Assessing Applicability of SWAT Calibrated at Multiple Spatial Scales from Field to Stream

다단계 수문과정을 고려하여 보정된 SWAT모형의 적용성 검토

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Abstract

유역 내부의 수문과정에 대한 관측치를 이용한 모형의 보정은 분산형 수문수질 모형의 적용성을 높이는 방법으로 권장되지만, 관측치가 충분하지 않은 경우가 많아 작용사례가 드문 실정이다. 본 연구는 경지에서 유역의 출구까지 여러 단계의 수문과정을 고려하여 분산형 수문수질모형을 보정하는 방법을 제시하고, 이와 같은 방법으로 보정된 모형의 작용성을 검토하고자 하였다. 이를 위해 본 연구는 SWAT 모형을 이용하여 미국 조지아의 South Atlantic Coastal Plain에 있는 한 농업유역의 수문과정을 모의하였다. 모형보정은 유역의 수문량 및 유사와 영양물질의 양 뿐만 아니라 경지에서 관측된 바이오매스, 토양 침식량 및 영양물질 발생량, 수변지역 (riparian buffer)에서 발생하는 유사와 영양물질 등의 과정을 고려하여 수행되었다. 모형의 보정 및 검증기간은 자료기간과 토양보전농법의 시행기간을 고려하여 선정되었으며, 보정된 모형의 적용성은 복수의 통계치 (NSE, RE, RSR 등)를 이용하여 분석하였다. 보정된 모형의 수문량 및 영양물질 운송과정은 각각 NSE 0.93 및 0.59의 정확도로 보정하였다. 그러나 유사 (NSE: 0.40)와 영양물질 (NSE: 0.45)의 경우는 상대적으로 낮은 정확도를 보였다. 본 연구에서 다단계 수문과정을 고려하여 보정된 SWAT 모형은 유역 내에서 발생하는 수문과정을 보정하는데 작용이 있는 것으로 나타났다. 하지만, 평균내용량 및 수문량의 변화를 고려하기 위해서는 오차를 감안하여 보정된 모형의 적용성을 고려하는 것이 필요하다. 이에선 SWAT 모형의 보정에 다단계 또는 유역내부의 수문과정을 고려하는 것이 유익하다고 판단되며, 이는 SWAT 모형의 보정에서 다단계 또는 수문과정을 고려하는 것이 유익하다고 판단된다. 또한 SWAT 모형의 보정에 다단계 또는 수문과정을 고려하는 것이 유익하다고 판단된다. 또한 SWAT 모형의 보정에 다단계 또는 수문과정을 고려하는 것이 유익하다고 판단된다.

Keywords: Calibration; Validation; SWAT; QUAL2E; Biomass; Sediment; Nutrient; Riparian buffer

I. Introduction

After substantial increases in USDA conservation programs through the Farm Security and Rural Investment Act of 2002, the Conservation Effects Assessment Project (CEAP) was initiated to quantify the environmental benefits of USDA conservation practices applied to agricultural land. As part of the CEAP effort, the impacts of conservation practices on hydrology and water quality have been evaluated by field and modeling studies at 14 USDA Agricultural Research Service (USDA-ARS) benchmark watersheds (USDA, 2007). These studies were designed to validate modeling output and quantify model prediction uncertainty at multiple scales (Richardson et al., 2008), utilizing the Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998) and Annualized Agricultural Non-Point Source (AnnAGNPS) (Bingner and Theurer, 2001).

The Little River watershed located in the coastal plain region of southern Georgia is one of the USDA-ARS CEAP benchmark watersheds. Many field and watershed-scale monitoring studies have been conducted in the watershed. Water quality of the watershed streamflow is generally considered good, believed to be largely controlled by riparian buffers surrounding the most part of the watershed stream system (Asmussen et al., 1979; Sheridan et al. 1983; Lowrance et al., 1984; Lowrance and Leonard, 1988; and Feyereisen et al., 2008). Feyereisen et al. (2008) reported average nitrate concentration of 0.21 mg/L, total Kjeldahl N (TKN) of 1.57 mg/L, and total P of 0.27 mg/L for streamflow of the Little River Watershed between 1974 and 2003.
Similar water quality observations have been reported for other subwatersheds of the Little River. Sheridan et al. (1983) found that average weighted nitrate and orthophosphate phosphorus concentrations of streamflow in subwatershed K ranged from 0.05 to 0.20 mg/L and from 0.29 to 0.45 mg/L, respectively, between 1974 and 1978. Lowrance and Leonard (1988) found mean daily nitrate concentrations to vary from 0.07 to 0.09 mg/L within subwatersheds between 1982 and 1986. These observations implicitly represent the watershed characteristics of having a plinthic layer of low hydraulic conductivity in the South Atlantic Coastal Plain, which are quite different from those of other regions. For instance, mean annual nitrate concentrations for some mid-western tile drained watersheds were reported to range from 6 to over 20 mg/L (Tomer et al., 2003; Tomer et al., 2008), and total phosphorus (TP) concentrations were as high as 2.8 mg/L (McDaniel et al. 2009). While considerable research has been implemented to relate the impacts of the riparian buffers to the in-stream water quality of the Little River watershed (Todd et al., 1983; Lowrance et al. 1983; Lowrance et al. 1984; Lowrance et al. 1985a; and Hubbard and Lowrance, 1997), little research has been conducted to examine the impacts of crop land management on in-stream water quality.

Hydrologic and water quality models have been used as cost-effective tools for analyzing the impacts of conservation practices on hydrology and water quality. Calibration and validation are common modeling practices used to evaluate the predictive capability of models for different watershed characteristics and increase the confidence in their predictions (Rogers et al., 1985; Klemes, 1986; Refsgaard, 1997; and Grunwald and Norton, 2000). Because of uncertainties associated with input, parameters, and model structure, modeling studies have shown that calibration can produce numerous parameter sets that result in similar model performance (Beven and Binley, 1992; Gupta et al., 1998; Beven, 2000; Trevisan et al., 2000; Beven and Freer, 2001; Beven, 2006; Van Griensven et al., 2008). The non-unique determination of parameter values is referred to as “equifinality” in hydrologic modeling (Beven and Freer, 2001). Several researchers emphasized the advantages of multi-site and multi-scale calibration for mitigating the equifinality problem in parameter calibration (Anderton et al., 2002; Madsen, 2000; Madsen, 2003; Vazquez et al., 2008; Rao et al., 2009).

Multi-scale calibration approaches are especially important for modeling studies using distributed models since the modeling practices try to simulate many hydrologic processes and predict multiple hydrologic variables with numerous parameters. Subsequently, simulated conservation practice effectiveness is subject to change depending on values of parameters used to represent the load reduction mechanisms of the practices. Considering transport processes of sediment and nutrient from fields to a watershed outlet along flow paths in parameter calibration would help improve credibility of the impact assessment results as well as the calibrated model. Previous SWAT modeling applications for the Little River watershed have examined accuracy of simulated water balance and parameter sensitivity with emphasis on appropriate partitioning of surface flow and baseflow (Bosch et al., 2004; Feyereisen et al., 2007a; Van Liew et al., 2005). The previous studies utilized measurements made at the watershed outlet for parameter calibration, but they did not incorporate field-scale observations into the water quality modeling and sensitivity analysis. Thus, it is required to evaluate SWAT water quality simulations for the watershed using additional field measurements in order to facilitate improved reliability of conservation practice impact assessment.

The objectives of this research are to 1) demonstrate a novel calibration procedure that considers upland and in-stream processes and transport pathways for runoff, sediment, and nutrients and 2) to examine the impacts of this calibration procedure on simulation results of the SWAT model developed for a shallow gradient Coastal Plain Watershed.

**II. Method and Procedures**

1. **Study Area and Datasets**

The Little River Experimental Watershed study area is located near Tifton, Georgia, in the South Atlantic Coastal Plain (Fig. 1). The hydrology and water quality of the watershed have been monitored by the Southeast Watershed Research Laboratory since 1968 and 1973, respectively (Bosch and Sheridan, 2007; Bosch et al., 2007a; Bosch et al., 2007b; Fayereisen et al., 2007b). Eight stream gauges and 31 rainfall gauges are currently active within the watershed.
Fig. 1 Location of the Little River watershed subwatershed K (LRK)

The climate of the watershed is humid subtropical with a long growing season (Sheridan, 1997). Rainfall is unevenly distributed during the year and often occurs as short-duration, high-intensity convective thunderstorms (Bosch et al., 1999).

Subwatershed K, draining 16.9 km², a headwater-catchment of the Little River watershed, was selected for this study. Mixed forest and pines cover approximately 60% of subwatershed K. The remainders are primarily row crops, including cotton, peanuts, corn, and fruit and vegetable. Almost year round production of vegetables and row crops such as peanuts and cotton on upland areas has led to extensive and sustained use of fertilizer and pesticides on the watershed (Bosch et al., 2004). The soils of subwatershed K are typically loamy sands with a plinthic layer of low hydraulic conductivity soil underneath the plow layer at a depth of 0.9 to 1.2 m (Rawls and Asmussen, 1973).

In preparation of the SWAT model (ver. 435) for the watershed, the weather input, including maximum and minimum daily air temperature, solar radiation, and relative humidity data, were obtained from a University of Georgia weather station located 5 km to the southeast of the watershed. Daily rainfall data from ten rainfall gauge stations were used to consider spatial rainfall distribution (Fig. 1). A digital elevation model (DEM) of 30 m grid size obtained from USGS webpage (http://seamless.usgs.gov/) was used to delineate subbasins, stream network, and topographic parameters for the SWAT input. Soil Survey Geographic (SSURGO) soil data obtained from the USDA-NRCS Soil Data Mart (http://soildatamart.nrcs.usda.gov/) were used to derive the soil related parameters. A land use layer digitized from agricultural field boundaries based on USGS 1999 ortho-photo quadrangles. Temporal changes in agricultural land use were identified based on harvested crop area data obtained from the USDA National Agricultural Statistics Service (NASS, http://www.nass.usda.gov/Data_and_Statistics/Quick_Stats/). Typical farming operations associated with each major crop were developed based on University of Georgia Extension recommendations. A minimum area to initiate the channel networks was set to 36 ha and then the burn-in option that the SWAT interface provides was used to match the channel networks defined based on DEM to stream lines digitized from USGS 1:24000-scale quadrangle maps. Hydrologic response units (HRUs) of SWAT were defined using land use and soil thresholds of 0%-0% so as to incorporate all land use and soil types in the modeling.

2. Hydrology and Water Quality Measurements

Hydrology, sediment, total N, and total P data collected in the watershed (Bosch et al., 2007a; Feyereisen et al., 2007b) were used for model calibration and validation. Daily estimates of sediment, total N, and total P concentrations were derived from existing weekly data (Feyereisen et al., 2007b) and then used to estimate daily loads based on daily streamflow measurements.

Stream water samples have been collected in subwatershed K on a weekly basis and analyzed for chloride, ammonium N, nitrate plus nitrite N, TKN, total P (total P), and dissolved molybdate reactive phosphorus (ortho-P) (Feyereisen et al., 2007b). Sediment data published by Sheridan and Hubbard (1987) (1974 to 1978 and 1979 to 1981) and by Hubbard et al. (1990) (1984 to 1986) were used in addition to a suspended sediment dataset for 2000 to 2004 (Feyereisen et al., 2007b). Nitrate plus nitrite N, ammonium N and ortho-P (make sure you refer to each of these chemical forms the same way throughout the manuscript) were determined using the EPA approved colorimetric techniques (Feyereisen et al., 2007b). Total Kjeldahl N and total P were determined
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Based on digestates of unfiltered samples. From the beginning of the record period through 1986, these analyses were conducted using a Technicon Autoanalyzer II instrument. In 1987, the instrument was changed to the Lachat flow injection analyzer. Estimates of daily TKN, total P, and sediment concentrations and loads derived from the weekly data by Feyereisen et al. (2007b) were used for comparison to model simulation results. In this study, the estimations of total N and total P daily loads were compared with the simulated ones. Total N was calculated as the sum of nitrate plus nitrite load and the TKN load.

3. Outlier Analysis

The daily TKN and total P datasets derived from the weekly sampling were examined for data quality. A previous study found considerable variability within the observed TKN and total P data, which could not be effectively explained with changes in climatic or land uses of the watershed (Feyereisen et al., 2008). The variability in the observed data can in part be explained by changes in the sampling and analytical techniques over the 26 year observation period (Bosch and Sheridan, 2007; Feyereisen et al., 2007b). Errors in measurement, sampling, and analytical techniques, can typically vary from +/- 3 to 42% for streamflow to +/- 4 to 421% for nitrate N (Harmel et al., 2006). These ranges constitute the best and worst case scenarios for various constituents. Manual sampling techniques and a lack of sample preservation can lead to particularly high measurement uncertainty (Harmel et al., 2006). As noted by Feyereisen et al. (2007b), grab sampling was used in subwatershed K from January 24, 1974 to August 15, 1974 and from April 2, 1993 to January 12, 1995. Preservation of the water samples has began being implemented since 2002 (Feyereisen et al., 2007b). Thus, measurements were reasonably expected to contain great errors and uncertainty due to the poor sampling and analysis processes.

The expected large measurement errors led us to conduct a statistical analysis to remove outliers from the following analysis so as to facilitate fair comparison between simulated and observed variables. The average TKN concentration observed at the outlet of subwatershed K from 1978 to 2003 was 2.06 mg/L with a coefficient of variation (CV) of 1.72 (n=2,887). Only nineteen (0.7%) out of the observations of the observed TKN concentrations were greater than 20 mg/L, and a disproportional number of these high values were observed in 1991 and 1997. A best fit theoretical probability distribution function was selected based on multiple goodness-of-fit tests including Kolmogorov-Smirnov, Anderson Darling, and Chi-Squared statistics, and it was used to identify outliers for TKN. The critical concentration for the upper limit was determined based on general extreme value distribution (Jenkinson, 1955) using weekly average TKN concentrations, and then it was used to identify outliers. A concentration of 2.68 mg/L corresponding to the 85% percentile of the fitted probability distribution function was selected as the critical upper limit concentration for TKN. This value is three times larger than the observed range of total N in subwatershed N of the Little River (Lowrance and Leonard, 1988). Outliers of the daily TKN concentration dataset were replaced with the long-term average concentration of subwatershed K (0.89 mg/L). The number of TKN concentrations exceeding 2.68 mg/L was 545 (19%) out of the total 2,887 samples collected from 1979 to 2003. The total P concentrations were also adjusted to obtain consistency across the daily dataset. The mean total P concentration for the same dataset was 0.15 mg/L (n=2,825), with a CV of 2.13. Total P measurements include organic P and ortho-P, and theoretically it should be greater or equal than the measured ortho-P. When measured total P concentration was less than ortho-P concentration due to measurement and lab analysis errors, total P was replaced by ortho-P. The number of replaced daily total P loads was 756 (27%) out of a 2,825.

4. Model Calibration and Validation

A step-wise (sequential) calibration procedure was conducted considering both in-stream and upland processes. In SWAT, the in-stream simulation module is activated when IWQ (the in-stream water quality code for considering in-stream nutrient and pesticide transformations) is set to 1. Measurements used for parameter calibration and SWAT parameters adjusted in the calibration steps are presented in Tables 1 and 2, respectively. The first step of the calibration process was to obtain an acceptable fit to the streamflow measured at the watershed outlet. The performance of calibrated model in the order of annual, monthly, and daily
Table 1. Measurements used for multi-scale model parameter calibration in this study

<table>
<thead>
<tr>
<th>Data type</th>
<th>Calibration</th>
<th>Validation 1</th>
<th>Validation 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow</td>
<td>○</td>
<td>○</td>
<td>○</td>
</tr>
<tr>
<td>Total Nitrogen (TN)</td>
<td>○</td>
<td>○</td>
<td>○</td>
</tr>
<tr>
<td>Total Phosphorus (TP)</td>
<td>○</td>
<td>○</td>
<td>○</td>
</tr>
<tr>
<td>Typical crop rotation*</td>
<td>Cr-Cr-Pn</td>
<td>Cr-Pn-Cr-Pn-Ct-Pn</td>
<td>Ct-Ct-Pn</td>
</tr>
</tbody>
</table>

* Cr: Corn, Ct: Cotton, Pn: Peanut

basis. An acceptable set of parameters for the streamflow simulation identified in the previous calibration practice was incorporated into the next calibration practice for parameters related to biomass growth at a field scale. The crop biomass calibration was conducted prior to the sediment calibration because crop residue on the soil surface that is estimated from total crop biomass was expected to greatly affect soil erosion estimates. Then, parameters for simulating upland erosion and nutrient transport estimates were calibrated. Upland sediment and nutrient processes were calibrated based on data obtained from the previous studies for the watershed (Sheridan et al., 1982; Lowrance et al., 1983; Lowrance et al., 1985b; Lowrance et al., 1986; Sheridan et al., 1999; Lowrance and Sheridan, 2005). The next calibration step considered sediment, total N, and total P transport through riparian forest buffers. Riparian buffers were reported to play a critical role of filtering sediment and thus reducing nutrient loads of the watershed (Sheridan et al., 1982; Lowrance et al., 1984). Thus, consideration of buffer transport processes is necessary to better represent watershed processes in parameter calibration. Finally, watershed-scale sediment and nutrient yields were calibrated considering in-stream processes represented by the default simulation of SWAT and the QUAL2E module in calibration. Model parameters were manually adjusted to maximize the overall agreement (represented by Nash-Sutcliffe efficiency coefficient and relative error) between observed and simulated time-series of variables of interest in calibration. It is worth noting that the objective of this study was to demonstrate a way to incorporate multi-scale hydrologic processes in model calibration rather than simply achieve the maximum model performance using a multi-objective optimization method.

Measurements made at the outlet of subwatershed K from 1979 to 1986 prior to implementation of recorded cropland conservation practices were used for calibration. Three-year (1884 to 1986) streamflow, total N and P measurements for in-stream processes were available for calibration. Sediment load measurements made between 1979 and 1981 were excluded in the calibration due to their extreme irregularity. The calibrated SWAT model was validated by comparing simulated and observed sediment and nutrient loads for two different 9-year simulation periods (1987 to 1995 and 1996 to 2004). Three different crop rotations including 3-year corn-corn-peanut rotation, 6-year corn-peanut-corn-peanut-cotton-peanut rotation, and 3-year cotton-cotton-peanut rotation were considered in the calibration and validation, and second validation periods, respectively. These rotations well represented crop type ratios reported by the USDA National Agricultural Statistics Service (NASS) (http://www.nass.usda.gov/Data_and_Statistics/Quick_Stats/). However, it is worth noting that spatial distributions of the rotations could not be considered in this modeling study due to lack of data. Thus, the lumped representation of the rotation practices would be one of the sources of modeling errors and uncertainty. Common management practices that were not associated cropping were incorporated into both calibration and validation periods. Simulation results for the first two years were not considered to reduce the impact of initial conditions on analysis results.

In addition to graphical comparisons, three statistics were used to evaluate the performance of the model, including percentage error (PE), monthly ratio of the root mean square error to the standard deviation of measured data (RSR)
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(Moriasi et al., 2007), and monthly Nash-Sutcliffe efficiency index (NSE) (Nash and Sutcliffe, 1970). For instance, model performance was considered satisfactory when monthly NSE is greater than 0.50, monthly RSR is less than 0.70, and PEs for streamflow, sediment, and nutrients are within ±25%, ±55%, ±70%, respectively, of measured values (Moriasi et al., 2007).

III. Results

1. Streamflow Calibration

In parameter calibration for streamflow, ESCO and GW_REVAP were used to minimize PE while trying to match the trends of simulated and observed annual streamflow. Subsequently, GWQMN was adjusted to maximize NSE for monthly streamflow. Finally, CN2, CH_N(2), and GW_DELAY were adjusted to maximize NSE for daily streamflow. Based on the previous modeling studies for the watershed, CN2 was adjusted to make the contribution of a baseflow to total streamflow 70% (Shirmohammadi, 1984), which led to decrease CN2 by 25.0% and 15.3% for non-crop and crop areas, respectively, from the default values. Field observations showed smaller variations of curve numbers over different types of row crops than that of curve numbers for forested areas (Neitsch et al., 2005a). An autocalibration study implemented for subwatershed F of the watershed (Fig. 1) showed wider changes in CN2 values from -27% to -50% (Van Liew et al., 2005). In addition, a manual calibration showed CN2 needed to be decreased by 9.9% to have a good agreement between the observed and simulated streamflow (Van Liew et al., 2007). Another SWAT calibration study for subwatershed K of the watershed suggested to decrease CN2 by 9.1% and 1.3% for non-crop and crop areas, respectively (Feyereisen et al., 2007a). The parameters identified from this calibration practice are presented in Table 2.

2. Biomass Calibration

Biomass simulation for the upland corn, cotton, and peanut crops was calibrated by adjusting radiation-use efficiency (BIO_E) and light extinction coefficient (EXT_COEF). There are other parameters that were believed to strongly affect biomass estimates such as optimal (T_OPT) and minimum temperature (T_BASE) for plant growth. However, they were not considered in the calibration because variability in those parameters for a specific crop was generally small. Target biomass values for corn, cotton, and peanut were set to 22,000, 7,000, and 11,000 kg/ha, respectively, based on the state database (http://www.swvt.uga.edu/). Simulation using the default values for BIO_E and EXT_COEF underestimated biomass compared to the target values. Typical ranges for BIO_E and EXT_COEF were determined based on studies of Flenet et al. (1996) for corn, Steglich et al. (2000) for cotton, Kiniry et al. (2005) for peanut, and Kiniry (2008- personal communication) for all three crops. Observed and simulated average crop biomass under the corn-corn-peanut rotation were 18,333 and 18,261 kg/ha, respectively (-0.4% error), in the calibration period.

3. Sediment and Nutrient Load Calibration

Sediment loads were calibrated by adjusting three different groups of parameters controlling: 1) field-scale erosion, 2) sediment delivery from fields to a stream, and 3) in-stream transport processes. The USLE factors of SWAT was used to simulate field-scale soil erosion. The USLE_LS factors determined by ArcSWAT based on the watershed topography were adapted to sediment and nutrient load simulation without adjustment. A support practice factor of 0.85 (USLE_P) was selected for crop areas, and it was determined considering USLE_P of 1.0 and 0.5 were used for plowing up-and-down slope and contour tillage, respectively (Lowrance et al., 1986). The default values of the soil erodibility (USLE_K) and average slope-length (SLSUBBSN) parameters were used because great variability was not found in their values when derived from the soil SSURGO database and DEM. Estimated USLE erosion based on previous monitoring studies ranged from 5.9 (Sheridan et al., 1982) to 7.5 tons/ha (Lowrance et al., 1986) for the subwatershed K. For this study, USLE soil erosion was simulated to 3.4 ton/ha without adjusting any other erosion-related parameters. A peak rate adjustment factor for sediment routing in the subbasin (ADJ_PKR) was used to calibrate sediment delivery from upland areas to streams. El values are estimated using an internal parameter (average fraction of total daily rainfall occurring in maximum half-hour period for month), which is
Table 2: Ranges and calibrated values of SWAT parameters

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Descriptions</th>
<th>Default</th>
<th>Range</th>
<th>Calibrated Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>ESCO.bsn</td>
<td>Soil evaporation compensation factor</td>
<td>0.95</td>
<td>0 1</td>
<td>0.518 0.518</td>
</tr>
<tr>
<td>ADJ_PKR.bsn</td>
<td>Peak rate adjustment factor for sediment routing in the subbasin (tributary channels)</td>
<td>1.0</td>
<td>0.5 2.0</td>
<td>1.75 1.75</td>
</tr>
<tr>
<td>PRF.bsn</td>
<td>Peak rate adjustment factor for sediment routing in the main channel</td>
<td>1.0</td>
<td>0.0 2.0</td>
<td>0.75 0.75</td>
</tr>
<tr>
<td>SCON.bsn</td>
<td>Linear parameter for calculating the maximum amount of sediment that can be re-entrained during channel sediment routing</td>
<td>0.0001</td>
<td>0.0001 0.01</td>
<td>0.0001 0.0001</td>
</tr>
<tr>
<td>PSP.bsn</td>
<td>Phosphorus availability index</td>
<td>0.4</td>
<td>0.01 0.7</td>
<td>0.040 0.053</td>
</tr>
<tr>
<td>BIO_E.crop</td>
<td>Radiation use efficiency or biomass-energy ratio ((\text{kg ha}^{-1})/(\text{MJm}^{-2}))</td>
<td>Variable</td>
<td>10 90</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Corn</td>
<td>39</td>
<td>39</td>
<td>39</td>
</tr>
<tr>
<td></td>
<td>Cotton</td>
<td>15</td>
<td>11</td>
<td>11</td>
</tr>
<tr>
<td></td>
<td>Peanut</td>
<td>20</td>
<td>17</td>
<td>17</td>
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<tr>
<td>EXT_COEF.crop</td>
<td>Light extinction coefficient</td>
<td>Variable</td>
<td>0 1</td>
<td>0.65 0.65</td>
</tr>
<tr>
<td></td>
<td>Corn</td>
<td>0.65</td>
<td>0.4 0.65</td>
<td>0.6 0.6</td>
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<tr>
<td></td>
<td>Cotton</td>
<td>0.65</td>
<td>0.5 1.3</td>
<td>0.5 0.5</td>
</tr>
<tr>
<td></td>
<td>Peanut</td>
<td>0.65</td>
<td>0.6 0.8</td>
<td>0.6 0.6</td>
</tr>
<tr>
<td>CH_N(2).rte</td>
<td>Manning’s “n” value for the main channel</td>
<td>0.014</td>
<td>0.01 0.3</td>
<td>0.015 0.015</td>
</tr>
<tr>
<td>GW_REVAP.gw</td>
<td>Rate of transfer from shallow aquifer to root zone</td>
<td>0.02</td>
<td>0.02 0.2</td>
<td>0.046 0.046</td>
</tr>
<tr>
<td>GW_DELAY.gw</td>
<td>Time required for water leaving the bottom of the root zone to reach the shallow aquifer (days)</td>
<td>31</td>
<td>0 500</td>
<td>0.036 0.036</td>
</tr>
<tr>
<td>GWQMN.gw</td>
<td>Threshold water depth in shallow aquifer for return to reach to occur (mm)</td>
<td>0</td>
<td>0 5000</td>
<td>66.7 66.7</td>
</tr>
<tr>
<td>HLIFE_NGW.gw</td>
<td>Half-life of nitrate in the shallow aquifer (days)</td>
<td>365</td>
<td>0 365</td>
<td>0.9 1.0</td>
</tr>
<tr>
<td>CN.mgt</td>
<td>Curve number for crop areas – Non-crop</td>
<td>Variable</td>
<td>35 98</td>
<td>-15.3% -15.3%</td>
</tr>
<tr>
<td></td>
<td>Crop</td>
<td></td>
<td></td>
<td>-25.0% -25.0%</td>
</tr>
<tr>
<td></td>
<td>Non-crop</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>FILTERW.mgt</td>
<td>Width of edge-of-field filter strip (m)</td>
<td>0</td>
<td>0 29</td>
<td>14 14</td>
</tr>
<tr>
<td>AI0.wwq</td>
<td>Ratio of chlorophyll-a to algal biomass ((\mu g\text{ chla} mg^{-1}\text{ algae}))</td>
<td>50</td>
<td>10 100</td>
<td>87</td>
</tr>
<tr>
<td>RHOQ.wwq</td>
<td>Algal respiration rate at 20°C (day^{-1})</td>
<td>0.3</td>
<td>0.05 0.5</td>
<td>0.08</td>
</tr>
<tr>
<td>AI1.wwq</td>
<td>Fraction of algal biomass that is N (mg N mg^{-1} algae)</td>
<td>0.08</td>
<td>0.07 0.09</td>
<td>0.07</td>
</tr>
<tr>
<td>AI2.wwq</td>
<td>Fraction of algal biomass that is phosphorus (mg P mg^{-1} algae)</td>
<td>0.015</td>
<td>0.01 0.02</td>
<td>0.02</td>
</tr>
</tbody>
</table>

IWQ: In-stream water quality code for considering in-stream nutrient and pesticide transformations

Calculated based on ADJ_PK. Increasing ADJ_PK to 1.75 increased both upland soil erosion and sediment delivery to the stream, resulted in HRU-level soil erosion of 6.72 ton/ha, which was very close to the target value of 6.70 ton/ha (average of 5.9 ton/ha and 7.5 ton/ha: Sheridan et al., 1982; Lowrance et al., 1986).

4. Sediment and Nutrient Transport through Riparian Buffer

Sediment reduction within the riparian forest buffers was examined while varying FILTERW, which represents edge-of-field filter strip width. A 14-m filter strip width produced
an 80% trapping efficiency in SWAT modeling (Neitsch et al., 2005b). Sheridan et al. (1999) showed that measured reductions of sediment loads by the RFB varied from 68% to 95% in the watershed, depending on types of applied management practices such as clear-cut, selectively thinned, and mature forest. FILTERW for each subbasin was determined by examining the fraction of the 14 m zone on each side of the stream with and without forest vegetation. The maximum FILTERW was set to 14 m in the case that all the buffer area consisted of forest, while the minimum of 0 m corresponded to no forest within the 14 m buffer zone. The variable FILTERW was considered by multiplying the fraction of forest area by 14 m to represent varying FILTERW over subbasins. For example, if 80% of the buffer area was forested, FILTERW would be 11 m (1.4 × 0.8). The fraction of forested area and FILTERW tend to increase for the higher order stream sections because of the greater floodplain areas found in the areas. A trapping efficiency of 76.0% was calculated by comparing HRU-scale sediment loads obtained using a FILTERW of 0 m and the variable FILTERW. The sediment loads delivered to the stream simulated were estimated to 0.41 ton/ha and 1.71 ton/ha with/without using the variable filter widths.

Filter strip and riparian buffer are vegetated areas located between crop fields and streams to reduce sediment and nutrient loadings from fields to stream. Riparian buffers usually represent areas forested along a stream while filter strips are placed immediately downstream from a source crop field. In this study, a filter strip simulation option of SWAT was used to represent riparian buffers of the study watershed since there is no option to directly simulate riparian buffers in SWAT. The riparian representation of this study could be justified by applying SWAT to a small watershed where most crop fields are immediately adjacent to streams.

5. In-Stream Sediment and Nutrient Process Calibration

In-stream sediment processes were considered after sediment enters streams from HRUs. Sediment was eliminated by adjusting parameters related to channel degradation processes, including the channel erodibility factor (CH_EROD) and the channel cover factor (CH_COV). Streamflow velocities are generally slow due to broad floodplains, low channel slopes, and dense vegetation in the watershed (Sheridan et al., 1982). The default values were used for parameters representing the cross-sectional channel geometry, including average width of main channel at top of bank (CH_W) and depth of main channel from top of bank to bottom (CH_D). An estimated CH_W at the watershed outlet was close to the width measured by Cathey (2005). Manning's roughness coefficient for the main channel (CH_N2) influenced sediment yield at the watershed outlet. Recommended ranges for natural streams are from 0.025 to 0.15 (Neitsch et al., 2005a). Recommended coefficients for channels in the Coastal Plain vary from 0.03 to 0.04 while coefficients for floodplains are slightly higher, 0.04 to 0.07 (Moyer and Bennett, 2007). Smaller ranges from 0.015 to 0.03 were reported for Coastal Plain streams in Maryland (Secrist et al., 2006). A coefficient of 0.015 was selected because increases in CH_N2 up to 0.12 increased sediment deposition in channels and substantially decreased daily flow peak for storm events. As a result, sediment deposition within the channel was simulated using a parameter related to channel sediment routing (SPCON) and the peak rate adjustment factor for sediment routing in the main channel (PRF) (Table 2). Sediment yields at the watershed outlet were overpredicted even when the minimum value of SCON (0.0001) was selected. Thus, PRF was adjusted to obtain the best simulation results. Simulated watershed-scale average annual sediment yield for the calibration period was 0.053 ton/ha, which resulted in a simulated annual sediment delivery ratio (SDR) of 0.79%. The estimated SDR value was relatively low compared to the estimated value (5.0%) for the subwatershed K using observed sediment yield and computed gross erosion using the USLE method (Sheridan et al., 1982).

6. Nutrient Yield Calibration

Results of the previous study implemented by Lowrance et al. (1985b) were used to compare simulated nutrient budgets. Watershed-scale average N input for subwatershed K was estimated to 56.6 kg/ha/year based on measured data from 1980 to 1981, which included 12.0 kg/ha/year of precipitation input and 44.6 kg/ha/year of agronomic input (Lowrance et al., 1985b). Simulated watershed-scale average...
N input was 86.6 kg/ha/year, including, 62.8 kg/ha/year of fertilizer input, 12.1 kg/ha/year of N loading from rainfall, and 11.7 kg/ha/year of symbiotic fixation. The difference in average N inputs could be explained by changes in fertilizer application rates over years. The fertilizer application rate used in the simulation was developed based on the current management practices while the measured application rates of Lowrance et al. (1985b) were based on early 1980’s management data. The study of Lowrance et al. (1985b) showed that 39.1% of the N inputs (22.1 kg/ha/year) was removed from the watershed through harvesting while 57.6% of N inputs (32.6 kg/ha/year) was retained in the watershed or lost in some unquantifiable way. Assuming 46% of total biomass is removed through harvest under a conventional tillage management (Khan et al., 2007; Kiniry et al., 2005), 37.4 kg/ha/year of N removal by harvest (43.1% of to N input) was indirectly estimated based on simulated uptake amount of 90.4 kg/ha/year.

The riparian ecosystem can be a nutrient sink. A previous study showed that about 67% of the waterborne N input, including precipitation and subsurface inputs from upland, were removed in the RFB (Lowrance et al., 1983) while 37% of N input in surface runoff was removed in the RFB (Lowrance and Sheridan, 2005). However, there is no option in SWAT to separately simulate N reductions within surface and subsurface regimes of RFBs. In this study, N reduction by the RFB was simulated based on the 14-m variable FILTERW, which was predefined from the sediment calibration. The simulated total N load from upland to stream without consideration of the riparian filters was 8.08 kg/ha/year, but it dropped to 4.45 kg/ha/yea with filters, which corresponds 44.9% reduction of N. For more detainted simulation of N transport processes through the riparian buffer, a parameter representing half-life of nitrate in the shallow aquifer (HLIFE_NGW) was used.

No measured in-stream N reduction rates are available in the watershed. However, Lowrance et al. (1985b) reported that 3.3% of the total watershed N inputs were transported by streamflow in subwatershed K. When the in-stream nutrient transformation was considered (IWQ=1), the associated parameters, such as ratio of chlorophyll-a to algal biomass (AI0), algal respiration rate at 20 °C (RHOQ), and fraction of algal biomass (AI1), were used to represent in-stream processes. These settings resulted in total N yield of 3.29 kg/ha/year at the watershed outlet, which is equivalent to 3.8% of the total N input. When IWQ is set to 0, no in-stream process is simulated. Thus, half-life of nitrate in the shallow aquifer (HLIFE_NGW) was adjusted to get a better watershed-scale total N yields, which led to total N yields of 2.96 kg/ha/year, equivalent to 3.4% of the total N input.

Lowrance et al. (1985b) reported that estimated watershed-scale average phosphorus input for the subwatershed K was 7.8 kg/ha/year, including 0.4 kg/ha/year from precipitation and 7.5 kg/ha/year from fertilizer application. The total watershed phosphorus input of 15.1 kg/ha/year was simulated without considering rainfall phosphorus inputs due to inability of SWAT. Lowrance et al. (1985b) estimated that 31.4% of the phosphorus input (2.5 kg/ha/year) was removed from the watershed through streamflow in subwatershed K. Similar to N, 34.6% of phosphorus removal by harvest (5.22 kg/ha/year) was indirectly calculated based on the simulated uptake amount (13.37 kg/ha/year) by assuming 46% of total biomass was removed during harvesting. About 56% of phosphorus input to RFB in surface runoff was removed through the RFB (Lowrance and Sheridan, 2005) while subsurface phosphorus reduction rate of RFB was about 24.8% (Lowrance et al., 1983). Opposite to the N, 99% of the simulated phosphorus was transported from upland to the channels through organic phosphorus (ORGP) and mineral phosphorus adsorbed to sediment (SEDP). In the study, a phosphorus reduction efficiency of the riparian filters was estimated to 77.1%, which was close to 76.0% for sediment using the variable FILTERW. Simulated phosphorus loads from the uplands to the stream were 0.71 kg/ha/year and 3.10 kg/ha/year with/without riparian filters. Phosphorus loads from upland area to the main channel was most sensitive to the phosphorus availability index (PSP). Lowrance et al. (1985b) showed that 14.7% of applied phosphorus (1.2 kg/ha/year) was transported by stream. In addition to AI0 and RHOQ, fraction of algal biomass (AI2) was used for calibrating in-stream phosphorus considering nutrient transformation (IWQ=1). Watershed-scale phosphorus yield was simulated to 0.63 kg/ha/year (4.2% of total P input), which meant in-stream reduction of 11.3% based on the phosphorus load rates of 0.71 kg/ha/year from upland to
stream after reductions in RFBs. When in-stream nutrient transformation was not considered (IWQ=0), PSP was adjusted to obtain the best total P yields at the watershed outlet, which resulted in 0.53 kg/ha/year (3.5% of the total P input) of total P yields at the watershed outlet, showing an in-stream reduction rate of 25.3%.

7. Performance of Calibrated SWAT in Calibration Period

For streamflow, PE for the calibration period was 1.2%, monthly RSR less than 0.7, and monthly NSE greater than 0.5 (Table 3). Fig. 2a displays the agreement between observed and simulated monthly and annual streamflow. Percent error, monthly RSR, and monthly NSE values for sediment yields were 1.6%, 0.78, and 0.40, respectively. Annual sediment loads were underestimated for a wet year (1984) and overestimated for dry years (1985 and 1986) with similar trends to the hydrology (Fig. 2b). For total N, PE for both settings with and without considering in-stream nutrient transformation was in the satisfactory range. Monthly RSR and NSE values considering in-stream nutrient transformation were better than those obtained without in-stream processes. The overall total N performance rate for the calibration with in-stream nutrient transformation was satisfactory while the performance rate for the calibration without the in-stream nutrient transformation process was unsatisfactory (Table 3). Calibration for total P showed similar results to total N; with better simulation results obtained utilizing in-stream nutrient transformation. While the goodness-of-fit statistics for total P indicated poor agreement, the model simulated trends in the total P fairly well (Fig. 2d). Fig. 2c and 2d show the monthly and annual trends of simulated and observed total N and total P obtained utilizing the QUAL2E in-stream algorithm, respectively. The annual trend of simulated and observed total N in Fig. 2c shows a similar trend to the hydrology results with overestimations for dry years such as 1981, 1985, and 1986. Fig. 2d shows a relatively good agreement between observed and simulated monthly total P loads except for February 1979. Based on the criteria established by Moriasi et al. (2007) for NSE and RSR, model performance for streamflow and total N were satisfactory while sediment and total P were unsatisfactory. Fig. 3 shows the comparison of monthly simulated total N and total P yields at the watershed outlet. For both total N and total P, in-stream nutrient processes increased nutrient yields when nutrient yields were small, while in-stream nutrient processes decreased yields when nutrient yields were high (Fig. 3).

8. Performance of Calibrated SWAT in Validation Period

The model performance statistics for the validation periods are provided in Table 4. Percent errors for streamflow were -9.0% for the first validation period and -19.9% for the second. Monthly RSR and NSE values for streamflow were within the satisfactory ranges for both validation periods (Table 4). Monthly and annual streamflow totals adequately represented the trends in the observed data for the two validation periods (Fig. 4a, Fig. 5a). Model performance for sediment was calculated for only the second validation period due to lack of the measurements for the first validation period. Percent error for sediment during the second validation period was -81.4%. Most of monthly sediment load measurements were underestimated in 2000, 2003, and 2004, which resulted in failure to reproduce the overall trend of observed sediment load for the years (Fig. 5b).
Fig. 2 Comparison of simulated and observed (a) total streamflow, (b) sediment loads, (c) total N, and (d) total P for the calibration period from 1979 to 1986.

Fig. 3 Comparison of simulated monthly (a) total N and (b) total P yields at the watershed outlet with/without in-stream nutrient transport simulation of QUAL2E for the calibration period from 1979 to 1986.
Table 4 Model performance measures for the two validation periods from 1987 to 1995 and 1996 to 2004

<table>
<thead>
<tr>
<th>Performance Measures</th>
<th>Streamflow (mm year⁻¹)</th>
<th>Sediment (ton year⁻¹)</th>
<th>Total N (kg year⁻¹)</th>
<th>Total P (kg year⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>87-95</td>
<td>96-04</td>
<td>87-95</td>
<td>96-04</td>
</tr>
<tr>
<td>Observed Total</td>
<td>456</td>
<td>326</td>
<td>-</td>
<td>225</td>
</tr>
<tr>
<td>Simulated Total</td>
<td>415</td>
<td>262</td>
<td>-</td>
<td>42</td>
</tr>
<tr>
<td>Percent Error (%)</td>
<td>-9.0</td>
<td>-19.9</td>
<td>-14</td>
<td>-81.4</td>
</tr>
<tr>
<td>Monthly RSR</td>
<td>0.28</td>
<td>0.33</td>
<td>-</td>
<td>1.00</td>
</tr>
<tr>
<td>Monthly NSE</td>
<td>0.92</td>
<td>0.89</td>
<td>-</td>
<td>0.01</td>
</tr>
</tbody>
</table>

Fig. 4 Comparison of simulated and observed (a) total streamflow, (b) total N, and (c) total P for the first validation period from 1987 to 1995

5b). The sediment loads for these three years were much greater than observations during the calibration period, which should lead to biased representation of the watershed hydrology and then the poor performance in the validation period. The reason for these higher loads has not known yet.

Percent errors for total N in both validation periods were satisfactory (±70%) with a PE of -35.1% for the first validation period and a PE of -26.4% for the second (Table 4). Based upon the criteria for RSR and NSE, both validation periods unsatisfactory. As mentioned earlier, measured TKN and subsequently measured total N were adjusted by replacing outliers in the TKN dataset with the long-term average TKN concentration. The measured annual total N loads for both validation periods without the outlier adjustments for TKN
are shown along with the adjusted data in Fig. 4b and Fig. 5c. There was a considerable deviation between the simulated and the unadjusted total N observations for both validation periods. For the unadjusted total N data, monthly total N loads increased substantially in the period from 1991 to 1994 and again in 1997 and 2004. These loads were significantly larger than the observed during the calibration period (Fig. 2c). Examination of land and cropping practices during the validation periods could not find a good explanation for the unexpected increase in the observed total N.

Percent errors in total P simulation were 32.8% and 8.5% for the first and second validation periods. Although simulated annual total P loads followed the overall fluctuations in the observed annual loads, simulated results failed to reproduce
Assessing Applicability of SWAT Calibrated at Multiple Spatial Scales from Field to Stream

the increase and decrease of measured annual loads in 1993 and 2001 (Fig. 4c and Fig. 5d). Measured annual total P loads for both validation periods without adjusting total P concentrations for outliers are shown in Fig. 4c and Fig. 5d. Total P loads were not significantly affected by removing the outliers, which is contrast to the case of the total N measurements. Even though annual total P loads were increased through the adjustment, simulated total P loads for 1993-1995 still were much greater than the adjusted data (Fig. 4d).

IV. Discussions

Overall, the hydrology and N simulations were satisfactory, but sediment and total P simulations were unsatisfactory for both validation periods. Percent error for the streamflow increased from 1.2% for calibration period to -9.0% and -19.9% for first and second validation periods, respectively. Hydrologic differences between the conditions observed during the calibration period and the two validation periods could contribute to the difference in the performance in the periods. Changes in land uses happened between the calibration and validation periods were not effectively reflected in the input data preparation, which might lead to the decrease in modeling performance for the validation periods. In addition, changes in sampling methods may cause inconsistency in the observed streamflow data. The device for measuring the depth of streamflow was changed from a float and pulley type device for 1967-1993 to a pressure transducer type device for 1993-present (Bosch and Sheridan, 2007). The error introduced into the streamflow data from these devices was expected to range from 0 to 2%, making up between 10 and 20% of the error in the validation results. Fig. 6 shows the comparison of correlations between annual rainfall and runoff/rainfall ratio for 1971-1992 and 1993-2006 on the subwatershed K, showing different correlations and variability in the two periods. Moreover, it was known that irrigated acres increased, which could have affected streamflow during the validation periods in the watershed.

As with the streamflow, consistency of observed water quality data is important for long-term model calibration and validation. As discussed, changes in sampling and analysis methods may cause variability in the observed data (Harmel et al., 2006). A summary of the long-term water quality monitoring program on the watershed is presented by Feyereisen et al. (2007b). The sampling methods in the subwatershed K included grab sampling (Jan-Aug 1974 and Feb 1993-Jan 1995), automated timed discrete (ATD) sampling (Aug 1974-Mar 1993), automated flow composite non-refrigerated (AFCN) sampling (Jan 1995-Dec 2002), and automated flow composite refrigerated (AFCR) sampling (Jan 2003-present) (Feyereisen et al., 2007b). Of these methods, grab and time based composite sampling have been shown to contribute the greatest sampling error (up to

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**Fig. 6** Correlation between annual rainfall and runoff/rainfall ratio for (a) 1971 - 1992 and (b) 1993 - 2006
Changes in the sediment sampling methods and analysis for the four different datasets may have also contributed to the different ranges of annual suspended sediment loads observed (Fig. 7). Samples collected from 1979 through 1986 were primarily from grab samples whereas those collected from 1999 through 2003 were from ATD samplers. The sediment loads during the calibration period (1979 through 1986) (Fig. 2) were approximately half of those observed during the second validation period (1996 through 2004) (Fig. 5), indicating a possible difference due to sampling methods.

Lab analysis techniques for N and P changed from the Technicon Autoanalyzer II (1974 through 1986) to Lachat flow injection (1987 to present) (Feyereisen et al., 2007b). Because of the change, measured TKN concentrations and their variations dramatically increased after 1987 (Fig. 8a). The increases in TKN concentrations led to increases in TKN load estimations from 1987 to present (Fig. 8b). Comparisons were also made between ortho-P and total P concentrations in the dataset. Ortho-P concentrations exceeding total P concentrations were detected more frequently after the instrumentation change in 1987, suggesting total phosphorous concentrations may have been underestimated since 1987.

Changes in personnel, sampling methods, and analytical methods could also affect observation data. Estimates of the expected error for streamflow was reported to vary from 7 to 65% (Harmel et al., 2006).
12% when both errors in the stage-discharge and the instrumentation were considered (Harmel et al., 2006). Estimates of errors associated with grab sampling vary from 25 to 50% while errors associated with automated sampling could vary from 0 to 33% (Harmel et al., 2006). Additional errors are introduced through methods of storing samples and analytical techniques. Thus, there is considerable uncertainty associated with model calibration and validation using long-term measurements. In addition, the model limitations for simulating dynamic N biogeochemical processes in riparian forest buffer areas might also contribute to difficulties in reproducing temporal trends of total N in streams.

V. Summary and Conclusions

The capability of SWAT for simulating long-term hydrology and water quality was evaluated using various data collected in subwatershed K of the Little River watershed. The SWAT model prepared for the study watershed was calibrated to measurements made at various spacial scales from 1979 to 1986, and then the calibrated model was validated by comparing the simulated and observed variables of interest made in two validation periods, 1987 to 1995 (first validation period) and 1996 to 2004 (second validation period). Two representative crop rotations of the watershed were assigned to each simulation period based on reported long-term changes in dominant crops of the watershed. Typical agronomic management schedules for conventional and conservation crop management practices were defined for each major crop. Watershed and field-scale sediment and nutrient transports along known loading pathways, including upland, riparian buffer, and in-stream processes, were considered in the calibration. The multi-scale calibration processes implemented in this study turned out to be useful in selecting appropriate parameters that represent detailed watershed processes. Riparian forest buffers were identified as an important hydrologic entity to be represented in the model. FILTERW and nitrate half-life in the shallow aquifer (HLIFE_NGW) were used to better represent complicated nitrogen transport processes through the riparian buffers.

The sediment and nutrient simulation results agreed with understandings about hydrologic characteristics of the watershed located in South Atlantic Coastal Plain. The multi-scale model evaluation processes described in this paper demonstrated a way to consider upland, riparian buffer, and in-stream processes and incorporate measurements made at different spatial scales into model calibration. HRU-scale outputs provided by the model calibrated considering multi-scale hydrologic processes showed good agreements with observations found in the literature, thus the calibrated model was expected to be useful in simulating field scale hydrologic processes. The calibration practices considering in-steam nutrient transformation processes represented by the QUAL2E module of SWAT provided better performance in predicting total N and total P than did the default simulation of SWAT. The calibrated model provided good performance in reproducing monthly streamflow of the watershed with PEs of -19.9 to 1.2 and NSEs of 0.89 to 0.93. However, the sediment and TP simulations provided unsatisfactory performance in the calibration and validation periods. Even when considering errors and uncertainty in measurements used for model evaluation, the unsatisfactory performance indicated that model failed to accurately simulate the sediment and nutrient transport processes, implying that the multi-scale calibration processes may not help obtain good model performance in predicting water quality at a watershed outlet.

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