Fecal Bacteria Source Characterization and Sensitivity Analysis of SWAT 2005

P. B. Parajuli, K. R. Douglas-Mankin, P. L. Barnes, C. G. Rossi

Abstract. *The Soil and Water Assessment Tool (SWAT) version 2005 includes a microbial sub-model to simulate fecal bacteria transport at the watershed scale. The objectives of this study were to demonstrate methods to characterize fecal coliform bacteria (FCB) source loads and to assess the model sensitivity to five user-defined model parameters (BACTKDQ: bacteria soil partition coefficient in surface runoff, TBACT: temperature adjustment factor, WDLPQ: less-persistent bacteria die-off in solution phase, WDLPS: less-persistent bacteria die-off in sorbed phase, and BACTKKDB: bacteria partition coefficient in manure) and one input parameter (BACTLPDB: FCB concentration in manure). Fecal bacterial source loads were described and applied spatially for confined livestock, seasonal grazing livestock, failing human septic systems, and indigenous large mammal, small mammal, and avian wildlife. The relative sensitivity index (S) was tested using the independent parameter perturbation (IPP) method. Validation results for an uncalibrated SWAT model using nine runoff events from Rock Creek watershed (77 km²) were considered adequate to proceed with sensitivity analyses. Flow simulation resulted in good coefficient of determination (R^2) of 0.67 and Nash-Sutcliffe Efficiency Index (E) of 0.55, and FCB source load characterization methods were sufficiently precise to result in fair correlation (R^2 = 0.29) and reasonable measured vs. predicted response slope (0.69). Within the ranges recommended for use in SWAT, BACTKDQ had moderate sensitivity (S < 2.67) within -99.5% from 175 (baseline value), BACTLPDB had low sensitivity (S < 0.25) within -90% from 3.29 × 10⁷ cfu 100 mL⁻¹, BACTKKDB had low sensitivity (S < 0.12) within -89% from 0.9, TBACT had low sensitivity (S < 0.36) ±20% from 1.07, WDLPQ had low sensitivity (S < 0.25) ±50% from 0.23, and WDLPS had no sensitivity (S < 0.06) ±50% from 0.023 when compared with all surface runoff events. This study recommends that SWAT could adopt default values of 0.23 for WDLPQ and 0.023 for WDLPS without adversely affecting results. Moderate sensitivity for BACTKDQ indicates that users should select these with caution considering locally relevant data. The sensitivity of BACTKDQ was found high when compared with nine measured surface runoff events.*

Keywords. *Fecal coliform bacteria, Sensitivity analysis, Water quality, Watershed modeling.*

Surface-water contamination from fecal bacteria is a major health issue in the U.S. (Benham et al., 2006). Fecal bacteria often are present in surface water at concentrations that have the potential to cause severe illnesses in humans (Craun and Frost, 2002). Nonpoint sources of fecal bacteria include land application of manures, grazing operations, winter feeding operations, failing septic systems, and wildlife (Zeckoski et al., 2005). Runoff following heavy storms increases the chance of bacteria from these sources reaching surface water systems and infecting humans (Curriero, 2001). Assessment of sources and transport mechanisms can help target source-control and management efforts; hydrologic or water-quality models provide an alternative to field monitoring that can save time, reduce costs, and maximize the impact of these efforts (Benham et al., 2006; Shirmohammadi et al., 2006). Such models can be used to assess water quality goals, even on large watersheds; however, the sensitivity of mathematical model simulation results to the input parameters is a concern because that sensitivity influences the model results.

The U.S. Environmental Protection Agency (EPA) has increasingly emphasized the importance of incorporating assessment of variability and uncertainty into the modeling process (USEPA, 1997). In watershed modeling, uncertainty can be associated with natural processes, models, or model parameters, and these uncertainties include effects of monitoring/measurement error, model error, model input parameter error, spatial variability, error in spatial data layers within a geographic information system (GIS), aggregation of spatial data, and temporal variability (Haan, 1989). For watershed-scale fecal bacteria modeling, many of these uncertainties are relevant in the selection of model parameters and model input values. The relationship of these uncertainties to model response must be understood for effective modeling.

Sensitivity analysis is used to describe the responsiveness of model outputs to model inputs. The most common form of sensitivity analysis is independent parameter perturbations

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(IPP), in which parameters vary individually by a fixed percentage around a base value (Ferreira et al., 1995). Model output responses to parameter perturbation can be quantified by percentage change of selected output variables and relative change of output versus input (Larocque and Banton, 1994). The overall model response can be obtained by measuring the average response of selected output variables (Nearing et al., 1989).

SWAT Model

This study applied the Soil and Water Assessment Tool (SWAT), a widely used, watershed-scale, process-based model developed by the USDA Agricultural Research Service (ARS) (Arnold et al., 1998; Neitsch et al., 2002, 2005; Gassman et al., 2007). The SWAT model simulates hydrological, sediment, nutrient, pesticide, and bacterial processes on a continuous, daily time step. Bacteria transport routines were added to the SWAT model in 2000 (Neitsch et al., 2002). In 2005, bacteria routines were improved (Sadeghi and Arnold, 2002) and the SWAT model was modified (Neitsch et al., 2005), which allowed it to be used as a tool for addressing microbial contamination of water caused by point and nonpoint sources.

The microbial component of SWAT simulates the fate and transport of bacterial organisms. This microbial sub-model uses the first-order decay equation as applied by Moore et al. (1989) to model fecal bacteria die-off and re-growth (eq. 1):

$$C_t = C_o\, e^{-K_{20}\, \theta^{(T-20)}\, t} \qquad (1)$$

where

C_t = bacteria concentration at time t (cfu 100 mL^{-1})

C_o = initial bacteria concentration (cfu 100 mL^{-1})

K_{20} = first-order die-off rate at 20°C (day^{-1})

θ = temperature adjustment factor (TBACT in SWAT)

T = temperature (°C)

t = exposure time (days).

The model partitions bacteria in manure into sorbed and solution phases using a bacteria partition coefficient (BACTKDDB). The BACTKDDB coefficient ranges from 0 (all bacteria sorbed to soil particles) to 1 (all bacteria in solution phase). For this study, baseline BACTKDDB was assumed to be 0.9 (Soupir et al., 2006). A bacteria soil partition coefficient (BACTKDQ), the ratio of bacteria concentrations in surface soil to surface runoff, serves to limit interaction of surface runoff with bacteria in the surface 10 mm of soil. As BACTKDQ increases, the bacteria concentration in surface runoff decreases for the same surface-soil bacteria concentration. BACTKDQ may vary from 0 to 500, with a default value of 175 in SWAT (Neitsch et al., 2005).

The temperature adjustment factor (TBACT) in the first-order bacteria decay rate function in equation 1 may range from 0.80 to 1.20 (Moore et al., 1989; Walker et al., 1990). A more typical range for biological reactions is 1.07 ± 0.05 (Reddy et al., 1981). The bacteria die-off factors in soil solution (WDLPQ) and sorbed to soil particles (WDLPS) are important in determining the net die-off rate in equation 1. The WDLPQ parameter may range from 0.23 to 0.693 (McFeters and Stuart, 1972; Baffaut and Benson, 2003), which correspond to half-lives of 3 to 1 days, respectively. The WDLPS has been assumed to be one-tenth of WDLPQ (Baffaut and Benson, 2003), which corresponds to a range of 0.023 to 0.069. The 3-day average bacteria half-life generally represents bacteria decay rate in solution for livestock and poultry wastes (Crane et al., 1980; Reddy et al., 1981). Bacteria concentration in manure (BACTLPDB) may range from 1.2×10^6 to 6.5×10^7 cfu $g_{dry\ manure}^{-1}$ based on the ± 1 standard deviation range for fecal coliform bacteria and the daily mean total solids production cited for beef livestock manure (*ASAE Standards*, 2003).

The SWAT water quality model has been applied and validated for runoff, sediment yield, and nutrient losses from watersheds at different geographic locations, and under different conditions and management practices (Saleh et al., 1999; Spruill et al., 2000; Santhi et al., 2001; Kirsch et al., 2002; White and Chaubey, 2005; Jha et al., 2007; Gassman et al., 2007). However, only limited research has been performed for the SWAT bacteria model in predicting bacteria movement. For example, Baffaut and Benson (2003) studied bacteria TMDLs (Total Maximum Daily Loads) for the Shoal Creek watershed in southwest Missouri using SWAT 2000. They calibrated the model using daily flow, weekly fecal coliform bacteria (FCB) concentration collected from water quality grab samples, and annual hay yield reported to the USDA. A frequency curve analysis method was used to compare measured vs. predicted data for daily flow and FCB concentration. The daily flow curve was reported to be reasonable except for overpredictions of peak flow. Then, the SWAT model predicted values were compared with a frequency distribution of 18 months of weekly measured FCB concentration data using average ± 1 standard deviation of measured means for 70% time of the frequency curve. However, direct daily or weekly comparison of measured versus predicted values was not determined.

Several authors have previously completed sensitivity and output-uncertainty analyses for the SWAT model (Lenhart et al., 2002; Eckhardt et al., 2002; Sohrabi et al., 2002; Benaman and Shoemaker, 2004; Huisman et al., 2004; Feyereisen et al., 2005). Parajuli et al. (2006) assessed sensitivity of the SWAT microbial sub-model, but for the previous version (SWAT 2000). They reported low ($0.10 < |S| \le 0.50$) to high ($|S| > 2.00$) relative sensitivity for TBACT; low relative sensitivity for BACTKDQ; no ($0 < |S| \le 0.10$) relative sensitivity for manure production rate, livestock stocking rate, and land application method of septic effluent; moderate ($0.50 < |S| \le 2.00$) relative sensitivity for the point-load application method of septic effluent; moderate relative sensitivity for applying wildlife bacteria source loads in the cropland, woodland, and cropland and woodland; and high relative sensitivity for bacteria concentration in livestock manure using SWAT 2000. Sensitivity analysis of SWAT 2005 focusing on the bacteria transport sub-model is needed to allow it to be used and parameterized appropriately, yet such analysis has not been assessed.

Objectives

The objectives of this study were to (1) demonstrate methods to characterize bacteria source loads and (2) assess the sensitivity of the SWAT 2005 version (modified 12 March 2009) to user-defined parameters of the bacterial sub-model.

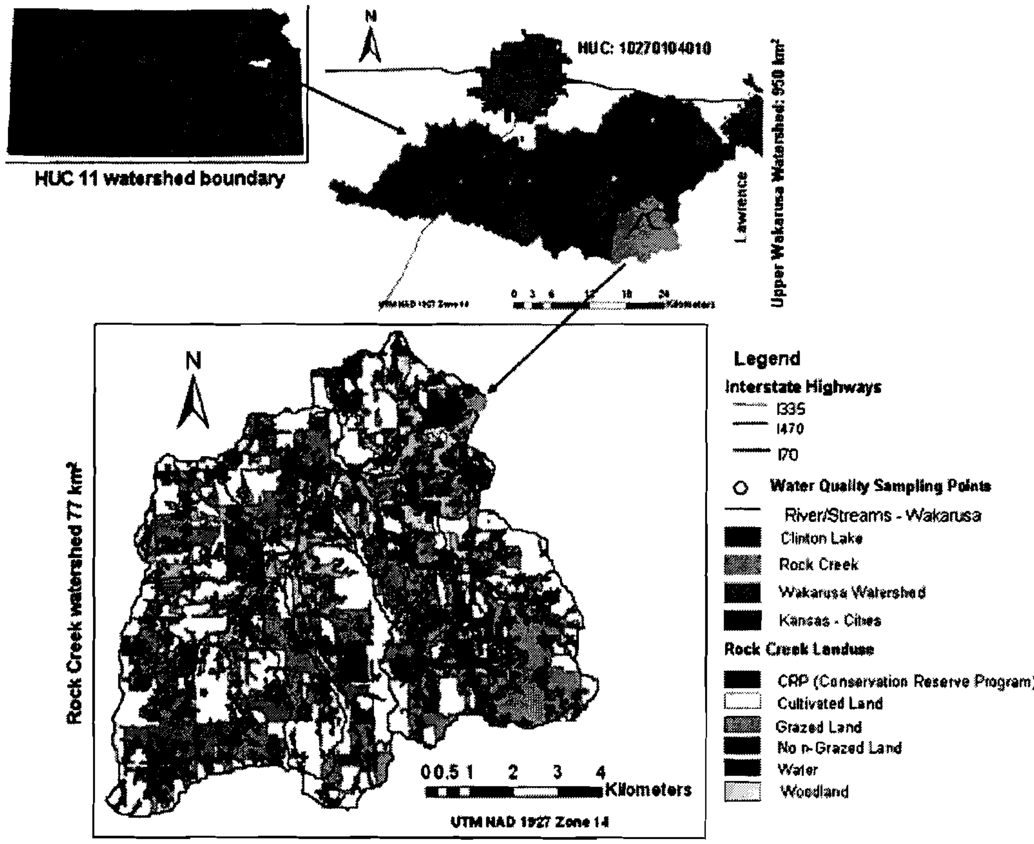

Figure 1. Location map of the Rock Creek watershed in northeast Kansas.

Materials and Methods

Watershed Setting

The study focused on the 77 km^2 Rock Creek watershed (fig. 1) in Douglas County, Kansas. The study area land uses were primarily grassland (52%), cropland (33%), woodland (14%), and other (water, urban; 1%), with grasslands categorized as a mixed-species native prairie, smooth bromegrass (*Bromus inermis*), or tall fescue (*Festuca arundinacea*). Soils were predominately silty-clay in texture (SSURGO: KS0457302, KS0457325, KS0458962). Average slope in the watershed sub-basins ranged from 3.8% to 6.3%.

Daily precipitation data for the watershed were taken from the Overbrook weather station located about 4.8 km south of the watershed. The 2004 annual rainfall for Overbrook was about 1,126 mm (fig. 2). Data from the Silver Lake weather station, which is located about 22.5 km south from the nearest

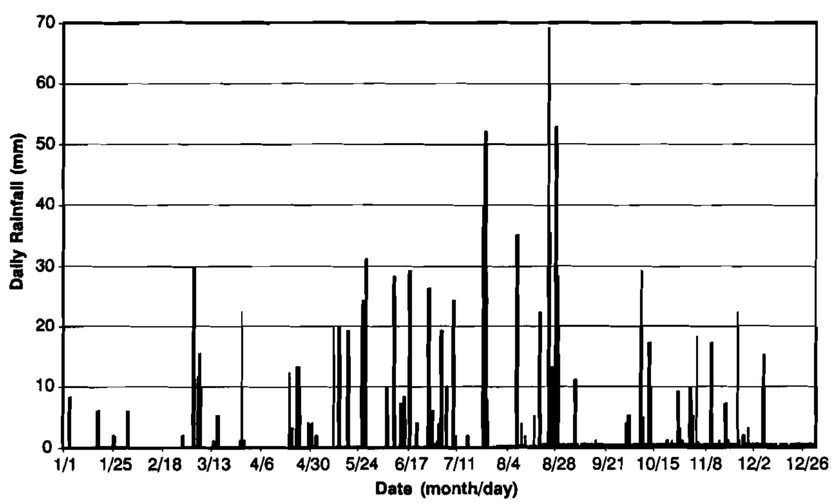

Figure 2. Distribution of daily rainfall data for the Overbrook weather station, 2004.

point of the watershed, were used for the daily temperature, daily solar radiation, daily wind speed, and daily relative humidity input. Any missing watershed data were adjusted using the SWAT weather generator, which uses data from the Ottawa weather station (Franklin County) located about 23 km southeast from the nearest point of the Rock Creek watershed. Precipitation in this region of Kansas does not exhibit notable local discontinuities; distance is the major consideration in extrapolating precipitation and other climate data from nearby recording stations.

Stream Sampling

Stream samples were collected at the Rock Creek watershed outlet throughout 2004, weekly during the growing season (April to September) and monthly during the dormant season (October to March). Only data collected during flow events were used in this study. Grab samples were collected from the midpoint of the flowing stream at the watershed outlet, placed immediately into an ice chest, and transferred to a laboratory refrigerator within 2 to 4 h. Flow at the time of sample collection was calculated using Manning's equation, as outlined by Ward and Elliot (1995). Flow depth, cross-sectional area, and channel slope were measured, and the channel roughness factor was estimated based on channel roughness characteristics and degree of meandering (Cowan, 1956). The calculated flow was validated using data from a nearby USGS streamgauge station (USGS 06891260 Wakarusa River near Richland, Kansas), which also drained to Clinton Lake, based on an area ratio of the Rock Creek watershed to the USGS station watershed. Ultimately, the calculated flow data showed very good correlation (>90%) with the weighted area flow data.

Bacteria enumeration procedures were started within 24 h of sampling. A serial dilution method was applied to enumerate FCB colonies, and samples typically required four serial dilutions to obtain reasonable colony counts. The membrane-filtration method was used for bacterial enumeration of fecal coliforms (APHA, 1998). Serial 1:10 dilutions were made in physiological saline solution. A 20 to 30 mL volume of sample and rinse water was filtered through a 0.45 μm gridded sterile membrane, the membrane was placed into mFC media (Difco Laboratories, Detroit, Mich.), and mFC plates were incubated at 44.5°C ±0.2°C for 24 h. Typically, the bacterial counts on each plate were within the recommended counting range (20 to 60 colony forming units, or cfu). If they were not, then the number of cfu was estimated by the method recommended in the Standard Methods (APHA, 1998). The detection limit for these methods was approximately 10 cfu 100 mL^{-1}.

SWAT Model Inputs

The SWAT 2005 model (revised 12 March 2009) was used in this study. The SWAT model uses geospatially referenced data to satisfy the necessary input parameters. In this study, U.S. Geological Survey 7.5-minute digital elevation data (USGS, 1999) were used to delineate the watershed boundaries and topography, the Soil Survey Geographic Database (SSURGO) was used to create a soil database (USDA-NRCS, 2005), and 2001 data from the Kansas Gap Analysis Project (GAP) that depicts 20 general land-cover classes (KARS, 2001) were used to define land cover. Wardlow and Egbert (2003) evaluated 1992 Kansas land use data from GAP and National Land Cover Data (NLCD). They found that GAP provided better discrimination of most land-cover classes than did NLCD. Specifically, their assessment found an overall accuracy of 87% for GAP and 81% for NLCD, and GAP had higher accuracy for most individual land-cover classes. In addition, while the GAP and NLCD products were compared in terms of characterizing broad-scale land-cover patterns, the Kansas GAP land-cover map appeared to be more appropriate for localized applications that require detailed and accurate land-cover information.

The land use categories were re-classified into eight classes (cropland, grazed grassland, non-grazed grassland/hay, Conservation Reserve Program (CRP) grassland, woodland, water, urban areas, and quarry) based on field-verified land use conditions (Mankin and Koelliker, 2001; Mankin et al., 2003). Parameters for each hydrologic response unit (HRU) in each watershed were defined based on soil, land use, and topographic characteristics of the watershed, as described in the SWAT 2005 documentation (Neitsch et al., 2005). The SWAT model identified eleven sub-basins and 539 HRUs (27 to 78 HRUs per sub-basin) ranging in size from 0.10 ha to 150 ha.

Model and Input Parameters Tested

Five key model parameters were tested for sensitivity: BACTKDQ, TBACT, WDLPQ, WDLPS, and BACTKDDB. One input parameter, BACTLPDB, was tested. Parajuli et al. (2006) found moderate to low relative sensitivity for other input parameters, such as manure production, stocking rate, land application and direct input of septic effluent, and wildlife source loads applied into cropland/woodland. Our preliminary analysis confirmed low sensitivity for these parameters in SWAT 2005, so these parameters were excluded from further study. Baseline values for each parameter were set based on the default used by SWAT or by an alternative value from the literature: BACTKDQ (175; Neitsch et al., 2005), TBACT (1.07; Reddy et al., 1981), WDLPQ (0.23; Baffaut and Benson, 2003), WDLPS (0.023; Baffaut and Benson, 2003), BACTKDDB (0.9; Soupir et al., 2006), and BACTLPDB (3.29×10^7 cfu $g_{dry\ manure}^{-1}$; ASAE, 2003).

Fecal Bacterial Source Characterization

Livestock

Manure applied from grazing, feeding operations, and winter feeding areas were major bacterial sources in this study. The total number of livestock in the study watershed included both confined (feedlots) and unconfined (pastures) fractions. The number of animal units (AUs) in feedlots within the watershed were estimated to be 130 AUs using active feedlot data (both federally permitted feedlots >1000 AUs and state registered feedlots >300 AUs) from the Kansas Department of Health and Environment (KDHE) (M. Jepson, 2005, personal communication). The number of livestock in pastures was estimated based on a stocking rate of 3.04 ha per cow-calf pair (0.545 AU), based on the bluestem pasture guidelines for grazing (KDA, 2004a) applied uniformly across all pastureland (32 km^2), resulting in 573 AUs. The stocking rate was confirmed using independent livestock population data. The total livestock census population of 6,158 cow-calf pair in 358 km^2 pastureland for Douglas County, Kansas (USDA-NASS, 2006) was confirmed with

Kansas Department of Agriculture Farm Facts data (KDA, 2004b) and scaled to the study watershed according to the proportion of the county land area in the study watershed, resulting in 552 total AUs. This is consistent (<4% difference) with the total grazing livestock (573 AUs) used in this study.

Livestock management in the Rock Creek watershed progressed through two distinct periods (W. Boyer, 2005, personal communication). During the 153-day grazing season, 573 beef AUs (AU = 1000 kg) were estimated to be in pastures (based on stocking rate). During the 212-day non-grazing season, the watershed held 130 beef AUs in feedlots and 229 beef AUs in winter feeding areas (40% of 573), with the remaining livestock being exported from the watershed. Manure production (58 kg day^{-1} AU^{-1}) and excreted FCB load (1.3×10^{11} cfu day^{-1} AU^{-1}) for each beef animal were estimated based on standard production rates (*ASAE Standards*, 2003). Bacteria concentration was converted into model-input units of cfu per gram of dry-weight manure using standard mean manure moisture content (85.3% moisture by weight; *ASAE Standards*, 2003).

Human

Septic systems typically fail by one of two mechanisms: (1) excessive soil conductivity in the soil absorption lateral field, which can lead to groundwater contamination; or (2) insufficient soil conductivity in the soil absorption field, which can lead to effluent surfacing. Soil types in the Rock Creek watershed commonly lead to failure by the second mode. Surfacing of effluent is observed in the field by greener vegetation (often grass lawn) occurring in the lateral field area. Generally, transport of contaminants from septic system failure is by runoff-related processes. Although there is no direct method to input septic system derived pollutants in the SWAT model, estimated septic system effluent have been applied as a fertilizer input in the SWAT model (Pradhan et al., 2004).

Digital orthophoto quarter quadrangles (DOQQ) of the watershed from 2002 (KGS, 2002) were digitized to identify rural houses likely to have septic systems. The physical context, including proximity to municipal treatment plants, type of roads, and type of houses, typically allowed unambiguous assignment of houses with septic systems. Each rural house was assumed to have one septic system, totaling 107 septic systems in the watershed, with about 20% of the estimated septic systems (22 septic systems) assumed to be failing (W. Boyer, 2006, personal communication; KDHE, 2000). Each septic system was assumed to be used by three persons in the household who contribute about 0.32 m^3 of sewage effluent per day (USEPA, 2001). Each failing septic system was modeled by assuming land application (4.7 kg^{-1} ha^{-1} day^{-1}) of all wastewater to non-grazed grassland (HRU of 19.5 ha) in the subwatershed, assuming that the land-applied "fertilizer" had FCB concentration of 6.3×10^6 cfu 100 mL^{-1} (Overcash and Davidson, 1980).

Wildlife

No comprehensive wildlife inventory was available for the Rock Creek watershed. Therefore, the wildlife population density was estimated based on information received from the Kansas Department of Wildlife and Parks (KDWP). The 2002 summer road kill indices survey data (M. Peek, 2005, personal communication) for Kansas were used to estimate small-mammal populations in the watershed: raccoon, opossum, striped skunk, coyote, badger, bobcat, red fox, gray fox, swift fox, beaver, mink, muskrat, river otter, spotted skunk, weasel, armadillo, woodchuck, and porcupine. Cumulatively, raccoon, opossum, striped skunk, and coyote populations constituted about 81% of the total small mammals in Kansas. Population of the predominant large mammal (white-tailed deer) in the watershed was based on expert opinion from the KDWP big-game coordinator (F. Lloyd, 2008, personal communication). Similarly, data were collected for the predominant indigenous avian species (turkey) from the KDWP small-game coordinator (J. Pitman, 2006, personal communication). No data were available for populations of migratory avian species (e.g., ducks, geese). Since these species are transient to the watershed and appear only during periods of low rainfall and runoff (November to March migratory periods), they were not included in this study.

To estimate the AUs of each wildlife species in the watershed, the population data were first distributed over the potential habitat for each species. Population data for small mammals and turkey were counted from a road survey, with most of the small mammals counted dead at the road shoulder. Sight distances of 5 m for small mammals and 50 m for turkey from each side of the road were assumed, and the population density of each species was estimated as number of animals per unit area using total length of the road driven during the survey. The number of deer harvested in northeastern Kansas (21,542 head, or 28% of the total deer harvested in Kansas [76,935 head]) was estimated and equally distributed in the total land area of northeastern Kansas (23,841 km^2) as a fraction of the total deer population of Kansas (330,000) to get the deer population density in the watershed (3.88 head per km^2). Overall, the current Rock Creek watershed scenario reflects wildlife populations (and corresponding 1000 kg AUs) of about 173 turkeys (1.2 AUs), 299 deer (25 AUs), and 20 small mammals (1.4 AUs) for this study. Animal weights were estimated based on information received from Mammals of Kansas (Timm et al., 2007) and personal communication with an expert (J. Pitman, 2006, personal communication).

Management Scenarios

Grassland

The grassland in the watershed consisted of four combinations of three major grass types and two management conditions: grazed native prairie grasses (68%), grazed tall fescue grass (12%), non-grazed smooth bromegrass (10%), and non-grazed tall fescue (10%) (W. Boyer, 2005, personal communication). The non-grazed fescue was used for haying. Non-grazed bromegrass included CRP grassland (about 5% of the watershed); the CRP grass was not hayed. The native prairie grass and bromegrass typically were not fertilized, but fescue was fertilized with 70-15-0 (NPK) (W. Boyer, 2005, personal communication). It was estimated that about 1.53 kg ha^{-1} day^{-1} dry weight of manure was applied to pastures due to grazing operations during the growing season. This estimation was based on *ASAE Standards* (2003).

Total air-dry forage required for 573 AUs in the pasture was estimated as 195,393 kg for 30 days using 341 kg of air-dry forage required for an AU for 30 days (Paul and Watson, 1994). Consequently, the consumed dry weight of biomass was estimated as 2.03 (kg ha^{-1} day^{-1}). The trampled dry weight of biomass (used as a model input) was estimated as 0.41 kg ha^{-1} day^{-1} assuming that 20% of the air-dry biomass consumed by livestock was trampled every day. Grazing was started about a month earlier on tall fescue grasslands (April 1) than on native prairie grass (May 1), but each assumed a grazing period of 153 days. About 3.7 Mg ha^{-1} of hay was harvested annually from the non-grazed fescue. Cattle density in the grazed grassland was estimated as 3.04 ha per cow-calf pair based on the bluestem pasture guidelines for grazing (KDA, 2004b). Because cattle did not graze pastures from October to March, no biomass uptake from pastures occurred, with no grass trampling and no manure deposition on the soil during this period.

Daily livestock manure load from a given confined animal feedlot was applied to a grazed grasslands HRU within the subwatershed in which the active, permitted feedlot was located using methods similar to manure application by grazing. Only one permitted feedlot was located in the watershed; it maintained 130 AUs and produced about 29.4 kg ha^{-1} day^{-1} of solid manure as an additional bacterial source to be accounted for in one 36 ha grazed grassland area.

The winter feeding areas were modeled based on the assumption that the estimated total number of AUs (40% of 573 AUs) was confined within 40% (12.8 km^2) of the grazed grassland of the watershed (32 km^2). It was estimated that about 1.53 kg ha^{-1} day^{-1} dry weight of cattle manure was applied in the respective pastures of the subwatersheds due to winter feeding operations. Animals in feedlots and winter feeding areas contributed fecal bacteria for 212 days during the dormant season of the year (October 1 to March 30).

Cropland and Woodland

Corn and soybean were the major warm-season crops (planted May 1, harvested October 1), and winter wheat was the primary cool-season crop (planted October 20, harvested July 30) grown in three-year rotations with crop residues remaining between crops (W. Boyer, 2005, personal communication). These dates represent typical planting and harvesting dates in the watershed. Conservation tillage is the most widely adopted tillage practice in the watershed for corn, soybean, and wheat production. All cropland HRUs were simulated to have 5 m wide filter strips, which were smaller than NRCS guidelines but selected to provide an average sediment removal rate of 59%, consistent with local field buffer strip data (Ngandu, 2004). The HRU sizes (0.10 to 150 ha) were roughly the same magnitude as the cropland fields.

Woodlands were primarily located in riparian corridors. All wildlife-generated manure and associated bacteria were applied in the woodland HRUs on a daily basis.

Statistical Analysis Methods

The SWAT model was validated using monitored flow and FCB concentrations from nine daily events. The statistical parameter used to evaluate measured vs. predicted daily mean flow includes coefficient of determination (R^2) and Nash-Sutcliffe efficiency index (E). The R^2 value indicates how consistently measured vs. predicted values follow a best fit line; therefore, if the R^2 value is less than or very close to zero, the model prediction is considered unacceptable or poor. If the value is 1.0, then the model prediction is perfect (Santhi et al., 2001). However, R^2 only describes how much of the observed dispersion is explained by the prediction; therefore, R^2 is not suggested to use alone. The E statistic indicates how consistently measured values (range $-\infty$ to 1.0) match predicted values (eq. 2; Nash and Sutcliffe, 1970). Moriasi et al. (2007) proposed performance categories for monthly time-step model results and recommended these categories be adjusted according to the evaluation time step and project scope. For validation of daily time-step results in this study, we classified model performance as excellent for R^2 or $E \geq 0.90$, very good for R^2 or E = 0.75 to 0.89, good for R^2 or E = 0.50 to 0.74, fair for R^2 or E = 0.25 to 0.49, poor for R^2 or E = 0 to 0.24, and unsatisfactory for R^2 or $E < 0$.

$$E = 1 - \frac{\sum_{i=1}^{n} (O_i - P_i)^2}{\sum_{i=1}^{n} (O_i - O_m)^2} \qquad (2)$$

where

E = model efficiency index

O_i = ith observed value

P_i = ith predicted value

O_m = observed mean value.

The E statistic is strongly overestimated by larger values (Legates and McCabe, 1999; Loague and Freeze, 1985). In addition, E is most accurate for normally distributed data, since it is a mean-based function (Coffey et al., 2004). For the assessment of model efficiency in this study, the extreme range and non-normality in distribution of stream FCB data were addressed by using $\log_{10}$ transformation of observed and modeled values, similar to other modeling studies using bacteria data having a range of several orders of magnitude (Hill and Sobsey, 2003; Sharma, et al., 2004; Benham et al., 2006; Mankin et al., 2007; Parajuli et al., 2009). Skewness of the nine original stream FCB data was 2.41, whereas skewness below 0.5 is considered fairly symmetric (Evans and Olson, 2002; Zhang, 2004). Skewness of $\log_{10}$-transformed observed data was -0.45.

Sensitivity Analysis

Model parameter and input values selected for IPP sensitivity analysis covered the range of values recommended for use in the model. The baseline values were selected to be consistent with the model default and literature values. For each parameter set, the SWAT model was run for two years (2003 to 2004). Model results were analyzed for 2004, consistent with available flow and bacteria data, and 2003 was used as an initialization period. Geometric means of predicted daily fecal bacteria concentrations for the 190 daily runoff events that generated non-zero watershed outlet bacterial concentrations and for the nine measured surface runoff events that generated bacterial concentrations in 2004, similar to Parajuli (2007), were used for sensitivity analyses.

The relative sensitivity (S) index for each modeled case was analyzed using equation 3 (James and Burges, 1982; Nearing et al., 1989; Jesiek and Wolfe, 2005; White and Chaubey, 2005):

$$S = \frac{(P_i - P_b)}{(I_i - I_b)} \frac{I_b}{P_b} \qquad (3)$$

Table 1. Relative sensitivity (S) index classes (Zerihun et al., 1996).

S Index Class	Symbol	S Index Range
No sensitivity	N	$0 < \lvert S \rvert \leq 0.10$
Low sensitivity	L	$0.10 < \lvert S \rvert \leq 0.50$
Moderate sensitivity	M	$0.50 < \lvert S \rvert \leq 2.0$
High sensitivity	H	$2.00 < \lvert S \rvert \leq 5.00$
Very high sensitivity	VH	$\lvert S \rvert > 5.00$

where

S = relative sensitivity index
I_i = ith model input parameter value
I_b = baseline model input parameter value
P_i = ith predicted value (model output)
P_b = predicted value (model output) for baseline.

An S index of 0 indicates that the output did not respond to changes in the input, while an S index of 1 indicates that the normalized output range was directly proportional to the normalized input range. Additionally, a negative value of S indicates that an increase in input value caused a decrease in output value. A greater absolute value of S indicates greater impact of an input parameter on a particular output (Walker et al., 2000). Relative sensitivity was classified based on table 1 (Zerihun et al., 1996; Walker et al., 2000; Graff et al., 2005).

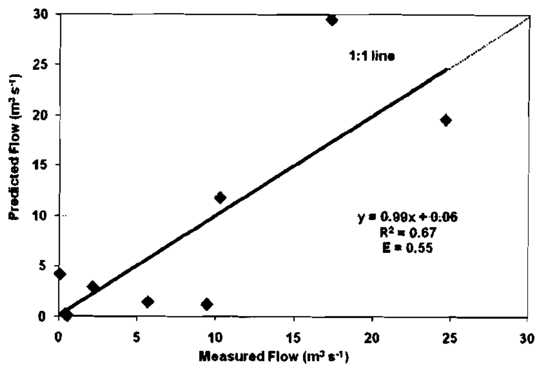

Figure 3. Measured daily flow and model responses for nine surface runoff events.

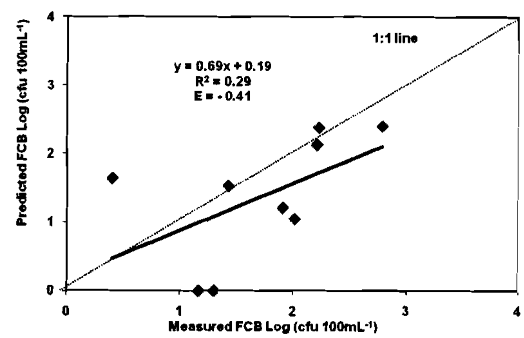

Figure 4. Measured daily fecal coliform bacteria (FCB) concentration and model response for nine surface runoff events.

Results and Discussion

Model Validation

The uncalibrated baseline SWAT model, using ground-truthed land use conditions and other parameters to define current conditions, predicted the daily average flow of the watershed with good correlation and model efficiency (R^2 = 0.67, E = 0.55) (fig. 3). Comparing this uncalibrated result against all daily model statistics reported by Gassman et al. (2007) from a literature review of more than 250 published SWAT studies, the R^2 was better than 13 (32%) of calibration values and 17 (44%) of validation values reported, and E was better than 42 (42%) of calibration values and 46 (58%) of validation values reported. Although it is likely that further model efficiency improvements would have been possible with calibration, the objectives of this study required reasonable, not optimum, model results. Confirmation of reasonable flow results provided confidence that the sensitivity analysis was being conducted with minimal bias from the flow-prediction algorithms of the model.

The uncalibrated baseline SWAT 2005 microbial sub-model (modified on 12 March 2009), using livestock, septic, and wildlife loadings to represent current conditions, underpredicted average daily FCB concentration by 20% (average log value of 1.716 vs. 1.364 cfu 100 mL^{-1}). The uncalibrated model produced fair correlation and unsatisfactory model efficiency (R^2 = 0.29, E = -0.41) (fig. 4). Although the model underpredicted bacteria concentration during five out of nine of the runoff events, the slope of predicted vs. measured regression (0.69) together with fair correlation indicated the model behaved at a level of stability that was considered adequate to allow sensitivity analysis. Errors associated with the model and with measurements of FCB concentration were not separated in this study.

Model Parameter Sensitivity

The SWAT model showed no to low S index for BACTKDQ values higher than the baseline value (175.00) but low to moderate S index for lower values (table 2). The geometric mean value of fecal bacteria output tripled from the baseline BACTKDQ (175) to 1.8, and increased by an additional 65% as BACTKDQ was reduced further to 0.9. The BACTKDQ parameter in SWAT provided an equilibrium constant to control the release of FCB from surface soil (top 10 mm) into surface runoff, with increasing BACTKDQ leading to decreasing FCB release to surface runoff for a given surface-soil FCB concentration. Increased sensitivity for lower BACTKDQ values would be consistent a non-limiting supply of FCB in surface soil and increasing release per unit of available FCB to surface runoff. The low to moderate model sensitivity for low BACTKDQ conditions, particularly less than 44, indicated that BACTKDQ will need to be more precisely defined if the user finds that it should be in this range.

Parajuli et al. (2006) reported that using BACTKDQ below 44 could be sensitive for SWAT 2000. They found that BACTKDQ had low sensitivity because the maximum change in percent output using SWAT 2000 was estimated as 51% based on base output (GM = 713 cfu 100 mL^{-1}) and maximum output (GM = 1077 cfu 100 mL^{-1}), whereas this study, using SWAT 2005, determined that the maximum change in percent output was 265% based on base output (4.75 cfu 100 mL^{-1}) and maximum output (17.35 cfu

Table 2. Relative sensitivity (*S*) analysis of BACTKDQ (bacteria partition coefficient in surface runoff).[a]

		All (190) Surface Runoff Events			Nine Surface Runoff Events		
Input	% Change from Base	GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class	GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class
0.9	-99.5	17.35	-2.67	M	123.06	-4.34	H
1.8	-99.0	14.30	-2.03	M	84.14	-2.66	H
43.8	-75.0	6.10	-0.38	L	25.33	-0.13	L
87.5	-50.0	5.32	-0.24	L	24.02	-0.08	N
131.3	-25.0	4.97	-0.19	L	24.00	-0.15	L
157.5	-10.0	4.83	-0.17	L	23.19	-0.02	N
175.0	0.0	4.75	Base	--	23.14	Base	--
192.5	10.0	4.69	-0.13	L	23.12	-0.01	N
218.8	25.0	4.60	-0.05	N	23.08	-0.01	N
262.5	50.0	4.49	-0.06	N	23.04	-0.01	N
350.0	100.0	4.34	-0.05	N	23.02	-0.01	N
525.0	200.0	4.16	-0.04	N	23.02	0.00	N

[a] Base model input values: BACTKDQ = 175, TBACT = 1.07, WDLPQ = 0.23, WDLPS = 0.023, BACTKDDB = 0.9, and BACTLPDB = 3.29 × 107 cfu g-1.

100 mL^{-1}), particularly at the extreme lower values of the BACTKDQ using 190 rainfall-runoff events (table 2).

The TBACT parameter generally demonstrated low *S* index throughout the range of values recommended for use in SWAT (table 3) when compared for both all 190 and nine surface runoff events. The TBACT parameter, as shown in equation 1, governs the degree to which the bacterial die-off rate changes in response to temperature. Wang et al. (2004) calculated the TBACT parameter in excreted manure as 1.026 for temperatures between 4°C and 27°C and 1.034 for temperatures between 27°C and 41°C, all within the range (1.07 ±0.05) given by Reddy et al. (1981) in which most biological reactions occur. Although using a TBACT value outside the range of 1.07 ±0.05 might increase model response sensitivity (Reddy et al., 1981), these extreme TBACT values might not be relevant for most environments for which SWAT modeling applies. Thus, the increased sensitivity of the SWAT 2000 model reported by Parajuli et al. (2006) at the extreme upper range of TBACT values (1.12 to 1.18) might not be relevant in practice. The SWAT 2005 model used in this study did not demonstrate a similar sensitivity response in this range.

The SWAT 2005 model uses only one TBACT value for the entire model simulation period (365 days in this study). Because the model demonstrated generally low sensitivity within the range of 1.07 ±0.05, changing TBACT over that range would not have a substantial effect on model results. This would support the use of the SWAT model TBACT default value of 1.07 (Neitsch et al., 2005).

The WDLPQ parameter had no to low *S* index throughout the range of values studied (table 4). Decreasing the WDLPQ value generally increased model relative sensitivity when compared with 190 surface runoff events. The WDLPQ governs the bacterial die-off rate in soil solution, so decreasing WDLPQ values would increase the population of bacteria available for transport in surface runoff. As anticipated, decreased WDLPQ values caused greater modeled bacterial concentrations. The decreased WDLPQ values also contributed to a greater relative sensitivity of the WDLPQ parameter for values less than 0.23, particularly for 190 surface runoff events. However, overall relative sensitivity for both 190 and nine surface runoff events were low (table 4).

The WDLPS parameter was the least sensitive parameter in this study (table 5). It was tested with additional model simulations (table 6) if the least sensitivity was caused by the assumption of 90% bacteria (BACTKDDB = 0.9) in the solution phase, leaving only 10% of bacteria in the sorbed phase. There was almost no difference in bacteria output regardless of the die-off rate in sorbed-phase bacteria (table 5). A bacteria GM output value of 5.17 (cfu 100 mL^{-1}) was found for the base condition in table 6 for 190 surface runoff events when using BACTKDDB = 0.10, which means assuming only 10% of the bacteria in solution phase and 90% of the bacteria in sorbed phase. The model sensitivity index

Table 3. Relative sensitivity (*S*) analysis of TBACT (temperature adjustment factor).[a]

		All (190) Surface Runoff Events			Nine Surface Runoff Events		
Input	% Change from Base	GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class	GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class
0.856	-20.0	5.09	-0.36	L	24.79	-0.36	L
0.963	-10.0	4.83	-0.17	L	24.12	-0.42	L
1.017	-5.0	4.74	0.04	N	23.71	-0.50	L
1.049	-2.0	4.75	0.00	N	23.24	-0.22	L
1.070	0.0	4.75	Base	--	23.14	Base	--
1.091	2.0	4.76	0.11	L	23.39	0.55	L
1.124	5.0	4.79	0.17	L	23.69	0.47	L
1.177	10.0	4.90	0.32	L	24.08	0.41	L
1.284	20.0	4.96	0.22	L	25.45	0.50	L

[a] Base model input values: BACTKDQ = 175, TBACT = 1.07, WDLPQ = 0.23, WDLPS = 0.023, BACTKDDB = 0.9, and BACTLPDB = 3.29 × 107 cfu g-1.

Table 4. Relative sensitivity (*S*) analysis of WDLPQ (less-persistent bacteria die-off in solution phase).[a]

Input	% Change from Base	All (190) Surface Runoff Events GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class	Nine Surface Runoff Events GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class
0.115	-50.0	5.35	-0.25	L	23.67	-0.05	N
0.173	-25.0	4.94	-0.16	L	23.48	-0.06	N
0.207	-10.0	4.82	-0.15	L	23.34	-0.09	N
0.230	0.0	4.75	Base	--	23.14	Base	--
0.253	10.0	4.70	-0.11	L	21.98	-0.50	L
0.288	25.0	4.62	-0.11	L	21.90	-0.21	L
0.345	50.0	4.52	-0.10	N	21.75	-0.12	L
0.460	100.0	4.36	-0.08	N	21.67	-0.06	N
0.690	200.0	4.15	-0.06	N	25.56	0.05	N

[a] Base model input values: BACTKDQ = 175, TBACT = 1.07, WDLPQ = 0.23, WDLPS = 0.023, BACTKDDB = 0.9, and BACTLPDB = 3.29 × 107 cfu g-1.

Table 5. Relative sensitivity (*S*) analysis of WDLPS (less-persistent bacteria die-off in sorbed phase).[a]

Input	% Change from Base	All (190) Surface Runoff Events GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class	Nine Surface Runoff Events GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class
0.012	-50.0	4.90	-0.06	N	23.42	-0.02	N
0.017	-25.0	4.82	-0.06	N	23.34	-0.03	N
0.021	-10.0	4.77	-0.04	N	23.28	-0.06	N
0.023	0.0	4.75	Base	--	23.14	Base	--
0.025	10.0	4.73	-0.04	N	23.12	-0.01	N
0.029	25.0	4.70	-0.04	N	23.09	-0.01	N
0.035	50.0	4.66	-0.04	N	23.05	-0.01	N
0.046	100.0	4.61	-0.03	N	23.03	-0.00	N
0.069	200.0	4.59	-0.02	N	23.02	-0.00	N

[a] Base model input values: BACTKDQ = 175, TBACT = 1.07, WDLPQ = 0.23, WDLPS = 0.023, BACTKDDB = 0.9, and BACTLPDB = 3.29 × 107 cfu g-1.

Table 6. Relative sensitivity (*S*) analysis of BACTKDDB (bacteria partition coefficient in manure).[a]

Input	% Change from Base	All (190) Surface Runoff Events GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class	Nine Surface Runoff Events GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class
0.90	0.00	4.75	Base	--	23.14	Base	--
0.70	-22.0	4.82	-0.07	N	23.75	-0.12	L
0.50	-44.0	4.89	-0.07	N	26.64	-0.34	L
0.30	-67.0	5.14	-0.12	L	27.72	-0.30	L
0.10	-89.0	5.17	-0.10	N	27.91	-0.23	L

[a] Base model input values: BACTKDQ = 175, TBACT = 1.07, WDLPQ = 0.23, WDLPS = 0.023, BACTKDDB = 0.9, and BACTLPDB = 3.29 × 107 cfu g-1.

was increased when tested with different solution and sorbed phase assumptions (table 6), but the WDLPS parameter was still found less sensitive.

The WDLPQ and WDLPS parameters had no recommended range of values in the SWAT documentation. However, WDLPS had no sensitivity in the model, and WDLPQ had low sensitivity within ±25% of the base value (0.23) used by Baffaut and Benson (2003). Results of this study support the use of 0.23 for WDLPQ and 0.023 for WDLPS for watershed conditions where BACTKDDB = 0.9 applies.

Input Parameter Sensitivity

Bacteria concentrations in manure (BACTLPDB) showed a direct relationship between bacteria concentration and bacteria prediction. The relative sensitivity was low (0.14 to 0.25) over a range of input values from 3.29×10^5 cfu g^{-1} (99% decrease from baseline) to 3.29×10^8 cfu g^{-1} (900% increase) when using 190 surface runoff events (table 7). The nine surface runoff events showed further decrease in the sensitivity, but ranged from no to low.

It is important to note that the current version of SWAT 2005 (prior to the one used in this study, but still widely used) contained an error accepting input values with more than eight integer digits. Bacteria concentration was stored in the model in floating-point 8.3 format (eight integer digits followed by three decimal digits: xxxxxxxx.xxx). When BACTLPDB values with nine integer digits were input, the model truncated one integer digit, resulting in a lower stream bacteria concentration. For example, a BACTLPDB of 99,999,999 cfu g^{-1} resulted in predicted bacteria concentration of about eight times more bacteria than for 100,000,000 cfu g^{-1}, since the 100,000,000 cfu g^{-1} was modeled as 10,000,000 cfu g^{-1}. Parajuli et al. (2006) reported a similar error with the SWAT 2000 model. This error was corrected in the version of SWAT 2005 used in this study.

Table 7. Relative sensitivity (*S*) analysis of BACTLPDB (bacteria concentration in manure).[a]

Input (cfu g^{-1})	% Change from Base	All (190) Surface Runoff Events GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class	Nine Surface Runoff Events GM Output (cfu 100 mL^{-1})	*S* Index	*S* Index Class
329,000	-99.0	3.60	0.24	L	21.10	0.09	N
3,290,000	-90.0	3.66	0.25	L	21.30	0.09	N
16,500,000	-50.0	4.22	0.22	L	22.08	0.09	N
24,700,000	-25.0	4.51	0.20	L	22.36	0.14	L
32,900,000	0.0	4.75	Base	--	23.14	Base	--
49,400,000	50.0	5.17	0.18	L	23.54	0.03	N
65,800,000	100.0	5.52	0.16	L	23.62	0.02	N
99,900,000	204.0	6.13	0.14	L	24.11	0.02	N
100,000,000	204.0	6.13	0.14	L	24.11	0.02	N
329,000,000	900.0	12.70	0.19	L	53.05	0.14	L

[a] Base model input values: BACTKDQ = 175, TBACT = 1.07, WDLPQ = 0.23, WDLPS = 0.023, BACTKDDB = 0.9, and BACTLPDB = 3.29 × 107 cfu g-1.

Conclusions

This study demonstrated methods used to characterize bacteria source loads and assess model sensitivity to user-defined model parameters, all essential to improve model accuracy. Results of this study can help researchers in watershed management and modeling decisions choose model parameters with due consideration of model sensitivity.

Sensitivity of model and input parameters generally ranked as BACTKDQ > TBACT > BACTKDDB > BACTLPDB > WDLPQ > WDLPS. In summary, the BACTKDQ parameter had moderate to high sensitivity, especially in the extreme range of lower values. Otherwise, generally low sensitivity was observed. The TBACT parameter generally showed low sensitivity, especially for either extremely low or high values. The WDLPQ and WDLPS parameters generally showed no sensitivity, except that WDLPQ showed low sensitivity for low WDLPQ values. The bacteria concentration input parameter generally indicated low sensitivity. This study suggested default WDLPQ (0.23) and WDLPS (0.023) values in the SWAT model to reflect natural life perspectives. The IPP method of sensitivity analysis used in this study was found simple and reasonable to use to test the performance of the SWAT 2005 model (revised 12 March 2009).

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