Modeling Flow and Pollutant Transport in a Karst Watershed with SWAT

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Abstract. *Karst hydrology is characterized by multiple springs, sinkholes, and losing streams resulting from acidic water percolating through limestone. These features provide direct connections between surface water and groundwater and increase the risk of groundwater, spring, and stream contamination. Anthropogenic activities (agriculture, tourism, urban and residential areas) accentuate the contamination potentials. The objectives of this article are to present a modification of the Soil and Water Assessment Tool (version 2005) that allows faster percolation through the soil substrate and recharge of the aquifer. This addition was necessary to simulate quick movement of water through vertical conduits that characterize karst topography. The model was calibrated for the James River basin, a large watershed (3,600 km^2) in southwest Missouri. Losing streams were simulated by specifying high soil conductivities in the channels, and sinkholes were simulated as ponds with a high hydraulic conductivity at the bottom. Results indicated that the changes improved the partition of stream flow between surface and return flow. Water quality results indicated that the SWAT model can be used to simulate the frequency of occurrence of pollutant concentrations and daily loads. This case study highlights the possibilities and limitations in modeling flow and water pollutant movement in a karst watershed.*

Keywords. *Fecal coliform, Groundwater, Hydrology, Karst, Modeling, Model performance, Phosphorus, Recharge, SWAT, Watershed.*

Karst results from the dissolution of carbonate rock (limestone and dolomite) by water. The resulting underground and vertical conduits manifest themselves on the surface by familiar elements: caves, holes, springs, and disappearing streams. The word "karst" is the name of the region that is now in Slovenia in which this specific hydrology was first observed and documented during the middle of the 19th century. German and Italian speaking engineers then referred to that region when describing hydrology in the carbonate mountains of Bosnia Herzegovina, Russia, and Italy. Karst became the word by which landforms of carbonate terrain and the associated hydrography were described.

Karst hydrology is schematized by some as a network of conduits along with a porous matrix (Bakalowicz, 2005). It is characterized by high heterogeneity, and it is therefore difficult to generalize what is understood in one karst watershed to another (Bakalowicz, 2005). However, all karst settings have in common that water circulates in large voids under flow conditions that may be similar to those of surface streams under normal conditions. Under flood conditions such that the capacity of these conduits is reached, their functioning becomes one of confined flow conditions.

Karst aquifers represent important water supply sources valued for the quantity of available water and the remarkably good quality of that water. This makes them economically valuable from both water supply and tourism standpoints. Agriculture is also a potential activity, in particular grazing animal agriculture and hay crops because they do not require deep and rich soils as do row crops. Because of the high storage capacity of karst aquifers, their hydraulic buffering function is large: high flow peaks are decreased and delayed, and springs sustain flow during drought. The underground storage of water also results in cooler water temperatures in springfed streams. The resource, however, is fragile: the system of cracks and conduits increases the risk of contamination of groundwater by surface waters because there is no filtering by soil layers before water enters the aquifer. Tourism and agriculture bring additional pressures on these systems, which conflict with a water supply source. Thus, karst systems increasingly need to be managed for sustained water quality. They also need to be managed for water quantity, as the water table can drop as a result of increased pumping and serious drought.

On the ground surface, physical manifestations of a karst system include sinkholes, losing streams, and springs. Sinkholes are subsidence of the surface, generally in areas with shallow soil surface cover. They can be plugged or open and connect the ground surface to the karst aquifer through a vertical conduit. In plugged sinkholes, a thin soil layer filters the surface runoff before it reaches the aquifer. In open sinkholes,

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there is no filtration at all. A losing stream has many cracks, small openings, or sink points in its bed and banks that allow water to directly recharge the groundwater aquifer. Some losing streams may appear as a dry channel bed during a large part of the year, only flowing when the subsurface karst system backs up and overflows during major storm events or when the flow rate exceeds the infiltration rate. Others may have water at all times because the flow rate is greater than the potential losses through the stream bed or the capacity of an underground channel. When these types of streams flow, they contribute to sediment, bacteria, organic material, and other debris being transported underground. Water that enters the groundwater aquifer through sinkholes and losing streams often resurfaces at springs.

The modeling of karst aquifers is important to understand the functioning of these aquifers and to have a tool to better manage these valuable resources. The functioning of a karst aquifer may be modeled by either a lumped parameter approach (black box model) or by a distributed or semi-distributed model. The use of a fully distributed model is only possible if there is complete knowledge of the conduit network. Developments in exploration techniques (Birk et al., 2004) and in computing methods (Nordqvist et al., 1996) are showing possibilities. However, it remains a challenging problem in any watershed and one whose solution is difficult to export from one watershed to another (Nordqvist et al., 1996; Bakalowicz, 2005). The commonly used groundwater flow MODFLOW model has been applied with some success to karst aquifers (Palmer et al., 1999). On the other end of the spectrum of modeling possibilities, the use of a completely lumped model to represent aboveground and underground reservoirs might be an efficient methodology. It would create a model suitable to study the effect of precipitation changes or to detect changes in the functioning of the aquifer but it would not be suitable to predict the effects of land use and land management changes. In addition, to be representative of the whole system, the model would need to be calibrated in part with spring flow data. This can be a challenge in watersheds that have many springs or when springs are located directly in a stream or a lake and not accessible.

Semi-distributed models might be an interesting compromise. The Soil and Water Assessment Tool (Neitsch et al., 2005), for example, is a daily time step hydrologic simulation model that simulates the impact of climate, land use, and land management in a watershed, which is usually divided into several subbasins. It is particularly interesting because certain features are already lumped together. Each hydrological response unit (HRU), for example, represents a unique combination of slope, land use, and soil within a subbasin. All tracts of land in that subbasin whose combination of soil, land use, and topography falls in that HRU category are lumped together. Similarly, all ponds within each subbasin are lumped and represented by one pond. Little work has been done on the calibration of the Soil and Water Assessment Tool for karst watersheds. Previous work by Afinowicz et al. (2005) led to a modification of the calculation of aquifer recharge and return flow to allow for a fraction of the infiltration to directly recharge the deep aquifer through cracks and fractures without going first to the shallow aquifer. Using the daily Nash-Sutcliffe efficiency as an indicator of the goodness of fit of the model, Afinowicz et al. (2005) obtained values of 0.4 and 0.09 for calibration and verification, respectively. They concluded that major changes to return flow calculations would be needed to accurately simulate baseflow during low flow periods, return flow during and after storm events, and potentially high peaks during storm events. Other efforts (Spruill et al., 2000; Coffey et al., 2004; Benham et al., 2006) are reported by Gassman et al. (2007) and show the difficulty of using that model to represent the baseflow of karst-fed streams.

The objective of this study is to determine if and to what extent a semi-distributed model like SWAT can be used to simulate sinkholes, losing streams, and springs and simulate the stream flow of a karst watershed. Our goals are to obtain good estimates of the flow and contaminant loads and concentrations in the streams of the watershed as a result of surface runoff, groundwater return flow, and infiltration and return of flow through larger conduits. A modification of the aquifer recharge is proposed to better represent quick vertical flow through the carbonate rock. To test the validity of our assumptions, the model was calibrated and validated for flow, fecal coliform, and phosphorus in a southwest Missouri karst watershed.

Study Area

Our application watershed was the James River basin in southwest Missouri. This 3,600 km^2 watershed lies within the Springfield plateau with a smaller portion in the northeast being part of the Salem plateau, two physiographic regions of the Ozark Plateau. The dolomite floor of the basin is covered by several layers of limestone that formed from the accumulation and sedimentation of small marine animal shells during the successive advances and recessions of sea waters. One of these layers, the Northview shale, acts as an aquitard, a restrictive layer that prevents the downward movement of water to the deeper aquifer and results in springs and seeps on its outskirts.

The river basin can be divided into five tributaries or subwatersheds: the upper James River in the northeast and the Finley River just south of it; Wilson Creek, which drains the city of Springfield and its surrounding towns; and Crane Creek and Flat Creek, which drain the central and southwest areas of the river basin, respectively (fig. 1). These perennial streams are fed by surface and groundwater flow. The James River basin drains into Table Rock Lake, a flood control and water supply lake that began operating in 1958. Flow gauges have been maintained by the U.S. Geological Survey (USGS) since 1970. Table 1 summarizes the daily flow data available at different stations and utilized in this study. Stream water quality was monitored on a monthly basis at some of these stations by the USGS.

The watershed is generally hilly and presents rock outcrops in some areas. Slopes are on average between 5% and 8%, with flatter areas in the north (2% slope) and steeper slopes adjacent to the streams (11% to 15%). The dissolution of limestone and dolomite resulted in karst topography. That topography, along with the existence of the Northview shale layer, resulted in many springs, sinkholes, and losing streams throughout the basin (fig. 1). Sinkholes are mostly located on the summit positions, while springs are on the back slopes and near the bottom of the valleys. Limited amounts of flow measurements were available for selected springs, located mostly around Springfield. While springs can easily be identified, sinkholes are more likely to be identified in areas of

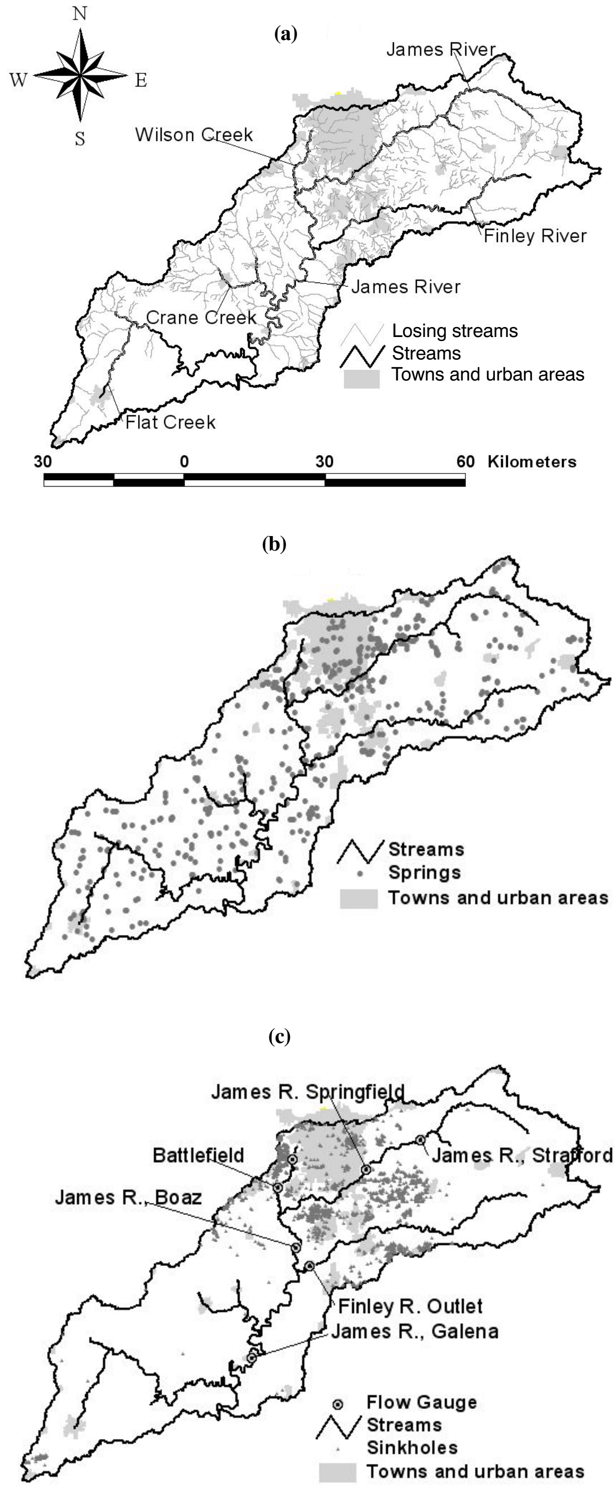

Figure 1. Karst features in the James River basin, Missouri: (a) losing streams, (b) springs, and (c) sinkholes.

greater population density. Sinkholes are typically identified when one stumbles upon them, sometimes at the expense of a sudden and unfortunate opening of the ground surface, or during construction of roads or buildings. Thus, greater building and population densities increase the likelihood of discovering sinkholes. The greater density of sinkholes in the center of the watershed may be an artifact of the heavier population density there. Almost all the streams are losing streams, which indicates that as the water moves downstream, some infiltrates through the stream bed into the aquifer. In the Ozarks, although the streams represent only 10% of the area, they are considered to be responsible for 40% of the groundwater recharge (Aley, 2007).

Soils are from the Noark-Rueter-Scholten association (NRCS, 2002). Half of the river basin is covered by Clarksville and similar soils, characterized by a very deep soil profile with high permeability (60 mm/h) and high rock fractions (20% to 80%). Another 20% of the watershed is covered by soils from the Scholten series. A fragipan located at a depth of 45 to 70 cm reduces this soil's permeability and increases the potential for surface runoff and lateral flow. Rock fractions are again very high (15% to 85%). Cedargap soils, very deep and well drained, are found in the flood plain.

The watershed's predominant land uses include grassland (pasture and hay fields) and forests (table 2). Grassland is divided into poor and fair condition pasture, and hay fields. Most rural households rely on septic tanks for their sanitary sewage treatment. The area assigned to septic fields was estimated from rural population densities by townships and by places obtained through the Missouri Spatial Data Information System (MSDIS, 2000). Urban and residential populations in communities with a sewer system were assumed to be connected to that system.

Four weather stations exist in the watershed to characterize climate: Springfield, Galena, Marshfield at the northeast end of the watershed, and Cassville in the Flat Creek subwatershed. Average annual precipitation at the four stations is very similar, approximately 1150 mm. Average annual temperature is 13°C, ranging from –0.5°C in January to 25.5°C in July.

METHODOLOGY

APPROACH FOR THE SIMULATION OF SINKHOLES, LOSING STREAMS, AND SPRINGS

Our purpose was to obtain a lumped representation of the karst features in each subbasin. We assumed that the location of these features was known throughout the watershed so that the number of sinkholes and the proportion of losing streams could be determined in each subbasin. While the prediction of water and contaminant transport through specific and defined karst features may be of interest in some cases, it re-

Table 1. Available flow and water quality data in the James River basin.

Station Name′	Latitude	Longitude	Period of Flow Monitoring	Period of Water Quality Monitoring
James River at Strafford	37° 12′ 13.5″	93° 04′ 45.6″	Oct. 1973 to Oct. 1986	NA
James River near Springfield	37° 08′ 59.9″	93° 12′ 12.2″	Oct. 1955 to present	Very limited
Wilson Creek near Battlefield	37° 07′ 03.9″	93° 24′ 13.9“	Oct. 1968 to Oct. 2004	Oct. 1999-2004
James River near Boaz	37° 00′ 23.7“	93° 21′ 52.8“	Oct. 1972 to present	Oct. 1969 to present
James River at Galena	36° 48′ 19.4“	93° 27′ 41.7“	Oct 1921 to present	1999 to present
Finley River at outlet	36° 58′ 29.6“	93° 19′ 40.4“	Oct 2001 to present	June 2001 to present

Table 2. Land use distribution in the James River basin.

Land Use Category	Percentage of Watershed
Pastures in fair condition	25.3%
Pastures in poor condition	25.3%
Forest	25.0%
Hay fields	21.5%
Urban high density	1.5%
Urban areas low-medium density	0.8%
Septic fields	0.6%

mains beyond the scope of this work as it would require knowledge of the network of groundwater conduits that link sinkholes and losing streams to springs. Our goal is to simulate and predict flow and pollutant concentrations in the streams and to obtain the correct balance between transport from surface waters, groundwater, and karst conduits using data that are available or that can be obtained at reasonable cost.

In the SWAT model, seepage losses can occur in tributary channels, main channels, ponds, potholes, wetlands, and reservoirs of a watershed. Tributary channels are smaller channels that drain only a fraction of a subbasin and route flow and pollutants to the main channel (Neitsch et al., 2005). Main channels route the flow and pollutants from one subbasin to the next. Where infiltrated water arrives depends on where it came from and is described in table 3. For tributary channels, ponds, and wetlands, the seepage goes to the shallow aquifer, where it should be taken into account when calculating return flow. In this study, sinkholes were represented by ponds with small drainage area and a large hydraulic conductivity at the bottom of the pond. Losing streams were represented by tributary channels with high hydraulic conductivity in the stream bed. Since seepage losses from main channels are lost from the system, we selected to represent losing streams only in tributary channels.

Springs were considered point sources. However, for springs recharged within a subbasin, the spring discharge is already calculated as part of the return flow to the stream and should not be added. The requirement to characterize a spring discharge exists only when the recharge area or the sinkholes that feed it lie outside the watershed boundary. An important assumption of the model is that water that infiltrates through soil, ponds, or losing streams in a subbasin recharges the aquifer and contributes to return flow within that same subbasin. The model does not allow for transfer of water from one subbasin to another. While we can suspect that such transfers do exist in this watershed, to our knowledge no dye trace has been performed to identify the recharge areas of any spring in the James River basin and there was no evidence of any transfer. Therefore, no spring was characterized as a point source. When and where it is determined that return flow is fed by infiltrations outside the watershed boundary or when water is transported from one subbasin to another or to a place outside the watershed, a different simulation strategy might be needed.

Table 3. Source and flow path of seepage losses in SWAT.

Source	Variable	Destination
Tributary channels	*tloss*	Shallow aquifer
Ponds	*twlpnd*	Shallow aquifer
Wetlands	*twlwet*	Shallow aquifer
Potholes	*potsep*	Soil profile
Reservoirs	*ressep*	Deep aquifer or bank storage
Main channels	*rttlc*	Deep aquifer

Model Description

The following model description is limited to the movement of seepage losses from streams and ponds, as these will be used to represent losing streams and sinkholes. For other aspects of SWAT, Neitsch et al. (2005) provide a complete description of the model in the theoretical documentation. We outline here the current model as well as the changes proposed to better represent quick vertical infiltration through cracks and conduits.

Aquifer Recharge

In the current version (SWAT 2005), the recharge for the day is calculated as a linear function of the daily seepage and of the recharge for the previous day. Daily seepage includes seepage through the soil profile and through ponds and losing streams. The equation used to calculate that recharge is:

$$rchrg(t) = \left[1 - \exp^{\left(\frac{-1}{gw\_delay}\right)}\right] * seepage(t) + \exp^{\left(\frac{-1}{gw\_delay}\right)} * rchrg(t-1) \quad (1)$$

where $rchrg(t)$ is the aquifer recharge on day t, gw_delay is the groundwater delay (a user-defined input parameter), $seepage(t)$ is the amount of water that exits the bottom of the soil profile on day t, and $rchrg(t-1)$ is the recharge of the previous day. All seepage through the soil and infiltrations from ponds and losing streams are added together and assumed to travel vertically to the aquifer with the same velocity. The groundwater delay represents the number of days required for water to reach the aquifer from the bottom of the soil profile. Increasing the groundwater delay increases the storage of water in the rock substrate and allows for sustained flows during drought periods. However, it delays the recharge of the aquifer and the subsequent increase in return flow following rain events. Thus, the specificity of a karst aquifer, i.e., large storage, quick infiltration, and fast return flow, cannot be taken into account.

To provide a means to represent the quick movement of water from the ground surface to the aquifer, we propose to split the recharge of the aquifer into two elements: the recharge from infiltrations through the soil profile, and the recharge from sinkholes and losing streams. Equation 1 is replaced by equations 2 and 3:

$$rchrg\_seep(t) = \left[1 - \exp^{\left(\frac{-1}{gw\_delay}\right)}\right] * soil\_seep(t) + \exp^{\left(\frac{-1}{gw\_delay}\right)} * rchrg\_seep(t-1) \quad (2)$$

$$rchrg\_krst(t) = \left[1 - \exp^{\left(\frac{-1}{krst\_delay}\right)}\right] * krst\_seep(t) + \exp^{\left(\frac{-1}{krst\_delay}\right)} * rchrg\_krst(t-1) \quad (3)$$

In equation 2, *rchrg_seep*(t) is the recharge from infiltrations through the soil (*soil_seep*(t)), and *gw_delay* is the time required for water to reach the aquifer from the bottom of the soil profile through the porous matrix of the rock substrate. This matrix includes small cracks but no direct conduits. In equation 3, *rchrg_krst*(t) is the recharge from sinkholes and losing streams via direct conduits to the aquifer, *krst_seep*(t) represents all losses from sinkholes and losing streams, and *krst_delay* is the time required for water to reach the aquifer from the bottom of sinkholes or stream beds via a direct conduit. The total recharge is the sum of the recharge by infiltrations and losses from sinkholes and losing streams.

Return Flow

No modification was introduced in how return flow is calculated as a function of the groundwater flow of the previous day and the aquifer recharge of that day. However, return flow was modified in our approach because the aquifer recharge was modified. The equation of the model presented by Neitsch et al. (2005) as modified by Afinowicz et al. (2005) is repeated here for convenience:

$$Q_{gw}(t) = [1 - \exp^{-\alpha_{gw}}] * rchrg(t) + \exp^{-\alpha_{gw}} * Q_{gw}(t-1) \quad (4)$$

where $Q_{gw}(t)$ is the return flow for the day, *rchrg*(t) is the aquifer recharge, and α_{gw} is the baseflow recession constant.

Model Configuration

Subbasins boundaries and HRUs were delineated using the ArcView AVSWATX interface. The digital elevation model (DEM) and the land use map were obtained from MSDIS (MSDIS, 1999a, 1999b). A 60 m DEM layer was utilized to determine the boundaries of the subbasins. Efforts were taken to match subbasin outlets with existing flow gauges and water quality stations. Subbasin size is 150 km^2 on average and varies from 35 to 325 km^2. Land use and land cover inputs were derived from a 30 m grid map built from 1991 to 1993 satellite images. Soil information was obtained from the Natural Resources Conservation Service (NRCS) State Soil Geographic (STATSGO) database, which provides soil data at an appropriate amount of detail for large areas. Using the splitting tool available in ArcSWAT, grassland was divided into fair-condition pasture, poor-condition pasture, hay fields, and septic fields. A threshold of 5% for land use and 25% for soil was used. Even though they do not represent 5% of the subbasin's area, septic tanks and the associated leach fields were forced into the model because we expect their impact on nutrient loadings will be significant. With 24 subbasins in the watershed, this configuration resulted in 243 HRUs.

The baseline management practices were estimated from previous work in the Shoal Creek and Little Sac watersheds, located on the other side of the west and north boundaries of the James River basin, respectively (Baffaut, 2004, 2006; Benham et al., 2006). Pastures were divided into two sets so that cattle could be moved between different pastures from month to month. Grazing periods alternate between these two sets. Hay land was assumed to be harvested in June and grazed later in the season. The grazing rates used in the model are based on cattle number, harvested hay acres, wood, and grass pastures from the National Agricultural Statistics Service (NASS) county summaries (USDA, 2004). Fertilizations, hay harvesting, and grazing of poor and fair condition pastures were based on information provided by local farm panels.

Septic fields were assumed to be in good-condition grass. A daily application of effluent was applied on these areas that reflected the estimated effluent production per household and the nutrient content of the effluent (USDA, 1992). The variation in population density across the watershed was represented by a larger or smaller fraction of the subbasin being used by these septic fields.

The management of urban areas was derived from the management of urban areas in North Springfield determined during the Little Sac water quality analysis (Baffaut, 2006). Forests were assumed to be mature forests; no forest harvesting or planting was represented in the model. Different values of the minimum value for the USLE cover factor (USLE_C) were given for wooded areas, fair and poor condition grazed pastures, and hay fields.

Numerous wastewater treatment plants (WWTPs) discharge their effluent into the James River or its tributaries. Permit records, personal communications, and the 2000 James River total maximum daily load (TMDL) document (MDNR, 2001) were utilized to estimate flow and phosphorus loadings from these WWTPs from 1970 to the present. Sources of information included the GIS layer on permitted facilities (MSDIS, 2006) and the daily discharges of the southwest wastewater plant of the city of Springfield, Missouri, from 2000 to 2007 (K. Highfill, Chemist, Southwest Wastewater Treatment Plant, Springfield, Missouri, April 2008, personal communication). The James River TMDL specifies the maximum allowed phosphorus concentration of major towns' effluents after 2003 and 2007. For other plants and for conditions prior to 2000, the phosphorus concentration was set at 5 mg/L, except for Springfield because data showed that it was closer to 7.3 mg/L and 0.3 mg/L before and after implementation of the phosphorus removal processes, respectively.

There is some uncertainty about the fecal coliform concentrations of the WWTP releases. Assuming permitted concentrations (400 cfu/100 mL) in all WWTP releases resulted in simulated concentrations that exceeded the monitoring results. The average concentration reported by the Springfield plant (10 cfu/100 mL) provided concentrations that were too low. In a previous study (Baffaut et al., 2005), weekly samples of the outflow of the North Springfield treatment plant over three months produced an average concentration of 70 cfu/100 mL. Since the same disinfection method is used at the southwest plant and other large plants in the James River basin, this value was selected as the outflow fecal coliform concentration for the WWTP discharges of those plants that used ozone disinfection. For other plants, the permitted average monthly value of 400 cfu/100 mL was used.

The watershed is characterized by many springs that have mean flow values from very small up to 1 m^3/s. The recharge areas for these springs are not known and may or may not be in the watershed. As indicated earlier, we did not attempt to represent the springs because we assumed that spring flow would be included in the return flow once it completely took into account losses from sinkholes and losing streams.

Losing streams were defined in the model by the hydraulic conductivity of tributary channels in each subbasin. Those were set as a function of the hydraulic conductivity of the flood plain soil (10 to 50 mm/h), with higher values being assigned in subbasins where the density of losing streams was highest. No losing stream was specified in the Flat Creek subbasins in which none had been identified. Sinkholes were simulated by ponds with a hydraulic conductivity of 100 mm/h. For each subbasin, the fraction of subbasin area draining into sinkholes varied from 0.001 to 0.15 with an average of 0.037.

Calibration and Validation

The model was manually calibrated for flow based on the 1973-1980 period and validated based on the years 2001-2007. For phosphorus and fecal coliform concentrations, data were reliable only for the years 2001 to 2007 and these were split into two periods: January 2001 to September 2004 for calibration, and October 2004 to September 2007 for validation. Samples were not collected in Battlefield after November 2003, and the number of samples collected near Boaz is smaller than in Galena and in the Finley River. The flow gauge on the Finley River was not operational in 2005, and no samples were collected during that period. Flow and water quality data were downloaded from the USGS website of daily surface water data (USGS, 2008).

Calibration and validation were performed with both the original code and after incorporation of the new model's components. This led to two sets of parameters that could influence the results. To test whether the model had a real impact, we then ran the original code on the final set of parameters obtained for the revised model.

The goodness of fit of the model was evaluated by visual comparison of measured and estimated hydrographs and plots of pollutant concentrations and loads with time. In addition, calibration indicators were used: the percent bias (PBIAS), which is the relative difference between measured and simulated results; the Nash-Sutcliffe efficiency (NSE) (Krause et al., 2005; Moriasi et al., 2007), a common indicator used to quantify how daily simulated values fit the measured values; and the prediction efficiency (P_E) (Santhi et al., 2001), which allows comparison between the distributions of measured and simulated values. According to Moriasi et al. (2007), an acceptable value of the Nash-Sutcliffe coefficient should be greater than 0.5, while a good value should be greater than 0.7. Acceptable values of PBIAS are less than 25% for flow and less than 70% for nutrient concentrations (Moriasi et al., 2007). There are no guidelines for prediction efficiencies.

To ensure that the model simulates the correct ratio between surface and total flow, PBIAS was also calculated on surface runoff. In this case, the SWAT estimates of base and surface flow could be significantly different from what is observed from total stream flow values because of the large impact of springs, discharges from wastewater treatment plants, and in-stream transmission losses. Therefore, the comparison between simulated and measured baseflow ratio was based on flow separation of both measured and simulated flow values. There are several techniques to separate baseflow from surface runoff in stream flow data. After verification that the USGS hydrograph separation (HYSEP) method (Sloto and Crouse, 1996) gave similar results to other baseflow estimation methods (Arnold and Allen, 1999), we selected the HYSEP method because it is easy to implement in a spreadsheet. The baseflow ratio, ratio of baseflow to total flow, further indicates whether the balance between base and surface flow is respected.

Phosphorus and fecal coliform input parameters were calibrated based on total phosphorus and fecal coliform concentrations in grab samples collected monthly from 2001 to 2007. Since both loads and concentrations are important water quality assessors, we evaluated how well loads were simulated after concentration-based calibration. While the Nash-Sutcliffe efficiency, PBIAS, and r^2 are typically calculated for measured and simulated values that correspond to the same day, the prediction efficiency compares the distributions of simulated and measured values. It is calculated as the r^2 between measured and simulated values that correspond to a same frequency of occurrence (Santhi et al., 2001). It is utilized to evaluate how the range and distribution of the simulated values match the range and distribution of measured values.

Results and Discussion

Flow Simulation

The calibration and validation indicators for flow are presented in table 4. Percent differences in total flow were all less than 25%. Simulated flows were underestimated during the calibration period but overestimated by 10% to 20% for the validation period, except in Galena. The validation period (2001-2007) was characterized by two extremely dry periods: one during summer 2002, and one from summer 2005 to winter 2007. In 2002, the precipitation in Galena (fig. 1) was less than 60% of the average 2001-2007 precipitation.

Surface runoff biases were all less than or close to 10%, indicating that the percent differences in total flow were due largely to a misrepresentation of baseflow. Nash-Sutcliffe values were around 0.5 for Wilson Creek and the James River near Boaz and in Galena. Nash-Sutcliffe values for the James River near Springfield and for the Finley River were lower, around 0.3. All values were less for the validation than they were for the calibration, except in Battlefield.

Table 4. Measures of flow goodness of fit for the James River basin SWAT model.

Measurement	Period	Wilson Creek Battlefield	James River near Springfield	James River near Boaz	Finley River at outlet	James River in Galena
PBIAS on total flow	1973-1980	20%	4%	12%	No data	9%
	2001-2007	-12%	-21%	-12%	-19%	-2%
PBIAS on surface runoff	1973-1980	3%	7%	11%	No data	-2%
	2001-2007	-9%	-1%	-9%	-5%	2%
Daily NSE	1973-1980	0.46	0.33	0.50	No data	0.56
	2001-2007	0.56	0.24	0.47	0.30	0.52
Comparison of baseflow ratio (measured/simulated)	1973-1980	0.85/0.82	0.75/0.66	0.79/0.72	No data	0.80/0.68
	2001-2007	0.73/0.66	0.40/0.51	0.54/0.55	0.51/0.57	0.54/0.56

Results in Battlefield were heavily influenced by the outflow of the wastewater treatment plant (WWTP). Validation results for Wilson Creek were improved compared to the calibration period because daily outflow data were available from 2000 to 2007 instead of an average annual outflow value for the calibration period. Due to infiltrations of soil water into the sewer pipes during rain events, the amounts of treated water and therefore outflow vary considerably on a daily and seasonal basis. In addition, when the capacity of the treatment plant is exceeded during large events, a fraction of the influent bypasses the treatment plant and is subjected to only decantation before being released into the stream. Outflow can therefore be larger than the plant maximum capacity, and contaminant concentration can be very high. No data were available for the other treatment plants in the watershed, and an average annual estimate of the discharge flow was utilized. The discharge from the Springfield WWTP represented on average 44% of the average annual flow in Wilson Creek and varied from 5% during storm events to 100% during very dry weather. The discharges from Springfield and other communities' WWTPs represented 14% and 7% of the stream flows in Boaz and Galena, respectively, and less than 1% of the remaining tributaries.

Comparisons of simulated and observed flows (fig. 2) show that the model sustained significant flows during summers even with a lack of precipitation, a result that could not be achieved before the code was modified because the recharge from losing streams and sinkholes was not returned to the streams. However, simulated flows during drought periods were still lower than measured flows. Indicators confirmed the visual assessment: while Nash-Sutcliffe efficiencies improved only slightly or even decreased in the upper reach of the James River, the partition of the flow between surface runoff and groundwater flow was improved (table 5) in comparison to what was obtained before incorporation of the new components. Percent bias on surface runoff was better at the three upstream stations (Wilson Creek and upper James River); it was similar for the James River near Boaz and in Galena. Percent bias on total flow did not change significantly and remained around 10%, except for Wilson Creek.

The improved partition between surface runoff and groundwater flow was not the result of a different calibration of the model after incorporation of the new components. A simulation run made with the original code on the final set of input parameters produced the following surface runoff biases: 12% in Battlefield, 14% in Strafford, 11% in Springfield, 17% near Boaz, and 3% in Galena. Except for the Strafford station, results were similar or worse than those presented in table 5.

In the SWAT model, only transmission losses from tributary streams recharged the shallow aquifer and possibly returned to surface waters. Losses from main channel reaches were calculated but did not recharge the shallow aquifer; they were lost from the simulated system. While this is legitimate for rivers and streams in most non-karst systems and is sometimes true for karst systems, there are cases when stream infiltrations return to a downstream reach or adjacent tributary of the same watershed through a direct underground conduit. Similarly, underground transfers from sinkholes in one subbasin to a spring located in a downstream or adjacent subbasin probably exist. We expect that the effect of these pathways would be more pronounced in the upper parts of the river basin because they represent a larger proportion of the stream flow. As flow moves downstream, integration of all the pathways takes place as all eventually reach the stream. This may explain why Nash-Sutcliffe values were better (around 0.5) in the downstream reach of the James River than in the upper reaches of the river basin (around 0.35 in the upper James River and in the Finley River). Pathways from one subbasin to another could be modeled if information on the endpoints and travel times of these underground conduits was available.

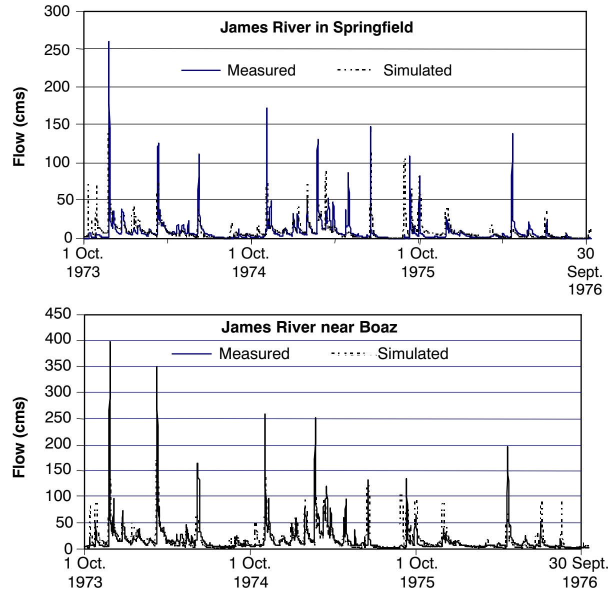

Figure 2. Measured and simulated flows of the James River in Springfield and near Boaz from 1973 to 1976.

Water Quality Simulation

Figures 3 and 4 show the measured and simulated total phosphorus concentrations and loads over time in the James River near Boaz and in the Finley River. These two streams are key examples of the model's behavior. Tables 6 to 8 present the calibration and validation indicators for total phosphorus concentrations and loads and for fecal coliform concentrations. Figures 5 to 8 allow a visual comparison of the concentration frequency curves for total phosphorus and fecal coliform for each sampling station. Since only monthly or bi-monthly grab sample concentrations were available, most grab samples were collected during baseflow condi-

Table 5. Values of 1973-1980 calibration indicators for the James River basin flow gauges before and after modification of the SWAT code.

	Wilson Creek Battlefield		James River in Strafford		James River near Springfield		James River near Boaz		James River in Galena	
	Before	After	Before	After	Before	After	Before	After	Before	After
NSE	0.36	0.46	0.35	0.33	0.36	0.33	0.47	0.50	0.54	0.56
Bias on surface runoff (%)	10%	3%	27%	10%	13%	4%	9%	11%	-3%	-2%
Bias on total flow (%)	23%	20%	-2%	-3%	4%	4%	11%	12%	6%	9%

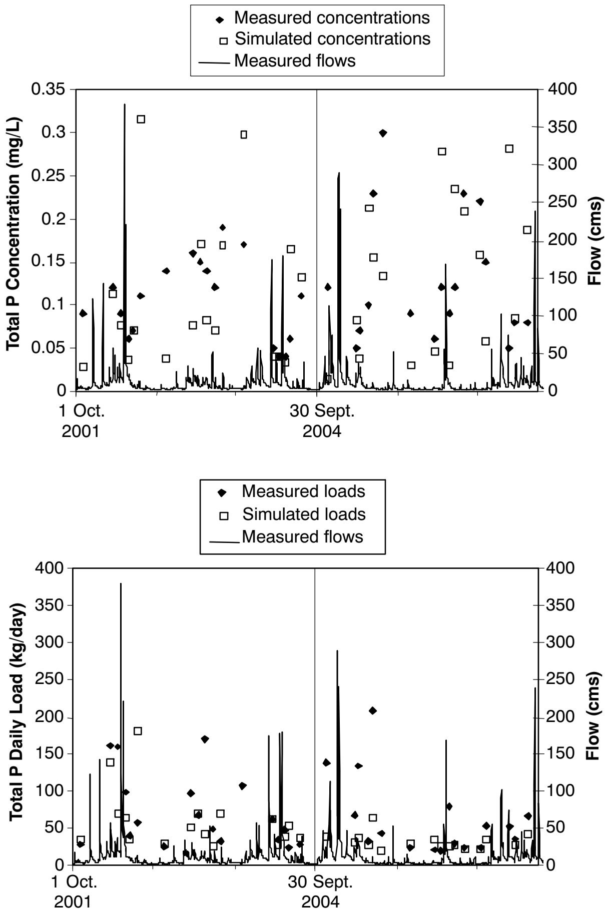

Figure 3. Plots of measured and simulated total phosphorus concentrations and loads in the James River near Boaz.

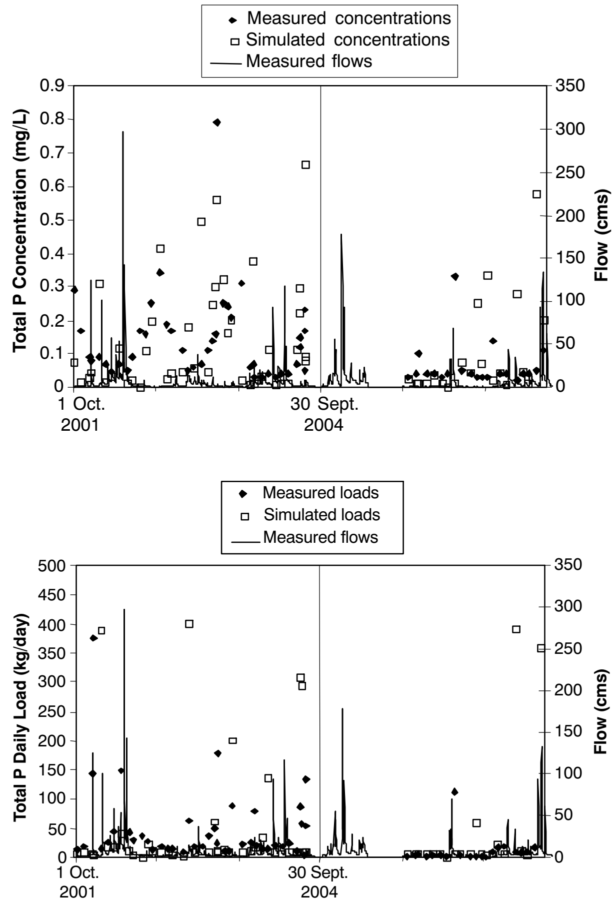

Figure 4. Plots of measured and simulated total phosphorus concentrations and loads in the Finley River.

Table 6. Total phosphorus concentrations goodness of fit for the James River basin SWAT model.

	Wilson Creek Battlefield		James River near Boaz		Finley River		James River in Galena	
	2001-2004	2004-2007	2001-2004	2004-2007	2001-2004	2004-2007	2001-2004	2004-2007
No. of samples	34	NA	23	17	46	24	43	31
NSE	0.34	NA	-2.1	-1.1	-0.4	-5.6	-1.8	-55
r^2	0.36	NA	0.05	0.0	0.20	0.0	0.07	0.0
PE	0.94	NA	0.50	0.84	0.87	0.81	0.96	0.62
PBIAS	16%	NA	4%	-3%	7%	30%	4%	-149%

Table 7. Total phosphorus loads goodness of fit for the James River basin SWAT model.

	Wilson Creek Battlefield		James River near Boaz		Finley River		James River in Galena	
	2001-2004	2004-2007	2001-2004	2004-2007	2001-2004	2004-2007	2001-2004	2004-2007
No. of samples	34	NA	18	17	42	24	43	31
NSE	-0.21	NA	-6.8	-27	-2.6	-0.9	0.23	-91
r^2	0.41	NA	0.07	0.0	0.04	0.0	0.24	0.07
PE	0.91	NA	0.61	0.54	0.92	0.68	0.91	0.59
PBIAS	-26%	NA	-26%	-58%	-101%	-52%	7%	-538%

Table 8. Fecal coliform concentrations goodness of fit for the James River basin SWAT model.

	Wilson Creek Battlefield		James River near Boaz		Finley River		James River in Galena	
	2001-2004	2004-2007	2001-2004	2004-2007	2001-2004	2004-2007	2001-2004	2004-2007
No. of samples	31	NA	30	18	42	21	43	33
NSE	0.11	NA	-6	-0.08	0.0	-0.08	0.03	0.21
r^2	0.24	NA	0.09	0.04	0.00	0.01	0.01	0.26
PE	0.66	NA	0.88	0.33	0.65	0.99	0.79	0.95
PBIAS	72%	NA	-56%	73%	92%	80%	75%	20%

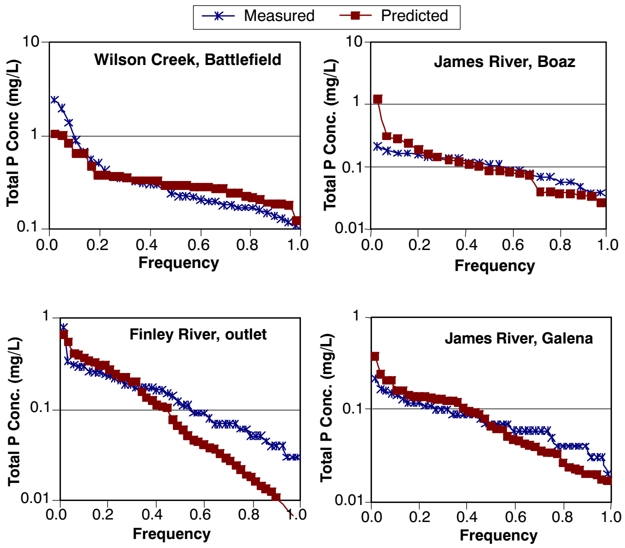

Figure 5. Total phosphorus concentrations in the James River basin, 2001-2004.

tions. In addition, the water quality samples collected during runoff events produce data that may misrepresent storm flow because concentrations on those days can vary rapidly. The calibration is therefore biased toward baseflow conditions because of the larger number of samples. Discrepancies between the measured and simulated concentrations at high flow (highest concentrations at lowest frequencies) are therefore expected because of scarce data to represent these conditions.

Nash-Sutcliffe efficiencies and r^2 values indicated poor simulation results everywhere, except for Wilson Creek in Battlefield, largely a result of the good knowledge of phosphorus releases at the wastewater treatment plant from 2000 to 2007. Prediction efficiencies were satisfactory, varying from 0.5 to 0.96 for phosphorus concentrations and loads, and from 0.33 to 0.97 for fecal coliform. The lowest values of prediction efficiencies were all obtained for the James River near Boaz, a possible result of the lower number of samples collected at that station.

In Boaz, prediction efficiencies indicated that the correct range and frequency of concentrations were simulated for

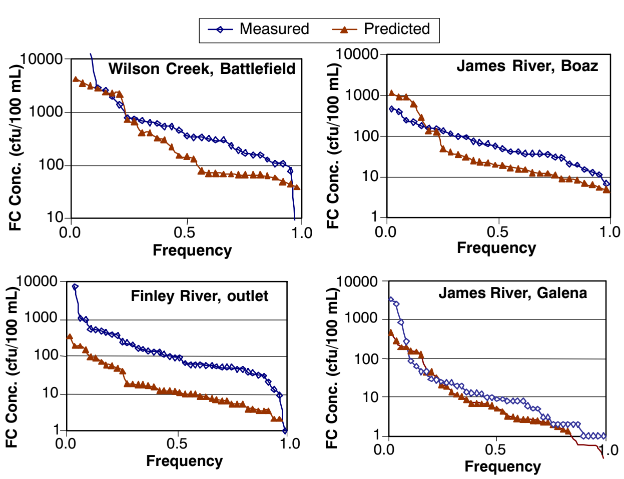

Figure 6. Fecal coliform concentrations in the James River basin, 2001-2004.

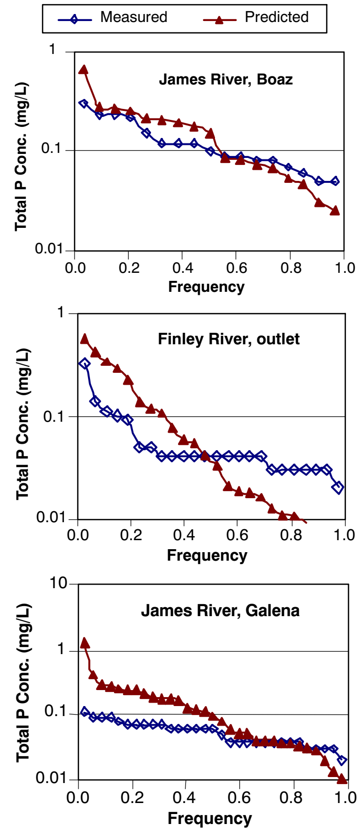

Figure 7. Total phosphorus concentrations in the James River basin, 2004-2007.

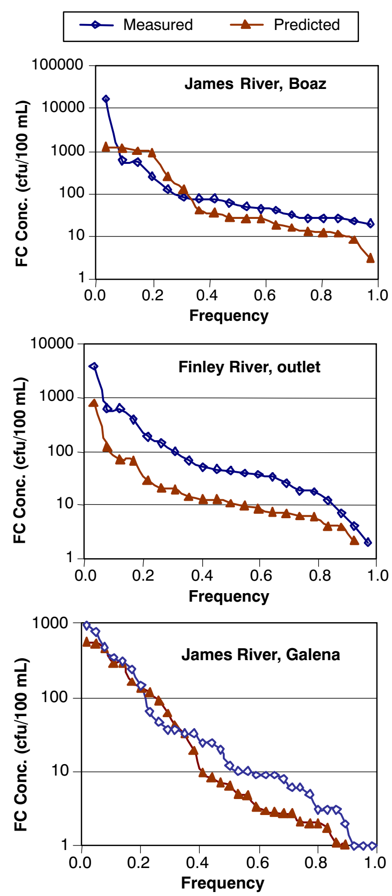

Figure 8. Fecal coliform concentrations in the James River basin, 2004-2007.

bacteria and total phosphorus during the calibration and validation periods. Although the prediction efficiency for fecal coliform concentrations during the validation period was low (0.33), this was largely because one high concentration during a storm event was underestimated by SWAT. Otherwise, the two curves were very close (fig. 8).

Different results were obtained for the Finley River. Both simulated fecal coliform and total phosphorus concentrations were low for all flow conditions. Given the good ranges and frequencies for phosphorus and fecal coliform in Boaz, we speculate that a source of contamination not taken into account in the model may be the cause: possibly illegal discharges of sanitary waters to the stream. There are many "weekend" trailer parks along the river that do not have adequate sanitation. These contributions decreased after 2004, and the frequency curves of measured concentrations in the Finley River for 2004-2007 are lower than for 2001-2004, especially for phosphorus. On the other hand, the curves for the calibration and validation periods are very similar for the Boaz site. The change in contributions to the Finley River also resulted in different phosphorus frequency curves in Galena, but the differences were smaller. This may explain the poorer prediction efficiency and the large percent bias of total phosphorus concentrations and loads in Galena for the validation period.

Phosphorus and bacteria are currently not routed through the groundwater aquifer in the SWAT model. Infiltrations of surface waters rich in pollutants with direct pathways to the shallow aquifer make these routines more critical. Quarterly spring monitoring undertaken by the Watershed Committee of the Ozarks (2001) in 2000 and 2001 showed that the average orthophosphate concentrations of rural and urban springs were 0.17 and 0.27 mg/L, respectively. The average rural and urban *E. coli* levels were 46 and 311 cfu/100 mL, respectively. Given that springs are responsible for a large fraction of the flow during baseflow conditions, such levels would significantly increase simulated concentrations at low flow.

CONCLUSIONS

After incorporating changes to the SWAT code to account for infiltrations from sinkholes and losing streams in the return flow to the surface waters, the SWAT model was calibrated to the James River basin, a mixed use, urban and rural 3600 km^2 watershed in a karst region of southern Missouri. Nash-Sutcliffe values of around 0.5 were obtained for the calibration and validation periods in the main stem of the stream and at the outlet. Lower values of 0.3 were obtained in smaller upstream tributaries, a possible consequence of transfers of water from one subbasin to another via underground conduits linking sinkholes to downstream or adjacent streams. Separation of surface from baseflow showed that surface flows were better simulated. In addition, similar results were obtained for calibration and validation, a significant achievement since both periods had different climatic characteristics, with the validation period being characterized by a drought for most of its duration. However, while simulated flows were sustained during normal dry periods, they were still too low during drought periods, as happened during the 2001-2007 validation period.

Monthly water quality grab samples were used to test the calibration of the model for phosphorus and fecal coliform. The simulations were successful on a frequency basis for both phosphorus and fecal coliform. Prediction efficiencies during calibration and validation varied from 0.33 to 0.97. We conclude that the model can be used to test the impact of management practices or land use changes on the frequency at which certain concentration thresholds are exceeded. This should prove valuable for watershed management and total maximum daily load studies.

In one area of the watershed, the results led to the speculation that some contamination sources were not incorporated in the model. We also speculate that routing of phosphorus and bacteria through the soil profile and through the rapid pathways to groundwater and the shallow aquifer must be modeled to achieve better results, especially at low flows.

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