Watershed Modeling to Evaluate Water Quality at Intakes of Small Drinking Water Systems

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Executive Summary

Water quantity and quality at surface water supply intakes are of serious concern nationwide. Existing studies have focused on characterization and assessment of public water supplies. However, there exists no research on evaluation of water quantity and quality at surface water supply intakes, and development of comprehensive watershed modeling tools to do so. Furthermore, no existing model is capable of comprehensively simulating all of the hydrologic, upland soil and streambank erosion, sediment transport, and fate and transport of nutrient and pesticide processes necessary to comprehensively assess the water quantity and quality problems and help make the best management decisions to eliminate or minimize those problems.

The current study addresses these issues extensively and, being a premier study, major efforts were devoted to the fundamental issues and goals, e.g., review and selection of the most suitable long-term continuous watershed simulation model; investigation of its theoretical bases, formulations, and major weaknesses; setting up the model, learning its procedures, and making it operational (normally a minimum of one-year process); and selecting a complementary and compatible storm-event watershed model to enhance storm-event simulations of the former, selecting a watershed in Illinois suitable for the study, applications (calibration and validation) of both the models to the selected watershed based on available limited data, and assessments of long-term water quantity and quality at intakes of all the small public water supply systems within the watershed, as an example, based on calibration and validation of the continuous model.

Recent review of leading watershed-scale hydrologic and nonpoint-source pollution models indicated various strengths and weaknesses in the models. It found that the most promising long-term continuous simulation model was the Soil and Water Assessment Tool (SWAT). This is the most established and commonly used model, and an excellent candidate for enhancement into a more comprehensive model with storm-event simulations. In this study, a compatible and complementary storm-event hydrologic model was selected to enhance SWAT's storm-event hydrologic simulations and tested on the selected watershed along with SWAT's long-term continuous simulations of hydrology, sediment, and nutrients.

The 8,400 square kilometer (km^2) Little Wabash River watershed in southeastern Illinois was chosen because its water supply and watershed attributes are favorable for this study. It has seven small (population < 10,000) and three large public surface water supply systems serving communities in it. Every major stream and river mile of the watershed has impairment from sediment, nutrient enrichment, and pesticides. The developmental history of the watershed shows that watershed growth was retarded by the low level of water resources development resulting in a very rural and sparsely populated agricultural watershed, the least developed major watershed in Illinois.

The SWAT was run on the Little Wabash River watershed using Geographic Information System data on topography, soil, and land use retrieved from links provided at the U.S. Environmental Protection Agency's Better Assessment of Science Integrating Point and Nonpoint Sources database and daily precipitation and air temperature data at 14 precipitation gages obtained from the National Climatic Data Center website. The

model was calibrated and validated using daily flow records at four gaging stations – Effingham (620 km^2), Wayne City (1,200 km^2), Clay City (2,930 km^2), and Carmi (8,000 km^2) – obtained from the U.S. Geological Survey (USGS) website, and water quality measurements made by the Illinois Environmental Protection Agency in cooperation with the Illinois Department of Natural Resources at these stations. The storm-event hydrology model was calibrated and validated only for the uppermost portion of the watershed draining to Effingham using storm event rainfall data at two of the gages located there and storm-event flow records at Effingham obtained from the USGS.

The long-term continuous hydrologic simulations were evaluated by comparing simulated monthly average flows with monthly observed flows at the four evenly distributed gaging stations. Coefficient of determination and Nash-Sutcliffe coefficient for the individual years, cumulatively for the 5-year calibration period (1995-1999), and cumulatively for the entire 8-year simulation period (1995-2002) were computed. These statistical values for three of the four stations – Effingham, Clay City, and Carmi – were above or near 0.5 and, therefore, had reasonable overall predictions of monthly flows. Although the overall statistical values for Wayne City were low (0.26-0.37), values for 5 out of the 8 individual years were above or near 0.5, with 0.91 as the highest value. Therefore, overall model performance at Wayne City in simulating monthly flows can also be considered reasonable. To our knowledge, such spatially and temporally distributed statistical evaluation of watershed model results is not available in the literature and, therefore, may have been done here for the first time. Visual comparisons of simulated and observed monthly flow hydrographs at the four stations also show that the model reasonably predicted monthly average flows throughout the watershed, although there were some discrepancies, especially with most of the peak flows (underpredictions). Therefore, the SWAT is a promising long-term continuous simulation model, which needs enhancements in storm-event simulations to improve its peak flow predictions.

Calibration of the water quality component of the SWAT was based on limited data, estimated from sporadic concentration measurements taken approximately once a month at only the four stations in the entire 8,400-km^2 watershed. Visual comparisons of simulated and observed (estimated) monthly sediment loads at each of the four monitoring stations during the 5-year period showed gross overpredictions. However, predictions of monthly total phosphorous (P), nitrate-nitrogen (N), and ammonia-N loads at the four stations were mixed and much better. Monthly total Kjeldahl N (TKN) load predictions at Carmi near the watershed outlet, the only station with TKN data, were found reasonable. Sampling of Ice Age near-stream lake-bottom sediments for the lower river stretches, ditches, and tile drains above Effingham, adjustment of model parameters, and inclusion of conservation structures throughout the watershed are necessary to improve predictions of these water quality parameters.

Using values of three hydrologic parameters from the SWAT calibration, the storm-event hydrologic simulations of three separate storms during the 8-year period resulted in comparable flow hydrographs with observed flows at Effingham, the uppermost station: peak flow errors of 1, 16, and 29 percent; volume errors of 5, 11, and 21 percent; and time-to-peak errors of 0, 3, and 3 percent. The SWAT daily flow simulations during the above three storms were mixed: underpredicted peak flows in two storms by 51 and 57 percent, but performed well for one storm for which peak flow was overpredicted by 6 percent. Comparisons of storm-event flow hydrographs at 15-minute intervals with SWAT daily flow hydrographs, along with their respective (15-minute and daily) observed hydrographs, revealed that storm-event hydrologic simulations predicted more accurate flows, especially high and peak flows, during storm events than the SWAT daily continuous simulations.

Use of a smaller time step and a unique combination of the runoff curve number method for rainfall excess computations and kinematic wave equations for flow routing and physical bases of these routines with the convenience of only three parameters (curve number or *CN*, Manning's roughness coefficient or *n*, and effective lateral saturated hydraulic conductivity or *ELSHC*) to calibrate are responsible for predicting more accurate high and peak flows during intense storms. Addition of these routines to SWAT would be a significant enhancement. Furthermore, recalibration of the three parameters for the storm event model was not necessary. Calibration of these parameters in SWAT was sufficient. Therefore, these parameters are interchangeable, and the models are compatible and complementary, which is unique.

For water quantity and quality evaluations at intakes of the seven small water supplies in the Little Wabash River watershed, yearly averages of precipitation, runoff, and loads of sediment, total P, nitrate-N, ammonia-N, and TKN at each intake under existing conditions were examined from model results. The calibrated and validated model can be used to evaluate impacts of changes in management practices towards improving or maintaining water quantities and qualities. In the present study, however, calibration of the water quality model was based on limited data as the water supplies no longer analyze water at their intakes. Uncertainty of the water quality results was a serious concern and, therefore, no attempt was made to evaluate any management practices. Once adequate water quality sampling resumed or equivalent monitoring was established, adequate calibration of the model and evaluation of management practices could be conducted.

To enhance the SWAT with National Oceanic and Atmospheric Administration (NOAA) high resolution (4 kilometer or km grid) daily radar precipitation data, radar and gage precipitation data at 7,235 stations around the continental United States were compared. Unfortunately, the comparisons showed wide scatter of the data, and radar precipitation consistently was underestimated with respect to the gage measurements. More research and efforts are necessary to improve radar estimates of precipitation and develop an interface to incorporate those high-resolution precipitation data or mesoscale Regional Climate Model simulations into the SWAT for more accurate simulation of intense storms, which generate and transport disproportionately large amounts of pollutants. Finer spatial resolution of the watershed also will be necessary for more realistic simulations, as well as capture of accurate point sources, sinks, and storm-event runoff, sediment, and contaminants.

Research needs to continue towards SWAT enhancements with storm-event soil erosion, streambed and bank erosion, and transport of sediment, nutrient, and pesticide simulations; ultimately developing a more comprehensive watershed simulation model. Also, uncertainties resulting from deterministic modeling of natural processes and measurements or observations of data used in modeling must be considered when using model results in management decisions and policymaking, the subject of future research.

The SWAT, a well-documented and user-friendly tool, is available on the USEPA website: http://www.epa.gov/waterscience/basins/bsnsdocs.html#swat. The storm event hydrologic model uses interchangeable parameters from the SWAT and watershed characteristics data are processed directly from SWAT data and, therefore, is also user friendly and can be developed into an even more user-friendly tool in future research. Training for use of these modeling tools by scientific or engineering consultants working for the small public water supply systems could be part of continued Extension work.

Dissemination of results of this study included presentations at two international and three regional conferences and two presentations to the officials of the Illinois Environmental Protection Agency: Division of Public Water Supplies and Bureau of Water. A paper from one of those conferences was published in its proceedings. One manuscript is currently under review for publication by a scientific journal.

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Introduction

Water quantity and quality at surface water supply intakes are of serious concern nationwide. Land degradation is widespread and many of the nation's surface waters are categorized as impaired (303d lists) by nonpoint pollution (USEPA, 2006). In the Mississippi River watershed, nonpoint-source pollution from agriculture is the most widespread source of impairment. Agriculture in Illinois has been identified as a particularly severe offender (Doering et al., 1999). Many Midwestern U.S. streams and rivers draining agricultural watersheds have elevated concentrations of nitrate-nitrogen (NO_3-N) and also drain into the Gulf of Mexico (Smith et al., 1993; Goolsby et al., 1999). In Illinois, some drinking water supplies suffer from high concentrations of NO_3-N and other agriculturally related chemicals that exceed health standard as well as from high concentrations of suspended sediment (e.g., Keefer et al., 1996; Mitchell et al., 2000; Borah et al., 2003). The sediment problem is so severe that it also seriously reduces water supply capacities of Illinois' lakes and reservoirs (e.g, Fitzpatrick et al., 1985, 1987).

Various studies have focused on characterization and assessments of public water supplies, e.g., Warner (2000), Eimers et al. (2000), and Delaware Division of Water Resources (2002), and numerous others on characterization of water quantities and qualities in watersheds through field monitoring at specific locations (e.g., Robertson and Roerish, 1999; Arheimer and Liden, 2000; Mitchell et al., 2000; Stone et al., 2000; and Borah et al., 2003). Many modeling studies focus primarily on calibration and validation of models on monitored watersheds, a few evaluate management strategies, as reviewed in Borah and Bera (2003, 2004). However, there exists no research on evaluation of water quantity and quality at surface water supply intakes and development of comprehensive watershed modeling tools to do so. No existing model is capable of comprehensively simulating all of the hydrologic, upland soil and stream bank erosion, sediment transport, and fate and transport of nutrient and pesticide processes necessary to assess water quantity and quality problems comprehensively and help make the best management decisions to eliminate or minimize these problems (Borah and Bera, 2003, 2004).

Small public surface water supply systems of the Midwest require watershed-scale hydrologic and nonpoint-source pollution models to support their water-supply needs. Such models can be enhanced and developed into useful watershed management tools for assessment of distributed water quantities and qualities, and evaluations of best management practices (BMPs) for protecting and improving water quantity and quality as well as the landscape of the watershed.

Thus, the overall goal of this project is to develop models as source-water protection assessment tools for operators of small Midwestern surface water supply systems. To meet this goal, the following specific objectives (goals) were proposed:

1. Find a suitable system of watershed modeling tools for evaluation of water quantity and quality at surface water sources of small drinking water systems, which also could be used for large systems, evaluate their potentials and weaknesses, and identify their potential enhancements.
2. Select a watershed most appropriate to the goal.

3. Test the modeling system on the selected watershed while assessing and evaluating water quantity and quality at surface water intakes of the small public water supply systems within the watershed.
4. Use the modeling system to evaluate watershed management and BMPs in protecting or improving water quantity and quality at the intakes.
5. Enable the modeling system simulations to incorporate digital input at high temporal- and spatial-resolution multi-sensor precipitation data (1 hour or hr, 4x4 square kilometer or km^2) of the National Centers for Environmental Predictions (NCEP) Environmental Modeling Center (EMC).
6. Enable the modeling system to input and simulate detailed point-source and sink water quantity and quality data, critical to small public water supply systems, using finer spatial resolution of the watershed.
7. Develop the modeling system to be a more user-friendly tool for small public water supply managers.

Extensive investigations were conducted to achieve these goals. For this premier study, major efforts were devoted to the fundamental issues and goals, e.g., reviewing and selecting the most suitable long-term continuous watershed simulation model; investigating its theoretical bases, formulations, and major weaknesses; setting up the model, learning its procedures, and making it operational (normally a minimum of one-year process); and selecting a complementary and compatible storm-event watershed model to enhance storm-event simulations of the former, selecting a watershed in Illinois suitable for the study, applications (calibration and validation) of both the models to the selected watershed based on available limited data, and assessments of long-term water quantity and quality at intakes of all the small public water supply systems within the watershed, as an example, based on the calibration and validation of the continuous model.

Recent review (Borah and Bera, 2003, 2004) of leading watershed-scale hydrologic and nonpoint-source pollution models indicated various strengths and weaknesses in the models. The Soil and Water Assessment Tool or SWAT (Arnold et al., 1998) was the most promising long-term continuous simulation model. SWAT, the most comprehensive and commonly used model, is an excellent candidate for enhancement as a more comprehensive model with storm-event simulations.

In this study, a compatible and complementary storm-event hydrologic model was selected to enhance SWAT storm-event hydrologic simulations and test on the 8,400 km^2 Little Wabash River watershed in southeastern Illinois along with SWAT long-term continuous simulations of hydrology, sediment, and nutrients. The Little Wabash River watershed in southeastern Illinois was chosen because its water supply and watershed attributes are favorable for this study. It has seven small (population < 10,000) and three large public surface water supply systems that serve communities. Every major stream and river mile of the watershed has impairment from sediment, nutrient enrichment, and pesticides. The developmental history of the watershed shows that watershed growth was retarded by the low level of water resources development. As a result, this became very rural and sparsely populated agricultural watershed and the least developed major watershed in Illinois.

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Finer spatial resolution of the watershed also will be necessary for more realistic simulations, as well as accurate capture of point sources, sinks, storm-event runoff, sediment, and contaminants.

The report describes the above operations and presents the results. The next section *Watershed Models* is devoted to review and comparisons of watershed models, selection of the SWAT model, and identification of SWAT weaknesses and potential enhancements. The section *The SWAT Model* describes the theoretical background and formulations in detail. The section *SWAT Storm-Event Enhancements* describes detailed theoretical background and formulations of a complementary and compatible storm event hydrology model selected for SWAT enhancement. The section *Little Wabash River Watershed* describes watershed characteristics, water supply systems, historical background, and suitability for this study. The section *Modeling the Little Wabash River Watershed* describes calibration and validation of the SWAT model for long-term continuous simulations of stream flows, sediment, and nutrients; presents these results along with water quantity and quality assessments at intakes of seven small public water supply systems within the watershed; and presents validation results of the storm-event hydrologic model. This section also includes discussions on evaluations of watershed management and BMPs, incorporation of high-resolution precipitation data into the model, using finer spatial resolution of watershed, and a user-friendly tool for small water supply managers. Finally, the *Conclusions* summarize the study, draw conclusions, and list future study recommendations.

Parts of the work were presented and disseminated at two international and three regional conferences (Borah et al., 2004b, 2004c, 2005, 2006a, 2006c). A paper from one of those conferences was published in its proceedings (Borah et al., 2005). One manuscript is currently under review for publication by a scientific journal (Borah et al., 2006b). Dissemination also included two presentations to officials of the IEPA: Division of Public Water supplies on August 25, 2005 and Bureau of Water on September 29, 2005.

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Watershed Models

Numerous watershed models are available today; most simulate hydrologic processes with or without nonpoint-source pollution, with a few having economic components (Singh, 1995; Singh and Frevert, 2002a, 2002b, 2006). The models have varying capabilities, strengths, and weaknesses; and none is capable of comprehensively simulating all hydrologic, upland soil and streambank erosion, sediment transport, and fate and transport of nutrient and pesticide processes necessary to assess water quantity and quality problems comprehensively and help make the best management decisions to eliminate or minimize these problems (Borah and Bera, 2003, 2004).

Model Reviews and Comparisons

Comprehensive review and comparison of 11 leading watershed-scale hydrologic and nonpoint-source pollution models was conducted (Borah and Bera, 2003). The models reviewed were Agricultural NonPoint Source pollution model or AGNPS (Young et al., 1987), Annualized AGNPS or AnnAGNPS (Bingner and Theurer, 2001), Areal Nonpoint Source Watershed Environment Response Simulation or ANSWERS (Beasley et al., 1980), ANSWERS-Continuous (Bouraoui et al., 2002), CASCade of planes in 2-Dimensions or CASC2D (Ogden and Julien, 2002), Dynamic Watershed Simulation Model or DWSM (Borah et al., 2002a), Hydrologic Simulation Program – Fortran or HSPF (Bicknell et al., 1993), KINematic runoff and EROSion model or KINEROS (Woolhiser et al., 1990), the European Hydrological System model or MIKE SHE (Refsgaard and Storm, 1995), Precipitation-Runoff Modeling System or PRMS (Leavesley et al., 1983), and Soil and Water Assessment Tool or SWAT (Arnold et al., 1998). The mathematical bases (the basic and most critical model elements) of these models were identified and compiled; and their flow-governing equations and solution methods were discussed. The comparison provides an objective guideline in determining the problems, situations, or conditions for which individual models are most suitable, the accuracies and uncertainties expected, their full potential uses and limitations, and directions for their enhancements or new developments.

The review found that AGNPS, AnnAGNPS, DWSM, HSPF, MIKE SHE, and SWAT were more fully developed than the other models having all the three major components: hydrology, sediment, and chemical. The AnnAGNPS, HSPF, and SWAT are mainly long-term continuous simulation models useful for analyzing long-term impacts of hydrological changes and watershed management practices. Both AGNPS and DWSM are storm-event simulation models useful for analyzing watershed responses from severe or extreme storm events and evaluating watershed management practices. The most physically based model, MIKE SHE, has both long-term continuous and storm-event simulation capabilities.

Among the long-term continuous simulation models, SWAT is a promising model for agricultural watersheds, and the HSPF is promising for mixed agricultural and urban watersheds. The AnnAGNPS model is similar to SWAT, and MIKE SHE uses intensive data and computations for efficient simulation of large watersheds (Borah and Bera, 2003).

Among the fully developed storm-event models, the simple and lumped AGNPS model generates overall responses from a storm, including surface water volume, peak flow, and yields or average concentrations of sediment and nutrients. It does not generate time-varying flows (hydrographs) and constituent discharges, which are critical in certain analyses. For example, peak flows, peak constituent concentrations, and their timing are crucial information in flood warning and management, watershed assessment, and BMP evaluations. On the other hand, MIKE SHE is too complicated and data and computation intensive. The DWSM provides a balance and compromise between the simple AGNPS and the complicated MIKE SHE storm-event models because of its physically based robust routines (Borah and Bera, 2003).

Reviews of applications of the SWAT, HSPF, and DWSM (Borah and Bera, 2004) showed that SWAT and HSPF were reliable for yearly and monthly (average or yield) predictions, except for the months having severe hydrologic conditions (storms). Daily predictions from SWAT and HSPF were less reliable, especially for the days with intense storms, and thus are not suitable for analyzing severe storm events. On the other hand, the DWSM, a storm-event model, performed well in simulations during storm events, including intense storms.

Model Selection and Potential Enhancements

These comparative studies suggested that research must continue to combine strengths of different models for developing more comprehensive models and enhancing their predictive capabilities. The SWAT was recognized as the most promising watershed model for enhancement as a more comprehensive tool. It was developed at the U.S. Department of Agriculture (USDA)-Agricultural Research Service (ARS) Grassland, Soil, and Water Research Laboratory in Temple, Texas (Arnold et al., 1998; Neitsch et al., 2002). This well-established model has GIS and graphical user interfaces, and is a part of the USEPA’s BASINS modeling system (USEPA, 2001; DiLuzio et al., 2002). As summarized in Borah and Bera (2004), and as may be found much more in recent literature (e.g., Sohrabi et al., 2003; Van Liew et al., 2003, 2005; Saleh and Du, 2004; Abbaspour and Srinivasan, 2005; Du et al., 2005; Vazquez-Amabile and Engel, 2005; Wang and Melesse, 2005; Wurbs et al., 2005), SWAT has been used nationally and internationally in watershed studies and management planning. Comparative investigations (Van Liew et al., 2003; Saleh and Du, 2004) of the SWAT with the HSPF, another widely used model from the BASINS (USEPA, 2001), found that the SWAT exhibited more robustness and proved to be a better predictor than the HSPF.

Based on these reviews, the SWAT was selected for long-term continuous simulations of the Little Wabash River watershed and for development into a more comprehensive watershed simulation model. The major shortcoming of the SWAT is that it is mostly a daily time-step model and is not formulated to simulate storm events (Arnold et al., 1998). Although a storm-event simulation option was added to the SWAT with sub-daily time steps (King et al., 1999) using the Green and Ampt (1911) infiltration equation, that option has not been widely used and one of the reasons could be its requirement of intensive soil-test data for parameterization.

Resolution of precipitation data input is becoming another concern in watershed modeling. Precipitation is a major driving force of watershed processes and is highly variable on spatial and temporal scales. Current watershed models, including the SWAT, are unable to capture distributions accurately because of limited resolution of precipitation data inputs from discrete, and usually sparse, rain gauges. On the other hand, recent availability of the NOAA multi-sensor (radar plus gauge) hourly, 4-km grid precipitation analysis and progress in mesoscale regional climate model (RCM) simulations (Liang et al., 2004a, 2004b) bring an unprecedented opportunity for substantially enhancing temporal and spatial resolutions of watershed model precipitation input.

Therefore, the principal enhancements envisioned for the development of a comprehensive SWAT are addition of storm-event simulations; addition of streambank erosion simulations, and addition of an interface to incorporate high-resolution precipitation data, such as the recently available NOAA near real-time multi-sensor (radar plus gauge) data at hourly, 4-km grid and mesoscale RCM simulations (Liang et al., 2004a, 2004b). These enhancements are necessary to simulate intense storm events – because such events generate and transport disproportionately large amounts of sediment and chemicals.

The SWAT Model

The SWAT is a watershed model for continuous simulations of hydrology and water quality in small to large watersheds. It operates on a daily time step and is designed to assess the impact of different management practices on water, sediment, and agricultural chemical yields. The model emerged mainly from the Simulator for Water Resources in Rural Basins or SWRRB (Arnold et al., 1990), and features from the Chemical, Runoff, and Erosion from Agricultural Management Systems or CREAMS (Knisel, 1980), the Groundwater Loading Effects on Agricultural Management Systems or GLEAMS (Leonard et al., 1987), the Erosion-Productivity Impact Calculator or EPIC (Williams et al., 1984), and the Routing Outputs to Outlets or ROTO (Arnold et al., 1995).

A watershed is divided into multiple subwatersheds, which are further subdivided into unique soil and land use characteristics called hydrologic response units (HRUs). The water balance of each HRU is represented by four storage volumes: snow, soil profile, shallow aquifer, and deep aquifer. Flow generation, sediment yield, and pollutant loadings are summed across all HRUs in a subwatershed, and resulting loads then are routed through channels, ponds, and/or reservoirs to the watershed outlet.

The SWAT has eight major components: hydrology, weather, sedimentation, soil temperature, plant growth, nutrients, pesticides, and land management. These components are described by the model developers in various publications and most recently in Neitsch et al. (2002) in great detail and with extensive background. A complete, streamlined presentation is necessary to provide a clear understanding of the scientific bases and, therefore, such a presentation of the hydrologic and sediment formulations is attempted here with consistent mathematical symbols. A similar level of presentation of the nutrient, pesticide, and management practices could not be completed due to time and budgetary constraints. However, all those components were used in this study.

Hydrology

The daily water budget in each HRU is computed as:

$$SW_t = SW_0 + \sum_{i=1}^{t} (R_i - Q_{surf,i} - Q_{et,i} - Q_{perc,i} - Q_{gwrf,i}) \tag{1}$$

where SW_t = final soil water content (mm), t = time (days), SW_0 = initial soil-water content (mm), R_i = precipitation on day i (mm), $Q_{surf,i}$ = surface runoff on day i (mm), $Q_{et,i}$ = evapotranspiration (ET) on day i (mm), $Q_{perc,i}$ = percolation on day i (mm), and $Q_{gwrf,i}$ = groundwater return flow, or base flow, on day i (mm).

Surface Runoff

Daily surface runoff is computed from daily rainfall using the Soil conservation Service (SCS, 1972) runoff curve number procedure, where the runoff volume is expressed as:

$$Q_{surf,i} = \frac{(R_i - 0.2S_r)^2}{R_i + 0.8S_r} \tag{2}$$

where S_r = retention parameter (mm), which depends upon soil cover complexes, including soil, land use, management, and initial soil moisture (antecedent conditions), and is expressed as:

$$S_r = \frac{25400}{CN} - 254 \tag{3}$$

where CN = the runoff curve number that indicates runoff potential, and its values are given by SCS (1972, 1986) for different soil-cover complexes. Practical CN values given by SCS range from 30 to 95; however, potential values may range from 1 to 100.

The amount of surface runoff from a subbasin reaching the main channel is computed as:

$$Q_{ch,i} = \left(Q_{surf,i} + Q_{stor,i-1}\right) \cdot \left[1 - e^{\frac{-surlag}{t_{conc}}}\right] \tag{4}$$

where $Q_{ch,i}$ = amount of surface runoff discharged into the main channel on day i (mm), $Q_{stor,i-1}$ = amount of surface runoff stored or lagged from day $i-1$ (mm) and is equal to ($Q_{stor,i-2} + Q_{surf,i-1} - Q_{ch,i-1}$), $surlag$ = surface runoff lag coefficient, and t_{conc} = time of concentration for the subbasin (hour or hr or h).

Evapotranspiration

Evapotranspiration (ET) includes all processes by which water on the earth's surface is converted to water vapor: evaporation from the plant canopy, transpiration, sublimation, and evaporation from the soil. The SWAT estimates ET ($Q_{et,i}$) from potential evapotranspiration (PET), which is calculated using three alternative methods: Hargreaves (Hargreaves and Samani, 1985), Priestley-Taylor (Priestley and Taylor, 1972), and Penman-Monteith (Monteith, 1965). The Priestley-Taylor method was used in this study.

Percolation

Percolation is calculated for each soil layer in the profile when water content exceeds field capacity. The amount of water that moves from one layer to the underlying layer is calculated using storage routing methodology and is expressed as:

$$Q_{perc,i} = SW_{excess,i} \cdot \left[1 - e^{\frac{-\Delta t}{TT_{perc}}}\right] \tag{5}$$

where $SW_{excess,i}$ = drainable volume of water in the soil layer on day i (mm), Δt = time interval (24 h), and TT_{perc} = percolation travel time for the soil layer (h). The percolation travel time for a soil layer is calculated as:

$$TT_{perc} = \frac{SW_{sat} - SW_{cap}}{K_{sat}} \tag{6}$$

where SW_{sat} = amount of water in the soil layer when completely saturated (mm), SW_{cap} = water content of the soil layer at field capacity (mm), and K_{sat} = saturated hydraulic conductivity for the layer (mm·h^{-1}).

Lateral Subsurface Flow

The SWAT calculates lateral subsurface flow in the soil layer simultaneously with percolation using a kinematic storage model (Sloan et al., 1983), which is expressed as:

$$Q_{lat,i} = 0.024 \cdot \frac{2 \cdot SW_{excess,i} \cdot K_{sat} \cdot S_{hill}}{\phi_d \cdot L_{hill}} \tag{7}$$

where $Q_{lat,i}$ = lateral subsurface flow in the soil layer (water discharged from the hillslope outlet) on day i (mm), S_{hill} = slope of the hillslope segment (m·m^{-1}), φ_d = drainable porosity of the soil layer (mm·mm^{-1}), which is the difference between porosity of the layer on day i and its porosity at field capacity, and L_{hill} = hillslope length (m). Once lateral flow is calculated, the amount of lateral flow released to the main channel is calculated using a lag equation similar to Equation 4.

Tile Flow

Tile drainage occurs when the soil water content exceeds field capacity. In the soil layer where the tile drains are installed, the amount of water entering the drain on a given day is calculated as:

$$Q_{tile,i} = \left(SW_t - SW_{cap}\right) \cdot \left[1 - e^{\frac{-24}{t_{drain}}}\right] \tag{8}$$

where $Q_{tile,i}$ = amount of water removed from the layer by tile on day i (mm) and t_{drain} = time required to drain the soil to field capacity (h). Water entering tiles is treated as lateral flow, which lags while discharging into the main channel and is treated similar to Equation 4.

Groundwater Flow

Groundwater return flow is derived from a water balance equation for the shallow aquifer, which is:

$$aq_{sh,i} = aq_{sh,i-1} + Q_{rchrg,i} - Q_{gwrf,i} - Q_{revap,i} - Q_{deep,i} - Q_{pump,i} \tag{9}$$

where $aq_{sh,i}$ = amount of water stored in the shallow aquifer on day i (mm), $Q_{rchrg,i}$ = amount of recharge entering the aquifer on day i (mm), $Q_{revap,i}$ = amount of water moving into the soil zone in response to water deficiencies on day i (mm), $Q_{deep,i}$ = amount of water percolating from shallow aquifer into the deep aquifer on day i (mm), and $Q_{pump,i}$ = amount of water removed from the shallow aquifer by pumping on day i (mm). The recharge to the aquifer on a given day is calculated as:

$$Q_{rchrg,i} = Q_{perc,i} \cdot \left[1 - e^{\frac{-1}{\delta_{gw}}}\right] + Q_{rchrg,i-1} \cdot e^{\frac{-1}{\delta_{gw}}} \tag{10}$$

where δ_{gw} = delay time or drainage time of the overlying geologic formations (day or d). Groundwater return or base flow is computed as:

$$Q_{gwrf,i} = Q_{rchrg,i} \cdot \left[1 - e^{-\alpha_{gw}\Delta t}\right] + Q_{gwrf,i-1} \cdot e^{-\alpha_{gw}\Delta t} \tag{11}$$

where α_{gw} = groundwater or base flow recession constant, and Δt = time step (1 d).

Channel Flow Routing

All the water ($Q_{ch,i}$, $Q_{lat,i}$, $Q_{tile,i}$, and $Q_{gwrf,i}$) reaching the main channels are routed through the channel network of the watershed using a variable storage coefficient method (Williams, 1969) or the Muskingum routing method (Linsley et al., 1958). Flow is computed assuming a trapezoidal shape of the channel having 0.5 side slope and using Manning's equation (Chow, 1959) with adjustments for transmission losses, evaporation, diversions, and return flow (Arnold et al., 1995), which accounts for reduction of runoff volume as water moves downstream through the channel network. Channel overbank flows are simulated over the floodplains similar to the channels and assuming width of each floodplain bottom to be five times the channel bankfull width and side slope of 0.25.

Flow Routing through Impoundment

Water is routed through four types of impoundments: depressions/potholes, ponds, wetlands, and reservoirs/lakes. Depressions/potholes, ponds, and wetlands are located within a subbasin receiving water only from the, or, part of the subbasin, whereas, reservoirs/lakes are located on the main channel network receiving water from all the upstream subbasins. Similar routing procedures are used for the impoundments based on water balance and user-provided measured or targeted outflows.

Erosion and Sedimentation

Sediment Yield

Sediment yield is computed using the Modified Universal Soil Loss Equation or MUSLE (Williams and Berndt, 1977), a modified equation of the Universal Soil Loss Equation or USLE (Wischmeier and Smith, 1978), where the rainfall energy factor was replaced with runoff factors to eliminate the need for delivery ratios. The MUSLE is expressed as:

$$Y_{sed,i} = 11.8 \cdot \left(Q_{surf,i} \cdot q_{peak,i} \cdot A_{hru}\right)^{0.56} \cdot K_{USLE} \cdot C_{USLE} \cdot P_{USLE} \cdot LS_{USLE} \cdot CF \tag{12}$$

where $Y_{sed,i}$ = sediment yield on day *i* (metric ton), $q_{peak,i}$ = peak runoff rate on day *i* (m^3/s), A_{hru} = area of the HRU (ha), K_{USLE} = USLE soil erodibility factor [0.013 metric ton m^2 hr/(m^3-metric ton cm)], C_{USLE} = USLE cover and management factor, P_{USLE} = USLE support practice factor, LS_{USLE} = USLE topographic factor, and CF = coarse fragment factor.

Peak Runoff Rate

The peak runoff rate is computed using the modified rational formula, expressed as:

$$q_{peak,i} = \frac{\alpha_{tc} \cdot Q_{surf,i} \cdot A_{sub}}{3.6 \cdot t_{conc}} \tag{13}$$

where α_{tc} = fraction of daily rainfall that occurs during the time of concentration, A_{sub} = subbasin area (km^2), and t_{conc} = time of concentration for the subbasin (hr).

$$\alpha_{tc} = 1 - \exp\left[2 \cdot t_{conc} \cdot \ln\left(1 - \alpha_{0.5}\right)\right] \tag{14}$$

where $\alpha_{0.5}$ = fraction of daily rain falling in the half-hour highest intensity rainfall.

$$t_{conc} = \frac{L_{sub}^{\ 0.6} \cdot n_{sub}^{\ 0.6}}{18 \cdot S_{sub}^{\ 0.3}} + \frac{0.62 \cdot L_{ch} \cdot n_{ch}^{\ 0.75}}{A_{sub}^{\ 0.125} \cdot S_{ch}^{\ 0.375}} \tag{15}$$

where L_{sub} = subbasin slope length (m), n_{sub} = Manning's roughness coefficient for the subbasin, S_{sub} = average slope in the subbasin (m m^{-1}), L_{ch} = channel length from the most distant point to the subbasin outlet (km), n_{ch} = Manning's roughness coefficient for the channel, and S_{ch} = channel slope (m m^{-1}). The right hand first and second terms of Equation 15 represents time of concentrations for overland flow and channel flow, respectively.

Soil Erodibility Factor

When the silt and very fine sand content make up less than 70 percent of the soil particle size distribution, the soil erodibility factor can be computed using the following equation developed by Wischmeier et al. (1971):

$$K_{USLE} = \frac{0.00021 \cdot M^{1.14} \cdot \left(12 - OM\right) + 3.25 \cdot \left(s_{str} - 2\right) + 2.5 \cdot \left(s_{perm} - 3\right)}{100} \tag{16}$$

where M = particle size parameter, OM = percent organic matter, s_{str} = soil structure class, and s_{str} = soil profile permeability class.

$$M = \left(m_{silt} + m_{vfs}\right) \cdot \left(100 - m_c\right) \tag{17}$$

where m_{silt} = percent silt content (0.002-0.05 mm), m_{vfs} = percent very fine sand content (0.05-0.10 mm), and m_c = percent clay content (<0.002 mm).

The OM term is equivalent to 1.72 times of percent organic carbon content of the soil layer. The s_{str} term is 1 for very fine granular, 2 for fine granular, 3 for medium or coarse granular, and 4 for blocky, platy, prismlike, or massive structure. The s_{perm} term is 1 for rapid (>150 mm/hr), 2 for moderate to rapid (50-150 mm/hr), 3 for moderate (15-50 mm/hr), 4 for slow to moderate (5-15 mm/hr), 5 for slow (1-5 mm/hr), and 6 for very slow (<1 mm/hr) saturated hydraulic conductivities of the soil profile.

The soil erodibility factor can also be computed by using an alternative equation proposed by Williams (1995):

$$K_{USLE} = f_{csand} \cdot f_{cl-si} \cdot f_{orgc} \cdot f_{hisand} \tag{18}$$

where f_{csand} = coarse-sand factor (low for high coarse-sand content and high for soils with little sand), f_{cl-si} = clay-silt factor (low for soils with high clay to silt ratios), f_{orgc} = organic carbon factor (low for soils with high organic carbon content), and f_{hisand} = high sand factor (low for soils with extremely high sand contents). The factors are calculated as:

$$f_{csand} = \left\{0.2 + 0.3 \cdot \exp\left[-0.256 \cdot m_s \cdot \left(1 - \frac{m_{silt}}{100}\right)\right]\right\} \quad (19)$$

$$f_{cl-si} = \left(\frac{m_{silt}}{m_c + m_{silt}}\right)^{0.3} \quad (20)$$

$$f_{orgc} = \left[1 - \frac{0.25 \cdot c_{org}}{c_{org} + \exp(3.72 - 2.95 \cdot c_{org})}\right] \quad (21)$$

$$f_{hisand} = \left\{1 - \frac{0.7 \cdot \left(1 - \frac{m_s}{100}\right)}{\left(1 - \frac{m_s}{100}\right) + \exp\left[-5.51 + 22.9 \cdot \left(1 - \frac{m_s}{100}\right)\right]}\right\} \quad (22)$$

where m_s = percent sand content (0.05-2.00 mm), and c_{org} = percent organic carbon content of the soil layer.

Cover and Management Factor

Because plant cover varies during the growth cycle of the plant, the cover and management factor (C_{USLE}) given by Wischmeier and Smith (1978) is updated daily using the equation:

$$C_{USLE} = \exp\{[\ln(0.8) - \ln(C_{USLE,mn})] \cdot \exp(-0.00115 \cdot rsd_{surf}) + \ln(C_{USLE,mn})\} \quad (23)$$

where $C_{USLE,mn}$ = minimum value for the cover and management factor for the land cover, and rsd_{surf} = amount of residue on the soil surface (kg/ha).

$$C_{USLE,mn} = 1.463 \cdot \ln(C_{USLE,aa}) + 0.1034 \quad (24)$$

where $C_{USLE,aa}$ = average annual C factor for the land cover given by Wischmeier and Smith (1978).

Support Practice Factor

Support practices include contour tillage, stripcropping on the contour, and terrace systems. Stabilized waterways for the disposal of excess rainfall are a necessary part of each of these practices. Numerical values of P_{USLE} for these practices are given in Wischmeier and Smith (1978) and reiterated by Neitsch et al. (2002) as used in the SWAT.

The P_{USLE} values for contour farming are given for seven ranges of slopes from 1 percent to 25 percent along with maximum valid slope lengths for each of the slope ranges. The P_{USLE} values for contour stripcropping also are given for seven ranges of slopes from 1 percent to 25 percent along with maximum valid slope lengths and strip widths for each of the slope ranges. In this case, three sets of P_{USLE} values are given. The first set is for 4-year rotation of row crop, small grain with meadow seeding, and 2 years of meadow. A second row crop can replace the small grain if meadow is established in it. The second set is for 4-year rotation of 2 years of row crop, winter grain with meadow seeding, and 1-year meadow. The third set is for alternate strips of row crop and winter grain.

The P_{USLE} values for contour farming terraced fields are given for six ranges of slopes from 1 percent to 25 percent. Slope length is the horizontal terrace interval. The values are for contour farming and, therefore, no additional contouring factor is needed. Four sets of P_{USLE} values are given. The first two sets are the same as the contour farming and contour stripcropping mentioned above and must be used for interterrace sediment yields. The third and fourth sets include entrapment efficiency of the terrace system and can be used to compute sediment yield from a terraced field. The third set is for terrace system with graded channels sod outlets and the fourth set for steep back slope underground outlets.

Topographic Factor

The topographic factor is calculated as:

$$LS_{USLE} = \left(\frac{L_{hill}}{22.1}\right)^{m} \cdot \left[65.41 \cdot \sin^{2}\left(\alpha_{hill}\right) + 4.56 \cdot \sin\alpha_{hill} + 0.065\right] \tag{25}$$

where m = exponent, and α_{hill} = angle of the slope.

$$m = 0.6 \cdot \left[1 - \exp\left(-35.835 \cdot S_{hru}\right)\right] \tag{26}$$

where S_{hru} = slope of the HRU (m m^{-1}) and is equal to tan α_{hill}.

Coarse Fragment Factor

The coarse fragment factor (CF) is calculated as:

$$CF = \exp\left(-0.053 \cdot rock\right) \tag{27}$$

where $rock$ = percent rock in the first soil layer.

Snow Cover Effects

During periods when snow is present in an HRU, the SWAT modifies the sediment yield using the following relationship:

$$Y_{sed,i} = \frac{Y'_{sed,i}}{\exp\left[\frac{3 \cdot sno}{25.4}\right]} \tag{28}$$

where $Y_{sed,i}$ = sediment yield on day i (metric ton), $Y'_{sed,i}$ = sediment yield on day i calculated with MUSLE (metric ton), and sno = water content of the snow cover (mm).

Sediment Lag in Surface Runoff

Similar to surface runoff lag (Equation 4), the amount of sediment released to the main channel is calculated as:

$$Y_{ch,i} = \left(Y_{sed,i} + Y_{stor,i-1}\right) \cdot \left[1 - e^{\frac{-surlag}{t_{conc}}}\right] \tag{29}$$

where $Y_{ch,i}$ = amount of sediment discharged to the main channel on day i (metric ton), $Y_{stor,i}$ = sediment stored or lagged from day $i-1$ (metric ton), $surlag$ = surface runoff lag coefficient, and t_{conc} = time of concentration for the HRU (hr).

Sediment in Lateral and Groundwater Flows

The amount of sediment contributed by lateral and groundwater flows is calculated as:

$$Y_{lat,i} = \frac{\left(Q_{lat,i} + Q_{gwrf,i}\right) \cdot A_{hru} \cdot conc_{sed}}{1000} \tag{30}$$

where $Y_{lat,i}$ = sediment loading in lateral and groundwater flows (metric ton), A_{hru} = area of the HRU (km^2), and $conc_{sed}$ = concentration of sediment in lateral and groundwater flow (mg L^{-1}).

Sediment Routing in Channel

As described in Arnold et al. (1995), sediment routing in the channel is based on Williams' (1980) modification of Bagnold's (1977) stream power concept for bed degradation and sediment transport and particle fall velocity for sediment deposition. Bed degradation is adjusted with USLE soil erodibility and cover factors. These procedures were simplified and an assumption was made that the maximum amount of sediment that can be transported from a channel segment is a function of peak channel velocity and can be expressed as:

$$conc_{sed,ch,mx} = c_{sp} \cdot \left(v_{ch,pk}\right)^{sp\exp} \tag{31}$$

where $conc_{sed,ch,mx}$ = maximum concentration of sediment that can be transported by the water (ton m^{-3} or kg L^{-1}), c_{sp} = user-defined coefficient, $v_{ch,pk}$ = peak channel velocity (m s^{-1}), and $spexp$ = user-defined exponent normally varies between 1.0 and 2.0 (Arnold et al., 1995).

$$v_{ch,pk} = prf \cdot \frac{q_{ch}}{A_{ch}} \tag{32}$$

where prf = peak rate adjustment factor, q_{ch} = average flow rate (m^3 s^{-1}), and A_{ch} = cross-sectional area of channel flow (m^2).

If the initial sediment concentration is higher than $conc_{sed,ch,mx}$, deposition is the dominant process in the channel segment, and the net amount of sediment deposited is calculated as:

$$sed_{dep} = \left(conc_{sed,ch,in} - conc_{sed,ch,mx}\right) \cdot V_{ch} \tag{33}$$

where sed_{dep} = amount of sediment deposited in the channel segment (metric ton), $conc_{sed,ch,in}$ = initial sediment concentration in the channel segment (ton m^{-3} or kg L^{-1}), and V_{ch} = volume of water in the channel segment (m^3).

If $conc_{sed,ch,in} < conc_{sed,ch,mx}$, degradation is the dominant process in the channel segment, and the net amount of sediment re-entrained is calculated as:

$$sed_{\deg} = \left(conc_{sed,ch,mx} - conc_{sed,ch,in}\right) \cdot V_{ch} \cdot K_{ch} \cdot C_{ch} \tag{34}$$

where sed_{deg} = amount of sediment re-entrained in the channel segment (metric ton), K_{ch} = channel erodibility factor (cm hr^{-1} Pa^{-1}), and C_{ch} = channel cover factor. Channel erodibility depends on channel bed and bank material properties. It can be measured with a submerged vertical jet device and determined using a method developed by Hanson (1990) and used by Allen et al. (1999). It is expressed as:

$$K_{ch} = 0.003 \cdot \exp[385 \cdot J_i] \tag{35}$$

where J_i = jet index as defined by Hanson (1991). In general, values for channel erodibility are an order of magnitude smaller than values for soil erodibility. The channel cover factor (C_{ch}) is the ratio of degradation from a channel with a specified vegetative cover to the corresponding degradation from a channel with no vegetation.

The net sediment load in the channel segment is:

$$sed_{ch} = sed_{ch,in} - sed_{dep} + sed_{\mathrm{deg}} \tag{36}$$

where sed_{ch} = amount of suspended sediment in the channel segment (metric ton) and $sed_{ch,in}$ = amount of suspended sediment in the channel segment at the beginning of time period (metric ton).

The amount of sediment transported out of the channel segment is calculated as:

$$sed_{out} = sed_{ch} \cdot \frac{V_{out}}{V_{ch}} \tag{37}$$

where sed_{out} = amount of sediment transported out of the channel segment (metric ton), and V_{out} = volume of outflow from the channel segment during the time step (m^3).

Channel Downcutting and Widening

Three channel dimensions are varied to simulate channel downcutting and widening: bankfull depth, channel width, and channel slope. These are updated using the following equations when water volume in the channel segment exceeds 1.4×10^6 m^3:

$$depth_{bnkfull} = depth_{bnkfull,in} + depth_{dcut} \tag{38}$$

$$W_{bnkfull} = ratio_{WD} \cdot depth_{bnkfull} \tag{39}$$

$$slp_{ch} = slp_{ch,in} - \frac{depth_{dcut}}{1000 \cdot L_{ch}} \tag{40}$$

where $depth_{bnkfull}$ = new bankfull depth (m), $depth_{bnkfull,in}$ = previous bankfull depth (m), $depth_{dcut}$ = new amount of downcutting (m), $W_{bnkfull}$ = new width of the channel at the top of the bank (m), $ratio_{WD}$ = channel width to depth ratio, slp_{ch} = new channel slope (m m^{-1}), $slp_{ch,in}$ = previous channel slope (m m^{-1}), and L_{ch} = channel length (km).

The amount of downcutting is calculated as:

$$depth_{dcut} = 358 \cdot depth \cdot slp_{ch} \cdot K_{ch} \tag{41}$$

where $depth$ = depth of water in the channel segment (m).

Sediment Routing in Impoundment

Sediment is routed through four types of impoundments: depressions/potholes, ponds, wetlands, and reservoirs/lakes. The same routing procedure is used based on a simple continuity equation for volumes and concentrations of inflow, outflow, and reservoir storage and assuming a completely mixed system. The mass balance equation is:

$$sed_{im} = sed_{im,in} + sed_{flowin} - sed_{stl} - sed_{flowout} \tag{42}$$

where sed_{im} = amount of sediment in the impounded water at the end of the day (metric ton), $sed_{im,in}$ = amount of sediment in the impounded water at the beginning of the day (metric ton), sed_{flowin} = amount of sediment added to the impounded water with inflow (metric ton), sed_{stl} = amount of sediment removed from the impounded water by settling (metric ton), and $sed_{flowout}$ = amount of sediment transported out of the impoundment with outflow (metric ton).

$$sed_{stl} = \left(conc_{sed,im,in} - conc_{sed,im,f}\right) \cdot V_{im} \tag{43}$$

$$sed_{flowout} = conc_{sed,im,f} \cdot V_{flowout} \tag{44}$$

where $conc_{sed,im,in}$ = initial sediment concentration in the impounded water (Mg m^{-3}), $conc_{sed,im,f}$ = final sediment concentration in the impounded water (Mg m^{-3}), V_{im} = volume of the impounded water which is equal to initial volume plus volume flowing in (m^3), and $V_{flowout}$ = volume of outflow from the impoundment (m^3).

$$conc_{sed,im,in} = \frac{\left(sed_{im,in} + sed_{flowin}\right)}{V_{im}} \tag{45}$$

The final sediment concentration ($conc_{sed,im,f}$) depends on sediment settling, which occurs only when the sediment concentration in the impounded water exceeds the equilibrium sediment concentration specified by the user, and is calculated as:

$$conc_{sed,im,f} = conc_{sed,im,in} \qquad \text{if } conc_{sed,im,in} \le conc_{sed,eq} \tag{46}$$

$$conc_{sed,im,f} = conc_{sed,eq} + \frac{\left(conc_{sed,im,in} - conc_{sed,eq}\right)}{e^{\left(k_s \cdot t \cdot d_{50}\right)}} \qquad \text{if } conc_{sed,im,in} > conc_{sed,eq} \tag{47}$$

where $conc_{sed,eq}$ = equilibrium sediment concentration in the impounded water (Mg m^{-3}), k_s = decay constant (day^{-1}), t = time step (1 day), and d_{50} = medium particle size of the inflow sediment (µm).

Assuming 99 percent of the 1 µm size particles settle out of solution within 25 days and k_s = 0.184, the median particle size of the inflow sediment is calculated as:

$$d_{50} = \exp\left(0.41 \cdot \frac{m_c}{100} + 2.71 \cdot \frac{m_{silt}}{100} + 5.7 \cdot \frac{m_s}{100}\right) \tag{48}$$

where m_c = percent clay in the surface soil layer in the subbasin, m_{silt} = percent silt in the surface soil layer in the subbasin, and m_s = percent sand in the surface soil layer in the subbasin.

Nutrients

The complete nutrient cycles for nitrogen and phosphorous are simulated. The nitrogen cycle is simulated using five different pools: two inorganic forms (ammonium and nitrate) and three organic forms (fresh, stable, and active). Similarly, six different pools of phosphorous in the soil are simulated: three inorganic forms and three organic forms. Mineralization, decomposition, and immobilization are important parts in both cycles. These processes are allowed to occur only if the temperature of the soil layer is above 0^{o}C.

Nutrient Simulations in Subbasin

Nitrate export with runoff, lateral flow, and percolation are estimated as products of the volume of water and the average concentration of nitrate in the soil layer. Organic N and organic P transport with sediment are calculated with a loading function developed by McElroy et al. (1976) and modified by Williams and Hann (1978) for application to individual runoff events. The loading function estimates daily organic N and P runoff loss based on the concentrations of constituents in the top soil layer, the sediment yield, and an enrichment ratio. The amount of soluble P removed in runoff is predicted using labile P concentration in the top 10 mm of the soil, the runoff volume, and a P soil partition coefficient (Knisel, 1980).

Stream Nutrient Simulations

In-stream nutrient transformations are simulated with a modified form of the QUAL2E model (Brown and Barnwell, 1987) with the components algae (as chlorophyll-*a*) dissolved oxygen, carbonaceous oxygen demand, organic N, ammonium-N, nitrite-N, nitrate-N, organic P, and soluble P (Ramanarayanan et al., 1996). Water temperature is estimated from air temperature using a regression relation (Stefan and Preud'homme, 1993) developed from numerous river observations.

Nutrient Simulations in Impoundment

A simple model for phosphorous mass balance from Thormann and Mueller (1987) is used. The model assumes: (1) completely mixed lake, (2) P limited, and (3) total P can be a measure of trophic status. The model is applicable when nonpoint sources dominate and a relationship exists between total P and biomass. It ignores lake stratification and the concentration of phytoplankton in the epilimnon. The P mass balance equation includes the concentration in the lake, inflow, outflow, and an overall loss rate.

Pesticides

Pesticides are simulated as per the GLEAMS model (Leonard et al., 1987), which is based on the plant leaf-area-index, application efficiency, wash-off fraction, organic carbon adsorption coefficient, and exponential decay according to pesticide half lives.

The model simulates pesticide transport by runoff, percolation, soil evaporation, and sediment. Pesticides may be applied at any time and rate to plant foliage or below the soil surface at any depth. The plant leaf-area-index determines what fraction of foliar-applied pesticide reaches the soil surface. Also, a fraction of the application rate (called application efficiency) is lost to the atmosphere. Each pesticide has a unique set of parameters, including solubility, half life in soil and on foliage, wash off fraction, organic carbon adsorption coefficient, and cost. Pesticide on plant foliage and in the soil degrades exponentially according to the appropriate half lives. Pesticide transported by water and sediment is calculated for each runoff event, and pesticide leaching is estimated for each soil layer when percolation occurs.

In-stream pesticide (toxic) transformations are simulated based on a model developed by Chapra (1997). The toxic is partitioned into dissolved and particulate states for both the water and sediment layers. The major processes simulated by the model are reactions, volatilization, settling, diffusion, re-suspension, and burial.

The lake toxic (pesticide) balance model also is based on Chapra (1997) and assumes well mixed conditions. The system is partitioned into a well-mixed surface water layer underlain by a well-mixed sediment layer. The toxic is partitioned into dissolved and particulate states for both the water and sediment layers. The major processes simulated by the model are loading, outflow, reactions, volatilization, settling, diffusion, re-suspension, and burial.

Management Practices

The SWAT incorporates detailed information on agricultural and urban land and water management into a simulation. A brief description of agricultural and water management practices are given below. Urban management is not discussed here.

General Agricultural Management

General agricultural management practices include tillage, planting, fertilizer, pesticide, grazing, harvest, kill, and filter strips. These management practices are incorporated into the model through various input data and parameters affected by the practices. For example, planting or beginning of growing season is incorporated by varying the curve number in the HRU throughout the year. A new curve number may be entered in a plant, tillage, harvest, or kill operation. The curve number entered is for moisture condition II and SWAT adjusts it daily to reflect change in water content.

Fertilization in an HRU may be scheduled by the user or automatically applied by the SWAT based on a nitrogen stress threshold, specified by the user. Filter strip trapping efficiencies for bacteria, sediment, nutrients, and pesticides are calculated based on power functions of filter strip width.

Water Management

Water management includes irrigation, tile drainage, impounded depressional areas, water transfer, consumptive water use, and point source loadings. Irrigation in an HRU may be scheduled by the user or automatically applied by the SWAT. In addition to specifying the timing and application amount, the user must specify the source of irrigation water, such as a channel reach, reservoir, shallow aquifer, deep aquifer, or outside source and location of the source within the watershed unless it is an outside source.

SWAT Storm-Event Enhancements

Storm events are critical in generating and carrying much, if not most, of yearly sediment and chemical loads (David et al., 1997; Borah et al., 2003). In this project, storm event enhancements of SWAT are investigated and a complimentary and compatible storm event hydrology model is selected, presented, and tested. Based on model reviews presented above, storm-event simulation routines (algorithms) described and verified by Borah et al. (2002a, 2002b, 2004a) are selected. These model routines simulate spatially and temporally varying (distributed: small time steps, minutes) surface and subsurface storm water runoff, propagation of flood waves, upland soil and streambed erosion, sediment transport, and agrochemical (nutrients and pesticides) transport in agricultural and suburban watersheds from spatially and temporally varying rainfall inputs resulting from rainfall events.

In this study, only the storm-event hydrologic model was investigated and tested for enhancement of the SWAT. The model uses interchangeable parameters with the SWAT and its watershed characteristics data are processed directly from SWAT data. Therefore, both models are complementary and compatible, which is unique. With the success of the hydrology enhancement, storm-event enhancements of soil erosion, streambed and bank erosion, and transport of sediment, nutrients, and pesticides could be done in future research.

Watershed Representation

The watershed is divided into one-dimensional overland planes, channel segments, and reservoir units (Borah et al., 2002a, 2004a). These divisions take into account the nonuniformities in topographic, soil, and land-use characteristics, which are treated as being uniform with representative characteristics within each of the divisions. An overland plane is represented as a rectangle, width is equal to the adjacent (receiving) channel length, and length is equal to the overland plane area divided by the width. Representative slope, soil, land cover, and roughness are based on physical measurements and observations. A channel segment is represented with a straight channel having the same length as in the field and having a representative cross-sectional shape, slope, and roughness based on physical measurements and observations. A reservoir unit is represented with a stage-storage-discharge relation (table) developed based on topographic data and discharge calculations using outlet measurements and established equations.

Storm-Event Hydrology

Overland planes are the primary sources of runoff. Two overland planes contribute surface runoff and subsurface flow to one channel segment laterally from each side. The excess rainfall is routed across an overland plane, resulting in variable flow along its slope length. However, cross-slope flow is assumed uniform. Thus, flow routing is only necessary within a unit width of the plane. Tile drain flows are combined with lateral subsurface flow using an effective lateral saturated hydraulic conductivity concept (discussed later). As a result, each channel segment receives time-varying, but spatially uniform, lateral inflows from the adjacent overland planes, in addition to inflows from upstream channel segment (or segments in the case of confluence), if any. The receiving waters from the overland planes throughout the watershed are routed through the network of channel segments, and reservoir units, if any, towards the watershed outlet.

Rainfall Excess and Infiltration

The storm-event model computes rainfall excess rates on overland planes at small time intervals (minutes) using the SCS (1972) runoff curve number equations (Equations 2 and 3), cumulative rainfall depths at each time step (total number of time intervals from beginning of simulation), and the following equation:

$$I_{e,i} = \frac{Q_{surf,i} - Q_{surf,i-1}}{\Delta t_i} \tag{49}$$

where i = time step: total number of time intervals from beginning of simulation (not day as in Eq. 2), Δt_i = time interval between time steps i-1 and i (h), $I_{e,i}$ = rainfall excess rate during time interval Δt_i (mm·h^{-1}), and $Q_{surf,i}$ = accumulated rainfall excess at time step i (mm).

Accumulated rainfall excess ($Q_{surf,i}$) is computed using Equation 2, but replacing daily rainfall (R_i) with cumulative rainfall depth at time step i. Assuming that evapotranspiration is negligible during a storm event, infiltration rates are computed by subtracting the rainfall excess rates ($I_{e,i}$) from rainfall intensities (rates) during the corresponding time intervals. This rainfall excess computation procedure is much simpler than any other procedure using interception and infiltration equations because of the single parameter, the runoff curve number *CN*, which is proven to be useful for half a century. The *CN* may be estimated based on soil-cover complexes (SCS, 1972, 1986), or calibrated using flow measurements (e.g., Borah, 1989; Borah et al., 2004a).

Surface Water Routing

The surface water routing algorithm for both overland planes and channel segments is based on kinematic wave approximations (Lighthill and Whitham, 1955) of the Saint-Venant or shallow water wave equations governing unsteady free surface flows. The governing equations are, respectively, the continuity and the approximate momentum equations in which the local and convective accelerations and pressure gradient terms are ignored:

$$\frac{\partial A}{\partial t} + \frac{\partial Q}{\partial x} = q \tag{50}$$

$$Q = \alpha A^m \tag{51}$$

where A = flow cross-sectional area (m^2), Q = flow rate of water discharge (m^3·s^{-1}), q = rate of lateral inflow per unit length (m^3·s^{-1}·m^{-1}), t = time (s), x = downslope position (m), α = kinematic wave parameter, and m = kinematic wave exponent.

Equations 50 and 51 are written for a channel, and also are used for overlands simply by substituting A, Q, and q with flow depth (m), rate of water discharge per unit width ($m^3 \cdot s^{-1} \cdot m^{-1}$), and rate of rainfall excess I_e ($m \cdot s^{-1}$), respectively. The kinematic wave parameter α and the exponent m are assumed independent of time and piecewise uniform in space (constant within each overland plane or channel segment), and are expressed as:

$$\alpha = \frac{S^{1/2}}{na^{2/3}} \tag{52}$$

$$m = (5 - 2b)/3 \tag{53}$$

$$P = aA^b \tag{54}$$

where S = longitudinal bed slope ($m \cdot m^{-1}$), n = Manning's roughness coefficient, P = wetted perimeter (m), and a and b = coefficient and exponent, respectively, in wetted perimeter versus flow area relation. For overland planes, a = 1.0 and b = 0.0, where P = 1.0 for unit width overland flow routing. For a channel segment, a and b are estimated from cross-sectional measurements. The lateral inflow q is assumed piecewise uniform in space and piecewise constant in time (constant over a time interval).

The water routing scheme is based on analytical and an approximate shock-fitting solutions of Equations 50 and 51 using the method of characteristics, as described in Borah et al. (1980), Borah (1989), or Borah et al. (2002a). The scheme is robust because of the closed-form solutions and only one calibration parameter, the Manning's roughness coefficient n.

It must be noted that the kinematic wave equations (Equations 50 and 51) generate only one system of characteristics, which means that they cannot represent waves traveling upstream as in the case of backwater flow. Therefore, the water routing scheme is not applicable when backwater flows are present, generally from downstream controls (e.g., dams and weirs), flooding, storm surge, or flood tide, which may occur and influence limited (small) flow lengths in an upland watershed; however, their overall influence may be quite negligible, especially during intense storms. Robustness of the scheme offers an excellent tradeoff to any error generated from the approximations, as the error can be corrected or compensated through adjustment (calibration) of the roughness parameter n.

Subsurface Flow Routing

A portion of the infiltrated water in an overland plane flows towards downstream as subsurface flow and ultimately discharges laterally into the contributing channel. The subsurface flow could be accelerated due to the presence of tile drains. The subsurface flow is computed using the kinematic storage equation (Equation 7) used in the SWAT. Although the equation was developed for mountainous watersheds (Sloan et al., 1983), it is also applicable to flatter slopes such as watersheds in Illinois (Borah et al., 2004a). Equation 7 is slightly modified for dimensional consistencies and expressed as:

$$q_s = K_s \sin\alpha \frac{2V}{L(\theta_s - \theta_d)} \tag{55}$$

where q_s = subsurface flow per unit overland width ($m^3 \cdot s^{-1} \cdot m^{-1}$), K_s = lateral saturated hydraulic conductivity ($m \cdot s^{-1}$), α = angle of the impermeable bed (^{o}), V = drainable volume of water stored in the saturated zone of a unit width of overland ($m^3 \cdot m^{-1}$), L = slope length (m), θ_s = saturated water content ($m^3 \cdot m^{-3}$), and θ_d = field capacity ($m^3 \cdot m^{-3}$).

Equation 55 is used with a modification to the K_s term to represent the lateral subsurface and tile-drain contributions from the overland planes to the channel flows, including base flows. In the presence of a tile drainage system, the overall hydraulic conductivity increases, and as a result the subsurface flow contribution to the channels (q_s) also increases. Therefore, tile drainage system in the model is represented through modifying the saturated hydraulic conductivity (K_s) to a combined hydraulic conductivity called the "effective lateral saturated hydraulic conductivity (*ELSHC*)." The *ELSHC* depends on porosity of the soil and the tile drainage system and may be different from field to field and overland to overland. In the model, the *ELSHC* is assumed time independent and its value for each overland plane is estimated through calibration and validation using monitored flow data and is the only parameter requiring calibration in the subsurface flow routing procedure.

Conservation of subsurface water mass is maintained by continuously updating the water volume (V) through solving the following spatially uniform and temporarily varying continuity equation:

$$fL - q_s = \frac{dV}{dt} \qquad (56)$$

where f = rate of infiltration: difference between rainfall intensity and rainfall excess rate I_e ($m \cdot s^{-1}$); and t = time (s).

Reservoir Flow Routing

Water through a reservoir unit is routed using the storage indication or modified Puls method (U.S. Bureau of Reclamation, 1949), as described in Hjelmfelt and Cassidy (1975).

Hydrology Parameters

The entire hydrology model requires calibration of only three parameters: runoff curve number *CN*, Manning's roughness coefficient *n*, and *ELSHC*. For spatially distributed modeling, calibration is required for all three parameters for each of the overland planes, and only the Manning's roughness coefficient for each of the channel segments. The remaining input variables are measurable or estimable. The *CN* and *n* are also estimable based on soil-cover complexes (SCS, 1972, 1986) and physical appearances (Chow, 1959), respectively, of the overland planes and channel segments; however, in watershed simulations, a combination of initial estimations and final calibrations is effective. Sensitivities of these two parameters and seasonal variations of *CN* were shown in Borah (1989) and Borah and Ashraf (1990), respectively, for a small watershed in Mississippi.

The above parameters are interchangeable with the SWAT. Calibrated values of these parameters from the SWAT are directly applicable and do not require recalibration, which is unique and, therefore, the models are compatible as well as complementary.

Coupling of Storm-Event Model with the SWAT

In this study, watershed characteristics data for the storm event model were processed from the SWAT data outside of the model, and the two models were run separately and independently. Coupling of both models will require modification of the SWAT user interface to process input data for the storm event model and then running it simultaneously with the SWAT, a subject of continuing (future) research.

Little Wabash River Watershed

Given that the Illinois State Water Survey (ISWS) has been conducting its mandate to characterize, and evaluate the availability, quality, and use of surface water resources of Illinois for more than 100 years, an Illinois watershed was selected to research. The 8,400 km^2 Little Wabash River watershed in southeastern Illinois (Figure 1) draining mostly from north to south was chosen because its water supply and watershed attributes are favorable for this research.

The Little Wabash River, a principal tributary of the Wabash River, which itself is a major tributary to the Ohio River, is a major river system in the eastern and central United States as may be seen in a satellite picture (Figure 2) taken on May 20, 2002. Flooding in the Little Wabash River, its major tributary the Skillet Fork, the Wabash River, and the Ohio River occurred on that day may be seen near the center of this satellite picture. The other major river system seen in this picture on that day is the Mississippi River from Minnesota at its upstream (north) end to its mouth in Louisiana and Gulf of Mexico at the south.

Water Supply Systems

For Illinois, the Little Wabash River watershed has a relatively high density of public intrastate surface water supplies. Seven small (population < 10,000) and three large public surface water supply systems serve communities in the Little Wabash River watershed (Knapp and Myers, 2001; USGS and IEPA, 2003).

The small water supply systems located north to south (Figure 1) and their populations are Neoga (1,854), Altamont (2,400), Flora (5,675), Clay City (1,033), Olney (9,016, Fairfield (5,421), and Wayne City (1,424). Similarly, the large systems are Mattoon (19,787), Effingham (18,065), and Rend Lake Intercity Water System (110,778). All but the last system are located within the Little Wabash River watershed drawing water captured and stored within it. Among the small systems, Flora, Clay City, and Fairfield draw water directly from the Little Wabash River where their intakes are located. In addition, Fairfield draws water from one side-channel storage. Figure 3b shows a cross-sectional view of the Little Wabash River under the bridge on Illinois State Route 15, approximately 0.5 km downstream of the Fairfield water supply intake. The river drains approximately 4,500 km^2 basin (54% of the Little Wabash River watershed) to this point. This large cross section of the river with very little base flow shows the flashy flow regime of the Little Wabash River watershed although there was very little release from the intake impoundment on the day of the photo (September 23, 2004).

Neoga, Altamont, and Olney currently draw water from impounded reservoirs: Neoga from Lake Mattoon, Altamont from Altamont Reservoir, and Olney from East Fork Lake or Olney Reservoir (Figure 1). Wayne City withdraws water from the Skillet Fork, a major tributary of the Little Wabash River (Figure 1), and a side-channel reservoir. Figure 4a shows the Skillet Fork impoundment for Wayne City withdrawal. The Skillet Fork drains approximately 1,200 km^2 basin (14% of the Little Wabash River watershed) to this point. Figure 4b shows the pump house of this second smallest water supply system (Wayne City, population 1,424).

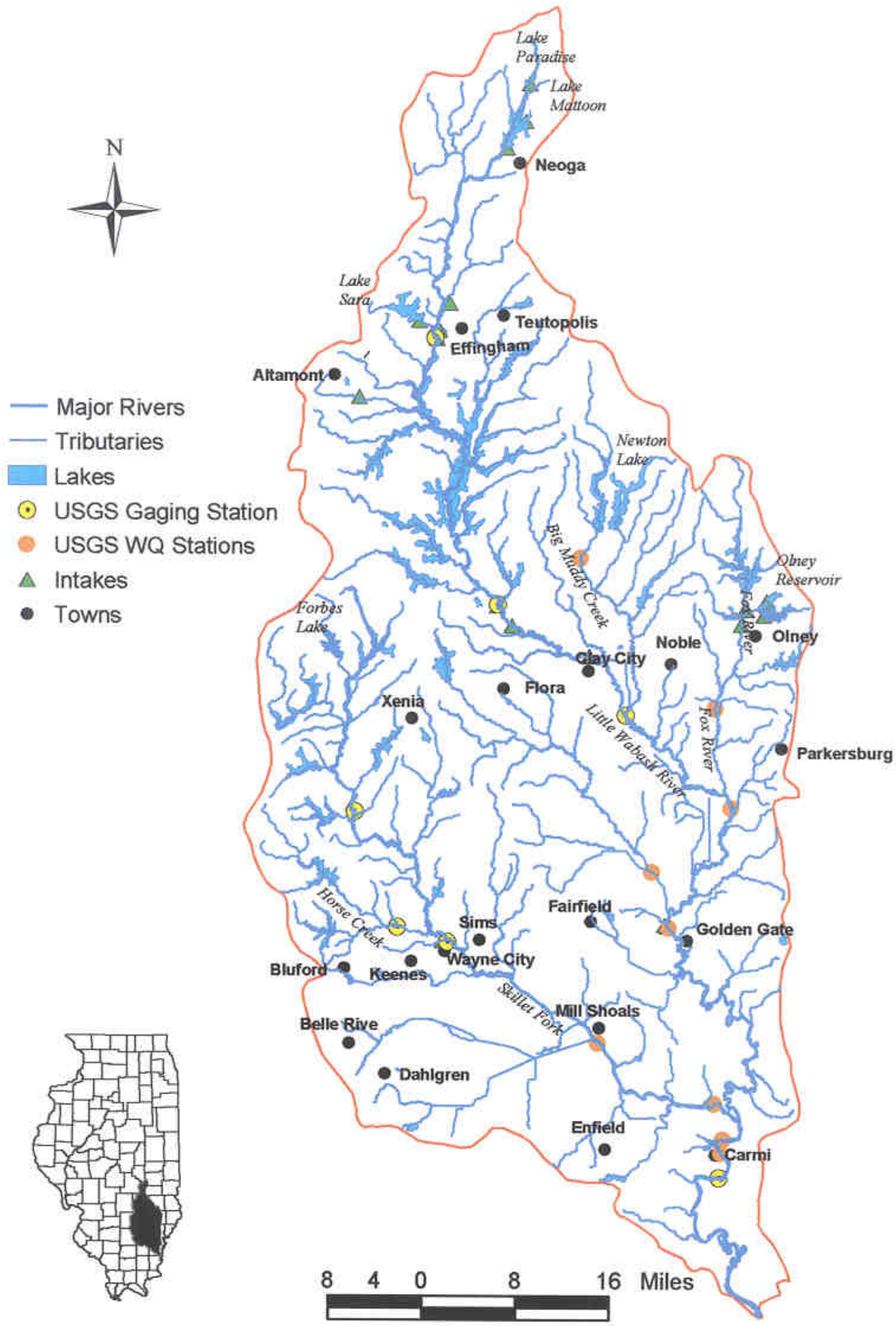


Figure 1. Little Wabash River watershed in Illinois (1 mile = 1.609 km)

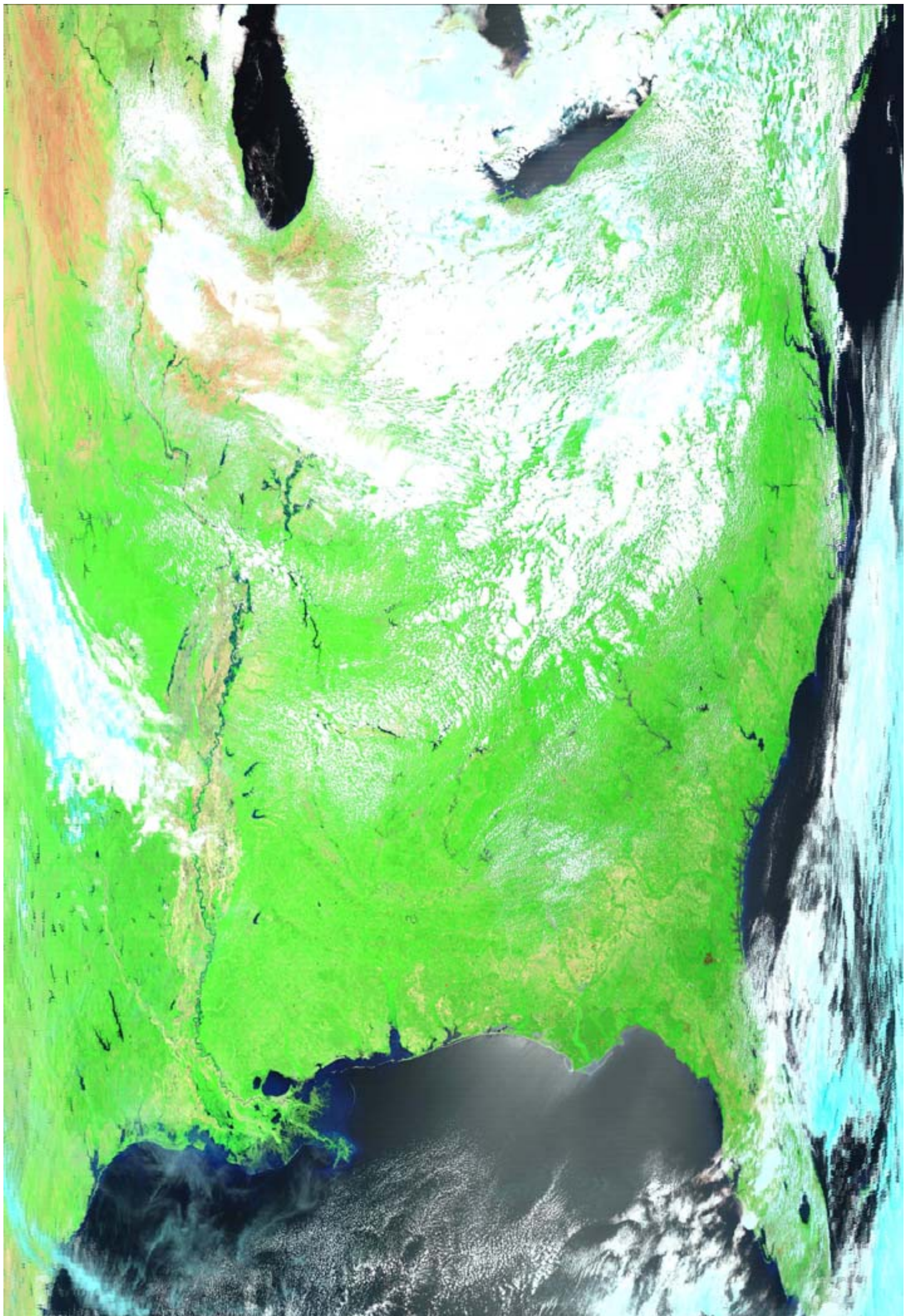


Figure 2. Satellite picture of eastern and central United States on May 20, 2002

(a) **(b)**

Figure 3. Little Wabash River: (a) I-57 at Effingham and (b) Route 15 at Fairfield (Photos 9/23/2004)

(a) **(b)**

Figure 4. Wayne City: (a) impoundment on Skillet Fork and (b) Water Works (Photos 9/23/2004)

Among the large water supply systems, Effingham withdraws water from the Little Wabash River, Lake Sara (Figure 1), and Central Illinois Public Service (CIPS) Lake, a 341-acre foot (ac-ft) water supply lake located near the Little Wabash River. Figure 3a shows the Little Wabash River looking upstream from the Interstate 57 southbound bridge, which is located approximately 5 km downstream of the Effingham intake. Drainage area up to this point is approximately 1,000 km^2 (12 % of the Wabash River watershed). As may be seen, the low base flow indicates a flashy flow regime during storm events and negligible contribution from groundwater.

Mattoon withdraws water from Lake Mattoon and Lake Paradise (Figure 1). The Rend Lake Intercity Water System is located outside of the Little Wabash River watershed on its southwest side and serves the southwest communities of the watershed. The Rend Lake Intercity Water System withdraws water from the Rend Lake located in the Big Muddy River watershed.

Hydrologic Characteristics and Information

The Little Wabash River watershed (Figure 1) consists of 2 U.S. Geological Survey (USGS) 8-digit watersheds: watersheds of the main-stem Little Wabash River (HUC No. 05120114) and the Skillet Fork River (HUC No. 05120115).

Figure 5 shows a land cover map of the Little Wabash River watershed. Land use (IDNR, 2001) is predominantly row-crop agriculture (62%) with a good mixture of forest (15%), grassland (19%), and wetland (4%); as exemplified by Figures 3, 4, and 6. Figure 6, a photo taken from Interstate 57 on September 23, 2004, shows a cornfield being harvested and trees along the horizon.

The watershed (Figure 1) is naturally parceled progressively downward in size from its one large tributary, Skillet Fork River (2,740 km^2) to Big Muddy Creek (820 km^2), Elm River (720 km^2), Fox River (530 km^2), Main Outlet (460 km^2), Horse Creek (270 km^2), Auxier Creek Drain (260 km^2), and finally Salt Creek (240 km^2). There are 15 artificial reservoirs and 5 low channel dams (immediately downstream of public water supply intakes to keep sufficient water pool during low flow) in the watershed (Knapp and Myers, 2001).

The 15 reservoirs and their storage capacities (acre-ft) from upstream to downstream of the watershed are: Lake Paradise (1,241), Lake Mattoon (11,820), Lake Sara (13,357), Central Illinois Public Service Lake (341), Altamont Reservoir (931), Pauline Lake (350), Newton Lake (28,500), East Fork Lake (12,460), Olney Lake (1,540), Vernor Lake (734), Greendale Lake (170), Patterson Lake (270), Forbes Lake (6,793), Sam Dale Lake (999), and Illinois Central Reservoir (421). In addition, Goose Pond, a natural lake with 34 acre surface area is located in Wayne County. The five low channel dams include three on the Little Wabash River, one on the Skillet Fork, and one on the Fox River (Figure 1). Two of the Little Wabash River dams are intakes to public water supplies, one near Effingham and the other near Louisville for Flora. The third dam is at Carmi. The Skillet Fork dam is at the Wayne City water supply intake. The Fox River dam is an inactive intake for the Olney public water supply.

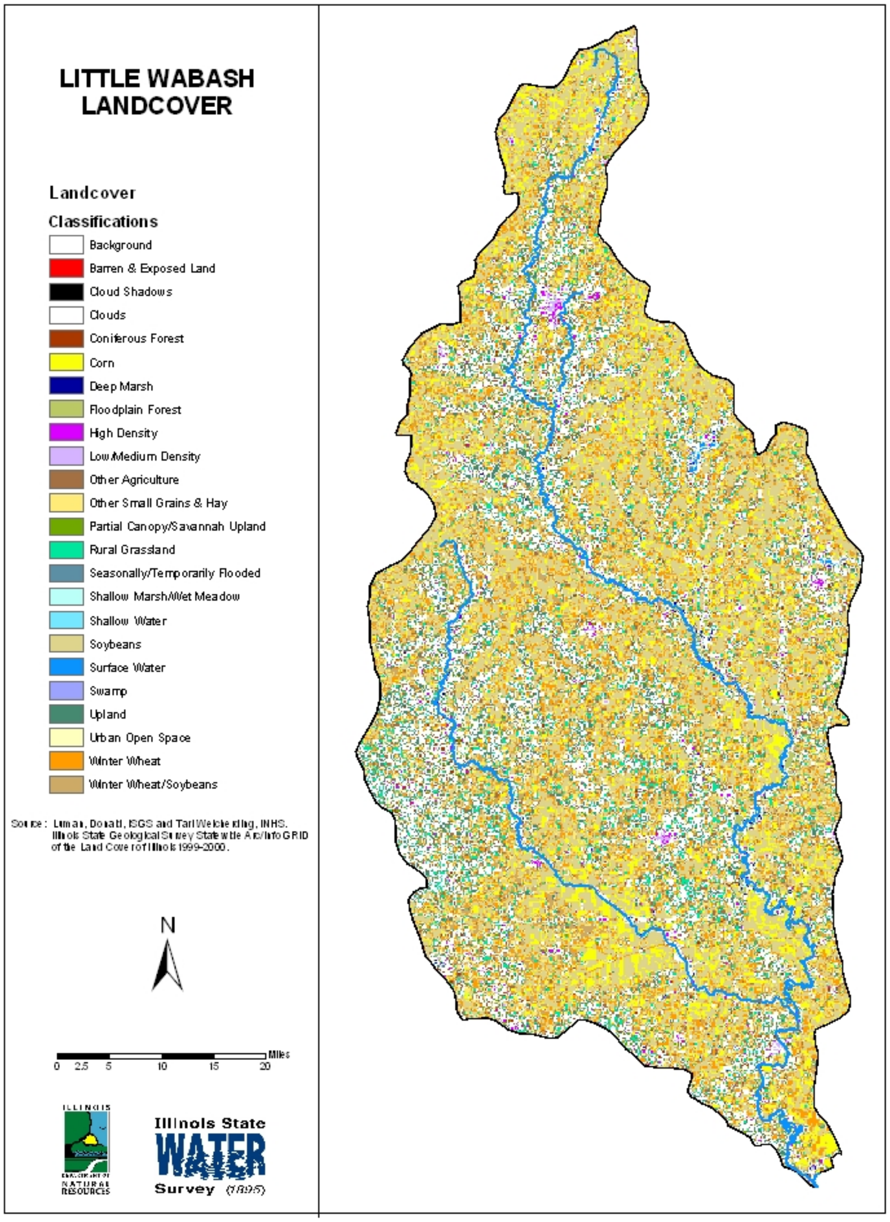


Figure 5. Little Wabash River watershed land cover (IDNR, 2001)

Figure 6. A cornfield along I-57 in the upper Little Wabash River watershed (Photo 9/23/2004)

Rainfall data at 14 National Weather Service (NWS) stations in and around the watershed (Mattoon, Effingham, Mason, Newton, Louisville, Salem, Iuka, Flora, Clay City, Olney, Fairfield, Mt. Vernon, Wayne City, and Carmi) are available from the National Climatic Data Center or NCDC (http://www.ncdc.noaa.gov/oa/ncdc.html, accessed December 1, 2005). Daily rainfall data are available at all the stations, but 15-minute interval data are available at only a few of the stations (e.g., Effingham, Mason, Flora, and Carmi). Other meteorological data (wind speed, air temperature, solar radiation, etc.) at these stations are available from the Midwest Regional Climate Center (MRCC) at the ISWS.

Four active USGS gaging stations in the watershed (Figure 1) have daily and 15-minute flow records: Little Wabash River near Effingham (620 km^2), below Clay City (2,930 km^2), and at Carmi (8,000 km^2, near the watershed outlet); and Skillet Fork at Wayne City (1,200 km^2). Daily and 15-minute flow records were obtained, respectively, from http://waterdata.usgs.gov/il/nwis/sw (last accessed December 20, 2005) and G. Johnson (Personal Communication, January 5, 2005, USGS, Urbana, IL).

Historically, stream water quality data in Illinois has been collected by the ISWS, IEPA, USGS, and the Illinois Department of Public Health (Short, 1999). The longest-term data are held by ISWS, which has published surface and groundwater quality data for the Little Wabash River dating back into the 1890s (e.g., Palmer, 1897; Bartow et al., 1909). Regarding the most modern data, seven USGS discharge measurement stations were used to measure water quality parameters. In addition to these stations, nine IEPA ambient water quality monitoring (AWQMN) stations were used to monitor water quality parameters periodically (e.g., Hite et al., 1993; Shasteen et al., 2002, 2003). These stations are shown as WQ stations in Figure 1. The IEPA (2003), in cooperation with the IDNR, conducted an intensive survey of the Little Wabash and Lower Wabash River basins. Stream conditions were evaluated by collection of water chemistry, sediment chemistry, in-stream habitat, macroinvertebrate community and fish community samples.

According to the USGS and IEPA (2003), all the water supply sources have detectable levels of atrazine, a commonly used herbicide, but few exceed the maximum allowable concentration (MCL) of one part per billion. Almost all the major stream segments are included in the Illinois 2004 Section 303(d) List (IEPA, 2004a) of the Federal Clean Water Act (CWA) as impaired. Types of impairment include elevated nutrient levels, habitat alterations, organic enrichment, low dissolved oxygen, and siltation (sedimentation). Principal sources of these impairments are nonpoint agricultural and point municipal sources, and hydrologic/habitat modifications. As recognized by Broeren and Singh (1989), continuing sedimentation of water supply reservoirs results in decreasing yields over time whereas water demand typically increases over time.

Historical Background

The rich land-use history of the Little Wabash River watershed shows that interrelated human and hydrologic factors have made this Illinois watershed an exemplary area for modeling effects of nonpoint pollution on small and rural surface water supplies. While row crop agriculture is still the predominant land, its land-use diversity and within-watershed land-use distribution makes research here relevant to a wide range of Midwest conditions. Past events in this historic watershed also illustrate the great human and ecological costs of inadequate water resources development.

The Little Wabash River watershed is the smallest of the 10 major Illinois watersheds (IDNR, 2001). This is one of the four areas in which the Corn Belt system of agriculture was developed (Spencer and Horvath, 1963). Most of the settlement in the watershed occurred between 1840 and 1880. Whereas the rest of Illinois continued to grow, the population of the watershed began decreasing shortly after 1900 (Barker et al., 1967; Langford, 1979). As the agricultural revolution progressed during the 20th Century, optimal farm size increased. Smaller farms were consolidated into bigger farms. The towns of the Little Wabash River watershed were unable to absorb people leaving the farm. Decreasing rural population also meant decreasing need for the goods and services that the towns provided. Like other rural areas lacking an adequate infrastructure to support an industrial base, and too far away to serve as a bedroom for metropolitan areas, the Little Wabash River watershed underwent depopulation. Thus, the developmental history of the Little Wabash River watershed is broadly important; it shows the importance of water resources development — watershed growth was retarded by the low level of development of water resources (Habermeyer, 1918; State Water Survey Staff, 1948; Barker et al., 1967; U.S. Army Corps of Engineers, 1979). That the history of water resources development has made the Little Wabash River watershed a very rural and sparsely populated agricultural watershed makes this watershed a great place in which to model the effects of nonpoint, agricultural pollution on the water quality needs of small rural communities.

The surface water and groundwater resources of the Little Wabash River watershed are severely constrained. Significant and usable groundwater resources are limited to the sand and gravel deposits underlying the southernmost portion of the watershed and sandy glacial tills underlying the northernmost portion of the watershed (Worthen et al., 1875; Barker et al., 1967; Zuehls, 1987). The uplands of the watershed are overlain by highly weathered soils underlain by weathered silt/clay glacial drift < 15 m thick. These surface deposits markedly limit surface recharge to potential underlying water-bearing strata. Indeed, subsoils are of such pure clay that watershed subsoils were an abundant economic geologic resource for the manufacture of bricks (Worthen et al., 1875). Bedrock underlying the clayey soils and glacial drift is generally of low porosity, even the sandstones and limestones. Not surprisingly, bedrock groundwater sources tend to be limited in yield. These groundwaters also tend to be highly mineralized with high iron content (Hansen and Hilscher, 1914; National Research Council, 1981; Zuehls, 1987) and, "Quite often the ground water contains methane and hydrogen sulfide" (Barker et al., 1967, p. 30). Thus, most of the watershed is compelled to import water or to use surface water.

In their undeveloped state, the surface water resources of the watershed are naturally unreliable. There are no natural lakes to speak of in the Little Wabash River watershed other than a few small and shallow abandoned river bottom lakes (oxbow lakes). Even though it is in the subhumid East, the Little Wabash River watershed suffers from extremely low flows during dry years: about 1 cfs/600 mi^2 of flow is generated at the 5-year interval level — about 6 cfs for the entire 3,200 mi^2 (8,400 km^2) watershed. Climate, soils, and geology make for flashy flow conditions — little to no groundwater baseflow during dry periods and relatively great amounts of surface runoff generally generated by convective storms of short duration and high intensity. However, the Little Wabash River watershed is surface water rich in that reservoirs could be built to capture storm runoff events. Such water resource management would ensure a plentiful year-round water supply as well as improve watershed water quality through low-flow augmentation (State Water Survey Staff, 1948; U.S. Public Health Service, 1965; Barker et al., 1967). The following historical watershed précis is one of underdevelopment of surface water resources.

As the Little Wabash River watershed was settled, people found that the wide floodplains of the Little Wabash River and its major tributary — the Skillet Fork — contained the best soil and access to water. But they could not be successfully farmed because of frequent severe flooding. Therefore, the nature of the watershed compelled settlers to farm uplands — the most soil-poor and water-poor portions of the watershed. They had to use shallow wells. Often these wells were of large diameter (dug) because of low infiltration rates. While these wells produced clear and cool water, they were readily contaminated and unsanitary. Typhoid fever, dysentery, and other water-borne diseases were endemic (Palmer, 1897, 1903; Bartow, 1911, 1912).

Communities that developed on favorable riverside locations and that used surface waters did not treat the water even though the rivers and streams were used as open sewers (Palmer, 1897, 1903; Collins, 1910; Bartow, 1936) and were choked with sediment from the highly erodible upland soils from which up to 9 inches (23 cm) of soil were eroded since their development (Demissie and Akanbi, 1994). Many town dwellers chose to dig shallow wells on their properties rather than use the murky, untreated surface waters supplied by municipalities. However, "The ground in our towns and villages is honeycombed with privy vaults, cesspools, and loose-jointed drains, and everywhere the soils more or less covered with refuse matters of vegetable and of animal origin, of which the proportion represented by barnyards, pigpens, and the like..." (Palmer, 1897, pp. 23). These well waters were especially unsanitary and further contributed to the unhealthy reputation of the area (Bartow, 1912; 1913; 1914), a condition that did not help to retain people in the watershed or to attract outsiders to it.

Watershed population peaked sometime between 1900 and 1910, after which watershed population declined. Farm size increased and fewer people were needed to farm. It was long recognized that, for the watershed to retain people leaving the farms, water resource development would be necessary to support urban population increase. Increased water supply would also be necessary to develop industries to process that produced from the watershed's natural resources: agriculture, forestry, mining and chemicals. However, the requisite water supply and flood control reservoirs were not built and the watershed depopulated; the majority of the watershed is able to support only small public surface water supplies (State of Illinois Rivers and Lakes Commission, 1914; Habermeyer, 1918; Pickels and Leonard, 1929; State Water Survey Staff, 1948; Executive Committee on Southern Illinois, 1949; Beimfohr, 1954; U.S. Public Health Service, 1963, 1965; Barker et al., 1967; Illinois Water Pollution and Water Resources Commission, 1967; U.S. Army Corps of Engineers, 1979). This seriously constrained human and economic development of the watershed (e.g. Habermeyer, 1918; Barker et al, 1967). Because of low-flow water quality problems, water supplies had to assume extra expense in treatment (State Water Survey Staff, 1948; State of Illinois Department of Business and Economic Development, 1970) and towns and waste-generating facilities had to assume extra expenses to treat and dispose of their wastewaters because of the extremely low waste assimilatory capacity of streams and rivers at their low flow conditions (Barker et al., 1967).

But lack of flood control did not keep out the oil wells, which were drilled in all types of terrain (Barker et al., 1967). Illinois has produced more than 4 billion barrels of oil (Ridgley, 1997). The largest concentration of Illinois' oil production fields occur in the Little Wabash River watershed (Flemal, 1981). Oil drilling resulted in a population rebound during the 1930s –1950s (Barker et al, 1967; Fisher, 1968; Langford, 1979). There was essentially pervasive pollution of the watershed's surface waters by oil and brine from oil drilling operations from the 1920s into the 1950s; floods also contributed to the pollution problem by damaging oil wells (Barker et al., 1967; Fisher, 1968). In the last decades of the 20th Century, the biological and chemical effects of the oil industry on Little Wabash surface waters appear to be negligible (Flemal, 1981; Hite et al., 1993; Shasteen et al., 2002; USGS and IEPA, 2003). Chloride (Cl^-) is an indication of oil well brine pollution and Cl^- concentrations are higher in the middle reaches of the watershed. However, geology (Flemal, 1981), the natural presence of salt licks (Jones and Hanson, 1985), and mineral springs (Worthen et al., 1875), large enough to be of national and commercial importance (Peale, 1886), indicate that any persistent oil brine pollution of groundwater may be superimposed upon naturally saline groundwaters in contributing Cl^- to these surface waters.

In conclusion, the historically poor level of watershed development of water resources severely hindered development of the Little Wabash River watershed. This is true for all but the northernmost end of the watershed. This part of the watershed is covered by young and highly fertile prairie soils and sandy/gravelly glacial deposits which provided ample, accessible, high quality water and economic wealth for development (Barker et al., 1967; Zuehls, 1987). Here water resources were developed. It is here in the north where two larger population (> 10,000) towns of the watershed (Mattoon and Effingham) were developed. The U.S. Environmental Protection Agency (USEPA)-funded Midwest Technology Assistance Center for Small Public Water Supplies (MTAC) supports research for small (population < 10,000) public water supplies and these two towns are excluded.

Regarding the modern state of Little Wabash River watershed surface waters, they can be said to be eutrophic waters that suffer from seasonal low levels of dissolved oxygen and taste and odor problems. Potable water quality standards for iron, manganese, and fecal coliforms are routinely exceeded and treated for, along with high levels of suspended solids. All the water supply sources have detectable levels of atrazine, a commonly used agricultural herbicide, but few exceed the maximum allowable concentration (MCL) of one part per billion (USGS and IEPA, 2003). While heavily polluted from oil well and domestic wastes in the past, surveys of the last decades suggest negligible point-source pollution (Flemal, 1981; Hite et al., 1993; Shasteen et al., 2002).

Finally, the literature shows that the ecological characteristics of the Little Wabash River watershed suggest that planners strike a balance between water use and ecological needs. Ecological research has led to nearly one-fourth of the length of the Little Wabash River to be listed as a Biologically Significant Stream. And 88 percent of the second largest Resource Rich Area of Illinois — the Southern Till Plain — lies within the Little Wabash River watershed (IDNR, 2001).

As discussed above, such planning was made to simultanteously improve the water quality and water supply capacities of the Skillet Fork and Little Wabash rivers in the decades following World War II. But these were not implemented. Today virtually every major stream and river mile of the Little Wabash watershed has impairment from sediment, nutrient enrichment, and other agricultural chemicals (IEPA, 1998, 2004a). There is insufficient water supply for population increase and industrial development; these rivers being capable of only supporting small public surface water supplies. The above review shows that interrelated human and hydrologic factors have made this Illinois watershed an exemplary area for modeling effects of nonpoint pollution on small, rural surface water supplies and, thus, justifies this modeling study.

Recent Studies

As part of the Wabash River Basin Comprehensive Study in Indiana, Illinois, and Ohio, the U.S. Army Corps of Engineers (1967) investigated reservoir sites in the Little Wabash River watershed. Barker et al. (1967) conducted a comprehensive plan for water resources development in the Little Wabash River watershed. The Wabash River Coordinating Committee (1971), a multi-agency group from U.S. Department of Agriculture, Federal Highway Administration, U.S. Department of Health, Education and Welfare, U.S. Department of the Interior, U.S. Department of Transportation, U.S.

Environmental Protection Agency, Federal Power Commission, Wabash Valley Interstate Commission, State of Illinois, State of Indiana, State of Ohio, and finally U.S. Department of the Army, published a 14-volume report on comprehensive survey of the Wabash River basin, which includes the Little Wabash River and its watershed.

Eheart and Libby (1980) investigated irrigation water use in the Little Wabash River watershed. Huff (1981) reported hydrometeorology data and information of heavy rainstorms during the early 1900s. Flemal (1981) conducted and reported a comprehensive water quality investigation in the Little Wabash River watershed more than two decades ago. Singh et al. (1988) computed and reported 7-day, 10-year low flows in the Little Wabash River. Knapp and Myers (2001) developed a streamflow assessment model of the watershed for statistical streamflow analyses.

Modeling the Little Wabash River Watershed

The SWAT was applied to the Little Wabash River watershed. Geographic Information System (GIS) data on topography, soil, and land use for the two USGS 8-digit watersheds in the Little Wabash River watershed were retrieved from links provided at the USEPA's BASINS database at http://www.epa.gov/OST/BASINS/. These data were used to define watershed and subwatershed boundaries, compute their dimensions and representative slopes, and estimate various model parameters.

Long-Term Continuous Modeling

Based on the BASINS GIS data, the Little Wabash River watershed was divided into 88 subwatersheds (Figure 7) for SWAT-continuous simulations. The model groups these subwatersheds based on climate, HRUs, ponds, ground waters, and main channels. The HRUs are lumped land areas within the subbasin with unique land cover, soil, and management combinations with uniform parameter values. Parameters are physically based, whose ranges of values are given by the model, and are manually adjusted within the given range during model calibration to best match the simulated values of the model variables with observed variables. Statistical parameters – coefficient of determination (COD or R^2) and Nash-Sutcliffe coefficient or NSC (Nash and Sutcliffe, 1970) – were computed to evaluate comparisons of simulated and observed values. The COD indicates strength of relationship and NSC indicates closeness between the simulated and observed values. If both equal one, model predictions are considered perfect.

Based on availability of data, a 5-year period (1995-1999) was chosen to calibrate the model, and a 3-year period (2000-2002) was chosen to validate it.

Continuous Flow Simulation

Daily precipitation and air temperature data at the 14 precipitation gages were obtained from the NCDC website given above. Any missing data are filled with estimates from available observations at neighboring stations. Daily flow records at the four gaging stations – Effingham (620 km^2), Wayne City (1,200 km^2), Clay City (2,930 km^2), and Carmi (8,000 km^2) – were obtained from the USGS. Carmi is located near the watershed outlet, and its flows may be considered as the watershed outflows.

While calibrating the model, adjustments were made to nine parameters for the sub-watersheds contributing to a gaging station, including (1) SCS runoff curve number *CN*, (2) soil evaporation compensation factor *ESCO*, (3) plant uptake compensation factor *EPCO*, (4) threshold water level in shallow aquifer for base flow *GWQMN*, (5) threshold water level in shallow aquifer for re-evaporation and/or deep percolation *REVAPMN*, (6) groundwater re-evaporation coefficient *GW_REVAP*, (7) groundwater delay *GW_DELAY*, (8) baseflow recession constant *ALPHA_BF*, and (9) deep aquifer percolation fraction *RCHRG_DP*. Once calibrated, parameters for upstream subwatersheds were kept the same while adjusting those on further downstream subwatersheds based on downstream flows.

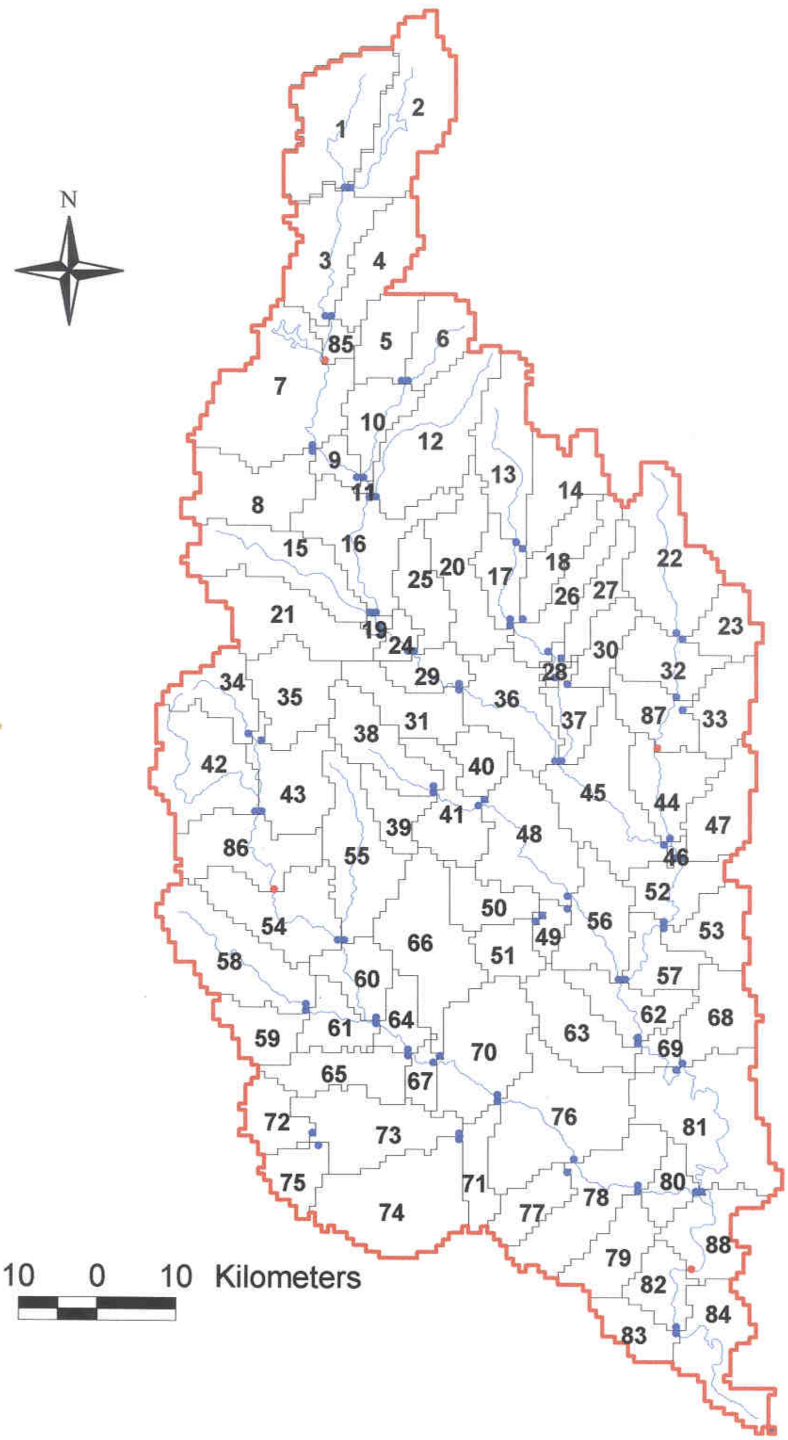


Figure 7. Little Wabash River watershed divided into 88 subwatersheds

Figures 8-11 compare simulated and observed monthly average flows (water discharges) at Effingham, Wayne City, Clay City, and Carmi, respectively, having increasing drainage areas (620, 1,200, 2,930, and 8,000 km^2, respectively), for both the calibration and validation periods (1995-2002), along with monthly average precipitation computed from the 14 precipitation stations and plots of COD and NSC values for individual and cumulative years. As may be seen in these plots, model performance in simulating monthly flows is quite variable with respect to the distributed stations and various years of the 8-year simulation.

For the first 5-year as calibration period (1995-1999), COD values for Effingham, Wayne City, Clay City, and Carmi are 0.58, 0.30, 0.65, and 0.53, respectively, and NSC values are 0.57, 0.26, 0.45, and 0.53, respectively. Although the overall parameter values for Wayne City are low, values for 3 of the 5 individual calibration years are above or near 0.5 (Figure 9), with COD and NSC values both as high as 0.85.

Parameter values were unchanged while running the model for the 3-year validation period (2000-2002). Respective model results for this period are extended from the calibration period and combined in Figures 8-11. The COD and NSC values for the individual years of the validation period are shown, and the values are reasonable except for the year 2001. As may be seen from the monthly flow hydrographs, year 2001 was a relatively dry year. Instead of computing cumulative COD and NSC values for the 3-year validation period, cumulative values for the entire 8-year simulation period were computed to show these values for a longer period of simulation.

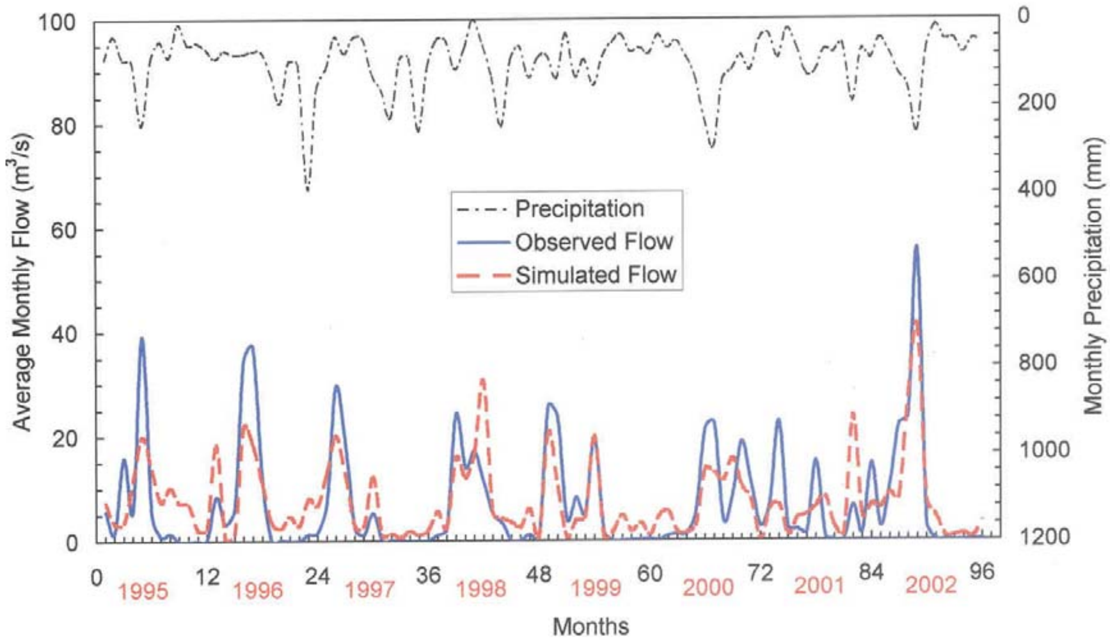
For the entire 8-year simulation period (1995-2002), COD values for the above stations (Effingham, Wayne City, Clay City, and Carmi) are 0.61, 0.37, 0.63, and 0.56; NSC values are 0.60, 0.33, 0.44, and 0.56, respectively. All stations except Wayne City have COD or NSC values above or near 0.5 and, therefore, provide reasonable overall predictions of monthly flows. Although the overall parameter values for Wayne City are low, values for 5 of the 8 individual years are more than or close to 0.5 (Figure 9), COD values are as high as 0.91, and NSC values are as high as 0.85. Therefore, overall model performance at Wayne City in simulating monthly flows also can be considered reasonable.

As can be seen in Figures 8-11, the model reasonably predicted the distributed monthly average flows in the Little Wabash River watershed although there were some discrepancies, especially most of the peak flows (underpredictions). Therefore, the SWAT is a promising long-term continuous simulation model, which needs enhancements in storm-event simulations to improve its peak flow predictions.

The spatially and temporally distributed parameters shown in Figures 8-11 were unique and were useful in the critical evaluations of the model results. To our knowledge, no other study has conducted this level of detailed evaluations and analyses.

Continuous Sediment Simulation

Figures 12-13 compare simulated and observed monthly sediment loads (metric tons) at Effingham, Wayne City, Clay City, and Carmi, for the period 1995-1999, along with irregularly and instantaneously measured (IEPA, 2003) sediment concentrations (mg L^{-1}). The period 1995-1999 was chosen because of the data availability.



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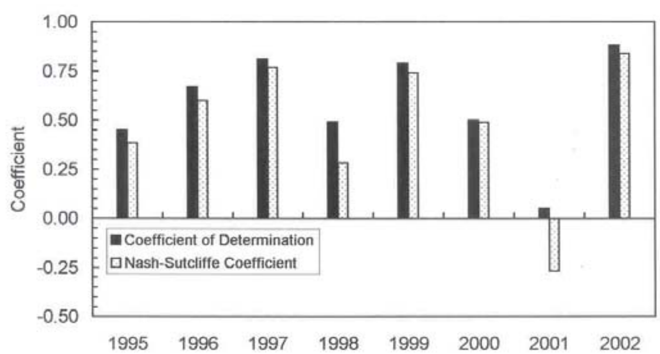

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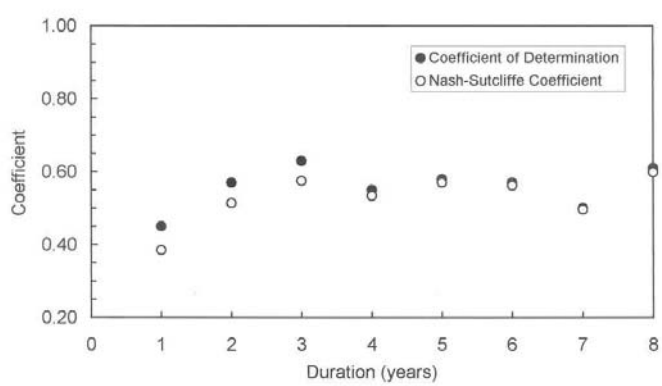
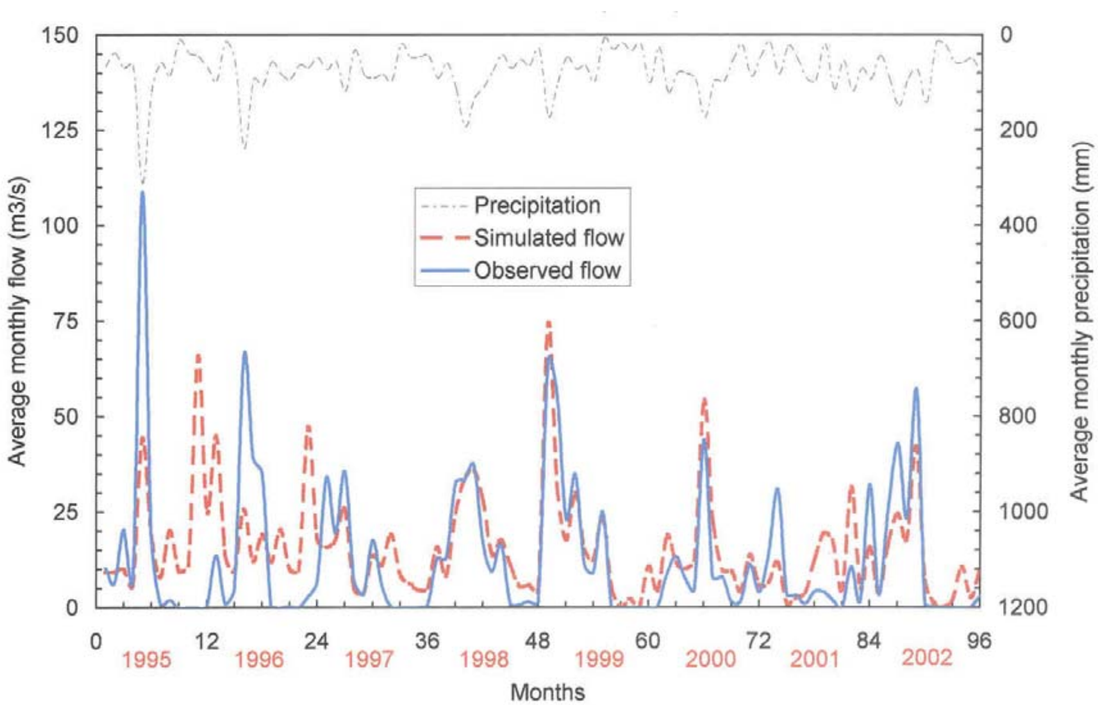


Figure 8. Comparisons of simulated and observed monthly flows of the Little Wabash River at Effingham: (a) flows and precipitation, (b) COD and NSC values for individual years, and (c) COD and NSC values for cumulative years (duration)



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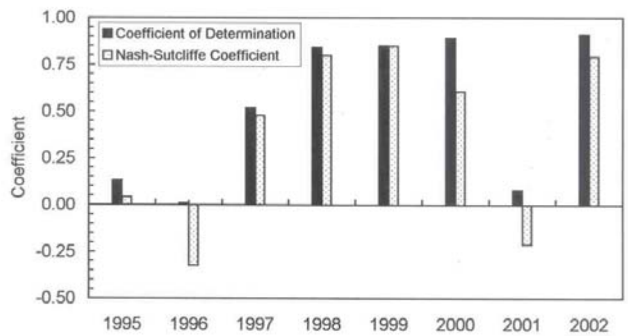

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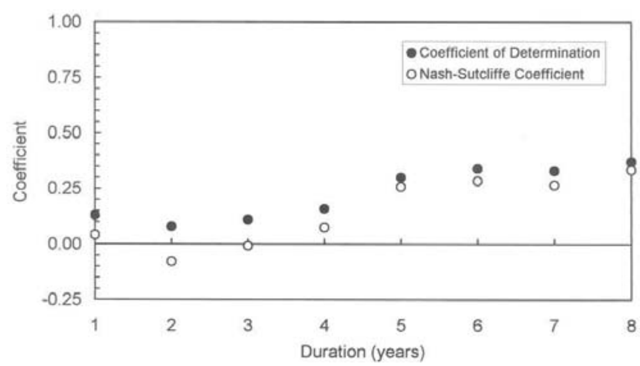
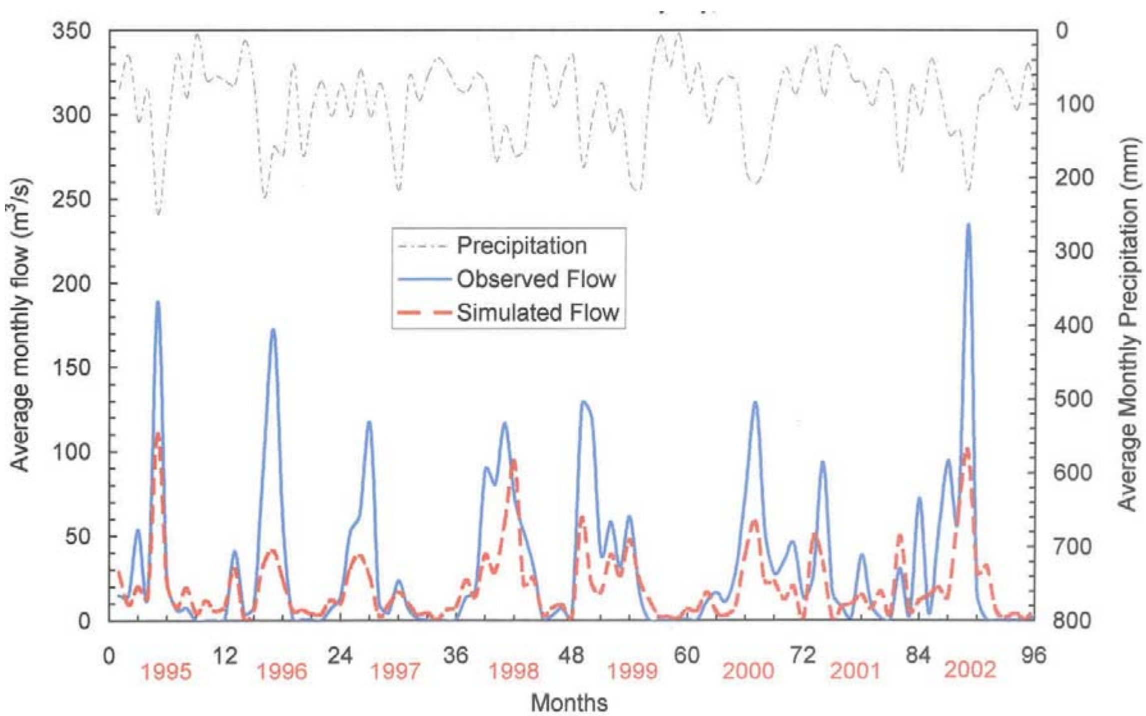


Figure 9. Comparisons of simulated and observed monthly flows of the Skillet Fork at Wayne City: (a) flows and precipitation, (b) COD and NSC values for individual years, and (c) COD and NSC values for cumulative years (duration)



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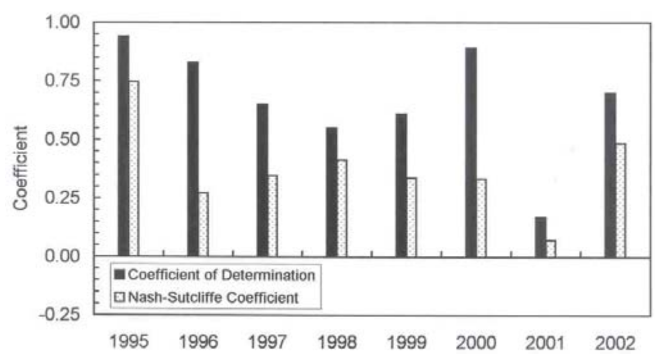

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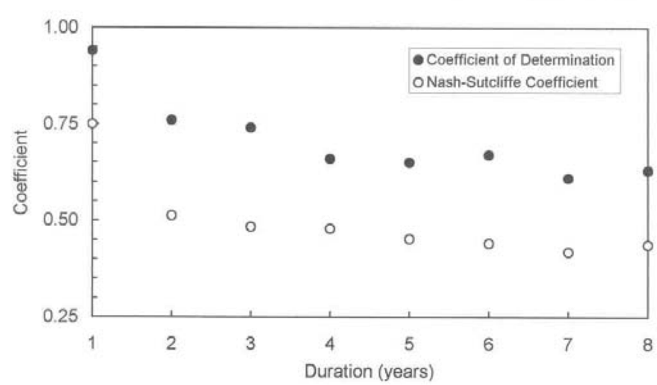
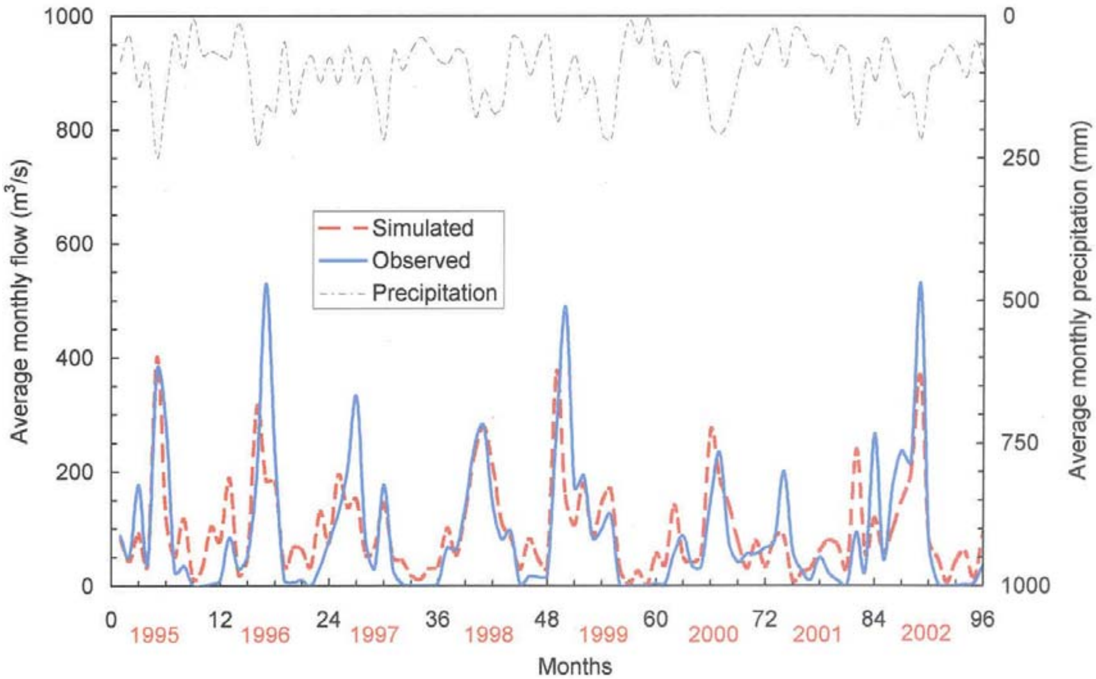


Figure 10. Comparisons of simulated and observed monthly flows of the Little Wabash River at Clay City: (a) flows and precipitation, (b) COD and NSC values for individual years, and (c) COD and NSC values for cumulative years (duration)



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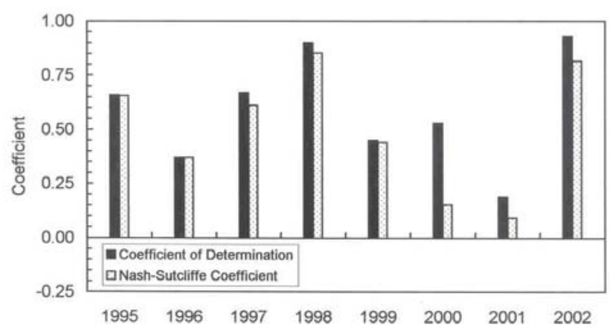

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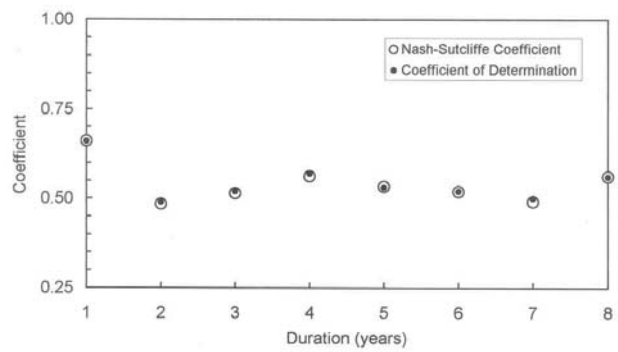


Figure 11. Comparisons of simulated and observed monthly flows of the Little Wabash River at Carmi: (a) flows and precipitation, (b) COD and NSC values for individual years, and (c) COD and NSC values for cumulative years (duration)

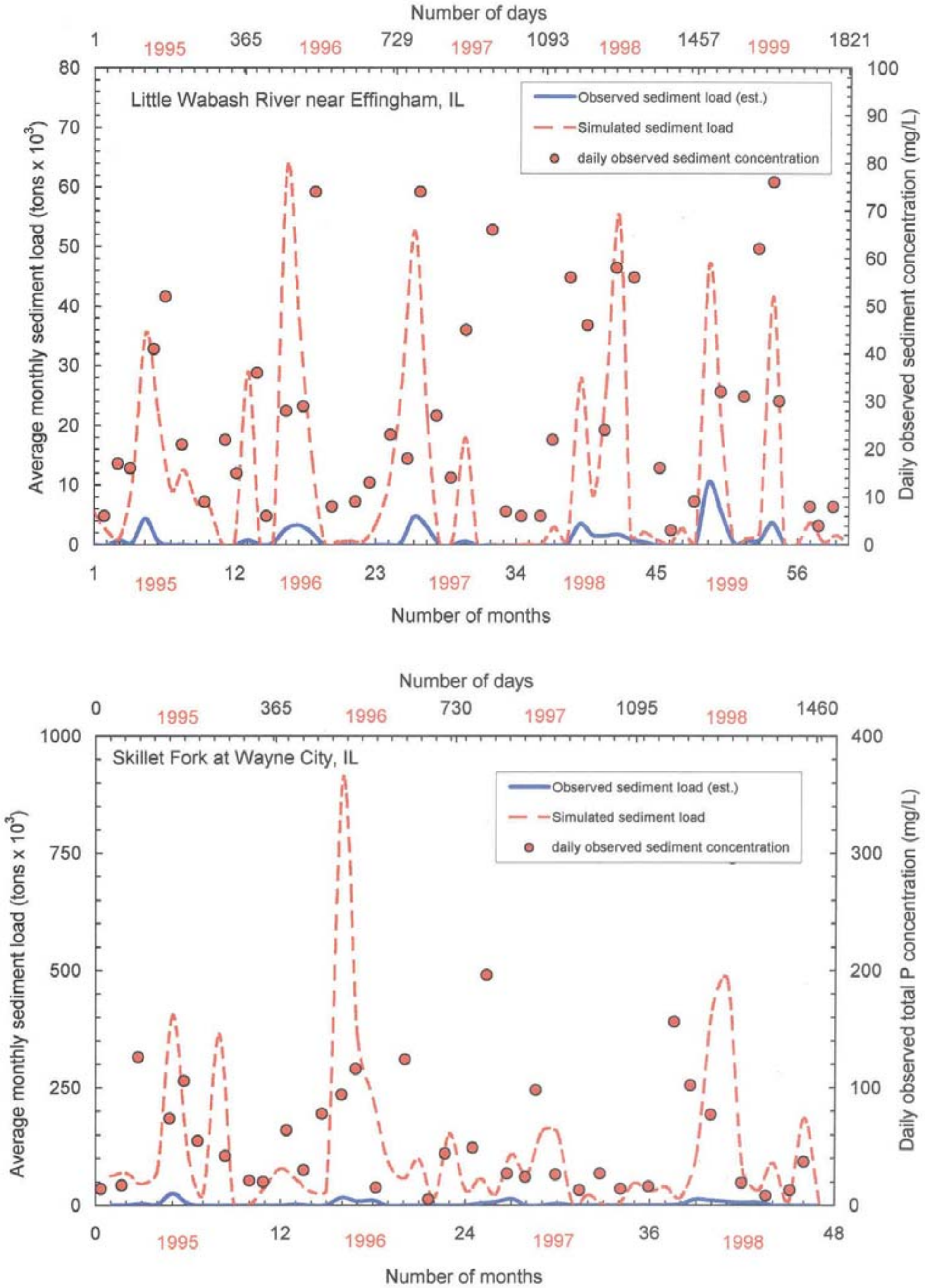


Figure 12. Simulated and observed monthly sediment loads and instantaneous sediment concentrations at Effingham and Wayne City

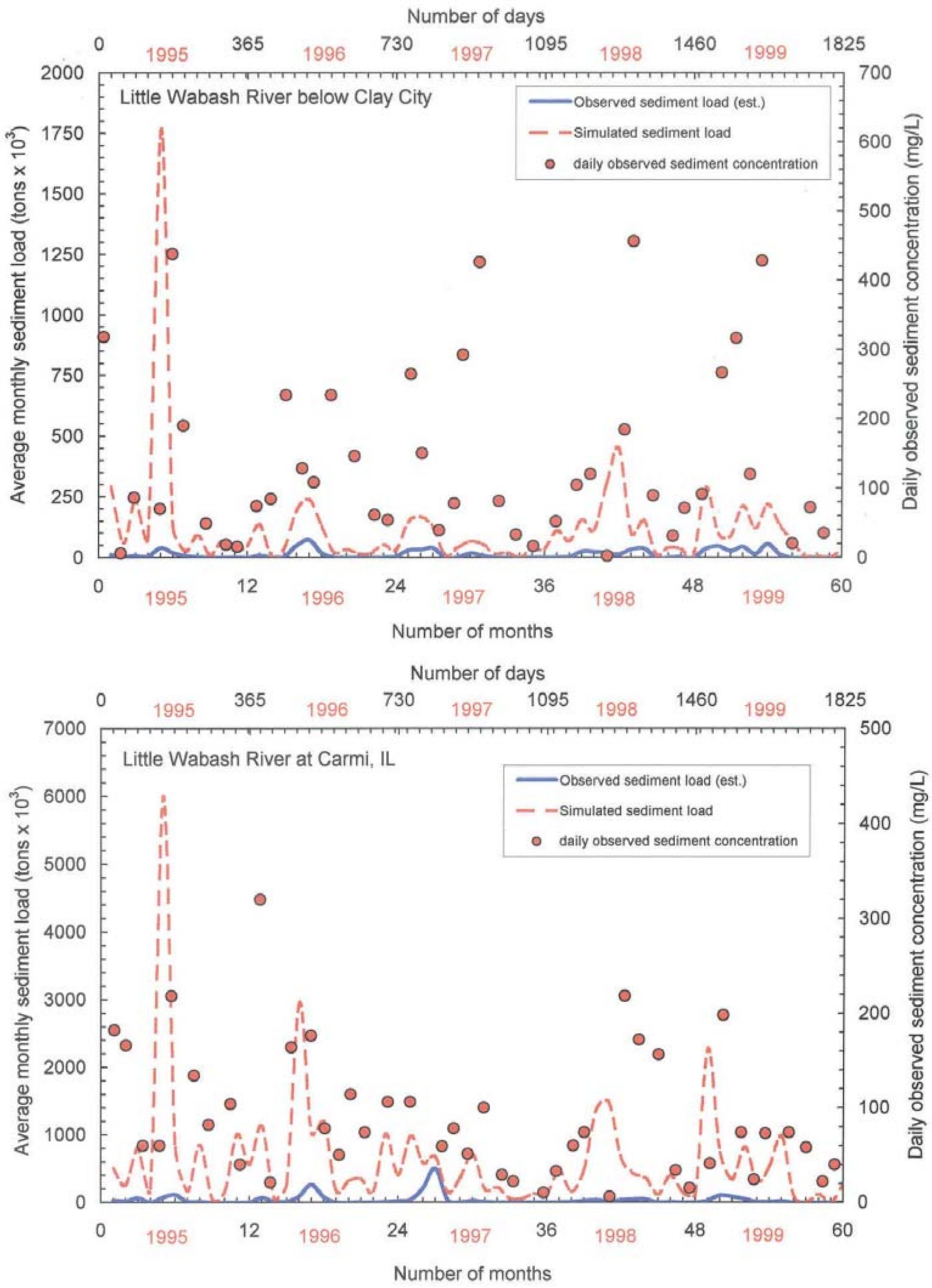


Figure 13. Simulated and observed monthly sediment loads and instantaneous sediment concentrations at Clay City and Carmi

The observed sediment loads were estimated from the instantaneous concentration measurements and observed flows. The concentrations and flows at all of the stations were found to have poor or no correlation. Therefore the observed concentrations were interpolated to get daily concentrations, which are multiplied by the respective daily observed flows to compute daily observed loads. By summing the daily loads during a month, monthly observed loads were computed. The same procedure was used to compute monthly loads of total P, nitrate-N, ammonia-N, and TKN presented later.

As may be seen in Figures 12 and 13, the model grossly overpredicted sediment loads at all the stations. Adjustments of the sediment parameters and inclusion of the sediment-retaining structures into the model are necessary to improve predictions.

Continuous Simulation of Total Phosphorous

Figures 14-15 compare simulated and observed monthly total P loads (tons) at Effingham, Wayne City, Clay City, and Carmi, for the period 1995-1999, along with irregularly and instantaneously measured (IEPA, 2003) total P concentrations (mg L^{-1}). The observed loads were estimated from the instantaneous concentration measurements and daily observed flows following the same procedure as the sediment load estimations, described above.

Unlike the sediment load simulations, total P simulations were much better (Figures 14 and 15). Predictions at Effingham, the smallest drainage area, are the best (Figure 14); mixed at Wayne City, the next larger drainage area (Figure 14), and gross underpredictions at Clay City and Carmi (Figure 15), large drainage areas. Similar to sediment, adjustments of model parameters and inclusion of the sediment-retaining structures into the model are necessary to improve predictions of total P. Specifically, the Clay City and Carmi sites were not rivers during the last Ice Age but lakes. These riverside glacial lake sediments would be more P- and N-rich than highly weathered, relatively nutrient-poor upland soils mantling most of the watershed used in the model, thus causing the model to underestimate total P and total N. Riverside sampling of glacial lake sediments and their derived soils would be necessary to improve model performance for these reaches.

Continuous Simulation of Nitrate-Nitrogen

Figures 16-17 compare simulated and observed monthly nitrate- (sum of nitrite or NO_2 and nitrate or NO_3) N loads (tons) at Effingham, Wayne City, Clay City, and Carmi, for the period 1995-1999 along with irregularly and instantaneously measured (IEPA, 2003) nitrate-N concentrations (mg L^{-1}). The observed loads were estimated from the instantaneous concentration measurements and daily observed flows following the same procedure as the sediment load estimations, described above.

Nitrate-N simulations are also mixed at different stations: best at Carmi (Figure 17), near the watershed outlet, somewhat reasonable at Wayne City and Clay City (Figures 16 and 17), and gross underpredictions at Effingham (Figure 16). Similar to the other water quality variables, adjustment of model parameters and refinement of input data are necessary to improve predictions of nitrate-N. Sampling of tile drains, ditches, and soil-N would help improve model performance here.

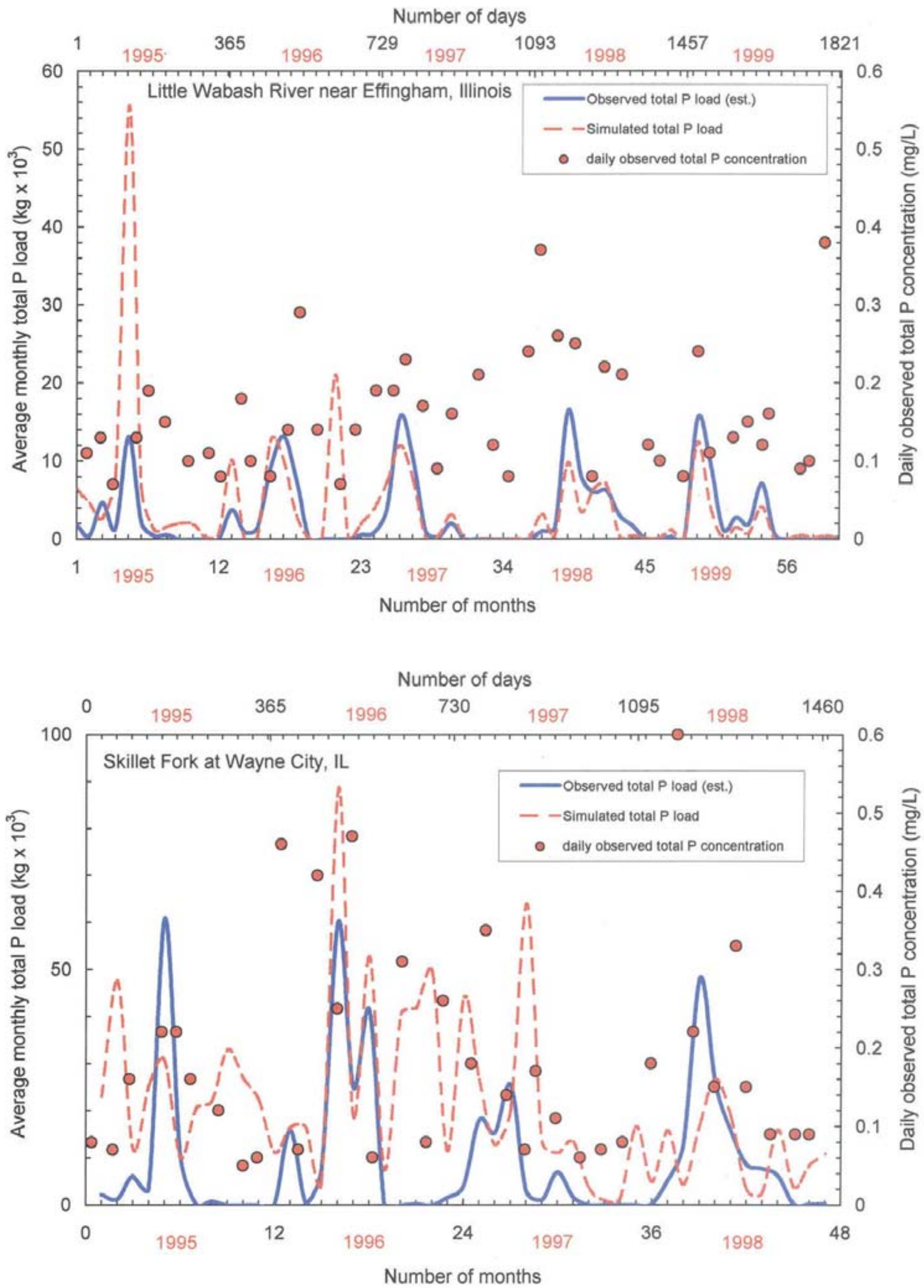


Figure 14. Simulated and observed monthly total phosphorous loads and instantaneous total phosphorous concentrations at Effingham and Wayne City

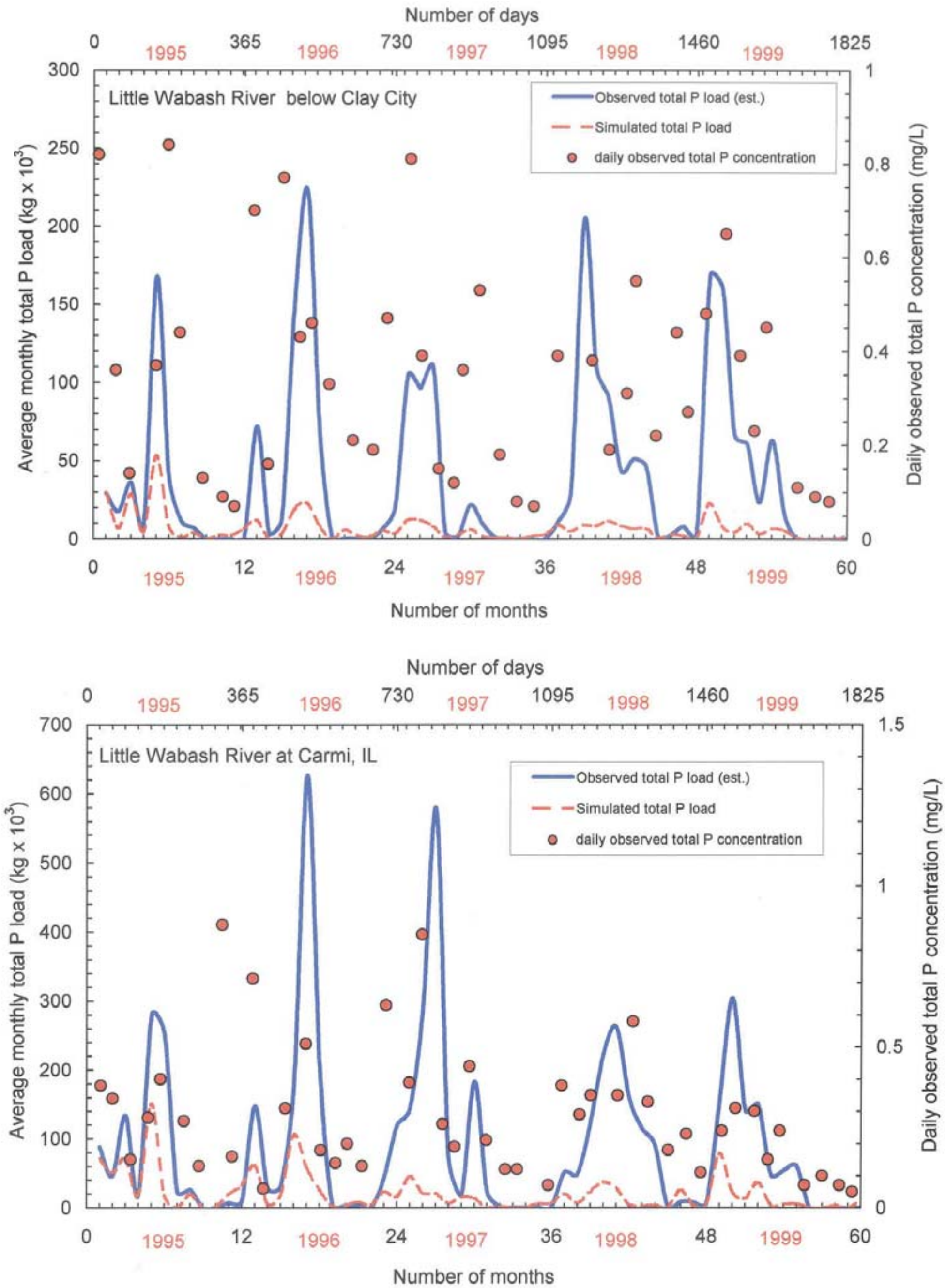


Figure 15. Simulated and observed monthly total phosphorous loads and instantaneous total phosphorous concentrations at Clay City and Carmi

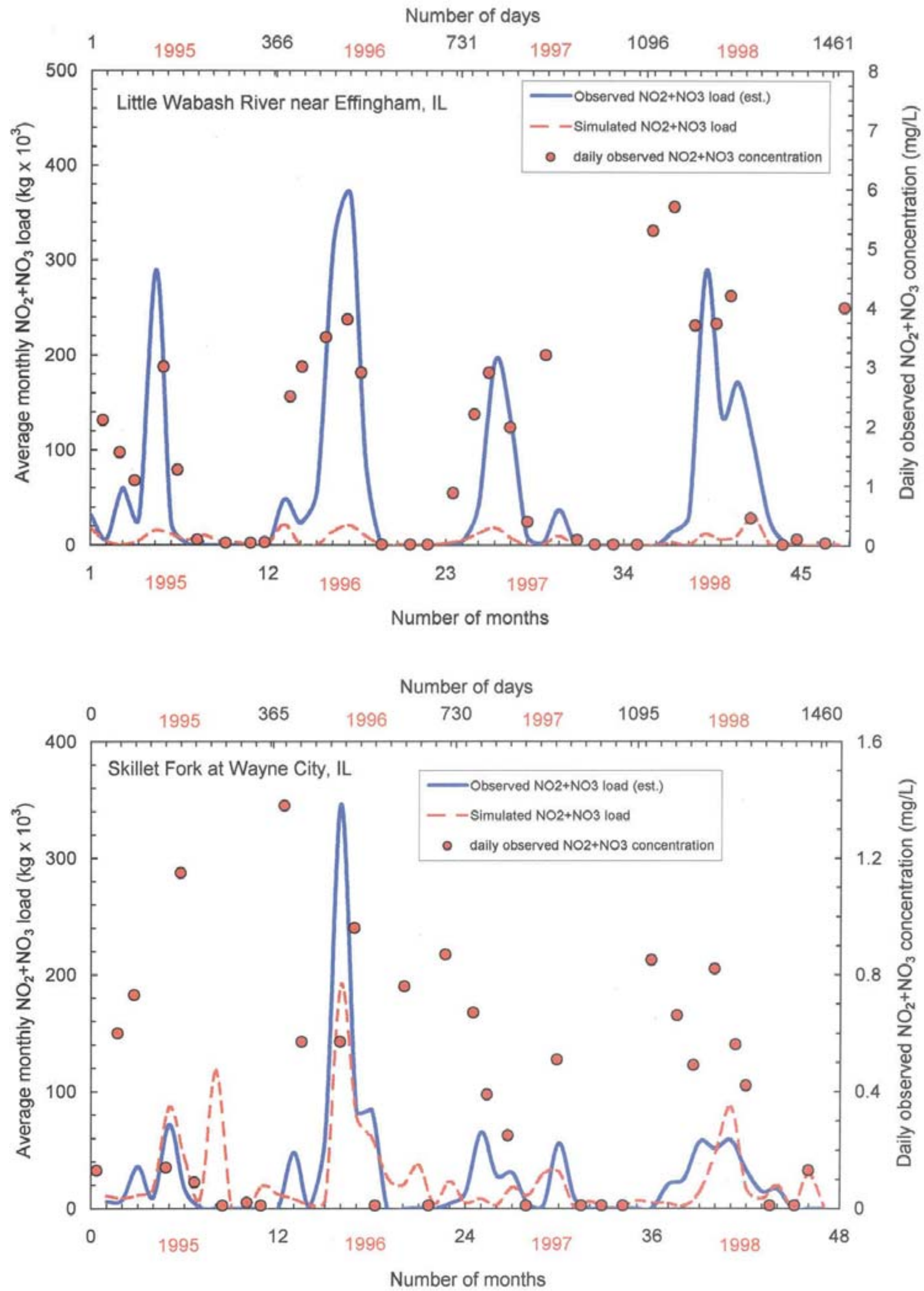


Figure 16. Simulated and observed monthly nitrate-nitrogen loads and instantaneous nitrate-nitrogen concentrations at Effingham and Wayne City

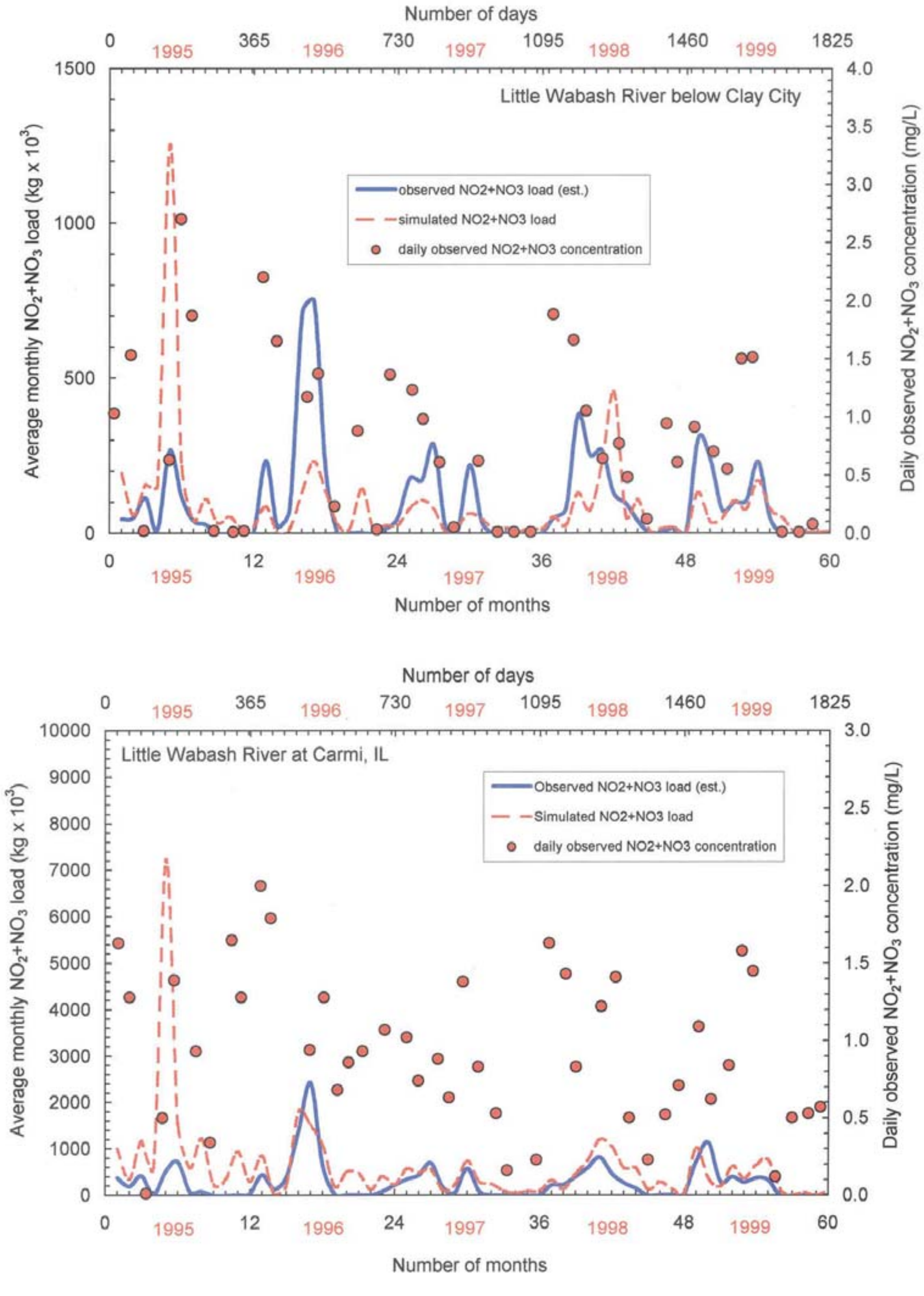


Figure 17. Simulated and observed monthly nitrate-nitrogen loads and instantaneous nitrate-nitrogen concentrations at Clay City and Carmi

Continuous Simulation of Ammonia-Nitrogen

Figures 18-19 compare simulated and observed monthly ammonia-N loads (tons) at Effingham, Wayne City, Clay City, and Carmi, for the period 1995-1999, along with irregularly and instantaneously measured (IEPA, 2003) ammonia-N concentrations (mg L^{-1}). The observed loads were estimated from the instantaneous concentration measurements and daily observed flows following the same procedure as the sediment load estimations, described above.

Ammonia-N simulations are also mixed at different stations: best at Clay City (Figure 19), somewhat reasonable at Carmi and Clay (Figure 19), mixed at Effingham (Figure 18), and gross overpredictions at Wayne City (Figure 18). Similar to the other water quality variables, adjustments of model parameters and refinement of input data are necessary to improve predictions of ammonia-N.

Continuous Simulation of Total Kjeldahl Nitrogen

Total Kjeldahl N (TKN) measurements were available only at the Carmi station, near the watershed outlet (IEPA, 2003) and, therefore, comparisons of simulated and observed TKN loads were made only at this station. Figure 20 shows comparisons of simulated and observed monthly TKN loads (tons) at Carmi, for the period 1995-1999, along with irregularly and instantaneously measured (IEPA, 2003) TKN concentrations (mg L^{-1}). The observed loads were estimated from the instantaneous concentration measurements and daily observed flows following the same procedure as the sediment load estimations, described above.

The TKN simulations at Carmi were found reasonable, if not somewhat underestimated (Figure 20). As with total P, adjustments of model parameters and refinement of input data would improve the predictions of TKN.

Storm-Event Modeling

Each of the 88 subwatersheds of the Little Wabash River watershed, used in the SWAT simulations, were further subdivided into two overland planes and one channel segment ─ totaling 176 overland planes and 88 channel segments (Figure 21). Areas, lengths, widths, and representative slopes of the overland planes, and channel lengths, slopes, widths, and depths were obtained from the same BASINS GIS data as used in the SWAT. Channel widths and depths given by these GIS data were used to develop relationships of wetted perimeters versus cross-sectional areas (a and b in Equation 54).

In this project, the uppermost station Effingham was chosen to test the proposed storm-event hydrology model as an initial attempt. Therefore, 15-minute precipitation data at Effingham and Mason, the two closest raingages to the watershed above Effingham (620 km^2), were retrieved from NCDC. Fifteen-minute flow data at the stream gage near Effingham were obtained from the USGS.

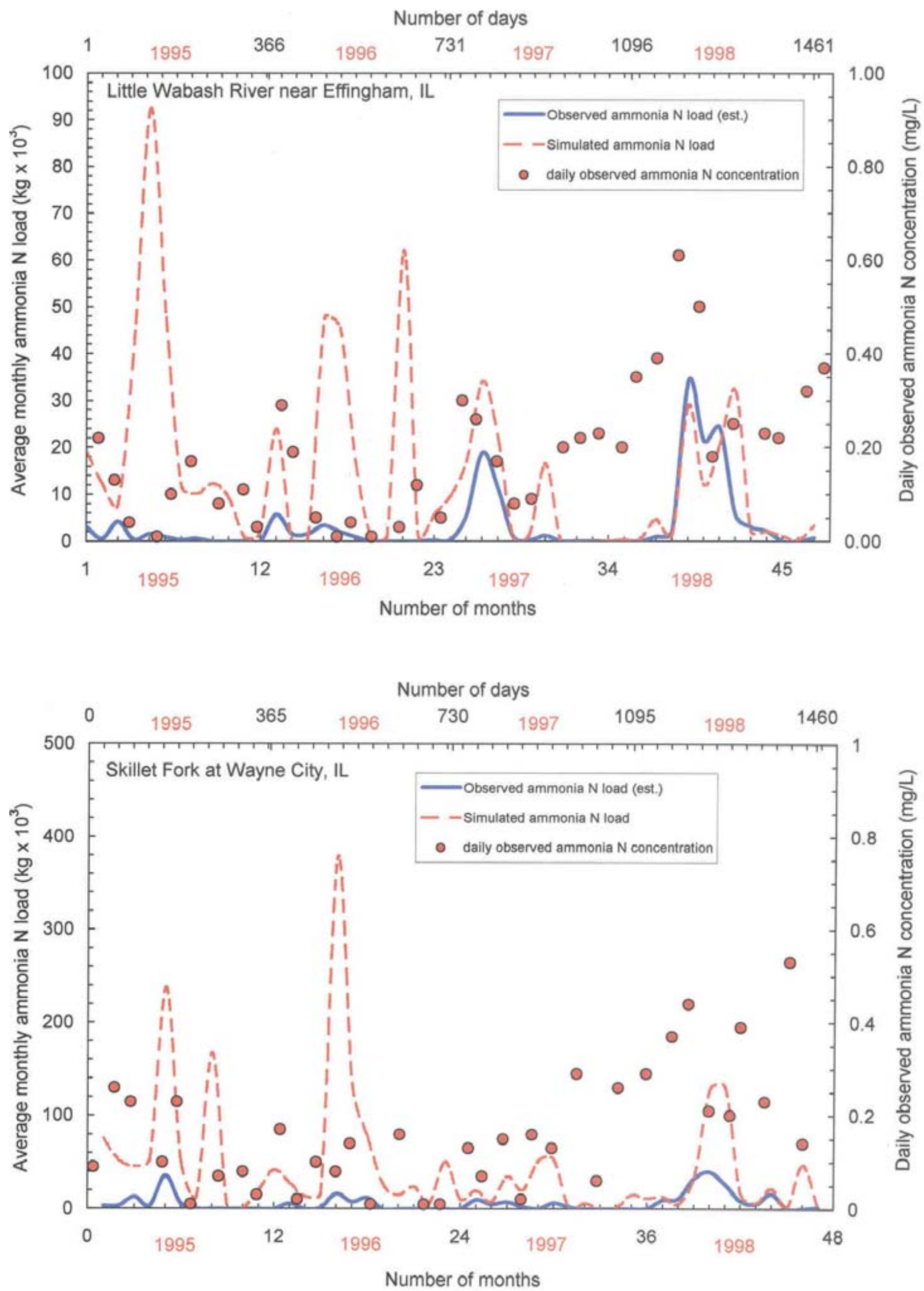
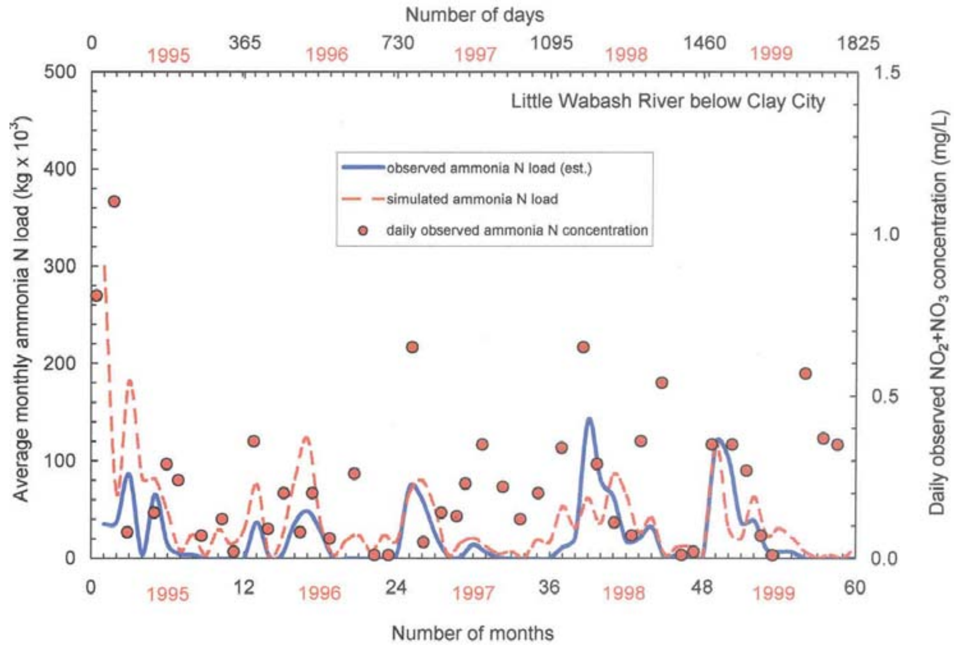


Figure 18. Simulated and observed monthly ammonia-nitrogen loads and instantaneous ammonia-nitrogen concentrations at Effingham and Wayne City



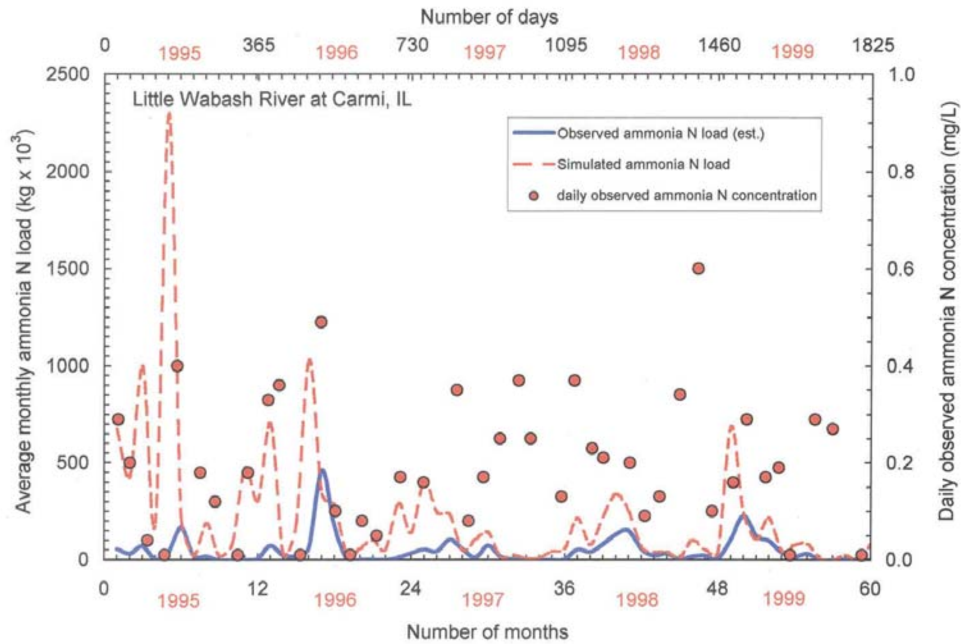


Figure 19. Simulated and observed monthly ammonia-nitrogen loads and instantaneous ammonia-nitrogen concentrations at Clay City and Carmi

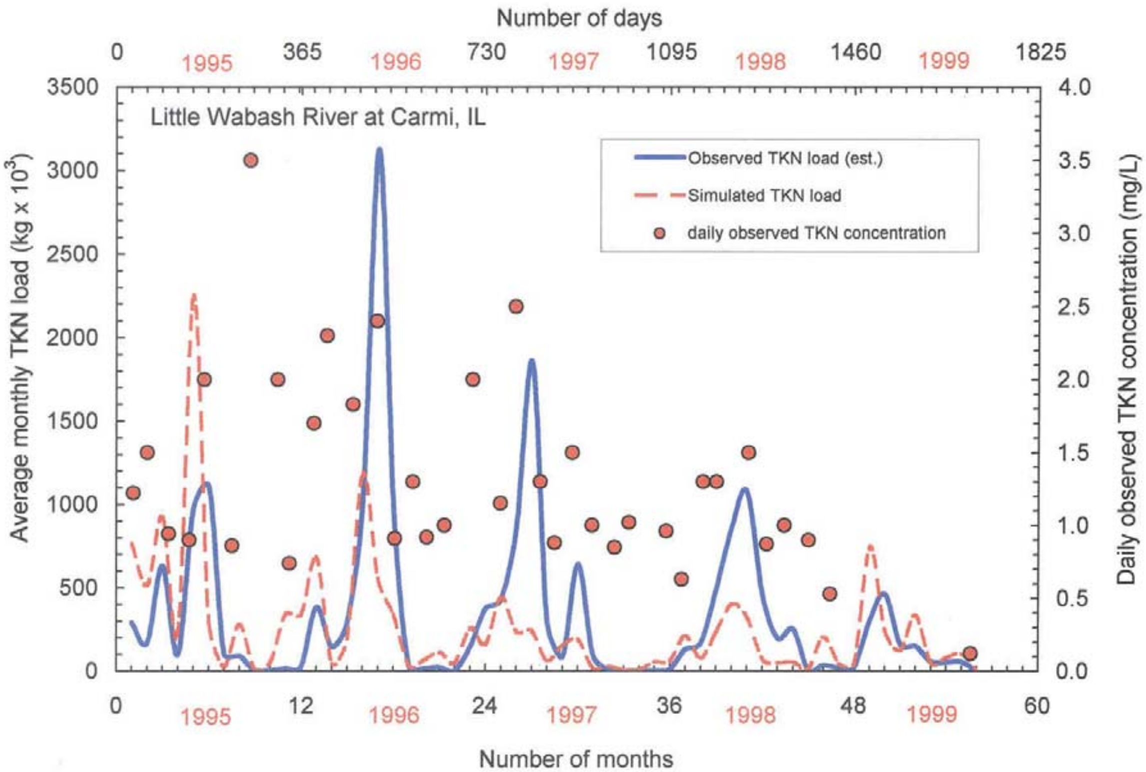


Figure 20. Simulated and observed monthly total Kjeldahl nitrogen loads and instantaneous total Kjeldahl nitrogen concentrations at Carmi

Hydrologic Simulations

A major mid-May 1995 storm event was used to test the storm-event hydrology model. Values for the three parameters – (1) SCS runoff curve number *CN*, (2) effective lateral saturated hydraulic conductivity *ELSHC*, and (3) Manning's roughness coefficient *n* for the overland planes and for the channel segments – were taken from the SWAT calibration. An overland plane may have several SWAT HRUs. Therefore, area-based weighted averages were used. The *ELSHC* values were chosen from SWAT's saturated hydraulic conductivities K_{sat} (*SOL_K*).

Figure 22(a) shows comparisons of observed and simulated hydrographs along with daily rainfall and 15-minute rainfall intensity data from May 1 to June 5, 1995 (35 days). In addition to the major storm event during days 15-20, as shown in Figure 22(a), there were smaller events before and after, which were also simulated. Figure 22(a) shows comparisons of observed daily flows with SWAT daily flow simulations on the Little Wabash River at Effingham in addition to the comparisons of observed 15-minute flows with 15-minute storm event flow simulations. Table 1 gives the simulated and observed peak flows, time-to-peak flows, runoff volumes, and percent differences (errors) of the respective observed and simulated values.

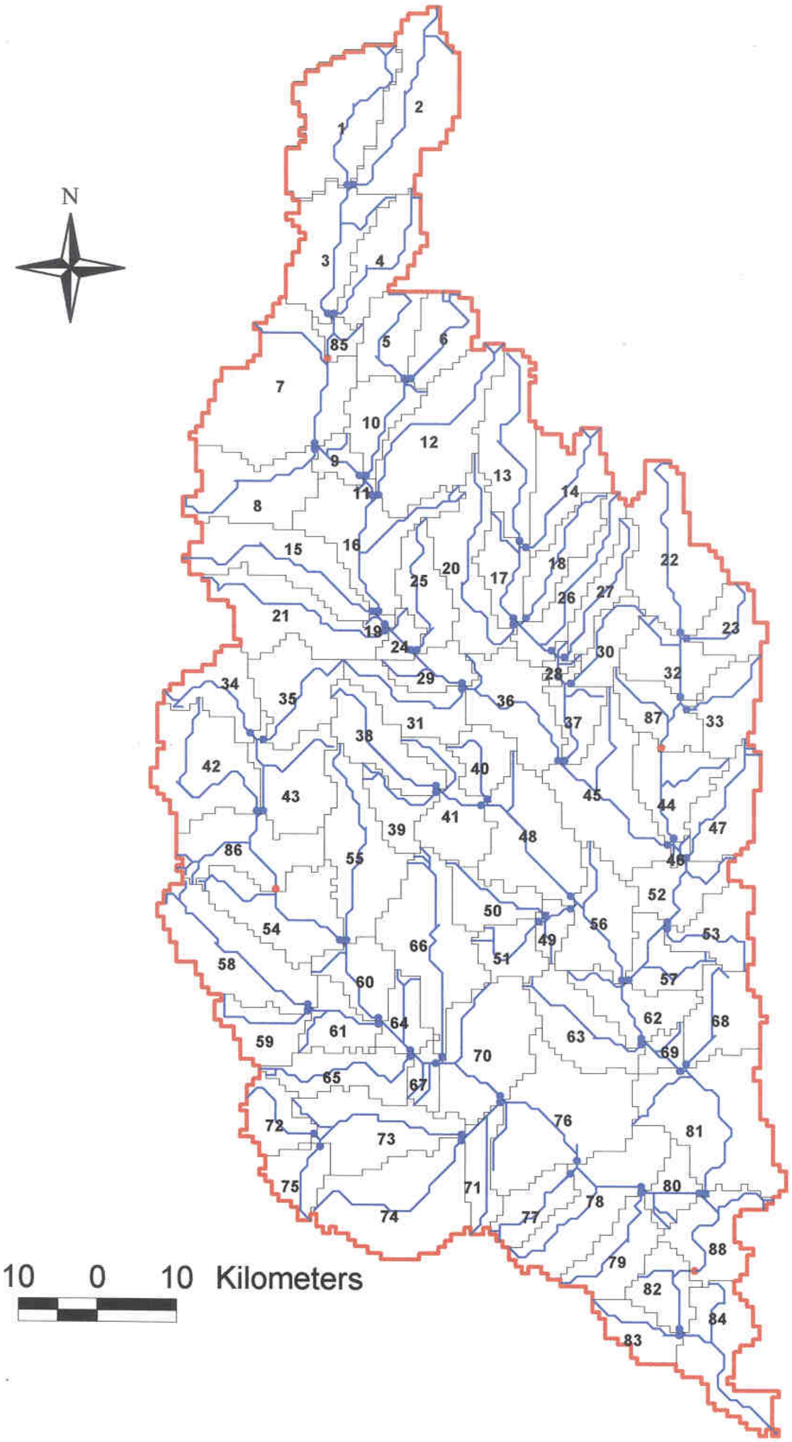


Figure 21. Little Wabash River watershed divided into 176 overland planes and 88 channel segments

(a)

(b)

(c)

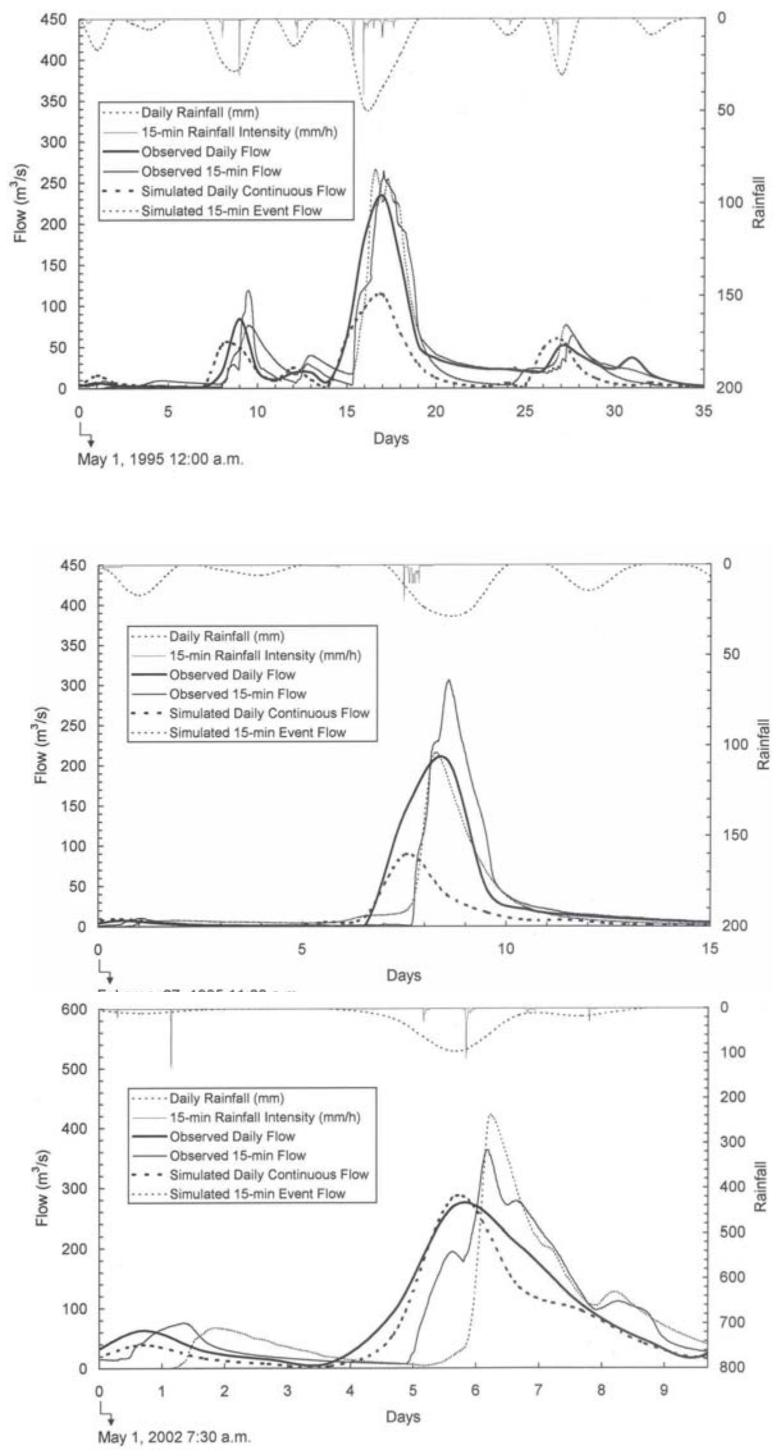


Figure 22. Daily and 15-minute storm rainfall and flows on the Little Wabash River at Effingham: (a) May 1-June 5, 1995; (b) February 27-March 14, 1995; and (c) May 1-11, 2002

Table 1. Comparisons of Observed and Simulated SWAT Daily Continuous and 15-Minute Storm Peak Flows, Time to Peak Flows, and Runoff Volumes on the Little Wabash River at Effingham

Period	*Parameter*	*SWAT daily continuous simulation*			*15-min storm-event simulation*		
	(unit)	*Simulated*	*Observed*	*% Error*	*Simulated*	*Observed*	*% Error*
May 1-June 5, 1995	Peak flow (m^3/s)	114	234	-51	266	264	1
	Time-to-peak flow (days)	17	17	0	17	17	0
	Runoff volume (ha-m)	6,719	11,140	-40	10,393	10,990	-5
February 27-March 14, 1995	Peak flow (m^3/s)	90	207	-57	216	306	-29
	Time-to-peak flow (days)	8	9	-11	8	9	-11
	Runoff volume (ha-m)	1,697	4,105	-59	3,225	4,087	-21
May 1-May 11, 2002	Peak flow (m^3/s)	286	271	6	421	362	16
	Time-to-peak flow (days)	7	6	17	6	6	0
	Runoff volume (ha-m)	6,150	7,926	-22	6,438	7,227	-11

As shown in Figure 22(a) and Table 1, the storm-event model predicted the peak flow and runoff volume for the simulation period better than the continuous model (SWAT) with daily time steps. The storm-event model results are more detailed than the continuous daily results. It shows the precise time of arrival of the peak flow – 15 minute resolution in this case. In this application (Figure 22a), the storm event simulations predicted the intense storm high flows (Days 15-20), much better than the daily continuous simulations. The storm-event-simulated peak flow 266 m^3s^{-1} has a deviation of 1 percent from its observed value 264 m^3s^{-1}. The simulated daily peak flow from the continuous model (114 m^3s^{-1}) is 51 percent underpredicted from the observed daily peak flow of 234 m^3s^{-1}, which is actually 57 percent less than the 15-minute observed peak flow of 264 m^3s^{-1}, a more realistic peak flow to be concerned with for flood warning, protection, or prevention.

Recalibration of the three parameters for the storm event model was not necessary. Calibration of these parameters in the SWAT was sufficient. Therefore, these parameters are interchangeable, and the models are compatible and complementary, which is unique. As a result, the storm-event runs presented here are all validation runs.

Using the same parameter values, the storm-event model was run for two other storm periods: February 27-March 14, 1995 and May 1-May 11, 2002. Similar comparisons are presented in Figures 22(b) and 22(c), and Table 1. Performances of the models during these two storm periods are similar to the May-June 1995 storm, as discussed above, but the SWAT also predicted daily flows exceptionally well during the May 2002 storm period (Figure 22c).

A shown in Figure 22 and Table 1 from the three storm simulations, storm-event model 15-minute peak-flow errors were 1, 29, and 16 percent; volume errors were 5, 21, and 11 percent (all underpredictions); and time-to-peak flow errors were 0, 11, and 0 percent, respectively. Similarly, the SWAT daily flow prediction errors were: peak flow of 51, 57, and 6 percent; volume of 40, 59, and 22 percent (all underpredictions), and time-to-peak flow of 0, 11, and 17 percent, respectively.

Water Assessments at the Small Water Supply Intakes

The calibrated and validated SWAT on the Little Wabash River watershed provides a tool to evaluate surface water quantities and qualities throughout the watershed. This includes small public water supply intakes under existing conditions, as well as, under future natural or human-made conditions, such as changes of climate, land use, and/or best management practices (BMPs).

The SWAT continuous model results generated in this study provide long-term (daily, monthly, and yearly) assessments of water quantities and qualities at the outlet of each of the 88 sub-watersheds of the Little Wabash River watershed (Figure 7) under existing (present) conditions. For example, Figures 8-11 shows the monthly average flows at the four gaging stations – Effingham, Wayne City, Clay City, and Carmi – for the period 1995-2002. Similarly, Figures 12-19 show monthly loads of sediment, total P, nitrate-N, and ammonia-N at those four stations, and Figure 20 shows monthly loads of TKN at Carmi for the period 1995-1999. The model also provides concentrations of the constituents.

For water quantity and quality evaluations at intakes of the seven small water supplies in the Little Wabash River watershed, yearly averages of precipitation, runoff, and loads of sediment, total P, nitrate-N, ammonia-N, and TKN at each of the intakes under existing conditions are presented in Table 2 from model results. The Wayne City intake coincides with the Wayne City USGS gaging station (Figure 1). Therefore, values of the above parameters (except TKN) calculated from concentration measurements at Wayne City, as described earlier, are also presented in Table 2 (in parentheses) for comparisons. As may be seen, simulated runoff volume and loads of total P and nitrate-N match very well with the observed values. As discussed earlier, sediment and ammonia N are grossly overpredicted. There was no measurement of TKN at the Wayne City station. At Carmi, simulated TKN is 310 kg km^{-2} as compared to 480 kg km^{-2} estimated from observed concentrations (35 percent underprediction).

Other model results can be used to extend the above assessments and detect any existing or potential water quantity or quality problems. In a similar manner, the storm-event hydrology model can be further developed to assess water quantities and qualities during intense or design storm events, which are critical in causing flooding and have disproportionately large influence on sediment and chemical loads. For example, Figure 22 shows storm-event high-flow predictions at Effingham during three intense historical storms in 1995 and 2002. The model results are useful in designing flood and sediment control structures, and planning management practices for keeping concentrations or loads of sediment, chemicals, and overall water quality at acceptable levels.

Evaluation of Watershed Management and BMPs

The calibrated and validated SWAT continuous model of the Little Wabash River watershed can evaluate long-term impacts of agricultural management, including tillage, planting, fertilizer, pesticide, grazing, harvest, kill, filter strips, irrigation, tile drainage, impoundments, water transfer, consumptive water use, and point-source loadings within the watershed through its built-in menu system.

Table 2. Annual Average Water Quantity and Quality Variables at the Little Wabash River Watershed's Small Water Supply Intakes

Variables	*Neoga*	*Altamont*	*Olney*	*Wayne City*	*Flora*	*Clay City*	*Fairfield*
Drainage area (km^2)	175	3	37	1217	1865	2193	4737
Precipitation (mm)	970	917	1278	1288	1114	1144	1133
Runoff (mm)	545	461	550	415 (376)	332	437	435
Sediment (ton/km^2)	1255	3275	1262	617 (35)	755	730	674
Total P (kg/km^2)	685	945	407	95 (92)	80	31	39
Nitrate-N (kg/km^2)	330	1162	124	242 (262)	802	970	847
Ammonia-N (kg/km^2)	583	2707	311	451 (62)	356	258	195
TKN (kg/km^2)	3884	5357	1624	928	542	395	294

Note: Numbers in parentheses were estimated from observed concentrations.

Various management options may be tested and implemented in the Little Wabash River watershed to improve its water quality. For example, in a nonpoint-source phosphorous management plan for the Altamont New Reservoir supplying water to the Altamont small public water supply system, the IEPA (2004b) recommended wetlands, filter strips, conservation tillage practices, and nutrient management. These practices can be entered into the model, and their impacts on water quality at the Altamont New Reservoir and other locations throughout the watershed, including at the remaining six small public water supply intakes, can be evaluated. In consultation with the IEPA and local, small water supply operators, more practices can be evaluated to improve water quantity and quality, especially if water quality data are collected at water supply intakes.

In order to evaluate management practices effectively or meaningfully, the model needs to be calibrated and validated further with data sets more complete than now exist. The data used to calibrate and validate the SWAT on the 8,400-km^2 watershed were very limited: data at only four gaging stations and water quality measurements made at sporadic time intervals, approximately once a month. Only one of the gaging stations was a water supply intake. Uncertainty of the model results based on the limited calibration and validation is of serious concern. Therefore, no attempt was made in this study to evaluate any management practices.

Unfortunately, water supply systems no longer determine water quality at their intakes. Furthermore, government water quality monitoring that may be used in their stead is decreasing. If water intake water quality sampling is resumed, or equivalent monitoring established, BMPs could be evaluated.

High-Resolution Precipitation Data

Resolution of precipitation data input is becoming a concern in watershed modeling. Precipitation is a major driving force of watershed processes and is highly variable on spatial and temporal scales. Current watershed models, including the SWAT, are unable to capture the distributions accurately because of limited resolution of precipitation data inputs from discrete, and usually sparse, raingages. On the other hand, recent availability of the National Oceanic and Atmospheric Administration (NOAA) multi-sensor (radar plus gauge) hourly, 4-km grid precipitation analysis and progress in mesoscale Regional Climate Model (RCM) simulations (Liang et al., 2004a,b) brings an unprecedented opportunity for substantial enhancement of the temporal and spatial resolutions of watershed model precipitation input.

One of the principal enhancements necessary for the development of an enhanced, comprehensive SWAT is addition of an interface to incorporate high-resolution precipitation data such as the NOAA near real-time, multi-sensor (radar plus gauge) data at hourly, 4-km grid and mesoscale RCM simulations (Liang et al., 2004a,b). Such enhancement will allow SWAT to simulate intense storm events more accurately. These events generate and transport disproportionately large amounts of sediment and chemicals.

Before making an effort to incorporate the NOAA high-resolution precipitation data, data accuracy with respect to gage data was investigated. The NOAA daily radar and gage precipitation data at 7,235 stations around the continental United States were compared. Figure 23 shows the comparisons for year 2001. As may be seen in Figure 23, the comparisons are widely scattered and radar precipitation were generally underestimated with respect to gage measurements.

More research and efforts are necessary to improve radar estimates of precipitation before using those to improve modeling the Little Wabash watershed. This was beyond the scope of this study and, however, must be continued as future research.

Finer Spatial Resolution of Watershed

Along with the high-resolution precipitation data, one of the goals was to use finer spatial resolution in the SWAT model for the Little Wabash River watershed. Figure 24a shows the stream network captured by the current resolution adopted in the model with 88 subwatersheds (Figure 7). As may be compared with Figure 1, the stream network from the current spatial resolution shown in Figure 24a does not capture all of the permanent streams. However, finer resolution of the Little Wabash River watershed and its 449 subwatersheds would capture all the permanent and ephemeral streams, as shown in Figures 24b. Such resolution would allow accurate capture of point sources and sinks, storm-event runoff, sediment, and contaminants, provided precipitation and other key data are available at that resolution or better.

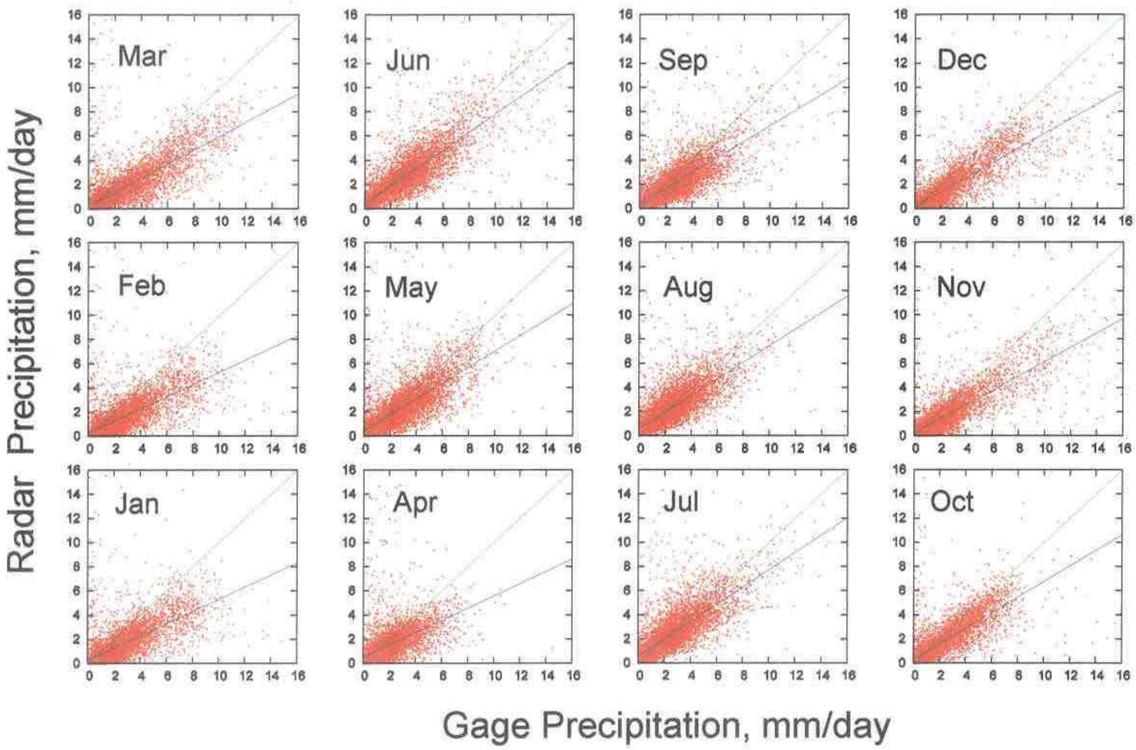


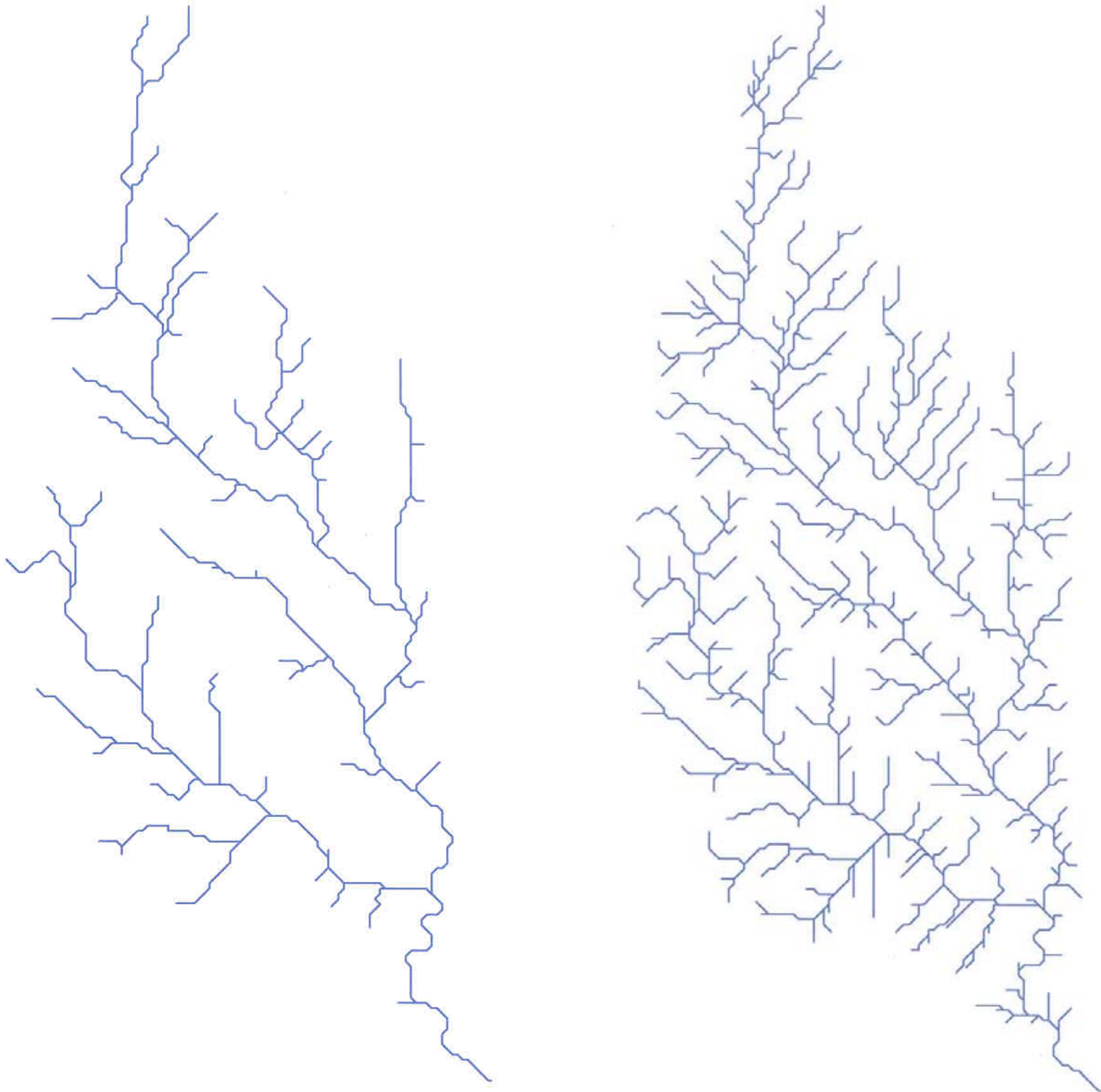
Figure 23. Comparisons of NOAA radar and gage precipitation data of the continental United States for the year 2001

As discussed above, higher resolution precipitation data are not ready to be used in improved modeling. Therefore, use of finer resolution data in the Little Wabash River watershed modeling is a subject for future work.

User-Friendly Tool for Small Water Supply Managers

The SWAT, a well-documented and user-friendly tool, is available for use on the USEPA website: http://www.epa.gov/waterscience/basins/bsnsdocs.html#swat (accessed July 11, 2006). The storm-event hydrologic model selected and tested here for enhancement of the SWAT uses interchangeable parameters from the SWAT, and watershed characteristics data are processed directly from SWAT data. Therefore, the storm-event hydrologic model is also user friendly and can be developed into a more user-friendly tool as part of future research.

The IEPA Division of Public Water Supplies officials (Richard Cobb, Anthony Dulka, and Wade Boring) on August 25, 2005 noted that water supply managers would use consultants to run the SWAT and its enhanced storm-event models for them. Thus, consultants working for the public water supplies require training, which could be a part of future Extension work. Mass production of a CD-ROM with the modeling tool for distribution to water supply managers is not warranted.



(a) **(b)**

Figure 24. Little Wabash River watershed stream network: (a) 88 subwatershed resolution, (b) 449 subwatershed resolution

Conclusions

Water quantity and quality at surface water supply intakes are of serious concern nationwide. Existing studies have focused on characterization and assessments of public water supplies. However, there exists no research on evaluation of water quantities and qualities at surface water supply intakes and development of comprehensive watershed modeling tools to do so. Furthermore, no existing model is capable of comprehensively simulating all of the hydrologic, upland soil and streambank erosion, sediment transport, and fate and transport of nutrient and pesticide processes that are necessary to comprehensively assess the water quantity and quality problems and help make the best management decisions to eliminate or minimize them. The current study, a premier study, has addressed these issues extensively and made significant advancements by selecting a suitable watershed model, making advancements in enhancing it, and testing it on a watershed in Illinois selected for the study.

Recent review of leading watershed-scale hydrologic and nonpoint-source pollution models indicated various strengths and weaknesses in existing models. It found that the SWAT was the most promising long-term continuous simulation model. The SWAT, currently the most established and commonly used model, is an excellent candidate for enhancement into a more comprehensive model with storm-event simulations. In this study, a compatible and complementary storm-event hydrologic model was selected to enhance the SWAT's storm-event hydrologic simulations and tested it on the selected watershed along with the SWAT's long-term continuous simulations of hydrology as well as sediment and nutrients.

The 8,400 km^2 Little Wabash River watershed in southeastern Illinois was chosen because its water supply and watershed attributes are favorable for this study. It has seven small (population < 10,000) and three large public surface water supply systems serving communities in it. Every major stream and river mile of the watershed has impairment from sediment, nutrient enrichment, and pesticides. The developmental history of the watershed shows that watershed growth was retarded by the low level of water resources development resulting in a very rural and sparsely populated agricultural watershed, the least developed major watershed in Illinois.

The long-term continuous hydrologic simulations in the Little Wabash River watershed were evaluated by comparing simulated monthly average flows with monthly observed flows at four evenly distributed gaging stations. The coefficient of determination and Nash-Sutcliffe coefficient for individual years, cumulatively for the 5-year calibration period, and cumulatively for the entire 8-year simulation period were calculated. These statistical values for three of the four stations – Effingham, Clay City, and Carmi – were above or near 0.5 and, therefore, had reasonable overall predictions of monthly flows. Although the overall statistical values for Wayne City were low (0.26-0.37), values for 5 of the 8 individual years were above or near 0.5, with 0.91 as the highest value. Therefore, overall model performance at Wayne City in simulating monthly flows also can be considered reasonable. These spatially and temporally distributed statistical evaluations were useful and unique. To our knowledge, no other modeling study has conducted a similar level of comparative evaluations and analyses.

From visual comparisons of simulated and observed monthly flow hydrographs at the four stations, it can be seen that the model reasonably predicted monthly average flows throughout the watershed although there were some discrepancies, especially most of the peak flows (underpredictions). Therefore, the SWAT is a promising long-term continuous simulation model, which needs enhancement in storm-event simulations to improve its peak flow predictions.

Calibration of the water quality component of the SWAT was based on limited data, estimated from sporadic concentration measurements taken approximately once a month at only four stations in the entire 8,400-km^2 watershed.

Visual comparisons of simulated and observed (estimated) monthly sediment loads at each of the four monitoring stations during the 5-year period showed gross overpredictions. However, predictions of monthly total P, nitrate-N, and ammonia-N loads at the four stations were mixed and much better. Monthly TKN load predictions at Carmi near the watershed outlet, the only station having TKN data, were found reasonable. Sampling of Ice Age near-stream lake-bottom sediments for the lower river stretches, ditches, and tile drains above Effingham, adjustment of model parameters, and inclusion of conservation structures throughout the watershed are necessary to improve predictions of these water quality parameters.

Using values of three hydrologic parameters from the SWAT calibration, storm-event hydrologic simulations of three separate storms during the 8-year period resulted in comparable flow hydrographs with observed flows at Effingham, the uppermost station: peak flow errors of 1, 29, and 16 percent; volume errors of 5, 21, and 11 percent; and time-to-peak flow errors of 0, 11, and 0 percent.

Recalibration of the three parameters (*CN*, *n*, and *ELSHC*) for the storm event model was not necessary. Calibration of these parameters in the SWAT was sufficient. Therefore, these parameters are interchangeable, and the models are compatible and complementary, which is unique.

The SWAT's daily flow simulations during the above three storms were found mixed. It underpredicted daily peak flows in two of the storms by 51 and 57 percent, but performed well for one of the storms, for which daily peak flow was overpredicted by 6 percent.

Comparisons of storm-event flow hydrographs at 15-minute intervals with SWAT daily flow hydrograph, along with their respective (15-minute and daily) observed hydrographs, revealed that storm-event hydrologic simulations predicted more accurate flows, especially high and peak flows, during storm events than the SWAT daily continuous simulations.

In addition to using a smaller time step, a unique combination of the runoff curve number method for rainfall excess computations and kinematic wave equations for flow routing and physical bases of these routines were responsible for predicting more accurate high and peak flows during intense storms, and addition of these routines to the SWAT would be a significant enhancement.

For water quantity and quality evaluations at intakes of the seven small water supplies in the Little Wabash River watershed, yearly averages of precipitation, runoff, and loads of sediment, total P, nitrate-N, ammonia-N, and TKN at each of the intakes under existing conditions were examined from model results.

The calibrated and validated model can be used to evaluate impacts of changes in management practices towards improving or maintaining water quantities and qualities. However, in the present study, calibration of the water quality model was based on limited data. Uncertainty of the water quality results was of a serious concern and, therefore, no attempt was made to evaluate any management practices. Once adequate water quality sampling resumed or equivalent monitoring was established, further calibration of the model and evaluation of management practices could be conducted.

More research and efforts are needed to improve radar estimates of precipitation, such as the NOAA near real-time multi-sensor (radar plus gauge) data at hourly, 4-km grid, and to develop the interface to incorporate those high-resolution precipitation data, or mesoscale Regional Climate Model simulations, for more accurate simulation of intense storms, which generate and transport disproportionately large amounts of pollutants.

Along with high-resolution precipitation data, finer spatial resolution of watershed will be achievable for more realistic simulations as well as capturing point sources, sinks, storm-event runoff, sediment, and contaminants more accurately.

Research needs to continue towards enhancements of the SWAT with storm-event soil erosion, stream bed and bank erosion, and transport of sediment, nutrient, and pesticide simulations, ultimately developing a more comprehensive watershed simulation model.

Uncertainties, resulting from deterministic modeling of natural processes and measurements or observations of data used in modeling, must be considered when using model results in management decisions and policymaking, the subject of future research.

The SWAT, a well-documented and user-friendly tool, is available on the USEPA website: http://www.epa.gov/waterscience/basins/bsnsdocs.html#swat. The storm-event hydrologic model uses interchangeable parameters from the SWAT and watershed characteristics data are processed directly from SWAT data. Therefore, the storm-event hydrology model is also user friendly and can be developed into a more user-friendly tool in future research. Scientific or engineering consultants working for the small public water supply systems require training on using these modeling tools, which can be given as part of continued Extension work.

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