A problem that is likely to worsen with increased rainfall is the reduced flood carrying capacity of the rivers due to excessive siltation of their bed. The river overflows its banks and causes flooding in the lower Nzoia. Severe degradation of the catchment caused by uncontrolled and unregulated human activity, especially large-scale deforestation could further aggravate the problem by increasing flood discharges.

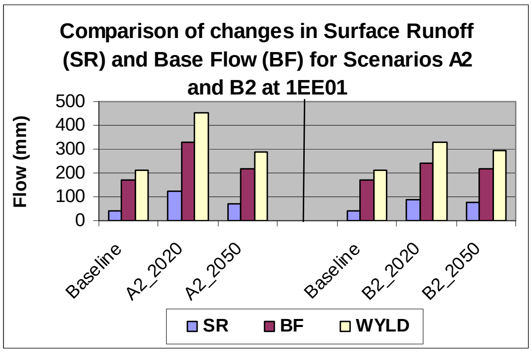


Figure 5: Surface runoff (SR), base flow (BF) and total water yields (WYLD) for Baseline, Scenario A2 and B2 at 1EE01

Table 7: Percentage changes in the Nzoia Catchment yields

	1EE01		
	SR	BF	WYLD
A2_2020	**190**	**94**	**113**
A2_2050	72	27	36
B2_2020	**108**	**40**	**54**
B2_2050	77	27	37

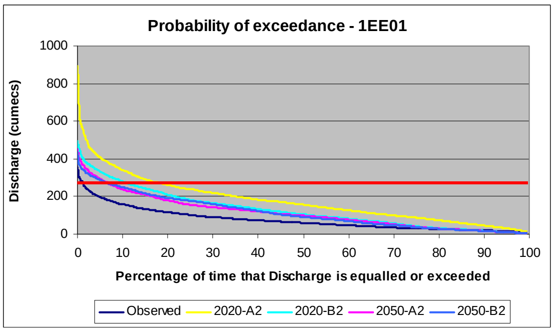


Figure 6: Probability of exceedance

Table 7: Number of times bank full discharge is exceeded at 1EE01

	% time bank full discharge is exceeded	No. of days/year
Observed	1.1	4.8
2020-A2	18.4	77.8
2020-B2	11.0	46.7
2050-A2	6.8	28.6
2050-B2	6.6	27.9

Conclusion

The impacts of climate change on the runoff of the Nzoia catchment were investigated using CGCM2, in combination with SWAT. For the baseline scenario, daily and monthly flows were simulated well, with NSE and R^2 above 0.7. The A2 2020 scenario GCM predicts increases in rainfall of 42% and a temperature change of about 0.8°C yielding an increase in surface runoff of 190%. The B2 2020 predicts rainfall increase of 12% and a temperature change similar to that of A2 2020 but increasing surface runoff by 108%. In the 2050s, A2 yields changes in rainfall, temperature and surface runoff of about 12%, 1.7^{o}C and 72% respectively while B2 yields 11%, 1.4^{o}C and 77%. The A2 scenario is warmer with more increase in rainfall than B2. The results show that the risk of flooding is likely to increase in the future with more flood events in the 2020s than in the 2050s and more so with the A2 than B2 scenario. Only one GCM model (CGCM2) was used and for two scenarios A2 and B2 as these were available from the IPCC data website. However more models and scenarios can be used in order to have a wider range of possible outcomes that can aid in decision making.

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Modeling of a River Basin Using SWAT Model and SUFI-2

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Abstract

This paper presents the hydrologic modeling for the development of management scenario and the simulation of the effect of management practices on water and sediment yielding in Gharasu watershed (5793 km^2) using the Soil and Water Assessment Tool (SWAT2000) model. The SWAT2000 interfaced with Arc View GIS data layers including Digital Elevation Model (DEM), land cover and soil map by AVSWAT2000 software. The model was calibrated from 1991 to 1996 and validated from 1997 to 2000. Then the model was calibrated again using SUFI-2. The results showed there is no considerable difference between the value of parameters that were obtained by SWAT and SUFI-2, but the duration of calibration was reduced from four months to one week. The calibrated model for hydrological conditions was used to assess suspended sediment load. Eventually, the model was used to predict the effect of changing land use and conservation practices on sediment yield within the basin.

KEYWORDS: Karkheh River Basin, sediment yield, simulation, SWAT, SUFI-2

Introduction

Because of geographical and climatic characteristics of Karkheh River Basin, the soil erosion is one of the main severe problems of this basin. The severity of this problem is more pronounced in arid and semi-arid land, where high rain fall intensities of short duration on grazing lands and rain-fed farms and human mismanagement of land has accelerated soil losses by erosion. 19% of the upper watershed's rangelands and 70% of its forests have been significantly degraded [5]. Unless erosion is controlled, sedimentation will significantly reduce the storage capacity of the Karkheh dam reservoir. The Karkheh River Basin has an average yearly sediment yield of 920 tones per km^2 each which is one of the country's highest [5]. In this paper, one of the sub-basins of Karkheh River Basin, Gharasu River Basin, was chosen to determine soil erosion and sedimentation transport loading pattern. The main problem of Gharasu basin is conversion of rangelands to rain fed crop in hilly lands without any conservation practices.

SWAT2000 Description

SWAT2000 has been chosen for this study because it can be used in large agricultural river basin scales and it is easy to use for simulating crop growth and agricultural management.

SWAT[1] incorporates features of several ARS[2] models and is a direct outgrowth of the SWRRB[3] model. SWAT can be used to simulate a single watershed or system of multiple hydrologically connected watersheds. Each watershed is first divided into sub-basins and then into hydrologic response units (HRUs) based on the land use and soil distribution. By using a DEM and stream network, the study area is divided into 66 sub-basins. Each sub-basin is further divided into 437 HRUs, which are determined by unique intersections of the land use-soils within each sub-basin. Each HRU within a given sub-basin can be characterized with a unique set of management practices such as crop growth and irrigation.

The water storage components are soil profile, shallow aquifer, deep aquifer and snow cover. A daily water budget is established for each HRU based on precipitation, surface runoff, evapotranspiration, base flow (groundwater and lateral flow), percolation and soil moisture change. A detailed theoretical description of SWAT and its major components can be found in Neitsch et al. (2002) [9].

SWAT is widely used in the United States and in other regions of the world; exploring the potential impact of reforestation on the hydrology of the upper Tana river catchments and the Masinga dam in Kenya (9753 km^2) [7], hydrologic modeling of the Iroquois River watershed, simulation of hydrologic and sediment loading in connonsville River Basin (1200 km^2) [3], water quality modeling for the Raccoon River watershed (9397 km^2) in west central Iowa [8], sediment, nitrogen and phosphorus loading simulation of Bosque River TMDL in Earth county, Texas [11]. In this study, simulation of hydrologic and sediment loading by SWAT has been performed in approximately large basin (5793 km^2). The model calibration by SWAT is time consuming, so in this study SUFI-2 (Sequential Uncertainly Fitting Ver. 2) [1] was used to evaluate SWAT by performing calibration and uncertainly analysis. SUFI-2 is a semi-automated inverse modeling procedure for combined calibration-uncertainly analysis [2].

Characterization of study area

Gharasu River Basin is located in the north west of Karkheh River Basin in the far western corner of Iran. The area of the basin is approximately 5793 km^2. The elevation changes from 1237 to 3350 m and the mean elevation is 1555 m. The average land–surface slope from DEM is 14%. Annual mean temperature of the study area is 14.6 °C, varying from 1.1 °C in February to 27.3 °C in August. annual average precipitation is about 447 mm, ranging from 215 mm to 785 mm. The predominate land use is agriculture which covers about 67% of the basin (Landsat 1993). Wheat and barley are the major crops grown in the basin. Some 5370 km^2 of the total area of basin is drained into the outlet, where the main gauge station, Gharabaghestan, is located. Soil is predominately a heterogeneous mix of silt or clay with some local deposits of sand in lowlands. Soil texture in lowland is clay to heavy clay and poor drainage.

Daily weather data for precipitation, maximum and minimum temperature were obtained from the records of the climate stations and rain gauge stations for the period of 1988 – 2000. Twenty years (1980–2000) of monthly rainfall, maximum and minimum temperature, relative humidity, wind speed and solar radiation data of the basin were obtained from two climate stations. Daily stream flow was obtained from three stations and Total Suspended Solids (TSS) were obtained from two stations for the period from 1991 to 2000 within the basin and the main station located at the outlet of the basin.

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[1] Soil and Water Assessment Tool
[2] Agricultural Research Service (USDA)
[3] Simulator for Water Resources in Rural Basins

In total, 1172 discharge and sediment samples were collected for generating monthly TSS. The monthly TSS was used for model calibration and validation. Figure 1 shows the location of the stream flow, TSS, rain gauges and climate stations used in the model calibration. Data layers include DEM (50×50 m), land use (Landsat 1993), soil map and streams shape file. The soil map includes 8 types of soils. Soil texture, percent of silt, clay and sand and organic carbon content information was available for different layers of soil. Six main classes of land use were: agriculture (rain-fed irrigated), range (poor-fair-good) and mixed-forest. Winter wheat is chosen as a main growing the crop basin. After a tillage operation, winter wheat is planted on the 20th of October, it is harvested on the 15th to 20th and the soil is tilled again. About 400 mm of water is used every 15 days for irrigation during 6 months. Table 1 summarizes the data used to develop, calibrate, and validate the model.

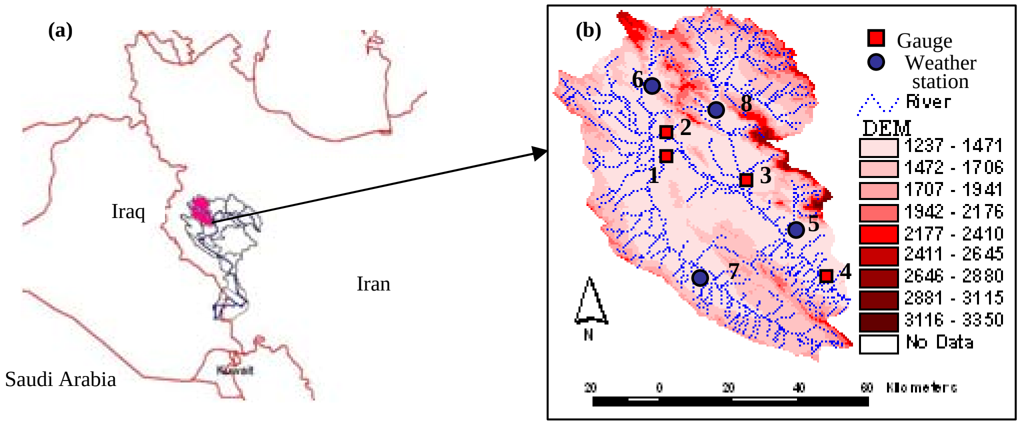


Figure 1 Study area: (a) Location of Karkheh river basin in Iran (b) Location of flow, climate and TSS stations in Gharasu sub-basin.

Table 1. Summary of data used in model development, calibration and validation.

Data	Location	Period of	Organization	Primary use
Stream flow Monitoring	Khers abad (1) Doab merek (2) Hojat abad (3) Gharabaghestan	1974-present 1954-present [illegible] 1954-present	IWRM	Calibration and validation
TSS Monitoring	Khers abad (1) Doab merek (2) Gharabaghestan	1974-present 1964-present 1962-present	IWRM	Calibration and validation
Climate	Kermanshah (5) Ravansar (6)	1951-present 1988-present	IRIMO	Model input
Rain gauge	Mahidasht (7) Jelogireh (8)	1975-present 1976-present	IRIMO	Model input
Land use	Basinwide	1993	RIAEP	Model input
Stream	Basinwide	Unknown	SCWMRC	Model input
Soils	Basinwide	Unknown	SWRI	Model input
Digital	Basinwide	Unknown	SCWMRC	Model input

Note: IWRM=Iran Water Resources Management; IRIMO=I. R. of Iran Meteorological Organization; RIAEP=Research Institute for Agricultural Economics and Planning;

Initial setting of parameters

After preparing required data files and information layers, the model was run. Then independent of numerical calibration, a number of model inputs and parameters adjusted to better represent known conditions in the basin. These parameters are presented in table 2. All data–driven input parameters in table 2 are constant in the calibration and validation periods. More details about the determination of these parameters can be found in [10].

Table 2. Summary of initial setting of the SWAT model parameters.

Parameter	SWAT variable name	Range	Default value	Final value
Snowfall temperature (ºC)	SFTMP	±5	+1	+2
Surface runoff lag coefficient	SURLAG	1-40	4	1
Manning's "n" value for overland	OV-N	0.01-	0.15	Engman, 1983 [6]
Manning's "n" value for the main	CH-N2	0.01-	0.01	Chow, 1959 [4]
Lateral flow travel time (days)	LAT-	0-180	0	Calculated and Varied
Temperature lapse rate (ºC/km)	TLAPS	0-50	6	5
Elevation at the center of the	ELEVB	0-	0	Determined from
Fraction of sub-basin area within	ELEVB-	0-1	0	AVSWAT elevation
USLE equation support practice	USLE-P	0.1-1	0	1
Soil erodibility (K) factor (units: 0.013 (metric ton m^2 hr)/(m^3 metric ton cm))	USLE-K	0-0.65	0	Mountain (0.3) Hill (0.4) Other areas (0.27)
Minimum value of USLE C factor for water erosionapplicable to the land cover/plant	USLE-C	0.001-0.5	Agricultural land	0.03
			Range (0.0	Good (0.002) Fair (0.003) Poor (0.004)
			Forest(0.0	0.001
Rock fragment content (% total	ROCK	0-100	0	Varied by soil type

Model calibration and validation

Continuous discharge data and a large number of TSS samples over 10 years from multiple locations within the basin were used for model calibration and validation. The model was calibrated over 6 years, from January 1991 to December 1996. Four years (1987 to 1990) were chosen as a warm-up period in which the model was allowed to initialize and then approach reasonable starting values for model state variables. Model predictions are not evaluated in accordance with the 4-year warm-up period until another 4 full years have been simulated. Some parameters used to simulate TSS were driven from available data or known conditions in the watershed.

In this study, the calibration process begun by 25 parameters in the SUFI-2 algorithm, but in the last iteration only 16 were found to be sensitive to discharge and sediment, because high correlated parameters with the smallest sensitivities were not changed any longer in the iteration process. In each iteration, 500 model calls were performed, for a total of 3000 simulations. The calibration parameters are presented in table 3. As shown in table 3, there is not considerable difference between the value of parameters that were calculated by SWAT and SUFI-2, but the duration of calibration was reduced 113 days and more numbers of parameter were determined, such as groundwater delay time (GW_DELAY), Manning's "n" value for overland flow (OV_N) and channel erosion parameters. In previous study [10] the channel erosion was ignored but by using SUFI-2 we could determine the stream channel erosion parameters. The parameters are ranked according to their sensitivities in table 3. Five parameters were found to be sensitive to sediment only. These included channel re-entrained exponent parameter (SPEXP), channel re-entrained linear parameter (SPCON), peak rate adjustment factor for sediment routing in the main channel (PRF), channel erodability factor (CH_EROD) and channel cover factor (CH_COV). Other parameters were found to be sensitive to both discharge and sediment; but the influence of two parameters (ALPHA_BF and GW_REVAP) on sediment load was negligible.

The results of the monthly discharge and TSS simulation are shown in figure 2. These simulations are based on a calibration that used monthly discharge and TSS in the objective function. The objective function used in this study is the sum of square errors. The shade region (95PPU), brackets a large amount of the measured data, which contains all uncertainties such as rainfall, soil properties and water consuming. SUFI-2 is a stochastic procedure, so statistics such as percent error, R^2 and Nash-Sutcliff, which compare two signals, are not applicable. Instead, the 95% prediction uncertainty (95PPU) was calculated for all the variables in the objective function [2]. This is calculated by the 2.5th (X_L) and 97.5th (X_U) percentiles of the cumulative distribution of every simulated point [2]. The parameter ranges leading to the 95PPU are presented in the table 3. The d-factor is the ratio of the average distance between the above percentiles and the standard deviation of the corresponding measured variable [2]. In discharge calibration, 83% of the measured data were bracketed by the 95PPU while the d-factor was 1.47.

The model was validated over 4 years, from January 1997 to December 2000. The longest–running flow gauge for the basin drains approximately 93% of the basin (station 4 in fig. 1). In addition, the three gauges that drain the smaller sub-basins were used during the calibration procedure (Station 1, 2 and 3 in fig. 1). The result of calibration and validation for TSS simulation at the main outlet of the basin is shown in figure 3.

Table 3. Initial and final values of SWAT calibration parameters for stream flow.

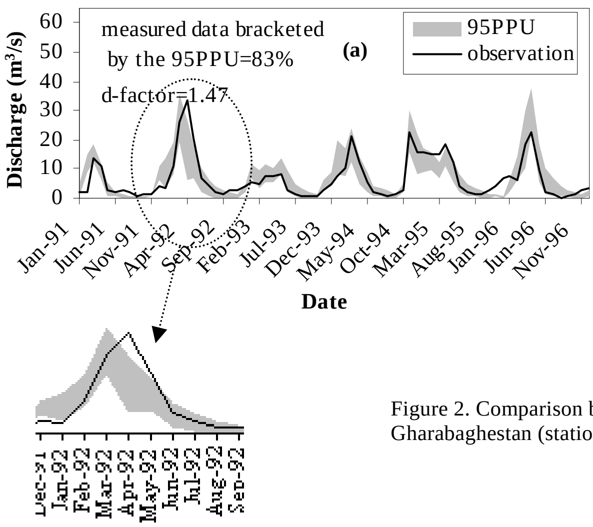
Parameter	SWAT variable name	Final value	
		SUFI-2	SWAT
ESCO	Soil evaporation compensation factor	0.48(a) 0.61(b)	0.40
SMFMN	Melt factor for snow on December 21	2.77(a)	2.5
SMFMX	Melt factor for snow on June 21 (mm	2.82(a)	2.6
GW_REV	Groundwater "revap" coefficient	0.06(a),(b) 0.04(	0.04(a) 0.06(b) 0
SMTMP	Snow melt base temperature (°C)	+3.55	+4
ALPHA	Base flow alpha factor (days)	[0.08 . 0.23]	0.118(a)
CH_K2	Effective hydraulic conductivity in the	45.71	40(a) 60(b),(c)
GW_DEL	Groundwater delay time (days)	[43 . 100]	Varied by HRU
GWQMN	Threshold depth of water in the shallow	[-20 . 171]	40(a),(b) 20(c)
OV_N	Manning's n value for the over land flow	[-0.13 . 0.24]	0.29(a) 0.3(b),(c)
SFTMP	Snow fall temperature (^{0}C)	1.91	2.0
REVAPM	Threshold depth of water in the shallow	[-33 . 118]	20(a),(b) 10(c)
PRF	peak rate adjustment factor for sediment	0.38	0.5
SPEXP	channel re-entrained exponent parameter	1.04	1.05
SPCON	channel re-entrained linear parameter	0.0016	0.002
CH_ERO	channel erodability factor	0.32	0.0
CH_COV	channel cover factor	0.49	0.0

(a) The area of basin that drained into Khers abad station (Station 1 on the map of Fig. 1) (1420 km^2).
(b) The area of basin that drained into Doab merek station (Station 2 on the map of Fig. 1) (1232 km^2).
(c) The area of basin that drained into Hojat abad and Gharabaghestan stations (Station 3 and 4 on the map of Fig. 1) (2718km^2).

The simulated flow of January, February and March is more than the observed flow in 1992, and it is less than the observed flow in April and May. It seems simulated snowmelt occurs sooner than actual time. Consistent with hydrology results, figure 3 demonstrates that at the main outlet of basin the model tends to increase TSS loading sooner in the winter of 1992 associated with snowmelt. The most severe errors in predicted TSS loads all occur in months where there are large predictive errors in the monthly flow.

In the previous study, average annual sediment yield of Gharasu basin was predicted 3.4 ton/ha by SWAT model, but in this study it is predicted 3.2 ton/ha by SUFI-2. Comparison of the values of hydrologic components that were calculated using SUFI-2 and SWAT showed the lateral flow and base flow changed more than other hydrologic components. Therefore, the change of sediment load was negligible, because it is not affected by lateral and base flows.

After sureness of model validity, the erosion map of sub-basins was provided. It is schematized in figure 4 from 1997 to 2000. By using this map the critical basin were specified (fig. 5). Comparison of erosion map and DEM showed that the critical sub-basins are located in mountainous and hilly areas. Moreover, comparison of sediment yield of HRUs indicates the most erosive areas are cultivated lands with steep slope. Land use type of hilly area is very important because most of the rain-fed lands are located in this area and the type of geology is low to medium resistance to erosion. So, vulnerability to soil erosion and sediment yield in these areas are high.



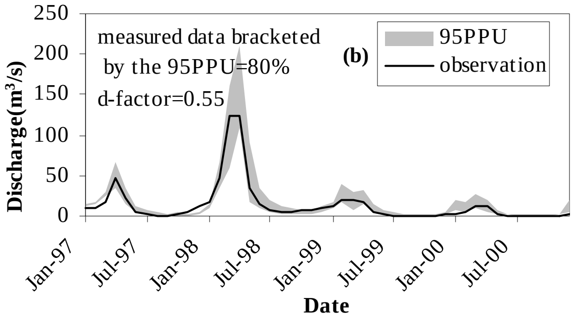


Figure 2. Comparison between observed and simulated monthly stream flow at Gharabaghestan (station 4) for: (a) Model calibration (b) Model validation.

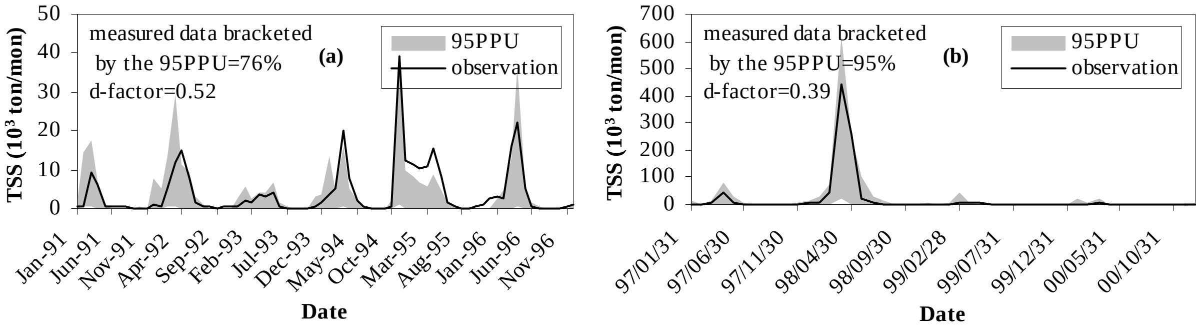
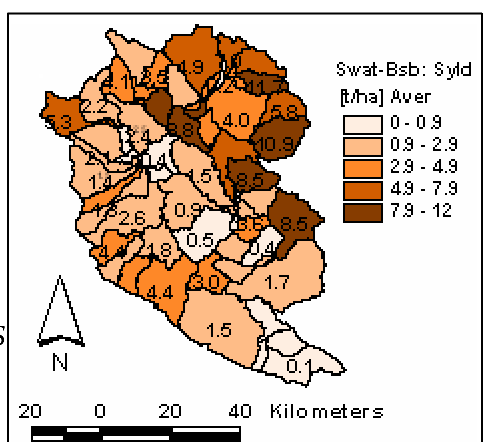
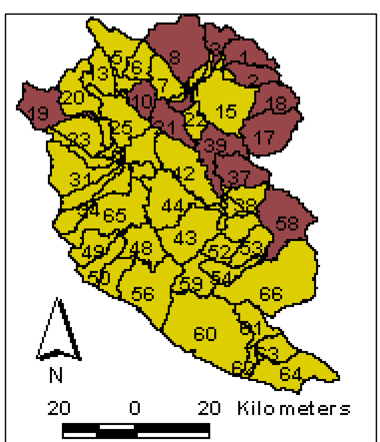


Figure 3. Comparison between observed and simulated monthly TSS at Gharabaghestan (station 4) for: (a) Model calibration (b) Model validation.





Irrigated agricultures are concentrated in the alluvial area and along the valley due to gentle slopes and its productive soils. Because of the gentle slope and heavy soil texture, little erosion occurs in these regions.

With consideration of the above explanations, some management scenarios are recommended for soil conservation:

1- Support practices such as contouring and terracing.

2- Land use change in hilly and mountainous areas of basin with due consideration of land capability.

First scenario: With due attention to topographic conditions and possibility of "contouring" or "contouring and terracing" the critical sub-basin 16, 17, 19, 37 and 39 are suitable for land management practices. Reduction of erosion in the agricultural HRUs located in lower parts of these critical sub-basins is presented in table 4. As shown in table 4, contouring and terracing is more effective than contouring.

Second scenario: Because land management practices in hilly and mountainous areas are impracticable, land cover changing of these areas is recommended for soil conservation. The hilly areas are suitable for afforestation. Therefore, rain-fed lands and other land uses located in hilly areas are converted to forest. The land cover of hillsides is converted to orchard. Finally, the mountainous areas are suitable for pasture and range.

The results of land use conversion are presented in table 5. The best effect of the land use conversion on sediment yield reduction occurs in sub-basins that rain-fed lands on hillsides are predominate land use (sub-basin 3, 8 and 19). Sediment yield reduction of mountainous sub-basins is negligible (sub-basin 10, 16, 37 and 39). In these sub-basins the main factor of erosion is steep slope, and land use conversion is not effective.

Table 4. Summary of support practices results on sediment yield.

Sub-basin	Area of HRU (%)	Initial sediment yield (ton/ha)	Predicted sediment yield (ton/ha)		Sediment yield reduction of sub basins (%)
			Contouring (Reduction %)	Contouring and Terracing (Reduction %)	
16	9	25.0	19.5 (22)	15.1 (40)	5
	3	0.6	0.28 (53)	0.28 (53)	
17	3	29.0	25.1 (13)	18.8 (35)	1
19	4	13.8	9.8 (29)	7.5 (46)	2
37	3	42.1	34.3 (19)	16.8 (60)	4
	4	22.6	20.3 (10)	13.7 (39)	
	1.5	1.3	0.0 (100)	0.65 (50)	
39	5	28.0	22.9 (18)	17.7 (37)	5
		6	7.8	5.1 (35)	

Table 5. Summary of land use conversion results on sediment yield.

Sub-basin	Initial sediment yield (ton/ha)	Predicted sediment yield after land cover changing (ton/ha)	Sediment yield reduction of sub-basins (%)
3	6.83	0.42	94
19	4.12	0.25	94
8	4.63	0.33	93
17	7.52	2.98	60
1	5.12	2.13	58
18	4.63	2.05	56
2	10.21	6.94	32
58	6.32	4.73	25
21	6.15	5.24	15
37	8.75	8.85	0.03
39	7.93	7.83	0.03
10	6.71	6.52	0.01
16	3.42	3.51	0.01

.Conclusions

In this study SUFI-2 was used for model calibration and validation. By using SUFI-2, we could perform uncertainly analysis and calibrate the model for more number of parameters. Also, the duration of model calibration was reduced from four months to one week. Two different management scenarios for soil conservation were considered in order to evaluate the effects on sediment yielding in Gharasu river basin. Contouring and terracing will effectively reduce sediment loading of rain-fed lands in hillsides. Changing agricultural practices such as increasing forest, conversion of rain-fed area in steep slope land to orchards and woods will reduce erosion about 5 percent within hilly and mountainous sub-basins. Finally, this study showed that the SWAT model is a capable tool for simulating hydrologic components and erosion in Gharasu river basin.

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Application of the SWAT Model to the Hii River Basin, Shimane Prefecture, Japan

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Abstract

We tried to apply the SWAT model to the Hii River basin from 1986 to 2005 by daily time step. As a first step of application of the SWAT model to the basin, we paid attention to discharge of the River. The Hii River basin is in the eastern part of Shimane Prefecture, Japan. It covers an area of about 900 km^2 and the length of the river from the source to Ootsu river discharge observation station, where is outlet of whole basin, is about 150 km. About 80% of the land use in the basin is forest and 10% is paddy fields. The parameters were calibrated from 1993 to 1996 and validated from 1986 to 1992 and from 1997 to 2005. The calibrated parameters automatically, which were CANMX, ALPHA_BF, SOL_AWC, SOL_Z, CH_K2, SMFMX, GWQMN, CN2, ESCO and SLOPE, were selected by ranking of sensitivity analysis. The both results of calibration and validation were represented fluctuations of discharge relatively well, though some peaks were overestimated. During the calibration period, R^2 varied from 0.65 to 0.77 and *NSI* did from 0.64 to 0.76. During the validation period from 1986 to 1992, R^2 varied from 0.58 to 0.74 and *NSI* did from 0.53 to 0.74. As well, from 1997 to 2005, R^2 varied from 0.51 to 0.71 and *NSI* did from 0.38 to 0.68. Due to advance a next step, it will be necessary to improve density of climatic gages and accuracy of soil and land use information.

KEYWORDS: Runoff analysis, watershed management, GIS

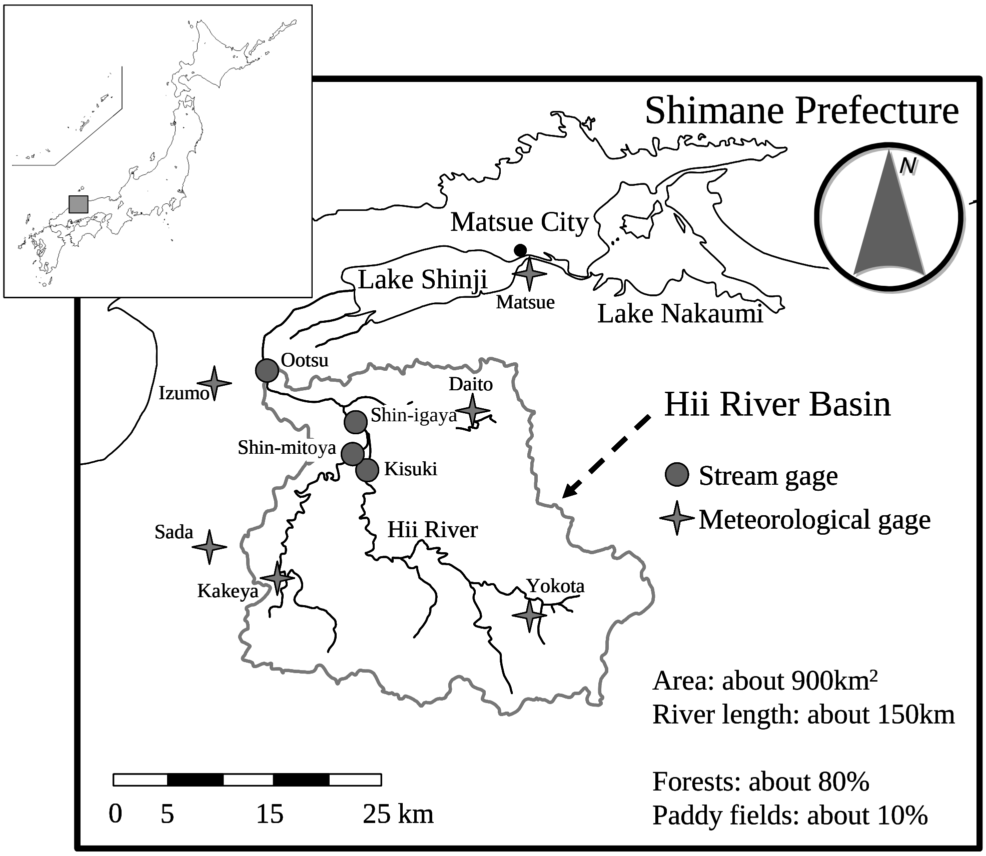
Introduction

Impact assessments of land use change, population growth / decrease and watershed development to water quantity and quality are one of the most important topics in a basin. As well, integrated managements of water environment from river basin to downstream such as lake are also very important for conservation and sustainable use of its resources. In recent years, water quality in lakes are tried to improve until under environmental standard by emission control of pollutant loads to lake and rivers through putting an adequate sewage system in place and development of laws, though water quality in lakes have not been improved well as we expected. One of the reasons is considered to be pollutant loads discharged from non-point sources such as agricultural land.

There are lakes called Lake Shinji and Lake Nakaumi where have not been improved water quality well in Shimane prefecture, Japan. The Lake Shinji and Lake Nakaumi have been designated as one of the Wetlands of International Importance by the Ramsar Convention in November 2005.

Many researchers study about water quality targeted at Lake Shinji and Lake Nakaumi from several perspectives (e.g. Seike et al., 2006; Sakuno et al., 2003). As well, there are some studies targeted at Hii River (Takeda et al., 1996; Ishitobi et al., 1988). However, few studies are done about runoff analysis and quantitative analysis of pollutant loads by a model in the Hii River basin. When considering watershed management and improvement of water environment in lakes, both information of lakes and rivers will be necessary. Thus, we tried to represent stream flow in the Hii River basin as a first step of water environment management.

Figure 2. Location of Hii River Basin.



Study Area

The Hii River basin is in the eastern part of Shimane Prefecture, Japan (Figure 1). It covers an area of 914.4 km^2 and the length of the river from the source to the Ootsu river discharge observation station, where is outlet of whole basin, is about 150 km. According to the Chugoku Regional Development Bureau in the Ministry of Land, Infrastructure and Transport Government of Japan (MLIT: http://www.cgr.mlit.go.jp/), Yearly averaged discharge is about 40 m^3/s and amount of total runoff is about 1270 Mm^3. About 80 % of the land use in the basin is forest and 10 % is paddy fields. As the Hii River dominates about 75 % of watershed area flowing into the Lake Shinji, it is considered that water quality and quantity of the river will affect the Lake a lot.

Methodology

We tried to apply the SWAT model to this basin from 1986 to 2005 by daily time step. As a first step of application of the SWAT model to the basin, we paid attention to discharge of the river. The Hii River basin was divided by four sub basins according to locations of stream gages in the basin (Ootsu, Shin-igaya, Shin-mitoya, and Kisuki). The parameters were calibrated from 1993 to 1996 and validated from 1986 to 1992 and from 1997 to 2005. The ten parameters selected by ranking of the sensitivity analysis were calibrated automatically for all sub basins using daily discharge data as shown in Table 1.

Table 2. Range and optimal values of SWAT2003 calibration parameters.

Parameter name	Lower bound	Upper bound	Optimal value	Imet
CANMX: Maximum canopy storage (mmH_2O)	0.0	10.0	0.009	1
ALPHA_BF: Baseflow alpha factor (days)	0.0	1.0	0.75	1
SOL_AWC: Available water capacity of the soil layer (mmH_2O/mm soil)	-0.04	0.04	0.04	2
SOL_Z: Depth from soil surface to bottom of layer (mm)	-50.0	600	588.2	2
CH_K2: Effective hydraulic conductivity in main channel alluvium (mm/hr)	0.0	150.0	150.0	1
SMFMX: Melt factor for snow on June 21 (mmH_2O/°C-day)	2.0	8.0	2.09	1
GWQMN: Threshold depth of water in the shallow aquifer required for return flow to occur (mmH_2O)	0.0	5000.0	0.35	1
CN2: Initial SCS runoff curve number for moisture condition II	-8.0	8.0	-6.6	2
ESCO: Soil evaporation compensation factor	0.001	1.0	0.89	1
SLOPE: Average slope steepness (m/m)	0.0	0.6	0.0002	1

Note: Imet means variation methods available in auto calibration (1: Replacement of initial parameter by value, 2: adding value to initial parameter)

Brief descriptions of SWAT model

The Soil and Water Assessment Tool (SWAT) has been widely applied for modeling watershed hydrology and simulating the movement of non-point source pollution. The SWAT is a physically-based continuous time hydrologic model with an ArcView GIS interface developed by the Blackland Research and Extension Center and the USDA-ARS (Arnold et al., 1998) to predict the impact of land management practices on water, sediment, and agricultural chemical yields in large complex basins with varying soil type, land use, management conditions over long periods of time. The main driving force behind the SWAT is the hydrological component. The hydrological processes are divided into two phases, the land phase, which controls the amount of water, sediment, and nutrient loading in receiving waters, and the water routing phase which simulates movement through the channel network. The SWAT considers both natural sources (e.g. mineralization of organic matter and N-fixation) and anthropogenic contributions (fertilizers, manures and point sources) as nutrient inputs. The SWAT delineates watersheds into sub basins interconnected by a stream network and each sub basin is divided further into hydrologic response units (HRUs) based upon unique soil / land class characteristics, without any specified location in the sub basin. Flow, sediment, and nutrient loading from each HRU in a sub basin are summed and the resulting loads are then routed through channels, ponds, and reservoirs to the watershed outlet (Arnold et al, 2001). The model includes a number of storage databases (i.e. soils, land cover/ plant growth, tillage, and fertilizer) which can be customized for an individual basin. A single growth model in SWAT is used for simulating all crops based on the simplification of the EPIC crop model (Williams et al., 1984). Phenological development of the crop is based on daily heat unit accumulation. The model can simulate up to 10 soil layers if sufficiently detailed information is available. The SWAT is expected to provide useful information across a range of timescales, i.e. hourly, daily, monthly, and yearly time-steps (Neitsch et al., 2002).

Input data descriptions

The SWAT requires meteorological data such as daily precipitation, maximum and minimum air temperature, wind speed, relative humidity and solar radiation data. As well, spatial data sets include a digital elevation map (DEM), land cover and soil maps are required. Since some holes were present in the climate data, the weather generator included in SWAT was used, based on statistical values (average monthly values of rain, maximum and minimum temperature, standard deviation, skew coefficient, probability of wet day following a dry day in the month, probability of wet day following a wet day, average number of rainy days in the month), and computed on the basis of available daily values.

Meteorological data was obtained from the Japan Meteorological Agency (JMA: http://www.jma.go.jp/jma/index.html). Measuring gages of precipitation, air temperature and wind speed were located in /around the basin. We chose five gages for precipitation and three gages for air temperature and wind speed. However there is no gage monitoring relative humidity data in the basin. So, relative humidity data observed in Matsue city where is located about 30 km away from the basin was used instead. As well, solar radiation data was calculated with the Angstrom formula (FAO, 1998) by using the data measured by Shimane University (http://www.ipc.shimane-u.ac.jp/weather/i/home.html) and actual sunshine duration in the basin obtained from the JMA because of no monitoring gage of solar radiation in the basin. The average values of climatic data at each gage are shown in Table 2.

Table 3. Average annual precipitation and climatic variables from 1985 to 2005 at each gage.

Gage name	EL. (m)	Annual Precip. (mm)	Max. Air temp. (deg. C)	Min. Air temp. (deg. C)	Wind speed (m/s)	Relative humidity (%)	Solar radiation (calculated) (MJ/m^2)
Matsue	16.9	-	-	-	-	75.6 (10.0)	-
Izumo	20	1726	18.9 (8.3)	10.3 (8.1)	2.2 (1.2)	-	11.1 (7.5)
Daito	56	1778	-	-	-	-	-
Sada	100	2072	-	-	-	-	-
Kakeya	215	2046	18.0 (9.0)	8.8 (8.3)	1.3 (0.7)	-	-
Yokota	369	1765	17.2 (9.3)	7.5 (8.8)	1.2 (0.7)	-	-

Note: The values in the parenthesis indicate a standard deviation

Discharge data was prepared at four monitoring stations named Ootsu, Shin-igaya, Shin-mitoya and Kisuki in the basin. The data was furnished from the Izumo River Office in the MLIT.

DEM data was prepared with 50 m grid created from 1:25,000 topographic map of the Geographical Survey Institute.

Land use data is digital national information categorized such as paddy field, upland field, orchard, denuded land, forest, water and others. The data was obtained from the National-Land Information Office in the MLIT (http://nlftp.mlit.go.jp/). The land use in each sub basin is almost very similar and impartial. Forest area changes from 59 % to 87 % and paddy fields area does from 9 % to 18 % spatially as shown in Table 3.

Table 4. Area and ratio of major land use in each sub basin.

Gage name	Sub basin No.	Drainage area	Sub basin			
			Area	Forests	Rice fields	Upland Fields and Orchard
		(Km^2)	(Km^2)	(%)	(%)	(%)
Ootsu	Sub 1	914.4	183.9	74	16	3
Shin-igaya	Sub 2	730.5	14.1	59	18	5
Shin-mitoya	Sub 3	206.8	206.8	86	9	3
Kisuki	Sub 4	509.6	509.6	87	10	2

Soil type data was clipped from a soil map GIS data in 1:500,000 Fundamental Land Classification Survey prepared by the MLIT (http://tochi.mlit.go.jp/tockok/index.htm). Soil type was categorized as ten groups of fourteen soils such as Dystric Rhegosols, Fluvic Gleysols, Gleysols, Haplic Andosols, Helvic Acrisols, Humic Cambisols, Lithosols, Ochric Cambisols, Rhodic Acrisols, and Vitric Andosols. Internal data of each soil such as the number of layers, soil depth and phsico-chemical properties was prepared based on soil profile in soil map and the data colleted up by Hirai (1995).

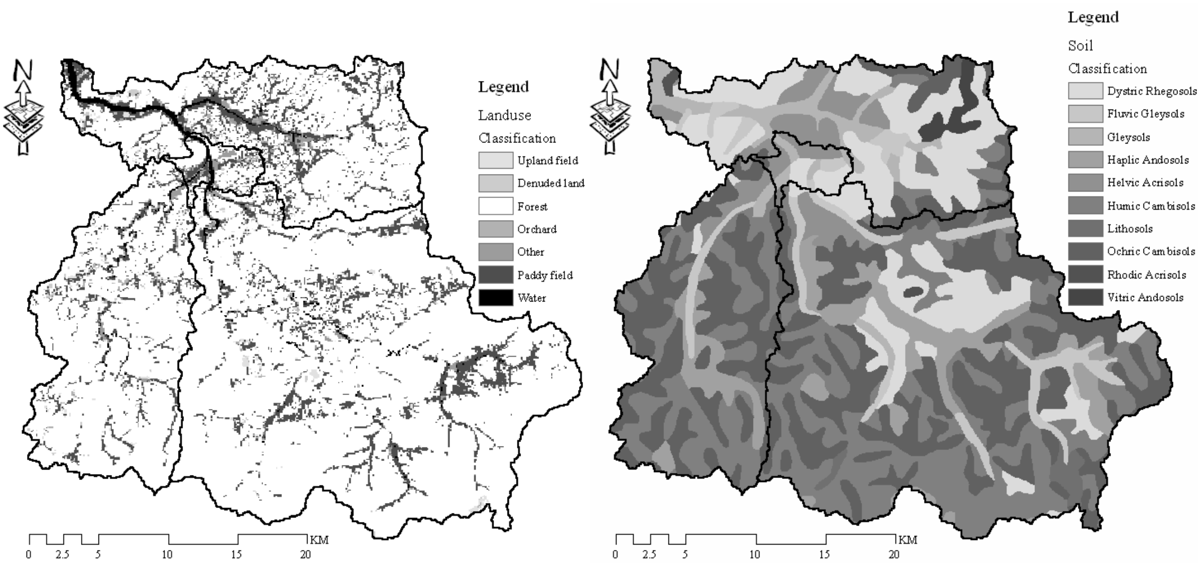


Figure 3. Land use and soil classification GIS data.

Model performance evaluation

The swat model was calibrated and validated using observed discharge data. The coefficient of determination (R^2) and Nash-Sutcliffe Index (*NSI*) were used to evaluate the model performance. The R^2 value is an indicator of strength of relationship between the observed and simulated values. The *NSI* value indicates how well the plot of the observed versus the simulated values fits the 1:1 line. The ranges of *NSI* value is between - ∞ and one. If the R^2 and *NSI* values are less than or very close to zero, the model performance is considered unacceptable or poor. If the values are equal to one, then the model prediction is considered to be perfect.

$$NSI = 1.0 - \left(\frac{\sum_{i=1}^{n} (Q_{obs,i} - Q_{cal,i})^2}{\sum_{i=1}^{n} (Q_{obs,i} - \overline{Q}_{obs})^2} \right) \text{ (Nash-Sutcliffe Index)}$$

where n represents the number of registered discharge data, $Q_{obs,i}$ is the observed discharge at time i, $Q_{cal,i}$ is the simulated discharge.

Result and Discussion

The model was applied to the Hii River basin, where has low densities of stream flow and climatic gages. The simulated and observed statistics for calibration and validation were shown in Table 4. The calibration procedures formulated consist of finding the most appropriate parameters for hydrologic routing model component. In this stage, the best fit was achieved with R^2: 0.65 at sub basin 1, 0.75 at sub basin 2, 0.77 at sub basin 3, and 0.69 at sub basin 4. As well, the best fit was done with *NSI*: 0.64 at sub basin1, 0.74 at sub basin 2, 0.76 at sub basin 3, and 0.67 at sub basin 4 for daily discharge. During the validation period (1986-1992), R^2 varied from 0.58 to 0.74 and *NSI* did from 0.53 to 0.74. As well, from 1997 to 2005, R^2 varied from 0.51 to 0.71 and *NSI* did from 0.38 to 0.68. During whole simulation period, sub basin 3, where is an independent sub basin, performed relatively high reproducibility among the sub basins. As well, sub basins 2 and 4 were also represented satisfactory except latter validation period (1997-2005) of sub basin 4.

Simulated and observed discharge on daily time step is reported in Figure 3. The gray line is observed flow and dotted black line is simulated flow. It was considered that the both results of calibration and validation at each sub basin were represented fluctuations of discharge relatively well, though some peaks were overestimated. Especially on 20th October 2004, the basin was struck by the big typhoon No.23 and observed precipitation was about 150 mm at Yokota rain gage (total about 200 mm for two days), 120 mm at Kakeya rain gage (total 165 mm for two days), and 100 mm at Daito rain gage (total 150 mm for three days). Therefore, it was considered that simulated discharge on/ around that day(s) at all sub basins became big, particularly at sub basin 4. If simulated result on the day were ignored, *NSI* value becomes 0.48 from 0.38.

Table 5. Simulated versus observed statistics for the Hii River calibration and validation.

	Calibration period		Validation period			
	1993-1996		1986-1992		1997-2005	
	R^2	*NSI*	R^2	*NSI*	R^2	*NSI*
Sub 1	0.65	0.64	0.58	0.53	0.51	0.50
Sub 2	0.75	0.74	0.67	0.60	0.64	0.62
Sub 3	0.77	0.76	0.74	0.74	0.71	0.68
Sub 4	0.69	0.67	0.70	0.69	0.59	0.38

Yearly averages for the water balance components are reported together with overall average for the simulated period (Table 5). The overall average of simulated river discharge (1,321 mm) was about 90 % of observed average discharge (1,473mm). The average water balance breaks down accordingly: precipitation 1,818 mm, percolation 921 mm, actual ET 428 mm, potential ET 985 mm, base flow 859 mm, lateral soil flow 400 mm, and surface flow 62 mm. It is considered that base flow accounts for about 65 % and lateral flow does for about 30 % of water yield in the simulation.

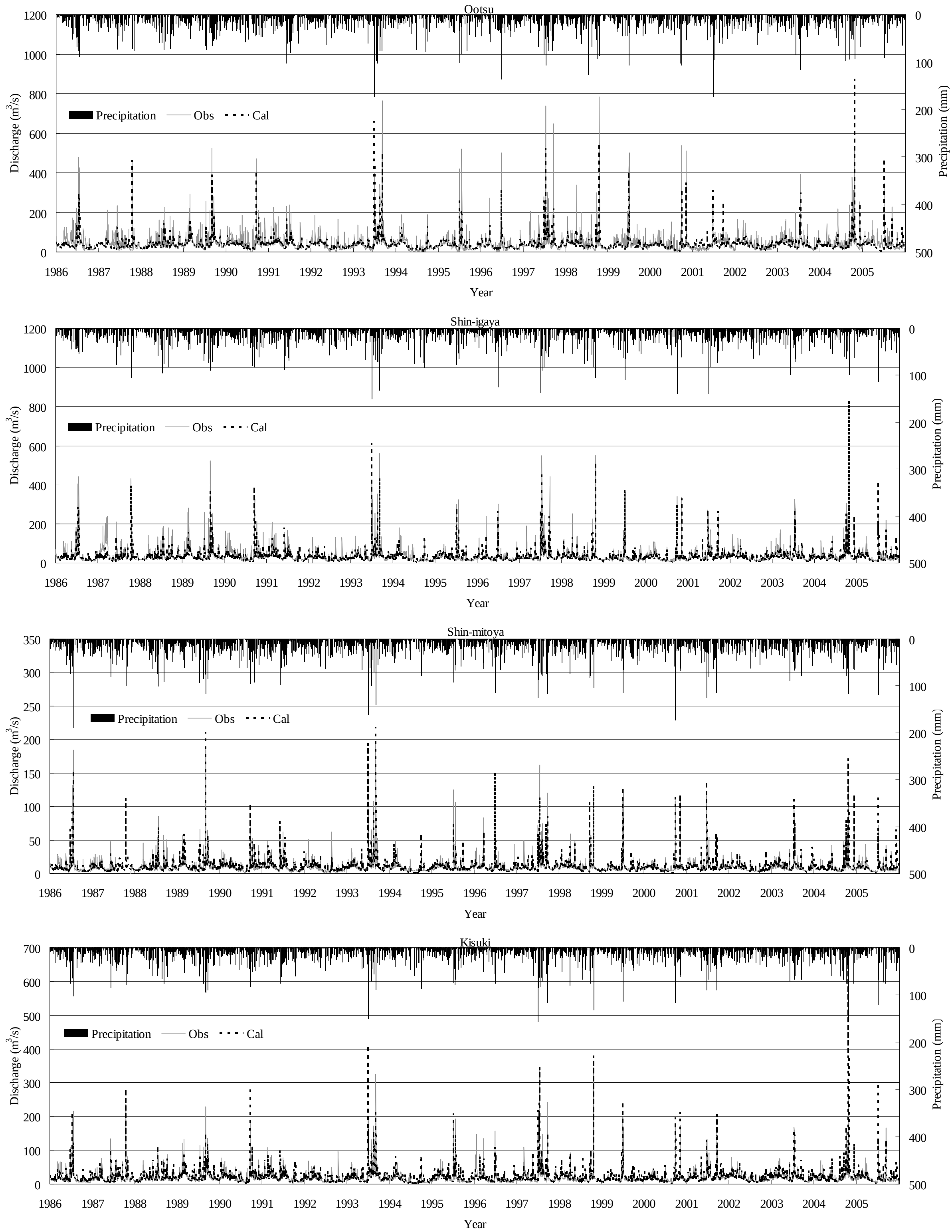


Figure 4. Simulated and observed discharge on a daily time step (calibration: 1993-1996, validation: 1986-1992, and 1997-2005).

Table 6. Yearly averages of simulated water balance.

Year	Precip.	Sur. flow	Lat. flow	Base flow	Perco.	Soil water	Actu. ET	Poten. ET	Water yield
	(mm)	(mm)	(mm)	(mm)	(mm)	(mm)	(mm)	(mm)	(mm)
1986	1649	59	366	760	808	77	412	1112	1185
1987	1819	64	385	828	868	68	511	1152	1277
1988	1768	34	406	858	934	69	392	862	1298
1989	2193	76	498	1096	1169	71	436	889	1670
1990	1916	54	419	890	975	75	467	1053	1363
1991	1843	37	406	899	950	73	443	874	1342
1992	1475	8	323	730	778	71	381	954	1061
1993	2258	148	505	1079	1169	79	426	835	1732
1994	1340	23	263	649	665	78	388	1122	935
1995	1877	58	430	890	979	72	376	932	1378
1996	1607	45	340	738	789	68	474	944	1123
1997	2189	113	490	1043	1112	73	467	1005	1646
1998	1862	80	391	886	911	73	479	908	1357
1999	1707	52	367	757	862	73	425	919	1176
2000	1545	68	320	725	749	69	413	1053	1113
2001	1996	54	449	912	1018	71	473	1004	1415
2002	1621	10	365	792	855	74	381	996	1167
2003	2017	57	457	961	1044	73	465	899	1475
2004	1998	130	434	891	929	72	480	1129	1455
2005	1674	81	390	797	862	72	285	1066	1268
Ave.	**1818**	**62**	**400**	**859**	**921**	**72**	**428**	**985**	**1321**

By using parameter values of the simulation, we tried to estimate a change in maximum and minimum discharge at each sub basin in case that annual total precipitation decreased or increased by 20% as shown in Table 6. The maximum and minimum flows were selected in 20 years of simulation period, and other parameters were not changed at all. It was calculated that the maximum discharge at sub basin 1 became 1,200 m^3/s if total precipitation amount increased 20 %. Oppositely, the maximum discharge became 558 m^3/s if the amount decreased 20%. As well, the minimum discharge at sub basin 1 became 3.76 m^3/s if the amount increased 20% and 2.1 m^3/s if the amount decreased 20%.

Table 7. Change in maximum and minimum discharge in case of decrease / increase of total precipitation amount (-20 %, 0%, and +20 %).

	Maximum flow (m^3/s)			Minimum flow (m^3/s)		
	-20%	0	+20%	-20%	0	+20%
Sub 1	558	876	1200	2.1	2.87	3.76
Sub 2	546	832	1120	1.5	2.07	2.70
Sub 3	133	218	311	0.3	0.55	0.82
Sub 4	440	656	872	0.9	1.32	1.72

Yearly averages in case of 20 % decrease / increase also are shown in Table 7. It is calculated that base flow accounts for about 66 % and lateral flow does for about 31 % of water yield in case of -20% of total precipitation amount. As well, base flow accounts for about 64 % and lateral flow does for about 30 % of water yield in case of +20% of the amount. Though it is natural, the ratio of surface flow increased to about 6.6 % compared with current condition (about 4.7 %) in case of increase of total precipitation.

Table 8. Yearly averages of simulated water balance in case of decrease / increase of total precipitation mount (-20 % and +20 %).

Year	Precip. (mm)	Sur. flow (mm)	Lat. flow (mm)	Base flow (mm)	Perco. (mm)	Soil water (mm)	Actu. ET (mm)	Poten. ET (mm)	Water yield (mm)
-20 %	1454	29	304	654	704	72	413	988	987
+20 %	2181	109	494	1059	1132	73	439	983	1662

In addition, the changes of maximum discharge in case of several return period of probability precipitation were estimated at each sub basin (Figure 4). The probability precipitation was calculated using software build by the Public Works Research Institute. Rainfall duration was set with 24-hour when the rainfall intensity was calculated. Simulation period was one year and daily average precipitation for 21 years from 1985 to 2005 was prepared. According to a ratio of monthly rainfall to yearly rainfall, the highest rainfall ratio was July. Thus, probability precipitation was set on the day recorded the highest amount of rainfall in July. As a result, the maximum discharge was 1,490 m^3/s at sub basin 1, 1,340 m^3/s at sub basin 2, 424 m^3/s at sub basin 3, and 906 m^3/s at sub basin 4 in case of 200-year return period. Actually, rainfall will continue for several days. Thus, maximum discharge will increase in that case.

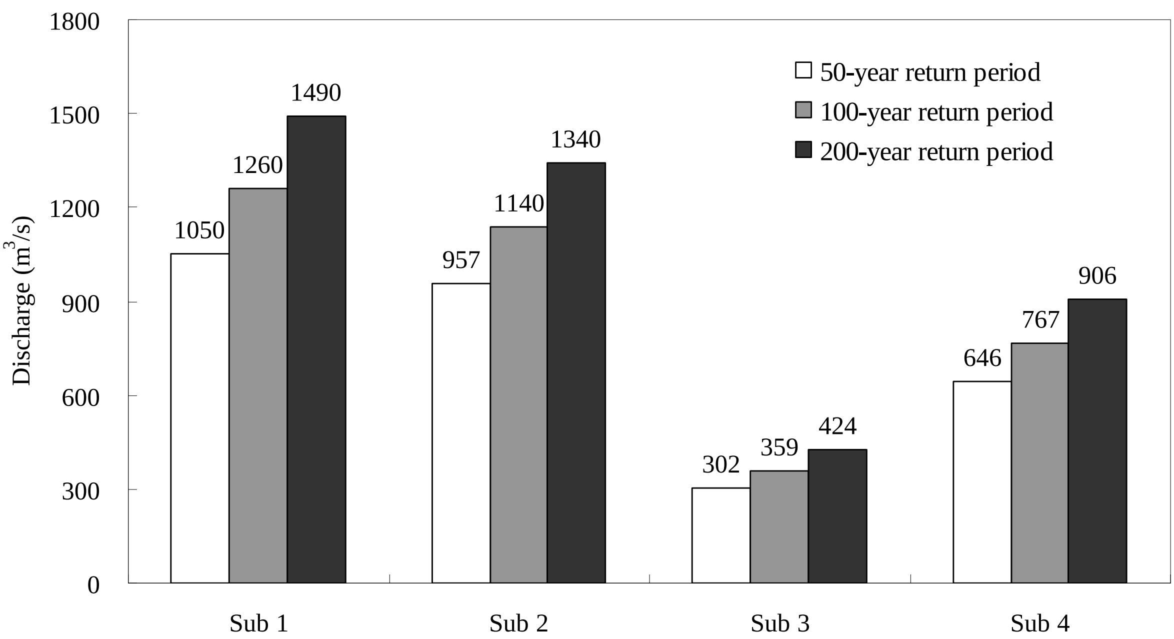


Figure 5. The change of maximum discharge at each sub basin in case of several return period of probability precipitation (rainfall duration was set with 24-hour)

Conclusion

The SWAT performed well in simulating the general trend of river discharges at all sub basins over time for daily time intervals. Thus, this study showed that the SWAT model can be used for Japanese mountainous river basin. However, for more accurate modeling of hydrology and simulating water quality component, a large effort will be needed to improve the quality of available information concerning soils, land use, agricultural activity, and climate of the basin.

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WaterBase: free, open source GIS support for SWAT

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Abstract WaterBase is a project of the United Nations University. Its aim is to advance the practice of Integrated Water Resources Management (IWRM) in developing countries, by providing (a) free, open source **tools** for modeling and decision support (b) a collection of IWRM **resources**: web sites, tools, literature, training material, etc. and (c) a community of **partners** who can provide advice, support, contribute to tools and resources. A first step in the project is a too to provide GIS support and a setup interface for SWAT. This paper describes the design of this tool, called MWSWAT.

Keywords: integrated water resources management; hydrological modelling; decision support; SWAT; MapWindow

1 Introduction

The WaterBase project (http://www.waterbase.org) of the United Nations University is aimed in particular (though not exclusively) at developing countries. Predictive modelling and decision support for water management in developing countries are plagued with a number of related problems: lack of money, lack of expertise, inadequate training capacity, dependence on experts from other countries. At the same time water resources are under increasing pressure, and aquatic ecosystems are being damaged by people who lack the resources to explore the consequences of decisions before they are made. WaterBase aims to improve this situation by providing (a) tools for decision support, (b) resources such as web sites, documentation, training material, and case studies, and (c) a community of partners who can advise and support other partners, and who can contribute to the tools and resources.

SWAT has a substantial reputation as a modelling tool, and has been used in many developing countries as well as in its home country, the US. Like other modeling tools, it requires a lot of data about terrain, landuse, soil, and climate. There are two essential components needed to set up SWAT models: (a) a GIS system to support the storage and display of the relevant maps, and to perform the terrain analysis needed to delineate watersheds, to identify the stream reaches and the associated subbasins, etc., and (b) a component that can generate all the files needed by SWAT, partly from the input maps and analyses, and partly by manual editing.

There is a substantial price tag on the current commercial GIS system that is currently used by SWAT. The WaterBase project decided to identify a suitable free, open source GIS system, and then to produce the additional component that would support the generation of SWAT input data. The use of open source is important: it gives users confidence that tools will not suddenly disappear with their original writers, or one day become something you have to pay for, and also gives users the possibility of adapting or extending them. This possibility ranges from the localization of the interface to the local language to the adding of significant functionality. The use of open source tools also implies the use of the corresponding open standards, such as those supported by the Open Geospatial Consortium [OGC].

There are a number of open source GIS systems available [OS GIS]. WaterBase eventually chose MapWindow (http://www.mapwindow.com) for three reasons. First (and critical when choosing any open source project) it was under active development. Second, unlike most open source projects, it is native to Microsoft Windows, which is the operating system we expect most of our users to be currently using and accustomed to. Third, it had just been chosen by the Environmental Protection Agency in the US as the basis for version 4 of BASINS [EPA], which gave us confidence in its future support. There were also technical issues to be considered, such as whether MapWindow, could support watershed delineation, and how easy it was to write an extension for it, but technical problems can often be overcome, while the basic issue of whether your chosen GIS system will still be available and supported in 5 years time is the most important issue.

As it happens, MapWindow does have watershed delineation, using David Tarboton's Taudem software [Tarboton 2001]. In fact Taudem's use of the Dinf approach to slope directions (instead of the normal D8) promised better watershed delineation than found in the current ArcSWAT interface. MapWindow is also intended to be extensible through the use of a "plugin" architecture, so it was in fact technically suitable.

So an interface for setting up SWAT was created based on MapWindow, and called MWSWAT.

The rest of this paper describes some of the details of MWSWAT, before considering what else needs to be done to provide decision support for IWRM.

2 Design Philosophy

Setting up a SWAT run is complicated. Generating a thousand input files is not unusual, and so there are a vast number of parameters to consider. The user can therefore easily get lost in the process, and we need to keep a balance between simplicity of the interface and access to everything which the user might need to see and perhaps change. The first priority is therefore to try to create a simple model of the process which the users can have in their minds. We based the interface around three basic steps:

1. Watershed delineation.
2. HRU definition
3. SWAT setup and run

This overall design is clear from the main MWSWAT form. **Error! Reference source not found.** shows the form when the first two steps are completed and the third can be started

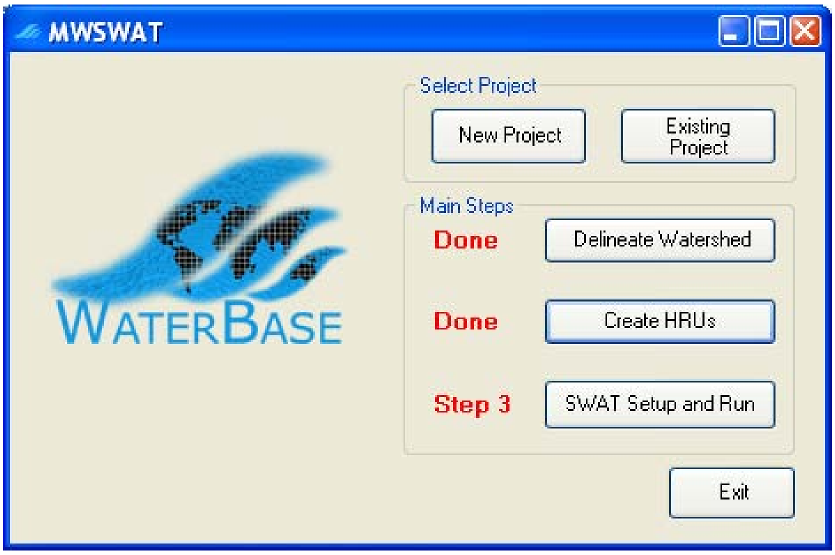

Figure 6: Main MWSWAT Form

2.1 Watershed Delineation

Watershed delineation uses the same form as BASINS: see Figure 2. First the digital elevation map (DEM) is chosen, and options to burn in existing streams, and to use a mask for the watershed, may be selected. Then the threshold (minimum area to be designated as a stream) is chosen. Finally outlets and inlets are selected, either from an existing point shapefile, or by creating one interactively.

Watershed delineation uses the Taudem code written by David Tarboton. MWSWAT by default uses the Dinf approach to slope directions, rather than the more common D8 method, which is expected to produce more accurate delineation.

A delineated watershed is illustrated in Figure 8.

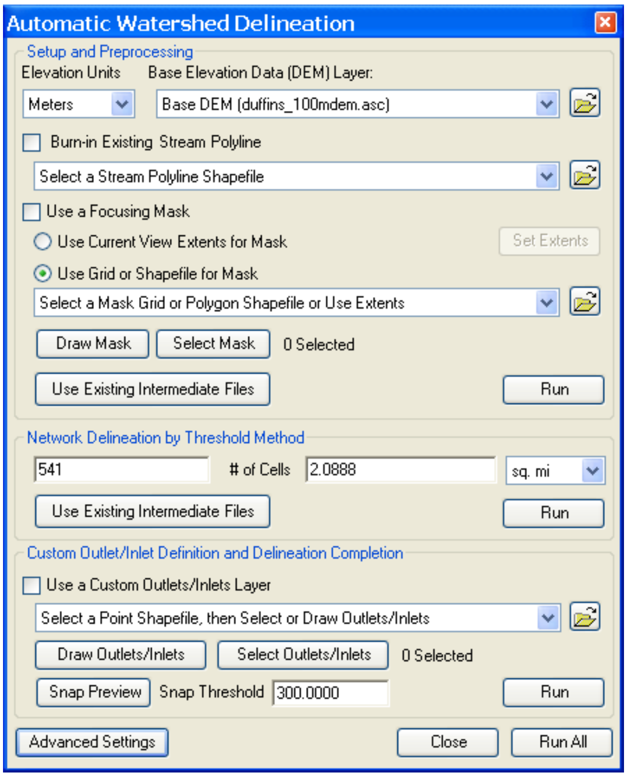

Figure 7 Watershed Delineation

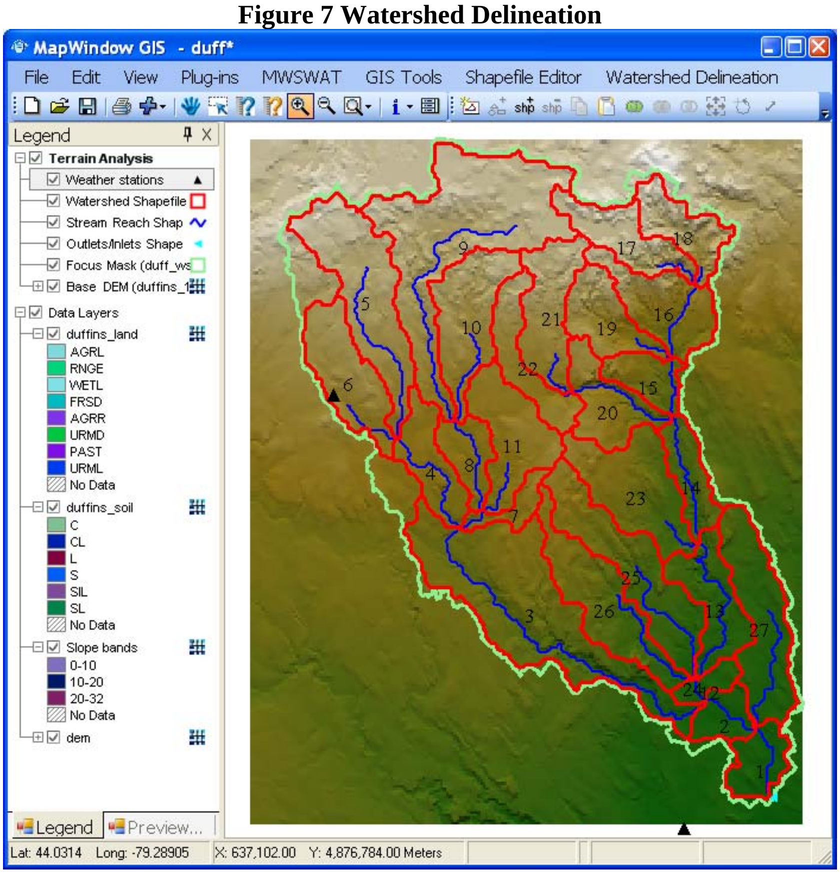

Figure 8: Delineated Watershed

2.2 HRU Creation

SWAT uses *Hydrological Response Units* (HRUs) as the basis for its modeling. HRUs may be formed 1 per subbasin (where a subbasin is the area that drains into a reach of the stream network), or as a division of a subbasin based on a particular combination of landuse, soil, and slope range. The *Create HRUs* form allows users to first select landuse and soil maps, together with database tables which relate the categories used in these maps to SWAT landuse and soil categories. Then users can select intermediate slope percentages so as to form bands of slopes. At this point the maps are read. Then the user can choose singe HRUs (i.e. one per subbasin) or multiple HRUs. In the second case the user removes small HRUs, either by a selecting a minimum area, or by selecting minimal percentages for landuse, soil and slope. Users may optionally also select subbasins at whose exit points reservoirs are situated, may choose to subdivide landuses into others, and may choose to exempt some landuses from the thresholds. See Figure 9.

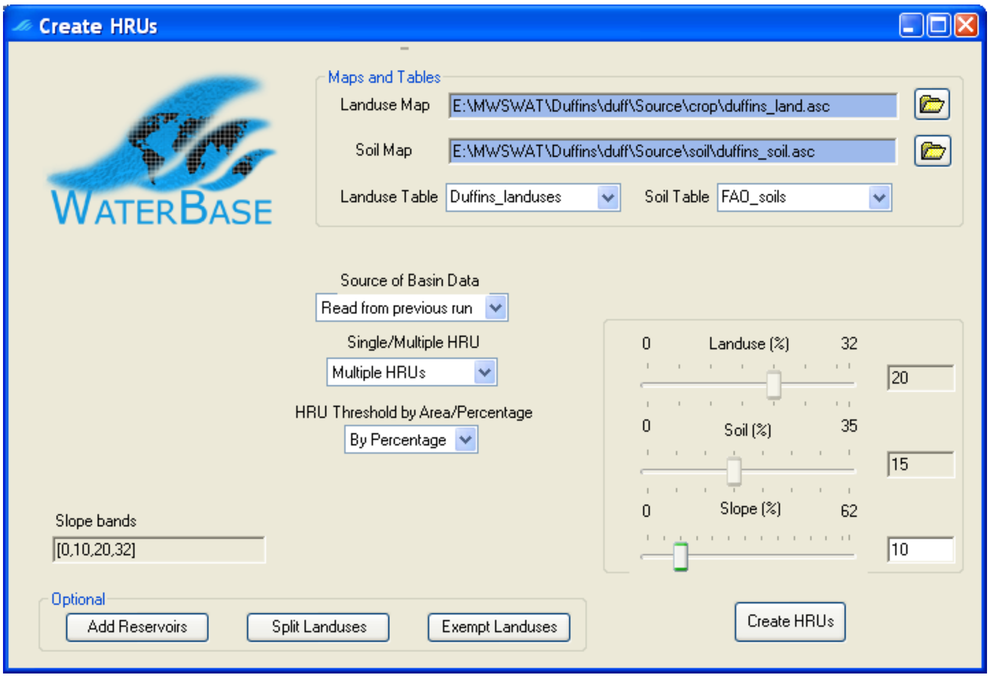

Figure 9: Create HRUs

2.3 SWAT Setup and Run

The final form allows the user to select weather sources (currently weather stations, plus precipitation and temperature gauges), to choose the period of simulation, and make a number of other choices: see **Error! Reference source not found.**.

Users can choose to make detailed edits to the input files using the SWAT Editor, can run SWAT itself, and can save the output from the latest SWAT run.

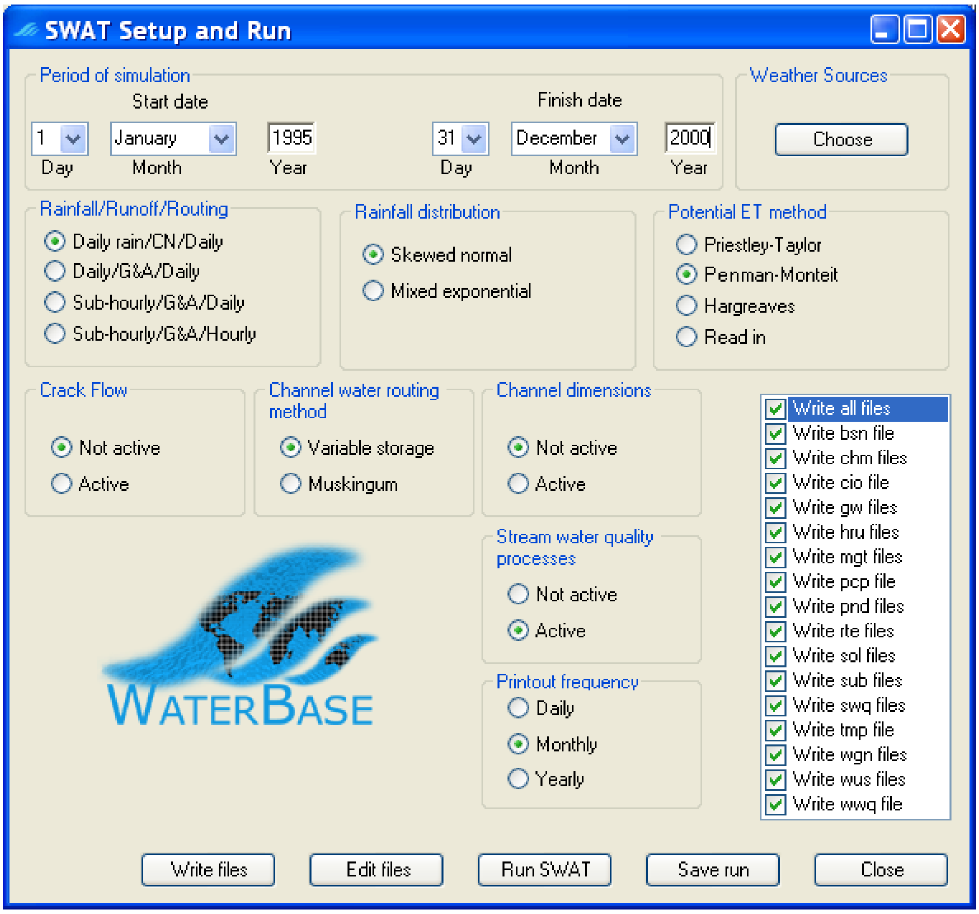

Figure 10: SWAT Setup and Run

3 Data sources

There is a considerable amount of data available on the internet, and MWSWAT is designed to take advantage of that. In particular it will be delivered along with global data:

1. DEM maps from the SRTM project [SRTM]
2. Landuse maps from the Global Land Cover Facility [Hansen 1998]
3. Soil maps from the FAO [FAO/UNESCO 2003]
4. Precipitation and temperature data [NCDC]

The increasing availability of such data opens a number of possibilities for its exploitation beyond water resource management. Of course, users should not be restricted to such data, because where local data exists it will generally be finer grained and more accurate. But at the same time they should not be prevented from doing some simulations even when there is no local data.

4 Future Work

In this section we consider future work in two categories: technical work concerning MWSWAT, and future tasks for WaterBase.

4.1 Technical Work

MWSWAT is almost complete at the time of writing, and will very soon be released. The next immediate technical task is to provide some decision support capability, especially some graphical support for viewing the SWAT outputs. This will certainly include capability for drawing graphs or histograms, especially for comparing outputs from runs with different input parameters, and also for showing, for example, watersheds coloured according to user-chosen characteristics such as sediment output.

Another technical aim is to support other kinds of models, such as event-based models which can analyse the effects of storm events.

4.2 WaterBase Objectives

MWSWAT is the first of, we hope, many tools to support IWRM. The next objective is to form a community of partners who are interested an using and/or contributing tools and other resources to the project. Partner organizations may be government departments, universities and research institutes, or even private companies. In particular partners can provide requirements for new tools and extensions or changes to existing ones. The existence of an active collection of users and developers will also be a critical factor in finding donor organizations to support the project financially.

5 Acknowledgements

The WaterBase project is a project of the United Nations University, in particular the International Institute for Software Technology (http://www.iist.unu.edu) and the International Network for Water, Environment and Health (http://www.inweh.unu.edu). We have received collaborative support from the Daniel Ames and the MapWindow team at Idaho State University, from David Tarboton at Utah State University, from Dave Swayne and his group at the University of Guelph, from Karim Abbaspour of Eawag, and, not least, from Raghavan Srinivasan and the SWAT team at Texas A&M University.

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THE BASHYT DSS: A WEB BASED DECISION SUPPORT SYSTEM FOR WATER RESOURCES MANAGEMENT

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Abstract

Prediction, prevention, or minimization of point and diffuse pollution is an important topic for scientific research. Environmental sciences are evolving from a simple, local-scale approach toward complex multilayered, spatially explicit regional ones. The advances in computer simulation and high performance computing in recent years have highly extended the possibilities in this field, and have changed the ways in which land management systems operate. The Basin Scale HYdrological Tool – BASHYT - is a Collaborative Working Environment (CWE) on the web that relies on hydrological models and web-GIS technologies to support decision makers, through a user-friendly Web interface, in the field of sustainable water resources management. The portal, for the general user, exposes hydrological applications based on the SWAT [*Neitsch et al, 2005*] and QUAL 2K [*Brown, L.C. and Barnwell, 1987*] models to quantify the impact of point/non point pollution. Within the experimental CWE environment, modules have been developed to run real-time applications based on numerical solvers, run pre- and post-processing codes, query and map results through the web browser.

Free software and in-house technologies are combined to transparently and automatically deploy the applications on the portal. Our objective is to build a development platform, made of a set of loosely coupled services, that promotes joint initiatives and encourages cooperation among interdisciplinary teams operating in the environmental sciences. To illustrate the potentiality of our decision support system (DSS), we present its application to a Sardinian case history where a complex watershed is threatened by point pollution and intensive agriculture.

1. Introduction

A sustainable environmental policy requires knowledge and easy access to up-to-date environmental information for politicians, administrators and citizens.

Large-scale networking can contribute to moving from a local scale to a global one, increasing exchanges and collaborations among scientific communities and end users sharing the same goals. Recent advances in distributed heterogeneous knowledge networks and the experience gained during many collaborative European projects led us to conclude that water and environmental management disciplines, largely based on GIS applications and hydrological numerical modeling, might draw a huge benefit from the use of web-based collaborative technologies. In this framework, research can also improve the exchange of good practices with regard to dissemination, training and transfer of knowledge, in agreement with the Environment Action programs.

Public and private environmental agencies often communicate little, even when solutions require an integrated interdisciplinary approach; this situation leads to higher costs and inefficiency in the decision process. A web-based environment constitutes the ideal terrain for sharing knowledge (data, models, procedures, physical and computing resources) by means of new tools. Client/server applications are the paradigm introduced in the Internet cyberspace, using ever more web services to extract meaningful information and grant services to the community.

This paper describes our efforts in building a web-based hydrological modeling system and implementation challenges. This overview illustrates the relevant technical platforms for state-of-the-art computing resources that can inform better management decisions. In this paper, we will explore the use of computer simulation models and related information technologies that aid in the assessment, analysis, and management of complex watersheds.

2. The BASHYT DSS

2.1 Description of the decision support system

The BASHYT DSS (http://era.hopto.org/bashyt/) is a software on the web, based on river basin, or watershed, scale model, that relies on a GIS to support decision makers, through a user-friendly Web interface, in the field of sustainable water resources management. The system, for the general user, exposes hydrological applications based on complex “physically based” models that make use of large volumes of distributed data. Free software and in-house technologies are combined to transparently and automatically deploy the applications on the web portal.

The DSS is based on the Driving forces-Pressures-State-Impact-Responses (DPSIR) paradigm, introduced by the European Environmental Agency (extension of the PSR model developed by the OECD, 1993) and also adopted in the EU Water Framework Directive (WFD). Such approach is useful to demonstrate the interconnectedness and estimate the effectiveness of the actions aimed (responses) at solving the problem.

Driving forces stand for processes underlying to environmental degradation such as land use or demographic development. Pressure indicators measure the level of environmental impairment (e.g. total quantity of phosphorous in chemical or biological fertilizers applied per hectare of agricultural land). State indicators are the condition of the environment (e.g. average concentration of phosphorous in surface waters). Impacts represent the ultimate effect of State indicators, or the damage caused (e.g. eutrophication of surface water or water becoming unsuitable for drinking). Responses are the policies and measures to solve the problems. They are represented by a set of alternative options to choose from (e.g. alternative designs for a water treatment plant).

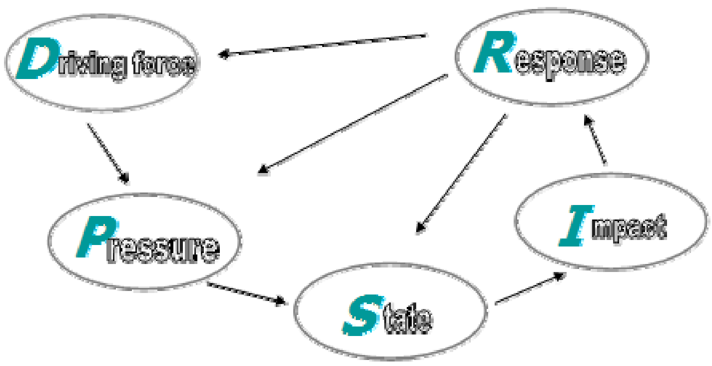

Figure 1. The DPSIR Model, adopted by the European Environmental Agency, is one of the frameworks based on the concept of causality chains for data synthesis, which links environmental information using indicators of different categories

In this context we reassess the current definitions of Indicators in the light of the WFD, proposing the design of modular procedures and computational practices to determine the most significant State/Impact indicators. To this end the DSS integrates QUAL2K and SWAT water quality models for the generation of quality data to assess differing DPSI scenarios, with the final aim to produce an integrated approach. The system does not require additional software or plugins, but works directly on a web browser. This improves the potential for its utilization by water management administrations, being assessable directly on the Internet/Intranet through a normal browser (Internet Explorer, Mozilla Firefox, Opera, ecc.). GIS technologies, hydrological models and/or meta-models are provided, and in the current version (BASHYT 1.0) of the software a full coupling dynamically links the conceptual framework to the hydrologic models and analysis tool.

The system incorporates a variety of physical, chemical, and biological processes that control the transport and transformation of pollutants and the state of water quality variables. Water quality models are driven by hydrodynamics, point and non-point source loadings, and key environmental forcing functions, such as temperature, precipitation, solar radiation, wind speed, and light attenuation coefficients. SWAT is a river basin, or watershed, scale model developed to predict the impact of land management practices on water, sediment, and agricultural chemical yields in large, complex watersheds with varying soils, land use, and management conditions over long periods of time. The model is computationally efficient and uses readily available inputs, enabling users to study long-term impacts. QUAL 2K is a river and stream water quality model which simulate the fate of pollutants and the state of selected water quality variables in water bodies to study short-term impacts.

The user interfaces have been optimized to be used also by non technicians. The environmental applications are exposed through a “user friendly” interface, which supports a coherent management of Drivers, Pressure, State, and Impact indicators as distributed (in space and time) catchment’s variables. This procedure encourages the user to increase the awareness of the effects of subjective judgments or misjudgments on the final result. The DSS enables to:

- store, manage, query the data collections;
- visualize data through the web GIS;
- interconnect directly to the AVSWAT [*Di luzio et al., 2002*] databases. BASHYT can be used in tandem with other GIS interfaces (e.g. ARCGIS SWAT [*Francisco et al, 2006]*, AV SWATX);
- run real time applications based on the SWAT and QUAL 2K models;
- create reports (graphs, maps, etc.) through automatic standardized procedures;
- design remediation strategies and evaluate their effectiveness using a standardized framework (DPSIR model).

2.2 The methodology

The application of the BASHYT decision support tool requires two phases of analysis. In the "Conceptual Phase", the user investigates and identifies causal links between current human activities (D), the pressures they exert (P), and the state (S) that produces Inpact (I) on the environment. This phase generates a formal description of activities and issues relevant to catchment’s management under current climate and management conditions and makes relationships between these phenomena explicit in the form of "DPSI chains". It is in this first phase that hydrological modeling and measured data can be used to explore the problem in different ways.
The identification of alternative options for courses of action (Responses), such as specific water management projects, follows in the "Scenarios Phase".

Lastly, in the "Scenarios Phase", the user elaborates a concise presentation of an environmental context to analyze. In this section the problem at hand is assessed and Responses are designed to solve the problems (e.g. new pressures on the environment).

Models are used for the evaluation of the performance of the responses, on the basis of the chosen indicators.

The Conceptual phase: users will browse and design the DPSI sections that are considered responsible for the current state of the environment. Within this area of the DSS the following default sections are found:

1. Driving forces and Pressures: this shows pollution sources, which are grouped in two main categories: point and diffuse;
2. State of the environment: the water balance and water quality state is assessed. IN the first user will browse average precipitation, water yield, groundwater recharge, potential and real evapotranspiration values on different time scales in the water balance section. In the latter, with regard to the European Water Directives, measured and/or simulated water quality concentrations under current land use (point and diffuse pollution) and climate conditions are shown. Results will be assessed spatially on the GIS and temporally through graphs.
3. The impacts are obtained assessing the states of the environment through automatic procedures.

The Scenarios phase: in the scenarios section the water manager can design management strategies to solve the environmental problem (responses) and evaluate their effects through the DPSI chains. Formally the scenarios section, formally designed as the Conceptual phase, although simulations are run under synthetic conditions designed by the user to evaluate a variety of environmental dynamics (e.g. effect of new climate conditions, new land management practices, etc.).

2.3 The Architecture and the Interoperability layer

BASHYT is an Integrated system made of datasets, visualization tools, numerical applications and analysis tools. It is made of different layers, where the web portal is its front end. The interoperability layer is called to transparently share heterogeneous data sources (distribuited database and filesystem) and computing resources, while the webservice layer commands the data management tools and the numerical applications. The webservice layer is expected to supply an abstraction layer between the archives/computing resources and the actual computing tasks performed for running the models, the geo-processing and the data dissemination mechanism. Computing resources can be used and jobs submitted from the Web portal. A Linux environment is used to run the computing and the geo-processing phases required: the SWAT and Qual 2K hydrologic models and the environmental applications process the dataset, produce new derived information , GIS maps, graphs, and reports. The data flow, the data storage, the way the applications work have been designed in a non-conventional fashion to hide the user the complexity of the infrastructure. This system is more then a mere sum of modules: it is a software to consume and expose Web services for data mapping, querying and sharing, processing and distributing, with a high degree of freedom (new contents, analysis and applications can be developed by each institution using the CWE framework).

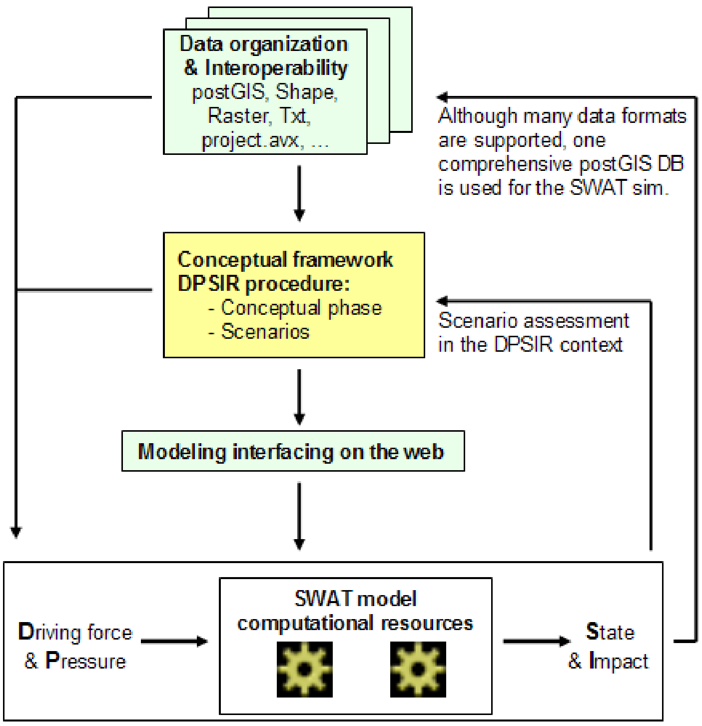

Figure 2. Collection of procedures required for the BASHYT DSS.

For security and organizational purposes a stand alone web-based interface grant users access to the contents (data, models, geo-tools). The portal provides institutions with complete control on the selection of data they wish to process and want to expose in the public sections. Contents are assessed on the basis of users and roles (e.g. “public”, “office”, “developer”, “administrator”): the “public” user can access an open environment on the internet where contents and services are freely given to the public (no authorization is required, contents can be only viewed); the “office” user freely uses the DSS (write texts, run the models, design remediation strategies, etc.) and decides which contents are to be fed in the public sections; the development user can modify the analysis tools and add new potentialities to the system; the administrator control the roles-users and in general is the manager of the system. In this way the software can be easily customized to respond to the necessity of different agencies. Security is guaranteed using the SSL protocols.

This stratification (interoperability layer, webservice layer, security layer) was necessary to develop an effective, authoritative, interoperable, and scaleable model oriented system.

2.4 Visualization and geo-processing tools

On the server side the GIS rendering has been optimized using the MapServer technology. The integration in the BASHYT framework has been accomplished, exploiting the scripting languages capabilities to access the MapServer CGI and OGC (WMS, WFS) interfaces. Mapserver works as a "map engine" for our integrated system, providing spatial context where needed. On the client side a Javascript cross-browser interface (web 2.0) has been developed and customized to allow user dynamically display and browse the geographical information layers.

The GRASS (Geographic Resources Analysis Support System) technology has been efficiently deployed within our system for the processing and analysis of the geo-databases. The BASHYT inherits all the Geographic Information System (GIS) capabilities granted by this technology. Through it, our client/server web system aspires to become desktop like for the geospatial data management and analysis, image processing, graphics/maps, spatial modeling, and visualization productions.

BASHYT supports a multitude of raster and vector data formats (ESRI shapfiles, PostGIS, Oracle Spatial, MySQL, Open Geospatial Consortium (OGC) web specifications WMS and WFS, etc.) using GDAL (Geospatial Data Abstraction Library) and OGR (Simple Feature Library).

Thus, it is proposed that a single and comprehensive postGIS geodatabase be used as the repository of the SWAT simulations.

The “user friendly” CWE environment, we developed, allows to design new applications and easily assess, customize and exploit the web services directly on the web browser. Through the web interface using secure connections, data objects and mapfiles are digested by the CWE environment to expose web services for data mapping, querying and sharing, processing and distributing. The CWE framework is an Open development environment for constructing spatially enabled Internet-webgis applications.

3. The case study

To illustrate the potentiality of the BASHYT DSS, we present its application to the San Sperate case history, where a complex watershed is threatened by point source discharges and diffuse pollution due to intensive agriculture.

High levels of contamination (mostly P and CBOD), due to civil, industrial and agro-zootechnical compartments are found in the two monitoring gages (PMP 20801 and PMP 20802) located on the river within the basin [*Contu et al. 2005*].

The study site is the 520 km^2 "Flumini Mannu of S. Sperate" basin (south part of Sardinia, Italy), which is part of the larger "Flumini Mannu of Cagliari" basin. The Flumini Mannu of S. Sperate basin is delimited by the Sarcidano plateau at north, the Sarrabus relief at east, the last layer of Iglesiente massif at West. The watershed is characterized by a significant variation in terms of altitude (from 13 to 972 m a.s.l.).

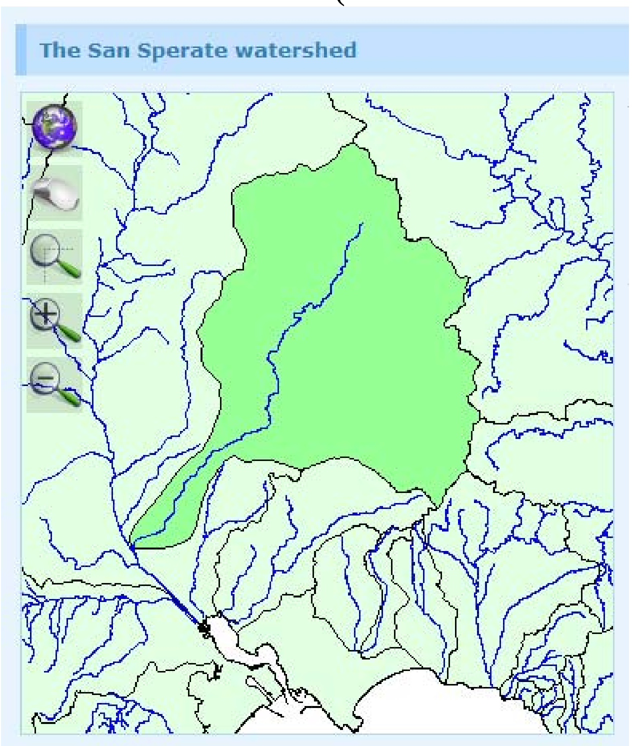

Figure 3. The S. Sperate basin viewed on the BASHYT interface.

The main river (the S. Sperate) is a tributary of Flumini Mannu of Cagliari river which discharges its waters into the Santa Gilla humid area near the gulf of Cagliari. The S. Sperate is characteristically fast flowing, with a relatively high water volume in winter, reduced to a trickle in summer. S. Gilla is a very important humid area in Sardinia and is among the largest wetlands in Europe. Such environments often provide favorable circumstances to facilitate physical and biogeochemical functions that are critical to the maintenance of healthy environmental conditions in waterways.

3.1 Current state and Scenarios: the DPSI chains

Under current land use (point and nonpoint source loadings) and climate condition the BASHYT DSS produces reports on the state of the environment (fig. 4,5,6,7).

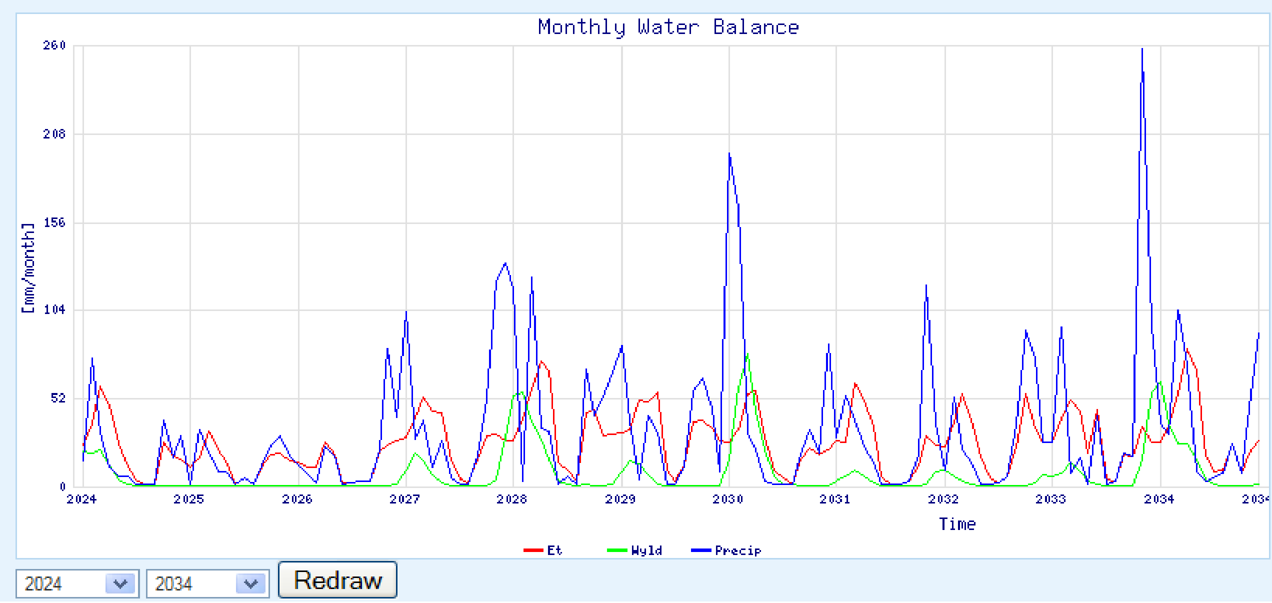

Figure 4. The water balance can be assessed on a yearly basis. Automatic procedures read the swat results stored in the PostGIS DB and produce reports in the form of graphs.

Providing the water quality model with the correct inputs requires a pre-processing stage which, starting with the Drivers, defines the resulting Pressures in terms of pollution load to be fed in the model. Different alternatives can be used:

1. The SWAT model is run locally using the AVSWAT-ARCGIS interface and the BASHYT is used for dissemination purposes and to create standardized reports on the environment. In this case the BASHYT connects directly with the local datasets.
2. The model is run directly from the web interface. In this case the input management is processed directly from the web portal (for the S. Sperate case, the model is run directly from the web).

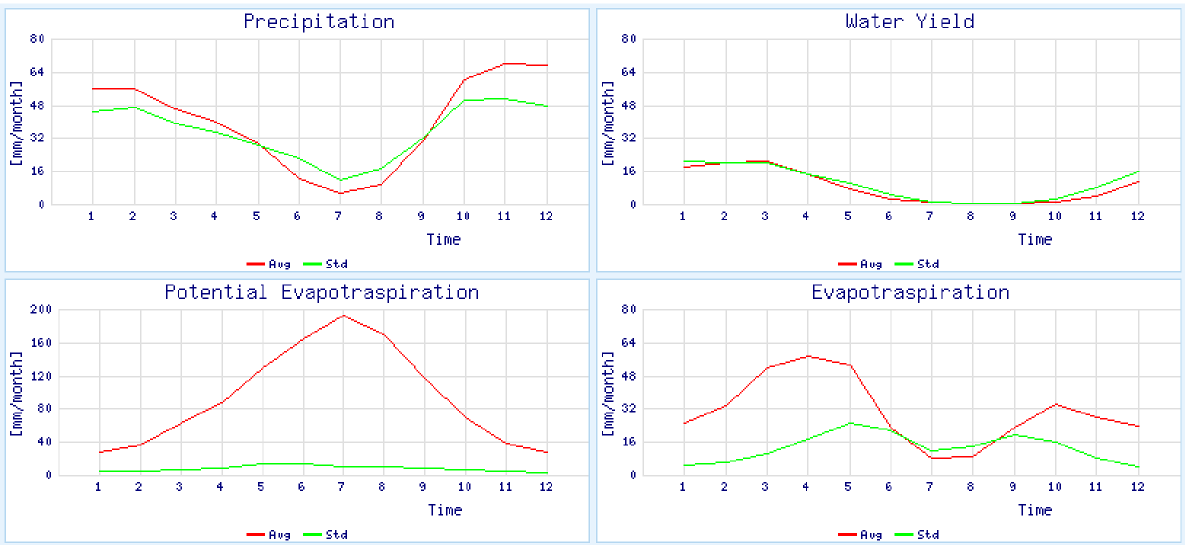

Figure 5. The water balance can be assessed on a monthly basis. Automatic procedures read the swat results stored in the PostGIS DB and produce reports in the form of graphs.

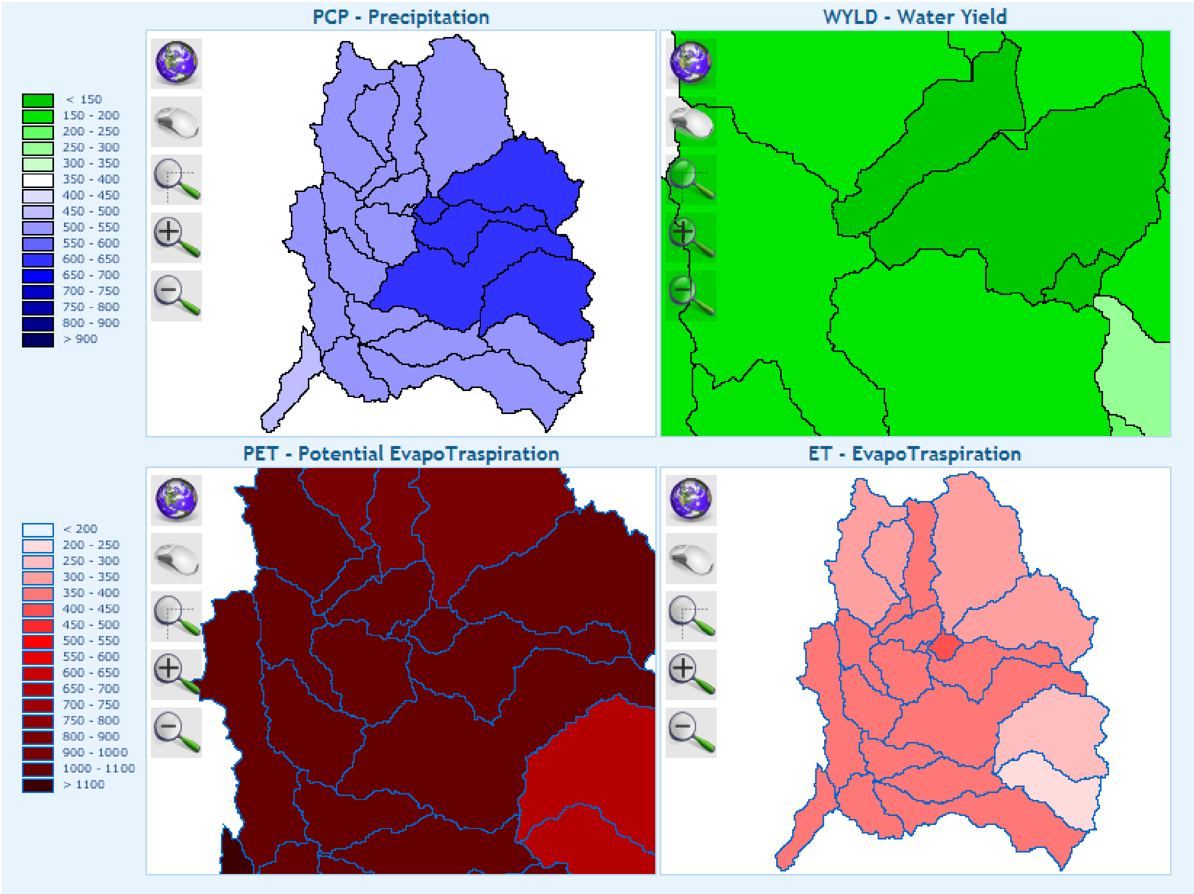

Figure 6. The water balance is mapped on the WEB GIS. Automatic procedures read the swat results and produce reports in the form of maps.

Natural resources and their management are intrinsically complex due to the dynamic balance and interactions among coexisting biotic agents and their abiotic environments within the ecosystem. The BASHYT DSS provided the analysis tools to explore the S. Sperate complex ecosystem, forming the basis for an integrated, system approach to management planning and implementation guidance. The CWE software framework improves model(s) usability to aid in making management decisions and improves watershed-scale (multi-scale) modeling to address more realistically the fate of multiple pollutants in multiple environmental media.

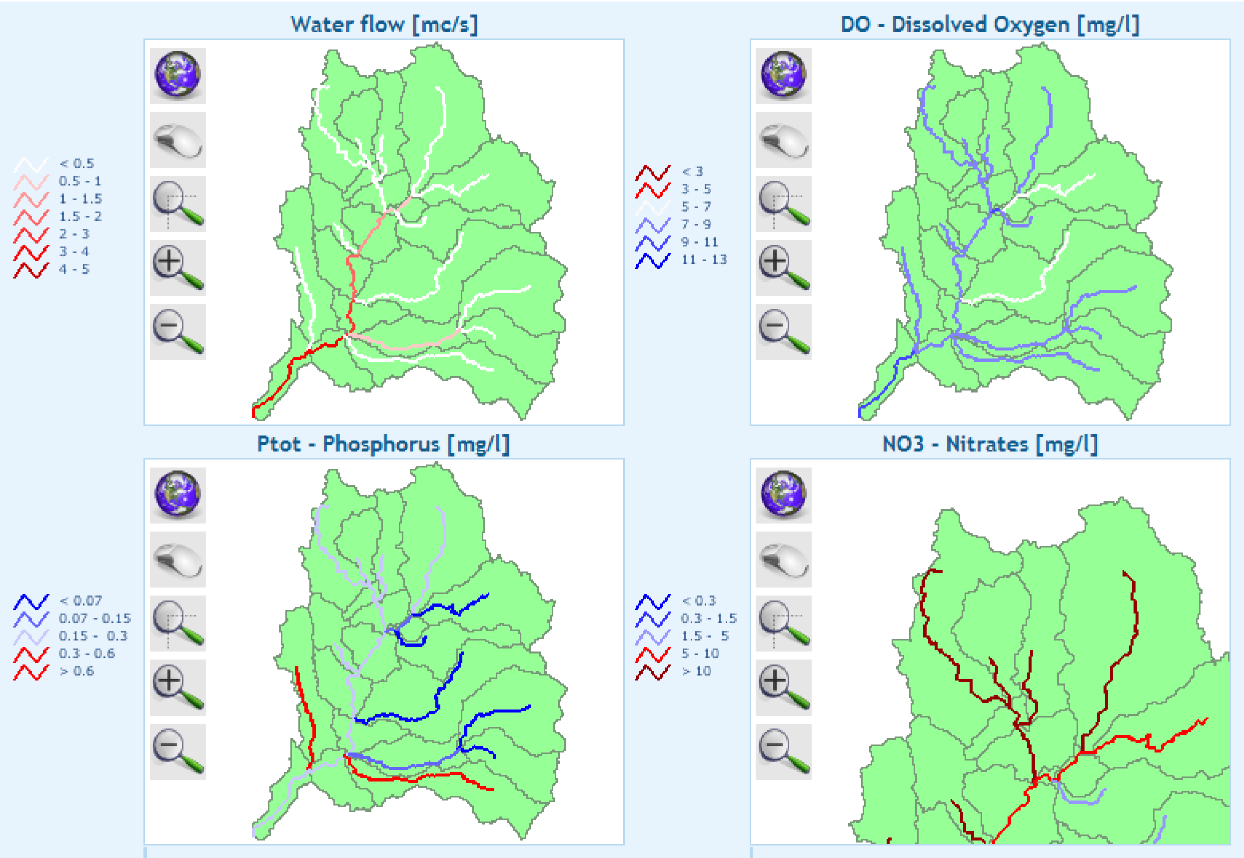

Figure 7. Water concentrations are mapped on the WEB GIS. Automatic procedures read the swat results and produce reports on the environment.

The DSS has been efficiently employed to identify the S. Sperate sub-watersheds, which are the major contributors to nutrient losses and prioritization of the critical sub-watershed in order to develop a multi-year management plan. This can be essential to reduce the nutrient impact from point and nonpoint source pollution to downstream water bodies.

4. Conclusions and future works

The BASHYT provides a framework for analyzing management scenarios based on valuable data and computing resources over the web. The system is based on a client/server architecture and can be used within the Internet/Intranet cyberspace, offering to the community services to extract meaningful information about the environment. The DPSIR framework can be used as a base for environmental management allowing the linkage between pressures and state-impact indicators, making it possible to integrate the conservation functions with sustainable developments. The application of this causality model, the use of GIS capabilities and of hydrological models have the advantage of allowing the spatial visualization and complex analysis and better integration/exploitation of the different indicators on which water and territorial management is based.

In general, the web interoperability is of paramount importance to control the redundancy of replicated datasets, and it allows the user to retrieve updated certified information, avoiding the latency due to administrative and technological barriers. As a matter of fact, environmental analysis will be enabled to benefit from real-time data processing, making territorial management and planning more efficient. The BASHYT integrated system can deeply contribute to the development and the exchange of information relative to the environment, offering the administration standardized procedures to manage water resources. The development of a CWE framework offers an infrastructure for optimizing data-sharing and solving application development problems in a multi-user environment.

The current version of the DSS can be easily applied to many common situations in the majority of real applications, i.e. watershed scale problems. The DSS was efficiently applied to a study site of south Sardinia, Italy.

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Modeling cost and impact of emission reduction measures

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Abstract

The assessment of cost and impact of nitrogen abatement measures is an essential element of a river basin management plan. Yet, modeling and consequently comparison of impact of measures is complex as measures can target different pollutant pathways, different water quality processes, at different scales and in different economic sectors. A broad variety of potential abatement techniques furthermore results in a wide range of costs. As a result, an evaluation of the cost-effectiveness of abatement measures is lacking in many pollution abatement plans.

In this paper, a generic framework is presented which allows to determine the required emission reduction in order to reach a concentration target and consequently the most cost-effective set of abatement measures to reach that target. The methodology is applied on the Kleine Nete river basin in Belgium with a focus on the abatement of point and non-point sources of nitrogen. The framework is based on the hydrological water quality model SWAT and the Environmental Costing Model. SWAT is used to assess the impact of emission reduction~~s~~ measures on the in-stream concentration of nitrate, organic nitrogen and total nitrogen. The Environmental Costing Model provides costs and nitrogen removal efficiencies of such measures. Despite the non-linear character of the modeled processes, a linear relationship is found between the in-stream concentration and the emission reduction. This paper aims to link the available scientific knowledge and tools to management needs that are required for the implementation of the European Union Water Framework Directive.

KEYWORDS: river basin management, nitrogen abatement, impact assessment, cost-effectiveness

Introduction

The main instrument of the European Union Water Framework Directive (WFD) to reach its target good water status is the river basin management plan (RBMP). A RBMP includes a description of the characteristics and status of the basin, the vision on river basin management and the translation of this vision into a concrete set of actions, including pollution abatement measures. According to the WFD, the selected measures should be the most cost-effective combination that will achieve the chemical and ecological good water quality (European Communities, 2003). The strict time frame as well as the ambitious environmental goals hinder the use of classical methods such as ‘trial and error’ and ‘worst polluter first’. The need therefore exists to assess cost and impact of pollution abatement measures from the planning phase.

Yet, values of cost-effectiveness of abatement measures are scarce. Scientific arguments are that an altered state cannot be attributed clearly to a source especially in the case of diffuse sources or ecological quality elements. Likewise, the comparison of the impact of measures is complex as these can target different pollutant pathways, different water quality processes and at different scales. Moreover, it is to be assessed whether the total required efforts can achieve the environmental objectives timely without disproportionate cost. To support the decision-making process in Flanders, the Environmental Costing Model is in development (Vito and Resource Analysis, 2006). The Environmental Costing Model is an economic optimization tool to select the most cost-efficient abatement measures to obtain a given surface water quality target. Currently, a database of measures including cost and abatement efficiency is available for the abatement of nitrogen, phosphorus and COD. However, the model is load based and a detailed simulation of the in-stream processes is missing. Hence, to assess the impact of abatement measures on in-stream water quality a linkage with surface water quality models such as SWAT is required.

This paper aims to link the available scientific knowledge and tools to management needs that are required for the WFD. Hence, a framework is presented that allows water managers to gain insight in the relationships between the objectives pursued, the measures that might be taken and the impacts and cost-effectiveness of such measures. A new decision support system (DSS) will not be developed as many tools are already available but hardly used. Reasons are the large amounts of required input data, a labour intensive development phase, slow availability of results and limited accessibility for water managers.

The methodology in this paper is based on the water quality model SWAT, but does not require water managers to model themselves. The impact of different scenarios is captured in relations between measure and output variable that can be represented in charts usable for management purposes. This paper focuses on point and non-point emission of nitrogen. The proposed methodology is applied in the Kleine Nete river basin in Belgium.

Study Area

The Kleine Nete river basin (320 km²) is located in the northeast of Belgium. The catchment is characterized by sandy soils, resulting in a high infiltration and a large baseflow component of river discharge. The study area is characterized by strong emission flows of both point and non-point sources. The major sources of pollution are (untreated) domestic waste water, industrial discharges and animal manure: up to 300kgN/ha.yr of manure is applied and 35% of the river basin population is not connected to waste water treatment. The Kleine Nete is furthermore characterized by a high density of animals bred for dairy and meat production (150 cows/km², 300 pigs/km² and 4000 chickens/km²) (Mestbank, 2005).

Model calibration

Impact assessment of pollution abatement measures on in-stream water quality is based on a calibrated SWAT2005 model (Neitsch et al., 2005). Flow and water quality variables are modeled on a daily time step. Point source data is introduced into SWAT as average daily nitrogen emissions (in kgN/day), manure applications as mass applied per ha (kg manure/ha). The model is calibrated for the period 2000-2002, based on monthly observations of organic nitrogen, nitrate, nitrite and ammonia and daily values for flow obtained from the Flemish environmental agency (VMM). Two years (1998-1999) are taken as warming up period. The SWAT model has been calibrated against flow, total nitrogen, nitrate and ammonia. Model performance is evaluated with the Nash-Sutcliff efficiency (NSE) and the coefficient of determination (R^2) as presented in Table 1. Model performance for flow is good and acceptable for water quality, considering its low data availability. Additionally, nitrogen emissions are partly estimated and therefore inherently uncertain. Pollutant loadings from households e.g. are estimated based on averaged Inhabitant Equivalents. Point sources entered into SWAT as average annual daily values can furthermore not represent daily dynamics. Total nitrogen values furthermore is determined for each nitrogen component separately. Measured measurement uncertainties of each nitrogen component are thus accumulated.

Table 9. SWAT model performance

Objective function	Flow	Nitrate (NO_3^-)	Total Nitrogen (TN)
NSE	0.84	-0.56	-1.69
R^2	0.88	0.58	0.42

Methodology for cost and impact assessment

Overview

To quantify cost and impact of nitrogen abatement measures, the methodology presented in Figure 1 is followed. Firstly, SWAT is used to model the impact of an emission reduction on the in-stream concentration and to setup a chart in which the total required emission reduction to reach a concentration target is visualized. Secondly, the Environmental Costing Model provides a comparison of the cost-effectiveness of measures across sectors. Hereby, cost-effectiveness (Euro/kgN removed) is defined as the ratio of cost over impact. A final step, which is out of scope of this paper, is combining the results of SWAT and the Environmental Costing Model, thereby selecting measures that can reach the required emission reduction effort at least-cost.

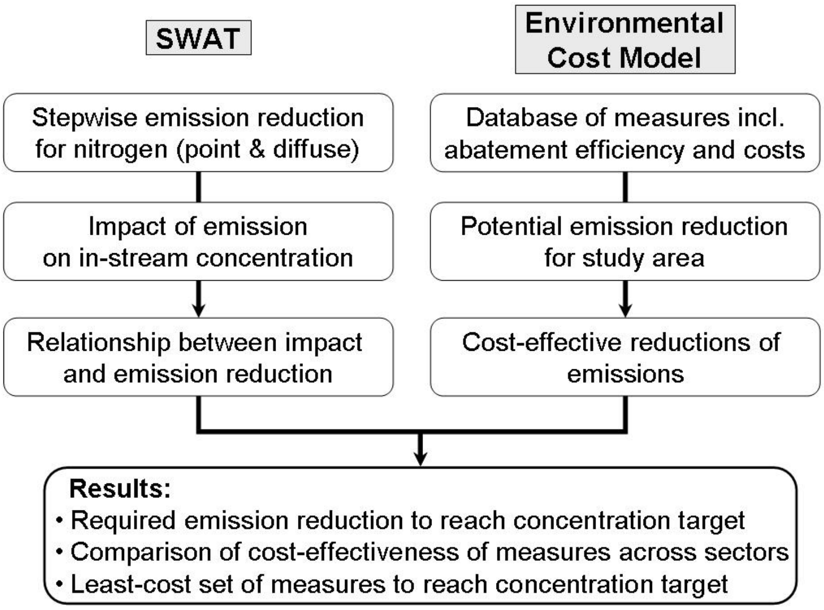


Figure 11. Methodology for cost and impact assessment of nitrogen abatement measures

Modeling the impact of emission reductions on in-stream water quality

The impact analysis is executed with the calibrated SWAT model. Impact is quantified as the sensitivity of SWAT outputs for a reduction in nitrogen inputs. The considered SWAT output variables are the in-stream concentration (mgN/l) of nitrate, organic nitrogen and total nitrogen at the watershed outlet as well as the export loads (kgN/ha) of the same nitrogen components of the hydrological response units (HRU). The first allows to gain insight in the overall impact of emission reductions from both point sources and diffuse sources on the river basin whereas the second assesses the response of the land phase to reduced manure application.

Nitrogen inputs for the watershed are reduced in steps of 10% from the current manure application rate down to 10% of this amount. Three scenarios are performed: 1) stepwise reduction of point source emissions only; 2) stepwise reduction of diffuse sources only; 3) stepwise reduction of point sources and diffuse sources. The reduction of diffuse sources consists of the simultaneous reduction of manure and fertilizer. To minimize interference in the modeled export loads due to interannual soil buffers, all modeled years are reduced simultaneously with an equal emission reduction percentage. Hereby, it is assumed that a new equilibrium in groundwater flow instantly exists after the emission reduction.

For linkage of the SWAT results to the Environmental Costing Model, three linkage parameters are used. The first two are derived from the relationship between the modeled in-stream concentration at the river basin outlet and the associated emission reduction. From this relationship, the required emission reduction effort to reach a concentration target is easily derived as the first linkage parameter. The second parameter, the sensitivity of the in-stream concentration to an emission reduction, is the slope of the above relationship. The third parameter is the export load coefficient (%). It is calculated with equation 1 and derived for the land use types corn and pasture. The export load coefficient represents the fraction of applied manure that arrives into the stream. Thereby, it integrates all land phase processes into one value.

$$\text{Export load coefficient} = \text{export load (kgN/ha)} / \text{manure applied (kgN/ha)} \quad [1]$$

Environmental Costing Model

The core of the Environmental Costing Model is a generic database of potential abatement measures, including their cost (Euro/year) and nitrogen removal efficiency (%). The database includes Best Available Techniques (BAT) and Best Management Practices (BMP). The cost and technical nitrogen removal efficiency of BATs are available in the BAT Reference documents (BREFs[4]). For BMPs, however, reference documents are not available. Both for BATs and BMPs, the resulting database is cross-checked and extended in discussion with sector-specific experts.

The removal efficiency and costs of agricultural BMPs are quantified with models used by the Flemish government. SENTWA is a semi-empirical model that simulates nutrient emissions from agriculture (mainly form manure application) to the surface water (Van Hoof, 2003). It quantifies the export load of total nitrogen and total phosphorus on an annual basis per watershed. The cost of a reduction in manure application is modeled with the SELES model. SELES models the response of a farmer to maximize his income e.g. under manure application restrictions (Gavilan et al., 2006).

[4] BREFs are available through the Energy and Environmental Information system for Flanders (EMIS): http://www.emis.vito.be/

Removal efficiencies are then used to derive the potential emission reduction of a measure for the study area. The potential emission reduction is based on the current emission load, available in the SWAT input, and the current and maximal degree of implementation of a specific measure on subcatchment level. By means of the export load coefficient, finally, the potential emission reduction is converted to the effective impact in-stream. Cost estimates are kept constant throughout the river basin.

Results and discussion

Relationship between in-stream concentration and emission reduction

Although the nitrogen pathways and the SWAT model are non-linear, model results show that the response of the in-stream concentration of nitrogen to emission reduction is linear. A linear relationship (R^2=1), as shown in Fig. 2, is observed for all nitrogen components both for the abatement of point sources and for manure and fertilizer. The sensitivity of the in-stream concentration to an emission reduction is presented in Table 2.

Table 10. Sensitivity coefficients of in-stream concentrations of organic nitrogen (ORGN), nitrate (NO3) and total nitrogen (TN) to a reduction in nitrogen input

Reduction scenario	ORGN	NO3	TN
Fertilizer only	-0.02	-0.41	-0.44
Point source only	-0.48	-0.30	-1.41
All emission	-0.51	-0.71	-1.84

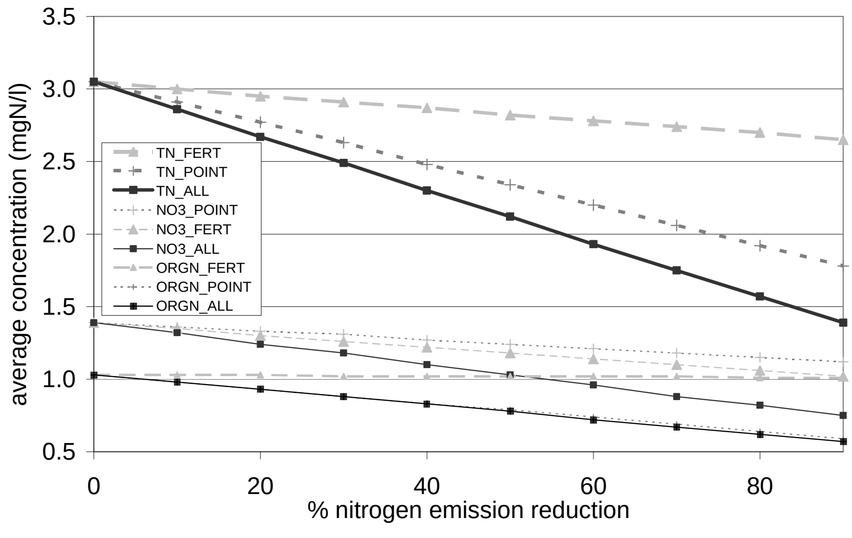


Figure 12: Relationship between in-stream concentration and nitrogen abatement for three groups of three scenarios: From top to bottom are shown: three scenarios for total nitrogen (TN), three for nitrate (NO3) and three for organic nitrogen (ORGN). For each nitrogen component, three scenarios are shown: FERT = reduction of applied fertilizer only; POINT=reduction of point sources emission only; ALL= reduction of both fertilizer and point source emissions.

The impact of point sources on the concentration of total nitrogen is three times larger than the impact of fertilizer and manure, mainly as a consequence of the large contributions of organic nitrogen and ammonia compared to the diffuse sources (Table 3). Alternatively, abatement can focus on specific nitrogen components: 46% of total nitrogen are nitrates, 34% is organic nitrogen. To abate nitrates, the reduction of manure is, contrary to total nitrogen, 30% more effective than the reduction of equal loads of point sources. The applied manure, a liquid manure composed of urine and dung, is the biggest source of nitrate. It is mainly composed out of ammonia and has only 5% of organic nitrogen. In SWAT, ammonia in manure is nitrified completely before it reaches the stream. The low organic nitrogen content of manure furthermore explains the insignificant response of the organic nitrogen concentration to a reduction in manure application. Organic nitrogen can effectively be abated through the reduction of point sources (mainly protein residues). The share of organic nitrogen, coming from plant residues is more difficult to address. Yet, organic nitrogen from natural sources contributes significantly to the in-stream concentration at the outlet. Figure 2 shows that a 90% reduction scenario still results in a concentration of 0.52mgN/l of organic nitrogen originating from eroded crop residues, runoff and sediments. The best concentration that can be achieved, under 90% reduction scenario, is 1.39 mgN/l for total nitrogen and 0.75 mgN/l for nitrate.

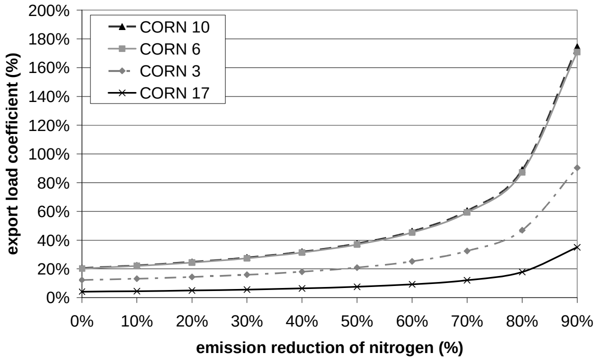
For the study area, the linear relationship is valid for impact assessment of nitrogen abatement in the modeled range of concentrations. Linearization of the prevailing non-linear processes is acceptable given the specific conditions of the study area. Firstly, in-stream conversions of nitrogen components are insignificant as travel time is less than one day. Secondly, the modeled times series (not visualized) show that baseflow is dominant and that nitrogen loads are mainly dissolved in baseflow as nitrate. Less manure application drastically reduces the nitrate loads in baseflow, especially in summer when the contribution of baseflow to total flow is maximal. Peak loads, mainly composed of organic nitrogen, decrease slightly. Thirdly, groundwater processes, especially travel time and denitrification, are poorly modeled in SWAT. As a consequence of the semi-lumped approach of SWAT, the position within the basin is lost and hence groundwater travel time is averaged and thus linearized. Hattermann et al. (2006) developed a SWAT variant in which the simulation of denitrification and travel time is improved for the lowland Nuthe basin (Germany). Finally, nutrient uptake by plants remains more or less constant in the scenarios. This signifies that nutrients are applied in excess. Nutrients originate not only form manure, but as well from crop residues.

SWAT2000 results for the intensively manured lowlands of Iowa (US) give comparable results. Jha et al. (2007) report fairly linear results for the Raccoon watershed in response to declining and increasing nitrogen application rates. Yet, as the Raccoon watershed is 10 times as big as the Kleine Nete, in-stream processes might cause the slight non-linearity. Chaplot et al. (2004) present a linear relationship for annual nitrate loadings for the 50 km² Walnut Creek.

Export load coefficients

Figure 3 presents results of the annually averaged export load coefficients for corn and pasture fields. Despite the linear response of the export load to emission reductions, the export load coefficient increases semi-exponentially in function of the emission reduction. The latter is caused by the fact that the fraction of organic nitrogen (not linked to manure application) remains constant when manure application is reduced extremely. A large variation is furthermore observed between the different response units of the same land use type (Figure 3 and Table 4). For corn, the export load coefficients are between 4.2% and 20%. Due to the grass cover, the response of pasture is much smaller, between 0.3% and 1.8%. The area-weighted average for the watershed is 10.8%. In the Environmental Costing Model, by using the SENTWA model, a comparable value of 9% is found for the whole study area.

(a)


200%
180%
160%
140%
120%
100%
80%
60%
40%
20%
0%
CORN 10
CORN 6
CORN 3
CORN 17
export load coefficient (%)
0%
10%
20%
30%
40%
50%
60%
70%
80%
90%
emission reduction of nitrogen (%)

(b)

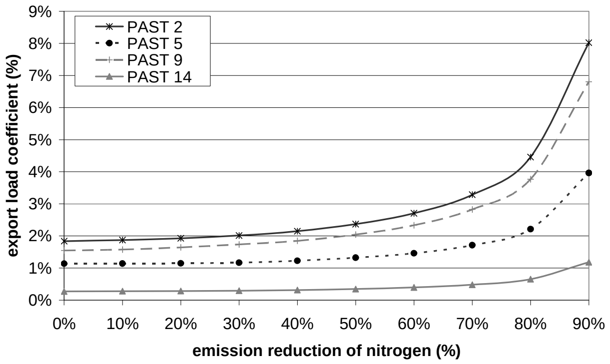

9%
8%
7%
6%
5%
4%
3%
2%
1%
0%
PAST 2
PAST 5
PAST 9
PAST 14
export load coefficient (%)
0%
10%
20%
30%
40%
50%
60%
70%
80%
90%
emission reduction of nitrogen (%)

Figure 13: Export load coefficients in function of the degree of emission reduction. Left (a) shows the export load coefficients for CORN, right (b) for pasture (PAST). The numbers signify the different HRUs modeled in SWAT.

Cost-effectiveness of nitrogen abatement measures

Following the above methodology, a selection of potential abatement measures including their total annual cost (Euro/year), the nitrogen removal efficiency and the cost-effectiveness CE (Euro/kgN removed) are presented in Table 3. The CE values allow to compare pollution abatement measures across sectors. Besides companies that closed down from the reference year 2002, the construction of riparian buffer zones and green manure application are most cost-effective. For household waste water treatment, mostly the connection to a waste water treatment plant (WWTP) is more cost-effective than individual treatment of waste water. Individual treatment is preferred when houses are too isolated and the cost of sewering becomes too expensive. As the industrial companies included in the model already installed some form of N-treatment, only a more advanced and less cost-effective treatment technique of membrane filtration is included. Data stated in the table below are average estimates for the study area. Estimates will differ for other study areas.

Table 11. Selection of nitrogen abatement measures with total cost, nitrogen removal efficiency and cost-effectiveness.

measure	target group	total cost	N removal efficiency	CE (Euro/kgN removed)
factory closure	industry	0	100%	0
membrane filtration (up to 500m³/day)	industry	$50\text{-}56*10^3$ Eur/installation.yr	65%	177-297
connect to WWTP	household wastewater	156-410 Eur/IE.yr	56-75%	65-172
individual treatment	household wastewater	172-326 Eur/IE.yr	10-55%	162-472
riparian buffer zones 5m	diffuse processes	1469 Eur/ha.yr	7%	9.6
riparian buffer zones 10m	diffuse processes	1469 Eur/ha.yr	10%	10.3
green manure application after harvest	diffuse processes	100 Eur/ha.yr	8%	12.8
reduce manure application to 170 kgN/ha	agriculture	653 Eur/ha.yr	27%	101.7

Conclusions

A linear relationship between the in-stream concentration and emission reductions can be validated for impact assessment of nitrogen for the specific conditions of the study area. This signifies that linearity can be accepted in the modeled range of concentrations for small, sandy lowland watersheds dominated by baseflow. Groundwater processes in SWAT, however, need to be improved for groundwater travel time and denitrification.

The cost-effectiveness of emission reduction measures is successfully assessed through linking SWAT to the Environmental Costing Model. A generic framework is presented which allows to determine the required emission reduction to reach a concentration target and consequently the most cost-effective set of abatement measures to reach that target. Due to a broad variety of technical efficiency and costs of abatement options as well on export load coefficients, large uncertainty on costs and effects exists. The presented results, however, are valid for first assessment of required emission reduction and comparison of cost-effectiveness of measures across sectors and processes. For effective use in water management, acceptance of ‘values’ by stakeholders is essential, comparison of export load coefficients and nitrogen processes. Finally, impact assessment cannot be limited to nutrients. In order to achieve the objectives of the European Union Water Framework Directive and to assess how good ecological status can be reached in a cost-effective manner by 2015, an analysis which is solely aimed at nitrate, organic nitrogen and total nitrogen will not suffice. How to extend this framework to determine cost-effective measures for multiple parameters at once remains a challenge. Especially on ecological quality advance has to be made of impact assessments of abatement measures.

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INCORPORATION of Hooghoudt and Kirkham Tile Drain Equations into SWAT2005

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Abstract

Agricultural tile drainage is a common water management practice in agricultural regions with seasonal high water tables such as the Midwest U.S. The goal of this study was to modify SWAT2005 to enable it to perform multiple scenario simulations to determine cost-effective water management systems at the watershed scale. This was accomplished by incorporating the Hooghoudt steady-state and Kirkham tile drain equations. The Hooghoudt steady-state and Kirkham equations have been successfully used in DRAINMOD, a computer simulation model that has been tested and widely used to simulate the performance of drainage and water table control systems on a continuous basis at field-scale. These equations depend on maximum depressional storage (Sd). Sd, which is assumed constant in DRAINMOD, is allowed to change, in the modified SWAT2005 model, as a function of dynamic soil random roughness. The dynamic random roughness is a function of tillage type and intensity and amount of rainfall. The drainage flux is calculated using a three-step approach. 1) Hooghoudt steady-state equation is used when the water table is below the surface or Sd; 2) Kirkham equation is used to compute the ponded surface drainage flux when the water table rises to completely fill the surface; and 3) when the drainage flux predicted by the appropriate equation is greater than the design drainage capacity (also known as drain coefficient (DC)), then the flux is set equal to DC. These algorithms were successfully incorporated into SWAT2005 and the enhanced SWAT2005 is currently undergoing validation. The modified SWAT2005 shows great potential for use in some of the conservation effects assessment project (CEAP) benchmark watersheds such as South Fork Watershed in north central Iowa and in other regions of the world with tile drainage.

KEYWORDS: SWAT, DRAINMOD, tile drainage, depressional storage

Introduction

Agricultural subsurface drainage, commonly known as tile drainage, is a common water management practice in agricultural regions with seasonal high water tables such as the in the Midwest U.S. There are several theories for steady-state drainage of layered soil (Hooghoudt, 1940; Luthin and Gaskell, 1950; Kirkham, 1951; Kirkham, 1957; Wesseling, 1964; Dagan, 1965; Toksoz and Kirkham, 1971; Ernst, 1978; and Wu and Chieng, 1991). However, DRAINMOD (Skaggs, 1978) was the first comprehensive computer model developed to simulate the water balance and the impact of subsurface water management of tile-drained fields. DRAINMOD predicts surface runoff, water table depth, drainage outflow, soil water content, evapotranspiration (ET), and infiltration on an hourly, daily, monthly, or annual basis in response to given soil properties, crop variables, climatological data, and site parameter inputs. Several components of DRAINMOD have been adopted and used by several point- and field-scale models such as CREAMS [Chemicals, Runoff, Erosion from Agricultural Management Systems] (Knisel, 1980), GLEAMS [Groundwater Loading Effects of Agricultural Management Systems] (Leonard et al., 1987), and RZWQM [Root Zone Water Quality Model] (Ahuja and Hebson, 1992).

One of the watershed-scale models that contain a tile drainage component is the Soil and Water Assessment Tool (SWAT) model (Arnold et al., 1998; Arnold and Fohrer, 2005). SWAT is a continuous-time physically-based watershed-scale model developed to predict the impact of land management practices on water, sediment, and agricultural chemical yields in watersheds with varying soils, land use, and management conditions over time. SWAT requires specific information about weather, soil properties, topography, vegetation, presence of ponds or reservoirs, groundwater, the main channel, and land management practices to simulate water quality and quantity. SWAT has become an effective means for evaluating nonpoint source water resource problems (flow, sediment, nutrients) for a large variety of water quality applications nationally and internationally. One of the reasons for this extensive use is because SWAT is continuously under development to meet the needs of its users. Gassman et al. (2007) gives a detailed description of the historical development and applications of SWAT.

For example, SWAT2000 was enhanced with a subsurface tile flow component and tested with data from a field-scale area with satisfactory results (Arnold et al., 1999). But when applied on a watershed-scale, SWAT2000 did not accurately simulate subsurface flow and stream discharge because the incorporated tile algorithms did not accurately represent the hydrologic processes at the watershed-scale and the effects of large depressional areas (potholes), prevalent within the watershed of interest, were not included in the algorithms (Arnold et al., 1999). Later, SWAT2000 was modified (SWAT-M)to simulate water table dynamics and linked with a simple tile flow equation in addition to inclusion of pothole algorithms (Du et al., 2005). These modifications resulted in improved predicted pattern and amount of monthly flow and subsurface drainage (Du et al., 2005). However, these modifications do not allow for scenario simulations, such as varying tile spacing, depth, and size, in order to aid in designing cost-effective water management systems. The objective of this study was to incorporate the Hooghoudt and Kirkham steady-state tile equations into SWAT2005 to allow for multiple simulations to determine the cost-effective design.

Model Modifications

DRAINMOD tile equations

Hooghoudt (1940) steady-state and Kirkham (1957) tile equations, which have been used in DRAINMOD model (Skaggs, 1978), were used in these modifications. The rate of subsurface water movement into drain tubes or ditches depends on the hydraulic conductivity of the soil, drain spacing and depth, soil profile depth, and water table elevation (Skaggs, 1980). Water moves toward drains in both the saturated and unsaturated zones. DRAINMOD computes the drainage rates by assuming that the lateral water movement occurs mainly in the saturated region. DRAINMOD uses effective horizontal saturated hydraulic conductivity and evaluates flux in terms of the water table elevation midway between the drains and the water level or hydraulic head in the drains.

The methods used in DRAINMOD to estimate the drain flux are the Hooghoudt (1940) steady-state and Kirkham (1957) equations. The Hooghoudt (1940) steady-state equation is used to compute both drainage and sub-irrigation flux. The drainage flux is calculated using a three-step approach as follows:

First, for water tables below the surface and for ponded depths less than S_1 (Fig. 1), when surface water can not move freely toward the drains, the Hooghoudt (1940) steady-state equation is used to compute flux as represented by equation 1:

$$q = \frac{8K_e d_e m + 4K_e m^2}{CL^2} \quad (1)$$

where q = flux in mm h^{-1}, m = midpoint water table height above the drain (mm)(Fig. 2), K_e = effective lateral hydraulic conductivity (mm h^{-1})(Fig. 2), L = distance between drains (mm)(Fig. 1), C is the ratio of the average flux between the drains to the flux midway between the drains and is assumed to be unity (C=1) in DRAINMOD model and in the modified SWAT2005 of this study, d_e = equivalent depth substituted for d (height of the drain from the impervious layer) in order to correct for convergence near the drains (mm) (Fig. 2). S (Fig. 1) is the maximum depressional storage (mm).

For layered soils (Fig. 2), parameter K_e, which used in equations 1, 3, and 5, is calculated as:

$$K_e = \frac{K_1 d_1 + K_2 D_2 + K_3 D_3 + K_4 D_4}{d_1 + D_2 + D_3 + D_4} \quad (2)$$

Because the thickness of the saturated zone in the upper layer is dependent on the water table position, K_e is determined before every flux calculation using the value of d_1 (Fig. 2) which depends on the water table position. If the water table is below layer 1, d_1 = 0 and a similarly defined d_2 is substituted for D_2 in the above equation.

The equivalent depth (d_e) is obtained using the equations developed from Hooghoudt's solutions by Moody (1966) as a function of L, d, and r. For real, rather than completely open drain tubes, there is an additional loss of hydraulic head due to convergence as water approaches the finite number of openings in the tube (Skaggs, 1978). The effect of various opening sizes and configurations can be approximated by defining an effective drain tube radius, r_e, such that a completely open drain tube with radius r_e will offer the same resistance to inflow as a real tube with radius r (Skaggs, 1978). Skaggs and Fernandez (1998) give a table of r_e for various tile tubing sizes, which can be used to determine d_e and g (Eq 3).

Secondly, for ponded depths greater than S_1, when water table rises to completely fill the surface and with ponded water remaining there for relatively long periods of time, the ponded surface flux drainage can be computed using Kirkham (1957) equation:

$$q = \frac{4\pi K_e\left(t + b - r\right)}{gL}$$

(3)

where g is a dimensionless factor which is determined using an equation developed by Kirkham (1957) as a function of d, L, actual depth of the profile (h, mm), m, and the radius of the tile tube (r, mm).

Thirdly, when the flux predicted by the appropriate equation is greater than the drainage coefficient (DC, mm day^{-1}), then the flux is set equal to the DC as:

$$q = DC$$

(4)

The DC is typically 10 to 20 mm day^{-1} depending on the geographic location and crops grown (Skaggs, 1980). The DC may also be estimated as a function of the surface inlet types, soil type, and crop type (Wrighr and Sands, 2001).

Finally, if subsurface irrigation is used, water is raised in the drainage outlet so as to maintain a pressure head at the drain tube of h_o (Fig. 2). Subsurface irrigation flux is estimated by equation 5.

$$q = \frac{4K_e m\left(2h_o + \frac{h_o}{D_o}m\right)}{L^2}$$

(5)

where $D_o = y_o + d$, d = distance from the drain tube to the impermeable layer, ho=yo+ de (Figs. 1 and 2).

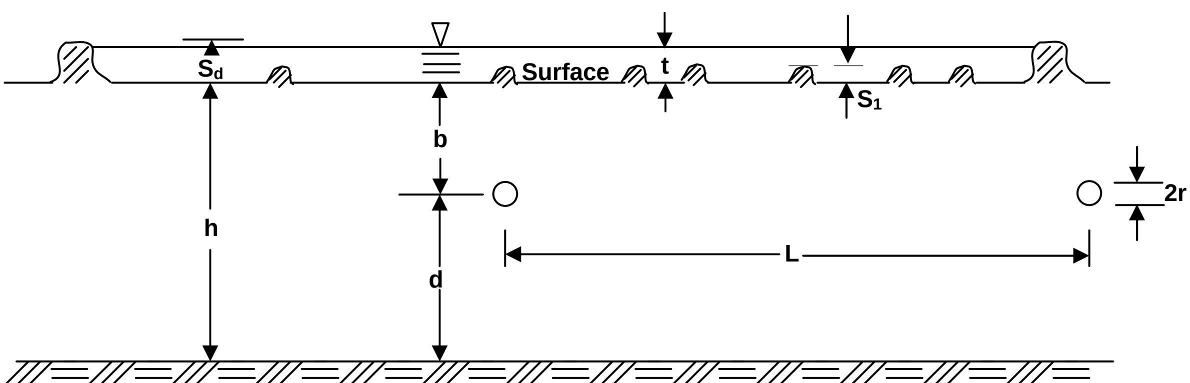

Figure 14. Schematic of drainage from a ponded surface. Water will move over the surface to the tile vicinity until the ponded depth becomes less than S_1 (redrawn from Skaggs, 1980). S_d is the maximum depressional storage.

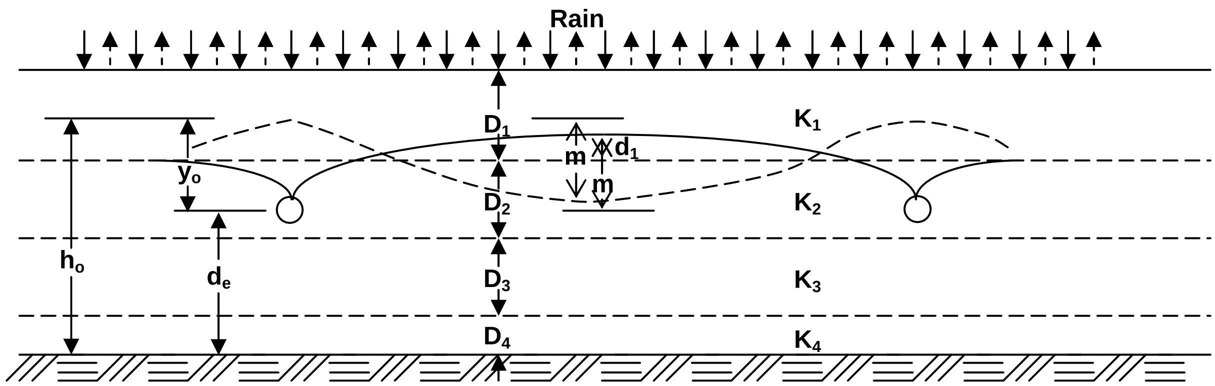

Figure 15. Schematic of water table drawdown to and subirrigation from tile tubes. Equivalent lateral hydraulic conductivity (K_e) is determined for soil profiles with up to 6 layers. K_n is lateral hydraulic conductivity for layer n, D_n islayer n thickness, d_n is the distance from the bottom of semi-saturated layer n to the water table elevation mid-way between the tile drains (modified from Skaggs, 1980).

Dynamic maximum depressional storage

Maximum depressional storage (S_d and hence S_1) is used to determine when to compute drainage flux using either equation 1 or 3 in DRAINMOD model and in this study. In addition, S_d partly influences rainfall or irrigation water infiltration and runoff and hence subsurface drainage. S_d is assumed constant in DRAINMOD although several studies show that it is dynamic. In this study, S_d is allowed to change as a function of the dynamic random roughness.

Several models have been developed and used to calculate S_d (Moore and Larson, 1979; Onstad, 1984; and Guzha, 2004). The model developed by Onstad (1984), in which S_d is computed based on the random roughness and slope of the depressions, was adopted in this study. S_d is estimated as:

$$S_d = 0.112RR + 0.031RR^2 - 0.012RR * S \tag{6}$$

where S_d is the maximum depressional storage (cm), RR is random roughness (cm), defined below, and S is the slope steepness (%).Generally, depressional storage decreases with decreasing random roughness and increasing slope steepness (Onstad, 1984).

Soil roughness is a dynamic factor readily modified by tillage, orientation of ridges, and weather, primarily intensity and amount of rainfall. The random roughness equation (Eq. 7) developed by Saleh and Fryrear (1999) was used in this study.

$$RR = 0.1RRi * e^{\left[DF\left(-0.0009CUMEI - 0.0007CUMR\right)\right]} \tag{7}$$

where RR is random roughness at any time t days after a tillage operation (cm), RR_i is the random roughness immediately after a tillage operation (mm), $CUMEI$ is cumulative rainfall erosivity (Mj mm ha^{-1} h^{-1}) and $CUMR$ is cumulative rainfall since last tillage operation (mm), and DF is decay factor based on percent clay in the soil ($CLAY$) and percent organic matter (OM) and is computed as follows:

$$DF = e^{\left[0.943 - 0.07CLAY + 0.0011CLAY^2 - 0.67OM + 0.12OM^2\right]} \tag{8}$$

RR_i is read from a given input file of random roughness values for different tillage types. *CLAY* and *S* are inputs in SWAT2005, whereas *OM* is estimated using amount of organic matter in the top soil layer classified as residue, depth to bottom of the first layer, and bulk density of top soil layer, which are all inputs in SWAT2005. *CUMEI* is computed using the Universal Soil Loss Equation erosion index, which is computed within SWAT2005 and finally *CUMR* is computed using the daily precipitation which is also an input in SWAT2005.

Model Inputs

Two subroutines, DRAINS.f and DEPSTOR.f for tile drainage and dynamic maximum depressional storage, respectively, were written and incorporated into SWAT2005 to effect the modifications using the equations discussed in this section. In addition to tile drainage prediction inputs for SWAT2005, which include depth to subsurface tile, time to drain soil to field capacity, drain tile lag time and depth to impervious layer, the modified SWAT2005 model requires several new inputs (Table 1). These include actual depth to impervious layer, drainage coefficient, factor g used in Kirkham (1957) equation, equivalent depth from water surface in tile tube to impervious layer, tile drainage flag/code used to switch between the old and new incorporated tile algorithms, pump capacity for subsurface irrigation, random roughness for a particular tillage operation, and distance between the mid-points of two tile tubes.

Table 12. SWAT2005 input parameters for tile and pothole drainage predictions.

SWAT2005 variable name	Parameter description
DDRAIN	Depth to subsurface tile (mm)
DEP_IMP	Time to drain soil to field capacity, time required to drain the water table to the tile depth (h)
GDRAIN	Drain tile lag time, the amount of time between the transfer of water from the soil to the drain time and the release of the water from the drain tile outlet to the channel (h)
TDRAIN	Depth to impervious layer (mm)
Additional inputs for the modified SWAT2005	
ADEPTH	Actual depth from surface to impervious layer (mm)
DC	Drainage coefficient (mm day^{-1})
GEE	Factor –g- in Kirkham (1957) equation (dimensionless)
HDRAIN	Effective depth from drain to impermeable layer (mm)
ITDRN	Tile drainage routines flag/code: 1 = DRAINMOD tile equations (Subroutine DRAINS); 0 = Original SWAT tile equations (Subroutine ORIGTILE)
PC	Pump capacity (default value = 1.042 mm h^{-1} or 25 mm day^{-1}) (mm h^{-1})
RANRNS	Random roughness for a given tillage operation (mm)
SDRAIN	Distance between two drain or tile tubes (mm)

Summary and Conclusion

Earlier on, SWAT2000 was enhanced with a subsurface tile flow component and tested with data from a field-scale area with satisfactory results. Later, SWAT2000 was modified (SWAT-M) to simulate water table dynamics and linked with a simple tile flow equation in addition to inclusion of pothole algorithms, which improved the predicted pattern and amount of monthly flow and subsurface drainage on a watershed-scale. However, these modifications do not allow for scenario simulations. In this study, the tile equations used in DRAINMOD model were incorporated into SWAT2005 to further improve the existing sub-surface tile flow component. The inclusion of the Hooghoudt steady-state and Kirkham tile equations is anticipated to enable scenario simulations, which will aid in designing cost-effective water management systems in agricultural regions with shallow water tables such as the Midwest US.

Based on these equations, the rate at which subsurface water moves into tiles depends on the hydraulic conductivity of the soil, drain spacing and depth, profile depth, and water table elevation. Water moves toward tiles in both the saturated and unsaturated zones in the modified SWAT2005 model. The drainage flux is calculated using a three-step approach. 1) Hooghoudt steady-state equation is used when the water table below the surface or below maximum depressional storage, when surface water can not move freely toward the tiles, 2) Kirkham equation is used to compute the ponded surface drainage flux when the water table rises to completely fill the surface with ponded water remaining on the surface for relatively long periods of time, and 3) when the drainage flux predicted by the appropriate equation is greater than DC, then the flux is set equal to the DC.

In addition, maximum depressional storage, which is assumed constant in the DRAINMOD model, is allowed to change in the modified SWAT 2005 as a function of the dynamic random roughness. The dynamic random roughness is a function of tillage types and the intensity and amount of rainfall. Next, the modified SWAT2005 will be calibrated and validated on the South Fork Watershed in north central Iowa and three watersheds located within Muscatatuck River Basin in southeast Indiana. Future developments include modification of these tile algorithms to allow for controlled drainage as a best management practice to reduce nitrates and soluble P in the surface and sub-surface water.

Acknowledgments

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Estimation of Nitrate Loss Through Tile Drains Using SWAT2005

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Abstract

Knowing the percentage of nitrate that comes from tile drains would be useful in estimating the potential of drainage-related conservation practices to reduce nitrate in streams. SWAT2005, with its modified tile drain component, was used to evaluate nitrate loss through tile drains in a heavily tile drained watershed, Sugar Creek watershed in Indiana. Stream flow predictions resulted in Nash-Sutcliff Efficiency values of 0.90 in the 5-year calibration period and 0.81 in the 6-year validation period. Monthly nitrate-N loads were predicted well with Nash-Sutcliff Efficiency values of 0.61 in the calibration period and 0.63 in the validation period. A method to output nitrate-N load from tiles was implemented, and the simulated nitrate loss through tile drains was compared with nitrate losses from the other flow paths (surface flow, percolation and lateral flow). The estimated median percentage of monthly nitrate loss entering Sugar Creek that flows through tile drains ranged from 0% to 65%, and was more than 40% for three months, April to June. These estimates can be used to determine the potential for tile-related management practices to reduce nitrate at the watershed level.

KEYWORDS. *Nitrate, Tile Drainage, Agriculture, Nutrients*

Introduction

Subsurface drainage, a common practice in the Midwestern U.S. and other agricultural areas where natural drainage is inadequate, often contributes to nitrate concerns in surface water. Many studies (e.g., Gilliam and Skaggs, 1986; Kladivko et al., 2004) have determined the impact of tile drains on nitrate loss at the field scale However, less is known about their impact on nitrate loads in rivers or streams. Watershed-scale simulation of nitrate loss could help determine the amount of nitrate from tile drains, and therefore the potential impact on watershed-scale nitrate loss of changing tile drain management.

The tile flow algorithm in SWAT has recently been improved by Du et al. (2005) to better simulate landscapes with tile and potholes drainage systems. They added a new function to predict the dynamic ground water table and tile flow by setting a restrictive soil layer at the bottom of the soil profile, and this improved tile drainage representation is included in SWAT2005 (Neitsch, 2005). Green et al. (2006) implemented SWAT2005 in a large tile drained watershed in north central Iowa, and found that the new function significantly improved the water balance and runoff simulation. Simulation of the highly tile-drained Walnut Creek watershed with the modified SWAT model improved monthly nitrate-N loads compared with the version without the new function (Du. et al., 2006).

The objective of this study is to simulate nitrate loads through tile drains in a heavily tile drained watershed, Sugar Creek watershed in Indiana, and compare it with nitrate loads through other flowpaths including surface runoff, lateral flow, and groundwater. Estimating the watershed-scale potential of conservation practices that reduce nitrate loads from tile drains, such as drainage water management (Evans et al., 1995, Frankenberger et al., 2006) or bioreactors (Greenan et al., 2006) requires an estimate of the nitrate contribution of tile drains at similar watershed scales.

Material and methods

The Sugar Creek Watershed

Sugar Creek drains a 242 km^2 agricultural (70%) watershed in the White River basin in central Indiana (Figure 1). The soils in the watershed are poorly drained and nearly level, and tile drain systems have been installed on most cropped fields where natural drainage is inadequate for high yields. The average annual precipitation in the Sugar Creek watershed is around 1000 mm, which is evenly distributed throughout the year. There are no major wastewater or industrial permitted facilities in the watershed.

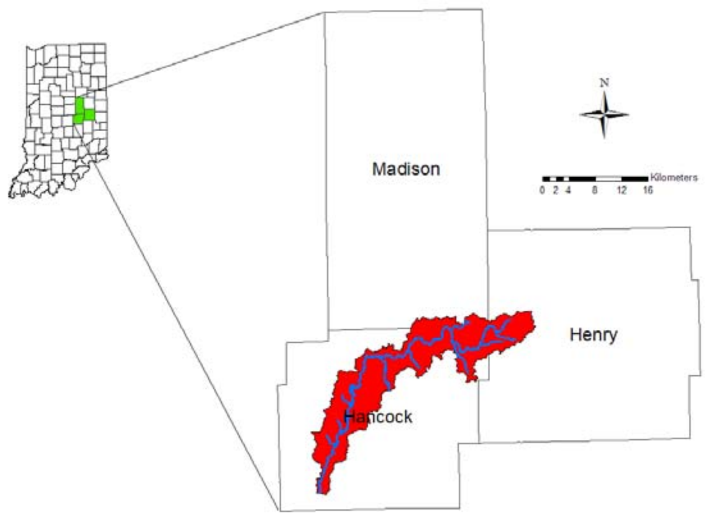


Figure 1. Location of the Sugar Creek Watershed in central Indiana

Flow and nitrate data from the United States Geological Survey (USGS, 2007) were used in this study. Pesticide data available at the same site were previously used by Neitsch et al. (2002). Nitrate concentrations were sampled bi-weekly or monthly, and in this study a total of 253 nitrate samples were used over the period of more than 11 years (05/07/1992 to 9/19/2003). For days without nitrate concentration measurements the nearest concentration value was used to calculate daily nitrate-N loads. Monthly nitrate-N loads were calculated by summarizing the daily nitrate-N loads. Flow and nitrate data from 05/1992 to 09/1997 were used for calibration, and 10/1997 to 9/2003 were used for validation. SWAT model inputs, including weather, soil, land use, HRU definition, and agricultural management inputs are described in Sui, 2007.

Nitrogen-related parameters and sensitivity

The initial soil values related to nitrate including SOL_NO_3, SOL_ORGN, and RSDIN were established by simulating the model for two years before the start of the calibration period. The sensitivities of other nitrogen parameters were assessed by evaluating the change in average monthly nitrate-N loads resulting from varying the parameters within bounds used by previous researchers (Santhi et al., 2001, Saleh, et al., 2004, Vandenberghe et al., 2001). The most sensitive parameters controlling the transformation of nitrogen in different pools were CMN and SDNCO. NPERCO, the parameter controlling the distribution of nitrogen in different layers, also played an important role. Other controlling parameters include the benthic source rate for NH_4-N

(RS3), rate coefficient for organic N settling (RS4), rate constant for biological oxidation from NH_4 to NO_2 and NO_2 to NO_3 (BC1 and BC2) and rate constant for hydrolysis of organic N to NH_4 (BC3). The sensitivity analysis showed that the rate constant for biological oxidation of NH_4 to NO_2 (BC1) had the greatest effect on nitrate among the in-stream nitrate controlling parameters.

Calculation of nitrate from tile drains

Tile drains were assumed to be present in all corn/soybean HRUs for which the soil drainage class was somewhat poorly, poorly, or very poorly drained, based on common practice in Indiana (Franzmeier et al., 2001). The depth of the tile drain from the top of the soil surface (DDRAIN) was set as 1000 mm, and the time to drain the soil profile to field capacity (TDRAIN) was 24 hours. The time it takes for water to enter the channel network after entering the tiles (GDRAIN) was 48 hours, following Neitsch et al. (2002).

In order to be able to quantify nitrate loss from tile drains, the SWAT model source code was modified to output nitrate from tile drains in the watershed along with the other flowpaths in the output.std file. The assumption was made that nitrate concentration in the tile flow is the same as the nitrate concentration in the soil layer where the tiles are put in. Nitrate load through tile drains was calculated by multiplying the nitrate concentration in the tile by the tile flow volume for each HRU, and then summarizing across the watershed. Additional information included the layer number where tiles were put in, the nitrate concentration in each layer and the volume of tile flow for each HRU were plotted in a new file.

Calibration

With the new tile drainage function in SWAT2005, the model is very sensitive to the depth of the impermeable layer in the tile-drained fields. The default value is 6000 mm, and 2000 mm was selected to best fit the stream observation data. The importance of this depth results not only from its role in controlling the volume above it, but also because in the current implementation of SWAT2005 the depth value also controls the extent to which the layer is impermeable (J. Arnold, personal communication). The runoff curve number for moisture condition II (CN2), was lowered by 7, Manning's "n" value for the main channel (CH_N) was changed to 0.009, and the soil evaporation compensation factor (ESCO) was changed to 0.90. Three parameters related to the Muskingum routing method for stream flow, the calibration coefficient used to control impact of the storage time constant for normal flow (MSK_CO1), low flow (MSK_CO2), and the weighting factor controlling the relative importance of inflow rate and outflow rate in determining water storage in the reach (MSK_X), were based on Neitsch et al. (2002).

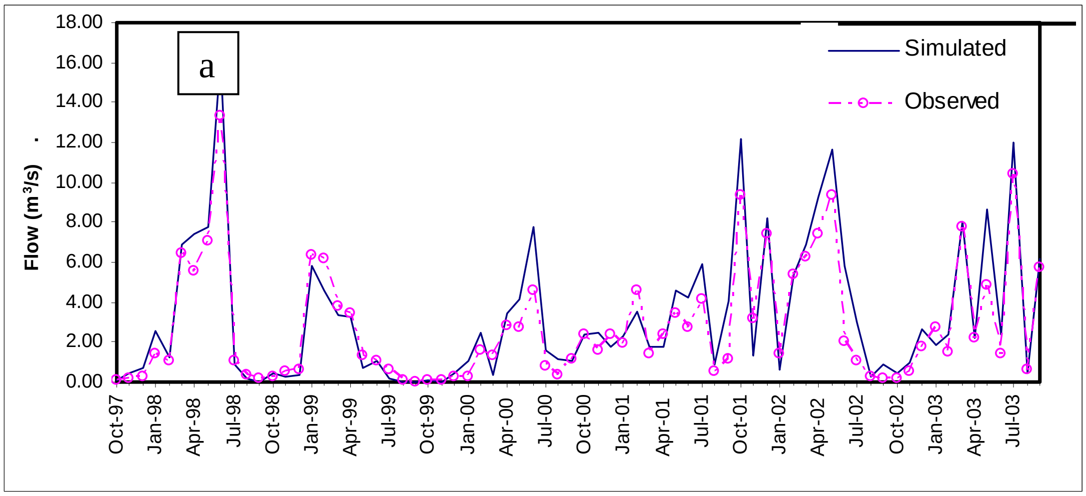
For nitrate calibration, the threshold value of nutrient cycling water factor for denitrification to occur (SDNCO) was set to 0.9 rather than the default value of 1.1 in the source code, because without this change no denitrification occurred. The nitrate percolation coefficient (NPERCO) was 0.01, the rate coefficient for mineralization of the humus active organic nutrients (CMN) was 0.019, and the in-stream nitrate oxidation parameter (BC1) was changed to 0.95.

Results and Discussion

Streamflow and Nitrate Load Predictions

Predicted average monthly flow rates showed a close relationship to the observed in the calibration period (R^2 = 0.96) and the validation period (R^2=0.95; Figure 2). The Nash-Sutcliffe coefficient, 0.90 in the calibration period and 0.81 in the validation period, showed that the observed and simulated were close to the 1:1 line.

Monthly nitrate-N load predictions were also strong, with Nash-Sutcliffe efficiencies of 0.61 in the calibration period and 0.63 in the validation period (Figure 2) easily exceeding the often-recommended minimum of 0.5 (i.e., Santhi et al., 2001). The R^2 value was 0.80 (calibration) and 0.81 (validation), which shows that the simulated monthly nitrate-N loads followed the observed values very well. The fact that the monthly nitrate-N loads were close to the observed values in the validation period suggests the processes are well represented.



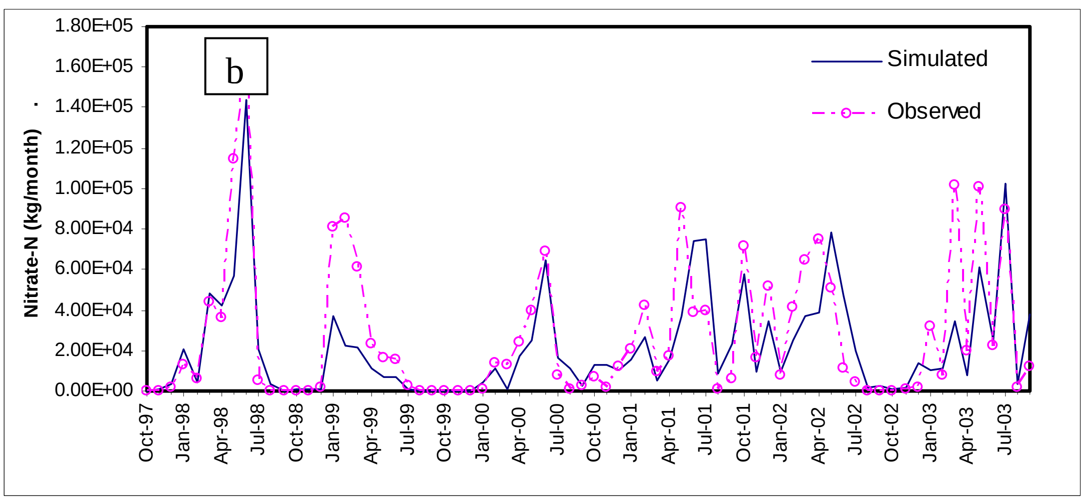


Figure 2: Observed and simulated (a) average monthly flow rate and (b) monthly nitrate-N load during the validation period, 1997-2003. (Calibration period not shown.)

Nitrate from the tile drains

The nitrate concentration in the four soil layers for a representative corn/soybean rotation HRU is shown in Figure 3 for one year. The nitrate concentration varied widely in Layer 1, the top 10 mm of soil. In Layer 2, the nitrate concentration was more stable. Layer 3 reached a depth of 1240 mm, and the nitrate concentration in this layer was assumed to be that in the tile drains. Nitrate concentrations averaged 3.2 mg/l when the tile drains were flowing, which is lower than the typical nitrate concentration in tiles in Indiana (e.g., Kladivko et al., 2004). Further study is needed to refine nitrate concentration estimates in each soil layer. Layer 4, located just above the impermeable layer, had a higher nitrate concentration.

The objective of this study was to estimate the nitrate input to Sugar Creek through tile drains in comparison with nitrate inputs through the other flow paths (surface flow, percolation and lateral flow). Based on the simulation described here, the median percentage ranged from 0% to 65% over the 12 months, with more than 40% of the nitrate input estimated to flow through tile drains in April, May and June (Figure 4). These estimates do not take into account the in-stream processes that can modify the nitrate concentrations prior to reaching the watershed outlet. Further research is also needed to explore the effect on the relative importance of tile drains and other flowpaths resulting from changing the depth to the impermeable layer. Because of its function in controlling deep percolation, this parameter has a critical role in influencing the estimated nitrate loss through tile drains.

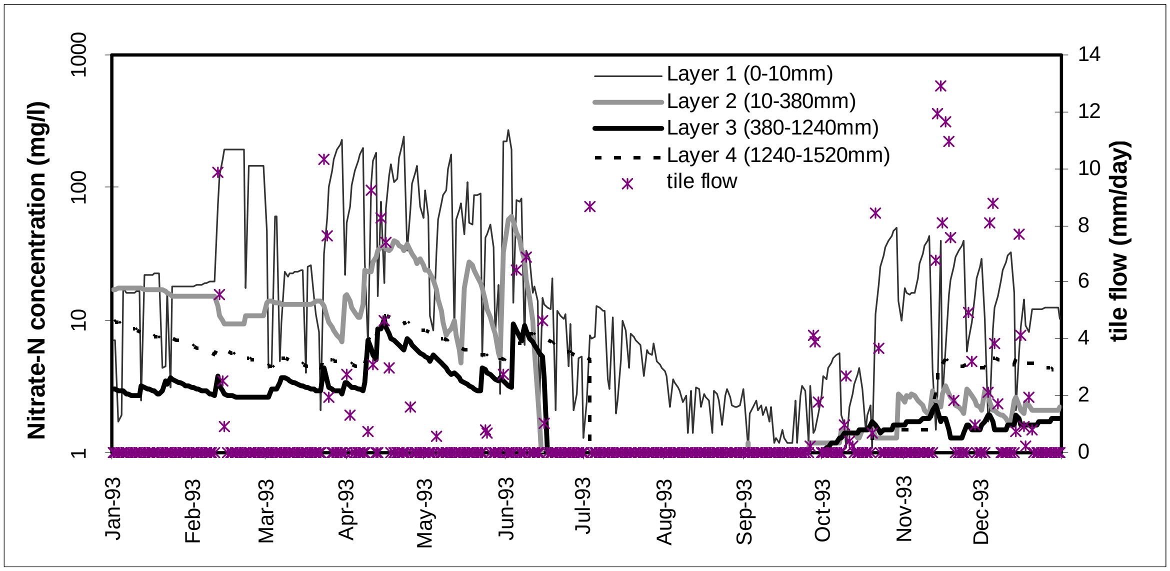


Figure 3. Nitrate-N concentration in the four soil layers and tile flow volume for a typical row crop HRU in one year

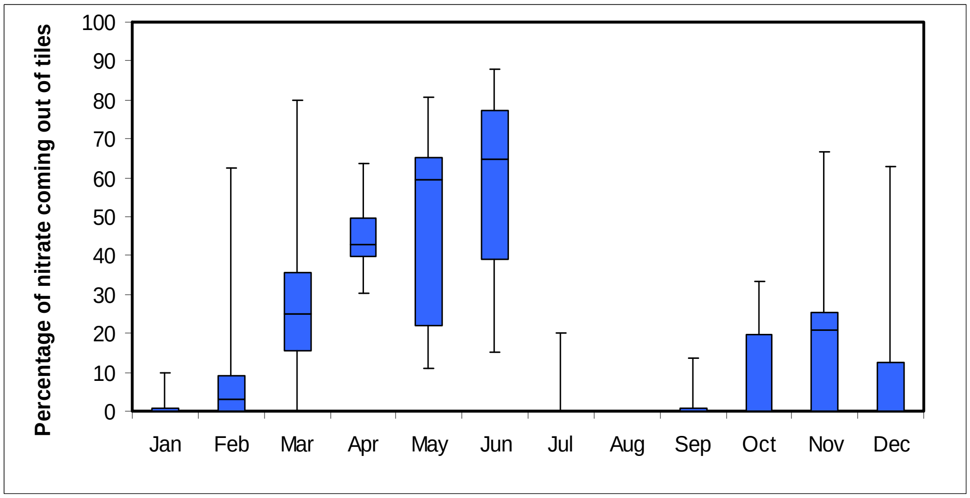


Figure 4: Estimated percentage of nitrate-N load entering Sugar Creek that flows through tiles, based on simulations from 5/1992 to 9/2003 (Black bar represents the median; boxes represent the 25th and 75th percentile.)

Summary and Conclusion

SWAT2005, with modified tile drain components, was used to study nitrate loss through tile drains in a heavily tile drained watershed in Indiana. Parameters controlling the nitrogen cycle were evaluated, and the model was found to be the most sensitive to CMN, SDNCO, NPERCO, and BC1 in the Sugar Creek watershed. Nash-Sutcliffe Efficiency values of the simulated monthly flows were 0.81 to 0.90, and for nitrate-N loads were 0.61 to 0.63. The source code was modified to output nitrate-N from tile drains, assuming that nitrate-N concentration in tile drains is the same as the mobile soil water in the layer in which the drains are located. The estimated median percentage of monthly nitrate input to Sugar Creek that came from tile drains ranged from 0% to 65% over the 12 months, and was more than 40% from April to June. These results could be used as a first step in assessing the potential of reduction in nitrate entering Sugar Creek that can be expected from implementation of conservation practices that reduce nitrate loss from tile drains, such as drainage water management, bioreactors, or wetlands at drainage outlets. The effectiveness of such practices is usually determined as a percentage of the nitrate from the drains, so estimates of the watershed nitrate load originating from tile drains will help in assessing overall nitrate reduction potential.

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Distributed model structures in catchment scale modeling

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Abstract

A prerequisite in the application of eco-hydrological models in the investigation of landuse and climate change is that controlling features in the catchment under study are well captured by the models. If the original model structure is not able to represent the dominant processes either the structure needs to be changed or a different model selected. We applied three different model structures of SWAT (original SWAT, SWAT-G and a new distributed version of SWAT-G) to the mesoscale catchment of the Wetter, Germany. SWAT-G was recently developed to improve model predictions for catchments that are mainly characterized by lateral flow components. The distributed version of SWAT-G now takes into account the various characteristics of sub-catchments, as it distinguishes between sub-catchments with either a dominance in lateral or base flow. All models were applied under a Monte Carlo framework to also investigate their parameter uncertainty. The analysis showed that the distributed version of SWAT-G provided the best results of all three structures compared to measured discharge, thereby reflecting the dominating hydrological processes and the expert's understanding of the catchment.

Keywords: lateral flow, Monte Carlo framework, parameter uncertainty

Introduction

Physically based, distributed models are often used to investigate the impacts of land use change or climate change on water resources and matter fluxes. A fundamental, though difficult to establish, feature of these models is that the processes in the catchment are well represented by the algorithms in the model. In some cases, the original model structure is not able to represent the dominating process in the catchment. Hence, the model structure needs to be revised and adapted to the catchment. State-of-the-art directions in watershed modeling follow a modularized approach, where various processes are switched on and off depending on the spatial characteristics of the watershed. The decision which module (respectively process) is used in a watershed can be made either on the basis of hard data and calibration, but mostly it depends on the expert's knowledge of the area and respectively on his or her perceptual model (Beven, 2001). One of the challenges of distributed modeling in large catchments is that the landscape and process heterogeneity commonly increase with increasing catchment size, so that an increased number of individual runoff processes may need to be captured.

In a first attempt to simulate the hydrological processes of the heterogeneous Wetter catchment, Germany, we applied the SWAT model. However, even the use of sophisticated automatic calibration routines did not result in satisfactory results. It could be shown, that this was mainly due to an incomplete description of run off generating processes in several parts of the catchments. The model failed to reflect the lateral flow conditions in those parts of the catchment that are characterized by shallow rocky aquifers. Applying SWAT-G (Eckhardt et al., 2002) a model version that explicitly considers anisotropy and thereby lateral flow processes for defined soil types did not really increase the models credibility as the available soil information did not allow for a detailed analysis of where exactly anisotropy occurs. Without refining the model structure we did not assume that a satisfactory process description of the dominant hydrological processes was achievable. Hence, the aim of this study was to implement a robust and simple approach that reflects the expert's understanding of the spatial distribution of dominant hydrological processes, here lateral flow dominance in the low mountainous parts of the catchment and base flow dominance in the flat regions.

Study Area

The model experiment was conducted in the 514 km² Wetter catchment, Germany. The landscape of this catchment is rather patchy but can be generally divided into two major sections. One part is characterized by a low mountain range with shallow soils over bedrock and steep slopes. This part consists mainly of the western catchment of the Usa (184 km²). The second part towards the east is represented by deep, loess-born soils with shallow slopes and the characteristics of a low land. Both landscape elements are dominated by different runoff generation processes. In the low mountainous part lateral subsurface stormflow appears to be the dominant runoff generation process (Figure1, process type I) whereas in the lowland part infiltration to groundwater aquifers and baseflow contribute mainly to runoff (Figure 1, process type II). The catchment is covered by 40% cropland, around 15% pasture, 34 % mixed forest and 11% urban respectively residential area, determined from ATKIS-Data from Hesse. Soil data were derived from the digital Hesse soil map 1:50000 (HLUG, 2002). The Catchment was divided into 33 subbasins and 585 HRUs.

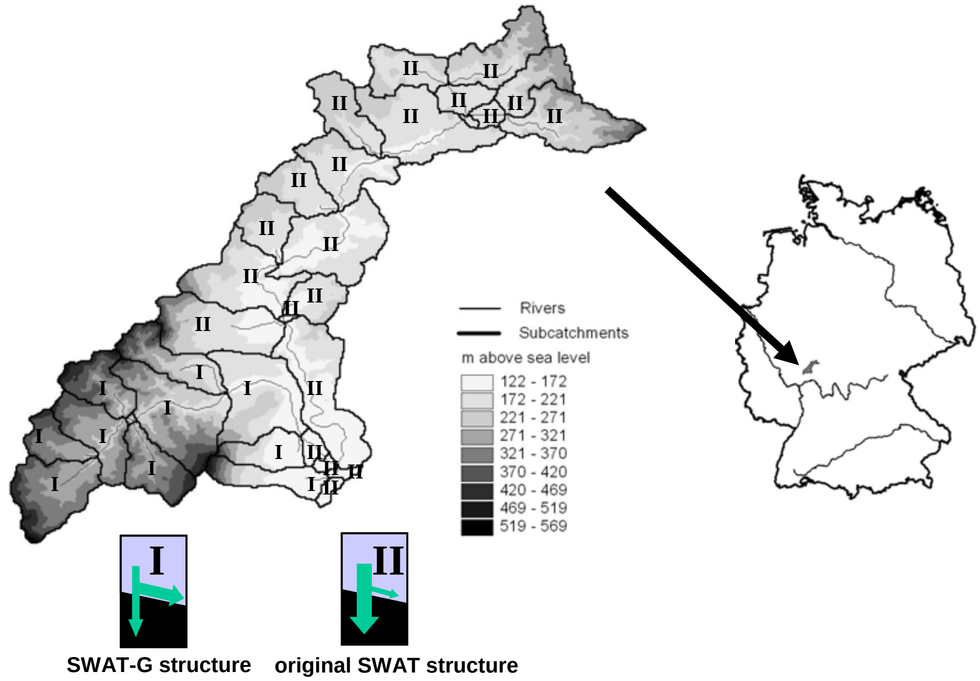

Figure 1. The Wetter catchment with its subcatchments and the dominating runoff generation processes subsurface stormflow (I) and baseflow (II)

Refining the model structure

Eckhardt et al. (2002) showed that the original SWAT structure (Arnold et al. 1998) was not able to capture the above mentioned runoff generation processes in a low mountain range. This led to the modification called SWAT-G, where the concept of anisotropy (Kleber, 1998) was implemented for certain soil types. Anisotropy differs between vertical and horizontal saturated hydraulic conductivity to account for the strong tendency for lateral flow which is typical for many low mountainous regions.

As in our case study, a disadvantage of the SWAT-G concept is that anisotropy is defined through soil type. In many instances soil digital information are limited, or, soil types across different topographical locations and geological parent material are summarized under the same soil type for the ease of simplification (e.g. “cambisols” in croplands of the Wetter low land and on pasture in the hilly mountainous areas). In both cases the necessary detail to describe the spatially distributed occurrence of anisotropy by soil type is not possible.

Hence, we changed the SWAT-G model structure. In the new concept, the decision of the occurrence of anisotropy is made on the subbasin-level. Both model structures, original SWAT and SWAT-G are now applied in a single SWAT run. As can be seen in Figure 1 there are catchments where the original SWAT approach is used and subcatchments where SWAT-G is switched on. To accomplish this, we changed the model code and introduced a new parameter in the *.sub-files, that defines the presence or absence of anisotropic flows.

In case of the Wetter catchment the decision to include or exclude anisotropic flows was made on the basis of the DEM, mapped soil depths, as well as expert’s knowledge. Steep slopes and shallow soils, which can be identified from these data, promote the development of lateral subsurface flows. In this case, these regions were chosen for the SWAT-G approach, otherwise the original SWAT approach was used.

Model calibration

All three model structures of SWAT were calibrated for a period of 3 years, by using the first year as warm-up period. For all SWAT-Runs the Hargreaves evapotranspiration scheme was implemented. During the calibration process, 25 parameters for the original SWAT model and 26 parameters (including an anisotropy parameter) for the SWAT-G and the distributed version of SWAT-G were changed within the parameters boundaries shown in Table 1. We used a Monte Carlo Framework for randomly drawing the parameter values from a uniform distribution. Approximately 7,500 runs for each model structure were developed, from which we compared the most highly efficient between model structures. These results provide significant insights into the three model structures, and their ability to capture measured dynamics The highest Nash-Sutcliffe-Efficiency (NSE) for the new structure was 0.65, for the SWAT-G structure 0.52 and for the original SWAT-structure it was 0.42 (Table 1). These results are consistent with our conceptual understanding of catchment dynamics, and suggest that the combined structure is more appropriate for the Wetter catchment.

Table 13. Upper and lower bounds of the parameter set used in the Monte Carlo framework.

model parameter	lower bound	upper bound
Snowfall temperature (SFTMP)	-1	2
Snow melt base temperature (SMTMP)	-1	2
Melt factor for snow on June 21 (SMFMX)	4	7
Melt factor for snow on Dec 21 (SMFMN)	1.4	4
Snow pack temperature lag factor (TIMP)	0.01	1
Surface runoff lag time (SURLAG)	1	4
Manning‘s n value for main channel	0.014	0.3
Baseflow alpha factor (ALPHA_BF)	0.01	0.07
Groundwater delay (GW_DELAY)	1	25
Groundwater „revap“ coefficient (GW_REVAP)	0.3	0.5
Deep aquifer percolation fraction (RCHRG_DP)	0.01	0.8
Lateral flow travel time (LAT_TIME)	1	4
Manning‘s „n“ value for overland flow (OV_N)	0.2	0.5
Bulk density (SOL_BD)	1	1.4
Available water content (SOL_AWC)	0.1	0.7
Saturated hydraulic conductivity (SOL_K)	10	250
Anisotropy factor (ANISO)	2	10
Albedo for landuse pasture (SOLALB)	0.2	0.3
Albedo for landuse mixed forest (SOLALB)	0.1	0.17

Albedo for landuse urban (SOLALB)	0.2	0.3
Maximum canopy storage for landuse pasture (CANMX)	2	7
Maximum canopy storage for landuse mixed forest (CANMX)	4	30
Maximum canopy storage for landuse urban (CANMX)	0.01	0.12
Curve number, general for landuse pasture (CN2)	45	60
Curve number, general for landuse mixed forest (CN2)	38	60
Curve number, general for landuse urban (CN2)	85	98

Model verification

To show the different behavior of all three model structures we sorted the model runs by NSE beginning from highest descending to lowest and plotted them against the number of runs (Figure 2). The new distributed version of SWAT-G has reached the highest efficiency and the most behavioral model runs. If you consider zero as a threshold between acceptable and unacceptable model runs, one can see that the new structure was able to produce a large set of acceptable model runs as compared to the other model structures. We therefore conclude, that the new model structure is superior in capturing the main run-off generating process in the region.

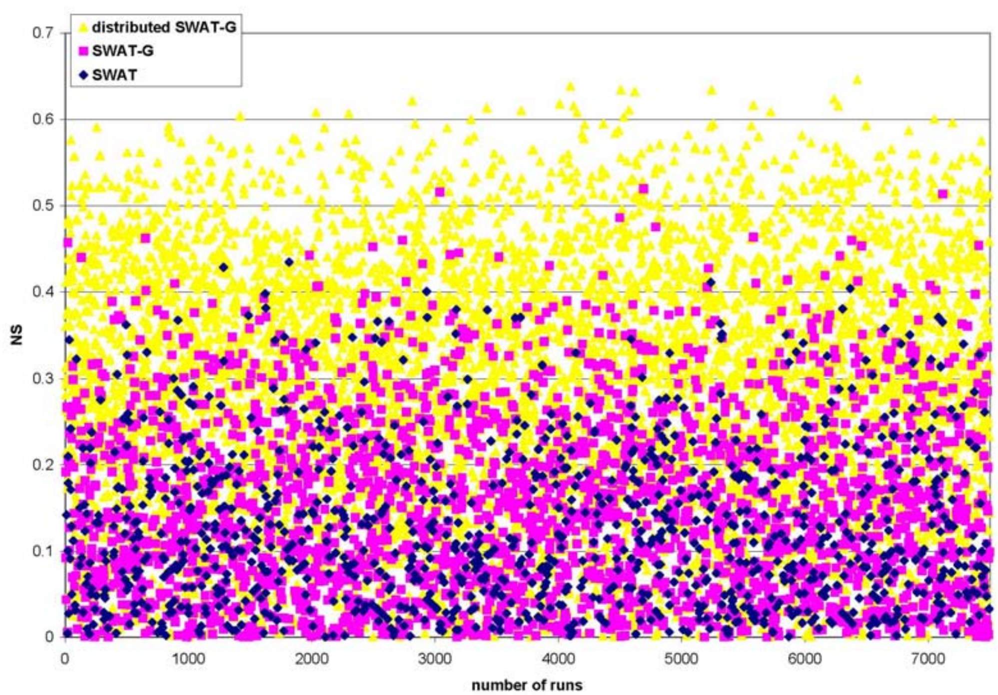

Figure 2. Number of model runs with a Nash-Sutcliffe-Efficiency >0 for SWAT, SWAT-G and the distributed version of SWAT-G

A more detailed analysis of the parameter distribution and the underlying parameter uncertainty for the three selected model structures is depicted in Figure 3, where scatter plots of four parameters impacting baseflow and subsurface stormflow are plotted against the NSE. A threshold criteria of 0.3 for NSE was selected as a (subjective) criteria for acceptable model performance. As can be seen, the distributed version of SWAT-G outperformed both other model structures across the original range of acceptable parameter values. Anisotropy (figure 3d) and the baseflow (figure 3c) factor are not well-constrained, but rather scattered within the boundaries of the parameter space. This indicates that there are not well defined parameter values given the current model structure. For the available water (figure 3a) content a drop of NSE in the center of the parameter space appears for the distributed version of SWAT-G. This apparent bimodality indicates that local preferable parameter spaces exists for the two different runoff generation processes I and II with values around 0.1-0.2 and 0.5-0.7, respectively. However, the mechanisms behind this behavior remain unclear and a more detailed analysis is needed for a more complete explanation. If valid, a distributed value for AWC might further improve the model structure. The only confined parameter from the selected four is saturated conductivity that tends to give better simulations for values >160 mm for both SWAT-G model structures.

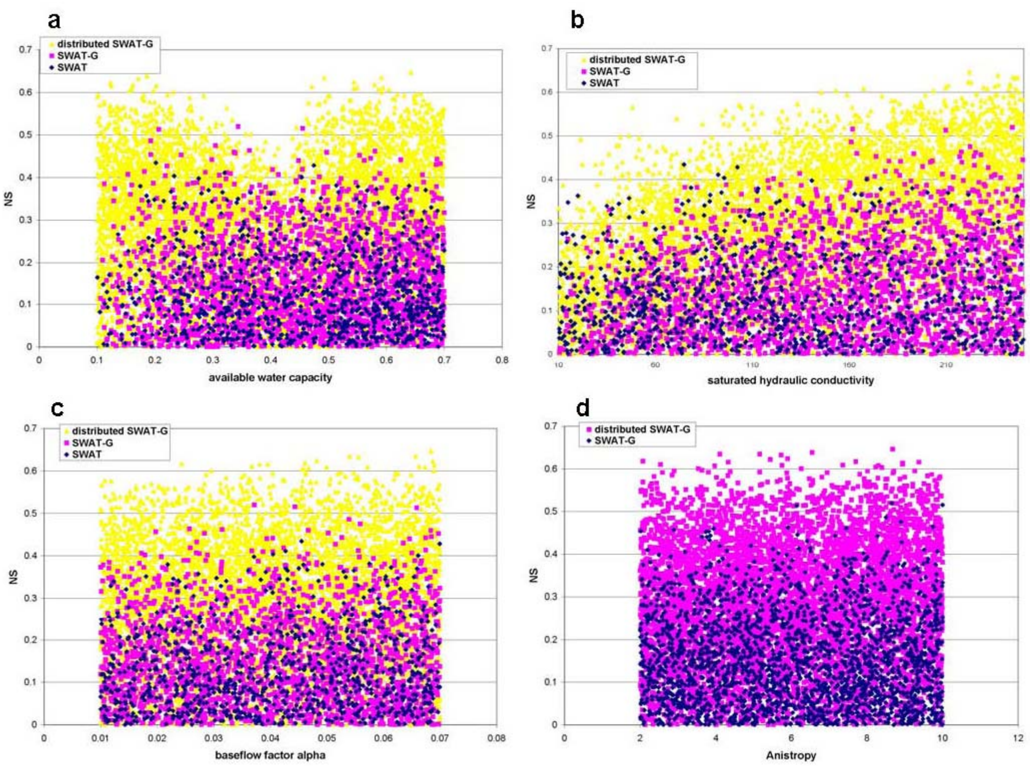

Figure 5. available water capacity (a), saturated hydraulic conductivity (b), baseflow factor alpha (c), anisotropy factor (d) plotted against Nash-Sutcliffe efficiency

As one can see in Figure 4 from all structures the distributed SWAT-G-model could represent the highflow and peak events best. These events are extremely affected by the confluence of the Usa, with the dominating runoff process type I (Figure 1). This subcatchment reacts quickly to rainstorm events because of its typical shallow soils overlaying hardrock. Here, infiltration to recharge the groundwater aquifers is limited, so lateral subsurface stormflow is the main runoff generation process.

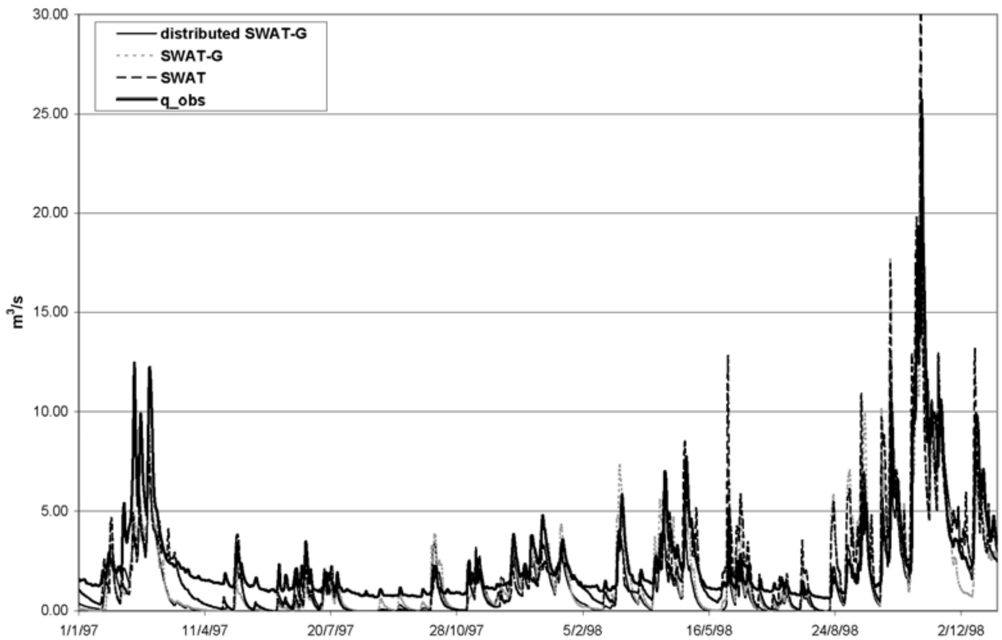

Figure 4. Measured and modeled daily discharge for three SWAT-structures (original SWAT, SWAT-G, and a distributed version of SWAT-G) in the Wetter catchment

What can also be depicted in Figure 4 is, that all selected model structures fail to represent the baseflow periods. This can be partly explained by the selection of NSE as the objective function in the Monte Carlo framework, as it puts the weight on peak events rather than baseflow conditions. Following a multi-objective approach considering also for example logarithmized discharge to account for low flow conditions within the Monte Carlo framework might improve this.

Though we want to point out that this modeling experiment was first of all aiming at a better spatial distribution of rainfall generating processes in the course of model application and therefore focused on peak discharges. Future work will focus on an improved description of the recession of the hydrographs. One should also be aware that many of parameters used in this Monte Carlo framework did not show a clear confined pattern. This points to a general high equifinality (Beven, 2001) for all the model structures investigated in this study.

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Modification of SWAT to Simulate Saturation Excess Runoff

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ABSTRACT

The variable source area concept of surface runoff generation is applicable in many catchments in southern Australia, where the widespread presence of duplex soils leads to the development of an ephemeral, perched water table. The intersection of the perched water table with the ground surface leads to saturation excess runoff being produced. The Soil and Water Assessment Tool (SWAT) computes surface runoff using the SCS curve number method which does not account for saturation excess runoff generated from variable source areas. To overcome this limitation, the kinematic storage model, which is embedded in SWAT to simulate subsurface flow, was extended to also simulate saturation excess runoff. This paper describes the extension of the kinematic storage model in SWAT to simulate saturation excess runoff. Results from the application of the modified model, referred to as SWAT-KSM, to a catchment in southern Australia are reported.

INTRODUCTION

The generation of surface runoff in many catchments across the world cannot be explained adequately by infiltration excess runoff because the infiltration capacity of the top layer of soil is usually much greater than the peak rainfall intensity of most flood-producing events. Research has shown that the variable source area concept is a suitable mechanism to explain the generation of surface runoff in a number of catchments located in different regions (Dunne, 1983). Variable source areas refer to the saturated portions of a catchment. Rain that falls directly onto variable source areas is readily transformed into surface runoff. Runoff produced by this process is commonly referred to as saturation excess runoff. The temporal and spatial extent of variable source areas is dynamic because they expand and contract on a seasonal basis as well as during storm events.

The variable source area concept of surface runoff generation is particularly relevant to a significant proportion of temperate and semi-arid Australian catchments where the presence of duplex soils leads to the development of an ephemeral, perched water table at shallow depth (Raper and Kuczera, 1991). Duplex soils consist of a thin layer of permeable soil (A horizon) overlying a relatively impermeable layer of soil at a shallow depth (B horizon). Duplex is a term that is widely used in Australia to describe soils which have an abrupt textural contrast between the A and B horizons. Soils that are characterized by a strong texture contrast between surface and subsurface horizons are found in other regions of the world (Chittleborough, 1992) but they are referred to by different terminology. The lack of vertical capacity in the B horizon of duplex soils causes water to pond above the boundary between the A and B horizons, which leads to waterlogging and the development of an ephemeral, perched water table (Cox and McFarlane, 1995). The intersection of the perched water table with the ground surface gives rise to the formation of variable source areas, from which saturation excess runoff is produced.

Given the prevalence of duplex soils in catchments across much of southern Australia, it is imperative that hydrologic models be able to account for the development of a perched water table and the role it plays in the generation of surface runoff when applied in this region. Models that account for runoff generated from variable sources areas are needed to ensure that the hydrologic response of a catchment is predicted accurately and to provide a rational interpretation of the origin and movement of nonpoint source pollution within a catchment (Ormsbee and Khan, 1989; Endreny and Wood, 1999). SWAT uses the SCS curve number method (USDA-SCS, 1972) to compute surface runoff. The curve number method is an empirical formulation that is unable to distinguish between infiltration excess runoff and saturation excess runoff (Endreny and Wood, 1999). The volume of runoff determined by the curve number method is assumed to be generated uniformly over the entire extent of the catchment area (Gburek et al., 2002) which is not consistent with the variable source area concept of runoff generation.

Despite a long-standing recognition of the importance of simulating saturation excess runoff in hydrologic models, relatively few methods have been developed to date that allow saturation excess runoff to be predicted in a simple and practical manner (Lyon et al., 2004). One hydrologic model developed in Australia to specifically account for the development of an ephemeral, perched water table in duplex soils is CATPRO (Raper and Kuczera, 1991). CATPRO conceptualizes the top hillslope layer as a narrow inclined cuboid, in which the phreatic surface of the perched water table is assumed to be flat (Kuczera and Mroczkowski, 1998). The intersection of the water table with the ground surface leads to the formation of a saturated length. Rain that falls directly onto the saturated length is transformed into surface runoff, while the remaining rain is assumed to infiltrate into the soil profile without any losses. Mroczkowski et al. (1996) reported that the hillslope conceptualization of CATPRO is in fact a generalization of the kinematic storage model.

The kinematic storage model is already embedded in SWAT to simulate subsurface flow. The kinematic storage model was originally developed by Sloan et al. (1983) to predict subsurface flow. However, they acknowledged that the original formulation "does not allow for surface runoff where the saturated zone reaches the surface." They went on to report that "surface runoff is easily accounted for in this model" and subsequently modified the model to account for saturation excess runoff. Extending the kinematic storage model in SWAT was regarded as a simple yet effective approach that would enable saturation excess runoff to be simulated in a manner that was consistent with the variable source area concept.

EXTENSION OF KINEMATIC STORAGE MODEL IN SWAT

Although the kinematic storage model has been incorporated into a number of hydrologic models, it is only used for the purpose of simulating subsurface flow. The one exception is the study of Ormsbee and Khan (1989), in which they embedded the kinematic storage model into HEC-1 (U.S. Army Corps of Engineers, 1985) to simulate both subsurface flow and saturation excess runoff.

The kinematic storage model is based on the mass continuity equation with the entire hillslope segment being the control volume. The idealized hillslope segment, represented as a rectangular storage element, has a permeable soil layer of constant depth *D* and an impermeable layer below of length *L*. The hillslope segment is oriented at an angle α to the horizontal. According to the kinematic wave assumption, the hydraulic gradient is assumed to be equal to the bed slope (Figure 1).

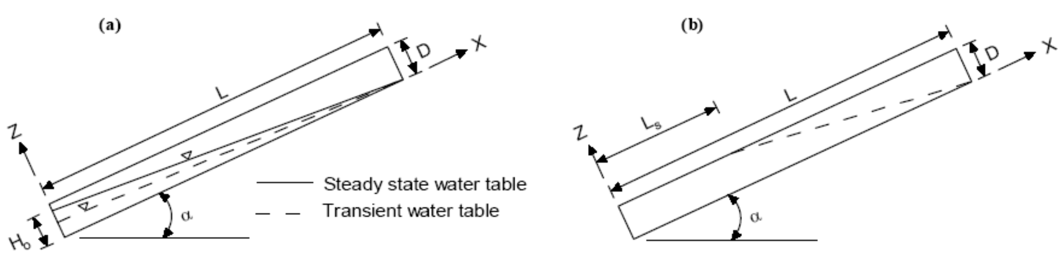

Figure 1. The kinematic storage model: (a) without saturation excess runoff (Sloan and Moore, 1984); and (b) with saturation excess runoff (Ormsbee and Khan, 1989).

The mass continuity equation for the saturated storage element may be expressed in mixed finite form as:

$$\frac{S_2 - S_1}{t_2 - t_1} = iL - \frac{(q_1 + q_2)}{2} \quad (1)$$

where S is the drainable volume of water stored in the saturated zone per unit width (mm^2), t is time (d), i is the vertical input rate to the saturated zone (mm/d), L is the hillslope length (mm), q is the discharge from the profile per unit width (mm^2/d) and the subscripts 1 and 2 denote the beginning and end of the time step respectively. The drainable volume of water stored in the saturated zone of the hillslope is found by:

$$S = \frac{H_o \theta_d L}{2} \quad (2)$$

where H_o is the depth of the water table at the outlet (mm) and θ_d is the drainable porosity of the soil (mm/mm). The discharge at the outlet is given by Darcy's Law:

$$q = H_o v \quad (3)$$

where v is the velocity of flow at the outlet (mm/d). The velocity is defined as:

$$v = K_{sat} \sin(\alpha) \quad (4)$$

where K_{sat} is the saturated hydraulic conductivity (mm/d). Substituting equations 2 and 3 into equation 1 allows the water table depth at the outlet at the end of the time step to be expressed explicitly as:

$$H_{o2} = \frac{H_{o1}(L\theta_d - v\Delta t) + 2Li\Delta t}{L\theta_d + v\Delta t} \quad (5)$$

Sloan and Moore (1984) showed that when the water table intersects the ground surface, equations 2 and 3 become:

$$S = \frac{D\theta_d (L + L_s)}{2} \quad (6)$$

$$q = iL_s + Dv \quad (7)$$

where L_s is the saturated slope length (mm) and D is the depth of the soil layer (mm). By substituting equations 6 and 7 into equation 1, the saturated slope length at the end of the time step can be determined explicitly:

$$L_{s2} = \frac{L_{s1}\left(D\theta_d / \Delta t - i\right) + 2iL - 2Dv}{D\theta_d / \Delta t + i} \qquad (8)$$

Once the saturated slope length is known, the discharge can be calculated via equation 7. Rain that falls directly onto the saturated slope length is transformed into surface runoff.

A complete description of the kinematic storage model can be found in Sloan et al. (1983) and Ormsbee and Khan (1989). These references provide a full account of the saturated and unsaturated zones of the storage element as well as the extended model in which saturation excess runoff is considered. The modified version of SWAT is referred to as SWAT-KSM in this paper. It is important to point out that the original version of SWAT automatically adds a 10 mm soil layer to the top of the soil profile. This is because SWAT assumes that surface runoff interacts with the top 10 mm of soil and that nutrients contained in this layer are available for transport to the main channel in surface runoff. However, this feature had to be deactivated in SWAT-KSM. Allowing the model to set this very thin soil layer as the top layer of the soil profile would lead to unrealistic values being calculated for the saturated slope length. The removal of the 10 mm layer of soil affects the simulation of nutrient transport and therefore the model cannot be used for this purpose until further modifications are carried out.

To account for the saturated region in the vicinity of the stream owing to groundwater flow from aquifers, Kuczera et al. (1993) incorporated a parameter, *wetfrac*, into CATPRO to represent the permanently saturated hillslope fraction. *wetfrac* has been incorporated into SWAT-KSM for the same purpose. The value of *wetfrac* is multiplied by L and then added to L_s. *wetfrac* is assumed to be constant over the entire year.

STUDY AREA

The Woady Yaloak River catchment is located in southern Australia. Streamflow is measured at two locations along the river: Pitfield (306 km^2) and Cressy (1157 km^2). The majority of land is used for agriculture with grazing livestock (beef cattle, sheep and prime lambs) and cereal crops (wheat, barely and oats) being the main commodities. Soils throughout the catchment are predominantly duplex. The climate of the region is distinctly Mediterranean, which is characterized by cold, wet winters and hot, dry summers. The mean annual rainfall decreases from 700 mm in the north of the catchment to 550 mm at the catchment outlet. The Woady Yaloak River catchment is a low-yielding catchment with ephemeral streams. The mean annual observed runoff at the Pitfield and Cressy gauging stations is 52.4 and 42.3 mm, respectively.

MODEL PERFORMANCE

SWAT-KSM was calibrated and validated for the periods 1978-1989 and 1990-2001, respectively. A long evaluation period was utilized in this study to determine whether the model was capable of capturing the dynamics of the long-term water balance of the Woady Yaloak River catchment. Adopting a short evaluation period would not test the model for the range of climatic conditions that the catchment is subject to over longer periods of time. The performance of the model for predicting runoff was assessed using the deviation of runoff volumes (D_V) and the coefficient of efficiency (E). The coefficient of efficiency was computed at annual, monthly and daily time scales. In addition, a visual comparison of the observed and predicted hydrographs at annual, monthly and daily time scales was performed.

RESULTS

Presented in Table 1 is the total observed and predicted runoff for the calibration and validation periods at Pitfield and Cressy. Also presented in Table 1 is the deviation of runoff volumes. It can be observed that the deviation of runoff volumes was less than 7% for the calibration and validation periods at both gauging stations. The validation period was considerably drier than the calibration period. This provides a stern test of model performance because when the model is independently tested using the data from the validation period, it is forced to extrapolate beyond the range of conditions encountered during the calibration period (Kuczera et al., 1993). Despite significantly less runoff being generated during the validation period, SWAT-KSM still managed to successfully reproduce the total runoff for this simulation period. This result indicates that SWAT-KSM was able to reproduce the total runoff for the Woady Yaloak River catchment reliably.

Table 1. Total observed and predicted runoff and deviation of runoff volumes (D_V).

Gauging station	**Simulation period**	**Observed (mm)**	**Predicted (mm)**	**D_V (%)**
Pitfield	Calibration	588.0	586.1	-0.3
	Validation	478.2	449.0	-6.1
Cressy	Calibration	475.3	487.8	2.6
	Validation	347.8	327.2	-5.9

The observed and predicted annual runoff is presented in Figures 2 and 3. It can be observed that SWAT-KSM reproduced the annual runoff at Pitfield and Cressy very well. It is apparent from the results presented in Figures 2 and 3 that there was considerable variability in the annual runoff at both gauging stations over the entire evaluation period. Despite the large fluctuations that occurred on an annual basis, SWAT-KSM managed to capture much of the variability at this time scale.

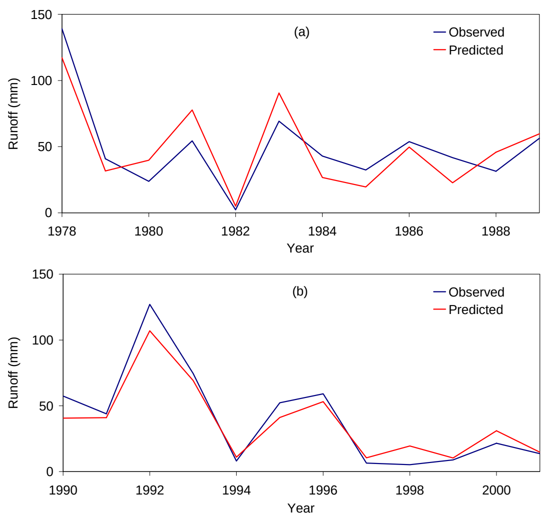

Figure 2. Observed and predicted annual runoff at Pitfield.

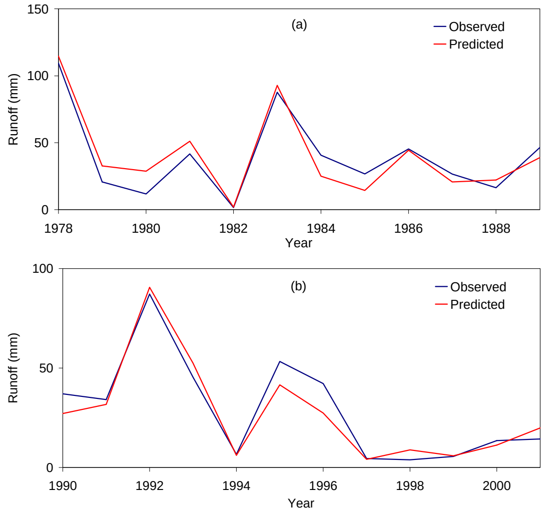

Figure 3. Observed and predicted annual runoff at Cressy.

The observed and predicted monthly runoff is presented in Figures 4 and 5. It can be observed that the monthly runoff fluctuated significantly at both gauging stations. The hydrographs on a monthly time step were characterised by sustained periods of low flow during summer and autumn and large quantities of runoff being generated in winter and spring for most years. The results indicate that the agreement between the observed and predicted monthly runoff at Pitfield and Cressy was very good. SWAT-KSM reproduced the strong seasonal variation in monthly runoff very well, although it did have a tendency to underestimate high monthly flows.

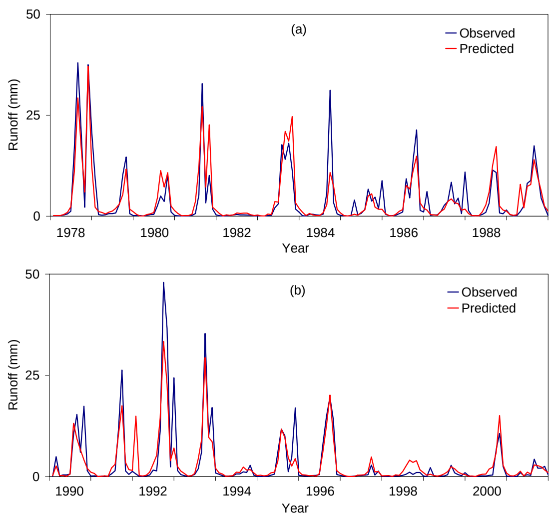

Figure 4. Observed and predicted monthly runoff at Pitfield.

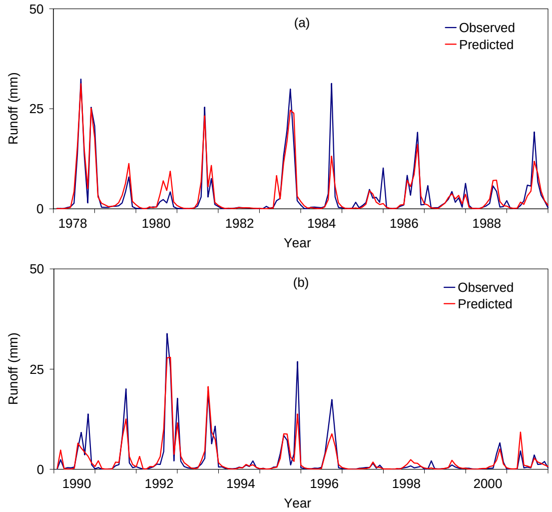

Figure 5. Observed and predicted monthly runoff at Cressy.

Due to space limitations, it is not feasible to present all the daily hydrographs in the main text below. Therefore, daily hydrographs for two years are presented for discussion. The results on a daily time step were mixed. The prediction of daily runoff for some years was relatively good while for other years the results were considerably poorer. To provide as much of an unbiased view as possible, years for which good and poor results were achieved are provided below. The observed and predicted daily runoff at Pitfield for 1993 is presented in Figure 6a while the observed and predicted daily runoff at Cressy for 1985 is presented in Figure 6b. The prediction of daily runoff by SWAT-KSM at Pitfield for 1993 was relatively good. However, it is evident that the daily runoff at Cressy for 1985 was not reproduced with the same degree of accuracy. The prediction of daily runoff at both gauging stations was found to be satisfactory overall, although the model clearly performed better for some years than others. SWAT-KSM had a tendency to underestimate peak flows for a number of years. This can be clearly seen in Figure 6b.

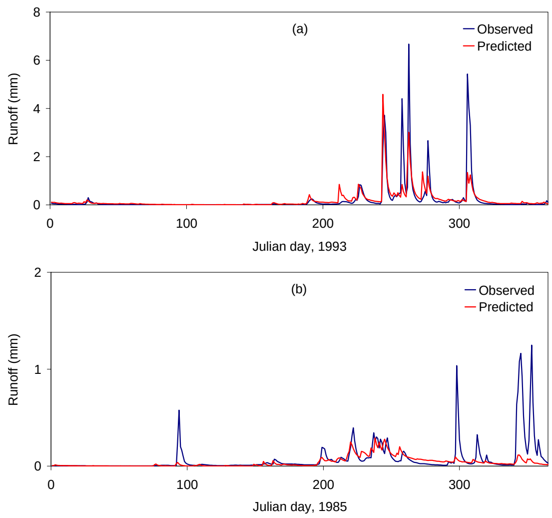

Figure 6. Observed and predicted daily runoff at (a) Pitfield for 1993; and (b) Cressy for 1985.

Presented in Table 2 are the coefficients of efficiency (*E*) achieved by SWAT-KSM for the prediction of runoff at annual, monthly and daily time scales. The performance of the model for predicting annual runoff at Cressy was excellent with the coefficients of efficiency for the calibration and validation periods being 0.90 and 0.92, respectively. The annual coefficient of efficiency at Pitfield for the calibration period was 0.77. For the validation period, this value increased to 0.92. The monthly coefficients of efficiency ranged from 0.77 to 0.85 at both gauging stations. These values indicate that the performance of the model for predicting monthly runoff was very good. The daily coefficients of efficiency varied between 0.45 and 0.56, which indicates that the performance of SWAT-KSM for predicting runoff on a daily time step at both gauging stations was satisfactory.

Table 2. Coefficients of efficiency (*E*) at Pitfield and Cressy.

Gauging station	**Simulation period**	***E***		
		Annual	**Monthly**	**Daily**
Pitfield	Calibration	0.77	0.77	0.54
	Validation	0.92	0.78	0.45
Cressy	Calibration	0.90	0.85	0.56
	Validation	0.92	0.84	0.46

DISCUSSION

The performance of SWAT for predicting annual and monthly runoff at Pitfield and Cressy was very good. The prediction of daily runoff at both gauging stations was found to be satisfactory overall, although the model clearly performed better for some years than others. It is important to point out that SWAT was developed to predict long-term yields and not detailed, single-event flood routing (Arnold and Williams, 1995). Model users are usually more interested in monthly runoff predictions instead of daily runoff predictions for the analysis of catchment water yield (Boughton, 2005). Therefore, it is not considered critical that the model reproduce the exact structure of the daily hydrographs as would be required in event-based modelling (Arnold and Williams, 1995). Motovilov et al. (1999) reported that according to common practice the performance of a hydrologic model is considered good for values greater than 0.75 and satisfactory for values between 0.75 and 0.36. Given that the daily coefficients of efficiency ranged from 0.45 to 0.56 in this application of SWAT-KSM, the performance of the model for predicting daily runoff was deemed to be sufficient.

In general, SWAT did not reproduce the peak flows very well at either gauging station. There a several reasons for this. Firstly, the underestimation of certain peak flows can be partly attributed to the amount of rain falling on the catchment at the time of these events being underestimated due to the poor coverage of rainfall stations across the study area. Secondly, it is likely that some peak flows were underestimated because the saturated portion of the catchment was too small for enough rainfall to be transformed to surface runoff. That is, the perched water table had not produced a saturated length that was sufficiently long for enough rainfall to be transformed to surface runoff. This problem could be overcome by changing the parameters that have the most influence on the soil water content of the A horizon including the soil depth, hillslope length, available water capacity and bulk density. This option will be pursued in future work with the model.

When analysing the predictive capabilities of SWAT-KSM for the study area, it is important to keep in mind that it is a low-yielding catchment with ephemeral streams. It is well recognised that there are inherent difficulties associated with the prediction of runoff from low-yielding catchments. The relationship between rainfall and runoff in low-yielding catchments is highly nonlinear unlike in high-yielding catchments where nonlinear processes do not manifest themselves strongly in the streamflow observations (Ye et al., 1997). Furthermore, low-yielding catchments undergo more complex and a wider range of hydrological processes than high-yielding catchments (Gan et al., 1997). Taking this into consideration, the performance of SWAT-KSM for predicting runoff at Pitfield and Cressy can be considered relatively good.

CONCLUSIONS

SWAT is unable to account for saturation excess runoff generated from variable source areas because it utilizes the SCS curve number method to compute surface runoff. Extension of the kinematic storage model in SWAT provides a simple yet effective approach for simulating saturation excess runoff that is consistent with the variable source area concept. The performance of SWAT-KSM for predicting annual and monthly runoff was very good with the model managing to capture much of the variability in runoff exhibited at these time scales. The results for the prediction of daily runoff were satisfactory. Although the model had a tendency to underestimate peak flows this was not deemed to be a critical problem. It is important to point out that several factors had implications for the results achieved in this study including the reliance on a low density of rainfall stations to calibrate the model and the study area being a low-yielding catchment with ephemeral streams.

Overall, SWAT-KSM provided a relatively good description of the long-term water yield of the Woady Yaloak River catchment. This indicates that the extended kinematic storage model can provide an alternative approach to the curve number method for computing surface runoff in SWAT. Since the surface runoff component of SWAT-KSM is consistent with the dominant runoff generation mechanism observed in catchments dominated by duplex soils, unlike that of the original version of SWAT which employs the curve number method, it may be regarded as a more suitable model to utilise for predicting the impacts of future land use change on the water yield and water quality for the study area. Sloan and Moore (1984) reported that the kinematic storage model had sufficient features that would enable it to be incorporated into comprehensive catchment models, which would then place these models on a more rational, physically correct and less empirical footing. Replacement of the curve number method with the kinematic storage model to calculate surface runoff removes some of the empiricism currently associated with SWAT while at the same time lending greater physical basis to the model. Although SWAT-KSM was developed principally for catchments in southern Australia that are dominated by duplex soils, it can be applied to other catchments around the world where an impermeable soil layer gives rise to a perched water table.

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Modelling mitigation measures for pesticide pollution control using SWAT

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Abstract

Pesticides are useful for agriculture because of their ability to protect crops against damaging organisms. At the same time, excessive loading of pesticides in streams and water bodies can produce toxic conditions that harm sensitive aquatic species, and render the water unfit for human consumption. Therefore measures need to be designed, evaluated and undertaken in order to reduce the pesticide pollution. In this study we focus on the Nil catchment, a small basin situated in the centre of Belgium. For this catchment, a SWAT model was available to predict the impact of several reduction measures. In agricultural areas, pesticides can enter surface water as point losses or by diffuse pollution. For point source pollution, adaptations needed to be done with regard to the efficiency of applying pesticides directly on the field and the pesticide transport in rivers. This source of pollution is easily controllable by taking precautions, and the effect of reduced point losses was simulated by increasing the value of the application efficiency parameter (apef). For diffuse source pollution, five management scenarios, namely sowing of cover crops, contour farming, strip-cropping, conservation tillage and construction of grassed buffer strips; were simulated and compared to the initial situation. In order to simulate the first three practices, the model parameters Moisture Condition II Curve Number (CN2) and/or USLE support practice factor (USLE-P) were adapted. The functioning of a buffer zone along the river was predicted by adding a part in the source code describing sedimentation in the grassed strip, and by extending the processes in the infiltration module. For user-friendly application of the infiltration processes, some modifications are required with regard to the representation of the landscape in the model. In order to simulate in-stream measures, a recommendation of several adaptations is given for the pesticide routing module.

KEYWORDS: Management, pesticides, point losses, diffuse losses, river water quality, modelling, SWAT

Introduction

Agricultural water pollution is becoming a major concern not only in developed regions such as the European Union (EU) but also in the developing countries. The intensification of agricultural practices – in particular, the growing use of fertilisers and pesticides, and the specialization and concentration of crop and livestock production – has had an increasing impact on water quality. The main agricultural water pollutants are nitrates, phosphorus and pesticides (Scheierling, 1995).

Pesticides are chemicals which are used to protect crops against damaging organisms. As a result, farmers have always sought ways to reduce this damage, like by using pesticides. Although pesticides are useful for society, they are a lot of problems associated with their use. One problem which became evident in early years is that the pests developed resistance to the chemicals; this in turn devastated crops. Another problem was the effect of pesticides on non-target organisms, which were inadvertently exposed through the food chain. As cause of this problem most modern pesticides are much lesser persistent and do not accumulate in the food chain. The most widespread and well known problem is the occurrence of pesticides in surface and groundwater, leading to toxicity to aquatic organisms (Dressing et al., 2003).

Controlling water pollution from agricultural pesticide application is made difficult by its particular nature. In most circumstances, pesticide pollution occurs over a wide area, and its sources are diffuse and difficult to identify. Pollution control measures must rely heavily on approaches that affect farmers' land use and protection decisions (Scheierling, 1995). These pollution prevention farming methods are known as Best Management Practices (BMPs). The simplest way to limit pesticide pollution is to reduce or eliminate pesticide use. This can be accomplished through guided pest control, biological control and by means of an integrated approach. If pesticides should be used, the most effective approach is to release lesser quantity of and/or lesser toxic pesticides in the environment while handling and, second, to use practices that minimize the movement of pesticides to surface waters (i.e. erosion control practices and drift reducing measures).

Managing the use of pesticides needs however a better understanding of the behaviour of the pesticides, its effects on the environment and the prediction of the effectiviness of pesticide abatement measures. This can be achieved by computer modelling. In this view, several studies have been conducted (e.g. Witchell, 2005). None of the existing modelling tools foresees the processes that are involved in the study area, the Nil Catchment, that is a small catchment where pesticides are applied directly on the field. For this reason, the Soil and Water Assessment Tool had been chosen in this study because it allows for the necessary adaptations.

Case study

The Nil catchment (Fig. 1) is a small and hilly basin situated in the central part of Belgium, Southeast of the capital. The average elevation is 151 m a.s.l., with the highest top reaching 167 m a.s.l. and the watershed outlet lying at 110 m a.s.l. The catchment drains an area of 32 km², is 14 km long and has a retention time of about 1 day. Seven % of the area is inhabited and the main crops grown are winter wheat (22% of the catchment area), corn (15%) and sugar beet (10%). There are no drainpipes and no waste water treatment plants situated in the catchment. Further, the basin is characterised by a low base flow which results from its specific geological structure. Highly permeable Brusselian sands, showing hydraulic conductivities between 10-3 and 10-5 m s-1, lay above a less permeable rock (Abdeslam, 1998). Hereby, an important part of the groundwater of the catchment is drained to the adjacent river ‘Train’.

The Nil catchment was selected because it is a well documented basin, studied in detail in terms of pesticide application by the Centre for Research in Animal Health and Agro-chemistry (CODA) (Beernaerts et al., 2002). They performed inquiries during springs of 1998 until 2002, in which they asked the farmers to give as detailed information concerning the amount of pesticide they applied, the application dates, the kinds of pesticides they utilized for their different crops and the treated surface. 42 % of the farmers could give such kind of information (CODA, 2003). The use of pesticides by the farmers leads unavoidably to water pollution of the river the Nil. To gain insight in the occurrence of these pollutants in the different compartments of the river, the CODA performed a monitoring campaign during this period. Two years later, the Flemish Institute for Technological Research (VITO) performed more intensive measurements in the Nil during spring i.e. the application period of pesticides. Further, detailed information about the two monitoring campaigns can be found in Beernaerts et al. (2002) and Holvoet et al. (2006). Several pesticides were measured in these two campaigns from which the herbicide atrazine was chosen as study object. High concentrations of this pesticide, applied on corn fields in the Nil catchment, could be found after rainfall events due to runoff and even more important in the absence of rain due to drift and point losses. The maximum concentration detected in the river during spring of 2004 was 40 µg/L.

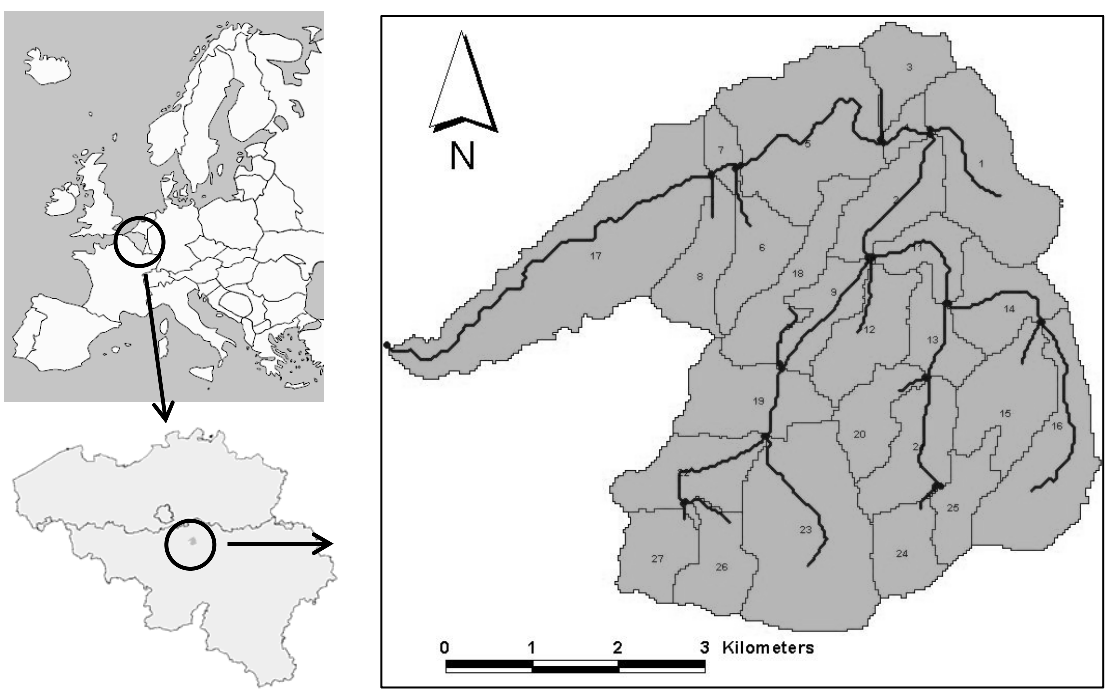

Figure 1. Situation of the Nil-catchment and sub-basin delineation automated by means of a DEM.

SWAT and the modifications

SWAT - the Soil and Water Assessment Tool- was developed by the USDA Agricultural Research Service (ARS) (Arnold et al., 1998) to predict the impact of land management practices on water, sediment and amount of chemicals originating from agriculture, in large complex river basins with varying soils, land use and management conditions over a long period of time. In order to simulate the pesticide processes in the Nil catchment, the following modifications were done to the model codes.

Pollution to the river

Pesticide water pollution occurs when a water body is adversely affected due to the addition of large amounts of pesticides to the water. They can be introduced into aquatic environment as point sources or by diffuse pollution. Holvoet et al. (2005) made a first distinction between direct (i.e. point losses and drift) and runoff losses. The runoff losses are simulated by the original SWAT source codes whereas extensions have been made for the point sources and drift, as occurring in the Nil catchment, and with the consideration of buffer strips.

(1) Point losses are caused by the cleaning of spray equipment on paved surfaces, the leakage of tools, spills, etc. Although these losses can simply be reduced by proper handling during pesticide use, they can contribute to 50-70% of the load of pesticides found in Belgian rivers like the Nil. Careful pesticide handling (e.g. avoiding of spills) and performing as many operations as possible on the field; are very effective pollution prevention measures. The point sources are simulated by using the original equation for application efficiency, with the change that the losses are now directed to the stream while originally, they disappeared from the system.

(2) Spray drift is defined as liquid drops formed by spray nozzles, that are carried out of the treated field by the wind; and there deposited. The calculation of these loadings was based on the German drift database (Ganzelmeier et al., 1995), in accordance to the recommendation of the FOCUS (FOrum for the Co-ordinaten of pesticide fate models and their USe) Surface Water workgroup. Performing a simple linear regression to these data, resulted in the following equation for arable cops:

$$\%drift = A \times z^B$$

where % drift is the percentage of drifted liquid on a certain distance z (m) from the latest crop row; and A and B are respectively the constant and exponential regression factor, dependent on the crop type and the growing stadium (De Schampheleire et al., 2005; FOCUS, 2001; Ganzelmeier et al., 1995).

Buffer strips

A more physically-based concept was aimed for the simulation of the bufferstrips, whereby a separation was made between solutes that undergo infiltration processes, and particular materials (suspended solids, adsorbed pollutants and particular organic matter) that undergoes settlement. Hereby, a distinction is made between the fractions of sand, loam and clay in relation to the top soil layer.

River routing

Due to previously mentioned modifications, significant pesticides loads are transported during low flow periods. This imposed more accurate calculations during low flow events what was proposed by van Griensven et al. (2006).

The model

Based on specific data and assumptions for the Nil catchment, the watershed was divided into 27 "hydrologically" connected sub-watersheds and 227 hydrologic response units (HRUs) (Holvoet et al., 2005). After making the necessary adaptations in the source code, described in section 3 and 4, the model could accurately predict the load of atrazine present in the river the Nil. In this study, we focus on the use of atrazine on corn during the growth season of 1998, when the application rate amounted 0.741 kg/ha. For this year the model predicts a total load of 2.515 kg at the mouth of the river. This initial atrazine load consists mainly of dissolved atrazine (2.376 kg).

In Fig. 2, the contribution of point, drift and runoff losses in the load of dissolved atrazine present in the river is represented for the mass coming from sub-basin 25 towards reach 25 during the application period of atrazine in 1998. Sub-basin 25 consists of many corn fields on which atrazine is applied. From Fig. 2, it is clear that point losses are the most important source of water pollution on application days. So, they need special attention in pesticide reduction strategies. Especially as point losses occur during low flow conditions, which can result in severe impacts on water ecosystems. Further, the fraction of drift towards the river system on those days amount between 1 and 2% of the dosage per unit area, which is a negligible amount compared to the contribution of point losses to the total pesticide load in the Nil. On rainy days, the figure (Fig. 1) shows clearly the importance of runoff as a transport route of pesticides towards the river.

Based on those results, measurements that can limit point losses and runoff flow will be of interest when we obtain to reduce atrazine fluxes towards the Nil. In this study, only these management strategies will be discussed, simulated with the SWAT model, and evaluated according to their environmental performance.

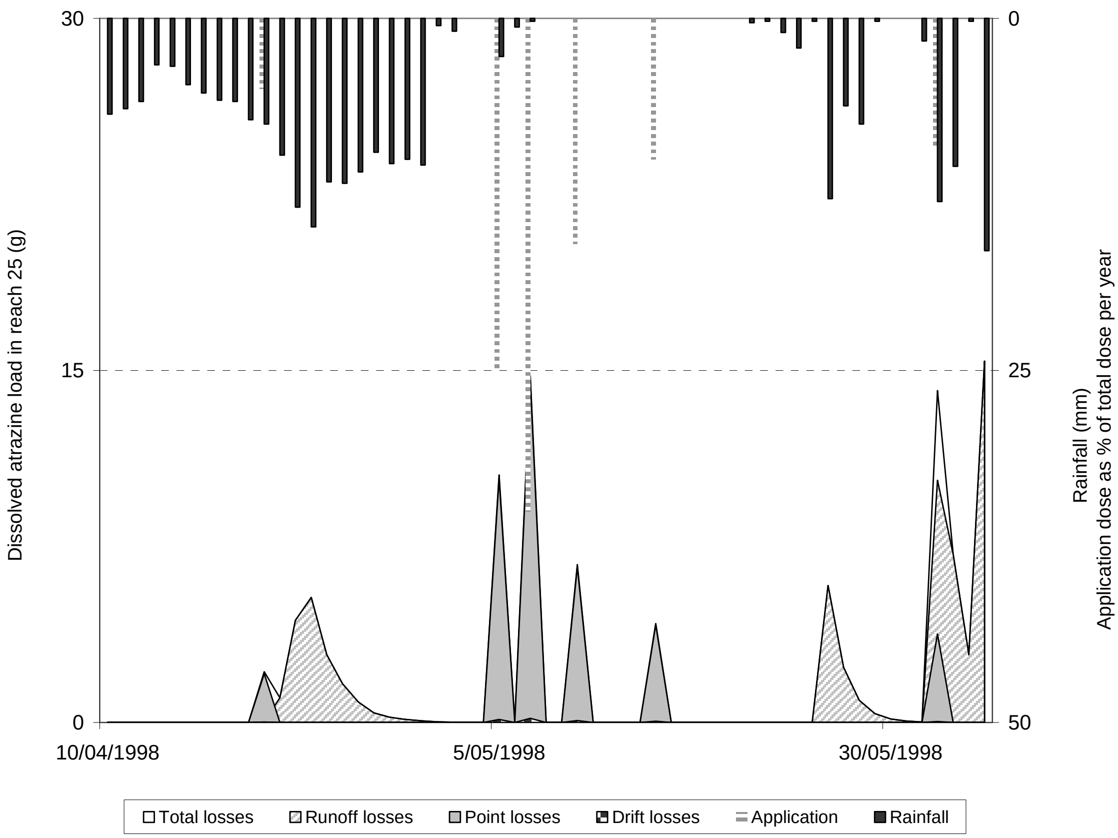

Figure 2. Predicted load of dissolved atrazine coming from sub-basin 25 during spring 1998, together with the measured rainfall and the application dose (showed on the secondary axis).

Modelling mitigation measures

Measures to reduce point pollution

A reduction of point losses can be simulated by setting the application efficiency parameter (apef) in the SWAT model on a higher value. This parameter stands for the fraction of the used pesticide dose that will be effectively applied on the agricultural field, and thus not be lost as point losses. The initial value for these parameter reached after calibration was 0.9985 (Holvoet et al., 2006b). During the simulations, it was assumed that the application efficiency parameter has a constant value over the simulation period. However, in reality there exists variability in time due to variability in farmers, in farmers' customs and daily differences.

A reduction of point source with 80% results in a decrease of the atrazine load with 21%. A reduction in point losses with 40% is estimated to result in almost 11% of load reduction in the river the Nil during the year 1998. It should be noticed that no conclusion could be given about the most efficient point source pollution prevention techniques. The application efficiency parameter (apef) covers the part of initial pesticide dose that is not lost by all types of point pollution.

So, it can be concluded that farmers can easily limit point source pollution by taking precaution measures, without relevant financial consequences. However, the measurements of the CODA in the Nil showed that a permanent sensitization of the farmers is necessary (Beernaerts et al., 2002).

Measures to reduce diffuse pollution

Several management practices can result in a reduction of pesticide fluxes towards the rivers. Within this study five erosion control practices were considered, keeping in mind the landscape feature of the Nil catchment: conservation tillage, sowing cover crops, contour farming, strip-cropping and construction of buffer strips. On can remark that the latter management techniques will also reduce drift losses to the river. Other drift reducing measures, e.g. drift reducing nozzles, are not included in this study due to minor importance; but can be found in De Schampheleire et al. (2005).

In order to the concept to account for buffers as presented in this paper, the following steps need to be done:

1. All fields/HRU's that will undergo buffers should be identified

In our study, this was established by manipulating the land use map using GIS software, whereby new land use classes were created for the areas near the river.

2. All these field will loose a small area to create a buffer.

Also using GIS software, new land use classes were defined for the buffers.

3. While the HRU's are calculated in the interface, it is essential that both the fields that get buffers, and the buffers itself are presented as HRU's. Within the interface, it is common practice to exclude marginal land use classes and soil classes by defining a threshold (in percentage) that the landuse or soils should cover in order to be represented in the HRU's. Since the land use classes that represent fields-with-buffer and the buffers in particular cover small areas, they would be excluded in this step. At present, there representation was guaranteed by setting 0 thresholds for the land uses, but at cost of a very high number of HRU's.

4. In the SWAT input files, a link between the HRU's representing the fields with buffer and the HRU's representing the buffers should be accounted.

In the adapted code, the link from the buffer HRU to the field-with-buffer HRU is made in the *.hru file of the field-with-buffer by providing the HRU number of the buffer. A requirement is that the buffer-HRU number is higher than the field-with-buffer HRU number.

The simulation results for the studied erosion control practices for corn cultivation in the Nil catchment are presented in Table 1 for the year 1998.

Before making any conclusions, it should be stressed that the changes of the parameter values CN2 and USLE-P is based on literature and the model manual. Sensitivity analysis revealed that model predictions are rather sensitive to the curve number value (Holvoet et al., 2006). As a consequence, the performed study should be considered as a guide to rank the measures in effectiveness rather than a quantitative assessment. To achieve the latter, field data would be necessary in order to be able to better parameterise the model for BMPs.

From Table 1 it is clear that strip-cropping is the most successful practice for atrazine reduction in the Nil basin, both for the dissolved and sorbed atrazine fractions. The alteration of low and high waned crops not only reduces the overland flow rate, but also decreases the transport of sediment particles and reduces the maximum slope length. Strip-cropping is followed by sowing cover crops and contour farming. The latter has the largest impact on the attached atrazine fraction. The construction of buffer strips results in a relatively limited decrease in pesticide load. Hereby it should be noticed that this measure is not applied on all corn fields in the catchment as was done with the other measures, but only on corn field situated along the river. Moreover, the model does not take into account runoff-water coming from fields situated above. This can result in underpredicting the amount of runoff-water entering the filter strip, and in underestimating the overland flow velocity. Also the buffering capacity is only calculated if a real (i.e. a field sowed with Bermuda grass) buffer strip is present along the water course. Modifications of ploughing practices, seems the least efficient measure in this study. Ploughing with a mouldboard plough leads even to increase in the total atrazine load in the river. This type of plough turns the soil around in such a manner that erosion sensitive soil is brought into the top layer. The mouldboard plough is not used in conservation agriculture, but is added to Table 1 for completion.

Table 14. Simulated results for mitigation measures.

BMP		% increase(+)/decrease(-)		
		dissolved atrazine	bounded atrazine	total atrazine load
Conservation agriculture	Mouldboard plough	-0.15	19.9	1.0
	Chisel plough	0.33	-23.3	-1.0
	Only seedbed preparation	0.57	-32.8	-1.3
	No-till management	1.02	-43.8	-1.5
Buffer strips	5 m width	-11.64	-12.1	-11.7
Contour farming		-25.93	-56.5	-27.6
Cover Crops: rye	Mouldboard plough	-32.12	-34.0	-32.2
	Chisel plough	-32.12	-47.2	-33.
	Only seedbed preparation	-32.08	-51.5	-33.5
	No-till management	-32.04	-58.4	-33.2
Strip-cropping		-37.27	-80.8	-39.7

Conclusions

From this study, a ranking in effectiveness of measures for atrazine load reduction could be obtained: strip-cropping seems to be more efficient than sowing cover crops, contour farming, construction of buffer strips, a 40% reduction in point losses and finally conservation agriculture. As indicated by Santhi et al. (2006), extensive monitoring data and intensive observation of BMPs are essential for assessing the effects of BMPs in a watershed. As for the moment there is no adequate literature available, showing the quantitative benefits of BMPs; a modelling approach is very useful. Furthermore, when choosing the most appropriate reduction technique; also the economic efficiency of the measure should be taken into account. Certain measures may have the best environmental performance; but are unachievable to perform due to high investment costs. A cost-benefit analysis would give insight in the costs and benefits of the proposed reduction measures, environmentally as well as economically. Herein, governments can play a big role. By given subsidies they can encourage farmers to take costly but environmentally friendly measures. Furthermore, by performing sensitization campaigns, farmers can be persuaded to take more precautions while performing operations with the product on the farm. It costs the farmers almost nothing, while those losses are responsible for almost 50 to 70% of the pesticide losses in the river the Nil.

Acknowledgement

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Using SWAT to support the Habitats Directive in the UK- a case study from the east of England

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Abstract

The Thurne catchment within the Broads National Park contains internationally important wetland sites, which are designated as Special Areas of Conservation (SAC's) under the EC Habitats Directive. As part of the national regulator's review of discharge consents in the catchment, the SWAT model is being used to assess the relative contributions of point and diffuse sources of phosphorous to the principle water bodies, and the likely impacts of phosphate removal at individual sewage treatment work discharges and land management change. This paper describes the use of the SWAT model within this policy context. It also considers the difficulties associated with applying the model in this flat coastal aquifer, in which much of the land is below sea level and drained by pumps which lift water from the artificial drainage networks into the river and lake system.

KEYWORDS: SWAT, drainage, lakes, Habitats Directive, SAC, Broads National Park

Introduction

Within Europe, natural habitats are continuing to deteriorate due to development and agricultural intensification. The EC Habitats Directive (Directive 92/43/EEC) aims to promote the maintenance of biodiversity by requiring Member States to take measures to maintain or restore natural habitats and species at a favourable conservation status. The 189 habitats and 788 species listed in the Directive are protected by a network of protected sites. The resultant Special Areas of Conservation (SACs), together with Special Protection Areas (SPAs) classified under the EC Birds Directive, form the Natura 2000 network.

The Environment Agency is the leading public body for protecting and improving the environment in England and Wales. It has a statutory duty under the Habitat Regulations to undertake a formal review of the effects on European features of existing consents, permissions and authorizations that it (or its predecessor bodies) has granted. The Agency has a four-stage process to implement this 'review of consents':

- **Stage 1** involved the identification of permissions that are relevant to the site;
- **Stage 2** assesses which permissions and authorizations, either alone or in combination are likely to have significant effect on the European site.
- Where Stage 2 concludes that there is likely to be a significant impact on the integrity of the European site, the Agency carries out a **Stage 3** "Appropriate Assessment" to assess whether permissions can be concluded not to have an adverse effect.
- At **Stage 4**, permissions are affirmed, or modified or revoked, subject to appeal.

The Broads is Britain's largest nationally protected wetland, comprising a complex network of rivers, broads (shallow lakes), marshes and fens. The Broads in Eastern England is the richest area for charophytes (or Stoneworts) in Britain (Stewart 1996). Twenty species have been recorded, which represents over 65% of the British flora. The core of this interest is the Upper Thurne Broads and Marshes Site of Special Scientific Interest (Figure 1), which forms part of the Broads/Broadland SAC and SPA, and particularly the large shallow Hickling Broad which is the richest site in the UK. The water bodies have been designated as "Hard oligo-mesotrophic waters with benthic vegetation of Chara spp." under the Habitats Directive.

Within the Upper Thurne, three of the four broads have been classified as being in "unfavourable status" (Hickling Broad- unfavourable declining; Heigham Sound - unfavourable recovering; Horsey Mere- Unfavourable – no change). Favourable conditon tables from Natural England give 0.03mg/l of total P as the limit, and these were used at stage 3 for the Appropriate Assessments. No target is set for nitrogen from the current state of knowledge (Broads Authority, 2006). Natural England are currently considering whether to adjust the P Limit for the stage 4 work. The complexity of the catchment prevents the use of existing empirical methods for quantifying impacts. This paper therefore describes ongoing work to use SWAT to support Stage 4 of the implementation of the Habitats Directive review of discharge consents in the Upper Thurne.

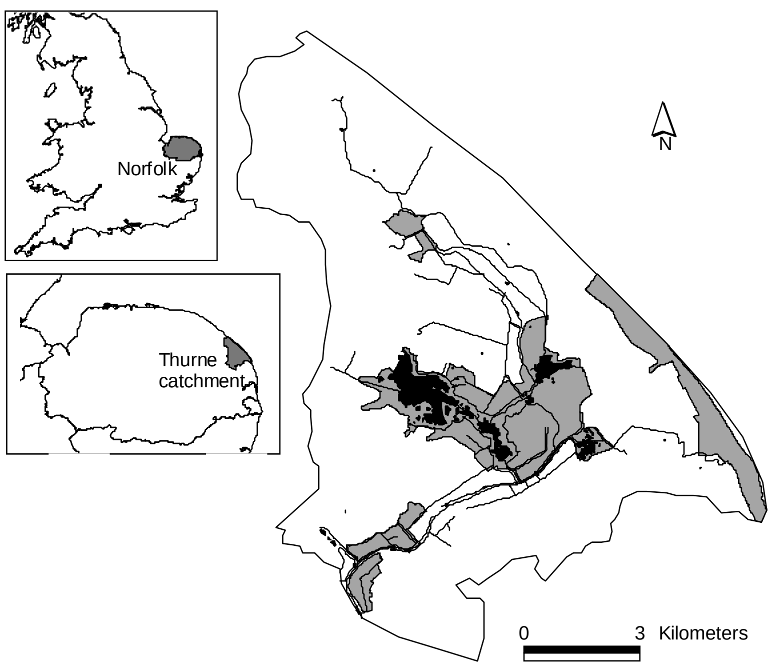


Figure 1 The Thurne catchment, showing the (black) broads and watercourses and the (grey) Upper Thurne Broads and Marshes SAC

Upper Thurne Catchment

The 110 square kilometers Thurne catchment in north east Norfolk contains several important Broads including Hickling Broad, Martham Broad and Horsey Mere.

The relief is very subdued with a height range of about 23 m. There are a number of significant challenges to modeling the flows of water and nutrients in the catchment:

- The water level in Hickling Broad is on average only +0.4 m above sea level, yet the eventual outlet to the sea is over 20 km away;
- Shrinkage of the alluvium due to drainage has left the river flowing above the level of the surrounding marshes;
- All of the marshes are artificially drained by a network of ditches, with the drainage water being raised up to 2.5 m into the river network by drainage pumps;
- Most of the river flow is due to land drainage pump discharges and tidal movements.

Although SWAT cannot represent the tidal fluctuations, SWAT has been set-up to try and replicate the hydrological behaviour of the system and, in particular, the effects of the pumped land drainage systems on water and diffuse source nutrient movement.

Materials and methods

Model set-up

Catchment delineation

Due to the low elevation and gradient of the river basin and SWAT's inability to model ground levels below sea level, all grid cells within the DEM were raised by 10 m prior to defining the basin and sub-basins using the automatic delineation function.

HRU delineation

The 1:250,000 scale National Soil Map (Hodge et al., 1984) has been combined with the Centre for Ecology and Hydrology's 1990 Land Cover Map of Great Britain.

Drainage pump systems

There is no module within SWAT to model drainage pumps. Each pump is connected to a network of upstream drains which, while the pump is not in operation, store water. The pumps could be modelled as point source discharge, but this will not represent the temporary storage of water in the drains or the change in flows into the broads, which occur with the turning on and off of the pumps. Both of these factors affect water quality, especially the sediment (and associated P) load reaching the Upper Thurne system.

The land drainage pumps have therefore been modelled as controlled outflow reservoirs. Reservoirs within SWAT are located on the main river channel network and receive water from all sub-basins upstream of the water body. They modify the movement of water in the channel network by lowering the peak flow. As the reservoirs slow down the flow of water, sediment will fall from suspension, removing nutrient and chemicals adsorbed to the soil particles. Electrical consumption data has been used to derive maximum and minimum average monthly discharge rates (using pump conversion factors of Holman, 1994) in order to set monthly target release rates for each controlled reservoir.

Attribute data

Soil attribute data for the dominant soil type in each soil polygon was provided by the National Soil Resources Institute. Crop parameter values were informed by Hough (1990). Management files were based upon Hough (1990), MAFF (2000) and Holman et al. (2004). A range of crop rotations based on Holman et al. (2004) were derived to capture the temporal variability in arable cropping. Thirteen rotations (Table 1) linked to soil type provided a good match with averaged crop statistics for 1994, 1997 and 2000 (Figure 2).

Table 1: Simulated crop rotations within the Thurne catchment

Soil type			
Other Mineral	**Sandy**	**Organic**	**Peaty**
osr/ww/wb/wbn/ww/wb osr/ww/sbt/wbn/ww/wb	pts/ww/sb/sbt/ww/wb pts/ww/wb/wbn/ww/wb wbn/ww/sb/sbt/ww/wb sbt/ww/sbt/wb sbt/ww/p/wb Maize sbt/ww/osr/wb sbt/ww/sb/wb	sbt/ww/sb/ww sbt/ww/p/ww	wbn/ww/p/ww

(Osr: oilseed rape; ww: winter wheat; sb: spring barley; wb: winter barley; wbn: winter field beans; sbt: sugar beet;; pts: potatoes; p: peas)

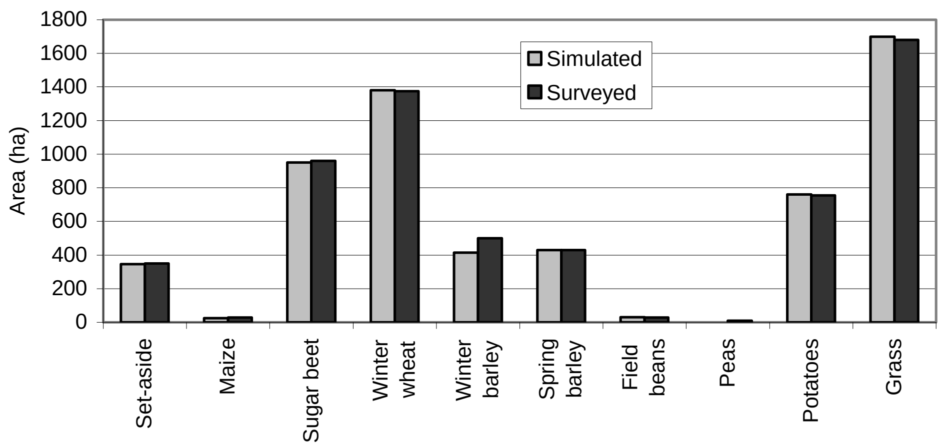


Figure 2 Comparison of simulated and actual crop areas within the Thurne catchment

Results and Discussion

Because of the tidal influence on the Thurne river, there are no flow data available to calibrate and validate the model. Input parameters were used from the successfully validated SWAT model of the neighboring non-tidal Bure and Ant catchments (Whitehead, 2006).

Pump discharges

Although an exact flow pattern is not achieved due to actual pump rate variability (Figure 3) a good comparison between annual pump rates has been seen (Table 2).

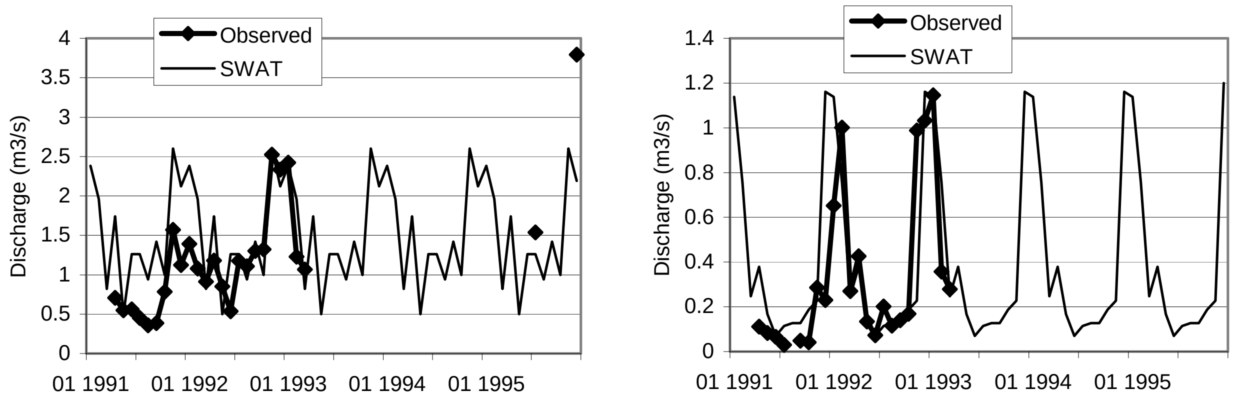


Figure 3 Comparison of observed and simulated discharge at (left) Brograve and (right) Catfield pumps

Table 2: Average annual observed and simulated pump flows in the Upper Thurne

Pump	Observed pump flow (10^4 m^3 yr^{-1})	Simulated pump flow (10^4 m^3 yr^{-1})
Catfield	520	470
Stubb Mill	953	978
Eastfield	2155	2310
Brograve	1572	1800
Horsey Mill	1711	1511

Surface water nutrient concentrations

Calibration and validation has only been undertaken for nutrients, with calibration over the period 1991-94 (Figure 3) and validation from 1995 onwards (Figure 4).

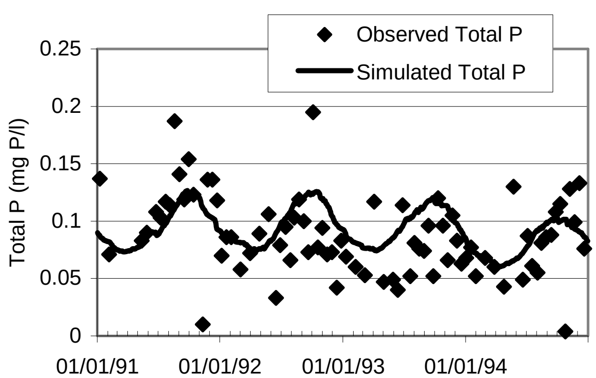
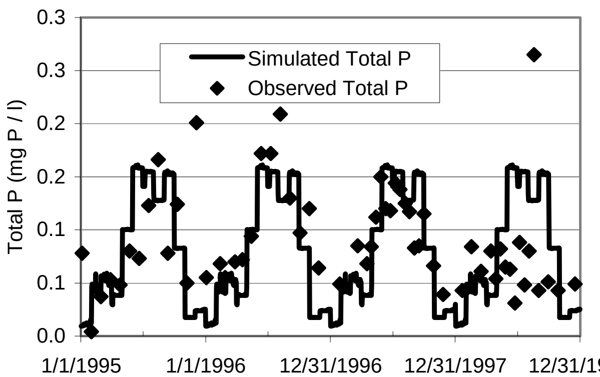


Figure 3 Nutrient calibration - observed and predicted total phosphorus in Hickling Broad (1991 – 1994).



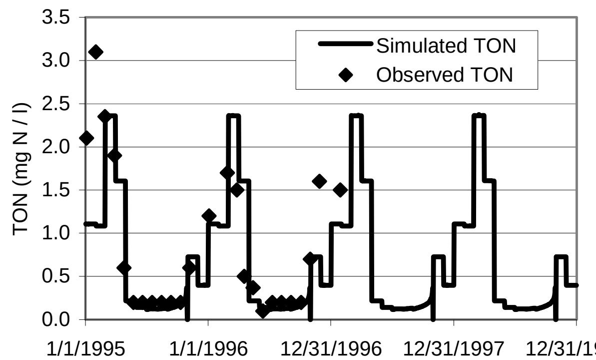


Figure 4 Nutrient validation at Heigham Sound (1995-1998) – (left) Total P and (right) Total Organic N.

Discussion

The SWAT model of the Thurne catchment (Whitehead, 2006) appears superficially to represent this man-modified catchment well. The variable land uses within the catchment have been accurately represented, based upon a range of realistic cropping rotations. The temporal concentrations of nutrients within the waterbodies are realistically simulated, despite the lack of simulated tidal influences. This suggests that the tidal influence on water quality is less important than the drainage pump discharges and internal nutrient cycling within the waterbodies.

However, there are a number of significant problems in this initial model build which limit the current practical utilization of the model to inform the review of consents in the Upper Thurne. The principal problem arises from the poor delineation of the basin (Figure 5) and sub-basins using the 50 m DEM. The sub-basins which feed into the drainage pumps are not topographically controlled, but divided by man-made barriers. Therefore the pump sub-basins as modelled in SWAT differ in size and shape to those of the actual pump sub-basins, thereby affecting the run-off and nutrient losses from each subbasin. The Catfield pump catchment is the only topographically defined catchment, and SWAT is therefore able to model reasonably well the flow pattern of this pump (Fig. 3). Current work is aiming to improve sub-basin delineation using a modified higher resolution DEM.

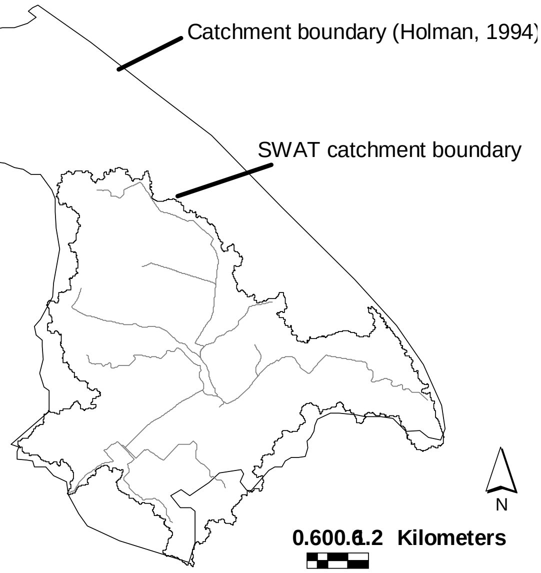


Figure 5 Comparison of the SWAT catchment boundary with the boundary of Holman (1994)

Conclusions

This paper reports the application of the SWAT model in the Thurne catchment within the Broads National Park in eastern England, UK. The Thurne catchment has been heavily modified by man's activities, so that most of the flows into the river enter via the pumped drainage systems. The pumped land drainage systems have been successfully represented within SWAT as reservoirs with controlled monthly outflows. SWAT is able to simulate nutrient concentrations with some degree of accuracy, although the monthly or bi-monthly observed water quality data prevents the model from capturing the daily variation of nutrient concentrations within the system. However, limitations in the basin and sub-basin delineation arising from the 50 m DEM resolution and very low topographical gradients means that further work is necessary to improve the model set-up prior to its use within Stage 3 of the Habitats Directive review of consents. Current ongoing work is improving the basin and sub-basin delineation to provide an improved representation of the internal hydrological behaviour of this complex catchment.

Acknowledgements

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An Alternative Approach for Analyzing Wetlands in SWAT for the Boone River Watershed in North Central Iowa

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Abstract

The Boone River Watershed (BRW) covers over 237,000 ha in north central Iowa. The watershed is dominated by corn and soybean production, which together account for almost 84% of the land use. Fertilizer and livestock manure applications to cropland are key sources of nutrient loads to the watershed stream system. Nitrate losses are of particular concern, much of which escapes the cropland via subsurface tiles that drain the predominantly flat landscapes throughout the watershed. A modeling framework using the Soil and Water Assessment Tool (SWAT) model (version 2005) to support analyses of alternative management practice and/or cropping system scenarios that could potentially result in reduced nonpoint source pollution in the watershed. Two SWAT configurations have been constructed for the Boone River watershed consisting of either 30 or 405 subwatersheds. The 30 subwatershed configuration is a typical SWAT approach that facilitates a variety of land use and management scenarios, but does not support detailed constructed wetland scenarios which are potentially a key nutrient loss mitigation strategy for the BRW and other watersheds in the region. However, the second configuration (405 subwatersheds) does provide the potential for a more realistic assessment of wetland impacts by providing the ability to spatially site wetlands in a more realistic manner within the watershed. The general framework for both approaches is discussed here, including differences between the two different subwatershed configurations. Alternative HRU configurations are also presented including the option to overlay common land units (field tracts) in the watershed, which results in over 20,000 HRUs.

KEYWORDS: SWAT, wetlands, tile drainage, nutrient applications, nitrate

Introduction

The Boone River Watershed (BRW) is an intensively cropped region located in north central Iowa which was identified by Libra et al. (2004) as discharging some of the highest nitrogen loads during 2000-2002 among the 68 Iowa watersheds that were analyzed within their study. They estimated that Iowa streams contributed approximately 20% of the long-term nitrogen load to the Gulf of Mexico based on in-stream measurements performed during 2000-2002. The nitrate load discharged from the mouth of the Mississippi River has been implicated as a key cause of the Gulf of Mexico seasonal oxygen-depleted hypoxic zone, which has covered an extent equal to or greater than 20,000 km^2 in recent years (Rabalais et al., 2002). The BRW has also been identified within the UMRB as both an area of freshwater biodiversity significance and a priority area for biodiversity conservation (Weitzell et al., 2003). The biodiversity conservation designation reflects the fact that the watershed has been identified as currently possessing a "relatively un-degraded stream ecosystem," but that it is also very vulnerable to future increased degradation (Neugarten and Braun, 2005). Potential biodiversity threats listed by Neugarten and Braun include consistently high in-stream nitrogen concentrations, farm production methods that may be ecologically harmful, and inadequate treatment of wastewater.

A simulation study has been initiated in response to these issues that is designed to evaluate the potential economic and environmental impacts of alternative land use and management practices in the BRW. The goal of the study is to identify strategies that can potentially mitigate loss of nitrates and other pollutants from agricultural cropland, which could lead to improved water quality in the Boone stream network as well as in downstream ecosystems such as the Gulf of Mexico. Insights gained from the study may also be transferable to other watersheds that drain parts of the Des Moines Lobe, which are generally characterized as regions of high nitrogen (N) export. Environmental impacts will be assessed within the study with the Soil and Water Assessment Tool (SWAT) model (Arnold et al., 2005), which has been used for a wide range of environmental conditions, watershed scales, and scenario analyses (Gassman et al., 2007).

Several practices are being investigated in the BRW project including constructed wetlands, which have been found to be very effective at removing nitrate (NO_3-N) from agricultural cropland drainage water (Mitsch et al., 2001). Studies that report simulation of wetlands in SWAT are very limited and focused only on wetland hydrologic impacts (Gassman et al., 2007). Constructed wetlands are potentially key nutrient loss mitigation structures for the BRW and other watersheds in the region, and represent partial restoration of the extensive wetland systems that existed prior to European settlement. Two different BRW SWAT configurations are described here that consist of 30 versus 405 subwatersheds. The 30 subwatershed configuration is a typical SWAT approach that facilitates a variety of land use and management scenarios, but does not support detailed spatial accounting of wetlands. However, the second configuration (405 subwatersheds) does provide the potential for a more realistic assessment of wetland hydrologic and nutrient impacts. Both simulation approaches are presented here, including alternative HRU configurations that allow for an additional range of spatial detail to be incorporated into the SWAT simulations.

Watershed Description

The BRW covers over 237,000 ha in six north central Iowa counties and is one of 131 U.S. Geological Survey (USGS) 8-digit hydrologic unit code (HUC) watersheds (http://www.nrcs.usda.gov/TECHNICAL/land/meta/m3862.html) that are located in the UMRB (Figure 1). It lies within the Des Moines Lobe geologic formation, which is the southern most portion of the central North American Prairie Pothole Region. An extensive network of subsurface tile drains and surface ditches have been installed throughout the watershed, resulting in the elimination of most wetland areas and an intensively cropped landscape. The watershed is dominated by corn and soybean production, which together account for almost 84% of the land use (http://www.igsb.uiowa.edu/nrgislibx/; other land use includes ungrazed grassland (8.6%), deciduous forest (2.2%), grazed grassland (1.5%), Conservation Reserve

Program (CRP) land (0.8%), and alfalfa (0.7%). An in-depth field-level survey of the BRW by Kiepe (2006) revealed that the use of conservation tillage, especially mulch tillage, is very extensive throughout the watershed. The survey also showed that only a small number of terraces, grassed waterways and other structural practices have been installed on cropland in the watershed, reflecting the relatively flat landscapes that dominant the watershed. A total of 128 confined animal feeding operations (CAFOs) are located in the BRW, which are partitioned between 13 cattle, 6 chicken (layer), and 109 swine operations that represent a total of about 4,250, 6,960,000, and 480,000 head, respectively (http://www.igsb.uiowa.edu/nrgislibx/). Observed stream flow data are collected by the USGS) at a gauge located south of Webster City and limited pollutant data are collected by the Iowa Department of Natural Resources at the watershed outlet (Figure 1).

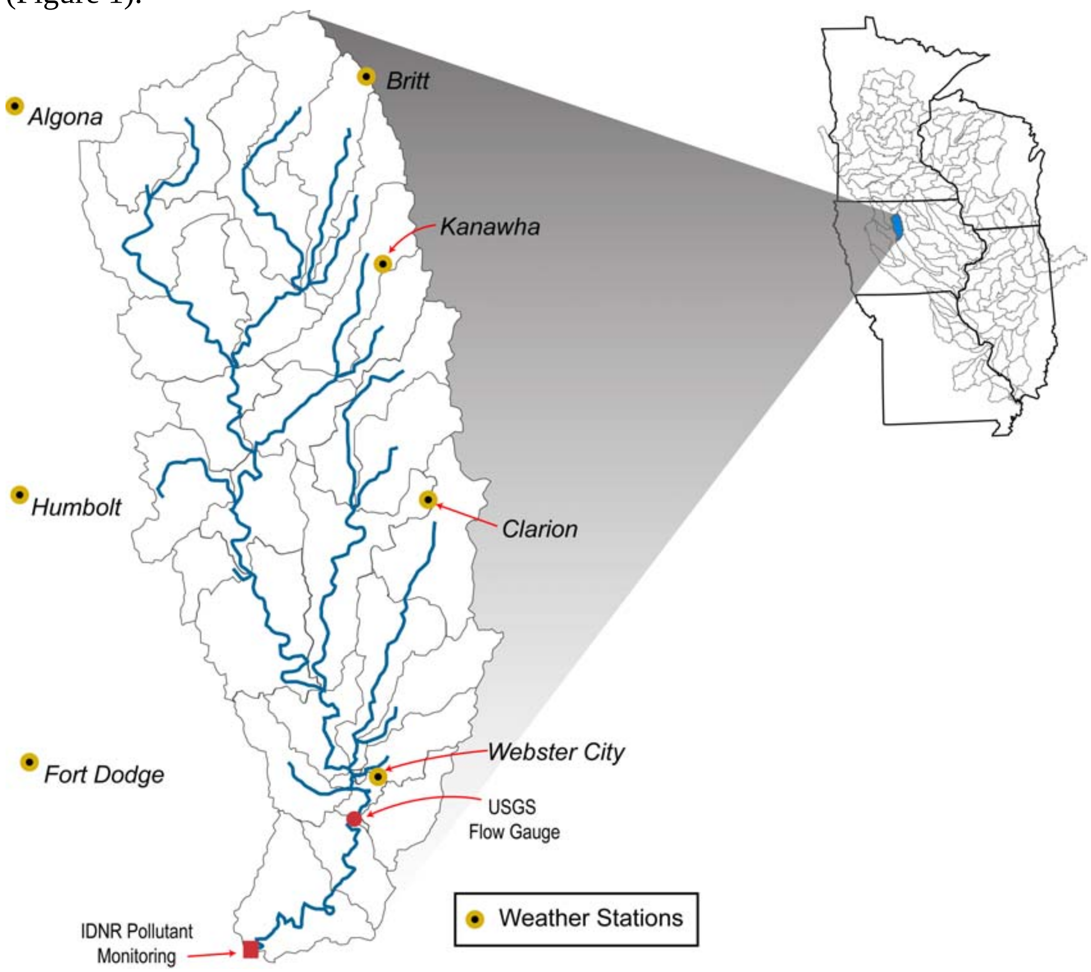

Figure 16. Location of the Boone River Watershed within Iowa and the Upper Mississippi River Basin, and the subwatersheds, climate stations, and monitoring sites used for the SWAT simulations.

Modeling System and Input Data

Figure 2 shows a schematic of the SWAT modeling system that has been constructed for the BRW simulations. The core of the system is SWAT2005, which is the latest release of the model that features several enhancements including an improved subsurface tile drainage component as described by Gassman et al. (2007). The BRW modeling system is initiated by processing topographic, land use, climate, and soil data (Table 1) within the ArcView SWAT-X (AVSWATX) interface (Di Luzio et al., 2004). The DEM layer was used in AVSWATX to initially delineate 30 subwatersheds (Figure 1), whose boundaries were generally aligned with standard 12-digit watershed boundaries (USDA-NRCS, 2007). A total of 531 hydrologic response units (HRUs) were then created that were distributed across the 30 subwatersheds. The initial 335 cropland HRUs consisted only of monoculture cropping systems dominated by continuous corn and continuous soybean. Some editing tools are provided in AVSWATX to convert such monoculture HRUs into crop rotations, and to add tillage, fertilizer application, and other management operations as appropriate. However, these editing tools are limited and did not provide the desired flexibility for building the cropping system and management inputs for the BRW simulations. Thus external software was developed to convert the monoculture HRUs into crop rotation HRUs (Figure 2); the desired fertilizer and tillage operations were also incorporated into the HRU management

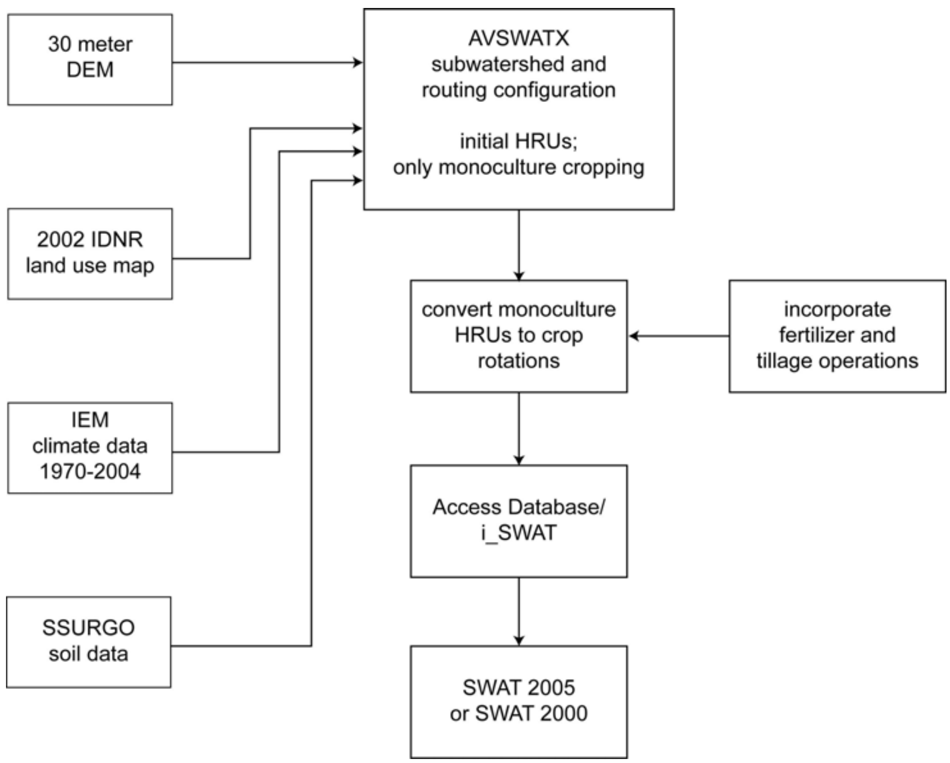

Figure 17. Schematic of the Boone River Watershed SWAT modeling system.

Table 15. Key categories of input data and respective data sources for the BRW simulation.

Data Type	Source
Soil[a]	Soil Survey Geographic (SSURGO) Database (http://www.ncgc.nrcs.usda.gov/products/datasets/statsgo/) Iowa Soil Properties and Interpretations Database (ISPAID) Version 7.2 (http://extension.agron.iastate.edu/soils/pdfs/ISP71MAN.pdf)
Climate	Primary: Iowa Environmental Mesonet (http://mesonet.agron.iastate.edu/COOP/) Secondary: NOAA Satellite and Information Service (http://www.ncdc.noaa.gov/oa/climate/stationlocator.html)[b]

Land use	2002 Land Cover GRID of Iowa (http://www.igsb.uiowa.edu/nrgislibx/)
Topographic	Resampled 30 m Digital Elevation Model (Wolter, 2006)

[a]SSURGO data was not available for two of the six counties that encompass the BRW; thus ISPAID data was converted into SSURGO format to complete the required soil input data layer.
[b]Source of data for the Kanawha climate station (Figure 1).
[c]The 30 m DEM was resampled from 10 m DEM topographic data.

schemes in this step. All of the required input data was then inserted into an Access database and the SWAT simulations were subsequently managed with the interactive SWAT (i_SWAT) software (http://www.card.iastate.edu/environment/interactive_programs.aspx), which translates the data in Access into the required input file formats, executes SWAT, and inserts output data back into the Access database. This approach provides increased flexibility for modifying SWAT inputs using Access queries and is in general a very straightforward method for managing the input and output data for a SWAT simulation.

The SWAT configuration shown in Figure 1 provides an adequate structure for performing a variety of alternative scenarios including variations in fertilizer and manure application rates, increased adoption of perennial vegetation in cropping systems, and shifts in tillage practices. However, there are definite limitations for using the 30-subwatershed approach to perform constructed wetland scenarios, due to the current SWAT structure that only allows a single, non-spatially defined wetland to be configured at the outlet of a subwatershed. Thus an alternative SWAT subwatershed configuration is being explored that potentially provides a more accurate assessment of wetland spatial placement in the BRW.

Alternative SWAT Configuration

The alternative SWAT configuration was inspired by current wetland placement efforts within the Iowa Conservation Reserve Enhancement Project (CREP) program (http://www.agriculture.state.ia.us/CREP.htm), which is supporting constructed wetland installation in tile-drained landscapes in 37 north central Iowa counties[5]. Three key wetland site selection criteria used within the CREP program are: (1) drainage areas must exceed 202 ha (500 ac), (2) wetland surface areas should equal 0.5 to 2.0% of the overall drainage area, and (3) the wetlands have to intercept subsurface tile drain discharge. Several other factors are also considered in CREP wetland placement decisions including topographic position and maintaining 25% open water[6] (versus the remaining 75% of the wetland being vegetated).

The alternative SWAT configuration was generated by using the smallest stream definition threshold area possible within AVSWATX for the BRW, using the previously discussed 30 m DEM. This resulted in a total of 405 subwatersheds (Figure 3) that average about 585 ha in size and are more consistent with the drainage areas targeted within the CREP program. It is assumed that any of these subwatersheds that exceed 202 ha can be included as potential wetland sites within the BRW simulation project, assuming the subwatershed is cropped and drained with subsurface tiles. However,

[5]The SWAT wetland simulation method described here is not intended to be an exact representation of the CREP approach, nor is there any intent to imply any official endorsement of the CREP program.
[6] According to Lemke (2007), the open water is maintained to provide suitable habitat for migratory water fowl.

additional criteria for siting potential wetlands can also be introduced, such as a threshold value based on a percentage of the area in the subwatershed that would be classified as hydric soils. For example, the subwatersheds that would be selected for constructed wetlands based on both the 202 ha criterion and a 60% hydric soil criterion[7] are shown in blue on the right-hand side in Figure 3a. Using these combined criteria results in 154 subwatersheds being selected for constructed wetlands, versus 319 subwatersheds that would be selected if just the 202 ha drainage area criterion is used.

Figure 4 also shows a comparison of wetland distribution between the 405- and 30-subwatershed configurations, using the 60% hydric soil threshold and assuming that the surface area of each wetland in Figure 4a was equal to 1.25% of the respective drainage area. The corresponding subwatershed wetland area ranges are given in Figure 4b for the 30-subwatershed configuration. Each of these total wetland areas has to be simulated as a single wetland in SWAT for the 30-subwatershed approach, with associated total drainage areas based on the total "wetland subwatersheds" (Figure 4a) that are located in each of the 30 subwatersheds. This comparison underscores that the alternative configuration of 405 subwatersheds provides a more realistic accounting of constructed wetland placement and potential impacts of those wetlands.

Hydrologic Response Unit (HRU) Configuration Options

Several options exist for delineating HRUs for the BRW SWAT simulations, most of which involve restructuring the HRUs external to AVSWATX as previously described for the 30-subwatershed configuration with 531 HRUs. Using dominant soil types and land use is a potentially viable option for the 405-subwatershed configuration, due to the relatively small subwatershed sizes incorporated in that approach. However, this results in some distortion of land use, with the cropland estimated to occupy over 200 km^2 more area than the 30-subwatershed/531 HRU approach. At the other extreme, a very detailed dataset has been

[7] The hydric soils were determined using an algorithm developed by Jaynes (2006) and the ISPAID soil database (Table 1).

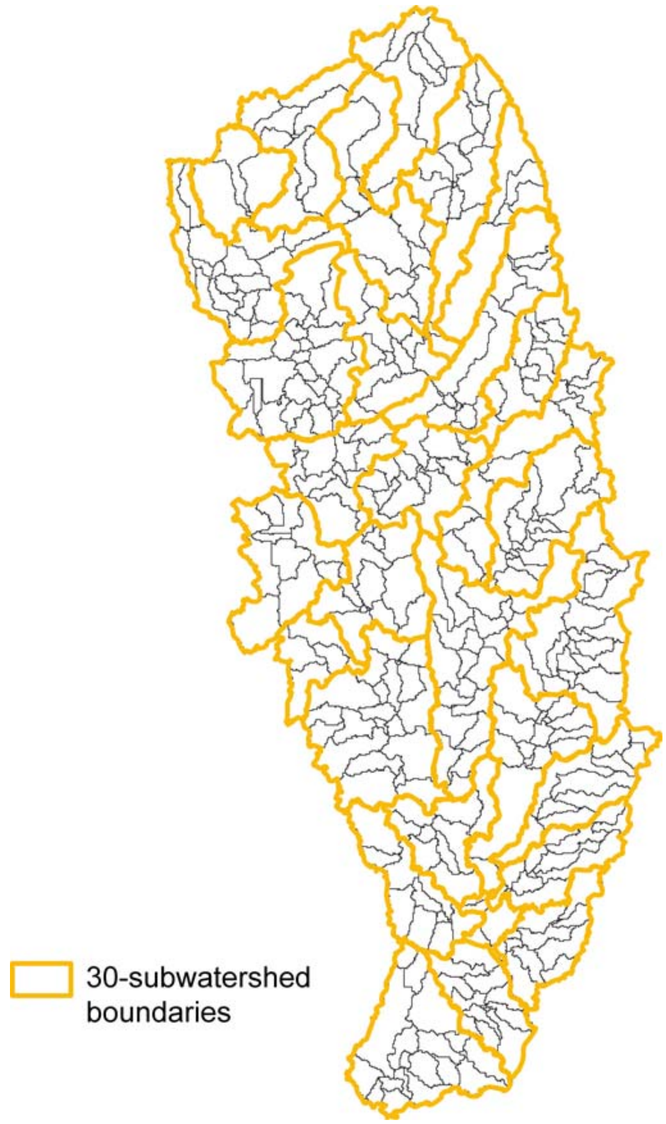

Figure 3. Alternative SWAT configuration of 405 subwatersheds for the Boone River Watershed (with overlay of the 30-subwatershed boundaries)

developed for the BRW at the Common Land Unit (CLU) level which are defined by individual field boundaries or combinations of fields for a whole farm (see http://www.itc.nrcs.usda.gov/scdm/docs/DMP-CLU-DataManagementPlan.pdf). An overlay of the 405 subwatershed boundaries on the CLU boundaries results in nearly 23,000 HRUs being delineated for the BRW. These detailed HRUs provide a greater level of soil and land use accuracy and also facilitate a more direct linkage with an economic model being developed for the BRW, which will be constructed using CLUs as the basic spatial unit. However, it is not clear how much the accuracy of the SWAT predictions will be improved with this set of detailed HRUs. This will be further explored in the next phase of work that will be focused on testing the different HRU delineations for both the baseline and wetland scenarios.

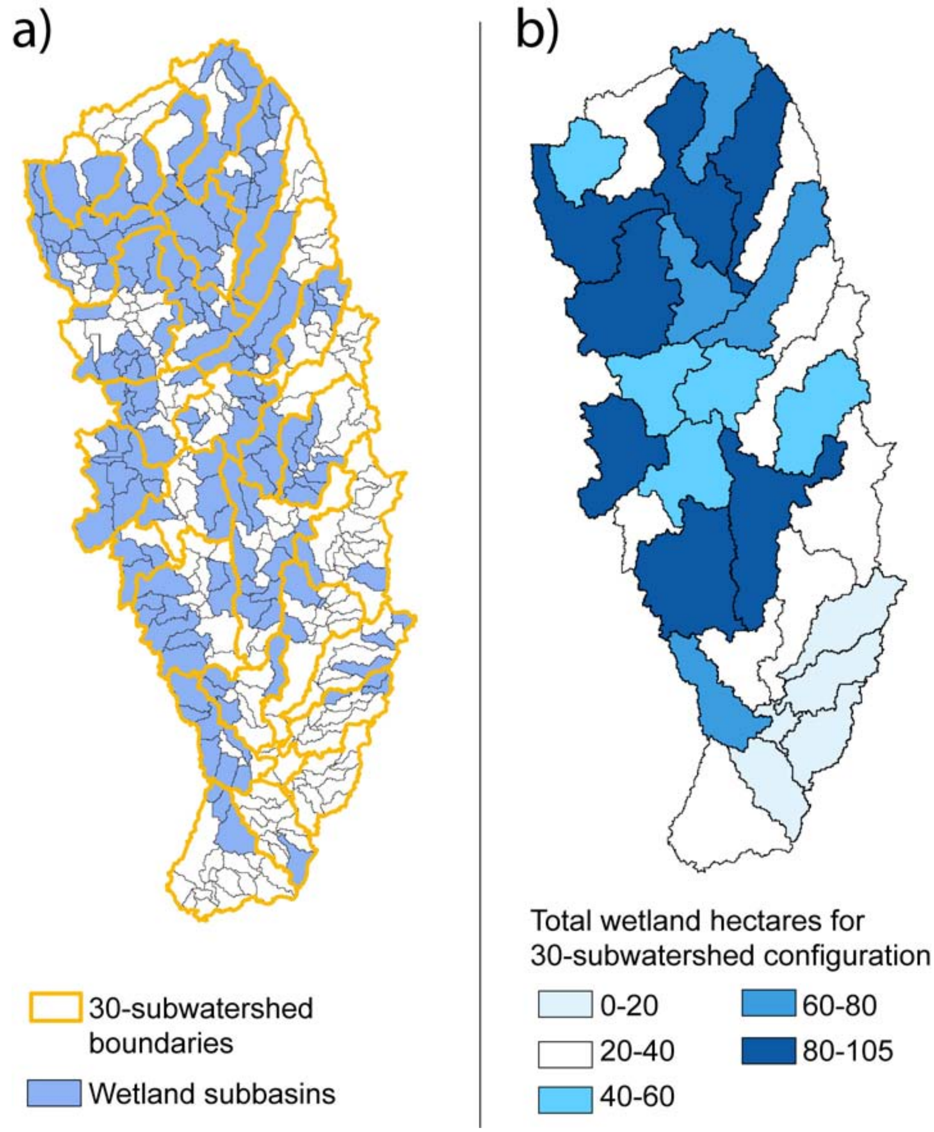

Figure 4. a) Example subwatersheds selected for constructed wetland placement within the alternative SWAT 405- subwatershed delineation of the BRW, versus b) the total areas that would be converted into wetlands for the 30-subwatershed configuration (based on wetland surface areas that equal 1.25% of each respective subwatershed drainage area).

Conclusions

The alternative SWAT configuration provides a more realistic framework in which to evaluate wetland scenarios in SWAT for the BRW, using either dominant HRUs or much more detailed HRUs that are based on the CLUs within the watershed. Testing of the different subwatershed and HRU combinations is needed to ascertain how sensitive SWAT is to the different configurations and whether the alternative approach definitely proves to be enhancement for simulating the effects of constructed wetlands.

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Reaction Kinetics for Modeling Non-Point Source Pollution of Nitrate with SWAT Model

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Abstract

Excessive use of nitrogenous fertilizer vis-à-vis phosphatic and potassium fertilizers in India has resulted in a shift of NPK ratio from an optimal value of 4:2:1 to a distorted 7.9:2.9:1. Nitrogenous fertilizers such as urea have high hydrolysing property and are not retained by the soil. They get leached into groundwater due to rainfall and irrigation water inputs and are thus distributed by the land phase of the hydrological cycle which requires knowledge of precipitation distribution, runoff generation, distribution, and fertilizer application rate in addition to residence time of water in each of the phases. Residence time is important as nitrates get transformed into gaseous and other stable forms in time and space. The reaction kinetics of the nitrogen transformation processes governs the overall movement of nitrates in each of the phases of the hydrological cycle.

SWAT ArcView GIS Version model has been used to simulate non-point source pollution of nitrate in a mountainous sub-catchment in India. The performance of the model in simulating daily discharge for the validation period was found to be good, measured both in terms of R^2 and Nash and Sutcliffe efficiency criteria. The model performance for estimating nitrates in surface runoff suggests a need for an improvement in the nitrogen transformation processes.

Further, this paper makes an analysis of the internal nitrogen fluxes simulated within SWAT and tries to conceptualize nitrification and denitrification kinetics in the unsaturated zone by making use of Michaelis-Menten mixed order kinetics. The comparison of the existing SWAT nitrate transformation first-order kinetics with the Michaelis-Menten mixed order kinetics show that Michaelis-Menten mixed order kinetics can be used to represent the nitrate transformation processes better.

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KEYWORDS: SWAT, Hydrology, Nitrate, Non Point, Pollution, Reaction Kinetics

Introduction

In India, amongst various sectors, the agriculture sector accounts for more than 80% of the total water use. It also accounts for excessive use of nitrate fertilizers in agricultural areas. Comprehensive analysis of the agriculture sector in the last three decades reveals that while the net sown area has remained almost constant the gross sown area is showing an upward trend. Crop intensification in the agriculture sector, measured in terms of cropping intensity and area under high-yielding varieties of crops, is on a rise. This in turn requires increased use of fertilizers to raise soil productivity and crop yields.

Amongst various fertilizers, namely, nitrogenous, phosphate, and potash, the use of nitrogenous (N) or nitrate fertilizers accounts for 60%–80% of the total use. Fertilizer use ratio of nitrate:phosphate:potash (N:P:K) comes out to be an average of 7.9:2.9:1 for last 50 years with a highest recorded value of 16.6:1.7:1. The optimal mix is 4:2:1 (FAI, 2001).

Nitrates being highly soluble in water are influenced by the land phase of hydrological cycle and get transported to surface and groundwater (Wendland et al., 1993). Moreover, N transformations, both in soil and water, are dynamic and involve microbial processes that are continuous in time. Process kinetics is dependent on a number of environmental factors such as soil pH, temperature, moisture content, oxygen, NO_3- content, etc., which requires accurate representation as these have a direct bearing on nitrate loadings into surface and groundwater systems. Since the medium that brings these transformations into effect is water, simulating N transformations in soil or unsaturated zone requires land phase of the hydrological cycle to be simulated accurately, and continuously.

This study makes use of SWAT model (Arnold et al., 1998), a physically based, time continuous, distributed, and widely applied land phase of the hydrological cycle model (Srinivasan et al., 1998; Gosain et al., 2005), to simulate runoff and nutrient (nitrate) transport in this study. Michaelis-Menten mixed-order kinetics, well suited for simulating microbial action and growth under various environmental conditions and substrate concentration, has been compared with the first-order kinetics used in SWAT. A mountainous sub catchment, namely the Lakhwar sub catchment of the Upper Yamuna catchment has been taken up for application of the SWAT model.

Methodology

The land phase of the hydrologic cycle as simulated by SWAT is based on the water balance equation (Neitsch et al., 2001).

Nitrification, defined as conversion of NH_{4+} to NO_3-, is a two-step bacteriological oxidation. Approach followed in SWAT to model nitrification is described by first-order reaction kinetics where the total amount of ammonium lost to nitrification ($N_{NH4+\rightarrow NO3-}$) is proportional to the amount of ammonium (N_{NH4+}) in the soil. A more realistic approach accounts for microbials where the growth equation of microorganisms (M) considers the utilization of the substrate ($N_{NH4+\rightarrow NO3-}$) (mg l^{-1}). This makes use of the Michaelis-Menten kinetics that substitutes for the microorganism concentration its equivalence in terms of substrate such that $\left(-\frac{dN_{NH4+}}{dt}\right)\alpha\left(a_n\frac{dM}{dt}\right)$ where, M is the microbial concentration (mg cells l^{-1}). This is finally represented as,

$$\left(\frac{dN_{NH4+\rightarrow NO3-}}{dt}\right)_{ly} = -a_n\, f_{t,ly}\, f_{w,ly}\, f(NH_{4+})_{ly}\, f_{pH,ly}\, M_{ly} \qquad \text{(in g m}^{-3}\text{)} \tag{1}$$

where $f(NH_{4+})_{ly}$ is the Michaelis-Menten based response function for soil ammonium given by

$$f(NH_{4+})_{ly} = \frac{N_{NH4+,ly}/\theta_{ly}}{N_{NH4+,ly}/\theta_{ly} + K_{s_{NH4+}}} \tag{2}$$

and, $M_{ly} = M_o \exp(k_{g,net}\, t)_{ly}$ (3)

where M_o is the initial microbial concentration (mg cells l^{-1}), $k_{g,net}$ (day^{-1}) is the net microbial growth rate, a_n is stoichiometric or yield constant (mg nitrogen N per mg cells), M_{ly} is the microbial concentration (mg cells per litre, mg cells l^{-1}), $K_{s,NH4+}$ is Michaelis or half saturation constant defined as the concentration of substrate (ammonium) at which the growth rate of microorganisms is one-half of the saturated rate (mg l^{-1}), $N_{NH4+,ly}$ is the ammonium load in the soil layer (g m^{-2}), θ_{ly} is the soil moisture content in layer (m), and t is time (day). Environmental factors that influence the kinetics include f_t (temperature factor), f_{pH} (soil pH factor), and f_w (soil moisture factor) (Johnsson et al., 1987).

The final time continuous equation can be represented as,

$$N_{NH4+\rightarrow NO3-,final,ly} = N_{NH4+\rightarrow NO3-,initial,ly} + \left(\frac{dN_{NH4+\rightarrow NO3-}}{dt}\right)_{ly} \Delta t \tag{4}$$

where, $N_{NH4+\rightarrow NO3-,\ initial,ly}$ and $N_{NH4+\rightarrow NO3-,\ final,ly}$ are initial and final soil ammonium concentrations (mg l^{-1} or g m^{-3}) available for nitrification, and Δt is time step (day).

The form of equation used by SWAT presently assumes limiting substrate (N_{NH4+}) in the soil 'without' microbial action with environmental factors. Michaelis-Menten based response functions are replaced by kinetics where total amount of ammonium lost is proportional to the amount of ammonium in soil. Incorporation of these limiting conditions in the governing equation, Eq. (1), above reduces it to first order kinetic rate equation.

Overall procedure followed for denitrification module is similar to that followed in nitrification module above where Michaelis-Menten kinetics is used to derive the nitrification rate equation. Denitrification process is a biological process where N oxides are reduced to gaseous form by facultative anaerobic bacteria. The process depends on the concentration of substrate N_{N03-} in N pool of soil layer (ly) and carbon content in soil layer (C_{ly}) in addition to various environmental conditions.

The Michaelis-Menten kinetic equation can be written as,

$$\left(\frac{dN_{NO3-}}{dt}\right)_{ly} = -k_{pot}\, f_{t,ly}\, f_{w,ly}\, f(NO_{3-})_{ly}\, f(C)_{ly}\, f_{pH,ly} \qquad (5)$$

where, $f(C)_{ly}$, and $f(NO_{3-})_{ly}$ are Michaelis-Menten based response function for soil carbon content nitrate nitrogen content in a soil layer 'ly'(g m^{-2}), $K_{dns,NO3-}$ is the half saturation rate for nitrate nitrogen concentration (mg l^{-1}), $K_{dnrate,c}$ is the half saturation rate for organic carbon (mg l^{-1}), and θ_{ly} is the soil moisture content in layer ly (m). $K_{n,ly}$ is the denitrification rate constant (g m^{-2}), and 't' is time (day). k_{pot} is the potential denitrification rate which is assumed to be related to maximum activity of the microbial activity in the soil and depends on soil type (g m^{-2} day^{-1}). The potential denitrification rate is reduced according to the oxygen status of the soil expressed in form of soil water content ($f_{w,ly}$). Other factors that influence denitrification are temperature ($f_{t,ly}$) and pH ($f_{pH,ly}$) (Hansen et al., 1999).

Actual denitrification rate is determined as reduced potential rate and as a rate at which nitrate and carbon in soil are available (Abbot et al., 1996; Hansen et al., 1991).

Taking initial nitrate concentration in soil layer as $N_{NO3-,\ initial,ly}$, the final nitrate concentration $N_{NO3-,\ final,ly}$ can be given as,

$$N_{NO3-,final,ly} = N_{NO3-,initial,ly} + \left(\frac{d_{NO3-}}{dt}\right)_{ly} \Delta t \qquad \text{(in g } m^{-2}\text{)} \qquad (6)$$

A case can be derived from the above equation that represents first order kinetics under favorable soil moisture conditions (that is, $f_{w,ly} \geq 0.95$) adopted by SWAT model.

Description of the study area

The Upper Yamuna catchment lies in the north of India, drains the Punjab–Kumaon Himalayas from Simla in the northwest to Mussoorie in the southeast. The river rises from the Jamnotri springs near Badar Punch (30°

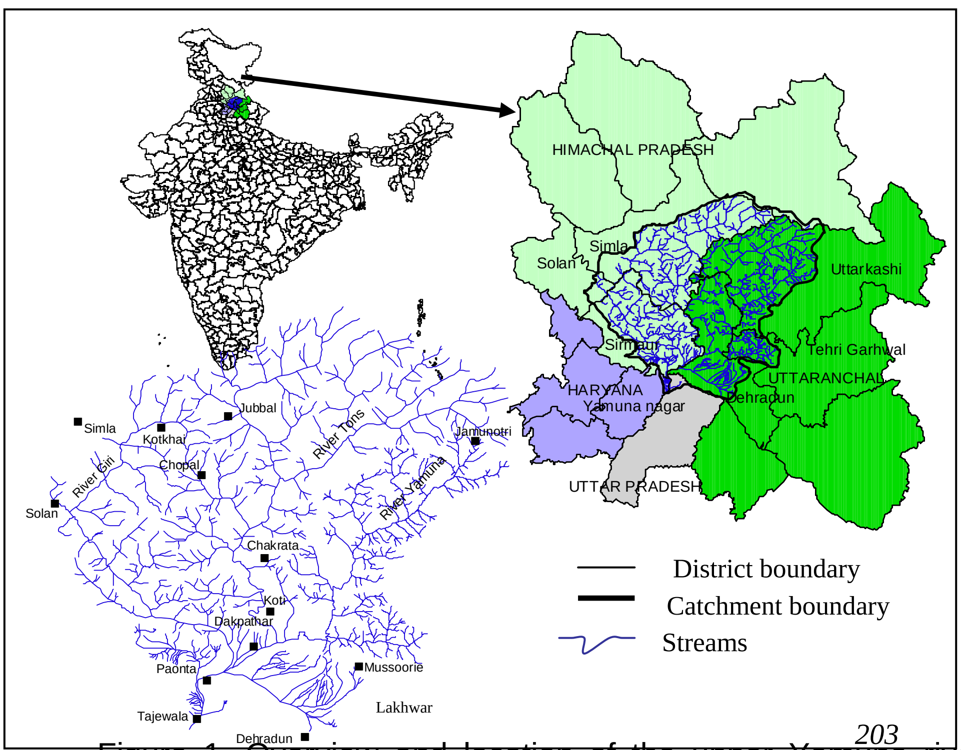

Figure 1. Overview and location of the upper Yamuna river catchment (Capital letters give the name of the State, and small letters give the name of the district in the map)

58' N, 78° 27' E) in the Lower Himalayas at an elevation of about 6320 m above mean sea level (msl) in the district of Uttarkashi of Uttaranchal. After flowing in the south-westerly direction for about 120 km, it is joined by its principal tributary, the Tons near Dakpathar. From the west another tributary, the Giri, that rises further northwest of the Tons, joins the main river near Paonta. The combined stream then forces its way through the lower Shiwalik range and enters the plains near Tajewala where a weir has been in existence for over 100 years at an elevation of 600 m above msl. The total length of the river till Tajewala is 172 km.

The Upper Yamuna catchment till Tajewala has a total catchment area of approximately 11 600 km^2. There are three major sub-catchments namely, the Yamuna sub-catchment from Jamnotri till Lakhwar, the Giri sub-catchment, and the Tons sub-catchment. Figure 1 gives the overview and location of the basin.

Rainfall spells in the catchment are generally associated with monsoon or late monsoon depressions either from Bay of Bengal or Arabian Sea. The catchment receives around 140 cm of rainfall throughout the year, out of which around 75% is received during the monsoon months only. In all, there are 14 daily rainfall recording raingauge stations in the catchment maintained by the Indian Meteorological Department. Daily observed temperature data obtained for a period of 1974–1992 for weather stations show that the minimum and maximum temperatures range from -6.0°C to 43.0°C.

Data preparation including compilation of climatic and hydrologic data

Inputs for simulating land phase of the hydrological cycle can be categorized into spatial and non-spatial data. Non-spatial data includes data on weather, crop, fertilizer use, irrigation, and groundwater table. Spatial datasets are pertaining to watershed characteristics, landuse, soil type, drainage network, and weather stations. Table 1 lists various input data required for modeling for the river basin.

Table 1. Various inputs and sources of information for the required datasets

Subject area	Data basis	Source and map scale
Basic data	Boundaries of the river basin, administrative boundaries, stream networks	Survey of India (SoI); 1: 50 000
Climatic data	Mean monthly and daily precipitation, maximum and minimum temperature, solar radiation, wind speed, potential evaporation	Indian Meteorological Department (IMD)
Soil-physical data	Soil characteristics (% silt, sand, clay, rocks), field capacity, wilting point, hydraulic conductivity, depth to water table, properties for different soil layers varying with depth	National Bureau of Soil Survey and Land use planning (NBSS& LUP); 1:250 000
Landuse data	Ground cover	SoI, Satellite Imageries, State Agricultural Board 1:50 000 and 1:250 000
Hydrogeological data and groundwater fluctuation	Groundwater-bearing lithologic units, transmissivity, hydraulic conductivity, groundwater levels, fluctuations, hydrochemical data, water use (pumping and extraction)	Geological Survey of India (GSI); National Thematic Map Organization (NATMO), CGWB, SGWB, 1: 250 000

Topography data	Elevation contours	SoI; 1: 50 000
Gauge data	Daily river flows	Chander et al. (1984), Central Water Commission, Ministry of Water Resources
Crop and fertilizer data	Nitrogenous fertilizers use, crop yields and types of crops; gross and net cropped areas	Fertilizer Statistics, Fertilizer Association of India (FAI)
Water quality data	Surface water quality (Nitrate concentrations), ground water quality (nitrate concentrations)	Central Pollution Control Board (CPCB); State PCB, Station-wise data

Results and discussion

The daily measured stream flow datasets were available for two gauge stations in the Lakhwar sub catchment, for a period of four years (1976–79) for five months in a year covering the complete monsoon cycle (June–October), namely Damta (an upstream station); and Lakhwar (downstream station). Three years data that is, 1976–1978 have been used for calibration and data for 1979 have been used for validation purposes. Nitrate loads in relation to surface water quality are calibrated and validated by using observed surface water quality data. Results of calibration and validation are presented in figures 2 to 4. Model performance is evaluated using R^2 criteria, and Nash and Sutcliffe simulation efficiency criteria (Nash and Sutcliffe,

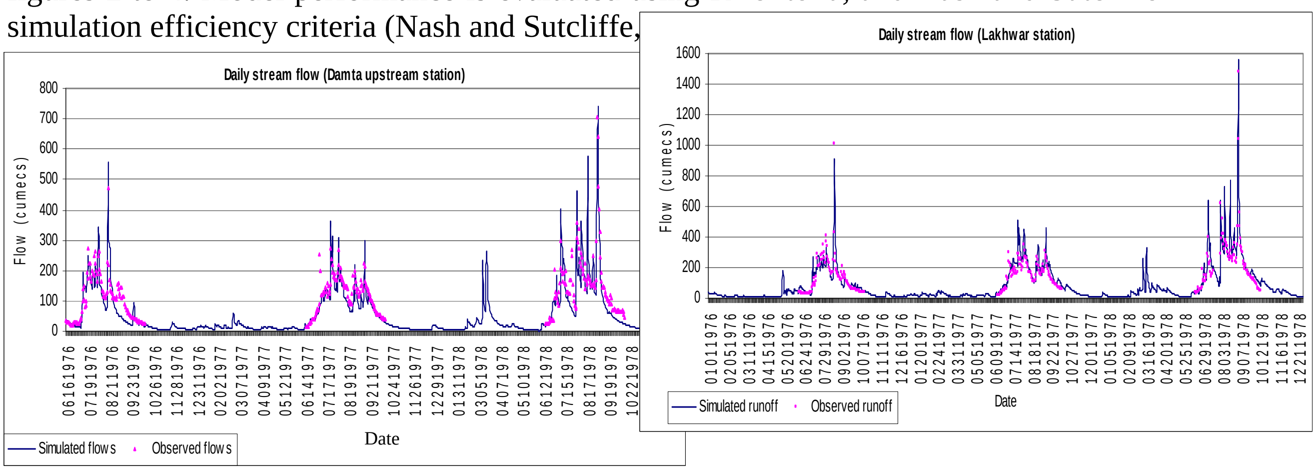

Figure 2. Comparison of daily flows (cumecs) - Damta and Lakhwar: calibration period (1976 to 1978)

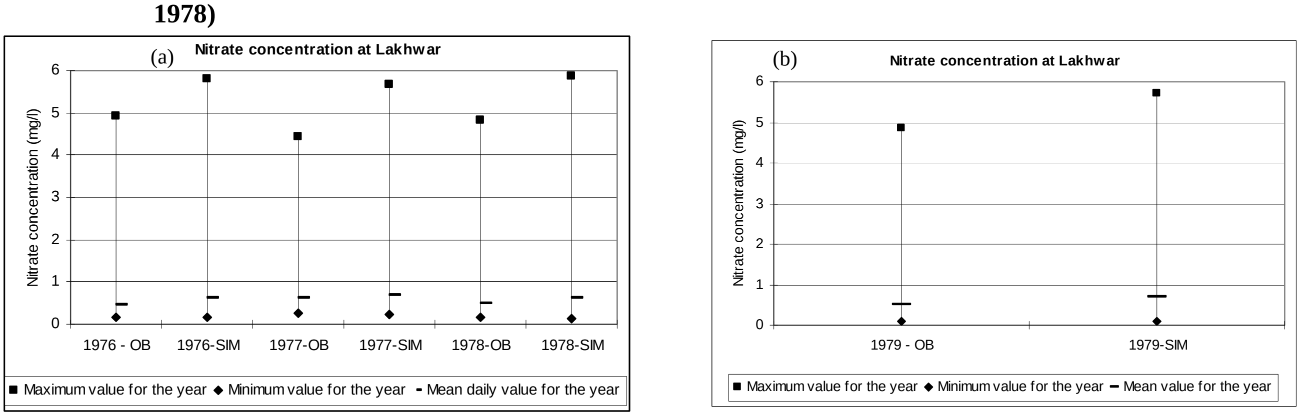

Figure 3. Mean daily nitrate concentrations (mg l^{-1}) in calibration (a) and validation (b) period at Lakhwar (OB – observed values, SIM-simulated values)

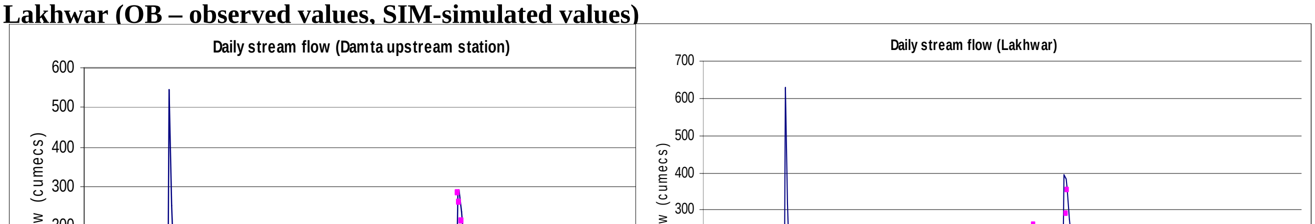

Figure 4. Comparison of daily flows (cumecs) - Damta and Lakhwar stations: validation period (1979)

Evaluation of the nitrification and denitrification processes

Published experimental data for 'nitrification' has been taken from the field work carried out under a doctoral programme (Tong, 2003) and data from Bremner and Shaw (1958) in Havlin et al. (1999) has been taken to study denitrification under various soil moisture conditions. Two cases have been described, one, where θ > 0.95 FC that is, θ = FC, and second, where θ < 0.95 FC that is, θ = 0.75 FC. A

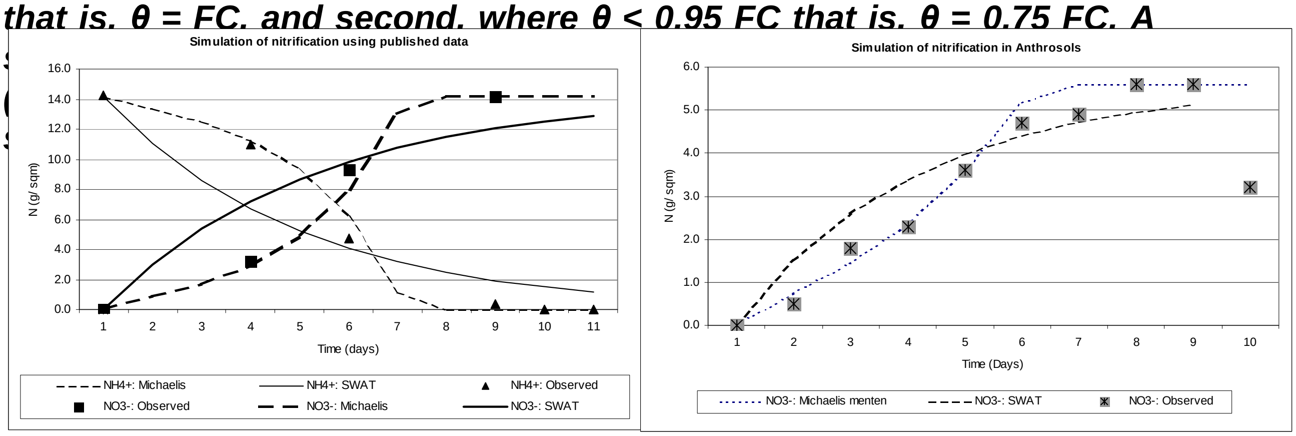

Figure 5. N transformation due to nitrification with Entisol and Anthrosol

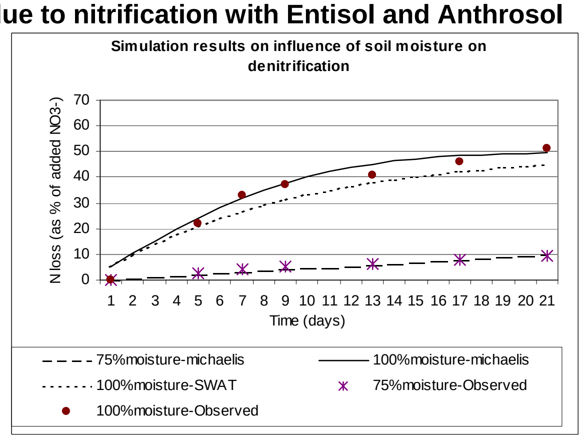

Figure 6. Simulation results of denitrification under soil moisture conditions (for 'soil water content (moisture) (θ)' < 0.95 FC, SWAT does not simulate for denitrification)

Discussion of results

SWAT model has been calibrated and validated in the above sections. Both the performance criteria namely R^2 criteria, and Nash and Sutcliffe model efficiency criteria were used to evaluate the simulation and validation of streamflow and nitrates in relation to surface water quality. The values of these components for calibration and validation periods are compared in table 2.

Table 2. Model performance comparison for calibration and validation periods

Component	Calibration		Validation	
	R^2	Nash and Sutcliffe	R^2	Nash and Sutcliffe
Stream flow				
Damta upstream on River Yamuna	0.725	0.710	0.935	0.925
Lakhwar downstream on River Yamuna	0.771	0.741	0.777	0.749

The simulation results of mean daily nitrates in surface runoff showed deviation in comparison with the observed values. The proposed kinetics for representing dominant N transformation processes showed that nitrification occurs as a result of microbial action and controlled by microbial population present in the soil layer. The microbial population grows in favourable environmental conditions, and takes time to stabilize once favourable conditions develop in soil. Thus the kinetics based on Michaelis-Menten equation controls the rate of reaction during the initial days followed by rapid rate of reaction after stabilization of population. Figure 5 shows the controlled release of nitrates using proposed Michaelis-Menten equation. This agrees well with the observed nitrates in soil that occur due to nitrification process. The model performance criteria results both for nitrification and denitrification modules are shown in table 3.

Table 3. Model performance comparison for nitrification and denitrification modules

Experimental station	R^2	Nash and Sutcliffe
Denitrification: SWAT, Case 1; (Case 2: θ < 0.95)	0.93; (does not simulate)	0.86; (does not simulate)
Michaelis-Menten, Case 1; (Case 2)	0.97; (0.97)	0.96; (0.96)
Nitrification: SWAT	0.75	0.83
Michaelis-Menten	0.98	0.98

The model performance criteria results show both R^2, and Nash and Sutcliffe coefficient for Michaelis-Menten-based equations improved compared to first order kinetics in the chosen land phase of the hydrological cycle model.

Denitrification occurs in low moisture conditions of less than 0.95 and upto 0.5 (measured in terms of percentage of field capacity) at low rates. This is especially true in the cases of heavy soils where microsites in soil continue to remain in anaerobic state at low soil moisture conditions occurring on a macro scale (Abbot et al., 1996; Wendland et al., 1993; Havlin et al., 1999; Hansen et al., 1991). SWAT presently does not model the denitrification process for θ < 0.95 of FC. Michaelis-Menten kinetics is able to model this process well as shown in figure 6.

Conclusions

The study makes use of SWAT model (Arnold et al., 1998, Neitsch et al., 2001) to simulate the land phase of the hydrological cycle. The study has been successful in calibrating and validating the model consisting of the streamflow, and nitrates in surface water components. Each of these components were separately calibrated and validated in a sequential manner. Three year data were used for calibration and one year data for validation. The chosen model SWAT is able to simulate the streamflow component of the land phase of the hydrological cycle. The performance of the model in simulating the streamflow for the validation period was found to be good.

Nitrate transport to surface water and groundwater aquifers takes place under the influence of the runoff components of the land phase of the hydrological cycle. SWAT is found to over-predict nitrates in surface runoff. The kinetics of nitrate transformation processes namely, nitrification and denitrification, in the unsaturated soil zone of the land phase of the hydrological cycle determines the transformation and release of nitrates in surface and groundwater aquifers. The comparison of the existing SWAT nitrate transformation first-order kinetics with the proposed Michaelis-Menten mixed order kinetics concludes that Michaelis-Menten mixed order kinetics represents the nitrate transformation processes better than the existing first-order kinetics in the chosen land phase of the hydrological cycle model.

Limitations of the study

The observed data for surface runoff was available for a total time period of only four years, 1976–79. Of these three years datasets were used for model calibration and one year dataset was used for validation. A longer length of record would have been desirable for calibration and validation of various components of the model.

Acknowledgement

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Application of SWAT to Quantify Internal Renewable Water Resources in Iran

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Abstract

Increasing water scarcity and growing demand for food give rise to the need to improve water use efficiency, or water productivity. This has been one of the important objectives in water resources management. A way of achieving this objective is through virtual water trade strategy. “Virtual water” is the amount of water used in producing agricultural products. The “virtual water trade strategy” in a region calls for producing crops of high water productivity and importing crops of low water productivity taking into account comparative advantage from physical, social, and economic points of view. The objective of this project is to provide a systematic assessment of the feasibility of promoting intra-country virtual water trade as a policy option to improve water productivity in Iran. We used the distributed hydrological model, Soil and Water Assessment Tool (SWAT), to quantify the provincial water resource availability. The model is under calibration/validation with SUFI-2 (Sequential Uncertainty Fitting, ver. 2) procedure using measured daily discharge data from 81 discharge stations across the country. The provincial-based water resources availability and crop water productivity as well as socio-economic conditions will be used in scenario analyses to improve the regional cropping structure. In this paper we only discuss the calibration of the hydrologic model for Iran.
Keywords: water scarcity, virtual water trade strategy, hydrologic modelling, provincial water resource availability, prediction uncertainty, Iran

Introduction

Increasing demand for food and competition for water is a growing problem as water scarcity increases due to population increase. As the largest water user, the agricultural sector is under pressure to produce more food with less water by increasing crop water productivity (CWP) (Kijne et al., 2003a). Enhancement of CWP has been one of the important objectives in water resources management (Postel 1999; Gleick, 2003; Rijsberman, 2006). Different methods have been suggested to improve CWP. 1) To improve crops at the genotypes level, 2) to improve water resources management locally to produce more crops with less water; 3) to improve cropping system regionally by adjusting cropping structure and promoting food trade (the latter is the so-called virtual water trade strategy). There is a large literature on the measurement of water productivity and the ways to improve it on scales ranging from individual farms, irrigation districts to river basins (Molden et al., 2005; Ahmad et al., 2004; Wallace and Gregory, 2002; Alizadeh and Keshavarz, 2005). In comparison, the virtual water approach for water stress alleviation is a rather new concept, although food trade is not. Studies linking virtual water trade with improving water productivity have been few and are mostly concerned with international trade. The rationale of the virtual water trade approach lies in the fact that water requirement for producing a unit of crop differs across regions due to variations in climatic conditions and agronomic practices. Water saving can be achieved when food is transferred from areas where water productivity is high to areas where water productivity is low. There are a number of studies that have quantified the scale of water saving embodied in the global food trade. The estimated volume of water saving ranges between 400 km^3 and 600 km^3 in the global trade of cereal crops (Zimmer and Renault, 2003; Oki and Kanae., 2004; Yang et al., 2005). The variations among different studies stemmed partly from different crop water requirement values used in the estimation. Systematic assessment of water resources availability and crop water requirement is the basis for estimating water demand in food production and the volume of virtual water flows. It also provides a foundation for examining water productivity across regions. This study aims to systematically assess the role of intra-country virtual water trade as a policy option to improve water productivity in individual regions and for the whole country of Iran, taking into account natural and socio-economic constraints.

Information concerning water resource availability, taking into account its spatial and temporal variation, is scarce in Iran. To our knowledge the national water planning report by Jamab Consulting Engineers Ministry of Power (1998) is the only available source which provides water resource availability data in surface water resources and harvestable ground water resources at regional scale. There is, however, a lack of detailed information concerning the hydrological components, affecting the water resource availability over the time and space in the country. Despite this availability of water resources information, the spatial and temporal resolution is not sufficient to fulfill the main objective of the current project in accounting for virtual water content and promoting the virtual water trade strategy. For this reason we use SWAT (Soil and water assessment Tool- Arnold et al., 1998) to estimate sub-provincial water resources availability and crop water requirement at a monthly time step. Explicit quantification of the hydrological components, e.g., surface water (blue water), soil water (green water storage), evapotranspiration (green water flow), and ground water recharge, etc. is one of the advantages of our modeling which provides a precise basis in our future scenario analysis in terms of water resource management in the country.

2. Description of the study area

Iran, with an area of 1,648,000 km^2 is located between 25 and 40 degrees north latitude and 44 to 63 degrees east longitude. The altitude varies form -40 m to 5670 m, which has a pronounced influence on the diversity of the climate. Although most part of the country can be classified as arid and semiarid, Iran has a wide spectrum of climatic conditions. The average annual precipitation is about 252 mm, while northern and western provinces receive about 2000 mm and central and eastern parts of the country receive less than 120 mm per year. A great range of extreme temperatures from -44 ºC in the south west to 56 ºC along the Persian Gulf coast is also observed in Iran.

According to the national water planning report by Jamab (1998), Iran could be divided into eight main hydrologic regions (HR) comprising 37 watersheds in each region. We considered the Jamab hydrologic regions as the basis in our study. Therefore, eight main hydrologic regions were delineated and used to simulate the hydrology of the country (fig. 1).

In HR1 Sefid Rud and Hraz are the main rivers. Sefid Rud River is 670 km long and rises in NW Iran and flows generally east to meet the Caspian Sea. It is Iran's second longest river after Karun. A storage dam on the river was completed in 1962. Haraz is a river in Northern Iran that flows northward from foot of Mount Damavand to the Caspian Sea cutting through Alborz. A storage dam has been constructed on Lar River which is an upstream tributary of the Haraz River. There are many other short rivers which originate from Alborz Mountains and flow toward Caspian Sea. This is a water-rich region in the country.

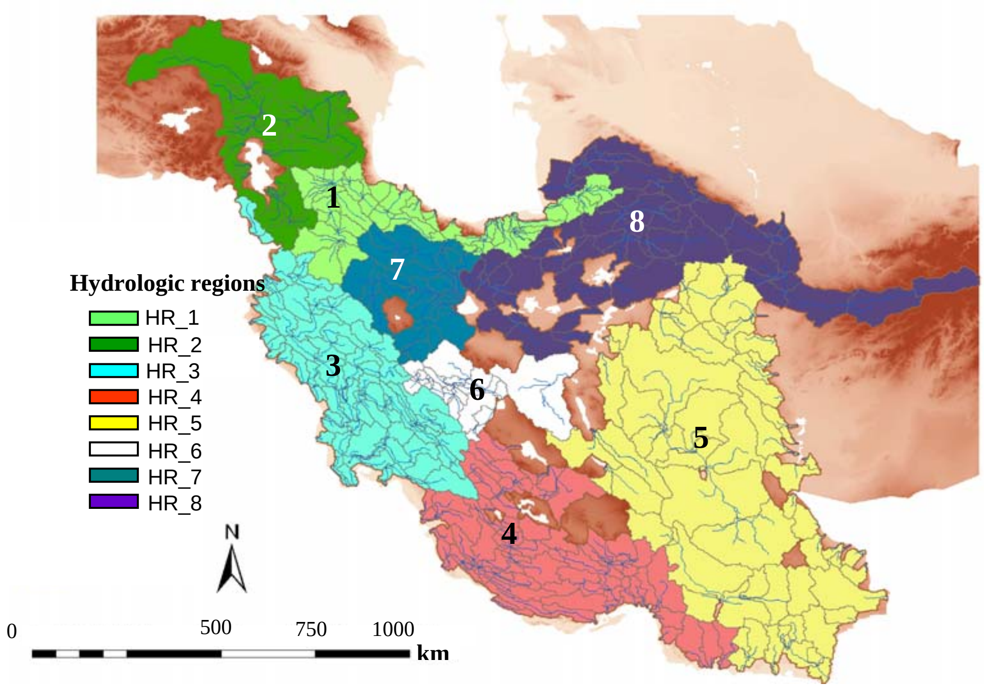


Figure 1. Main hydrologic regions in Iran

In HR2 exists a permanent salt lake, Lake Urmiyeh, to which several permanent and ephemeral rivers flow. Aras is an international river in HR2, which originates from Turkey and flows along Turkish-Armenian border to Iranian-Armenian border and then to Iranian-Azerbaijan meeting finally with the Kura River, which flows into the Caspian Sea. This hydrologic region is one the most important areas of the country for agricultural activities as the water resource availability is high.

In HR3, Karkheh and Karoun are the main rivers, which are the most effluent and navigable rivers in Iran, receiving many tributaries. HR3 is an arid and semi-arid region of the country. Jarahi, Zohreh, and Sirvan are the other main rivers in the region. Several storage dams have been constructed on the rivers and operated for many years. Water resource availability is high in the region but due to poor climatic conditions agricultural performance is moderate.

In HR4 all the rivers and streams provide relatively moderate water resources availability in terms of the agricultural activities. Except the Kor River that meets the

Bakhtegan Lake at the end of its journey, Dalki, Mond, Kol, and southern coastal tributaries flow through this hydrologic region and end in Persian Gulf.

In HR5 there are no major rivers. The region is classified as very arid. The only important rivers of the region are Halil Rud and Bampoor.

In HR6, the famous Zayandeh Rud is the only main river, which originates from Zagros Mountains and flows west to end in the Gavkhooni marsh after passing a meandering 420 km. There is a storage reservoir on the river from which the average annual outflow is 47.5 $m^3\ s^{-1}$.

In HR7, Karaj, Jaj Rud, Ghom Rud, and Shor Rud are the main tributaries. The rivers originate from both Alborz and Zagros Mountains and flow toward a Salt Lake at the central plateau of Iran.

In HR8, of six river basins Atrak, and Hari Rud are the most important. Atrak is a fast-moving river which begins in the mountains of Northeastern Iran, and flows westward to end at the south-eastern corner of the Caspian Sea. Hari Rud is a riparian river recharged from tributaries of both Iran and Afghanistan. Table 1 shows some pertinent characteristics of the eight watersheds.

Table 1. Watershed characteristics of eight main hydrologic regions in Iran

Hydrologic region	Area (km^2)	Mean Zonal precipitation[a]	Number of sub-basins	Dominant Land use[b]	
				Landuse	%
HR 1	97478	454-1300	66	Grassland	61.9
HR 2	131973	367-424	58	Grassland	54.2
HR 3	185042	356-728	92	Shrubland	53.4
HR 4	196329	191-369	87	Shrubland	71.1
HR 5	459309	70-196	68	Baren/Sparesly vegetated (BSVG)	75.1
HR 6	66654	120-217	26	BSVG	65.2
HR 7	82268	287	43	Shrubland	52.8
HR 8	256553	161-359	67	Shrubland	52.2

[a] Available from Jamab (1998) report.
[b] Extracted from USGS land use data base using SWAT.

3. Model inputs and model setup

3.1 Model input

We used the ArcGIS-SWAT in this project (Olivera et al., 2006).The data required for this study was obtained from different available sources. Digital Elevation Model (DEM) was extracted from Global U.S. Geological Survey's (USGS) public domain geographic database (spatial resolution 1 km). Landuse map extracted from USGS landuse map (1 km spatial resolution) with 24 landuse/land cover classes. Soil map was obtained from the global soil map of Food and Agriculture Organization of the United Nations (FAO,1995) where soil data is provided for 5000 soil types comprising two layers (0-30 cm and 30-100 cm depth) at a spatial resolution of 10 km. Further soil physical properties required for SWAT were obtained from estimates of Schoul and Abbaspour (2007). The digital stream network, administrative boundaries depicting country and provinces boundaries, and reservoirs/dams were available from National Cartographic Center of Iran which provides information at a spatial resolution of 1 km.

Weather input data needed by SWAT in this study (daily precipitation, maximum and minimum temperature) were obtained from the Public Weather Service of Iran Meteorological Organization for more than 150 synoptic stations. Distribution of the selected stations across the country was reliable as the gauging station network is dense in mountain areas. Time duration of the available data varied from 1980 to 2004 depending on the age of the weather stations. Most of the stations provided data from 1983 to 2004. We used the SWAT-WXGEN weather generator model (Sharpley and Williams, 1990) to fill in the gaps in measured records. Daily reservoir/dam inflow-outflow data were obtained from Water Resources Management Organization (WRMO) of Iran. The river discharge data required for calibration-validation were obtained from Jamab Consulting Engineers and Ministry of Power in Iran for more than 90 hydrometric stations. The available time period of the river discharge data for most of the stations was from 1977 to 2002.

A threshold drainage area of 600 km^2 was selected in watershed delineation process. A dominant soil, landuse and slope type within each sub-basin was used to develop the soil and landuse data in the model. To make a more realistic simulation of hydrology, reservoirs/dams were added to the model. Only large reservoirs/dams were included in the model because of a lack of information for all the reservoirs and also of their negligible influence on the hydrological water budget.

The simulation period for calibration processes was set to 16 years from 1987-2003 considering 3 years as the warm up period. Half of the data at each station was used for calibration and the other half for validation.

3.2. Calibration setup

Based on literature, 23 parameters integrally related to stream flow were selected to calibration the model (Table 2).

Table 2. Initially selected input parameters involved in calibration process

Name[a]	Descrioption	Range of values Min.	Max.
v__SURLAG.bsn	Surface runoff lag time (days)	1	24
v__SMTMP.bsn	Snow melt base temperature (ºC)	-5	5
v__SFTMP.bsn	Snowfall temperature (ºC)	-5	5
v__SMFMX.bsn	Maximum melt rate for snow during the year (mm/ºC-day)	0	10
v__SMFMN.bsn	Minimum melt rate for snow during the year (mm/ºC-day)	0	10
v__TIMP.bsn	Snow pack temperature lag factor	0.01	1
r__CN2.mgt	SCS runoff curve number for moisture condition II	-0.5	0.5
v__ALPHA_BF.gw	Base flow alpha factor (days)	0	1
v__REVAPMN.gw	Threshold depth of water in the shallow aquifer for ‘revap’ to occur (mm)	0	100
v__GW_DELAY.gw	Groundwater delay time (days)	0	500
v__GW_REVAP.gw	Groundwater revap. coefficient	0.02	0.2
v__GWQMN.gw	Threshold depth of water in the shallow aquifer required for return flow to occur (mm)	0	100
v__RCHRG_DP.gw	Deep aquifer percolation fraction	0.01	1
v__ESCO.hru	Soil evaporation compensation factor	0.01	1
v__EPCO.hru	Plant uptake compensation factor	0.01	1
r__OV_N.hru	Manning’s n value for overland flow	-0.6	0.6
r__SOL_AWC.sol	Soil available water storage capacity (mm H_2O/mm soil)	-0.6	0.6
r__SOL_K.sol	Soil conductivity (mm/hr)	-0.8	0.8

r__SOL_BD.sol	Soil bulk density (g/cm^3)	-0.5	0.5
r__SOL_ALB.sol	Moist soil albedo	-0.5	0.5
v__CH_N2.rte	Maning's n value for the main channel	0	0.3
v__CH_K2.rte	Effective hydraulic conductivity in the main channel (mm/hr)	0	150

[a] v__: The parameter value is replaced by given value or absolute change; r__: parameter value is multiplied by (1+ a given value) or relative change (See Abbaspour (2007) for more detail).

Three different parameterizations were used to perform the calibration/uncertainty analysis:

i. *Global approach*: relative or absolute change of each parameter was applied globally to the entire country.
ii. *Scaling approach*: relative or absolute change of each parameter was applied based on landuse and soil types. In this approach the same soil type, for example, can be spatially treated differently.
iii. *Regional approach*: in this approach the eight hydrologic regions were treated independently (each having its own objective function) and the relative or absolute change of each parameter was applied globally in each region.

FUFI-2 (Abbaspour et al., 2007, 2004) algorithm was used for calibration-uncertainty analysis. SUFI-2 needs the smallest number of model runs to achieve a similarly good prediction uncertainty in comparison with the other uncertainty analysis techniques (Yang et al., 2007). Conceptually, SUFI-2 seeks parameter ranges where upon propagation most of the measured data falls in the 95 percent prediction uncertainty (95PPU) with the smallest 95PPU band. Hence two indices are used to quantify the strength of the calibration/uncertainty analysis. These are *P-factor*, which is the percentage data bracketed by the 95PPU band (ideal value 100%), and the *R-factor*, which is the average thickness of the band (ideal value 0). The objective function chosen in this project is defined as (Krause et al., 2005):

$$\phi = \begin{cases} bR^2 & \text{for} \quad 0 < b \le 1 \\ b^{-1}R^2 & \text{for} \quad b > 1 \end{cases} \tag{1}$$

and to calibrate an entire region the objective function is simply an un-weighted average of ϕ for all stations within the region:

$$g = \frac{1}{n}\sum_{i=1}^{n}\phi_i \tag{2}$$

Where, R^2 is the coefficient of determination of the simulated and observed stream flow, and b is the slope of the regression line, and n is the number of stations in a region of interest. In the above formulation the objective function is always between (0-1).

4. Results and Discussions

So far we have only performed preliminary calibration of Iran’s hydrologic model. Figure 2 illustrates the average monthly objective function value in the eight hydrologic regions of Iran for the three different parameterization schemes. The Figure shows that regional approach performs the best in all hydrologic regions. Most of the stations located in hydrologic regions 3, 4 and 6 showed quite good simulations as indicated also in Figure 3 where ϕ is shown for individual discharge stations.

Figures 4a, b, and c illustrate the *P-factor* and the *R-factor* for a good, an intermediate, and a poorly simulated station, respectively. According to our preliminary results 7 out of 81 stations have poor simulations, which need to be further

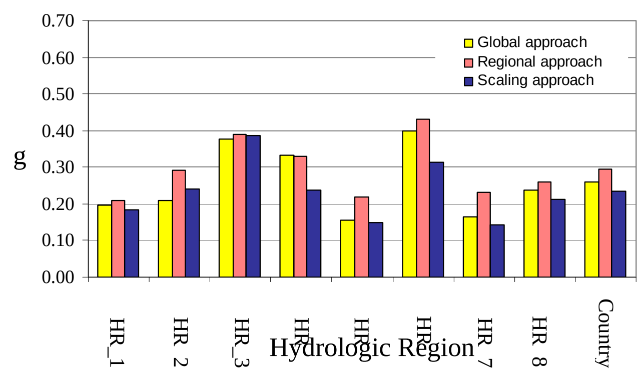


Figure 2. Average monthly value of the objective function for the eight hydrologic regions and the whole country

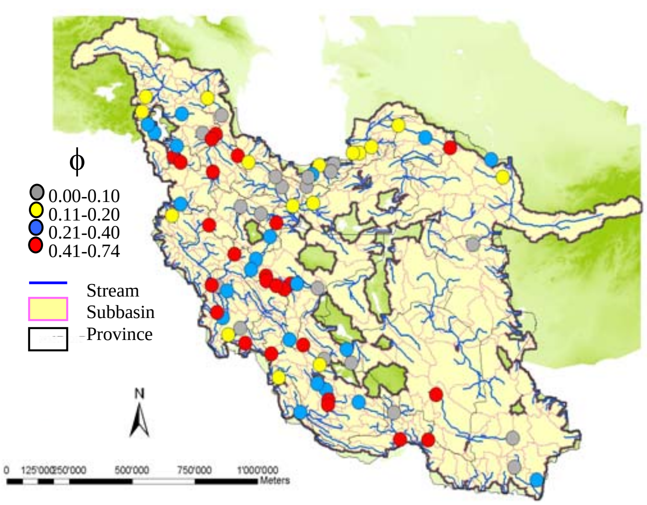


Figure 3. Weighted coefficient of determination (ϕ) at 81 stations across the country

investigated. For some of these stations the main reason is the large "conceptual model uncertainty" (Abbaspour, 2007) due to the presence of processes occurring in the watershed (e.g. water transfer, water use, etc.) that are not included in the model. As there is a lack of information with respect to water transfer and water use, we probably can not improve the results very much in these stations. It is noteworthy that the choice of the objective function is such that these stations have no effect on the calibration process as their contribution to the objective function is rather small because of a small R^2 value.

We calculated the renewable water resources (RWR, calculated as the sum of water yield, WYLD, and deep aquifer recharge, DA_RCHG) of each hydrologic region by aggregating the results from subbasins. The simulated RWR was compared with the existing data reported by Jamab (1998) as shown in Figure 5. It can be seen that in all the hydrologic regions, the Jamab estimated values fall within the simulated uncertainty band.

Conclusion

The hydrologic model SWAT was used for developing a long-term renewable water resources data at monthly temporal resolution and at subbasin level. Assessment of all the hydrological components affecting the renewable water resources are under consideration. The current study showed that ArcGIS interface is a powerful tool in developing the model for such a large area and was used efficiently for parameterization

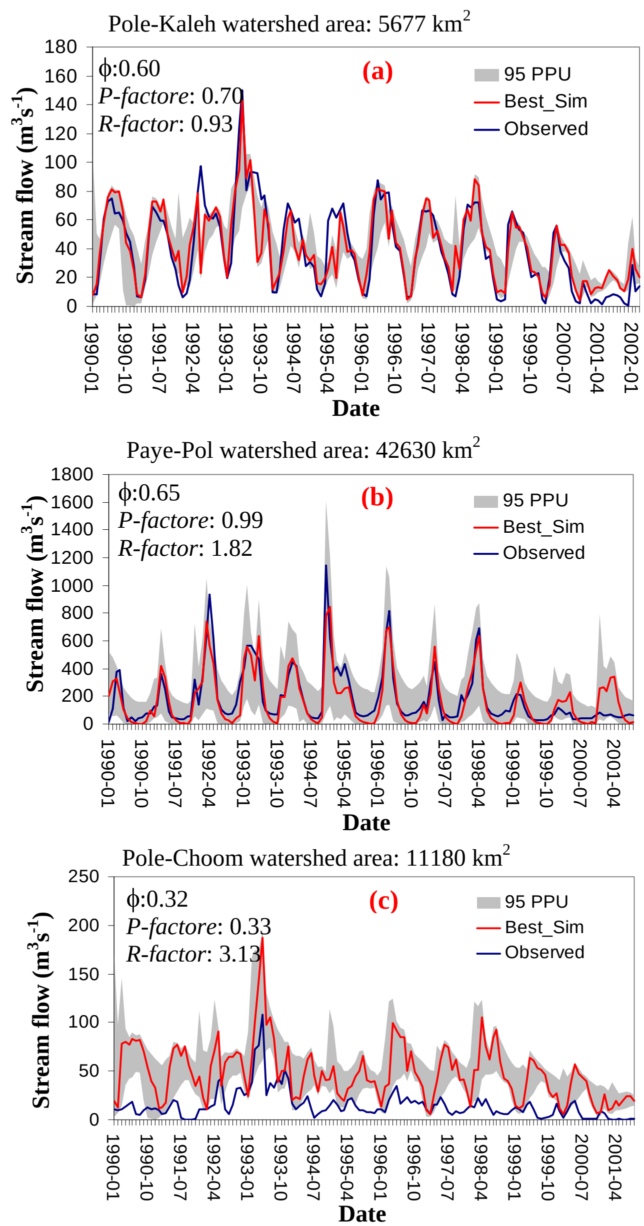
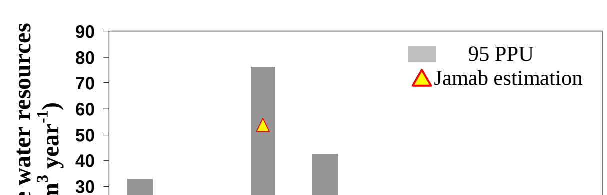


Figure 4. Calibration results for 3 of 81 stations



of the whole area. The final calibrated model will be used to account for actual evapotranspiration, crop yield, and regional water productivity for the purpose of intra-country virtual water trade studies.

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Water Quality Modelling in a Highly Regulated Lowland Catchment

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Abstract

Water quality modelling in the Rhin catchment was done to answer some river basin specific questions dealing with identification of point and diffuse sources in the catchment, simulating the influences of an expected climate and land use change on water quantity and quality and suggestions of possible measures to be done in order to achieve a "good ecological status" of the river and its lakes as required by the Water Framework Directive (WFD).

The Rhin river catchment is a typical lowland river basin, which is highly regulated. These regulations complicate water quantity and quality modelling in the catchment. The research was done by using the ecohydrological model SWIM (Soil and Water Integrated Model), which simulates water and nutrient fluxes in soil and vegetation, as well as transport of water and nutrients in the river network. The modelling period was from 1981 until 2005. After calibrating and validating the hydrological processes at different gauges within the basin with satisfactory results, water quality (nitrogen) modelling was started taking into account the emissions of different point sources (e.g. sewage treatment plants) and identifying the amount of diffuse pollution caused especially by agriculture.

For suggesting some feasible measures to improve water quality and to reduce diffuse pollution considering possible climate and land use changes different reasonable scenarios will be applied in consultation with the Federal Environmental Agency of Brandenburg (LUA).

KEYWORDS: water quality modelling, nitrogen, lowland river system, SWIM, WFD

1. Introduction

Lowland river systems and their catchments are typical ecosystems with small amplitude in altitude, low flow velocity, high ground water tables and a substantial share of typical organic soils, e.g. fens. In former centuries different melioration measures were applied in order to provide areas for agriculture. That's why the lowland areas are often heavily drained. This leads to a reduction of retention time of nutrients, which contributes to eutrophication problems in the river network and coupled lakes, as the nitrogen, introduced to the system by fertilization, has not enough time to be decomposed during its pathway to the river network (mainly by denitrification).

Knowing these problems and being interested in finding solutions for a better ecological status of such a lowland river, LUA requested a modelling support for implementation of the WFD in the Rhin catchment. The WFD requires that all European water systems should achieve a "good ecological status" until 2015. This "good ecological status" should be as near to the reference status of the water body as possible. For this purpose model scenarios can be helpful in order to find reasonable measures for achieving a better ecological status taking into account possible changes of land use, water use or climate conditions.

In general, future water management decisions should be more adaptive, as we are living in a rapidly changing world. It can not be expected that the former conditions will stay unchanged. Some changes in temperature and/or intensity and frequency of extreme events have already been observed in the region. That's why scenarios as different options for a possible future should be taken into account by implementing the WFD.

The LUA was especially interested in answering the following questions:

- What are the shares of point and diffuse sources of pollution, and where are the main areas of diffuse pollution located?
- What is the reference status of the Rhin river system, which should be reached by the WFD?
- What are the best measures to reach the requirements of the WFD?
- Which consequences have possible changes in land use and climate for the river system in the future?

2. Materials and methods

2.1. Study area and data preparation

The study area was the Rhin catchment (drainage area 1716 km²) in the north of the German federal state Brandenburg. The Rhin river drains into the Havel river, a tributary of the Elbe river. The catchment belongs to the lowland part of the Elbe basin, and its altitude is between 19 m and 116 m above sea level. Main average climate and water characteristics measured in the Rhin catchment are listed in Table 16. The location of the basin, discharge gauges and climate station are shown in Figure 18.

Table 16. Average climate characteristics of the Rhin catchment (P, T; station Neuruppin), and mean water discharge and NO_3-N concentration in the outlet (Q, N; gauge Kietz)

variable		unit	period	value
P	precipitation	mm a^{-1}	01/1981-12/2005	524.4
T	temperature	°C	01/1981-12/2005	9.4
Q	discharge	$m^3\ s^{-1}$	11/2001-10/2005	3.73
N	nitrate nitrogen	mg l^{-1}	01/1981-12/2005	0.62

41% of the area is used for agriculture (especially winter crops and maize), 34% is covered by forests (mainly evergreen forests) and 19% by intensive used pastures. The pasture area practically coincides with the fen soil type area (ca. 23% of all soils), which can be used as pasture only due to the intense water regulation system.

The Rhin river network is influenced by more than 300 dams and weirs (see Figure 18). 27 of them are located within the main river course. The large fens and wetland areas near the middle course of the river are strongly drained. A lot of ditches for irrigation and drainage together with the barrages influence the hydrological cycle. Natural discharge behaviour at the five observation gauges within the Rhin catchment can not be recognised. Additionally, the water dynamics is heavily influenced by storage of water in winter time (mainly in the upper course of the Rhin river), which can be later used for irrigation during dry periods, and several in- or outflows of water from or to adjacent catchments.

There are a lot of lakes within the stream (3% of the area is covered by surface water bodies). On the basis of their chemical (e.g. oxygen demand, total phosphorus, and sulphate) and biological (e.g. diatoms) quality they can be clearly distinguished as a less polluted upper part and a higher polluted lower part. Measures to improve water quality and to reach the aims of the WFD are necessary especially in the lower part of the basin, which is classified as an endangered water body due to its trophic status and oxygen demand. But looking at the actual status, one can ask whether it is possible at all for the lower part of the river to reach the "good ecological status" and the "good chemical status" as required by the WFD. However, the modelling could help to clarify these questions.

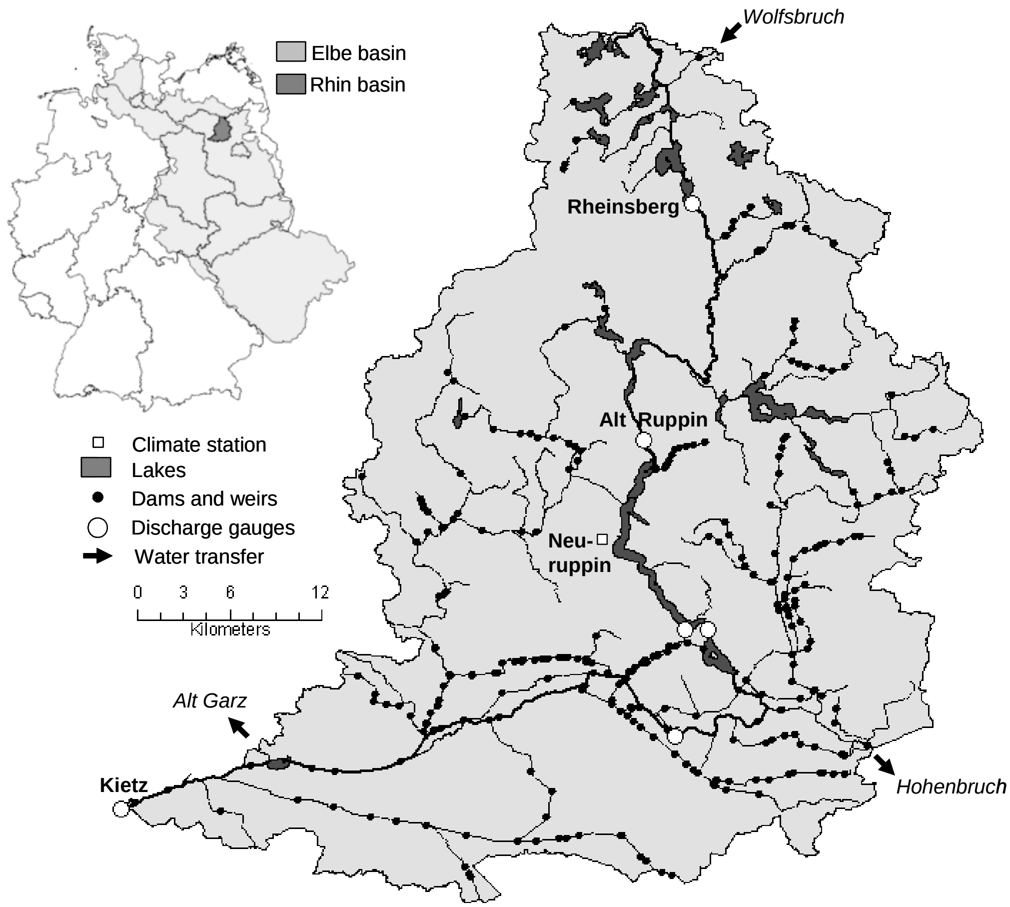

Figure 18. Location of the Rhin catchment within the Elbe basin together with borders of the federal states of Germany (small map) and location of rivers, lakes, discharge gauges, climate station, water transfer points and weirs within the Rhin catchment (large map)

To setup the model the study area was defined by several raster maps (digital elevation (DEM), soil (BÜK 1000), land use (Corine2000) and subbasin) with a resolution of 50 m x 50 m. Climate data (temperature, precipitation, solar radiation and air humidity) provided by the German Weather Service were interpolated to the centroids of every subbasin by an inverse distance method using 37 climate stations in and around the Rhin catchment and 11 additional precipitation stations within the basin. According to the subbasin map delivered by the LUA the Rhin basin was divided into 218 subbasins.

For calibrating the model following measured data provided by the LUA were used: daily measurements of discharge at the three gauges Rheinsberg, Alt Ruppin and Kietz and fortnightly measurements of nitrate nitrogen concentrations at the last Rhin gauge Kietz for different periods between 1981 and 2005. Linear interpolation was necessary for calculating the daily nitrogen loads. The LUA provided one detailed and two general informations about three water transfer points from or to adjacent catchments (Wolfsbruch, Hohenbruch and Alt Garz) as well as information about point sources.

Fertilization data for winter wheat and intensive grassland were taken from the Havel management project (Voß, 2007).

2.2. Model SWIM

The dynamic eco-hydrological model SWIM (Soil and Water Integrated Model) was developed on the basis of SWAT (Arnold et al., 1994) and MATSALU (Krysanova et al., 1989). SWIM simulates hydrological processes, vegetation and nutrient cycles at the river basin scale (Krysanova et al., 1998, 2000) by disaggregating the basins to subbasins and hydrotopes, where the hydrotopes are the highest disaggregated units (sets of elementary units in a subbasin with the same soil and land use types). Up to ten vertical soil layers can be considered for hydrotopes. It is assumed that a hydrotope behaves uniformly regarding hydrological processes and nutrient cycles.

Water fluxes, nutrient dynamics and plant growth are calculated for every hydrotope and then lateral fluxes of water and nutrients are moving to the river network. After reaching the river system, water and nutrients are routed along the river network to the outlet of the simulated basin.

The hydrological system is split into four compartments in the model: the soil surface, the soil layers, the shallow aquifer and the deep aquifer. Processes taken into account for the soil zone are surface runoff, infiltration, evapotranspiration, percolation and interflow. Hydrological processes in the aquifer zone are groundwater recharge, capillary rise to the soil profile, lateral flow and percolation to the deep aquifer.

The nutrient modules include pools of active and stable phases, inorganic and organic phases and nutrients in the plant residue for nitrogen and for phosphorus. The following processes: mineral and organic fertilization, input with precipitation, mineralization, denitrification, plant uptake, leaching to groundwater, and losses with surface runoff, interflow and erosion for both nitrogen and phosphorus are taken into account. While passing the soil by flowing to the river system the nutrients are subject to retention and decomposition processes, whose rate and intensity are described by special parameters (Hattermann et al., 2006). The retention parameters are taken from literature and can be calibrated.

Mineralization of fresh organic nitrogen (crop residue) and active organic nitrogen (soil humus) is a function of C:N ratio, C:P ratio, temperature and water content in soil.

Denitrification occurs in times with oxygen deficit, usually associated with high water content in soil. In addition it is a function of soil temperature and carbon content.

The vegetation is allowed to take nutrients from any soil layer that has roots. Uptake starts at the upper layer and proceeds downwards until the daily demand is met or until all available nitrogen is used.

For simulating the Rhin catchment some additional model assumptions had to be set. Firstly, it was necessary to define subbasins with additional in- or output of water and nutrients from or to adjacent catchments according to the information delivered by the LUA. The water transfer point Wolfsbruch (inflow) was defined by adding daily measurements of water discharge and the corresponding calculated nutrient load. The outflow point Hohenbruch could only be taken into account by using long time mean values of discharge and load. Alt Garz was defined by the constraint that in winter time and in times with discharge higher than 5 m^3/s, the outflow and nutrient load of the corresponding subbasin are nearly halved.

Secondly, the LUA delivered information on location and output of point sources (sewage treatment plants), which were added to the daily nutrient amount of the corresponding subbasins. Unfortunately these data have a high uncertainty, as they are annual values estimated from one or two measurements per year. As there were no better data on point sources, they were used during calibrating the nitrate nitrogen concentration.

Thirdly, a simple wetland method (Hattermann et. al, 2007) was introduced in the model, as about 40% of the catchment belongs to fens or ground water influenced soils. This method allows increasing the plant uptake of water and nitrogen from groundwater in wetland areas in times, when the supply of water and nitrogen in soil is limited.

2.3. Evaluation of model results

To describe the quality of simulated model results different criteria of fit can be taken into account. In this study the efficiency criteria of Nash and Sutcliffe (1970) (E) and the relative difference in balance (B) were used. E is a measure to describe the squared differences between the observed and the simulated values on a daily time step. B shows the long-term differences of the observed values against the simulated ones in percent for the whole modelling period. The efficiency can vary from minus infinity to 1 and should be as near as possible to 1, while the deviation in balance has its best values near 0. The Nash-and-Sutcliffe-Efficiency and deviation in balance were used for discharge as well as for nutrient loads.

3. Results and Discussion

3.1. Discharge calibration

The discharge gauge Kietz is partly influenced by the Havel and Elbe rivers. That’s why there are only little discharge measurements available and only a short time period could be used for calibration.

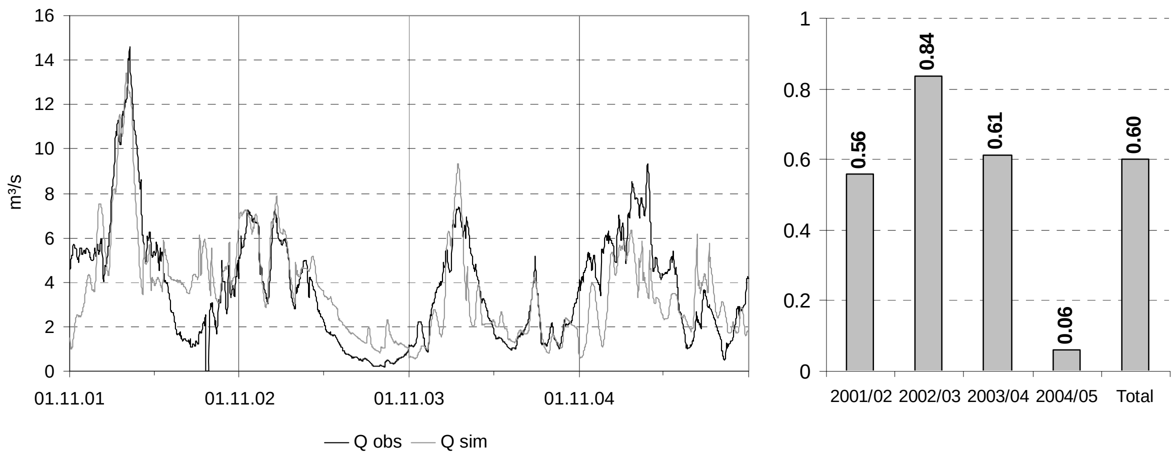

Figure 19. Comparison of daily observed and simulated discharges at the gauge Kietz (left), and corresponding efficiencies for the hydrological years and the whole period (right)

For the hydrological four-years-period November 2001 to October 2005 the best obtained model efficiency was 0.6 with large differences between the single years (0.06 to 0.84) as shown in Figure 19. The problems are more pronounced during the summer months (especially first two summers). This could result from the water management system in the Rhin river. Water is stored in the upper part of the basin and then used for irrigation (artificially increased evapotranspiration), which can lead to lower discharge than expected according to the precipitation amount. In August 2002 the extreme Elbe flood occurred. The lower parts of the Havel and partly Rhin lowlands were used to cut the flood peak of the Elbe river by opening some polder areas for flooding to protect downstream Elbe regions. That's why the Rhin discharge was strongly influenced and for some days even interrupted. But the next year 2003 had very dry conditions, and the overestimation can not be explained with flood events. Most probably, here another factor should be taken into account: an underestimation of evapotransration processes in the lower part of the Rhin catchment with high ground water tables. In general, it seems that evapotranspiration should have different intensities in the upper and the lower parts of the basin due to natural conditions and/or human interventions described above. Calibrating the three gauges Rheinsberg, Alt Ruppin and Kietz (see Figure 18) and comparing their best parameter settings a continuous decrease of the evapotranspiration correction parameter *thc* (meaning an increase in evapotranspiration) was necessary to get the results shown in Figure 20.

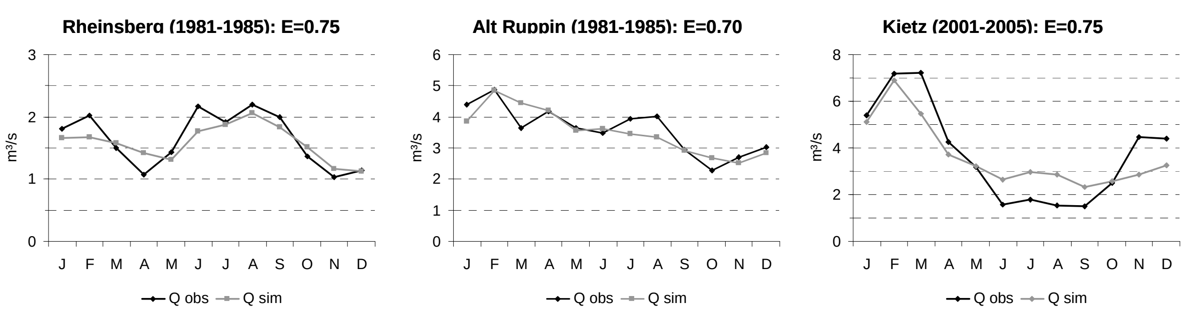

Figure 20. Comparison of the observed and simulated monthly average discharges for three discharge gauges of the Rhin basin (different periods due to data availability)

Figure 20 shows comparison of the mean monthly average discharges for three gauges along the Rhin river. Quite good efficiencies and typical seasonal dynamics could be achieved (Rheinsberg: maximum flow in summer; Kietz: maximum flow in winter, minimum in summer; Alt Ruppin: more or less continuous decline in discharge during the year). For a lowland catchment with regulations the obtained efficiencies between 0.7 and 0.75 are very good. But still the question is, whether the different evapotranspiration parameters were needed due to different locations within the basin, or due to different time periods compared (time periods of dissimilar political and water management regimes). However, the most important problem was probably missing or imprecise information about real water management in the Rhin catchment.

3.2. Nitrogen calibration

Nitrate nitrogen calibration was done for the gauge Kietz by taking into account point sources (sewage treatment plants). However, more important for the amount of nutrients in a river is diffuse pollution coming mainly from fertilized fields. The nutrient transformation processes are highly influenced by soil conditions (water content and temperature).

Figure 21 shows model results for the calibration period from November 2001 to October 2005. Especially the two years in the middle of the period are well reproduced, whereas the peak in the first year is highly underestimated, and concentrations in the last year are overestimated. It seems that it is difficult to reproduce better measured data during extreme events (like high flood in the beginning of 2002 or the winter following an extremely dry period in 2003).

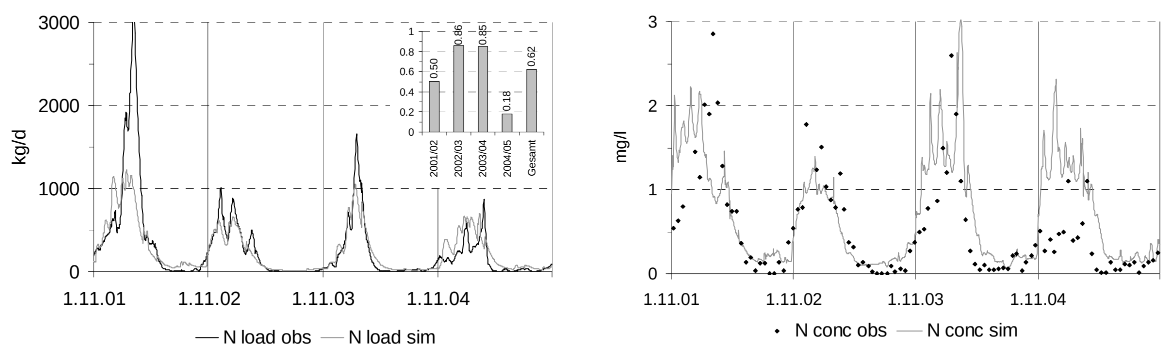

Figure 21. Observed and simulated nitrate nitrogen loads with model efficiencies (left) and NO_3-N-concentrations (right) for the gauge Kietz in the calibration period November 2001 to October 2005

During the nitrogen calibration it was seen, that the measured concentrations could not be reproduced without assuming retention of nutrients derived from point sources while passing through the river network. Of course, this assumption is highly arguable due to the high uncertainty connected to the provided point source data, but it was necessary for good model results.

3.3. Validation

After finding the best parameter combination for the four-years-period in the 21st century, the model was tested for the 25-years-period from 1981 until 2005. Due to lack of discharge data at the gauge Kietz for almost 20 years, the validation was only done by comparing the measured nitrate nitrogen concentrations for these 25 years with the simulated ones. Figure 22 shows both water discharge and nitrogen concentrations to facilitate interpretation of the results for concentrations.

In general, the seasonal dynamics of the measured nitrate nitrogen concentration is simulated satisfactorily. There are peaks in winter time and low amounts of nitrogen in summer months. No trends can be observed. But it can be seen, that in two years of low winter discharge (1989 and 1996) the measured nitrogen concentration is reproduced quite well, whereas in each case in the following two years the concentration peaks are highly overestimated.

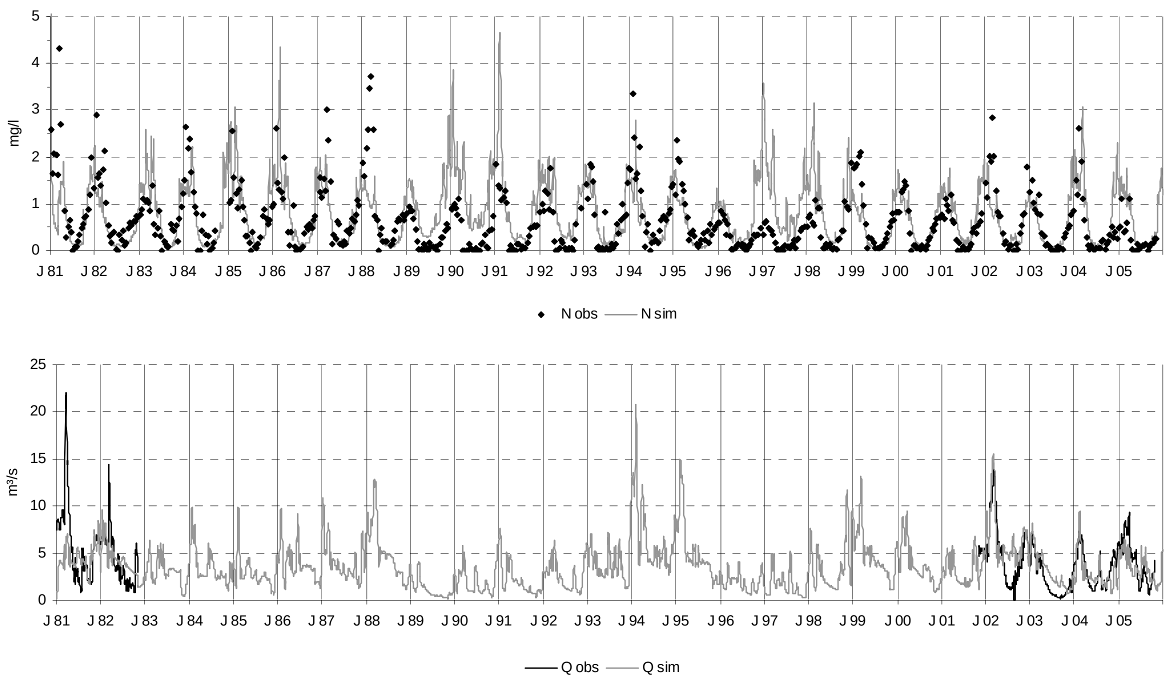

Figure 22. Validation of nitrate nitrogen concentrations in Kietz (upper graph) and the corresponding measured water discharge (lower graph)

It is not quite clear why the model behaves in such a way, and it is not easy to explain without knowing whether the simulated discharge is correct or not. Perhaps the reason for the overestimation of nitrogen can be found in a wrong assumption of a threshold parameter of water content for denitrification. Denitrification starts in case the water content is higher than the threshold. This threshold was set to 0.7 for the calibration period (in Scheffer and Schachtschabel (1998): denitrification processes start with a water content of soil above 70-80%). Perhaps denitrification processes occur also in times with lower soil water content. It is known that in soils with easy available carbon (as in the fens of the Rhin catchment) denitrification is also possible with water contents lower than 60-70% because of the high activity of microorganisms (Scheffer & Schachtschabel, 1998). Another possibility is different water management in years with lower discharge (e.g. no water transfer during winter time), and as a consequence more water with lower concentrations at the outflow Kietz.

3.4. Sources of nitrogen load and climate sensitivity

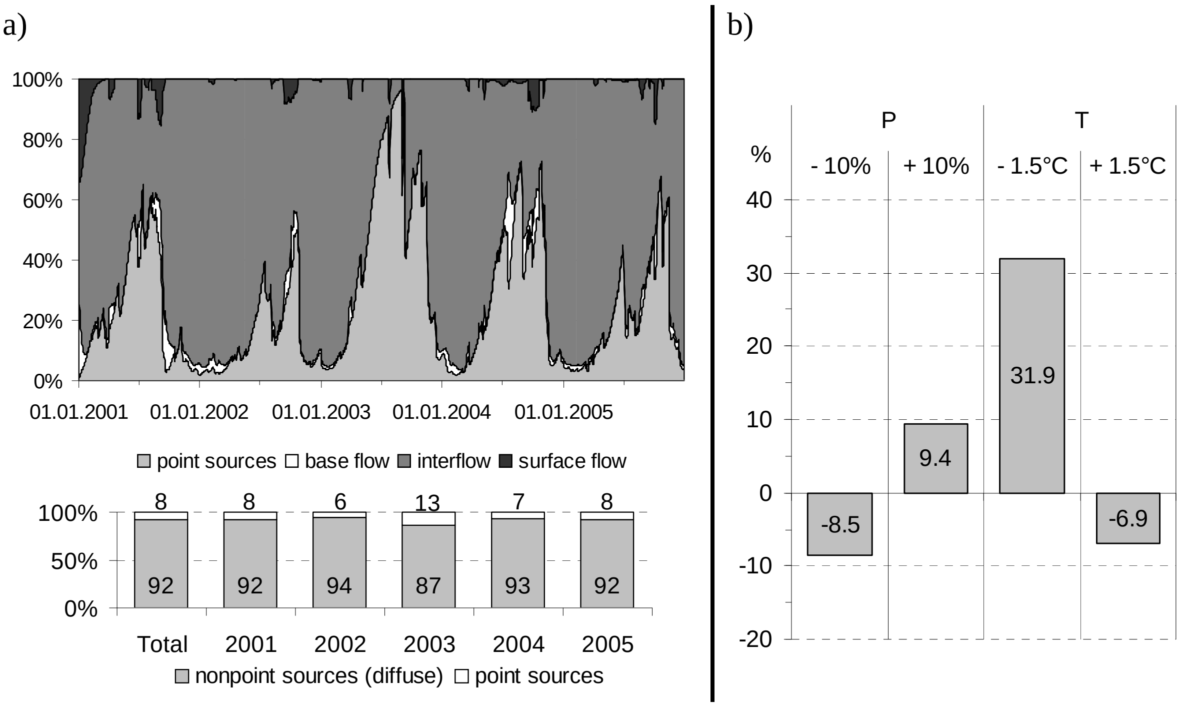

Figure 23. Two model experiments: a) Percental composition of the nitrate nitrogen load at the gauge Kietz coming with different pathways (above) and mean annual fractions of point and diffuse sources of the total nitrogen load at Kietz for the years 2001 to 2005 (below); and b) Influence of changing precipitation (P) and temperature (T) on nitrate nitrogen load at the outflow in Kietz (period 2001 to 2005)

A special interest of the study was to identify sources of nutrient loads in the Rhin river and to define the reference status of the catchment. For this purpose different simulation experiments were done. Some results can be seen in Figure 23 a). A clear seasonality is obvious. In summer about 50% of the nitrogen simulated at the outflow of the Rhin catchment originates from point sources, and in winter almost 100% of the measured nitrogen loads comes with interflow. For the whole five years period 87% of the nitrogen load comes with interflow, 3% with base flow, 2% with surface flow, and 8% from point sources. These percentages differ slightly between the years mainly due to differences in precipitation amount and occurrence of dry or wet conditions. The very dry year 2003 with rare precipitation and low water level in the river has the highest percentage of point sources; whereas in the very wet year 2002 the fraction of point sources is lower than average. Especially in winter the contribution of diffuse pollution from arable land is high caused by washing out soils by precipitation. In summer the most of soil nitrogen on agricultural and vegetation covered land is uptaken by plants. Additionally, there is lower discharge, so that the percentage of point sources is higher. Taking into account the map outputs of the SWIM model it can be seen that denitrification and mineralization are maximal in areas with organic soils and high ground water table, whereas loss of nitrogen with interflow reaches its maximum on the agricultural areas on parabrown soils.

For defining the reference status of the river understood as uninfluenced by human nutrient input, a model experiment was done with no fertilization and no point sources. The resulting nitrogen load at the gauge Kietz for the period 2001 to 2005 was reduced by about 65%. Although this can be assumed as the potential natural conditions, it is not realistic to define this as a reference status in an inhabited area with agricultural use. However, the “good agricultural practices” aimed in reduction of nutrient losses from arable land should be applied in order to reduce diffuse pollution as much as possible. Further scenario simulations will be done in this respect.

To investigate the influence of a changing climate on water quality (nitrogen loads) of the Rhin river, two simple climate sensitivity analyses were performed. Precipitation was first reduced and increased by 10% with the same temperature, and then temperature was reduced and increased by 1.5°C with unchanged precipitation. The results are shown in Figure 23 b). A reduced precipitation leads to reduced nitrogen loads (less diffuse pollution as a result of less washing out) and an increase of precipitation behaves in the opposite way (higher nitrogen loads at the gauge Kietz). Lower temperatures cause higher discharge and more leaching (as a consequence of lower evapotranspiration) and result in higher nitrogen loads in the Rhin river. With higher temperatures a decrease in loads can be observed because of higher evapotranspiration and decreased leaching.

4. Conclusions

Simulation of the discharge behaviour and water quality with eco-hydrological catchment models is much more difficult for regulated lowland rivers than for rivers in mountainous areas due to the special characteristics of the former ones (e.g. high percentage of wetland areas, water management and melioration activities). This leads to problems in reproducing water amount and water quality by models. In our case study these problems were partly solved by introducing available information about water management. However it has to be admitted that further improvement of the model results can not be achieved without a better knowledge of human impacts and management activities.

Nevertheless such modelling experiments help to understand the river system behaviour better than before. Especially for identifying the fractions of point and diffuse sources at the outlet of the river system and the areas of highest diffuse pollution the model can be very useful. Knowing these sources and hotspot areas, it is easier to identify useful measures for reducing nutrient load in the river network and for achieving the “good ecological status” as required by the WFD.

As a further step it is planned to simulate some scenarios for the Rhin catchment. Possible scenarios are a land use change with no agriculture or agriculture with decreased fertilization on areas with a high rate of leaching nutrients to the ground water and surface water. Another possible scenario is the reduction of the input from point sources by improvement of the processes in water sewage plants or a combination of these two approaches. A very important scenario in view of future implementation of the WFD is a scenario taking into account possible climate changes in the Rhin region with a trend to warmer and dryer summers and more frequent extreme events.

This Rhin study was a pilot project with a close cooperation of researchers and representatives of the decision-making government to support implementation of WFD with research results. The LUA stakeholders can benefit from the model results as well as the researchers benefit from their special knowledge of the basin and the available data. It should be the aim of every research activity to provide results, which are requested and will be used by water managers and politicians to improve the adaptability of river systems to the future world.

Acknowledgements

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How Realistic is the Implementation of the European Water Framework Directive in River Basins Dominated by Agriculture? The Example of the Upper Ems River Basin (Germany)

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Abstract

The main objective of the European Water Framework Directive (WFD) is the achievement of a good ecological and chemical status of the water environment (water bodies). The objective of our study was to find a land use and land management scenario that would reduce the total nitrogen concentration in the rivers of the Upper Ems River Basin (North-Western Germany) to the WFD water quality class II (3 mg/l). The rivers in the agricultural used basin show total nitrogen concentrations of partly over 9 mg/l. We took policy instruments such as WFD, CAP and landscape development programs into account and developed consecutive land use scenarios. Results of the SWAT scenario calculations showed that the needed measures to achieve the water quality target in the basin would be unrealistic from a socio-economic point of view (reduction of arable land from 77% to 46% [13% organic farming], increase of pasture from 4% to 15%, afforestation from 10% to 21%, increase of protected wetlands from 0% to 9%, etc.). The example shows that the achievement of the environmental targets of the WFD is only possible with a consideration of regional distinctions. A general problem to be addressed is the lack of available water quality data. Strategies on water quality monitoring and water quality data availability must be improved and data access has to be facilitated. Under the current circumstances water quality modelling remains highly uncertain, since model results can not be validated sufficiently and the water quality models can not be proved adequately.

KEYWORDS: Water Framework Directive, land use scenarios, SWAT, water quality

Introduction

The application of large amounts of mineral and organic fertilizers in intensely agricultural used regions of Europe causes nutrient loads in soils, ground and surface water bodies. In contrast, other regions experienced in the last years a decreasing intensity of the agricultural use (EC, 1998, Zebisch, 2002). Such land use trends are influenced by policy instruments such as Common Agricultural Policy (CAP), Water Framework Directive (WFD) (EC, 2000) or national and regional land development plans. An example for an agricultural region with intensive use and high numbers of livestock is the Upper Ems River Basin in North-Western Germany. About 77% of the watershed is covered by agricultural land. Concentrations of partly over 10 mg/l of total

nitrogen substantially impair the water quality of the Ems River. The Working Group of the Federal States on Water Issues (LAWA) requires for instance 3 mg/l of total nitrogen as limit value for surface waters ("good ecological status", water quality class II). The LAWA water quality classification corresponds to the classification used by the WFD. Thus the current situation in the Ems River Basin is far from the postulated environmental targets of the WFD. In order to improve the water quality in the river network according to requirements of LAWA and WFD both the proportion of agricultural land and the amount of the applied fertilizers have to be reduced.
The main question to be addressed for us was thus which kind of land use and land management change scenarios in such an intensively used basin would result in the required reduction to 3 mg/l of total nitrogen in the rivers. Another question to be addressed was if the available water quality data would be sufficient to validate the simulation. We used the model SWAT2000 (Soil and Water Assessment Tool) (Arnold, et. al., 1998) to simulate and evaluate the impact of various land use and land management scenarios as potential measures. Taking the above mentioned policies into account, consecutive scenarios were developed. The studies presented in the paper were realized within the frame of the project FLUMAGIS where we developed a Spatial Decision Support System (SDSS) (Volk et al., 2007).

Methods

Study area

The study area is the Upper Ems River Basin in North-Western Germany (State of North Rhine Westphalia), which covers an area of 3,740 km². The entire basin has low relief terrain, and represents the macro-scale level in the project. The hydrological processes in the region are characterized by increasing precipitation amounts from the Northwest and Central basin (700 mm/year) to the Southeast (1,200 mm/year) but also by the widespread permeable sandy soils.
The river basin belongs to one of the most intensive used agricultural regions in Europe. Arable land covers approximately 77% of the area (average of Germany: 50%) and has lead to a dramatic loss of landscape diversity. The proportions of the other land use types are 9.9% for forest, 8.9% for urban areas, 3.9% for pasture and 0.2% for others. The cultivation of forage crops is used predominantly for the highly intensive livestock farming in the basin. The mentioned uses cause high nutrient inputs into the Ems river system. Total nitrogen concentrations of over 9 to 14mg/l (measured at gauge Rheine) substantially impair the water quality of the Ems River.

Land Use Scenarios

The high infiltration rates of the sandy soils favour the nitrogen leaching to the groundwater and the nutrient transport to the river. As a consequence, it has to be investigated in how far the change of land use and management practices could be a measure to improve the water quality and meet the standards of the WFD concerning the chemical water quality.

The objective was to develop a final land use scenario with the corresponding land management practices in order to achieve the limit value for total nitrogen concentration of the water quality class II. The limit values for selected nutrients are listed in Table 1.

Table 1. Limit values for nitrogen, water quality class II (LAWA, 1998)

	Total N	Nitrate-N	Nitrite-N	Ammonium-N
Maximum concentration [mg/l]	<= 3.0	<= 2.5	<= 0.1	<= 0.3

We included the main targets of relevant programmes such as Common Agricultural Policy (CAP), Water Framework Directive (WFD) or the landscape program of the State of North Rhine Westphalia (KULAP) to include realistic trends of land use changes and management practices (EC, 1998; MUNLV, 2001; Zebisch, 2002; BMVEL, 2004) in our scenarios. The measures and changes implemented in one scenario were consecutively integrated in the following scenario. Various options are possible for developing such measures. The objective of all measures is the reduction of nutrient inputs into the ecosystem and we made a distinction between following three types of measures:

Reduction of agricultural land

We assumed that a reduction of agricultural used land in the basin (currently 77% of the basin) could be considered as an important measure to reduce the nutrient inputs into the ecosystem. In the scenarios we decreased the proportion of agricultural land stepwise to the benefit of pasture and forest.

Extensification

This type of measure was focused on alternative land management practices. The land use configuration has not been changed in these scenarios. Extensification measures included a) reduction of livestock intensity on pasture land, b) reduction of the applied amounts of mineral and organic fertilizer applications, and c) application of conservational and eco-farming practices.

Renaturation measures

In the past several meanders have been artificially cut-off and the river has been heavily regulated for flood protection and to use the floodplains for agricultural production. Several weirs have been built to control the discharge. Currently, some river sections are under reconstruction. We tried to implement the following renaturation measures in the floodplains in the last scenarios:

- Extensification (reduced livestock on pasture) of or no land management in floodplains, respectively
- Implementation of filter strips / buffer zones
- Reconnection of oxbow lakes to the channel network

Model set up and calibration

SWAT2000 was used in this study to simulate the impact of land use and management scenarios on water quantity and quality in the Upper Ems River Basin. Before developing the scenarios and implementing them into the model, a comprehensive sensitivity analysis of selected model parameters and different management practices has been carried out using a virtual catchment. In addition, model modifications concerning the leaf area index and the canopy height of forests were carried out. The modifications are not topic of the paper, so please contact the authors for more information about it.

Implementation and preparation of the scenarios

This section gives a brief overview of the scenario implementation in the model SWAT2000. Figure 2 presents the details of the scenario settings.

In order to implement scenarios with reduced arable land, it was necessary to create new land use maps by using GIS operations. For scenario 1, we reduced agricultural land from 77% to 64% to the benefit of pasture (3.5% to 16.5%). Therefore, floodplains with typical alluvial soils currently used by agriculture were selected and the use was converted to pasture.

For scenario 2 an extensification measure of pasture was simulated: We reduced the livestock units from 2.6 to 1.4 as suggested by the landscape development program (MUNLV, 2001). This measure was simulated by using the amount of manure accordant to 1.4 livestock units. Moreover, and in contrast to the conventional pasture management, no additional mineral fertilizer was applied.

In scenario 3 agricultural land was converted into forest. Areas with potential low production capacity were identified by using the soil map. Where applicable, the use of these soils was converted from agriculture to forest evergreen.

For scenarios 4 and 5, the management practices were changed from conventional farming to eco-farming practices on selected agricultural land. The widespread distribution of poor sandy soils requires the application of huge amounts of fertilizers to achieve a reasonable agricultural production. Since this situation would not meet the requirements of organic farming, it was necessary to choose areas in the watershed with more fertile soils and comparatively low sand contents. In these scenarios we i) reduced the amount of applied mineral fertilizers, ii) implemented soil conservational tillage practices, and iii) changed the management with a focus on reduced time of bare soil (three-year crop rotation and intercropping).

In scenario 6, renaturation measures such as the reconnection of oxbow lakes were implemented, and filter strips or buffer zones were created. In order to simulate the reconnection of oxbow lakes to the channel network, the river length was increased about 10 kilometres. Due to the modified river morphology the roughness of the river bed has been increased by changing the parameter CH_N2 (Manning’s “n”) from 0.044 to 0.06. Buffer zones around the river were simulated by increasing the parameter FILTERW from 0 to 10 meters in the corresponding HRU files.

The floodplain areas in scenario 6 were still used by pasture. In scenario 7 these floodplain areas (over alluvial soil types) at the border of the river network were taken out of management. To simulate this, the land use type PAST (pasture) was converted to WETL (wetlands) with its corresponding default plant parameters.

The area of pasture land was crucially reduced in the former scenario. In the final scenario arable land was converted to pasture (sites have been chosen randomly this time).

Model Calibration

The first simulation (simulation 0) was a status quo scenario with the current land use situation which was also used to calibrate the model.
The simulated discharge was calibrated at six gauges using two objective functions, the Nash-Sutcliffe efficiency (Nash and Sutcliff, 1970), and the bias. In the calibration period an average efficiency of 0.75 and a bias of 2% were achieved. In the validation period the efficiency was 0.65 and the average bias 2.5%. Taking into account that the discharge regime of the Ems River is heavily modified, these results can be considered as satisfactory.
In contrast to the availability of long-term daily discharge time series in the study area, there is a lack of water quality data. Unfortunately, it was almost impossible to calibrate the model for water quality. We were forced to calibrate the model on the basis of five average annual nitrogen concentration values. Only for the year 2001 three data samples per month were available. Thus, it is dispensable to present the calibration results of an objective function in this case. Anyway, it was possible to fit the model to the average annual values adequately. Noticeable is that the simulated nitrogen concentrations have been slightly underestimated by the model.

Results and discussion

The simulation of the current conditions was based on the recent land use distribution (cp. Figure 1) and a land management with a crop rotation of fodder corn, barley and wheat which is applied on 90% of the arable land (LDS, 2001). The required limit value for total nitrogen (3 mg/l) is recently exceeded by 100% in an annual average.

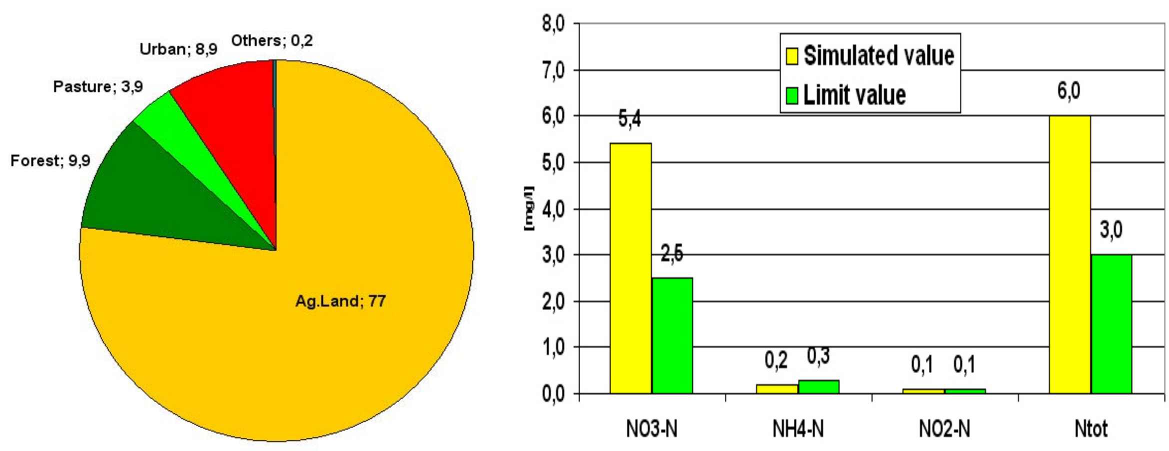


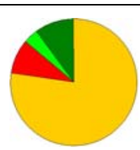
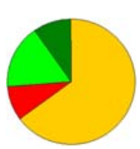
Figure 1. Current land use distribution (in percent) and results for the simulation of the nitrogen concentration in the Ems River at the outlet gauge. The darker columns show the target values of the WFD.

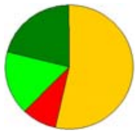
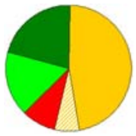
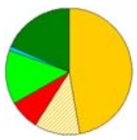
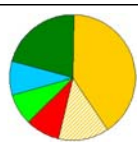
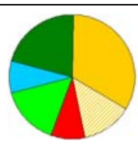
Starting from the simulation of the current conditions, eight further land use scenarios have been calculated. They were developed successively in direction of a target scenario that finally would come close to the water quality objective of the WFD. The scenarios are listed in detail in Figure 2. The simulation results have been used for cost assessment of selected management measures (Volk et al., 2007).

Figure 2 shows the simulation results of all scenarios for hydrological and nitrogen components (first row) as well as their change to the previous scenario in percent (italic numbers in the second row). At first sight, the most effective measures to reduce the total nitrogen concentrations are changes in management practices such as in scenario 4 and 5 (conventional farming to eco-farming practices). But also scenario 8, representing a land use change measure, has a large effect on nutrient reductions. Noticeable is that similar measures (arable land to pasture) represented by scenario 1 and 8 can result in very different degrees of effectiveness. Where in scenario 1 the total area of pasture was increased from 3.5% to 16.5% the pasture area in scenario 8 was increased from 8.1% to 15.2% only. But the reduction effect of scenario 8 is much higher than in scenario 1. An explanation for these discrepancies could be that in scenario 1 the nutrient inputs in the entire catchment are still very high and exceed critical thresholds. Under these circumstances the effect of reduced nutrient inputs remains rather small. Where in scenario 8 the nutrient inputs probably fall below a critical catchment threshold, and thus increase the effect of the measure. This would also explain the relative “un-effectiveness” of the measures implemented in scenarios 1 to 3.

The impact of renaturation measures on the simulated water quality, represented by scenario 6, can almost be neglected. In reality one would expect a larger influence of an implementation of riparian zones and river channel renaturation measures. Generally, complex processes of accumulation and decomposition of nutrients in riparian zones are influencing the nutrient outputs out of these buffer zones. The simulation of the impact of riparian zones on water quality in rivers is not yet solved adequately in SWAT2000 (realized by a simple function yet).

Another topic to be stressed here is the cause and effect delay between catchment response to implemented measures in the model and in reality. In the model the impact of measures takes effect immediately, because the scenarios starts always in the same year with same initial conditions. On the one hand this is necessary to compare the results of different scenarios.

Nr.		Scenario	Effect on					
			Runoff *tot*	**runoff *surf***	**Base flow**	**N*tot* [mg/l]**	**Nitrate**	**Ammonium**
0		**Current conditions** Arable 77% Forest 9.9% Urban 8.9% Others 0.2%	377.5	121.6	266.2	**5.98**	5.41	0.21
1		**Land use change I** Arable 77% →64.4% Pasture 3.5% → 16.5%	363.2 *-3.8*	104.8 *-13.8*	267.7 *+0.6*	**5.89** ***-1.5***	5.4 *-0.7*	0.19 *-10.5*
2		**Land management change I** Reduction of live stock units pasture 2.6 → 1.4	363.2 *0.0*	104.8 *0.0*	267.7 *0.0*	**5.72** ***-2.9***	5.2 *-3.1*	0.19 *0.0*

3		**Land use change II** Arable 64.4% →53.2% Forest 9.9% → 21.1%	355.8 *-2.1*	96.5 *-7.9*	265.7 *-0.8*	**5.56** ***-2.7***	5.06 *-2.7*	0.19 *0.0*
4		**Land management change II** Arable 53.2% →47.0% Arable (cons.) 0% →6.2%	357.4 *+0.5*	89.1 *-7.7*	276.0 *+3.8*	**5.11** ***-8.1***	4.63 *-8.7*	0.18 *-4.3*
5		**Land management change III** Arable 47.0% →40.2% Arable(cons.) 6.2% →13.0%	363.4 *+1.7*	84.7 *-4.9*	285.6 *+3.6*	**4.51** ***-11.8***	4.06 *-12.2*	0.16 *-7.9*
6		**River channel changes** Extension of river length 10 km Riparian buffers 10 m Pasture 16.5%→ 15.4% Floodplains (not used) 0% → 1.2%	363.7 *+0.1*	85.3 *+0.6*	285.4 *+0.1*	**4.48** ***-0.6***	4.04 *-0.5*	0.16 *0.0*
7		**Land management change IV** Pasture 15.4%→ 8.1% Floodplains (not used) 1.2% → 8.5%	364.3 *+0.2*	85.2 *-0.1*	286.1 *+0.2*	**4.33** ***-3.4***	3.91 *-3.3*	0.15 *-0.6*
8		**Land use change III** Arable 40.2% →33.2% Pasture 8.1% → 15.2%	364.5 *-0.8*	74.6 *-12.5*	292.4 *+0.9*	**3.83** ***-11.5***	3.42 *-12.5*	0.15 *0.0*

Arable (conventional) Arable (conservation) Urban / Traffic Pasture Floodplain, Wetlands Forest

Runoff*tot* = total runoff [mm/a]; runoff*surf* = surface runoff [mm/a]; N*tot* = Total nitrogen [mg/l]

Figure 2. Developed land use scenarios to finally receive the required water quality situation of the WFD. The effects of the scenarios on hydrological and nitrogen components (first row) are presented. Italic numbers in the second row indicate their change to the previous scenario in percent.

But on the other hand this procedure is far from reality. Depending on catchment characteristics, such as permeability of soils and initial nutrient loads, the impact of land use and management changes in the real world will usually take effect delayed. Maybe this delay is in the range of many years – which represents another problem for the implementation of the WFD.

In order to achieve the good ecological status for nitrogen at the Upper Ems River, the nitrogen concentration has to be reduced by 50% of the mean annual average. This would require aggravating changes of the agricultural land use and the management intensity. Figure 3 shows the target scenario (scenario 8) that comes close to the requirements of the water quality class II for nitrogen.

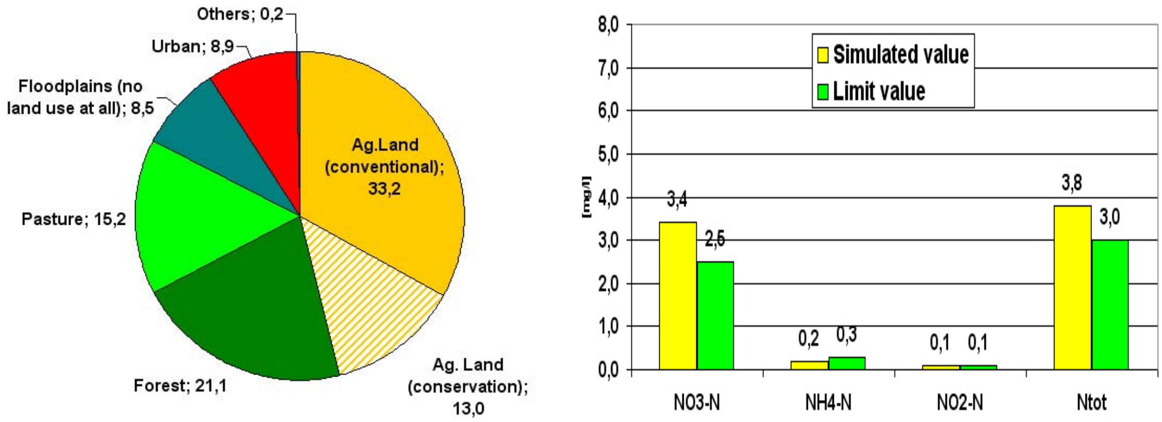


Figure 3. Land use distribution and simulated nitrogen concentrations to achieve the good ecological status for the Upper Ems River (scenario 8).

Beside changes of the river channels, this scenario includes a general reduction of the agricultural land and the fields with conventional management, an instruction of conservation management on 13% of the agricultural land, forestation, an increase of pasture, and the floodplains would need buffer zones. In addition, any other use has been taken out of the floodplains. However, this is not realistic from an economic point of view, since the incisions for the farmers would be so strong that most of them would have to give up their farms. Agro-economic calculations have shown, for instance, that only the mentioned changes in the floodplains would cost around 500 Euro per hectare and year what would amount to 29.6 Million Euro for the entire area.

Conclusions

The results have shown that SWAT is able to represent adequately general trends of water quality changes resulting from measures based on land use and management change. Especially area-related measures, such as changes in tillage practices, applications of fertilizers etc., can be described reasonably. But more sensitivity analysis is required to answer the question how detailed we have to parameterize management operation for large area applications of SWAT. In addition, measures based on linear structures (riparian zones) or spatially explicit measures are not represented satisfactorily and need to be improved.

In general the lack of long time series of water quality data with daily time step and higher spatial resolution has limited our capacity to evaluate the simulations – which represents a general problem and results in uncertainty. In addition, the existing monitoring programs for water quality in Europe are not suitable yet to deliver a sound database for the simulations (EEB and WWF, 2005). This is caused by a) the high costs of the needed procedures which results in sparse water sampling (every two to five weeks), and b) by sometimes insufficient cooperation between the relevant authorities, NGO’s and research institutes.

The results of our investigations show that there is an urgent need to reduce the nonpoint nutrient inputs from agriculture within the study areas. What we learned from the scenario simulations is that taking economical aspects into account, it will be almost impossible to achieve the environmental objectives of the WFD in our agricultural intensively used study area up to the year 2015. The results suggest that the achievement of the environmental targets of the WFD is only possible with a consideration of regional distinctions (different natural conditions, intensively used areas, areas with decreasing land use intensity, etc.), which would be more realistic. A “balanced” approach could be also taken into account where we could ask if it possible to balance areas of pollution with areas without any or only less pollution.

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Catchment Modelling using Internet based Global Data

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Abstract

The biggest problems in water resources occur often in areas with limited data collection (e.g. infrastructure problems) or with limited accessibility to data (e.g. some international basins). Thorough modelling studies to predict and evaluate water resources management scenarios, are hence not possible. Last decade however, there is an important growth of websites allowing for free of charge downloads of geophysical data such as GIS type data, weather data, etc., often referred to as global data. Such initiatives give new potentials for modelling data-sparce areas.

In this paper, global data is used for hydrological modelling using the Soil and Water Assessment Tool (SWAT), with applications to 3 very different catchments with large water resources problems: the River Kagera in Rwanda, Burundi, Uganda and Tanzania, the River Blue Nile in Ethiopia and the River Ganges in India, Bangladesh and Nepal. The model performance was very different for these catchments. The River Ganges, where the water resources are heavily influenced by human activities and which no data could be found, gave poor modelling results compared to observations, while in more natural areas, such as the Blue Nile, good monthly Nash-Sutcliffe efficiencies could be obtained. The current available global data is mainly originating of remote sensing derived data that provides support to the model inputs with regard to the geophysical characteristics. Rainfall data derived from satellite data could not replace the basic needs of rainfall ground data. For catchments with large human impacts, global data on water use (e.g. dams and reservoirs, irrigation schemes) may still be lacking.

KEYWORDS: SWAT, global data, Ganges, Kagera, Blue Nile

Introduction

A proper water management may safe lives by anticipating to flooding, droughts or reducing water born diseases. Modelling can be a useful tool for better understanding the water (quality) processes and for predicting the effects of management options (as decision support tool), or by controlling operations of hydraulic constructions such as reservoirs and dams. However, modelling needs, according to the problem and the modelling tool data on weather, pollutant sources, geomorphology, landuse, soil data, soil water data, crop yield data, evapotranspiration data and output data for calibration and validation (flow data/water quality data)... These data may not be locally available in areas with limited data collection (e.g. infrastructure problems) or with limited accessibility to data (e.g. some international basins).

The 3 catchments that are discussed in this paper are characterised by a lack of data as well as by major water resources problems: the Kagera catchement (sub-basin of Nile basin) is known for having water quality problems in a region of high population density, the Blue Nile catchment in Ethiopia has problems with severe droughts, as well as flooding, and Bangladesh is suffering of severe flooding by the Ganges river.

Models have been built for those cases using the Soil and Water Assessment Tool that aims at supporting water management by modelling the hydrological and water quality processes in relation to pressures by human activities or climate change. SWAT is embedded in an ArcView pre-processor that subdivides the catchement in sub-catchements using the Digital Elevation Model (DEM) and the Digitized Stream Network (DSN) and further into Hydrological Response Units (HRU's) using the Soil and Land Use map.

Global data

The models were built mainly by means of global and in most cases freely available information. The collection of the data was followed by an accurate compilation and analysis of the quality and integrity.

(i) Digital elevation model (DEM)

- **DEM-HYDRO1k**: the U.S. Geological Survey's (USGS) public domain geographic database HYDRO1k (http://edc.usgs.gov/products/elevation/gtopo30/hydro/index.html), which is derived from their 30 arc-second digital elevation model of the world GTOPO30. HYDRO1k has a consistent coverage of topography at a resolution of 1 kilometer.
- DEM-SRTM (Shuttle Radar Topography Mission) is a joint project between NASA and NGA (National Geospatial-Intelligence Agency) to map the world in three dimensions (http://www2.jpl.nasa.gov/srtm/index.html). The SRTM digital elevation data has a spatial resolution of 3 arc-second for Global coverage of latitude and longitude (approximately 90 meters). The SRTM "finished" data meet the absolute horizontal and vertical accuracies of 20 meters and 16 meters respectively.

(ii) Digital stream network (DSN):

- **DSN-HYDRO1k**: The USGS' HYDRO1k stream network database is derived from the flow accumulation layer for areas with an upstream drainage area greater than 1000 km^2.

(iii) Soil map:

-**SOIL-FAO**: Food and Agriculture Organization of the United Nations (FAO, 1995) provides almost 5000 soil types at a spatial resolution of 10 kilometres with soil properties for two layers (0-30 cm and 30-100 cm depth). Further soil properties (e.g. particle-size distribution, bulk density, organic carbon content, available water capacity, and saturated hydraulic conductivity) were obtained from Reynolds et al. (1999) or by using pedotransfer functions implemented in the model Rosetta (http://www.ars.usda.gov/Services/docs.htm?docid=8953).

- **SOIL-ISRIC**: Soil parameter data of International Soil Reference and Information Centre (**ISRIC**) can be downloaded from the website, 1 degree spatial resolution. (http://islscp2.sesda.com/ISLSCP2_1/html_pages/groups/hyd/islscp2_soils_1deg.html).

(iv) Landuse map
- **LANDUSE-GLCC**: the USGS Global Land Cover Characterization (GLCC) database (http://edcsns17.cr.usgs.gov/glcc/glcc.html) has a spatial resolution of 1 kilometre and 24 classes of landuse representation. The parameterization of the landuse classes (e.g. leaf area index, maximum stomatal conductance, maximum root depth, optimal and minimum temperature for plant growth) is based on the available SWAT landuse classes and literature research.

(v) Weather data:
- **Weather-NCDC**: the National Climatic Data Centre provides daily maximum and minimum temperatures as well as precipitation data at about 40000 stations worldwide (NCDC, 1994, 2002). However, the global distribution of the climate stations is quite uneven and both quantity and quality of the data vary noticeably.
-**WEATHER-CRU/dGen**: The Climatic Research Unit (CRU, http://www.cru.uea.ac.uk/cru/data/hrg.htm) data-sets TS 1.0 (wet days per month; New et al., 2000) and TS 2.0 provides precipitation and average minimum and maximum temperature per month (Mitchell et al., 2004). These monthly climate grids have a resolution of 0.5°. Daily weather values for each sub-basin were generated based on CRU data using the semi-automated daily weather generator algorithm dGen (Schuol and Abbaspour, 2007). To make use of the climate grid in SWAT, dGen changed the grid data into synthetic station data by overlaying the climate grids with SWAT subbasins shape-file and then aggregating into point data. This program creates Thiessen polygons around each value point representing the center of the grid cell, overlays and intersects the subbasin layer with the climate grids and finally computes the area-weighted average which is then assigned to each subbasin centroid.

Table 17. Data sources for the three catchments.

	Ganges	Kagera	Blue Nile
DEM	DEM-HYDRO1k	DEM-SRTM	DEM-HYDRO1k
DSM	DSN-HYDRO1k	DSN-HYDRO1k	DSN-HYDRO1k
SOIL map	SOIL-ISRIC	SOIL-FAO	SOIL-FAO
LANDUSE map	LANDUSE-GLCC	LANDUSE-GLCC	LANDUSE-GLCC
WEATHER data	WEATHER-TRMM	WEATHER-CRU/dGen WEATHER-TRMM **WEATHER-NCDC**	WEATHER-CRU/dGen
DISCHARGE data	Local data of the Institute of Water Modelling (IWM) and Bangladesh Water Development Board	Average monthly discharges from The Global River Discharge Database Daily	

	(BWDB)	discharges from East Africa hydrologic database	
WATERBODIES	No data available	WETLANDS-FAO	*Local data*

- WEATHER- TRMM (Tropical Rainfall Measuring Mission). TRMM is joint project of NASA and National Space Development Agency of Japan, which objectives are observing and understand the tropical rainfall and how this rainfall affects the global climate. TRRM is interested in tropical rainfall because most of the rainfall is found in tropical zones and have a significant influence on the global climate change. Daily accumulated rainfall or rain rate data are available for the period from 1998 to present, with a spatial resolution varying from 0.25 degree to 5.0 degree resolution. (http://disc2.nascom.nasa.gov/Giovanni/tovas/TRMM_V6.3B42_daily.shtml)
- WEATHER - NCEP/NCAR: The NCEP/NCAR Reanalysis Project is a joint project between the National Center for Environmental Prediction (NCEP) and the National Center for Atmospheric Research (NCAR). The NCEP/NCAR Reanalysis data are divided into 6 different sections: pressure level, surface, surface fluxes, tropause sections, derived data and spectral coefficients. Precipitation data are in surface fluxes section. This section comprises 4-times daily, daily and monthly time series from January 1st, 1948 to present. The data sets spatial coverage is T62 Gaussian grid with 192x94 points. Other climate variables available through this dataset are air temperature, surface lifted index, pressure, relative humidity, and wind velocity. The levels of measurement are surface or near the surface, or entire atmosphere. (*http://www.cdc.noaa.gov)*
(vi) River discharge data:
- DISCHARGE-GRDC: The Global Runoff Data Centre (GRDC, http://grdc.bafg.de) provides flow data at montly/daily time basis.
(vii) Wetlands and reservoir data:
- **WETLANDS-FAO**: information on wetlands is gathered at the site http://www.fao.org/DOCREP/003/X6611E/x6611e00.htm#TopOfPage

The Ganges catchment

Description

Bangladesh is a low-lying country in the Meghna delta located at the confluence of three major rivers; the Ganges, the Brahmaputra and the Meghna. Flood occurs almost every year in Bangladesh with varying intensity and magnitude. In a normal year, 20% of the country is inundated by river spills and drainage congestions. Approximately 37% of the country is inundated by floods of 10 years return period. Devastating floods of 1988 and 1998 inundated more than 60% of the country. In a deltaic region like Bangladesh, flood forecasting is very complicated. Moreover considerable computer support and sophisticated models are required for hydrological modelling.
The river Ganges is a major river (The total length is about 2,507km.), which originates as the Bhagirathi from the Gangotri Glacier in the Uttaranchal Himalaya and joins the Alaknanda near Deoprayang to form the Ganga. Further downstream, the Ganges flows across the large plains of north India and empties in to the bay of Bengal after dividing up

into many distributaries. The main barrier to establish a hydrological model for Ganges basin is the absence or unavailability of hydrological information of the cross-boundary Ganges-Brahmaputra basins which together are around 10 times larger than the Bangladesh portion.

Model set-up

The Ganges basin was subdivided into 11 sub-catchments based on the DEM processing and 11 HRU's representing the dominant land use and soil classes within each sub-catchment. The models ability to reproduce the derived hydrological response is determined from the calibration. As the main objective of this work is flood forecasting, so in this study a good agreement of peak flow is considered. Both manual and auto-calibrations were performed.

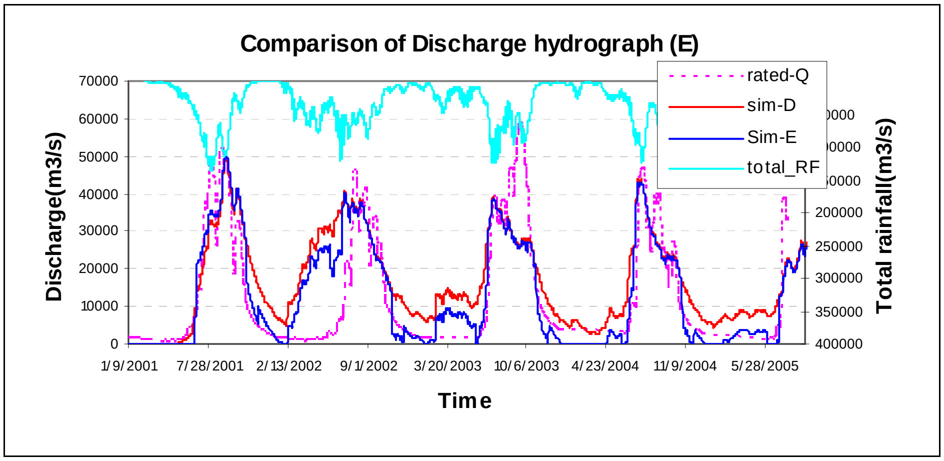

Figure 1. Last modelling (sim D and E) results compared to observations (rated-Q).

Results

While not being very successful, model improvements were sought by including hydraulic conductivity and orographic effect by using different elevation bands for each sub-catchment (sim D). At last the water distribution agreement between India and Bangladesh is also put into account for the flow distribution (version sim E of figure 1). However, all the options couldn't provide satisfactory result for flood forecasting purpose.

It concludes that no calibration can substitute the missing processes of a catchment (e.g. human impacts) to develop a physical based hydrologic model like SWAT, when hydrometerological information is too scarce.

The Kagera catchment

Description

The Kagera River basin is distributed in four countries: Burundi, Rwanda, Tanzania and Uganda The study area covers an area of approximately 58000 km^2. The Kagera River is the largest of the 23 rivers that drain into Lake Victoria and it is the most remote head stream of Nile River. It is formed at the confluence of two rivers: Nyabarongo (Rwanda) and Ruvubu (Burundi). The elevation in the basin varies from around 1100 m above mean sea level in east to 4500 m in the Northern-west. Rainfall varies from less than 1000mm over the eastern part of the basin up to 1800 mm and above in the west, where most of the runoff is generated.

There are two rainfall seasons, with the longer south-easterly monsoon bringing rain between about February and May, and the shorter north-easterly monsoon from about September to November. The runoff responds to the rainfall with a higher peak in May and a smaller peak in November. However, the river flows are attenuated by a number of lakes, and wetlands and associated lakes (Holmberg et al., 2003). Heavy rainfall during rainfall seasons and steep slopes cause erosion hence degradation of ecosystem. On the other hand, there are also droughts during prolonged dry seasons. Moreover, Kagera Basin faces problems related to water resources management of trans-boundary watersheds.

Model set-up

The watershed delineation of Kagera River basin was done using a DEM of spatial resolution of 92mx92m from SRTM, and the basin was subdivided into 12 sub-catchments, which surface area ranges from 11900 to 16745 km^2. The other data used are **SOIL-FAO and LANDUSE-GLCC**. Monthly discharge from The Global River Discharge Database for 5 gauging stations was used for model evaluation. Several sources of weather data were compared.

The model calibration period runs from 1974 to 1979. The following period up to 1985 was used to validate the calibrated model. The calibration process was done in three phases: resizing of subbasins channels, specification of wetlands, modification of ground water flow parameters and then auto-calibration with sensitive parameters. Wetlands have a significant influence on hydrologic processes in Kagera River basin and they occupy in average 6% of Burundi and Rwanda surface area (FAO-SAFR, 1998). An automated base flow separation filter (Arnold, 1995) was used to determine the recession constant and ground water delay for the subbasins from stream flow data.

A model of the Nyabarongo sub-watershed was build consisting of 6 sub-basins and 36 HRUs. Only ground stations precipitation data were used. The model was calibrated and validated at Kigali gauging station. Monthly water flow was calibrated for a period of 6 year from 1974 to 1979, and validated from 1980 to 1984.

Results

The evaluation is based on the coefficient of determination (R^2), Nash-Sutcliffe efficiency coefficient (Ns) and Deviation of stream flow volume (Dv) to measure the model performance.

Table 2. Performance of the Kagera model for different sources of rainfall data (Ns=Nash-Sutcliffe efficiency, R2= coefficient of determination, RMSE=Root mean squared error)

	CRU/dGen		NCEP/NCAR		Ground data	
	Calibration (1974-1979)	Validation (1980-1985)	Calibration (1974-1979)	Validation (1980-1984)	Calibration (1974-1979)	Validation (1980-1985)
Ns	0.41	0.01	0.43	-21.03	0.63	-1.36

R2	0.48	0.26	0.45	0.03	0.65	0.08
Dv	-7.67%	-10.83%	-6.96%	39.49%	-7.31%	11.24%

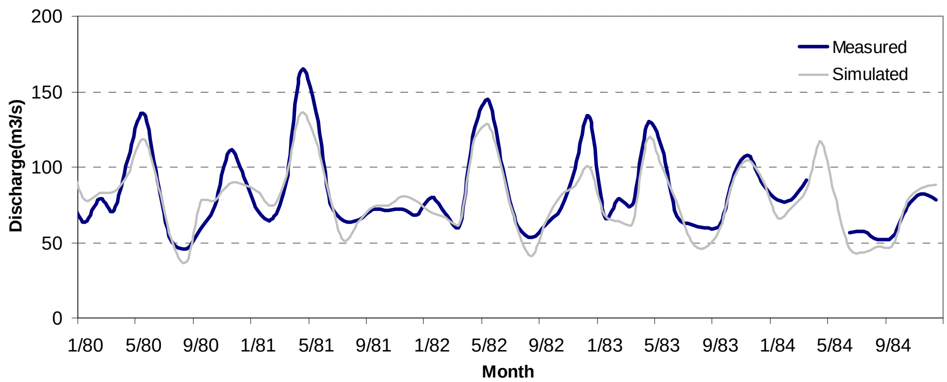

Figure 2 Simulations for the Nyarabongo watershed, validation period using (Kigali station)

The simulated stream flow using TRMM gave an underestimation, where the best simulation gave only 18% of the total volume was simulated. Due to underestimation of precipitation, the simulated stream flow was very low so that no calibration was done, except manual adjustment of ground water parameters. A calibration and validation was performed for the CRU/dGen and the NCEP/NCAR giving reasonable results, but poor results for predictions of the validation period (table 2).

Promising results were obtained for the Nyarabongo watershed at Kigali station using ground data, but only after including the wetlands in the model (figure 2). The Nash-Sutcliffe efficiencies of the calibration and validation were 0.65 and 0.78 respectively.

The Blue Nile

Description

The study area, Upper Blue Nile river basin, is known as Abbay in Ethiopia. It is the largest in terms of volume of discharge, second largest in terms of area in the country. The river contributes over 50 % of the long-term river flow of the Main Nile (Conway, 2000). The basin covers an area of 175,000 km^2 having 14 sub basins. High population growth and environmental degradation, limited resources, and inadequate land use and water policies have increased Ethiopia's vulnerability to drought disaster.

However, only a few studies of this portion of the watershed have been conducted and the information available on the hydraulic and hydrologic characteristics of the river and the basin is incomplete. Ethiopian agriculture and Nile River flows are heavily dependent upon precipitation in the upper Blue Nile basin as a means of irrigation and stream flow contribution, respectively. Such a state of high dependency on rainfall for food production has made Ethiopia very vulnerable to drought.

The annual rainfall over the basin decreases from the south - west (>2000 mm) to the north - east (around 1000 mm), with about 70 per cent occurring between June and September (Conway, 2000). Conway (1997) explained that the Blue Nile is characterized by severe seasonality with average annual flow of about 50 km^3 measured at the basin outlet at El Diem station near the Sudan-Ethiopia border. More than 80% of this flow occurs during the flood season between July and September, while only 4% of that flow occurs during the driest period January-April.

Model set-up

SWAT model requires at least three elements of climate data for simulation. These are monthly average precipitation, monthly maximum temperature and monthly minimum temperature. The global data obtained from Climate Research Unit (CRU) from 1971 – 1994. In this study stream flow data from Kessi station and at Sudan border station will be used in the calibration and validation process later.

The whole catchment is sub divided in to 22 sub-basins and reservoirs have not been included due to data limitation however two lakes were included Lake Tana in sub-basin 21 having 3156km^2 and Lake Fincha that is an artificial lake at Fincha hydroelectric project site in sub-basin 8. 295 HRUs were created for the whole basin based on the 2% threshold areas for land use and soil.

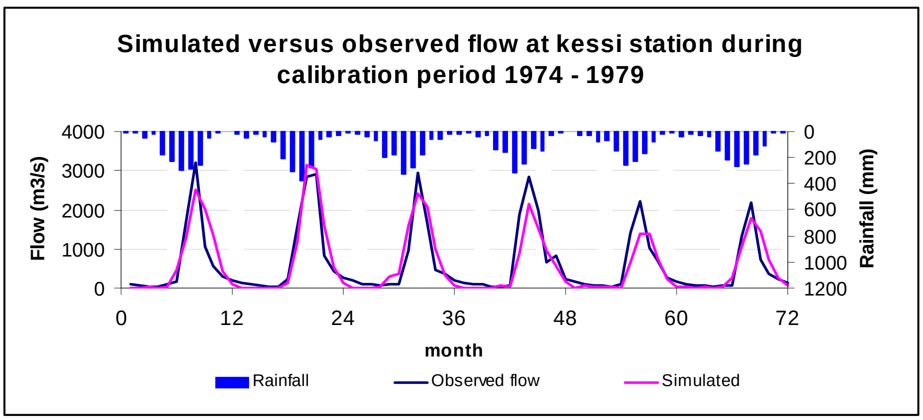

Figure 3: Simulated versus observed flow at Sudan border and Kessi station during calibration period

Results

During calibration period, both stations show good performance while the Sudan border station performs less during validation period. The reason could be the extension made over the global data as of 1995 with observed climate data from nearby meteorological stations. This leads to lesser annual rainfall in the catchment. Consequently it was difficult to obtain good validation result with the model calibrated relatively with higher rainfall. Such result was expected at Sudan border.

	Kessi		Sudan border	
	Calibration (1974-1979)	Validation (1980-1985)	Calibration (1974-1979)	Validation (1980-1984)
Ns	0.83	0.80	0.77	0.74
R2	0.83	0.81	0.79	0.76

Dv	4.36%	14.45%	-6.46%	31.1%

Table 18: Deviation of runoff volume for calibration and validation period

The result showed that the model better simulate runoff volume at Kessi station than Sudan border station. The drainage area of smaller catchment with Kessi outlet is one-third of the entire drainage area of Upper Blue Nile basin. Most of the area upstream of Kessi outlet is at higher elevation than downstream, which indicates that floodwater quickly collects in the drainage channels. The reason for underestimation of runoff volume at Kessi station could be due to the fact that sources like springs are not included in the model.

Conclusions

Global data are an important source for setting-up a model by providing GIS data (DEM, land use maps, soil maps and databases), weather data and information on water bodies (e.g. wetlands or reservoirs). In addition, flow data to calibrate or verify a model can be found in global data-bases. In general, the GIS data are found to be very useful, for building hydrological models, for instance by using SWAT. The weather data seem to be the limiting factor in getting accurate model results. Our studies could not prove that satellite born data for estimating rainfall can replace groundwater data. Also, catchments with large human activities that disturb the natural flow of the river can not be modelled properly without information about these activities.

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Taking the Step from a Large-Scale Hydrological Model (West-Africa) to a Continental Model (Africa)

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Abstract

The information on the amount of country-based freshwater availability is of critical importance for national strategic water planning and management, particularly concerning water and food security. This information is much more valuable if it accounts for spatial and temporal variations in a country and incorporates the input and modeling uncertainties. Based on globally available data, we implemented and calibrated a SWAT model for water quantity investigations in a 4-million-km^2 area in West-Africa. The model setup was performed using the ArcView interface and the “Sequential Uncertainty Fitting Algorithm” (SUFI-2) program, which was used for calibration and uncertainty prediction concerning input, output and model parameters. The West-African study showed that SWAT and the selected calibration procedure were applicable to very large areas. Considering the scarcity of data in the region, the results were very satisfying and provided notable insight into the freshwater availability and the associated uncertainties in this vulnerable region. All components of the freshwater availability (blue water and green water) were quantified explicitly on a sub-country level and with a monthly interval.

This study is part of a larger project to assess the global freshwater availability. Up to now, the limiting factor for the setup of a continental model was the inability of the ArcView-interface to calculate the geomorphic subbasin-parameters for such large areas with about 1500 subbasins. Using the new ArcSWAT interface this problem was overcome and we succeeded to setup and run a model for the whole continent of Africa. The large variation, e.g. in climate and landuse, major local interferences caused by, amongst others, reservoirs and wetlands, and last but not least the long computation time make the calibration procedure especially challenging. Different ways to handle these continental scale hydrological modelling problems will be shown.

Keywords: hydrological modelling, freshwater availability, blue water, green water, prediction uncertainty, (West) Africa

Introduction

Estimates of the temporally and spatially unevenly distributed water resources are essential for many global studies, ranging from food security in the face of climate change to conflicts related to regional freshwater availability. Providing a global coverage, there only exist estimates for the long term annual averages of the country-based internal renewable water resources (IRWR, e.g. Shiklomanov, 2000; Alcamo et al., 2003). These IRWR equal the internal blue water availability of different countries but disregard the green water, which is the main source for sustaining rainfed agriculture and ecosystems (Falkenmark, 1995). A further major drawback of the existing freshwater availability estimates is their lack of quantification of uncertainties associated with the estimates, which is generally quite large in distributed models and global assessments.

The presented work is part of a modelling project with the goal of assessing the monthly sub-country-based global freshwater availability for both, the blue and the green water. The uncertainties associated with the model results will be clearly shown and made available to advanced studies and decision making.

To simulate hydrology, we selected the distributed watershed model “Soil and Water Assessment Tool” (SWAT, Arnold et al., 1998). With this model it is possible to quantify the different components of the water cycle (evapotranspiration, river discharge, soil moisture, shallow and deep aquifer recharge) even for very large watersheds with an adequate spatial resolution (subbasins) and a monthly temporal resolution. This capability is necessary in order to estimate next to blue water (sum of river discharge and deep aquifer recharge), also the by Falkenmark and Rockstrom (2006) defined “green water resource” (or, as we prefer to call it “green water storage”), which corresponds to the soil water, and the “green water flow”, which corresponds the actual evapotranspiration (AET; Fig. 1).

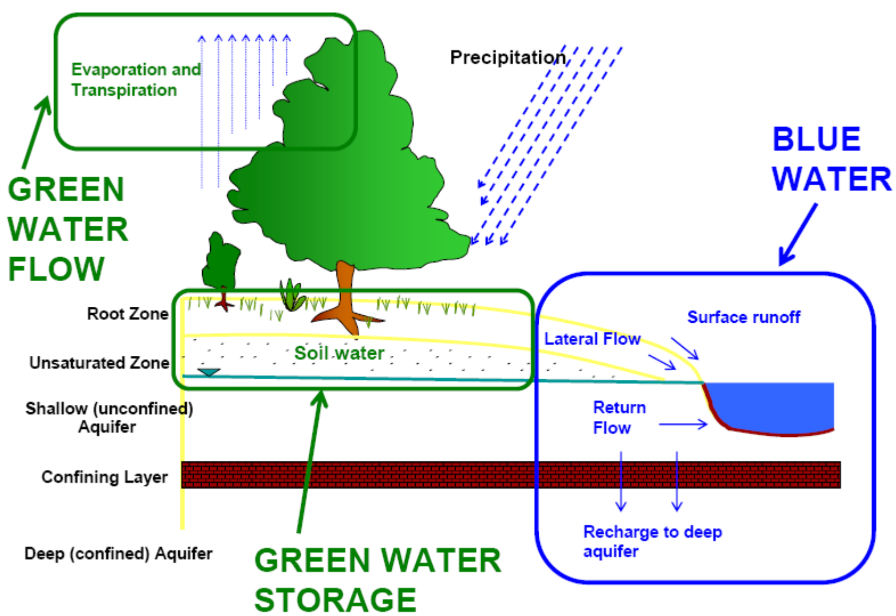


Figure 1: Blue and green water differentiation using SWAT (modified from Neitsch et al., 2002)

In order to facilitate the pre- and post-processing of the Swat’s in- and output (e.g. watershed delineation, manipulation of the spatial and tabular data, visualization), the model is linked to GIS. Up to the beginning of 2007 there existed only a link to ArcView GIS (Di Luzio et al., 2001). With the help of this interface we setup a model for West-

Africa which was also successfully calibrated and validated (Schuol et al., 2008) and provided much information on the freshwater availability in this vulnerable region. With this interface it was not possible to handle the whole continents at once – most probably due to some internal ESRI ArcView memory problem. Since the beginning of 2007 an ArcGIS interface has become available for SWAT (ArcSWAT, Olivera et al., 2006, Winchell et al., 2007, http://www.brc.tamus.edu/swat/arcswat.html). With this tool, it was now possible to overcome the previous limitation of the maximum model-area and to setup a model for the whole of Africa. The ArcSWAT core provides the same model setup possibilities as the ArcView GIS interface but it adds some new options. Amongst others these are the use of predefined watersheds instead of a minimum sub-basin-threshold based delineation, the inclusion of the slope characteristics in the HRU determination and the additional choice of selecting the dominant HRU instead of the dominant landuse, soil and slope independently.

The output of any (hydrological) model, especially at this large scale, can't be used directly in further studies or for decision making. It is essential to calibrate and validate the model and also to quantify the prediction uncertainty. For this purpose we selected the program "Sequential Uncertainty Fitting Algorithm" (SUFI-2, Abbaspour et al., 2004, 2007). Yang et al. (2007) compared different uncertainty analysis techniques in connection to SWAT and found that SUFI-2 needed the smallest number of model runs to achieve a similarly good solution and prediction uncertainty. As our continental model is computationally quite intensive, this efficiency is of great importance. In addition SUFI-2 can easily be linked to SWAT and it can handle a large number of parameters and many calibration stations at the same time.

Lessons learned from the model setup and the calibration procedure of the West-Africa model as well as from the Africa continental model will be shown and discussed. Furthermore we highlight some results from the West-Africa model to point out what could be achieved using a SWAT model for the whole continent of Africa and in the near future for the whole world.

The ArcSWAT Continental Africa Model Setup

The West Africa model, as well as the continental Africa model, was constructed using globally and in most cases freely available information. The collection of the data was followed by an accurate compilation and analysis of the quality and integrity. A detailed description of the database can be found in Schuol et al. (2008).

The ArcView interface of SWAT has limitations concerning the size of the delineated area. This restriction is relaxed in the new ArcSWAT interface and therefore, in contrast to the West-Africa model, the continental Africa model uses ArcGIS interface (Winchell et al., 2007). Selecting a minimum drainage area of 10000 km^2 and making some additional manual modifications (e.g. deleting many outlets in the Sahara desert, adding outlets at gauging stations with observed runoff) resulted in a total of 1496 sub-basins for the whole continent. The geomorphology and stream reaches parameterization was done automatically by the interface. In addition to dominant landuse and soil maps, we made use of the new option to differentiate different slope classes (in this case three) to create HRUs. The choice of dominant soil and landuse was due to the very large scale of the model area and the consequently long computational time. For the simulation time we selected the period from 1968 to 1995 and provided for these years daily generated precipitation as well as minimum and maximum temperature for each sub-basin. The superiority of this generated climate input compared to the available unevenly distributed and often poor quality measured data input was already shown in a previous study (Schuol and Abbaspour, 2007).

Lakes, wetlands and (large) reservoirs affect the river discharge to a great extent. For this reason we included lakes and reservoirs with a storage volume greater than 1 km^3 in the model and parameterized them based on the existing information and assumptions, where ever necessary (Schuol et al., 2008). Major wetlands on the main channel network were also handled as reservoirs and parameterized in such a way that the simulated outflow pattern resembles the observed one. With this step, the basic model setup within the ArcSWAT interface was finished and simulations could be carried out.

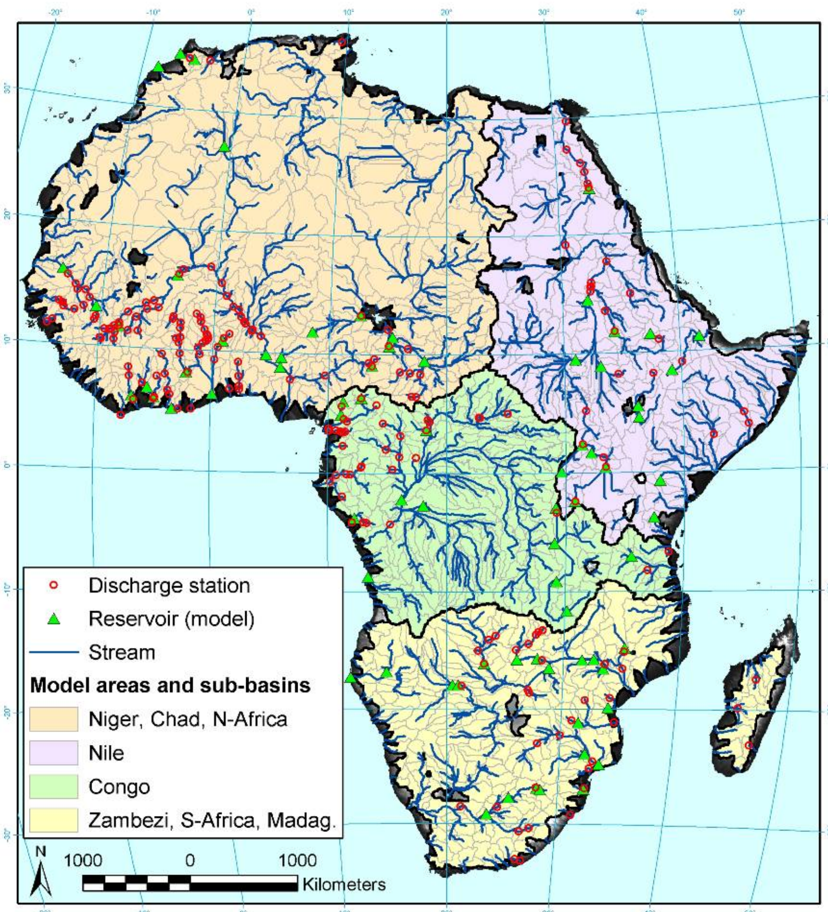


Figure 2: Model areas, sub-basins, reservoirs and discharge stations used for calibration in Africa

Model Calibration

For the calibration of the continental Africa model, which was performed on a monthly time step, we used the available observed river discharge data at more than 200 stations. Figure 2 shows the location of these stations and also the very uneven distribution throughout the continent. Most stations don't have data for the whole analysis period from 1971 to 1995 and therefore it was inevitable to include different time periods and time lengths for calibration at different stations. The whole continent shows a great variation in climate zones as well as dominant landuse and soil types. For this reason we believe that it might not be reasonable to have only one parameter set for the whole continent and consequently divided the continent into four areas: (1) Niger, Chad and North Africa, (2) Nile, (3) Congo and (4) Zambezi, South Africa and Madagascar (Fig. 2). Theoretically it would be possible to divide into even more areas but this is limited on the one hand by the large basins, like the Nile or the Congo, which we would like to treat as a whole unit in the same way and on the other hand by the model approach with a simultaneous model run and calibration for all regions. For each of these areas we optimize a slightly different parameter set, depending on the predominant landuse and soil types.

In contrast to the West-Africa study, where we selected the Nash-Sutcliff (NS) coefficient between the monthly measured and the monthly simulated discharges as efficiency criteria (objective function), we selected this time a weighted version of the coefficient of determination: *Φ* (Eq. 1, Krause et al., 2005; slightly modified). The weighting is performed based on the gradient *b* of the regression between the monthly observed and simulated runoff and guarantees that under- or over-predictions are reflected together with the R^2, which represents the discharge dynamics.

$$\Phi = \begin{cases} |b| \cdot R^2 & \text{for } |\mathrm{b}| \le 1 \\ |b|^{-1} \cdot R^2 & \text{for } |\mathrm{b}| > 1 \end{cases} \tag{1}$$

The major advantage of this efficiency criterion is its range (0 to 1), which isn't as sensitive to a few bad simulations as the NS, which ranges from minus infinity and 1.

The objective function *g* (Eq. 2) is formulated in each model area independently as the *n*-station-sum of *Φ* where each station is weighted with *w* (Eq. 3) depending on the contributing area (*A [km²]*) and the number of monthly observations used for calibration (*s*) at the certain station *i* and at the upstream stations *j*.

$$g = \sum_{i=1}^{n} \left(w_i \cdot \Phi_i \right) \tag{2}$$

$$w_i = \sqrt{\frac{\left(A_i - \sum_{j=1}^{n} A_{up,j} \right) \cdot s_i}{s_i + \sum_{j=1}^{n} s_{up,j}}} \tag{3}$$

The reason for this weighting is based on the uneven distribution of the stations (Fig. 2) and the different long time periods with observed data that were used for calibration. A runoff station with a long data series and a large basin without further stations upstream provides more new information for calibration and should obtain a larger weight than a station in a densely gauged area or a station with a short time series.

While the lowest station weight w_i was set to 1, the highest station weight had a value of 61 and was determined for the furthest downstream station on the river Congo at Kinshasa.

Starting with a physically meaningful intervals for the parameters included in calibration and using two contradictory criteria, the iterative Latin hypercube sampling based calibration with SUFI-2 aims to maximize the objective function *g* and to minimize the parameter ranges while bracketing a reasonable percentage of the measured data in the 95% prediction uncertainty (95PPU) band. As all sources of uncertainties (input, parameter, conceptual) are reflected in the measurements (in this case the discharge), the parameter ranges (uncertainties) are directly linked to the model uncertainty, represented by the 95PPU. For a model of this size calibration procedure is quite computationally demanding but we found SUFI-2 to be quite efficient compared to other methods. In each iteration step we performed “only” about 500 model runs and the new, narrower parameter ranges were obtained based on the one percent of the simulations with the highest objective function values instead of the “best parameters” as originally performed in SUFI-2 (Abbaspour et al., 2004).

Results and Discussion

As calibration of the continental Africa model is still ongoing, in the following we will show and discuss some results of the West-Africa model. In the near future similar results and information are expected to be provided for all African countries.

Due to the inevitable uncertainties in all hydrological models and particularly also in large scale applications as in our case, it is not appropriate to show solely the results of the “best” simulation but rather the 95PPU intervals of the last iteration, representing the calibrated model. The different SWAT outputs for each sub-basin and reach were summed up for different areas (countries) and time steps.

In Fig. 3 we present for selected countries in West Africa the 95PPU intervals of the computed annual average (1971-1995) of the IRWR (blue water) and the AET (green water flow). For the sake of comparison and model verification we also show the IRWR estimates of two other global assessments: (a) WaterGAP 2.1e (1961-1995 average; Esty et al., 2005) and (b) FAO (2003), where long-term averages are reported from multiple sources and years. The SWAT 95PPU intervals for the IRWR are large but they cover in most cases the FAO and WaterGAP estimates. These single-value estimates are in many countries quite different, while the 95PPU ranges seem to be a realistic estimation of the existing uncertainties. The uncertainty ranges of the AET are clearly smaller as only one out of 16 calibrated parameters (ESCO) affects its value.

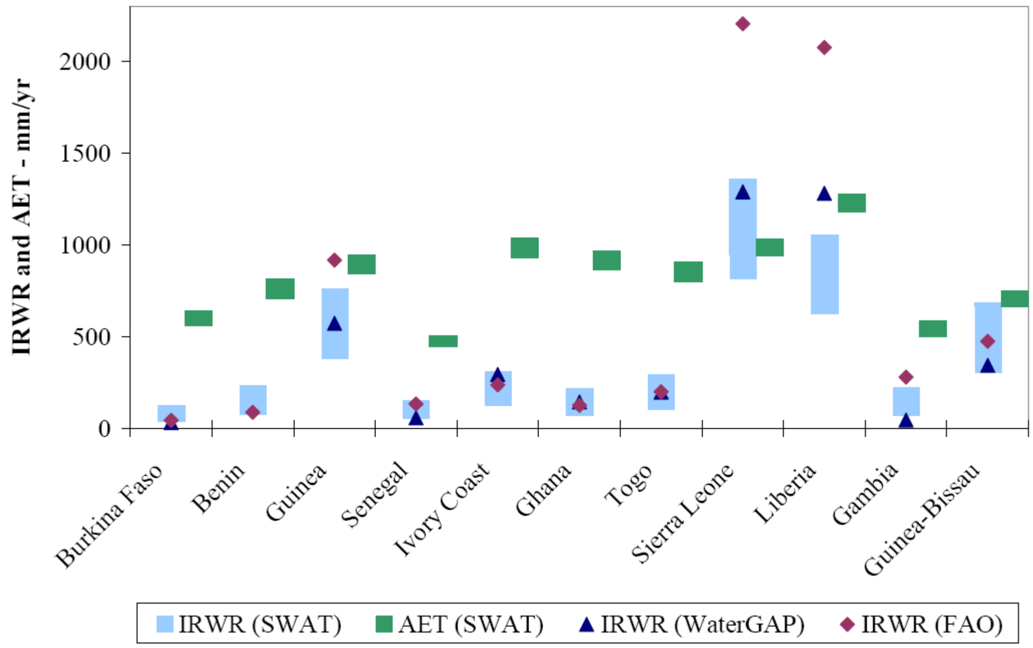
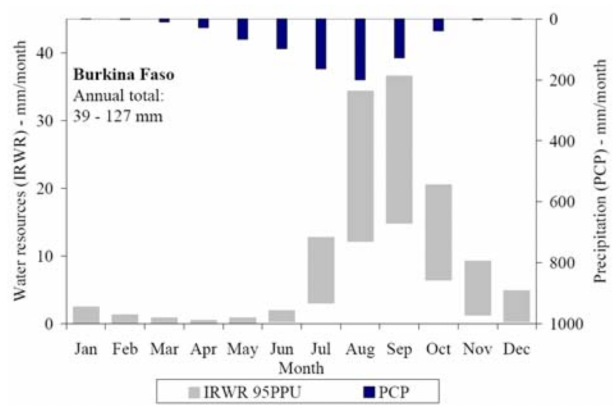


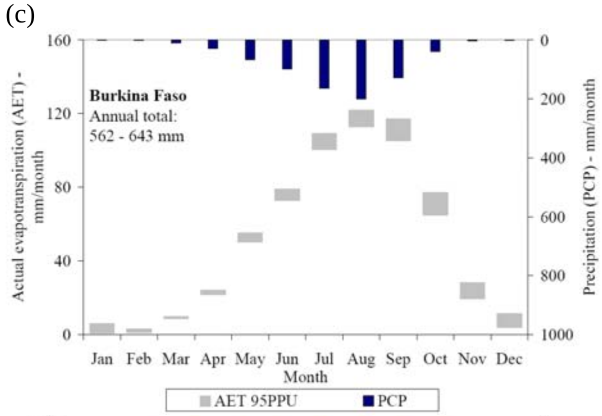
Figure 3: The SWAT computed 95PPU intervals for the internal renewable water resources (IRWR, blue water) and the actual evapotranspiration (AET, green water flow) for selected West African countries compared with the IRWR estimates from the FAO assessment and the WaterGAP model

In addition to the annual averages, we also assessed the monthly country-based 95PPU averages (1971 to 1995) of the IRWR, the AET and the plant available soil water (SW, green water storage) and present it in Fig. 4 together with the monthly precipitation for Burkina Faso. The uncertainty ranges of the IRWR are, compared to the AET, quite large especially for the wet months but the annual pattern is nicely represented. Furthermore it can be seen that in the months after the rainy period there is still a remarkable contribution to the annual IRWR resulting from base-flow and deep groundwater recharge components. The monthly variation of the 95PPU interval of the SW is narrower than that of the IRWR and the annual pattern is similar to the precipitation pattern but with a somewhat delayed depletion after the rainy period. It should be mentioned that a direct validation of the modelled soil water was not possible as suitable soil water observations for the region of interest were not available. While we solely present country based results, our calculations are based on subbasins and it would be also possible to show intra-country differences which are, especially in larger countries, quite significant.

(a)



(b)
(c)



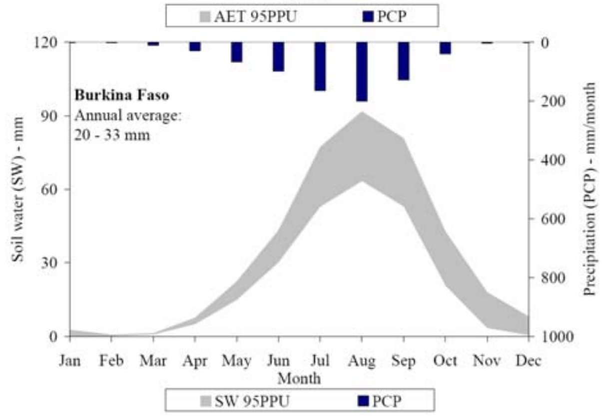


Figure 4: Average (1971-1995) monthly 95PPU bands of the (a) internal renewable water resources (IRWR, blue water), (b) actual evapotranspiration (AET, green water flow), and (c) soil water (SW, green water storage) in Burkina Faso

Conclusion and Outlook

While SWAT models are in most cases used for small scale investigations, we showed that SWAT can also be used for very large scale water quantity assessments. With the new ArcSWAT interface previous area limitations are overcome and we successfully setup and run a model for the whole African continent. We do not show the results of the continental model here as it is still in progress, but we showed its use and expected outcomes based on the results of the calibrated and validated West-African model. We provide at (sub-) country-level annual and monthly information not only on the blue water but also on the mostly neglected green water flow and green water storage. These freshwater availability components are always shown as prediction uncertainty ranges, as one number could be quite misleading due to the uncertainties in the available data (model input), the parameters and in the conceptual model.

After the calibration and evaluation of the African model is finished we will continue with other continents in order to finally obtain a global picture. The results of the spatial and temporal variations of the freshwater availability could and will be used in global and national strategic water planning and management, e.g. in advanced studies concerning the water and food security, virtual water flow, and effects of climate change.

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Evaluation of the SWAT Model Setup Process Through A Case Study in Roxo Catchment, Portugal

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Abstract

Acquisition of input data at a very detailed temporal and spatial scale represents one of the main limiting factors for the successful application of the Soil and Water Assessment Tool (SWAT) model, which is highly physically based and process oriented. Furthermore, model input preparation and setup usually require substantially more time compared to the actual run-time of the model. In this respect, the main focus of this study was the initial stages of model setup, including data gathering, processing, formatting and preparation, catchment discretisation, model parameterisation, etc. The applicability of the GIS coupled SWAT model as an integrated catchment management tool was evaluated through a case study outside the US (Roxo Catchment, Portugal), specifically in terms of data requirements and model input preparation. To perform this task, available spatial and non-spatial data were processed and formatted, a field campaign undertaken, laboratory analyses performed and post-processing of the data carried out. Focusing on “how to get data” rather than “how to model using default values”, it was found that when planned and implemented properly, fieldwork and consequent laboratory analyses combined with other tools, such as mobile global positioning systems (GPS) for fieldwork, freeware, spreadsheets and geographic information systems (GIS), were very effective and rendered it feasible to establish a SWAT input dataset within a reasonable time and budget.

KEYWORDS: SWAT, data acquisitions, model setup, fieldwork, GIS, Roxo Catchment

Introduction

Effective decision-making processes leading to the choice of the best management options, such as the implementation of the EU water framework directive, require relatively reliable, inexpensive and timely information. Conventional monitoring methods, like field stream water quality data collection and analysis as a decision-making tool, are expensive and time consuming (Santhi et al., 2005). In this regard it is generally considered that a watershed-based modeling approach, preferably integrating hydrological, erosion and water quality processes and with spatial or geographic information system capability, allows for the consideration of such attribute variations, and quantification of the impacts at different spatial and temporal scales. The application of watershed models as management and decision support tools have been discussed in many research studies (Arheimer et al., 2004; Arnold et al., 1998; Bhuyan et al., 2003;

Borah and Bera, 2003; Grizzetti et al., 2003; Jayakrishnan et al., 2005; Jha et al., 2004; Jonsson et al., 2002; Santhi et al., 2005; Tripathi et al., 2003; White et al., 1992).

The Soil and Water Assessment Tool (SWAT) is a comprehensive model which is extensively used worldwide. SWAT serves as a powerful tool to simulate long term impacts of land use practices in a watershed varying in soil types, land uses and climatic conditions. According to a classification of the models based on the spatial scale and physical detail by Droogers, 2001, SWAT is considered a basin scale and highly physical model (Figure 1).

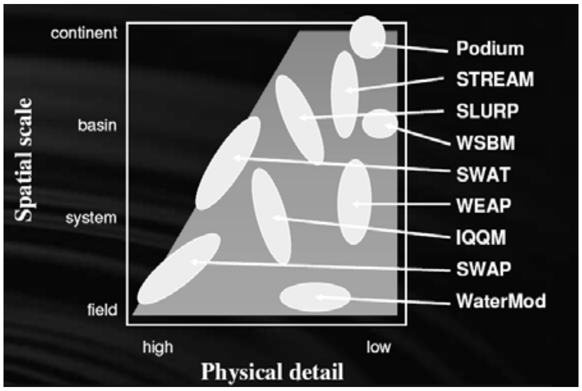

Figure 1: Classification of models based on spatial scale and physical detail (Droogers, 2001).

SWAT has the capability of analyzing large watersheds and river basins by subdividing the area into homogeneous sub-basins or sub-watersheds (Santhi et al., 2005). Each sub-basin is further discretized into several hydrologic response units (HRUs) that have distinctive land use and soil combinations. The semi-distributed nature of SWAT (spatial heterogeneity on subcatchments scale but not on Hydrologic Response Units) allows for both the simulation of spatial variability of the catchment characteristics and computational efficiency at the same time.

However, all the capabilities of the SWAT model are limited by availability of a sufficient amount of appropriate input data. SWAT requires a diversity of information in order to run. While some of the data are required input, the model can also utilize optional input data. Basically, SWAT requires specific information about the topography (for catchment delineation), land use, soil characteristics (physical and chemical) and climatic conditions of a watershed. According to the needs of the study, the SWAT model allows to utilize information on plant growth, pesticide, fertilizer, and management practices (timing and amounts of fertilizer, pesticide, irrigation, crop rotations, planting and harvest dates). Also, streamflow and water quality data are needed for calibration and validation purposes.

Di Luzio et al. (2002) give the sequence of the key procedures for a SWAT model application as follows:

1) Delineate the watershed and define the HRUs
2) (Optional) Edit weather and soil databases (which is in fact necessary for the SWAT applications outside the U.S.)
3) Define the weather data
4) Apply the default input files writer ????

5) (Optional) Edit the default input files
6) Set up (requires specification of simulation period, PET calculation method, etc.) and run SWAT
7) (Optional) Apply a calibration tool
   (Optional) Analyze, plot and graph SWAT output

Although the choice of the most suitable model for a particular application depends on the purpose of the study (e.g. identifying risk areas, quantification of nutrient of losses from land, scenario analysis etc.) and on the quality (accuracy and precision) needed from model results, a review by Schoumans and Silgram, (eds.) (2003) shows that the possibilities for scenario analyses tend to increase as the complexity of the model increases, but so does the relatively high costs and efforts associated with setting up these more complex models, too. In many cases, the detailed data required by such models may not be available, necessitating some assumptions or default values to be made, or transfer functions to be developed.

As a good example to those more complex data-hungry models (see Figure 2.), SWAT was evaluated in this study based on a case study (Roxo catchment, Portugal), specifically in terms of its data requirements and model input preparation. While analyzing the complete cycle of “how to get data” and the setup processes, also the sensitivity of the model to some parameters was checked in terms of data quality needs.

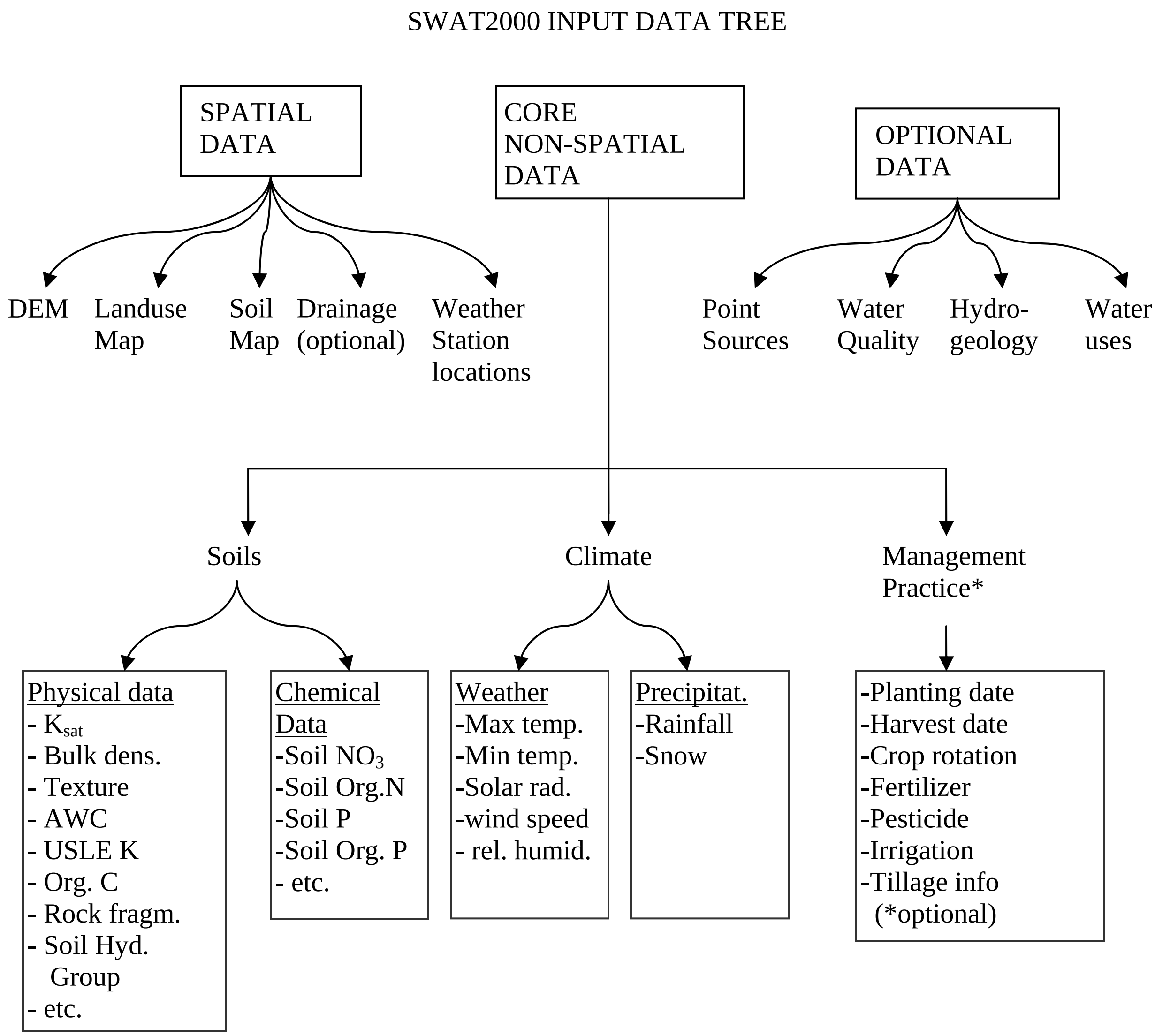

Figure 2. A schematic overview of the SWAT input requirements

Methodology

Study Area

The study area, the Roxo reservoir catchment, is located in the Beja district of Alentejo Province in southern Portugal (Figure 3). The Roxo catchment covers a total area of 353.18 km^2 being mainly drained by the Ribere do Roxo river (http://www.inag.pt). The Roxo reservoir forms the outlet of the catchment with an average area of 13.8 km^2, varying season to season. The reservoir is of high economic importance to the area as it provides water for irrigation and domestic use to the towns of Beja and Aljustrel.

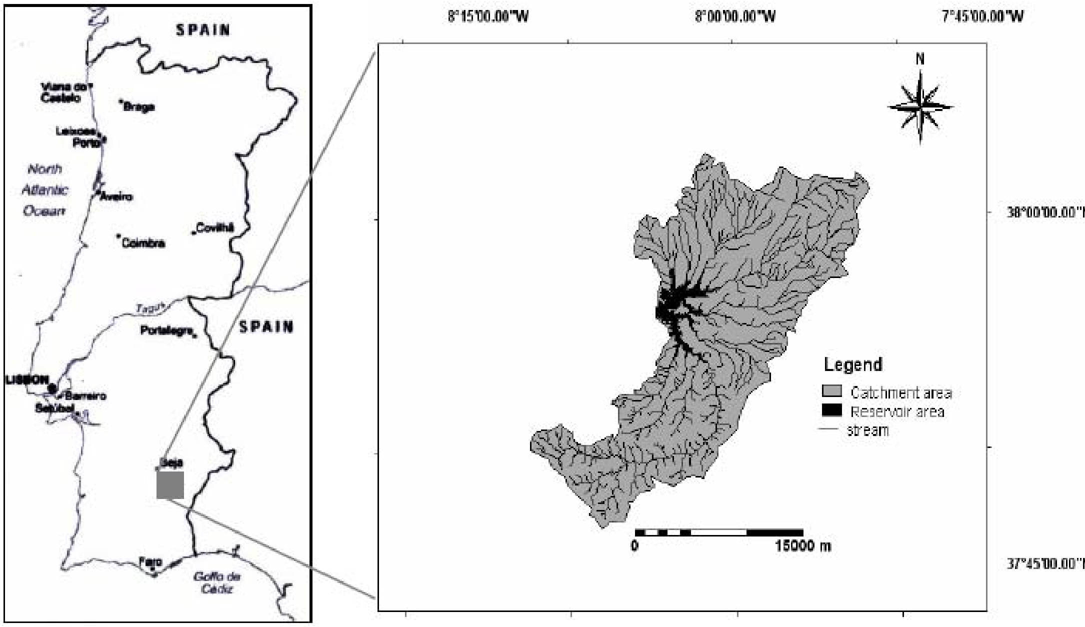

Figure 3. Location of the Roxo Reservoir Catchment.

The Roxo catchment has a topography ranging from nearly flat to gently sloped with an average elevation of 192.6m above sea level. Lying in the southern part of Portugal, the climate of the study area is influenced by Mediterranean with relatively warmer winter and drier summer. Maximum temperature during the summer goes up to 40^{o}C and minimum temperature in the winter drops down to 5 ^{o}C. The average annual rainfall in the study area is estimated to be around 550 mm (Sen, 2004).

The predominant landuse in the catchment is comprised of agricultural land, both rain-fed and irrigated. The forest area is limited to some patches of lands and along the stream course, the large part of which is occupied by eucalyptus trees. The grass land also constitutes some part of the area.

Model Preparation and Setup

In this study, the model input preparation for the Roxo catchment has followed the steps below:

- A screening of the available spatial and non-spatial data for the model was performed;
- The available data were processed and formatted according to the AVSWAT requirements;
- A fieldwork campaign was planned and undertaken to obtain the soil data and update climate data;
- Laboratory analyses were carried out to obtain physical and chemical soil parameters;
- Finally, the main required data-set was compiled through post-processing and making also use of freeware, spreadsheets and geographic information systems (GIS);

- An initial setup and run of SWAT was achieved for Roxo Catchment.

The main data components, processes as well as a flowchart for SWAT Modeling are summarized in Figure 3.

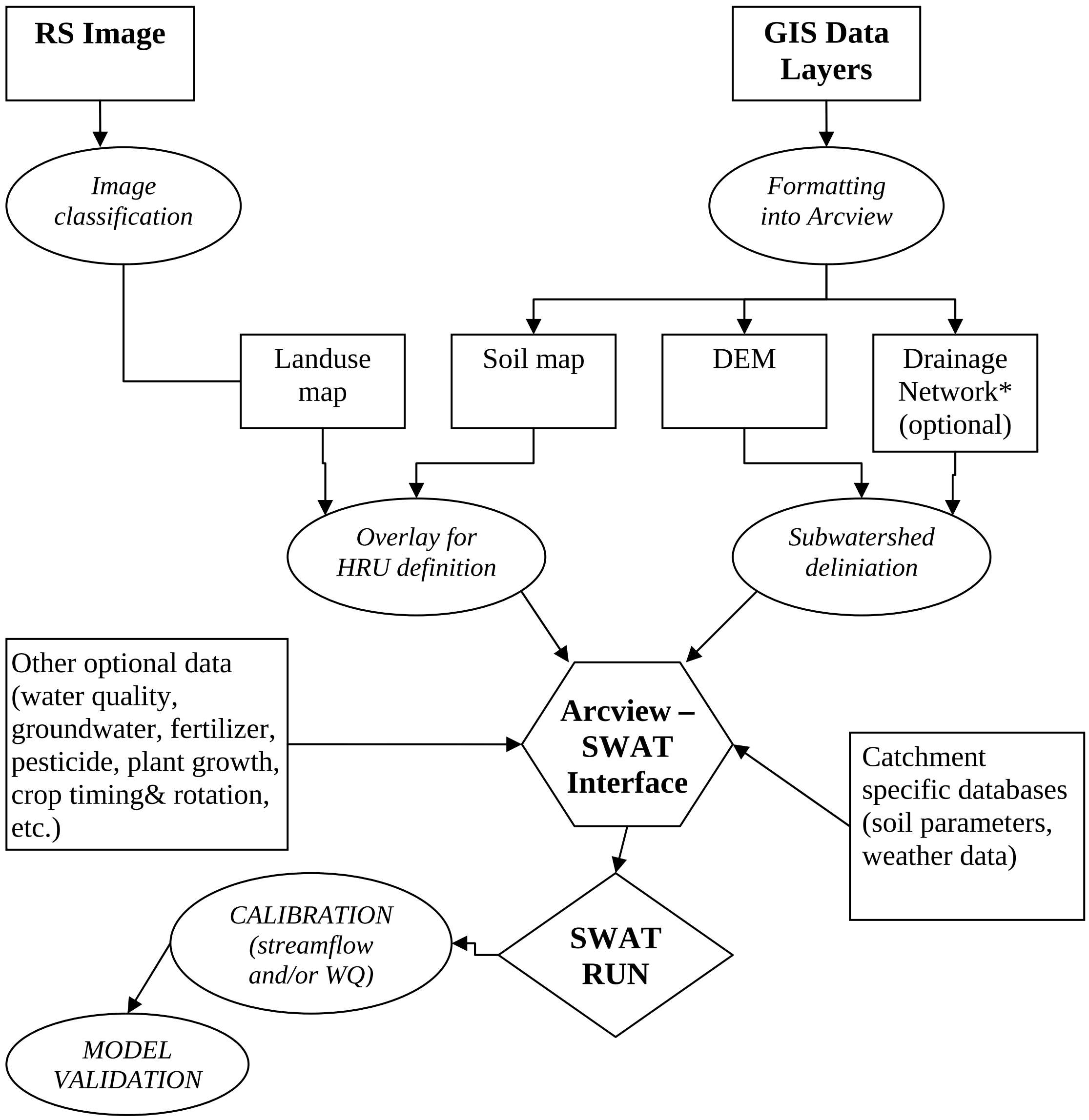

Figure 3. A summary flow chart for SWAT model input and processes.

Results

Data Sources

Though SWAT has the capability of assuming default values for many of the input parameters and simulating time series of climate data based on statistics, such abilities would not help much in real case applications as a management tool, it being a heavily physical model. Therefore, "how to get data" forms one of the main limitations for SWAT modeling. Figure 4 and 5 present a preliminary inventory of possible data sources to serve as a guide for the main SWAT input parameters.

SPATIAL DATA SOURCES:

Digital Elevation Model (DEM)

1) Contour map of the area

2) Toposheets

3) Global SRTM data (90m resolution)
http://srtm.usgs.gov/index.htm

Landuse Map

1) RS image (image classification)
2) Internet resources
- European Corine Landcover database (EEA), 1:100,000 or 1:500,000 scale
http://dataservice.eea.europa.eu/dataservice/metadetails.asp?id=667
- Global landcover (1992-1996), 1km resolution,
http://edcdaac.usgs.gov/1km/comp10d.asp

Soil Map

1) Soil maps (digital preferably) from national, regional authorities

2) Internet resources
- European digital archive on soil maps of the world
http://eusoils.jrc.it/esbd_archive/EUDASM/indexes/access.htm
- World Soil Information (global, regional, national maps)
www.isric.org/UK/About+Soils/Soil

River network

1) Digitizing from toposheets

2) Drainage extraction from available DEM

3) An online global hydrograph, derived from 1km resolution
http://edcdaac.usgs.gov/gtopo30/hydro/index.asp
4) JRC River and Catchment Database for Europe
http://agrienv.jrc.it/activities/catchments/

Figure 4: A list of possible resources for SWAT spatial data input

With the advances in internet, more and more online data warehouses are becoming available with different scales and quality. However, special care needs to be paid especially in the case of spatial data when internet data resources are to be utilized. Finding up to date and suitable scale (digital) maps and data in appropriate accuracy are still a big problem at the catchment scale for many countries. Therefore, it can be suggested that a complementary fieldwork would always be useful and necessary when online spatial data is utilized, especially in the case of landcover maps which are more prone to changes comparing to other spatial data. However, internet still serves (with improving quality) as an important resource for data scarce catchments. An example would be the globally covered 90m-resolution DEM by the SRTM mission, which always serves as a good alternative if a better option is not available.

With regard to main non-spatial data requirements like climate and soil, climate data can be obtained relatively easily by including the large databases available on the internet. As it was also the case in this study, soil physico-chemical data constitute one of the main bottlenecks in the preparation of SWAT input files and usually require utilization of different sources and methods.

SOURCES FOR CORE NON-SPATIAL DATA:

Climate

1) Meteorology data providers (national, regional)

2) Internet resources
- www.worldclim.org (global climate grids with 1km2 resolution)

- World's largest archive of climate data www.ncdc.noaa.gov/oa/ncdc.html

Soil

1) Fieldwork & laboratory analyses

2) Literature data (especially on texture) in combination with Soil Water Characteristics program.

3) Internet resources
- World Soil Information www.isric.org/UK/About+Soils/Soil+data/

Figure 5. A list of possible data sources for main non-spatial data

Data Accuracy

While preparing SWAT soil input data for the Roxo catchment, the Saturated Hydraulic Conductivity (K_{sat}), one of the main physical soil parameters, was measured or estimated by different methods for comparison and accuracy purposes. The soil samples collected during the fieldwork were first used to measure the particle size distribution (texture) of Roxo soils for different classes. Then, the particle size distribution data were utilized to estimate soil water parameters including K_{sat} with the application of Soil Water Characteristics freeware (http://hydrolab.arsusda.gov/soilwater/Index.htm). On the other hand, K_{sat} was also directly measured by the Falling Head Test, using the same soil samples. In addition to these, literature particle size distribution data obtained from the

Soils of Portugal Report (Cardoso, 1965) was used to estimate Ksat values. The results of the three methods are presented in Figure 6.

As seen from Figure 6, the Falling Head Test and K_{sat} estimation from laboratory texture data gave consistent results. While the K_{sat} estimation from the literature texture data (based on Cardoso, 1965) gave significantly higher results than the other two methods, the distribution of K_{sat} values among the different soil types was still similar.

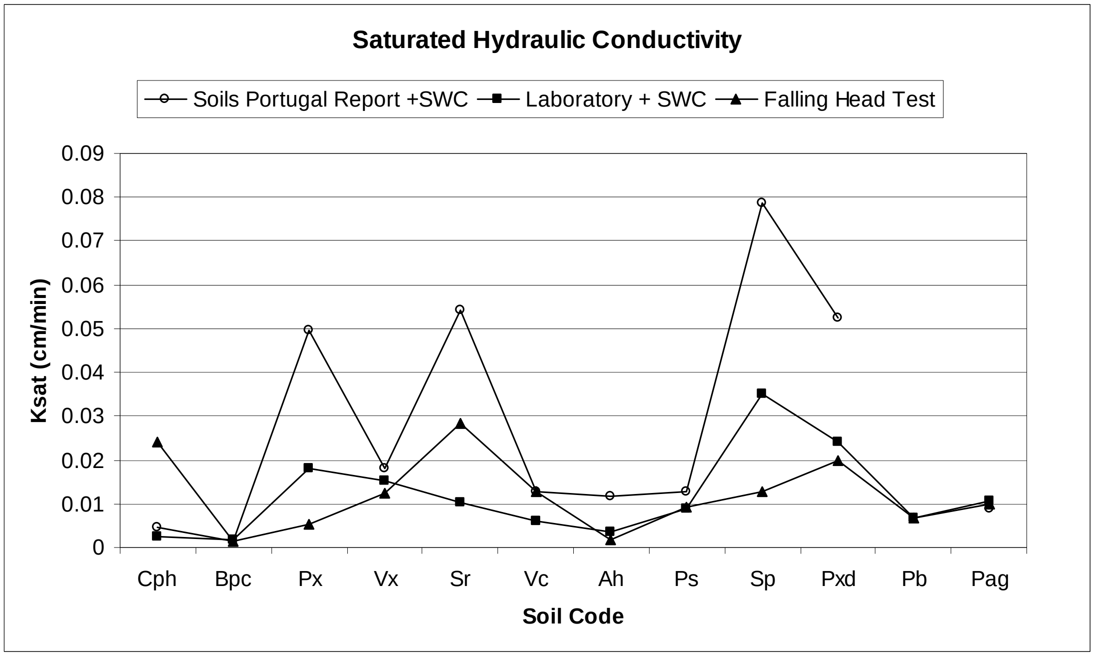

Figure 6: A graphical comparison of the methods used to estimate K_{sat} values

Data sensitivity

The quality (accuracy and precision) expected from a model result is usually determined by the purpose of the study. As the complexity of the model increases, not only the extent of the data demand but also the need for higher quality data increases due to higher complexity of model calibration and validation. This necessitates to concentrate the efforts for collecting better quality data on the relatively more sensitive model parameters. In this respect, a good strategy for data gathering would be to carry out a sensitivity analysis first, and to pay special attention to those parameters in data collection.

The calibration tool available under the simulation menu of AVSWAT2000 interface allows to analyze the sensitivity of certain SWAT parameters. The only available soil parameter, Available Water Capacity (AWC), was analyzed for its sensitivity through the "Edit Parameter Variation" tool. Figure 7 shows the change in the model results with ± 0.5 change (which is the maximum range allowed by the calibration tool) for the value of AWC.

Table 1. The Sensitivity of the SWAT Model to the Soil AWC Parameter

Qout (m^3/s)				Qout (m^3/s)			
Sub basin	AWCini	AWCini +0.05	% change	Sub basin	AWCini	AWCini -0.05	% change

1	2.397	2.185	-0.09	1	2.397	2.704	0.14
2	0.099	0.092	-0.08	2	0.099	0.110	0.13
3	0.091	0.084	-0.08	3	0.091	0.102	0.13
4	0.090	0.084	-0.08	4	0.090	0.100	0.12
5	0.606	0.560	-0.08	5	0.606	0.674	0.12
6	0.102	0.091	-0.12	6	0.102	0.118	0.16
7	0.137	0.123	-0.11	7	0.137	0.156	0.15
8	0.297	0.269	-0.10	8	0.297	0.338	0.15
9	0.496	0.449	-0.10	9	0.496	0.564	0.14
10	0.227	0.205	-0.11	10	0.227	0.258	0.14
11	0.198	0.179	-0.10	11	0.198	0.224	0.14
		Avg=	**-0.10**			**Avg=**	**0.14**

A data set of 5 years duration (2001-2005) was used for setting up an initial run of the SWAT Model for the Roxo catchment. In the sensitivity analysis, the average model outputs for the last 4 years were taken, excluding the first year 2001 used for priming the model. Although the sensitivity of the AWC is not very much highlighted in the results in Table 1, its effect can be still significant in the model outputs considering its role in the water balance. The AWC parameter determines the water holding capacity of the soil layer, affecting the infiltration rate and runoff amount, and finally the water balance, which is the driving force of most of the model processes.

Conclusion

An FP5 Research Project called EUROHARP makes an assessment of 9 contemporary modeling tools in terms of their applicability under different climatic, agricultural, geophysical and hydrological conditions in Europe and classifies SWAT as “suitable” or “very suitable” in almost all the conditions, except for North (N) & Northeast (NE) climates (relatively colder) and mountainous & delta landscapes, for which SWAT is classified as “uncertain”. However, acquisition of input data at a very detailed temporal and spatial scale represents one of the main limiting factors for the successful application of the Soil and Water Assessment Tool (SWAT).

The earlier stages of model setup, including data processing, formatting and preparation, catchment discretisation, and model parameterisation etc, constituted the main components of this study, which usually takes much more time compared to the actual run-time of a model. Although the availability of free of charge online data warehouses is increasing (Figures 4 and 5), the need for up-to-date and high quality data is still not mostly covered by those sources, especially for the main components and sensitive parameters of SWAT. Therefore, a successful application of the SWAT Model for scenario analysis and similar complicated purposes necessitates implementation of an effective data collection strategy: determination of sensitive parameters, utilization of appropriate data warehouses and GIS, carrying out the necessary complementary fieldwork and laboratory analyses.

Focusing on “how to get data” rather than “how to model using default values”, it was found that when planned and implemented properly, fieldwork and consequent laboratory analyses combined with other tools, such as mobile global positioning systems (GPS) for fieldwork, freeware, spreadsheets and geographic information systems (GIS), were very effective and rendered it feasible to establish a SWAT input dataset within a reasonable time and budget.

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Modelling Streamflow under Different Land Use Conditions with SWAT: Preliminary Results from a Chilean Case Study

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Abstract

Water resources from the Biobío Basin are of high strategic importance for Chilean economic development. In this context, and under the current scenario of increasing pressures, advances in the general understanding and capacity to describe and predict, in a spatially explicit way, the impact of climate and anthropogenic forcing on the hydrology of the Biobío River Basin are urgently needed. The Soil and Water Assessment Tool (SWAT) was chosen to model the hydrology of the Vergara Basin, a sub-basin of the Biobío. The model was calibrated (2000 -2002) using the PARASOL automated calibration procedure implemented in SWAT2005. Validation took place using monthly output data for the time periods contained between 1994 –1999 and 1977 -1982, which represent current and historic land use conditions, respectively. Validation for the historic time period was done in order to evaluate the capability of the model to accurately describe the basin hydrology under considerably different conditions of land use (the presence of forestry plantations in the basin increased from 0% to 39% between 1979 and 1996).

The results show that model performance can be considered as satisfactory for most part of the basin during calibration and both validation periods. In order to further evaluate the sensitivity of the SWAT model application for Vergara, fictitious “extreme” land use scenarios were generated (e.g. 100% forestry plantations) and modelled. Results were analyzed at the yearly and monthly output level, for both wet and dry season river flows. Results from these analyses are discussed.

KEYWORDS: SWAT, Chile, Vergara Basin, Streamflow, Land use.

Introduction

The adequate allocation of water resources between different water uses under changing conditions of land use and climate is a challenge which many societies already face, or will need to face during the next decades (Simonovic 2002). In this context, the use of spatially distributed rainfall-runoff models which allow to describe the temporal variability and spatial distribution of river hydrology and water availability becomes increasingly relevant, as these tools allow to describe the impact of not only past or present situations but also future scenarios (Klöcking and Haberlandt 2002, Haverkamp, et al. 2005).

Over the past decades, considerable changes in land use have occurred in the Vergara Basin (e.g. 39% increase of exotic forestry plantations). To a certain extend - although probably at a slower pace- additional changes may also be expected in the upcoming decades. However, evaluations of the impact of these changes on basin hydrology are currently not available. In what follows, a quantitative evaluation was made of the potential influence on the local water balance of past, current and hypothetic future land use conditions, by means of the application of a calibrated and validated version of the SWAT model for the Vergara Basin.

Materials and Methods

Study area

The rainfall-runoff modelling application described in this paper focuses on the Vergara River Basin. It is located in Central Chile, between 37°29’ - 38°14’ S and 71º36’ - 73º20’ W (Fig. 1); covering an area of 4.265 km^2. Minimum and maximum mean monthly discharges occur during the month of July and February-March respectively (Table 1). The Biobio Basin to which the study area belongs, constitutes one of the country’s most important centres for forestry activities (eg. exotic species forestry plantations) and contains a major portion of the Chilean agricultural soils. It is influenced by the temperate climates of the South as well as by the Mediterranean climate of Central Chile.

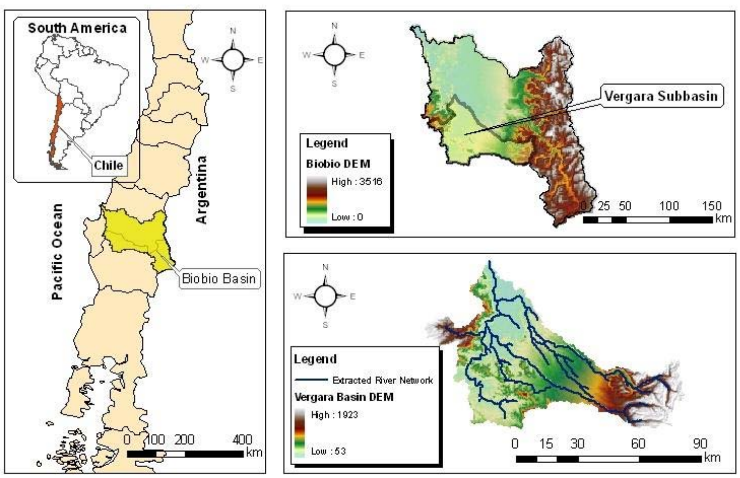


Figure 1. Location of the Vergara basin.

Selection of Vergara as a test area for the application of the SWAT model is based on the following criteria: (*i*) the availability of a typical set of basic input data which should allow for decent model calibration and validation; (*ii*) the absence of flow regulation due to hydropower infrastructure or operation of major irrigation works; (*iv*)the reduced portion of the basin with snowmelt contribution to river flow; and (*v*)the fact that the selected sub-basin constitutes an interesting test-case for evaluating impacts of land use change on basin hydrology, as major conversions between agriculture and forestry land use have been experienced in this area over the past decades(see e.g. Table 3).

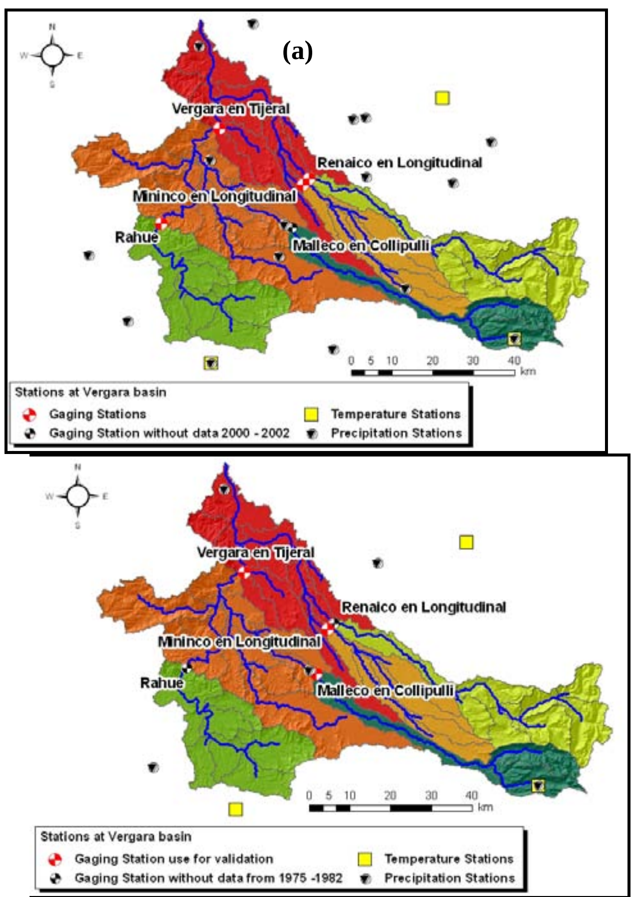
Table 1. Mean monthly discharges (m^3/s) at the different control points in the Vergara basin

	Tijeral	Rehue	Mininco	Renaico
imum	153.70 (July)	16.79 (July)	43.57 (July)	90.17 (June)

imum	7.52 (February)	0.28 (February)	2.16 (February)	6.71 (March)
n	56.32	5.85	16.21	42.66

Model Setup

For the application of the SWAT model (Di Luzio, et al. 2002, Neitsch, et al. 2005[a,] Neitsch, et al. 2005[b]) to the Vergara Basin, a 90m x 90 m Digital Elevation Model (DEM) (based on the final SRTM data sets) was used as a basis for the delineation of the river basin. Meteorological and fluviometric input data were obtained from the National Water DataBank ("*Banco Nacional de Aguas*") of the Chilean General Water Directorate DGA (Fig. 2a). First, an 11 years meteorological time series corresponding to the period 1992 – 2002 was used, together with a land use description for the basin (Fig. 5) based on the interpretation of aerial photographs (scale 1:70.000 and 1:115.000) from 1996-1998 (INE, 1999), which were combined with information from the "National Inventory of Vegetational Resouces of Chile" (CONAF - CONAMA – BIRF, 1995). The corresponding hydrological groups of the different soil series in the basin were derived from soil texture information contained in the "Official Description of Soil Series" of the "Agrological Study of the VIII and IX Region" (CIREN 1999[a,] CIREN 1999[b]). This was done by following the recommendations given by the US Department of Agriculture (USDA 1986, CIREN 1999[a,] CIREN 1999[b]).



(b)

Figure 2. Meteorological and gaging stations used for the (a) 1994 -2002 calibration and validation, and the (b) 1975-1982 validation.

Results

Calibration

Calibration was done for the period 2000 - 2002. Prior to the calibration exercise, a sensitivity analysis (LH-OAT analysis - Latin Hypercube Sampling/One at A Time; incorporated in the latest model version, SWAT2005; (van Griensven, et al. 2006) was executed for each control point where limnigraph were available, this in order to determine the top 8 parameters to which the model results are most sensitive. At each point, these 8 parameters are then used in the calibration process. The SWAT2003 "PARASOL" automated calibration procedure (Parameter Solution Method;(van Griensven and Bauwens 2003)) was applied separately to each one of the 4 subbasins. Results from the calibration process are given in Figure 3 and Table2.

From the results illustrated in Table 2, it can bee seen that best model performance is obtained for the basin closing at Tijeral (the biggest of the studied subbasins). Poorest results are obtained for the Renaico subbasin. Performance for Rehue (subbasin of Tijeral) is poorer than for Tijeral as a whole, but water yield for Rehue (observed & modeled) is proportionally much lower than for the remaining part of the Tijeral basin, so the performance of the model for Rehue has a relatively low impact on model performance at Tijeral. In general, the performance over the 3-year calibration period ranges from "very good" to "acceptable", based on an interpretation of statistical indicators using criteria from (Abu El-Nasr, et al. 2005).

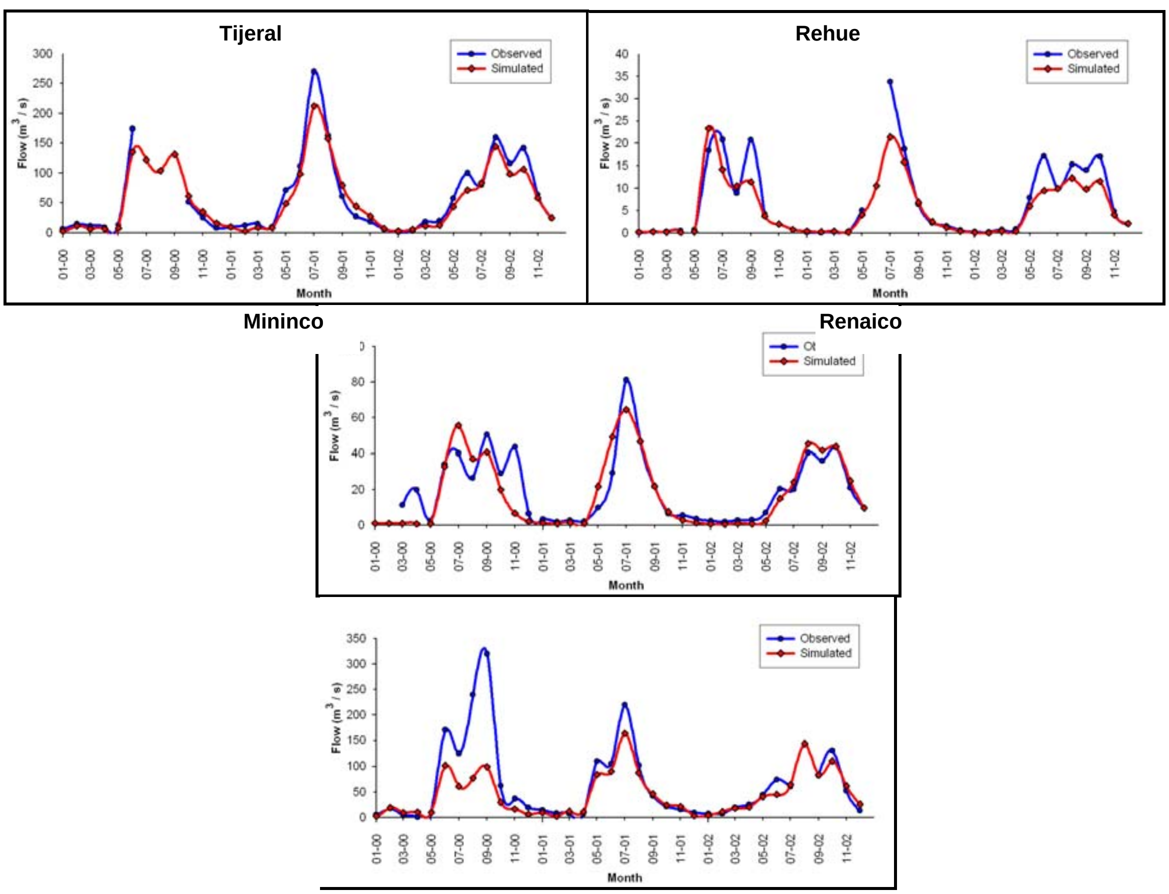


Figure 3. Observed versus simulated flow at the Tijeral, Rehue, Mininco and Renaico gaging station (calibration period)

Table 2: Statistical Indicators of model performance (monthly output) calculated at the different control points within the Vergara Basin: model calibration/model validation

x	Tijeral	Rehue	Mininco	Renaico
MSE	0.30 / 0.31	0.51 / 0.63*	0.50 / 0.33	0.82 / 0.42
ERR	11.64 / 8.24	1.99 / 2.15*	6.31 / 2.98	24.16 / 9.12
	0.93 / 0.93	**0.82 / 0.75***	**0.72 / 0.92**	**0.54 / 0.82**
	0.96 / 0.93	0.88 / 0.80*	0.76 / 0.94	0.71 / 0.83
AS	11.78 / 2.77	21.35 / 32.75*	8.32 / 9.13	32.04 / 7.88

RRMSE: Relative Root Mean Square Error; ABSERR: Mean Absolute Error; EF: Nash-Sutcliffe modelling efficiency; R^2: goodnes of fit: PBIAS: volumetric error. *Only considering data from July 1997

Validation

In order to assess the capability of the model to adequately reproduce basin hydrology under different scenarios of land use, model performance was evaluated under both “current” (land use map from 1994) and “historic” land use (based on an interpretation of satellite imagery from 1979; Fig. 4) conditions. Simulations were run using 1994-1999 and 1975-1982 time series of meteorological data, respectively. Table 2 gives the values of the different statistical indicators for the first validation period. General model performance over the first validation period is good.

For the second validation period, five precipitation stations and three temperature stations (Fig. 2b) were used. An additional flow gaging station (records available for this period only) was also incorporated. The model was run from 1975 to 1982. The first two years were considered to correspond to model warm up. The model was then run once more for the 1994-1999 period, using time series from the same stations as used for the second validation period, this in order to evaluate the effect of different configurations of meteorological input data sets on model outcome.

(a) (b)

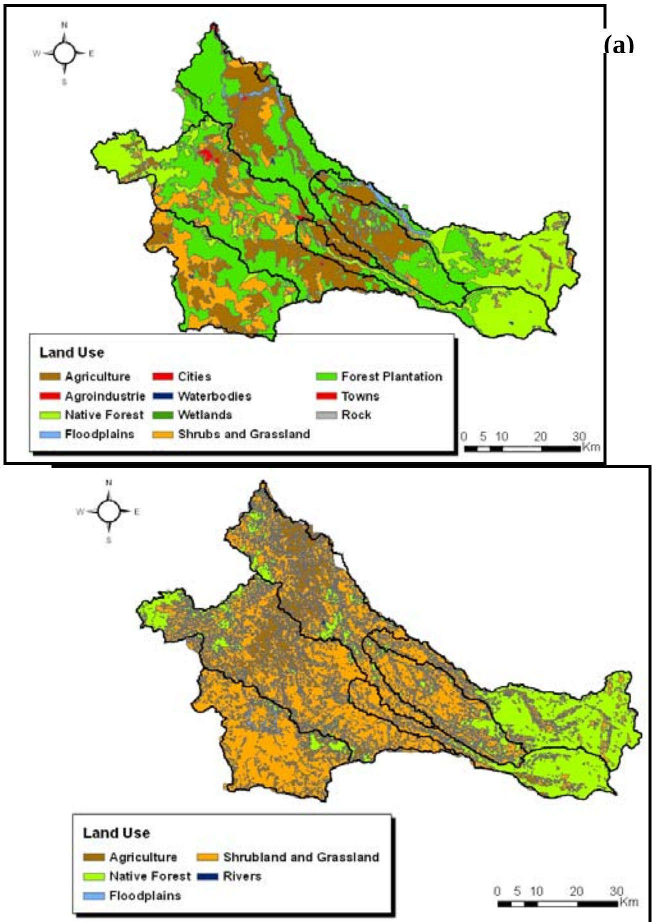


Figure 4. Land use classification (a) 1994 (b) 1979

Table 3. Percentage of area covered by the different land uses for the entire basin and the different sub-basins.

Land use	Basin		Tijeral		Mininco		Malleco	
	1979	1996	1979	1996	1979	1996	1979	1996
geland	**12.77**	**12.79**	**9.56**	**19.71**	**13.09**	**2.31**	**6.72**	**3.71**
ure	**48.83**	**1.66**	**57.01**	**0.01**	**64.40**	**0.00**	**28.76**	**0.08**
ve Forests	**26.29**	**23.37**	**19.01**	**22.64**	**20.41**	**9.49**	**59.36**	**60.25**
:stry Plantations	**0.00**	**39.44**	**0.00**	**35.99**	**0.00**	**49.07**	**0.00**	**23.81**
iculture	**11.42**	**21.11**	**13.28**	**20.92**	**2.08**	**39.12**	**4.87**	**11.89**

From Figure 5b and Table 3, it can be observed how in 1979 no exotic species forestry plantations were present in the basin. The percentages given in Table 3 are indicative of major land use changes observed in the Basin over this period.

Table 4. Statistical Indicators of model performance (monthly output) calculated at the different control points within the Vergara Basin, comparing the results from the periods 1979 – 1982 with those from 1994 – 1999 (Nash Sutcliffe efficiency / PBIAS):

od	sidered Meteorological Stations	Tijeral	Mininco	Malleco
99	All available stations	0.93 / 2.77	0.92 / 9.13	0.87 / 10.44
99	5 PP 3 T (2T extrapolated)	0.90 / 9.28	0.90 / 21.28	0.80 / 22.55
82	5 PP 3 T (2T extrapolated)	0.88 / 10.95	0.74 / 19.47	0.77 / 17.15

The results currently available from the double validation exercise are a first indication that the model continues to give an acceptable representation of basin hydrology under different conditions of land use.

Evaluation of model sensitivity to land use

Model sensitivity to land use changes was tested using 4 hypothetic scenarios (Fig. 5; Table 5). Scenario 1 (a) and 3 (c) are both based on the concept of “no change” to the native forest cover. This concept is inspired on the current Chilean legislation (Decreto Ley 701, 1974). “Extreme” scenarios 2 (b) and 4 (d) consider complete forestation with exotic species, and complete conversion to agriculture, respectively. The calibrated and validated SWAT model was used to simulate basin hydrology under different land use scenarios using a common meteorological time series (i.e., time series of the 1994 – 1999 reference period). As such, each simulation differed only from the 1994-1999 reference model run by its land use conditions. This was done with the specific purpose of comparing and evaluating land use impacts on river flow.

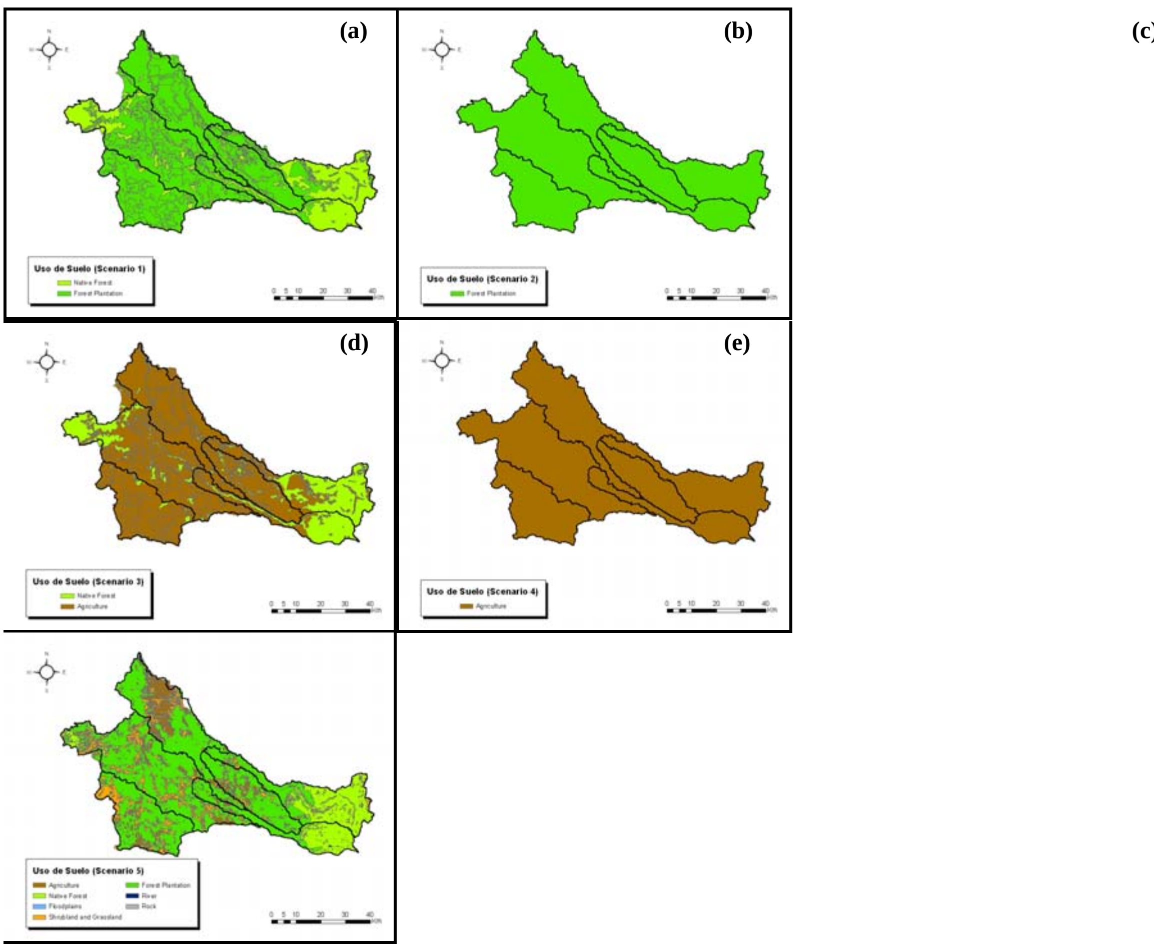


Figure 5. Different land use scenarios generated for the Vergara Basin (a) Scenario 1, in which all non-native forest areas are occupied by forestry plantations; (b) Scenario 2 consisting of 100% coverage by forestry plantations; (c) Scenario 3, in which all non-native forest areas are occupied by agriculture; (d) Scenario 4 consisting of 100% coverage by agricultural areas, and (e) baseline scenario

Table 5. Area (km^2) of the different land uses for all the basins and sub-basins

	Actual				Scenario 1				Scenario 2				Scenario 3				Scenario 4			
	R	NF	FP	A	R	NF	FP	A	R	NF	FP	A	R	NF	FP	A	R	NF	FP	A
Basin	546	997	1682	900	0	997	3269	0	0	0	4265	0	0	997	0	3269	0	0	0	4265
Tijeral	455	523	832	483	0	523	1788	0	0	0	2311	0	0	523	0	1788	0	0	0	2311
Mininco	10	42	215	172	0	42	397	0	0	0	438	0	0	42	0	397	0	0	0	438
Renaico	36	417	164	44	0	417	266	0	0	0	683	0	0	417	0	266	0	0	0	683
Rehue	226	20	270	148	0	20	648	0	0	0	668	0	0	20	0	648	0	0	0	668
Malleco	15	245	97	48	0	245	162	0	0	0	407	0	0	245	0	162	0	0	0	407

Rangeland: R; Native Forests: NF; Forestry Plantations: FP; Agriculture: A

Discussion

The currently available analysis of the outcome of this modelling exercise consisted of a comparison, at each one of the 5 considered sub-basins, of the mean of mean annual discharges from the baseline scenario with those obtained from the simulation of the different hypothetic land use scenario (Fig.6 and 7). It can bee seen that increasing the area covered by forestry plantations will in general produce a reduction of the mean annual flow, whereas an increase of the of area under agriculture will produce a increase in mean annual flows, this is consequent with the results obtained by Thanapakpawin, et al (2007). Additionally, a specific comparison of scenario results obtained for both the dry (mean values for November to April) and wet period (mean values from May to October) was made (Table 6), Mayor relative changes are expected to happen at Rehue sub-basin followed by Malleco sub-and Tijeral subbasin (Rehue and Malleco are nested sub-basins of Tijeral). Scenario four will in general provoke the mayor changes in all the basins. Scenario 2 at Rehue shows an high increment in the flows, that is opposite to the changes shown in all basin for this case, were a reduction of the flows occurs. The same occurs at Malleco but for scenario 4, in this case all basins show an increment in the flows and for malleco there is a reduction of them. Siriwardena, et al. (2006) expressed that non-uniform variations in land uses over the basin and spatial and temporal variation in rainfall, are some of the factors that inhibit consistent conclusions from large basins regarding land use changes.

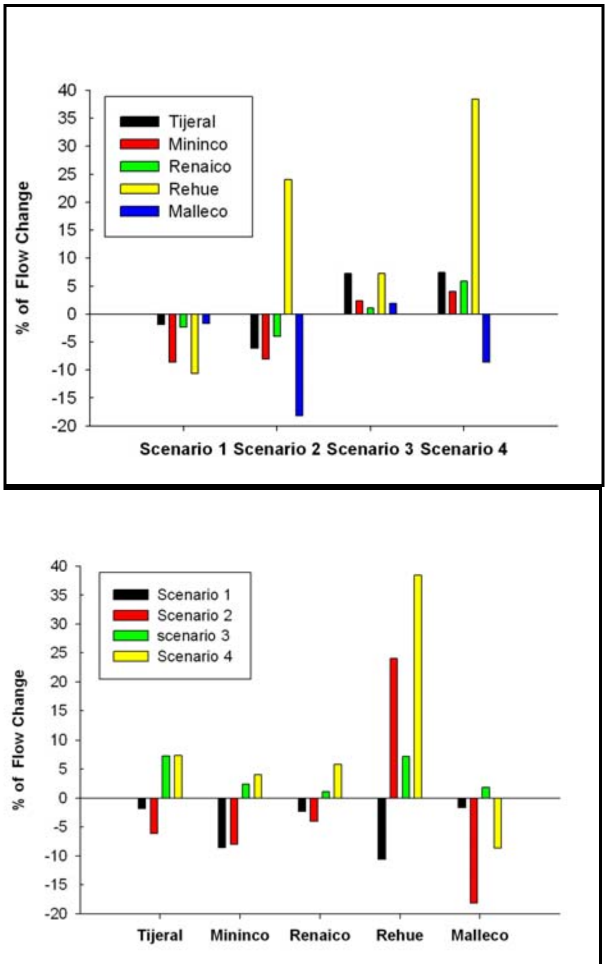


Figure 6. Percentage of change in mean annual flow with respect to the baseline scenario comparison between (a) each sub-basin for the different scenarios (b) each scenarios for the different sub-basin.

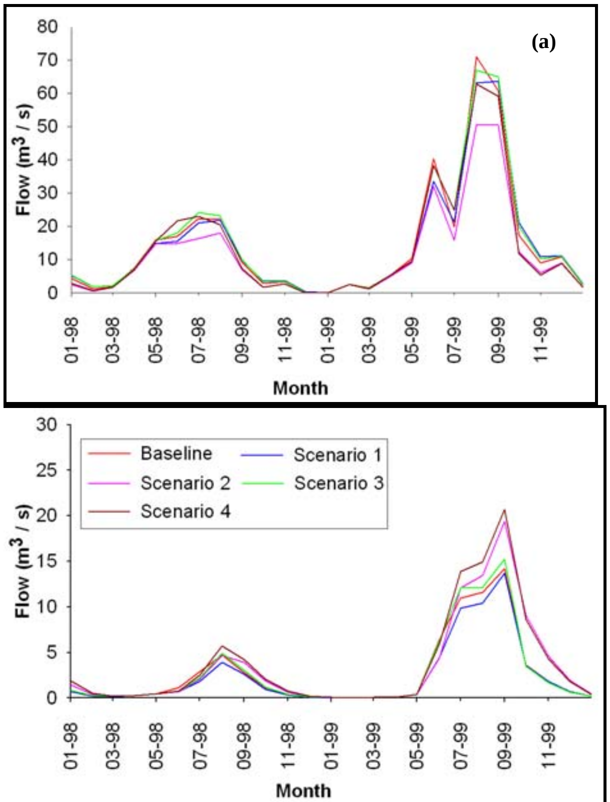


Figure 7. 1998-99 monthly hydrographs for the different land use scenarios (a) Tijeral sub-basins (b) Rehue sub-basin

Table 6. Percentage of change from the baseline scenario for mean annual, wet season (May – October) and dry season (November –April) flows. (Annual (A), Wet season (W) and Dry season (D))

	Tijeral			Mininco			Renaico			Rehue			Malleco		
	A	**W**	**D**	**A**	**W**	**D**	**A**	**W**	**D**	**A**	**W**	**D**	**A**	**W**	**D**
Scenario 1	-1.86	-2.81	2.93	-8.57	-9.02	-4.42	-2.32	-2.81	0.06	-10.61	-10.67	-9.94	-1.69	-2.48	1.19
Scenario 2	-6.14	-6.63	-3.33	-8.04	-8.46	-4.13	-3.99	-4.95	0.36	24.04	20.66	59.98	-18.21	-18.36	-16.68
Scenario 3	7.30	7.28	6.32	2.38	2.63	-1.06	1.13	1.27	0.81	7.23	7.70	1.55	1.86	1.37	3.86
Scenario 4	7.40	8.42	0.75	4.03	4.38	-0.35	5.85	6.67	2.51	38.41	34.96	76.02	-8.65	-7.27	-13.54

Conclusions

The current model version successfully passed two validation exercises considering monthly outputs, in which 2 different land use conditions were considered (actual vs. past). However, further research is required in order to confirm the explanatory power of the model with respect to land use impacts on basin hydrology, considering daily outputs. Taking into consideration this last observation, the model can already be used for making a first preliminary assessment of the potential impacts of land use changes.

The results shows in this study are an examination of the sensitivity of hydrology to a particular aspect of the ecosystem dynamics. This results only includes the response of the hydrology to changes in vegetation, to have more accurate results for plausible future scenarios changes in precipitation and temperature must also be included. However, this is a first step which provides an approach into how hydrology responds to land use change and will be valuable when analyzing relations between land use change, climate and hydrology.

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SWAT in Land Use Planning: Simulating Impacts of Density and Physical Layout of Residential Subdivisions on the Hydrology of an Urbanizing Watershed

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Abstract

This paper describes the use of SWAT (Soil and Water Assessment Tool) to assess the impact of different residential-development scenarios on watershed behaviour. Calibration of the SWAT simulations is done using the Generalized Likelihood Uncertainty Estimation (GLUE) method. The pre- and post-processing of SWAT's input/outputs are done in ArcView® while impervious surface is estimated from remote sensing images. Using SWAT and ArcView GIS, the study investigates the relationship between stream flow and runoff ratio as a function of percent impervious cover under eleven different residential-development scenarios, varying in density and physical layout. The results indicate that there are differences in the potential runoffs generated by the different scenarios and their subsequent impacts on stream flows. However, at the same design capacity (gross dwelling units/acre) the amount of runoff from high-density compact developments is similar to that from low-density sprawl developments. The study thus confirms a relationship between runoff and pattern of urban development and demonstrates that watershed model can be used to understand the impact of development characteristics on the hydrology of watersheds. The model can generate *what-if* scenarios of a watershed under study, something that is useful to land use planners in making decisions on a variety of land-use options.

KEYWORDS: SWAT, residential density, impervious surface, watershed

Introduction

Land use regulations are one of several tools urban planners use to control physical characteristics of developing landscapes. Through these regulations, planners can impose restrictions on various development variables including land use types and activities, density, open space and even extent of impervious surface. The intrinsic relationships between land use regulations and the physical structure of an urban watershed suggest that there should be mechanisms through which planners can evaluate the environmental consequences of existing regulations and improve the scientific basis of future decision making to further minimize negative effects on the watershed. An integrated application of geographical information system (GIS), remote sensing and numerical modeling has the potential to contribute to this informed decision-making process by providing synoptic data collection capabilities and analysis techniques that can generate pertinent information about watershed implications of planning decisions.

One example of such integrated uses is the hydrological modeling of a watershed. For this, data from GIS and remote sensing can be together used to parameterise the simulation of a hydrological model to evaluate the impact of planning decisions on watersheds. This paper discusses the hydrological simulation part of the integrated application. This is part of the study carried out by the author to investigate the application of remote sensing and GIS in assessing the influence of density and subdivision design on residential-area imperviousness with a demonstration on the application of the results for hydrological simulation of a watershed. For techniques on estimating impervious surface using remote sensing, readers are encouraged to refer to Majid (2006), Yang et al. (2003), Wu and Murray (2002) and Ward et al. (2000). A good review of the techniques is provided by Slonecker et al. (2001).

The objectives of the paper are: 1) to analyse the change in the flow behaviour of an urbanizing watershed due to different residential densities and subdivision designs; and 2) to demonstrate an application of GIS coupled with numerical modeling (i.e. SWAT) in predicting the implications of urban planning (land use) regulations. The findings from this study can help one understand how a combination of density and subdivision design affects the hydrology of a watershed.

Subdivision Design and Impervious Surface

In hydrology impervious surface can generally be defined as any material that prevents the infiltration of water into the soil. Subdivision design affects the amount of impervious surface in many ways, both directly and indirectly. For a start, the density of a subdivision determines the extent of building footprints and driveways, two major components of impervious surface, in a subdivision. Impervious surface from the streets serving these buildings meanwhile is influenced by both the street standards and the design of street network, which is very much influenced by the design or layout of the subdivision. Conventional or standard design, as contrasted to cluster subdivision in Figure 1, typically leads to the development of residential subdivisions that completely blanket a parcel with evenly-spaced lots. This results from zoning provisions that require minimum lot sizes and widths, and streets that front every lot. In addition, large lots and front yard setbacks necessitate even more pavement to connect garages and front doors to streets. With the entire pavement connected, there is much less opportunity for runoff to soak into the ground.

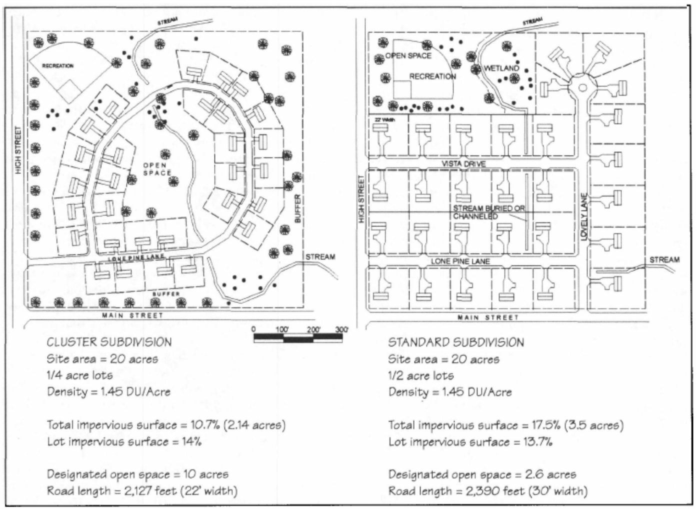

Figure 1: Conventional (standard) versus cluster subdivisions
Source: Arnold & Gibbon, 1996

Although originally intended for open space preservation, cluster subdivisions have of late been promoted as an alternative design to reduce imperviousness (Arnold and Gibbons, 1996; Schueler, 1996; Arendt, 1996). In *density-neutral* provision for cluster development, a developer is still limited to the total units as in conventional development but has the flexibility to place them in a way that is more responsive to the physical characteristics of the site. In spite of more open space, gross density is maintained by allowing smaller lots than the conventional developments. Smaller lot size that results in narrower and shallower lots also helps reduce the lineal length of streets and total length of driveways. Clustering lots closer together also results in shorter roads. Through proper site planning, roads and houses can be located at higher elevation away from natural drainageways, leaving them as open space and giving runoff opportunity to infiltrate while slowly traveling downhill towards the drainage.

In terms of its hydraulics, impervious surface/area within residential subdivisions can be distinguished into two categories, i.e. effective impervious area (EIA) and non-effective impervious area (non-EIA). EIA comprises those impervious surfaces that are hydraulically connected to the stream (through impermeable storm sewers, culverts, etc.) while non-EIA consists of those impervious areas that drain to pervious ground such as a roof drains onto a lawn or a street draining onto a grass swale, allowing some infiltration of runoff into the soil (Alley & Veenhuis, 1983). The sum of both EIA and non-EIA is the total impervious area (TIA). For assessing watershed impact, EIA is a better indicator than TIA but more difficult to quantify.

Majid (2005) conducted a study on the effects of density and subdivision design on the total amount of impervious surface (TIA) and its components, i.e. EIA and non-EIA, for 115 subdivisions in North Carolina, USA. The results of the study confirmed findings by other researchers that the amount of impervious surface is positively correlated with density, regardless of subdivision design. Even though he found that overall imperviousness was lower in cluster subdivisions, the significant difference was in impervious surface originating from within residential lots rather than from streets right-of-way. Detailed description on the methods and results of the impervious surface study can be found in Majid (2005). The level of imperviousness according to density and subdivision design (conventional vs. cluster) obtained from the study is utilized in this study, with all impervious surface from streets right-of-way assumed to be EIA if the streets are curbed.

Materials and Methods

Study Area

The study area is the watershed of a tributary of Swift Creek, located within the jurisdiction of the City of Raleigh and Wake County, North Carolina, USA (Figure 2). The existing land use/land cover of the 4030-acre (1631 ha) watershed is mostly vegetation (forest and agriculture) which accounts for about 63% of the area (Figure 3). Residential area, concentrated along the boundary, makes up about 29% of the area, with 90% of the total residential area having a density of 1 d.u./acre or less. Commercial area is located at the northern part of the watershed and constitutes only about 3% of the watershed. The remaining land use is main roads (3%) and water bodies (2%). The current policy on development within the area limits future development to only residential uses with an average of 1 d.u./acre (City of Raleigh, 2002).

The watershed lies between eighty to one hundred fifty meters above sea level with the high points generally located on the northeast corner and low points on the southeast corner. Based on the SSURGO soil database, there are about 33 types of soil uniformly distributed throughout the watershed. Figure 2 also shows the stream networks within and around the watershed and the location of the USGS gage that acts as the watershed outlet point. The gage was operated by USGS for only about one and a half year, from May 2002 - September 2003. A climate station recording daily climate data (precipitation, temperature, solar radiation, etc.) is located at Lake Wheeler Road adjacent to the site (Figure 2).

Model Set Up

The watershed boundary of the study area was delineated using the automatic watershed delineation process embedded in AvSWAT based on the 6m DEM of the study area and the 1:14000-scale stream networks. The location of the USGS gage acts as the

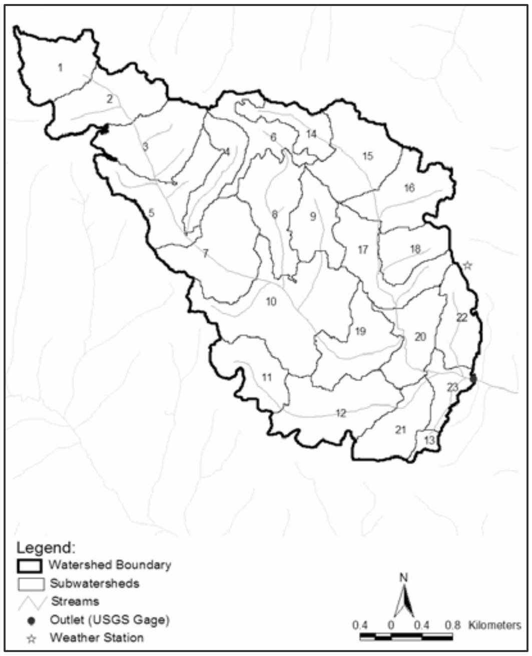

Figure 2: Study Area

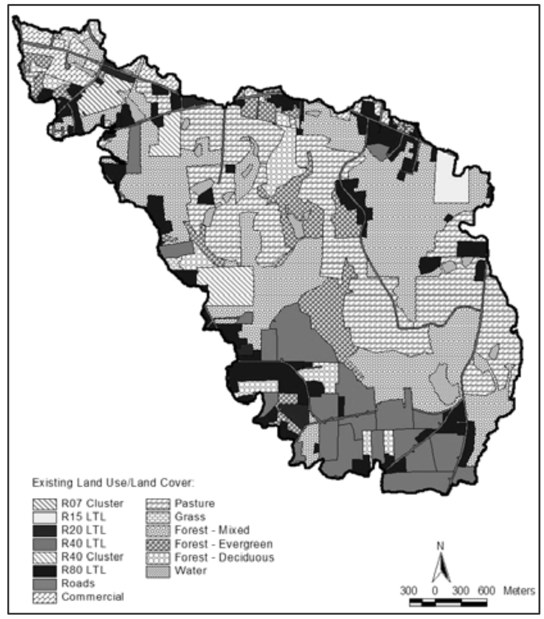

Figure 3: Existing LULC of the study area

Note: R15 LTL = Conventional subdivision with 15000ft^2/unit houses (~ 3 d.u./acre density)

downstream outlet of the watershed and the resulting watershed is as previously shown in Figure 2. A total of 23 subwatersheds were created ranging from 22 to 437 acres. Variations in topography and land use were considered when deciding the coverage of each subwatershed. Through the GIS overlay process embedded in AvSWAT, Hydrological Response Units (HRUs) are determined based on user-defined minimum threshold percentage of land use/land cover (LULC) in a subwatershed and minimum threshold percentage of soil type in the each accounted LULC type. For this study, the threshold values used are 0% and 5%, meaning that all LULC in a subwatershed and all soil types that make up at least 5% of each LULC are included in overlaying the two layers (LULC and soils) to create HRUs. This resulted in a total of 625 HRUs for the whole watershed with the number of HRUs per subwatershed ranges from twelve to forty eight.

In SWAT, requirements for climate data depends on the calculation mode of potential evapotranspiration (PET) chosen by the user and SWAT offers four modes of PET calculations. For this study, the Priestly-Taylor method was used and the data required by this method are minimum and maximum temperature, daily solar radiation and relative humidity. All of these data can either be input by the user or generated by SWAT using its built-in weather generator. This study, however, used observed data from the Lake Wheeler Road Field Lab for all of the climate inputs except for a few dates for which solar radiation and relative humidity data were missing. The dates involved are January to May of 1998, the first few days of the 5-year spin-up period. For these dates, the built-in weather generator was used. Since the dates concerned are within the spin-up period instead of the calibration period, the impact of not using the actual field data during these dates on the calibration results is minimal.

Once all the required inputs are in place and the model is set up, calibration of the stream flow at the outlet of the watershed is carried out against the daily observed stream flows obtained from the USGS Gage Station Number 0208762750 located at the outlet for the period of one water year from 10/1/2002 to 9/30/2003. The simulation is carried out under the existing LULC condition at the time of the simulation year. The beginning date of simulation is however set at 1/1/1998 since the model requires the initial few years to be used as 'spinning up' years. Hence, the results of the simulation from 1/1/1998 up to 9/30/2002 are not used in the analysis and only the results for the period of 10/1/2002 to 9/30/2003 are used.

Calibration of the model is done using the Generalized Likelihood Uncertainty Estimation (GLUE) approach (Beven, 2001; Beven & Binley, 1992). In this study, the hydrological model SWAT is coupled with the GLUE method in LINUX environment using a Python scripting system (Shin, Band & Hwang, 2005). The GLUE method is an extension of the generalized sensitivity analysis of Spear and Hornberger (1980) and is used for calibration and uncertainty estimation of models based on generalized likelihood. The method incorporates information from multiple acceptable model parameter sets within a Bayesian Monte Carlo framework. To do this, a large number of model runs are carried out with each run parameterized with randomly chosen parameter values from uniform distributions across the range of each parameter. The acceptability of each run is assessed in this study by the Nash-Sutcliffe Coefficient of Efficiency (COE) between log values of simulated and observed stream flows. The Nash-Sutcliffe COE was first defined by Nash and Sutcliffe (1970) as the coefficient of efficiency that ranges from minus infinity to 1.0, with higher values indicating better agreement between the observed and the simulated flows. For this study, the parameter set that gives the highest value of Nash-Sutcliffe COE is taken as the optimal parameter set and used in subsequent modeling of the watershed under different LULC scenarios.

Based on past simulation studies using SWAT (Spruill et al., 2000; Eckhardt et al., 2003) and recommendations from the model's manual (Neitsch et al., 2002), initial batch of parameters that are potentially sensitive and their range of values are identified. These parameters are: i) groundwater delay (days); ii) the minimum threshold depth of water (mm) in the shallow aquifer for return flow to occur; iii) the groundwater "revap" coefficient; and iv) the minimum threshold depth (mm) of water in the shallow aquifer for "revap" or percolation to the deep aquifer to occur. "Revap" is defined as the movement of water from shallow aquifer into the overlying unsaturated zone as a function of water demand for evapotranspiration. With the parameters confined within their pre-selected ranges, a total of 1000 simulations are run, each with random parameter values selected from the ranges. The acceptability of each run is then assessed by comparing simulated to observed stream flows through a chosen likelihood measure, in this case the Nash-Sutcliffe COE. The run that gives the highest likelihood value is then identified. This run is the best simulation to represent the observed stream flow and it gives the optimal parameter set (excluding imperviousness level) to be used in running flow simulations under different LULC scenarios.

Different LULC Scenarios for Flow Simulations

Once the model has been calibrated, flow simulations are carried out under different LULC scenarios to compare the impacts of different imperviousness levels (due to different subdivision designs) on watershed hydrology. A set of hypothetical LULC scenarios representing different levels of imperviousness are formulated by assuming additional subdivision development in the watershed. Incremental effects of imperviousness from these different hypothetical scenarios are compared to the effects of imperviousness from the baseline scenario. The baseline scenario is the existing LULC in the watershed with no additional subdivision development. A total of eleven LULC scenarios (one baseline scenario plus ten hypothetical scenarios) were selected on the basis of the variations in the imperviousness-related variables of the new development. Table 1 summarizes the characteristics of all the scenarios including the imperviousness levels. Important variations among the scenarios are the size and density of new development which determine the amount of TIA, the type of subdivision design (conventional vs. cluster) which also affects the amount of TIA and the practice of street curbing which influence the amount of EIA.

Table 1: Brief characteristics of simulated development scenarios

		LULC		New Development					Imperviousness	
Scenario	Description	Developed (%)	Undeveloped (%)	Area (acres)	Density (d.u./acre)	Units	Design	Curb	TIA (%)	EIA (%)
1	Existing development (Baseline scenario).	35.0	65.0	-	-	-	-	-	8.0	3.0
2	Additional 2400 acres of residential area at 1 d.u./acre as zoned. Streets are not curbed.	94.5	5.5	2400	1	2400	*	No	16.0	3.0
3	Additional 2400 acres of residential area at 1 d.u./acre as zoned. Streets are curbed.	94.5	5.5	2400	1	2400	*	Yes	16.0	6.5
4	Additional 2400 acres of conventional residential area at 4 d.u./acre. Streets not curbed.	94.5	5.5	2400	4	9600	Conv.	No	29.0	3.0
5	Additional 2400 acres of conventional residential area at 4 d.u./acre. Streets are curbed.	94.5	5.5	2400	4	9600	Conv.	Yes	29.0	11.0
6	Additional 2400 acres of cluster residential area at 4 d.u./acre. Streets are not curbed.	94.5	5.5	2400	4	9600	Cluster	No	27.0	3.0
7	Additional 2400 acres of cluster residential area at 4 d.u./acre. Streets are curbed.	94.5	5.5	2400	4	9600	Cluster	Yes	27.0	10.0
8	Additional 600 acres of conventional residential area at 4 d.u./acre. Streets not curbed.	50.0	50.0	600	4	2400	Conv.	No	13.0	3.0
9	Additional 600 acres of conventional residential area at 4 d.u./acre. Streets are curbed.	50.0	50.0	600	4	2400	Conv.	Yes	13.0	5.0
10	Additional 600 acres of cluster residential area at 4 d.u./acre. Streets are not curbed.	50.0	50.0	600	4	2400	Cluster	No	12.5	3.0
11	Additional 600 acres of cluster residential area at 4 d.u./acre. Streets are curbed.	50.0	50.0	600	4	2400	Cluster	Yes	12.5	4.5

Note: Conv. = Conventional.
* - Conventional or cluster design since no difference in TIA and EIA between the two designs at this density.

Results and Discussion

Calibration and Simulation of Flows

Calibration of the model for daily stream flows indicates sufficient performance of the model with a Nash-Sutcliffe COE of 0.78. Figure 4 shows the time series of the observed and simulated daily stream flows. The relatively high Nash-Sutcliffe COE qualifies the model for simulating the daily stream flows within that one year period under different LULC scenarios. The simulated daily stream flows of Scenarios 2-11 are shown against the existing stream flow (Scenario 1) in Figure 5. The time series of stream flows depicted in the figure seem to fall into three distinct groups. The first is the time series for the existing scenario which have higher volume of low flows and lower peak flows. The second group is made up of flows from Scenarios 2-3 and 8-11 whose time series stay very close to that of the existing scenario with no obvious differences within the group. The third group is formed by flows from Scenarios 4-7 whose low flows are distinctively lower and peaks higher. Differences in flows are noticeable even within the group itself where Scenario 5 which has the highest TIA and EIA generates more extreme flows than the other scenarios. Thus, through observation of the time series alone it is evident that higher imperviousness levels (Scenarios 4-7) generate more impact on stream flows.

Impact on Annual Flows

Table 2 lists the annual values of stream flow, baseflow, surface flow and runoff ratio against the TIA and EIA values for each scenario. Under the existing land use (Scenario 1) where only 35% of the watershed is developed and 8% of the watershed is impervious surface, the annual stream flow measures 26.4 inch (671 mm) with baseflow makes up 62% of it and the remaining 38% is from surface flow or stormflow. The annual runoff ratio of the

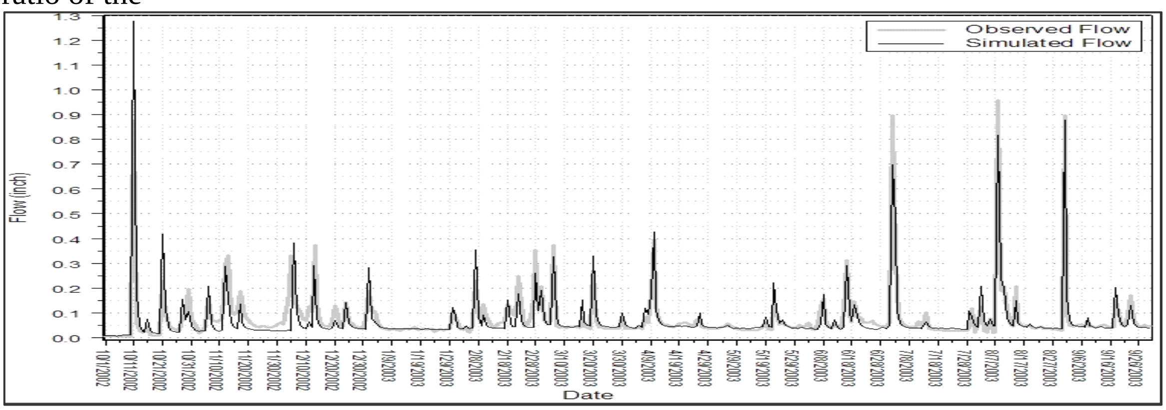

Figure 4: Observed and simulated daily stream flows for the demonstration watershed from 10/1/2002 - 9/30/2003. Nash-Sutcliffe COE between the two flows is 0.78.

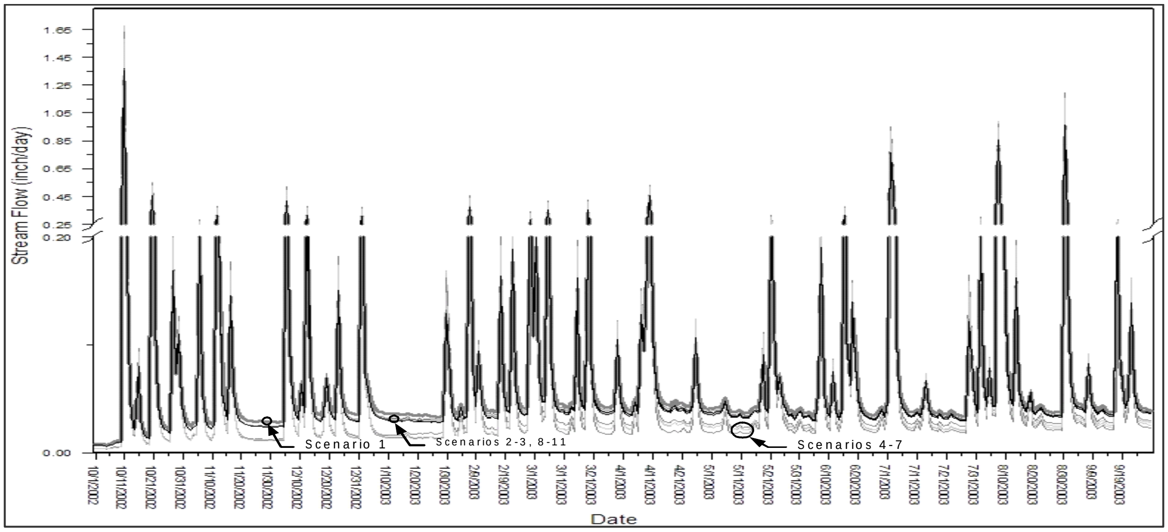

Figure 5: Simulated daily stream flows for the demonstration watershed from 10/1/2002 - 9/30/2003 under different LULC scenarios.

watershed under the existing land use observed at its outlet is 14.5%, which means that 10 inches (254 mm) of the total 69 inches (1753 mm) of the precipitation falling within the watershed that year becomes runoff. Even though the annual stream flows differ very slightly among scenarios, there are significant differences in the values of the components of stream flows and those of runoff ratios. The percentage of surface flow increases with increasing TIA, accompanied by corresponding reduction in the percentage of baseflow. The increasing surface flow results in increased runoff ratio as shown in the table where runoff ratio increases from 14.5% when TIA is 8% to almost 23% for TIA of 29%.

Table 2: Annual values of flow components and runoff ratio under different LULC scenarios

	Scenario										
	1	2	3	4	5	6	7	8	9	10	11
TIA (%)	8	16	16	29	29	27	27	13	13	12.5	12.5
EIA(%)	3	3	6.5	3	11	3	10	3	5	3	4.5
Stream Flow (in)	26.4	26.8	26.8	25.1	24.7	25.5	25.0	26.0	25.9	26.1	26.0
Baseflow (%)	62.2	56.3	56.3	38.9	36.2	42.9	39.9	56.4	55.8	57.4	56.7
Surface Flow (%)	38.1	43.7	43.7	61.1	63.8	57.1	60.1	43.9	44.6	42.9	43.6
Runoff Ratio (%)	14.5	16.9	16.9	22.2	22.8	21.1	21.7	16.5	16.7	16.2	16.4

Impact on Daily Flows

In spite of the small differences in annual stream flows between the scenarios, there are significant differences in how some scenarios respond to individual precipitation events. In general flow volumes increase with precipitation except in a few cases where low stream flows are generated by high precipitations occurring after a period of dry days or high flows by low precipitations immediately following other precipitation events (Figure 6). The data appear to form two distinct groupings of slopes that are significantly different for precipitations equal to or higher than 1 in/day. It seems that precipitations less than 1 in/day are not high enough to make any difference in runoff between scenarios. For precipitations equal to or higher than 1 in/day, Scenarios 4-7, with higher TIAs, show higher stream flow responses to increasing precipitation than the other scenarios (Figure 7). On the other hand, the responses of Scenarios 2, 3, 8-11 to precipitation are not significantly different from the baseline scenario.

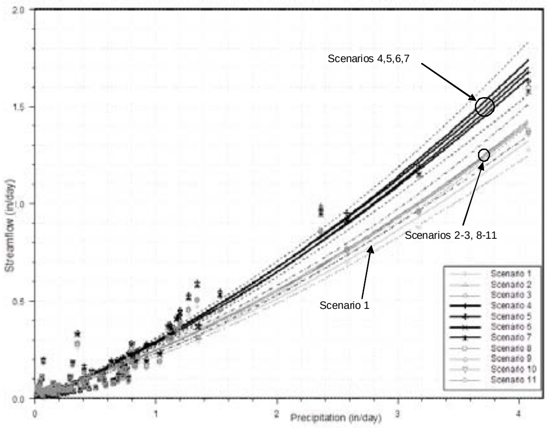

Figure 6: Stream flows versus precipitation for days with precipitation

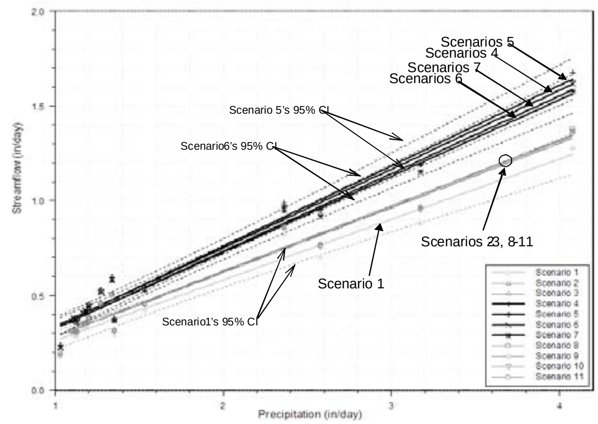

Figure 7: Stream flows versus precipitation for days with precipitation more than 1 inch/day

The distribution of daily stream flows from each scenario is plotted in Figure 8 for all days of the simulated year (a), days with measurable rainfalls within the simulated year (b), and dry days within the simulated years (c). For the whole simulated year, there are significant differences in the amount of stream flows among the scenarios. Test of non-parametric Kruskal-Wallis rank test on this data set gives a statistically significant Kruskal-Wallis chi-square of 19.9 (df = 10, p-value = 0.03). The Kruskal-Wallis rank test is employed since the distribution of stream flows is not normal, i.e. positively skewed as can be seen in the plots. Figure 8(a) shows the differences in the stream flows where flows from the more impervious Scenarios 4-7 have higher peaks but lower medians. Higher flows in Scenarios 4-7 during precipitation events are confirmed by Figure 8(b) which shows only stream flows for days with rainfalls. The difference in the amount of stream flows among the scenarios is tested to be statistically significant (Kruskal-Wallis chi-square = 19.9, df = 10, p-value = 0.03). The differences in amount of stream flows are also tested for days of no rain and are found to be statistically significant (Kruskal-Wallis chi-square = 761.8, df = 10, p-value = 0). The distribution of the flows is as shown in Figure 8(c) and it can be seen that during period of low flows (dry days) Scenarios 4-7 record lower flows than the other scenarios. Thus, in comparison to the other scenarios, Scenarios 4-7 are more likely to have lower flows during dry period and higher flows during wet period. This characteristic is common in an urbanized watershed where stormflow periods are brief but with high volumes and rapid recession rates (Konrad and Booth, 2002).

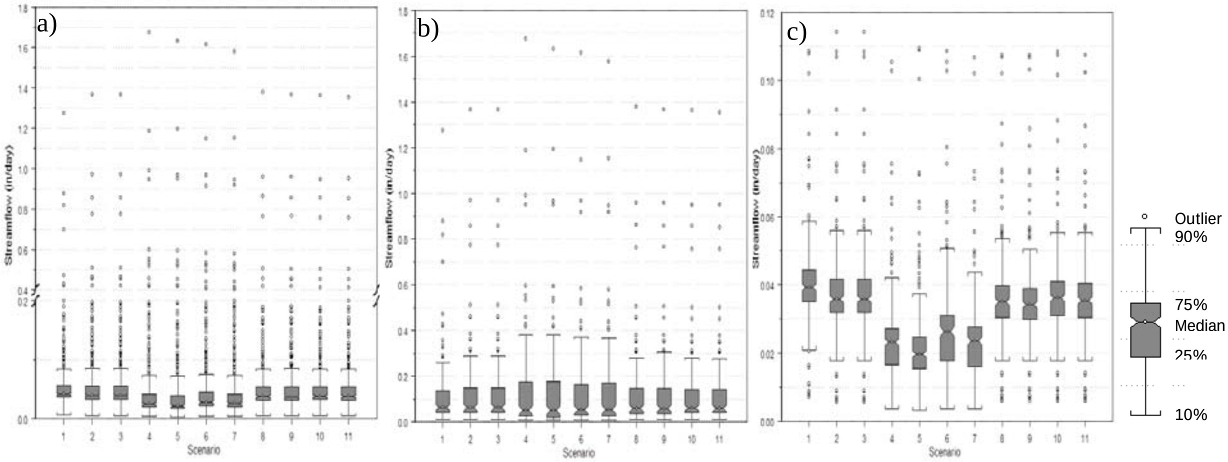

Figure 8: Distribution of stream flows according to scenarios for a) the whole simulated year; b) days with precipitation event; and c) days with no precipitation event.

It is also interesting to investigate what happens to the impact of imperviousness if design capacity is taken into consideration. This is done by comparing Scenarios 8-11 to Scenario 2 & 3. While generating the same amount of new units (2400), Scenarios 8-11 require only one fourth of the new development area required for Scenarios 2 & 3 since they are four times as dense. Interestingly, the results of the simulated flows for Scenarios 8-11 are very close to those of Scenarios 2 & 3 (Figures 6-8) but Scenarios 8-11 conserve about 50% of the watershed against 5% by Scenario 2 & 3. The results thus support the position taken by Richards et al. (2003) and Stone Jr. (2004) that design capacity should be taken into consideration when considering imperviousness since considering only coverage percentage alone would unfairly give the wrong impression of high density development.

Conclusions

All of the flow parameters discussed above are affected by the amount of impervious cover which is influenced by density and subdivision design. In addition to its amount, impervious cover also affects runoff through the nature of its connectivity to drainage system, directly or indirectly connected, and through its location within the watershed. The amount according to type (connectivity) of impervious surface was handled well in the model since the author had the option of inputting observed values instead of using the default values. Being a semi-distributed model, however, SWAT can only handle the location problem of the impervious surface to a certain degree. If subwatersheds within a watershed are delineated small enough and the land cover resolution is high enough to have only binary impervious and pervious classifications then the model technically could address this location problem. This, however, would require tremendous computing time even for a small watershed like this one, not to mention the time required to prepare the detailed land cover data. Since the LULC information available and inputted into the model was only detailed enough to give the lumped percentage of impervious surface and its type within a subdivision, not the spatial location of the impervious cover, subdivision designs treated could only capture the difference in impervious surface amount. Classifying streets as curbed or non-curbed meanwhile determined only the distribution of different types of impervious cover, i.e. EIA and non-EIA. Hence any difference in the results due to subdivision design and type of street curbing could only be attributed to the difference in imperviousness amount and the composition of its types, but not to the location of the impervious surface.

In spite of the above shortcoming, the study successfully shows the impacts of different levels of imperviousness on watershed hydrology and the potential of coupling GIS with numerical modeling in urban planning applications. The impacts are less on the total amount of annual stream flow within a watershed but more on how the watershed responds to precipitation events as reflected in the daily stream flows. For the same precipitation events, a more impervious watershed would have higher peak flows with rapid recession rates during wet periods, resulting in lower flows during dry periods. This is due to higher runoff ratio and less infiltration because of higher imperviousness. The impact of different subdivision designs and the practice of street curbing, as translated into the differences in the amount of TIA and EIA among the simulated scenarios, is rather small. This is because of two factors: 1) the small difference in imperviousness level between the two designs; and 2) the impacts of subdivision design are only represented by the level of imperviousness without taking into account the location of impervious surface. The outcome would likely be different if location were taken into account. Cluster subdivisions, for example, are theoretically more flexible than conventional subdivisions in accommodating site design that would allow for longer period and distance of runoff travel over pervious ground, hence encouraging infiltration.

On the other hand, holding subdivision design and area constant, density seems to have greater influence on flows. This is simply because imperviousness level varies greater across density than between subdivision designs. With or without curbed streets, higher density development generates noticeably higher rate of runoff and lower baseflow than its lower density counterpart. However, if design capacity is taken into consideration, higher density development has almost the same effects on hydrology per dwelling unit as the lower density one. This fact seems to make higher density development more favorable when a situation reveals that developable land is limited and the number of dwelling units is fixed. Thus, comparing imperviousness among residential development options should be done with design capacity taken into consideration to give an unbiased report of the imperviousness level of each option per dwelling unit.

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DEFINITION of HRU using Area Fraction Images derived from Spectral Unmixing

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Abstract

Since the rainfall-discharge relationship is modeled by SWAT in a semi-distributed way, there is no need for the land use information to be fully spatially explicit. This feature opens up the possibility of defining HRU by combining a soil map with land use information in the form of area fraction images (AFI). Such AFI are the typical end product of sub-pixel classification of multi-spectral imagery of low to middle spatial resolution. We derived AFI for 5 land use classes from a time series of SPOT-VGT-imagery with a spatial resolution of 1 km^2 by means of linear spectral mixture analysis, with emphasis on the fastly expanding sealed area class. Although the resulting AFI are not accurate enough for operational use, the approach is promising for rapid and low cost assessment of land use and land use changes. Combination with a larger scale soil map for definition of HRU is possible both through downscaling of the soil data and upscaling of the land use data.

Keywords

Spectral unmixing, area fraction image, land use, sealed area, hydrological response unit

Introduction

The rainfall-discharge relationship for a river basin is heavily influenced by the extent, location and type of impervious or sealed surfaces. In densely populated areas, sealed surfaces are increasing steadily resulting in less infiltration of rainfall, more and more rapid surface run-off to the river system and higher peak discharges.

Rainfall-discharge modeling can serve various purposes among which (i) the design of hydraulic infrastructures and (ii) early warning for flood risk. Especially for the latter application, information on the extent and type of sealed areas must be regularly updated. Hence a rapid and low-cost method for monitoring land use is required.

Models describing the rainfall-discharge relationship in a spatially semi-distributed way like the SWA-Tool, do not require the land use information to be fully spatially explicit. This feature opens up the possibility of defining HRU by combining a soil map with land use information in the form of area fraction images (AFI). Such AFI are the typical end product of sub-pixel classification of multi-spectral imagery of low (LR) or middle (MR) spatial resolution. Sub-pixel classification by means of spectral unmixing addresses the mixed pixel problem which is inherent to LR- and MR-imagery: the spectral reflectance values recorded for each pixel is related to several real world entity types present within the pixel boundaries on the ground (Figure 1).

In addition of being cheap or even free, the temporal resolution of typical LR- and MR-imagery is high, facilitating periodic updates of the land use information.

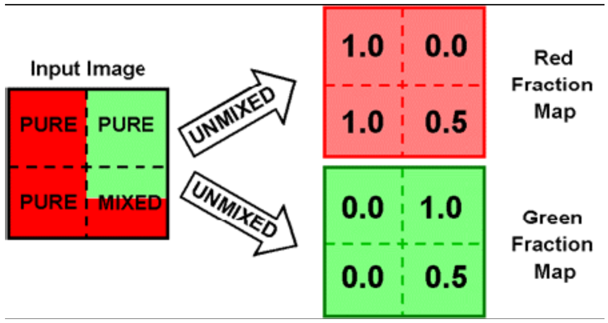

Figure 24 Illustration of mixed pixels and fraction images (Newland, 1999)

Popular techniques for sub-pixel classification of multi-spectral images are linear mixture analysis (LMA) and artificial neural networks (ANN) (Verbeiren et al., in press).

The major objective of this study was to perform and evaluate the LMA of a time series of SPOT-VEGETATION imagery as to obtain AFI for various land use classes including a sealed class. In addition we aimed at exploring how the resulting AFI can be combined with a soil map to generate HRU which can be input into SWAT.

Materials

Study area

The study area encompasses the administrative regions of Flanders and Brussels, located in northern Belgium, and covers a total area of 13.756 km^2. Land cover and land use types occur in a spatially very fragmented fashion. According to a LANDAT-derived geodataset (GIS-Vlaanderen, 2001, 15*15 m^2 spatial resolution), the total build-up area is 2.082 km^2 . It is expanding by 34 km^2/year.

Datasets

Low resolution image data

Decadal maximum NDVI-value composites (S10) of SPOT-VEGETATION-imagery (VGT) were the subject of the unmixing. A VGT-sensor is on board of both the SPOT-4 and SPOT-5-satellites and operates in 4 spectral bands: blue (0.43-0.47 µm), red (0.61-0.68 µm), near infrared (NIR; 0.78 – 0.89 µm) and short wave infrared (SWIR; 1.58-1.78 µm). Its spatial resolution is 1 km^2 and its temporal resolution is 1 day.

S10-VGT-products contain for each pixel and each of the 4 bands the spectral value of the day within the decade for which the normalized difference vegetation index (NDVI) is highest. A ten-day synthesis image is thus composed of pixels with reflectance values from different dates but, in principle, free of clouds. S10-VGT-products are corrected atmospherically and geometrically prior to dissemination.

Eight winter composites were selected. They are from the 3rd decade of December 2000 to the 1st decade of March 2001. The 5 summer images cover the 3rd decade of June 2001 to the 2nd decade of July 2001 and also comprise the first two decades of August 2001. The summer and winter VEGETATION images, as downloaded from http://free.vgt.vito.be, were clipped to the study area and projected to the standard Belgian coordinate system.

Since all the selected images have data errors in the SWIR-band (band 4) due to blind and/or aberrant SWIR detectors, the SWIR-band was removed so that 39 bands were available in the image dataset.

In order to cope with the correlation in time of the reflectance values and reduce the noise, a Minimum Noise Transformation (MNF) was performed on all image data. MNF-transformation is commonly used to determine the inherent dimensionality of image data, to segregate noise in the data, and to reduce the computational requirements for subsequent processing (Boardman and Kruse, 1994). It consists essentially of two cascaded Principal Components transformations. The first transformation, based on an estimated noise covariance matrix, decorrelates and rescales the noise in the data. This first step results in transformed data in which the noise has unit variance and no band-to-band correlations. The second step is a standard Principal Components transformation of the noise-whitened data. For the purposes of further spectral processing, the inherent dimensionality of the data is determined by examination of the final eigenvalues and the associated images. The data space can be divided into two parts: one part associated with large eigenvalues and coherent eigenimages, and a complementary part with near-unity eigenvalues and noise-dominated images. By using only the coherent portions, the noise is separated from the data, thus improving spectral processing results (Raymaekers et al., 2005). The 10 MNF-bands with the highest eigenvalues were retained for further processing.

Ground truth data

The ground truth or reference data required for calibrating the spectral unmixing procedure were extracted from two existing geodatasets: (i) a land cover/use geodataset 2001 of Flanders and Brussels (raster, 20 classes, 15 m × 15 m spatial resolution) produced by the Flemish Agency for geographic information and (ii) an agricultural parcel geodataset 2001 of Flanders and Brussels (vector, 41 classes). This dataset contains a total of 489.176 parcels, with parcel areas ranging from 0.14 ha to 660 ha. The dataset is produced by the agency implementing the nitrates directive in Flanders.

Methods

Linear mixture analysis (LMA)

Several methods are used to perform sub-pixel classification. The most prominent ones are artificial neural networks (ANN) and linear mixture analysis (LMA) (Verbeiren et al., in press). LMA was chosen for this study. The basic assumption on which LMA is based is that the spectral reflectance of a mixed, typically low resolution, pixel is equal to the linear combination of the spectra of its pure components, so-called endmembers, and that the weights in the linear combination, for each of the endmembers, are equal to the fraction of the mixel area covered by the considered endmember (Equation 1, Xiao and Moody, 2005).

$$R_k = \sum_{i=1}^{M} f_i r_{i,k} + e_k \qquad \text{(Equation 1)}$$

In equation 1, R_k is the LR-pixel reflectance for band k, M is the number of endmembers, f_i is the area fraction of endmember i, $r_{i,k}$ is the reflectance of endmember i at band k, and e_k is the residual term at band k

The purpose of spectral unmixing in general and LMA in particular is to estimate the area fraction (AF) for each endmember in each pixel. In this way an area fraction image (AFI) is created for each endmember. To this end, a system of equations is established per pixel with one equation like equation 1 for each image band and with two additional constraints (Equations 2 and 3):

$$\sum_{i=1}^{m} f_i = 1 \qquad \textit{and} \qquad 0 \le f_i \le 1 \qquad \text{(Equations 2 and 3)}$$

In order to find a least-squares solution for each pixel, the number M of unknowns (f_i) should be smaller than the number of equations (Bajjouk et al., 1998).

Determination of endmember spectra: reversed linear unmixing (RLUM)

Solving the LMA-system of equations for the area fractions of the endmembers of interest in each pixel requires that the endmember spectra are known. Endmember spectra can be derived either from pure pixels within the imagery to be unmixed, from spectral reference libraries or by reversed linear unmixing (RLUM; Raymaekers et al., 2005). The latter approach is adopted here since no spectral libraries are available for the endmembers of interest and since at a 1 km^2 spatial resolution, pure pixels are almost impossible to find in fragmented landscapes as the ones covering the study area.

If in the general system of equations for LMA, area fractions are known, endmember spectra can be derived according to Equation 4:

$$P = (F^T F)^{-1} F^T R \qquad \text{(Equation 4)}$$

In equation 4, P is the MNF transformed matrix of endmember spectra, F is the matrix of known endmember fractions and R is the maxtrix of MNF transformed input reflectance values.

The MNF transformed S10-VGT-imagery and the reference AFI were rearranged to a (n*m) matrix and a (i*j) matrix like following:

$$R = \begin{bmatrix} R_{11} & R_{12} & \cdots & R_{1m} \\ R_{21} & \ddots & \cdots & R_{2m} \\ \vdots & \cdots & \ddots & \vdots \\ R_{n1} & R_{n2} & \cdots & R_{nm} \end{bmatrix} \quad F = \begin{bmatrix} F_{11} & F_{12} & \cdots & R_{1j} \\ F_{21} & \ddots & \cdots & R_{2j} \\ \vdots & \cdots & \ddots & \vdots \\ F_{i1} & F_{i2} & \cdots & F_{ij} \end{bmatrix} \quad P = \begin{bmatrix} P_{11} & P_{12} & \cdots & P_{1m} \\ P_{21} & \ddots & \cdots & P_{2m} \\ \vdots & \cdots & \ddots & \vdots \\ P_{j1} & P_{i2} & \cdots & P_{jm} \end{bmatrix}$$

With n the number of considered pixels of the image and m the number of bands (10 in this case); i the number of pixels for which reference area fractions are available (i = n) and j the number of endmembers (j=<m).

Whereas in the matrix R each column represents an MNF transformed band value for each pixel, each column in the matrix F represents the area fractions of a given endmember in each pixel. Each row in matrix P corresponds to one endmember and contains the spectral values for each band.

The determination of the endmember spectra by means of RLUM requires reference area fraction (rAF) data to be available for at least a subset of pixels of the imagery to be unmixed. We performed the RLUM using two of such sets: (i) all pixels which are not in the boundary zone of the study area (10.443 pixels, 73% of study area) and (ii) a sample composed of the upper left pixel of each 5 * 5 window of 1 km^2 pixels (416 pixels, 3% of study area).

A set of 5 reference Area Fraction Images were composed for the entire study area from where both subsets were extracted. The considered endmembers were Sealed, Agriculture, Grass, Forest and Water.

Reference AF-data

In order to produce rAFI at a resolution of 1 km^2, a detailed reference land cover/land use geodataset was compiled by combining the GIS-Vlaanderen land cover/land use geodataset (15 m * 15 m resolution) and the vectorial agricultural parcel geodataset. The latter was converted from vector to raster at the same resolution and superimposed on the former. The result corresponds to the original land cover/use geodataset but is enhanced for the agricultural classes. Its 61 classes were aggregated into the considered 5 broader classes. The sealed class encompasses 2.029,75 km^2.

The created reference dataset is a raster with a 15 m × 15 m resolution. In order to use it for endmember determination it had to be converted to a raster geodataset with a 1 by 1 km resolution matching the VGT-imagery. This is done by computing the AF of every class in every pixel and storing the AF-values in a separate geodataset per class, the so-called reference AFI (rAFI). The rAFI for the sealed class is shown in Figure 3.

Accuracy assessment

The AFI resulting from the LMA were compared pixel by pixel with the rAFI. Coefficients of determination of the linear relationship REF = a+b*EST (not shown) and RMSE were computed. The same statistics were computed after aggregating the pixel-results at the level of municipalities.

Results and discussion

Endmembers

Endmembers were estimated by means of RLUM from all available pixels and from a systematic sample of 1 out of each 5*5 pixel window. The first option is not practical because ground truth should be collected for every pixel at each update. In that case, unmixing becomes superfluous of course. We can however assume that the more pixels are involved in the endmember estimation process, the higher the accuracy. We used the results of the first option merely as a reference.

Figure 2 shows that the endmember values (MNF-transformed) estimated based on a systematic sample of pixels are quite comparable with those estimated from all available pixels.

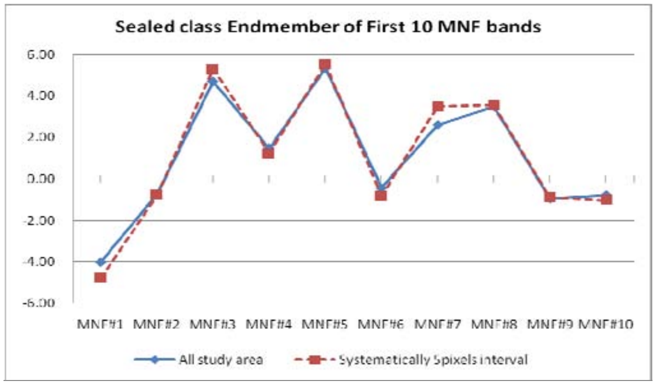

Figure 2: Endmember values estimated from all and from a sample of pixels

Area estimates and accuracy

Using both groups of end-members two unmixing exercises were performed. The results for the sealed class are presented in Figures 4 and 5 and Table 1. Figure 3 shows the rAFI for the sealed class.

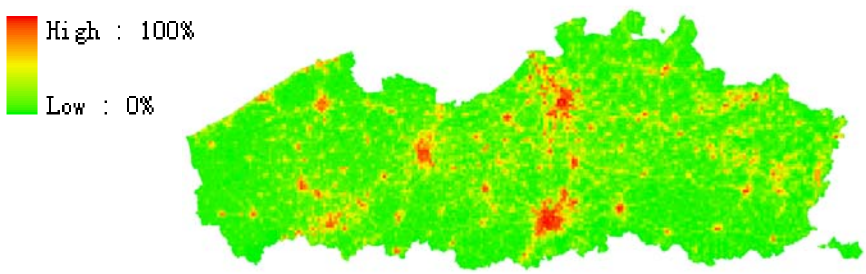

Figure 3: Reference area fraction image for the sealed class

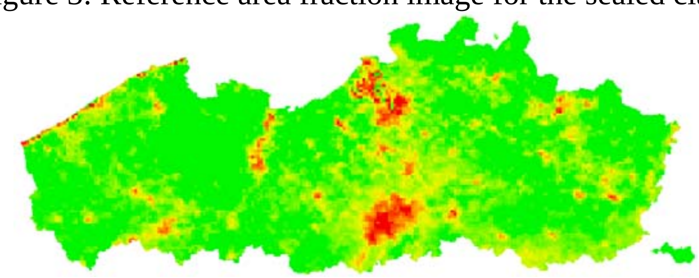

Figure 4: Estimated area fraction image for the sealed class with endmembers derived from all available pixels

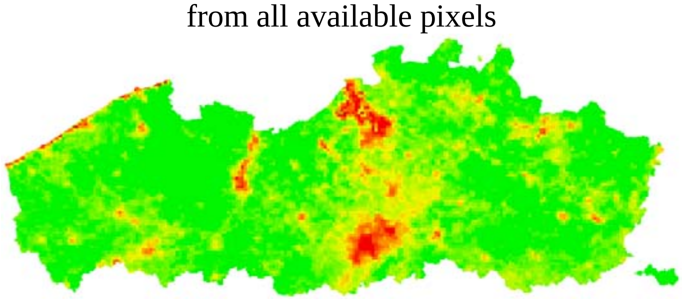

Figure 5: Estimated area fraction image for the sealed class with endmembers derived from the sample of pixels.

Table 19: Accuracy of estimated AFI for the sealed class as obtained with endmembers derived from all available pixels and from a systematic sample of pixels

End-member estimation method	RMSE			Total of Sealed Area (km^2)	Absolute Difference Estimated vs. Reference (km^2)	Relative difference Estimated vs. Reference (%)
	Per Pixel (%)	Per Municipality (km^2)	Per Municipality (%)			
From all available pixels	16.77	4.85	11.61	2020.54	3.42	0.17
From systematic 5*5 sample	16.87	4.91	11.84	2116.46	92.50	4.57

The pixel-based RMSE for the AF of the sealed class are comparable for the two approaches but high. A typical deviation between the true and estimated AF is 17%. As can be expected, RMSE drops considerably when the pixel-based results are aggregated to higher levels, i.c. the municipality. When aggregated to the complete region, the total area of sealed surfaces is very well estimated. From the lower level results it is obvious however that the spatial distribution is not well captured.

Conclusions and recommendations

Sub-pixel classification of sealed areas

The accuracy of the AFI obtained for the sealed land use class are too low for operational use. The pixel-by-pixel RMSE is high. The spatial distribution of the sealed areas is not well captured as also shown by the severe drop in RMSE after aggregation to the municipal level. Due to this lack of positional accuracy, time series, e.g. annual, of such AFI will show even more inconsistencies. Explanatory factors for the low performance may be the too coarse spatial resolution of the SPOT-VGT-sensor, the too limited spectral information used (blue, red and infrared-bands only) and the MNF-transformation possibly violating the assumption of linear mixture (Lobell and Asner, 2004). The approach as such does seem sound however and a number of possible improvements are worth examining:

- The use of middle resolution imagery, e.g. TERRA-MODIS (250 m * 250 m and 500 m * 500 m), ENVISAT-MERIS (300 m * 300 m);
- The use of more spectral bands;
- The use of one summer and one winter image only so that MNF-transformation would no longer be required;
- The use of alternative unmixing methods. For agricultural land use ANN seem to be most promising in this respect (Verbeiren et al., in press).

How to combine AFI and a soil map to generate HRU ?

AFI of sealed land use classes cannot directly be combined with a soil map to generate HRU. In the case e.g., that AFI are at a lower spatial resolution (LR) than the soil map (HR) , we see following possibilities:

- Downscale the soil map information towards the resolution of the AFI or
- Upscale the AFI to the resolution of the soil map.

A possible downscaling approach consists of attributing to each of the LR-pixels the land use class with the highest AF and combining it with the dominant soil unit within that LR-pixel.

In an upscaling approach, land use classes can be attributed to the HR-pixels within each LR-pixel. Spatial allocation to the HR-pixels can be done on a random basis with every land use class standing a probability of being attributed to a HR-sub-pixel which is in agreement with the respective AF. The random allocation can be fine-tuned taking into account assumed or extrapolated spatial dependence between neighbouring sub-pixels. The probability for attributing a land use class to a sub-pixel would then be co-determined by the probability of having adjacent pixels belonging to the same class (Verhoeye and De Wulf, 2002).

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Impact of Point Rainfall Data Uncertainties on SWAT Simulations

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Abstract In this uncertainty analysis study we investigated the impact of systematic and random point rainfall measurement errors on the simulation of discharge and nitrogen yield with the complex spatially distributed nutrient transport model SWAT. The model study was conducted in the Weisse Elster River basin which is located in the middle part of the Elbe watershed, Germany, and covers an area of about 5200 km². We randomly generated 200 rainfall time series for each of the 49 precipitation gauging stations in the Weiße Elster catchment using the Data Uncertainty Engine (DUE). The length of each time series was 12 years. The DUE can handle random errors in the measured data in a statistical framework considering a selected error probability distribution type. Systematic and random precipitation errors were investigated using mean correction values of annual precipitation of different precipitation shelter classes.

Input data uncertainty analysis showed in small catchments with only one precipitation gauge station that the selection of a wrong wind shelter class can lead to large relative discharge errors in the case of low flow conditions. The importance of detailed information on wind shelter classes of precipitation gauge sites compared to the use of mean correction factors will decline with an increase of catchment size and an associated increased number of rainfall gauge stations. Point measurement rainfall errors tend to compensate if a large number of precipitation stations is used. Systematic precipitation errors can significantly be increased by additional random rainfall errors. However the impact of random point measurement errors rapidly decreases with an increase in number of precipitation stations. This suggests that a careful consideration of random point measurement precipitation errors is necessary only if a small number of precipitation stations (< 10) are included in the study.

Keywords: nitrate, modelling, precipitation uncertainties, SWAT

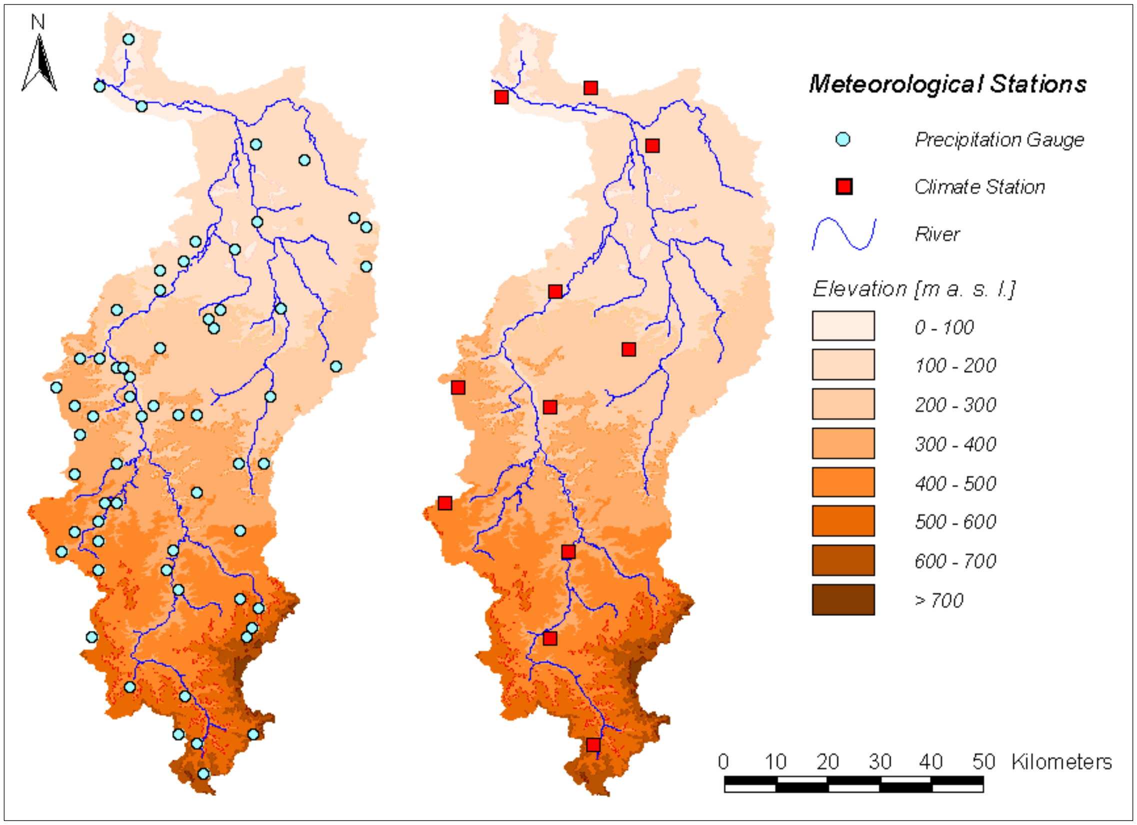
1 Introduction

Due to considerable uncertainties associated with the hydrological nutrient transport and water quality models, there is a need to quantify these uncertainties and generate a basis for the decision making process. Uncertainties in the modelling approach can be divided between input data uncertainties and model parameter uncertainties. Due to the time consuming aspect of a Bayesian approach in model uncertainty analysis, we concentrated on the following specific objectives in our case study i) investigate the impact of systematic and random point rainfall measurement errors on the simulation of discharge and nitrogen yield with the complex spatially distributed nutrient transport model SWAT, ii) analyse how these rainfall measurement uncertainties change with increasing size of a catchment and number of precipitation gauges.

2 Material and Methods

2.1 Case study

The Weisse Elster river basin is a subcatchment of the Saale basin which is the second largest tributary of the Elbe River. The catchment area is about 5200 km². The river is 250 km long and has a mean discharge of 26 m³/s (gauging station Oberthau). Land use in the basin is dominated by agricultural activities (43 % cropland, 16 % pasture),



especially in the lower part,

Figure 1: Precipitation gauging stations (left) and climate stations (right) situated in the Weisse Elster River basin

and forest (21 %), mainly in the upper part. The upper part of the basin is mountainous characterised by steep slopes and narrow valleys with hardly any floodplains and scarce groundwater resources. Precipitation varies between 500 mm in the northern part of the basin (lowlands) to 1000 mm in the southern part (mountains) with annual runoff varying approximately between 50 and 600 mm. Meteorological data are available from the German Meteorological Service. There are 49 precipitation and 11 climate stations in and around the Weisse Elster basin (Figure 1). Daily data are available for the most precipitation gauging stations, while six-hourly or hourly data are available for the climate stations. Time series have been collected from 1960 to 2004. Daily water levels measurements are available for 12 gauging stations in the Weisse Elster catchment. These stations are run by the environmental agencies of the federal states; several regional authorities run additional gauging stations. Time series of water level and discharge data have been used from 1990 to 2002. A digital elevation model at 50 m resolution is available. For the Weisse Elster basin there is a land use map derived from Landsat imagery at a spatial resolution of 30 m for 1999.

2.2 Model setup

For the water and nutrient transport modelling of the entire Weisse Elster basin, the Soil and Water Assessment Tool (SWAT) was used. SWAT is a deterministic continuous process-based model coupling hydrological, biogeochemical and ecological processes at the river basin-scale (Arnold *et al*., 1994; Krysanova & Haberlandt, 2002). According to the hydrological characteristics of the Weisse Elster River basin the whole catchment is divided into 108 subcatchments in the SWAT-Model. With regard to the spatial distribution of discharge and water quality gauging stations and the definition of water bodies for the implementation of the WFD these 108 subcatchments are combined to 12 hydrological calibration areas. A discharge gauge station is situated at each of these calibration areas. Hydrological model calibration and validation was carried out using time series from 1991 to 2000. After the determination of the sensitivity of the most important model parameters, the calibration was carried out manually. The calibration runs of the hydrological model were assessed by the visual comparison of the simulated and the observed hydrograph and objective criteria (e.g. Pearson's Product-Moment Correlation Coefficient, the coefficient of determination, the Nash & Sutcliffe coefficient of efficiency, yearly absolute volumetric error measures). They quantify the degree of agreement between the observed and simulated values.

2.3 SWAT Uncertainty analysis

Since rainfall is the most important input for any precipitation runoff modelling, precipitation data have been chosen for uncertainty analysis. Furthermore relative rainfall data errors are large compared to other meteorological data. This is also true when standardised measurement procedures are used. Systematic measurement errors are the most important source of uncertainty and there has been intensive research on this topic. The most important problem for assessing systematic errors is the lack of information on systematic measurement errors of precipitation gauges, which can be expressed in precipitation shelter classes. These shelter classes are often unknown for a specific precipitation site; although the choice of a shelter class strongly determines the correction of the measured rainfall data (see Table 1).

In this uncertainty analysis study using SWAT we analysed selected effects of point rainfall measurement errors on discharge and nitrogen loads. Specific objectives were to investigated i) the effect of the selected shelter class expressed by the mean correction factor ii) the effect of random errors according to the selected correction factor and iii) the effect of these point measurement errors with increase in catchment size and hence, the number of rain gauges stations. The SWAT model always uses one rainfall gauging station as rainfall input for each subcatchment. In the present study no spatial interpolation of point measurements was carried out. Spatial autocorrelation as well as temporal autocorrelation of point rainfall data have not been considered.

The uncertainty of precipitation data has been evaluated by adding a random error to the measured time series respectively. Measured precipitation data have a systematic bias.

This bias is site specific and depends on local wind exposure conditions. This point measurement error can be defined after to four different precipitation shelter classes according to their specific rainfall measurement correction factors after Richter (1995) (see Table 1). This systematic bias can be expressed as a mean correction value on a monthly or yearly basis. We added a randomly generated correction value to each uncorrected precipitation measurement value. The correction value is not constant and varies randomly in time. We analysed the uncertainties associated with this random variation defining a probability distribution function of the correction factor. This correction value can be defined as the relative error due to wind velocity and local disturbances of air flow. Willems (2001) set the standard deviation of the total point measurement error equal to the relative measurement error. Additional he considered errors resulting from the resolution of the rain gauge measurement. Willems (2001) assumed a normal error probability distribution. The resolution error was assumed to be small and is defined by a value of 0.1 mm.

The true precipitation value can not be smaller than the uncorrected precipitation measurement value if we assume that reading errors in the rainfall gauge measurement and the resolution error can be ignored. Therefore we defined the standard deviation of the pdf of the point measurement error as 50 % of mean measurement error, which is defined by the correction factor. The mean of the pdf of the point measurement error is defined by the

Table 1: Mean correction (%) of the average monthly and annual precipitation total (1961/90) for precipitation gauging station locations in the Weisse Elster River Basin (according to Richter, 1995)

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Year
a	31.6	33.5	26.9	18.3	12.5	10.4	10.8	10.5	12.6	15.5	21.8	26.5	18.2
b	23.3	24.5	20.3	15.1	11.2	9.8	10.0	9.5	11.5	12.7	16.8	19.8	14.6
c	17.3	17.9	15.5	12.7	10.1	8.8	9.1	8.5	10.2	11.0	13.3	15.0	12.0
d	11.5	11.8	10.7	10.0	8.6	7.7	8.0	7.5	8.7	8.8	9.5	10.3	9.3

a) free, not wind-sheltered sites, b) slightly wind-sheltered sites, c) moderate wind-sheltered sites, d) strong wind-sheltered sites

correction factor. If we assume a normal distribution of the pdf more than 97 % of the values
are larger than the uncorrected precipitation value. With respect to resolution errors less than 3 % of the values are allowed to be smaller than the uncorrected measurement value. We calculated the pdfs assuming a normal distribution and alternatively a Gumbel distribution. Since very little information of the correct distribution type is available in the literature we analysed this two distribution function types to investigate their impact on the randomly generated precipitation data. The Gumbel distribution allows the consideration of larger extreme values of the precipitation error compared to the normal distribution. The location of the Gumbel distribution is defined by the mean correction value and the scale is defined by 50 % of the mean measurement error. Using this definition of the Gumbel pdf, 99 % of the randomly generated precipitation values are larger than the uncorrected measurement values. We randomly generated 200 rainfall

time series for every precipitation gauging station in the Weiße Elster catchment using the Data Uncertainty Engine (DUE) (Brown and Heuvelink, 2007). The length of each time series is 45 years. The DUE can handle random errors in the measured data in a statistical framework considering a selected error probability distribution type. We used the mean correction values of annual precipitation of all four shelter classes a,b,c and d to investigate the systematic as well as a random precipitation error (see Table 1).

Simulation runs were carried out for the entire Weiße Elster catchment including several subcatchments with different catchment size according to the gauging stations. We investigated scaling effects of the point rainfall measurement uncertainties based on the analysis of simulated discharge and nitrogen load with the increase in catchment size and associated number of rainfall gauge stations. In the Weiße Elster catchment the mean density of rainfall gauging stations is one per 85 km².

3 Results

Uncertainties in rainfall data and their impact on discharge and nitrogen loads were analysed with respect to systematic errors due to different types of precipitation shelter classes and due to random rainfall errors. Special focus has been put on the impact of varying catchment size and station numbers on simulated discharge and nitrogen load. Systematic errors have been considered using correction factors ranging between 9.3 and 18.2 %. The relative differences of mean values between the selected correction factors vary in relation to total discharge (Fig. 2).

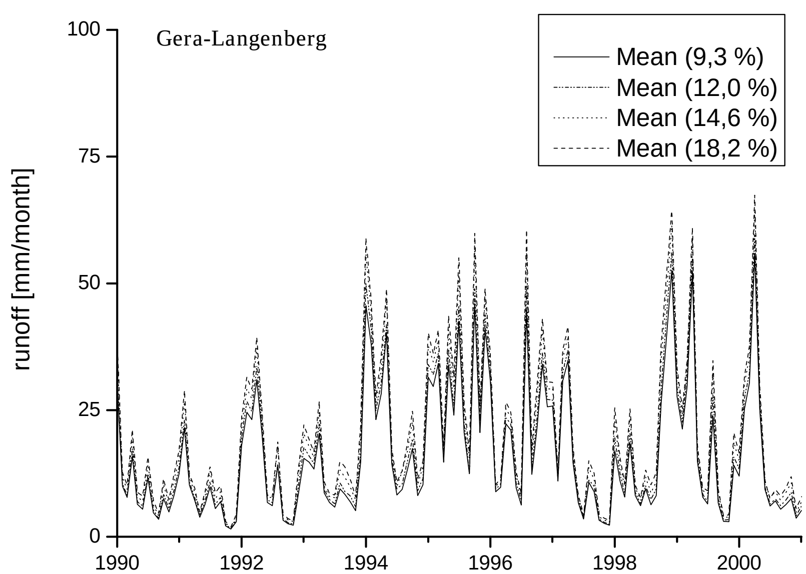


Figure 2: Mean simulated discharge using four different correction factors from 9.3 to 18.2 % based on 200 randomly (Gumbel distribution) generated rainfall time series for the time period from 1990 to 2001 of discharge gauging station Gera-Langenberg

Simulated discharge at gauging station Gera-Langenberg with a catchment size of 2146 km² showed considerable lower differences between the selected correction factors especially during low flow conditions compared with small catchments. The effect of randomly generated rainfall measurement errors on catchment discharge and total nitrogen load with respect to the catchment size is shown in Figure 3. Simulated errors are expressed as mean and maximum monthly error ranges in selected subcatchments of the Weiße Elster using a rainfall correction factor of 18.2 %. In this simulation study a Gumbel probability distribution function for the generation of random rainfall measurement errors was used. The maximum error range of discharge can exceed 46 mm per month in the smallest catchment with only one precipitation station. This value is high compared to maximum simulated monthly discharge of 92 mm. For the same catchment the mean of the error range of monthly discharge values is lower with 9.5 mm.

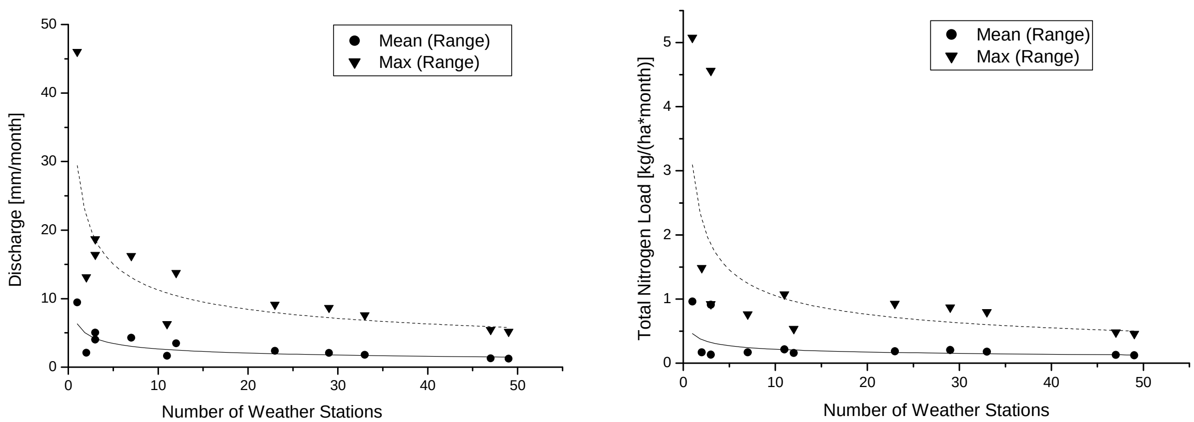


Figure 3: Mean and maximum monthly error ranges of simulated discharge and nitrogen load in selcted subcatchments of the Weiße Elster using a rainfall correction factor of 18.2 % and randomly generated rainfall time series (Gumbel distribution)

With increasing catchment size the effect of random measurement errors on simulated discharge error ranges decreases rapidly. If more than 10 precipitation stations are included in the simulations study, which relates in our study to a catchment size of 1255 km², error range of monthly values are smaller than 4 mm.

In our investigation we did not explore the spatial representatives of a single precipitation gauge station of the given rain gauge network. This could be investigated if a much denser precipitation network would have been available, at least in a part of the Weiße Elster catchment (compare Willems, 2001, Refsgaard et al. 2006). If spatial correlation length scale is larger or in the same order of magnitude as the distance between stations, we are underestimating the precipitation uncertainty. Since we only considered point measurement errors the computed values express the minimum errors due to precipitation.

The results of simulated nitrogen load are comparable to the discharge values. Significant error ranges due to random rainfall errors could only be computed in the case of small catchments and corresponding small number of precipitation stations. It has to be kept in mind that we always investigated an increase of precipitation gauge stations in connection with an associated increase in catchment size. If more than three precipitation stations are included in the simulation study, no considerable change of the error range of nitrogen loads could be observed. In these cases error ranges are lesser than 0.22 kg N /(ha*month) which is low compared to simulated total nitrogen loads. These nitrogen loads can exceed 4.3 kg N/(ha*month) in mesoscale catchments (gauging station Gera-Langenberg, 2146 km²). However, these small absolute error ranges of nitrogen loads may not only be explained with an insignificant influence of random precipitation error. Also the amount of mean area weighted precipitation and discharge decreases with increasing catchment size because of subsequent inclusion of the lowland areas with comparably low precipitation. The decrease of area weighted discharge with increasing catchment size leads also to a decrease of area weighted nitrogen load and hence to a decrease of absolute error ranges.

4 Conclusions

With regard to the SWAT modelling study we can conclude that systematic rainfall measurement errors can have considerable impact on simulated discharge as well as simulated nitrogen load. In small catchments with only one precipitation gauge station the selection of a wrong wind shelter class can lead to large relative discharge errors in the case of low flow conditions. The importance of detailed information on wind shelter classes of precipitation gauge sites compared to the use of mean correction factors will decline with an increase in catchment size and associated increase of rainfall gauge stations. Point measurement rainfall errors tend to compensate if a large number of precipitation stations is used.

We investigated these uncertainties by the use of mean yearly correction factors. The results of Richter (1995) show that monthly correction factors can be much larger especially in winter time in connection with snow induced errors. Furthermore the effect of precipitation errors will increase if daily discharge and nitrogen loads are calculated instead of monthly values in the present study. This may be important if nitrogen concentrations are of interest (water quality standards are often based on 90 percentiles of matter concentrations). Systematic precipitation errors can significantly be increased by additional random rainfall errors. However the impact of random point measurement errors rapidly decreases with an increase in number of precipitation stations. This suggests that a careful consideration of random point measurement precipitation errors is necessary only if a small number of precipitation stations (< 10) are included in the study. It has to be taken into account that we did not investigate the effect of additional spatial and temporal autocorrelation on precipitation uncertainties. Including these additional uncertainties would possibly increase total rainfall uncertainties.

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Consideration of Measurement Uncertainty in the Evaluation of Goodness-of-Fit in Hydrologic and Water Quality Modeling

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Abstract

As hydrologic and water quality (H/WQ) models are increasingly used to guide water resource policy, management, and regulation, it is no longer appropriate to disregard uncertainty in model calibration, validation, and evaluation. In the present research, the method of calculating the error term in pairwise comparisons of measured and predicted values was modified to consider measurement uncertainty with the goal of facilitating enhanced evaluation of H/WQ models. The basis of this method was the theory that H/WQ models should not be evaluated against the values of measured data, which are uncertain, but against the inherent measurement uncertainty. Specifically, the deviation calculations of several goodness-of-fit indicators were modified based on the uncertainty boundaries (Modification 1) or the probability distribution of measured data (Modification 2). These modifications require estimation of measurement uncertainty with a method such as described in Harmel et al. (2006). The choice between these two modifications is based on absence or presence of distributional information on measurement uncertainty. Modification 1, which is appropriate in the absence of distributional information, minimizes the calculated deviations and thus produced substantial improvements in goodness-of-fit indicators for each example data set. Modification 2, which provides a more realistic uncertainty estimate but requires distributional information on uncertainty, resulted in smaller improvements. Modification 2 produced small goodness-of-fit improvement for measured data with little uncertainty but produced modest improvement when data with substantial uncertainty were compared with both poor and good model predictions. This limited improvement is important because poor model goodness-of-fit, especially due to model structure deficiencies, should not appear satisfactory simply by including measurement uncertainty. A full description of these modifications and results of their application to example data sets appears in Harmel and Smith (2007).

KEYWORDS: Model calibration, validation; Statistics; Nash-Sutcliffe; Index of agreement

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Sensitivity analysis of sediment processes with SWAT

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Abstract

This paper presents a computational analysis for evaluating critical nonpoint source sediment processes and management actions at the watershed scale. In the analysis, model parameters that bear key uncertainties are presumed to reflect the importance of natural processes and/or management actions that they represent. A hybrid of multivariate sensitivity analysis techniques was integrated with the Soil and Water Assessment Tool (SWAT) to investigate correlation structure in the parameter space. The computational analysis was applied to the Dreisbach watershed in Indiana in the Midwestern portion of the United States. Results showed that incorporation of parameter interactions is essential to obtaining conclusive information about critical system processes and management actions. Interactions between surface runoff volume and within-channel processes were critical to describe transport of sediments in the study watershed. The sensitivity analysis reported herein could be used to derive a list of key nonpoint source best management practices for development of watershed management plans.

KEYWORDS: SWAT, Water quality, nonpoint source pollution, best management practices, uncertainty analysis, modeling.

Introduction

Identification of natural processes and management actions that control nonpoint source pollution is essential for development of watershed management plans. The current handbook of the Environmental Protection Agency for developing watershed plans (US EPA, 2005) recommends implementation of management practices in portions of the watershed that are believed to contribute intensively to nonpoint source pollution. Locating critical areas, however, is complicated because contaminants are carried along with the flow, and water movement over a watershed tends to be fairly dynamic, behaving in a nonlinear fashion. Potential interactions among hydrologic, fluvial, and nutrient processes and nonpoint source pollution control measures should be identified and included in designing watershed management scenarios. Such information could be obtained by integration of computational techniques with hydrologic/water quality models.

Inverse modeling approaches based on sensitivity analysis have been used in the past to obtain information about important system processes in a variety of disciplines. Osidele et al. (2003) utilized a sensitivity analysis to investigate the importance of sediment and nutrient processes in a section of the Chattahoochee River (nearly 115 river miles) in Atlanta. The univariate regionalized sensitivity analysis (RSA; Spear et al., 1980) was used in conjunction with the multivariate tree structured density estimation (TSDE; Spear et al., 1994) to account for the correlation structure in the parameter space. The regionalized sensitivity analysis was also utilized by Zheng and Keller (2006) for sensitivity analysis of the Watershed Analysis Risk Management Framework (WARMF) model and its management implications. The one-at-a time sensitivity analysis (Pitman, 1994), factorial design, and the Fourier amplitude sensitivity test (FAST) (Saltelli et al., 2000) are some other sensitivity analysis methods. These sensitivity analysis methods determine parameter sensitivities for a single model response (e.g. average annual sediment yield).

In this study, we investigate the utility of computational approaches for identifying critical NPS processes and key management actions that are likely to control fate and transport of sediments in watershed systems. To this end, the RSA and TSDE procedures are linked with a comprehensive watershed model (Soil and Water Assessment Tool) to (i) develop a sensitivity analysis framework that can handle interactions in the parameter space as well as multiple trajectories of model outputs, and (ii) demonstrate the application of the computational analysis to highlight critical sediment processes in an agricultural watershed in Indiana. The developed framework provides a novel capability for dealing with important issues for holistic watershed management. Critical nonpoint source sediment processes and their potential interactions can be evaluated. Watershed management programs such as the total maximum daily load (TMDL) and the USDA’s environmental quality incentive program (EQIP) would significantly benefit from such analysis. Another feature of the present work is that real system response data are utilized in the analysis.

Theoretical Background

Regionalized Sensitivity Analysis (MOGSA)

Regionalized sensitivity analysis (Spear and Hornberger, 1980) aims at identification of critical uncertainties in order to evaluate the relative importance of individual parameters that exert the most influence on system behavior. The procedure utilizes a uniform sampling of the parameter space and involves (i) a qualitative definition of the behavior of the system under study, and (ii) a binary classification of the parameter space (*S*) into good (behavior B) or bad (nonbehavior $\bar{B}$) regions. The strength of the method lies in the classification scheme, which facilitates the application of multivariate statistical methods to explore the level of significance that the posterior probability distribution function of each element of the parameter vector $\vec{\alpha}$ in the behavior region $f_m(\vec{\alpha} \mid B)$ deviates from the one in the nonbehavior region $f_n(\vec{\alpha} \mid \bar{B})$. A Kolmogorov-Smirnov two sample test is performed to test the hypothesis that for a given parameter $\alpha_k \in \vec{\alpha}$, $f_m(\alpha_k \mid B)$ differs from $f_n(\alpha_k \mid \overline{B})$. The Kolmogorov-Smirnov statistic ($d_{m,n}$) can be used to determine the relative importance of the uncertainty associated with each element of the parameter vector, with higher values indicating higher influence on model outputs.

Tree Structured Density Estimation (TSDE)

Spear et al. (1994) recognized that despite the conceptual simplicity of the regionalized sensitivity analysis, its applicability may be limited as a result of different correlation structures among parameters. The tree structured density estimation (TSDE) methodology was developed to obtain information relevant to the interactions between model parameters in the complex non-uniform behavior region. In the TSDE procedure, the *m* behavior parameter vectors are treated as independent samples from an unknown probability distribution function. To construct an adequate approximation of this unknown distribution, the behavior parameter space is partitioned into *q* sub-spaces with relatively similar densities. The relative density of a subspace (or a node) is defined as the fraction of behavior parameter vectors in the subspace divided by its volume. The splitting process is performed successively, and begins with splitting the parameter space into two sub-spaces. The parameter space is split on the axis of the parameter that produces the largest increase in the accuracy criterion (i.e., a loss functions). Likewise, in the second recursion, each of the two sub-spaces constructed in the first step is split to two new sub-spaces. This procedure is repeated until either the accuracy of density estimate does not increase significantly or the density of each sub-space is less than some critical value. The TSDE procedure finally yields a tree structure that reveals the multivariate correlations among model parameters. The top (origin) node in the tree represents the original sample space with relative density of 1. The density of end (terminal) nodes indicates the relative importance of the corresponding intermediate parameters and their interactions for matching the behavior of the system under study. Beginning from the origin node and ending with a terminal node, each branch graphically depicts the interactions between model parameters. In a TSDE diagram, terminal nodes are further grouped into high density terminal nodes or low density terminal nodes. The sequence of nodes and branches that ends with a high density terminal node reveals the important or sensitive parameters. The closer a parameter is to the origin in the sequence, the more important is its influence on the objective function.

Methods

A computational procedure (Fig. 1) was developed aiming at identifying critical processes that control the behavior of a given system. In this framework, model parameters serve as surrogates for internal system processes and external controls that they represent in the model's mathematical structure. Defining the behavior of natural systems typically involves multiple criteria that are likely to interact in a nonlinear fashion. Extracting credible inferences pertaining critical processes in such systems necessitates the incorporation of interactions in the parameter space. In the present work, the RSA and TSDE methods are applied in conjunction to rank model input parameters according to the importance of their uncertainties for defining the system behavior. While the main effects of input parameters, independent of others are sought using the RSA results, the TSDE reveals multivariate interactions. The system behavior is defined based on historical observations or a desired future state of the system. Implementation of the uncertainty analysis framework in Figure 1 is demonstrated through application to sediment nonpoint source processes in a small watershed in Indiana, USA.

Case Study: Dreisbach Watershed, Indiana

The application of the methods used in this study is demonstrated in the Dreisbach watershed in the Maumee River basin that drains into Lake Erie in the Midwestern portion of the United States. Figure 2 depicts the location and 1975 land use maps of this primarily agricultural watershed. The majority of the soils in the watershed fall in the type C hydrologic group with low to moderate infiltration characteristic. The Dreisbach watershed was subject to an extensive water quality monitoring program during 1974 through 1978. Daily precipitation, temperature, streamflow, sediment, and nutrient data were collected at the outlet of the watershed. A detailed description of the available data for the watershed can be obtained from Arabi et al. (2004).

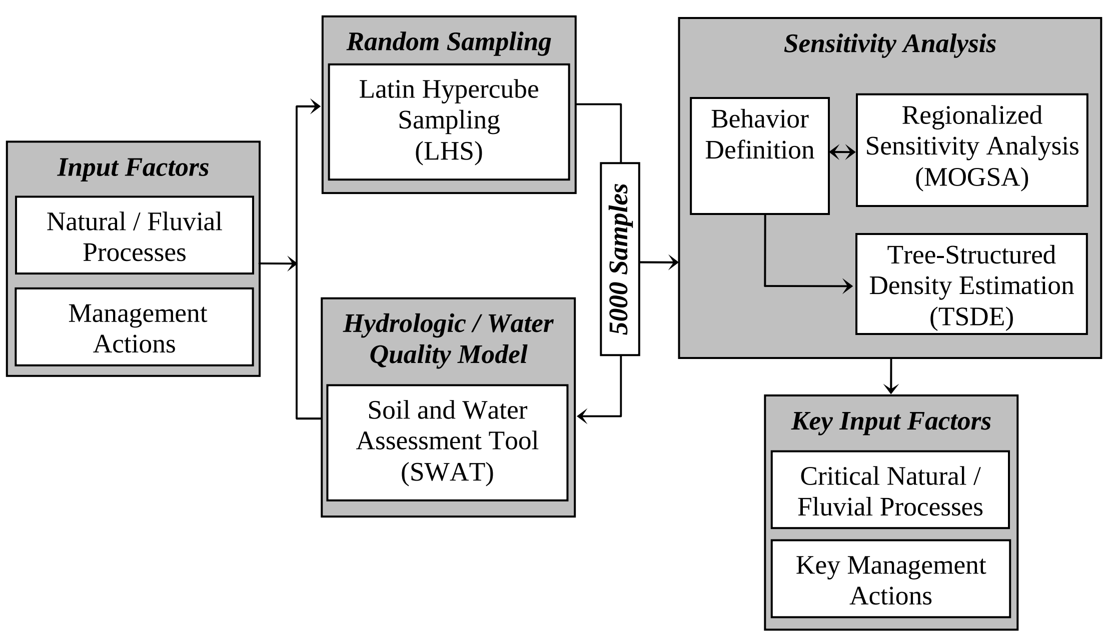

Figure 1. Computational framework for sensitivity analysis of sediment processes.

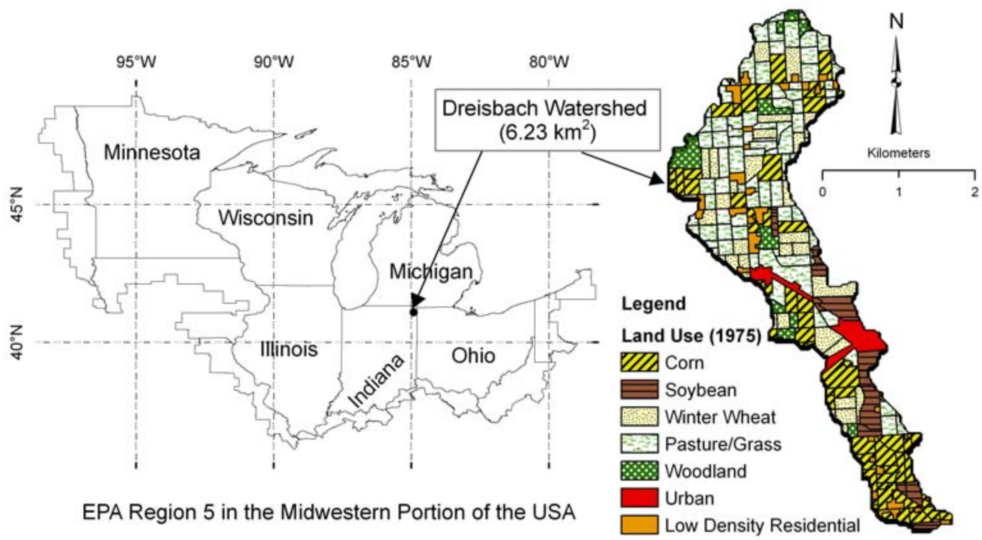

Figure 2. Location and 1975 land use maps for the Dreisbach watershed.

Choice of Watershed Model: Soil and water Assessment Tool (SWAT)

The framework outlined above is fairly general. In this paper, it was applied to identify nonpoint source processes and management actions that are likely to control transport of sediments at the outlet of a small agricultural watershed in the state of Indiana in the US. The Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998, Arnold and Fohrer, 2005) was chosen as the watershed model. SWAT is a process-based simulation model, operating on a daily time step. The model was originally developed to quantify the impact of land management practices in large, complex watersheds with varying soils, land use, and management conditions over long periods of time. SWAT uses readily available inputs and has the capability of routing runoff and chemicals through streams and reservoirs, and allows for addition of flows and inclusion of measured data from point sources. Moreover, SWAT has the capability to evaluate the relative effects of different management scenarios on water quality, sediment, and agricultural chemical yields in large, ungaged basins. Major components of the model include weather, surface runoff, return flow, percolation, evapotranspiration, transmission losses, pond and reservoir storage, crop growth and irrigation, groundwater flow, reach routing, nutrient and pesticide loads, and water transfer.

Application to nonpoint source sediment and nitrogen processes

The computational procedure was implemented as follows. First, the natural process and/or management action represented by each input parameter of the SWAT model was identified. Model parameters, their suggested ranges, and the processes they represent were obtained from Arabi et al. (2006). These parameters can be categorized in two classes: (1) parameters that represent internal natural processes or external management controls such as curve number that represents surface runoff, and (2) parameters that represent external management actions such as USLE practice factor. USLE practice factor indicates the importance of upland farming practices such as implementation of parallel terraces (Renard et al., 1997).

Next, 5000 parameter vectors were randomly generated with a Latin Hypercube Sampling (LHS; McKay et al., 1979) strategy. The SWAT model was used to simulate monthly flow, sediment and nutrient outputs corresponding to each parameter vector. Model inputs and outputs were integrated with the RSA and the TSDE methods to investigate significance of input factors. In the present work, we adopted the Nash-Sutcliffe coefficient (Nash and Sutcliffe, 1970) for behavior/non-behavior classifications. A parameter set was classified as behavior giving if the Nash-Sutcliffe coefficient computed based on observed and simulated monthly sediment loads was larger than zero.

Results

Results of the RSA analysis in Table 1 showed that curve number (*CN*) was the top rank for sediment yield, followed by the linear coefficient for channel transport capacity estimation *SPCON*, channel erodibility factor *CH_EROD*, and channel Manning's number *CH_N2*. In the analysis, 360 out of 5000 parameter sets comprised the behavior set.

Table 1. Regionalized sensitivity analysis ranking of input factors

Symbol	*Description*	*dm,n*
CN_f	Fraction change in SCS runoff curve number (%)	0.40
SPCON	Linear coefficient for sediment routing in the channel network	0.23
CH_EROD	Channel erodibility factor	0.16
CH_N2	Manning's "n" value for the main channel	0.14
SOL_AWC	Available soil water capacity (m/m)	0.11
USLE_P	USLE equation support practice factor	0.10

Main channel sediment processes within the SWAT model include channel deposition and channel erosion (degradation). Whether a channel segment is undergoing erosion or deposition is determined by comparing its estimated transport capacity with sediment load in the streamflow. The transport capacity of channel segments is estimated as a function of peak flow velocity. Manning's equation is employed to estimate flow velocity in the channel network. When the estimated transport capacity is larger than the existing sediment load, in the channel segment is assumed to undergo erosion that is estimated as a linear function of channel erodibility factor.

The TSDE procedure complemented the RSA results accounting for multivariable interactions in the parameter space. Two high density terminal nodes, S_8 and S_{11}, were identified as depicted in Figure 3. Terminal nodes were selected such that they contained collectively at least 75% of behavior giving parameter vectors. Both high density terminal nodes were defined by curve number and parameters that represent fluvial processes within the channel network {CN_f; *SPCON*; *CH_N2*}. All of these input factors appeared in the RSA results. However, the TSDE results led to important inferences about the control processes and management actions in the Dreisbach watershed.

The TSDE diagram in Figure 3 reveals the critical role of interactions between runoff volume and channel processes. Terminals nodes S_4 and S_5 defined by only curve number (CN_f) contained only about 10% of behavior parameter space. This indicated that surface runoff volume alone was not adequate to describe the system behavior. High density terminal nodes S_8 and S_{11} indicated that the interactions among surface runoff volume and channel processes controlled the system behavior for sediment yield. Thus, management actions that decrease the transport capacity and erodibility of channel segments were identified as key actions for preventing or minimizing sediment transport to the outlet. Examples of these management options include grassed waterways and grade stabilization structures. Implementation of grassed waterways will decrease erodibility of the channel network and will increase flow resistance in the channel segments (Bracmort et al., 2006). Grade stabilization structures will minimize channel erodibility and will reduce transport capacity of the channel segment by reducing bed slope, thus minimizing bed/bank erosion and sediment transport in the channel network.

Implications for calibration of watershed models

The computational analysis in Figure 1 could be employed to obtain information relevant to posterior distribution of process parameters. The behavior parameter set obtained from the sensitivity analysis can be used to estimate the posterior probability distribution function of the parameters with critical uncertainties. As an example, Figure 4 shows the posterior distribution of the channel Manning's number *CH_N2* constructed from the behavior parameter set. It is evident that the predefined uniform distribution in the [0.0,0.3] range for *CH_N2* did not adequately describe its distribution. The distribution is clearly skewed to the left and peaks at 0.04-0.05 that is in concurrence with

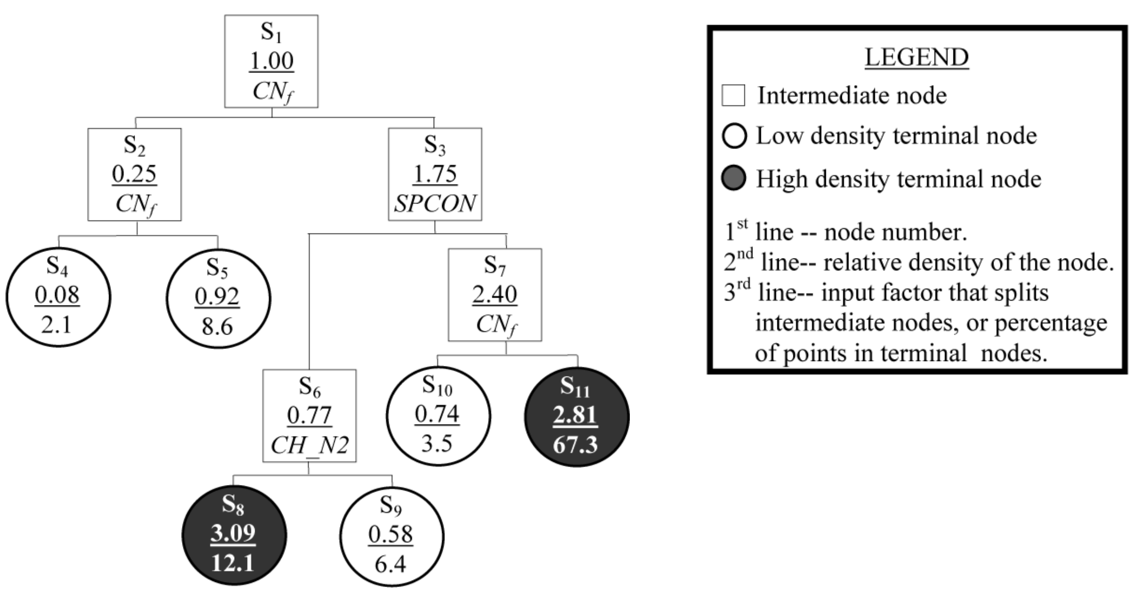

Figure 3. TSDE tree diagram.

recommended values for natural streams (Chow, 1959). Thus posterior parameter distributions from the multivariate sensitivity analysis can improve parameterization of watershed models.

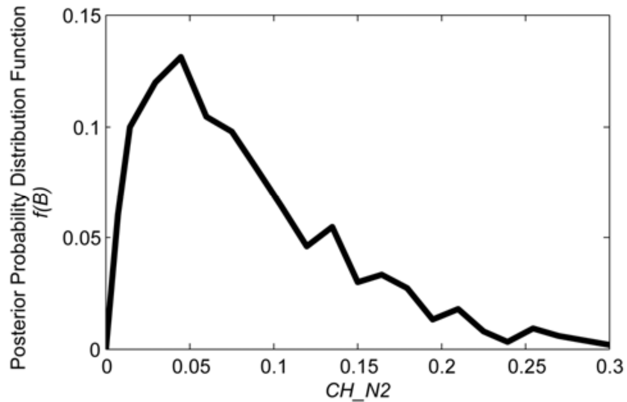

Figure 4. Posterior density function of *CH_N2* -- channel Manning's coefficient.

Conclusions

A framework has been presented to obtain information about critical NPS sediment processes at the watershed scale. In this framework, joint multivariate and multiobjective approaches point to natural processes, management actions and interactions thereof at the watershed scale. The computational analysis can be utilized to abridge the long list of management practices recommended for sediment and nutrient control. Inferences can be drawn relevant to the practices likely to achieve desired water quality standards at the watershed scale.

The application of the methodology was demonstrated in a case study watershed in Indiana. Results indicated the dominance of interactions among surface runoff volume and channel processes in controlling fate and transport of sediments. It became evident that within-channel management practices would significantly influence sediment loads at the outlet of the watershed. Although the use of these results may be limited to watersheds with similar characteristics as the Dreisbach watershed, the developed methodology can be used in other watershed systems.

We also demonstrated the utility of the computational analysis for obtaining information pertinent to the posterior distribution function of model parameters. Better estimates of parameter distribution functions would assist faster and more accurate calibration of the model.

There is often a temptation to develop more complex simulation models, with increasing number of input and output parameters, to address a variety of water quality problems at the watershed scale. Increasing number of processes parameters combined with inclusion of more model outputs in the evaluation of model performance will likely exacerbate problems of parameter identifiability. The computational analysis we have employed in this study holds a great promise for assessing the trade-off between improving the performance of watershed models and parameter identifiability.

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Integrated Modeling of Surface Water and Groundwater by Using Combined SWAT-MODFLOW

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Abstract
It is essential for the integrated watershed hydrologic model to be able to examine the hydrologic effects and at the same time, allowing hydraulic interaction between surface water and groundwater. To compute the quantity of groundwater recharge and discharge determined by runoff process from the watershed, SWAT model and MODFLOW model were fully combined together. Since the SWAT model has semi distributed features, it is difficult to represent groundwater recharge, head distribution and pumping effect etc. To solve these problems, the method of exchanging the characteristics of the hydrologic response units (HRUs) in SWAT with cells in MODFLOW by fully coupled manner is proposed. The linkage is completed by considering the interaction between the stream network and the aquifer to reflect boundary flow and enhancement of water transfer module in SWAT. This approach is applied to Gyungancheon basin in Korea and it demonstrates the combined model enables interaction between saturated zone and channel reaches, estimation of distributed groundwater recharge and head, which play an essential role in the generation in the Gyungancheon basin. The comprehensive results show that wide applicability of model which represents the various features of surface water and groundwater simultaneously.

KEYWORDS: SWAT, MODFLOW, Groundwater recharge, River-Aquifer Interaction.

Introduction
Up until now, hydrologic component analysis in Korea has concentrated on surface water management, so problems related to groundwater were not dealt with in a rigorous manner. Additionally, the groundwater model was not adequately linked to surface water analysis, and thus the main focus was primarily on aquifer management. For instance, groundwater recharge could not be considered in terms of hydrological processes, which are directly related to precipitation, evapotranspiration and surface runoff. Groundwater recharge rate was an input to the groundwater model and thus has been determined from trial and error during calibration. The best solution for solving this problem is the construction of a long term rainfall runoff model, which can effectively produce an integrated analysis for both the groundwater and surface water. The main factors to consider for these kinds of models are: the land use, surface runoff, and other factors such as climate change. It is essential for the model to be able to examine the hydrologic effects and at the same time, allowing hydraulic interaction between surface water and groundwater. To compute the quantity of groundwater runoff determined by

runoff analysis from the watershed, SWAT(Arnold et al., 1993; Arnold, 1998) model and MODFLOW (McDonald and Harbaugh, 1988) model were fully combined(Kim et al. 2004a, b; Kim et al. 2006). The SWAT model is widely used for long-term runoff and water quality simulations. It was originally developed from the CREAMS(Knisel, 1980) and SWRRB (Williams et al., 1985) models with channel routing and groundwater components added for larger watersheds. Although, SWAT has its own module for groundwater components, the model itself is semi-distributed and thus distributed parameters such as hydraulic conductivity distribution could not be represented. Moreover it causes difficulties in expressing the spatial distribution of groundwater levels. One of the most essential components of an efficient groundwater model is the accuracy of recharge rates amongst the input data. The conventional groundwater flow analysis performed by MODFLOW often overlooks the accuracy of the recharge rates that are required input to the model. Consequently, there is considerable uncertainty in the simulated runoff results. To overcome these disadvantages, we developed subroutines which exchanges flow data between the cells in MODFLOW and the HRUs (hydrologic response units) of SWAT. HRUs are defined by overlaying soil and landuse and lumping similar soil/land use combinations. On the basis of these modifications, the groundwater model in SWAT was replaced with MODFLOW. Therefore, it was possible to establish a fully combined modeling program which is able to form a linkage in each time step. Sophocleous et al.(1997, 1999) already presented the development and implementation of a computer model SWATMOD which is capable of simulating the flow of surface-water by SWAT, ground-water and stream aquifer interactions by MODFLOW on a continuous basis for the Rattlesnake Creek basin in south-central Kansas. In this study, the integrated SWAT-MODFLOW model is described and demonstrated. SWAT-MODFLOW is quite different to SWATMOD which involves separate modeling by sequential linkages. Additionally, results obtained from the initial test in Gyungancheon basin are analyzed.

Development of combined SWAT-MODFLOW

The SWAT model is particularly limited in terms of dealing with groundwater flow, due to its lumped nature. Conversely, MODFLOW has difficulty in computing the distributed groundwater recharge, which is a major input data for groundwater modeling. Therefore, by sustaining the advantages of the two models, it is possible for the hydrological components to be quantified reasonably. If HRU based groundwater recharge is used for input data of MODFLOW, and the groundwater flow between aquifer and stream is computed and exchanged to SWAT, then the spatial-temporal characteristics in watershed will be reflected properly.

Main features of combined SWAT-MODFLOW

SWAT and MODFLOW are divided into two components: the input and computation components. The purpose of this division is to include MODFLOW into the groundwater module of SWAT. This process is shown in Fig. 1. For this purpose, SWAT is divided into two modules before and after the subroutine 'simulate' shown as S-1 and S-2 in Fig. 1. MODFLOW is divided into two parts before and after the 'Stress loop' shown as M-1 and M-2 in Fig.1. Subroutine 'gwmod' is associated with groundwater flow computed based recharge from each HRU in SWAT. As MODFLOW does not have any division of subbasins or HRU, an alternative method is needed to use HRU based groundwater recharge in SWAT as MODFLOW's input. Therefore, SWAT is split before and after the 'gwmod' subroutine (Fig.2). As 'gwmod' subroutine is called by HRU for each time step (one day), gwmod is not easily disassembled into two parts such as input and computation parts. We reconstructed the original 'gwmod' so the variables calculated before calling the gwmod subroutine(S-2-1 in Fig. 2) could be used after calling the gwmod subroutine(S-2-2 in Fig. 2). This modification makes the exchange of variables possible.

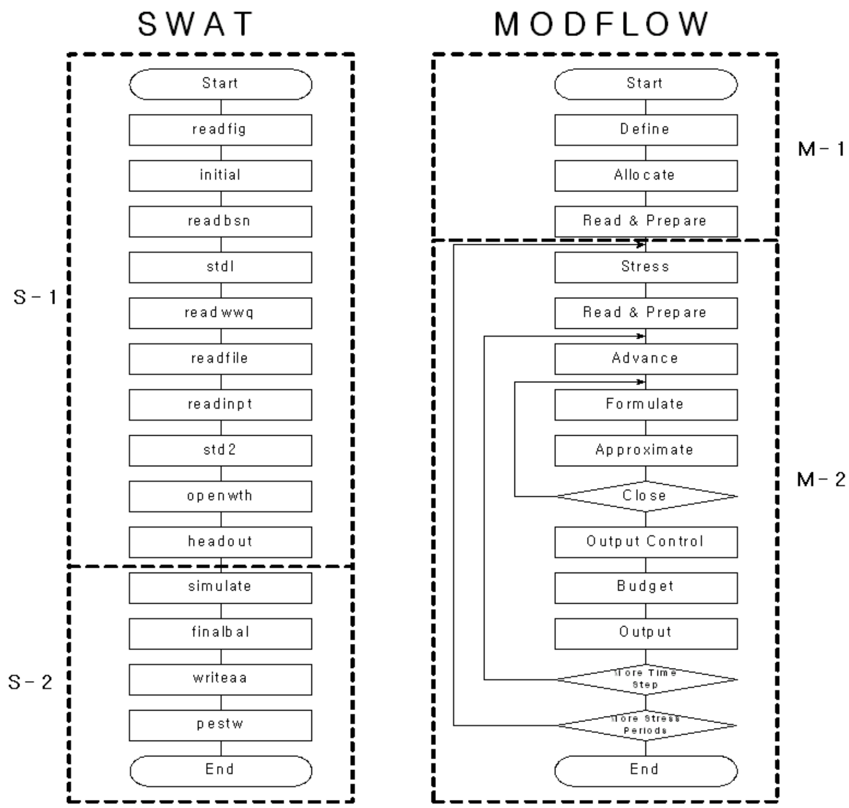


Fig. 1. Division of SWAT and MODFLOW

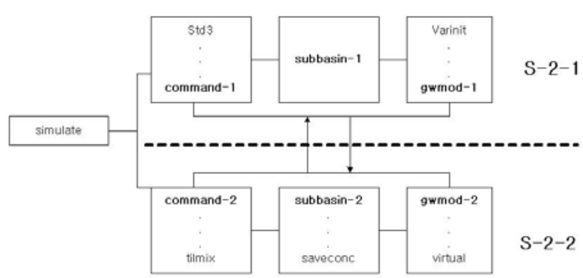


Fig. 2. Specific division of SWAT model

The SWAT-MODFLOW's main program is simply a modified version of SWAT's main program as shown in Fig.3. The SWAT-MODFLOW starts by initiating the S-1 part from SWAT, and reads the input data which is required to initiate the SWAT model. This allows us to select between the two groundwater models, SWAT and MODFLOW. If MODFLOW is initiated, then M-1 is implemented and data for MODFLOW is used. Main computation of SWAT is implemented in the 'simulate' subprogram, and different subroutines are implemented according to the use of SWAT or

MODFLOW. On one hand, if MODFLOW is not used, then SWAT's groundwater module is implemented. On the other hand, if MODFLOW is used, S-2-1 is implemented. When S-2-1 is implemented, the groundwater recharge by cell from HRU and river stage by cell is used for input of M-2 in MODFLOW. After implementing M-2, the outputs by cells are added by HRU and channel, which is then sent to S-2-2. In the combined model, the recharge rate by cell and river stage is used as input data for MODFLOW from SWAT. Whereas, input data for SWAT and MODFLOW implementations are HRU averaged recharge rates. These rates are determined from the cell based recharge rate and flow rate between the river and aquifer. These processes are necessary for the S-2-1 → M-2 → S-2-2 stages in Fig.3. Cell based recharge rate and river stage are computed when S-2-1 to M-2 are processed. The HRU based recharge rate and flow rate between the river and the aquifer are computed through M-2 → S-2-2.

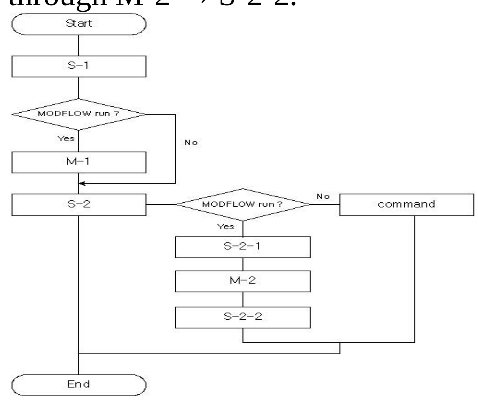


Fig. 3. Flow chart of SWAT-MODFLOW

Exchange of variables in both models

In SWAT, groundwater recharge rates by HRU within a subbasin do not contain spatial information. The HRU spatial distribution map is made by overlaying the soil map and the land use map to distribute the recharge values by HRU to cells in MODFLOW. For this purpose, HRU number is used as zoning number in Basic package of MODFLOW(*.bas file). HRU distribution maps are used to assign the recharge values of HRU from S-2-1 to each cell as the cell's averaged area values. All variables associated with groundwater, which are needed for each time step, are computed during the M-2 process. The inverse process of the previous section is needed to return the values to S-2-2 process. Thus, it is necessary for the groundwater quantity by cell to be changed into groundwater quantity by HRU. By using the HRU distribution map, cell based groundwater quantity is computed to determine the groundwater quantity by HRU. In SWAT, if the depth of a shallow aquifer increases above the user defined threshold value, it is assumed that groundwater runoff is occurring. Conversely, the groundwater flow rate from river to aquifer cannot be computed if the depth of the shallow aquifer is lower than the threshold value or if flow occurs in the channel. This problem is caused by SWAT's inability to use river bed elevation and aquifer depth values. However, MODFLOW is capable of dealing with river-aquifer flow exchange by comparing river stage and groundwater level which are computed by using river bed elevation and aquifer depth. The major input data used in the experiment conducted in the MODFLOW river package were: row and column of cells for river, river stage, conductance of river bed and, river bed elevation. Among these variables, river stage and conductance of river bed are read directly from SWAT and they can be modified by the user. The river bed elevation and river stage of previous day is computed by SWAT. Conductance of river bed is computed by using conductance, width and length of the channel, which are the input data for SWAT. The thickness of river bed can be read by the user. To convert SWAT's channel length to MODFLOW's river length, the length of main channel in SWAT is divided by the number of river cells in MODFLOW. These data are the variables for exchange from SWAT to MODFLOW during the process from S-2-1 to M-2. Exchange rate in each cell during M-2 process is computed by summing the contributed groundwater flow to river and the contributed river flow to aquifer for each respective SWAT's channels. Therefore, another process is required to match SWAT's channel with MODFLOW's river cells. By making use of this process, the exchange flow rate between river and aquifer is converted to flowrate in SWAT's channel, and returned to S-2-2. The HRU based groundwater flow is computed if the difference between the contributed groundwater flow to river and the contributed river flow to aquifer is divided by the HRU area.

Water transfer module

In this study, water transfer methods are enhanced considering the different types of water source and destination. In SWAT, if the source type is aquifer or reach or reservoir and the destination type is outside of watershed, then SWAT can handle this water transfer by using consumptive water use option. In this case, water can be removed from the watershed as a constant monthly rate. To solve this limitation, the method of removing water from the watershed by using the "water transfer" command is proposed. This method uses daily/monthly/yearly options as well as constant amount or constant rate or minimum value which already used for water transfer from water source. The source code "transfer.f" was modified to consider the direct water transfer from various water sources such as aquifer, reach, and reservoir without using the command of "point source and inlet discharge". In this study, the enhanced water transfer method when the source type is shallow aquifer and the destination is the reach of different subbasin is only considered. If the source is the shallow aquifer, then the water transfer method is related to MODFLOW's Well package. The "water transfer command" in SWAT, which is linked with the Well package in MODFLOW, was used in order to simulate the water transfer by pumping. The Well Package in MODFLOW can be used to simulate wells which withdraw water from the aquifer or add water to it at a specified rate during a given stress period. If an amount of water is removed from the aquifer by pumping, the discharged water can be transferred to any destination. This is performed with a water transfer command in the watershed configuration file(basins. fig) of the SWAT. In this study, the locations of water source are matched to the well cells in MODFLOW. The locations of the receiving water bodies are set to be specified reaches in different subbasins.

Application

The combined SWAT-MODFLOW model is tested in the Gyungancheon basin in Korea, where the area of the basin is 259.2 km^2. As shown in Fig. 4, this drainage basin is divided into 9 sub basins, and the area of each subbasins range from 7 to 60 km^2. The channel lengths of each subbasins range from 5 to 20 km.

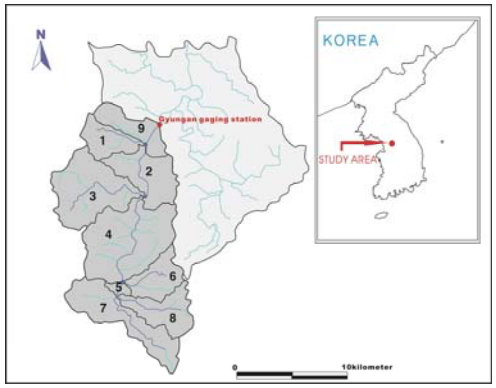
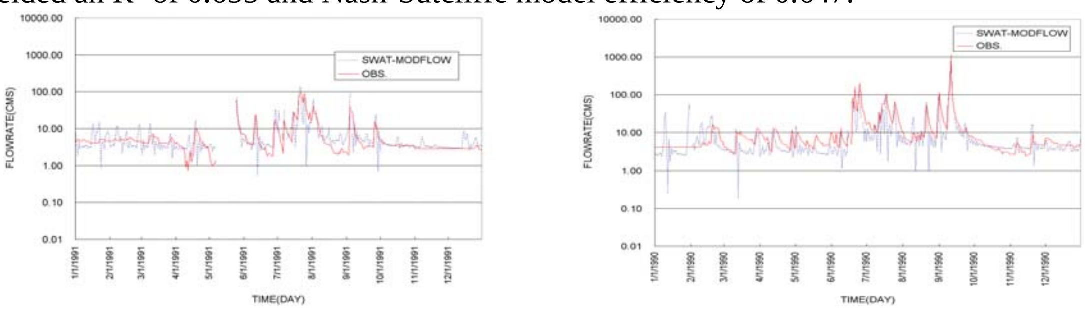


Fig. 4. Division of subbasins

AVS2000 (DiLuzio et al., 2001) was used to automate the development of model input parameters. Daily precipitation for Suwon gauging station which covers the entire watershed were obtained from the hydrologic database of MOCT (Ministry of Construction and Transportation). Daily values of maximum and minimum temperatures, solar radiation, wind speed, and relative humidity were collected from the weather service data of KMA(Korea Meteorological Administration). Land use digital data(1:25,000) from the National Geographic Information Institute of MOCT were used. The detailed soil association map (1:25,000) from the NIAST (National Institute of Agricultural Science and Technology) was used for selection of soil attributes. Thirty-eight hydrologic soil groups within the Gyungancheon watershed were used for analysis. Relational soil physical properties such as texture, bulk density, available water capacity, saturated conductivity, soil albedo etc. were obtained from the Agricultural Soil Information System(http://asis.rda.go.kr) of NIAST(2005). HRUs in SWAT are formed based on the hydrologic soil group and land use. However, due to the semi distributed features of SWAT, spatial locations of each HRU within subbasins is not determined. Hence, to reflect HRU locations to MODFLOW, spatially distributed HRUs using the DEM, with a cell-size of 300m, are used to match the discretized watershed with MODFLOW grids. An HRU-Grid conversion tool is made for this purpose. Within MODFLOW, the aquifers are represented as two layers, discretised into a grid of 126 rows and 123 columns. Groundwater information from National Groundwater Information Management and Service Center was used to determine the aquifer characteristics for MODFLOW inputs. Hydraulic conductivity in alluvial aquifer is 2.113×10^{-3}cm/sec. Additionally, an assumption was made, according to a literature by Freeze and Cherry(1979), that the specific yield ranges from 0.1 to 0.3. Conductance of river bed was determined as one tenth of the alluvial aquifer by trial and error procedure. The watershed boundaries were designated as no-flow cells. Recharge was distributed according to SWAT simulation outputs for each day. River-aquifer interaction was simulated using a RIVER package for MODFLOW. River stage of MODFLOW is imported from SWAT's daily simulation outputs.

Model Validation

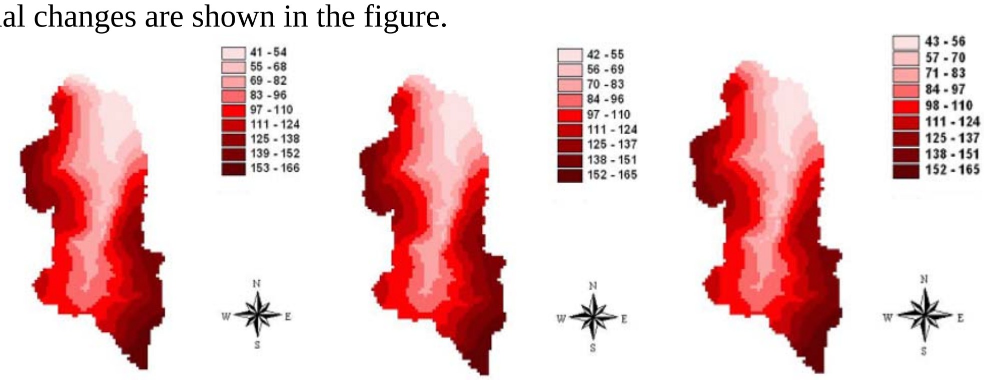
Daily stream flow for year 1990 was calibrated against measured daily stream flow. Inputs to the model are physically based (i.e. based on readily observed or measured information). Several variables such as ESCO, AWC, CN2 were used for calibration. For the groundwater model, primary calibration parameters were the aquifer hydraulic conductivity and storativity. The hydraulic conductivity, the storativity and river bed conductance were then optimized by trial and error procedure. These variables were optimized by minimizing the low flow error during dry season. Calibration was performed on total stream flow. If simulated and measured flows are within 10% calibration was terminated. Observed and calibrated flow is shown in Fig. 5(a). Total flow for the entire basin yielded an R^2 of 0.79. During 1991, daily stream flows were simulated by SWAT-MODFLOW model at the Gyungan gaging station in order to verify the performance of the calibrated SWAT-MODFLOW. The hydrograph was plotted using a log scale in order to emphasize the quality of low flow simulation. Fig. 5(b) shows the SWAT-MODFLOW simulation during 1991. Total flow for the entire basin yielded an R^2 of 0.655 and Nash-Sutcliffe model efficiency of 0.647.



(a) calibration **(b) verification**

Fig. 5. Simulation results by SWAT-MODFLOW

Since SWAT-MODFLOW is a grid base model, capable of calculating the spatially distributed groundwater table as shown in Fig. 6. Fig. 6 shows the distributions of groundwater head at 500, 800, 1000 days after running SWAT-MODFLOW. The gradual changes are shown in the figure.

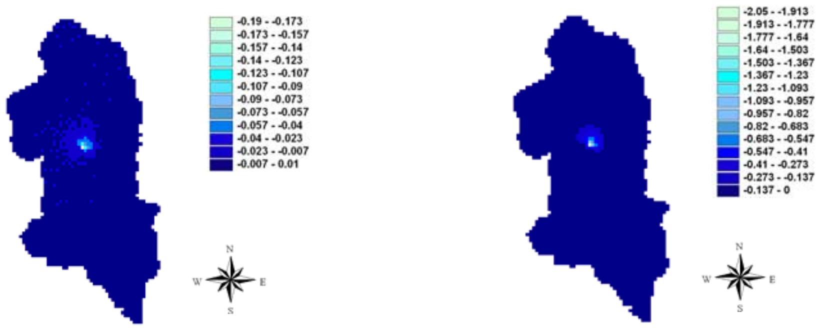


(a) 14 May, 1990 **(b) 11 March, 1991** **(c) 26 Sep, 1991**

Fig. 6. Spatial distribution of groundwater head simulated by SWAT-MODFLOW

The Application of SWAT-MODFLOW pumping module

The advanced pumping module, which is added to the SWAT-MODFLOW, is initially tested to Gyungancheon watershed in Korea. For this purpose, the transfer command should be inserted to the data file ‘basins. fig. We located a single discharge well at a certain cell(column=68, row=43) of MODFLOW in subbasin 4. The source is the shallow aquifer in subbasin 4, and the destination is the reach of subbasin 1. Two pumping scenarios, 100m^3/day and 1000m^3/day, are applied. The groundwater variation and water budget variation according to pumping were examined. Fig. 7 shows the groundwater drawdown contour lines with pumping rates of $100m^3$/day and $1000m^3$/day. These results demonstrate that the advanced pumping module in the combined SWAT-MODFLOW model could effectively describe the water transfer in the watershed.



(a) pumping rate Q=100CMD(Unit:m) (b) pumping rate Q=1000CMD(Unit:m)

Fig. 7 Groundwater drawdown contour lines

Fig. 8 shows the comparison of the results between natural water budget and water budget with pumping. Fig. 8 shows that the reduced groundwater discharge by pumping is transferred to stream, so resultant stream flow would be increased just as much.

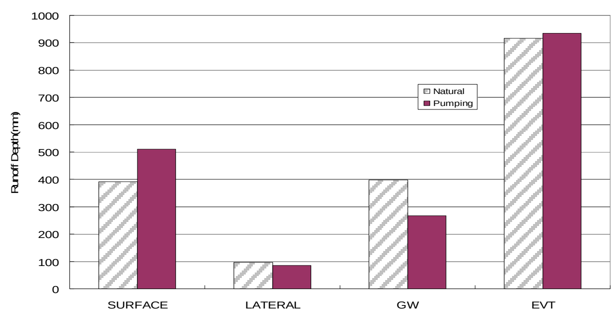


Fig. 21 Water budget results with or without pumping

Conclusion

In this study, the combined watershed and groundwater model, SWAT-MODFLOW is introduced and applied to Gyungancheon basin in Korea. As model is fully combined, it is very useful to compute surface and groundwater components altogether. The application demonstrates a combined model which enables an interaction between saturated zones and channel reaches. This interaction plays an essential role in the runoff generation in the Gyungancheon basin. The comprehensive results show a wide applicability of the model which represents the temporal-spatial groundwater head distribution.

Acknowledgments

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Comparing SWAT and WetSpa on the River Grote Laak, Belgium

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Abstract

SWAT and WetSpa are two hydrologic models that can be used for the simulation of discharges. In order to improve the performance of the models in the future, we compared these two models on the catchment of the low land river Grote Laak. This basin has an area of 56 km² and is situated in Belgium. An automatic calibration (SCE-UA for SWAT and PEST for WetSpa) was performed on a period of four continuous years. Both models provided acceptable results, although SWAT showed slightly better calibration and WetSpa better validation results. Both were not able to simulate an extreme dry event.

KEYWORDS: distributed modelling, semi-distributed modelling, model comparison, SWAT, WetSpa.

Introduction

Hundreds of hydrologic models have been developed in the past, all with different features and with different advantages and limitations. Distributed models, like WetSpa (Liu et al., 2003, Gebremeskel et al., 2002) and SHE (Abbott et al., 1986a,b), largely maintain a physical background because of the high level of detail. Modelling these details, however, requires a lot of computer time and data. Due to the spatial lumping, semi-distributed models, like SWAT (Arnold et al., 1993) and TOPMODEL (Beven and Freer, 2001), reduce the calculation time and the data requirements significantly, when compared to distributed models. As a result of the lumping, however, the modelling of the processes becomes less detailed and the relation with reality becomes weaker. A typical example of the latter is the use of the equations of de Saint Venant for river routing in distributed models and the use of hydrologic routing methods in semi-distributed models. As a consequence of these differences, the user must find a compromise between the computation time and the data requirements on the one hand and the detail of the representation of the system and the processes on the other hand. While it is generally recognised that such a compromise will be case and problem specific, no objective criteria exist so far to select one or another approach.

In order to shed some light on the problem, this paper presents a comparison of a distributed model (WetSpa) and a semi-distributed model (SWAT) for a watershed in Flanders, i.e. the watershed of the river Grote Laak. Refsgaard and Knudsen (1996) reported earlier on a comparison of three different models in three catchments and SWAT was compared with MIKESHE (also a distributed model) by El-Nasr et al. (2005).

[1] Parameters related to snowfall and snowmelt are negligible for this catchment and not taken into account.

Based on the information provided in this paper, future research should be performed on the processes that are included in the two models and that lead to the resulting output. In that way, the performance of the models can be improved, when adapted to these findings.

Materials and methods

Model description

SWAT

SWAT (Soil and Water Assessment Tool) is a physically based, time continuous, semi-distributed river basin scale model, developed by the USDA Agricultural Research Service (ARS) in order to quantify the impact of land management practices on water quantity, sediment and water quality in large complex watersheds with varying soil, land use and management conditions over long periods of time. The semi-distributed characteristics of the model are linked to the division of the catchment into subcatchments, which are divided into HRU's (Hydrological Response Units): portions of the subbasin containing a unique combination of land use and soil. The processes are lumped at the HRU level and the discharge of the subcatchments is routed through the river network to the main channel and the basin outlet (Neitsch et al., 2002).

We used the most recent version of SWAT (SWAT2005) for the modelling presented in this paper.

WetSpa

WetSpa is a physically based, distributed hydrological model for predicting the Water and Energy Transfer between Soil, Plants and Atmosphere on regional or basin scale and was developed at the Vrije Universiteit Brussel, Belgium. The model conceptualizes a basin hydrological system as being composed of atmosphere, canopy, root zone, transmission zone and saturation zone. The processes within WetSpa are based on the division of the catchment into grid cells to maintain the distributed characteristics (Wang et al., 1996).

For the research presented in this paper, we used the WetSpa Extension. This version is capable of predicting outflow hydrograph at basin outlet or any converging point in a watershed with a variable time step (De Smedt et al., 2000).

Site and Data description

Site and maps

The river Grote Laak is a typical lowland river and is situated in the North Eastern part of Belgium. Grote Laak is a sub-catchment of the Nete basin, a part of the larger Schelde catchment and has a surface of 56 km². The mean elevation is 36.6 m with a maximum of 65 m and a minimum of 17 m. A specific feature of the basin are the canals that cross the catchment. The catchment's yearly precipitation is around 900 mm.

The topography of the study area was digitised from 1/10,000 maps and divided into grid cells of 50 by 50 m. Based on the resulting Digital Elevation Model (DEM), we performed an initial delineation of the watershed with the ArcView GIS SWAT user interface (Di Luzio et al., 2002), with an outlet close to the gauging station of Laakdal. Figure 1 presents the topography of the area, with the main river courses and measuring stations.

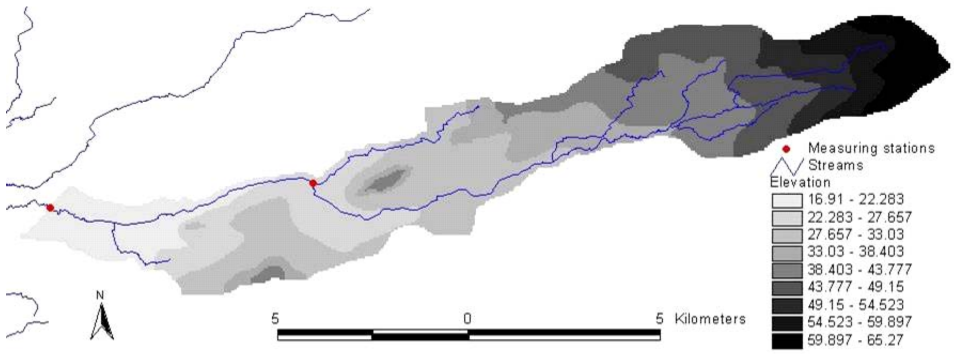

Figure 25: Topography of the study area.

Based on the mask, created out of this generated watershed, we defined the study area for the use in WetSpa. Because of the small differences in altitude and the small slopes in the catchment of the Grote Laak, WetSpa was not able to generate the correct main river channel and some areas were not routing towards the outlet of the basin. In order to get a better relation between the digitised and the modelled streams, we created a secondary elevation map. We performed a burn in of the streams on this second elevation map with the "CRWR-prepro"-tool (Olivera, 1998) by increasing the elevation of all cells but those that coincide with the digitised streams, with 5000 m. We also added an artificial wall around the area to become a better delineation and a total coverage of the area with subwatersheds, in WetSpa. The original elevation map was used to generate the correct value of the slopes and the elevations. The secondary map was used for the stream order, stream network and subwatersheds. Wetspa generated 185 subwatersheds in this way; in SWAT we only defined three subcatchments.

The predominant soil types in the catchment have a sandy texture and the predominant land use classes are residential (high density), pasture, agricultural land (corn) and mixed forest. Figure 2 shows both the land use and soil map, adapted to the extent of the catchment, which are based on maps ordered from "GIS Vlaanderen" and were resampled to four soiltypes (landdune, sand, sandyloam, clay) and five land use classes (corn, mixed forest, pasture, urban, water).

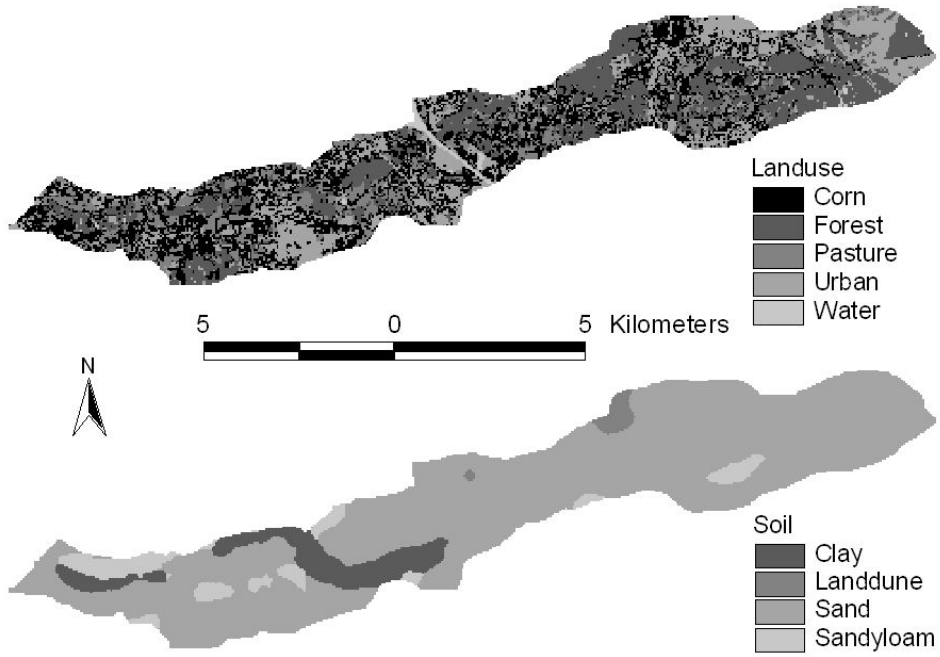

Figure 26: Land use and Soil map of the study area

Table 1 presents the distribution of soil and land use over the study area. In SWAT we performed a redistribution on subcatchment scale, leaving out the land uses accounting for 2% or less of the area and 8% or less for the soil classes. This resulted in a total number of 23 HRU's with varying sizes between 0.2 and 6.9 km².

Table 20: Distribution of soil and landuse over the study area

	Soil types				Land use				
Class	Clay	Landdune	Sand	Sandyloam	Corn	Forest	Pasture	Urban	Water
Percentage[a]	8.29	1.11	84.19	6.41	32.28	28.89	16.39	21.71	0.73
Percentage[b]	8.36	0	87.52	4.11	32.51	29.11	16.50	21.87	0

[a] Actual percentage of soil and landuse in the area.
[b] Percentage after redistribution in SWAT (not used in WetSpa).

Hydrometeorological data

For the gauging station of Laakdal, discharge data is available from October 1998 until December 2003 from the HYDRONET database (IVA-VMM, 2006). We used the discharge data from January 2000 on for calibration of the models and the year 1999 for validation. The main reason for this choice is that we wanted to include the very dry year 2003 (precipitation around 700 mm) in the calibration period.

Meteorological data are all available from the Royal Meteorological Institute (KMI). WetSpa uses daily time series of precipitation, potential evapotranspiration and average temperature data. For SWAT we provided daily time series of precipitation, minimum and maximum temperature, relative humidity, solar radiation and windspeed data in order to use the Penman-Monteith equation. Table 2 presents the available data from January 1996 until December 2003 for different measuring stations.

Table 21: Available meteorological data

	Ukkel	Kleine Brogel	Lichtaart	Turnhout	Geel
Precipitation	X	X	X	X	X

PET	X				
Av. temperature	X	X			X
Min.-Max. temperature	X				
Windspeed	X				
Solar radiation	X				
Rel. Humidity	X				

Calibration techniques

Theoretically, the parameters of physically based models do not need to be calibrated. However, due to uncertainty on model input and measurements and because of the spatial variability in both horizontal and vertical direction, we performed a calibration on discharge data for certain parameters for both models.

For SWAT we used the model incorporated LH-OAT – method (Latin Hypercube-One factor At a Time) (van Griensven and Meixner, 2003, Holvoet et al., 2005) to perform a sensitivity analysis. We performed an automatic calibration with the SCE-UA-algorithm (Shuffled Complex Evolution), also incorporated in SWAT (Eckhardt and Arnold, 2001), on nine sensitive parameters and a manual calibration on three other parameters afterwards. Table 3 shows the most sensitive[1] and the calibrated parameters.

Table 22: SWAT parameters

Parameter		Sensitivity	Automatic Calibration	Manual Calibration
CN2	SCS runoff Curve Number	1	HRU split[a]	
ALPHA_BF	Baseflow alpha factor	2	HRU split[a]	
SURLAG	Surface runoff lag coefficient	3	X	
CH_K2	Effective hydraulic conductivity (main channel)	4	X	
GWQMN	Threshold depth of water for return flow to occur	5	HRU split[a]	
CH_N	Manning coefficient for channel	6		X
CANMX	Maximum canopy index	7	X	
RCHRG_DP	Groundwater recharge to deep aquifer	8	X	
SLSUBBSN	Average slope length	9		X
ESCO	Plant evaporation compensation factor	10	X	
SOL_AWC	Available water capacity	11	X	
SLOPE	Average slope steepness	12		X

[a] HRU split: the parameter was calibrated for the different HRU's separately.

We performed a manual and an automatic calibration, using PEST (Doherty, 2001), on the WetSpa model on eight (out of eleven) global model parameters. The three global model parameters related to snowfall and snowmelt were not taken into account.

Criteria for the evaluation of the model performance

Although graphical presentation of the overall shape of the time series of discharges (simulated vs. observed) can be useful for interpreting the model results, objective and quantitative information is needed for the evaluation of the model performance and the comparison of models. In this study, we evaluated the performance compared to measured data by following criteria:

1. NSE (Nash-Sutcliffe efficiency):

The Nash-Sutcliffe efficiency (Nash and Sutcliffe, 1970) describes how well the stream flows are simulated by the model. A value of one for the NSE indicates a perfect fit between simulated and observed hydrographs, a value of zero indicates that average measured stream flow would have been as good a predictor as the modelled flow. Equation 1 presents the NSE equation:

$$NSE = 1 - \frac{\sum_{i=1}^{N}\left(Qs_i - Qo_i\right)^2}{\sum_{i=1}^{N}\left(Qo_i - \overline{Qo}\right)^2} \tag{1}$$

where Qs_i is the simulated stream flow, Qo_i is the observed stream flow and $\overline{Qo}$ is the mean observed stream flow.

2. LNSE (Logarithmic version of Nash-Sutcliffe efficiency):
This logarithmic transformed Nash-Sutcliffe efficiency is a criterion for the time evolution of the low flow simulations (Smakhtin et al., 1998). A value of one gives a perfect representation. Equation 2 presents the LNSE equation

$$LNSE = 1 - \frac{\sum_{i=1}^{N}\left(\ln\left(Qs_i\right) - \ln\left(Qo_i\right)\right)^2}{\sum_{i=1}^{N}\left(\ln\left(Qo_i\right) - \ln\left(\overline{Qo}\right)\right)^2} \tag{2}$$

3. ANSE (Adapted version of Nash-Sutcliffe efficiency):
An adapted version of the Nash-Sutcliffe efficiency is presented in equation 3. It gives a criterion for the evaluation of the high flow simulations. A value of one gives a perfect representation (Liu and De Smedt, 2004).

$$ANSE = 1 - \frac{\sum_{i=1}^{N}\left(Qo_i + \overline{Qo}\right)\left(Qs_i - Qo_i\right)^2}{\sum_{i=1}^{N}\left(Qo_i + \overline{Qo}\right)\left(Qo_i - \overline{Qo}\right)^2} \tag{3}$$

Results and Discussion

Figure 3 presents the comparison of simulated and observed discharges for the calibration period using both models. As one can see, both models have comparable and acceptable results for the daily discharges. Table 4 states these findings, although it can be noticed that the quantitative performance of SWAT is slightly higher than that of WetSpa, especially on the low flows. The water balance was closed in both cases, with corresponding components (Table 5).

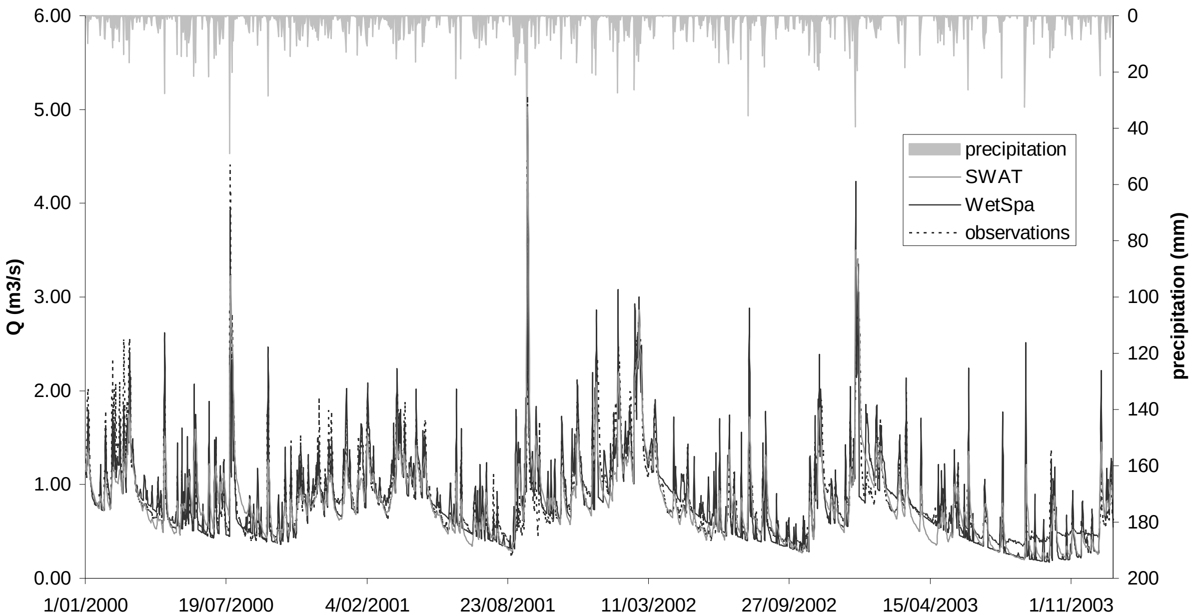

Figure 27: Calibration of SWAT and WetSpa (2000-2003)

Table 23: Model performance after calibration (2000-2003)

	NSE	LNSE	ANSE
SWAT[a]	0.739	0.658	0.788
WetSpa (manual) [a]	0.701	0.588	0.752
WetSpa (automatic) [a]	0.701	0.588	0.752

[a] One day of observed discharge is adapted to a known flooding.

Table 24: Water balance components (2000-2003)

	Precipitation (mm/y)	Runoff (mm/y)	Evapotranspiration (mm/y)
SWAT	918.1	466.0	434.6
WetSpa	920.8	464.1	476.2

Although SWAT performs better in this case, it is not possible to extend this to other basins. Due to the limited area of the catchment and the availability of sufficient discharge data for calibration, scale effects are probably less important for the semi-distributed model (Refsgaard, 1997). It will be necessary to extend the study in the future to other catchments with other properties to reveal this.

The lower performances on low flows (for SWAT and WetSpa) are partly generated by the very dry year 2003. It can be seen in the previous figure that both SWAT and WetSpa are underestimating the baseflow during this extreme event.

The previous table also shows that the results for the manual and automatic calibration in WetSpa are identical. Due to the little number of parameters, WetSpa is more stable and less prone to equifinality. Besides the calibration method that was used, the number of parameters also has its consequences on the calculation time for the calibration: for WetSpa it took less than four hours to calibrate the model; in SWAT more than four days were needed to get these results. We can conclude that, known that the area is limited, in this case the advantage of less calculation time for semi-distributed models was not respected.

The stability of the model with respect to the parameters also has its effects on the quantitative performance of WetSpa for the validation (Table 6). The latter performance is significantly higher than for SWAT and even better than during the calibration period. Although the previously discussed dry year 2003 influenced the results of the calibration, one can still conclude that due to the stable performance, WetSpa is better suited to make predictions on future events. Here again future research should state this.

Table 25: Model performance after validation (1999)

	NSE	LNSE	ANSE
SWAT	0.563	0.580	0.809
WetSpa	0.752	0.690	0.919

Conclusions

SWAT and WetSpa have acceptable and comparable model results for the basin of the Grote Laak, although they both have problems predicting extreme dry events and resulting low flows. The stability of WetSpa with respect to the parameters leads to a better validation and capability to make predictions and has its effects on the calibration time. Future studies on other catchments with other properties are needed to extend the findings of this study.

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Integration of MOHID Model and Tools with SWAT Model

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Abstract

MOHID Water Modelling System is an integrated state of the art modular system, composed by a series of models that simulate surface water bodies, streams and watersheds. MOHID's code development follows a methodology which improves its robustness related to programming errors. MOHID is written in ANSI FORTRAN 95, profiting from all its new features, including the ability to produce object oriented code, although it is not an object oriented language. It includes object oriented features. This results in a series of object oriented models for simulating the water cycle which integrates several different scales and processes.

SWAT source code was partially modified, namely in the inputs and outputs of the model, using MOHID's code and programming philosophy. These changes maintained the integrity of the original model, thus guarantying that results remain equal to the original version of SWAT. This allowed to output results in MOHID format, thus making it possible to immediately process it with MOHID visualization and data analysis tools.

MOHID River Network is a river model developed in European funded project (TempQSim). A link between SWAT and MOHID River Network was developed. Presently, two modified versions exist, based on the two SWAT releases (SWAT2000 and SWAT2005).

The inclusion of output result files in HDF5 format is currently under development. This allows the visualization of watershed properties (modeled by SWAT) in animated maps using MOHID GIS and animation tools. These tools allow the production of animated files showing the spatial and time evolution of the modeled properties.

The modified version of SWAT described here has been applied to various national and European projects. Results of the application of this modified version of SWAT to estimate diffuse nutrients loads to estuaries and water bodies will be shown.

MOHID source code and its support tools are freely available under the GNU Public License.

KEYWORDS: SWAT-MOHID, Hydrological, Statistics, Mondego

Introduction

SWAT has demonstrated its potential in analyzing the water cycle and the related hydrologic fluxes at the catchment scale and the associated nutrient transport in study sites all around the world (Arnold & Fohrer, 2005). The release of the source code is surely one of the reasons of SWAT success. The free access to the code allowed also the many modified versions of SWAT allowing its adaptation to users needs.

Like SWAT, MOHID is an open source code model under the GNU Public License, and has users all around the world. MOHID is written in ANSI FORTRAN 95, profiting from all its new features, including the ability to produce object oriented code with it, although it is not an object oriented language. It includes object oriented features like those described in Decyk et al (1997). This development strategy resulted in a series of object oriented models for simulating the water cycle which integrates several different scales and processes. This type of programming allowed a straightforward integration of SWAT and MOHID models and tools.

Two modified versions of SWAT were developed, based on the two SWAT releases (SWAT2000 and SWAT2005). The functionalities described herein are equal in both versions. So we refer to this modified versions in this paper has SWAT-MOHID. However, all the results showed here were obtained with SWAT2005 version. The aim of these developments is not to replace the use of SWAT graphical user interfaces for input or output, but to complement it. This also means that the developed outputs do not include all the SWAT variables but instead, they were developed on a need basis either for easier exploring, results analysis, model coupling, etc. This development benefits from many current developments and improvements of MOHID tools made by its users and will continue to benefit in the future. To make this development sustainable two important aspects were taken into account: i) the use of a source code version control software to keep track of changes made by many users in the code ii) the use of online discussion forum and wiki (using Wikipedia technology) that together allows the users to help each other and to participate in the documentation of the models and tools.

One of the reasons to develop SWAT-MOHID was to compute nutrient loads onto surface water bodies (e.g. reservoirs, estuaries), using them has boundary conditions for models like MOHID Water and CeQualW2.

So far the coupled version of MOHID and SWAT has been used to study the dynamics of eutrophication in several Portuguese reservoirs and estuaries.

This paper shows the developments made in SWAT using Mohid tools and presents the results of the application of SWAT-MOHID to the Portuguese Mondego Watershed. The application of the model was made in the framework of HARP-NUT Guidelines, which allows making an annual comparison between the load oriented approach to the source oriented approach (Borgvang & Selvik, 2000; Schoumans & Silgram, 2003). Only the hydrodyamic calibration for Mondego is shown in this paper.

SWAT-MOHID

SWAT source code was partially modified, namely in the inputs and outputs of the model, using MOHID’s code and programming philosophy. These changes maintained the integrity of the original model, thus guarantying that results are equal to the original version of SWAT. A first modification was the implementation of time series outputs. The modified SWAT source code enables the user to output time series for two geometric entities found in SWAT: (i) hydrological response units (HRU) and (ii) or sub-basins/reaches.. The output of the time series is done with a frequency which is independent of the “normal” SWAT output and the format of the time series is done in MOHID format.

Time series written for selected HRU’s are organized in four categoiries: (i) meteorological information, (ii) plant growth, (iii) nutrient concentration and (iv) erosion.

Time series written fro selected sub-basins/reaches are nutrient concentration, sediments concentrations and the water flows into the reaches.

Since the time series are written in MOHID format, the user has two new possibilities: (i) use MOHID tools to analyze them or (ii) use them as boundary conditions for a detail river network model (MOHID River Network). These features are described in more detail later in this paper.

MOHID time series are stored as ASCII files, with syntax very similar to XML files. Figure 28 shows an example of a MOHID time series. The header section contains general information (reference date, time units used and the identification of each column), followed by a block which contains columns with the actual data. The first column represents the time passed since the reference date.

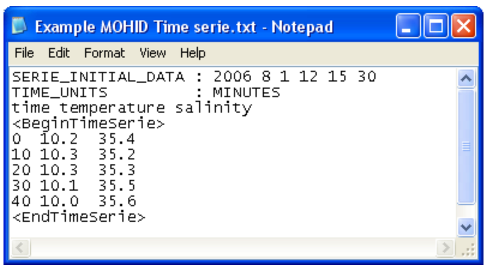

Figure 28. Example file of a MOHID Time series.

MOHID Time Series editor

MOHID Time Series Editor is a graphical user interface written in VB.NET which allows the user to visualize in a straightforward way MOHID time series required or produced by the MOHID numerical programs.

In each time series graphic window it is possible to change the features of the graphic using the available buttons in the menu located above the graph, namely: 1) Chart Type: opens a Command and Options window allowing the selection of several actions: General, Border/Fill, Data Sheet, Type, Series Groups, Show/Hide; 2) Show/Hide Legend: selecting the button causes the appearance of the legend, clicking on it again makes the legend disappear; 3) Commands and Options: opens a Command and Options window allowing the selection of several actions: General, Border/Fill, Data Sheet, Type, Series Groups, Show/Hide; 4) Chart Wizard: opens a Command and Options window allowing the selection of actions concerning Data Source, Data Sheet and Type; 5) CheckBox1: checking it allows the selection in Window Width [days] the number of days to appear in the graph and in the ruler the location in the time interval of that number of days; 6) Save Image: saves the graph as a GIF or JPEG file in the path provided by the user. This sample representation of a time series file shows.

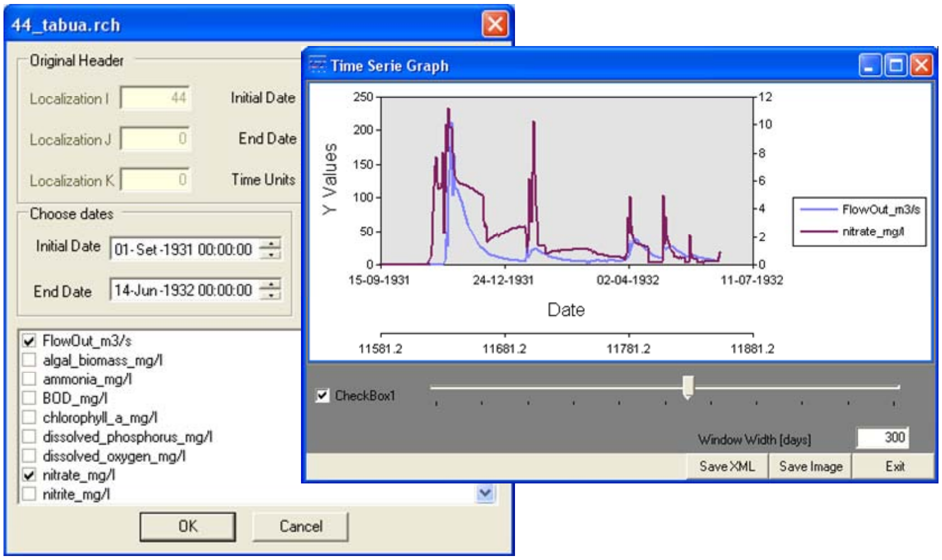

Figure 29. Example of the visualization of two reach SWAT results: flow and nitrate concentration.

MOHID Time Series Analyzer

The MOHID Time Series Analyzer application calculates statistics and obtains graphs of comparison between two time series data. Running options for this application are specified in a graphical interface. For model validation or model comparison studies it is useful to calculate comparison statistics and graphs. The MOHID Time Series Analyzer allows the computation of such information for one or more properties with two site specific time series files in a user-friendly way.

The information is obtained based on data located on a time window. That time window is defined as the time period for which data in both time series exists and any further constrain indicated by the user in the interface. If time instants in the two time series are different then the second time series data is linearly interpolated for the first time series data time instants.

The MOHID Time Series Analyzer statistics output is organized in the Statistic (All), Statistic (Daily) and Statistic (Monthly) tabs, for statistics considering all data, daily averages and monthly averages, respectively.

This output is only produced after the data input by the user in the General tab and the successful data processing. In each of the statistics tabs the information content is similar: 1) Observed Average (average of the first time series); 2) Modeled Average (average of the second time series); 3) Bias; 4) RMSE (root mean square error); 5) R2 (Pearson product-moment correlation coefficient); 6) Model Efficiency (Nash-Sutcliffe coefficient). These parameters are proposed by Evans et al. (2003) to evaluate model efficiency.

MOHID HDF

MOHID's HDF class is a class on the top of the Hierarchical Data Format (HDF library). HDF is a general purpose library and file format for storing scientific data, developed and maintained by the United States National Center for Supercomputing Applications. The main functionality of this library is to store matrix data in a structured way (Braunschweig et al., 2004).

The inclusion of output result files in HDF5 format is currently under development. This allows the visualization of watershed properties (modeled by SWAT) in animated maps using MOHID animation tools like MOHID-GIS (Figure 30). These tools allow the production of animated graphic files showing the spatial and time evolution of the modeled properties. The user decides on the time step output. For each output a vector with the values of soil water content of each sub-basin is stored.

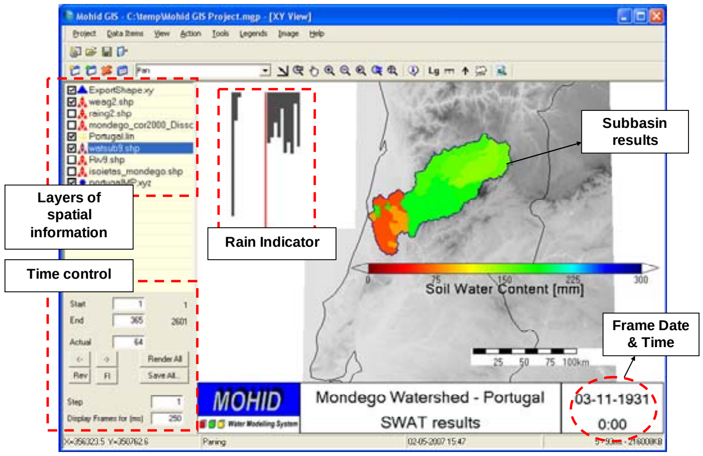

Figure 30. SWAT Soil Water content results and visualization in MOHID-GIS using the HDF result files produced by SWAT-MOHID (see complete animation in: http://www.mohid.com/Gallery/Swat_HDF_view.gif)

Coupling SWAT with Mohid River Network

Common problems for the application of basin models in the temporary catchments are related to: i) Periods without runoff (which results in numerical problems for most models) ii) Extreme first flush effects with the beginning of the rain period (sub-hourly time steps required in simulations) iii) Quality of the water (sediments, solutes) is frequently poor described in the models and highly variable in time.

In order to respond to such demands a physical based model, the TempQsim STREAM model, was developed (Galvão et al., 2005). This model is currently maintained by MOHID group under the name MOHID River Network (MRN). MRN computes water, sediments and properties transport in a river network. The model is written in FORTRAN 95 and follows an object oriented programming philosophy with a finite volume approach (Braunschweig et al, 2004). The different processes occurring in the river are programmed in different modules. This model has been calibrated for Vène watershed (France) with a special focus on the transport of particulates for the first significant flood events (Obermann, 2007).

Fluid flow in this model is governed by conservation equations for mass, momentum, energy and any additional constituents and the numerical algorithm is based on the finite volume approach. Following this strategy it is easier to build conservative transport models and coupling between modules is also simpler because it is based on fluxes. Object oriented programming was also used, which facilitates model coupling. An interface between this model and SWAT was also developed in order to simulate agriculture in the catchment using MOHID for simulating the river network and the corresponding sediment transport and biochemistry (Chambel-Leitão et al, 2006) (Figure 31). SWAT source code was slightly changed so time series of flow / properties are produced for each sub-basin. A “watsub” theme with the location of the outlets of each sub-basin (created by SWAT ESRI ® ArcView extension) is read by MOHID GIS and dynamically construct links between the time series locations of SWAT and discharge nodes of the MRN. MRN runs using surface runoff, lateral flow and groundwater flow from SWAT as input discharges. Property concentrations can be considered constant (user supplied) or can be an output of modified SWAT.

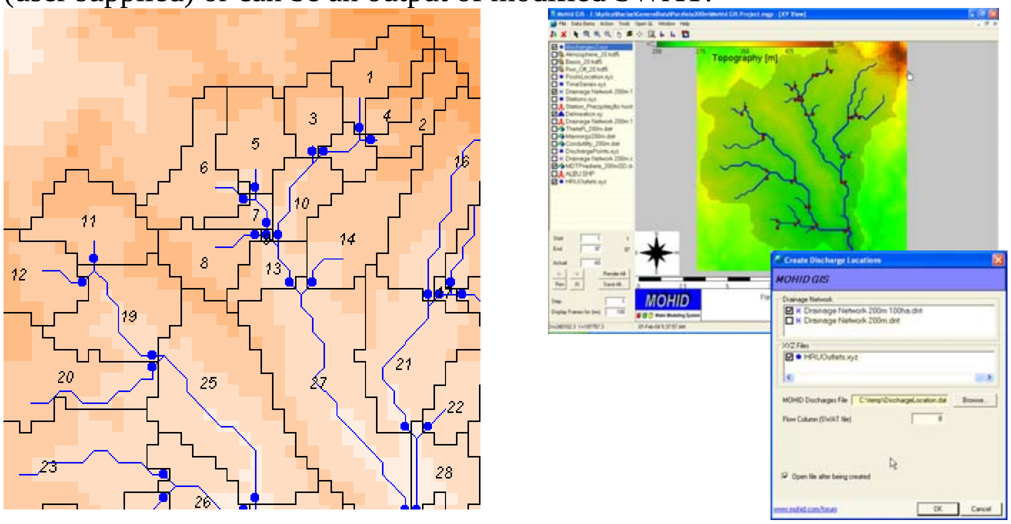

Figure 31. Connection of SWAT and MOHID River Network using MOHID-GIS interface.

In the framework of the TempQsim project (www.tempqsim.net) the problem of temporary waters and the role of rainy events for the total river budget was addressed. Rainy events are the major challenge of catchment models and specifically of SWAT. For this reason MOHID Land (www.mohid.com) was developed allowing to the use of variable fine grids and a dynamic time step determined by the iterative procedure.

SWAT-MOHID coupling was tested for Pardiela Basin. Pardiela is a temporary river located in the south Mediterranean part of Portugal called Alentejo covered by undulating plains ranging from 50 m to 400 m elevation. A first run was made with SWAT model and MOHID coupled with SWAT, for similar river networks, to see if they were producing the same results. For similar channel manning's N coefficient, flow results were the same (Figure 32). In both models the Kinematic wave equation was used (Neitsch et al., 2000).

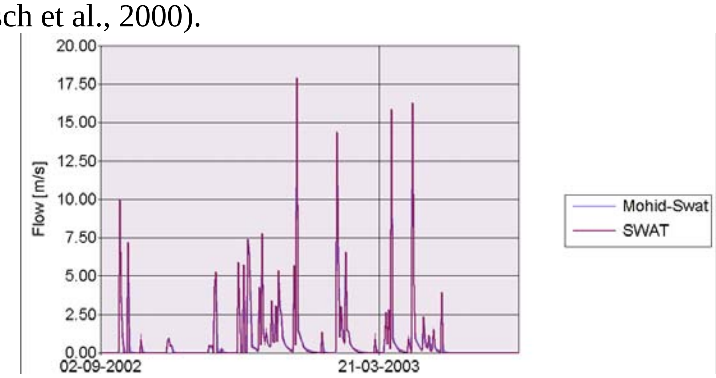

Figure 32. River flow in Swat compared with flow in Mohid coupled with Swat.

SWAT-MOHID application

The modified version of SWAT described here, SWAT-MOHID, has been applied to various national and European projects. Results of SWAT-MOHID will be shown for the Mondego watershed. Mondego River is located in the central region of Portugal drains to the Atlantic Ocean a basin with an area of about 6700 km^2. This river basin has an elongated shape with the longest axis orientated NE–SW and the maximum altitude is nearly 2000 meters. The Mondego itself is the largest entirely Portuguese river with a length of 234 km and is located in a region of transition between Atlantic to Mediterranean climate. In the early 90's the estuary was object of important geomorphologic modifications in order to improve navigability and the upper communication between the two arms was closed. In the late 90's symptoms of eutrophication have been identified. However comparing land use from CORINE land cover from 1990 and 2000 the agriculture area has decreased. In order to simulate the nutrient dynamics of the watershed and its effects on the estuary, data was gathered to setup SWAT-MOHID model.

The digital elevation model (DEM) is in a raster format with a grid resolution of 70 m, which has been clipped from the Shuttle Radar Topography Mission (SRTM) DEM data (Hounam & Werner, 1999).

The land use map (Figure 33) has been clipped from the CORINE (released in 2000 obtained in http://dataservice.eea.europa.eu/dataservice/) whose legend is based on the CORINE level 3 legend. The original legend entries were reclassified and, in some cases, aggregated to conform to the land use database present in the SWAT model (the watershed results included 15 land use classes). For simplicity of representation Figure 33 shows only four classes composed from level 1 classes of CORINE.

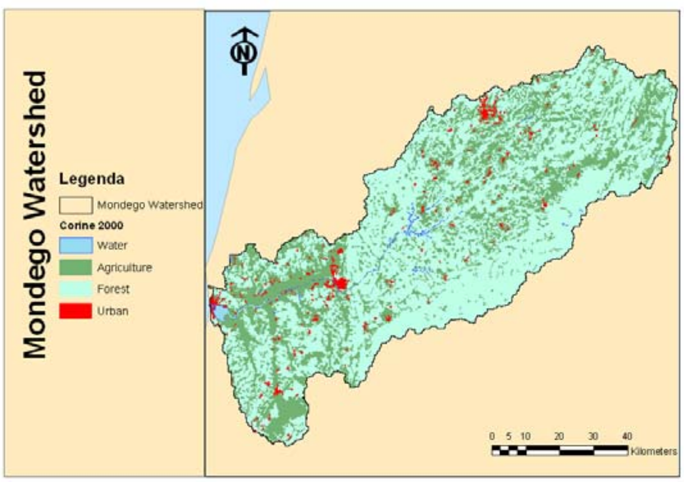

Figure 33. Simplified land use map. LU/LC in the Mondego watershed area (Corine 2000). i) Water - Wetlands + Water bodies; ii) Agriculture - Agricultural areas; iii) Forest - Forest and semi natural areas; iv) Urban - Artificial surfaces

The soil map 1:1 000 000 was gathered from EEA data center in vector form covering the entire Mondego Watershed. This data set was first digitalized by Platou et al (1989) and further improved by Vossen & Meyer-Roux (1995)[13]. The physical-chemical parameters needed to fill the SWAT soil database were produced using pedotransfer functions based on texture (Saxton et al., 1986). Daily precipitation data were obtained for several stations in the area; only those having long near-complete time series were retained. Daily precipitation values from 1931 to 2002 were obtained for seven stations (http://snirh.pt). Monthly values of maximum and minimum temperature, solar radiation,

[13] http://dataservice.eea.europa.eu/dataservice/metadetails.asp?id=196

wind speed, and relative humidity were available for four meteorological stations for a period of 30 years (ISA, 2004): Caramulo, Penhas Douradas and Viseu. Several daily flow stations were available for model calibration. Three of them were chosen (Tábua, Mucela & Coimbra) because they had long near-complete time series (thirty to fourthly year data), which were included in the time period of precipitation data selected (http://snirh.pt).

Flow results were evaluated using MOHID Time Series Analyzer and the results are shown in Figure 34, Figure 35 for Coimbra gauge station and in Table 26 for the three gauge stations the obtained statistic parameters. To obtain these results the changes made in relation to SWAT defaults were: GW_DELAY=10, ALPHA_BF=0.5 and distribution of the precipitation stations in the sub-basins according with Isohyets.

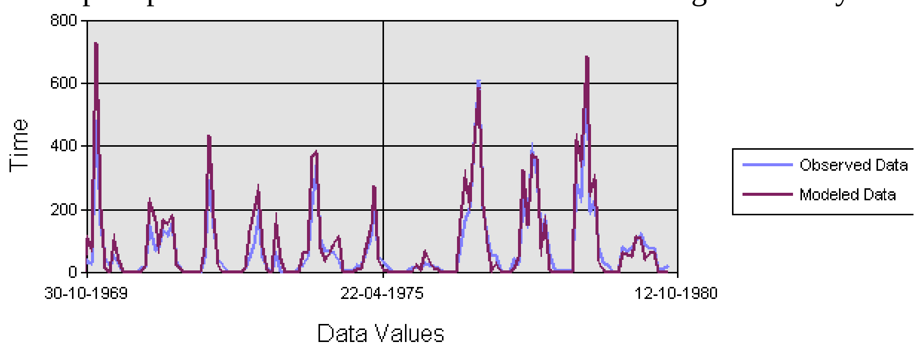

Figure 34. Monthly values of observed and modeled data in Coimbra gage station. Graph produced with MOHID Time Series Analyzer

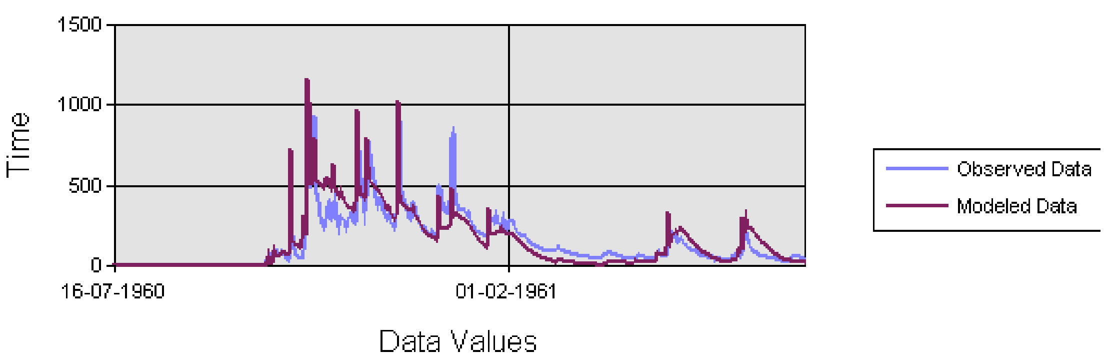

Figure 35. Daily values of observed and modeled data in Coimbra gage station. Graph produced with MOHID Time Series Analyzer.

Flows simulated show a correlation (R2) always higher than 0.69 with 30 year measurements of the three gauge stations (Table 26). This means that the general dynamics of the watershed is satisfactory. However in the case of the Tábua gauge station the model efficiency is low, basically due to an underestimation of flows. This tendency also happens in Mucela and Coimbra but the differences are smaller (Figure 34 for the case of Coimbra). The excess simulated flow can be related either to an underestimation of evapotranspiration or it can be related with the underestimation of water lost to the deep aquifer. This last value is difficult to estimate. However, the evapotranspiration estimation can be significantly improved if the soil depth is known (as well as other soil parameters). In this simulation a constant soil depth in all watershed of 1 meter was assumed.

Table 26. Evaluation of flow results after calibration for a thirty years period. Values produced with MOHID Time Series Analyzer.

Flow gage station	Coimbra		Tabua		Mucela	
	Daily	Monthly	Daily	Monthly	Daily	Monthly
RMSE - Root Mean Squared Error [m3/s]	88	51	37	25	15	9
R2 - Pearson Product-Moment Correlation Coefficient [-]	0.78	0.91	0.76	0.9	0.69	0.83
E - Model eficiency (Nash-Sutcliffe) [-]	0.69	0.82	0.05	0.27	0.52	0.68

Figure 35 shows that it is possible that base flow is being underestimated. This could also be related with an underestimation of soil retention of water or with a wrong parameterization of the aquifer in SWAT.

A wrong estimation of evapotranspiration could be a reason for wrong flow results. This could be related with a wrong input to the SWAT weather generator or with wrong estimations of this model. To analyze this, a search was made to find a long period of historical values of some climatic parameter (temperature, relative humidity, wind or radiation). A period of daily temperature between 1982 and 1990 was obtained for meteorological station of Alagoa (reference 12G_05) which is situated in the center of the watershed. These values were compared with the output of average temperature of SWAT-MOHID of the subbaisin where Alagoa meteorological station was located (Figure 36). The model efficiency for the daily values of temperature was 0.54 while for monthly values the efficiency increased to 0.8. These are considered good results taking in consideration that they were obtained from a meteorological station different from the ones used as input for SWAT weather generators.

No values of relative humidity, wind or radiation were found for long periods of historical measurements for this area.

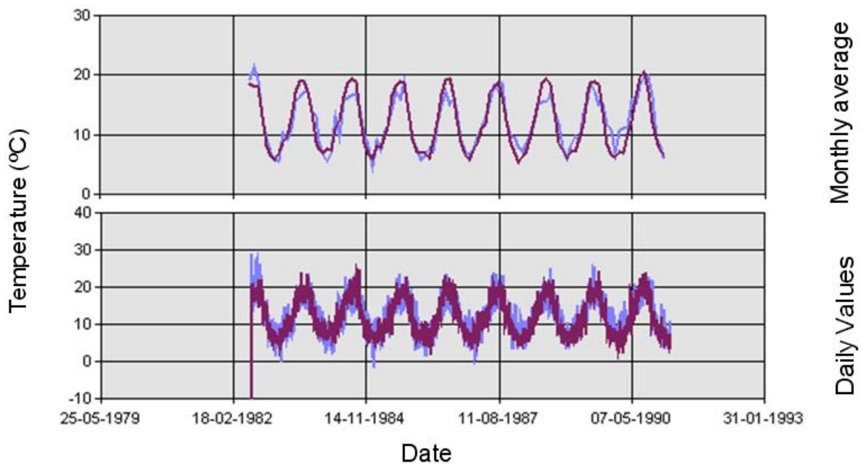

Figure 36. Monthly and Daily values of observed (blue) and modeled (reddish) data in Alagoa meteorological station. Graphs produced with MOHID Time Series Analyzer.

The next step in the application of SWAT will be to find a better soil map for the area, because the present one, though it allowed reasonable results of hydrodynamic, it is surely not enough to estimate the nutrient transport. For example, a constant value of 0.8 % of organic carbon content was assumed. Though this is a typical value in Portuguese soils, it is surely not adequate to estimate the nutrient dynamics in the soil. A second step will be to compile the most frequent agricultural practices in the area and introduce them in to the model.

In November 2006 European Soil Database V2[14] was made available with the associated maps at a scale of 1:1.000.000. This will allow producing a comprehensive map of soil parameters to make a more realistic simulation of nutrients. National soil specialists are also be contacted, to try to find better soil data. In Portugal there is very few soil maps published, and mainly in the South of Portugal (Gonçalves et al., 2005).

With a new soil map it will be possible to obtain HARP-NUT results (loads of total nitrogen and phosphorus per year) originated on anthropogenic diffuse sources and Background sources of nutrients. To estimate Background loads of nutrients the model will be run assuming that the all watershed is covered with oak tree forest.

Conclusions

MOHID and SWAT have free access source code. This has allowed the integration of some aspects of both models. This allowed the use of the advantages of each model. Current version of SWAT-MOHID allows: i) the output of time series and allows them to be compared statistically using MOHID Time Series Analyzer ii) a link between SWAT and MOHID River Network iii) storing of subbabsin and reach results in HDF format which, associated with MOHID-GIS, allows to produce animated graphic files showing the spatial and time evolution of the modeled properties.

The advantage of application of SWAT-MOHID is mainly the improved capabilities to analyze results. The aim of these developments is not to replace the use of SWAT graphical user interfaces for input or output, but to complement it. This also means that the developed outputs do not include all the SWAT variables but instead, they were developed on a need basis either for easier results exploring, results analysis, model coupling, etc. This development benefits from many current developments and improvements of MOHID tools made by its users and will continue to benefit in the future.

Flows simulated in Mondego watershed (considering both daily and monthly averages) show a correlation (R2) always higher than 0.69 with 30 year measurements of the three gage stations. However the model seems to be underestimation flows, though the simulation results in the most downstream station (Coimbra) has an monthly Efficiency of 0.82 and of 0.69 for daily results.

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[14] http://eusoils.jrc.it/ESDB_Archive/ESDBv2/index.htm

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Improved Rainfall-Runoff Modeling Combining a Semi-Distributed Model with Artificial Neural Networks

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Abstract

The research presented herein explores the simultaneous calibration of river component flows of a watershed using the integrated automatic calibration of the Soil Water Assessment Tool (SWAT) in combination with artificial neural networks, with application to the Grote Nete basin, 383 km^2 in size, located in the sandy plane region of north-eastern Belgium. For model calibration the Shuffled Complex Evolution algorithm (SCEM-UA) was applied using historical data of the period 1994-2002. The simulated daily component flows were further improved applying a multi-layer, feed forward Artificial Neural Network (ANN) trained with a Levenberg-Marquart (LM) backpropagation algorithm. The choice of appropriate ANN topologies for total flow and slow flow, in terms of hidden layers and nodes, was investigated. It was observed that the integration of the SWAT model with ANN improves significantly the model performance.

Keywords: component flows, SWAT, SCE, ANN

1. Introduction

Mathematical computer models have become indispensable for rainfall-runoff prediction. Such models are used, for example, to simulate the flow of water and the transport of sediments and chemicals in catchments in all its phases (surface runoff, soil water and groundwater). In addition they are used for flood prediction, drought mitigation, and the design of hydraulic structures.

The analysis of the rainfall-runoff process changed drastically since Sherman presented in 1932 the unit hydrograph. Various models, ranging from moving average and exponential smoothing to linear and nonlinear, lumped to distributed models have been developed, all with the same aim, i.e. to describe at catchment level the transformation of the rainfall hyetograph to the discharge hydrograph. Models represent in varying degree existing or hypothesized knowledge of the processes underlying the transformation of rainfall into river flow. According to Beck (1985) models contain either mechanistic or/and empirical descriptions of those processes, whereby the mechanistic descriptions consist in general of fundamental physical and chemical laws, and the empirical formulations in general are derived from observations or data analysis (Klemes, 1982). Recently, we notice an increasing interest in conceptual models with the aim to:

(1) synthesize the complexity of detailed numerical and experimental analyses into a general description of the governing hydrological processes, and (2) to formulate testable research questions and hypotheses which can be assessed only in terms of its instrumental and heuristic value.

Hydrological models can be classified as either lumped or distributed models. The lumped model does not explicitly take into account the spatial variability of inputs, outputs, or parameters. It considers the catchment as an undivided entity and use lumped values of input variables and parameters. The outputs of a lumped model, which are essentially averaged across the entire basin, satisfy less and less the requirements and needs of the individual stakeholders. To this end distributed models were designed to work at finer spatial and temporal scales than the one used in lumped models as to be able of determining local hydrologic behavior. The process of physical-based distributed models, initiated by Freeze and Harlan (1969), was strongly accelerated by the increasing availability of innovative techniques such as geographic information systems and satellite-based data.

In contrast, practicing hydrologists face major difficulties applying numerical models; in particular the parameter estimation remains a challenging task. Traditionally, parameters are indirectly estimated by calibrating the model output to available hydrological variables. The errors and incompleteness of hydrological information and the constraints inherent to most models are responsible that it is difficult to define a unique global parameter set (Beven and Young, 2003). The difficulties in obtaining unique global parameter estimates arise because of the presence of multiple local optima, the non-linear interaction between model parameters, and the shape and roughness of the response surface defined by the selected objective function (Feyen et al., 2005).

Optimization problems are frequently encountered in practice, with distinct relevance to many scientific and engineering applications. Most of the problems in optimization are nondeterministic polynomial hard, which means that no classical algorithm is known that solves the problem significantly faster than the exhaustive search of the domain. Researchers have focused on mathematical modeling techniques to determine optimal or near-optimal parameters of simulation models with respect to various objective criteria. Several of the modeling techniques proposed and implemented in literature are based on statistical regression (Robin, 1994), empirical formulae (Sherman, 1932), and artificial neural networks (Bishop, 1995).

The technique of artificial neural networks (ANNs) has been proven to be an efficient tool in many different fields of modeling and forecasting. Their widespread adoption is due to their ability to model highly non-linear relationships accurately without requiring explicit knowledge of the underlying system equations. In addition, they have been proven to be robust to noise and outliers (Zhang and Govindaraju, 2000). ANNs are used in numerous real world applications, such as reservoir operation (Chandramouli and Deka, 2004), rainfall-runoff modeling (Lauzon et al., 2006), the prediction and forecasting of water resource variables (Bodri and Cermak, 2000), and water quality modeling (Kuo et al., 2007).

Although several studies demonstrated that ANNs perform superior in rainfall-runoff prediction, hydrologists still have the tendency to rank artificial neural networks as black box approaches (Hsu et al., 1995). Recently hybrid models, that combine ANNs with process driven models, integrating the advantages of both, have been proposed

(Kingston et al., 2005; Chen and Adams, 2006). Within the hybrid model structure, the mechanistic model specifies the basic dynamics of the relevant process variables, whereas the neural model describes the unknowns and nonlinear parts of the mechanistic model.

The objective of the study presented in this paper is the integration of SWAT, a semi-distributed rainfall-runoff model, with an ANN operating as post-processing module.

2. Applications

2.1. Site of study and data

For the testing of the combined approach, consisting of the SWAT mathematical model and an ANN as post-processing module, data of the study basin the Grote Nete (383 km²), located in the north-eastern part of Belgium, were used. The elevation of the area varies from approximately 12 m above sea level in the western part to 68 m in the eastern part. Slopes are in general less than 1 percent. The land coverage is mainly agricultural (both pasture and cropland) with some local forested areas although a significant proportion of the area is urbanized. According to the Belgian soil map (Vander Poorten and Deckers, 1994; Vázquez, 2003), sandy soils (49.57%) are the dominant in the catchment. Light sand loamy soils and clayey soils can also be distinguished. The average rainfall in the region is 800 mm. The model was developed using the daily rainfall and flow data in the period 1 January 1998-2002 for model calibration and in the period 1 September 1994-1997 for model validation. A detailed description of the study basin is given in Rouhani et al. (2005).

Daily observations of precipitation, air temperature, evaporation, and daily streamflow data were obtained from the Royal Meteorological Institute and the Flemish Administration for Land and Water, Belgium. The soil map was available at a scale of 1:25.000; the soil physical data was derived from the Aardewerk-SIBIS Soil Information System (Van Orshoven et al., 1993); and landuse was derived from the multi-temporal LANDSAT 5 TM image of 18 July 1997.

2.2. Input data and sub-watershed delineation

The SWAT (Soil Water Assessment Tool) requires spatial information about topography, river/stream reaches, landuse, soil and climate to accurately simulate the streamflow. The climatic inputs in SWAT include daily precipitation measured in 5 stations scattered in and outside the study area, and the potential evapotranspiration and min/max temperature collected in a station at the northern boundary of the catchment. Details of input data are given in Rouhani et al. (2005). The catchment was subdivided in 8 subcatchments and 65 HRUs. The latter were created based on the various combinations of landuse and soil types present in the catchment. Climate data were assigned to each HRU using the centroid method. The daily streamflows in the Varendonk outlet station were used for model calibration and verification.

2.3. Conceptual modeling

The SWAT is a semi-distributed, conceptual model that combines spatially distributed physical attributes into hydrologic response units (HRUs), each of which, in response to meteorological inputs (such as precipitation, potential evapotranspiration) is assumed to behave in an uniform manner. The model operates in a continuous mode and has been widely used to estimate catchment runoff, nutrient and sediment loads. The SWAT model development, operation, limitations, and assumptions are extensively discussed by Arnold et al. (1998). Also the user's manual and theoretical documentation are available on the SWAT website (http://www.brc.tamus.edu/swat/index.html).

The spatial heterogeneity of the study area is represented by dividing the catchment into subbasins. Each subbasin is further discretised into a series of hydrologic response units (HRUs), which are unique landuse-soil combinations. The model represents the water balance for each HRU in four vertical water storage components as: snow, soil profile (0-2 m), shallow aquifer (2-20 m) and deep aquifer (> 20 m). Soil water content, surface runoff, nutrient cycles, sediment yield, crop growth and management practices are generated for each HRU and then aggregated for the subbasin by a weighted average and routed to the watershed outlet through channels, ponds and/or reservoirs. Surface runoff from daily rainfall is estimated using a modified SCS curve number method. Peak runoff predictions are based on a modification of the Rational Formula and groundwater flow is calculated using empirical relations. Actual soil water evaporation is estimated by using exponential functions of soil depth and water content. Plant transpiration is simulated as a linear function of potential evapotranspiration and leaf area index.

2.4. Selection of neural network architecture

An ANN operates like a 'black box' tool and does not require detailed information about the system. Instead, they learn the relationship among the input parameters, the controlled variables and the uncontrolled variables by studying previously recorded data, similar to the approach used in a non-linear regression model. Another advantage of ANNs is their ability to handle large and complex systems with many interrelated parameters (Haykin, 1994). Among the various types of ANNs used (Self Organizing Maps, RBF networks, …) the Multi-Layer feed-forward neural network trained with backpropagation (BP) is so far the most popular network topology in the field of water resources (Adeloye and Munari, 2006).

The multi-layer perceptron (MLP) is based on a multi-layered feed forward topology with supervised learning, i.e., the network constructs a model based on example data with known outputs (training). In this case the training was conducted using the fast Levenberg-Marquardt backpropagation algorithm with Bayesian regularisation. The Levenberg-Marquardt backpropagation algorithm (Hagan and Menhaj, 1994) is selected for its fast convergence and its ability to give better results than other backpropagation variants. Backpropagation works as follows: training examples are presented to the network one at a time (sequential training) or in groups (batch training). For each input vector the network calculates an output which is then compared to the known ouput value. Based on the discrepancy between the two values an error value is obtained and used to update the network weights. The error is effectively backpropagated through the network. In optimization terms, what really happens is that the backpropagation algorithm performs a gradient descent (in case of the LM algorithm, using second order information from the Hessian) of the error surface until a local optimum has been reached.

The main downside of this method is that it is very prone to converging to sub-optimal local minima. The ANN error landscape is known to be extremely multi-modal, meaning a network must be trained many times from different starting points to ensure gradient descent finds a good solution. An additional problem is the danger of overtraining (overfitting). If the network is trained too long or contains too many parameters it will perform well on the training set but poorly on unseen data (poor generalization). This is the so called Bias-Variance Tradeoff.

In order to improve generalization we therefore employed regularization during training. Another method which could have been used is early stopping. Regularization means the error function is changed by adding a penalty term proportional to the sum of the weights. This means that, besides minimizing the error, the weights are also kept small, allowing for a smoother surface. Regularization introduces a new parameter which specifies how much effort should be spent on minimizing the prediction error and how much on minimizing the weights. In order to avoid manually setting of this parameter the optimal value was selected using Bayesian theory (MacKay, 1992).

In addition, to regularization a portion of the available data was kept aside and used to judge the effective generalization of the model after training. In this study the daily discharge data from 1998-2002 was used to train the ANN model and the remaining data, 1994-1997, was reserved for testing.

The first step in developing an ANN model involves identifying input and output variables. As an additional preprocessing step the data is normalized between -1 and 1 in order to improve training. The next problem is then to determine a suitable network topology in order to find an optimal bias-variance tradeoff. In this paper, networks with up to two hidden layers were considered with a maximum of 20 units in each hidden layer. The number of units in each layer was varied linearly and the performance of each network on the test set was calculated (taking care to retrain multiple times to avoid local minima).

Our objective was to train the ANN for daily flow components with the output of SCE as the ANN inputs and observation data as the ANN targets. Simulated input signals are generated with the SWAT numerical model. The performance of the model parameters obtained from the SCE and Hybrid (SCE+ANN) technique was then evaluated by computing various standard statistical performance evaluation criteria (Rouhani et al., 2005), using both training and testing data.

3. Analysis of the result

Parameter sensitivity analysis (PSA) was applied to identify the parameters of the SWAT model that contribute most to the variability of component flows. The Latin Hypercube and One-factor-At-a-Time (LH-OAT) sensitivity analysis yielded the 5 most sensitive parameters (Rouhani et al., 2006). Those parameters were optimized in the calibration process using a multi-automatic calibration scheme, based on the Shuffled Complex Evolution algorithm (SCE-UA; Duan et al., 1992). The SCE-UA global search procedure is based on the downhill simplex method, combined with a random search procedure and the idea of complex shuffling. However, to reduce the chance of premature convergence of the algorithm, it has been suggested by Kuczera (1997) that the number of complexes (p) be set equal to the number of parameters to be optimized. Therefore p was set equal to 5.

The optimal size of the hidden layer was found by systematically increasing the number of hidden neurons until the network performance on the test set no longer improved significantly. Comparing the performance of network configurations, we found that a network with two hidden layers and 10 neurons gave the best results for both total flow and slow flow (a different network was used in each case).

The simulation results obtained with the SCE and hybrid (SCE + Neural network) method, using eight year daily total flow data measured at the Varendonk outlet station of the Grote Nete catchment, are plotted versus the observed flow data in Figs. 1 (calibration period) and 2 (validation period). A comparison of daily total flow reveals that the magnitude and trend in the predicted stream flows obtained by both methods agree well with the measured data.

The two hydrographs show acceptable agreement, although the simulated winter flows appear to be more accurately predicted than the simulated summer flows in the validation period. Simulated peak flows are reasonably consistent with measured peaks. However on some events SWAT simulated peak flows show overestimation. Only small differences can be seen: e.g. the hybrid method is slightly closer to the observations in validation period (1994-97); also the hybrid method seems to fit parts of the drier periods better.

Analyzing the figure more closely reveals that the hybrid method, regardless the period, seems to match parts of the slow flow and recessions more accurately than the SCE optimization alone. The results in terms of various standard statistical performance evaluation criteria for the daily total and slow flows are presented in Table 1. The performance of the model was evaluated using the root mean square error (RMSE), the coefficient of determination (R^2), the Nash-Sutcliffe simulation efficiency (E) (Nash and Sutcliffe, 1970) and the bias. No significant difference between both methods, using the data of the calibration period, was found in simulated daily total and slow flow. The hybrid method seems to perform slightly better than the SCE approach. In contrast, the performance of the hybrid model was found to significantly improve the daily total and slow flow prediction during the validation period. As can be seen in Table 1, the RMSE of 1.38 and 1.11 using the SCE method decreased to 0.88 and 0.76 using the hybrid method for daily total and slow flow, respectively. Similarly, the EF value of 0.69 and 0.63 using the SCE technique increased to 0.76 and 83 applying the hybrid method. The performance of the hybrid method was consistently superior to that of the SCE in terms of the other statistical criteria as well.

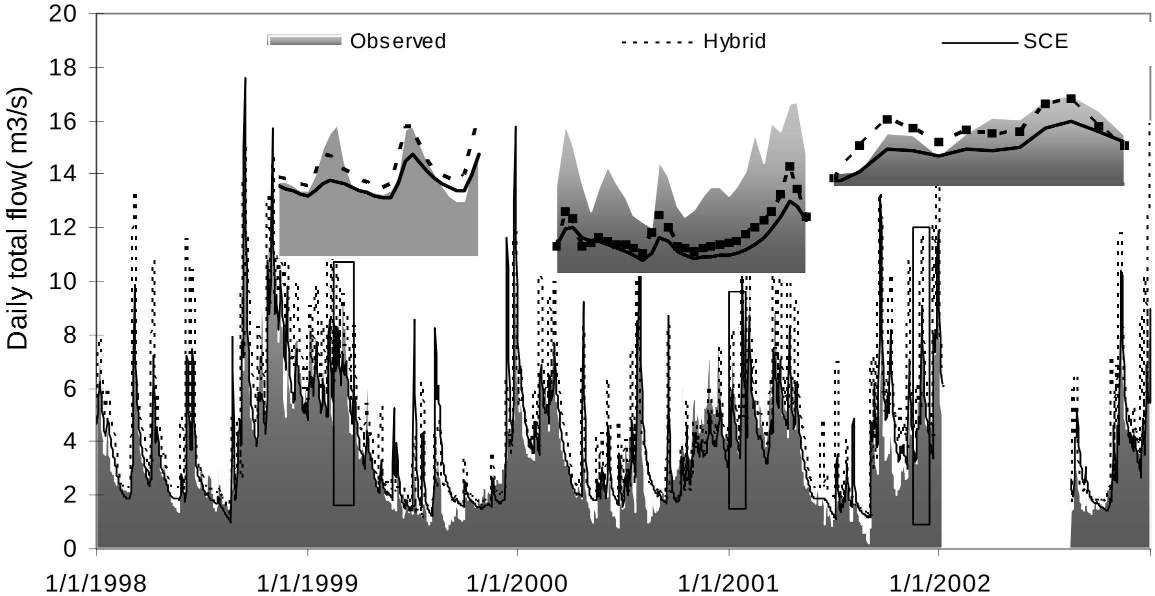

Figure 1. Comparison of observed and simulated daily total flows at Varendonk outlet during the model calibration (1998-2002), for observed, hybrid and SCE methods.

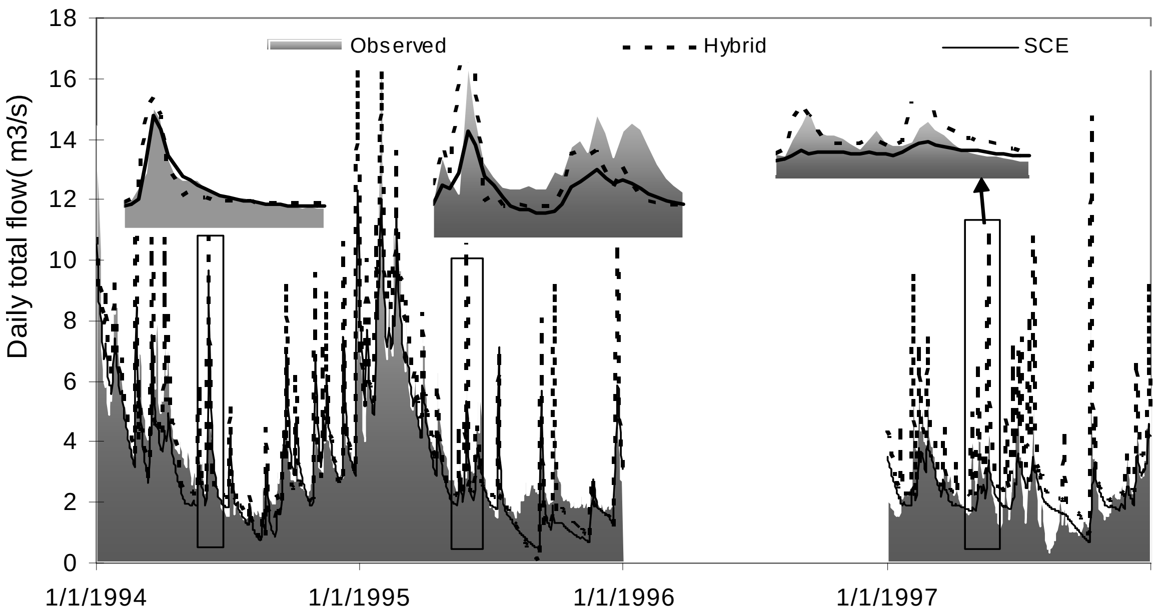

Figure 2. Comparison of observed and simulated daily total flows at Varendonk outlet during the model calibration (1994-1997), for observed, hybrid and SCE methods.

Results in Table 2 indicate that the estimated runoff volumes by the hybrid method are very close to the observed values. The simulated daily total flow using the SCE method during calibration and validation were estimated to be 3.96 $m^3 s^{-1}$ with bias of -10% and 3.06 $m^3 s^{-1}$ with bias of -19%, respectively. The simulated daily slow flows using the SCE method were underestimated by 10 and 20% during the calibration and validation period, respectively.

Table 1. Summary of the statistics for the daily total water and slow flows in the calibration and validation periods.

Statistical criteria	Average daily total water flow ($m^3 s^{-1}$)				Average daily slow flow ($m^3 s^{-1}$)			
	Calibration		Validation		Calibration		Validation	
	SCE	Hybrid	SCE	Hybrid	SCE	Hybrid	SCE	Hybrid
RMSE	1.45	1.33	1.38	0.88	0.70	0.74	1.11	0.76
EF	0.80	0.83	0.69	0.76	0.89	0.88	0.63	0.83
R^2	0.80	0.81	0.88	0.90	0.81	0.81	0.87	0.91
Bias	-10%	0%	-19%	0%	-10%	0%	-20%	-4%

Table 2. Comparisons between measured and predicted average daily total water and slow flow.

Period	Average daily total water flow ($m^3 s^{-1}$)			Average daily slow flow ($m^3 s^{-1}$)		
	Observed	SCE	Hybrid	Observed	SCE	Hybrid
Calibration	4.39	3.96	4.39	3.36	3.02	3.36
Validation	3.75	3.06	3.75	3.34	2.68	3.21

4. Conclusions

Two automatic hybrid calibration methods were developed in order to improve the daily component flows generated by the SWAT model using the hydrological data of a medium size catchment, situated in the north-eastern part of Belgium. A feed forward artificial neural network with two hidden layers was trained on the output result of the SCE method using historic data and a fast backpropagation algorithm. Application results of the validation period indicate a significant improvement in both daily total and slow flow moving from the SCE to the hybrid method. The results obtained in this study indicate that the hybrid technique is suitable for the purpose of rainfall-runoff simulation.

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Uncertainties in calibrating SWAT for a semi-arid catchment in NSW (Australia)

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Abstract

Management of catchment scale processes often relies on predictions of large scale hydrological models to study “what-if” scenarios. Large scale models generally need calibration to match parameters to observed values as the models contain a large number of parameters. It is then interesting to study how the underlying conceptual models (determining the choice of calibration parameters) might influence the final calibration. This is particularly an issue in semi-arid catchments as the hydrology is much less understood and calibration is more difficult, even more so if the model outcomes are subsequently used for an economic or social study. SWAT 2000 was calibrated to 10 years of streamflow data from three stations in a 4500 km^2 semi-arid catchment in NSW (Australia). Different conceptual models of the groundwater surface water interaction were calibrated, all with similar calibration results. Overall calibration efficiency was low due to major difficulties in representing the semi-arid system. This was also compared to a much simpler lumped conceptual model again with similar calibration results. The implications of the lack of difference in the calibration results is that this does not allow separation of the conceptual models, all of which would have different consequences for the planned socio-economic modelling of policies to control recharge. This is troublesome as the conceptual parameters in the groundwater module of SWAT are also not easily verified by field measurements. Possibly using a distribution of calibration outcomes would deliver the best input into a socio-economic analysis.

KEYWORDS: Semi-arid hydrology, SWAT, calibration, uncertainty

Introduction

Large scale hydrological models are important tools in catchment management as they can be used for what-if scenarios for policy development and economic analysis. One of the problems with large scale models is that they need to be based on credible conceptual models (Beven 2001), but that each of these conceptual models include a series of assumptions. This means there needs to be a reasonable understanding of the flow processes in a catchment. In particular in semi-arid areas, in which flow processes are often flashy, unclear and difficult to describe and the forcing variable, rainfall, can be highly spatially variable this assumption might not be true (Beven 2002; Pilgrim et al. 1988).

The Soil Water Assessment Tool (SWAT) is essentially a semi-distributed surface water model. The hydrological response units (HRUs), based on a combination of landuse and soil properties, allow a spatial representation of the runoff processes. Groundwater is represented conceptually by two interconnected “buckets”, one of which can interact with the surface water system (Arnold and Fohrer 2005). The spatial representation of the catchment properties is important for economic analysis of trade-offs in human and environmental needs beyond the "representative farm". The biophysical model can be used to produce spatially distributed production functions which can be used in an economical model to identify trade offs in both time and space. An example of this is spatially targeting environmental policies to reduce deep drainage, which, in Australia, is seen as the cause of dryland salinity. As deep drainage is not spatially homogeneous, including spatial variability is important and SWAT is able to deliver outcomes at a spatially detailed level. However, the increase in spatial representation also increases the number of model parameters. With increasing model complexity and spatial detail there is a trade-off as input uncertainty will increase with decreasing model uncertainty (i.e. is the model parsimonious?) (Jakeman and Hornberger 1993; Silberstein 2006).

Because most large scale models include a high level of conceptual interpretation of the catchment, parameters are basically "effective biophysical representations" (Grayson et al. 1992). As a result and similar to all large scale catchment models, calibration of SWAT is needed to equate observed flows to predicted flows, either by hand or using some sort of automated procedure as in the parameter estimation model PEST (Doherty and Johnston 2003). In semi-arid basins calibration can be particularly difficult due to the large number of zero flows, and due to the gaps in understanding of the physical processes (Pilgrim et al. 1988; Ye et al. 1997). During calibration of SWAT in semi-arid basins the shallow groundwater "bucket" was made very small, meaning that the flow into the river was very rapid (Sun and Cornish 2005; Van Liew and Garbrech 2003). This would mimick a losing river over a disconnected groundwater system with some delayed runoff occurring via some form of subsurface lateral flow. However the process might be more complex as during wet periods, the groundwater can also become connected and contribute to flows, thus behaving as "ephemeral baseflow" (Rassam et al. 2006). Similar to the conceptual definition of the hydrological system, landuse can also only be defined at a conceptual level. In Australia, this means that "dryland cropping" can contain anything from a sorghum or wheat rotation to growing of chick peas to capture summer rains. In addition, the rotations from different farmers might not be synchronised.

This paper aims to differences in hydrological outcomes using SWAT on a large scale semi-arid catchment in northwest NSW, Australia as a pre-cursor to economic modelling. In particular the effect of different conceptual hydrological models on calibrated hydrological outcomes using the automatic parameter estimation model PEST (Doherty and Johnston 2003) are investigated. Differences in hydrological outcomes would have major implications for economic and water quality evaluations based on these model outcomes.

Methodology

The Mooki catchment is approximately 4500 km^2 in northwestern NSW, Australia. The soils in the area are mainly heavy clays in the Vertosol and Chromosol classes (Isbell 1996). The catchment is part of the Liverpool plains area, which is considered one of the most productive agricultural areas of Australia. It is therefore relatively intensively cropped with several grain crops and more recently irrigated cotton and lucerne. The landscape is very flat, with only the sides of the catchment having significant relief.

Much of the research interest in the catchment has been due to the occurrence of dryland salinity and its relationship to the use of European farming techniques (Abbs and Littleboy 1998). Several modelling studies have been undertaken in the area (Abbs and Littleboy 1998; Sun and Cornish 2005) to estimate recharge and this provides some background information for this study. The older study has used a 1-D point model to investigate deep drainage and the effect on dryland salinity risks, while the later study used SWAT on a much smaller subcatchment.

The time period for this study is from 1994 – 2003, partly to capture the increasing influence of the irrigated cotton industry, which is important for the economic study which will build on the results of the hydrological model, and partly because of the streamflow record. Rainfall data from 7 weather stations in the area and temperature and ET data for two of the stations was used to drive the climate model. Landuse was mainly based on the 1999 NSW Department of Natural Resources survey, while soils were based on soil landscape units in the area. Because the landuse data did not have detailed data for dryland cropping, typical wheat and grain sorghum rotations were assigned randomly to half of the HRU's. As the curve number approach used in SWAT is more sensitive to landuse than to soil type (NRCS 2004), this was deemed sufficient. Most of the soils in the area are in the "C" and "D" classes of the curve number method. The DEM is derived from the 1:25,000 DEM for the area, which is the best avialable. These spatial layers resulted in a model with 32 subcatchments and 732 HRUs based on an 8000 ha threshold and set to a minimum of 10% for landuse and soil type (Fig. 1). Currently a SWAT 2005 version of the model is being calibrated.

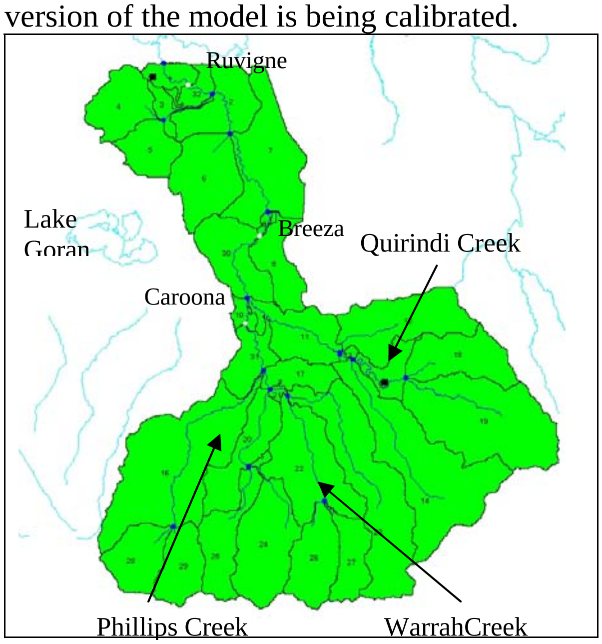

Figure 37 Overview of the catchment indicating 32 subcatchments resulting in 688 HRU's. The streamflow measuring stations at Ruvigne, Breeza and Caroona are indicated, as well as the main creeks in the upper catchment.

Daily stream flow data for the period is available for 3 gauging stations in the area (Ruvigne, Breeza and Caroona). The data cannot be split easily in a calibration and validation set as several large floods occurred in 1998, followed by a prolonged drought from 2000. No long-term data are available for the Quirindi creek inflows below Caroona (Fig. 1). Overall around 3500 calibration points were available for each station. Extraction of water from the streams by irrigators is limited by the water sharing rules established for the area (DIPNR 2004). These rules have been included in the model based on the broad catchments within the model. A complication is that the irrigators mainly extract water during high flows and use large storage dams to store the water until needed for irrigation. Most irrigators also extract both groundwater and surface water with preference for the latter due to quality and lower pumping costs. The groundwater supply is however more reliable. This is not a system which can currently be modelled in SWAT and could influence the outcomes of the model.

The automatic parameter estimation model PEST was used to calibrate the model over the time period. Calibration was based on the log(flow + 1) values and weights were assigned to the observations to give extra weight to the low flows. Baseflow parameters were derived from the data at each flow gauging station and assumed to be unchanged over the period. Calibration parameters included curve numbers for the different landuses, the limit for baseflow from the shallow groundwater store, channel parameters such as Manning's n and hydraulic conductivity (Table 1). Again most of these parameters were split by stream gauging station to allow more flexibility in the calibration. The curve numbers were related to each other using the module PAR2PAR forcing all curve numbers to be linked to the curve number with maximum runoff (pasture on "D" soils).

As there has been considerable work suggesting that there is insufficient information in the stream flow timeseries and that a low parameter model should be used (Ye et al. 1997), the 8 parameter catchment moisture deficit module of IHACRES (Croke and Jakeman 2004; Evans and Jakeman 1998) was used to also calibrate streamflow at the Ruvigne, Breeza and Caroona stations using the "optim" procedure in R (R Development Core Team 2006) to maximise the Nash Sutclife Efficiency (NSE). The NSE is defined as:

$$NSE = 1 - \frac{\sigma_e^2}{\sigma_o^2}$$

Where σ_e^2 is the variance of the error and σ_o^2 is the variance of the observations. As the parameters in the model are relatively correlated, only the fractions of fast and quick flow and delays and the parameter converting temperature to evapotranspiration (e) were calibrated (Croke and Jakeman, 2004). The remaining parameters were set to the suggested values in Croke and Jakeman (2004).

Table 27 Calibration parameters for the different calibration scenarios in SWAT 2000

Parameters	**SWAT 2000**
Curvenumbers	10 different soil and landuses
REVAP	4 subcatchments
Transmission K	4 subcatchments
Mannings n	4 subcatchments
GWQMN (GW threshold)	4 subcatchments
EPCO	10 soils

Table 28 Overview of the parameters varied in the different calibration runs to represent the different conceptual models of groundwater and surface water interaction.

Run	**SHALLST**	**RCHRG_**	**REVAPMIN**	**GWQMN**	**ESCO**

		DP			
Run 1	500	0.5	200	500	0.95
Run 2	500	0.01	500	Calibrated	0.95
Run 3	500	0.1	500	Calibrated	0.95
Run 4	1	0.1	10	1	0.95
Run 5	500	0.1	510, Calibrated, tied to GWQMN	500, Calibrated	0.001
Run 6	500	0.25	510, Calibrated, tied to GWQMN	500, Calibrated	0.95
Run 7	500	0.25	300, Calibrated, tied to GWQMN	600, Calibrated	0.95

Seven different conceptual models were incorporated in the calibration attempts for the SWAT 2000 model (Table 2): Large shallow groundwater store, high leakage to deep groundwater; large shallow groundwater store, low leakage to deep groundwater; small shallow groundwater store, high leakage to deep groundwater; and additional models depending on the calibrated values of the parameters. The output of the different calibrations was compared in terms of the correlation between observed and predicted values, as well as using the NSE.

Results and Discussion

Calibration results for both runs clearly indicated the strong non-linearity of the hydrological response in the catchment (Fig. 2). While only two runs are presented in Fig 2, the pattern was similar for all runs. The calibrated model in general underestimates the peak runoff and over predicts many of the lower flows and some of the smaller peaks. The PEST model calibration generally decreased transmission losses (by lowering the hydraulic conductivity in the reach file) to adjust for the under prediction of the peak flows, but this of course results in an over prediction of the low flows. Similarly, IHACRES will adjust the effective rainfall up to match the peaks. Some of the over estimation on the recession curve could be due to underestimation of the extraction of water by irrigators, since the timing of the extraction is difficult to assess due to the use of storage tanks. The irrigators could also be using groundwater. In a future model a random assignment of both types might assist in the calibration. In contrast, Sun and Cornish (2005) generally overestimated the annual runoff, however the authors do not supply any data on the goodness of fit of their results.

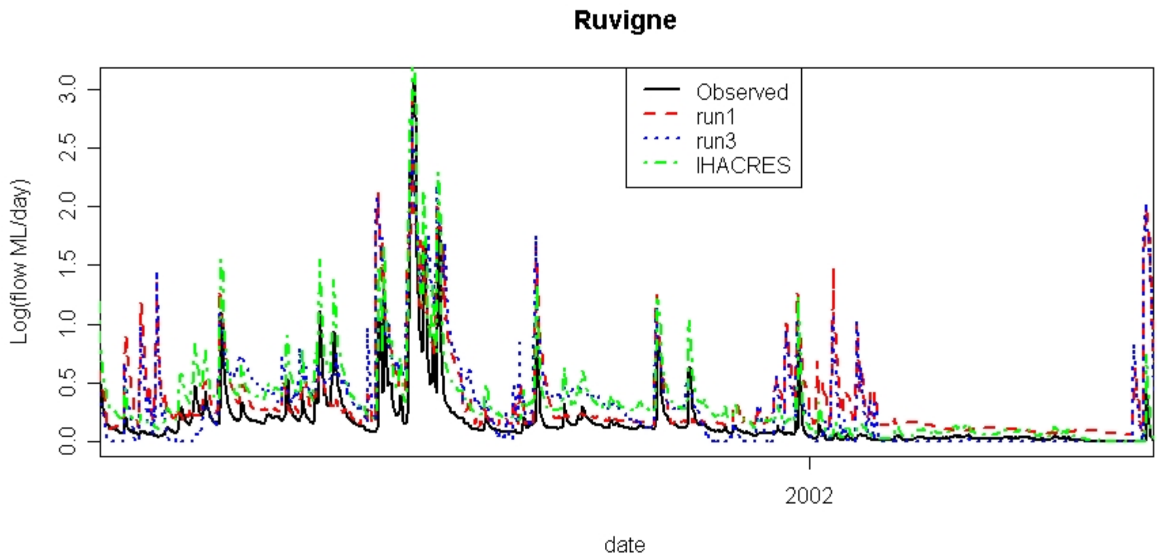

Figure 38 Example calibration results for run 3 and 7. The results clearly indicate the over prediction of recessions and under prediction of the peaks. Note the log scale, this is log(flow+1) in ML/day to be able to plot the zero flows.

The under prediction of the peak flows can have several other reasons, which are more related to the specific behaviour of semi-arid hydrology. Firstly, transmission losses are often dynamic (Dunkerley and Brown 1999; Lange 2005), with large losses occurring during low flows and small floods and much lower losses occurring during large floods (only in the floodplain). In addition, during the flood recession, transmission losses might actually be negative as water might be added to the river from floodplain storage (Rassam et al. 2006). In SWAT the routing and reach file parameters in the model are however static and apply for the whole 10 years. The question is whether this dynamic behaviour is somehow accounted for in a different way by the model structure.

Secondly, a neighbouring catchment on the West side with a terminal lake (Lake Goran) is generally disconnected, but at very high rainfall events and floods (such as in 1998), there is the possibility of the two catchments being connected, either through surface flow, or through subsurface flow. This is also a common feature of many semi-arid catchments in flat landscapes, such as the Mooki. This means that the catchment area might be somewhat dynamic and this might explain the underestimation of the larger flows.

Thirdly, the uncertainty in the actual landuse patterns might play a significant role, this will be investigated in future research. Lastly, the number of no-flows (0) in the observation set might also make it difficult for PEST to actually calibrate the groundwater parameters, as there are insufficient low flow observations even in the 10 year record. The groundwater store in the SWAT model is a “bucket”, and this means that the baseflow generation might not capture the dynamic nature of ephemeral groundwater behaviour. The threshold for groundwater flow was calibrated to try and mimic the ephemeral behaviour rather than making the groundwater bucket very small (Sun and Cornish 2005). However this only restricts water flowing out, not water flowing in as transmission losses (as was raised in the first point).

All the Nash-Sutfcliffe efficiencies and r^2 values for all model calibrations were relatively low, reflecting the lack of calibration of the model (Table 3). The results indicate that the calibrations are different at the different gauging stations, with the Caroona (the most upstream station) having the best calibration results overall. In contrast, the first two runs calibrated somewhat better at the downstream station Ruvigne and less at the upstream stations Breeza and Caroona. Interestingly however, looking at the "average" r-squared values and the NSE values between the SWAT 2000 model runs, there appeared to be little difference, with only the Ruvigne station performing poorly. This might be indicating that these conceptual models are somewhat equivalent. Given the uncertainty in representing the processes in the conceptual model, this result is troublesome. It means that the calibration results are not a good discriminator between conceptual models. For water resource predictions at the downstream end of the river, this might be not such a problem, as a more lumped model (e.g. Croke and Jakeman 2001) might also suffice, and the IHACRES model gave similar correlations and NSE values (Table 3). However for the coupling of a biophysical model to an economic model including spatial analysis, which was the underlying goal of the broader study, this result is a problem. It is similarly a problem from a water quality perspective. In both these cases the actual pathway and underlying conceptual model have a direct influence on the actual outcomes of the economic or water quality analysis.

Table 29 Nash Sutcliffe (N-S) efficiencies and r^2 for the comparison of predicted versus observed values at the different streamflow measuring stations and for the calibration runs.

Station	Ruvigne		Breeza		Caroona	
Model run (Table 2)	NSE	r^2	NSE	r^2	NSE	r^2
run 1	0.25	0.52	0.26	0.37	0.16	0.38
run 2	0.12	0.48	0.34	0.48	0.34	0.44
run 3	0.09	0.54	0.35	0.49	0.40	0.55
run 4	-0.13	0.46	0.24	0.44	0.17	0.46
run 5	-0.10	0.47	0.23	0.43	0.23	0.48
run 6	-0.24	0.38	0.15	0.42	0.14	0.44
run 7	-0.11	0.49	0.28	0.48	0.29	0.53
IHACRES	0.40	0.40	0.38	0.38	0.39	0.39

The cause of this similarity between the runs is the fact that the parameters which define the groundwater system are correlated and changing one will have implications for the others. For example, a high leakage to the deep groundwater will result in a lower REVAP and GWQMN etc. This means that without independent measurements of these parameters, the uncertainty in many of the processes and the flexibility of the SWAT model coupled with calibration results in a range of equivalent outcomes, each of which has different implications for the subsequent economic modelling.

How do we manage this problem? And is this a problem? There are several different approaches to managing this issue. One way would be by developing a better understanding of the system under investigation to decrease the uncertainty. This can be seen as the traditional scientific deterministic approach, in which further knowledge and detail should allow a better model. This approach has however always been challenged (i.e. Beven 2001; Grayson et al. 1992), as more knowledge does not always decrease the uncertainty and does not make the model transferable to different environments. It is also somewhat impossible to actually measure the parameters, as these represent a "conceptual" groundwater system. Calibrating the overall model to additional economic or water quality measures would possibly also fall in this category, but might work better. Another approach could be to quantify the uncertainty (for example using regularisation in PEST) and investigating whether the different conceptual models have different levels of uncertainty in the economic or water quality outcomes. It is possible that this would allow discrimination between the different conceptual models.

A final approach is to consider all models as components of an equifinality framework (Beven 2002). In this case all outcomes are valid (if not rejected for normal reasons) and the family of outcomes should reflect the "true" behaviour of the catchment, for example through some sort of Bayesian average (Vrugt and Robinson 2007) (Fig. 3). Again this means that there is also a family of economic and water quality outcomes, some of which might be rejected through standard scientific reasoning.

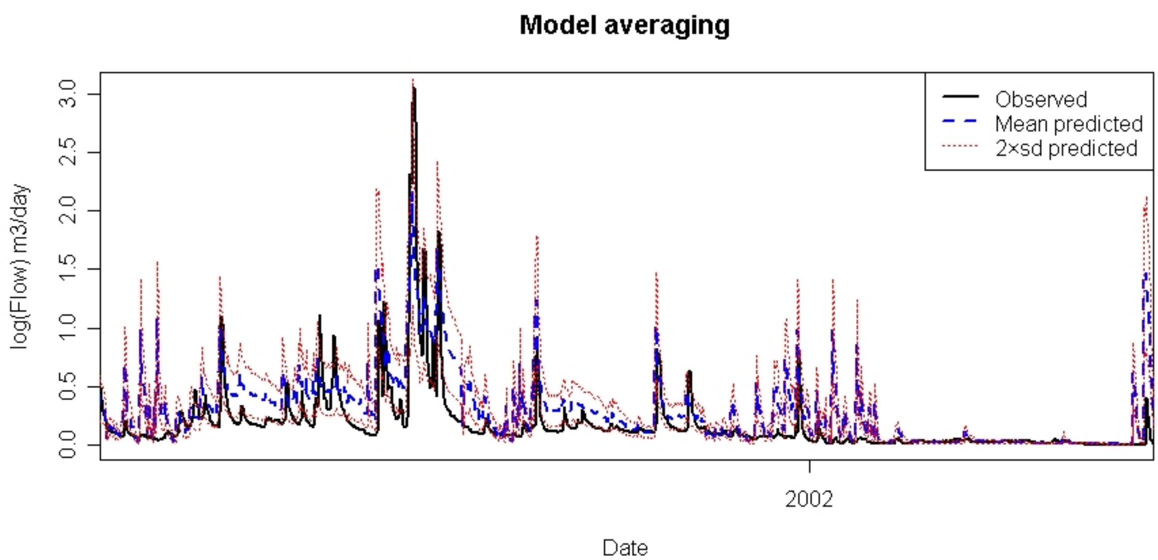

Figure 39 Average and 95% confidence interval of all SWAT predictions. Note that the mean predicted and confidence intervals (plotted as 2 × standard deviations) are well outside the observed values, clearly indicating the difficulty SWAT has in modelling this system.

Conclusions

This study is far from complete and the calibration of the model for the Mooki catchment can still be improved based on a better (or different) understanding of the processes and possible calibration of different parameters. The semi-arid system itself provides some significant difficulties in modelling using SWAT. However, the point of this paper is that automatic calibration and reasonable efficiencies do not necessarily validate the conceptual model. As this paper shows, different conceptual models can have similar calibration results. For stand alone water resource predictions in a catchment, this is not such a problem, but for further modelling which is based on the outcomes of the water quantity modelling this is a major problem.

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CALIBRATION and Validation of SWAT2005/ArcSWAT in Anjeni Gauged Watershed, Northern Highlands of Ethiopia

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Abstract

Poor land use practices and improper management system had played an important role for causing high soil erosion rates, sediment depositions and loss of agricultural nutrients of the soil in the Ethiopian highlands. Limited measures are taken to combat the surface erosion and sedimentation/pollution problems. To improve the situation decision making tools are needed for the assessment of the hydrology, soil erosion and sedimentation processes. The main study objective was to assess the impacts of land management practices on the surface runoff in Anjeni gauged watershed, Northern highlands of Ethiopia. For this purpose a spatially distributed river basin model, the Soil and Water Assessment Tool (SWAT2005) was used. A 2m by 2m grid DEM, Land use and Soil layers, ten years climatic and stream flow data were used for the delineation and simulation of the hydrology of the watershed. Sensitivity analysis was done to identify the most sensitive flow parameters for the specific landuse and agro-climatic condition of the Anjeni watershed. These sensitive model parameters were adjusted within their allowable ranges during calibration to optimize model prediction. The model was calibrated using eight years hydrometric measurements, from 01 January 1984 to 31 December 1991. Validation of the model was also done with independent measured stream flow data from 01 January 1992 to 31 December 1993. The model performance evaluation statistics such as Nash–Sutcliffe model efficiency (NS>0.91) and coefficient of determination (R^2>0.92) showed that the model can produce reasonable estimates of monthly discharge. The study showed that the SWAT model is a useful modeling tool for analyzing the hydrological processes. It can be used to design appropriate land and water resources conservation strategies.

KEYWORDS: ArcSWAT, Calibration; Hydrology; Modeling; SWAT; Validation,

Introduction

Land degradation and soil erosion is a major problem in Ethiopian highlands. Poor land use practices and improper management systems have played a major role for causing the high soil erosion rates, sediment depositions and loss of agricultural nutrients of the soil in the Ethiopian highlands. Little effective measures are taken to combat the surface erosion and pollution problems. To improve the situation tools are needed for the better assessment of the hydrology, soil erosion and sediment transport processes as well as planning and implementations of appropriate measures. The tools concern various hydrological and soil erosion models.

The focus of hydrological models is to establish a relationship between various hydrological components such as precipitation, evapotranspiration, surface runoff, ground water flow, soil water movement (infiltration) and so on. Models range from simple unit hydrograph based models to more complex models that are based on the fully dynamic flow equations. To make reliable prediction of flows accurate representations of the hydrologic processes occurring in the system are needed. An effective way to improve the accuracy is to use spatially distributed models. Physically based distributed watershed models play a major role in analyzing the impact of land management practices on water, sediment, and agricultural chemical yields in large complex watersheds. SWAT (Soil and Water Assessment Tool) model is one of the appropriate watershed models for long-term impact analysis. It is widely applied in many parts of United States (Bingner 1996, Peterson and Hamlett 1998; Srinivasan et al. 1998; Arnold et al. 1998; Neitsch et al. 2001; Benaman et al. (2005) and many other countries (Dilnesaw 2006; Heuvelmans et al 2004; Bouraoui 2005) .

The ability of a watershed model to sufficiently predict stream flow and sediment yield is evaluated through sensitivity analysis, model calibration, and model validation. The sensitive parameters are further used to find the most reasonable parameter values for optimum estimations of the stream flows.

Model outputs should be calibrated to fall within an acceptable range of model performance statistics, coefficient of correlation (R^2) and Nash-Sutcliffe Coefficient (NS) (Nash and Sutcliffe 1970). Santhi et. al., (2001) assumed an acceptable calibration for hydrology is $R^2 > 0.6$ and NS > 0.5. Kati and Indrajeet (2005) summarized the flow calibration performance statistics of studies conducted by different researchers as listed in Table 1.

Table 1. Summary of Monthly Calibrations Performed on SWAT Models with their respective Statistic of Measurements: R^2 and NS; Kati L.W and Indrajeet C. (2005)

Reference	*Base flow*	*Runoff-Flow*	*Total Flow*	*Sediment*
Arnold and Allen (1996)	(0.38 to 0.51)	(0.79 to 0.94)	(0.63 to 0.95)	
Arnold and Allen (1996)	(0.62 to 0.98)			
Arnold et al. (2000)			(0.63)	
Spruill et al. (2000)			0.58, 0.89	
Santhi et al. (2001b)			0.79, 0.83	0.8, 0.69
			(0.80, 0.89)	(0.81, 0.87)
Cotter (2002)			0.76(0.77)	0.50(0.69)
Hanratty and Stefan (1998)			0.78	0.59
Luzio et al. (2002)			0.82	0.78
Tripathi et al.. (2003)			0.98(0.97)	0.79(0.89)
Srinivasan et al. (1998)			0.77, 0.84	
			(0.87, 0.84)	
Srinivasan and Arnold (1994)			0.86	

Dilnesaw (2006) conducted monthly flow calibration in different sub watersheds of Awash river basin of Ethiopia (Table 2). According to Benaman et al. (2005) study in Cannonsville reservoir watershed, Upstate New York, the hydrological calibration and validation results using SWAT displayed an NS of 0.63 to 0.78 and R^2 of 0.72 to 0.80.

Table 2. SWAT model calibration and Validation statistics for monthly stream flow in Awash river basin, Ethiopia Dilesaw (2006).

Watersheds	Parameters	Nash-Sucrcliffe	R^2 coefficient

Berga	Calibrated	0.8	0.87
	Validated	0.78	0.84
Melkakuntere	Calibrated	0.83	0.83
	Validated	0.77	0.78
Hombole	Calibrated	0.87	0.87
	Validated	0.79	0.79

In the present study the SWAT model (SWAT2005 with ArcSWAT interface) is applied to Anjeni gauged watershed area. The main objectives were to evaluate the applicability and performance of the model in predicting the monthly water yield and to study the long- term impacts of agricultural management practices on water yield in Lake Tana watershed of Blue Nile River basin. To achieve the objectives sensitivity analysis, calibration and validation of the model were essential steps for testing the model as well as extending the application area. The ultimate aim of the study is to up-scale the model to Lake Tana watershed of Blue Nile river basin, Ethiopia which is located in a similar agro-climatical zone.

Materials and Methods

SWAT Model

SWAT is a river basin, or watershed scale model developed to predict the impact of land management practices on water, sediment and agricultural chemical yields in large complex watersheds with varying soils, land use and management conditions over long periods of time (Neitsch et al., 2002). It is a public domain distributed model developed and actively supported by the USDA Agricultural Research Service at the Grassland, Soil and Water Research Laboratory in Temple, Texas, USA. SWAT uses hydrologic response units (HRUs) to describe spatial heterogeneity in terms of land cover and soil type within a watershed. The model estimates relevant hydrologic components such as evapotranspiration, surface runoff, groundwater flow and sediment yield at each HRUs. SWAT requires specific information about weather, topography, vegetation, land management and soil properties. The details of input data needed for SWAT model are listed in the SWAT input/output file documentation (Neitsch S.L. et al, 2005). Currently, SWAT is imbedded in an ArcGIS interface.

Simulation of the hydrology of a watershed is separated into two major divisions. One is the land phase of the hydrological cycle that controls the amount of water, sediment, nutrient and pesticide loadings to the main channel in each subbasin, the second is routing phase of the hydrologic cycle that can be defined as the movement of water, sediments through the channel network of the watershed to the outlet. Runoff is predicted separately for each HRU using SCS curve number method and routed to obtain the total runoff for the watershed. Potential evapotranspiration was estimated using the Penman Monteith equation.

The watershed model SWAT gives an option to perform automatic calibration using an optimization algorithm. The automatic calibration procedure is based on the Shuffled Complex Evolution algorithm developed at the University of Arizona (SCE-UA). It is applied with success to SWAT for hydrologic parameters (Eckhardt and Arnold, 2001) and hydrologic and water quality parameters (van Griensven et al., 2002).

Model Application

Watershed Description

Anjeni gauged watershed is situated in 37°31’E / 10°40’N, in the Northern part of Ethiopia. Its altitude ranges from 2407 - 2507 m above sea level. Hydrological catchment area is 113.4 ha. Mean annual rainfall and temperature is 1690 mm and 16°C, respectively (Figure1). The watershed was established by the Soil Conservation Research Programme (SCRP) in 1981 with the support of the Swiss Agency for Development and Cooperation (SDC). The land is highly cultivated with different field crops and vegetables.

Model Input

The dataset used for this study was obtained from Soil Conservation Research Programme, University of Bern, Switzerland and SCRP Project office, Addis Ababa, Ethiopia. GIS input files used for the application of SWAT model include the digital elevation model (DEM), landuse, and soil layers (Figure 3). The DEM utilized by ArcSWAT to delineate watershed and subbasin boundaries, calculate subbasin average slopes and delineate the stream network.

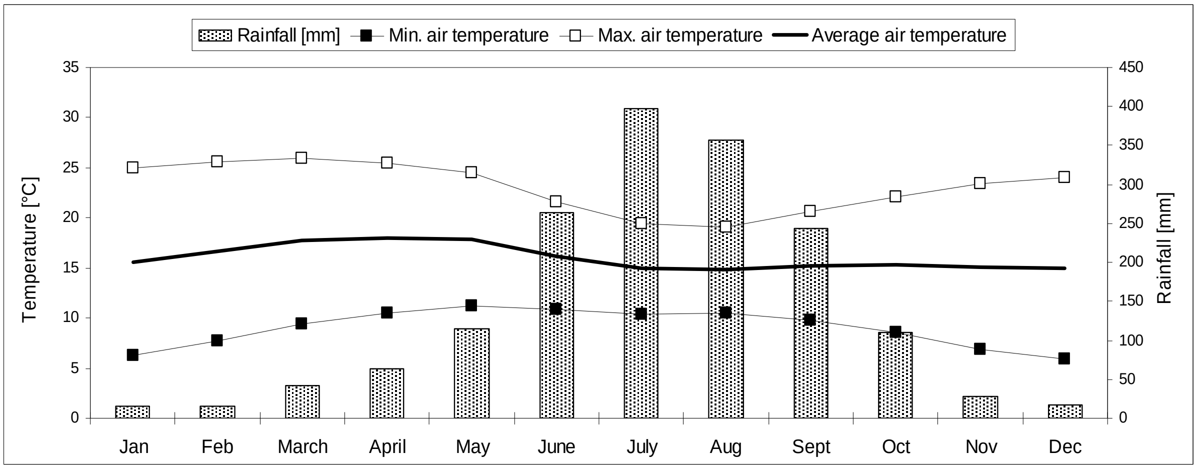
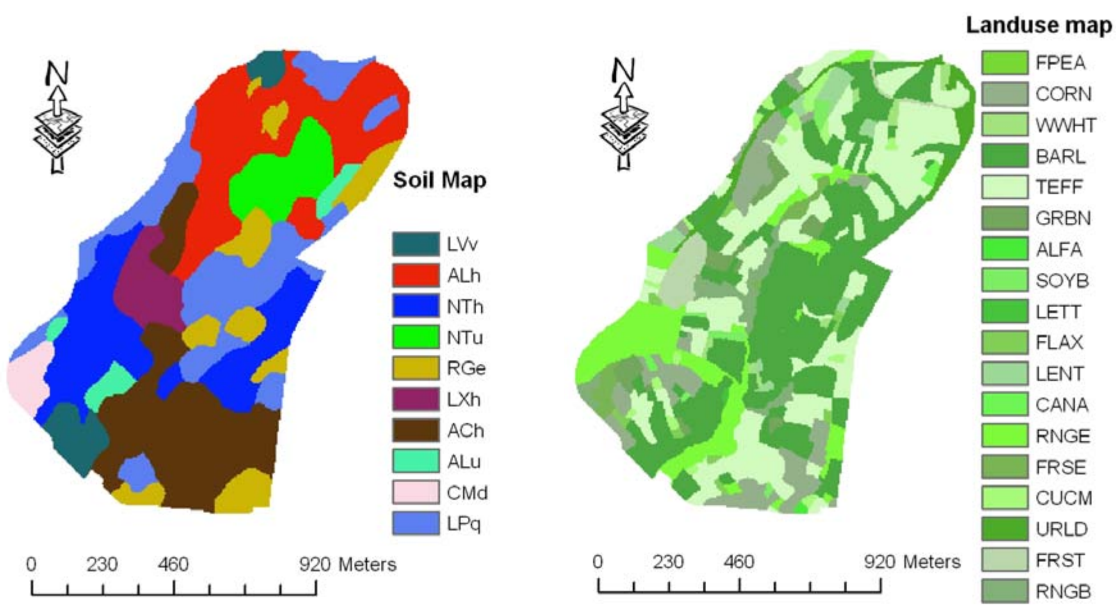


Figure1. Average monthly rainfall, minimum, maximum and average air temperature of Anjeni watershed

For creation of HRU's SWAT requires land use, soil and slop layers and their threshold inputs to define the level of spatial detail (Neitsch et al. 2005). We have used ten years of weather data and hydrometric measurements on Minchet river in the Anjeni gauged watershed for the simulation of the hydrology of the area. For HRU definition we have determined 10 classless of soil type and 17 classless of landuse categories (Figure 3).



(a) Landuse Map (b) Soil Map

Figure 3. Soil and Land use map of Anjeni gauged watershed (original source SCRP)

Results and Discussion

The application of the model involved data processing, model setup, sensitivity analysis, calibration and validation of the model. We conducted the simulations of stream flow on a monthly basis to compare the modeling output with the observed flow data for the period 1984 to 1993. The simulation from 1984 to 1986 was considered as a “warm-up” period for the model to allow hydrologic processes to reach a certain level of equilibrium. The watershed was divided into 13 subwatersheds. For HRU definition we have used dominant HRU type option so that each subbasin consisted of one specific HRU (Table 3).

Table 3. Description of each subbasin with the corresponding landuse, soil and topographic characteristics

SUBBASIN	*HRU*	*LANDUSE*	*SOIL*	*Slope Class*	*Average slope length, m*	*Average slope steepness (m/m)*
1	1	TEFF	ALh	15-9999	121.9512	0.1946517
2	1	BARL	NTu	15-9999	121.9512	0.2197215
3	1	TEFF	ALh	15-9999	121.9512	0.1984594
4	1	BARL	ALh	15-9999	121.9512	0.2530086
5	1	FRST	NTh	15-9999	121.9512	0.2074802
6	1	BARL	LPq	15-9999	121.9512	0.2734874
7	1	BARL	NTh	15-9999	121.9512	0.1942956
8	1	BARL	NTh	10-15	121.9512	0.1270579
9	1	BARL	ACh	10-15	121.9512	0.1234175
10	1	CORN	ACh	15-9999	121.9512	0.181459
11	1	RNGE	NTh	15-9999	121.9512	0.1910231
12	1	TEFF	ACh	15-9999	121.9512	0.1877687
13	1	BARL	LVv	15-9999	121.9512	0.1558671

ALh - Haplic Alisols, NTu - Humic Nitisols, NTh - Haplich Nitisols, LPq - Lithic Leptosols, Ach - Haplic Acrisols, Ach - Haplic Acrisols, LVv-Vertic Luvisols

Sensitivity analysis:

For the setup of the ‘sensitivity analysis input’ we have defined the analysis location, the algorithm input settings and the SWAT model parameters to be evaluated. Twenty six hydrological parameters were tested for sensitivity analysis for the simulation of the stream flow. For this processes we adapted the default lower and upper bound parameter values. We have used ‘multiply by value (%)’ option of the variation method. For ‘sensitivity analysis output’ we have selected ‘average criteria’ options. Eighteen parameters were found to be sensitive and the most eight sensitive ones were considered for calibration processes (Table 4). The remaining eight parameters had no significant effect on the monthly stream flow simulations. The details of all hydrological parameters are found in the ArcSWAT interface for SWAT user’s manual (Winchell 2007).

Table 4. Sensitive parameters and their rankings.

Parameter	*Lower and Upper bound*	*Relative Sensitivity*	*Rank*
Initial SCS CN II value, Cn2	±25	3.2900	1
Base flow alpha factor [days], Alpha_Bf	0 - 1	0.7640	4
Groundwater delay [days], Gw_Delay	±10	0.5600	3
Available water capacity [mm WATER/mm soil], Sol_Awc	±25	0.2630	2
Channel effective hydraulic conductivity [mm/hr], Ch_K2	0 - 150	0.2420	5
Soil evaporation compensation factor, ESCO	0 - 1	0.1140	6
Groundwater "revap" coefficient, Gw_Revap	±0.036	0.1030	7
Soil depth [mm], Sol_Z	±25	0.0806	8

Model Calibration

We have used a combination of manual and automatic calibration method for the calibration of SWAT2005 model using the measured stream flow data. For this analysis a seven years, from 01 January 1984 to 31 December 1991, meteorological and hydrometric flow data were utilized, including three years of 'worm-up' period. After many trials we have found a good agreement between observed and simulated flows at Anjeni station as shown in Figure 4 and indicated by the coefficient of determinations (R^2), 0.92 and the Nash-Suttcliffe simulation efficiency (NS), 0.91 (Figure 6a).

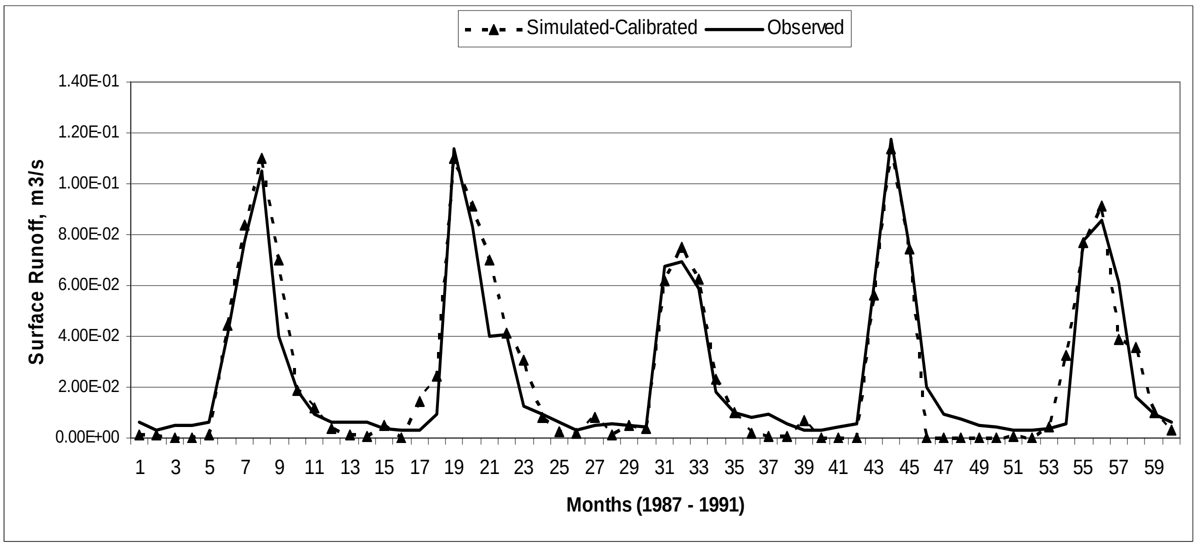


Figure 4. Comparison between observed month flow, SWAT default simulation and Simulation using calibrated parameters for the calibration period (1987 – 1991)

Model Validation

Model validation was done using the calibrated parameters. Model validation involved re-running the model using input data independent of data used in calibration. Two years observed flow data from 01 January 1992 to 31 December, 1993 from Anjeni hydrometric measurement were used to validate the model. The validation process led to found the coefficient of determinations (R^2) and the Nash-Suttcliffe simulation efficiency (NS) are 0.96 and 0.93 respectively (Figure 6b). This showed that there is a good agreement between monthly measured and simulated flows, (Figure 5).

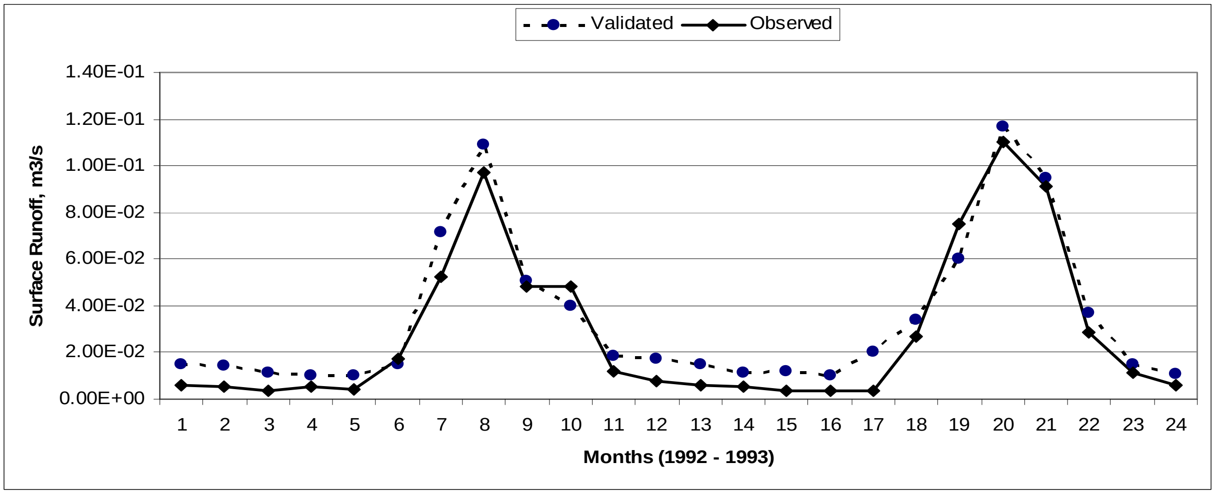
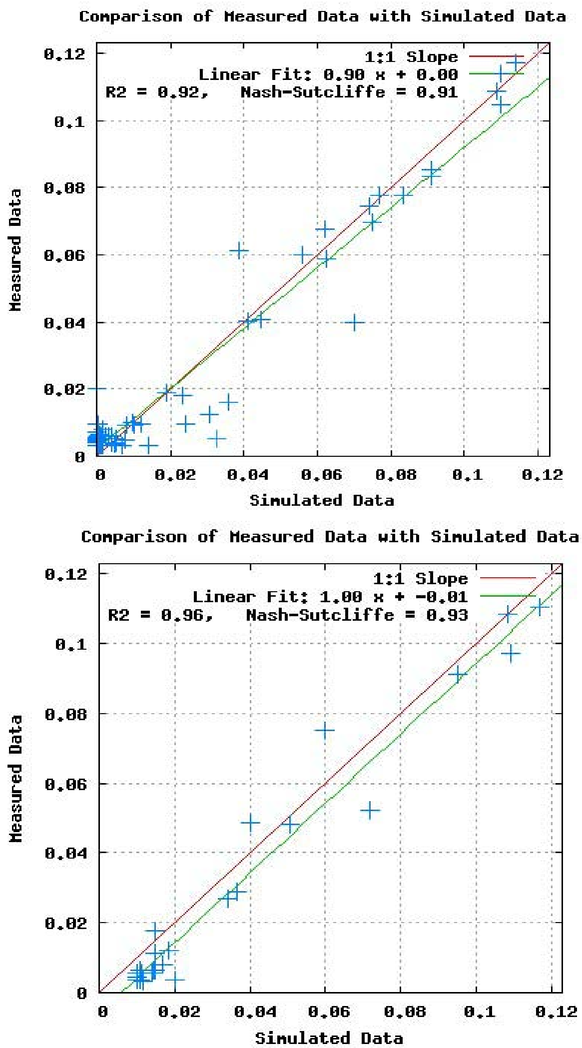


Figure 5. Comparison between observed monthly flow and Simulation result using calibrated parameters for validation period (from 01/01/1993 to 31/12/1993).



(a) **(b)**

Figure 6. Scatter plot of monthly simulated versus observed flow at Anjeni gauged station (a) after calibration processes and (b) after validation process

Conclusion

The main study objective was to evaluate the performance and applicability of the SWAT model in predicting the hydrology of the Anjeni gauged watershed. The ability of this model to sufficiently predict stream flow was evaluated through sensitivity analysis, model calibration, and model validation. The sensitive parameters were used to find the most reasonable parameter values for optimum estimations of the stream flows. The analysis showed that Baseflow alpha factor (days), initial SCS curve number for moisture condition II and groundwater delay time (days) are the most sensitive parameters in Anjeni watershed.

Model performance evaluation statistics for simulating monthly flows at Anjeni gauged watershed for calibration (NS=0.92 and R^2=0.91) and validation (NS=0.96 and R^2=0.93,) periods showed that the simulated trends matched well with the observed data. Looking into the calibrated and validated statistical results we can conclude that the set of optimized parameters during calibration process can be taken as the representative set of best parameters for Anjeni watershed and surrounding areas which have similar agro-climatic condition.

The study has shown that the SWAT model can produce reliable estimates of monthly runoff. SWAT is a good modeling tool for analyzing hydrologic processes and water resources planning and management in the Anjeni gauged watershed.

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Calibrating SWAT using satellite evapotranspiration in the Upper Bhima catchment, India

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Abstract

A common issue in hydrological modeling is the lack of reliable data to calibrate the model. Calibration is also constrained by uncertainty in representing the physical features of a river catchment, and the implementation of hydrological processes in a simulation model. With the advent of remote sensing algorithms that enable quantification of actual evapotranspiration (ET_{act}) in time and space calibration of these models can take a novel direction. In this paper, an innovative approach is presented which incorporates Remote Sensing derived evapotranspiration in the calibration of the Soil and Water Assessment Tool (SWAT). SWAT was calibrated using an ET_{act} time series of eight months with high spatial detail in the Upper Bhima catchment in India. In the best performing optimisation, the r^2 between monthly sub-basin simulated and measured ET_{act} was increased from 0.40 to 0.81. ET_{act} was more sensitive to the groundwater and meteorological parameters than the soil and land use parameters. This innovative approach to calibrate on remotely sensed ET_{act} is a promising approach for similar data scarce catchments across the globe.

KEYWORDS: Evapotranspiration, Remote Sensing, Calibration, SWAT

Introduction

Calibration of physically based, distributed hydrological models is complex given limitations of the input data, complexity of the mathematical representation of hydrological processes, and incomplete knowledge of basin characteristics. A priori, it is not clear whether a unique set of model parameters exists for a given catchment. While comparing model outputs to observations the main question is what the causes of these differences are (Duan *et al*., 2003)? Model calibration is usually based on the comparison between modelled and observed hydrographs for a limited number of locations and a small number of input parameters are varied in a trial and error mode to achieve a desired response (Gupta *et al*., 1998; Anderston *et al.*, 2002). In a complex distributed hydrological model with numerous parameters with a high spatial and temporal heterogeneity this approach can be cumbersome. To overcome these difficulties a number of different auto-calibration or parameter optimisation methods have been developed that deploy a systematic approach to parameter estimation.

The type of optimisation algorithm applied is the dominant distinguishing factor in parameter estimation (Singh and Woolhiser, 2002).The objective function describes the difference between the observed and model simulated values. RMSE statistics and the Nash-Sutcliffe criterion (Nash and Sutcliffe, 1970) are amongst the most commonly used.

Minimizing the objective function is complex; because most discharge based objective functions in distributed hydrological models have multiple extremes. Optimisation algorithms adopt either a gradient (Levenberg (1944); Marquard (1963)) or a global search method such as the shuffled complex evolution algorithm (Kuczera, 1997) and genetic algorithms (Wang, 1991). Global optimisation algorithms are designed for locating the global optimum. Local search algorithms have been criticized for getting trapped in local minima. The most important advantage of a local search is nonetheless its efficiency; e.g., the number of model calls required to find the optimum set of parameters. Skahill and Doherty (2006) show how the algorithms underlying the Gauss–Marquardt–Levenberg (GML) method of computer-based parameter estimation can be improved to enhance the possibility to find the global minimum while retaining the model run efficiency.

Hydrological parameters measured using Remote Sensing, which has a high spatial and temporal observational resolution, could provide a suitable solution in this respect. Measuring hydrological parameters such as evapotranspiration and soil moisture using Remote Sensing is a growing field of research (Bastiaanssen *et al.*, 1998; Hall *et al.*, 1992; Kite and Droogers, 2000; Su, 2000).

The objective of this study was to evaluate the method of using Remote Sensing derived actual evapotranspiration (ET_{act}), based on the Surface Energy BALance algorithm (SEBAL; Bastiaanssen *et al*., 1998), to calibrate the process-based hydrological model Soil and Water Assessment Tool (SWAT; Arnold *et al.*, 1998), in the water scarce Upper Bhima catchment in southern India. The Parameter ESTimation (PEST) software (Doherty, 2005), incorporating the gradient search GML optimisation method, was used for this purpose.

Study area

The Upper Bhima catchment (45,678 km2) is located in the upstream part of the Krishna basin in southern India and originates in the Western Ghat mountains and covers part of the Maharasthra state (Figure 1). The catchment is located between 16.5^o-19.5^o latitude and 73.0^o -76.5^o longitude. The elevation ranges from 414 meter in the east to 1458 meter in the Western Ghat mountains and 95% of the catchment is below 800 meter and relatively flat. The average slope of the catchment is 2%.

The catchment has two main tributaries, the Sina River which drains the north eastern part and the Bhima River which drains the remainder. The catchment is an important source of water for the entire Krishna basin as a major part of the precipitation falls in the Western Ghat range in the east of the catchment and is retained and released to downstream areas through an intricate set of reservoirs, especially along the Bhima tributary. Flows in the rivers are, therefore, mainly human controlled and respond less directly to variations in the climate excitations and biophysical conditions and are hence less suitable to use for calibration. The reservoirs accumulate water during the monsoon season (June to September), and this is gradually released throughout the irrigation season (October to May).

The catchment has a highly diverse climate mainly caused by the interaction between the monsoon and the Western Ghat mountain range (Gunnel, 1997). The precipitation ranges from less than 600 mm in the eastern part of the basin to over 1800 mm in the mountains in the west with an average of 941 mm during the averagely wet irrigation year 2004-2005 (Figure 40).

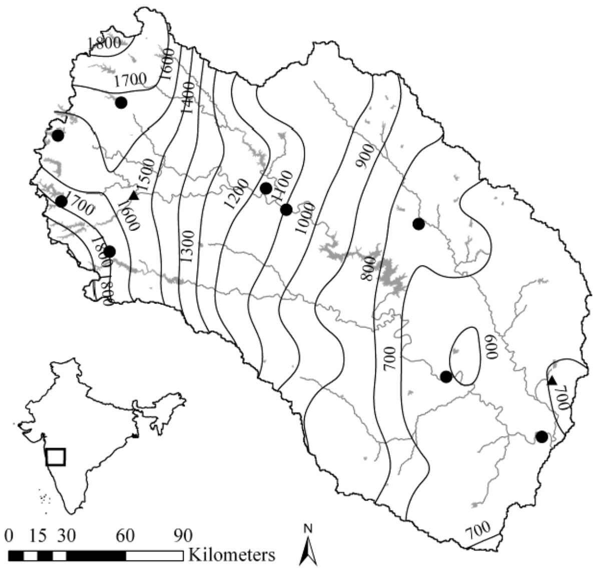


Figure 40. Upper Bhima catchment boundary, contours of the precipitation sum from June 2004 to May 2005, river network, major reservoirs, and meteorological station (circles represent precipitation stations; triangles represent precipitation, temperature, wind speed, relative humidity and radiation stations)

Reference evapotranspiration (ET_{ref}) in the basin. ET_{ref} is calculated using Penman-Monteith (Monteith, 1965) and alfalfa as reference crop. The catchment has a high annual ET_{ref} (1814 mm) ranging from 224 mm/month in May to 108 mm/month in December. More than 75% of the annual precipitation occurs during the monsoon. Between October and May, large precipitation deficits occur with the peak in May (179 mm) just before the onset of the monsoon.

The state of Maharasthra has a diverse cropping pattern characterized by cultivation of sugarcane, sorghum, wheat, corn, millet, groundnut, grass fodder, and a variety of horticultural crops (Neena, 1998). Three main types of agricultural systems were identified in the catchment: (i) rain fed agriculture with a single crop (e.g. sorghum) cultivated during the monsoon, (ii) supplemental irrigated agriculture with one rain fed crop during the monsoon (e.g. sorghum) and a (groundwater) irrigated crop planted in October and harvested in February (e.g. winter wheat) and (iii) irrigated perennial sugarcane which has a growing period of 11 months and which is grown throughout the year and irrigated from water released by the reservoir system. Other natural land covers include rangelands, mixed forests, evergreen forests and water surfaces.

Methods

SEBAL

The Surface Energy Balance Algorithm for Land (SEBAL) formulated by Bastiaanssen et al. (1998) was used to calculate bi-weekly ET_{act} from October 2004 to

May 2005. Spectral radiances in the visible, near-infrared, and thermal infrared part of the spectrum derived from 16 MODIS satellite images were used. SEBAL converts satellite radiances into land surface characteristics such as surface albedo, leaf area index, vegetation index, and surface temperature, which were used in solving the instantaneous energy budget equation given by

$$L_v E = Q^* - G_0 - H \qquad (1.)$$

Where L_vE is the latent heat flux (W m-2), Q* is the net radiation flux at the surface (W/m2), G_0 is the soil heat flux (W/m2), and H is the sensible heat flux to the air (W/m2). Knowing the instantaneous soil, latent, and sensible heat fluxes made it possible to calculate the evaporative fraction given by:

$$\Lambda = \frac{L_v E}{Q^* - G} \qquad (2.)$$

The most important assumption of SEBAL is that the evaporative fraction is constant during the day and this assumption allows the conversion of an instantaneous L_vE value to a daily value. Experimental work has demonstrated that this holds true for environmental conditions where soil moisture does not significantly change (e.g., Shuttleworth et al., 1989; Brutsaert and Sugita, 1992; Nicols and Cuenca, 1993; Kustas et al., 1994; Crago, 1996; Franks and Beven, 1997). For periods longer than one day it may be assumed that the soil heat flux equals 0. The 24hr latent heat flux could therefore be determined by

$$L_v E_{24hr} = \Lambda Q^*_{24hr} \qquad (3.)$$

The final step was to calculate biweekly evapotranspiration data. This was achieved by inserting L_vE_{24hr} into the Penman-Monteith equation (Monteith, 1965). SEBAL has been extensively validated (Bastiaanssen *et al.*, 1998[b]).The 16 biweekly ET_{act} images of the Upper Bhima catchment were accumulated to eight monthly images from October 2004 to May 2005, which were all used in the calibration of the SWAT model. SEBAL could not be applied during the monsoon months (June to September) due to the lack of cloud free imagery.

SWAT

A SWAT model was built and simulations were run on a daily basis from June 2004 to May 2005. SWAT is a distributed hydrological model providing spatial coverage of the integral hydrological cycle including atmosphere, plants, unsaturated zone, groundwater, and surface water. The model is comprehensively described in literature (Arnold *et al*., 1998; Srinivasan *et al.*, 1998).

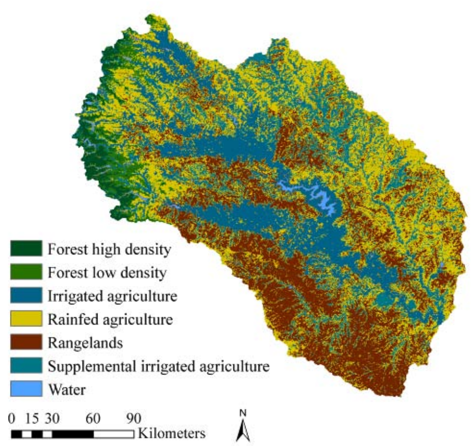
Conceptually SWAT subdivides the catchment into sub-basins and a river network based on a digital elevation model (DEM). Based on unique combinations of soil and land use, the sub-basins were further detailed into hydrological response units (HRUs), which were the fundamental units of calculation. A total of 115 sub basins and 768 HRUs were delineated in the Upper Bhima catchment.

The Penman-Monteith method (Monteith, 1965) was used in SWAT to calculate daily reference evapotranspiration for alfalfa (ET_{ref}) and potential plant transpiration (ET_p). Daily data on radiation, wind speed, relative humidity, and air temperature for two different meteorological stations were used (Figure 1). Potential daily plant transpiration is calculated using actual daily crop height and leaf area index (LAI), required to determine the aerodynamic resistances and canopy resistances respectively. Potential soil evaporation is an exponential function of ET_{ref} and the soil cover and is further reduced during periods with high plant water use. Actual soil evaporation is limited by the soil water content (θ) and is reduced exponentially when θ drops below field capacity.
To calculate actual plant transpiration the potential plant water uptake is defined by

$$w_{p,z} = \frac{ET_p}{\left[1-e^{\beta_w}\right]} \cdot \left[1 - e^{\left(-\beta_w \cdot \frac{z}{z_{root}}\right)}\right] \qquad (4.)$$

Where $w_{p,z}$ (mm H_2O) is the potential plant water uptake from the soil surface to a specified depth from the soil surface on a given day, ET_p (mm H_2O) is the maximum plant transpiration on a given day, β_w (-) is the water use distribution parameter, z is the depth from the soil surface (mm), and z_{root} is the depth of root development in the soil (mm). Actual plant water uptake equals actual plant transpiration and is, similarly to soil evaporation, reduced exponentially when θ drops below field capacity. Actual evapotranspiration (ET_{act}) is the sum of interception, actual soil evaporation, and actual plant transpiration.

Land used in this study was based on a 15 class unsupervised land use classification using a time series of 16 MODIS derived Normalized Difference Vegetation Index (NDVI) images with a spatial resolution of 250 meter. Based on existing land use map and field surveys 15 classes were clustered into the land use classes defined in Figure 41. The results were visual verified with high resolution satellite imagery. The final land use map is shown in Figure 41.



Class name	Code	Area (10^3 ha)	Area (%)
Water surfaces	WATR	4111	1
Rangelands	RNGB	122417	27
Rain fed agriculture	AGR1	153477	34
Supplemental irrigated agriculture	AGR2	61208	13
Irrigated sugarcane	AGR3	90899	20
Evergreen forest	FRSE	14160	3
Mixed forest	FRST	10506	2

Figure 41. Land use in the Upper Bhima catchment

PEST

In this study PEST (2005 version) was used to calibrate the SWAT model using ET_{act} acquired with SEBAL. PEST is a non-linear parameter estimation package that can be used to estimate parameters for just about any existing computer model (Doherty, 2005). PEST uses the GML algorithm is to optimize the model. The goal of PEST is to minimise the objective function (Φ), which is defined as the sum of squared deviations between SWAT generated values and experimental observations based on SEBAL.

Based on the analysis of time series of ΔET_{act} (ET_{act} SEBAL – ET_{act} SWAT) a number of PEST optimization runs were identified. Each PEST run optimizes a set of variables related to either of these explaining factors. The results of each PEST run (Φ reduction and r^2) were used to identify a number of combination PEST runs. Final results were analyzed at basin, sub-basin, and HRU level. All ET_{act} optimisation runs calculated the Φ based on monthly data at sub basin level. A total of 920 observations (8 months times 115 sub basins) were used in the objective function. The following optimisation runs were formulated:

- Available water capacity (AWC). The AWC is defined as the difference between the field capacity of the soil and the permanent wilting point. It is defined per soil layer per soil type and determines, to a large extent, the water holding capacity of the soil. Ten different soil types with two layers each resulted in 20 different parameters to be optimized. AWC should is bound by the range 0.05 mm/mm and 0.60 mm/mm.
- Maximum potential leaf area index (BLAI). The LAI is the leaf area divided by the land area. The BLAI is one of six parameters that determine leaf area development of a crop in SWAT and determines the maximum threshold. BLAI is specified per land use type (Table 2) and, excluding water surfaces, resulted in 6 parameters to be optimized. The bounding range is between 2.0 and 8.0 for AGR1, AGR2, FRSE and FRST, between 5.0 and 12.0 for AGR3 and between 1.0 and 8.0 for RNGE.
- Monthly rainfall increment (RFINC). The RFINC is specified per month and per sub-basin and is defined as the relative monthly adaptation in rainfall. The assumption was made that the spatial distribution of the TRMM derived precipitation was correct, however that for specific months the scaled absolute amounts of precipitation could be incorrect. This led to one variable to be optimized per month. RFINC is allowed to vary between -200% and 200% of the original monthly precipitation.
- Groundwater revap coefficient (REVAP1 and REVAP2). Water may conceptually move from the shallow aquifer into the overlying unsaturated zone. The process of water being evaporated from the capillary fringe in dry periods is referred to as groundwater revap and in SWAT quantified by the revap coefficient (β_r) multiplied by ET_{ref}. Two optimisation runs were designed. For the first optimisation run one β_r for each land use, except water, was defined (REVAP1, 6 variables). For the second optimisation run it was assumed that β_r varies per land use and per elevation zone. Four different elevation zones were defined (0-500 m, 500-600 m, 600-700m and >700m). In combination with land use this resulted in 21 unique β_r resulting from unique combinations of elevation zone and land use

class (REVAP2, 21 variables). For both optimisation runs the revap coefficient may range between 0.0 and 0.5.

Two combined optimisation runs were performed based on the results of the individual runs (COM1 and COM2). COM1 combined AWC, RFINC and GWREVAP and COM2 combined AWC, RFINC and GWREVAP2.
The results of the best performing combination run were validated using historical stream flow data.

Results

Table 3 presents the results for the different optimisation runs. In the base run the RMSE was equal to 24 mm/month and ET_{act} SWAT iwas generally higher than the ET_{act} SEBAL (ε = 5.2 mm/month). The AWC run reduced the RMSE to 22 mm and the average value for the AWC after optimisation was 0.22 mm/mm, while the average initial value was 0.15 mm/mm. The model was relatively insensitive to maximum plant leaf area index (BLAI). The RMSE was not reduced and the average of the residuals only decreased by 0.4 mm/month. The RFINC optimisation run had a significant effect and the r^2 increased from 0.40 to 0.70. The average adjustment in monthly precipitation during the calibration months was limited (-13 mm/month) with a maximum in December (21 mm) and a minimum in April (-29 mm). ET_{act} SWAT was also sensitive to the groundwater revap coefficient. The GWREVAP run (6 variables) resulted in a RMSE of 17 mm and ε was reduced to 1.6 mm/month. GWREVAP2 (21 variables) yielded a slightly higher r^2 but no significant improvements in RMSE and ε were observed. The best results were achieved by the combination runs (COM1 and COM2). COM2 yielded the best results in the optimisations: 2987 model calls were required to increase r^2 from 0.40 to 0.81. The RMSE in that case equalled 13 mm/month and ET_{act} SWAT was on average only 0.5 mm/month higher. The COM1 results were also good, but significantly less model calls were required (1610). In the COM1 run the average available water content increased from 0.15 mm/mm to 0.21 mm/mm, the rainfall adjustment on average was -2% during the calibration months and the groundwater revap coefficient equalled 1.1.

Table 30: Results of different optimisations runs. #var and #obs are the number of variables and observations used in the optimisations. RMSE is the Root Mean Square Error, ε is the average of the residuals and # model calls is the number of model calls required to reach the optimisation results.

PEST run	Variable	# var	# obs	Φ	RMSE (mm/month)	r^2	ε (mm/month)	# model calls
BASE	-	0	920	5.29E+05	24.0	0.40	5.9	-
AWC	Available water content	20	920	4.49E+05	22.1	0.49	5.2	664
BLAI	Maximum plant leaf area index	6	920	5.19E+05	23.8	0.41	5.5	68
RFINC	Monthly rainfall increment	12	920	2.54E+05	16.6	0.70	0.8	324
GWREVAP	Groundwater revap coefficient	6	920	2.78E+05	17.4	0.68	1.6	55
GWREVAP2	Groundwater revap	21	920	2.66E+05	17.0	0.70	1.7	792

	coefficient							
COM1	Available water content, monthly rainfall increment, groundwater revap coefficient	53	920	1.63E+05	13.3	0.81	0.5	2987
COM2	Available water content, monthly rainfall increment, groundwater revap coefficient	38	920	1.77E+05	13.9	0.79	-0.5	1610

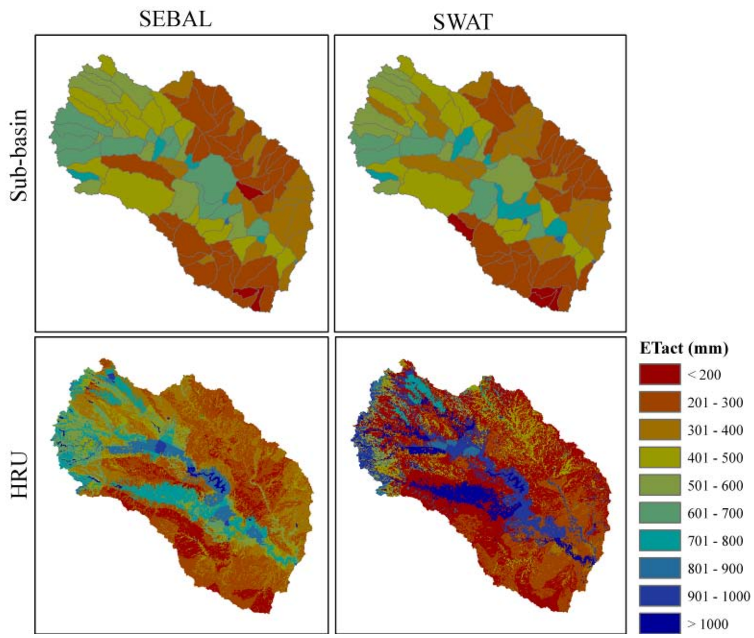


Figure 42. Eight month sum of ETact for SWAT and SEBAL on sub basin and HRU level respectively. SWAT results relate to COM2 optimisation.

Figure 8 maps the eight month ET_{act} sum for SWAT and SEBAL at sub-basin and at HRU level. At sub-basin level, the spatial patterns between SWAT and SEBAL were highly consistent. At HRU level there were considerable differences. The general spatial patterns were well depicted. However, some HRUs within a sub-basin evaporated more than measured with SEBAL and some evaporated less, however aggregated over the entire sub basin these differences were levelled out. For example most sub basins with a significant area of AGR3 also contained a large area of RNGE. ET_{act} of AGR3 seems to be overestimated and ET_{act} of RNGE seems to be underestimated. On subbasin scale these differences level out and the results are good, however the variation in SWAT ET_{act} at HRU level was larger. This was a result of the fact that PEST optimises monthly ET_{act} at sub basin level and not at HRU level.

There are several ways to evaluate the reliability of any optimisation of a distributed hydrological model across time and space. Figure 7 shows the scatter plots for run COM1 between SEBAL ET_{act} and SWAT ET_{act} on catchment, sub-basin and HRU level respectively. It also shows the individual monthly data and the eight month sum of $ET_{act.}$ The figure shows that the goodness of fit decreases with spatial and temporal detail. The r^2 of monthly catchment ET_{act} for example is as high as 0.90, while at HRU level the

r^2 is only 0.35. In time we see similar patterns. The r^2 at monthly sub basin level is 0.81 while if the eight month sum is analyzed the r^2 increase to 0.92.

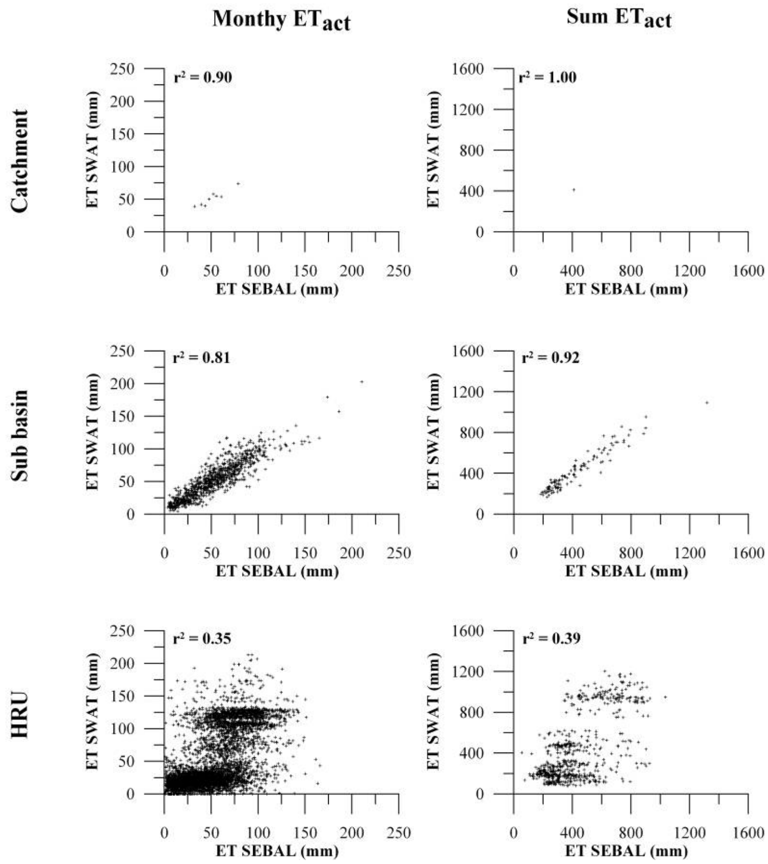


Figure 43. Scatter plots of SEBAL and SWAT ETact. Monthly data are shown on the left side graphs and the eight month sum is on the right side of the graphs. Spatial detail increases from top to bottom and ranges from catchment, sub basin to HRU level respectively. SWAT results relate to COM1 optimisation.

Discussion and conclusions

This study showed that the spatially distributed hydrological model SWAT can be successfully calibrated using the GML algorithm and remotely sensed derived evapotranspiration from a time series of MODIS images in a data scarce area. The best results were obtained by optimising a combination of soil, meteorological, and groundwater related parameters for an eight month time series of sub-basin actual evapotranspiration. Optimising a total of 53 variables using 920 monthly observations increased the r^2, significantly, from 0.40 to 0.81. Separate optimisation runs revealed that ET_{act} is more sensitive to the groundwater and meteorological parameters than to soil and land use parameters. On sub-basin level the ET_{act} showed least response to the land cover dependent maximum leaf area index. Furthermore, it can be concluded that at the HRU level more work is required to fine-tune the calibration procedure. The calibration was only reliable at the spatial and temporal scale on which the observations, used in the optimisation, were based. Future work should focus on calibration strategy that incorporates HRU level ET_{act} observations and discharges at a high temporal resolution in the objective function.

In this study, the gradient search GML algorithm was used in the optimisation although this method is sometimes sensitive to local minima, especially when non-linear processes are modelled. However, a time series of spatially distributed ET_{act} exhibits more linear behaviour than discharge at a limited number of locations. Moreover, it has been shown that global search algorithms require much more function calls to identify the global minimum. SWAT was recently used in the evaluation of a number of optimisation algoritms; Shuffled Complex Evolution, real-valued simple Genetic Algorithm, multi-start Simplex and Monte Carlo Sampling and a new algorithm called the Global Greedy Search algorithm (Tolson and Schoemaker, 2006). For two case studies a maximum of 2500 (6 parameters) respectively 6000 (14 parameters) SWAT model calls were required. The GML algorithm is much more efficient in this respect (Skahill and Doherty, 2006) and for the best performing optimisation (53 parameters) in this study 2987 model calls were required. We therefore conclude that PEST and the GML algorithm served our objectives best.

The calibration period covers only eight months and a longer time scale would be preferable. Presently longer time series of Remotely Sensed ET_{act} are unavailable, but this might changes in the future with the advent of a standard MODIS ET product (Running *et al.*, 1995). Realistic simulations during the dry period from October to May are also more important than the monsoon period. Runoff is not a critical issue, but ET management, water shortage and irrigation are the dominant hydrological issues relevant to water managers.

The monsoon period is not covered in the calibration, because of the absence cloud free imagery. SEBAL depends on the visible, NIR en thermal IR part of the spectrum which is hampered by clouds. A viable alternative could be to incorporate radar based soil moisture in a combined objective function during the monsoon months in order to appropriately calibrate the model in the monsoon season.

In developing countries, where lack of data is an issue and the planning process needs to be supported by scientific sound measures, the innovative use of Remote

Sensing in hydrological model calibration as presented in this study will contribute to the prevention of disasters and improve sustainable management in the long term. Recently, interest in using simulation models in ungauged or sparsely gauged basins has increased leading to some concerted actions. The most relevant is the Prediction in Ungauged Basin (PUB) initiative; an International Association for Hydrological Sciences (IAHS) initiative for the decade of 2003-2012, aimed at uncertainty reduction in hydrological practice (Sivapalani *et al.*, 2003). PUB focuses the development of new predictive approaches that are based on "understanding" of hydrological functioning at multiple space-time scales. This study provided an ET based innovative approach at different temporal and spatial scale that fits well into the PUB science program.

Acknowledgements

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Problems and Solutions in Applying SWAT in the Upper Midwest USA

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Abstract

We are applying the Soil and Water Assessment Tool (SWAT2000) to the Willow River watershed in western Wisconsin to assess the effects of land use and management changes. The Willow River drains about 735 km^2, much of which is agricultural land yielding substantial nonpoint-source loads of sediment and nutrients. The upper Midwest USA is a geologically young landscape with many closed drainages, intensive cultivation amid patches of forest, and expanding urban centers. Principal crops include corn, soybeans, and alfalfa typically grown in rotation to support both dairy and cash-crop operations. SWAT had problems with rotations that included alfalfa, wherein alfalfa could not be removed from the landscape once planted. Code changes to the SWAT engine were required to correct this problem. Corn yields were underestimated because of nitrogen stress due to excessive denitrification. Again, code changes allowed parameterization of the nitrification process. About 29% of the landscape drained to closed depressions; the fraction of closed drainage in each subbasin was routed to the Pond routine in SWAT and parameterized to trap all sediment and phosphorus. However, seepage from Ponds was trapped by SWAT in shallow aquifer storage and not included as groundwater recharge, and therefore did not contribute appropriately to stream baseflow. As a surrogate for this missing groundwater discharge, we disallowed Pond seepage and forced slow surficial outflow by increasing the days to reach target storage. Phosphorus loading from subbasins to the channel reaches was complicated by the addition of a subbasin chlorophyll load by SWAT. When the stream water-quality routine was activated, this chlorophyll load was apparently interpreted as algae with significant phosphorus content, and the subsequent release of this phosphorus constituted an additional load unconnected to the land-surface phosphorus budget. To avoid this extraneous phosphorus load, either the stream water-quality routine had to be de-activated, or the phosphorus content of algae had to be reduced to a negligible fraction. Subbasin sediment yields were consistently overpredicted by SWAT with default parameterization. Sediment calibration could be achieved by parameterizing the soil-loss equation to reduce erosion, or by parameterizing the channel to trap excess sediment. Despite these problems encountered during model construction and calibration, the solutions given above have resulted in workable SWAT models for our purposes. We acknowledge other members of the SWAT Midwest America Users Group (SMAUG) in identifying and solving the above problems. Some of these problems have already been addressed in SWAT2005, and we are confident that the code will continue to improve.

KEYWORDS: SWAT, nonpoint source pollution, sediment, nutrients, runoff, agriculture

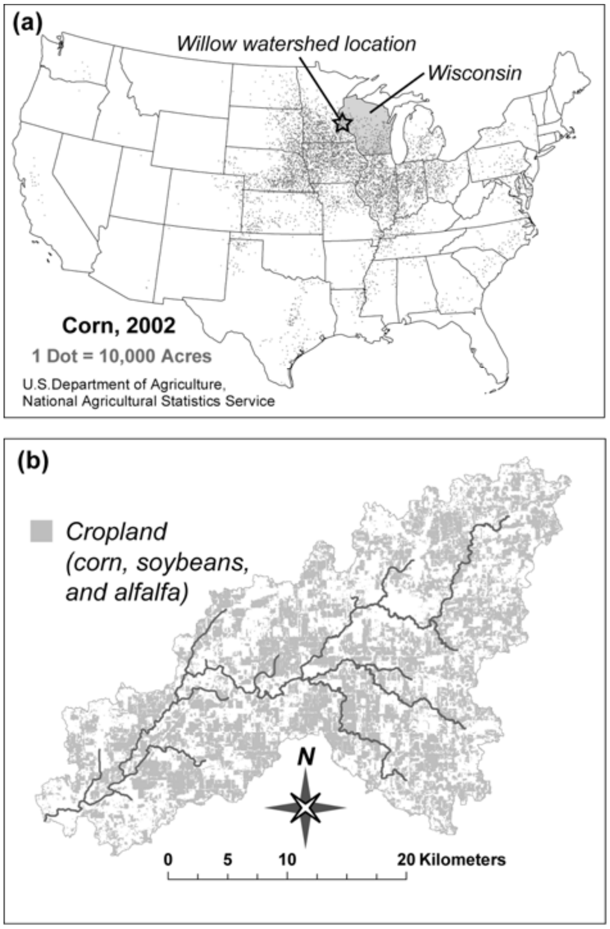

Figure 1. (a) Location of Willow River watershed (star) in Wisconsin (shaded), relative to area of corn harvested in the Midwest USA in 2002 and (b) cropland in the Willow River watershed in 1992-93.

Introduction

Row-crop agriculture, as typified by corn, is common in the Midwest USA (Figure 1a), and nonpoint-source (NP-S) loads of sediment and nutrient from these lands cause significant ecological problems for receiving waters. Suspended sediment is one of the most prevalent pollutants in streams in the USA (USEPA, 2002). Increased phosphorus loading from watersheds is generally considered the primary cause of eutrophication of freshwater ecosystems (Carlson, 1977; Schindler, 1978). Excessive nitrogen loading has been linked to hypoxia in the bottom waters of the Gulf of Mexico (Goolsby, 2000; Rabalais *et al.*, 2001). Addressing these problems will require improved agricultural practices that reduce soil erosion and the loss of applied fertilizers.

Computer models of watersheds provide an important management tool for planning remedial actions. In conjunction with monitoring programs, modeling can help target critical source areas of NP-S pollution for remediation. More importantly, models can help predict effectiveness of remedial actions. However, the predictive power of models depends on their ability to simulate hydrological mechanisms of flow and transport. The art of successful model application relies on knowing model strengths and limitations in simulating these mechanisms.

The purpose of this paper is to describe several perceived limitations in a model of an agricultural watershed in the upper Midwest, and to provide methods to circumvent these limitations to achieve successful model calibration and application. The modeling program selected was the Soil and Water Assessment Tool (SWAT2000), a widely used program developed to simulate long-term loading of NP-S pollutants in large basins with diverse land use (Arnold *et al.*, 1998; Di Luzio *et al.*, 2002). In some cases, overcoming model limitations required changes to the engine code; in other cases, alternative parameterizations allowed acceptable model performance. Knowledge of model problems and solutions has implication for all users of SWAT and can lead to improved versions of the model code such as SWAT2005, which addresses some of the issues with SWAT2000 identified here.

Study Site

The Willow River in western Wisconsin, USA, is about 91 km long and drains about 735 km^2 of rural lands which are substantially agricultural (Figure 1b). It is tributary to the St. Croix River, a nationally recognized Scenic and Recreational Riverway in the upper Mississippi River drainage. The Willow watershed was chosen for study because it has been identified as a significant contributor of NP-S pollution (Lenz *et al.*, 2003) and has thus been targeted for remediation by implementation of agricultural best management practices (BMPs). The climate is humid mid-continental with most rainfall in summer; the 1971-2000 normal precipitation was about 813 mm, averaged from two nearby weather stations (NCDC, 2005). Flow averages about 4.7 cms (cubic meters per second) (Lenz *et al.*, 2003). Two reservoirs on the main channel effectively trap sediment and ameliorate flood peaks, though these reservoirs are relatively small (96 and 70 ha) and shallow (1.1 and 2.4 m mean depth).

Most of the watershed has 15–60 m of Pleistocene glacial drift overlying Ordovician sandstone and dolostone (Feinstein *et al.*, 2005). This geologically young landscape has many closed drainages totaling about 29% of the watershed area. Soils derived from this glacial parent material are predominately loamy and include well-drained to moderately poorly-drained areas (USDA, 1978). From 1992-93 satellite data, land use in the study area was estimated to be 43% agriculture cropland, 30% grassland, 18% forest, 7% water/wetland, and only 2% urban (WDNR, 1998). By 1999, about 10% of this cropland was converted to other land uses, largely rural-residential developments (Almendinger and Murphy, 2005). Principal crops are corn (*Zea mays* L.), soybeans (*Glycine max* L.), and alfalfa (*Medicago sativa* L.) typically grown in rotation to support both dairy and cash-crop operations.

Methods – *Model Construction and Data Comparisons*

A SWAT model was constructed for the Willow River watershed to help guide remediation efforts. The AVSWAT interface discretized the watershed into 27 subbasins based on 10-m digital elevation model datasets (USGS, 2005). The intersection of the subbasin boundaries, soils data set (NRCS, 2006), and 30-m land use grid (WDNR, 1998) produced 532 hydrologic response units (HRUs). The soils data were simplified by averaging the properties of soils in the same hydrologic group within each subbasin prior to the intersection to avoid an excessive number of HRUs. The primary calibration data set consisted of daily flows and monthly loads of sediment and total phosphorus from water year 1999 (October 1998 through September 1999) (Lenz *et al.*, 2003). Because model calibration to a single year of data may be unreliable, additional data were used to constrain the model, including crop yields, estimates of reservoir sedimentation, and phosphorus content of trapped sediment. To simplify interpretation, the model was parameterized to preclude both sediment erosion or deposition in the channel. In comparison to the water-year 1999 data set, the model simulated daily flows with a Nash-Sutcliffe coefficient of efficiency (E_{NS}) of 0.70, monthly sediment loads with an E_{NS} of

0.63, and monthly total phosphorus loads with an E_{NS} of 0.47 (Nash and Sutcliffe, 1970; Legates and McCabe, 1999).

A number of problems arose during model calibration, where the model did not behave as described, or where model output deviated from observed data or expected values enough to bring into question model algorithms. We describe below some of the more important of these problems, and how each problem was either solved or avoided in our construction of a workable SWAT model of the Willow River watershed, hereafter referred to as the calibrated model. In most examples, results are presented as a contrast between the problematic model runs and the calibrated model runs. Typically the models were run for 15 years, with annual average output being calculated from the last 10 years, ignoring the first 5 years for model warm up.

Results and Discussion – *Modeling Pitfalls to Avoid*

Persistent Alfalfa

Dairy farmers commonly grow alfalfa in rotation with corn and sometimes soybeans. Representative rotations included in the Willow model are two years of corn followed by three years of alfalfa (C2A3), and three years of corn, one year of soybeans, and three years of alfalfa (C3S1A3). Both corn and soybeans require annual tillage and planting, whereas alfalfa is a perennial that requires no tillage after planting the first year. Consequently, sediment and nutrient loads are typically much greater from corn and soybean fields than from alfalfa fields. Because of the large areas of these crops grown in the Midwest USA, and because of the large differences in sediment and nutrient yields between alfalfa and row crops (corn and soybeans), it is critical that SWAT be able to simulate rotations of these crops properly.

However, in simulations using the downloaded SWAT2000 engine, once alfalfa was planted it persisted for all remaining years of the simulation. In some respects, parts of the model may be rotating corn with alfalfa, as calculated nitrogen fixation for a C2A3 rotation rises and falls in a 5-year sequence as might be expected. Nonetheless for HRUs with either C2A3 or C3S1A3 rotations, model yields of sediment and phosphorus approach those for alfalfa alone and are thus critically underestimated.

To avoid the problem, one could disallow rotations and grow only continuous corn, soybeans, and alfalfa. However, continuous monocultures could result in unrealistic soil nutrients balances and cause significant errors in nutrient yields. A better solution was to correct the model code, as was done by Baumgart (2005), who kindly provided the executable copy that was used to build the calibrated model of the Willow watershed. Compared to Baumgart's SWAT version, the original SWAT code underpredicted sediment yields by 75% and phosphorus yields by 63% for the C2A3 rotation.

Excessive Denitrification

During model calibration runs, corn yields were consistently underpredicted because of nitrogen stress. Model output indicated that about 75% of nitrogen applied to corn HRUs was being lost to denitrification, whereas literature values were closer to 15%. The original SWAT2000 engine did not give access to denitrification parameters.

Again Baumgart (2005) provided a modified SWAT executable that allowed denitrification to be tuned. The Willow model was then parameterized to allow only about 11% denitrification, which improved corn yields substantially. Even still, the model had problems with nitrogen stress caused by loss to leaching, and to bring corn yields up to observed values the nitrogen auto-fertilization routine in SWAT was activated. Auto-fertilization is very useful, but it does complicate interpretation of soil nitrogen budgets and consequent loads to aquatic systems. Because nitrate loads to the Mississippi River from the Midwest are the primary cause of hypoxia in the Gulf of Mexico (Goolsby, 2000), refining model algorithms that simulate nitrogen transport is important. Apparently SWAT2005 has incorporated access to denitrification parameters, which is a useful step in this direction.

Closed Drainages and Loss of Infiltration

Closed drainages are a common landscape feature in the glaciated Midwest USA. About 29% of the Willow River watershed drains to closed depressions identifiable from 1.5-m contours of 1:24,000-scale topographic maps; conceivably additional areas drain to unidentified closed depressions shallower than 1.5 m. Under current conditions, such depressions rarely if ever spill. Consequently they play an important role in trapping sediment and phosphorus that otherwise would be delivered to the channelized network. However, during the processing of the digital elevation grid by the AVSWAT interface, such depressions are artificially filled in order to clarify the delineation of subbasins. Closed depressions, and their important water-quality function, are thereby lost from the model.

Fortunately, SWAT provides tools to address such a problem. In SWAT, Ponds are conceptual water bodies, one per subbasin, that receive drainage from a specified fractional area of that subbasin. In the Willow model, the aggregate area of closed drainage within each subbasin was routed to a Pond, which was parameterized to trap all sediment and phosphorus, and to infiltrate all influent water. In theory, this would allow the model to closely mimic closed drainages by trapping all sediment and phosphorus while passing water to groundwater recharge, which after some time should smoothly contribute to baseflow in the main channel.

In practice, Ponds in the model did trap sediment and phosphorus correctly; however, they also incorrectly trapped all infiltrated water as well, rather than contributing to groundwater recharge. When Ponds were added to the Willow model to simulate the 29% of landscape occupied by closed depressions, the basin-wide yields of sediment, phosphorus, and water all declined by about the same percent (29-31%). The infiltrated water was apparently being trapped in the model's shallow aquifer storage component, rather than being passed along to groundwater recharge. According to model output, the water in shallow aquifer storage continued to increase by about 50 mm each year the model was run, corresponding approximately to the annual reduction in water yield from the basin. Infiltration from other surface-water bodies in SWAT may suffer the same fate, though we investigated only Ponds.

This problem appears to be a relatively simple error in model code wherein water in shallow aquifer storage is not passed along to groundwater recharge. In the absence of a code correction, we developed a work-around by allowing Ponds to spill as slowly as possible, in an attempt to mimic groundwater discharge to the river. To do this, Pond infiltration was disallowed by setting bottom hydraulic conductivity to zero, and each Pond was given a large emergency volume and a long hydraulic response time (parameter NDTARG set to 500-1000 days) to smooth outflow. The result was marginally successful: Pond outflow could be slightly smoothed, but not to the degree we presume necessary to mimic groundwater discharge well. Still, overall the Pond tool in SWAT did allow closed drainages to be effectively simulated: sediment and phosphorus were trapped, and slow surficial outflow allowed water yields to match observed values.

Extraneous Phosphorus Loads as Chlorophyll from Subbasins

In a SWAT model, if reservoirs are removed and stream water-quality routines deactivated, the amount of phosphorus leaving the land (subbasins) should be effectively the same as that leaving the watershed outlet (lowermost reach), because the channel simply passively transports phosphorus loads without alteration. This was true for the Willow model. However, when stream water-quality routines were activated, phosphorus yields from the Willow watershed suddenly increased by 19%, which was very odd, because the losses from the land remained the same as before. The watershed was losing more phosphorus at its outlet than was being lost from the land. What was the source of the extra phosphorus reaching the outlet?

The answer is that in addition to the phosphorus load transported from the land to the channel, SWAT also adds a chlorophyll load. From the perspective of the land, this chlorophyll load appears to be phosphorus-free and unrelated to the phosphorus budget of the soil. If stream water-quality routines are not activated, then the channel likewise ignores this chlorophyll load; it is simply passed downstream, unaltered. However, if stream water-quality routines are activated, then the stream interprets this chlorophyll load as algae, and further presumes that this algae has a phosphorus content. Upon decomposition of this perceived algal load, its presumed phosphorus content is released, thereby providing the source of phosphorus loading that suddenly appeared in the Willow model upon simply activating the stream water-quality routines. This phosphorus load appears to be extraneous and unrelated to the actual phosphorus budget of the land surface.

Two options exist to avoid this problem. The first is to avoid activating the stream water-quality routines. The second is to reduce the phosphorus content of algae in the model (parameter AI2) to a very small value, thereby making the added phosphorus from the subbasin chlorophyll load negligible. Both options are functional, though neither is satisfying. We believe that the algorithm that adds a chlorophyll load from the subbasins to the channel needs to be re-examined, or at least integrated with the phosphorus fractions transported from the subbasins.

Large Sediment Yields and Alternate Calibrations

Under default parameterization, SWAT produced sediment yields nearly twice those expected from the landscape in the Willow model, about 44 t km^{-2} compared to the expected 23 t km^{-2}. In our case, the “expected” value is admittedly poorly known, as it was determined from a series of considerations including approximate trapping efficiencies of reservoirs, estimated sediment thicknesses in the reservoirs, sediment delivery ratios from watersheds the size of the Willow, and estimated gross erosion rates from cropland and pasture. Yet it is our belief that other colleagues in the Midwest have had similar experiences, and that typically SWAT has had to be parameterized to dial down sediment yield, rather than the converse. In the Willow model, sediment yield was reduced by two main parameterizations. First, another code modification by Baumgart (2005) changed runoff time-of-concentration calculations to be based on subbasin area and channel length, rather than on HRU area and channel length. Second, the modified universal soil-loss equation cropping practice (MUSLE P) factor was reduced from 1.0 to 0.7, which we used simply as a scaling factor to reduce sediment yields by 30%.

We offer this observation more as a comment rather than critique of SWAT. In the model, HRUs are depicted as contiguous uniformly sloping landscape units draining directly to channels without intermediate sediment traps. In reality, however, an HRU likely comprises many disaggregated, non-contiguous land parcels, and runoff from these parcels has a tortuous path from upland to stream channel with many potential intermediate sediment traps. One might therefore expect that net erosion from real landscape units would be less than that predicted by MUSLE calculations and the HRU concept.

Alternatively, if one accepts the sediment yields from the landscape as calculated by SWAT, yet the consequent modeled loads in the stream are greater than measured loads, then the excess sediment must be trapped at the upland/stream interface, namely in the floodplain and channel system. Up to this juncture our calibrated model had disallowed any channel processes, hence it was called the “passive channel” model. We next created an alternate calibrated model, called the “active channel” model, which received greater sediment and nutrient loads from the uplands (by re-setting the MUSLE P factor back up to its default value of 1.0), but then deposited these excess loads in the channel and floodplain system by adjusting sediment entrainment and phosphorus settling parameters. The net effect on loads at the watershed outlet was minimal, and the active channel model achieved nearly identical Nash-Sutcliffe calibration statistics as the passive channel model. In using SWAT to test relative effectiveness of agricultural BMPs, conclusions will likely depend on which calibrated model – the passive channel, or active channel, version – is chosen as the baseline scenario.

Summary and Conclusions

Application of SWAT2000 to an agricultural watershed in the upper Midwest USA uncovered several issues that needed to be addressed to achieve model calibration. Changes in model code allowed proper rotation of alfalfa with other crops and a reduction in denitrification loss of applied nitrogen. While the Pond tool in SWAT was very useful in simulating sediment and phosphorus trapping by closed depressions, infiltration from these depressions was excluded from groundwater recharge and thus did not correctly contribute to stream baseflow. SWAT added a chlorophyll load from the subbasins to the channel, which was converted to phosphorus when stream water-quality routines were activated, thereby resulting in an extraneous phosphorus load that was unrelated to the phosphorus budget of the soil column. SWAT appeared to deliver greater than expected sediment yields under default parameterization, and whether the modeler decides to trap this excess sediment in the uplands or floodplain can lead to alternate calibrated models. This problem points to a fundamental gap in our knowledge base. There are many field-scale studies that have verified the USLE and its variants in calculating gross erosion. There are many watershed-scale studies that have measured net erosion (total sediment yield) leaving the watershed at its outlet. But what happens between the field and the watershed outlet remains difficult to characterize.

The Midwest USA is one of the most agriculturally productive and intensively cultivated regions of the world. Nonpoint-source loads of sediment and nutrients from this agricultural activity have had significant impacts on the water quality of both freshwater and marine systems. Watershed modeling is an important tool to facilitate balancing the need for agricultural production with the desire to protect aquatic resources. The development of SWAT has made important contributions to the science of watershed modeling. The issues with SWAT raised in this paper are mostly soluble by model code revisions, which we feel would incrementally, but significantly, increase the power of SWAT to solve real problems in watershed management.

Acknowledgements

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Application of SWAT Model on Three Watersheds within the Venice Lagoon Watershed (Italy): Source Apportionment and Scenario Analysis

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Abstract

Aim of this study was the analysis of the source apportionment of three river basins within the Venice Lagoon Watershed (VLW), in the North-Eastern part of Italy, which on the whole cover about 700 km^2 (i.e. about 35% of the VLW surface area). The three watersheds are characterised by a very intensive agriculture and landfarming systems. Moreover, in this area the groundwater recharge to surface waters significantly contributes to the total load discharged into the Venice Lagoon. SWAT model application allowed to describe the hydrologic and agricultural characteristics of the watersheds and to assess the source apportionment in terms of point and diffuse sources. Furthermore, SWAT model allowed the evaluation of the benefit due to the application of agri-environmental measures through the simulation of a “better-business” agricultural scenario versus the “business-as-usual” scenario. Out of a total annual nutrient load of about 2200 tNy^{-1} and of about 140 tPy^{-1}, the dry weather diffuse sources (i.e. groundwater/spring recharge and tributary/irrigation channels coming from bordering watersheds) resulted to be the most important (65% in case of nitrogen and 35% in case of phosphorus). SWAT outputs indicated a runoff load contribution of about 20% of the total nitrogen load and of about 30% of the total phosphorus load; agricultural runoff resulted to be about 2/3 of this runoff load. The simulation of the Better Business scenario indicated a possible reduction in the agricultural runoff loads of about 50% in

case of nitrogen and of about 15% in case of phosphorus, which corresponds to a decrease in the total annual load of about 5-7%.

KEYWORDS: SWAT, Venice Lagoon Watershed (VLW), point and non point sources, source apportionment, scenario analysis.

Introduction

River basin source apportionment is an important issue for water quality management. In fact, point and non point sources are quite different in terms of pathways and temporal dynamics and instream measures do not give information about the contribution of point and diffuse sources on the total load. In particular, the assessment of diffuse sources may be uncertain, since their transport dynamics are strictly related to the hydrologic cycle and agricultural practices of the watershed. Diffuse loads reach the surface waters through different pathways, which could be surface runoff and erosion processes during rainstorm events, but also groundwater recharge during dry-weather conditions (i.e. when groundwater flows into surface waters). However, it is very difficult to quantify the not rain-driven diffuse loads because the temporal scale of the subsurface runoff processes is generally much longer than the surface runoff time scale. Therefore, the analysis of the principles of interactions between groundwater and surface waters should be the basis for an effective water resource management (Sophocleous, 2002).
This study analyses the source apportionment of three watersheds within the Venice Lagoon Watershed (VLW). The nutrient discharge of the VLW has been widely studied because of the critical effect on the eutrophication of the Venice Lagoon (Bendoricchio et al., 1999).

Since the VLW is characterised by a very intensive agricultural land use, a multicriteria approach was used in order to map the risks of agricultural pollution for water resources (Giupponi et al., 1999) and agri-environmental policy measures were analysed.
Moreover, the Northern part of the VLW is characterised by a significant groundwater and springs recharge area, which significantly contributes to the nitrogen load discharged into the Lagoon.

SWAT model was applied in order to describe the agricultural characteristics of the watersheds and to assess the source apportionment in terms of point and diffuse sources. In particular, it was possible to split non point sources into a rain-driven and a not-rain-driven component. Finally, SWAT model allowed the simulation of a better-business agricultural scenario, in which alternative management strategies were implemented.

Materials and Methods

Study area

The Venice Lagoon Watershed (VLW) is located in the North-Eastern part of Italy. Three river basins were studied within the VLW: the Naviglio Brenta-Bondante (NBB), the Dese-Zero (DZ) and the Vela (VL) watersheds (Fig.1). On the whole, the three basins cover about 35% of the VLW surface area and are characterised by a very complex network of irrigation channels which very often receive also direct sewage discharges. Moreover, the three watersheds are located within the groundwater recharge area (Fig.2), which significantly contributes to the hydraulic and nutrient load transported by the main channels. As shown in Table 1, agricultural activities constitute the dominant land use within the area.

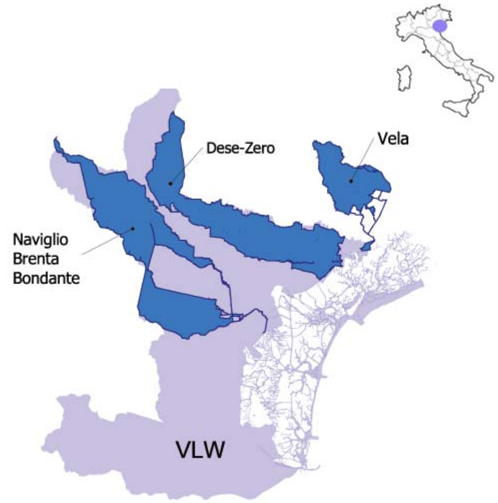


Figure 1. Study area: Dese-Zero watershed within the Venice Lagoon Watershed (VLW).

Table 1. Characteristics of the three watersheds.

	NBB	DZ	VL
Drainage area (km^2)	293	310	102
Urban Area	24%	26%	13%
Agricultural Area	69%	72%	86%
Dominant crops	Corn, soy, wheat, sugar beet	Corn, soy, wheat, sugar beet	Corn, soy, wheat, sugar beet

Input data

Data available for the modelling of the watersheds were:

1. A 50 x 50m Digital Elevation Model grid;
2. Daily series of meteorological data (i.e. precipitation, wind speed, solar radiation and dew point temperature) from 1993 to 2004-2005;
3. Soil geomorphologic and textural characteristics aggregated into 20 soil classes;
4. To better describe the main agricultural uses within the watershed, 9 four-year crop rotation classes were defined out of a total of 13 land use classes. Year 2001 Census data of livestock categories were used to compute the respective loads on a municipal scale.
5. Streamflow daily data were available at 11 gauging stations during different years;

6. Water quality data on monthly basis, were available from 2001 to 2005 at 23 monitoring stations;
7. An hourly water quality data series was also available for specific rainstorm events on Dese river;
8. Anthropogenic load measurements were available for all significant point sources: 8 wastewater treatment plants (WWTPs) and 86 industrial facilities (IFs);
9. Nitrate in precipitation measures were also available.

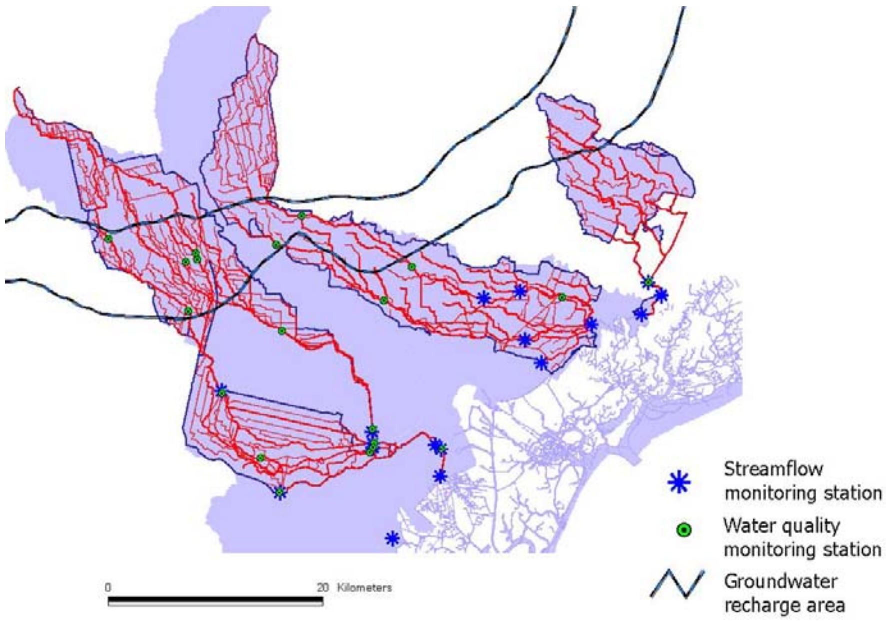


Figure 2. NBB, DZ and VL watershed characteristics: hydrography, water quality monitoring stations, streamflow monitoring stations, spring and groundwater recharge area.

Loads computation

The analysis of the available data allowed to identify two main classes of sources:

1. *The point loads,* which consisted on all localized emissions, in terms of Waste Water Treatment Plants (WWTPs), industrial facilities' discharges and direct sewer discharges into surface waters.
2. *The diffuse loads,* which were splitted in*:*
   - *Dry weather loads*, deriving from the irrigation channel network (mostly coming from bordering basins) and from groundwater recharge zone. Although irrigation channels cannot properly be considered diffuse sources, they were considered as a diffuse anyway, since their polluting load to the streams cannot be easily identified as a localized emission.
   - *Rain driven diffuse loads*, i.e. nitrogen loads carried by surface runoff.

Concentration and flowrate measures enabled to quantify the load coming from the irrigation channel system. BOD load from direct sewer discharge was estimated by difference from the instream organic load balance. as follows:

$$BOD_{measured} = BOD_{WWT+Industries} + BOD_{Irr.channels} + BOD_{sewage}$$

where:
BOD $_{measured}$ and BOD $_{Irr.channels}$ were calculated from water quality instream measurements, BOD $_{WWT+Industries}$ was calculated from the discharge data measurements, and BOD $_{sewage}$ was the unknown parameter estimated by difference from the others.

Such an organic load attributed to the direct sewer discharges was eventually converted into nitrogen load by means of a specific emission factor (i.e. 60 g BOD PE^{-1} d^{-1}, 12 g N PE^{-1} d^{-1}, PE = People Equivalents). Therefore, it was possible to estimate the groundwater recharge loads:

$$\text{N-tot}_{measured} = \text{N-tot}_{WWT+Industries} + \text{N-tot}_{Irr.channels} + \text{N-tot}_{sewage} + \text{N-tot}_{groundwater}$$

where:
N-tot $_{measured}$ and N-tot $_{Irr.channels}$ were calculated from water quality instream measurements, N-tot $_{WWT+Industries}$ was calculated from the discharge data measurements, N-tot $_{sewage}$ was estimated on the basis of the PE_{BOD} and N-tot $_{groundwater}$ was the unknown parameter estimated by difference from the others.
The dry weather source estimates were validated by means of QUAL2E model (Brown and Barnwell, 1987) and were used as input data for SWAT model according to the methodology used in Salvetti et al. (in press).

Model set-up

In this study the Basins 3.1 version of SWAT model was applied, because a subbasin predefined delineation was needed.
The NBB basin was subdivided into 33 subbasins and 169 HRUs, the DZ into 42 subbasins and 212 HRUs and the VL into 18 subbasins and 105 HRUs.
As already said, the three basins receive a significant groundwater recharge from bordering watersheds. Although SWAT model enables to simulate the groundwater recharge to surface water bodies, it is not able to consider the contribution to the groundwater recharge of the bordering watersheds. SWAT groundwater recharge simulations were in fact sistematically underestimated. Therefore, the groundwater recharge area was modelled with an additional inlet contribution.

As far as parameter calibration is concerned, the cultural parameters BLAI, BIO_E and HVSTI were incremented for corn and wheat, in order to obtain a better description of the Italian cultures and land management.
As already observed, the studied basins are quite flat areas (average slope of about 0.005) and characterized by a very complex irrigation/channel network. Therefore, the resulting watershed hydrology is quite different from the “natural” watershed hydrology. The hydrological calibration was performed to obtain mm_{rain}/mm_{runoff} ratios typical for the study area (i.e. about 20-30%). As a result, the Curve Number and the OV_N parameter were accordingly adjusted.
Sediment and nutrient calibration was carried out considering the following SWAT parameters: USLE_P, ERORGN, ERORGP, PHOSKD, NPERCO and FRT_LY1.
Table 2 summarises the calibrated parameters.

Table 2. Values for the calibrated parameters adopted during SWAT simulations

Parameter identifier	Calibrated value	Parameter identifier	Calibrated value
BLAI (corn)	7	USLE_P	0.3
BLAI (wheat)	6.5	ERORGN	1
BIO_E (corn)	75	ERORGP	0.1
BIO_E (wheat)	90	PHOSKD	200
HVSTI (corn)	0.76	NPERCO	0.5
HVSTI (wheat)	0.6	FRT_LY1	0.5

Results and Discussion

SWAT simulations

The results of the hydrologic calibration were evaluated by means of the *Nash-Sutcliffe* coefficient of efficiency (Nash and Sutcliffe, 1970), that showed a good fit on a monthly basis (median E of 0.65). On a daily basis, the simulations at the basin closures showed a good agreement with measured data. The Nash-Sutcliffe efficiency (E) was of about 0.4, ranging from 0.11 to 0.67 from year to year (In Figure 3 an example for the Dese-Zero closure is reported).

Water quality data for the only Dese river coming from measurements during rainstorm events allowed to quantify the daily nitrogen load and were used for the water quality calibration. As Figure 4 shows, the instream simulated loads fitted well the measured actual loads. A series of sequential rainy days can be considered as a single stormflow event. In Figure 4 both rainy days and stormflow events are shown. The cumulative loads, corresponding to these stormflow events, showed a good agreement with the actual measurements (mean error: ± 17%).
For the three watersheds SWAT outputs indicate a total annual load of about 2200 tNy^{-1} and of about 140 tPy^{-1} (Table 3). The most important contribution to the total nutrient load is given by NBB (about 50%), because of its higher streamflow.

In Figure 5 the global source apportionment is reported. As it can be seen, the dry weather diffuse load is the most important source.

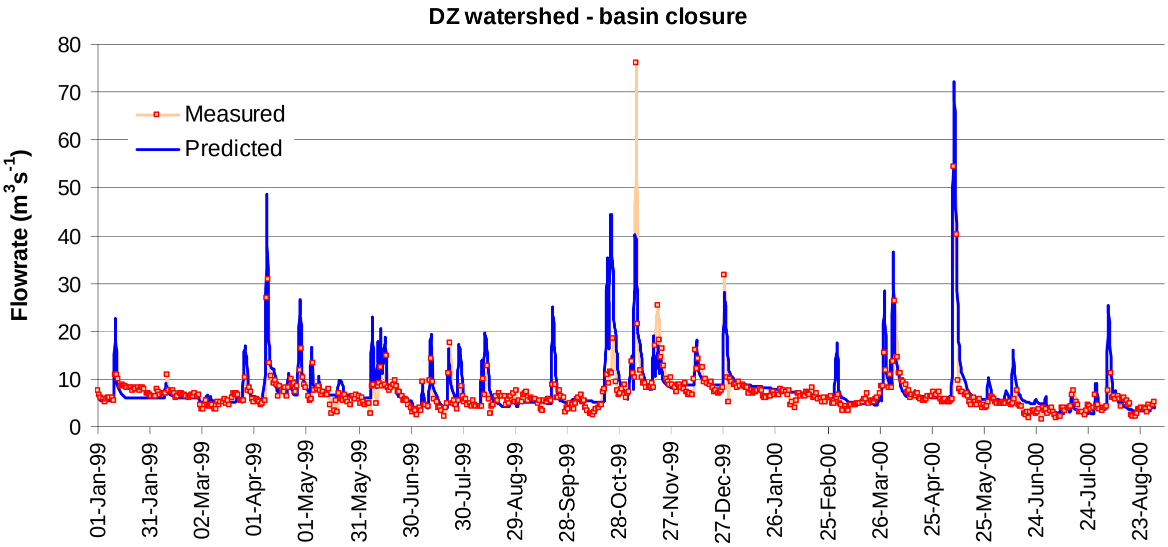


Figure 3. Daily streamflow measured and simulated by SWAT model at the Dese-Zero river basin closure

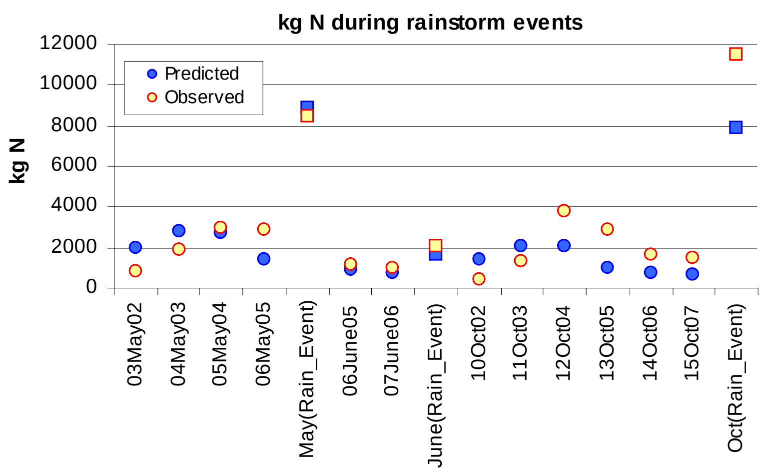
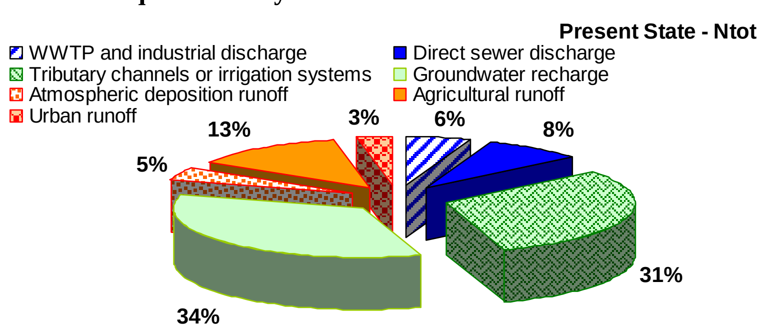


Figure 4. Measured and simulated loads during rainstorm events. The circles represent the daily nitrogen loads (kg N d^{-1}), while the squares indicate the cumulative stormflow event loads (kg N event^{-1}, i.e. the sum of the sequential daily loads that constitute the whole stormflow event).



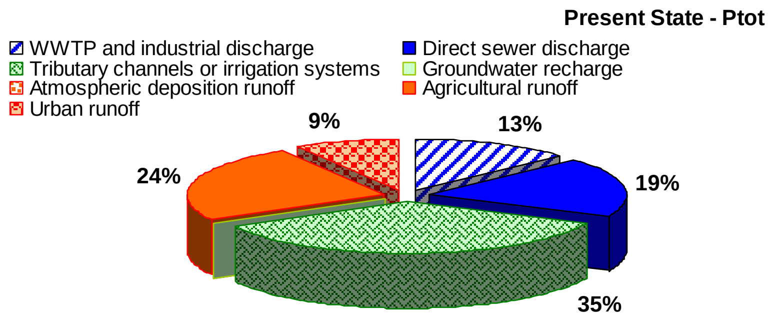


Figure 5. Source apportionment of the total annual load.

Table 3. Total annual load and runoff load at the present state and in the agricultural scenario.

		Total annual load		Annual runoffload	
		N tot	P tot	N Tot	P Tot
		t N y^{-1}	t P y^{-1}	t N y^{-1}	t P y^{-1}
NBB	Business-as-usual	1155	64	181	23
	Better-Business	1101	60	134	19.5
DZ	Business-as-usual	696	51	199	19
	Better-Business	636	49	133	17
VL	Business-as-usual	335	25	89	7
	Better-Business	310	24	63	6
Sum	Business-as-usual	2186	140	469	49
	Better-Business	2047	133	330	42.5

As far as nitrogen is concerned, nitrogen from groundwater recharge resulted to be about 35% of the total annual load. A similar contribution derived from channel load coming from external basins, while point loads (WWTP, industrial discharge and direct sewer discharge) constituted about 15% of the total annual load. SWAT ouputs indicated for the runoff loads the remaining 20%, in which the most important source derived from agricultural runoff (63% of the runoff load), followed by atmospheric deposition (22%) and urban runoff (15%).
The results in terms of runoff apportionment (i.e. 26% of the total annual load) are coherent with what was found for other Italian watersheds (Salvetti et al., 2006). Runoff loads, in fact, if during rainstorm events constitute the most of the instream total load (up to 80-90%, Azzellino et al., 2006), on an annual basis contribute much less.
As far as phosphorus apportionment is concerned, external channel load constituted about 35% of the total annual load and a similar contribution derived from point loads (because of the absence of groundwater recharge load). Runoff loads amount to about 33%, in which about 2/3 comes from agricultural runoff and 1/3 from urban runoff.

In order to test the potential of the SWAT implementation in the VLW for supporting future environmental policies, a scenario analysis was carried out by comparing business-as-usual management with the implementation of better-business agricultural management strategies. Hence, the indications of environmental legislations (both European and national) for reducing nitrate discharges were used as reference for the scenario of the full implementation of such agricultural policy measures.
The scenario analysis showed the effectiveness of the proposed measures imposing changes of agricultural management, especially for nitrogen (Table 3). In fact,

agricultural nitrogen runoff load was almost halved (being reduced from about 300 t N y^{-1} to about 150 t N y^{-1}) and become about 7% of the total annual nitrogen load. Phosphorus runoff load decreased of about 16% and become about 21% of the total annual phosphorus load. Therefore, the better-business scenario assessment allowed to obtain a total annual nutrient load decrease of about 5-7%, corresponding to a total annual load of about 2047 t N y^{-1} and 133 t P y^{-1}.

Finally, a selection of the most significant model outputs were passed to a decision support system software (mDSS), in order to provide an effective interface with policy makers, allowing to display the main feature of the territorial system and the cause-effect links affecting nutrient pollution loads, together with a facilitated comparative analysis of the alternative scenarios.

Conclusions

The application of SWAT model allowed to assess a source apportionment for three watersheds within the Venice Lagoon Watershed. Model outputs pointed out the following conclusions:

- At the present state, the three basins discharge a total annual nutrient load of about 2200 tNy^{-1} and of about 140 tPy^{-1};
- Out of this total load, the dry weather diffuse sources (i.e. groundwater/spring recharge and tributary/irrigation channels coming from bordering watersheds) constitute the most important source (65% in case of nitrogen and 35% in case of phosphorus);
- Runoff loads cover about 20% of the total nitrogen load and about 30% of the total phosphorus load. Agricultural runoff constitute about 2/3 of the runoff load;
- SWAT model allowed to simulate better-business agricultural scenario, in which alternative management strategies were implemented. SWAT outputs indicate a reduction in agricultural runoff loads of about 50% for nitrogen and of about 15% for phosphorus, with a decrease in the total annual load of about 5-7%.
- A selection of the most significant model outputs has being implemented in a decision support system software (mDSS), in order to provide an effective interface with policy makers.

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Phosphorus Model Development with the Soil and Water Assessment Tool

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Abstract

The ability to accurately simulate phosphorus transport at watershed scales is essential to environmental managers for making pollutant reduction decisions. Computer models are inexpensive tools that can quickly identify environmentally sensitive areas. Phosphorus transport is dependent on hydrologic properties inherent to soil and land management. In order to effectively model P transport, an efficient hydrologic model is necessary. The SWAT model simulates P movement in surface runoff and in-stream which includes soluble and sediment attached P. The SWAT model has been updated to include the linking of P surface and subsurface algorithms so that P can travel throughout the entire soil profile. SWAT PBIAS results for the calibration and validation periods for the South Fork watershed in Iowa for lateral flow were -33.3 and -38.9. The tile flow calibration and validation periods PBIAS SWAT results were -2.7 and 35.6. These PBIAS results range from very good to good when considering P water quality data. Overall, the new P algorithms have proved to be a useful since P can now move throughout the soil profile projecting more realistic values.

Keywords: SWAT, phosphorus, subsurface flow, water quality

Introduction

Phosphorus is recognized as a contaminant that causes adverse conditions in surface water bodies and is one of the most important nutrients in agricultural systems, along with nitrogen. Phosphorus is an essential plant growth requirement for many physiological processes. Fertilizer is generally applied to crops based on the nitrogen requirement resulting in an excess of P application. With the establishment of increasing livestock production facilities, P runoff from agricultural fields has become scrutinized with growing public concern. Watershed simulation models are a relatively inexpensive and are a rapid method to track P transport pathways and their sources and sinks. Simulation models reflect the state of scientific knowledge and rely on measured data collection to adequately reflect environmental processes. A general model limitation is the availability of measured water quantity and quality data.

Conceptual models of P transfer are presented in Haygarth and Jarvis (1999) and Dougherty et al. (2004) that basically emphasize how hydrology heavily impacts potentially mobile P in which both hydrology and mobile P are impacted by the properties inherent in soil and land management. Sauer et al. (2003) evaluated Tipton Creek (202 km^2), which is a portion of the South Fork Watershed (SFW), for surface P transport predictions in which 84% is in row crop production (corn and soybeans). This author found that after combining the flow data with P concentrations an average of 0.52 kg ha^{-1} soluble P (taken via stream samples) was lost from the application site from April 1, 2000 to April 1, 2001.

Having the necessary hydrologic pathways is imperative to predicting P loads or concentrations. Subsurface drainage systems can be a significant source of pollutants (Cambardella et al., 1999; Northcott et al., 2001). Over 25% of U.S. cropland requires drainage alterations (Pavelis, 1987) with artificial drainage needed on more than 50% of the agricultural land in some states (Skaggs et al., 1992). Tile flow studies are imperative to nutrient transport since hydrology has a large role in their movement (Du et al., 2005; Green et al., 2006). Du et al. (2005) assessed the applicability of tile flow in SWAT2000 for nine years for Walnut Creek, a 51 km^2 watershed in Iowa. SWAT2000 estimated monthly and daily flow with Nash-Sutcliffe efficiency (NSE) values up to 0.72 and 0.47, respectively. Differences between the models that include tile flow lie in the simulation of flow and water quality.

This study updates the soluble P lateral, percolation, and tile flow soil processes in the Soil and Water Assessment Tool (version 2005). The objective of this study is to evaluate the model's ability to adequately simulate soluble P transport and fractionation between percolation, lateral and tile flow pathways.

Phosphorus Chemistry

Phosphorus can be transferred within pools in the soil-plant continuum that can result in available runoff P. The rate and amount of P transferred between pools depends on environmental factors such as biological, chemical, physical, and hydrologic components as well as the contributing source. The amount of P available for transport is constantly changing due to establishing chemical equilibrium. Phosphorus can easily move when sediment-containing P is transported from agricultural land via runoff and erosion. This transport of P is why agricultural industries are being targeted as areas in need of immediate attention, and regulation, for better nutrient management practices. In many soils, P content is greatest in the surface horizons, when compared to the subsoil due to greater biological activity, fixation of P, and most organic material residing in the surface layers.

Methodology

SWAT Modification for Restrictive Layer

To accompany tile drainage systems, soil water routing is modified to predict water table depth by creating a restrictive soil layer at the bottom of the profile for a SWAT version 2005 (Green et al., 2006).

SWAT Phosphorus Model Background

The SWAT model versions earlier than 2005 transported P only through the top 10 mm and into the second soil profile layer after which it was unaccounted for. In response to concern of the mobility of P, SWAT2005 has been altered to accommodate its subsurface movement. When P is applied in amounts higher than the P soil sorption capacity or during large rain events, it can be transported via multiple pathways (i.e. leaching, lateral flow, and surface runoff). Figure 1 illustrates the major components of the P cycle modeled in SWAT before the changes to soluble P transport pathways.

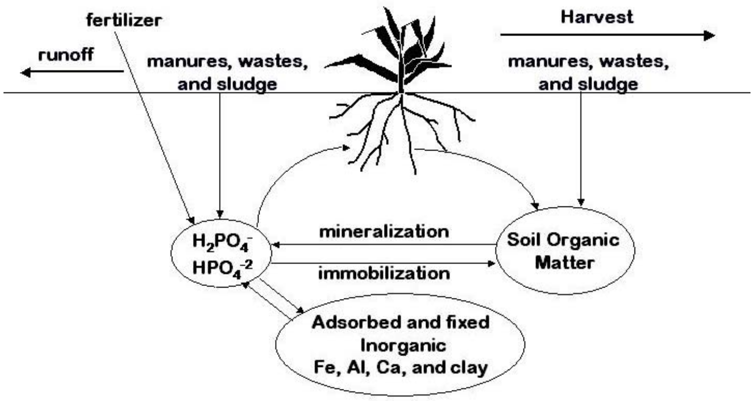

Figure 1. Phosphorus fractionation in SWAT versions prior to SWAT2005.

The original labile P SWAT module was adapted from the EPIC version as described in Jones et al. (1984) and Sharpley et al. (1984) but the movement of P with soil depth beyond 10 mm was not incorporated. The areas that are covered in depth below are those that have been altered, otherwise the process equations remain the same as published by Neitsch et al. (2001a).

Leaching

When plants take P from the soil solution present in the root zone, a concentration gradient is created in the soil solution matrix. Diffusion, the migration of $H_2PO_4^-$ and HPO_4^{2-} ions in the soil solution in response to a concentration gradient, is the primary mechanism of P movement in the soil. Before the revision made for SWAT2005, the soluble P was simulated by the model to leach only from the top 10 mm of soil into the first soil layer. The mass of solution P leaching into the first soil layer was calculated as:

$$\mathrm{P_{perc}} = \frac{\mathrm{P_{solution,surf}} \ \ \mathrm{w_{perc,surf}}}{10 \ \ \rho_{\mathrm{b}} \ \mathrm{depth_{surf}} \ \ \mathrm{k_{d,perc}}} \qquad (2)$$

where P_{perc} is the amount of P moving from the top 10 mm into the first soil layer (kg P ha^{-1}), $P_{solution,surf}$ is the amount of labile P in the top 10 mm (kg P ha^{-1}), $w_{perc,surf}$ is the amount of water percolating to the first soil layer from the top 10 mm on a given day (mm), ρ_b is the soil bulk density of the top 10 mm (Mg m^{-3}), $depth_{surf}$ is the depth of the surface layer, and $k_{d,perc}$ is the P percolation coefficient. The $k_{d,perc}$ is calculated as the ratio of the labile P concentration in the surface 10 mm of soil to the concentration of P in percolate.

In this study, the soluble P can move throughout the depth of the soil profile. The depth of each soil layer and the associated soluble P in each soil layer is calculated as:

$$Psol_{zi} = (sol\_z - sol\_zi) \qquad (3)$$

where $Psol_{zi}$ is the soluble P available in the soil layer between sol_z (the soil layer above) and sol_zi, the soil layer beneath.

Phosphorus Movement in Surface Runoff

The SWAT model simulates the movement of P from the landscape into surface runoff as:

$$P_{surf} = \frac{P_{solution,surf} \; Q_{surf}}{\rho_b \; depth_{surf} \; k_{d,surf}} \tag{4}$$

where P_{surf} is the amount of soluble P transported by surface runoff (kg P ha^{-1}), $P_{solution,surf}$ is the amount of labile P in the top 10 mm (kg P ha^{-1}), Q_{surf} is the amount of surface runoff on a given day (mm), and $k_{d,surf}$ is the P soil partitioning coefficient (m^3 Mg^{-1}) (Neitsch et al., 2001b). The $P_{solution\ surf}$ is impacted by the k_{dsurf} and the layers below are now impacted by a coefficient entitled "$phos_{kdsub}$." The soluble phoskd for the subsurface soil layers ($phos_{kdsub}$) are then multiplied by the soil depth and the soil bulk density in order to obtain an intermediate term that can be used in the percolation, lateral flow and tile flow of soluble P accounting.

$$xx = sol\_bd * (sol\_z - sol\_zi) * phos_{kdsub} \tag{5}$$

where *xx* is an intermediate term used in accounting for the bulk density of the soil (*sol_bd*), the difference between the upper (*sol_zi*) and the next lower soil layer (*sol_z*) and the phoskd subsurface phosphorus soil partitioning coefficient ($phos_{kdsub}$).

Model Evaluation Methods

The performance of SWAT was evaluated using the Nash-Sutcliffe efficiency (NSE) (Nash and Sutcliffe, 1970) and PBIAS statistical analyses to determine the quality and reliability of the predictions when compared to observed values. Percent bias (PBIAS) measures the average tendency of the simulated data to be larger or smaller than their observed counterparts (Gupta et al., 1999). Table 1 describes the ratings for recommended streamflow and P (as phosphate) assuming average uncertainty in measured data based on Harmel et al. (2006). This provides the basis for evaluating how well the altered SWAT algorithms performed.

Table 1. General performance ratings for recommended quantitative criteria, assuming typical uncertainty in measured data based on Harmel et al. (2006).

Rating	**NSE**	**Streamflow**	**P**
Very good	0.75<NSE≤1.00	PBIAS < ±10	PBIAS< ±25
Good	0.65<NSE≤0.75	±10≤PBIAS<±15	±25≤PBIAS<±40
Satisfactory	0.50<NSE≤0.65	±15≤PBIAS<±25	±40≤PBIAS<±70
Unsatisfactory	NSE ≤ 0.50	PBIAS ≥± 25	PBIAS ≥ ±70

Input Data

Watershed Description

The South Fork of the Iowa River covers 775 km^2, including tributaries of Tipton and Beaver Creeks (fig. 3). It is representative of the Des Moines Lobe, the dominant landform region of north-central Iowa. The terrain is young (about 10^4 years since the last glacial retreat), and therefore natural stream incision and development of alluvial valleys has only occurred in the lower parts of the watershed.

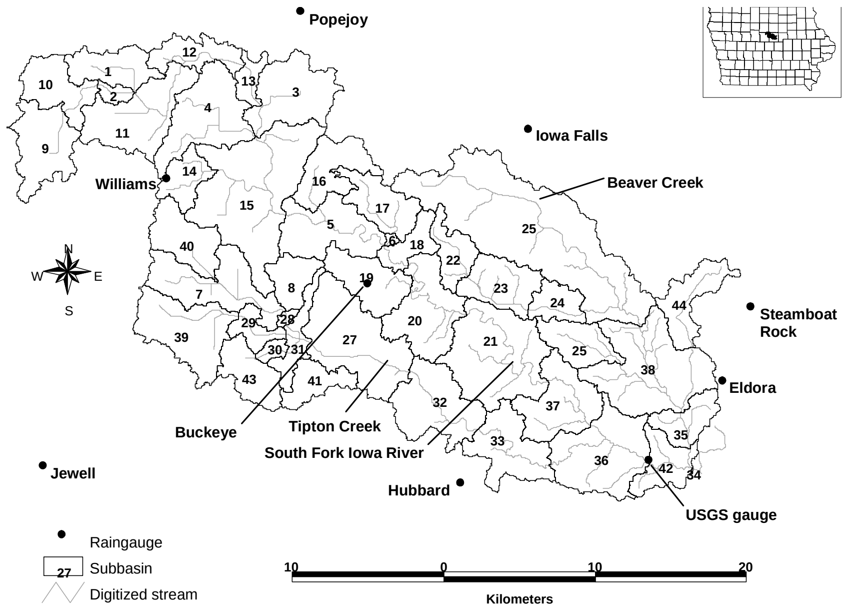

Figure 3. Distribution of rain and temperature gauges, USGS gauge (site 05451210), and subbasins in the South Fork watershed.

Details regarding the topographic, weather, precipitation, temperature, soils, hydrologic and land use data and calibration parameter values are presented in Green et al. (2006). The combination of land use and soil type resulted in 727 HRUs.

Tile Drains

Approximately 80% of the agricultural watershed is tile drained. This estimated value includes all of the soils that are not well drained and a few that are well drained but are surrounded by poorly drained soil. A 2.5 m depth to impermeable layer (depimp) and a standard tile drain depth of 1.0 m were used for the entire basin in this study to account for tile flow. The tile drains were used to reduce the water content to field capacity within 24 h; therefore, the time to drain (t_{drain}) soil to field capacity was set at 24 h. The drain tile lag time (g_{drain}) and depth to drainage (d_{drain}) were set to 96 h and 1 m, respectively. The best available information for the values of t_{drain}, g_{drain}, and d_{drain} was obtained from a representative of the South Fork Watershed Alliance (M. Tomer, personal communication, January 2005). Table 3 lists the ranges of adjusted parameters suggested by the SWAT model and the calibrated values of the adjusted parameters used for discharge calibration of the SWAT2005 model for the SFW. All other parameters were kept at the SWAT default values.

Table 3. Calibrated values of adjusted parameters for discharge calibration of the SWAT2005 model for the SFW.

Parameter	Description	Range	Calibrated Value
ESCO	Soil evaporation compensation factor	0.01 to 1.0	0.95
FFCB	Initial soil water storage expressed as a fraction of field capacity water content	0 to 1.0	0.8
Surlag	Surface runoff lag coefficient (days)	0 to 4	0.2
ICN	Based on the SCS runoff curve number procedure and a soil moisture accounting technique	0 or 1	1
CNcoeff[a]	Curve number coefficient	0.5 to 2.0	0.2
CN2	Initial SCS runoff curve number to moisture condition II	30 to 100	66-78
PHU	Potential heat unit (used for corn and soybeans)	1000 to 2000	1800

[a] Williams and LaSeuer, 1976.

Fertilizer and Manure Applications

The South Fork watershed contains some of the most intensive row-crop and livestock production in the Midwest. There are nearly 100 confined animal feeding operations in the watershed (433,500 km^2), most producing swine. An N-based manure application of 200 kg N ha^{-1} for each year of corn is assumed. *Results and Discussion*

Water Balance ***Table 4 indicates the importance of having the necessary water balance components accounted for in order to adequately allocate water and its constituents. The SFW with and without the tile drainage clearly indicate the misdirection policy managers would have in making decisions if this component was not included in the model. The tile drains allow for the movement of water (and its constituents) alleviating the upper soil layers of water therefore reducing surface runoff. Figure 3 illustrates the major components of the P cycle modeled with the recent soluble P developments including the restrictive layer that enables tile flow. The SWAT model, version 2005, allows for lateral subsurface flow, which is the zone between the restrictive layer below and the soil surface above (fig. 3).***

Table 4. Predicted hydrologic and P budget for the SFW from 1995 through 2004 with and without tile drainage and its calibration (1995-1998) and validation (1999-2004) simulations using the SWAT2005 model.

Hydrologic Component	With Tile Flow (mm) 1995-2004	Without Tile Flow (mm) 1995-2004	Calibration (mm) 1995-1998	Validation (mm) 1999-2004
Precipitation	770.6	768.0	787.5	758
Surface runoff	38.1	117.4	44.3	34.3
Lateral flow	5.2	0.40	5.5	5.3
Tile flow	158.79	0.0	196.55	182.0
Groundwater flow	0.0	11.7	0.0	0.0
Evapotranspiration	499.0	638.6	437.6	515.0
Potential ET	760.0	1191.6	698.2	775.5
Soluble P in Surface Runoff kg ha^{-1*}	0.185	0.259	0.018	0.017
Soluble P in Surface Runoff kg ha$^{-1\dagger}$	0.113	n/a	0.012	0.020
P Leached kg ha^{-1*}	0.562	0.104	0.362	0.537
P Leached kg ha$^{-1\dagger}$	0.581	n/a	0.378	0.524
Lateral Soluble P kg ha$^{-1\dagger}$	0.004	n/a	0.004	0.004
Tile Soluble P kg ha$^{-1\dagger}$	0.077	n/a	0.148	0.086

*SWAT model before P alteration to allow transport below 10 mm soil depth.

†SWAT model after P change that allows transport through the soil profile and through tiles.

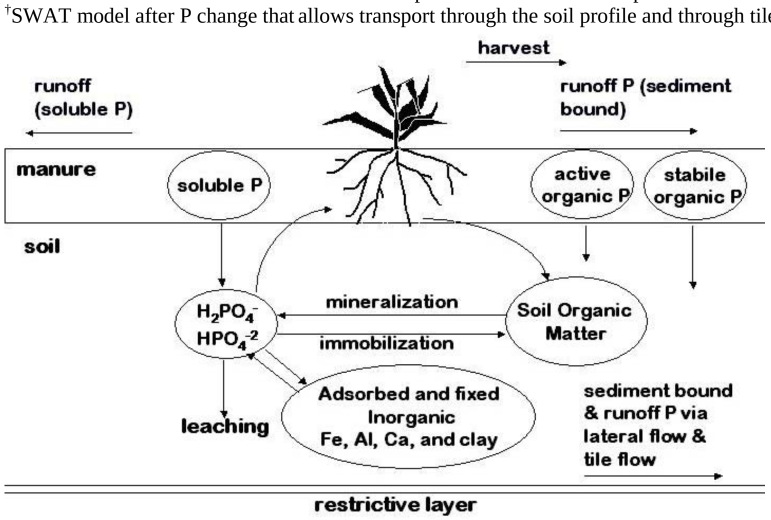

Figure 3. The adapted SWAT2005 phosphorus transport model under development.

The annual NSE values for the calibration and validation periods were 0.7 and 0.6, respectively with observed and simulated means with less than a 20% difference (Green et al., 2006). Tile samples were taken approximately bimonthly during the periods of July, 2003 to October, 2003; March, 2004 to November, 2004; and March, 2005 to September, 2005 while in-stream grab samples have been collected since 2001 during the same months that the tile samples were taken.

An average of 100 kg manure ha^{-1} applied resulted in average annual lateral soluble P and tile soluble P concentrations for 1995-1998 and 1999-2004 of 0.072 mg P L^{-1}/0.075 mg P L^{-1} and 0.075 mg P L^{-1}/0.047 mg P L^{-1}, respectively. The sum of lateral, percolate, and tile soluble P loads annually averaged estimated simulated losses from the fertilizer application sites for 1995-1998 and 1999-2004 were 0.53 kg P ha^{-1} and 0.614 kg P ha^{-1}. The average concentration for the measured in-stream grab samples for the time periods mentioned above was 0.054 mg P L^{-1} and the tile soluble P average was 0.073 mg L^{-1}. Representative soluble P averages from the measured in-stream and tile flow data were used to determine the PBIAS values with the calibration and validation periods. The lateral flow simulated soluble P values were assumed to be similar to the in-stream measured value and resulted in PBIAS values as follows: calibration lateral/tile is -33.3/-2.7 and validation lateral/tile is -38.9/35.6. With only a difference of approximately 0.02 mg P L^{-1}, the PBIAS values per Table 1 indicate that the calibration of the tile soluble P is “very good” while the remaining validation of the lateral and tile flow soluble P and the calibrated lateral P are “good.”

Conclusions

SWAT model simulations and continued updates allow water resource managers access to a tool that enables them to plan and make decisions in evaluating water supplies and non-point source pollution impacts in large river basins. The new P algorithms for the SWAT model version 2005 has made alterations that will only improve P tracking by including its transport pathways and have shown promise with the limited data available in the South Fork watershed in Iowa. Additional changes will be made as scientific knowledge progresses and additional data become available.

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Simulating *Nothfagus* forests in the Chilean Patagonia: a test and analysis of tree growth and nutrient cycling in SWAT

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Abstract

The SWAT model was applied to the Aysén River Basin of Southern Chile as part of ECOManage, an international project aimed at make available modeling and decision-making tools for managers of coastal zones. This heterogeneous basin of 11,456 km^2 presents a modeling challenge due to the mountainous terrain, strong precipitation gradient and the importance of non-agricultural land cover types. The low nitrogen deposition and cold climate of Chilean Patagonia place restrictions on the nitrogen cycle in the forests of the region. After presenting the results of the hydrodynamic calibration of SWAT for the Aysén Basin, we analyze the importance of vegetation growth and litter production in the nitrogen cycle of evergreen and deciduous *Nothofagus* (Southern Beech) forests. We compare SWAT2000 and SWAT2005 in terms of tree growth and the nitrogen cycle. Finally, we present our conclusions as to the most appropriate way to simulate *Nothofagus* forests while trying to balance model complexity and data requirements with the realism of the simulations.

KEYWORDS: SWAT, nitrogen cycle, forest growth, Patagonia

Introduction

The application of complex watershed simulation models to basins that are quite distinct from those used for model development can lead to unexpected difficulties. One potential pitfall is the issue of equifinality where numerous parameter sets can lead to statistically acceptable results. However, unless one delves into the details, examining the behavior of different modeled processes in different land use/soil combinations, using the model as a predictive tool is not recommended. In this paper, we examine in detail nutrient cycling and plant growth in a remote basin in the Chilean Patagonia. This basin presents a case study that is highly divergent from the low-topography agricultural basins originally used as references for the creation of SWAT and related models. Additionally, ecological research in the past two decades has uncovered several singularities in the nutrient cycling of Patagonian forests. This paper addresses the challenges and solutions involved in applying the SWAT model in the Aysén Basin.

The SWAT model was used in European project EcoManage as a tool to increase the capacity of assisting managers of Coastal Zones to join knowledge horizontally from ecological and socio-economic disciplines. The Aysén Fjord is an interesting study site because it shows conflicting interests between urban, industrial and agricultural pressures and environmental maintenance. The three key aspects of EcoManage are (1) the consideration that a coastal zone depends on local pressures, but also on pressures originated in the drainage basin, transported mostly by rivers and by groundwater, (2) that socio-economic activities are the driving forces of those pressures and that their impacts on the Coastal ecosystem have feedback on socio-economics and (3) the impacts depend on physical characteristics of the Coastal ecosystem that together with the loads determine its ecological state.

Review: Nitrogen Cycling in Southern Chile

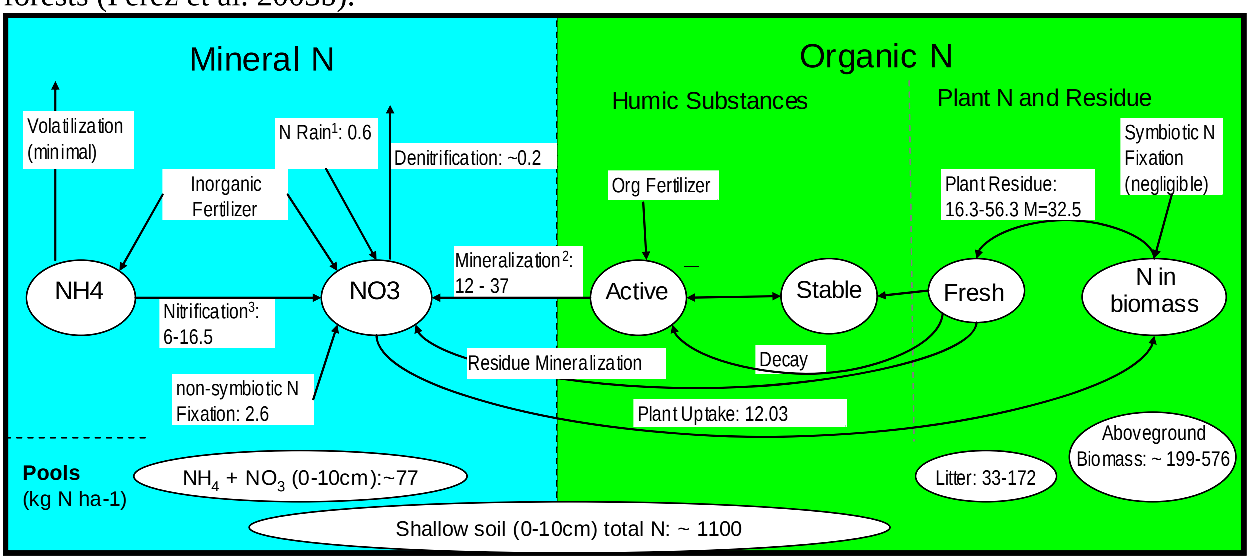
Nitrogen and phosphorus are thought to be the key elements in limiting primary productivity in temperate forests (Attiwell & Adams 1993). Thus, it is important to gain an understanding of these cycles in any particular forest type or geographic location. The Aysén Basin is located at about 45° S in the XI region of Chile. However, the bulk of what is known about nutrient dynamics in Patagonian forests comes from other geographical areas such as Chiloé Island, (approximately 300km northeast) of the Aysén basin and Puyehue National Park (500km to the north).

Perhaps the most outstanding feature of the Southern Patagonian forests is their isolation. In a biogeographical sense, this isolation has led to high rates of endemism as well as restricted biodiversity. Currently, a major effect of isolation is that rates of wet and dry nutrient deposition remain similar to historic rates, while in the northern hemisphere these rates have been severely altered (Godoy et al. 2003; Oyarzún et al. 2004). As a first approximation, forests in the Chilean Patagonia would be expected to have high internal nutrient cycling resulting in low nutrient loading to rivers. However, the creation of pastures by burning and the subsequent introduction of cattle and sheep into many areas (especially valleys) of Patagonia have altered watershed dynamics in ways that are not yet fully understood.

Several processes and pools of the nitrogen cycle of the Patagonian forests have been studied a good deal, while for others, published information is scarce (see Figure 1). Several authors have commented on the limited inorganic N wet deposition to forested ecosystems in Southern Chile, where concentrations range from 0.015 to 0.071 mg/L (Hedin et al. 1995; Perez et al. 1998; Godoy et al. 2003). However, organic N in rainwater and especially cloudwater from biogenic and anthropogenic sources can contribute significantly to the total N deposition (Godoy et al. 2003). This appears especially true in coastal areas, where cloudwater can deliver up to 9 kg ha-1 yr-1 of organic N (Godoy et al. 2003). Net N mineralization has also been studied in Patagonia and reported values of between 12 to 37 kg N ha-1 yr-1 have been reported, significantly less than forested systems with anthropogenic N inputs (Pérez et al. 1998; Pérez et al. 2003a). Nitrification was demonstrated to be between 50% and 62% of total N mineralization for forests in Chiloe (Pérez et al. 1998). SWAT directly calculates mineralization of organic N fractions to nitrate, therefore including a nitrification step. Ammonium in the terrestrial SWAT2000 N cycle only derives from fertilizer (including animal grazing), and thus NH_4 loading to rivers and volatilization will be zero unless fertilizers are used or water quality processes are activated. This contrasts with data from Patagonia which indicate that the average ratio of nitrate: ammonium is low in (a) rainwater (0.2 in Puyehue Park), (b) in soil water (ca 0.44 for shallow water and 0.03 for deep water) and (c) runoff (consistently below 1 and often below 0.1) (Perakis & Hedin 2002; Oyarzún et al. 2004, Perakis & Hedin 2005). Because these basins are largely forested, they have little to no fertilizer input and thus in SWAT there is no NH_4 in the terrestrial N cycle. In SWAT 2005, a new natural source of N was included: NH_4 in rain.

Denitrification can be generally characterized as low in well-drained forested soils and higher in wetter floodplain areas. Pérez and colleagues (2003a) show that denitrification in temperate forests of Chiloe Island are low: roughly 0.2 kg ha-1 yr-1. Although, measured data are not available for the Aysén Basin, it is hypothesized that denitrification maybe higher in the western fringe where precipitation can surpass 4m yr-1, or in specific patches of wetland soils in the flatter and drier eastern fringe. While the *Nothofagus* genus, the dominant trees in the Aysén, does not have symbiotic N-fixation, non-symbiotic N-fixation takes place in the woody debris and litter on the forest floor. Pérez and colleagues (2003a) report that non-symbiotic fixation in Chiloe ranges between 1.5-3.6 kg N ha-1 yr-1. Volatilization of nitrogen does not appear to be important in the nutrient cycle of Patagonian forests, according to an N tracer study by Perakis and Hedin (2001).

To synthesize the above: tight N cycles have been documented in several Patagonian temperate forests and would be expected in areas not yet intensively studied. Wet deposition of N is generally very low, especially for inorganic fractions (Godoy et al. 2003). Mineralization and nitrification were consistently lower than most forest from the Northern Hemisphere (Pérez et al. 1998). Inorganic N appears to be rapidly taken up by microbes and vegetation, and soil and stream measures of nitrate are often very low (Perakis & Hedin 2001; Perakis & Hedin 2005). Dissolved organic nitrogen makes up the bulk of exported N in forested watersheds and appears to be controlled by hydrological factors instead of by microbial processes or plant uptake (Perakis & Hedin 2001). Because of scare available N, turnover rates of canopy foliage tend to be low: on the order of 5 years for broad leaf evergreen forests and 15 years for conifer-dominated forests (Pérez et al. 2003b).



[1]This is a value for NO3, total N contribution via precipitation can reach 11.8 kg ha-1 yr-1
[2]net mineralization in 0-10cm of soil, (includes organic to NH4)
[3]nitrification estimated as 50% of net mineralization (Perez et al. 1998)

Figure 1: Diagram of N cycle in SWAT2005 with pools and process rates for Patagonian forests taken from the literature. (Diagram adapted from SWAT2005 Theory)

Potential Problems Modeling Patagonian Forests with SWAT

The SWAT model is now widely applied around the world; however, it has been most frequently tested using data from low topography agricultural watershed in the American Midwest. Users applying the model to unique basins should proceed with caution in order to make sure modeled processes in SWAT fit with what is known about a particular basin. Because the Aysén basin is heterogeneous basin with a predominance of non-agricultural land cover types we examine potential problems that may arise in modeling plant growth and nutrients with SWAT.

Tree growth and organic residue

SWAT inherited plant growth algorithms from the EPIC model, which were created for use in highly agricultural settings (Gassmann et al. 2005). Fundamental differences in agricultural and forest systems and the fact that SWAT does not explicitly model the carbon cycle have lead several researchers to incorporate more realistic forest algorithms into SWAT (Watson et al. 2005; McDonald et al. 2005). This work is certainly valuable; however, due to lack of data in many parts of the world, adding more complexity to SWAT may not necessarily improve results. We attempt to improve nutrient cycling in SWAT by fine-tuning existing algorithms and without incorporating a full accounting of the carbon cycle.

Because exogenous nutrient inputs into forests systems in southern Chile are minimal, litterfall is an important source of bioavailable P and N. It is thus important to examine the residue production in each landuse type and to verify the fate of this residue.

Too much N in wet deposition

The default value of the RCN parameter (nitrate in precipitation) in SWAT is 1 mg/L. However, Oyarzún and colleagues (2004) indicate that in the Andes range of the X region of Chile the concentration of total N in precipitation is 0.174 mg/L. Furthermore, unlike the northern hemisphere where the majority of wet deposition N is inorganic, in southern Chile most deposition occurs as dissolved organic nitrogen; Oyarzún et al. (2004) found 66% of N in precipitation was organic N. Other authors have found similar patters (Godoy et al. 2003). Thus, it would be more realistic and likely to provide better results if the N in precipitation could be divided into its major fractions and introduced directly into the appropriate pools.

The objective of this work was to address the following problems and concerns in the context of the Aysén basin. A final objective is to produce a SWAT output file that expedites the visualization and analysis of nutrient cycles for the entire basin and for different types of HRU.

Calibration and Validation of SWAT for the Aysén Basin

Data and Model Setup

The SWAT model was set up in AVSWAT-X and pre-existing data for the region was used. A descriptive study of the regions soils by IREN-CORFO (1979) was augmented by data provided to ECOManage by SAG (Agriculture and Livestock Service) in 2006. Pedotransfer functions which allow the estimation of hydraulic soil properties like porosity, field capacity, wilting point and conductivity were used (Saxton et al., 1986). The official land register of native forests (CONAF-CONAMA 1997) was used to characterize the current landuse. Because the database includes a large number of land cover classes, the information had to be converted (and simplified) into a format usable by SWAT. Additionally, because of the dominance and diversity of forest cover in the basin, new SWAT classes were created based on an extensive literature review and expert opinion (Table 1). Meteorologic and streamflow data were obtained from the General Water Authority (DGA). Due to the extreme spatial heterogeneity of precipitation in the Aysén Basin, the simple algorithm used by AVSWAT-X to assign precipitation stations to subbasins was not adequate. First, 'synthetic' were added: multiplying an observed precipitation station by a constant factor in order to bring the annual average precipitation in line with the isohyets generated by the DGA. Second, precipitation stations were manually assigned to subbasins in order to approximate the isohyets.

Table 1: New SWAT Land Cover Classes Used in Aysén SWAT Model

SWAT Code	Vegetation Type	Principal Species	Range of litterfall (Mg ha-1 yr-1)	Simulated residue, range and mean (Mg ha-1 yr-1)
BCAY	Deciduous forest of Aysén	*Nothfagus pumilio*	2.0 - 3.6	1.9 - 4.2; 2.93
MCAM	Montane deciduous forest	*N. Antarctica, N. pumilio, Berberis spp.*	1.4 - 2.5	1.2 – 2.8; 1.94
BSNB	Montane evergreen forest	*Nothfagus Betuloides, Laurelia philippiana*	2.8 - 3.8	2.3 – 3.82; 3.19

Sources: Caldentey et al. 2001; Austin & Osvaldo 2002; Vann et al. 2002; Pérez et al. 2003b

Model Calibration and Validation

A sensitivity analysis was run for the model setup and autocalibration was carried out using the ten most sensitive parameters. Only the basin output point (Aysén River monitoring station) was used for autocalibration; however, limited manual calibration was carried out in specific subbasins. SWAT hydrodynamic calibration results can be seen in Table 2.

Table 2. Comparison of observed and modeled average monthly flows and daily and monthly R2 and model efficiency statistics

	Station	Time period	Observed / modeled flow (m^3/s)	R2 Day/Month	ME Day/Month
Calibrati	Aysén	Jan '96– May '98	556/582	0.54/0.86	0.5/0.73
	Mañihu ales	Jan '96– May '98	187/173	0.53/0.75	0.51/0.73

	Simpson	Jan '96–May '98	47/62	0.67/0.83	0.55/0.59
Validation	Aysén	Sep '02-Sep '05	599/507	0.48/0.67	0.33/0.41
	Mañihuales	Jul '02-Jul '05	201/167	0.52/0.79	0.5/0.7
	Simpson	Feb '02-May'05	60/43	0.56/0.61	0.42/0.41

Modeling the Nutrient Cycles of *Nothofagus*-dominated watersheds

Tree growth and residue production

SWAT outputs the biomass generated in each HRU, and though for agricultural landuses this figure may be directly compared to crop growth and harvest, the situation for forests is more complex. In SWAT2000, interannual tree growth did not occur: all forest HRUs were considered to be mature. In SWAT2005, tree growth from sampling to mature tree is able to occur (Figure 2). However, under default tree parameter sets (e.g. PINE) a large fraction of annual biomass production is removed as yield or converted to residue, resulting in minimal growth of persistent biomass such as trunks and large roots. If the primary objective for calibrating biomass is to achieve realistic nutrient cycling, then the total biomass of a forest system can be ignored and the focus can be placed on residue production. This is our strategy. In Table 1, the ranges of litterfall for the three principal forest types in the Aysén basin were gathered from the literature and compared to SWAT output after calibration. The most sensitive parameter modified was BIO_LEAF which controls the fraction of biomass converted to residue at the end of each season. Other parameters changed—according to values found in the literature—were the nitrogen and phosphorus uptake parameters, LAI, canopy height, rooting depth, and the harvest index.

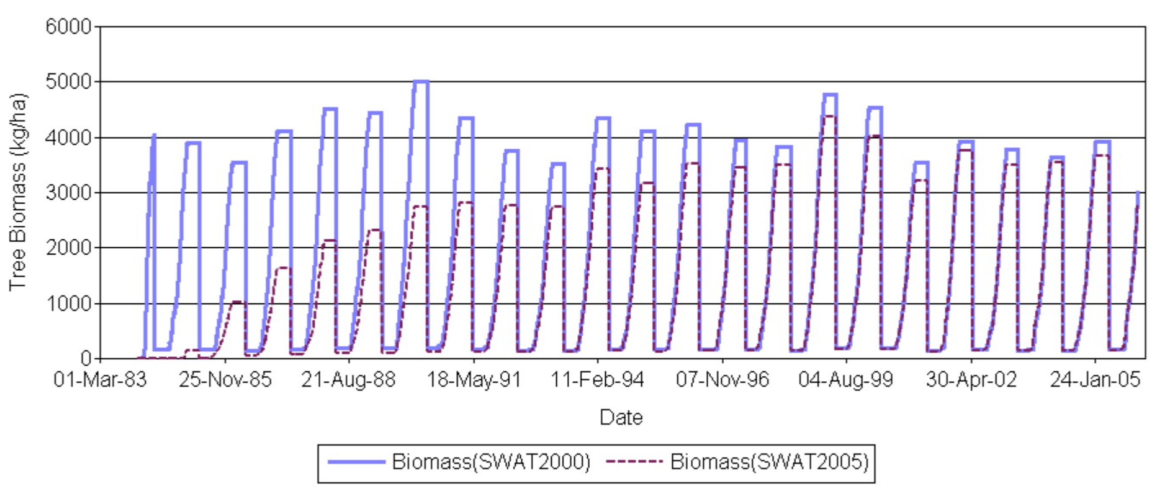


Figure 2. Difference in biomass production between SWAT2000 and SWAT2005 for Nothofagus pumilio.

Modification of wet deposition

An option in SWAT2005 is to split wet deposition of N into NH_4 and NO_3 components. However, the literature from Patagonia indicates that dissolved organic nitrogen (DON) can make up one third of wet deposition (Oyarzún et al. 2004). Thus, source code was changed to allow DON to be added as a component of wet N deposition.

After a spin-up period of 6 years, we compared the average daily NO_3 concentration at the subbasin that corresponds to the Aysén River monitoring station with 28 measurements of NO_3-N taken between 1997 and 2004 by the local authorities. The average measured value was 0.048 mg/l while SWAT NO_3 output for the corresponding reach was 0.053 mg/l. The difference between modeled and measured values was not statistically significant (t= 0.91, p=0.37, gl=27). The same simulation set up, run with RCN = 1 gives a NO_3 concentration in the reach of 0.23 mg/L, which is higher than even the highest measured value. Table 3 indicates that reducing NO_3 in rain (to levels measured in Patagonia), adding DON as input, and adjusting related parameters can produce good results.

Table 3. Estimated annual nitrogen and phosphorus loads from diffuse sources.

Simulation Run	N (tons/year)	P (tons/year)	% Org N
RCN = 1	7674	436	29%
RCN=1, N parameter set	4592	231	24%
NH4=0.049, NO3=0.01, DON=0.115, N parameter set	2776	288	56%

(N parameter set: RSDCO = 0.005, SDNCO = 0.95, NPERCO = 0.005)

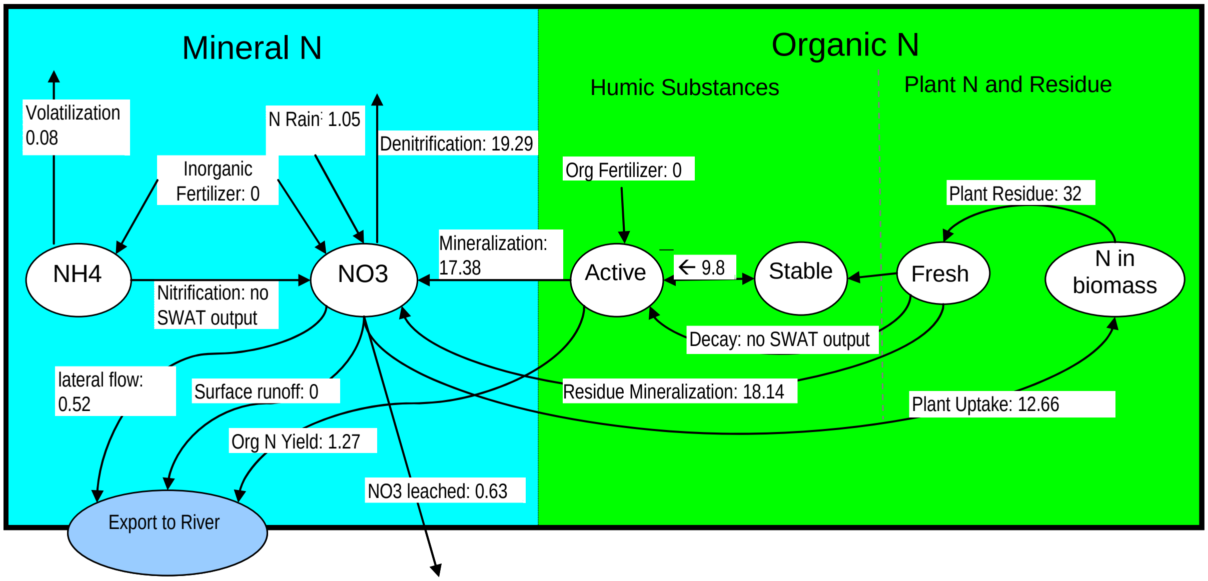


Figure 3. Output of SWAT2005 after calibration and changes for BCAY forest type (kg N ha-1 yr-1)

A more detailed look at the model results after changes in the N cycle parameterization and wet deposition can be seen in Figure 3. In general, the values of the annual processes in the N cycle are within the range of literature values given in Figure 1. One process where the model and field data do not match is denitrification. This might partially explained by the fact that SDNCO (denitrification threshold water content), has been adjusted down slightly and that CDN (denitrification exponential rate coefficient) was left at its default value. It is likely that further adjustment to existing parameters can help decrease the model value; nevertheless, it is also important to consider if the little published denitrification data (which is hard to measure reliably in the field and often is variable in time) are too low. In Aysén, the high rainfall and high organic soil matter should allow for more denitrification. Another point is that the active organic to stable organic N value is negative, and more generally, net mineralization occurs consistently during the simulation period. This might act to drive more denitrification than would otherwise occur. A qualitative indicator that the current SWAT setup is appropriate for Patagonian forests is that the ratio of external:internal N cycle fluxes is 0.4. This fits with statements by Pérez and colleagues (2003a) that the N cycle in Patagonian forests tends to be tight with much internal cycling.

Conclusion

In this paper we have taken steps to improve the ability to model watersheds dominated by relatively unpolluted temperate forests with SWAT2005. Our strategy has been to make small modification instead of adding more complex routines requiring additional parameterization or input data. We have seen results improve: the ratio of organic N to inorganic N in river water has decreased as we have calibrated and then modified the model. Apart from a few processes, the annual fluxes in the SWAT N cycle for the BCAY cover class corresponded to what were gleaned from the literature. As more data becomes available on the soils, forest dynamics, and nutrient cycles for the

Patagonian region, it will be desirable to increase the complexity of the model for those routines where results are sub par. This work has already begun (e.g. Watson et al. 2005; McDonald et al. 2005). However, with the limited data currently available, we conclude that SWAT2005 is capable of simulating the N cycle in a unique forested system.

We used a SWAT2005 version in which the source code was partially modified—the inputs and outputs of the model—using MOHID's code and programming philosophy (Chambel-Leitão et al. 2007). This has improved analysis and visualization of admittedly complicated N cycles in large basins. Furthermore, a macro created in Excel allows diagrams such as the ones in this paper to be rapidly produced. Further work is continuing on adjusting the N cycle and P cycles in SWAT for Patagonian forests. One expected outcome is to be able to identify the most pressing gaps in field data. A final outcome for the ECOManage Project will be to produce a working set of tools and models for managers and policy makers for the Aysén Basin.

In concluding, we wish to mention the utility of using SWAT with the three wet deposition compartments (NH_4, NO_3, and DON) as a way to study the potential effects of increasing anthropogenic N emissions worldwide and the interactions between climate change and biogeochemical cycles.

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Evaluating the role of shrub, grass and forb growth after harvest in forested catchment water balance using SWAT coupled with the ALMANAC model

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ABSTRACT

To accurately simulate watershed hydrology after forest harvest using SWAT, it is important to understand the factors that potentially make certain sites more sensitive to disturbance. The growth model in SWAT has been modified to provide a more precise description of forest growth dynamics, by integrating a multi-species growth model, $ALMANAC_{BF}$. We collected field data to develop parameters for the multi-species growth model. The biomass, leaf area index and light interception was measured on hummock tops and in depressions of three sites that differed in pre-harvest forest stand composition. Among the sites, the overall light extinction coefficient (k), of sites was observed to be 0.49 ± 0.22 and the estimated radiation use efficiency (RUE) was 4.9 ± 1.7 for annual species and 3.3 ± 2 for shrubs. We observed significant differences in percent cover of vegetation, biomass and leaf area index among sites representing differing pre-harvest forest stands and associated with topography, between hummocks and depressions. LIDAR imagery was used to provide estimates of leaf area and biomass that accounted for variations in vegetation associated with landscape characteristics. Our results demonstrate that on the hummocky terrain of the Boreal Plain, there are important variations in vegetation recovery after disturbance among different ecosites and among landscape positions. Research to identify trends in vegetation reestablishment after harvest requires sampling designs that capture differences in vegetation associated with these factors. LIDAR imagery and targeted sampling of obvious landscape variations was effective in correcting vegetation cover estimates to account for differences in cover observed between depressions and hummocks in a variety of sites.

INTRODUCTION

One of the objectives of the **For**est **W**atershed **a**nd **R**iparian **D**isturbance (FORWARD) project is to simulate the effects of fire and harvest disturbance upon streamflow and water quality in forested watersheds. The project is being conducted in the Boreal Plains forest of Northern Alberta, Canada (Figure 1). The Soil and Water Assessment Tool (SWAT) is an effective tool for land managers to model streamflow and water quality effects where agriculture is the dominant landuse. It is increasingly of interest for land managers in forested areas (King and Bologh 2001; Miller et al. 2002; Putz et al. 2002). However, it is also recognized that the vegetation model requires significant modification to effectively simulate forested environments (Wattenbach et al. 2005; Watson et al. 2005) and forest management (Lorz et al. in press; MacDonald et al. in prep.; Kiniry et al. submitted).

Forest management practices cause important quantitative changes to evapotranspiration, runoff and watershed discharge. The changes to watershed hydrology take many years to return to a steady state mature forest condition (Hornbeck et al. 1997; Buttle and Metcalfe 2000; Devito et al. 2005; Burke et al. 2003; Prepas et al. 2006). The impact of forest management practices will be buffered or augmented depending upon landscape characteristics. For a hydrological model to provide useful information about the impact of localized disturbance at the watershed scale, whether natural or anthropogenic, it must effectively simulate the variation associated with physical landscape characteristics such as slope and soil type, but also the variability associated with different vegetation cover.

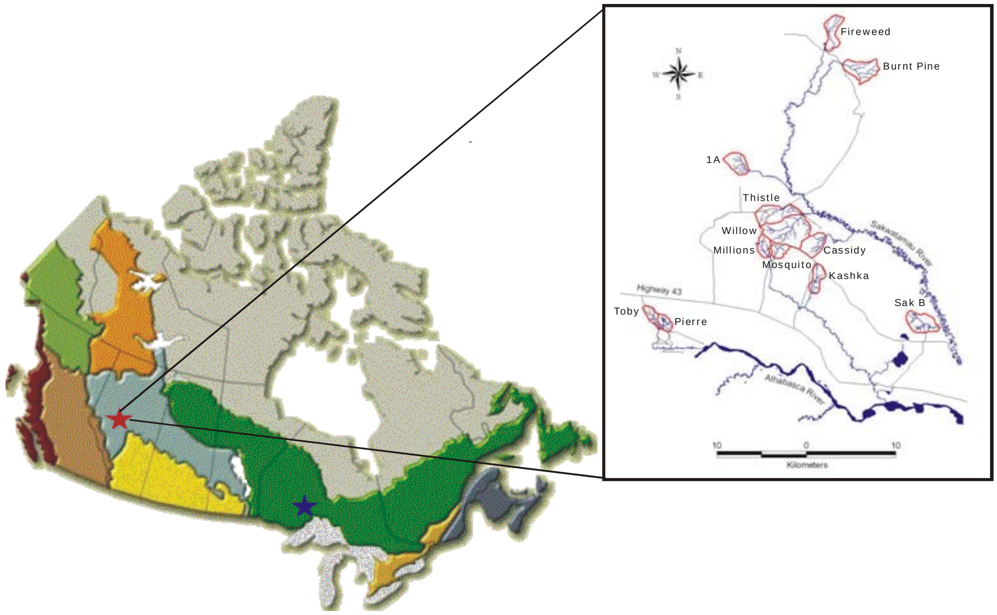

Figure 44 Location of FORWARD project research watersheds on the Boreal Plain.

The western Boreal Plain of North America is characterized by flat to gently sloping relief with large catchment areas. First-order watersheds range around 5 km^2. Water flow is dominated by spring run-off and rainfall occurs largely in June and early July. Annual precipitation ranges from 325 to 625 mm (Smith et al. 2002). Forest growth is limited on dry uplands whereas forest stands occurring in low lying areas are adequately sustained by shallow aquifers occurring close to the surface. Forest stands of the Boreal Plain are complex ecosystems that contain a variety of pure or mixed stands of lodgepole pine (*pinus contorta*) white and black spruce (*picea glauca and mariana*), balsam fir (*Abies balsamea*) and rapid growing deciduous forests dominated by *Populus* species.

After disturbance, pioneer vegetation such as grasses, forbs and woody shrubs compete strongly with crop trees. Research on the growth of vegetation on sites that have been disturbed, either naturally through fire and disease or through forest harvesting activities, has mainly focused on silviculture and biodiversity questions. Impacts of forest disturbance on species diversity are complex. However, research that has focused on silvicultural questions has noted the importance of the amount of competition by grasses, forbs and shrubs on the regrowth of the forest (Hogg and Lieffers 1991; Bell et al. 2000; Shropshire et al. 2001; Wagner, 2000). Furthermore, other research has demonstrated that the amount of vegetation growing on a site after disturbance can play an important role in the site water balance (Landhausser et al. 2003; Pothier et al. 2003; Jutras and Plamondon 2005).

Research carried out with the objective of improving silvicultural techniques provides us with our best indication of vegetation variability. Godin et al. (in prep) notes that there are definite gradients in pioneer vegetative cover of grasses, forbs and shrubs that establish on harvested sites on the Boreal Plain and the cover varies according to a site's position on the edatopic grid (i.e. the moisture regime and the nutrient richness of a site). Silvicultural treatments are varied as a consequence of the successional changes in competitive vegetation associated with the moisture regime and richness of a site. This indicates that pioneer vegetative cover does vary significantly among different sites, and thus differences could occur in site hydrology.

Therefore a significant challenge in modelling the impacts of forest harvesting and management on watershed hydrology is to simulate the variability in vegetative cover after disturbance, as well as the successional process of forest recovery after disturbance. Modelling techniques to simulate post-disturbance vegetation recovery have often simply relied on multi-year leaf area development curves (McMicheal et al. 2004). However, recovering Boreal forest sites not only undergo an increase in leaf area over time, but also a conversion from grasses, forbs, shrubs and deciduous species to coniferous species. Hence, a multi-species approach with several inter-annual cycles is required (MacDonald et al. in prep).

MacDonald et al. (in prep) demonstrate that Boreal forest recovery can be modelled using growth characteristics of generic annual and short shrub species that grow in conjunction with crop tree species. The objective of the work described in this paper was to evaluate the variability in observed vegetative cover among Boreal Plain forest sites three years after harvest, and to develop parameters that can be used in the simulation of vegetation growth in the initial stages of forest recovery on sites with similar characteristics.

Material and Methods

Site Description

The sites discussed are located on the Boreal Plain of Northern Alberta, Canada in the forest management area (FMA) of Millar Western Forest Products Ltd. (MW) in the lower foothills (LF) ecozone. Further, all sites are located within two watersheds regularly monitored as a part of the FORWARD project (Figure 1) The sites were clearcut harvested in the winter of 2003/2004 and received an application of herbicide in the spring of 2004. Each site had previously been characterized as a part of a riparian vegetation research project (Luke 2007).

Luke (2007) established transects between riparian areas and upland forests at 22 sites for vegetation characterization before and after harvest. Three sites among those characterized by Luke were chosen for this study. The sites were selected based on a visual evaluation of the different possible sites, and the requirement that they demonstrate very definitive differences in vegetative cover before harvest. Each of the three sites has undergone identical silviculture treatments since harvest.

Site 1 was located in FORWARD watershed Pierre (Figure 1), a small first-order watershed of 2.79 km^2 occurring on hummocky moraine material. According to a pre-harvest aerial photo interpretation survey of Alberta forests, the Alberta Vegetation Inventory (Alberta Environmental Protection 1996), Site 1 is located where a former stand with an overstorey dominated by 100% lodgepole pine (*pinus contorta,*), (Pl), and a 100% white birch (*betula papyrifera*), (Bw), understorey existed*.* The complete ecosite classification is lower foothills (LF), bracted honeysuckle (f), phase 1, Pl. The Luke (2007) survey classifies the site as a Pl/bracted honeysuckle/fern community type (Table 1). While the overstorey is effectively dominated by Pl, the localized sites examined by Luke indicated the tree cover was much more heavily dominated by a black spruce (Sb) understorey as opposed to Bw as is suggested by the aerial photo survey.

Site 2 was also located in Pierre on hummocky moraine material. According to the pre-harvest Alberta Vegetation Inventory (Alberta Environmental Protection 1996), Site 2 was formerly covered by an overstorey dominated by 80% Pl with some trembling aspen (*populus tremuloides),* (Aw), 20%. The understorey was mixed with 50% paper birch (*betula papyrifera*), (Bw), 40% white spruce (*picea glauca)*, (Sw), and 10% black spruce (*picea mariana)*, (Sb)*.* These aerial observations correspond to ground observations by Luke (2007). However, the ecosite classification based on aerial photo interpretation is inaccurate according to Luke's (2007) survey. Site 2 appears to be fall in a zone that has understorey vegetation associated with poorer site conditions as indicated by a strong presence of Labrador tea, bog cranberry and blueberry. Understorey vegetation corresponds more closely with a d1, mesic, Labrador tea site with a type 2, Pl-Sb/green alder plant community. Whereas the aerial photo survey suggests a mesic medium, low-bush cranberry (e), phase 3, Aw-Sw-Pl ecosite.

Site 3 was located in Millions (Figure 1), another of the FORWARD project's small first order watersheds of 3.35 km^2 occurring on hummocky moraine material that has been dissected by glaciofluvial channeling and deposits. The site is on a south west facing slope. Site 3 was formerly covered by an overstorey consisting of 100% Aw. No understorey was observed. These observations correspond to results observed by Luke (2007). The complete ecosite classification is lower foothills (LF), low-bush cranberry (e), phase 2, Aw. Luke's (2007) survey suggests that the site is consistent with the Aw green alder plant community type (Table 1).

Table 1. Summary of characteristics of chosen study sites, preharvest, forest stand types and ecosite classifications, from Alberta Vegetation Index and compared to Luke (2007) vegetation inventory.

	1996 Alberta Vegetation Inventory			Luke (2007)			
Site and Watershed	**Original Stand Classification**	**Ecosite**	**Ecosite Phase**	**[1]Original Stand Classification**	**Ecosite**	**Ecosite Phase**	**Community Type**
Pierre, Site 1	$Pl_{(o)}$-100 $Bw_{(u)}$-100	e. Low bush cranberry mesic-medium	3. Pl-Aw-Sw	Pl – 69, Sb – 27, Bw -2	f1. bracted honeysuckle mesic-medium	f1. Pl	f1.1 Pl – bracted honeysuckle/fern
Pierre, Site 2	$Pl_{(o)}$-80%/$Aw_{(o)}$-20% $Bw_{(u)}$-50%/$Sw_{(u)}$40% / $Sb_{(u)}$10%	e. Low bush cranberry mesic-medium	3. Pl-Aw-Sw	Pl – 65, Aw – 22, Sb – 13	d1 labrador tea mesic	d1 Pl-Sb	d1.2 Pl –Sb- green Alder-feather moss
Millions, Site 3	$Aw_{(o)}$-100%	e. Low bush cranberry, mesic-medium	2. Aw	Aw – 100	e. Low bush cranberry mesic-medium	e2. Aw	e2.3 Aw/green alder

Aw = Trembling Aspen Bw = White birch Pl = lodgepole pine Sb = black spruce Sw = white spruce

Subscript (o) represents overstorey and subscript (u) represents understorey
1 Numbers represented the basal area of trees in plot, unlike aerial photo interpretation where numbers represent the percent cover. No separation of understorey and overstorey was reported.

Data Collection and Analysis

The dominant landform in this region of the Boreal Plain is largely the hummocky moraine. During initial visits to the sites we noted visually that there were very obvious differences in the growth of vegetation on most of the harvested sites between depressions (moist areas) and the ridges or hilltops (dry areas). We chose to carry out a targeted sampling program aimed at providing a good estimate of the amount of vegetation growing in depressions vs. hilltops (see Figure 2).

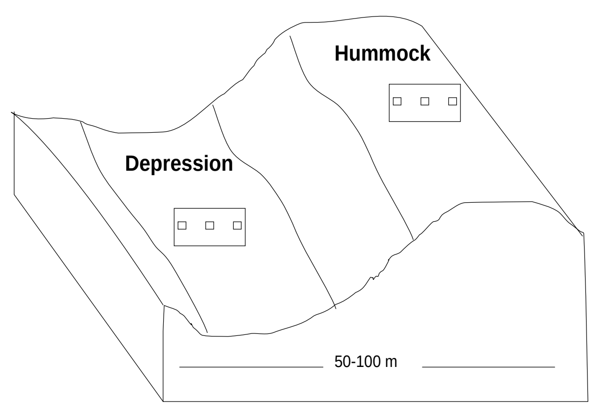

Figure 45 General layout of sample plots at a site.

In total we established 18 plots (9 paired plots in total, 3 depression and 3 hilltop plots per site). The coordinates of each plot were recorded with GPS. Each plot consisted of three 1m x 1m subplots.

Field Measurements

In each 1 m^2 subplot the leaf area index, total dry weight biomass, percent cover and PAR was measured. A visual estimate of percent cover for individual species was made to the nearest percentage up to ten percent and, thereafter, five percent intervals were used. Light measurements were taken using a 0.8 m long Sunfleck Ceptometer (Decagon, Pullman, WA 99163). One light measurement was taken every 10 cm (total of 10 measurements) at ground level, and above the plant canopy according to Kiniry et al. (1999). Ten measurements within the 60 m^2 plot were also taken to confirm that 1 m^2 subplots were representative of leaf area in the larger area. The median difference observed between light interception averages on large plots relative to subplot averages was 17% of the mean of subplots.

Leaf area was measured by calculating the ratio of leaf area to moist weight for each of the species observed in the plots. Outside of the plots, an example of each of the plants observed in the plot was taken for analysis. The plants removed were weighed. Then all leaves were removed, pressed between two pieces of plexi-glass, placed on white poster board and digital photos taken. The digital photos were analyzed according to methodology outlined by O'Neal et al. (2002). Briefly, the digital images of sub samples of plants of known weight were converted to black and white images and pixilated using public domain image software (Scion Image). The Scion Image software, calibrated by relating a pixel count to a known area, traces the outline of leaves and automatically calculates surface areas. Destructive sampling in each subplot was carried out and the moist weight of each species present was recorded immediately after sampling. Samples were retained, stored in paper bags and later dried in convection ovens at 100º C to provide the total dry biomass.

Table 2. Measurements carried out at each site and information supplied for hydrological modelling.

Plot Type	Number	Summary Notes	Measurement	Information Supplied
Paired Plots (60 m^2)	9	Three paired plots per site (hummock and depression).	PAR[1].	Medium-scale variability comparison.
Subplots (1m^2)	54	Three subplots (1 m^2) per 60 m^2 paired plot.	PAR[1].	Model Parameter (light extinction coefficient)
			Total Biomass	Model Parameter (radiation use efficiency)
			Total Leaf Area	Potential evapotranspiration
			Percent cover	Links to forestry data collection procedures.

[1] Photosynthetically active radiation.

Parameter Calculations

Leaf area index was calculated by multiplying the measured leaf area per gram wet weight by the total weight of each individual species collected within each individual subplot. The light extinction coefficient (k) for Beer's law (Monsi and Saeki, 1953) was calculated from the fraction of PAR intercepted (FIPAR) and the LAI. Average values of k were calculated for full plots for the date of destructive sampling as:

$$k = [\ln(1 - FIPAR)]/LAI$$

Radiation use efficiency was estimated by fitting SWAT/ALMANAC modelling results to both LAI and total annual biomass data as described below.

Modelling

Model simulations were carried out using a simplified watershed representation with three subbasins and three HRUs to maximize processing speed. Input data consisted of meteorological data from a weather station installed as a part of the FORWARD project's permanent monitoring program. The weather station is in close proximity (10 km radius) to the watersheds in which vegetation data was collected. FORWARD weather station data was available for years 2002 to 2006. Additional meteorological data was available from Environment Canada's Whitecourt weather station (approx. 40 km away) for years from 1990 to 2002. Soils data was taken from the Alberta AGRASID soil database (AGRASID, 1999) and a typical glacial moraine orthic gray luvisol of the Hubalta series, common in this region, was used in all simulations. Simulations were carried out from 1990 to 2006 with a simulated forest harvest in the winter of 2003/2004. Multiple runs were carried out varying the radiation use efficiency of shrubs and forbs to minimize the difference between observed and measured leaf area index and biomass for the sample plots.

Statistics and LIDAR

Basic statistics such as means, medians and standard deviations of samples were calculated using SYSTAT© (Wilkinson, 2000). Vegetation parameters were analyzed for significant differences using a two way ANOVA test with site and landscape position as the independent parameters, and means were compared based on Bonferonni contrasts. Geo-referenced LIDAR imagery files were supplied by Millar Western Forest Products Ltd. The image files were converted to grids and sample sites were located on the grids using GPS coordinates. The forest polygon in which sites were located was isolated and the LIDAR based elevation grid was used to calculate the proportion of depressions and hummocks at the sites. Depression areas were identified as being those zones less than 50% of the total elevation difference in the forest polygon. In the case of Site 3 located on a slope, the polygon was broken into zones in order to calculate the total area. Areas identified as depressions were assigned average values of biomass, LAI and percent cover measured at depression sites. Similarly hummock areas were identified and assigned their respective average values. For each site an area weighted LAI, biomass and percent cover was calculated corrected for the proportions of hummock and depressions.

Results and Discussion

Vegetation Characteristics, Distribution Among Sites and Landscape Position

Over 54 species of plants were observed at the three sites (Figure 3). The dominant species with respect to cover were *Calamagrostis canadensis* (Marshreed grass), *Epilobium angustifolium* (Fireweed), *Galium trifolium* (Sweet Scented Bedstraw), *Petasites palmatus* (Palmate-leaved coltsfoot), *Lonicera involucrata* (Bracted Honeysuckle) and *Rubus spp* (Raspberry). However, across the three sites *Vaccinium spp*. (Blueberry), *Rubus spp* (Raspberry) *Populus tremuloides* (Trembling aspen), *Epilobium angustifolium* (Fireweed), *Salix spp.* (Willow), *Equiseteum spp* (horsetail) and *Rosa acicularis* (prickly rose) were observed to occur in a greater number of plots. Dominant species varied among sites as would be expected between ecosites with different

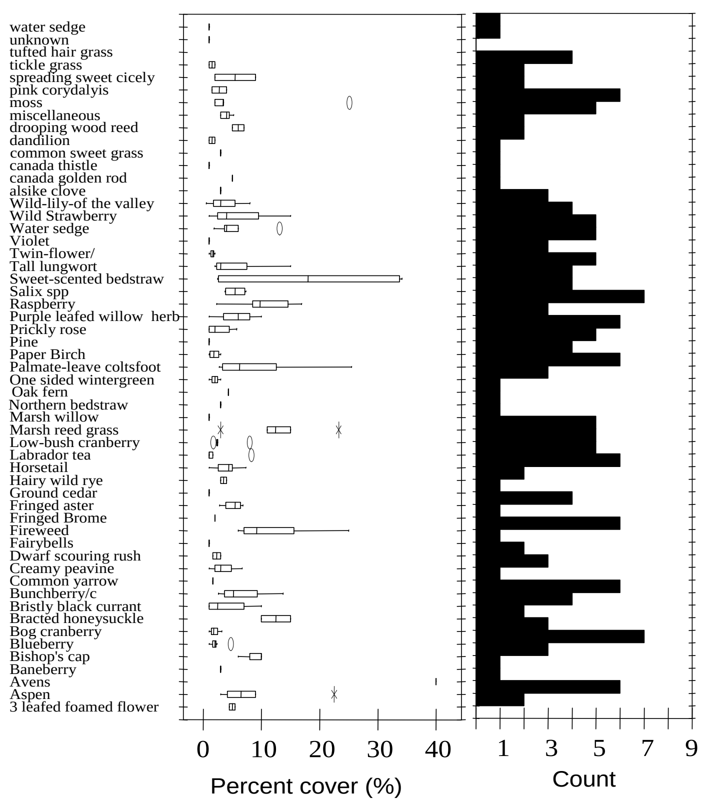

Figure 46 Distribution of species, boxplots of percent cover and frequency of occurrence in plots

plant communities. The proportional distributions between grasses, forbs, shrubs and trees did not demonstrate any obvious patterns. Grass was more prevalent in depressions and in several cases dominated certain plots. Shrubs and trees also tended to be larger in depressions as opposed to hummocks (Table 3). However, due to high variability among sites, the differences were not significant.

Total biomass, LAI and percent cover was significantly greater in Sites 1 and 3 than Site 2 and in depressions relative to the tops of hummocks (Figure 4). Pre-harvest vegetation surveys suggested that Site 2, was not as rich an ecosite and that less growth was expected at this site. Godin et al. (in prep) suggests that vegetation recovery after harvest varies strongly with ecosite phase and these results would appear to agree with that hypothesis. Differences between depressions and hummocks were very large in Sites 1 and 2, whereas there was little difference between depressions and hummocks in Site 3 (Table 3). Site 3 demonstrated very consistent biomass, cover and leaf area in both depressions and uplands. It was the only site where we observed higher biomass, and leaf area index in uplands in some cases. The lack of a significant difference between Sites 1 and 3 was due to the heavy vegetation cover, mainly grass species, in depressions at Site 1 (cover, biomass and LAI). This heavy grass cover had a greater LAI and biomass than depressions in Site 3. The hummocks of Site 1 were however less productive relative to Site 3. Consequently, the lack of a significant difference between these sites is an artefact of the sampling design. In reality, the total biomass and LAI will be largely dependant on the relative proportion of depressions and hummocks on the landscape in Site 1 whereas, in Site 3 due to the similarity between depressions and hummocks, an average LAI and biomass will be sufficiently accurate. This result stresses the importance of recognizing landscape position in sampling of vegetation. The greater vegetation cover in depressions between hummocks is probably due to differences in moisture availability. On hummocky and ridged landscapes with finer soils such as in this region of the Boreal Plain, infiltration will be slow and surface runoff and shallow lateral flow will move from hummocks and inevitably collect and infiltrate in lower areas resulting in greater availability of water for plant growth. The presence of depressions probably plays a critical role in the recolonization of vegetation in this ecosystem.

Table 3. Summary of vegetation characteristics among sites and landscape positions.

Site	Plot	Position	LAI $m^2\ m^{-2}$			Percent Cover			Biomass Mg ha^{-1}		
			Annual	Shrub	Tree	Annual	Shrub	Tree	Annual	Shrub	Tree
Site 1	1	Hummock	1.2	0.0	0.0	36.0	2.0	1.0	1052.8	25.4	6.5
		Depression	2.0	0.4	0.0	52.8	7.0	1.0	2194.8	280.8	13.0
	2	Hummock	1.1	0.3	0.2	29.5	6.5	4.3	848.3	193.3	333.1
		Depression	1.7	0.5	0.9	57.3	15.3	22.5	1198.7	1222.6	2746.0
	3	Hummock	0.6	0.2	0.0	17.8	8.7	1.0	386.2	128.1	14.2
		Depression	1.3	0.4	0.1	30.8	6.5	7.0	1065.7	241.0	30.3
Site 2	1	Hummock	0.2	0.1	0.2	9.0	5.3	9.0	66.2	156.8	500.1
		Depression	0.6	0.1	0.1	24.2	2.7	4.0	558.9	149.5	132.5
	2	Hummock	0.1	0.0	0.1	5.2	3.0	2.3	88.2	69.8	133.7
		Depression	0.2	0.8	0.1	9.8	26.0	1.0	199.9	1162.6	40.9
	3	Hummock	0.3	0.0	0.0	11.3	2.3	0.0	247.8	37.9	0.0
		Depression	0.7	0.3	0.3	27.6	13.0	15.0	864.8	473.5	605.4
Site 3	1	Hummock	1.2	0.4	0.2	37.7	13.6	3.0	1199.1	302.7	140.2
		Depression	1.8	0.7	0.1	44.6	18.3	2.0	2165.8	554.8	123.4
	2	Hummock	1.2	0.1	0.0	50.8	5.3	1.0	1259.5	114.5	26.7
		Depression	1.2	0.4	0.1	40.2	16.8	1.8	1106.7	299.3	116.7

3	**Hummock**	1.2	0.3	0.2	27.6	10.3	2.5	1280.2	320.7	299.3
	Depression	1.1	0.5	0.2	30.0	11.3	5.0	1044.2	345.0	283.7

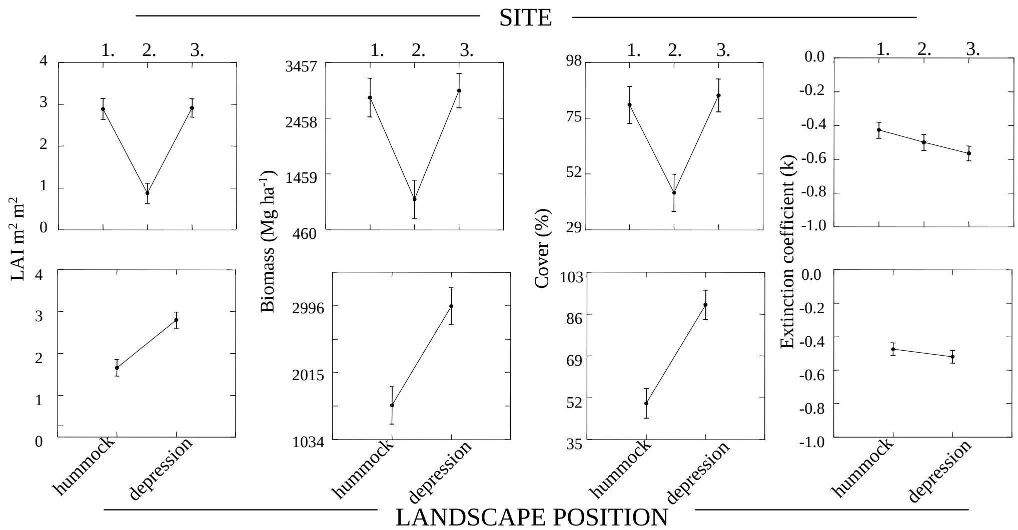

Figure 47 Least square means of leaf area index (LAI), dry biomass, percent cover and Beer`s extinction coefficient for different sites and landscape positions, error bars represent mean squared error.

Interestingly, unlike Sites 1 and 2, Site 3 was observed to have relatively heavy cover of fireweed in depressions and on hummocks or ridges. Before harvest the site was an aspen forest and due to the richness and more neutral pH of deciduous litter fall, surface soil conditions are probably more conducive to seed germination at this site relative to the more acidic litter left by coniferous vegetation. The potential differences in litter quality on the soil surface may have resulted in the greater vegetative cover on uplands at Site 3.

Biomass and Leaf area index, corrected for landscape position

Since leaf area and biomass were significantly different between depressions and hummocks (Figure 4) reasonable estimates of parameters for the entire area must take into account these variations in landscape position. LIDAR imagery, because of its capability to accurately identify elevation differences of 35 cm allowed estimates of small scale topographic differences and consequently avoided the necessity to carry out large grid based sampling regimes to capture landscape heterogeneity. Based on LIDAR imagery (Figure 5) we were able to calculate the percentage of total area the depressions occupied at the three sites, with Site 1: 17% Site 2: 37% Site 3: 24%. Correspondingly, hummocks occupied 83%, 63% and 76% of Sites 1, 2 and 3 respectively. This procedure provides a means of creating an area weighted estimate of leaf area index, biomass and percent cover that is in certain cases, such as in Site 1, where the weighted average LAI, for example is approximately 20% lower than an estimate based on a simple mathematical average (Figure 6). On the other hand, at Site 3 where there is little difference between vegetation growth in depressions and on hummocks, there was little difference between area weighted averages and mathematical averages. The result suggests that as vegetation heterogeneity increases at a site, the importance of incorporating techniques to correct vegetation parameters for heterogeneity increases.

Model Parameters (k and RUE)

While the characteristics of vegetation growth, such as LAI and biomass varied from site to site, the parameters used to describe growth such as extinction coefficients (k) and did not vary significantly among sites (note Beer's light extinction coefficient in Figure 4). Extinction coefficients for Beer's equation demonstrated considerable variability ranging from nearly 0.77 to 0.35, however as could be expected the average value was close to 0.49, a value that is typically

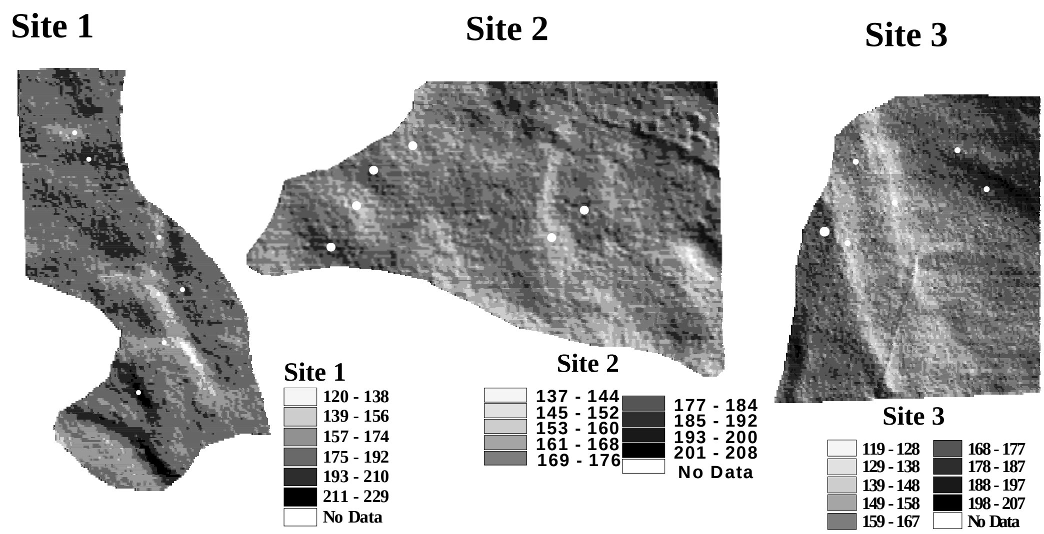

Figure 48 Delineation of forest polygons with consistent vegetation characteristics. Pixilation of LIDAR imagery for distribution of landscape features, light colours represent depressions

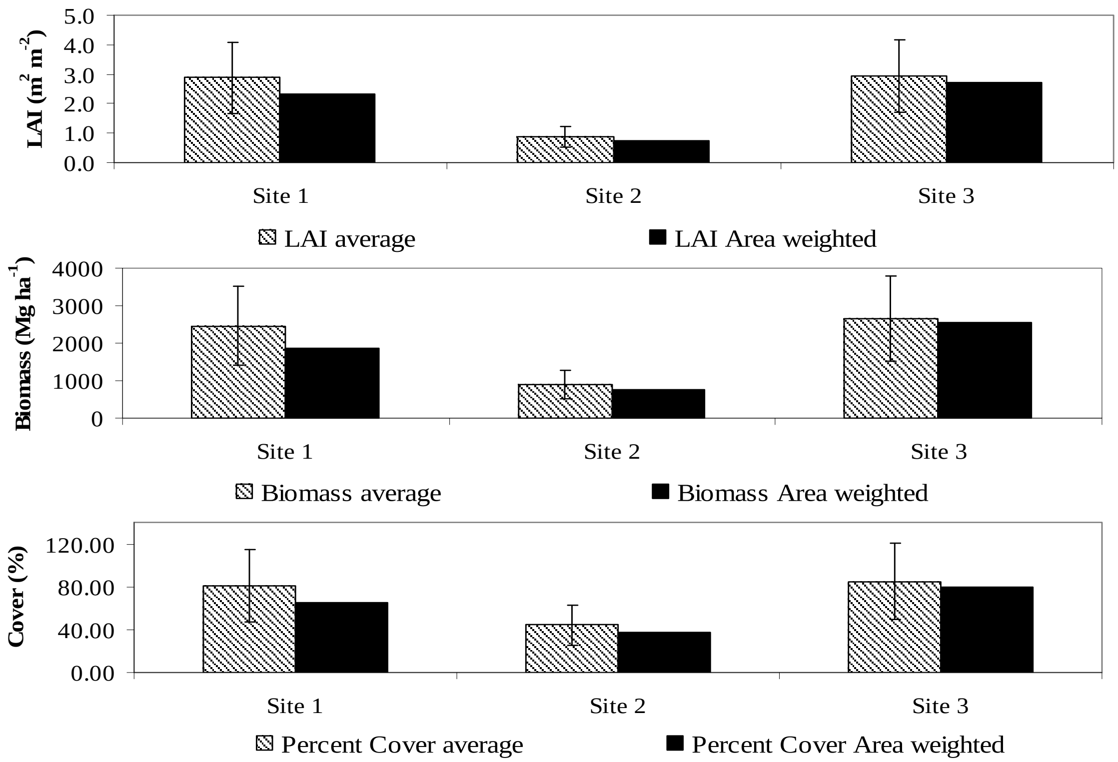

Figure 49 Corrections to leaf area index (LAI), biomass and percent cover based on the distribution of area between hummocks and depressions

used for forest vegetation (Waring and Running 1998). The high variability in measurements is due to the very wide variety of plants that exist within the sites as well as the variability in plant heights. Plants could be very heavily clumped together in certain plots resulting in very high light extinction. In other cases very short plants or plants that essentially creep along the surface may not have contributed significantly to light absorption measurements.

Model runs optimizing the fit of simulated and measured biomass and LAI data simultaneously allowed estimates of radiation use efficiency (RUE). We observed very low RUE for both annual forbs and perennial shrubs colonizing the sites relative to those typically observed for agricultural crops (Neitsch et al. 2002). Estimated mean values of 4.9 ± 1.7 Kg ha^{-1} per MJ m^{-2} for annual species and 3.3 ± 2 Kg ha^{-1} per MJ m^{-2} for shrubs were obtained from the model runs. Typically very dry land species such as mesquite may have RUE values as low as 10 Kg ha^{-1} per MJ m^{-2} (Neitsch et al. 2002). However, RUE values are rarely observed below 10 Kg ha^{-1} per MJ m^{-2}. It is possible that model simulations underestimate vegetation stresses. However, it should be considered that these species are typically understorey plants that are adapted to difficult growing conditions and consequently may have developed strategies that allot greater energy towards LAI as opposed to biomass.

Conclusions and Future Directions

We observed important differences between vegetation cover three years after harvest. Our results agree with hypotheses by Godin et al. (in prep) that vegetation reestablishment after harvest varies according to the edatopic grid. Consequently LAI, and biomass will increase as we move from poor xeric (dry) sites to rich hygric (moist) sites. Within sites, we observed important variations in vegetation cover on hummocks and in depressions associated with in-site moisture distributions. Consequently sampling regimes intended to characterize vegetation in clear-cuts must take into account vegetation heterogeneity associated with topographical features. The use of LIDAR and targeted sampling regimes proved effective but this hypothesis should be verified against complete grid sampling procedures.

The work carried out as a part of this study provides data on important parameters required in simulating forest growth and regrowth after harvest. Important differences in vegetation establishment after harvest exist among ecosites; however, there does not appear to be a consistent pattern to the variability of vegetation parameters used to describe plant growth. Consequently the most effective approach to modeling growth in these complex sites is to use average parameters, but vary maximum plant cover among ecosites.

Future work will focus on using the SWAT-C/ALMANAC$_{BF}$ coupled model to carry out sensitivity analysis that will focus on identifying the impact of variations in the magnitude and rate of vegetation recolonization on the water balance in forested watersheds. Consequently, we will identify if hydrological impacts to watershed discharge caused by forest harvest can be managed or modified by taking into consideration the early trends in vegetation reestablishment after disturbance in forested watersheds.

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