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Non-point source pollution modeling for Muddy Creek watershed.

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NON-POINT SOURCE POLLUTION MODELING FOR MUDDY CREEK WATERSHED

by

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ABSTRACT

Muddy Creek is a small agricultural watershed located in Essex County, in the south-western part of Ontario between Lake Erie and Lake St. Clair. Muddy Creek delivers runoff to Wheatley Harbour, which is identified as one of the Areas of Concern (AOC). Sediments and nutrients from agricultural runoff have been identified as environmental concerns in Wheatley Harbour. In this study, the Annualized AGricultural NonPoint Source (AnnAGNPS) model was used to estimate the loadings and their contributing areas of surface runoff, sediment and nutrients from Muddy Creek watershed. Simulated runoff volume is found to be within the acceptable limit and the sediment loading is fairly comparable to the loadings produced from the watersheds with similar nature found in the literature.

Due to lack of observed data, the model could not be calibrated. However, a sensitivity analysis is performed to increase the confidence in model predictions and to improve the understanding of the model behaviour. Sensitivity analysis showed that sediment yield is highly sensitive to the scale of cell discretization.

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I would finally like to share this piece of work with the members of my family.

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GLOSSARY OF ACRONYMS

AGNPS	Agricultural NonPoint Source
AGFLOW	AGricultural watershed FLOWnet generation program
AnnAGNPS	Annualized Agricultural NonPoint Source
ANSWERS	Areal Nonpoint Source Watershed Environment Response Simulation
ANSWERS	Areal Nonpoint Source Watershed Environment Response Simulation
AOC	Area of Concern
ARS	Agricultural Research Service
BMPs	Best Management Practices
CN	Curve Number
CSA	Critical Source Area
DEM	Digital Elevation Model
DWSM	Dynamic Watershed Simulation Model
ERCA	Essex Region Conservation Authority
GEM	Generation of weather Elements for Multiple applications computer program
GIS	Geographic Information System
HRU	Hydrologic Response Unit
HSPF	Hydrological Simulation Program- Fortran
HUSLE	Hydro-geomorphic Universal Soil Loss Equation
MOE	Ministry of the Environment
MSCL	Minimum Source Channel Length
NPS	Non-Point Source
NRCS	Natural Resources Conservation Authority
OMAFRA	Ontario Ministry of Agriculture, Food and Rural Affairs
PWQMN	Provincial Water Quality Monitoring Network
RAP	Remedial Action Plan
RASFOR	RASter FORmatting computer program
RASPRO	RASter PROperties computer program
RCN	Runoff Curve Number

RUSLE	Revised Universal Soil Loss Equation
SCS	Soil Conservation Service
SWAT	Soil and Water Assessment Tool
SWRRB	Simulator for Water Resources in Rural Basins
TOPAGNPS	A computer model which is a subset of TOPAZ written for AGNPS
TOPAZ	Topographic Parameterization computer model
TR 55	Technical Release 55
TRCA	Toronto Region Conservation Authority
USDA	United States Department of Agriculture.
USDA-ARS	USDA Agricultural Research Service
USEPA	U.S. Environmental Protection Agency
VFS	Vegetative filter strips

CHAPTER 1

INTRODUCTION

Nonpoint source water pollution causes widespread water quality problems worldwide. The U.S. Environmental Protection Agency (USEPA) has concluded that the risks of nonpoint source pollution are much greater than the risks posed by point sources (USEPA, 2000). Agriculture is the largest contributor of nonpoint source pollutants in US, affecting 50-70% of the waters assessed by the USEPA (USEPA, 2000). Agricultural practices such as tilling make soil available to flow overland with storm waters, causing soil erosion. Soil erosion not only influences soil fertility that reduce crop productivity, but also adds sediments and degrades the quality of receiving waters. Sediments also act as a carrier and a storage agent for other kinds of pollutants such as phosphorous, nitrogen and organic compounds. Watershed models can be used in the assessment of these pollutant loads and to identify specific areas that have high potential for pollutants.

1.1 STATEMENT OF THE PROBLEM

Muddy Creek watershed is located in Essex County, in the southwestern part of Ontario between Lake Erie and Lake St. Clair. It is predominantly an agricultural watershed with a watershed area of about 8.4 km^2. Muddy Creek delivers runoff to Wheatley Harbour, which is a small, confined harbour on the north shore of Lake Erie.

Sediments and nutrients carried by the agricultural runoff are identified as the major source of Non-Point Source (NPS) pollution particularly in agricultural watersheds, such as the Muddy Creek. NPS pollution can originate from several locations in an agricultural watershed as opposed to point sources that are easily identifiable due to its localized nature. Point source discharges affect the water quality conditions during times of low flow when there is less water to dilute incoming effluents. NPS contribute pollutant loads that are washed off and transported mainly during precipitation events when higher surface runoff exist. NPS pollutants that accumulate in water bodies create water quality problems and make several beneficial use impairments.

In the 1980's, Wheatley Harbour was identified as an Area of Concern (AOC), the only one on the Canadian shores of Lake Erie. The Wheatley Harbour AOC encompasses the harbour area and the adjacent Muddy Creek watershed. Under the Great Lakes Water Quality Agreement, the governments of Canada and the United States have agreed to develop Remedial Action Plans (RAPs) for AOCs.

Water quality levels in Muddy Creek and downstream water body have been degraded due to NPS pollutants from Muddy Creek watershed. The total phosphorus concentrations in the sediments and in the waters in Wheatley Harbour have been found to exceed provincial guidelines. The agricultural runoff from Muddy Creek is identified as the main source of sediment and nutrients (Environment Canada, 2005).

RAP process has identified the following beneficial use impairments in Wheatley Harbour AOC.

- Restrictions on specific kinds of fish and wildlife consumption
- Degradation of fish and wildlife populations
- Restrictions on dredging activities
- Undesirable algae formation
- Loss of fish and wildlife habitat

1.2 OBJECTIVES OF THE STUDY

The overall objective of this study is to model the hydrological processes and assess the impact of land management practices on water quality and quantity of the Muddy Creek watershed using Annualized AGricultural Non-Point Source (AnnAGNPS) Pollutant Loading (PL) model.

The specific objectives of the study are to:

- Investigate the adaptability of AnnAGNPS model for the watersheds in Essex region.
- Identify the areas susceptible to soil erosion within the watershed and estimate sediment loading in an effort to prioritize the subwatersheds for treatment/management.
- Assess agricultural nutrient loadings that are responsible for water quality degradation.

1.3 STRUCTURE OF THE THESIS

This thesis report is mainly divided into six chapters to present the key tasks that are required to perform this simulation study.

Chapter 1 includes an introduction to the existing problem in the Muddy Creek watershed and describes the specific objectives of the study.

Chapter 2 presents a literature review on the topics that are relevant to the study. This includes an introduction to non-point source pollution, watershed models that are mostly used for non-point source pollution modeling, AnnAGNPS model applications and sensitivity of the model. It also includes an introduction to the agricultural Best Management Practices that are used to minimize the impacts of NPS pollutants on surface water quality.

Chapter 3 presents a review of the AnnAGNPS model. Some of the concepts and major processes within the AnnAGNPS model are reviewed in this chapter.

Chapter 4 describes, the development of the computer-based model to represent the Muddy Creek watershed. This includes an overview of the watershed, modules used and methodology adopted in developing the necessary input database for the model. This chapter further describes the parameter selection process conducted in the study and the methodology used in the modeling of agricultural best management practices.

Chapter 5 presents the results and review of various simulations performed for sensitivity analysis, assessing model loadings and their contributing areas and the effectiveness of best management practices.

Chapter 6 presents the conclusions derived from the results of the simulation study and recommendations for further actions with a view to monitor non-point source pollution within the Muddy Creek watershed.

CHAPTER 2

LITERATURE REVIEW

2.1 NON-POINT SOURCE POLLUTION

Non-point source (NPS) pollution is the introduction of pollutants into a system through an indirect or unidentified route. It is generated in a variety of land uses including agriculture, construction and forestry practices. NPS pollution is often associated with precipitation events, snowmelts or irrigated land activities that generate surface runoff, pick up pollutants and deposit them into water bodies. The term non-point source is used to distinguish it from point source pollution, which comes from localized, easily identifiable sources such as sewage treatment plants or industrial facilities.

NPS pollutants have particular characteristics that separate them from point source pollutants. They are (Ambrosio et al., 2001 and Leeds et al., 1994):

- NPS pollutants enter receiving waters in a dissipated manner at intermittent intervals mostly due to variations in meteorological conditions
- NPS pollution occurs over a large area of land and moves overland before reaching waterways
- NPS pollution sources are difficult to monitor at the point of origin
- The effect of NPS pollution may be minor when considered individually but will be significant when considered collectively
- The extent and significance of NPS pollution is related to parameters that are uncontrollable by humans such as climatic conditions (storm intensity and frequency) and geographical conditions (soil type, erodibility).

NPS pollution has been associated with water quality standard violations and the contamination of aquatic ecosystems that lead to unsafe drinking water, destroyed habitat, fish kills, property loss and many other environmental and human health problems (Ambrosio et al., 2001). According to the 3rd World Water Forum – 2003, every year at least five million people die from water related diseases worldwide (Pollution Probe, 2006).

A report (MOE, 2004), indicated that "Water treatment alone cannot ensure that we can meet our needs for good quality water. Even with the best water treatment technology money can buy, a community is at risk if it relies on a water source that is susceptible to contamination." Once water sources get contaminated, it is often very difficult to remove them, because some of the contaminants may stay for decades or even centuries (MOE, 2004).

National Water Quality Inventory Report to US Congress - 2000, indicated that 45% of the surveyed waters in the United States were contaminated with pollutants such as sediment, nutrients, bacteria and metals. Primary sources of impairment were identified as runoff from agricultural lands, municipal point sources and hydrologic modifications such as channelization and dredging (USEPA, 2002). A study on economic impact of erosion on surface waters (Osterkamp et al., 1998) indicated that annual costs of damages due to physical, chemical and biological sediment discharge in North America exceed $16 billion.

2.2 SOIL EROSION PROCESS

Soil erosion and sediment delivery are the key processes controlling NPS pollution in agricultural watersheds. Furthermore, erosion reduces production potential by removing nutrients needed for crop production, deteriorates the soil structure by deposition and increases flood hazards by reducing the infiltration rate and water holding capacity of the soil. Sedimentation reduces capacity of downstream channels and waterbodies, destroys fish and wild life habitat and increases cost of maintaining downstream waterbodies such as harbours.

During precipitation events, the energy from the impact of raindrops and the shear force of water flowing over the land surface causes detachment of soil particles. The detached soil particles with nutrients, such as nitrogen and phosphorous attached to them, are transported to waterways by the surface runoff while a portion of the detached soil particles deposit in the field before reaching streams. Sediment transport is a function of rainfall intensity, sediment characteristics and hydraulic parameters. Renard et al., (1997) noted that sediment transport is largely a function of topography and runoff velocity while deposition is a function of runoff velocity and sediment particle sizes.

Erosion occurred by water is generally recognized in three different forms: sheet and rill erosion, stream bank erosion and stream bed erosion. Sheet erosion is a process in which soil detached by rainfall energy, is moved across the soil surface by sheet flow, often in the early stages of runoff. Rill erosion occurs as runoff water begins to concentrate in small channels or streamlets. Inter-rill erosion takes place between rills. Sheet and interrill erosion can go unnoticed because it removes sediment in a uniform layer. For a susceptible soil, rill erosion is immediately visible since flow concentrates in many small streamlets or rills.

Sheet and rill erosion carry mostly fine-textured, small particles and aggregates. These sediments will contain higher proportions of nutrients, pesticides, or other adsorbed pollutants than are contained in the surface soil as a whole. Sheet and rill erosion are generally active only during or immediately after rainstorms or snowmelt.

2.3 WATERSHED MODELS

As non-point source pollution (NPS) problem has gained more attention throughout the world in recent years, various methods have been developed to evaluate the magnitude and extent of NPS pollution. Most of these methods involved development of computer-based models to analyze stormwater quality and quantity in watersheds. Generally, models are simplified mathematical representations of real systems or processes that can be used for simulations or predictions. Watershed models simulate the generation and movement of stormwater runoff as well as the pollutants it carries from the source areas to downstream waterbodies. NPS pollution models can be used in estimating loadings of chemicals, sediments and nutrients that degrade water quality, establishing critical source areas and ranking alternative measures. Hence they form effective tools for watershed planning and management (Novotny, 2003). Use of these models have been limited due to difficulties involved in simulating large areas having heterogeneous properties such as land use, land cover, soils, and topography and gathering large amounts of input data. Linking models with GIS technology, which has the capability to handle large volumes of spatial and non-spatial data, has helped in overcoming many of these difficulties.

2.3.1 Classification of models

Models are generally classified based on their functionality, method by which inputs and outputs are manipulated and whether they simulate single event or continuous processes. There are two general types of models, physical or analogue models and mathematical models. Physical models are simpler physical representations of complex systems that are assumed to have similar properties to the prototype system or reduced-dimension representation of real world system. Diskin (1970) stated that the "mathematical models are simplified systems that are used to represent real-life systems and may be substitutes of the real systems for certain purposes." According to Woolhiser and Brakensiek (1982), mathematical model is a "symbolic, usually a mathematical representation of an idealized situation that has the important structural properties of the real system." Mathematical models can be subdivided into analytical and numerical models. Analytical models provide a direct solution of the governing equations for homogeneous systems. Numerical models simulate more complex systems by solving the governing equations that represent the physical processes approximately (Woolhiser and Brakensiek, 1982).

Models can also be classified as deterministic or stochastic depending on the character of the model outputs. If all the input data, parameters and processes are considered free of random variation and known with certainty, then the model is referred to as a deterministic model. Deterministic hydrologic simulation models are well established and widely used in watershed management activities. Stochastic models have the capability of representing the random variability of input parameters, whereby known probability distributions are used to determine statistical probabilities of output parameters.

According to the degree of spatial variability that is represented and simulated, models may be classified as lumped parameter or distributed parameter models. Lumped parameter models ignore spatial variability of land uses, soil types, and other land surface properties within a computational unit. Effective parameter values are typically estimated based on area weighted averages. A distributed parameter model is one in which the spatial variations of characteristics are considered explicitly. In this approach, the watershed is divided into relatively smaller elements that may be considered as homogeneous units or cells. Each unit is modeled separately and the output is obtained by routing the flow or loadings from cell to cell. In these models, changes in the watershed and their effects on the output can be modeled effectively. During the past decade, distributed parameter models are linked with GIS in reducing manual data input requirements.

Hydrologic and water quality models account for water, sediment and chemical transport through watersheds. NPS models are concerned with generation, transport and tracking of pollutant loadings such as sediment, nutrients and chemicals into waterways. These models typically simulate either on an event basis or on a continuous basis. An event model simulates over a single storm that may range from a few hours to few days. Continuous simulation models are useful in analyzing long-term effects of hydrological changes and watershed management practices since they take into consideration of both during and between precipitation events.

2.3.2 NPS pollution models

To model NPS pollutant loading from a watershed, a model of sufficient complexity to simulate the diffuse nature of NPS pollution is required. During past three decades, several watershed-scale, hydrologic and NPS pollution models have been developed. These models cover a large range of complexity depending on the extent to which hydrologic, sediment erosion and chemical processes are modeled in a mechanistic manner or based on empirical procedures (Donigian and Huber, 1991). These models employ wide ranges of techniques, from simple annual loading functions to detailed process simulation models. Most of the commonly used models were developed in 1970s and 1980s. Since 1980s and early 1990s, most of the research in model development was on integration of geographic information systems (GIS) and remote sensing data and development of graphical user interfaces (Borah and Bera, 2003).

It is necessary to have a clear understanding of an appropriate model for an application and for a certain watershed. Parsons et al. (2001) noted that if a model is to be truly practical and applicable for the purpose, a potential user has to understand: the original purpose of the model, under what conditions the model will perform correctly, the accuracy that can be expected under the best conditions and the limitations of the model. The advantages and limitations of three distributed parameter models: AnnAGNPS, ANSWERS-2000 and SWAT and a lumped parameter model HSPF are discussed below.

2.3.3 AnnAGNPS pollutant loading model

AnnAGNPS is a continuous simulation, multi event modification of single event model AGNPS, which was first developed in the early 1980's by the Agricultural Research Service (ARS) in cooperation with Natural Resources Conservation Authority (NRCS) (Bingner and Theurer, 2003). AnnAGNPS was first released in 1998 and is intended to be used as a tool to evaluate non-point source pollution from agricultural watersheds ranging in size up to 300,000 ha. In AnnAGNPS, watershed is subdivided into homogeneous land areas (cells) based on land use, soil type and land management. A separate Window based flow network generator using DEMs can be used to subdivide the watershed into hydrologically derived cells of different shapes. The model can be used to examine current conditions in a watershed or to compare effects of different conservation alternatives within a watershed (Bingner and Theurer, 2003).

AnnAGNPS calculations are performed on a daily time step. AnnAGNPS simulates water, sediment, nutrient and pesticide transport at the cell and watershed level. Each day the applied water and resulting runoff are routed through the watershed system before the next day is considered. Runoff is calculated using SCS Runoff Curve Number (RCN) equation where the curve numbers are modified daily, based upon soil moisture, crop stages and tillage operations. Separate input files for watershed data and simulation period climate data allows for quick changing of climate data. Overland sheet and rill erosion of sediment for each cell is determined using RUSLE (Renard et al., 1997).

Peak flow calculations are performed using TR-55 graphical peak discharge method. Sediment routing is calculated based upon transport capacity relationships using the Bagnold stream power equation (Bingner and Theurer, 2003). A daily mass balance for Nitrogen (N), phosphorous (P) and organic carbon (C) is calculated for each cell. Nutrients and pesticides are subdivided into soluble and sediment attached components for routing. Each nutrient component is decayed based upon the reach travel time, water temperature, and an appropriate decay constant (Theurer and Bingner, 2005).

The limitations of the model are identified as: all runoff and associated sediment, nutrient and pesticide loads for a single day are routed to the watershed outlet before the next day's simulation begins regardless of how many days this may actually take. Also point sources are limited to constant loading rates for entire simulation period.

2.3.4 ANSWERS-2000

ANSWERS (Areal Non-point Source Watershed Environment Response Simulation) - 2000 is a continuous simulation, distributed parameter, physically-based model developed in mid 1990s for evaluating the effectiveness of agricultural and urban BMPs in reducing sediment and nutrient delivery to streams in surface runoff. The original ANSWERS was developed in late 1970s as an event-based, distributed parameter model (Dillaha et al., 2001).

The model divides the area simulated into a uniform, square grid cells. Typical cell sizes range from 0.4 to 1 ha. with smaller cells providing more accurate simulations. Within a cell, all properties such as soil, land use and management are assumed homogeneous. The model can simulate BMPs such as conservation tillage, ponds, grassed waterways and tile drainage. The model simulates soil detachment, transport and deposition. N and P are simulated using correlation relationship between chemical concentration, sediment yield and runoff volume (Zhen, 2002).

Limitations of the model are identified as (Parsons et al., 2001): the sediment detachment submodel is empirical and out of date, the model does not simulate snow pack and melt and is thus not suitable for use in areas with significant winter snow accumulation and snowmelt.

2.3.5 SWAT model

Soil and Water Assessment Tool (SWAT) is a continuous, distributed parameter river basin or watershed scale model developed by Agricultural Research Services of US Department of Agriculture (Neitsch et al. 2002). Model can be used to predict the impact of land management practices on water, sediment and agricultural pollutants in large complex watersheds. The model emerged mainly from SWRRB (Simulator for Water Resources in Rural Basins) (Arnold et al., 1993) and inherits features from CREAMS (Chemicals, Runoff and Erosion from Agricultural Management Systems), GLEAMS (Groundwater Loading Effects on Agricultural Management Systems), and EPIC (Erosion – Productivity Impact Calculator) models (Borah and Bera, 2003). SWAT model divides the watershed into a number of subwatersheds or subbasins called Hydrologic Response Units (HRU's). HRU's are lumped land areas having unique land cover, soil and management combinations.

The model uses a Geographic Information System (GIS) interface to facilitate the automatic development of input parameters that are required to operate the model (Srinivasan and Arnold, 1994). SWAT model has hydrology, sediment, chemical and microbiological components and uses SCS runoff curve number and other empirical relationships to compute runoff volumes and peak flows. Erosion caused by rainfall and runoff is computed using Modified Universal Soil Loss Equation (MUSLE), a modified version of USLE, developed by Wischmeier and Smith (1978). SWAT does not make a distinction between sediment originating over the landscape (sheet, rill and gully erosion) and sediment originating within the stream system (bed and bank erosion). This may result in attributing all of the sediment load to the USLE parameters which are only related to sheet and rill erosion and considered as a serious deficiency with the model (Bingner et al., 2005).

2.3.6 HSPF model

Hydrological Simulation Program - FORTRAN (HSPF), developed by USEPA is a one-dimensional stream network, lumped parameter, continuous simulation model that can simulate watershed hydrology and water quality for both conventional and toxic organic pollutants. HSPF produces a time history of the runoff flow rate, sediment load, and nutrient and pesticide concentrations, along with a time history of water quantity and quality at any point in a watershed (Donigian and Huber, 1991). HSPF simulates three sediment types (sand, silt, and clay) in addition to a single organic chemical and transformation products of that chemical.

HSPF is originally developed from the Stanford watershed model. HSPF needs extensive input data and requires considerable effort when applied to a watershed and is not user friendly. In HSPF, overland flow is treated as a turbulent flow process and simulated using the Chezy-Manning equation. Reach routing is performed by using kinetic wave method. A number of pollutant transport processes can be modeled including chemical partitioning, hydrolysis and volatization. Since HSPF performs simulations on a lumped parameter basis, magnitudes of parameters are to be determined by calibration and hence model requires considerable amount of time for calibration.

2.3.7 Model selection

In 1993, Natural Resource Conservation Service (NRCS) completed a study of thirty-eight available water quality models. Two watershed scale, agricultural non-point source pollution models: AGNPS (precursor to AnnAGNPS) and SWRRB (precursor to SWAT) were selected for further analysis. AGNPS contained US Department of Agriculture, NRCS approved science and it was selected for further development (Bingner et al., 2005).

The processes such as gully erosion and tile drains in agricultural fields are unique processes in AnnAGNPS pollutant loading model (Bingner et al., 2005). Model's capability to display pollutant loadings in GIS environment is also advantageous. For the current study, AnnAGNPS pollutant loading model was selected on the basis of its capability of identifying and evaluating the sources of water, sediment and nutrient within the watershed.

2.4 AGNPS/AnnAGNPS MODEL APPLICATIONS

AGNPS has been used as a simulation model for prediction of non-point source pollution, studying the impacts of land use management on water quality and assessment of BMPs for more than a decade, in United States and in several other countries.

Bhuyan et al. (2003) applied the model to assess nutrient loadings from five watersheds in Kansas, US and concluded that the model is useful as a decision support system for resource managers. Since AGNPS assumes a uniform rainfall over the entire watershed, which is certainly not the case in a large watershed, they recommended that the large watershds be divided into smaller sub-watersheds to increase estimation and prediction accuracy. Ma et al. (2002) applied AGNPS to a watershed in Kalamazoo, Michigan in the US to evaluate the parameters that are most sensitive for phosphorous sediment loading. Their results showed that phosphorus sediment is the most sensitive to soil texture while sediment loss is the most sensitive to the SCS curve number.

Vennix and Northcott (2004) applied the AGNPS model to prioritize vegetative buffer strips placement on reducing sediment loading in an agricultural watershed in Michigan, US. Results of this study may be helpful for watershed managers to implement vegetative buffer strips in site-specific areas within the watershed to employ efficient implementation of conservation management programs. Smith (2002) applied AGNPS model to identify critical source areas of sediment and nutrient runoff in a watershed in South Dakota, US. Also the model was used to develop management alternatives to reduce sediment and nutrient loads. Sugiharto et al. (1994) applied the AGNPS to evaluate twenty different management practices in reducing sediment and phosphorus yields. Kausman and Mitchell (1997) applied AGNPS to assess erosion and sediment in Indonesian watersheds. They found that AGNPS model simulations gave realistic results for erosion rates and sediment yields. Grunwald and Norton (1999) applied AGNPS model to predict runoff and sediment yield in two small-ungaged watersheds in Germany. Results indicated that there was considerable under and over prediction of surface runoff and sediment yield. The model has been applied in many countries in different circumstances to estimate surface runoff, soil erosion, sediment yield and nutrient loading due to non-point source pollution (Lenzi and Luzio, 1997; Mohammed et al. 2004).

Mitchell et al. (1993) evaluated the suitability of the model for predicting surface runoff and sediment yield from watersheds in Illinois, US. Results indicated that variation of runoff between modeled and observed data was reasonable but sediment yield varied about five times the observed data. Despite these discrepancies, it was concluded that the model is a valuable tool for water quality management. Pekarova et al. (1999) tested the possibility of applying AGNPS model in Slovakia and simulated runoff and nutrient loading in surface water in two experimental watersheds. Comparison of results indicated that the AGNPS was suitable to model runoff and nutrient loading with sufficient accuracy.

AGNPS model has been tested and validated for Ontario watersheds. Leon et al. (2003) applied the model for a southern Ontario watershed to validate the model for nutrients and runoff. The results showed that the model is well suited for applications in Southern Ontario. Perrone and Madramootoo (1997) applied AGNPS to a watershed in Quebec to determine its predictive capability with respect to surface runoff, peak flow, and sediment yield. Booty et al. (2005) used AGNPS to study surface water quality conditions during dry and wet weather in a watershed in Ontario. Results of the study may be used to establish a methodology for assessing the sensitivity of water quality to the climatic changes.

AnnAGNPS model has had fewer applications. Baginska et al. (2003) applied AnnAGNPS in a watershed in Sydney, Australia to model nutrient transport. Bingner et al. (2005) applied AnnAGNPS to Upper Auglaize watershed in Ohio, US for assessing and reducing pollution from agricultural runoff and other non-point sources. AnnAGNPS has been applied in a watershed in Mississippi, US to estimate sediment yields to develop water quality targets (Simon et al., 2002). Results showed that flows and sediment loads estimated from the AnnAGNPS were in close agreement with the measured data. AnnAGNPS model has been applied in a Southern Ontario watershed (Das et al., 2006) to evaluate the hydrology and sediment loadings from non-point sources. Model results showed that simulated runoff was under predicted and sediment yield was over predicted. Their calibration and validation results showed that the AnnAGNPS model is capable of simulating the runoff and sediment yield fairly well for a cold and temperate region like Ontario.

2.5 MODEL SENSITIVITY

Extensive number of input parameter requirement is a well-known problem that exists in hydrological models, especially with distributed models. They contain hundreds of parameters that represent hydrologic and water quality processes in watersheds. Due to spatial variability and some other constraints, model input parameters always contain uncertainty to some extent. Therefore it is important to understand the input parameters that are sensitive to model output and will be beneficial in model development and application. This knowledge will result in better estimated values and thus reducing model uncertainty.

Sensitivity analysis is a formalized procedure that can be employed to identify parameters that have a significant influence on model simulation results. Those parameters, to which the model output is highly sensitive, require a special care in their estimation in order to produce reliable conclusions. Newham et al. (2003) stated that sensitivity analysis might be used to increase the confidence in a model and its predictions. Sensitivity analysis is an important part of the model validation, which draws attention to those components where further research and development should be focused to enhance model performance. Though several researchers have accepted the importance of sensitivity analysis, there is no single, well accepted procedure for sensitivity analysis.

Vieux and Needham (1993) investigated the effects of grid cell size on sediment yield predictions. Results indicate that grid cell sizes are the most important factor affecting sediment yield. Sediment yield is mostly dependent on flow-path length and as the grid cell sizes increase, stream meanders get shortened and cause increased sediment yield. Therefore, cell sizes should not be selected arbitrarily and should be based on a scale appropriate for capturing the spatial variability of the watershed. Qiu et al. (1997) reported AGNPS outputs with two different cell resolution (100x100 meter and 200x200 meter). Results showed that cell resolution did not cause significant difference in estimating the soil loss. However, estimated nutrient loading was 20 percent higher in the 200 meter cell discretization than the 100 meter cell discretization.

Baginska et al. (2003) applied AnnAGNPS model to predict export of nitrogen and phosphorous from a small experimental watershed in Sydney, Australia. They applied a model independent, nonlinear parameter estimation code PEST for sensitivity testing and to assess the relative importance of key parameters of the model. Results indicated that predicted phosphorous loads had a high level of sensitivity to assigned pH values for topsoil and also increase of pH value by one unit resulted in up to 34% increase in model generated particulate phosphorus load. Yuan et al. (2003) applied the AnnAGNPS model to a watershed in Mississippi, US and performed a sensitivity analysis on parameters that are critical to nitrogen loading. Their study revealed that initial nitrogen concentration in the soil and crop nitrogen uptake had the most impact on the nitrogen loadings.

Yuan et al. (2003) performed a sensitivity analysis to identify parameters that most significantly affect on N loading in an agricultural watershed in Mississippi, US. Yuan et al. (2003) concluded that, initial organic and inorganic N concentrations in the soil and crop N uptake had the most impact on N loadings.

Fisher et al. (1997) pointed two types of sensitivity analysis that are recognized in the watershed modeling. The first method, attribute sensitivity analysis, is widely used in the model validation and is commonly called, sensitivity analysis. This method is used to examine percentage change in model output results by changing input parameters. This is performed by systematically changing individual input parameters one at a time by some constant percentage and percentage variation of output is observed. There are several limitations in this method. The second type, resolution sensitivity analysis, is appropriate to any distributed model where the parameters are being sampled over space or time. In this method, sensitivity of a model is assessed by varying the sampling interval of input parameters in geographical space. Input spatial data are subjected to random mixing spatially, to varying degrees, such that the organized landscape become disorganized. They applied spatial sensitivity analysis for two models AGNPS and ANSWERS. The results showed that model outputs are insensitive to the spatial pattern of single input variables and the most of output variables showed absolutely no change as a result of the mixing. Only infiltration-related inputs produced variations in sediment and nutrient yield output.

2.6 AGRICULTURAL BEST MANAGEMENT PRACTICES

NPS pollution is often associated with precipitation events, snowmelts or irrigated land activities that generate surface runoffs, pick up pollutants and deposit them into water bodies. The primary NPS pollutants that move with agricultural surface water runoff are sediments and nutrients. Best Management Practices (BMPs) are pollution control measures or practices selected and implemented to minimize the impacts of these NPS pollutants on surface water quality (Novotny, 2003).

BMPs can be categorized as structural or nonstructural practices, or a combination of practices, designed to act as effective and practicable means to minimize soil and nutrient loss from agricultural fields. Structural BMPs are practices that involve construction of facilities that typically capture runoff, detain it until sediment and nutrients are settled out or be filtered through the underlying soil. Structural practices can be vegetative, such as, soil bioengineering techniques, or non-vegetative, such as, riprap or gabions (USEPA, 2005). The most commonly used structural agricultural BMPs are contour buffer strips, vegetative filter strips along waterways, contouring, terracing, and sediment basins. Nonstructural practices usually involve changes in activities or behavior and focus on controlling pollutants by reducing the generation of pollutants at their source. Some of the most commonly used nonstructural agricultural BMPs are conservation tillage, fertilizer and pesticide management, rotational grazing and residue management. These structural and nonstructural agricultural BMPs can be used individually or in combination to control specific pollutant sources and for particular sites to meet water quality goals.

Vennix and Northcott (2004) studied the effectiveness of vegetative buffer strips on reducing sediment load in the East Bad Creek watershed located in Michigan, US. There results indicated that incorporation of buffer strips along the stream segments reduced sediment load at watershed outlet by 17%. A study carried out by South Dakota Department of Environment and Natural Resources in 2004, indicated that reduction of total phosphorus loading at watershed outlet could be achieved through standard BMPs such as conservation tillage, decreased fertilization rates, and installation of buffer strips. Gaynor and Findlay (1995) reported that although conservation tillage reduced soil loss by 49%, concentrations of soluble phosphorus increased by 2.2 times. Overall it appeared that conservation tillage reduces total phosphorus although the soluble phosphorus delivery increases.

Results of field research conducted in Virginia, US, using grass as the buffer/filter strip material, indicate that buffer strips are very effective in removing sediment from runoff, with an average reduction ranging from 56 to 95 percent (Leeds et al., 1994). This reduction in sediment removal mainly depends on soil characteristics, field slope, runoff conditions and width of the filter strip. For buffer strips wider than 30 feet, no improvement in filter effectiveness has been observed. The results for nitrogen and phosphorus removal indicated that buffer strips are not as much effective in removing N and P. Results for P removal ranged from 0 to 83 percent.

2.7 SUMMARY

Hydrologic models have become essential tools for estimating and managing NPS pollution. Model complexities depend on the extent to which hydrologic, sediment erosion and chemical processes modeled by these models. Among the available hydrologic models, AnnAGNPS has proven reliable in predicting NPS pollution, studying the impacts of land use management on water quality and assessment of BMPs. A general review of AnnAGNPS model and its processes are presented in the following chapter.

CHAPTER 3

REVIEW OF THE AnnAGNPS MODEL

3.1 ANNAGNPS MODEL REVIEW

AnnAGNPS is a continuous, distributed parameter watershed model developed by US Department of Agriculture (Bingner and Theurer, 2005). This model was developed based on the original single event model AGricultural Non-Point Source (AGNPS) (Young et al., 1989) developed by Agricultural Research Services (ARS) and NRCS to predict non-point source pollutant loadings within agricultural watersheds. AnnAGNPS is suitable to simulate long term sediment and chemical transport from ungaged agricultural watersheds. Source accounting function is one of the distinctive features of the model. AnnAGNPS can generate loadings for the entire simulation period at user defined locations and calculate the contribution of each location as a ratio to the loadings at watershed outlet. This feature is particularly useful in identifying critical areas in a watershed and can be used to assist in determining BMPs and for risk and cost/benefit analyses (Bingner and Theurer, 2005).

3.2 ANNAGNPS MODEL STRUCTURE

A number of modules that are supplied with AnnAGNPS model can be used in the preparation of AnnAGNPS database. The input output structure and the suite of modeling components contained within AnnAGNPS model are shown in Figure 3.1. The basic modeling components are hydrology, sediment, nutrient and pesticide transport. The model requires physical parameters of the watershed, soil data, climate data, land use and management data.

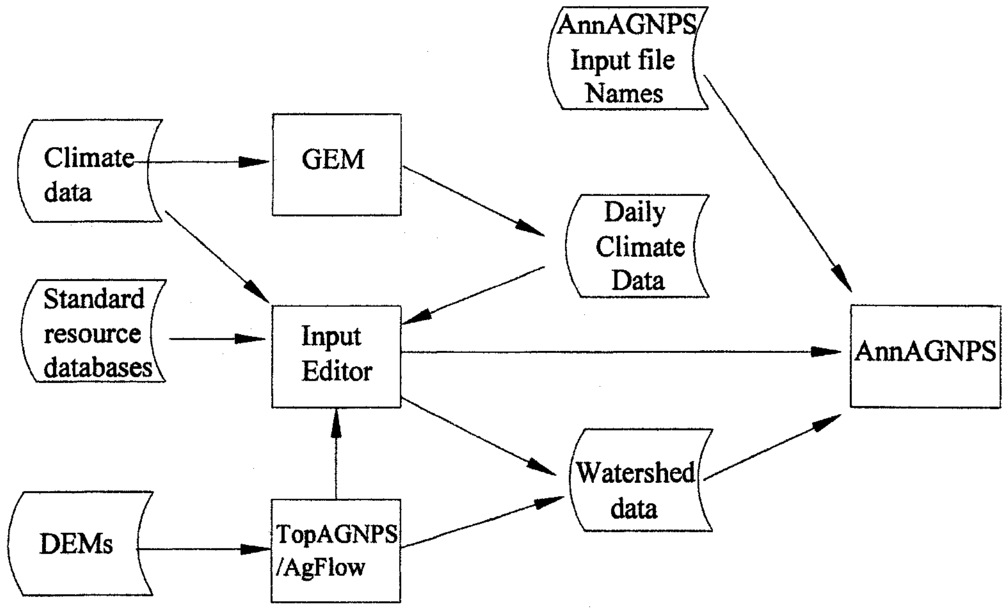


Figure 3.1: Input-Output structure of the AnnAGNPS model (adapted from Bingner and Theurer, 2005)

The physical parameters of the watershed such as cell and stream network information can be extracted from watershed digital elevation models (DEMs) using TOPAGNPS. AGFLOW is used to determine the topographic related input parameters for AnnAGNPS and to format the TOPAGNPS output in the form needed by AnnAGNPS (Bingner and Theurer, 2005). These physical parameters of the watershed are held constant throughout the simulation period. Climate data can be either generated using the Generation of Weather Elements for Multiple Applications (GEM) program or can be generated manually using historical data. Graphical input editor assists in developing the AnnAGNPS database. Seasonal data will change according to human activities and will rarely change during a season such as soil, land use and management data are typically specified manually for each cell using AnnAGNPS input editor. AnnAGNPS-ArcView interface program has been utilized to facilitate TOPAGNPS and AGFLOW programs and to export data to AnnAGNPS input editor. Output processor has been used to analyze the results from AnnAGNPS by generating summary of the results in tabular or GIS format (Bingner and Theurer, 2005).

3.3 AnnAGNPS THEORY

3.3.1 Watershed concept

The spatial variability within a watershed is accounted for by dividing the watershed into many homogeneous drainage areas referred to as "cells." Each cell is homogeneous in soil type, land use and land management and represents the landscape within its respective drainage area boundary. A daily soil water balance is maintained, so runoff can be determined when a precipitation event that includes rainfall, snowmelt or irrigation application occurs (Bingner and Theurer, 2005). Simulated drainage areas are integrated together by a network of "reaches" which collectively represent the stream system in the watershed. Basic AnnAGNPS watershed concept is illustrated in Figure 3.2.

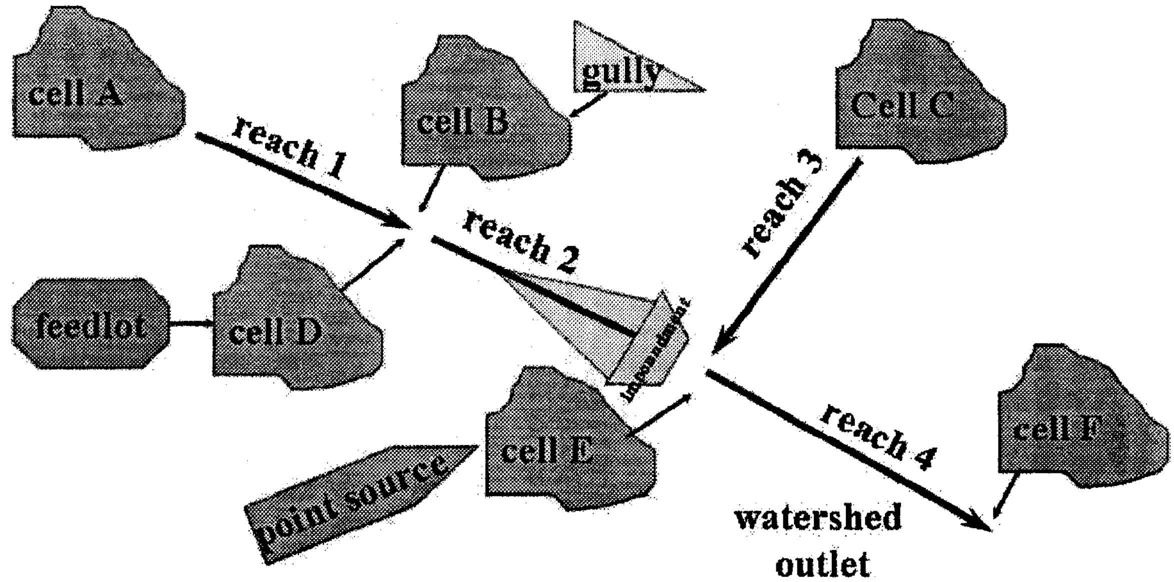


Figure 3.2: AnnAGNPS watershed concept (adapted from Bingner and Theurer, 2005)

3.3.2 Major processes

a. Water Balance

The hydrology component of the model is based on a water balance equation. A schematic of water balance is shown in Fig. 3.3. The erosion calculation is based on whether there has been any runoff for each day. The amount of soil moisture is used to determine the effect of the SCS curve number and is thus the basis for the surface and subsurface runoff in the system (Bingner and Theurer, 2005). AnnAGNPS simulates the soil profile into two layers. First layer is the top 200 mm and the second layer is from the bottom of the first layer to depth of the soil profile. Water balance in a soil layer can be shown as in the following figure. Soil moisture for each time step (i. e., daily time step) is calculated using the equation 3.1.

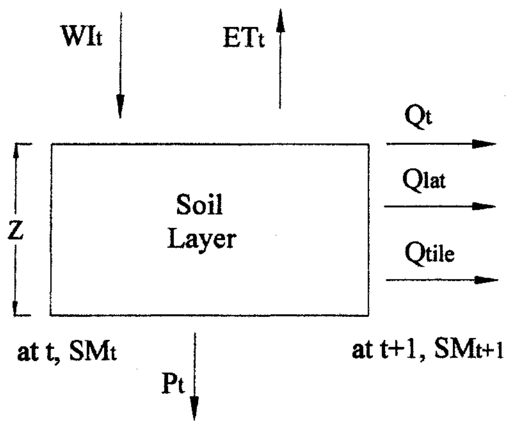


Figure 3.3: Water balance in a soil layer

$$SM_{t+1} = SM_t + \frac{WI_t - Q_t - P_t - ET_t - Q_{lat} - Q_{tile}}{Z} \tag{3.1}$$

where SM_t and SM_{t+1} are the moisture contents for each soil layer at the beginning (t) and end of the time period (t+1), WI_t is the water input consisting of precipitation, snowmelt or irrigation water (mm), Q_t is surface runoff (mm), P_t is percolation of water out of soil layer (mm), ET_t is potential evapotranspiration (mm), Q_{lat} is subsurface lateral flow (mm), Q_{tile} is tile drainage flow (mm), Z is thickness of soil layer (mm) and t is the time period.

b. Surface Runoff

The Soil Conservation Service (SCS) curve number technique is used within AnnAGNPS to determine the surface runoff from a field. The model first calculates a number of parameters in soil moisture calculations such as soil porosity and hydraulic properties, that will remain constant throughout the simulation period. Additional curve number parameters are calculated to vary the curve number for a given day between the dry and wet condition curve numbers based on soil moisture storage.

The average curve number (CN_2) can change due to operation events, which make significant changes to the land surface such as crop harvesting or during the active growth phase of a crop. These operations primarily change the ground cover and affect the hydraulic properties of the soil and have impact on runoff. The model calculates SCS curve numbers for each cell for a given day corresponding to dry condition or wilting point (CN_1) and wet condition or field capacity (CN_3) as a function of CN_2. The actual curve number (CN) associated in calculating runoff is allowed to vary depending on the available soil moisture content.

Surface runoff for each cell is calculated for the current day using the retention variable, S:

$$S = 254\left[\frac{100}{CN} - 1\right] \tag{3.2}$$

where S is retention variable (mm) and is related to the soil and cover conditions of the cell through the CN; CN is the SCS curve number. With the value of S calculated for the current day, runoff is calculated as,

$$Q = \frac{[WI - 0.2S]^2}{WI + 0.8S} \tag{3.3}$$

where Q is runoff (mm) and WI is water input to soil (mm). This equation is valid as long as WI is greater than 0.2 S, otherwise Q is set to zero. WI is equal to the snowmelt amount if a snowpack exists, or the daily precipitation, if no snow is present plus any irrigation water applied. Runoff volume for a cell (Q_t) is obtained by multiplying Q by the cell area.

c. Potential Evapotranspiration

The model uses the Penman equation, a commonly accepted method, to calculate potential evapotranspiration (ET_p). The equation uses standard climatological records of solar radiation (sunshine), air temperature, humidity and wind speed and is given as (Bingner and Theurer, 2005).

$$ET_p = \frac{1}{H_v}\left\{\left[\frac{\Lambda}{\Delta+\gamma}\right](R-G)+\left[\frac{\gamma}{\Delta+\gamma}\right]W\,(e_{sat}-e)\right\} \tag{3.4}$$

where ET_p is potential evapotranspiration (mm), H_v is latent heat of vaporization (MJ/kg) which is a function of the mean air temperature for a day, Δ is the slope of saturation vapor pressure-temperature curve, γ is psychrometric constant (kPa/ ^{0}C), R is net radiation (MJ/m^2), G is soil heat flux (MJ/m^2) and is calculated as a function of air temperature for the current day and three previous days, W is wind function, e_{sat} is saturation vapor pressure that is a function of air temperature, e is actual vapor pressure (kPa) and is a function of relative humidity. W is calculated using wind speed. Volume of evapotranspiration for a cell (ET_t) is obtained by multiplying ET_p by the cell area.

d. Subsuface Flow

The model calculates the lateral subsurface flow and tile drain flow to determine the contribution of subsurface flow from each cell to the corresponding reach. Subsurface flow calculation is done only when there is an impervious layer present in the soil profile. In case of tile drainage, the model assumes that a steady constant flow occurs through the soil to the drains (Figure 3.4). When the water table is above tile drains, the model calculates drainage flux through pipes using the widely applicable Hooghoudt's equation (Bingner and Theurer, 2005) shown below:

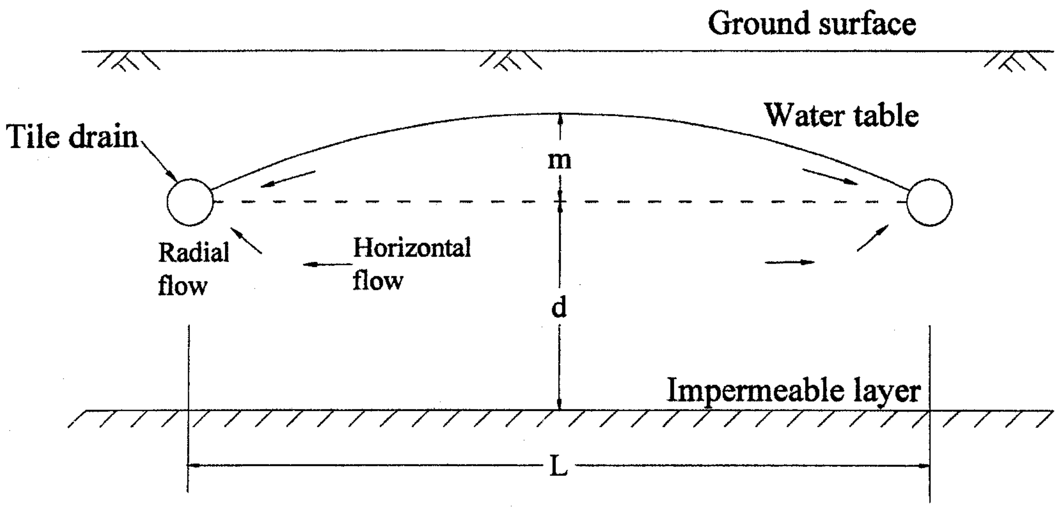


Figure 3.4: Schematic diagram for Houghoudt tile drainage

$$q_{drain} = \frac{8\, K_s\, d_e\, m + 4\, K_s\, m^2}{L^2} \tag{3.5}$$

where q_{drain} is drainage flux (mm per time period), K_s is saturated lateral hydraulic conductivity (mm per time period), L is the distance between tile drains (m), m is midpoint water table height above tile drains (m) and d_e is the equivalent depth of impermeable layer below the tile drain which is a function of L, d and radius of tile drain tube. The total tile drainage flow out of each cell (Q_{tile}) to corresponding reach is obtained by multiplying q_{drain} for the cell by area of the cell.

When the water table is below the depth of the tile drainage system, the model calculates lateral flow using Darcy's equation. Darcy's equation is widely used and provides an accurate description of subsurface flow. In the model, only the saturated flow condition is considered and subsurface flow is assumed to be homogeneous through the entire soil profile.

$$q_{lat} = -K_s \frac{dh}{dl} \tag{3.6}$$

where q_{lat} is subsurface lateral flow (mm per time period), K_s is saturated hydraulic conductivity for each soil layer (mm per time period), *dh/dl* is hydraulic gradient where stream length *dl* represents the length of the cell. The total volume of lateral flow out of each cell (Q_{lat}) is obtained by the product of q_{lat} and the lateral flow cross section area.

e. Channel Hydrology

AnnAGNPS uses TR55 methodology to calculate peak water discharge for each cell. Flowpath in a cell is divided into a section of overland flow, followed by a section of shallow concentrated flow and a section of concentrated or open channel flow. Length of overland flow and shallow concentrated flow is assumed to be no longer than a maximum length of 50 m each. The length of the in-cell, concentrated flow is the remainder of the in-cell flow length. In-cell time of concentration for flow to each cell outlet (T_{c,in_cell}) is calculated using travel time for flow in these sections using the equation 3.7:

$$T_{c,in\_cell} = T_{t,ov} + T_{t,scf} + T_{t,cf} \tag{3.7}$$

where T_{c,in_cell} is time of concentration for the in-cell processes, $T_{t,ov}$ is travel time for the overland flow period, $T_{t,scf}$ is travel time for the shallow concentrated flow period and $T_{t,cf}$ is travel time for the concentrated flow period. Time of concentration to channel reach outlet is the maximum value of time of concentrations for all reaches flowing into a reach being considered.

AnnAGNPS assumes a triangular shaped hydrograph. The time to base of the hydrograph (i.e. duration of each surface runoff event) is calculated using equation 3.8,

$$t_b = 20\,(R_q\,D_a / Q_p) \tag{3.8}$$

where Q_p is peak discharge (m^3/s), R_q is total runoff volume from upstream drainage area (mm), D_a is total drainage area (ha.) and t_b is the time base of the hydrograph (s). The ratio of initial abstraction, I_a to 24-hour precipitation is needed to calculate the peak discharge for each cell during each runoff event. The model calculates initial abstraction using the equation 3.9,

$$I_a = (P_{24} + 2\,Q_{24}) - (5\,Q_{24}\,P_{24} + 4\,Q_{24}^2)^{0.5} \tag{3.9}$$

where Q_{24} is 24-hour runoff. The model calculates the peak discharge for each runoff event using the following equation,

$$q_p = 2.7778 \times 10^{-3}\,P_{24}\,D_a \left[\frac{a + (c\,T_c) + (e\,T_c^2)}{1 + (b\,T_c) + (d\,T_c^2) + (f\,T_c^3)} \right] \tag{3.10}$$

where q_p is peak discharge (m^3/s), D_a is total drainage area (ha.), T_c is time of concentration (hr.) and *a, b, c, d, e* and *f* are the unit peak discharge regression coefficients for a given (I_a/P_{24}) and rainfall distribution type.

f. <u>Sediment</u>

AnnAGNPS uses Revised Universal Soils Loss Equation (RUSLE) (Renard et al., 1997) technology to predict sheet and rill erosion from cells. RUSLE technology within AnnAGNPS calculates LS, C, P factors for each cell in the watershed and a K factor for each soil in the watershed. When factors other than rainfall are held constant, soil losses from agricultural fields are directly proportional to a rainstorm parameter called EI. The value of EI for a given rainstorm equals the product of total storm energy (E) times the maximum 30-minute rainfall intensity (I_{30}) (Renard et al., 1997). RUSLE uses EI value to determine the erosion within a cell and is calculated for a given rainfall distribution type and the rainfall amount using equation 3.11,

$$\mathrm{EI} = \frac{A \exp(2.119 \log(\mathrm{P}) \exp(0.0086 \log (24)))}{\exp(B \log(24))} \tag{3.11}$$

where EI is energy intensity, P is precipitation or snowmelt, A and B are EI coefficients used in AnnAGNPS for different cumulative rainfall distributions. The total potential erosion within each cell is calculated using

$$A = R * LS * K * C * P \tag{3.12}$$

where A is total potential erosion, R is the annual rainfall-runoff erosivity factor, LS, K, C and P are RUSLE coefficients. Potential erosion is multiplied by the sediment delivery ratio to determine the amount of sediment delivered to the edge of the field. The Hydro-geomorphic Universal Soil Loss Equation (HUSLE) is used to determine the delivery ratio for total sediment. The sediment delivered into the edge of the cell is broken into five particle size classes: clay, silt, sand, large aggregate and small aggregate. All sediment routing in the concentrated flow channels used within AnnAGNPS are performed using five particle size classes and for each increment of the hydrograph.

g. Chemical Routing

AnnAGNPS recognizes three nutrients: nitrogen, phosphorous and organic carbon. Nitrogen and phosphorous are considered to be able to exist in both the dissolved and attached forms. AnnAGNPS uses a simple first order model for partitioning between chemical forms as shown in equation 3.13,

$$M_s = \frac{M_c}{1 + K_d} \qquad (3.13)$$

where M_s is total mass of chemical in solution (Mg), M_c is total mass of chemical both attached and in solution (Mg) and K_d is partition coefficient of chemical (Mg).

3.4 SUMMARY

The structure and the processes of AnnAGNPS model are briefly described in this chapter. AnnAGNPS requires about 400 parameters in 34 different data categories to describe a watershed and there are a number of modules supplied with AnnAGNPS model to aid in the preparation of the required input database. A brief description of these supporting modules and the methodology used in the development of AnnAGNPS input database is discussed in the following chapter. Sensitivity analysis was performed to increase the confidence in model predictions and to improve the understanding of the model behaviour also discussed in the following chapter.

CHAPTER 4

MODEL DEVELOPMENT

4.1 MUDDY CREEK WATERSHED – AN OVERVIEW

Muddy Creek watershed is located in Essex County, in the Southwestern part of Ontario, Canada between Lake Erie and Lake St. Clair (Figure 4.1). Watershed consists of an area of about 840 ha. (Figure 4.2). Lake Erie to the east of the watershed has an altitude of 174 meters above mean sea level (MSL). Topography of the watershed is gently rolling to flat and land-surface elevations in the Muddy Creek watershed range from about 174 meters to 188 meters above sea level. Watershed delivers runoff to Wheatley Harbour, which is a small, confined harbour on the North shore of Lake Erie.

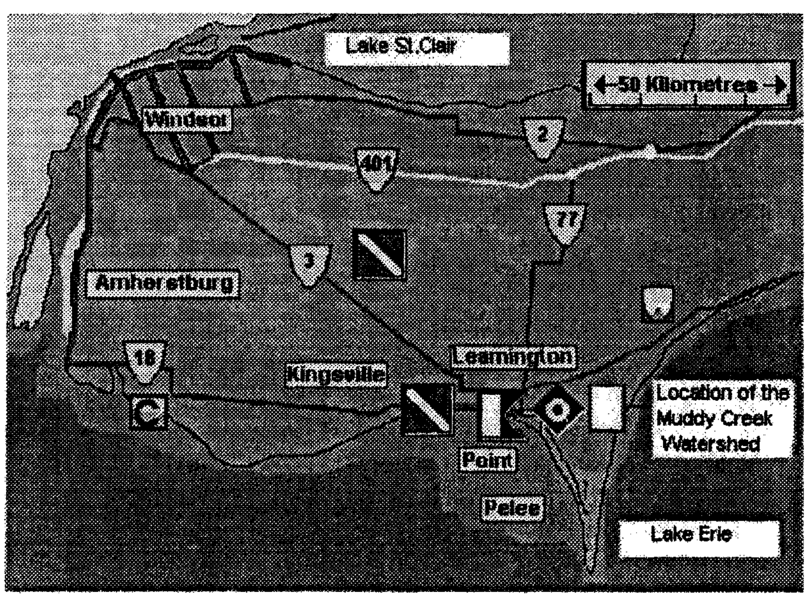


Figure 4.1: Location map of Muddy Creek Watershed
(Adapted from www.ontarioexplorer.com)

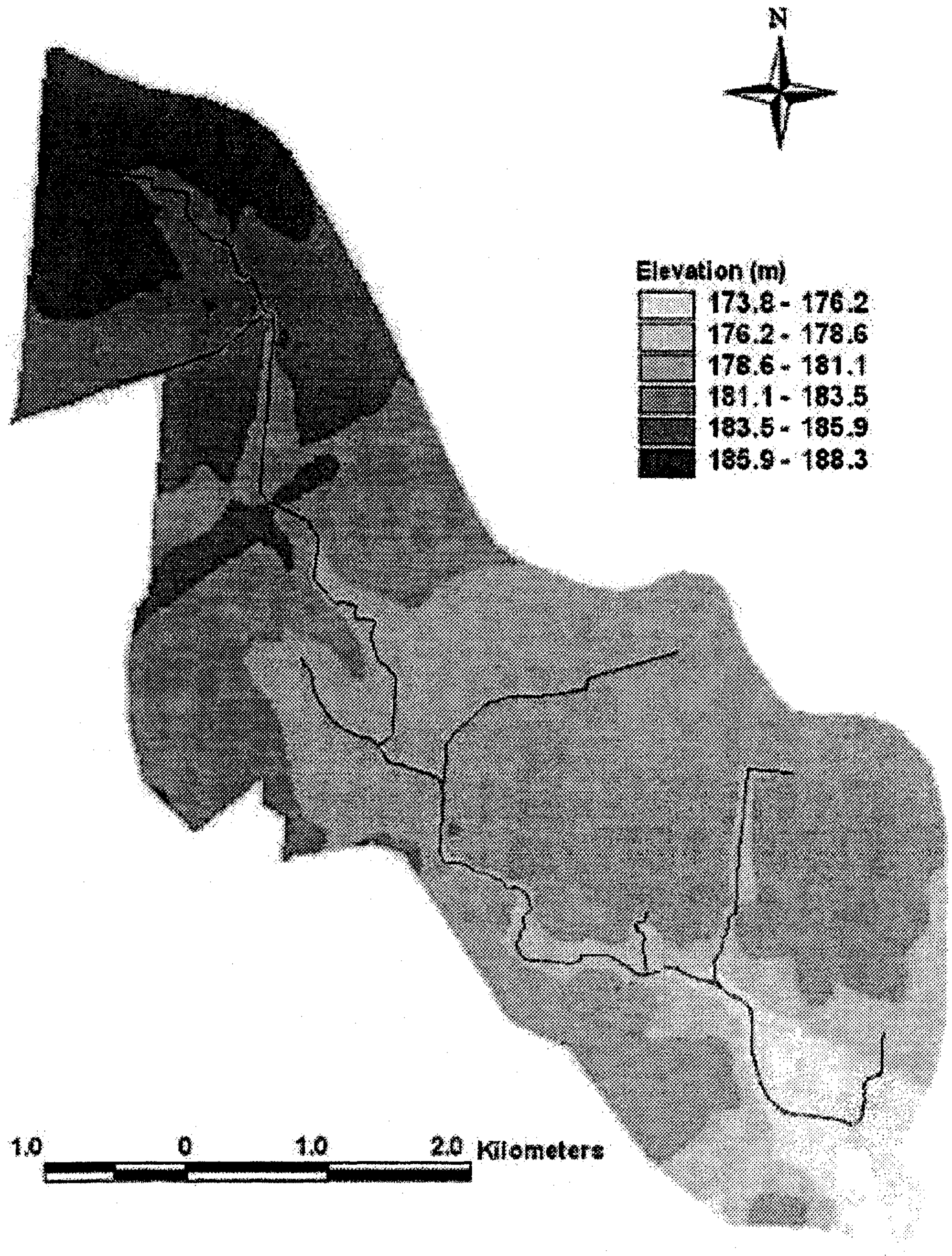


Figure 4.2: Drainage network and Digital Elevations Model (DEM) of the Muddy Creek Watershed

The Muddy Creek watershed has an extensive network of open drains, which expedite the delivery of sediment and other contaminants from agricultural lands to receiving water bodies. Nearly 80% of the watershed area is tile drained. Sediments and nutrients from agricultural runoff, as well as discharge from faulty septic systems, have been identified as environmental concerns in Wheatley Harbour. Most drains in the watershed are channeled, with little or no vegetative buffer to filter runoff and this has caused siltation and reduced water depths in the wetlands (ERCA, 2000). Filling in of wetlands to create land for residential and farm land also have altered many components of the natural ecosystem in the Muddy Creek watershed. The wetland is often nearly dry during Summer times from lack of stream runoff which might be the result of sediment deposition that are washed off from the upstream agricultural lands (Environment Canada, 2005).

4.1.1 Land use

Muddy Creek watershed is predominantly an agricultural watershed with cultivated croplands and pasture or fallow conditions. Urban and rural residential land uses are about five percent of the watershed area that consist of Former Township of Mersea and parts of Wheatley Township. Wooded areas constitute approximately five percent of the watershed area and are mostly located at the lower end of the Muddy Creek. Agriculture constitutes over 90% of the watershed area and the predominant types of crops produced include corn, soybeans and wheat.

4.1.2 Climate

Essex County is located in a region that receives an average annual precipitation of about 850 mm. The annual mean temperature in Essex County is about 14°C. Winter mean of 2.3°C, spring mean of 18 °C, summer mean of 25 °C and fall mean of 9°C, are generally the warmest temperatures in the province.

4.1.3 Stream Characteristics

Muddy Creek watershed consists of channelized streams at the upper part of the watershed and natural streams at the lower part of the watershed. Bank erosion is observed during field surveys throughout the natural streams having sinuous alignment. Some parts of the streams have vegetation that result in reduced erosion and sediment loads to the receiving waters. During field visits, evidence of bank-toe erosion resulting in higher stream banks was also noticed (Figure 4.3). Increase in flow of surface water and decrease in soil resistance to erosion could cause this kind of erosion. An increase in water flow can be due to several reasons such as, improvement of drainage channels upstream, removal of water holding areas upstream (such as bushes), change in tillage and cropping practices in the watershed. Bank failures occur as the resistance of the bank materials to shearing is exceeded by gravitational forces. It is learnt that there are no flow periods in some of the streams. However, presently there is no measured sediment and runoff data available for the Muddy Creek watershed to quantify flow amounts.

Figure 4.3: Erosion of natural stream banks and toe-erosion – March 2006

4.1.4 Soil Characteristics

Generally, soils with faster infiltration rates, higher levels of organic matter and improved soil structure have a greater resistance to erosion. Sand, sandy loam and loam textured soils tend to be less erodible than silt, very fine sand, and certain clay textured soils. According to Richards et al. (1949), Brookston and Berrien are the major soil types in the Muddy Creek watershed. The Brookston series is the poorly drained member of the Huron catena. This series has fairly high organic matter content in the surface soil and it exhibits the characteristic of the Dark Grey Gleisolic soils. Brookston clay covers about 25% of the watershed area. Brookston clay (sand spot phase) occurs in about 48% area of the watershed and is the predominant soil type in the watershed. The sand spot phase is a condition where shallow sandy knolls similar to the Berrien are scattered over an area of Brookston clay. Usually the sand does not exceed three feet in depth at the center of the spot. These knolls are somewhat acidic having a pH of 6.3 to 6.5. Berrien Sandy Loam occurs in about 16% of the watershed area. The Berrien sandy loam is fairly well suited to the growing of a wide range of cash crops. Plainfield Sand that has excessive natural drainage characteristics covers about 16% of the watershed area. The coarse nature of this soil type allows for ready percolation of soil moisture (Richards et al., 1949).

4.2 OVERVIEW OF THE METHODOLOGY

4.2.1 Introduction

AnnAGNPS requires about 400 parameters in 34 different data categories to describe a watershed including topographic data, soil and land-use related data, and climate data. These data are prepared and organized using the component modules provided with the AnnAGNPS program. The modules TopAGNPS, a subset of the Topographic Parameterization (TOPAZ) and AGricultural watershed FLOWnet generation program (AGFLOW) are used to generate spatially varying drainage densities and subcatchment areas. AnnAGNPS cells are hydrologically determined by varying critical source area and minimum source channel length values. Physical cell parameters such as area, length, slopes and LS parameters are determined by AGFLOW module. AnnAGNPS cells are homogeneous in soil and land use type. Dominant soil and land use type for cells are assigned by superimposing the soil and land use shape files over the delineated subwatershed shape file.

AnnAGNPS Input Editor and AnnAGNPS-Arcview interface programs are used to develop the input file "AnnAGNPS.inp", required for AnnAGNPS pollutant loading model. Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997) has been implemented in AnnAGNPS to predict the sheet and rill erosion resulting from raindrop splash and surface runoff. Soil-Plant-Air-Water (SPAW) (Saxton and Wiley, 2005) computer model that simulate the daily hydrologic water budgets of agricultural lands is used to derive soil water relationships such as wilting point, field capacity and hydraulic conductivity using soil characteristics such as soil texture and organic matter.

Climate data is of great importance for AnnAGNPS simulation and results are largely dependent on the quality of the climate data used in the simulation. Historic data such as daily precipitation, maximum and minimum temperature, dew-point temperature and sky cover data are obtained from Environment Canada on-line climate data website for the nearest climate station of the Muddy Creek watershed Kingsville, for the 13 year simulation period from 1991 to 2003.

4.2.2 Modules for model development

a. <u>TOpographic PArameteriZation program (TOPAZ)</u>

TOpographic PArameteriZation program (TOPAZ) (Garbrecht and Martz, 1999) is used to generate spatially varying drainage densities and subcatchment areas from digital elevation models (DEMs). DEMs) are numerical representations of the elevations of a surface that has been divided into regularly spaced grids (Martz and Garbrecht, 1993). TOPAZ generates raster output files of the drainage network, subcatchment areas and a variety of drainage-related topographic variables that can be easily imported to Geographic Information Systems (GIS) for display.

TOPAZ consists of three programs: Digital Elevation Drainage Network Model (DEDNM), RASter PROperties (RASPRO) and RASter FORmatting (RASFOR) and each program requires, input files provided by the user and input files that are generated internally. For hydrographic landscape discretization and channel network generation, two important parameters: the critical source area (CSA) and the minimum source channel length (MSCL) are defined. These two parameters control the topology and properties of the network and subcatchments generated by TOPAZ. For example, a higher CSA parameter value results in a drainage network with a lower drainage density, and a higher MSCL parameter value results in a drainage network without short, first order source channels (Garbrecht and Martz, 1999).

b. AGricultural watershed FLOWnet generation program (AGFLOW)

The AGricultural watershed FLOWnet generation program (AGFLOW) (Bingner et al., 1997) is used to determine the topographic related input parameters for AnnAGNPS and to format the TOPAGNPS output for importing into the form needed by AnnAGNPS. This program uses FLOWGEN.INP as input data file. The input file consists of data such as: AnnAGNPS input and output path, AGFlow output path, values for CSA, MSCL and watershed outlet location.

c. Input Editor

AnnAGNPS Pollutant Loading (PL) model requires two input files: AnnAGNPS.inp and DayClim.inp. AnnAGNPS input editor is used to import data created from TOPAGNPS and AGFLOW, in order to develop AnnAGNPS.inp input file. DayClim.inp, input file created with historically recorded data are imported into the daily climate data section of the input editor program. Also Input Editor is used to compile other input parameters required by PL model. Input Editor program facilitates exporting data as text, comma separated or as database files.

d. AnnAGNPS-Arcview interface

The AnnAGNPS-Arcview interface is used to simplify the use of TOPAGNPS and AGFLOW modules and to derive cell and reach data required by the AnnAGNPS Input Editor. The interface uses information extracted from the DEM to identify and quantify topographic features and land surface processes based on values of hydrology, drainage characteristics and elevation data and user defined outlet row and column values to generate the TOPAGNPS files (USDA-ARS, 2005).

e. Revised Universal Soil Loss Equation (RUSLE)

AnnAGNPS uses RUSLE (Renard et al., 1997) module to estimate soil loss resulting from sheet and rill erosion in farm fields. RUSLE is the modified version of USLE developed by Wischmeier and Smith in 1978. This equation is very robust and has been used in many research studies related to sedimentation in watersheds (Montgomery et al., 1997). RUSLE computes the average annual erosion by using a functional relationship of six factors, as is given by equation 3.12.

f. Hydro-geomorphic Universal Soil Loss Equation (HUSLE)

Whenever a runoff event occurs, RUSLE calculates the amount of sheet and rill erosion in a field but does not calculate field deposition. In AnnAGNPS, Hydro-geomorphic Universal Soil Loss Equation (HUSLE) is used to determine the amount of sediment delivered to the stream that is generated from sheet and rill erosion. HUSLE calculates the total sediment yield at a point, for a given storm event using average RUSLE parameters, upstream drainage area, peak discharge and volume of surface runoff.

4.3 DEVELOPMENT OF INPUT DATABASE

To model runoff, sediment erosion and nutrient transport at the watershed scale, a large amount of input data had to be acquired, organized, and stored. This is achieved using the computer modules described in section 4.2.2 and assembling many sources of information such as soil data, landuse details, field management practices and weather information. Topographic information is crucial in determining the watershed and subwatershed boundaries, channel locations, channel slopes, flow routing, field slopes, flow travel times, the RUSLE LS-factor, aspect and elevation of fields. Most of the required parameters are obtained as described above, while some data such as channel geometry parameters, were to be measured in the field in order to represent them realistically. Data related to agricultural activities such as crop growth parameters, crop rotations, operation management data and fertilizer application data are obtained in consultation with the staff of Agriculture Canada at Harrow in Essex County, ERCA and from several related databases while some other parameters were assumed with the best available data.

4.3.1 Digital Elevation Models (DEMs)

The use of DEMs provides a convenient source of topographic information. The Essex Region Conservation Authority (ERCA) provided the DEM for the watershed having a resolution of 10 m x 10 m. Stream locations are important in generating AnnAGNPS components. During the field visits to Muddy Creek watershed, it was observed that some modifications to the GIS layers were necessary with regard to the stream network.

4.3.2 Drainage boundary and sub-drainage areas

TOPAGNPS program is used to generate drainage boundary and watershed outlet from the DEM for the watershed. AnnAGNPS-Arcview interface accesses the TOPAGNPS and AGFLOW files, and the DEM in generating the required Arcview shape files from which the necessary data can be extracted. The sub-drainage areas of the watershed were discretized into AnnAGNPS cells based on the spatial variation of landuse and the location of the digitized stream network. Delineation process started with an assumption of the critical source area (CSA) and minimum source channel length (MSCL) required with the use of TOPAGNPS. The CSA parameter represents the threshold drainage area below which a channel is assumed to form (Garbrecht and Martz, 1999). Larger CSA values produce sparse drainage network and smaller values produce dense drainage network. Also, larger CSA and MCSL values produce large sub-catchment areas and smaller values produce small sub-catchment areas. Figure 4.4 illustrates the difference in the drainage density of the generated network by changing the value of the CSA from a larger to a smaller value. Figure 4.5 illustrates the differences in the generated sub-catchment resolution by changing CSA and MCSL values from larger to smaller.

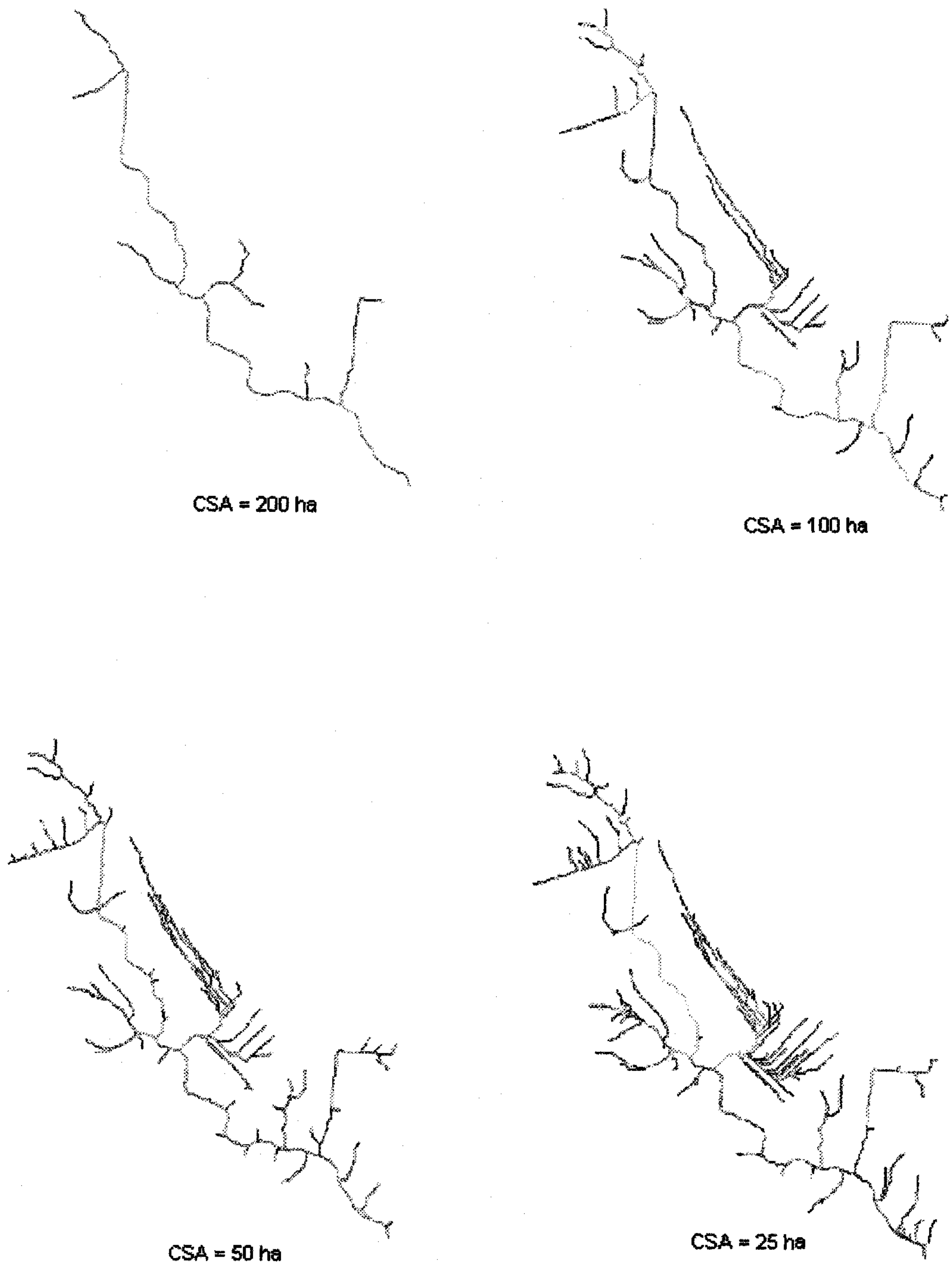


Figure 4.4: Effect of Critical Source Area (CSA) on the drainage network

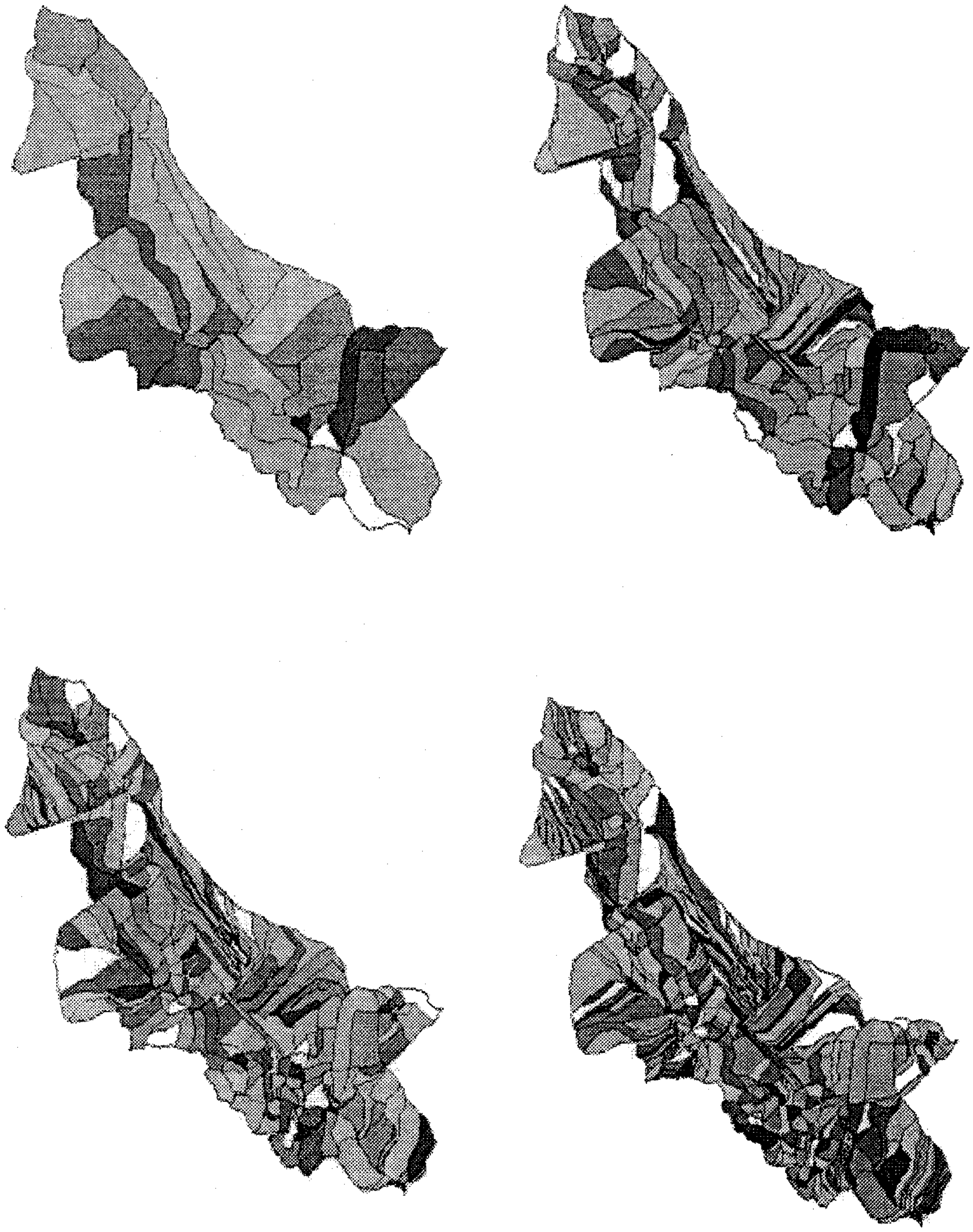


Figure 4.5: Generated sub-catchment resolutions by changing CSA and MCSL values from larger to smaller values.

For the Muddy Creek watershed, an initial 8.0 hectare CSA and 100 meter MSCL values were selected to produce AnnAGNPS cells. This initial subdivision produced 62 AnnAGNPS cells distributed throughout the watershed. Since watershed was not being adequately divided to capture the spatial variability of land use, some of the AnnAGNPS cells were selected for further subdivision using different CSA and MSCL values. The process of starting with larger cell sizes and working to subdivide only those areas needed to capture landuse features, provides the simplest approach in deriving AnnAGNPS cells. Several combinations of CSA and MCSL values were tested and final subdivision of Muddy Creek watershed produced 346 AnnAGNPS cells and 154 stream segments or reaches with an average cell area of 1.9 hectares.

4.3.3 Land use data

Soil erosion from agricultural lands is heavily dependent on the landuse type and hence it is critical to define the landuse with a greater accuracy. For the Muddy Creek watershed historical landuse data were not readily available. Arcview shape files obtained from ERCA for urban, wooded and wetland areas were superimposed in developing a composite landuse GIS layer. Since this is predominantly agricultural watershed, the remaining area is treated as agriculture.

Crop distribution obtained from Ontario Ministry of Agriculture for the Essex Region shows soybean as the dominating crop with about 60% of distribution in the region. For the simulation, crop distribution within the watershed was considered as soybean 60%, corn 25% and wheat 15% of the agricultural area. In developing the GIS layer of the landuse, a purely arbitrary approach was used for the crop distribution, maintaining the above ratios at the beginning of the simulation period. Landuse is assigned to each cell, based on the predominant landuse from the landuse GIS layer and the subwatershed GIS layer derived from the delineation procedure. Approximately 10% of the drainage area is covered by urban, wooded and wetland landuses and the rest were assigned for agricultural crops with the above distribution. Figure 4.6 illustrates the land use at the beginning of the simulation.

Crop rotation is an integral part of an agricultural crop production system. Hence agricultural land use allocated for cells is changed during the simulation period as different crops are grown in sequence. For the simulation, crop rotations are considered in such a way that no crop is followed by the same crop as recommended in OMAFRA (2002b). In Essex Region, producers plant soybean about two weeks later than corn, when soil temperatures are higher and thus nutrient availability is greater. Based on the discussions with the officers of Agriculture Canada, Harrow office, and integrating available data, the crop planting and harvesting dates are identified as shown in Table 4.1.

Table 4.1: Crop planting and harvesting dates

Crop	Planting	Harvesting
Soybean	1st week of June	3rd week of October
Corn	2nd week of May	1st week of October
Wheat	2nd week of October	2nd week of July-next year

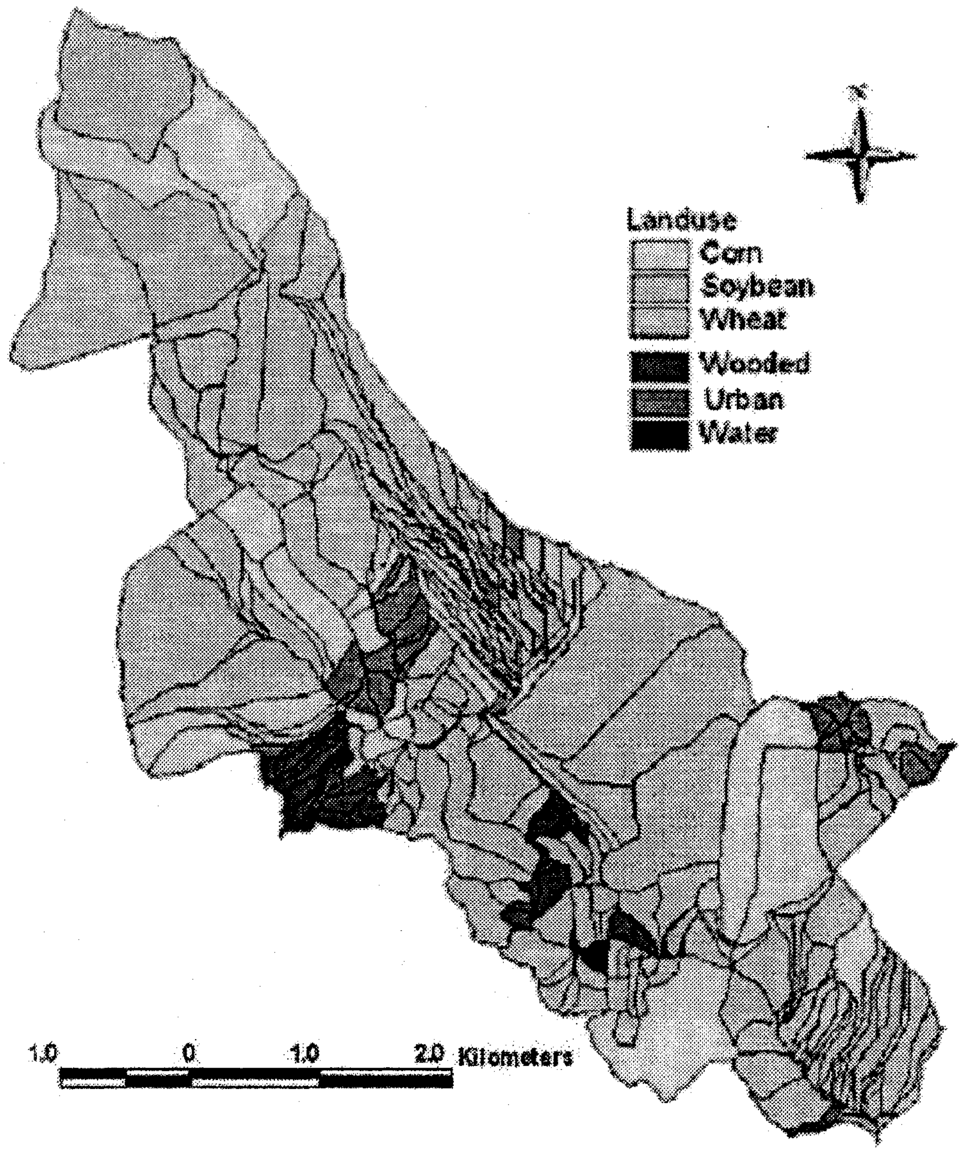


Figure 4.6: Land use at the beginning of the simulation period

4.3.4 Management data

Field management data were obtained from a variety of sources including discussions with ERCA personnel. Generally, reduced and no tillage methods are in operation throughout the watershed depending on crop types. Management operations may change the soil cover conditions with the time and hence change the SCS curve number that is a key factor in obtaining an accurate prediction of runoff and sediment yields. Management operation information for each cell was set up using RUSLE guidelines and databases provided with the AnnAGNPS module. Initial curve numbers are selected based on the guide to conservation planning with the Revised Universal Soil Loss Equation (Renard et al., 1997).

4.3.5 Soil data

Within the Muddy Creek watershed, five different soil types are identified from the soil GIS layer. The predominant soil type is Brookston sandy clay loam. Figure 4.7 illustrates the distribution of soil types to AnnAGNPS cells. Soil textures are identified from the soil data and corresponding composition of soils are obtained from Richards et al. (1949). The various layers in each soil type were defined. In AnnAGNPS, for the purposes of runoff generation and soil water storage, the soil profile is divided into two layers. The top 200 mm is used as a tillage layer whose properties such as, bulk density, can change. The remaining soil profile comprises the second layer whose properties remain static. For each layer in the soil profile, clay, silt, sand, rock and very fine sand ratio was input. In addition, saturated hydraulic conductivity, field capacity, wilting point, base saturation, pH and organic matter were input. Most of these data were obtained from the available databases and a program developed by Agricultural Research Service of United States Department of Agriculture (Saxton and Wiley, 2005). Soil organic matter, which is an important component of the nutrient cycle, holding soil moisture and soil structure, was also obtained from Richards et al. (1949) for surface soil layers. The above parameters for sub-soil layers were obtained assuming decreasing organic matter contents for sublayers.

Default values provided in the model were used for initial nutrient content (organic and inorganic nitrogen, organic and inorganic phosphorus). Hydrologic soil group for each soil was obtained from the soil GIS layer.

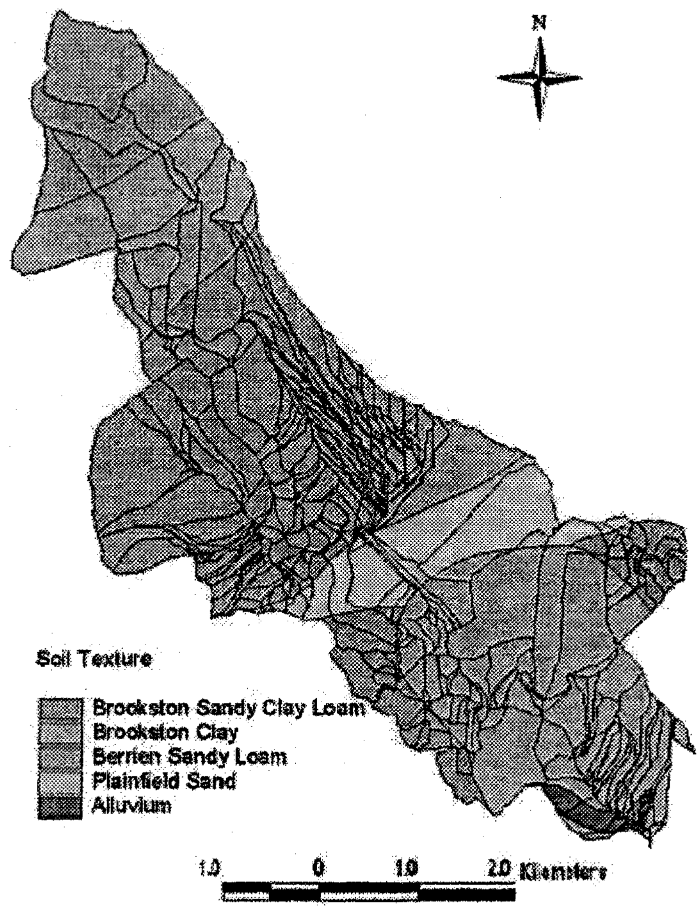


Figure 4.7: Soils identified by AnnAGNPS cells

4.3.6 Weather data

AnnAGNPS model has a supporting module GEM for synthetic weather generation based on historical values for nearby weather stations. For this simulation, historical climate data obtained from Kingsville weather station, which is the nearest station to the Muddy Creek watershed, is used. Daily climate data required for the model such as minimum and maximum temperature and precipitation data are obtained from Environment Canada's online climate data website. Other required climate data such as dew point temperature, wind speed and sky cover that were not available for Kingsville station were obtained from the Windsor Airport weather station. The available hourly data on these parameters was converted to daily data as required for the model input. Daily precipitation is the prime driver of the hydrologic cycle; temperatures are used to define frozen conditions and remaining climate parameters are used in the model to compute potential evapotranspiration using Penman equation.

To represent the current weather conditions in the simulation, all the above climate parameters were obtained for the 13-year period from 1991 to 2003. The average annual rainfall for the period is 859 mm. Figure 4.8 represents the variation of annual precipitation values for the period of simulation.

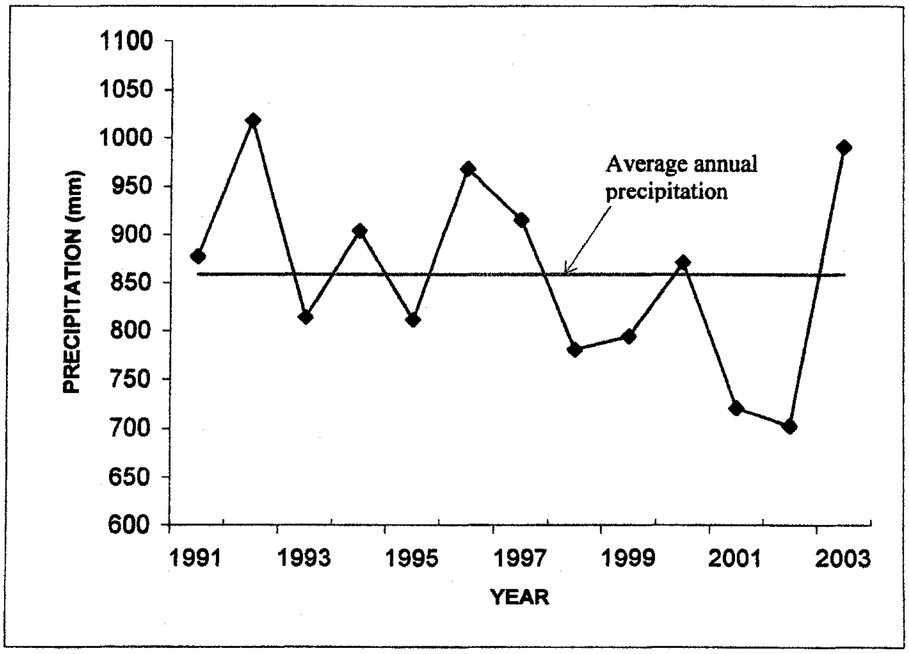


Figure 4.8: Variation of annual precipitation values

4.3.7 RUSLE parameters

a. <u>Rainfall-runoff erosivity factor (R)</u>

The value "R" quantifies the effect of raindrop impact and reflects the amount and rate of runoff likely to be associated with a given rainfall event. It represents two most important characteristics of storm erosivity: amount of rainfall and peak intensity sustained over an extended period of time. The greater the intensity and duration of the rain storm, the higher the erosion potential. "R" is the average annual total of the storm Energy Intensity (EI) values for a given area. The value of EI for a given rainstorm is equal to the product of total storm energy (E) times the maximum 30-min intensity (I_{30}). The storm energy indicates the volume of rainfall and runoff and I_{30} component reflects the prolonged peak rates of detachment and runoff. The product EI is a statistical interaction term that reflects how total energy and peak intensity are combined in each particular storm (Renard et al., 1997).

In AnnAGNPS, EI value for a given rainfall distribution type and the rainfall amount is determined using the equation 3.11. The annual rainfall-runoff erosivity factor R is the sum of the energy intensity values for all the storms in a given year. R factor is derived based on rainfall intensity data over extended periods (Renard et al., 1997) and is expressed by the equation:

$$R = \sum_{i=1}^{j} (EI_{30}) / N \qquad (4.1)$$

where R is rainfall-runoff erosivity factor in $MJ.mm.ha^{-1}.h^{-1}.\ yr^{-1}$, EI_{30} for i^{th} storm and j is the number of storms in an N year period. The distribution of erosive rains differs significantly with the geographical locations. R values for Ontario locations have been published by OMAFRA (2000a) and some extracted values are shown in the following table.

Table 4.2: Average annual R values for Ontario locations (OMAFRA, 2000a)

Location of weather station	County	R Factor
Essex	Essex	110
Windsor	Essex	110
Toronto	Metro-Toronto	90

b. Soil erodibility factor (K)

Soil erodibility may be thought as the ease with which soil is detached by splash during rainfall or by the surface flow. The K factor represents the average long-term soil and soil profile response to the erosive powers of rainstorms (Renard et al., 1997). The physical, chemical and mineralogical soil properties and their interactions affect K value. K factor is affected by antecedent soil-water and soil surface conditions and seasonal variations of soil properties (Renard et al., 1997). When the soil surface contains rock fragments, it reduces soil detachment by rainfall and reduces soil erosion. When rock fragments present in a coarse textured soil profile (having sand and loamy sand textures), it reduces infiltration and increases soil erosion. In RUSLE, rock fragments in surface soil is accounted in C factor. Subsurface component is accounted in K factor through adjustments of the permeability of soil (Renard et al., 1997).

Coarse textured soils such as sandy soils, have low K values, about 0.05 to 0.2, since they have low runoff even though they can get detached easily. Clayey soils are resistant to detachment and they also have low K values, about 0.05 to 0.15. Medium textured soils, such as silt loam soils are moderately susceptible to detachment and they produce moderate runoff and have K values, about 0.25 to 0.4. Soils with high silt contents are easily detached and are most erodible of all soils. They produce high rates of runoff and K values for these soils tend to be greater than 0.4. Soils with high organic matter content are less erodible because it increases infiltration and reduces the susceptibility of the soil to detachment by reducing overland flow and thus erosion. Soils that are most prone to sheet and rill erosion are those with relatively high sand content, low in organic matter and clay.

AnnAGNPS uses soil nomograph equations to calculate K factor for each soil in the watershed or optionally user can input K factor values for each soil (Bingner and Theurer, 2003). Stone and Hillborn (2002) suggested soil erodibility factors for Ontario soils with different soil textural classes and having average organic matter content. Novotny (2003) suggested values of K depending on soil texture and having organic matter contents of 0.5, 2 and 4 percent. Table 4.3 lists general magnitudes of K values for different soil textural classes and for average organic matter content.

c. Topographic factor (LS)

Erosion increases as slope length increases and is accounted in slope length factor (L). As the slope steepness increases, soil loss increases and considered in slope steepness factor (S). In erosion prediction calculations the factors L and S are usually evaluated together and the effect of topography on soil erosion is accounted for by the combined topographic factor (LS). The LS factor represents a ratio of soil loss under given conditions to that at a site with the standard slope steepness of 9% and slope length of 72.6 feet.

Table 4.3: Soil erodibility factor –K for different soil textures

Soil textural class	Soil erodibility factor (K)	
	Stone and Hillborn (2002)	Novotny (2003)
Clay	0.22	0.17
Silty Clay	0.26	0.23
Clay Loam	0.30	0.25
Silty Clay Loam	0.32	0.32
Sandy Loam	0.13	0.24
Sandy Clay Loam	0.20	0.25
Fine Sandy Loam	0.18	0.30
Sand	0.02	0.03
Loamy Sand	0.04	0.10
Fine Sand	0.08	0.14
Very Fine Sand	0.43	0.36
Silt Loam	0.38	0.42
Very Fine Sandy Loam	0.35	0.41
Silt	--	0.52

Renard et al. (1997) showed, for average erosion, slope length factor (L) varies with slope length λ as:

$$L = (\lambda / 72.6)^m \tag{4.2}$$

where 72.6 is the RUSLE unit plot length in feet, m is a variable slope length exponent. The slope length λ is the horizontal projection of the slope in feet.

Procedures have been developed to calculate the LS factor for multiple cells using the slope geometry from DEMs for the watershed. Moore (1992), based on erosion theory, developed a relationship to determine the LS factor using the sub-watershed area (A) and average slope (δ). Moore's equation to calculate the LS factor is:

$$LS = (A / 22.13)^{0.4} * (\sin \delta / 0.0896)^{1.3} \tag{4.3}$$

where A is the subwatershed area and δ is the average slope angle. During input database development, LS values generated for each cell from the AnnAGNPS-ArcView interface procedure were extracted into AnnAGNPS input editor.

d. <u>Cover management factor (C)</u>

Cover management factor reflects the effect of cropping and management practices on erosion rates and is the factor mostly used to compare the impacts of management options on conservation plans. "C" represents the ratio of soil loss from an area with specific cover and management to soil loss from a standard plot. In this case, standard plot is considered as an area under clean-tilled and continuous fallow condition (Renard et al., 1997). C values range from 1.0 where there is little soil cover to values less than 0.10 where there is dense cover and large amounts of crop residues left on the soil surface.

In evaluating C factor, impact of cropping and management on soil loss is generally divided into series of subfactors such as, impact of previous cropping and management, impact of vegetative canopy on soil surface, the reduction in soil loss due to surface cover and surface roughness and impact of low soil moisture on reduction of runoff from low intensity rainfall. Each of the subfactors is assigned a value and Soil Loss Ratio (SLR) is obtained by the following equation:

$$SLR = PLU * CC * SC * SR * SM \tag{4.4}$$

where PLU is the prior land use subfactor, CC is the canopy cover subfactor, SC is the surface cover subfactor, SR is the surface roughness subfactor and SM is the soil moisture subfactor. Each of the subfactors contains cropping and management variables that affect soil erosion.

Usually, in croplands, soil and crop parameters vary with time due to either specific management practices or due to climate changes. This makes that the SLR values be calculated frequently enough to capture those variations. Average annual soil loss can be high if cropping and management operations occur during higher rainfall erosivity. RUSLE module in AnnAGNPS calculates SLR values every 15 days throughout the year to incorporate this effect. Once SLR for each time interval is calculated they are multiplied by their corresponding percentage of annual EI values. Cover management factor (C) can be calculated using the following equation:

$$C = \left[SLR_1 * EI_1 + SLR_2 * EI_2 + . . + SLR_j * EI_j + . . + SLR_n * EI_n \right] / EI_t \qquad (4.5)$$

where C is the average annual cover management factor, SLR_j is the soil loss ratio value for time period j, EI is the percentage annual EI occurring during that time period, n is the number of periods considered and EI_t is the sum of EI percentages for the entire time period. In the development of input database, canopy cover, surface cover, surface roughness and droplet fall heights for various crops are obtained from Renard et al., (1997).

e. <u>Support practice factor (P)</u>

Support practice factor (P) is the ratio of soil loss with a specific support practice such as contouring, strip-cropping or terracing, to the soil loss with straight-row farming with rows oriented parallel to the slope gradient. Generally support practices affect erosion by modifying the flow pattern, grade or direction of subsurface runoff. Support practice factor reflects the effects of practices that will reduce the amount and rate of the water runoff and thus reduce the amount of erosion. Support practice factor varies from 1.0, when there are no support practices to 0.1 to 0.05 for areas with practices such as terracing.

4.4 SENSITIVITY ANALYSIS

4.4.1 Introduction

In the Muddy Creek watershed, there are no measured values for surface runoff, sediment yield and nutrient loadings and hence no calibration and validation of the model can be performed. In such circumstances, a high uncertainty in model simulations could be expected. To increase the confidence in the model predictions and to improve the understanding of the model behaviour, a sensitivity analysis was conducted as part of the model development. Several parameters, that may have significant influence on the runoff, sediment and nutrient loadings, are selected for sensitivity analysis, based on the model processes and the results of previous studies reported in literature.

4.4.2 Method of analysis

The most common method used in sensitivity analysis is to examine percentage change in model output results by changing input parameters one at a time by some constant percentage. The method used in sensitivity analysis reported by Vieux and Needam (1993), was to keep all other variables constant while varying one parameter at a time, by ±25% and ±50% and measuring the change relative to a base value. Ma et al. (2002) performed sensitivity analysis to test eight parameters related to soil and phosphorus sediment, on model output results. In their study they kept all other variables constant while varying one parameter at a time, by ±10%.

Variation by a fixed percentage of the initial parameter value may sometimes cause unrealistic results. If the initial parameter value is located close to the upper or lower bound of the valid parameter range, the variation by a fixed percentage can cause inadmissible values beyond the valid parameter range. Therefore, in place of this conventional method, an alternative approach to define the parameter variation is to be considered. Chaubey et al. (1999) performed a study on uncertainty in the model parameters due to spatial variability of rainfall parameters. In their study, a relative sensitivity index was used to rank the model parameters in terms of their sensitivities in affecting the model outputs. In Lenhart et al. (2002) approach, the parameter values are not varied by a fixed percentage of the initial value, but they are varied by a fixed percentage within the valid domain of the parameter value. Sensitivity is expressed by a dimensionless index I, which is calculated as the ratio between the relative change of model output and the relative change of an input parameter. Mathematically, the partial derivative $\partial y/\partial x$ is used to represent the dependence of a variable y on a parameter x. This expression may be numerically approximated by a finite difference. Let y_0 be the model output calculated with an initial value x_0 of the parameter x. Let this initial parameter value be varied by $\pm \Delta x$, yielding $x_1 = x_0 - \Delta x$ and $x_2 = x_0 + \Delta x$. Let y_1 and y_2 are the corresponding values for $x_1 = x_0 - \Delta x$ and $x_2 = x_0 + \Delta x$ (Figure 4.9). Then the finite approximation of the partial derivative $\partial y/\partial x$ can be written as, $I' = (y_2 - y_1)/ (2\,\Delta x)$.

To get a dimensionless index, I' is normalized by dividing with the corresponding initial values. The expression for the sensitivity index I then assumes the form

$$I = \frac{(y_2 - y_1)/y_0}{2\Delta x/x_0} \tag{4.6}$$

The sign of the sensitivity index I, indicates whether the model output change co-directionally to the input parameter change, i.e., if an increase in the parameter leads to an increase of the output variable and a decrease of the parameter to a decrease of the variable. In order to assess the effect of parameter sensitivity, the calculated sensitivity indices are ranked into four different classes as shown in Table 4.3.

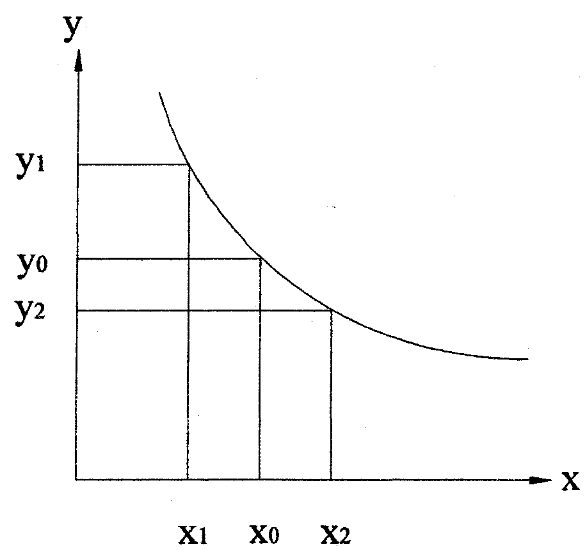


Figure 4.9: Behavior of an output variable y with an input parameter x.

Table 4.4: Sensitivity classes (Lenhart et al. 2002)

Class	Index	Sensitivity level
I	$0.00 \leq \lvert I \rvert < 0.05$	Small to negligible
II	$0.05 \leq \lvert I \rvert < 0.20$	Medium
III	$0.20 \leq \lvert I \rvert < 1.00$	High
IV	$\lvert I \rvert \geq 1.00$	Very high

This approach is followed in the present study to determine the sensitivity of the parameters selected in the following section, except for the cell size. In this approach, parameter value, x is varied by $\Delta x = \pm 10\%$ and $\pm 20\%$ as applicable within the entire valid domain of the parameter. The value x_0 is the parameter value assigned for the base case, based on the available data sources.

4.4.3 Parameters selected for the analysis

It has been recognized by several studies that the scale of cell discretization affects the model results significantly with respect to runoff and sediment loading (Vieux and Needam, 1993, Qiu et al., 1997). Therefore, the sensitivity of cell discretization on model results has been investigated to determine the critical cell size that is to be used in this simulation. In AnnAGNPS, each cell or subcatchment is assumed to be homogeneous in landuse, land management and soil type. Predominant landuse and soil type within a cell is considered distributed homogeneously within that cell. Therefore cells sizes should be selected such that they are able to capture the spatial variability of land uses and other features and hence cell size selection should not be done arbitrarily. Several parameters such as RUSLE topographic (LS) factor, channel network and flow path lengths, cell time of concentration are dependent on cell sizes. Different cell discretization would produce different model inputs, thus causing differences in the model output results.

Based on the reviews performed and the reported literature, thirteen AnnAGNPS parameters are selected for sensitivity analysis. They are listed in Table 4.4 with the corresponding run configurations. Parameters 1 to 6 are primarily represent soil properties. Parameters 9 to 11 are related to crop properties while parameters 7 and 8 are related to both soil and crop properties. Parameter 12 and 13 are related to the condition of the field, land use and cover type.

Table 4.5: List of parameters used in the sensitivity analysis

No.	Parameter	Run no.	No.	Parameter	Run no.
1	K factor	002-004	8	Inorganic N	029-032
2	Wilting point	005-008	9	Plant N uptake	033-036
3	Field capacity	009-012	10	Plant P uptake	037-040
4	Hydraulic conductivity	013-016	11	Fertilizer mixing code	041-044
5	pH value	017-020	12	Surface roughness	045-052
6	Organic matter content	021-024	13	Curve number	053-060
7	Organic N	025-028			

4.4.4 Simulation method

In order to observe the effect of cell sizes on model output results and to determine appropriate cell discretization that is to be applied in the simulation study, sensitivity analysis is performed for seven different cell discretizations. Watershed is divided into 33 cells to 700 cells having average cell sizes ranging from 1.0 ha. to 20 ha. for different cell discretizations.

Soil related parameters, K factor, wilting point, field capacity, hydraulic conductivity, pH value and organic matter content of soils are varied by $\Delta x = \pm 10\%$ of their base case values. Soil initial organic and inorganic nitrogen ratios were set at model default values in the base case. They were tested with the values 50 ppm for the top layer and 5 ppm for the subsequent layers for organic nitrogen ratio and 5 ppm for the top layer and 0.5 ppm for the subsequent layers for initial inorganic nitrogen ratio.

Crop related parameters, plant N uptake, plant P uptake and the parameter

"Fertilizer mixing code" was tested to see the effects on N and P loading. Fertilizer mixing code parameter reflects whether the applied fertilizer is mixed well within the depth of fertilizer application. Plant N and P uptake were set to model default values in the base case. N and P uptake values for soybean, wheat and corn were set to literature values and tested individually and also tested together to see the effect on N and P loading. Plant uptake values for N and P used in the simulation are shown in the table 4.6.

Table 4.6: Plant N and P uptake values

Crop	N uptake	P uptake
	(kg/kg of harvest)	
Soybean	0.092	0.0095
Wheat	0.022	0.0025
Corn	0.0017	0.0023

Surface roughness resulting from roots or any other vegetative effects on the surface is represented by random roughness parameter. This parameter was set to baseline value and tested with $\Delta x = +10\%$ and $+20\%$ to see the effects on erosion.

For the base case, runoff curve numbers for different land use and management and hydrologic soil groups were set to the values shown in Table 4.6. Runoff CNs are varied such that they do not go beyond the valid parameter range for respective hydrologic soil group. Range of parameters considered in the analysis are shown in Table 4.7. Simulations are carried out by varying curve numbers by 50% and 100% within the valid range.

Table 4.7: Curve numbers for hydrologic soil cover conditions (AMC II)

Land use or cover manageme nt	Treatment or practice	Hydrologic condition	Hydrologic soil group A	B	C	D
Fallow	Bare soil	-----	77 *77 – 85*	86 *86 - 90*	91 *91 – 93*	94 *94 -94*
	Crop residue cover	Poor	76 *76 – 84*	85 *85 - 89*	90 *90 – 92*	93 *93 - 94*
		Good	74 *74 – 82*	83 *83 – 87*	88 *88 – 89*	90 *90 - 94*
Row crops	Straight row	Poor	72 *72 – 80*	81 *81 – 87*	88 *88 – 90*	91 *91 - 94*
		Good	67 *67 – 77*	78 *78 – 84*	85 *85 – 88*	89 *89 - 94*
Small grain	Straight row	Poor	65 *65 – 75*	76 *76 - 80*	84 *84 – 87*	88 *88 - 94*
		Good	63 *63 – 74*	75 *75 – 82*	83 *83 – 86*	87 *87 - 94*
Pasture or range		Poor	68 *68 – 78*	79 *79 - 85*	86 *86 - 88*	89 *89 - 94*
		Good	39 *39 – 60*	61 *61 – 73*	74 *74 – 79*	80 *80 - 94*
Developing urban areas			77 *77 – 85*	86 *86 – 90*	91 *91 93*	94 *94 - 94*
No cover or minimal roughness or both			64 *64 – 77*	78 *78 – 84*	85 *85 – 87*	88 *88 - 94*

Note: CNs in italics show the range of values used in the sensitivity analysis.

4.5 AGRICULTURAL BEST MANAGEMENT PRACTICES

4.5.1 An overview

In general, not all the areas in a watershed, contribute sediment and nutrients to receiving waters. Small areas of the landscape with specific soil characteristics and agricultural practices are often responsible for a majority of the sediment and nutrient loading to surface waters. For best results, an effective Best Management Practices (BMP) or a combination of BMP's must be implemented in these areas that are most critical in exporting NPS pollutants. Therefore, in the selection of one or several BMP's to implement, the ability of BMP's to achieve the water quality goal should be considered. Also, the economic feasibility of implementing such BMP's too should be considered. There are several accepted BMP's that are used frequently in controlling soil erosion and reducing nutrient loadings at receiving waters (Appendix A). In this study, two BMP alternatives that are meant for reducing the sediment erosion within the watershed and that had a reasonable chance of being implemented were considered.

4.5.2 Vegetative filter strips:

Vegetative filter strips (VFS) are permanent grass borders of dense, tall, stiff grass on field boundaries or along stream segments that help in reducing soil input into streams. Filter/buffer strips act as porous dams to temporarily pond surface runoff and allow slowing down and reducing surface runoff from fields. Ponding allows sediments to settle and buffers traps the soil particles from surface runoff and gradually release water to down slope. The effectiveness of buffer strips is dependent on the buffer width, slope of the land, type of vegetation and most importantly on the particle size (Dosskey, 2001).

The AnnAGNPS model does not have a riparian buffer or filter strip component to evaluate the effectiveness of this BMP. Work is under way to develop this capability (Bingner et al., 2005). Though the model cannot model VFS BMP, it can account for the changes in erosion when a cropland is converted to a permanent grass border. In this simulation, streamside cells were converted to buffer strips to evaluate the effectiveness of VFS. Parameters such as, curve number, RUSLE C-factor, overland flow Manning's coefficient and surface condition constant are assigned to represent VFS in streamside cells.

4.5.3 Tile drainage:

Drainage is an important conservation practice. A properly designed drainage system should remove excess water from agricultural fields. Tile drains reduce surface runoff as well as increase the amount of water available for plants by allowing more water to soak into the soil (OMAFRA, 2002b). About 75% of the cropland in the Muddy creek watershed is tile drained. AnnAGNPS model allows for tile drainage to be turned on or off for any given cell during simulation. In this simulation, tile drain BMP was modeled by considering all the cropland in Muddy creek watershed as tile drained.

4.6 SUMMARY

In this chapter, the methodology adapted in the model development process and the development of input database required running the AnnAGNPS model was presented. As Muddy Creek watershed is an ungaged watershed, a high uncertainty could be involved in model results. In order to increase the confidence in model predictions and to improve the understanding of the model behaviour, a sensitivity analysis was conducted as part of the model development. Also, two BMP alternatives that are focused on reducing sediment erosion within the watershed are modeled. In the following chapter, sensitivity analysis, model simulation results and the effectiveness of BMPs on reducing the sediment erosion are presented.

CHAPTER 5

RESULTS AND DISCUSSIONS

5.1 OVERVIEW

AnnAGNPS simulation is performed to predict runoff volume, sediment and nutrient loadings at the outlet of the Muddy Creek watershed, over the thirteen-year period from 1991 to 2003. In this watershed, there are no measured values for surface runoff, sediment yield and nutrient loadings and hence no calibration and validation of the model is performed. To increase the confidence in model predictions and to improve the understanding of the model behaviour, a sensitivity analysis is performed. Results of the sensitivity analysis are presented in section 5.2. Results for runoff volume, sediment and nutrient loadings at the outlet of Muddy Creek watershed are presented in section 5.3. Vegetative buffer strips along stream segments and tile drainage are modeled as BMP alternatives in reducing soil input into streams and the results are presented in section 5.4.

5.2 SENSITIVITY ANALYSIS RESULTS

5.2.1 Effect of cell discretization

In order to observe the effect of cell sizes on the model output results, sensitivity analysis is performed on seven different cell discretizations. The watershed is divided into 33 cells to 700 cells having average cell sizes ranging from 1.0 ha. to 20 ha. for different cell discretizations. Simulated watershed runoff, sediment and nutrient loadings for different cell discretizations, obtained at the watershed outlet are summarized in Table 5.1.

Table 5.1: AnnAGNPS output results for different average cell sizes

Input/Output parameter	Average Cell Size (ha)						
	1	1.4	1.9	2.4	3	8	20
Number of cells	700	450	350	275	216	80	33
Runoff (mm)	254	255	259	254	258	259	257
Sediment loading at outlet (t/yr)	6035	4278	2800	1988	1246	367	90.5
Sediment yield at outlet (t/ha/yr)	9.2	6.58	4.31	3.06	1.92	0.56	0.14
N loading at outlet (kg/yr)	2744	3077	3221	3317	3418	3357	3358
P loading at outlet (kg/yr)	177	188	183	191	169	166	162
Flow path length (m)	51598	32870	27973	24762	23333	14991	10027

The most significant variation with respect to cell size is the change of sediment loading at the watershed outlet. As the cell size increased from 1.0 ha. to 20.0 ha., sediment loading reduced approximately by 66 times. Vieux and Needam (1993) found that as the cell size increases, the stream length decreases due to short-circuiting of flow paths. It can be seen from Table 5.1 that length of flow path has been reduced significantly as the cell size increased. Variation of N loading showed a 22% increase as the average cell size increased from 1.0 ha. to 20.0 ha., while P loading did not show such a trend with the change of cell sizes. Figure 5.1 shows the relative variation of sediment yield, flow path length, N and P loadings with the average cell size. This figure exhibits a trend in sediment yield and flow path length with respect to cell size. As the cell size increases, relative change in sediment yield and flow path length increases. N and P loadings did not exhibit such a trend with respect to change in cell size. Figure 5.2 shows the variation in sediment yield and flow path length for different cell sizes.

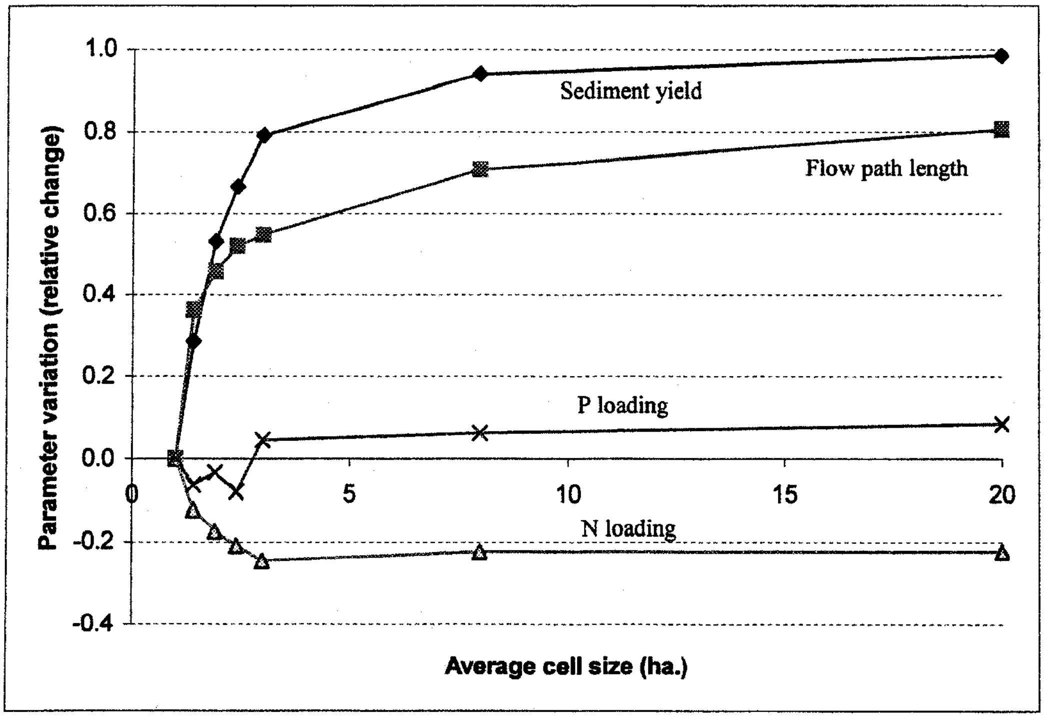


Figure 5.1: Effect of cell size on the relative change of sediment, N and P loadings

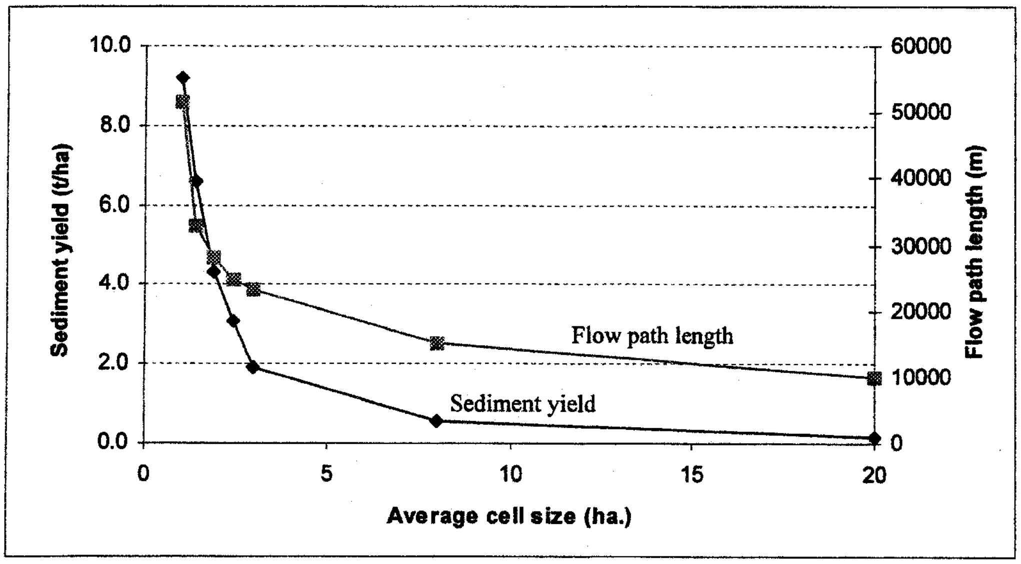


Figure 5.2: Effect of cell size on the sediment yield and flow path length

Figure 5.3 shows the sensitivity of sediment yield for the flow path length on a relative change basis. Sediment yield and flow path length are normalized by the respective values at 1.0 ha. cell size. Variation in relative sediment yield is high for cell sizes less than 1.4 ha. and for cell sizes greater than 3.0 ha. For the cell sizes in the range of 1.9 ha. to 3.0 ha., the variation of sediment yield is less dependent on the flow path length. It can be assumed that, this range of cell sizes represent the flow path length accurately. Due to reduced delivery ratio, sediment yield is under predicted for larger cell sizes when compared to the yields obtained for smaller cell sizes. It is clear from this study, that the cell discretization that produces smallest cell size is not necessarily the best and the cell discretization that shows less variation in sediment yield to change in flow path length is appropriate for use in the simulation. In this simulation study, average cell size of 2.0 ha was used. Range of cell sizes varied from 0.3 ha to 16.5 ha with a standard deviation of 2.2.

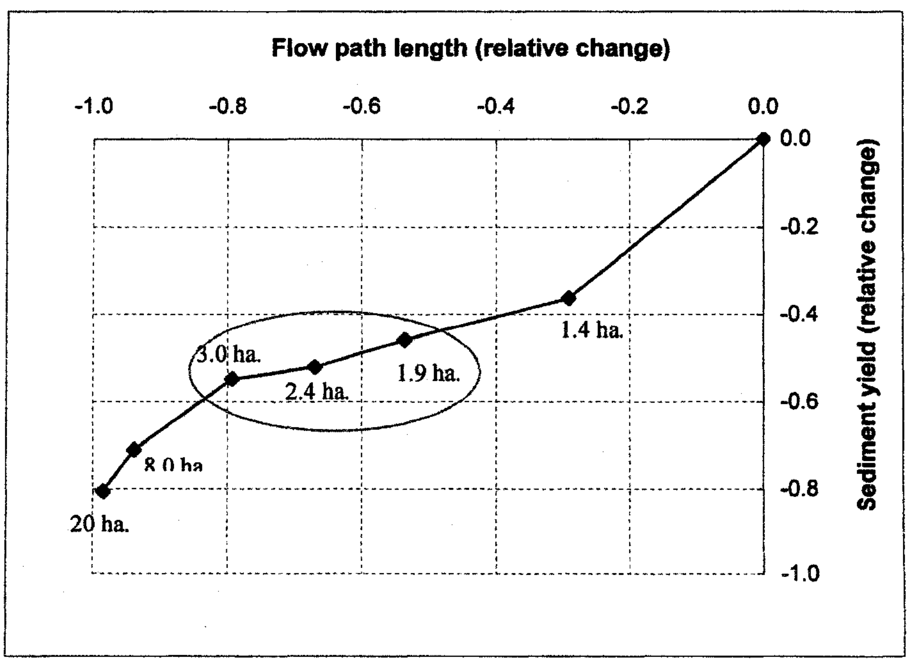


Figure 5.3: Sensitivity of sediment yield to flow path length

It can be concluded that, sediment yield is highly sensitive to the scale of cell discretization and hence estimating the sediment yield without considering the effect of cell discretization, could drastically alter the decisions made concerning non-point source pollution control. Clearly, cell size selection shall not be done arbitrarily and should be based on the scale necessary to capture the spatial variability. Vieux and Needam (1993) suggested that the grid cell sizes should be chosen such that the flow path lengths in the drainage network are closely approximated.

5.2.2 Effects of the soil, crop and other selected parameters

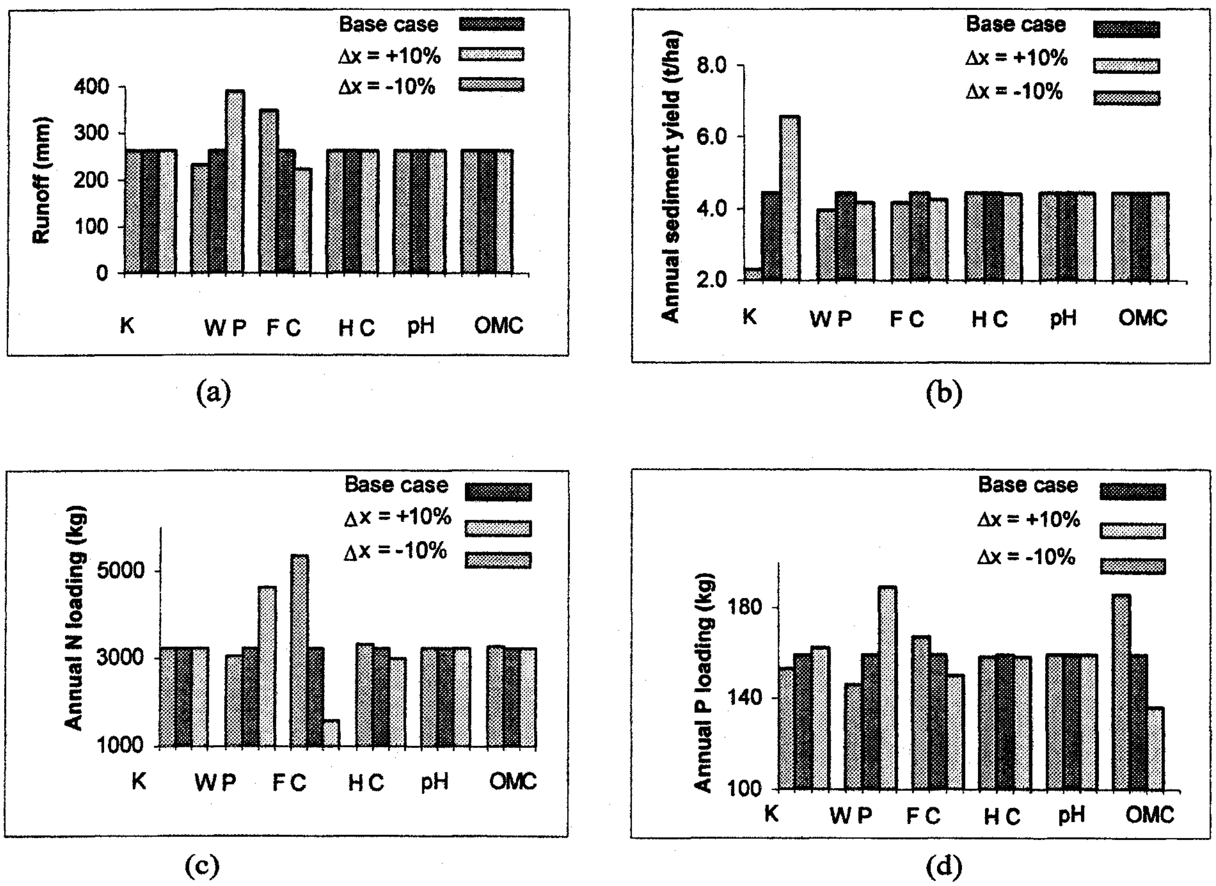
Results of the sensitivity analysis is shown in Table 5.2. Sensitivity index, I in Table 5.2 indicates that the effect of change of the parameter on the simulation results is small and sensitivity index IV indicates that the effect of change of the parameter on the simulation results very high. Indices II and III indicate medium and high sensitivity, respectively.

Table 5.2: Sensitivity indices

Parameter	Runoff	Sediment	N load	P load
K factor	I	III	I	I
Wilting point	IV	II	III	II
Field capacity	III	II	IV	II
Hydraulic conductivity	I	I	III	I
pH value	I	I	I	I
Organic matter content	I	I	I	III
Organic N	I	I	I	III
Inorganic N	I	I	II	III
Plant N uptake - Mix code "Y"	I	I	II	I
Plant P uptake - Mix code "Y"	I	I	I	I
Plant N uptake - Mix code "N"	I	I	I	III
Plant P uptake - Mix code "N"	I	I	III	III
Surface roughness	I	I	I	I
CN (+50% within range)	I	I	III	I
CN (+100% within range)	II	I	IV	I

a. Soil properties

Sensitivity indices indicate that surface runoff is most sensitive to change of wilting point. Wilting point represents the fraction of water volume at wilting point to the soil volume in the soil layer. Effect of change in soil properties on runoff, sediment yield, N and P loading are shown in figure 5.4. It can be seen from 5.4.(a), that increase in wilting point and decrease in field capacity increased runoff by over 35%. As the wilting point is increased, less moisture is required by the soil layer to reach to the field capacity resulting more runoff.



Note: In these figures: K = K factor, WP = Wilting point, FC = Field capacity, HC = Hydraulic conductivity, pH = Soil pH value, OMC = Soil organic matter content.

Figure 5.4: Effect of change in soil properties on the (a) runoff, (b) sediment, (c) N loading and (d) P loadings

The same phenomena occur when the field capacity is lowered, resulting in a higher runoff. An increase in field capacity and decrease in wilting point resulted in a lower runoff since more moisture is needed to be absorbed by the soil layer before producing any runoff.

From figure 5.4(b), it can be seen that the sediment yield is highly sensitive to the K factor. K factor represents the ease of soil particle to detach from the soil surface and the transportability of the sediment. Change in K factor by $\Delta x = \pm 10\%$ resulted in annual sediment yields of 6.6 tons/ha and 2.3 tons/ha, respectively. The base annual sediment yield is estimated as 4.4 tons/ha. Wilting point and field capacity showed a moderate effect on the soil erosion. Increase in wilting point and decrease in field capacity has an effect on soil moisture which in turn increase the runoff and hence the sediment detachability and transportability. Other parameters tested showed low to negligible effect on the surface runoff and sediment yield.

Figure 5.4 (c) and (d) shows that field capacity and wilting point has a greater effect on nutrient loss that is resulting from the increased runoff where surface runoff is the major transport mechanism for nutrient loading. Hydraulic conductivity of the soil showed a moderate effect on the nutrient loadings while organic matter content showed a greater effect on P loading. Soils with high organic matter content, tends to reduce overland flow by improved non-capillary porosity thus reducing transport of nutrient loading. Soil moisture has a substantial influence on the infiltration capacity of the soil and the surface runoff, thus on the soil erosion. When the soil profile is at or near field capacity, soil moisture is maximum and infiltration capacity is less, thus more runoff and more soil erosion occurs. When the soil profile is near to the wilting point, then the soil moisture is minimum and thus less or no runoff and erosion are expected. Soil water movement is affected by hydraulic conductivity and transmissivity that are related to particle size, morphological properties (such as bulk density, organic matter content) and water retention properties.

In conclusion, soil related parameters such as; field capacity, wilting point, K factor and hydraulic conductivity are to be selected with a greater accuracy.

b. Crop related parameters

Effects of change in crop related properties on runoff, sediment yield, N and P loading are shown in figure 5.5. Cases 1 to 9, shown in the figures are described as:

Case 1: Organic N was tested by using 10% and 50% of the value used in the base case.

Case 2: Inorganic N was tested by increasing base case value by +50% and +100%.

Case 3: Plant N uptake was tested by increasing base case value by +50% and +100%.

Case 4: Plant P uptake was tested by increasing base case value by +50% and +100%.

Case 5: Plant N uptake was tested with fertilizer mixing code set to "N".

Case 6: Plant P uptake was tested with fertilizer mixing code set to "N".

Case 7, 8 and 9: Plant N and P uptake was tested by changing values for soybean, wheat and corn respectively.

It can be seen, from figures 5.5 (a) and (b), crop related properties have little to no effect on runoff and sediment yield. P loadings are found to be highly sensitive to initial amount of organic and inorganic N in the soil. In case 1, as organic N amount was reduced to 10% and 50% of the default value of 500 ppm, P loading increased by about 70%. The same pattern was observed in case 2. As inorganic N amount was increased by 50% and 100% of its base default value of 5 ppm, loadings increased by about 70%. As plant N uptake increased, in case 3, N loading reduced by nearly 25% while no effect was shown on P loading.

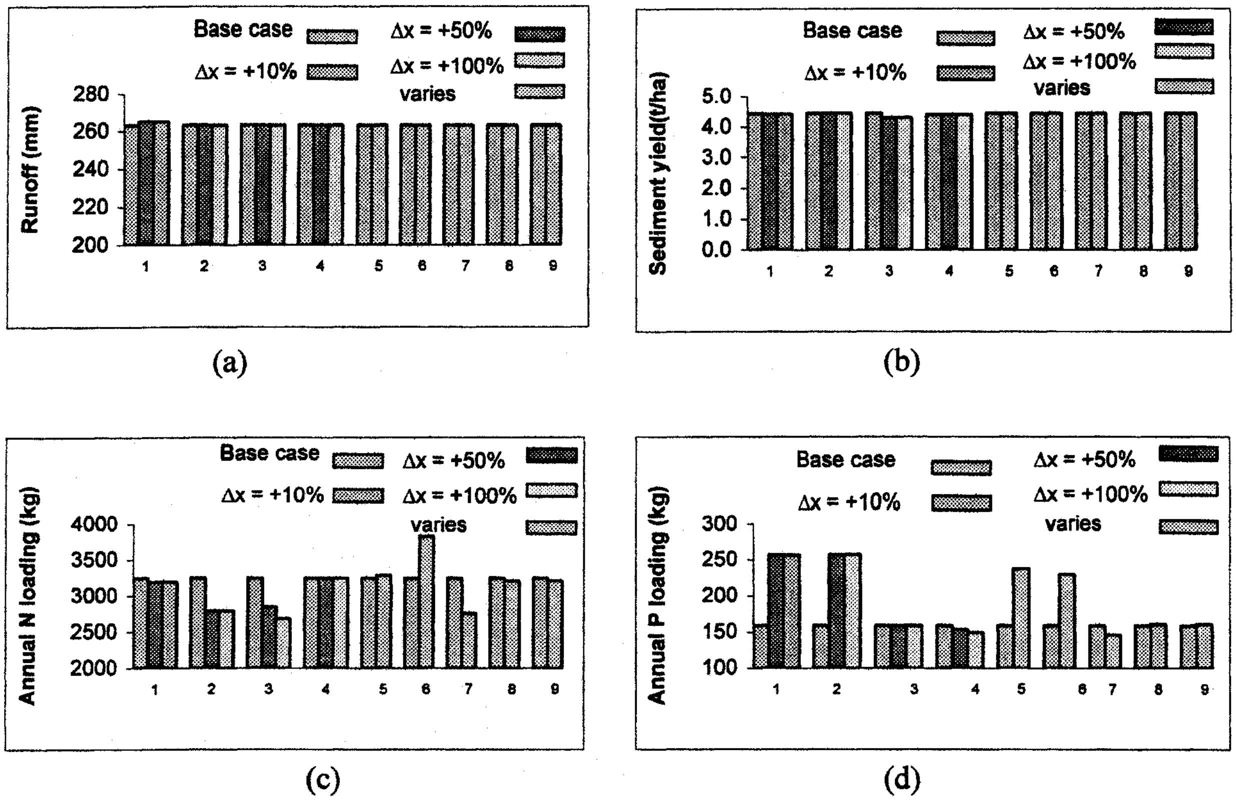


Figure 5.5: Effect of change in crop related properties on the runoff, sediment, N and P loadings

In case 4, as plant P uptake was increased by 50% and 100%, P loading was decreased by about 5% while no effect is observed on the N loading. Cases 5 and 6 indicate that, fertilizer mixing code has a greater effect on N and P loading which depict that fertilizers are not mixed well within the depth of fertilizer application thus allowing to move with surface runoff.

It can be noted from cases 7, 8 and 9, that N and P loading from surface runoff is most sensitive to soybean plant uptake and less sensitive to wheat and corn plant uptake. Usually, little or no fertilizer is applied to soybean fields. Soybean obtains up to 75% of plant N from soil residual nitrate and from soil organic matter (Ferguson et al., 2002). This may be the reason that soybean N uptake was the most sensitive parameter for N loss, which is very sensitive to the amount of fertilizer applied Yuan et al. (2003).

c. Surface roughness and CN

Effect of change in surface roughness and CN on runoff, sediment yield, N and P loading are shown in figure 5.6. As surface roughness (SR) is increased by 10% from its base value, variation showed a little decrease in runoff, sediment yield, N and P loadings. As the change is further increased to 20%, as can be seen from figure 5.6, the effect was considerably large on sediment yield and P loadings.

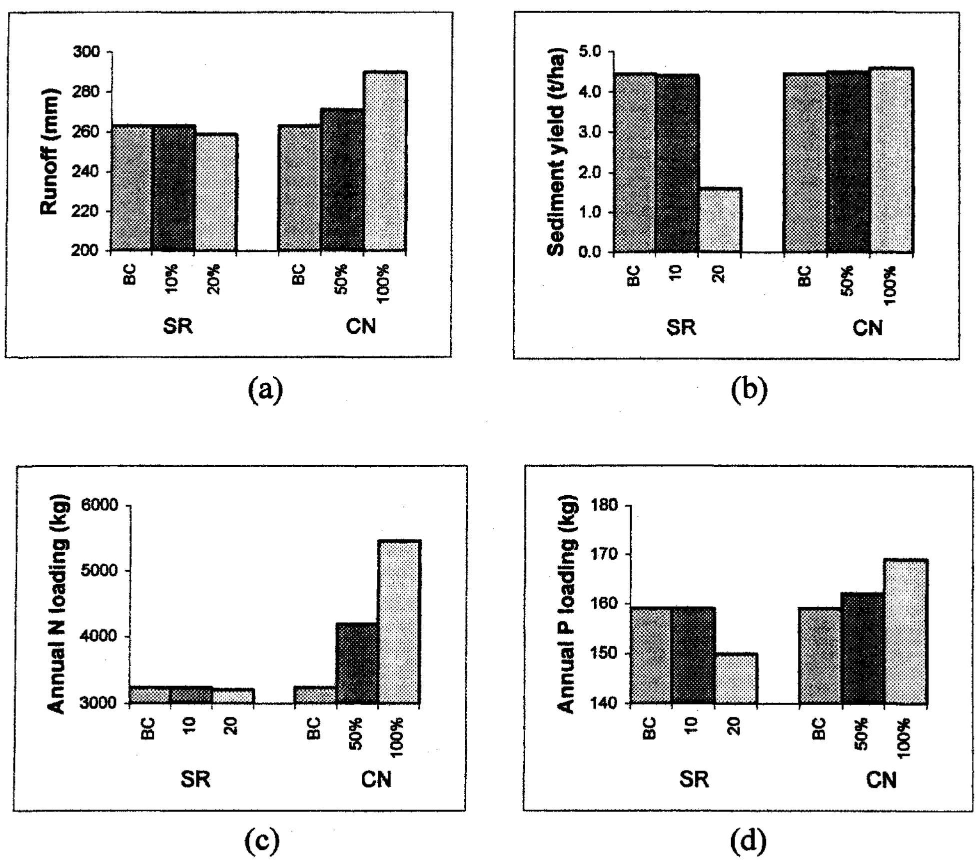


Figure 5.6: Effect of change in surface roughness and CN on (a) runoff, (b) sediment, (c) N loading and (d) P loading

Surface random roughness is mostly affects on sediment yield and P loading as can be seen from figure 5.6 (b) and (d). A rough surface with depressions acts as barriers and trap water and sediment, thus reducing detachment by surface runoff and sediment transport (Renard et al, 1997). Runoff CN's are varied by 50% and 100% of their base values within the valid range as described in section 4.4. Increase in CN, showed a moderate, about 10%, increase in runoff as shown in figure 5.6(a). As indicated in Table 4.6, increase in CN values mostly occurred in the soil group, which has low runoff potential and a high infiltration potential. From figure 5.6 (b), it can be seen that the effect of change in CN on sediment yield is minimal. Though the effect of change in CN on P loading is moderate, it showed a high sensitivity on N loading. Changing CN by 50% and by 100% within the range showed an increase in N loading by 30% and 65%, respectively.

5.3 RUNOFF VOLUME, SEDIMENT AND NUTRIENT LOADINGS

The AnnAGNPS simulation is performed to predict runoff volume, sediment, total N and total P loadings at the outlet of Muddy Creek watershed. Randomly distributed crop pattern with crop rotations is considered in the simulation. Rainfall and climate variables are obtained from the closest weather station, located at Kingsville in Essex County, which is 17 km away from the Muddy Creek watershed. There are no flow data available to be compared with predicted simulated loadings. However the data from the Provincial Water Quality Monitoring Network (PWQMN) site located closer to the watershed outlet is used to compare simulated total N and P loadings. Since no flow data is available, the predicted loads were converted into concentrations, based on the predicted runoff at the watershed outlet.

5.3.1 Water loading – Runoff volume

The average annual precipitation for the simulation period is 856 mm and average annual water loading at the outlet is 1,783,995 m^3. This results in an average annual water yield of 275 mm. Average annual precipitation and runoff are presented in Table 5.3. The average annual runoff is approximately 32% of the total precipitation for the simulation period. The model runoff is considered little under-predicted compared to the nearby watersheds, which are typically on the order of 35% to 40% of the total annual precipitation. AnnAGNPS does not account for base flow and this could be a reason for this under prediction.

Table 5.3: Average annual precipitation and runoff

Year	Precipitation (mm)	Runoff (mm)	% Runoff (mm)
1991	877	345	39
1992	1021	429	42
1993	814	298	37
1994	904	328	36
1995	812	257	32
1996	968	324	33
1997	915	317	35
1998	781	251	32
1999	800	201	25
2000	871	185	21
2001	721	182	25
2002	703	200	28
2003	991	294	30

Figure 5.7 shows the simulated average annual runoff at the watershed outlet and the observed average annual precipitation for the simulation period.

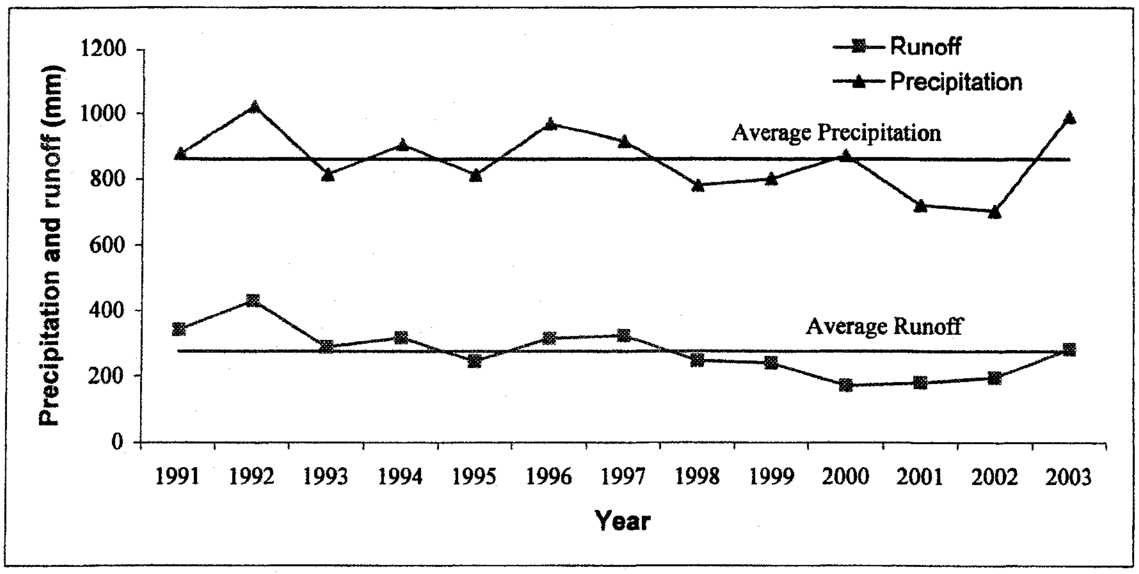


Figure 5.7: Variation of average annual precipitation and average annual runoff

The simulated average monthly runoff at watershed outlet and observed average monthly precipitation for the simulation period are plotted as shown in Figure 5.8. The runoff variation pattern agrees with the precipitation variation pattern except for the last two years of the simulation period. Similarly, on the average, peaks and troughs in monthly variations of predicted runoff agree with the monthly precipitations as shown in Figure 5.8. It can be observed that for smaller precipitation events, the model had produced very low or no surface runoff since the runoff is highly sensitive to antecedent conditions.

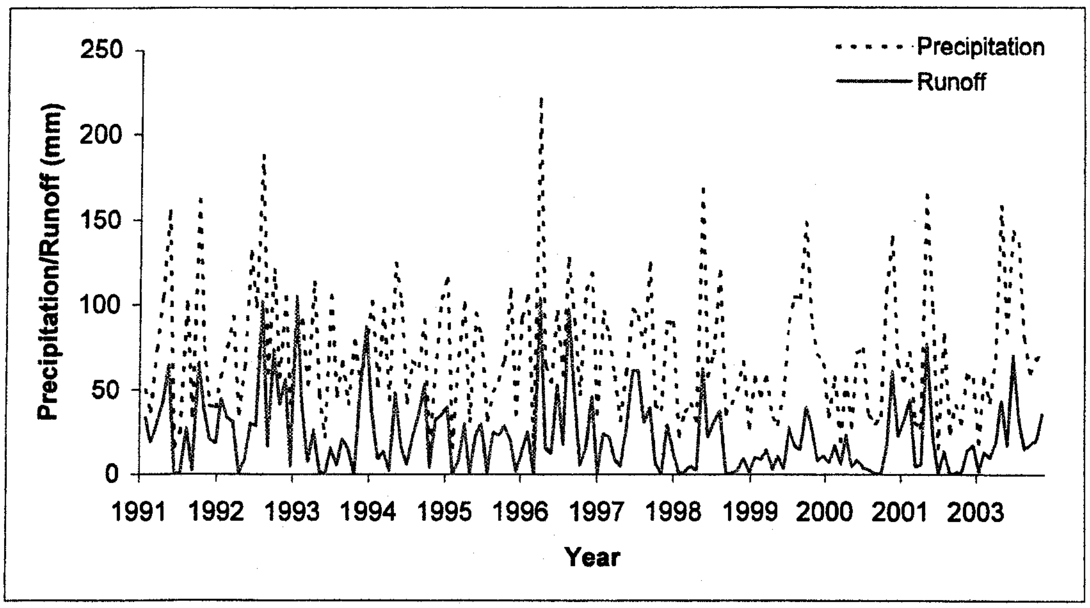


Figure 5.8: Variation in monthly precipitation and monthly runoff

The simulated runoff from AnnAGNPS cells can be used to describe the contribution of runoff from the various locations within the watershed. Figure 5.9 depicts the spatial distribution of annual water yield from different cells. Though the model predicted watershed annual water yield is averaged to 275 mm, it can be noted from this figure that the areas at the southern part of the watershed have produced a yield greater than 300 mm.

These higher runoff rates may be attributed to the presence of Brookston Clay, which has a low infiltration and high runoff potential. Areas designated with sandy soils (Plainfield Sand), which has poor runoff potential, produced a runoff of 225 mm or less which is much less than the average annual water yield.

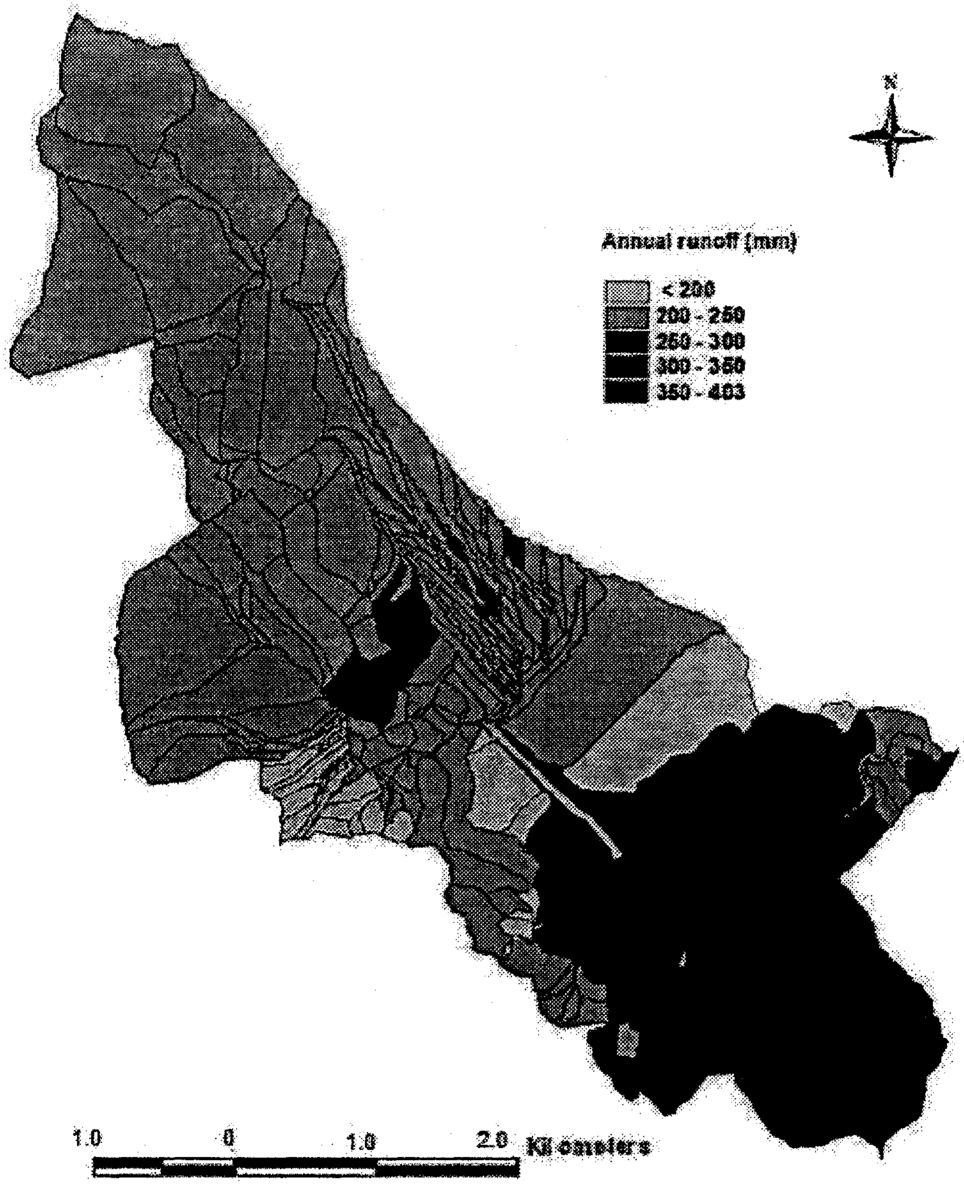


Figure 5.9: Spatial distribution of average annual water yield

Areas having Brookston clay loam and Berrien sandy loam produced moderate average annual water yield. Average annual water loading at the outlet is approximately 1.8 million m^3.

5.3.2 Sediment loading

The total annual sediment load at the watershed outlet is obtained as 2739 tons. This results in an annual sediment yield of 4.2 tons/ha. This sediment load is mainly contributed by sheet and rill erosion and gully erosion. Table 5.4 presents simulated average annual sediment loads at watershed outlet. Sediment loadings produced from the AnnAGNPS simulation at the watershed outlet are fairly comparable to the ones reported in the literature although the watersheds are of different sizes (TRCA, 2003, Bingner et al., 2005).

Table 5.4: Average annual sediment loading at the watershed outlet

Year	Runoff (mm)	Sediment load (tons)	Sediment yield (tons/ha)
1991	345	2931	4.5
1992	429	1944	3.0
1993	298	1957	3.0
1994	328	3224	5.0
1995	257	2181	3.4
1996	324	2700	4.2
1997	317	3366	5.2
1998	251	3438	5.3
1999	201	1931	3.0
2000	185	3192	4.9
2001	182	1189	1.8
2002	200	1713	2.6
2003	294	5840	8.9

Variation of average monthly sediment loadings for the simulation period is shown in Figure 5.10. On the average, peaks and troughs in monthly simulated sediment load agrees with the variation of simulated monthly runoff.

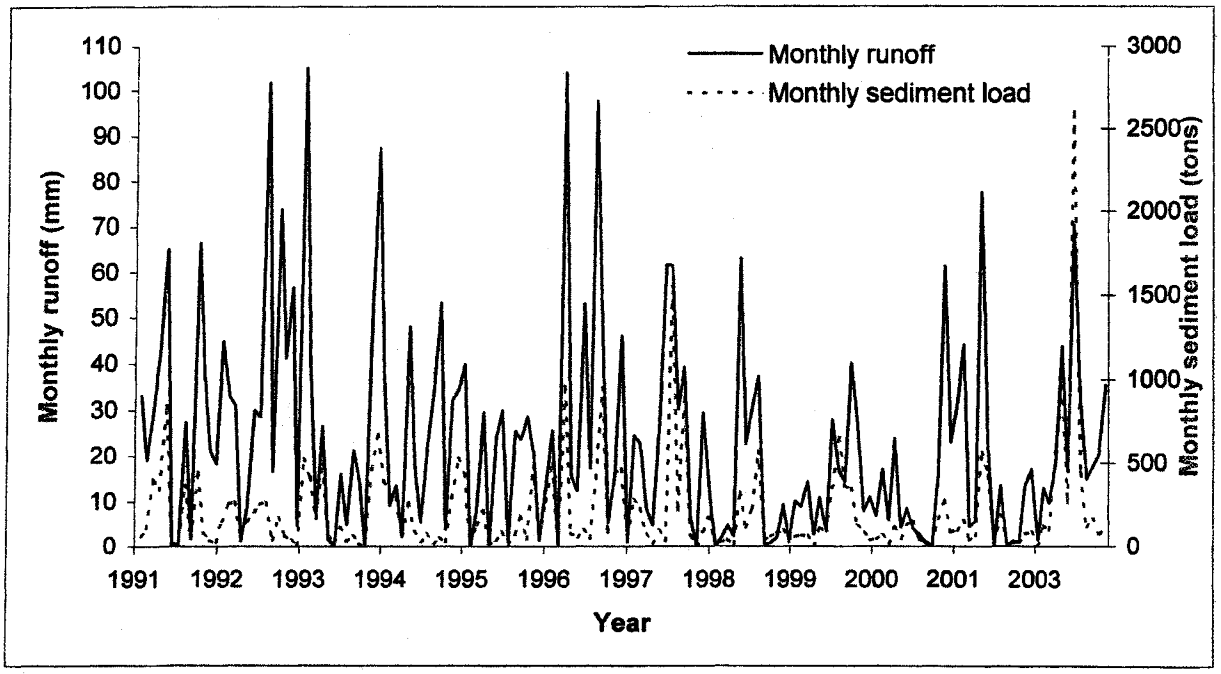


Figure 5.10: Average monthly sediment load at the watershed outlet

Simulated sediment loads from AnnAGNPS cells can be used to describe the contribution of sediment load from the various locations within the watershed. Figure 5.11 presents the annual sediment yielding from cells to reach segments. A significant portion of the sediment loadings occurs from cells that are along the main channel. The soils along the main channel consist of typical floodplain soils, which include clay and silty clay loam. These soils produce a high amount of sediment erosion and consequent loadings to the receiving waterbodies.

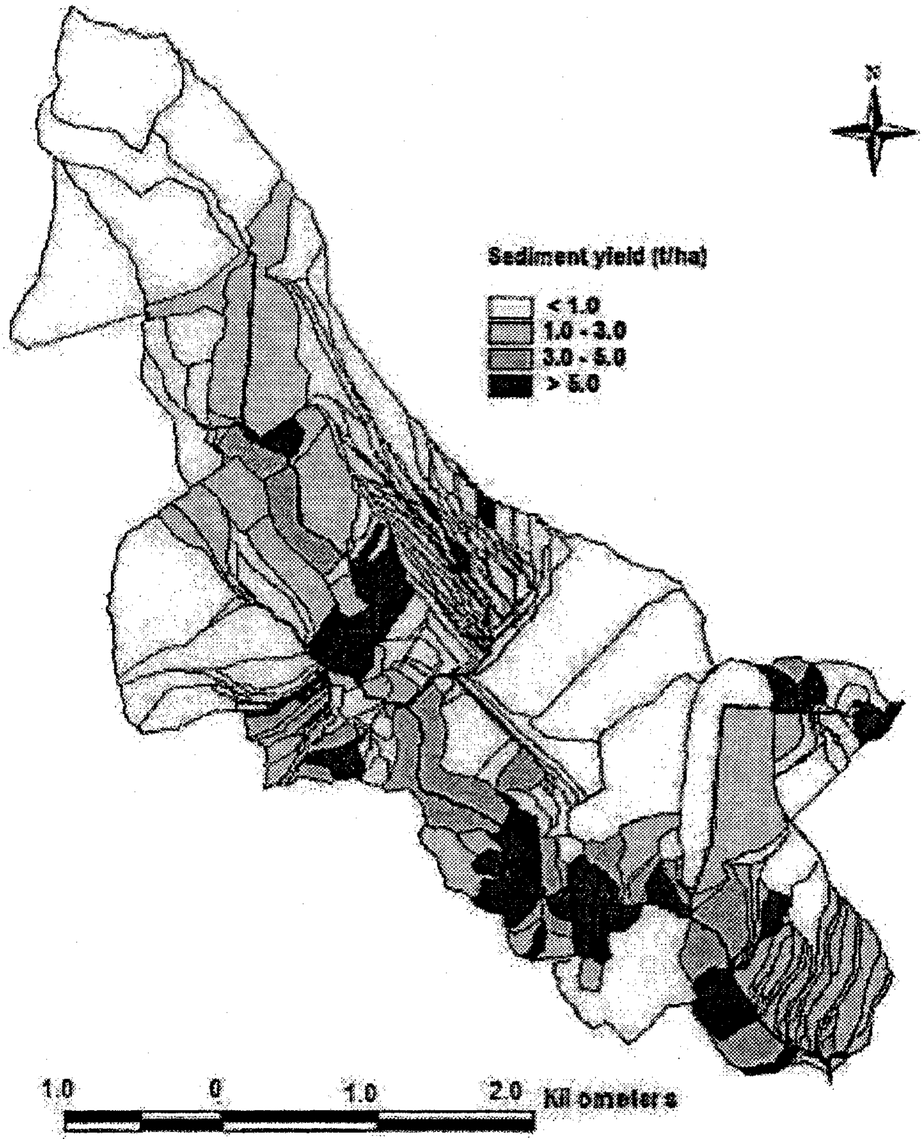


Figure 5.11: Average annual sediment yield (tons/ha/year) from cells to the watershed outlet

Figure 5.12 represents the average monthly sediment concentrations at the watershed outlet. The highest sediment concentration of nearly 110 mg/l occurs during the period of low flow following a heavy precipitation in 1993. Average sediment concentration for the simulation period is approximately 30 mg/l.

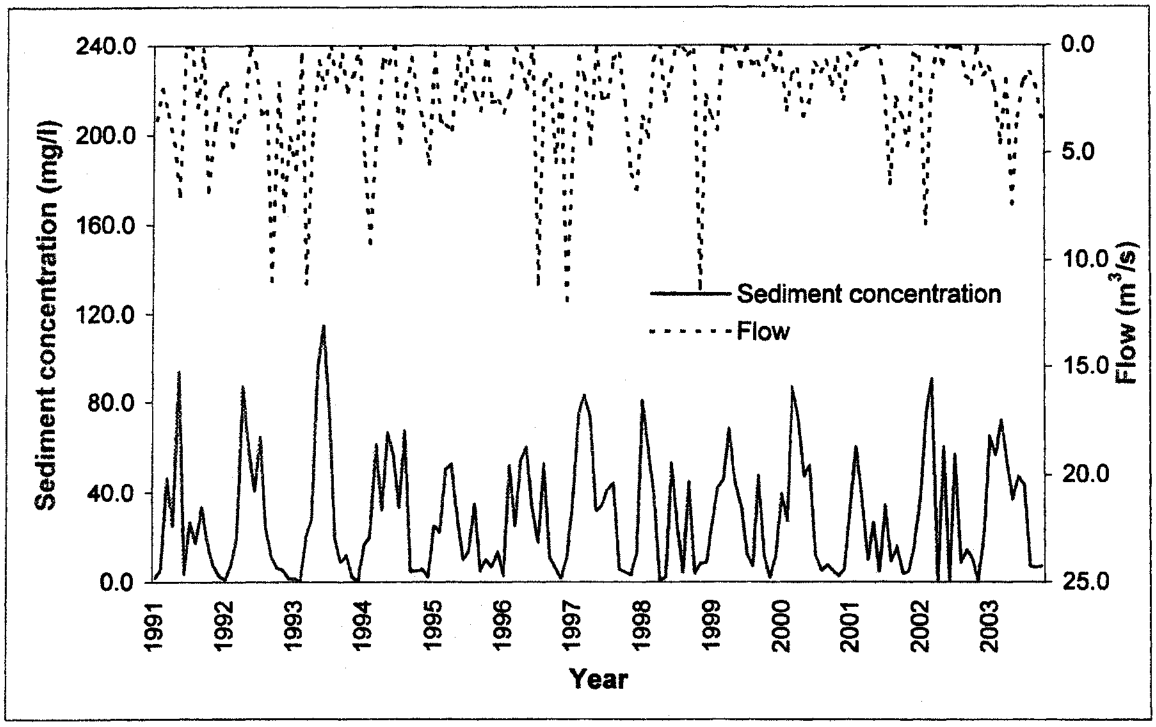


Figure 5.12: Monthly sediment concentrations at the watershed outlet

5.3.3 Nutrient loading

Simulated average annual total N and P loadings at the watershed outlet are 3773 kg and 786 kg respectively. These numbers result in N and P yield at the watershed outlet as 5.8 kg/ha and 1.2 kg/ha, respectively. These values are somewhat less than the anticipated loading rates. Since the results obtained are based on the existing data and knowledge, there has been no effort made to calibrate the model and hence these results are indicative only. Simulated annual total N and P loadings at watershed outlet are presented in Table 5.5.

The simulated monthly total N loadings are plotted with avreage monthly runoff in Figure 5.13. As can be seen from this figure, the monthly simulated nitrogen loading did not match well with the simulated runoff.

Table 5.5: Annual total N and P loading

Year	Nitrogen load (kg)	Phosphorous load (kg)
1991	861	508
1992	2450	284
1993	4876	363
1994	8242	1413
1995	5178	535
1996	7187	688
1997	767	1085
1998	4793	1197
1999	1323	966
2000	6423	885
2001	4428	878
2002	1360	673
2003	1158	732

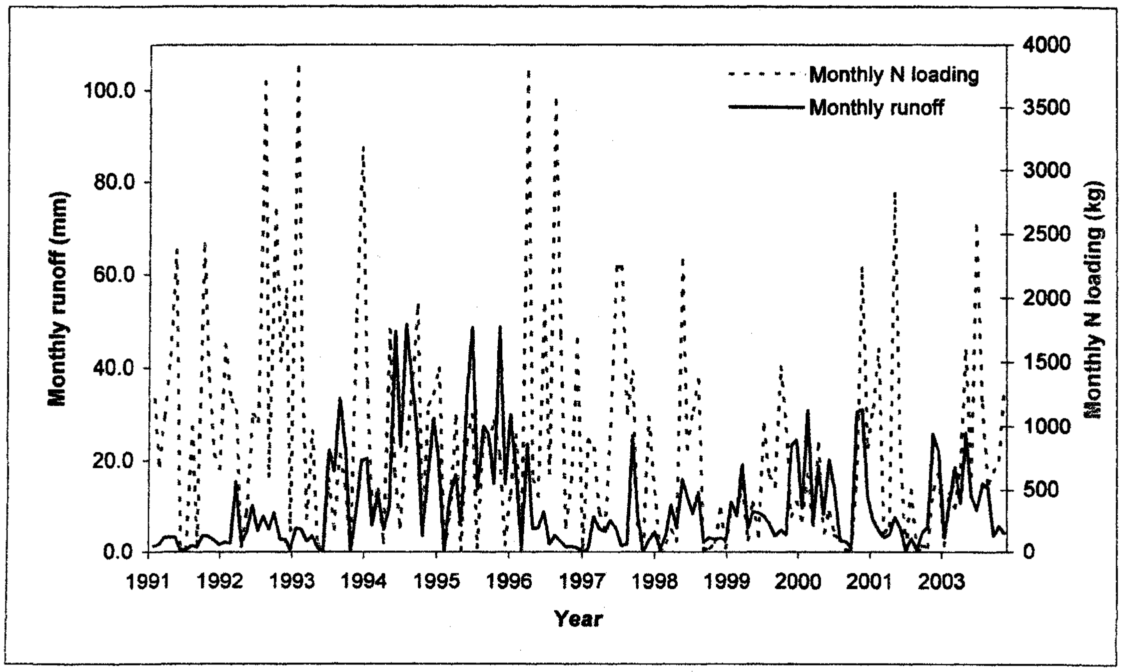


Figure 5.13: Average monthly N load at the watershed outlet

Figures 5.14 shows the average monthly N and P concentrations at the watershed outlet. It can be seen from this figure that during some months average monthly N and P concentrations exceeded 40 mg/l and 18 mg/l, respectively. Average N and P concentrations for the simulation period are 3.5 mg/l and 0.7 mg/l, respectively.

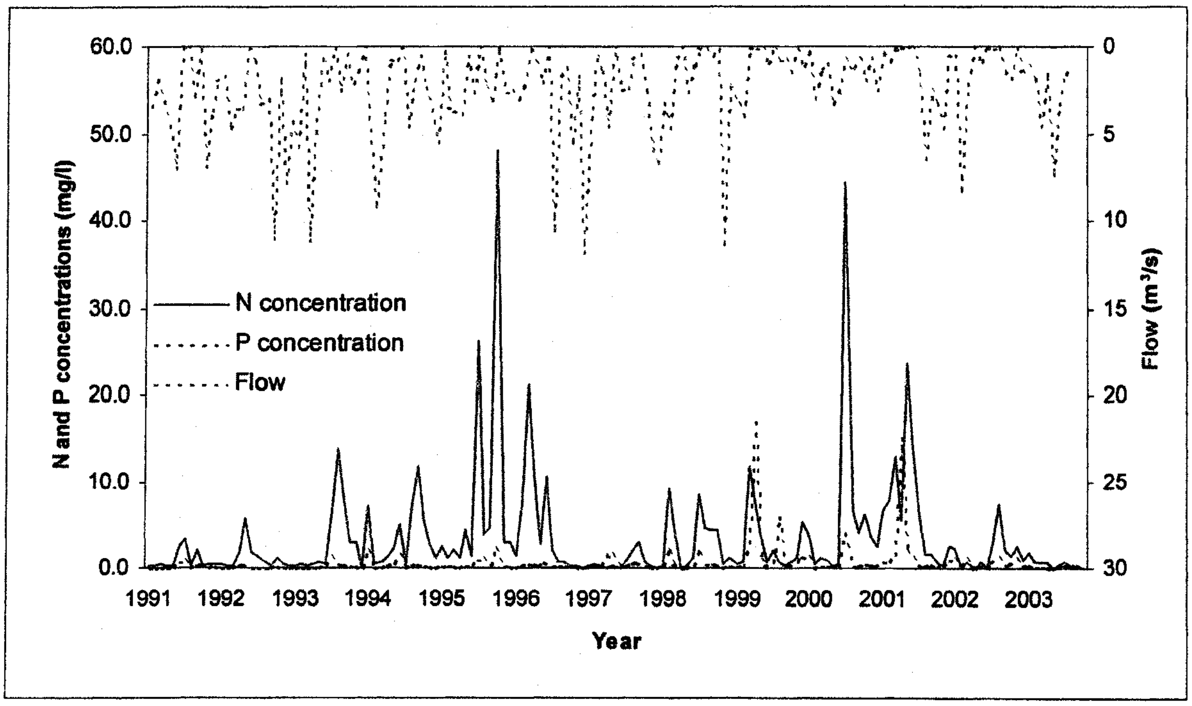


Figure 5.14: Average monthly N and P concentrations at the watershed outlet

Simulated nutrient loadings from AnnAGNPS cells can be used to describe the contribution of N and P loads from the various locations within the watershed. Figure 5.15 represents the areas that are contributing N loading to the watershed outlet. Areas in the southern part of the watershed yielded greater than 8 kg/ha of N, while the northern part of the watershed has yielded of 3.0 to 5.0 kg/ha of N in an average year during the simulation period.

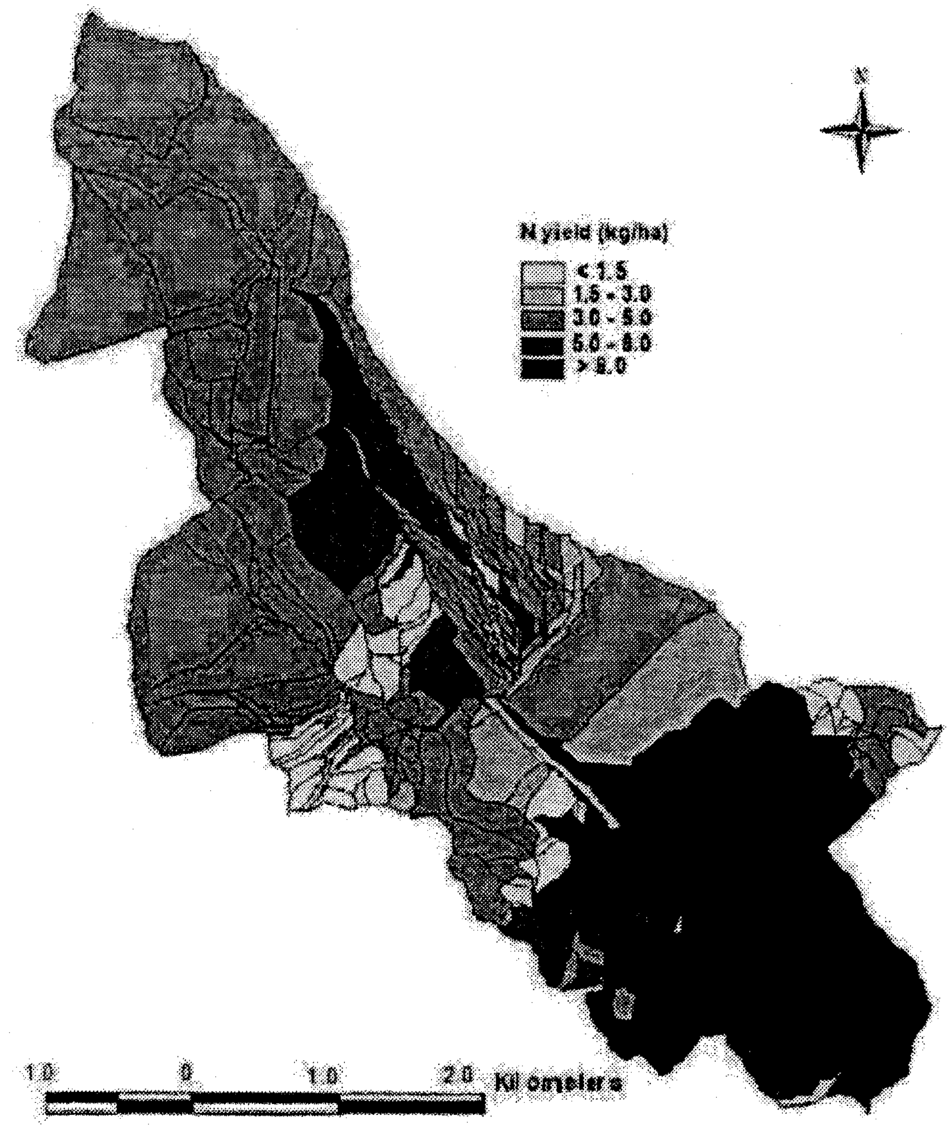


Figure 5.15: Average annual N yield (kg/ha/year) from cells to the watershed outlet

Average annual P yields are represented in Figure 5.16. It can be observed that more than 60% of the watershed area yield less than 0.5 kg/ha of P, annually. Nearly 15% of the watershed area yields 1.2 to 2.2 kg/ha annually. Pockets at the southern part of the watershed yield more than 3.0 kg/ha annually.

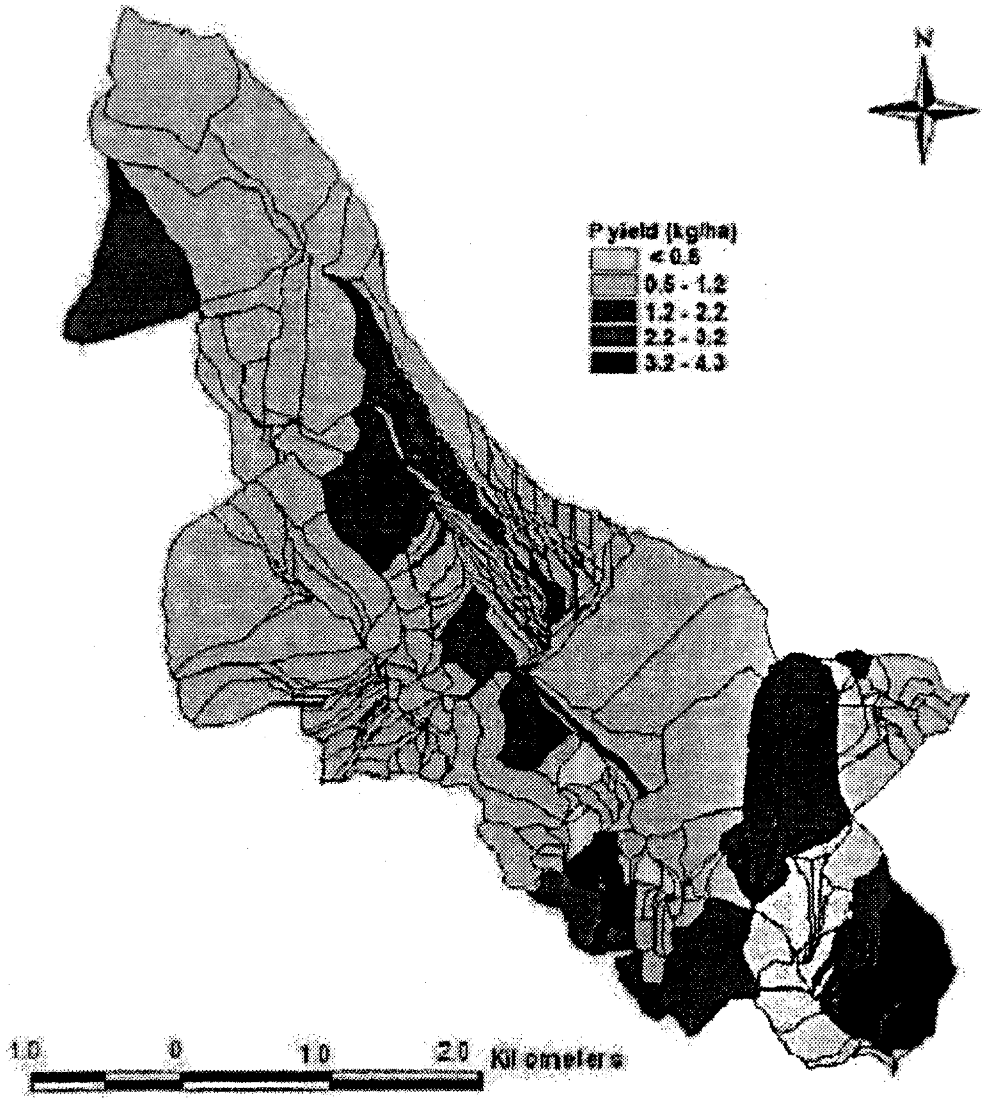


Figure 5.16: Average annual P yield (kg/ha/year) from the cells to the watershed outlet

There are no continuously monitored data available to be compared with predicted loadings. However, the data from the Provincial Water Quality Monitoring Network (PWQMN) site located close to the watershed outlet is used to compare simulated total N and P loadings. Since no flow data is available, the predicted loads were converted into concentrations, based on the predicted event runoff at the watershed outlet. The simulated/calculated and observed nitrogen and phosphorous concentrations are presented in Table 5.6 and Table 5.7, respectively for five runoff events in the year 2003.

It can be seen that simulated/calculated nitrogen concentration does not match well with the PWQMN observed nitrogen concentrations. One reason would be that N concentrations are calculated based on the average daily flow and average N loading during that day. The concentration due to previous day's N loading is not considered.

Table 5.6: Comparison of simulated/calculated total N concentrations with PWQMN data

Date	N concentrations (mg/l)	
	Observed	Simulated/calculated
6/18/2003	1.46	1.22
7/21/2003	2.18	2.45
8/13/2003	0.84	0.39
9/17/2003	0.95	0.31
10/22/2003	1.67	0.76

Table 5.7: Comparison of simulated/calculated total P concentrations with PWQMN data

Date	P concentrations (mg/l)	
	Observed	Simulated/calculated
6/18/2003	0.59	0.96
7/21/2003	1.83	1.16
8/13/2003	2.04	0.26
9/17/2003	1.10	0.21
10/22/2003	0.45	0.62

It can be observed that the P concentrations observed from the modeling study are generally lower than the long term monitoring data at the watershed outlet. PWQMN monitoring is usually done during low flow seasons resulting in more nutrient concentrations in the downstream waters. Significant increase in P concentrations could be expected by incorporating the septic discharge information for a given flow condition. Also, actual nutrient management data such as, types of fertilizers used in the watershed, fertilizer application rates and dates of application would produce more reliable results of the nutrient concentrations.

5.3.4 Summary

The predicted runoff volume (Table 5.3) during early years of the simulation was fairly acceptable while during the latter part of the simulation period, the water loadings were under predicted. On the average, predicted runoff volume is within the lower margin of acceptable limit. Sediment loading at the outlet is within the acceptable range and it is fairly comparable with the loadings produced from the watersheds with similar nature found in the literature. Predicted nutrient loadings are compared with the long term average concentration values of N and P observed at a station located within the watershed. Predicted N concentration can be considered within the acceptable lower limit, since the observed concentrations correspond to low flow and high concentration situations. Also, the faulty septic system that existed in the watershed must have contributed to the observed data causing the difference. Summary of the results for runoff, sediment and nutrient loadings from AnnAGNPS simulation at the watershed outlet are presented in Table 5.8.

Table 5.8: Summary of simulation results at the watershed outlet

Item	Amount	Units
Annual average rainfall	856	mm
Annual average runoff	275	mm
Annual average rainfall : runoff ratio	32	%
Water loading	1.8	million-m^3/year
Sediment loading	2,739	t /year
Sediment loading rate	4.2	t /ha/year
Phosphorous (P) loading	786	kg/year
Phosphorous (P) yield	1.2	kg/ha/year
Nitrogen (N) loading	3,773	kg/year
Nitrogen (N) yield	5.8	kg/ha/year

5.4 EFFECTIVENESS OF BEST MANAGEMENT PRACTICES

5.4.1 Vegetative Filter Strips (VFS)

From the model runs, the simulated sediment and nutrient loadings are obtained and tabulated in Table 5.8. It was observed that a significant portion of the sediment loading occur from cells that are along the main stream (Figure 5.11). Therefore, in order to assess the effectiveness of BMP's in reducing sediment and nutrient loadings to downstream waters, VFS are modeled mainly along the main stream. The stream system was divided into five segments in order to prioritize the effectiveness of the buffer placement (Figure 5.17). Buffer placement was modeled first by placing VFS to section 1 and then to sections 1 and 2 and continued until it covered all the five sections. Sediment, runoff and nutrient loadings were obtained at watershed outlet for five different buffer placement scenarios. Efficiency of buffer placement was obtained by calculating the amount of sediment and nutrient filtered by five different buffer placement scenarios.

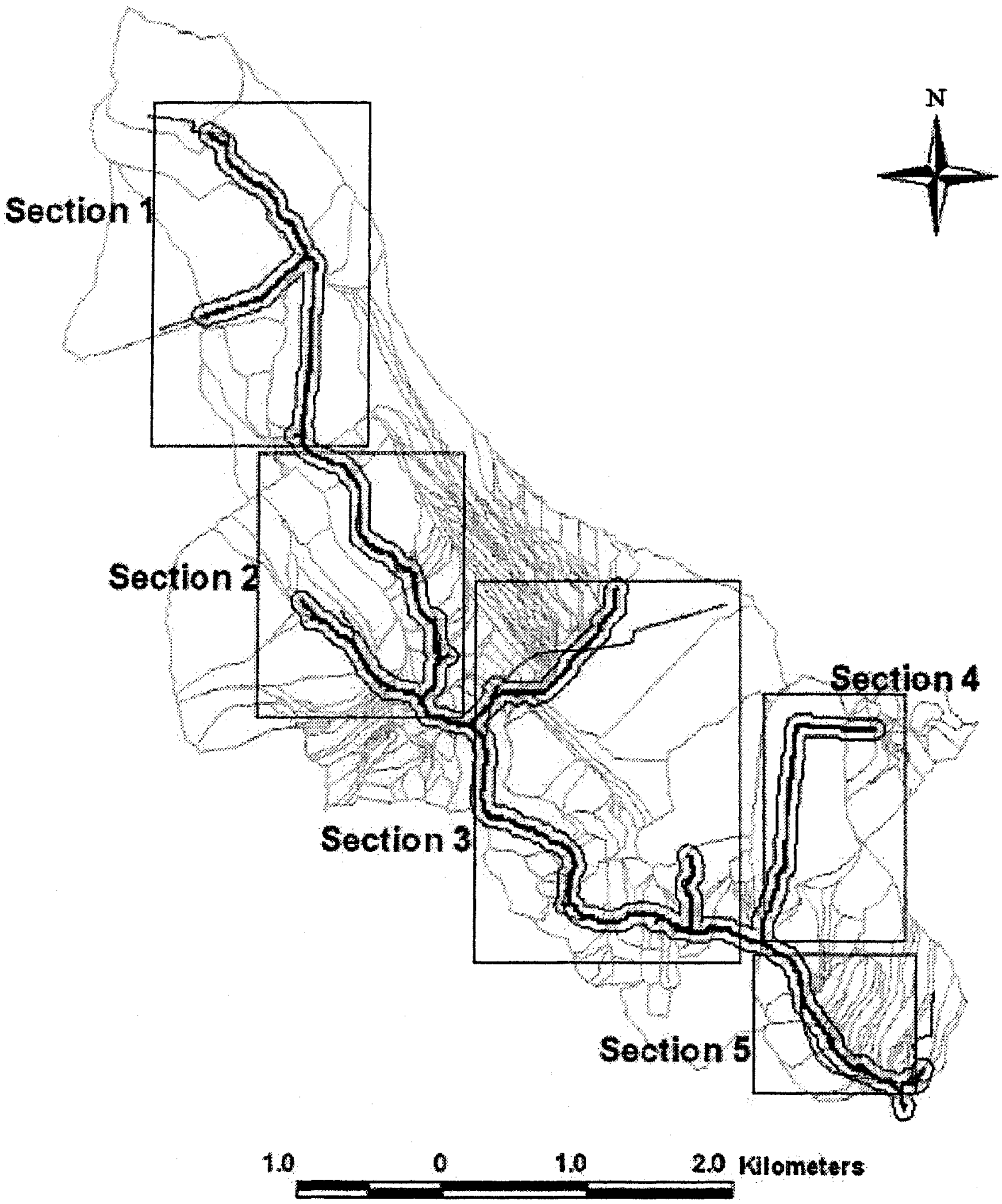


Figure 5.17: Stream segment division for modeling vegetative buffer strips

Analysis results for five vegetative buffer strip modeling scenarios, is summarized in Table 5.9 (a) to 5.9 (e). Placement of buffer strips along all the stream segments reduced annual average sediment loading at watershed outlet by 20.6 %. Placement of buffers only along stream segment 3, reduced sediment loading at watershed outlet by 6 % while placement of buffers along stream segments 2 and 3 together reduced sediment loading by 11.3 %. Lowest amount of sediment filtered by the buffers placed around stream segment 4 and this is due to less sediment loading from this area as shown in Figure 5.11. Buffer placement efficiency, in reducing the sediment loading at the watershed, outlet can be ranked in ascending order as section 3 (6%), section 2 (5.3%), section 1 (4.6%), section 5 (3.2%) and section 4 (1.5%), respectively.

Table 5.9 (a): Reduction in loadings due to buffer placement in stream segment – section 1

Item	Without buffers	With buffers in section 1	Amount filtered	% Reduction
Annual average runoff (mm)	275	266	9	3.3
Sediment loading (t/year)	2,739	2,613	126	4.6
Sediment loading rate (t/ha/year)	4.20	4.02		
Phosphorous (P) loading (kg/year)	786	775	11	1.4
Phosphorous (P) yield (kg/ha/year)	1.20	1.19		
Nitrogen (N) loading (kg/year)	3,773	3,540	233	6.2
Nitrogen (N) yield (kg/ha/year)	5.80	5.45		

Table 5.9 (b): Reduction in loadings due to buffer placement in stream segment – sections 1 and 2

Item	Without buffers	With buffers in sections 1 and 2	Amount filtered	% Reduction
Annual average runoff (mm)	275	259	16	5.8
Sediment loading (t/year)	2,739	2,467	272	9.9
Sediment loading rate (t/ha/year)	4.20	3.80		
Phosphorous (P) loading (kg/year)	786	731	55	7.0
Phosphorous (P) yield (kg/ha/year)	1.20	1.13		
Nitrogen (N) loading (kg/year)	3,773	3,389	384	10.2
Nitrogen (N) yield (kg/ha/year)	5.80	5.22		

Table 5.9 (c): Reduction in loadings due to buffer placement in stream segment – sections 1,2 and 3

Item	Without buffers	With buffers in sections 1,2 and 3	Amount filtered	% Reduction
Annual average runoff (mm)	275	252	23	8.4
Sediment loading (t/year)	2,739	2,304	435	15.9
Sediment loading rate (t/ha/year)	4.20	3.55		
Phosphorous (P) loading (kg/year)	786	679	107	13.6
Phosphorous (P) yield (kg/ha/year)	1.20	1.05		
Nitrogen (N) loading (kg/year)	3,773	3,002	771	20.4
Nitrogen (N) yield (kg/ha/year)	5.80	4.62		

Table 5.9 (d): Reduction in loadings due to buffer placement in stream segment – sections 1,2,3 and 4

Item	Without buffers	With buffers in sections 1,2,3 and 4	Amount filtered	% Reduction
Annual average runoff (mm)	275	247	28	10.2
Sediment loading (t/year)	2,739	2,263	476	17.4
Sediment loading rate (t/ha/year)	4.20	3.48		
Phosphorous (P) loading (kg/year)	786	645	141	17.9
Phosphorous (P) yield (kg/ha/year)	1.20	0.99		
Nitrogen (N) loading (kg/year)	3,773	2,673	1100	29.2
Nitrogen (N) yield (kg/ha/year)	5.80	4.11		

Table 5.9 (e): Reduction in loadings due to buffer placement in stream segment – sections 1,2,3,4 and 5

Item	Without buffers	With buffers in sections 1,2,3,4 and 5	Amount filtered	% Reduction
Annual average runoff (mm)	275	244	31	11.3
Sediment loading (t/year)	2,739	2,174	565	20.6
Sediment loading rate (t/ha/year)	4.20	3.35		
Phosphorous (P) loading (kg/year)	786	638	148	18.8
Phosphorous (P) yield (kg/ha/year)	1.20	0.98		
Nitrogen (N) loading (kg/year)	3,773	2,485	1288	34.1
Nitrogen (N) yield (kg/ha/year)	5.80	3.82		

Analysis of results showed that placement of VFS reduced N and P loadings at the watershed outlet by 34.1 % and 18.8 %, respectively. Buffer placement around stream segment 3, reduced N loading at watershed outlet by 10.2 % while buffer placement around stream segments 3, 4 and 5 together reduced N loading by 24%. This large reduction is due to the reason that this area has the greatest N yield as shown in Figure 5.15. Figure 5.16 shows the northern part of the watershed yield less P loading to the watershed outlet and hence buffer placement in stream segment 1 only reduced P loading by 1.4 %. Buffer placement in the stream segment 3 showed a reduction of 6.6 % in P loading at the watershed outlet, which is the highest among the five segments. It has also been observed from the analysis that, buffer placement resulted in reducing the annual average runoff by 11.3 % at the watershed outlet.

5.4.2 Tile drainage

Due to the nature of the soils in Muddy Creek watershed, about 80% of the agricultural lands in the watershed are tiled drained. Some of the fields might not be used as cropland if they were not tiled. An effort was made to see the effects on sediment load reduction by considering all the cropland in Muddy creek watershed are tile drained. Table 5.10 presents a comparison of the loadings at the watershed outlet for existing condition and for 100% tile drainage in agricultural areas.

Table 5.10: Comparison of loads at watershed outlet

Item	Existing condition	100% Tile drainage
Annual average runoff (mm)	275	273
Sediment loading (t/year)	2,739	2,578
Phosphorous (P) loading (kg/year)	786	769
Nitrogen (N) loading (kg/year)	3,773	3,773

The results indicate that, increasing tile drainage area from 80% to 100% could reduce annual sediment loading at watershed outlet only by 6% and annual P loading by 2%. There was no reduction in N loading was observed, however tile drains reduced surface runoff by a small amount by allowing more water to infiltrate into the soil. It was observed from the simulation results that the areas which are not tile drained presently are less prone to soil erosion.

5.5 SUMMARY

This is the first effort in quantifying pollutant loadings from the Muddy Creek watershed. The predicted runoff volume, on average, is within the lower margin of acceptable limit for similar watersheds. Sediment loading at the outlet is within the acceptable range and it is fairly comparable to the loadings produced from the watersheds with similar nature found in the literature. Predicted nutrient loadings were converted into concentrations and are compared with the long-term average concentration values of N and P observed at a station located within the watershed. Predicted N concentration can be considered within the acceptable lower limit, since observed concentrations correspond to the low flow and high concentration situations.

Selection of model input parameters needs a great care, especially when performing long term simulations as the results are much sensitive to the input parameters. Sensitivity analysis showed that sediment yield is highly sensitive to the scale of cell discretization and hence estimating sediment yield without considering the effect of cell discretization, could drastically alter the decisions made concerning non-point source pollution control. Among the soil parameters tested, K factor, field capacity and wilting point showed the most sensitivity to the model output. These parameters are to be estimated with greater care in order to obtain the reliable conclusions from the model simulations. The N loading may be affected by many factors such as, land use type, crop rotation, soil types, farming method, amount and timing of the fertilizer application. Many of these factors are interactive and their combined effects are often unpredictable. Simulation results may be improved by incorporating actual field data of crop information such as plant uptake parameters that are sensitive to nutrient loading. The model input database consisted of about 400 parameters in 34 data categories and the preparation required extensive field information that had been collected during the discussions with various people from different organizations in order to obtain reliable simulation results. Incorporating actual crop distribution data throughout the watershed and nutrient application data would improve simulation results. Simulation of BMPs showed that placement of buffer strips along stream segments are effective in reducing sediment and nutrient loadings.

It is evident from this study that, in general, the AnnAGNPS model can be adopted in simulating the surface runoff, sediment and nutrient loadings from the Muddy Creek watershed that has mostly agricultural land use and can be used in prioritizing watershed management activities.

CHAPTER 6

CONCLUSIONS AND RECOMMENDATIONS

6.1 CONCLUSIONS

This modelling study was conducted to investigate the adaptability of AnnAGNPS model in Muddy Creek watershed, to identify the areas susceptible to soil erosion within the watershed and to estimate sediment and nutrient loadings in an effort to prioritize the subwatersheds for treatment/management. The modeled runoff, though within the acceptable limit, is slightly under-predicted compared to the runoff volumes of nearby watersheds. AnnAGNPS does not account for base-flow and this could be a reason for this under prediction. Sediment loading at the outlet is within the acceptable range and it is fairly comparable to the loadings reported from the watersheds with similar nature found in the literature. Predicted N concentration can also be considered to be within the acceptable lower limit.

The analysis of placement of vegetative filter strips revealed that some sections are more effective in reducing sediment loading. Placement of VFS along all the stream segments could reduce 21% of the total average annual sediment loading at watershed outlet.

In conclusion, it is evident from this study that, the AnnAGNPS model can be adopted in simulating surface runoff, sediment and nutrient loadings in the Muddy Creek watershed that has mostly agricultural land use and can be used in prioritizing watershed management activities.

6.2 RECOMMENDATIONS

From the simulation results it is recommended that at least one experimental, continuously monitoring station be set up to monitor the stream flow, sediment and nutrient concentrations within the watershed. The quantities of water, sediment and nutrient obtained from this simulation may be considered as qualitative indicators. These quantities do indicate the relative quantities of the sediment and nutrient loadings from different cells (pockets of lands) within the watershed. These relative quantities or percentages may be used in prioritizing watershed management activities for soil and water conservation.

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APPENDIX A

Table A.1: Best Management Practices Summary (Brown et al., 1991)

Best Management Practices	Surface Water Quality						Ground Water Quality	
	Salin -ity	Sedim-ent	Soluble Nutrie -nts	Absorbed Nutrients	Soluble Pestici -des	Absorbed Pesticides	Nutrie -nts	Pestici de
Management practices								
Nutrient Management	C	C	A	A	C	C	A	C
Pest Management	C	C	C	C	A	A	C	A
Irrigation Water Management	A	A	A	A	A	A	B	B
Soil Salinity Management	A	B	B	B	B	B	C	C
Runoff Management System	C	A	A	A	C	C	A	C
Vegetative and Tillage Practices								
Conservation Tillage	C	A	C	A	C	A	N	N
Contour Farming	C	A	B	A	B	A	N	N
Contour Stripcropping	C	A	B	A	B	A	C	C
Buffer Strip	C	B	C	B	C	B	N	N
Structural Practices								
Water and Sediment Control Basin	C	A	C	A	C	A	N	N
Grade Stabilization Structure	C	B	C	B	C	C	C	C
Grassed Waterway	C	B	C	B	C	B	C	C
Streambank and Shoreline Protection	C	A	C	A	C	C	C	C

A : Medium to high effectiveness, **B** : Low to medium effectiveness, **C** : No control to low effectiveness, **N** : May increase or decrease impact

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