WATER QUALITY MODELING FOR THE RACCOON RIVER WATERSHED USING SWAT

M. K. Jha, P. W. Gassman, J. G. Arnold

ABSTRACT. *The Raccoon River watershed (RRW) in west-central Iowa has been recognized as exporting some of the highest nitrate-nitrogen loadings in the U.S. and is a major source of sediment and other nutrient loadings. An integrated modeling framework has been constructed for the 9,400 km² RRW that consists of the SWAT (Soil and Water Assessment Tool) model, the interactive SWAT (i_SWAT) software package, the Load Estimator (LOADEST) computer program, and other supporting software and databases. The simulation framework includes detailed land use and management data, such as different crop rotations, and an array of nutrient and tillage management schemes, derived from the USDA National Resources Inventory (NRI) databases and other sources. This article presents the calibration and validation of SWAT for the streamflow, sediment losses, and nutrient loadings in the watershed, and an assessment of land use and management practice shifts in controlling pollution. Streamflow, sediment yield, and nitrate loadings were calibrated for the period 1981-1992 and validated for the period 1993-2003. Limited field data on organic nitrogen, organic phosphorus, and mineral phosphorus allowed model validation for the period 2001-2003. Model predictions generally performed very well on both an annual and monthly basis during the calibration and validation periods, as indicated by R^2 and Nash-Sutcliffe efficiency (E) values that exceeded 0.7 in most cases. A set of land use change scenarios depicting conversion of cropland into land set-aside resulted in large reductions of sediment yield at the watershed outlet. A second scenario set found that reductions in nutrient applications of 10% to 20% resulted in similar predicted percentage reductions in nitrate loadings at the watershed outlet and in corresponding corn yield reductions of 3% to 6%.*

Keywords. *Calibration, Management practices, Modeling, Nutrients, Raccoon River watershed, SWAT, Water quality.*

Excess nitrogen, phosphorus, and sediment loadings have resulted in water quality degradation within the upper Mississippi River and its tributaries. This is particularly true for watersheds draining portions of Iowa, which are generally greatly impacted from agricultural nonpoint-source pollution. Kalkoff et al. (2000) report that nitrogen and phosphorus levels measured in several large eastern Iowa watersheds, which drain to the Mississippi River, were among the highest found in the Corn Belt region and in the entire U.S. as part of the U.S. Geological Survey (USGS) National Water-Quality Assessment Program. Sediment loads discharged from Iowa watersheds to the Mississippi River are also reported to be among the highest in the upper Mississippi River basin (USGS, 2006a). The annual export of nitrate from surface waters in Iowa was estimated to be about 20% of the total nitrate load delivered by the Mississippi River to the Gulf of Mexico during 2000-2002 (IDNR-IGS, 2004), which is a disproportionate amount considering that Iowa covers less than 5% of the Mississippi River drainage area. The nitrate load discharged from the mouth of the Mississippi River has been implicated as the primary cause of the seasonal oxygen-depleted hypoxic zone that occurs in the Gulf of Mexico, which has covered upwards of 20,000 km² in recent years (Rabalais et al., 2002).

The Raccoon River watershed (RRW) is located in an intensive agricultural production region in west-central Iowa (fig. 1). The river is impacted by sediment, phosphorus, and nitrogen pollution (Lutz, 2005), which originate primarily from nonpoint sources. The nutrient input sources include widespread use of fertilizers, livestock manure applications, legume fixation, and mineralization of soil nitrogen. Nitrate pollution is a particularly acute problem in the RRW and is transported primarily through groundwater discharge via baseflow and tile drainage. Schilling and Zhang (2004) reported that nitrate export from the RRW is among the highest in the interior U.S. The watershed's high concentrations of nitrates have exceeded the federal maximum contaminant level (MCL) standard of 10 mg L^{-1} with enough frequency since the late 1980s to warrant the Des Moines Waterworks' (DMWW) installation and operation of the world's largest nitrate removal facility. Sections of the Raccoon River have also been listed in Iowa's Federal Clean Water Act 303(d) list of impaired waters, due to the elevated nitrate levels.

Several studies have been performed in the RRW to quantify nitrate concentration patterns and corresponding streamflow relationships. Schilling and Lutz (2004) examined a 28-year record (1972-2000) of streamflow and nitrate concentrations measured in the Raccoon River and reported evidence of strong seasonal patterns in annual nitrate

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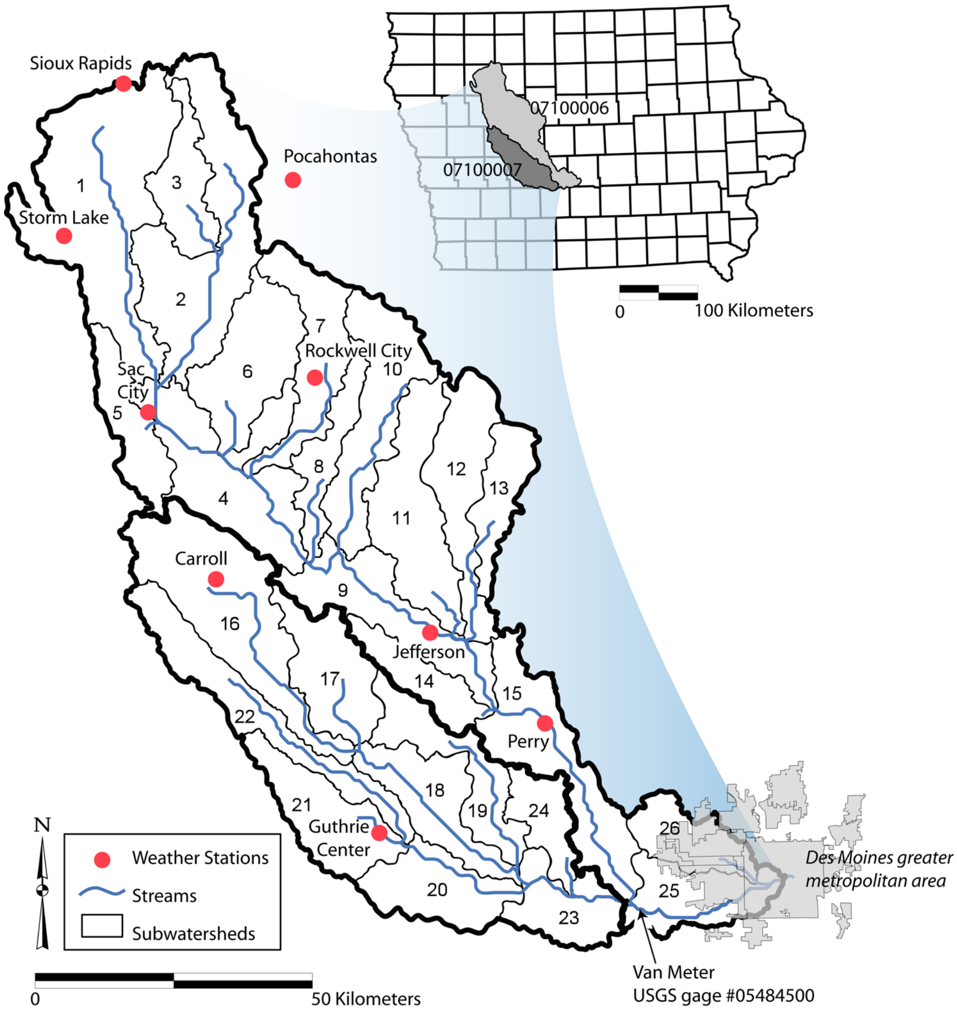

Figure 1. Raccoon River watershed SWAT configuration, showing the outline of the two 8-digit watersheds, the delineated 10-digit subwatersheds, and the location of the weather stations.

concentrations, with higher concentrations occurring in the spring and fall. No long-term trends in nitrate concentrations were noted in the entire period. Schilling and Zhang (2004) described nitrate loading patterns in the Raccoon River and found that nitrate losses in baseflow comprised nearly two-thirds of the total nitrate load over the same 28-year monitoring period. They also found that seasonal patterns of nitrate loads were similar to nitrate concentration patterns, with baseflow contributions to nitrate loads greatest in the spring and later fall, when baseflow contributed more than 80% of the total nitrate export.

The focus of this study was to assess the ability of the SWAT (Soil and Water Assessment Tool) model version 2000 (Arnold et al., 1998; Arnold and Fohrer, 2005; Gassman et al., 2007) to simulate stream flow and associated movement of nitrogen, phosphorus, and sediment in the RRW. No previous studies have been reported in the literature regarding an in-depth simulation study of the RRW. Likewise, previous simulation studies performed totally or in part with SWAT for watersheds in Iowa (Chaplot et al., 2004; Chaplot, 2005; Chaplot et al., 2005; Gassman et al., 2002; Gassman et al., 2006; Santelman et al., 2004; Vaché et al., 2002) did not report a comprehensive evaluation of the accuracy of predicted sediment, nitrogen, and phosphorus losses. It is further important to perform a comprehensive test of the model for the RRW because the Iowa Department of Natural Resources (IDNR) has chosen SWAT to develop a nitrate Total Maximum Daily Load (TMDL) for the watershed (K. Schilling, Geological Survey, Iowa Department of Natural Resources, Iowa City, Iowa; personal communication). Developing a reliable SWAT simulation tool could provide very useful insight into the movement and potential mitigation of nonpoint-source pollution in the RRW, which is especially important with regards to the pervasive high nitrate loadings in the watershed. The results could also provide useful insight into the application of SWAT and similar tools for other similarly impacted agricultural watersheds in Iowa and the Midwest U.S. Thus, the objectives of this study were to: (1) calibrate and validate the SWAT model for stream flow, sediment, and nutrients for the entire watershed, and (2) evaluate the sensitivity of SWAT sediment and nutrient predictions in response to several land use and nutrient management scenarios.

Materials and Methods

SWAT Model

SWAT is a hydrologic and water quality model developed by the USDA Agricultural Research Service (USDA-ARS). It is a long-term, continuous, watershed-scale simulation model that operates on a daily time step and is designed to assess the impact of different management practices on water, sediment, and agricultural chemical yields. The model is physically based, computationally efficient, and capable of simulating a high level of spatial detail. Major model components include weather, hydrology, soil temperature, crop growth, sediment, nutrients, pesticides, and land management. In SWAT, a watershed is divided into multiple subwatersheds, which are further subdivided into unique soil/land use characteristics called hydrologic response units (HRUs). The water balance of each HRU is represented by four storage volumes: snow, soil profile, shallow aquifer, and deep aquifer. Flow generation, sediment yield, and pollutant loadings are summed across all HRUs in a subwatershed, and the resulting flow and loads are then routed through channels, ponds, and/or reservoirs to the watershed outlet.

Surface runoff from daily rainfall is estimated with the modified SCS curve number method (Neitsch et al., 2002b), which estimates the amount of runoff based on local land use, soil type, and antecedent moisture condition. The Green-Ampt method (Green and Ampt, 1911) of estimating infiltration is an alternative option for estimating surface runoff and infiltration that requires sub-daily weather data. Melted snow is treated the same as rainfall for estimating runoff and percolation. Channel routing is simulated using either the variable-storage method or the Muskingum method; both methods are variations of the kinematic wave model (Chow et al., 1988). Three methods of estimating potential evapotranspiration are available: Priestley-Taylor (Priestley and Taylor, 1972), Hargreaves (Hargreaves and Samani, 1985), and Penman-Monteith (Monteith, 1965; Allen et al., 1989).

Erosion and sediment yield are estimated for each HRU with the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1995). The channel sediment routing equation uses a modification of Bagnold's sediment transport equation (Bagnold, 1977) that estimates the transport concentration capacity as a function of velocity. The model either deposits excess sediment or re-entrains sediment through channel erosion depending on the sediment load entering the channel.

A generic crop growth submodel is used in SWAT that is a simplified version of the crop growth functions developed for the EPIC (Environmental Impact Policy Climate) model (Williams et al., 1989; Gassman et al., 2005). A wide range of crop rotations can be simulated in the model, as well as different grassland and forest systems. Yields and/or biomass output are estimated at the HRU level in SWAT.

SWAT simulates the complete nutrient cycle for nitrogen and phosphorus. The nitrogen cycle is simulated using five different pools; two are inorganic forms (ammonium and nitrate), while the other three are organic forms (fresh, stable, and active). Similarly, SWAT monitors six different pools of phosphorus in soil; three are inorganic forms, and the rest are organic forms. Mineralization, decomposition, and immobilization are important parts in both cycles. These processes are allowed to occur only when the temperature of the soil layer exceeds 0°C. Nitrate export from runoff, lateral flow, and percolation are estimated as products of the volume of water and the average concentration of nitrate in the soil layer. Organic N and organic P transport with sediment is calculated with a loading function developed by McElroy et al. (1976) and modified by Williams and Hann (1978) for application to individual runoff events. The loading function estimates daily organic N and P runoff loss based on the concentrations of constituents in the top soil layer, the sediment yield, and an enrichment ratio. The amount of soluble P removed in runoff is predicted using the labile P concentration in the top 10 mm of the soil, the runoff volume, and a phosphorus soil partitioning coefficient. In-stream nutrient dynamics are simulated in SWAT using the kinetic routines from the QUAL2E in-stream water quality model (Brown and Barnwell, 1987).

A detailed theoretical description of SWAT and its major components can be found in Neitsch et al. (2002b). SWAT has been widely validated across the U.S. and in other regions of the world for a variety of applications including hydrologic, pollutant loss, and climate change studies. An extensive set of SWAT applications are documented in Gassman et al. (2007).

Watershed Description

The RRW (fig. 1) encompasses approximately 9,400 km^2 of prime agricultural land in west-central Iowa. It is comprised of two U.S. Geological Survey (USGS) 8-digit Hydrologic Cataloging Unit (HCU) watersheds (Seaber et al., 1987): HCU 0710006 (North Raccoon) covers approximately 5,950 km^2, whereas HCU 0710007 (South Raccoon) covers approximately 3,450 km^2. Land use in the RRW is dominated by agriculture and is comprised of cropland (75.3%), grassland (16.3%), forest (4.4%), and urban (4.0%). The watershed is a part of the Des Moines lobe of the Wisconsin Glacier, which is a swampy, prairie pothole region.

The Raccoon River and its tributaries drain all or parts of 17 of Iowa's 99 counties before emptying into the Des Moines River in the city of Des Moines. It is the primary source of drinking water for more than 370,000 residents in Des Moines and other central Iowa communities. The primary sources of nitrates in the RRW are high organic matter soils and applications of fertilizer and livestock manure to cropland. Cropland production areas are also the primary sources of sediment losses and other nutrient loadings to the Raccoon River (Woolson, 2002).

Input Data

Basic input data required for a SWAT simulation include topography, weather, land use, soil, and management data. Topography data are used to delineate a watershed into multiple subwatersheds and also to calculate watershed/subwatersheds parameters such as slope and slope length. Topography data were obtained in the form of a digital elevation model (DEM) at 90 m resolution from the U.S. Environmental Protection Agency (USEPA) Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) modeling package version 3.1 (USEPA, 2006). Daily climatic data include precipitation, maximum and minimum air temperature, solar radiation, wind speed, and relative humidity for each subwatershed. These climatic inputs can be entered from historical records and/or generated internally in SWAT using monthly climate statistics that are based on long-term weather records. In this study, daily precipitation and temperature data were collected from the National Climatic Data Center (NCDC, 2006) for ten weather stations located in and around the watershed (fig. 1).

Land use and conservation practice data were obtained from the USDA 1997 National Resources Inventory (NRI) database (USDA-NRCS, 2006a; Nusser and Goebel, 1997).

The NRI is a statistically based survey database that contains information for the entire U.S. including landscape features, soil type, cropping histories, and conservation practices for roughly 800,000 non-federal land "points." Each NRI point represents an area, generally ranging from a few hundred to several thousand hectares in size, which consists of homogeneous land use, soil, and other characteristics. These points are spatially referenced at the state level, major land resource area (MLRA) level, 8-digit watershed level, and county level. Major land use categories represented in the 1997 NRI include row crop, forest, urban, pasture, and range land. Conservation practice data are also provided for terraces, contouring, grassed waterways, and strip cropping. Tile drainage distribution data was obtained by linking the survey points to the 1992 NRI survey because the 1997 NRI survey does not report tile drainage usage. It was assumed that tile drains were installed on about 51% of the entire cropland area, based on the 1992 NRI data. The information on tillage implements simulated for different levels of tillage (conventional, reduced, mulch, and no-till) was obtained from data reported in the USDA 1990-95 Cropping Practices Survey (CPS) data (USDA-ERS, 2006b). Soil layer data were obtained from a soil database that contains soil properties consistent with those described by Baumer et al. (1994) and includes soil identification codes that allow linkage to NRI points.

Nutrient inputs to corn were simulated in the form of fertilizer and manure applications. Explicit fertilizer application rate data are not available for the RRW. Thus, a nitrogen application rate of 145.6 kg ha^{-1} (130 lb ac^{-1}) was assumed applied to corn regardless of rotation sequence. This rate is consistent with a suggested "average application rate range of 120 to 140 lb ac^{-1} for the RRW" as quoted in Woolson (2002) and is also consistent with 2003 Iowa statewide survey and sales average application rates (ISU, 2004a). The nitrogen was applied in either a single amount or in a split application, based on weighted random draws of surveyed nitrogen application practices in the CPS. The corn phosphorus fertilizer application rates were based on values reported in the CPS, which ranged between 28 and 67.2 kg ha^{-1}. No fertilizer applications were simulated for soybean, which is consistent with recent survey results showing that only 3% of the Iowa soybean acreage received nitrogen fertilizer in 2002 (USDA-ERS, 2006a).

The choice of appropriate manure application rates for the RRW is even more uncertain than those regarding fertilizer application rates. The manure-based nutrient application rates and associated application areas used for the study are listed in table 1, which were obtained from the USDA Natural Resources Conservation Service (R. Kellogg, USDA-NRCS, Washington, D.C.; personal communication, 2004) and were based on a previous national-level assessment of Comprehensive Nutrient Management Plans (USDA-NRCS, 2003). The total amount of nutrients that were assumed to be applied from livestock manure is based on the livestock numbers reported in the 1997 census of agriculture (USDA-NASS, 1997) for the counties that comprise the RRW and various nitrogen loss assumptions, as discussed in USDA-NRCS (2003). The county-level livestock numbers, which were obtained from the census data, were converted into livestock distributions at the 8-digit watershed level; the total areas receiving manure-derived nutrients from these livestock at the 8-digit level are listed in table 1.

Table 1. Simulated manure application rates and cropland areas receiving manure by 8-digit watershed.

8-Digit Watershed (HCU) ID Crop		Area (km^2)	Application Rates (kg ha^{-1})	
			Nitrogen	Phosphorus
07100006	Corn	310	314.9	125.4
07100006	Corn	303	173.8	78.0
07100006	Soybean	14	390.0	187.4
07100006	Pasture	12	151.8	59.9
07100007	Corn	31	53.3	23.6
07100007	Corn	37	162.6	69.9

These assumed manure application rates result in about 10% of the simulated cropland receiving manure and reflect assumptions that much of the manure will be applied at higher than agronomic rates. It was also assumed that manured cropland received fertilizer during years that corn was planted at the same rate as the HRUs planted to corn that did not receive manure. These generally high nutrients application rates reflect conditions of little or no manure nutrient crediting, such as described by Gassman et al. (2002) for a watershed in northeast Iowa and to a lesser extent by Shepard (1999) for two watersheds in Wisconsin. Actual manure management across the RRW likely reflects a broader spectrum of nutrient crediting, which would include cropland that only receives manure applied at appropriate agronomic rates. Two alternative manure management scenarios have been included in this study to provide further insight into the impacts of the manure applications.

Other input options that were used for the RRW study included the modified CN method to calculate partitioning of surface runoff versus infiltration, the Muskingum method for simulating the channel routing process, and the Hargreaves method for estimating potential evapotranspiration.

SWAT Baseline Scenario Configuration

The process of constructing the HRUs and subwatersheds for the SWAT simulation is shown in figure 2. The delineation of the 26 subwatersheds for the SWAT simulation (fig. 1) and the corresponding routing structure was performed with the ArcView SWAT (AVSWAT) interface (Di Luzio et al., 2004). The watershed was delineated so that the boundaries of the simulated subwatersheds were coincident with the 10-digit HCU watershed boundaries (USDA-NRCS, 2006b) that have been determined for the RRW. AVSWAT was also used to automatically assign one of the ten weather stations to each of the delineated 26 subwatersheds based on the proximity of the weather station to the centroid of the subwatershed.

The HRUs were constructed by aggregating NRI points that possess common soil type, land use, and management characteristics. The HRUs were first created for the two 8-digit watersheds that comprise the RRW (fig. 1). Common soil types were aggregated at the 8-digit level via a statistically based soil clustering process that was performed for NRI-linked soils for most of the U.S. (Sanabria and Goss, 1997). For land use, all of the points within a given category such as forest, urban, pasture, and Conservation Reserve Program (CRP) were clustered together, except for the cultivated cropland. For the cultivated cropland, the NRI points were first aggregated into several crop rotation land use clusters based on the NRI cropping histories. The final step of developing HRUs required aggregation across NRI points according to the management characteristics such as tile drainage (yes or

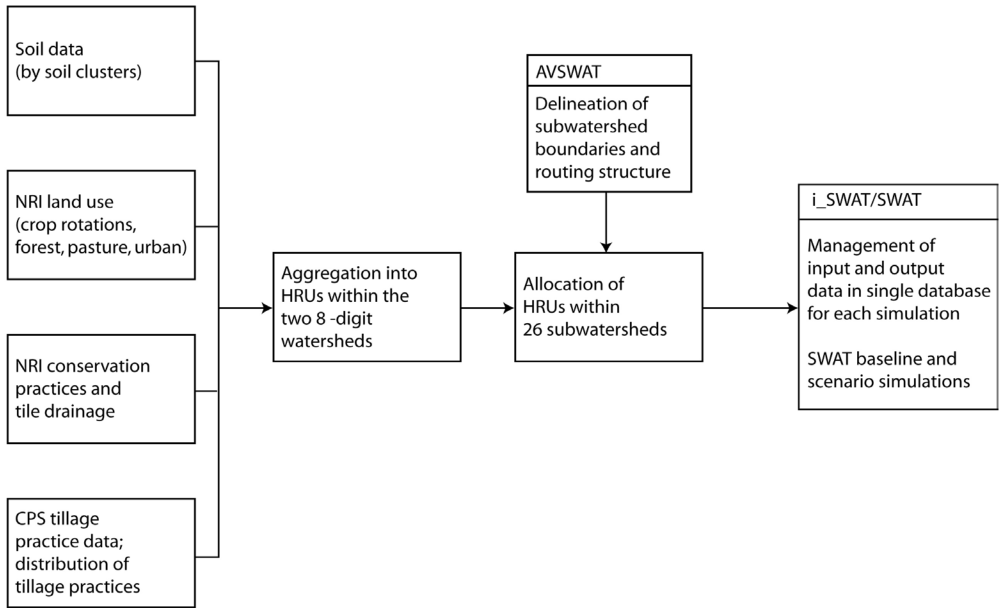

Figure 2. Schematic showing the data flows and software components of the RRW SWAT modeling system.

no), conservation practices (terracing, contouring, and/or strip cropping), and type of tillage (conventional, reduced, mulch, or no-till). A total of 307 HRUs were created between the two 8-digit watersheds. The 307 HRUs were then allocated to each of the 26 subwatersheds (fig. 2) based on guidance provided by 30 m resolution 2002 IDNR land use data (IDNR-IGS, 2004) and Iowa soil properties and interpretations database (ISPAID) soil data (ISU, 2004b), similar to the process described by Kling et al. (2005). A few of the HRUs had to be split due to overlaying of subwatershed boundaries within each 8-digit watershed, which resulted in a final number of 321 HRUs. The SWAT simulations, including the input and output data, were managed with the interactive SWAT (i_SWAT) software (fig. 2). A single Access database is used to manage both the input and output data of each SWAT simulation within i_SWAT. Once the input data have been constructed, the SWAT simulation can be executed within i_SWAT. Output data for each simulation are scanned from the standard SWAT output files and also stored in the database. A more detailed description of i_SWAT is given in Gassman et al. (2003); further documentation and software downloads are provided by Campbell (2006).

SWAT Calibration and Validation

Measured data collected at a sampling site located at Van Meter (fig. 1) were used for calibration and validation in this study; approximately 95% of the entire watershed drains to this location. An extensive amount of measured data has been collected at this location, especially for flow, suspended sediment, and nitrate. Daily USGS streamflow data (USGS, 2006b) were obtained for station 05484500 at Van Meter for the period 1981-2003. Water quality data, including suspended sediment, nitrate, organic N, organic P, and soluble orthophosate P (referred to as mineral P for the SWAT output), for the Raccoon River at Van Meter were obtained from the Des Moines River Water Quality Network, as described by Lutz (2005). These samples were collected on a weekly or biweekly basis and were analyzed by the Analytical Services Laboratory at Iowa State University. Suspended sediment and nitrate data were available for the entire period 1981-2003, but organic N, organic P, and mineral P data were available only from May of 2000 to December 2003.

Grab samples of water quality data were extrapolated into continuous monthly data using the USGS Load Estimator (LOADEST) regression model (Runkel et al., 2004). LOADEST estimates constituent loads in streams and rivers by developing a regression model, given a time series of streamflow, constituent concentration, and additional data inputs. LOADEST is based on two previous models: LOADEST2 (Crawford, 1996) and ESTIMATOR (Cohn et al., 1989). The model is well documented and is accepted as a valid means of calculating annual solute load from a limited number of water quality measurements. However, the load estimation process of the model is complicated by the same problems experienced with other approaches; e.g., retransformation bias, data censoring, and non-normality. For example, Ferguson (1986) reported that the rating curve estimates of instantaneous load were biased and may have underestimated the true load by as much as 50%.

SWAT was executed for a total simulation period of 23 years, which included 1981-1992 as the calibration period and 1993-2003 as the validation period. The simulated outputs at the outlets of subwatersheds 15 and 23 were summed together, and the resulting values were compared with the measured data at Van Meter (fig. 1) to perform the SWAT calibration and validation. Parameter adjustment was performed only during the calibration period; the validation process was performed by simply executing the model for the different time period using the previously calibrated input parameters. The calibration process was performed manually by adjusting key hydrologic, sediment, and nutrient related parameters (described below), including several suggested by Neitsch et al. (2002a), Santhi et al. (2001), Santhi et al. (2006), and Green et al. (2006), and then comparing model output with measured data. Santhi et al. (2006) pointed out that there is no formal optimization procedure that can be

Table 2. SWAT calibration parameters and their final values for the Raccoon River watershed.

	SWAT Calibration Parameter[a]		Final Calibrated Value
Streamflow Calibration	Curve number	CN2	−6.0 (change in values)[b]
	Soil available water capacity	SOL_AWC	−0.02 (change in values)
	Evaporation compensation coefficient	ESCO	0.85
	Revap coefficient	REVAP	0.02
	Groundwater delay	GW_DELAY	60 days
	Groundwater recession coefficient	GW_ALPHA	0.2
	Snowfall temperature	SFTMP	1.0°C
	Snowmelt base temperature	SMTMP	−1.0°C
	Melt factor for snow on 21 June	SMFMX	2.5 mm H_2O °C-day^{-1}
	Melt factor for snow on 21 December	SMFMN	2.5 mm H_2O °C-day^{-1}
	Surface runoff lag coefficient	SURLAG	1
Sediment Calibration	Linear components	SPCON	0.0004
	Exponent component	SPEXP	2.5
	Channel cover factor	CH_COV	0.5
Nutrient Calibration	Initial organic nitrogen	SOL_ORGN	1200 mg kg^{-1}
	Initial organic phosphorus	SOL_ORGP	240 mg kg^{-1}
	Initial mineral phosphorus	SOL_SOLP	1 mg kg^{-1}
	Biological mixing efficiency	BIOMIX	0.3
	Nitrogen percolation coefficient	NPERCO	0.20
	Phosphorus percolation coefficient	PPERCO	10
	Phosphorus soil partitioning coefficient	PHOSKD	100
	Residue decomposition factor	RSDCO	0.05

[a] Adjustments of CN2, SOL_AWC, ESCO, REVAP, GW_DELAY, and GW_ALPHA occurred at the HRU level; the other variables were adjusted at the watershed level (i.e., the same value was used across the entire watershed).

[b] For example, an initial CN2 value of 78 was reduced to 72 following calibration; each curve number was reduced from standard table values as reported in USDA-NRCS (2004) for row crop (straight row, contoured, or terraced and contoured), pasture, forest, and urban land use categories.

used to calibrate SWAT. Thus, subjective decisions are inherent in calibrating the model.

The calibration process was initiated by calibrating the water balance and streamflow for average annual conditions. Baseflow is an important component of the overall streamflow and had to be calibrated before the model was subsequently calibrated for stream flow and other components. An automated digital filter technique (Arnold and Allen, 1999) was used to separate baseflow from the measured streamflow, which resulted in a baseflow estimate of about 58% of the total streamflow on an average annual basis for 1981-2003. A similar ratio of 54% was found for the period 1972-2000 by Schilling and Zhang (2004) for the RRW using an automated hydrograph separation program developed by Sloto and Crouse (1996). The streamflow calibration process was then completed by varying several SWAT hydrologic calibration parameters within their acceptable ranges (table 2) to match the model-predicted baseflow fraction, average annual streamflow, and monthly streamflow time series with corresponding measured values. These parameters include the soil available water capacity (SOL_AWC), evaporation compensation coefficient (ESCO), groundwater delay (GW_DELAY), groundwater recession coefficient (GW_ALPHA), surface runoff lag coefficient (SURLAG), snow parameters, and the curve number (CN2).

Jacobs and Srinivasan (2005) cite Hawkins (1998), who stated that the standard CN2 tables should be viewed as guidelines and that specific curve numbers (and their empirical relationships) should be based on local and regional data. Adjustment of the CN2 value and/or other components of the SCS curve number method have been reported to be necessary to more accurately simulate runoff for ten watersheds in Texas (Jacobs and Srinivasan, 2005) and watersheds in west-central Florida (Trommer et al., 1996). Adjustment of CN2 values has also been found necessary for many SWAT studies, including Green et al. (2006), Santhi et al. (2001), Santhi et al. (2006), Qi and Grunwald (2005), and Stewart et al. (2006). Stewart et al. (2006) reduced the curve numbers by −8, which compares similarly to the reduction of −6 found for this study (table 2).

The streamflow calibration (1981-1992) and validation (1993-2003) periods were also used for assessing the accuracy of the SWAT sediment and nitrate predictions. However, only limited measured data for organic N, organic P, and mineral P were available for May 2000 to December 2003, which precluded any formal validation for those constituents. Sediment yield calibration was performed following completion of the flow calibration process. There are two sources of sediment in a SWAT simulation: loadings from the HRUs, and channel degradation/deposition. Model parameters such as the linear (SPCON) and exponential (SPEXP) components of the sediment transport equation, and the channel cover factor (CH_COV), were adjusted within their acceptable ranges to match simulated sediment loadings with the measured loadings (table 2). Several model parameters were also adjusted during the nutrient transport calibration process (table 2). These included the initial soil nutrient concentrations, biological mixing efficiency (BIOMIX), nitrogen percolation coefficient (NPERCO), phosphorus percolation coefficient (PPERCO), phosphorus soil partitioning coefficient (PHOSKD), and residue decomposition factor (RSDCO).

The model predictions were evaluated for both the calibration and validation periods using graphical comparisons and two statistical measures: the coefficient of determination (R^2), and the Nash-Sutcliffe simulation efficiency (E), developed by Nash and Sutcliffe (1970). The R^2 value is an indicator of the strength of the relationship between the measured and simulated values. The E value measures how well the simulated values agree with the measured values. The model prediction is considered unacceptable if the R^2 values are close to zero and the *E* values are less than or close to zero. If the values equal one, the model predictions are considered perfect. Coffey et al. (2004) discuss the fact that explicit statistical criteria have not been established for judging model results. However, Moriasi et al. (2006) have proposed several statistical standards for assessing simulation result accuracy, including a minimum value of 0.5 for *E*. This proposed standard was viewed as a minimum goal for both the R^2 and *E* statistics in this study; ultimately, calibration was performed until the best achievable results were obtained based on the adjustments to the calibration parameters (table 2) that were described above.

Sensitivity Analysis

Sensitivity analyses were performed with the calibrated model by simulating long-term scenarios of land use and nutrient input changes. SWAT was first executed for a total of 23 years (1981-2003) to establish baseline average annual values for the flow and other water quality indicators, which form the basis of comparison for scenario results. The different scenarios were then executed for the same 23-year period. The sensitivity results were evaluated on the basis of model output at the overall watershed outlet (outlet of subwatershed 25 in fig. 1).

The first set of scenarios focused on taking cropland out of production, i.e., increasing the amount of CRP land in the RRW. Increasing the amount of CRP land in a watershed can be a very effective soil and water conservation practice, because cropland is usually converted into perennial grass, which results in reduced surface runoff and erosion. Five CRP scenarios were executed with SWAT runs that depicted successively increasing amounts of CRP land, which were selected as a function of the slopes of the HRUs (table 3).

The second set of scenarios was performed to assess the impacts of hypothetical increases or decreases in overall nutrient applications (both fertilizer and manure) to corn in the RRW, i.e., to assess the sensitivity of different nutrient application rates on nitrogen losses to the stream system and on crop yield. The scenarios reflected 10%, 20%, 30%, and 50% increases and in turn 10%, 20%, 30%, and 50% decreases in the nutrient application rates on corn, relative to the baseline application rates.

A final set of scenarios was performed to provide further insight into how the manure application rate assumptions impacted the total nutrient loadings predicted at the watershed outlet. Two manure application-related scenarios were performed using: (1) the same rates reported in table 1 but with no fertilizer applied to the areas that receive manure, and (2) the baseline fertilizer rates without manure applications for the cropland areas shown in table 1.

Table 3. Slope cutoffs and corresponding amounts of converted cropland for the CRP scenarios.

CRP Scenario	HRU Slope Cutoff (%)	Cropland Affected (%)
1	7	6
2	4	17
3	2	41
4	1	88
5	0	100

Results and Discussion

Calibration and Validation

The overall 1981-2003 average annual water balance components are shown in table 4. The total 1981-2003 annual average water yield was predicted to be about 235 mm, which consisted of 105 mm of surface runoff, 133 mm of baseflow (combined tile flow and groundwater flow), and transmission losses of 3 mm. The baseflow fraction was 56%, which was consistent with the baseflow separation model estimate of 58% and the value of 54% found by Schilling and Zhang (2004).

Tile flow was estimated to contribute an annual average flow of 21 mm, which was 15% of the overall baseflow estimate. Exact measurements of the tile flow contribution to the total RRW baseflow were not available to confirm the accuracy of this estimate. However, Green et al. (2006) report an overall water balance including tile flow for the 775 km^2 South Fork of the Iowa River watershed in north-central Iowa, which was estimated to be 80% tile-drained, using an experimental version of SWAT that contains improved tile drainage routine initially developed by Du et al. (2005). They found that the predicted 10-year average annual water yield was partitioned between 38 mm of surface runoff and 154 mm of baseflow (tile, lateral, and groundwater flow) and that the estimated tile flow of 136 mm was the dominant baseflow component. Although the extent of tiled cropland in this study was only 51% of the RRW, it is probable that the tile flow portion of the overall RRW baseflow estimate is underestimated and that the groundwater contribution is correspondingly overestimated, based on the results reported by Green et al. (2006) for the South Fork of the Iowa River watershed.

Figures 3 and 4 show graphical representations of the streamflow comparisons at Van Meter (fig. 1) on an annual and monthly basis, respectively. The graphical results indicate that SWAT accurately tracked the annual and monthly flow trends across both the calibration and validation periods. However, streamflow was clearly underpredicted in 1993, while the largest overpredictions occurred in 1986, 1991, and 1998. The measured annual average streamflow of 222 mm over the entire 1981-2003 period at Van Meter was closely matched by the corresponding predicted value of 227 mm. This was further confirmed by the strong R^2 and *E* statistics (table 5), which ranged from 0.87 to 0.97.

Figures 5 and 6 show the annual and monthly comparisons of measured and simulated sediment yields for the calibration and validation periods at Van Meter (fig. 1). The statistical evaluation revealed a strong correlation between the measured and simulated values, as indicated by the R^2 and *E*

Table 4. 1981-2003 average annual water balance components for the entire Raccoon River watershed.

Water Balance Component	Depth (mm)
Precipitation	840.2
Surface runoff	104.6
Groundwater (shallow aquifer) flow	112.5
Tile flow	21.2
Evapotranspiration	598.3
Transmission loss	3.3
Total water yield[a]	235.1

[a] Total water yield = surface runoff + groundwater flow + tile flow − transmission loss.

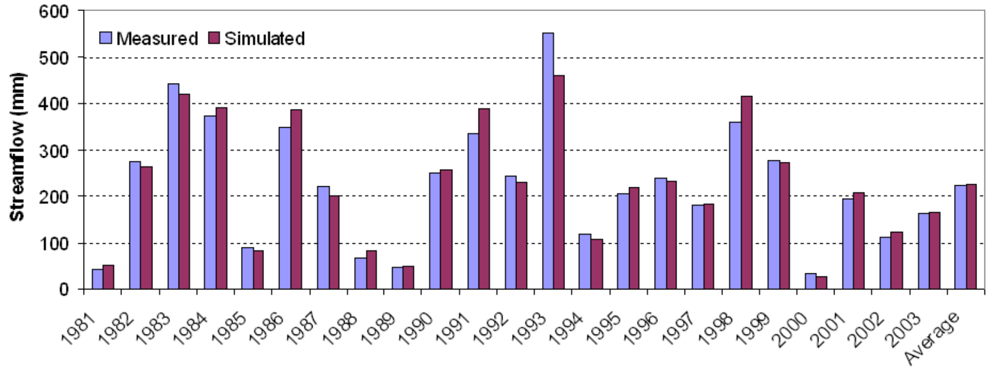

Figure 3. Annual flow calibration and validation for the Raccoon River watershed at Van Meter.

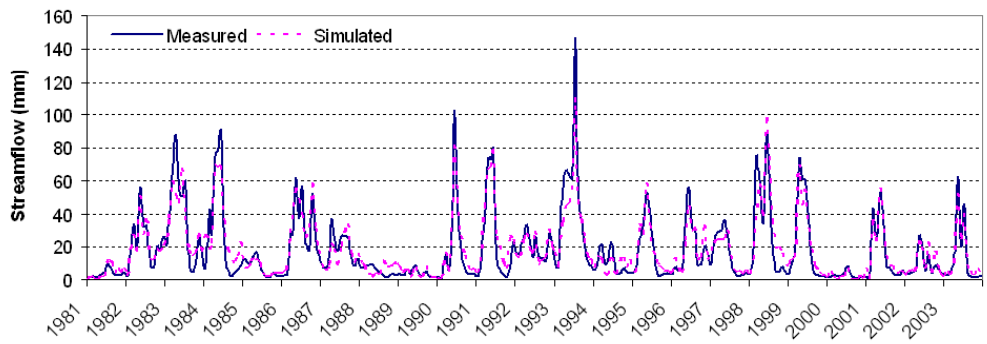

Figure 4. Monthly flow calibration and validation for the Raccoon River watershed at Van Meter.

Table 5. R^2 and *E* values of SWAT predictions versus measured data.

		Calibration (1981-1992)		Validation (1993-2003)	
Variable		R^2	*E*	R^2	*E*
Streamflow	Annual	0.97	0.97	0.94	0.94
	Monthly	0.87	0.87	0.89	0.88
Sediment	Annual	0.97	0.93	0.89	0.79
	Monthly	0.55	0.53	0.80	0.78
Nitrate	Annual	0.83	0.78	0.91	0.84
	Monthly	0.76	0.73	0.79	0.78

values (table 5) that ranged between 0.78 and 0.97, except for the monthly R^2 and *E* statistics during the calibration period, which were 0.55 and 0.53. The graphical comparison (fig. 6) indicates that the peak monthly sediment loads were less accurately predicted for the calibration period, which resulted in the weaker monthly results for the calibration period relative to the validation period. However, the monthly calibration statistics still meet the standard proposed by Moriasi et al. (2006). Overall, the model was able to simulate sediment yield with reasonable accuracy.

Figures 7 and 8 show the comparisons between the predicted and measured nitrate loads at Van Meter (fig. 1) across the calibration and validation periods. A strong correlation was observed in both the calibration and validation periods, as evidenced by the values of 0.73 to 0.91 that were computed for the R^2 and *E* statistics (table 5), indicating that the model predicted the nitrate loadings accurately. However, it is probable that some of the nitrate loss predicted via the groundwater flow portion of the baseflow should in fact have been simulated as nitrate loss through the subsurface tile drains.

Figure 9 shows that SWAT accurately replicated both the annual aggregate and monthly time-series of observed organic N values. Comparisons between the measured and

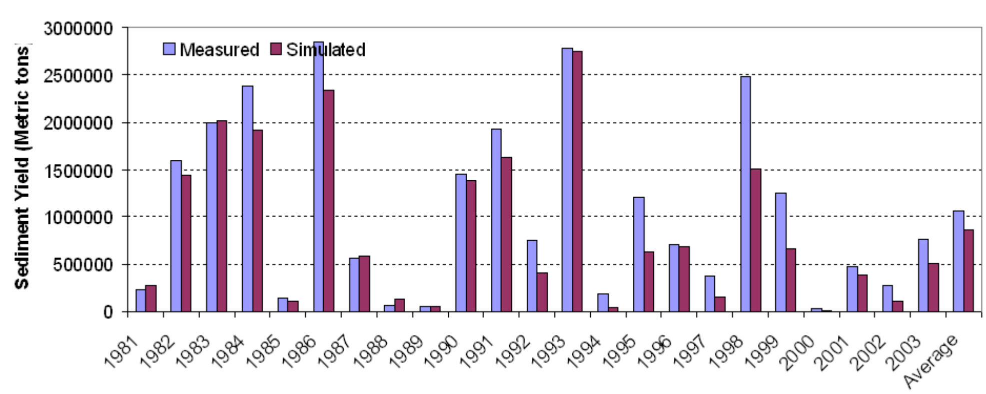

Figure 5. Annual sediment yield calibration and validation for the Raccoon River watershed at Van Meter.

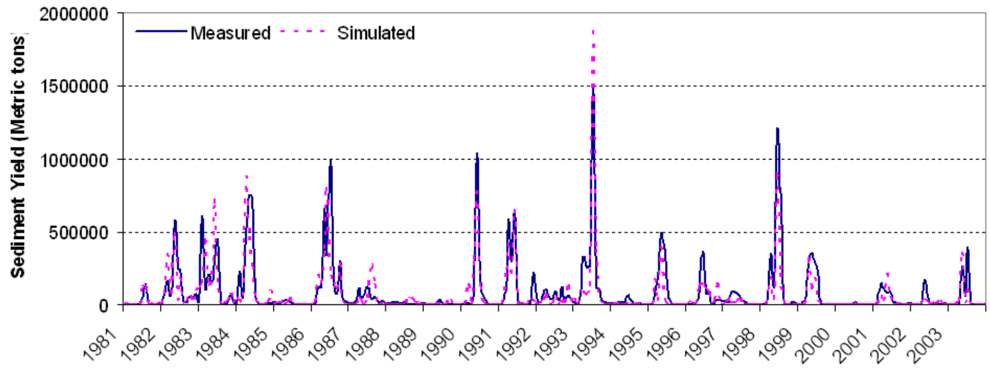

Figure 6. Monthly sediment yield calibration and validation for the Raccoon River watershed at Van Meter.

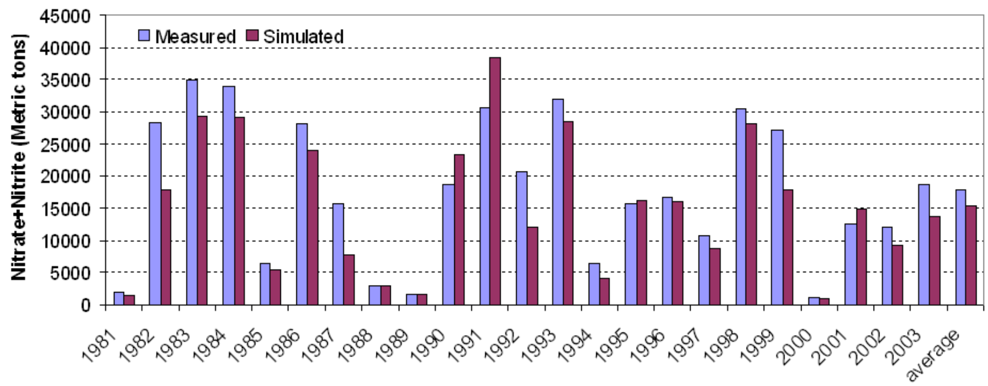

Figure 7. Annual nitrate loadings calibration and validation for the Raccoon River watershed at Van Meter.

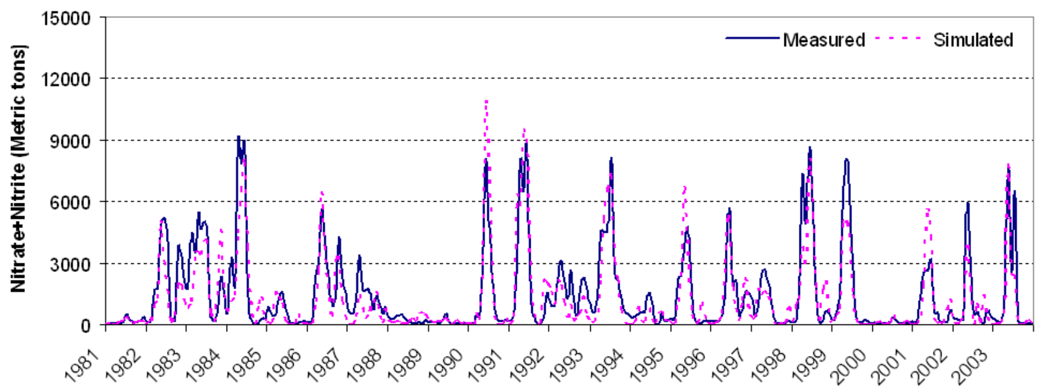

Figure 8. Monthly nitrate loadings calibration and validation for the Raccoon River watershed at Van Meter.

simulated organic N levels resulted in R^2 and *E* values of 0.80 and 0.79 on an annual basis, and 0.86 and 0.85 on a monthly basis. Similar comparisons are shown in figures 10 and 11 for organic P and mineral P, which reveal that SWAT tracked both indicators well. Organic P R^2 and *E* statistics were found to be 0.96 and 0.54, and 0.68 and 0.74, for comparisons between measured and simulated annual and monthly values, respectively. Similar corresponding values for mineral P were computed to be 0.92 and 0.51 on an annual basis, and 0.85 and 0.86 on a monthly basis. In general, the temporal patterns and statistics indicated that the predictions of organic N, organic P, and mineral P at Van Meter corresponded well with measured values. The annual R^2 values found for organic P and mineral P indicate a strong linear relationship between the measured and simulated loadings. However, the *E* values for the annual calibration for organic P and mineral P reveal less correspondence of measured values versus simulated values, although they do meet the criteria suggested by Moriasi et al. (2006).

Sensitivity Analysis Scenarios

The results of the five CRP land increase scenarios are shown in figure 12 for both sediment and nitrate losses at the overall watershed outlet. Sediment yield decreased as CRP land area increased, as expected. Nearly half of the predicted sediment decrease occurred within the first three increments of CRP conversion (41% of the total cropland), reflecting the impact of converting the land with high slopes from cropland into CRP, as shown in table 1. The maximum sediment reduction of 71% was achieved when all cropland was converted into CRP land. The increase in CRP land resulted in decreased surface runoff and hence erosion, but also

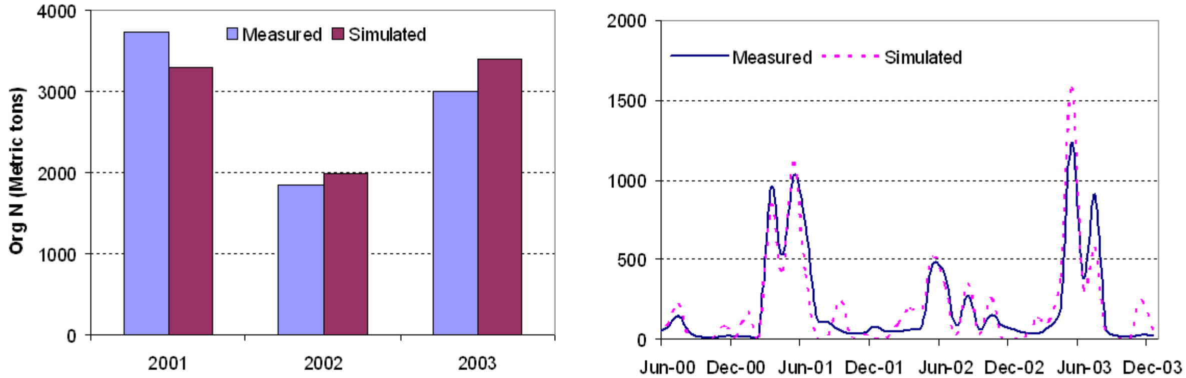

Figure 9. Annual and monthly organic N comparisons for the Raccoon River watershed at Van Meter.

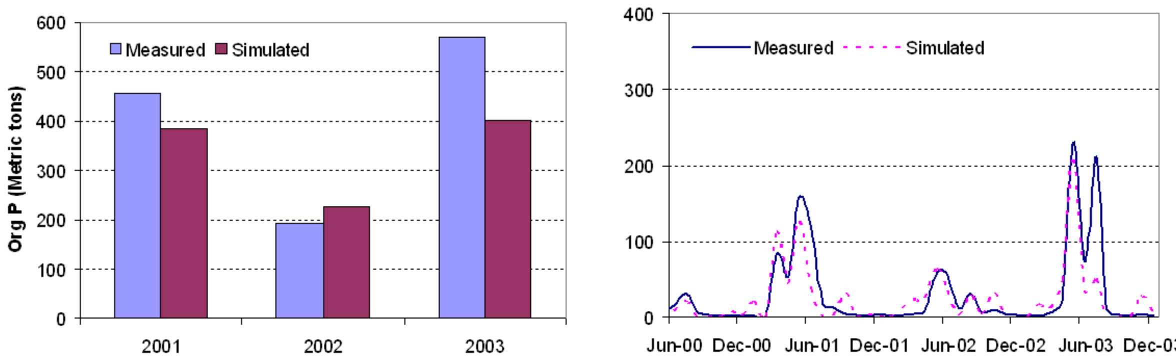

Figure 10. Annual and monthly organic P comparisons for the Raccoon River watershed at Van Meter.

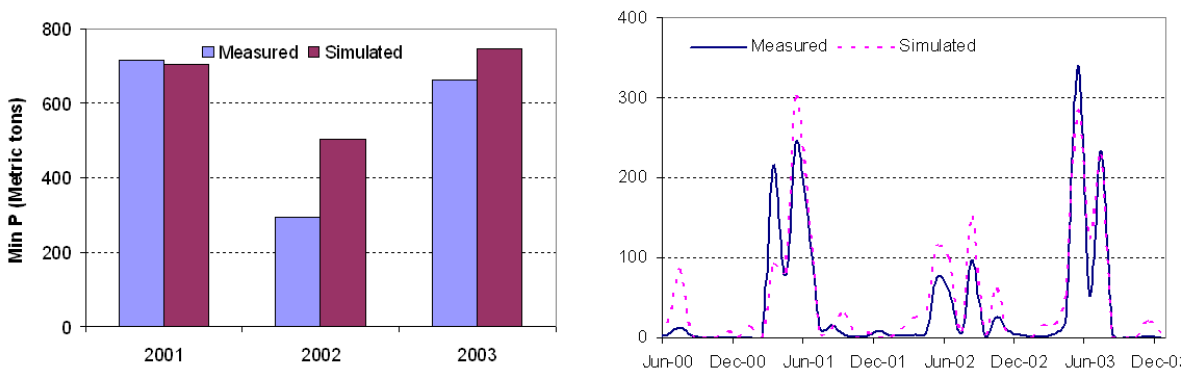

Figure 11. Annual and monthly mineral P comparisons for the Raccoon River watershed at Van Meter.

resulted in increased water movement to the groundwater. The predicted nitrate loadings also decreased, which again follows expectations because no fertilizer applications were applied on the CRP land. These results indicate that significant reductions in sediment and nutrient loadings can be achieved by converting cropland into CRP land.

Figure 13 shows the changes in nitrate loadings at the watershed outlet in response to the changes in nutrient application, as well as the impacts on mean corn yield across the entire RRW. As the application rates decreased, the nitrate loadings at the watershed outlet decreased, and vice versa. However, the predicted rate of change in nitrate loading is different for the decreasing application rates as compared to the increasing application rates. Decreases in the nutrient application rates of 10% and 50% resulted in approximately 12% and 50% reductions in the nitrate loadings at the RRW outlet. An increase in the nutrient application rates of 10% resulted in approximately the same relative impact as the 10% decrease, but a 50% increase in the nutrient application rates resulted in almost an 80% increase in nitrate loadings at the RRW outlet.

The yield impacts shown in figure 13 were determined as a function of the simulated average SWAT baseline corn yield, which was computed to be 7.1 t ha^{-1} across the entire RRW (on a dry weight basis). Measured yields that would provide direct comparisons with the SWAT-predicted yields were not available. However, average USDA-NASS yields previously computed for 1992-2002 (excluding 1993, which was an extreme wet year) for the two 8-digit watersheds that comprise the RRW were obtained to provide an approximate comparison between the SWAT output and measured yields (M. Nurmakhanova, Department of Economics, Iowa State University, Ames, Iowa; personal communication, 2006). An

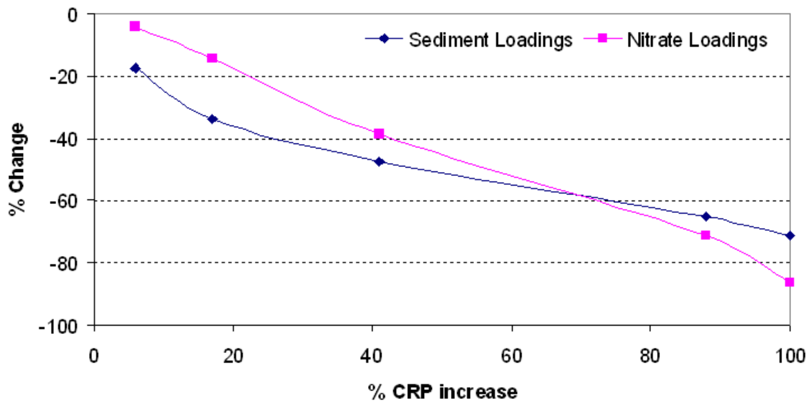

Figure 12. Reduction in sediment and nitrate loadings due to increase in CRP lands.

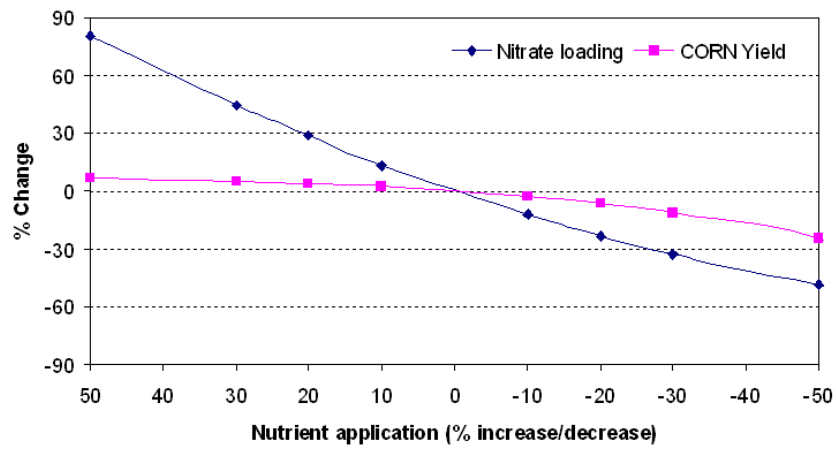

Figure 13. Effect of nutrient application in nitrate loadings and corn yield.

areal weighting approach was used to translate the county-level NASS yields to the 8-digit watershed level. The average NASS yields were determined to be 7.8 and 8.2 t ha^{-1} on a dry weight basis (assuming 15% moisture content at harvest) for the 07100007 and 07100006 8-digit watersheds (fig. 1), respectively. These measured yields were underestimated by the average SWAT-predicted yield by 9% and 13%.

The predicted corn yields were relatively insensitive to the majority of hypothetical nutrient application rate changes, as compared to the nitrate losses (fig. 12). The yield was estimated to decline by less than 10% for nutrient application rate reductions of up to 28%. In contrast, a 50% increase in the nutrient application rate was estimated to result in a yield increase of less than 10%.

Overall, the corn yield versus nitrate loading loss relationship suggests that decreases in RRW nitrate loadings can be achieved with minimal effects on crop yield with relatively low nitrogen application rate reductions (e.g., 10% to 20%). Conversely, minimal increases in corn yield were predicted for nitrogen application rate increases of up to 50%.

Table 6 shows the impacts of the two alternative manure-related scenarios on the predicted annual average nutrient loadings, relative to the observed loadings and the loadings estimated for the previously discussed baseline simulation. The largest predicted impacts were reductions of roughly 25% and 23% in the nitrate and mineral P loads for alternative scenario 1 (no application of manure), as compared to the baseline simulation. The nitrate load was predicted to decline by about 18% when fertilizer was not applied to the manured HRUs (alternative scenario 2), relative to the baseline. Other predicted impacts were generally minor; no impact was predicted for the organic P loadings for alternative scenario 2 because the P fertilizer consists only of inorganic P. These scenario results clearly reveal that the model is sensitive to the manure and fertilizer application rates that are assumed for the HRUs that are managed with manure.

Table 6. Comparison of average annual Raccoon River watershed nutrient loadings (t $year^{-1}$) between observed levels, the standard baseline simulation, and two alternative manure management scenarios.

Scenario or Observed	Nitrate (1981-2003)	Organic N (2001-2003)	Organic P (2001-2003)	Mineral P (2001-2003)
Observed levels	17,743	2,863	406	556
Standard baseline simulation	15,898	3,189	387	726
Scenario 1: Manure not applied	12,068	3,100	344	557
Scenario 2: Fertilizer not applied to manured HRUs	13,055	3,094	387	721

Input and Simulation Uncertainty

Shirmohammadi et al. (2006) provide an in-depth discussion of modeling uncertainty, particularly in the context of TMDL applications. They list several sources of modeling uncertainty inherent in a modeling process, including model input data, model algorithms, measured data used for calibration and validation, and simulation scale. The RRW study described here underscores the reality of these uncertainties, including management input assumptions such as the distribution of tile drainage, fertilizer application rates and timing, and manure application rates and areas.

The 1992 NRI is the only data source available at present that provides an estimate of the regional distribution of tile-drained lands in the RRW, and in most other major watersheds in the Upper Midwest. The fact that collection of tile-drained acreage was discontinued for the 1997 NRI is an indicator of the difficulty of accurately estimating such areas, and that there may be large error in the 1992 estimates. Anecdotal evidence also suggests that tile-drained areas have increased across much of Iowa during the past 15 years, most of which would not have been accounted for in the 1992 NRI. Thus, there is clearly a need for improved estimates of the percentage of tile-drained cropland in the RRW and in other heavily tile-drained regions, especially considering the fact that tile drains are a key conduit of nitrate to surface water. Green et al. (2006) estimated the percentage of cropland that is tile-drained in the South Fork of the Iowa River watershed based on the presence of poorly drained soils and other factors. A similar approach may prove useful for future RRW simulation studies.

The use of a single spring nitrogen fertilizer application rate in this study is obviously an oversimplification of the actual distribution of nitrogen application rates used across the RRW. However, it is a major challenge to establish what the true distribution is and the associated areas that the different application rates are applied to. A survey of RRW producers could provide valuable insight regarding fertilizer management in the watershed, but such an approach would still likely not capture the full range of fertilizer input practices. Further investigation is needed to identify any other potential sources of data that can provide improved understanding of fertilizer practices in the watershed. An even greater range of manure application rates probably exists in the RRW. As stated previously, the manure application assumptions used in this study reflect “worst-case conditions” of no manure crediting and corresponding fertilizer use at rates similar to surrounding non-manured fields. Such intensive nutrient input areas likely exist in the RRW, and establishing their location would be very useful because they can potentially be much higher sources of nutrient losses. However, identification of these fields, as well as the full range of manure management application rates, is again very difficult. Improved manure application assumptions can be developed by using IDNR confined animal feeding operation (CAFO) inventory data and associated manure management plan information. A survey approach could also provide additional insight into typical RRW manure management practices, such as was reported by Gassman et al. (2002). Simulating a distribution of manure and/or fertilizer application rates for the watershed would further yield important quantification of the uncertainty of these practices.

Other aspects of uncertainty should also be further investigated in future RRW and similar studies, including determining the relative contribution of upland areas versus stream channels to in-stream sediment loads. To date, few studies have been conducted that provide a quantitative estimate of these two sources of suspended sediment. Thoma et al. (2005) found that 23% to 56% of in-stream sediment load could be attributed to bank erosion in their analysis of sediment sources for the Blue Earth River in southern Minnesota. Better estimates of in-stream sediment sources could provide a more accurate accounting of land management needs, as suggested by Thoma et al. (2005).

Conclusions

Simulated output generated with the SWAT model were compared with measured data at the assumed outlet (Van Meter, Iowa) of the Raccoon River watershed, for both calibration (1981-1992) and validation (1993-2003) periods. The R^2 and E values (>0.7 in most cases) indicated that the model was able to replicate annual and monthly streamflow, sediment, nitrate, organic N, organic P, and mineral P with reasonable accuracy. The calibrated model was used to study the effects of CRP land and nutrient application on sediment and nutrient loadings. The results show that the sediment and nutrient loadings at the watershed outlet can be significantly reduced by converting cropland to CRP land. Similarly, reductions in nutrient fertilizer application rates were predicted to have a significant effect on reducing nitrate loadings at the watershed without affecting crop yield significantly. Conversely, increases in application rates were also predicted to have minor impacts on corn yields but resulted in sizeable increases in nitrate loadings. The results were also found to be sensitive to the simulated manure application and fertilizer rates, for those HRUs that were assumed to receive manure applications. Further research is needed to confirm whether the impacts of different nutrient application rates on nitrate loading losses at the watershed outlet and corresponding crop yields are consistent with measured data.

The overall results of this study also point to the importance of accurate input data. Future simulation work for the Raccoon River watershed should incorporate improved estimates of fertilizer and manure nutrient inputs and associated application rates, including total manure estimates based on updated livestock inventory data available from the IDNR. There is also a need to obtain better estimates of the percentage of cropland that is tile-drained, if such data are available. Further, there is a need to more clearly understand how much of the in-stream sediment load is being contributed from the stream channels relative to upland contributions. Also, additional in-stream flow and pollutant loss monitoring data available for the North and South Forks of the Raccoon River (K. E. Schilling, Iowa Department of Natural Resources, Iowa City, Iowa; personal communication) should be included for a more comprehensive validation of the model. This would allow for testing of SWAT at multiple points within the RRW, which has been performed in several previous modeling studies (e.g., Borah et al., 2002; Santhi et al., 2001; Santhi et al., 2006; Qi and Grunwald, 2005).

Finally, the results of this study also underscore the need to perform further simulation research for the Raccoon River watershed with the recently released SWAT2005, which contains several enhancements, including the previously discussed improved tile drainage component developed by Du et al. (2005) and applied by Green et al. (2006). This will allow more accurate simulation of flow and nitrate discharge via subsurface tiles, which would be expected to result in overall improved simulation results.

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