Evaluation of Best Management Practices (BMPs) in Impaired Watersheds Using the SWAT Model

A report by:

Solomon Folle

Brent Dalzell

& David Mulla

Department of Soil, Water and Climate
University of Minnesota

Project Manager:

Adam Birr

Funded by:

2007 Minnesota Department of Agriculture (MDA) Clean Water Legacy Fund

The SWAT model

The model selected for this work is the Soil and Water Assessment Tool (SWAT). SWAT is a watershed-scale model that functions on a daily time step; it is primarily applied to predict and evaluate long-term land cover and land management practices on the quantity and quality of water that is exported from watersheds with agricultural land use. The model is physically-based and relies on environmental parameters and plant growth to estimate the amount of water available in the landscape to contribute to stream flow and delivery of sediment, nutrients, and pesticides to the watershed outlet. The SWAT model was selected for this work because it is freely available, it has a large user base and is actively being supported and developed. Further, it has a great degree of flexibility to allow simulation and evaluation of a wide variety of alternative crops and land management practices. SWAT has been used widely for the study of water quality in agricultural regions and has been applied to TMDL studies. For a comprehensive review of the SWAT model in the scientific literature, please refer to Gassman et al., 2007.

Despite the strengths of the SWAT model, it is important to note model weaknesses:

- There is no routing of flow and pollutants with in a sub-watershed (i.e., routing between HRUs is currently not possible).
- The SWAT model does not simulate non-field sources of sediment such as erosion of streambanks, ravines and/or bluffs. These can be more important than field sources in some watersheds.
- Targeted placement of BMPs like filter strips, grassed water ways, riparian buffer zones, wetlands, grassland or other land use within a given sub-watershed is not possible.
- The model does not contain routines for concentrated animal feeding operations .
- Stream channel degradation and sediment deposition routines are simplified and still under development and testing.
- The tile drainage routines in SWAT are based on empirical parameters related to timing of field drainage and they do not explicitly account for the spacing of tile drains or depth of shallow water table.

In addition to these specific weaknesses of the SWAT model, it is important to highlight more general shortcomings to watershed modeling approaches that are not model-specific. These weaknesses are generally the result of the tradeoff that occurs between having a field study that can be detailed but limited in space and time vs. applying general principles to a broader geographic area in order to estimate and predict landscape-scale fluxes of water, sediment, and nutrients.

Quality of input data: The SWAT model relies heavily on accurate precipitation data in order to satisfy demands for plant growth and predict runoff and infiltration from precipitation events. In the upper Midwest, summertime convective thunderstorms can generate a lot of precipitation over a small area that can be important in generating runoff and erosion. The inclusion or exclusion of these rainfall events can result in spurious model results, even though the model may be simulating processes accurately. Ideally, each study watershed would be fitted with a network of multiple rain gauges to capture this variability. In reality, it is often necessary to rely on a small number of rain gauges to interpolate rainfall estimates for the entire watershed.

Simulation of actual watershed conditions: In large watersheds with multiple landowners and land uses, it is prohibitive to accurately simulate all of the management practices present in the watershed. In order to develop and calibrate the model for a watershed, a suite of typical land management practices are developed based on local knowledge and they are applied uniformly to the watershed. For example, crop planting and harvest dates are based on information available in weekly crop reports and an average value is used to guide the model. In reality, these practices are distributed over multiple days/weeks by different farmers throughout the watershed. Similar simplifications are used for crop rotations as well as the fertilizer application (timing and rate) and tillage (timing and implement). In sum, these simplifications do not permit the model user to accurately predict behavior of a specific field (this would not be an appropriate use of results from a watershed-scale model in most circumstances). These simplified average conditions are appropriate and useful when they are used to predict how the watershed will respond to broad shifts in management practices averaged over long time periods.

How to use model results

Given the complexity of the SWAT model and the level of detail available in the model outputs, it can be tempting to rely wholly on data provided by the model. It is important to not over-interpret model results. Rather, output from model scenarios it most valuable when applied in the following ways:

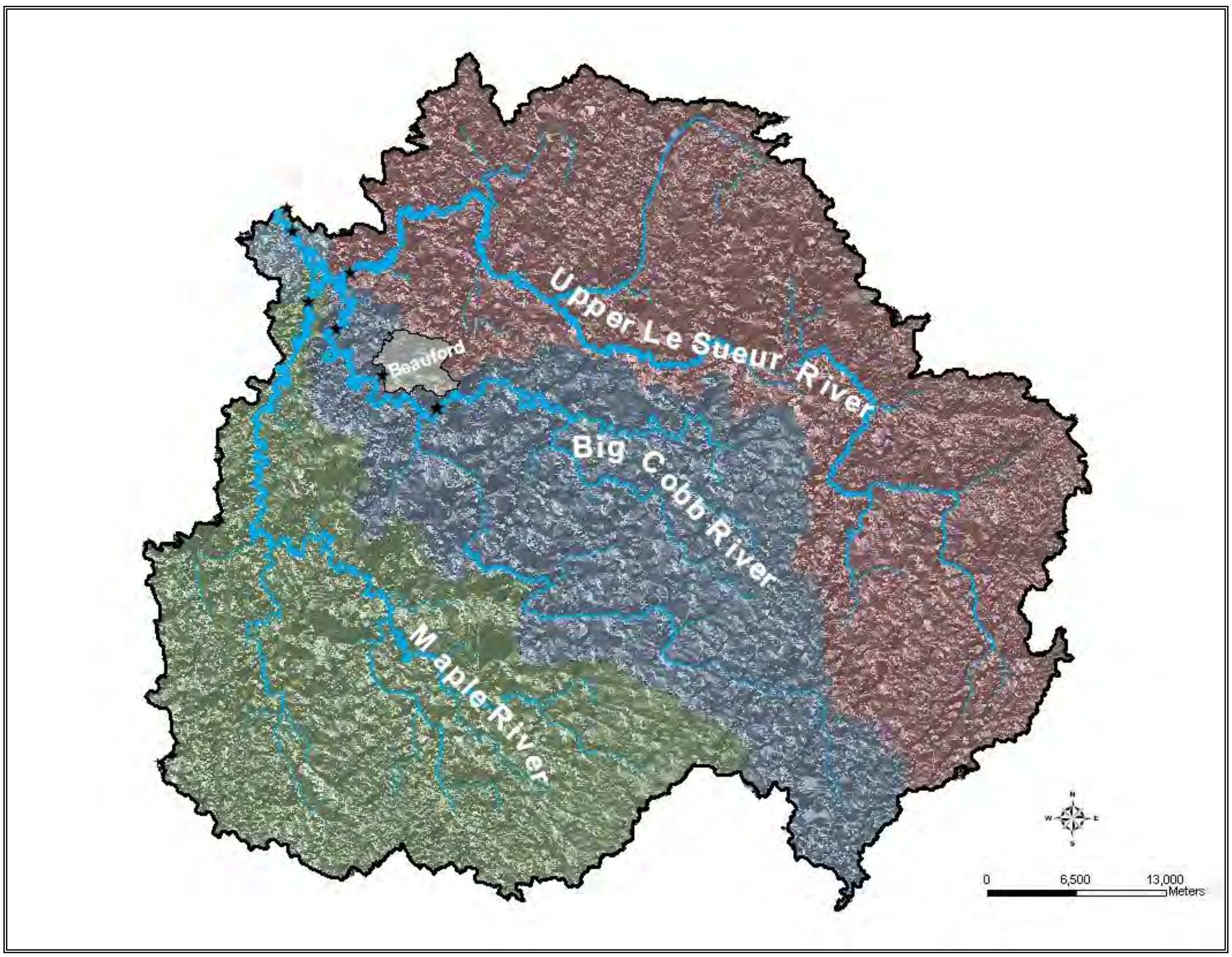
1) **Models allow evaluation of the <u>relative</u> effectiveness of alternative management scenarios.** Model results become easier to interpret when only one or two parameters have been changed between simulations. In this manner, the SWAT model (or any watershed model) is very useful in evaluating the relative effectiveness of alternative management scenarios. Less emphasis should be given to the absolute concentration or loading of a pollutant and more attention should be placed on the relative effectiveness of one management scenario over another. In this manner, it becomes feasible to employ the model results as a way to identify candidates for alternative management practices that should be more effective in the study watershed.
2) **Model results should be compared to experimental (i.e., field and laboratory) data when possible.** Actual measurements of water quality parameters in the field can be expensive, and time-consuming, but they provide the best way to assess the effects of management practices on water quality parameters (assuming that the experiments were adequately designed and the measurements were properly taken). The resource-intensive process of field based measurements provides a good means by which to determine if model results are within the realm of reason and help to provide an important context in which model results can be meaningfully evaluated. In scenarios where model results agree with field- and lab-based studies, then model results become more meaningful and can be interpreted and applied with greater confidence. In contrast, if the model results do not agree with measured data, this can provide substantial insight to help identify areas where the model does not perform well. In these cases, disagreement between model data and field- or lab-based data can help to identify areas where model results should be treated with greater skepticism and applied only with necessary caveats (or used to focus efforts for additional field studies and future model improvement).
3) **Model simulations and alternative management scenarios are most helpful when coupled with stakeholder input and expert judgment**. The simple fact that a

management scenario can be simulated in a model does not mean that it is a suitable to achieve water quality goals in a particular watershed. Other factors must be taken into consideration in order to ensure that alternative scenarios are meaningful. For example, it may be possible to simulate practices that involve changing the timing or location of manure application, however, manure storage facilities and logistics associated with manure transportation are likely to preclude these practices in most areas. In those cases, the model may be able to simulate the practice but the likelihood that actual management practices will change are slim. By relying on expert knowledge from local landowners and local and state agency representatives, more meaningful management scenarios can be developed that will be in a better position to provide useful information and guide water quality improvement practices.

PART 1: SEDIMENT, NUTRIENTS AND PESTICIDES MODELING IN THE LE SUEUR RIVER WATERSHED, SOUTH-CENTRAL MINNESOTA

Solomon M. Folle and David J. Mulla
Department of Soil, Water and Climate
University of Minnesota

Submitted to: Minnesota Department of Agriculture



September 2009, Minnesota

Table of Contents

Part 1: Sediment, nutrients and pesticides modeling in the LeSueur River watershed, south-central Minnesota.

List of Tables: Part 1

List of Tables: Part 2

List of Figures: Part 1

List of Figures: Part 2

1. Introduction

The Minnesota River Basin drains about 4.4 million ha of productive agricultural land that covers 20 percent of Minnesota. The Minnesota River flows 539 km from its source, and enters the Mississippi River in St. Paul, where its discharge represents nearly half of the Mississippi River flow. The Minnesota River has several water quality concerns, including excess sediment, nutrients (phosphorus and nitrogen) and pathogens (bacteria and viruses). The river often violates federal standards for water quality, and contaminants from the Minnesota River enter the Mississippi River and subsequently flow to Lake Pepin, where fish kills and severe nuisance algal blooms have been reported. The twelve major watersheds of the Minnesota River Basin differ greatly in the amount of contaminants they transport. The Le Sueur River Watershed transports a disproportionately high load of sediment to the Minnesota River.

The Le Sueur River Watershed covers a total area of about 2,850 square km (Figure 1), which represents 7% of the area in the Minnesota River Basin. According to the estimates of Mulla (1997) and data from Minnesota State University at Mankato, this watershed contributes 53% of the sediment load, 20% of the nitrate-nitrogen load, 31% of the phosphorus load and a large load of pesticides to the Minnesota River Basin. The MPCA listed the Le Sueur River as an impaired water body in 2006. Thus, it is imperative to conduct research that can contribute towards mitigating the contaminant loads from this watershed. For this purpose, the Soil and Water Assessment Tool (SWAT) model developed by USDA-ARS was selected to study the watershed with the following basic objectives

- To investigate and test applicability of the SWAT hydrologic model under the climate, farming systems, hydrologic and physiographic conditions of Minnesota
- To accurately and efficiently quantify sediment, nutrient (nitrate-nitrogen, phosphorus) and pesticide (atrazine, acetochlor and metolachlor) losses from the watershed

- To identify and prioritize critical sub-watersheds and to evaluate the relative importance of managing them
- To evaluate the effectiveness of alternative best management practices (BMPs) at reducing pollutant loads from the Le Sueur River watershed

2. The Study Area

The study was conducted in the Le Sueur River Watershed (LRW), which is designated by an 8-Digit Hydrologic Unit Code (HUC) 7020011. It is located in the south central Minnesota covering a total area of about 2,850 square km in the counties of Blue Earth (33%), Waseca (31.8%), Faribault (22%), Freeborn (9.7%), Steele (3.2%), and Le Sueur (0.3%).

The population in the LRW is about 56,100. Agriculture is the primary land use of the watershed, accounting for approximately 87% of the available acres. A two-year corn/soybean rotation comprises approximately 93% of cropped lands within the watershed; small grains, hay, grasslands, and lands enrolled in the Conservation Reserve Program (CRP) make up the rest. There are about 1.1 million livestock in the LRW, of which 59.3% are swine, 28.2% turkey, 4.2% beef and dairy cattle and 8.3% chicken and other animals (USDA, 2008).

The Le Sueur Watershed has a continental climate with cold dry winters and warm wet summers. Based on the long term weather averages (from 1971 to 2000) recorded at the Southern Research and Outreach center of Waseca, the average monthly temperatures range from 11^0 F in January to 71^0 F in July. The average annual precipitation of LRW ranges from 737 mm to 838 mm.

Elevation in the watershed ranges from 233 to 418 m a.s.l. The highest values are located in the eastern and southeastern portions of the watershed, while the lowest are found across the central and western parts towards its outlet. Flat landscape (0-2% slope) covers 80% of the total watershed area (Figure 21).

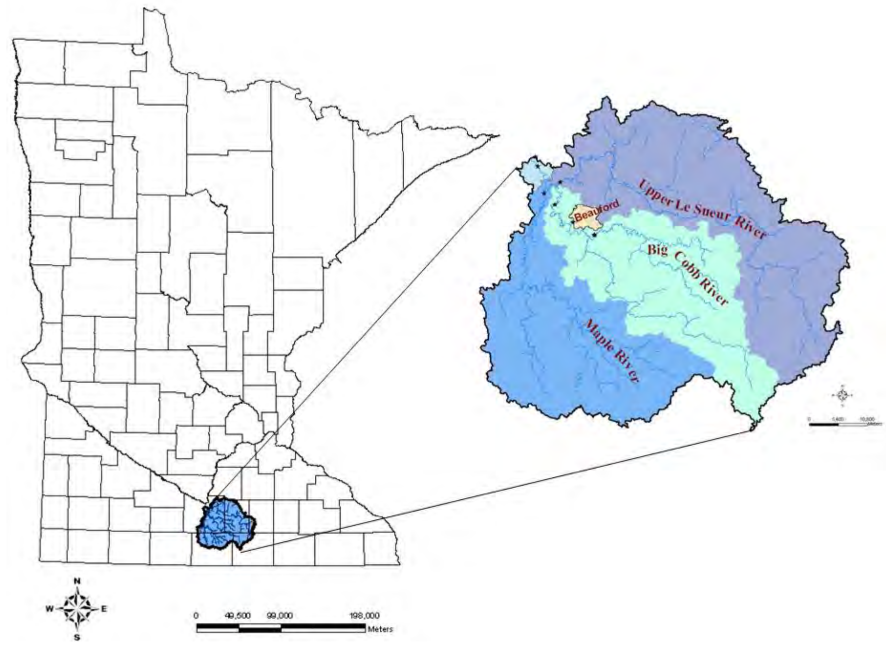


Figure 1: Location Map of Le Sueur River Watershed

Table 1: Distribution of Slope Steepness in the Le Sueur Watershed

Percent Slope	**Area [ha]**	**% Watershed Area**
0-2	231,254	80.3
2-6	48,042	16.7
6-12	6,826	2.3
> 12	1,901	0.7
TOTAL	**288,023**	**100**

The Le Sueur River Major Watershed drainage network is defined by the Le Sueur River and its major tributaries of the Maple River, Big Cobb River, the Upper Le Sueur River, several smaller streams, public and private drainage systems, lakes, and wetlands. The watershed has a stream network 1,933 km long, of which 41% is perennial (Minnesota State University, 2000). The streams of LRW flow through gentle landscapes at the headwaters and then they flow in very deep incised stream channels near the outlet.

The MPCA 2008 list of impaired water bodies included the Le Sueur River and its tributaries of the Cobb River, Little Cobb River, Maple River, Rice Creek and Little Beauford ditch, and the Madison, Lura, Elysian and Bass lakes (Minnesota Pollution Control Agency, 2009). Thus, it is imperative to conduct research that can contribute towards mitigating the contaminant loads from this watershed. The Soil and Water Assessment Tool (SWAT) model developed by USDA-ARS was selected to study the impacts of sediment, nutrients and pesticides in the Le Sueur River watershed, and then to develop management strategies to reduce contaminant loads.

3. SWAT Model

The SWAT model is a distributed parameter model that operates on a daily time step so as to predict the impact of management measures on flow, sediment and agricultural chemical yields of watersheds (Neitsch et al., 2002). SWAT works on a continuous time scale to simulate long-term effects of management changes. Among the many advantages of this model are; it has incorporated several environmental processes, it uses readily available inputs, it is user friendly, it is physically based and distributed, and it is computationally efficient to operate on large basins in a reasonable time. Despite the strengths mentioned above SWAT model has some known weaknesses:

- There is no routing of flow and pollutants with in a sub-watershed
- Targeted placement of BMPs like filter strips, grassed water ways, riparian buffer zones, wetlands, grassland or other land use within a given sub-watershed is not possible
- No routines for concentrated animal feeding operation
- Simplified stream channel degradation and sediment deposition routines
- The tile drainage routine of SWAT does not account for the drain spacing and depth of shallow water table.

The Le Sueur River Watershed SWAT modeling project was developed using ARCSWAT version 2.1.3. Detailed sensitivity analysis, calibration and validation of the SWAT model was initially made at a relatively small landscape scale of the Beauford sub-watershed. After evaluating model performance at the scale of Beauford sub-watershed, the calibrated input parameters were transferred to the entire LRW modeling of hydrology, sediment and chemical pollutants. This modeling approach helped reduce the complexity of parameter optimization that comes with upscaling the model.

4. Model Input Data Organization

All the necessary spatial datasets and input database files for the LRW SWAT model were organized following the guidelines of Neitsch et al. (2004). USGS 30 m DEM, the most detailed soils data of SSURGO, crop land data (CLD) of the USDA for 2006 and the stream network from the Minnesota River Basin Data Center (MRBDC) were the GIS data layers used to build the model. (Table 3 & Figure 2).

The other SWAT model input database files used were farm management operations, stream water quality, point sources and weather data. Weather data for the LRW SWAT modeling was taken from nine different gauging stations of the MN State Climatology Office (Table 3 & Figure 3). The selected stations are distributed all over the watershed to effectively capture the spatial variability of the LRW precipitation. Measurements of stream flow and water quality recorded by USGS, MN Metropolitan Council, MPCA, DNR and MDA were used for model simulation, calibration and validation.

Tile drained lands in the LRW were identified based on the following assumptions:

- No subsurface tile drainage exists in crop land with slopes greater than 6%
- All crop land with slopes less than 2% are tile drained
- Crop land with slopes of 2-6% and hydrologic soil group “C” or “D” are tile drained

A two year rotation of corn and soybean was used as the baseline scenario, and as the framework for management operations over the simulation period.

5. Modeling Assumptions

The following basic assumptions have been used during the LRW SWAT modeling.

- Management operations including tillage, crop rotations, grazing, nutrient application, planting and harvest are assumed to happen on fixed dates. The model does not modify these dates based on precipitation events or on the annual weather condition (Table 2).

- It was assumed that manure applications are confined to the sub-basin that each feedlot was located in. The manure was applied to corn fields within a one mile radius of the feedlot.
- The animal population of 2006 is assumed to be representative during the entire modeling period, and is assumed to be constant over the calibration and validation periods.
- The land use area was assumed constant during the calibration and validation periods
- A two year rotation of corn and soybean is assumed to be the default scenario

Table 2: Baseline Scheduled Management Operations for Corn-Soybean Rotation

Year	Crop type	Management operation	Date
Year 1	Corn	- Secondary tillage Cultivation (Field Cultivator) - Planting corn - 18-46-00 @ 163 lb/ac - Metolachlor application (2.21 lb/ac) - Acetochlor application (1.6 lb/ac) - Atrazine application (0.59 lb/ac) - Harvest/kill - Primary Tillage (Chisel Plow)	April 28 May 1 May 1 May 3 April 29 April 29 Oct 20 Oct 25
Year 2	Soybean	- Secondary tillage Cultivation (Field Cultivator) - Planting soybean - Metolachlor (0.89 lb/ac) - Harvest/kill - Primary Tillage (Chisel Plow) - Swine fresh manure application - Broiler Fresh Manure - Dairy fresh manure - Anhydrous ammonia @ 120 lb/ac (injected)	May 12 May 15 May 14 Oct 7 Oct 12 Oct 30 April 24 Oct 30 Nov 1

Note:

- Acetochlor was applied to 35% of the corn acreage
- Atrazine was applied to 15% of the corn acreage
- Metolachlor was applied to 4% of corn and 1% of soy acreage

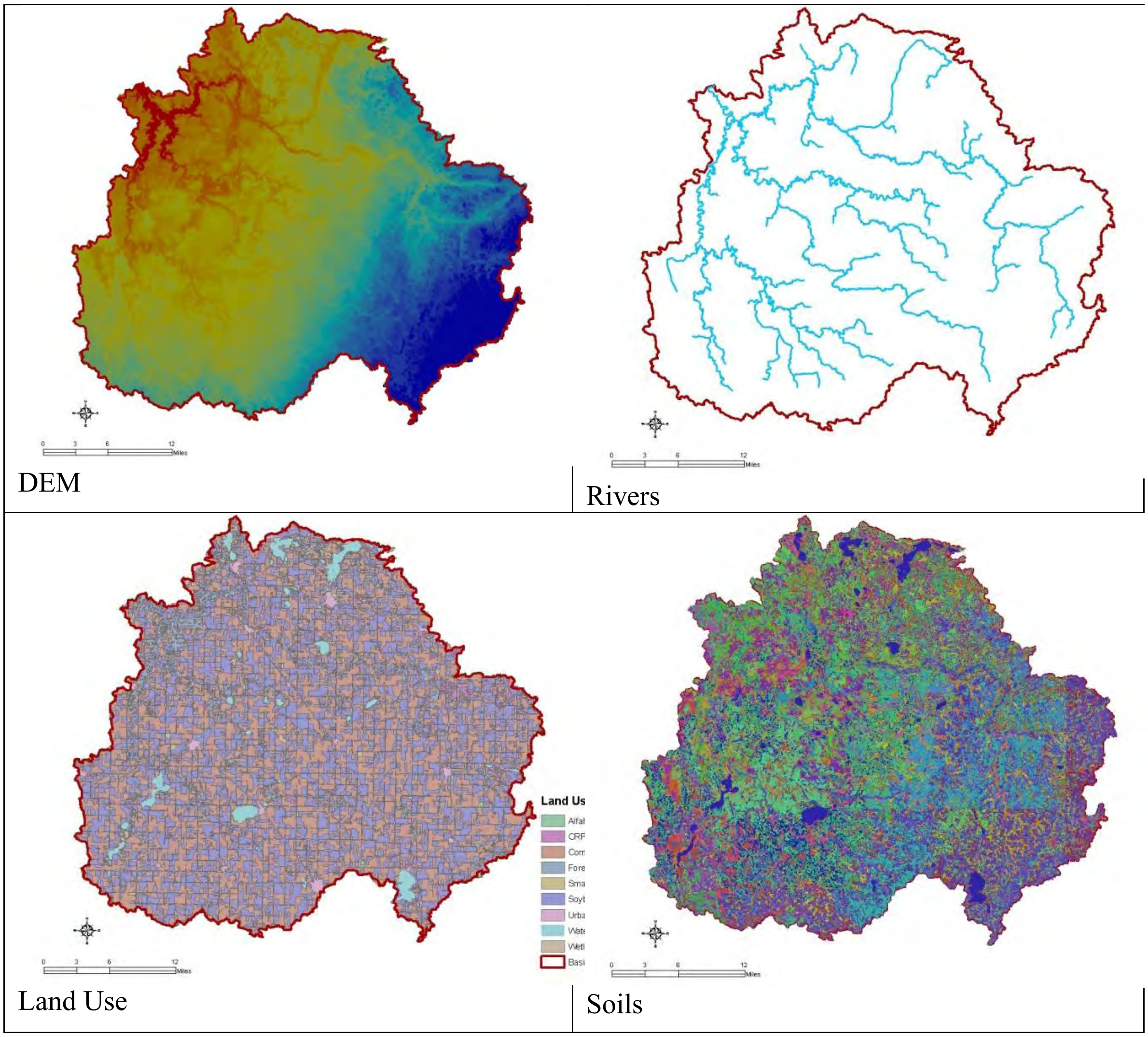


Figure 2: Input data layers to build the LRW SWAT Model

Table 3: Data sources for the Le Sueur Watershed SWAT modeling

Data Type	Source	
Digital Elevation Model (DEM)	http://seamless.usgs.gov	USGS
SSURGO soil	http://www.ftw.nrcs.usda.gov	USDA
Land use	http://datagateway.nrcs.usda.gov	USDA
Stream network	http://mrbdc.mnsu.edu/gis/lesueur	MRBDC
Weather	http://climate.umn.edu	U of M
Point Sources		MPCA

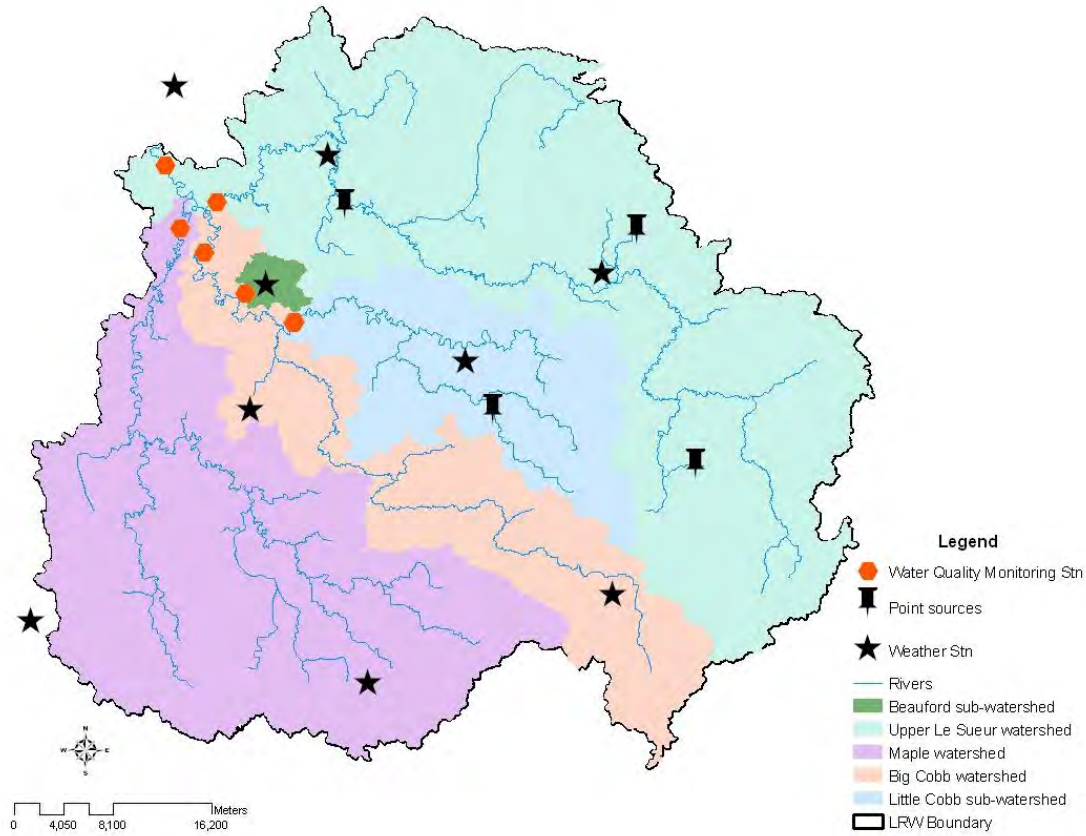


Figure 3: Monitoring Stations in LRW

6. Model configuration and Setup

The Le Sueur watershed was subdivided into a total of 84 sub-watersheds and 4,818 Hydrologic Response Unit (HRUs) based on a USGS 30 m DEM and Minnesota Department of Natural Resources (DNR) watershed subdivisions.

Weather data from 2000-2006 were taken from nine different gauging stations provided by the MN State Climatology Office. Measurements of stream flow and water quality recorded in 2006 by the USGS, MN Metropolitan Council, MPCA, DNR and MDA were used for model simulation, calibration and validation. LOADEST (Runkel et al., 2004) was used along with measured stream flow and water quality data to estimate sediment loads. Detailed USDA NASS crop land data (CLD) for the year 2006, SSURGO soils data from the USDA NRCS and the stream network from the Minnesota River Basin Data Center (MRBDC) were used to build the model. All necessary spatial datasets and database input files for the LRW SWAT model were organized following SWAT model guidelines (Neitsch et al., 2005). Evaluation of the model performance to appropriately predict the hydrology was evaluated through qualitative and quantitative measures involving both graphical comparisons and statistical tests of the Nash and Sutcliffe Efficiency goodness-of-fit. The same comparisons were performed during both the calibration and validation phases.

7. Model Evaluation Criteria

The LRW model performance was evaluated on the basis of test criteria recommended by (Moriasi et al., 2007). The Nash–Sutcliffe efficiency (NSE) statistic was used for model evaluation. The NSE indicates how well the plot of observed versus simulated data fits the 1:1 line (Nash and Sutcliffe, 1970). Model simulation results were considered "satisfactory" if NSE > 0.50. NSE is given by:

Where Yprd and Yobs are predicted and measured values of the dependent variable Y, respectively; and Yavg is mean of the measured values of Y.

8. LRW Critical Contributing Areas (CCAs)

The 84 sub-watersheds of the LRW, as delineated for the LRW SWAT modeling, have different amounts of discharge depending on their topographic location, soil types, proximity to streams, land use and land cover. The concept behind Critical Contributing Areas (CCAs) is that there are small localized saturation excess areas in each sub-watershed of the LRW that contribute high amounts of sediment and nutrients. Besides the improvement in the SWAT model capability to accurately simulate the runoff source areas in the LRW, the inclusion of CCAs is very important for cost effective implementation of CCA targeted best management practices. CCAs were identified based on GIS analysis of terrain attributes, proximity to streams and soil properties. Identification of these critical areas is essential for the effective and efficient implementation of LRW management programs.

About 29 % (559 ha) of the Beauford sub-watershed, encompassing 152 of the 357 HRUs were identified as CCA HRUs (Figure 4). In the case of LRW, 28% (79,868 ha) of the land area is in the CCAs.

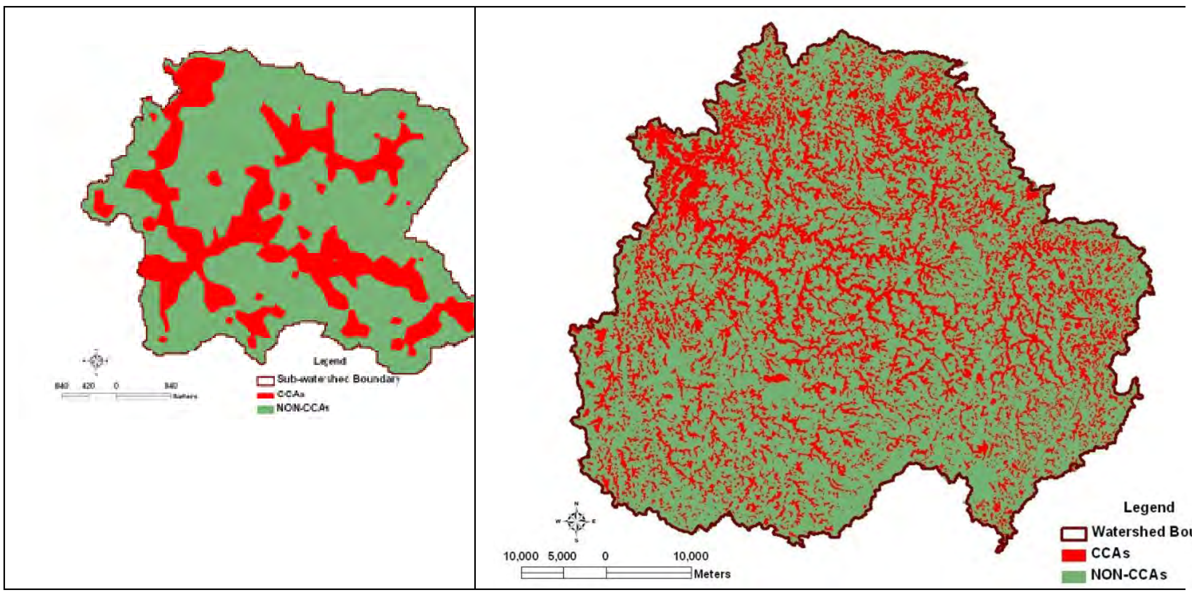


Figure 4: Critical Contributing Areas (CCAs) in Beauford Sub-watershed and LRW

9. Modeling Hydrology

The movement of ground and surface water and associated pollutants are governed by hydrology. Therefore, understanding hydrological processes is very important for assessing the environmental and economical well-being of the LRW and all receiving water bodies downstream . The Le Sueur River drains an area of 2,850 square km and has an annual mean flow of 21 m^3/sec. The three months of April to June are the peak flow seasons, and flow is reduced during the months of Dec-Jan.

There are a number of processes controlling the spatial and temporal variability of the LRW hydrology. In order to separate surface and subsurface hydrological processes, a sensitivity analysis of SWAT model input parameters was followed by baseflow separation.

9.1. Sensitivity Analysis of LRW Hydrology

A sensitivity analysis was used as a screening tool for reducing the number of parameters to be adjusted during calibration. This is further used in building and understanding the SWAT model. The sensitivity of different parameters is impacted by topography, geomorphology of the landscape, size of the watershed, land-use variations and human impacts. A summary of the twelve most sensitive parameters in the Beauford and Le Sueur watersheds is shown in Table 4.

Table 4: SWAT Model Sensitive Parameters for Hydrology

Parameter	UNIT	Description
Ch_K2	mm/hr	Effective hyd. cond. in the main channel
Surlag	days	Surface lag coefficient
Alpha_Bf	decimal	Baseflow alpha factor
Esco	fraction	Soil evaporation compensation factor
Ch_N	[ ]	Manning coefficient for channel
Sol_Awc	mm/mm	Available soil water capacity
Blai	[ ]	Leaf area index for crop
Sol_Z	mm	Soil depth
Timp	[ ]	Snow pack temperature lag factor
Canmx	[ ]	Maximum canopy index
Cn2	[ ]	SCS curve number, antecedent moisture condition II,
Epco	fraction	Plant evaporation compensation factor

9.2. Baseflow Separation

Baseflow separation procedures in the Beauford sub-watershed showed that 49% of the total stream flow was contributed by baseflow. 87% of the total baseflow occurred in the five months from March to July, which account for 84% of the annual stream flow. The month of May has the largest baseflow (28%), while the month of August has the smallest (0.28%)(Figure 5).

The baseflow contribution to the LRW is about 66% of the total flow. In the period from March to July, baseflow represented 71% of the streamflow. The same period accounts for 72% of the total stream flow. The month of April has the largest baseflow (19%), while the month of January has the smallest (0.28%, Figure 6). The monthly distribution of baseflow in the LRW is more consistent from month to month than in small sub-watersheds like Beauford. This is what sustains flow in the LRW throughout the year. The high baseflow of the LRW, compared to the Beauford sub-watershed is due to the large contributing area that increases baseflow, as well as snowmelt that increases hydrostatic pressure.

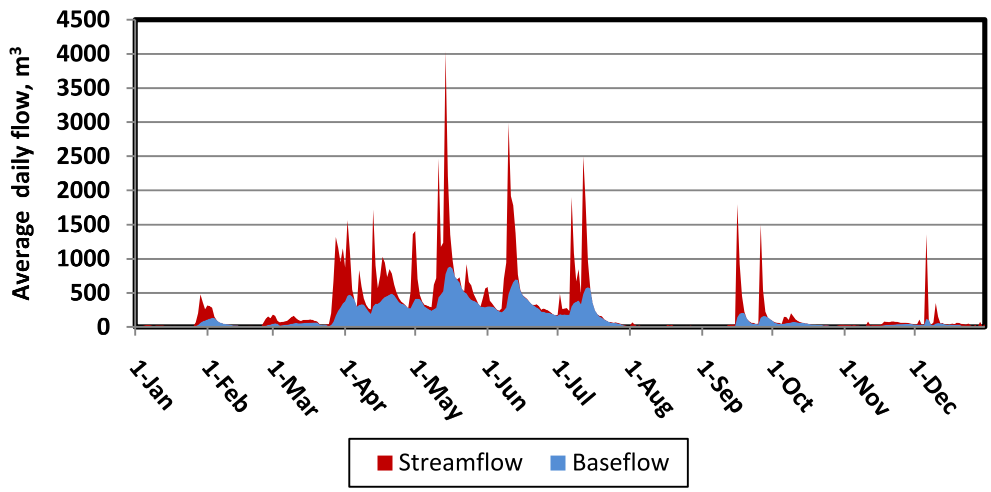


Figure 5: Average daily streamflow and baseflow in the Beauford sub-watershed

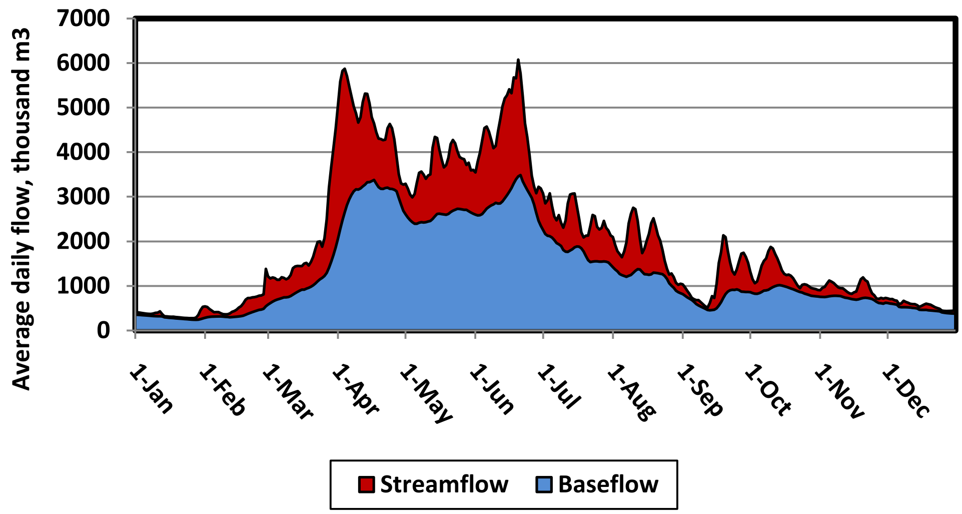


Figure 6: Le Sueur River average daily streamflow and baseflow

9.3. Calibration of Flow in the Beauford sub-watershed

To minimize the complexity of model calibration, the Beauford sub-watershed was selected as a representative small sub-watershed of the LRW. Beauford sub-watershed is located at the center of the LRW (Figure 3) and has an area of 2,096 ha. The fact that this sub-watershed has no significant channel erosion and no bluff or ravine erosion is very important for proper calibration of the SWAT model. The calibrated model parameters for the Beauford were considered transferable to the entire upland region of the LRW if the corresponding model performance as defined by the Nash-Sutclife efficiency on the Beauford sub-watershed is good.

The selected twelve most sensitive model input parameters were adjusted based on available measured data, knowledge about the watershed and an extensive literature review of SWAT model applications.

Timing of occurrence of both low and peak flows as predicted by the SWAT model generally agreed with observed data. The total simulated annual flow volume for the calibration year (2000) was less than the observed by 14%, with an NSE value of 0.77. Overall, model predictions showed very good agreement with field measured data. The stream flow calibration results are shown in Figure 7.

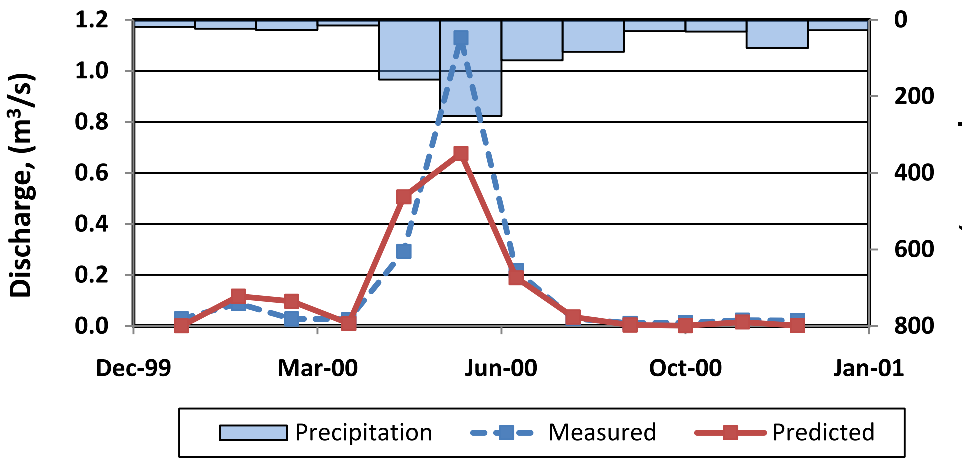


Figure 7: Calibration of monthly stream flow

9.4. Validation of Stream Flow

Validation of stream flow was conducted both in the Beauford and the entire LRW. Validation of monthly flow over the years 2001 - 2006 in Beauford gave an NSE of 0.89. Validation in the LRW from 1994 to 2006 showed good agreement between measured and predicted monthly flow, with an NSE of 0.73. These results indicate that the SWAT model can accurately simulate the hydrology of the LRW (Figure 8 and Figure 9).

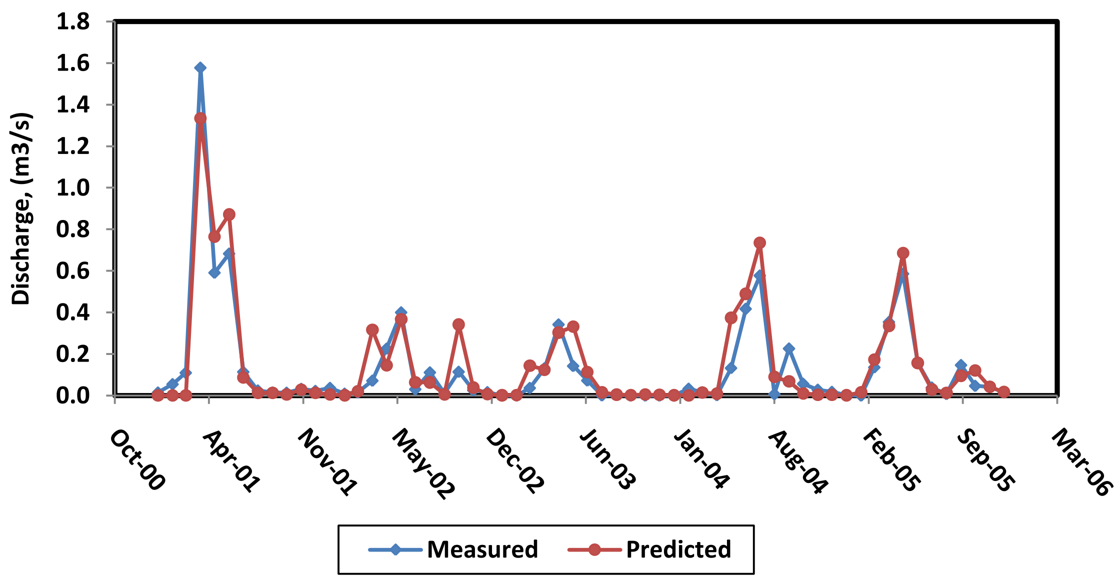


Figure 8: Validation of monthly flow in the Beauford sub-watershed

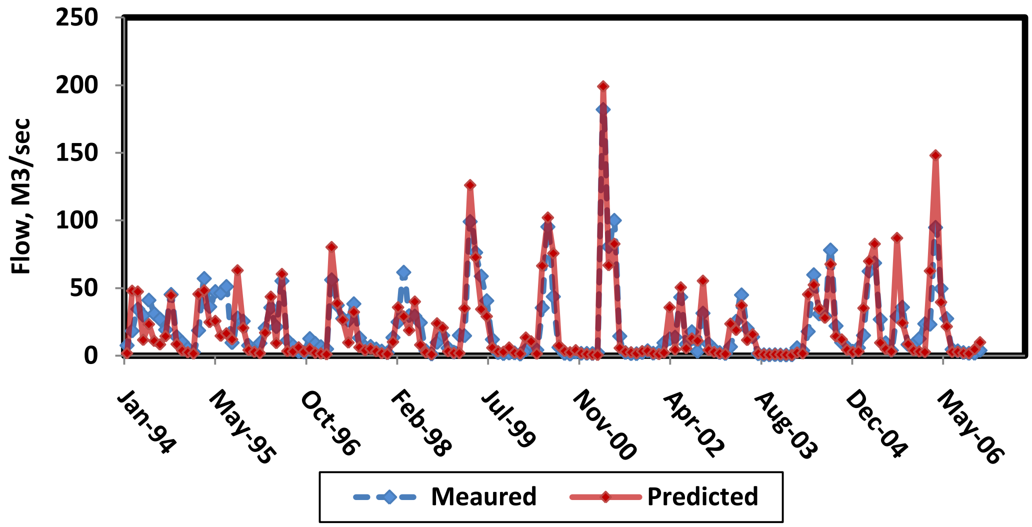


Figure 9: Validation of monthly flow in the LRW

The predicted average annual flows in the Beauford and Le Sueur watersheds were very accurate. The model over-predicted annual flow by 10% in the Beauford sub-watershed. Over the simulation period from 1994 to 2006 there was a 6% under-prediction in some years and a 7% over-prediction in other years, indicating no systematic error. The predicted flows in the Beauford and LRW have NSE values of 0.93 and 0.7, respectively (Figure 10 & Figure 11).

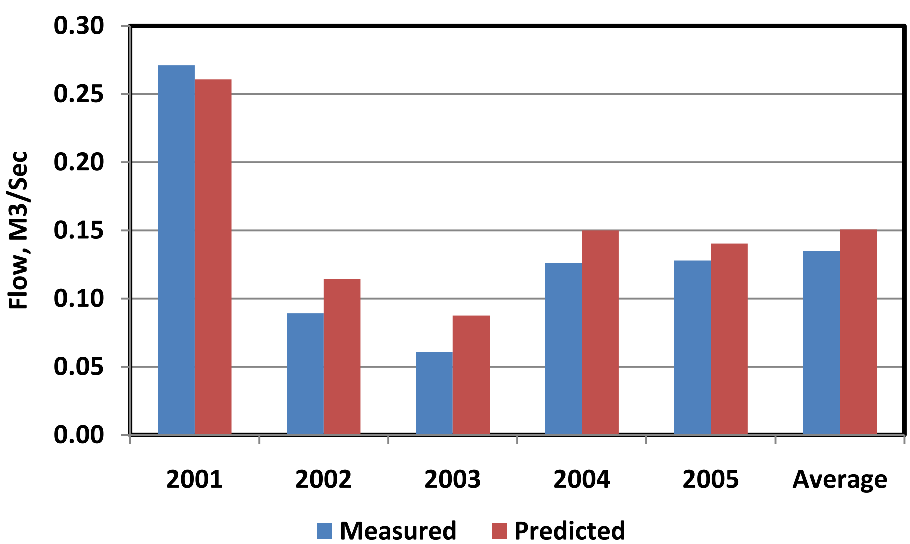


Figure 10: Annual flow in Beauford sub-watershed

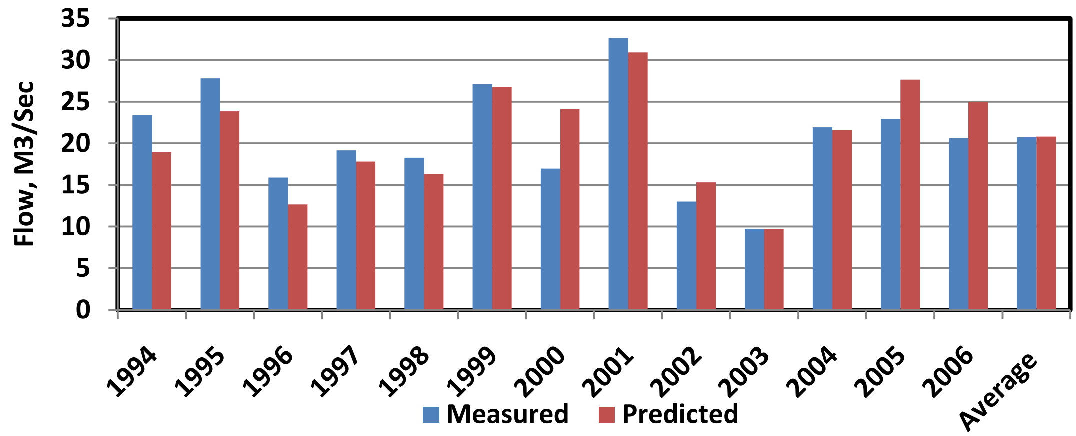


Figure 11: Annual flow in the LRW

9.5. Water Budget

The principle of conservation of mass was applied to compute the annual water budget of the LRW.

Where SW is the change in soil water storage, P is the total annual precipitation, ET the evapotranspiration, Qtile the tile flow, Qsurf the surface runoff flow, Qgw the groundwater flow and Qdaq the deep aquifer recharge.

ET accounts for 71% of the water budget, the largest of all components (Figure 12 & Table 5). Tile flow is the second largest component, accounting for 13% of the water budget. Surface runoff accounts for another 11% of the water budget. Available water holding capacity of the LRW soils varies considerably, the average annual soil water storage is estimated at about 245 mm, while the annual water yield is 230 mm.

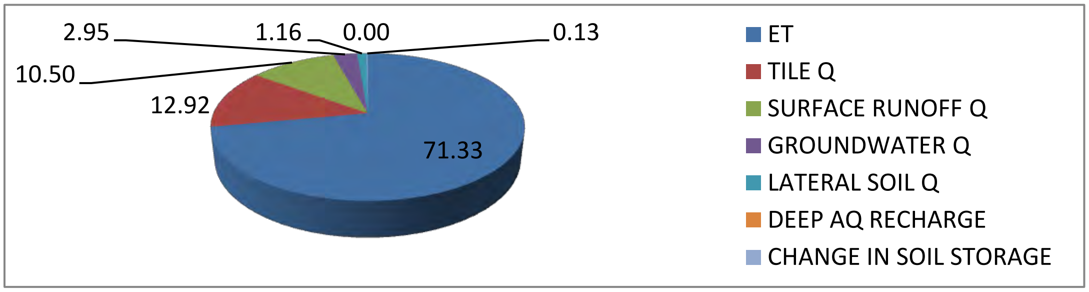


Figure 12: Average annual water budget

Table 5: Water budget components of the LRW (1994-2006)

YEAR	PREC	SURQ	LATQ	GWQ	TILEQ	SW	Δ SW	ET	WYLD
1994	854.7	60.8	11.1	25.6	111.1	262.1	0.0	641.3	207.3
1995	868.1	56.9	12.2	32.5	157.1	265.3	3.2	597.4	256.9
1996	799.9	48.8	9.8	17.6	69.0	262.0	-3.3	597.8	144.6
1997	687.0	64.3	10.1	21.1	100.4	225.3	-36.7	574.3	195.0
1998	809.8	69.8	8.4	20.4	81.6	253.4	28.1	598.3	178.8
1999	897.9	99.6	11.4	31.9	154.6	234.8	-18.5	607.2	296.2
2000	923.4	144.8	8.7	27.6	87.2	224.6	-10.3	596.6	267.1
2001	866.1	159.0	9.8	24.1	145.9	236.0	11.4	561.7	337.1
2002	805.9	64.4	8.4	22.7	79.3	255.0	19.1	609.5	173.6
2003	589.5	22.8	6.6	17.2	64.6	149.0	-106.0	567.0	110.7
2004	980.7	127.4	7.8	32.5	74.9	264.6	115.6	619.1	241.3
2005	1001.8	136.9	11.0	29.1	132.0	281.8	17.2	634.5	307.5
2006	845.1	91.8	11.1	20.4	155.0	276.0	-5.7	591.9	276.5
Average	840.8	88.2	9.7	24.8	108.7	245.4	1.1	599.7	230.2

The major water yielding areas of the watershed are concentrated in the western portion of the LRW. The source areas and their relative contributions are shown in Table 6 and Figure 13 below.

Table 6: Water yield versus areal coverage

WYLD, mm	**Area coverage, %**	**WYLD Contribution, %**
> 350	**2**	**3**
250-350	**34**	**41**
200-250	**45**	**43**
131-200	**17**	**13**
< 131	**2**	**0**

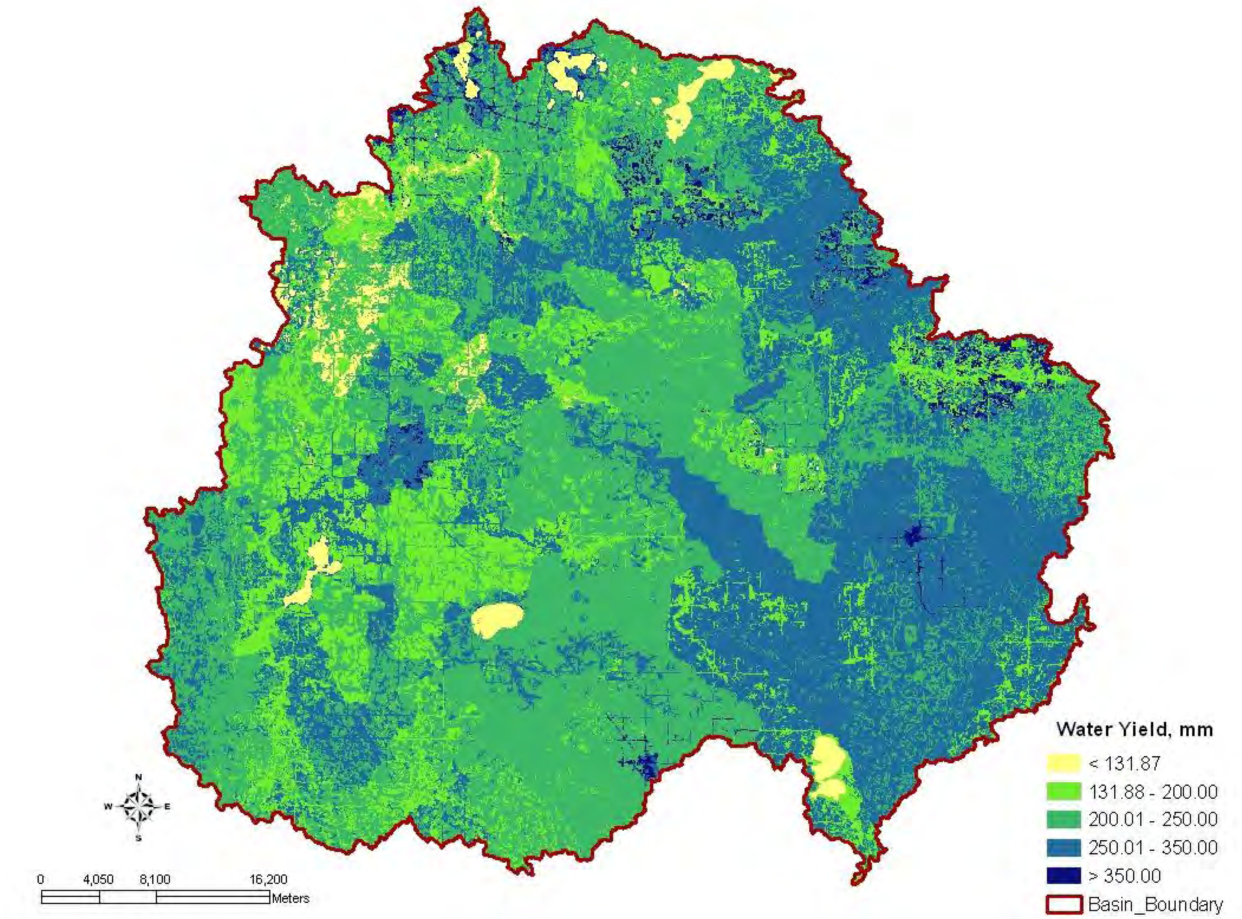


Figure 13: Spatial Variation in Water Yield in the LRW

10. Modeling Sediment

Sediment losses from the LRW have important implications for water quality of the watershed and all receiving waters downstream, including the Minnesota River and Lake Pepin. Sediment is a major pollutant and a transporter of pollutants that affects the quality of water resources. Characterizing the consequences of soil erosion, transport, and deposition is vital to identify the sediment source areas of the LRW. Sediment sources in the LRW are spatially heterogeneous, and include slumping river bluffs, ravines, stream banks and eroding upland agricultural lands. In this study of LRW, SWAT modeling was used to quantify the contribution of upland areas to sediment loads at various locations within the Le Sueur River Watershed. Predicted upland sediment loads were compared to measured stream sediment loads to indirectly estimate the sediment contributions from channel sources of sediment, including river bluffs, ravines and stream banks.

The SWAT model soil erosion algorithm uses the MUSLE equation to estimate the total amount of sediment delivered to the stream network within a watershed.

$$11.8*(Q_{surf} * qp)^{0.56}*K * LS *C * P*CFRG$$

where SYLD is the sediment yield to the stream network in metric tons, Q_{surf} is the surface runoff volume in mm, qp is the peak flow rate in m^3/s, K is the soil erodibility factor which is a soil property available from the Soil Survey Geographic (SSURGO) data, LS is the slope length and gradient factor, C is the cover management factor and can be derived from land cover data, P is the erosion control practice factor which is a field specific value and *CFRG* is the coarse fragment factor.

10.1. Calibration

The SWAT model was calibrated and validated from 2000-2006 in the Beauford sub-watershed, where the landscape has no bluffs or ravines. Calibrated model parameters other than channel scour factors were applied to the entire LRW in order to estimate sediment losses from upland regions of the watershed. Channel scour factors at the scale of the LRW (PRF and SPCON) were selected based on a sensitivity analysis. The contribution of the channel sources was estimated by difference from the amount measured at the outlet of the LRW.

The calibration results showed good agreement between measured and predicted sediment load at the mouth of Beauford sub-watershed with an NSE of 0.71 (Figure 14). The predicted sediment load was less than the measured by about 32%, which could be the contribution from near ditch sources of concentrated flow or ditch bank slumping.

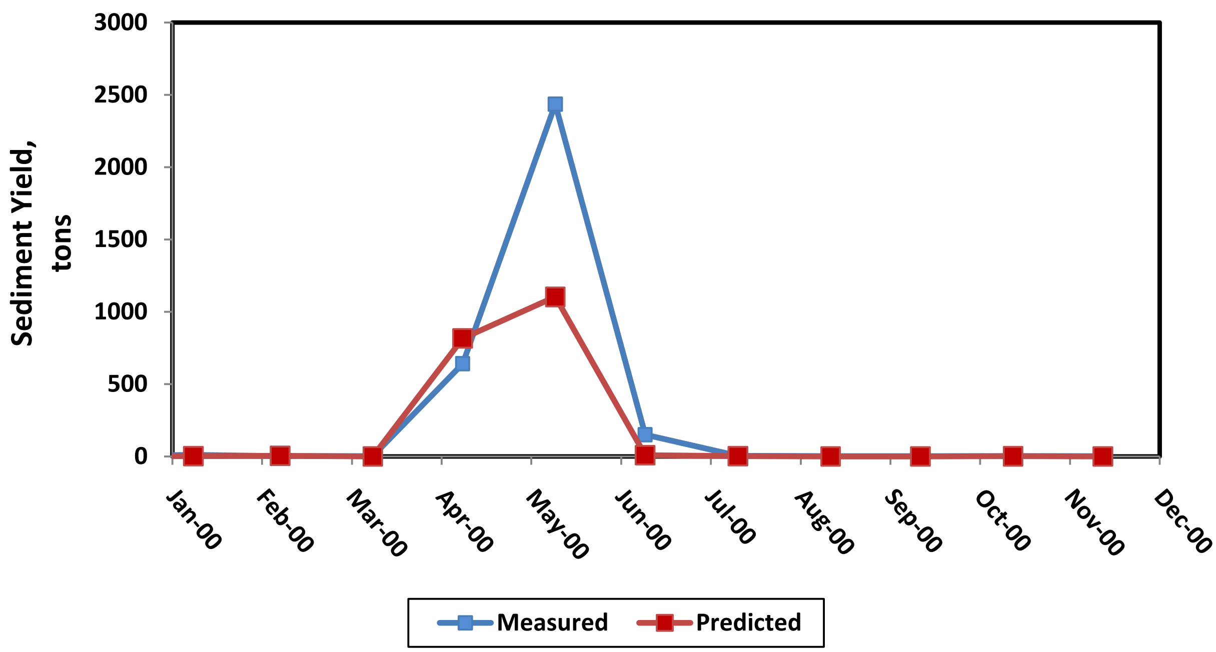


Figure 14: Calibration of Sediment Load in Beauford Sub-watershed (Year 2000)

10.2. Validation

The predicted sediment load in the Beauford sub-watershed for the validation years of 2001-2006 was satisfactory, with NSE values of 0.61 and 0.75 for monthly and annual loads, respectively (Figure 15 & Figure 16). In the LRW, the measured and simulated sediment yields have an average annual deviation of 86%. Overall, the time to peaks of the simulated sediment yield consistently matched the time to measured peaks of sediment yield in different seasons. Since the model only predicts upland sediment sources, the predicted sediment load was consistently under predicted. The predicted average annual sediment yield of the LRW ranged from 13% to 30% of the total measured sediment load. Most of the sediment loss (70% to 87%) from the LRW was contributed by the channel sources, probably from bluffs, streambanks and ravines.

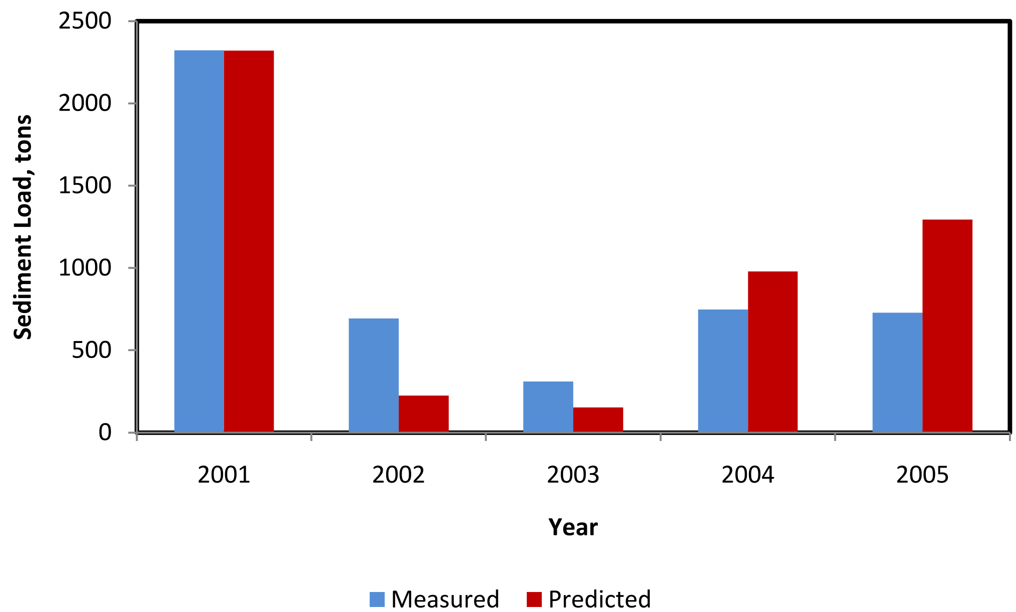


Figure 15: Validation of Annual Sediment Yield in the Beauford Watershed

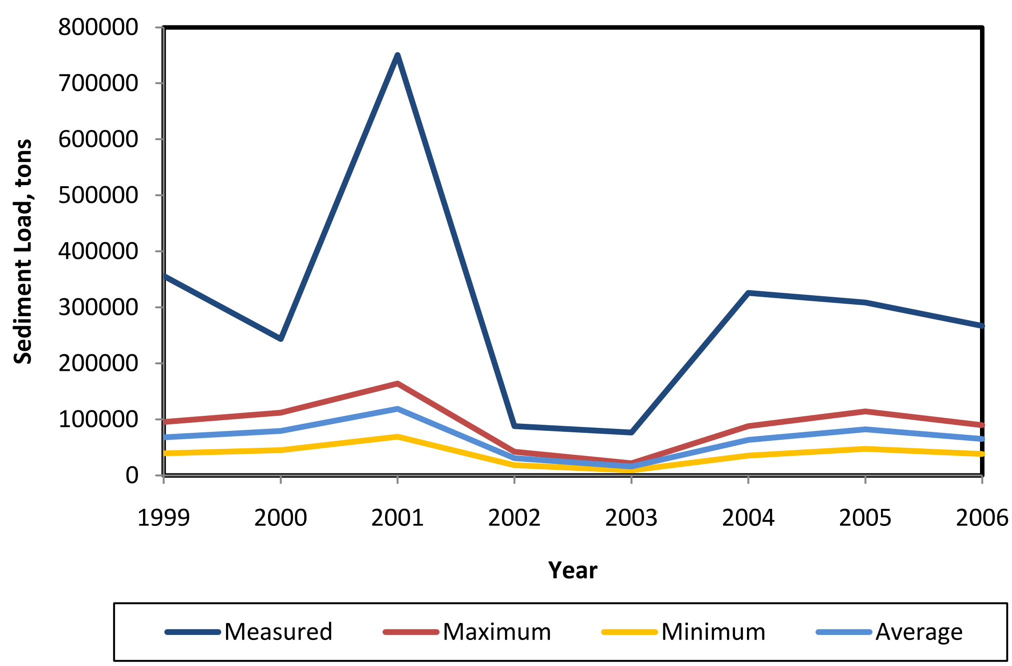


Figure 16: Validation of Annual Sediment Yield in the LRW.

10.3. Spatial and Temporal Distribution of Sediment Yield

Analysis of the monthly distribution of sediment yield shown in Figure 17, indicated that 72% of the sediment yield occurs in the three months from April to June. There is no substantial sediment yield in the four months from November to February.

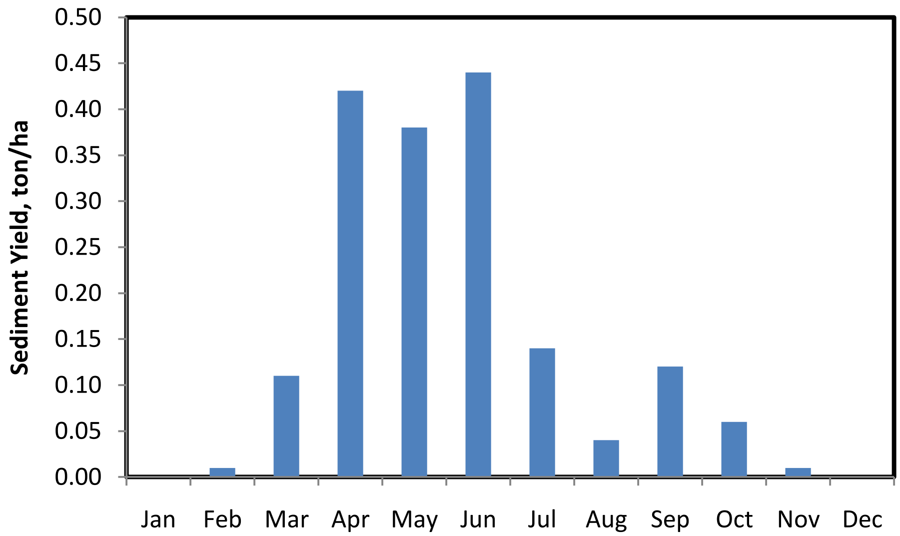


Figure 17: Average monthly sediment yield of LRW (1994-2006), ton/ha

Annual sediment yield in the LRW was below average in 8 of 13 years of simulation. Annual sediment yield was 30% less than the average for the years 1994-1998, and was very similar over those years. however, the sediment yield has shown over 50% increase in the years after 1998, with the exception of 2002 and 2003. Correlation coefficients of the linear regression of average sediment yield to surface runoff over the years 1994 to 2006 (Figure 18) suggest that the increase in sediment yield was primarily due to increases in surface runoff. Increased surface runoff is related to increased total amounts of precipitation received in those years. The minimum surface runoff (21.6 mm) occurred in the dry year of 2003, when the precipitation was 581 mm, leading to a sediment yield of only 0.4 ton/ha. In contrast, 910 mm of rainfall in the wet year of 2000 caused 159 mm of surface runoff flow that generated a sediment yield of 3.5 ton/ha. The sediment yield response to amount of annual precipitation was not as strong as the response to the surface runoff. Changes in precipitation only explained 62% of the variation in sediment yield (Figure 19).

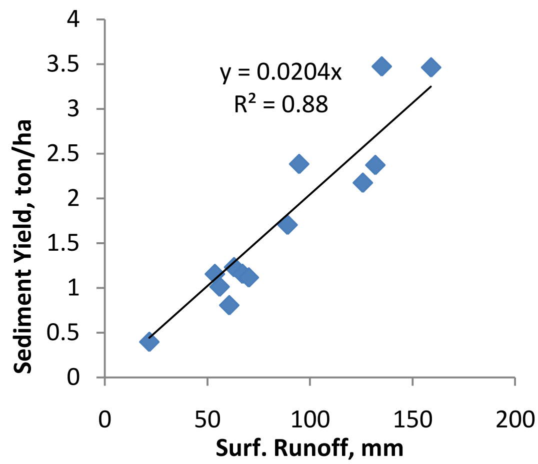


Figure 18: Sediment yield runoff relationship

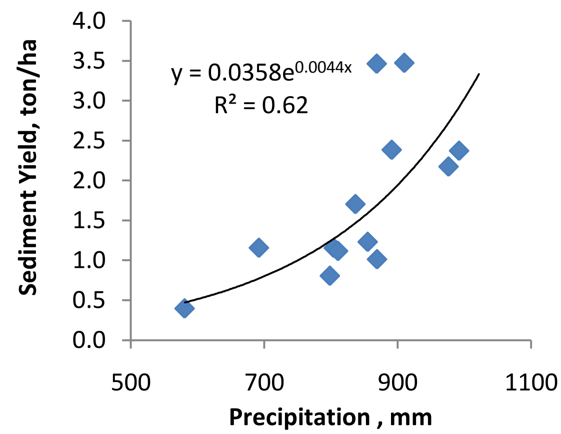


Figure 19: Sediment yield versus precipitation

Table 7: Average annual hydrology and sediment yield of LRW

Year	Precipitation, mm	WYLD, mm	Surf. Runoff, mm	SYLD, ton/ha
1994	855.2	212.7	62.9	1.2
1995	869.2	261.3	55.9	1.0
1996	804.5	153.7	53.6	1.2
1997	692.3	203.8	67.0	1.2
1998	810.8	184.3	70.1	1.1
1999	891.2	295.9	94.7	2.4
2000	910.1	262.1	134.9	3.5
2001	868.8	343.0	159.0	3.5
2002	798.6	170.9	60.6	0.8
2003	581.1	107.9	21.6	0.4
2004	976.3	241.9	125.7	2.2
2005	992.0	304.7	131.8	2.4
2006	836.9	275.2	89.0	1.7
Average	837.5	232.1	86.7	1.7

10.4. Upland Sediment Source Areas

Identifying sediment source areas is crucial to design proper abatement strategies and develop TMDL initiatives. In the case of the LRW, the CCAs contributed disproportionately high water yield, but CCAs did not show a large difference in sediment yield from the rest of the watershed area. On average, 91% of the upland areas of the LRW delivered less than 5 tons/ha and 9% delivered over 5 tons/ha (Figure 20 & Figure 22). Roughly 25% of the LRW generated half of the upland sediment losses (Figure 21).

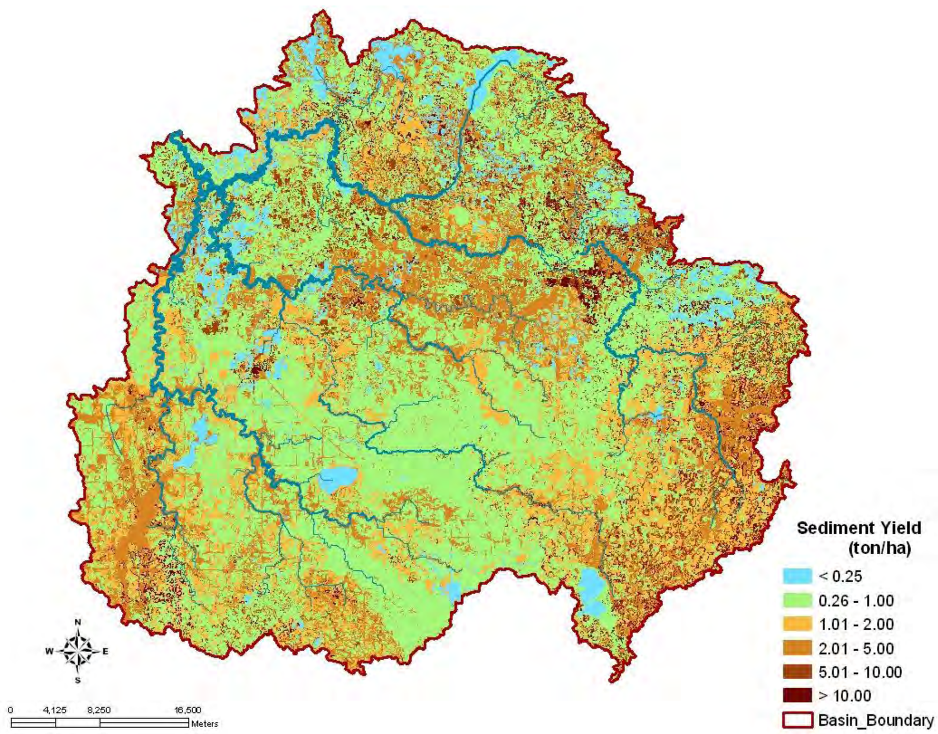


Figure 20: Spatial Distribution of Sediment yield in the LRW

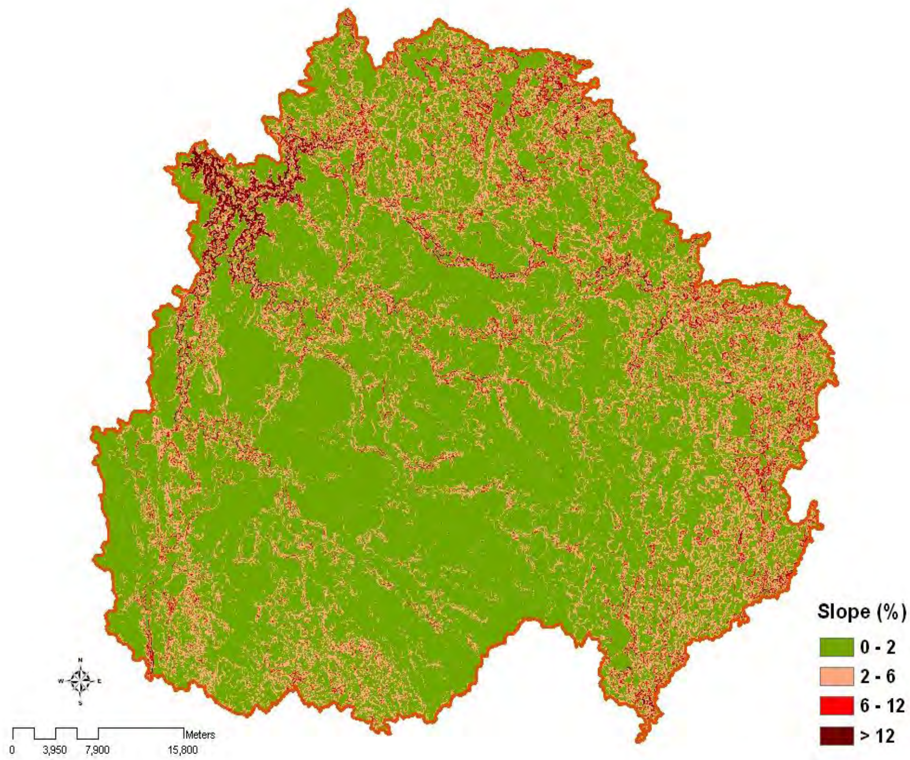


Figure 21: LRW slope map

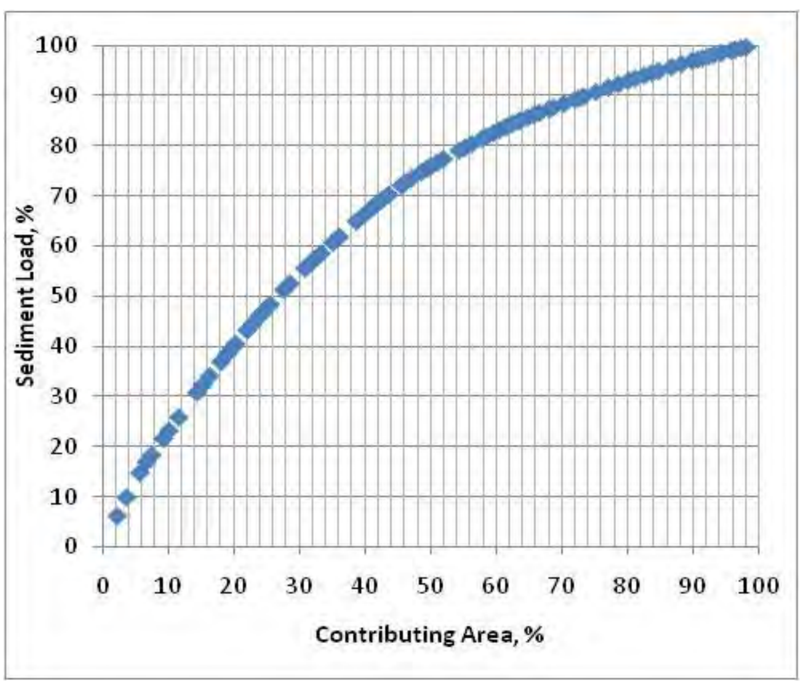


Figure 22: Cumulative Sediment Yield vs Contributing Watershed Area of LRW

Table 8: Summary of Sediment Yield Vs Contributing Areas of LRW

SYLD, ton/ha	Area, ha	Area, %
< 5	258692.21	91
5 to 10	19000.55	7
10 to 20	7512.59	3
> 20	470.72	0
Total	285676.06	100

10.5. Sediment BMPs

Various BMPs to reduce upland sediment loads to the LRW and its tributaries were evaluated using the SWAT model. The BMPs were tested for specific land use and potential sediment source areas.

A. <u>Tillage BMPs</u>

Three different scenarios conservation tillage were tested in the corn residue from the year before soybeans were planted:

- On all fields under corn residue going from corn to soybean
- Corn going into soybeans on land over 2% slope.

Each scenario resulted in sediment load reductions of 13% in the LRW compared to the baseline scenario of conventional tillage. Application of conservation tillage on 50% of randomly selected soy land (73,852 ha) decreased the sediment yield by 9%. Application of no-till on the same randomly selected 50% soy land decreased the sediment yield by 31% (Table 9).

Table 9: Sediment Loss Under Different Tillage BMP Scenarios

Sed. load, tons (% of baseline)

Year	Baseline Scenario	Conservation tillage			No till
		All Corn land	Land > 2% slope	50% Corn land	50% Corn land
1994	351623	309445(85.6)	309846(85.7)	323940(89.8)	228480(65.5)
1995	289349	253626(83.6)	255653(84.2)	270498(89.8)	204231(60.2)
1996	329939	284724(90.2)	285448(90.4)	304293(93.3)	220709(72)
1997	330432	282700(88.6)	283078(89)	296739(97.1)	216413(69.6)
1998	318872	266662(86.4)	268522(86.5)	286298(91.1)	191940(66.7)
1999	680854	614465(84.2)	615686(84.4)	635388(89.5)	490084(66.7)
2000	992003	878960(92.3)	882518(92.4)	963282(98.5)	690503(79.2)
2001	988949	854127(84.8)	855280(85)	900747(90.6)	659891(67.3)
2002	230289	193831(90.4)	194270(90.5)	206115(93.5)	153680(78.8)
2003	113424	104710(84.3)	104765(84.4)	111703(88.6)	89849(61.9)
2004	621019	526314(87.2)	527603(87.5)	562554(92.4)	417805(68.6)
2005	677579	309445(85.6)	309846(85.7)	323940(89.8)	228480(65.5)
2006	486717	253626(83.6)	255653(84.2)	270498(89.8)	204231(60.2)
Average	493158	284724(90.2)	285448(90.4)	304293(93.3)	220709(72)

B. Filter Strips

The vegetative filter strip (VFS) is thought to be one of the most effective methods to trap sediment effectively. Establishing VFS at the edge of agricultural fields or adjacent to streams or drainage ditches has been shown to be effective in removing sediment loss from upland runoff.

The SWAT model algorithm of VFS sediment trapping efficiency equation is given by:

$$_{\text{filtstrip}})^{0.2967}$$

where $trap_{eff}$ is the fraction of the constituent loading trapped by the filter strip, and $width_{filtstrip}$ is the width of the filter strip (m).

This equation has some limitations. Even though there are many factors affecting sediment trapping efficiency, such as runoff volume, soil properties, and vegetative properties, the SWAT model algorithm to simulate VFS effects on sediment reduction is set only as a function of width. Despite this limitation, three different alternative scenarios of VFS were tested in the LRW (Table 10). The simulation results revealed that application of VFS on all corn -soy land over 2% slope reduces sediment loss by 56% and VFS installation on all corn-soy CCAs reduces the sediment loss by about 20% as compared to the baseline scenario.

Table 10: Sediment Loss Under Different VFS BMP Scenarios

Sed. load, tons (% of baseline)

Year	Baseline	On CS land > 2% slope	On all CS CCAs
1994	351623	121382(34.5)	276936(78.8)
1995	289349	106761(36.9)	234533(81.1)
1996	329939	139655(42.3)	261836(79.4)
1997	330432	139791(42.3)	264995(80.2)
1998	318872	137301(43.1)	252456(79.2)
1999	680854	277106(40.7)	550127(80.8)
2000	992003	458741(46.2)	784208(79.1)
2001	988949	450535(45.6)	809755(81.9)
2002	230289	97497(42.3)	164468(71.4)
2003	113424	49782(43.9)	98725(87)
2004	621019	287666(46.3)	488343(78.6)
2005	677579	340709(50.3)	557473(82.3)
2006	486717	208850(42.9)	384537(79)
Average	493158	216598(43.9)	394492(80)

C. <u>Cover Crops</u>

The vegetative biomass of rye as cover crop increases the amount of transpiration and decreases the impact of rain drops that can break soil aggregates. As a result of this, there is an increase in water infiltration and decrease in surface runoff and runoff velocity. Planting rye after the harvest of soybeans in the LRW reduced the sediment loss into the streams by an average of 32% (Table 11).

Table 11: Effects of planting rye as a cover crop in reducing sediment loss

	Sed. load, tons (% of baseline)	
Year	Baseline	Rye as cover crop
1994	351623	233265(66.34)
1995	289349	197150(68.14)
1996	329939	263284(79.8)
1997	330432	192726(58.33)
1998	318872	196497(61.62)
1999	680854	453698(66.64)
2000	992003	768043(77.42)
2001	988949	609187(61.6)
2002	230289	163233(70.88)
2003	113424	84193(74.23)
2004	621019	459333(73.96)
2005	677579	484285(71.47)
2006	486717	257258(52.86)
Average	493158	335550(67.95)

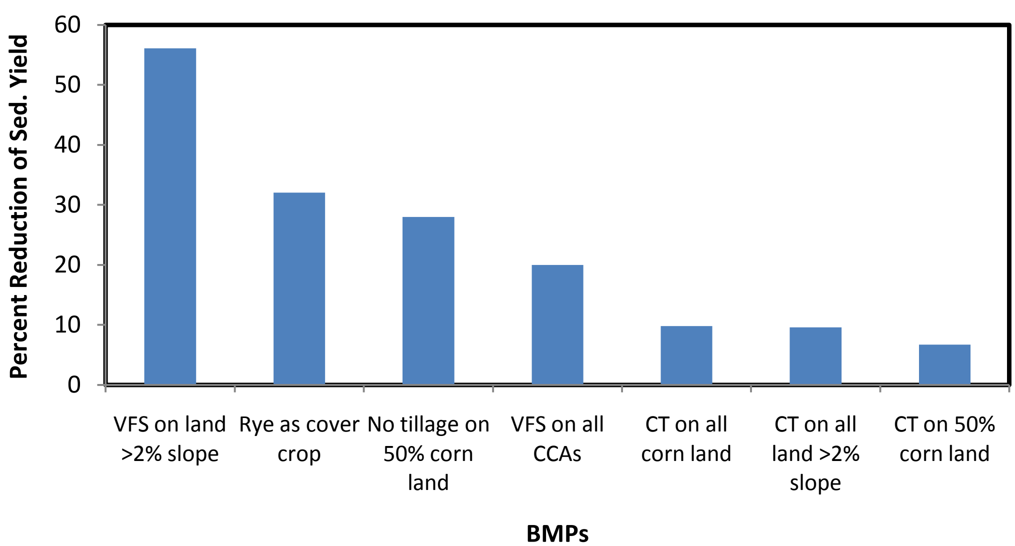


Figure 23: Comparison of Different BMPs Sediment Loss Reduction Potential

11. Modeling Phosphorus

The LRW water quality has been impacted by phosphorus enrichment, mainly from applied fertilizer and animal manure for crop production. The LRW SWAT phosphorus modeling effort was intended to evaluate spatial trends in phosphorus loss from the watershed and then to develop mitigation strategies by applying best management practices.

11.1. Calibration

The calibration and validation of phosphorus focused on total phosphorus coming out of the Beauford sub-watershed. The period of calibration (2000) and validation (2001-2006) represented the most recent time period for which required input data and calibration/validation data were available.

Phosphorus calibration of the SWAT model in the Beauford sub-watershed showed good agreement between predicted and measured total phosphorus at the outlet of the watershed, with an NSE of 0.79 (Figure 22). Even though the observed NSE value is acceptable, the predicted P load was 39% less than measured. The under-prediction was mainly during peak snowmelt months and was consistent with model under-predictions of hydrology and sediment in those same months.

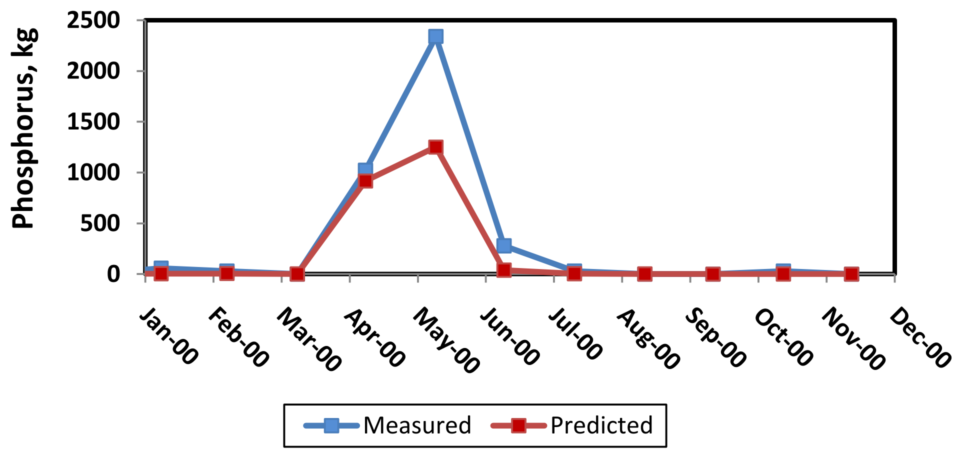


Figure 24: Phosphorus calibration in the Beauford Sub-watershed.

11.2. Validation

The predicted average monthly total phosphorus loading in the Beauford and LRW were 56% and 77% of their respective measured average monthly loadings (Figures 24 & 26). Validation NSE for the total monthly phosphorus loadings were 0.76 and 0.67 in the Beauford sub-watershed and LRW, respectively (Figures 23 and 25) . The largest errors in SWAT model phosphorus predictions were always associated with peak flow prediction errors. Moreover, the fact that the P from stream banks and deposition was not counted has also contributed for the gap in between the measured and predicted amounts.

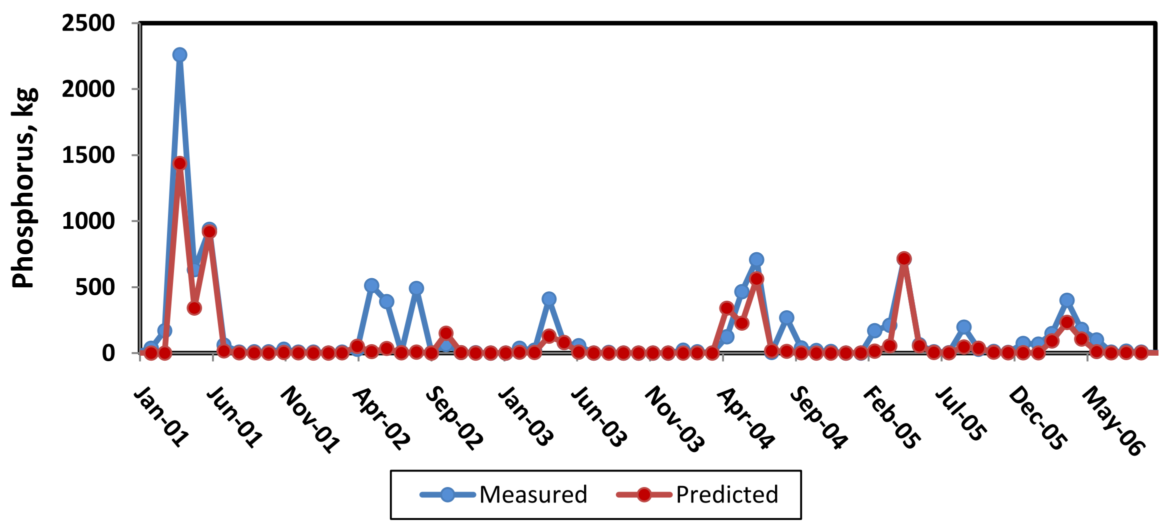


Figure 25: Monthly Phosphorus Loss Validation in the Beauford Sub-watershed

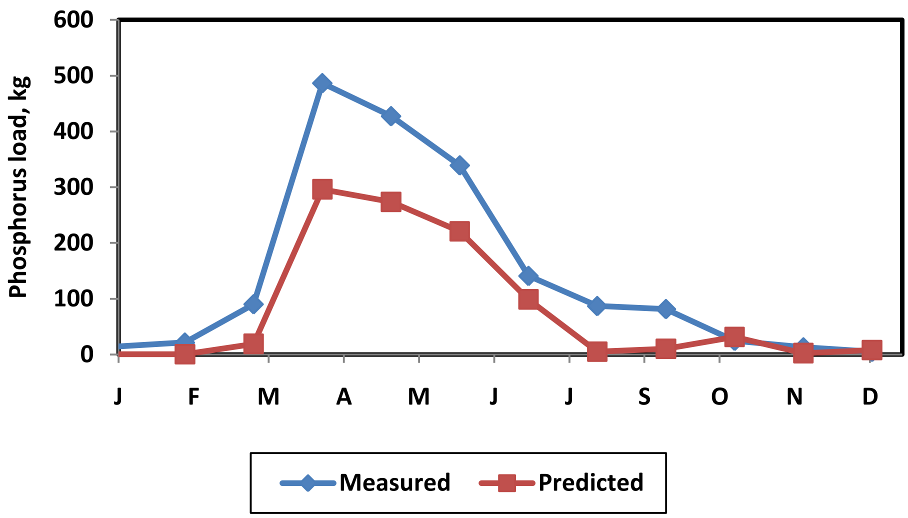


Figure 26: Average Monthly Phosphorus Loss Validation in the Beauford Sub-watershed (2001-2006)

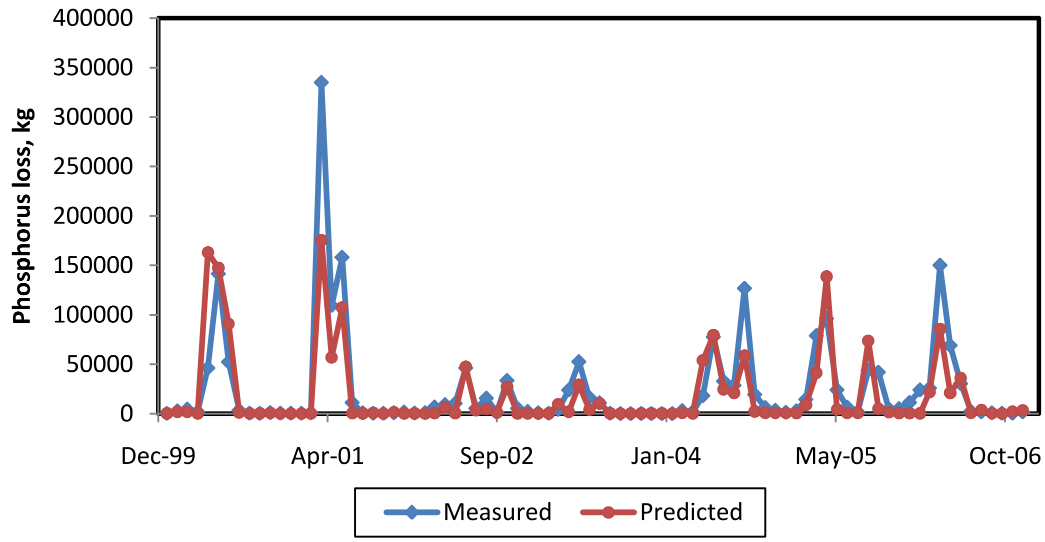


Figure 27: Monthly Phosphorus Loss Validations in the LRW

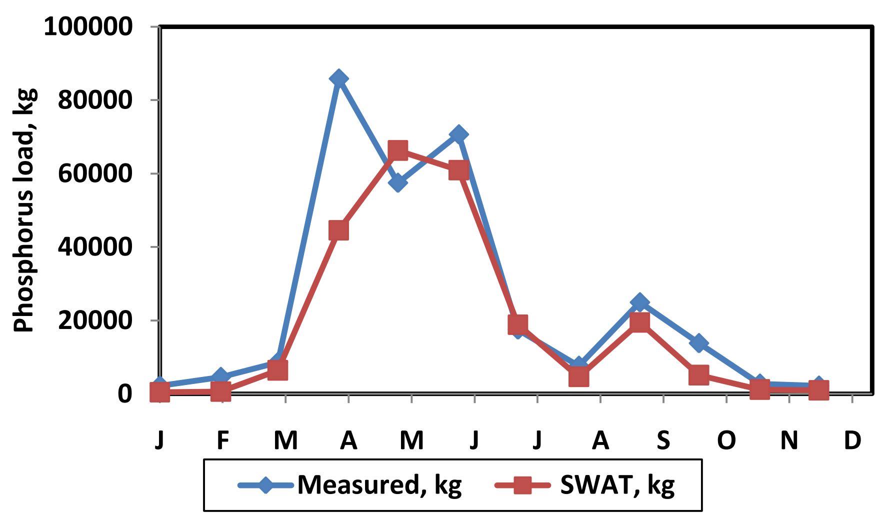


Figure 28: Average Monthly Phosphorus Loss Validation in the LRW (2000-2006)

11.3. Spatial and Temporal Distribution of Phosphorus Loss

The SWAT simulation results showed that the highest phosphorus losses (0.22-0.28 kg/ha/month) occur in the peak flow months of April to June. This accounts for 75% of the total annual loss. About 85% of the P losses occurred in the particulate or sediment bound form. The dissolved P was only about 15% (Table 12). Agricultural row crop producing fields account for 99% of the predicted total phosphorus loading to surface waters of the LRW.

Average monthly phosphorus load in the LRW was estimated using the USGS LOADEST program. The monthly maximum loss (83,000 kg) occurred in the month of April (Figure 28). Based on 446 samples tested from 1999-2006, the maximum average monthly concentration (0.61 mg/L) was recorded in the month of June. The loss of phosphorus is affected by several factors, including the occurrence, amount and intensity of rainfall and runoff, P application amount and timing, and land management practices such as tillage etc.

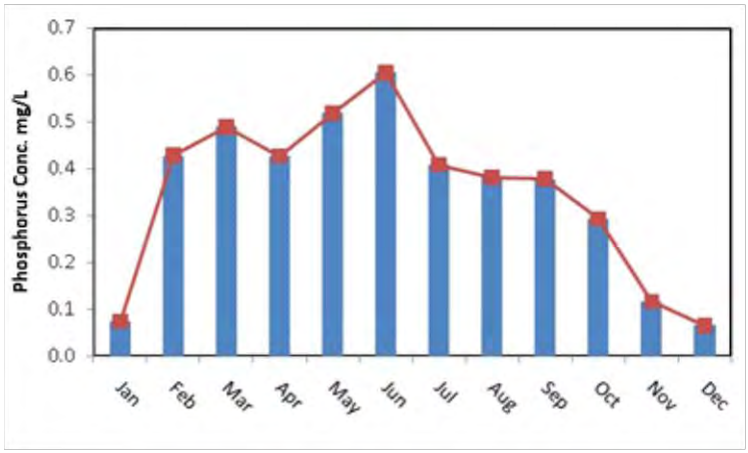


Figure 29: Monthly phosphorus concentration in the LRW

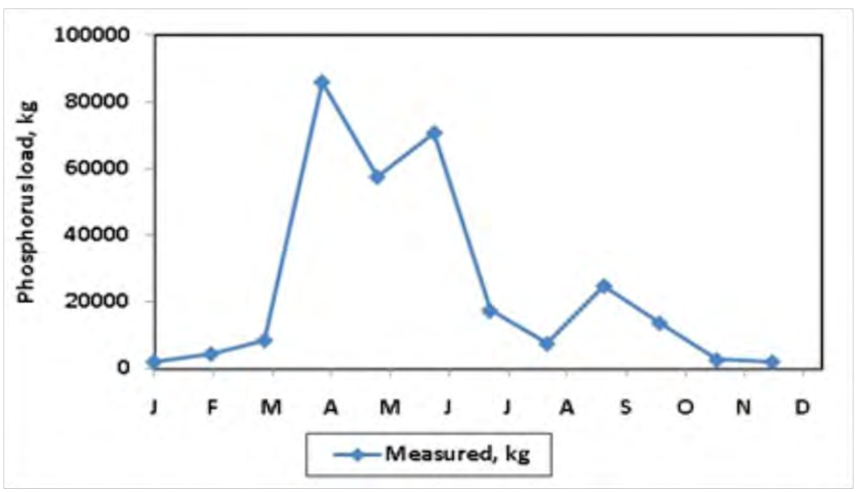


Figure 30: Monthly phosphorus load in the LRW

The annual discharge of total P from the LRW varied across years during the period from 1994-2006. The predicted lowest annual phosphorus loss of 0.44 kg ha^{-1} occurred in 2002, and the highest annual losses of 2.01 kg ha^{-1} occurred in 2001. The mean annual total P loss across all years was 1.02 kg ha^{-1} (Table 12).

Table 12: Annual Phosphorus Loss in the LRW (kg/ha)

Year	ORGANIC P	SEDIMENT P	SOLUBLE P	Total P
1994	0.19	0.31	0.08	0.59
1995	0.18	0.31	0.10	0.59
1996	0.24	0.43	0.12	0.79
1997	0.23	0.34	0.10	0.66
1998	0.20	0.31	0.10	0.62
1999	0.42	0.76	0.18	1.37
2000	0.57	1.14	0.27	1.98
2001	0.68	1.07	0.26	2.01
2002	0.13	0.23	0.07	0.44
2003	0.07	0.14	0.05	0.27
2004	0.41	0.75	0.20	1.37
2005	0.46	0.85	0.23	1.55
2006	0.33	0.52	0.14	0.99
Average	0.32	0.55	0.15	1.02

The southwestern and southeastern portions of the watershed had the highest P loss rates (Figure 30). Those are areas that have steep slopes, high surface runoff and sediment loss (Figures 13, 20 & 21). The Upper Le Sueur major sub-watershed has the highest annual sediment yield of 1.35 kg/ha and contributes about 64% of the P loss, also it has about 40% of the total area. The other two major sub-watersheds, Maple and Big Cobb, contribute 20% and 16%, respectively. Roughly 30% of the LRW contributes 50% of the phosphorus load (Figure 29)

Table 13: Phosphorus loss in the major sub-watersheds of LRW

Reach	Area, ha	ORGANIC P, kg	MINERAL P, kg	Total P, kg
LRW	285676	104100	102000	206100
Upper Le Sueur	115286	79180	76420	155600
Big Cobb	79955	19720	19670	39390
Maple	88081	25930	23130	49060

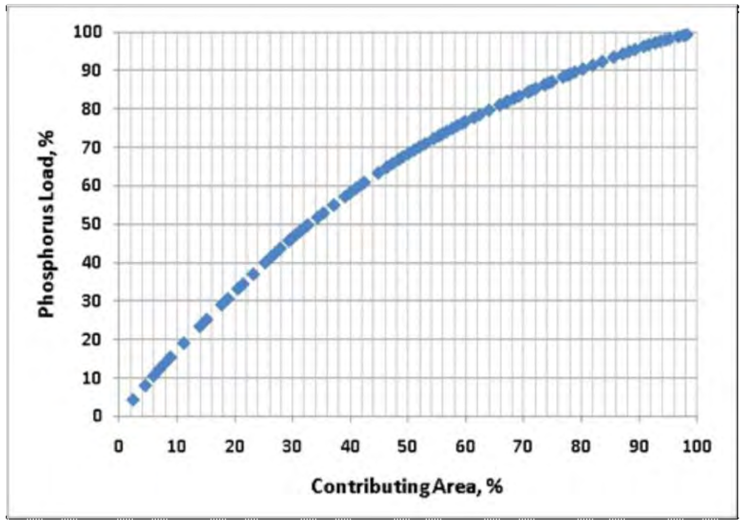


Figure 31: Phosphorus contributing area vs load.

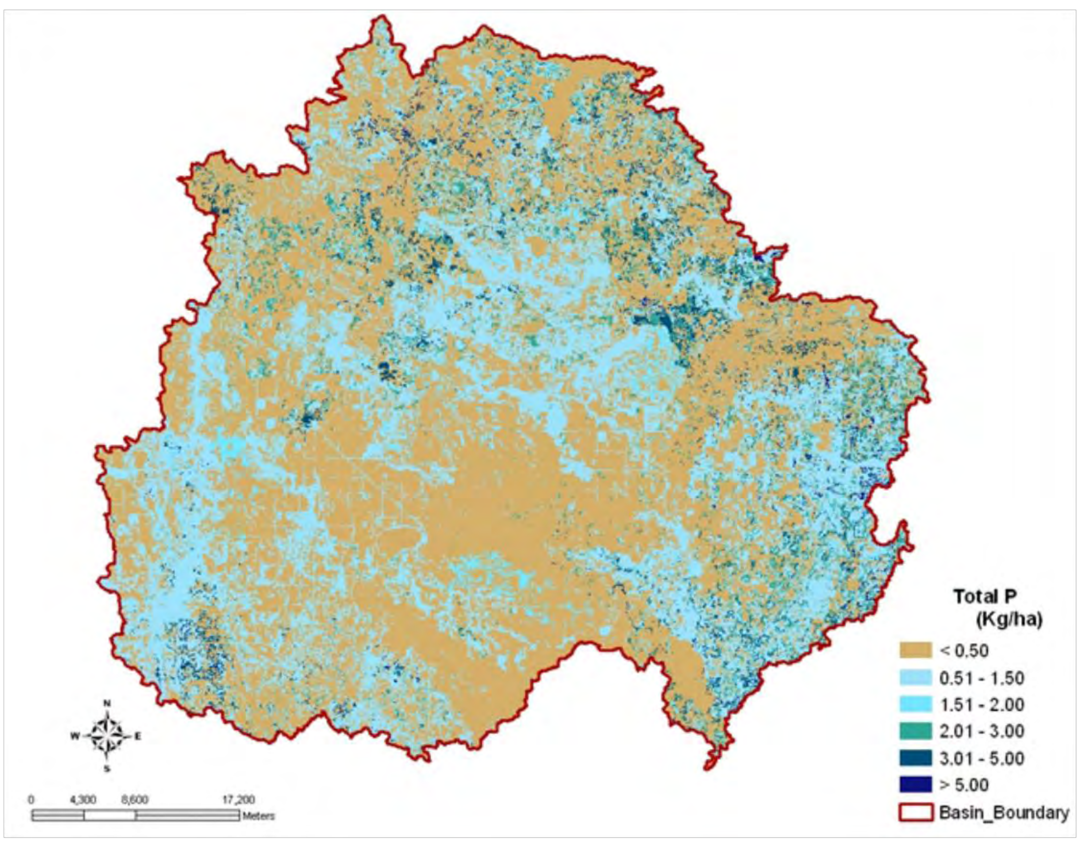


Figure 32: Spatial distribution of phosphorus loss in the LRW

Phosphorus Budget

The mineralization of organic P from soil humus, crop residue and microbial biomass of the LRW was estimated at about 24.3 kg/ha. Since crop uptake was 29 kg/ha/year and losses were about 1kg/ha/year, the application should not have exceeded 7 kg/ha/year. However, the annual phosphorus application to the LRW is about 38 kg/ha, and animal manure accounts for 15% of the total. Considering the annual uptake and the losses in runoff, there is an accumulation of 11 kg/ha/year phosphorus in the LRW (Table 14).

The role of point sources has also been accounted for in the loss estimates at the mouth of LRW. The average annual mineral phosphorus released from point sources was about 185 kg. This is about 0.001% of the annual loss from the entire watershed.

Table 14: LRW Phosphorus Budget

INPUTS	(KG/HA)	%
P FERTILIZER APPLIED (Min Fertilizer and Manure)	38.36	61.21
MIN OF FRESH ORG P (Residue and Microbial biomass)	14.54	23.20
MIN OF ORG P (HUMUS)	9.77	15.59
TOTAL	62.67	100.00
OUTPUTS	(KG/HA)	%
P UPTAKE	29.03	55.8
Adsorbed P	21.94	42.2
Soluble P loss	0.15	0.3
Sediment P loss	0.55	1.1
Organic P loss	0.32	0.6
Point Source P loss	0.001	0.002
TOTAL	51.99	100.00

Change in storage = Inputs - Outputs = 10.8

The annual accumulation of 10.8 kg/ha phosphorus in upland soils of the LRW may affect freshwater ecosystems in the long run. The adverse effects of P on freshwater at downstream of sources could occur after several years of accumulation of P in soil (Reed-Andersen, 2000). Daniel et al. (1994) and Sharpley et al. (1994) reported that all factors that increase erosion or the concentration of P in the soil also increase the potential for P losses. Therefore, the current P accumulation rate in the LRW could have a very dangerous effect on the LRW water quality in the long run.

11.4. Phosphorus BMPs

Phosphorus concentration at the soil surface and increases in runoff are the two major factors that increase phosphorus transport in runoff (Sharpley et al., 2003). Thus, all practices that can reduce these two factors can reduce the loss of P.

A. Reduced rate phosphorus application

Application of 56 kg/ha of P_2O_5, from both animal manure and applied DAP fertilizer, has been evaluated against the baseline application rate of 84 kg/ha P_2O_5. The 34% reduction in application rate reduced the total P loss by about 28% (Table 15).

Table 15: Relative effects of application of different BMPs in reducing P loss of the LRW

Scenario	SEDIMENT P	SOLUBLE P	ORGANIC P	Total P	% Reduction
Baseline	0.55	0.15	0.32	1.02	
Reduced Rate Application	0.36	0.09	0.28	0.73	28
Conservation Tillage on 50% corn land	0.52	0.15	0.3	0.87	15
No Tillage on 50% corn land	0.4	0.15	0.21	0.76	25
VFS on CS over 2% slope	0.18	0.08	0.1	0.36	64
Rye as cover crop	0.38	0.13	0.22	0.73	29

B. Conservation tillage

Cropping systems that maintain at least 30 percent of the soil surface covered with residue after planting help to reduce soil erosion and the associated P loss. Application of conservation tillage on 50% of the corn residue in the year before soybean planting reduced P losses by 15%. The reduction was mainly due to reductions in the particulate phosphorus loss.

C. Vegetative filter strips (VFS)

Establishment of vegetative grass strips on corn-soybean fields with slopes steeper than 2% gave a 64% reduction in P losses.

D. No tillage

Direct seeding of soybean into corn residue without tillage showed a 25% reduction in P losses. This is mainly a result of an increase in infiltration rates and reduction in soil erosion. Application of no-till reduced sediment adsorbed P, but increased soluble P losses. No-tillage is not recommended on most soils in the Minnesota River Basin because of negative effects on crop yield.

E. Cover Crops

Planting rye as a fall cover crop reduced phosphorus losses by 29%, mainly through reducing surface runoff from the soil.

12. Modeling Nitrogen

High nitrogen losses in tile drainage from upland agricultural areas of the LRW detrimentally affect downstream water quality. According to Randall et al., (1995) the annual nitrate-N loss through subsurface drainage ranges from 1.4 to 139 kg/ha at Waseca in the LRW, depending on variations in climate and cropping system. Thus, it is important to understand the sources, transport and fate of nitrogen from LRW using a watershed modeling approach.

SWAT describes the nitrogen transport and loss processes by simulating nitrogen availability, transport, and attenuation processes using mechanistic functions. The model describes the spatial and temporal variations of sources and sinks within a watershed. Nitrate losses from the LRW occur largely in tile drainage. Nitrate removed through the tile drainage is calculated using:

$$N03_{tile} = Conc_{NO3,mobile} * Q_{tile}$$

where $N03_{tile}$ is the nitrate removed in tile flow from a layer (kg N/ha),

$Conc_{NO3\ mobile}$ is the concentration of nitrate in the mobile water through tiles (kg N/mm H20), and Q_{tile} the water discharged from the layer by tile drainage (mm H20).

The amount of water removed from a layer in the tile drain on a given day is calculated using:

where $tile_{wtr}$, is the amount of water removed from the layer on a given day by tile drainage (mm H20), h_{wtbl} is the height of the water table above the impervious zone (mm), h_{drain} is the height of the tile drain above the impervious zone (mm), SW is the water content of the profile on a given day (mm H20), FC is the field capacity water content of the profile (mm H20), and t_{drain} is the time required to drain the soil to field capacity (hrs).

12.1. Calibration

Due to the complexity of the SWAT model nitrogen component and its intensive input data, calibration and validation of the model are vital. Calibration and validation of the LRW SWAT model was based on monthly model predictions for total nitrogen loads compared against measured monthly data.

Calibration of nitrate-N in the Beauford sub-watershed gave an NSE at the outlet of the watershed of 0.74 (Figure 31). The ratio of predicted to measured annual N loads was about 89%.

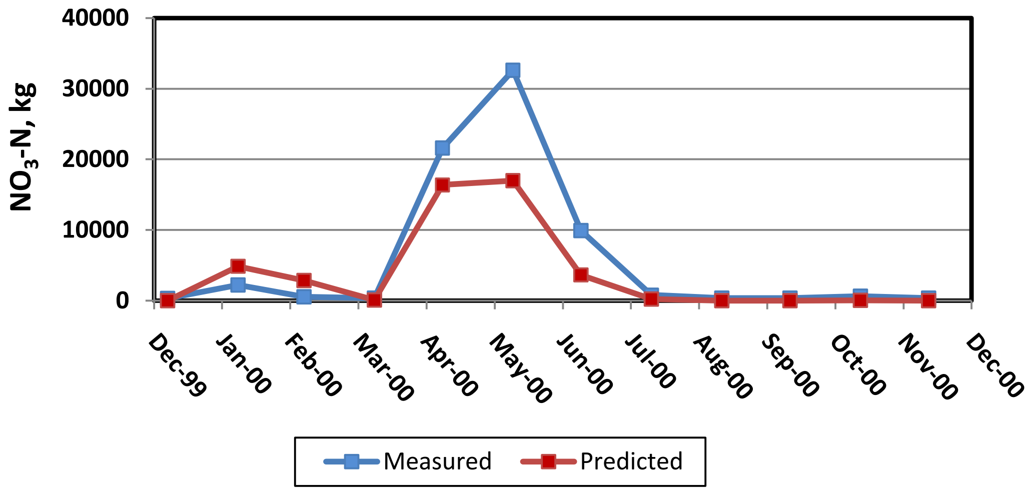


Figure 33: Nitrate-Nitrogen calibration in the Beauford Sub-watershed.

12.2. Validation

Validation of SWAT for nitrate-N loads was evaluated using monthly nitrate-N measurements in both the Beauford sub-watershed and the entire LRW. The monthly validation NSE values for nitrate-nitrogen loading were 0.77 and 0.74 in the Beauford sub-watershed and LRW, respectively. The ratio of predicted to measured nitrate-N loads were 0.81 and 0.76 in the Beauford sub-watershed and LRW, respectively (Figures 32 and 34).

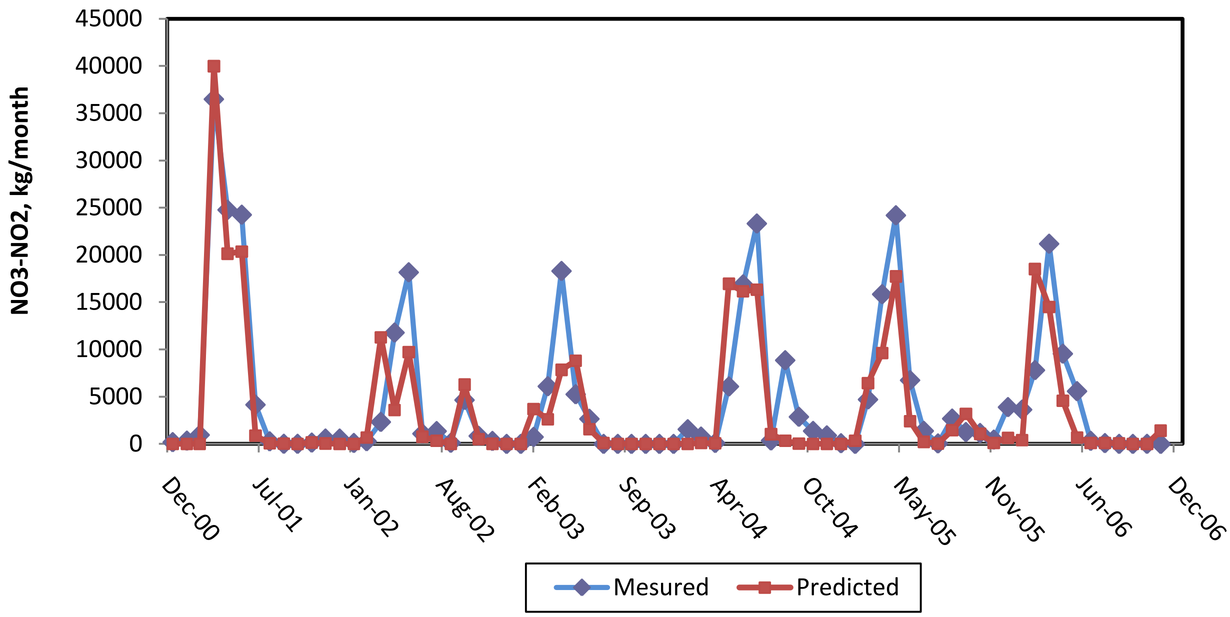


Figure 34: Average Monthly Nitrogen Loss Validation in the Beauford Sub-watershed

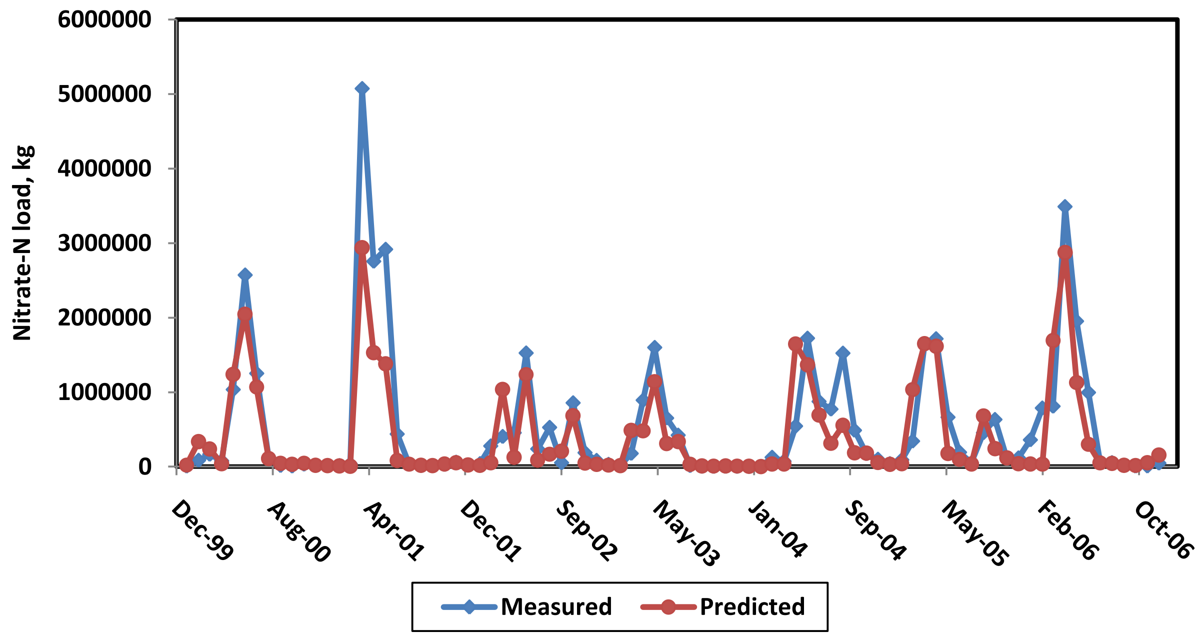


Figure 35: Monthly Nitrogen Loss Validations in the LRW

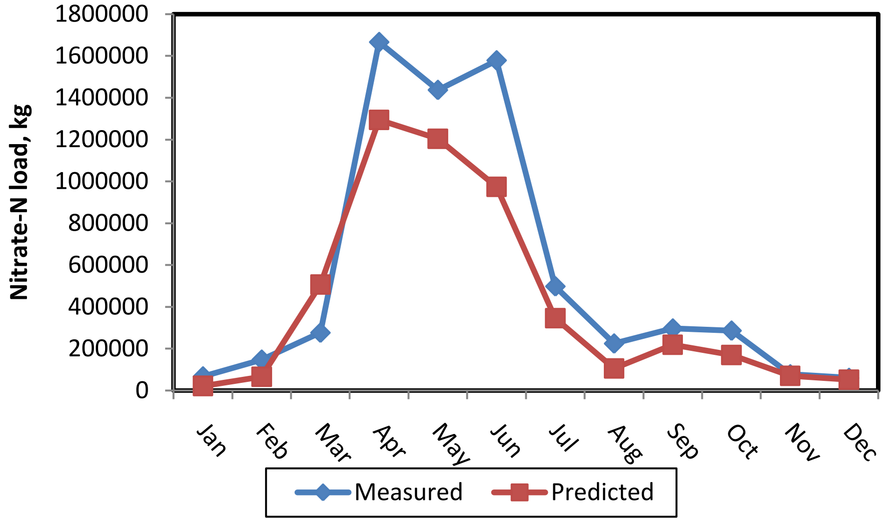


Figure 36: Average Monthly Nitrogen Loss Validation in the LRW (2000-2006)

12.3. Spatial and Temporal Distribution of Nitrogen Loss

The average annual total nitrogen load from the LRW was 15.7 kg/ha. Tile drainage accounts for 69% of the total loss. The organic and mineral nitrogen losses accounted for 8% and 92% of the total loss, respectively. Crop lands contributed 99% of the predicted total nitrogen loading to surface waters in the LRW. About 30% of the watershed area contributes half of the losses.

The months of March to July are when 86% of the LRW nitrogen losses occur. From 1994-2006, only 30% of the LRW contributed about 50% of the nitrogen loss. The general spatial distribution of nitrogen loss is shown in Figure 36. The Upper Le Sueur sub-watershed contributed 57% of the total nitrogen loss. This was disproportional to its areal coverage of 40% of the LRW. The Big Cobb (28% of LRW area) and the Maple major sub-watershed (31% of LRW) contributed 22% and 21% of the annual average nitrogen losses (Table 16).

Table 16: Nitrogen Losses from the Sub-watersheds of the LRW

kg

Reach	Area, ha	Organic-N	Nitrate- N	Total N	%
LRW	285676	41746	4879837	4921583	100
Upper Le Sueur	115286	30246	3575743	3605989	57
Big Cobb	79955	18855	1386439	1405294	22
Maple	88081	28475	1276237	1304712	21

Table 17: Annual Nitrogen Loss in the LRW (kg/ha)

Year	Organic-N	Surface-N	Tile-N	Groundwater-N	Total N
1994	0.83	2.27	8.66	1.02	12.77
1995	0.70	1.50	13.54	1.42	17.16
1996	0.98	1.72	7.49	0.69	10.88
1997	0.88	1.81	10.68	1.04	14.41
1998	0.84	1.61	9.18	0.75	12.38
1999	1.71	3.78	17.93	2.27	25.70
2000	2.60	1.98	10.21	1.53	16.32
2001	2.70	4.43	12.38	1.13	20.64
2002	0.62	0.97	8.22	0.86	10.68
2003	0.28	0.83	6.83	0.55	8.49
2004	1.80	1.78	10.68	1.83	16.09
2005	1.92	3.35	12.02	1.32	18.61
2006	1.19	4.04	13.19	1.10	19.52
Average	**1.31**	**2.31**	**10.85**	**1.19**	15.67

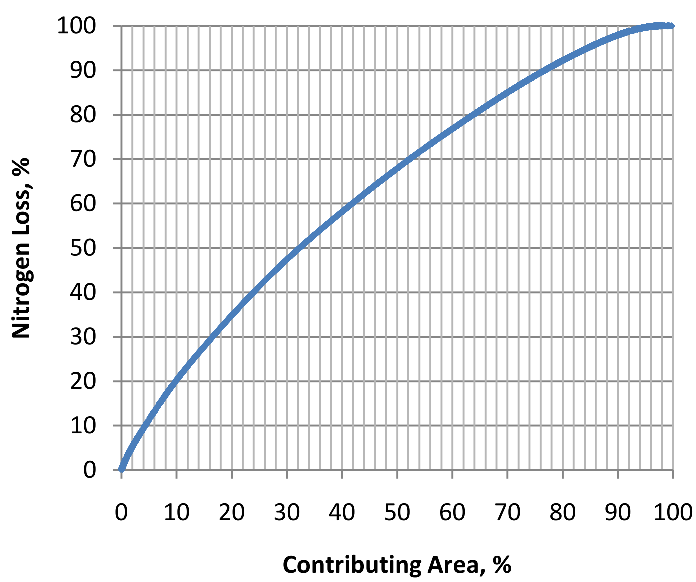


Figure 37: Contributing area vs Nitrogen loss

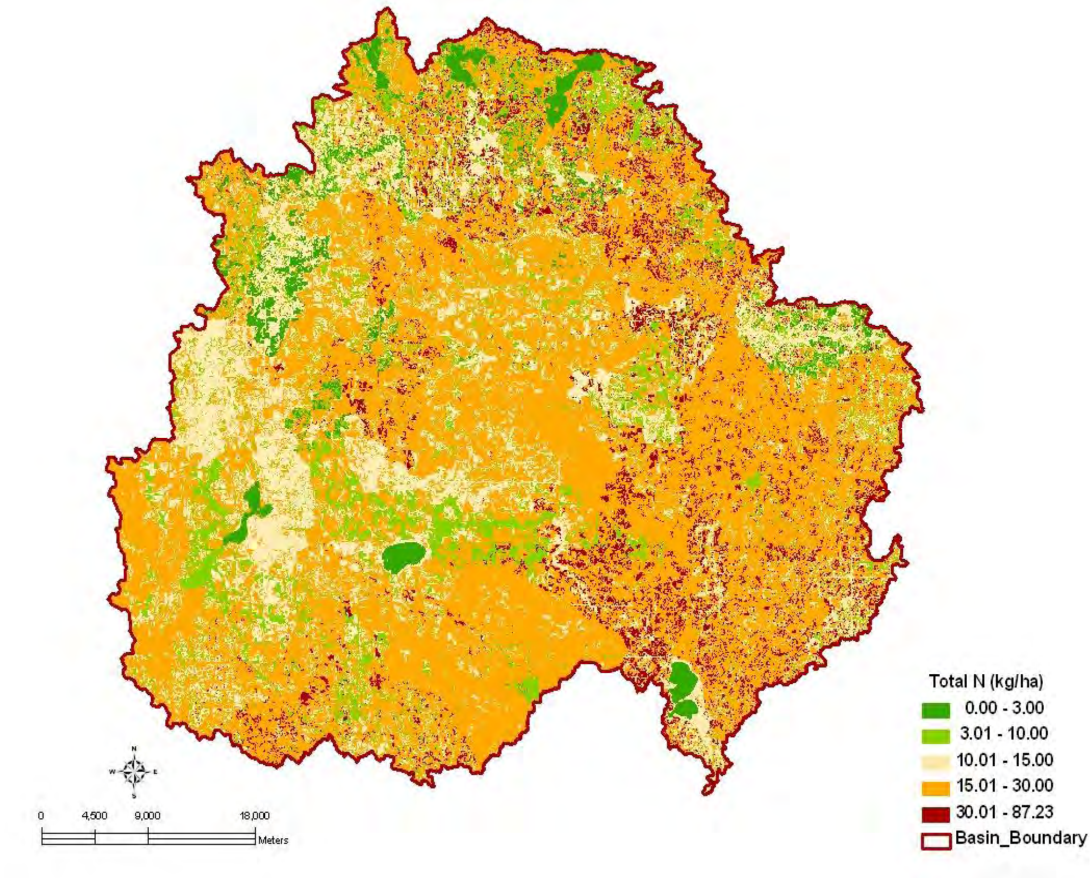


Figure 38: Nitrogen contributing areas in the LRW

12.4. Nitrogen BMPs

A. Reduced rate fall/spring nitrogen application

Fall application of 134 kg/ha of nitrogen, from both animal manure and applied AA fertilizer was evaluated against the baseline application rate of 167 kg/ha. This 20% reduction in application rate reduced total N losses by about 20%. When the reduced application was made in spring using urea fertilizer, there was only a 6.8% reduction in loss (Table 18). In the case of urea, the model predicted more tile loss and substantial increases in surface runoff losses of nitrogen. These may not be realistic for urea.

B. Conservation tillage

SWAT simulation of conservation tillage on 50% of corn residue increased the total nitrogen loss by 3%. An increase in surface residue cover reduced soil erosion and increased infiltration. Consequently, organic N losses decreased and tile drainage losses increased.

Table 18: Relative effects of application of different BMPs in reducing N loss of the LRW

Scenario	kg/ha					% Reduction
	Organic-N	Surface-N	Tile-N	Groundwater-N	Total N	
Baseline	1.3	2.3	10.8	1.2	15.7	
Reduced rate fall nitrogen application (AA)	1.1	2	8.4	0.9	12.5	20.4
Reduced rate spring nitrogen application (Urea)	1.1	2.6	10.1	1.1	14.6	6.8
Conservation Tillage on 50% corn residue	1.2	2.5	11.2	1.2	16.1	-3
No Tillage on 50% corn residue	1	2.3	10.9	1.2	15.4	1.8
VFS on CS over 2% slope	0.8	1.1	11.3	1.1	14.4	8.3
Rye as cover crop	0.9	2.5	8.7	1	13.1	16.4

C. Vegetative filter strips (VFS)

Application of VFS on corn-soy land over 2% slope showed an 8% reduction in total N loss. This was mainly due to a decrease in total organic nitrogen and surface runoff losses.

D. No-tillage

There is no substantial reduction in total nitrogen loss from the application of no-tillage on 50% corn residue. There was only a 2% reduction in the total loss.

E. Cover Crops

Planting rye after corn harvest covered the soil during the winter season and reduced the total nitrogen loss by 16%. Rye reduced the amount lost in tile drainage. Rye was planted after the October corn harvest and produced high above-ground biomass until the 3rd week of April. The high rye biomass production resulted in increased uptake of nitrogen from the soil, which otherwise would have been lost in tile drainage.

13. Pesticides

The extent of pesticide pollution of water resources can vary with application rate, properties of the pesticide, vegetation and soil types, topography and climatic conditions in which the pesticide is applied. Efforts to reduce pesticide pollution of water resources can be better informed by understanding pesticide transport processes. The LRW SWAT pesticide modeling was meant to identify pathways of pesticide losses and impacts of potential BMPS designed to minimize offsite effects. Three different pesticides with different properties were studied (Table 19).

Table 19: Physico-Chemical Properties of Selected Pesticides

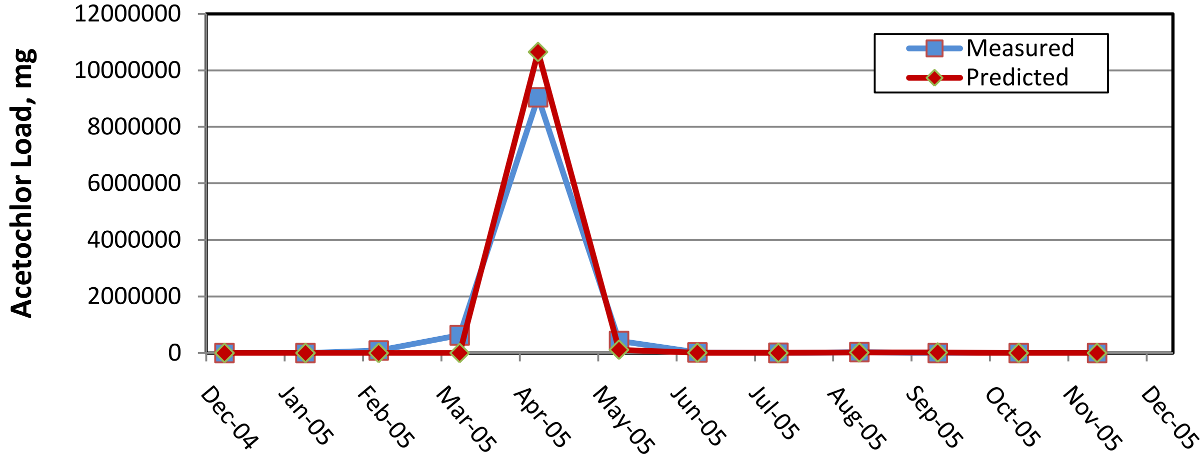
PESTICIDE	Atrazine	Metolachlor	Acetochlor
SOIL-KOC (mg/kg)/(mg/L)	100	190	150
HALF LIFE FOLIAR (days)	5	5	5
HALF-LIFE SOIL (days)	60	110	12
WATER-SOLUBLITY (mg/L)	33	530	223

13.1. Calibration

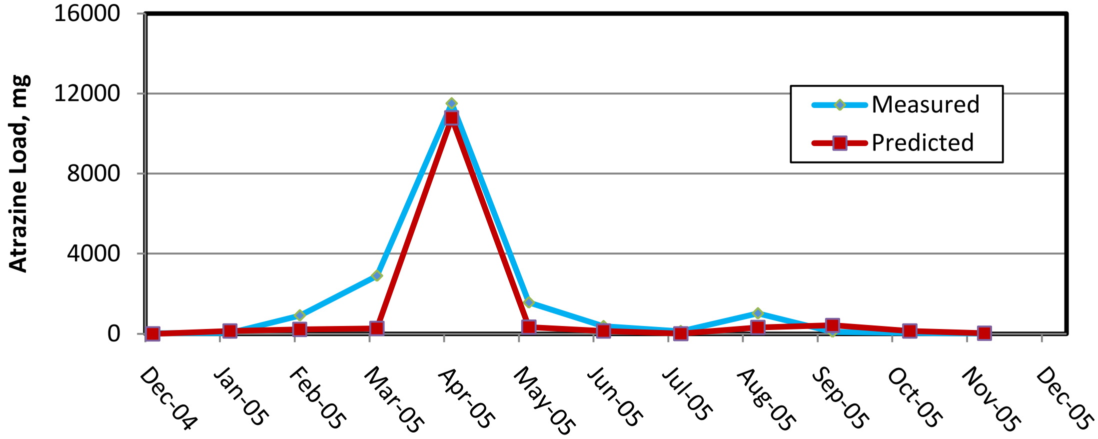
Calibration of the three pesticides was accomplished for year 2005 in the Beauford sub-watershed. The LOADEST program was used to estimate monthly losses against which the SWAT prediction was calibrated.

The predicted and observed monthly average losses of acetochlor, atrazine and metolachlor (Figure 37) have NSE values of 0.96, 0.91 and 0.71, respectively. This indicates the SWAT model can predict the monthly losses of these pesticides with great accuracy. The loss of all the three pesticides is for the most part in solution form. Comparing the three pesticides, acetochlor has the largest (93%) loss in solution form, and metolachor has the largest sorbed loss of 18% (Table 20). Of the total amount applied in 2005, about 4.1% of the applied acetochlor and 0.01% of atrazine and metolachlor reached the mouth of the Beauford sub-watershed.

A) Acetochlor



B) Atrazine



C) Metolacholr

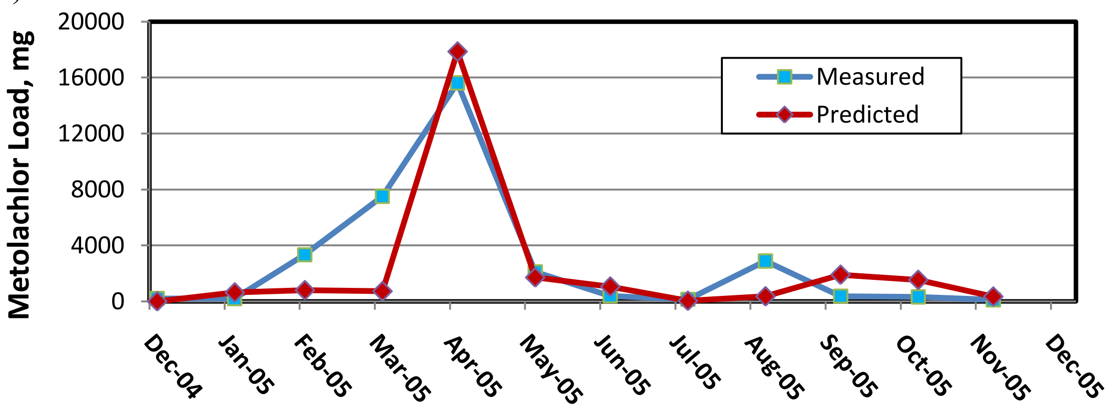


Figure 39: Pesticide Calibration Results in the Beauford Sub-Watershed

Table 20: Proportion of pesticide losses in solution and adsorbed forms

Pesticide	Solution, %	Sorbed, %
Acetochlor	93	7
Atrazine	86	14
Metolachlor	82	18

13.2. LRW Estimated Pesticide Losses

13.2.1. Acetochlor

Based on MDA field surveys, it was estimated that 35% of corn fields in the LRW receive acetochlor applications. The estimated average annual loss of acetochlor from the LRW was 0.47% of the total applied amount. Maximum losses occurred in the years 2000, 2005 and 2006, with percent losses of 1.56%, 1.59% and 1.75%, respectively. Acetochlor has a solubility of 223 mg/L and a short half life of 12 days, so it is largely lost in solution form. In the LRW, 94% of this pesticide was lost in solution. Thus, the three months of April to June are critical periods in which 99% of the acetochlor loss occurs (Figure 40 & Figure 41).

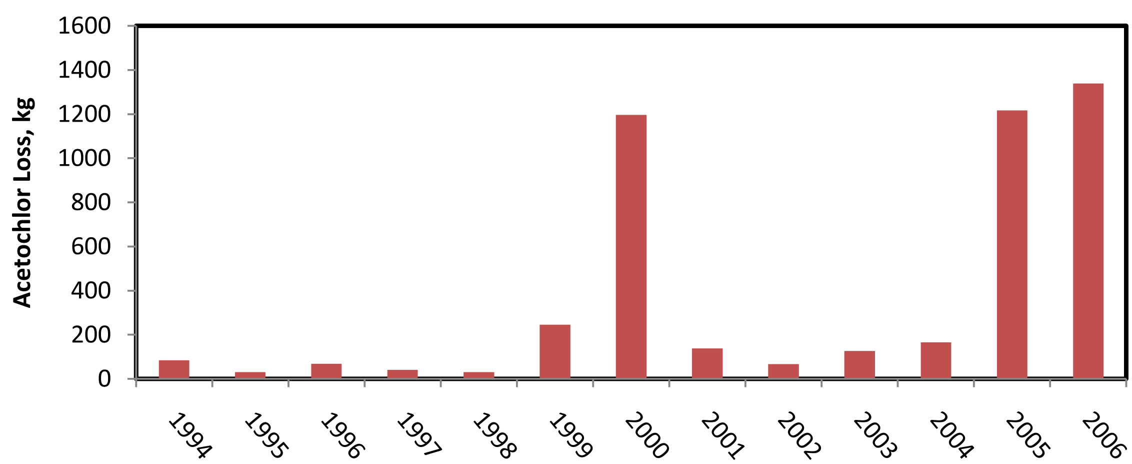


Figure 40: Annual loss of Acetochlor in the LRW

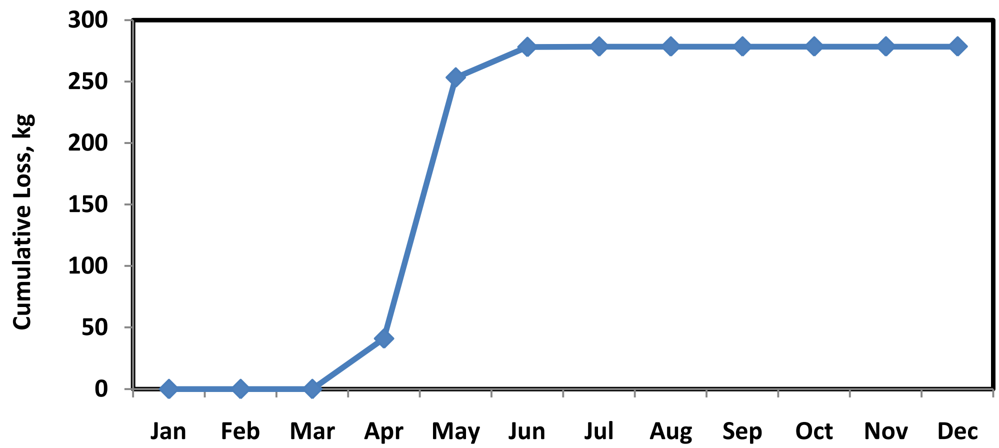


Figure 41: Average monthly cumulative losses of acetochlor in the LRW

13.2.2. Atrazine

Based on field visits and surveys made in the LRW, 15% of the corn fields in the LRW receive atrazine applications. The average application rate was 0.66 kg/ha of active ingredient.

SWAT simulation showed that 85% of the atrazine loss in the LRW occurs in the four months from April to July (Figure 43). About 82% of this loss occurs in solution form, and the remaining 18% is adsorbed to soil particles. The variability in the incidence of precipitation after application of atrazine is one of the most important factors that affects the loss in different years. The LRW has an average annual atrazine loss of about 3.2 kg or 0.02% of the total applied. The maximum loss was 4.8% in the years 2000 and 2005 (Figure 42).

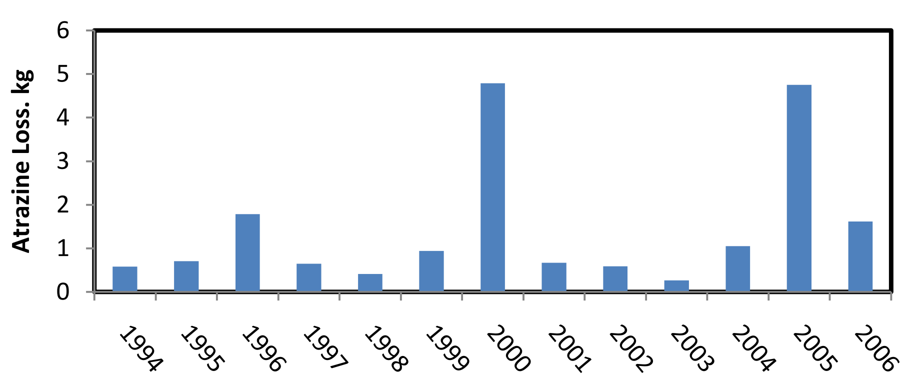


Figure 42: Annual Loss of Atrazine in the LRW

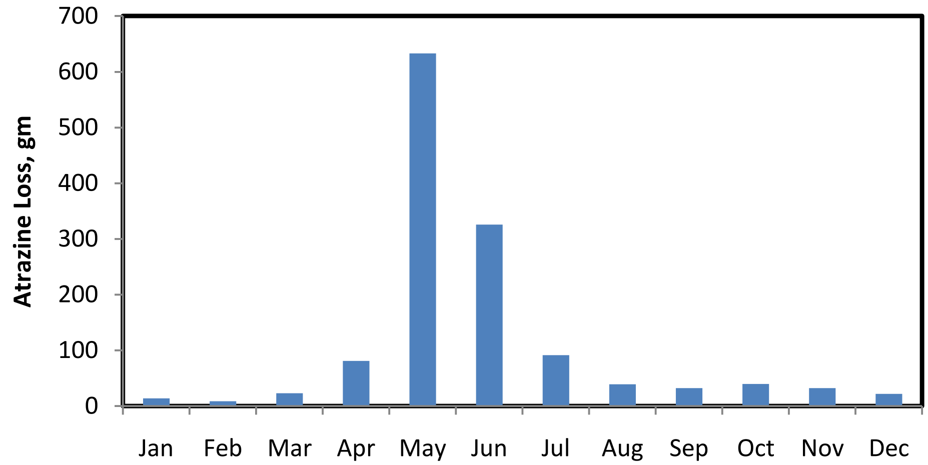


Figure 43: Average Monthly Loss of Atrazine in the LRW

13.2.3. Metolachlor

SWAT simulation of metolachlor was made assuming that 4% of the corn fields and 1% of soybean fields in the LRW receive this pesticide. The average application rate was 2.48 kg/ha for corn and 1 kg/ha for soybean.

The simulated average annual loss of metolachlor was 5.54 kg or 0.034%. The maximum loss occurred in the year 2000, and was about 33 kg. The months of April to July are critical periods in which 91% of the loss occurs (Figure 44 & Figure 45).

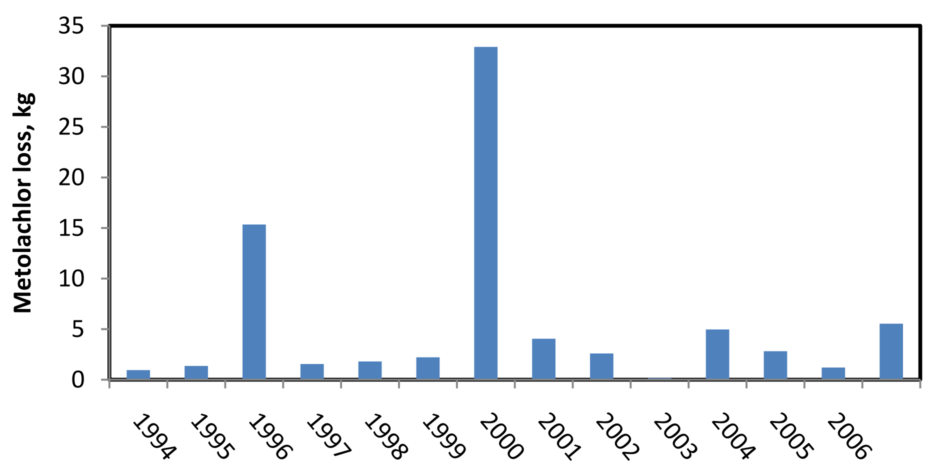


Figure 44: Annual Loss of Metolachlor in the LRW

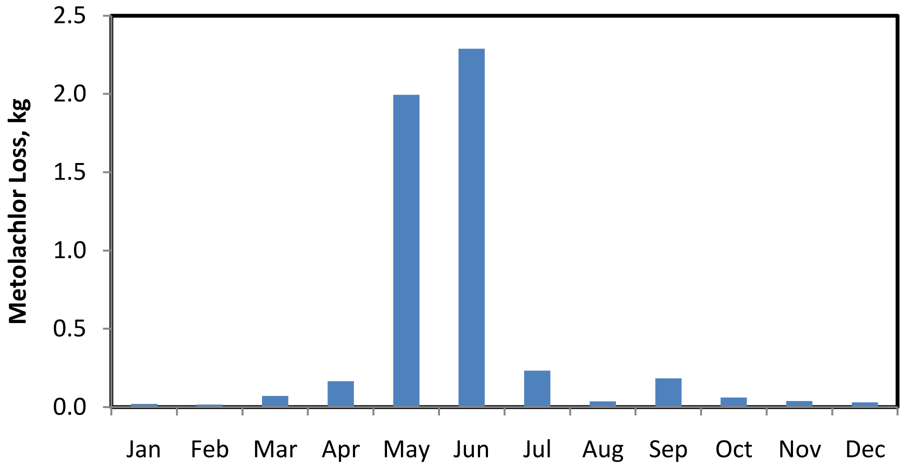


Figure 45: Average Monthly Loss of Metolachlor in the lRW

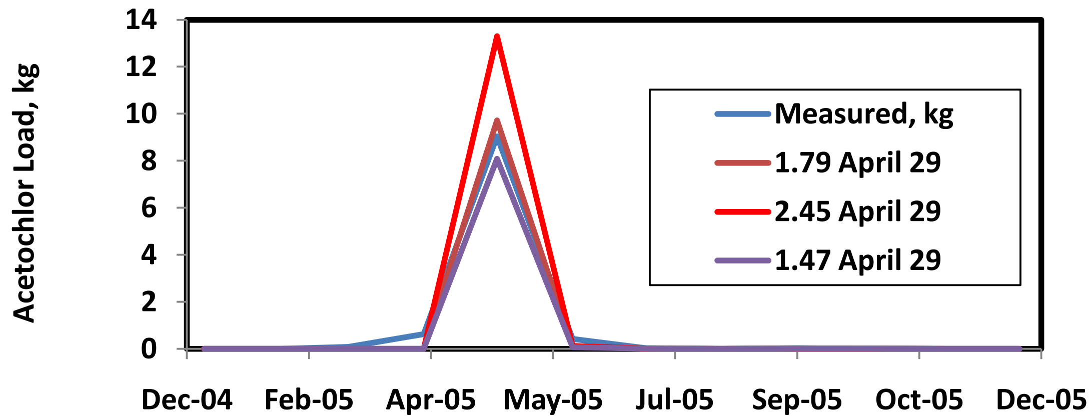
13.3. Acetochlor BMPs

A. Beauford Sub-Watershed

BMPs evaluated in the Beauford watershed for acetochlor included 1) rate of application, 2) watershed area of application, 3) timing of application, 4) incorporation of acetochlor and 5) effects of buffer strips.

The first set of alternative scenarios involved rate of acetochlor application. Rate of acetochlor was either the low or high label rates of 1.47 or 2.45 kg/ha in comparison with the baseline rate of 1.79 kg/ha. The second set of scenarios involved area receiving acetochlor. Watershed area receiving acetochlor applications was studied in two ways. The first was to apply acetochlor at a rate of 1.47 kg/ha on all CCAs and 1.79 kg/ha on all non-CCAs or to apply a rate of 1.47 kg/ha on all CCAs and 2.45 kg/ha on all non-CCAs. The second was to apply 1.79 kg/ha on 20% of the corn land or 1.79 kg/ha on 50% of all corn land. The third set of scenarios involved date of application. Application date was varied from April 29 to May 3. The fourth set of scenarios involved incorporation of acetochlor. The fifth set of scenarios involved field buffer strips. Buffer strips were investigated with different rates of acetochlor on all land receiving acetochlor, or only on CCAs.

Application rate had a significant effect on acetochlor losses (Figure 46). In comparison with the baseline simulations, applying 1.47 kg/ha reduced acetochlor losses by 17%. Applying 2.45 kg/ha increased acetochlor losses by 37%. Concentrations of acetochlor at the watershed scale were also affected by rate of application (Figure 46). Maximum concentrations were reached in the month of May. With an application rate of 1.47 kg/ha, the maximum concentration of acetochlor was 4.39 ppb, with a rate of 2.45 kg/ha the maximum concentration was 7.23 ppb.



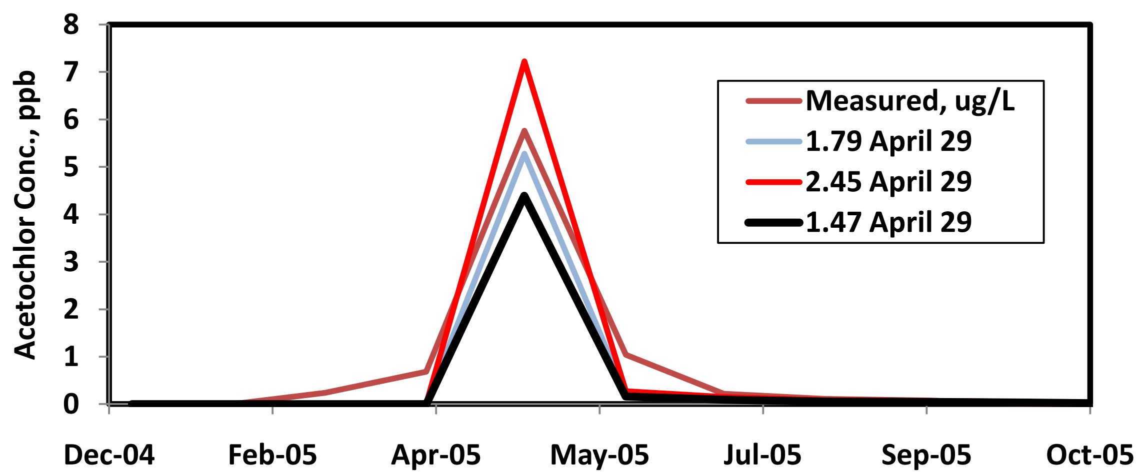


Figure 46: Effect of Changing Application Rate of Acetochlor in the Beauford Watershed.

Application of acetochlor at low label rates to CCAs had a significant effect on acetochlor losses (Figure 47). Applying 1.47 kg/ha to CCAs and 1.79 kg/ha to non-CCAs resulted in an 11% reduction of acetochlor losses relative to the baseline scenario. Applying 1.47 kg/ha to CCAs and 2.45 kg/ha to non-CCAs resulted in a 3% reduction of acetochlor losses relative to the baseline scenario. Application of acetochlor at a rate of 1.79 kg/ha to 20% of the land planted to corn reduced acetochlor losses by 56% relative to the default application on 35% of the corn land. Increasing the area receiving acetochlor to 50% of the corn land increased acetochlor losses by 117% relative to the baseline scenario.

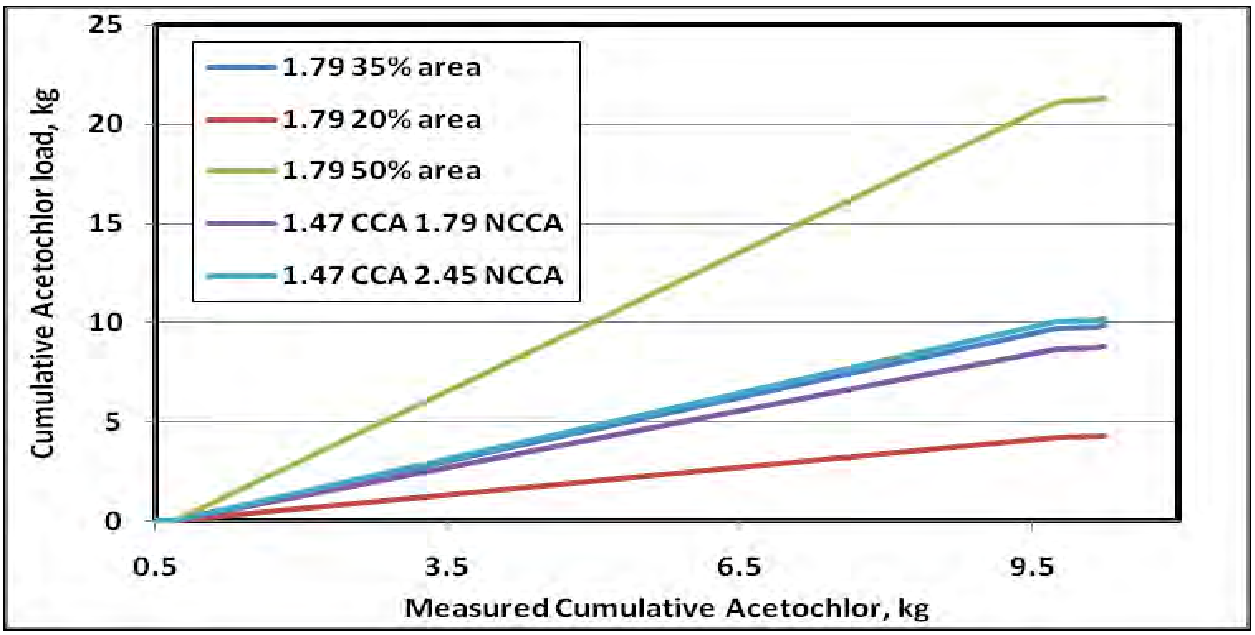


Figure 47: Effect of Watershed Application Area on Acetochlor Losses.

Application date had a significant effect on acetochlor losses, relative to the baseline application date of April 29 (Figure 48: Acetochlor Losses in Response to Application Date and Rate.). Delaying acetochlor applications until May 3 increased the losses by 31% with an application rate of 1.79 kg/ha and increased losses by 8% with an application rate of 1.47 kg/ha. This increase is due to the occurrence of storms shortly after May 3. Acetochlor should not be applied shortly before rainstorms to avoid losses.

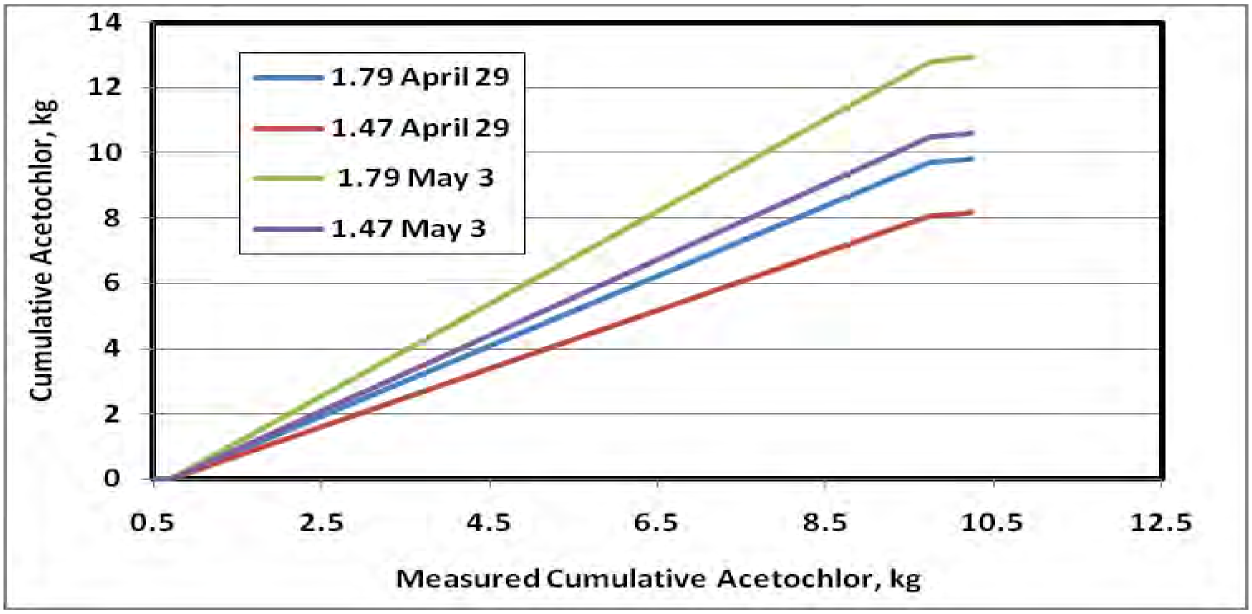


Figure 48: Acetochlor Losses in Response to Application Date and Rate.

Incorporation of acetochlor produced significant reductions in acetochlor losses relative to the baseline scenario with no incorporation (Fig. 47). At the lowest rate of acetochlor application (1.47 kg/ha), incorporation reduced acetochlor losses by 95% relative to the baseline scenario with an application rate of 1.79 kg/ha.

Field buffer strips had a significant effect on acetochlor losses (Figure 50: Effect of Buffer Strips on Acetochlor Losses.). Width of the buffer was somewhat important. With a 33 ft. wide buffer applied throughout the watershed, acetochlor losses were reduced by 68% relative to the situation without buffers. With a 66 ft. wide buffer everywhere, losses were reduced by 89%. Buffers were more effective at lower rates of acetochlor application. Installing buffer strips only in CCAs reduced acetochlor losses by roughly 50%, relative to the baseline scenario.

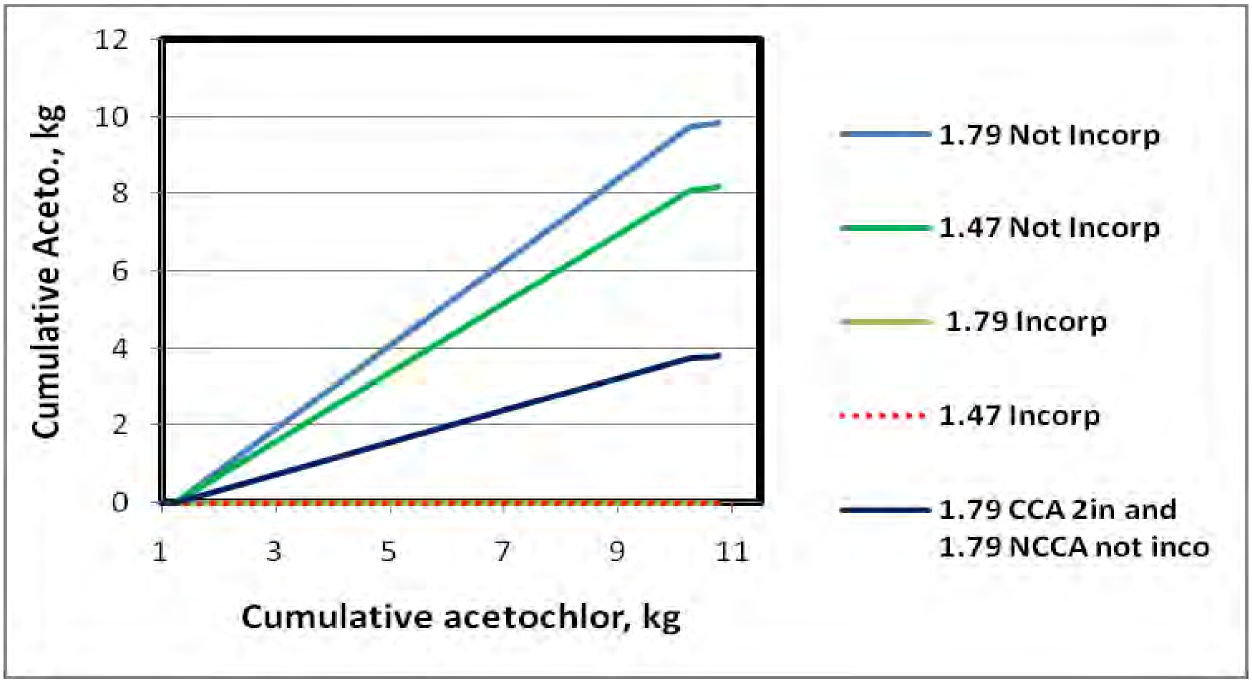


Figure 49: Effect of Acetochlor Incorporation on Acetochlor Losses.

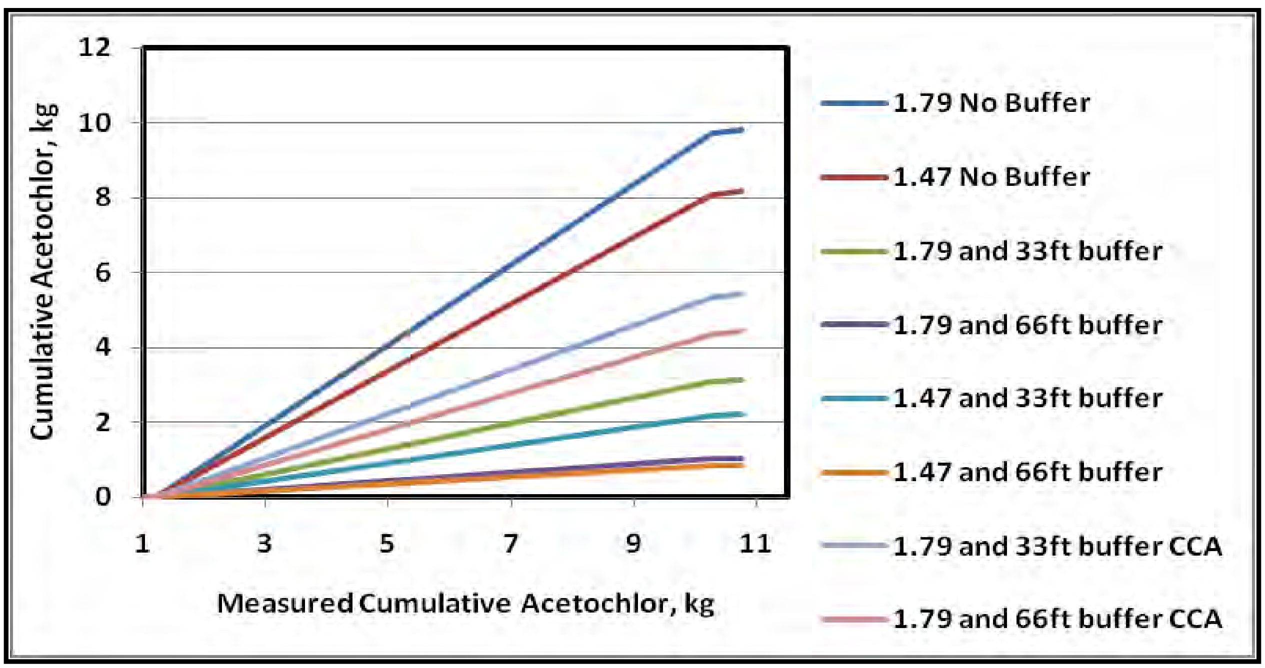


Figure 50: Effect of Buffer Strips on Acetochlor Losses.

B. Acetochlor BMPs for the LRW

Based on the model simulation results at the Beauford sub-watershed, six acetochlor best management practices were selected and applied to the LRW. The BMPs were:

1) reduced rate of application, 1.47 kg/ha
2) reducing application area from 35% of the watershed to 20%
3) change in application time, post emergence on May 3rd
4) incorporation of acetochlor
5) 33ft buffer strips on all corn and soybean fields
6) 33ft buffer strips on CCAs.

The simulation results showed that incorporation of acetochlor reduces the loss by over 95%. This practice may not be feasible for adoption by farmers, as it requires significant changes in tillage and manure management. Reducing application rate to 1.47 kg/ha and application area to 20% of the corn land, and establishing buffer strips reduces the loss by 18%, 62% and 73%, respectively. Buffer strip establishment exclusively on CCAs reduced the loss by 14%. A change in application time to post emergence on May 3 reduces the loss by only 9% (: Relative Importance of Acetochlor Management Practices in the LRW).

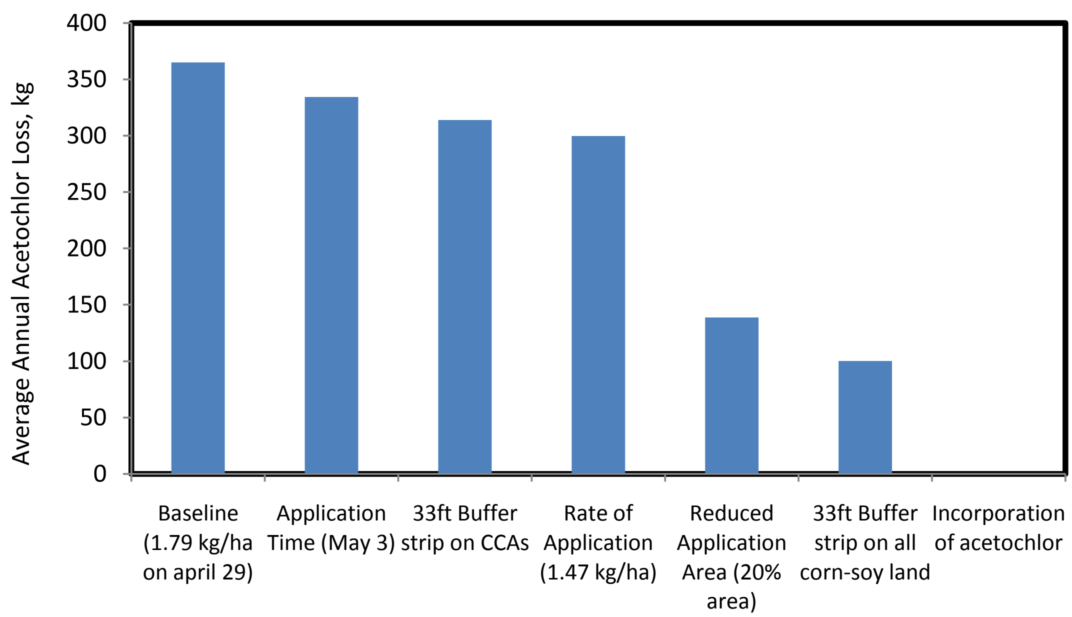


Figure 51: Relative Importance of Acetochlor Management Practices in the LRW

14. CONCLUSIONS

Calibration and validation of the SWAT model in the Beauford sub-watershed of the Le Sueur River watershed was satisfactory for all parameters investigated with the exception of sediment at the scale of the entire Le Sueur River watershed. Calibration of the SWAT model for discharge gave NSE values of 0.77 in the Beauford sub-watershed. Validation for discharge gave NSE values of 0.89 in the Beauford sub-watershed and 0.73 in the LSR watershed. Calibration of the SWAT model for sediment in the Beauford sub-watershed gave an NSE value of 0.71. Extrapolating the calibrated model to the LSR watershed showed that upland areas contributed as little as 14% of the overall sediment load measured in the LSR watershed. Over 90% of the upland areas generate less than 5 t/ha of sediment. The remaining sediment is assumed to come from bluffs, stream banks and ravines which are not simulated by SWAT. Upland sediment loads can be reduced by 12% by implementing conservation tillage on all corn residues from slopes steeper than 2%. Installing grass filter strips on all cropped landscapes steeper than 2% gives a 56% reduction in sediment losses.

Calibration of the SWAT model for phosphorus losses gave NSE values of 0.79 in the Beauford sub-watershed and 0.67 in the LSR watershed. Roughly 75% of the phosphorus losses were in the particulate form, the remaining 25% was lost as soluble P. Upland agricultural areas had average phosphorus losses of 1 kg/ha. Phosphorus losses could be reduced by 28% using reduced application rates of fertilizer. Conservation tillage on corn residue in fields steeper than 2% reduced P losses by 15%. Installation of grass filter strips on all cropped landscapes steeper than 2% reduced P losses by 64%. Planting a fall cover crop of rye reduced P losses by 29%.

The SWAT model accurately predicted nitrate-N losses from the LSR watershed. Calibration NSE values for nitrate-N were 0.74 in the Beauford sub-watershed and 0.74 in the LSR watershed. The average loss of nitrogen from upland areas was 15.7 kg/ha, and 69% of this loss occurred in tile drains. Reducing the rate of fall applied anhydrous ammonia reduced nitrate-N losses by 20%. Planting a fall cover crop of rye reduced nitrate-N losses by 16%.

Acetochlor is primarily lost in the soluble phase through surface runoff and tile drainage. Buffer strips installed in CCAs are very effective at reducing acetochlor losses. Incorporation of aceotchlor is very effective at reducing losses. The adoption of this practice has to be considered in the context of associated nitrogen fertilizer and tillage practices. Reductions in the area receiving acetochlor are very effective at reducing acetochlor losses. Reductions in application rate are somewhat effective at reducing acetochlor losses, especially if practiced in critical areas. It is important to apply acetochlor well in advance of spring rainstorms to reduce losses. There is no systematic benefit of delaying pre-plant acetochlor application from late April to early May due to the high frequency of spring rainstorms. Growers have a wide range of acetochlor BMPs to choose from depending on their specific management approach and availability of money, time and equipment.

References

Minnesota State University. October, 2000. LE SUEUR RIVER MAJOR WATERSHED DIAGNOSTIC REPORT. Rep. 951-1-194-07. Minnesota State University, Mankato, Mankato.

Moriasi D., Arnold, J., Van Liew M., Bingner R., Harmel R. and Veith T. 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. Trans. ASAE 50:885-900.

Mulla, D.J. 1997. Minnesota river basin water quality overview. Minnesota Extension Service, University of Minnesota, .

Nash, J.E. and Sutcliffe, J.V. 1970. River flow forecasting through conceptual models; part 1–A discussion of principles. *J. hydrol.* 10:3, pp. 282–290.

Neitsch, S.L., Arnold J.G., Kiniry J.R., Srinivasan R., and Williams J.R.2002. Soil and water assessment tool User's manual version 2000. Rep. GSWRL Report 02-02, BRC Report 02- 06. Texas Water Resources Institute TR-192, College Station, TX.

Neitsch, S., Arnold J.G., Kiniry J.R., and Williams J.R.2005. Soil and water assessment tool theoretical documentation, version 2005. Temple, TX.USDA Agricultural Research Service and Texas A&M Blackland Research Center.

Neitsch, S.L., Arnold J.G., Kiniry J.R., Srinivasan R., and Williams J.R.2004. Soil and water assessment tool Input/Output file documentation. version 2005. Temple, TX: Blackland Research Center, USDA Agricultural Research Service.

Randall, G.W., and Iragavarapu T.K. 1995. Impact of long-term tillage systems for continuous corn on nitrate leaching to tile drainage. J. Environ. Qual. 24:360–366.

Runkel, R.L., Crawford C.G., Cohn T.A. and Geological Survey (US. 2004). Load estimator (LOADEST) a FORTRAN program for estimating constituent loads in streams and rivers. US Dept. of the Interior, US Geological Survey.

Sharpley, A.N., Weld J.L., Beegle D.B., Kleinman P.J.A., Gburek W.J., Moore P.A., and Mullins G.2003. Development of phosphorus indices for nutrient management planning strategies in the U.S. J. Soil Water Conserv. 58:137–152.

USDA, N.M. 2008. Rapid watershed assessment resource profile le sueur (MN) HUC . 7020011. www.mn.nrcs.usda.gov/technical/rwa/Assessments/reports/le_sueur.pdf. USDA, NRCS MN, Minnesota.

PART 2: SEDIMENT, NUTRIENTS AND PESTICIDES MODELING IN THE SOUTH BRANCH OF THE ROOT RIVER, MINNESOTA

Brent J. Dalzell and David J. Mulla
Department of Soil, Water and Climate
University of Minnesota

Submitted to: Minnesota Department of Agriculture

South Branch of the Root River: Watershed scale modeling and evaluation of alternative management practices.

Summary: A watershed modeling approach was applied to assess current water quality in the South Branch of the Root River (SBRR) watershed located in southeastern Minnesota. Observed water quality data were used to calibrate the model for 2004-2005 and validate the model for 2006-2008. Model calibration/validation ranged from good to fair for mean monthly flow as well as monthly sediment and phosphorus loads. The ability to predict monthly nitrogen loads was less successful and the model failed the calibration period; primarily due to over-predicted loads in 2004 (the model did perform satisfactorily during the validation period, however). Problems with N prediction in 2004 may be due to weather and/or the fact that SBRR is influenced by karst geology and has a dominant groundwater component.

Assessment of baseline conditions (average annual results from 10 years of simulation data) shows that agricultural lands occurring on slopes greater than 4% are disproportionately the greatest contributors of sediment and phosphorus to the stream. These landscape elements comprise 8% of the area of SBRR yet they contribute 50% of the edge-of-field sediment losses predicted by the model.

Edge-of-field phosphorus losses followed a similar trend to sediment losses and baseline results showed that agricultural lands with slopes steeper than 4% contributed 63% of the average annual phosphorus loss. Simulated export of NO_3-N was primarily influenced by differences in flow characteristics in various sub-basins. Sub-basins in which subsurface tile drainage was common (presence assumed in poorly-drained soils) exhibited high NO_3-N export via lateral flow (which includes tile drainage). In sub-basins where karst geology dominates, simulated NO_3-N export occurs predominately via flow of the shallow aquifer. Overall, the primary means by which NO_3-N is exported in SBRR is shallow groundwater while subsurface tile drainage has a secondary influence.

The calibrated and validated model was used to evaluate the effectiveness of various alternative management practices intended to reduce the delivery of sediment and nutrients from agricultural fields to the SBRR. For sediment and phosphorus, alternative management scenarios show that steep landscape elements (slope > 4%) represent the greatest potential for reductions in sediment and phosphorus loading. When targeted to croplands on slopes of greater than 4%, individual practices of: conservation tillage, cover cropping, or 10m filter strips reduced

simulated field erosion by 8,32, and36% respectively. In a comprehensive scenario: croplands steeper than 4% received winter cover crops in addition to filter strips while the flatter cropland in the watershed received conservation tillage. This scenario resulted in a reduction of field erosion by 53% when compared to the baseline scenario.

Overall, substantial sediment and phosphorus reductions (53 and 28%, respectively) can be achieved via implementation of cover crops and filter strips on steep croplands (slopes > 4%) and conservation tillage on all remaining cropland. Reductions in nitrogen losses are approximately equal to reductions in commercial fertilizer application rates with further benefits expected from filter strips and cover crops.

Organization of this chapter:

This chapter details the application of a watershed-scale modeling approach to assess current conditions as well as evaluate alternative management scenarios in the South Branch of the Root River (SBRR) watershed located in southeastern Minnesota. The primary focus of this modeling effort was to identify which portions of SBRR are most likely contributing to water quality problems as well as to test the effectiveness of alternative management scenarios in reducing nonpoint source pollution in the watershed. The chapter is divided into three main sections:

1) Model description, study area, and background.
2) SWAT model calibration and validation.
3) Evaluation of current watershed conditions and alternative management scenarios.

Model Description, Study Area, Model Inputs

Model Description

The model selected for this work is the Soil and Water Assessment Tool (SWAT). SWAT is a watershed-scale model that functions on a daily time step; it is primarily applied to predict and evaluate land cover and land management practices on the quantity and quality of water that is exported from watersheds with agricultural land use. The model is physically-based and relies on environmental parameters and plant growth to estimate the amount of water available in the landscape to contribute to stream flow and delivery of sediment, nutrients, and pesticides to the watershed outlet. The SWAT model was selected for this work because it is freely available, it has a large user base and is actively being supported and developed. Further, it has a great degree of flexibility to allow simulation and evaluation of a wide variety of alternative crops and land management practices. SWAT has been used widely for the study of water quality in agricultural regions and has been applied to TMDL studies. For a comprehensive review of the SWAT model in the scientific literature, please refer to Gassman et al., 2007.

Study Area

This study is being performed on the South Branch of the Root River (SBRR) Watershed, a tributary to the Root River located in southeastern Minnesota (Fig 52). The western portion of the watershed is generally flat and dominated by corn and soybean row crop agriculture. The eastern portion of the watershed includes steeper slopes and increasing amounts of forest, range, and grassland land cover; row crops are still the dominant land use, however, with some animal operations also present. An important characteristic of this watershed is the fact that the eastern portion of SBRR is influenced by karst geology and many surface features such as sinkholes serve as a reminder of the rapid communication between the surface and the groundwater in this landscape (Fig 52). The outlet of SBRR is situated within Forestville State Park, where daily flow measurement and periodic water quality monitoring is being conducted and serves as a benchmark for comparison of model results.

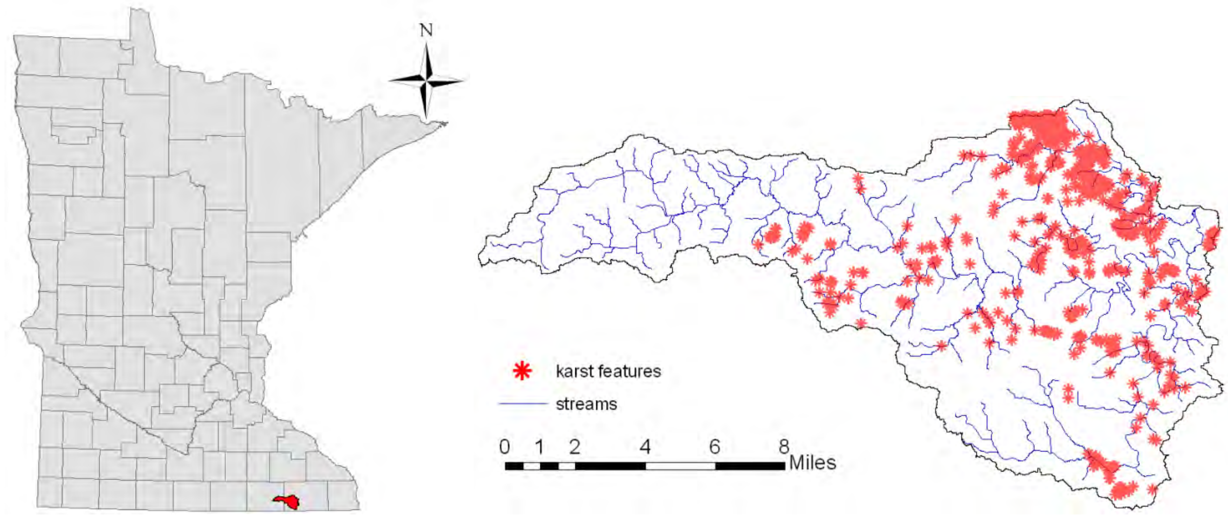


Figure 52. Location map showing the South Branch Root River watershed in southeastern Minnesota. Right: stream network and location of karst features in the watershed.

Input Data

Spatial Data – all data used for this project were projected in UTM coordinates (NAD 83, Zone 15N).

Digital Elevation Model – Stream network and slope data were determined from a 10-meter (1/3 arc second) Digital Elevation Model (DEM; Figs 53 and 54) obtained from the National Map Seamless Server (http://seamless.usgs.gov/index.php). More information about the National Elevation Dataset is available at: http://ned.usgs.gov/.

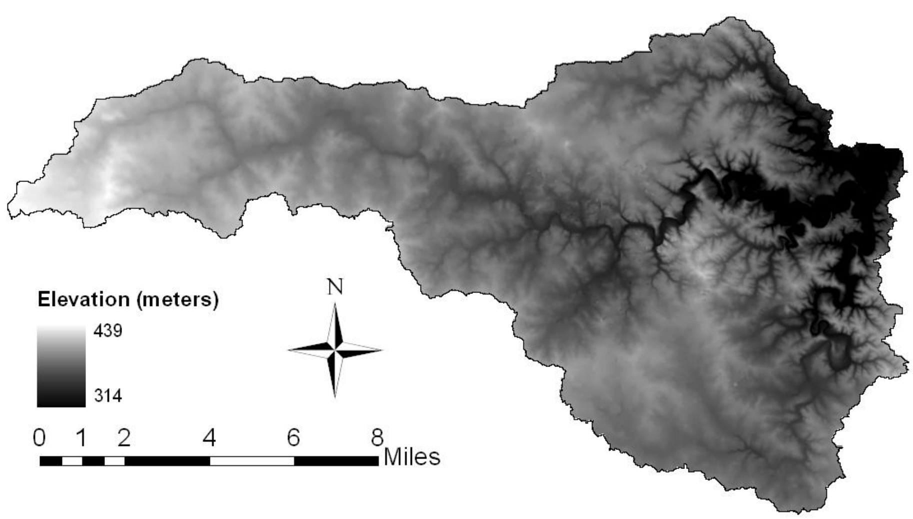


Figure 53. Digital elevation model of SBRR watershed (pixel size = 10m)

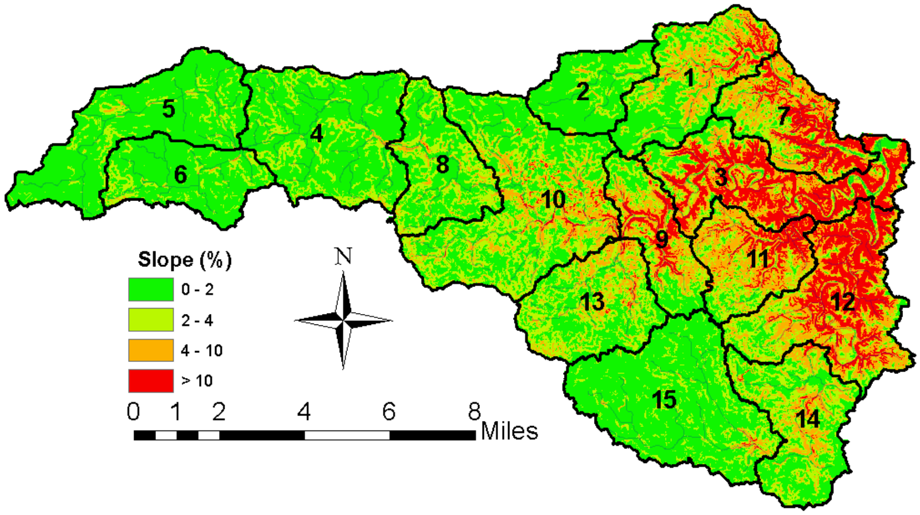


Figure 54. Slope map of SBRR (calculated from DEM).

Land Cover and Land Use information were determined from the 2001 National Land Cover Database (NLCD: http://www.mrlc.gov/index.php); also available via the National Map Seamless Server. Pixel size for the 2001 NLCD is 30-meters. In order to reduce the number of functional units handled by the SWAT model, some of the smallest land cover classes were aggregated into similar classes. These changes are summarized in Table 19; the resulting land cover map was used as input for the SWAT model (Fig 55). The aggregation step was taken as a way to maintain model computation efficiency while simplifying some of the least prevalent land cover classes.

Table 19. Original and revised classifications for land cover / land use input for the SWAT model.

Initial 2001 NLCD Classification	Watershed Area (%)	New Land Cover Class
Urban - Medium Density	0.96%	Urban - Low Density
Urban - High Density	0.05%	Urban - Low Density
Urban - Industrial	0.01%	Urban - Low Density
Barren Land	0.02%	Urban - Low Density
Forest - Evergreen	0.08%	Forest - Deciduous
Wetland - Non Woody	0.15%	Wetland - Woody

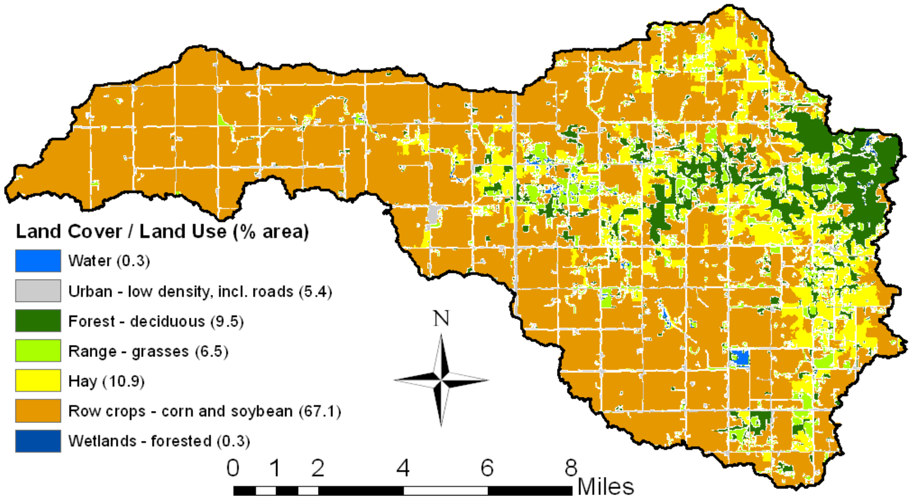


Figure 55. Land cover / land use classification for SBRR watershed. (2001 national land cover dataset)

County level ***Soils Data*** are derived from the SSURGO soils database. Data were downloaded from the web soil survey maintained by the USDA-NRCS (http://websoilsurvey.nrcs.usda.gov/app/) and processed into a format amenable to the SWAT model. User-defined soils data tables were provided by the SWAT development group (R. Srinivasan) located at Texas A&M University, College Station, TX. Four (out of 63) of the soil map units present in the SSURGO data were not found in the user-defined soil tables:

- *Alluvial land*
- *Escarpments*
- *Mantorville*
- *Mixed alluvial land*

These soils were re-named to match adjacent soil map units that had similar hydrologic groups in order to successfully generate input for the SWAT model.

Weather data – the SWAT model requires daily values of:

- Precipitation
- Temperature
- Relative humidity
- Wind speed, and
- Incoming solar radiation

in order to simulate plant growth and water movement. Weather data were collected from the closest available monitoring stations; this is especially important for precipitation which can vary greatly over small distances.

Long term ***precipitation and temperature data*** are available for stations located near the towns of Grand Meadow and Preston, MN, which are located approximately 2 and 7 miles from the watershed boundary, respectively. Data are also available at Spring Valley (about 1 mile from the watershed); unfortunately, these data are only complete after October 2004. For periods where Spring Valley precipitation data were available, they were used. For periods where the Spring Valley data were not complete, missing values were filled in as the average of the next two closest gauge stations (Grand Meadow and Preston). Based on the location of precipitation gauges, data from Grand Meadow and Spring Valley were used as model inputs. Daily minimum and maximum temperature were taken from Grand Meadow and Preston. Missing temperature values were filled in with data from the closest available station.

Wind speed and relative humidity data were taken from stations in LaCrosse, WI and Minneapolis, respectively. ***Solar radiation*** data were generously provided by the Minnesota Climatology Working Group (St. Paul).

Sub basins – Within the SWAT model environment, the watershed can be further divided into smaller watersheds called sub basins. This permits some flexibility for model calibration as well as making modifications to management scheduling and targeting alternative management practices. For SBRR, sub basins were delineated based on major land cover and slope characteristics as well as the presence/absence of karst features. The resulting sub basins are presented in Figure 56.

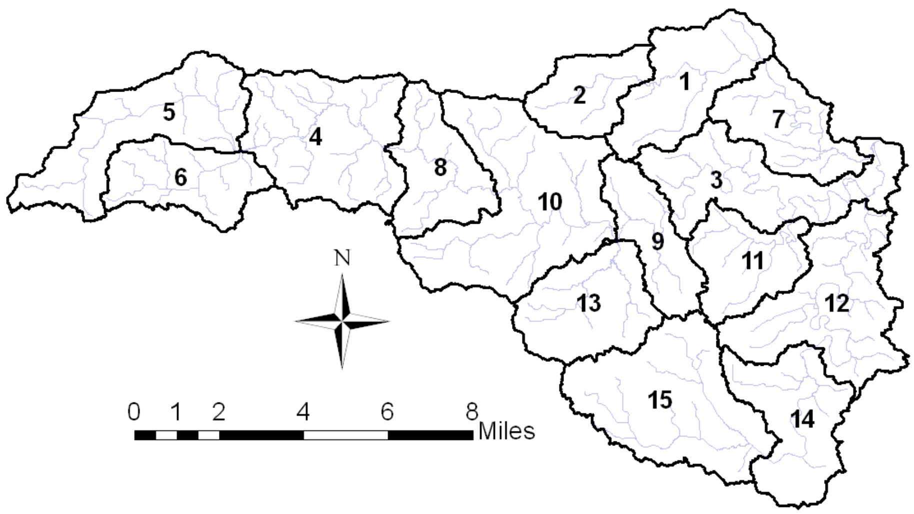


Figure 56. Map of SBRR showing sub basin boundaries.

Physical Watershed Characteristics

The SWAT model includes many parameters that directly reflect physical watershed characteristics including measures of stream channel dimensions, roughness, and hydraulic conductivity of the channel bed material. For this study, these values were directly measured at selected sites in the watershed by Dr. Toby Dogwiler (Winona State University, Department of Geoscience). Values for channel dimensions and roughness are summarized in Table 20.

Table 21. Physical channel dimensions and Manning's roughness values for several sites in SBRR.

Site Number*	Site Name	SWAT SubBasin	Bankfull width of main channel (m)	depth of main channel (m)	width/depth ratio	mannings roughness coefficient (n) for the main channel
1	SBRR SR2	5	12.47	2.54	4.91	0.060
2	SBRR SR3	6	12.08	1.27	9.51	0.040
3	SBRR Site 3 at Hwy 14	4	22.50	2.96	7.60	0.035
4	SBRR Trebiste Reach	8	18.50	1.22	15.16	0.025
6	SBRR 151st Ave	10	22.80	1.41	16.17	0.035
7	Etna Cr 153rd Ave	13	11.00	1.04	10.58	0.055
8	SBRR Tart Site	9	20.60	1.23	16.75	0.065
12	SBRR at confluence with Canfield Cr	3	19.30	1.01	19.11	0.030
13	Forestville Cr	7	12.00	1.45	8.28	0.040
14	Canfield Cr confluence with SBRR	12	16.10	1.18	13.64	0.024
15	Canfield Cr 201st Ave	14	11.40	1.89	6.03	0.055
17	Canfield Cr Stockdale site	15	56.20	1.49	37.72	0.065

* corresponds with site number from report by Dr. Toby Dogwiler. (does not correspont with SWAT sub basin numbers)

Measurements of the hydraulic conductivity of the channel material were made at 2 different watershed sites during varying hydrograph conditions (e.g., rising vs. fall hydrograph). Even though a range of hydraulic conductivity values were measured, SWAT model input allows one value for each sub basin; average values were used. The average hydraulic conductivity measured in Etna creek was 37 mm hr^{-1}; this value was applied to all sub basins considered to be more representative of areas not strongly influenced by karst geology. The average value measured at the Mystery Cave site was 66 mm hr^{-1}; this value was applied to sub basins influenced by karst (karst and non-karst influenced sub basins are discussed in the section on model calibration).

A series of ***management schedules*** were developed to represent typical practices within SBRR watershed. Planting and harvesting dates for corn and soybeans were based on average values determined from weekly crop reports from 1997 to 2007. Tillage practices and rates and timing of fertilizer and manure application were based on valuable information contained in the two FANMAP (Farm Nutrient Management Assessment Program) surveys conducted on SBRR in 2003 and 2007. It should be noted that these management schedules are representative of an average farm in SBRR based on the data available for the period studied and do not necessarily reflect any individual farm in SBRR.

During the 2003 cropping season, there were roughly equal amounts of corn and soybean land planted in SBRR (2003 FAMAP report). The general trend in SBRR may be towards greater corn acreage as reflected in the 2007 FANMAP report, (this may reflect a temporary trend due to fluctuating corn prices, however). Increases in corn acreage are not simulated here and row crop land is in SBRR is assumed to be in a corn-soybean rotation for the duration of the study.

The basic corn/soybean management schedule in SBRR is summarized in Figure 57.

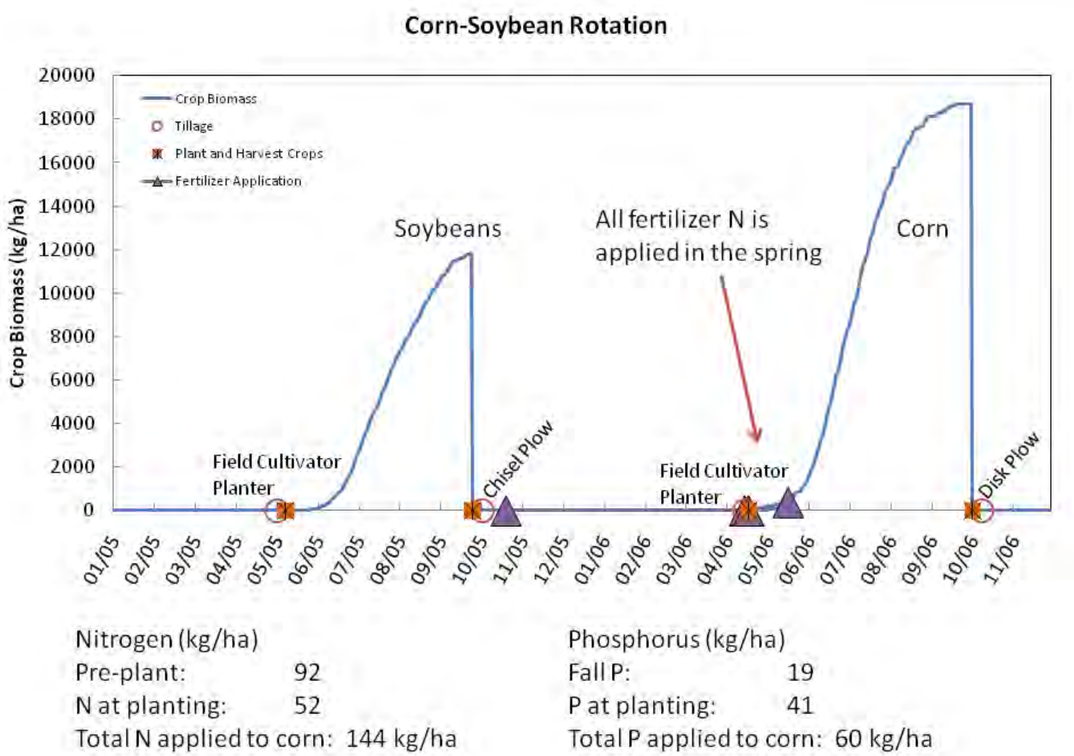


Figure 57. Typical management schedule for a corn-soybean rotation field in SBRR.

In order to account for anhydrous ammonia (A.A.) applied to corn acres (described in the FANMAP surveys), a separate management schedule was developed to allow fall application of A.A. (Fig 58). This management schedule was applied to three sub basins in the western portion of the watershed (based on local knowledge) and includes enough corn acreage to roughly account for the amount of A.A. reported in the 2003 and 2007 FANMAP surveys.

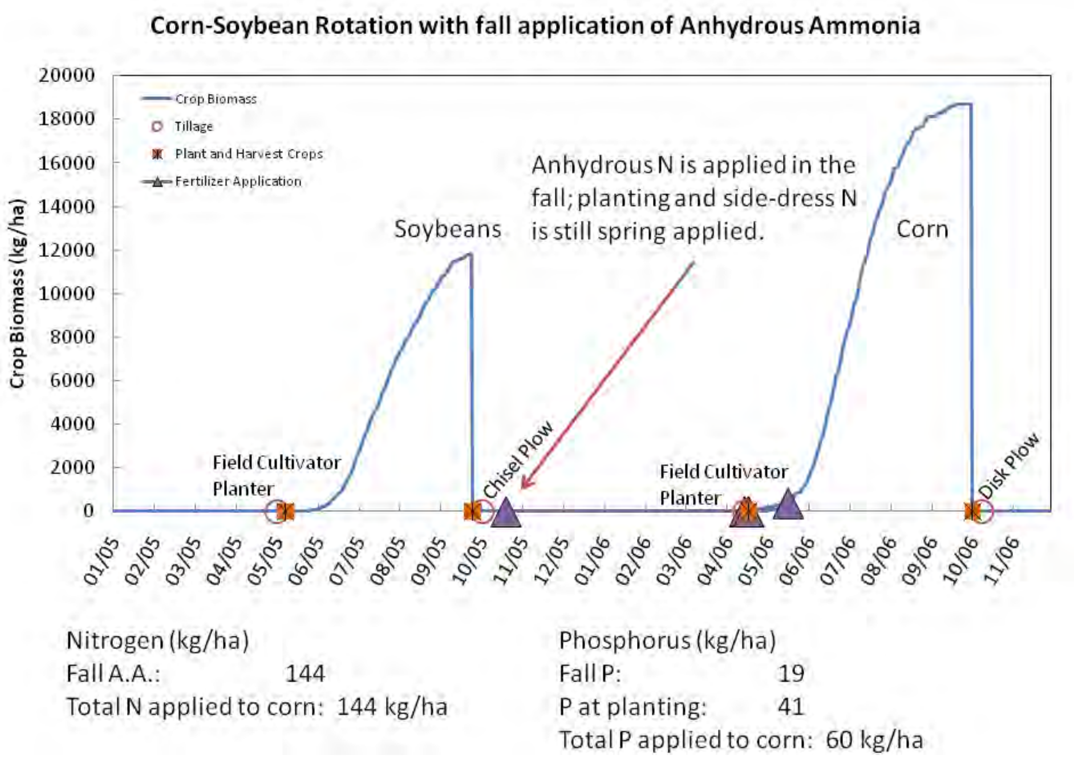


Figure 58. Typical management schedule for a corn-soybean rotation of fields receiving fall application of anhydrous ammonia in SBRR.

Manure Application

Based on the 2003 and 2007 FANMAP surveys, it is estimated that animal manure is applied to approximately 8% of crop land in SBRR during any given year. The major sources of manure in SBRR are Swine, Dairy, and Beef operations. A large portion of manure from beef cattle, however, is not collected and it is assumed that the main sources of manure to cropped lands in SBRR are from Dairy cattle and Swine. The FANMAP survey is not a comprehensive survey of all livestock producers in the watershed; it is uncertain how representative the results from the survey are when applied to the entire watershed. The amount and nutrient content of manure produced was based animal numbers obtained from the FANMAP survey and University of Minnesota Extension Documents *(summarized in a Minnesota Department of Agriculture information sheet titled: Useful Nutrient Management Data).*

Based on information from the FANMAP surveys, Swine Manure was applied to sub basins 4 and 15 while Dairy manure was applied to sub basins 1 and 7. It is important to note that this is a simplification of actual manure management practices within SBRR and fields receiving manure are more distributed throughout the watershed. However, this approach is based on the general distribution of animals in SBRR and results provide insight into how manure application may influence nutrient losses from row cropped fields in varying portions of SBRR. A crop management schedule was established such that manure is applied to corn acres once every four years. Manure application is divided between fall (67%) and spring (33%) according to the FANMAP survey. For the baseline scenario, it is assumed that commercial N and P fertilizer rates are not changed in response to manure application. This results in these fields receiving excess N and P once every 4 years. Manure was applied to achieve a rate of 88.2 lb/ac (98.8 kg/ha) based on P application rates reported in the FANMAP survey. Manure N:P ratios taken from the Minnesota Department of Agriculture information sheet titled: *Useful Nutrient Management Data* were preserved within the model; as a result, manure N application rates were dependent on the amount of manure required to achieve the estimated manure P rate and was different for swine and dairy manure.

Management schedules including manure application for swine and dairy manure are presented in Figures 59 and 60, respectively.

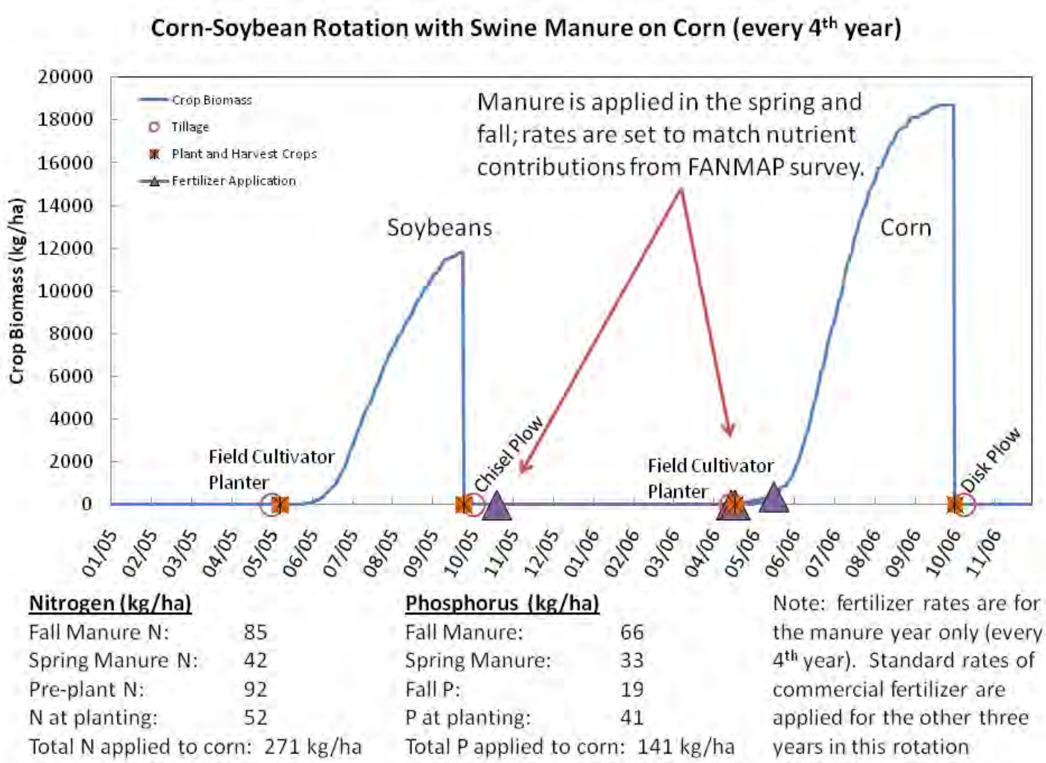


Figure 59. Typical management schedule for a field receiving swine manure in SBRR.

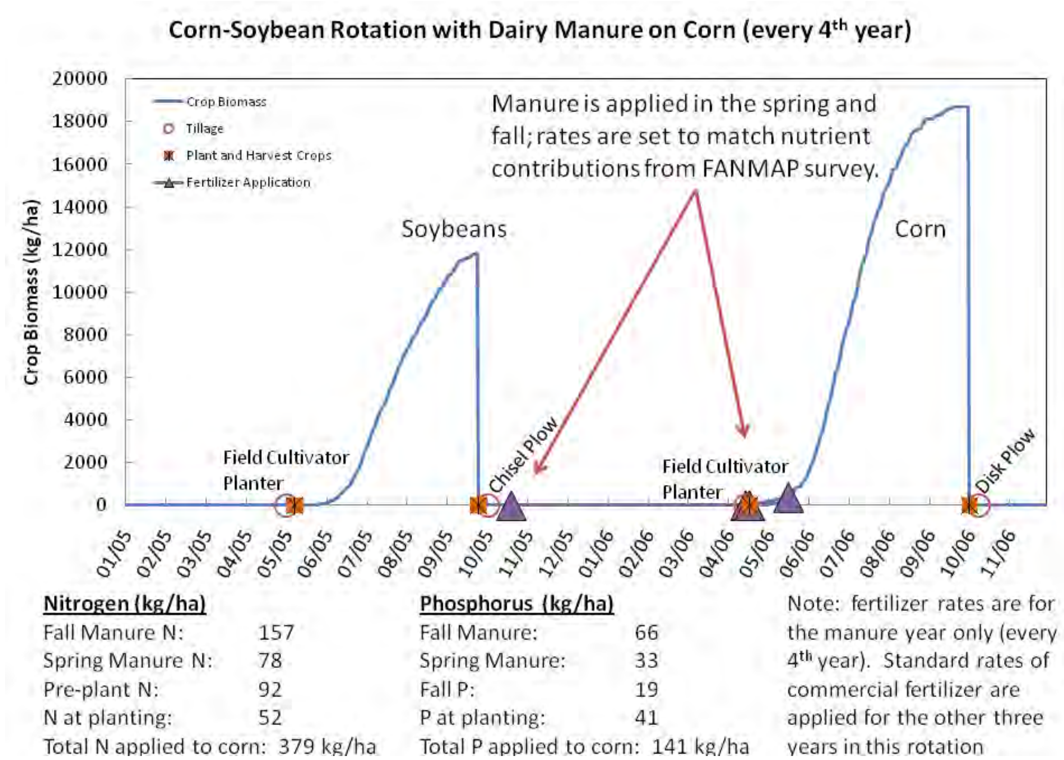


Figure 60. Typical management schedule for a field receiving dairy manure in SBRR.

An overview of how different management schedules were applied to SBRR is presented in Table 22.

Table 22. Summary table showing how different management schedules are applied to the SWAT model of SBRR. Sub basin numbers correspond with Fig. 55.

		Commercial N application		Manure	
Sub Basin	Karst	Spring	Fall	Swine	Dairy
1	Y	X			X
2	N	X			
3	Y	X			
4	N	X		X	
5	N		X		
6	N		X		
7	Y	X			X
8	N		X		
9	Y	X			
10	Y	X			
11	Y	X			
12	Y	X			
13	N	X			
14	Y	X			
15	N	X		X	

Observed flow and water quality data.

Five years of flow and water quality monitoring data (from 2004-2008, inclusive) are available for SBRR. Ideally, the model calibration and validation periods encompass a range of climatic conditions in order to provide for a more robust model calibration. Compared against the period from 1980 to 2008, the five-year period of observed data for this study includes annual rainfall amounts that are average or wetter than average (Fig. 61). The SWAT model was calibrated based on data from 2004-2005 and validated for the period from 2005-2008.

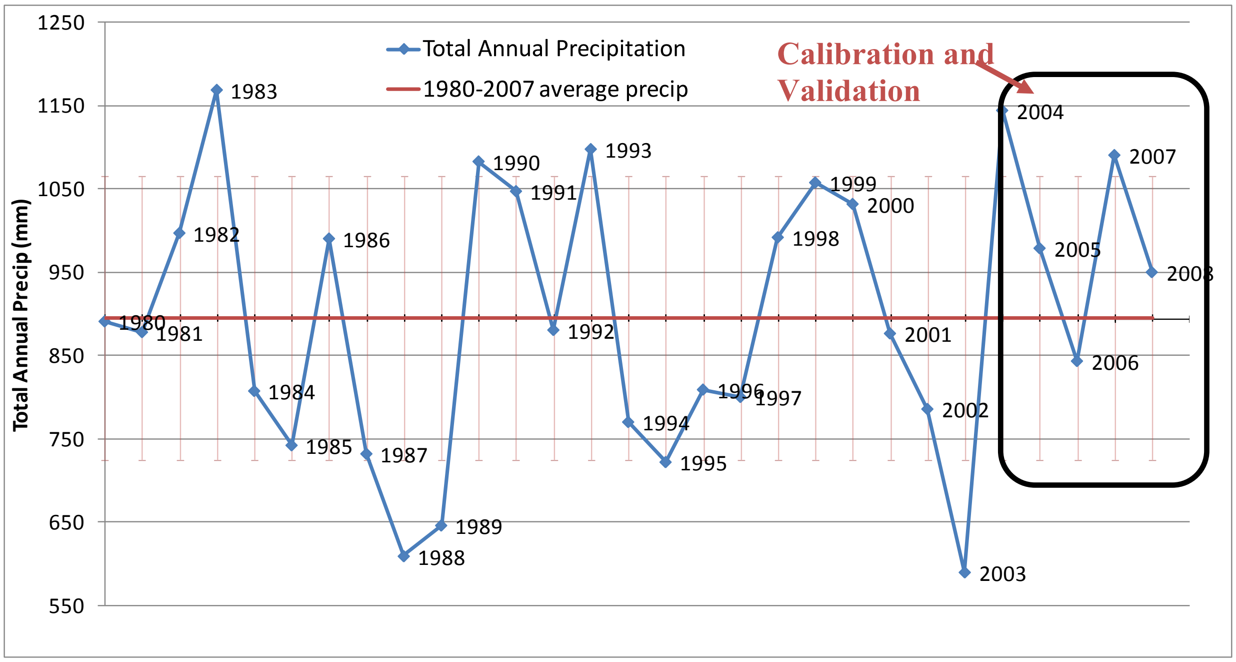


Figure 61. Total annual precipitation for SBRR from 1980-2008. The model calibration and validation period from 2004-2008 includes average and wet years, but no dry years.

Daily flow data were coupled with periodic sampling of water quality parameters in order to estimate monthly loads exported from SBRR. Monthly loads were estimated using FLUX, a model developed and maintained by the U.S. Army Corps of Engineers (Walker, 1996). Estimated monthly loads are provided in an appendix. Model calibration is a long process that involves collection of observed data, research of primary and secondary literature, sensitivity analysis, as well as trial and error. The key aspects of model calibration are summarized and annotated in Table 22, below. One aspect of model calibration that is important to highlight is the influence of karst geology on the response of shallow groundwater and hydraulic conductivity of bed material in some SBRR sub basins. In order to simulate this in SWAT, sub basins were classified as either karst or non-karst based on the predominance of karst features observed in the area (Fig 62). There are likely many more intermediate steps to the distinction between areas with the presence/absence of karst features and some sub basins in this approach do contain some karst features. For this study, the decision was made for two classes (i.e., karst vs. non-karst) in order to avoid adding more complexity to the calibration process in the absence of additional supporting data. The karst-influenced sub basins were calibrated based on the assumption of stronger contributions from shallow groundwater and shorter delay in groundwater response time (when compared to non-karst sub basins). Specific calibration parameters for the SWAT model are provided in Table 22; parameters that are not listed here were not changed during calibration.

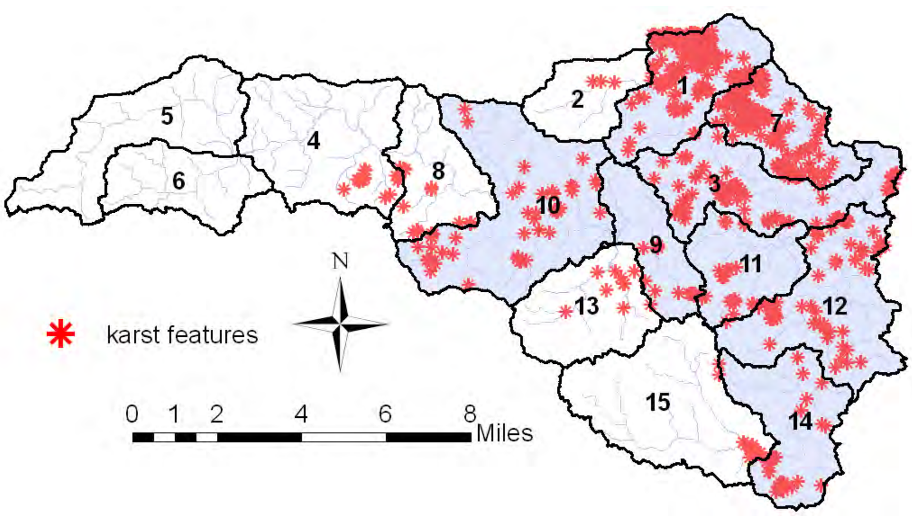


Figure 62. SBRR sub basin map showing observed karst features. Shaded sub basins are considered to be karst influenced for purposes of model calibration.

Table 23. Summary of SWAT model calibration parameters for SBRR.

Parameter	Description	Default Value	Calibrated Value	Notes
SWAT location = **.bsn** file				
TIMP	snow temperature lag factor	1	0	calibrated for timing of snowmelt
PET method	method for estimating potential evapotranspiration	Penman Monteith	Hargreaves	Wang et al., 2006
ESCO	soil evaporaton compenstation factor	0.95	0.7	calibrated based on water budget
EPCO	plant uptake compensation factor	1	0.95	calibrated based on water budget
CN_FROZ	allows applicatoin of curve number approach to frozen	inactive	active	appropriate for local conditions
Crack flow	simulates crack development in soils	inactive	active	too much surface runoff when this parameter is inactive
SURLAG	surface runoff lag coefficient	4	3	calibrated value
PRF	peak rate adjustment factor for sediment routing	1	0.8	calibrated value
SPCON	sediment entrainment factor - linear	0.0001	0.001	estimated from observed sediment concentrations
EPEXP	sediment entrainment factor - exponent	1	1.5	
CMN	rate factor for humus mineralization	0.0003	0.002	default is too low (outside the range of values recommended in the SWAT documentation); calibrated to N budget
CDN	denitrification exponential rate coefficient	0	0.05	Herridge et al., 2008; David et al., 2009; calibrated to N budget
SDNCO	denitrification threshold water coefficient	0	0.95	Herridge et al., 2008; David et al., 2009; calibrated to N budget
SWAT location = **.hru** file				
OV_N	Manning's roughness coefficient for overland flow	0.14	0.4	value for farm fields based on fall disk from SWAT literature (compiled from Engman, 1986).
		0.14	0.25	value for all other HRUs; Engman, 1986
DEP_IMP	depth to impervious layer in soil profile (mm)	inactive	3750	used as a calibration parameter for soils that are well to moderately-well drained (A, B).
		inactive	1500	for poorly drained soils (A/D, B/D, C, D) where water table is defined (by NRCS) as being less than 0.6 m from the surface. Set to 1.5m here in order to be below the depth of subsurface tile drainage.
CANMX	maximum canopy storage (mm)	0	4	Kozak et al., 2007.

Table 22 continued on next page…

Table 22 continued…

Parameter	Description	Default Value	Calibrated Value	Notes
SWAT location = **.gw** file				
GW_DELAY	groundwater delay time (days)	31	1	for karst sub basins only, based on spring data provided by Adam Birr (timed response to rainfall events)
Alpha_BF	baseflow recession constant, groundwater response to changes in recharge	0.048	0.1	for non-karst sub basins, initial watershed value was compiled from baseflow separation program as a starting point. Smaller for non-karst basins
		0.048	0.8	for karst sub basins, based on starting value, literature, and calibration
GWQMIN	threshold depth of water in shallow aquifer required for return flow to occur	0	150	calibration parameter
SWAT location = **.mgt** file				
FRSD	initial age of trees	0	50	arbitrarily set to represent a somewhat established forest (so trees aren't growing from scratch)
Cn2	SCS curve number	varies	Decr. by 20%	Cn2 was decreased as a calibration parameter to increase infiltration (small peaks from rainfall events were too sharp, indicating too much surface runoff.
SWAT location = **.rte** file				
Ch_K2	hydraulic conductivity of channel bed material	0	37	For non-Karst basins; ave of values measured by Toby Dogwiler
		0	66	for Karst basins; ave of values measured by Toby Dogwiler
CH_W	channel width at bankful conditions	varies	see table 2	measured by Toby Dogwiler
CH_D	channel depth at bankful conditions	varies	see table 2	measured by Toby Dogwiler
W/D	width/depth ratio	varies	see table 2	measured by Toby Dogwiler
CH_N2	Manning's roughness coefficient for channel flow	0.014	see table 2	measured by Toby Dogwiler

Measuring model performance

Performance of the SWAT model was assessed by comparing the models ability to match monthly values of observed flow (mean monthly discharge) and water quality parameters (total monthly loads).

In addition to comparing mean values for the calibration and validation periods, model performance is evaluated with the Nash-Sutcliffe Efficiency metric (NSE; Nash and Sutcliffe, 1970):

$$E = 1 - \frac{\Sigma(Y_o - Y_m)^2}{\Sigma(Y_o - \overline{Y_o})^2}$$

Where Y_o is the observed monthly value (discharge or load), Y_m is the modeled value of the same parameter, and $\overline{Y_o}$ is the mean value of the observed data. NSE values can range from $-\infty$ to 1. Perfect agreement between predicted and observed data results in NSE = 1; an NSE value of 0 indicates that the mean of the observed data is as accurate as the model predictions. For watershed scale modeling, NSE values of 0.36 to 0.75 are generally considered fair, while values greater than 0.75 indicate good model performance (Motovilov et al., 1999).

Flow – Daily Values

Average daily flow values were 3.47 and 3.79 m^3 sec^{-1} for the calibration and validation periods, respectively; the modeled averages for the same periods were 3.06 and 3.24 m^3 sec^{-1}. Model performance is fair with NSE values of 0.49 and 0.66 for the calibration and validation periods. Prediction of timing and magnitude of flow events is generally in close agreement with observed values (Fig. 63). There are some notable exceptions, however, during large flow events during which the model generally under-predicts discharge. This may be partly due to the nature of intense rainfall events (thunderstorms) that typically generate large flow events during summer months. Variability in rainfall patterns may not be adequately captured by the two gauges used by the model. Additionally, interactions with groundwater in SBRR are variable (as indicated by channel hydraulic conductivity measurements that vary over different hydrograph conditions) and it is likely that some model parameters controlling groundwater response may be over simplified for this watershed.

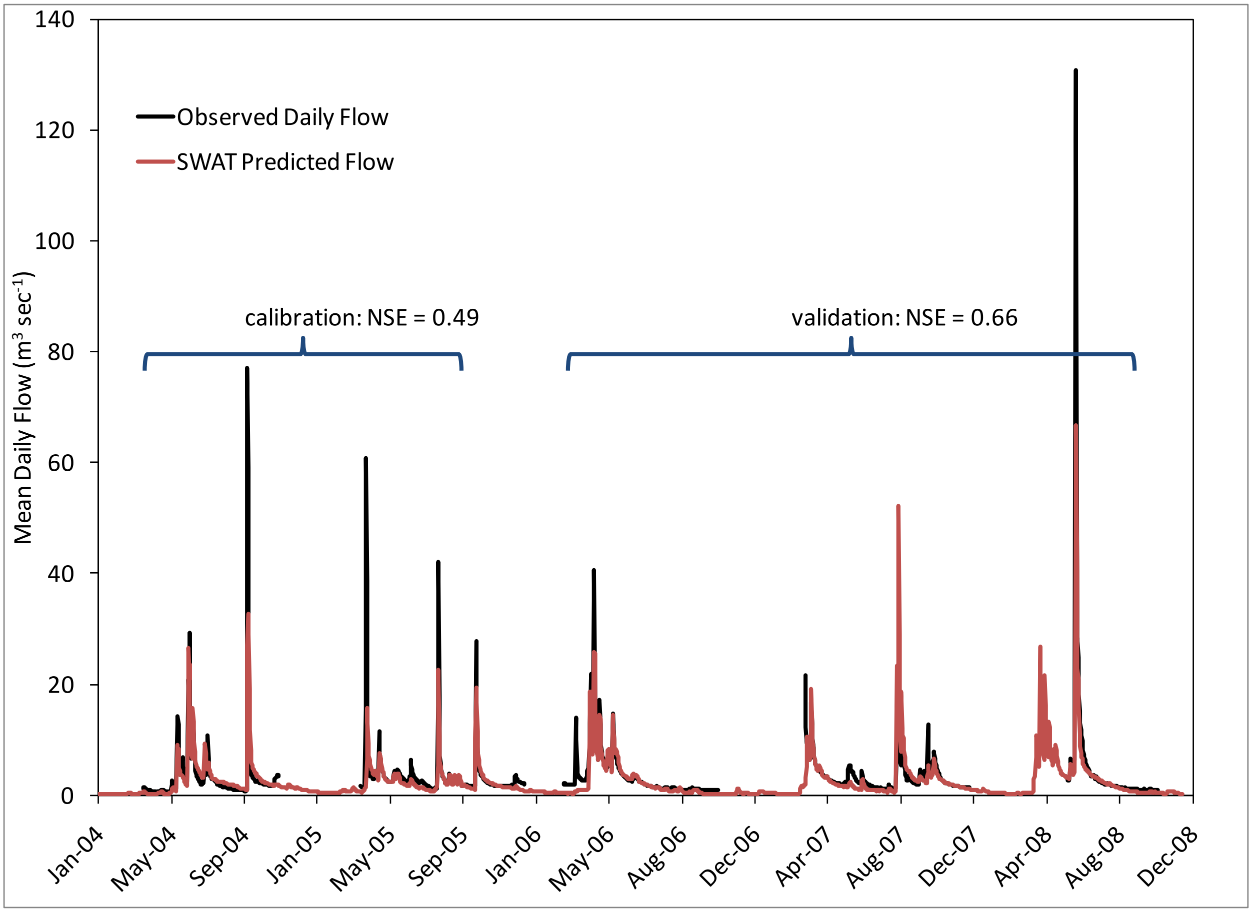


Figure 63. Observed and predicted mean daily flow for SBRR from 2004 to 2008.

Flow – Monthly Values

Average observed monthly flows were 3.34 and 3.91 m^3 sec^{-1} for the calibration and validation periods while predicted values were 3.18 and 3.39 m^3 sec^{-1}, respectively (Fig. 64). Similar to the daily flow, the model is slightly under predicting mean monthly flow. Agreement of observed and predicted flow for SBRR is good, with NSE values of 0.81 for both the calibration and validation periods.

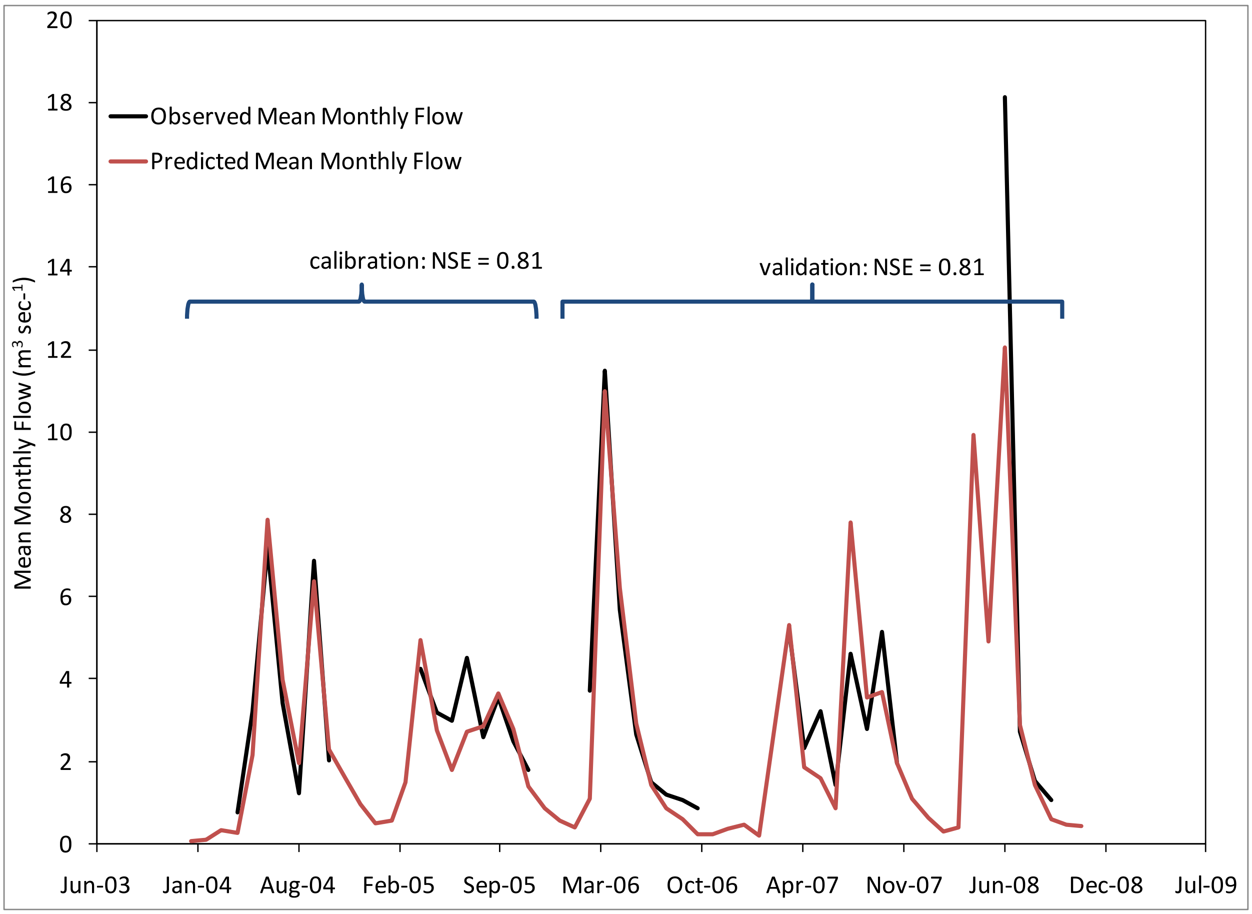


Figure 64. Observed and predicted mean monthly flow for SBRR from 2004 to 2008.

Total Suspended Solids

The mean monthly sediment loads from SBRR were 998 and 1477 tons for the calibration and validation periods, respectively. The SWAT model slightly under predicted sediment loads during calibration (777 tons) and slightly over predicted sediment loads during the validation period (1,557 tons). The model generally did a good job of predicting the overall timing and magnitude of sediment export (Fig. 65). However, the model calibration period includes two months in 2004 (June and September) during which the model over- and under-predicted sediment loads, respectively. This lead to a poor calibration value (NSE = 0.26). In contrast, the validation values were quite good with NSE = 0.85. The role of the year 2004 in calibration values will be discussed at the end of this section.

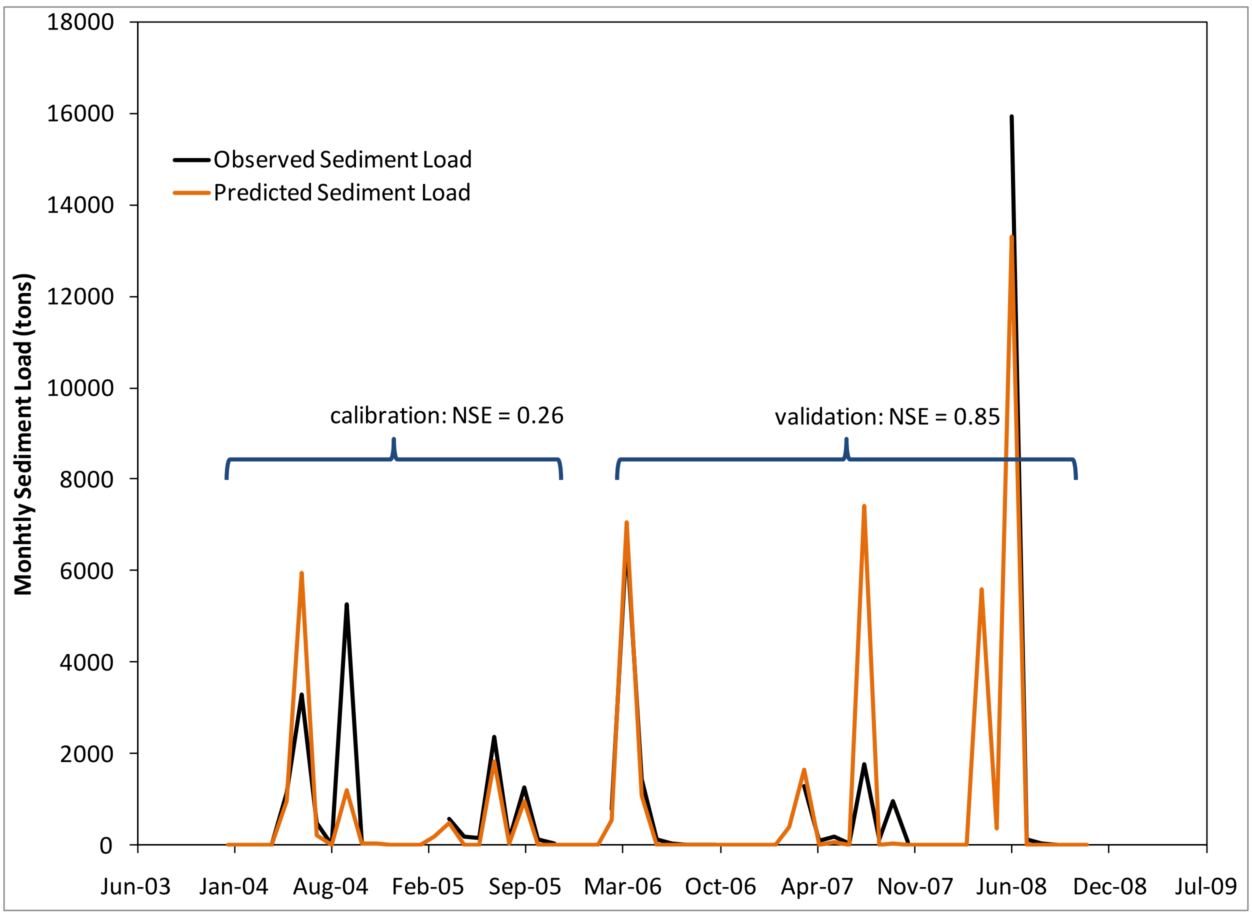


Figure 65. Observed and predicted monthly sediment loads for SBRR from 2004 to 2008.

Total Phosphorus

Mean monthly phosphorus loads values for the calibration and validation periods were 1,820 and 2,544 kg, respectively. Model predicted monthly averages for the same periods were 1,462 and 2,372 kg. Model performance was fair for the calibration period (NSE = 0.48) and good for the validation period (NSE = 0.78). Model estimated phosphorus loads (Fig. 66) tracked the over/under prediction trends of the sediment loads (above), indicating that soil erosion is the primary mechanism by which phosphorus is exported from SBRR.

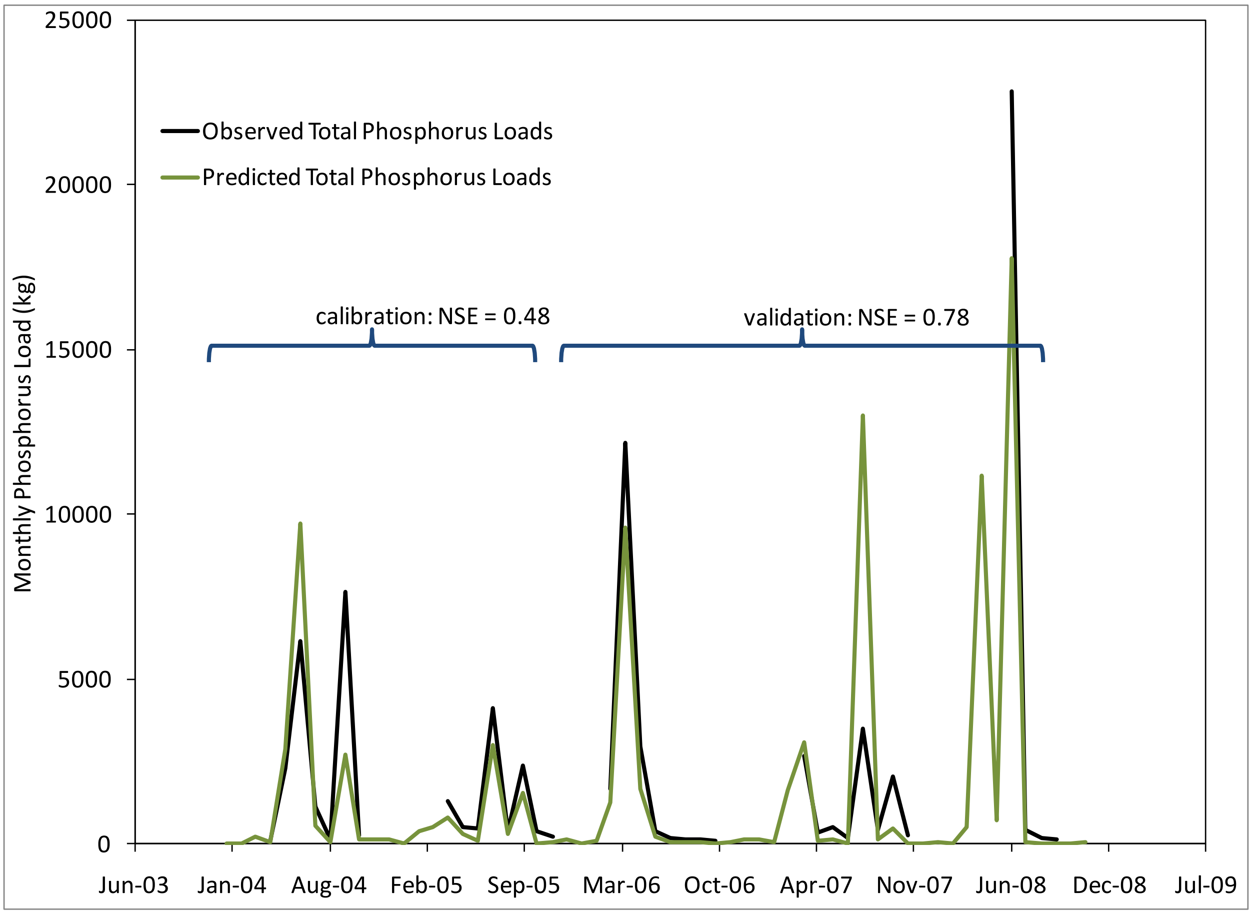


Figure 66. Observed and predicted monthly phosphorus loads for SBRR from 2004 to 2008.

Nitrate/Nitrite

The SWAT model had the greatest difficulty with correctly estimating nitrogen export (discussed here as the sum of nitrate, NO_3^-, and nitrite, NO_2^-). This is particularly the case with the calibration period, during which the predicted monthly mean (102,188 kg) was 48% greater than the observed monthly mean (68,957 kg). This is primarily the result of poor model performance during 2004 (Fig. 67). Performance was better during the validation period, however, with the predicted mean monthly load (80,911 kg) being 5.6% less than the observed mean monthly load of 85,712 kg. The model failed the calibration period with a NSE value of -3.63. Surprisingly, the NSE value for the validation period was fair (NSE = 0.66). While the model generally predicted the timing and magnitude of monthly NO_3^-/NO_2^- loads, it often missed the more detailed shifts in month-to-month values.

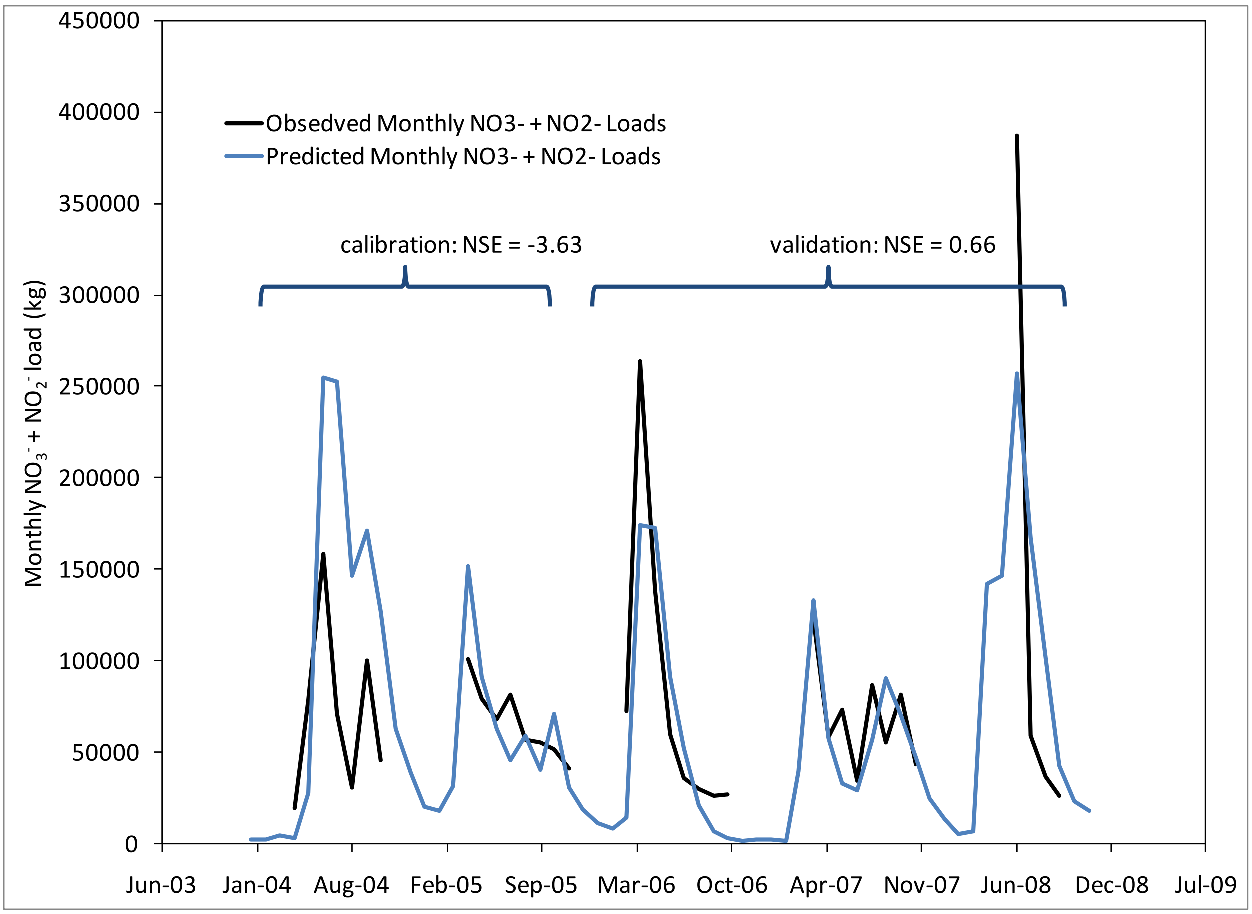


Figure 67. Observed and predicted monthly nitrogen loads flow for SBRR from 2004 to 2008.

Discussion of model calibration and validation.

The calibration of the SWAT model for SBRR was complicated by difficulties in successfully predicting sediment, phosphorus, and nitrogen loads in 2004. This is likely due to accumulation of several factors that resulted in 2004 being a challenging year to model successfully. When compared against the past 28 years of rainfall data from local gauges, 2004 was the 2^{nd} wettest year (annual precipitation = 1144mm) and was following the driest year (2003 had 590mm of annual precipitation). Storage of water in the shallow aquifer and also within the soil profile can influence springtime flow resulting from snowmelt and soil thawing. The poor model performance may indicate that SWAT doesn't perform adequately at the far ranges of dry/wet conditions for this watershed.

Results from 2004 are likely further complicated by intense rainfall events that occurred the months of June and September, the two months during which the model over- and under-predicted, respectively. During both months, rainfall events occurred during which there was a large difference between individual rain gauges: 2.9" on June 9 and 3.2" on September 15, 2004. This large difference between gauges indicates that daily precipitation was heterogeneous and it's likely that the actual rainfall in SBRR may not be accurately reflected by the 2 gauges used for this study. If this is the case, then the problem is not necessarily with model performance, but rather, with quality of input data. This is challenge for watershed-scale modeling is not uncommon, potential solutions are developing a more dense rain gauge network or switching to more spatially-explicit rainfall datasets such as those collected by radar (i.e., NEXRAD).

Finally, these results (especially the nitrogen results) are likely to be complicated by the fact that a large proportion of SBRR is influenced by karst geology and complex interactions between surface and groundwater. The SWAT model assumes that groundwater follows flow paths that reflect an extension of surface topography and watershed boundaries. In karst landscapes, this assumption is not likely to hold true, as has been shown by dye-tracing studies in SBRR and delineated spring-shed boundaries that do not necessarily correlate with watershed boundaries expressed topographically. This disconnect between surface and groundwater in SBRR is likely the reason behind the fact that sediment and phosphorus calibration/validation values are stronger than those for nitrogen.

Extending the record of monitoring data to additional years will help to develop more representative dataset which should help with a more robust calibration and validation in future studies. For the present study, calibration and validation values are fair to good for flow, sediment, and phosphorus. While the validation performance was acceptable for NO_3^-/NO_2^-, the model failed the calibration period. As such, it is recommended that nitrogen results from alternative management scenarios be interpreted with caution; it would be more appropriate to view nitrogen results in light of relative differences between scenarios, rather than placing confidence in actual values.

Current watershed conditions and alternative management scenarios

Evaluating nonpoint source pollution in SBRR – current conditions.

The calibrated/validated model was run for the period from 1997 through 2008 and results from 1999-2008 are used to assess current sources of nonpoint source pollution in SBRR (results from 1997 & 1998 are considered part of the model warm-up period and discarded).

Flow – water balance

For the 10-yr evaluation period, 72.7% of average annual precipitation was removed from the watershed via evapotranspiration (ET) while 25.5% of annual precipitation contributed to water yield at the outlet of SBRR (the 1% difference is due to soil storage). This partitioning between ET and water yield is comparable to other hydrologic studies in the region and suggests that the SWAT model is doing an adequate job at simulating plant growth and water use.

Of the water that reaches the outlet of SBRR, the largest proportion is comprised of shallow groundwater (58.8%) with the balance coming in the form of: tile drainage (16.6%), surface runoff (15.9%), and lateral soil flow (8.8%; Fig 68).

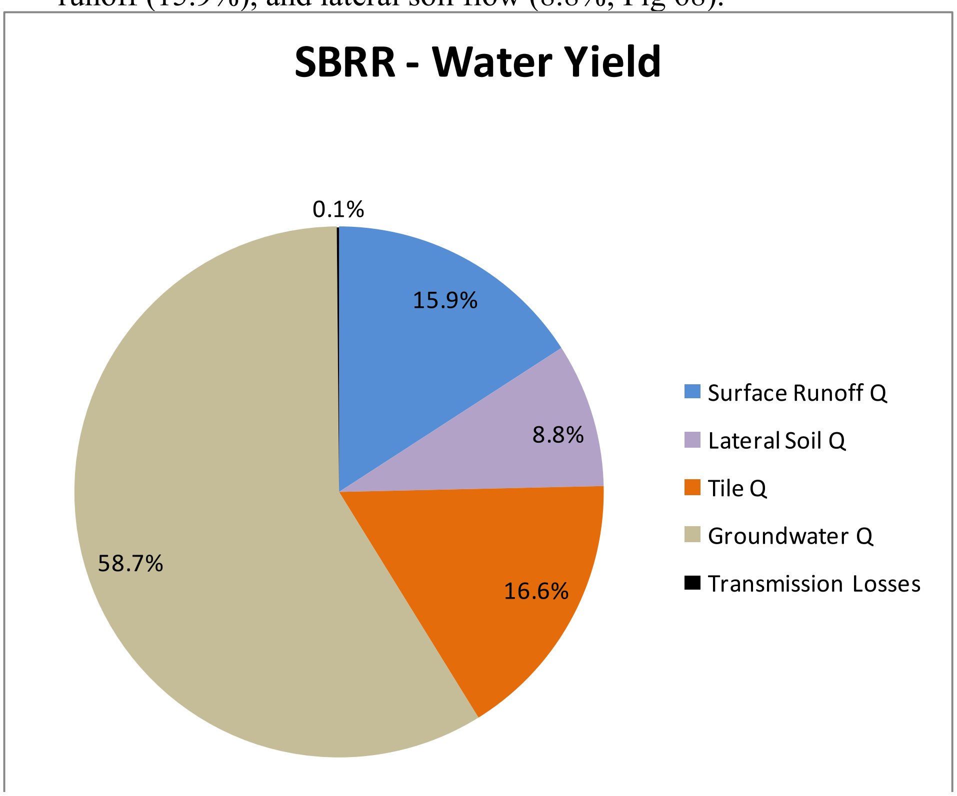


Figure 68. Average annual water yield components from SBRR for the period from 1999 to 2008.

Sediment

The SWAT model predicts sediment loss from field sources only; additional sources of sediment such as gullies and streambanks within SBRR are not predicted by the model and not considered in the results presented here. Determining the relative importance of field vs. non-field sources of sediment in SBRR would require additional field-based study. The predicted 10-yr average annual sediment yield from SBRR was 1.11 t ha^{-1}. When considering only lands used for row crop production, average sediment yield was 1.60 t ha^{-1}. The range of predicted sediment loss varied widely, however and was greatly dependent upon land management practices as well as slope characteristics. Over 65% of HRUs in SBRR averaged less than 1 t ha^{-1} yr^{-1} of sediment erosion. Corollary to this observation is that a small number of HRUs are characterized by high sediment yields which are responsible for a large proportion of the annual load. For the 10-year period studied here, approximately 75% of the average annual sediment erosion occurred in less than 25% of the watershed (Fig 69).

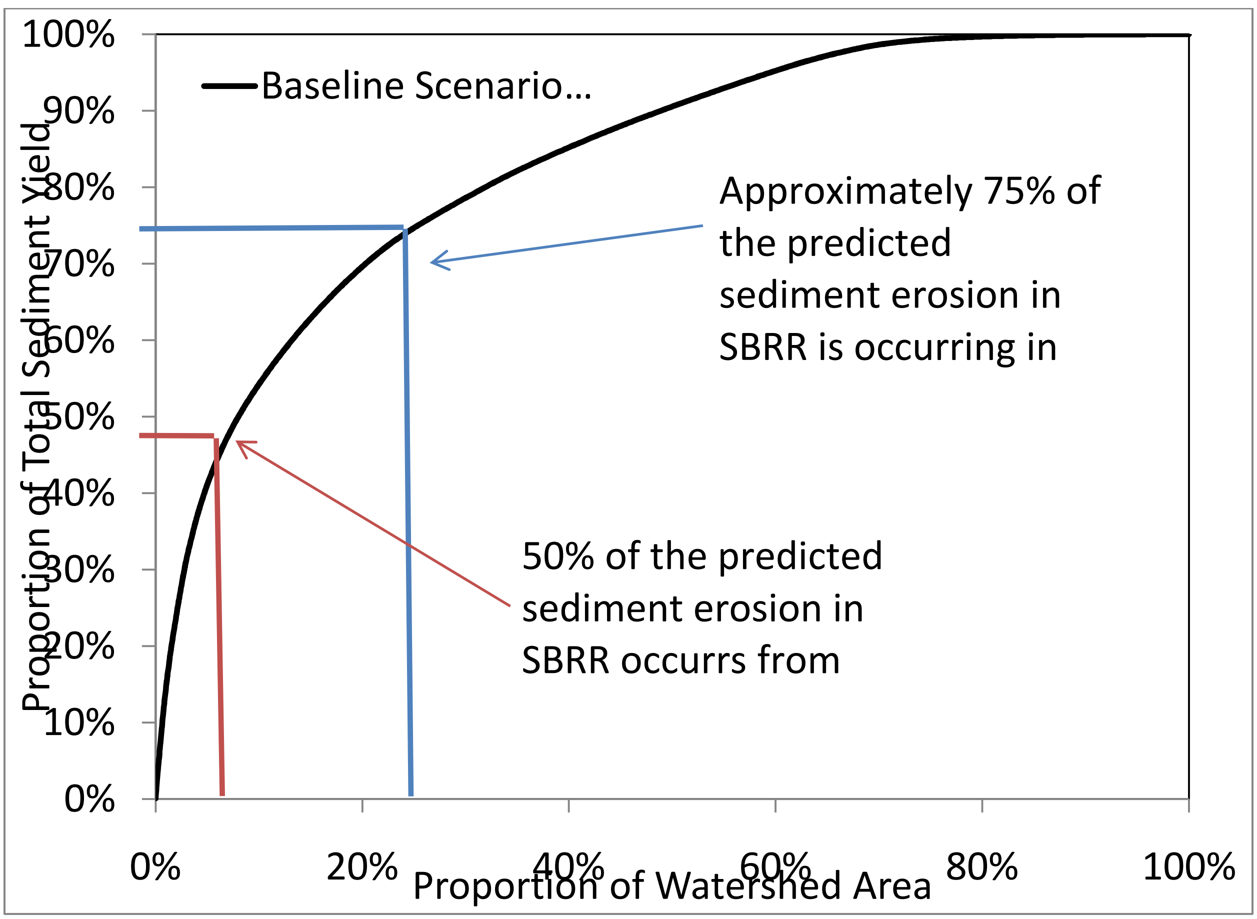


Figure 69. Cumulative upland sediment yield plotted as a function of cumulative watershed area for SBRR (based on annual averages from 1999 to 2008).

The HRUs that are contributing the greatest edge-of-field sediment losses are those with row crop land use occurring on slopes steeper than 4%. These areas are not common in SBRR; they comprise only 8% of the total watershed area but disproportionately contribute to high sediment erosion (Fig 70). This provides insight into identifying targeted alternative management practices for reducing sediment in SBRR.

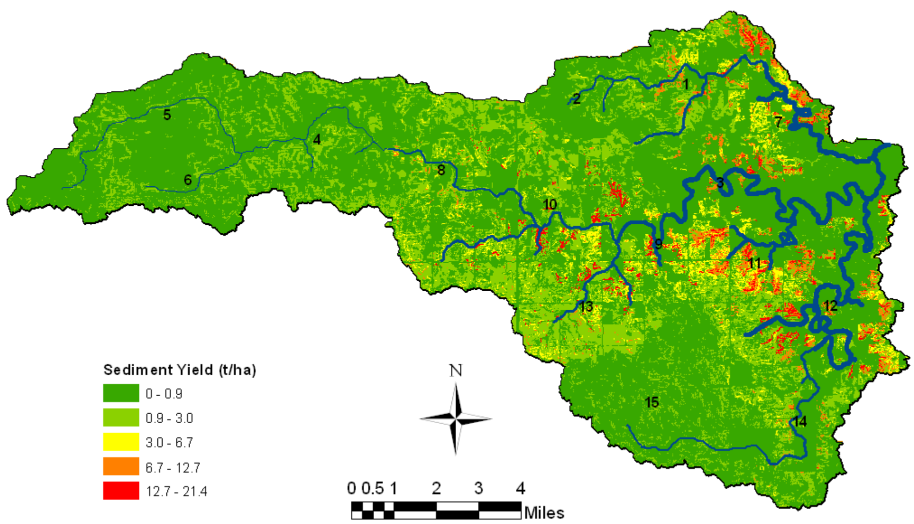


Figure 70. Predicted edge of field sediment loss under the baseline scenario for SBRR. Values are averages of annual sediment yield predicted from 1999-2008.

Phosphorus

Edge-of-field phosphorus losses in SBRR for the period from 1999-2008 averaged 0.84 kg ha^{-1}. Phosphorus losses from lands with rowcrops averaged 1.2 kg ha^{-1}. Phosphorus loss in SBRR followed similar patterns as model predictions for sediment; this is not surprising since phosphorus is typically associated with sediments and the primary mode of phosphorus loss in most landscapes occurs via overland runoff (Sharpley and Syers, 1979). In SBRR, approximately 63% of total phosphorus is contributed by 25% of the landscape (Figs 71 and 72). It is important to note that this study does not account for variability in soil test P levels which can be particularly high in areas that support livestock production. The greatest risk for P loss from fields will occur when potential for transport (i.e., high erosion) is co-located with manure P and/or high soil test P.

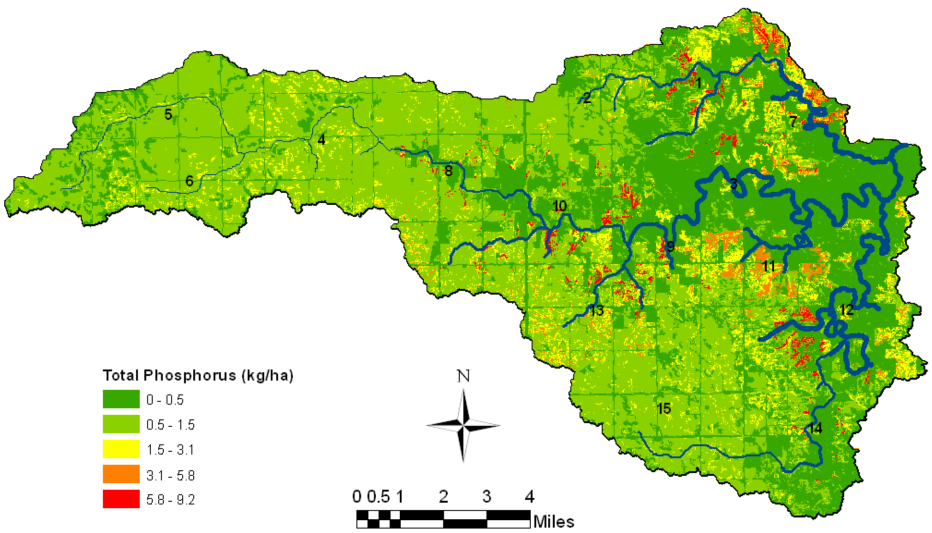


Figure 71. Predicted edge of field phosphorus loss under the baseline scenario for SBRR. Values are averages of annual phosphorus yield predicted from 1999-2008.

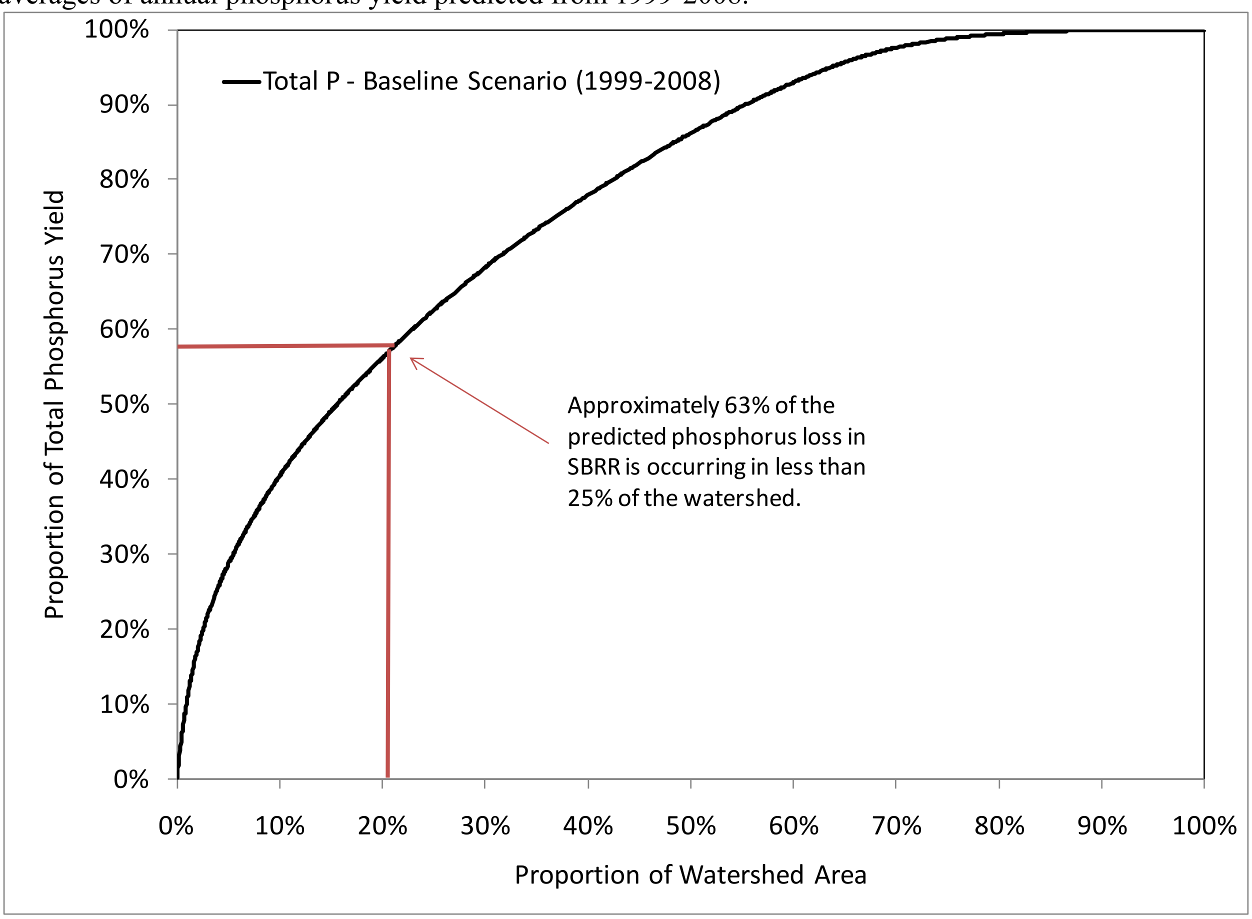


Figure 72. Cumulative upland phosphorus yield plotted as a function of cumulative watershed area for SBRR (based on annual averages from 1999 to 2008).

Nitrogen

In contrast to sediment and phosphorus sources in SBRR, nitrogen loss occurred more uniformly from all croplands. Average predicted NO_3^- export from SBRR was 21 kg ha^{-1}; average export from row cropped lands was 29.7 kg ha^{-1}. The pathway for NO_3^- delivery did vary spatially across the watershed, however. Predicted nitrogen export via subsurface tile drainage systems was prevalent in portions of the watershed that contained poorly drained soils (subsurface tile drainage was assumed to be present in HRUs where cropland occurred on poorly-drained soils and corresponded to FANMAP survey estimates of the extent of drainage in SBRR). This occurred primarily in the western and southern portions of the watershed (Fig 73).

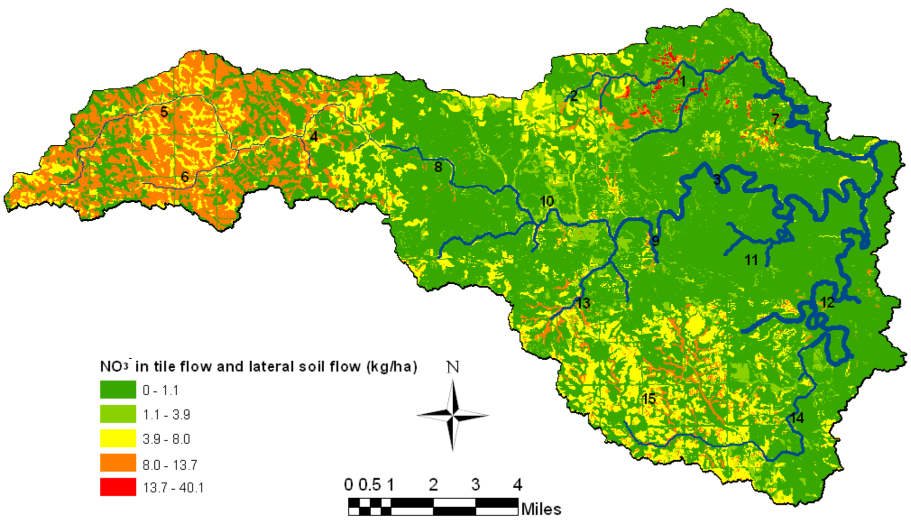


Figure 73. Predicted nitrogen loss from tile flow and lateral soil flow under the baseline scenario for SBRR. Values are averages of annual NO_3^- yield predicted from 1999-2008.

In sub basins where karst geology is present, a large proportion of NO_3^- export occurred via the shallow aquifer (Fig 74). Particularly high NO_3^- loads are predicted from sub basins 1 and 7 where karst geology and manure application coincide. Overall, groundwater flow from the shallow aquifer contributes the greatest proportion of nitrogen export in SBRR; this is primarily driven by the geology of the watershed (as it is simulated in this model). Where manure application is present, however, there is a very strong secondary effect of this management practice on a landscape that is already predisposed to deliver NO_3^- from fields to streams and rivers.

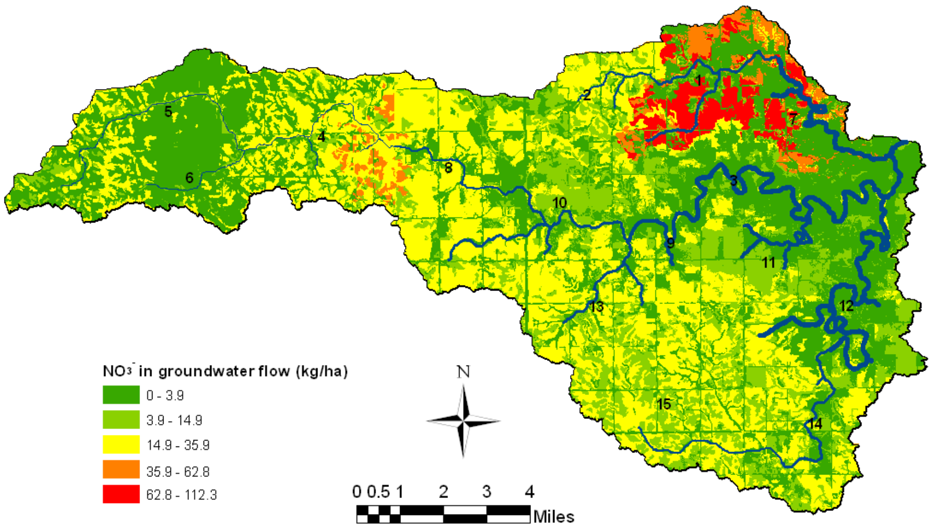


Figure 74. Predicted nitrogen loss from shallow groundwater flow under the baseline scenario for SBRR. Values are averages of annual NO_3^- yield predicted from 1999-2008.

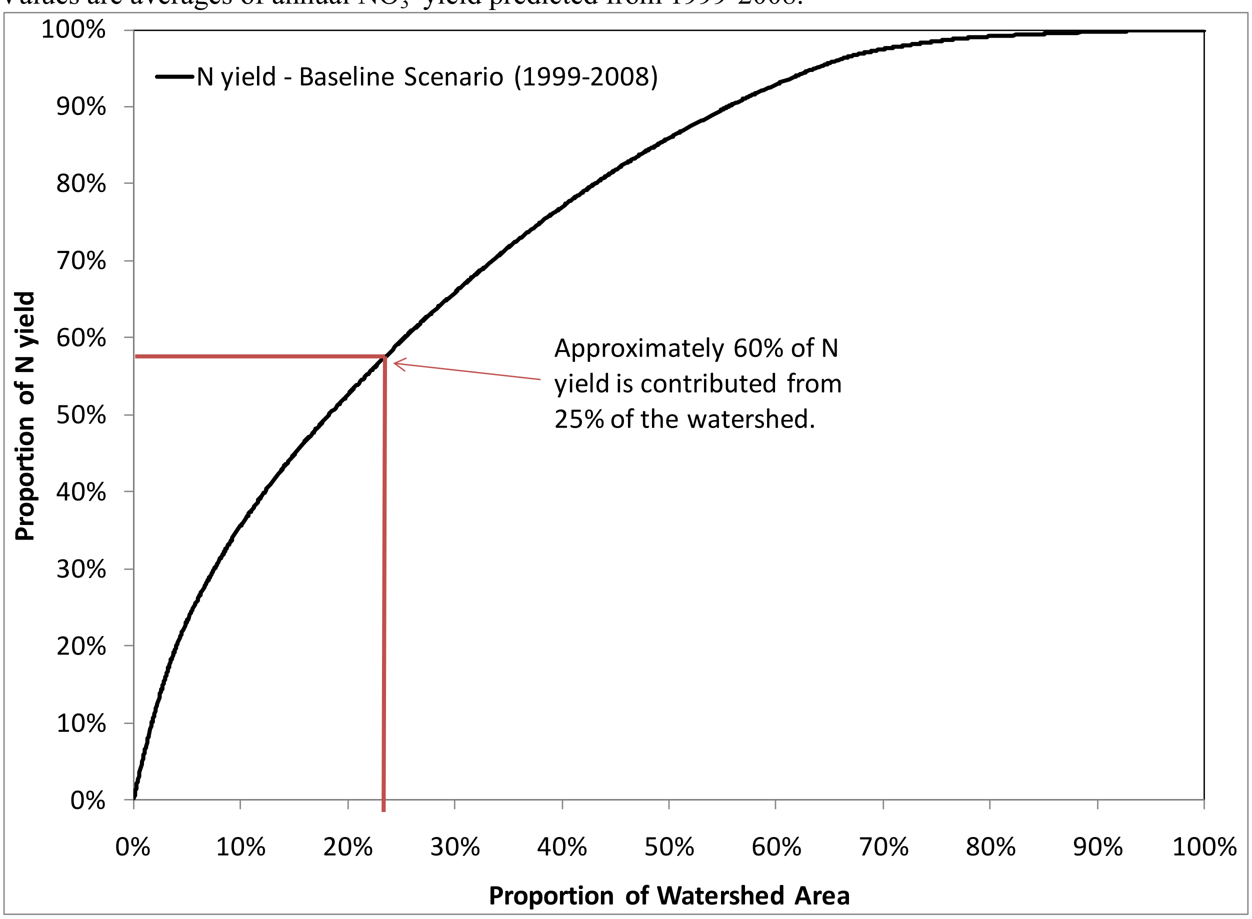


Figure 75. Cumulative nitrogen yield (from shallow groundwater and tile flow) plotted as a function of cumulative watershed area for SBRR (based on annual averages from 1999 to 2008).

Alternative Management Scenarios

Alternative management scenarios in SBRR focused primarily on reducing sediment and phosphorus export. Scenarios that evaluate reduced fertilizer application rates should be interpreted with the caveat that the model calibration failed to adequately predict monthly NO_3^-/NO_2^- loads (despite fair performance during the validation period). Alternative management practices are summarized below.

Conservation Tillage: Chisel and disk tillage practices are replaced with a generic conservation tillage practice. Field cultivators are still used for planting. Conservation tillage practices are not as deep or well-mixed as conventional practices. This allows more crop residue to remain on the soil surface, reducing soil erosion.

Cover Crop: Rye is planted immediately following fall harvest of corn or soybeans and allowed to grow in the fall and spring (as allowed by temperature). Immediately prior to spring field preparation (for corn or soybeans), the rye crop is harvested/killed and field preparations resume with primary tillage, field cultivation, and planting.

Edge-of-Field Filter Strips: A 10m wide filter strip is applied to select fields.

Managing Commercial Fertilizer: Fall application of anhydrous ammonia is replaced with spring application of urea (this only occurs in three sub basins in the western portion of the watershed). Commercial fertilizer application rates for a corn/soybean rotation are reduced to 56 kg/ha (50 lbs/ac) for phosphorus and 135kg/ha (120 lbs/ac) for nitrogen. Commercial fertilizer is not applied to fields receiving manure.

Targeted Manure Application: In sub basins with dairy manure (sub basins 1 & 7), manure is not applied to land steeper than 4%. Rather, flatter portions of the sub basin receive manure at an increased rate to facilitate handling the same amount of manure with a smaller area. Sub basins with swine manure did not have cropland on slopes greater than 4%. *(note: applying excess manure to other non-manured fields was not considered because of the increased cost and effort associated with transporting manure over longer distances).*

A suite of alternative management scenarios were developed that involve various combinations of the alternative practices described above. Results for alternative scenarios are presented below.

Sediment

Alternative management scenarios that targeted landscape elements that were the greatest sources of sediment were, not surprisingly, the most effective at reducing it. A comparison of average sediment yields (for the entire SBRR watershed) is presented in Figures 76 and 77. Only minimal improvements are seen when conservation tillage is applied uniformly to 25% of the cropland in the watershed; this resulted in a 2% reduction in predicted sediment yields. Applying conservation tillage to the 8% of the watershed with cropland on slopes greater than 4% is more effective, resulting in a predicted 8% reduction in sediment yields when compared against the baseline scenario. Even greater improvements are predicted when winter cover crops or vegetated filter strips are targeted to the steep (>4% slope) croplands with average reductions of 32 and 36%, respectively. Average predictions of edge-of-field sediment yield decreased by 53% when a combined approach was simulated which employed both cover crops and filter strips on croplands steeper than 4% and conservation tillage on all remaining cropland (Figs 76 and 77). The alternative management scenarios evaluated here focus on practices that occur in (or adjacent to) crop fields – scenarios for which SWAT is well suited. There are additional measures that can be employed to reduce sediment loads in SBRR streams that focus on structural practices such as terracing and construction of earthen dams. These structural practices were not evaluated in the present study.

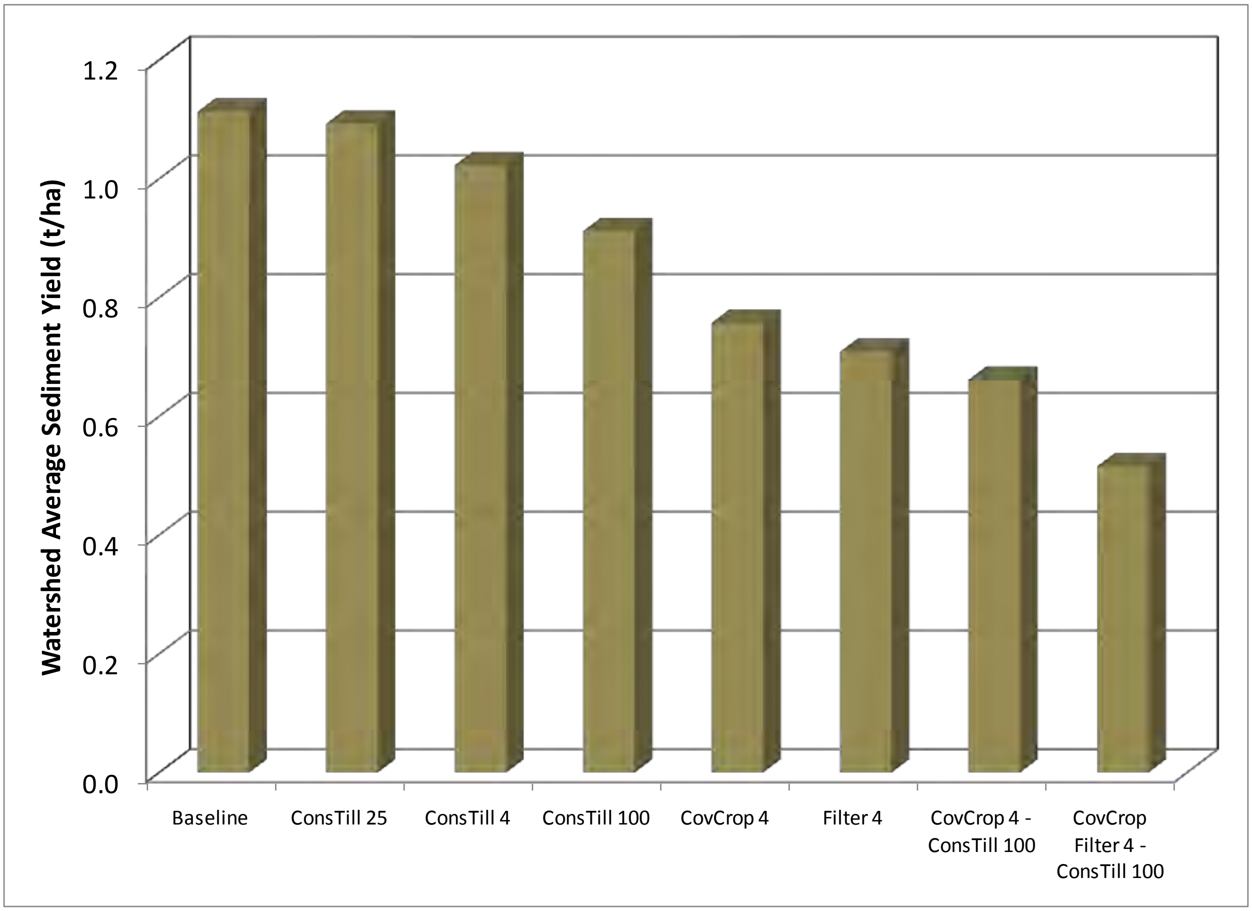


Figure 76. Watershed average sediment yield (edge-of-field losses) for a suite of alternative management scenarios in SBRR. Scenario abbreviations: **Baseline:** Baseline scenario with current conditions. **ConsTill25:** Conservation tillage applied to 25% of cropland in a non-targeted approach. **ConsTill4**: Conservation tillage applied to all cropland with a slope of greater than 4%. **ConsTill100:** Conservation tillage applied to all cropland. **CovCrop4:** Cover crops on cropland with a slope greater than 4%. **Filter4:** 10 meter filter strip on all cropland with a slope greater than 4%. **CovCrop4-ConsTill100:** Cover crops on all cropland with a slope greater than 4% and conservation tillage on all remaining cropland. **CovCropFilter4-ConsTill100:** Cover crops and conservation tillage on all cropland with a slope greater than 4%; conservation tillage on all remaining cropland.

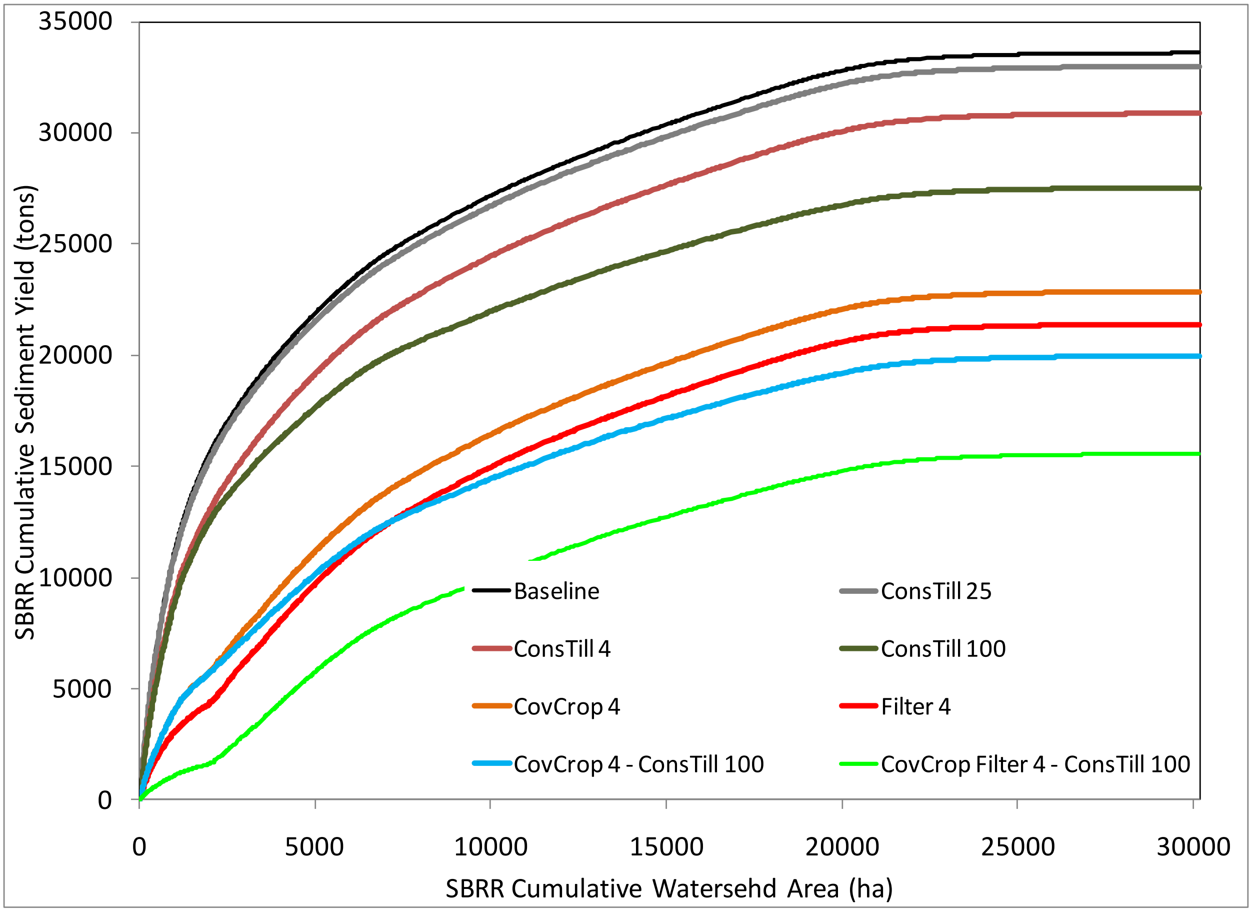


Figure 77. Cumulative sediment yield (average annual yield from 1999-2008) for a suite of alternative management practices in SBRR. A large proportion of the potential reductions in sediment yield occur on the 8% of the watershed comprised of croplands with slopes steeper than 4%. **Baseline:** Baseline scenario with current conditions. Scenario abbreviations: **ConsTill25:** Conservation tillage applied to 25% of cropland in a non-targeted approach. **ConsTill4**: Conservation tillage applied to all cropland with a slope of greater than 4%. **ConsTill100:** Conservation tillage applied to all cropland. **CovCrop4:** Cover crops on cropland with a slope greater than 4%. **Filter4:** 10 meter filter strip on all cropland with a slope greater than 4%. **CovCrop4-ConsTill100:** Cover crops on all cropland with a slope greater than 4% and conservation tillage on all remaining cropland. **CovCropFilter4-ConsTill100:** Cover crops and conservation tillage on all cropland with a slope greater than 4%; conservation tillage on all remaining cropland.

It's important to note that the SWAT model simulates in-channel sediment deposition and re-suspension which is primarily dependent upon the concentrations of suspended sediment and predicted channel velocity. As a result, reductions in edge-of-field sediment yield do not necessarily translate to equivalent reductions in suspended load predicted at the watershed outlet. In SBRR, a 53% reduction predicted in edge-of-field losses produces a 35% reduction in total suspended solid predicted at the watershed outlet.

This has important implications for efforts to reduce sediment loads to achieve water quality goals for trout streams in SBRR such as Forestville Creek and Canfield Creek. Edge-of-field sediment reductions as a result of alternative management practices are predicted to be most effective in Canfield Creek with in-channel reductions approximately equivalent to reductions in predicted edge-of-field losses. Suspended sediment loads in Forestville Creek, on the other hand, are predicted to be less sensitive to alternative management practices with in-channel reductions only reflecting roughly half of the predicted edge-of-field losses (Fig 78). It is worth noting that these predictions are sensitive to estimates of channel dimensions, slope, and roughness; additional collection of physical channel data could change the relationships observed here.

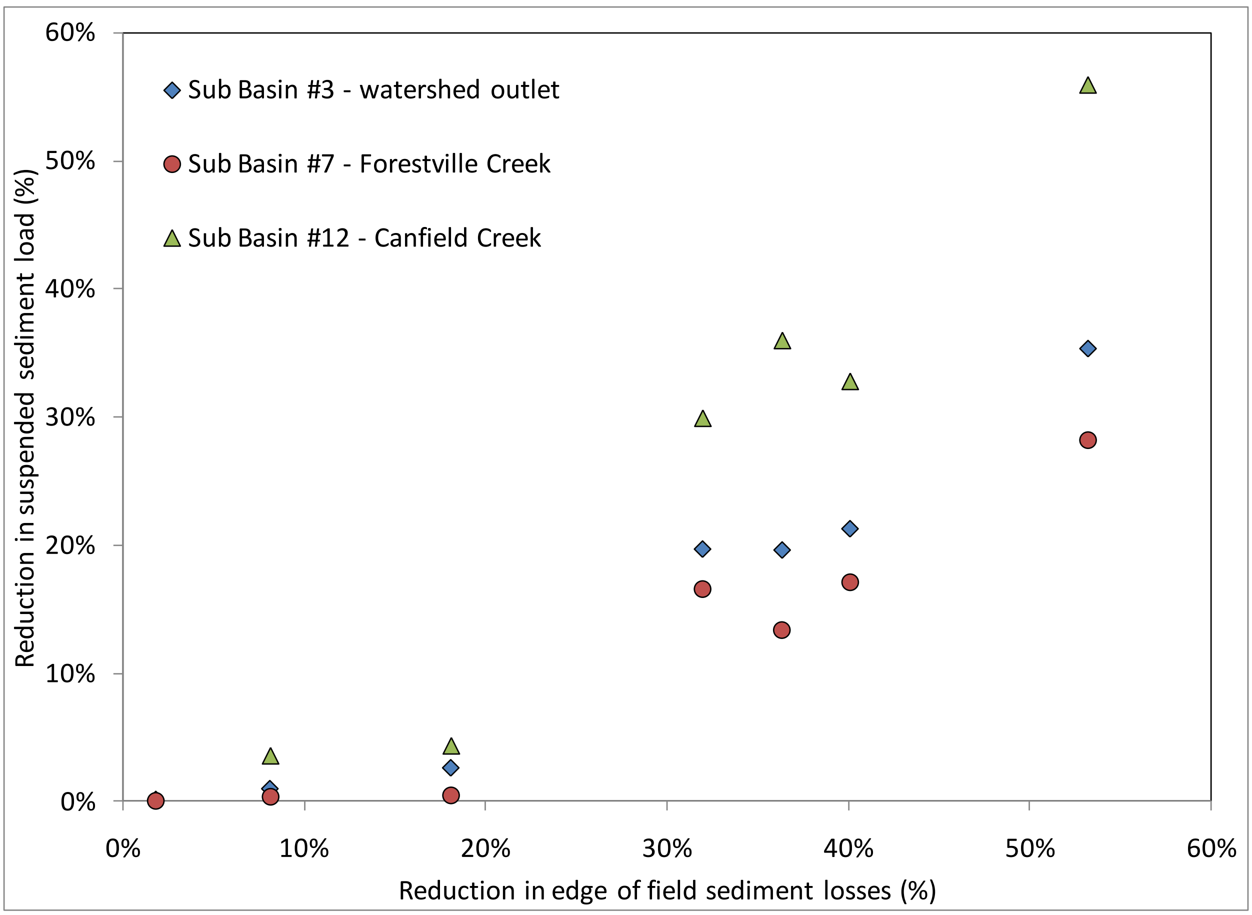


Figure 78. Comparison of predicted reductions in suspended sediment load to reductions in edge-of-field sediment losses for alternative management scenarios. Canfield Creek is predicted to be the most sensitive to upland reductions in sediment loss.

Phosphorus

Similar to the alternative management simulation results for sediment yield, reductions in phosphorus loss from farmlands were greatest in scenarios that employed cover crops and vegetated filter strips on croplands with sloped steeper than 4%. In contrast to the sediment results, however, is the observation that benefits from conservation tillage do little to reduce phosphorus loss and actually increases phosphorus loss in some scenarios (Fig 79). This is the result of crop residue decomposition within the SWAT model framework. Under less efficient (and more shallow) tillage, a greater proportion of crop residue remains on the soil surface where it is allowed to decompose and transition from organic to mineral phosphorus; thus increasing the potential source of phosphorus from farm fields, even though sediment erosion is diminished. SWAT-predicted losses of soluble phosphorus are minor and generally comprised less than 8% of the total predicted phosphorus losses. This fraction of the phosphorus pool is not sensitive to alternative measures to reduce sediment loss, however, resulting in total phosphorus reductions that are more modest than reductions observed with sediment yield. Reducing the rate of commercial P fertilizer applied to corn fields by 8% (from 60.7 to 56 kg ha^{-1}) resulted in a 5% reduction in predicted P losses. This is partially due to the fact that total manure applied in SBRR was not changed in alternative management scenarios. Additionally, watershed averages also incorporate P losses from alternative sources beyond corn fields and will not be influenced by reducing fertilizer application rates.

Measures that target areas that disproportionately contributing P include over crops and filter strips on croplands with slopes steeper than 4%. In sub basins with manure application, manure was not applied to slopes steeper than 4%; this resulted in an approximately 49% increase rates of dairy manure application on fields with slopes less than 4% because the total amount of manure was held constant. In should be noted that simulated manure application rates are already in excess of plant requirements and this redistribution of manure will result in increased nutrient loss from those fields receiving additional manure. The current approach is based on the assumption that manure is not likely to be transported longer distances to additional fields due to the logistics and cost of manure transportation. Sub basins with simulated swine manure application do not have cropland with slopes over 4% so manure application rates are unchanged in alternative scenarios.

The use of cover crops or filter strips (10m wide) on crop land with slopes greater than 4% (in addition to manure redistribution) resulted in reductions of P loss by 17 and 25%, respectively.

An approach that relies on a combination of cover crops and filter strips achieves a 28% reduction in P losses (Fig. 79).

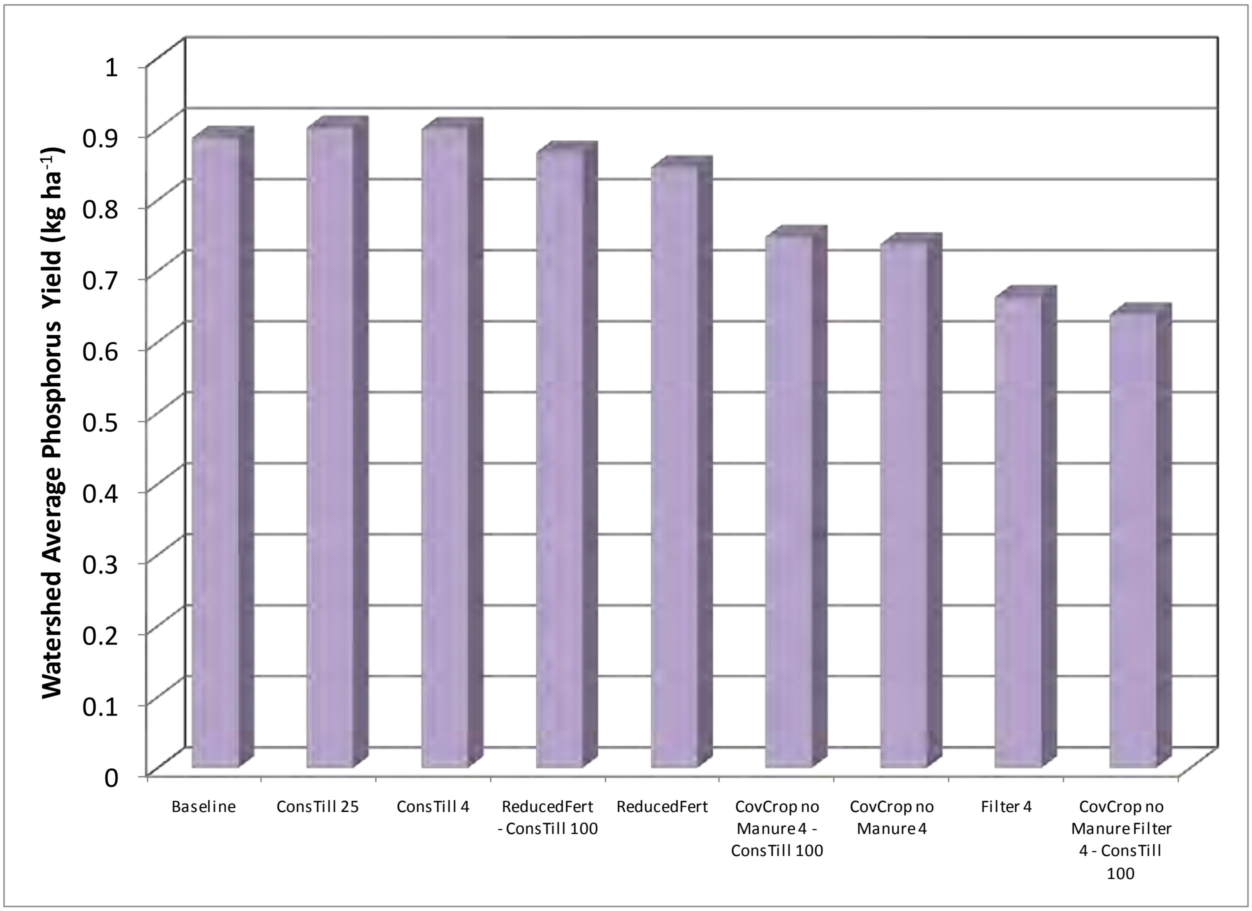


Figure 79. Watershed average phosphorus yield (edge-of-field) losses for a suite of alternative management scenarios in SBRR. Scenario abbreviations: **Baseline:** Baseline scenario with current conditions. **ConsTill25:** Conservation tillage applied to 25% of cropland in a non-targeted approach. **ConsTill4**: Conservation tillage applied to all cropland with a slope of greater than 4%. **ReducedFert-ConsTill100:** Reduced application rate of commercial N and P; conservation tillage applied to all cropland. **Reduced Fert**: Reduced application rate of commercial N and P. **CovCrop no Manure 4-ConsTill100**: Cover crops and no manure on cropland with a slope greater than 4%; conservation tillage on all remaining cropland. **CovCrop no Manure4**: Cover crops and no manure on cropland with a slope greater than 4%. **Filter4:** 10 meter filter strip on all cropland with a slope greater than 4%. **CovCrop no Manure Filter 4-ConsTill 100**: Cover crops, no manure, and 10 meter filter strips on all cropland with a slope greater than 4%; conservation tillage on all remaining cropland.

Nitrogen

Nitrogen losses reported here are the sum of: N in surface runoff (generally negligible), organic N losses, NO_3^- in lateral soil flow (including tile drainage), and NO_3^- in shallow groundwater flow. Compared to the effectiveness of alternative management scenarios on reducing sediment and phosphorus, nitrogen reductions simulated here are more modest. Reductions in commercial application rates were among the more effective practices, with a 6% reduction in fertilizer producing an approximately 7% reduction in watershed export of N (the 1% difference is attributed to differences in denitrification and crop uptake between scenarios). The implementation of filter strips and cover crops were also effective in reducing N export via increased plant uptake of N for a longer proportion of the year. When used individually, cover crops and filter strips resulted in reductions of 5 and 6%, respectively. Combining cover crops and filter strips together (on cropped lands with slopes steeper than 4%) resulted in an 8% reduction in predicted N losses (Fig 80).

It is important to note that calibration of the SWAT model for nitrogen was unsuccessful for SBRR and these results should be interpreted with caution. It is recommended that more attention be paid to relative differences amongst scenarios as opposed to predicted values.

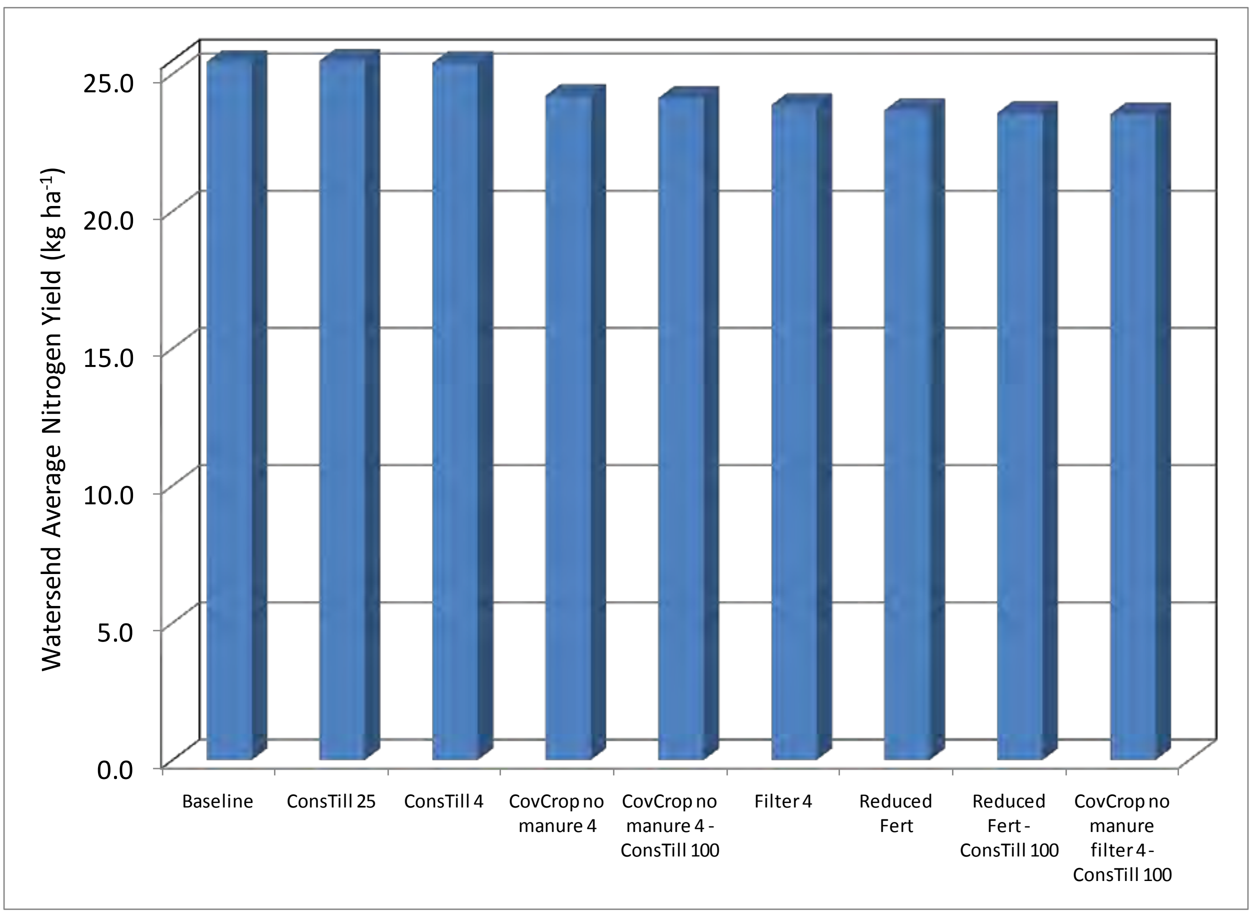


Figure 80. Watershed average nitrogen losses for a suite of alternative management scenarios in SBRR. Scenario abbreviations: **Baseline:** Baseline scenario with current conditions. **ConsTill25:** Conservation tillage applied to 25% of cropland in a non-targeted approach. **ConsTill4**: Conservation tillage applied to all cropland with a slope of greater than 4%. **CovCrop no Manure4**: Cover crops and no manure on cropland with a slope greater than 4%.

CovCrop no Manure 4-ConsTill100: Cover crops and no manure on cropland with a slope greater than 4%; conservation tillage on all remaining cropland. **Filter4:** 10 meter filter strip on all cropland with a slope greater than 4%. **Reduced Fert**: Reduced application rate of commercial N and P. **ReducedFert-ConsTill100:** Reduced application rate of commercial N and P; conservation tillage applied to all cropland. **CovCrop no Manure Filter 4-ConsTill 100**: Cover crops, no manure, and 10 meter filter strips on all cropland with a slope greater than 4%; conservation tillage on all remaining cropland.

Summary

The SWAT model was calibrated and validated to predict monthly flow, sediment, phosphorus, and nitrogen loss from the South Branch of the Root River watershed. Model calibration values ranged from fair to good for flow, sediment, and phosphorus. Despite acceptable validation values, the model failed calibration for nitrogen. Variability in precipitation data that was not captured by the rain gauges is likely to be the source of at least some of the disagreement between observed and model-predicted values. For nitrogen, poor performance is also likely to be the result of the karst geology in some portions of the watershed as well as a large inter-annual swing in total precipitation from a very dry year (2003) to a very wet year (2004 – the first year of model calibration). Results from this modeling effort pertaining to nitrogen should be interpreted with caution.

The calibrated and validated model was used to assess the current sources of sediment, phosphorus, and nitrogen from SBRR. Sediment and phosphorus were predicted to originate disproportionately from a small proportion of the landscape where row crops were grown on lands with slopes steeper than 4%. Nitrogen delivery to surface waters occurred via two main pathways: (1) subsurface tile drainage from poorly-drained soils in the western and southern portions of the watershed, and (2) shallow groundwater flow in the eastern and northern portions of the watershed. Nitrogen in shallow groundwater was particularly high in sub basins where dairy manure was applied to fields. These sub basins were located in the karst-influenced northwestern portion of SBRR.

Simulation of alternative management practices to reduce sediment loss in SBRR were most effective when targeted to the most sensitive portions of the landscape. In fact, applying conservation tillage to fields with slopes greater than 4% (representing 8% of the watershed) was four times more effective at reducing sediment erosion than applying conservation tillage to 25%

of the cropland in the watershed in a non-targeted manner. For reducing sediment loss from steep croplands, implementing 10m filter strips was the most effective practice, followed by planting cover crops, and lastly by conservation tillage. The scenarios that were most effective at reducing sediment loss (53% reduction from the baseline scenario) included a combination of cover crops and filter strips on steep croplands as well as conservation tillage on all cropland in the watershed.

Similar to sediment, reductions in phosphorus loss were greatest under scenarios that focused on croplands with slopes greater than 4% with the greatest overall reductions (28% reduction from the baseline scenario) coming from combined approaches of cover crops and filter strips. In contrast to sediment, increasing conservation tillage in SBRR had a negligible effect on phosphorus loss. This is due to the fact that, despite reductions in sediment loss, conservation tillage results in greater simulated crop residue decomposition in the SWAT model, effectively generating more phosphorus from the decomposed crop residue.

Reductions in nitrogen loss from SBRR were achieved via simulated reductions in commercial fertilizer application rates by 6% (to 120 lb/ac, 135 kg/ha) which produced a roughly equivalent reduction in nitrogen loss. Additional reductions in nitrogen loss were also predicted in scenarios with cover crops and filter strips as a result of increased nutrient uptake from the vegetation associated with these practices. While dramatic changes in manure application were not simulated here, model results from the baseline scenario suggest that applying manure to fields in karst regions of the watershed can result in excessive nitrogen reaching the shallow groundwater and streams. Additional management scenarios that address the costs and benefits of more targeted manure management approaches have the potential to dramatically reduce nitrogen loss in the karst-influenced regions of SBRR.

Overall, there is great potential for reducing sediment, phosphorus, and nitrogen loads in the South Branch of the Root River and its tributaries. The most effective approaches simulated with SWAT for this study are those that target crop lands that occur on slopes greater than 4% (there are additional structural practices that can also be used to achieve reductions of sediment and nutrient reductions in SBRR). Combined practices of cover crops and 10m filter strips on these steep fields have the potential to reduce sediment and phosphorus through reductions in surface runoff and edge-of-field losses; these practices can also reduce nitrogen losses via increased plant uptake of excess nitrogen in soil water.

References

David, M.B., S.J. Del Grosso, X. Hu, E.P. Marshall, G.F. McIsaac, W.J. Parton, C. Tonitto, M.A. Youssef. 2009. Modeling denitrification in a tile-drained, corn and soybean agroecosystem of Illinois, USA. *Biogeochemistry* 93: 7-30.

Engman, E.T. 1986. Roughness coefficients for routing surface runoff. *J. Irrigat. Drain. Eng.* 112(1): 39-53.

Herridge, D.F., M.B. Peoples, R.M. Boddey. 2008. Global inputs of biological nitrogen fixation in agricultural systems. *Plant Soil*. 311: 1-18.

Kozak, J.A., L.R. Ahuja, T.R. Green, L. Ma. 2007. Modelling crop canopy and residue rainfall interception effects on soil hydrological components for semi-arid agriculture. *Hydro. Proc.* 21: 229-241.

Motovilov, Y.G., L. Gottschalk, K. England, and A Rodhe. 1999. Validation of a distributed hydrological model against spatial observations. *Agric. Forest Meteorology 98-99:* 257-277.

Nash, J.E., and J.V. Sutcliffe. 1970. River flow forecasting through conceptual models: Part 1. A discussion of principles. *J. Hydrology* 10(3): 282-290.

Sharpley, A.N. and J.K. Syers. 1979. Phosphorus inputs into a stream draining an agricultural watershed: II. Amounts and relative significance of runoff types. Water, Air, and Soil Pollution 11: 417-428.

Walker, W.W. 1996. Simplified procedures for eutrophication assessment and prediction: User Manual, Instruction Report W-96-2, U.S. Army Engineer Waterways Experiment Station. Vicksburg, MS.

Appendix: FLUX model estimates of monthly loads for the South Branch of the Root River watershed. Months highlighted in green contained observed water quality data and were used for model calibration and validation.

Year	Month	Average Monthly Flow (m^3 sec^{-1})	Total Suspended Solids (tons)	NO_3^- & NO_2^- (kg)	Total Phosphorus (kg)
2004	Jan	1.04	8	29989	109
	Feb	1.09	8	28847	105
	Mar	3.94	1310	63474	2705
	Apr	0.76	8	19036	81
	May	3.22	1121	78554	2286
	Jun	7.19	3300	158129	6151
	Jul	3.42	475	70686	1100
	Aug	1.22	8	30156	122
	Sep	6.87	5255	99626	7640
	Oct	2.02	63	45501	260
	Nov	2.55	99	50999	351
	Dec	2.31	84	49522	315
2005	Jan	1.63	20	39839	171
	Feb	4.30	1320	61761	2646
	Mar	5.90	3792	132231	5941
	Apr	4.23	569	100652	1298
	May	3.20	167	79351	503
	Jun	2.99	148	67605	448
	Jul	4.52	2366	81531	4118
	Aug	2.58	106	56587	370
	Sep	3.56	1246	54864	2373
	Oct	2.50	104	51497	357
	Nov	1.79	33	40588	197
	Dec	2.06	67	46001	269
2006	Jan	1.89	53	43700	235
	Feb	1.90	49	39629	214
	Mar	3.71	770	72491	1662
	Apr	11.49	6694	263484	12153
	May	5.67	1422	137810	2943
	Jun	2.64	106	59793	369
	Jul	1.49	14	35790	153
	Aug	1.20	8	29829	121
	Sep	1.04	8	25804	105
	Oct	0.87	8	26598	93
	Nov	1.84	40	41432	209
	Dec	2.18	73	47702	289

Year	Month	Average Monthly Flow ($m^3\ sec^{-1}$)	Total Suspended Solids (tons)	NO_3^- & NO_2^- (kg)	Total Phosphorus (kg)
2007	Jan	2.45	99	50920	347
	Feb	1.31	7	31225	117
	Mar	8.09	4890	134594	8950
	Apr	5.19	1282	122253	2651
	May	2.32	86	58261	319
	Jun	3.22	173	72700	499
	Jul	1.42	17	34297	151
	Aug	4.60	1760	86881	3485
	Sep	2.80	127	55491	407
	Oct	5.15	943	81445	2037
	Nov	1.94	49	42906	230
	Dec	1.64	31	39889	187
2008	Jan	1.66	24	40236	178
	Feb	1.57	7	36396	141
	Mar	4.01	808	74837	1717
	Apr	8.80	4357	204181	8677
	May	4.63	365	113603	858
	Jun	18.13	15929	386998	22840
	Jul	2.71	120	58985	400
	Aug	1.51	17	36221	158
	Sep	1.05	8	26216	105
	Oct	1.19	13	32346	125
	Nov	1.71	29	38426	179