ENVIRONMENTAL AND ECONOMIC MODELING OF NON-POINT SOURCE POLLUTION CONTROL: OPTIMIZED LAND USE SYSTEMS AND VEGETATIVE FILTER STRIPS

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Chapter I

BACKGROUND, PROBLEMS, AND OBJECTIVES

Introduction

Agricultural production activities have long been identified as a major contributor to non-point source (NPS) pollution of streams, lakes and reservoirs. Such NPS pollution by sediment, nutrient, pesticides and pathogens is a significant threat to water supplies, waterways and wildlife habitats in many parts of the United States (EPA, 2003). Examples of studies illustrating the threat from agricultural pollution include;

> A U.S. Geological Survey (USGS) study estimated that 71% of U.S. cropland (nearly 300 million acres) is located in watersheds where the concentration of at least one of four common surface water contaminants (nitrate, phosphorus, fecal coliform bacteria and suspended sediment) exceeded criteria for supporting water-based recreation activities (USDA/ ERS, 2000).
>
> The presence of agricultural chemicals in drinking water supplies creates public health concerns and health risks increase in regions with geologic features conducive to rapid movement of water from the soil surface to aquifers used for drinking water (Bosch and Truman, 2000). The types and concentrations of nutrients and pesticides found in streams and groundwater are closely linked to land use and the chemicals applied. For example, in studies completed in 1998 by

the National Water–Quality Assessment (NAWQA) program, nitrate and pesticides were most frequently detected in shallow groundwater, less than 30 m below the land surface (USGS, 1999).

Comprehensive estimates of the damages from agricultural pollution are lacking, but soil erosion alone is estimated to cost water users $2 billion to $8 billion annually (Ribaudo et al., 1999). Sediment deposited in lakes and reservoirs tends to degrade water quality, destroying aquatic organisms and disrupting fish populations (Nelson, 2001). Sediment deposits also contribute to loss of storage, impedance of flood control measures and reduction in long-term water supplies. Sediment can also transport other pollutants, like phosphorus and nitrogen (Neitsch et al., 2002), two elements involved in lake eutrophication and a resultant unpleasant odor. Fish are killed in the eutrophied lakes because of reduced dissolved oxygen in the water, and recreation is deterred.

In recent years, the agricultural community has started to address NPS pollution problems through the use of best management practices (Nelson, 2001). Significant conservation and environmental gains were made in terms of introducing and refining conservation policy tools enhancing water quality and maintaining wildlife habitat through conservation tools like compliance, land retirement and cost-sharing (Anderson, 1995). The Conservation Reserve Program (CRP), the largest land retirement program with an annual budget about $1.6 billion, currently enrolls about 10% of the country's cropland (Feng et al., 2004).

Clearly NPS control measures involve costs to landowners. However, as concern for the environmental impacts of agricultural production increase, non-economic factors must be considered in the decision process to minimize conflicts with land users over watershed management plans that fail to reflect economic uses (Yanggen et al., 2002).

Since economic and environmental factors interact with each other, it is necessary to investigate possible tradeoffs in the decision making progress. One alternative for proper watershed management would be selection of appropriate land uses for each land unit and using management practices that maximize profit with minimum environmental impact. Ultimately, however management takes place on a landscape in response to the desires of private landowners and their decisions will, in large part, reflect economic criteria such as income maximization (Beaulieu et al., 2000). On the other hand, there is increasing pressure to have watershed management planners' focus on how resources can be used to maintain or enhance water quality and ecological integrity. Thus, there is a need to search for methodologies that serve both environmental and economic issues.

One of the mechanisms currently being used to induce a voluntary shift to an environmentally friendly land use system or to land retirement is through government incentive payments. The Conservation Reserve Program (CRP) is one such voluntary program which offers an annual rental payment for 10 to 15 years along with cost share assistance to eligible producers (USDA, 2003). The payments are intended to establish long-term resource conserving covers to reduce soil erosion, improve water quality and enhance wildlife habitat on eligible land. Thus, CRP, enacted in the 1985 Farm Bill, removes sensitive croplands from production. CRP acres must be maintained and not harvested. In return, farmers receive an annual rental payment from the government (Walsh et al., 1996). Alternatively, biomass energy crops, such as switchgrass, can be grown and harvested for alternative energy purposes (ethanol production) and still provide environmental benefits. One such cover, switchgrass, was chosen by the Department of Energy (DOE) as the model herbaceous species for development as a bioenergy feedstock crop (Fuentes and Taliaferro, 2002) and has been proposed as a crop that minimizes sediment problems.

Biomass has the potential to provide significant sources of energy and fiber in selected regions of the country while providing both economic and environmental benefits to the agricultural community (Tolbert and Downing, 1995). Initial studies of the small-scale plantings of short-rotation woody crops and herbaceous energy crops indicate that these crops can provide environmental benefits (e.g., soil conservation, increased biodiversity and reduced fertilizer runoff) while improving farm income (Tolbert and Schiller, 1995). No known analysis has been made on a watershed scale to evaluate the effect of optimal replacement of agricultural crop land with bio-energy crop production.

A reasonable question is “will producers shift to environmentally friendly crops?” One assumption is that producers will readily shift from the highly profitable crops to environmentally friendly crops such as switchgrass as long as they receive incentives greater than or equal to the difference in net returns from two production systems. An alternative approach is the use of regulations such as pollution taxes to control NPS pollution, giving the producer the option of paying tax or reducing the amount of pollution to the required level (Hartwick and Olewiler, 1998). The option of reducing pollution by switching to conservation crops such as switchgrass over paying tax could be attractive to many producers. Another policy option is the use of uniform government regulation in which each producer is expected to reduce pollution by the same amount.

As an alternative to voluntary or mandatory land allocation over the watershed producers are encouraged by the use of incentives to place buffer/filter strips adjacent to the crop field (Wyatt, 1999). These filter strips can be undeveloped land where the existing vegetation is left intact, or may be land converted from cropland to vegetation such as native grasses. The purpose of vegetative filter strips or riparian buffers is to protect streams and lakes from pollutants such as sediment, nutrients and organic matter buffer/filter strips often provide several benefits to wildlife, such as travel corridors, nesting sites and food sources.

One needed task which has not been evaluated on its environmental and economic merits and demerits is the use of buffer or vegetative filter strips over targeting programs like CRP. An alternative that has not been previously proposed is the use of filter strips within the field, located adjacent to field drains and waterways (Barfield, 2005, personal communication).

Problem Statement

Protecting water bodies from NPS pollution must be accomplished in a cost effective way. The Fort Cobb watershed in Oklahoma is one example of one of the most intensive agricultural farming areas of Oklahoma according to Smolen and Lee (1999). They indicate that peanuts, wheat, alfalfa and other row crops are grown throughout the watershed. Due to its high dollar value, much of the farmland has been continuously planted to peanuts for several years using clean cultivation combined with soils that are very coarse and fragile. This clean cultivation allows excessive erosion. Peanuts can generate more income to the farmers than crops like hay that reduce soil erosion, however, the present problems for the farmers and the public as a whole are pesticide and nutrient runoff, soil erosion and destabilization of riparian areas. The shift to environmentally friendly production systems such as switchgrass has a long-term positive impact on soil productivity, which is also a concern for the producers. Converting relatively more erodible parts of the watershed to crops that generate less sediment and nutrient is an option that could have the highest environmental impact and must be evaluated. However, it is also important to look at how this conversion would affect the income that accrues to the producers.

A social problem confronting all types of pollution is that the polluter enjoys exclusive benefits to the economic activity causing pollution, while the costs of that pollution are shared with society at large (Stoecker, 2005). By imposing the costs on others, the polluter has insufficient incentive to minimize pollution i.e. the producers would prefer to stay in a production system that causes more pollution as long as they can earn more benefits than an environmentally friendly production system would generate.

In view of all these conflicting issues, an integrated watershed management approach is generally recognized as the most practical and efficient way to improve water quality while maintaining economic viability (Yanggen et al., 2002). Single discipline centered approaches are inadequate in NPS pollution control since most watershed problems are complex and the solutions need to satisfy many stakeholders. A desirable focus, therefore, is to determine the optimal land allocation on the watershed that: 1) minimizes sediment and nutrient yield while maximizing income through proper land selection or 2) use of BMPs such as buffers and vegetative filter strips to trap sediment and nutrient.

Research indicates that filter strips are effective in the control of many agricultural and urban non-point source pollutants, especially sediment. Field research on filter-strip width, using grass as the filter material, conducted in Kentucky, Indiana, Iowa, Maryland and Virginia indicate that filter strips are very effective in removing sediment from runoff, with the average reduction ranging from 56% to 95%, depending on soil characteristics, slope, rainfall and runoff conditions and filter width (Barfield et al., 1979; Leeds et al., 1994). To determine the feasibility of using filter strips, the economic impact of taking land out of production to construct filter strips must be minimized concurrent with making reduction in sediment load. This requires knowledge of appropriate filter size and associated change in sediment trapping to maximize reductions in sediment and nutrient loss.

In summary, environmental problems can only be solved holistically by capturing the interactions among social, economic and hydrologic systems. Stated differently, non-point source pollution problems are complex in nature and comprehensive solutions cannot be achieved based on a single-discipline approach. Integrated scientific approaches are required to satisfy multi-stake holders including watershed planners and landowners. Planning alternatives to reduce non-point source pollution problems such as replacement of agricultural crop lands by conservation crops and the use of vegetative filter strips is presented in this study. The goal is how to make these alternatives effective from both environmental and economic perspectives while using them as a means to reduce non-point source pollution. Since land allocation over the watershed has an effect on the income and water quality, this study aims at determining methods of allocating land cover types over the watershed and best management practices such that sediment and nutrient load to the streams is reduced with minimum possible impact on the total income generated. The study also evaluates ways for achieving the goal of pollutant reduction with minimum possible government water quality incentive payment on a watershed scale.

Objectives of the Study

Given the stated problem, the overall objective of the study was to develop a methodology that could be used to address the environmental and economic goals of reducing sediment and nutrient load taking the Fort Cobb basin as the example watershed. Specific objectives of the study were to:

1) Construct and demonstrate a Land Use Decision Model (LUDM) using mathematical programming approach that will be used to :
    a. Determine the optimal land allocation that maximizes net returns, subject to sediment and nutrient load constraints.
    b. Determine optimal land allocation over the watershed for efficient utilization of limited government water quality incentives to meet varying target levels of sediment and nutrient load to streams by inducing a shift to either switchgrass or CRP.
    c. Compare the economic and environmental benefits of conversion from row crops to bioenergy crop (switchgrass) production and CRP.
    d. Compare sub-watershed based uniform regulation to whole watershed based non-uniform regulation.
2) Develop simplified procedures to be used on a watershed scale for computing sediment trapping efficiency of vegetative filter strips used in conjunction with field drains.

3) Evaluate/compare the economic and environmental impact of varying size of vegetative filter strips along field drains to optimal land distribution on the watershed (prevention and control approach).

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Chapter II

LITERATURE REVIEW

Agricultural Non-Point Source Problems

In the last quarter century the United States has made progress in reducing water pollution, especially from point sources and hazardous waste sites. However, according to the EPA nearly 40% of surveyed waters remain too polluted for fishing, swimming and other uses (Kerr et al., 2002). Attention has turned from major point source polluters to reducing non-point source pollution. The 1970 Clean Water Act (CWA) and its subsequent amendments in 1987 clearly considered NPS pollution as one of the most serious water quality problems.

The primary water quality problems from agricultural non-point source pollution are sediment and nutrients. It has been estimated that non-point source pollution from agricultural land contributes 64% of total suspended sediment and 76% of total phosphorus (Duda and Johnson, 1985). Since primary point source pollution has been increasingly controlled during the past decade, regulatory attention has shifted toward reducing non-point source pollution associated with agricultural production (Vukina and Pasternak, 1997).

It is estimated that the economic damage to surface water quality caused by sediment and nutrients from agricultural cropland ranges from $2.2 to $7 billion each year in the United States (Lovejoy, et al., 1997). An example of the damage is over-enrichment of nutrients from non-point freshwater sources which stimulates algal and rooted aquatic plant growth, and results in oxygen depletion, fish kills, odor problem and eutrophication.

Controlling agricultural non-point source pollution requires that numerous minor polluters coordinate their actions. Farmers working together in a watershed will need to adopt tillage and cultivation practices that generate less runoff and erosion, and install land use measures such as grass filter strips that capture eroding soil before it can be deposited into waterways.

It has long been known that the costs of soil erosion in the United States are borne disproportionately off-farm (Crosson and Stout, 1983). In other words, erosion has relatively little impact on agricultural production and its costs are manifested mainly in the form of soil erosion downstream. The onsite impact of soil and nutrient loss is masked by increased yield from technical changes and use of fertilizer (Stoecker, 2005). Off-site impacts of land degradation due to soil erosion are often much harder to evaluate, because the off-site benefits provided by land resources are not traded at all (Barbier, 1995). In this literature review, currently used non-point source (NPS) pollution control approaches are presented with significant emphasis on vegetative filter strips. Since watershed hydrology and water quality is an important component of this research, most commonly used watershed models will be reviewed. Finally, since sustainable agricultural production systems require balance between economic and environmental impacts of policies, integrated approaches that combine these interactions are reviewed. Such integrated approaches require integration of biophysical and economic models, therefore, these types of studies are also included in the review.

NPS Pollution Control Approaches

Several approaches to control NPS pollution are possible, either individually or in combination, imposing regulations requiring farmers to adopt conservation practices, subsidizing their cost and appealing to farmers through education. The history of conservation programs in the United States and around the world shows elements of all of these approaches (Kerr et al., 2002). In the United States, programs have focused primarily on helping pay for the cost of conservation practices and paying farmers not to cultivate land that bears a high risk of erosion (Horan and Ribaudo, 1999).

There is widespread acceptance of the proposition that farmers will need financial assistance to adopt soil conservation practices since they will only partially accrue benefits from soil conservation. Horan and Ribaudo (1999) recommend incentive-based approaches as the most efficient way to encourage soil conservation. However, the main question is about how to design programs such that financial assistance will be as cost effective as possible. Cost effectiveness entails achieving the greatest reduction in pollution at a given level of cost, thus identification is needed of areas in the watershed where subsidies should be paid to get the maximum benefits from conservation practices. Current programs select certain eligibility targets for recruiting farmers and for sharing investment costs (Kerr et al., 2002). A common approach is to pay farmers 75% of the cost of approved conservation practices like buffer strips, grass waterways, and stream bank protection.

Beginning in 1936, U.S. Department of Agriculture (USDA) provided cost sharing to farmers on selected conservation practices through the Agricultural Conservation Program (ACP) (Helms, 2003). This program, which was introduced in 1936, offered farmers cost-sharing for land conservation measures. The program evolved over the years and was augmented and ultimately replaced by other programs. Today, several major programs such as the Sodbuster, the Conservation reserve Program (CRP), Environmental Quality Incentive Program (EQIP), Small Watershed Program, and Clean Water Act (CWA) help farmers make conservation investments. Of these, the Conservation Reserve Program (CRP) is the largest environmental program (Allen and Vandever, 2003). The most common types of NPS pollution control approaches are presented in the following section.

Conservation Reserve Program (CRP)

The Conservation Reserve Program (CRP), enacted in the 1985 Farm Bill, removes environmentally sensitive cropland from production. While enrolled in the program, CRP acres must be maintained in conservation uses and not harvested. In return, farmers receive an annual rental payment from the government (Walsh et al., 1996). Under the CRP, producers can bid to enroll highly erodible or environmentally sensitive lands into the reserve during signup periods, retiring it from production for 10 years or longer. Enrollment is limited to 36.4 million acres in total, and to 25% of the crop land in a county (Zinn, 1995). The USDA estimates that the average erosion rate on enrolled acres was reduced from 21 to less than 2 tons per acre, per year. Retiring these lands also expanded wildlife habitat, enhanced water quality and restored soil quality.

The annual value of these benefits has been estimated from less than $1 billion to more than $1.5 billion; some estimates of these benefits exceed annual costs, especially in areas of heavy participation (Zinn, 2001). USDA economists estimate that CRP land generates far more savings than it costs. Kerr et al., (2002) indicated that the CRP program is particularly attractive to farmers because, in addition to paying for 50% of the cost of installing conservation measures, it pays them up to 90% of the annual rental value of land taken out of production.

Environmental Quality Incentives Program (EQIP)

The Environmental Quality Incentives Program (EQIP), reauthorized in the Farm Security and Rural Investment Act of 2002, is a voluntary USDA conservation program for farmers and ranchers to implement soil, water and related natural resource problems on eligible lands. It is the second largest conservation program in the history of U.S. agriculture (Khan, 2003). Land retirement is not involved, but rather conservation farming on working farms is the focus. Farmers are asked to engage in five or ten year contracts involving financial and technical assistance and education. EQIP was introduced with the 1996 Farm Act, updating and bringing under one umbrella a number of previous programs. It was initially funded at $200 million per year for 1997 through 2002, and then the total funding was raised to $325 million in 2001 (Kerr et al., 2002).

EQIP was reauthorized in the Farm Security and Rural Investment Act (FSRIA) of 2002. Since EQIP began in 1997, the USDA has entered into 117,625 contracts and enrolled more than 51.5 million acres into the program (NRCS/USDA, 2004). These efforts have concentrated on improving water quality, conserving both ground and surface water, reducing soil erosion from cropland and forestland and improving rangeland.

Highly Erodible Land Conservation (Sodbuster)

The Sodbuster Program applies to any highly erodible field that was not planted to an annual crop or was designated as set-aside or diverted acreage under government commodity supply programs for at least one of the five crop years. Under the Sodbuster program, established in the 1985 farm bill, producers who cultivate highly erodible land (HEL) are ineligible for most major farm program benefits, including price supports and related payments (Zinn, 2001). The Conservation Compliance and Sodbuster Programs require that producers implement an approved conservation plan on their highly erodible cropland to remain eligible for a wide range of USDA program benefits (Osborn, 1996).

Managed Harvesting of Bio-Energy Crops

To increase biofuel production, the number of acres in bio-energy crop production such as switchgrass needs to be increased. CRP lands are potential lands for biofuel production since the environmental objectives that can be obtained through CRP can also be achieved through energy crop production. In 2003 the USDA began a policy allowing managed haying and grazing of land under the Conservation Reserve Program (CRP). The USDA reduces the CRP payment by 25% on any acres harvested under the program. In addition to the annually recurring managed harvesting option on CRP acres, in 2004 the USDA opened up portions of the CRP land for emergency grazing. When CRP land is available for harvesting or grazing, producers must consider whether it is economical to do so since the producers receive 25% less payment (Diersen, 2004). This new program opens up the use of bio-energy crops such as switchgrass since bio-energy crops can be grown and harvested as a renewable energy source while also providing environmental benefits of improving water quality.

Various organizations have conducted plot and field level experimental studies of economic and environmental impact of growing switchgrass for potential biomass energy feed stocks. The National Audubon Society investigated the impact of displacing annual agricultural crops with perennial biomass crops (Beyea et al., 1992). It was concluded from this study that displacing annual agricultural crops with native perennial biomass crops would help restore natural ecosystem functions in worked landscapes, and help to preserve natural biodiversity. In another on farm study, environmental impacts of rotation of short woody crops, was carried out to quantify sediment production, nutrient runoff, wildlife impact, groundwater impact and soil quality impact (Joslin, 1996). The study concluded that agricultural crops are generally more erodible than tree crops. Nitrate runoff was higher under the agricultural crops and ammonium runoff was higher under the trees.

Another plot level study on the environmental impact of conversion of cropland to biomass production compared treatments of row crop (no-till corn), short rotation woody crop (SRWC) production with sweet gum (Liquidambar styraciflua L.), SRWC with a tall fescue (Festuca eliator L.); and switchgrass (Panicum virgatum L.) as a biomass energy crop (Green et al., 1996). Although switchgrass plots eroded more early in the growing season, erosion was low once it became well established. Nutrient runoff was related to fertilization.

Another field level modeling study explored the feasibility for the Missouri–Iowa–Nebraska–Kansas (MINK) region of the U.S. of converting some agricultural land to the production of switchgrass, a perennial warm season grass, as a biomass energy crop (Brown et al., 2000). The Erosion Productivity Impact Calculator (EPIC) crop growth model was used to simulate production of corn, sorghum, soybean, winter wheat and switchgrass. Precipitation increases resulted in greater runoff from the traditional crops but not from switchgrass due to the crop's increased growth and longer growing season. Simulated soil erosion rates under switchgrass and wheat cultivation were less severe than under corn management. Another farm plot level study was carried out on the feasibility of EPIC to assess long term impacts of switchgrass, cottonwood, sweetgum and sycamore production systems on runoff quality (Choi, 1999). The study showed 31% and 37% less runoff than for no-till corn and no-till cotton plots, respectively. The average magnitudes of predicted and measured TSS discharges from woody plots were small. Twenty-year TSS simulation for woody crops showed no TSS discharges, indicating that TSS discharges from these plots were negligible compared to agricultural production plots. The average magnitudes of predicted and measured NO3-N and T-P losses from woody plots were small compared with agricultural crops. Twenty-year NO3-N simulations showed that woody plots had the lowest NO3-N losses.

Another model study was carried out to develop SWAT model predictions of reductions in sediment yield, surface runoff, nitrate nitrogen in surface runoff and edge-of field erosion associated with switchgrass production on cropland in the Delaware Basin in northeast Kansas, and evaluated switchgrass grass break even prices (Nelson, 2001). The study showed the magnitude of environmental benefits and how switchgrass can compete with other commodities while providing environmental benefits. The predicted reduction in sediment yield, edge-of-field erosion and surface runoff as a result of switchgrass plantings was 99%, 98%, and 55%, respectively. The study also predicted that that magnitude of switchgrass water quality payments ranged from a low of $10.06 per ton ($61.59 per acre) to a high of $24.71 per ton ($52.35 per acre), depending upon the switchgrass yield level and competing cropping rotation. The values are based on break even price analysis for each plot.

Best Management Practices for Controlling NPS Pollution

Best Management Practices (BMPs) are alternatives to using land retirement programs to control delivery of pollutants from agricultural activities to water resources and to prevent impacts to the physical and biological integrity of surface and ground water. BMPs can be grouped into structural and non-structural. In non-structural BMPs there are no physical structures. Non-structural BMPs are designed to limit the amount of pollutants available in the environment that would potentially end up in stormwater runoff. Non-structural BMPs typically lessen the need for the more costly structural BMPs and can be achieved through such things as education, management and development practices. Some examples include ordinances and practices associated with land use and comprehensive site planning. Structural BMPs on the other hand can be thought of as engineering solutions to runoff management. They are used to treat storm runoff at the point of generation, the point of discharge, or at any point along the stormwater "treatment train." Structural BMPs can serve many different functions based on their design. In the following section some of the structural BMPs used to control agricultural non-point source will be discussed. Structural best management practices include small ponds (mostly used in urban areas), silt fences used in construction sites, bio-retention cells, vegetative filter strips, riparian buffer strips and grass waterways. Limitations and capabilities of some of structural BMPs are given in Table 2-1. Vegetative filter strip (VFS) is simple to place on agricultural lands and can be harvested if needed. Thus is included in this study and will be presented in some detail.

Table 2-1. Common BMPs for NPS control, limitations and advantages.

BMP type	Limitations	Capabilities
Ponds	loss of infiltrative capacity low removal of dissolved pollutants possible nuisance (odor, mosquito) frequent maintenance requirement high land use requirement	achieves high levels of particulate pollutant removal an effective runoff control can serve relatively large tributary areas
Bioretention cells	cold climate hinders infiltrative capacity clogging may occur in high sediment load areas	enhance quality of downstream water bodies improves area's landscaping
Grass waterways	inefficient nutrient removal can become mosquito breeding areas. not appropriate for steep topography, very flat slopes. Area limited to a maximum of 5 acres difficult to avoid channelization ineffective in large storms due to high velocity flows	reduction of peak flows lower capital cost promotion of runoff infiltration low land requirements
VFS	sheet flow may be difficult to attain not appropriate for very steep slopes tributary area limited to 5 acres	slows runoff flow removes particulate pollutants and some dissolved pollutants

Bioretention Cells. Bioretention areas function as soil and plant-based filtration devices that remove pollutants through a variety of physical, biological and chemical treatment processes. By intercepting, detaining and infiltrating runoff, bioretention cells reduce the energy of stormwater flows and reduce on-site erosion. They may be designed on-line or off-line from the primary stormwater conveyance system (Yu et al., 1999).

Riparian Buffer Strips. A riparian buffer strip area is unmowed, undisturbed and naturally occurring vegetation that buffers the water body and riparian ecosystem from the impacts of adjacent land uses. Buffer functions include protecting water quality and providing for aquatic and terrestrial habitats. As corridors, riparian areas provide travel and dispersal routes for wildlife and plants and sustain long-term river and stream channel functions, such as lateral channel migration and floodwater dissipation (Agency of Natural Resources, 2005). Concentrated flow, sediment accumulation and buffer zone disturbances can reduce the sediment-trapping ability.

Grass Waterways (GWWs). GWWs are important components of a sound soil and water conservation planning (McVay et al., 2004). They play an important role in improving water quality and preventing channel gully erosion.

Vegetative Filter Strips. Vegetative filter strips (VFS) are areas that are seeded to close growing grasses at locations where runoff water leaves a field to remove sediment, organic material, nutrients and chemicals from the flow. Formation of concentrated flow channels with the VFS can reduce effectiveness (Barfield et al., 1979). VFS are also placed along main water courses, streams, ponds and lakes to protect surface water.

Vegetative filter strips are most effective at removing sediment, nitrogen, phosphorous and pesticides bound to soil particles and through infiltration. Recent research at the University of Nebraska-Lincoln evaluated filter strips using simulated rainfall and runoff on silty clay loam soils with 6% to 7% slopes and land area ratios of 15 acres of cropland to 1 acre of filter. Results indicate a 25-foot wide grass filter strip can reduce off-site movement of total nitrogen and atrazine by 70% and total phosphorous by 85%. The reduction in the amounts of herbicide and nitrogen was the result of increased infiltration within the filter strip. Total phosphorous reduction was a result of sediment removal (Franti, 1997).

Studies were conducted in Kentucky on the effectiveness of natural riparian grass buffer strips in removing sediment, atrazine, nitrogen and phosphorus from surface runoff (Barfield et al., 1998) in a karst watershed. No-till and conventional-tillage erosion plots served as the sediment and chemical source area. Runoff from the plots was directed onto 4.57, 9.14, and 13.72 m filter strips where the inflow and outflow concentrations and flow rates were measured. Trapping percentages for sediment and chemicals typically ranged above 90%. An evaluation was made of the distribution of trapped chemicals among infiltrated mass and mass stored in the surface layer. The analysis showed that most of the chemicals were trapped by infiltration into the soil matrix and that trapping efficiency increased with filter strip length and with fraction of water infiltrated.

Barfield et al. (1979) reported that grass filter strips have high sediment trapping efficiencies as long as the flow is shallow and uniform and the filter is not submerged. Researchers (Dillaha, et al., 1989) have found that the filter length controls sediment trapping up to an effective maximum length value, thereafter, additional length does not significantly improve filter performance. VFS performance is inversely related to slope for several reasons. Velocity increases with increasing slope, causing a decrease in residence time within the VFS and a corresponding decrease in the opportunity for sediment to settle out (Hayes et al., 1984). Topography should be relatively flat to maintain sheet flow conditions. Secondly, an increase in slope increases the bed load transport capacity of sediment in the filter, increasing the distance over which bed load is transported into the filter. Finally, the increase in slope results in increased shear force within the concentrated flow areas, causing an increased propensity for erosion and possible VFS failure.

Topography should be relatively flat to maintain sheet flow conditions. When filter strips are used on steep or unstable slopes, the formation of rills and gullies can disrupt sheet flow. As a result the Washington State Department of Transportation (1995) states that filter strips will not function at all on slopes greater than 15% and may have reduced effectiveness on slopes between 6% and 15%. It was further recommended that performance is best with longitudinal grades of 5% or less to maintain uniform sheet flow conditions. Conversely, experimental results by Barfield and Hayes (1988) have shown that VFS designs have been successful in steeper slopes ranging from 15% to 20%.

Several modeling efforts have been undertaken to simulate VFS efficiency in removing pollutants from surface waters. Researchers at the University of Kentucky (Barfield et al., 1979; Hayes et al., 1979; Tollner et al., 1976) developed and tested a model (GRASSF) for filtration of suspended solids by artificial and real grass media. The model is based on the hydraulics of flow, and transport and deposition profiles of sediment in laboratory and field conditions. This physically based model takes into account a number of important field parameters that affect sediment transport and deposition through the filter (sediment type, concentration and particle size distribution, vegetation type and density, slope and length of the filter and infiltration rate).

The model was modified and incorporated into SEDIMOT II and SEDIMOT III, a hydrology and sedimentology watershed model. A modification of the model has also been incorporated into Integrated Design and Evaluation Assessment of Loads (IDEAL) model. GRASSF is an event-based model developed for designing vegetative filter strips with respect to sediment removal (Barfield et al., 1979; Hayes et al., 1984, Hayes and Hairston, 1983). The model was evaluated using data from experimental field plots for multiple storm events and predictions were in good agreement with observed sediment discharge values (Hayes and Hairston, 1983).

Inamdar (1993) developed a model that could predict sediment trapping in natural grass filters where flows have become channalized. The channel network was decided stochastically since occurrence of channels in the filter was random. Channel densities, channel flows and channel shapes were variables selected to represent channel network. Probability density functions for the variables were determined from data and by fitting standard distributions to the data. Deposition/detachment in each channel was modeled using physically based fundamental methods. Both these approaches were combined to determine the expected trapping for a given filter length subject to a known storm event. Model evaluation was done for selected values of Manning's n to give predicted filter trapping efficiencies with in 2% of the observed, indicating model validity. Another modification of GRASSF model for VFS trapping efficiency is VFSMOD (Muñoz and Parsons, 2004). VFSMOD is a computer simulation model created to study hydrology and sediment transport through VFS. The sediment deposition and filtration is modeled using an implementation of the University of Kentucky grass filtration model, GRASSF.

The model is targeted at studying VFS performance on an event by event basis and uses Green Ampt approach to estimate infiltration.

The Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS) model can also be used to evaluate the trapping of sediment by grass filter strips from overland and concentrated flow (Williams et al., 1988) and from deposition where the upper edge of a vegetative filter strip has redirected runoff from overland to concentrated flow. If grass filter strips are so narrow that the strips completely fill with deposited sediment, CREAMS overestimates the trapping of sediment because the model does not account for sediment deposited in the grass strip. Another weakness is that it does not account for vegetation density.

The event based Agricultural Non-point Source Pollution Model (AGNPS) is an example of another approach to estimate trapping in VFS by using hydrologic calculations. AGNPS was used to determine locations of vegetative buffer strip effectiveness on reducing sediment load within the East Bad Creek (EBC) watershed, a 690 ha agricultural watershed located in mid Michigan. To simulate a buffer strip within AGNPS, four input parameters were manipulated on the streamside cells: the CN, C-factor, n value and surface condition constant (SC). Each land cover class was assigned a value for the SCS curve number (CN), crop management factor (C), overland Manning's coefficient (n) value and surface condition constant (SC) based on the digitized land use/cover database (Vennix and Northcott, 2004). However this study accounted for reduction in sediment generation on streamside cells and didn't consider trapping by buffer strips.

An equation for filter strip trapping efficiency is also included in SWAT for sediment and nutrient leaving each Hydrologic Response Unit. Edge-of field filter strips can be defined in a hydrologic response unit (HRU) and sediment, nutrient, pesticide and bacteria loads in surface runoff are reduced as the surface runoff passes through the filter strip (Neitsch et al., 2002). The equation is used to estimate trapping efficiency based on filter strip width alone and does not consider other parameters such as particle size distribution and grass properties and slope which are important for sediment trapping. The buffer strip width can be input for each HRU. To improve the accuracy of predicting sediment trapping, the VFS routine in SWAT needs to be improved by incorporating other parameters affecting trapping efficiency in VFS. The VFS algorithms in SEDIMOT II and III, single storm simulations, are based on GRASSF discussed earlier. To be included in continuous simulation and watershed models such as SWAT, it is important that VFS algorithms be simplified, yet have accuracy approaching that of GRASSF.

Targeting in Conservation Programs

Targeting applies to a variety of payment practices. The common element among these schemes is that not all farmers or ranchers necessarily receive the same payment for a given practice or action. Instead, some criteria are used to differentiate among the sources. Approaches proposed by the USDA to limit the expenditures associated with the conservation programs include targeting conservation funds to parts of watersheds identified as high priority, enrolling farmers who are willing to participate at the lowest cost and using the Environmental Benefits Index. Since 1996 there has been growing emphasis on improving the targeting of the program by using the Environmental Benefits Index (EBI) to enroll land that maximizes conservation and environmental benefits relative to the government cost of enrollment (USDA, 1997). Similarly, Feng et al. (2004) demonstrated that at the beginning of CRP, when erosion reduction was a major goal of the program, if payments were targeted at land with the highest erodibility indices, the average erodibility index of enrolled land in Iowa would be more than twice as high as that of the actually enrolled land. Additionally, supplementary programs have been developed, namely the continuous CRP to target enrollments of acreage in specific conservation practices in environmentally sensitive areas and the Conservation Reserve Enhancement Program (CREP) to achieve specific environmental objectives (Smith, 2000). Unlike the CRP that considers all cropland to be eligible and enrolls selectively on the basis of the EBI, CREP seeks to target land more specifically by limiting the eligible region to environmentally sensitive areas. Another significant conservation program that has employed various targeting tools is the Environmental Quality Incentives Program (EQIP).

Watershed Hydrologic, Water Quality and Biophysical Modeling

Watershed Hydrologic and Water Quality modeling is an important part of the methodology used in this dissertation, thus is included in the literature review. Models that predict all the components including sediment, runoff, water quality and biomass growth are called biophysical models. This section provides an overview of the general literature on the biophysical models and discusses the model adapted for this study. A summary of model capabilities and limitations is given in Table 2-2. Biophysical models specially used for rural watersheds include Agricultural Non-point Source Pollution (AGNPS), Areal Non-point Source Watershed Environment Response Simulation Model (ANSWERS), Soil and Water Assessment Tool (SWAT), Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS), and Environmental Productivity-Impact Calculator (EPIC).

The U.S. Department of Agriculture (USDA) developed the Agricultural Non-Point Source (AGNPS) pollution model of watershed hydrology in response to the complex problem of managing non-point sources of pollution. AGNPS simulates the behavior of runoff, sediment and nutrient transport from watersheds that have agriculture as their prime use. AGNPS is a distributed parameter, event-based model (Young et al., 1995) that operates on a cell basis. It was developed to evaluate the effect of management decision impacts in agricultural watershed-scale systems and addresses concerns related to the potential impacts of point and non-point source pollution on surface and groundwater quality. It uses the universal soil loss equation to predict erosion.

Using AGNPS as a basis, AnnAGNPS model was later developed as a continuous simulation model. It includes all the features that were in the original AGNPS version plus pesticides, source accounting, settling of sediments due to in-stream impoundments, and utilizes the Revised Universal Soil Loss Equation (RUSLE). AnnAGNPS also has limitations. There are no mass balance calculations tracking inflow and outflow of water. The model considers surface hydrology, stream flow and infiltration, but sub-surface hydrology is ignored. This can be a serious limitation with sandy soils, high water table soils, or soils with other unfavorable characteristics. The model does not allow the input of spatially variable rainfall data. This can be a severe limitation as the size of the watershed increases. Storm event precipitation is considered uniform throughout the watershed. As mentioned in the limitations, this can become a serious problem as the size of the watershed increases (León et al., 2004).

Beasley and Huggins (1980) developed the original ANSWERS (Areal Non-Point Source Watershed Environment Response Simulation) model in the late 1970s (Dillaha et al., 2001). ANSWERS can be used to evaluate the effects of land use, management schemes and conservation practices or structures on the quantity and quality of water from both agricultural and non-agricultural watersheds. The distributed structure of this model allows handling spatial as well as temporal variability of pollution sources and loads. It was initially developed on a storm event basis to enhance the physical description of erosion and sediment transport processes. The program has been used to evaluate management practices for agricultural watersheds and construction sites in Indiana. Recent model revisions include improvements to the nutrient transport and transformation subroutines (Dillaha et al., 2001). Some of the limitations of ANSWERS are: It is not well adapted for large watersheds nor for extremely long simulations due to computational requirements, the nutrient transformations and transport simulation relies on the empirical statistical equations. Thus, it works better for certain land uses and soil types than others, model simulation is time consuming and computationally intensive.

The CREAMS model can simulate pollutant movement on and from a field site, including such constituents as fertilizers (N and P), pesticides and sediment (Knisel, 1980). The effects of various agricultural practices can be assessed by simulation of the potential water, soil, nutrient and pesticide losses in runoff from agricultural fields. The spatial scale of the model is intended to be the size of an agricultural field. The model structure consists of three major components: hydrology, sedimentation and chemistry. The hydrology component estimates the volume and rate of runoff, evapotranspiration, soil moisture content and percolation. In spite of its wide use, limitations of the model became apparent when CREAMS was used for hydrologic simulation of flat topography, sand soils and high water-table watersheds in South Florida. In evaluating the suitability of the model for simulating nutrient yield from Coastal Plain watersheds in South Florida, it was determined that assumptions made in developing the model were not valid for sandy soil prevalent in this region. Conceptual changes were led to the development of the CREAMS-WT version which better represents the low phosphorus buffering capacity of these sandy soils and to better represent the hydrology of flat, sandy, high water- table watersheds (Heatwole et al., 1987).

EPIC is a comprehensive model developed to determine the relationship between soil erosion and soil productivity throughout the United States. It continuously simulates the processes associated with erosion, using a daily time step and readily available inputs. EPIC is capable of computing the effects of management changes on outputs. It is composed of physically and biologically based components for simulating erosion, plant growth, and related processes and economic components for assessing the cost of erosion and for determining optimal management strategies. The EPIC physical and biological components include hydrology, climate simulation, erosion-sedimentation, nutrient cycling, plant growth and tillage.

A detailed description of SWAT is given in (Neitsch et al., 2002). An overview is given here. The Soil and Water Assessment Tool (SWAT) is a river basin, or watershed scale model developed by the USDA-ARS. Developed to predict the impact of land management practices on water, sediment and agricultural chemical yields in large complex watersheds with varying soils, land use and management conditions over long periods of time, SWAT is an operational model that operates on daily time step (Arnold et al., 1998).

The first level of subdivision in SWAT is the sub-basin. Sub-basins possess a geographic position in the watershed and are spatially related to one another. A sub-basin contains at least one HRU, a tributary channel and a main channel or reach. The land area in a sub-basin may be divided into hydrologic response units (HRUs). Hydrologic response units are portions of a sub-basin that possess unique land use/management/soil attributes (Neitsch et al., 2002). The assumption is there is no hydrologic interaction between HRUs in one sub-basin. Loads (runoff with sediment, nutrients, etc. transported by the runoff) from each HRU are calculated separately and then summed together to determine the total loads from the sub-basin. If the interaction of one land use area with another is important, rather than defining those land use areas as HRUs they should be defined as sub-basins. It is only at the sub-basin level that spatial relationships can be specified. The benefit of HRUs is the increase in accuracy they add to the prediction of loads from the sub-basin. One reach or main channel is associated with each sub-basin in a watershed. Loads from the sub-basin enter the channel network of the watershed in the associated reach segment. Outflow from the upstream reach segment(s) will also enter the downstream reach segment.

Using daily rainfall amounts, SWAT simulates surface runoff volumes and peak runoff rates for each HRU using a modification of the SCS curve number method for runoff volume and a modified rational method for peak discharge. The possibility of having small and relatively uniform land units (HRU) can be used to reduce the error due to lumping effects. Another advantage of using SWAT is that it is a continuous simulation model. The initial conditions for each day are determined by the model based on the conditions on the previous day. SWAT utilizes a single plant growth model to simulate all types of land covers, which is a limitation. Annual plants grow from the planting date to the harvest date or until the accumulated heat units equal the potential heat units for the plant. Perennial plants maintain their root systems throughout the year, becoming dormant after frost. The plant growth model is used to assess removal of water and nutrients from the root zone, transpiration and biomass/yield production (Neitsch et al., 2002). Sediment yield is estimated for each HRU with the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1975). While the USLE uses rainfall as an indicator of erosive energy, the MUSLE uses the amount of runoff to simulate erosion and sediment yield. The substitution results in a number of benefits: the prediction accuracy of the model is increased, the need for a delivery ratio is eliminated and single storm estimates of sediment yield can be calculated. The hydrology model supplies estimates of runoff volume and peak runoff rate which, with the sub-basin area, are used to calculate the runoff erosive energy variable. Other factors of the erosion equation are evaluated as described by Wischmeier and Smith (1978).

SWAT tracks the movement and transformation of several forms of nitrogen in the watershed. Nutrients may be introduced to the main channel and transported downstream through surface runoff and lateral subsurface flow. The three major forms of nitrogen in mineral soils are organic nitrogen associated with humus, mineral forms of nitrogen held by soil colloids, and mineral forms of nitrogen in solution. Plant use of nitrogen is estimated using the supply and demand approach. In addition to plant use, nitrate and organic N may be removed from the soil via mass flow of water. Amounts of NO3-N contained in runoff, lateral flow and percolation are estimated as products of the volume of water and the average concentration of nitrate in the layer. If users do not specify the initial nitrogen concentrations, SWAT will initialize initial levels of nitrogen in different pools. Organic N transport with sediment is calculated with a load function developed by McElroy and Nebgen (1976) and modified by Williams and Hann (1972) for application to individual runoff events. The load function estimates the daily organic N runoff loss based on the concentration of organic N in the topsoil layer, the sediment yield and the enrichment ratio. Organic nitrogen levels are assigned based on C: N ratio for humic material. The enrichment ratio is the concentration of organic N in the sediment divided by that in the soil (Neitsch et al., 2002).

The movement and transformation of several forms of phosphorous is simulated by the model. The three major forms of phosphorus in mineral soils are organic phosphorus associated with humus, insoluble forms of mineral phosphorus and plant-available phosphorus in soil solution. The amount of soluble P and organic phosphorus contained in humic substances for all soil layers is defined by the user at the beginning of the simulation. If the user does not specify initial phosphorus concentrations, SWAT will initialize levels of phosphorus in the different pools. The concentration of solution phosphorus in all layers is initially set to 5 mg/kg soil, representative of unmanaged land under native vegetation. A concentration of 25 mg/kg soil in the plow layer is considered representative of cropland. The amount of soluble P removed in runoff is predicted using labile P concentration in the top 10 mm of soil, the runoff volume and a partitioning factor. Phosphorus in soil is mostly associated with the sediment phase. Organic and mineral P attached to soil particles is transported by surface runoff to the main channel. These forms of P are associated with sediment load from the HRU and changes in sediment load are reflected in the load of these forms of P.

SWAT allows the user to define management practices taking place in every HRU. The user may define the beginning and the ending of the growing season; specify timing and amounts of fertilizer, pesticide and irrigation applications as well as timing of tillage operations. In addition to these basic management practices, operations such as grazing, automated fertilizer and water applications, and incorporation of management options for water use are available.

Once SWAT determines the loads of water, sediment, nutrients and pesticides to the main channel, the loads are routed through the stream network of the watershed. The transport of sediment in the channel is controlled by deposition and degradation.

Economics of Agricultural Pollution

The economic literature on agricultural pollution has been developed somewhat later than the literature on the general environmental economics, because of the fact that agriculture was traditionally not seen as a source of pollution from a regulatory stand point. An exception to this is the Concentrated Animal Feeding Operations (CAFOs) that are considered as point sources of pollution, and as such are subject to the Clean Water Act provisions (EPA, 2003). Nonetheless, the general principles of environmental economics were adopted for the analysis of agricultural externalities. Externalities in

Table 2-2. Hydrologic models capabilities and limitation

Model type	AGNPS	AnnAGNPS	ANSWERS	CREAMS	EPIC	SWAT
Purpose/ Capabilities	Simulates runoff, sediment, and nutrient transport primarily from agricultural watersheds Event based model Distributed model Uses USLE	Includes features of AGNPS plus pesticides, settling of sediments due to in-stream impoundments Continuous model Distributed model Uses MUSLE	Simulates runoff, erosion, nutrients and effectiveness of BMPs in reducing sediment and nutrients. Continuous model Physically based Distributed	Simulates nutrients, pesticides, and sediment. Continuous model	Physically Based model for erosion productivity relation, hydrology, nutrients tillage, plant growth economics	Predicts water, sediment and chemical yields Can be used in complex watersheds with varying soils and land uses continues, distributed parameter Hydrologic response units (HRUs) allows SWAT to account for the diversity within a sub-basin.
Model Limitations	No day to day tracking of sediment attached chemicals deposited in stream reaches Considers only surface water including runoff, stream flow, and infiltration but not subsurface flow Areal extent limited by the assumption of spatially uniform distributed rainfall	Same limitations as AGNPS but it is continuous and uses MUSLE instead of USLE	Not good for large watershed and long simulations Nutrient transformations and transport relies on the empirical statistical equations. Doesn't work equally good for all land uses and soil types	Applicable to field size, homogeneous areas	Applicable to small and homogeneous areas	HRUs may not be spatially contingent No interaction between HRUs Uses unvalidated assumptions for in-stream-processes Stream process algorithms are poor

production are the economic effects of one's activity on the other, for instance the agricultural activities by the people from upstream affecting people down stream especially when the externalities are not internalized (Hartwick and Olewiler, 1998).

Economic theory and applied studies show that when there are different non-point sources of pollution in a watershed, opportunities for tradeoffs in abatement between the two different sources exist (Hartwick and Olewiler, 1998). There is an economically optimal, least-cost allocation of abatement between sources for any given level of pollutant emissions. This optimal abatement corresponds to the point where the marginal abatement costs at one of the sources is just equal to the marginal abatement cost of another source of pollution. Stated differently, the optimal abatement for one of the sources is where the cost of removing another unit of pollution from the one of the sources is equal to the cost of removing another unit of pollution from the second source. Factors affecting sediment and nutrient load such as topography and soil conditions vary on the watershed. To be cost effective in controlling pollutants, regulations should vary depending on these factors. Land units with lower cost of pollution abatement need to do much of the abatement.

If, however, uniform regulations are imposed, Hartwick and Olewiler (1998) indicate that a producer on a location with the higher cost of abatement has to abate an amount equal to another producer with a lower cost of abatement. This is not a cost effective approach to pollution control. The cost effective policy is the one that equates marginal costs across all the sources (non-uniform approach). Cost effective policies need to be designed and implemented. If standards are set without considering costs and benefits, it is not possible to achieve the desired goals at least cost. At economic optimum, marginal costs across all polluters need to be equal and marginal abatement cost needs to be equal to the marginal damage cost. Integrated economic and biophysical models are useful tools to achieve this.

Integrated Economic and Biophysical Modeling

A key component in achieving more sustainable agricultural production systems is the capability to assess the impacts of changes in policy or technology on land use and on the economic and environmental consequences of farmers' related production decisions (Stoorvogel and Antle, 1999).

The economic analysis of surface water pollution has been conducted to some extent on the watershed level by using a combination of economic and biophysical modeling. The integration of economic models with a biophysical simulation model is suitable for conducting watershed level studies of agricultural pollution since the processes that need to be modeled are both bio-physical (biomass yield, runoff, sediment and nutrient load) and economic (returns and costs).

An example of an integrated biophysical and economic study was carried out by (Ancev, 2003). In this study a methodology was developed that could be used to address the economics of phosphorous pollution in the Eucha-Spavinaw Watershed in North East Oklahoma, to determine the socially optimal level of phosphorous abatement and determine cost effective policies to reduce phosphorous load. The SWAT model and a linear programming model were used to determine a socially optimal level of phosphorous load. One result was an indication that soil test phosphorous (STP) criterion is not an effective policy to reduce phosphorous load.

In another study an integrated framework that combines economic, environmental and GIS modeling was used to evaluate the opportunity costs, in terms of forgone benefits from crop production, and the sediment abatement benefits from land enrolled in the Conservation Reserve Enhancement Program in the lower Sangamon Watershed in Illinois (Khanna et al., 2003). The results showed that the program has been successful in achieving a 20% sediment abatement goal in the watershed but that its costs could be lowered without sacrificing effectiveness if the program could be targeted to a narrow buffer along the streams and tributaries of the Illinois River. This would require, as pointed out by Khanna et al. (2003), the design of a parcel-specific land retirement instrument that would target parcels that are more sloping, closer to water bodies and have lower quasi-rents.

In another study, a Watershed Management Decision Support System was developed and used to evaluate the economic and environmental consequences of alternative land use/management practices. The modeling system consisted of three components: GIS, an economic model and environmental simulation model. The model presents tabular and spatial results that were then viewed side-by-side for comparison (Fulcher et al., 1997). The model presents scenarios of input and output but not an optimized output. Model components include a cost estimator and the Agricultural Non-Point-Source (AGNPS) pollution model for simulating sediment, runoff and nutrient transport from agricultural watersheds. Output scenarios are generated based on input scenarios.

Studies have also been undertaken to determine the minimum incentives needed to induce farmer's participation in conservation programs. The minimum incentive rates were defined as the farmer's actual costs when switching from base scenario to conservation practices. The common feature of these studies is that they emphasize the incentive required to induce land owner's participation into a conservation program to achieve a fixed acreage goal, rather than design a policy instrument based on environmental benefit contribution of land parcels (Khanna, 2003).

Developing a Land Use Decision Model (LUDM)

For this study a LUDM was developed that integrates both the environmental and economic aspect of NPS pollution. When making land use decisions, it is desirable not to make the decision based on subjective assessment of watershed features as only a few physical criteria such as slope, soil characteristics or sediment yield alone will not yield an optimal solution to the problem as pointed out earlier. Real world decision problems in management and engineering often involve multiple, potentially conflicting objectives with highly non-linear responses (Eschenaer et al., 1990). The scope of environmental management is to develop a procedure to reach, as much as possible, an acceptable balance between economic benefits and resulting environmental quality. Such a balance is defined in terms of established criteria and goals. Optimization problems involve objective functions, decision variables and constraints. In the optimization process the decision variables are altered to satisfy any given constraints and to find extreme values of the objective functions (Pike, 1986). The optimization process begins with a set of independent variables or parameters and often includes conditions or restrictions (constraints) that define the acceptable values of the variables and a measure of goodness termed as objective function (Gill et al., 1981). The solution is a set of allowed values of the variables for which the objective function assumes an optimal value.

Watershed managers, farmers and other resource users must respond to policy initiatives. Using a LUDM goal for each of the constraints e.g. sediment yield and nutrient load in the receiving water bodies can be set and land use can be selected to maximize income from the watershed while meeting the constraints. A framework of the general structure of a LUDM is given in Figure 2-1. The structural components of the model include a load model, an optimization model and farm income model. The output from load model and farm return data from the farm income model are input into the optimization model which is used to make land use decision that satisfies constraints while achieving economic goals.

There is a growing consensus that an effective way to control non-point source pollution and enhance the long-term sustainability of agriculture and rural communities is through locally-based planning and management at the watershed scale (Fulcher et al., 1997). According to Kneese (1989), the study of resource economics has required and motivated researchers to reach out beyond their own disciplines and to integrate ideas from other fields. Gottfried (1992) asserts that few economists have addressed the interrelated nature of ecological goods and services, that is, the relationship among spatial units. Ecological Economics is an emerging branch of applied economics that deals with studying ecosystems as integral components of the landscape. Some examples of land use decision models are given in Table 2-3.

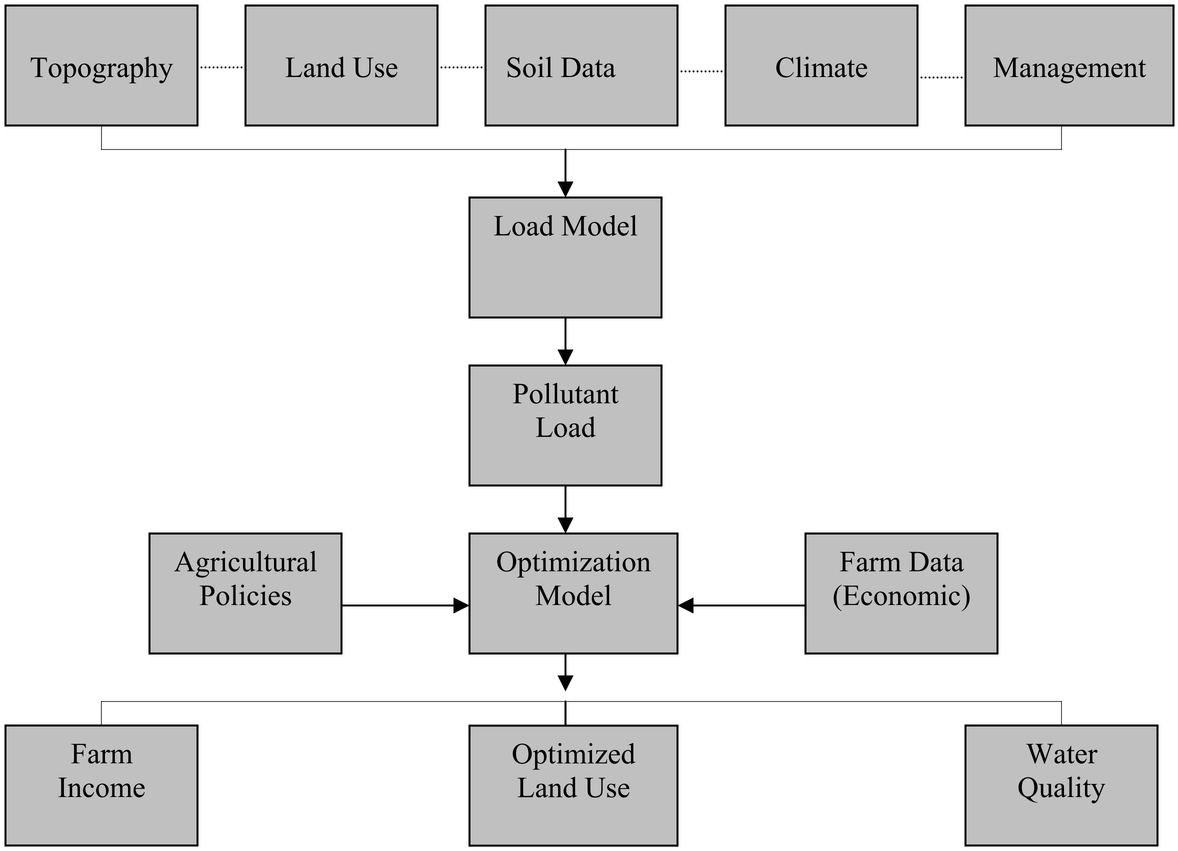

Figure 1. LUDM framework and components.

Mathematical System Programming

A component of the LUDM is a mathematical system programming model. The subject of system programming has received wide spread attention and is avdvanced discipline. As such sophisticated modeling tools are now available. Several modeling languages are available such as MATLAB, Advanced Interactive Mathematical Modeling Software (AIMMS), and a Modeling Language for Mathematical Programming (AMPL), Linear, Nonlinear and Integer programming solver with Mathematical Modeling Language (LINGO) etc. Each of these languages contains a variety of solvers. One such

Table 2-3 Examples of land use decision models.

Model Name	Model Type	Components	What It Explains	Other Variables	Comments
NELUP	General system framework, Economic component uses a linear planning model	Regional agricultural economic model of land use at catchment levels Hydrologic model Ecological model	Explains patterns of agricultural and forestry land use under different scenarios	Soil characteristics, meteorological data land cover	Uses land cover to link market forces, hydrology and ecology in a biophysical model of land use Uses most publicly available data
NELUP-Extension	Linear planning model at farm level	Sub models Lowland and mainly arable Lowland and mainly grazing livestock	Maximizes income Profit is the dependent variable	Level of farm activity Gross Margin per unit of farm activity Fixed resources represents as physical constraints	Detailed farm level model Farmers shown as rational profit maximizing beings but also includes the impact of off farm income
FASOM (Forest and Agriculture Sector Optimization Model)	Dynamic, non-linear, mathematical programming model	Sub models Forest sector transition timber supply model Agricultural sector that is optimized with forest sector sub model	Allocation of land in the forest and agricultural sectors Maximizes discounted economic welfare	Forest sector Variables Demand functions for forest products , Timberland area, age-class dynamics, costs Agricultural sector variables Water, grazing, labor, agricultural demand	Incorporates both agriculture and forest land uses
SDN	Spatial dynamic model	Several subroutines for different tasks	Predicts sites used for shifting cultivation in terms of topography and proximity to population centers	Site productivity, ease of clearing, erosion hazard, site proximity to population centers	Tries to mimic expansion of cultivation over time

Source : Agrawal et al., 2002

a tool is the General Algebraic Modeling System (GAMS) specifically designed for modeling linear, nonlinear and mixed integer optimization problems (Dellink et al., 2001). It consists of a software package including a language compiler and a number of integrated solvers used to solve systems of linear, nonlinear and mixed integer optimization problems and get an optimum solution subject to constraints. GAMS contains different solvers for different purposes.

Various kinds of models can be written down as a system of equations including systems analysis and non-linear optimization modeling (Dellink, 2001). The model can be used to handle environmental economics modeling by writing a standard economic model and then add equations for emission, abatement and economic damages from pollution. It is also possible to write an environmental model using GAMS without economics in it.

The first step in GAMS modeling is to define the problem and write an input file. The general structure of an input GAMS file contains parameters, variables and equations. Parameters are exogenous coefficients that are not determined within the model but which need to be provided to the model as fixed values. Variables are values that are determined endogenously within the model and values which cannot be calculated beforehand. The values of the variables are determined by solving the equations. Equations need to be declared first before writing them down. The core of the model is given by the equations that have to be solved. More details on the use of GAMS is given in chapter III.

Summary and Conclusion

The targeting process is continually being improved to achieve maximum possible environmental benefits from conservation programs and to reduce conservation expenses. Targeting highly erodible lands (HEL) has usually been based on USLE soil loss estimation. It is desirable to develop an analytical framework to determine a cost effective targeting pattern for achieving an off-site sediment yield goal instead of on-site soil erosion goal. In such a framework the contribution of each land parcel to off-site sediment load is needed. An analytical framework for making the calculations will need a biophysical model. SWAT is a model that makes these calculations. It uses the MUSLE equations for estimating sediment load to the streams, thus takes delivery ratio into account and could help to make the targeting process on off-site yield based rather than on-site gross erosion. SWAT was chosen among other models because it is a continuous simulation model and uses the smallest homogenous units (HRUs). SWAT also estimates crop yield from each HRU. Continuous models improve accuracy because they keep track of moisture and nutrients and therefore determine the initial conditions. The use of HRUs makes the model relatively distributed and reduces lumping errors.

Watershed based land use planning in economically and environmentally optimal manner which has been undertaken in this study is an improvement over previous targeting approaches used in CRP. Targeting should not only consider sediment and offsite nutrient contribution of each land parcel but also the opportunity cost of converting each land unit which depends on the productivity of each parcel. This has been taken into account in Chapter III using the LUDM which includes the economic and environmental benefits of putting a given land unit in to conservation program. The

GAMS model was used in chapter III to build the LUDM model because GAMS can handle large, complex, linear, nonlinear and mixed integer optimization problems. It also easy to create closely related constrains in one statement using GAMS.

It was discussed that managed harvesting of bio-energy is an alternative that needs to be evaluated for its economic and environmental benefits over the CRP program. CRP lands are potential lands for bio-fuel production since the environmental benefits of CRP may also be obtained through energy crop production. In addition to finding out the cost effective approach to implementing conservation measures, it was found desirable to compare the effectiveness of producing bioenergy crop (switchgrass) to the current CRP program to predict the potentials of replacing CRP lands by bio-energy (Chapter III).

Land retirement through enrollment in programs such as CRP has been an important policy tool to achieve conservation and protection of water quality. Land retirement is basically a prevention option which limits the generation of sediment and nutrients. When prevention is not feasible, one must look into ways of removing pollutants using BMPs such as ponds, silt fences, bio-retention cells, vegetative filter strips, riparian buffer strips and grass waterways to improve water quality. The major problems of using ponds on farms include potential for flooding after runoff event, inconvenience for working around with farm equipment which is also a limitation in implementing GWWs and unlike VFS part of the farm land will be put out of production. The advantage of VFS is that the sediment and nutrient doesn't leave the field. Hence, VFS is thought to be more appropriate on-site sediment and nutrient control from small fields. However, the problem encountered in doing this comparison was that the hydrologic models currently used do not have a good routine to evaluate the effect of

VFS on sediment trapping. It was found necessary to simplify the existing VFS models to simulate the effect of VFS on sediment trapping. Hence, a simplified procedure was developed to compute sediment trapping efficiency of vegetative filter strips based on the Kentucky filter strip model, GRASSF (Chapter IV). The GRASSF procedure was chosen because it is the most comprehensive currently available model that considers the effect of both flow and sediment properties such as flow depth, velocity, sediment particle size distribution and width of filter strip on sediment trapping.

Both targeting and replacing sensitive area in the watershed and putting them into conservation crops such as CRP or switchgrass to reduce pollutant generation and using vegetative filter strips along field drains to remove agricultural pollutants from runoff are alternatives that need to be evaluated for their economic and environmental benefits. In Chapter V the effectiveness of the two alternatives have been compared.

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Chapter III

MODELING ECONOMIC AND ENVIRONMENTAL IMPACTS OF LAND USE CHANGE IN THE FORT COBB BASIN

Abstract

Watershed management plans must reflect the economic interests of landowners. Row crops such as peanuts grown on the Fort Cobb basin generate more income to the farmers; however, there are concerns about excessive sediment and nutrient load to the streams. The Conservation Reserve Program (CRP) and bioenergy crop (switchgrass) were alternatives considered as replacement for row crops on parts of the watershed to reduce sediment and nutrient load to the streams. A Land Use Decision Model (LUDM) was written for this analysis using General Algebraic Modeling System (GAMS) to make land use decisions. A biophysical model, the Soil and Water Assessment Tool (SWAT), was used to determine sediment and nutrient load. The outputs from SWAT and economic data were used to construct the LUDM input data base. Crops and tillage methods analyzed were switchgrass, conventional and minimum tillage wheat, peanuts, grain sorghum and CRP lands. Two approaches were used in the decision making process: 1) land use decision-making using income maximization subject to defined sediment and nutrient load and 2) incentive minimization subject to sediment and nutrient load. Using an income maximization approach, a non-uniform sediment and nutrient

reduction goal across Hydrologic Response Units (HRUs) was compared with a sediment reduction goal that is uniform across all HRUs. The predicted reduction in sediment yield as a result of replacement of minimum tillage wheat by switchgrass was 95% and the predicted reduction for replacement of other crops and tillage methods such as conventional tillage wheat, grain sorghum and peanuts was more than 98%. The predicted reduction in total P load varied from 80% for minimum tillage wheat to 95% for peanuts. The reduction for total N load was slightly lower than sediment and phosphorous in the range of 65% to 90% for minimum tillage wheat and peanuts respectively. The analysis further indicates that: 1) the loss in income for the same amount of load reduction, as a result of replacement of peanuts by switchgrass is less than it is for replacement by CRP, 2) the incentive required per ton of sediment or nutrient reduced as a result of replacement of croplands by CRP and minimum tillage wheat is higher than the payment required for replacement by switchgrass, and 3) with incentive payments lower than required for CRP, it is possible to have farmers produce and sell switchgrass to generate income and make biomass available for energy purpose and get more water quality benefits.

Introduction

One definition of proper watershed management would be selection of land uses that are appropriate for each sub-watershed and using management practices that maximize profit while minimizing environmental impact. Ultimately, however, watershed management takes place on a landscape in response to the desires of private landowners whose decisions will, in large part, reflect economic criteria such as income maximization (Beaulieu et al., 2000). However, there is increasing pressure to have watershed planners' focus on how resources can be used to maintain or enhance water quality and ecological integrity. Conflicts with land users can occur if the watershed management plans fail to reflect economic uses by landowners. Since these factors interact with each other, it is important to investigate tradeoffs in the decision making process (Yanggen et al., 2002).

One of the mechanisms to induce a voluntary shift to an environmentally friendly land use system is through government incentive payments. The Conservation Reserve Program (CRP) is a voluntary program which offers an annual rental payment for 10 to 15 years and cost share assistance to eligible producers (Walsh et al., 1996). The payments are intended to reduce erosion, improve water quality and enhance wildlife habitat by establishing long-term resource conserving covers on eligible marginal lands. Thus, the CRP, enacted in the 1985 Farm Bill, removes sensitive croplands from production. In CRP program, acres converted must be maintained and not harvested.

An alternative to CRP as a conservation practice is conversion of traditional agricultural crops to biomass energy crops, such as switchgrass on marginal lands. These biomass crops can be grown for energy purposes (ethanol production, direct combustion, etc) and still provide environmental benefits. Due to the greater root and above ground biomass, switchgrass provides more surface area for absorbing nutrients and for trapping sediment (Prairie Resource, 1999). Switchgrass was chosen by the Department of Energy (DOE) as the model herbaceous species for development as a bioenergy feedstock crop. Initial studies of the small-scale plantings of short-rotation woody crops and herbaceous energy crops such as switchgrass indicate that these crops can provide environmental benefits (e.g., soil conservation, increased biodiversity and reduced fertilizer runoff) while improving farm income (Tolbert and Downing, 1995).

Assuming that a ready market is available for energy crops, an appropriate analysis can be made to determine the impacts of bioenergy crop production on environment and water quality. In this report the analysis is made on the Fort Cobb watershed in Oklahoma, one of the most intensive agricultural farming areas of Oklahoma. Peanuts are currently grown on the watershed, along with wheat, alfalfa and numerous other row crops. Due to the high dollar value, much of the farmland has been continuously planted to peanuts for several years. The soils are very coarse and fragile, allowing for high infiltration rates and excessive erosion (Smolen and Lee, 1999). Peanuts can generate more income to the farmers; however their production causes problems for the farmers and the public as a whole by causing pesticide and nutrient runoff, soil erosion and resulting in destabilization of riparian areas. A desirable focus in this study, therefore, was to find out the optimal land use allocation on the watershed to minimize sediment and nutrient yield while maximizing income through proper land selection. The shift to environmentally-friendly production systems such as switchgrass has a long-term impact that increases soil productivity, which is also a concern for the producers. In the analysis, it is assumed that producers will readily shift from the most profitable crop to switchgrass production as long as they receive incentives greater than or equal to the difference in net returns from two production systems.

In order to determine the optimum land use system to maximize income and minimize environmental impacts, a LUDM model is needed. The overall objectives of the studies being conducted were to construct a Land Use Decision Model (LUDM) that can be used to:

1) Determine the optimal land use distribution that maximizes net returns subject to sediment and nutrient load constraints.
2) Determine land distribution over the watershed for efficient utilization of limited government water quality incentives to minimize sediment and nutrient loads by inducing a shift to either switchgrass or CRP.
3) Compare the economic and environmental benefits of conversion to switchgrass production and CRP.
4) Compare Hydrologic Response Unit (HRU) based uniform sediment and nutrient reduction and whole watershed based non-uniform reduction.

Methodology

A Land Use Decision Model, LUDM, was written using GAMS (Dellinket al., 2001) to achieve the listed objectives. In addition to environmental data and hydrologic data, the LUDM requires economic data on costs and benefits. The Soil and Water Assessment Tool (SWAT) was used to generate sediment and nutrient loads and crop yield from each Hydrologic Response Unit (HRU). The output data from SWAT was input into the LUDM, written using GAMS, which is used as an aid in making land use decision. Various kinds of models can be programmed as a system of equations using GAMS. A variety of solvers are used in GAMS for different purposes. Description of SWAT model used in this study, demonstration watershed, SWAT model calibration, SWAT inputs and outputs, environmental and economic data used in LUDM are discussed first, followed by a description of the LUDM models used to achieve the objectives.

SWAT Model

SWAT is a continuous simulation model developed by the USDA-ARS to predict the impact of land management practices on water, sediment and agricultural chemical yields in large watersheds with varying soils, land use, and management conditions over long periods of time. SWAT is an operational model that operates on daily time step (Arnold et al., 1998). A watershed in SWAT is divided into sub-basins, each of which contain at least one hydrologic response unit (HRU), a tributary channel and a main channel. HRUs are portions of a sub-basin that possess unique land use, management, soil attributes and are the smallest homogeneous units (Neitsch et al., 2002).

The benefit of HRU is it allows SWAT to be a distributed model and the possibility of having small and relatively uniform land units reduces the error due to lumping effects. Another advantage of using SWAT is that it is a continuous simulation model. The initial conditions for each day are determined by the model based on the conditions of the previous day. Continuous models improve accuracy as compared to event models because they keep account of the basin moisture condition and determine the initial conditions unlike event models for which the initial conditions are assumed.

Runoff and Peak Discharge. Using daily rainfall amounts, SWAT simulates surface runoff volumes and peak runoff rates for each HRU using a modification of the SCS curve number method and a modified rational method. Two options are also available for estimating the peak runoff rate: the modified rational method and the SCS TR-55 method (Soil Conservation Service, 1986).

Evapotranspiration. The model offers three options for estimating potential ET namely Hargreaves, Priestly-Taylor and Penman-Monteith. The Penman-Monteith method requires solar radiation, air temperature, wind speed and relative humidity as input. If wind speed, relative humidity and solar radiation data are not available, the Hargreaves and Priestly-Taylor methods can be used.

Erosion and Sediment Yield. Erosion and sediment yield are estimated for each HRU using both the Universal Soil Loss Equation (USLE) and the Modified Universal Soil Loss Equation (MUSLE). Sediment yield is estimated for each HRU with the Modified Universal Soil Loss Equation (MUSLE) (Williams and Berndt, 1977).

Nitrogen. SWAT tracks the movement of several forms of nitrogen in the watershed. The Amount of nitrate (NO3-N) contained in runoff, lateral flow and percolation are estimated as products of the volume of water and the average concentration of nitrate in the layer. If users do not specify the initial nitrogen concentrations, SWAT will initialize levels of nitrogen in different pools. Organic N transport with sediment is calculated with a load function developed by McElroy (1976) and modified by Williams and Hann (1972) for application to individual runoff events. This transport function estimates the daily organic N runoff loss based on the concentration of organic N in the top soil layer, sediment yield and enrichment ratio. Organic nitrogen levels are assigned based on C: N ratio for humic material. Enrichment ratio is the concentration of organic N in sediment divided by that in soil (Neitsch et.al, 2002).

Phosphorous. Soluble P and organic phosphorus contained in humic substances for all soil layers is user defined at the beginning of the simulation. If the user does not specify initial phosphorus concentrations, SWAT initializes levels of phosphorus in the different pools. For this study, swat initialized default values were used. The total phosphorous load at the Fort Cobb reservoir was adjusted to previous SWAT predicted load (Storm et al., 2003) at the reservoir using parameters given in Table 3-1. In the study by Storm et al. (2003) total phosphorous and nitrogen were calibrated using water quality data collected throughout the basin. The model was calibrated by comparing individual water quality observations at the same location and time in the model as they were actually taken. Soluble P removed in runoff is predicted using labile P concentration in the top 10 mm of soil, runoff volume and a partitioning factor. Organic and mineral P attached to soil particles is transported by surface runoff to the main channel. These forms of P are associated with sediment load from the HRU and are affected by changes in sediment load.

Management Inputs. User defined management practices can be specified for every HRU. These include beginning and ending of the growing season; timing and amounts of fertilizer applied and pesticide and irrigation applications as well as timing of tillage operations. Grazing operations, automated fertilizer and water applications, and incorporation of management options for water use can also be specified**.**

Plant Growth. SWAT utilizes a single plant growth model to simulate all types of land covers which is a limitation. Annual plants grow from the planting date to the harvest date or until accumulated heat units equal the potential heat units for the plant. Perennial plants maintain their root systems throughout the year, becoming dormant after frost. The plant growth model is used to assess removal of water and nutrients from the root zone, transpiration and biomass/yield production (Neitsch et al., 2002).

Flow, Sediment and Nutrient Routing in Channels. Once SWAT determines the loads to the main channel, the loads are routed through the stream network using a command structure similar to that of HYMO (Williams and Hann, 1972). SWAT tracks mass flow in the channel and models the transformation of chemicals in the stream. Water, sediment, nutrients and organic chemicals are routed in the main channel.

Flow is routed using a variable storage coefficient method developed by Williams (1969) or the Muskingum routing method (Chow et al., 1988). Sediment transport in the channel is controlled by deposition and degradation. Sediment transported from a reach segment is a function of the peak channel velocity. Nutrient transformations in the stream are controlled by the in-stream water quality component of the model, using nutrient routing kinetics adapted from QUAL2E (Brown and Barnwell, 1987). The focus of this study was to optimize land use on each sub-watershed based on total load to the streams. Hence, in stream components of SWAT were turned off and these processes were not simulated.

Demonstration Watershed

Modeling procedures were demonstrated on the Fort Cobb Basin located in Caddo, Washita and Cluster counties in Southwestern Oklahoma. The basin area is 308 square miles which drains into Fort Cobb reservoir. The current land use in the watershed is 41.3% grazing pasture, 50.0% cultivated crops (wheat, peanuts and sorghum), 6.0% forest and 2.6% water.

The basin was subdivided into 154 sub-basins and 1819 HRUs. The GIS data used are the 10m US Geological Survey (USGS) Digital Elevation Model (DEM), 200m Oklahoma Natural Resource Conservation Commission (NRCS) Map Information Assembly and Display System (MIADS) soils data, along with 30m Applied Analysis Incorporated (AAI) land use Data Layer. The DEM and land cover maps are shown in Figure 3-1 and Figure 3-2.

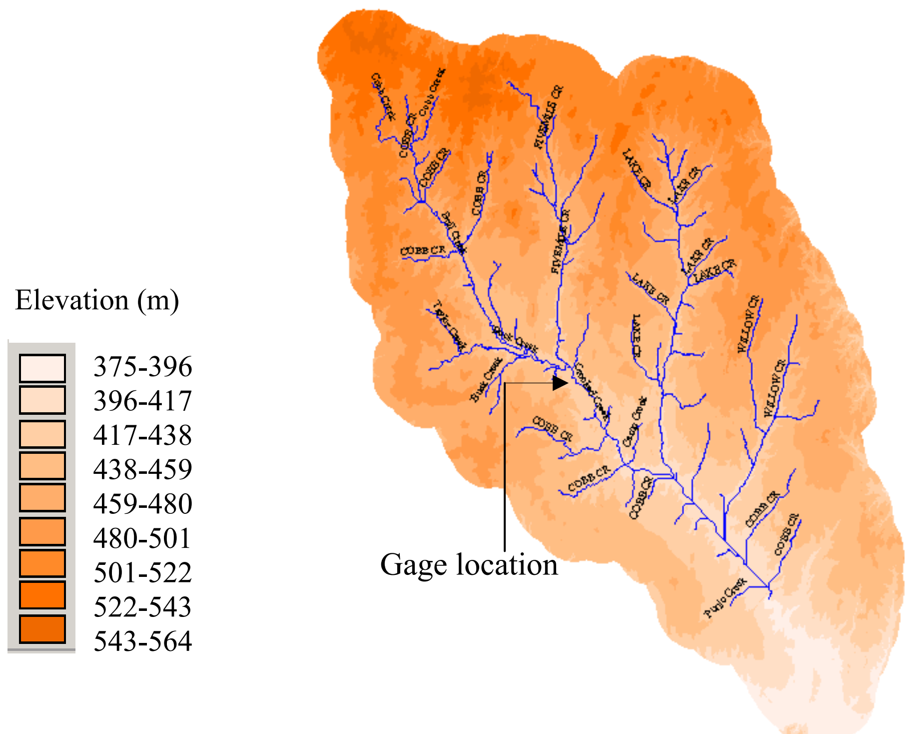


Figure 3-1. USGS digital elevation model (DEM) for Fort Cobb basin**.**

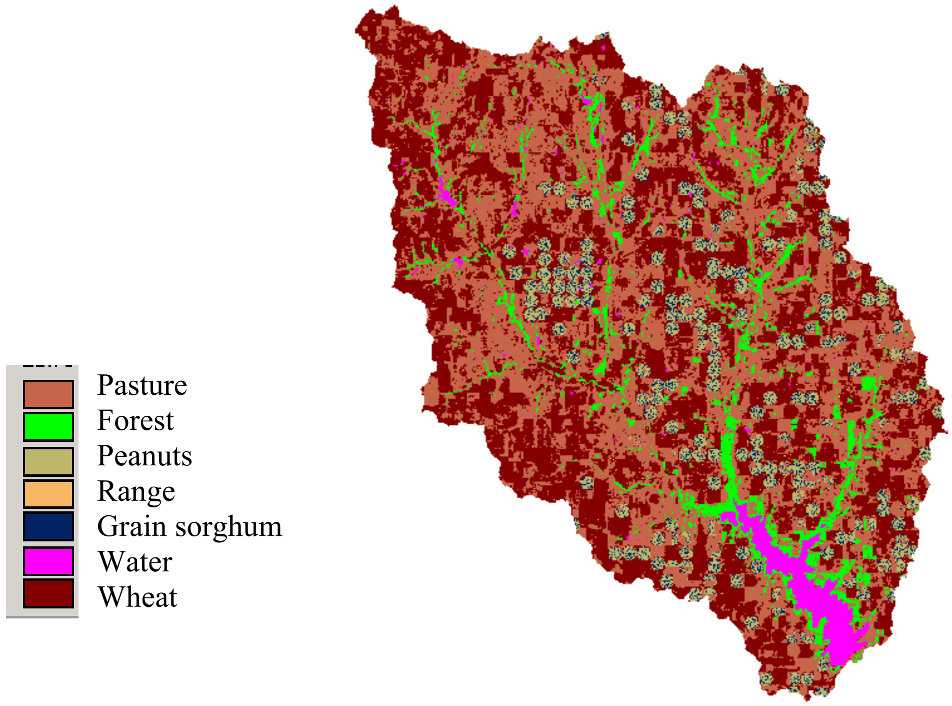


Figure 3-2 Land use map for Fort Cobb basin.

SWAT requires daily values of precipitation, maximum and minimum temperature, solar radiation, relative humidity and wind speed. SWAT can use observed metrological data or simulated data using a database of weather statistics from stations across the US. For this study, the SWAT model was used to generate runoff, sediment, nutrient as well as crop and biomass yields.

Model Calibration and Validation

Calibration is the process by which a model is adjusted to make its predictions agree with observed data. Calibration improves the reliability of the model predictions. Validation is similar to calibration except the model is not modified and tests the model with observed data that are not used in the calibration.

Hydrologic and Flow Calibration. Hydrologic data for the period of Jan 1990 – Dec 2000 from the USGS flow gage at Cobb Creek Near Eakley (USGS 07325800) were used to calibrate SWAT. The same calibration parameters developed for calibrated areas were used for ungaged areas as well. This assumption was made based on similarities between gaged and ungaged areas in the type of soil, slope and rainfall distribution. Ten year average rainfall for ungaged areas is slightly higher (870mm) than gaged areas (850mm). GIS soil data shows that the soils in both gaged and ungaged locations are predominantly silty loam soils. The slopes are similar for ungaged and gaged areas with a range of 2% to 11% especially in the upstream and along the periphery of the watershed. About 10% of the watershed downstream of the gage station is relatively flat with slopes less than 3%.

The total flow data from the gage station were separated using Sliding-Interval Method. In the Sliding Interval Method, the duration of surface runoff is calculated from the empirical relation $N = A^{0.2}$ where N is the number of days after which surface runoff ceases, and A is the drainage area in square miles (Linsley et al., 1982). The interval 2N used for hydrograph separations is the odd integer between 3 and 11 nearest to 2N (Pettyjohn and Roger, 1979). The method determines the lowest discharge in the interval (2N) and takes this minimum value as base flow. Surface flow is computed by subtracting the base flow from total flow.

To adjust surface flow the curve number and soil evaporation compensation factor (ESCO) were used. ESCO is a coefficient that allows SWAT to modify the depth distribution used to meet the soil evaporative demand. As the value for *ESCO* is reduced, the model is able to extract more of the evaporative demand from lower levels, thus altering water balance and reducing surface and base flow.

For groundwater three calibration parameters were used: groundwater "revap" coefficient (GWREVAP), threshold depth of water in the shallow aquifer for revap (REVAPMN), and threshold depth of water in the shallow aquifer required for base flow (GWQMN).

> Water may move from the shallow aquifer into the overlying unsaturated zone. In periods when the material overlying the aquifer is dry, water in the capillary fringe that separates the saturated and unsaturated zones will evaporate and diffuse upward. As GW_REVAP increases movement of water from the shallow aquifer to the root zone is restricted affecting the depth of water in the shallow aquifer.
>
> REVAPMN is the threshold depth of water in the shallow aquifer for movement of water from the shallow aquifer to the unsaturated zone (revap) to occur. Revap is allowed only if the volume of water in the shallow aquifer is equal to or greater than REVAPMN. Increasing REVAPMN effectively increases the depth of water in the shallow aquifer.
>
> GWQMN is threshold depth of water in the shallow aquifer required for base flow to occur (mm H2O). Groundwater flow to the reach is allowed only if the depth of water in the shallow aquifer is equal to or greater than GWQMN. Increasing this threshold value reduces ground water contribution to the total flow.

Table 3-1. Parameter values for SWAT model calibration for Fort Cobb basin**.**

Parameter	Description	Value
GWQMN	Threshold depth of water in shallow aquifer for return flow (mm)	100
REVAPMN	Threshold depth of water in the shallow aquifer for "revap" (mm)	20
RCHRG_DP	Deep aquifer percolation fraction	0.1
ESCO	Soil evaporation compensation factor	0.4
USLEP	Universal Soil Loss Equation conservation practice factor	0.8
NPERCO	Nitrogen percolation coefficient	0.1
PPERCO	Phosphorus percolation coefficient	10
PHOSKD	Phosphorus soil partitioning coefficient	180
PSP	Phosphorus sorption coefficient	0.5

To compare the simulated data to the observed data and to guide the whole calibration process relative error was used. Hydrologic calibration parameters for surface and base flow were adjusted to reduce relative error. Relative error is given by:

$$\text{Relative Error} = 100\,(\text{Observed - Simulated}) \,/\, \text{Observed} \tag{3.1}$$

The results of the flow calibration are shown in Table 3-2 below. Average relative errors were less than 10% for the total flow, base flow and surface flow.

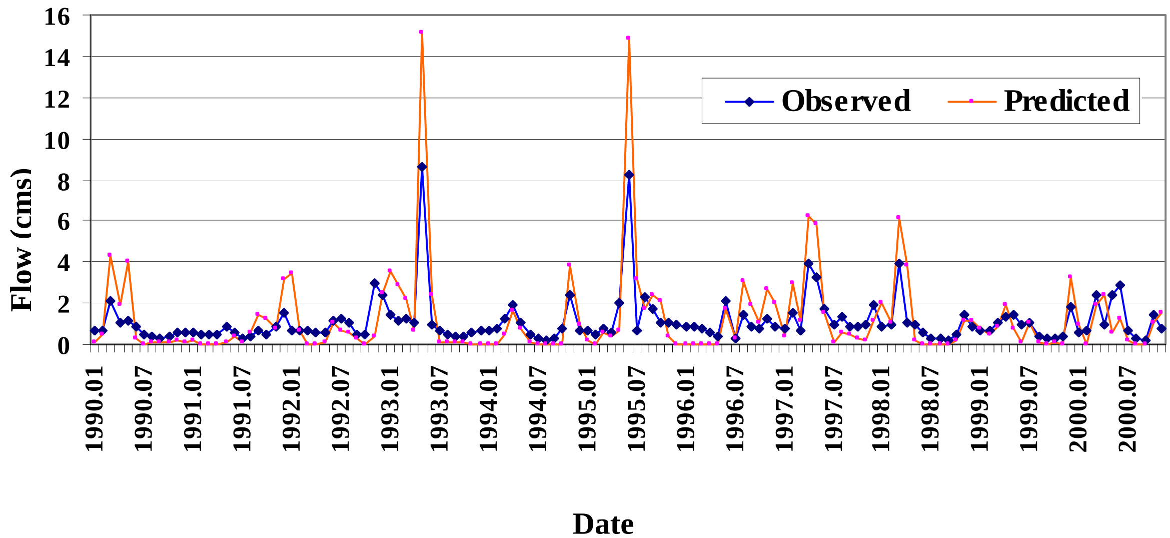
Table 3-2. Annual hydrologic calibration results on Fort Cobb basin**.**

Item	Surface flow (cms)	Base flow (cms)	Total flow (cms)
Observed	0.44	0.46	0.91
Predicted	0.48	0.50	0.98
Relative error (%)	-7.83	-7.26	-7.13

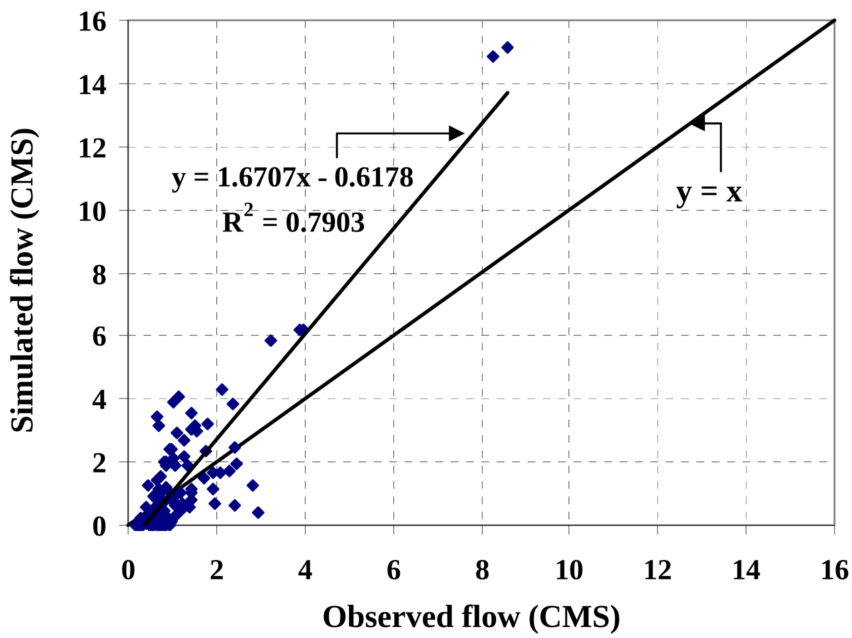
Time series of predicted monthly flows were compared with the observed monthly flows. Time-series of monthly flows shows similar patterns between predicted and observed flows; however, the model in general overpredicts peak flows and underpredicts base flow as shown in Figure 3.3 (A). Effects of ponds were not simulated and this could be the reason for predicted peak flows to be higher than observed as pond storage would tend decrease peak flows. Underpredicting the base flow during the dry periods may not have significant effect especially on sediment and phosphorous load since sediment and phosphorous load are primarily associated with surface flow and erosion. However, over predicting the peak flow during storms has an effect on the amount of sediment and nutrient load. For relative studies like this one, however, the effect is minimized since the effects apply to each land use type.

Scatter plots of average monthly observed and simulated flows are shown in Figure 3.3 (B). Simulated total stream flow matched the observed total stream flow fairly well as shown by the scatter plots, with an R^2 value of 0.79.

Scatter plots of average daily observed and simulated flows are shown in Figure 3.4. The R^2 value for daily observed and simulated flows is 0.60 slightly lower than the values for average monthly flows. The results also show that SWAT overpredicts daily flows. Daily sediment load from SWAT are used in vegetative filter strip evaluation in Chapter V. The vegetative filter strip study in chapter is a comparative study. The study in Chapter V compares placement of vegetative filter strips along field drains to total replacement of parts of the watershed by grass. The error due to overpredicted flows affects both options evaluated and the effect on over all results is negligible.



(A) Time series monthly average observed and SWAT predicted flow.



(B) Scatter plot of monthly average observed and SWAT predicted flow

Figure 3-3. Monthly average observed and SWAT predicted flow.

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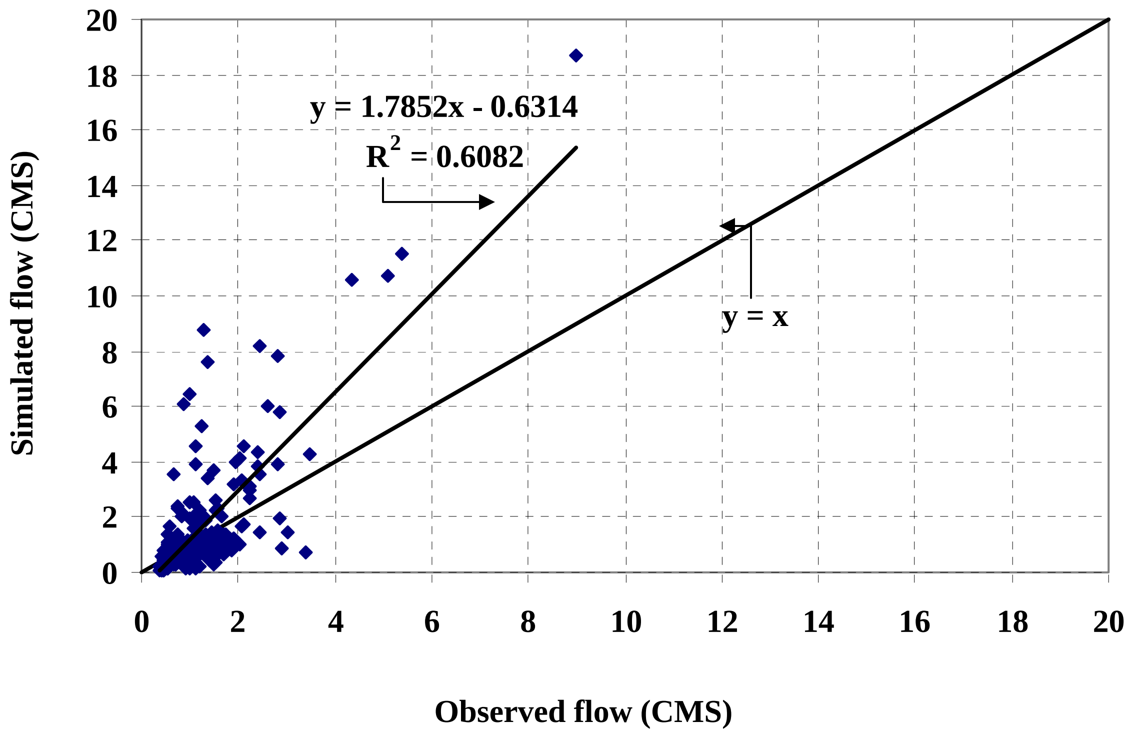

Figure 3-4. Scatter plot of daily average observed and SWAT predicted flow at Cobb Creek Near Eakley.

Nash-Sutcliffe Coefficient of Efficiency (NSE) was used to evaluate the closeness of fit of the observed data with calibrated data. NSE determines the model efficiency as a fraction of the measured stream flow variance that is reproduced by the model:

$$NSE = 1 - \frac{\sum (Q_o - Q_s)^2}{\sum (Q_o - \overline{Q_o})^2} \tag{3.2}$$

where Q_o is the observed stream flow, Q_s is the simulated stream flow and $\overline{Q_o}$ is the observed mean stream flow. The closer the NSE value is to 1.0, the better the estimation of the stream flow by the model. The NSE can be negative when the scatter is large. This does not mean that the prediction is invalid, nor the correlation between observed and predicted values is negative.

The SWAT Model was validated for total flow at the Cobb Creek Near Eakley gage for a validation period of 1980-1989 using the calibration paramteres determined for the period 1990-2000. The Nash-Sutcliff efficiency value for the period of calibration and validation was -0.2 and 0.3 respectively. The model performed better during the validation period than the calibration period.

Nutrient Calibration. The total phosphorous load at Fort Cobb reservoir were adjusted to previous SWAT predicted load (Storm et al., 2003) at the reservoir by varying parameters given in Table 3-1. Justification of the values obtained was also given by comparison to other studies in the literature. Total P and total N load from Storm et al. (2003) study were 102,000 and 734,000 kg/ha respectively. The total P and total N loads obtained by adjusting nutrient parameters in this study were 125,000 and 740,000 kg/ha respectively. The study by Storm et al. (2003) was used to adjust total P and N load because the model was calibrated for nutrients water quality data collected throughout the basin. The report indicates insufficient data was available at any given location to accurately estimate nutrient load. Thus, the model was calibrated by comparing individual water quality observations at the same location and time in the model as they were actually taken. The vast majority of these samples were taken under base flow conditions; thus their utility is limited. Due to limited water quality data with which to calibrate, the utility of nutrient predictions is limited especially when absolute values of nutrient load. However, the analysis can still be useful to carry out a comparative study like this one where the focus is to evaluate the effect of one land use to another in a given location.

Comparisons to other supporting studies were also made as shown in Table 3-3. Effects of precipitation, runoff and management on total phosphorus (TP) loss from three adjacent, row-cropped watersheds in Northeastern Missouri were examined from 1991 to 1997 to understand factors affecting P loss (Udawatta et.al., 2004). Runoff samples from each individual runoff event were analyzed for TP and sediment concentration. The annual TP loss ranged from 0.29 to 3.59 kg/ ha with a mean of 1.36 kg/ha across all the watersheds during the study period.

A study by Mutchler et al. (2002) compared total N and P losses for fertilizer inserted with the planter and broadcasted at planting. Total N lost each year in runoff and sediment from the insert-fertilizer plot averaged 9.4 kg/ha; broadcasting increased this loss to 15.3 kg/ha. Phosphorus loss was 2.4 kg/ha from the insert- fertilizer plot and 3.8 kg/ha from broadcasting. McDowell and McGregor (1984) measured 37.9 kg/ha of N lost in runoff and sediment from conventionally-tilled corn on standard 5% erosion plots.

Also shown in Table 3-3 are results from Reckhow et al (1980) in which a rough estimate of the effects of land use activities on nutrient load to water resources was simulated using export coefficient model. The export coefficient model is the simplest type of pollutant runoff model because all factors that effect pollutant movement are combined into one term, the export coefficient.

As can be seen in Table 3-3, a comaprison to other supporting studies show that the results obtained in this study using SWAT model are similar to previous modeling and experimental studies.

Table 3-3. Literature values for annual nitrogen and phosphorous load**.**

Investigator	Nitrogen (kg/ha)	Phosphorous (kg/ha)
Udawatta et al (2004)		
Range	-	0.29 to 3.59
Mean	-	1.36
Mutchler et al (2002)		
Fertilizer application method		
Insert	9.4	2.4
Broadcast	15.3	3.8
Reckhow et al (1984)		
Land use		
Forest	1.8	0.11
Corn	11.1	2.0
Cotton	10.0	4.3
Soybeans	12.5	4.6
Small grains	5.3	1.5
Pasture	3.1	0.1
Feedlot or Dairy	2900	220
Idle	3.4	0.1
Residential	7.5	1.2
Business	13.8	3.0
Industrial	4.4	3.8
SWAT predicted values (original land cover)	7.09	1.58

Crop yield calibration: Minor adjustment was made to crop yield parameters in SWAT to get crop yield values within the range of National Agricultural Statistical Service (NASS) crop yield data for Caddo County, Oklahoma. NASS 10 year average crop yield for wheat, grain sorghum and peanuts is 2560, 3765 and 3580 kg/ha respectively. The average yields obtained from SWAT model are 2640, 3950 and 3700 kg/ ha respectively. Simulated average yield for switchgrass was 10 t/ha as compared to a measured mean biomass yield from experimental plots at Caddo County for switch grass of 11.4 t/ha (Fuentes and Taliaferro, 2002).

SWAT Model Management and Cover Factor Inputs

Minimum C Factors. Minimum C-factor is used to reflect the effect of cropping and management practices on erosion rates. It is the factor used most often to compare the relative impacts of management options on conservation plans. The minimum C factor indicates how the conservation plan will affect the average annual soil loss and sediment yield.

Minimum C factors were chosen based on crop and tillage type. The minimum C factor for grain sorghum, minimum tillage wheat, conventional tillage wheat and peanuts given in Table 3-4 were obtained from the minimum C factor used in erosion prediction

Table 3-4. Minimum C values for crops.

Crop and tillage type	Minimum C factor	Reference
Peanuts	0.2	NRCS, USDA, http://www.iwr.msu.edu/rusle/doc/factors.pdf
Sorghum	0.18	NRCS, USDA, http://www.iwr.msu.edu/rusle/doc/factors.pdf
Conventional tillage wheat	0.1	NRCS, USDA,2002 http://www.iwr.msu.edu/rusle/doc/factors.pdf
Minimum tillage wheat	0.05	NRCS, USDA, 2002 http://www.iwr.msu.edu/rusle/doc/factors.pdf
Switchgrass	0.005	Wischmeir, W.H., and Smith, D.D. 1978
CRP lands	0.003	Wischmeir, W.H., and Smith, D.D. 1978
Grazed pasture	0.1	Wischmeir, W.H., and Smith, D.D. 1978

in (NRCS-USDA, 2002). The values are close to default minimum C values used in SWAT database. The minimum C factor used for undisturbed Bermudagrass (CRP) is based on the minimum C value recommended for permanent pasture and range land under 95% ground cover (Wischmeier and Smith, 1978). The value is the same as the default value given in SWAT database. The minimum C factor used for harvested switchgrass is slightly higher than the minimum C factor for permanent grass under CRP assuming low residue grass due to harvesting.

It is important to determine the total sediment load from the whole watershed under original land cover condition to identify the relative change in sediment and nutrient load as a result of change in land use. The minimum C factor for other land cover types has been discussed. As stated earlier, 41.3% of original land cover in the watershed is grazed pasture. Since grazed pasture is the dominant land cover, it is necessary to have a good estimate of sediment load from this land use in order to get a good estimate of the total sediment load from the whole watershed under original land cover condition. Hence, the minimum C value used for grazed pasture given in Table 3-3 was estimated based on pasture for an estimated 60% ground cover where a minimum C value of 0.1 was recommended (Wischmeier and Smith, 1978).

To justify the use of minimum C of 0.1 an estimate of percent ground cover for grazed area in the Fort Cobb basin is needed and was developed using a relationship by Shelton el al. (1995). The relationship uses the number of cattle in the basin, number of grazing days and average live weight of cattle and calves to estimate percent ground cover. The number of cattle and calves in the basin is estimated to be 38,700 from NASS statistical data (USDA, 2002). These data are also available in Storm et al (2003). The number of grazing days was assumed to be 365 days and live weight of 500 lb was used. Percent ground cover reduction due to grazing (PR) is calculated as a function of number of animals (NA), average animal weight in pounds (W), number of grazing days (D), number of acres grazed, AC. Percent ground cover (PG) is then calculated from percent ground cover reduction due to grazing. The equation for PR and PG and estimated values for the Fort Cobb grazed land (Shelton et al., 1995) are given by:

$$PR = \frac{0.5WDNA}{1000AC} = \frac{0.5(500)(365)(38700)}{1000(0.41)(308)(640)} \approx 40 \qquad (3.3)$$

$$PG = 100 - 40 = 60 \qquad (3.4)$$

Thus, the use of 0.1 for a minimum C is justified based on calculated PG of 60%. Using a minimum C value of 0.1 for grazed pasture and the values given in Table 3-3 for other crops in the watershed the total sediment load at the Fort Cobb reservoir was 210, 000 Mg/year. This value is close to SWAT predicted value of 245,000 Mg/year from a previous study by Storm et al (2003).

Management Inputs. Table 3-5 shows management inputs used for each crop. The land cover types and tillage practices included in the study were switchgrass, conventional and minimum tillage wheat, grain sorghum, peanuts and CRP lands under bermudagrass. Management inputs for switchgrass are from management guide for the production of switchgrass (Teel et al., 1997). It is recommended that if an acceptable stand was achieved during the seeding year, 90 to 120 lb/ of N is applied annually depending on the amount of rainfall in the area to assure productive yields, i.e. to get as much biomass each year as possible (Teel et al., 1997).

Management practices for biomass production of bioenergy grasses may differ from management for forage. Fertilizer rates and harvest dates vary depending on climatic conditions. Information on optimal harvest periods and N fertilization rates for witchgrass grown as a biomass or bioenergy crop in the Midwest USA is limited.

Table 3-5. Management operations for each crop.

Conventional tillage wheat	Minimum tillage wheat
Harvesting, June 1	Harvesting: June 1
Fert: 120 lb/ac N, 40lb/ac P_2O_5 Sept 20	Fert:120 lb/acre N, 40lb/ac P_2O_5, Sept 20
Disk plowing and Harrowing : Sept 22	Minimum tillage: at planting only Sept 25
Spring tooth harrowing: Sept 24	Switchgrass during establishment year
Planting: Sept 25	Land clearing: March 1
Grain sorghum	Fert:120 lb/acre N, 40lb/ac P_2O_5 Sept 15
Harvesting, Oct 15	Disk plowing and harrowing : April 15
Fertilizer:120lb/ac N, 40 lb/acre P May 27	Planting:April 18, No harvesting during 1st year
Disk plowing and harrowing: May 28	Operation repeated every 10 years
Spring tooth harrowing: May 29	Switchgrass management (other years)
Peanuts	Fert.:120lb/acre N,40lb/acre P_2O_5 April 15
Fertilizer: 30 lb/ac N, 70 lb/ac P_2O_5 April 16	Harvest operation: July 30
Disk plowing and harrowing: April 17	
Spring tooth April 18	
Planting: April 19	
Harvesting: Oct 15	

A study was conducted at Stephenville, TX and 1993 to 1995 at Beeville, TX to determine the yield and stand responses of 'Alamo' switchgrass (Panicum virgatum L.) to N and P fertilization as affected by row spacing (Muir et al., 2001). Biomass production was not influenced by the addition of P. Biomass production responses to N were quadratic. A maximum yield of 22.5 Mg ha^{-1} occurred during 1995 at Stephenville at 168 kg N ha^{-1}. Lodging occurred only at the 224 kg N ha^{-1} rate. Average biomass production at 168 kg N ha^{-1} yr^{-1} was 14.5 and 10.7 Mg ha^{-1} yr^{-1} at Stephenville and Beeville, respectively. Biomass production without applied N tended to decline over the years. In this study, the fertilizer rate used was 120 N lb/ acre. SWAT predictioions showed that the increament in biomass yield after this point is very small.

A study was made by Vogel et al (2002) to determine optimum harvest periods and N rates for biomass production in the region. Established stands of switchgrass at Ames, IA, and Mead, NE, were fertilized at 0, 60, 120, 180, 240, or 300 kg N ha^{-1}. Harvest treatments were two- or one-cut treatments per year, with initial harvest starting in late June or early July (Harvest 1) and continuing at approximately 7-d intervals until the latter part of August (Harvest 7). A final eighth harvest was completed after a killing frost. Regrowth was harvested on previously harvested plots at that time. Averaged over years, optimum biomass yields were obtained when switchgrass was harvested at the maturity stages R3 to R5 and fertilized with 120 N kg/ha (106 lb/acre). Biomass yields with these treatments averaged 10.5 to 11.2 Mg ha^{-1} at Mead and 11.6 to 12.6 Mg ha^{-1} at Ames.

Seasonality of the biomass growth must be considered when determining scheduling of harvest. Alternatives include harvesting the required amount of biomass once per annum during low-moisture, non-growth period (usually late summer/ early fall), or harvesting nearly year round (Thorsell et al., 2004).

SWAT predictions show that delayed harvesting didn't improve crop yield. The increase in crop yield is less than one percent as a result of delaying harvesting by one or two months (from July 30 to August or September 30). However, the sediment yield increased by about five percent when harvesting is delayed. This can be attributed to change in plant density. Hence, the July 30 date used in this study is more appropriate to reduce sediment and nutrient load since there is no significant increase in biomass yield.

SWAT predictions also show that harvesting twice with first harvesting at the end of July and second harvesting mid-August increases total P and sediment yield but reduced nitrogen yield probably due to nitrogen removal with biomass.

Fertilization and management practices for wheat, grain sorghum and peanuts are from Fort Cobb Basin modeling and land cover classification, final report (Storm et. al, 2003). Storm et al obtained their data from OSU recommendations and knowledge of local OSU cooperative Extension Service and Soil Conservation District personnel (primarily Monty Ramming).

SWAT did not automatically change management parameters when changing tillage practices, for instance, from conventional tillage to minimum tillage. It is necessary to manually input the changes in C factor and curve number as necessary to get reasonable results for each tillage practice. In this study, different curve numbers and C factors were used for minimum tillage wheat and conventional tillage wheat.

Land Use Decision Model (LUDM)

When making land use decisions, it is desirable not to make the decision based on subjective assessment of watershed features as only few physical criteria such as slope or soil characteristics or sediment yield alone cannot be used to yield an optimal solution to the problem. Real world decision problems in management and engineering often involve potentially conflicting objectives with highly non-linear responses. The scope of environmental management is to develop a procedure to reach, as much as possible, an acceptable balance between economic benefits and resulting environmental quality.

Using the LUDM discussed in Chapter II, constraints for sediment yield and nutrient load in the receiving water bodies can be set and the land use be optimized until intended goals are achieved. A framework of the general structure of the LUDM and its components was given earlier in Figure 2-1. The output from load model, SWAT along with farm return data from the farm income model are input into the optimization model which is used to make land use decisions that satisfies constraints while achieving economic goals.

Environmental Data for LUDM

The first step in model execution is running SWAT simulations for land cover types included in the study to generate a database for the LUDM model which requires selection of a period of simulation. In this study, the SWAT model was run for 10 years plus a 3 year warm up period. A 10 year simulation period was chosen after comparing the results for the 10 year and 20 years simulation period. For a 10 year period average sediment, total P and total N loads were 2.74 Mg/ha, 1.58 kg/ha and 7.1 kg/ha respectively. The results for 20 year simulation period were 2.65 Mg/ha, 1.5 kg/ha and 6.95 kg/ha respectively. For a 10 year simulation period SWAT predicted percent reduction in sediment yield, total P and total N loads resulting from conversion from row crop to switchgrass were 99.3%, 95% and 91.30% respectively. The corresponding reductions were 99.6%, 99.5% and 91.31% for a 20 year simulation. Sediment and nutrient load obtained with a 10 and 20 year simulation are similar and perecent reductions as a result of land conversion did not change much with change in length of period of simulation. Hence a 10 year period was chosen to reduce simulation time. As cited earlier, the land cover types included in the study are peanuts, grain sorghum, minimum and conventional tillage wheat, switchgrass and bermudagrass for CRP. Also, the watershed has been subdivided into 154 sub-basins and 1819 HRUs. For each run, all the area in the watershed is converted to one land cover type except for areas currently under forest which accounts for only 6.68% of the watershed, and is primarily located on riparian zones. The sediment, nutrient and crop yield data for each HRU under each land cover type is obtained from SWAT simulation. Examples of the data generated from SWAT are given in Tables 3-6 to 3-9. The data are used as database to construct the LUDM using GAMS.

An example of crop yield data from SWAT model is given in Table 3-6. Tables 3-7 to 3-9 are examples of sediment and nutrient data for each crop and corresponding management in each HRU. SWAT predicted average annual basin values for each crop are shown in Table 3-10.

The reduction in sediment and nutrient yield for converting each HRU from row crop to switchgrass can be calculated using the data in Tables 3-7 to 3-9. This reduction expressed as an average percentage is given in Table 3-11. The percent reduction in total N is relatively low compared to total P and sediment load. This can be attributed to the fact that nitrogen is highly mobile compound compared to phosphorous. Hence, integrating conservation crops such as switchgrass into agricultural productions is an alternative approach to increase farm income while addressing environmental concerns. The potentials for grass biomass for conversion to energy products and how these components fit into conservation management plans is also important issues needs to be evaluated along side the environemtal benefits.

Table 3-6. Example crop yield data for each land cover (Kg/ha).

HRU	MT wheat	CT wheat	Grain sorghum	Peanuts	Switchgrass	CRP
1	2,780	2,617	3,262	3,828	11,092	NH
2	2,622	2,602	3,370	3,828	11,166	NH
5	1,730	1,537	3,529	3,551	5,241	NH
6	2,622	2,602	3,370	3,828	11,166	NH
7	2,651	2,599	3,373	3,828	11,166	NH
8	3,237	3,079	4,359	3,808	9,343	NH
11	2,651	2,599	3,373	3,828	11,166	NH
12	2,780	2,617	3,262	3,828	11,092	NH
13	2,523	2,498	3,825	3,652	7,351	NH
14	3,319	3,069	4,013	3,820	11,300	NH
20	3,319	3,069	4,013	3,820	11,300	NH
21	2,916	2,898	3,109	3,820	10,981	NH
22	2,943	3,022	2,782	3,820	11,031	NH
23	2,987	2,787	3,284	3,820	10,877	NH
24	3,316	3,071	4,020	3,820	11,290	NH
32	2,916	2,898	3,109	3,820	10,979	NH
33	3,022	3,047	4,967	3,771	8,728	NH

MT = Minimum Tillage, CT = Conventional Tillage, NH = not harvested

Table 3-7. Example sediment yield data for each land cover (Kg/ha).

HRU	MT wheat	CT wheat	Grain sorghum	Peanuts	Switchgrass	CRP
1	1,619	5,790	11,811	16,904	62	42
2	1,441	3,993	8,288	12,988	82	42
5	16,217	26,089	27,581	35,323	608	168
6	1,475	4,080	8,387	13,225	84	42
7	1,376	3,860	8,093	12,610	79	40
8	2,128	5,844	17,334	21,967	96	79
11	1,517	4,245	8,802	13,810	86	44
12	1,552	5,555	11,369	16,252	57	40
13	6,432	13,096	19,931	30,262	398	141
14	4,015	10,267	20,428	26,474	109	69
20	3,620	9,239	18,337	23,751	99	62
21	1,745	4,522	9,696	16,563	173	42
22	2,434	5,337	11,337	17,576	131	54
23	922	2,866	6,842	9,778	42	27
24	1,552	3,981	7,771	10,064	42	27
32	746	1,915	4,072	6,951	74	17
33	14,013	34,068	64,340	88,022	1,186	388

MT = Minimum Tillage, CT = Conventional Tillage

Table 3-8. Example total P yield data for each land cover (Kg/ha).

HRU	MT wheat	CT wheat	Grain sorghum	Peanuts	Switchgrass	CRP
1	1.36	2.00	3.90	7.26	0.14	0.52
2	1.30	1.55	3.05	6.31	0.20	0.60
5	5.97	5.79	6.03	9.74	1.14	1.62
6	1.32	1.57	3.08	6.39	0.20	0.60
7	1.26	1.51	3.00	6.18	0.19	0.59
8	1.45	1.64	3.96	7.98	0.20	0.96
11	1.35	1.61	3.18	6.57	0.21	0.61
12	1.32	1.95	3.80	7.07	0.14	0.52
13	4.28	4.78	6.47	11.98	1.01	2.00
14	2.68	3.34	6.20	10.26	0.26	0.83
20	2.50	3.11	5.81	9.63	0.24	0.80
21	1.47	1.68	3.54	7.65	0.41	0.46
22	1.92	2.04	4.17	8.39	0.33	0.81
23	0.98	1.39	3.18	5.95	0.15	0.60
24	1.42	1.75	3.36	5.59	0.15	0.65
32	0.83	0.93	1.99	4.37	0.24	0.37
33	6.02	6.05	8.88	17.94	2.10	3.38

MT = Minimum Tillage, CT = Conventional Tillage

Table 3-9. Example total N yield data for each land cover (Kg/ha).

HRU	MT wheat	CT wheat	Grain sorghum	Peanuts	Switchgrass	CRP
1	3.60	9.31	17.00	23.08	0.32	0.64
2	3.28	6.99	12.80	18.89	0.43	0.67
5	11.45	19.60	20.71	21.32	5.87	6.99
6	3.33	7.10	12.92	19.12	0.44	0.68
7	3.19	6.83	12.58	18.53	0.43	0.67
8	3.16	6.31	12.90	14.53	1.80	2.69
11	3.41	7.31	13.32	19.64	0.47	0.71
12	3.50	9.05	16.59	22.53	0.33	0.64
13	9.94	18.23	24.04	27.49	3.21	6.76
14	7.18	15.44	27.18	32.46	0.74	1.39
20	6.70	14.40	25.53	30.54	0.69	1.33
21	3.58	7.09	14.29	21.62	0.83	0.47
22	4.80	9.15	16.62	23.56	0.67	0.97
23	2.44	6.05	13.02	17.16	0.28	0.73
24	3.73	8.11	15.01	18.10	0.36	0.98
32	1.97	3.91	8.14	12.60	0.40	0.30
33	11.09	19.18	23.55	22.69	8.90	12.40

MT = Minimum Tillage, CT = Conventional Tillage

Table 3-10. SWAT predicted watershed annual average sediment and nutrient load for different crops and tillage practices.

Land cover type	Sediment yield (kg/ha)	Total N (kg/ha)	Total P (kg/ha)
Conventional tillage wheat	3,630	4.87	1.23
Minimum tillage wheat	1,480	2.38	1.14
Grain sorghum	6,640	8.03	2.22
Peanuts	9,700	9.30	4.20
Switchgrass	70	0.81	0.21
CRP lands (bermudagrass)	40	1.69	0.78
Original land cover	2,740	7.09	1.58

The percent reduction in sediment and nutrient load for replacement of agricultural crops by switchgrass is given in Table 3-11 showing more than 95% reduction in sediment yield can be achieved by replacing row crops by switchgrass. Phosphorous and nitrogen load are also significantly reduced as a result of replacement by swtchgrass.

Table 3-11. SWAT predicted percent reduction in sediment and nutrient load for replacement by switchgrass.

Land cover type	% reduction in sediment yield	% reduction in total P yield	% reduction in total N yield
Minimum till wheat	95.27	81.58	65.97
Conventional till wheat	98.07	82.93	83.97
Grain sorghum	98.95	90.54	89.91
Peanuts	99.28	95.00	91.29

Economic Data for LUDM

The cost benefit analysis in this study was conducted with input from the Department of Agricultural Economics, Oklahoma State University. The OSU enterprise budget software (Doye, et al., 2001) was used to estimate the cost per acre data required for each land use. The total benefit is determined using the SWAT crop yield output and crop price data. Data for the different cultural practices used in the production of each crop was gathered from many different sources. Switchgrass management data required for calculating switchgrass production costs were obtained from the management guide for switchgrass (Teel et al., 1997). The establishment costs were distributed over 10 years assuming a stand renewal once in 10 years. The machinery costs were calculated using the Machsel program (Kletke and Sestak, 1991). Estimates of expected prices for bioenergy crops vary widely by crop, region and estimation methods, including notably whether transportation costs are included. Walsh et al. (1996) estimated production costs to vary from $22/dry ton to $110/dry ton and transportation costs to range from $5/dry ton to $8/ dry ton for a 25-mile transport distance. On a national scale Oak Ridge National Laboratory estimates of bioenergy supply prices were $30–40/ dry ton at low near term demand rates (McLaughlin, et al., 1999). The analysis in this study is done at a price of $39/ton. The information for the peanuts budgets was obtained from the Agricultural Extension Agent (Nowlin, 2004) in Caddo County. The sorghum information was from the Southwest Oklahoma Agricultural Specialist (Gregory, 2004).

Information for establishing CRP acreages came from the Oklahoma Conservation Practice Job Sheet for Range Planting. Wheat data are from extension data base (Peeper, 2004). Sorghum and wheat prices are from average price received for Oklahoma published in the November and August 1999-2003 issues of Oklahoma Agricultural Prices respectively (USDA, 2003).

The peanut program changed significantly in the Farm Security and Rural Investment Act (FSRIA) 2002. Previously, peanuts were produced using a quota system to support farm incomes whereas under the new law peanut producer's incomes are supported with the same type program as all other commodities. The five-year average price was 29 cents per pound for the Oklahoma Marketing years 1997-2001 before the program change (USDA, 2002). In 2002 after FSRIA took effect the Oklahoma average price was 17 cents per pound (USDA, 2003). Because, the program has changed it was assumed that the prices before FSRIA would not accurately predict the future and only the 2002 price was used in the analysis. Since prices vary each year, the analysis was made on average prices to make the comparison among land uses. The average CRP rental rate of $43 per acre in Caddo County was used based on data from the 26th CRP signup (Agapoff et al., 2003). Costs and prices for each crop are given in Table 3-12.

The government system of decoupled direct and counter cyclical payment designed to allow producers to make production decisions were included in this study. The 2002 Farm Bill authorizes direct and countercyclical payments to enrolled producers. Countercyclical payment rates vary depending on market prices and are issued only when the effective price for a crop is below the target price. Direct payments, DP, are given by:

$$DP = 0.85[DPR]\ [Base\ Yield] \tag{3.5}$$

where DPR is direct payment rate and base yield is crop yield on base year chosen for calculation. Counter cyclical payments, CCP, are computed from:

$$CCP = 0.85[(TP - (LR\ or\ MYA) - DP)][Base\ Yield] \tag{3.6}$$

where TP is target price, LR is loan rate and MYA is marketing year average price. The average total direct and counter cyclical payments (DCP) per hectare of wheat, grain sorghum, and peanuts are given in Table 3-12.

Table 3-12. Revenue, Costs, and Returns for Crop Production ($/ha).

Item	Peanuts	Grain Sorghum	Conventional Tillage Wheat	Minimum Tillage Wheat	CRP	Switch-grass
Fixed Cost/ac	175.50	56.20	63.90	58.60	6.60	69.00
Variable Cost	1084.0	294.51	253.80	285.44	10.10	197.58
Prices/ton	355.72	70.00	91.60	91.60	-	39.00
DCP ($/ha)	409.77	53.24	78.91	78.91	-	-

LUDM Modeling Approaches

Modeling of the impact of land use change requires some method of allocating land use to units within the watershed. In this study, the allocations are made at the Hydrologic Response Unit (HRU) level. Various approaches used to make land use decision within LUDM model are presented in the subsequent sections. In the first approach, the land use decision is made based on achieving maximum net returns subject to environmental constraints, assuming producers will be willing to implement the decision for the greater good of minimizing pollution. The second approach uses water quality incentives to cause landowners switch to switchgrass production or enroll in CRP. A uniform reduction approach in which each land unit in the watershed is subject to reducing sediment load in the same proportion is compared to a non-uniform load reduction approach in which the focus is to reduce the total sediment and nutrient yield at the watershed outlet. Optimal land use distribution for varying level of erosion charges was also investigated.

LUDM: Maximum Net Return Model. In this approach the total income from the watershed is maximized while meeting allowable sediment and nutrient load constraints at the watershed outlet. It is assumed that the grower will make the necessary conversion for the greater good of minimizing the environmental impact. Total loads are evaluated at the watershed outlet. The algebraic expressions are given below.

The objective function for LUDM maximum net return model is given by:

$$Max \sum_{i=1}^{N} \sum_{j=1}^{M} NI_{ij} X_{ij} \tag{3.7}$$

The objective function given by expression (3.7) is the net income from the watershed which is maximized. NI_{ij} is the net income from HRU j for land cover i. X_{ij} is the fraction of land cover type i assigned to HRU j. N is the number of land cover types compared and M is the number of HRUs in the watershed.

The constraints used in LUDM maximum return model are:

$$\sum_{i} X_{ij} \leq 1 \tag{3.8}$$

$$X_{ij} \geq 0 \tag{3.9}$$

$$\sum_{i=1}^{N} \sum_{j=1}^{M} Sed_{ij} X_{ij} \leq Sedyld \tag{3.10}$$

$$\sum_{i=1}^{N} \sum_{j=1}^{M} P_{ij} X_{ij} \leq Pyld \tag{3.11}$$

$$\sum_{i=1}^{N} \sum_{j=1}^{M} N_{ij} X_{ij} \leq Nyld \tag{3.12}$$

where X_{ij} is 1 if i^{th} land cover is selected in the j^{th} HRU and it is zero if the land cover is not selected, N is the number of land cover types compared, NI_{ij} is the net revenue form the i^{th} land cover in the j^{th} HRU, Sed_{ij} represents the sediment yield from HRU j when land cover i is selected, Sedyld is the maximum amount of sediment allowed to leave the watershed, P_{ij} represents the phosphorus yield from HRU j if land cover i is selected, Pyld is the maximum amount of phosphorous allowed to leave the watershed, N_{ij} represents the nitrogen yield from HRU j when land cover i is selected, and Nyld is the maximum amount of nitrogen allowed leave the watershed.

X_{ij} is the decision variable in the model. The constraint function given by (3.8) implies that the sum of fractions of areas assigned to the different land uses in a given HRU should be equal to unity. If for instance, the LUDM model selects switchgrass on HRU1, i.e. if 100% of the area is assigned to switchgrass, the value assigned to switchgrass is 1. Other land covers will be automatically set at zero value since the sum has to be equal to one to satisfy constraint (3.8). The selection depends on maximizing the net return from the watershed and meeting the load requirements. The net return for other land covers will therefore be zero since the product NI_{ij} X_{ij} is zero. GAMS has a provision for linear, non-linear or mixed integer programming. If the model chosen is a mixed integer in cases where the land use planner chooses to have only one land cover type on each land unit or HRU, X_{ij} will be either 0 or 1. For other models such as linear programming or non linear programs any value between 0 and 1 can be assigned. It is also possible that the variable X_{ij} is declared as binary so X_{ij} is either 0 or 1 to give same result as mixed integer programming.

The constraint given by (3.9) requires that the fraction of area under any land cover is greater than or equal to zero, i.e. the decision variable should be non-negative. Constraints (3.10) to (3.12) restrict the maximum levels of sediment, phosphorous and nitrogen levels allowed to leave the watershed. $\sum_{i=1}^{N}\sum_{j=1}^{M}\mathrm{Sed}_{ij}X_{ij}$ is the total sediment yield from all HRUs and Sedyld is the allowable sediment load level. Constraint (3.10) implies that the total sediment load to the stream from all HRUs in the watershed should be less than the allowable level. Constraints (3.11) and (3.12) are similar constraints for total P and total N respectively. The model can be run with one or more constraints at a time, i.e. only one or combination of all pollutants can be constrained. Constraining any of the pollutants, sediment nitrogen or phosphorous has an effect on the other. The LUDM maximum return model gives the maximum return for given sediment and nutrient constraint level and decides the land cover type on each HRU to achieve this.

LUDM: Minimum Incentive Model. In this model, the objective is to minimize the total incentive payment required to achieve an allowable level of sediment and nutrient load. Unlike maximum return model, this approach uses economic incentives to effect land use change. The method determines the best way to efficiently utilize limited water quality incentives to achieve maximum environmental benefits by inducing a voluntary shift in land use. The model determines the most effective land distribution that requires minimum government water quality incentives for achieving allowable sediment and nutrient load levels. The algebraic expression of the modeling approach is given by:

$$Min\sum_{i=1}^{N}\sum_{j=1}^{M}INC_{ij}X_{ij} \tag{3.13}$$

The objective function given by expression (3.13) is the total incentive payment required to achieve sediment and nutrient load goals. The LUDM model decides the land cover type for each HRU to achieve the sediment nutrient load goals with minimum possible government water quality payments.

The constraints used in LUDM minimum incentive model are:

$$\sum_{i} X_{ij} \leq 1 \tag{3.14}$$

$$X_{ij} \geq 0 \quad \text{and} \quad INC_{ij} \geq 0 \tag{3.15}$$

$$\sum_{i=1}^{N}\sum_{j=1}^{M} NI_{ij}X_{ij} + \sum_{i=1}^{N}\sum_{j=1}^{M} INC_{ij}X_{ij} \leq T\arg et\, Income\, Level \tag{3.16}$$

$$INC_{ij} \leq NI_{ij\{crop\}} - NI_{ij\{grass\}} \tag{3.17}$$

$$\sum_{i=1}^{N}\sum_{j=1}^{M} Sed_{ij}X_{ij} \leq Sedyld \tag{3.18}$$

$$\sum_{i=1}^{N}\sum_{j=1}^{M} P_{ij}X_{ij} \leq Pyld \tag{3.19}$$

$$\sum_{i=1}^{N}\sum_{j=1}^{M} N_{ij}X_{ij} \leq Nyld \tag{3.20}$$

INC_{ij} is incentive needed to induce a shift in land use. The product INC_{ij} X_{ij} is zero if HRU j is not under switchgrass since X_{ij} is zero.

X_{ij} and INC_{ij} are decision variables. Constraint (3.15) assures that the decision variables are non-negative. Constraint (3.16) assures that the target watershed income level is met. The target income level was chosen as the maximum income that could be achieved if there were no constraints. This helps determine the minimum incentive levels for land owners to voluntarily shift to the less profitable but environmentally friendly land use option. The underlying assumption is producers will be willing to switch to environmentally friendly land cover if they receive incentives equal to the difference in net income between the new crop and the most profitable crop they would produce without the incentive as given by constraint (3.17). Constraint (3.17) requires that the incentive payment needed for switching to switchgrass or CRP is the difference in net income between the most profitable crop and the replacement crop. Constraints (3.18) to (3.20) are sediment and nutrient load constraints as in (3.10) to (3.12) in the LUDM maximum return model.

LUDM: Maximum Return Uniform and Non-Uniform Abatement Models. The difference between the two approaches considered in this section is that under non-uniform pollutant reduction approach requires that the total pollutant load to the streams be below the allowable level while in the uniform reduction approach the load from each of the HRUs is reduced by the same amount. For instance, in the uniform reduction approach each land unit or HRU is required to reduce sediment or nutrient load by 20% while in the non-uniform abatement approach it is required that the total sediment or nutrient load is reduced by 20% regardless of the amount of reduction from each HRU. Because of the heterogeneity in pollution abatement efficiency across HRUs, the land distribution pattern determined in the two approaches will be different. In the uniform reduction approach sediment load from each HRU is reduced uniformly in proportion to the area of the HRU regardless of location or soil type. A comparison was made to evaluate the relative economic advantages of the uniform and non-uniform reduction approaches to achieve the same level of total load to the stream.

LUDM: Non-Uniform Abatement Model. The LUDM maximum return model discussed above is essentially a non-uniform abatement model since the sediment and nutrient load constraints given by (3.10) to (3.12) put a limit on the sum of total loads to streams. The algebraic expressions (3.7) to (3.12) apply to this model.

LUDM: Uniform Abatement Model. This approach refers to uniform regulation where by each land unit is expected to reduce pollutant load in the same proportion. Contrary to the non-uniform abatement approach in which the objective is to reduce total sediment load, in this case constraints are put on each land unit. Similar modeling approach is used as the non-uniform abatement approach, the difference being on how the sediment and nutrient load constraints are set. The objective function and the constraints are given by:

$$Max \sum_{i=1}^{N} \sum_{j=1}^{M} NI_{ij} X_{ij} \tag{3.21}$$

$$\sum_{i} X_{ij} \leq 1 \tag{3.22}$$

$$X_{ij} \geq 0 \tag{3.23}$$

$$\sum_{i=1}^{N} Sed_{ij} X_{ij} \leq \frac{A_{ij}}{A} Sedyld \tag{3.24}$$

$$\sum_{i=1}^{N} P_{ij} X_{ij} \leq \frac{A_{ij}}{A} Pyld \tag{3.25}$$

$$\sum_{i=1}^{N} N_{ij} X_{ij} \leq \frac{A_{ij}}{A} Nyld \tag{3.26}$$

The constraints given by (3.24) to (3.26) dictate that the sediment and nutrient load from each HRU is below a fraction of the total allowable sediment and nutrient loads Syld, Nyld, and Pyld. The fraction is the ratio of the area of each HRU and the total area of the watershed.

LUDM: Erosion Charge Model. The objective of this document is not to consider offsite impacts. However, one would be remiss if it were not mentioned. The following is a few general comments, not included to draw any conclusion. Offsite water quality effects of land use directly affect downstream population. The net returns from a social perspective will be the net returns from production less the damage costs associated with land use decisions. To truly access the economics of erosion control, the downstream damage cost should be considered in the calculation of net social benefits. However, the offsite economic costs of soil erosion can be difficult to quantify. It is known that costly offsite damage can cause severe threats to the environment (Barbier, 1995). Lake and reservoir capacity is lost to sedimentation each year which can result in a need for costly dredging. Other problems include sediments interfering with the breeding and feeding of various aquatic species, and the severity of flooding is affected by siltation (National Academy of Sciences, 2000). The erosion that occurs on the farm can reduce the productivity of land, labor, and capital on the farm, and increase the need for fertilizer and other inputs. The net social benefits is the total watershed income minus damages cost if damage cost or erosion charges if damage costs are paid by the land owners in the form of erosion charges. The objective should be to maximize the net social benefits, or:

$$Max \sum_{i=1}^{N} \sum_{j=1}^{1819} NI_{ij} X_{ij} - DC[Sedyld] \tag{3.27}$$

where DC is the damage cost per ton of sediment discharged which can also be paid by landowner in the form of erosion charge. The constraints are given by:

$$\sum_{i} X_{ij} = 1 \tag{3.28}$$

$$X_{ij} \geq 0 \tag{3.29}$$

$$\sum_{i=1}^{N} \sum_{j=1}^{M} Sed_{ij} X_{ij} \leq Sedyld \tag{3.30}$$

$$\sum_{i=1}^{N}\sum_{j=1}^{M} P_{ij} X_{ij} \leq Pyld \quad (3.31)$$

$$\sum_{i=1}^{N}\sum_{j=1}^{M} N_{ij} X_{ij} \leq Nyld \quad (3.32)$$

Results and Discussion

General

As discussed previously, land use decision models are used in this study to determine land use distribution that maximize economic returns and minimize incentives required to achieve sediment and nutrient loading goals. Each of the methods used will be discussed in subsequent sections.

Optimal land use allocation changes with constraint level. The optimal land distribution depends on sediment yield form each land unit and the opportunity cost of converting the land use to less erosive cover type. Sediment yield map and opportunity cost of converting peanuts by switchgrass shown in Figures 3-5 and 3-6. Sub-basins that are highly erodible and less productive are converted first by the model to less erosive cover type such as switchgrass or CRP land.

The LUDM assigns a land cover type for each HRU and maximizes (e.g. LUDM maximum return model) or minimizes (e.g. LUDM minimum incentive model) the objective function value while meeting allowable level of sediment load to the streams. For a lower level of allowable load, most of the HRUs are assigned to a land cover type that is less erosive such as switchgrass or CRP. The total net return under this condition is lower than what can be obtained at higher allowable sediment load since switchgrass and CRP generate relatively lower net return per acre compared to peanuts.

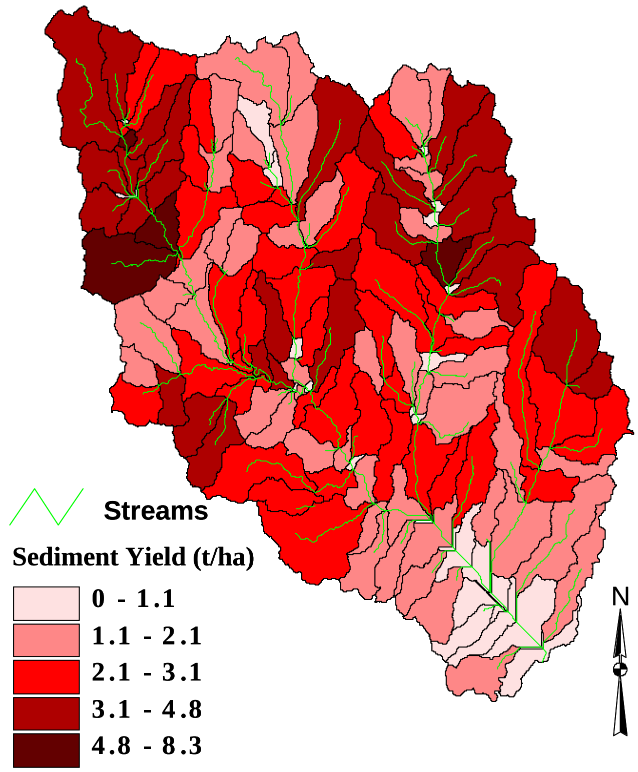

Figure 3-5. Sediment yield map for Fort Cobb basin original land cover.

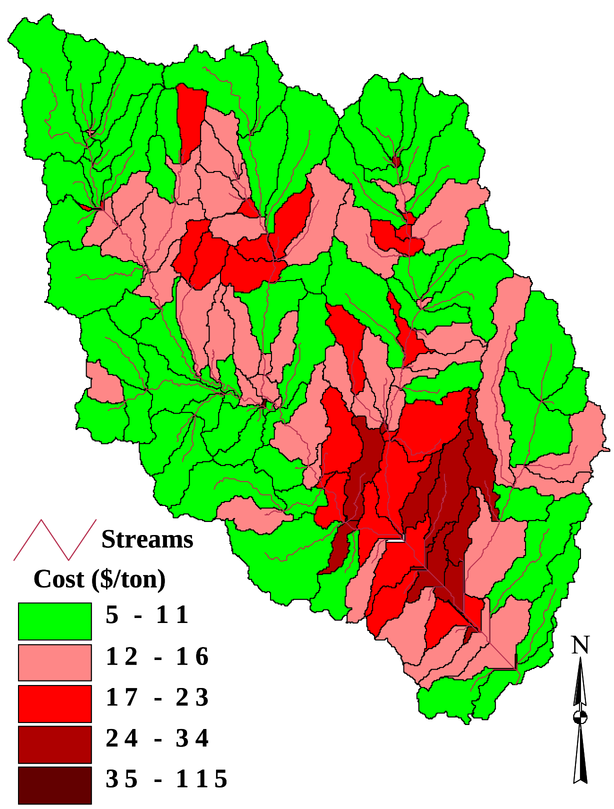

Figure 3-6. Opportunity cost of replacement by switchgrass ($/ton).

The LUDM model selects land cover type for each HRU based on the amount of sediment yield from each cover type, the net return obtained from each cover type which depends on yield, cost of production and price of each crop. The efficient land use for each HRU maximizes net returns or minimizes incentive required for a given level of load. In this study, sediment and nutrient load levels were varied by equal amounts from low to high, and optimum land use combinations and impacts on the net income or incentive required are identified at varying levels of allowable sediment and nutrient load.

Switchgrass production on conventional agricultural cropland has distinct environmental advantages compared to conventional and minimum tillage wheat, grain sorghum, or peanuts. Thus average sediment yield (MUSLE), total P and total N load were reduced by an average of 99%, 95%, and 91%, respectively, for land cover change from peanuts to switchgrass. The reduction in nutrient load to a large extent is associated with reduction in sediment load and runoff. Highly erodible sub-basins are assigned to land cover types which result in less soil and nutrient loss. Crops with high economic value such as peanuts are assigned by the model to less sensitive locations in the watershed so that the total income from the watershed can be maintained or possibly increased. The results from each of the modeling approaches are presented next.

Comparing Conversion to Switchgrass and CRP Lands Using LUDM

LUDM: Maximum Return Model. The LUDM maximum return model was run to evaluate the economic and environmental impact of replacement of croplands both by CRP and switchgrass. The model determines optimal land distribution and maximum return for a given allowable level of sediment and nutrient load. An example of profit maximizing land distribution for a 20% reduction in sediment load from base scenario as a result of replacement by switchgrass is given in Figure 3-7. The optimal land distribution for the same level of reduction for replacement by CRP is given in Figure 3-8. The CRP acreage was limited to 25% of the total watershed area following the regulation which states that maximum acreage which may be placed in the CRP may not exceed 25% of the total cropland in a county (Zinn, 1995). As result, in the case of replacement by CRP, part of the minimum tillage wheat appeared in the optimal solution. Subbasins converted to switchgrass, CRP or minimum tillage wheat are areas with high sediment yield and low opportunity cost of replacement as shown in Figures 3-5 and 3-6.

Switchgrass and CRP lands under bermudagrass were compared for their relative economic and environmental advantages to minimize sediment and nutrient load to the streams. The allowable sediment and nutrient load level was varied and the effect on land allocation and income was investigated under both options i.e. replacement by switchgrass and CRP. The income for switchgrass is the sale of the harvest. The income from CRP lands to the landowners is the rental payment ($/acre).

The percent total area allocated for each cover type and the corresponding total income that can be obtained from the watershed for varying levels of allowable sediment load is shown in Figure 3-9A to 3-9F. Most of the abatement is accomplished through the conversion of HRUs that are more erodible and less productive since the objective is to maximize the net returns.

For a comparison between switchgrass and peanuts, Figure 3-9A, 3-98C, and 3-9E show how land allocation and total income change with change in allowable sediment and nutrient load from the watershed. As the allowable sediment and nutrient load level is decreased, more land is devoted to switchgrass production. The income curve has a steep slope at the lower allowable sediment and nutrient level and becomes flat as the allowable sediment and nutrient level is increased. This is because the more erodible HRUs with low opportunity cost of conversion are converted to switchgrass first and as the allowable levels are further decreased, HRUs that are less erodible and with high opportunity cost of conversion are converted. Other crops such as grain sorghum and wheat were compared in the study were not competitive with switchgrass both from economic and environmental perspective, therefore, they are not selected in the higher or lower sediment and nutrient constraint levels.

The model was run for CRP instead of switchgrass with the additional constraint that the total area under CRP should not exceed 25%. Unlike the case for switchgrass, minimum tillage wheat appeared in the optimal solution because the area under CRP was limited to 25% and minimum tillage wheat was the best alternative among the remaining land cover types to achieve the sediment and nutrient load goal.

Again, the net return from grain sorghum and conventional wheat are relatively low and sediment and nutrient load from these crops are high compared to minimum tillage wheat and switchgrass, therefore, they did not appear in the solution although included in the mix of crops. Figures 3-9B, 3-9D, and 3-9F show the change in land allocation between CRP, minimum tillage wheat, and peanuts with allowable sediment, total phosphorous, and total nitrogen load levels. The total income from the watershed changes with change in allowable sediment and nutrient load. The pattern of change for replacement by minimum tillage wheat and CRP is similar to replacement by switchgrass, however as shown in Figure 3-9B, 3-9D, and 3-9F, the income curve is steeper compared to replacement by switchgrass. The implication is that the loss in income to the producers from putting land in switchgrass production is less than the loss as a result of enrolling it in CRP for the same amount of reduction in sediment and nutrient load.

The model was also run to see the land allocation for a 20% and 10% decrease in sediment load from the base scenario and the corresponding change in net return for replacement by switchgrass and CRP. The results are shown in Table 3-13. The base scenario refers to the loads under original land cover conditions given in Table 3-8.

Table 3-13. Sediment load, land cover distribution and average annual income for a 20%, 10%, and 0% reduction in sediment load from base scenario.

Alternatives	Item	Load reduction		
		0%	10%	20%
I. Replacement by switchgrass	% area peanuts	36	34	31
	% area switchgrass	64	66	69
	Sediment load (Mg)	210	189	168
	Income (Million USD)	18.0	17.5	17.0
II. Replacement by CRP and other crops	% area peanuts	32	28	25
	% area CRP	25	25	25
	% area min.till wheat	43	47	50
	Sediment load (Mg)	210	189	168
	Income (Million USD)	12.0	11.0	10.0

In case I the reduction in sediment load is achieved by converting more area into switchgrass production. Similarly, in case II the sediment reduction goal is achieved by converting to CRP and minimum tillage wheat, since CRP is limited to 25% of the watershed and minimum tillage wheat is the next less erosive crop among others. In both the LUDM model assigns sub-basins with high sediment yield to less erosive cover type, switchgrass in the first case and conservation reserve program and minimum tillage wheat in the second case.

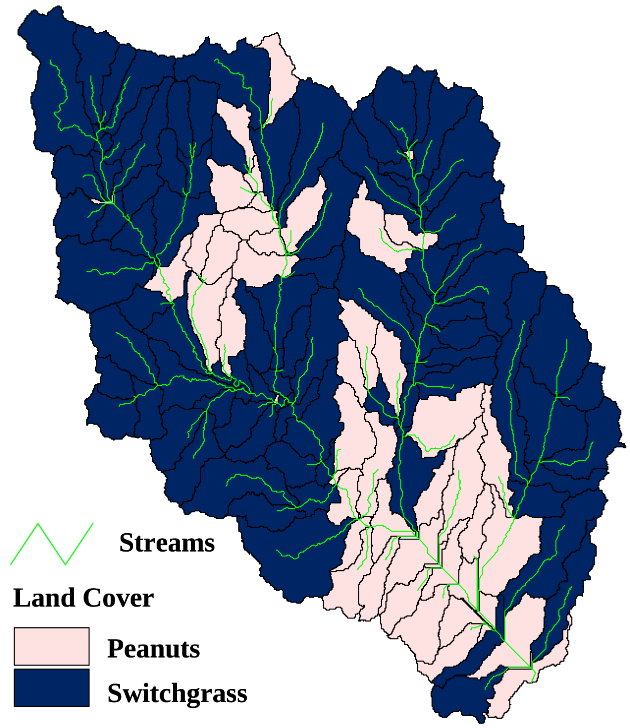

Figure 3-7. Optimal land allocation beween sitchgrass and peanuts for 20% reduction in sediment yield from base scenario for 1991-2000.

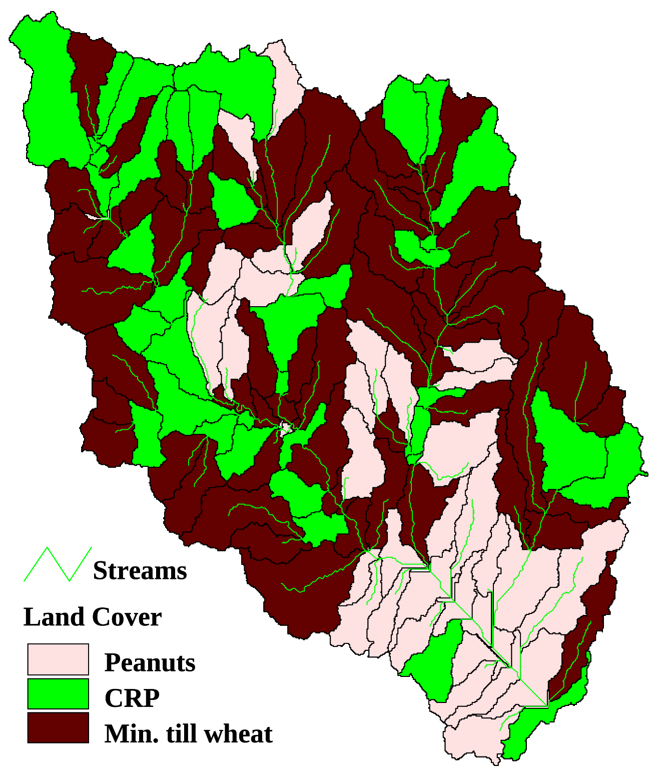

Figure 3-8. Optimal land distribution for minimum tillage wheat, CRP and peanuts for a 20% reduction in sediment load from base scenario for 1991-2000.

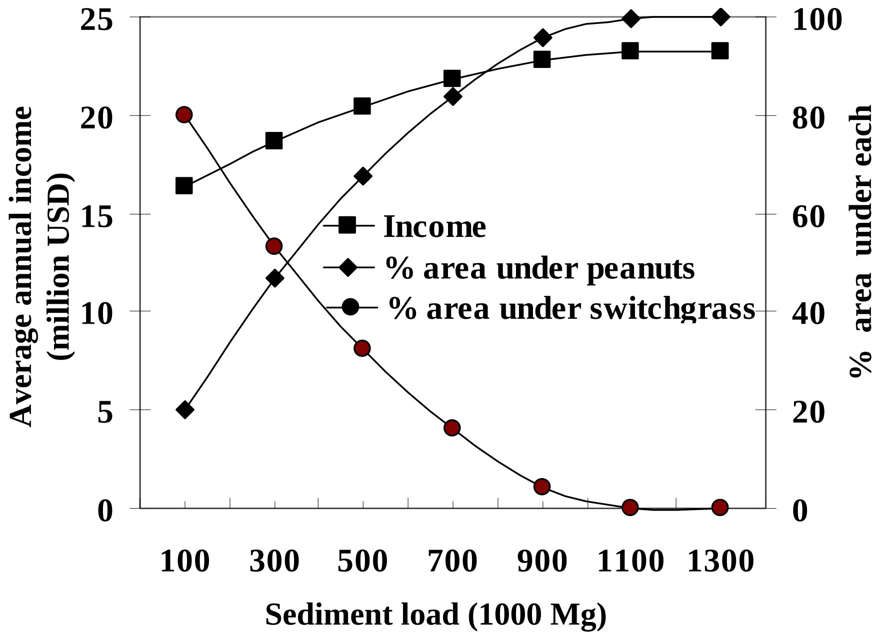

Figure 3-9A. Income, sediment load and land use interactions for the year 1991-2000 (replacement of peanuts by switchgrass).

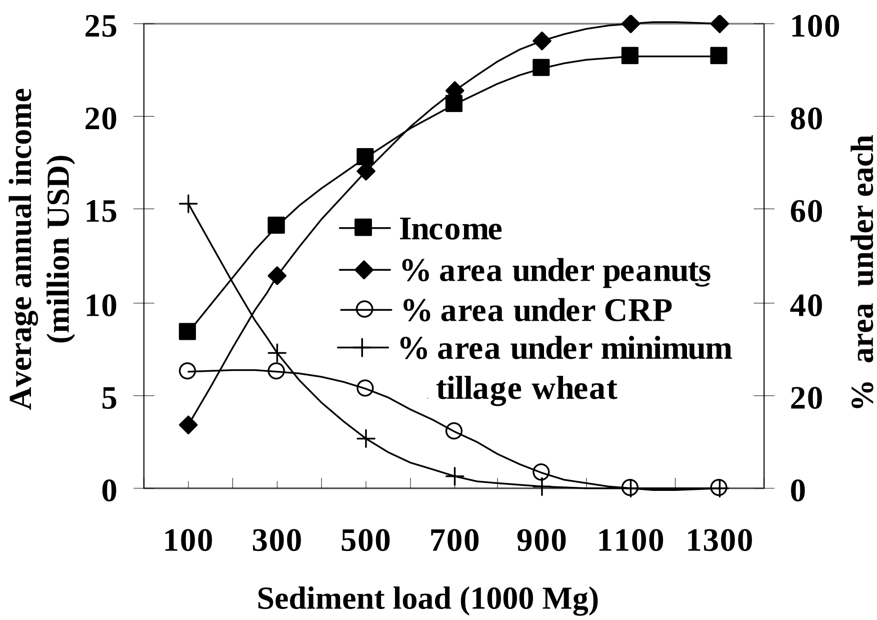

Figure 3-9B. Income, sediment load and land use interactions for the year 1991-2000 (replacement of peanuts by CRP and minimum tillage wheat).

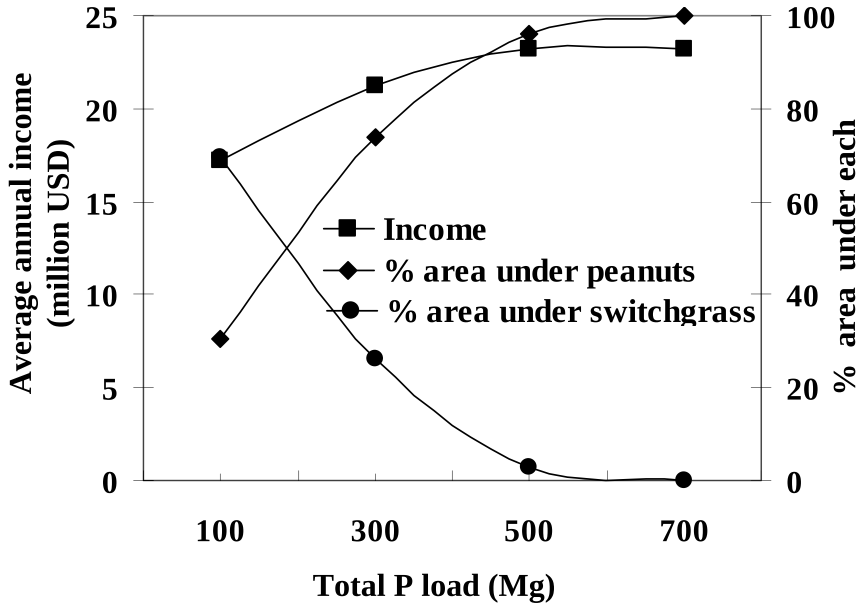

Figure 3-9C. Income, total P load and land use interactions for the year 1991-2000 (replacement of peanuts by switchgrass).

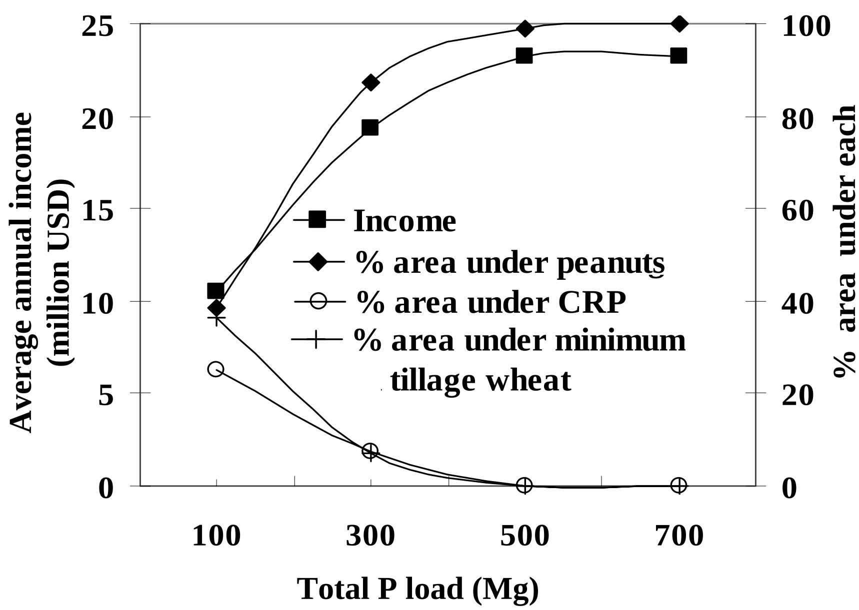

Figure 3-9D. Income, total P load and land use interactions for the year 1991-2000 (replacement of peanuts by CRP and minimum tillage wheat).

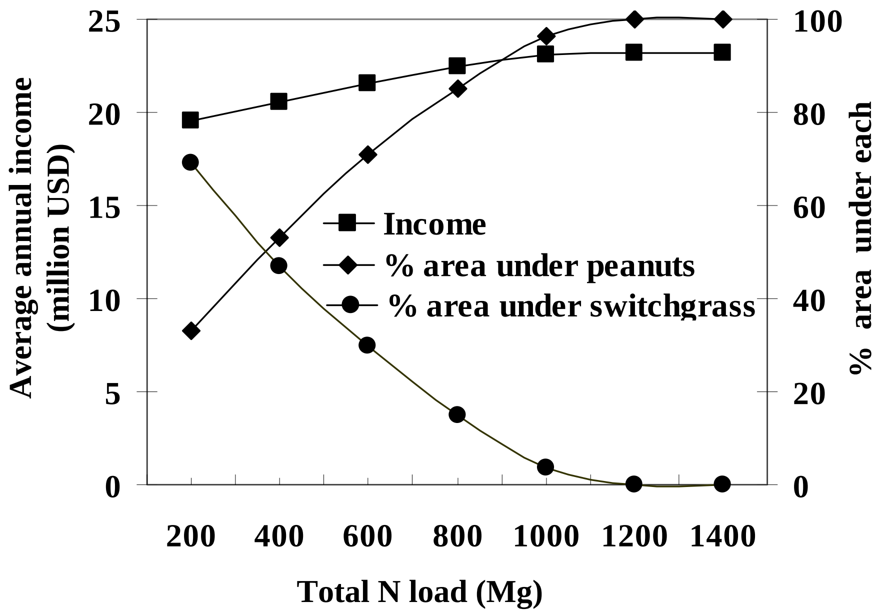

Figure 3-9E. Income, total N load and land use interactions for the year 1991-2000 (replacement of peanuts by switchgrass).

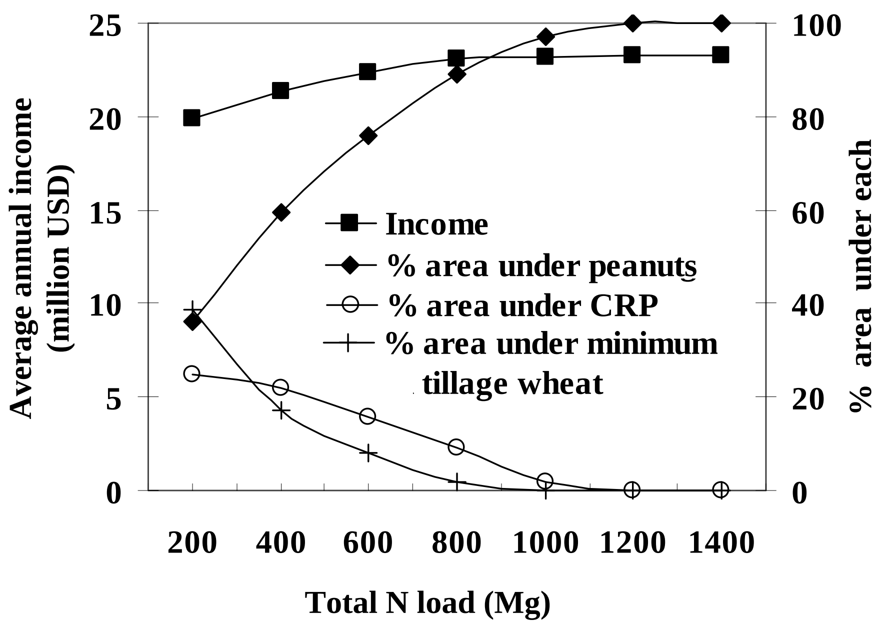

Figure 3-9F. Income, total N load and land use interactions for the year 1991-2000 (replacement of peanuts by CRP and minimum tillage wheat).

LUDM: Minimum Incentive Model. Figure 3-10 shows minimum sediment load to the streams that could be achieved for a given amount of water quality incentives. The government payments per acre are calculated as the difference in net returns per acre between the most profitable crop in the watershed (peanuts) and switchgrass or CRP land. This is based on the assumption that a landowner is willing to shift to switchgrass or enrolls in CRP provided the difference in potential income is paid, in addition to the long-term soil conservation benefits. In the case of CRP, since the crop is not harvested, the producer receives no income other than rental payments. As shown in Figure 3-10 the incentive required to result in the same level of load level is higher for CRP land as compared to switchgrass production. This is assuming a ready market for switchgrass.

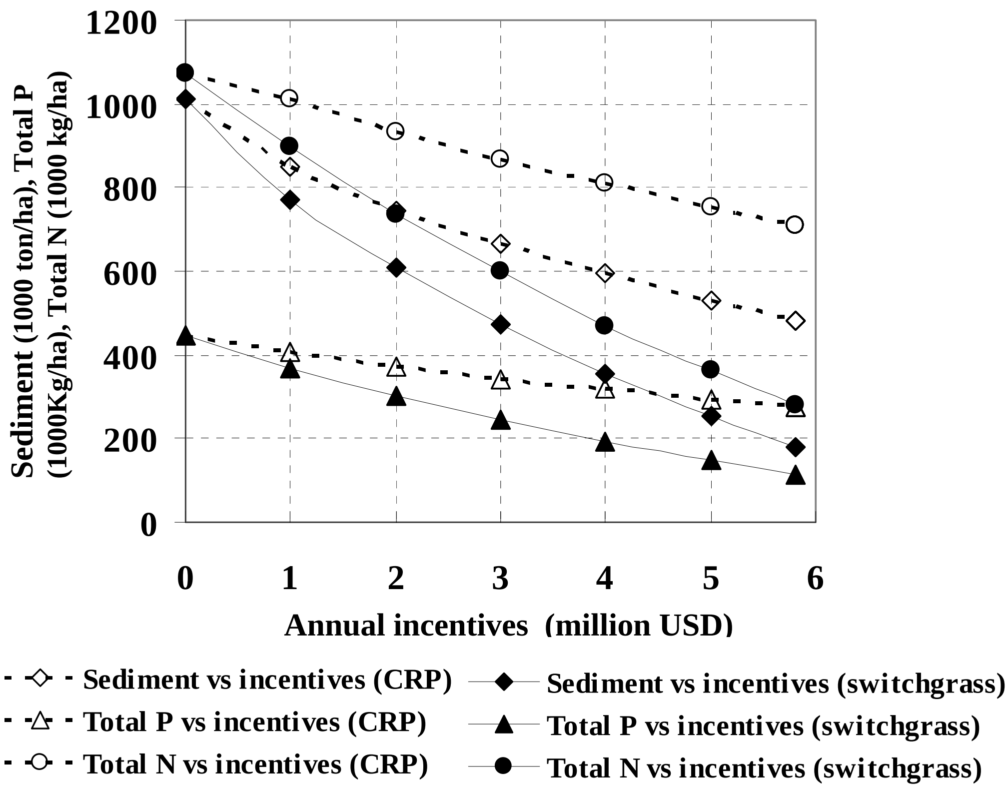

Figure 3-10. Comparison of annual incentives for switching to CRP and switchgras for the year 1991-2000.

LUDM: Non-Uniform and Uniform Abatement Models

As stated earlier, in the non-uniform load reduction approach in which the concern was to reduce total sediment and nutrient load to the streams, most of the abatement is accomplished through a few of the HRUs. In contrast, the uniform reduction approach pollution abatement is shared equally by all HRUs. In the uniform reduction approach, each HRU is required to reduce load in proportion to the area of the HRU.

As shown by the income curves, in all the cases in Figures 3-11A to 3-10C the uniform reduction approach is less cost effective. The total area that needs to be under switchgrass is higher for uniform reduction approach for a given level of sediment and nutrient load as compared to non-uniform reduction approach. This makes the uniform reduction approach less attractive to producers.

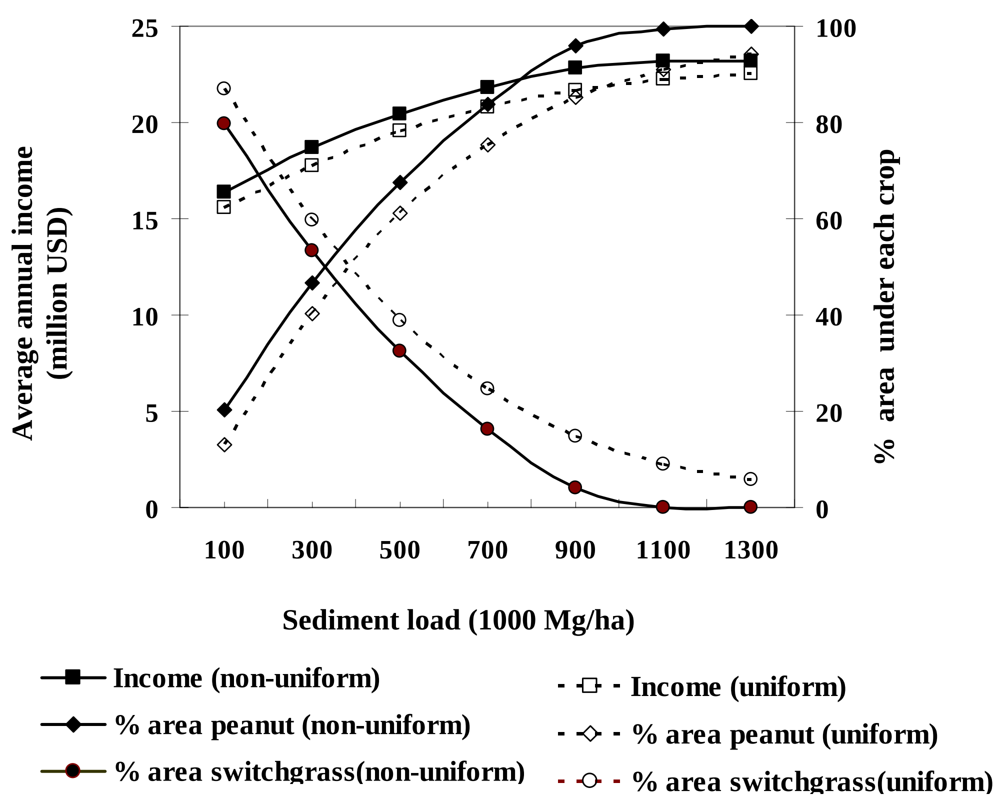

Figure 3-11A. Income, sediment load and land use interactions for the year 1991-2000 under uniform and non-uniform reduction (replacement of peanuts by switchgrass).

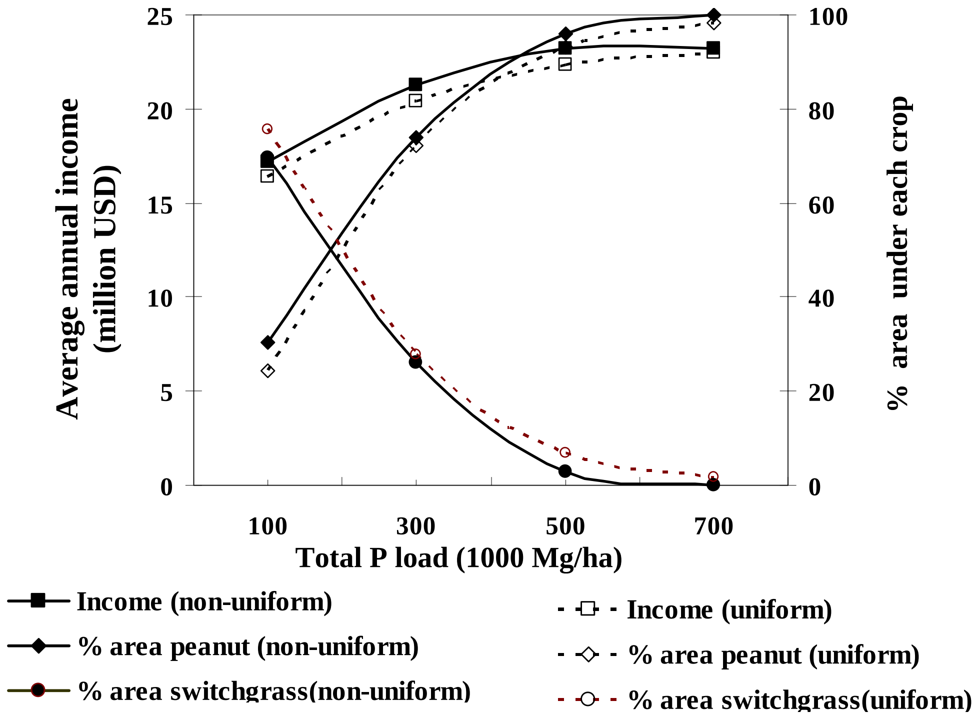

Figure 3-11B. Income, total P load and land use interactions for the year 1991-2000 under uniform and non-uniform reduction (replacement of peanuts by switchgrass).

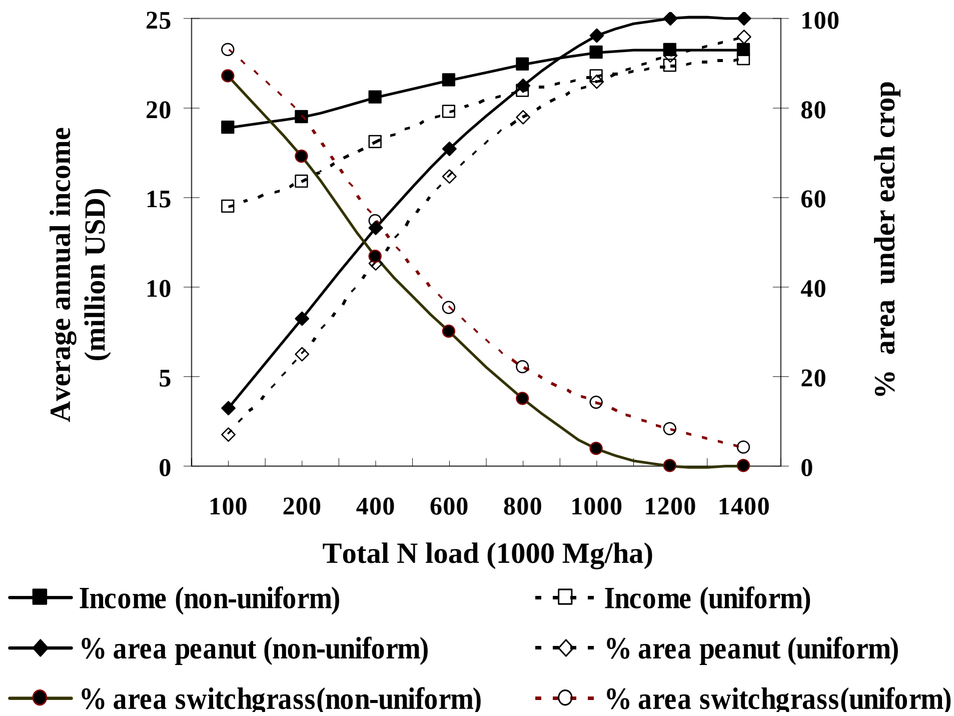

Figure 3-11C. Income, sediment load and land use interactions for the year 1991-2000 under uniform and non-uniform reduction (replacement of peanuts by switchgrass).

LUDM: Erosion Charge Model

The actual off-site damage costs have not been assessed in this study, however, a surrogate analysis shows the effects of instituting offsite erosion charges per ton of sediment load to the streams on land allocation and net social benefits by varying the erosion charges assuming the damage cost are paid by landowners in the form of erosion charges. The results are shown in Figure 3-12. The net social benefits are the total watershed income minus the erosion charges.

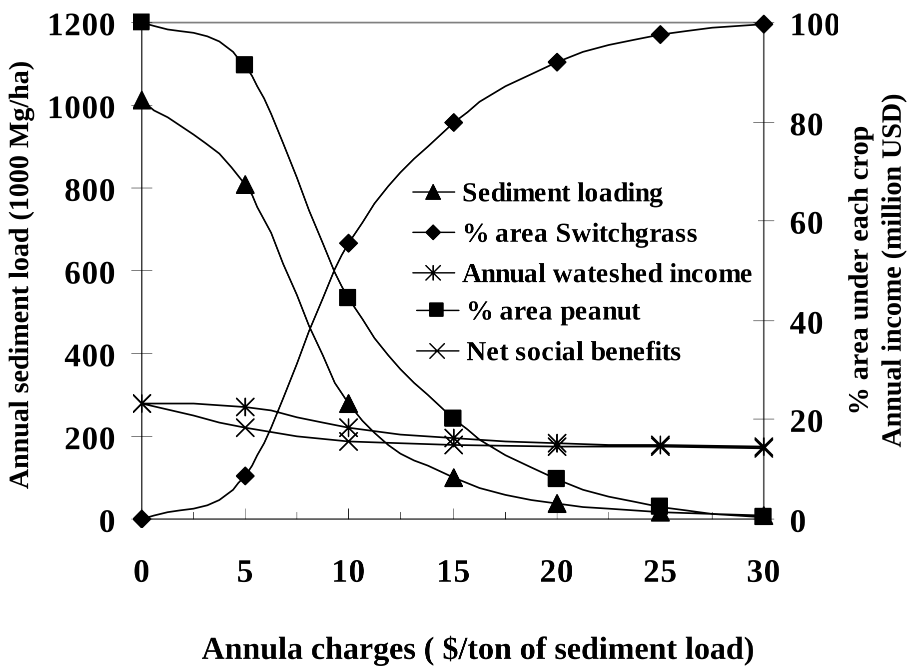

Figure 3-12. Effect of charges on sediment load on land allocation, annual income and sediment load for switchgrass for year 1991-2000.

Higher erosion charges would cause a shift to environmentally-friendly land cover type (switchgrass) since abatement would be a cheaper option to landowners as compared to paying erosion charges. This is not presented as a recommendation, but as an example of analysis that could be run with tools discussed in this dissertation. It is interesting to note that the net social benefits did not decline dramatically as the charge per ton increased from 0 to $30 while the total sediment load decreased considerably. This is because the net social benefit is the total income from the watershed less the damage costs, and the total damage costs decreases as the erosion charge is increased, because the model allocates more land to less erosive cover type (switchgrass). As the erosion charge is increased most of the land is allocated to switchgrass and the sediment load decreases. The result is that the net social benefits approach the total watershed income from crop sales. The drop in income also depends on the relative value of the crop used to replace the erosive cover type and its effectiveness in reducing sediment load per unit area of land converted.

Summary and Conclusions

The Conservation Reserve Program (CRP) and bioenergy crop (switchgrass) were evaluated as alternatives as replacements for row crops on parts of the watershed to reduce sediment and nutrient load to the streams. A Land Use Decision Model (LUDM) was written for this analysis using General Algebraic Modeling System (GAMS). SWAT model was used to determine sediment and nutrient load. Crops and tillage methods analyzed were switchgrass, conventional and minimum tillage wheat, peanuts, grain sorghum, and CRP lands. Land use decisions were made based on maximizing income subject to defined sediment and nutrient load and also based minimizing government water quality incentive payment subject to sediment and nutrient load. Using the income maximization approach, a non-uniform sediment and nutrient reduction goal across Hydrologic Response Units (HRUs) was compared with a sediment reduction goal that is uniform across all HRUs. The predicted reduction in sediment yield as a result of replacement of minimum tillage wheat by switchgrass was 95%. The predicted reduction for replacement of other crops and tillage methods such as conventional tillage wheat, grain sorghum and peanuts was more than 98%. The predicted reduction in total P load varied from 80% for minimum tillage wheat to 95% for peanuts. The reduction for total N load was slightly lower ranging from 65% to 90% for minimum tillage wheat and peanuts respectively. The analysis predicted that the loss in income for the same amount of load reduction, as a result of replacement of peanuts by switchgrass, is less than it is for replacement by CRP and the incentive required per ton of sediment or nutrient reduced as a result of replacement of croplands by CRP and minimum tillage wheat is higher than the payment required for replacement by switchgrass.

The LUDM written using mathematical programming is a valuable tool for modeling land use in conjunction with SWAT model. The model can be used to generate different land use scenarios based on environmental and economic goals. With the help of the model, multiple relationships between the decision variables and the constraints can be interpreted. The SWAT model generates data useful for the LUDM.

For each crop, loss in income per ton of reduced sediment load increases with the total amount of sediment abated since the model selects less productive and highly erodible lands first and gradually moves to more productive and less erodible lands as the constraint level is increased. This is desirable since the objective is to obtain the highest possible load reduction per dollar lost as a result of replacing a more profitable land cover type by less profitable ones.

The non-uniform pollutant reduction approach which equalizes the marginal cost of abatement across the sub watersheds is more cost effective compared to uniform reduction approach.

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Chapter IV

SGRASSF: A SIMPLIFIED PROCEDURE FOR COMPUTING VEGETATIVE FILTER STRIP TRAPPING EFFICIENCY

Abstract

A simplified procedure SGRASSF was developed to compute sediment trapping efficiency using vegetative filter strips (VFS) based on the Kentucky filter strip model GRASSF. The model is used in SEDIMOT III and other models to compute sediment trapping on a watershed scale. In the original GRASSF, the impact of flow, infiltration and sediment properties are predicted. These properties include flow depth, velocity, sediment particle size distribution, width of filter strip, density of vegetation, height of vegetation, and slope. In GRASSF, the depth of flow required for trapping efficiency calculation is determined in an implicit equation developed from continuity equation and a calibrated Manning's equation for overland flow velocity. In SGRASSF, the computation of flow depth was simplified by developing an explicit equation using regression techniques on a large number of simulated data points. The effect of advance of deposition wedge as a result of sediment inundation and grass recovery during the growth period on trapping efficiency was taken into account. The computation of equilibrium hydraulic radius used to calculate the advance of the deposition wedge has also been simplified. The modified model gives similar results to SEDIMOT III with an

R^2 value equal to 0.92. The Nash-Sutcliffe coefficient, used as an indicator of goodness-of-fit of the simplified model to the original model, was determined to be 0.9. The modified model can be used to calculate sediment trapping from multiple sub-watersheds using the daily precipitation data, sediment yield data, and sub-watershed characteristics such as soil type, vegetation cover, slope and size.

Introduction

Runoff carrying sediment from non-point sources has long been recognized as a major pollutant of surface water. Sediment-bound pollutants, such as phosphorous and pesticides, are also a major pollution concern. Several management practices have been suggested to control runoff quantity and quality from disturbed areas. One of these practices discussed in Chapter III as optimal land distribution is to replace land uses generating high sediment loads with conservation crops.

A second alternative to the optimal land distribution approach or total replacement approach described in Chapter III would be to trap or filter-out as much pollutants as possible using best management practices including vegetative filter strips and buffer strips located just upslope of all concentrated flow channels. Filter strips are land areas of either planted or indigenous vegetation, situated between a potential pollutant source area and a receiving surface-water body.

Research verifies that filter strips are effective in the control of many agricultural and urban non-point source pollutants, but are most effective controlling sediment (Ohio State University Extension, 1994). Sediment bound materials could be effectively reduced using filter strips both by settling and infiltration but dissolved substances can only be trapped by infiltration, thus have a better chance to flow through. The idea of implementing best management practices (BMPs) is widely accepted as a desirable solution to the problem of non-point source pollution from agricultural sources. Runoff may carry sediment and organic matter, plant nutrients and pesticides that are either bound to the sediment or dissolved in the water. A filter strip provides water quality protection by reducing the amount of sediment, organic matter, nutrients and pesticides in runoff before the runoff enters a concentrated flow channel.

Several mechanisms cause VFS to be effective in trapping sediment. First of all the VFS may effectively reduce runoff volume by infiltration. Also, flow velocities are deceased primarily because the VFS hydraulic roughness resulting in enhanced settling and reduce sediment transport (Barfield et al., 1975, Hayes et al., 1984). Grass filter strips generally have high sediment trapping efficiencies as long as the flow is shallow and uniform and the filter is not submerged. For submerged flow, the increased turbulence reduces trapping. Also if the surface is not uniform from side-to-side, flow can be concentrated in one location, decreasing the area where infiltration is occurring and increasing turbulence both of which reduce trapping of suspended sediment (Haan, et al., 1994).

Studies were conducted by Barfield et al. (1998) on the effectiveness of natural riparian grass buffer strips in removing sediment, atrazine, nitrogen and phosphorus from surface runoff. They showed that trapping percentages for sediment and chemicals typically ranged above 90% and that most of the chemicals were trapped by infiltration into the soil matrix. Further, trapping efficiency was shown to increase with filter strip length and with fraction of water infiltrated. Chaubey et al. (1994) observed a mass reduction of TSS, TN, and TP in surface runoff by 66%, 0% and 27%, respectively, with a 4.6-m long filter strips. Studies by Edwards et al. (1996), Srivastava et al. (1996), and Lim et al. (1998) showed the reductions in the concentration of soluble pollutants is not as significant as settleable pollutants. Young et al. (1980) concluded that 10-m wide grass filter strips could reduce up to 70% of the amount of fecal coliforms bacteria in runoff. Fajardo et al. (2001) reported that with filter strips up to 99% nitrogen removal efficiency and up to 87% fecal coliforms removal efficiency are possible from runoff originated from stockpiled manure. According to Ikenberry and Mankin (2000), vegetated filter strips can reduce pollutant concentrations from 70% to 90% in runoff from open animal feedlots.

Several modeling efforts have been undertaken to simulate VFS efficiency in removing pollutants from surface waters. Researchers at the University of Kentucky (Barfield et al. 1978, 1979; Hayes et al. 1984) developed and tested a model (GRASSF) for filtration and infiltration of suspended solids by artificial grass media. This physically-based model takes into account a number of important field parameters that affect sediment transport and deposition through the filter (sediment type and concentration, vegetation type, slope and length of the filter). The GRASSF model can be used for determining the sediment filtration capacity of grass media as a function of flow, sediment load, particle size, flow rate, slope, and media density. Another vegetative strip model (VFSMOD) was also developed to study hydrology and sediment transport through vegetative filter strips. VFSMOD is based on GRASSF algorithm developed specifically for the filtration of suspended solids by grass (Muñoz and Parsons, 2000). Filter strip subroutines must be included in large scale hydrologic models to predict sediment and nutrient trapped. A very simple relationship for predicting filter strip trapping efficiency is in the SWAT model for sediment and nutrient leaving each HRU (Neitsch et al., 2002). Edge-of field filter strips are defined in a hydrologic response unit (HRU). Sediment, nutrient, pesticide and bacteria loads in surface runoff are predicted to be reduced as the surface runoff passes through the filter strip using the equation:

$$TE = 0.367(W_f)^{0.2967} \tag{4.1}$$

where TE is the fraction of the constituent load trapped by the filter strip, and W_f is the width of the filter strip in meters (Neitsch et al., 2002). The buffer strip width can be input for each HRU. The equation is used to estimate trapping efficiency based on filter strip width alone and does not consider other parameters such as particle size distribution and grass properties and slope which are important for sediment trapping. The trapping efficiency using the above equation approaches 100% for a filter strip width equal to 30m, regardless of particle size and flow rate. The validity of that approximation is questionable.

The VFS algorithms in SEDIMOT III are based on GRASSF. The model takes both properties of grass, soil and land slope in addition to buffer width and can be adapted to large scale watershed models such as SWAT but the computation time becomes excessive for 20-plus year simulations because of the implicit nature of many of the equations. By simplifying the equations to explicit forms computational time can be greatly reduced.

The objective of this study is to develop a simplified series of explicit equations to predict sediment trapping efficiency and effluent sediment load for grass filter strips. The

SGRASSF procedures can be incorporated into large scale hydrologic models or be used with a hydrology-sedimentology load function as stand-alone model to evaluate VFS sediment trapping on a watershed scale for a number of years.

Methodology

In GRASSF, the filter strip is divided into four zones A(t), B(t), C(t), and D(t) as shown in Figure 4.1. Because the sediment is moving downstream, all are given as a function of time. The upstream zone A(t) is the region where sediment has been deposited up to the level of the VFS, hence all the sediment that flows into this zone flows in to the next zone B(t). Zone B(t) is the area where deposition occurs uniformly along the deposition wedge. Zone C(t), downstream zone B(t), is the zone in the filter where there is sufficient sediment deposition to allow bed load transport but not sufficient to alter the bed slope. Zone D(t) represents that area within the filter in which the layer of litter on the bed has not been totally filled with sediment, therefore there is no bed load transport in this zone. This is the zone where trapping of relatively fine particles occurs by settling and infiltration. Trapping in this zone D(t) will be discussed first, followed by procedures used to determine trapping in the deposition wedge. Modification has been made to transform implicit equation in GRASSF for calculating trapping in each zone into explicit equations.

The total trapping efficiency equation for VFS in GRASSF model is the sum of fractions of coarse, medium and fine particles trapped. The total trapping efficiency is given by:

$$TE = \left[f + f_d^c (1 - f) \right](1 - f_{ri}^1) + f_d^m (f_{ri}^1 - f_{ri}^0) + f_d^f f_{ri}^0 \qquad (4.2)$$

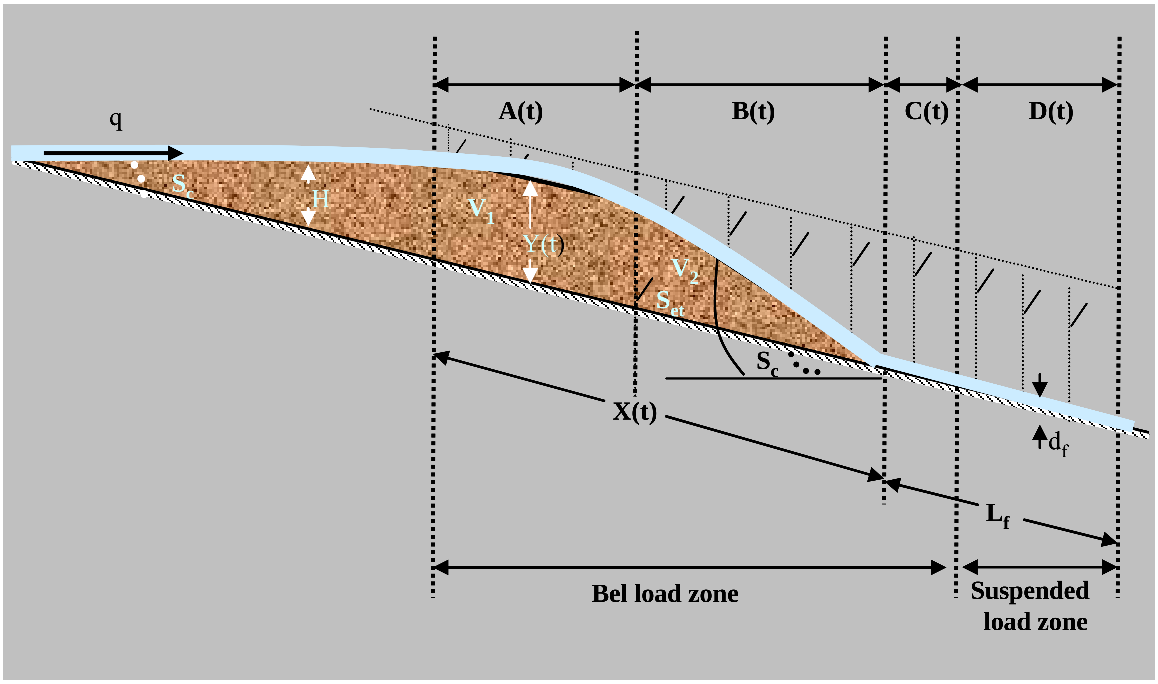


Figure 4-1. Illustration of trapping mechanisms in VFS**.**

where TE is total trapping efficiency, f is fraction of sediment trapped in deposition wedge, f_d^c is total trapping efficiency for coarse particles, f_d^m is total trapping efficiency for medium size particles, f_d^f is total trapping efficiency for fine particles, f_{ri}^1 is fraction corresponding to 0.037 mm, f_{ri}^0 is fraction corresponding to 0.004 mm.

Equation (4.2) has also been used in SGRASSF, however the method in which each of the parameters in the equation is determined has been modified. The trapping efficiency obtained using the two methods are compared. Sediment trapping is divided into coarse, medium and fine particle trapping. The coarse particles are trapped in the deposition wedge by settling. Coarse, medium and fine particles are trapped in the settling zone, zone D(t) by settling or infiltration.

Calculating Trapping by Settling in the Settling Zone

The discussions will start with trapping in the settling zone, zone D(t), which occurs as a result of infiltration and settling. Sediment that settles to the bed in zone D(t) is considered trapped since there is no bed transport in this zone. Procedures used for computing trapping by settling and infiltration are discussed below.

Trapping efficiencies for coarse, medium and fine particles are calculated separately as a function of the Reynolds number Re and fall number, $N_{f,i}$, a parameter related to the number of times a particle could settle from the water surface to the bed. The fall number given is given by:

$$N_{f,i} = \frac{V_{s,i} L}{V_m d_f} \tag{4.3}$$

where $N_{f,i}$ is dimensionless fall number for particle class i (coarse, medium or fine particles). $V_{s,i}$ is the settling velocity (fps), V_m is the overland flow velocity (fps), L is length of settling zone (ft), d_f is overland flow depth (ft).

The trapping by settling for coarse, medium and fine particles, T_s, in zone D(t) is given by:

$$T_s = EXP(-0.00105 R_e^{0.82} N_f^{-0.91}) \tag{4.4}$$

where R_e is Reynolds number given by:

$$R_e = \frac{V_m R_s}{\nu} \tag{4.5}$$

where V_m is overland flow velocity (fps) and ν is kinematic viscosity (ft^2/s).

Settling Velocities, $V_{s,i.}$ Settling velocity, $V_{s,i}$, for a given diameter d (mm) in the above equation is determined for each particle class i, i.e. for coarse, medium and fine particles, or:

$$V_s = \left| \begin{array}{ll} 2.81 d^2 & d \leq 0.1 mm \\ \log Vs = -0.34246 (\log d)^2 + 0.98912 \log d + 1.14613 & d > 0.1 mm \end{array} \right| \quad (4.6)$$

where d is representative particle diameter (mm).

A particle size distribution is required to determine the representative diameter to compute the settling velocity for each particle class. The Kentucky model treats particles greater than 0.037 mm as coarser particles. Particles smaller than 0.037 mm are not trapped in the deposition wedge. All particles less than 0.037 are transported to zone D(t).

Representative diameters are needed to calculate transport capacity in the deposition wedge in Einstein's bed load transport equation and settling velocity in the settling zone. In SGRASSF, representative particle diameters are determined using the CREAMS approach with soil texture data (% sand, silt, and clay as input). The use of soil texture data in SGRASSF simplifies the method used in the GRASSF model to get representative diameters.

The CREAMS model defines five particle classes for eroded material (primary clay, silt, sand, and small and large aggregate). However for the purpose of calculating trapping efficiency in SGRASSF, the five classes were reclassified into three classes. Sand and coarse aggregates were treated as coarse particles, silt and small aggregates as medium particles and clay as fine particles. Representative diameters for each of the three classes were then calculated based on CREAMS equations given in Table 4-1. The weighted average of the representative diameters for sand and large aggregate was taken as representative diameter for the coarser particles and it was used as a representative

Table 4-1. Representative diameters by classes based on soil matrix fractions [based on Foster et al. (1985)].

Class	Representative diameter (mm)	Range of limits of clay in soil matrix	Specific gravity	
Clay	$D_{Cl} = 0.002$		2.65	(4.7)
Silt	$D_{si} = 0.010$		2.65	(4.8)
Sand	$D_{sa} = 0.200$		2.65	(4.9)
Small aggregate	$D_{sg} = 0.030$ $D_{sg} = 0.2(O_{cl}-0.25)+0.030$ $D_{sg} = 0.100$	$O_{Cl} < 0.25$ $0.25 \leq O_{Cl} \leq 0.6$ $O_{Cl} > 0.60$	1.80	(4.10)
Large Aggregate	$D_{lg} = 0.30$ $D_{lg} = 2\ O_{cl}$	$O_{Cl} \leq 0.15$ $O_{Cl} > 0.25$	1.60	(4.11)

Table 4-2. Fractions of sediment by classes based on soil matrix particle size distribution [after Foster et al. (1985)].

Class	Representative diameter (mm)	Range of limits of clay in soil matrix	
Clay	$F_{Cl} = 0.26\ O_{cl}$		(4.12)
Small aggregate	$F_{sg} = 1.8\ O_{cl}$ $F_{sg} = 0.45\text{-}0.6(O_{cl} - 0.25)$ $F_{sg} = 0.6\ O_{cl}$	$O_{Cl} < 0.25$ $0.25 \leq O_{Cl} \leq 0.5$ $O_{Cl} > 0.50$	(4.13)
Silt	$F_{si} = O_{si} - F_{sg}$		(4.14)
Sand	$F_{sa} = O_{sa}(1\text{-}Ocl)^5$	$O_{Cl} < 0.25$ $0.25 \leq O_{Cl} \leq 0.6$ $O_{Cl} > 0.60$	(4.15)
Large Aggregate	$F_{lg} = 1\text{-}F_{cl}\text{-}F_{si}\text{-}F_{sg}\text{-}F_{sa}$	$O_{Cl} \leq 0.15$ $O_{Cl} > 0.25$	(4.16)

diameter to calculate settling velocity for coarse particles in equation (4.6). It was also used as a representative diameter to calculate bed load transport capacity using Einstein's method as given subsequently in equation (4.52). The weighted average diameter of small aggregate and silt was used as a representative diameter for medium particles and diameter of clay was used for fine particles in equation (4.6).

Overland Flow Velocity, V_m. Overland flow velocity is given as a function of grass and flow properties. In the original model overland flow velocity was predicted in VFS by using a specially calibrated form of Manning's equation in which an analogy is made between flow in vegetation with spacing of S_s and flow in a rectangular channel with a flow depth of d_f and width of S_s using the following equation (Hayes et al., 1978):

$$V_m = \frac{1.5 R_s^{2/3} S_c^{1/2}}{xn} \tag{4.17}$$

where R_s is the spacing hydraulic radius (ft), S_c is slope of the channel (ft/ft), xn is calibrated value of Manning's n value for particular vegetation (Haan et al., 1994, Table 9.10). Barfield et al. (1979) and Tollner et al. (1976) used a constant Manning's roughness coefficient to describe sediment laden non-submerged overland flow in vegetative filter strips. In their model, a "spacing hydraulic radius" replaced the hydraulic radius in the velocity equation to account for the effect of vegetation.

The spacing hydraulic radius R_s is given by:

$$R_s = \frac{S_s d_f}{2d_f + S_s} \tag{4.18}$$

where, S_s is grass spacing (ft). Hence, the continuity equation is given by:

$$q_w = \frac{1.49}{xn}\left(\frac{S_s d_f}{2d_f + S_s}\right)^{2/3} S_c^{1/2} d_f \tag{4.19}$$

A value for d_f in the above equation cannot be explicitly expressed as a function of q_w and must be determined by trial and error with more than one positive root possible. The equation was converted to an explicit form by simplifying the velocity term, using procedure described below.

In the velocity equation, the roughness parameter, xn, varies based on grass type. R_s is also affected by the property of the grass and the depth of flow. The type of grass affects the resistance to flow and the flow depth and the slope affects the energy required to overcome the resistance. Therefore, the assumption here is that it is possible to get a simplified equation that relates overland flow velocity to depth of flow and slope along with some parametric values reflecting grass properties that would give the same result as the calibrated Manning's form in equation (4.17). It should be noted that equation (4.17) was calibrated based on observed data. Hence overland flow velocity in the modified equation was described as a function of depth of flow and slope for a given vegetation type. Data was generated for overland flow velocity using the original formula given by equation (4.17) and the relationship between overland flow velocity, slope and/or depth of flow was investigated for different grass types. Using different power functions the velocity equation was simplified using non-linear regression (NLIN) procedure in SAS.

Two simplified explicit models were compared as an alternative to the implicit form in equation (4.17): a) one in which velocity was expressed as a function of slope alone with a constant depending on the type of vegetation and b) another in which velocity is expressed as a function of both depth of flow and slope again with a constant depending on the type of vegetation, i.e. $V_m = f(Sc)$ and $V_m = f(S_c, d_f)$. Data patterns indicated that power functions are more appropriate in both cases given by:

$$V_m = \mu S^m \tag{4.20}$$

and

$$V_m = \alpha (df)^{\beta} S_c^{\gamma} \tag{4.21}$$

In the first approach parameters and m were determined using the SAS NLIN procedure for each grass type using the data generated using equation (4.17). Similarly in the second approach, parameters α, β and γ were determined using the same procedure. In both cases the power term for slope m and γ approached 0.5 for all grass types. The NLIN procedure was rerun using a constant value of 0.5 for the power term for slope to determine the remaining parameters in both cases. The parameter values are presented in Table 4-3 and Table 4-4 for varying grass types. A graphical illustration of the differences in the two models is shown in Figures 4-2 (A) and (B) and Figures 4-3 (C) and (D) for bermudagrass and tall fescue, respectively.

Number of simulations, ranges of slope used, depths of flow used, and other grass parameters used in the development of explicit relations for overland flow velocity are given in Table 4-5. Topography should be relatively flat to maintain shallow flow conditions. Performance is best with longitudinal grades of five percent or less to maintain uniform sheet flow conditions (Washington State Department of Transportation, 1995), although VFS designs have been successful in steeper slopes ranging from 15 to 20% (Barfield and Hayes, 1988). Rainfall patterns and intensity also play a role. A 15% slope in arid and semi-arid climates would result in erosion rills because of rainfall intensity. Similarly for the second approach α, and β were determined for each grass.

Table 4-3. Parameter values and R^2 values for model 1 ($V_m = \mu S^{0.5}$).

Vegetation type	μ	m	xn	Spacing S_s (ft)	Correlation coefficient, R^2
Bermudagrass	1.39	0.5	0.074	0.045	0.85
Ryegrass	1.97	0.5	0.056	0.056	0.77
Tall fescue	1.93	0.5	0.056	0.053	0.80
Kentucky bluegrass	1.93	0.5	0.056	0.053	0.80
Blue grama	1.93	0.5	0.056	0.053	0.80
Centipedegrass	1.39	0.5	0.074	0.045	0.85
Buffalograss	1.88	0.5	0.056	0.050	0.79
Grass mixture	2.24	0.5	0.050	0.071	0.72
Alfalfa	3.39	0.5	0.037	0.100	0.59
Sericea lespedeza	3.61	0.5	0.037	0.129	0.55
Common lespedeza	3.86	0.5	0.037	0.183	0.50
Sudangrass	4.15	0.5	0.037	0.317	0.44

Table 4-4. Parameter values and R^2 values for model 2 ($V_m = \alpha d_f^{\beta} S_c^{0.5}$).

Vegetation type	α	β	γ	xn	Spacing S_s (ft)	Correlation coefficient, R^2
Bermudagrass	1.85	0.13	0.5	0.074	0.045	0.98
Ryegrass	3.10	0.18	0.5	0.056	0.056	0.98
Tall fescue	2.93	0.17	0.5	0.056	0.053	0.98
Kentucky bluegrass	2.93	0.17	0.5	0.056	0.053	0.98
Blue grama	2.93	0.17	0.5	0.056	0.053	0.98
Centipedegrass	1.85	0.13	0.5	0.074	0.045	0.98
Buffalograss	2.82	0.13	0.5	0.056	0.050	0.98
Grass mixture	4.40	0.23	0.5	0.050	0.071	0.98
Alfalfa	8.99	0.33	0.5	0.037	0.100	0.99
Sericea lespedeza	11.27	0.37	0.5	0.037	0.129	0.99
Common lespedeza	14.85	0.43	0.5	0.037	0.183	0.99
Sudangrass	21.13	0.52	0.5	0.037	0.317	0.99

Table 4-5. Range of parametric values for computing over land flow velocity.

Vegetation type	No. of [1] Simulations	Slope [2] range (%)	Flow depth [3] (ft)	xn [4]	Spacing [5] S_s (in)
See Table 4-3	5000	0 - 10	0 – 0.4	0.037-0.074	0.04-0.32

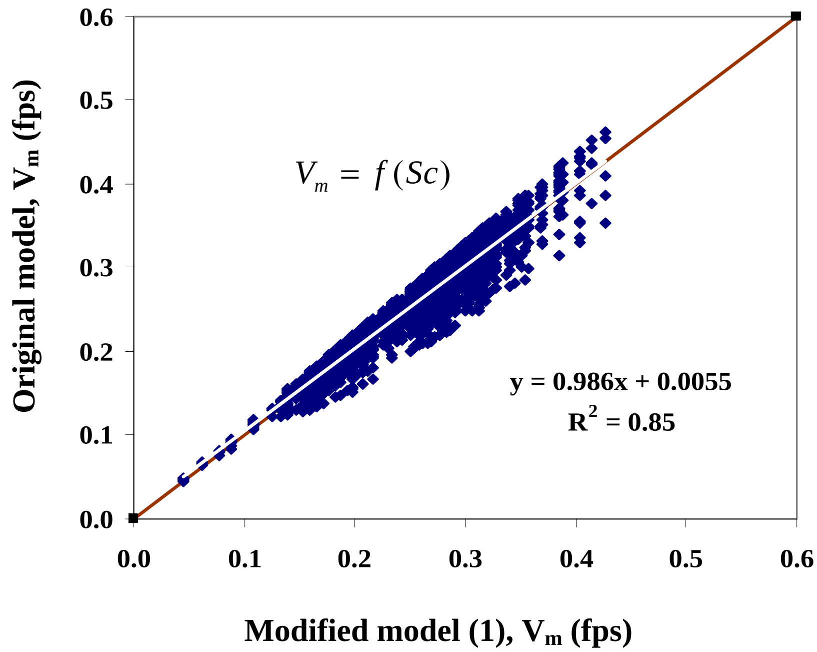
[1] No. of simulations used for model development.

[2] Ranges of slopes used in simulation based on recommended slope for VFS.

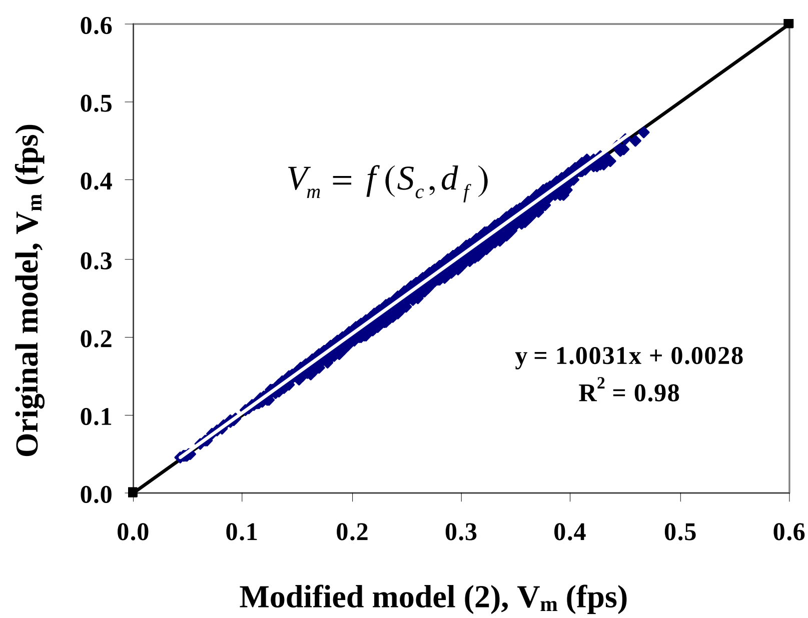
[3] Flow depth calculated using equation equation (4.33).

[4] Calibrated Manning's roughness coefficient from Haan et al (1994), p364, Table 9.10.

[5] Spacing of vegetative elements from Haan et al (1994), p364, Table 9.10.

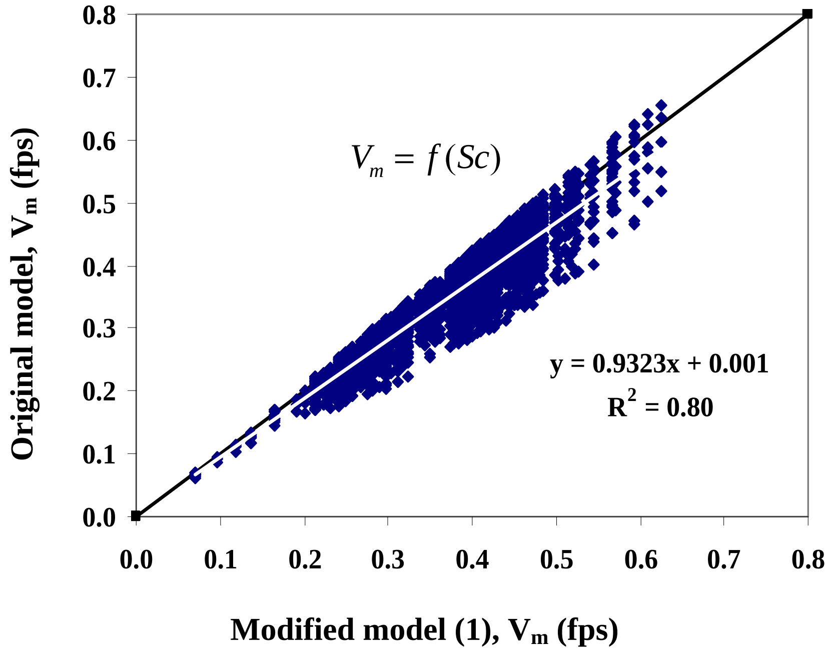


(A) Original vs modified model (1) (Bermudagrass)

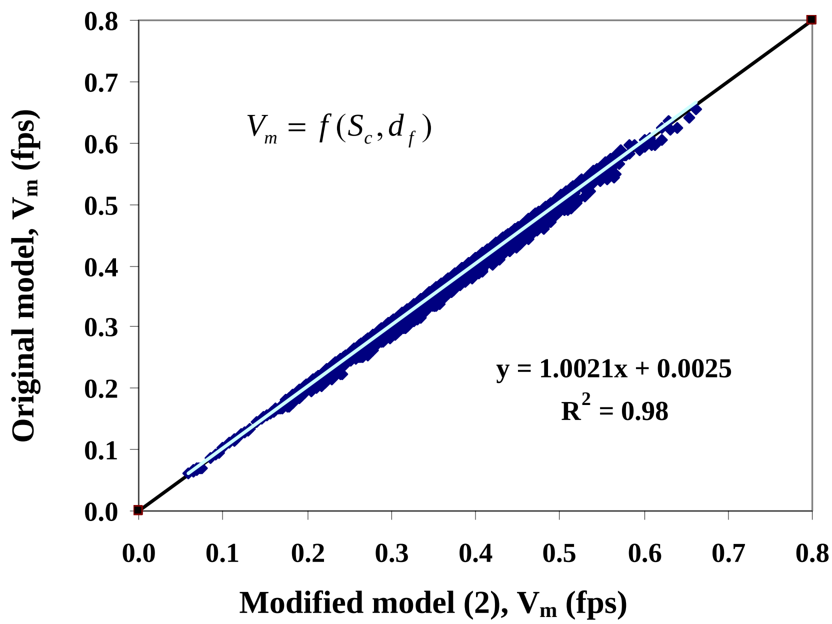


(B) Original vs modified model (2) (Bermudagrass)

Figure 4-2. Comparison of velocity predictions using original and modified VFS models for Bermudagrass.



(C) Original vs modified model (1) (Tall fescue)



(D) Original vs modified model (2) (Tall fescue)

Figure 4-3. Comparison of velocity predictions using original and modified VFS models for tall fescue.

Flow Depth, d_f. The computation of depth of flow required in the fall number calculation was simplified using the modified velocity equation. The depth of flow required for TE in equation (4.2) is determined using the continuity equation given by:

$$q_w = V_m d_f \tag{4.22}$$

where q_w is discharge per unit length (cfs/ft), V_m is flow velocity (fps) and d_f is flow depth (ft). The methodology for calculating d_f is given after a discussion of the peak discharge equation.

The GRASSF routine in SEDIMOT III uses storm hydrographs to calculate the depth of flow. Since depth of flow changes with time during storm, so does trapping efficiency with minimum trapping efficiency typically occurring at or near peak flow. In the modified procedure, the objective is to determine trapping efficiency during the storm that will give the same results as SEDIMOT III, using simple explicit relationships but hydrograph information is not usually as readily available as is peak discharge which can be computed using TR 55 or rational method. If the peak discharge is directly used, the computed depth of flow will be much greater than that occurring during most of the storm and the trapping efficiency will be underestimated. To adjust the trapping efficiency to the one that is computed from SEDIMOT III, the peak discharge should be adjusted using a correction factor. To develop a correction factor, a data set was needed relating trapping efficiency calculated with peak discharge to that determined with a hydrograph.

For consistency for the use with SWAT output, the modified rational method used in SWAT model was used to estimate the peak flow rate. The peak flow rate in the original rational method is given by:

$$q_p = 645CiA \tag{4.23}$$

where q_p is peak runoff rate (cfs), C is the runoff coefficient, i is the rainfall intensity (in/hr) and A is area in (mi^2). The modified rational equation used to estimate the peak flow rate in SWAT model is given by

$$q_p = \frac{645\alpha_{tc} Q_{surf} A}{t_c} \tag{4.24}$$

Q_{surf} (in runoff) in equation (4.24) replaces the C factor. t_c is time of concentration in (hr) given by:

$$t_c = \frac{0.027L^{0.6}n^{0.6}}{S_c^{0.6}} \tag{4.25}$$

where L is the subbasin slope length (ft), S_c is the average slope in the subbasin (m/m) n is Manning's roughness coefficient for the subbasin and 18 is a unit conversion factor. α_{tc} in equation (4.24) is the fraction of daily rainfall that occurs during the time of concentration, α_{tc} is given by:

$$\alpha_{tc} = 1 - \exp[2t_c \ln(1 - \alpha_{0.5})]; \quad t_c/24 \le \alpha_{tc} \le 1 \tag{4.26}$$

$\alpha_{0.5}$ is the fraction of daily rain falling in the half-hour of highest intensity rainfall. $\alpha_{0.5}$ in SWAT is calculated from triangular distribution and probabilistic approach. In SGRASSF, $\alpha_{0.5}$ was taken from NRCS Type Curves and is 0.21 for Type I, 0.38 for Type II and 0.20 for Type III distribution (Haan et al., 1994).

Using the peak discharge as representative of the entire storm will underestimate trapping efficiency. Therefore, a correction factor C' is needed to adjust the peak discharge to obtain a more accurate trapping efficiency. In SGRASSF, the trapping efficiency obtained in SEDIMOT III is used as a reference and a correction factor required to adjust TE from modified SGRASSF model to that of SEDIMOT III model was found to be well related to the peak discharge per unit length (perpendicular to the flow).

To develop a correction factor prediction equation the following steps were followed:

1) Trapping efficiency for VFS was determined from SEDIMOT III in which storm hydrographs were used for 2000 simulations.
2) For the same inputs on drainage area characteristics and VFS parameters trapping efficiencies were determined using trapping predicted from peak discharge.
3) Using the data set, various prediction relationships were evaluated with different parameters. Ranges of parameters used in deriving the C' factor are shown in Table 4-6.

Table 4-6. Range of values for development of C' factor.

No. of Simula tions	Slope (%)	Drainage area (acre)	Precip- itation (in)	Flow depth (in)	Dischrge (cfs/ft)	Spacing S_s (in)	Calibrated Manning's rougness (xn)
2000	0 - 10	0 - 5	0 - 6	0 - 3	0 - 0.2	0.54-3.80	0.037-0.074

[1] No. of simulations used for model development.
[2] Ranges of slopes used in simulation based on recommended slope for VFS.
[3] Ranges of area used in simulation based on recommended drainage area for VFS effectiveness.
[4] Ranges of precipitation values used in simulation.
[5] Flow depth calculated using equation equation (4.33).
[6] Discharge per unit width calculated using equation equation (4.31).
[7] Spacing of vegetative elements from Haan et al (1994), p364, Table 9.10.
[8] Calibrated Manning's roughness coefficient Haan et al (1994), p364, Table 9.10.

Based on the analysis the correction factor C' required to get the same trapping efficiency from using the peak discharge and storm hydrograph was determined for different VFS parameters and watershed characteristics. The best prediction of peak discharge adjustment was obtained by relating the correction factor to peak discharge per unit length, or:

$$C' = 0.0417(0.005 + q_w)^{-0.7157} \tag{4.27}$$

where q_w is discharge per unit length of filter perpendicular to the direction of flow is given by:

$$q_w = \frac{q_p}{W} \tag{4.28}$$

and W is the length of filter strip (ft). Thus,

$$q_{padj} = 0.0417(0.005 + q_w)^{-0.7157} q_p \tag{4.29}$$

where q_p is the peak flow rate given by equation (4.24) above and q_{padj} (cfs) is the adjusted flow rate. Thus, q_{padj} can be used to calculate the depth of flow such that same trapping efficiency is calculated as that obtained using a hydrograph in SEDIMOT III. The data relating peak discharge per unit length and the correction factor is given in Figure 4-4 below.

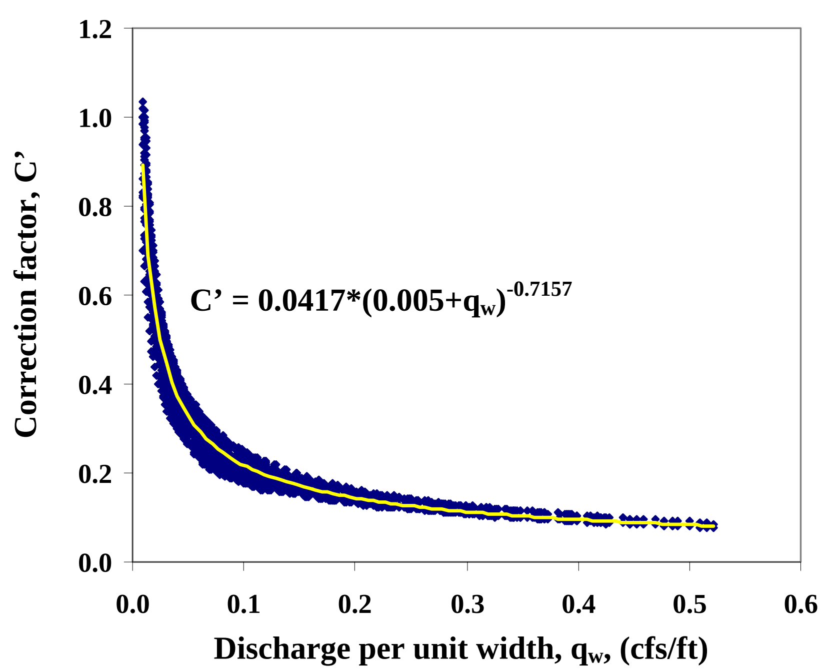


Figure 4-4. Correction factor for peak discharge.

Once a calibrated peak is calculated, the depth of flow d_f can be determined as follows:

$$q_{wadj} = V_m d_f \tag{4.30}$$

$$q_{wadj} = \frac{q_{padj}}{W} \tag{4.31}$$

In GRASSF V_m is determined using equation (4.17). Using the modified equation for V_m, the continuity equation becomes:

$$q_{wadj} = \alpha d_f^{\beta} S^{0.5} d_f \tag{4.32}$$

or, solving for d_f:

$$d_f = \left[\frac{q_{wadj}}{\alpha S^{0.5}}\right]^{\left(\frac{1}{1+\beta}\right)} \tag{4.33}$$

The above simplified equation can be used to solve for d_f for any standard grass type once the discharge and slope are known. Once d_f and V_m are determined the values can be used to calculate the fall number N_f in equation (4.3) and Reynolds number, R_e, and trapping efficiency for each particle size, T_s, in equation (4.4) .

Calculating Size Distribution Parameters f_{ri}^0 and f_{ri}^1

In addition to the parameters already discussed two other parameters are required to predict total trapping efficiency, TE, from equation (4.2). These are fractions corresponding to fine particle f_{ri}^0 that are trapped by settling in zone D and f_{ri}^1, the fraction of medium size particles that are not trapped in the deposition wedge. f_{ri}^1 in the model is the lower limit of the coarser materials. f_{ri}^0 and f_{ri}^1 determine the proportion of sediment that are fine and coarse, and thus affect trapping efficiency. The higher the proportion of fines, f_{ri}^{0}, the lower the trapping efficiency since the trapping efficiency for fines material is low. In SGRASSF, the CREAMS model, equations (4.12) to (4.16) are used to estimate particle diameters and fraction of clay, silt, small and large aggregates and sand in the eroded material. Due to deposition of coarser materials in the deposition zone the sediment size gets finer as sediment enters the settling zone.

Based on the classification given in Table 4-2 the fraction corresponding to lower limit of the coarser materials, f_{ri}^{1}, was calculated as:

$$f_{ri}^{1} = F_{cl} + F_{si} + F_{sg} + 0.5F_{sa} \tag{4.34}$$

and the fraction corresponding to the fine particles, f_{ri}^{0}, was calculated as:

$$f_{ri}^{0} = F_{cl} + F_{si} + F_{sg} \tag{4.35}$$

where F_{cl} is fraction of clay, F_{si} is fraction of silt, F_{sg} is fraction of small aggregates, F_{sa} is fraction of sand, and F_{lg} is fraction of large aggregates

It was discussed earlier that the peak flow rate was adjusted such that the trapping efficiency from SEDIMOT III with GRASSF and that predicted from SGRASSF are as close as possible. What was not mentioned earlier is that the values for f_{ri}^{o} and f_{ri}^{1} affect trapping efficiency since these parameters reflect the size distribution of the eroded material. Thus when developing the correction factor for peak discharge, C', in equation (4.27), in addition to adjusting the peak flow rate, the relations for f_{ri}^{o} and f_{ri}^{1} were also developed and optimized at the same time in order to achieve similar trapping efficiency prediction using SEDIMOT III with GRASSF and SGRASSF. The result is given by equations (4.34) and (4.35).

Calculating Trapping by Infiltration

In GRASSF, trapping by infiltration is determined at each time step of the hydrograph. The terms f_d^c, f_d^m, and f_d^f in equation (4.2) are the total trapping efficiencies for coarse, medium and fine materials in zone D respectively considering trapping both by settling and infiltration. Hayes et al. (1984) developed a prediction equation for trapping accounting for both settling and infiltration, or:

$$f_d = \frac{T_s + 2I(1-T_s)}{1 + I(1-T_s)} \tag{4.36}$$

where T_s and I account for trapping by settling and infiltration respectively. I is a dimensionless term related to the average infiltration rate:

$$I = \frac{q_{wadj} - q_{wo}}{q_{wadj} + q_{wo}} \tag{4.37}$$

where q_{wadj} given in equation (4.31) is inflow rate adjusted to obtain same trapping efficiency as the original model and q_{wo} is outflow rate. The outflow rate is obtained by subtracting the infiltration rate from the inflow rate, or:

$$q_{wo} = q_{wadj} - i_{av}L \tag{4.38}$$

where L is the length of VFS (ft) and i_{av} is average infiltration rate (ft/s).

In SGRASSF, calculations are made for an entire storm based on peak discharge. The procedure discussed above was used to account for infiltration. The following section discusses an alternative method to account for infiltration to the method described above. The percentage of particles trapped by infiltration can be calculated as the ratio of the total mass infiltrated to the total mass in the incoming flow. Total mass can be computed as the product of concentration and infiltration, or:

$$M_{\inf} = C_{\inf} I_{vol} \tag{4.39}$$

where M_{inf} is mass infiltrated per unit time, C_{inf} is concentration in the infiltration volume, and I_{vol} is i_{av} times A_{VFS} where A_{VFS} is the area under VFS. Similarly incoming mass per unit time is:

$$M_f = C_f q_{av} \tag{4.40}$$

where M_f is incoming mass per unit time, C_f is concentration in the incoming flow, and q_{av} is the flow rate.

Trapping by infiltration is given by:

$$TE_i = \frac{M_{\inf}}{M_f} = \frac{c_{\inf} I_{vol}}{c_f q_{av}} \tag{4.41}$$

where:

$$q_{av} = \frac{(q_{padj} + q_{po})}{2} \tag{4.42}$$

and:

$$q_{po} = q_{padj} - i_{av} A_{VFS} \tag{4.43}$$

If concentration in the infiltration volume is assumed to be the same as the concentration in the flow, then trapping by infiltration is the ratio of infiltration rate to flow rate, or:

$$TE_i = \frac{M_{\inf}}{M_f} = \frac{I_{vol}}{q_{av}} \tag{4.44}$$

The equation is modified assuming that only those particles which settle to the bottom of VFS are carried into the soil by infiltration or the concentration in the infiltration is directly proportional to settling. A further assumption made is that deposition in the settling zone is uniform. The fraction of sediment incoming mass deposited in the settling zone is equal to T_s. Another factor considered is that flow rate and concentration in the flow change as the flow moves through the VFS. Coarse, medium and fine particles were treated separately.

The fraction of particles in the flow when the flow enters VFS is 1 and the fraction of particles at the VFS exit is 1-T_s i.e. the fraction of incoming mass carried by the flow when the flow reaches the downstream end of VFS is 1-T_s. The average is (2-T_s)/2. Concentration in the infiltration is assumed to be proportional to settling and is taken to be T_s times incoming concentration. On the other hand, the average concentration in the flow decreases when settling increases and is given by incoming concentration times (2-T_s)/2.

$$TE_{\inf} = \frac{T_s C_f}{\left[(2 - T_s)/2\right] C_f} \frac{I_{vol}}{q_{av}} = \frac{2T_s}{(2 - T_s)} \frac{I_{vol}}{q_{av}} \tag{4.45}$$

$$f_d = T_s + \frac{2T_s}{2 - T_s} \frac{I_{vol}}{q_{av}} = T_s \left[1 + \frac{2I_{vol}}{(2 - T_s) q_{av}} \right] \tag{4.46}$$

The average infiltration rate was used in equations (4.38) and (4.46). Horton's equation was used to estimate the average infiltration rate during the runoff period. Horton's method is an empirical relation with parametric values selected based on experiment (Maidment, 1992). Infiltration rate in the Horton's Method is given by

$$f(t) = f_c + (f_0 - f_c) e^{-kt} \tag{4.47}$$

where f_c is the final infiltration rate, a value equal to the water transmission rate. f_0 is the initial infiltration rate which varies with soil type and vegetation cover and k is a measure of rate of decrease of infiltration rate. In its usual form, Horton's equation is most applicable to events for which the rainfall intensity exceeds the infiltration capacity. Infiltration rate decreases with time during the runoff period. By the time the flow from upland area reaches the VFS, the soil is already wet and the infiltration rate in the VFS is assumed to be lower than the rate at the beginning of the storm. Because there is a depth of flow in the VFS, it is reasonable to assume that infiltration is lower than the rainfall intensity to satisfy Horton's conditions even if the rainfall intensities are very light by the time the runoff reaches the VFS. This will justify the use of Horton's method. The use of variable infiltration rate in SGRASSF made trapping efficiency adjustment simpler. The objective in this case is not to determine the portion of rainfall that infiltrates but the portion of the incoming runoff from contributing area that is lost as a result of infiltration in the VFS. It is important to note that runoff from the contributing area into the VFS area does not start at the same time as the rainfall, i.e. some time T_L is required before onset of runoff. T_L can be estimated as the fraction of the duration of storm. By the time runoff starts the infiltration rate in the VFS is expected to go down as a result of rainfall occurring on the VFS.

Another important point is the fact that runoff does not reach all parts of the VFS at the same time even after runoff started, with upstream VFS area receiving runoff as soon as the runoff starts and the downstream end of VFS receiving flow after T_t, where T_t the time required for the flow to travel through the VFS. The average time required for the flow to reach VFS is $T_t/2$. Hence the total time T_{tot} required is $T_L + T_t/2$. The total volume of infiltration in the filter strip was obtained by evaluating the integral of the Horton's function over the interval (t_b-T_{tot}) where t_b is the duration of runoff, or:

$$V = \int_{T_{tot}}^{t_b} f(t)\,dt = f_c t + \frac{(f_c - f_0)(e^{-kt} - 1)}{k}\Bigg|_{T_{tot}}^{t_b} \qquad (4.48)$$

If T_{tot} is taken as fraction of t_b, T_{tot} is given by t_b where t_b is the time base or duration of runoff. The infiltration volume can be simplified as;

$$V = f_c t_b + \frac{(f_c - f_0)}{K}(e^{-kt_b} - 1) - \left[\mu f_c t_b + \frac{(f_c - f_0)}{K}(e^{-\mu kt_b} - 1) \right] \tag{4.49}$$

The time average infiltration rate during a runoff period can be estimated by dividing the total infiltration volume given by equation (4.49) by the interval (t_b- T_{tot}), or:

$$i_{av} = \frac{f_c}{1-\mu} + \frac{(f_c - f_0)}{(1-\mu)Kt_b}(e^{-kt_b} - e^{-\mu kt_b}) = \frac{1}{1-\mu}\left[f_c + \frac{(f_c - f_0)}{Kt_b}(e^{-kt_b} - e^{-\mu kt_b}) \right] \tag{4.50}$$

The i_{av} values obtained using equation (4.50) are used to calculate the outflow rate q_{wo} in equation (4.38). As a side note the values obtained using equation (4.50) are close to hydraulic conductivity as shown in Table 4.7 especially for longer duration runoff. In summary trapping efficiency by infiltration can be calculated by using an input average value for infiltration or by using a variable infiltration rate as calculated by the Horton equation (4.50).

Table 4-7. Infiltration constants, average infiltration rates, and total volume of infiltration for Bluegrass Turf (Terstriep and Stall, 1974), k = 2/hr.

Time	Initial and final infiltration rates (iph)								Infiltration volume				Average infiltration rate			
	A		B		C		D		A	B	C	D	A	B	C	D
D_{st}	f_c	f_o	f_c	f_o	f_c	f_o	f_c	f_o	in	in	in	in	iph	iph	iph	iph
2	1	10	0.5	8	0.25	5	0.1	3	2.8	1.9	1.1	0.6	1.9	1.3	0.76	0.41
4	1	10	0.5	8	0.25	5	0.1	3	3.4	1.9	0.9	0.4	1.2	0.6	0.34	0.15
6	1	10	0.5	8	0.25	5	0.1	3	4.5	2.3	1.2	0.4	1.0	0.5	0.27	0.11
8	1	10	0.5	8	0.25	5	0.1	3	5.8	2.9	1.4	0.5	1.0	0.5	0.25	0.10
10	1	10	0.5	8	0.25	5	0.1	3	7.2	3.6	1.8	0.7	1.0	0.5	0.25	0.10
12	1	10	0.5	8	0.25	5	0.1	3	8.7	4.3	2.2	0.8	1.0	0.5	0.25	0.10
14	1	10	0.5	8	0.25	5	0.1	3	10.1	5.0	2.5	1.0	1.0	0.5	0.25	0.10
16	1	10	0.5	8	0.25	5	0.1	3	11.5	5.7	2.8	1.1	1.0	0.5	0.25	0.10
18	1	10	0.5	8	0.25	5	0.1	3	13.0	6.5	3.2	1.3	1.0	0.5	0.25	0.10
20	1	10	0.5	8	0.25	5	0.1	3	14.4	7.2	3.6	1.4	1.0	0.5	0.25	0.10
22	1	10	0.5	8	0.25	5	0.1	3	15.9	7.9	3.9	1.5	1.0	0.5	0.25	0.10
24	1	10	0.5	8	0.25	5	0.1	3	17.3	8.7	4.3	1.7	1.0	0.5	0.25	0.10

The time base depends on storm runoff volume in addition to the watershed characteristics. If a triangular hydrograph is assumed, the time base can be calculated from the runoff volume and peak discharge, or:

$$t_b = \frac{2V}{q_p} \tag{4.51}$$

Calculating Trapping in the Deposition Wedge

Trapping in the deposition wedge (see Figure 4-1) is calculated in GRASSF with an implicit equation based on the Einstein's bed load formula (Tollner et. al., 1976). Further simplification was necessary in order to compute the fraction of sediment trapped in the deposition wedge with an explicit relation. Transport in the deposition wedge occurs if the transport capacity for bed load is higher than the incoming load. If the transport capacity is lower than the incoming sediment inflow rate into the VFS, sediment is deposited over the deposition wedge. Some coarse particles that are not trapped in the deposition zone, can still be trapped in the settling zone. Computation of sediment trapping in the deposition wedge is important to calculate the advance of the deposition wedge and the decrease in the effective length of the filter strip. The trapping efficiency has to be adjusted for the change in VFS length.

Calculating Transport Capacity in a VFS

The transport capacity for bed load within a VFS is calculated in GRASSF by calibrated Einstein's transport rate function given by:

$$q_{sd} = \frac{K(R_{sd} S_c)^{3.57}}{d_{pd}^{2.07}} \tag{4.52}$$

where:

$$K = (1.08)^{3.57} \gamma_w g^{1/2} SG(SG - 1)^{-3.07} \tag{4.53}$$

and R_{sd} is determined from flow depth given subsequently by equation (4.76).

If q_{sd}^{c} is in lb/sec-ft width, R_{sd} is in ft, and d_{pd} particle diameter in millimeters, then:

$$K = (6.462)10^{7} SG(SG - 1)^{-3.07} \tag{4.54}$$

Calculating Sediment Inflow Rate over Time

The incoming coarse material load rate is given by:

$$q_{si}^{c} = q_{si}^{t}(1 - f_{ri}^{1}) \tag{4.55}$$

where q_{si}^{t} is estimated from water flow rate, q, based on the assumption that the concentration c of sediment is a power function of water discharge, or:

$$c = kq^{a} \tag{4.56}$$

$$q_{si}^{t} = kq^{a+1} \tag{4.57}$$

In the calculation, it is desirable to use the adjusted peak discharge in instead of q in equation (4.57), hence:

$$q_{si}^{t} = kq_{adj}^{a+1} = k(C' q_p)^{a+1} \tag{4.58}$$

where C' is given by equation (4.27).

To calculate the sediment inflow rate using the above equation k needs to be determined from:

$$Y = \int_0^{t_b} kq^{a+1} dt = k\sum_{i=1}^{n} q_i^{a+1} \Delta t_i \tag{4.59}$$

To perform the integration over time using the gamma function relationship flow rate q, needs to be expressed as function of time. Haan (1970) developed the relationship describing q as a function of peak discharge and time, or:

$$q = \left[\frac{t}{t_p} e^{1-t/t_p} \right]^K q_p \tag{4.60}$$

where q (cfs) is the hydrograph ordinate at any time t (hr), q_p (cfs) is the peak flow rate and K is a dimensionless parameter defined by the equation:

$$K = 6.5 \left(\frac{q_p t_p}{V} \right)^{1.92} \tag{4.61}$$

where q_p (iph) is peak discharge, t_p (hr) is the hydrograph time to peak and V (in) is runoff volume and K is dimensionless (Haan et al., 1994). Using equation (4.60) in equation (4.59), the sediment yield becomes:

$$Y = \int_0^{t_b} k \left[q_p \left(\frac{t}{t_p} e^{1-t/t_p} \right)^K \right]^{a+1} dt \tag{4.62}$$

and:

$$k = \frac{Y}{\int_0^{t_b} \left[q_p \left(\frac{t}{t_p} e^{1-t/t_p} \right)^K \right]^{a+1} dt} . \tag{4.63}$$

The coefficient k can be computed if the denominator is determined in the above equation. The Simpson's rule was used to determine the integral. In the Simpson's rule:

$$\int_a^b f(x)dx \approx \frac{\Delta x}{3} (f(x_0) + 4 f(x_1) + 2 f(x_2) + 4 f(x_3) + ... + 4 f(x_{n-1}) + f(x_n)) \tag{4.64}$$

For n panels $\Delta x = t_b/n$ where t_b is time base of hydrograph and n is the number of panels. $f(x_0)$, $f(x_1)$, ..$f(x_n)$ can be evaluated from the gamma function. Hence:

$$\int_0^{t_b} q^{a+1}dt = \frac{t_b}{3n}\left(\frac{t_b.e}{nt_p}\right)^{K(a+1)} q_p^{a+1}\left[0+4e^{\frac{-K(a+1)t_b}{nt_p}}+2e^{\frac{-2K(a+1)t_b}{nt_p}}+...+4e^{\frac{-(n-1)k(a+1)t_b}{nt_p}}+e^{\frac{-nK(a+1)t_b}{nt_p}}\right] \quad (4.65)$$

Further simplifying:

$$k = \frac{1}{\frac{1}{3n}\left(\frac{t_b e}{nt_p}\right)^{K(a+1)} q_p^{a+1}\left[0+4e^{\frac{-K(a+1)t_b}{nt_p}}+2e^{\frac{-2K(a+1)t_b}{nt_p}}+...+4e^{\frac{-(n-1)K(a+1)t_b}{nt_p}}+e^{\frac{-nK(a+1)t_b}{nt_p}}\right]}\frac{Y}{t_b} \quad (4.66)$$

Replacing k in equation (4.58) and simplifying, an equation for sediment inflow rate is:

$$q_{si}^t = \frac{C'^{(a+1)}}{\frac{3600}{3n}\left(\frac{t_b.e}{nt_p}\right)^{k(a+1)}\left[0+4e^{\frac{-k(a+1)t_b}{nt_p}}+2e^{\frac{-2k(a+1)t_b}{nt_p}}+...+4e^{\frac{-(n-1)k(a+1)t_b}{nt_p}}+e^{\frac{-nk(a+1)t_b}{nt_p}}\right]}\frac{Y}{t_b} \quad (4.67)$$

If $$sed_f = \frac{C'^{(a+1)}}{\frac{3600}{3n}\left(\frac{t_b e}{nt_p}\right)^{k(a+1)}\left[0+4e^{\frac{-k(a+1)t_b}{nt_p}}+2e^{\frac{-2k(a+1)t_b}{nt_p}}+...+4e^{\frac{-(n-1)k(a+1)t_b}{nt_p}}+e^{\frac{-nk(a+1)t_b}{nt_p}}\right]} \quad (4.68)$$

then:

$$q_{si}^t = sed_f \frac{Y}{t_b} \quad (4.69)$$

where sed_f is sediment load rate adjustment factor, q_{si}^t is sediment load rate (lb/sec), Y is sediment load in (lbs), t_b is duration of storm (hr), a is a coefficient that varies from 0.5 to 1, C' is a peak discharge correction factor less than or equal to 1 given by equation (4.27) . Equation (4.69) can be used to estimate the sediment inflow rate into the VFS. The sediment load rate determined using the above equation has similar trend with an average sediment load rate, total sediment load over the duration of storm, Y/t_b.

The incoming coarse material load rate per unit width is given by :

$$q_s^c = \frac{q_s(1 - f_{ri}^1)}{L} \tag{4.70}$$

where L is filter strip width. Using equation (4.70), the fraction of sediment trapped in the deposition wedge is given by:

$$f = \frac{q_s^c - q_{sd}^c}{q_s^c} \tag{4.71}$$

where q_{sd}^c is the bed load transport capacity given by equation (4.52).

Predicting the Advance of the Deposition Wedge

The deposition wedge advances down the slope as sediment is deposited at the downstream edge as shown in Figure 4-1. The filter strip length changes with time as sediment gets deposited and this causes a drop in trapping efficiency. Therefore trapping efficiency must be corrected for this effect. The correction is made by calculating the advance distance x (t) and the corrected length L_f shown in Figure 4-1. This calculation is based on mass balance in zone B shown Figure 4-1. The incoming sediment load to zone B is that coming into the filter, q_{si}, minus that which is deposited in zone zone A. The sediment load leaving zone B is equal to transport capacity of zone B, q_{sd}. Hence, assuming that there is no deposition in the upstream delta, the average sediment load on the deposition wedge q_{sba} should be the average of q_{si} and $q_{sd,}$ or:

$$q_{sba} = \frac{q_{si} + q_{sd}}{2} \tag{4.72}$$

and the trapping efficiency is given by:

$$f = \frac{q_{si} - q_{sd}}{q_{si}} \tag{4.73}$$

A correction factor is discussed later for the time period when deposition is occurring in the upstream delta.

An equilibrium slope is calculated based on sediment transport rate equation (4.52) and (4.53) and continuity equation. The equilibrium slope is the slope required for the flow to transport the sediment load q_{sba}. Equation (4.52) which is calibrated for Einstein's bed load equation can be used to solve for the equilibrium slope S_{et}. It is written here using parameters for equilibrium slope, or:

$$q_{sba} = \frac{K(R_{sba}S_{et})^{3.57}}{d_{pba}^{\ 2.07}} \tag{4.74}$$

$$q_{wadj} = \alpha d_f^{(\beta+1)} S_{et}^{\ 0.5} \tag{4.75}$$

which is a modified form of Manning's equation. Also a simplified relation between hydraulic radius and flow depth can be given by:

$$R_{sba} = a_R d_f^b \tag{4.76}$$

where a and b are empirical parameters for each grass species. Paramteres a_R and b were determined for each grass species. For the recommended grass species the values are similar as shown in Table 4. Hence the data were rerun in SAS to get parameteric values applicable to all the grass of the grass types in Table 4.8 and the values for a_R and b are 0.0516 and 0.3670 respectively.

Table 4-8. Parameter values for for computing hydraulic radius $(R_s = a_R\, d_f^b)$.

Vegetation type	a_R	b
Bermudagrass	0.0376	0.3141
Ryegrass	0.0491	0.3566
Tall fescue	0.0455	0.3443
Kentucky bluegrass	0.0461	0.3462
Blue grama	0.0461	0.3462
Centipedegrass	0.0376	0.3141
Buffalograss	0.0428	0.3346
Grass mixture	0.0659	0.4064
Alfalfa	0.0999	0.4821
Vegetation not typically recommended for VFS		
Sericea lespedeza	0.1338	0.5390
Common lespedeza	0.1940	0.6154
Sudangrass	0.32140	0.4821

Equation (4.76) gives results very close to equation (4.18) with an R^2 value equal to 0.96 for a shallow flow depth recommended for VFS. The hydraulic radius at equilibrium slope is computed by solving equations (4.74) to (4.76) simultaneously. In the original GRASSF equations (4.18) and (4.19) were used for computing hydraulic radius and discharge and R_{sb} can only be determined by trial and error. In SGRASSF the simplification makes it possible to get an explicit solution for equilibrium hydraulic radius and slope.

S_{et}, the equilibrium slope required to transport q_{sba} can be calculated from equation (4.74) if R_{sba} is determined. Equations (4.74) to (4.76) can be simultaneously solved to obtain:

$$R_{sba} = \left[\left(\frac{q_{sba}}{K} \right)^{0.28} \left(\frac{\alpha}{q_{wadj}} \right)^{2} d_{pba}^{0.5798} \left(\frac{1}{a_R} \right)^{\frac{2(\beta+1)}{b}} \right]^{\frac{b}{b-2(\beta+1)}} \qquad (4.77)$$

Using the value for R_{sba}, equilibrium slope can be calculated from:

$$S_{et} = \left(\frac{q_{sba}}{K} \right)^{0.28} \frac{d_{pba}^{0.5798}}{R_{sb}} \tag{4.78}$$

Details of the derivation are given in Appendix (B-2). The R_{sba} value from the above equation is close to the value obtained using the original iterative method. The result is shown in Figure (4.5).

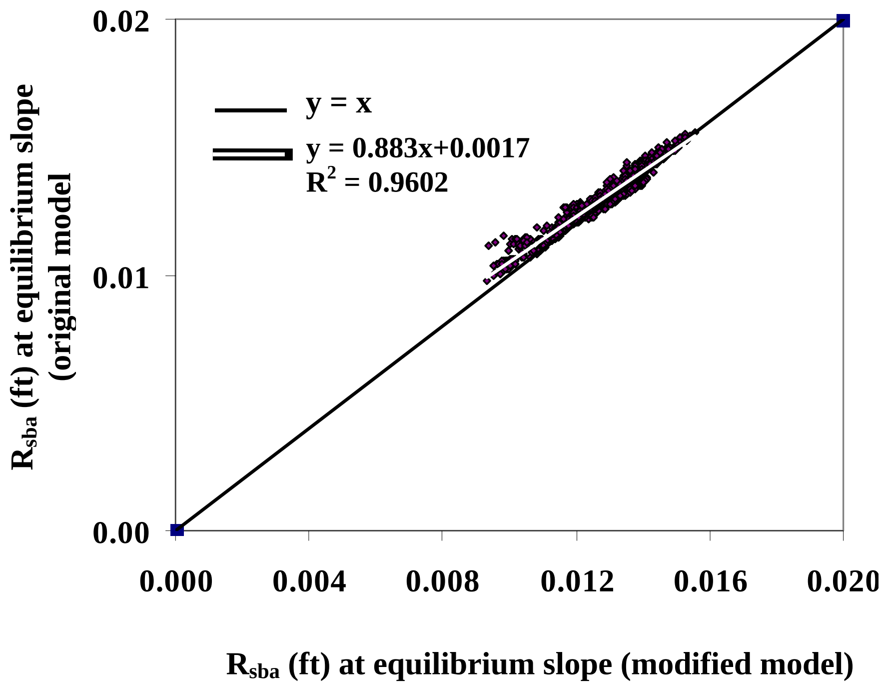


Figure 4-5. Comparison of hydraulic radius at equilibrium slope using original model and modified.

Figure (4-6) shows comparison of equilibrium slopes using the original method and the modified procedure given by equation (4.78). Determining equilibrium slope enables the definition of the geometry of the deposition wedge. Based on the geometry of the deposition wedge and the mass of sediment deposited, the advance of the deposition wedge is determined and the trapping efficiency is corrected for change in filter length.

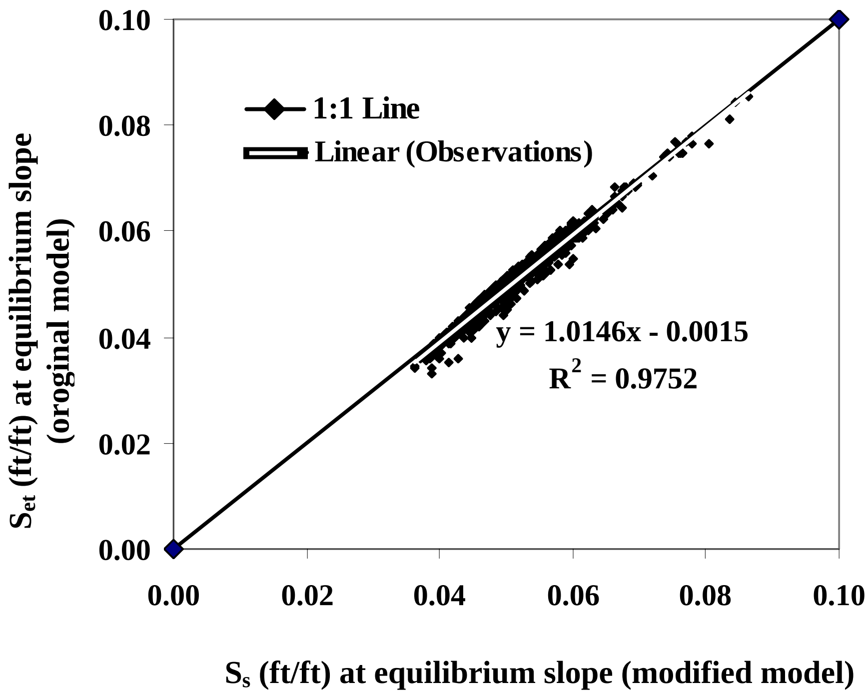


Figure 4-6. Comparison of equilibrium slope using original and modified model.

The volume of sediment deposited during each storm is calculated based on the incoming sediment load and bulk density of soil. The total volume deposited (ft^3) is given by:

$$V_{tot} = \frac{f\ (M_{sed})\ (1 - f_{ri}^{1})}{\gamma_{sb}} \tag{4.79}$$

where M_{sed} is sediment load (lb), and $_{sb}$ is bulk density of the material deposited (lb/ft^3).

The location of the leading edge of the deposition wedge x(t) (See Figure 4.1) can be determined by sediment mass continuity relationships. Tollner et al. (1976) derived equation for depth of deposition and the advance distance x(t) based on simple mass continuity. Two different equations are used to calculated x(t) depending on whether depth of deposition is smaller that or equal to the height of the grass media:

$$Y_f(t_f) = \begin{vmatrix} \left(\dfrac{2}{\gamma_{sb}} f' q_{si} S_e (t_f - t_i) + Y_i(t_i)^2 \right)^{1/2} & Y_f(t_f) < H \\ H & Y_f(t_f) = H \end{vmatrix} \tag{4.80}$$

$$X_f(t_f) = \begin{vmatrix} \left(\dfrac{2}{\gamma_{sb}} \dfrac{f' q_{si}}{S_e} (t_f - t_i) + X_i(t_i)^2 \right)^{1/2} & Y_t(t_f) < H \\ X_i(t_i) + (t_f - t_i) \dfrac{f' q_{si}}{H \gamma_{sb}} & Y_t(t_f) = H \end{vmatrix} \tag{4.81}$$

where H is height of media, $_{sb}$ is the bulk density of the deposited sediment ,$Y_t(t_f)$ and $X_t(t_f)$ are the depth of deposition and advance distance respectively at time t_f.

The correction for deposition in the upstream delta is only made prior to the height of deposition reaching the grass height. This requires the calculation of the ratio between sediment deposition in the upstream delta, V_1, to that in zone B(t), V_2 is given by:

$$\frac{V_2}{V_1} = \frac{S_{et} - S_c}{S_c} = r \tag{4.82}$$

In the above equations, if f is the total fraction of coarse particles trapped in the deposition wedge, then the portion trapped in zone B would be:

$$f' = f\left(\frac{r}{1+r}\right) \tag{4.83}$$

where r is given by equation (4.82). f' is used if $Y_t(t_f)$ less than H since only part of sediment flows into zone B and f is used when the depth of deposition reaches the height of the grass media, i.e. if $Y_t(t_f)$ is equal to H, then f' is replaced with f. If the initial advance distance of the deposition wedge from the previous storm is $x_i(t_i)$ the net advance after each storm is given by:

$$\Delta x = x_f(t_f) - x_i(t_i) \tag{4.84}$$

Accounting for Grass Recovery

The filter strip length changes with time as sediment gets deposited causing a drop in trapping efficiency. However, the grass may recover during the growing season. The length of time required for grass recovery after burial by sediment will be a function of variables associated with rainfall, runoff, vegetation growth rate and depth of sediment accumulation. If D represents the number of days required for recovery, and L_0 is the total length of filter strip, the following relation was developed to estimate the net filter strip length L_n for each day in the season used to calculate the trapping efficiency considering both advance of the deposition wedge and recovery for each day during the growth period, or:

$$L_n = L_{n-1} - \text{Advance of the deposition wedge} + \text{Recovery} \qquad (4.85)$$

where L_n is effective length of filter strip on a given day and L_{n-1} is length of filter strip on the previous day. The D value depends on the grass type. A report on rotational grazing by (Henning et al., 2000) indicates that switchgrass requires 30 to 45 rest days before grazing it again to recover after first round grazing. Another study on the design of filter strips to trap sediment and nutrients by (Prosser and Karssies, 2001) made an observation on filter renewal by vegetation germination and growth on and through the trapped sediment. They observed that in warm climates, dense grass cover re-establishes within three months of its burial. Like other warm-season grasses, switchgrass is noted for its heavy growth during summer season. There are no procedures accounting for grass recovery after inundation. A relationship accounting for the effect of grass recovery was developed based on assumed linear plant recovery shown in Figure 4.7. Details of the derivation are given in Appendix (B-3).

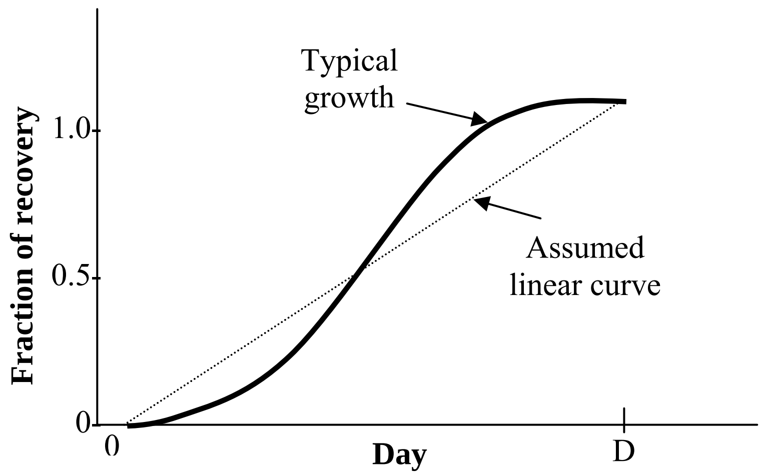


Figure 4-7. Plant growth characterstic curve.

During the growing season, both deposition and recovery are expected and the net length of filter strip given in equation (4.85) can be re-written as:

$$L_n = L_{n\text{-}1} - (\Delta X_1 + \Delta X_2 + ... + \Delta X_n) + \frac{1}{D}\left[(n-1)\Delta X_1 + (n-2)\Delta X_2 ... + \Delta X_{n\text{-}1}\right] \quad (4.86)$$

for n less than D and:

$$\begin{aligned} L_n = L_{n\text{-}1} - (\Delta X_1 + \Delta X_2 + ... + \Delta X_n) + (\Delta X_1 + \Delta X_2 + ... + \Delta X_{n\text{-}D+1}) \\ + \frac{1}{D}\left[(D-1)\Delta X_{n\text{-}D+2} + (D-2)\Delta X_{n\text{-}D+3} ... + \Delta X_{n\text{-}1}\right] \end{aligned} \quad (4.87)$$

for n greater than or equal to D where Δx_i is the advance in the deposition wedge after each day and D is the number of days required for grass recovery.

During the winter season for warm season grasses, there will be no grass re-growth. So for the dormant season L_n is given by:

$$L_n = L_{n\text{-}1} - (\Delta X_1 + \Delta X_2 + ... + \Delta X_n) \quad (4.88)$$

Advance in the deposition wedge ΔX_i occurs only on those days with significant storm to cause runoff and sediment yield, otherwise ΔX_i is zero.

An example of the computation for recovered length can be calculated if it is assumed that D=10, then the effective length of filter strip on a 9th day is, then $L_9 = L_8 - (\Delta X_1 + \Delta X_2 + ... + \Delta X_9) + \frac{1}{10}\left[9\Delta X_1 + 8\Delta X_2 ... + \Delta X_9\right]$. The advance of the deposition wedge on the 9th day, for example, is the sum of all the deposition length lost to sediment inundation during the 9 days, which is equal to $\Delta X_1 + \Delta X_2 + ... + \Delta X_9$. The advance on any day is zero if there is no storm. The recovery on 9th day is $\frac{1}{10}\left[9\Delta X_1 + 8\Delta X_2 ... + \Delta X_9\right]$. Note that the recovery rate on day 9 is 90% for part of VFS inundated on day 1, 80% for part of VFS inundated on day 2, and only 10% for part of VFS inundated on day 9.

Model Validation

The modified VFS model, SGRASSF, was tested using an input data different from the data used in the determination of the parameters and was compared again with SEDIMOT III trapping efficiency. Since the representative particle diameter and fractions of fine and coarse materials in the eroded material affect trapping efficiency, different soil types were considered in the validation, namely sandy loam, loam, sandy clay loam and clay loam representing hydrologic soil groups A, B, C, and D, respectively. The discharge per unit width and filter length also affect trapping efficiency. The discharge was varied by varying the precipitation level and area contributing to the flow. The filter length was also varied. The same data range given in Table 4-4 was used but with a new set of data points.

Figure 4-8 shows comparison of trapping efficiency using GRASSF routine and modified VFS model, SGRASSF. The trapping efficiency from SGRASSF is correlated to SEDIMOT trapping efficiency values with R^2 value equal to 0.92. The Nash-Sutcliffe efficiency was used as an indicator of goodness of fit. The Nash Sutcliffe coefficient obtained is 0.9.

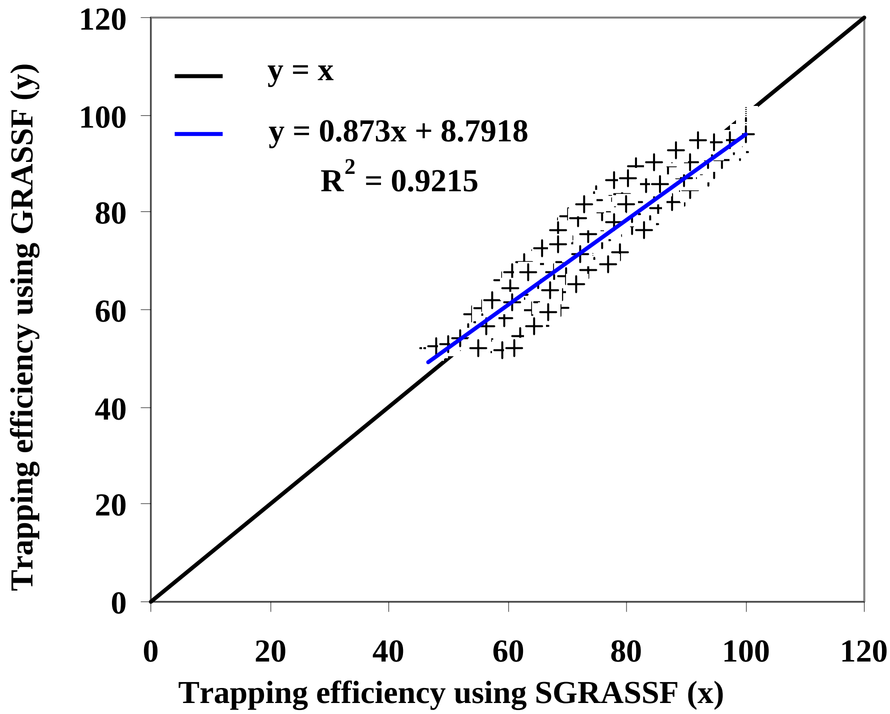


Figure 4-8. Comparison of trapping efficiency using SEDIMOT with GRASSF and SGRASSF.

Summary and Conclusion

A simplified procedure SGRASSF was developed to compute sediment trapping efficiency using vegetative filter strips based on Kentucky filter strip model GRASSF. A flow chart for procedures for loading and equations used in sediment trapping in VFS is given in Figure 4-9 below. In the original GRASSF, implicit equations were routinely used for many processes. In SGRASSF, explicit equations were developed for average overland flow velocity as a function of depth of flow, slope and grass type, average depth of flow based on continuity equation, trapping in the deposition wedge, advance of deposition wedge, infiltration in VFS, and grass recovery. Summary of equations and procedures used to calculate trapping efficiency is given in Appendix B-1. As in the original GRASSF the impact of flow, infiltration and sediment properties are predicted. These properties include flow depth, velocity, sediment particle size distribution, width of filter strip, density of vegetation, height of vegetation and slope.

The calibrated Manning's equation is used for overland flow velocity in GRASSF. The overland flow velocity in SGRASSF was described as a function of depth of flow and slope for a given vegetation type. The results from the modified relationships were similar to original model with an R^2 value equal to 0.98. This enabled development of an explicit relationship for flow depth. The GRASSF routine uses storm hydrographs to calculate the depth of flow. In SGRASSF peak discharge is used. The advantage of using peak discharge is that it can readily be computed using TR 55 or rational method. However, if the peak discharge is directly used, the computed depth of flow will be much greater than that occurring during most of the storm and the trapping efficiency will be underestimated. To adjust the trapping efficiency to the one that is computed using GRASSF, the peak discharge should be adjusted using a correction factor. The correction factor was found to be well related to the peak discharge per unit length (perpendicular to the flow).

The effect of advance of the deposition wedge on filter length and trapping efficiency as a result of sediment inundation and grass recovery during the growth period was taken into account. The implicit relations used in computation of equilibrium hydraulic radius and slope of the deposition wedge used to calculate the advance of the deposition wedge are replaced by simple explicit relationships with R^2 value equal to 0.97 for equilibrium slope.

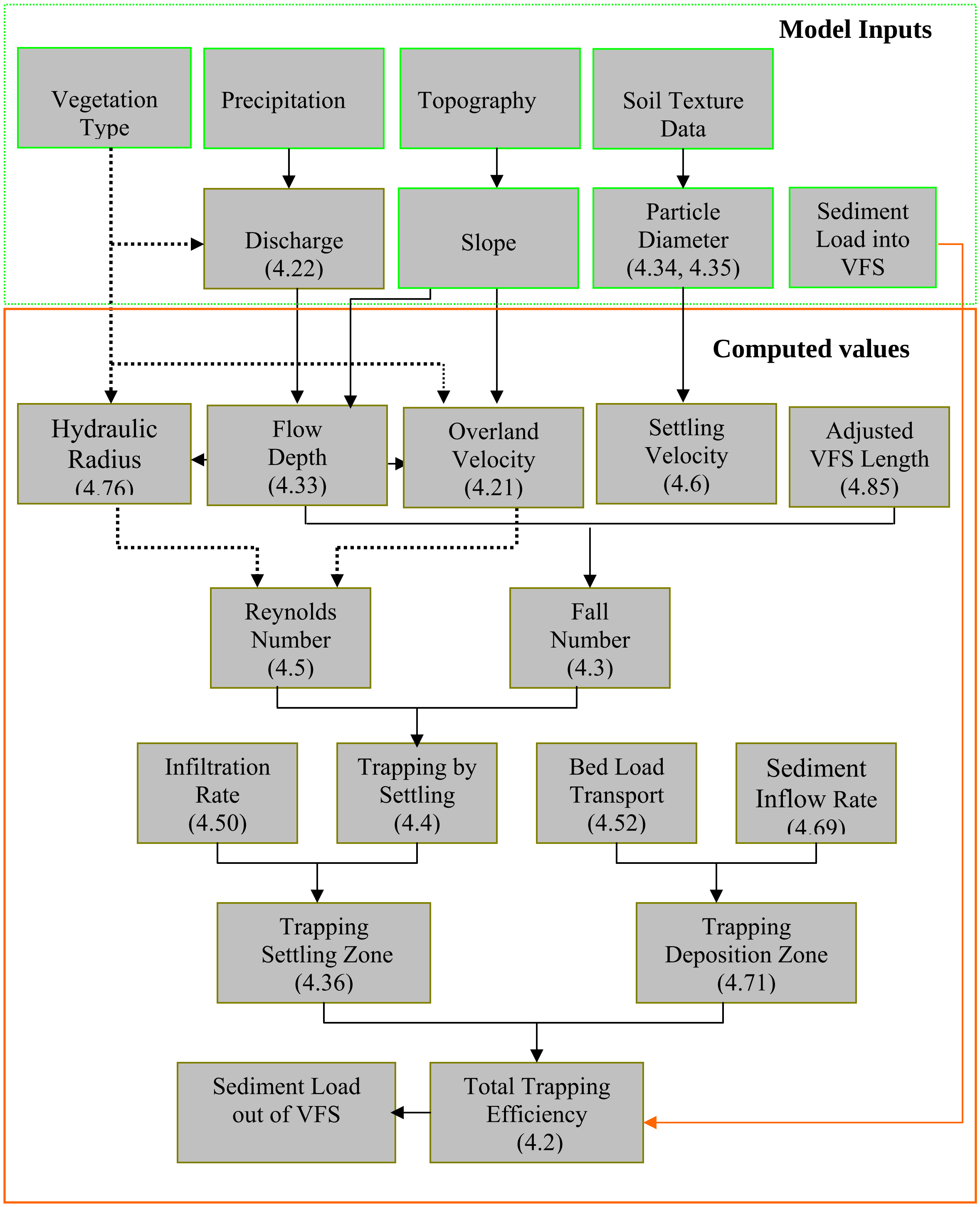


Figure 4-9. Flow chart of procedures for calculating sediment trapping in VFS.

SGRASSF gives similar trapping efficiency results to GRASSF in SEDIMOT III with an R^2 value equal to 0.92. The Nash-Sutcliffe coefficient, used as an indicator of goodness of fit of the simplified model to the original model, was determined to be 0.9. The modified model can be use to calculate sediment trapping from multiple sub-watersheds using the daily precipitation data, sediment yield data and subwaterhed characteristics such as soil type, vegetation cover, slope and size.

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Chapter V

COMPARISON OF ENVIRONMENTAL AND ECONOMIC BENEFITS OF EROSION CONTROL METHODS: VEGETATIVE FILTER STRIPS AND OPTIMAL LAND USE SYSTEMS

Abstract

Vegetative filter strips (VFS) can be placed between agricultural crop lands and environmentally sensitive areas and reduce pollutant loads, dependent on the VFS area and other parameters. Watershed income varies with the size of the VFS since the VFS will replace part of each land unit, which may or may not be harvested, or if harvested may generate lower income compared to the agricultural crop. As an alternative approach for reducing pollutant load, land cover distribution over the watershed could be planned in economically and environmentally efficient manner using the LUDM model discussed in Chapter III such that watershed income is maximized while maintaining sediment load at a desired level. The Soil and Water Assessment Tool (SWAT) was used to estimate sediment load in both approaches. A comparison has been made between these two methods based on effectiveness in removing pollutants versus potential income.

Further analysis was made to compare the relative environmental and economic benefits of placing vegetative filter strips selectively on the watershed based on sediment yield. The comparison was made between placing VFS in all the fields and placing VFS on 75%, 50% and 25% of the fields with higher sediment loads. For very low allowable

levels of sediment load the optimal replacement approach is more effective compared to VFS approach, especially if VFS is not harvested. In both cases, the income from the watershed changes with change in the amount of sediment allowed to leave the watershed, again at a very low allowable sediment load levels, the rate of change in income with respect to allowable sediment load is lower for the optimal replacement approach than it is for the VFS approach.

The environmental and economic benefits of using constant width of VFS along the flow across all land units were compared with variable VFS width. It was found to be slightly more beneficial to vary the width of the VFS in proportion to the size of the field contributing to the sediment inflow, however, the difference was small. In addition, the effect of harvesting the VFS was compared to paying incentives to landowners. The results show that producers could obtain more income from harvesting the VFS than they would earn from the water quality incentive program while maintaining the same level of sediment load. The results indicate that conservation expenditures could be reduced without sacrificing the environmental benefits through managed harvesting of conservation crops such as switchgrass used as vegetative filter strips.

Introduction

Erosion and nutrient control plans may reduce farmers' net income. This paper was written to examine the premise that incorporating spatial information into non-point source pollution (NPS) control policies can help target critical areas within the watershed and reduce NPS control costs. A number of studies have developed targeting procedures to enable watershed-specific evaluation of NPS pollution control. An early economic analysis of environmental targeting indicated that the first environmental benefit index (EBI) substantially increased environmental benefits relative to costs, compared with the program's original, erosion-based design (Osborn, 1993). A more recent study shows that moving to environmental targeting provided a $370-million/year increase in CRP benefits with program acreage and costs virtually unchanged (Feather et al., 1999). Additionally, spatial variability at the field and farm level has been shown to be an important aspect of effective targeting (Bricker et al., 1999)

The public downstream of the source area obtains benefits through reduced sediment flows, improved stream water quality, additional fish and wildlife habitat, and better scenery along streams. The buffer/filter area may be natural, undeveloped land where the existing vegetation is left intact, or it may be land planted with vegetation. Its purpose is to protect streams and lakes from pollutants such as sediment, nutrients and organic matter. Filter strips can also provide several benefits to wildlife, such as travel corridors, nesting sites and food sources (Ohio State University Extension, 1999).

Sediment and Nutrient Removal

Extensive literature is available on VFS; including primary sources such as: Wilson (1967), Young et al. (1980), Hayes et al. (1984), Dillaha et al. (1989), Daniels and Gilliam (1993), Robinson et al. (1996), and Patty et al. (1997). The experiments reported deal with vegetative filter strips with lengths of flow path ranging from less than one meter to more than thirty meters, slopes ranging from 2 to 16%, and various types of grasses and pollution load. Performance of the VFS in treatment of runoff in these studies was typically evaluated based on comparing pollutant concentrations in runoff samples at the inlet and outlet of the VFS. If properly installed and maintained, VFS have been shown to have the capacity to remove up to 75% or more of the sediments and sediment-bound pollutants from cropland runoff.

While sediment-removal studies are abundant, research studies that have dealt with P removal in VFS are very limited and the sparse results are somewhat contradictory. In a VFS field experiment, Dillaha et al. (1987) found that total P removal was closely related to sediment removal when runoff had high particulate P concentration. They found that P removal efficiency in 4.6-m-long filters varied from 49% to 73%, while corresponding sediment removal was slightly higher at 53% to 86%. Longer filters of 9.1 m were more efficient, with P removal ranging from 65% to 93% and sediment removal ranging from 70R to 98%. In the Dillaha et al (1987) study more than 90% of the total phosphorus content was sediment bound. Another study (Magette et al., 1989) reported that VFS were less efficient in P removal compared with that of sediment removal. They found that the average total P removal for the 4.6- and 9.1-m-long filters was only 27 and 46%, respectively. The corresponding sediment removal efficiencies for the same study were 66% and 82%, respectively. In a two-year VFS study under natural rainfall conditions, Daniels and Gilliam (1993) found that 6-m-long filters retained, on average, 60% of the total P load, and retained about 50% of the soluble P load. Many other studies have suggested that infiltration is the primary mechanism of P removal, especially for runoff with high soluble P content such as runoff from land area receiving manure applications (Overcash et al., 1981; Chaubey et al., 1994; Srivastava et al., 1996, Barfield et al., 1998). All of the studies above were on common lawn and pasture grasses.

In this study switchgrass has been used as grass filter which has also been shown to have a very good potential as filter media. For instance, studies by Lee (2000) on sediment and nutrient trapping showed that switchgrass VFS removed 97% of the sediment, 94% of the total-N, 85% of the NO3-N, 91% of the total-P, and 80% of the PO4-P in the runoff. VFS needs to be wider when slopes become steeper, because the velocity of surface runoff increases. In hilly topographies VFS appeared ineffective because flow was more concentrated following specific routes (Dillaha et al., 1989).

Effect of VFS Width

As the width of the VFS perpendicular to the flow increases VFS effectiveness increases. Dillaha et al. (1989) used grass VFS widths of 4.6 and 9.1 m with 70% and 84% of the sediment being removed, respectively. Barfield et. al (1998) also conducted studies on the effective width of natural riparian grass buffer strips in removing sediment, atrazine, nitrogen and phosphorus from surface runoff. No-till and conventional-tillage erosion plots served as the sediment and chemical source area. Runoff from the plots was directed onto 4.57, 9.14, and 13.72 m filter strips where the inflow and outflow concentrations and flow rates were measured. Trapping percentages for sediment and chemicals typically ranged above 90%. The analysis showed that most of the chemicals were trapped by infiltration into the soil matrix and that trapping efficiency increased with filter strip length and with fraction of water infiltrated. Wilson (1967) found an inverse relationship between width of the VFS and the maximum deposition of a particle size of a given diameter. A study conducted by Patty et al. (1997), investigated the removal of soluble P load in VFS with 12 filters with lengths of 6, 12, and 18 m under natural rainfall conditions. They found that the average soluble P removal was 40%, 52%, and 87% for lengths of 6, 12, and 18 m, respectively. Corresponding average sediment load removal was 92%, 98%, and 99%, respectively. The studies above show that the relationships between width and trapping efficiency are not linear. The existence of no evident linear relationship between length along the flow path and sediment reduction shows that other factors are also very influential. However, for a specific site, the length along the flow path of the VFS required to trap a given fraction of sediment is closely related to the ratio of sediment source area length to the VFS length.

Effect of Sediment Inundation

Since coarse sediment deposits in the upstream edge of the VFS, the filter strip width eventually changes with time (Haan et al. 1994). This results in a decrease in the effective length along the flow path along with a drop in trapping efficiency. However the grass may recover during the growing season. Since some guidance to understanding the potential of recovery from inundation by sediment can be found in response to flooding. Gamble and Rhoades (1964) investigated various grasses along a reservoir shoreline for tolerance to inundation by flood. The studies showed that factors such as duration and depth of submergence, season and frequency of flooding are associated with satisfactory survival of grasses when inundated. The grasses studied showed variable tolerance to inundation associated with shoreline fluctuation. Switchgrass was found to be one of the grasses grouped under strong tolerance for flooding (10 to 20 days). When used within field filter strips, the chance of grass being inundated by water for several days is low compared to buffer strips along creeks and lakes. For relatively strong tolerant crops damage due to inundation by flood may not be a problem; however, ineffectiveness of vegetative filter strips can occur from sediment inundation. For example, Dillaha et al. (1989) saw a decrease in effectiveness of the VFS up to 39% from the first to the second simulation runoff event. The effect of sediment inundation in trapping depends on the length of the filter strip along the flow path. If the filter strip has long flow path, the change in flow path length due to deposition might not have sufficient effect to cause a significant change in trapping efficiency.

The effectiveness of the VFS was shown to decrease in the Dillaha et al (1989) studies when the surface runoff water level exceeds the height of the grasses. This makes it important to choose grass species that recover fast from inundation and with a relative tall height. The Dillaha et al (1989) study was conducted on a research plot with high rainfall amount and intensity. In a real world, this may or may not be a long term problem because filter strip vegetation should be able to grow through most sediment accumulations. The success of VFS in surviving burial by sediment will be a function of variables associated with rainfall, runoff, vegetation growth rate and depth of sediment accumulation, all of which are stochastic in nature.

Buffer Strips

Previous studies on placement of vegetation for controlling sediment have usually been on buffer strips. Alliance for the Chesapeake Bay (1996) found that spatial placement of buffer strips within a watershed can have profound effects on water quality. Riparian buffers in headwater streams (i.e., those adjacent to first, second and third-order systems) have much greater influences on overall water quality within a watershed than those buffers occurring in downstream reaches. Downstream buffers have proportionally less impact on polluted water already in the stream Buffer strips along larger rivers and streams cannot significantly improve water that has been degraded by improper buffer practices higher in the watershed. Many US Army Corps of Engineers projects occur along the higher order streams and rivers and have little or no control over water quality resulting from land-use practices higher in the watershed (Richard et al., 2000). Although the buffer strips along these larger systems are typically not effective in controlling channel sediment loads, they tend to be longer and wider than low-order systems, thus potentially providing significant wildlife habitat and movement corridors.

Buffer strips at lower elevations of fields intercept surface runoff water from crop fields. These might be ordinary grassed fence rows that runoff water crosses as it leaves fields, or strips of grasses, shrubs, and trees lining the banks of streams. Since these areas often have fewer slopes than waterways they can be more effective to remove sediments (Regehr et al., 1996).

The Agricultural Non-Point Source Pollution Model (AGNPS) was used to determine locations of vegetative buffer strip effectiveness on reducing sediment load within the East Bad Creek (EBC) watershed, a 690 ha agricultural watershed located mid Michigan (Vennix and Northcott, 2004). The placement of buffer strips within the watershed was prioritized on three different scales. The reduction of sediment due to buffer strips was analyzed on a stream segment level, a field boundary level, and on a cell-by cell basis. The stream segment buffers and field buffers were ranked on their overall ability to reduce sediment load into the stream. The reduction in sediment yield from the stream segments (along field drains) and the fields varied from 3.49 to 58.54

tons and 0 to 19.31 tons respectively. The cell-by-cell evaluations highlighted specific critical areas of buffer efficiency on a 30-meter resolution where the stream segment and field evaluations identified specific stream segments and fields to target for buffer placement.

Alternative Approaches to Limiting Sediment Load

Land cover distribution over the watershed using total replacement of row crops with dense cover grass crops on selected hydrologic units can be planned in an economically and environmentally optimal manner such that watershed income is maximized while maintaining sediment load at a desired level. Alternatively, vegetative filter strips can be placed along field drains to trap sediment and nutrients which have already been displaced by sheet and rill erosion. These are two contrasting approaches. In the first approach, optimal land distribution approach, the amount of sediment generated is minimized by assigning environmentally sensitive areas to less erodible cover types in contrast to the second approach which uses vegetative filter strips to trap sediment which has already been displaced. The distribution of the land cover over the watershed affects the rate of soil erosion. Sediment load can be reduced if the highly erodible parts (hilly slopes) of the watershed are assigned to grasses or forest and less erodible (flat slopes) are used for conventional agricultural crops. Targeting hilly areas to less erosive cover types also has economic advantage because hilly areas are usually marginal and less fertile. Therefore in the optimal land distribution approach the ultimate goal is to find out the land use pattern over the watershed that is most advantageous from both economic and environmental perspectives. In Chapter III, an optimization approach was discussed that makes it possible to determine the distribution of land cover types over the watershed that maximizes watershed income while keeping the sediment load to the streams below some specified value for maintaining water quality. Since the producer's tendency is to grow more profitable crops that may result in water quality problems regardless of environmental outcomes, it is important to find ways to minimize the sediment load level while putting most of the land into profitable crops.

Alternatively, vegetative filter strips can be placed between cropland, grazing land, or disturbed land and environmentally sensitive areas with the purpose of reducing sediment, particulate organics and sediment adsorbed contaminant loads in runoff. Because of their potential environmental benefits, filter strips are recommended by a number of state and federal agencies as an urban and agricultural best management practice (Ohio State University Extension, 1994). By using vegetative filter strips pollutants can be trapped while letting most of the flow go through to sustain river ecology. Although dissolved substances still have the chance to flow through, sediment bound materials could be effectively reduced.

There has been growing emphasis on improving the targeting or selection of land uses to put in the retirement program and to make it cost effective. The Targeting process is being continually improved to make it more cost effective. Details of targeting are discussed in Chapter II and III.

Hence, the alternatives are: 1) Total replacement of selected land units by siwtchgrass 2) Continuation of row crop production with VFS along field drains with water quality incentives and 3) Same as 2 without incentives but switchgrass is harvested and sold.

Objective

The objective of the study is to compare the effectiveness of optimal land distribution or replacement approach to the placement of vegetative filter strips along field drains.

Methodology

The paper examines the effect of varying VFS width on sediment trapping and on the opportunity cost of the lost agricultural income due to replacement of crops with VFS. This approach is compared to total replacement approach discussed in Chapter III.

A 2,800 acre watershed within Fort Cobb basin subdivided into 271 subbasins was delineated within the Fort Cobb basin to compare the cost effectiveness of two different approaches. The same parameters used in Fort Cobb watershed calibration discussed in Chapter III were used in the smaller watershed. Sediment and nutrient load for each sub-basin was generated by Soil and Water Assessment Tool (SWAT). Output data from SWAT was used as input.

The two approaches compared are:

1. Optimal land distribution on the watershed to maximize income from watershed subject to sediment load constraint. This involves cost effective total replacement of parts of the watershed by switchgrass. In this case the LUDM programming model written using GAMS assigns land cover type for each sub-basin such that the income from the watershed is maximized while meeting an allowable level of sediment load to the streams using the LUDM non-uniform model described in

Chapter III. The land cover types compared are conventional and minimum tillage wheat, sorghum, peanuts and switchgrass.

2. The use of vegetative filter strips along field drains to trap sediment load from each sub-basin. The simplified VFS trapping efficiency procedure described in Chapter IV, SGRASSF, was used to estimate the sediment trapping efficiency. The sediment yield from each sub-basin was calculated for each day from 1991 to 2000 and the total annual sediment yield was calculated by adding the sediment yield for each day. The sediment yield from all sub-basins for a given year is given by:

$$\sum_{i=1}^{365}\sum_{j=1}^{271}(1-TE_{ij})Sed_{ij} \tag{5.1}$$

where TE_{ij} is the trapping efficiency on jth field on the ith day.

A 10 year simulation period with two years for warm was used to calculate average sediment yield to compare the sediment yield from the two approaches. The 10 year period of simulation was chosen after comparing the results with a longer 20 year simulation period. The comparison is presented in Chapter III. The SWAT model was used to generate sediment flowing into the VFS. For the VFS approach sediment trapped in the VFS was deducted to obtain the final sediment yield from each sub-basin. Switchgrass was used as filter media and the VFS parameters used to calculate trapping efficiency using SGRASSF, described in Chapter IV, were chosen based on switchgrass as filter media. To consider the effect of the deposition wedge, it was assumed that switchgrass re-establishes at the beginning of the growing season following dormancy.

Traping efficiency was calculated using equation (4.2) in chapter IV. Equations (4.86) to (4.88) are used to estimate the effective length of trapping on a given day. In equation (4.86) to (4.88), D is the number of days required for grass recovery. The higher the D value used the smaller the effective trapping length and the lower the trapping efficiency. If the number of days between significant storms is more than the time required for complete recovery, the effect of deposition is almost negligible.

The D value depends on the grass type. A report on rotational grazing by Henning et al. (2000) indicates that switchgrass requires 30 to 45 rest days before grazing it again to recover after first round grazing. Another study on the design of filter strips to trap sediment and nutrients by Prosser and Karssies (2001) made an observation on filter renewal by vegetation germination and growth on and through the trapped sediment. They observed that in warm climates, dense grass cover re-establishes within three months of its burial. Like other warm-season grasses, switchgrass is noted for its heavy growth during summer season. Switchgrass is planted in mid April and harvested in July. Oklahoma statistics data shows that this period is also a period during which much of the rainfall in the year is expected, therefore, the highest sediment inundation and recovery is expected during this period. During the winter season, there will be no grass re-growth for warm season grasses, hence equation (4.88) has been used which accounts for deposition only.

The watershed income for varying allowable sediment load levels was calculated. Sediment load and trapping in VFS were calculated with simplified grass filter model, SGRASSF discussed in chapter IV. Two cases were evaluated, one in which VFS is harvested and another in which VFS is not harvested. The allowable sediment yield and corresponding watershed income from VFS approach are compared with the sediment yield and income from the optimal land use approach in which land use decision for each sub-basin was made to maximize income while meeting the sediment load requirements.

An analysis was also made to compare the relative environmental and economic benefits of putting vegetative filter strips selectively on the watershed. The selection of sub-basins for installing VFS was based on sediment yield. The comparison was made between placing VFS in all the fields to placing VFS on the 75%, 50% and 25%% of the fields with higher sediment loads. The effect of harvesting the VFS on sediment load and watershed income was also compared with sediment load and watershed income for the case in which the VFS is not harvested. The environmental and economic benefits of using constant width VFS across all land units was compared with variable width VFS.

Calculating VFS Width

In this section, VFS width refers to the length along the flow path. A variable VFS width across the different sub-basins was obtained by using a fraction of the width of each sub-watershed as the width of the VFS. Since sub-basins vary in size, taking a constant fraction of the width of each of sub-watershed as VFS width gives a variable filter width. On the contrary, to get a constant width across all sub-basins, a variable fraction of the length (distance perpendicular to the flow path) of the sub-watershed was used. The fraction was varied from 0 to 1 corresponding to VFS area varying from zero acres to the size of the sub-watershed.

Parameters available from SWAT and GIS used in this study are subbasin area and length. The width of each HRU (sub-basin) was determined based on the length and area of HRU and assuming a rectangular shape to estimate the width. There are two options in SWAT in determining the HRU distribution. One of the options is the use multiple HRUs within a sub-basin. The user may specify sensitivities for the land use and soil data that will be used to determine the number and kind of HRUs in each sub-basin (Neitsch et al., 2002). Using the concept of virtual land units (HRUs), the different soil and land uses in the sub-basin can be simulated to the level of detail desired and accuracy can be increased due to discretization. However, if this option is used the assumption is there is no hydrologic interaction between HRUs in one sub-basin. HRUs in a sub-basin are not necessarily spatially connected. If the interaction of one land use area with another is important or if dimensions of HRU are required, rather than defining those land use areas as HRUs they should be defined as sub-basins. It is only at the sub-basin level that spatial relationships can be specified.

The second option is to assign a single HRU to each sub-watershed. If a single HRU per sub-basin is selected, the HRU is determined by the dominant land use category and soil type within each watershed. Spatial location and dimensions of HRU can be specified only when the dominant land option is used. Hence for this study, a dominant land use option was used in defining HRU distribution in SWAT. For the purpose of this study, the dominant land use class and soil type for each sub-watershed were used, resulting in one HRU per sub-watershed and making it possible to define HRU dimensions and locations.

It is expected that the accuracy of this option can be affected as a result of lumping land use and soil types in the sub-basin. However, the study uses only one land use (peanuts) through out the sub-basin along with switchgrass as a conservation crop and there could be no benefit from land cover discretization. For the purpose of reducing the effect of flow concentration each HRU was limited to less than ten acres with approximately half the area on both sides of field drains, at this size range the variation in soil type within an HRU is assumed to be very low. In addition to this, the study is comparative. Overestimating or underestimating sediment load from each HRU affects both alternatives compared and the over all conclusions from this study are not expected to change with slight change in sediment load generated from each HRU.

The environmental and economic benefits of the water quality incentive payment were also evaluated. If VFS is not harvested the producer receives incentive payments. Income could be obtained from the area occupied by VFS by selling the harvest or by receiving the incentive payment the producer would receive if the VFS is not harvested. The corresponding effects on sediment load were compared. This would help determine if the environmental benefits would be achieved from VFS while harvesting the VFS and without incentive payments. The incentive used was $43 per acre based on data from the 26th CRP signup in Caddo County (Agapoff et al, 2003).

Results and Discussion

The environmental and economic benefits of the two approaches (VFS and Optimal Replacement) approaches are given in Figure 5-1 and Figure 5-2. In both approaches, the amount of income from watershed decreases as the allowable sediment load is decreased, since an increasing part of the watershed has to be replaced by a less profitable crop. However, for lower allowable levels of sediment load the optimal total replacement approach is more effective than the VFS approach especially if VFS is not harvested for VFS placed on 100%, 75%, 50% or 25% of the land units. This is because trapping efficiency of VFS does not increase linearly with the width of VFS i.e. the slope of the trapping efficiency with respect to sediment load is not as shown in Figure 5-3.

In both cases as the amount of allowable sediment load is decreased, the income goes down. In the total replacement approach more area has to be converted to less erodible land cover (switchgrass) as the allowable sediment yield decreases which yields low returns. Also, in the case of the VFS approach the part of the land under the VFS has to be put out of production if water quality incentives are used. Alternatively if VFS is harvested and sold, it generates a lower income per acre compared to the row crop.

However as given by slopes of the curves in the optimal replacement and VFS approaches in Figure 5-1 and Figure 5-2 the change in income as a result of change in the allowable sediment load level differs in the two approaches. The optimal replacement approach is less sensitive to the change in the allowable sediment load level at lower allowable load levels as compared to the VFS approach, i.e. dollars lost per ton of sediment abated, is relatively lower in the optimal replacement approach.

Further evaluation was made to compare the environmental and economic benefits of installing vegetative filter strips in all fields and on selected fields based on the sediment load from each field. The comparison was made between installing VFS on 100%, 75%, 50% and 25%% of the fields with higher sediment load rates being selected for installation. The results are shown in Figure 5.1 and 5.2. Figure 5.1 shows the income at given sediment loading if VFS is not harvested and sold. The reason for the curves in this case to be below the optimal land distribution is because of the assumption that no income is generated from VFS unlike the results shown in Figure 5.2 in which VFS is harvested and sold.

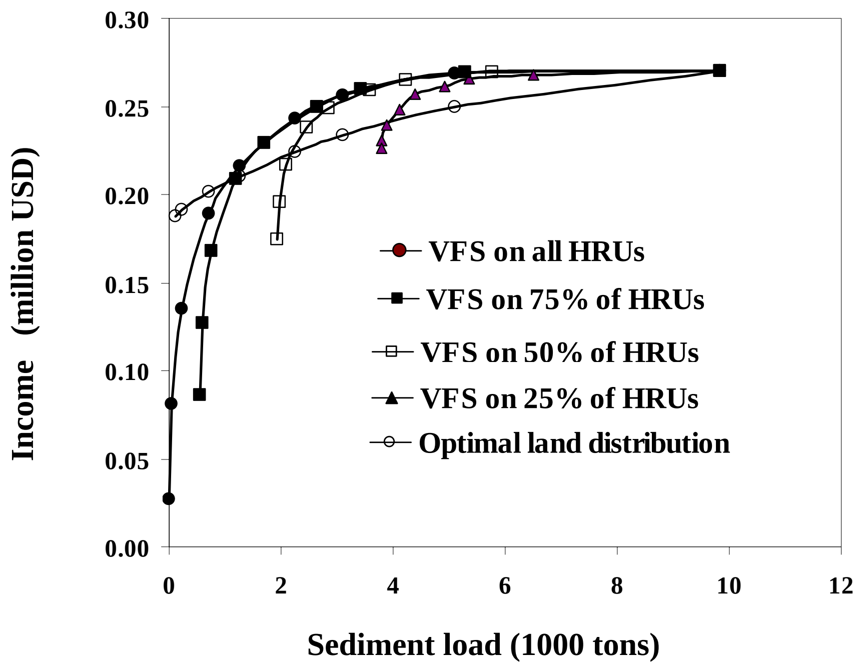

Figure 5-1. Income versus sediment load under optimal land distribution and VFS approaches (VFS not-harvested and sold).

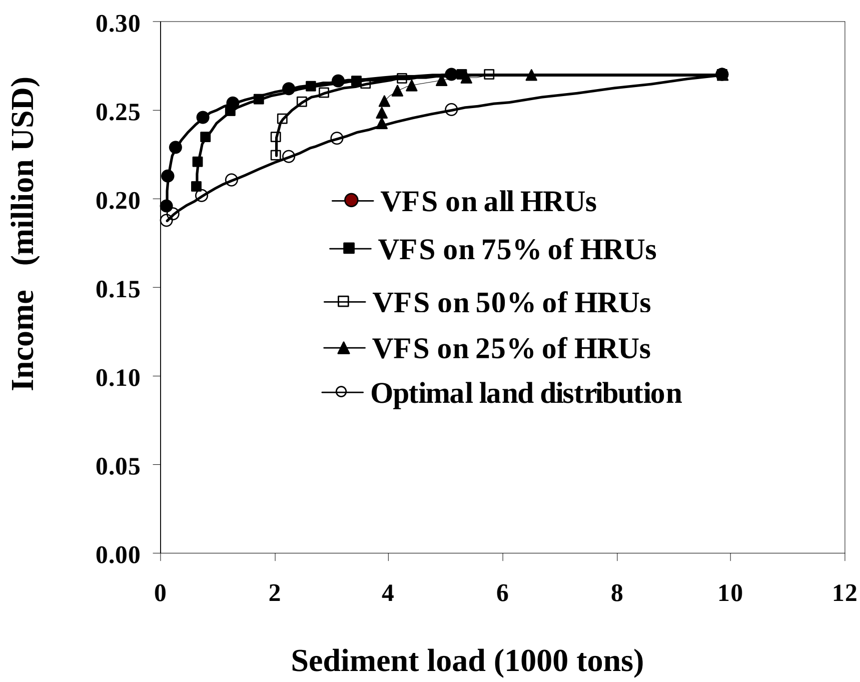

Figure 5-2. Income versus sediment load under optimal land distribution and VFS approaches (VFS harvested).

Figure 5-3 shows that the VFS reaches maximum trapping efficiency after a certain width. Increasing VFS width beyond this limit does not reduce sediment load further. From an economic stand point, it is not advisable to increase width beyond this point, since this involves opportunity cost as a result of replacement.

The effects of variable versus constant width VFS are shown in Figure 5.4 and 5.5. The results in Figure 5-4 show that it is beneficial to vary the width of the VFS in proportion to the size of the field contributing to the sediment inflow, however the difference is small. For the same level of allowable sediment load, the income obtained, if VFS width is varied in proportion to size of contributing area, is slightly higher than the income obtained if the constant VFS width is used across all land units. This suggests that to increase the environmental and economic efficiency of use of VFS it is advisable to choose the VFS width in proportion to the size of the contributing area; however, from point of view of implementation of recommended VFS width and given that the benefits of varying VFS width is only slightly higher, constant buffer width across all fields might be preferred. Figure 5-5 shows that the sediment load from constant VFS width and variable VFS width averaged over the whole watershed.

A comparison of income from harvesting and selling the VFS versus receiving the water quality incentive are shown in Figure 5.6. This shows that producers could obtain more income from harvesting the VFS than they would earn from the water quality incentive program while maintaining the same level of sediment load. The income obtained from harvesting the VFS is 43% higher than the income obtained from incentive payments.

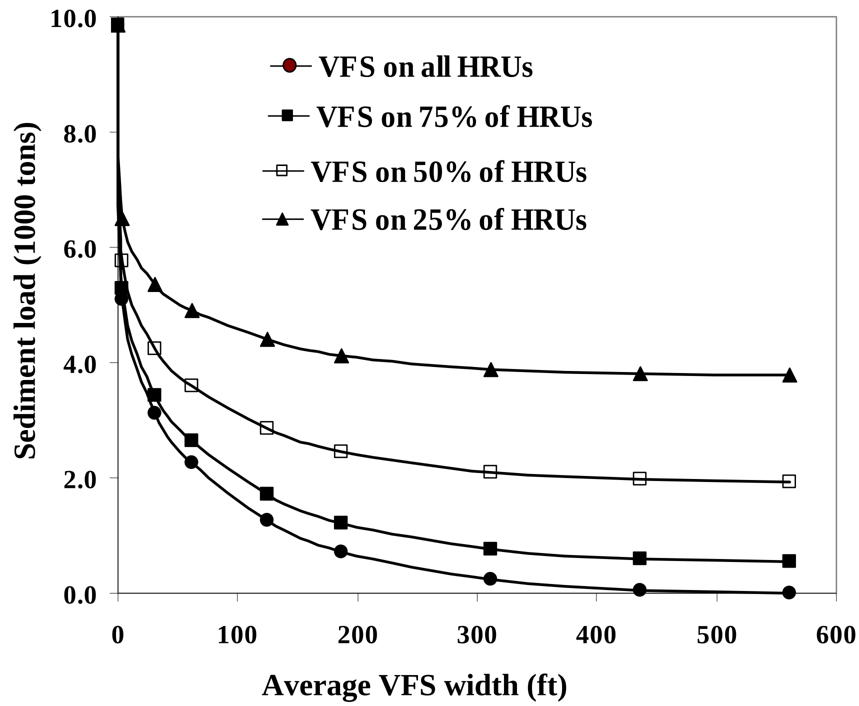

Figure 5-3. Average VFS width versus sediment load under each scenario

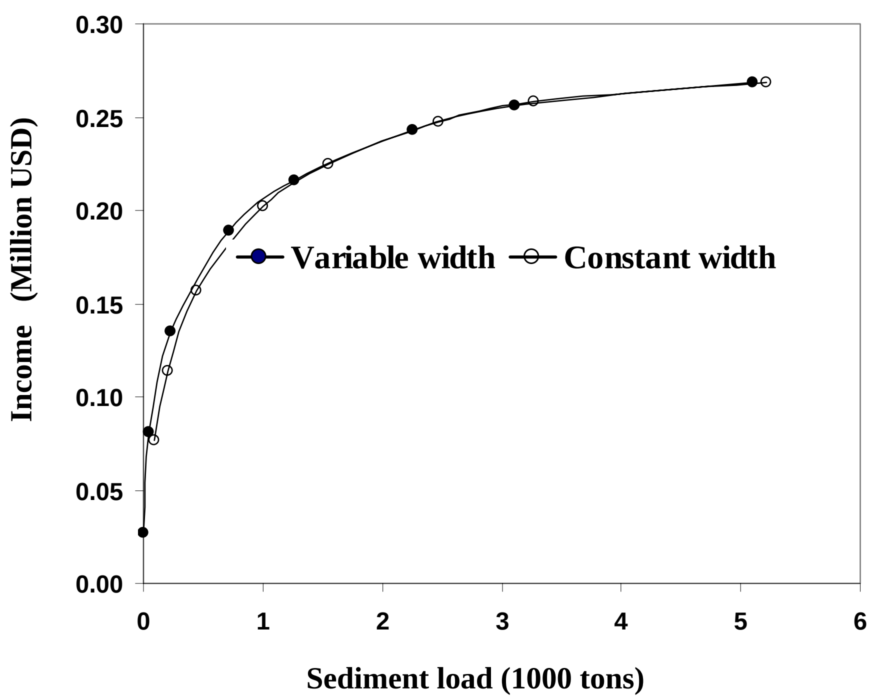

Figure 5-4. Income versus sediment load (variable and constant width).

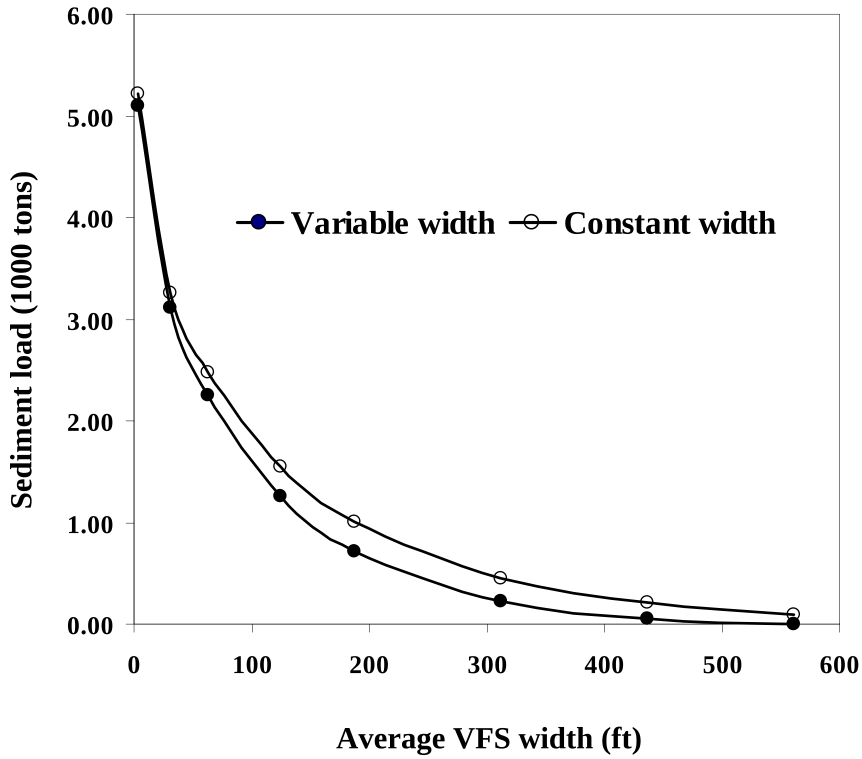

Figure 5-5. VFS width versus sediment load under each scenario (variable and constant width).

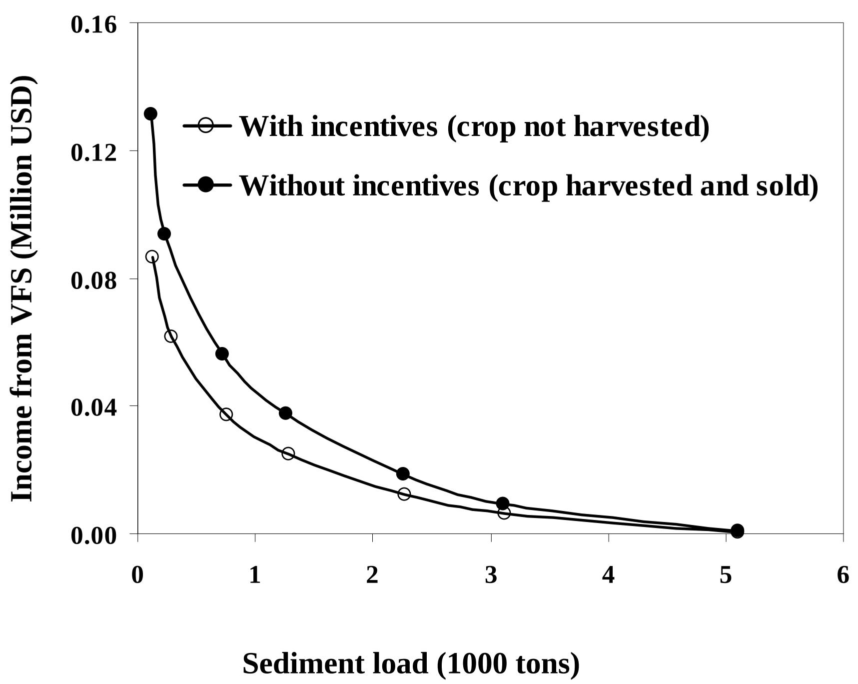

Figure 5-6. Comparison of environmental and economic benefits with and without water quality incentives.

Example application of the models: Economic and Environmental Implications of Switchgrass Production as Energy Crop

An example of how the VFS model and LUDM optimal land distribution model can be used to evaluate energy alternatives is presented. The discussion here is speculative and can only be used as a theoretical example. As pointed out earlier, displacement of row crops with perennial grasses such as switchgrass will have major environmental and economic implications. Thus, perennial grass production for biofuels offers significant economic advantages to a national energy strategy which considers both agricultural and environmental issues. Switchgrass is a high yielding crop, grows well in diverse geographical growing range and has high net energy yields. Also, as shown earlier in this chapter and Chapter III, it has high soil and water conservation potential. Ethanol production from switchgrass requires conversion of agricultural crop lands to switchgrass. It is important to determine land use planning approaches that are effective from both environmental perspective.

A ton (2,000 lb or 980 kg) of corn stover could yield about 80-90 US gallons (300-340 liters) of ethanol, and a ton of switchgrass could yield in the range of 75-100 US gallons (285-380 liters) (Oak Ridge National Laboratory, 2001). Nyren and Mattern (2003) reported that assuming a conversion rate of 75% at the processing plant, a ton of switchgrass would produce approximately 80 gallons of ethanol.

A conservative estimate of 50 gallons/ton of switchgrass and a 4 ton/acre of biomass yield was used to compare the sediment load reduction for conversion from agricultural crop (peanuts) to switchgrass using the two erosion control methods discussed, namely LUDM approach and VFS approach. The analysis was made on small watershed and projected to an area that can support an ethanol production plant with capacity as high as 100 million gallons. The results are given in Figures 5-7 and 5-8 for a small watershed and a projected watershed.

An estimated 250,000 to 500,000 acres are required to produce biomass sufficient to support an ethanol production plant with a capacity of 50 to 100 million gallons respectively. This is equivalent to 2 to 4 watersheds of size equal to the size of the Fort Cobb basin (200,000 acres) if only 50% of the watershed along the field drains is used to produce switchgrass. The corresponding sediment load reduction as a result of converting 50% of the watershed area along the filed drains from peanuts to switchgrass is greater than 97%. The reduction will be over 76% if the LUDM approach is used. Hence, the analysis suggests that placement of vegetative fiter strips aong field drains is more effective compared to total replacement of parts of the watershed by conservation crop.

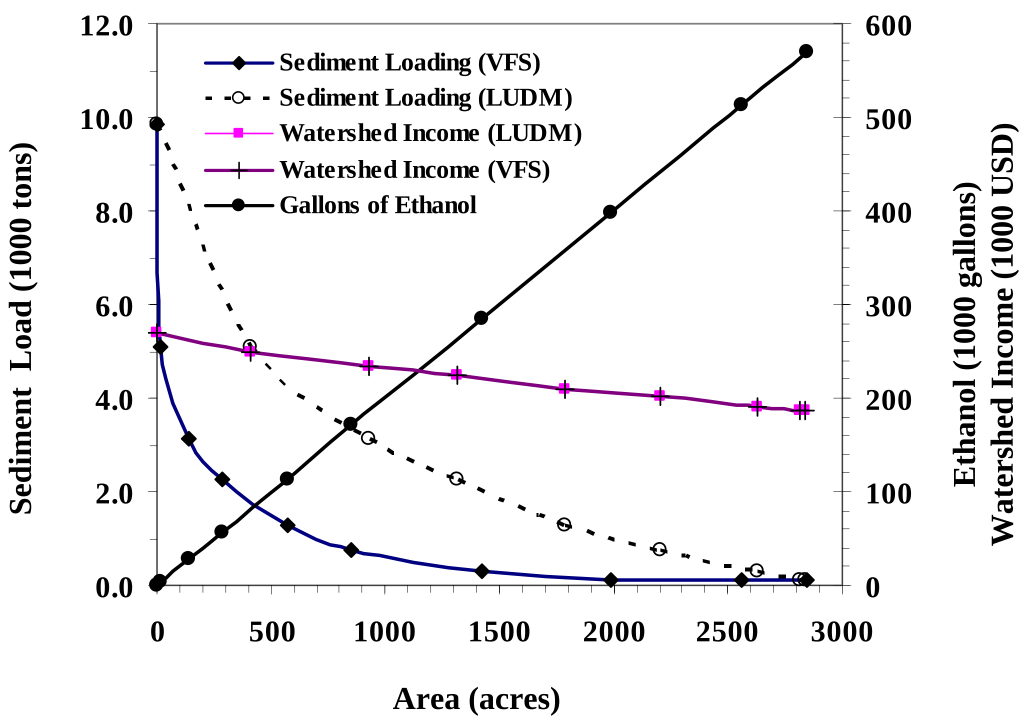

Figure 5-7. Comparison of environmental benefits LUDM and VFS along drains in a subbasin within Fort Cobb watershed.

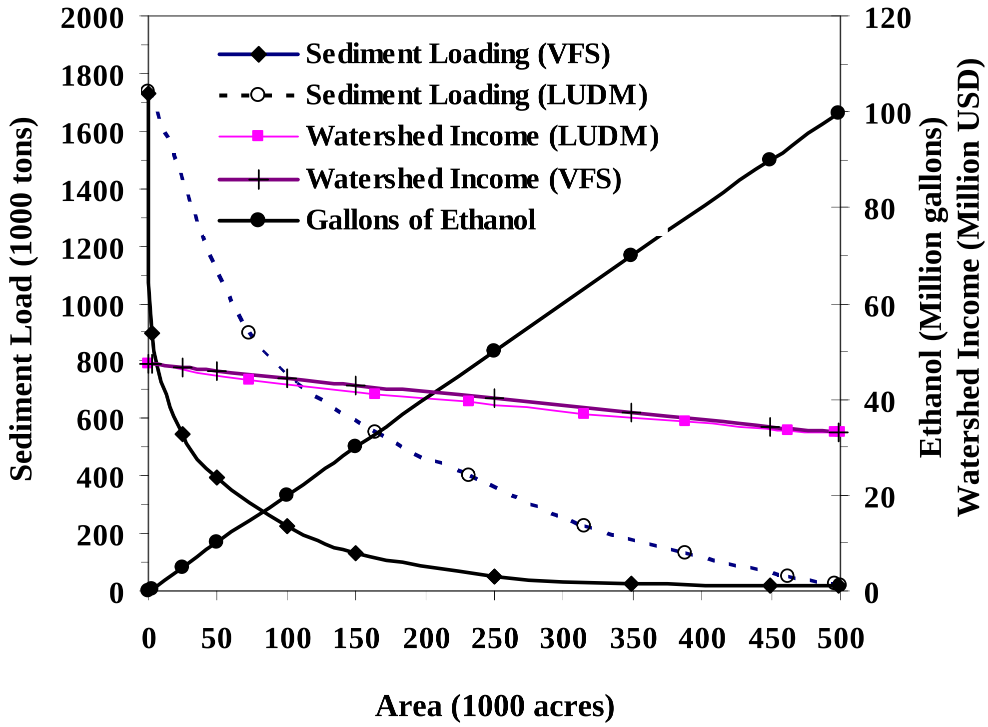

Figure 5-8 Comparison of environmental benefits of LUDM and VFS along drains determined by projection from Figure 5-7.

Summary and Conclusion

Two erosion control methods were compared. The first method involves targeting environmentally sensitive areas and assigning them to more protective land cover types such as switchgrass using the land use decision making model (LUDM) discussed in Chapter III. This was compared with a second approach in which vegetative filter strips (VFS) are placed along field drains to trap sediment. In the VFS approach, VFS were placed on all or a percentage (75%, 50% and 25%) of sub-basins in the watershed based on sediment yield from a contributing area. The selection of sub-basins for placing VFS was based on sediment yield from a contributing area.

For very low allowable levels of sediment load the LUDM approach is slightly more effective compared to the VFS approach especially if VFS is not harvested. In both cases, the income from the watershed changes with change in the amount of sediment allowed to leave the watershed; however, the change in income for the same amount of change in sediment load is less at very low allowable loads for optimal replacement approach than it is for the VFS approach. But as the allowable sediment loading is increased, the second approach, i.e. placement of VFS along the field drains is more effective compared to total replacement approach.

The results show that it is beneficial to vary the width of the VFS in proportion to the size of the field contributing to the sediment inflow; however the difference is small. For the same level of allowable sediment load the income obtained if VFS width is varied in proportion to size of contributing area is slightly higher than the income obtained if the constant VFS width is used across all land units. This suggests that to increase the environmental and economic efficiency of VFS, it is advisable to choose the VFS width in proportion to the size of the contributing area; however from the point of view of implementation of recommended VFS width and given that the benefits of varying VFS width is only slightly higher, constant buffer width across all fields might be preferred.

The results show that producers could obtain more income from harvesting the VFS than they would earn from water quality incentive program while maintaining the same level of sediment load. The results indicate that conservation expenditures could be reduced without sacrificing the environmental benefits through managed harvesting of conservation crops such as switchgrass used as vegetative filter strips.

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Chapter VI

SUMMARY AND FUTURE RECOMMENDATION

The effect of non-point source pollution as a result of sediment, nutrient and chemical transport is becoming widely recognized. Land use planning is one of many alternatives to minimize water quality problems. The overall objectives of the study were to evaluate environmental and economic impacts of Non-Point Source (NPS) pollution control approaches on a watershed scale. A Land Use Decision Model, LUDM, was developed to determine the optimal land use system in combination with a hydrologic model. A modified procedure, SGRASSF, was developed based on GRASSF vegetative filter strip model to evaluate sediment trapping efficiency on a watershed scale. Economic and environmental impacts on the use vegetative filter strips (VFS) were compared to an optimal total land cover replacement approach. The results of this study should prove useful in planning NPS pollution control alternatives with consideration of both environmental and economic constraints.

Single discipline scientific approaches are most useful when the system being modeled is relatively simple. The best solution is to combine a number of discipline-centric models that, together, capture the complex interactions among social, economic and hydrologic systems. The importance of integrated approaches in environmental studies is well recognized; the difficulty is how to bring diverse disciplines together to give useful results.

The mathematical programming method used in this study allows an efficient land use plan to be identified by combining economic and environmental objectives. Land cover change can be as effective as other BMP practices in controlling non-point source pollution if planned efficiently. Land use decisions in this study are made based on achieving sediment and nutrient load requirements.

LUDM was written for this analysis using General Algebraic Modeling System (GAMS). The SWAT model was used to determine sediment and nutrient loads. The LUDM is a valuable tool for modeling land use change in conjunction with the SWAT model. The model can be used to generate different land use scenarios based on environmental and economic goals. With the help of the model, multiple relationships between the decision variables and the constraints can be developed.

To achieve lower sediment and nutrient loads, more area needs to be converted to conservation crops such as CRP and switchgrass despite the relatively lower income generated from these land uses. The proportion of land cover under conservation crop and conventional crops for a given allowable level of sediment load would change if actual damage costs are considered. If damage costs are considered, it could be more profitable to grow a crop that generates less income to the producers compared to one that generates a higher income but with more damage to the environment. Future research should focus to include damage costs.

Vegetative filter strips can be effective in removing nutrient and sediment inputs to streams that have been transported into the vegetative filter strips. A simplified procedure, SGRASSF, was developed to compute sediment trapping efficiency using vegetative filter strips (VFS) based on the Kentucky filter strip model GRASSF. As in the original GRASSF, the impact of flow depth, velocity, sediment particle size distribution, width of filter strip, density of vegetation, height of vegetation and slope are predicted. SGRASSF gives similar trapping efficiency results to GRASSF. The SGRASSF can be used to calculate sediment trapping from multiple sub-watersheds using daily precipitation, sediment yields and subwatershed characteristics such as soil type, vegetation cover, slope and area.

Unlike sediment studies on the removal of nutrients by vegetative filter strips is very limited. Nutrients are removed in vegetative filter strips by infiltration and with sediment trapped in the vegetative filter strips. Infiltration is the primary mechanism for removal of soluble pollutants. Nutrients that are sediment bound are removed with sediment trapped in the vegetative filter strips. Future research should develop procedures to estimate nutrient trapping based on the modified equations developed in this study for infiltration rate and sediment trapping by settling since nutrient trapping in vegetative filter strips is dependent on infiltration rate and sediment trapping. The conceptual frame work for the development of SGRASSF can be used to develop routines for other BMPs such as ponds and bioretention cells since similar factors influence sediment trapping. Future research should focus on developing efficient routines for other BMP routines as well.

The results indicate that conservation expenditures could be reduced without sacrificing the environmental benefits through managed harvesting of conservation crops such as switchgrass used as vegetative filter strips. If filter strips are harvested incentive programs may not be used or if used only the differences between previous incentive amounts and returns from crop sales could be paid since the landowner expects to obtain returns through harvests. The modeling results show that producers could obtain more income from harvesting the VFS than they would earn from the water quality incentive program while maintaining the same level of sediment load depending of the production costs and price of conservation crop used. Further field level research should be carried out to support the results of this modeling study to determine the possibility of minimizing conservation expenditures without sacrificing the environmental benefits through managed harvesting of conservation crops such as switchgrass used as vegetative filter strips.

Appendix-A

Land Use Decision Models (LUDM)

Appendix A-1

```
*Model 1: LUDM, Maximum Return Model.
*Replacement of conventional crops by switchgrass
*Whole watershed based non-uniform reduction approach
*This model is used to determine optimal land distribution on the
*watershed to maximize income subject to sediment and nutrient load
*constraints

 Sets
*WMT=minimum tillage wheat, SOR=grain sorghum, NUT=peanuts,
SWG=Switchgrass, CRP=Bermudagrass
J activities (land use alternatives for each hru)
/WMTHRU1,….,WMTHRU1819, SORHRU1,….,SORHRU1819, NUTHRU1,….,NUTHRU1819,
SWGHRU1,….,SWGHRU1819, CRPHRU1,….,CRPHRU1819/
*list of HRUs (constraints)
I
/HRU1,…HRU1819/

Sed  sediment yield (tons/year)
/sedyld/
Phos  phosphorus yield (kilograms/year)
/Pyld  /
Nit  nitrogen yield (kilograms/year)
/Nyld  /

*Define values for area of each HRU (ha) used to compute watershed income,
total sediment and *nutrient yield

Parameter k(J) area  Min tillage wheat

/WMTHRU1        78.79
.
.
CRPHRU1819       6.58/

*Crop yield data for each crop

Parameter        P(J) yield data for minimum tillage wheat (tons/ha)
/WMTHRU1        2.78,…,WMTHRU1819        3.22/
Parameter        S(J) yield data for grain sorghm(tons/ha)
/SORHRU1        3.26,…,  SORHRU1819        4.49/
Parameter        N(J)
/NUTHRU1        3.83,….,NUTHRU1819         3.44/

Parameter        g(J)
/SWGHRU1        11.09,…,SWGHRU1819        10.06/
parameter W(J)
/WCTHRU1        2.65,…, WCTHRU1819        3.11/

*Crop payments (payments farmers receive for growing agricultural crops
based on previous year)

Parameter WMTPAY(J)
/WMTHRU1        0,…,WMTHRU8        78.91,…..,WMTHRU1819        78.91/
Parameter SORPAY(J)
*area allocation constraint (sum of area allocated to each of land cover
types should be equal to 1.
Parameter B(I)

/HRU1        1,….,HRU1819        1 /

Table A(I,J)
        WMTHRU1    SORHRU1    NUTHRU1    SWGHRU1    WCTHRU1
```

```
HRU1    1          1          1          1          1
+       WMTHRU2    SORHRU2    NUTHRU2    SWGHRU2    WCTHRU2
HRU2    1          1          1          1          1
        .          .          .          .          .
        .          .          .          .          .
+       WMTHRU1819 SORHRU1819 NUTHRU1819 SWGHRU1819 WCTHRU1819
HRU1819 1          1          1          1          1

*Sediment yield data from each HRU

Table sedi(sed,J)
       WMTHRU1     SORHRU1     NUTHRU1     SWGHRU1     WCTHRU1
sedyld 127.77346   932.71467   1334.98785  4.81717     457.2363
+      WMTHRU2     SORHRU2     NUTHRU2     SWGHRU2     WCTHRU2
sedyld 235.1376    1353.34752  2120.64723  13.38978    652.01697
        .          .           .           .           .
        .          .           .           .           .
+      WMTHRU1819  SORHRU1819  NUTHRU1819  SWGHRU1819  WCTHRU1819
sedyld 9.1293727   44.5542349  65.6630296  0.3159408   23.8798588

*Phosphorous yield data from each HRU

Table P(phos,J)
     WMTHRU1     SORHRU1     NUTHRU1     SWGHRU1     WCTHRU1
Pyld 107.47817   307.90403   573.24323   11.0558     158.33485
+    WMTHRU2     SORHRU2     NUTHRU2     SWGHRU2     WCTHRU2
Pyld 212.44029   498.52437   1030.68648  32.00484    252.44634
     .           .           .           .           .
     .           .           .           .           .
+    WMTHRU1819  SORHRU1819  NUTHRU1819  SWGHRU1819  WCTHRU1819
Pyld 8.622551    18.9893585  35.8000419  1.0728823   10.070613

*Nitrogen yield data from each HRU

Table N(nit,J)
     WMTHRU1     SORHRU1     NUTHRU1     SWGHRU1     WCTHRU1
Nyld 284.292     1342.41103  1822.6276   25.50731    734.89482
+    WMTHRU2     SORHRU2     NUTHRU2     SWGHRU2     WCTHRU2
Nyld 535.75449   2089.62213  3084.87468  69.88812    1142.05026
     .           .           .           .           .
     .           .           .           .           .

+    WMTHRU1819  SORHRU1819  NUTHRU1819  SWGHRU1819  WCTHRU1819
Nyld 19.0354332  73.2258625  90.3656509  3.8636927   41.0196472 ;

*price data ($/ton) and cost per hectar for each crop
*pp = price of min. Tillage wheat, ps = price of grain sorghum, pN = *price
of peanuts, pG = price of switchgrass, pW = price of conv. *Tillage wheat
*cp = cost for min. Tillage wheat, cs = cost for grain sorghum, cN = *cost
for peanuts, cG = cost for switchgrass, cW = cost for conv. *Tillage wheat,
cW = cost for min. Tillage wheat,

Scalar pp /91.60/
Scalar ps /70/
Scalar pN /355.72/
Scalar pG /39.05/
Scalar pW /91.60/
Scalar cp /344.04/
Scalar cs /350.71/
Scalar cN /1259.52/
Scalar cG /266.58/
Scalar cW /317.74/
```

```
parameter area;
area=73559;

*Income from hay

parameter PNUTHAY;
PNUTHAY=185.25;
parameter WHHTPASTmin;
WHHTPASTmin=91.884;
parameter WHHTPASTcon;
WHHTPASTcon=87.018;

variables
Z            'objective function (total income)'
Y            'total area'
V            'total area assigned to min.till wheat
rr           'total area assigned to grain sorghum '
h            'total area assigned to peanuts'
f            'total area assigned to switchgrass'
t            'total area assigned to conv. till wheat'
X(J)         'decision variable'
Totinc'total income'
totN         'total N from all HRUs'
totP         'total P from all HRUS'
totsed'total sediment yield from all HRUs'
arealimit    'Maximum area'
positive variable X;

Equations
Totalarea
areamintillwheat
areasorghum
areapeanut
areaswg
areaconvenwheat
totalN
totalP
totalsed
Totalincome
Cons1(sed)
*cons2(phos)
*Cons3(nit)
areacons
Obj
Watershed(I);

*Objective function

obj.. Z=E=Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+ sum (J,q(J)*(pW*W(j)-
cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum (J,q(J)*WHHTPASTcon*X(j));

*Sum of percentages of area assigned to each land cover equal to 1.

watershed(I)..sum(j,A(I,J)*X(J))=L=B(I);

*Sediment and nutrient load constraints

cons1(sed)..sum(j,sedi(sed,J)*X(J))=L=sedyld;
*cons2(phos)..sum(j,P(phos,J)*X(J))=L=pyld;
*cons3(nit)..sum(j,N(nit,J)*X(J))=L=Nyld;
```

```
* Total area is equal to the watershed area

areacons..   Sum(J, k(J)*X(j))+ sum(J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))+sum (J,R(J)*X(j)) =E=area;

*Information to be displayed

Totalarea..Y=E=Sum(J, k(J)*X(j))+ sum(J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))+sum (J,R(J)*X(j));
areamintillwheat..V=E=Sum(J, k(J)*X(j));
areasorghum..rr=E=Sum(J, l(J)*X(j));
areapeanut..h=E=Sum(J, m(J)*X(j));
areaswg..f=E=Sum(J, o(J)*X(j));
areaconvenwheat..t=E=Sum(J, q(J)*X(j));
totalN(nit) ..      totN=E=sum(j,N(nit,J)*X(J));
totalP(phos) ..     totP=E=sum(j,P(phos,J)*X(J));
totalsed(sed) ..  totsed=E=sum(j,sedi(sed,J)*X(J));
Totalincome..totinc=E=Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+ sum (J,q(J)*(pW*W(j)-
cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum (J,q(J)*WHHTPASTcon*X(j));
Model landuse /all/;
solve landuse using LP Maximizing z;
display Y.L,V.L,rr.L,h.L,f.L,t.L,totinc.L,totN.L,totP.L,totsed.L;
```

Appendix A-2

```
*Model 2: LUDM, Maximum Return Model.
*Replacement of conventional crops by switch grass
*HRU based uniform load reduction approach
*This model is used to determine optimal land distribution on the
*watershed tomaximize income subject to sediment and nutrient load
*constraints
Sets
*WMT=minimum tillage wheat, SOR=grain sorghum, NUT=peanuts,
*SWG=Switchgrass, *CRP=Bermudagrass
J activities (land use alternatives for each hru)
/WMTHRU1,….,WMTHRU1819, SORHRU1,….,SORHRU1819, NUTHRU1,….,NUTHRU1819,
SWGHRU1,….,SWGHRU1819, CRPHRU1,….,CRPHRU1819/
*list of HRUs (constraints)
I
/HRU1,…HRU1819/
area
/areaHRU1,…,areaHRU1819/
Sed  sediment yield (tons/year)
/sedyld/
Phos  phosphorus yield (kilograms/year)
/Pyld  /
Nit  nitrogen yield (kilograms/year)
/Nyld  /
*area of each HRU (ha) used to compute watershed income, total sediment and
nutrient yield
Parameter k(J) area  Min tillage wheat
/WMTHRU1        78.79
CRPHRU1819       6.58/
*Crop yield
Parameter        P(J) 'yield data for minimum tillage wheat (tons/ha)'
/WMTHRU1        2.78,…,WMTHRU1819        3.22/
Parameter        S(J) yield data for grain sorghm(tons/ha)
/SORHRU1        3.26,…,  SORHRU1819        4.49/
Parameter        N(J)
/NUTHRU1        3.83,….,NUTHRU1819        3.44/

Parameter        g(J)
/SWGHRU1        11.09,…,SWGHRU1819        10.06/
parameter W(J)
/WCTHRU1        2.65,…, WCTHRU1819        3.11/

* Crop payments (payments farmers receive for growing agricultural crops
based on previous year)
Parameter WMTPAY(J)
/WMTHRU1        0,…,WMTHRU8        78.91,…..,WMTHRU1819        78.91/
Parameter SORPAY(J)
*area allocation constraint (sum of area allocated to each of land cover
types should be equal to 1.
Parameter B(I)
/HRU1       1,….,HRU1819        1 /
subarea(area)
/areaHRU1        168.1,…areaHRU1819 6.58/


Table A(I,J)
        WMTHRU1    SORHRU1    NUTHRU1    SWGHRU1    WCTHRU1
HRU1    1          1          1          1          1
+       WMTHRU2    SORHRU2    NUTHRU2    SWGHRU2    WCTHRU2
HRU2    1          1          1          1          1
        .          .          .          .          .
        .          .          .          .          .
+       WMTHRU1819 SORHRU1819 NUTHRU1819 SWGHRU1819 WCTHRU1819
```

```
HRU1819 1          1          1          1          1

*Sediment yield from each HRU

Table sedi(sed,J)
             WMTHRU1     SORHRU1     NUTHRU1     SWGHRU1     WCTHRU1
Sedyld1      127.77346   932.71467   1334.98785  4.81717     457.2363
+            WMTHRU2     SORHRU2     NUTHRU2     SWGHRU2     WCTHRU2
Sedyld2      235.1376    1353.34752  2120.64723  13.38978    652.01697
             .           .           .           .           .
             .           .           .           .           .
+            WMTHRU1819  SORHRU1819  NUTHRU1819  SWGHRU1819  WCTHRU1819
Sedyld1819   9.1293727   44.5542349  65.6630296  0.3159408   23.8798588

Table P(phos,J)
             WMTHRU1     SORHRU1     NUTHRU1     SWGHRU1     WCTHRU1
Pyld1        107.47817   307.90403   573.24323   11.0558     158.33485
+            WMTHRU2     SORHRU2     NUTHRU2     SWGHRU2     WCTHRU2
Pyld2        212.44029   498.52437   1030.68648  32.00484    252.44634
             .           .           .           .           .
             .           .           .           .           .
+            WMTHRU1819  SORHRU1819  NUTHRU1819  SWGHRU1819  WCTHRU1819
Pyld1819           8.622551    18.9893585  35.8000419  1.0728823
10.070613

Table N(nit,J)
             WMTHRU1     SORHRU1     NUTHRU1     SWGHRU1     WCTHRU1
Nyld1        284.292     1342.41103  1822.6276   25.50731    734.89482
+            WMTHRU2     SORHRU2     NUTHRU2     SWGHRU2     WCTHRU2
Nyld2        535.75449   2089.62213  3084.87468  69.88812    1142.05026
             .           .           .           .           .
             .           .           .           .           .

+            WMTHRU1819  SORHRU1819  NUTHRU1819  SWGHRU1819  WCTHRU1819
Nyld1819     19.0354332  73.2258625  90.3656509  3.8636927   41.0196472 ;

*price ($/ton) and cost ($/ha) for each crop

Scalar pp /91.60/
Scalar ps /70/
Scalar pN /355.72/
Scalar pG /39.05/
Scalar pW /91.60/
Scalar cp /344.04/
Scalar cs /350.71/
Scalar cN /1259.52/
Scalar cG /266.58/
Scalar cW /317.74/
parameter area;
area=73559;
parameter PNUTHAY;
PNUTHAY=185.25;
parameter WHHTPASTmin;
WHHTPASTmin=91.884;
parameter WHHTPASTcon;
WHHTPASTcon=87.018;

variables
Z
Y
V
rr
h
f
```

```
t
X(J)
totinc
totN
totP
totsed
arealimit
positive variable X;
Equations
Totalarea
areamintillwheat
areasorghum
areapeanut
areaswg
areaconvenwheat
totalN
totalP
totalsed
Totalincome
Cons1(sed)
*cons2(phos)
*Cons3(nit)
areacons
Obj
Watershed(I);

obj.. Z=E=Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+ sum (J,q(J)*(pW*W(j)-
cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum (J,q(J)*WHHTPASTcon*X(j));

*Sum of percentages of area assigned to each land cover equal to 1.

watershed(I)..sum(j,A(I,J)*X(J))=L=B(I);

*Sediment and nutrient load constraints

cons1(sed)..sum(j,sedi(sed,J)*X(J))=L=sedyld*subarea(area)/area;
*cons2(phos)..sum(j,P(phos,J)*X(J))=L=pyld* subarea(area)/area;
*cons3(nit)..sum(j,N(nit,J)*X(J))=L=Nyld*subarea(area)/area;

* Total area is equal to the watershed area

areacons..   Sum(J, k(J)*X(j))+ sum(J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))+sum (J,R(J)*X(j)) =E=area;

*Information to be displayed

Totalarea..Y=E=Sum(J, k(J)*X(j))+ sum(J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))+sum (J,R(J)*X(j));
areamintillwheat..V=E=Sum(J, k(J)*X(j));
areasorghum..rr=E=Sum(J, l(J)*X(j));
areapeanut..h=E=Sum(J, m(J)*X(j));
areaswg..f=E=Sum(J, o(J)*X(j));
areaconvenwheat..t=E=Sum(J, q(J)*X(j));
totalN(nit) ..     totN=E=sum(j,N(nit,J)*X(J));
totalP(phos) ..    totP=E=sum(j,P(phos,J)*X(J));
totalsed(sed) ..  totsed=E=sum(j,sedi(sed,J)*X(J));
Totalincome..totinc=E=Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
```

```
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+ sum (J,q(J)*(pW*W(j)-
cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum (J,q(J)*WHHTPASTcon*X(j));
Model landuse /all/;
solve landuse using LP Maximizing z;
display Y.L,V.L,rr.L,h.L,f.L,t.L,totinc.L,totN.L,totP.L,totsed.L;
```

Appendix A-3

```
*Model 3: LUDM, Maximum Return Model
*Replacement of conventional crops by CRP (Bermudagrass)
*Whole watershed based non-uniform load reduction approach.
*Same inputs were used as in Model 1. Switchgrass was not used in this
*case.Instead Bermudagrass was used.
*price ($/ton) and cost ($/ha) for each crop
Scalar pp /91.60/
Scalar ps /70/
Scalar pN /355.72/
Scalar pW /91.60/
Scalar cp /344.04/
Scalar cs /350.71/
Scalar cN /1259.52/
Scalar cW /317.74/
scalar cR/40.58/
*Payemnt received per ha of land enrolled in CRP
parameter payment;
payment=107.32;
parameter area;
area=73559;
parameter PNUTHAY;
PNUTHAY=185.25;
parameter WHHTPASTmin;
WHHTPASTmin=91.884;
parameter WHHTPASTcon;
WHHTPASTcon=87.018;
variables
Z
Y
V
rr
h
t
ac
crp
X(J)
totinc
totN
totP
totsed
positive variable X;
Equations
Totalarea
areamintillwheat
areasorghum
areapeanut
areaconvenwheat
areacrp1
areacrp2
totalN
totalP
totalsed
Totalincome
Cons1(sed)
*cons2(phos)
*Cons3(nit)
areacons
Incomecrp
Obj
Watershed(I);

*objective function (Total net return from watershed)
```

```
obj.. Z=E=Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,q(J)*(pW*W(j)-
cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum (J,q(J)*WHHTPASTcon*X(j))+sum
(J,R(J)*(payment-cR)*X(j));

*Sum of percentages of area assigned to each land cover equal to 1.
watershed(I)..sum(j,A(I,J)*X(J))=L=B(I);
*sediment and nutrient load constraints
cons1(sed)..sum(j,sedi(sed,J)*X(J))=L=195008;
*cons2(phos)..sum(j,P(phos,J)*X(J))=L=200000;
*cons3(nit)..sum(j,N(nit,J)*X(J))=L=800000;

* Total area is equal to the watershed area
areacons.. Sum(J, k(J)*X(j))+ sum(J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))+sum (J,R(J)*X(j))=E=area;
*Only 25% of the total area can be enrolled
areacrp2        ..Sum(J, R(J)*X(j))=L=0.25*Y;

*Information to be displayed
Totalarea..Y=E=Sum(J, k(J)*X(j))+ sum(J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))+sum (J,R(J)*X(j));
areamintillwheat..V=E=Sum(J, k(J)*X(j));
areasorghum..rr=E=Sum(J, l(J)*X(j));
areapeanut..h=E=Sum(J, m(J)*X(j));
areaconvenwheat..t=E=Sum(J, q(J)*X(j));
areacrp1        ..ac=E=Sum(J, R(J)*X(j));
totalN(nit) ..     totN=E=sum(j,N(nit,J)*X(J));
totalP(phos) ..    totP=E=sum(j,P(phos,J)*X(J));
totalsed(sed) ..  totsed=E=sum(j,sedi(sed,J)*X(J));
Incomecrp..crp=E=sum (J,R(J)*(payment-cR)*X(j));
Totalincome..totinc=E=Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+ sum (J,q(J)*(pW*W(j)-
cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum (J,q(J)*WHHTPASTcon*X(j))+sum
(J,R(J)*(payment-cR)*X(j));
Model landuse /all/;
solve landuse using LP Maximizing z;
display Y.L,V.L,rr.L,h.L,,ac.L,t.L,crp.L,totinc.L,totN.L,totP.L,totsed.L;
```

Appendix A-4

```
*Model 4: LUDM, Minimum Incentive Model.
*Replacement by switchgrass. Subsidies are based on the differences
*between switchgrass and the most profitable crop in the watershed.
*Objective:optimal land distribution to minimize incentive payment
*price ($/ton) and cost ($/ha) for each crop
Scalar pp /91.60/
Scalar ps /70/
Scalar pN /355.72/
Scalar pG /39.05/
Scalar pW /91.60/
Scalar cp /344.04/
Scalar cs /350.71/
Scalar cN /1259.52/
Scalar cG /266.58/
Scalar cW /317.74/
parameter PNUTHAY;
PNUTHAY=185.25;
parameter WHHTPASTmin;
WHHTPASTmin=91.884;
parameter WHHTPASTcon;
WHHTPASTcon=87.018;
parameter area;
area= 73559;
*incentive payment ($/ha)
parameter diff(J);
diff(J)=(P_income-SG_income)*o(J)/(area*(o(J)+0.000001));
Display diff;
variables
SWGINC
PNUTINC
WMTINC
Income
Z
y
V
rr
h
f
t
Inc_used
pwmt
syld
pyld
Nyld
X(J)
positive variable X,iv;
Equations
limi(j)
totinc
areatot
areatot1
areawmt
areasor
areanut
areaswg
areawct
TargetIncome
row(I)
cons1(sed)
*cons2(phos)
*cons3(nit)
totincen(sed)
```

```
usedincen
swgincome
pnutincome
wmtincome
crpincome
Totalsedyld(sed)
TotalPyld(phos)
TotalNyld(nit);
TargetIncome..Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+sum(j,o(j)*iv(j)*X(J))+ sum
(J,q(J)*(pW*W(j)-cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum
(J,q(J)*WHHTPASTcon*X(j)) =G=22002303 ;
row(I)..sum(j,A(I,J)*X(J))=l=B(I);
cons1(sed)..sum(j,sedi(sed,J)*X(J))=L=600000;
*cons2(phos)..sum(j,P(phos,J)*X(J))=L=100000;
*cons3(nit)..sum(j,N(nit,J)*X(J))=L=nitro(nit);
areatot..Y=E=Sum(J, k(J)*X(j))+ sum (J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j));
areatot1..Sum(J, k(J)*X(j))+ sum (J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))=E=area;
areawmt..V=E=Sum(J, k(J)*X(j));
areasor..rr=E=Sum(J, l(J)*X(j));
areanut..h=E=Sum(J, m(J)*X(j));
areaswg..f=E=Sum(J, o(J)*X(j));
areawct..t=E=Sum(J, q(J)*X(j));
limi(j)..iv(j)=G=diff(J);
*total incentive required
totincen(sed)..Z=E=sum (J,o(J)*iv(J)*X(j));
totinc..Income=E=  Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+sum(j,o(j)*iv(j)*X(J))+ sum
(J,q(J)*(pW*W(j)-cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum
(J,q(J)*WHHTPASTcon*X(j))
swgincome..SWGINC=E=sum (J,o(J)*(pG*G(j)-cG)*X(j))+Sum(J,
o(J)*SWGPAY(J)*X(j));
pnutincome..PNUTINC=E= sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J,m(J)*NUTPAY(J)*X(j));
Totalsedyld(sed).. syld=E=sum(j,sedi(sed,J)*X(J));
TotalPyld(phos)..   pyld=E=sum(j,P(phos,J)*X(J));
TotalNyld(nit)..    Nyld=E=sum(j,N(nit,J)*X(J));
Model landuse /all/;
solve landuse using NLP Minimizing z;
```

Appendix A-5

```
*Model 5: LUDM, Minimum incentive model.
*Replacement by CRP. The assumption here is producers will be will be
*enroll their land in CRP if they are apid an amount equal to the net
*income they would obtain from producing the most profitable crop in the
*watershed.
*The objective to determine the optimal land distribution on the *watershed
to *minimize incentive payment while meeting sediment and *nutrient load
constraints

Scalar pp /91.60/
Scalar ps /70/
Scalar pN /355.72/
Scalar pG /39.05/
Scalar pW /91.60/
Scalar cp /344.04/
Scalar cs /350.71/
Scalar cN /1259.52/
Scalar cG /266.58/
Scalar cW /317.74/
scalar cR/40.58/
parameter payment;
payment=0;
parameter PNUTHAY;
PNUTHAY=185.25;
parameter WHHTPASTmin;
WHHTPASTmin=91.884;
parameter WHHTPASTcon;
WHHTPASTcon=87.018;
parameter wheat1_income(J);
wheat1_income(J)= k(J)*(pp*P(j)-cp)+k(J)*WHHTPASTmin;
display  wheat1_income;
parameter Sorghum_income(J);
Sorghum_income(J)= l(J)*(ps*S(j)-cs);
display  Sorghum_income;
parameter Peanut_income(J);
Peanut_income(J)= m(J)*(pN*N(j)-cN)+ m(J)*PNUTHAY;
display  Peanut_income;
parameter Switchgrass_income(J);
Switchgrass_income(J)= o(J)*(pG*G(j)-cG);
display  Switchgrass_income;
parameter Wheat2_income(J);
Wheat2_income(J)= q(J)*(pW*W(j)-cW)+ q(J)*WHHTPASTcon;
display  Wheat2_income;
parameter P_income;
P_income=sum(J,m(J)*(pN*N(j)-cN))+sum
(J,m(J)*PNUTHAY)+Sum(J,m(J)*NUTPAY(J));
display  P_income;
parameter SG_income;
SG_income=sum(J,o(J)*(pG*G(j)-cG))+Sum(J, o(J)*SWGPAY(J));
display  SG_income;
parameter diff(J);
parameter area;
area=73559;
diff(J)=(P_income)*R(J)/(area*(R(J)+0.000001));
Display diff;
variables
SWGINC
PNUTINC
WMTINC
WCTINC
Income
ac
```

```
Z
y
V
rr
h
f
t
crp
Inc_used
pwmt
syld
pyld
Nyld
X(J)
sub
positive variable X,iv;
Equations
limi(j)
totinc
areatot
areatot1
areawmt
areasor
areanut
areaswg1
areaswg2
areawct
areacrp
TargetIncome
row(I)
cons1(sed)
*cons2(phos)
*cons3(nit)
totincen(sed)
usedincen
swgincome
pnutincome
wmtincome
wctincome
crpincome
crpareacons
Totalsedyld(sed)
TotalPyld(phos)
TotalNyld(nit);
TargetIncome..Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+sum(j,R(j)*iv(j)*X(J))+ sum
(J,q(J)*(pW*W(j)-cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum
(J,q(J)*WHHTPASTcon*X(j))=G=22002303.7675;
row(I)..sum(j,A(I,J)*X(J))=l=B(I);
cons1(sed)..sum(j,sedi(sed,J)*X(J))=L=100000;
*cons2(phos)..sum(j,P(phos,J)*X(J))=L=100000;
*cons3(nit)..sum(j,N(nit,J)*X(J))=L=nitro(nit);
areatot..Y=E=Sum(J, k(J)*X(j))+ sum (J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))+Sum(J, R(J)*X(j));
areatot1..Sum(J, k(J)*X(j))+ sum (J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))+Sum(J, R(J)*X(j))=E=73314.9;
areawmt..V=E=Sum(J, k(J)*X(j));
areasor..rr=E=Sum(J, l(J)*X(j));
areanut..h=E=Sum(J, m(J)*X(j));
areaswg1..f=E=Sum(J, o(J)*X(j));
areawct..t=E=Sum(J, q(J)*X(j));
```

```
areacrp..ac=E=Sum(J, R(J)*X(j));
*Total area enrolled should be less than 25% of the total area
crpareacons.. Sum(J, R(J)*X(j))=L=0.25*area;
limi(j)..iv(j)=G=diff(J);
totincen(sed)..Z=E=sum (J,R(J)*iv(J)*X(j));
totinc..Income=E=  Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+sum(j,R(j)*iv(j)*X(J))+ sum
(J,q(J)*(pW*W(j)-cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum
(J,q(J)*WHHTPASTcon*X(j));
usedincen.. Inc_used=E= sum (J,R(J)*iv(j)*X(j));
swgincome..SWGINC=E=sum (J,o(J)*(pG*G(j)-cG)*X(j))+Sum(J,
o(J)*SWGPAY(J)*X(j));
pnutincome..PNUTINC=E= sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J,m(J)*NUTPAY(J)*X(j));
crpincome..crp=E= sum (J,R(J)*iv(j)*X(j));
wmtincome..  WMTINC=E= Sum(J, k(J)*(pp*P(j)-cp)*X(j))+sum
(J,k(J)*WMTPAY(J)*X(j))+ Sum(J, k(J)*WHHTPASTmin*X(j));
wctincome..  WCTINC=E= Sum(J, q(J)*(pW*W(j)-cW)*X(j))+sum
(J,q(J)*WCTPAY(J)*X(j))+ Sum(J, q(J)*WHHTPASTmin*X(j));
Totalsedyld(sed).. syld=E=sum(j,sedi(sed,J)*X(J));
TotalPyld(phos)..   pyld=E=sum(j,P(phos,J)*X(J));
TotalNyld(nit)..    Nyld=E=sum(j,N(nit,J)*X(J));
Model landuse /all/;
solve landuse using NLP Minimizing z;
display
Y.L,V.L,rr.L,h.L,f.L,t.L,ac.L,Income.L,Inc_used.L,crp.L,SWGINC.L,PNUTINC.L,
WMTINC.L,WCTINC.L,syld.L,pyld.L,Nyld.L;
iv.UP(J)=600;
```

Appendix A-6

```
*Model 6: Effect of Erosion Charges (replacement by switchgrass).
*Net social benefit is the net income after deduction the total damage
*cost.
Scalar pp /91.60/
Scalar ps /70/
Scalar pN /355.72/
Scalar pG /39.05/
Scalar pW /91.60/
Scalar cp /344.04/
Scalar cs /350.71/
Scalar cN /1259.52/
Scalar cG /266.58/
Scalar cW /317.74/
parameter DC;
DC=5;
parameter area;
area=73559;
parameter PNUTHAY;
PNUTHAY=185.25;
parameter WHHTPASTmin;
WHHTPASTmin=91.884;
parameter WHHTPASTcon;
WHHTPASTcon=87.018;
variables
sedyield
Pyield
Nyield
Z
Y
V
rr
h
f
t
X(J)
totinc
dirinc
positive variable X;
Equations
Totalarea
Totalarea1
areamintillwheat
areasorghum
areapeanut
areaswitchgrass
areaconvenwheat
Directinc
Totalincome
Cons1(sed)
cons2(phos)
Cons3(nit)
Obj
Watershed(I);
*sub-watershed(un);
watershed(I)..sum(j,A(I,J)*X(J))=L=B(I);
*sub-watershed(un)..sum(j,uni(un,J)*X(J))=L=bu(un);
cons1(sed)..sedyield=E=sum(j,sedi(sed,J)*X(J));
obj.. Z=E=Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+ sum (J,q(J)*(pW*W(j)-
```

```
cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum (J,q(J)*WHHTPASTcon*X(j))-
DC*sedyield;
cons2(phos)..Pyield=E=sum(j,P(phos,J)*X(J));
cons3(nit)..Nyield=E=sum(j,N(nit,J)*X(J));
Totalarea..Y=E=Sum(J, k(J)*X(j))+ sum(J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))+sum (J,R(J)*X(j));
Totalarea1..Sum(J, k(J)*X(j))+ sum(J,l(J)*X(j))+ sum (J,m(J)*X(j))+ sum
(J,o(J)*X(j))+ sum (J,q(J)*X(j))+sum (J,R(J)*X(j))=E=area;
areamintillwheat..V=E=Sum(J, k(J)*X(j));
areasorghum..rr=E=Sum(J, l(J)*X(j));
areapeanut..h=E=Sum(J, m(J)*X(j));
areaconvenwheat..t=E=Sum(J, q(J)*X(j));
areaswitchgrass..f=E=Sum(J, o(J)*X(j));
*Net social benefits (NSB)
Totalincome..totinc=E=Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+ sum (J,q(J)*(pW*W(j)-
cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum (J,q(J)*WHHTPASTcon*X(j))-
DC*sedyield;
*Direct income (without considering damage costs)
Directinc..dirinc=E=Sum(J, k(J)*(pp*P(j)-cp)*X(j))+Sum(J,
k(J)*WHHTPASTmin*X(j))+Sum(J, k(J)*WMTPAY(J)*X(j))+ sum (J,l(J)*(ps*S(j)-
cs)*X(j))+Sum(J, l(J)*SORPAY(J)*X(j))+ sum (J,m(J)*(pN*N(j)-cN)*X(j))+sum
(J,m(J)*PNUTHAY*X(j))+Sum(J, m(J)*NUTPAY(J)*X(j))+ sum (J,o(J)*(pG*G(j)-
cG)*X(j))+Sum(J, o(J)*SWGPAY(J)*X(j))+ sum (J,q(J)*(pW*W(j)-
cW)*X(j))+Sum(J, q(J)*WcTPAY(J)*X(j))+sum (J,q(J)*WHHTPASTcon*X(j));
Model landuse /all/;
solve landuse using LP Maximizing z;
display
Y.L,V.L,rr.L,h.L,f.L,t.L,totinc.L,sedyield.L,Pyield.L,Nyield.L,dirinc.L;
```

Appendix -B

Procedures for Trapping Efficiency Calculation

Appendix B-1

Summary of VFS Trapping Efficiency Computation Procedures

1) **Determine representative diameter of coarse, medium and fine particles using soil texture data and CREAMS method.**

For coarse particles:

$$dp_c = \frac{F_{sa} d_{sa} + F_{\lg} d_{\lg}}{F_{sa} + F_{\lg}}$$

where d_{sa} is 0.2 mm and:

$$d_{\lg} = \begin{vmatrix} 0.3 & O_{cl} \le 0.15 \\ 2O_{cl} & O_{cl} > 0.15 \end{vmatrix}$$

and O_{cl} is fraction of clay in the parent material.

For medium size particles:

$$dp_m = \frac{F_{si} d_{si} + F_{sg} d_{sg}}{F_{si} + F_{sg}}$$

where d_{si} is 0.01 and:

$$d_{\lg} = \begin{vmatrix} 0.03 & O_{cl} < 0.25 \\ 2(O_{cl} - 0.25) + 0.03 & 0.25 \le O_{cl} \le 0.6 \\ 0.1 & O_{cl} > 0.6 \end{vmatrix}$$

For fine particles, dp_f is 0.002 mm.

2) **Compute V_s for each particle size class using diameters from (1)**

For medium and fine particles:

$$V_s = 2.81d^2 \qquad \text{[fps]}$$

For coarse particles:

$$\log V_s = -0.34246(\log d)^2 + 0.98912 \log d + 1.14613 \quad \text{[fps]}, \text{ d in [mm]}.$$

3) Calculate overland flow depth, d_f:

$$d_f = \left[\frac{q_{wadj}}{\alpha.S^{0.5}}\right]^{\left(\frac{1}{1+\beta}\right)} \quad \text{[ft]}$$

$$q_{wadj} = \frac{q_{padj}}{W} \quad \text{[fps/ft]}$$

W is width of filter strip [ft]

$$q_{padj} = C'.q_{peak}$$

where:

$$C' = 0.0417(0.005 + q_w)^{-0.7157}$$

q_{peak} is determined using modified using rational method or TR 55.

4) Calculate overland flow velocity, V_m

$$V_m = \alpha(df)^{\beta}S^{0.5} \quad \text{[fps]}$$

α and β are taken from Table 4-4.

5) Trapping in the deposition wedge:

$$f = \frac{q_s^C - q_{sd}^C}{q_s^C}$$

q_s^c is the coarse material transport rate, q_{sd}^c is the transport capacity for bed load and:

$$q_{sd} = \frac{K(R_{sd}S_c)^{3.57}}{d_{pd}^{2.07}}$$

where:

$$K = (1.08)^{3.57}\gamma_w g^{1/2} SG(SG - 1)^{-3.07}$$

$$q_s^c = \frac{q_s^t(1 - f_{ri}^l)}{L}$$

$$q_{si}^t = sed_f \frac{Y}{t_b}$$

and:

$$sed_f = \frac{C'^{(a+1)}}{\frac{3600}{3n}\left(\frac{t_b.e}{nt_p}\right)^{k(a+1)}\left[0 + 4e^{\frac{-k(a+1)t_b}{nt_p}} + 2e^{\frac{-2k(a+1)t_b}{nt_p}} + ... + 4e^{\frac{-(n-1)k(a+1)t_b}{nt_p}} + e^{\frac{-nk(a+1)t_b}{nt_p}}\right]}$$

q_s^t is sediment inflow rate into VFS in lb/sec, q_s^c coarse material inflow rate into VFS in lb/sec-ft, Y is sediment load in lb,t_b is duration of storm (hr), C' is a correction factor (see #3), a varies from 0.5 to1, and L is filter strip length perpendicular to the flow in (ft). The fraction of sediment trapped in the deposition wedge is given by:

$$f = \frac{q_s^c - q_{sd}^c}{q_s^c}$$

where q_{sd}^c is the bed load transport capacity downstream of sediment wedge (lb/sec-ft) and q_s^c is bed load transport capacity at the enterance to the VFS.

6) Predicting the advance of the deposition wedge:

$$q_{sba} = \frac{q_s + q_{sd}}{2}$$

q_{si} is incoming sediment load to zone B and q_{sd} is sediment load leaving zone B.

$$f = \frac{q_s - q_{sd}}{q_{si}}$$

$$R_{sb} = \left[\left(\frac{q_{sba}}{K} \right)^{0.28} \left(\frac{\alpha}{q_{wadj}} \right)^{2} d_{pba}^{0.5798} \left(\frac{1}{a_R} \right)^{\frac{2(\beta+1)}{b}} \right]^{\frac{b}{b-2(\beta+1)}}$$

$$S_{et} = \left(\frac{q_{sba}}{K} \right)^{0.28} \frac{d_{pba}^{0.5798}}{R_{sb}}$$

$$V_{tot} = \frac{(f)(sedyld)(1-f_{ri}^{1})}{\gamma_{sb}}$$

a_R is 0.0516 and b is 0.3670, V_{tot} is total volume of sediment deposited (ft^3), Sedyield is Sediment load (lb), $_{sb}$ is bulk density of the material deposited (lb/ft^3), R_{sb} is equilibrium hydraulic radius, K is given in #5, S_{et} is equilibrium slope and Sc is Channel slope

$$\frac{V_2}{V_1} = \frac{S_{et} - S_c}{S_c} = r \qquad \text{(see Figure 4-1)}$$

$$Y_f(t_f) = \begin{vmatrix} \left[\frac{2}{\gamma_{sb}} f' q_{si} S_e (t_f - t_i) + Y_i(t_i)^2 \right]^{1/2} & Y_f(t_f) < H \\ H & Y_f(t_f) = H \end{vmatrix}$$

$$X_f(t_f) = \begin{vmatrix} \left[\frac{2}{\gamma_{sb}} \frac{f' q_{si}}{S_e} (t_f - t_i) + X_i(t_i)^2 \right]^{1/2} & Y_t(t_f) < H \\ X_i(t_i) + (t_f - t_i) \frac{f q_{si}}{H \gamma_{sb}} & Y_t(t_f) = H \end{vmatrix}$$

where H is height of media , $_{sb}$ is the bulk density of the deposited sediment ,$Y_t(t_f)$ and $X_t(t_f)$ are the depth of depositon and advance distance respectively at time $t_{f.}$

If f is the total fraction of coarse particles trapped in the deposition wedge, then the portion trapped in zone B would be:

$$f' = f\left(\frac{r}{1+r}\right)$$

f' is used if $Y_t(t_f) < H$ since only part of sediment flows in to zone B and f is used when the depth of deposition reaches the height of the grass media i.e if $Y_t(t_f) = H$.

7) Determine effective length of settling zone L_n :

For growing season efllective length L_n is given by:

$$\left| \begin{array}{ll} L_n = L_{n-1} - (\Delta X_1 + \Delta X_2 + ... + \Delta X_n) + \frac{1}{D}\left[(n-1)\Delta X_1 + (n-2)\Delta X_2 + ... + \Delta X_{n-1}\right. & n < D] \\ L_n = L_{n-1} - (\Delta X_1 + \Delta X_2 + ... + \Delta X_n) + (\Delta X_1 + \Delta X_2 + ... + \Delta X_{n-D+1}) & \\ \quad + \frac{1}{D}\left[(D-1)\Delta X_{n-D+2} + (D-2)\Delta X_{n-D+3} ... + \Delta X_{n-1}\right] & n \geq D \end{array} \right|$$

For the dormant season L_n is given by:

$$L_n = L_{n-1} - (\Delta X_1 + \Delta X_2 + ... + \Delta X_n)$$

where ΔX_i is advance in the deposition wedge

8) Calculate fall number for coarse, medium and fine size particles:

$$N_f^{c,m,f} = \frac{V_s L_n}{V_m d_f} \qquad \text{[Dimensionless]}$$

9) Calculate Reynolds number, Re:

$$R_e = \frac{V_m R_s}{v} \qquad \text{[Dimensionless]}$$

$$R_s = a_R d_f^b$$

where a_R is 0.0516 and b is 0.3670, Re is Reynolds number, V_m is overland flow velocity [fps], Rs is hydraulic radius [ft] and v is kinematic viscosity [ft^2/s]

10) Calculate T_s for each particle size:

$$T_s = EXP((-1.05)10^{-3} R_e^{0.82} N_f^{-0.91})$$

11) Determine the infiltration rate

$$i_{av} \ = \ \frac{1}{1-\mu}\left[f_c \ + \ \frac{(f_c \ - f_0)}{Kt_b}(e^{-kt_b} \ - \ e^{-\mu kt_b}) \right]$$

i_{av} is the average infiltration rate for a duration of runoff equal to t_b and   is the ratio of time required for the flow to reach VFS to the time base.

12) Determine the dimensionless infiltration term I

$$I = \frac{q_{wadj} - q_{out}}{q_{wadj} + q_{out}}$$

$$q_{out} = q_{wadj} - i.L$$

where i is infiltration rate (in/hr) and L is width of filter strip along flow path.

12) Determine total trapping efficiencies $\mathrm{f}_\mathrm{d}^\mathrm{c}$, $\mathrm{f}_\mathrm{d}^\mathrm{m}$ and $\mathrm{f}_\mathrm{d}^\mathrm{f}$ for coarse, medium and fine particles

$$f_d^{c,m,f} = \frac{T_s + 2I(1-T_s)}{1 + I(1-T_s)}$$

13) Determine f_{ri}^{1} and f_{ri}^{o} based on CREAMS model

Using CREAMS model, fraction of clay is given by:

$$F_{cl} = 0.26 O_{cl}$$

Fraction of small aggregates:

$$\left| \begin{array}{ll} F_{sg} = 1.8 O_{cl} & O_{cl} < 0.25 \\ F_{sg} = 0.45 - 0.6(O_{cl} - 0.25) & 0.25 \leq Ocl \leq 0.5 \\ F_{sg} = 0.6 O_{cl} & O_{cl} > 0.5 \end{array} \right|$$

Fraction of silt:

$$F_{si} = O_{si} - F_{sg}$$

Fraction of sand:

$$F_{si} = O_{sa}(1 - O_{cl})^5$$

Fraction of large aggregates:

$$F_{\lg} = 1 - F_{cl} - F_{si} - F_{sg} - F_{sa}$$

Based on fractions of clay, small aggregates, silt, sand and large aggregates, f_{ri}^1 is calculated as:

$$f_{ri}^1 = 0.5F_{sa} + F_{cl} + F_{si} + F_{sg}$$

and f_{ri}^o is given by:

$$f_{ri}^0 = Fsg + Fcl + Fsi$$

14) Determine total trapping efficiency TE

$$TE = \left[f + f_d^c(1 - f) \right](1 - f_{ri}^1) + f_d^m(f_{ri}^1 - f_{ri}^0) + f_d^f f_{ri}^0$$

Appendix B-2

Derivation of an Explicit Equation for Equilibrium Slope.

1) Solve for R_{sba} from calibrated for Einstein's bed load equation

$$R_{sba} = \left(\frac{q_{sba}}{K}\right)^{0.28} \frac{d_{pba}^{0.5798}}{S_{et}}$$

2) Solve for S_{et} from equation (1)

$$S_{et} = \left[\frac{q_{wadj}}{\alpha}\right]^{2} \left[\frac{1}{d_f}\right]^{2(\beta+1)}$$

3) Solve for d_f from equation for hydraulic radius $\left(R_{sba} = a d_f^{b}\right)$

$$d_f = \left[\frac{R_{sba}}{a}\right]^{1/b}$$

4) Solve for S_{et} as a function of R_{sba} by replacing d_f in equation (2) by the expression in step 3

$$S_{et} = \left[\frac{q_{wadj}}{\alpha}\right]^{2} \left[\frac{a}{R_{sba}}\right]^{2(\beta+1)/b}$$

5) Replace S_{et} in step (1) by the expression given in step and solve for R_{sba} to get the hydraulic radius at equilibrium slope in the deposition zone, or:

$$R_{sba} = \left[\left(\frac{q_{sba}}{K}\right)^{0.28} \left(\frac{\alpha}{q_{wadj}}\right)^{2} d_{pba}^{0.5798} \left(\frac{1}{a}\right)^{\frac{2(\beta+1)}{b}}\right]^{\frac{b}{b-2(\beta+1)}}$$

6) Use the R_{sba} obtained using the above equation to solve for equilibrium slope.

$$S_{et} = \left(\frac{q_{sba}}{K}\right)^{0.28} \frac{d_{pba}^{0.5798}}{R_{sb}}$$

Appendix B-3

Derivation of Grass Recovery Equation

$L_n = L_{n-1}$ - Advance of the deposition wedge + Recovery

L_n = effective Length of filter strip on a given day and L_{n-1} is Length of filter strip on the previous day.

Growing season

Both deposition and recovery are expected during growing season.

Assume D = the number of days required for complete recovery

n = day count

Δx_i is the advance in the deposition wedge after each day

Case 1) n < D

Day	Advance	Recovery
1	X_1	0
2	$X_1 + X_2$	$\frac{1}{D} X_1$
3	$X_1 + X_2 + X_3$	$\frac{2}{D} X_1 + \frac{1}{D} X_2 = \frac{1}{D}(2 X_1 + 1 X_2)$
4	$X_1 + X_2 + X_3 + X_4$	$\frac{3}{D} X_1 + \frac{2}{D} X_2 + \frac{1}{D} X_3 = \frac{1}{D}(3 X_1 + 2 X_2 + 1 X_3)$
. . .		
n = D	$X_1 + X_2 + X_3 + \ldots X_n$	$\frac{(n-1)}{D} X_1 + \frac{(n-2)}{D} X_2 + \frac{(n-3)}{D} X_3 + \ldots + 1 X_{n-1}$ $= \frac{1}{D}((n-1) X_1 + (n-2) X_2 + \ldots + 1 X_{n-1})$

Case 2) $n > D$, if $n=D+1$, advance on day 1, X_1 is assumed to recover completely and if $n = D+2$, both X_1 and X_2 are recovered and so on.

Day	Advance	Recovery
D+1	$X_1+ X_2+ X_3+\ldots+ X_n$	$(X_1)+\frac{(D-1)}{D} X_2+\frac{(D-2)}{D} X_3+\ldots+\frac{1}{D} X_{n-1}$
D+2	$X_1+ X_2+ X_3+\ldots+ X_n$	$(X_1+ X_2)+\frac{(D-1)}{D} X_3+\frac{(D-2)}{D} X_4+\ldots+\frac{1}{D} X_{n-1}$
D+3	$X_1+ X_2+ X_3+\ldots+ X_n$	$(X_1+ X_2+ X_3)+\frac{(D-1)}{D} X_4+\frac{(D-2)}{D} X_5+\ldots+\frac{1}{D} X_{n-1}$
.		
.		
.		
n	$X_1+ X_2+ X_3+\ldots+ X_n$	$(X_1+ X_2+ X_3+\ldots+ X_n)+(X_1+ X_2+ X_3+\ldots+ X_{n-D+1})$ $+\frac{1}{D}\left[(D-1)\Delta X_{n-D+2} + (D-2)\Delta X_{n-D+3}\ldots + \Delta X_{n-1}\right]$

VITA

Mengistu Geza Nisrani

Candidate of the Degree of

Doctor of Philosophy

Dissertation: ENVIRONMENTAL AND ECONOMIC MODELING OF NON-POINT SOURCE POLLUTION CONTROL: OPTIMIZED LAND USE SYSTEMS AND VEGETATIVE FILTER STRIPS

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Pages in Study: 229 Candidate for the Degree of Doctor of Philosophy

Major Field: Biosystems Engineering

Scope and Method of Study: The objectives of the study were to evaluate environmental and economic impacts of Non-Point Source Pollution (NPS) control approaches on a watershed scale. A Land Use Decision Model (LUDM) written using mathematical programming model (GAMS) was used to determine the optimal land use systems that maximize net returns subject to sediment and nutrient load constraints and to determine the optimal land use systems for efficient utilization of water quality incentives to minimize sediment and nutrient loads. Uniform and non-uniform sediment and nutrient load reduction approaches were also compared. Environmental and economic benefits of switchgrass production and currently used Conservation Reserve Program (CRP) were compared using LUDM model. A modified procedure for computing sediment trapping in grass filters, SGRASSF, was developed based on previously developed GRASSF vegetative filter strip model to evaluate sediment trapping efficiency on a watershed scale. Economic and environmental impact of use vegetative filter strips (VFS) was compared to total optimal replacement of parts of the watershed by grass. Soil and Water Assessment Tool (SWAT) was used to estimate sediment load in both approaches. In the optimal replacement approach, the LUDM model was used to determine the optimal land distribution.

Finding and Conclusion: LUDM built using GAMS is a useful tool to make cost effective land use decision to achieve environemtnal goals. The loss in income for the same amount of load reduction, as a result of replacement of peanuts by switchgrass is less than it is for replacement by CRP. The incentive required per ton of sediment or nutrient reduced as a result of replacement by CRP and minimum tillage wheat is higher than the payment required for replacement by switchgrass. The results show that whole watershed based non-uniform sediment and nutrient load reduction approach is more cost effective than Hydrologic Response Unit based uniform reduction approach. SGRASSF gives similar results to GRASSF model with an R^2 value equal to 0.92. The Nash-Sutcliffe coefficient, used as an indicator of goodness of fit, was determined to be 0.9. Placement of vegetative filter strips along field drains is more cost effective compared to optimal replacement of parts of the watershed using LUDM approach.

ADVISOR'S APPROVAL: Dr. Bill Barfield_________