Simulation of a Low-Gradient Coastal Plain Watershed Using the SWAT Landscape Model

D. D. Bosch, J. G. Arnold, M. Volk, P. M. Allen

Abstract. *Accurate simulation of landscape processes in natural resource models requires spatial distribution of basin hydrology and transport processes. To better represent these processes, a landscape version of the SWAT model has been developed to simulate the runoff, run-on, and infiltration processes that typically occur in different parts of the landscape. The model addresses flow and transport across hydrologic response units prior to concentration in streams, and is capable of simulating flow and transport from higher landscape positions to lower positions. The SWAT landscape model was tested using data collected from a heavily vegetated riparian buffer system in the Atlantic Coastal Plain near Tifton, Georgia. Simulations of surface runoff, lateral subsurface runoff, and groundwater flow for an upland field, a grass buffer, and a sub-divided forested buffer floodplain were generated. Model results and field data indicate that surface runoff was dominant in the upland field, while groundwater flow was dominant in the grass buffer and the floodplain. While average annual surface runoff agreed satisfactorily with observations from the site, annual and monthly simulated values varied considerably from observed values. Simulated surface runoff tracked general trends in the observed data, but winter months and extreme events were overestimated while summer months were underestimated. Annual surface runoff predictions at the edge of the upland field varied from the observed data by 11% to 44%. The Nash-Sutcliffe efficiency for annual estimates of surface runoff at the field edge was 0.83 for the three-year calibration period. The results demonstrate the ability of the model to simulate the surface runoff and enhanced infiltration typically associated with riparian buffer systems. Additional revision of the model will likely be necessary to adequately represent redistribution of water between surface, lateral subsurface flow, and groundwater flow.*

Keywords. *Natural resource modeling, Surface hydrology, Watershed modeling.*

Watershed models are valuable tools for examining the impact of land use on hydrology and water quality. While extensive research has been done to describe the impact of management practices on field and farm runoff, less is known about how these changes are reflected at the watershed scale. The success of the Total Maximum Daily Load (TMDL) program in the U.S. will be based on water quality improvements that result at the watershed scale. Additionally, a national assessment of the effects of conservation practices on watershed-scale water quality is underway that relies heavily on the reliability of watershed flow and transport models (Mausbach and Dedrick, 2004).

The SWAT model has been applied to watersheds throughout the world (Arnold and Fohrer, 2005; Gassman et al., 2007). The model has received extensive testing in Texas (Srinivasan et al., 1997; Saleh et al., 2000; Santhi et al., 2001), Kentucky (Spruill et al., 2000), Wisconsin (Kirsch et al., 2002), Mississippi (Bingner, 1996), Indiana (Smithers and Engel, 1996), Pennsylvania (Peterson and Hamlett, 1998), and Georgia (Bosch et al., 2004; Van Liew et al., 2007) in the U.S. In most cases, the prediction accuracy was satisfactory to obtain working knowledge of the hydrologic system and the processes occurring in the watersheds. One of the shortcomings of SWAT has been an inability to model flow and transport from one position in the landscape to a lower position prior to entry into the stream. The model utilizes a hydrologic response unit (HRU) concept. HRUs are lumped land areas within each subbasin that are comprised of unique land cover, soil, and management combinations. Transported water, sediment, and chemicals from the HRU are currently routed directly into the stream channel by SWAT, bypassing lower landscape units. As currently configured, SWAT does not simulate transport from upslope HRUs to flow through lower landscape position HRUs prior to entry into the stream.

Submitted for review in October 2009 as manuscript number SW 8271; approved for publication by the Soil & Water Division of ASABE in July 2010.

Contribution from the USDA-ARS Southeast Watershed Research Laboratory, Tifton, Georgia, in cooperation with the University of Georgia Coastal Plain Experiment Station.

The authors are **David D. Bosch, ASABE Member Engineer,** Research Hydrologist, USDA-ARS Southeast Watershed Research Laboratory, Tifton, Georgia; **Jeffrey G. Arnold, ASABE Member Engineer,** Agricultural Engineer, USDA-ARS Grassland Soil and Water Research Laboratory, Temple, Texas; **Martin Volk,** Geographer, Department of Landscape Ecology, UFZ-Helmholtz Center for Environmental Research, Leipzig, Germany; and **Peter M. Allen, ASABE Member,** Geologist, Department of Geology, Baylor University, Waco, Texas. **Corresponding author:** David D. Bosch, USDA-ARS SEWRL, P.O. Box 748, Tifton, GA 31794; phone: 229-386-3515; fax: 229-386-7294; e-mail: David.bosch@ars.usda.gov.

While the intended use of the SWAT model is for river basin scale applications, calculations within the model at the HRU level are based on the CREAMS, GLEAMS, and EPIC models (Gassman et al., 2007) and are applicable for smaller field-sized areas. At the HRU level, surface runoff is computed using the SCS curve number method and erosion using MUSLE (Arnold et al., 1998), both of which are frequently applied on plot and field sized areas. While most testing of the SWAT model has been for large areas (Gassman et al., 2007), the model has also been successfully applied to small-scale watersheds. Arnold et al. (2005) applied the SWAT model to the 53 ha USDA-ARS Y-2 watershed in Texas. HRUs for the Texas application were on the order of 10 ha. Bingner (1996) applied the model to a 5 ha watershed in Mississippi. Runoff parameters for the Mississippi watershed were derived using data from a 3.1 ha field. Chanasyk et al. (2003) applied SWAT to three watersheds from 1.5 to 226 ha in Saskatchewan, Canada. Van Liew et al. (2007) applied the model to a 40 ha watershed in Pennsylvania. The Pennsylvania watershed was divided into five HRUs, an average of 8 ha each.

The importance of landscape units between upland fields and streams is well documented in the literature. Grass filter strips have been shown to reduce sediment and agrichemical loading and enhance infiltration (Barfield et al., 1979; Magette et al., 1989; Dillaha et al., 1989; Arora et al., 1996; Sheridan et al., 1999). Early research on the quality of streamflow from Atlantic Coastal Plain agricultural watersheds indicated that lower landscape riparian forests played an important role in reducing agrichemical transport from these watersheds (Asmussen et al., 1979; Sheridan et al., 1982; Yates and Sheridan, 1983). Further research on these riparian buffers at several locations in the Atlantic Coastal Plain indicated that the riparian buffers enhanced infiltration and subsurface flow, and reduced nutrient loading from upland agricultural fields (Lowrance et al., 1983, 1984, 1986; Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985). Because of the significance of grass filter strips, riparian forests, and other landscape units that interact with surface and subsurface runoff from upland fields in many watersheds, it is important to incorporate these concepts into watershed-scale natural resource models.

Arnold et al. (2010) developed a modification to the SWAT model that facilitates the simulation of transport from one landscape unit to the next prior to flow into the stream. One of the methods of subwatershed discretizations explored by Arnold et al. (2010) was the catena approach (Lane and Nearing, 1989; Kirkby et al., 1998). The catena approach divides the catchment into upslope and downslope landscape units, conceptually the upland divide, the hillslope, and the valley bottom (fig. 1) at the subwatershed scale. This model structure more closely reflects the complex controls on infiltration, runoff generation, run-on, and subsurface flow. The landscape positions can differ in terms of soil type, slope, vegetation, and management. The modified model routes surface runoff, lateral subsurface flow, and shallow groundwater flow from the upland divide, through the hillslope, through the floodplain, and eventually to the stream. Through this approach, the impact of the upslope management on downslope landscape positions can be assessed. In-stream processes are currently not simulated by the revision nor are water quality functions (Arnold et al., 2010).

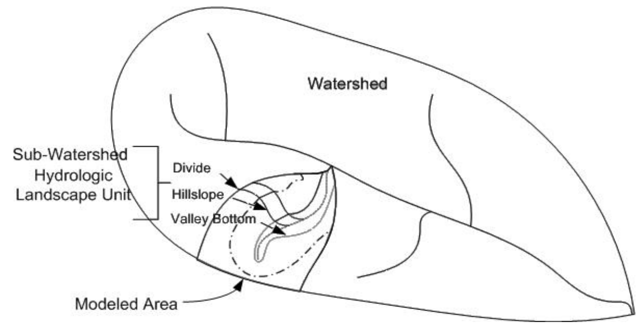

Figure 1. Subwatershed landscape delineation for the SWAT landscape revision following the catena approach.

While the SWAT landscape revision has undergone limited testing (Arnold et al., 2010), further testing and review are necessary to confirm model operation. To adequately understand the capabilities of the model, each component of the revision must be tested. Arnold et al. (2010) tested the landscape model on the 17.3 km^2 Brushy Creek subwatershed in the Blackland Experimental Watershed near Riesel, Texas (Arnold et al., 2010). Results indicated that the new landscape model, after routing across the landscape units, was comparable to the existing SWAT model structure for simulating total water flow. The emphasis of the analysis was on comparison of the different methods of landscape delineation. While comparisons were made between predicted and observed streamflow at the watershed outlet, detailed comparisons to observed data were not made at the landscape scale. Detailed testing of the simulation of transport processes between the landscape components and outputs of each landscape unit is necessary to complete model validation. To examine the landscape routing in the SWAT revision, appropriate testing must be conducted at the HRU level. This analysis separates the testing of the channel routing from the testing of the HRU processes and linkages between the HRUs, which provides for a more thorough test of the landscape SWAT model.

Stepwise testing will be developed to examine first the hydrologic components at the landscape scale, then the water quality functions at the landscape scale, and finally the instream processes. Here we examine the hydrologic components of the SWAT landscape model at the landscape scale. Specifically, the objectives are to test the hydrologic component of the SWAT landscape model at the HRU level for a four-component hillslope in south-central Georgia. Riparian buffers within this region are an important and dominant part of the landscape. For this geographic region, the drainage density is typically high compared to other regions, and the farmed components between these streams are typically less than 100 ha. The fields typically drain via nonconcentrated surface runoff through edge-of-field vegetated buffers. Contributing areas to each buffer section along the streams are typically a small fraction of the field areas. For the analysis presented here, surface, lateral subsurface, and groundwater flow at the outlet of the landscape units will be examined and compared to observed data. In addition, comparisons are made between simulations obtained using the original SWAT 2005 for a single HRU and simulations obtained using the landscape revision. The comparison between results obtained with the SWAT landscape model and the single-HRU SWAT 2005 provides guidance on the conditions that the landscape revision may be able to represent more accurately than the original SWAT model.

Methods

Site Description

The study site consisted of an upland agricultural field draining through a riparian forest buffer into a first-order stream (fig. 2). Surface and subsurface runoff originating within the upland field, transported through the buffer at the edge of the stream, and discharging into the stream were examined using the SWAT landscape revision. The Fox Den field study site is located on the University of Georgia Gibbs Farm near Tifton, Georgia. The Fox Den field and the riparian buffer surrounding it were the focus of extensive research from 1992 to 2004 (Sheridan et al., 1999; Hubbard and Lowrance, 1997; Inamdar et al., 1999a, 1999b; Bosch et al., 1994, 1996, 2003; Lowrance et al., 2007). SWAT simulations were configured to match the drainage characteristics of the field and the riparian buffer the field drained into. For this study, data collected from the buffer study area and the field area draining into the buffer were used (fig. 2). Rainfall data were obtained from an on-site recording tipping-bucket rain gauge (Texas Electronics, Inc.) reported in prior studies (Sheridan et al., 1999). Surface runoff data have been reported in Sheridan et al. (1996), Sheridan et al. (1999), and Inamdar et al. (1999a).

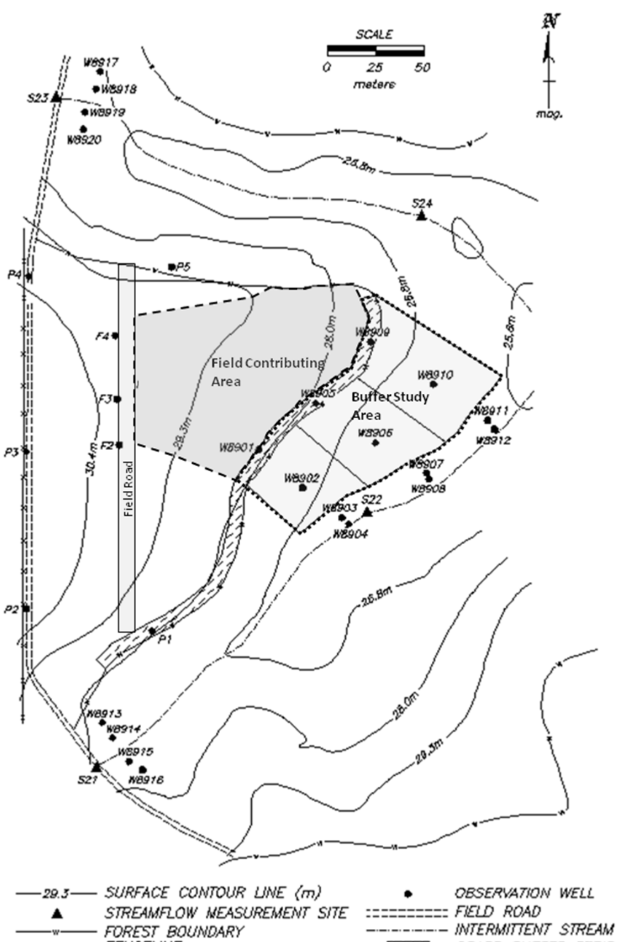

Figure 2. Simulated hillslope at the Gibbs Farm Fox Den field.

The hillslope study area consists of an upland tilled field, an 8 m wide grass buffer at the lower boundary of the field, and a 55 to 65 m wide forested buffer between the grass buffer and the stream (fig. 2). Upland and riparian buffer data collected at the site, used for this analysis, were obtained from measurements collected at the edge of the upland field and within three plot areas covering an approximate total area of 0.84 ha (Sheridan et al., 1999; Inamdar et al., 1999a). The upland field is divided by a field road crossing the middle of the field such that only 50% of the field surface runoff drains into the buffer along the eastern edge of the field (fig. 2). For this analysis, subsurface drainage was assumed to conform to surface drainage patterns. Sheridan et al. (1999) reported that the contributing area from the upland field to the studied buffer area was 0.93 ha, yielding approximately a 1.1:1 field to buffer ratio. Conventionally tilled corn, peanuts, and pearl millet were grown in the upland field. The grass buffer was seeded in common bermudagrass (*Cynodon dactylon* L. Pers.) and bahiagrass (*Paspalum notatum* Flugge). The grass buffer was interplanted with perennial ryegrass (*Lolium perrene* L.) to provide additional biomass production and nutrient uptake during the first winter. The biomass in the grass buffer was harvested twice annually. The riparian forest buffer consisted of a 45 to 55 m wide band of mature slash pine (*Pinus elliottii* Engelm.) and longleaf pine (*Pinus palustris* Mill.) and a 10 m wide band of hardwoods, including yellow poplar (*Liriodendron tulipifera* L.) and swamp black gum (*Nyssa sylvatica* var *biflora* Marsh.). The width of the entire buffer, from the edge of the stream to the edge of the tilled field, averaged 70 m and the length parallel to the stream was approximately 120 m. The soil type within the upland field and the grass buffer is a Tifton loamy sand, while that in the forested area is an Alapaha loamy sand.

The geology and soils of the region are conducive to lateral subsurface flow of water within the vadose zone and shallow groundwater. The Tifton soil contains subsurface horizons with reduced infiltration rates that perch water and initiate lateral flow during wet conditions (Hubbard and Sheridan, 1983). The Tifton soil contains 7% to 14% plinthite from 0.8 to 1.4 m. The region is in the outcrop area of the Miocene Hawthorn formation (Asmussen, 1971). This formation is the geologic parent material and is overlain by Quaternary sands. The Hawthorn formation is believed to be continuous throughout the region and serves as an aquiclude in the Tifton Upland (Stringfield, 1966). The surficial aquifer formed by the Hawthorn formation and the lateral flow initiated by the plinthic soils generate baseflow within regional streams and lead to saturation excess conditions along the stream channels. In the area of the Fox Den field, the Hawthorne confining layer varies from 4.5 m below the land surface at the top of the hillslope to approximately 2 m below the stream bottom (Bosch et al., 2003).

Surface Runoff Data

Surface runoff data from the Fox Den field and the downslope riparian buffer were collected from February 1992 through December 1996 (Sheridan et al., 1999). Surface runoff was characterized at four positions: the upland field edge, the interface between the grass buffer and the pine buffer, mid-way into the pine buffer, and at the interface between the pine buffer and the yellow poplar buffer along the stream edge. No surface runoff data were collected at the interface between the yellow poplars and the stream. Measurement of surface runoff at the site required the use of low-impact surface flow samplers (Sheridan et al., 1996) to minimize disturbance of the surface runoff between landscape positions. The low-impact sampling approach is based on the assumption that surface flow entering the riparian buffer system is shallow and is not concentrated, considered a good approximation for the region (Sheridan et al., 1996). The samplers provide estimates of surface runoff while providing little disturbance to the ground surface, vegetation, or flows (Sheridan et al., 1996; Sheridan et al., 1999). Surface flow volumes estimated using the approach are likely less precise than estimates using more conventional runoff plot measurement devices (Sheridan et al., 1996). Further examination of the samplers indicated that they tended to underpredict extreme events (Sheridan et al., 1996). The surface runoff collectors provide reliable long-term estimates for relative treatment comparison but are not adequate for event-based comparison. While the riparian buffer field study examined three treatments in the buffer area (clear cutting, thinning, and no cutting), only the data from the mature forest treatment were used for this analysis where treatment effects were found to have a significant impact on the surface runoff results. Sheridan et al. (1999) reported no significant difference between runoff or sediment concentrations at the field edge or at the interface of the grass/pine forest buffers collected from the three different plots. For the edge of the field and the grass/pine forest interface, all data from all samplers were used. Use of the data from all of the samplers increased replication and improved data quality. Treatment effects were found to be significant for surface runoff and sediment concentrations at the pine forest and poplar forest interface (Sheridan et al., 1999). For the position below the pine forest buffer, only the data for the mature forest treatment was used in order to avoid treatment effects.

Direct observations of lateral flow are not available for the site. Lateral flow within the vadose zone was estimated using data collected within a nearby tilled field (Bosch et al., 2005). The study characterized surface runoff and shallow subsurface flow from plot-sized upland areas in strip and conventional tillage from 1999 to 2003. Lateral subsurface flow averaged 9% of precipitation for conventional tillage plots and 16% of precipitation for strip tillage plots studied (Bosch et al., 2005). Soils at the site were classified as a Tifton loamy sand with a 3% to 4% slope, similar to that found in the Fox Den field. Because the upland Fox Den field was in conventional tillage, the measurement of lateral flow from the conventionally tilled plots (9% of annual precipitation) was used to estimate lateral flow contributions for the upland field.

Subsurface hydrology at the riparian buffer site has been extensively studied (Bosch et al., 1994, 1996, 2003). Annual groundwater yield to streamflow from this side of the Fox Den field varied from 7% of annual precipitation to 32% (Bosch et al., 2003). The shallow groundwater aquifer from this area of the Fox Den field discharges into the stream for 70% of the year and accounts for approximately 60% of the total streamflow in this watershed. Matric potential (Bosch et al., 1994), surface runoff (Sheridan et al., 1999), and groundwater (Bosch et al., 1996, 2003) data collected at the site indicate significant infiltration of surface runoff at the interface between the grass and riparian buffers. Subsurface hydrologic data indicate that the riparian buffer contributes significantly to streamflow during periods of alluvial saturation.

SWAT Landscape Revision

The landscape version of the SWAT model was developed to simulate the flow of water from one landscape unit into another prior to entering the stream channel. Currently, the model simulates hydrologic flow through the hillslope only, with no in-stream or water quality components (Arnold et al., 2010). Specific details on the model are provided by Arnold et al. (2010). An overview of the model calculations are provided here for completeness. The landscape model simulates surface runoff, lateral subsurface flow, and shallow aquifer flow between landscape units (fig. 3). Calculations within the landscape version of SWAT are similar to those within the original version. Surface runoff is simulated using the curve number or Green and Ampt infiltration equation. Run-on to an adjacent downslope landscape unit is estimated using a coefficient to partition the amount of flow that is channelized before leaving the landscape unit and the amount that is direct surface run-on. The amount of surface run-on that infiltrates is determined by multiplying the travel time by the saturated conductivity of the soil. Percolation is modeled with a layered storage routing technique combined with a crack flow model.

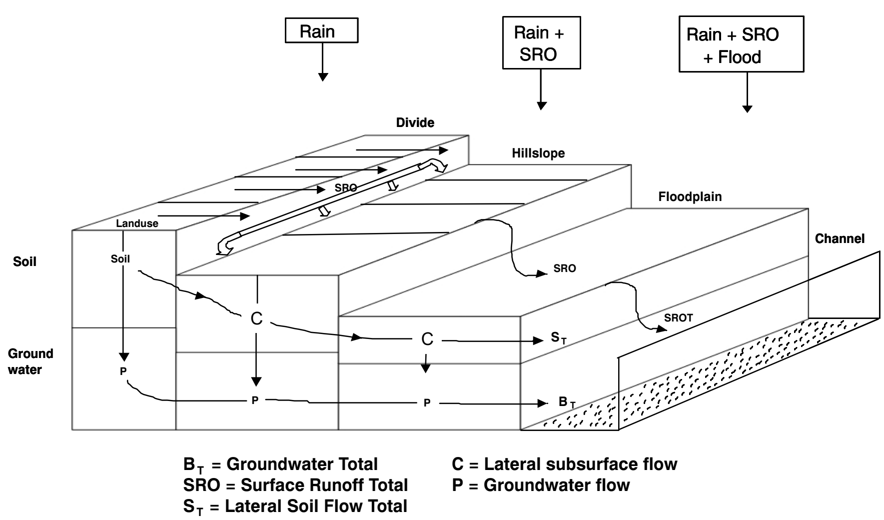

Figure 3. Processes considered in routing units of the revised SWAT landscape model (Arnold et al., 2010).

The model accommodates multiple soil layers as required to account for vertical heterogeneity and soil horizons typically defined in U.S. soil surveys. Lateral flow volumes are calculated using a kinematic storage model (Arnold et al., 1998). Conceptually, groundwater flow is simulated as routing through a series of linear storage elements based on the classic linear tank storage model (Brutsaert, 2005). Groundwater flow from the valley bottom landscape unit contributes directly to the streamflow. During low flow, channel seepage or transmission losses recharge the shallow aquifer of the floodplain unit.

Simulations

The Fox Den field and the down-gradient riparian buffer were simulated with the SWAT landscape model utilizing the catena delineation method (fig. 2). The catchment was manually configured to simulate one subbasin divided into three landscape units (upland divide, hillslope, and a sub-divided valley bottom), as depicted in figure 4. The valley bottom, or floodplain, was divided into two HRUs to accommodate a change in vegetation (fig. 4). Row crops were simulated for the upland divide (HRU1), bermudagrass in the hillslope (HRU2), pine trees in the upslope floodplain position (HRU3), and yellow poplar trees in the portion of the floodplain nearest the stream (HRU4).

The 0.93 ha contributing area from the upland field and the 0.84 ha riparian buffer area flowing into a 120 m length section parallel to the stream were simulated (fig. 2). The simulated upland was dictated by the area contributing to the riparian buffer study area. The total simulated area was 1.77 ha (table 1). This configuration closely resembles the drainage at the Fox Den field. Slope lengths and slopes were determined from site characteristics and published reports (Sheridan et al., 1999; Hubbard and Lowrance, 1997; Inamdar et al., 1999a, 1999b; Bosch et al., 1994, 1996, 2003). The upland and the grass buffer were simulated with the soil type of Tifton loamy sand, while the forested buffer was simulated with a soil type of Alapaha loamy sand. The SCS curve number method was used because of a lack of site-specific Green and Ampt parameters for the study site. Based on prior results with the SWAT watershed model (Bosch et al., 2004; Feyereisen et al., 2007; Van Liew et al., 2007), the Hargreaves method was selected to estimate evapotranspiration.

Calibration and Validation

Data collected from February 1992 through December 1996 from the riparian buffer were used for model testing

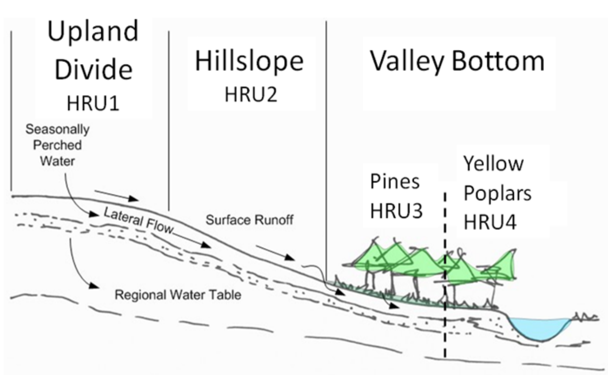

Figure 4. Schematic of simulated flow system.

Table 1. Characteristics of the HRUs in the Gibbs Farm Fox Den field simulation.

Landscape Unit	Slope Length (m)	Slope (%)	Area (ha)	Fraction of Simulated Area (%)
HRU1 Upland field	40	3.0	0.93	53
HRU2 Grass buffer	30	2.4	0.10	5
HRU3 Pine buffer	30	2.0	0.62	35
HRU4 Poplar buffer (yellow poplar)	30	1.0	0.12	7
Total watershed			1.77	100

(Sheridan et al., 1999). Surface runoff data collected from February 1992 through December 1994 were used for calibration, while data collected from January 1995 through December 1996 were used for model validation. A simulation period from January 1991 through January 1992 was used as a warm-up period for the simulation. Corn was grown in the upland field in 1990, 1992, 1993, and 1994; peanuts in 1991 and 1996; and pearl millet in 1995. Planting and harvest dates for each crop type were kept consistent with local practices but were the same for each individual crop across the different years. All of the corn was simulated with a planting date of March 12 and a harvest date of July 30. The planting date for the peanuts was simulated as June 7 and the harvest date as October 21. The planting date for the pearl millet was simulated as May 12 and the harvest date as September 27. Fallow conditions were assumed for periods between harvest and planting. A daily rainfall data file was created for the period from 1991 to 1996 using available precipitation data collected at the site. Other climatic inputs were simulated by the model.

Limited model calibration was conducted to fit the simulations to surface runoff observations from the upland field and the riparian buffer. Some of the parameters for the four HRUs were obtained from prior calibration within similar watersheds. Following the parameter analysis conducted by Feyereisen et al. (2007), the parameters GW_DELAY, GW_REVAP, and ALPHA_BF were defined for the four HRUs. The parameters ESCO, OV_N (Manning's n), and initial CNII were varied by HRU. The OV_N parameter was determined based on tabular values for each landcover. ESCO and CNII were used as calibration parameters to adjust evapotranspiration and surface runoff. A lower bound for CNII was set at 48 for the grass buffer, while lower bounds of 50 were assumed for CNII for the forested HRUs. In order to change infiltration in lower landscape positions of surface runoff generated upslope, soil parameters had to be modified. The saturated hydraulic conductivities (SOL_K) for the first two layers of the soil were used as calibration parameters to modify infiltration. These layers were partitioned from 0 to 254 mm and from 254 to 457 mm for the Tifton soil, and from 0 to 838 and from 838 to 1219 mm for the Alapaha soil. The parameters were adjusted for the grass buffer and the pine buffer HRUs in order to obtain the desired infiltration rate of upland runoff. Because there was no significant channelized flow from one landscape position to the adjacent landscape position, the coefficient to partition the amount of flow that is channelized before leaving each landscape unit was set to 0. For these simulations, the fraction of water losses into the deep aquifer was set to 0.01 based on estimates from Sheridan (1997).

Adjustments were made to the calibration parameters within the four HRUs to obtain a surface runoff from each HRU that best approximated the observed data. Target average annual surface runoff values were obtained from the Gibbs farm riparian buffer plots for each landscape position, the upland field edge, the grass buffer edge, and the pine buffer edge for the period from February 1992 to December 1994. The observed surface runoff from the upland field only reflects what is coming from the field itself. The observed surface runoff from the grass buffer and the pine buffer reflects the sum of surface runoff generated within that landscape position, surface runoff from upslope that has not infiltrated, and groundwater seeps.

Test statistics used to evaluate goodness of fit of the surface runoff simulations included average annual surface runoff, residuals between observed and predicted annual surface runoff, the Nash-Sutcliffe efficiency (NSE) index (Nash and Sutcliffe, 1970) for the annual surface runoff (NSE_{ann}), residuals between the observed and the predicted monthly surface runoff, and the NSE for the monthly surface runoff volumes (NSE_{mon}). NSE can range between $-\infty$ and 1.0, with 1.0 being the optimal value. Based on Moriasi et al. (2007), model performance for streamflow can be considered satisfactory if NSE_{mon} values are >0.50. No comparisons between simulated and observed surface runoff at the edge of the yellow poplar area were made because there were no observations made at this interface.

Model Comparison

Simulations were also run to make a model comparison between SWAT 2005 and the SWAT landscape revision. For the SWAT 2005 simulations, only the upland HRU was simulated. For SWAT 2005, the model would normally be configured to simulate the dominant land use and soil type configuration. The upland is the dominant portion of the landscape unit simulated (53%). The land slope for the upland HRU for the SWAT 2005 simulation was set at 3% to reflect the slope from the upland to the stream. The area of the upland HRU for the SWAT 2005 simulation was set at 1.77 ha to match the acreage for the landscape simulation. All other parameters for the upland HRU were set the same for this landscape unit as those obtained through the calibration of the landscape model.

Results and Discussion

Calibration

The calibration period was from February 1992 through December 1994. The resultant parameters obtained through the calibration are shown in table 2. Calibration for the upland field required adjustments to the parameters related to surface runoff (ESCO and CNII). Calibration for the grass buffer and the pine buffer required adjustments to the surface runoff related parameters as well as adjustments to infiltration related parameters within the HRUs. To obtain the desired infiltration within the grass buffer, SOL_K for layer 1 was adjusted to 8000 mm h^{-1} and SOL_K for layer 2 was adjusted to 1000 mm h^{-1} (table 2). These values are an order of magnitude greater than what would typically be expected for the Tifton soil type, but they are indicative of the rapid infiltration observed in the grass buffer. For the pine buffer and poplar buffer HRUs, only SOL_K for layer 1 was adjusted. For these HRUs, SOL_K for layer 1 was increased to 600 mm h^{-1} from 230 mm h^{-1}.

Table 2. Calibration parameters for the four-HRU study areas.

Parameter	HRU1 Upland Field	HRU2 Grass Buffer	HRU3 Pine Buffer	HRU4 Poplar Buffer
ESCO	0.95	0.74	0.74	0.74
OV_N	0.09	0.41	0.50	0.50
CNII	88	48	50	63
SOL_K layer 1 (mm h^{-1})	400	8000	600	600
SOL_K layer 2 (mm h^{-1})	44	1000	22	22

The observed and simulated average annual surface runoff for the upland, grass filter, pine buffer, and poplar buffer HRUs are shown in table 3. For the calibration period, the average annual surface runoff simulated for the edge of the upland field was 29% of precipitation, while the average annual observed value was 27%. The predicted volume of annual surface runoff at the edge of the upland field varied from the observed data by 11% to 44% over the three-year calibration period (table 4).

Annual precipitation over the calibration period varied from 1124 mm observed from February 1992 to December 1992 to 1428 mm observed in 1994 (table 4). The NSE_{ann} for the surface runoff from the upland field HRU was 0.83. The annual average was skewed somewhat by an attempt to fit the high runoff observed in 1994, which led to an overestimation for 1992 and 1993 (table 4). For the calibration period, simulated surface runoff at the upland field edge (HRU1) varied from a high of 505 mm (35% of precipitation) simulated for 1994 to a low of 263 mm (25% of precipitation) simulated for 1993 (table 4). Residuals for the annual predictions during the calibration period varied from 5% low for 1994 to 8% high for 1992.

An attempt was made during the calibration process to fit the simulated monthly surface runoff at the upland field edge to the observed data at that same point (fig. 5). As figure 5 illustrates, there was a relatively poor fit to the monthly surface runoff observations (negative NSE_{mon}). While the trends

Table 3. Observed and simulated average annual surface runoff volume (mm) and percentage of annual rainfall total generated within each HRU for the upland, grass buffer, pine buffer, and poplar buffer HRUs for the calibration and validation periods.

	Calibration Period (Feb. 1992 to Dec. 1994)		Validation Period (Jan. 1995 to Dec. 1996)	
HRU	Avg. Annual Observed Surface Runoff, mm (%)	Avg. Annual Simulated Surface Runoff, mm (%)	Avg. Annual Observed Surface Runoff, mm (%)	Avg. Annual Simulated Surface Runoff, mm (%)
HRU1 Upland field	326 (27)	354 (29)	309 (32)	247 (26)
HRU2 Grass buffer	103 (9)	101 (8)	90 (9)	55 (6)
HRU3 Pine buffer	64 (5)	67 (6)	54 (6)	18 (2)
HRU4 Poplar buffer	no data	67 (6)	no data	15 (2)

Table 4. Observed precipitation and observed and simulated surface runoff in mm and as a percentage of annual precipitation at three positions in the riparian buffer.

Period	Precipitation, mm	Upland Field Edge		Grass Buffer Edge		Pine Buffer Edge	
		Observed, mm (%)	Simulated, mm (%)	Observed, mm (%)	Simulated, mm (%)	Observed, mm (%)	Simulated, mm (%)
Feb. 1992 to Dec. 1992	1124	203 (18)	293 (26)	121 (11)	62 (5)	22 (2)	18 (2)
Jan. 1993 to Dec. 1993	1062	206 (19)	263 (25)	75 (7)	66 (6)	66 (6)	28 (3)
Jan. 1994 to Dec. 1994	1428	570 (40)	505 (35)	112 (8)	176 (12)	105 (7)	154 (11)
Jan. 1995 to Dec. 1995	842	229 (27)	191 (23)	39 (5)	43 (5)	34 (4)	6 (1)
Jan. 1996 to Dec. 1996	1093	389 (36)	303 (28)	141 (13)	66 (6)	74 (7)	31 (3)

were in general correct, there were periods of large overprediction (February 1992, January 1993, and October 1994) as well as periods of large underprediction (June 1993 and June 1994). The model typically overpredicted winter runoff and underpredicted summer runoff at the upland field edge. The February 1992 and January 1993 periods represented winter months with relatively saturated conditions. June 1993 and June 1994 were summer months with relatively dry conditions. In contrast to the results obtained here with the landscape model, prior applications with the SWAT watershed model have shown a tendency to underpredict total watershed yield during wetter (typically winter) months and overpredict total water yield during drier months (typically late summer and early fall) (King et al., 1999; Arnold et al., 2000; Bosch et al., 2004; Van Liew et al., 2005; Feyereisen et al., 2007; Van Liew et al., 2007). The October 1994 results with the landscape model were dominated by an extreme event on October 2 when 191 mm of rainfall was measured. Prior research has indicated that the SWAT watershed model also has a tendency to overpredict seasonal events during drier periods of the year (Bosch et al., 2004), similar to what was observed here for October 2, 1994.

As indicated by Sheridan et al. (1996) observed surface runoff from extreme events was underestimated due to the type of surface runoff collectors that had to be used for the study. This could have contributed to the difficulty in obtaining good fits for the wetter months. In reducing simulated winter runoff to fit the observed data, summer simulation results were reduced as well, increasing the error for the summer periods. With the months of January 1993 and October 1994 removed, the NSE_{mon} improved considerably, although still not in a satisfactory range. The difficulty in obtaining a good monthly fit may be an indication that the landscape SWAT model will require more detail to adequately describe interactions between surface runoff, lateral subsurface flow, and groundwater. The fluctuations of the shallow groundwater in this field site have a dramatic influence on surface hydrology (Bosch et al., 2003), and these interactions will need to be more dynamic in the landscape simulations.

For the grass buffer (HRU2), exiting surface runoff was reduced by increasing the infiltration capacity of the HRU through adjustments to SOL_K. For the calibration period, the average annual surface runoff simulated for the edge of the grass buffer was 101 mm (8% of precipitation), while the average annual observed value was 103 mm (9% of precipitation) (table 3). While the simulated and observed annual averages agreed well, the NSE_{ann} for the surface runoff from the grass buffer HRU was negative, indicating a poor fit to the observed data. The negative NSE_{ann} was attributed to a high degree of variability in the observed data and an inability to adequately simulate this variability (table 4). Because of the relatively high runoff values entering the grass buffer and the low runoff values leaving the grass buffer, it is believed the runoff exiting the grass buffer is largely controlled by the runoff from the upland field. The observed annual runoff did not correlate well with annual precipitation, indicating that there were other processes controlling runoff leaving the grass buffer. Proportionally, little runoff is likely generated within the grass buffer itself. More accurate representations of the surface runoff at the edge of the grass buffer would require further modifications to the characteristics of the soil in the grass buffer as well as improved simulations of seasonal evapotranspiration. Similar to what was observed at the field edge in the monthly simulations, summer runoff was slightly underpredicted while winter runoff was overpredicted (fig. 6). Surface runoff at the edge of the grass buffer was considerably overpredicted for the month of October 1994.

Average annual surface runoff predicted at the pine buffer and poplar buffer interface averaged 67 mm (6% of precipitation), while the observed averaged 64 mm (5% of precipitation) (table 3). On a percentage of annual rainfall basis, the landscape model underestimated the surface runoff at this interface by 3% in 1993 and overestimated it by 4% in 1994 (table 4). As with the other HRUs for the monthly simulations, the surface runoff was overpredicted at the edge of the

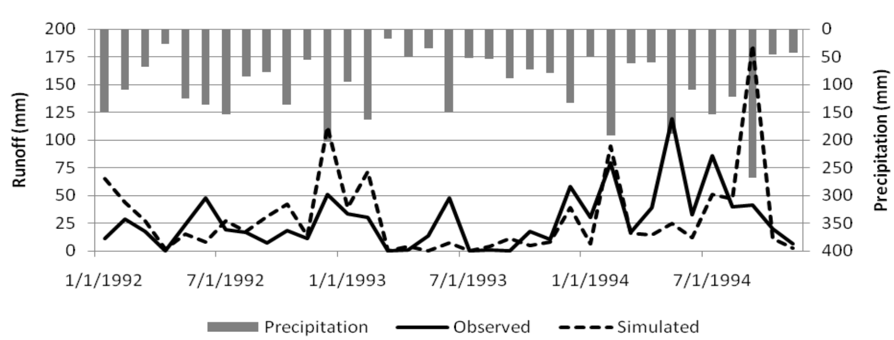

Figure 5. Observed and simulated monthly surface runoff at the upland field edge for the calibration period (Feb. 1992 through Dec. 1994).

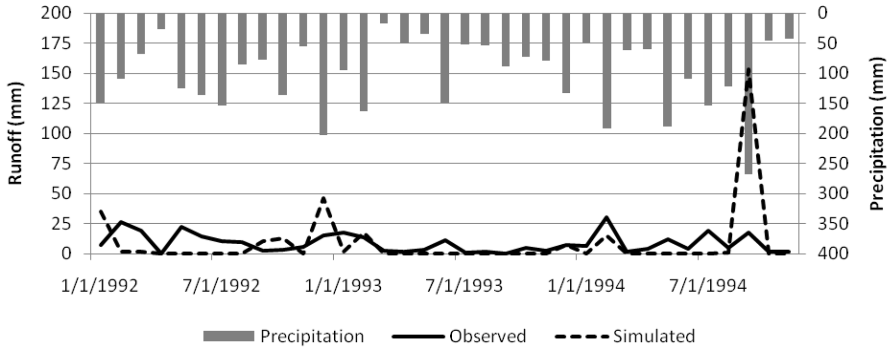

Figure 6. Observed and simulated monthly surface runoff at the downslope edge of the grass buffer for the calibration period (Feb. 1992 through Dec. 1994).

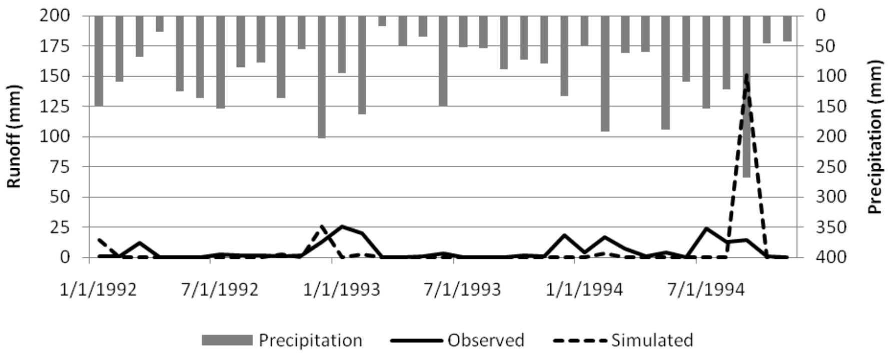

Figure 7. Observed and simulated monthly surface runoff at the downslope edge of the pine buffer for the calibration period (Feb. 1992 through Dec. 1994).

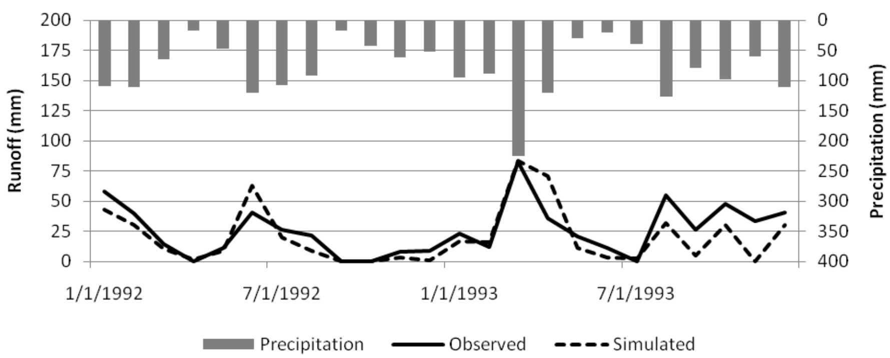

Figure 8. Observed and simulated monthly surface runoff at the downslope edge of the upland field for the validation period (Jan. 1995 through Dec. 1996).

pine buffer for the month of October 1994 (fig. 7). This may have been due to an under-representation of the actual surface runoff for the larger events, as described earlier. Other trends were not as consistent as those observed at the field or the grass buffer edges. Aside from the October 1994 event, observed and simulated runoff for the pine buffer and poplar buffer interface was relatively small (<25 mm) (fig. 7).

Validation

The validation period was from January 1995 through December 1996. For the validation period, the simulated annual runoff underestimated the observed annual runoff at the edge of the upland field by 4% in 1995 and by 8% in 1996 (table 4). At the edge of the grass buffer, the simulated annual runoff matched the observed runoff for 1995 and underestimated it by 7% in 1996 (table 4). Simulated surface runoff at the edge of the pine buffer also underestimated the observed runoff by 3% in 1995 and 4% in 1996 (table 4).

Results at the grass buffer and the downslope edge of the pine buffer were largely controlled by the results at the upland field edge. Monthly simulated surface runoff at the upland field edge tracked the observed data fairly closely (fig. 8). The improvement over the calibration period was likely due to a lack of extreme events for the validation period (fig. 8). Very little monthly runoff was observed at the edge of the grass buffer for the validation period (fig. 9). In contrast to what was observed for the calibration period, no general seasonal trend in terms of over- or under-simulation was observed. Similar monthly results were obtained during the validation period at the edge of the pine buffer (data not shown).

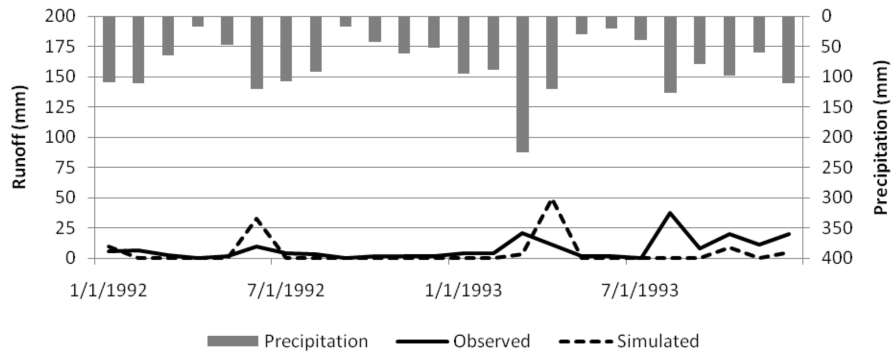

Figure 9. Observed and simulated monthly surface runoff at the downslope edge of the grass buffer for the validation period (Jan. 1995 through Dec. 1996).

Model Comparison

Total water yield simulated with SWAT 2005 for a single-unit HRU, comparable to the hillslope multiple-HRU simulation, was 34% of precipitation for the period from February 1992 to December 1996. For the multiple-HRU simulation with the landscape version of SWAT, the total yield was 44%. The expected total water yield for this landscape averages 34% of the annual precipitation from data sets on surface runoff (Sheridan et al., 1999), lateral subsurface flow (Bosch et al., 2005), and groundwater flow (Bosch et al., 2003). There were large differences in the hydrologic components of the flow (table 5). Because there is no re-distribution of the water, the single-unit HRU simulation generates considerably more surface runoff and less groundwater flow. In contrast, the SWAT landscape model allows re-distribution of the water into the subsurface, resulting in considerable groundwater flow at the bottom of the landscape (table 5).

While surface runoff data were not collected at the edge of the poplar buffer, observations of surface runoff at the edge of the pine buffer were on the order of 2% to 7% of annual rainfall (table 4). Surface runoff exiting the poplar buffer may be greater than that observed exiting the pine buffer due to increased saturation in the lower landscape positions and the proximity of the groundwater to the soil surface. Estimates of surface runoff obtained using the single-unit HRU simulated with SWAT 2005 were 9% of annual rainfall, while estimates obtained using the landscape SWAT model were 4%.

Lateral subsurface flow was not measured within the riparian buffer study area. Upland field edge measurements collected by Bosch et al. (2005) found that lateral flow from conventionally tilled fields averaged 9% of annual precipitation. Simulated estimates of lateral flow obtained with the SWAT landscape model at the upland field edge were 15% of annual precipitation, while they were 0% at the edge of the pine and poplar buffers. The SWAT 2005 single-HRU simulation yielded an estimate of 17% for lateral flow at the edge of the bottom of the landscape. At the riparian buffer site, the shallow water table and the lateral subsurface flow merge at the edge of the stream throughout wetter portions of the year, making it difficult to distinguish between lateral subsurface flow and groundwater flow at the bottom of the landscape (Bosch et al., 2003). While lateral flow at the stream edge is expected to be less than that observed at the upland field edge, there is still likely a lateral flow component during some portions of the year. Thus, simulated lateral flow obtained using the landscape SWAT model is likely too low, while that simulated with SWAT 2005 is likely too high.

Table 5. Water budget as a percentage of precipitation obtained for the poplar buffer (HRU4) using the SWAT landscape model and that obtained for the single-unit upland HRU obtained using the SWAT 2005 original configuration for the period from February 1992 to December 1996.

	Landscape Model (HRU4)	Single-Unit HRU
Surface runoff (%)	4	9
Lateral subsurface flow (%)	0	17
Groundwater flow (%)	40	8
Total water flow (%)	44	34

For the landscape SWAT model, simulated groundwater accounted for 90% of the streamflow, while for the single-unit HRU simulated with SWAT 2005, the groundwater accounted for 24% of the total streamflow (table 5). Estimates of groundwater flow for this portion of the Fox Den field obtained by Bosch et al. (2003) ranged from 7% to 32% of annual precipitation depending on the season. The simulated estimate of groundwater flow (40% of annual precipitation) obtained with the landscape SWAT model is greater than this observed range.

Simulated Buffer Outflow

While no observed data were available at the bottom of the poplar forest buffer, the edge of the stream, examinations of the simulated values are useful for evaluating stream contributions obtained using the SWAT landscape model. Summary water budget results for the entire simulation period (calibration and validation) are shown in table 6. The average annual precipitation for the period was 1110 mm. The average annual simulated total water yield on a per area basis at the bottom of the poplar buffer (HRU4) catchment was 492 mm, or 44% of the annual precipitation. Based on watershed observations, the actual average annual total water loss is believed to be around 34%.

Simulated annual total water yields from HRU4 (poplar buffer) varied from 603 mm (55% of precipitation) to 237 mm (28% of precipitation) for the entire simulation period (fig. 10). The largest portion of the simulated water budget out of the hillslope (HRU4) was from the groundwater component, which varied from 571 mm (1996) to 231 mm (1995) (fig. 10).

There is a shift in the water budget from the upland field to the stream edge. The largest contributor to flow from the upland field was surface runoff, while the largest contributor from the floodplain was groundwater (table 6). Surface runoff and lateral subsurface flow both decrease while moving

Table 6. Average annual simulated results for water flow leaving each HRU area (mm) and percentage of precipitation over the period from February 1992 to December 1996 using the SWAT landscape model.

Landscape Unit	Precipitation, mm	Surface Runoff, mm (%)	Lateral Flow Contribution, mm (%)	Groundwater Contribution, mm (%)	Total Yield, mm (%)
HRU1 Upland field	1110	311 (28)	63 (6)	14 (1)	388 (35)
HRU2 Grass buffer	1110	83 (7)	210 (19)	73 (7)	365 (33)
HRU3 Pine buffer	1110	47 (4)	39 (4)	407 (37)	493 (44)
HRU4 Poplar buffer	1110	46 (4)	3 (0)	443 (40)	492 (44)

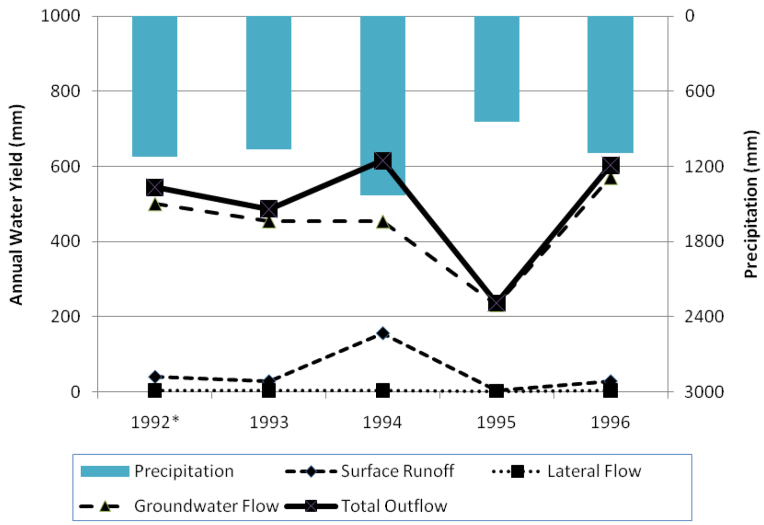

Figure 10. Simulated average annual water balance at HRU4 for the simulation period from February 1992 to December 1996.

down the landscape, while groundwater flow increases. This agrees with the findings of Sheridan et al. (1999), who reported a 40% to 70% reduction in surface runoff from the upland field edge to the edge of the pine buffer. Surface runoff within HRU4 (the lower floodplain) was reduced despite the contributions of overland flow from the upland field, indicating considerable infiltration within HRU2 and HRU3. There was a large increase in the groundwater component of the flow within the floodplain HRUs (HRU3 and HRU4) (table 6). This large increase in groundwater flow is caused by redistribution from surface runoff into groundwater. Groundwater within the floodplain includes contributions from both the upland and the hillslope units. While the rapid shift from surface and lateral flow to groundwater flow simulated within the pine buffer HRU agrees with observations from the site, simulated groundwater yields (40%) overestimate observations from the site (7% to 32%).

Total water loss simulated at the bottom of the landscape averaged 44% of precipitation (table 6). Assuming that the soil water storage changed little over the entire long-term simulation, evapotranspiration for the hillslope can be estimated as 56% of the annual precipitation. Estimates for ET in watersheds dominated by pine forests range from 60% to 80% of precipitation per year (Riekerk, 1985). Knisel et al. (1991) reported ET from an upland field in this region as 69% of precipitation for a corn/soybean rotation with a winter cover of oats. Bosch et al. (1996) reported ET for the riparian forests in this watershed as 67%. Estimates for ET losses for the corn and fallow upland fields obtained using the GLEAMS model ranged from 58% to 83% of annual precipitation per year for the observation period (Bosch et al., 1996). Thus, estimates of ET simulated with the SWAT landscape model appear to be low for this landscape. An increase in evapotranspiration would reduce the overall groundwater yield. In particular, an increase in cool season evapotranspiration would yield the reduction in winter surface runoff required without further reducing summer runoff predictions.

Summary and Conclusions

The SWAT landscape revision provided reasonable long-term estimates of surface runoff and groundwater flow for the simulated hillslope in south-central Georgia. Estimates of lateral subsurface flow at the bottom of the landscape were less than expected. The annual total water yield at the bottom landscape unit averaged 492 mm, or 44% of the annual precipitation (table 6). The actual average annual total water yield for this same landscape position was estimated to equal 34% of the annual precipitation from data sets on surface runoff (Sheridan et al., 1999), lateral subsurface flow (Bosch et al., 2005), and groundwater flow (Bosch et al., 2003). Simulated annual surface runoff at the edge of the upland field, the edge of the grass buffer, and the edge of the pine buffer followed general trends in the observed data (table 4). However, residuals between the observed and simulated annual runoff at the edge of the grass and pine buffers varied by up to 75% from the observed runoff. While annual simulations of surface runoff provided reasonable estimates of annual trends, the SWAT landscape model did not do a good job of tracking monthly observed surface runoff at the upland field edge (fig. 5). This could possibly be due to an underestimation of extreme events in the field observations (Sheridan et al., 1999) or a poor tracking of seasonal variations in water table elevation, evapotranspiration, or soil water storage.

Estimates of groundwater flow at the edge of the poplar buffer (table 6) were greater than what would be expected for this site. Field estimates of groundwater flow from the site indicate that groundwater contributions range from 7% to 32% of annual precipitation (Bosch et al., 2003). Lower groundwater yields could be obtained by increasing evapotranspiration rates in the forested buffer. In particular, higher wintertime evapotranspiration would reduce the high runoff rates predicted for the winter period.

The SWAT landscape model simulates redistribution of water calculated by landscape position, a feature not available within SWAT 2005. Without the landscape configuration, the SWAT model does not simulate the large groundwater component observed at this site. In many cases, such as the one examined here, the groundwater component can be a dominant part of the hydrologic budget and play an increasingly large role as the water flows down the landscape. The SWAT landscape model is able to adequately redistribute the water. However, the landscape model may require additional detail to properly describe interactions between the soil surface, the vadose zone, and groundwater. In order to reduce the simulated total water loss and accurately simulate the seasonal effects observed at this site, the SWAT landscape model may require a more detailed simulation of interactions between the vadose zone and shallow groundwater. In particular, an accurate representation of water storage within the vadose zone and the surficial aquifer appears to be critical to accurately representing the hydrology in these landscapes where subsurface processes dominate. While calibration of SWAT 2005 could yield more appropriate redistribution, it could not represent the higher fraction of surface runoff observed in one landscape position while allowing infiltration and redistribution of this water into groundwater at lower landscape positions, as the landscape model is capable of doing. This would be expected to have dramatic implications on water quality.

While additional calibration and testing of the SWAT landscape model are necessary, the results are encouraging. The modifications will allow the model to more realistically represent actual landscape flow and transport processes. The movement of water between the soil surface, vadose zone, and shallow aquifers is represented by the SWAT landscape model. This framework will provide a more realistic basis for water quality simulation. This feature is particularly important for landscapes, such as the one examined here, where significant buffering takes place between upland fields and nearby streams. As testing of the model is expanded, the full utility of the model will be realized.

References

Arnold, J. G., and N. Fohrer. 2005. SWAT2000: Current capabilities and research opportunities in applied watershed modelling. *Hydrol. Proc.* 19(3): 563-572.

Arnold, J. G., R. Srinivasan, R. S. Muttiah, and J. R. Williams. 1998. Large-area hydrologic modelling and assessment: Part I. Model development. *J. American Water Resources Assoc.* 34(1): 73-89.

Arnold, J. G., R. S. Muttiah, R. Srinivasan, and P. M. Allen. 2000. Regional estimation of base flow and groundwater recharge in the Upper Mississippi river basin. *J. Hydrol.* 227: 21-40.

Arnold, J. G., K. N. Potter, K. W. King, and P. M. Allen. 2005. Estimation of soil cracking and the effect on surface runoff in a Texas Blackland Prairie watershed. *Hydrol. Proc.* 19(3): 589-603.

Arnold, J. G., P. M. Allen, M. Volk, J. R. Williams, and D. D. Bosch. 2010. Assessment of different representations of spatial variability on SWAT model performance. *Trans. ASABE* 53(5): 1433–1443.

Arora, K., S. K. Mickelson, J. L. Baker, D. P. Tierney, and C. J. Peters. 1996. Herbicide retention by vegetative buffer strips from runoff under natural rainfall. *Trans. ASAE* 39(6): 2155-2162.

Asmussen, L. E. 1971. Hydrologic effects of quaternary sediments above the marine terraces in the Georgia Coastal Plain. *Southeastern Geology* 12(3): 189-201.

Asmussen, L. E., J. M. Sheridan, and C. V. Booram. 1979. Nutrient movement in streamflow from agricultural watersheds in the Georgia Coastal Plain. *Trans. ASAE* 22(4): 809-815, 821.

Barfield, B. J., E. W. Tollner, and J. C. Hayes. 1979. Filtration of sediment by simulated vegetation: I. Steady-state flow and homogeneous sediment. *Trans. ASAE* 22(5): 540-545, 548.

Bingner, R. L. 1996. Runoff simulated from Goodwin Creek watershed using SWAT. *Trans. ASAE* 39(1): 85-90.

Bosch, D. D., R. K. Hubbard, L. T. West, and R. R. Lowrance. 1994. Subsurface flow patterns in a riparian buffer system. *Trans. ASAE* 37(6): 1783-1790.

Bosch, D. D., J. M. Sheridan, and R. R. Lowrance. 1996. Hydraulic gradients and flow rates of a shallow coastal plain aquifer in a forested riparian buffer. *Trans. ASAE* 39(3): 865-871.

Bosch, D. D., R. R. Lowrance, J. M. Sheridan, and R. G. Williams. 2003. Groundwater storage effect on streamflow for a southeastern Coastal Plain watershed. *Ground Water* 41(7): 903-912.

Bosch, D. D., J. M. Sheridan, H. L. Batten, and J. G. Arnold. 2004. Evaluation of the SWAT model on a Coastal Plain agricultural watershed. *Trans. ASAE* 47(5): 1493-1506.

Bosch, D. D., T. L. Potter, C. C. Truman, C. Bednarz, and T. C. Strickland. 2005. Surface runoff and lateral subsurface flow as a response to conservation tillage and soil-water conditions. *Trans. ASAE* 48(6): 2137-2144.

Brutsaert, W. 2005. *Hydrology: An Introduction.* Cambridge, U.K.: Cambridge University Press.

Chanasyk, D. S., E. Mapfumo, and W. Williams. 2003. Quantification and simulation of surface runoff from fescue grassland watershed. *Agric. Water Mgmt.* 59(2): 137-153.

Dillaha, T. A., J. H. Renueau, S. Mostaghimi, and D. Lee. 1989. Vegetative filter strips for agricultural nonpoint-source pollution control. *Trans. ASAE* 32(2): 513-519.

Feyereisen, G. W., T. C. Strickland, D. D. Bosch, and D. G. Sullivan. 2007. Evaluation of SWAT manual calibration and input parameter sensitivity in the Little River watershed. *Trans. ASABE* 50(3): 843-855.

Gassman, P. W., M. R. Reyes, C. H. Green, and J. G. Arnold. 2007. The soil and water assessment tool: Historical development, applications, and future research directions. *Trans. ASABE* 50(4): 1211-1250.

Hubbard, R. K., and R. Lowrance. 1997. Assessment of forest management effects on nitrate removal by riparian buffer systems. *Trans. ASAE* 40(2): 383-391.

Hubbard, R. K., and J. M. Sheridan. 1983. Water and nitrate-nitrogen losses from a small, upland, coastal plain watershed. *J. Environ. Qual.* 12(2): 291-295.

Inamdar, S. P., J. M. Sheridan, D. D. Bosch, R. G. Williams, R. R. Lowrance, L. S. Altier, and D. L. Thomas. 1999a. Riparian ecosystem management model (REMM): I. Hydrology evaluation. *Trans. ASAE* 42(6): 1679-1689.

Inamdar, S. P., R. R. Lowrance, L. S. Altier, R. G. Williams, and R. K. Hubbard. 1999b. Riparian ecosystem management model (REMM): II. Testing of the water quality and nutrient cycling component for a coastal plain riparian system. *Trans. ASAE* 42(6): 1691-1707.

Jacobs, T. C., and J. W. Gilliam. 1985. Riparian losses of nitrate from agricultural drainage waters. *J. Environ. Qual.* 14(4): 472-478.

King, K. W., J. G. Arnold, and R. L. Bingner. 1999. Comparison of Green-Ampt and curve number methods on Goodwin Creek watershed using SWAT. *Trans. ASAE* 42(4): 919-925.

Kirkby, M. J., R. Abrahart, M. D. McMahon, J. Shao, and J. B. Thornes. 1998. MEDALUS soil erosion models for global change. *Geomorphology* 24(1): 35-49.

Kirsch, K., A. Kirsch, and J. G. Arnold. 2002. Predicting sediment and phosphorus loads in the Rock River basin using SWAT. *Trans. ASAE* 45(6): 1757-1769.

Knisel, W. G., R. A. Leonard, F. M. Davis, and J. M. Sheridan. 1991. Water balance components in the Georgia Coastal Plain: A GLEAMS model validation and simulation. *J. Soil and Water Cons.* 46(6): 450-456.

Lane, L. J., and M. A. Nearing, eds. 1989. USDA-Water erosion prediction project: Hillslope profile model documentation. NSERL Report No. 2. W. Lafayette, Ind.: USDA-ARS National Soil Erosion Research Laboratory.

Lowrance, R. R., R. L. Todd, and L. E. Asmussen. 1983. Waterborne nutrient budgets for the riparian zone of an agricultural watershed. *Agric., Ecosystems and Environ.* 10(4): 371-384.

Lowrance, R. R., R. L. Todd, J. Fail, O. Henderson Jr., R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. *Bioscience* 34(6): 374-377.

Lowrance, R., J. K. Sharpe, and J. M. Sheridan. 1986. Long-term sediment deposition in the riparian zone of a coastal plain watershed. *J. Soil and Water Cons.* 42(4): 266-271.

Lowrance, R., J. M. Sheridan, R. G. Williams, D. D. Bosch, D. G. Sullivan, D. R. Blanchett, L. M. Hargett, and C. M. Clegg. 2007. Water quality and hydrology in farm-scale Coastal Plain watersheds: Effects of agriculture, impoundments, and riparian zones. *J. Soil and Water Cons.* 62(2): 65-76.

Magette, W. L., R. B. Brinsfield, R. E. Palmer, and J. D. Wood. 1989. Nutrient and sediment removal by vegetated filter strips. *Trans. ASAE* 32(2): 663-667.

Mausbach, M. J., and A. R. Dedrick. 2004. The length we go: Measuring environmental benefits of conservation practices. *J. Soil and Water Cons.* 59(5): 97A-103A.

Moriasi, D. N., J. G. Arnold, M. W. Van Liew, R. L. Bingner, R. D. Harmel, and T. L. Veith. 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Trans. ASABE* 50(3): 885-900.

Nash, J. E., and J. V. Sutcliffe. 1970. River flow forecasting through conceptual models: Part 1. A discussion of principles. *J. Hydrol.* 10(3): 282-292.

Peterjohn, W. T., and D. L. Correll. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology* 65(6): 1466-1475.

Peterson, J. R., and J. M. Hamlett. 1998. Hydrologic calibration of the SWAT model in a watershed containing fragipan soils and wetlands. *J. American Water Res. Assoc.* 34(3): 531-544.

Riekerk, H. 1985. Lysimetric measurement of pine evapotranspiration for water balances. In *Advances in Evapotranspiration*, 276-281. St. Joseph, Mich.: ASAE.

Saleh, A., J. G. Arnold, P. W. Gassman, L. M. Hauck, W. D. Rosenthal, J. R. Williams, and A. M. S. McFarland. 2000. Application of SWAT for the upper North Bosque River watershed. *Trans. ASAE* 43(5): 1077-1087.

Santhi, C., J. G. Arnold, J. R. Williams, L. M. Hauck, and W. A. Dugas. 2001. Application of a watershed model to evaluate management effects on point and nonpoint source pollution. *Trans. ASAE* 44(6): 1559-1570.

Sheridan, J. M. 1997. Rainfall-streamflow relations for Coastal Plain watersheds. *Applied Eng. in Agric.* 13(3): 333-113.

Sheridan, J. M., C. V. Booram, and L. E. Asmussen. 1982. Sediment-delivery ratios for a small Coastal Plain watershed. *Trans. ASAE* 25(3): 610-615, 622.

Sheridan, J. M., R. R. Lowrance, and H. H. Henry. 1996. Surface flow sampler for riparian studies. *Applied Eng. in Agric.* 12(2): 183-188.

Sheridan, J. M., R. Lowrance, and D. D. Bosch. 1999. Management effects on runoff and sediment transport in riparian forest buffers. *Trans. ASAE* 42(1): 55-64.

Smithers, J. C., and B. A. Engel. 1996. An initial assessment of SWAT as a hydrological modeling tool for the Midwest USA. ASAE Paper No. 962065. St. Joseph, Mich.: ASAE.

Spruill, C. A., S. R. Workman, and J. L. Taraba. 2000. Simulation of daily and monthly stream discharge from small watersheds using the SWAT model. *Trans. ASAE* 43(6): 1431-1439.

Srinivasan, R., T. S. Ramanarayanan, R. Jayakrishnan, and H. Wang. 1997. Hydrologic modeling of Rio Grande/Rio Bravo basin. ASAE Paper No. 972236. St. Joseph, Mich.: ASAE.

Stringfield, V. T. 1966. Artesian water in Tertiary limestone in the southeastern states. USGS Professional Paper No. 517. Reston, Va.: U.S. Geological Survey.

Van Liew, M. W., J. G. Arnold, and D. D. Bosch. 2005. Problems and potential of autocalibrating a hydrologic model. *Trans. ASAE* 48(3): 1025-1040.

Van Liew, M. W., T. L. Veith, D. D. Bosch, and J. G. Arnold. 2007. Suitability of SWAT for the Conservation Effects Assessment Project: Comparison on USDA Agricultural Research Service watersheds. *J. Hydrol. Eng.* 12(2): 173-189.

Yates, P., and J. M. Sheridan. 1983. Estimating the effectiveness of vegetated floodplains/wetlands as nitrate-nitrite and orthophosphorus filters. *Agric., Ecosystems and Environ.* 9(3): 303-314.