

Simulating hydrological and nonpoint source pollution processes in a karst watershed: A variable source area hydrology model evaluation



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ARTICLE INFO

Article history:

Received 31 January 2016

Received in revised form 4 July 2016

Accepted 6 July 2016

Available online 16 July 2016

ABSTRACT

An ecohydrological watershed model can be used to develop an efficient watershed management plan for improving water quality. However, karst geology poses unique challenges in accurately simulating management impacts to both surface and groundwater hydrology. Two versions of the Soil and Water Assessment Tool (SWAT), Regular-SWAT and Topo-SWAT (which incorporates variable source area hydrology), were assessed for their robustness in simulating hydrology of the karstic Spring Creek watershed of Centre County, Pennsylvania, USA. Appropriate representations of surface water – groundwater interactions and of spring recharge – discharge areas were critical for simulating this karst watershed. Both Regular-SWAT and Topo-SWAT described the watershed discharge adequately with daily Nash-Sutcliffe efficiencies (NSE) ranging from 0.77 to 0.79 for calibration and 0.68–0.73 for validation, respectively. Because Topo-SWAT more accurately represented measured daily streamflow, with statistically significant improvement of NSE over Regular-SWAT during validation (p -value = 0.05) and, unlike Regular-SWAT, had the capability of spatially mapping recharge/infiltration and runoff generation areas within the watershed, Topo-SWAT was selected to predict nutrient and sediment loads. Total watershed load estimates (518 t nitrogen/year, 45 t phosphorus/year, and 13600 t sediment/year) were within 10% of observed values (−9.2% percent bias for nitrogen, 6.6% for phosphorous, and 5.4% for sediment). Nutrient distributions among transport pathways, such as leaching and overland flow, corresponded with observed values. This study demonstrates that Topo-SWAT can be a valuable tool in future studies of agricultural land management change in karst regions.

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1. Introduction

Hydrologic and pollution transport processes in watersheds with karst hydrology are complicated by underground networks of bedrock fractures and solution cavities. The subsurface heterogeneity and presence of preferential flow paths enhance groundwater recharge (Hartmann et al., 2015). Resulting high infiltration capacities limit surface runoff and reduce actual evapotranspiration during wet conditions (Malard et al., 2016). Sub-surface channels within karst aquifers accelerate groundwater flow (Parizek, 1984; Fulton et al., 2005) as compared to water movement through non-karst aquifers. Moreover, springs recharged outside the topographic watershed boundary can discharge inside the basin and

vice versa. However, the high storage capacities of karst aquifers tend to sustain stream channel baseflow and decrease hydrograph peaks as compared to the hydrology of similar non-karst watersheds (Fulton et al., 2005; Baffaut and Benson, 2009; Amatya et al., 2013). Additionally, the karst geology of northeastern USA promotes rapid infiltration in areas with karst features like sink holes and solution cavities along with saturation excess surface runoff in topographic lows like near-stream areas, where soils tend to be poorly drained and regional groundwater systems are most likely to intersect the land surface (Fulton et al., 2005; O'Driscoll and DeWalle, 2006; Buda and DeWalle, 2009). This non-uniform spatial arrangement of runoff generation processes, or variable source area (VSA) hydrology, becomes a primary driver of surface runoff generation and nutrient loss throughout the region (Srinivasan et al., 2002; Easton et al., 2008).

Hydrologic and water quality models of karst watersheds must incorporate increased complexity in the simulation of hydrolog-

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Table 1

Summary of relevant modeling studies in karst watersheds.

Study	Model used	Key issue	Techniques	Key results
1 – Afinowicz et al. (2005)	SWAT	Brush management on water budget	Modification of baseflow code for rapid groundwater movement	NSE = 0.09–0.4
2 – Amatya et al. (2011)	SWAT	Streamflow and embayment	Addition of subsurface point source (spring) and modification of baseflow estimation method	NSE = 0.29–0.91
3 – Amatya et al. (2013)	SWAT	Phosphorous loading	Addition of subsurface point source and modification of baseflow estimation method	PBIAS = 13
4 – Baffaut and Benson (2009)	SWAT	Flow and pollutant transport	Splitting groundwater recharge and using high hydraulic conductivity values	NSE = 0.24–0.56
5 – Fleury et al. (2007)	2-Reservoir	Rainfall-discharge relationship	Transfer function consisting of two reservoirs: a slow discharge reservoir for low flow and a rapid discharge reservoir for high flow	NSE = 0.92
6 – Jourde et al. (2007)	Rainfall-runoff	Groundwater contribution to surface flow	Hydrodynamic analysis of groundwater flow into a standard rainfall-runoff model	Partly good fit
7 – Kourgialas et al. (2010)	Integrated karstic	Hydrology of complex geomorphology	A 2-reservoir model: water flow through the karst network was determined as a function of karstic area and as a fraction of inflow	Good agreement
8 – Martinez-Santos and Andreu (2010)	Lumped & distributed	Natural recharge in semiarid climate	Lumped model approach and distributed model approach with application of a standard finite-difference code	Reproduced recharge accurately
9 – Nikolaidis et al. (2013)	SWAT	Hydrological and geochemical processes	A modified karst flow model: upper reservoir and lower reservoir system	NSE = 0.62
10 – Palanisamy and Workman (2015)	SWAT	Flow through sinkholes located in streambeds	Application of orifice flow method incorporated into SWAT model for sinkhole modeling	PBIAS = –22.3 NSE = 0.57–0.87
11 – Rozos and Koutsoyiannis (2006)	Multicell/MODFLOW	Groundwater level	Application of conduit flow approach using Manning's equation	Improved model performance
12 – Salerno and Tartari (2009)	Wavelet analysis	Baseflow component	Application of the statistical method of Wavelet analysis	NSE = 0.56–0.66
13 – Spruill et al. (2000)	SWAT	Stream discharge	Regular SWAT model parameterization	NSE = –0.04–0.19
14 – Yactayo (2009)	SWAT	Hydrological process	Allowing overland flow and lateral flow from upstream areas to recharge the sinkholes	NSE = –34.8–0.37
15 – Zhang et al. (2011)	Distributed	Hydrological process	Integrating mathematical routings of porous Darcy flow, fissure flow, and underground channel flow	R ² > 0.75

ical processes, as compared to those of non-karst watersheds, in order to accurately capture the influence of karst groundwater flow on surface water flow and quality (Jourde et al., 2007; Salerno and Tartari, 2009). In particular, many topographically-driven hydrologic models tend to overestimate actual evapotranspiration and surface runoff and thereby underestimate karst recharge (Hartmann et al., 2015). Moreover, a fully distributed model cannot be used if the complete network of the subsurface conduit system is unknown. A summary of relevant simulation studies in karst watersheds (Table 1) illustrates five distinct modeling approaches: (a) conduit flow via Manning's equation, (b) distributed groundwater pools, (c) groundwater storage represented as a reservoir, (d) distributed hydrologic models coupled with conduit routing, and (e) semi-distributed hydrologic response units that interact independently. Some of these studies focused mainly on prediction of flow through sinkholes (Palanisamy and Workman, 2015) and some were unable to incorporate the impacts of land use changes on hydrology and water quality at the watershed scale (Fleury et al., 2007; Jourde et al., 2007).

The semi-distributed Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998; Neitsch et al., 2011; Arnold et al., 2012; Winchell et al., 2013) is capable of simulating stream discharges and nonpoint source pollution in a watershed using long-term cli-

mate and land use data (Kaini et al., 2012; Jeong et al., 2013). Different versions of SWAT have been used for watersheds with karst features in a limited number of cases with a wide range of daily Nash–Sutcliffe efficiencies (NSE) (Table 1). A modified version of SWAT, initially termed SWAT-VSA by Easton et al. (2008) but hereafter called Topo-SWAT, incorporates the topographic wetness index (Beven and Kirkby, 1979) and has been used satisfactorily in a number of cases for simulating hydrology and phosphorus (P) transport (White et al., 2011; Pradhanang et al., 2013; Woodbury et al., 2014; Collick et al., 2015; Winchell et al., 2015) for watersheds with VSA hydrology. However, to our knowledge, Topo-SWAT has not yet been applied in a watershed with substantial karst geology. In the current study, we hypothesized that the surface runoff and baseflow processes in a karst watershed with VSA hydrology would be better simulated by Topo-SWAT than standard SWAT (hereafter Regular-SWAT) due to Topo-SWAT's ability to capture spatial differences in recharge/infiltration and runoff generation throughout the basin. Approaches taken in this SWAT modeling study could be adopted in karst regions of the northeastern US and elsewhere.

The overall goal of the study was to develop a simulation tool for a karst watershed with VSA hydrology that dynamically links surface water, groundwater, and field-level land use changes to

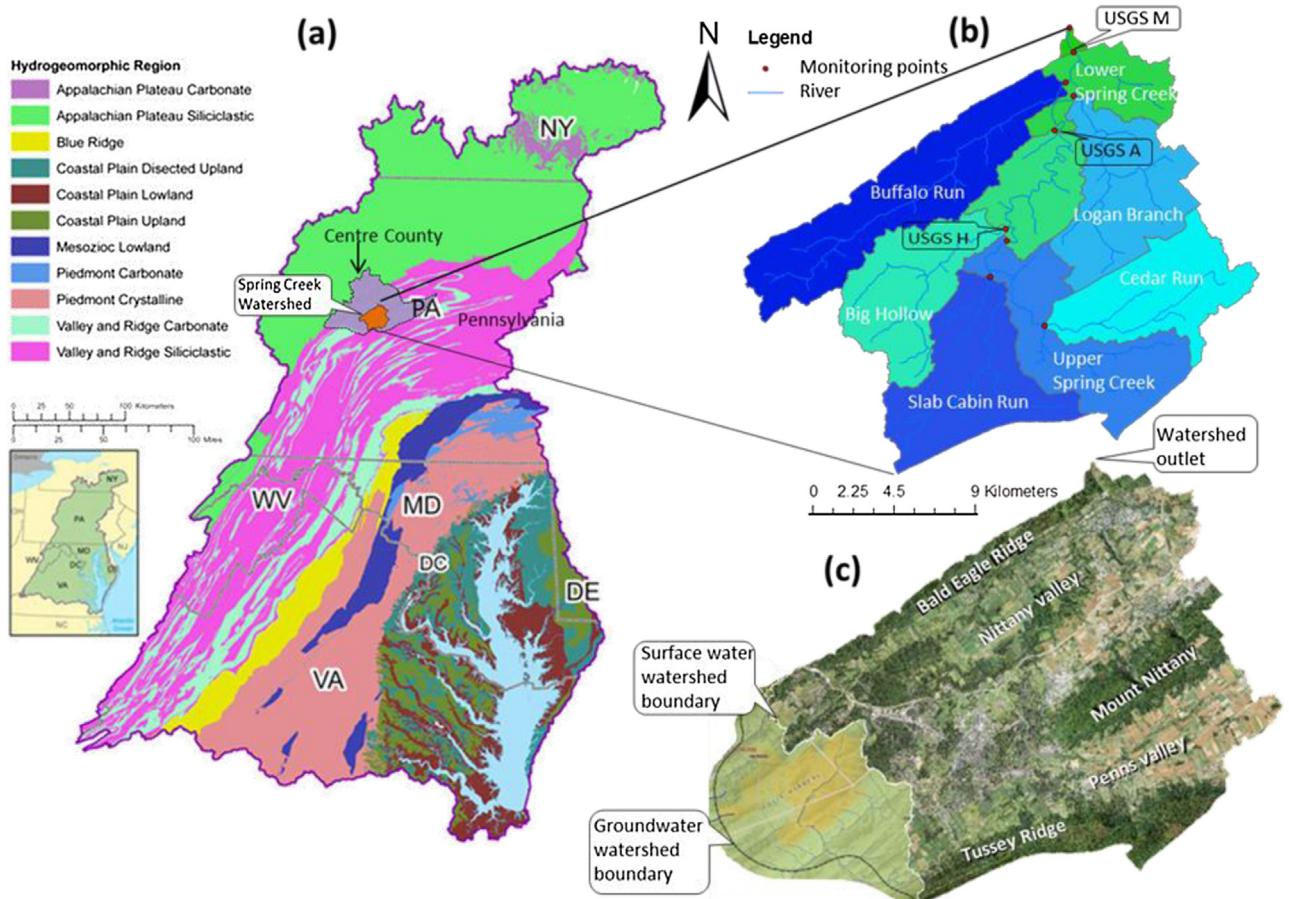


Fig. 1. Location of Spring Creek watershed, Centre County, Pennsylvania, USA: (a) Hydrogeomorphic regions in Chesapeake Bay Watershed, (b) Major subbasins and USGS gage stations (M– Milesburg, A– Axemann, and H– Houserville) in Spring Creek watershed, and (c) Location of valleys, mountains, and watershed boundaries (Source: United States Geological Survey and the Chesapeake Bay Program website <http://www.chesapeakebay.net/maps/map/hydrogeomorphic.regions>).

predict nutrient and sediment losses at both field-management and watershed scales. The specific objectives of the study were to:

- (i) assess the ability of Regular-SWAT and Topo-SWAT to adequately simulate the VSA hydrology of a karst watershed; and
- (ii) examine the accuracy of the best-fit hydrological model in estimating nutrient and sediment loads while taking into consideration karst hydrological processes.

2. Materials and methods

2.1. Study watershed overview

Spring Creek watershed (Hydrologic Unit Code 02050204; 40°40'–40°59'N, 77°38' – 78°00'W), located in Centre County, Pennsylvania in the northeastern USA (Fig. 1a), was chosen for this study as a representative karst watershed with VSA hydrology. Spring Creek is a 370 km² basin situated in the Appalachian Ridge and Valley physiographic province of the upper Chesapeake Bay watershed. Spring Creek is a fourth-order stream that discharges into Bald Eagle Creek, a tributary to the West Branch Susquehanna River and ultimately, the Chesapeake Bay estuary. Spring Creek watershed has three US Geological Survey (USGS) gauging stations, namely Milesburg (USGS ID: 1547100), Axemann (USGS ID: 1546500), and Houserville (USGS ID: 1546400) (Fig. 1b).

2.1.1. Topography and climate

The watershed mean elevation is approximately 370 m above mean sea level (amsl). The most prominent topographic features in the watershed are the Bald Eagle, Tussey, and Nittany Mountain ridges with reliefs of 550–675 m amsl (Fig. 1c), whereas the stream channel bottoms are approximately 280 m amsl. Climate is temperate with hot, humid summers and cold winters with a mean annual temperature of 10.1 °C and an average annual precipitation of 1060 mm. Annual actual evapotranspiration varied between 268 mm and 758 mm during 1968–1994, and an average annual pan evaporation coefficient of nearby area was 0.76 (Farnsworth et al., 1982).

2.1.2. Karst geology and hydrology

Karst geologic formations dominate the Spring Creek watershed (Buda and DeWalle, 2009; Brooks et al., 2011; Piechnik et al., 2012). The area of the Spring Creek watershed as defined by its groundwater boundary is 450 km², which is 22 percent larger than the surface-water watershed area (Giddings, 1974; Wood, 1980) (Fig. 1c). There is only one known, large karstic flow that recharges outside and discharges inside the watershed (Fulton et al., 2005; SCWA, 2013). Other local small karstic subsurface flow systems, perched, and losing streams are regularly seen, especially in headwater regions during dry periods (O'Driscoll and DeWalle, 2006). The shallow soils in the forested uplands are underlain by low permeability sandstone and shale bedrock, which drain runoff as subsurface flow along the bedrock surface. This subsurface flow eventually returns to the surface in springs at the base of hillslopes

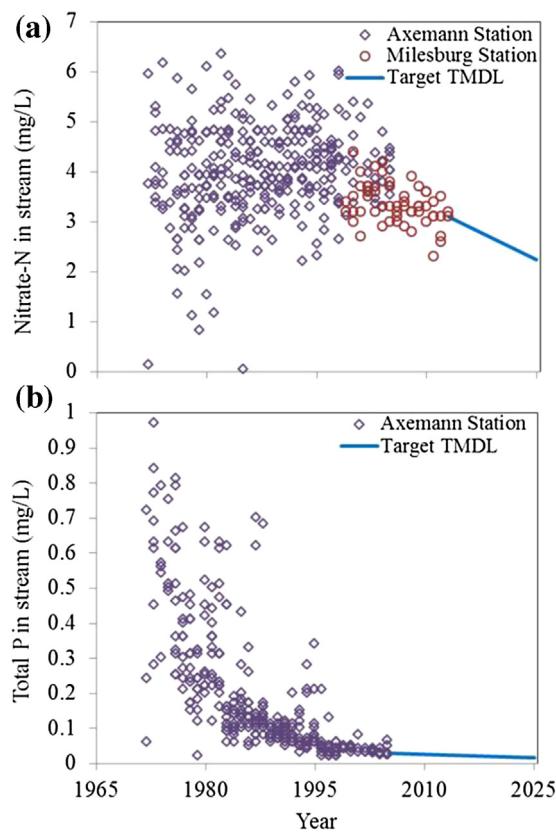


Fig. 2. Concentration of (a) nitrate-N in stream at Axemann and Milesburg gage and (b) total P in stream at Axemann gage; each with corresponding target Total Maximum Daily Load (TMDL) in Spring Creek watershed to meet the Chesapeake Bay TMDL target by 2025 (Source: United States Geological Survey; CCMP, 2015; SCWA, 2013).

and in the near-stream zone (Fulton et al., 2005; Buda and DeWalle, 2009) creating the variably saturated areas that increase the likelihood of saturation excess runoff during rain storms. A water balance study of Spring Creek watershed based on 1968–1994 data showed that 33–60% of precipitation became actual evapotranspiration and 33–58% entered the stream, of which 9–16% was direct runoff and 87–91% baseflow (Taylor, 1997).

2.1.3. Dominant land use

Land use in Spring Creek watershed is 34% agriculture, 23% developed, and 43% forest. General crop rotations in the watershed include corn, soybean, winter wheat, small grain (barley, sorghum, rye, canola, and oat), alfalfa, and dry bean. Hay and pasture are sustainable options in the areas with karst formations because they do not require the deep, rich soils that productive row crops do. Substantial urban development increased the impervious surface of the watershed from 3.1% in 1938 to 13.3% in 2006 (Brooks et al., 2011). Detention ponds have been added in some places to mitigate the quantity of runoff generated from areas of intensive impervious development. In response to various watershed management initiatives nutrient concentration in stream has decreased over the last three to four decades (Fig. 2).

2.2. Model descriptions

2.2.1. Regular-SWAT

The Soil and Water Assessment Tool (SWAT) is a continuous, physically-based, semi-distributed watershed model developed by USDA-ARS to predict the impact of agricultural management practices on water, sediments, and agricultural chemical yields among

different soil types and land uses (Arnold et al., 1998, 2012; Neitsch et al., 2011). The model uses spatial information on topography, soil properties, land use, climate, and management practices. The model subdivides a watershed into a number of subbasins; each subbasin is characterized by one or more hydrological response units (HRUs—particular combinations of land uses, soils, and surface slopes). In SWAT, unit areas within a subbasin having the same land use, soil type, and slope class have identical hydrological properties and are grouped into a single HRU, ignoring location within a subbasin, unless explicitly configured otherwise. In this study, agricultural management practices are explicitly defined at the field-level of the land use layer so that the spatial locations of HRUs are explicitly maintained. This enables comparison between Regular-SWAT and Topo-SWAT in their spatial predictions of hydrology and water quality.

Daily precipitation, maximum and minimum daily temperature, solar radiation, wind speed, and relative humidity values are used by SWAT for hydrological balance. Within SWAT, runoff volume is estimated daily from each HRU using the modified Soil Conservation Service (SCS) curve number (CN) method (Williams and Laseur, 1976), and sediment yield is estimated daily from each HRU using the modified universal soil loss equation (MUSLE; Williams and Berndt, 1977). The nutrient (N and P) cycle representations and crop growth modeling in SWAT are similar to those in the Environmental Policy Integrated Climate (EPIC) model (Williams et al., 1984).

The model provides hydrological and nutrient balance and sediment loss for each HRU, reach/river section, and subbasin. Different nutrient pools, such as total applied, atmospheric deposition, fixation, denitrification (for nitrogen), leaching, mineralization, immobilization, crop uptake, runoff, and groundwater contribution to surface water, are output separately at a daily, monthly, or annual time step. Definitions of model outputs and detailed descriptions can be found in Arnold et al. (2012) and Winchell et al. (2013).

2.2.2. Topo-SWAT modifications to Regular-SWAT

Saturation excess surface runoff from VSAs is a common runoff generation mechanism in the Spring Creek watershed (Fulton et al., 2005; Buda and Dewalle, 2009). Incorporating a topographic wetness index (TI) (Beven and Kirkby, 1979) into a hydrologic model is the most common way of representing the VSA hydrology (Easton et al., 2008). Topographic indices (TI) can be generated from each upslope contributing area (α) draining through any given point and the local slope gradient ($\tan \beta$) using the following equation (Beven and Kirkby, 1979; Easton et al., 2008):

$$TI = \ln(\alpha / \tan \beta) \quad (1)$$

The TI classes indicate the saturation potential of a landscape unit and subsequent likelihood of runoff generation. Following the method of Easton et al. (2008) and Collick et al. (2015), a TI layer was prepared and reclassified into ten equal-area wetness classes ranging from a wetness class of "1" (10% of the watershed with the lowest runoff potential) to a wetness class of "10" (10% of the watershed with the highest runoff potential). An automated ESRI ArcMap toolbox 'TopoSWAT' (Fuka, 2013) was used to overlay the FAO-UNESCO Digital Soil Map of the World layer (FAO, 2007) with the wetness class layer, thereby generating a single substitute GIS layer and associated lookup tables for the Regular-SWAT slope class and soil layers. In Regular-SWAT, hydrologic differences were simulated at a much coarser scale, i.e., the HRU scale instead of the TI scale.

2.3. Datasets for parameterization, calibration, and validation

A digital elevation model (DEM) with a resolution of 1/3 arc-second (10-m) was prepared for the watershed by mosaicking nine

DEM sections downloaded from the USDA-NRCS geospatial data gateway (<https://gdg.sc.egov.usda.gov/GDGOrder.aspx>). The elevation accuracy (root mean square error) for nationwide data of USGS DEMs is ± 2.44 m. Cropland Data Layers (CDL) from 2008 to 2014, with 30-m resolution, were collected from the USDA-NASS geospatial data gateway (<http://nassgeodata.gmu.edu/CropScape/>). The same elevation and cropland data were used for both Regular-SWAT and Topo-SWAT.

Soil attributes were imported from the Soil Survey Geographic Database (SSURGO) for SWAT (<http://swat.tamu.edu/>), and the 30-m soil spatial layer and its lookup table were formatted for use in Regular-SWAT. For Topo-SWAT (Easton et al., 2008), a spatial combination of the FAO-UNESCO Digital Soil Map of the World (FAO, 2007) and topographically-derived wetness classes was used instead of the SSURGO soil layer, as discussed in Section 2.2.

Weather data required for the study were obtained from the Chesapeake Community Modeling Program (<http://ches.communitymodeling.org/>) and from the Pennsylvania State Climatologist (PSC, 2014). Daily total streamflow data for the three USGS gage stations in the watershed were obtained from the USGS website (<http://waterdata.usgs.gov/pa/nwis/rt>). Seasonal sediment, nitrogen (N), nitrate-N, phosphorous (P), and dissolved oxygen concentrations in the stream during 1972–2005 were obtained from the Chesapeake Community Modeling Program (CCMP, 2015) and seasonal nitrate-N concentrations in the stream during 1999–2013 were obtained from a local community water monitoring program operated by the Spring Creek Watershed Association (SCWA, 2013).

Crop types and field-scale crop rotations were derived from the CDLs for years 2008–2014. Details on agricultural operations, such as tillage, manure/fertilizer application, sowing, starter fertilizer, top dressing/side dressing of fertilizer, harvesting, and killing/end of growing season, were determined from the Agronomy Guide of Pennsylvania (The Agronomy Guide, 2015).

2.4. Model preparation, calibration, and validation

A DEM-based overland flow network calculated through the ArcSWAT interface was used to define the resolution of the stream network. A stream generation threshold of 25 ha indicates that each stream cell is formed by an overland contributing area of at least 25 ha (i.e., 2500 10-m contributing cells). At a 25-ha threshold, the stream network for Spring Creek watershed closely mirrored that of the USGS topographic maps, aerial photos, and known permanent streams. The USGS gage stations and point sources were added to the stream network manually. For both models, all soil, land use, and slope classes were maintained in the final HRU layer by using a 0% deletion threshold in the ArcSWAT interface, resulting in 8754 total HRUs. After defining eight 8-year crop rotations for the watershed, each agricultural HRU of the model was populated with an appropriate crop rotation based on the standing crop in the CDL in 2012 so that the actual spatial and temporal distribution of the crops on the cultivated part of the watershed was maintained over the simulation period. All data were processed and incorporated into the model through the Arc-SWAT interface as described in detail by Neitsch et al. (2011), Arnold et al. (2012), and Winchell et al. (2013).

Both SWAT projects for Spring Creek watershed were manually calibrated, using trial-and-error, by adjusting one parameter at a time. At each step, hydrological component responses (surface runoff, baseflow, lateral flow, groundwater recharge, groundwater contribution to stream), individually and as a whole, were evaluated through visual and analytical evaluation of the spatial and temporal output. The calibration process was guided by local knowledge of the watershed processes, previous literature, and suggestions from the SWAT documentation manuals. The water-

shed model was calibrated for the 2002–2007 period and validated from 2008 to 2013, against daily streamflow data from the USGS gage stations, all available measurement data, and literature for the watershed. Important parameterization decisions are summarized in Table 2 and explained at length in the remainder of this section, followed by explanation of evaluation methods.

To represent the processes of surface water loss to groundwater and reappearance of groundwater in the karst aquifer, the initial curve number was reduced by 25% (final average curve number CN2 = 47) and the surface water lag coefficient (SURLAG) was reduced from 4.0 to 0.2 (Table 2). The SWAT algorithm determines baseflow from the total amount of water infiltrating through the soil profile but does not allow deep groundwater to become baseflow (Neitsch et al., 2011). To reproduce the highly permeable karst aquifer and sustained baseflow, the GW_DELAY factor was decreased from the default (31 days) to a value of 1 day. This adjustment was also made by Afinowicz et al. (2005) for another karst type watershed. The baseflow recession factor ALPHA_BF (0.011 day) was calculated from the observed daily streamflow of the preceding 15 years. Routing of runoff or lateral flows from HRU to HRU is currently absent in both SWAT and Topo-SWAT; rather the lateral flow is calculated for each HRU and is added directly to the subbasin reach. These lateral flows were lagged by specifying the lateral flow travel time (LAT_TTIME) to a value of 10 days. The FAO soils were shallower than the corresponding SSURGO soils and experimental knowledge of the watershed; thus, FAO soil depths used in Topo-SWAT were adjusted accordingly. All these adjustments helped to predict the sustained spring discharges and groundwater contributions to the streams.

To further augment the baseflow and match the observed data, the deep aquifer percolation fraction (RCHRG_DP) and groundwater evaporation coefficient (GW_REVAP) were tuned to 0.001 and 1.0, respectively. Springs that recharge outside of the watershed but discharge inside of Spring Creek watershed were added as point sources in the model. The lateral flow lag time adjustment (LAT_TTIME = 10 days) was important for predicting nutrient cycling because lateral water movement from the karst HRUs directly controls soil moisture. In a preliminary run of the hydrologically calibrated Topo-SWAT model, very little nitrate-N was coming through the groundwater to the streams despite a considerable amount of nitrate-N leaching. Increasing the half-life of nitrate-N in groundwater from zero (default) to 200 days, based on regional karst studies by Fishel and Lietman (1986) and USGS (1997), improved prediction of nitrate-N concentration in groundwater. Parameter sensitivity indices were calculated by sequential uncertainty domain parameter fitting (SUFI-2), a built-in algorithm in calibration and uncertainty program SWAT-CUP (Abbaspour, 2014).

Output was evaluated using standard statistical methods (t-test at the 95% confidence level) in Microsoft Excel and ESRI ArcGIS. Predicted hydrological accuracy was primarily evaluated by the visual fitness of the hydrograph and two frequently used measures of accuracy: Nash–Sutcliffe efficiency (NSE) and percent of bias (PBIAS) (Moriasi et al., 2007). In addition, precision of the simulated values with respect to the observed was assessed using the coefficient of determination (R^2). Observed in-stream sediment, N, nitrate-N, P, and dissolved oxygen concentrations were compared with modeled values to examine the accuracy of the model predictions but we did not include NSE due to the brevity and infrequency of the observed water quality data. Since the Spring Creek watershed is one of the delineated Chesapeake Bay subwatersheds modeled by the Chesapeake Bay program, we compared the annual nutrient and sediment loads from SWAT model output with the estimated annual load from the Chesapeake Bay Program (CCMP, 2015). Components of nutrient loss, such as loss by individual land use type and pathways of losses, were also compared with liter-

Table 2

Calibrated parameters and parameter sensitivities of SWAT for Spring Creek watershed.

Parameter name	Definition of parameter	Calibrated Value	Parameter sensitivity indices ^a		
			Ranking	t-Stat	P-value
SURLAG	Surface runoff lag time (day)	0.2	1	-162	0
SOL_AWC	Available water capacity of the soil layer (mm/mm)	0.1	2	-23.8	0
ALPHA_BF	Baseflow alpha factor: Baseflow recession constant	0.011	3	-2.71	0.01
CN2 ^b	SCS runoff curve number for average moisture	47	4	-2.18	0.03
ESCO	Soil evaporation compensation factor	0.86	5	1.46	0.15
EPCO	Plant uptake compensation factor	0.52	6	-1.35	0.18
GW_REVAP	Groundwater re-vaporization coefficient	0.02	7	-1.18	0.24
LAT_TTIME	Lateral flow travel time (day)	10	8	0.8	0.43
GW_DELAY	Groundwater delay time (day)	1	9	-0.77	0.44
DEPTH_IMP	Depth to impervious layer of water table (mm)	3450	10	0.74	0.46
RCHRG_DP	Deep aquifer percolation fraction	0.001	11	0.52	0.6
GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur (mm)	77	12	0.44	0.66

^a Parameter sensitivity indices were calculated by sequential uncertainty domain parameter fitting (SUFI-2), a built-in algorithm in calibration and uncertainty program SWAT-CUP (Abaspour, 2014).

^b CN2 is a function of the soil's permeability, land use, and antecedent soil water conditions.

ature values to confirm that the model adequately described the watershed processes.

3. Results

3.1. Watershed discharge estimation

3.1.1. Model performance for hydrology

Parameters dealing with groundwater recharge and its contribution, in terms of both time and magnitude, to the sustained baseflow of the stream were very important for capturing the karst nature of the watershed (Table 2). The calibrated values of curve number, the surface water lag coefficient, lateral flow travel time, groundwater delay factor, adjusted FAO soil depth, and calculated baseflow recession factor adequately reproduced the observed ratio of runoff and infiltration and the sustained baseflow. Both versions of SWAT (Topo-SWAT and Regular-SWAT) satisfactorily simulated daily streamflow at all three USGS gauging stations in Spring Creek watershed (Milesburg, Axemann, and Houserville) for both the calibration period of Jan 2002–Dec 2007 and validation period of Jan 2008–Dec 2013 (Fig. 3). The models also predicted the pattern and magnitude of baseflow and captured most of the streamflow peaks satisfactorily. In a few cases (2004, 2005, 2007, and 2011) discharge peaks were underestimated by both models. A tendency for baseflow over-prediction in the upper catchment (Houserville) was observed for Regular-SWAT. On the other hand, Topo-SWAT predicted the baseflow at all three gauging stations with equivalent accuracy.

The Topo-SWAT model performed more accurately, compared with the measured data, than did Regular-SWAT for daily streamflow prediction in the validation period: NSE values of 0.73 and 0.68, respectively (Table 3). The PBIAS values for Topo-SWAT for simulating daily streamflow were also within an acceptable range, -2.8 and -3.7 for calibration and validation period, respectively (Table 3). The negative values indicate that the flow was under-predicted. The PBIAS values were lower at the upper gauging stations (-0.5 to -1.0) than the final outlet at Milesburg (-5.6 to -7.4) (data not shown in Table). Flow was only overestimated at the Houserville station during the validation period (PBIAS of 3.5 for Topo-SWAT and 5.5 for Regular-SWAT).

3.1.2. Hydrological balance estimation

Simulation results showed that 49% of total precipitation was converted to streamflow, and only 17% of streamflow was attributed to surface runoff. Those predictions were very similar to the observed values (based on a catchment water balance): 53% of precipitation became streamflow during 2002–2013 (USGS

gauge data). Parameters related to the groundwater-surface water interaction (water balance ratio), such as surface runoff lag time, available water capacity of the soil layer, and SCS curve number were highly sensitive (Table 2). Groundwater dynamics represented the primary influence on the hydrology of the watershed because of its karst geology. Most of the sensitive parameters were groundwater related parameters, such as base-flow recession, groundwater evaporation coefficient, lateral flow lag, delay factor for groundwater leaving the lumped aquifer reservoir, soil depth to impervious layer, deep aquifer percolation fraction, and threshold depth of water in the shallow aquifer required for return flow to occur (Table 2).

3.2. Watershed nutrient and sediment load estimation

3.2.1. Model performance for water quality

The mean simulated values of in-stream N, P, sediment, and dissolved oxygen concentrations were close to the observed mean values (Fig. 4) with a consistently negative percent bias in Topo-SWAT versus the observed mean value: -12.9 for N, -9.1 for P, -4.0 for sediment, and -2.8 for dissolved oxygen. Simulated ranges were wider than the observed for all water quality constituents because daily simulated values, including extreme events, were compared with the available observed values of only 6–8 seasonal sampling events per year (Fig. 4). It appears that Topo-SWAT only slightly under-predicted yearly total N load and marginally over-predicted total P and sediment load, as compared to the observed (Fig. 5). The mean value of total N simulated by Topo-SWAT (2.9 mg/L) was also lower than the observed mean value (3.3 mg/L) (Fig. 4). However, the overall match between predictions of Topo-SWAT and the observed were considered satisfactory with a percent bias in Topo-SWAT versus the observed of -9.2 for N, 6.6 for P, and 5.4 for sediment.

3.2.2. Water quality estimation

Sediment and nutrient losses varied considerably among land uses (Fig. 6). The two major crops in the watershed, corn and soybean, produced the highest sediment and N losses. As expected, the losses were comparatively low from grass land, and the lowest losses of both sediment and organic N were found in forested land. With the current nutrient application rates, both N and P accumulated in soil over the simulation period. Organic nutrient pools in the soil were enriched by 1.8% for N and 3.8% for P per year.

The ten topographic index (TI) (or wetness index) classes produced varying rates of hydrological and chemical activities (Fig. 7). Actual evapotranspiration and surface runoff from the wet part of the watershed increased during simulation, compared to that from

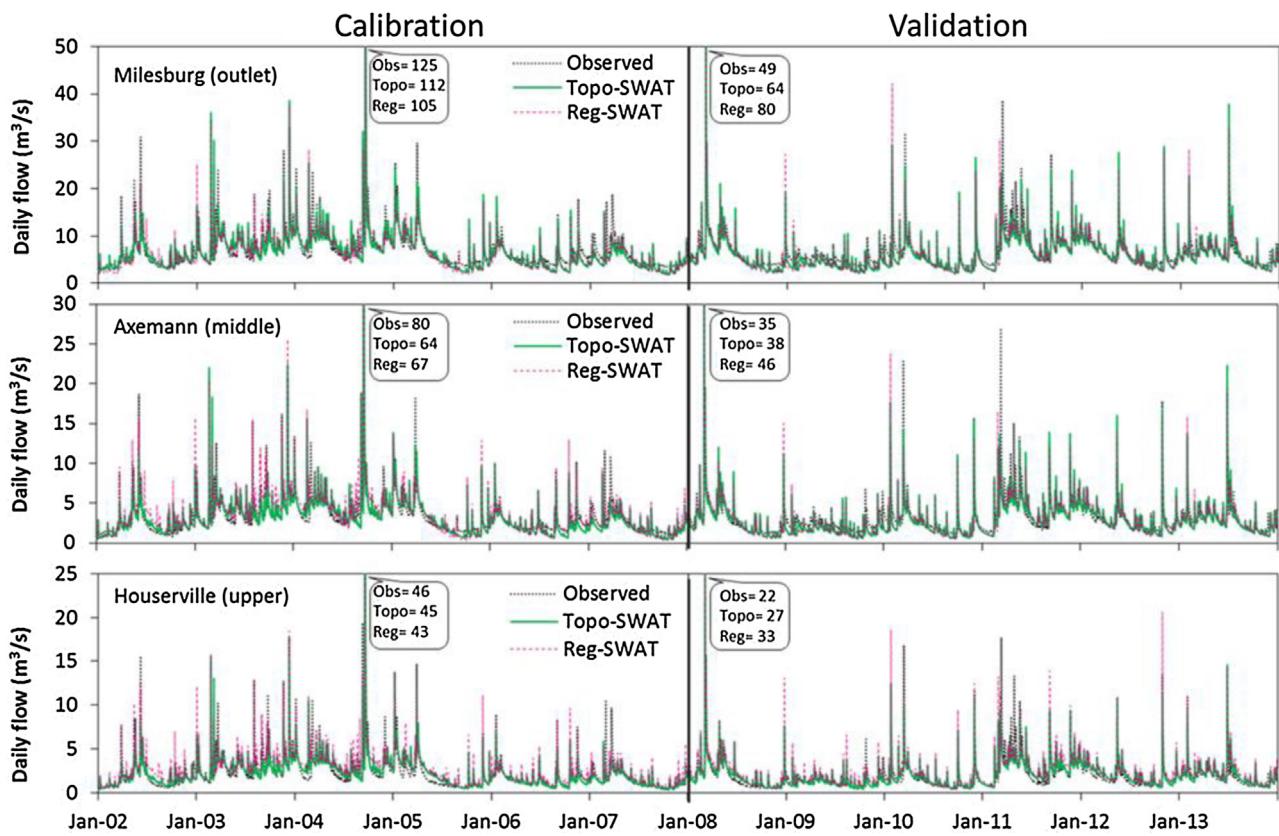


Fig. 3. Simulated and observed daily streamflow at USGS gage stations (Milesburg, Axemann, and Houserville) of Spring Creek watershed for a calibration period of Jan 2002–Dec 2007 and validation period of Jan 2008–Dec 2013. (Two data points are beyond the upper limit of the Y-axis, and their values are stated on the plots).

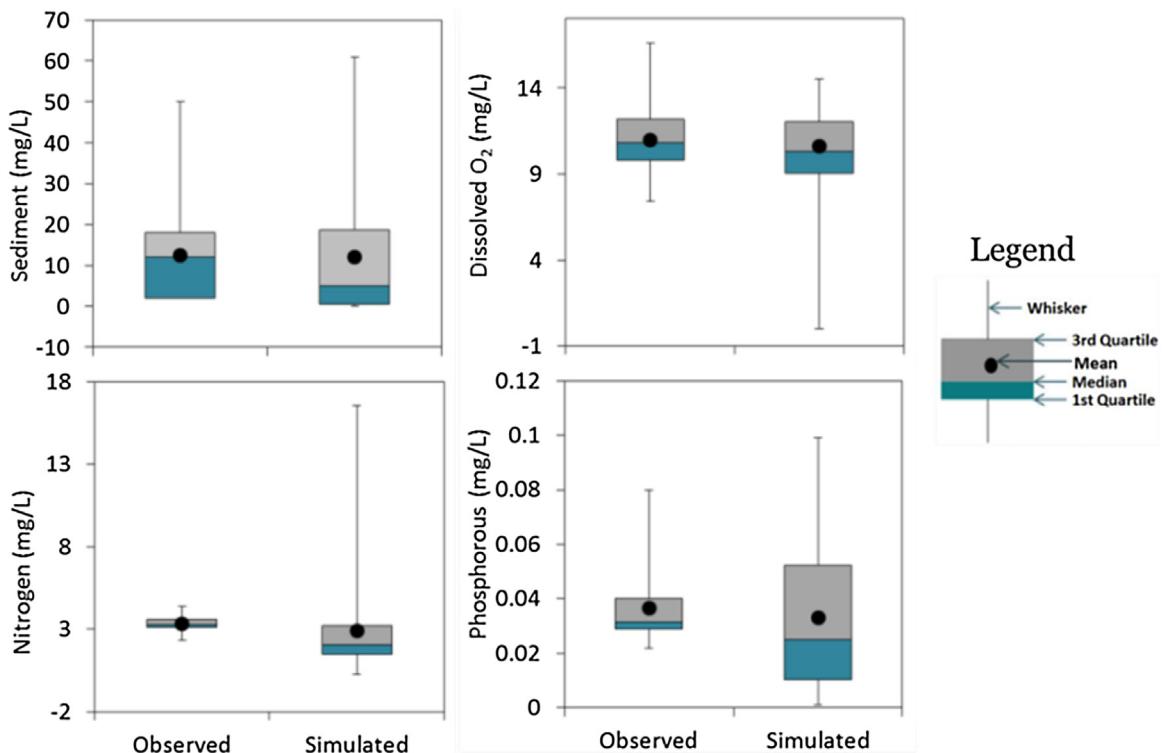


Fig. 4. Simulated (number of data points = 2920) and observed (number of observations = 57) sediment, total nitrogen (N), and total phosphorous (P) concentration at USGS Axemann gage station of Spring Creek watershed for the period of 2000–2008. (Solid circles in the figure indicate mean value).

Table 3

Performance indicators of SWAT streamflow predictions of Spring Creek watershed. Small letters and double bars indicate that NSE values are significantly different ($p < 0.05$).

SWAT version	Calibration (2002–2007)			Validation (2008–2013)		
	NSE	PBIAS	R ²	NSE	PBIAS	R ²
Statistics for daily streamflow prediction						
Topo-SWAT	0.79	-2.8	0.77	0.73 a	-3.7	0.71
Regular-SWAT	0.77	-3.6	0.75	0.68 b	-2.7	0.71
Statistics for monthly streamflow prediction						
Topo-SWAT	0.85	-2.8	0.85	0.82	-3.7	0.75
Regular-SWAT	0.84	-3.6	0.86	0.83	-2.7	0.81

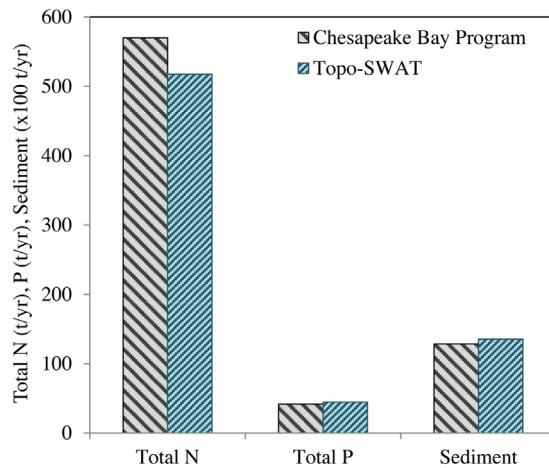


Fig. 5. Nutrient and sediment loadings from Spring Creek watershed outlet as estimated from observed values by the Chesapeake Bay Program (CCMP, 2015) and predicted by the Topo-SWAT model.

the drier areas, due to higher soil water content before and after rain events (Fig. 7a). On the other hand, groundwater contribution to streamflow was less in areas with high TI classes because of the low permeability in the fine textured soil (30–39% clay and 30–36% silt). The largest soil loss, occurring in TI class 10, was attributed to this class having the most surface runoff. However, soil loss also occurred from row crop fields in TI class 1 due to its more coarsely textured, more mobile soil (sand:silt:clay = 78:9:13). Organic N loss was mainly influenced by surface runoff. Thus, organic N loss increased with increase in TI class (Fig. 7b). In contrast, the largest nitrate-N loss was observed in TI class 1 because nitrate-N leaching was augmented by the increased infiltration and percolation in the coarse textured soil of TI class 1. The main path of nitrate-N loss was leaching through the soil profile, as described earlier. Crop yields were typically greater in the wetter parts of the watershed, due to a sustained supply of soil water.

4. Discussion

4.1. Watershed discharge estimation

The Topo-SWAT model performed well in daily watershed discharge estimation (Table 3) because the VSA hydrology of the watershed was well represented by the TI class distribution over the watershed. Spring Creek watershed has three major land covers—forested upland, urbanized, and agricultural valley floor. Perched and losing streams are regularly seen in the headwater regions of the watershed, especially during dry periods (O'Driscoll and DeWalle, 2006). The lack of surface runoff in the forested uplands, as a result of quick infiltration through shallow soils was represented by the lower TI classes in Topo-SWAT (Fig. 7). On the other hand, the higher TI classes appropriately predicted the overland flow generated at the base of hillslopes, recharged from the upland subsurface

flow and upwelling from the clay-rich, low-lying land that feeds the streams in this karst valley (Fulton et al., 2005; O'Driscoll and DeWalle, 2006).

Our study suggests that parameterizing the groundwater contribution both in terms of time and magnitude to the sustained baseflow of stream is critically important for a karst watershed. Amatya et al. (2013) echoed the importance of properly parameterizing the groundwater-surface water interaction for a karst watershed. The curve number (CN2), effective hydraulic conductivity in the main and tributary alluvium (CH_W and CH_K, respectively), and the deep aquifer percolation coefficient (RCHRG_DP) were also identified as the most sensitive hydrologic parameters for a 1555-ha karst watershed in South Carolina, USA (Amatya et al., 2013). Baffaut and Benson (2009) and Amatya et al. (2011) used high soil conductivity values both for the land surface and the tributaries to represent karst geology and assigned a higher value for the deep percolation coefficient to mimic the surface water loss through sink-holes, depressions, and losing streams. However, increasing soil hydraulic conductivity for the Spring Creek watershed caused an over-prediction of flow at the end of the recession period. In addition, over calibrating intrinsic soil properties, such as soil hydraulic conductivity, for only hydrology may hinder accurate simulation of the geochemical processes (Abbaspour, 2014).

Baffaut and Benson (2009) proposed simulating the fast movement of water from the ground surface to the aquifer by splitting the recharge of the aquifer into two parts: the faster recharge from sink-holes and losing streams, and the slower recharge from infiltration through the bulk soil. With the parameter adjustments described above, the simulated baseflow contribution of the Spring Creek watershed to the total streamflow was 83%, only 4% less than the observed value of 87%.

The soil depth adjustment in the FAO soil database with corresponding values from the SSURGO soil database augmented groundwater recharge and consequently improved the accuracy of surface runoff magnitude and groundwater recharge predictions. The results are in agreement with a field study in the region by Duncan et al. (2014) who found that shallow soil depths produced higher surface runoff volumes than deep soil layers in field-scale lysimeters. They also observed substantial variation of the soil depth to bedrock (0.14 m → 1.33 m) within a field of 15 m × 27 m. However, it is difficult to incorporate all the soil depth variations and subsequent surface runoff generation into the watershed-scale simulation.

Correctly locating the recharge area of a spring, whether inside or outside of the watershed, is vital to developing an accurate hydrologic and chemical balance of the system (Tzoraki and Nikolaidis, 2007; Moraetis et al., 2010). Springs recharged from outside of the watershed were considered point sources, but springs recharged inside were not because their discharges are already included in the baseflow of the stream, as suggested by Baffaut and Benson (2009). The simulated underestimation of flow at the watershed outlet indicates that not all inputs into the watershed water balance from all springs were captured.

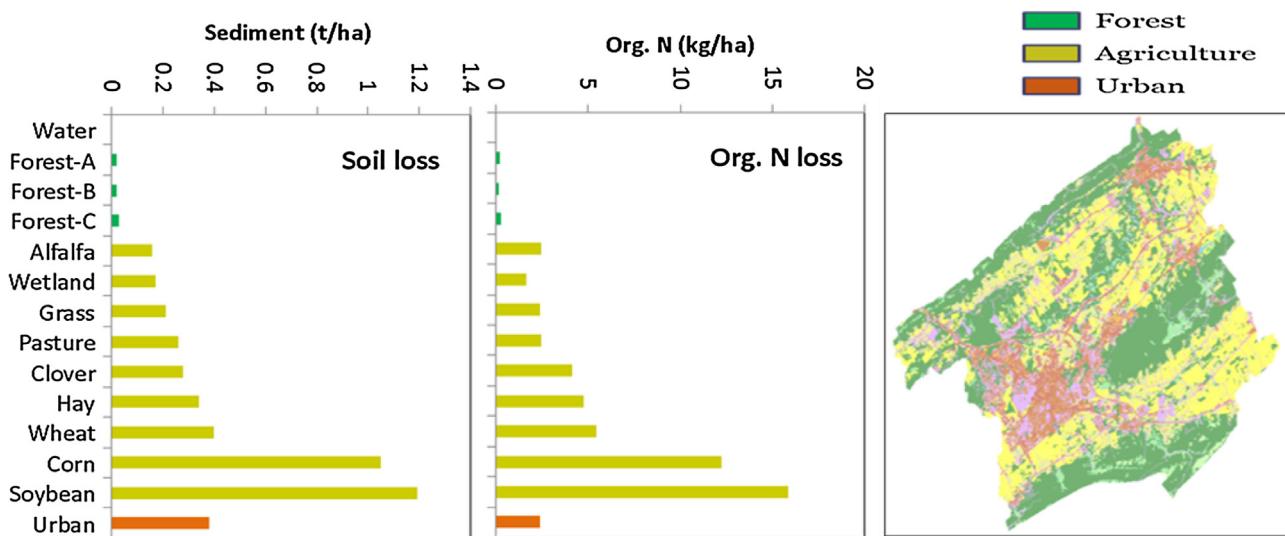


Fig. 6. Topo-SWAT predicted sediment and organic nitrogen (N) losses for the land uses in Spring Creek watershed (Forest A, B, and C represent evergreen, mixed, and deciduous forest, respectively).

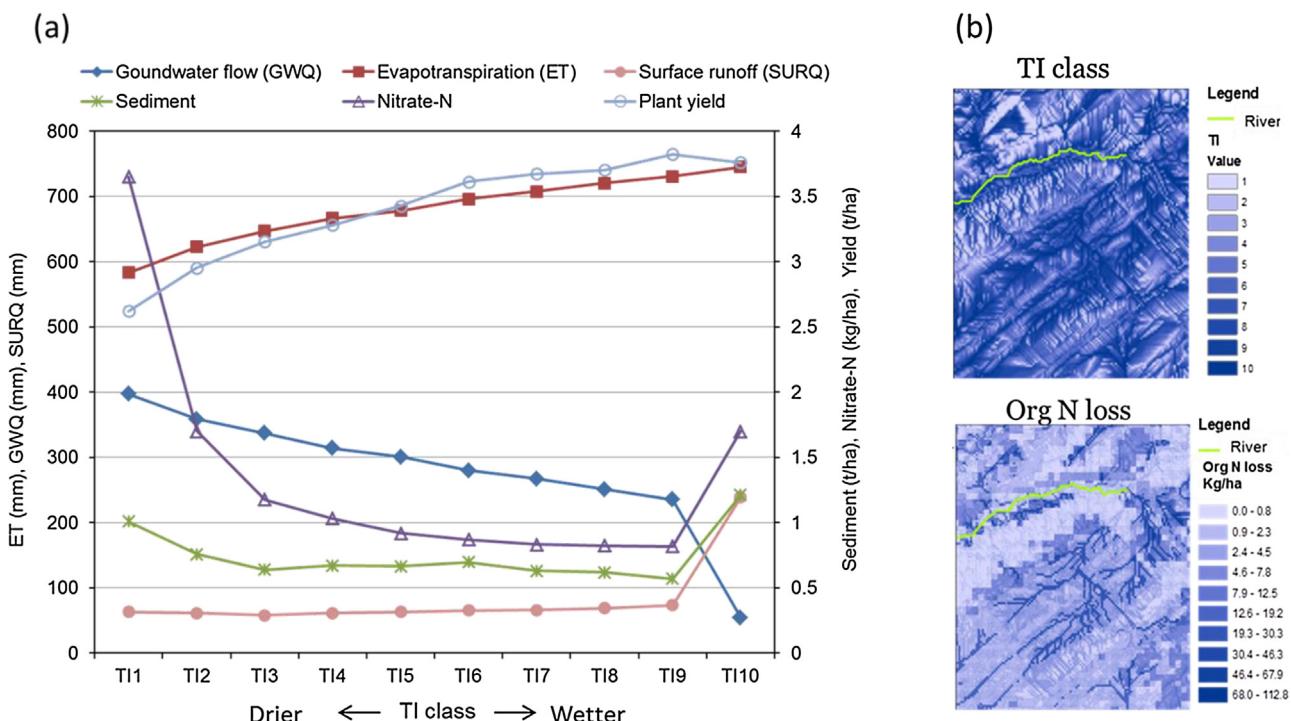


Fig. 7. (a) Average annual runoff, erosion, nutrient losses, and crop yield for different TI (topographic wetness) classes within Spring Creek watershed (ET, GWQ, and SURQ denote actual evapotranspiration, groundwater contribution to streamflow, and surface runoff, respectively) and (b) relationship between spatial distribution of TI classes (wetness index) and organic nitrogen (N) loss for an excerpt of Cedar Run subbasin in Spring Creek watershed.

4.2. Watershed nutrient and sediment load estimation

Adequate simulation of watershed hydrology is the first step toward realistic prediction of nutrient transport (Abbaspour, 2014). The hydrologically calibrated Topo-SWAT model was used for simulating water quality because Topo-SWAT predicted hydrology relatively well, as described earlier, and has the ability to locate critical source areas for nutrient loss. Collick et al. (2015) found also that Topo-SWAT performed more accurately than Regular-SWAT in predicting watershed-scale P losses in a nearby, non-karst agricultural watershed with VSA hydrology.

The hydrologically calibrated models simulated sediment and P loads more accurately than N loads (Fig. 4). The sediment load per unit area in the Spring Creek watershed was much lower than sediment fluxes in the adjacent non-karst Juniata River watershed (Sevon, 1989) because more than 80% of the stream flow in Spring Creek was generated through the subsurface karstic system. Average observed sediment concentration in the major karstic spring for our watershed was 3.5 mg/L (SCWA, 2013). The amount of sediment carried by the spring that recharges outside of the watershed was added in the model as a point source. Other sediments generated from the karst surface move primarily with subsurface flow below a shallow soil layer and cannot completely avoid the filtering effect

of such complex soil formation (Baffaut and Benson, 2009), and there is probably both filtration and settling. The flow systems were therefore represented in the model either as lateral flow or shallow groundwater flow based on the soil profile depth. The resulting simulated sediment concentration in the baseflow was very similar to the observed values (≤ 2 mg/L). Nerantzaki et al. (2015) also found low concentration of suspended sediment in karstic springs (1–10 mg/L) whereas the concentration of suspended sediment was between 2 and 1000 mg/L in other non-karst sections of their study stream.

The average annual sediment load from Spring Creek watershed predicted by the Topo-SWAT model was 0.34 t/ha. The sediment loading rate was 0.97–1.1 t/ha for corn and soybean field, 0.36 t/ha for small grain, 0.4–0.56 t/ha for hay and grasses, 0.28 t/ha for range and brush, 0.03–0.04 t/ha for forest, and 0.37 t/ha for developed area (Fig. 6). The sediment loading rates predicted by Topo-SWAT were within the range of the literature values, such as 0.03–2.8 t/ha for grass (Meeuwig, 1970), 0.1–1.4 t/ha for brush (Johnson and Gordon, 1988), 0.01–0.08 t/ha for evergreen forest (Elliot, 2013), 0.005–1 t/ha for roads, and 0.01–0.13 t/ha for timber harvest forest (Covert, 2003; Robichaud et al., 2010; Robichaud et al., 2011).

Nutrient dynamics, especially N dynamics, in a karst aquifer differ considerably from those in a non-karst aquifer, as observed in previous studies (Fishel and Lietman, 1986; USGS, 1997; SCWA, 2013). Particular attention was paid to simulating the residence time of N in groundwater as accurately as possible. Highly weathered bedrock, a common feature of karst geology, allows for short flow path times, facilitating aerated conditions in groundwater because of increased dissolved oxygen concentrations. In addition, atmospheric pressure and temperature changes in the conduit of a karst system can increase mixing of N (Moraetis et al., 2010). These aerated conditions can increase the half-life of nitrate-N by inhibiting denitrification. Nitrate-N levels are likely to be higher in groundwater underlain with carbonate bedrock than in sandstone and shale bedrock areas (Fishel and Lietman, 1986; USGS, 1997). For example, monitored nitrate-N concentrations in baseflow and stormflow (direct runoff) for a 350-ha low-agriculture drainage area in Spring Creek watershed in 2011 were 3.5 and 0.42 mg/L, respectively (SCWA, 2013). Concentration of nitrate-N in rainwater ranged 0.15–0.36 mg/L for the period 2000–2014 in the watershed (CCMP, 2015). During 1999–2013 the overall mean nitrate-N concentration was 3.1 mg/L in surface water and 3.6 mg/L in groundwater at the Milesburg USGS gage (SCWA, 2013). Nitrate-N transport to groundwater through leaching and subsequent release from groundwater to surface water was therefore a major source of N loading to the streams in this watershed. However, P did not follow the same pathway due to its sorptive behavior. Unlike N, P is likely moving in overland flow and shallow subsurface flow. Groundwater N flow pathways also are supported by long-term observations of nitrate-N in Spring Creek (Fig. 2). In response to various watershed management initiatives (e.g., sewage treatment processes, riparian buffer installation, conservation tillage, etc.), both magnitude and range of the concentration of total P have decreased rapidly over the last three to four decades, but nitrate-N has decreased less rapidly (Fig. 2).

The average rate of total N loss from the watershed was 14.02 kg/ha, of which organic N loss was 4.3 kg/ha, nitrate-N loss with surface runoff was 0.89 kg/ha, nitrate-N leached was 11 kg/ha, nitrate-N loss with lateral flow was 0.53 kg/ha, and nitrate-N loss through groundwater yield was 8.3 kg/ha. The ratio of predicted inorganic to organic N loss was 24:76, which was close to the observed value in storm water of 30:70 (SCWA, 2013). An average 75.8 kg/ha mineral N became available annually from prior residues, particularly previous legume crops and manure applications, which can contribute 45–123 and 22–39 kg N/ha/year, respectively, in this area of Pennsylvania (The Agronomy Guide,

2015). Organic N loss from the two major crops of the watershed, corn and soybean, ranged from 13.0–16.4 kg/ha (Fig. 6), well within the 7.27–21.9 kg/ha range for those crops reported by Harmel et al. (2006). Organic N loss for small grain (5.3 kg/ha) simulated by the model (Fig. 6) was within 1 kg/ha of the literature value (5.9 kg/ha by Harmel et al., 2006).

In the watershed the predicted average rate of total P loss was 1.21 kg/ha, of which 0.63 kg/ha was organic P, 0.05 kg/ha was soluble P with surface runoff, and 0.54 kg/ha was sediment bound. Harmel et al. (2006) reported that total P loss from agricultural soil from different management practices varied 0.22–1.18 kg/ha in different states of USA. They also stated that dissolved P loss varied 0.15–1.0 kg/ha, and sediment bound P loss ranged 0–1 kg/ha. The sediment bound P loss simulated for the major crop of the watershed (corn) was 1.1 kg/ha. A median value of 0.85 kg/ha with a wide range (0–8 kg/ha) of this value was reported by Harmel et al. (2006).

4.3. Implications of the model

This study suggests that Topo-SWAT, when properly applied, can be effective in simulating karst watersheds with VSA hydrology and in evaluating comparative water quality impacts among scenarios with various crop types and agricultural management practices. This study supports implementation of more sustainable soil and nutrient management practices in order to improve water quality to meet the Chesapeake Bay TMDL. The critical source areas, where highly concentrated nutrient source areas and localized high transport areas meet, that can be targeted for placement of best management practices, enabling development of a more cost-effective watershed management plan. However, to use this model successfully the recharge and discharge locations of any large karstic springs in the watershed need to be known beforehand. The Spring Creek watershed has only one large karstic spring, so more studies need to be done for more complex karstic watersheds.

5. Conclusion

Topo-SWAT simulated daily streamflow slightly more accurately than did Regular-SWAT (validation $NSE = 0.73$ and 0.68, respectively), when compared with measured data at three gauging stations across the watershed. The Topo-SWAT model performed well in daily discharge estimation because the VSA hydrology of the watershed was accurately reflected by the wetness class distribution over the watershed. Proper representation of a spring that recharges outside of the watershed but discharges inside as a point source in the model was critically important for hydrological and chemical balance. Topo-SWAT predicted water quality parameters satisfactorily with a percent bias against observed mean values of -12.9 for N, -9.1 for P, -4.0 for sediment, and -2.8 for dissolved oxygen. Nitrogen dynamics in groundwater and groundwater contribution of N to the streams were the critical components of predicting overall N loading in the watershed. Additionally, water and nutrient transport within the watershed matched expected and literature-supported patterns. The simulated nutrient and sediment losses among different land uses were reasonably supported by the literature values, indicating that the crop types and agricultural management practices were represented by the model and supporting the use of the model in evaluating the water quality impacts of land use changes.

Acknowledgements

This publication was developed under Assistance Agreement No. RD 83556801-0 awarded by the US Environmental Protec-

tion Agency (EPA) to the Center for Nutrient Solutions, Penn State University, USA. It has not been formally reviewed by EPA. We also acknowledge the support from the Department of Plant Science, Penn State University and USDA-ARS, Pasture Systems and Watershed Management Research Unit, University Park, PA 16802, USA. We are also grateful to the Chesapeake Community Modeling Program. Mention of trade names or commercial products in this publication is solely for the purpose of providing specific information and does not imply recommendation or endorsement by the US Department of Agriculture, EPA, Penn State University, or University of Maryland – Eastern Shore. All entities involved are equal opportunity providers and employers.

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