DEVELOPMENT OF ALGORITHMS FOR MODELING ONSITE WASTEWATER SYSTEMS WITHIN SWAT

J. Jeong, C. Santhi, J. G. Arnold, R. Srinivasan, S. Pradhan, K. Flynn

ABSTRACT. *Onsite wastewater systems (OWSs) are a significant source of nonpoint-source pollution to surface and groundwater in both rural and suburban settings. Methods to quantify their effect are therefore important. The mechanics of OWS biogeochemical processes are well studied. However, tools for their assessment, especially at the watershed scale, are limited. As part of this work, modeling capabilities were developed within the Soil Water Assessment Tool (SWAT) such that OWSs and their subsequent environmental impacts can be evaluated A case study was initiated on the Hoods Creek watershed in North Carolina to test the new SWAT algorithms. Included were: (1) field-scale simulations of groundwater quantity (water table height) and quality (N, P), (2) Monte Carlo evaluations of OWS service life to evaluate suggested calibration parameters, and (3) assessments of watershed-scale pollutant loadings within the model. Results were then analyzed at both the field and watershed scales. The model performed well in predicting both site groundwater table levels (R^2 = 0.82 and PBIAS = -0.8%) and NO_3-N concentration in the groundwater (R^2 = 0.76, PBIAS = 2.5%). However, the performance for PO_4-P simulations was less reliable due to difficulty in representing the mobility of soluble P in the soil. An advanced P algorithm is recommended to address the sophisticated physiochemical properties of soil particles and improve the model's performance.*

Keywords. *Biozone, Nitrogen, Nutrient, Onsite wastewater systems, Phosphorus, Septic, SWAT.*

In rural areas, it is often inefficient to operate centralized wastewater systems for the purpose of domestic sewage treatment due to the sparse residential densities, uneven terrain, and/or limited water or energy supplies. In such instances, decentralized types of wastewater treatment systems (i.e., onsite wastewater systems) are typically used. In fact, more than 25% of existing homes and 37% of new developments in the U.S. use onsite wastewater systems (OWSs) for wastewater disposal (USEPA, 1997). The U.S. Census Bureau (1999) estimates that more than 60 million people depend on decentralized systems for their treatment needs. Hence, understanding the biophysical processes within OWSs, and the environmental impacts thereof, is of great importance to scientists, watershed managers, and regulatory agencies.

A thin, biologically active layer of soil, called the biozone, develops underneath the drainfield near the OWS infiltrative surface. This layer is of primary interest in this study as it contains a large population of naturally existing microorganisms that digest nutrients and are the primary mechanism of treatment in an OWS. The fate and transport of nutrients, chemicals, and pathogens within this layer have been investigated (Van Cuyk et al., 2005; Heatwole and McCray, 2007; McCray et al., 2005; Lee et al., 1998; Van Cuyk et al., 2001; Andreadakis, 1987; Van Cuyk and Siegrist, 2007; Dimick et al., 2006; Lowe et al., 2006; Weintraub et al., 2002; Lemonds and McCray, 2003; Weintraub et al., 2004), and theories describing the clogging mechanism near the infiltrative surfaces (where the wastewater is distributed to the soil) have also been developed (Kristiansen, 1981; Siegrist, 1987; Weintraub et al., 2002; Beach and McCray, 2003; Siegrist et al., 2005). Both are of great importance because of their capacity to alter nutrient loads to groundwater.

A number of physically based and empirical models for predicting biozone hydraulic conductivity have also been developed (USEPA, 1980; Clement et al., 1996; Weintraub et al., 2002; Beach and McCray, 2003; Siegrist et al., 2004; Beach et al., 2005; Bumgarner and McCray, 2007). However, this and much of the previous literature focuses on micro-scale processes (lab-scale column tests) to field-scale systems (typically a single unit). Very little modeling work has been done at the watershed scale, most likely due to uncertainties introduced by the complex of subsurface hydrology, spatial and temporal variation in soil and water, and the lack of data regarding OWSs in the first place. However, some efforts have been completed. For example, Weintraub et al. (2002) developed a biozone algorithm and integrated it into the Watershed

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Analysis Risk Management Framework (WARMF), a decision tool that was designed to support the watershed approach involving stakeholders (Chen et al., 2001). Lemonds and McCray (2003) applied SWAT to model phosphorus transport in the Blue River watershed in Colorado. They simulated OWSs by manipulating fertilizer management practices in SWAT because SWAT did not have a built-in module for simulating OWS processes. Similarly, Pradhan et al. (2005) used auto fertilizer management operation within SWAT to mimic nutrient loads from OWSs. However, in both the SWAT efforts mentioned previously, i.e., Lemonds and McCray (2003) and Pradhan et al. (2005), the simulations of OWSs were done for the lack of a better alternative at the time. Our work has been initiated to finally address that deficiency.

SWAT Model Description

SWAT is a distributed parameter, continuous simulation, watershed-scale model that was originally developed by the USDA Agricultural Research Service for the long-term simulation of the impact of land management practices, land use changes, and agricultural chemical yields on downstream water bodies (Arnold et al., 1998). The model has been widely used in water quality modeling studies, including TMDL analyses (Benham et al., 2006; Borah and Bera, 2004; Radcliffe et al., 2009), evaluations of the USDA Conservation Effects Assessment Project (Harmel et al., 2008; Richardson et al., 2008; Van Liew et al., 2007), and nonpoint-source pollution analyses (Borah and Bera, 2003; Santhi et al., 2001). Hence, it is a very suitable tool for consideration of OWS processes given its strong nutrient routines.

Spatial heterogeneity and connectivity in SWAT are described by partitioning a watershed into a number of subwatersheds that are homogeneous in terms of climate and topography, but are distributed in the context that they are linked spatially with other subwatersheds. Discretization is further completed into hydrologic response units (HRUs), which are lumped soil and landcover combinations having no spatial context. Soil water, surface runoff, sediment yield, and nutrients are computed at the HRU level and then aggregated for subsequent routing through the channel network. Six hydrologic compartments are incorporated into the model to describe the flux of water in the HRUs. These include: (1) snow accumulation and melt, (2) surface runoff, (3) evapotranspiration and unsaturated zone processes, (4) lateral flow, (5) shallow groundwater flow, and (6) deep aquifer flow. Nutrient processes in SWAT include fate and transport of a number of different nitrogen and phosphorus pools including organic and inorganic forms, the latter of which are a primary consideration in biozone development.

Methods and Materials

Biozone Processes and Adaptation

The biozone algorithm proposed by Siegrist et al. (2005) was adapted to SWAT in this study to simulate the fate and transport of domestic pollutants discharged from septic tank effluent (STE) into the soil absorption system. The algorithm was modified such that (1) the net growth of septic biomass is dependent upon soil temperature, and (2) the leaching of soluble phosphorus through soil layers is regulated not only by a linear isotherm but also by a linear function in which the effluent P concentration is a function of soil type and the total amount of P in the soil.

The conceptual model for the biozone is a biologically active layer in the soil absorption system directly below the infiltrative surface where STE organic matter (BOD) is reduced by microorganisms (fig. 1). The biozone layer is assumed as a control volume that receives the septic effluent from the OWS and infiltration from above layers while allowing percolation to subsoil layers. The amount of live bacteria biomass in the biozone layer in SWAT is estimated using a mass balance equation modified from Siegrist et al. (2005):

$$\frac{d(Bio)}{dt} = \alpha \cdot \gamma_{tmp} \cdot \left[\sum Q_{STE} \cdot C_{BOD,in} - I_p \cdot C_{BOD} \right] - \gamma_{tmp} \cdot \left(R_{resp} + R_{mort} \right) - R_{slough} \tag{1}$$

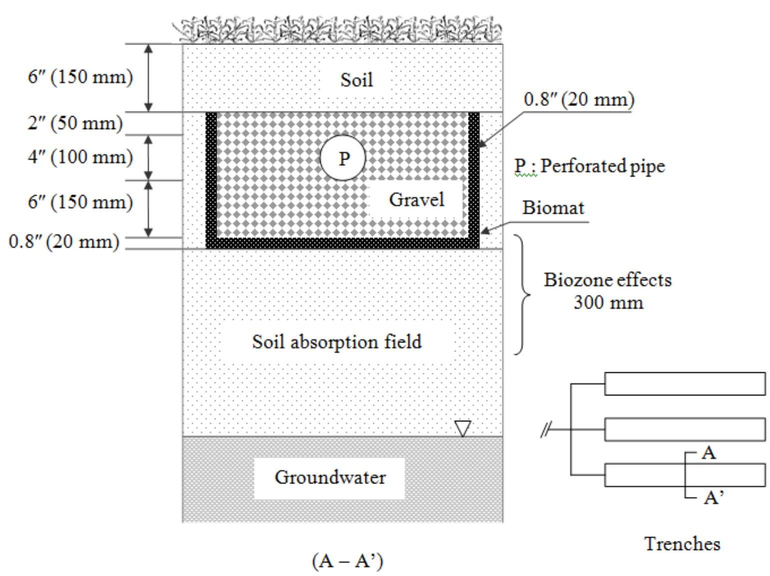

Figure 1. Configuration of STE distribution chamber and soil absorption system.

where Bio is the amount of live bacteria biomass in the biozone (kg ha^{-1}), $C_{BOD,in}$ is the BOD concentration in the STE (mg L^{-1}), C_{BOD} is the BOD concentration in the biozone (mg L^{-1}), α is the ratio of live bacteria growth to BOD in the STE (g g^{-1}), Q_{STE} is the flow rate of the STE (m^3 d^{-1}), I_p is the amount of percolation out of the biozone (m^3 d^{-1}), R_{resp} is the amount of respiration of bacteria (kg ha^{-1}), R_{mort} is the amount of mortality of bacteria (kg ha^{-1}), and R_{slough} is the amount of sloughed bacteria (kg ha^{-1}).

In equation 1, both the bacteria conversion rate (growth) and die-off rate (respiration and mortality) are temperature corrected (γ_{tmp}). The adjusted temperature (γ_{tmp}) was adapted from Eppley (1972):

$$\gamma_{tmp} = \zeta^{(T-20)} \tag{2}$$

where T is soil temperature (°C), and ζ is a temperature correction coefficient that is available in SWAT (default ζ = 1.07).

As bacteria die, a portion of the dead biomass becomes plaque. Total solids in the STE can also contribute to plaque accumulation. The rate of change in plaque is computed by:

$$\frac{d(plaque)}{dt} = R_{mort} + \frac{\sigma \cdot \sum Q_{STE} \cdot TS}{1000 \cdot A_d} - R_{slough} \tag{3}$$

where $plaque$ is the amount of dead bacteria biomass and residue (kg ha^{-1}), σ is a calibration coefficient that affects the conversion of total solids in the STE to plaque (unitless), TS is the total solids contained in the STE (mg L^{-1}), and A_d is the area of the drainfield (ha).

The accumulation of plaque affects soil porosity and slows down soil water percolation (USEPA, 1980). Field-scale experiments suggest that reductions in hydraulic conductivity are primarily influenced by the STE loading rate and the type of infiltrative surface (Bumgarner and McCray, 2007). Similarly, live bacterial biomass further alters hydrologic flux by allowing the biozone layer to retain additional water (through the development of filamentous organic material). Unlike in natural soils, the field capacity of the biozone layer is proportional to the live bacterial biomass, as it is formed with water-absorbing filamentous material, allowing the biozone layer to retain additional water. Weintraub et al. (2002) proposed a relationship between biozone field capacity and the amount of biomass:

$$\theta_f^{\,t} = \theta_f^{\,t-1} + \Phi \cdot \left(\theta_s^{\,t-1} - \theta_f^{\,t-1}\right)^{\xi} \frac{\left[\frac{d(Bio)}{dt}\right]}{10 \cdot \rho_{bm}} \tag{4}$$

where $\theta_f^{\,t}$ is field capacity at the end of the day (mm), $\theta_f^{\,t-1}$ is field capacity at the beginning of the day (mm), $\theta_s^{\,t-1}$ is saturated moisture content at the beginning of the day (mm), ρ_{bm} is the density of live bacterial biomass (~1000 kg m^{-3}), Φ is field capacity coefficient 1 (unitless), and ξ is field capacity coefficient 2 (unitless).

The transformation and removal of pollutants in the biozone in SWAT is directly related to the population of live bacteria and biophysical processes in the biozone layer. Changes in nitrogen, BOD, and fecal coliform are estimated by a first-order reaction equation:

$$C_{k,end} = C_{k,i} \cdot e^{-K_k \Delta t} \tag{5}$$

where $C_{k,end}$ is the concentration of constituent k in the biozone at the end of the day (mg L^{-1}), $C_{k,i}$ is the concentration of constituent k at the beginning of the day (mg L^{-1}), and K_k is a first-order reaction rate (d^{-1}) that is a function of the total biomass of live bacteria and a reaction rate coefficient.

The primary mechanism of phosphorus (P) removal in the biozone is adsorption, which takes place in the soil medium below the STE distribution field. Since concentrations of phosphorus in the soil medium are often in the linear range of reported nonlinear isotherms (McCray et al., 2005), P sorption in the biozone is described directly by the linear isotherm:

$$S = K_D C \tag{6}$$

where S is the mass of solute sorbed per unit dry weight of solid (mg kg^{-1}), C is the concentration of the solute P in equilibrium with the mass of P sorbed onto the soil particles (mg L^{-1}), and K_D is a linear distribution coefficient (L kg^{-1}). For modeling purposes, McCray et al. (2005) recommended K_D = 15.1 L kg^{-1} (median value), although it can vary from 5 to 128 L kg^{-1} (10th to 90th percentile) according to local conditions. Similarly, a median value for the maximum of S = 237 mg kg^{-1} is recommended. Should the median value pose a significant underestimation of the maximum S, a larger value (~800 mg kg^{-1}) can be used instead (Zanini et al., 1998).

It is implied in the linear isotherm concept that the effluent P concentration leaching to subsoil layers is zero until the soil is saturated with P. However, a small amount of P may leach to the groundwater while the soil is not fully saturated (perhaps through preferential flow). Therefore, the linear relationship proposed by Bond et al. (2006) was adapted to allow P percolation to the groundwater:

$$P_{sol,leaching} = a \cdot P_{sol,total} + b \tag{7}$$

where the concentration of soluble P leaching to the subsoil layer ($P_{sol,leaching}$) is a function of the soil type and the total amount of the soluble P ($P_{sol,total}$) in the soil. The slope and intercept vary for different soil types (e.g., a = 0.14 and b = -0.9 for loamy sand, a = 0.09 and b = 1.3 for silt loam, and a = 0.09 and b = 3.06 for sandy clay loam).

Soil Temperature Effects on the Biozone

Soil temperature in SWAT is calculated as a function of air temperature, ground cover, soil depth, soil moisture, and other contributing factors. During the winter, the soil temperature in the drainfield is affected by the warm temperature of the domestic wastewater, which precludes the soil from freezing. The temperature algorithm of the biozone layer and subsoil layers was therefore modified by the following linear relationship to ensure that soil temperatures remain above freezing over the course of a simulation:

$$T_{soil,ly} = C_1 - \left(C_1 - T_{soil,ly}\right) \cdot C_2 \text{ if } T_{soil,ly} < C_1 \tag{8}$$

where $T_{soil,ly}$ is the soil temperature in the soil layer ly, C_1 is temperature correction factor 1 representing the minimum temperature that equation 8 does not apply, and C_2 is a multiplier that slows the temperature decrease below C_1. We suggest C_1 = 10°C and C_2 = 0.1 to 0.2, as these guarantee that the soil temperature below the biozone layer does not reach freezing.

Integration of Biozone Algorithm in SWAT

The biozone algorithm described previously was incorporated into the SWAT model (SWAT2009) with appropriate linkage to the soil moisture routines and nutrient routines. It was also linked with the most recent ArcSWAT GIS interface (to generate inputs files for simulating the biozone processes). Each onsite septic system was represented as a "septic or OWS" HRU that was inclusive of the soil type, average drainfield area, average number of people in the house, and type of septic system. Because multiple onsite septic systems may fall on a single soil type, they are aggregated and configured to a single HRU. In other words, a septic HRU could represent one or multiple OWSs, depending on their spatial distribution in the watershed.

A flowchart of the SWAT biozone algorithm is shown in figure 2. Biozone processes are called within the HRU loop (i.e., daily simulation so that it operates seamlessly with other hydrologic and water quality processes within SWAT), and the entire model is predicated on the question of whether the system is active (functioning properly) or failing (altered by plaque buildup to the point where hydraulic conductivity is compromised). An active system is simulated according to the mass balance described previously, while failing septic HRUs have no biozone processes implemented. A failing system is returned to active status only after maintenance or repair, and the time to maintenance is counted by the model based on a user-specified number of days. Once the number of days exceeds the specified maximum number of failing days, the failing system is automatically returned to active status and related septic properties are re-initialized.

In the active OWS HRUs, plaque buildup over time is simulated until system hydraulic failure occurs. The time of

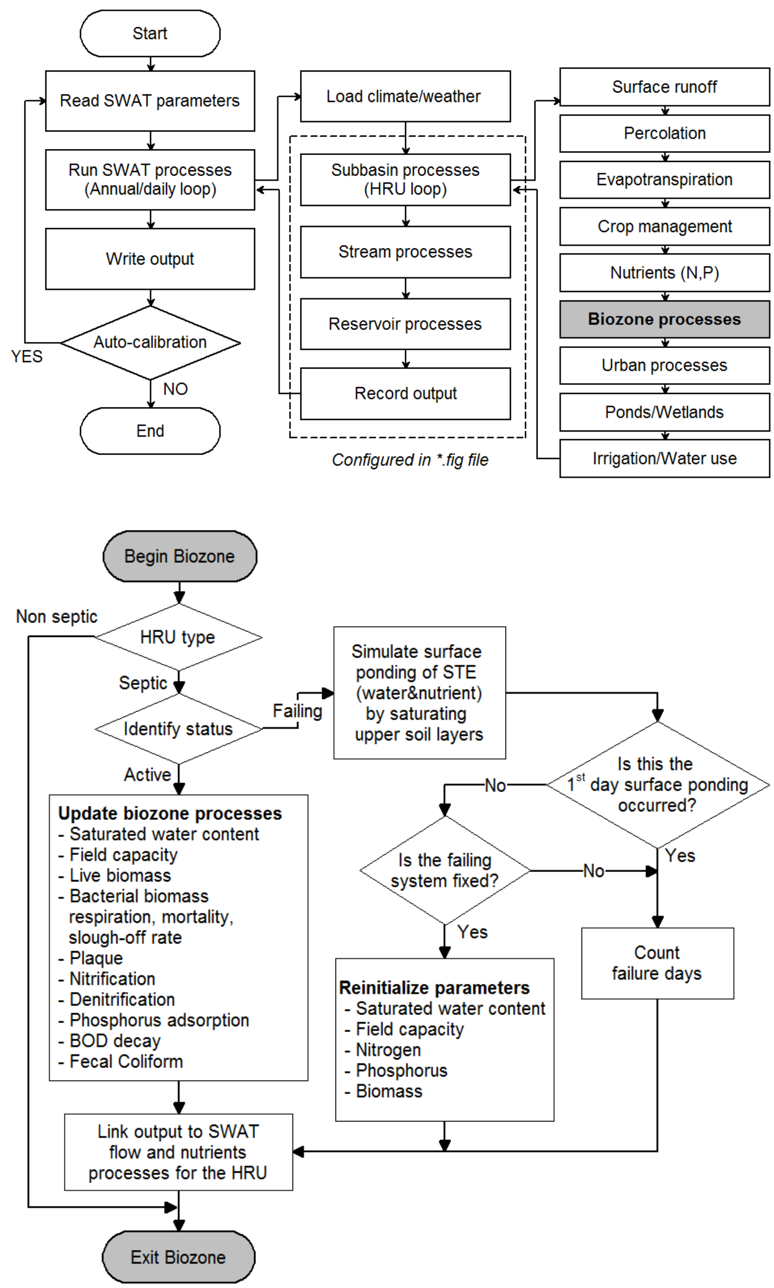

Figure 2. Flowchart of the biozone algorithm in SWAT.

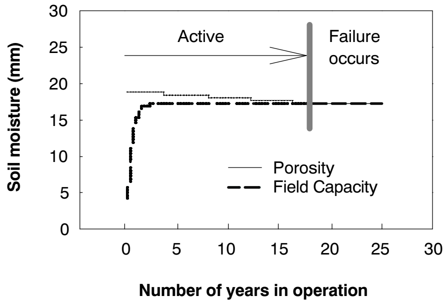

Figure 3. Predicted changes in porosity and field capacity in the biozone layer.

failure is denoted as the point when the updated saturated water content and field capacity equal the porosity (thereby allowing no infiltration, see fig. 3). At this time, ponding of septic effluent occurs on the ground surface due to hydraulic failure within the biozone layer. The amount of nutrients that are transported to the upper soil layers is estimated based on the nutrient concentration in the STE and the amount of water that moves to the upper soil layer. There are no special treatment processes that apply to the nutrients in failing septic systems. Once the number of failing days counted exceeds the user-specified number of failing days (*isep_tfail*), the failing system is updated to an active system and related properties are reinitialized to a fresh active system. The transition between active and failing systems may repeat several times during a SWAT simulation in long-term simulations.

Study Area

The performance of the SWAT biozone algorithm was tested in a case study on the Hoods Creek watershed in North Carolina. The Hoods Creek watershed was selected for evaluation of the biozone algorithm due to the availability of septic field data collected by North Carolina State University (NCSU) researchers (Humphrey, 2002). It is located south of the Trent River near its confluence with the Neuse River in the lower Coastal Plain physiographic region of eastern North Carolina. The watershed area is small (172 ha) and has been recently developed with 227 housing units (fig. 4). The residences only use OWSs for domestic wastewater treatment, and there are believed to be no other significant point or nonpoint sources that would interfere with our model testing.

The predominant soil type in the Hoods Creek watershed is Autryville loamy sand (70%). It is a well-drained sandy soil that has several clay layers at approximately 95 cm depth, which results in a perched water layer and lateral flow. The other predominant soil is Masontown Mucky fine sandy loam (7.5%), which is a mineral-organic soil that has very high organic carbon content and occurs primarily within riparian areas along the creek (Pradhan et al., 2005). The whole area is underlain by limestone about 2.5 to 3.5 m below the stream bottom of Hoods Creek, and major land uses are rangeland (35%), urban (28%), and forest (26%).

GIS and Other Data Sources

SWAT requires GIS data such as land use, soil, and DEM and time series. Data used for model development are as follows: daily precipitation and maximum/minimum temperatures were obtained from a local weather station approximately 3 km away from the watershed with a record of approximately 60 years (1950-2008). The total annual rainfall over the test period (2000 and 2001) was 1250 mm, with a daily maximum of 62.5 mm. No significant seasonal trends were observed; however, summer had more wet days than winter. Historical temperature data show that the watershed experiences hot temperatures during the summer (approaching 35°C) and mild winters. Daytime temperatures during the winter months are mostly above zero, while nighttime lows drop below zero (-9°C).

Several high-resolution spatial map layers were used for watershed delineation, including: (1) a one-meter resolution LIDAR DEM prepared by FEMA, (2) a 1:24,000-scale hydrography layer from the North Carolina Center for Geographic Information and Analysis (NCCGIA), (3) a SSURGO 2.0 soil map downloaded from the NRCS Data Gateway, and (4) a 1:24,000-scale land use map obtained from NCCGIA for map overlay and discretization. The geographic location of OWSs was assumed to be identical to the location of the housing units, which were identified manually using digital orthophotos. Houses were digitized as polygons and then imported into the model (Pradhan, 2004).

Data for calibration of groundwater height and biozone nutrient concentrations in SWAT were obtained from piezometers monitored over the period of September 2000 to November 2001 at three residential sites listed in table 1 (Humphrey, 2002). Nutrient data including PO_4^-, NH_4^+, and NO_3^- were collected at a number of wells (118 monitoring wells in total), and only data collected within 1 m of an STE drainfield were considered for use in the study. At these sites, groundwater grab samples were collected at two different depths: one near the groundwater table and the other at a lower depth. These are the data presented in model evaluation.

Model Setup

The location of 227 septic systems in the Hoods Creek watershed were merged into the land use map and assigned a typical area of a septic system drainfield (100 m^2). The information was integrated into a SWAT project that specifically represented the characteristics of the watershed (and associated OWS sites) including observed STE flux and associated quality, landcover, soil type, etc. The final SWAT model had 13 subbasins and 310 HRUs, of which 52 were septic.

Determination of OWS Parameter Values

In calibration, it was identified that the original biozone parameter recommendations (Siegrist et al., 2005) did not function suitably in SWAT. This occurred because the biozone parameter values recommended by Siegrist et al. (2005) were decided with the WARMF model, which differed in many aspects from SWAT, including the simulation time interval, watershed model configuration, and actual model processes. Therefore, new biozone coefficients were calibrated, adopted, and recommended for implementation in SWAT. Site 1 was used for calibration, and sites 2 and 3 were used for validation (not shown, but in the same vicinity as site 1). Both sites 1 and 2 were conventional drainfield systems, and site 3 was an advanced system with a geosynthetic textile used as the distribution medium in the drainfield trenches (instead of gravel aggregate). The STE

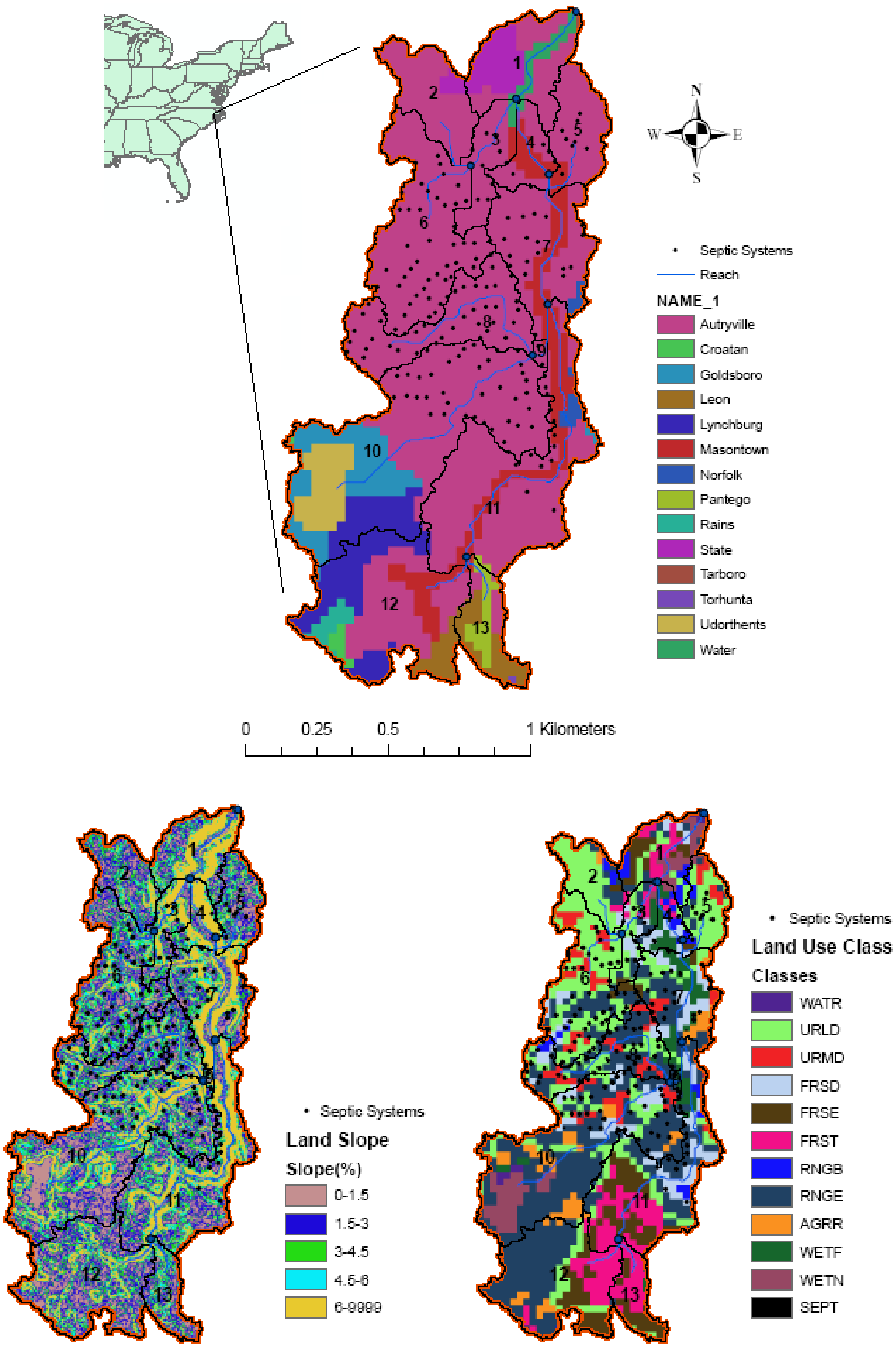

Figure 4. Hood's Creek watershed showing soil types (top), land slopes (bottom left), and land uses (bottom right).

Table 1. Summary of sampling sites characteristics.

Site	Septic System Type	No. of Bedrooms	No. of Occupants	Slope (%)	Age[a] (years)	Q_{STE} (L d^{-1})
1	Conventional	3	3	2	13	930
2	Conventional	3	6	3	13	1385
3	Advanced	3	3	4	2	798

[a] Age of septic system at year 2000 since installed.

rate at site 2 was 40% higher than at site 1, and the N concentration was 10% higher. The calibration involved evaluation of groundwater level, septic system service life, and nutrient data. In the validation, site-specific properties such as STE hydraulic loading rates and inflow nutrient concentrations were modified to correctly represent the validation sites, while other general septic parameters calibrated at site 1 were retained. The OWS parameter values determined from this exercise are shown in table 2 and are based on comparisons of time-series field data collected at the three residential sites mentioned previously. The results and discussion regarding these sites are presented in the next section.

Table 2. Recommended values for SWAT biozone parameters.

Variable	Description	Default	Range
isep_typ	Septic system type	1	1 to 28
isep_iyr	Year the septic system began operation	0	0 to 9999
isep_opt	System status (1 = active, 2 = failing, 0 = non-septic)	1	0 to 2
isep_cap	Number of permanent residents in the house	2.5	1 to 10,000
bz_area	Average area of individual septic drainfields (m^2)	100	10 to 1,000,000
isep_tfail	Time until a failing system gets fixed (days)	70	10 to 100,000
bz_z	Depth to the biozone layer (mm)	500	10 to 10,000
bz_thk	Biozone layer thickness (mm)	50	5 to 100
sep_strm_dist[a]	Average distance to the stream from the septic systems (km)	0.5	0.01 to 100
sep_den[a]	Number of septic systems per square kilometer	1.5	0.001 to 500
bio_bd	Density of biomass (kg m^{-3})	1000	900 to 1100
coeff_bod_dc	BOD decay rate coefficient	0.5	0.1 to 5
coeff_bod_conv	Ratio of live bacteria growth to BOD in STE (g g^{-1})	0.32	0.1 to 0.5
coeff_fc1	Field capacity coefficient 1	30	0 to 50
coeff_fc2	Field capacity coefficient 2	0.8	0.5 to 1
coeff_fecal	Fecal coliform bacteria decay rate coefficient	1.3	0.5 to 2
coeff_plq	Conversion factor for plaque from total dissolved solids	0.1	0.08 to 0.95
coeff_mrt	Mortality rate coefficient	0.5	0.01 to 1
coeff_rsp	Respiration rate coefficient	0.16	0.01 to 1
coeff_slg1	Sloughing coefficient 1	0.3	0.01 to 0.5
coeff_slg2	Sloughing coefficient 2	0.5	0.1 to 2.5
coeff_nitr	Nitrification rate coefficient	1.5	0.1 to 300
coeff_denitr	Denitrification rate coefficient	0.32	0.1 to 50
coeff_pdistrb	Linear P sorption distribution coefficient (L kg^{-1})	128	1.4 to 478
coeff_psorpmax	Maximum P sorption capacity (mg P kg^{-1} soil)	850	0 to 17,600
coeff_solpslp	Slope of the linear effluent soluble P equation	0.04	0 to 0.3
coeff_solpintc	Intercept of the linear effluent soluble P equation	3.1	0 to 10

[a] Place holders for future update.

RESULTS AND DISCUSSION

SWAT results including groundwater height, OWS service life, and nutrients (N and P) were evaluated at the three sites as described previously. Our results are presented below.

GROUNDWATER HEIGHT

Hoods Creek is ungauged, and thus calibration of streamflow was not possible. However, because accurate representation of infiltration, runoff, percolation, and evapotranspiration processes are important in the model, the model was calibrated to groundwater height instead. SWAT estimates groundwater height (mm) by evaluating the daily shallow aquifer recharge, groundwater recession, and specific yield of the shallow aquifer, where groundwater height is the depth of the groundwater from the bottom of the shallow aquifer to the unsaturated zone. In this instance, the reference elevation for the bottom of the groundwater table was assumed to be the bottom of Hoods Creek (i.e., the confining layer mentioned previously intersected Hoods Creek a short distance from the site). By taking the difference in elevation between site 1 (ground surface) and the creek (which was 4 m on the project DEM), the bottom of our monitoring wells were between 0.8 and 1.0 m above the creek bed (assuming groundwater table depths of 2 to 3 m in the project vicinity, as indicated by USGS NWIS for Craven County).

Humphrey (2002) recorded the groundwater height near drainfields over the study period, and these values were used in comparison with the model. Because the groundwater table was only minimally influenced by daily aquifer recharge (variation of <0.5 m temporally), seven-day moving average comparisons were made instead of daily output. SWAT-predicted groundwater heights for the calibration at site 1 are shown in figure 5. The simulations were quite good (R^2 = 0.82 and PBIAS = -0.8%) and reflected the general trend of the observed head over time. Similarly, hydrologic flux was well represented. The model indicated that 80% of the rainfall infiltrated into the soil surface (mainly due to the dominant highly infiltrative loamy sandy soils) and that potential evapotranspiration (PET) was quite high. Using the Penmen-Monteith method, PET was computed to be 1400 mm annually, which compares well with published actual pan evaporation data (Farnsworth and Thompson, 1983). Hence, from both lines of evidence (i.e., groundwater height and infiltration/PET representation), the model simulations were successful.

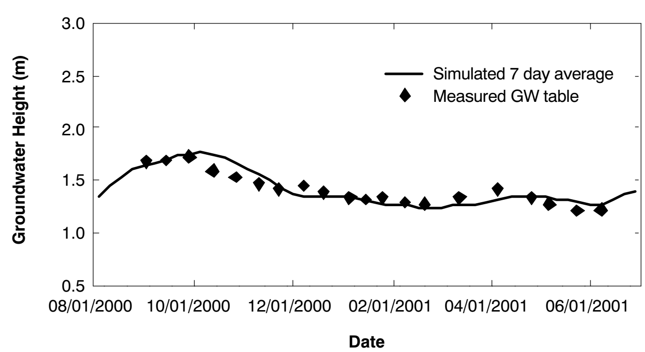

Figure 5. Groundwater levels at site 1 calibrated to the field data.

OWS Service Life

The service life of the OWSs in the Hoods Creek watershed (encompassing sites 1, 2, and 3, and others) were also evaluated. This was done as an additional check on the model parameterization. OWS service life varies significantly with local conditions, such as STE loading rate, septic maintenance, soils, and hydrologic properties of the site, and septic systems will function for several decades if properly constructed and maintained. Studies show that observed service life ranges from 11 to 30 years (Siegrist et al., 2001). For design purposes, the U.S. EPA recommends 20 years or less (assuming most household systems are not well maintained). Consequently, there is no specific guideline for modeling the service life of an OWS, despite the fact that such information is critically important in evaluating pollution from the untreated STE that is released from failing systems.

No direct field data are available for calibration or validation of OWS service life (Siegrist et al., 2005). However, probabilistic approaches are seemingly fruitful (Weintraub et al., 2004). In this work, Monte Carlo simulation techniques were used to create distributional variability in STE loading rates and associated service life response in SWAT. An expected STE flow rate of 227 L per person per day and a standard deviation of 50 L per person per day were used, and then random values were selected from the normal cumulative probability distribution function of STE loading rates to evaluate the SWAT response (fig. 6). Samples ranged from 82 to 339 L per person per day, with the 25th, 50th, and 75th percentiles being 194, 225, and 260 L per person per day, respectively. Other values (such as medians of total solids, BOD, and nutrients) were taken directly from McCray et al. (2005). The subsequent statistical distribution of the predicted OWS service life with the Monte Carlo approach is presented in figure 7.

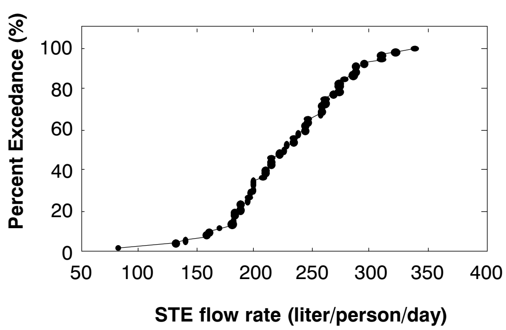

Figure 6. STE loading rates sampled by a Monte Carlo simulation.

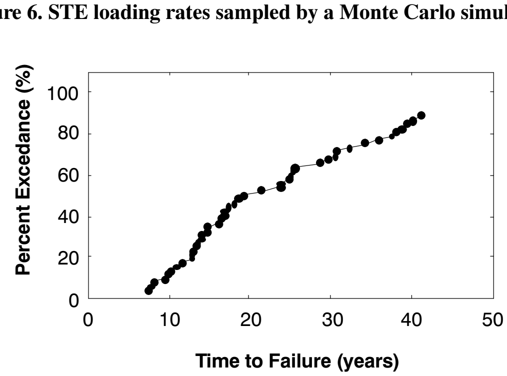

Figure 7. Frequency distribution of OWS service life.

As observed, in a 50-year SWAT simulation (1952-2002) among the 52 septic HRUs evaluated, the time to failure ranged from 5 to >35 years. No correlation was found between HRU area and time to failure, which is mostly because SWAT uses a depth unit for the STE loading rate after normalizing the volume rate with the HRU area. It was estimated that 75% of the conventional OWSs would fail in less than 35 years, with quartiles at 10 and 30 years and the median at 20 years. This result is close to the reported typical OWS life span (10 to >30 years) (Siegrist et al., 2001; USEPA, 2002), and the probabilistic exceedance curve also compares well with a similar effort made by Siegrist et al. (2005) with the WARMF model. If the variability in STE rate is removed (i.e., set at a constant value of 227 L per person per day, so only the influence of soil type is considered), the model predicts a much narrower range of OWS life spans, with a median of 20 $\pm$2 years. Hence, we feel that these two tests of service life provide additional evidence that our modeling approach is valid.

Nutrients

Nutrients (N and P) were also evaluated in the SWAT biozone algorithm. Each nutrient is addressed separately below.

Nitrogen Transport and Removal

The transport of NO_3^- through surface runoff, percolation, or groundwater in SWAT is estimated in terms of a daily mass load (not as solute concentration), and we used the seven-day average of the N yields through the soil profile divided by the volume of soil water percolation to make our model comparisons (these, in effect, are an N concentration, i.e., mass/volume). Initial nitrogen concentration in the groundwater was determined with a ten-year warm-up, which greatly reduced the initial condition error. Simulated N concentrations at site 1 (calibration) and site 2 (validation) are shown in figure 8. The observed concentrations showed large variability, ranging from <1 to 25 mg L^{-1}, and the central tendency of the data was used in the calibration (after removing outliers). The calibrated model reproduced the seasonal variation effectively, as evidenced by good statistical efficiencies (R^2 = 0.76, PBIAS = 2.5%). The predicted N concentrations also varied within the standard error of the data. Predicted N concentrations for the validation were also comparable with the field data throughout the test period (R^2 = 0.39, PBIAS = 0.58%). Interestingly, site 2 had nearly 50% more STE flux to the drainfield than site 1 (STE was similar in nitrogen concentration to site 1), but the mean values remained comparable to site 1 (although there were some higher maximum concentrations, nearly 41 mg L^{-1} below the drainfield). Based on these results, we feel that the model successfully reproduced the temporal mean profile of nitrate in the groundwater.

A detailed mass balance of N was also reported for the watershed (table 3). N in domestic wastewater accounted for about 85% of the total N input to the soil system at each OWS site (e.g., 500 kg N ha^{-1} is from STE, while the total N input is 583 kg N ha^{-1}). However, OWS contributed to only 25% of N inflow at the watershed level (9.3 kg ha^{-1} $year^{-1}$) as the 227 drainfields occupy relatively small areas compared to non-septic areas. It was estimated that 5.2 kg N ha^{-1} $year^{-1}$ was lost through denitrification, and 24.7 kg N ha^{-1} $year^{-1}$

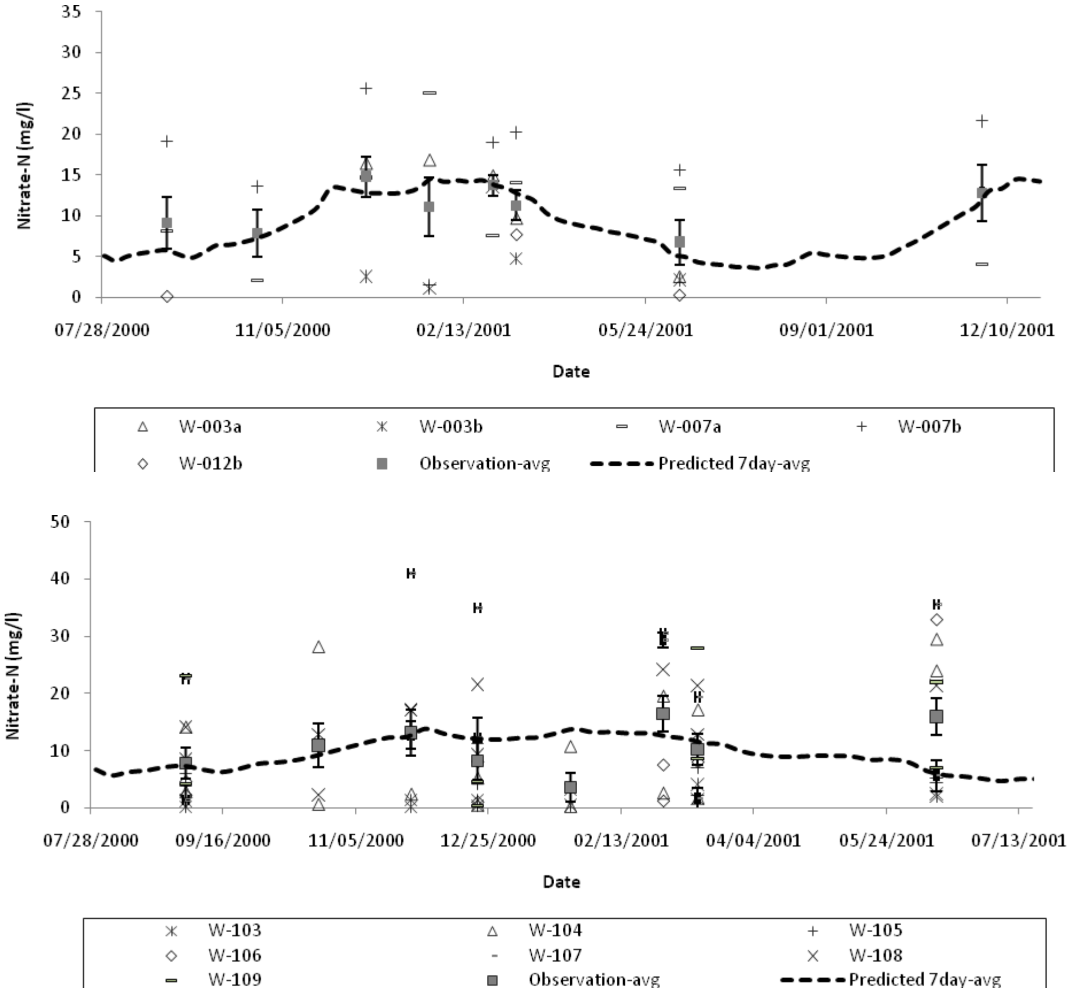

Figure 8. Predicted nitrate concentrations at site 1 (calibration, top) and site 2 (validation, bottom).

was removed by plant uptake (combining to 80% removal of the total N input). N loads from OWSs were estimated to be two orders of magnitude higher than from any other land use (fig. 9), and after denitrification and plant uptake, loadings from OWSs were slightly less than the N loading from atmospheric deposition of rainfall (1 mg L^{-1}). The results are consistent with the findings of Pradhan et al. (2005).

Flux of N through the soil profile was evaluated within 5%. N percolation to the shallow aquifer was twice that of the baseline scenario (where the watershed was assumed OWS-free), and it is not surprising that N concentrations in the groundwater (at least in the vicinity of OWS drainfields) are high (the data show nitrate concentration ranging from <1 mg L^{-1} to above 25 mg L^{-1}). Therefore, the OWSs in the Hoods Creek watershed may cause a noticeable increase of N concentration in the groundwater, at least in the vicinity of OWS drainfields. However, N removal for the entire watershed is very efficient, estimated at 82% (under baseline conditions). As a reference, the North Carolina Department of Environment and Natural Resources regulates TN removal of >60% for advanced septic systems (NCDENR, 2011). The predicted N removal efficiency was overall slightly underestimated for all septic sites (<5%) and ranged from 69% to 73%. For non-septic HRUs, N removal was much higher; for example, forest was 96%. The analysis suggests that while biological removal plays a significant role in N

Table 3. Simulated N balance and removal efficiency.

	N Input (kg ha^{-1} year^{-1})[a]				N Yield/Loss (kg ha^{-1} year^{-1})					% Removal	
Site	STE	Rainfall	F-MN	A-MN	Denit.	Uptake	Surf. *Q*	Lat. *Q*	Perc.	Pred.	Obs.
1	500.0	12.9	62.9	7.0	205.9	177.9	0.1	4.4	174.3	69	73
2	937.5	12.9	65.0	7.1	532.9	181.3	0.1	10.8	265.4	73	75
3	360.0	12.8	61.6	6.4	153.6	173.6	0.1	6.4	119.6	71	75
Forest	--	12.7	9.6	5.0	--	25.4	0.1	0.0	1.2	96	n/a
Total[b]	9.3	12.8	9.3	5.8	5.2	24.7	1.5	0.2	5.2	82	n/a
No OWS[c]	--	12.8	8.7	5.9	--	22.9	1.5	0.1	2.5	85	n/a

[a] STE = septic tank effluent, F-MN = fresh organic to mineral N, and A-MN = active to mineral N.
[b] Watershed averages (current condition).
[c] Assumes no nitrogen inflow from OWSs.

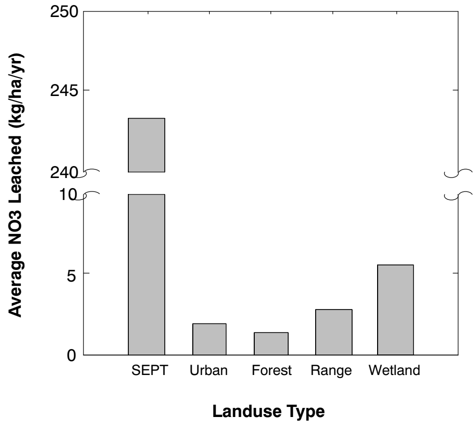

Figure 9. Amount of nitrate leaching to the groundwater in different land use types in the Hoods Creek watershed.

reduction of STE through denitrification, N loading is still much greater than that of natural conditions.

Phosphorus

P output in SWAT is similar to N (in relation to mass output, not concentration). Therefore, the same assumptions used in the N evaluation were made for P. Observed soluble P concentrations were around 6 mg L^{-1} in the STE and 1 mg L^{-1} in the groundwater, and the calibration of soluble P was made by fitting the model output to the P time series field data. Unfortunately, though, the calibration was not successful, and the predicted values from the model were two orders of magnitude lower than the field data, even when using the best combination of related model parameters. This is believed to be related to two factors: the soil type of the Hoods Creek watershed, and the current limitations of SWAT. First, note that the dominant soil type was loamy sand (Autryville), which field fractions suggest comprises about 90% of the soil at site 1 (in the biozone layer) and >80% at site 2. Given that P sorption is related to cation exchange

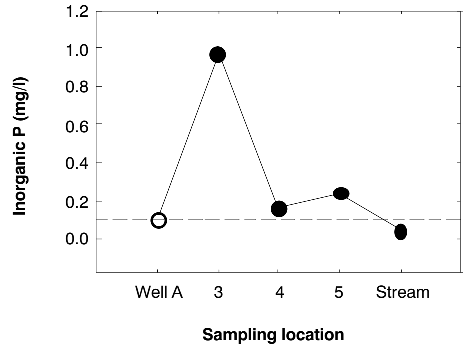

Figure 11. Spatial distribution of P along the groundwater path (shown in fig. 10) from septic drainfield to the creek at site 1.

capacity (CEC, i.e., the ability to sorb P), and CEC is generally high in clay soils and low in sandy soils, it is not surprising that there was very little P sorption capacity (i.e., low CEC) at our site because of the high percentage of sand, thereby explaining the high P concentrations (>1 mg L^{-1}) in the groundwater. Secondly, the SWAT biozone P algorithm currently does not take the percent ratio of sand, silt, and clay into consideration in predicting the fate of P in soils, which is a major limitation. Instead, it depends on a parameter defined at the watershed level (i.e., one parameter applies to all HRUs). This limitation is recommended to be altered in future SWAT development work.

To further evaluate the P simulation, we reviewed P concentrations along the groundwater flow path at the project site. As indicated in figure 10, the groundwater flows northeasterly, and those piezometers that intersect the drainfield flow path (both upstream and downstream) show a distinct signature P concentration. Beginning at well A (upstream of the septic drainfield), the background P concentration in the groundwater is approximately 0.1 mg L^{-1} (fig. 11, dotted line). It quickly spikes at the drainfield (well 3 = 1.0 mg L^{-1}) and then returns to background levels

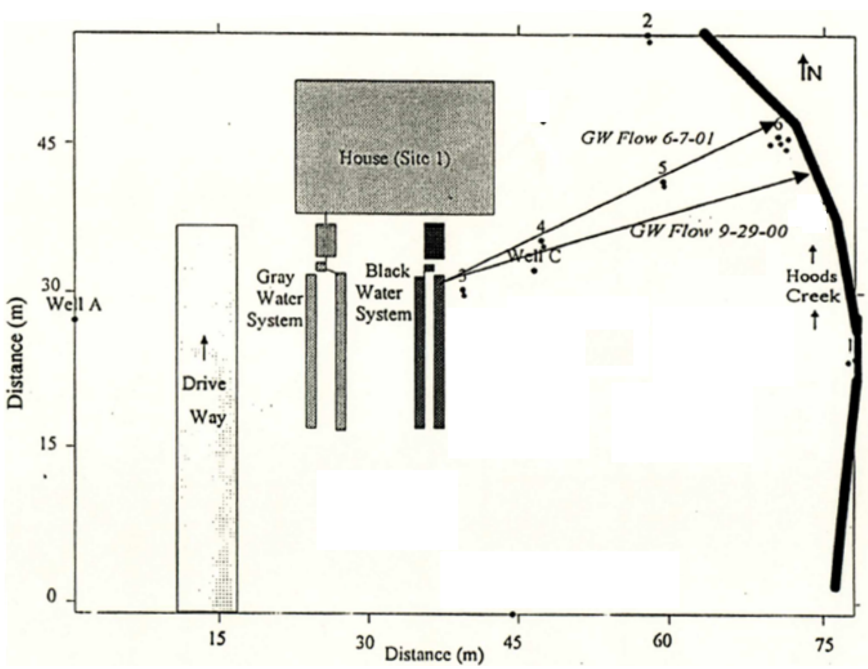

Figure 10. Observed groundwater flow path at site 1 (September 2000-highest water level, June 2001-lowest water level).

a mere 30 m farther down-gradient (e.g., well 4 = 0.16 mg L^{-1}, well 5 = 0.24 mg L^{-1}, stream = 0.04 mg L^{-1}). Therefore, P removal is very quick, and despite the inability to simulate P removal directly in the biozone, nearly all of the increase in concentration from the drainfield STE is mitigated through either soil sorption or groundwater dilution (removal is estimated to 90% by wells 4 and 5).

SUMMARY

A process-based biozone algorithm was developed in SWAT to simulate the environmental effect of onsite wastewater systems (OWSs) at the watershed scale. This represents a significant improvement in the capability of SWAT to characterize point and nonpoint source pollutant loads. The overall performance of the SWAT biozone algorithm was tested in a case study in the Hoods Creek watershed in North Carolina. Based on our findings (including evaluations of groundwater height, OWS service life, and nutrients), we feel that groundwater simulations of N concentration in OWS settings are reliable, as evidenced by the calibration and validation tests presented herein. OWS loads are a significant source of N in the watershed evaluated (~25%), although more than 80% of the input to the watershed was found to be removed before reaching the creek (from denitrification and plant uptake). In regard to P, the model simulations were less reliable. This is believed to be primarily related to the cation exchange capacities of the soil in the test watershed and the fact that SWAT currently does not have the ability to vary the mobility of soluble P (linear sorption isotherm coefficient) by soil type (i.e., sand, clay, etc.). Hence, the soil layers did not allow P leaching through the soil absorption field in the model, and P concentrations in groundwater were greatly underpredicted. However, P loss did occur over longer lengths (30 m). In the future, additional work should be done to improve the P routines in the SWAT model.

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