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Modeling Riverine Nitrate Export from an East-Central Illinois Watershed Using SWAT

X. Hu,* G. F. McIsaac, M. B. David, and C. A. L. Louwers

ABSTRACT

Reliable water quality models are needed to forecast the water quality consequences of different agricultural nutrient management scenarios. In this study, the Soil and Water Assessment Tool (SWAT), version 2000, was applied to simulate streamflow, riverine nitrate (NO_3) export, crop yield, and watershed nitrogen (N) budgets in the upper Embarras River (UER) watershed in east-central Illinois, which has extensive maize-soybean cultivation, large N fertilizer input, and extensive tile drainage. During the calibration (1994–2002) and validation (1985–1993) periods, SWAT simulated monthly and annual stream flows with Nash-Sutcliffe coefficients (E) ranging from 0.67 to 0.94 and R^2 from 0.75 to 0.95. For monthly and annual NO_3 loads, E ranged from –0.16 to 0.45 and R^2 from 0.36 to 0.74. Annual maize and soybean yields were simulated with relative errors ranging from –10 to 6%. The model was then used to predict the changes in NO_3 output with N fertilizer application rates 10 to 50% lower than original application rates in UER. The calibrated SWAT predicted a 10 to 43% decrease in NO_3 export from UER and a 6 to 38% reduction in maize yield in response to the reduction in N fertilizer. The SWAT model markedly overestimated NO_3 export during major wet periods. Moreover, SWAT estimated soybean N fixation rates considerably greater than literature values, and some simulated changes in the N cycle in response to fertilizer reduction seemed to be unrealistic. Improving these aspects of SWAT could lead to more reliable predictions in the water quality outcomes of nutrient management practices in tile-drained watersheds.

Nitrate (NO_3) export from the intensively cultivated midwestern U.S. corn belt has been linked to hypoxia in the northern Gulf of Mexico (Rabalais et al., 2001). As one of the most intensively fertilized regions in the USA, east-central Illinois has been identified as a major contributor to NO_3 flux in the Mississippi River (Goolsby et al., 1999). The average annual NO_3 export from east-central Illinois watersheds typically ranges from 25 to 35 kg N ha^{-1} yr^{-1} (David et al., 1997; Demissie and Keefer, 1996; Mitchell et al., 2000; McIsaac and Hu, 2004; Kalita et al., 2006). Tile drainage systems in the east-central Illinois region also promote the NO_3 export from agricultural fields (McIsaac and Hu, 2004). Field observations have shown that NO_3–N concentration in tile drainage and receiving streams often exceeds the U.S. Environmental Protection Agency (USEPA) drinking water standard (10 mg L^{-1}), with individual tile NO_3–N concentrations at times >30 mg L^{-1} (Keefer et al., 1996; Keefer et al., 1997; David et al., 1997; Gentry et al., 2000; Kalita et al., 2006).

Among the different nutrient management practices, eliminating excess N fertilizer application has been regarded as one of the most effective means to decrease NO_3 loadings to streams. Jaynes et al. (2001) showed that fertilizer application rates had a significant impact on NO_3 loss from tile drainage in central Iowa. Using a mass balance–based regression model, McIsaac et al. (2001) suggested that a 12% reduction in N fertilizer application in the Mississippi River basin could result in a 33% of reduction in NO_3 flux from the river into the Gulf of Mexico. A modeling study on two watersheds in Iowa indicated that reducing N fertilizer application by 10 to 33% along with implementation of other best management practices could result in a riverine N export reduction of more than 50% (Vaché et al., 2002).

A comprehensive decision-making tool for water quality management, the Soil and Water Assessment Tool (SWAT), was developed to simulate water budgets, sediment yield, and nutrient fluxes by accounting for a wide variety of climatic, soil, topographic, and land use factors. Since its creation in early 1990s, SWAT has been continuously revised (Arnold et al., 1998; Neitsch et al., 2002). Recently, a tile drainage component was added to the model (Neitsch et al., 2002).

A number of SWAT studies have focused on regions with tile drainage systems. Arnold and Allen (1996) used SWAT to simulate hydrologic budgets from three watersheds, two of which were located in the extensively tile-drained Central Illinois region. The tile drainage component of SWAT had not been developed when that study was conducted. They found values of R^2 ranging from 0.63 to 0.95 for monthly streamflow simulation, with systematic overestimation in low-flow months and underestimation in high-flow months during a 3-yr period. Manguerra and Engel (1998) used SWAT on a 113.4 km^2 tile drained Indiana watershed and obtained a Nash-Sutcliffe coefficient (E) of 0.48 and an R^2 of 0.82 for monthly flow from 1991 to 1995. Without available formulations to account for the tile drainage effects, they adjusted curve number and the return flow parameters to offset the impacts of subsurface drainage on the rainfall-runoff response. They further suggested an extension of SWAT to directly handle the subsurface drainage processes. Vaché et al. (2002) applied SWAT on a 51-km^2, tile-drained agricultural watershed in Iowa and reported an R^2 value of 0.67 for monthly flow in a 7-yr period, without mentioning the configuration of the tile drainage simulation. Singh et al. (2005) used SWAT

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Abbreviations: CN, curve number; $\overline{D}$, mean relative error; E, Nash-Sutcliffe coefficient; HRU, hydrologic response unit; MSE, mean squared error; SWAT, Soil and Water Assessment Tool; UER, Upper Embarras River.

with the new tile drainage component to simulate stream flows in a 5600 km^2 river basin in northern Illinois and Indiana, which includes intensively tile-drained subbasins. They obtained E values ranging from 0.80 to 0.93 for monthly streamflow simulation in wet and dry validation periods, respectively.

Several SWAT studies have focused on the simulation of pollutant loads in different regions in the USA (Saleh et al., 2000; Santhi et al., 2001; Vaché et al., 2002; Chu et al., 2004). However, the model's accuracy in simulating the riverine NO_3 flux from intensively cultivated and tile-drained midwestern regions, such as east-central Illinois, has not been tested adequately. Also, few studies have focused on the model's effectiveness in simulating crop yield and N budgets, which are important aspects in agricultural nutrient management programs.

A comprehensive model structure makes SWAT a broadly applicable tool for predicting the water quantity and quality outcomes from alternative land management practices. However, only a relatively small number of studies have reported using the model to evaluate how alternative management practices might affect water quality. Fohrer et al. (2001) used SWAT to investigate the hydrologic response to land use change in a German watershed. Vaché et al. (2002) used SWAT to simulate a reduction of N fertilizer application by 10 to 33% along with implementation of other best management practices, and the model suggested that this would result in more than 50% reduction in riverine N export from tile-drained agricultural watersheds.

The objectives of our study were (i) to calibrate and evaluate the SWAT model (version 2000) for simulating stream flow, riverine NO_3 flux, crop yield, and watershed N budgets for a watershed in east-central Illinois and (ii) to use SWAT to predict the impact of reducing N application rates on watershed NO_3 export and crop yield.

MATERIALS AND METHODS

Watershed Description

The upper Embarras River watershed (UER) located in east-central Illinois (Fig. 1) is 482 km^2 in area, with 97% in maize and soybean rotation (Table 1). The dominant soils (Table 1) in UER are poorly and somewhat poorly drained Mollisols developed from loess over Wisconsinan glacial till. The majority of the soils covering 88% of the watershed are identified as Drummer and Flanagan, which are closely related and belong to the Drummer-Flanagan Association (Mount et al., 1980). The watershed is nearly flat, with slopes mostly less than 1%. Irrigation is not significant, but tile drainage systems are widely installed in the watershed. David et al. (1997) estimated that 75 to 80% of the fields have tile drainage and later estimated drain tile density ranging from 2.8 to 5.3 km tile km^2 in different portions of the watershed, based on analysis of infrared photography (David et al., 2003). Based on the climatic data measured at Urbana weather station in Champaign County (National Climatic Data Center, 2004), the long-term (1954–2003) average temperature and annual precipitation in the studied region were 11.0°C and 996 mm, respectively. On average, approximately 5% of the annual precipitation falls as snow, and 60% of the annual precipitation occurs between March and August.

Data Sources and Watershed Delineation

The watershed boundary was delineated using digital elevation data and the spatial analysis tools provided by the

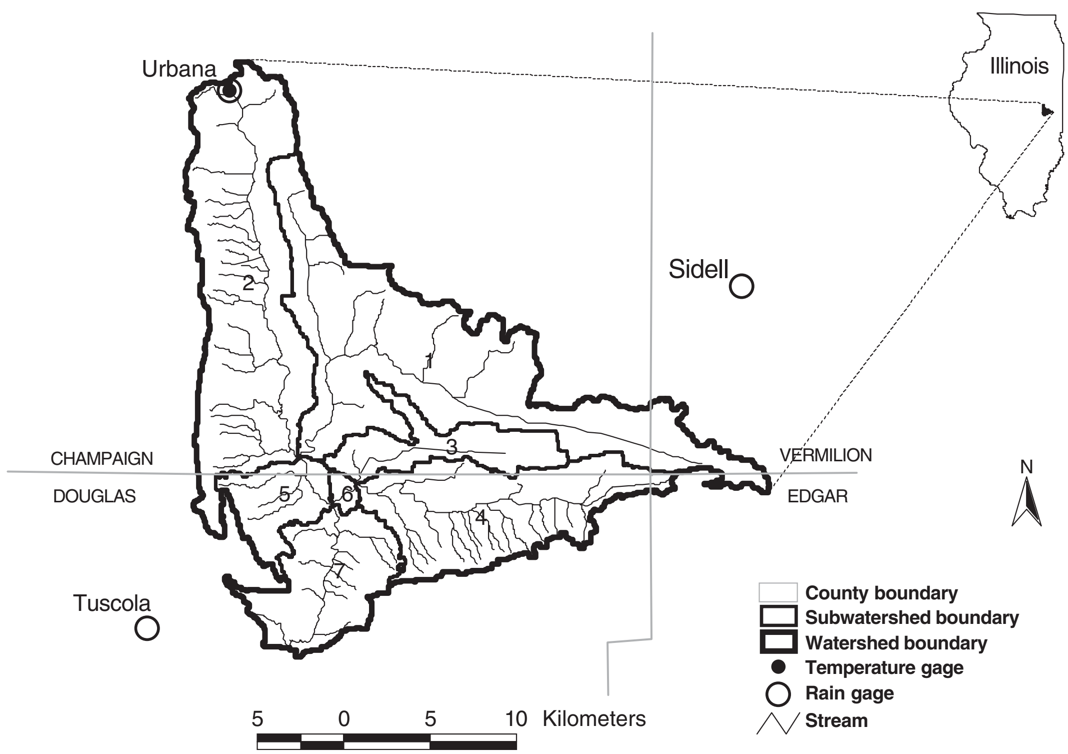

Fig. 1. Upper Embarras River watershed and subwatershed divisions used in this study.

Table 1. Land cover and soil types classified by BASINS for the study area.

Land cover/soil type	Before HRU¶ delineation		After HRU delineation	
	Area	Watershed area	Area	Watershed area
	ha	%	ha	%
Upper Embarras River watershed	47800	100	47800	100
Land use				
Row crop	46500	97.3	47800	100
Forest	150	0.3	0	0
Orchard	250	0.5	0	0
Urban	900	1.9	0	0
Soil series				
Flanagan†	17100	52.2	31700	66.4
Drummer‡	24900	35.8	15700	32.9
Camden§	5700	12.0	300	0.7

† Fine, smectitic, mesic Aquic Argiudolls.
‡ Fine-silty, mixed, superactive, mesic Typic Endoaquolls.
§ Fine-silty, mixed, superactive, mesic Typic Hapludalfs.
¶ Hydrologic response unit.

BASINS 3.0 software. The watershed was divided into seven subwatersheds (Fig. 1) using the default threshold area (the minimum drainage area required to support a permanent headwater stream) of 2900 ha suggested by BASINS. Because the study watershed is largely uniform in land use, relief, and soil properties, the single hydrologic response unit (HRU) option was used for the HRU delineation (i.e., each subwatershed is defined as a single unit with a homogeneous soil type and land use). Because of this treatment, the land use and soil type data changed slightly after HRU delineation (Table 1).

Precipitation and temperature for UER were represented by three rain gauges at Urbana, Tuscola, and Sidell and one temperature gauge at Urbana (Fig. 1). Daily precipitation and temperature data were obtained from the National Climatic Data Center (2004). This study assumes that the three rain gauges, located near the three angles of the triangle-shaped UER, have accounted adequately for the spatial variability of the precipitation within the watershed. Because temperature usually does not vary significantly over a geographically uniform region such as UER, the Urbana station alone was thought to provide sufficiently accurate temperature information. For missing data (<1%) at each station, SWAT automatically generated values based on historical weather records.

All geographic information about the watershed, including boundary, elevation, soil type, and land cover, were obtained from the USEPA BASINS database (USEPA, 2002), where the GIS data were categorized by the US Geological Survey (USGS) eight-digit hydrologic unit code. According to the USGS hydrologic unit map, UER is located in the Embarras River basin (USGS cataloging unit 05120112).

Annual crop yields and the areas of land planted with maize and soybean in the counties where the watershed was located were obtained from the National Agricultural Statistics Service (USDA-NASS, 2003). Crop yields of the watershed were calculated in an area-weighted fashion based on the proportions of the watershed in each involved county. The statistics indicate that the acreage planted with maize in this region is generally equal to the acreage with soybean every year. This condition was reflected in SWAT configuration.

Nitrogen fertilizer application rates were estimated based on the county-level fertilizer sales data from the Illinois Commercial Fertilizer Tonnage Reports (July through June basis) from 1978 to 2002. It was assumed that all N fertilizer sold was used for maize production only because there is a negligible amount of land cultivated with other crops requiring N fertilizer. The per area fertilizer input for maize in the watershed was estimated by averaging the fertilizer application rates over the four surrounding counties (total fertilizer amount divided by the total maize acreages). A 3-yr moving average value was used for each year to adjust for late or inaccurate reporting. It was assumed that half of the fertilizer was applied in fall and the other half in spring.

Daily streamflow data were obtained from the USGS station at Camargo (site no. 03343400), which has been operated continuously since 1960. River NO_3 concentrations were measured by two different agencies. From the late 1970s, the Illinois Environmental Protection Agency measured river NO_3 concentration (cadmium reduction method) at Camargo at a frequency of about nine samples per year. From December 1992 through December 2002, the Department of Natural Resources and Environmental Sciences at the University of Illinois has measured NO_3 concentration (ion chromatography) at Camargo on a daily, weekly, or biweekly basis depending on flow (Royer et al., 2006), resulting in a total of 492 samples.

Nitrate loads were estimated using the linear interpolation method. The method assumes that NO_3 concentration changes linearly over time between successive observations. Thus, daily NO_3 load can be estimated by the multiplying interpolated daily concentrations and the corresponding daily stream flows.

Initial Parameter Configurations

Although SWAT provides default values for all its parameters, it is recommended that users specify parameter values based on known characteristics of study area to make the simulations more reliable and physically meaningful. Table 2 displays the parameters that were specified based on the phys-

Table 2. Adjusted Soil and Water Assessment Tool, version 2000, parameters before calibration based on the physical conditions of the studied watershed.

Parameter name	Physical meaning	Original value	Adjusted value	Reference
SMFMX	The maximum snow melt factor (mm d^{-1} °C^{-1})	4.5	7.1	Neitsch et al., 2002
SMFMN	The minimum snow melt factor (mm d^{-1} °C^{-1})	4.5	2.0	Neitsch et al., 2002
CH_N(1)	Manning's coefficient for the tributary channel	0.014	0.03	Chow, 1959
CANMX	Maximum canopy water storage (mm)	0	0.8	Burt et al., 2002
TDRAIN	Time required to drain soil from saturation to field capacity (h)	0	48	Marshall et al., 1996
GDRAIN	Drain tile lag time (the amount of time for water to transport through the drain tile) (h)	0	1	Drablos and Moe, 1984
GW_SPYLD	Specific yield of shallow aquifer	0.003	0.14	Dingman, 2002
GWNO3	Average NO_3–N concentration in groundwater (mg L^{-1})	0	0.1	IEPA, 2002
BLAI	The maximum potential leaf area index for maize	3	5	Sogbedji and McIsaac, 2002
CN_YLD for maize	Nitrogen content in maize seed	0.014	0.013	McIsaac et al., 2002
CN_YLD for soybean	Nitrogen content in soybean seed	0.065	0.059	McIsaac et al., 2002
FRT_LY1	The fraction of fertilizer applied to top 10 mm of soil	0.2	0.01	Hoeft and Peck, 2002

ical conditions of UER. These parameters were not further adjusted during the calibration procedures.

The initialization of SWAT parameters also includes the description of a farming schedule. In this study, we assume that maize and soybean were rotated on an annual basis. The farming schedule was created based on the Illinois Agronomy Handbook (Hoeft and Nafziger, 2003) and Illinois Agricultural Statistics Service (2003) on the timing of planting and harvest. For maize years, spring N fertilizer was applied on 1 April, tillage occurred on 20 April, planting on 27 April, and harvest on 15 October. For soybean years, tillage occurred on 14 May, planting on 21 May, harvest on 10 October, and fall fertilizer application for the subsequent maize crop on 1 November. Anhydrous ammonia was chosen to be the only form of fertilizer for both spring and fall application. As suggested by the Illinois Agronomy Handbook (Hoeft and Nafziger, 2003), anhydrous ammonia was set to be placed 15 to 20 cm deep into the soil. Accordingly, the fraction of N applied to top 10 mm of soil (an input parameter) was assumed to be 1% to account for upward diffusion of the applied ammonia.

Model Calibration and Validation

To calibrate the model, a total of 13 parameters were adjusted, including seven hydrologic parameters, four nutrient parameters, and two plant growth parameters (Table 3). Most existing SWAT studies conducted a manual calibration, which can be tedious and inefficient and can lead to mislocation of the global optimum. In this study, we developed an automated screening technique to perform SWAT calibration. The technique is basically a systematic trial-and-error strategy that guides the computer to find sufficient approximations to the global optimum. The calibration was designed with three stages: hydrologic calibration, riverine NO_3 load calibration, and crop yield calibration. At each stage, the following steps were performed.

Step 1: Selection of Parameters

A number of important parameters were selected, and each was assigned with a value range based on the SWAT manual (Neitsch et al., 2002), existing literature (Sophocleous et al., 1999; Arnold et al., 2000; Spruill et al., 2000; Santhi et al., 2001), and personal judgments. For parameters that have multiple values due to spatial and temporal variation (e.g., soil hydraulic conductivity and available water holding capacity vary across soil types and layers; curve numbers change between growing and nongrowing seasons), their values were increased or decreased proportionately. For instance, a change in a value in one soil type or soil layer was accompanied by a change in all others by the same ratio.

Table 3. Soil and Water Assessment Tool, version 2000, parameter values before and after calibration based on 1994–2002 observations in the upper Embarras River watershed.

Parameters description (name in SWAT)	Initial value	Calibrated value
Hydrologic calibration		
Curve number (CN)		
Growing season	82	57.4
Nongrowing season	78	54.6
Soil evaporation compensation coefficient (ESCO)	0.95	0.9
Plant transpiration compensation coefficient (EPCO)	1	0.9
Soil saturated hydraulic conductivity (SOL_K) adjusting factor (%)	0	+30
Soil available water capacity (SOL_AWC) adjusting factor (%)	0	−5
Drain tile depth (DDRAIN) (mm)	1200	1100
Nutrient calibration		
Mineralization rate coefficient for the fresh crop residue N (RSDCO)	0.05	0.02
Mineralization rate coefficient for the humus active organic N (CMN)	0.003	0.001
Fraction of porosity from which anions are excluded (ANION_EXCL)	0.5	0.3
Denitrification rate coefficient (PARM3)	0.8	0.3
Crop yield calibration		
Plant radiation use efficiency for maize (RUE) (10^{-1} g MJ^{-1})	39	46
Plant radiation use efficiency for soybean (RUE) (10^{-1} g MJ^{-1})	25	26

Step 2: Formation of Grid

The value range for each selected parameter was divided into a certain number of intervals so that a grid was formed in the parameter space. For example, if N parameters were used for calibration, and their value ranges were divided into n_1, n_2, . . ., n_N intervals, respectively, then a N-dimension grid with $(n_1 + 1) \times (n_2 + 1) \times \ldots \times (n_N + 1)$ intersections would be formed. Each of the intersections in the grid would specify a unique combination of different parameter values, which produces a corresponding set of simulation outputs.

Step 3: Enumeration

The SWAT model was run with each set of parameter values specified by the grid created in the previous step. After each run, the SWAT outputs were read out and evaluated for goodness-of-fit. The goodness-of-fit was measured by four indicators: mean relative error ($\overline{D}$), mean squared error (MSE), Nash-Sutcliffe coefficient (E) (Nash and Sutcliffe, 1970), and R^2. They were calculated as follows:

$$\overline{D} = \frac{\sum(\hat{X}_i - X_i)}{\sum X_i} \quad [1]$$

$$\mathrm{MSE} = \frac{\sum(\hat{X}_i - X_i)^2}{N} \quad [2]$$

$$E = 1 - \frac{\sum(\hat{X}_i - X_i)^2}{\sum(X_i - \overline{X})^2} \quad [3]$$

$$R^2 = \frac{\left[\sum\left(\hat{X}_i - \overline{\hat{X}}\right)(X_i - \overline{X})\right]^2}{\sum\left(\hat{X}_i - \overline{\hat{X}}\right)^2 \sum(X_i - \overline{X})^2} \quad [4]$$

where $\hat{X}_i$ and X_i are individual simulated and individual observed values, respectively; $\overline{\hat{X}}$ and $\overline{X}$ are the mean of simulated and the mean of observed values, respectively; and N is the number of observations.

Step 4: Identify the Optimum

The optimum set of parameter values for each calibration stage should achieve the minimal MSE value subject to the following constraints:

For hydrologic calibration:

The ratio of mean annual tile flow to mean annual water yield >75%. Because there are no direct tile flow measurements available for this study, the acceptable average tile to total flow ratio was set to 75% for the calibration based on observations in a neighboring watershed (Mitchell et al., 2000).

$\overline{D}$ in mean annual water yield within ±15% range.

$E > 0$ for monthly NO_3 flux simulation. Different parameter combinations may result in acceptable hydrologic simulations but unacceptable NO_3 flux simulations. Because hydrologic processes influence NO_3 transformations and transport, this restriction was set to reject the hydrologic parameter sets that lead to poor nutrient calibration. This same concept applies to the crop yield restrictions in the hydrologic and the NO_3 calibrations.

$E > 0$ or $\overline{D}$ within ±20% range for maize and soybean yields.

For NO_3 calibration:

Error in mean annual NO_3 flux within ±25% of the value.

$E > 0$ or $\overline{D}$ within ±15% range for maize and soybean yields.

For crop yield calibration:

$\overline{D}$ within ±10% range.

Once one stage was accomplished, the calibration proceeded to the next stage. The three calibration stages were iterated several times to allow the parameter values to converge. The selected calibration period was from 1994 to 2002, when the water quality measurements were more frequent than other time periods. The accuracy of the calibrated model was then validated by comparing simulation output to measured values under time periods not used in the calibration, specifically, 1985 to 1993.

Evaluation of Fertilizer Reduction Scenarios

Four fertilizer reduction scenarios were evaluated in this study. The N application rates were reduced by 10, 20, 30, and 50% from the recorded values during the simulation periods to represent different nutrient management programs. Riverine NO_3 fluxes, crop yields, and soil N budgets under each level of fertilizer reduction were then predicted using the calibrated SWAT model to evaluate water quality and agricultural consequences.

RESULTS AND DISCUSSION

Calibrated Parameter Values

The most notable adjustment to parameter values during calibration (Table 3) was a reduction of the curve numbers (CN) from the initial range of 78 to 82 to a range of 55 to 58, which is lower than the values typically used for continuous row cropping on fine textured soils. This is because the existence of tile drainage largely reduces the runoff and nutrient washout occurring at soil surface. The SWAT model would not be able to capture this change with the typical CN values. Early simulations showed that using typical CN values produced relatively high quantities of surface runoff with substantially lower NO_3 concentrations than observed values. Singh et al. (2005) calibrated the hydrologic component of SWAT to a basin similar to UER (~100 km to the north) and reported an optimal CN of 67, which is also lower than values commonly applied to annual row crops.

Stream Flow

Using the calibrated parameter values, SWAT successfully reproduced the stream runoff from UER during the calibration period. Simulated monthly and annual flows were in good agreement with the observations (Fig. 2a and 3a), with E values of 0.85 and 0.67 and R^2 values of 0.87 and 0.76, respectively (Table 4). During the validation period, SWAT simulations of the UER were nearly as accurate (Fig. 2b and 3b), with E values of 0.69 and 0.94 and R^2 values of 0.75 and 0.95 for monthly and annual stream runoff, respectively (Table 4). There was a tendency for SWAT to overestimate by approximately 20% monthly flows in excess of 50 mm during the validation period. Examination of residuals did not allow us to identify a single cause for this overestimation, but most of the overestimation occurred from September to March, outside of the main growing season. This could be due to underestimation of dormant season ET and inadequate simulation of freeze-thaw effects. There was about 26% more September to March precipitation during the validation period than in the calibration period, which led to 80% greater observed water yield. The greater water input and output evidently magnified inaccuracies in the calibration that were not significant under the lower flow regime.

The monthly simulation results were comparable to those of Arnold and Allen (1996), who reported R^2 values between 0.63 and 0.95 for monthly flow in three Illinois watersheds during a 22-mo period. However, Arnold and Allen (1996) reported a tendency for overestimation of low flow months and underestimation in high flow months, whereas we observed an overestimation of high flow months. Our monthly simulation results

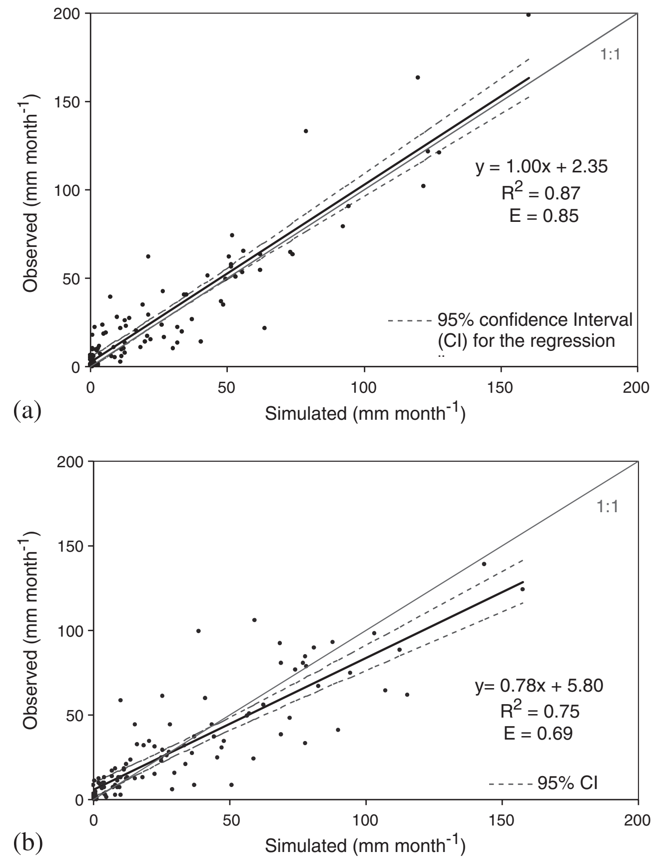

Fig. 2. Simulated vs. observed monthly stream flow in the upper Embarras River for (a) the calibration period (1994–2002) and (b) the validation period (1985–1993).

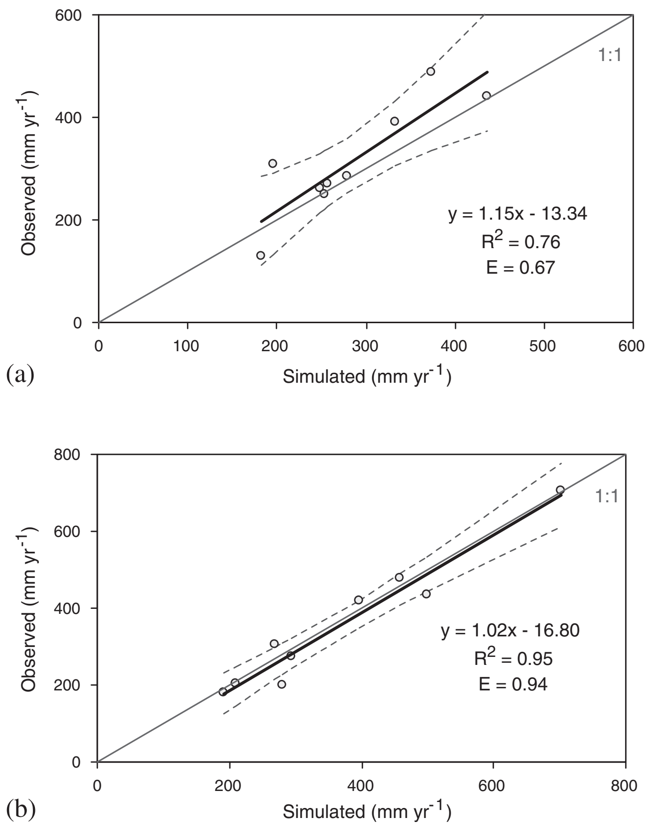

Fig. 3. Simulated vs. measured annual stream flow in the upper Embarras River for (a) the calibration period (1994–2002) and (b) the validation period (1985–1993).

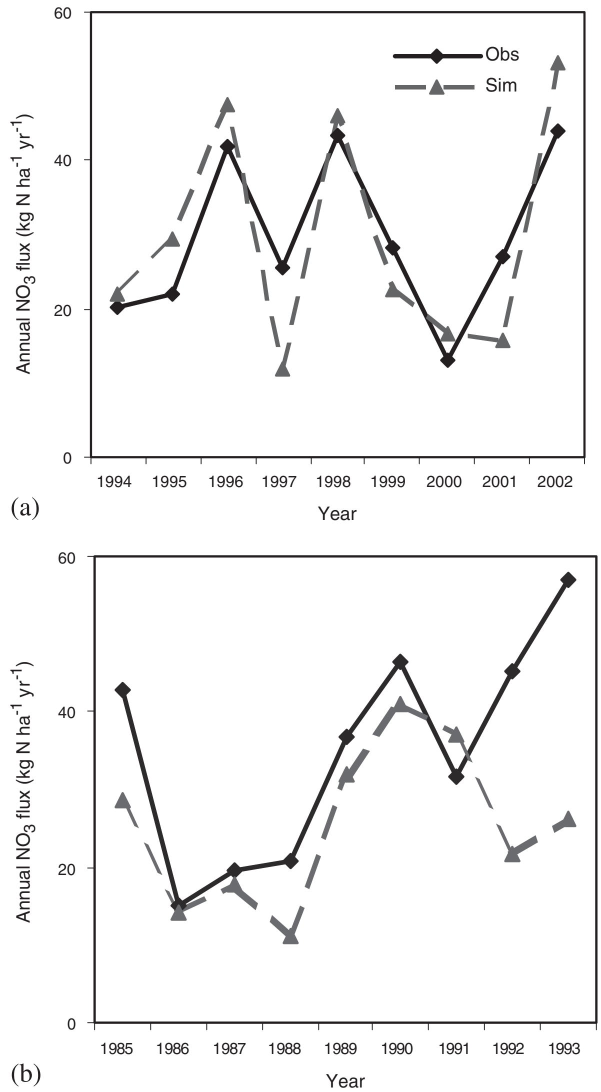

Fig. 4. Time series of simulated and measured annual riverine NO_3 flux in the upper Embarras River for (a) the calibration period (1994–2002) and (b) the validation period (1985–1993).

are more accurate than Manguerra and Engel (1998), who applied an earlier version of SWAT to two tile drained Indiana watersheds and obtained an E values between 0.48 and 0.77 for monthly simulations with little or no calibration.

Riverine Nitrate Flux

Compared with the stream runoff results, the simulations on riverine NO_3 flux were less accurate for calibration and validation periods (Table 4). During the calibration period, the simulated monthly and annual NO_3 fluxes from the UER produced E values of 0.2 and 0.45 and R^2 values of 0.72 and 0.74, respectively. The average annual NO_3 flux was estimated adequately with a mean relative error close to zero. The simulated annual NO_3 flux generally followed the year-to-year trend of the observations (Fig. 4a).

The seasonal variation of the monthly riverine NO_3 flux was fairly well reproduced in the calibration period (Fig. 5a), except for four large overestimations in 1995, 1996, 1998, and 2002. All of these overestimations occurred in the month of May, when the observed precipitation was greater than 185 mm, which is about 70% greater than the 1971 to 2000 average May precipitation. Although May precipitation of this magnitude occurred in 4 of the 9 yr of the calibration period, it occurred only

Table 4. Model performance statistics from observed vs. simulated for calibration and validation scenarios.†

	Stream flow					Riverine NO_3 flux					Crop yield	
	Monthly		Annual			Monthly		Annual			Maize	Soybean
Scenario	R^2	E‡	R^2	E	$\overline{D}$§	R^2	E	R^2	E	$\overline{D}$	$\overline{D}$	$\overline{D}$
Calibration	**0.87**	**0.85**	**0.76**	**0.67**	**−0.09**	**0.72 (0.8)**	**0.20 (0.53)**	**0.74**	**0.45**	**~0**	**−0.1**	**−0.02**
Validation	**0.75**	**0.69**	**0.95**	**0.94**	**0.03**	**0.51**	**0.31**	**0.36**	**−0.16**	**−0.27**	**0.05**	**0.06**

† Values in the parentheses indicate results when extreme months are removed.
‡ Nash-Sutcliffe coefficient.
§ Mean relative error.

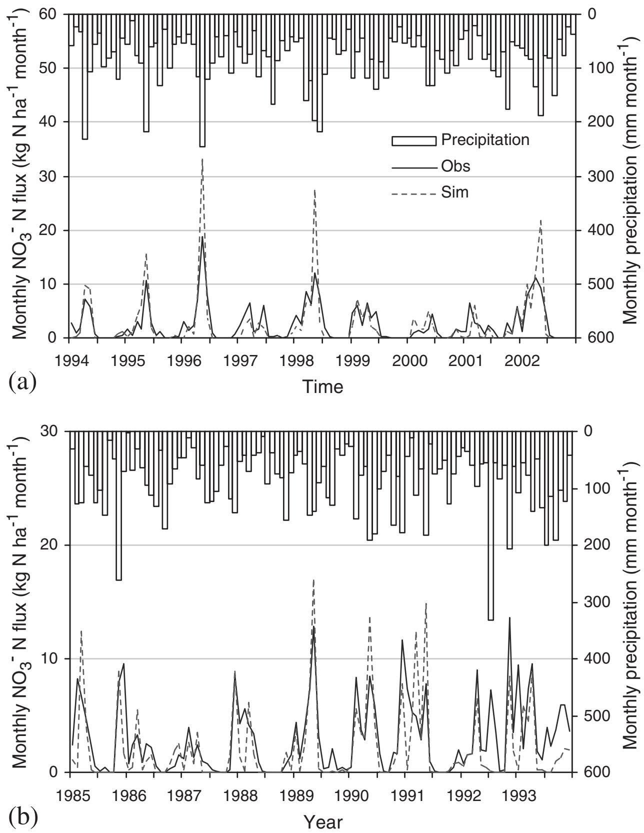

Fig. 5. Time series of simulated and measured monthly riverine NO_3 flux in the upper Embarras River for (a) the calibration period (1994–2002) and (b) the validation period (1985–1993).

one other time since 1949 (the first year in which there is continuous precipitation from all three stations used to characterize precipitation in the watershed). Frequency analysis of May precipitation between 1949 and 2005 indicates a 10% probability of May precipitation in excess of 185 mm in any given year, and a 0.3% probability that four such events would occur in any 9-yr period. Thus, the sequence of high May precipitation in the calibration period is either a statistical anomaly or a manifestation of the intensification of the hydrologic cycle, possibly a consequence of global climate change, as reported by Groisman et al. (2004). When the four outlying simulated NO_3 flux values were removed from the evaluation of the SWAT performance, the E and R^2 for monthly simulation increased to 0.53 and 0.80 (Table 4), respectively.

The large overestimations of NO_3 flux in wet months of May might be a consequence of a number of factors. In our simulations, we set half of the N fertilizer to be applied on 1 d in April as NH_3. In reality, this N is applied over several weeks, and a significant portion may be incorporated into soil organic matter. Due to the large pulsed application on a single day, the immobilization of fertilizer N into organic N may have been underestimated, leaving an excessive NO_3 pool created by nitrification of the applied NH_3. If large rainfall occurred in the following month, the excessive NO_3 would likely cause considerable overestimates of NO_3 concentration in the subsurface drainage, which would exaggerate NO_3 export. Such an effect may also explain why precipitation in excess of 185 mm that occurred in other months did not lead to large overpredication of NO_3 leaching. Second, the overprediction may be associated with the simplification in the tile drainage processes. The model simplifies the tile drainage as an underground "layer" that can immediately remove the vertically moving water from the soil profile once the water reaches the layer. This assumption largely ignores the horizontal movement of soil water between drain tiles that may take a much longer time in reality. Thus, SWAT may largely underestimate the delay and mixing of subsurface flow, which have important smoothing effects on NO_3 transport. As a consequence of both effects, SWAT may overpredict NO_3 transport in drainage water if a large rainfall event occurs soon after N fertilizer application.

During the validation period (Fig. 5b), May precipitation never exceeded 185 mm, but the riverine NO_3 flux was not as accurately reproduced as in the calibration. The E values (Table 4) were 0.31 and −0.16, and the R^2 values were 0.51 and 0.36 for monthly and annual NO_3 fluxes, respectively. Annual average NO_3 flux was underestimated by 27%. The differences between simulated and observed NO_3 fluxes were not clustered in any particular month, but large and persistent discrepancies occurred in 1992 and 1993 (Fig. 6), which further caused substantial divergence in the annual NO_3 values (Fig. 4b). This may have been a consequence of the model overestimating maize yield in previous years (1991 and 1988, discussed below). Precipitation in July 1992 was greater than 300 mm, which is the largest monthly precipitation recorded since 1949, and annual precipitation in 1993 was the greatest recorded since 1949 and 35% greater than the 1971 to 2000 average. The SWAT model accurately simulated corn yields in these years but may have overestimated denitrification.

It is possible that the unusually high precipitation in May during the calibration period led to an optimum parameter set that was not applicable to other time

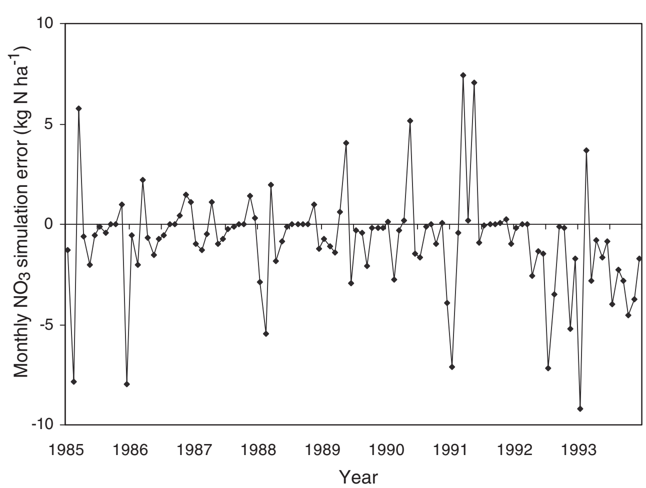

Fig. 6. Model residuals (simulated minus observed) for the validation period.

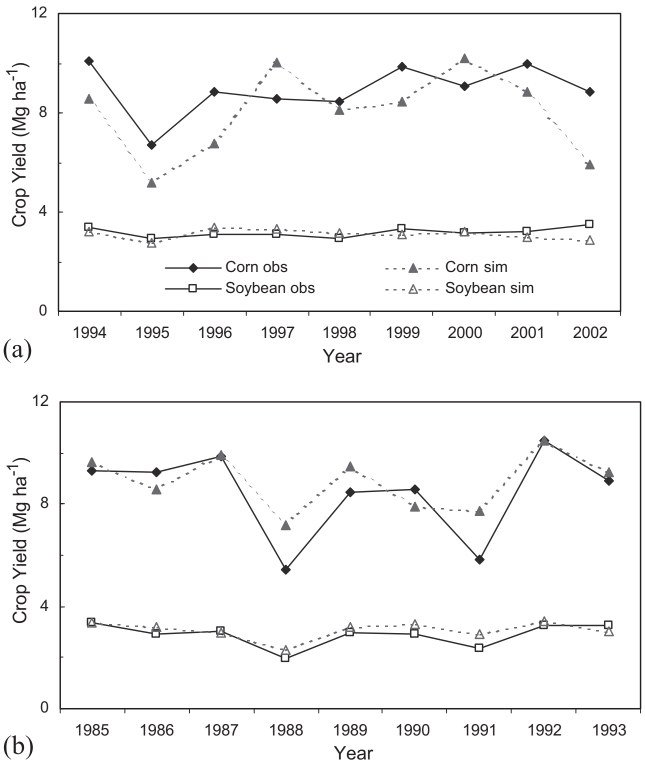

Fig. 7. Time series of simulated and measured maize and soybean yield in the upper Embarras River watershed for (a) the calibration period (1994–2002) and (b) the validation period (1985–1993).

periods. However, Louwers (2003) also used SWAT to simulate water and NO_3 fluxes in this watershed using different periods of time for calibration and obtained results that are similar to those reported here.

Crop Yield

The simulated soybean yields were generally similar to the observed values, with average relative errors of −2 and 6% for the calibration and validation, respectively (Table 4). Soybean yield varied little during the calibration period (Fig. 7). The E and R^2 values were low (−1.36 and 0.07, respectively), partly because there was little variation for the model to simulate. During the validation period, soybean yields were more variable and better simulated with an E value of 0.53.

Maize yields were more variable than soybeans but were not well simulated during calibration, with E of −0.49 and average relative error of −10%. In the validation period, the model simulated maize yield variation better (E = 0.8) but overestimated maize yields by 5% on average. The overestimation of yields was greatest during the droughts of 1988 and 1991. By overestimating maize yield, the model simulated more N removal from the soil, and less was available for leaching. This probably contributed to the underestimation of riverine NO_3 flux in the validation period.

Observed yields were estimated from the county level statistics, and the UER watershed is only 15% of the combined areas of Champaign and Douglas Counties. We would expect crop yields at the county level to be less variable than at the watershed level due to spatial averaging. This seemed to be reflected in the calibration period, where simulated yields were more variable than observed yields. In the validation period, however, simulated yields were less variable than observed yields. This may indicate that the effect of spatial averaging may be relatively small.

Nitrogen Budgets

Although SWAT has the ability to simulate all the major N inputs to and outputs from an agricultural watershed, few studies have examined the simulated N budgets. In this study, SWAT generated positive N budgets for the calibration and validation periods. The sum of the major N inputs (fertilizer application, fixation, and deposition of nitrogen oxides from atmospheric sources) was 8 to 10 kg N ha^{-1} yr^{-1} greater than the major N outputs (crop harvest, denitrification, and riverine NO_3 flux) (Table 5). This budget does not include organic N or ammonia in riverine total Kjehldahl N flux, which McIsaac and Hu (2004) estimated to be 3.3 kg N ha^{-1} yr^{-1}. It also does not take into account the net deposition of reduced forms of N, which are often assumed to be largely derived from local volatilization. Finally, it ignores N in animal and human wastes, which McIsaac and Hu (2004) estimated to be 7 kg N ha^{-1} yr^{-1} for the 1977 to 1997 period. Including these terms would further increase the net N inputs to the watershed by approximately 4 kg N ha^{-1} yr^{-1}.

The SWAT model estimated the N fixation in soybeans fields to be 172 to 206 kg N ha^{-1} yr^{-1}, which is considerably greater than the generally accepted values for central Illinois (102–124 kg N ha^{-1} yr^{-1}) (McIsaac and Hu, 2004; Gentry et al., 1998; David et al., 1997). In this study, extensive calibration failed to adjust the N fixation to a reasonable level because there is no N fixation parameter in SWAT that is directly user adjustable. Nitrogen fixation is affected by other processes, such as crop growth and soil nutrient transformations, but to a limited extent.

The simulated N in harvested crops was about 20 to 30 kg N ha^{-1} yr^{-1} greater than a previous estimate by McIsaac and Hu (2004), who calculated the average N accumulation in crop grain in several east-central Illinois

Table 5. Model simulated average annual N budgets for calibration and validation periods.

Period	Inputs			Outputs			Net gain or loss
	Fertilizer	Fixation	NO_y Deposition	Harvest	Denitrification	Riverine NO_3 flux	
	kg N ha^{-1} yr^{-1}						
Calibration	**95**	**99**	**10**	**144**	**23**	**29**	**+8**
Validation	**98**	**97**	**10**	**149**	**20**	**26**	**+10**

watersheds to be 116 kg N ha^{-1} yr^{-1} for the period 1977 to 1997. This overestimation is likely due to the following reasons: (i) The simulations assumed a higher percentage (nearly 100%) of agricultural land than reality (about 90%), and (ii) the average crop yield in the east-central Illinois region in 1985 to 2002 is about 10% higher than that in 1977 to 1997.

Evaluation of Fertilizer Reduction Scenarios

The calibrated SWAT was used to evaluate hypothetical nutrient management programs, which would reduce the fertilizer input by 10, 20, 30, and 50% from the existing application rate. Simulations suggested that the fertilizer reduction during 1985 to 2002 could decrease the average riverine NO_3 flux by 10, 18, 29, and 43%, respectively, for the same time period (Table 6). This prediction is similar to the results reported by Jaynes et al. (2001), who found that moderately fertilized fields with 33% reduction in N fertilizer input rate (from highly fertilized fields to moderately fertilizer) resulted in about 28% less NO_3 export from a 22-ha maize-soybean rotation field in Iowa.

Simulated soybean yields did not change with reduced fertilizer application, but N fixation in soybeans increased as a result of less freely available NO_3 in the soil. On the other hand, SWAT predicted a 6% reduction in maize yield from a 10% reduction of N fertilizer application (from 190 to 171 kg N ha^{-1} yr^{-1}), even though these fertilizer application rates are considered sufficient to achieve maximum yields. If the agricultural land in east-central Illinois has, as widely believed, been over fertilized, we would not expect any significant change in maize yields in response to a reduction in excessive N. It is possible that the mineralization of the soil organic N pool was underestimated or that the simulated leaching or denitrification were overestimated, which depleted the N before the maize roots are able to acquire it. Alternatively, the relationships describing the maize yield response to available N may require refinement.

Regardless of the fertilizer N rate reduction, SWAT simulated a positive N budget residual, indicating an accumulation of N in the soil. It seems more likely that producing maize with insufficient fertilizer would lead to a depletion of soil N, as reported by Jaynes et al. (2004). This result may be a consequence of overestimation of N fixation in soybeans and underestimation of N mineralization of soil organic N and subsequent losses by leaching or denitrification.

Table 6. Predicted N budgets and crop yields under reduced N fertilizer rates in the upper Embarras watershed for 1985–2002.

Parameter	% of N fertilizer reduction				
	0	10	20	30	50
	kg N ha^{-1} yr^{-1}				
N inputs					
Fertilizer	97	87	77	68	48
N_2 fixation	98	99	101	101	102
Atmospheric deposition	10	10	10	10	10
N outputs					
N removed in harvest	147	143	140	135	126
Denitrification	22	20	18	16	13
NO_3 loss to stream	28	25	23	20	16
N budget residual	+9	+8	+8	+8	+6
	Mg ha^{-1}				
Crop yields					
Maize yield	8.5	8.0	7.4	6.8	5.3
Soybean yield	3.1	3.1	3.1	3.1	3.1

SUMMARY AND CONCLUSIONS

The SWAT model was calibrated to a central Illinois watershed for a 9-yr period, and then simulations of stream flow, riverine NO_3 flux, watershed N budgets, and crop yields for a different 9-yr period were compared with observed values. The calibrated model was also used to simulate the impact of fertilizer reduction scenarios on riverine NO_3 export.

The SWAT model successfully simulated monthly stream flow, but simulation of monthly riverine NO_3 fluxes was less accurate. During the calibration period, the model largely overestimated NO_3 flux when precipitation in May was greater than a 10-yr return frequency. The model did not simulate annual variation in maize and soybean yields well in the calibration period, although there was little variation in soybean yield to simulate. In the validation period, SWAT overestimated maize yields during droughts and underestimated riverine NO_3 flux during an unusually wet period. The annual average soybean yields were successfully predicted, but the simulated soybean N fixation was substantially higher than existing estimates.

When N fertilizer rates were hypothetically reduced by 10 to 50% from the existing values, SWAT predicted a nearly proportional decrease (10–41%) in riverine NO_3 flux. Those predictions are supported by observations reported in the literature. However, a simulated 6% decline in maize yields with a reduction in fertilizer N from 190 to 171 kg N ha^{-1} yr^{-1} and the simulated N budget indicating accumulation of N in the soil regardless of the extent of N reduction seems to be inconsistent with field observations reported in the literature.

These results suggest that SWAT had a limited ability to simulate N transformations and transport, especially during some of the low-probability hydrologic conditions encountered in this study. Although these conditions have been rare in the past, they may be an indication of a changing climate. To address this, further improvements seem to be needed in SWAT's simulation of soil N processes, soybean N fixation, and maize yield simulations to better simulate the N budgets in tile-drained agricultural lands in east-central Illinois.

REFERENCES

Arnold, J.G., and P.M. Allen. 1996. Estimating hydrologic budgets for three Illinois watersheds. J. Hydrol. 176:57–77.

Arnold, J.G., R.S. Muttiah, R. Srinivasan, and P.M. Allen. 2000. Regional estimation of base flow and groundwater recharge in the upper Mississippi river basin. J. Hydrol. 227:21–40.

Arnold, J.G., R. Srinivasan, R.S. Muttiah, and J.R. Williams. 1998. Large area hydrologic modeling and assessment part I: Model development. J. Am. Water Resour. Assoc. 34:73–89.

Burt, C.M., A.J. Mutziger, R.G. Allen, and T.A. Howell. 2002. Review and interpretation of evaporation research. p. 17–66. *In* C.M. Burt et al. (ed.) Evaporation from irrigated agricultural land in California. ITRC Rep. No. R 02-001. Irrigation Training and Research Center, Calif. Polytech. State Univ., San Luis Obispo.

Chow, W.-T. 1959. Open channel hydraulics. McGraw-Hill, New York.

Chu, T.W., A. Shirmohammadi, H. Montas, and A. Sadeghi. 2004. Evaluation of the SWAT model's sediment and nutrient components in the piedmont physiographic region of Maryland. Trans. ASAE 47:1523–1538.

David, M.B., L.E. Gentry, D.A. Kovacic, and K.M. Smith. 1997. N balance in and export from an agricultural watershed. J. Environ. Qual. 26:1038–1048.

David, M.B., L.E. Gentry, K.M. Starks, and R.A. Cooke. 2003. Stream transport of herbicides and metabolites in a tile drained, agricultural watershed. J. Environ. Qual. 32:1790–1801.

Demissie, M., and L. Keefer. 1996. Watershed monitoring and land use evaluation for the Lake Decatur watershed. Illinois State Water Survey Miscellaneous Publ. 169. Illinois State Water Survey, Champaign.

Dingman, S.L. 2002. Physical hydrology. 2nd ed. Prentice Hall Inc., Upper Saddle River, NJ.

Drablos, C.J.W., and R.C. Moe. 1984. Illinois drainage guide. Circular 1226. Cooperative Extension Service, College of Agric., Consumer and Environ., Univ. of Illinois, Urbana.

Fohrer, N., S. Haverkamp, K. Eckhardt, and H.-G. Frede. 2001. Hydrologic response to land use changes on the catchment scale. Phys. Chem. Earth B 26:577–582.

Gentry, L.E., M.B. David, K.M. Smith, and D.A. Kovacic. 1998. N cycling and tile drainage NO_3^- loss in a corn/soybean rotation. Agric. Ecosyst. Environ. 68:85–97.

Gentry, L.E., M.B. David, K.M. Starks-Smith, and D.A. Kovacic. 2000. Nitrogen fertilizer and herbicide transport from tile drained fields. J. Environ. Qual. 29:232–240.

Goolsby, D.A., W.A. Battaglin, G.B. Lawrence, R.S. Artz, B.T. Aulenbach, R.P. Hooper, D.R. Keeney, and G.J. Stensland. 1999. Flux and sources of nutrients in the Mississippi-Atchafalaya River Basin. NOAA Coastal Ocean Program. Decision Analysis Series No. 17. National Oceanic and Atmospheric Administration, Silver Spring, MD.

Groisman, P.A., R.W. Knight, T.R. Karl, D.R. Easterling, B. Sun, and J.H. Lawrimore. 2004. Contemporary changes of the hydrological cycle over the contiguous United States: Trends derived from in situ observations. J. Hydrometeorol. 5:64–85.

Hoeft, R.G., and T.R. Peck. 2002. Soil testing and fertilizer. p. 91–131. *In* Illinois agronomy handbook. 23rd ed. Cooperative Extension Service, College of Agric., Consumer and Environ., Univ. of Illinois, Urbana.

IEPA. 2002. Illinois water quality report 2000. Illinois Environmental Protection Agency, Springfield.

Illinois Agricultural Statistics Service. 2003. Illinois annual summaries [online]. Available at http://www.agstats.state.il.us/website/reports.htm (accessed May 2003; verified 8 Feb. 2007).

Jaynes, D.B., T.S. Colvin, D.L. Karlen, C.A. Cambardella, and D.W. Meek. 2001. Nitrate loss in subsurface drainage as affected by nitrogen fertilizer rate. J. Environ. Qual. 30:1305–1314.

Jaynes, D.B., D.L. Dinnes, D.W. Meek, D.L. Karlen, C.A. Cambardella, and T.S. Colvin. 2004. Using the late spring nitrate test to reduce nitrate loss within a watershed. J. Environ. Qual. 33:669–677.

Kalita, P.K., A.S. Algoazany, J.K. Mitchell, R.A.C. Cooke, and M.C. Hirschi. 2006. Subsurface water quality from a flat tile-drained watershed in Illinois, USA. Agric. Ecosyst. Environ. 115:183–193.

Keefer, L., M. Demissie, S. Shaw, and S. Howard. 1997. Watershed monitoring for the Lake Decatur Watershed, 1996–1997. Illinois State Water Survey Contract Rep. 620. Illinois State Water Survey, Champaign.

Keefer, L., M. Demissie, S. Shaw, and K. Nichols. 1996. Watershed monitoring for the Lake Decatur Watershed. Illinois State Water Survey Contract Rep. 602. Illinois State Water Survey, Champaign.

Louwers, C.A. 2003. Application of the SWAT model to examine a N management program on East-Central Illinois watersheds. M.S. thesis. Univ. of Illinois, Urbana.

Manguerra, H.B., and B.A. Engel. 1998. Hydrologic parameterization of watersheds for runoff prediction using SWAT. J. Am. Water Resour. Assoc. 34:1149–1162.

Marshall, T.J., J.W. Holmes, and C.W. Rose. 1996. Soil physics. 3rd ed. Cambridge Univ. Press, New York.

McIsaac, G.F., M.B. David, G.Z. Gertner, and D.A. Goolsby. 2001. Eutrophication–nitrate flux in the Mississippi river. Nature 414: 166–167.

McIsaac, G.F., M.B. David, G.Z. Gertner, and D.A. Goolsby. 2002. Relating net N input in the Mississippi River Basin to nitrate flux in the lower Mississippi River: A comparison of approaches. J. Environ. Qual. 31:1610–1622.

McIsaac, G.F., and X. Hu. 2004. Net N input and riverine N export from Illinois agricultural watersheds with and without extensive tile drainage. Biogeochemistry 70:251–271.

Mitchell, J.K., G.F. McIsaac, S.E. Walker, and M.C. Hirschi. 2000. Nitrate in river and subsurface drainage flows from an east central Illinois watershed. Trans. ASAE 43:337–342.

Mount, H.R., C.C. Cochran, C.E. Wacker, and S. Engel. 1980. Soil survey of Champaign county, Illinois. Illinois Agric. Exp. Stn. Soil Rep. 114. USDA-SCS, Washington, DC.

Hoeft, R.G., and E.D. Nafziger (ed.) 2003. Illinois agronomy handbook. 23rd ed. Univ. of Illinois at Urbana Champaign.

Nash, J.E., and J.V. Sutcliffe. 1970. River flow forecasting through conceptual models. Part I–A discussion of principles. J. Hydrol. 10:282–290.

National Climatic Data Center. 2004. Locate weather observation station record [Online]. Available at http://www.ncdc.noaa.gov/oa/climate/stationlocator.html (accessed Oct. 2004; verified 8 Feb. 2007).

Neitsch, S.L., J.G. Arnold, J.R. Kiniry, J.R. Williams, and K.W. King. 2002. Soil and water assessment tool user's manual. TWRI Rep. TR-191. Texas Water Resources Inst., College Station, TX.

Rabalais, N.N., R.E. Turner, and W.J. Wiseman, Jr. 2001. Hypoxia in the Gulf of Mexico. J. Environ. Qual. 30:320–329.

Royer, T.V., M.B. David, and L.E. Gentry. 2006. Timing of riverine export of nitrate and phosphorus from agricultural watersheds in Illinois: Implications for reducing nutrient loading to the Mississippi River. Environ. Sci. Technol. 40:4126–4131.

Saleh, A., J.G. Arnold, P.W. Gassman, L.M. Hauck, W.D. Rosenthal, J.R. Williams, and A.M.S. McFarland. 2000. Application of SWAT for the upper north Bosque River watershed. Trans. ASAE 43: 1077–1087.

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

Singh, J., H.V. Knapp, J.G. Arnold, and M. Demissie. 2005. Hydrological modeling of the Iroquois River Watershed using HSPF and SWAT. J. Am. Water Resour. Assoc. 41:343–360.

Sogbedji, J.M., and G.F. McIsaac. 2002. Evaluation of the adapt model for simulating water outflow from agricultural watersheds with extensive tile drainage. Trans. ASAE 45:649–659.

Sophocleous, M.A., J.K. Koelliker, R.S. Govindaraju, T. Birdie, S.R. Ramireddygari, and S.P. Perkins. 1999. Integrated numerical modeling for basin-wide water management: The case of Rattlesnake Creek basin in south-central Kansas. J. Hydrol. 214: 179–196.

Spruill, C.A., S.R. Workman, and J.L. Taraba. 2000. Simulation of daily and monthly stream discharge from small watersheds using the SWAT model. Trans. ASAE 43:1431–1439.

USDA-NASS. 2003. Published estimates data base (PEDB) [Online]. Available at http://www.nass.usda.gov/ (accessed May 2003; verified 8 Feb. 2006).

USEPA. 2002. BASINS Database [Online]. Available at http://www.epa.gov/waterscience/ftp/basins/wdm_data/ (accessed October 2002; verified 8 Feb. 2007).

Vaché, K.B., J.E. Eilers, and M.V. Santelmann. 2002. Water quality modeling of alternative agricultural scenarios in the U. S. Corn Belt. J. Am. Water. Resour. Assoc. 38:773–787.