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Impact of Best Management Practices on Water Quality of Two Small Watersheds in Indiana: Role of Spatial Scale

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Impact of Best Management Practices on Water Quality of Two Small Watersheds in Indiana: Role of Spatial Scale

Prepared By

Mazdak Arabi
School of Civil Engineering
Purdue University
West Lafayette, Indiana 47907

Rao S. Govindaraju
School of Civil Engineering
Purdue University
West Lafayette, Indiana 47907

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Project Officer

Mohamed M. Hantush
National Risk Management Research Laboratory
Office of Research and Development
U.S. Environmental Protection Agency
Cincinnati, Ohio 45268

Notice

The U.S. Environmental Protection Agency through its Office of Research and Development funded the research described here. This report has been subjected to the Agency's peer and administrative review and has been approved for publication as an EPA document. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

All research projects making conclusions or recommendations based on environmentally related measurements and funded by the Environmental Protection Agency are required to comply with the Agency Quality Assurance Program. This report has been subjected to QA/QC review. The report presented a mathematical framework for modeling water quality in eutrophic water bodies and did not involve collection and analysis of environmental measurements.

Foreword

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This publication has been produced as part of the Laboratory's strategic long-term research plan. It is published and made available by EPA's Office of Research and Development to assist the user community and to link researchers with their clients.

Sally Gutierrez, Director
National Risk Management Research Laboratory

Abstract

Transport and fate of sediments and nutrients within watersheds have important implications for water quality and water resources. Water quality issues often arise because sediments serve as carriers for various pollutants such as nutrients, pathogens, and toxic substances. The Clean Water Act provision (CWA) [Section 303(d)] requires all states to develop and implement a Total Maximum Daily Load (TMDL) for their impaired water bodies, and water bodies that are likely to join this list. Implementation of Best Management Practices (BMPs) is a conventional approach for controlling nonpoint sources of sediments and nutrients. However, implementation of BMPs has rarely been followed by a good long-term data monitoring program in place to study how effective they have been in meeting their original goals. Long-term data on flow and water quality within watersheds, before and after placement of BMPs, is not generally available. Utility of mathematical models provides an effective and powerful tool for evaluation of long-term performance of BMPs (especially new ones that have had little or no history of use). In this study, a process-based modeling framework is developed to evaluate the effectiveness of parallel terraces, field borders, grassed waterways, and grade stabilization structures in reducing sediment and nutrient yields in two small agricultural watersheds (<10 km^2) in Indiana, with Soil and Water Assessment Tool (SWAT) serving as the watershed model. Based on the functionality of each BMP, appropriate model parameters are selected and altered to represent the effect of the BMP on hydrologic and water quality processes. A sensitivity analysis is performed to evaluate the sensitivity of model computations to selected parameters. Results indicated that parallel terraces and field borders were effective at a field scale, while grassed waterways and grade stabilization structures were the more effective BMPs at a watershed scale.

Distributed-parameter models partition the watershed into subunits (subwatersheds/hyrologic response units/grids) during computations to represent heterogeneity within the watershed. Homogeneous properties are assumed over each computational unit. Identification of the stream network and partitioning of the study area into subunits may significantly affect hydrologic and waters quality simulations of a distributed-parameter model. Because model outputs are affected by geomorphologic resolution, the evaluation of performance of BMPs based on model predictions will be influenced as well. Thus, examination of the efficacy of

BMPs must be conducted in conjunction with studies performed at multiple spatial scales. In this study, sediment and nutrient outputs from the calibrated SWAT model are compared at various watershed discretization levels both with and without implementation of these BMPs. Results indicated that evaluation of the impacts of these BMPs on sediment and nutrient yields at the outlet of the two agricultural watersheds in Indiana was very sensitive to the level of discretization that was applied for modeling. An optimal watershed discretization level for representation of the BMPs was identified through numerical simulations. It would appear that the average subwatershed area corresponding to approximately 4% of total watershed area is needed to represent the influence of BMPs in a modeling effort.

It should be noted that the results of this study are location-dependent, and also depend on the type of BMPs. However, the methodology can be utilized for similar studies in other watersheds with different BMPs.

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Acknowledgements

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Section 1.0
Introduction

1.1 Background and Rationale

Sediment and nutrient yield from a watershed have important implications for water quality and water resources. Water quality issues often arise because sediments serve as carriers for various pollutants such as nutrients, pathogens, and toxic substances. Surface water quality is important not only for protection of fish and aquatic life, but it is often used as an indicator of the environmental health of a watershed. Increased sediment load to a watershed can be detrimental to an entire ecosystem. Land use changes over the years have had an enormous effect on sediment levels in surface waters throughout the United States. Important sources of sediments include erosion from agricultural fields, construction sites and reclaimed mining areas. Estimates of sediment and nutrient yield are required for a wide spectrum of problems dealing with dams and reservoirs, fate and transport of pollutants in surface waters, design of stable channels, protection of fish and other aquatic life, watershed management and for environmental impact statements.

Often, sediments in surface water bodies are contaminated with chemicals that sorb onto fine-grained organic and inorganic soil particles. Sources of such contamination can result from either existing point or non-point sources, historical spills, or discharges. When such contamination exceeds critical levels, ecological and human health risks require appropriate remedial actions. Such remedial measures take the form of isolating the contaminated sediments, reducing their exposure to other parts of the ecosystem, complete removal of the contaminated sediment, or some combination of the above. For all such measures, an accurate understanding of the fate and transport of sediments/contaminants is crucial for designing suitable remediation measures.

The Clean Water Act provision (CWA) [Section 303(d)] requires all states to develop and implement a Total Maximum Daily Load (TMDL) for their impaired water bodies, and water bodies that are likely to join this list. Implementation of the TMDL program is now considered to be pivotal in securing the nation's water quality goals (NRC, 2001). A TMD is the maximum of point and nonpoint source loads that can enter a water body without exceeding specified water quality standards. Over the past 30 years, some success has been achieved by reducing pollution from point sources such as sewage treatment plants and industrial discharges. However, controlling the pollution from nonpoint sources, which is essential to the successful implementation of TMDL, still requires more study. According to the most recent lists submitted to EPA, there are nearly 26,000 impaired water bodies in the nation. Sediment/siltation and nutrients together are the major concern for approximately 11,000 of these water bodies, thus the most common impairments are sediment related.

Once a water body is listed as impaired and its type of impairment is classified as sediment and nutrients, water quality modeling is required to make predictions that support the TMDL process. Water quality modeling for TMDL development usually involves watershed modeling and waterbody modeling. While the latter is necessary to determine pollutant concentrations as a function of pollutant loads into the waterbody, the former is employed to predict the pollutant loads into a waterbody as a function of watershed characteristics such as slope of the watershed, land use, soil series, and management practices. Watershed models are also utilized to evaluate the effectiveness of abatement strategies such as implementation of Best Management Practices (BMPs). Watershed models have been classified into various categories including empirical vs. physically-based, event-based vs. continuous, and lumped vs. distributed-parameter models. Selection of a suitable model depends on several factors such as capability to simulate design variables (runoff, groundwater, sediment yield, nutrient yield, etc.), accuracy, available data, and temporal and spatial scales.

Spatial scale is an important consideration in watershed modeling. In large watersheds, channel processes tend to become more important while in small watersheds hydrology is usually dominated by overland flow. The validity of the predictions of a watershed model depends on how well the spatially heterogeneous characteristics of the watershed are represented by the model inputs. Lumped models consider a watershed as a single unit for computations, and watershed parameters are averaged over this unit. The ability to represent spatial variability inherent in watershed characteristics is the reason that distributed models have been favored over lumped models. Distributed models partition a watershed into subunits (subwatersheds, HRUs, or grids) for simulation purposes, and homogeneous properties are assumed for each subunit. Because model inputs are averaged over a subunit, model simulations are greatly influenced by the size and number of subunits.

Currently, watershed delineation and extraction of stream networks are accomplished with GIS databases of Digital Elevation Models (DEMs). The most common method for extracting channel networks requires the *a-priori* specification of a critical source area that is required for channel initiation. The nature of the channel network is very sensitive to this critical source area, with drainage density decreasing exponentially with increasing critical source area. Thus, the channel network could be viewed at multiple scales within the same watershed. There are no established guidelines on how to select this critical source area. Thus, for the same watershed and Digital Elevation Model (DEM), users may obtain markedly different channel networks, and subsequently the watershed model results based on the channel network could be affected as well. The challenge is to identify an optimal scale of geomorphologic resolution such that further refinement in spatial scale does not contribute to a significant improvement in predicting design parameters at the watershed outlet. Such an optimal spatial scale, if identifiable, can be further used for identification of nonpoint sources of sediments and nutrients.

Natural sources of sediment are primarily upland areas where erosion, including both sheet and rill erosion, is dominated by overland flow, or in ephemeral gullies. Sheet erosion results in removal of a fairly uniform layer of sediment from an area, while rill erosion is restricted to concentrated channel flows. Large runoff events, like those that occur during a flood event, can lead to mass sediment and nutrient removal. Anthropogenic activities may lead to creation of important sources of sediments and nutrient, among which agricultural tillage has the strongest influence. Highway construction, timber cutting, mining, urbanization, land development for recreational use and animal grazing also contribute to varying degrees. Large channels within a watershed not only serve as the source for movement of contaminant-laden sediments, but may also act as a source because of erosion from streambeds or banks. On the other hand, depending on the main channel geometry, sediment particles could be deposited in the main channel. In the latter case, there is a significant difference between sediment and nutrient loads generated from upland areas and the ones measured at the outlet of the watershed. Considering this phenomenon, implementation of sediment and nutrient reduction plans will be highly affected by the control processes within the watershed. For example, in a transport limited watershed, the transport capacity of the watershed stream network is less than the sediment generated in upland areas (Keller et al., 1997).

Identification of sources of sediment and nutrients within a watershed is necessary for developing control measures. Most modeling strategies have focused on the forward problem of predicting sediment and/or contaminant concentrations given the source locations and strengths. While good geomorphologic data on stream networks and soil types are available within watersheds from GIS databases, most monitoring programs are located at the watershed outlet. Thus, detailed information within a watershed is rarely available at a resolution that would enable proper identification of sediment or contaminant sources. This problem is complicated because sediment and contaminants are carried along with the flow, and water movement over a watershed tends to be fairly dynamic, behaving in a nonlinear fashion. Previous studies do not provide a good modeling framework for identification of sediment and nutrient sources within a watershed. Specifically, a methodology that could be utilized to identify the control processes and management actions on sediment and nutrient movement have not been developed.

1.2 Objectives

The overall objectives of this study are:

1. <u>Evaluation of effectiveness of Best Management Practices (BMPS) in reduction of sediment and nutrient yields</u>: BMPs are conventional tools used widely as sediment and nutrient reduction plans. While a few studies have addressed the effectiveness of some BMPs (Mostaghimi et al., 1997; Williams et al., 2000; Vache et al., 2002; Yuan et al., 2002a,b; Dybala, 2003; Santhi et al., 2003), the importance of scale (i.e. watershed or farm scale) has been neglected in the appraisal of the BMPs. In this research, the long term impacts of BMPs on water

quality will be studied. The effectiveness of BMPs will be evaluated at the watershed scale and farm (subwatershed) scale.

2. Investigation of role of watershed discretization on model simulations and evaluation of effectiveness of BMPs in reduction of sediment and nutrient yields: There are two sub-problems that result from multi-scale effects. First, an optimal scale of geomorphologic resolution needs to be identified such that further refinement in spatial scale does not contribute to a significant improvement in simulating design quantities at the watershed outlet. This optimal geomorphologic resolution, along with the associated drainage density, can then be utilized to determine the appropriate critical source area. Second, the role of spatial scale on evaluation of the efficacy of BMPs will be investigated. Because model results are affected by the geomorphologic resolution, the predicted performance of BMPs will be influenced by model parameters.

The remainder of this report is organized in six sections. Section 2.0 reviews the characteristics of the study area and the watershed model that were used in this study. The following criteria are utilized for selecting a watershed that will support the proposed objectives: (i) The watershed should have been listed as and impaired waterbody by EPA or the state authorities, (ii) BMPs must have been implemented for nonpoint source pollution control and (iii) Daily water quality data (streamflow, and sediment and nutrient loads) should have been collected at the outlet of the watershed for a reasonable period of time. Various components of the selected watershed model are also discussed in this section. The effect of watershed discretization on various hydrologic and water quality components of the selected model is presented in Section 3.0. The possibility of identifying an optimal critical source area for the model simulations and the conditions when such an identification is relevant will be examined. A simple process-based index will be developed to help identification of a proper watershed configuration prior to model calibration. The procedure adopted for representation of Best Management Practices (BMPs) and model calibration is described in Section 4.0. A discussion on the role of watershed discretization effects on evaluation of the effectiveness of BMPs is provided in Section 5.0. Section 6.0 presents the utility of the watershed model in generation of sediment and nutrient source maps that can be used in TMDL development. Overall conclusions of the study are summarized in Section 7.0.

Section 2.0
Watershed Description and Model Selection

2.1 Introduction

To achieve the goals of this project, a suitable watershed model is selected, calibrated, and validated for a study area where adequate water quality data are available. The availability of a rather unique dataset for a particular watershed, and how this will be utilized to meet the project goals will be described briefly.

2.2 The Study Area and Available Data

For the objectives of this research to be successfully fulfilled a watershed must be selected where BMPs have been implemented and adequate hydrologic and water quality data including rainfall, streamflow, and sediment and nutrient yields are available. Various watersheds in the United States have been studied for evaluation of the effects of BMPs on water quality (Batchelor et al., 1994; Park et al., 1994; Griffin, 1995; Edwards et al., 1996; Saleh et al., 2000; Santhi et al., 2001; Saleh and Du, 2002; Vache et al. 2002; Santhi et al., 2003). However none of these studies had the data needed for a thorough evaluation of the influence of BMPs. After an exhausting search, the Black Creek watershed, northeast Indiana, was identified as perhaps one of the very few watersheds with both daily measured water quality data and with detailed information on various implemented BMPs. This watershed is also preferred because daily water quality data were measured at two outlets within the watershed (Figure 2.1). This allows for further validation of the conclusions of this study.

A study on the Black Creek watershed, funded by EPA, was conducted in 1970s and early 1980s to examine the short-term effects of soil and water conservation techniques on improving water quality by reducing sediment and nutrient loads leaving the watershed. This watershed, located in Allen County, northeast Indiana (see Figure 2.1) is an approximately 50 km^2 (12,000 acre) watershed in the Maumee River basin. In this previous study, detailed water quality monitoring was carried out during the duration of the project. Nineteen major

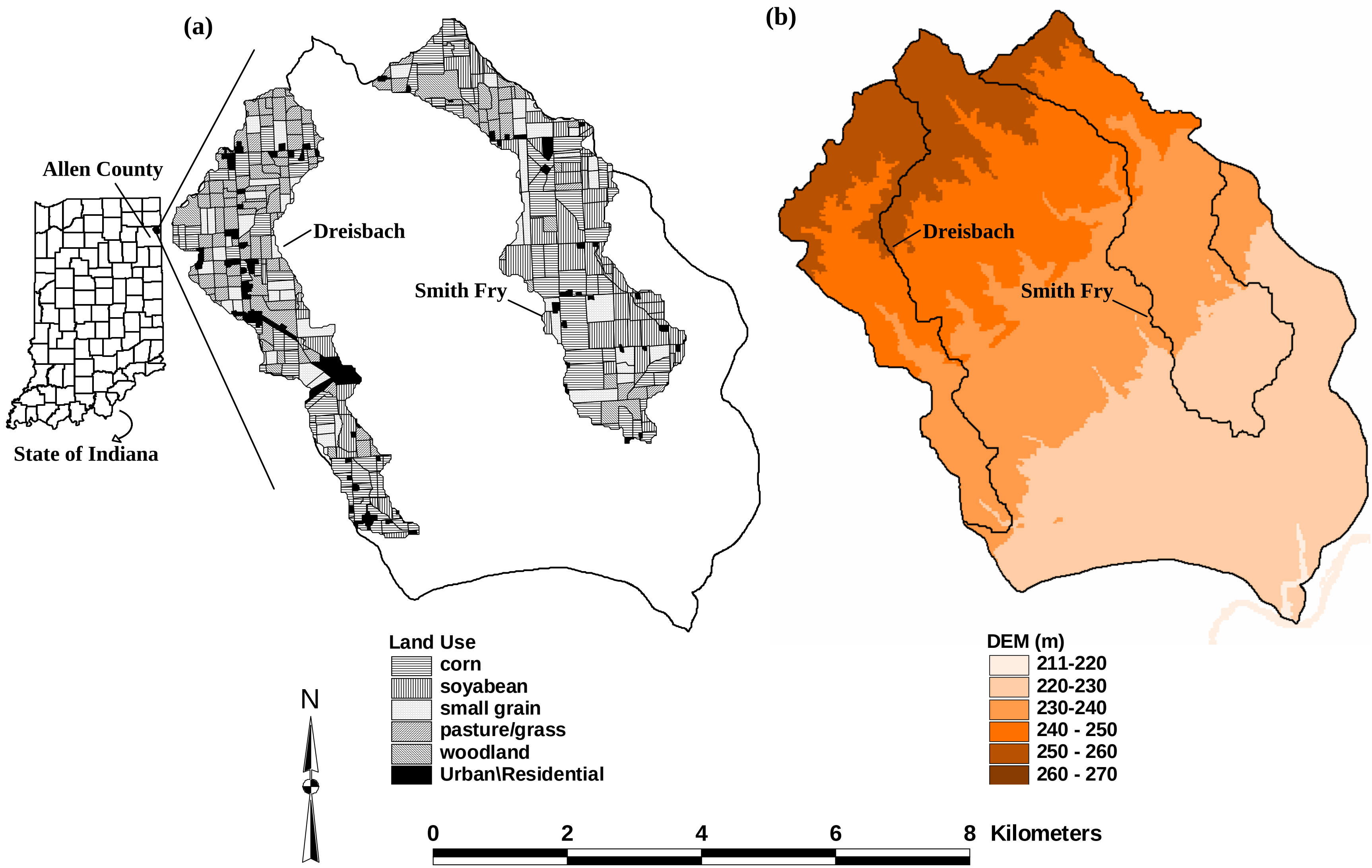


Figure 2.1. (a) Land Use, (b) Digital Elevation Model (DEM) for the Dreisbach and Smith Fry Watersheds, Allen County, Indiana.

monitoring stations were established within the watershed. However, data collected from automated samplers located at Smith Fry and Dreisbach outlets were the most complete and were used for most of the analysis reported in the project. The areas of the Smith Fry and Dreisbach watersheds shown in Figure 2.1 are 7.3 km^2 and 6.23 km^2, respectively. Daily precipitation, streamflow, and sediment and nutrient loads were recorded at the outlet of these two watersheds. Land use in the Dreisbach watershed (Figure 2.1a) is mostly pasture in the upper portion, while cropland is wide spread in remainder of the watershed. Land use in the Smith Fry watershed is mostly croplands (see Figure 2.1a). Table 2.1 presents land use distributions for the two watersheds.

The Digital Elevation Model (DEM) for the study area is shown in Figure 2.1(b). The dominant hydrological soil group of soil series in both watersheds is type C. Major soil series in the two watersheds are listed in Table 2.2.

Table 2.1 Land Use in the Dreisbach and Smith Fry Watersheds, Indiana.

Land Use	% Dreisbach Area	% Smith Fry Area
Pasture (PAST)	37.55	8.72
Corn (CORN)	23.38	33.59
Soybean (SOYB)	7.22	31.84
Winter Wheat (WWHT)	16.97	14.28
Forest (FRSD)	5.83	8.93
Residential- Low Density (URLD)	9.06	2.64

Table 2.2. Major Soil Series in the Dreisbach and Smith Fry Watersheds, Indiana.

Soil	% Dreisbach Area	% Smith Fry Area	Hydrologic group
WHITAKER	3.77	11.25	C
RENSSELAER	9.1	20.64	B
MORLEY	40.88	8.35	C
PEWAMO	13.31	5.77	C
NAPPANEE	3.68	4.23	D
HOYTVILLE	5.52	10.7	C
BLOUNT	14.87	22.21	C

There were 26 best management practices (BMPs) installed in the Dreisbach watershed in 1974 while this number was 6 for the Smith Fry watershed. The BMPs were installed in the Smith Fry watershed in 1975. The types and locations of the BMPs in the Dreisbach and Smith Fry watersheds are shown in Figure 2.2.

2.3 Model Selection

2.3.1 Background

Watershed models are utilized to better understand the role of hydrological processes that govern surface and subsurface water movement. Moreover, they provide assessment tools for decision making in regard to water quality issues. Watershed models have been classified into various categories including empirical vs. physically-based, event-based vs. continuous, and lumped vs. distributed-parameter models. Selection of a suitable model depends on several factors such as capability to simulate design variables (runoff, groundwater, sediment yield, nutrient yield, etc.), accuracy, available data, and temporal and spatial scales.

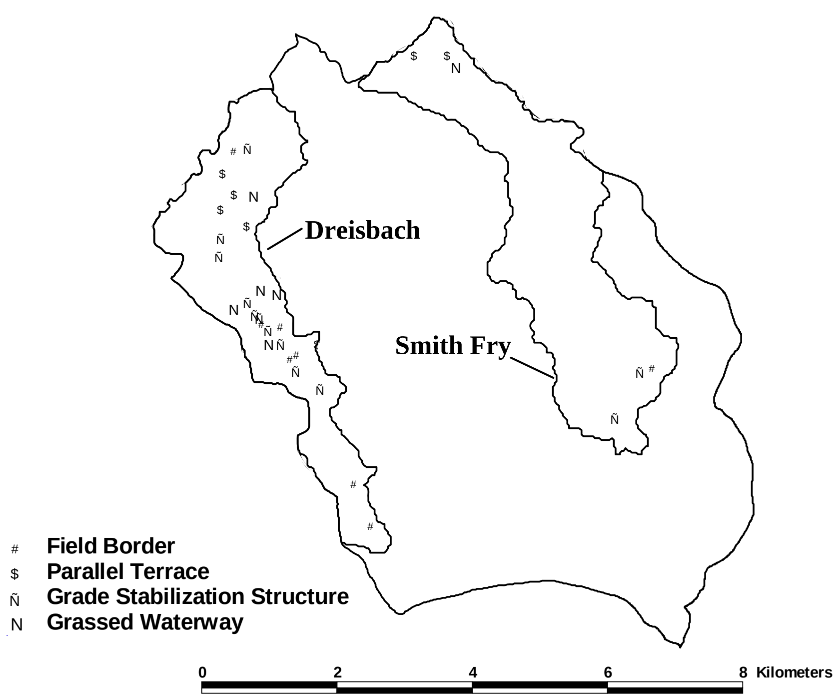


Figure 2.2. Type and Location of BMPs in the Dreisbach and Smith Fry Watersheds, Indiana.

Empirical models are developed based on statistical relationships between design parameters and watershed characteristics. These relationships are obtained from regression analysis using observed data. Application of these models will likely be limited to the same statistical conditions over which the observed data were acquired. For example, the well-known Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) was developed based on statistical analysis of many years of rainfall, runoff, and soil loss data from many small plots around the United States, and is suitable for estimation of average annual soil loss from a field based on steepness, soil series, land use, and management practice. Application of the USLE for daily and/or monthly estimation of soil loss may not yield realistic results. These limitations do not hold for physically-based models as they are grounded in physical principles of conservation of mass, energy, and momentum. These models are preferred because they provide a better understanding of the processes in the watershed. Many models utilize both empirical and physically-based relationships to represent hydrologic and water quality processes within a watershed, and may be labeled as process-based models.

Lumped models consider a watershed as a single unit for computations, and watershed parameters are averaged over this unit, while distributed-parameter models partition a watershed into subunits (subwatersheds, HRUs, or grids) for simulation purposes, and homogeneous properties are assumed for each subunit. As a result, the number of input parameters increases significantly. However, the spatial variability of watershed parameters such as land use, soil series, and management actions are more easily represented in distributed-parameter models.

In addition to spatial scale, watershed models utilize different temporal scales for computations. Event-based models usually require small time steps, at times in the order of seconds. These models are suitable for analyzing influence of design storms. Larger time steps, in the order of days, are usually sufficient for continuous models that are appropriate for long term assessment of hydrological and land use change and watershed management practices.

Water quality models estimate sediment and nutrients loads through prediction algorithms. These are primarily empirical in nature and use versions of the Universal Soil Loss Equation

(USLE) (Wischmeier and Smith, 1978) for sediment loads as in AGNPS (Young et al., 1987, 1989) and SWRRBQ (Renard et al., 1997). Particle detachment and wash equations are utilized in HSPF (Bicknell et al., 1993), ANSWERS (Beasley et al., 1980), and other models. AGNPS and ANSWERS evaluate sediment transport associated with individual events, while models like HSPF and SWAT (Neitsch et al., 2001a,b) utilize hourly or daily time steps and are better suited for long-term simulations. Some of these models can be used to estimate sediment erosion and nutrient loads from multiple source categories and can track the fate and transport of sediments and nutrients. Therefore, they are well-suited to providing useful information on sediment and nutrient yields from different regions of a watershed (Reid and Dunne, 1996). While these models can delineate sediment and nutrients sources at the point-scale in principle, the problem would have to be posed in an inverse sense, and would entail very substantial amounts of data requirements and computer effort.

Borah (2002) reviewed eleven continuous-simulation and single-event watershed scale models including the ones mentioned above. The study provides a better understanding of the mathematical bases of the models. Among all, the Soil and Water Assessment Tool (SWAT) model is the only continuous/process-based/distributed-parameter model that contains both sediment and nutrient components and is capable of representing BMPs at a watershed scale.

Implementation of the Best Management Practices (BMPs) is a conventional approach for nonpoint source pollution control. Various watershed and field scale models have been used to simulate the effectiveness of BMPs (Bachelor et al., 1994; Park et al., 1994; Edwards et al. 1996). The WEPP model (Flanagan and Livingston, 1995) has the most mechanistic sediment transport component and can simulate various BMPs including agricultural practices (e.g. tillage, contouring, irrigation, drainage, crop rotation, etc.), ponds, terraces, culverts, filter fences and check dams (Kalin and Hantush, 2003). However, the application of the model is limited to field scale studies or very small watersheds (<3 km^2). Most of models with good representation of BMPs, such as WEPP, are more applicable to field scale studies.

Kalin and Hantush (2003) reviewed key features and capabilities of widely cited watershed scale hydrologic and water quality models with emphasis on the ability of the models in representation of Best Management Practices (BMPs) and TMDL development. The review indicated that the SWAT and AGNPS models offer the most management alternatives for modeling of agricultural watersheds. In this study, the Soil and Water Assessment Tool (SWAT) model is selected as the watershed model.

Saleh and Du (2002) compared the performances of the SWAT and HSPF models, both integrated into BASINS framework, in predictions of streamflow, sediment yield, and nutrient loads. The authors suggested that SWAT is more user-friendly and had a better prediction of nutrient loads while HSPF streamflow and sediment predictions were closer to the measured data. The SWAT model and HSPF model streamflow predictions were also compared by Van Liew et al. (2003). They found that although the modeling errors were smaller for HSPF in the calibration period, SWAT exhibited more robustness during the calibration and validation periods. The robustness refers to more acceptable error statistics during validation period. They also concluded that SWAT might be more suitable for long-term assessment of the effects of climate variability on surface water resources.

The SWAT model has been widely used for streamflow, sediment yield, and nutrient load predictions. The SWAT model development, operation, limitations, and assumptions were discussed by Arnold et al. (1998). Srinivasan et al. (1998) reviewed the applications of the SWAT model in streamflow prediction, sediment and nutrients transport, and effects of management practices on water quality. Arnold and Allen (1996) evaluated the performance of different hydrologic components of the SWAT model for three watersheds in Illinois (100-250 km^2). Comparing the model outputs to measured data, the calibrated model reasonably simulated runoff, groundwater, and other components of hydrologic cycle for the study watersheds. Most simulated average monthly outputs were within 5% of the historical data and nearly all of them were within 25%. ^{R2} (correlation coefficient) statistic was used to compare the correlation between the observed and simulated average monthly variables. Also, the interaction among various components of hydrologic budgets was recognized to be realistic. SWAT was utilized in a study by Arnold et al. (2000) to compare the performance of two baseflow and groundwater recharge models. The first model was the water balance components of the SWAT model. A combination of a digital hydrograph separation tool and a modified hydrograph recession curve displacement technique composed the second model. The results of the two models were in general agreement in the Upper Mississippi river basin. A detailed procedure for calibration of SWAT was laid out by Santhi et al. (2001a). Jha et al. (2003) found curve Number (CN) as the most sensitive parameter in streamflow prediction. A series of studies have been carried out with SWAT to model sediment and nutrients transport within Upper North Bosque River watershed (4277 km^2), TX (Saleh et al., 2000; Santhi et al., 2001a; Santhi et al., 2001b; Saleh and Du, 2002; Santhi et al., 2003). Manure application to pasture and cropland is the main nonpoint source pollution concern in this large watershed. Dairy management practices have been utilized for phosphorus load control. In conclusion, SWAT performance has been extensively validated for streamflow, and sediment and nutrients yield predictions for different regions of United States.

SWAT has been applied to evaluate the effects of a number of BMPs such as waterways, filter strips, and field boarders on streamflow and sediment and nutrients annual loads from U.S. Corn Belt (Vache et al. 2002). The study indicated that implementation of BMPs resulted in 30 to 60% reduction in sediment and nutrients loads. The SWAT model was utilized by Kirsch et al. (2002) to appraise the effectiveness of BMPs on reduction of sediment and phosphorus load over Rock River Basin (9708 km^2), WI. The BMP practices analyzed included modifications in tillage operations, and adoption of recommended nutrient application rates. They concluded that implementation of modified tillage practices would result in almost 20% sediment reduction. Additional in-stream modeling, and field scale water quality screening was recommended.

In conclusion, SWAT performance has been extensively validated for streamflow, and sediment and nutrients yield predictions for different regions of the United States. The model has also been successfully utilized for representation of various management scenarios. In this study, the Soil and Water Assessment Tool (SWAT) model is selected to simulate fate and transport of sediments and nutrients in the Dreisbach and Smith Fry watersheds.

2.3.2 SWAT Model Description

Soil and Water Assessment Tool (SWAT) (Neitsch et al., 2001a,b) has been widely used for watershed scale studies dealing with water quantity and quality. SWAT is a process-based distributed-parameter simulation model, operating on a daily time step. SWAT partitions the watershed into subwatersheds, each of which is treated as an individual unit. The model has also been integrated into USEPA's modeling framework, Better Assessment Science Integrating Point and Nonpoint Sources (BASINS). This framework provides users with a watershed delineation tool that enables users to automatically or manually delineate the watershed based on a Digital Elevation Model (DEM). A stream definition value is required by the delineation tool for watershed delineation. Selecting several different values for stream definition and by comparing the predicted sediment and nutrient yields, the role of subwatershed division on predicted responses of water and contaminant fluxes from the watershed can be examined to address the issue of spatial resolution required for modeling purposes. The SWAT model needs to be calibrated and validated for the study area to ensure that model parameters are representative for the study region.

SWAT is a process-based based model, operating on a daily time step. The model was originally developed to quantify the impact of land management practices in large, complex watersheds with varying soils, land use, and management conditions over a long period of time. SWAT uses readily available inputs and has the capability of routing runoff and chemicals through streams and reservoirs, and allows for addition of flows and inclusion of measured data from point sources. The model is capable of simulating long periods for comparing the effect of management changes. Moreover, SWAT has the capability to evaluate the relative effects of different management scenarios on water quality, sediment, and agricultural chemical yield in large, ungaged basins. Major components of the model include weather, surface runoff, return flow, percolation, evapotranspiration (ET), transmission losses, pond and reservoir storage, crop growth and irrigation, groundwater flow, reach routing, nutrient and pesticide loads, and water transfer.

For simulation purposes, SWAT partitions the watershed into subunits including subbasins, reach/main channel segments, impoundments on main channel network, and point sources to set up a watershed. Subbasins are divided into hydrologic response units (HRUs) that are portions of subbasins with unique land use/management/soil attributes.

SWAT uses a modification of the SCS curve number method (USDA Soil Conservation Service, 1972) or Green and Ampt infiltration method (Green and Ampt, 1911) to compute surface runoff volume for each HRU. The SCS curve number equation is:

$$Q_{surf} = \frac{(R_{day} - I_a)^2}{(R_{day} - I_a + S)} \tag{2.1}$$

where Q_{surf} is the accumulated runoff or rainfall excess (mm water), R_{day} is the rainfall depth for the day (mm water), I_a is initial abstraction which includes surface storage, interception and infiltration prior to runoff (mm water), and S is the retention parameter (mm water).

$$S = 25.4(\frac{1000}{CN} - 10) \tag{2.2}$$

where CN is the SCS runoff curve number. The initial abstraction, I_a, is commonly approximated as 0.2S:

$$Q_{surf} = \frac{(R_{day} - 0.2S)^2}{(R_{day} + 0.8S)} \tag{2.3}$$

Runoff will only occur when $R_{day} > I_a$.

Peak runoff rate is estimated using a modification of the rational method. Daily or sub-daily rainfall data is used for calculations. The rational formula is:

$$q_{peak} = \frac{C.i.Area}{3.6} \tag{2.4}$$

where q_{peak} is the peak runoff rate (m^3/s), C is the runoff coefficient, i is the rainfall intensity (mm/hr), $Area$ is the HRU area (km^2), and 3.6 is a unit conversion factor. Flow is routed through the channel using a variable storage coefficient method developed by Williams (1969) or the Muskingum routing method.

Erosion and sediment yield are estimated for each HRU with the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1975):

$$sed = 11.8(Q_{surf}.q_{peak}.area_{hru})^{0.56}.K_{USLE}.C_{USLE}.P_{USLE}.LS_{USLE}.CFRG \tag{2.5}$$

where sed is the sediment yield on a given day (metric tons), Q_{surf} is the surface runoff volume (mm water), q_{peak} is the peak runoff rate (m^3/s), $area_{hru}$ is the area of the HRU (ha), K_{USLE} is the USLE soil erodibility factor, C_{USLE} is the USLE cover and management factor, P_{USLE} is the USLE support practice factor, LS_{USLE} is the USLE topographic factor, and $CFRG$ is the coarse fragment factor.

Sediment deposition and degradation are the two dominant channel processes that affect sediment yield at the outlet of the watershed. Whether channel deposition or channel degradation occurs depends on the sediment loads from upland areas and transport capacity of the channel network. If sediment load in a channel segment is larger than its sediment transport capacity, channel deposition will be the dominant process. Otherwise, channel degradation (i.e. channel erosion) occurs over the channel segment. SWAT estimates the transport capacity of a channel segment as a function of the peak channel velocity:

$$T_{ch} = a.v^b \tag{2.6}$$

where T_{ch} (ton/m^3) is the maximum concentration of sediment that can be transported by streamflow (i.e. transport capacity), a and b are user defined coefficients, and v (m/s) is the peak channel velocity. The peak velocity in a reach segment is calculated:

$$v = \frac{\alpha}{n} R_{ch}{}^{2/3} S_{ch}{}^{1/2} \tag{2.7}$$

where α is the peak rate adjustment factor with a default value of unity, n is Manning's coefficient, R_{ch} is the hydraulic radius (m), and S_{ch} is the channel invert slope (m/m).

Channel degradation (Sed_{deg}) and deposition (Sed_{dep}) in tons are computed as:

$$sed_i > T_{ch} \quad : \quad sed_{dep} = (sed_i - T_{ch}) \times V_{ch} \quad \& \quad sed_{\deg} = 0 \tag{2.8}$$

$$sed_i < T_{ch} \quad : \quad sed_{\deg} = (T_{ch} - sed_i) \times V_{ch} \times K_{ch} \times C_{ch} \quad \& \quad sed_{dep} = 0 \tag{2.9}$$

where sed_i is the initial sediment concentration in the channel segment (ton/m^3), V_{ch} is the volume of water in the channel segment (m^3), K_{ch} is the channel erodibility factor (cm/hr/Pa), and C_{ch} is the channel cover factor. The total amount of sediment that is transported out of the channel segment (sed_{out}) in tons is computed as:

$$sed_{out} = (sed_i + sed_{\deg} - sed_{dep}) \times \frac{V_{out}}{V_{ch}} \tag{2.10}$$

In (5), V_{out} is the volume of water leaving the channel segment (m^3) at each time step.

Movement and transformation of several forms of nitrogen and phosphorus over the watershed are accounted within the SWAT model. Nutrients are introduced into the main channel and transported downstream through surface runoff and lateral subsurface flow. Major phosphorous sources in mineral soil include organic phosphorus available in humus, mineral phosphorus that is not soluble, and plant available phosphorus. Phosphorus may be added to the soil in the form of fertilizer, manure, and residue application. Surface runoff is the major carrier of phosphorous out of most catchments (Sharpley and Syers, 1979). The transformation of phosphorus in the soil is controlled by the phosphorus cycle (see Figure 2.3). Unlike phosphorus that has low solubility, nitrogen is highly mobile. Major nitrogen sources in mineral soil include organic nitrogen available in humus, mineral nitrogen in soil colloids, and mineral nitrogen in solution. Nitrogen may be added to the soil in the form of fertilizer, manure, or residue application. Plant uptake, denitrification, volatilization, leaching, and soil erosion are the major mechanisms of nitrogen removal from a field. In the

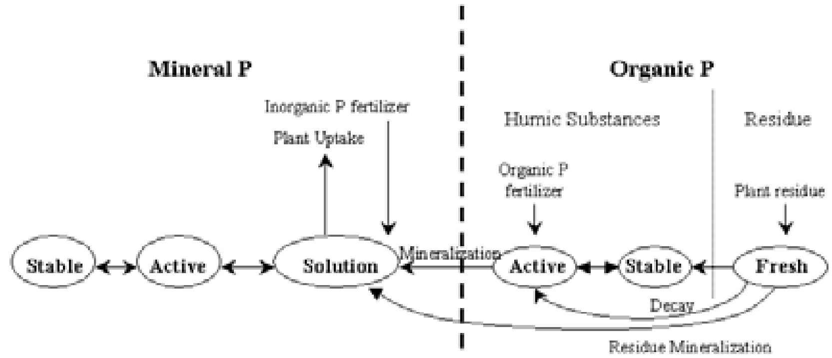


Figure 2.3. Phosphorus Processes Modeled in SWAT (USDA-ARS, 1999).

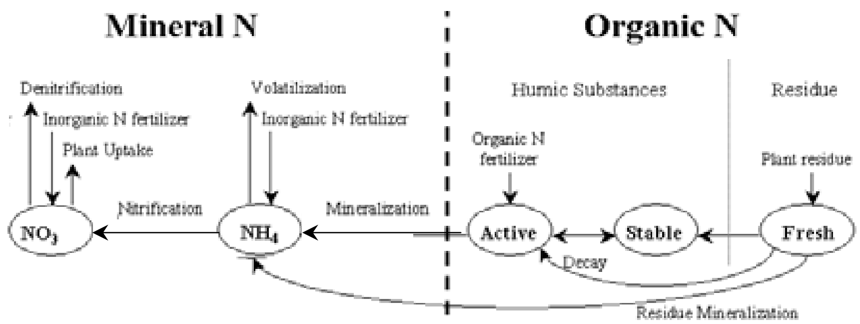


Figure 2.4. Nitrogen Processes Modeled in SWAT (USDA-ARS, 1999).

soil, transformation of nitrogen from one form to another is governed by the nitrogen cycle (see Figure 2.4).

SWAT simulates pesticide movement into the stream network via surface runoff (in solution and absorbed to sediment transported by runoff), and into the soil profile of the underlying aquifer by percolation (in solution). The equations used to model the movement of the pesticide were adopted from GLEAMS (Leonard et al., 1987).

The movement of water, sediment, and nutrients through the channel network of the watershed to the outlet is simulated by routing in main channel and reservoirs.

Very detailed management input data are required for a SWAT simulation including general management practices such as tillage, harvest and killing, pesticide application, fertilizer application, and irrigation management. Input data needed to run the SWAT model include soil, land use, weather, rainfall, management conditions, stream network, and watershed configuration. The summary output file (output.std), the HRU output file (.sbs), the subbasin output file (.bsb), and the main channel or reach output file (.rch) are the primary output files generated in every SWAT simulation. Users can refer to Soil and Water Assessment Tool User's Manual and theoretical documentation version 2000 (Neitsch et al., 2001a,b), published by the Agricultural Research Service and The Texas Agricultural Experiment Station, Temple, Texas for a detailed description of SWAT model. Various documents are also available at the SWAT website: *www.brc.tamus.edu/swat/index.html*.

SWAT has been integrated into USEPA's modeling framework, Better Assessment Science Integrating Point and Non-point Sources (BASINS). This framework provides users with a watershed delineation tool that allows automatic or manual watershed delineation based on Digital Elevation Model (DEM) data. BASINS is available for free download at: *www.epa.gov/waterscience/basins*.

2.3.3 Model Inputs

The SWAT model requires inputs on weather, topography, soil, land use, management, stream network, ponds, and reservoirs. The BASINS framework is used to develop the input parameters.

Climate Inputs

Daily precipitation from January 1974 to June 1977 was obtained from the monitoring station located at the outlet of the Dreisbach and Smith Fry watersheds. The elevation of the outlet of the Dreisbach watershed is 230 m above see level while the elevation of the outlet of the Smith Fry watershed is 222 m. The recorded daily precipitation for the two watersheds was published in the Black Creek project data report (Lake and Morrison, 1978). This information was converted to a tabular form and is available at: *http://pasture.ecn.purdue.edu /ABE/blackcreek/original_ data/Weather*.

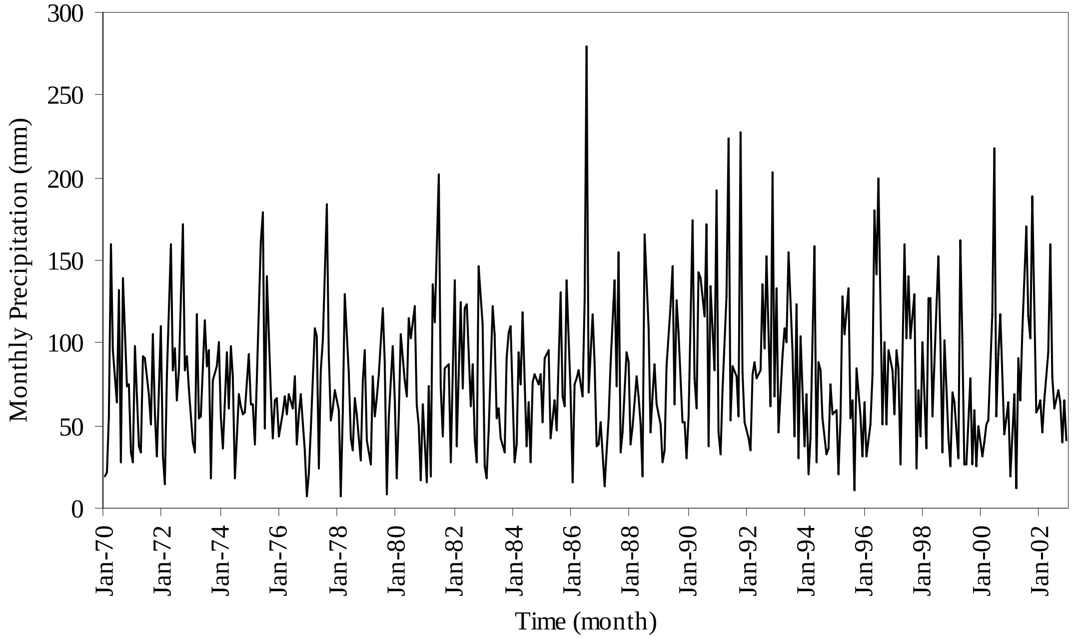


Figure 2.5. Monthly Precipitation Time Series from January 1970 to December 2002, Black Creek Watershed, Indiana.

Daily precipitation was obtained from the Fort Wayne disposal plant station (Station ID: 123037) monitored by Purdue University Applied Meteorology Group for 1902 to 1973 and 1978 to 2002. This station is located at 41°06'N / 85°07'W (LAT/LONG) which is approximately 32 kilometers southwest of the outlet of the Black Creek watershed. The elevation of Fort Wayne disposal plant station is 240 m above the sea level. The daily precipitation and temperature data for this station from 1900 to 2003 are available at: *http://shadow.agry.purdue.edu/sc.index.html*. Information on daily temperatures was obtained from the Fort Wayne station. Figure 2.5 depicts monthly precipitation time series for the 1970-2002 period at the outlet of the Black Creek watershed.

Elevation Map

A 30-m resolution, UTM NAD83 projected Digital Elevation Model (DEM) was obtained from the National Elevation Dataset dated 2001. The DEM for the whole state of Indiana is available at: *http://pasture.ecn.purdue.edu/ABE/Indina*.

Soils

Soil data were obtained from the Soil Survey Geographic Database. Detailed digital representation of County Soil Survey maps was published by the Natural Resources Conservation Service (USDA-NRCS). The Soil Survey Geographical Database (SSURGO) soil map dated 2002 for the whole state of Indiana is available at: *http://pasture.ecn.purdue .edu/ABE/Indina*.

Land Use

Land use map was digitized into ArcView shapefile format from the Black Creek project historical files. The land use maps for 1975, 1976, 1977, and 1978 were extracted from aerial photos dated 1975, 1976, 1977, and 1978, respectively. This information is available at: *http://pasture.ecn.purdue.edu/ABE/blackcreek*.

The information on the best management practices (BMPs) such as type, location, and date of installment were obtained from the Black Creek project technical report (Lake and Morrison, 1977a,b). A BMP shapefile was built to locate the BMPs in the watersheds. Individual subbasins were determined based on the location of the BMPs. The main goal was to locate each BMP in a different subbasin, although in some cases there is more than one BMP in a subbasin. The historical crop rotation in the Black Creek watershed is presented in Table 2.3 and Table 2.4.

Table 2.3. Corn-Soybean Rotation for the Dreisbach and Smith Fry Watersheds in 1975-1978.

year	operation	crop	date	
			month	day
1	tillage		May	3
1	fertilizer		May	6
1	plant/begin. Growing season	CORN	May	10
1	pesticide application	CORN	May	10
1	harvest and kill	CORN	October	15
2	plant/begin. Growing season	SOYB	May	20
2	pesticide application	SOYB	June	15
2	harvest and kill	SOYB	October	1
2	tillage		October	10

Table 2.4. Corn-Soybean-Winter Wheat Rotation for the Dreisbach and Smith Fry Watersheds in 1975-1978.				
year	operation	crop	date	
			month	day
1	tillage		October	12
1	plant/begin. Growing season	WWHT	October	15
2	fertilizer	WWHT	April	5
2	harvest and kill	WWHT	July	15
2	tillage		July	30
3	tillage		May	3
3	fertilizer		May	6
3	plant/begin. Growing season	CORN	May	10
3	pesticide application	CORN	May	10
3	harvest and kill	CORN	October	15
4	plant/begin. Growing season	SOYB	May	20
4	pesticide application	SOYB	June	15
4	harvest and kill	SOYB	October	1
4	tillage		October	10

Flow, Sediment, and Nutrient Data

Streamflow discharge, sediment, and nutrient yields were measured at the two monitoring stations at the outlet of the Dreisbach and Smith Fry watersheds. Complete discussion of all monitoring sites, laboratory methods, and supporting study designs were contained in the Black Creek project technical report (Lake and Morrison, 1977a,b) and the Black Creek project final report (Lake et al., 1981). The measured daily streamflow discharge, sediment, and nutrient yields were reported in the Black Creek project data report (Lake and Morrison, 1978) and the Black Creek project final report (Lake et al., 1981). The available set of measured data include daily streamflow discharge from January 1975 to December 1978, sediment yield from April 1973 to June 1977, and nutrient yields from April 1973 to June 1977. All the above information was converted to tabular form and is available at: *http://pasture.ecn.purdue.edu/ABE/blackcreek/original_data*. The available data compiled for use in SWAT along with their sources are summarized in Table 2.5.

Table 2.5. List of Available Input Data and Their Sources.			
Data Type	Source	Date	Description
Digital Elevation Model (DEM)	National Elevation Data	2001	30-m resolution, U.S. Geological Survey
Soils	Soil Survey Geographic Database	2002	Digital representation of County Soil Survey maps
Land Use	USDA-NRCS	2003	Digitized into GIS from aerial photos
Land Use	Black Creek Project	1975	Digitized into GIS from aerial photos
Weather	Black Creek Project[1]	1974-1977	Daily precipitation graphs
Weather	Purdue Applied Meteorology Group	1902-2002	Minimum and maximum daily temperature and
Crop Management	Engel & Lim (2001)	1975	Management scenarios for crops
Streamflow	Black Creek Project[2]	1975-1978	Daily streamflow
Water Quality	Black Creek Project[1]	1974-May 1977	Daily sediment, mineral P, total P, and total N

[1]Lake and Morrison (1978), [2]Morrison and Lake (1981).

2.4 Base Flow Separation Model

An automated hydrograph separation model "ISEP" was used to determine the relative contribution of surface runoff and ground water to total streamflow. This model was developed in the Department of Agricultural and Biological Engineering, Purdue University.

The "ISEP" program is available at: *http://danpatch.ecn.purdue.edu/~sprawl/iSep*. To further validate the separation model, the determined hydrographs were confirmed with another flow separation model (Arnold and Allen, 1999). The results of the two models were consistent in their determinations of contributions to surface runoff and baseflow parts of the total stream flow. Figures 2.6 and 2.7 show the baseflow volume (mm) estimated by the two flow separation models for the Dreisbach and Smith Fry watersheds, respectively.

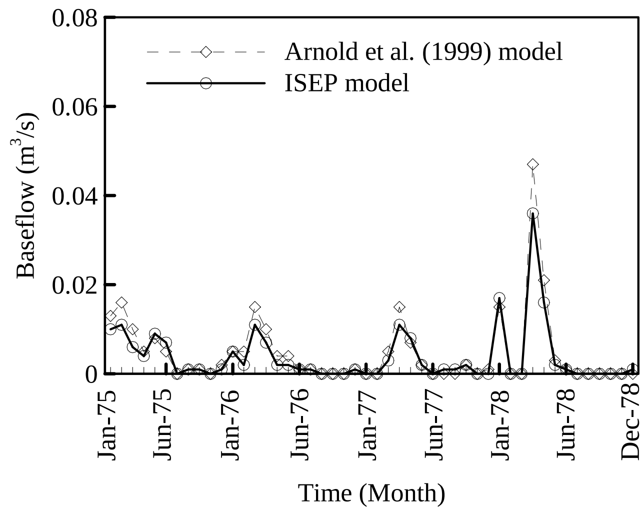


Figure 2.6. Comparison of the Estimated Baseflow Using “ISEP” Model and the Model Adapted from Arnold et al. (1999), Dreisbach Watershed, Indiana.

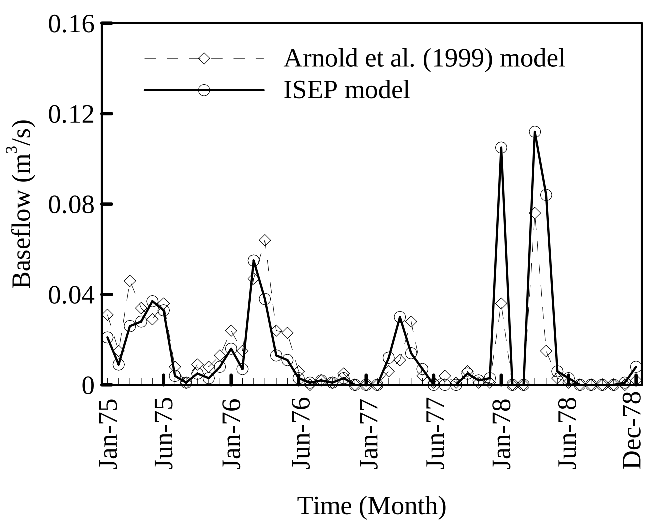


Figure 2.7. Comparison of the Estimated Baseflow Using “ISEP” Model and the Model Adapted from Arnold et al. (1999), Smith Fry Watershed, Indiana.

Section 3.0
Role of Watershed Discretization on SWAT Computations

3.1 Introduction

The ability of a nonpoint source pollution model to simulate design parameters including streamflow, sediment yield, and nutrient loads depends on how well the watershed characteristics are represented by the model inputs. The ability to represent spatial variability inherent in watershed characteristics is the reason that distributed hydrological and water quality models have been favored over the lump models. Distributed models partition a watershed into subunits (subwatersheds, hydrologic response units, or grids) for simulation purposes, and homogeneous properties are assumed for each subunit. As the model inputs are averaged over a subunit, model simulations are greatly influenced by the size and number of the computational units.

The question of spatial resolution can be posed in two ways. First, the spatial resolutions and attributes of input data such as soil series, land use, and Digital Elevation Model (DEM) might significantly influence the model computations. Utilizing finer resolutions of these input data, if available, will result in more accurate simulations although it might be computationally more demanding. Secondly, spatial resolution in the form of watershed discretization is an important consideration in watershed modeling. Currently, watershed delineation and extraction of stream networks are accomplished with GIS databases of Digital Elevation Models (DEMs). The most common method for extracting channel networks requires the *a-priori* specification of a critical source area (CSA) that is required for channel initiation. For the same watershed and Digital Elevation Model (DEM), users may obtain markedly different channel networks, and watershed configurations (i.e. the number and size of subunits). The input parameters are averaged over the computational units. Subsequently the watershed model computations based on the channel network and watershed configuration could be affected as well. This study is an attempt to assess the latter problem - that is given specific soil series, land use, management scenarios, and Digital Elevation Model (DEM) how are model outputs affected by watershed discretization?

Effect of watershed discretization on model outputs has been the motivation of several studies in the past. Norris and Haan (1993) demonstrated that increasing the number of subwatersheds beyond a certain threshold level did not improve runoff generation significantly. Other studies established a threshold value for critical source area for channel initiation (Goodrich, 1992; Zhang and Montgomery, 1994). Miller et al. (1999) concluded that the hydrologic response of small watersheds was more sensitive to changes in topography within the subwatersheds. Kalin et al. (2003) studied the effect of catchment scale on runoff generation and sediment yield over small watersheds. They concluded that a critical source area could be identified for particular combinations of rainfall events and watershed characteristics.

Bingner et al. (1997) utilized the SWAT model to evaluate the impact of the number and size of subwatersheds on runoff generation and fine sediment loads. They found simulated runoff to be rather insensitive to the subwatershed scale. They could identify a critical source area for fine sediment yield. In contrast, Mamillapalli (1998) found that the SWAT model runoff simulations tended to be more accurate with finer discretization of the watershed into subwatersheds or by increasing the number of hydrologic response units (HRUs) in the watershed. It was concluded that the model accuracy does not improve beyond a certain level of discretization. Further, land use and soil distributions were found to have a more significant effect on streamflow simulation than topography. The simultaneous impacts of watershed characteristics, channel parameters, and spatial resolution on sediment generation were studied by FitzHugh and McKay (2000). They concluded that due to limited transport capacity of the channel network downstream of the study area, the streamflow and sediment yield simulated by the SWAT model were not sensitive to changes in the number and size of the subwatersheds. Thus, the role of spatial discretization on SWAT outputs is still unclear, with conflicting viewpoints being expressed by researchers. Effect of watershed discretization on some nutrient components of the SWAT model has been addressed by Jha et al (2004). The results indicated that simulated nitrate (NO_3-N) at the outlet of the watershed increased with the number of subwatersheds while mineral phosphorus (MIN P) was unaffected. These authors recommended further research on evaluation of the effect of watershed discretization on nutrient components of the SWAT model.

3.2 Objectives

An optimal scale of geomorphologic resolution needs to be identified such that further refinement in spatial scale does not contribute to a significant improvement in predicting design quantities at the watershed outlet. This optimal geomorphologic resolution, along with the associated drainage density, can then be utilized to determine the appropriate critical source area prior to calibration and validation of the model.

The following questions are posed to address the effects of spatial resolution in the form of watershed discretization on SWAT model simulations:

(i) To investigate how the number and size of subwatersheds impact SWAT simulations of streamflow, sediment yield, and nutrient load.

(ii) To evaluate the possibility of identifying an optimal critical source area for these quantities, and the conditions when it is available.

(iii) To develop a simple process-based index that is solely a function of the watershed discretization level (i.e. does not require any information on soil, land use, and management data, and HRU distribution level) to serve as a surrogate for sediment and nutrient outputs in evaluation of the effect of watershed discretization level on SWAT computations.

Previous studies have only partially addressed these objectives. Specifically, the impact of watershed discretization on nutrient loads from upland areas has not been discussed at all. With the exception of mineral phosphorus and nitrate, the impact on various pools of phosphorus and nitrogen at the outlet has not been addressed either. The conditions when a critical source area can be identified as in objective (ii) have not been studied, while objective (iii) is completely novel.

3.3 Methodology

The SWAT model integrated into the BASINS framework was utilized to evaluate the effect of watershed discretization on SWAT computations. SWAT simulations were performed with various watershed configurations for a 30 years time horizon from 1971 to 2000. The characteristics of some of the watershed configurations that were utilized in this study (see Figures 3.1-3.2) are summarized in Tables 3.1 and 3.2 for the Dreisbach and Smith Fry watersheds, respectively. The tables include information on the applied critical source area and corresponding number of subwatersheds, total number of HRUs, drainage density (the ratio of total channel length over total watershed area), and average subwatershed area.

The SWAT model streamflow simulations are very sensitive to HRU distribution levels for soil and land use areas (Mamillapalli, 1998). These user-specified thresholds control the number of hydrologic response units (HRUs) in the watershed. For example, if a 10% soil area is defined in HRU distribution, only soils that occupy more than 10% of a subwatershed area are considered in HRU distributions. Subsequently, the number of HRUs in the watershed decreases with increasing threshold values. Since the goal of this study was to evaluate only the effect of watershed discretization, and not the effects of spatial resolutions of soil series, land use, and Digital Elevation Model (DEM), a 0% threshold value was assigned for both soil area and land use area in HRU distribution.

Table 3.1. Properties of the Watershed Configurations Used for the Dreisbach Watershed.

Critical Source Area (km^2)	0.03	0.035	0.045	0.06	0.10	0.30	0.40	2.5
Number of Subwatersheds	103	81	59	45	23	13	5	1
Number of HRUs	647	587	502	445	314	231	135	73
Drainage Density (km/km^2)	3.91	3.57	3.19	2.83	2.28	1.55	1.30	0.91
Average Subwatershed Area (km^2)	0.06	0.08	0.11	0.14	0.27	0.48	1.25	6.23

Table 3.2. Properties of the Watershed Configurations Used for the Smith Fry Watershed.

Critical Source Area (km^2)	0.03	0.050	0.060	0.10	0.25	0.40	0.60	2.9
Number of Subwatersheds	89	63	49	33	15	9	5	1
Number of HRUs	676	577	522	429	308	248	198	93
Drainage Density (km/km^2)	4.09	3.28	3.06	2.55	1.89	1.56	1.35	0.65
Average Subwatershed Area (km^2)	0.08	0.12	0.15	0.22	0.49	0.82	1.47	7.30

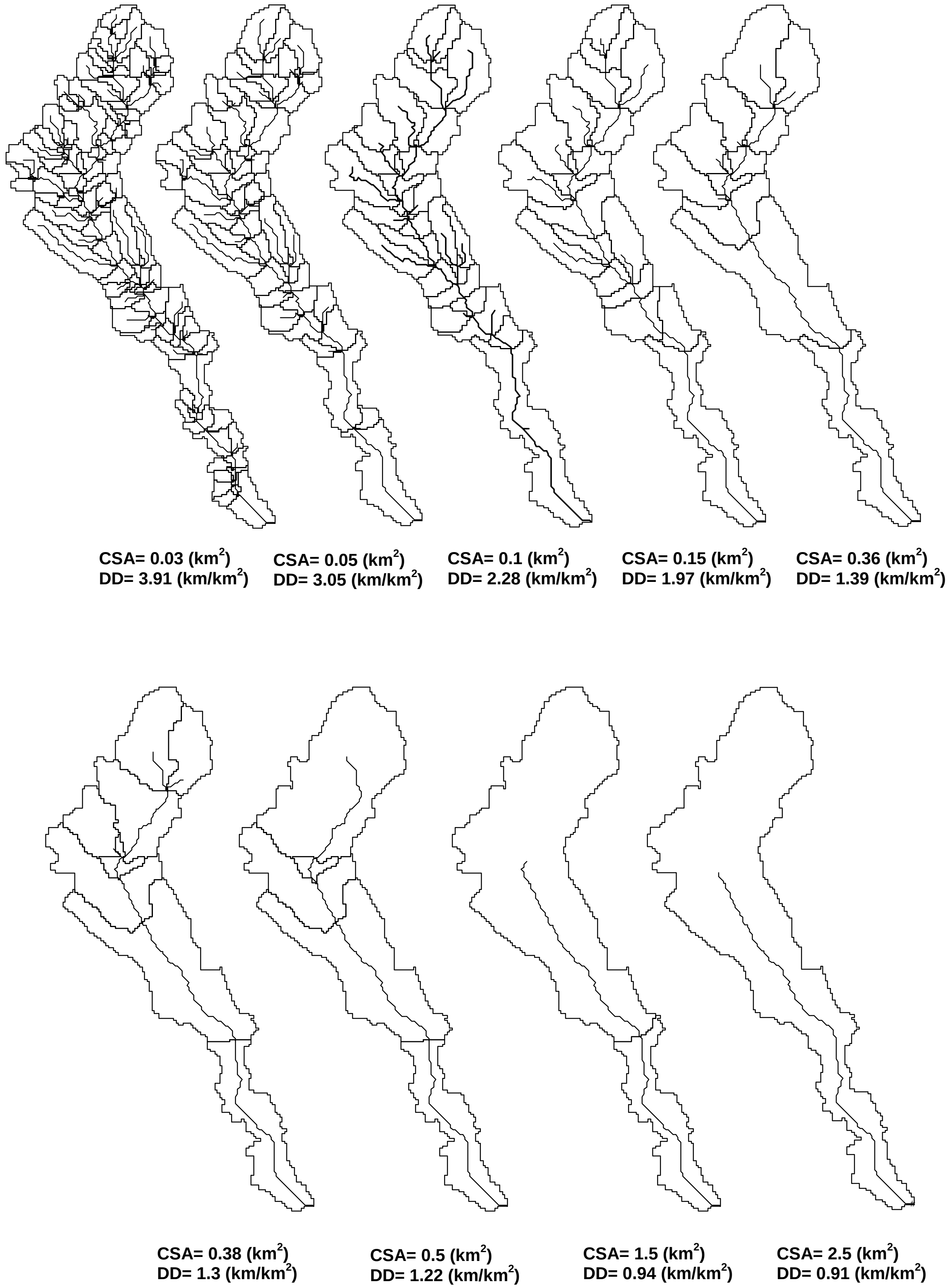


Figure 3.1. Watershed Configurations Used for the Dreisbach Watershed, Indiana.

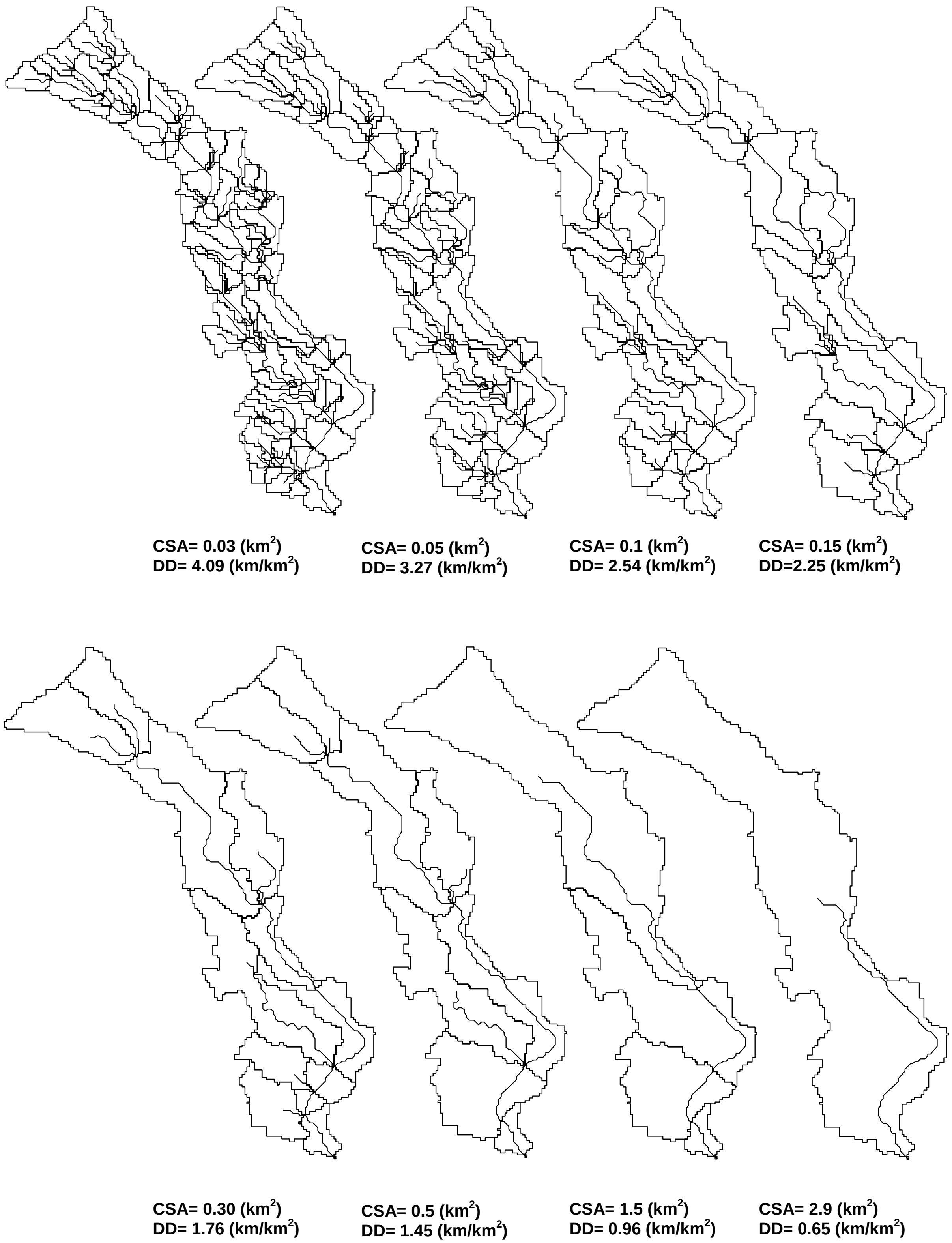


Figure 3.2. Watershed Configurations Used for the Smith Fry Watershed, Indiana.

3.4 Effects of Watershed Discretization on Model Outputs

Spatial resolution in the form of watershed discretization might influence estimation of streamflows, sediment, and nutrient loads generated from upland areas in a different way from the ones computed at the outlet of the watershed. The difference between the two is that the loads generated from upland areas do not include main channel processes such as channel degradation and deposition. Here, the impacts of watershed discretization level on both sediment and nutrient loads from upland areas and the ones at the outlet are discussed.

3.4.1 Streamflow

The effect of watershed discretization on simulated water yield for each HRU can be evaluated by quantifying its effects on surface runoff and transmission losses. SWAT uses the SCS curve number method to compute surface runoff for each HRU. If the HRU distribution levels are set at 0 percent, the overall soil, land use, and management attributes of the HRUs will be the same for various watershed configurations. Therefore, the number and size of subwatersheds will not influence surface runoff computations. Bingner et al. (1997) observed that simulated annual runoff varied by nearly 5% for various watershed configurations. The small variations were perhaps because they applied nonzero threshold levels for soil and land use areas. FitzHugh and Mackay (2000) reported that surface runoff was practically identical for all watershed configurations although a 10 percent threshold level was selected for both soil and land use areas. The results of our study revealed that surface runoff computations were unaffected by the watershed discretization.

At a HRU level, transmission losses (i.e. water lost from ephemeral channels through the bed) are the only mechanism in water yield simulations that may be affected by the watershed discretization. The structure and properties of the ephemeral channels vary with the number and size of subwatersheds that may affect computations of transmission losses. FitzHugh and Mackay (2000) concluded that the 12 percent variation in streamflow simulations at the outlet was due to the impact of watershed discretization on transmission losses. Jha et al. (2004) also came to this conclusion. We observed that transmission losses

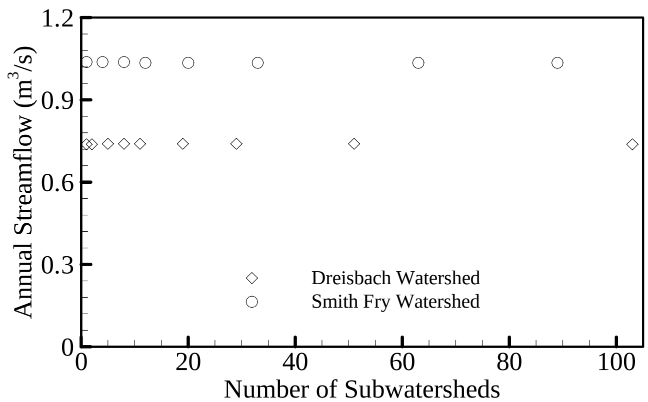


Figure 3.3. Effects of Watershed Discretization on SWAT Streamflow Computations.

simulated by the SWAT model for various watershed configurations were identical for the Dreisbach and Smith Fry watersheds.

The SWAT model employs Manning's equation to estimate flow velocity in a given main channel. The variable storage or the Muskingum channel routing method is applied to route water through the channel network. Flow losses through evaporation and channel losses are the only processes that may result in a difference between water yield from upland areas (i.e. HRUs) and streamflow at the outlet of the watershed. The results of this study indicate that there was no significant difference between the two. This aspect can be explained by considering the size of the watersheds and the fact that the simulations were performed over a 30 year period (1971-2000). The difference between streamflows computed for the coarsest and finest watershed discretization levels was quite small. Figure 3.3 graphically depicts the insensitivity of streamflow simulations to watershed discretization in the Dreisbach and Smith Fry watersheds.

3.4.2 Sediment

An amalgamation of the studies on the impacts of watershed discretization on sheet erosion computations and sediment routing components of SWAT is required for appraisal of the effects on sediment yield at the outlet of the watershed. For this aspect, conflicting results have been reported in previous studies. Bingner et al. (1997) and Jha et al. (2004) only studied the effects of watershed delineation on sediment yield at the outlet without making a distinction between the effects on sediment loads generated at upland areas and the effects on in-stream processes (i.e. channel deposition or degradation). Both studies indicated that sediment yield at the outlet is very sensitive to the number and size of subwatersheds. FitzHugh and Mackay (2000) observed that sediment loads (i.e. sheet erosion) from upland areas decreased with the number of subwatersheds (the graphs presented in the paper show that sediment generation increases with average subwatershed size) while sediment yield at the outlet was almost unaffected by the number and size of subwatersheds.

SWAT model applies the Modified Universal Soil Loss Equation (Equation 2.5) at a HRU level to compute sheet erosion from upland areas. In this study, a constant *P* factor was applied to the whole watershed. *C*, *K*, and *CFRG* parameters were estimated by the BASINS framework for each HRU based on its soil, land use, and management attributes. Since a 0% threshold level was considered for both soil and land use areas in HRU distribution, unlike the number of HRUs, their overall attributes were not affected by variations in the number and size of subwatersheds. Thus, spatial average of *P*, *C*, *K*, and *CFRG* factors were identical for various watershed configurations. The only parameter in Equation 2.5 that was influenced by altering watershed configuration was USLE topographic factor, *LS*. SWAT calculates this parameter for each subwatershed based on its slope and slope length, and applies it to all HRUs located in that particular subwatershed. Figure 3.4 demonstrates the effect of watershed discretization on weighted average *LS* factor. The weighted average of *LS* decreased with the number of subwatersheds in both Dreisbach and Smith fry watersheds. The rate of reduction plateaued once the number of subwatersheds was more than 20. This level of watershed discretization corresponds to a 15 (ha) CSA that is approximately 2% of

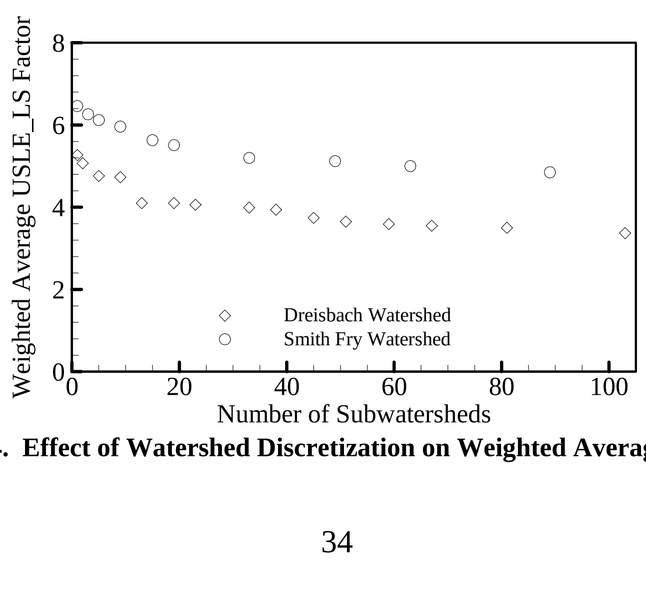


Figure 3.4. Effect of Watershed Discretization on Weighted Average *LS* factor.

Dreisbach and Smith Fry areas. Likewise, sediment loads from upland areas decreased by 28.9% and 22.7% in the Dreisbach and Smith Fry watersheds, respectively, between the coarsest and finest watershed discretization levels (see Figure 3.5).

A comparison of sediment yields at the outlet of the watershed simulated for various watershed configurations would reveal how channel processes are influenced by the CSA. Further, comparing sediment loads from upland areas and sediment yield at the outlet for each watershed discretization would be helpful to identify whether the watershed is "supply-limited" or "transport-limited". Supply-limited refers to watersheds whose transport capacity of the channel network is greater than sheet erosion from upland areas. In this type of watersheds, channel deposition tends to be the overall dominant main channel processes influencing sediment yield at the outlet. The results of this study revealed that sediment loads from upland areas were larger than the sediment yields at the outlet indicating that Dreisbach and Smith Fry are "transport-limited" watersheds. In a transport-limited watershed, sheet erosion from upland areas is the major source of sediments in the watershed. Simulated sediment yields at the outlet of the study watersheds were within 10% of the simulated sediment loads from upland areas. The correlation coefficients between sheet erosion and sediment yield at the outlet of the Dreisbach and Smith Fry watershed were respectively 0.97 and 0.99, indicating that simulated sediment yield at the outlet tended to behave in accordance with sheet erosion from upland areas. The impact of watershed discretization on both sediment loads, i.e. sheet erosion, from upland areas and sediment yield at the outlet of the study watersheds is shown in Figure 3.5. It should be noted that channel deposition is the

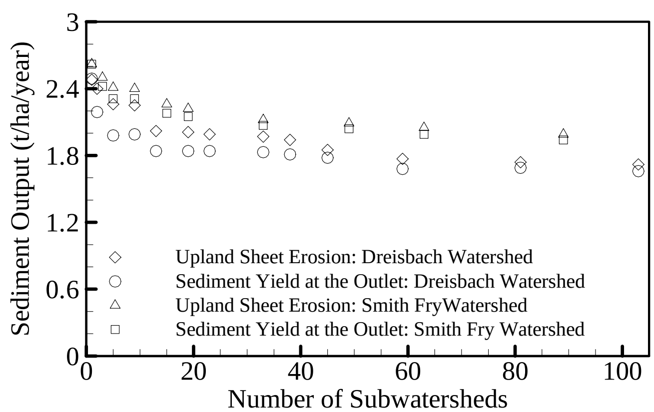


Figure 3.5. Effects of Watershed Discretization on SWAT Sediment Computations.

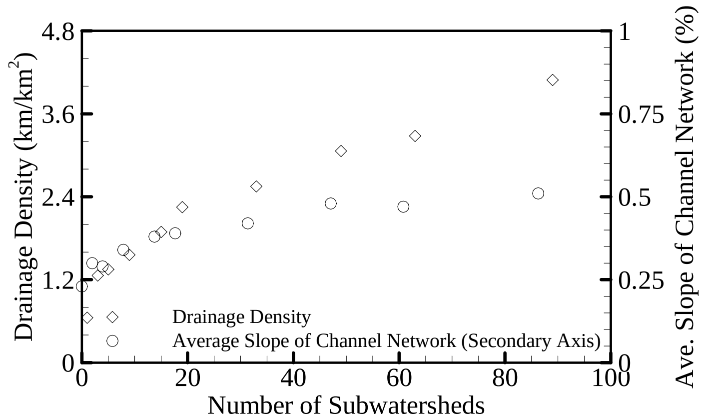


Figure 3.6. Effects of Watershed Discretization on Drainage Density (DD) and Average Slope of Channel Network, Smith Fry Watershed.

overall dominant main channel process in transport-limited watersheds. Dominance of channel deposition indicates that channel erosion does not significantly contribute to sediment yield at the outlet in a transport-limited, and thus sediment yield at the outlet does not increase with drainage density. Although Drainage Density (DD) and average slope of the channel network of the Dreisbach and Smith Fry watersheds increased with finer watershed discretization (Figure 3.6), they did not influence sediment yield at the outlets.

3.4.3 Nutrients

Similar to sediment outputs, the effects of watershed discretization on nutrient outputs of SWAT model were studied by examining the effects on nutrient loads from upland areas and effects on in-stream processes. These relationships have been partly examined by Jha et al. (2004). Here, we evaluate the effects of watershed discretization on total phosphorus (total P) and total nitrogen (total N) loads from upland areas as well as at the outlets of the Dreisbach and Smith Fry watersheds shown in Figures 3.7 and 3.8.

Total P (sum of all phosphorus pools) and total N (sum of all nitrogen pools) loads from upland areas differ by nearly 30 percent between coarsest to finest watershed discretization levels (Figures 3.7 and 3.8). These outputs were highly correlated to sheet erosion from upland areas. A comparison of nutrient loads from upland areas and nutrient yields at the outlet revealed that in-stream processes did not dramatically change the nutrient yields at the outlet of the Dreisbach and Smith fry watersheds. Thus, nutrient yields at the outlet exhibited

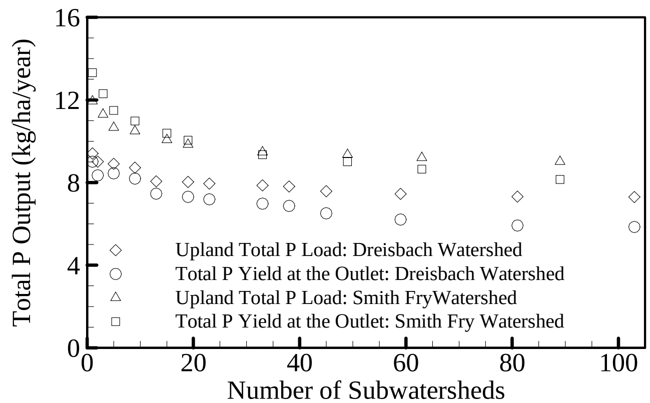


Figure 3.7. Effects of Watershed Discretization on SWAT Total P Computations.

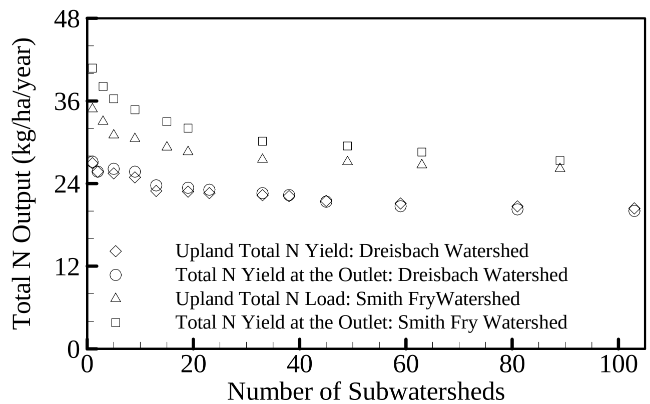


Figure 3.8. Effects of Watershed Discretization on SWAT Total N Computations.

trends similar to nutrient loads from upland areas. Total P and total N yields at the outlet decreased by nearly 40 percent between coarsest to finest watershed discretization levels. The rate of reductions were considerably smaller once the number of subwatersheds was more than 20 corresponding to 2 percent of the Dreisbach and Smith Fry watershed area. It would appear that in-stream processes did not play a significant role in nutrient loads at the outlet of the study watersheds.

3.5 Identification of an Optimal Watershed Discretization Level

A comparison of sediment and nutrient loads from upland areas of the Dreisbach and Smith Fry watersheds for various watershed configurations revealed that 2 percent of the total watershed area could be considered as the optimal critical source area. Furthermore, it was shown that this optimal watershed discretization level could be applied to the sediment yield at the outlet as well. Similar results were reported by Jha et al. (2004). These results along with the ones reported by Bingner et al. (1997), and FitzHugh and Mackay (2000) provide modelers with valuable insight into effects of watershed discretization on SWAT computations. We now examine the nature of averaging that the SWAT model does in order to elucidate the role of sub-grid processes. Results indicated that sheet erosion estimates by SWAT are affected by watershed discretization because USLE topographic factor, *LS*, is averaged over subwatersheds. The effect of averaging the LS_{USLE} over subwatersheds can be assessed by rewriting Eq. (2.5):

$$sed = 11.8 \times \sum_{i=1}^{N} \left\{ \left[\sum_{j=1}^{p} C_{i,j} \times K_{i,j} . P_{i,j} \times CFRG_{i,j} \times f_{i,j}(CN_{i,j}, A_{i,j}) \right] \times LS_i \right\} \tag{3.1}$$

where N is the total number of subwatersheds, p is the total number of HRUs in subwatershed i, A (ha) is total watershed area, $A_{i,j}$ (ha) is the area of HRU j in subwatershed i, $LS_{,i}$ is the USLE topographic factor averaged over subwatershed i, and $C_{i,j}$, $K_{i,j}$, $P_{i,j}$, and $CFRG_{i,j}$ are soil erosion parameters for HRU j in subwatershed i as defined in Eq. (2.5). The quantity $f_{i,j}$ for each HRU is computed as:

$$f_{i,j} = (Q_{i,j} \times q_{i,j} \times A_{i,j})^{0.56} \tag{3.2}$$

In Eq. (3.2), all parameters are defined as in Eq. (2.5). The runoff volume for HRU j in subwatershed i ($Q_{i,j}$) is not affected by watershed discretization as discussed in Section 3.4.1. Rational Method (Equation 2.4) is applied for computation of peak runoff rate for each HRU. The area of HRUs is the only parameter in this equation that varies with the number and size of subwatersheds. Therefore the effect of watershed discretization on parameter $f_{i,j}$ can be sought through the effect on $A_{i,j}^{1.12}$ (i.e. $[A_{i,j} \times A_{i,j}]^{0.56}$). Sheet erosion from upland areas by SWAT can be represented with an Erosion Index (*EI*) defined as:

$$EI = \sum_{i=1}^{N} \left\{ \left[\sum_{j=1}^{p} A_{i,j}^{1.12} \times C_{i,j} \times K_{i,j} \times P_{i,j} \times CFRG_{i,j} \right] \times LS_i \right\} \tag{3.3}$$

The parameter *EI* is essentially a weighted average of USLE topographic factor over the whole watershed that can reasonably represent sediment generated from upland areas for investigation of watershed discretization effects. In a watershed with one land use/management, and a single soil type this weighted average would not depend on the soil and land use attributes and only Digital Elevation Model (DEM) attributes would be important. In that case parameter *EI* can be written as:

$$EI = K \times \sum_{i=1}^{N} \left\{ \left[\sum_{j=1}^{p} A_{i,j}{}^{1.12} \right] \times LS_i \right\} \tag{3.4}$$

where K is a constant ($C_j \times K_j \times P_i \times CFRG_j$). If all HRUs in subwatershed i have the same size, and the number of HRUs in different subwatersheds are the same, *EI* can be rewritten:

$$EI = K \times \frac{1}{p^{1.12}} \times \sum_{i=1}^{N} \left\{ A_i{}^{1.12} \times LS_i \right\} \tag{3.5}$$

More insight into sheet erosion computations would be provided by computing another index, namely the Area Index *(AI)*, defined as:

$$AI = \sum_{i=1}^{N} A_i{}^{1.12} \tag{3.6}$$

Figure 3.9 presents the Area Index for various watershed configurations for both the

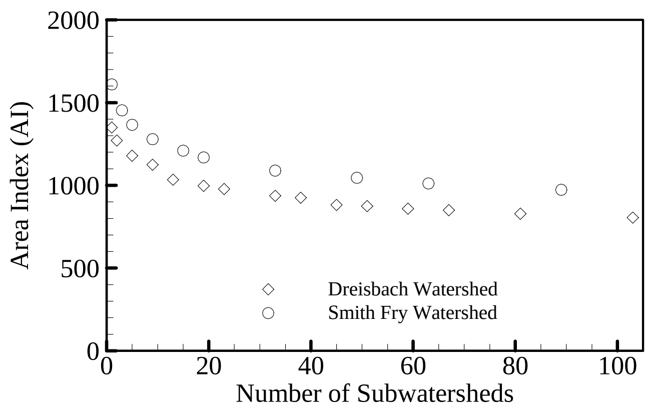


Figure 3.9. Effects of Watershed Discretization on Area Index (*AI*).

watersheds. This index is simple to compute, and does not require any information on soil, land use, and management data. Also, *AI* is computed at a subwatershed level and not a HRU level. Thus, the important HRU distribution levels for soil and land use areas do not affect its computation. *AI* enables users to identify an optimal critical source area for a given watershed utilizing only Digital Elevation Model (DEM) data and is independent of soil, land use, and management attributes.

The limitations of utilizing *AI* for identification of an optimal critical source area arise from the assumptions that were made in arriving at Equations (3.3)-(3.6). As critical source area decreases, subwatershed scale approaches HRU scale. It was assumed that the effect of soil, land use, and management properties could be factored out in the watershed. The validity of this assumption depends on the importance of topographic attributes of the watershed that are represented by a Digital Elevation Model (DEM) versus the importance of soil, land use, and management properties.

The results of this study indicated that the effect of Digital Elevation Model (DEM) attributes of the study area on runoff term of MUSLE equation, parameter *f* (Equation 3.2), dominated the heterogeneity of soil and land use attributes. Thus, *AI* could represent sediment loads from upland areas in identification of an optimal critical source area. In addition, nutrient loads from upland areas and sediment yield at the outlet of the Dreisbach and Smith Fry watersheds were strongly correlated to sediment loads from upland areas for various watershed configurations. If these assumptions do not hold, then the more complicated Erosion Index (Equation 3.3) needs to be used. The high correlation between the Erosion Index (*EI*) and the Area Index (*AI*), depicted in Figure 3.10, indicates that these assumptions were valid for both Dreisbach and Smith Fry watersheds.

The correlation between sediment loads from upland areas and sediment yield at the outlet depends on whether the watershed is transport- or supply-limited. In a transport-limited watershed, upland areas are the major source of sediments. Therefore, application of the Area Index would be adequate for identification of a proper watershed discretization level. In a supply-limited watershed, not only upland areas contribute to sediment yield at the outlet, but channel degradation also serves as a major source of sediment. Channel degradation depends

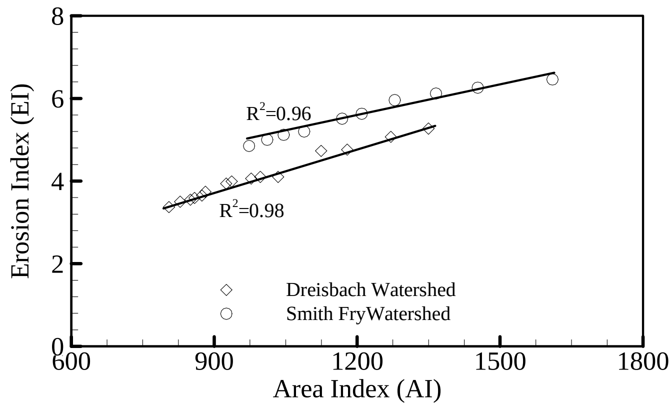


Figure 3.10. Correlation between the Erosion Index (*EI*) and the Area Index (*AI*).

on drainage density and slope of the channel network. Computation of the Area Index does not include the effects of drainage density. Thus, application of *AI* would not be appropriate if the watershed is supply-limited and channel degradation significantly contributes to sediment yield at the outlet.

3.6 Conclusions

The main conclusions of examination of the effect of watershed discretization on un-calibrated model computations for the Dreisbach and Smith Fry watersheds are as follows:

Surface runoff computations of the SWAT model were virtually unaffected by the number and size of subwatersheds. Transmission losses and losses in the main channel were mostly unchanged between the coarsest to finest watershed discretization levels.

Sediment loads from upland areas are affected by watershed discretization. In both Dreisbach and Smith Fry watersheds these loads decreased with the number of subwatersheds. The rate of reduction plateaued once the number of subwatersheds was more than 20 in both watersheds. This watershed discretization level corresponded to a critical source area about 2 percent of total area of the watersheds. Nutrient loads from upland areas were highly correlated to sheet erosion.

Identification of control processes and key management actions within a watershed is essential to obtain an optimal watershed discretization level for SWAT computations at the outlet of the watershed. Substantially different conclusions can be drawn for transport-limited versus supply-limited watersheds. Both Dreisbach and Smith Fry watersheds exhibited the behavior associated with transport-limited watersheds. However, BMPs were not represented in this phase of the study. In-stream processes did not significantly influence nutrient predictions at the outlet of the watersheds. Total P and total N yields at the outlets were highly correlated to the nutrient loads from upland areas of the Dreisbach and Smith Fry watersheds.

Computation of the Area Index (*AI*) appears to be a useful alternative for identification of an appropriate watershed discretization level prior to model calibration. However it is cautioned that application of the Area Index might not be appropriate for supply-limited watersheds. If channel degradation contributes to sediment and nutrient yields at the outlet, a more accurate measure for estimation of optimal drainage density is required. To overcome this limitation, we recommend that the drainage density corresponding to the optimal watershed discretization level be based on the Area Index approach be computed initially. This drainage density could be compared to the channel network defined by USGS 7.5-min quadrangle maps. The watershed discretization level corresponding to the one providing more detailed channel network should then be utilized for modeling purposes.

Section 4.0
Model Calibration and Validation

4.1 Introduction

Application of simulation modeling in research and decision making requires establishing credibility, i.e., "a sufficient degree of belief in the validity of the model" (Rykiel, 1996), for model simulations. The term validity has been defined in so many different ways that no single literature has been able to embrace all of the methods employed to address the issue of validation. However, it is reasonable to agree on the three fundamental attributes of a valid model as described by Beck et al. (1997): (i) soundness of mathematical representation of processes, (ii) sufficient correspondence between model outputs and observations, and (iii) fulfillment of the designated task.

Peer-review is commonly practiced to deal with the first attribute, and is often followed by model calibration. Model calibration is the exercise of adjusting model parameters manually or automatically for the system of interest until model outputs adequately match the observed data. The credibility of model simulations is further evaluated by investigating whether model predictions are satisfactory on different data sets. The semantic of appropriate terminology (validation, verification, corroboration, confirmation, etc.) for this procedure has been disputed, although in practice these terms have been used interchangeably. The bottom line is that all of these terms refer to truth and accuracy of the model (Konikow and Bredehoeft, 1992; Oreskes et al., 1994). Here, the term "validation" will be used with no attempt to clarify the appropriateness of these words.

Although model simulations can be conducted on various temporal and spatial scales, representation of natural processes through the device of a model will always be macroscopic in comparison to reality. Models provide nothing beyond an approximation of reality. A certain degree of confidence in model predictions can be obtained by minimizing the errors associated with such approximation through a calibration procedure. Calibration of a watershed model is essentially the exercise of adjusting model parameters such that model predictions sufficiently match observations. In this section, the calibration and validation of the SWAT model for the study watersheds is discussed.

4.1 Indicators of Model Performance

Various measures including the coefficient of determination R^2 and the coefficient of efficiency $E_{N\text{-}S}$ (Nash and Sutcliffe, 1970) have been utilized to evaluate the accuracy of model predictions (Srinivasan et al., 1998; Eckhardt and Arnold, 2001; Santhi et al., 2001a; Chung et al., 2002). The coefficient of determination is the square of the Pearson's product-moment correlation coefficient. This coefficient describes the proportion of the total variances in the observed data that can be explained by the model, and is defined as:

$$R^2 = \frac{\left[\sum_{i=1}^{N}(O_i - \overline{O})(P_i - \overline{P})\right]^2}{\left[\sum_{i=1}^{N}(O_i - \overline{O})^2\right]\left[\sum_{i=1}^{N}(P_i - \overline{P})^2\right]} \tag{4.1}$$

where O_i and P_i are observed and predicted data points, respectively. $\overline{O}$ is the average of observed data and $\overline{P}$ is the average of predicted values.

R^2 values range from 0 to 1. An R^2 value equal to one is indicative of a perfect correlation between measured data and model predictions. The coefficient of determination is insensitive to additive and proportional differences between the predicted and observed values. On the other hand, R^2 is more sensitive to outliers than to the values near the mean. This oversensitivity leads to a bias toward extreme streamflow values.

To overcome the limitations associated with using the coefficient of determination, the coefficient of efficiency $E_{N\text{-}S}$ has been widely used to evaluate the performance of hydrologic models. The coefficient of efficiency is defined as:

$$E_{N-S} = 1.0 - \frac{\left[\sum_{i=1}^{N}(O_i - P_i)^2\right]}{\left[\sum_{i=1}^{N}(O_i - \overline{O})^2\right]} \qquad (4.2)$$

$E_{N\text{-}S}$ ranges from $-\infty$ to 1, with higher values indicating a better prediction. If $E_{N\text{-}S}$ is negative or very close to zero the model prediction is considered "unacceptable" (Santhi et al., 2001a). The coefficient of efficiency is indicative of how well the plot of observed versus predicted values fit the 1:1 line.

4.2 Sensitivity Analysis

Large complex watershed models contain hundreds of parameters that represent hydrologic and water quality processes in watersheds. Model predictions are more sensitive to perturbation of some input parameters than others, even though the insensitive parameters may bear a larger uncertain range. Thereby, adjustment of all model parameters for a given study area not only is cumbersome, but is not essential. The main objective of sensitivity analysis is to explore the most sensitive parameters to facilitate model calibration procedure.

4.2.1 Sensitivity Index

The SWAT model outputs depend on many input parameters related to the soil, land use, management, weather, channels, aquifer, and reservoirs. Table 4.1 summarizes the 36 SWAT parameters selected out of for sensitivity analysis in this study. These parameters were chosen based on the results of previous studies by Arnold et al. (2000), Eckhardt and Arnold (2001), Santhi et al. (2001a), Vandenberghe (2001), Sohrabi et al. (2003), and Benaman and Shoemaker (2004). Sensitivity of streamflow, sediment, and nutrient outputs of the SWAT model to the selected parameters is sought by perturbing model parameters "one-at-a-time" and determining a linear sensitivity parameter (S_i), defined as (adapted from Gu and Li, 2002):

Table 4.1. List of SWAT Parameters Considered in Sensitivity Analysis						
No.	Parameter	Description	Min	Max	Units	SWAT input file
1	CN2	Initial SCS runoff curve number for moisture condition II	35	98		.MGT
2	SLOPE	Average Slope steepness	0	0.6	m/m	.HRU
3	SLSUBBSN	Average slope length	10	150	m	.HRU
4	ESCO	Soil evaporation compensation factor	0	1		.HRU
5	CH-N1	Manning's "n" value for tributary channels	0.008	30		.SUB
6	CH-S1	Average slope of tributary channels	0	10	m/m	.SUB
7	CH-K1	Effective hydraulic conductivity in tributary channel alluvium	0	150	mm/hr	.SUB
8	CH-N2	Manning's "n" value for the main channel	0.008	0.3		.RTE
9	CH-S2	Average slope of the main channel along the channel length	0	10	m/m	.RTE
10	CH-K2	Effective hydraulic conductivity in main channel alluvium	0	150	mm/hr	.RTE
11	GWQMN	Threshold depth of water in shallow aquifer for return flow to occur	0	5000	mm	.GW
12	ALPHA-BF	Baseflow alpha factor	0	1	days	.GW
13	GW-DELAY	Groundwater delay time	0	500	days	.GW
14	GW-REVAP	Groundwater "revap" time	0.02	0.2		.GW
15	SOL-AWC	Available water capacity of the soil layer	0	1	mm/mm	.SOL
16	CH_EROD	Channel erodibility factor	0	0.6	cm/hr/Pa	.RTE
17	CH_COV	Channel cover factor	0	1		.RTE
18	SPCON	Linear coefficient for calculating maximum sediment re-entrained	0.001	0.01		.BSN
19	SPEXP	Exponent coefficient for calculating maximum sediment re-entrained	1	1.5		.BSN
20	PRF	Peak rate adjustment factor for sediment routing in channel network	0	2		.BSN
21	USLE_P	USLE equation support practice factor	0.1	1		.MGT
22	USLE_C	Maximum value of USLE equation cover factor for water erosion	0.001	0.5		CROP.DAT
23	SOL_LABP	Initial soluble P concentration in soil layer	0	100	mg/kg	.CHM
24	SOL_ORGP	Initial organic P concentration in soil layer	0	4000	mg/kg	.CHM
25	SOL_NO3N	Initial NO_3 concentration in soil layer	0	5	mg/kg	.CHM
26	SOL_ORGN	Initial organic N concentration in soil layer	0	10000	mg/kg	.CHM
27	RS1	Local algae settling rate at 20^{oc}	0	2	m/day	.SWQ
28	RS2	Benthic (sediment) source rate for dissolved P in the reach at 20^{oc}	0.001	0.1	mg/m^2.day	.SWQ

Table 4.1. (Continued)						
29	RS4	Rate coefficient for organic N settling in the reach at 20^{oc}	0.001	0.1	1/day	.SWQ
30	RS5	Organic P settling rate in the reach at 20^{oc}	0.001	0.1	1/day	.SWQ
31	BC4	Rate constant for mineralization of P to dissolved P in the reach at 20^{oc}	0	1	1/day	.SWQ
32	AI0	Ratio of chlorophyll-a to algae biomass	0.001	0.01	μg/mg	.WWQ
33	AI1	Fraction of algal biomass that is nitrogen	0.07	0.09	mg N/mg	.WWQ
34	AI2	Fraction of algal biomass that is phosphorus	0.01	0.02	mg P/mg	.WWQ
35	RHOQ	Algal respiration rate at 20^{oc}	0.05	0.5	1/day	.WWQ
36	K-P	Michaelis-Menton half-saturation constant for phosphorus	0.001	0.5	mp P/l	.WWQ

$$S_i = \max\left\{ \left| \frac{\dfrac{(O_i^{2+} - O_i^1)}{(O_i^{2+} + O_i^1)/2}}{\dfrac{(P_i^{2+} - P_i^1)}{(P_i^{2+} + P_i^1)/2}} \right|, \left| \frac{\dfrac{(O_i^{2-} - O_i^1)}{(O_i^{2-} + O_i^1)/2}}{\dfrac{(P_i^{2-} - P_i^1)}{(P_i^{2-} + P_i^1)/2}} \right| \right\} \tag{4.3}$$

where $O_i^1$and O_i^2 are model outputs corresponding to perturbation of parameter *i* from P_i^1 to P_i^2, respectively. A "+" sign corresponds to parameter changes in positive direction, i.e., $P_i^1 < P_i^2$, whereas a "-"sign indicates parameter changes in negative direction, i.e., $P_i^1 > P_i^2$. In (4.3), it is assumed that the response of model outputs to parameter perturbation is linear. S_i is essentially a normalized estimate of sensitivity of design variables (streamflow, sediment yield, etc.) to a parameter perturbation, with higher values indicating higher sensitivity.

The sensitivity of various outputs of the SWAT model to the parameters listed in Table 4.1 for the study watersheds is depicted in Figure 4.1(a-d). The indices shown in the figure were calculated by incorporating the results of the sensitivity analysis on both Dreisbach and Smith Fry watersheds:

$$S_i = 0.5 \times S_{i,Dreisbach} + 0.5 \times S_{i,Smith\ Fry} \tag{4.4}$$

where $S_{i,Dreisbach}$ and $S_{i,Smith\ Fry}$ are the sensitivity indices determined for parameter *i* at the outlet of the Dreisbach and smith Fry watersheds, respectively.

4.2.2 Additional analysis

The magnitude of the sensitivity index, S_i (4.3), corresponding to each model parameter is rendered subjective to the initial set of parameters that are used in the analysis. Figure 4.3 illustrates sensitivity of sediment output of SWAT to various input parameters listed in Table 4.1 for two cases. In case one, corresponding to the results shown in Figure 4.1(b), the default value was used for the USLE practice factor, i.e., USLE_P=1. It was observed that in this case the parameters that affect the magnitude of channel degradation such as PRF, CH_COV, and CH_EROD (see Table 4.1 for definitions) did not bear a high sensitivity for sediment outputs. However, when the USLE practice factor was altered to 0.3, i.e.,

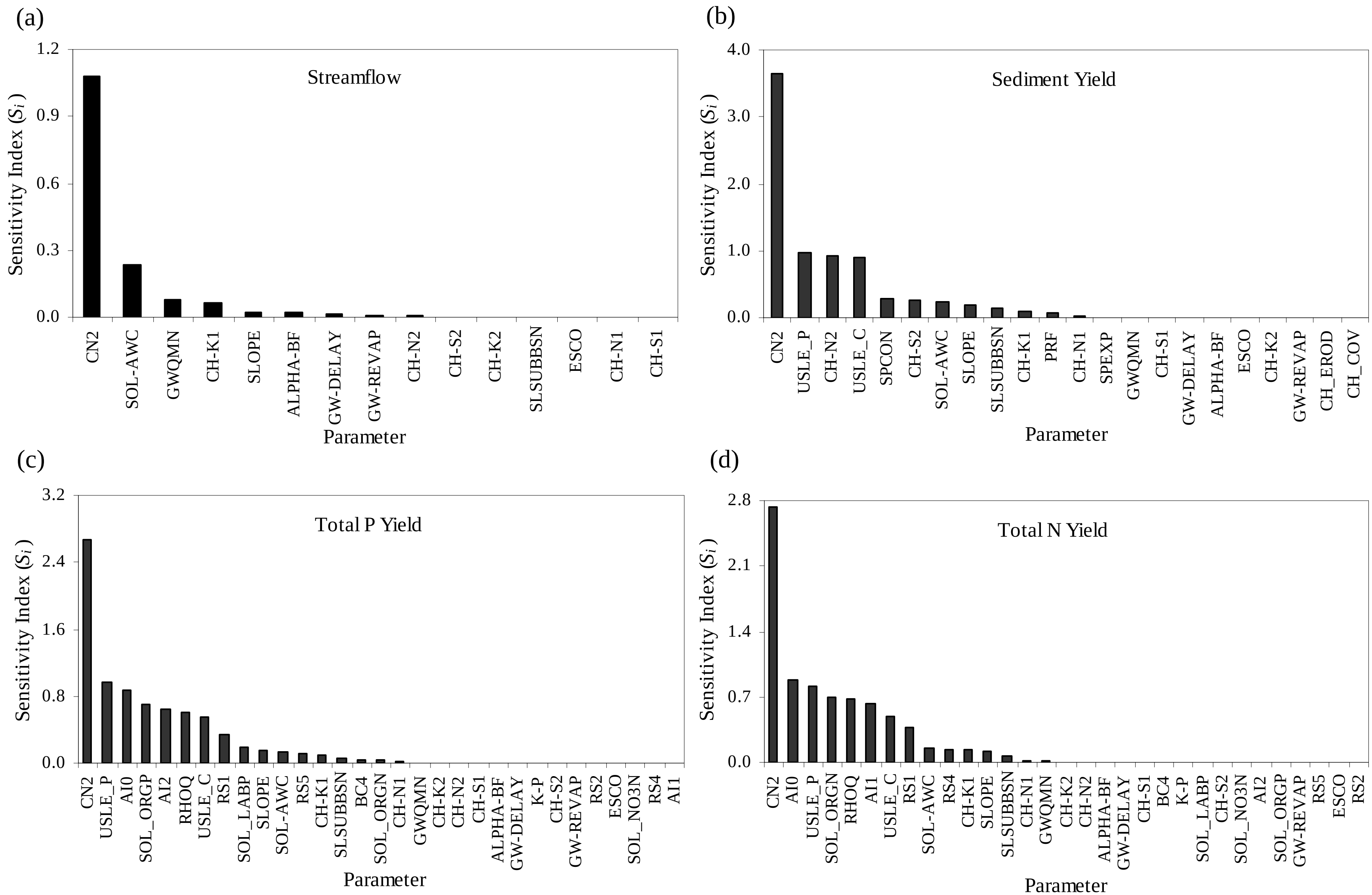


Figure 4.1. Sensitivity of SWAT Parameters Listed in Table 4.1Determined Based on (a) Streamflow, (b) Sediment, (c) Total P, and (d) Total N.

USLE_P=0.3, the parameters corresponding to sediment transport in channel network were among the most sensitive parameters as demonstrated in Figure 4.3. The procedure that is utilized within the SWAT code for representation of sediment transport in the channel network is the primary reason that the sensitivity index (Equation 4.3) for channel sediment parameters varied with USLE practice factor that is utilized for estimation of sheet erosion.

Channel sediment processes within the SWAT code are represented by Equations 2.6-2.10. At each time step, for each channel segment, the initial sediment concentration that depends on both sheet erosion from upland areas and sediment processes (degradation or deposition) in the upstream channel segments is compared to the transport capacity of the channel segment. When a USLE practice factor equal to 1.0 was utilized, initial sediment concentration in the channel network was greater than transport capacity of the channel network and channel deposition was dominant in the channel network. In this case, channel degradation is set to zero by the model and therefore, the sediment output was not sensitive to model parameters that correspond to channel degradation (Figure 4.2, USLE_P=1.0). When USLE practice factor was set at 0.3, sheet erosion from upland areas and subsequently initial sediment concentration in the channel network decreased and sediment degradation was the dominant channel processes. The sensitivity of sediment output to the channel sediment parameters such as CH_N2, CH_S2, CH_EROD, CH_COV, SPCON, SPEXP, and

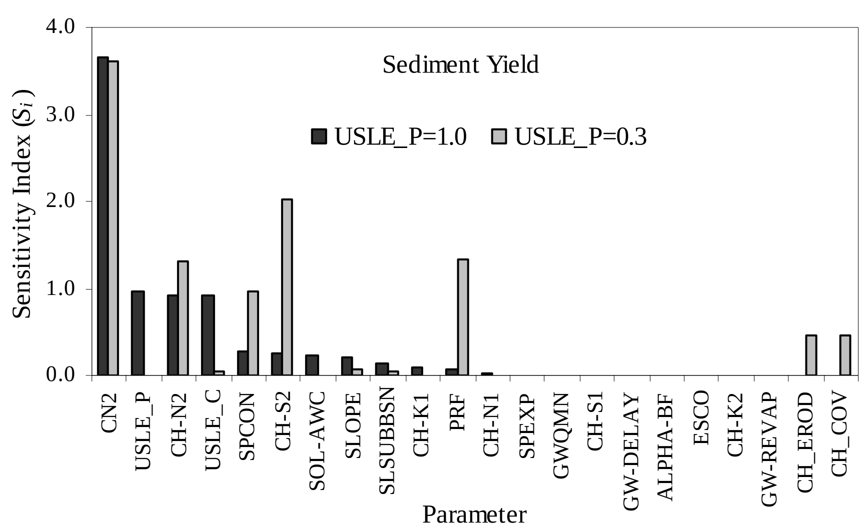


Figure 4.2. Sensitivity of SWAT Parameters Listed in Table 4.1Determined Based on Sediment Yield.

PRF (see Table 4.1 for definition of the parameters) significantly increased as a result as shown in Figure 4.2 for USLE_P=0.3.

4.2.3 Limitations

In computing the sensitivity index (S_i) (Eq. 4.3), the underlying assumption that the response of the model to parameter perturbation is linear may not hold for all of model parameters. For example, Figure 4.3 shows the response of streamflow computations of the SWAT model to GWQMN (threshold depth of water in shallow aquifer for return flow to occur) parameter at the outlet of the Dreisbach watershed. Streamflow output of the model is very sensitive to the parameter changes in the range of 0-500 (mm), whereas changes beyond 1000 (mm) do not result in any appreciable variation in model output.

Moreover, correlations between model parameters that should be elicited and encoded in a comprehensive sensitivity analysis are neglected in (4.3). A change in one parameter would result in a subsequent change in the correlated parameter. The combined changes perhaps results in a different response in the design variable.

4.2.3 Conclusions

A linear sensitivity index was applied to the Dreisbach and Smith Fry watersheds in Indiana to determine the most sensitive SWAT parameters for calibration purposes. The most sensitive parameters identified for various design variables are listed in Table 4.2. It should

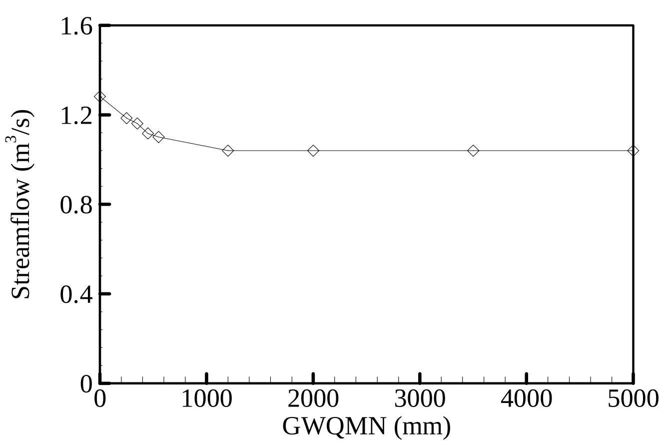


Figure 4.3. Sensitivity of Streamflow Output of the SWAT Model at the Outlet of Dreisbach Watershed to GWOMN Parameter.

Table 4.2. Parameters Identified as Being Important from Sensitivity Analysis for Calibration.				
Design Variable	Streamflow	Sediment	Total P	Total N
Parameter	CN2	CN2	CN2	CN2
	SOL-AWC	USLE_P	USLE_P	AI0
	GWQMN	CH-S2	AI0	USLE_P
	CH-K1	USLE_C	SOL_ORGP	SOL_ORGN
	SLOPE	CH-N2	AI2	RHOQ
	ALPHA-BF	CH_EROD	RHOQ	AI1
	GW-DELAY	CH_COV	USLE_C	USLE_C
	GW-REVAP	PRF	RS1	RS1
	CH-N2	SPCON	SOL_LABP	SOL-AWC
	CH-S2	SOL_AWC	RS5	RS4

be noted that these results are location- and size-dependent and may vary for watersheds with different characteristics.

4.3 Representation of Best Management Practices (BMPs) with SWAT

There were four different types of structural BMPs installed on the Dreisbach and Smith fry watersheds, namely grassed waterways, field borders, parallel terraces, and grade stabilization structures. The BMPs were implemented in 1974 and 1975 in the Dreisbach and Smith Fry Watersheds, respectively. Figure 2.2 depicts the location of these BMPs in the watersheds. SWAT has previously been used to model the impact of some structural BMPs in good condition. Vache et al. (2002) simulated riparian buffers, grassed waterways, filter strips and field borders by modifying the channel cover factor and channel erodibility factor in SWAT to model the cover density and erosion resistant ability of the structures. Santhi et al. (2003) simulated grade stabilization structures in SWAT by modifying the slope and soil erodibility factor and used a program that simulates filter strips based on the filter strip's ability to trap sediment and nutrients based on the strip's width.

For this study, a method was developed to evaluate the ability of grassed waterways, grade stabilization structures, field borders and parallel terraces in SWAT to reduce sediment and nutrients loads from non-gully erosion, based on published literature pertaining to BMP simulation in hydrological models and considering the hydrologic and water quality processes simulated in SWAT. Based on the function of the BMPs and hydrologic and water quality processes that are modified by their implementation, corresponding SWAT parameters were selected and altered as discussed below.

Field Borders

Field borders are strips of vegetation established at the borders of a field where excessive sheet and rill erosion is known to occur. The vegetative cover slows down surface runoff and reduces sheet and rill erosion, and nutrient and pesticide loads in surface runoff. “FILTERW” (width of edge-of-field filter strip) parameter in .hru input file is used in SWAT to calculate the filter strip’s trapping efficiency for sediment, nutrients, and pesticides. The default value for this parameter is zero. The width of the field borders installed in the study watersheds was 5 m. Therefore, FILTERW was modified to 5 m for the HRUs where the field borders have been implemented

Parallel Terraces

Parallel terraces are often used to reduce the peak runoff rate and soil erosion, decrease sediment content of runoff water, and improve water quality. Figure 4.4 illustrates a schematic of a parallel terrace. The horizontal spacing between terraces is determined as (ASAE 2003):

$$H = (X.S + Y)\frac{100}{S} \tag{4.5}$$

where H (SLSUBBSN in Table 4.1) is horizontal spacing between terraces, S (SLOPE in

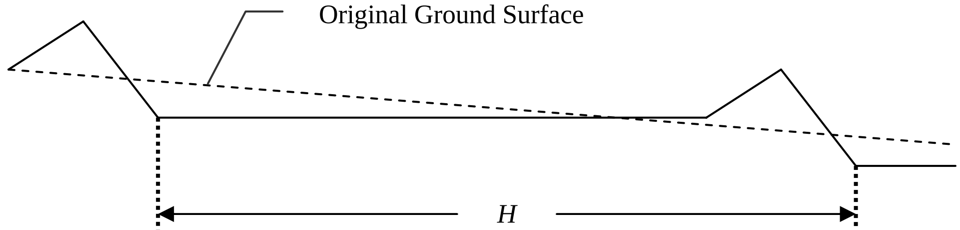


Figure 4.4. Schematic of Parallel Terraces.

Table 4.1) is the weighted average land slope of the land draining into the terrace, Y is a variable with values of 0.3, 0.6, 0.9, or 1.2 influenced by soil erodibility, cropping systems, and crop management practices. X is a variable with values from 0.12 to 0.24. This value for the study area is 0.21 (ASAE 2003). Equation (4.5) with the slope (S) assigned by SWAT based on the Digital Elevation Model (DEM) and Y=0.9 was used to determine the SLSUBBSN parameter for the HRUs with parallel terraces.

Streamflow, sediment, and nutrient computations of the SWAT model are most sensitive to the SCS curve number (CN2 in Table 4.1). CN2 and consequently simulated surface runoff volume (Equation 2.1) decrease significantly for terraced conditions. The CN2 values for the HRUs with parallel terraces were altered to the values for terraced condition obtained from Neitsch et al. (2001a,b).

USLE support practice factor (USLE_P) in (2.5) accounts for the impact of a specific support practice on soil loss from a field. Support practices include contour tillage, strip cropping on the contour, and terrace systems. Figures 4.1(b)-(c) indicate that sediment and nutrient computations of the SWAT model are very sensitive to this parameter. While the default value for USLE_P is unity, this value was altered to 0.2 (Neitsch et al., 2001a,b) for the HRUs with parallel terraces.

Grassed Waterways

Grassed waterways are used to protect a stream from gully erosion, and act as a filter to absorb some of chemicals and nutrients being carried in surface runoff. A natural stream is graded and seeded by grass to form a parabolic shape channel covered by grass. Surface runoff flows down across the grass rather than eroding soils from the channel perimeter. To represent grassed waterways in the SWAT model three parameters-- channel erodibility factor (CH_EROD), channel cover factor (CH_COV), and channel Manning's "*n*" value (CH_N2) -- were modified.

SWAT uses Manning's equation to compute the velocity of flow in the channel segments. Flow velocity decreases with channel Manning's "*n*" value (CH_N2). The sensitivity of sediment computations of SWAT to Manning's number is shown in Figure 4.2. The default value for CH_N2 in SWAT is 0.014. This value was modified to 0.24 for the channel segments with grassed waterways (Chow, 1956). These channel segments were considered fully protected by the vegetative cover (CH_COV=0), and non-erosive (CH_EROD=0).

Grade Stabilization Structures

A dam or an embankment built across a waterway or an existing gully reduces water flow and gully erosion. The height of the grade stabilization structures installed on the Dreisbach and Smith Fry watersheds was 1.2 m. Figure 4.5 shows the schematic of a grade stabilization structure. The new slope (S_{mod}) of channel segments with grade stabilization structures was calculated as:

$$S_{\mathrm{mod}} = S_{org} - \frac{1.2}{L} \tag{4.6}$$

where S_{org} is the original channel slope, and L is the length of the channel segment in meters. The channel segments with grade stabilization structure were also considered non-erosive (CH_EROD=0).

The representation of BMPs discussed above is summarized in Table 4.3. Once the BMPs were represented by fixing the corresponding parameters at the values shown in Table 4.3, the rest of model parameters were calibrated for the study watersheds.

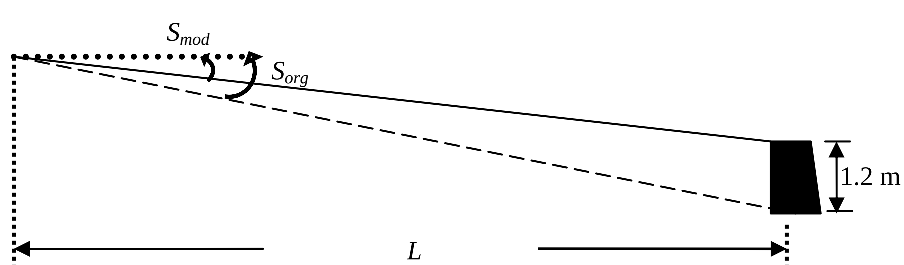


Figure 4.5. Schematic of Grade Stabilization Structures.

Table 4.3. Representation of Field Borders, Parallel Terraces, Grassed Waterways, and Grade Stabilization Structures in SWAT.				
BMP	Function	Representing SWAT Parameter		
		Variable (input file)	Range	Value when BMP implemented
Field Border	Increase sediment trapping	FILTERW (.hru)	0-5 (m)	5 (m)
Parallel Terrace	Reduce overland flow	CN(2) (.mgt)	0-100	*
	Reduce sheet erosion	USLE_P (.mgt)	0-1	0.2 (terraced condition)
	Reduce slope length	SLSUBBSN (.hru)	10-150	From Eq. (4.5)
Grassed Waterway	Increase channel cover	CH_COV (.rch)	0-1	0.0 (completely protected)
	Reduce channel erodibility	CH_EROD (.rch)	0-1	0.0 (non-erosive channel)
	Increase channel roughness	CH_N(2) (.rch)	0-0.3	0.24
Grade Stabilization Structure	Reduce gully erosion	CH_EROD (.rch)	0-1	0.0 (non-erosive channel)
	Reduce slope steepness	CH_S(2) (.rch)	-	From Eq. (4.6)

*Estimated based on land use and hydrologic soil group of the HRU where it is installed for terraced condition.

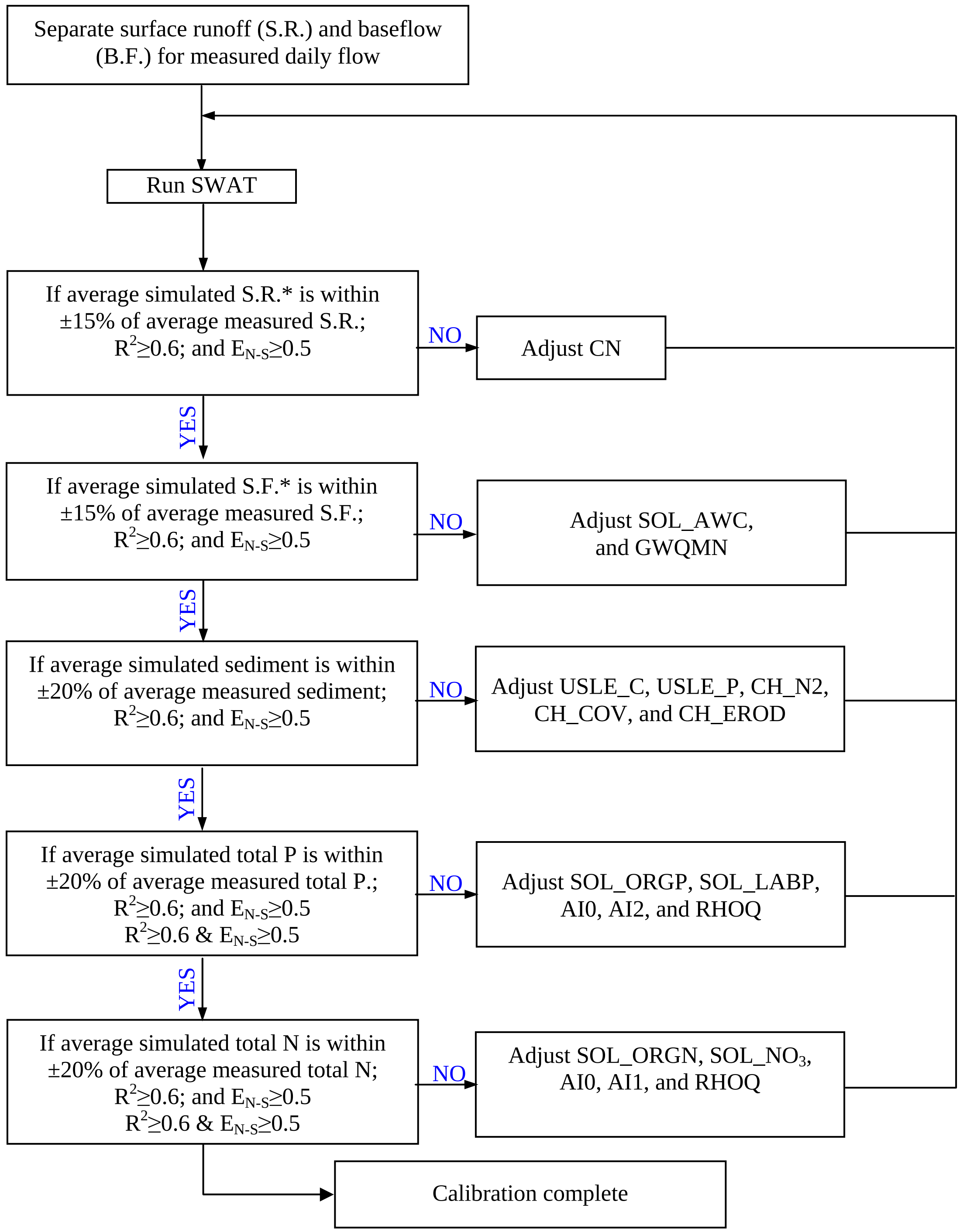
4.4 Model Calibration

The characteristics of a good calibration data set have been subject of much discussion and debate (James and Burges, 1982; Gupta and Sorooshian, 1985; Beck, 1987; Sorooshian and Gupta, 1995). However, there are only general, qualitative guidelines for the selection of the calibration data set. A good calibration data set contains sufficient information to fulfill the goals of the study. Sorooshian et al. (1983) showed that a single year of measured stream flow data could be adequate to calibrate a hydrologic model if it contains the right information. Typically three to five years of data are required in calibration of a hydrologic model.

In this study, hydrologic components of the SWAT model were calibrate and validated on a monthly basis for a time period from January 1975 to December 1978. Average, minimum, and maximum monthly precipitations at the outlet of the Black Creek watershed for this period were 70, 7, and 184 mm, respectively. The calibration and validation period contains the lowest precipitation in 1970-2000 time period (see Figure 2.6). Only 2.5% of monthly precipitations during 1970-2000 period exceeded 184 mm. The average monthly precipitation during 1970-2000 period was 77 mm, slightly larger than 70 mm. The monthly precipitation time series depicted in Figure 2.6 shows that the 1975-1978 time period encompasses adequate information for calibration and validation of SWAT that will be utilized for long-term (1970-2000 period) evaluation of the objectives of the study. The available data for calibration and validation of SWAT for the study watershed are summarized in Table 2.5.

For calibration and validation of the SWAT model, three steps were implemented. First, the optimal watershed discretization level obtained from Section 3.0 was utilized for watershed subdivision and extraction of channel networks of the study watersheds. It was concluded that application of 2 percent of the watershed area as critical source area is sufficient for the Dreisbach and Smith Fry watersheds. Further, HRU distribution levels for soil and land use areas were set at 0%. These user-specified thresholds control the number of HRUs in the watershed. For example, if a 10% soil area is defined in HRU distribution, only soils that occupy more than 10% of a subwatershed area are considered in HRU distributions. Moreover, parameters of the HRUs and channel segments where the BMPs have been installed were accordingly set to the values specified in Table 4.3 and were not altered during calibration. The rest of model parameters were calibrated for streamflow, sediment, and nutrient yields.

For flow calibration, the measured daily stream flow series from January 1975 to December 1978 was split into two sets. The first set of streamflows from January 1975 to June 1977 (30 months) was utilized for calibration. The rest of the time series containing 18 months of measured streamflow was used for validation of the model. Sediment and nutrient components of SWAT were calibrated for the time period from January 1974 to December 1975, and validated from January 1975 to May 1977. Both calibration and validation procedures were performed on a monthly basis. A flowchart describing the procedure for calibration of the SWAT model is shown in Figure 4.6 (adapted from Santhi et al., 2001a).



*S.R.: surface runoff, S.F.: streamflow, and B.F.: baseflow.

Figure 4.6. Calibration Flowchart (Adapted from Santhi et al., 2001a).

Initially, baseflow was separated from surface runoff using the "ISEP" hydrograph separation model. Surface runoff was calibrated until the average monthly simulated surface runoff was within ±15% of average observed surface runoff, $R^2 \geq 0.6$, and $E_{N-S} \geq 0.5$ for the calibration period. The same criteria were used for the total streamflow. Sediment and nutrient yields were calibrated until the average simulated quantities were within ±20% of average observed ones, $R^2 \geq 0.6$, and $E_{N-S} \geq 0.5$. The results of the calibration procedure are summarized in Table 4.4.

Once calibration of the model was completed, validation was performed to evaluate the performance of the model for a data set different from the one used for calibration. The optimal parameter values obtained from model calibration were used in model validation. Predicted and observed data were compared using coefficient of efficiency (E_{N-S}) and coefficient of determination (R^2) to test the validity of the model. The summary results of model validation are summarized in Table 4.5.

Satisfactory model calibration and validation results were obtained for both watersheds (Tables 4.4 and 4.5). In general, the calibrated model was able to adequately predict both low and high streamflow, sediment, and nutrient yields in both watersheds. However, streamflows for March 1978 were underpredicted and a low coefficient of efficiency was obtained for total P in the Smith Fry watershed in the validation period. While the model slightly overpredicts mineral and total phosphorus yields at the outlets for the months with low phosphorus yield, the high yield months were underpredicted.

Table 4.4. Results of Calibration of SWAT for Streamflow, Sediment and Nutrient Simulations.

Variable[1]	Dreisbach				Smith Fry			
	Obs[2]	Sim[3]	R^2	E_{N-S}	Obs[2]	Sim[3]	R^2	E_{N-S}
Streamflow (m^3/s)	0.039	0.04	0.92	0.84	0.054	0.052	0.86	0.73
Surface Runoff (m^3/s)	0.035	0.037	0.91	0.80	0.045	0.049	0.84	0.62
Suspended Solids (t/ha)	0.027	0.024	0.97	0.92	0.151	0.16	0.94	0.86
Mineral P (kg/ha)	0.070	0.070	0.92	0.84	0.46	0.55	0.92	0.73
Total P (kg/ha)	0.077	0.094	0.93	0.78	0.587	0.708	0.91	0.82
Total N (kg/ha)	1.35	1.53	0.76	0.54	8.81	7.29	0.82	0.64

[1] Monthly simulations, [2] Observed; [3] Simulated.

Table 4.5. Results of Validation of SWAT for Streamflow, Sediment and Nutrient Simulations.								
Variable[1]	Dreisbach				Smith Fry			
	Obs[2]	Sim[3]	R^2	$E_{N\text{-}S}$	Obs[2]	Sim[3]	R^2	$E_{N\text{-}S}$
Streamflow (m^3/s)	0.042	0.047	0.87	0.73	0.053	0.069	0.81	0.63
Surface Runoff (m^3/s)	0.038	0.045	0.88	0.75	0.051	0.065	0.84	0.63
Suspended Solids (t/ha)	0.032	0.033	0.86	0.75	0.052	0.073	0.85	0.68
Mineral P (kg/ha)	0.067	0.067	0.86	0.74	0.139	0.133	0.73	0.51
Total P (kg/ha)	0.074	0.09	0.90	0.79	0.241	0.159	0.73	0.37
Total N (kg/ha)	1.227	1.20	0.75	0.52	2.59	2.45	0.85	0.72

[1] Monthly simulations, [2] Observed; [3] Simulated.

The observed and simulated monthly surface runoff, streamflow, sediment, mineral phosphorus, total phosphorus, and total nitrogen for the calibration and validation period at the outlet of the Dreisbach and Smith Fry watersheds are shown in Figures 4.7 to 4.10. Based on thee results, it was assumed that the SWAT model was calibrated and validated for the study watersheds.

4.6 Discussion

A total of 26 different BMPs were implemented in the Dreisbach watershed while only 6 were implemented in the Smith Fry watershed (see Figure 1). After application of the same method to represent the BMPs in the watersheds, the same set of calibrated parameters was obtained for each of the Dreisbach and Smith Fry watersheds, except for USLE practice factor (USLE_P). This provided further confirmation for the calibration procedure and the method that was utilized to represent the BMPs. The reason for different optimal (calibrated) USLE_P parameter is that a major portion of the Dreisbach watershed is cultivated by a community that practices a more traditional method for farming.

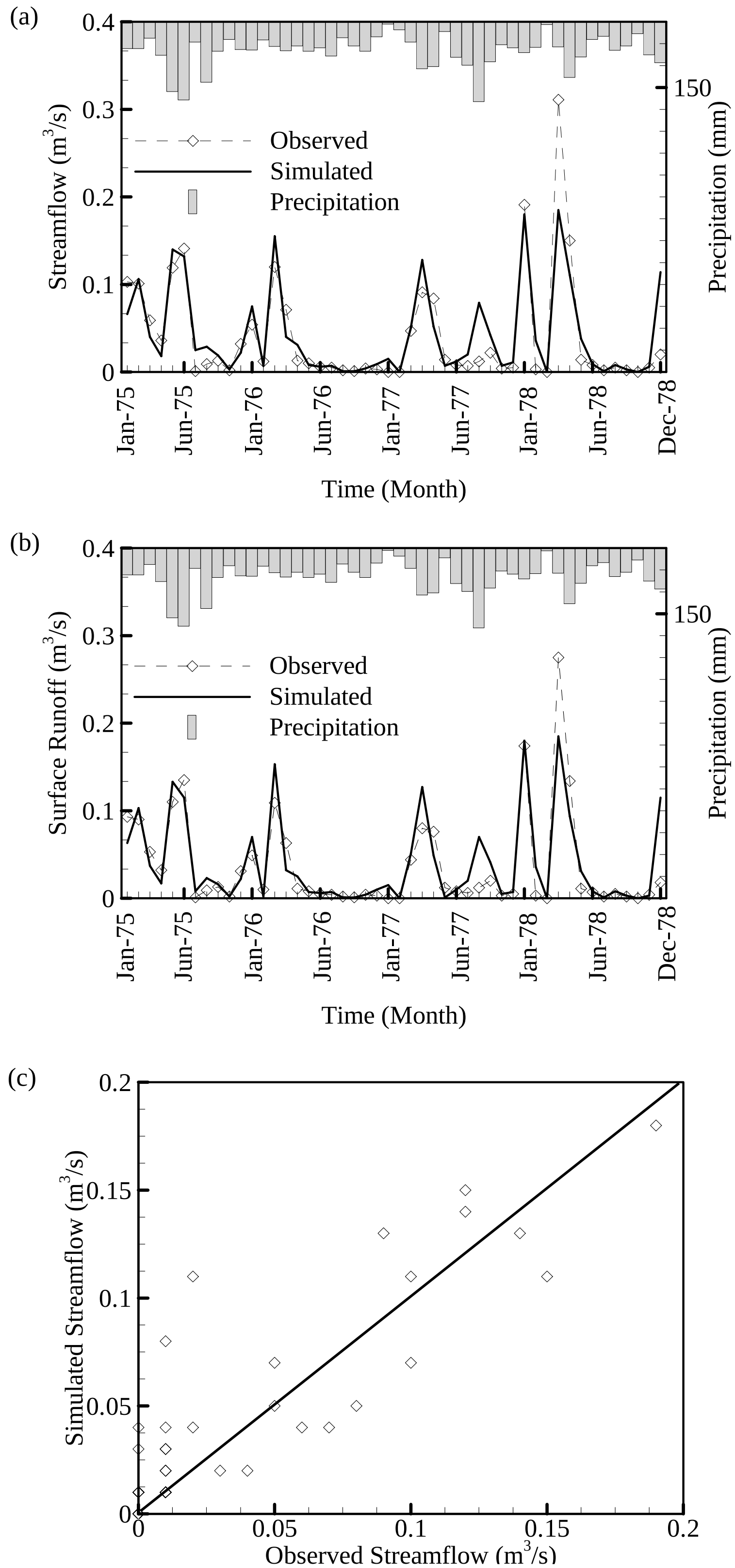


Figure 4.7. Measured and Simulated (a) Streamflow, (b) Surface Runoff, and (c) Plot 1:1 Streamflow, Calibration and Validation Period, Dreisbach Watershed, Indiana.

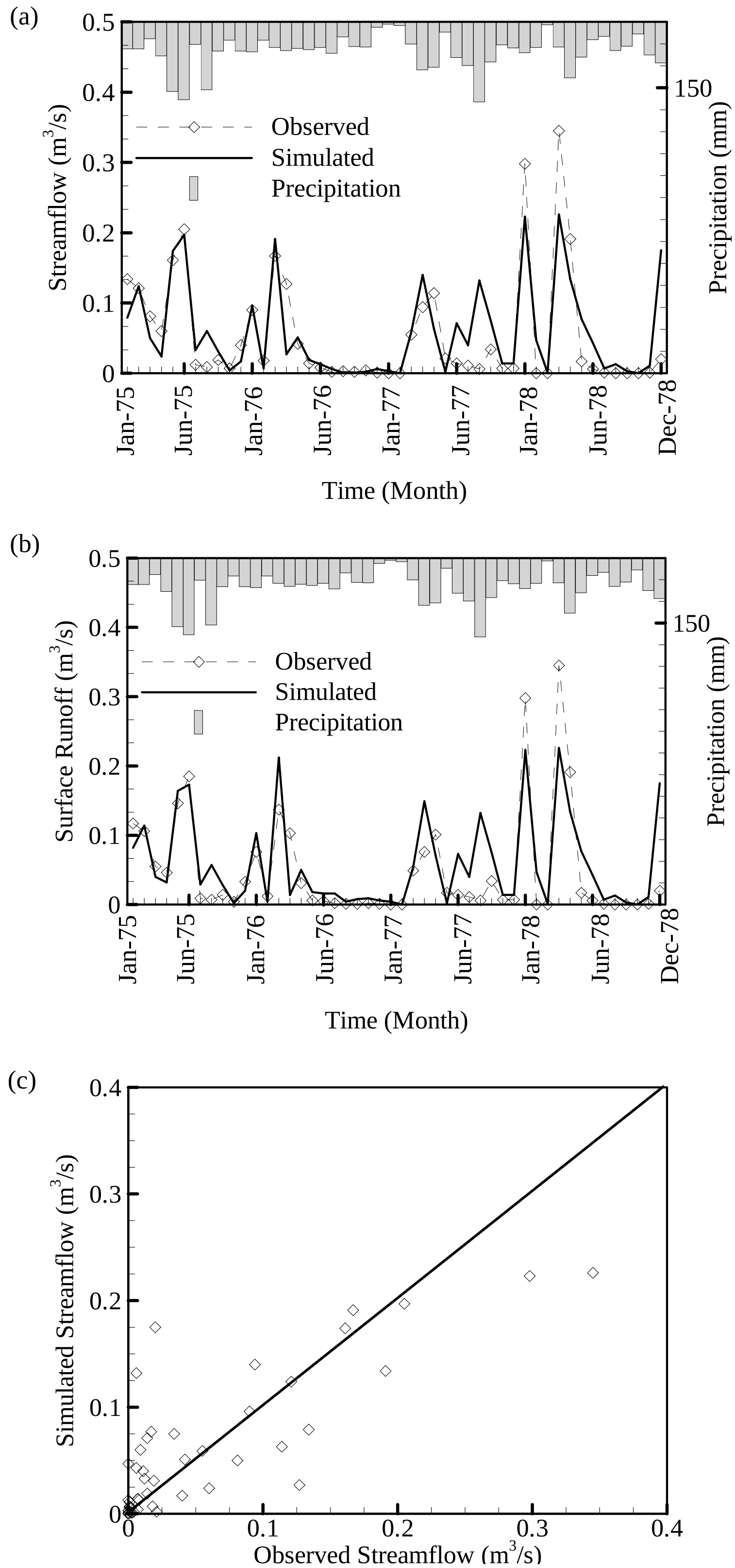


Figure 4.8. Measured and Simulated (a) Streamflow, (b) Surface Runoff, and (c) Plot 1:1 Streamflow, Calibration and Validation Period, Smith Fry Watershed, Indiana.

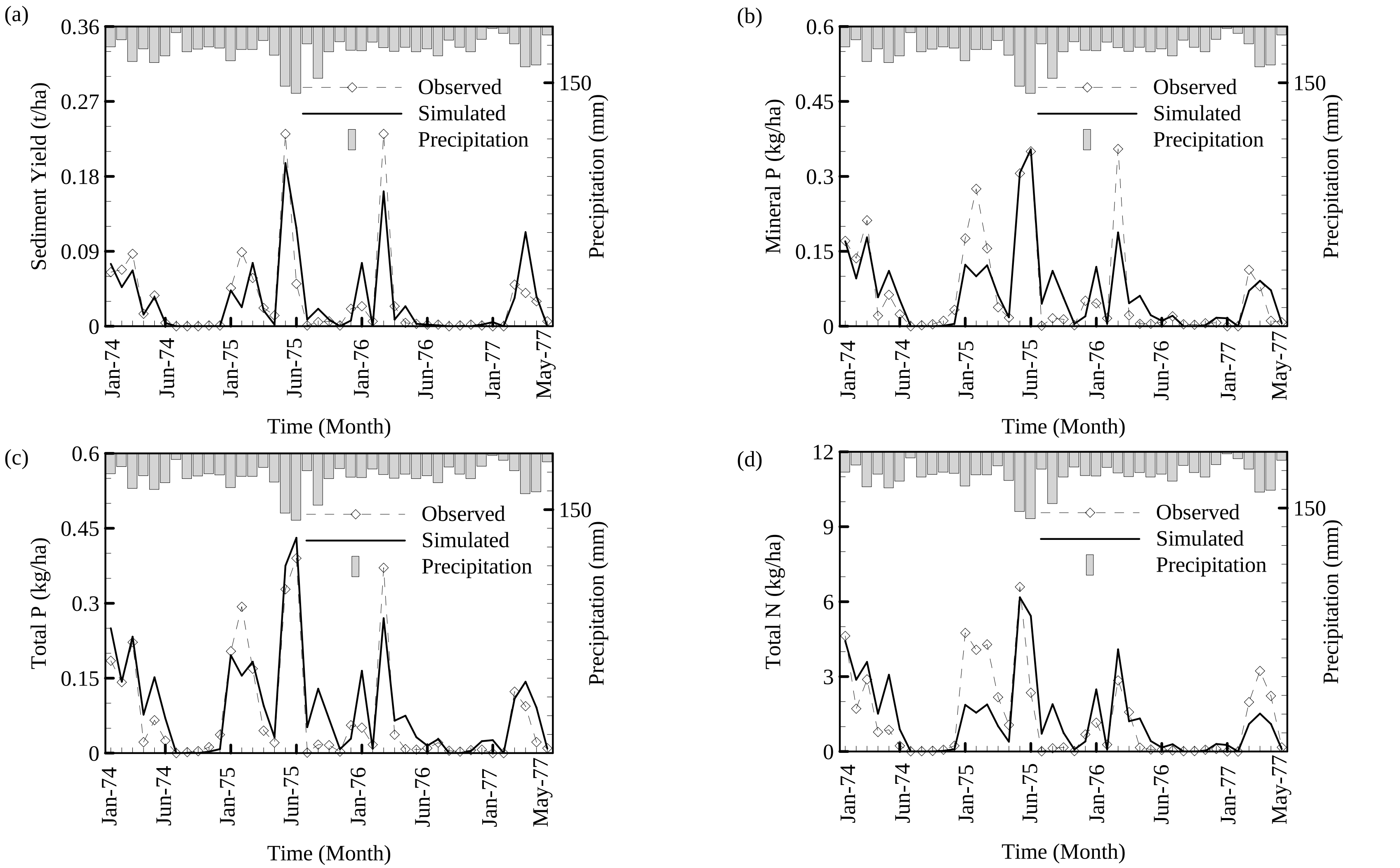


Figure 4.9. Measured and Simulated (a) Sediment, (b) Mineral P, and (c) Total P, (d) Total N, Calibration and Validation Period, Dreisbach Watershed, Indiana.

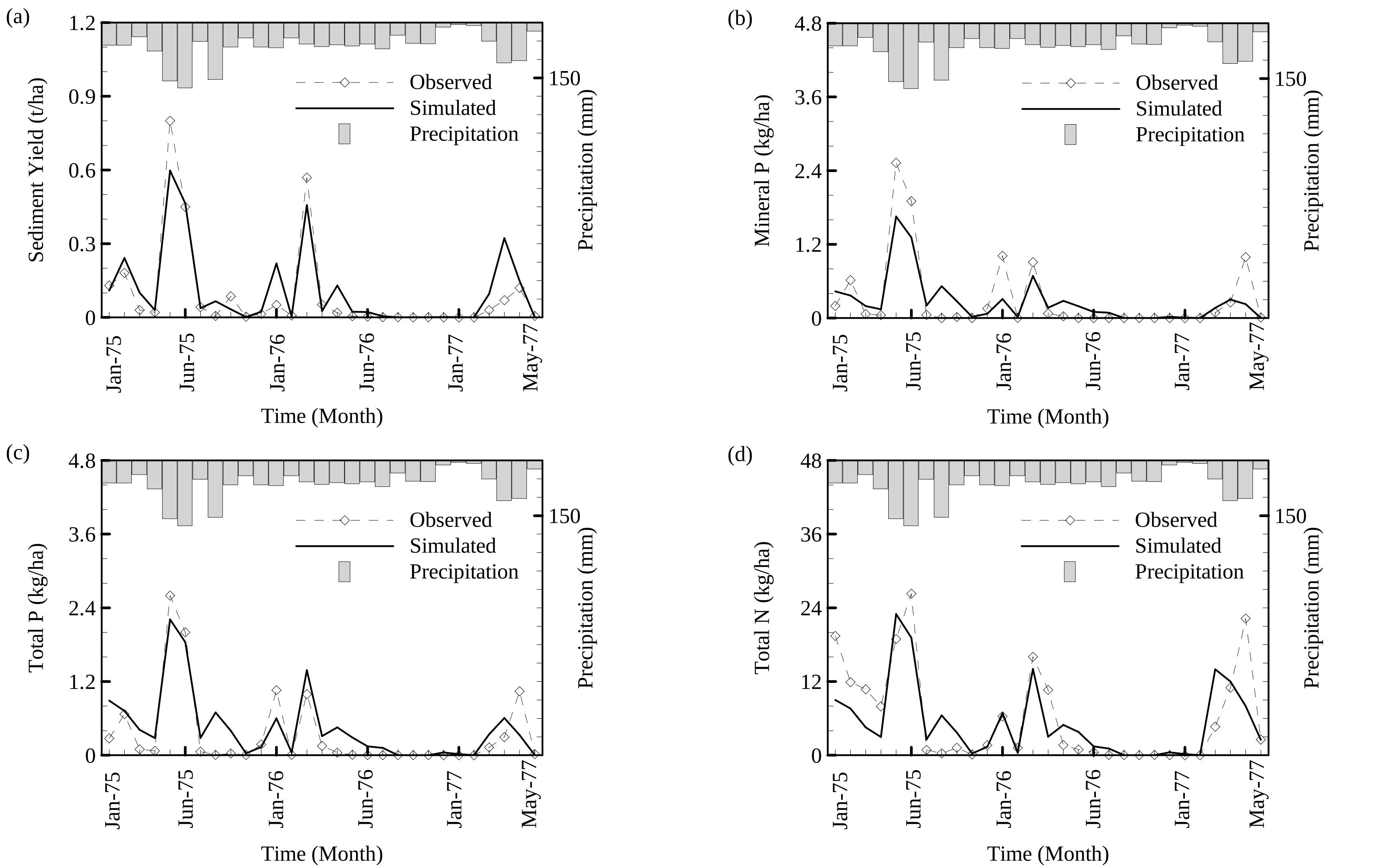


Figure 4.10. Measured and Simulated (a) Sediment, (b) Mineral P, and (c) Total P, (d) Total N, Calibration and Validation Period, Smith Fry Watershed, Indiana.

Section 5.0
Evaluation of Long-Term Impact of Best Management Practices on Water Quality with a Watershed Model: Role of Spatial Resolution

5.1 Introduction

Implementation of Best Management Practices (BMPs) is a conventional approach for controlling nonpoint sources of sediments and nutrients. However, implementation of BMPs is rarely followed by a good long-term data monitoring program in place to study how effective they have been in meeting their original goals. Long-term data on flow and water quality within watersheds, before and after placement of BMPs, is not generally available. Therefore, evaluation of BMPs (especially new ones that have had little or no history of use) must be necessarily conducted through watershed models. In this regard, various watershed and field scale models have been used to asses the effectiveness of BMPs (Moore et al., 1992; Batchelor et al., 1994; Park et al., 1994; Griffin, 1995; Edwards et al., 1996; Mostaghimi et al., 1997). A number of studies have been performed with the Soil and Water Assessment Tool (SWAT) model to study the effects of different BMPs on sediment and nutrient transport within watersheds (Saleh et al., 2000; Santhi et al., 2001b; Kirsch et al., 2002; Saleh and Du, 2002; Vache et al. 2002; Santhi et al., 2003).

Distributed models partition the watershed into smaller units (subwatersheds/hyrologic response units, or grids) to represent heterogeneity within the watershed. Delineation of the watershed, identification of the stream network, and partitioning of the study area into smaller units is generally accomplished through Geographic Information System (GIS) databases that help automate this process and make it convenient for modeling purposes. However, division into subwatersheds and identification of stream networks are extremely sensitive to spatial scale. The number and size of computational units varies with a user-defined critical source area (CSA), the minimum area required for channel initiation. Results of Section 3.0 indicate that the SWAT model sediment and nutrient simulations vary quite dramatically with the number and size of subwatersheds. Because model outputs are affected by geomorphologic resolution, the predicted performance of BMPs will be influenced as well. Thus, examination of the efficacy of BMPs must be conducted in conjunction with studies performed at multiple spatial scales. Previous research on evaluation of the effectiveness of BMPs has not incorporated the effects of geomorphologic resolution.

In this section, the long-term water quality impact of BMPs is analyzed through the device of a watershed model. The analysis is conducted in conjunction with investigating the role of spatial resolution effects resulting from watershed discretization.

5.2 Methodology

Calibration of hydrologic and water quality components of the SWAT model for the Dreisbach and Smith Fry watersheds was discussed in Section 4.0. Calibrated model simulations were performed for a 30 year period (1971-2000) for two scenarios (scenarios A and B). Scenario A corresponded to model results without BMPs, while scenario B simulated the design variables (sediment and nutrient yields) with BMPs in place. Scenarios A and B were compared at various watershed discretization levels in order to determine the efficiency of the BMPs at each watershed discretization level. All of the input parameters for the two scenarios were exactly the same over the study watersheds with the exception of the parameters of the hydrologic response units (HRUs) with parallel terraces and field borders, and the parameters of the channel segments with grassed waterways and stabilization structures. In scenario A, these parameters were assumed to be the same as the rest of the study area for which calibrated values are available. The values specified for different BMPs in Table 4.3 were utilized for these parameters in scenario B. A comparison of model predictions for these two scenarios enabled the determination of the long-term impacts of the BMPs on sediment, and nutrient yields at the outlet of the Dreisbach and Smith Fry watersheds.

5.2.1 Watershed Discretization

SWAT simulations were performed with various watershed configurations for a 30 year time horizon from 1971 to 2000. The characteristics of the watershed configurations that were utilized in this part are summarized in Tables 5.1 and 5.2 for the Dreisbach and Smith Fry watersheds, respectively. The tables include information on the applied critical source area

Table 5.1. Properties of the Watershed Configurations Used for the Dreisbach Watershed.								
Critical Source Area (km^2)	0.03	0.05	0.10	0.15	0.36	0.50	1.5	2.5
Number of Subwatersheds	103	51	29	19	11	5	2	1
Number of HRUs	647	470	359	301	204	138	91	73
Drainage Density (km/km^2)	3.91	3.05	2.28	1.97	1.39	1.22	0.94	0.91
Average Subwatershed Area (km^2)	0.06	0.12	0.22	0.33	0.57	1.26	3.11	6.23

Table 5.2. Properties of the Watershed Configurations Used for the Smith Fry Watershed.								
Critical Source Area (km^2)	0.03	0.05	0.10	0.15	0.30	0.50	1.5	2.9
Number of Subwatersheds	89	63	33	20	12	8	4	1
Number of HRUs	676	577	429	358	278	239	159	95
Drainage Density (km/km^2)	4.09	3.27	2.54	2.25	1.76	1.45	0.96	0.65
Average Subwatershed Area (km^2)	0.08	0.12	0.22	0.37	0.61	0.92	1.83	7.30

(km^2) and corresponding number of subwatersheds, drainage density (km/km^2), and average subwatershed area (km^2). Drainage density is defined as the ratio of total channel length to the total watershed area. Note that the some of the discretization levels in Tables 5.1 and 5.2 are different from the ones reported in Tables 3.1 and 3.2. The corresponding watershed configurations used for the Dreisbach watershed are shown in Figures 5.1 and 5.2, respectively.

5.3 Impact of Best Management Practices on Water Quality

5.3.1 Effects of BMPs on Streamflow

Simulated runoff volume and streamflow at the outlet of the Dreisbach and Smith Fry watersheds were not affected by implementation of the BMPs. This was anticipated, because the BMP selection for the Black Creek project was targeted at sediment and phosphorus reduction (Lake and Morrison, 1977a; Lake and Morrison, 1977b; Morrison and Lake, 1983). Parallel terraces, the only type of BMPs in the study watersheds that influence runoff parameters (see Table 4.3), cover less than 2% and 1% of the Dreisbach and Smith Fry watersheds, respectively. Thus, their impact on simulated streamflow at the outlet of the study watersheds was negligible.

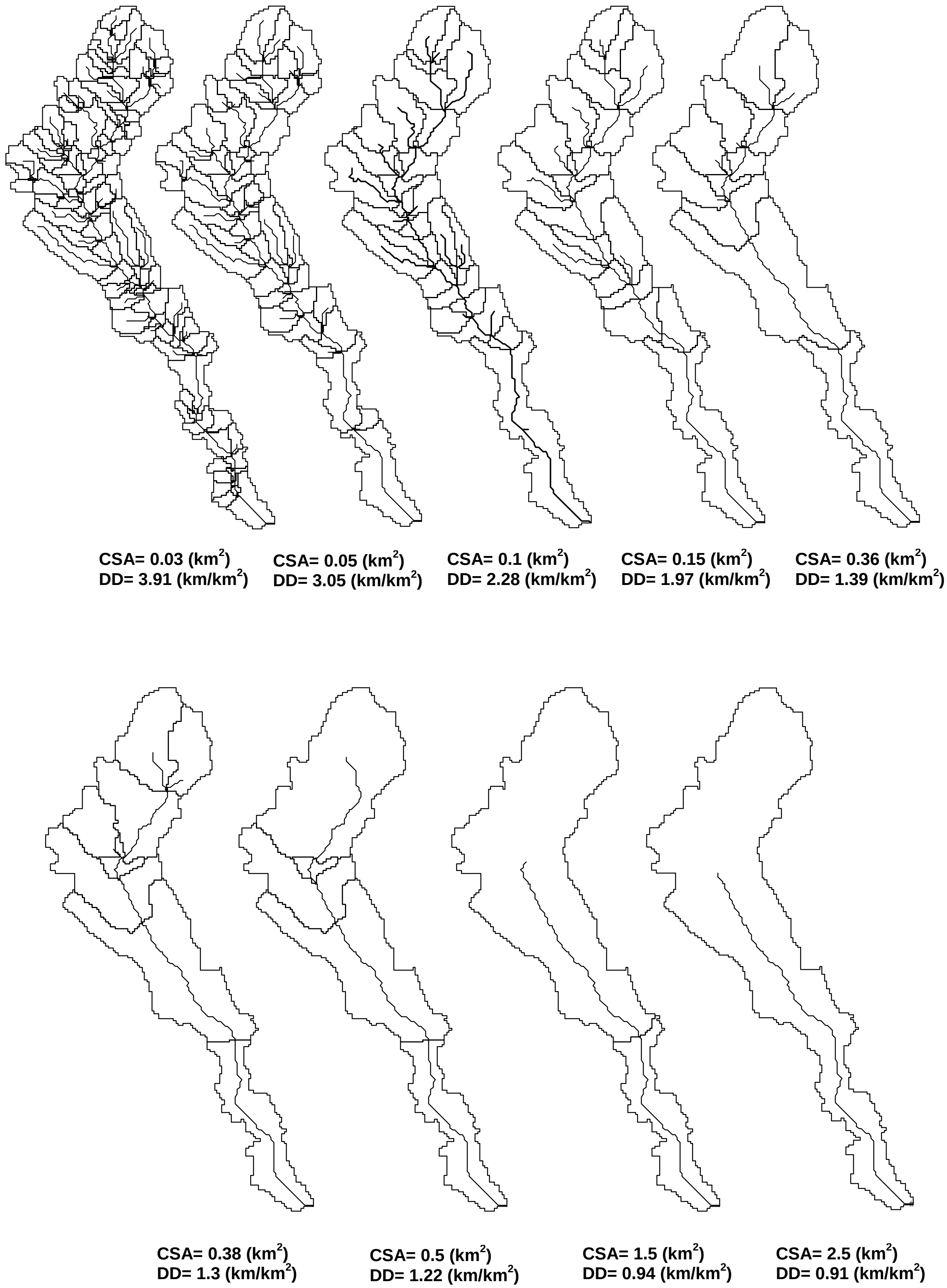
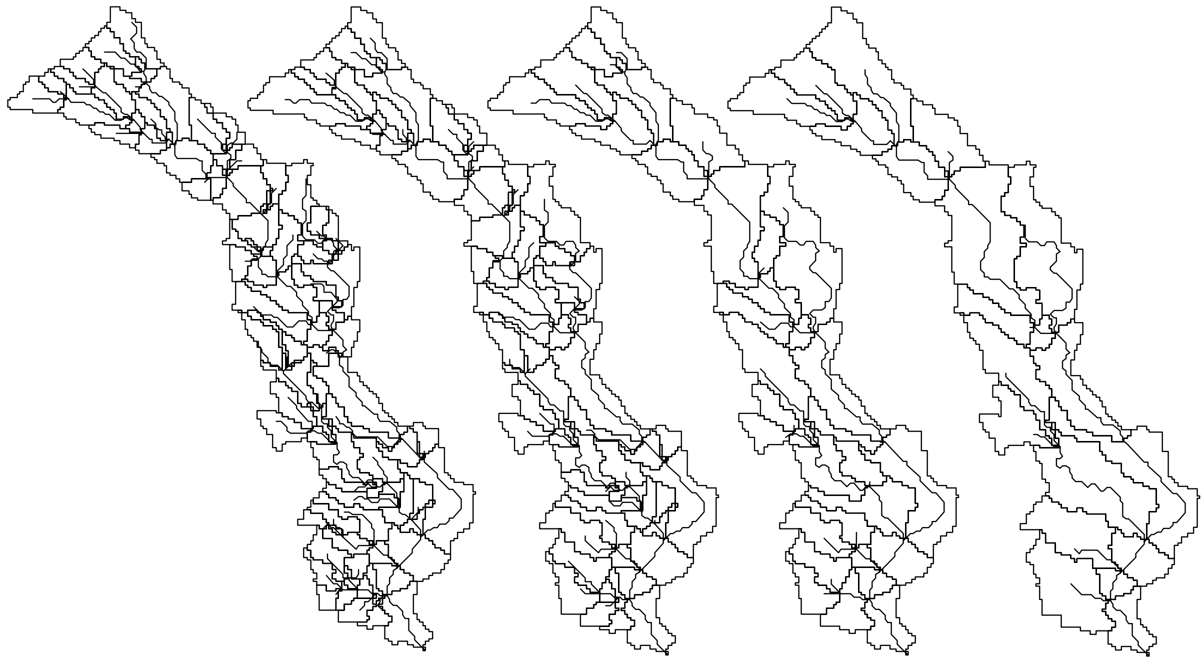


Figure 5.1. Watershed Configurations Used for the Dreisbach Watershed, Indiana.

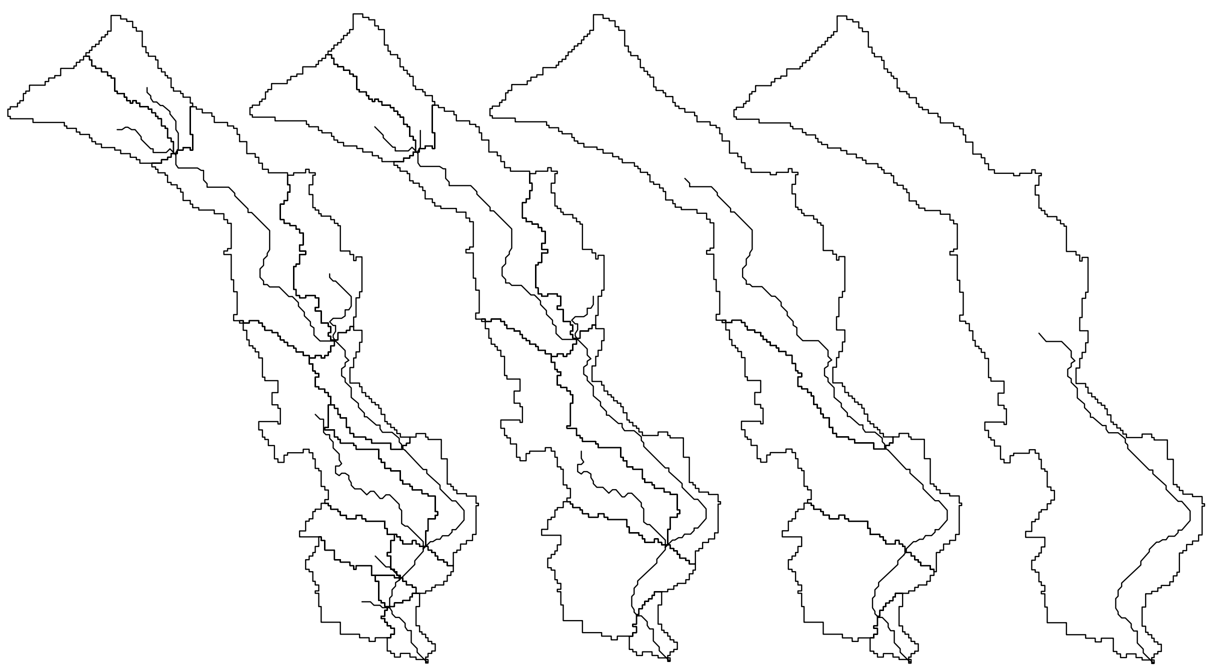


CSA= 0.03 (km^2) DD= 4.09 (km/km^2)

CSA= 0.05 (km^2) DD= 3.27 (km/km^2)

CSA= 0.1 (km^2) DD= 2.54 (km/km^2)

CSA= 0.15 (km^2) DD=2.25 (km/km^2)



CSA= 0.30 (km^2) DD= 1.76 (km/km^2)

CSA= 0.5 (km^2) DD= 1.45 (km/km^2)

CSA= 1.5 (km^2) DD= 0.96 (km/km^2)

CSA= 2.9 (km^2) DD= 0.65 (km/km^2)

Figure 5.2. Watershed Configurations Used for the Smith Fry Watershed, Indiana.

5.3.2 Impact of BMPs on Sediment Yield

The effect of watershed discretization on sediment output of the SWAT model at the outlet of the study watersheds is depicted in Figure 5.3. Under scenario A without the BMPs, average annual sediment yield at the outlet of the watersheds increased by nearly 200% between the coarsest and the finest discretization levels. The increase could be due to two processes: higher sheet erosion from upland areas and/or more intense channel erosion.

The SWAT model employs the Modified Universal Soil Loss Equation (MUSLE) (Eq. 2.5) to estimate sheet erosion. All of the parameters in the MUSLE equation are estimated for each HRU with the exception of USLE topographic factor, *LS*, which is determined for each subwatershed and applied to the HRUs contained in the subwatershed. The results of this study presented in Section 3.0 revealed that the weighted average USLE topographic factor, *LS*, was reduced by nearly 25% between the coarsest and finest discretization levels. The rate of reduction plateaued at finer discretization levels. Similar trends were observed for the computed sheet erosion from upland areas. Consequently, the model predicted that variation of sheet erosion was not the reason for higher sediment yield at the outlet due to finer watershed discretization.

When the impacts of the BMPs were not included (scenario A), sediment yield at the outlet of the watersheds was computed by SWAT to be larger than estimated sheet erosion from upland areas. Because estimated transport capacity of the channel network (Equation 2.6) exceeded sediment loads from upland areas. Thus, channel degradation was predicted by the model to be the dominant channel process and contributed to the sediment yield at the outlet. Dominance of channel degradation indicated that sediment yield at the outlet would increase with drainage density, which increased with finer discretization levels (Figure 3.4). At finer discretization levels, higher drainage density provided longer channel network that would be subject to channel degradation. This resulted in significantly higher sediment yields at the outlets. The correlation coefficient between sediment yield at the outlet and drainage density of the Dreisbach and Smith Fry watersheds was 0.98 and 0.97, respectively. The correlation was extremely poor for scenario B which simulates the presence of the BMPs.

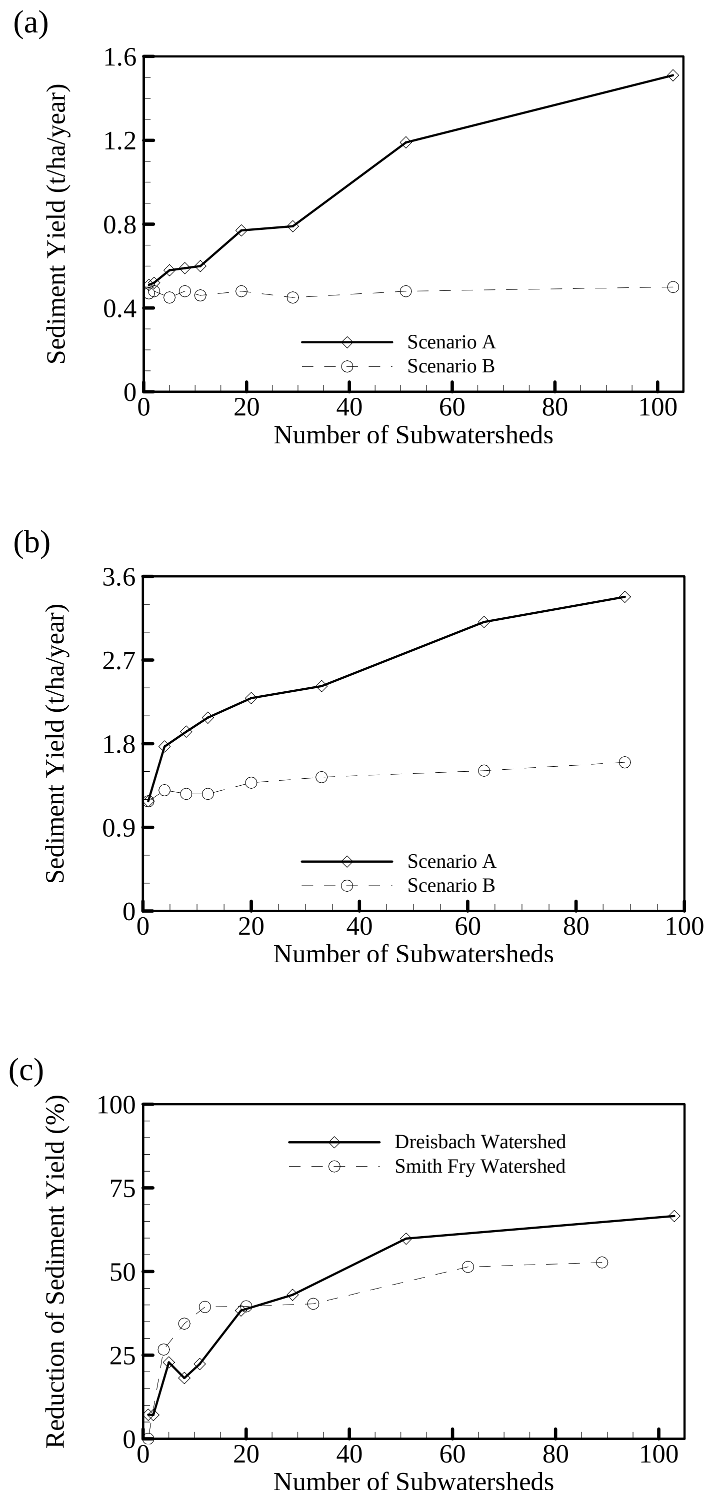


Figure 5.3. Average Annual Sediment Yield at the Outlet of (a) Dreisbach Watershed, (b) Smith Fry Watershed, (c) Percent Sediment Reduction. Scenario A: Simulations with No BMP; Scenario B: Simulations with BMPs in Place.

Predicted sediment yield at the outlet was comparatively stable at various discretization levels when model simulations were performed under scenario B. Transport capacity of the channel network is a function of the peak channel velocity as indicated in Equation 2.6. Implementation of grade stabilization structures in the watersheds resulted in lower main channel slopes while implementation of grassed waterways increased channel resistance, both of which lowered the peak channel velocity. Subsequently, transport capacity of the channel network was significantly lower after implementation of the grassed waterways and grade stabilization structures. With the BMPs, both Dreisbach and Smith Fry watersheds exhibited the characteristics of "transport-limited" watersheds. For such watersheds, estimated transport capacity of the channel network is less than sediment loads from upland areas and sediment deposition is the dominant main channel process. Dominance of channel deposition indicated that sediment yield at the outlet did not increase with drainage density. The results presented in Figure 5.3 confirm that sediment yield at the outlet was relatively insensitive to finer watershed discretization under scenario B when influence of BMPs was included in the model simulations.

As discussed above, several factors contribute to determine the impact of the BMPs on abatement of sediment yield at the outlet of watersheds. An overall evaluation was therefore made by estimating BMP efficacy at any particular discretization level as:

$$\text{Reduction}(\%) = \frac{\text{Model output from scenario A - Model output from scenario B}}{\text{Model output from scenario A}} \tag{5.1}$$

In the Dreisbach watershed, the efficacy of the BMPs for abating sediment yield was evaluated to be only 7% at the coarsest discretization level, while the efficacy was nearly 70% at the finest discretization level. The corresponding efficacy values in the Smith Fry watershed were nearly zero and 50 %, respectively (see Figure 5.3 (c)).

An optimal watershed discretization level for representation of the BMPs and their validity could be identified from Figure 5.3 at a CSA corresponding to 2 % of the total watershed areas. The average subwatershed area at this discretization level was approximately 4% of the total watershed area. There are two major reasons for this recommendation. First, the estimated sheet erosion from upland areas did not vary significantly beyond this discretization level (see Section 3.0). Second, the asymptotic behavior of the average slope of channel network (Figure 3.4) indicated that channel degradation and its contribution to the sediment yield at the outlet also tended to stabilize at finer discretization levels. These trends are more apparent in the Smith Fry watershed where upstream channel network is relatively flatter than the one in the Dreisbach watershed.

5.3.3 Impact of BMPs on Nutrient Yields

Figures 5.4 and 5.5 depict simulated total P and total N yields at the outlet of the Dreisbach and Smith Fry watersheds, respectively. Without BMPs (scenario A), total P predictions by the SWAT model were 200% higher at the finest discretization level in comparison to the coarsest level utilized for both watersheds. However, the rate of change stabilized at finer discretization levels (Figures 5.4(a) and 5.4(b)). Total N predictions of the model exhibited similar trends as evidenced in Figure 5.5. The installed BMPs were estimated to effectively reduce total P yield at the outlet of the Dreisbach watershed by 30% when the finest discretization level was utilized. The reduction (predicted by the SWAT model) corresponding to the coarsest discretization level was 0% (see Figure 5.4(c)). The results presented in Figure 5.5(c) demonstrate that simulated impact of BMPs in alleviating total N yield at the outlet of the Dreisbach watershed also depended on the utilized watershed discretization level. A 25% reduction was obtained at the finest discretization level while the reduction was negligible at the coarsest level. Similar trends were observed for simulated reduction of total P and total N in the Smith Fry watershed as depicted in Figures 5.4(b) and 5.5(b), respectively. From Figures 5.4(c) and 5.5(c), an optimal critical source area corresponding to 2% of total areas of the respective watersheds continues to serve as an appropriate discretization level for evaluation of effectiveness of the BMPs for reduction of total P and total N. This was partly anticipated because the same optimal discretization level was identified earlier for sediment yield.

The reduction in total P load was consistent with the reduction of sediment yield at the outlet of the watersheds. This was anticipated for two reasons. First, in relatively small watersheds like Dreisbach and Smith Fry, the role of in-stream nutrient processes that are simulated by

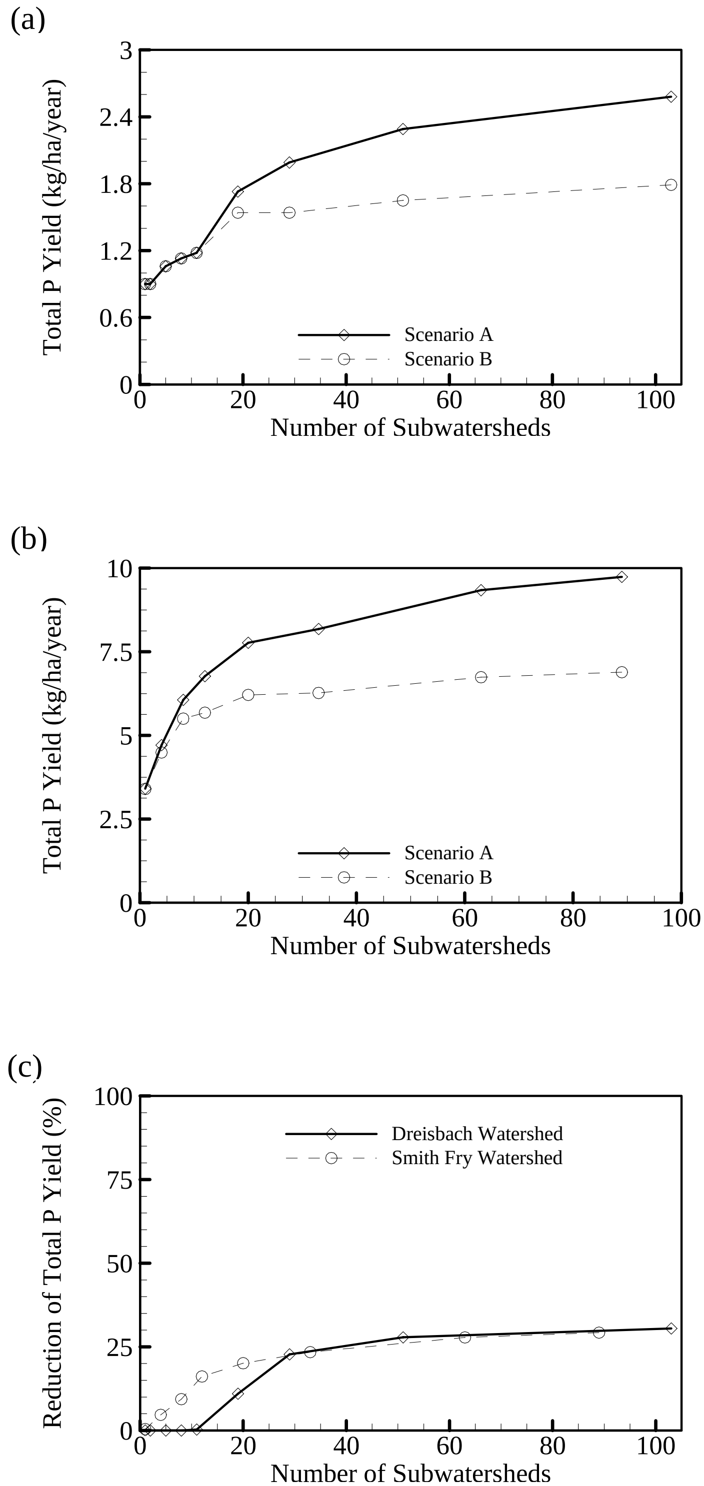


Figure 5.4. Average Annual Total P Yield at the Outlet of (a) Dreisbach Watershed, (b) Smith Fry Watershed, (c) Percent Sediment Reduction. Scenario A: Simulations with No BMP; Scenario B: Simulations with BMPs in Place.

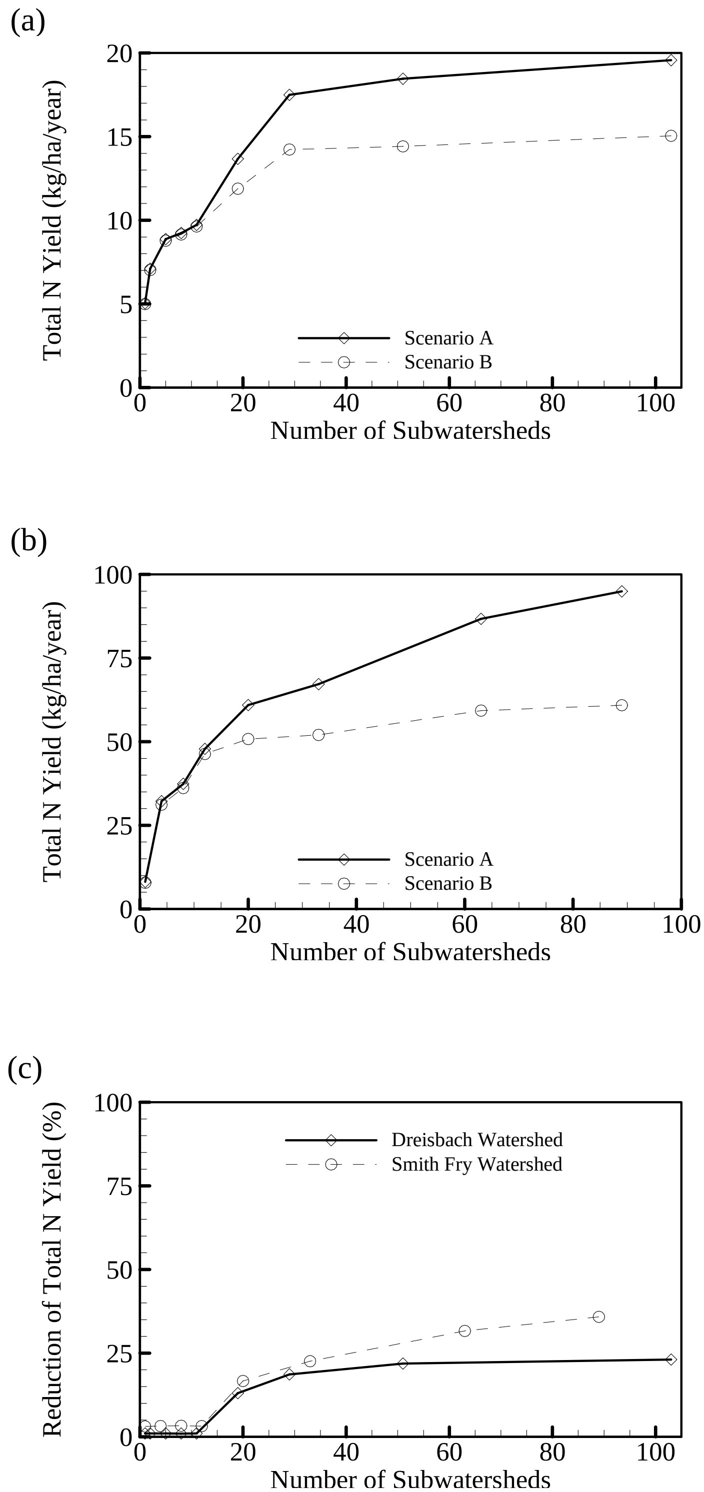


Figure 5.5. Average Annual Total N Yield at the Outlet of (a) Dreisbach Watershed, (b) Smith Fry Watershed, (c) Percent Sediment Reduction. Scenario A: Simulations with No BMP; Scenario B: Simulations with BMPs in Place.

SWAT, such as algal decay on phosphorus yield, is negligible compared to soil loss from upland areas and channel erosion. In such watersheds, it can be claimed that sediment and nutrient yields are correlated. The correlation coefficient between observed sediment yield and nutrient loads (Table 2.5) at the outlets of the study watersheds was 0.72. Moreover, the BMPs installed in the study watersheds were basically sediment control structures. The impact of the BMPs on nutrient loads was as a consequence of reduction of sediment yield.

5.4. Field Scale versus Watershed Scale Evaluation

The impacts of the BMPs in the Dreisbach and Smith Fry watersheds were examined at two spatial scales based on their functionality. Parallel terraces and field borders are implemented to reduce soil loss from upland areas. Therefore, their efficacy may be evaluated at a HRU (or field) scale as well as a watershed scale. The effect of grassed waterways and grade stabilization structures must be discussed at a larger watershed scale because they are implemented in channels and their effects can not be felt on upland areas. Model predictions at the finest discretization level, i.e., critical source area equal to 0.03 (km^2) were applied to compare the efficacy of the BMPs at watershed and field scales. The sediment, total P, and total N reduction rates determined by comparing model simulations with and without inclusion of parallel terraces and field borders are summarized in Table 5.3. In this table, the presented results at HRU scale correspond to reduction rates of model outputs averaged over the particular field-plots where the parallel terraces and field borders have been implemented (shown in Figure 2.2). At a watershed scale, these BMPs did not contribute to appreciable sediment, total P, and total N reductions. This was anticipated because they have been placed

Table 5.3. Reduction of Sediment, Total P, and Total N loads Resulted from Implementation of Parallel Terraces and Field Borders.

Watershed	Scale	% Reduction		
		Sediment	Total P	Total N
Dreisbach	HRU[1]	57	50	55
	Watershed	2	2	2
Smith Fry	HRU[1]	45	30	35
	Watershed	1	1	1

[1]Obtained by averaging the reduction rates over the HRUs (i.e. fields) where the parallel terraces and field borders have been installed.

to target small portions of the study watersheds. On the contrary, sediment, total P, and total N loadings from the fields where the terraces and field borders were installed decrease by nearly 57%, 50%, and 55% in the Dreisbach watershed and by 45%, 30%, and 35% in the Smith Fry watershed, respectively, which implies that land owners would substantially benefit from their implementation, if the regulation were to be imposed immediately downstream of the upland area.

Grassed waterways and grade stabilization structures would likely be more beneficial to development of a sediment and nutrient TMDL at the outlet of the study watersheds. As illustrated in Figures 5.3(c), 5.4(c), and 5.5(c), sediment, total P, and total N yields at the outlet of the Dreisbach watershed decreased by nearly 70, 25, and 30% as a result of the installation of the waterways and stabilization structures. The corresponding values in the Smith Fry watersheds were approximately 50, 30, and 35%.

Interestingly, although the number of the BMPs implemented in the Smith Fry watershed was significantly less than that for the Dreisbach watershed, the estimated sediment and nutrient reduction rates were comparable. This indicates that not only the number of the BMPs, but also their location in the watershed plays a significant role. Our assessment of the impact of individual BMPs revealed that the two grade stabilization structures at the downstream portion of the channel network in the Smith Fry watershed were the primary reason for such reduction rates. These structures lowered the transport capacity of upstream channel segments that resulted in deposition of a large amount of the sediments and nutrients in the channel network. Thus, the simulated sediment and nutrient yields at the outlet were dramatically reduced. This would imply that for maximum benefits, the BMPs should be placed as close upstream as possible to where the regulation will be imposed. It also suggests that with proper implementation of BMPs, managers are able to exert enough control to convert a supply-limited watershed to a transport-limited one.

5.5. Conclusions

For the study watersheds, sediment, total phosphorus, and total nitrogen outputs of the SWAT model were highly influenced by watershed discretization before representation of the BMPs. Predicted dominance of channel degradation in simulations without BMPs resulted in an increase of these outputs with drainage density, which increased with finer discretization levels.

The implemented grassed waterways and grade stabilization structures appreciably reduced the transport capacity of the channel network of the watersheds. After implementation of the BMPs, sediment deposition was the dominant channel process in the study watersheds. The predicted sediment yield at the outlet of the study watersheds was relatively stable and did not vary with finer discretization.

The predicted reduction of sediment and nutrient yields as a result of implementation of the BMPs were insignificant when more coarse levels of discretization were applied. Utilization of the finer discretization levels resulted in substantial sediment and nutrient reductions according to the model. An optimal discretization level at a critical source area corresponding to 2% of the total watershed area was identified to be adequate for representation of the BMPs and assessment of their validity. Study results indicated that a proper assessment of the efficacy of the BMPs must be conducted in conjunction with multiple watershed discretization levels.

The management implications of this study were found to be scale dependent. Implementation of parallel terraces and field borders significantly alleviated estimated sediment and nutrient loadings from the fields where they have been installed. The reduction was negligible at the outlet of the study watersheds. While land owners may identify parallel terraces and field borders as being very effective for controlling downstream discharges, watershed managers may not appreciate their impact on water quality at the outlet of the Dreisbach and Smith Fry watersheds. Based on the SWAT model simulations, at a watershed scale, grassed waterways and grade stabilization structures appeared to more effectively reduce sediment and nutrient yields at the outlets. In particular, grassed waterways and grade stabilization structures located in the downstream portion of the channel network increased channel deposition in upstream segments. It may be concluded that placement of the BMPs plays an important role in improving the water quality at the outlet of the watersheds.

Identification of the most appropriate locations for implementation of abatement strategies requires a better understanding of control processes in a watershed.

Since different sets of calibrated parameters may be obtained from the calibration procedure, applying sensitivity and uncertainty analysis techniques would be valuable for identification of control processes and key management actions such as sheet erosion, channel degradation, and channel deposition within a watershed. In a watershed where channel degradation is the dominant main channel process, implementation of grassed waterways and grade stabilization structures would be highly successful in reducing sediment and nutrient loads to the extent of converting a supply-limited watershed to a transport-limited one. Application of BMPs such as parallel terraces and field borders would be more successful for watersheds where upland areas are the dominant sources of sediments and nutrients. Their role in changing the overall nature of the watershed is likely to be minimal.

The results of this study, which was conducted on small watersheds, should be verified by other studies focused on evaluation of effectiveness of BMPs at various watershed discretization levels. Sediment and nutrient yields from larger watersheds may exhibit different trends with watershed discretization. The method presented in this paper for evaluation of effectiveness of BMPs at various discretization levels is recommended for other watershed studies because uncertainties resulting from spatial resolution deserve more attention than has been devoted to them in the past.

Section 6.0
Source Identification

6.1 Introduction

Identification of sediment and nutrient sources within a watershed has many important implications for watershed management. Once nonpoint sources of sediment and nutrients are identified, managers will be able to examine whether abatement strategies such as implementation of Best Management Practices (BMPs) effectively reduce sediment and nutrient losses. The question of where to place BMPs for maximum benefits also depends on being able to identify contaminant sources. Major sources of sediment and nutrients within a watershed can be categorized into sheet erosion from upland areas, and channel degradation. Anthropogenic activities are known to contribute to both of these sources of erosion.

Sheet erosion results in removal of a fairly uniform layer of sediment and agrichemicals that adhere to sediment particles from upland areas. Tillage and fertilizer application are perhaps the most important anthropogenic activities that directly increase nutrient loads from upland areas. An accurate estimate of sediment and nutrient loads from upland areas will be beneficial to land owners and field-scale managers as well as watershed-scale managers.

In addition to sheet erosion, channel degradation and other channel processes that influence nutrient yield at the outlet have significant roles in watershed-scale management. Channels within a watershed not only serve as a conduit for movement of contaminant-laden sediments, but may also act as a source because of erosion from streambeds and bank erosion. Channel erosion may significantly contribute to sediment and nutrient yields at the outlet, especially in supply-limited watersheds where transport capacity of channel network is larger than sediment concentration in channel flow due to sheet erosion from upland areas. Algal growth, transformation and respiration rates, and other in-stream processes may influence transport of organic and inorganic forms of phosphorus and nitrogen. A proper assessment of sediment and nutrient sources should include the influence of these activities.

Most of the available data from monitoring programs have been collected at the outlet of watersheds. These data are not adequate to directly identify nonpoint sources within watersheds since sediment and nutrient movement over a watershed tends to be fairly dynamic, often behaving in a nonlinear fashion. Flow, sediment, and nutrient monitoring programs should be conducted at both field and watershed scales to help decision making and management at various spatial scales. In doing so, however, there are two issues that need to be addressed. First, installation, maintenance, and operation of monitoring stations are usually expensive and time consuming. It is almost impossible to conduct such a program for every single system of interest. Second, analysis of historical data may not be adequate for evaluation of the impact (s) of certain management actions on the system, especially the ones that have not been implemented yet.

Modeling studies not only provide a versatile tool for assessing the future of a given system under various scenarios, but can also be used to examine whether a certain future state is attainable for the system. Thus, they can be used for development and implementation of a TMDL for the design variable (s) of concern. In this study, the SWAT model was selected to assess the impact of implementation of various best management practices (BMPs) in the Dreisbach and Smith Fry watersheds in Indiana. Good soil, land use, and management data are available for the study watersheds and can be utilized by BASINS to prepare the required input files for SWAT simulations. Application of SWAT for source identification and evaluation of performance of BMPs are discussed in this section.

6.2 Objective

In Section 3.0, a method was developed to obtain an optimal watershed discretization level such that further refinement would not change SWAT computations for sediment and nutrient loadings from upland areas. The conditions under which such an “optimal” resolution would be available were also determined. The goal of this section is to develop sediment and nutrient maps for the study area. These maps will be generated based on best resolutions available for soil, land use and management data and are indicators of the sources within the study area. Two scenarios are examined: sediment and nutrient sources before implementation of BMPs (scenario A), and sediment and nutrient sources after implementation of BMPs (scenario B). Furthermore, the effectiveness of BMPs in reducing sediment and nutrient loads will be evaluated.

6.3 Methodology

To fulfill the source identification objectives, SWAT was calibrated and validated for the Dreisbach and Smith Fry watersheds. The results of calibration procedure were presented in Section 4.0. These two watersheds are located in the Black Creek watershed almost 10 km apart. Calibration of the SWAT model resulted in the same model inputs for both watersheds except for USLE practice factor (USLE_P). In Section 3.0, an optimal watershed discretization level was identified for both watersheds. It was shown that SWAT computations are not sensitive to critical source area (CSA) for resolutions finer than 20 (ha). The experience gained form previous sections provides more confidence in application of SWAT computations for source identification purposes. The mechanisms utilized by SWAT to compute sheet erosion and channel processes were explained in detail in Section 2.

6.3.1 Sediment and Nutrient Source Maps

SWAT model simulations were performed over a 30 year time period from January 1970 to December 2000. Average annual quantities predicted for subwatersheds were utilized as sediment and nutrient source indicators. Figures 6.1-6.2 depict the source maps for sediment, total P and total N loads from upland areas of the Dreisbach watershed before and after implementation of BMPs, respectively. Similar maps are presented in Figures 6.3-6.4 for the Smith Fry Watershed.

Various segments of channel network can be sources of sediment and nutrient. Sediment and nutrient loads from the channel networks of Dreisbach and Smith Fry before and after implementation of BMPs are presented in Figures 6.5-6.8. A positive value indicates that the channel segment serves as a source of sediment or nutrient while a negative value is an indicator of sediment or nutrient deposition in the channel segment.

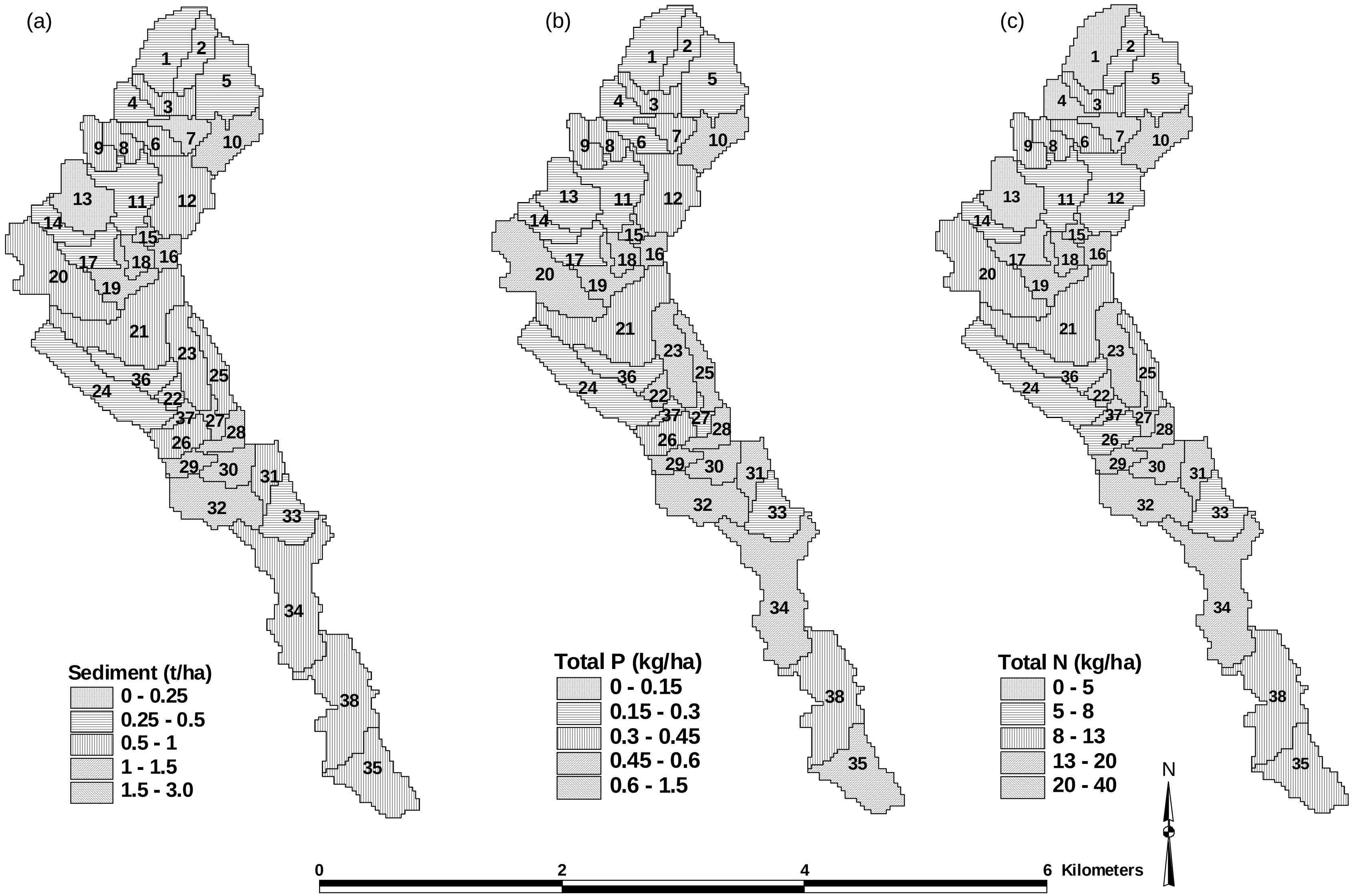


Figure 6.1. Simulated Average Annual Loads Generated at Upland Areas before Implementation of BMPs for Dreisbach Watershed, 1971-2000: (a) Sediment, (b) Total P, and (c) Total N.

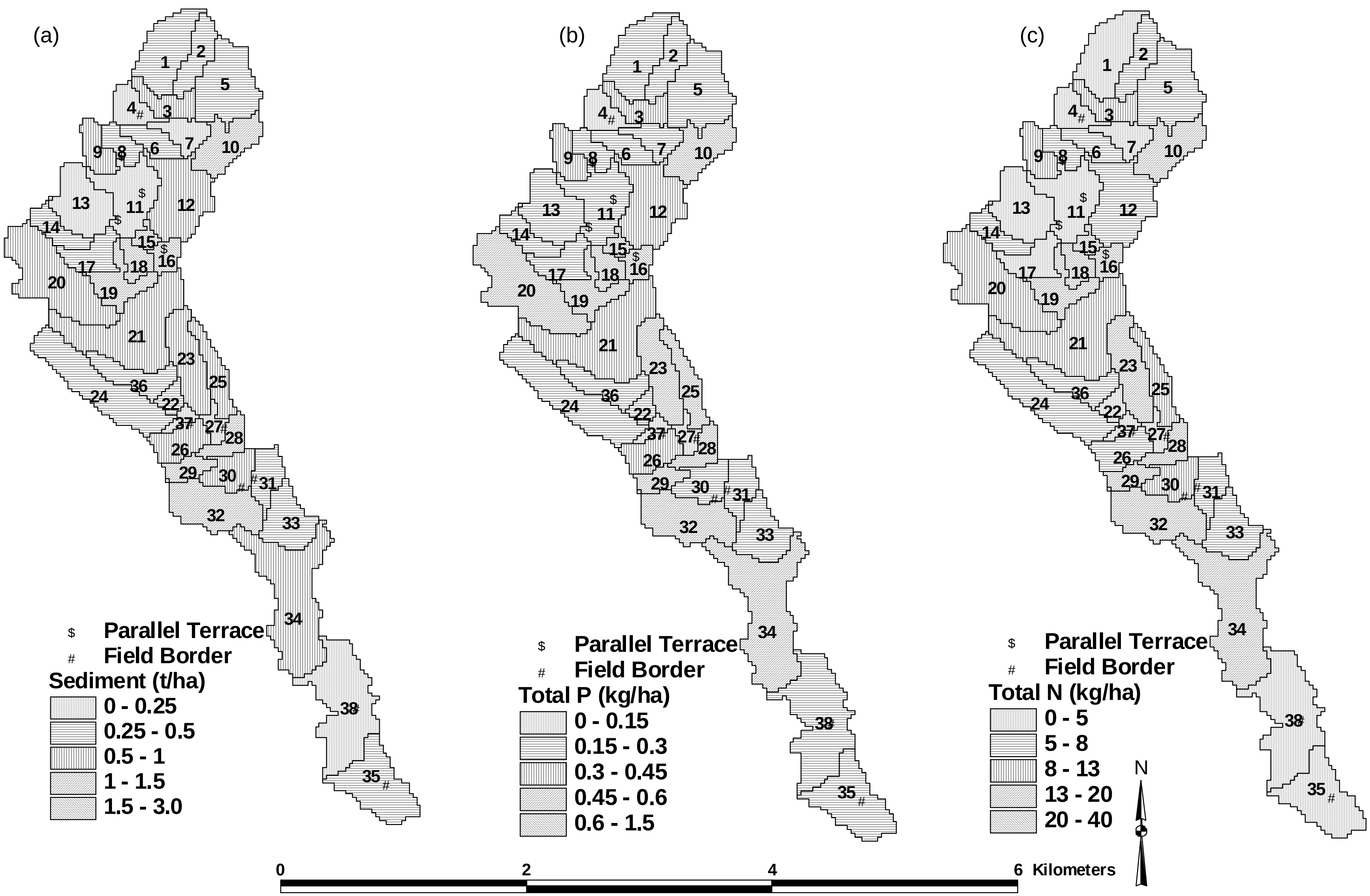


Figure 6.2. Simulated Average Annual Loads Generated at Upland Areas after Implementation of BMPs for Dreisbach Watershed, 1971-2000: (a) Sediment, (b) Total P, and (c) Total N.

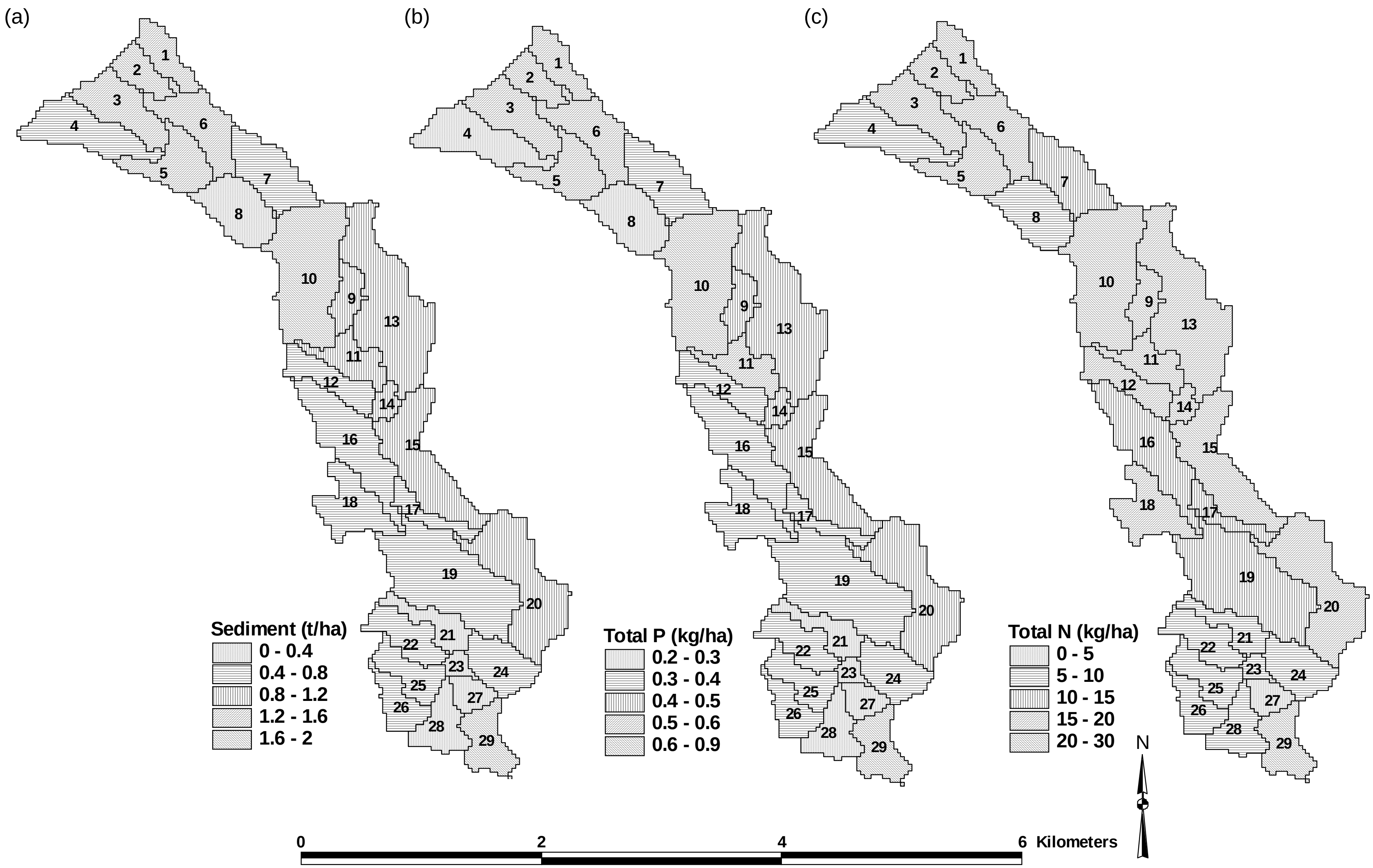


Figure 6.3. Simulated Average Annual Loads Generated at Upland Areas before Implementation of BMPs for Smith Fry Watershed, 1971-2000: (a) Sediment, (b) Total P, and (c) Total N.

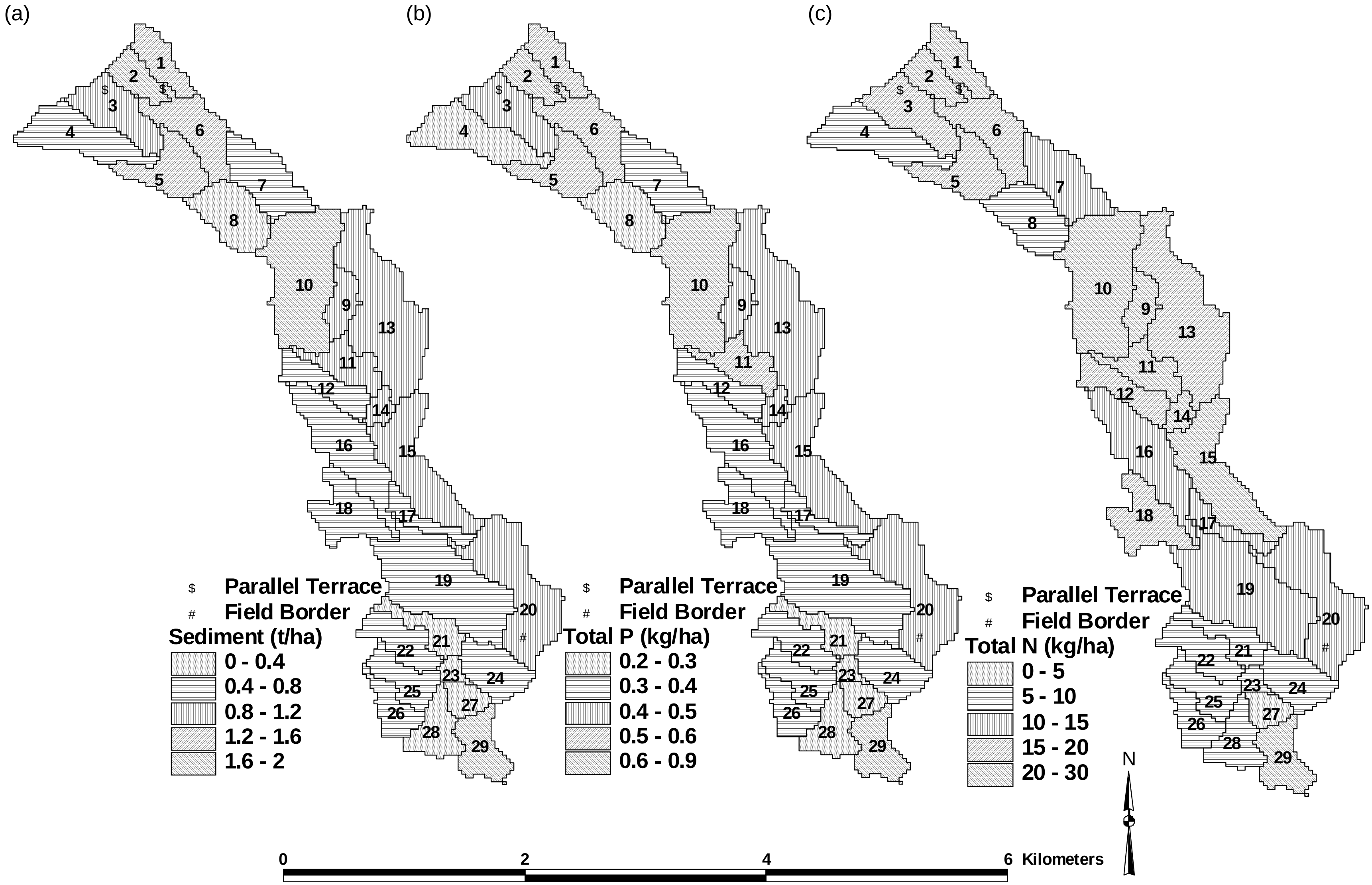


Figure 6.4. Simulated Average Annual Loads Generated at Upland Areas after Implementation of BMPs for Smith Fry Watershed, 1971-2000: (a) Sediment, (b) Total P, and (c) Total N.

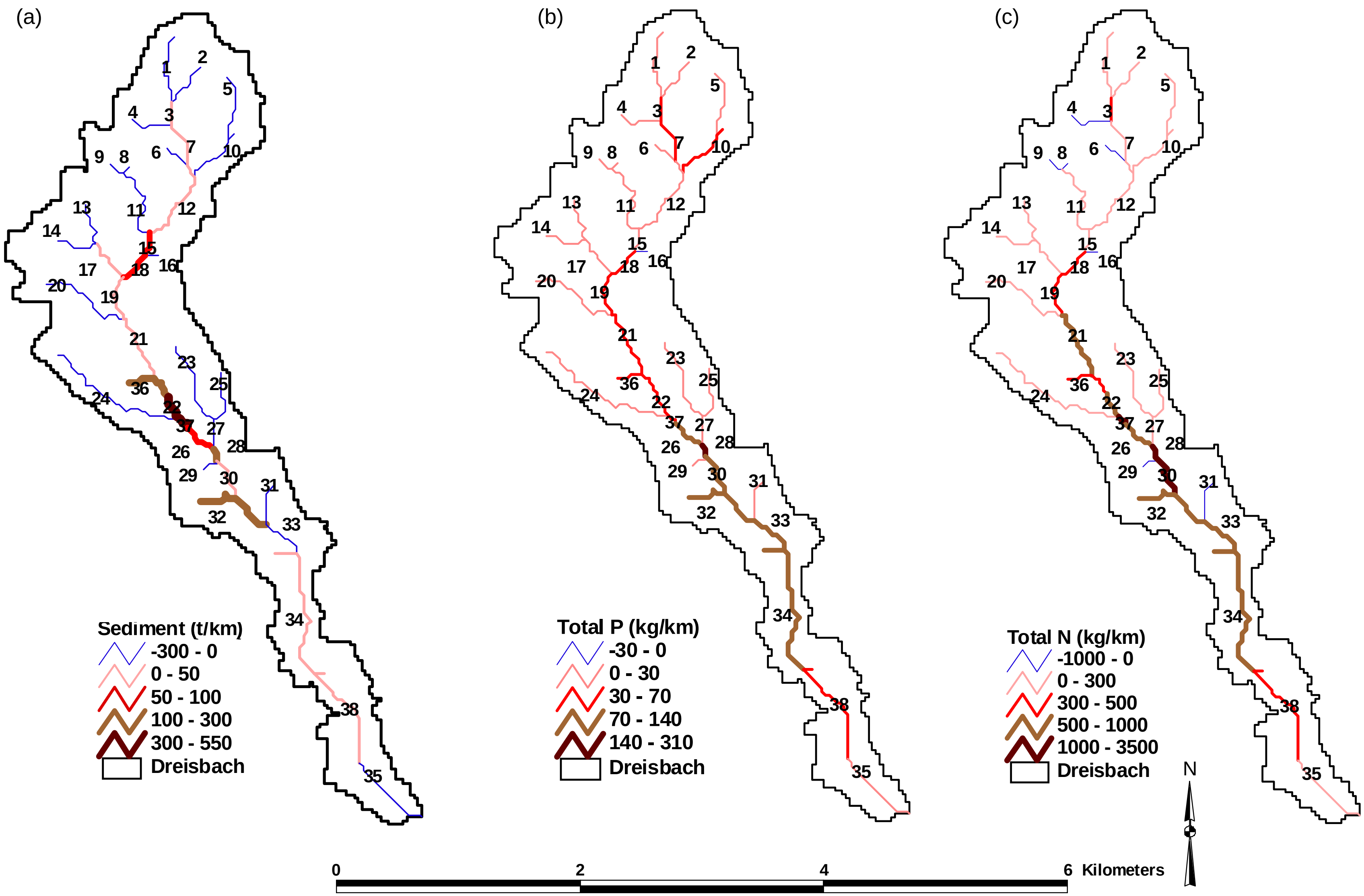


Figure 6.5. Simulated Average Annual Loads from Channel Network before Implementation of BMPs for Dreisbach Watershed, 1971-2000: a. Sediment, b. Total P, and c. Total N.

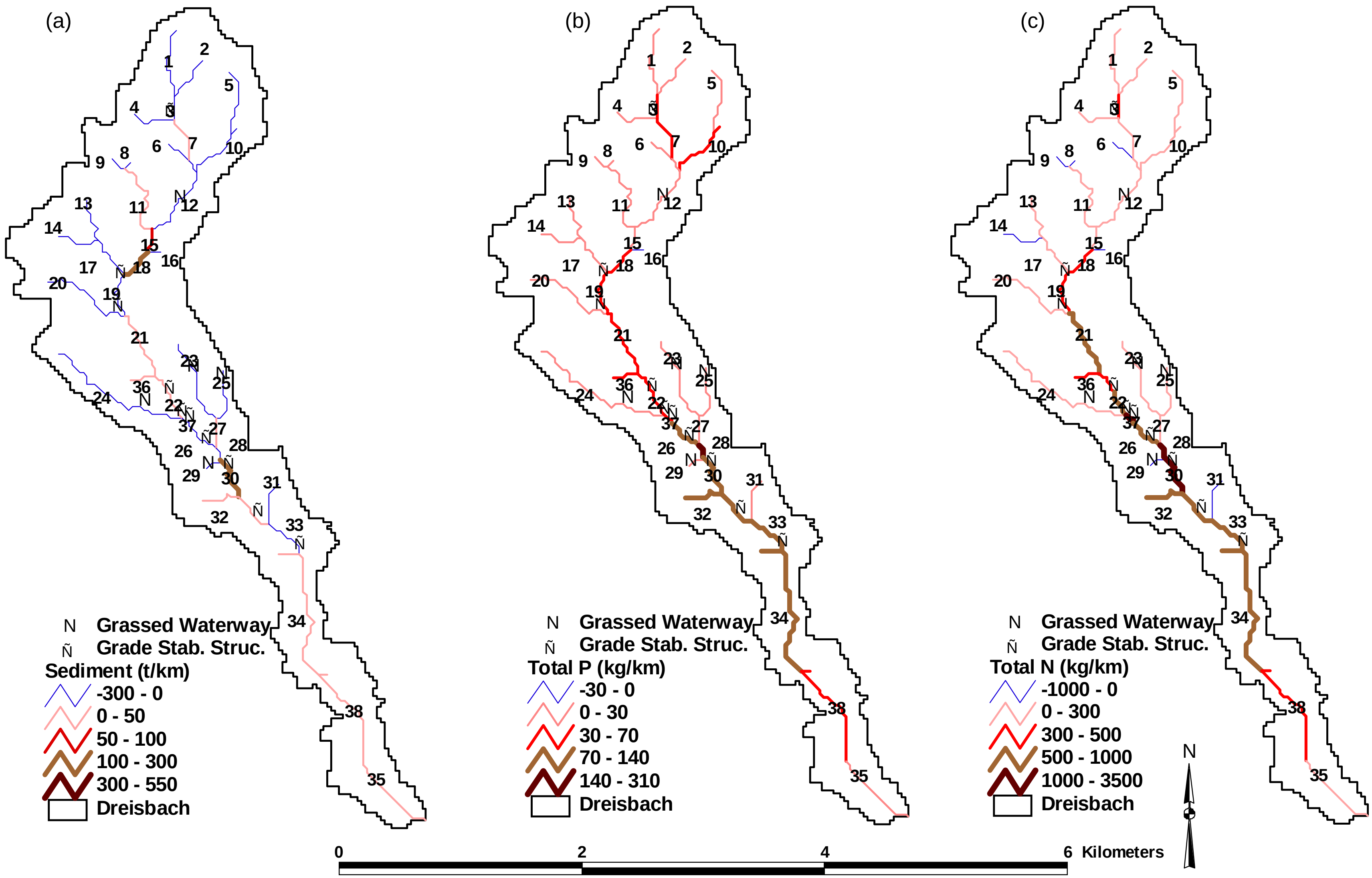


Figure 6.6. Simulated Average Annual Loads from Channel Network after Implementation of BMPs for Dreisbach Watershed, 1971-2000: a. Sediment, b. Total P, and c. Total N.

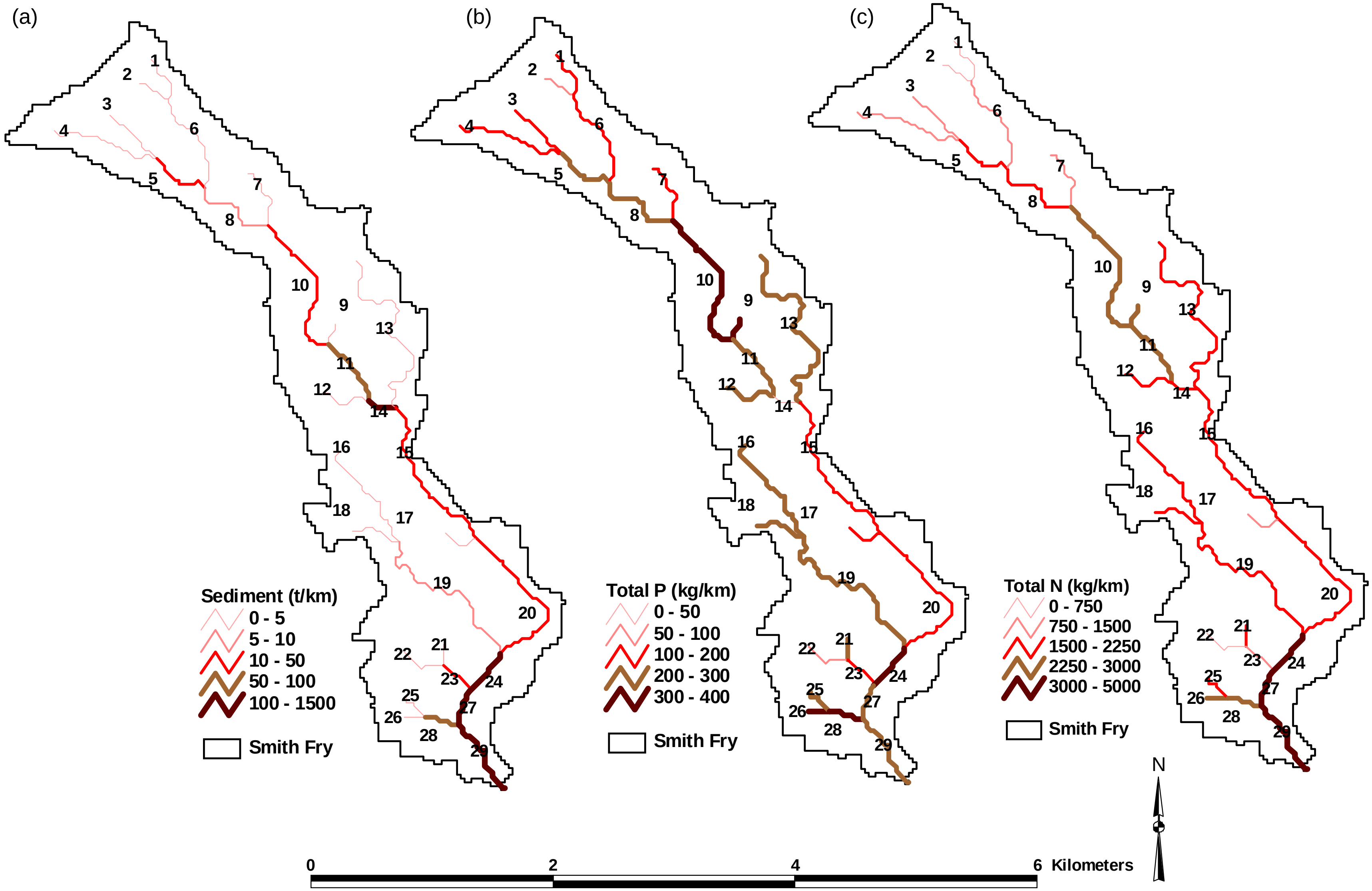


Figure 6.7. Simulated Average Annual Loads from Channel Network before Implementation of BMPs for Smith Fry Watershed, 1971-2000: 1971-2000: a. Sediment, b. Total P, and c. Total N.

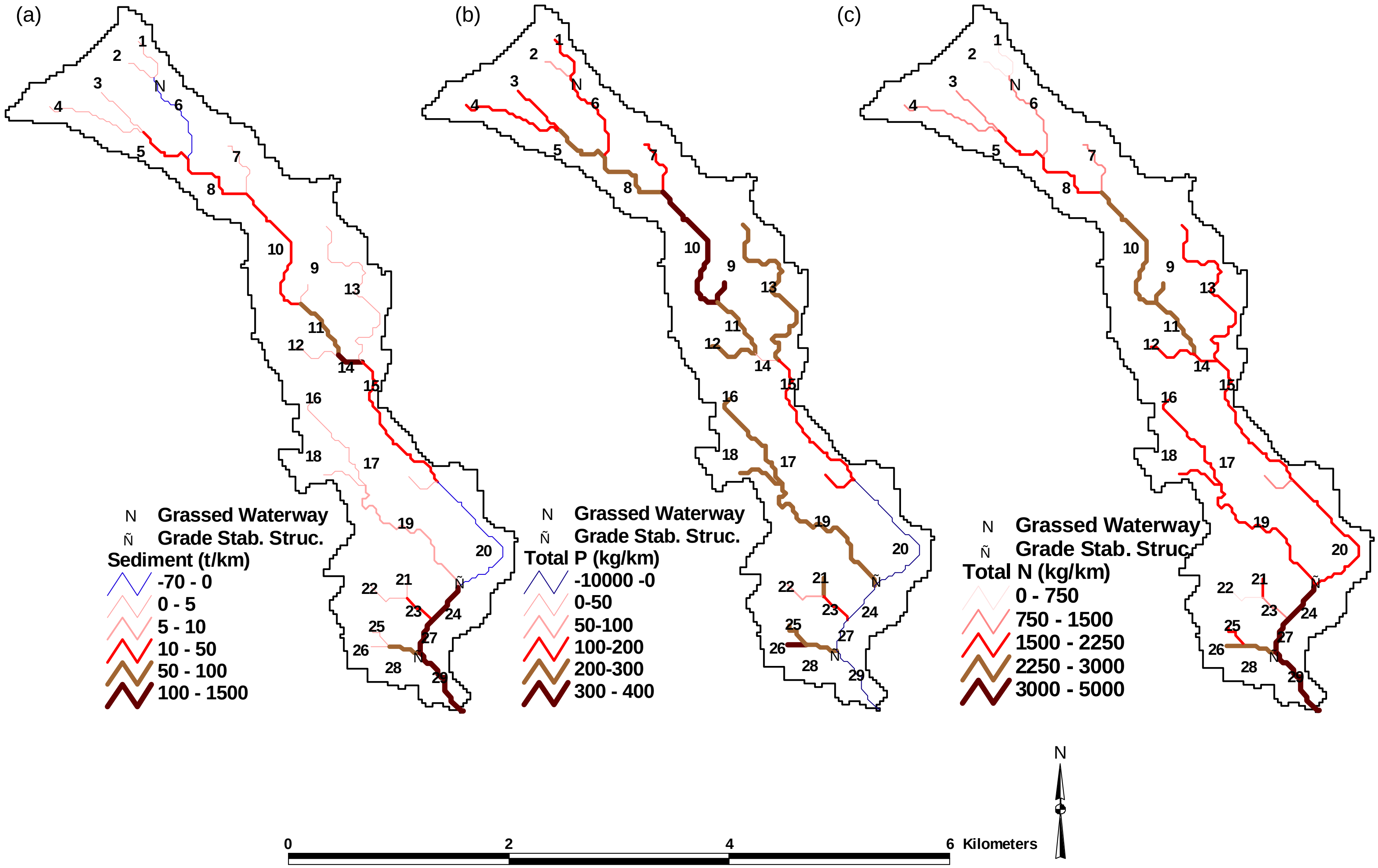


Figure 6.8. Simulated Average Annual Loads from Channel Network after Implementation of BMPs for Smith Fry Watershed, 1971-2000: a. Sediment, b. Total P, and c. Total N.

6.3.2 Long-Term Performance of BMPs

Long-term impacts of the field borders, parallel terraces, grassed waterways, and grade stabilization structures on water quality of the Dreisbach and Smith Fry watersheds were discussed in Section 5.0. Here these impacts are assessed for the particular HRUs and channel segments where the BMPs have been installed.

Table 6.1 includes simulated sediment, total P, and total N loads from particular fields in the Dreisbach and Smith Fry watersheds with field borders for scenarios A and B (defined in Section 5.2). Scenario A corresponded to model results without BMPs, while scenario B simulated the design variables (sediment and nutrient yields) with the particular BMP in place. The location of the field borders and the number of the subwatersheds where they are located is presented in Figures 6.2 and 6.4 for the Dreisbach and Smith Fry watersheds, respectively. The results of model predictions presented in Table 6.1 indicate that implemented field borders resulted in a nearly 60% reduction of sediment and nutrient loads from the corresponding fields in the Dreisbach watershed. In Smith Fry watershed, there was only one field border that reduced sediment, and total P loads by 50% and total N by 40%.

The impact of parallel terraces on sediment and nutrient loads from the field where they have been installed is presented in Table 6.2. Based on model simulations, the average reduction

Table 6.1. Impact of Field Borders on Sediment, Total P, and Total N Loads at A Field Scale.

Watershed	Location (subwatershed)	Sediment (t/ha)		Total P (kg/ha)		Total N (kg/ha)	
		Scen. A	Scen. B	Scen. A	Scen. B	Scen. A	Scen. B
Dreisbach	4	0.262	0.102	0.3	0.1	4.0	1.6
	27	0.136	0.056	0.3	0.1	2.4	1.0
	30	1.321	0.532	0.7	0.3	20.1	8.1
	31	0.746	0.304	0.5	0.2	15.0	6.1
	35	0.645	0.251	0.5	0.2	10.7	4.2
	37	0.111	0.046	0.2	0.1	1.8	0.7
	38	0.540	0.212	0.4	0.2	9.3	3.7
Smith Fry	20	1.038	0.532	0.5	0.3	16.5	9.6

Table 6.2. Impact of Parallel Terraces on Sediment, Total P, and Total N Loads at A Field Scale.							
Watershed	Location (subwatershed)	Sediment (t/ha)		Total P (kg/ha)		Total N (kg/ha)	
		Scen. A	Scen. B	Scen. A	Scen. B	Scen. A	Scen. B
Dreisbach	8	0.719	0.350	0.4	0.3	11.7	6.3
	11	0.347	0.173	0.3	0.2	5.2	3.1
	16	2.957	1.561	1.3	0.8	36.2	21.2
Smith Fry	2	1.910	0.812	1.0	0.5	27.1	14.3
	3	1.313	1.073	0.6	0.5	19.1	15.7

of sediment, total P, and total N loads from the corresponding fields in the Dreisbach watershed was approximately 50%, 30%, and 40%, respectively. The corresponding reduction of sediment, total P, and total N loads in the Smith Fry watershed were nearly 40%.

It was discussed in Section 5.0 that based on SWAT computations, grassed waterways and grade stabilization structures were the effective BMPs in the study watersheds, mainly because they significantly lower the transport capacity of the channel segments where they have been implemented. Table 6.3 shows the predicted sediment loads from the channel segments before and after inclusion of the grassed waterways and grade stabilization structures in the study watersheds. The values in the table were computed by subtracting sediment loads at the beginning of the channel segment from the ones at the end. It is evident that based on SWAT computations, sediment erosion was the dominant channel process in most of these segments prior to implementation of the BMPs. After implementation of the grassed waterways and grade stabilization structures (see Figures 4.6 and 4.8 for their locations), channel deposition was dominant in most of these segments. In Table 6.3, a positive value refers to channel degradation (erosion), while a negative value indicates channel deposition. It is observed that implementation of grade stabilization structures almost in all of the cases (except for the one installed on channel segment 29 in the Dreisbach watershed) resulted either in channel deposition or a significant reduction of channel erosion in both watersheds. Model predictions imply that implementation of grassed waterways and stabilization structures on the channel segments in equilibrium, i.e., no channel degradation and/or channel erosion in the segment, did not change channel characteristics. This indicated that sediment loads in and out of the channel segment were the same even after installation of these BMPs.

Table 6.3. Impact of Grassed Waterways on Sediment Loads (t/km) from Channel Segments.						
BMP	Dreisbach			Smith Fry		
	Channel Segment	Scen. A	Scen. B	Channel Segment	Scen. A	Scen. B
Grassed Waterway	12	2.403	-39.779	6	1.944	-6.389
	23	0	0			
	24	0	0			
	25	0	0			
	29	0	0			
Grade Stabilization Structure	3	25.684	-24.766	20	28.522	-18.412
	17	6.042	-3.241	27	1378.12	238.327
	19	31.210	-240.008			
	22	432.699	16.986			
	26	60.436	-121.837			
	28	286.374	-9.483			
	29	0.000	0.000			
	32	115.939	11.069			
	33	-164.403	-231.304			
	36	185.232	14.407			
	37	522.489	-283.529			

It should be noted that implementation of sediment reduction BMPs at a particular point of a channel network does not only affect the upstream channel segments. If implementation of a grassed waterway or grade stabilization structure result in significant reduction in concentration of sediments in channel flow, cannel degradation may happen in downstream segments. This can be observed in channel segment 35 in the Dreisbach watershed (refer to Figures 6.5a and 6.6a). Slope of channel network in this part of the watershed is very small. Therefore, estimated transport capacity of this segment is very low. Before implementation of the grassed waterways and grade stabilization structures upstream of this channel segment, simulated sediment concentration in channel flow was more than its transport capacity. However, after implementation of these BMPs, sediment concentration in channel flow was significantly reduced and was smaller than transport capacity of the channel segment. Thus, model simulations indicated that channel degradation occurred after implementation of the BMPs. This would imply that for maximum benefits, the BMPs should be placed as close upstream as possible to where the regulation will be imposed.

6.4 Conclusions

A calibrated and validated SWAT model was utilized to identify sediment and nutrient sources within the Dreisbach and Smith Fry watersheds. Average annual quantities predicted by the model were used to generate sediment and nutrient source maps. The results of this study based on model simulations revealed that before implementation of BMPs, channel processes, namely streambed or/and bank erosion, would contribute to sediment, total P and total N loads at the outlet of the study watersheds. It was observed that implementation of the BMPs in the Dreisbach watershed would result in a significant reduction of sediment and nutrient yields. These reductions would be mainly due to implementation of grade stabilization structures and grassed waterways that appreciably reduce the transport capacity of channel network. Parallel terraces and field borders would be more effective at a farm scale (i.e. subwatershed scale) while their effect on the sediment and nutrient yields at the outlet would be relatively small. Also, the results indicate that spatial scale has a significant role in the appraisal of the effectiveness of BMPs.

The attributes of the selected watershed and the watershed model, upland sediment and nutrient loading, in-stream processes, or a combination thereof will control estimates of sediment and nutrient yield at the outlet of the watershed. Calibration of a model while essential to sediment and nutrient source identification, may not be sufficient since it usually does not result in a unique set of input parameters. Performing an uncertainty analysis will be critical for accurate interpretation of model results. Key control processes and management actions can be identified by applying a proper sensitivity analysis.

In conclusion, application of watershed models such as SWAT in identification of sediment and nonpoint sources requires two major steps. First, the model should be calibrated and validated for the study area. Model simulations are performed to predict sediment and nutrient loads from upland areas and at the outlet. A comparison of the two will provide a good assessment of control processes in the watershed. Furthermore, a detailed sensitivity and uncertainty analysis will be useful in confirmation and interpretation of the results from the previous step. In this study, sediment and nutrient sources within Dreisbach and Smith Fry watersheds were identified by utilizing SWAT simulations after model calibration. A thorough sensitivity analysis is required for further verification and interpretation of the results.

Section 7.0
Conclusions

The regulations stipulated for the Total Maximum Daily Load (TMDL) program require all of the states to identify impaired water bodies within the states, and develop abatement strategies for the impairment (s) of concern. NRC (2001) reported that implementation of TMDL program is pivotal in securing the nation's water quality goals and should be the target of management and decision making in watershed systems. Successful development of the TMDL program depends to a large extent on the ability of managers and analysts (i) to understand the transport and fate of contaminants within watersheds, and (ii) to evaluate the outcome (s) of a certain management action on water quality of the system. Modeling proves to be a useful tool for such purposes. Simulation models not only facilitate contemplating the future of a given system under various management scenarios, but can also be used to examine whether a certain future state is attainable for the system.

According to the latest list submitted to EPA, sediment and nutrients are the most encountered cases of impairment in watersheds. Natural sources of sediment and nutrients are primarily upland areas, including both sheet and rill erosion, and channel segments under streambed and/or bank erosion. Anthropogenic activities are known to contribute to both of these sources of sediments and nutrients. Over the past 30 years, Best Management Practices (BMPs) have been installed in watersheds to reduce sediment and nutrient from various sources. However, their implementation has been rarely followed by a long-term monitoring program to study the performance of the BMPs. In the absence of good measured data, watershed models can be utilized for such an evaluation. In this study, performance of various BMPs in abatement of sediments and nutrients in two agricultural watersheds in Indiana was investigated through the device of a watershed model. The management implications of the study are site-specific and may not hold for other watershed systems. However, the developed methodology for evaluation of the efficacy of BMPs can be applied for other watersheds.

7.1 Management Implications

Four different types of agricultural BMPs were installed in the Dreisbach and Smith Fry watersheds, including field borders, parallel terraces, grassed waterways, and grade stabilization structures in the early 1970's. A modeling framework was developed to represent the BMPs with the Soil and Water Assessment Tool (SWAT) model and evaluate their long-term impact (s) on the water quality of the study watersheds. First, the soundness of various components of the SWAT model was evaluated through peer review. SWAT has been widely used for streamflow, sediment, and nutrient simulations for a variety of watersheds of different sizes (5-100,000 km^2) throughout the world. Second, a certain level of credibility for model computations was established by calibration of model parameters for the study watersheds based on a set of observed data, and further confirming the validity of model simulations for another dataset. Based on the function of the BMPs and hydrologic and water quality processes that are modified by their implementation, corresponding model parameters were altered to encode the impact of the BMPs on flow, sediment, and nutrient simulations of the model. Finally, the calibrated model was used for comparison of two scenarios, scenario A and scenario B. Scenario A represented model predictions over 1971-2000 time period without inclusion of the BMPs, while scenario B reflected model predictions for the same period with BMPs. These scenarios were compared at a field scale as well as a watershed scale to evaluate the impact of the BMPs on sediment and nutrient yields.

Field borders and parallel terraces were installed on the upland areas and were intended to reduce sheet erosion from upland areas. Based on model predictions, implementation of these BMPs would reduce sediment and nutrient loads from the fields where they have been installed by nearly 50%. However, their impacts would not be felt at the outlet of the study watersheds, primarily because they have been installed to influence less than 2% of total area of the Dreisbach and Smith Fry watersheds.

Grassed waterways and grade stabilization structures would be the more effective BMPs at watershed scales. Comparison of scenarios A and B revealed that implementation of these BMPs would significantly reduce sediment and nutrient yields at the outlets of the watersheds. Under scenario A, the watersheds tended to behave like a supply-limited watershed, i.e., simulated sediment and nutrient loads from upland areas were less than estimated transport capacity of the channel network. Thus, the channel network would undergo bed and bank erosion. The transport capacity of the channel networks would be significantly lowered due to implementation of grassed waterways and grade stabilization structures. Under scenario B, the study watersheds would show the characteristics of a transport-limited watershed, i.e., simulated sediment and nutrient loads from upland areas would be more than estimated transport capacity of the channel network. Thus, channel deposition would be the overall dominant main channel process in the watersheds, indicating that the channel network would not contribute to the sediment and nutrient yields at the outlets. It was also observed that the grade stabilization structures that have been placed at the downstream portion of the channel network would be the most effective ones. This would imply that for maximum benefits, these BMPs should be placed as close upstream as possible to where the regulation will be imposed

In a watershed where channel degradation is the dominant main channel process, implementation of grassed waterways and grade stabilization structures would be highly successful in reducing sediment and nutrient loads, perhaps to the extent of converting a supply-limited watershed to a transport-limited one. Application of BMPs such as parallel terraces and field borders would be more successful for watersheds where upland areas are the dominant sources of sediments and nutrients.

7.2 Modeling Implications

Utility of a distributed-parameter watershed model for simulating sediments and nutrients was discussed in this study. Also, a process-based method for representation of Best Management Practices (BMPs) was developed. SWAT model was selected not only because the model has both sediment and nutrient components in addition to hydrologic components, but because of the model structure that allows representation of BMPs in a process-based fashion. Similar to other distributed-parameter models, SWAT subdivides the watershed into sub-units including subwatersheds and channel segments for computations. Further, subwatersheds are partitioned into hydrologic response units (HRUs) that are used for computation of runoff, sheet erosion, and nutrient loads from upland areas. Thus, representation of BMPs such as field borders and parallel terraces that are installed in a particular field to reduce runoff, sediment, and nutrient loads can be easily done within SWAT by altering appropriate model parameters for the corresponding HRU (field). These estimated loads are routed through the channel network that is divided into various segments for computation purposes. Subdivision of the channel segment into smaller segments provides the option for alteration of model parameters for the particular segments with BMPs such as grassed waterways and grade stabilization structures.

Evaluation of the performance of BMPs can be facilitated by utilizing distributed-parameter watershed models that partition the watershed into fields (HRUs) and channel segments for computations. In doing so, however, model computations are rendered subjective to the level of watershed discretization. The results of this study revealed that sediment and nutrient simulations of the SWAT model may be very sensitive to the number and size of subwatersheds as well as the drainage density of the channel network (drainage density of the channel network is defined as the ratio of length of channel network to the total watershed area). As a result, a proper assessment of the efficacy of the BMPs must be conducted in conjunction with multiple watershed discretization levels.

While size of subwatersheds influences sediment and nutrient loads from upland areas, drainage density of the channel network is important in computing these loads eroded from the bed and bank of the channel network. In Section 3.0, two indices i.e., Erosion Index and Area Index were recommended to be applied for estimation of a proper watershed discretization level for sheet erosion computations. These indices were derived based on the Modified Universal Soil Loss Equation (MUSLE), which is used by SWAT for estimation of sheet erosion from upland areas. The applicability of the two indices was confirmed for the Dreisbach and Smith Fry watersheds. It was concluded that in transport-limited watersheds where channel network does not contribute to the sediment and nutrient yields at the outlet, application of the Erosion Index and Area Index is likely adequate for obtaining a proper watershed discretization level. However, when the channel network contributes to the sediment and nutrient loads at the outlet, an accurate estimation of the length and characteristics of the channel network is required. Examination of the effect of watershed discretization on average slope of channel network was found useful for such an estimation.

7.3 Closing Remarks

A methodological framework for representation of BMPs with a watershed model was developed in this study. A watershed model was selected and calibrated for the study area, and was utilized for predicting the impact(s) of implementation of BMPs on water quality. Calibration procedure is often used for establishing credibility for simulations of a model. This common practice embraces the critical issue of non-uniqueness of the optimal (calibrated) set of model parameters. The hydrological and water quality processes that are represented by the model parameters may be affected by the choice of the calibrated parameter data set. More credibility in the developed methodology could be established by employing uncertainty techniques. The uncertainty of input parameters should be elicited and encoded in the modeling approach to provide a better understanding of the processes that control transport and fate of sediments and nutrients in a watershed for a comprehensive management and decision making.

In addition to including uncertainty of input parameters in the modeling approach, an accurate estimation of drainage density of the channel network is required. This problem is complex because drainage density varies with different storm events. Application of remote sensing techniques for extraction of the characteristics of the channel network from aerial photos and satellite images at the time of large storm events as well as low flow conditions would be helpful. Also, hydrologic and water quality monitoring programs at various locations of the channel network should be conducted for such purposes.

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