EVALUATING THE AGNPS MODEL FOR PRIORITIZING VEGETATIVE FILTER STRIPS WITHIN AGRICULTURAL WATERSHEDS

By

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ABSTRACT

Evaluating the AGNPS Model to Prioritize Vegetative Filter Strips within Agricultural Watersheds

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Recent developments in GIS interfaces have greatly improved hydrologic water quality modeling; larger watersheds can accurately be assessed at higher spatial resolutions for prioritizing site-specific areas where best management practices, such as vegetative filter strips, would be most effective. In this study, the AGNPS model in conjunction with the AGNPS GIS interface was used to prioritize locations of vegetative filter strip effectiveness within areas of a 45,000 ha (111,000 ac) agricultural watershed located in mid Michigan. Vegetative filter strips alone were found to be relatively inefficient (<25%) especially in areas where concentrated flow occurs (drainage area > 4 ha or 10 ac), which are areas of primary concern in terms of sediment load. Therefore conservation efforts should focus on reducing sediment load in areas of concentrated flow.

In addition, an unconditional stochastic simulation was preformed to identify the uncertainty in the AGNPS model due to errors in 30 m USGS DEM. Comparing the AGNPS sediment yield at the outlet of a 107 ha (264 ac) watershed when using 30 m USGS DEM and 10 m simulated elevation data resulted in a 37% decrease. However, the analysis also identified that change in cell size (from 30 m to 10 m) reduces the sediment load by 41%. These uncertainties are profoundly affecting the AGNPS estimates of sediment yield, which may dramatically influence the analysis of prioritizing conservation practices within watersheds.

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To my Advisors…

You have shaped my life significantly.

To my Family…

You have loved me unconditionally.

To my Friends…

You have brought me lifelong happiness and hope.

and especially to Frank…

You are my inspiration.

My accomplishments belong to you.

Thank you everyone for all that you
have brought to my life.

~Sharon

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NOMENCLATURE

USDA, Soil Conservation Service Curve Number Method, 1972 (Section 2.4)

CN = Curve number (dimensionless)
P = Precipitation rate (in/hr)
Q = Runoff volume (in)
S = Retention parameter (dimensionless)

Smith and Williams, 1980 (Section 2.4)

A = Area (ac)
CS = Channel slope (ft/ft)
LW = Watershed length width ratio (dimensionless)
Q_p = Peak Runoff Rate (cfs)
RO = Runoff volume (in)

Wischmeier and Simth, 1978 (Section 2.4)

C = Cover management factor (dimensionless)
EI = Rainfall energy-intensity (100ft-ton inch/acre hour)
K = Soil erodibility factor (ton-acre hour/100-acre foot-ton inches)
LS = Slope length factor (dimensionless)
P = Practice factor (dimensionless)
SL = Soil loss (tons/acre)
SSF = Factor to adjust for slope shape within cell (dimensionless)

Foster et al., 1981 and Lane 1982 (Section 2.4)

$D(x)$ = Deposition rate (lb/sec-ft^2)
$g'_s(x)$ = Effective transport capacity per unit width (lb/sec-ft)
L_r = Reach length (ft)
$q(x)$ = Discharge per unit width (cfs)
$Q_s(O)$ = Sediment discharge into the upstream end of the channel reach (lb/sec)
$Q_s(x)$ = Sediment discharge at the downstream end of the channel reach (lb/sec)
$q_s(x)$ = Sediment load per unit width (lb/ft^2)
Q_{sl} = Lateral sediment inflow rate (lb/sec)
V_{ss} = Particle fall velocity (fps)
w = Channel width (ft)
x = Downstream distance (ft)

Bagnold, 1990 (Section 2.4)

g'_s = Effective transport capacity per unit width (lb/sec-ft)
g_s = Transport capacity (dimensionless)
k = Transport capacity factor (dimensionless)
n = Effective transport factor (dimensionless)
v = Average channel flow velocity (fps)
V_{ss} = Particle fall velocity (fps)
τ = Shear stress (lb/ft^2)

Variogram Equation (Section 3.3)

h = Lag distance (units described by data)
Var = Variance of the argument
γ(h) = Semivariance
Z(**x**) = Value of the regionalized variable of interest at location **x**
Z(**x**+h) = Value at the location **x**+h

Error Equation (Section 3.3)

C = Coarse DEM (units described by data)
E = Error (units described by data)
H = Higher Accuracy DEM (units described by data)
u = A set of locations (possibly gridded) in a spatial dataset

Chapter 1

1 Introduction

1.1 Background

Non-point source (NPS) pollution is the nation's number one source of water quality problems (EPA, 2002a). Thirty-three percent of U.S. surface waters were surveyed in 2000; the survey determined that 40% of streams, 45% of lakes, and 50% of estuaries are contaminated to the degree that they cannot meet their standard uses (EPA, 2002b). NPS pollution dramatically impacts the quality of aquatic habitats and the species that they sustain; therefore wildlife populations and human uses, such as drinking water and recreation, are also being affected.

NPS pollutants are defined as pollutants carried by rainfall or snowmelt moving over and through the ground (Novotny and Olem, 1994). This runoff carries natural and human made pollutants such as fertilizers; pesticides; toxic chemicals; acid drainage from abandoned mines; bacteria and nutrients from livestock; and sediment from cropland, forestland, and eroding stream banks (Novotny and Olem, 1994). The pollutants are eventually deposited into lakes, rivers, estuaries, wetlands, and also reach sources of underground drinking water.

The Environmental Protection Agency (EPA) issued a 2000 National Water Quality Inventory Report stating that agriculture is the leading source of NPS pollution in surveyed rivers and lakes. Approximately 60% of surface waters are polluted by agricultural activities where sediment is one of the major pollutants (EPA, 2002c). Sediments and sediment-bound nutrients increase turbidity and eutrophication, which decreases dissolved oxygen in surface water. Erosion and sedimentation also reduces the life expectancy of roads, ditches, culverts, and bridges. The damage due to deposited sediment is estimated to be between 2.2 and 7 billion dollars a year (Lovejoy et al., 1997).

Conservation practices, also known as Best Management Practices (BMP) such as conservation tillage, reconstructed wetlands, and vegetative filter strips have been vital in reducing surface water pollution from agricultural runoff. Vegetative filter strips in particular are crucial to water quality because they have a wide range of applications (Lawrence et al., 2002). Defined as an area of herbaceous vegetation situated in-field, at the edge of fields, or adjacent to streams, rivers, or small lakes, vegetative filters slow runoff and filter sediments and nutrients from agricultural fields and animal production systems (NRCS, 1999). Depending on the type of vegetative filter strip, research has indicated that filters must be between 1 and 10 m (3.3 and 33 ft) wide to filter 30-80% of sediments and nutrients that could potentially reach surface waters (Ghadiri, 2001).

Federal or state funded cost-sharing programs developed to encourage landowners to implement conservation practices have existed since the 1930's. Approximately 1.3 million acres of vegetative filter strips are currently covered under various cost-sharing programs (NRCS, 2002a). Each program has different incentives but the same objective: taking farmland out of production to filter runoff. Unfortunately, financial incentives are limited, the eligibility guidelines are broad, and most conservation programs have little or no evaluation procedure. Existing technology does not allow for inexpensive large-scale analysis of non-point source pollutants, therefore under current guidelines 75% of cropland that lie adjacent to streams or rivers is eligible for conservation funding.

Evaluating non-point source pollution is a complex process. Changes throughout agricultural watersheds are mainly occurring on three scales: in the field, on the farm, and throughout the watershed. Environmentally critical land along with conservation practices can accurately be assessed on the three levels with the help of hydrologic water quality models (Lawrence et al., 2002). These models estimate the hydrology and transport of sediments and nutrients deposited into streams or rivers throughout a watershed. As a result, environmentally critical land can be assessed at high resolutions with or without the use of conservation practices. Therefore, conservation efforts can be analyzed both economically and on an environmental level, locally or throughout the watershed.

Hydrologic water quality models such as those of Beasley et al. (1980), Knisel (1980), Skaggs (1980), Leavesley et al. (1983), Williams et al. (1984), Abbott et al (1986), Lenord et al. (1987), Young et al. (1989), Lane and Nearing (1989), Woolhiser et al. (1990), Chung et al. (1992), Bicknell et al. (1993), Arnold et al. (1998), Ahuja et al. (1999), Bingner and Theurer (2001), Borah et al. (2002), and Ogden and Julien (2002) have been developed with different objectives in mind; thus simulated sediment results have varying degrees of accuracy. With the capability of higher computing systems and the recent integration of geographic information systems (GIS), these models have gained widespread acceptance as accurate and cost-effective tools for evaluating agricultural BMPs such as vegetative filter strips (Tim and Jolly, 1994, Mitchell et al., 1993, Srinivasan and Arnold, 1994).

In this study, the Agricultural Non-point Source Pollution (AGNPS) model developed by Young et al. (1989) is used as a tool to evaluate and prioritize areas of vegetative filter strip effectiveness throughout an agricultural watershed. AGNPS version 5.0, in conjunction with its recently developed geographic information systems (GIS) interface (AVNPSM) is a user-friendly, single storm event based, model developed by the United States Department of Agriculture-Agricultural Research Service (USDA-ARS) in cooperation with the Minnesota Pollution Control Agency and Soil Conservation Service (Young et al., 1989). Unlike many models, AGNPS has been validated on watersheds throughout the U.S. and in various countries around the world. The research has stated that the AGNPS model compares well with field data (Young et al., 1989; Mitchell et al., 1993; Perrone and Madramootoo, 1999).

In addition, AGNPS is a distributed parameter model that subdivides watersheds into uniform cells to capture the spatial variability of the physical characteristics of a watershed. The input database, consisting of 21 input parameters for every homogeneous cell, is compiled with the help of a GIS interface called ArcView Non-Point Source Pollution Model (AVNPSM) (He et al., 2001).

The AVNPSM interface was developed to easily compile AGNPS 21 input parameters from four GIS layers; watershed boundary, soil, land use/cover, and a digital elevation model (DEM), so multiple higher resolution scenarios can efficiently be evaluated (He et al., 2001). The grid layout for the AGNPS model is usually based on the GIS layer with the lowest resolution. USGS DEMs have 30 m accuracy in some parts of the nation and are readily available on the Internet for free download. Therefore, because the DEM resolution is usually larger than the soil and land use/cover GIS layers, the DEM resolution defines the size of the grid cell for the AGNPS model.

The main objectives of the AGNPS model are to simulate runoff, sediment and nutrient yields of agricultural watersheds ranging from a few hectares to 20,000 ha (49,421 ac) (Young et al., 1989). The cell-by-cell analysis not only pinpoints areas of excessive sedimentation, but also identifies areas where BMPs such as vegetative filter strips are effective in reducing NPS pollutants (He et al., 1993).

The study focuses on using the AGNPS model with the AVNPSM interface as a tool to prioritize vegetative filter strips within the Stony Creek watershed. The watershed, located in Clinton County, Michigan, drains 45,452 ha (112,314 acres) and is a subbasin of the Grand River, a major tributary of Lake Michigan (NRCS, 2001). Recently, water quality concerns prompted an environmental assessment of Stony Creek which identified sediment as the primary pollutant in the watershed (NRCS, 2001). The reoccurring sedimentation problem is contributing to the degradation of habitat not only in Stony Creek, but also the

Grand River and ultimately Lake Michigan. Erosion and sediment deposition are also reducing the longevity of roads, ditches, culverts, and bridges throughout the watershed. An estimated 90,500 tons of sediment enters the stream each year, where damages are estimated at approximately $15,500 per km ($25,000 per mile) (NRCS, 2001).

Government subsidized cost-sharing programs are aiding in the rapid installation of BMPs throughout the watershed to increase water quality. Vegetative filter strips are one of the main BMPs that are implemented throughout Stony Creek to stabilize streambanks and decrease sediment delivery carried by agricultural runoff (NRCS, 2001). Unfortunately there are limited funds for the cost sharing programs and therefore it's necessary to evaluate areas where vegetative filter strips would decrease the largest amount of sediment throughout the watershed.

1.2 Objectives

The objective of this study is to prioritize areas along a stream segment where a vegetative filter strip would significantly reduce sediment delivery into Stony Creek, by using the AGNPS model with the AVNPSM GIS interface. Prioritizing critical areas of filter effectiveness will help watershed managers install filter strips with greater certainty. This analysis will help determine where funds from cost-sharing programs should be allocated, so future water quality goals can be met.

In addition, a digital elevation model (DEM) uncertainty analysis was also preformed to identify the ambiguity of the AGNPS sediment output when using a 30 m USGS DEM. The parameters derived from elevation data, such as aspect, slope, slope length, and slope shape, are sensitive parameters when estimating sediment yield throughout a watershed. Higher accuracy elevation data was used to identify the errors in a 30 m USGS DEM. Using higher accuracy "true" elevation data when evaluating sediment yield within the AGNPS model will not only improve the accuracy of the output, but will explain the uncertainty in the AGNPS model due to the error in the original elevation dataset (USGS 30 m DEM).

It is important to note that the study only focuses on AGNPS modeling results, there was no in-field data measured to calibrate or validate the data presented in this study. The AGNPS model has been calibrated and validated on many watersheds worldwide; therefore, it is assumed that the model results are representative of realistic scenarios.

Chapter 2

2 Literature Review

2.1 Vegetative Filter Strips

2.1.1 Overview

Sediment, nutrients, pesticides, and other NPS pollutants carried by agricultural runoff are major water quality concerns. As farming operations intensify and the landscape changes, implementing conservation practices or BMPs in agricultural watersheds is becoming the main focus toward alleviating NPS pollution. Vegetative filter strips for example, are used to reduce a wide range of NPS pollutants from various sources, such as herbicide runoff from cropland (Misra et al., 1996; Arora et al., 1996) and sediment and nutrient delivery from feedlots (Young et al., 1980, 1980; Edwards et al., 1983; Dickey and Vanderholm, 1981), dairy facilities (Livingston and Hegg, 1981), swine operations (Sievers et al., 1975; Chaubey et al., 1994), and poultry production areas (Srivastava et al., 1996; Edwards, et al., 1997).

In particular, vegetative filter strips have been studied extensively in small agricultural research plots for removal of sediment delivery from cropland to improve water quality (Meyer et al., 1995; Magette et al., 1989; Raffaelle et al., 1997). Experiments have tested various grasses, filter widths, and slopes to determine the influence of these variables on certain pollutant loads. The results have been consistent in showing that under optimal conditions 75% or more of total sediments are filtered (Gharabaghi, et al., 2001). The following section will focus on issues that involve the reduction of sediments by incorporating a vegetative filter strip on cropland adjacent to streams to improve water quality.

2.1.2 Vegetative Filter Strip Effectiveness

Significant reductions in sediment load take place when flow passes through a filter strip. Dillaha et al. (1989) found that 84% and 70% of sediment was filtered by incorporating a 9.1 m and 4.6 m (30 and 15 ft) wide filter strip. Parsons et al. (1994) noted that filter strips of varying widths resulted in sediment reductions of greater than 50% from field runoff. Daniels and Gilliam (1996) studied total sediment removal through a filter strip and found that approximately 80% was filtered.

The effectiveness of filter strips to reduce sediment load from runoff has been attributed to their filtering capacity (Ghadiri et al., 2001). For the filtration process to work efficiently, flow should be shallow and uniform (Dillaha et al., 1989). Dillaha et al. (1989) observed in flatter regions of a watershed a filter strip can retain greater than 50% of sediment. Jin et al. (2001) similarly noted that the type of vegetation, width, slope, flow rate, and sediment type of the contributing area determine sediment entrapment in vegetative filter strips. Magette et al. (1989) also indicated that when the ratio of filter width to pollutant-contributing area decreases, the effectiveness of the filter strips also decrease. The effectiveness of a filter strip, i.e., filtering capacity, thus is a complex function of the characteristics of the delivery area and the streamside area where the filter is installed.

2.1.3 Design Considerations

Not all stream segments within a watershed are candidates for installing a filter strip. The design of a filter strip depends on many factors including the purpose of the filter (nutrient, sediment, or pesticide removal); seasonal weather conditions; type of vegetation; and soil, slope, size, and land use of the contributing area (NRCS, 1999a). Vegetative filter strips are also only one part of an overall system of conservation practices that control the source and transport of contaminants throughout the watershed (NRCS, 1999a).

Slope. Filter strips are appropriate down-slope of cropland where pollutants are more likely carried. Steep slopes increase flow velocity so that concentrated flow will rapidly exit through the filter, decreasing the interaction time between pollutants and the vegetation and soil. Therefore, filters are most efficient where no slope or gradual slopes exist so that shallow, uniform flow can slowly pass through. Dillaha et al. (1989) found that slope and filter width affect sediment yield and sediment concentration. In addition, Robinson et al. (1996) identified that filter strips were less effective for filtering sediment from a 12% slope than a 7% slope. Jin and Romkens (2001) also studied optimal slopes for sediment removal and noted that, as the slope of the contributing area increased to 4% or 6%, the filter strip failed. The Natural Resource Conservation Service (NRCS) has suggested filter strip locations on slopes greater than 1% but less than 10% for optimal filter strip effectiveness (NRCS, 1999b).

Width. Dense, narrow filters may be the most cost-effective way to reduce sediment delivery in most areas throughout a watershed, but wider filters may be needed elsewhere depending on the characteristics and size of the contributing area. For the purpose of sediment removal, the slope, land use, and soil type of the contributing area determine the filter width. NRCS suggestions for filter width are identified in Table 2.1 (NRCS, 1999a). These suggestions are based on hydrologic class of the soil and percent slope of the contributing area. Other research, has identified that filter widths of 5 to 10 m (16 to 33 ft) are sufficient in filtering up to 80% of sediment (Dillaha et al., 1989; Dabney et al., 1993; Srivastava et al., 1996). Line (1991) studied trapping efficiencies with respect to width and found that filter width greater than 6.1 m (20 ft) produces a small amount of change.

Table 2.1: NRCS suggestions for filter strip length (NRCS, 1999a).

% slope of the filter strips contributing area	Length of Flow (Feet) Hydrologic Soil Group of Filter Area			
	A	B	C	D
0-1	20	20	22	24
1-3	20	25	28	30
3-5	24	30	33	36
5-8	28	35	40	42
8-12	32	40	44	48
12-15	40	50	55	60
15-20	48	60	66	72
>20	May need additional guidance			

Filter strips are more effective at removing coarse aggregate sediment than clay-sized sediment or fine organic particles. Neibling and Alberts (1979) found that filter strips removed over 90% of total sediment at widths ranging from 0.6 - 4.9 m (2 -16 ft). Thirty-seven, 78, 82, and 83% of the clay-sized fraction was removed by the 0.6, 1.2, 2.4, and 4.9 m (2, 4, 8, and 16 ft) filter strips respectively. Wilson (1967) found that a 3.1 m (10 ft) wide filter was sufficient to remove the maximum percentage of sand that went through the filter, 15.2 m (50 ft) for silt, and 122 m (400 ft) for clay. Gharabaghi et al. (2001) indicated that aggregates larger than 40 mm (1.6 in) in diameter were captured in the upper half of a filter strip. In addition, approximately 90% or more of aggregates smaller than 40 mm (1.6 in) were removed when low to moderate flow rates existed (Gharabaghi et al., 2001). Therefore, the soil type is an important factor when determining the width of the filter strip, i.e., if the soil type of the contributing area is primarily clay the filter strip should be wider than if the soil type was primarily sand.

Studies have also shown that approximately 70% of the sediment is deposited between the contributing area and the up-gradient area of the filter (NRCS, 1999b). Robinson et al. (1996) found that the most effective part of the filter strip was in the first 3 to 4 m (10 to 13 ft), suggesting that filter strips act as grass barriers or buffers instead of filters. Dillaha et al. (1989) also indicated that 20 to 50% of the trapped sediment, in a 9.1 and 4.6 m (29 and 15 ft) filter strip, was deposited above the filter strip. In addition, Ghadiri et al. (2001) determined that the deposited sediment in the backwater of the filter was independent of strip width no matter how high the slope of the contributing area. This analysis suggests that vegetation characteristics and planting rates are important aspects when designing an effective filter strip.

Vegetation. Identifying the correct vegetative species for the filter depends on the characteristics of the contributing area as well as the planned purpose of the filter. Grasses or legumes (alfalfa) are usually used as the vegetation for the filter strip (Table 2.2). The NRCS suggests that vegetation species should have stiff stems and high stem density near the ground surface (NRCS, 1999a). Overall the planted vegetation should slow runoff, increase infiltration, reduce erosion, and trap contaminants (NRCS, 1999b).

The NRCS CORE 4 Manual suggests to plant stem densities ranging from 1,500 to 2,500 bunches/m^2 (139 to 231 bunches/ft^2) for optimal conditions (NRCS, 1999b). In contrast, Jin and Romkens (2001) found that 2,500 to 10,000 bunches/m^2 (231 to 926 bunches/ft^2) increase trapping efficiency by 45%. Table 2.2 shows the planting rates of different grasses and their function when planted for vegetative filter strip installation (NRCS, 1999a).

Table 2.2: Vegetation species and seeding rates for trapping sediment within a filter strip (NRCS, 1999a).

Species or Seeding Mixture	Cool/Warm Season	Seeding Rate (lb/Acre)	Seeding Rate (kg/ha)	Established Density (Stems per ft^2)	Established Density (Stems per m^2)
Smooth Bromegrass	Cool	15-30	17-34	50	540
Orchardgrass	Cool	10-15	11-17	70	758
Reed Canarygrass	Cool	10	11	50	540
Tall Fescue	Cool	15-25	17-28	60	648
Tall Wheatgrass	Cool	8-12	9-13	60	648
Prairie grasses Intermediate Wheatgrass	Cool	8-12	9-13	60	648
Eastern Gamagrass	Warm	8	9	40	432
Switchgrass	Warm	5-10	6-11	50	540
Timothy Alfalfa	Cool	5-10 6-10	6-11 7-11	60	648
Bromegrass Alfalfa	Cool	6-12 6-10	7-13 7-11	60	648
Orchardgrass Alfalfa	Cool	2-5 6-10	2-6 7-11	60	648

2.1.4 Maintenance

Vegetative filter strip effectiveness decreases with time. Dillaha et al. (1989) found that the efficiency of a 4.6 m and 9.1 m (29 and 15 ft) filter strip that removed 81% and 91% of total sediments respectively, dropped by an average of 9% between the first and second rainfall simulations. Maintenance of the filter strip throughout the year is an important aspect when achieving filter effectiveness. Optimal conditions for sediment removal is shallow uniform flow. Shallow, uniform flow across the filter must be encouraged to avoid the formation of channels or rills (NRCS, 1999a). When channels or rills do develop the areas must be repaired immediately. Filter strips with cross-slopes, indicating channelized or concentrated flow, are 40-95% less effective at removing sediment (Dillaha et al., 1989). Infiltration can be enhanced by filling the channels or rills with porous or absorbent material (crushed limestone or wood products) to reduce runoff and absorb pollutants contained in the runoff. Installing a grassed waterway or some other conservation practice can aid in reducing sediment in areas where concentrated flow cannot be redirected.

Sediment often accumulates within a filter strip or along the upper part of a filter. If accumulation becomes a problem the sediment should be removed before it reaches a height where flow is diverted around the filter. Machinery may be needed to re-establish the interface between the contributing area and the filter strip so concentrated flow does not occur.

2.2 Conservation Programs

2.2.1 Overview

Financial incentives for vegetative filter strips are available through six USDA conservation programs: Conservation Reserve Program (CRP), Conservation Reserve Enhancement Program (CREP), Environmental Quality Incentives Program (EQIP), Wildlife Habitat Incentives Program (WHIP), Wetlands Reserve Program (WRP), and Stewardship Incentives Program (SIP). These programs were developed by the USDA to encourage the use of conservation practices. Cost-share assistance is provided to any landowner that establishes approved cover on eligible cropland.

2.2.2 Conservation Reserve Program (CRP)

CRP is the nation's largest private lands conservation program and is implemented through the USDA Farm Service Agency (FSA). The current version of CRP was enacted in 1985. It reached 13.7 million hectors (33.9 million acres) through September 2002, at an approximate cost of $1.6 billion (FSA, 2002). USDA economists estimate that it generates far more savings than it costs. CRP is particularly attractive to farmers because in addition to paying for 50% of the cost of installation, it pays 100% of the annual soil rental rate and an extra 20% of the soil rental rate if grassed waterways, field windbreaks, filter strips, and riparian buffers are installed (FSA, 1999a). CRP contracts require landowners to enroll into the program for 10-15 years.

Land that is accepted for enrollment has to meet certain eligibility requirements. Cropland that is in production, pastureland that is suitable for use as riparian buffers, and EPA designated wellhead protection areas are all eligible areas (FSA, 1999a). In addition, highly erodible or environmentally sensitive land is eligible for installation of the following practices: riparian buffers, filter strips, grass waterways, shelter belts, field windbreaks, living snow fences, contour grass strips, salt tolerant vegetation, or shallow water areas for wildlife. The

Environmental Benefits Index (EBI) is used to prioritize land offered for enrollment (FSA, 1999b). Scores are based on cost, six environmental factors: wildlife, water quality, erosion, enduring benefits, air quality benefits from reduced wind erosion, and state or national conservation priority areas (CPAs).

2.2.3 Conservation Reserve Enhancement Program (CREP)

CREP is a federal-state conservation partnership program that targets significant environmental effects related to agriculture and is based on state participation (FSA, 2000). CREP is a spin-off from the original CRP program. The USDA's objective is to share costs and resources to address specific local environmental problems. For example, the Michigan CREP has been designed to reduce the amount of sediment by over 784,000 metric tons (864,000 tons), nitrogen by 726,000 kg (1.6 million pounds), and phosphorus by 363,000 kg (0.8 million pounds) (FSA, 2000). In addition, the Michigan CREP is intended to protect water supplies used by over one million people, protect over 8,100 linear km (5,000 miles) of streams from sedimentation, and improve wildlife habitat in the project areas over the next 15 to 20 years (FSA, 2000). Participating states receive funding for 40,500 ha (100,000 acres). Incentives for CREP participants are very high. Landowners are eligible for five types of payments: base annual rental payments (soil rental rate and consolidated annual CRP rental payment), incentive payments (consolidated annual CRP rental payment), maintenance payments (consolidated annual CRP rental payment), state cost-share assistance payment, and state lump sum one-time payment (FSA, 2000). In addition to the normal rental payment, an extra incentive of $12 per ha ($5/acre) is also given to the landowner for any lands that are highly erodible.

2.2.4 Environmental Quality Incentives Program (EQIP)

Under the EQIP program farmers and ranchers may receive financial and technical help to install a conservation practice or implement structures to manage conservation practices. EQIP does not involve land retirement but rather conservation farming on working farms (NRCS, 2002b). Landowners that are interested in the program are subject to a minimum of a one-year contract and up to a 10-year contract involving financial and technical assistance and education. EQIP may pay up to 75% of the cost of certain conservation practices. Limited resource farmers or beginning farmers may be eligible for up to 90% of the cost of conservation. The total cost-share and incentive payments are limited to $450,000 per individual (NRCS, 2002b).

2.2.5 Wildlife Habitat Incentives Program (WHIP)

The WHIP program is a voluntary program for people who want to develop and improve wildlife habitat on their own land (NRCS, 2002d). NRCS provides technical and financial assistance to improve fish and wildlife habitat. Up to 75% of the cost is paid to the landowner to establish the habitat (NRCS, 2002d). Agreements generally span 5 to 10 years.

2.2.6 Wetlands Reserve Program (WRP)

WRP offers landowners the opportunity to protect, restore, and enhance wetlands on their property. NRCS provides technical and financial support to help landowners. The program offers three enrollment options: permanent easement, 30 year easement, and restoration. Each enrollment option has different cost-sharing advantages to the landowner, up to 100% of the soil rental rate and also 75-100% of the cost to restore the land (NRCS, 2002c). The goal of the program is to achieve the natural wetland functions combined with optimum wildlife habitat on every acre enrolled in the program.

2.2.7 Emergency Watershed Protection (EWP)

The EWP protects the lives and property threatened by natural disasters such as floods, hurricanes, tornadoes, and wildfires (NRCS, 2002e). The NRCS provides technical and financial assistance to preserve life and property threatened by excessive erosion and flooding. EWP provides funding for clearing debris from clogged waterways, restoring vegetation, and stabilizing riverbanks, for example by installing vegetative filter strips. The measures that are taken must comply with environmental and economic regulations. The benefits are payable only too individual property owners. Seventy-five percent of funds needed to restore the property are provided. The community or local sponsor pays the remaining twenty-five percent.

2.3 Current Hydrologic Water Quality Models

2.3.1 Overview

Modeling the performance of vegetative filter strips, or any other BMP, involves simulating multiple complex mechanisms that take place within a watershed. Hydrologic/water quality models are used as tools to analyze the hydrology, sediment, and nutrient transport throughout watersheds and to evaluation these processes with or without the use of BMPs. There are three types of models that are used for NPS pollution modeling, continuous watershed scale, single event based watershed scale, and field based models. Most of these models that exist today were developed in the 1970s and 1980s (Borah, 2002a).

The most commonly used continuous simulation models include: Chemical, Runoff, and Erosion from Agricultural Management or CREAMS (Knisel, 1980), Hydrological Simulation Program or HSPF (Bicknell et al., 1993), Areal Nonpoint Source Watershed Environmental Response Simulator or ANSWERS-continuous (Beasley et al., 1980), Soil and Water Assessment Tool or SWAT (Arnold et al., 1998), and Annualized Agricultural NonPoint Source Pollution Model or ANNAGNPS (Bingner and Theurer, 2001). CASC2D (Ogden and Julien, 2002), Precipitation-Runoff Modeling System or PRMS (Leavesley et al., 1983), and European Hydrologic System or SHE (Abbott et al., 1986a,b) have long-term and single-event based modes.

Continuous models are useful for analyzing long-term effects of hydrological changes and watershed management practices. Due to their complexity these models require large amounts of data for simulations which also require higher computing systems. Continuous models take a long time to calibrate and validate, therefore, the results from these models are not yet robust, which makes it harder to rely on the output as an accurate interpretation of the processes that take place within the watershed.

Commonly used field-based models include Drainage Model or DRAINMOD (Skaggs, 1980), Groundwater Loading Effects on Agricultural Management Systems or GLEAMS (Leonard et al., 1987), Root Zone Water Quality Model or RZWQM (Ahuja et al., 1999), Erosion-Productivity Impact Calculator or EPIC (Williams et al., 1984), Water Erosion Prediction Project or WEPP (Lane and Nearing, 1989), and Agricultural Drainage and Pesticide Transport or ADAPT (Chung et al., 1992). Field based models only analyze the sediment and nutrient mechanisms within an agricultural field boundary. This simplistic field scale assessment ignores the soil and land use/cover of the upland area beyond the boundary.

Many of the equations and concepts from the field-based models have evolved into watershed scale models. For example, the SWAT model emerged mainly from SWRRB, which is a single storm event based model developed by Arnold et al. (1990), but features from CREAMS, GLEAMS, and EPIC are important elements of the model as well. The field-based models have often been used as the basic framework for the watershed scale models. Each field-based model has varying degrees of accuracy where model validations from in-field analysis have been analyzed extensively.

Single event models, such as AGNPS (Young et al., 1989), ANSWERS (Beasley et al., 1980), KINEROS (Woolhiser et al., 1990), and DWSM (Borah 2002a) can be used to evaluate watershed NPS pollutants and design BMPs for severe or actual storm events. The single storm event based models are user-friendlier than continuous models but have the luxury of simulating the effects of pollutants throughout a watershed unlike the field-based models. The main strengths and weaknesses of the single event-based models are described in the sections below.

2.3.2 Areal Nonpoint Source Watershed Environmental Response Simulator (ANSWERS)

The ANSWERS model was developed to evaluate the effects of BMPs on surface runoff and sediment loss from agriculture watersheds. The original ANSWERS model was developed in the late 1970s (Beasley and Huggins, 1982). The model is based on one of the first true distributed parameter hydrologic models (Huggins and Monke, 1966). The model components and capabilities are runoff, infiltration, subsurface drainage, soil erosion, and overland sediment transport. ANSWERS principle application is watershed planning for erosion and sediment yield control on complex watersheds and water quality analysis associated with sediment bound chemicals (Beasley et al., 1980).

The watershed representation consists of square grids with uniform hydrologic characteristics, some having companioned channel elements (Boroh et al., 2001). Rainfall excess is calculated by surface detention with empirical relations. Infiltration and runoff is calculated using the Mannings equation and continuity equations. The main weakness of ANSWERS is its erosion model. The erosion component is largely empirical and simulates only gross sediment distribution of eroded sediment using Yalin's method (Dillaha and Beasley, 1983).

2.3.3 KINematic runoff and EROSion (KINEROS) model

KINEROS is an event oriented, physically based model developed by the USDA-ARS. The model describes the processes of interception, infiltration, surface runoff, and erosion from small agricultural and urban watersheds. The partial differential equations describing overland flow, channel flow, erosion, and sediment transport are solved by finite difference techniques (USDA-ARS, 2002). The spatial variation of rainfall, infiltration runoff, and erosion parameters can be accommodated.

Applications of this model are limited to small watersheds and specific combinations of space and time increments. KINEROS does a relatively good job simulating runoff and sediment yield at watershed scales of up to approximately 1000 ha (4 sq. miles) (USDA-ARS, 2002). BMP evaluation is based on detention basins and alterations to hydrologic and hydraulic conditions (Borah, 2002b).

2.3.4 Dynamic Watershed Simulation Model (DWSM)

DWSM was developed to simulate surface and subsurface storm water runoff, propagation of flood waves, upland soil erosion, sediment transport, and nutrient and pesticide transport in agricultural watersheds (Borah, 2000a). Watershed representation is overland, channel, and reservoir segments defined by topographic-based natural boundaries (Borah, 2002a). The model has routing schemes developed using approximate analytical solutions of physically based governing equations preserving the dynamic behaviors of water, sediment, and the accompanying chemical movements within a watershed (Borah, 2002a). The DWSM is a newly developed model therefore validations are in process. The model is still in the development stages but has a potential for simulating and predicting accurate results.

2.4 Agriculture Non-Point Source Pollution (AGNPS) Model

2.4.1 Overview

The Agricultural Non-Point Source pollution (AGNPS) model version 5.0 is a single event empirically based distributed parameter model (Young et al., 1989). The model uses one time step (storm duration) to generate one value for each of the output variables: runoff volume, peak flow, sediment yield, and average concentrations of nutrients (Young et al., 1989). The AGNPS model's distributed parameter approach allows for every area within the watershed to be analyzed. AGNPS is a suitable tool to use to study the overall response of BMPs from a single severe or design storm. Therefore, BMPs such as vegetative filter strips can accurately be prioritized throughout watersheds.

Recently, the development of a GIS interface called AVNPSM has enhanced the capabilities of the model (He et al., 2001). AVNPSM has allowed the AGNPS model to assess larger watersheds with higher resolutions, yielding an accurate description of the physical characteristics of the watershed. The interface decreases the time and labor involved in producing the input for the model, thus allowing the model to become widely accepted by watershed managers.

2.4.2 Model Structure

The AGNPS model components use equations and methodologies that have been well established and are extensively used by agencies such as the USDA-NRCS (Young et al., 1989). Runoff volume and peak flow rates are estimated using the SCS runoff curve number method. Peak runoff rate for each cell is calculated by an empirical relationship proposed by Smith and Williams (1980). The Universal Soil Loss Equation or USLE described by Wischmeier and Smith (1978) estimates upland erosion and sediment transport. Sediment is routed from cell to cell through the watershed to the outlet using a sediment transport and depositional relationship described by Foster et al. (1981) which is based on a steady-state continuity equation (Young et al., 1989). These equations are described in the sections below.

Hydrology. Runoff volume and peak flow rate is calculated based on the SCS curve number method.

$$Q = \frac{(P - 0.2S)^2}{P + 0.8S} \qquad \textbf{(2.1)}$$

Where Q is the runoff volume (in), P is the precipitation rate (in/hr), and S is the retention parameter (dimensionless).

The retention parameter (S) is expressed in terms of a curve number (CN) as follows:

$$S = \frac{1000}{CN} - 10 \qquad \textbf{(2.2)}$$

The curve number (CN) depends on the land use, soil type, and hydrologic soil condition.

Peak runoff rate (Q_p) in units of cfs for each cell is estimated by using an empirical relationship proposed by Smith and Williams (1980).

$$Q_p = 3.79A^{0.7}CS^{0.16}(RO/25.4)^{(0.903A\wedge 0.017)}LW^{-0.19} \qquad \textbf{(2.3)}$$

Where, A is the Area in Acres, CS is the channel slope in ft/ft, RO is the runoff volume in in, and LW is the watershed length width ratio (dimensionless). The coefficient values for the peak runoff rate (Q_p) were determined from field measurements (Young et al., 1989).

Erosion and sediment transport. A modified form of the universal soil loss equation (USLE) is used to estimate upland erosion for single storms.

$$SL = (EI)KLSCP(SSF) \qquad \textbf{(2.4)}$$

Where SL = soil loss (tons/acre), EI = rainfall and runoff erosivity index (100 ft ton inch/acre inch), K = the soil erodibility factor (dimensionless), LS = topographic factor (dimensionless), C = the cover and management factor (dimensionless), P = the supporting practice factor (dimensionless), and SSF = factor to adjust for slope shape within the cell (dimensionless).

Soil loss is calculated for each cell in the watershed. Five particle size classes –clay, silt, small aggregates, large aggregates, and sand for eroded soil and sediment yield is estimated for the watershed (Young et al., 1989). Detached sediment is routed from cell to cell through the watershed to the outlet. The basic routing equation is derived from the steady-state continuity equation as described by Foster et al. (1981) and Lane (1982):

$$Q_s(x) = Q_s(O) + Q_{sl}(x/L_r) - \int_0^x (x)wdx \qquad \textbf{(2.5)}$$

Where $Q_s(x)$ is sediment discharge at the downstream end of the channel reach (lb/sec), $Q_s(O)$ is the sediment discharge into the upstream end of the channel reach (lb/sec), Q_{sl} is the lateral sediment inflow rate (lb/sec), x is the downstream distance (ft), L_r is the reach length (ft), w is the channel width (ft), and D(x) is the deposition rate (lb/sec-ft). Deposition rate (D(x)) is estimated as follows:

$$D(x) = [V_{ss} / q(x)][q_s(x) - g'_s(x)] \qquad \textbf{(2.6)}$$

Where V_{ss} is the particle fall velocity (fps), q(x) is the discharge per unit width (cfs), $q_s(x)$ is the sediment load per unit width (lb/sec-ft), and $g'_s(x)$ is the effective transport capacity per unit width (lb/sec-ft).

Effective transport capacity is computed using a modification of the Bagnold (1990) stream power equation, as follows:

$$g'_s = \eta g_s = \eta k \frac{\tau v^2}{V_{ss}} \qquad \textbf{(2.7)}$$

Where g_s is the transport capacity (dimensionless), *n* is an effective transport factor (dimensionless), k is the transport capacity factor (dimensionless), τ is the shear stress (lb/ft^2), and *v* is the average channel flow velocity (fps) determined by Manning's equation. Young et al. (1987) describes values for the effective transport capacity.

Sediment load for each of the five-particle size classes leaving a cell is calculated as follows:

$$Q_s(x)=\left[\frac{2q(x)}{2q(x)+\Delta x V_{ss}}\right]\left[Q_s(o)+Q_{sl}\frac{x}{L}-\frac{w\Delta x}{2}\left[\frac{V_{ss}}{q(o)}[q_s(o)-g'_s(o)]-\frac{V_{ss}}{q(x)}g'_s(x)\right]\right] \quad \textbf{(2.8)}$$

Symbols for equation 2.8 are defined above. Equation 2.8 is the basic routing equation that drives the sediment transport model.

2.4.3 Input and Output Database

The AGNPS model subdivides the watershed into uniform cells to capture the spatial variability of its landscape. A database consisting of 21 input parameters for every cell representing the watershed is required to run the model (Table 2.3). This distributed parameter approach allows the AGNPS to take into account the spatial variability of the landscape and also examine sedimentation, runoff or nutrients, either for entire watershed or on a cell-by-cell basis. The cell-by-cell analysis not only pinpoints excessively polluted areas, but also is significant for evaluating where best management practices are effective.

Table 2.3: AGNPS Cell Input Parameters

1. Cell Number	8. SCS curve number (CN)	15. Fertilization incorporation
2. Overland flow direction	9. Mannings roughness coeff. (n)	16. Fertilization level
3. Receiving Cell number	10. USLE C factor	17. Pest Indicator
4. Average slope (%)	11. USLE P factor (P)	18. Point source indicator
5. Average slope length	12. Surface condition constant (SC)	19. Gully source indicator
6. Slope shape factor	13. Chemical oxygen demand factor	20. Impoundment factor
7. USLE K factor (K)	14. Soil texture	21. Channel indicator

Developing the AGNPS input database is extremely time consuming and labor intensive, especially for large watersheds simulated with small cell sizes. Capturing the spatial variability of a watershed is difficult when a large cell size is used. Previous research has concentrated on identifying an optimal cell size to adequately capture the landscape variability within a watershed so the model's estimates for runoff, sediment yield, and nutrients are reasonably close to the measured data (Bhuyan et al, 2001).

Brannan and Hemlet (1998) used a Jack-Knifing procedure, a geostatistical method, for selecting a base cell size for the entire watershed and also locating areas of cell sub-divisions within the watershed. The base cell size was predicted according to the layout of the SCS Curve Number data, which represents the land use characteristics, within the watershed. The base cell size of 2.14 ha (5.3 ac) was selected based on the mean square residual of the krigged estimates, less than 2.14 ha (5.3 ac) was not used because the average mean square residuals were not significantly lower. Locating cell sub-divisions were also used for indicating an enhanced representation of the land use/cover characteristics within the watershed. This analysis found that when 0.03 ha (0.07 ac) cell sub-divisions were included in the 2.14 ha (5.3 ac) grid layout the variance increased by 10%, indicating a better representation of the watershed.

The labor and time constraints of subdividing watersheds into smaller grid cell sizes for accurate representation of the physical characteristics is a limiting factor when evaluating NPS pollutants in the AGNPS model. Fortunately, GIS interfaces have been developed to aid in analyzing larger watersheds with higher resolutions. Since the 1990s, GIS interfaces have aided in the development of the AGNPS input database allowing larger watersheds to be modeled at a higher resolution (smaller cell size) leading towards a more accurate physical representation of the watershed.

The GIS interfaces allow grid cell size to be comparable to the resolution of the raster GIS data that is used to derive the input parameters for the model. Therefore, The grid layout for the AGNPS model is usually based on the DEM datasets because the grid resolution is larger (30 m or 98 ft) than the soil and land use/cover GIS layers (1 m or 3.3 ft). Readily available soil and land use/cover data is digitized from 1 m (3.3 ft) USGS digital ortho photo quadrangles and readily available DEMs are have resolutions of 30 m (98 ft) in various parts of the nation. Therefore, the coarse grid layout is limited to the resolution of the elevation data and the accuracy of the spatial resolution of the soil and land use/cover for the watershed is limited to the size of the grids as well.

Over the years, there have been a number of AGNPS GIS interfaces developed (Srinivasan and Engel, 1991; Cronshey et al., 1993; Engel et al., 1993; He et al., 1993; Jankowski and Haddock, 1993; Klaghoffer et al., 1993; Mitchell et al., 1993; Yoon et al., 1993). Tim and Jolly (1994) demonstrated the use of an Arc/Info – AGNPS interface for assessing the effectiveness of best management practices. In their study, they demonstrated that sediment load was reduced by 41% by implementing a vegetative filter strip around all stream segments within a 417 ha (1030 ac) agricultural watershed located in southern

Iowa. The grid cell size for the entire watershed was based on the manual labor involved, time constraints, and computing capability, thus a 100 m by 100 m (328 ft by 328 ft) grid was used for the analysis. The 100 m (329 ft) grid cells located adjacent to the stream were subdivided to create 20 m (66 ft) grids. The 20 m (66 ft) grid cells that touch the stream were buffered, simulating the buffer strip scenario. The analysis was a modeling scenario where no in-field data was presented. The 100 m (329 ft) grid cell size may not have defined the "true" landscape characteristics of the watershed, therefore, the results at the outlet may not have been accurate.

Most recently, He et al. (2001) developed AVNPSM, an ArcView GIS interface for AGNPS. The AVNPSM, a Windows based ArcView (version 3.0a or later) GIS interface, developed to easily collect and manipulate the 21 input parameters needed for the AGNPS input database so multiple scenarios can be evaluated. The input database, created by AVNPSM, is a database file that is imported into the AGNPS model, version 5.0. The interface, which was written in ArcView Avenue scripts, uses three GIS layers to develop the database. These layers include soil, land use/cover, and a DEM.

AGNPS creates a tabular output that can easily be imported into ArcView for evaluation. The outputs can be examined for each cell throughout the watershed or at the watershed outlet. The tabular output provides estimates of runoff volume (inches), peak runoff rate (cfs), sediment yield (tons), sediment concentration (ppm), upland erosion (tons/acre), amount of deposition (%), sediment generated within each cell (tons), mass of sediment attached and multiple chemical outputs associated with nitrogen, phosphorus and chemical oxygen demand (Young et al., 1989).

2.4.4 Validation

The AGNPS model has been validated on several watersheds all over the world resulting in varying degrees of accuracy. Young et al. (1989) found a coefficient of determination (r^2) to be 0.81 for sediment yield estimates on three different watersheds ranging from 1,000 to 4,000 ha (2,500 to 10,000 ac) located in north central United States. A 4 ha (10 ac) cell size was used for each watershed. Perrone and Madramootoo (1999) tested AGNPS on a 2,700 ha (6,700 ac) watershed in Quebec. A 9.25 ha (23 ac) cell size was used for the evaluation. The storm events that were modeled resulted in an average error of 28.2%.

Mitchell et al. (1993) evaluated 50 sediment yield events from two watersheds and five sub watersheds located in East Central Illinois. Half of the events were used to calibrate the model and the other half were used for validation. A 20 m by 20 m (66 ft by 66 ft) cell size was used to predict sediment yield from small, mild-sloped watersheds ranging from 30 to 1.6 ha (74 to 4 ac). Predicted vs. observed resulted in a mean error of 35% for the validated data and 23% for the calibrated data.

The AGNPS model was used to assess runoff, soil erosion, and associated NPS pollutants in a watershed located in Italy. Lenzi and Luzio (1995) found that the predicted runoff volume was estimated precisely but peak flow rates were poorly predicted at high and low storm events. Predicted sediment and nutrient loads were overestimated and underestimated in the same events. The average error for predicted sediment and nutrient load resulted in 26% and 14% respectively. This analysis demonstrated that the AGNPS model is capable of predicting sediment and nutrient loads for large storm events but the authors suggest further investigation of the model.

2.4.5 Sensitivity Analysis

A sensitivity analysis is an essential process when understanding model responses to parameter changes. Finding the parameters that have the largest or no impact on the model is of great importance. Knowing the sensitivity of the model parameters will aid in understanding errors that occur in the output.

Young et al. (1987) demonstrated a sensitivity analysis for all of the parameters of the AGNPS model. The sensitivity analysis of varying all the parameters by -50%, -25%, 25%, and 50% reported that the slope-associated parameters (LS) were the most sensitive when evaluating sediment yield (Figure 2.1).

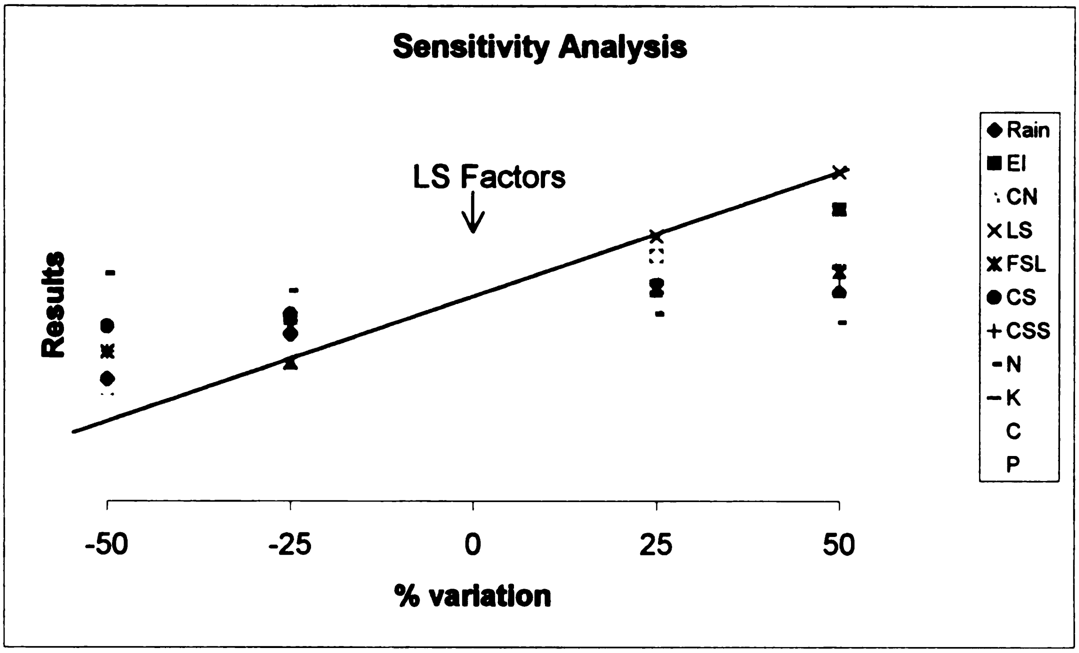


Figure 2.1: Young et al. (1987) sensitivity analysis indicating slope-associated parameters (LS) are the most sensitive in the AGNPS model when predicting sediment yield.

Topographic attributes are important factors in predicting sediment loss when using hydrologic models as shown by the sensitivity analysis in Figure 2.1 for the AGNPS model. Most watershed analyses, including evaluations using the AGNPS model, use 30 m USGS DEMs to derive the slope associated parameters (aspect, slope length, slope shape, percent slope). These elevation datasets are readily available for free download off of the Internet for various parts of the U.S.

The 30 m USGS DEM data is in 7.5-minute units and corresponds to the USGS 1:24,000 scale topographic quadrangle map series for the U.S. with a spatial resolution of 30 by 30 meters (USGS, 2002a). The uncertainty of USGS

DEMs have been evaluated in certain spatial modeling applications by using various geostatistical methods (Fisher, 1999) but no data has been reported for evaluating the 30 m USGS DEM uncertainty in the AGNPS model.

Chapter 3

3 Materials and Methods

3.1 Study Area

The Stony Creek watershed, located in Clinton County, Michigan drains 45,452 ha (112,314 acres) and is a subbasin of the Grand River, a major tributary of Lake Michigan (Figure 3.1). Land use in the Stony Creek watershed is 85% agricultural land consisting of corn, soybeans and wheat, as well as more diverse crops such as mint, with the remaining 15% a mixture of urban areas, forests, shrubland, and wetlands or water (Figure 3.2). Soil types vary extensively throughout the watershed from well-drained sandy loam soils to poorly drained clay soils (Figure 3.3). The topography is predominantly flat but in some areas gentle slopes exist, with the elevation ranging from 198 to 277 m (650 to 909 ft) above sea level (Figure 3.4). Please note the following images throughout this thesis are presented in color.

Figure 3.1: Location of the Stony Creek watershed.

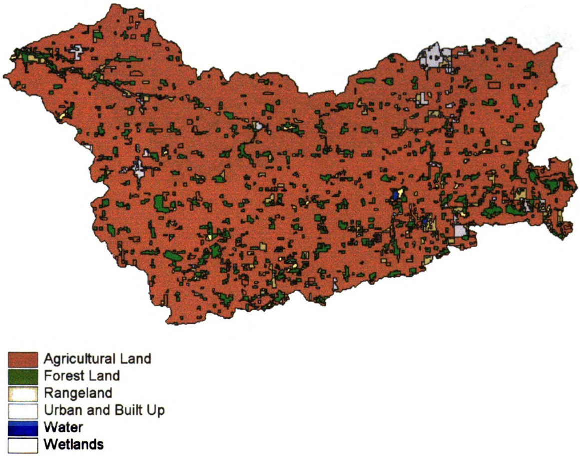


Figure 3.2: Land use in the Stony Creek watershed.

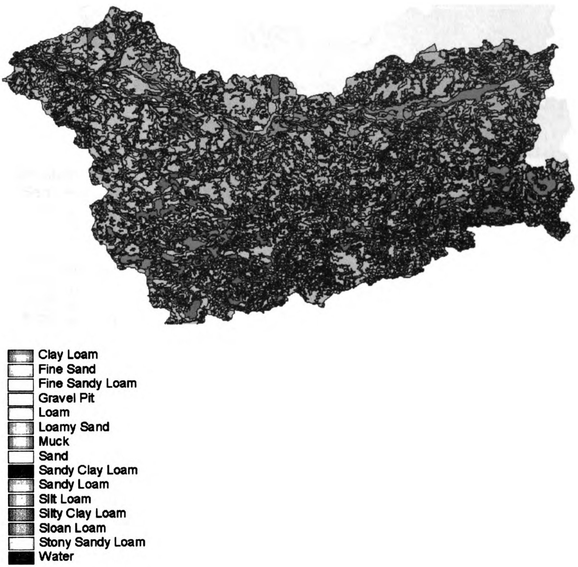


Figure 3.3: Soil textures of the Stony Creek watershed.

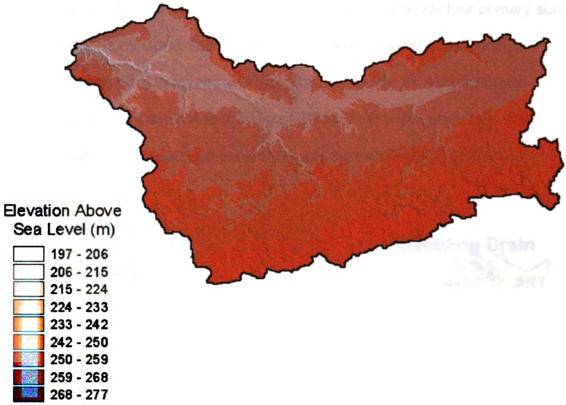


Figure 3.4: Topography of the Stony Creek watershed.

3.2 AGNPS Modeling Scenarios

3.2.1 Overview

The AGNPS distributed parameter approach allows analysis at any point throughout the watershed at the resolution of the specified grid cell size. The AGNPS model is potentially capable of analyzing watersheds consisting of 30,000 cells, but experience has shown that the model tends to crash when the database exceeds 15,000 cells, therefore only allowing analysis of smaller watersheds. Larger cell sizes can be used when analyzing extensive watersheds greater than 4,047 ha (10,000 ac) but this analysis sacrifices detailed characteristics of the watershed. For the purposes of this study four primary sub-characteristics of the watershed. For the purposes of this study four primary sub-watersheds of Stony Creek (Bad Creek, Lost Creek, Muskrat Creek, and Spaulding Drain) were analyzed and then subdivided into 31 secondary sub-watersheds ranging from 284 to 1,214 ha (702 to 3000 ac) (Figure 3.5). The grid-cell resolution for each sub-watershed was 30 meters (0.22 ac).

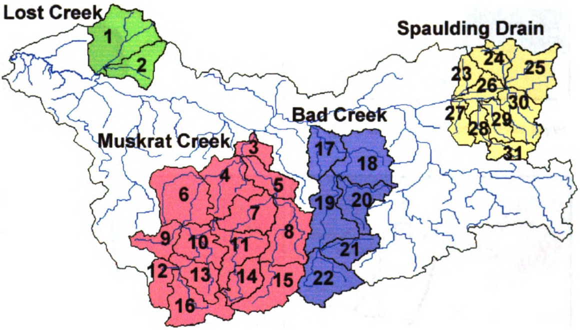


Figure 3.5: Thirty-one secondary sub-watersheds of the Stony Creek watershed.

The AGNPS GIS interface, AVNPSM, was used to develop the AGNPS input database for the 31 sub-watersheds. The interface uses three raster GIS layers (soil, land use/cover, and a digital elevation model (DEM)) and a boundary layer to develop the AGNPS input database for each watershed. An example of the four GIS layers needed to develop the input database is shown in Figure 3.6.

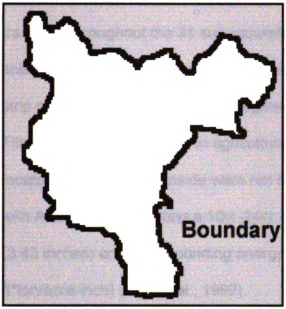


Figure 3.6a: Boundary layer

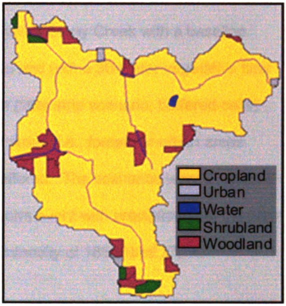


Figure 3.6b: Land use/cover layer

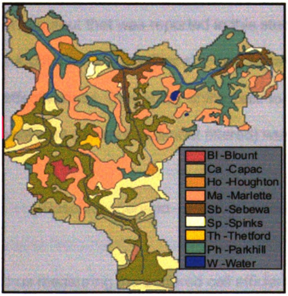


Figure 3.6c: Soil type layer

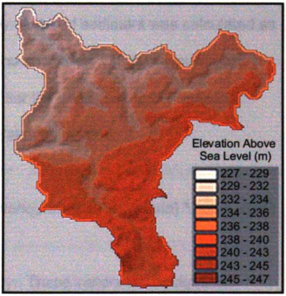


Figure 3.6d: 30 m USGS DEM

Figures 3.6a-d: Four GIS layers for the AVNPSM interface.

Modeling scenarios consisted of simulating the hydrology and sediment transport throughout the 31 sub-watersheds of Stony Creek with a baseline scenario (no filter strip, non-buffered cells) and with a 30-meter vegetative filter strip placed around each stream segment (filter strip scenario, buffered cells). Filters were only simulated in agricultural areas, i.e., forests or urban areas located along the streamside were not buffered. The scenarios were evaluated with AGNPS by simulating a 10yr-24hr storm event with precipitation of 87.1 mm (3.43 inches) and corresponding energy intensity of 1859 N/m^2 (70.47-ft*ton/acre-inch) (Huff et al., 1992).

AGNPS creates a tabular output that can easily be imported into ArcView for evaluation. Although AGNPS creates several outputs, sediment yield was the only output that was reported in this study. Filtered sediment was calculated as the difference in the filter strip and baseline scenarios, which measured cell effectiveness. The ability of the cells to filter the entire amount of entering sediment (percent sediment filtered) was also calculated,

$$[(\text{Baseline scenario} - \text{Filter strip scenario}) / \text{Baseline scenario}] * 100 \quad \textbf{(3.1)}$$

thus measuring the buffered cell efficiency. These watershed evaluation procedures led to identifying and prioritizing areas of filter strip effectiveness and efficiency throughout the 31 sub-watersheds in Stony Creek.

3.2.2 Filter Strip and No Filter Strip Input Database

The data sets used for the study area were obtained from a variety of different sources (Table 1). The NRCS county soil survey database (SSURGO) for Clinton County was used to identify soil texture and soil erodibility factor (K). The SSURGO soils database is generally the most detailed level of soil geographic data developed by the National Cooperative Soil Survey. The digital vector data is collected and archived in a 7.5-minute topographic quadrangle unit, mapped on a 1:15,840 scale using aerial maps or remotely sensed images (USGS, 2002b)

A 7.5-minute digital elevation model provided by USGS was used to determine slope, slope length, slope shape, and flow direction for the AGNPS database. The USGS DEMs are digital representations of cartographic information in raster form (USGS, 2002a). The 7.5-minute units correspond to the USGS 1:24,000 scale topographic quadrangle map series, which provide elevation values sampled at 30 by 30 m intervals (USGS, 2002a).

The land use database, digitized and reprojected by the State of Michigan Center for Geographic Information Systems consists of the 1992 National Land Cover Data (NLCD). The vector NLCD is derived from the mid-1990s 30 m spatial resolution Landsat Thematic Mapper satellite data and aerial maps (USGS, 2002c). The Stony Creek's land use included five different land cover classes: woodland, shrubland/wetlands, water, farmsteads, and cropland. Each land cover class was assigned a value for the SCS curve number (CN), crop management factor (C), overland Manning's value (n), and surface condition constant (SC) based on the digitized land use database (Table 3.1).

Table 3.1: AGNPS input parameters and the corresponding data sources.

Parameter	Data Type	Data Source
1. Cell Number	Topography	USGS DEM
2. Overland flow direction (1-8)	Topography	USGS DEM
3. Receiving Cell number	Topography	USGS DEM
4. Average slope (%)	Topography	USGS DEM
5. Average slope length	Topography	USGS DEM
6. Slope shape factor (1, 2, or 3)	Topography	USGS DEM
7. USLE K factor (K)	Soil	SSURGO county soil database
8. SCS curve number (CN)	Land use	1992 National Land Cover
9. Mannings roughness coeff. (n)	Land cover	1992 National Land Cover
10. USLE C factor (C)	Land use	1992 National Land Cover
11. USLE P factor (P)	Land cover	1992 National Land Cover
12. Surface condition constant (SC)	Land use	1992 National Land Cover
13. Chemical oxygen demand factor (COD)	Land use	1992 National Land Cover
14. Soil texture (1, 2, or 3)	Soil	SSURGO county soil database
15. Fertilization incorporation (1 or 0)	Assume none	-
16. Fertilization level	Assume none	-
17. Pest Indicator (1 or 0)	Assume none	-
18. Point source indicator (1 or 0)	Assume none	-
19. Gully source indicator	Topography	USGS DEM
20. Impoundment factor (1 or 0)	Assume none	-
21. Channel indicator (1 or 0)	Hydrology	1992 National Land Cover

Many of the 21 parameters were assumed for both scenarios. The USLE conservation practice factor (P) was assumed to be 1 to simulate worst-case occurrences. The soil texture number (sand=1, silt=2, clay=3) was set at a 2, the closest soil texture number simulating loam and silt loam soils. The fertilizer, pesticide, point source, and impoundment factors were set at zero, since the study did not focus on nutrient or pesticide pollution.

To simulate a filter strip within AGNPS, four input parameters were manipulated for the streamside cells: the curve number, C-factor, overland Manning's value and surface condition constant. Tim and Jolly (1994) also chose these parameters for their filter strip analysis. The curve number for the filter strip was defined as brush-weed-grass mixture with good hydrologic condition (SCS, 1986). The C-factor for the filter strips was assigned a value of 0.003, representing a filter with 95% vegetative density and a vegetative canopy of 75% grass or grass-like plants (Wischmeier and Smith, 1978). The overland Manning's value was set at 0.25, representing a grass pasture (NRCS, 1979) and the surface condition constant was set to 1.0, an internal indicator in AGNPS for simulating a filter strip (Young et al., 1994). These values are listed in Table 3.2.

Table 3.2: Land use parameters for the Stony Creek watershed.

Land Cover Class	Hydro Class	CN	C	P	n	SC
Cropland (85.1%)	A	64	0.22	1	0.04	0.05
	B	75	0.22	1	0.04	0.05
	C	82	0.22	1	0.04	0.05
	D	85	0.22	1	0.04	0.05
Urban (2.3%)	A	59	0.01	1	0.015	0.01
	B	74	0.01	1	0.015	0.01
	C	82	0.01	1	0.015	0.01
	D	86	0.01	1	0.015	0.01
Shrubland -including wetlands (3.81%)	A	30	0.08	1	0.2	0.29
	B	58	0.08	1	0.2	0.29
	C	70	0.08	1	0.2	0.29
	D	78	0.08	1	0.2	0.29
Water (0.09%)	-	100	0	0	0.99	0
Woodland (8.7%)	A	30	0.002	1	0.4	0.29
	B	55	0.002	1	0.4	0.29
	C	70	0.002	1	0.4	0.29
	D	77	0.002	1	0.4	0.29
Buffer	A	30	0.003	1	0.25	1
	B	48	0.003	1	0.25	1
	C	65	0.003	1	0.25	1
	D	73	0.003	1	0.25	1

3.3 AGNPS Uncertainty Due to DEM Error

3.3.1 Method for Evaluating Error Propagation

In this section, elevation uncertainty is assessed to measure its effect on estimates of sediment yield within the AGNPS model. The uncertainty of terrain data is a factor of the distance that a particular spot on the landscape is from the nearest data point, the variation of terrain between the data points, and the accuracy of the elevation measure in the datasets (Isaaks and Srivastava, 1989). Therefore, as the elevation data points become more generalized, e.g., 30 m (0.22 ac) elevation data as opposed to 10 m (0.025 ac) elevation data, the uncertainty is greater. This generalization effect has repercussions when using the data in spatial modeling applications.

Measuring the uncertainty in the spatial modeling application (AGNPS) due to errors in the input data (elevation data) helps to understand the confidence limits associated with the result, i.e., determining the error in the output, given the operation and the errors in the input attributes (Heuvelink, 1999). There are many methods used to evaluate this uncertainty, such as the Monte Carlo method (Heuvelink, 1999).

The Monte Carlo method (Hammersley and Handscomb, 1979; Lewis and Orav, 1989) used in this study, computes the result of a spatial modeling application (AGNPS) repeatedly, with randomly generated input data (elevation data) sampled from their joint distribution (Heuvelink, 1999). The distribution of the results is then statistically assessed, which reflects the uncertainty in the spatial modeling application due to the error in the input data. Figure 3.7 characterizes the framework for the Monte Carlo process.

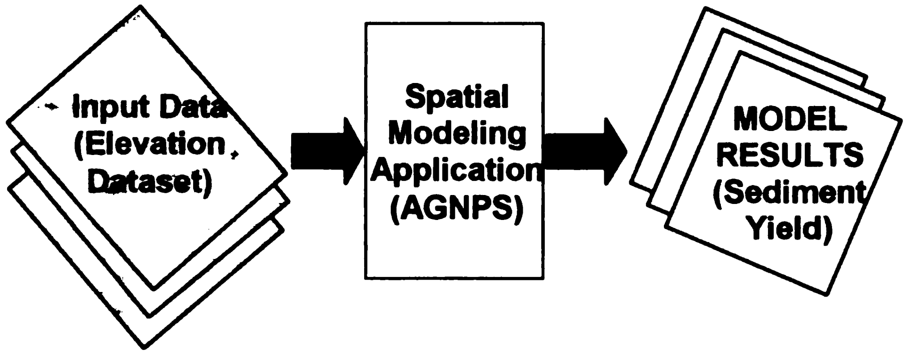


Figure 3.7: Monte Carlo framework for evaluating uncertainty in the spatial modeling application (AGNPS model) due to the error in the input data (elevation dataset).

To assess uncertainty in the elevation data, error must be measured. In this study, a higher accuracy elevation dataset was compared to a coarse elevation dataset:

$$E_{\mathbf{u}} = H_{\mathbf{u}} - C_{\mathbf{u}},\ \mathbf{u} = \{(x_1, y_1), (x_2, y_2), \ldots (x_n, y_n)\} \quad \textbf{(3.2)}$$

where E is the error, C is the coarse elevation dataset, H is the higher accuracy elevation dataset, and **u** are a set of locations (possibly gridded) in a spatial dataset. In addition, E, C, and H are shorthand for spatial datasets consisting of grids of values. Stochastic simulation approach was used to develop a set of error realizations. From the distributions of the error maps a mean and a standard deviation can be obtained which relates to the "true" values of the higher accuracy elevation data (Wechsler, 2000).

A sequential Gaussian simulation was used to produce a set of error realizations based on E. This process is commonly used in geostatistics to evaluate error in the input data for any spatial modeling application (Gooverts, 1997). The elevation error was assumed to have a normal distribution to employ this process. The following steps describe how the sequential Gaussian simulation constructs the error realizations.

Step 1. A variogram must be employed to the error data to account for its spatial structure. Accounting for the spatial structure in the error data allows analysis of the "true" spatial patterns of the error in the elevation for that specific geographic region. A variogram summarizes the relationship between differences in pairs of measurements and the distance of the corresponding points from each other within the dataset. The variogram is calculated by equation 3.3:

$$\gamma(h) = (1/2)\mathrm{Var}[Z(\mathbf{x})-Z(\mathbf{x}+h)] \qquad \textbf{(3.3)}$$

where, h is the lag distance separating pairs of data points, Var is the variance of the argument, $Z(\mathbf{x})$ is the value of the regionalized variable of interest at location $\mathbf{x}$, and $Z(\mathbf{x}+h)$ is the value at the location $\mathbf{x}+h$ (Lin and Teng, 2000). Generally, as the lag distance increases the plot should rise simulating at larger distances the variability is greater.

At large values of the lag-distance (h) the plot tends to level off becoming horizontal. At this distance the variogram has reached a sill, which is theoretically the sample variance. The distance to the sill is called the range where data points are spatially autocorrelated (Bailey and Gatrell, 1995). The nugget is the magnitude of discontinuity at the origin. A variogram model type is decided by plotting the empirical variogram and examining the behavior of the sill, range, and nugget.

Step 2. A randomly generated path is defined for visiting each of the nodes (**u**) or cells throughout the error data.

Step 3. Sequentially visit each node (**u**).

Step 3a. Simple kriging is performed for grid node (**u**) using the variogram model developed in step 1. The result is both an expected value and a kriging variance. The expected value is a best linear unbiased estimate (BLUE), in the least squares sense. The kriging variance is the minimized value of the estimation variance or the variance of the error of estimation. The kriging variance is a measure of the certainty the model has in its estimate and is a function of the variogram model and the distance to surrounding known points. The variance is higher for locations remote from unknown nodes. Kriging is a geostatistical estimation method by which optimal weights are assigned to unknown values based on the variogram model, since the variogram changes with distance the weights depend on the known sample distribution.

Step 3b. Next, a random number from a normal (Gaussian) distribution is drawn for "u" that has a variance equivalent to the kriged variance and a mean equivalent to the kriged value. This number will be the simulated number for that grid node. When all nodes have been simulated this is the first realization.

Step 4. Repeat steps 2 and 3 for all the other realizations using a different random number sequence to generate multiple error realizations of the original error map.

The error realizations match the statistical characteristics specified by the error model. They are equiprobable in the sense that each is representation of the true error in the dataset. The error realizations are then added to the coarse elevation data of the study area. Each new elevation realization for the study area is used for evaluating sediment load in the AGNPS model. The sedimentation results from using the new elevation realizations in the AGNPS model are separately compared to the sedimentation results from the coarse elevation data. The results from the simulations construct a distribution of possible outcomes. The width of this distribution reflects the uncertainty in the AGNPS model due to coarse elevation error.

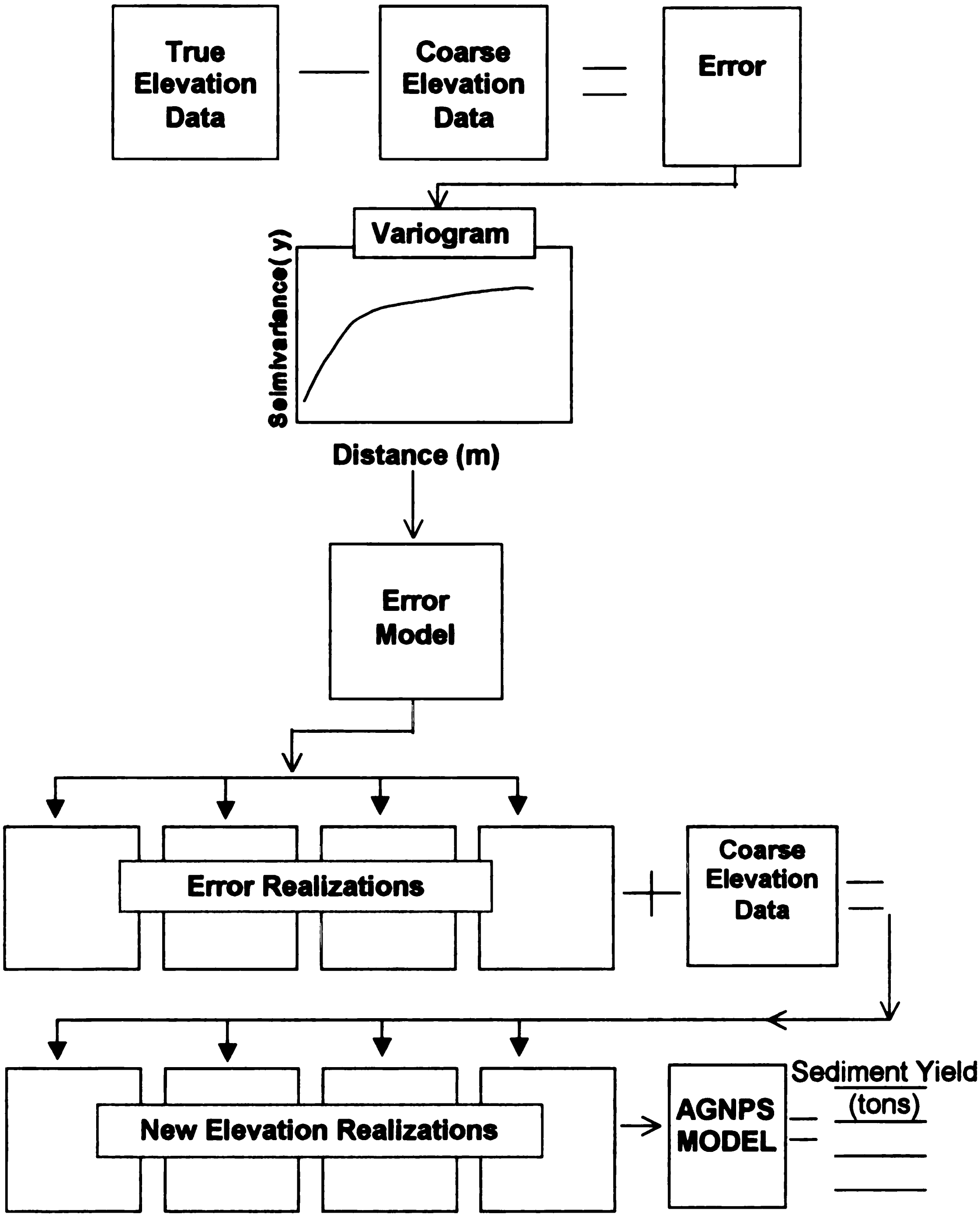


Figure 3.8: Stochastic simulation approach for evaluating uncertainty in the AGNPS model due to error in the coarse elevation data.

3.3.2 Unconditional Stochastic Simulation

If the resulting realizations honor the data values at the sampled locations the process is said to be a conditional simulation. Since the data values in this study do not represent the actual study area values the process is unconditional. The unconditional approach, used in this study, is applied because no information about true elevation is available within the study region. Therefore, an error model is developed by comparing higher accuracy "true" elevation data with a coarse elevation dataset from a geographic region that has similar geomorphological characteristics to the study area. A statistical error model characterizing error magnitudes and spatial patterns observed for the coarse elevation data can be developed by assuming that the error characteristics for the two geographic areas are the same.

3.3.3 Methodology Using the Study Area's Watershed

The unconditional stochastic simulation approach was employed on a 107 ha (264 ac) sub-watershed of Stony Creek (Figure 3.9). Thirty-meter USGS DEM data is available for the study area but higher accuracy elevation data is not. However, a 30 m USGS DEM and higher accuracy elevation data, characterizing the "true" elevation, is available for a watershed located at Michigan State University Farms (MSU Farm watershed), which is 43 km (27 miles) south of the Stony Creek study area (Figure 3.9). The higher accuracy digital elevation data for the MSU Farm watershed (Survey DEM) is generated from ground truth data surveyed at various spacing and gridded to a 0.6 m (2 ft) resolution. The 0.6 m (2 ft) DEM was resampled to 10 m because the 10 m grid for this watershed consisted of 10,968 cells, which was the largest database that operated efficiently in the AGNPS model.

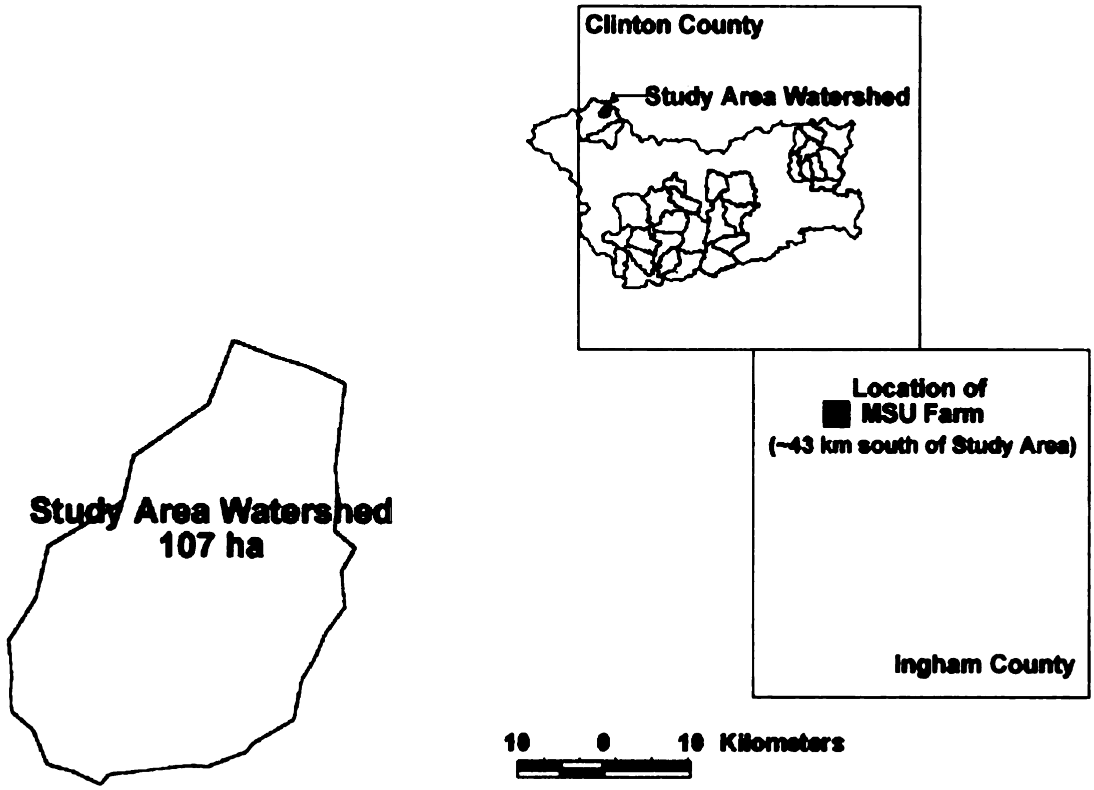


Figure 3.9: Location of study area and MSU Farm watershed site.

The MSU Farm watershed and the study area have similar geomorphological characteristics. Therefore, it is assumed that the error characteristics for the USGS DEM covering the MSU Farm watershed are similar to the unknown characteristics of the USGS DEM overlying the study area in Stony Creek. By assuming that the error characteristics are the same a statistical error model characterizes error magnitudes and spatial patterns observed for the MSU Farm DEM. The objective of this analysis is to employ an error model for the derivation of the USGS DEM of the MSU Farm watershed from higher accuracy elevation data characterizing the "true" elevation. The error model can then be used to generate realizations of error for the 30 m USGS DEM.

The USGS DEM of the MSU Farm watershed was interpolated or resampled to 10 m by nearest neighbor interpolation so it could be compared to its 10 m Survey DEM. The resampled DEM is termed the USGS resampled DEM. The difference between the two data sets was calculated and termed the error (Equation 3.4). Error is a spatially extensive variable; thus, an error magnitude is present for every cell in the watershed.

Thus,

$$\text{USGS resampled DEM} - \text{Survey DEM} = \text{Error} \qquad \textbf{(3.4)}$$

Gstat, a geostatistics modeling program, was used to fit a variogram model to the error (Pebesma, 1999). A variogram model was fit to the empirical semivariogram by a weighted least squares method and yielded a Gaussian model with a sill of 1.76 and a range of 53.8 plus an exponential model with a sill of 6.8 and a range of 115, where each lag had thousands of point pairs (Figure 3.10).

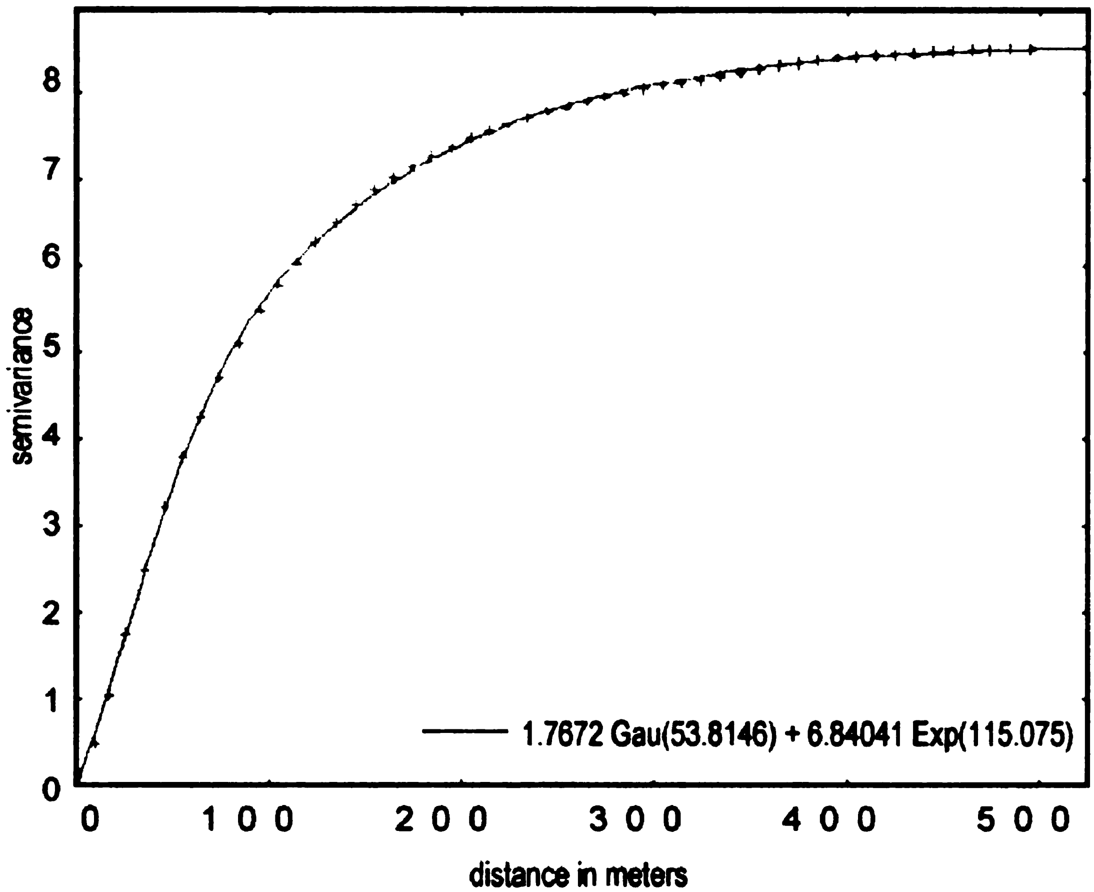


Figure 3.10: The variogram model of the error between the Survey DEM and the USGS resampled DEM of the MSU Farm watershed.

The variogram model, a mean error of 0.6 m (2 ft), and a mask map (ascii grid) identifying the watershed area were used to create a set of error realizations for the study area. The USGS DEM of the study area was interpolated bilinearly to 10 m so each random error realization could be added to it (Equation 3.5). The resampled USGS DEM for the study area is termed the Study Area USGS resampled DEM. The DEM realizations create a collection of alternative equally probable models of spatially distributed variable uncertainty or error (Wechsler, 2000). For the scope of this study, nine new DEM error realizations were created (Figure 3.11a-i).

Thus the study area DEM realizations yield:

Study Area USGS resampled DEM + Error realizations = DEM realizations **(3.5)**

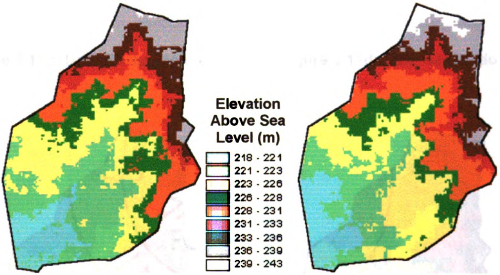


Figure 3.11a: DEM Realization 1 **Figure 3.11b: DEM Realization 2**

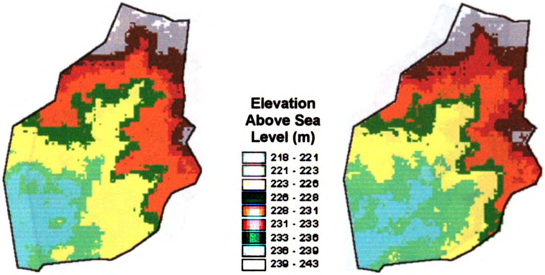


Figure 3.11c: DEM Realization 3

Figure 3.11d: DEM Realization 4

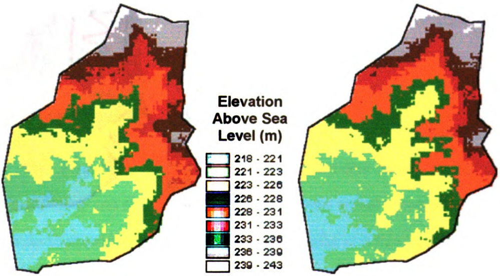


Figure 3.11e: DEM Realization 5

Figure 3.11f: DEM Realization 6

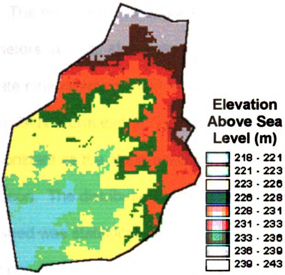


Figure 3.11g: DEM Realization 7

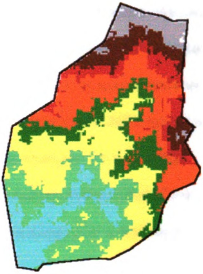


Figure 3.11h: DEM Realization 8

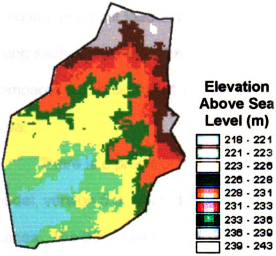
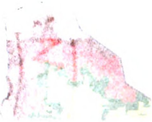


Figure 3.11i: DEM Realization 9

Figure 3.11a-i: DEM Realizations 1-9 of Study Area watershed.



The nine DEM realizations were used to derive the four slope associated parameters (aspect, slope, slope length, slope shape) for the AGNPS model to simulate nine sediment yield results at the outlet of the study area watershed for a 10 yr-24 hr storm event. The other parameters within the AGNPS model were kept constant as the slope-associated parameters changed for each DEM realization. The distribution of the sedimentation results at the outlet of the watershed was statistically assessed to calculate a mean and a standard deviation.

To identify the uncertainty in changing grid cell size from 30 m to 10 m, the USGS resampled DEM for the study area was also used in the AGNPS model to simulate sediment load at the outlet of the watershed. The sediment yield from using each of the nine DEM realizations and the USGS resampled DEM were compared to the sediment yield from using the 30 m USGS DEM for the study area.

There is a precision problem with characterizing cell size in the AGNPS model, version 5.0: when defining cell size the model only allows two decimal places. The 30 m cell size (0.22 acres) matches the actual acreage well enough with two decimal places. However, the 10 m cell size of 0.0247 acres does need more decimal places to characterize its true acreage. The rounding effect increases the cell size to 0.03 acres, which would inflate the watershed size and also increase the results at the outlet significantly over a number of cells, i.e., 10,968 cells. Therefore, to account for these effects each simulation including the 30 m data was increased in cell size by a factor of 100. The 10 m and 30 m cell size yielded 2.47 acres and 22.44 acres respectively. The sediment yield at the outlet will not result in actual sedimentation estimates but can be compared on the same scale to identify uncertainty in the AGNPS model when estimating sediment yield due to error in the DEM.

Chapter 4

4 Results and Discussion

4.1 Stony Creek Secondary Sub-watershed Results

The sediment yield within the 31 Stony Creek secondary sub-watersheds were analyzed by using the AGNPS model to identify site-specific areas along stream segments where filter strips were effective (tons of sediment filtered) as well as efficient (% sediment filtered). Four secondary sub-watersheds were not completed due to errors in their AGNPS input databases, three located in Muskrat Creek (sub-watershed number 10, 15, and 16) and one located in Bad Creek (sub-watershed number 19). The data for the remaining 27 secondary sub-watersheds are presented in this analysis.

There was not a significant difference in the predicted sediment loads at the outlet of each secondary sub-watershed between the baseline and the filter strip scenarios. Filtered sediment at the sub-watershed outlets varied from 3 to 50 tons as shown in Table 4.1. This analysis presented the overall effectiveness or lack of effectiveness of filter strips throughout the watershed. However, throughout this study, the main focus is to prioritize vegetative filter strips by comparing areas of filter strip effectiveness and efficiency along stream segments within the watershed.

Table 4.1: Filter strip effectiveness at the outlet of the 27 secondary sub-watersheds.

Primary Sub-watershed	Secondary Sub-watershed	Sub-watershed Size (ha)	No Filter Strip/Baseline Results (tons)	Filter Strip Results (tons)	Filtered Sediment (tons)	% Reduction
Lost Creek	1	1,320	500	473	27	5.4
	2	604	225	212	13	5.8
Muskrat Creek	3	468	104	101	3	3.0
	4	302	422	373	50	11.8
	5	959	290	264	26	8.9
	6	540	451	425	26	5.7
	7	1,255	371	350	20	5.5
	9	755	230	216	15	6.4
	11	766	276	260	16	5.9
	12	420	215	200	15	7.0
	13	568	231	213	19	8.0
	14	521	179	170	9	5.0
	15	872	261	252	9	3.5
Bad Creek	17	624	222	199	23	10.2
	18	1,107	256	244	13	5.0
	20	694	280	251	29	10.2
	21	522	183	171	12	6.6
	22	918	340	316	24	7.0
Spaulding Drain	23	301	124	114	10	7.9
	24	370	149	136	13	9.0
	25	1,030	268	247	22	8.1
	26	415	104	97	7	6.7
	27	349	147	135	12	8.1
	28	264	119	106	12	10.5
	29	358	142	125	17	12.2
	30	732	229	213	16	7.2
	31	323	116	107	9	7.8

The cell-by-cell analysis did not show one or two distinct areas of filter efficiency or effectiveness throughout the 27 sub-watersheds. Therefore, it was important to assess the reasons for the scattered results. In ArcView, version

3.2, a brushing technique was used to examine the filter strips that were efficiently filtering sediment (>50% reduction) throughout the 27 secondary sub-watersheds (Figure 4.1). In these areas, the tons of sediment produced was minimal compared to other areas within the watersheds and in most cases the upland contributing area entering the filter strip was less than 4 ha (10 ac). This analysis is simulating the effect of filter strips only being able to filter shallow, uniform flow as described by Dillaha et al. (1989). In the areas where the filter strips upland contributing area is greater than 4 ha (10 ac) the majority of the cells have a percent filtered sediment of less than 25%. Therefore, the effect of concentrated flow is defining areas where the filter strips are less effective in reducing sediment, which indicates that filter efficiency is dependant on drainage area.

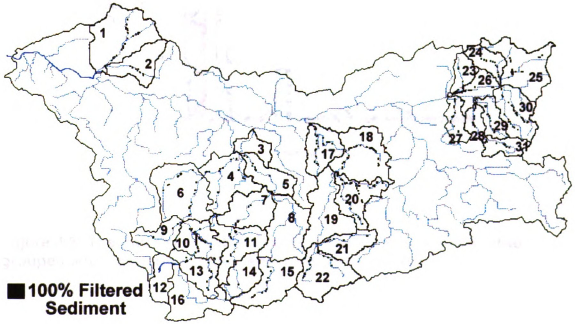


Figure 4.1: 100% filter efficiency for the 27 secondary sub-watersheds of Stony Creek.

Drainage area was compared to percent filtered sediment for the 27 sub-watersheds. The data was compiled into separate bar graphs for the four primary sub-watersheds of Stony Creek to analyze the buffered cell's distribution of filter efficiency with respect to drainage area. Filter strip efficiency was grouped into four categories (0-25%, 25-50%, 50-75%, 75-100%) and drainage area was classified by natural breaks in ArcView into five different categories as shown in the Figures 4.2, 4.3, 4.4, and 4.5. The cells that did not accumulate any sediment in the no filter strip/baseline scenario were excluded from the data.

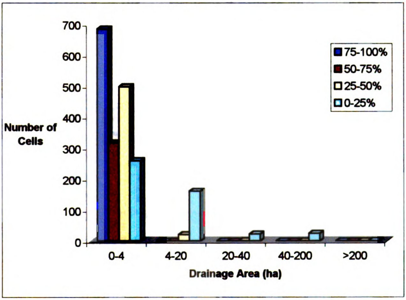


Figure 4.2: The Bad Creek sub-watershed –cells within the filter strip grouped into categories of filter efficiency and drainage area.

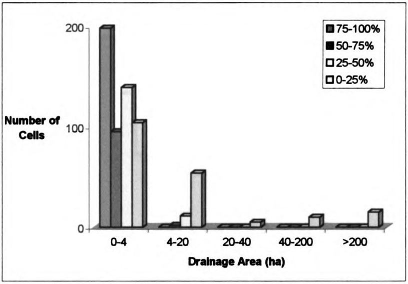


Figure 4.3: The Lost Creek sub watershed –cells within the filter strip grouped into categories of filter efficiency and drainage area.

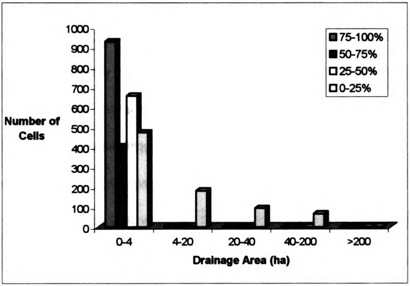


Figure 4.4: The Muskrat Creek sub watershed –cells within the filter strip grouped into categories of filter efficiency and drainage area.

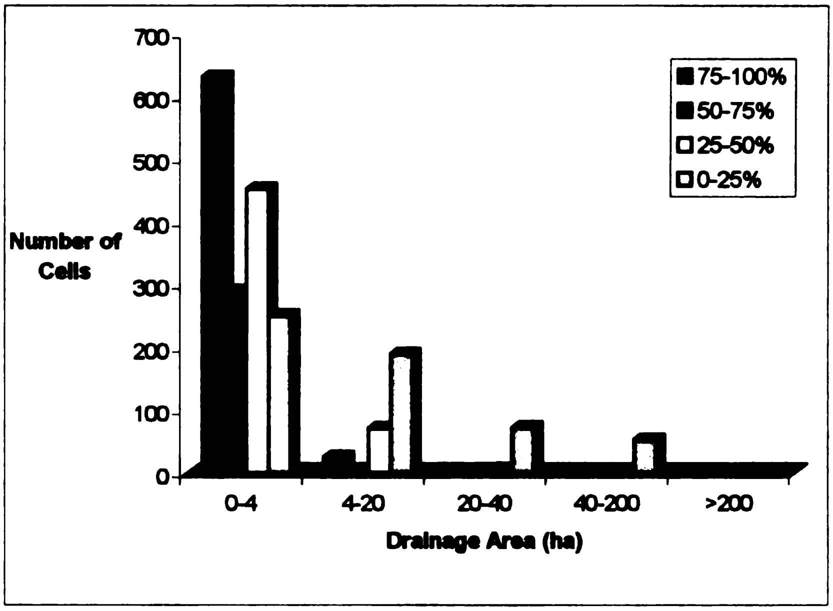


Figure 4.5: The Spaulding Drain sub watershed –cells within the filter strip grouped into categories of filter efficiency and drainage area.

As shown in the figures above the filter efficiency from the four primary sub-watersheds are closely related to drainage area. The cells within the filter strip are more efficient at lower drainage areas (0-4 ha or 0-10 ac) but aren't necessarily filtering much sediment because the draining areas are so small. An analysis of variance (ANOVA) was conducted on the filter strip data of the four primary sub-watersheds to test whether filter efficiency is directly related to drainage area. In each of the data sets the ANOVA indicated that filter efficiency is significantly related to drainage area (p < 0.01) (Tables 4.2-4.5 and Figures 4.6- 4.9).

Table 4.2: ANOVA results for Bad Creek, filter strip efficiency verses drainage area.

Anova: Single Factor

SUMMARY

Groups (Drainage Area in ha)	*Count*	*Sum*	*Average Filter strip efficiency*	*Variance*
0-4	1755	107691.06	61.36	970.91
4-20	181	2662.18	14.71	56.25
20-40	23	240.14	10.44	42.59
40-200	24	148.99	6.21	30.69
>200	1	14.08	14.08-	

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	468011.8	4	117002.95	135.03	4.2E-102	2.38
Within Groups	1714743	1979	866.47			
Total	2182755	1983				

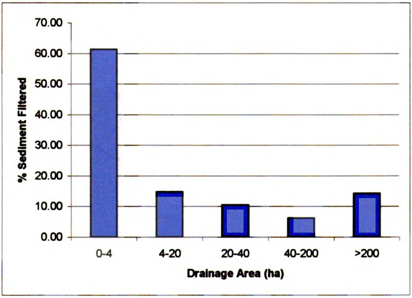


Figure 4.6: The Bad Creek average filter strip efficiency vs. drainage area of the cells within the filter strip.

Table 4.3: ANOVA results for Lost Creek, filter strip efficiency verses drainage area.

Anova: Single Factor

SUMMARY

Groups (Drainage Area in ha)	*Count*	*Sum*	*Average filter strip efficiency*	*Variance*
0-4	536	31440.85	58.66	979.94
4-20	67	1122.88	16.76	155.67
20-40	5	39.93	7.99	19.37
40-200	10	22.93	2.29	0.95
>200	15	84.34	5.62	7.59

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	173559.7	4	43389.91	50.96	3.63E-37	2.39
Within Groups	534734.4	628	851.49			
Total	708294	632				

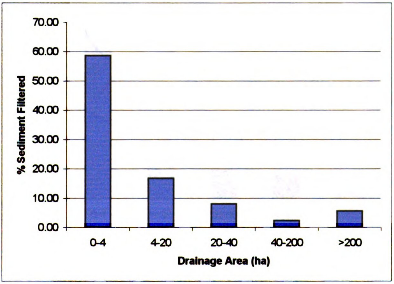


Figure 4.7: The Lost Creek average filter strip efficiency vs. drainage area of the cells within the filter strip.

Table 4.4: ANOVA results for Muskrat Creek, filter strip efficiency verses drainage area.

Anova: Single Factor

SUMMARY

Groups (Drainage Area in ha)	*Count*	*Sum*	*Average filter strip efficiency*	*Variance*
0-4	2460	144923.21	58.91	1025.33
4-20	191	2440.84	12.78	101.41
20-40	95	887.72	9.34	31.05
40-200	66	507.48	7.69	32.65
>200	2	4	2	0.67

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	716665.8	4	179166.46	197.71	1.5E-149	2.38
Within Groups	2545592	2809	906.23			
Total	3262257	2813				

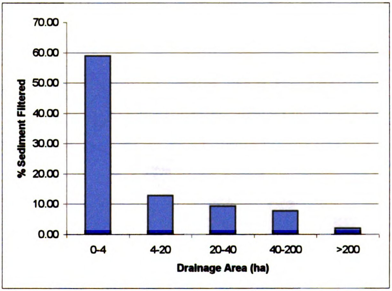


Figure 4.8: The Muskrat Creek average filter strip efficiency vs. drainage area of the cells within the filter strip.

Table 4.5: ANOVA results for Spaulding Drain, filter strip efficiency verses drainage area.

Anova: Single Factor

SUMMARY

Groups (Drainage Area in ha)	*Count*	*Sum*	*Average filter strip efficiency*	*Variance*
0-4	1621	97909.99	60.40	913.82
4-20	288	7410.46	25.73	422.38
20-40	77	925.24	12.02	54.67
40-200	52	604.46	11.62	34.65
>200	4	23.35	5.84	35.64

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	529307.9	4	132326.97	167.67	3.2E-124	2.38
Within Groups	1607636	2037	789.22			
Total	2136943	2041				

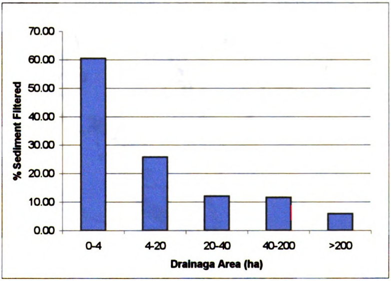


Figure 4.9: The Spaulding Drain average filter strip efficiency vs. drainage area of the cells within the filter strip.

4.2 *Prioritizing Vegetative Filter Strips*

4.2.1 Overview

Vegetative filter strips along stream segments were analyzed throughout the 722 ha (1,784 ac) secondary sub-watershed 20 or also known as the East Bad Creek (EBC) watershed. The EBC watershed was divided into 8,107 cells at a resolution of 30-meters. The filter strip throughout the EBC watershed consisted of 680 cells, which was divided into 500 m (1640 ft) lengths on each side of the stream (Figure 4.10). A detailed analysis of the tons of sediment filtered as well as average filter efficiency for each 500 m filter strip segment was assessed to identify a method of filter strip placement.

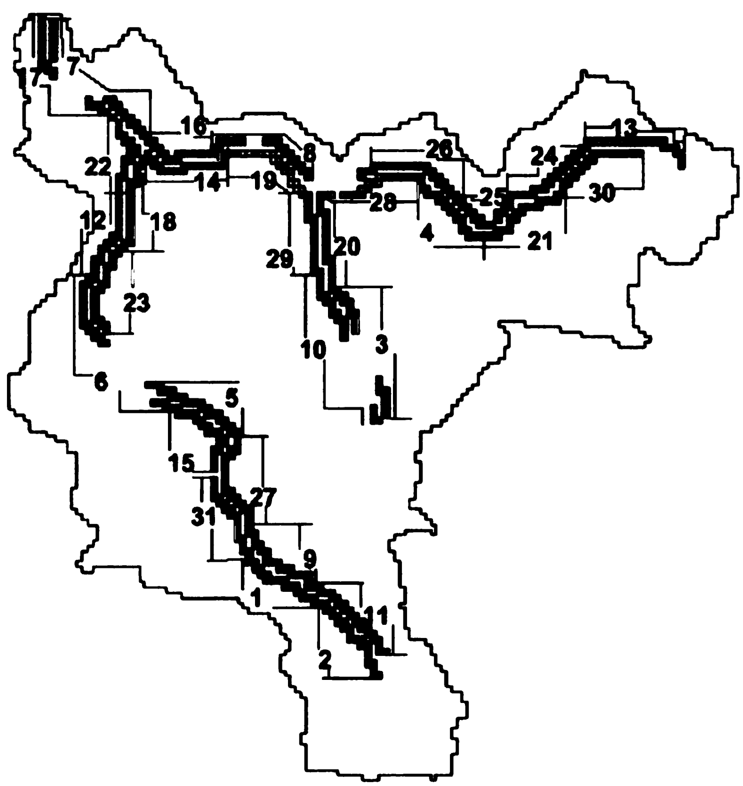


Figure 4.10: The EBC watershed filter strip segments of equal length (500 m), numbered from 1-31 corresponding to a ranking of filter strip effectiveness (filtered sediment in tons) with 1 the most effective and 31 the least effective.

4.2.2 Filter Strip Segment Results

The sediment reduction of each filter strip segments varied from 2 to 6.4 tons, with a mean of 3.7 tons (Table 4.6). The average efficiency of each filter strip segment also fluctuated throughout the watershed, from 25 to 78%, with a mean of 56% (Table 4.6). The filter strip segments were ranked based on the amount of total sediment each segment filtered, where one is the most effective and 31 the least (Figure 4.10 and Table 4.6).

Table 4.6: Thirty-one filter strip segments ranked by the amount of sediment filtered within the EBC watershed.

Ranking Number	Total Amount of Sediment Filtered (tons)	Average Percent of Sediment Filtered (%)	Total Drainage Area (ha)
1	6.4	62	8.5
2	5.5	57	32.0
3	4.9	65	36.0
4	4.9	69	21.5
5	4.8	29	34.4
6	4.5	63	19.8
7	4.3	78	7.3
8	4.3	60	9.7
9	4.2	70	6.5
10	4.2	38	29.1
11	4.1	49	14.2
12	4.0	69	6.9
13	3.8	40	37.7
14	3.8	58	44.5
15	3.6	69	4.5
16	3.5	69	3.6
17	3.5	60	7.7
18	3.2	40	13.0
19	3.1	59	6.9
20	3.1	58	7.3
21	3.1	51	46.2
22	3.1	73	10.5
23	3.0	25	14.2
24	3.0	73	10.5
25	3.0	71	4.5
26	2.9	40	14.2
27	2.9	52	22.7
28	2.4	51	8.5
29	2.2	63	4.5
30	2.2	29	5.7
31	2.0	36	6.1

The average percent of sediment filtered in Table 4.6 identifies the most efficient filter strip segments. However, the averaging is inaccurately representing the actual efficiency because within most filter strip segments the majority of the filter strip drains small areas (0-4 ha), and in most cases the total amount of sediment entering the stream from the smaller drainage areas was filtered, i.e., >50% of sediment filtered. Within certain stream segments there may be one or two distinct areas within the filter strip segment where the filter efficiency was very low, i.e., less than 25%, with a large corresponding drainage area indicating concentrated flow. By averaging the efficiency throughout the filter strip segment, shallow, uniform flow areas that make up the majority of the segment was diluting its inefficiencies.

For example, filter strip segment number 14 filtered 58% with a total drainage area of 44.5 ha (110 ac). In this particular segment, one cell filtered 39.2 ha (97 ac) and the rest of the drainage area was distributed evenly across the remaining cells within the segment. The efficiency of this cell was 5%, which did not significantly lower the overall filter efficiency for the filter strip segment because 19 other cell's filter efficiency was higher than 30%. This analysis indicates the importance of assessing filter strip efficiency in each cell within the filter strip segment instead of using the segment's average.

To determine why filter strips within the EBC watershed filtered better than others the drainage area and slope were analyzed. First, an ANOVA was preformed to assess the influence of filter efficiency and drainage area in the EBC watershed (Table 4.7 and Figure 4.11). This analysis was comparable to the 27 sub-watershed ANOVA results throughout Stony Creek, where the filter efficiency is significantly influenced ($p < 0.01$) by drainage area. Therefore indicating as the drainage area of the filter strip increased, suggesting concentrated flow, filter strip efficiency decreases.

Table 4.7: ANOVA results for the EBC watershed, filter strip efficiency verses drainage area.

Anova: Single Factor

SUMMARY

Groups (Drainage Area in ha)	*Count*	*Sum*	*Average of filter strip efficiency*	*Variance*
0-4	615	36895.73	59.99	951.58
4-20	58	807.80	13.93	50.47
20-40	6	40.49	6.75	33.20
40-200	2	35.81	17.91	41.68
>200	1	0	0	-

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	132814.7	4	33203.67	38.27	6.81E-29	2.39
Within Groups	587353.6	677	867.58			
Total	720168.3	681				

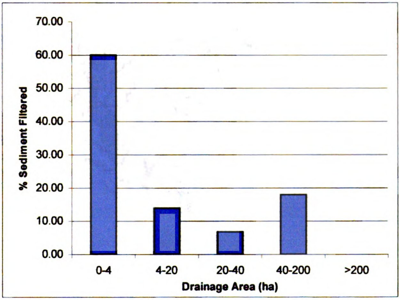


Figure 4.11: The EBC watershed average filter strip efficiency verses drainage area of the cells within the filter strip.

The slope of the EBC watershed is an insignificant parameter to analyze for filter strip efficiency because the watershed is very flat (Figure 4.12). The average slope across the watershed is 0.82% with an overall elevation change of 6 m or 20 ft (244 m to 230 m).

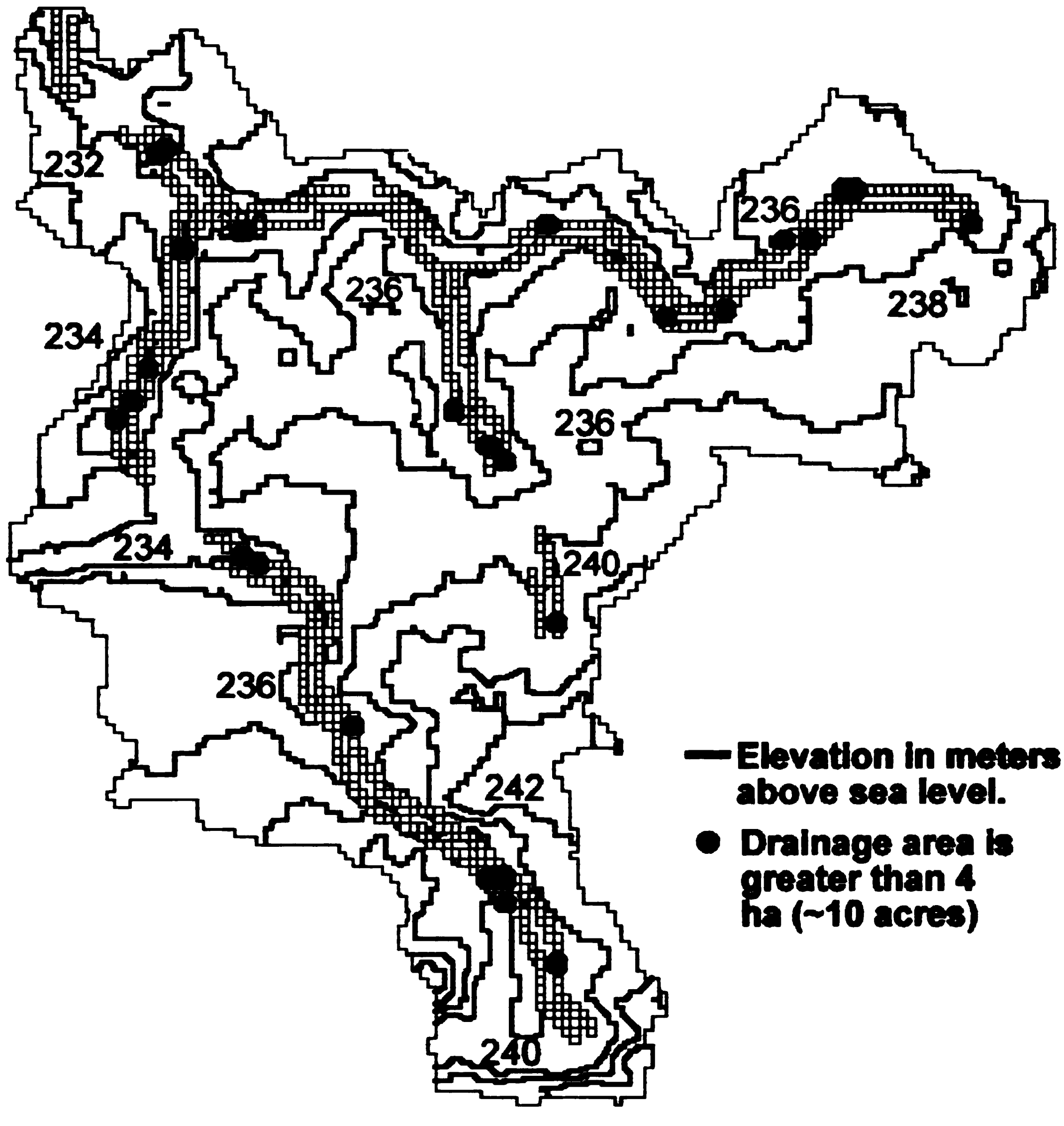


Figure 4.12: Topography and drainage areas greater than 4 ha (~10 acres) within the filter strip of the EBC watershed.

The filter strip segment results show the importance of identifying the efficiency in site-specific areas within the filter strip, especially the areas where concentrated flow exists. The corresponding 18 filter strip segments that have at least one or more cells that drain greater than 4 ha (~10 acres) are segments: 2, 3. 4, 5, 6, 7, 10, 11, 13, 14, 18, 21, 22, 23, 24, 26, 27, and 30. The effects of concentrated flow are detrimental to filter strip efficiency. Individual cells within the filter strips that have drainage areas greater than 4 ha generally were inefficient. The 24 cells that drained greater than 4 ha, defining concentrated flow areas, are highlighted in Figure 4.12. The total amount of sediment produced in the 24 areas from the no filter strip/baseline scenario resulted in 209 tons, which is 47% of the total amount of sediment entering the stream. The total amount of filtered sediment from installing the filter strips in the concentrated flow areas resulted in a reduction of 25 tons. The average filter strip efficiency in these areas was only 15%.

The filter strips are very efficient in the remaining areas throughout the watershed where uniform flow occurs (drainage area is < 4 ha). The average efficiency is 63%. However, the total sediment delivered to the stream in these areas is 231 tons. By incorporating a filter strip in these areas the sediment load was reduced by 93.4 tons.

This analysis indicates that the majority of the sediment delivered to the stream originates from the areas of concentrated flow. When the drainage area is greater than 4 ha excessive sedimentation exists as show by the 47% sediment load. Filter strips are not effective in areas of concentrated flow, e.g., 15% average efficiency, therefore it is imperative to identify areas of concentrated flow and model the effects of reducing sediment by installing other conservation practices, such as sedimentation basins, reconstructed wetlands, or grass waterways. Vegetative filter strips should then be analyzed in areas where shallow, uniform flow occurs.

The remaining filter strip segments that do not have any areas of concentrated flow should then be analyzed. For example, removing the 24 concentrated flow areas from the 17 filter strip segments would create an entirely different ranking of the 31 segments as shown in Table 4.8. Filter strip segment 1 did not contain any concentrated flow areas and also filtered the most amount of sediment therefore its rank remains the same. This scenario recognizes that filter strips should be prioritized based on the amount of sediment each segment filters but the concentrated flow areas must separately be evaluated.

Table 4.8 Ranking order from 1 to 31 of vegetative filter strip effectiveness without the concentrated flow areas.

Rank without Concentrated Flow Areas	First Ranking Order	Total Amount of Sediment Filtered (tons)	Filter sediment in Concentrated Flow Areas (tons)	Filtered Sediment without the Concentrated Flow Areas (tons)
1	1	6.4	0.0	6.4
2	4	4.9	0.2	4.7
3	8	4.3	0.0	4.3
4	9	4.2	0.0	4.2
5	12	4.0	0.0	4.0
6	15	3.6	0.0	3.6
7	7	4.3	0.8	3.5
8	16	3.5	0.0	3.5
9	17	3.5	0.0	3.5
10	6	4.5	1.1	3.4
11	2	5.5	2.3	3.2
12	19	3.1	0.0	3.1
13	20	3.1	0.0	3.1
14	25	3.0	0.0	3.0
15	24	3.0	0.1	2.9
16	5	4.8	1.9	2.9
17	27	2.9	0.2	2.7
18	11	4.1	1.5	2.6
19	22	3.1	0.5	2.6
20	21	3.1	0.5	2.6
21	10	4.2	1.7	2.6
22	3	4.9	2.5	2.5
23	28	2.4	0.0	2.4
24	14	3.8	1.5	2.3
25	29	2.2	0.0	2.2
26	31	2.0	0.0	2.0
27	30	2.2	0.3	1.9
28	26	2.9	1.1	1.8
29	13	3.8	2.5	1.3
30	18	3.2	2.0	1.3
31	23	3.0	2.1	1.0

4.3 Uncertainty in the AGNPS Model Due to DEM Error

4.3.1 Overview

Visually comparing the 30 m USGS DEM and the Survey DEM of the MSU Farm watershed suggest that there is a substantial amount of terrain information missing from the 30 m USGS DEM (Figure 4.13). For example, shown in the Survey DEM the roads, railroads, and hills are obvious, whereas in the 30 m USGS DEM these areas are generalized and do not emerge from the image (Figure 4.13). The error map measures the errors associated to the terrain variation from the 30 m USGS DEM and the Survey DEM. The negative error values are the lower areas (roads or railroad tracks) and positive error values are the higher areas or hills. The elevation variability between the two datasets has a range from -8.5 to 3.9 m (-28 to 13 ft), with a mean difference of -0.6 m (-2 ft) and a standard deviation of 0.93 m (3 ft). Therefore, on average the 30 m USGS DEM underestimates the Survey DEM by 0.6 m (2 ft) (Figure 4.14).

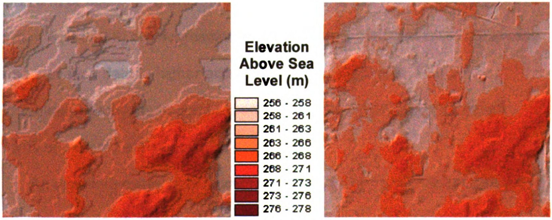


Figure 4.13: 30 m USGS DEM (Left) and 10 m Survey DEM (Right) of MSU Farm watershed.

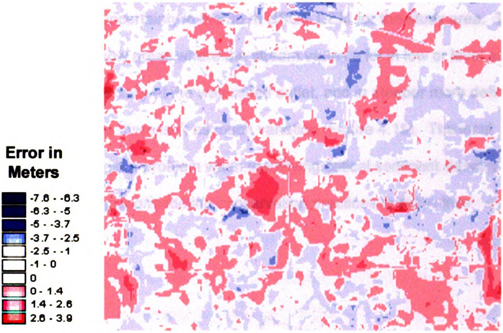


Figure 4.14: Error Map –difference of USGS DEM and Survey DEM of MSU Farm watershed (white=no error, red=positive error, blue=negative error)

4.3.2 Local Effects

Figure 4.14 depicts areas in the 30 m USGS DEM where high elevations (273-278 m or 896-912 ft) are underestimated, whereas lower elevations (261-256 m or 856-840 ft) are overestimated. This bias is standard when comparing coarse elevation data to higher accuracy sources (Gooverts, 1997). Unfortunately, such bias becomes a serious problem when trying to detect patterns of extreme elevation values.

An excellent example of local uncertainty is identifying stream location. Stream location is an important aspect of hydrologic modeling especially for identifying areas of excessive NPS pollution, i.e., major sediment delivery areas. Each stream was derived from the nine DEM realizations, which was developed to help explain local uncertainty in the 30 m USGS DEM for the study area. The stream derivations show areas where the stream varies a lot in low or level areas of the watershed especially at the watershed outlet, and in steeper more defined areas upstream where there isn't as much variation (Figure 4.15). The local uncertainty can become a problem when identifying critical sediment delivery areas as well as designing/modeling conservation practices to aid in their reduction.

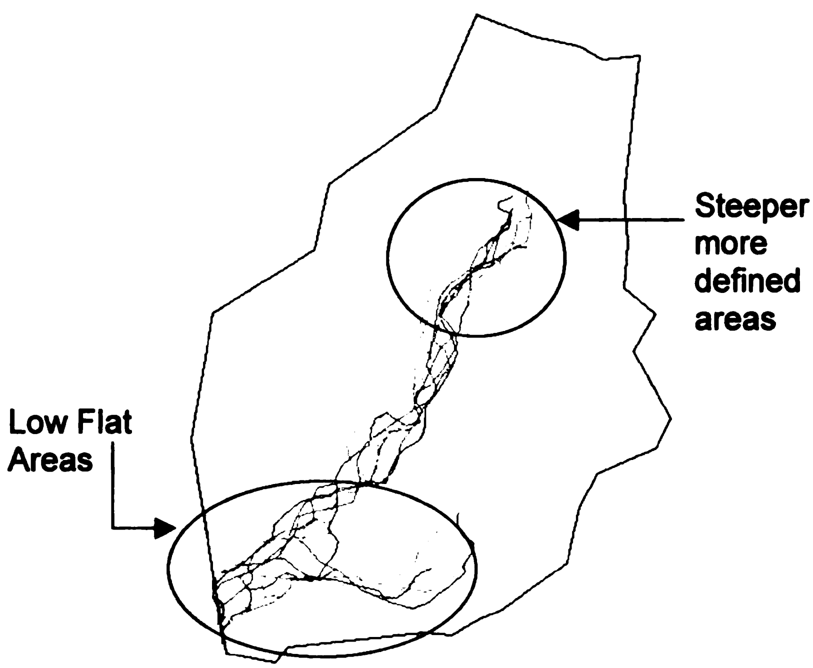


Figure 4.15: The streams derived from the nine DEM realizations are showing locations of local uncertainty in the 30 m USGS DEM.

In distributed parameter models there is a local uncertainty for every output in every cell representing the watershed due to DEM error. The local uncertainty is very difficult to quantify for every cell. The effect of local uncertainty propagates to the outlet; therefore, this effect can be measured at the outlet, which produces a global uncertainty measurement.

4.3.3 Global Effects

The overall uncertainty in the coarse DEM causes problems when using the AGNPS model or any NPS pollution model to assess sediment yield at the outlet. Comparing the 30 m USGS DEM and the nine 10 m DEM realizations used in the AGNPS model to estimate sediment load resulted in an average reduction of 37% with a standard deviation of 118 (Table 4.9). The standard deviation identified that randomly changing the slope-associated parameters had a large effect in simulating sediment yield at the outlet when using the AGNPS model. The 37% reduction suggests that the 30 m USGS DEM is profoundly overestimating the sediment yield.

The decrease in sediment load was logically consistent with the findings of Wolock and Price (1994). They reported that as grid cell size decreased, the depth to water table increased. The denser elevations produced detailed slope attributes thus, identifying more areas where sediment is deposited upstream, which accounts for the decrease in sediment load for each simulation.

Table 4.9: Sedimentation results at the outlet from the nine simulations.

DEMs used in the AGNPS Model	Sediment Load at the Watershed Outlet	% Difference From the Original 30 m Data
30 m	3358	
DEM Realization 1	1926	43
DEM Realization 2	2059	39
DEM Realization 3	2046	39
DEM Realization 4	2151	36
DEM Realization 5	2122	37
DEM Realization 6	2125	37
DEM Realization 7	2078	38
DEM Realization 8	2211	34
DEM Realization 9	2351	30
Mean	2119	37
SD	118	

4.3.4 Resampled USGS DEM Results

The resampled USGS DEM was used to measure the effect of changing cell size from 30 m to 10 m when predicting sediment load in the AGNPS model. As the slope-associated parameters derived from the elevation data remain the same the 10 m cell size redefines the soil and land use/cover characteristics and better represents the actual size of the watershed. This analysis has shown that the cell resolution greatly reduces the sediment load by 41% (Table 4.10). A z-score test was used to compare the sediment yield from the resampled USGS DEM and the DEM realizations, which identified that they were significantly different at an alpha of 0.01. The analysis suggests that incorporating the effects of uncertainty into the resampled DEM, e.g., DEM realizations, increases estimates of sediment yield. This indicates that 30 m DEM error is affecting the actual estimates of sediment yield. However, the results for sediment yield are not actual tons because of the cell size precision problems described in the materials and methods section.

Table 4.10: Sedimentation results at the outlet from the resampled DEM.

DEMs used in the AGNPS Model	Sediment Load at the Watershed Outlet	% Difference From the Original 30 m Data
30 m	3358	
Resampled DEM	1993	41

4.3.5 Cell Size Effects in the AGNPS Model

The change in cell size, original 30 m DEM cell size to 10 m resampled or interpolated DEM, not only affects the representation of the soil and land use/cover characteristics within the watershed, but it redefines the actual size of the watershed. When larger cell size is used the estimates of sediment yield may be larger because not all the cells are entirely within the actual boundary of the watershed (Figure 4.16). The cells that lie on the boundary overestimate the size of the watershed, which increases the sediment load. When smaller cell size is used, the watershed is closer to its actual size, which decreases the sediment load at the outlet and represents a closer actual estimate of sediment load for the watershed.

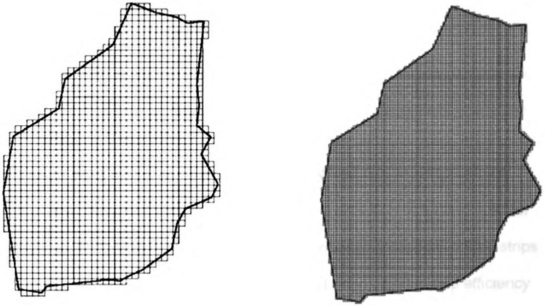


Figure 4.16: 30 m grid cell layout (Left) and 10 m grid cell layout (Right) for the study area watershed.

The coarser grid layout (30 m cell size) estimates the size of the watershed at 116 ha (287 acres), while the higher density 10 m cell size estimates the area of the watershed at 110 ha (271 acres). This effect is a major concern because in addition to all of the other cell size effects (redefining the land use and soils) it too enhances the uncertainty of the sediment load not only at the outlet but also throughout the entire watershed.

Chapter 5

5 Conclusions

5.1 General Conclusions

5.1.1 Prioritizing Vegetative Filter Strips

The AGNPS model combined with the AVNPSM GIS Interface allows evaluation of multiple watershed scenarios of high resolution to be analyzed for BMP placement. This research has shown that prioritizing vegetative filter strips within agricultural watersheds primarily should be based on filter strip efficiency and effectiveness. The size of the upland contributing area and the flow path through the filter strip is directly related to filter efficiency (% sediment filtered) and thus filter strip effectiveness (tons of filtered sediment). The model results identified vegetative filter strip inefficiencies (<25%) when drainage area was greater than 4 ha (10 ac). Filter strips were inefficient in these areas because the increase in drainage area simulated where concentrated flow paths would occur.

The results of this study were similar to vegetative filter strip field research conducted by Dillaha et al. (1989) and Robinson et al. (1996), in which the size of the upland contributing area affected filter strip performance. Therefore, it is imperative to assess the effects of vegetative filter strips in areas where shallow, uniform flow occurs. The main problem with this analysis is that areas of shallow, uniform flow are usually not critical delivery areas of concern, i.e., excessive NPS pollutant loading areas.

Vegetative filter strips alone were found to be relatively inefficient especially in areas where concentrated flow occurs. For these potential risk areas, other conservation practices, such as grassed waterways, reconstructed wetlands, or sedimentation basins, should be considered. Evaluating other conservation practices in the AGNPS model will help to reduce the pollutant load in the most efficient way possible. The results at the watershed outlet may then show a higher reduction in sediment than if filter strips alone are the sole conservation practice considered.

The AGNPS model in conjunction with the AVNPSM interface proved to be an efficient, user-friendly tool for evaluating conservation efforts on a watershed scale. The model in conjunction with prioritizing potential risk areas should be incorporated into the conservation planning process when allocating federal funds from cost-sharing programs. This type of watershed scale analysis establishes the foundation for conservation decisions in critical areas subjected to NPS sediment delivery.

5.1.2 AGNPS Uncertainty Due to DEM Error

The unconditional stochastic simulation technique does not ensure an actual "real" elevation map of the study area but identifies many equal probable elevation maps within which we can state where the true elevation map may lie (Ehlschlaeger and Shortridge, 1996). As shown in this study, by using higher accuracy elevation data, of a small watershed with little terrain variation, the AGNPS sedimentation estimates are dramatically decreased (37%) from the estimates when using the coarse elevation data. Cell size effects were also evaluated to identify the difference in sediment estimates when only changing cell size from 30 m to 10 m, all other parameters stayed constant. The difference of sediment yield at the outlet of the watershed from changing cell size resulted in a 41% reduction. The percent reduction from changing the cell size (41%) and from the DEM uncertainty (37%) were compared, which identified that, they were statistically significant. Although, this data states that by using a smaller grid cell will allow a closer estimate of the actual sedimentation results by better representing the watershed's size and the soil and land use/cover characteristics it is equally important to state the significance of the DEM uncertainty.

In the present analysis, the overall difference due to 30 m USGS DEM error and cell size effects was significant and could become a detrimental problem when using the results from the AGNPS model for identifying areas of BMP placement within watersheds. Therefore, pinpointing potential risk areas in terms of sediment load do require increase accuracy in elevation data but in particular further analysis needs to be explored on the effects of changing cell size.

5.2 Future Considerations

Future work should include the use of water quality models to first identify the critical delivery areas within the watershed, prioritize them by the amount of sediment that is delivered into the stream, and then evaluate which best management practice would be most appropriate. The technique for prioritizing critical sediment delivery areas can be incorporated into the AGNPS AVNPSM GIS interface by writing an ArcView Avenue script. The conservation practice costs and soil rental rates can also be incorporated into the AVNPSM interface to evaluate the economics of installing effective conservation practices.

Accurate elevation data is an important variable when using water quality models. For example, the DEM analysis suggested an increased uncertainty in the results generated by the AGNPS model due to the error in the 30 m USGS DEM. Although the error was small it was significant. Using actual higher accuracy or ground truth elevation data for the study area to simulate sediment yield in the AGNPS model should test the unconditional stochastic simulation approach that was used in this study. The sedimentation results from using the two elevation datasets in AGNPS should be compared as well as the simulated error and the actual errors in the 30 m USGS DEM.

The grid cell size effects contributed significantly in the uncertainty of estimating sediment at the outlet of a watershed. These effects require an in-depth evaluation to characterize the reasons why estimates of sedimentation at the outlet and throughout the watershed differ. This in-depth analysis will help identify which of the model limitations (accuracy in watershed size or soil/land use parameters) is the greatest source of error in the final estimate.

In addition, the AGNPS model must be very precise when characterizing the cell size. The model characterizes grid cell size in acres, but only allows for two decimal places. This becomes a problem when using a small cell size such as 10 m. Over a large number of cells the acreage of the watershed will be very different than the true size of the watershed if the cell size is rounded down or up. The increase or decrease in the "true" cell size will increase the error throughout the watershed and propagate to the outlet results. This is one of the major limitations of the AGNPS model when using a small cell size. This precision problem must be analyzed further.

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APPENDICES

Appendix A: Gstat program

APPENDIX A

Gstat Program

```
#
# Unconditional Gaussian simulation on a mask
# (local neighborhoods, simple kriging)
#
# defines empty variable:
data(mserr): dummy, sk_mean=0.6, max=30;
variogram(mserr): 1.7672 Gau(53.8146)+6.84041 Exp(115.075);
mask: 'lc1mask.asc';
method: gs; # Gaussian simulation instead of kriging
predictions(mserr): 'lc1sim';
set nsim=10;
```