STOCHASTIC OPTIMIZATION OF NON-POINT SOURCE POLLUTION ABATEMENT FOR THE EUCHA-SPAVINAW WATERSHED

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CHAPTER I

INTRODUCTION

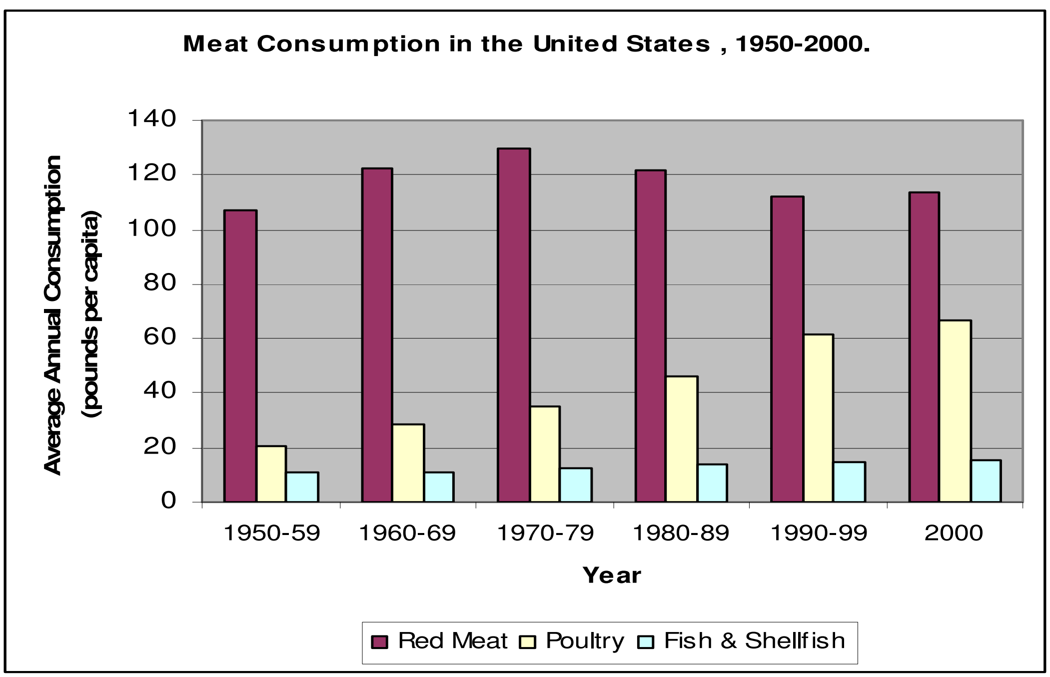
Background

The National Academy of Science (NAS 2001) reported that since the early 1970s, water quality management in the United States hinged on the control of point sources of pollution and the use of effluent-based water quality standards. The quality of U.S. water bodies generally improved as point sources (for instance, wastewater treatment plants and industrial dischargers) complied with requirements of the 1972 Clean Water Act. Polluters have been required to meet effluent-based standards for criteria pollutants spelled out in National Pollutant Discharge Elimination System (NPDES) permits issued by respective states with the approval of the United States Environmental Protection Agency (USEPA). However, the NPDES program failed to achieve water quality goals of "fishable and swimmable" waters largely because of unsuccessfully controlled pollution coming from unregulated non-point sources. The Clean Water Act did not consider pollutants such as nutrients and sediment (often associated with non-point sources) as criteria pollutants. These unsuccessfully controlled nutrient- and sediment discharges from non-point sources continue to jeopardize water quality and the environment across the United States such that the focus of water quality management has shifted from effluent-based to ambient-based water quality standards.

The U.S. Environmental Protection Agency implements the Total Maximum Daily Load (TMDL) program with the objective of attaining ambient water quality standards by controlling both point and nonpoint sources of pollution (NAS, 2001; Younos, 2005).

Degradation of water quality and the environment is a major concern in the Eucha-Spavinaw basin. Spavinaw Lake and downstream Lake Eucha supply drinking water to Tulsa and surrounding communities in Oklahoma. Excessive amounts of nutrients coming from the watershed into the lakes promote growth of algae that degrade the water quality. The Oklahoma Conservation Commission (OCC, 1997) pointed out that the major nutrient of concern is phosphorus which runs off from cropland in the watershed on which poultry litter is applied as fertilizer. It is estimated that annually about 41 tons of phosphorus and 959 tons of nitrogen entered Lake Eucha from the watershed in the early 1990s. Poultry is the main industry in the Eucha-Spavinaw watershed.

Studies conducted by the USDA Economic Research Service (USDA-ERS) show that annual average total meat consumption (red meat, poultry, fish) in the United States increased substantially in the period from 1950 to 2000. Per capita total meat consumption was estimated at 195 pounds in the year 2000, representing an increase of 41.2 percent above average annual consumption recorded in the 1950s. A closer look at the per capita consumption levels of the individual meat categories (see Figure 1) suggests that each person ate an average of 4 pounds more fish and shellfish, 7 pounds more red meat and 46 pounds more poultry in 2000 than in the 1950s (USDA-ERS, 2000). However, the total amount of red meat consumed by each person increased substantially between the 1950s and 1960s and continued to rise at a slower rate until it reached a maximum of 129.4 pounds in the 1970s. USDA-ERS (2000) attributed this increase in consumption of red meat to rising consumer incomes and lower real prices for meats during that period. Per capita consumption of red meat then declined from the 1970s onwards due to nutritional concern among consumers about fat and cholesterol in their diet that necessitated substitution of other meats (poultry and fish) for red meats in order to lower total fat and saturated fat intake. Though per capita consumption of poultry increased steadily between the 1950s and 1970s, the increase was substantial thereafter such that the average total annual consumption reached 68.4 pounds by the year 2000 from 19.8 pounds in the 1950s. Per capita consumption of fish and shellfish increased steadily between the 1950s and 2000 (USDA-ERS, 2000).



Source : United States Department of Agriculture / Economic Research Service , 2000

Figure 1 Average Annual Meat Consumption Per Capita in the U.S., 1950-2000.

The continuous growth in the demand for poultry since the 1950s resulted in the rapid expansion of the poultry industry in the United States. It is estimated that the number of poultry farms in the U.S. reached 51,423 in 1997, with recorded production of approximately 8 billion birds (See Table 1). However, in half a decade later, the number of poultry farms and birds produced had risen to 67,256 and about 9 billion, respectively. This represents about 14.4 percent increase in poultry production by the year 2002. Broiler production accounts for about 92 percent and 93 percent of U.S. total poultry production in 1997 and 2002, respectively. The States of Arkansas and Oklahoma combined produced about 1.3 billion and 1.5 billion birds in 1997 and 2002, respectively. These production levels represent about 16 percent of the national poultry production in the respective years. Arkansas produced about 84 percent of the combined total poultry production for the two states in the years 1997 and 2002. On the basis of dry manure estimation assumptions suggested by Sims et al. (1989), the American nation produced roughly 41 million tons of dry poultry manure in 1997. The national manure production reached 47 million tons by 2002, an increase of about 14.4 percent. Evers (1996) reported that farmers use more than 95 percent of national poultry litter as crop and pasture fertilizer. The states of Oklahoma and Arkansas combined produced about 6.5 million and 7.4 million tons of poultry manure in 1997 and 2002 respectively representing approximately 16 percent of the national dry poultry manure production in the respective years.

Table 1 Poultry and Dry Manure Production in the U.S., 1997 and 2002.

	No. of Farms			No. of Birds			Dry Manure (tons)		
1997	US	AR	OK	US	AR	OK	US	AR	OK
Layers	12,789	853	423	194,945,215	12,985,428	3,330,062	1,364,617	90,898	23,310
Broilers	27,737	3,882	751	7,366,526,456	1,046,510,017	197,077,480	36,095,980	5,127,899	965,680
Pullets	4,052	315	110	142,094,811	13,849,439	1,797,309	383,656	37,393	4,853
Turkeys	6,845	291	73	307,605,599	25,453,838	1,748,693	3,352,901	277,447	19,061
Total	51,423	5,341	1,357	8,011,172,081	1,098,798,722	203,953,544	41,197,153	5,533,637	1,012,904
2002									
Layers	18,621	643	635	202,947,490	9,124,085	3,027,523	1,420,632	63,869	21,193
Broilers	32,006	3,520	871	8,500,313,357	1,181,907,700	231,877,714	41,651,535	5,791,348	1,136,201
Pullets	8,193	324	373	174,916,701	14,811,501	3,316,431	472,275	39,991	8,954
Turkeys	8,436	292	115	283,247,649	28,459,783	933,382	3,087,399	310,212	10,174
Total	67,256	4,779	1,994	9,161,425,197	1,234,303,069	239,155,050	46,631,842	6,205,419	1,176,522

Source : Adapted from US Census of Agriculture, 1997 and 2002

The Eucha-Spavinaw watershed is located in the northeastern Oklahoma and western Arkansas (Ancev, 2003). Table 2 shows this watershed drains 245,591 acres in Mayes County and Delaware County, Oklahoma (64.2 percent), and Benton County, Arkansas (35.8 percent). However, the watershed covers about 1 percent of Mayes, OK, 17 percent of Benton, AR and 30 percent of Delaware, OK.

Table 2 Eucha - Spavinaw Watershed Area by County.

County / State	Area of County (acres)	County Area in the watershed (acres)	Share of Total County Area (%)
Benton, AR	534,424	87,952	16.5
Delaware, OK	511,698	153,171	29.9
Mayes, OK	425,768	4,468	1.0
Total	1,471,890	245,591	47.4

Source: Adapted from Ancev (2003);

Storm et al.(2002) found that just over half of the Eucha-Spavinaw watershed area is occupied by forests while pastures account for 42.7 percent (see table 3). They classified pastures into three categories: hayed, poorly- and well-maintained pastures. They found 53.8 percent of the pastureland was well-maintained, that 30.9 percent was hayed, that 15.2 percent was poorly-managed and that 2.7 percent of the watershed area was used for row crop. Brushy rangeland, urban and water together occupied 3.7 percent of the watershed area (Storm et al. 2002).

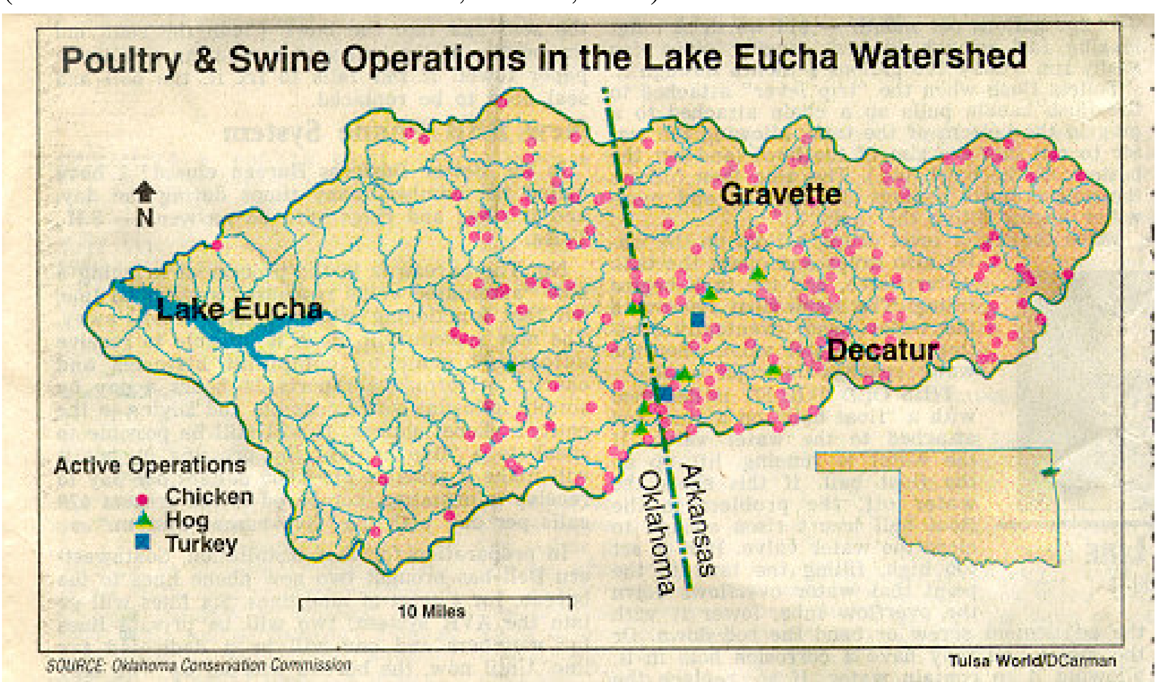
Table 3 Land Use in the Eucha-Spavinaw Watershed.

Land Use	Watershed Area Covered (%)
Forest	50.9
Hayed Pastures	13.2
Well Managed Pastures	23.0
Poorly Managed Pastures	6.5
Brushy Rangeland	0.3
Urban	1.5
Water	1.9
Row Crop	2.7

Source: Storm et al. (2002)

Problem Statement

Total Maximum Daily Loads (TMDLs) are being implemented to prevent eutrophication of public water supplies by phosphorus runoff from manure applications in many watersheds in the United States. Agricultural pollution attributed to excessive land application of poultry manure as fertilizer is a serious environmental problem for surface water quality in the Eucha-Spavinaw watershed situated on the border of the states of Oklahoma and Arkansas as shown in Figure 2 below. The Eucha-Spavinaw watershed has been troubled by water pollution for years and has created considerable controversy between the two states. The city of Tulsa, Oklahoma and the Tulsa Metropolitan Utility Board filed a lawsuit in December 2001, naming several poultry integrators as defendants (Oklahoma Water Resources Board, OWRB, 2002).



Source : Oklahoma Conservation Commission

Figure 2 Poultry and Swine Operations in the Lake Eucha Watershed.

The Eucha-Spavinaw watershed is of interest because Lake Eucha and Spavinaw Lake are currently on the Environmental Protection Agency (EPA) 303(d) Impaired Water List due to low dissolved oxygen and excessive phosphorus from municipal point source discharges, agriculture, and other unknown sources (ODEQ, 2004). The Oklahoma Water Quality Standard specifies the designated beneficial uses of Lake Eucha and Spavinaw Lake as including public and private water supply, aquatic community, agricultural irrigation, recreation and aesthetics, and sensitive drinking water supply (OWRB 2004; 2006). There is rapid urban expansion in an adjacent watershed, historical growth of poultry production and very little cropland within the Eucha-Spavinaw watershed. The rate at which poultry litter is currently being produced and land applied is most likely to lead to excessive phosphorus runoff levels from agricultural land into the water bodies during storm events in the watershed. Most of the non-point nutrient pollution comes from poultry manure fertilized pastures (OWRB, 2002; Storm et al., 2003).

Eutrophication threatens the Tulsa metropolitan water supply. Excessive phosphorus loading has led to excessive algae blooms in Lake Eucha and Spavinaw Lake that cause oxygen depletion in both lakes. Excessive levels of phosphorus and algal growth impair the designated aesthetics, recreational and drinking water beneficial uses of the two lakes by causing undesirable taste and bad odor. Municipal water treatment facilities that treat the water to achieve established drinking water standards find it difficult and prohibitively expensive to remove the bad taste and odor in drinking water. The City of Tulsa reported additional water treatment costs due to excessive algae exceeding $72.78 per million gallons. Should their current treatment system be unable to eliminate the taste and odor problems, the City of Tulsa will have to either increase water treatment costs or abandon lake Eucha and Spavinaw lake as a water supply entirely and look for alternative drinking water supply such as Lake Hudson. The additional costs of using Lake Hudson water was estimated to exceed $7,000 per day whereas the cost of abandoning lakes Eucha and Spavinaw as a water supply and using Lake Hudson was estimated to exceed $250 million (City of Tulsa 2006; OWRB, 2006).

There is also a public health risk associated with excessive phosphorus and algae levels in both lakes. Lakes Eucha and Spavinaw are reported as supporting sufficiently large amounts of Bluegreen algae. This algae species release microcystins that can cause liver damage; other forms can be neurotoxic and cytotoxic. Both lakes are also impacted by increases in disinfection byproduct precursors such as total organic carbon (City of Tulsa 2006; OWRB, 2006).

There is need for phosphorus reduction programs and nutrient management plans to reduce both point and nonpoint source nutrient pollution in the Eucha-Spavinaw watershed, especially that coming from agriculture. Therefore best management practices (BMPs) to reduce phosphorus loading in the watershed are of high interest, not only to poultry integrators and farmers using poultry manure, but also to municipal authorities, recreation managers, regulators, policy makers and the general public. Although several studies have analyzed nitrogen and phosphorus loading in the watershed, few studies have analyzed the role of grazing management systems as a profitable economic enterprise and a phosphorus reduction strategy under stochastic conditions from a watershed where large quantities of litter were available for use as fertilizer on pastures to achieve the recommended phosphorus load reductions for the watershed at minimum cost to society. The research to be undertaken in this dissertation addresses the question, "What is the most efficient set of litter and grazing management practices that can be used to maximize net agricultural income while meeting the recommended phosphorus load reductions for the Eucha-Spavinaw watershed within specified margins of safety?" The answer to this question will help the many interested groups identify least-cost BMPs and policy instruments to implement to reduce water pollution from phosphorus runoff in the watershed. The results of this study will allow all affected parties to make better decisions concerning cost effective phosphorus pollution abatement from non-point sources in the Eucha-Spavinaw watershed.

Objectives of the Study

This study analyses the economic and environmental impacts of watershed-scale adoption of various pasture management practices in the Eucha-Spavinaw watershed. The main focus is on maximizing net agricultural income from grazing while meeting specified environmental-improving phosphorus pollution restrictions for the Eucha-Spavinaw watershed at least social cost within specified margins of safety. The specific objectives are:

a) To develop an integrated biophysical - economic optimization model for cost efficient non-point source pollution abatement in the Eucha - Spavinaw watershed.

b) To determine the least cost mix, location, and magnitude of grazing management practices to reduce phosphorus loading under various phosphorus loading targets and margins of safety for the Eucha-Spavinaw watershed.

c) To determine the optimal transportation pattern for poultry litter under various phosphorus loading targets and margins of safety for the Eucha-Spavinaw watershed.

d) To determine the efficiency of changes in pasture management practices in reducing phosphorus runoff relative to the use in a possible litter-to-energy power plant under various phosphorus loading targets and margins of safety for the Eucha-Spavinaw watershed with and without an alum-treated poultry litter option.

e) To determine the effect of different soil types, hydrology and management practice variables on phosphorus runoff in the Eucha-Spavinaw watershed.

Contribution of the Dissertation

This dissertation research developed a comprehensive decision-support tool, an integrated biophysical-hydrologic – economic watershed model, with the ability to reflect the dynamic interactions of essential biophysical, hydrologic, agronomic, and economic components and to explore both the economic and environmental consequences of a wide variety of management practices and policy choices for the Eucha-Spavinaw watershed. The study built on, improved and extended some of the hydrological and economic studies that have been conducted in the Eucha-Spavinaw watershed and provided a simple and more realistic framework that combined GIS-based hydrological simulations with mathematical programming to effectively determine optimal amounts of control at non-point pollutant sources to meet the recommended phosphorus load reductions for a watershed at minimum social cost. The Soil and Water Assessment Tool (SWAT) model that has been used to simulate effects of various BMPs on nutrient and sediment discharges in watersheds allowed researchers to use consistent scientific methods to estimate the effects of each possible management practice in a given hydraulic response unit (HRU). An HRU refers to an area of land representing a combination of a major soil type and land use within a subbasin. However, the SWAT simulation is not an optimization model. This study introduced optimization into the analysis by integrating SWAT simulation model with a mathematical programming model to determine site-specific management practices that would meet the recommended phosphorus load reductions for the watershed at minimum social cost.

Studies that have investigated the economic feasibility of converting litter-to-energy and commercially saleable fertilizer in the Eucha-Spavinaw watershed focused on private firm-level costs and benefits and did not take into account pollution from point and non-point sources in their analyses. This study built on this previous research work and incorporated non-point source polluters in the watershed to capture societal costs and benefits consistent with meeting the recommended phosphorus load reductions for the watershed. We developed and applied an integrated biophysical and economic methodology to determine the costs of investing in abatement efforts and converting litter-to-energy in the watershed.

The research determined management practices for particular areas within the watershed that would effectively control phosphorus pollution in a way that is least costly to society using an updated SWAT model, recent and larger datasets with much greater spatial detail. This improved on a few studies in the watershed that had employed a similar approach but at a lesser degree of spatial detail. The results obtained from the simulation of the long term effects of management practices on the amount of phosphorus runoff and production levels of land in specific areas in the watershed would provide farmers and watershed managers with information to aid them make better decisions about production methods and levels that can sustain both water quality of the lakes and productivity of the farmland.

We incorporated both environmental impacts and costs of meeting recommended phosphorus load reductions in the analysis and demonstrated that TMDL programs can be improved by using economic analysis of costs and benefits to set and implement TMDL goals and standards so as to achieve efficient targeting of pollution reductions while distributing costs among polluters (both point and non-point sources) equitably. This watershed-level economic study of agricultural pollution would serve as an additional resource in the growing public debate surrounding agricultural pollution in the watershed and the implementation of market-based mechanisms in environmental policy.

CHAPTER II

LITERATURE REVIEW

Background

Lakes Eucha and Spavinaw make up a single surface water system. They are two impoundments on Spavinaw Creek with Lake Spavinaw downstream (approximately four miles) of Lake Eucha. Lake Eucha (established in 1952) receives a great majority of its water and nutrients from Spavinaw Creek. Lake Spavinaw (impounded in 1924) receives most of its water and nutrients from the Lake Eucha dam discharge; therefore, Lake Eucha provides a continuous water supply to Lake Spavinaw (Oklahoma Conservation Commission, OCC, 1997). The Tulsa Metropolitan area and other small communities receive their drinking water from the Lake Eucha-Spavinaw complex (Oklahoma Water Resources Board, OWRB, 2002).

Prior to the creation of Lake Eucha, the general land usage of the watershed was farming corn, wheat, and oats (Kesler, 1936). In the western Arkansas portion of the watershed, apple and peach orchards along with vineyards were abundant. Nearly 80 percent of the watershed's land area was scrub timber before Lake Eucha was created (Kesler, 1936). Since Lake Eucha was created, the primary land use is forest and pasture, and is in the Ozark Plateau of northwest Arkansas and northeastern Oklahoma where the underlying geology is karstic. Agricultural practices in the Eucha-Spavinaw basin include grazing cattle, small dairy production, confined animal operations (poultry and swine), land application of animal wastes and some row crop production. Land within the watershed has been used to support the commercial poultry industry, with the capacity to produce over 84 million birds, along with some 1500 tons of phosphorus rich waste per year (Tulsa Metropolitan Utility Authority, TMUA, 2001).

The Eucha-Spavinaw system is designated by the OWRB (1996, 2002) as a system for public water supply along with recreation, fish and wildlife, and aesthetics (OCC, 1997; TMUA, 2001). These designated uses are important because they ultimately determine the "acceptable" pollution level. The Eucha-Spavinaw basin contains two rural wastewater treatment plants situated in Gravette and Decatur, Arkansas. The facility at Gravette treats water from a residential community and has a design discharge of 0.56 million gallons per day (mgd). The stream receiving the point source discharge is an intermittent system which frequently has no surface flow entering Spavinaw Creek. The Decatur facility treats wastewater from a residential community and a poultry processing plant. This facility discharges approximately 1.6 mgd (OCC, 1997) into the receiving stream (Columbia hollow) and Spavinaw Creek. While point source pollution is a significant contributor to reservoir nutrient loading, nonpoint sources still contribute a greater proportion of the nutrients entering the Eucha-Spavinaw water system (OCC, 1997).

A diagnostic study of Lakes Eucha and Spavinaw conducted by OWRB (2002) indicated these lakes are eutrophic (i.e. Lakes Eucha and Spavinaw are nutrient-enriched and display high or excessive levels of algal production). Phosphorus was the limiting nutrient during most of the project period. The annual phosphorus budget analysis for Lake Eucha showed that 93 percent of the phosphorus entering the lake originated in the drainage basin, and most of that entered the lake through Spavinaw Creek, the lake's main tributary. The remaining 7 percent of the phosphorus entering the lake came from lake sediments. On an annual basis, the phosphorus in the discharge from Lake Eucha accounts for about 85 percent of the phosphorus entering Spavinaw Lake (OWRB, 2002).

Several agricultural best management practices (BMPs) have been implemented in other watersheds in the United States to deal with non-point source nutrient pollution of water bodies. A BMP is a practice or combination of practices chosen the most effective, economical, and practical means of preventing or reducing the amount of pollution generated by nonpoint sources to a level compatible with state and local water quality goals. It is a BMP (instead of just an "MP") because: it works; it is possible to use it (i.e., not unduly complicated); and it is a "good buy" compared to alternatives. However, the selection of an appropriate BMP will depend greatly upon the site conditions (land use, topography, slope, water table elevation, and geology) (Cestti, Srivastava and Jung, 2003). In general, these practices are designed to effectively use agricultural chemicals; increase ground cover, decrease the velocity of surface runoff, and improve the management of livestock waste. Controlling erosion is an essential aspect of preventing nutrient non-point source pollution of surface waters as eroding soil particles will carry excess nutrients, particularly phosphorus, into water bodies (Cestti, Srivastava and Jung, 2003). To address agriculture nutrient non-point source pollution, farmers can use either structural measures (i.e. waste containment tanks / lagoons, sediment basins, terraces, diversion, fencing, tree plantings) or managerial ones (i.e. nutrient budgeting, rotational grazing, and conservation tillage). In either case, good management is always a necessary condition for reducing farm pollution (Gale et al., 1993).

Agriculture best management practices can be grouped according to their functions. The USEPA (1993) guidelines identify the following categories:

- Managing sedimentation. Measures to control the volume and flow rate of surface water runoff, keep the soil in place, and reduce soil transport. Such BMPs include permanent vegetative cover, strip cropping systems, terrace and diversion systems, grazing and cropland protection systems, waterway and stream protection systems, conservation tillage systems, sediment retention and erosion.
- Managing nutrients. Measures to help to keep the nutrients in the soil, minimizing their movement into the water bodies. Such measures include permanent vegetative cover, animal waste management systems, strip cropping systems, terrace system, grazing and cropland protection systems, conservation tillage systems, tree plantings and fertilizer management.
- Managing pesticides. Measures to reduce non-point source contamination from pesticides, by helping limiting pesticide use and managing its application. Such BMPs include strip cropping systems, terrace systems, and pesticide management plans.

- Managing confined animal facility. Measures to reduce or limit the discharge from confined animal facilities. Animal waste management systems fall in this category.
- Managing livestock grazing. Measures to reduce impacts of grazing on water quality. BMPs in this category include permanent vegetative cover, diversion systems, grazing land protection systems, waterway and stream protection systems, conservation tillage systems, and tree plantings or riparian forest buffer.

Wheat Production and Management Practices Used in Oklahoma

Krenzer (1994) reported that more than 6 million acres of cropland in Oklahoma are seeded to winter wheat every year for grain-only, forage-only, or as a dual purpose forage and grain crop. Wheat forage has high nutritive value and provides excellent weight gains for livestock. Forage is available at different times of the year depending on the forage system. Forage-only systems have wheat forage available for grazing by livestock in late fall, winter and early spring. These are times when other forage sources would be low in quantity and quality. Dual purpose systems have wheat forage available for grazing by livestock from mid-November. Livestock graze on the wheat until development of the first hollow stem. This allows the wheat to mature and produce a grain crop, usually harvested in June (Krenzer, 1994).

Krenzer (2000) noted that wheat production practices differ according to intended use. Recommended planting dates for forage-only wheat are usually 2-6 weeks before planting dates for grain-only wheat. Forage-only wheat also has greater seeding rates than grain-only wheat (Krenzer 2000). Hossain et al (2004) surveyed Oklahoma wheat, wheat pasture and wheat pasture livestock producers to determine production methods, management practices, and lease arrangements they use. The study found that Oklahoma farmers planted 6.1 million acres to wheat in the fall of 1999. About 61 percent of the wheat acres were grazed mostly to stocker cattle and cows and /or replacement heifers. The average stocking rates were 2.1 acres / steer and 2.0 acres / heifer whereas the animals gained about 2 lbs on average daily. Statewide, 20 percent of the wheat acreage was intended for forage only, 49 percent for dual purpose, and 31 percent for grain only, but due to weather constraints use was 22 percent, 39 percent, and 39 percent, respectively. Farmers intended and actually used more acreage for forage-only than for dual purpose and grain-only in 1999-2000. Respondents indicated average target planting dates of September 13 for forage-only, September 20 for dual purpose, and October 2 for grain-only. Average reported seeding rates were 94lb/acre for forage-only, 84 lb / acre for dual purpose, and 77 lb/acre for grain-only. Nitrogen use averaged 69 lb/acre, 69 lb/acre, and 63 lb/acre for forage-only, dual purpose, and grain-only, respectively. Approximately 886,000 steers and 466,000 heifers were stocked on Oklahoma wheat pasture during the 1999-2000 season. On average, the beginning weights for steers and heifers were 460 lb and 447 lb, respectively (Hossain et al., 2004).

Possible Uses of Poultry Litter

Most poultry houses use wood shavings or sawdust as bedding material. The mixture of manure, feed, feathers, and bedding material from these houses is commonly referred to as "poultry litter." Several studies conducted in the Eucha-Spavinaw watershed indicate that poultry litter application rates to agricultural land to supply crop nutrient are high and exceed crop nutrient requirements. For instance, Storm et al. (2003) showed that there is a high correlation between phosphorus loading and litter application in the Eucha-Spavinaw basin. They estimated the average annual total phosphorus loading in the watershed at 47.6 metric tons. However, a study conducted by Ancev (2003) determined that the socially optimum phosphorus loading for the area falls between the range of 23 to 26 tons per year. Thus, there is greater need to reduce phosphorus loading in the Eucha-Spavinaw watershed. The OWRB (2004) recommended reductions in annual total phosphorus loading into lakes Eucha and Spavinaw by 54 and 47 percent, respectively. The literature suggests several ways of using or disposing of poultry litter that may be adopted to abate phosphorus pollution. The following sections outline some of the possible poultry litter disposal practices.

Use of Fresh Poultry Litter as Fertilizer

Several studies have determined that poultry litter is rich in nutrients and organic matter. USDA (1995) estimated that annual litter from a typical broiler housing 22,000 birds contains as much phosphorus as is in the sewage from a community of 6,000 people. Given that land application of litter is relatively simple and inexpensive compared to commercial fertilizers, farmers use poultry litter as a low-cost crop fertilizer with the potential of returning essential nutrients and organic matter to the soil to improve its structure and fertility. However, poultry litter is generally applied to meet the nitrogen requirement of the crop. The problem that arises from such a practice is excessive application of phosphorus owing to the lower nitrogen-to-phosphorus ratio in poultry litter compared to most crop nutrient requirements. In some areas, farmers have been encouraged to apply poultry manure on the basis of the soil test phosphorus (STP) index to reduce the chances of phosphorus runoff. The STP index represents the required amount of phosphorus beyond which additional phosphorus will not increase the maximum yield. That means the soil has sufficient phosphorous for plant uptake at that index such that continued application of manure above this level may lead to phosphorous runoff during a storm event. Evers (1998) estimated that 95 percent of U.S. poultry litter is applied to agricultural land as fertilizer. Controlling the runoff from the farms is a very important component of the total program to control pollution from non-point sources. It is further observed that land application is confined to areas near poultry production. The poultry are produced in spatially concentrated areas to minimize feed and transportation costs (Cestti et al., 2003; Storm et al. 2003).

Use of Alum-Treated Poultry Litter as Fertilizer

A study has found that adding aluminum sulfate to poultry litter provides benefits for both the farmer and the environment (Cestti et al., 2003). Addition of alum to poultry litter traps the nitrogen in the fertilizer. This in turn, reduces nitrogen losses through ammonia volatilization and increases the level of nitrogen available to plants. The alum would also tie up soluble phosphorous in the fertilizer. This chemical process transforms the soluble phosphorus into more stable aluminum phosphate compounds that are insoluble and thereby significantly reducing the potential for soluble phosphorus runoff once the litter is applied to agricultural land. The aluminum phosphate compounds are also not readily available for plant and algae uptake in water bodies. Though alum can be applied directly to agricultural land, research has shown that the alum can easily be added to litter in the poultry house in alum-to-litter ratio of 1 part alum to 10 parts litter (Moore and Miller, 1994; Moore 1999 and Moore et al., 2000). However, despite these beneficial effects of alum-treated litter, this practice may not be sustainable given the high cost of litter management owing to both large quantities of litter to be treated and availability and price of the aluminum sulfate. Simpson (1998) found that financial incentives such as a cost-sharing, tax credit, tax incentive and low interest rate loans have been offered to farmers in various States to encourage them adopt this litter management practice.

Use of Composted Poultry Litter as Fertilizer

As the poultry industry expands in eastern Oklahoma and western Arkansas, farmers are faced with the challenge of disposing of increasing volumes of poultry litter. Walker (2002) recommended composting as one option that producers may consider as a way of increasing the value and potential markets for their litter, while moving excess nutrients from their operations. Composting is a simple, natural process (an aerobic degradation of biodegradable organic waste) poultry producers can use to produce a marketable product. Composted poultry litter has few, if any, odors. It is a more stable and more consistent material than fresh litter, so is less likely to damage plants. Many potential on-farm and off-farm uses and markets exist for compost as an organic fertilizer, including the nursery industry, organic growers and vegetable producers, homeowners, golf courses, highways and land reclamation. The off-farm removal of poultry litter as compost is an environmentally sound method of removing excess nutrients from many land-limited operations, and can be an important way of protecting agricultural land and surface waters from the excessive loading of litter nutrients. Studies conducted by Tyson (1994) and Preusch et al.(2002) found that composted litter releases nitrogen more slowly and over a longer period of time than fresh litter, with only 10 percent of the total nitrogen in the composted litter available for plant uptake in the first year, compared to 30-50 percent in the fresh broiler litter. Thus, composting has the potential of limiting nitrogen leaching to ground water through formation of more stable organic components. Composted broiler litter produces less phosphorus runoff than fresh broiler litter. However, economic analysis studies conducted by VerVoort and Keeler (1998) and Kelleher et al. (2002) showed that the practice of using composted broiler litter lead to a loss of nutrients (especially nitrogen) required by plants and was much more expensive compared to the alternative of applying fresh broiler litter directly to cropland owing to increased demand for land and additional equipment and labor costs.

Use of Poultry Litter as Cattle Feed Ingredient

Broiler litter can be good feed source for cattle during the winter or times of drought, particularly for brood cows and stocker cattle. Good-quality broiler litter is approximately equal to good-quality alfalfa hay, based on nutrient analysis. Broiler litter fed to cattle is usually mixed with a more palatable feed, such as corn. Any number of palatable feeds in addition to corn can be used to mix with broiler litter, such as wheat, milo, commercial grain mixes, and soybean hulls to increase consumption. Diets containing broiler litter can produce acceptable levels of performance by beef cattle. Using a lower-energy-based diet, Cross and Jenny found gains of feedlot steers were similar between cattle fed diets containing corn silage with either 0, 10, or 30 percent broiler litter substituted for corn silage. A study conducted by McCaskey et al. (1994) found that beef steer gains were 2.53 pounds per day on a concentrate diet as compared with 2.12 pounds per day on a diet of 50 percent broiler litter and 50 percent corn. Burdine, et al. (1993) and Bagley, et al.(1994) report that animal performance was the same with broiler litter diets mixed with either corn or soybean hulls. These studies demonstrated that beneficial uses of broiler poultry litter as a cattle feed, among others, include environmental protection through responsible use of an animal by-product, increased sale value of the by-product for poultry producers, and economic benefit for production of beef cattle as a low-cost protein feed source. Despite these benefits, the feeding of poultry litter, however, was not a widespread practice. It was estimated that less than 1 percent of the total amount of poultry litter generated in the United States was fed to cattle. However, fresh broiler litter should be processed to ensure its safety from potentially harmful pathogens (Davis, 1999; U.S. FDA, 2004).

Use of Poultry Litter as a Bio-Fuel Source

Studies have shown that energy production from poultry litter is one potentially beneficial use. Poultry litter is increasingly becoming one of the readily available agricultural by-products from which renewable energy can be created. The process is such that poultry litter is transported to the power plant where it is combusted in a furnace at high temperature. The heat produced heats water in a boiler to produce high pressure steam that drives a turbine and generator to produce electricity. A report by Fibrominn

(2002) indicated that their poultry litter fired power plant in Benson, Minnesota utilized approximately 500,000 tons of poultry litter per year to produce 50 MW of electricity. When poultry litter is combusted to produce electricity, a nutrient-rich ash is produced and can be used as fertilizer. Alternatively, as demonstrated in a study conducted by Kelleher et al. (2002), the poultry litter may be degraded and stabilized under anaerobic conditions by microbial organisms to produce methane that can be used to replace natural gas or fuel oil as a fuel for boilers to produce steam and electricity. This process also produces stable methane sludge that can be used as fertilizer. Therefore, energy conversion in poultry litter fired power plants does not only provide electricity, it also produces ash or methane sludge that is cheaper to transport to other locations for use as a concentrated nutrient-rich fertilizer compared to fresh poultry litter.

Shipping Surplus Poultry Litter to Deficit Areas

Transportation of poultry litter out of problem areas is another alternative for reducing excessive application of nutrients to cropland. A study conducted by Pelletier et al. (2000) to examine the economic feasibility of litter transport, and the potential for a cost-share program to encourage shipment from litter-rich regions in Virginia concluded that if poultry-producing farms must apply litter on a phosphorus basis, more litter will be available for sale. They found that in regions of intense poultry production and limited land application alternatives, litter prices will likely be lower, unless new litter markets become available. A study conducted in Northern Arkansas found that both surface and groundwater can be improved by transporting litter from areas where there is high poultry concentration to areas with lower potential for contamination (Govindasamy and

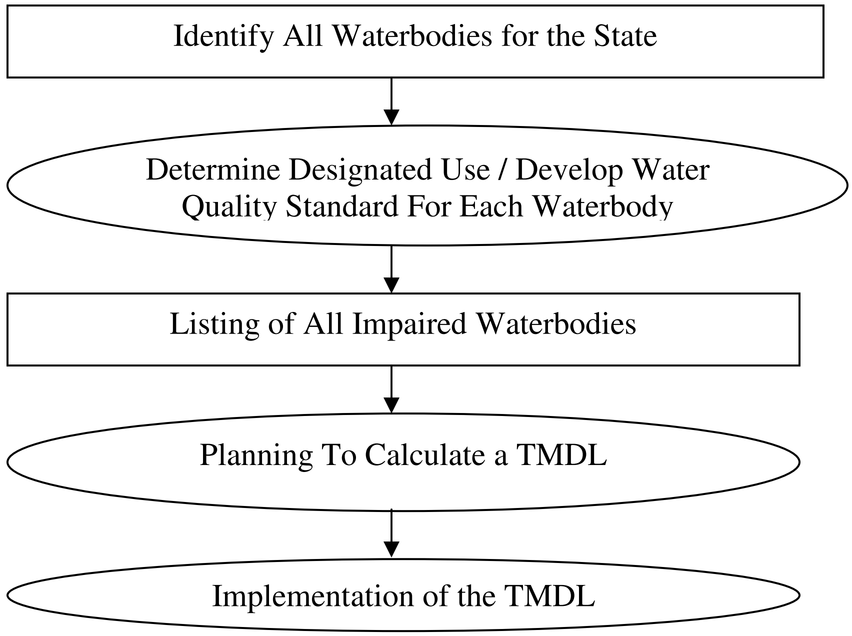
Cochran, 1995). These studies, however, revealed that farmers who prefer using poultry litter as fertilizer needed high volumes of the manure to cover cropland given its low nutrient content. Thus, it might not be economical to transport such large quantities of poultry litter beyond immediate areas for use as crop fertilizer. Cost-share programs have been designed in some States to encourage shipment from litter-rich regions (Simpson, 1998; Pelletier et al., 2000; Ancev, 2003).

A combination of some of these management practices could be implemented by agricultural production enterprises in a given watershed to reduce nutrient loadings to levels that are consistent with the set TMDL for the area. The next section provides an overview of the TMDL development process in the state of Oklahoma.

The Total Maximum Daily Load Development Process in Oklahoma

TMDLs are required under Section 303(d), "List of Impaired Waters," of the Federal Clean Water Act (CWA) for U.S. water bodies that are not attaining ambient water quality standards after application of the technology-based effluent limitations required by the Act (Younos, 2005). By definition, a TMDL specifies the allowable pollutant loading from all contributing sources (that is, point sources, non-point sources, and natural background) at a level necessary to attain the applicable water quality standards with seasonal variations and a margin of safety that takes into account any lack of knowledge concerning the relationship between the sources of the pollutant and water quality (USEPA, 2003; Younos, 2005). In essence, a TMDL defines the assimilative capacity of the waterbody to absorb a pollutant and still meet water quality standards.

The National Academy of Sciences (NAS) defines the TMDL process as the plan to develop and implement the TMDL (NAS, 2001). The objective of a TMDL is to allocate allowable loads among different pollutant sources so that the appropriate control actions can be taken to achieve water quality standards. All states must determine the required reductions in pollutant loadings from point and non-point sources to meet the state water quality standards (Younos, 2005). Not only do approaches to creating a TMDL for a particular pollutant vary throughout the nation, but also the approaches to developing a TMDL vary between pollutants. Because of the wide range of approaches to TMDLs in the United States, no one standard approach can be cited as the best criterion for setting TMDLs in the United States. States, territories, and authorized tribes have limited autonomy, and thus can create TMDL development approaches best suited to the unique nature of their own water quality conditions and water quality standards (Younos, 2005). However, NAS (2001) outlined conceptualized basic steps in the TMDL process as shown in Figure 3 below. It can be noted from Figure 3 that generally the major components of TMDL development are assessment of existing conditions of all waterbodies, determination of maximum allowable loads, allocation of loadings among point and nonpoint sources, and implementation of the TMDL plan.



Source: Adapted from NAS (2001); Younos (2005).

Figure 3 Conceptualized Basic Steps of the TMDL Process.

The Oklahoma Water Resources Board (OWRB) holds the statutory authority to develop Oklahoma's Water Quality Standards (OWQS), a set of rules (Oklahoma Administrative Code, Title 785, Chapter 45) that provide the baseline against which the quality of waters of the state are measured. At the beginning of the TMDL process, the OWRB identifies all waterbodies for the state and develops water quality standards for each waterbody. The Department of Environmental Quality is expected to gather available data and information on the conditions of the water body, pollutant, and pollutant sources throughout typical geographical and temporal conditions with reasonable certainty. The standards comprise of three components: beneficial uses, criteria and anti-degradation policy. Every waterbody has multiple designated beneficial uses (e.g., fish and wildlife propagation, drinking water, or recreation) that are assigned by the OWRB. A waterbody's beneficial uses are determined statistically or through the use attainability analysis, a procedure that requires obtaining physical, chemical, and biological field measurements. These measurements are compared to a set of conditions that describe a waterbody's ability to support different beneficial uses. If a waterbody currently supports or has the potential to support a particular use, that use is designated to the waterbody in the Oklahoma Water Quality Standards. The current list of beneficial uses include : fish and wildlife propagation, public and private water supply (drinking water) or emergency water supply ; primary body contact recreation or secondary body contact recreation; fish consumption, agriculture, and hydroelectric power generation; industrial and municipal process and cooling water, navigation and aesthetics (Oklahoma Department of Environmental Quality , ODEQ, 2004).

Once the beneficial uses for a waterbody are determined, the OWRB determines the specific water quality criteria that apply to the waterbody. The criteria can be numerical or narrative. Numerical criteria are associated with specific numerical values, usually in the form of concentration of a particular water quality characteristic (usually measured in milligrams per liter, mg/L, or micrograms per liter, μg/L). Some numerical criteria are dependent on other factors such as season, temperature, pH, or hardness. Narrative criteria are only defined by description of the desired condition (e.g. to be aesthetically enjoyable, Spavinaw creek must be free from floating materials and suspended substances that produce objectionable color and turbidity). In cases where a single constituent is associated with more than one beneficial use or has more than one criterion, the most stringent of the applicable criteria is what drives a TMDL. The last component of the Oklahoma Water Quality Standards (OWQS) is the anti-degradation policy. This policy describes the conditions under which a waterbody's quality may or may not be decreased. Special designations in the OWQS are used to define how the anti-degradation policy is applied. These designations include Outstanding Resource Waters (ORW), High Quality Waters (HQW), and Sensitive Water Supplies (SWS). The limitations associated with each special designation are described in the OWQS. To fully assess a waterbody, each one of its designated beneficial uses must be assessed. Once each of the beneficial uses has been assessed, an overall category can be assigned. The OWRB places an assessed waterbody into one of five categories described below:

Category 1	-	all beneficial uses assessed and attained
Category 2	-	some beneficial uses assessed, no impaired uses
Category 3	-	not enough information to assess beneficial uses
Category 4	-	one or more uses impaired, but no TMDL required
Category 5	-	one or more uses impared, TMDL required.

Category 5 waterbodies make up the State's 303(d) List of Impaired Waters. Listing of an impaired waterbody is done by a comparative analysis of existing water quality data to the relevant water quality standard. If known, the cause, source and extent of the impairment(s) are identified in this process. The OWRB establishes a priority ranking for the waterbodies on the list and dates by which TMDLs should be developed to address causes of impairment based on the availability of data needed, severity of impairment, presence of endangered or threatened species, public health issues, public interest, and efficiency in public participation. Listing of waters for TMDL development is an integrated process involving monitoring, water quality standards, and Oklahoma Pollution Discharge Elimination System (OPDES) permits (NAS, 2001; ODEQ, 2004). Next in the sequence is the planning step in which the TMDL is calculated. The OWRB determines a numerical quantity representing an estimate of the maximum amount of pollutant loading a water body can receive over time from both point and non-point sources without violating the Oklahoma Water Quality Standards with an adequate margin of safety. Then this maximum permissible pollutant load is allocated among various point sources, nonpoint sources, natural background sources and a margin of safety (MOS) in the watershed according to the following equation:

$$TMDL = WLA + LA + MOS \quad (1)$$

where TMDL is loading capacity of the receiving water body (an estimate of the maximum amount of pollutant loading a water body can receive over time without violating water quality standards); WLA is wasteload allocation (the portion of a receiving water body's loading capacity that is allocated to existing and future point sources; LA is load allocation (the portion of a receiving water body's loading capacity that is allocated to existing and future nonpoint sources and to natural background source; and MOS is margin of safety (the prescribed mechanism to account for the uncertainty in determining the amount of pollutant load and its effect on water quality (ODEQ, 2004). Hydrological, biological, chemical, and pollutant fate and transport data are required to calculate a water body's loading capacity. Before pollutant loads are allocated among sources, the location and types of sources, and the current and projected pollutant load for each source are identified. Current loading and source contributions are established by measuring pollutant loads directly, calculating or estimating loads from water quality and flow data, estimating loads with mathematical models, or using a combination of these methods. The data needed for pollutant source analysis may include watershed and subbasin boundaries, hydrologic interaction between surface water and groundwater, locations of stream segments, locations of pollutant sources, types of pollutant sources, anticipated growth of discharges, meteorological/rainfall data and runoff coefficients, land uses and land cover, and soil types. Information on factors that influence water quality such as permitted industrial and municipal wastewater discharges, concentrated animal feeding operations (CAFOs), waste application sites, cropland, forestry operations, industrial storm water runoff, urban runoff, construction activities, and other sources such as natural background may be collected and used to determine cause-and-effect relationships in a given watershed (NAS, 2001; ODEQ, 2004).

There are three common methods for allocating pollutant loads; equal percent removal, equal effluent concentrations, and a hybrid method. Equal percent removal exists in two forms. In one, the overall removal efficiencies of the sources are set so that they are all equal. In the other, the incremental removal efficiencies beyond the current discharge are equal. The equal effluent concentration method is similar to equal percent removal and requires that influent concentrations at all sources must be the same. With the hybrid method, the criteria for waste reduction may not be the same from one source to another. One source may be allowed to operate unchanged while another may be required to provide the entire load reduction. More generally, however, a proportionality rule may be assigned that requires the percent removal to be proportional to the input source loading or flow rate (ODEQ, 2004).

The public participation process starts shortly after the TMDL project starts. A public notice is placed in the Oklahoma Register and local newspaper(s) announcing the start of TMDL development. Staff from the participating agencies attends stakeholder and focus group meetings upon request. The ODEQ has the statutory authority to lead the development of TMDLs. It conducts the initial review and approval process for TMDLs before sending them to the EPA. After completion of the public participation process, the ODEQ submits the completed TMDL report to EPA for approval. EPA has 30 days for this approval process. The EPA has to review and approve the TMDLs conducted for waterbodies that appear on the State's 303(d) list before any TMDL can be implemented. Following EPA approval, the results of the TMDL must be incorporated in the State's Water Quality Management Plan, also called the 208 Plan after the Clean Water Act section that requires it. The Implementation Plan will go through a similar public participation process as the TMDL (ODEQ, 2004).

The last step in the process is the implementation of the TMDL. This is conducted through a variety of mechanisms and programs. In Oklahoma, the recommendation for point sources and non-point sources are applied in different ways. Point sources are regulated through the Oklahoma Pollution Discharge Elimination System (OPDES) program. The 208 Plan contains information about all of the regulated point sources in the state. Wasteload allocations for each point source are made part of the 208 Plan and permittees must have a wasteload allocation before a discharge permit is issued. OPDES permits must be in compliance with the TMDLs. Non-point source implementation is managed by the Oklahoma Conservation Commission (OCC) through Watershed Base

Plans. Non-point source pollution controls are implemented on a voluntary basis. The primary mechanisms used by the OCC are incentive programs for the installation of Best Management Practices and public education and outreach programs. TMDLs set the stage for the implementation of voluntary and existing regulatory reduction measures to reduce the pollutant loads for the attainment of water quality standards (ODEQ, 2004; Younos, 2005).

Potential Contribution of Economic Models in TMDL Program

The TMDL process as outlined in the previous section includes specific steps that must be followed in identifying impaired water bodies, setting and implementing the TMDLs. All states must set water quality standards for water bodies based on designated uses and numeric and / or narrative criteria. However, the institutional context within which the current design and implementation of TMDL plans take place does not recognize that watershed stakeholders may have multiple and diverse objectives related to improving water quality and thus constrains the way they can be achieved effectively and efficiently. Neither does the current TMDL process recognize that setting a pollution standard that maximizes net social benefits requires an economic valuation of both abatement and damage costs that occur within and beyond the watershed. The current TMDL process requires that a TMDL be set first, and then waste load allocations be determined next. This approach is not consistent with the maximization of net social benefits in the watershed, a condition that requires minimization of the sum of total pollution treatment costs and total environmental damage costs. Economic theory suggests that setting of efficient pollution standards and the determination of waste load allocations to polluters (that incorporates transfer coefficients that reflect proportions of emissions from each polluter) must be done simultaneously taking into account both treatment and environmental damage costs for a given pollutant in the watershed (Tietenberg, 2003). The use of economic optimization models integrated with environmental models allow environmental benefits and costs of TMDL standards to be assessed, both treatment and damage costs to be considered, while simultaneously determining the desired standard and allocation of pollution reductions among sources to minimize costs of reducing pollution to society. The integrated environmental-economic approach to TMDL setting and implementation has the ability to take into account multiple and diverse watershed stakeholder objectives and responses. Thus, the current TMDL design and implementation process may be improved by using economic analysis of costs and benefits to set and implement TMDL goals and standards efficiently. A number of influences are cited in the literature regarding phosphorus pollution in watersheds. The next section highlights such influences in the case of the Eucha-Spavinaw watershed.

Legislative, Regulatory and Legal Influences

There are several factors that are cited to have contributed to the current problem of phosphorus pollution in the Eucha-Spavinaw watershed. Commentators point to legislative, regulatory and legal developments in Oklahoma and Arkansas, among other factors, as major influences on nutrient pollution in the watershed. Hipp (2002) attributes the rapid growth of the poultry industry in the Eucha-Spavinaw watershed to the relaxation of laws prohibiting corporate farming in the two states. This led to increased corporate swine and poultry farms in the area. Corporate swine farms are primarily concentrated in Western Oklahoma though some are located in the study area. Poultry farms are concentrated in both Eastern Oklahoma and Western Arkansas. The intensity of use within the specified areas increases the number of birds and swine and thus litter. The litter from these concentrated animal feeding enterprises caused public concerns related to odor and water quality. Because of these concerns, the Oklahoma-Arkansas River Compact Commission adopted a goal to reduce phosphorus pollution in the nearby Illinois River by 40 percent.

In 2001, the Oklahoma Poultry Waste Transfer Act was enacted to provide tax relief to those who transport poultry waste from the regions where it is abundant and creates environmental problems to regions where phosphorus is in deficit. The OWRB (1996) designated public and private water supply, cool water aquatic community agricultural irrigation, primary body contact recreation, and aesthetics as beneficial uses for the Lakes Eucha and Spavinaw. However, OWRB (2002) conducted a water quality study whose results indicate that several of the designated beneficial uses of the lakes were impaired, most importantly the water supply and recreational uses. The OCC (1997) and OWRB (2002) identified the main cause of impairment as external phosphorus loading from non-point agricultural sources and a municipal point source in Arkansas. Crop production, swine, poultry and cattle production agricultural enterprises have been cited as the main contributors to excessive phosphorus pollution in the Eucha-Spavinaw watershed. Municipal discharges have been identified as the major point source of phosphorus pollution in the area. A variety of these upstream uses accumulate in a number of downstream ecological effects, such as eutrophication and species extinction

(OWRB, 2002). Due to the unidirectional flow of water and matter from the catchment to the lakes, the ecological and economic effects of measures may be distributed unevenly across space in the watershed. Since the watershed is shared by two states, there are structural, legal and administrative issues that limit possibilities for regulating phosphorus pollution in the Eucha-Spavinaw watershed.

Taylor et al. (2004) pointed out that most policies to reduce agricultural nutrient run-off have relied upon voluntary technology-based approaches rather than market-based approaches such as tradable permits or taxes, such as the United States Department of Agriculture (USDA) Environmental Quality Incentive Program. The United States Environmental Protection Agency (USEPA) currently lists thirty-seven water quality trading markets, in existence or development, that allow privately owned point sources to meet their regulatory burden through the purchase of non-point source abatement offsets from landowners (USEPA 2002). Woodward (2003) observed that market-based programs for pollution control were rapidly on the rise in the United States. Not only were SO_2 permits bought and sold on the Chicago Board of Trade, but volatile organic compounds, nitrogen oxides, and other air pollutants were traded in local markets throughout the country. Markets involving water pollution trading rights are also growing in number and scope. In 1996, the U.S. Environmental Agency (EPA) released a draft framework for water pollution trading that implicitly sanctioned the development of nutrient trading programs. An Environomics report to the EPA, titled “A Summary of U.S. Effluent Trading and Offset Projects”, listed 16 market-based programs for controlling water pollution in various stages of implementation and nine more programs that are under development (Environomics, 1999). Despite this burgeoning interest, the experience with effluent trading was quite limited. According to the Environomics report, less than 10 trades had actually taken place in the nation's history of effluent trading. Only one or two trades had taken place in each of the trading programs in existence for more than a decade. The experience suggested that there were characteristics of water pollution problems that posed serious barriers to trading such as verifiability, transaction costs of the trading process, or both. Several studies identified transaction costs in pollution trading markets as a significant factor in determining success or failure of pollution abatement efforts.

Transactions Costs in Pollution Trading Markets

O'Neil et al. (1983) considered transaction costs to include those required for monitoring of emissions, enforcement of environmental standard, and information costs associated within a tradable market system. They claimed that often a permit system where no trades occur is likely to be less costly than a technology standard. A later study conducted by Stavins (1995) confirmed transaction costs are not negligible for permit markets, but concluded that even if transaction costs prevent a permit system from realizing a high number of trades, the aggregate costs of compliance will be less costly than a command-and-control policy. However, he pointed out that it is difficult to know, a priori, whether the transactions costs of market based solutions will be greater or less than non-market solutions. Therefore, options need to be evaluated on a case-by-case basis to incorporate transactions costs in policy evaluation. The literature on transaction costs and environmental policy cites several factors that influence the level of transaction costs such as the number and diversity of agents, available technology, policy under consideration, and the amount of abatement or the size of the transaction (Coase, 1960; Stavins, 1995; Thompson, 1996 and McCann and Easter, 1999).

Thompson (1996) proposed an institutional transaction cost framework that could be used to measure and incorporate transaction costs in policy evaluation. McCann and Easter (1999) applied the framework with modifications in their study to determine transaction costs of implementing various agricultural pollution reduction policies. They defined transactions cost to include research, information gathering and analysis; enactment of enabling legislation including lobbying costs; design and implementation of policy; support and administration of the on-going program; monitoring / detection; and prosecution / inducement costs. McCann and Easter (1999) measured the magnitude of transaction costs associated with policies to reduce agricultural non-point source pollution in the Minnesota River. Their findings indicate that taxes may have advantages with respect to transaction costs and abatement costs compared to educational programs on BMPs, the requirement for conservation tillage on cropped land, and expansion of a permanent conservation easement program. Gangadharan (2000) studied the Regional Clean Air Act Incentives Market (RECLAIM), a program which uses emissions trading to reduce smog creating pollutants in Los Angeles. The study measured the impact of transaction costs (defined to include search costs and information costs) on trading probabilities of firms in RECLAIM. Results showed that transaction costs are substantial in the initial years of the program; search costs and information costs are high, as the firms do not participate in similar input and output markets; the presence of transaction costs reduces the probability of trading by about 32 percent. The study concluded that transaction costs are significant in explaining non-participation of some firms in the market (Gangadharan, 2000). A similar view was held by Stavins (1995) and Nagurney and Dhanda (2000) who emphasized that transaction costs can obstruct performance efficiency of the permit market by impeding the trading process of permits. This occurs when information requirements significantly raise transaction costs in the trading process because parties to an exchange must find one another, communicate, and exchange information (Stavins, 1995). The preceding discussion suggests that the success of tradable emissions permit system required that attention be paid to technology of monitoring pollutants, ensuring effective monitoring of the quantity and quality of effluents, define transactions, establish emission ceilings and have effective enforcement systems. Emphasis is also laid on the necessity to have enough buyers and sellers, clear rules, especially for transactions and emission limits. Effluent limits must depend partially on installed technology and require minimum pollution control, which will allow for effluents with different treatment cost; there must be a future trading market with enough buying and selling potential; it is important for the smooth and efficient functioning of the market that the permit market be very broad and deep and transactions costs must be low; and enforcement must be credible and sustainable over time.

Agriculture and Water Quality Standards

Watershed management involves the making of informed choices about the desired level of economic activities and ecosystem functioning in the catchment. Information on the economic and ecological effects of measures as well as their spatial distribution is therefore needed. Agricultural production has increasingly been identified as a leading cause of water quality impairment in the United States (USDA and USEPA, 1998; Poe et al. 2004). Agricultural sources of pollution are generally classified as non-point sources. Tietenberg (2000) observed that point sources discharge into surface waters at a specific location through a pipe, outflow or ditch while non-point sources pollute the waters in a more diffuse and indirect way. Loehr (1984) further noted that non-point source pollution is intermittent and affected by random meteorological events while pollution from point sources is more or less constant and dependent on the level of production activities.

Poe et al. (2004) stressed that non-point source pollution, particularly from agriculture, is the largest reason that the United States is not meeting “fishable / swimmable” objectives of the 1972 Clean Water Act. This means that there is a growing need for environmental policy instruments that are effective and efficient in generating significant reductions in nutrient runoff from farming. However, as Poe et al. (2004) noted, the development of such policies is complicated by the disperse nature of agricultural runoff. The conveyance of nutrients from farming practices to waterways, across and through the physical landscape, makes identifying the contribution of individual sources difficult and costly. Farm-specific emissions are difficult to measure, the relationship between emissions and ambient water pollution levels is stochastic and difficult to model, and there are adverse selection and moral hazard problems.

The major economic policy question is how to create incentives for both point and non-point sources to cost-effectively meet water quality standards. Economic theory suggests that cost-effective environmental policy should impose the same Pigouvian tax rate or quota price on all polluters (Poe et al., 2004). The cost-effective solution would result from equating marginal abatement costs across pollution sources on condition that marginal damage costs are independent of the pollution source.

Segerson (1988) established that socially efficient outcomes can be achieved through ambient-based approaches. These approaches regulate non-point source polluters based on ambient concentrations. Incentive schemes are designed such that a group of polluters (also called collective or team contractors) pays penalties if ambient pollution at a common monitoring point exceeds a water quality target or receives subsidies if pollution is below the target. Similar recent auction experiments conducted by Taylor et al. (2003) tested the effectiveness of team contracts as a strategy to induce farm-level nutrient abatement in a watershed. The contract for nutrient reduction tied the payments of individual farmers to the collective performance of the entire group in a small sub-watershed. Farmers receive a monetary reward when their collective abatements (measured at a common monitoring point) are equal or greater than the collective bid quantity, otherwise they are penalized for under-abating (payments are withheld). The liability of each polluter depends upon the abatement effort of all polluters as well as stochastic environmental factors such as weather. The study establishes that the bidding mechanism that allows individual farmers to decide both the technology they will use for pollution abatement as well as the quantity, could be efficient in meeting pollution targets. An important theoretical assumption underlying these ambient-based approaches is that firms undertake non-cooperative Nash behavior.

However, the results from Taylor et al.'s (2003) work should be treated with caution. The study used a relatively small data set comprising of two groups of agricultural producers, and three groups of students at Ohio State University in testing the effectiveness of the team contract bidding mechanism. The students and agricultural producers are two different groups with different characteristics. More experiments with farmers would provide more relevant information to determine the extent to which they know how their individual farming practices (and their neighbors') contribute to water quality. It is important to note that this study did not adequately consider stochastic environmental factors such as weather. Also, the participants in this study had difficulties understanding the contract and were not informed of the Nash equilibrium strategies. Neither did the study take into consideration the socioeconomic and structural differences of the participants. The need to consider such differences is demonstrated in an earlier study conducted by Renwick and Archibald (1998) using household-level panel data for two communities in Southern California. The authors assess the performance of alternative demand side management (DSM) price and non-price policy instruments in terms of their effectiveness in reducing aggregate demand and distribution of water savings among households. The findings suggest that even though DSM policies are effective in reducing aggregate demand, the magnitude of the reduction in demand associated with different policy instruments varied significantly with the characteristics of the households. The composition of aggregate demand (or program participants) matters in the assessment of the potential for policies as a water resource management tool (Renwick and Archibald, 1998). However, this study did not address the lags in responsiveness of aggregate demand to policy changes; neither does it account for interaction effects on demand of different policy instruments used during the study period. In reality, it is most unlikely that there will be no interaction among different policy instruments used and that no lags in responsiveness to policy changes will occur in a setting used in this study. Therefore, more research is needed to measure policy interaction effects on demand and determine the circumstances under which potential synergistic policy interaction would yield maximum reduction in aggregate demand or maximum water savings by households than if each policy were implemented in isolation. This would apply as well to environmental policy instruments geared at providing incentives to farmers to reduce nutrient discharges into ambient water bodies.

Romstad (2003) warned that theoretical team mechanisms are difficult to implement because they impose a high information requirement on the regulatory agency or point source. The contractor needs to possess privately held, farm-level abatement cost information in order to set the appropriate incentives. Though these experimental economic studies have established that ambient-based pollution mechanisms are effective in meeting pollution targets, their effectiveness is limited to small group settings of homogeneous, non-cooperative agents. However, there is a policy concern because this "theoretical non-cooperative agents" assumption may not be appropriate under field settings where a small group of firms with similar interests are likely to collude.

Poe et al. (2004) investigated the performance of ambient-based approaches under conditions of group cooperation. They used different programs that included mechanisms such as tax and subsidy, fixed penalty, tax only, subsidy only and combined approach. The study establishes that when firms are allowed to cooperate, as is likely in a real world setting where a small group of polluting firms face the threat of potentially large tax liabilities or the possibility of large subsidies, the experimental outcomes deviate quite substantially from non-cooperative settings and corresponding theoretical predictions. Kebede et al. (2002) used an integrated economic-environmental model to assess the point source pollution from major industries in Northern Alabama. They apply an input-output economic model with pollution emission coefficients to assess direct and indirect pollutant emission for several major industries. This approach provides a useful analytical tool for direct and cumulative emission estimation and generates insights on the complexity in choice of industries or enterprises. A variation of this approach was developed and applied by Schou, Skop and Jensen (2000). Their method combines a partial equilibrium sector model for agriculture, farm accounts statistics, a GIS-based procedure for spatial disaggregation of agricultural production structure, a procedure for calculating farm economic input, and a nitrate loading model. The method is applied by analyzing two alternative tax policies in terms of their effects on farm value, nitrate leaching, and nitrate loading of costal waters. Scheren, Zanting and Lemmens (2000) applied a system of pollution inventory methods to estimate waste loads from pollution sources on the basis of functional variables and pollution intensities and use penetration factors to incorporate the effects of treatment facilities and natural 'purification' in rivers and wetlands.

Point – Non Point Source Pollution Trading

"Among the most important EPA initiatives to address agricultural and other nonpoint source contributions to water quality problems is the Total Maximum Daily Load

(TMDL) program. The program requires states to develop and implement watershed-based plans for water resources that are too polluted to meet designated uses. In many watersheds, achieving designated uses will require that states tackle long unregulated nonpoint sources. As the leading nonpoint source, agriculture will likely be a major target of TMDL initiatives (USEPA, 2002). There is substantial interest in using point-nonpoint trading to achieve nonpoint source reductions. Several fully implemented and pilot point-nonpoint trading programs have emerged over the past decade, the best-known being Tar-Pamlico (NC), Cherry Creek (CO), Dillon Reservoir (CO), and Fox River Basin (WI)." (Horan and Shortle, 2005, AJAE 87(2): 340).

Lake Dillon, Colorado is a high water quality reservoir used for recreation purposes. It is the first point-nonpoint trading program in the U.S. (Jarvie and Solomon, 1998). The state of Colorado used a total phosphorus standard of 7.4 μg / L to allocate a total annual phosphorus load of 4,610 kg per year for the watershed. The state implemented controls for existing urban nonpoint sources rather than upgrading their publicly owned treatment works (POTWs) discharging to the lake and approved a water quality management plan with a trading ratio of 2:1. The Lake Dillon point-nonpoint trading program did not allow for the purchase or sale of phosphorus credits. However, it demonstrated that trading can simultaneously allow development in the watershed and still improve or maintain lake quality (EPA 1992; Jarvie and Solomon, 1998).

The Tar-Pamlico river basin experienced high levels of nitrogen and phosphorus emissions in the 1980s. Sixty-six percent of the phosphorus came from nonpoint sources while 25 percent came from waste water treatment plants. Eighty-three percent of the nitrogen came from nonpoint sources. The Tar-Pamlico river basin was classified as Nutrient Sensitive Waters due to rapid eutrophication in the basin. Municipal plants in the basin formed the Tar-Pamlico basin association and worked with the State of North Carolina to develop a nutrient budget and reduction strategy for the basin. The dischargers in the Tar-Pamlico basin had an option to either trade 10-year life reduction credits at a 3: 1 trade ratio with other point sources or pay to implement best management practices at nonpoint sources to reduce the basin's nutrient load (Hall and Howett, 1994; NCDEM, 1995; and Jarvie and Solomon, 1998).

From the preceding discussion, the lesson learned is that "point-nonpoint trading works as follows: Pollution sources are required to hold permits that define their allowable discharges. For metered point sources, the permits define allowable measured discharges. Because nonpoint discharges are generally unobservable, the permits define allowable "estimated" discharges, where the estimates are derived from models linking observable land use and management practices to nonpoint loads. With tradable permits, each source can adjust its allowances by buying or selling permits subject to rules governing trades. Among these rules is a trading ratio that defines how many nonpoint source permits trade for one point source permit." (Horan and Shortle, 2005, AJAE 87 (2): 340).

Jarvie and Solomon (1998) pointed out some of the conditions whose presence might increase the likelihood of a successful implementation of an effluent trading program. There must be a need for improved water quality in the watershed coupled with an existence of a strong economic incentive for effluent sources in the basin to engage in trading. The point-nonpoint effluent trade must not bring additional risk for point sources in the watershed to reduce effluent at nonpoint sources not under their control. Trading may be encouraged if there were community benefits resulting from the effluent trade. Previous studies have shown that trading programs that had public support were most likely to be implemented with much success. Cost-effective improvements in water quality may be achieved if all these conditions were present in a given basin (Jarvie and Solomon, 1998).

The Need for Environmental Policy

Policymakers must understand why environmental policy is needed. A basic microeconomic principle is that the equating of marginal benefits and marginal costs will maximize total net benefits. Thus, free markets will lead to a socially efficient allocation of resources. However, it is important that policymakers understand what kind of costs and benefits are generated by the good or service in question. When private costs are identical to social costs, and private benefits are identical to social benefits, then the free market will produce optimum welfare and resources will automatically be allocated efficiently. Evidence abound in economic literature suggests that markets are not necessarily perfectly competitive in the real world. There exist circumstances that create a disparity between private costs and social costs, and private benefits and social benefits. Several reasons are identified, based on the preceding argument, why environmental policy is necessary. The reasons include market and policy failures that are interlinked with the evolution of property rights. Market failure is a technical term that is used to refer to conditions under which the free market does not produce optimal welfare and fails to allocate resources efficiently. The market clearing forces do not maximize social net benefits by equating marginal social benefits with marginal social costs. Important examples of such failure include external effects (externalities), public goods, common pool resources, poorly defined or defended property rights, imperfectly competitive markets, and imperfect (or asymmetric) information (Khan, 1998; Freeman, 2003; Sterner, 2003).

Some environmental problems have arisen from a failure of political rather than economic institutions to allocate environmental resources efficiently. Policies reflect economic interests, and in some cases, there may not be a single policy that is "optimal" for every group in society. However, improper incentives are the root cause of policy failure. Inappropriate government intervention in the economy may be a source of disparity between private and social values. This divergence between private and social costs, or private and social benefits, may lead to non-optimal social welfare. Sterner (2003) distinguished two types of policy failure: corrupt policy and bad policy. The former is a policy that claims to be in the interest of the whole country but actually serves the interest of one group. Policy resulting from special interest groups is most likely to fall in this category. Special interest groups use the political process to engage in rent seeking, defined as the use of resources in lobbying and other activities directed at securing protective legislation (Tietenberg, 2000). A bad policy is one that intends to enhance welfare in a reasonable way but fails due to ineptitude (Sterner, 2003). Public policies and the actions of individuals and firms can lead to changes in the flow of services from natural resources, thereby creating benefits and costs. Public policy will lead to misallocation of environmental goods and services if property rights are affected in such a manner that results in a divergence between private and social costs and benefits in the economy (Freeman, 2003).

Jones (2002) pointed out that the economics of goods and services depends on their attributes. Goods and services that are excludable allow their producers to capture the benefits they produce; goods and services that are not excludable involve substantial ‘spillovers” of benefits that are not captured by producers. Such spillover costs or benefits, unintended consequences, or unintended side effects (either beneficial or detrimental) associated with market transactions are called externalities. Examples include soil erosion caused by unsuitable agricultural practices. The silting of dams and the destruction of coral reefs are real costs, but these costs are not borne by the individuals or corporations that cause the damage. These externalities lead to a divergence between private and social costs and benefits. This results in a misallocation of resources in the economy. Goods and services with positive spillovers tend to be underproduced by markets because not all preferences are revealed. The marginal private benefits are less than marginal social benefits in this case. This provides an opportunity for government intervention to improve welfare. Goods with negative spillovers may be overproduced by markets. The marginal private costs are less than marginal social costs in this case. Examples include water pollution and a factory the smoke from which has harmful effects on those occupying neighboring properties. Such situations can be seen as consequences of incomplete property rights: if waterways and air had owners with a right to clean water and clean air respectively, then those owners could sue those who caused the soil erosion and air pollution and thus internalize the effects. Government regulation may be needed if property rights cannot be well defined. The tragedy of the commons is a good example (Stavins, 2000; Sterner, 2003).

Several common environmental resources such as clean water, clean air, tropical forests, world's fisheries, natural scenic landscape, and biological biodiversity are examples of public goods. Tietenberg (2000) defined public goods as those that exhibit both consumption indivisibilities and nonexcludability. These goods or services are enjoyed in common. Once the good or service is provided, even those who fail to pay for it cannot be excluded from enjoying the benefits it confers. One person's consumption of a good does not diminish the amount available for others. The market tends to undersupply public goods because it is hard to exclude those who do not pay and each person is able to become a free rider on the other's contribution. When this happens it tends to diminish incentives to contribute, and the contributions are not sufficiently large to finance the efficient amount of the public good. Sterner (2003) emphasized that political processes are needed, such as the election of a government that collects taxes and finances public goods. He further noted that common pool resources also have costly exclusion, but the goods produced with these resources are consumed individually (as private goods). Examples include firewood and fodder, and the resources are often managed as common property. Free riding and other mechanisms that lead to the undersupply of public goods may also lead to the overuse of common pool resources unless institutions are strong enough to limit access by the users.

Imperfect competition is the term used for markets where individual actions of particular buyers or sellers have an effect on market price. In such markets, the marginal revenue of the firm becomes different from the market price, and this tends to generate an equilibrium where marginal social cost is not equal to marginal social benefit. Noncompetitive markets, monopolies, and oligopolies usually result in nonoptimal supply of good and services. These market structures usually result in lower-than-normal production and higher-than-normal prices. That is, situations where too little of the good or service is sold at too high a price compared with outputs and prices usually achieved under perfectly competitive market conditions (Tietenberg 2000; Sterner, 2003). Thus, imperfect markets contribute to environmental problems by inefficiently allocating environmental resources to competing uses. Many extractive industries may be characterized by imperfect competition. Some industries, such as electric power and natural gas distribution, are regulated monopolies. Other industries, such as oil and coal, are regarded by the general public as oligopolistic (only a few sellers who have price-setting ability) (Khan, 1998).

Asymmetric or imperfect information is another cause of market failure. This is a situation where some segment of the market - consumers or producers or both - does not know the true costs or benefits associated with the good or service. Economists typically point out that there are no "free lunches" yet commonly assume that information is freely available to everyone. This does not necessarily hold in the real world. Empirical evidence suggests information is costly, and lack of information stops the market from operating perfectly. Under these circumstances, the forces of supply and demand are most likely not to equate marginal social benefits with marginal social costs. Imperfect information may be an important factor when dealing with global warming, acid rain, the effect of exposure to hazardous chemicals in the home (pesticides, asbestos, detergents, radon, etc), air and water pollution. Because policymakers do not have reliable data on pollution damages and abatement costs, for instance, they cannot design policies that are both efficient (with respect to resource allocation) and fair (in sharing the burdens of all the costs involved). Understanding information asymmetries would help policymakers design policy instruments to address monitoring difficulties, and promote social goals such as equity without destroying incentives for work and efficiency (Khan, 1998; Sterner, 2003).

Policy Instruments Used for Environmental and Natural Resources Policy

Given market and policy failures outlined in the previous section, this section presents the main categories of policy instruments used for environmental and natural resources policy. However, Sterner (2003) warned that no policy instrument will work perfectly (although some will work better than others) if the economy is not competitive and if the bureaucracies are not honest, well-informed, and sufficiently well funded to carry out their responsibilities. Various kinds of policy instruments are applicable in the area of natural resource management (water, fisheries, land, forests, agriculture, biodiversity, and minerals) and in pollution control (air, water, and solid and hazardous wastes). The policy instruments may be divided into four categories: using markets, creating markets, environmental regulation, and engaging the public (World Bank, 1997; Sterner 2003) as shown in table 4 below. As can be seen from table 4, the first category of policy instruments, "using markets," includes subsidy reduction; environmental charges on emissions, inputs, or products; user charges (taxes or fees), performance bonds, deposit-refund systems, and targeted subsidies.

Table 4 Classification of Instruments in the Policy Matrix.

Using Markets	**Creating Markets**	**Environmental Regulation**	**Engaging the Public**
Subsidy reduction	Property rights and decentralization	Standards	Public participation
Environmental taxes and charges	Tradable permits and rights	Bans	Information disclosure
User charges	International offset systems	Permits and quotas	
Deposit-refund systems		Zoning	
Targeted subsidies		Liability	

Source: Adapted from World Bank 1997; Sterner 2003.

The first category of instruments also includes instruments such as refunded emissions payments and subsidized credits. There are many forms of subsidies, from tax expenditures, to more classical direct, budget-financed payments in support of certain activities (or people). A per-unit pollution subsidy is an incentive that pays the polluter a fixed amount of money for each unit of pollution that is reduced. The polluter will reduce pollution to the point where the per-unit subsidy is equal to the marginal cost of abatement. Under the tax system, pollution is reduced until the point where the per-unit tax is equal to marginal abatement cost. Therefore, if the amount of the per-unit subsidy is equal to the amount of the per-unit tax, the two systems will generate identical behavior on the part of an individual polluter. Subsidies may apply to payment for certain “services,” prices for certain inputs or technology, loans, or access to credit markets. The most practical argument against subsidies is that they are too expensive as a policy instrument, especially in developing countries where the opportunity cost of public funds is high. Baumol and Oates (1988) argue that while taxes and subsidies have equivalent effects on an individual polluter, they have quite different effects on the number of polluters. Subsidies make firms more profitable, so there will be more firms under a subsidy system than under a tax system. A subsidy system whose aim was to reduce the amount of pollution per polluter in the watershed could actually result in an increase in the amount of pollution if a large number of firms entered the market due to the increased profitability generated by the subsidy. The deposit-refund system instrument encompasses a charge on some particular item and a subsidy for its return. The polluters (i.e. those who do not return the item) pay a charge, whereas those who return the item collect a refund and thus pay nothing. A deposit-refund system is similar to a tax, but instead of making the individual pay for undesirable acts as they occur, the individual pays up front and then is rewarded if he or she acts properly. The refund is paid when the potential polluter demonstrates compliance by returning the item that carries the refund, thus making the monitoring of illegal disposal unnecessary. Deposit-refund systems have been used mainly for waste management and recycling of beverage containers, tires, batteries, and containers of toxic household products (Khan,1998; Tietenberg, 2000; Sterner, 2003). A refunded emissions payment encompasses a charge, the revenues of which are returned to the aggregate of taxed firms. Thus, polluters pay a charge on pollution, and the revenues are returned to the same group of polluters, not in proportion to payments made but in proportion to another measure, such as output. The net effect of the payment and refund is that the firms with above-average emissions make net payments to the cleaner-than-average firms.

A pure environmental charge is referred to as a Pigouvian tax if it is set equal to marginal social damage (e.g. of some pollution). At least under several classical assumptions (including fully informed, honest, welfare-maximizing regulators and appropriate concepts of property rights), they have certain optimality properties. However, in many cases, the pure environmental tax is hard to use (e.g. if pollution is unobservable). The reasons for this difficulty include the lack of understanding of the multiservice and public good characteristics of ecosystems. The available proxies or substitutes (such as input or output taxes) are more or less suitable. Setting the level of tax or charge is far from trivial. It is important that an environmental protection agency set the tax or charge equal to marginal damages at the optimal pollution level, which may be different from marginal damages at the time of the decision itself. However, taxes have a couple of disadvantages, one of which is the relatively complex legal process involved in passing and modifying tax laws, which can make the tax instrument somewhat blunt. Furthermore, many politicians have encountered considerable resistance to environmental taxes, and local or sectoral charges typically are more readily acceptable.

The second category of policy instruments, "creating markets," consists of mechanisms for delineating rights. The most fundamental of these mechanisms has particular relevance in developing and transitional economies: the creation of private property rights for land and other natural resources. A mechanism that is relevant at the local level is common property resource management. Special kinds of property rights in environmental or natural resource management are emissions permits and catch permits. In an international context, such mechanisms are often referred to as "international offset systems." One way to control aggregate levels of emissions or harvest is to set a total number of permits or quotas to adapt to the assimilative capacity of the environment or the sustainable harvest yield, respectively. Setting totals while allowing for some dynamics in the economy due to population growth, changing technology, mobility, and economic growth means that the allocated permits must be transferable. Otherwise, the allocation of all available rights would make all new activities impossible by definition. Tradability also allows the efficiency of the market mechanism to be harnessed to ensure that marginal benefits and costs are equalized. The resulting instrument is called tradable emissions permits in pollution management and individual transferable quotas in fisheries management. Similarly, transferable grazing rights, development rights, and other mechanisms apply to other areas of natural resources management. The creation of tradable permits helps remove the externalities implied by the absence of property rights or the “public good” character of the environment. Essentially, this mechanism creates property rights to new resources or shares in the assimilative capacity or the sustainable rent production of ecosystems. The fact that these property rights internalize externalities and create incentives for protection means that resources have a good chance of being put to their most efficient use. However, severe conceptual and practical problems must be overcome. For the permits or quotas to work, they must acquire the characteristics of property rights, such as permanence and reliability. It takes time and commitment to develop permanence and trust, and in the case of natural resources management, a lack of knowledge about the underlying ecosystems and lack of agreement about how they should be managed create additional difficulties (Coase,1960; Dale, 1968; Montgomery, 1972; Sterner, 2003).

The category "environmental regulations" includes standards, bans, (non-tradable) permits or quotas, and regulations that concern the temporal or spatial extent of an activity (zoning). Licenses and liability rules also belong in this category, connecting it to a large area of lawmaking and to the politics of enforcement. Such instruments as liability bonds, performance bonds, (more generally) enforcement policies and penalties are all part of the instrument arsenal. One way of regulating the behavior of firms, households, agencies, and other agents in the economy is by prescribing the technology to be used or the conditions (through zoning or timing). Regulations that restrict location or timing are bans and zoning. A ban is a form of technology regulation in which a specific process or product is not allowed whereas zoning refers to a kind of regulation whereby certain methods or technology are banned in or limited to a certain area. Examples are bans on certain kinds of chemicals, fuels or energy technologies, and vehicle types. Examples in natural resources management include mandatory technology (or restrictions on technology) for management, catch, hunting, farming, and so forth. With full information about abatement and damage costs, the regulator could specify the necessary individual technologies to achieve maximum welfare.

The reasons why technology standards, restrictions, and zoning are the most commonly used instruments are their intuitive simplicity and perhaps the short time perspective of many policy decisions. These kinds of regulation may also suit the interests of both regulators and polluters. The technology must achieve a significant reduction in pollution, but at reasonable costs. Conditions that might motivate the use of design standards include: technical and ecological information is complex; crucial knowledge is available at the central level of authorities rather than at the firm; firms are unresponsive to price signals and investments will have long-run irreversible effects; the standardization of technology holds major advantages; of only a few competing technologies available, one is superior; and monitoring costs are high: monitoring emissions is difficult, but monitoring technology is easy (Sterner, 2003). However, the design standard is a mandatory technology that leaves enterprises with little choice. They are not encouraged to explore cost-efficient ways of achieving pollution control. They cannot trade reductions betweens sources, and they are not given any incentive to develop cleaner technology. Furthermore, it typically is not feasible for a regulator to have knowledge about individual abatement levels or technologies for each firm. The information requirements and administrative costs are prohibitive. Typically, an environmental protection agency wants a standard technology that is the same for all and easy to monitor. In this case, abatement and emissions levels typically will not be optimal. A regulation that imposes a certain limit to harvests or to emissions instead of requiring a particular technology is called a performance standard (as distinguished from a design standard). Performance standards are significantly different from mandatory technology because they give firms considerable flexibility in the choice of abatement method by which to meet the mandated goal. They also leave the firms a choice between output reduction and abatement level, and trade-offs between polluting units are possible. With performance standards, a firm has the additional flexibility to reduce emissions not only by abatement investments but also by reducing output. This flexibility is usually part of the socially optimal outcome because the cost at the margin of reducing output to lower emissions may be smaller than the costs associated with additional investments in abatement. In many real-world cases, industrial pollution is controlled by licensing procedures that are a mixture of set emission levels (total or relative to output) and mandated technology. Licensing procedures do not allow for flexibility in attaining goals through trading between sources.

Liability systems are based on defining legal liability for the damages caused by certain types of pollution discharges and facilitating the collection of these damages. The Comprehensive Environmental Response, Compensation and Liability Act of 1980 defines legal rights to natural resources for local, state, and federal governments and specifies methods by which damages may be measured. The provisions of this act provide a means of facilitating the incorporation of the expected social cost of spills into the private cost calculation by potential polluters or internalizing the expected damages of spills. The legislation increases the probability that the firm will have to pay the social costs of its spills, so the firm is more likely to take appropriate safety measures. A variation of this system will involve defining legal liability, and then require potential polluters to obtain full insurance against any damages they might generate. The insurance industry would then require appropriate safety measures on the part of potential polluters. This type of system is generally utilized by generators, haulers, and disposers of toxic waste (Khan, 1998). With bonding systems, a potential degrader of the environment is required to place a large sum of money in an escrow account. The money will be returned if the environment is left undamaged (or returned to its original condition) and will otherwise be forfeited. The size of the bond should be large enough to provide appropriate safeguards by those posting the bond, or large enough that the government can use the funds to clean up damage if it occurs. Bonds have been employed in strip mining areas, where mining companies forfeit the bond if the land is not returned to its original condition. Other applications include companies that have leases to cut trees in public forests, and companies that transport oil or toxic substances (Khan, 1998).

The fourth category, "engaging the public," includes such mechanisms as information disclosure, labeling, and community participation in environmental or natural resources management. Dialogue and collaboration among the environmental protection agency, the public, and polluters may lead to voluntary agreements or voluntary approaches, which have become a fairly popular instrument recently. Voluntary agreements refer to a form of negotiated (and verifiable) contract between environmental regulators and polluting firms. A firm agrees to invest, clean up, or undergo changes to reduce negative environmental effects. In exchange, the firm may receive some subsidies or perhaps some other favor, such as positive publicity, a good relationship with the environmental protection agency, and perhaps speedier and less formal treatment of other environmental controls. This agreement is formalized in a model in which the polluter agrees to adopt a cleaner technology in exchange for more lenient regulation.

The direct production of environmental services (public goods) is another way in which the government can use its own personnel, know-how and resources to mitigate environmental market failures. Activities such as waste disposal, planting trees, stocking fish, creating wetlands, providing and maintaining national parks, treating sewage, and cleaning up toxic sites are common examples of government direct provision of environmental quality. However, government production of environmental quality is largely an ameliorative action, and in many cases it would have been better for society if the environmental degradation had been prevented in the first place. It is unlikely that such pressing problems such as air and water pollution, global warming, and the depletion of the ozone would be adequately addressed by direct provision by government or moral suasion (governmental attempts to influence behavior without actually stipulating any rules that constrain behavior (Khan, 1998; Sterner, 2003).

The lessons learned form the preceding discussion is that most U.S. environmental regulation has been in the form of command-and-control requirements. Command-and-control approaches require groups of similar sources to use a specific control technology or comply with a uniform emission rate requirement. Economists and policymakers have been critical of the command-and-control approach because : (i) some low cost emission reduction measures are not pursued; (ii) uniform requirements for broad categories of sources ignore differences in the costs of control at and the environmental impacts of emissions from different facilities; (iii) regulators are not in a position to identify the most cost –effective portfolio of control measures or how that mix may change over time; (iv) it creates disincentives to technology development, in that new, potentially more efficient facilities are typically subject to more stringent requirements and that sources may be required to place any new emission control technology on all their facilities; and

(v) development of detailed technology standards has been time-consuming, politically controversial, and administratively costly (Hobbs and Centolella, 1995).

There are two basic types of environmental regulation: emission taxes or effluent fees and marketable permit or allowance systems. Such systems provide affected sources the incentive and flexibility to achieve the lowest cost mix of pollution prevention and emission reduction measures. One classical solution to the problem of environmental externalities is a Pigouvian tax, a tax on emissions equal to the marginal environmental damage cost at the point where the marginal control cost and marginal damage cost functions intersect. To date, emission taxes have not been popular in the United States because of the costs which can be imposed on affected sources. Sources pay both for their emission reductions (with an incremental cost below the tax rate) and taxes on any residual emissions. Emissions taxes can increase economic efficiency by moving prices towards societal marginal costs and redistributing demand to less polluting substitutes. Marketable permit or allowance systems distribute limited authorizations to emit, which can be traded among sources as fungible commodities and are exhausted when a specified quantity of pollutant is emitted. In some systems, unused allowances may be banked from period to period. Given unhindered trading, actual emission reductions are made by the sources which can most cost-effectively do so. The distribution of compliance costs, however, depends on the original distribution of allowances. Allowances may be either distributed to specified sources-existing sources may receive allowances at no cost –or sold at auction. If allowances are auctioned off, sources' compliance costs may resemble their costs under an equivalent emissions tax. In some marketable permit systems, a small fraction of allowances is held back from distribution and sold in zero revenue auction to ensure market liquidity, provide price discovery, and inhibit oligopoly power. In a “zero revenue” auction, auction revenues would not be retained by the government and may be distributed in proportion to the original distribution of allowances (Hobbs and Centolella, 1995). From 1977 to 1986, the U.S. Environmental Protection Agency began to supplement command-and-control requirements with limited Emission Reduction Credit (ERC) trading programs: netting, offsets, bubbles, and banking. In each of the trading programs, to create a tradeable ERC the underlying emission reduction ha d to be: surplus to that required to meet existing requirements; enforceable by state and federal authorities; permanent; and quantifiable in comparison to an established level of baseline emissions. The cost and difficulty of identifying potential transactions and securing prior regulatory approval has substantially limited the creation and transfer of such credits. Despite limited use, these mechanisms reduced air and water pollution control costs

The basic difference between the tax and allowance approaches is that a tax system limits the maximum amount that any source is likely to spend on emission reductions, while an allowance system ensures that emissions from covered sources will not exceed a specified (annual or cumulative) quantity without regard to the cost of the last unit of emission reduction. If policymakers had perfect knowledge, either approach could be structured to achieve an equivalent result. Under conditions of uncertainty, however, if the marginal cost of further emission reductions is rising more rapidly than the rate of change in the value of marginal emission reductions (i.e. slope of the marginal cost function exceeds the slope of the marginal benefit function), an emissions tax approach may tend to produce smaller distortions from an economically efficient result (Baumol and Oates,1988). Researchers have used mathematical programming techniques in watershed studies to establish the TMDLs, discharge permit prices and efficient trading patterns.

Risk Programming Models in Agriculture

Various risk-programming techniques have been developed to address risk in decision making. A number of these models have been applied in agriculture to incorporate risk in farm management decisions. The various risk-programming techniques help farmers make production decisions or select optimal farm plans or management strategies that maximize net farm income under conditions of risk and uncertainty. However, most risk-programming applications in agriculture are based on either mean-variance or minimization of total absolute deviations (MOTAD) decision criteria. Markowitz (1959) developed quadratic programming methods to address risk and uncertainty issues. However, application of these methods did not only prove to be computationally difficult, but also require certain assumptions on the part of the decision maker. Hazell (1971) developed the MOTAD model for farm planning under uncertainty as an alternative to quadratic and semi-variance programming. Hazell's MOTAD approach uses mean absolute deviations as the risk measure. This model has been widely applied because it allows the development of a linear programming model with a parametric approach to risk and requires no particular assumptions about the behavior of the decision maker (Hazell, 1971). The MOTAD model has since become a commonly accepted approach to risk modeling. However, Tauer (1983) developed the Target MOTAD, a variation of Hazell's model that incorporates a safety level of income (or target income level) and a risk parameter that allows negative deviations from that safety (target) income level or represents the maximum allowable average income shortfall. The Target MOTAD calls for maximization of expected income subject to the requirement that income deviations below the target income not exceed some specified level. This model has been used for computing stochastically efficient mixtures of risky alternatives. Results from several studies indicate that Target MOTAD is computationally efficient. The model also has theoretical appeal because it generates a set of efficient choice alternatives that are members of the second-degree stochastic dominance (SSD) efficient set (Tauer, 1983; MacCamly and Kliebenstein, 1987). Teague et al. (1995) employed a farm-level risk programming model using a time-series of environmental risk indices to incorporate the stochastic, multi-attribute characteristics of environmental outcomes associated with agricultural production practices (Teague et al., 1995). This framework was used to evaluate the tradeoffs between environmental risk and net returns at farm level. "Environmental indices were developed which aggregated water quality effects across environments (surface water and groundwater) for a given form of contaminant (pesticides and nitrates). Restrictions on environmental outcomes were specified based on a target (or maximum) level of the environmental indices, and / or the acceptable level of compliance with that target." (Teague et al., AJAE 77 (Feb. 1995), p. 18). The findings of this study suggested that expected income is sensitive to nitrate loading restrictions, and relatively less sensitive to pesticide loading restrictions. However, it should be noted that these findings could be basin and crop specific. Results also indicated that prescriptions derived using deterministic environmental risk measures may ignore significant probabilities of exceeding an environmental standard (Teague et al., 1995). This study demonstrated that Target MOTAD could be used to find maximum expected farm income while insuring the probability that maximum contaminate levels (MCL) for any of several pollutants (pesticides, nitrates, phosphates) were not exceeded was less than a specified tolerance. An environmental risk income frontier can be traced out if the probability that a particular MCL not be exceeded can be varied upward from zero (Teague et al., 1995). Later, Qui, Prato and Kaylen (1998) extended Teague et al.'s (1995) farm-level Target MOTAD model to allow incorporation of both economic and environmental risks in agricultural production at watershed-scale level. The modified Target MOTAD model was used to evaluate the economic and environmental tradeoffs in a watershed by imposing a probability-constrained objective function to capture the yield uncertainty caused by random allocation of farming systems to soil types and by introducing environmental targets to incorporate environmental risk due to random storm events. Using the modified Target MOTAD framework, Qiu, Prato and Kaylen (1998) determined the tradeoff frontier between watershed net return and sediment yield and nitrogen concentration in runoff in a watershed. The findings of their study showed that the tradeoff frontier is significantly affected by environmental risk preference (Qiu, Prato and Kaylen, 1998). The preceding discussion demonstrates that the desire to reflect uncertainty of future events within decision-making problems has led to a number of risk models. Many of these risk models attempt to reflect the decision maker's expectations of possible outcomes and their probabilities, along with the decision maker's attitude toward assuming risk (Lambert and McCarl, 1985). For the sake of compactness, the next section focuses on and describes the theoretical environmental Target MOTAD model because of its usefulness and appropriateness to meeting the objectives of this dissertation research.

Environmental Target MOTAD Model

Tauer (1983) developed and describes the Target MOTAD as a two-attribute risk and return model. In this model, economic return is measured by the sum of the expected economic returns per unit of activity multiplied by individual activity levels. The riskiness of returns is measured by the probability-weighted average of the negative deviations of the resulting economic returns from a target return level under the different states of nature. Risk is varied parametrically so that a risk-return frontier is traced out. Based on the works of Tauer (1983), Teague et al. (1995) and Qiu, Prato and Kaylen (1998), the theoretical Target MOTAD model may be mathematically expressed as follows:

$$\max \ E(z) = \sum_{j=1}^{n} C_j X_j \qquad (2)$$

subject to

$$\sum_{j=1}^{n} a_{kj} X_j \leq b_k \qquad k = 1,\ldots,K \qquad (3)$$

$$T_e - \sum_{j=1}^{n} V_{rj} X_j + d_r \geq 0 \qquad r = 1,\ldots,s \qquad (4)$$

$$\sum_{r=1}^{s} p_r d_r = \lambda_e \qquad \lambda_e = M \to 0 \qquad (5)$$

for all $\mathrm{X_j}$ and $\mathrm{d_r} \geq 0$.

where $E(z)$ is the expected return of the farm plan; C_j, expected return of activity j; X_j, level of activity j; a_{kj}, amount of resource k used per unit of activity j; b_k, level of resource k available; T_e, target identified for the environmental indicator (the total annual maximum phosphorus load) ; V_{rj}, value of the environmental indicator for activity j in state of nature r (phosphorus runoff from hru j from the state of nature r) ; d_r, deviation above T_e for state of nature r (the phosphorus runoff deviation above Te for the state of nature r); p_r, probability that state of nature r will occur; λ_e, permissible level of compliance to T_e parameterized from M to 0 (risk aversion parameter); n, the number of activities; K, number of resource equations or constraints; s, number of states of nature (state of nature r refers to the HRU specifications and weather patterns that affect phosphorus runoff); and M is a large number. Equation (2) maximizes expected return of the solution set. Equation (3) fulfills the technical constraints. Equation (4) measures the revenue of a solution under state r. If that revenue is less than the target T, the difference is transferred to equation (5) via variable y_r. Equation (5) sums the positive deviations after weighing them by their probability of occurring, p_r. This Target MOTAD model identifies farm plans which maximize net returns but maintain environmental risk below a critical level or target.

Application of Mathematical Programming in Watershed Studies

Economic optimization models seeking efficient allocation of limited resources have been developed and used to identify optimal management strategies conducive to maximization of producer income in the agricultural sector. Watershed managers have employed economic optimization models to determine cost efficient nitrogen and phosphorus pollution abatement in various watersheds in the U.S. The optimal level of phosphorus load can be achieved by constructing an economic model that maximizes social welfare by equating estimated marginal abatement with marginal damage costs in the watershed. Though costs of reducing phosphorus loading fall on agricultural producers, industries, and municipalities in the watershed, a reduction in phosphorus loading may reduce costs of treating municipal water supplies and improve water quality in the region. It will also result in increased recreational benefits in the watershed.

Westra and Olson (2001) have used mathematical programming to determine the most efficient methods of reducing phosphorus loading to the Minnesota River by 40 percent. The authors found that by targeting specific areas in the watershed, the goal could be reached with less reduction in income than was possible if restrictions were uniformly applied to all producers (Westra and Olson, 2001). Ancev (2003) and Ancev, Stoecker and Storm (2003) developed an optimization model that was used to estimate the feasibility and cost of meeting various total maximum annual loads (TMAL) and the damage cost incurred by water treatment plants and recreation losses from each possible TMAL (Ancev, 2003). The results indicated the minimum sum of abatement costs (costs to reduce soluble phosphorus) from point and non-point sources plus damage costs from phosphorus pollution required a reduction of phosphorus loading from a current 51 tons to 23-26 tons per year (Ancev, 2003). The Oklahoma Water Resources Board (OWRB 2002) study called for an annual loading of not more than 18 tons per year. Ancev (2003) found that when the phosphorus load was limited to 18 tons, the marginal damage cost avoided from removing a kilogram of phosphorus was $11 while the cost of removing the kilogram of phosphorus was $27. The results also indicated that a combination of methods such as treating litter with alum, litter trading within the watershed , and land use changes would be required to achieve this objective at least cost (Ancev, 2003).

Ancev's (2003) approach employs a static spatial programming model. His approach does not adequately provide for linkages between sub-basins in the Eucha-Spavinaw watershed throughout the year. The flow of water and nutrients from one sub-watershed to the next is not linked to ensure that TMDLs are met at critical points along the streams within the watershed. The model does not account for total and soluble phosphorus runoff from each of the hydraulic response units on a monthly basis to determine the degree to which the TMDL is met throughout the year. The most likely points for total phosphorus constraints may be at the outlets of some sub-watersheds. These may particularly be where the low TMDLs are established for specific reaches within the watershed.

Another problem with Ancev's (2003) programming model is that it does not incorporate a risk analysis component. Similar studies have used models that accommodate risk measures to ensure the probability of exceeding the TMDL under variable weather conditions at various points in the watershed is less than a stated tolerance. Teague et al. (1995) and Qui et al. (1998) developed and demonstrated environmental risk programming models that minimize the cost of meeting the TMDL while insuring that the probability of violating the TMDL is less than a tolerance level. The work by Ancev, Stoecker, and Storm (2003) demonstrated that it was optimal to ship poultry litter from areas of excess soil test phosphorus (STP) to areas where soils were still low in phosphorus. However if an STP policy is adopted where producers must first test soils before applying litter, this option will exist only as long as the STP remains below the upper limit. If litter applications were limited to agronomic rates then such buildup might not occur although surface accumulations of phosphorus are vulnerable to runoff. The model does not account for phosphorus accumulation over time from land applications of organic and inorganic fertilizer. It does not predict the effect on STP levels and expected phosphorus runoff when producers change BMPs over time.

Overview of Hydrologic / Watershed Models

Rao (1996) defined a model as a simplification of nature representing a set of objects and their relationships and often describes a phenomenon that cannot be directly observed. Environmental models are developed to better understand natural phenomena and to better manage the natural resources. There is growing concern about the negative impact of various economic activities on the environment. Most of these environmental problems have a spatial dimension. On-site monitoring of the impacts of say, agricultural activities, on the natural environment is labour intensive and often expensive. Therefore, simulation modeling of nonpoint source pollution can provide useful information for decision-makers and planners to take appropriate land management measures and provide guidelines in the development and planning of agricultural management strategies (Liao and Tim, 1994). Watershed models abound in the hydrological literature. Numerous computer models have been developed to perform watershed assessments integrating hydrologic models with erosion models and identifying point and nonpoint pollution sources and assessing their impact on water quality within the watershed. Only hydrological models that are relevant to the problem being considered in this dissertation are described. Srinivasan and Arnold (1994) pointed out that each model addresses specific issues along with a set of assumptions and variable input requirements. They described models as either non-spatially distributed (e.g CREAMS and EPIC), or spatially distributed (e.g SWRRB and SWAT); single event (e.g. AGNPS and ANSWERS), or continuous time-scale (e.g. CREAMS, EPIC, SWRRB, ROTO and SWAT); field-scale (e.g CREAMS, EPIC and GLEAMS), or basin-wide (e.g AGNPS, ANSWERS, SWRRB and SWAT) (Srinivasan and Arnold,1994).

CREAMS is the acronym for Chemical, Runoff, and Erosion from Agricultural Management Systems, a field-scale model developed by the USDA-Agricultural Research Service (ARS) to simulate the impact of agricultural management systems on water, sediment, nutrients, and pesticides leaving the edge of a field (Knisel, 1980). Several other field-scale hydrology models evolved from the original CREAMS, including the Groundwater Loading Effects of Agricultural Management System (GLEAMS), and the Erosion Productivity Impact Calculator (EPIC). The GLEAMS model was developed for field-size areas to simulate pesticide groundwater loadings and evaluate the effects of agricultural management systems on the movement of agricultural chemicals within and through the plant root zone (Knisel, 1980; Leonard et al., 1987). The EPIC model is a continuous simulation model that can be used to determine the effect of management strategies on agricultural production and soil and water resources. The drainage area considered by EPIC is generally a field-size area, up to 100ha (weather, soils, and management systems are assumed to be homogeneous). The major components of EPIC are weather simulation, hydrology, erosion-sedimentation, nutrient cycling, pesticide fate, plant growth, soil temperature, tillage, economics, and plant environmental control (Williams et al., 1985; Arnold et al., 1990).

SWRRB is the acronym for Simulator for Water Resources in Rural Basins, a model that was developed by modifying the CREAMS daily rainfall model for large, complex basins. SWRRB operates on a daily time step and is capable of simulations up to 100 years or more. The model provides the efficient computation of sediment yield from small to large, complex watersheds. Major additions to the CREAMS models include allowing simultaneous computations for several subbasins within a large basin and adding components to simulate weather, return flow, pond and reservoir storage, crop growth, transmission losses, groundwater, and sediment routing. SWRRB allows basins to be divided according to landuse, soils, and topography. Since SWRRB places a limit on the number of subbasins within a watershed, some lumping of input parameters is necessary (GSWRL). However, a model was needed in the late 1980s to estimate the downstream impact of water management within Indian reservation lands in Arizona and New Mexico. Limitations in the size and number of sub basins and the methods employed to model the water and sediment transported out of the sub basins in which both routed directly to the watershed outlet led to the development of another model. The Routing Outputs to Outlet (ROTO) model took output from multiple SWRRB runs and routed the flows through channels and reservoirs. This model overcame the SWRRB sub basin limitation by linking multiple SWRRB runs together.

The input and output of multiple independent SWRRB runs was cumbersome and required considerable computer storage. In order to remove the difficulty of running the SWRRB model multiple times and then entering the output into ROTO, these two models were combined to create one new model, the Soil and Water Assessment Tool (SWAT). This allows simulations of very extensive areas but retains all of the features that made SWRRB a valuable simulation model. Since the early 1990s when SWAT was developed it has undergone continued review and expansion of capabilities. Each release has provided more features enabling greater analytical opportunities. In addition to the expanded capabilities, SWAT has also undergone extensive validation. Some of the features added include multiple hydrologic response units; auto-fertilization and auto-irrigation management options; canopy storage of water; addition of carbon dioxide component to crop growth model for climatic change studies; potential evapo-transpiration equation, lateral flow of water in the soil , and in-stream water quality equations and pesticide routing. Based on the usefulness of this tool, its users include Natural Resources Conservation Service (NRCS), Environmental Protection Agency (EPA), Environmental Consulting Firms, and Universities. SWAT model has been widely applied in various scenarios and watersheds (Spruill et al. 2000). Previous applications of SWAT in other parts of the United States have shown promising results (Srinivasan and Arnold 1994; Rosenthal et al., 1995). The model not only provides opportunities to improve watershed modeling accuracy and better long-term prediction of hydrologic components (Arnold et al. 1998), but also allows a great deal of flexibility in watershed configuration (Peterson and Hamlett ,1997).

Rosenthal et al. (1995) tested SWAT predictions of stream flow volume for the Lower Colorado River Basin in Texas. The model closely simulated monthly stream flow with a regression coefficient of 0.75. The model underestimated stream flow volume during extreme events, where precipitation was scattered with high intensity. Bingner et al. (1997) evaluated the SWAT model in Goodwin Creek Watershed in Northern Mississippi. The Nash-Sutcliffe coefficients, R2, values computed with observed monthly flow were all close to 0.80 except one measuring station, which was predominantly in forest. Smithers and Engel (1996) used the SWAT model to monitor the Animal Science and Greenhill watersheds in west central Indiana. The model underestimated totals for both while simulating none or very little base flow. Possible reasons of poor simulation were inappropriate soil input parameters or water budgeting procedures, which resulted in little drainage. Several hydrological studies have been conducted on the Eucha-Spavinaw watershed. Storm et al. (2001, 2003) conducted two hydrological studies in the watershed. In these studies they have developed, calibrated, and further refined the SWAT simulation model so it successfully tracks water and phosphorus movements in the watershed. Arnold (1998) points out that the SWAT model can predict the effect of alternative agricultural management practices on water, sediment and chemical yields from river basins.

SWAT Model Limitations

SWAT model limitations may be the result of the data we used in the model, inadequacies in the model itself, or our application of the model to simulate scenarios for which the SWAT was not designed. The SWAT model, like any other model, is a system of equations that represent a simplification of real world processes. Modeling requires many assumptions about real world processes because we do not quite understand them. This lack of knowledge on all the variables involved creates a great deal of uncertainty associated with modeling. For example, the nutrient loading for next year is unpredictable as next year's weather. There is great uncertainty associated with rainfall variability. The calibration of the SWAT model is based on the GIS data, water quality, and stream flow data. These data are not free from errors. However, methods are currently not available to quantify the uncertainty from sources other than weather. Weather is the driving force for any hydrologic model. Great care must be taken to include as much accurate observed weather data as possible. The SWAT model assumes and simulates litter applications as simple linear nutrient additions applied to the top 10mm of the soil surface in a uniform manner. In reality poultry litter could be incorporated into the soil after application or might lie on the soil surface until rainfall moves it into the soil or washes it away. There is more and closer litter-surface runoff interaction in the first few storm events after application than simulated by SWAT. Normally there would be higher phosphorus concentrations in surface runoff if it rained immediately following litter application in the fields. The SWAT model does not adequately simulate or allow for dramatic increases in phosphorus concentrations when litter is applied in the field. This implies that model output on a daily basis or monthly basis should be used with caution. However, if the SWAT model was well calibrated, these phosphorus loading discrepancies would have less influence on an average annual basis since they are typically not additive (Storm et al. 2001). The SWAT model can simulate and predict the relative impacts of long term agricultural management and land use on water quality in receiving water bodies in a basin. However, the model cannot properly simulate the effects of a single storm event.

CHAPTER III

CONCEPTUAL FRAMEWORK

The water quality problem resulting from excessive emissions of nutrients (e.g. phosphorus and nitrogen) into Lakes Eucha and Spavinaw is viewed as a case of market failure. The water pollution problem exists because property rights for clean water in the area are not clearly defined. Polluters, especially agricultural producers using inputs that have adverse effects on the environment such as pesticides and fertilizers (especially poultry manure) do not internalize the social costs associated with the use of such inputs in their private cost calculations. The divergence between private and social costs gives the polluters an incentive to use the inputs (e.g. poultry litter) in quantities exceeding socially optimal levels and thus excessive phosphorus runoff into water bodies. However, based on Coase's (1960) argument, agricultural enterprises (and all polluters in general) in the Eucha-Spavinaw watershed should strive to maximize the difference between total benefits resulting from cleaner waters and the total cost of achieving the environmental quality. It is not necessary to force environmental polluters to cease production activities resulting in the negative externality.

This research used a GIS-based hydrological simulation model and mathematical programming to assign management practices to particular areas within the watershed to effectively control non-point pollution at least cost to society. Estimated non-point source coefficients resulting from the simulation model can be input into a mathematical programming model to select most efficient management practices for each location in the watershed so that an overall pollution target is reached at least cost (Stoecker, Ramariz, and Ancev 2004). This study approaches the problem of phosphorus pollution from the social perspective, that is, with the objective of choosing a level of phosphorus control that maximizes total benefits to the society. In their earlier work on pollution abatement, Freeman, Haveman and Kneese (1973) suggest that the conceptual framework for determining optimal abatement levels be based on the concept of minimizing the sum of total pollution abatement cost and total environmental damage cost. The concept assumes that there exists a social welfare function with which to work. Tietenberg (2003) demonstrated that general social welfare can be maximized by minimizing the sum of total pollution abatement cost and total environmental damage cost. This dissertation research is conducted based on the same concept and assumes existence of a social welfare function to be maximized from consumption of market or economic output and environmental services. This relationship can be mathematically expressed as:

$$W = M + E \tag{6}$$

Where W is the social welfare function; M is the value of the market goods and services consumed by society and E is the value of environmental service consumed by society. If we let E* be maximum potential value of environmental services from pristine environment, D be costs of environmental damages from production and consumption of market goods and services, M* be maximum value of market goods and services with no pollution treatment, and T be costs associated with treating pollution, then we may state the actual values of market goods and services and environmental services as follows:

$$M = M^* - T \quad (7)$$

$$E = E^* - D \quad (8)$$

Substituting equations (7) and (8) into equation (6) redefines the total social welfare function as:

$$W = (M^* - T) + (E^* - D) = M^* + E^* - (T + D) \quad (9)$$

Given that M* and T* are fixed, equation (9) shows that we can maximize total welfare function by minimizing (T + D), the sum of pollution treatment costs and environmental damage costs. If we assume that both T and D are functions of a given pollutant (p), equation (9) may be recast to show that total welfare function will also be a function of pollutant (p) as follows:

$$W (p) = M^* + E^* - (T(p) + D(p)) \quad (10)$$

Maximizing total social welfare function in this form requires differentiating equation (10) with respect to p and setting the derivative equal to zero:

$$\frac{\partial W}{\partial p} = -\frac{\partial T}{\partial p} - \frac{\partial D}{\partial p} = 0 \quad (11)$$

$$\Rightarrow \; -\frac{\partial T}{\partial p} = \frac{\partial D}{\partial p} \quad (12)$$

Where $\partial T/\partial p$ is the marginal treatment cost, the change in total treatment costs from an additional unit of pollutant treated; and $\partial D/\partial p$ is the marginal environmental damage costs, the change in total environmental cost due to an additional untreated unit of pollutant emitted into the environment. The result in equation (12) implies that total social welfare is maximum when marginal treatment costs are equal to marginal environmental damage costs.

The second order conditions with respect to p are:

$$\frac{\partial^2 W}{\partial p^2} = -\frac{\partial^2 T}{\partial p^2} - \frac{\partial^2 D}{\partial p^2} \leq 0 \tag{13}$$

Equation (13) shows that the second order derivative is non-positive and thus consistent with the requirement for the point of maximum of the social welfare function. The implicit assumption here is that both $\partial^2T/\partial p^2$ and $\partial^2D/\partial p^2$ are non-negative at the optimal point for the second order derivative to be non-positive. Equation (13) implies that the treatment cost function should be increasing at a non-decreasing rate as the amount of pollution treatment increases. On the other hand, the environmental damage cost function should be increasing at a non-decreasing rate as the amount of pollution treatment decreases as illustrated in the example provided in Figure 4 below.

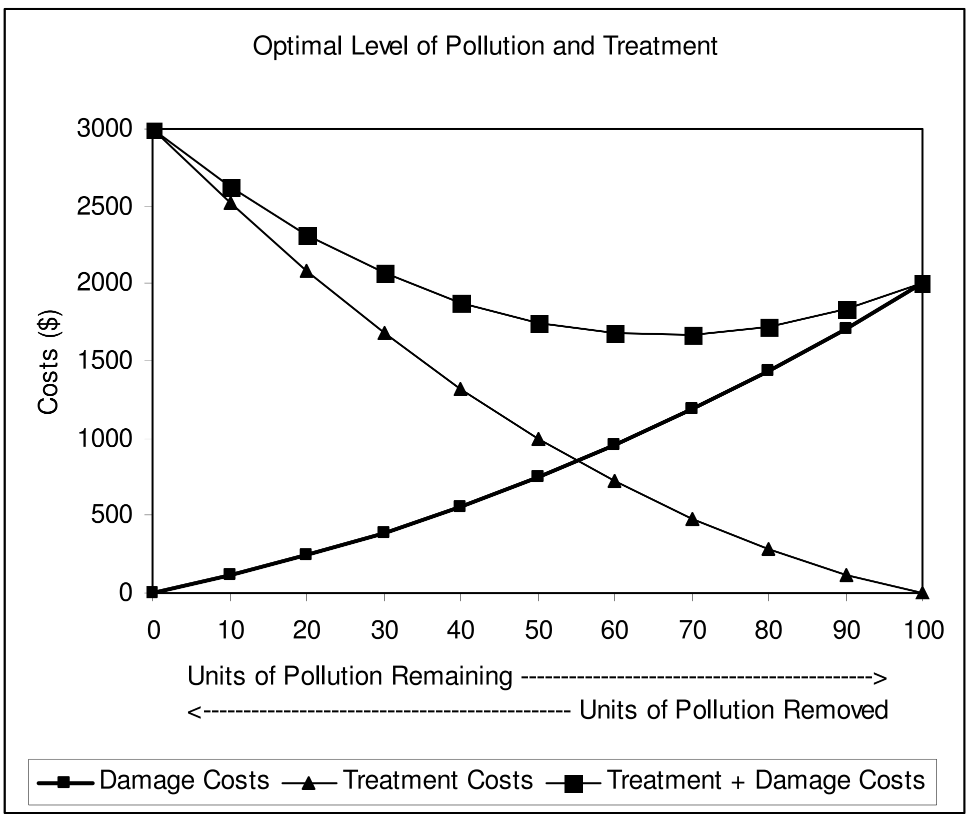


Figure 4 Damage Costs and Treatment Costs for Pollution Emissions.

In the case of water pollution as in the Eucha-Spavinaw watershed, the damage cost function represents the cost to the environment (such as dead fish, reduced recreational values, increased downstream water treatment costs) if various amounts of the pollutant (phosphorus) enters into the water supply. The treatment cost function represents all the costs incurred in the process of removing and / or preventing the pollutant (phosphorus) from entering the water course (Lakes Eucha and Spavinaw). The U-shaped total damage and treatment cost curve is obtained by vertical summation of the damage and treatment cost curves. The optimal level of pollution and treatment occurs at the minimum point of the total damage and treatment cost curve, a point at which the marginal treatment cost equals the marginal damage cost as illustrated in figure 5 below. Both Figures 4 and 5 indicate that the optimal level of pollution remaining and removed is 67 and 33 units, respectively.

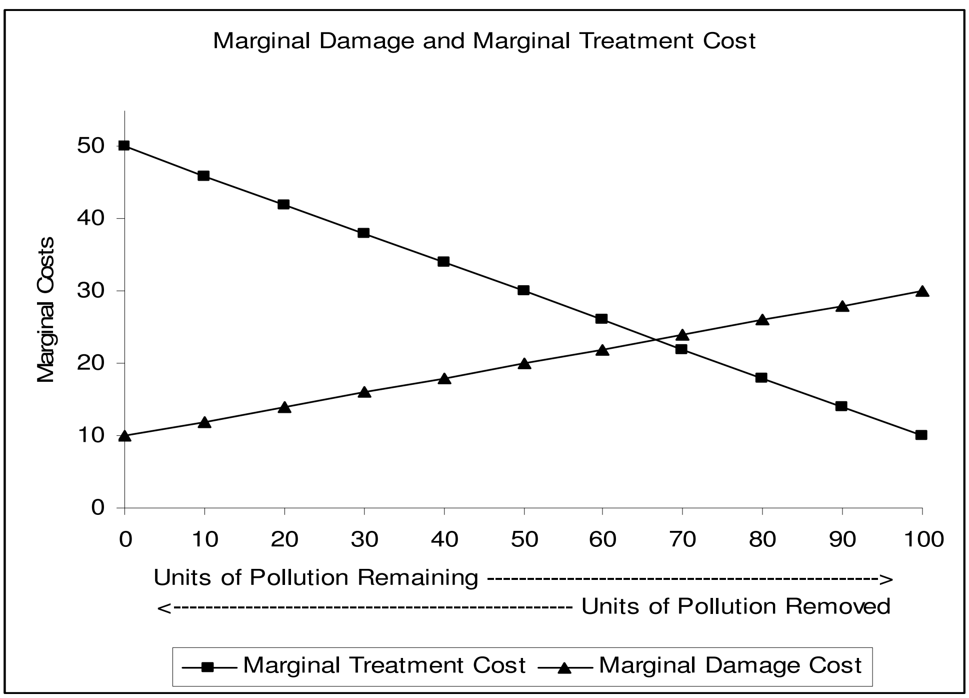


Figure 5 Marginal Damage and Marginal Treatment Costs for Pollution Emissions.

CHAPTER IV

METHODS AND PROCEDURES

Background

The main purpose of the study was to determine optimal pasture management practices within the Eucha-Spavinaw watershed that will effectively control phosphorus, nitrogen, and sediment runoff in a way that is least costly to society. The study used a two-step modeling approach that combines Geographical Information Systems (GIS) data-based biophysical simulations with mathematical programming to estimate the change in pasture management practices and producer income from the implementation of different environmental pollution standards or Total Maximum Daily Loads (TMDL) and policy instruments in the Eucha-Spavinaw watershed.

Simulation of Pasture Management Practices in the Watershed

A calibrated GIS-based Soil Water Assessment Tool (SWAT) model (Storm et al., 2002; Storm and White, 2005) was used to simulate hydrological and biophysical characteristics, production, and sediment, nitrogen and phosphorus runoff for 105 alternative pasture management practices. The study used daily weather records for temperature and rainfall for the period 1950 to 2004, from which three sets of 23 years of daily weather (rainfall and temperature) were selected for use in all simulations performed in this study. The first three years in each set comprised of daily weather data for the period 1993-95 and were used for warm-up and the base run of the simulation model. The other 20 years in each of the three weather data sets consisted of randomly selected sequence of years between 1950 and 2004. A series of simulation runs were performed for sixty (60) feasible pasture management practices in each HRU. The results of each simulation were then used to generate HRU specific coefficients for production, phosphorus runoff, nitrogen runoff and sediment runoff for each pasture management practice in each HRU. The respective coefficients obtained from the SWAT model were then used to develop a mathematical programming model used to select the most efficient pasture management practice for each HRU. The SWAT model delineated the Eucha-Spavinaw watershed into 90 subbasins with a total of 2416 hydraulic response units (HRUs) and 27 major soil types as indicated in Table 5 and Table 6 below.

Clarksville is the dominant soil type, covering about 44 percent of the watershed area, followed by Nixa which accounts for approximately 14 percent. Captina and Doniphan cover approximately 7 percent of the watershed area each. The soil types Razort and Tonti account for about 6 and 4 percent of the area, respectively. The other 21 major soil types account for about 18 percent of the Eucha-Spavinaw watershed area.

Table 5 Number of Subbasins and HRUs in the Eucha Watershed by Soil Type.

Soil Type	Number of Subbasins*	Number of Hrus	Area (hectares)	Share of Total Area (%)
Britwater	50	178	1,974	2.12
Captina	48	241	6,456	6.93
Carytown	4	11	152	0.16
Cherokee	3	4	20	0.02
Clarksville	78	605	41,388	44.45
Doniphan	36	167	6,160	6.62
Eldorado	2	2	26	0.03
Elsah	21	39	444	0.48
Healing	14	28	216	0.23
Hector	1	1	6	0.006
Jay	15	43	1,134	1.22
Linker	4	5	44	0.05
Macedonia	21	109	2,116	2.27
Mountainburg	1	1	1	0.001
Newtonia	14	83	3,063	3.29
Nixa	33	249	12,785	13.73
Noark	16	65	2,417	2.60
Peridge	19	78	1,544	1.66
Razort	49	159	5,165	5.55
Secesh	20	49	537	0.58
Shidler	1	1	1	0.001
Stigler	13	36	427	0.46
Taft	2	2	1	0.001
Taloka	16	63	2,324	2.50
Tonti	34	165	3,724	4.00
Waben	5	9	47	0.05
Water	14	23	942	1.01
Grand Total	90	2416	93,115	100.0

** Some soil types are found in more than one subbasin in the watershed.*

Table 6 Soil and Land Use Delineation in the Eucha-Spavinaw Watershed.

	Land Use and Area (Hectares)					
Soil Type	Pasture	Forest	Range	Crop	Urban	Water
Britwater	1,111	593	145	5	27	91
Captina	5,150	404	201	316	383	2
Carytown	127	0	16	0	8	1
Cherokee	19	0	0	0	1	0
Clarksville	5,932	32,810	2,327	11	152	156
Doniphan	4,353	1,161	398	73	172	3
Eldorado	26	0	0	0	0	0
Elsah	85	313	33	0	4	9
Healing	175	15	7	17	2	0
Hector	6	0	0	0	0	0
Jay	985	0	32	89	27	1
Linker	44	0	0	0	0	0
Macedonia	1,460	291	111	168	86	0
Mountainburg	1	0	0	0	0	0
Newtonia	2,224	128	84	566	61	0
Nixa	5,752	5,659	994	1	377	2
Noark	394	1,793	220	0	9	2
Peridge	1,339	0	2	122	80	0
Razort	1,118	3,716	306	2	22	2
Secesh	210	258	30	2	33	4
Shidler	0	0	0	0	1	0
Stigler	368	0	0	36	23	1
Taft	0	0	0	1	0	0
Taloka	1,948	0	60	271	44	2
Tonti	3,039	237	145	70	230	3
Waben	44	0	0	1	2	0
Water	5	19	3	0	2	912
Total	35,916	47,396	5,113	1,751	1,747	1,191
Share (%)	38.57	50.90	5.49	1.88	1.87	1.28

Table 6 shows land uses in the Eucha-Spavinaw watershed by major soil types. An HRU represents a combination of a major soil type and land use within a subbasin. It can be seen from Table 6 that about 51 percent of the watershed area is forest. Pastureland accounts for approximately 39 percent while rangeland is about 5 percent of the watershed area. Cropland and urban area account for about 2 percent of the watershed area each. Water accounts for about 1 percent of the total watershed area. A series of simulation runs were performed for a total of one hundred and five feasible pasture management practices in each hydraulic response unit (HRU) in the Eucha-Spavinaw watershed. Potential alternative pasture management practices were simulated using different combinations of land use/land cover, rate of poultry litter application, commercial nitrogen, minimum biomass retained during grazing, and stocking rates as shown in tables 7, table 8 and table 9 below.

Table 7 Codes for Various Levels of Management Practice Variables Used .

Code*	Land Use	Litter Application Rate (kg/ha)	Nitrogen Application Rate (kg/ha)	Minimum Plant Biomass for Grazing (kg/ha)	Stocking Rate (AU/ acre)
0	-	0	0	-	-
1	AGRL	2,000	50	1,100	0.63
2	HPAS	4,000	100	1,600	1.00
3	LHPA	6,000	150	2,000	1.26
4	LLPA	1,765	200		
5	LMPA	3,529			
6	LPAS	5,294			
7	MPAS				
8	RNGB				
9	FRST				

* Codes in column 1 mean different things in columns 2 - 6.

Table 7 shows the codes assigned to each management activity simulated. There are eight types of land use / land cover, seven levels of litter application rate, five levels of nitrogen application rate, three levels of minimum biomass for grazing, and three levels of stocking rate. For instance, the numeric activity code 62133 should be interpreted as follows: The first digit (6) represents land use (LPAS in this case); the second digit (2) represents litter application rate (4,000 kg/ha); the third digit (1) represents nitrogen application rate (50 kg/ha); the fourth digit (3) represents minimum biomass for grazing

(2,000 kg/ha); and the fifth digit (3) represents stocking rate (1.26 AU/acre).

Table 8 Simulated Fertilizer and Minimum Biomass Maintained During Grazing.

Pasture No.	Pasture Condition	Litter Applied (kg/ha)	Elem-N (kg/ha)	Total N (kg/ha)	Total P (kg/ha)	Plant Biomass Retained (kg/ha)	Hydrologic Group / SCS-Curve Number A	B	C	D
1	P	0	0	0	0	1,100	68	79	86	89
2	P	0	50	50	0	1,100	68	79	86	89
3	P	2,000	0	60	28	1,100	68	79	86	89
4	F	0	100	100	0	1,600	49	69	79	84
5	F	2,000	50	110	28	1,600	49	69	79	84
6	F	4,000	0	120	56	1,600	49	69	79	84
7	G	0	150	150	0	2,000	39	61	74	80
8	G	0	200	200	0	2,000	39	61	74	80
9	G	2,000	100	160	28	2,000	39	61	74	80
10	G	2,000	150	210	28	2,000	39	61	74	80
11	G	2,000	200	260	28	2,000	39	61	74	80
12	G	4,000	50	170	56	2,000	39	61	74	80
13	G	4,000	100	220	56	2,000	39	61	74	80
14	G	4,000	150	270	56	2,000	39	61	74	80
15	G	4,000	200	320	56	2,000	39	61	74	80
16	G	6,000	0	180	84	2,000	39	61	74	80
17	G	6,000	50	230	84	2,000	39	61	74	80
18	G	6,000	100	280	84	2,000	39	61	74	80
19	G	6,000	150	330	84	2,000	39	61	74	80
20	G	6,000	200	380	84	2,000	39	61	74	80

Table 8 shows simulated pasture conditions resulting from alternative combinations of fertilizer and minimum biomass maintained during grazing. These pasture management scenarios were evaluated assuming a low stocking rate of 0.63 AU/acre and a medium stocking rate of 1.0 AU/acre. A management scenario that maintained minimum biomass during grazing of 1,100, 1,600 and 2,000 kg/ha was considered to represent a poor, fair, or good pasture, respectively.

Table 9. Simulated Fertilizer and Minimum Biomass Maintained During Grazing.

Pasture No.	Pasture Condition	Litter Applied (kg/ha)	Elem-N (kg/ha)	Total N (kg/ha)	Total P (kg/ha)	Plant Biomass Retained (kg/ha)	Hydrologic Group / SCS-Curve Number A	B	C	D
1	P	0	0	0	0	1,100	68	79	86	89
2	P	0	50	50	0	1,100	68	79	86	89
3	P	2,000	0	60	28	1,100	68	79	86	89
4	F	0	100	100	0	1,600	49	69	79	84
5	F	2,000	50	110	28	1,600	49	69	79	84
6	F	4,000	0	120	56	1,600	49	69	79	84
7	F/G	0	150	150	0	2,000	44	65	76	82
8	F/G	0	200	200	0	2,000	44	65	76	82
9	F/G	2,000	100	160	28	2,000	44	65	76	82
10	F/G	2,000	150	210	28	2,000	44	65	76	82
11	F/G	2,000	200	260	28	2,000	44	65	76	82
12	F/G	4,000	50	170	56	2,000	44	65	76	82
13	F/G	4,000	100	220	56	2,000	44	65	76	82
14	F/G	4,000	150	270	56	2,000	44	65	76	82
15	F/G	4,000	200	320	56	2,000	44	65	76	82
16	F/G	6,000	0	180	84	2,000	44	65	76	82
17	F/G	6,000	50	230	84	2,000	44	65	76	82
18	F/G	6,000	100	280	84	2,000	44	65	76	82
19	F/G	6,000	150	330	84	2,000	44	65	76	82
20	F/G	6,000	200	380	84	2,000	44	65	76	82

Table 9 shows simulated pasture conditions evaluated using the stocking rate of 1.26 AU/acre. However, a maintained minimum biomass of 2,000 kg/ha was assumed to represent a good / fair pasture at the stocking rate of 1.26 AU/acre as shown in Table 9.

The NRCS curve numbers were adjusted according to the pasture condition and hydrologic group (A, B, C, or D) assigned to each soil type. It is generally assumed that poor pastures are more susceptible to runoff because of less land cover. Thus, poor pastures were assigned a higher curve number. A total of 105 alternative pasture management scenarios were simulated. Each of the 105 pasture management practices was applied to each of the 2,416 HRUs in the watershed.

Row Crops Management (Green Beans and Winter Wheat)

The row crops, that is, winter wheat and green beans were modeled as a graze-out wheat-and-green bean rotation (green beans followed by winter wheat). It was assumed that farmers in the Eucha-Spavinaw watershed undertake a generic spring plowing operation, apply commercial nitrogen fertilizer to the cropland and then plant green beans in early May. Commercial nitrogen fertilizer will be applied at the rate of 0-200 kg / ha irrespective and independent of the amount of poultry litter applied. A metric ton of poultry litter was assumed to contain 14 kg of phosphorus and 30 kg of nitrogen. They will then harvest and kill the green beans, apply commercial nitrogen fertilizer and till (generic fall plowing operation) the cropland in early August. Commercial nitrogen fertilizer will be applied at the rate of 0-200 kg / ha irrespective and independent of the amount of poultry litter applied. Farmers are expected to plant winter wheat in early September and start a grazing operation in early November and another grazing operation in mid February at stocking rates of 0.63, 1.00 and 1.26 animal units (AU) per acre with 7.4, 11.8 and 14.9 kg of dry biomass consumed per day. Three levels of biomass maintained during grazing were assumed depending on the condition of the pasture. It is assumed that grazing is suspended when dry biomass per hectare falls below 1,100, 1,600, and 2,000 kg for pastures in poor, fair, and good condition respectively (OSU Extension Publication F-2586).

Another grazing operation will be undertaken in mid February. Farmers will then apply poultry manure to the winter wheat in early March. Based on fertilization recommendations, four litter application rates ranging from 0 to 6,000 kg / ha would be modeled, with 2,000 kg / ha as the base application rate. Phosphorus applied on cropland is assumed to come solely from poultry litter. Each ton of litter was assumed to contain 30 kg of nitrogen and 14 kg of phosphorus. Farmers are expected to harvest and kill the winter wheat in early May before the spring plowing, fertilizer application and planting the green beans. The inclusion of green beans (a legume crop) in the rotation improves the soil structure, reduces erosion, and offers farmers the possibility of reducing commercial nitrogen fertilizer needs as well as fertilizer costs for the subsequent crop. This is because legume crops can replace some of the nitrogen in the soil by fixation. The rotation would help reduce the farmers' cost of production by naturally breaking the cycle of weeds, insects, and diseases that are limited by their plant hosts on the cropland (USDA, 1993).

Reducing Poultry Litter Application Rate

Nutrient management plans help farmers use nutrients (mainly nitrogen and phosphorus) efficiently for optimum economic benefit to the farmer, while minimizing impact on the environment. However, excessive application of fertilizers resulted in nutrient contamination of water bodies. Recent studies have shown farms where poultry manure is the major or only fertilizer source may be suffering from excessive application of phosphorus, especially in those where their soils have extremely high phosphorus content caused by previous application of manure. Row crops contribute 49 percent of the total phosphorus loading to Lake Eucha. Majority of the non-point source total phosphorus loading is attributed to the elevated soil test phosphorus from row crops. The majority of soluble phosphorus is due to the application of poultry litter to pastures. Total phosphorus and soluble phosphorus loads are expected to decline by 16 percent and 27 percent respectively should application of poultry litter cease (Storm et al., 2002). Farmers using poultry litter can reduce phosphorus loading in the watershed by reducing the amount of poultry litter applied on the cropland. This objective is attainable if farmers could reduce the poultry litter application rates applied on the crops they grow. Farmers grow different crops on different soils and topography. Therefore, each distinct agricultural HRU would have a different optimal litter application rate, crop yield response to nutrients applied with the litter, and nutrient run off. The land uses modeled are low-biomass pasture (LPAS), medium-biomass pasture (MPAS), high-biomass pasture (HPAS), litter low-biomass pasture (LLPA), litter medium-biomass pasture (LMPA), and litter high-biomass pasture (LHPA), winter wheat (WWHT), green beans (GRBN), rangeland (RNGB) and forests (FRST). It is assumed that poultry litter is applied only to pastures and row crops in the management simulations.

Low, Medium, and High-Biomass Pasture Management

The model simulated low-biomass pasture (LPAS), medium-biomass pasture (MPAS), and high-biomass pasture (HPAS) management systems as described in Table 10.

Table 10 Low, Medium and High Biomass Pasture Management Scenarios.

Item	LPAS	MPAS	HPAS
Crop used on pastures	Tall Fescue	Tall Fescue	Tall Fescue
Type of fertilizer	Litter/Nitrogen	Litter/Nitrogen	Litter/Nitrogen
Grazing period (days)	270.0	270.0	270.0
BIO_MIN (kg/ha/day)	1,100.0	1,600.0	2,000.0
BMEAT (kg/ha/day)	7.4	11.8	14.9
BMTRMP (kg/ha/day)	7.4	11.8	14.9
WMANURE (kg/ha/day)	2.4	3.8	4.8

Table 10 shows that farmers in the Eucha-Spavinaw watershed were assumed to graze their cattle on Tall Fescue in all three pasture management systems. In the cases of LPAS and MPAS systems, we assumed that the tall fescue was seeded in early January and cattle put on it beginning of March. The length of the grazing period is 270 days. However, the minimum plant biomass for grazing to occur (BIO_MIN) on the LPAS and MPAS was maintained at 1,100 kg/ha and 1600 kg/ha, respectively. Due to this difference in minimum plant biomass before grazing could be suspended, it was assumed that the LPAS yielded more runoff due to less plant cover compared to the MPAS system. This was achieved by setting SCS-curve numbers (CN2) for the LPAS higher than MPAS depending on the soil hydrologic group. However, it was assumed that the amount of biomass removed (BMEAT) and trampled (BMTRMP) were 7.4 kg / ha and 11.8 kg / ha (dry weight), respectively. The HPAS system also involved tall fescue seeded in early January. Farmers were expected to apply commercial nitrogen fertilizer at rates ranging from 0-200kg / ha and then start grazing in March. Grazing would be suspended when plant biomass was less than 2,000 kg / ha. The length of the grazing period was the same as in the other two systems, but the amount of biomass removed, manure deposited and plant biomass trampled daily was set at twice that of the LPAS. The SCS-curve number was assumed lower compared to the other two systems. This accounted for the fact that the HPAS system had more plant biomass left after grazing and thus would have lesser runoff than the LPAS and MPAS.

<u>Litter Low, Medium and High-Biomass Pasture Management</u>

The model simulated three pasture management systems that apply poultry manure on tall fescue as shown in Table 11. These are litter low-biomass pasture (LLPA), litter medium-biomass pasture (LMPA), and litter high-biomass pasture (LHPA). The planting, grazing and harvesting operations in these three systems are the same and are carried out as described under the HPAS system. In the LLPA, LMPA, and LHPA systems, farmers are assumed to apply poultry manure to tall fescue in March before grazing. Based on fertilization recommendations, four litter application rates ranging from 0 to 6,000 kg/ha will be modeled, with 4,000 kg/ha as the base application rate for grasses. The model assumes a choice of nitrogen replacement by commercial fertilizer at litter application rates less than the base application rate to maintain the current total nitrogen rate and forage production.

Table 11 Litter Low, Medium, and High Biomass Pasture Management Systems.

Item	LLPA	LMPA	LHPA
Crop planted (Jan 1)	Tall Fescue	Tall Fescue	Tall Fescue
Fertilization (Mar 1)	Broiler-Fresh Manure	Broiler-Fresh Manure	Broiler-Fresh Manure
Grazing (Mar 1) (days)	270.0	270.0	270.0
BIO_MIN (kg/ha/day)	1,100.0	1,600.0	2,000.0
BMEAT (kg/ha/day)	14.9	14.9	14.9
BMTRMP (kg/ha/day)	14.9	14.9	14.9
WMANURE (kg/ha/day)	4.8	4.8	4.8

For application rates that exceeded the base rate, the nitrogen applied on the grasses was assumed to come from the poultry litter. Both litter and nitrogen application rates were based on fertilization recommendations. Phosphorus applied on cropland was assumed to come solely from poultry litter. Each ton of litter was assumed to contain 30 kg of nitrogen and 14 kg of phosphorus. Grazing was suspended when plant biomass was less than 1,100, 1,600 and 2,000 kg /ha for the LLPA, LMPA and the LHPA systems respectively. The respective SCS-curve numbers were adjusted depending on the level of maintained biomass during grazing and the soil hydrologic group.

Range Management

The Eucha-Spavinaw basin was modeled as having brushy rangeland. Much of the default management parameters for rangeland areas remained unchanged. Two management operations were included in this system: planting and harvesting operations. The planting operation initiated plant growth in January when the leaf area index of the canopy equals 0.20. The harvesting operation was undertaken at a plant biomass target of 1,100 kg / ha. The total number of heat units required for the plants to reach maturity was estimated at 1,800. The amount of heat units accumulated each day is equal to the average daily temperature minus the base temperature of the plant. The base temperature is the minimum temperature required by the plant to grow.

Forest Management

The Eucha-Spavinaw basin was modeled as having mixed forests (existence of both deciduous and evergreen trees in the same locality). Much of the default management parameters for forested areas remained unchanged. Two management operations were included in this system: planting and harvesting / kill operations. The model assumed the trees were growing at the beginning of the simulation. The planting operation initiated growth of trees at a fraction of base zero potential heat units of 0.150, with a leaf area index of the canopy of 0.200. The total number of heat units required for the trees to reach maturity was estimated at 2,082. The harvest and kill operation stoped the growth of trees at a fraction of base zero potential heat units of 1.200, with a biomass target of 9.0 kg/ha. It was assumed that no nutrient would be removed with harvested material.

Using Aluminum Sulfate (Alum) to Reduce Phosphorus Loading

Given elevated phosphorus levels in runoff from agricultural land on which poultry manure was used, there is need to determine alternative methods for controlling either available phosphorus content of the poultry litter or the phosphorus holding capacity of the soil. Our model allows for treatment of poultry litter with alum. A study conducted by

Cestti et al, 2003 found that adding aluminum sulfate to poultry litter provided benefits for both the farmer and the environment. They reported that the presence of alum in the poultry litter allowed it to trap nitrogen in the fertilizer and reduce nitrogen losses through ammonia volatilization (Cestti et al, 2003). This increased the level of nitrogen available to plants. Moore and Miller (1994) reported that when alum was added to poultry litter, it reduced phosphorus runoff by tying up and transforming soil labile phosphorus into more stable aluminum phosphate compounds that are insoluble. The resultant compounds were not soluble (unless the lake became acidic) and hence were not readily available to promote algae growth in the water body. Moore (1999) claimed that farmers using alum-treated poultry litter on their cropland could produce runoff with less than 75 percent phosphorus content.

Based on studies conducted by Moore (1999) and Moore, Daniel and Edwards (1999; 2000), it was assumed in this study that Alum is added to litter in the poultry house in a ratio of 1 part alum to 10 parts poultry litter. We also assumed that the Alum would reduce soluble phosphorus runoff by 75 percent. Since Alum would reduce nitrogen loss in the poultry house the average ton of litter would contain 34 kg of nitrogen rather than 30 kg for untreated litter. Hence only 88 percent as much litter had to be applied for the same amount of nitrogen. Alum-treated poultry litter was assumed to impose an additional cost of $5.00 per ton to farmers undertaking the agricultural activities in HRUs where alum-treated litter is applied. Poultry litter was applied to pastures at levels consistent with meeting the nitrogen requirement of the crop. The quantity of poultry litter applied was reduced by 88 percent to account for increased nitrogen in the litter while the amount of soluble phosphorus runoff from the SWAT simulation runs was reduced by 75 percent to account for the tied up phosphorus in Alum-treated poultry litter and not readily available for plant uptake.

Mandatory (Command-and-Control) Phosphorus Abatement Policies

Soil Test Phosphorus (STP) – Based Litter Application Policy

This criterion allowed litter application only to those soils where the STP was not higher than a specified cut-off value. Crop yield reaches a plateau at this critical STP value. Continued application of phosphorus above the critical STP would not increase crop yield. However, it would increase phosphorus levels in the soil, a high proportion of which may runoff during storm events. Farmers may improve crop yields while preventing the runoff of the excess phosphorus during the storm events by applying poultry litter only to soils with STP values lower than the given threshold. This is a "command-and-control" regulatory approach, where threshold standards are set and enforced to reduce total phosphorus loading in the entire Eucha-Spavinaw watershed.

The amount of phosphorus needed in a fertilizer or manure program for obtaining optimum yield is measured using an STP index. This is a value at which the soil has sufficient phosphorus that could be used for plant uptake. Based on OSU fertilization recommendations, Oklahoma soils require, on average, 120 lbs of available P per acre (STP value of 120). It was assumed in this study that farmers do not apply poultry manure to pastures and row crop on soils with STP higher than 120, but had an option of meeting nitrogen requirement using commercial nitrogen fertilizer. The STP for all pastures and row crop was assumed to be the same across sub-basins. Forest STP was assumed constant in all the management practice simulations.

Uniform Conversion vs Targeted Land Use Conversion Policy

Previous studies have concluded that overgrazed pasture and row crop contribute relatively more to the phosphorus loading compared to that from hay and well-maintained pasture. Storm et al. (2002) estimated row crop contributed 49 percent of the total phosphorus loading in the Eucha-Spavinaw basin. The study found that this contribution of total phosphorus loading from row crop fields is disproportionately high relative to pasture. It further indicated that changing row crop to pastures would reduce total phosphorus loads by almost 50 percent. Producers, therefore, may significantly reduce total phosphorus loading in the watershed by changing land use. Thus, it is assumed in this study that low-biomass pasture (LPAS) and medium-biomass pasture (MPAS) will be converted to high-biomass pasture (HPAS); litter low-biomass pasture (LLPA), litter medium-biomass pasture (LMPA), and row crops will be converted to litter high-biomass pasture (LHPA). These conversions were modeled assuming both mandatory uniform conversion and site-specific land use conversion policy which takes into account economic characteristics, biophysical conditions and phosphorus runoff potential of the agricultural land.

Determination of the Value of Biomass Consumed During Grazing

The value of hay and pasture consumed during grazing was derived based on a 100 cow unit size cow-calf enterprise budget obtained from Oklahoma State University

Cooperative Extension Service as shown in Table 12.

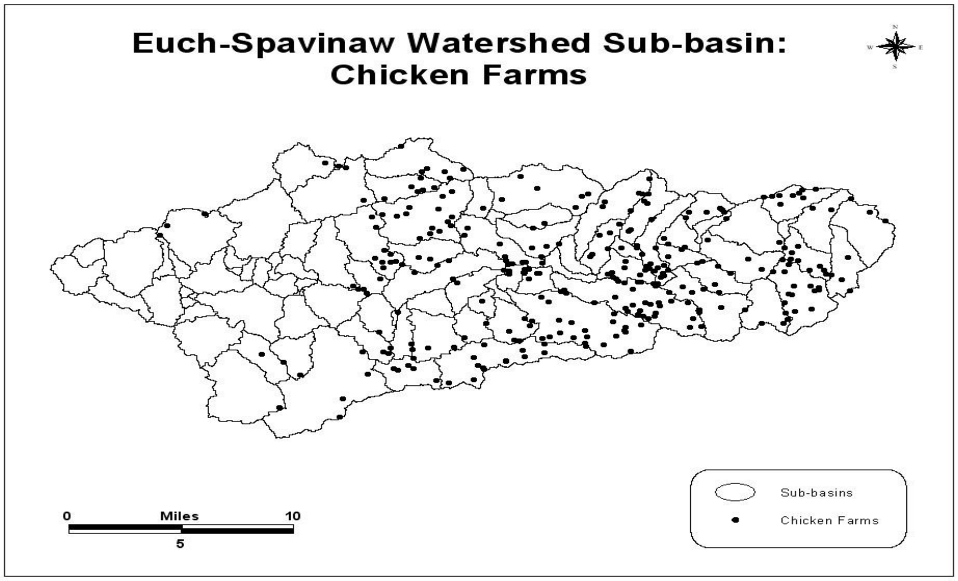
Table 12. 100 Herd Cow Calf Enterprise Budget.

Production	Weight	Unit	Price / Cwt	Qty	Revenue
Steer Calves	470	Lbs./hd	$107.42	18.91	$9,547
Heifer Calves	470	Lbs./hd	$100.04	7.49	$3,522
Cull Cows	1150	Lbs./hd	$44.27	12	$6,109
Cull Replacement	825	Lbs./hd	$84.34	12	$8,350
Cull Bulls	1750	Lbs./hd	$58.58	1	$1,025
Stockers	623	Lbs./hd	$112.00	40	$27,910
Total Receipts					$56,463
Protein Supp. $ Salt	1	hd.	$44.40	1.1	$4,884
Minerals	1	hd.	$14.07	1.1	$1,548
Vet Services	1	hd.	$7.14	1.1	$785
Vet Supplies	1	hd.	$1.16	1.1	$128
Marketing	1	hd.	$6.91	1	$691
Mach. Fuel,Oil, Repairs	1	hd.	$24.09	1.1	$2,650
Machinery labor	1	hrs.	$9.25	2.65	$2,451
Other labor	1	hrs.	$9.25	3	$2,775
Other expense	1	hd.	-	1.1	
Annual Operating Capital		Dollars	0.0825	184.62	$1,523
Total Operating Costs					$17,435
Other Fixed Costs					$12,926
Net Return to Hay and Pasture					$26,102
		lbs/day	days/yr	lbs/yr	kg/yr
Cow		25	365	9125	4139
Bull		25	365	365	166
Replacement Heifer		18	365	788	358
Stocker		14	100	560	254
Hay and Pasture Required Per Cow Unit					4916
Net Revenue per Mg Biomass Consumed ($26,102/100hd/4.92)					$53.05

We assumed that part of the calf crop were kept beyond weaning and sold later as stockers. The cattle prices used were based on Oklahoma direct feeder cattle sales for the trade period May 12 through May 18, 2007. Table 12 above shows the modified OSU 100 herd cow-calf enterprise budget with the net return to hay and pasture estimated at $26,102. The hay and pasture required per cow unit was estimated at 4916 kg per year. Based on these estimates, the value of biomass consumed during grazing was estimated at $53.05 per metric ton.

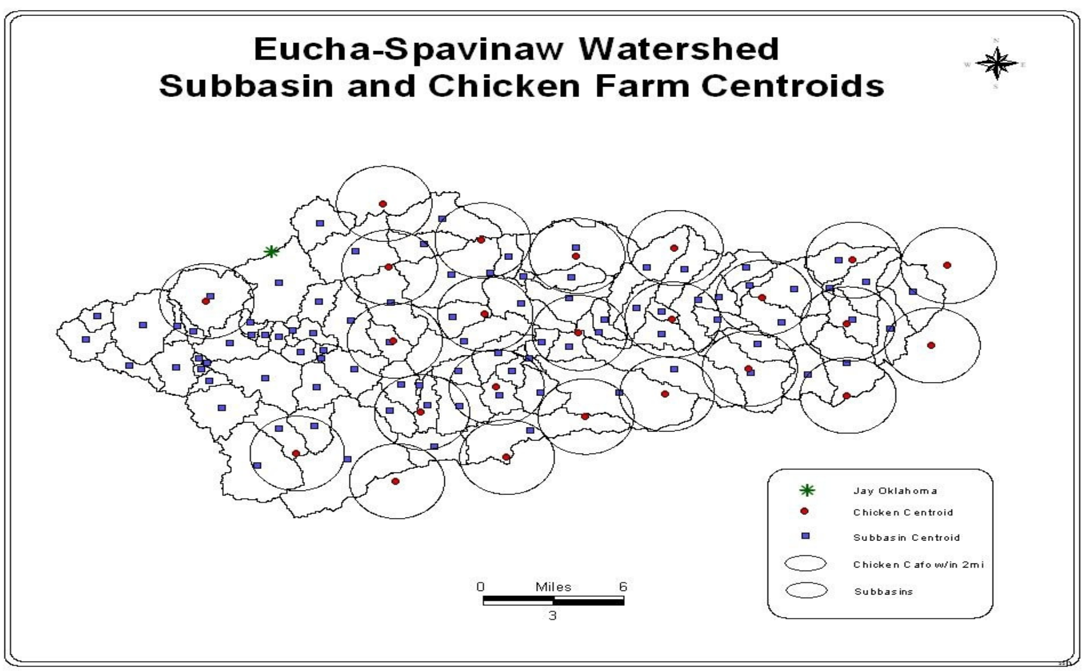
Development of the Transportation Matrix

The development of the transportation matrix required knowledge and information regarding the location and amount of litter produced from poultry operations in the Eucha-Spavinaw watershed. Based on the work done by Storm and White (2001), this study assumed that there are 1,053 broiler houses in the Eucha-Spavinaw watershed with an estimated output of approximately 89,500 tons of litter per year. Figure 6 and Figure 7 below show chicken farms as well as sub-basins and chicken farm centroids that were established as litter shipment points in the Eucha-Spavinaw watershed for the purpose of developing the transportation matrix. Three hundred chicken farms were assigned into twenty four groups ensuring that no chicken farm was located more than two miles from a group centroid as shown in Fig. 7. This exercise allowed us to limit the number of transportation activities in the mathematical programming model.



Source: Storm et al. (2002).

Figure 6 Chicken Farms in the Eucha-Spavinaw Watershed.



Source: Adapted from Storm et al. (2002).

Figure 7 Sub-basins and Chicken Farm Centroids in Eucha-Spavinaw Watershed.

Four distance calculations were performed. The average distance from each chicken farm

to the centroid of the group to which it was assigned was determined using ArcView Version 3.3; the distance from each chicken farm centroid to a point on the nearest road was estimated using the nearest feature algorithm; the distance from the point on the road nearest each chicken farm to a point on the road nearest each sub-basin centroid was estimated using a multi-path script; and lastly the nearest feature algorithm was used to determine the distance from the road to the sub-basin centroid. We used the same process to create a transportation matrix from each chicken farm centroid to Jay, Oklahoma for location of a possible litter-to-energy processing plant. This approach resulted in a matrix with 2208 possible transportation activities constituted from each of the 24 chicken farm centroids supplying litter to each of the 92 sub-basin centroids. Cost estimates for transporting litter from chicken farm centroids to subbasin centroids were based on information supplied by BMPs Inc. The cost for loading and coordinating a haul ranged from \$7.50 to \$8.00 per ton. The cost of hauling ranged from \$3.25 to \$3.50 per loaded mile per truckload. Each truck averaged 23 tons per load. The loaded mileage was a one-way distance. No direct cost for spreading, but BMPs, Inc. would coordinate spreading at an average of \$6 per short ton
(BMPs, Inc 2006).

The estimated litter transportation costs, value of biomass consumed during grazing as well as crop yield or grazed plant biomass, and nutrient runoff estimates from each HRU for each simulated pasture were then input into a spatial mathematical programming model discussed in the next section. The programming model was then used to select the most efficient pasture management practice and litter transportation pattern for each HRU in the Eucha-Spavinaw watershed that maximized producer income while ensuring that the overall agricultural phosphorus pollution target is reached at least social cost.

The Stochastic Optimization Model for the Watershed

The HRU specific coefficients for production, sediment yield, nitrogen runoff and phosphorus runoff for each pasture management practice obtained from the SWAT simulations were used to develop a spatial mathematical programming model. The linear programming model was used to determine the optimal pasture management practice for each HRU and pattern of litter shipments so that total watershed pollution target is met at least social cost. Based on the works of Tauer (1983), Teague et al. (1995) and Qiu, Prato and Kaylen (1998), this study employed a modified Target MOTAD model to determine the optimal spatial allocation of the alternative pasture management practices and a pattern of litter shipments that maximizes producer income subject to not exceeding maximum allowable total sediment yield, total phosphorus runoff and total nitrogen runoff for the Eucha-Spavinaw watershed. The theoretical Target MOTAD model (Tauer, 1983) may be mathematically expressed as:

$$\max \; E(z) = \sum_{j=1}^{n} C_j X_j \tag{14}$$

subject to

$$\sum_{j=1}^{n} a_{kj} X_j \leq b_k \qquad k = 1,\ldots,m \tag{15}$$

$$T - \sum_{j=1}^{n} C_{rj} X_j + y_r \leq 0 \qquad r = 1,\ldots,s \tag{16}$$

$$\sum_{r=1}^{s} p_r y_r = \lambda \qquad \lambda = M \to 0 \tag{17}$$

$$\text{for all } X_j \text{ and } y_r \geq 0.$$

where $E(z)$ is the expected net return of the farm plan; C_j is expected net return of activity j; X_j is level of activity j; a_{kj} is the amount of resource k used per unit of activity j; b_k is level of resource k available; T is target level of return; C_{rj} is return of activity j for state of nature r; y_r is return deviation below T for state of nature r; p_r is probability that state of nature r will occur; λ is risk aversion measure parameterized from M to 0; n is the number of activities; m is the number of resource equations or constraints; s is the number of states of nature; and M is a large number. This structure of the Target MOTAD model allows the decision maker to choose an optimal farm plan that maximizes expected return subject to the constraint that the probability of income being lower than the target income does not exceed a specified value. Thus the model identifies farm plans which maximize net returns but maintain risk below a critical level or target. Equation (14) maximizes expected return of the solution set. Equation (15) fulfills the technical constraints. Equation (16) measures the revenue of a solution under state r. If that revenue is less than the target T, the difference is transferred to equation (17) via variable y_r. Equation (17) sums the negative deviations after weighing them by their probability of occurring, p_r (Tauer, 1983; Teague et al., 1995; and Qiu et al., 1998).

For purposes of this study, the Target MOTAD specified in equations (14) to (17) was modified into an environmental Target MOTAD, incorporating environmental risks associated with nitrogen and phosphorus losses in runoff from each HRU and allowing for litter transportation activities within and outside the watershed. The objective function (equation 14) was modified and specified to represent all the feasible pasture management practices in each HRU and all possible litter shipment activities between chicken farm centroids and subbasin centroids in the Eucha-Spavinaw watershed. Thus, this study maximizes net returns from grazing less subbasin transportation costs for poultry litter subject to a limit on total nitrogen and phosphorus loading from the entire watershed within a specified tolerance level. The reformulated objective function may be mathematically expressed as:

$$\max_{X_{ij},\,T_{kb}} \; E(z) = \sum_{i=1}^{2416}\sum_{j=1}^{105} R_{ij} X_{ij} - \sum_{k=1}^{24}\sum_{b=1}^{92} T_{kb} C_{kb} \tag{14a}$$

subject to a set of resource and operational constraints specified below in equation (15a) to equation (20); where E(z) is the expected net agricultural income for the watershed; R_{ij} is the net income from the j^{th} management practice in the i^{th} HRU; X_{ij} represents amount of land allocated for the j^{th} management practice in the i^{th} HRU; T_{kb} is the quantity of litter transported from the k^{th} chicken farm centroid to the b^{th} subbasin centroid; C_{kb} is the cost of transporting poultry litter from the k^{th} chicken farm centroid to the b^{th} subbasin centroid. Equation 2 was modified to represent the amount of available land resource (Area) in each HRU that can be allocated for use under any feasible pasture management system as follows:

$$\sum_{j=1}^{105} X_{ij} = Area_i \,, \quad \forall_j \tag{15a}$$

Equations 16 and 17 were modified to incorporate the environmental risks associated with phosphorus and nitrogen losses in runoff in the watershed. The new equations reflect the relationship between total allowable phosphorus loading, $PH_{max,}$ (NIT_{max} , for allowable nitrogen losses) for the entire watershed, the amount of phosphorus runoff (measured as elemental phosphorus), PH_{ij} (NIT_{ij} , for amount of nitrogen runoff) from each HRU (i) under the different pasture management practices (j) and phosphorus runoff deviation, δp_r , (δ_{Nr}, nitrogen runoff deviation) above the maximum allowable load for the watershed under each state of nature (r). The average annual phosphorus and nitrogen runoff levels must not exceed a specified limit or target for the entire watershed while average annual phosphorus and nitrogen runoff deviations above a set target must not exceed a specified tolerance level (λ). The modifications to the equations representing environmental risk may be mathematically expressed as:

$$PH_{\max} - \sum_{i=1}^{2416}\sum_{j=1}^{105}\overline{PH}_{ij} X_{ij} + \delta_{PHr} \geq 0, \qquad \forall i, j, r \tag{16a}$$

$$NIT_{\max} - \sum_{i=1}^{2416}\sum_{j=1}^{105}\overline{NIT}_{ij} X_{ij} + \delta_{NITr} \geq 0, \qquad \forall i, j, r \tag{16b}$$

$$\sum_{i=1}^{2416}\sum_{j=1}^{105} PH_{ijr} X_{ij} - \delta_{PHr} \leq PH_{\max}, \qquad \forall i, j, r \tag{17a}$$

$$\sum_{r=1}^{s} p_r \delta_{PHr} \leq \lambda_{PH}, \qquad \forall r \qquad \lambda_{PH} = M \to 0 \tag{17b}$$

Furthermore, a set of constraints is added to the model to balance demand and supply of poultry litter in the entire watershed. The model allows for shipments of litter from chicken house centroids to subbasin centroids in the watershed. All litter must be shipped from each chicken farm centroid such that the amount of litter shipped to each subbasin must be equal to the quantity of litter applied in each subbasin. These constraints may be mathematically expressed as:

$$\sum_{b=1}^{92} T_{kb} = S_k, \quad \forall k \tag{18}$$

$$\sum_{b=1}^{92}\sum_{j=1}^{105} Q_{jb} X_{jb} = \sum_{k=1}^{24} T_{kb}, \quad \forall b, j, k \tag{19}$$

where S_k is the quantity of litter supplied at the k^{th} chicken farm centroid; Q_{jb} is the quantity of litter required by the j^{th} management practice in the b^{th} subbasin; and X_{jb} is the amount of land allocated to the j^{th} management practice in the b^{th} subbasin. The non-negativity constraints for production and transportation activities are:

$$X_{ij} \geq 0,\ T_{kb} \geq 0 \tag{20}$$

Figure 8 below presents a schematic diagram of the integrated simulation-optimization model used in this study. The diagram shows the SWAT model input data as well as the outputs generated by the SWAT simulation model. The SWAT model outputs for specific management practices in a given HRU were used in the input-output coefficient matrix of the Target MOTAD risk programming model together with other necessary production and economic input data. Figure 8 also outlines the expected output from the Target MOTAD optimization model as applied for purposes of achieving the objectives of this study.

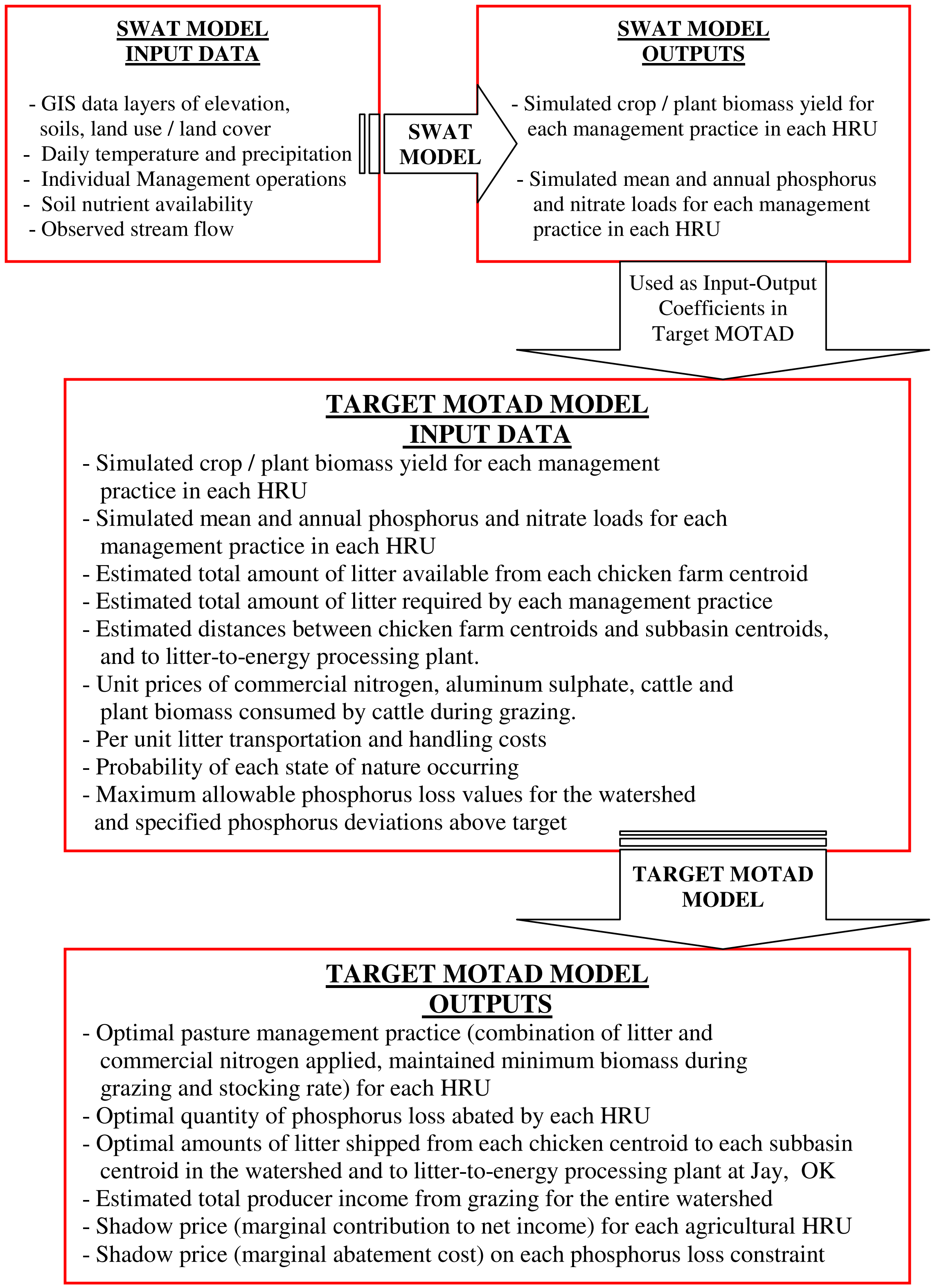


Figure 8 Schematic Diagram of the Integrated Simulation-Optimization Model

Determination of Phosphorus Pollution Abatement Costs

This study applied the environmental Target MOTAD risk programming model described above to select the best litter and pasture management practice for each HRU in the Eucha-Spavinaw watershed so as to reduce aggregate phosphorus emissions from pastures from the current undesirable level (estimated at about 40 tons per year) to alternative lower phosphorus runoff levels (35, 30, 25, and 20 tons per year) at least social cost. These socially optimum points are assumed to occur at points where the marginal abatement cost equals the marginal cost of damage to the environment and the sum of treatment and damage costs are minimized. However, various policy instruments and pollution reduction programs not only cause different allocative effects but also impose different financial burdens for polluters, victims, and society. We should note that there is no such thing as “the cost” of abatement or of environmental damage; both values depend on the level of pollution emissions or pollution emissions reduction, respectively. Thus, the (implicit) price of pollution and pollution prevention vary-often considerably-with the intensity of pollution (Sterner, 2003).

Total Phosphorus Pollution Abatement Costs

In the case of water pollution from phosphorus emissions as is the case in the Eucha-Spavinaw watershed, the treatment or abatement cost function represents all the costs incurred in the process of removing and / or preventing the pollutant (phosphorus) from entering the water course (Lakes Eucha and Spavinaw). However, for purposes of this study, we determined total abatement costs in terms of reduction in producer income from crops, pasture and range. Total abatement costs were estimated as the difference in the value of the objective function (representing total agricultural net returns for the watershed) of the Target MOTAD programming model (specified above) subject to the estimated current level of phosphorus loading for the Eucha-Spavinaw watershed (40 tons per year) and the value of the objective function at each of the alternative annual phosphorus loading targets (that is, at 35, 30, 25, and 20 tons per year) and a specified phosphorus deviation limit above a given phosphorus loading target. The upper limit on the phosphorus runoff deviation above annual phosphorus loading was varied from 10 tons to 2 tons per year. For instance, if you subtract the value of the objective function when the maximum allowable phosphorus loading is reduced to 35 tons per year from the value of the objective function under the current estimated phosphorus loading of 40 tons per year, you obtain the total cost of treating/abating 5 tons of phosphorus loading per year in the Eucha-Spavinaw watershed for a given phosphorus pollution reduction strategy. The total abatement cost curve could be traced out using a mathematical function that maps the alternative annual phosphorus runoff targets to corresponding reductions in the value of the objective function (representing reductions in total agricultural net returns due to a given pollution reduction strategy or total pollution abatement costs). Typically, the total abatement cost function increases with increased abatement (or decreases with increased pollution emissions).

Marginal Phosphorus Pollution Abatement Costs

The marginal phosphorus treatment/abatement cost may be defined as the change in total phosphorus pollution abatement costs from an additional unit of phosphorus treated/abated. Optimal pollution abatement requires that the marginal abatement costs in production be set equal to the marginal benefit of the abatements as measured by a reduction in environmental damage (Tietenberg, 2003; Sterner, 2003). For purposes of this study, we determined the marginal phosphorus pollution abatement cost using the shadow price on the binding average annual phosphorus runoff constraint obtained from the solution of the economic model specified above. This shadow price may be interpreted in economic terms to represent the amount by which the value of the objective function (or the total agricultural net return for the Eucha-Spavinaw watershed) is reduced as the maximum allowable annual phosphorus runoff is restricted by an additional unit per year. The marginal abatement cost curve could be traced using a mathematical function that maps the alternative annual phosphorus runoff targets to corresponding shadow prices or marginal abatement costs. Typically, the marginal abatement cost function increases with increased abatement (or decreases with increased pollution emissions) whereas the marginal damage cost function increases with pollution emissions. The intersection of the curves for the marginal costs of pollution damage and the marginal costs of pollution abatement determines the optimal levels of pollution emissions and their shadow cost (Steiner, 2003; Tietenberg, 2003).

Generalized Linear Econometric Model Specification

A generalized linear econometric model was specified to summarize the simulation results of the SWAT in terms of the controlled management and weather variables. The regression model was specifically used to determine the relationship between phosphorus runoff in the current period and soil type, RKLS-factor, curve number (CurV), minimum biomass maintained during grazing (BmMin), stocking rate (StkRate), amount of litter/phosphorus applied (Papl), amount of commercial nitrogen applied (Napl) and the amount of phosphorus runoff in the previous period (LagPloss). The general econometric model may be mathematically specified as:

$$P_{it} = \sum_{k=1}^{K} X_{itk} \beta_k + u_{it} \qquad i = 1,\ldots,N; \quad t = 1,\ldots,T \tag{21}$$

$$u_{it} = v_i + e_t + \varepsilon_{it} \tag{22}$$

Where P_{it} represent expected phosphorus runoff in the current period, X_{it} represent the independent variables outlined above, β_k are parameters to be estimated, v_i is a cross-section specific residual, e_t is a time-series specific residual, ε_{it} is a classical error term with zero mean and a homoskedastic covariance matrix, N is the number of cross-sections, T is the length of the time series for each cross section, and K is the number of explanatory variables included in the model.

Data and Sources

Large amounts of spatial and non-spatial data are required in the two-step GIS-based modeling approach used in this study. The study required data on topography, land use or land cover, soil types, weather, management systems and stream flow. The most current GIS data for topography, soils, land cover, and stream flows were used in the SWAT model. These data were obtained from various sources including public agencies, County extension offices, and via personal communications. The inputs to the SWAT model were accumulated from hydrographic and geographic databases and maps.

Topographic data are necessary for delineation of the watershed and its subbasins. 30-meter digital elevation model (DEM) (1:24,000) for the Eucha-Spavinaw watershed were used to define topography in this study. These were obtained from the United States Geological Survey (USGS) topographical database. A DEM is a digital record of terrain elevations for ground positions at regularly spaced horizontal intervals that is derived from USGS maps. The DEM was used to define layout and number of subbasins, the stream network and its characteristics and derive subbasin parameters such as slope, slope length, and aspect of catchments. Land cover can change spatially and temporally over a short period of time. It is important that these data be based on the most current data available. Land cover data used in the model was derived from the Arkansas and Oklahoma Gap Analysis Program (GAP) data available online at http://www.gap.uidaho.edu/About/Overview/GapDescription/default.htm#Products . The GAP data was simplified into land cover categories suitable for this study namely; Generic Agricultural Land (AGRL), low-biomass pasture (LPAS), medium-biomass pasture (MPAS), high-biomass pasture (HPAS), litter low-biomass pasture (LLPA), litter medium-biomass pasture (LMPA), litter high-biomass pasture (LHPA), rangeland (RNGB), forests (FRST), urban (URBN) and water (WATR).

SWAT requires soil GIS data to define soil types. The Natural Resource Conservation Service (NRCS) developed a GIS coverage for soils nationwide. STATSGO are the default soil data used with SWAT. The model uses STATSGO data to define soil attributes for all soils. This study used the soils layer representing the Oklahoma and Arkansas portion of the watershed derived from the STASGO data. The GIS data used in this study contained S5ID (Soils5id number for USDA soil series) that linked an area to the STASGO database. When no observed weather data are available, SWAT can stochastically simulate the weather data using a database of weather statistics from stations across the United States (Storm et al., 2001). The National Oceanic and Atmospheric Administration (NOAA) maintains records from numerous National Weather Service Cooperative Observing Network (COOP) station data. Weather data used in this study were obtained from websites of the Oklahoma Mesonet and the NOAA. Because of the spatial and temporal variability of precipitation, multiple climate stations with at least 20-year period of records were selected. This study used observed daily precipitation data from 50 stations located within, or near Eucha-Spavinaw watershed for the period 1/1/1950-12/31/2004. Daily maximum and minimum temperatures were obtained from 7 weather stations. Each subbasin was assigned the nearest climate station. However, all of the stations used had time periods where data were missing. SWAT generates simulated weather when missing data are detected at a station. However, missing data in the Eucha-Spavinaw model were filled with corresponding data from the nearest station or estimated.

Stream flow data were obtained from USGS stream gauge stations. The measured stream flow time series was split into calibration and validation periods. During the calibration period, model inputs were allowed to vary across the basin until acceptable fit to measured flow at the basin outlet was obtained. The model was calibrated for surface runoff, base flow and for phosphorus loads. The model was then run using the same input parameters for the validation period and goodness-to-fit was determined (Arnold et al.

2000). Observed water quality data collected by the City of Tulsa and stream flow records from the U.S Geologic Survey (USGS) were used to estimate phosphorus loads in the Lake Eucha and Lake Spavinaw basin. The net income from agricultural activities was estimated by using data from the SWAT model (yield and biomass data), cost estimates from Oklahoma State University Enterprise Budgets and various USDA published and unpublished sources.

CHAPTER V

RESULTS

A total of 105 grazing management practices were simulated and tested in each of the agricultural HRUs in the Eucha-Spavinaw watershed. Three 20-year simulations were made for each management practice. The weather years were randomly selected with replacement from 55 years of annual weather data from 1950 to 2005. We examined the effects of limiting total phosphorus runoff for the watershed to 40, 35, 30, 25, and 20 tons per year on optimal litter and pasture management systems under two scenarios. The first scenario assumed three possibilities: that all or part of the poultry litter produced within a given subbasin can be applied on land as fertilizer, or all or part of the poultry litter produced within a given subbasin can be sold to producers in another subbasin for use as crop fertilizer, or all or part of the poultry litter produced within a given subbasin can be sold to a possible litter-to-energy power plant located at Jay, Oklahoma. The second scenario allowed for the possibility of using Alum-treated poultry litter on pastures in addition to the three possibilities stipulated in the first scenario. The environmental target MOTAD risk programming model was solved for each of the possible mean annual phosphorus runoff targets (40Mg, 35Mg, 30Mg, 25Mg, and 20Mg) and phosphorus runoff deviation limits (10Mg, 8Mg, 6Mg, 4Mg, and 2Mg) above target for the Eucha-Spavinaw watershed to maximize producer income while meeting the mean phosphorus load limits within a specified margin of safety at minimum cost to society. The optimization model assigned a specific management practice to each of the 2,416 HRUs in the watershed given conditions stipulated in the two scenarios outlined above.

Empirical Estimation of the Generalized Linear Econometric Model

The ANOVA regression model was used as a convenient method of summarizing the SWAT simulation results. Table 13 shows the analysis of variance (ANOVA) results and Table 14 below presents parameter estimates of the generalized linear econometric model fitted using the SAS Generalized Linear Model (GLM) Procedure. Current phosphorus runoff was regressed on soil type, KLSCP- factor, curve number (CurV), phosphorus applied in the current period (Papl), phosphorus runoff in the previous period(LagPloss), nitrogen applied in the current period (Napl), the minimum biomass maintained during grazing (BmMin) and the stocking rate (StkRate). The overall model had a significant F-Value and explained 96 percent of the variation in simulated phosphorus loss in the watershed. The Type III sum of squares were statistically significant, implying that individual independent variables contribute significantly to phosphorus runoff in the watershed. Specifically the ANOVA shows the soil types explained a significant portion of the variation in phosphorus loss from each HRU in the presence of the KLSCP- factor, curve number, and management variables. The soil type parameter estimates shown in Table 14 have been sorted in descending order to show the relative contribution of each soil type, KLSCP-factor, curve number and management variables to phosphorus runoff in the Eucha-Spavinaw watershed. The SWAT estimates the amount of soil erosion using the modified universal soil loss equation (MUSLE) (Williams, 1975), an adaptation of the Universal Soil Loss Equation (USLE). The KLSCP-factor is the value calculated in the

USLE (Wischmeiner and Smith, 1978). This factor incorporates a soil erodibility factor (K), topographic or soil slope length gradient factor (LS), cover and management factor (C), and a support practice factor (P) (Williams, 1975; Arnold, 1992).

Table 13 Analysis of Variance (ANOVA).

The GLM Procedure :Dependent Variable: Ploss

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	30	88933889.72	2964462.99	128444	<.0001
Error	143586	3313925.44	23.08		
Corrected Total	143616	92247815.17			

R-Square	Coeff Var	Root MSE	Ploss Mean
0.964076	47.86849	4.804136	1.003611

Source	DF	Type III SS	Mean Square	F Value	Pr > F
Soil	23	2013399.115	87539.092	3792.90	<.0001
KLSCP	1	1846277.007	1846277.007	79995.6	<.0001
CurV	1	93480.874	93480.874	4050.35	<.0001
BmMin	1	14547.613	14547.613	630.32	<.0001
StkRate	1	4333374.674	4333374.674	187757	<.0001
Papl	1	8487.762	8487.762	367.76	<.0001
Napl	1	4995.617	4995.617	216.45	<.0001
LagPloss	1	24242.337	24242.337	1050.37	<.0001

Table 14 Estimated Regression Coefficients of the Linear Econometric Model.

Effect	Estimate	Standard Error	t Value	Pr>|t|
Intercept	-47.8379	5.0665	-9.44	<.0001
Britwater*	3.0772	1.3424	2.29	0.0219
Razort*	2.3981	1.1362	2.11	0.0348
Clarksville*	2.0253	0.9638	2.10	0.0356
Captina*	1.6864	0.4134	4.08	<.0001
Secesh*	1.4292	0.7230	1.98	0.0481
Healing*	1.4078	0.4822	2.92	0.0035
Cherokee	1.3653	0.7232	1.89	0.0590
Noark	1.3210	0.6886	1.92	0.0551
Nixa	1.0697	0.6202	1.72	0.0846
Macedonia*	0.9691	0.3104	3.12	0.0018
Peridge*	0.8471	0.2761	3.07	0.0022
Tonti	0.7468	0.4137	1.81	0.0711
Stigler*	0.6987	0.1757	3.98	<.0001
Doniphan	0.2024	0.1395	1.45	0.1467
Jay	0.1896	0.2092	0.91	0.3648
Eldorado*	0.1441	0.0202	7.15	<.0001
Taloka	0.1133	0.1100	1.03	0.3033
Elsah	0.0615	0.1052	0.58	0.5590
Hector	-0.1190	0.4494	-0.26	0.7912
Newtonia	-0.2618	0.1368	-1.89	0.0593
Linker*	-0.6358	0.2075	-3.06	0.0022
Carytown*	-1.4204	0.3809	-3.73	0.0020
Mountainburg*	-2.0916	0.7242	-2.89	0.0039
Waben	0.0000	.	.	.
KLSCP*	32.6432	0.1154	282.83	<.0001
StkRate*	14.3768	0.0332	433.31	<.0001
LagPloss*	0.2650	0.0082	32.41	<.0001
CurV*	0.2492	0.0039	63.64	<.0001
Papl*	0.0108	0.0006	19.18	<.0001
BmMin*	-0.0020	0.0001	-25.11	<.0001
Napl*	-0.0036	0.0002	-14.71	<.0001

* regression coefficient is statistically significant at the 5% significance level.

All the "effects" in Table 14 marked with an asterisk (*) have regression coefficients that are significantly different from zero at the 5% significance level. This means a change in any of these variables will have a statistically significantly higher or lower phosphorus loss from pastures than the Waben soil. Only 13 of the 24 soil types (marked with an asterisk) had a significant effect on phosphorus runoff in the watershed. Six of the 13 soils had significantly higher phosphorus loss (Britwater, Razort, Clarksville, Captina, Secesh and Healing) than the Waben soil. An additional hectare of pasture on any of these soils was predicted to increase phosphorus loss by at least 1.5 kg per hectare over the Waben soil. For instance, the model predicted that putting one more hectare of Britwater under pasture will increase phosphorus loss by 3 kg per hectare while an additional hectare of pasture on Razort, Clarksville and Captina soils will increase phosphorus runoff by 2.4, 2.0, and 1.7 kg per hectare over the Waben soil, respectively. One more hectare of pasture on either Secesh or Healing will increase phosphorus runoff by 1.4 kg per hectare over the Waben soil. Eleven of the 24 soils used in this study did not have significant different phosphorus loss values than the Waben soil when the curve number, KLSCP-factor, and other management variables were accounted for. This implies that it may be necessary to consider soil type and develop phosphorus reduction programs that target specific soil types within the Eucha-Spavinaw watershed rather than continue with the current uniform policy of limiting litter application rates strictly by soil test phosphorus. Besides soil type, all the other explanatory variables in the model were found to affect phosphorus runoff significantly. As expected, the results show a significant positive relationship between phosphorus loss (Ploss) and the KLSCP-factor, curve number (CurV), stocking rate (StkRate), the amount of phosphorus loss in the previous period (LagPloss) and the amount of phosphorus applied (Papl) in the current period. The results also suggest a significant inverse relationship between phosphorus loss (Ploss) and the minimum biomass maintained during grazing (BmMin) and amount of nitrogen applied on the pastures (Napl). For example, the model predicted that if the stocking rate increases by 1AU / ha, phosphorus runoff will increase by about 14 kg per hectare. An increase of carryover phosphorus from the previous year by one kilogram per hectare would increase phosphorus loss in the current period by 0.27 kg per hectare while a one unit increase in the KLSCP factor would increase phosphorus loss by about 33 kg per hectare.

The Efficient Allocation of Phosphorus Pollution

Phosphorus is one of the nutrients for which the environment has some absorptive capacity. That is, the environment has some ability to absorb phosphorus. However, phosphorus does accumulate in the environment when the emissions load exceeds its absorptive capacity. The presence of excessive amounts of phosphorus in the environment leads to environmental degradation, a damage cost incurred by the society. The damage costs are expected to rise with the cumulative phosphorus emissions load in the environment. Therefore, phosphorus pollution control measures need to be put in place to counter this negative externality. This intervention would impose pollution control or abatement costs on the polluter and thus reduce phosphorus emissions load in the environment. The treatment or abatement costs incurred by the polluter are expected to rise with the cumulative phosphorus emissions treated or controlled. An efficient policy response must therefore not only determine the appropriate level of phosphorus pollution that balance control and damage costs. It must also specify how the responsibility for achieving that phosphorus level should be allocated among the various sources of phosphorus emissions when reductions are needed in the watershed. The mathematical programming model used in this study addressed the two issues based on the general framework illustrated in Figure 9 to determine the efficient amount of phosphorus pollution for the Eucha-Spavinaw watershed. In Figure 9, MC represents marginal costs, MAC represents marginal abatement cost, MDC represents marginal damage costs, TAC represents total abatement costs, TDC represents total damage costs; Z is the total amount of phosphorus pollution emissions. MC* represents the marginal cost at which MAC = MDC and T* represents the optimal phosphorus pollution tax rate.

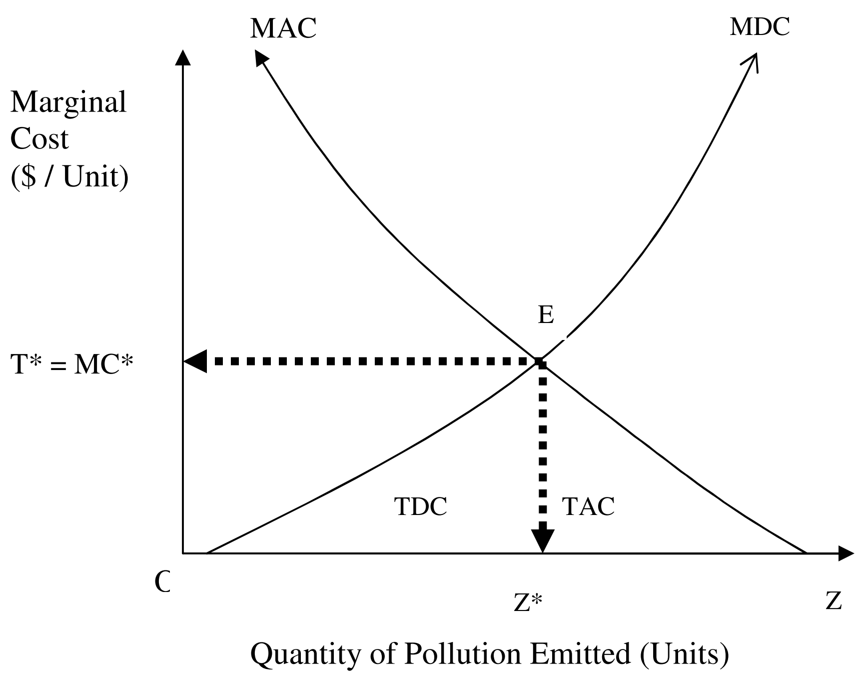


Figure 9 Optimal Level of Phosphorus Pollution (Z*).

A movement along the X-axis of Figure 9 from left to right refers to less control and more pollution emitted. The marginal damage cost (MDC) resulting from an additional unit of pollution is shown to increase with the quantity of pollution emitted. A movement in the opposite direction implies greater control and less pollution emitted into the environment. The marginal abatement cost (MAC) resulting from an additional unit of pollution is shown to increase with the quantity of pollution abated. Under these conditions, Figure 9 demonstrates that the optimal level of phosphorus pollution is not zero. An efficient allocation of phosphorus pollution requires that marginal benefit be equated to marginal cost. This efficiency condition holds at the quantity of pollution where the damage cost resulting from an additional unit of phosphorus pollution is equal to the marginal cost of abating that extra unit of pollution. Thus, point Z* in Figure 9 represents the optimal level of phosphorus pollution in the watershed. The area of triangle OEZ* represents the total damage costs (TDC) while the area of triangle ZEZ* represents the total abatement costs (TAC).

Cost-Effective Allocation of Phosphorus Pollution in the Watershed

We assumed five alternative levels of maximum allowable mean annual phosphorus loads for the Eucha-Spavinaw watershed in this study. The base mean annual phosphorus load was assumed to be 40 Mg per year. The mean phosphorus load was then gradually reduced by 5 Mg per year until it reached 20 Mg per year. This study aimed at achieving a specific phosphorus emission reduction for the entire watershed. At each mean annual phosphorus load, the "pollution authority" had to decide how to allocate the responsibility for meeting these predetermined pollution levels among all the polluters in the watershed at minimum cost to society. Polluters had different options for controlling the amount of phosphorus pollution they emitted into the environment. A total of 105 possible pasture management practices were available for reducing phosphorus emissions in each of the agricultural HRUs in the watershed. Some of these pasture management practices would be relatively cheaper to implement than others. Given these alternative pasture management practices, different soil types and hydrology characteristics of the 2,416 HRUs in the watershed, the cheapest method of phosphorus control will differ widely not only among subbasins but also among HRUs in the same subbasin.

An environmental Target MOTAD risk programming model was used as a systematic method for finding the lowest cost means of maximizing producer income from grazing while meeting the maximum allowable mean annual phosphorus load for the watershed within a specified margin of safety. The optimization model applies the cost-effectiveness equimarginal principle (illustrated in Figure 10) to select least-cost management practices for each HRU such that total cost of pollution reduction for the watershed is minimized. Figure 9 assumes that there are only two polluters (Hru_1, Hru_2) in the watershed emitting a total of Z units of phosphorus into the environment. The marginal control costs for Hru1 (shown as MAC_1) are relatively lower than those for Hru2 (shown as MAC_2). The cost-effective allocation of phosphorus pollution control among the two HRUs requires that their marginal abatement costs be equal (that is, $MAC_1 = MAC_2$). This condition holds at point Q* in Figure 10, implying that Hru1 will abate phosphorus emissions amounting to ZQ* while Hru2 abates the remaining OQ*.

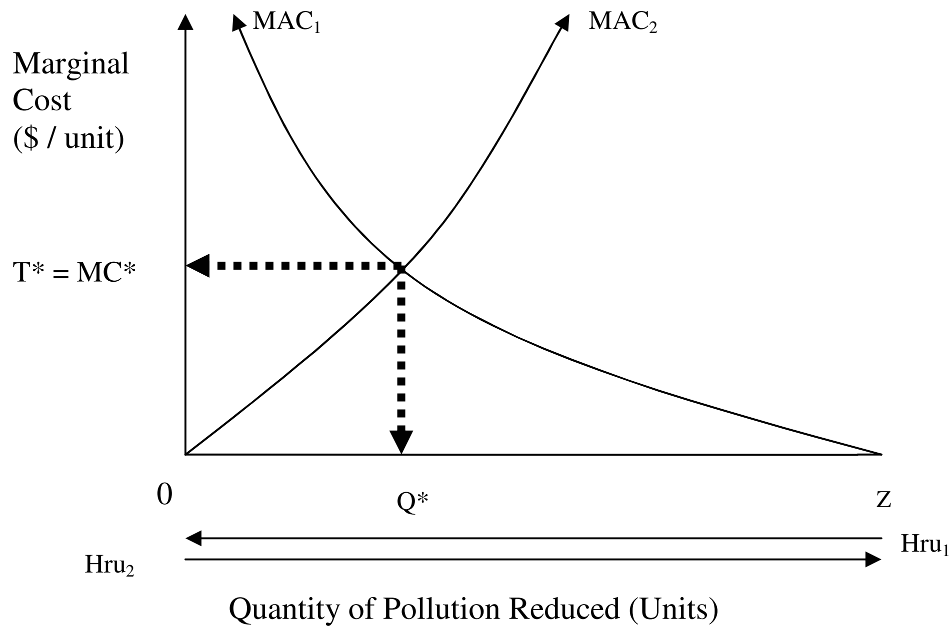


Figure 10 Cost-Effective Allocation of Phosphorus Pollution Among Two HRUs.

It is worth noting that the total costs of pollution reduction were minimized by allocating more phosphorus control to Hru$_2$ with lower marginal abatement costs and less to Hru$_1$ with higher marginal treatment costs. The optimization model used a similar technique to allocate phosphorus control across the 2,416 agricultural HRUs in the Eucha-Spavinaw watershed to minimize total cost of pollution control. The cost of achieving a given reduction in phosphorus emissions was minimized by selecting management practices that equalized the marginal costs of control across all HRUs in the watershed.

One of the constraints imposed on the optimization model specified in equation (14a) was to select a specific pasture management practice for each of the HRUs in the watershed. The value of the Lagrange multiplier (or the shadow price) on that constraint for a given HRU represented the additional monetary contribution from that agricultural land that would be added to the objective function were an additional identical HRU created in the watershed. Given this environmental-economic optimization model, the more profitable agricultural production was on a particular HRU, the higher the associated shadow price. Similarly, the higher the phosphorus loss from a particular HRU, the lower the corresponding shadow price. These two situations imply that HRUs may contribute differently toward net agricultural income and total phosphorus loading for the watershed. Therefore, in this optimization model, preference for selection into the optimal solution set is given to HRUs with both the highest contribution to net agricultural income and least contribution to total phosphorus loading in the watershed. Such HRUs will have high shadow prices per hectare. On the other hand, HRUs that contribute marginally to net agricultural income and heavily to total phosphorus loss in the watershed will be characterized by very low per hectare shadow prices. These shadow prices may even be negative for some HRUs if their contribution to total phosphorus loss outweighed their contribution to net agricultural income. All HRUs in the watershed that exhibit negative and very low per hectare shadow prices could be targeted for inclusion in a phosphorus pollution reduction program for the Eucha-Spavinaw watershed. The following sections present the effects of limiting total phosphorus runoff for the watershed to 40, 35, 30, 25, and 20 tons per year on optimal litter and pasture management systems under the first scenario (Scenario I).

Scenario I: Land Application and Trading of Untreated Poultry Litter

In this option we examined the effects of limiting total phosphorus runoff for the Eucha-Spavinaw watershed to 40, 35, 30, 25, and 20 tons per year on optimal litter and pasture management systems when the available method of litter allocation is hauling within the watershed and to a possible litter-to-energy power plant located at Jay, Oklahoma. The poultry litter was untreated (no Alum was added to the poultry litter).

Optimal Grazing Management Practices for the Watershed

The model selected the best grazing management practice for each HRU in the watershed that maximized total net agricultural income from grazing less transportation costs while total watershed phosphorus pollution target was met at least cost within a specified deviation over the target. Table 15 below shows a wide range of grazing management practices identified for optimal level of phosphorus abatement in the Eucha-Spavinaw watershed. No single grazing management practice dominated in all the HRUs of the watershed. Instead, optimal phosphorus abatement for the watershed was achieved through a combination of various site-specific grazing management practices at each mean annual phosphorus loading target and phosphorus runoff deviation limit tested in this study.

Table 16 to Table 20 below show the optimal grazing management practices selected by the economic optimization model and amount of land allocated for each selected management practice when the mean annual total phosphorus runoff for the Eucha-Spavinaw watershed was limited to 40 Mg, 35Mg, 30Mg, 25Mg and 20Mg per year, respectively, with phosphorus deviation limits above target varied from 10 Mg to 2 Mg per year.

Table 15 Data Coding for Alternative Grazing Management Practices

BMP Code	Poultry Litter Applied (tons/ha)	Elemental Nitrogen Applied (kg/ha)	Minimum Biomass Maintained During Grazing (tons/ha)	Stocking Rate (AU/ha)
0011	0	0	1.1	0.63
0012	0	0	1.1	1.00
0013	0	0	1.1	1.26
0221	0	100	1.6	0.63
0222	0	100	1.6	1.00
0223	0	100	1.6	1.26
0331	0	150	2.0	0.63
0332	0	150	2.0	1.00
0333	0	150	2.0	1.26
0111	0	50	1.1	0.63
0112	0	50	1.1	1.00
0113	0	50	1.1	1.26
1011	2	0	1.1	0.63
1012	2	0	1.1	1.00
1013	2	0	1.1	1.26
1231	2	100	2.0	0.63
1232	2	100	2.0	1.00
1233	2	100	2.0	1.26
1121	2	50	1.6	0.63
1122	2	50	1.6	1.00
1123	2	50	1.6	1.26
2021	4	0	1.6	0.63
2022	4	0	1.6	1.00
2023	4	0	1.6	1.26
3031	6	0	2.0	0.63
3032	6	0	2.0	1.00
3033	6	0	2.0	1.26
3231	6	100	2.0	0.63
3431	6	200	2.0	0.63
3432	6	200	2.0	1.00
3433	6	200	2.0	1.26

Table 16 Comparison of Optimal Management Practices (Ha) When Maximum Average Phosphorus Target is 40 Mg / year as Average P Loss Deviations Above the Mean Phosphorus Load were Reduced from 10 Mg to 2 Mg per year.

Optimal Management Practice					Deviation Above Maximum 40Mg Phosphorus Loss (Mg)				
Code	PL	Nit.	MB	SR	10	8	6	4	2
	Mg	kg			Hectares Where Management Practice was Optimal				
0011	0	0	L	L	1753	1923	963	694	126
0012	0	0	L	M	6608	7397	6947	6335	6896
0013	0	0	L	H	3933	3793	3005	2470	1086
0221	0	100	M	L	1	2	2	164	337
0222	0	100	M	M	0	0	1	645	2432
0223	0	100	M	H	1	1	1	3	129
0331	0	150	H	L	1	3	0	56	883
0332	0	150	H	M	1	0	0	112	33
0333	0	150	H	H	0	0	1	127	327
0111	0	50	L	L	1	2	55	23	512
0112	0	50	L	M	1401	1100	4217	5777	4375
0113	0	50	L	H	1	1	2	2	1
1011	2	0	L	L	4	32	124	15	23
1012	2	0	L	M	4541	4294	2853	1186	115
1013	2	0	L	H	2	2	1	0	4
1231	2	100	H	L	2	1	0	4	3
1232	2	100	H	M	0	1	0	0	0
1233	2	100	H	H	1	1	0	0	0
1121	2	50	M	L	1	1	2	1	0
1122	2	50	M	M	2	222	362	66	0
1123	2	50	M	H	0	0	0	1	1
2021	4	0	M	L	1759	1939	2288	3821	5778
2022	4	0	M	M	15988	13962	12676	10694	6373
2023	4	0	M	H	0	1	0	47	1707
3031	6	0	H	L	1	1065	1597	2424	2652
3032	6	0	H	M	1	1	4	3	2
3033	6	0	H	H	1	1	1	1	2
3231	6	100	H	L	171	309	938	1353	1855
3431	6	200	H	L	1	0	0	4	118
3432	6	200	H	M	0	1	1	49	133
3433	6	200	H	H	3	6	6	44	646
Ave. P Loss (Mg/yr)					40	40	38	35	31
Ave. P Deviation(Mg/yr)					9.4	8.0	6.0	4.0	2.0

** Abbreviations used: PL=Poultry Litter Applied (Mg/ha); Nit. = Commercial Nitrogen Applied (kg/ha); MB= Minimum Biomass (L=1.1,M=1.6,H=2.0 Mg/ha); SR = Stocking Rate (L=.63, M=1.0, H=1.26 Animal units/ha). See Table 13 for the best management practice (BMP) associated with this BMP code.*

Table 16 above shows that when mean annual phosphorus load for the Eucha-Spavinaw watershed is limited to 40 Mg per year with an upper limit on phosphorus deviation above mean load of not more than 10 Mg per year, BMP 2022 received the largest land allocation of about 16,000 hectares of pastureland. Under this grazing management practice, the pasture received 4 tons of poultry litter per hectare and no commercial nitrogen fertilizer was applied at all. However, producers maintained minimum biomass of 1,600 kilograms per hectare during grazing at a stocking rate of 1.00 AU per hectare. When the mean phosphorus loss was restricted to 40 Mg, the average deviations above this mean averaged 9.4 Mg per year. As the upper limit on phosphorus deviations above the maximum mean load was reduced from 10 Mg to 2 Mg per year, more land was transferred from BMP 2022, BMP 1012, and BMP 0013 and put under BMP 2021, BMP 0112, BMP 0222 and BMP 0012. These changes represented combinations of less litter, more commercial nitrogen, more biomass retained after grazing, and reduced stocking rates. The amount of pastureland that received no poultry litter at all increased from about 14,000 to 17,000 ha whereas the amount of land that received 4 tons of poultry litter per hectare declined from about 18,000 to 14,000 ha. The amount of land that received from 50-150 kg/ha of commercial nitrogen fertilizer increased from approximately 1,400 to 11,000 hectares. However, the amount of pastureland on which a minimum biomass of 1,100 kg/ha was maintained during grazing declined from 18,000 to 13,000 hectares whereas the land on which a minimum biomass of 1,600 kg/ha and above was maintained during grazing increased from about 18,000 to 27,000 hectares. The amount of land that was stocked at a rate of 1.00 AU/ha and above declined from approximately 33,000 to 24,000 hectares while that which was stocked at a lower rate of 0.63 AU/ha increased from 4,000 to 12,000 hectares. Actual mean phosphorus loss declined from 40 Mg to 31 Mg per year. Thus the model shows the most cost effective method to reduce average positive deviations to 2 Mg over 40 Mg is to reduce average annual phosphorus loss to

31 Mg per year.

Table 17 below shows the optimal grazing management practices selected by the economic optimization model and amount of land allocated for each selected management practice when the mean annual total phosphorus runoff for the Eucha-Spavinaw watershed was reduced from 40 Mg to 35 Mg per year, with phosphorus deviation limits above target were systematically reduced from not more than 10 Mg to 2 Mg per year. The results indicate the least costly changes in management practices to meet the limits in each case. For example the area allocated for BMP 2022 (4MgL,0N,mMB,mSR) declined from 16,000 to 12,000 hectares and remained the largest share of total area under pasture until the phosphorus deviation limit above target was reduced from not more than 10 Mg to 4 Mg per year. However, both the reduction of the mean annual phosphorus runoff target and phosphorus runoff deviation limits further transferred more land from BMP 2022, BMP 1012, and BMP 0013 and put it under BMP 2021, BMP 0112, BMP 0222 and BMP 0012. The later management activities represent less poultry litter, more commercial N, and less intensive grazing than the former. As shown in Table 17, the amount of pastureland that received no poultry litter at all increased further from about 14,000 to 18,000 ha whereas the amount of land that received 4 tons of poultry litter per hectare declined from about 18000 to 11000 ha.

Table 17 Comparison of Optimal Management Practices (Ha) When Maximum Average Phosphorus Target is 35 Mg / year as Average P Loss Deviations Above the Mean Phosphorus Load were Reduced from 10 Mg to 2 Mg per year

Optimal Management Practice					Deviation Above Maximum 35Mg Phosphorus Loss (Mg)				
Code	PL	Nit.	MB	SR	10	8	6	4	2
	Mg	kg			Hectares Where Management Practice was Optimal				
0011	0	0	L	L	2057	2104	1203	1351	915
0012	0	0	L	M	7593	7901	7773	7528	5683
0013	0	0	L	H	2708	2715	2612	1180	731
0221	0	100	M	L	13	12	51	252	35
0222	0	100	M	M	152	132	456	1114	4046
0223	0	100	M	H	20	21	2	5	344
0331	0	150	H	L	2	0	56	342	954
0332	0	150	H	M	1	0	11	79	113
0333	0	150	H	H	0	0	3	357	2083
0111	0	50	L	L	4	4	128	16	483
0112	0	50	L	M	2669	2672	3861	4796	2665
0113	0	50	L	H	0	0	4	2	1
1011	2	0	L	L	3	4	10	11	6
1012	2	0	L	M	1159	1149	1337	447	9
1013	2	0	L	H	2	4	2	1	1
1231	2	100	H	L	0	3	1	1	1
1232	2	100	H	M	0	0	1	0	0
1233	2	100	H	H	0	0	0	0	0
1121	2	50	M	L	1	0	0	0	0
1122	2	50	M	M	5	7	40	6	4
1123	2	50	M	H	1	1	1	1	1
2021	4	0	M	L	5846	5892	4222	5034	7637
2022	4	0	M	M	12256	12308	11189	8222	2906
2023	4	0	M	H	99	101	60	572	883
3031	6	0	H	L	1115	1049	2236	3311	4367
3032	6	0	H	M	3	4	2	2	1
3033	6	0	H	H	1	2	0	1	0
3231	6	100	H	L	224	245	973	1583	1892
3431	6	200	H	L	1	0	0	12	122
3432	6	200	H	M	0	1	1	79	218
3433	6	200	H	H	2	3	6	65	571
Ave. P Loss (Mg/yr)					35	35	35	32	26
Ave. P Deviation(Mg/yr)					8.4	8.0	6.0	4.0	2.0

** Abbreviations used: PL=Poultry Litter Applied (Mg/ha); Nit. = Commercial Nitrogen Applied (kg/ha); MB= Minimum Biomass (L=1.1,M=1.6,H=2.0 Mg/ha); SR = Stocking Rate (L=.63, M=1.0, H=1.26 Animal units/ha). See Table 13 for the best management practice (BMP) associated with this BMP code.*

The amount of land that received from 100-150 kg/ha of commercial nitrogen fertilizer increased from approximately 5 to 9,000 hectares. However, the amount of pastureland on which a minimum biomass of 1,100 kg/ha was maintained during grazing declined from 18,000 to 10,000 hectares whereas the land on which a minimum biomass of 2,000 kg/ha was maintained during grazing increased drastically from about 181 to 10,000 hectares. The amount of land that was stocked at a rate of 1.00 AU/ha and above further declined from approximately 33,000 to 20,000 ha while that which was stocked at a lower rate of 0.63 AU/ha further increased from about 4,000 to 16,000 ha.

Table 18 below shows the optimal grazing management practices selected by the economic optimization model and amount of land allocated for each selected management practice when the mean annual total phosphorus runoff for the Eucha-Spavinaw watershed was reduced from 40 Mg to 30 Mg per year, with phosphorus deviation limits above target varied from not more than 10 Mg to 2 Mg per year.
The area allocated for BMP 2021 drastically increased from 1800 to 9000 hectares, the largest share of total area under pasture. Under this grazing management practice, the pasture received 4 tons of poultry litter per hectare and no commercial nitrogen fertilizer was applied at all. However, producers maintained minimum biomass of 1600 kilograms per hectare during grazing at a stocking rate of 0.63 AU per hectare. BMP 2022 and BMP 0012 are second, each of them allocated about 6000 ha. The grazing management practices BMP 0011 and BMP 3031 were each allocated about 3000 hectares of land.

Table 18 Comparison of Optimal Management Practices (Ha) When Maximum Average Phosphorus Target is 30 Mg / year as Average P Loss Deviations Above the Mean Phosphorus Load were Reduced from 10 Mg to 2 Mg per year

Optimal Management Practice					Deviation Above Maximum 30Mg Phosphorus Loss (Mg)				
Code	PL	Nit.	MB	SR	10	8	6	4	2
	Mg	kg			Hectares Where Management Practice was Optimal				
0011	0	0	L	L	3470	3470	3470	1606	1568
0012	0	0	L	M	5954	5954	5903	6603	5349
0013	0	0	L	H	838	838	831	822	2
0221	0	100	M	L	360	360	365	98	990
0222	0	100	M	M	2575	2575	2447	3957	3870
0223	0	100	M	H	189	189	168	188	1736
0331	0	150	H	L	66	66	59	10	729
0332	0	150	H	M	13	13	14	42	499
0333	0	150	H	H	69	69	221	1214	4009
0111	0	50	L	L	1	1	124	1	7
0112	0	50	L	M	2717	2717	2897	2631	1782
0113	0	50	L	H	0	0	2	1	1
1011	2	0	L	L	4	4	8	7	7
1012	2	0	L	M	109	109	100	11	5
1013	2	0	L	H	2	2	3	2	1
1231	2	100	H	L	0	0	1	0	1
1232	2	100	H	M	0	0	0	0	0
1233	2	100	H	H	0	0	0	0	0
1121	2	50	M	L	0	0	1	0	0
1122	2	50	M	M	1	1	2	2	2
1123	2	50	M	H	0	0	0	0	0
2021	4	0	M	L	8911	8911	8115	6782	5939
2022	4	0	M	M	6091	6091	6463	5101	910
2023	4	0	M	H	1245	1245	939	1140	632
3031	6	0	H	L	3041	3041	3504	3805	4903
3032	6	0	H	M	2	2	7	4	1
3033	6	0	H	H	0	0	1	0	0
3231	6	100	H	L	379	379	424	1853	2432
3431	6	200	H	L	0	0	0	78	191
3432	6	200	H	M	1	1	1	60	199
3433	6	200	H	H	1	1	4	31	502
Ave. P Loss (Mg/yr)					30	30	30	29	26
Ave. P Deviation(Mg/yr)					7.5	7.5	6.0	4.0	2.0

** Abbreviations used: PL=Poultry Litter Applied (Mg/ha); Nit. = Commercial Nitrogen Applied (kg/ha); MB= Minimum Biomass (L=1.1,M=1.6,H=2.0 Mg/ha); SR = Stocking Rate (L=.63, M=1.0, H=1.26 Animal units/ha). See Table 13 for the best management practice (BMP) associated with this BMP code.*

Table 19 Comparison of Optimal Management Practices (Ha) When Maximum Average Phosphorus Target is 25 Mg / year as Average P Loss Deviations Above the Mean Phosphorus Load were Reduced from 10 Mg to 2 Mg per year.

Optimal Management Practice					Deviation Above Maximum 25Mg Phosphorus Loss (Mg)				
Code	PL	Nit.	MB	SR	10	8	6	4	
	Mg	kg			Hectares Where Management Practice was Optimal				
0011	0	0	L	L	4356	4356	4354	2917	
0012	0	0	L	M	3081	3081	3096	4441	
0013	0	0	L	H	1	1	2	3	
0221	0	100	M	L	1107	1107	1080	1185	
0222	0	100	M	M	3926	3926	3813	3605	
0223	0	100	M	H	824	824	828	994	
0331	0	150	H	L	1	1	1	15	
0332	0	150	H	M	576	576	573	542	
0333	0	150	H	H	3367	3367	3600	3949	
0111	0	50	L	L	2	2	1	1	
0112	0	50	L	M	1007	1007	1020	1070	
0113	0	50	L	H	1	1	0	1	
1011	2	0	L	L	2	2	3	10	
1012	2	0	L	M	7	7	8	6	
1013	2	0	L	H	2	2	3	1	
1231	2	100	H	L	0	0	1	0	
1232	2	100	H	M	0	0	0	0	
1233	2	100	H	H	0	0	0	0	
1121	2	50	M	L	0	0	0	0	
1122	2	50	M	M	3	3	2	2	
1123	2	50	M	H	0	0	0.4	1	
2021	4	0	M	L	8174	8174	8338	7655	
2022	4	0	M	M	1620	1620	1613	1304	
2023	4	0	M	H	1679	1679	1671	1218	
3031	6	0	H	L	6582	6582	6579	5576	
3032	6	0	H	M	4	4	4	2	
3033	6	0	H	H	1	1	1	18	
3231	6	100	H	L	441	441	441	1658	
3431	6	200	H	L	27	27	26	112	
3432	6	200	H	M	0	0	1	37	
3433	6	200	H	H	1	1	1	6	
Ave. P Loss (Mg/yr)					25	25	25	25	
Ave. P Deviation(Mg/yr)					6.2	6.2	6.0	4.0	6.2

** Abbreviations used: PL=Poultry Litter Applied (Mg/ha); Nit. = Commercial Nitrogen Applied (kg/ha); MB= Minimum Biomass (L=1.1,M=1.6,H=2.0 Mg/ha); SR = Stocking Rate (L=.63, M=1.0, H=1.26 Animal units/ha). See Table 13 for the best management practice (BMP) associated with this BMP code.*

Table 19 shows the optimal grazing management practices selected by the economic optimization model and amount of land allocated for each selected management practice when the mean annual total phosphorus runoff for the Eucha-Spavinaw watershed was reduced from 40 Mg to 25 Mg per year, with phosphorus deviation limits above target varied from not more than 10 Mg to 4 Mg per year. When the mean annual phosphorus runoff was limited to 25 Mg per year, the area allocated for BMP 2021 declined slightly, but it remained the largest share of total area under pasture followed by BMP 3031 and BMP 0011. The amount of land allocated for BMP 0011, BMP 0222, BMP 0333, and BMP 3031 increased. Table 20 shows the optimal grazing management practices selected by the economic optimization model and amount of land allocated for each selected management practice when the mean annual total phosphorus runoff for the Eucha-Spavinaw watershed was reduced from 40 Mg to 20 Mg per year, with phosphorus deviation limits above target varied from not more than 10 Mg to 4 Mg per year. When the mean annual phosphorus runoff was limited to 20 Mg per year, the area allocated for BMP 0333 drastically increased to about 9,000 hectares, receiving the largest share of total area under pasture. The amount of land allocated for BMP 2021 declined to about 5,000 hectares, but ranked second to BMP 0333. Land allocated for BMP 3031 declined while that allocated for BMP 0222 remained relatively the same. The amount of land allocated for BMP 0221 increased significantly. The amount of pastureland that received no poultry litter at all increased from about 14,000 to 26,000 hectares whereas the amount of land that received 4 tons of poultry litter per hectare declined from about 18,000 to 5,000 hectares.

Table 20 Comparison of Optimal Management Practices(Ha) When Maximum Average Phosphorus Target is 20 Mg / year as Average P Loss Deviations Above the Mean Phosphorus Load were Reduced from 10 Mg to 2 Mg per year.

Optimal Management Practice					Deviation Above Maximum 20Mg Phosphorus Loss (Mg)			
Code	PL	Nit.	MB	SR	10	8	6	4
	Mg	kg			Hectares Where Management Practice was Optimal			
0011	0	0	L	L	1772	1772	1772	1775
0012	0	0	L	M	1966	1966	1966	1978
0013	0	0	L	H	2	2	2	4
0221	0	100	M	L	4129	4129	4129	4158
0222	0	100	M	M	3287	3287	3287	3369
0223	0	100	M	H	3124	3124	3124	3145
0331	0	150	H	L	1	1	1	8
0332	0	150	H	M	1884	1884	1884	1872
0333	0	150	H	H	8881	8881	8881	8879
0111	0	50	L	L	7	7	7	7
0112	0	50	L	M	554	554	554	569
0113	0	50	L	H	0	0	0	3
1011	2	0	L	L	3	3	3	9
1012	2	0	L	M	6	6	6	7
1013	2	0	L	H	2	2	2	4
1231	2	100	H	L	0	0	0	1
1232	2	100	H	M	0	0	0	1
1233	2	100	H	H	1	1	1	0
1121	2	50	M	L	0	0	0	1
1122	2	50	M	M	4	4	4	5
1123	2	50	M	H	0	0	0	1
2021	4	0	M	L	5278	5278	5278	5079
2022	4	0	M	M	13	13	13	150
2023	4	0	M	H	144	144	144	164
3031	6	0	H	L	4674	4674	4674	4382
3032	6	0	H	M	3	3	3	3
3033	6	0	H	H	1	1	1	1
3231	6	100	H	L	301	301	301	368
3431	6	200	H	L	31	31	31	31
3432	6	200	H	M	25	25	25	26
3433	6	200	H	H	1	1	1	4
Ave. P Loss (Mg/yr)					20	20	20	20
Ave. P Deviation(Mg/yr)					5.3	5.3	5.3	4.0

** Abbreviations used: PL=Poultry Litter Applied (Mg/ha); Nit. = Commercial Nitrogen Applied (kg/ha); MB= Minimum Biomass (L=1.1,M=1.6,H=2.0 Mg/ha); SR = Stocking Rate (L=.63, M=1.0, H=1.26 Animal units/ha). See Table 13 for the best management practice (BMP) associated with this BMP code.*

The amount of land that received no commercial nitrogen fertilizer dropped from approximately 35,000 to 14,000 hectares whereas the land that received from 100 - 150 kg / ha of commercial nitrogen fertilizer increased from approximately 5 to 22,000 hectares. However, the amount of pastureland on which a minimum biomass of 1,100 kg/ha was maintained during grazing declined from 18,000 to 4,000 hectares whereas the land on which a minimum biomass of 1,600 kg/ha and above was maintained during grazing increased from about 18,000 to 32,000 hectares. The amount of land that was stocked at a rate of 1.00 AU/ha and above declined from approximately 33000 to 19000 hectares while that which was stocked at a lower rate of 0.63 AU/ha increased drastically from 4,000 to 16,000 hectares.

Total Annual Phosphorus Runoff from Pastures Under Alternative Mean Annual Phosphorus Loads and Deviations Above Mean Phosphorus

Figure 11 shows the effect of alternative phosphorus runoff targets and deviation limits above target on predicted mean total annual phosphorus runoff from pastureland in the Eucha-Spavinaw Watershed.

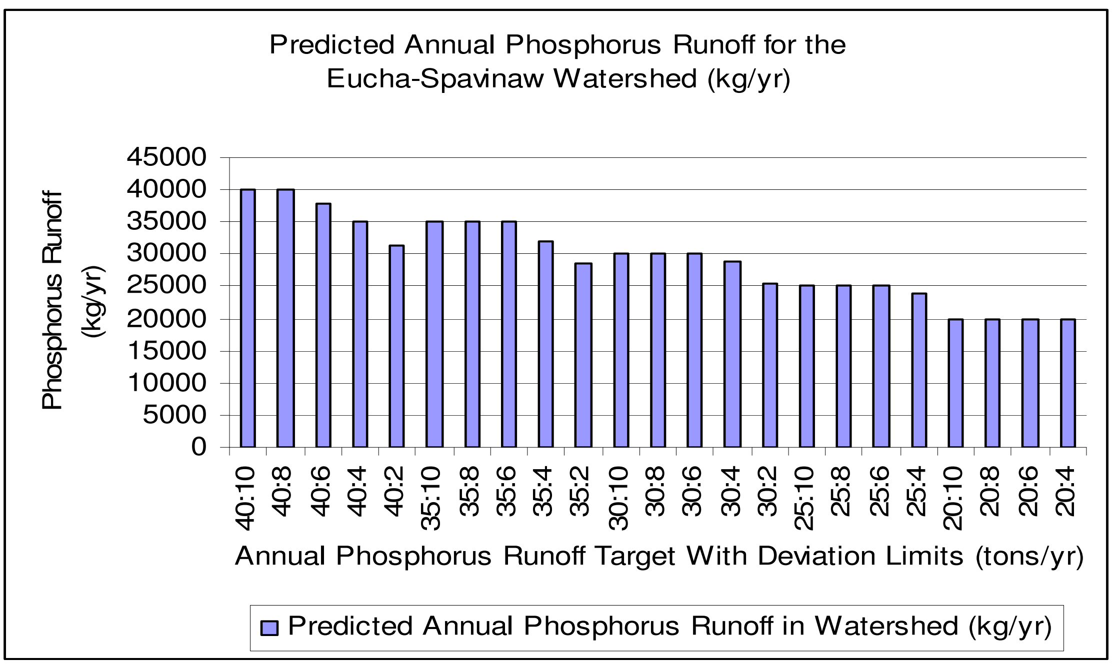


Figure 11 Predicted Annual Phosphorus Runoff for the Eucha-Spavinaw Watershed.

The annual phosphorus runoff target was reduced from 40 to 20 tons per year. The maximum allowable phosphorus deviation above the set annual phosphorus runoff target was varied in reductions of 2 tons from 10 to 2 tons per year. To meet the maximum deviations above the target the model could adopt management practices with small positive deviations and / or had a lower mean phosphorus loss. However, it was not feasible to impose a phosphorus deviation limit lower than 4 tons per year above target in the case of lower phosphorus runoff targets such as 25 tons and 20 tons per year. The imposition of an upper limit on phosphorus deviation above mean annual phosphorus runoff target affected estimated total annual phosphorus runoff from pastureland differently depending on the specified annual phosphorus runoff target and maximum allowable phosphorus deviation above that target. In general, the imposition of an upper limit on phosphorus deviation above target resulted in further reduction of predicted annual phosphorus runoff when the maximum allowable total phosphorus runoff for the entire watershed was set equal to or greater than 25 tons per year. When the target level of annual phosphorus runoff was set at 40 tons per year, setting a maximum allowable phosphorus deviation equal to or greater than 8 tons per year above this target did not affect the estimated total annual phosphorus runoff from pastureland. The imposition of upper limits on phosphorus deviation of 6, 4, and 2 tons per year above target resulted in further reduction of predicted annual phosphorus runoff from pastureland to 38 tons per year (5 percent reduction), 35 tons per year (13 percent reduction) , and 31 tons per year (23 percent reduction), respectively. The total annual phosphorus runoff from pastureland remained unchanged when the annual phosphorus loss targets were set at 35, 30 and 25 tons per year until the maximum allowable phosphorus deviation above these targets was reduced from 10 to 4 tons per year.

When the target level of annual phosphorus runoff was reduced by 25 percent to 30 tons per year, setting a maximum allowable phosphorus deviation equal to or greater than 6 tons per year above this target did not affect the estimated total annual phosphorus runoff from pastureland. The imposition of upper limits on phosphorus deviation of 4 and 2 tons per year above target resulted in further reduction of predicted annual phosphorus runoff from pastureland to 29 tons per year (28 percent reduction) and 26 tons per year (36 percent reduction), respectively. When the target level of annual phosphorus runoff was reduced by 50 percent to 20 tons per year, the imposition of an upper limit on phosphorus deviation above this annual phosphorus runoff target for the entire watershed had no effect on the estimated total annual phosphorus runoff from pastureland and yielded an amount of phosphorus loss abatement for the entire watershed equivalent to the specified annual phosphorus loading target for the entire watershed. The 50 percent reduction in the phosphorus loading target for the watershed resulted in a 58 percent reduction of the optimal annual phosphorus loss from 1.2 to 0.5 kilograms per hectare.

Estimated Poultry Litter Use for the Watershed

Figure 12 and Figure 13 below show the effect of alternative annual phosphorus runoff targets and phosphorus deviation limits above target on optimal poultry litter use in the Eucha-Spavinaw watershed. As the maximum allowable total annual phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year without imposing an upper limit on the phosphorus deviation above target, the amount of poultry litter applied on pastures in the entire watershed declined from about 43,000 to 11,000 tons per year (approximately 76 percent reduction in litter applied as fertilizer) or from an average of about 1.2 to 0.3 tons per hectare). The imposition of an upper limit on phosphorus deviation above the set phosphorus loading target for the watershed resulted in further reduction of the optimal amount of poultry litter applied in the entire watershed depending on the set phosphorus loading target and the tolerance or maximum allowable phosphorus runoff deviation above the specified phosphorus runoff target. The lower the maximum allowable phosphorus runoff deviation above a specified target the larger the reduction in the estimated annual phosphorus runoff from pastureland in the watershed.

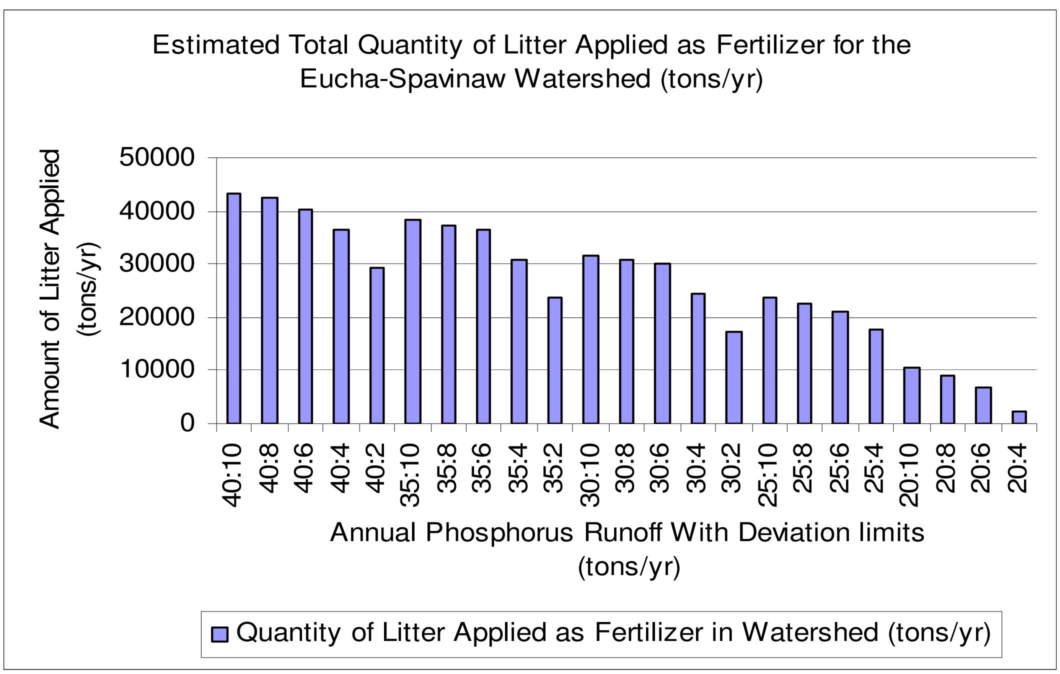


Figure 12 Estimated Total Quantity of Litter Applied in Eucha-Spavinaw Watershed.

Also, the lower the maximum allowable total annual phosphorus runoff target for the watershed the lesser the reductions in the estimated annual phosphorus runoff from pastureland resulting from the imposition of an upper phosphorus runoff deviation above the specified target. As shown in Figure 12 and Figure 13, as the maximum allowable total annual phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year with an upper limit of 4 tons per year imposed on the phosphorus deviation above target, the amount of poultry litter applied on pastures in the entire watershed declined from about 43,000 to 2,300 tons per year (approximately 95 percent reduction in litter applied as fertilizer) or from an average of about 1.2 to 0.06 tons per hectare).

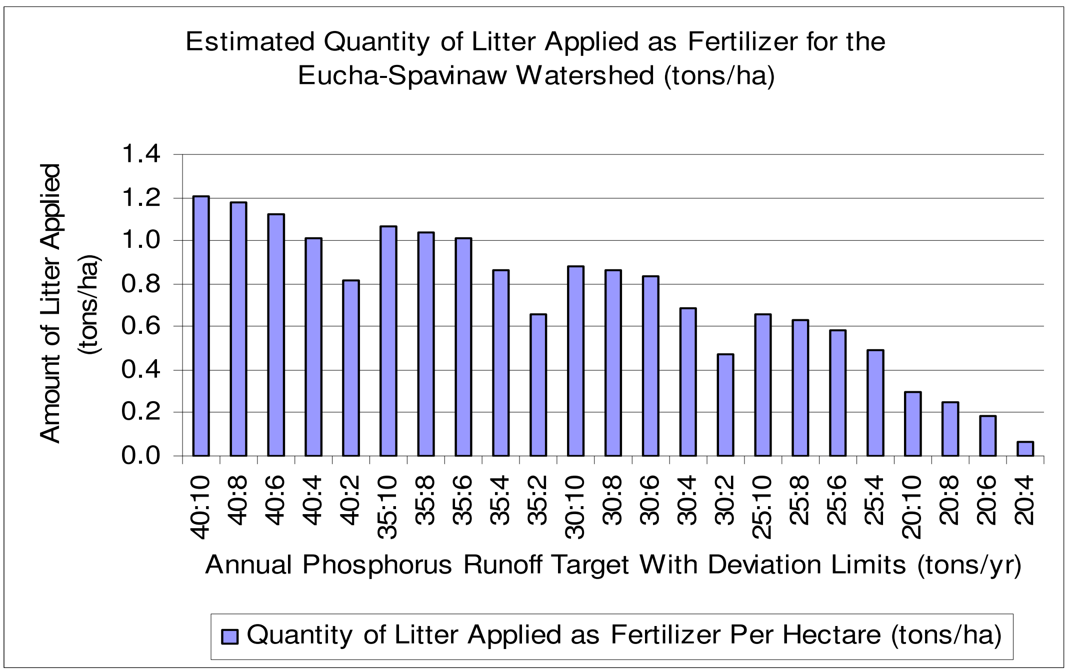


Figure 13 Estimated Quantity of Litter Used Per Ha in Eucha-Spavinaw Watershed.

Estimated Poultry Litter Shipments to Litter-to-Energy Power Plant

Figure 14 below shows the effect of alternative annual phosphorus runoff targets and phosphorus deviation limits above target on optimal litter shipments from chicken farm centroids in the watershed to the possible litter-to-energy processing plant with and without upper phosphorus deviation limits above target. As the allowable total annual phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year, the optimal amount of poultry litter shipped to the litter-to-energy processing plant (located at Jay, Oklahoma), increased depending on the tolerance or allowed deviation above the specified average total phosphorus target. When the mean phosphorus loading target was reduced from 40 to 20 tons per year without imposing an upper limit on the phosphorus deviation, the estimated amount of poultry litter shipped to the litter-to-energy processing plant increased from 46 to 79 thousand tons per year. The imposition of an upper limit on phosphorus deviation of not more than 4 tons per year above the phosphorus loading target of 20 tons per year for the watershed resulted in further increases of the optimal amount of poultry litter shipped to the processing plant to about 87 thousand tons per year.

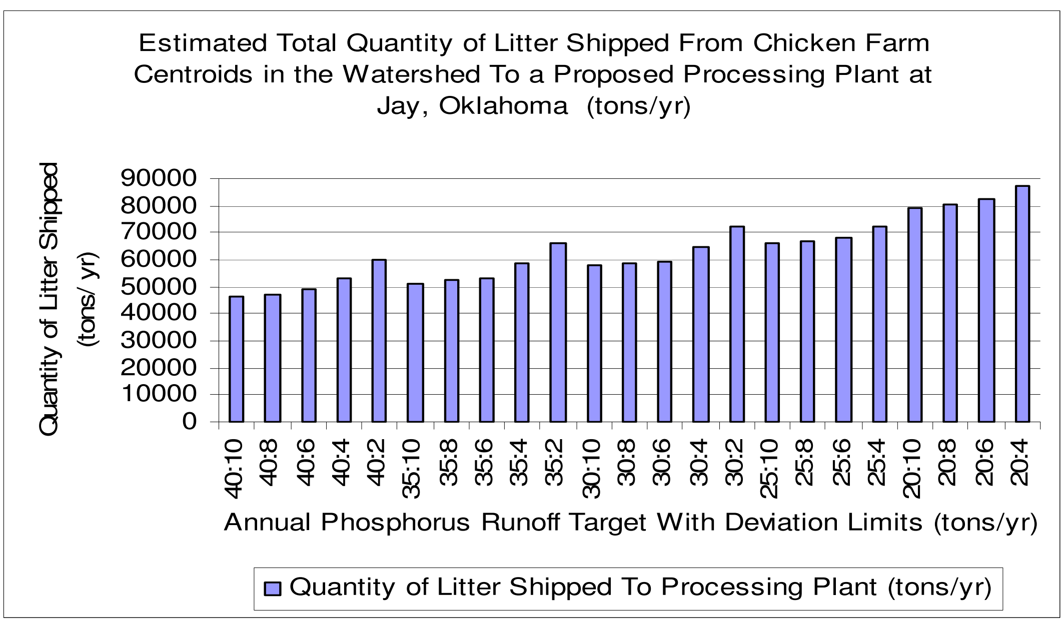


Figure 14 Quantity of Litter Shipped From Chicken Farm Centroids to Energy Plant.

The lower the maximum allowable phosphorus runoff deviation above a specified target the larger the increase in the estimated amount of poultry litter shipped to the litter- to-energy processing plant. However, the lower the maximum allowable total annual phosphorus runoff target for the watershed the lesser the increase in the amount of poultry litter shipped to the litter-to-energy processing plant resulting from the imposition of upper phosphorus runoff deviations above the specified target.

Estimated Litter Application Rates For Selected Major Soil Types

A closer look at how the different soil types in the Eucha-Spavinaw Watershed performed in terms of optimal litter application rates and phosphorus runoff under alternative phosphorus loss targets revealed considerable variation within and across the different targets. The SWAT model delineated 27 soil types in the Eucha-Spavinaw watershed. Six of the identified soil types are shown in Table 21 below and cover about 73 percent of the total area of the Eucha-Spavinaw watershed.

Table 21 Major Soil Types in the Eucha-Spavinaw Watershed

Soil Type	Share of Total Watershed Area (%)
Clarksville	17
Nixa	16
Captina	14
Doniphan	12
Tonti	8
Newtonia	6

For discussion purposes, we focus on these six major soils to highlight the variation between the amounts of litter that can be applied to and amount of predicted phosphorus runoff from different soil types given alternative phosphorus runoff targets and phosphorus runoff deviations above the specified targets. Figure 15 below shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on optimal litter application rates for the Clarksville soil. The overall quantity of poultry litter applied per hectare on the Clarksville soil declined as the annual phosphorus runoff target was reduced from 40 to 20 tons per year.

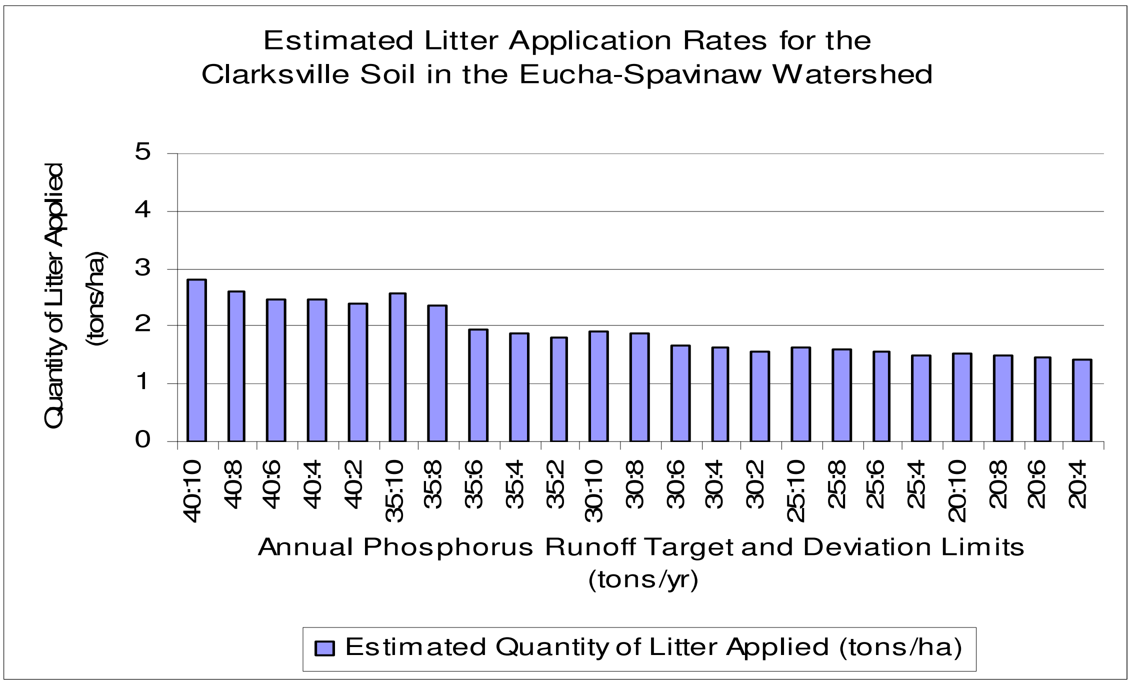


Figure 15 Estimated Litter Application Rates for the Clarksville Soil.

However, the amount of litter applied per hectare declined rapidly when the phosphorus runoff target was reduced from 40 to 30 tons per hectare and then continued to decline steadily at lower phosphorus runoff target levels. Reducing the target phosphorus runoff from 40 to 30 tons per year with a phosphorus deviation limit of not more than 2 tons per year above the target reduced the average amount of litter applied on pastures from 2.8 to 1.6 tons per hectare. A 50 percent reduction in the phosphorus loss target from 40 to 20 tons per year with a phosphorus deviation limit of not more than 4 tons per year above the target reduced the average amount of litter applied on pastures from 2.8 to 1.4 tons per hectare. Figure 16 below shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on optimal litter application rates for the Doniphan soil.

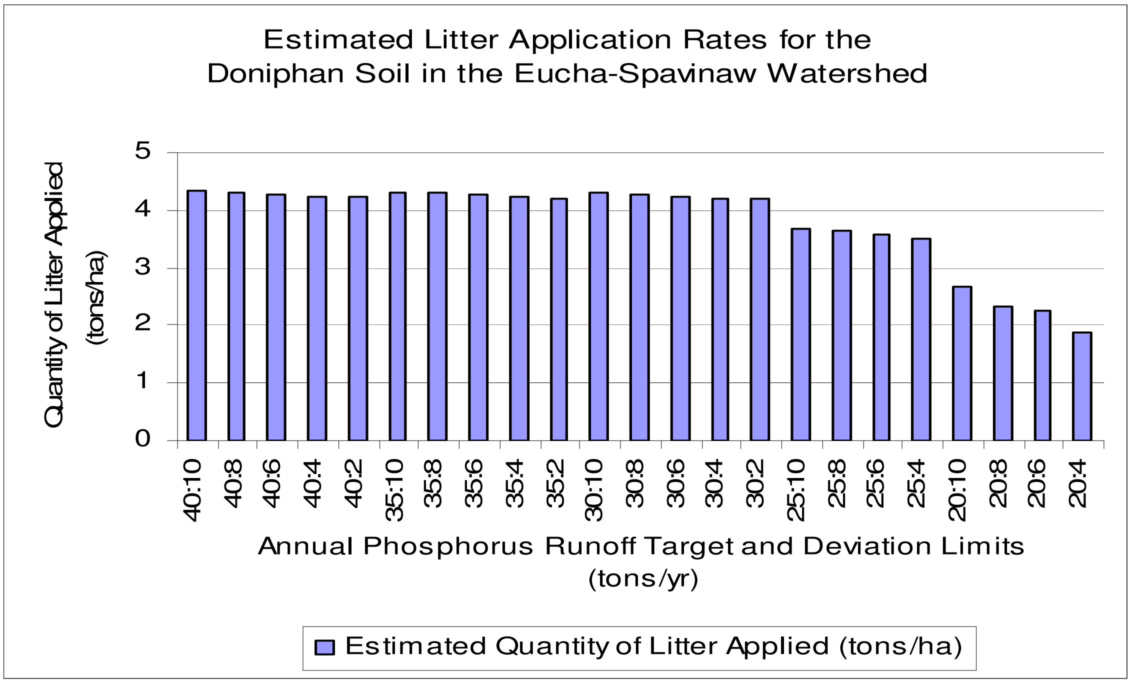


Figure 16 Estimated Litter Application Rates for the Doniphan Soil.

The overall quantity of litter applied per hectare on the Doniphan soil declined as the phosphorus loss target was reduced from 40 to 20 tons per year. However, the amount of litter applied for this soil declined slightly from an average of about 4.3 to 4.1 tons per hectare when the phosphorus loss target was reduced from 40 to 30 tons per year. The amount of litter applied per hectare declined rapidly for phosphorus runoff targets lower than 30 tons per year. A 50 percent reduction in the phosphorus loss target with an upper phosphorus deviation limit of not more than 4 tons per year yielded an optimal litter application rate of 1.8 tons per hectare for the Doniphan soil. The optimal litter application rate is higher on a Doniphan soil than a Clarksville. The optimization model allocated a higher litter application rate on a Doniphan soil because the marginal cost of removing an additional unit of phosphorus runoff on a Doniphan soil is lower than the marginal cost of abating an additional unit of phosphorus runoff on a Clarksville soil.

Figure 17 below shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on optimal litter application rates for the Captina soil.

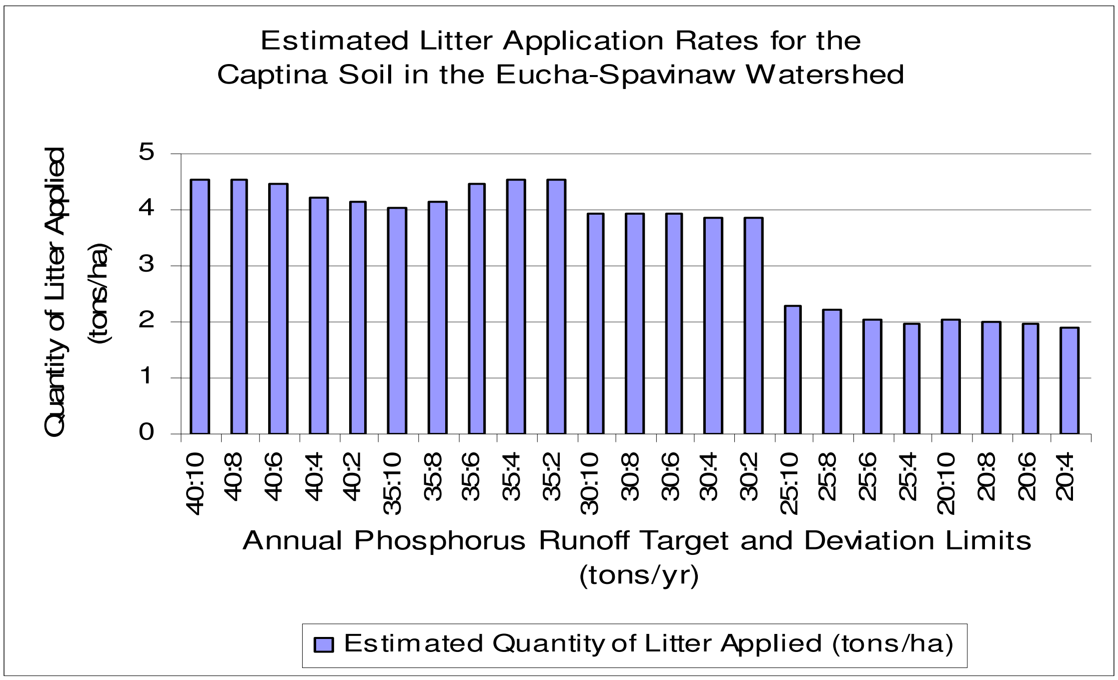


Figure 17 Estimated Litter Application Rates for the Captina Soil.

The Captina soil showed a response pattern different from Clarksville and Doniphan soils. Though the overall quantity of poultry litter applied declined from 4.5 to 1.9 tons per hectare as the phosphorus loss target was reduced from 40 to 20 tons per year with an upper phosphorus deviation limit of not more than 4 tons per year, the amount of litter applied per hectare increased when the phosphorus loss target was reduced from 40 to 35 tons per year. The amount of litter applied per hectare declined rapidly for phosphorus runoff targets lower than 35 tons per year.

Figure 18 below shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on optimal litter application rates for the Nixa soil.

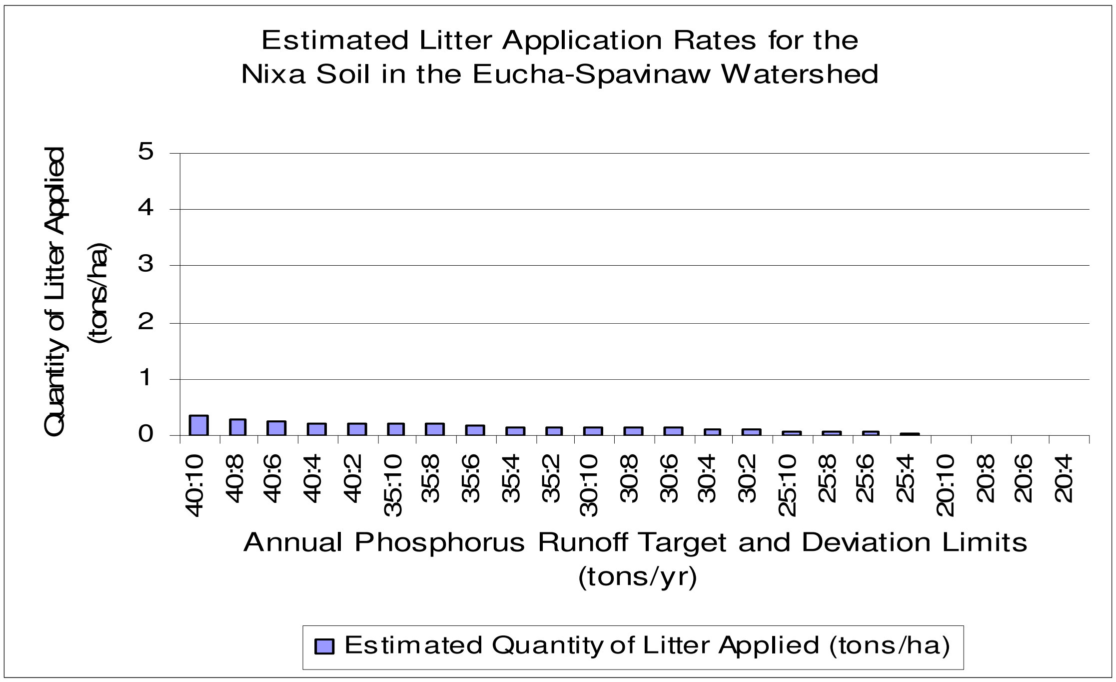


Figure 18 Estimated Litter Application Rates for the Nixa Soil.

The Nixa soil showed a response pattern very sensitive to the annual phosphorus runoff target for the watershed and phosphorus deviation limits above target. The overall quantity of litter applied declined drastically as the phosphorus runoff target was reduced from 40 to 20 tons per year and the phosphorus deviation limit above target reduced from 10 to 2 tons per year. For this soil, a 25 percent reduction in the annual phosphorus runoff target from 40 to 30 tons per year with a phosphorus deviation limit above target of not more that 2 tons per year reduced the litter application rate from 0.34 to 0.10 tons per hectare. However, a 50 percent reduction in the annual phosphorus runoff target resulted in complete cessation of litter application on pastures.

Figure 19 below shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on optimal litter application rates for the Tonti soil.

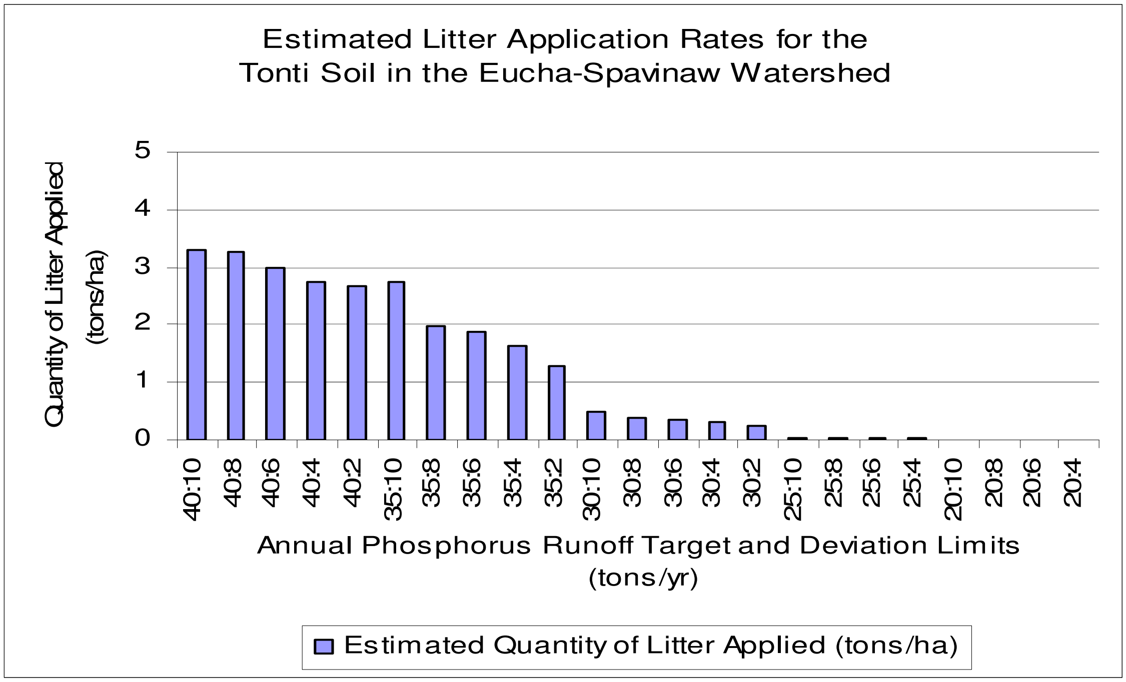


Figure 19 Estimated Litter Application Rates for the Tonti Soil.

The Tonti soil showed a response pattern very similar to that observed in the case of the Nixa soil. The amount of poultry litter applied was very sensitive to the annual phosphorus runoff target for the watershed and phosphorus deviation limits above target. The overall quantity of litter applied declined drastically as the phosphorus loss target was reduced, such that very minimal amounts of litter were applied on pastures at a phosphorus runoff target of 25 tons per year. No litter was applied at all when the phosphorus loss target reached 20 tons per year for the Tonti soil.

Figure 20 below shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on optimal litter application rates for the Newtonia soil. The quantity of poultry litter applied on the Newtonia soil declined steadily from 3.9 to 2.6 tons per hectare as the annual phosphorus runoff target was reduced from 40 to 20 tons per year. The imposition of an upper limit on phosphorus deviation above the set phosphorus loading target for the watershed further reduced the optimal amount of litter applied per hectare at all phosphorus loading targets. However, the amount of litter applied per hectare remained higher than in other soils when the phosphorus runoff target reached 20 tons per year.

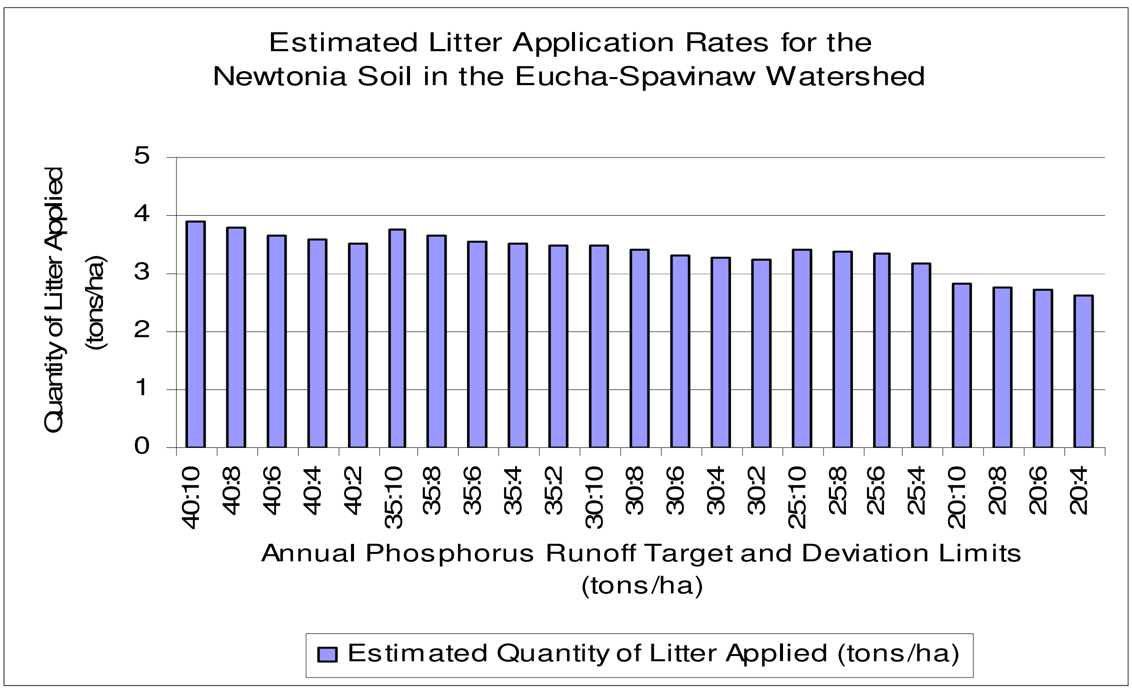


Figure 20 Estimated Litter Application Rates for the Newtonia Soil.

The preceding discussion shows the variation in the amount of poultry litter applied by soil type in the Eucha-Spavinaw watershed. A 50 percent reduction of the phosphorus loss target from 40 to 20 tons per year for the entire watershed resulted in an overall decline in the quantity of litter applied per hectare for all major soil types in the watershed. However, the degree and response pattern to reductions of the phosphorus runoff target from 40 to 20 tons per year and reduced upper limits on phosphorus runoff deviations from 10 to 2 tons per year was different for different soils. This result suggests that uniform phosphorus reduction policies and programs in the case of these major soil types in the Eucha-Spavinaw watershed are not cost effective and efficient in achieving the desired phosphorus reduction goals to ensure clean water in the lakes at least cost.

<u>Optimal Minimum Biomass Retained During Grazing For Major Soil Types</u>

Figure 21 shows the effects of alternative annual phosphorus runoff targets and deviation limits on minimum biomass maintained during grazing for the Clarksville soil in the Eucha-Spavinaw Watershed.

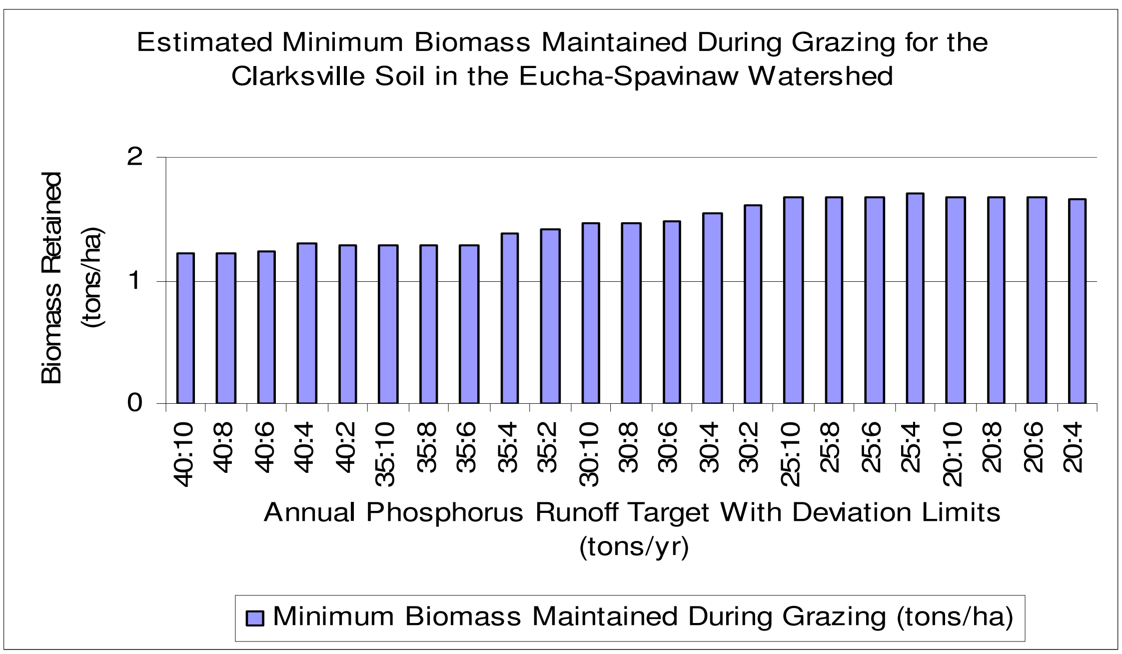


Figure 21 Minimum Biomass Maintained During Grazing for Clarksville Soil.

The Clarksville soil is one of the six major soil types in the watershed and accounts for about 17 percent of the total land area of the watershed. The minimum biomass maintained during grazing on this soil steadily increased as the maximum allowable total annual phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year with or without imposing an upper limit on the phosphorus deviation above target. The average amount of minimum biomass maintained during grazing rose from an average of approximately 1.2 to 1.7 tons per hectare.

The imposition of an upper limit on phosphorus deviation above the set phosphorus loading target for the watershed further increased the optimal amount of minimum biomass maintained during grazing depending on the set phosphorus loading target and the tolerance or maximum allowable phosphorus runoff deviation above the specified phosphorus runoff target. The higher the annual phosphorus runoff target the lesser the effect of imposing an upper phosphorus runoff deviation limit equal to or greater than 6 tons per year on the estimated minimum biomass maintained during grazing. When the target level of annual phosphorus runoff was reduced by 50 percent to 20 tons per year, the imposition of an upper limit on phosphorus deviation above this annual phosphorus runoff target for the entire watershed had no effect on the estimated minimum biomass maintained during grazing.

Figure 22 below shows the effects of alternative annual phosphorus runoff targets and deviation limits on minimum biomass maintained during grazing for the Nixa soil in the Eucha-Spavinaw Watershed. The Nixa soil accounts for about 16 percent of the total land area of the watershed. The minimum biomass maintained during grazing on this soil increased as the maximum allowable total annual phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year with or without imposing an upper limit on the phosphorus deviation above target. The amount of minimum biomass maintained during grazing rose from an average of approximately 1.1 to 1.5 tons per hectare.

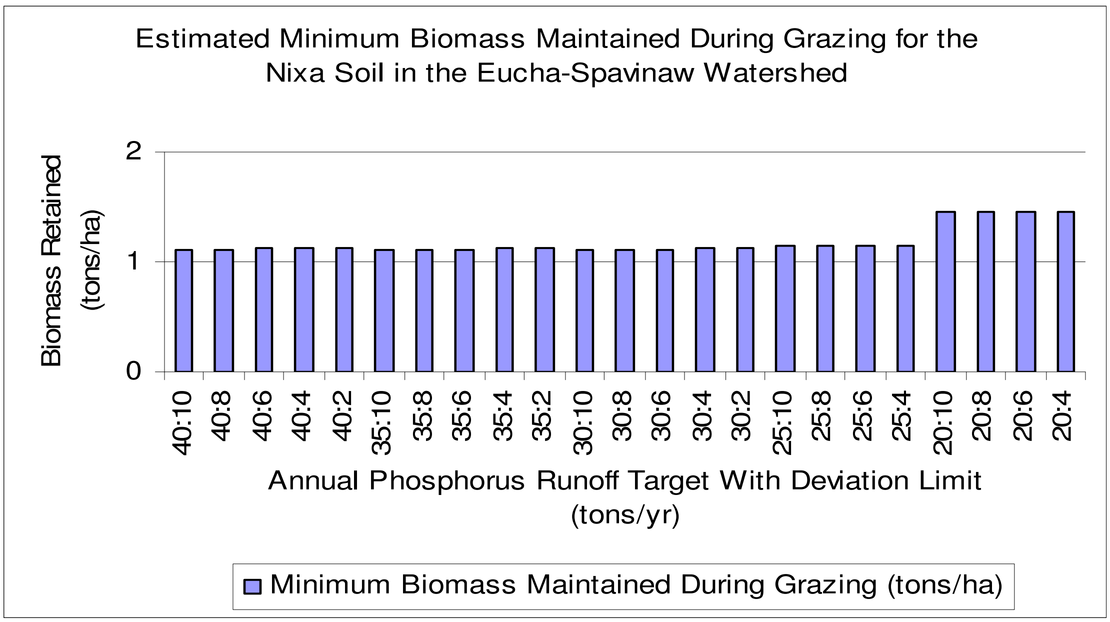


Figure 22 Minimum Biomass Maintained During Grazing for Nixa Soil.

However, the estimated amount of minimum biomass maintained during grazing remained relatively the same as the target level of annual phosphorus runoff varied from 40 to 25 tons per year with and without an upper limit imposed on the phosphorus deviation above target. The imposition of an upper limit on phosphorus deviation above phosphorus loading targets of 40, 35, 30, and 25 tons per year further increased the optimal amount of minimum biomass maintained during grazing depending on the set phosphorus loading target and the tolerance or maximum allowable phosphorus runoff deviation above the specified phosphorus runoff target. The estimated minimum biomass maintained during grazing was highest (1.5 tons/ha) when the target level of annual phosphorus runoff was reduced by 50 percent to 20 tons per year. The imposition of an upper limit on phosphorus deviation above this annual phosphorus runoff target for the entire watershed had no effect on the estimated minimum biomass maintained during grazing. Figure 23 below shows the effects of alternative annual phosphorus runoff targets and deviation limits on minimum biomass maintained during grazing for the Captina soil in the Eucha-Spavinaw Watershed.

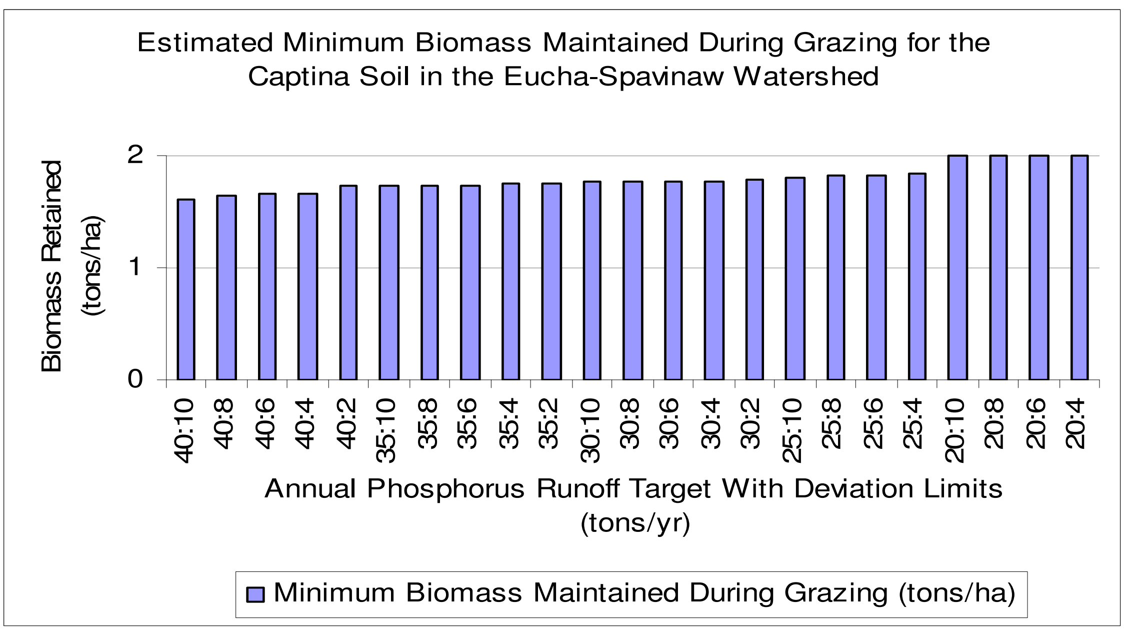


Figure 23 Minimum Biomass Maintained During Grazing for Captina Soil.

The Captina soil accounts for about 14 percent of the total land area of the watershed. The minimum biomass maintained during grazing on this soil steadily increased as the maximum allowable total annual phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year with and without imposing an upper limit on the phosphorus deviation above target. The amount of minimum biomass maintained during grazing rose from an average of approximately 1.6 to 2.0 tons per hectare. The estimated minimum biomass maintained during grazing was highest (2.0 tons/ha) when the target level of annual phosphorus runoff was reduced by 50 percent to 20 tons per year. The imposition of an upper limit on phosphorus deviation above this annual phosphorus runoff target for the entire watershed had no effect on the estimated minimum biomass maintained during grazing. Figure 24 below shows the effects of alternative annual phosphorus runoff targets and deviation limits on minimum biomass maintained during grazing for the Doniphan soil in the Eucha-Spavinaw Watershed.

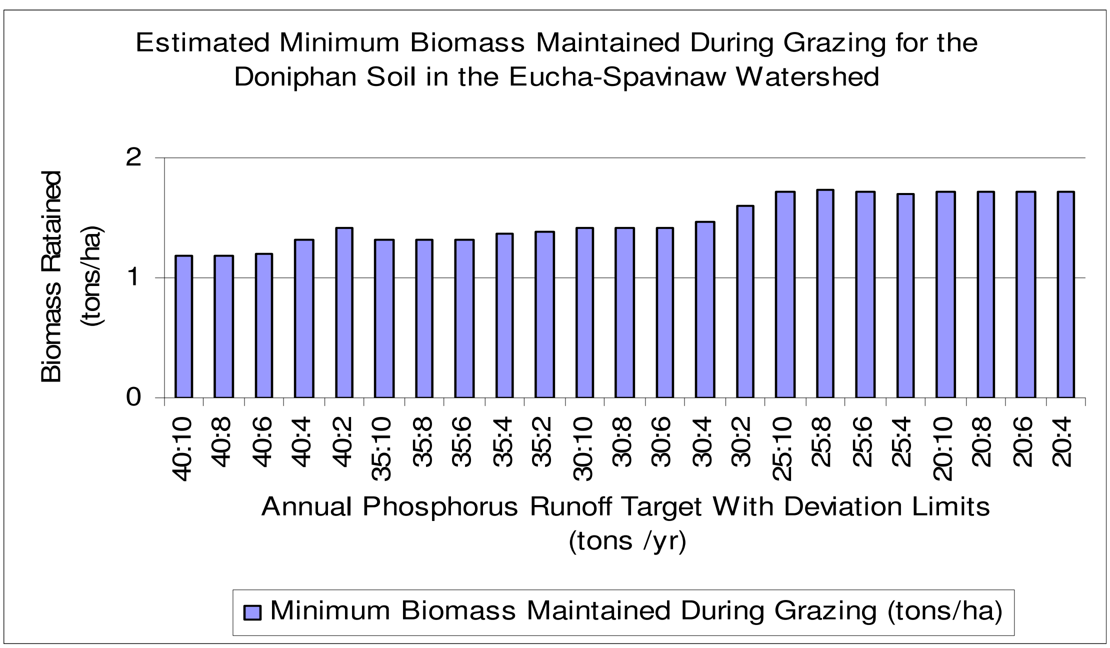


Figure 24 Minimum Biomass Maintained During Grazing for Doniphan Soil.

The Doniphan soil accounts for about 12 percent of the total land area of the watershed. The minimum biomass maintained during grazing on this soil increased as the maximum allowable total annual phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year with and without imposing an upper limit on the phosphorus deviation above target. The amount of minimum biomass maintained during grazing rose from an average of approximately 1.2 to 1.7 tons per hectare. The estimated minimum biomass maintained during grazing remained relatively the same when the target level of the total annual phosphorus runoff for the watershed was reduced to 25 and 20 tons per year. Figure 25 below shows the effects of alternative annual phosphorus runoff targets and deviation limits on minimum biomass maintained during grazing for the Tonti soil in the Eucha-Spavinaw Watershed. The Tonti soil accounts for about 8 percent of the total land area of the watershed. The minimum biomass maintained during grazing on this soil increased as the maximum allowable total annual phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year with and without imposing an upper limit on the phosphorus deviation above target.

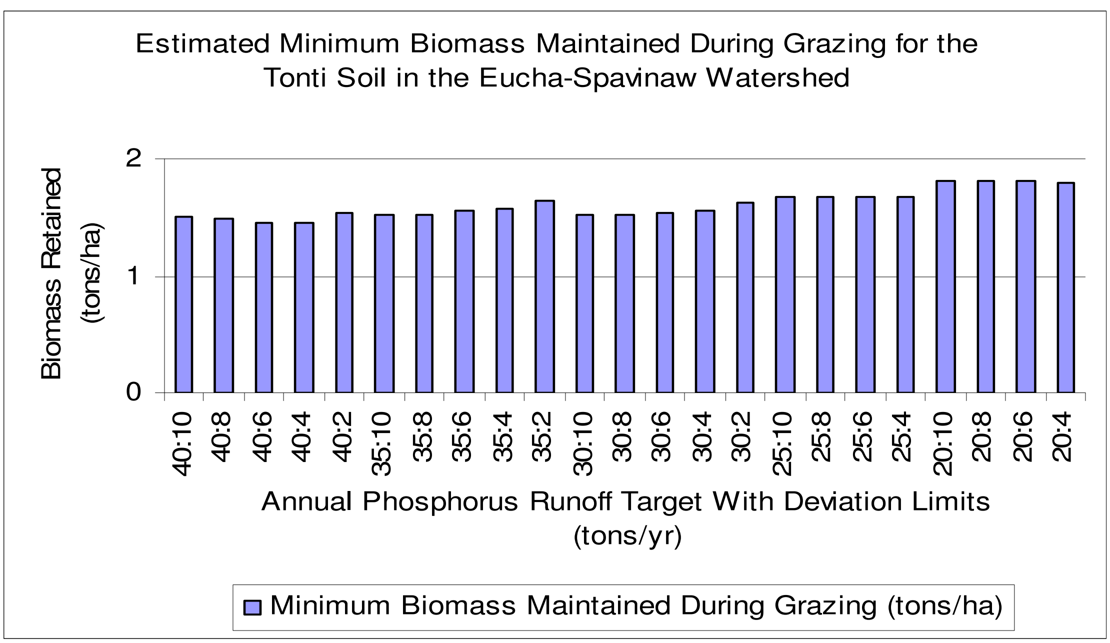


Figure 25 Minimum Biomass Maintained During Grazing for Tonti Soil.

The amount of minimum biomass maintained during grazing rose from an average of approximately 1.5 to 1.8 tons per hectare. The imposition of an upper limit on phosphorus deviation above the annual phosphorus runoff target of 20 tons per year had no effect on the estimated minimum biomass maintained during grazing. Figure 26 below shows the effects of alternative annual phosphorus runoff targets and deviation limits on minimum biomass maintained during grazing for the Newtonia soil in the Eucha-Spavinaw Watershed. The Newtonia soil accounts for about 6 percent of the total land area of the watershed. The minimum biomass maintained during grazing on this soil increased slightly as the maximum allowable total annual phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year with and without imposing an upper limit on the phosphorus deviation above target.

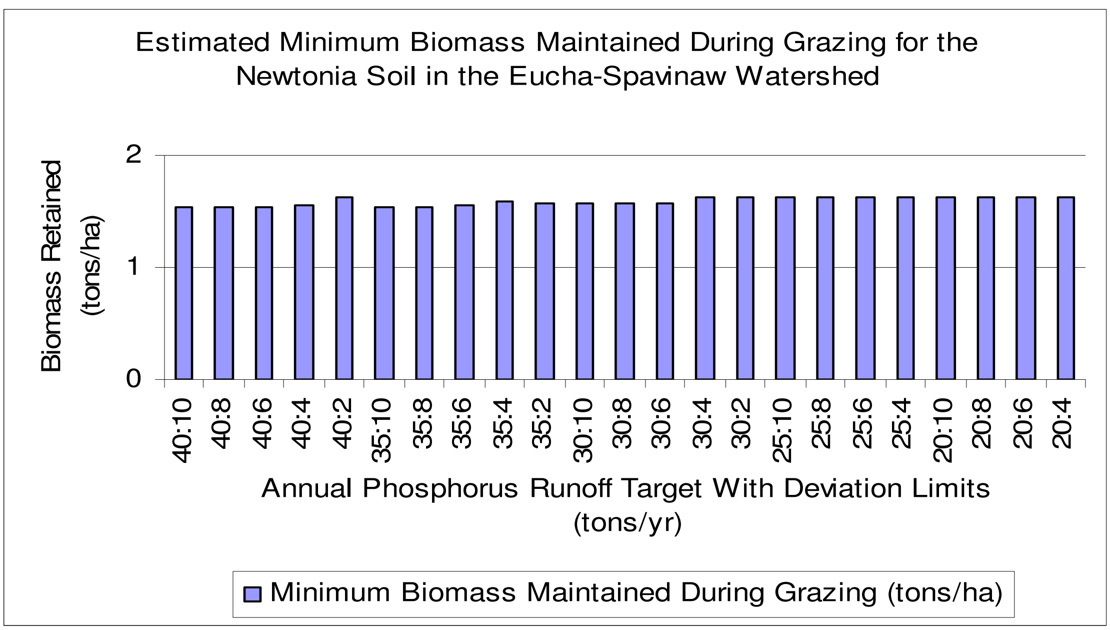


Figure 26 Minimum Biomass Maintained During Grazing for Newtonia Soil.

The amount of minimum biomass maintained during grazing rose from an average of approximately 1.5 to 1.6 tons per hectare. The estimated minimum biomass maintained during grazing remained unchanged at 1.6 tons per hectare when the target level of the total annual phosphorus runoff for the watershed was reduced to 25 and 20 tons per year.

Optimal Amount of Phosphorus Loss For Selected Major Soil Types

Figure 27 below shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of phosphorus runoff on the Clarksville soil. The estimated quantity of phosphorus runoff per hectare on the Clarksville soil declined steadily as the annual phosphorus runoff target was reduced from 40 to 25 tons per year and then declined significantly when the target reached 20 tons per year with an upper phosphorus deviation limit of not more than 4 tons per year.

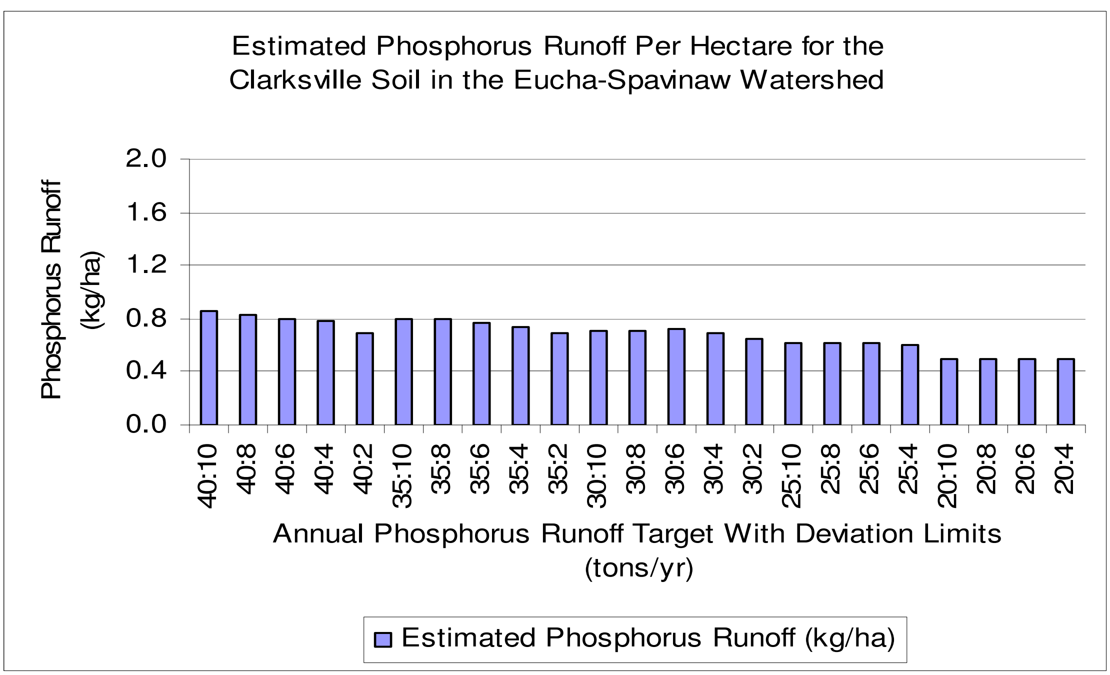


Figure 27 Estimated Amount of Phosphorus Runoff on the Clarksville Soil.

Reducing the target phosphorus runoff from 40 to 30 tons per year with a phosphorus deviation limit of not more than 2 tons per year above the target reduced the average amount of phosphorus runoff from pastures from about 0.9 to 0.7 kilograms per hectare. A 50 percent reduction in the phosphorus loss target from 40 to 20 tons per year with a phosphorus deviation limit of not more than 4 tons per year above the target reduced the average amount of phosphorus runoff on pastures from 0.9 to 0.5 kilograms per hectare.

Figure 28 below shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of phosphorus runoff for the Nixa soil. The estimated quantity of phosphorus runoff per hectare on the Nixa soil declined relatively faster than in the case of the Clarksville soil as the annual phosphorus runoff target was reduced from 40 to 25 tons per year and the phosphorus runoff deviation limit above target was varied from 10 to 2 tons per year.

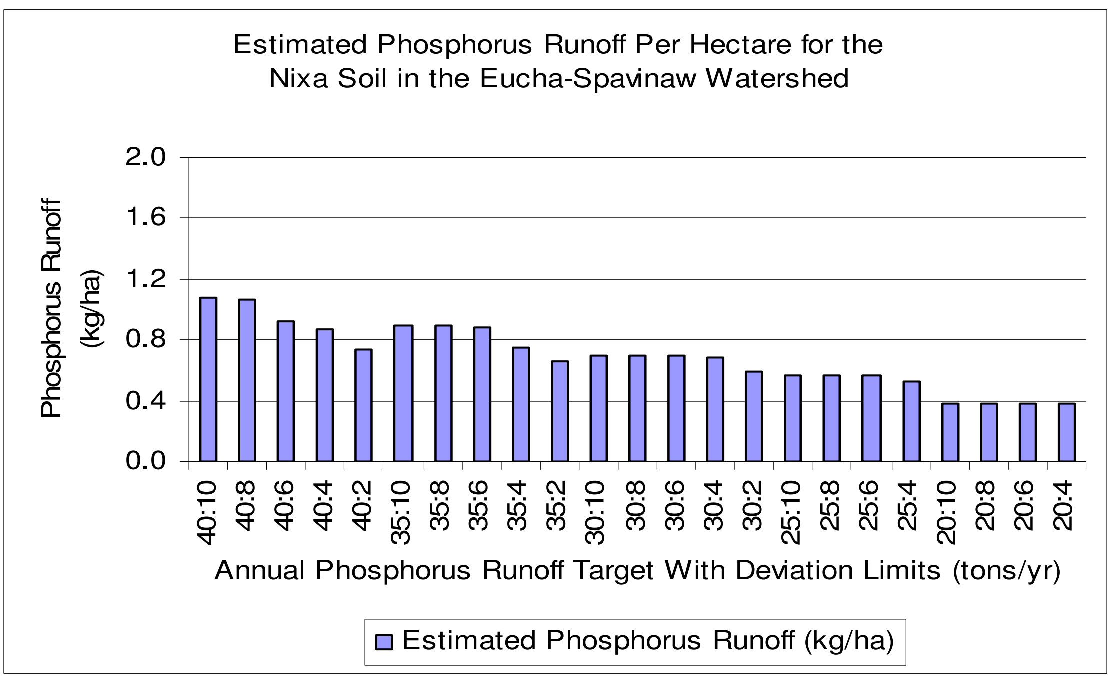


Figure 28 Estimated Amount of Phosphorus Runoff on the Nixa Soil.

The estimated amount of phosphorus runoff was sensitive to both the annual phosphorus runoff target and phosphorus deviation limit above target. Reducing the target phosphorus runoff from 40 to 30 tons per year with a phosphorus deviation limit of not more than 2 tons per year above the target reduced the average amount of phosphorus runoff from pastures from about 0.7 to 0.6 kilograms per hectare. A 50 percent reduction in the phosphorus loss target from 40 to 20 tons per year with a phosphorus deviation limit of not more than 4 tons per year above the target reduced the average amount of phosphorus runoff on pastures from 0.9 to 0.4 kilograms per hectare. Figure 29 below shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of phosphorus runoff for the Captina soil.

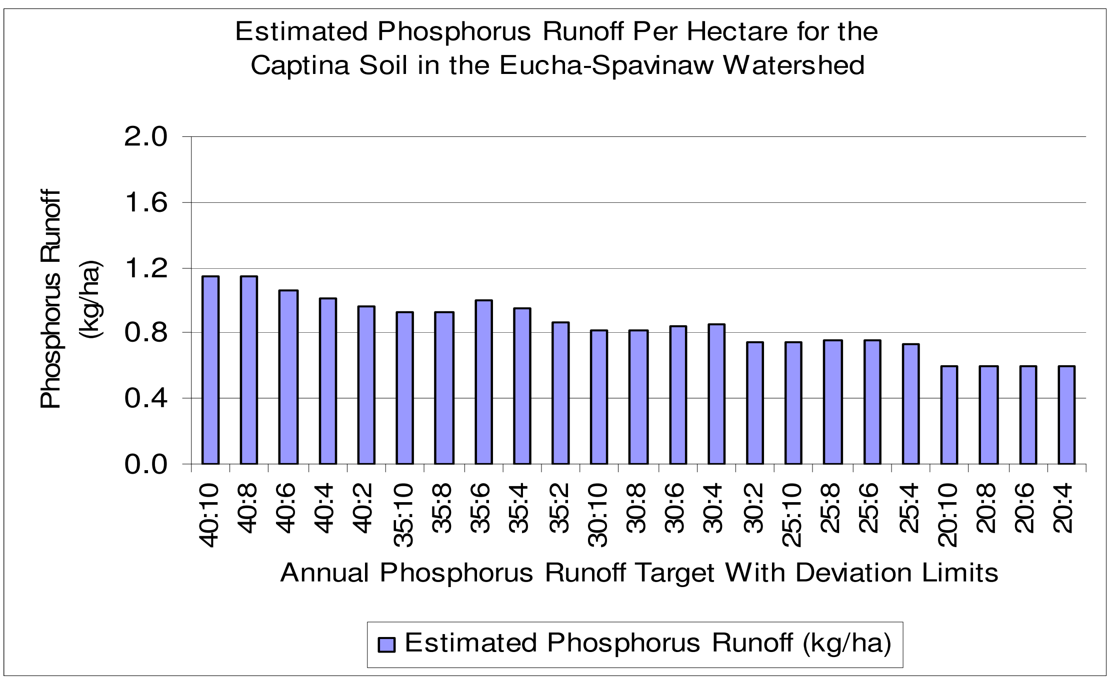


Figure 29 Estimated Amount of Phosphorus Runoff on the Captina Soil.

The estimated quantity of phosphorus runoff per hectare on the Captina soil exhibited a response pattern to changes in the annual phosphorus runoff target and phosphorus runoff deviations limits similar to that observed in the case of the Clarksville soil. The amount of phosphorus runoff declined steadily as the annual phosphorus runoff target was reduced from 40 to 30 tons per year and then declined rapidly at lower annual phosphorus runoff target levels and phosphorus runoff deviation limits above target. Reducing the target phosphorus runoff from 40 to 30 tons per year with a phosphorus deviation limit of not more than 2 tons per year above the target reduced the average amount of phosphorus runoff from pastures from about 1.0 to 0.7 kilograms per hectare. A 50 percent reduction in the phosphorus loss target from 40 to 20 tons per year with a phosphorus deviation limit of not more than 4 tons per year above the target reduced the average amount of phosphorus runoff on pastures from 1.0 to 0.6 kilograms per hectare.

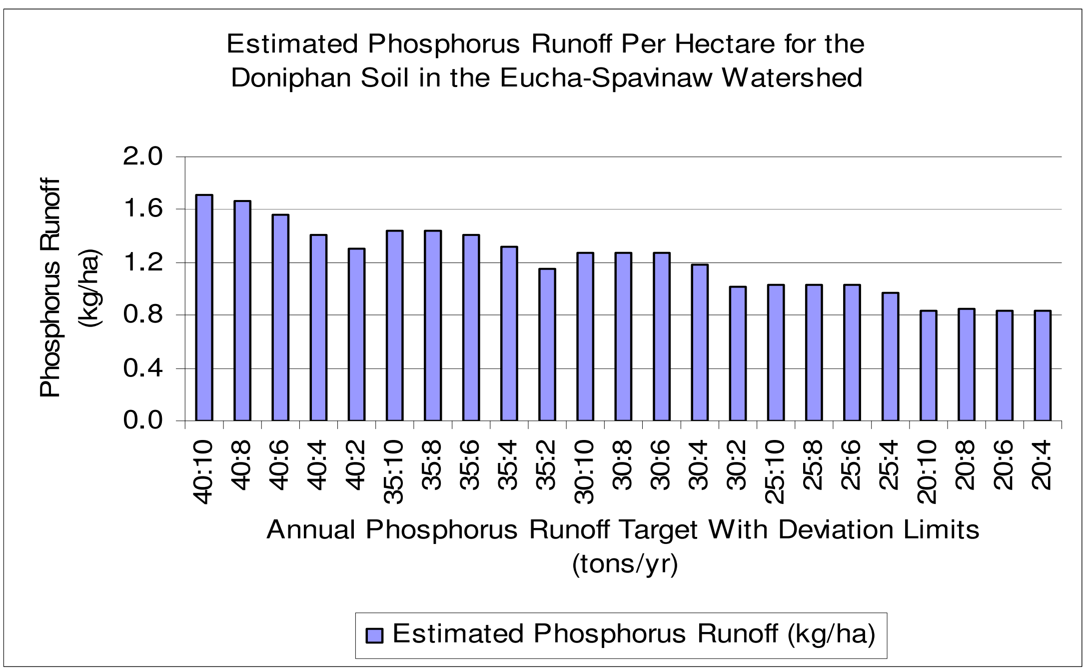


Figure 30 Estimated Amount of Phosphorus Runoff on the Doniphan Soil.

Figure 30 above shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of phosphorus runoff for the

Doniphan soil. The estimated quantity of phosphorus runoff per hectare on the Doniphan soil was not very sensitive to changes in both the annual phosphorus runoff target and phosphorus runoff deviation limits. The estimated amount of phosphorus runoff declined steadily from 1.7 to 1.3 kilograms per hectare when the target phosphorus runoff was reduced from 40 to 30 tons per year and then dropped much faster at annual phosphorus runoff targets lower than 30 tons per year.

Figure 31 below shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of phosphorus runoff for the Tonti soil. The estimated quantity of phosphorus runoff per hectare on the Tonti soil exhibited a response pattern similar to that observed in the case of Nixa.

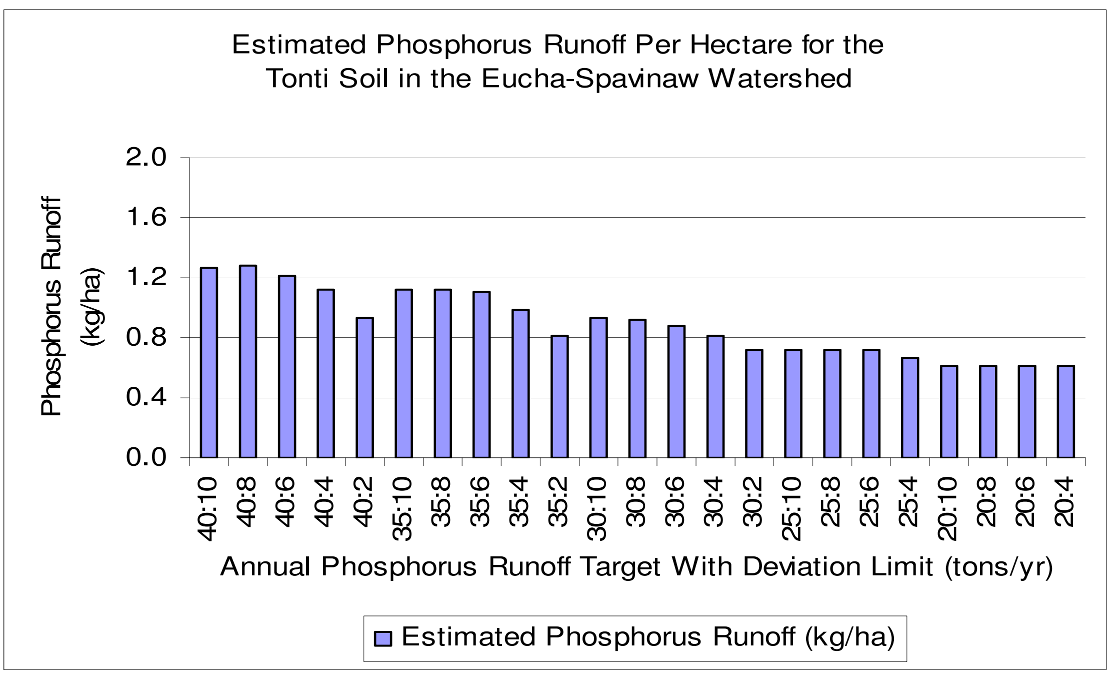


Figure 31 Estimated Amount of Phosphorus Runoff on the Tonti Soil.

The estimated amount of phosphorus runoff was sensitive to changes in both the annual phosphorus runoff target and mean phosphorus runoff deviation limits above target. The amount of phosphorus runoff per hectare declined relatively faster than in other major soils as the annual phosphorus runoff target was reduced from 40 to 25 tons per year and the mean phosphorus runoff deviation limit above target was varied from 10 to 2 tons per year. Reducing the target phosphorus runoff from 40 to 30 tons per year with a phosphorus deviation limit of not more than 2 tons per year above the target reduced the average amount of phosphorus runoff from pastures from about 0.9 to 0.7 kilograms per hectare. A 50 percent reduction in the phosphorus loss target from 40 to 20 tons per year with a phosphorus deviation limit of not more than 4 tons per year above the target reduced the average amount of phosphorus runoff on pastures from 1.0 to 0.6 kilograms per hectare.

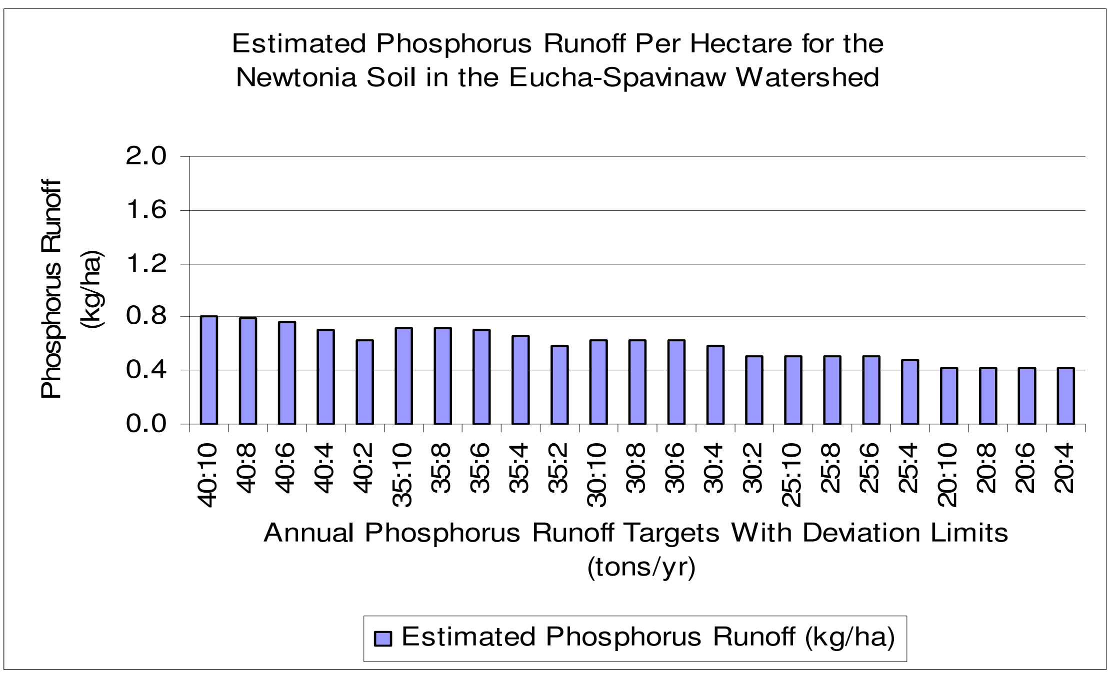


Figure 32 Estimated Amount of Phosphorus Runoff on the Newtonia Soil.

Figure 32 above shows the effect of alternative phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of phosphorus runoff for the

Newtonia soil. The overall quantity of phosphorus runoff per hectare on the Newtonia soil declined as the phosphorus loss target was reduced from 40 to 20 tons per year. However, the amount of phosphorus runoff for this soil declined slightly from an average of about 0.8 to 0.6 kilograms per hectare when the phosphorus loss target was reduced from 40 to 30 tons per year. Then the amount of phosphorus runoff declined much faster at target levels of 25 tons per year and lower such that the average amount of phosphorus runoff was estimated at 0.4 kilograms per hectare when the target annual phosphorus loading for the watershed reached 20 tons per year with an upper phosphorus deviation limit of not more than 4 tons per year.

Optimal Elemental Nitrogen Application Rates

Figure 33 below shows the effect of alternative phosphorus loss targets and phosphorus runoff deviation limit above target on optimal elemental nitrogen use in the watershed.

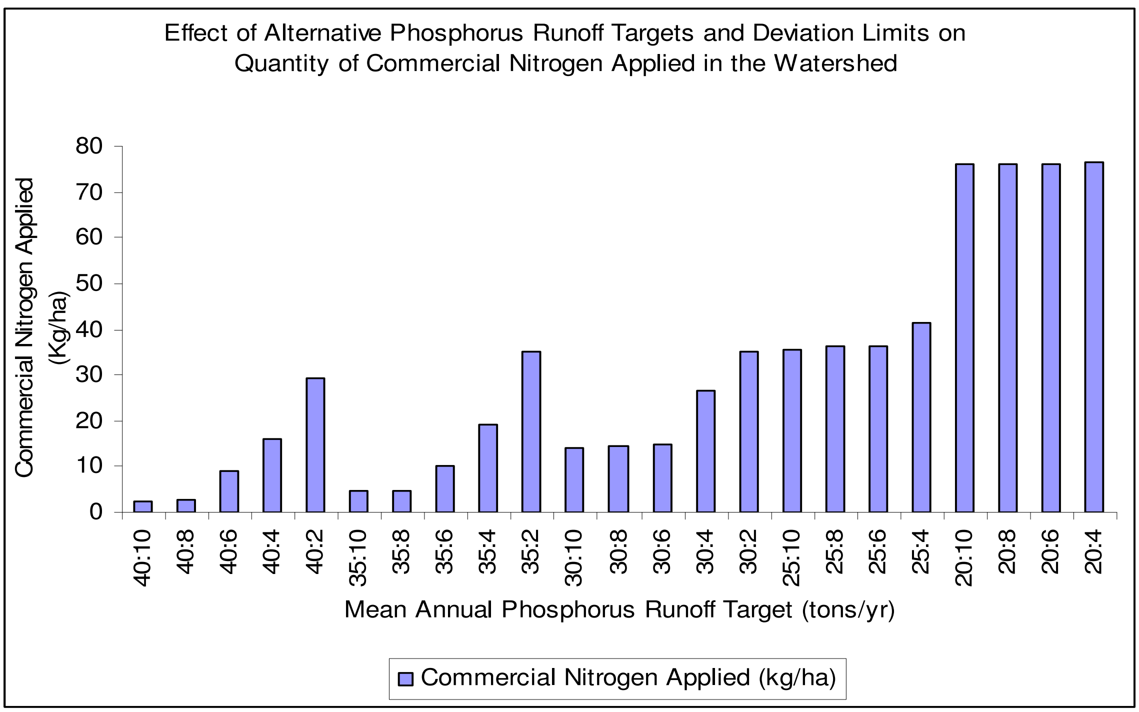


Figure 33 Optimal Elemental Nitrogen Application Rates in the Watershed.

As the allowable total annual phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year, the optimal amount of elemental nitrogen applied in the entire watershed increased depending on the tolerance or allowed deviation above the specified average total phosphorus target. Reducing the phosphorus loading target from 40 to 20 tons per year increased the optimal amount of elemental nitrogen used from about an area-weighted average of 2 kg/ha to 75 kg/ha for the entire watershed. Generally, the imposition of an upper limit on phosphorus deviation above the set loading target for the watershed resulted in further increases of the optimal amount of elemental nitrogen applied in the watershed.

Total Agricultural Income from Grazing

Figure 34 below provides an aggregate summary of the effects on agricultural income from grazing of limiting total phosphorus runoff for the entire Eucha-Spavinaw watershed to 40, 35, 30, 25 and 20 tons per year with mean phosphorus runoff deviation limits above these targets varied from 10, 8, 6, 4 and 2 tons per year. The total producer income from grazing for the entire watershed declined from about \$3.1 million to \$1.3 million as the phosphorus runoff limit was reduced from 40 tons to 30 tons per year. A further reduction of the phosphorus runoff limit to 20 tons per year with an upper mean phosphorus runoff deviation limit of not more than 4 tons per year further reduced total agricultural income from grazing to \$0.6 million. Reducing the upper limit on the phosphorus runoff deviation above the target loading from 10 tons to as low as 2 tons per year further reduced agricultural income at all load levels. However, reductions in total agricultural income from grazing were larger at higher phosphorus runoff limits than at lower limits. Also, the lower the upper limit on phosphorus deviation above the target phosphorus loading the larger the reduction in agricultural income from grazing at that target load.

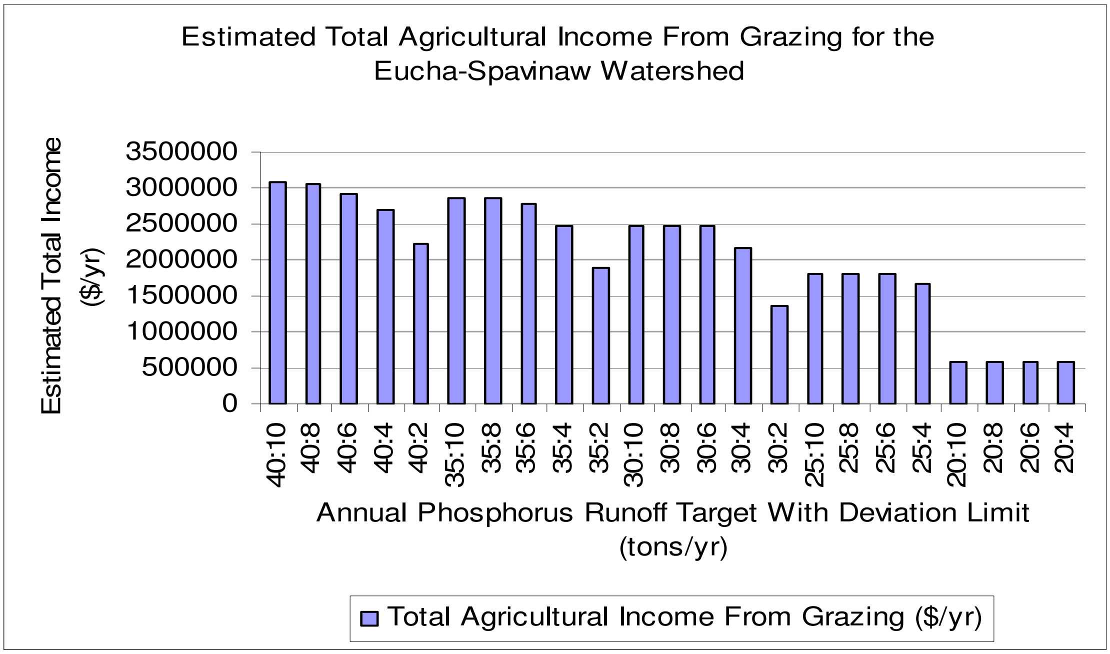


Figure 34 Estimated Producer Income from Grazing in Eucha-Spavinaw Watershed.

<u>Total Phosphorus Pollution Abatement Costs for the Watershed</u>

Figure 35 and Figure 36 below provide an aggregate summary of the effects on phosphorus pollution abatement costs of limiting total phosphorus runoff for the entire Eucha-Spavinaw watershed to 40, 35, 30, 25 and 20 tons per year with phosphorus runoff deviation limits above these targets varied from 10, 8, 6, 4 and 2 tons per year. The total phosphorus pollution abatement costs in the watershed increased rapidly as the phosphorus runoff limit was reduced from 40 tons to 20 tons per year.

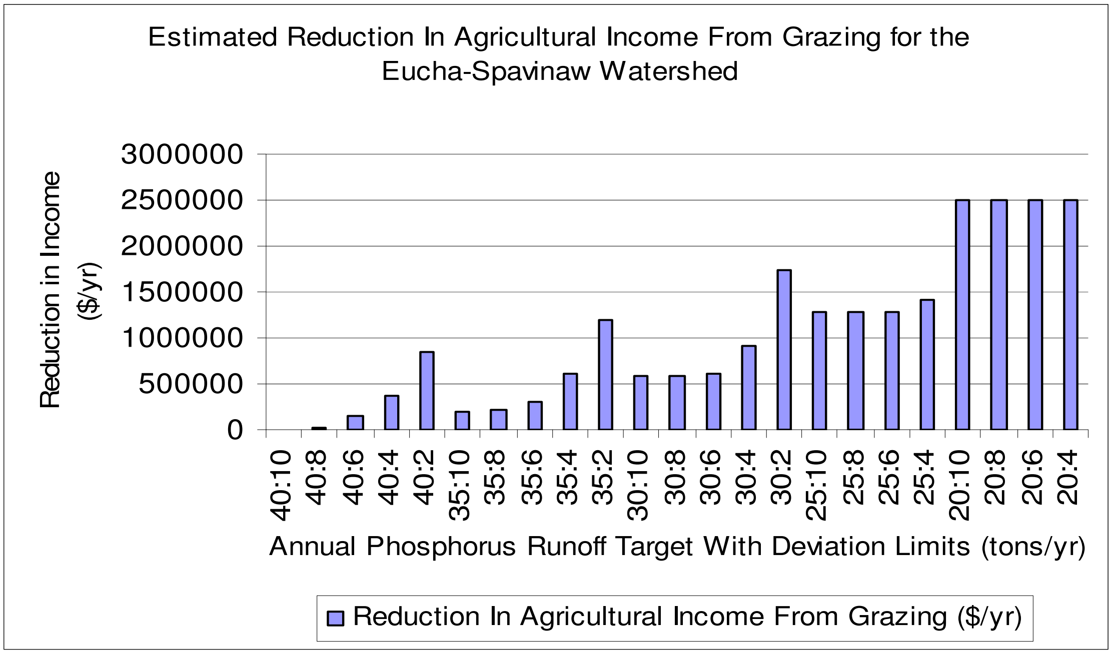


Figure 35 Reductions in Total Net Returns from Grazing in the Watershed.

Total abatement costs rose from zero to $1.7 million when phosphorus limit was reduced from 40 tons to 30 tons per year. The total abatement costs for the watershed further increased to $2.5 million when the phosphorus runoff limit was further reduced to 20 tons per year with an upper phosphorus runoff deviation limit of not more than 4 tons per year. Reducing the upper limit on the phosphorus runoff deviation above the target loading from 10 tons to as low as 2 tons per year further increased total abatement costs at all load levels. However, increases in total abatement costs due to reductions in upper phosphorus runoff deviation limits above target were larger at higher phosphorus runoff limits than at lower limits.

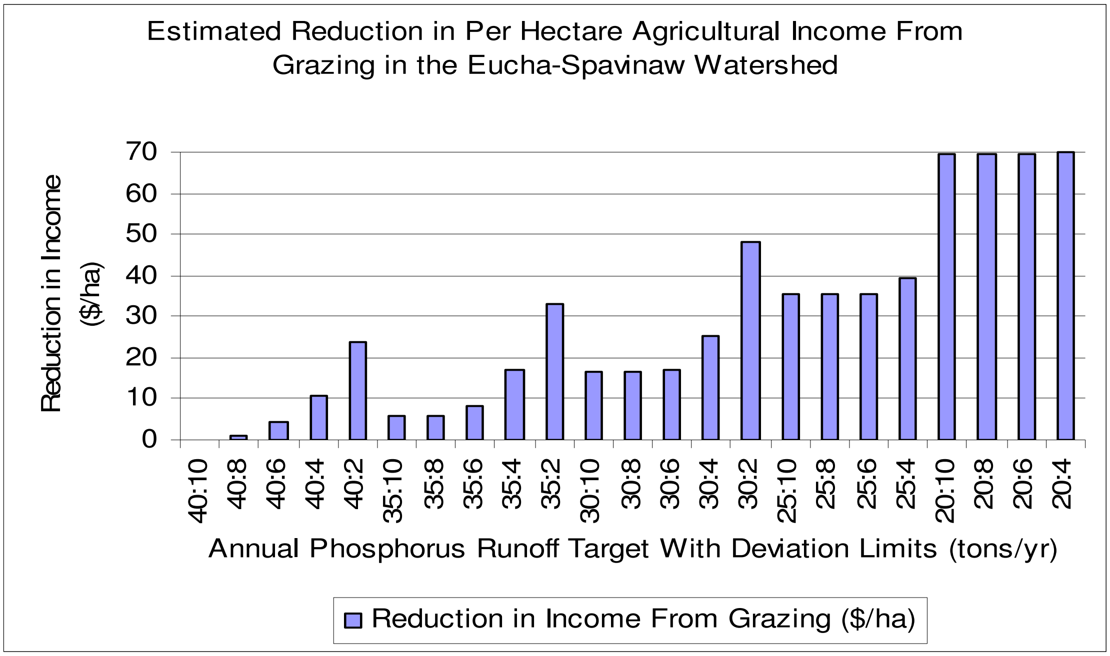


Figure 36 Reductions in Per Hectare Net Returns from Grazing in the Watershed.

Marginal Phosphorus Pollution Abatement Costs for the Watershed

Figure 37 below shows the cost of removing one additional kilogram of phosphorus as the mean annual phosphorus runoff was limited to 40, 35, 30, 25 and 20 tons per year and the phosphorus runoff deviation above target was limited to 10, 8, 6, 4, and 2 tons per year. The cost of removing an additional kilogram of phosphorus increased as the mean annual phosphorus runoff limit was reduced from 40 to 20 tons per year. It further increased at all total phosphorus runoff targets for the watershed when an upper limit imposed on phosphorus runoff deviation above the target was reduced from 10 to 2 tons per year. The marginal abatement cost rose from $28 to $385 when the upper limit on phosphorus runoff deviation limit was reduced from not more than 10 to 2 tons per year given an annual phosphorus runoff limit of 40 tons per year for the watershed. Given a phosphorus runoff deviation limit of not more than 10 tons per year above the target phosphorus loading, reducing the total phosphorus runoff limit for the watershed from 40 to 20 tons per year increased the cost of removing an additional kilogram of phosphorus from \$28 to \$334. However, the cost of removing an additional kilogram of phosphorus from \$334 to \$456 as the upper limit on phosphorus runoff deviation was reduced from not more than 10 to 4 tons per year above the phosphorus runoff limit of 20 tons per year for the watershed. The marginal abatement cost reached a high of \$635 when the annual total phosphorus runoff limit for the watershed was reduced from 40 to 30 tons per year with phosphorus runoff deviation limit of not more than 2 tons per year above the target phosphorus loading. Ancev et al. (2006) also found that marginal abatement costs rose rapidly when annual phosphorus loadings were reduced from 25 Mg to 20 Mg in the Eucha-Spavinaw watershed.

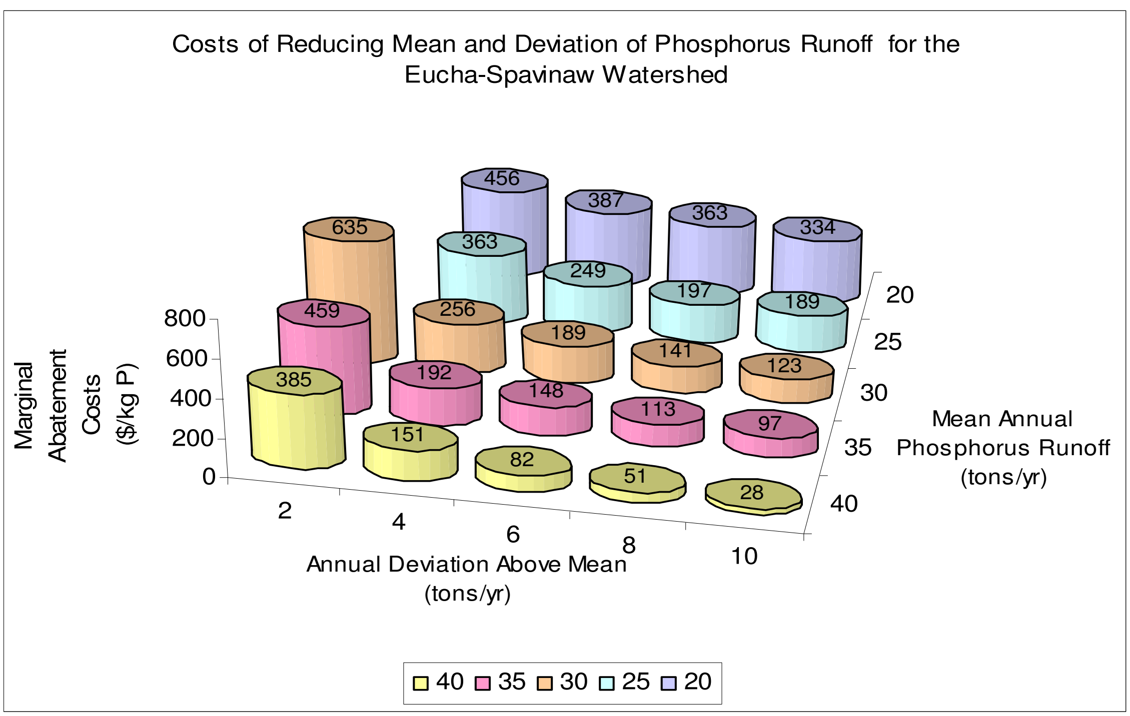


Figure 37 Marginal Abatement Costs in the Eucha-Spavinaw watershed.

Optimal Shipment Pattern of Poultry Litter Between Subbasins

Figure 38 and Figure 39 below show the effect of alternative phosphorus loss targets for the Eucha-Spavinaw watershed on optimal shipment pattern of poultry litter from chicken farm centroids to subbasin centroids and to the possible litter-to-energy processing plant at Jay, Oklahoma. The allocation of poultry litter between the processing plant and the use on pasture land within the watershed subbasins varied with the different phosphorus loss targets.

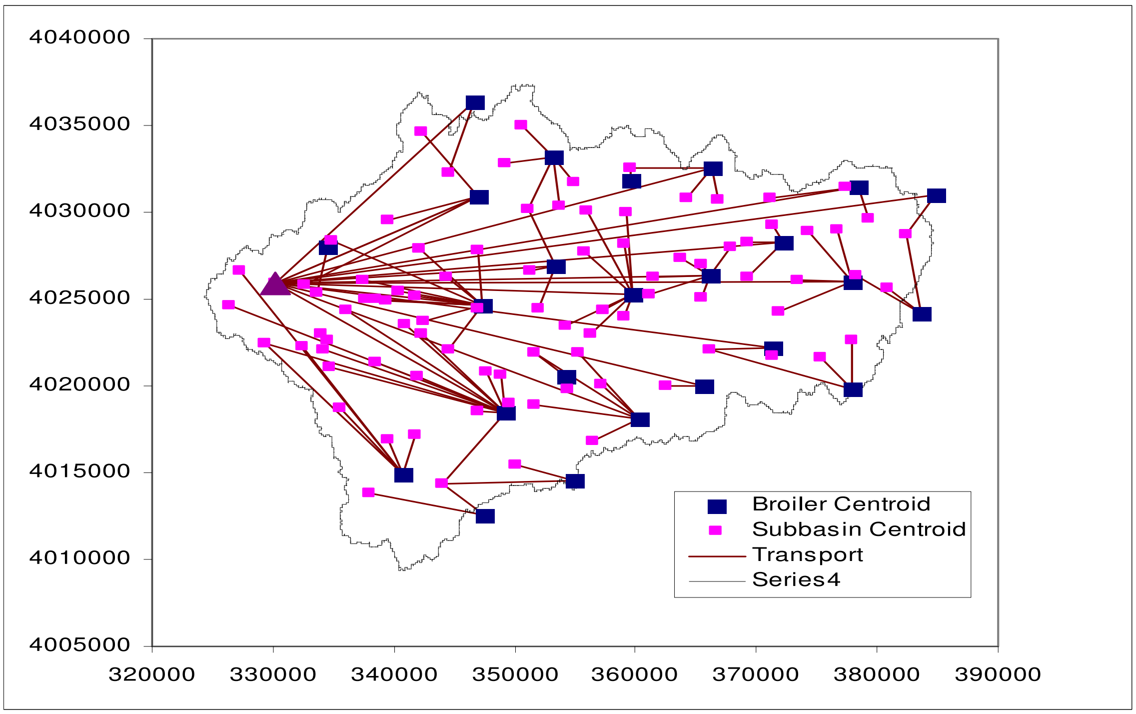


Figure 38 Litter Shipment Pattern Given Phosphorus Loss Target of 40,000 kg /yr.

As can be seen from the both figures, as the allowable maximum phosphorus loss target was reduced from 40 to 20 tons per year, the amount of litter transported from the chicken farm centroids to the litter-to-energy plant drastically increased (while the amount of litter applied as fertilizer on pasture land in the watershed subbasins declined).

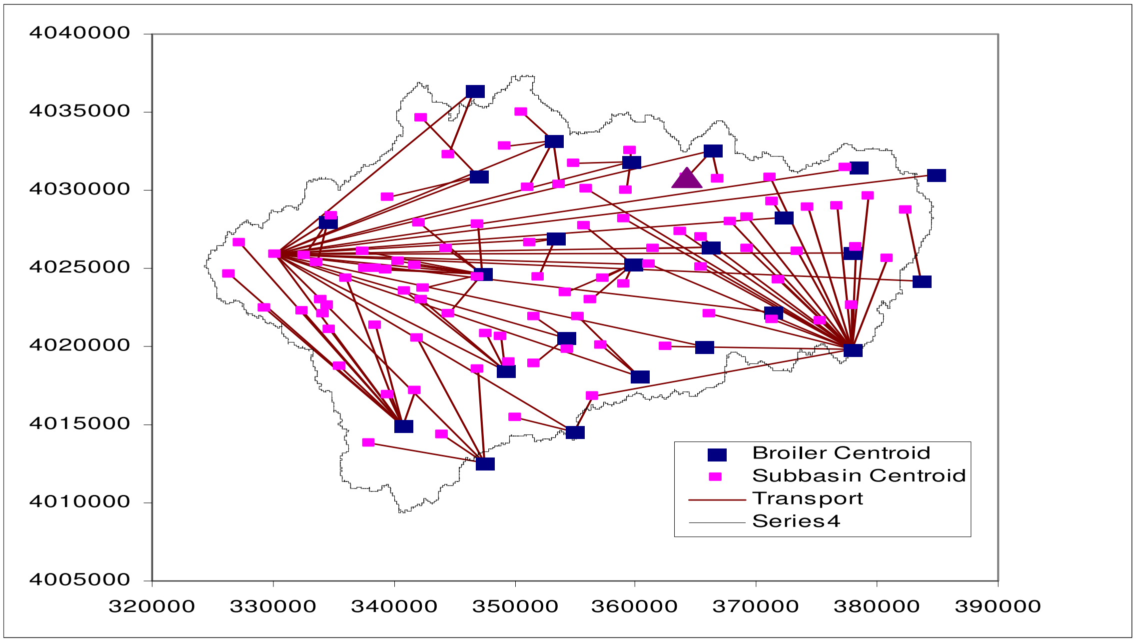


Figure 39 Litter Shipment Pattern Given Phosphorus Loss Target of 20,000 kg /yr.

The increase in the amount of litter shipped to the processing plant is shown by the increased intensity of transportation lines that directly connect chicken farm centroids to the processing plant at Jay, Oklahoma as the allowable level of phosphorus runoff is reduced by 25 percent and 50 percent, respectively. As indicated previously, reducing the phosphorus loading target from 40 to 20 tons per year without imposing an upper limit on the deviation above the target, increased the amount of poultry litter transported from the chicken farms in the Eucha-Spavinaw watershed to the possible litter-to-energy processing plant at Jay, Oklahoma by about 140 percent. This is a rise from the initial simulated amount of about 35,000 to 84,000 kilograms per year. However, the imposition of an upper limit on phosphorus deviation above the set loading target for the watershed resulted in further increase of the optimal amount of poultry litter transported to the litter-to-energy processing plant from chicken farm centroids in the entire watershed. Larger increments in optimal amount of litter shipped were predicted at phosphorus loading targets of 40 tons (litter shipments rose by 56 percent) and 30 tons (litter shipments rose by 23 percent) per year compared to that obtained when the loading target was set at 20 tons (litter shipments increased by 1 percent) per year for the entire watershed. The overall phosphorus loss in the watershed declined because most of the poultry litter was shipped to the litter-to-energy processing plant and the rest was hauled and applied on pasture land in the subbasins within the Eucha-Spavinaw watershed.

Scenario II: Land Application of Alum-Treated Litter and Trading Option

In this option we examined the effects of limiting total phosphorus runoff for the Eucha-Spavinaw watershed to 40, 35, 30, 25, and 20 tons per year on optimal litter and pasture management systems with an option to use alum-treated litter on pastures as well as hauling litter within the watershed and to a possible litter-to-energy power plant located at Jay, Oklahoma. The basic assumptions were that Alum would reduce soluble phosphorus runoff by 75 percent. Also, since Alum would reduce nitrogen loss in the poultry house the average ton of litter would contain 34 kg of nitrogen rather than 30 kg for untreated litter. Hence only 88 percent as much litter had to be applied for the same amount of nitrogen. Alum-treated poultry litter was assumed to impose an additional cost of $5.00 per ton to farmers undertaking the agricultural activities in HRUs where alum-treated litter was applied. Poultry litter was applied to pastures at levels consistent with meeting the nitrogen requirement of the crop.

Optimal Grazing Management Practices for the Watershed

Table 22 below shows the codes and description of management activities that entered the solution set at different levels of soluble phosphorus runoff. The addition of the possibility to use alum-treated litter on pastures reduced the number of optimal management practices in the solution set at all levels of soluble phosphorus runoff for the Eucha-Spavinaw watershed. No commercial nitrogen was applied to pastures in this scenario. Poultry litter was applied to pastures at levels consistent with meeting the nitrogen requirement of the crop. There are only 2 pasture management systems in the solution set (codes 46 and 56) that do not involve the use of alum-treated poultry litter.

Table 22. Optimal Management Activities Given Alum-Treated Litter Option.

BMP Code	Poultry Litter Applied (tons/ha)	Elemental Nitrogen Applied (kg/ha)	Minimum Biomass Maintained During Grazing (tons/ha)	Stocking Rate (AU/ha)
46	4	0	1.6	1.26
56	6	0	2.0	1.26
61	1.765	0	1.1	0.63
66	1.765	0	1.1	1.00
76	3.529	0	1.6	0.63
81	3.529	0	1.6	1.00
86	3.529	0	1.6	1.26
91	5.294	0	2.0	0.63
96	5.294	0	2.0	1.00
101	5.294	0	2.0	1.26

Table 23, Table 24, and Table 25 show the range of pasture management practices that entered the solution when the annual mean soluble phosphorus runoff was limited to 40, 30, and 20 tons per year respectively, with mean phosphorus deviation limits above target varied from 10 to 2 tons per year. When the soluble phosphorus runoff was limited to 40 tons per year, 21,000 ha of land was allocated to pasture that received 4 tons of untreated litter per ha, stocked at 1.26 AU/ha and the biomass maintained during grazing was 1,600kg/ha. Approximately 15,000 ha of pastureland were allocated to management 96. This management practice represented application of alum-treated poultry litter at the rate of about 5 tons per ha, with cattle put on pasture at the stocking rate of 1.00 AU/ha. Biomass maintained during grazing was estimated at 2,000kg/ha. However, as the soluble phosphorus runoff limit was reduced to 20 tons per year, more land was moved out of management 46 and 56 (both use untreated litter) and allocated largely to management systems 96, 81 and 66 in that order. All these three management systems that came into the solution set represented the use of alum-treated litter, maintaining at least 1,600kg/ha of biomass during grazing and a stocking rate of 1.00 AU/ha.

Table 23. Comparison of Optimal Management Practices (Ha) When Maximum Average Soluble Phosphorus Target is 40 Mg / year as Average P Loss Deviations Above the Mean were Reduced from 10 Mg to 2 Mg Per Year.

Optimal Management Practice					Deviation Above Maximum 40Mg Phosphorus Loss (Mg)				
Code	PL	Nit.	MB	SR	10	8	6	4	2
	Mg	kg			Hectares Where Management Practice was Optimal				
46	4.0	0	M	H	20774	18873	14027	12009	4583
56	6.0	0	H	H	5473	6173	4954	3020	1025
61	1.8	0	L	L	0	0	0	0	0
66	1.8	0	L	M	0	0	0	0	58
76	3.5	0	M	L	0	0	0	0	350
81	3.5	0	M	M	0	0	0	1303	11715
86	3.5	0	M	H	0	0	0	0	53
91	5.3	0	H	L	0	0	316	1174	21029
96	5.3	0	H	M	15208	16797	22833	23951	21029
101	5.3	0	H	H	1647	1611	1689	1445	3394
Ave. P Loss (Mg/yr)					40	40	36	32	22
Ave. P. Deviations(Mg/yr)					8.4	8.0	6.0	4.0	2.0

** Abbreviations used: PL=Poultry Litter Applied (Mg/ha); Nit. = Commercial Nitrogen Applied (kg/ha); MB= Minimum Biomass (L=1.1,M=1.6,H=2.0 Mg/ha); SR = Stocking Rate (L=.63, M=1.0, H=1.26 Animal Units/ha). See Table 22 for the best management practice (BMP) associated with this BMP code.*

Table 24. Comparison of Optimal Management Practices (Ha) When Maximum Average Soluble Phosphorus Target is 30 Mg / year as Average P Loss Deviations Above the Mean were Reduced from 10 Mg to 2 Mg Per Year.

Optimal Management Practice					Deviation Above Maximum 30Mg Phosphorus Loss (Mg)				
Code	PL	Nit.	MB	SR	10	8	6	4	2
	Mg	kg			Hectares Where Management Practice was Optimal				
46	4.0	0	M	H	14724	14724	10012	7314	71
56	6.0	0	H	H	1284	1284	1821	720	176
61	1.8	0	L	L	0	0	0	0	350
66	1.8	0	L	M	0	0	0	0	1433
76	3.5	0	M	L	0	0	0	125	797
81	3.5	0	M	M	36	36	34	1468	21696
86	3.5	0	M	H	0	0	0	32	1799
91	5.3	0	H	L	0	0	316	1050	226
96	5.3	0	H	M	24270	24270	27862	29607	16291
101	5.3	0	H	H	2739	2739	3144	2908	108
Ave. P Loss (Mg/yr)					30	30	30	26	18
Ave. P. Deviations(Mg/yr)					6.5	6.5	6.0	4.0	2.0

** Abbreviations used: PL=Poultry Litter Applied (Mg/ha); Nit. = Commercial Nitrogen Applied (kg/ha); MB= Minimum Biomass (L=1.1,M=1.6,H=2.0 Mg/ha); SR = Stocking Rate (L=.63, M=1.0, H=1.26 Animal Units/ha). See Table 22 for the best management practice (BMP) associated with this BMP code.*

Table 25. Comparison of Optimal Management Practices(Ha) When Maximum Average Soluble Phosphorus Target is 20 Mg / year as Average P Loss Deviations Above the Mean were Reduced from 10 Mg to 2 Mg Per Year.

Optimal Management Practice					Deviation Above Maximum 20Mg Phosphorus Loss (Mg)				
Code	PL	Nit.	MB	SR	10	8	6	4	2
	Mg	kg			Hectares Where Management Practice was Optimal				
46	4.0	0	M	H	1676	1676	1676	402	0
56	6.0	0	H	H	375	375	375	283	0
61	1.8	0	L	L	7	7	7	7	0
66	1.8	0	L	M	0	0	0	58	20679
76	3.5	0	M	L	176	176	176	125	0
81	3.5	0	M	M	9258	9258	9258	10396	17852
86	3.5	0	M	H	848	848	848	1702	0
91	5.3	0	H	L	647	647	647	1277	0
96	5.3	0	H	M	29682	29682	29682	28191	4605
101	5.3	0	H	H	252	252	252	595	0
Ave. P Loss (Mg/yr)					20	20	20	20	13
Ave. P. Deviations(Mg/yr)					4.2	4.2	4.2	4.0	2.0

** Abbreviations used: PL=Poultry Litter Applied (Mg/ha); Nit. = Commercial Nitrogen Applied (kg/ha); MB= Minimum Biomass (L=1.1,M=1.6,H=2.0 Mg/ha); SR = Stocking Rate (L=.63, M=1.0, H=1.26 Animal Units/ha). See Table 22 for the best management practice (BMP) associated with this BMP code.*

Optimal Amount of Poultry Litter Used As Fertilizer on Pastures

The option of using alum-treated poultry litter on pastures lead to a drastic reduction in the litter shipments from the watershed to the possible litter-to-energy power plant in Jay, Oklahoma. Most of the poultry litter produced in the watershed was hauled between subbasins within the watershed and applied on land as crop fertilizer. Figure 40 and Figure 41 below show the effect of alternative annual soluble phosphorus runoff targets and phosphorus deviation limits above target on optimal poultry litter use in the Eucha-Spavinaw watershed. As the maximum allowable total annual soluble phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year, the amount of poultry litter applied on pastures in the entire watershed declined from about 46,000 to 17,000 tons per year (approximately 63 percent drop in litter applied as fertilizer) or from an average of about 1.3 to 0.5 tons per hectare). The imposition of an upper limit on phosphorus deviation above the set phosphorus loading target for the watershed resulted in further decline of the optimal amount of poultry litter applied in the entire watershed depending on the set soluble phosphorus loading target and the tolerance or maximum allowable phosphorus runoff deviation above the specified phosphorus runoff target. The lower the maximum allowable phosphorus runoff deviation above a specified target the larger the decrease in the estimated amount of poultry litter applied in the watershed. Also, the lower the maximum allowable total annual soluble phosphorus runoff target for the watershed the lesser the decrease in the estimated amount of poultry litter applied in the watershed for a given upper phosphorus runoff deviation above the specified target.

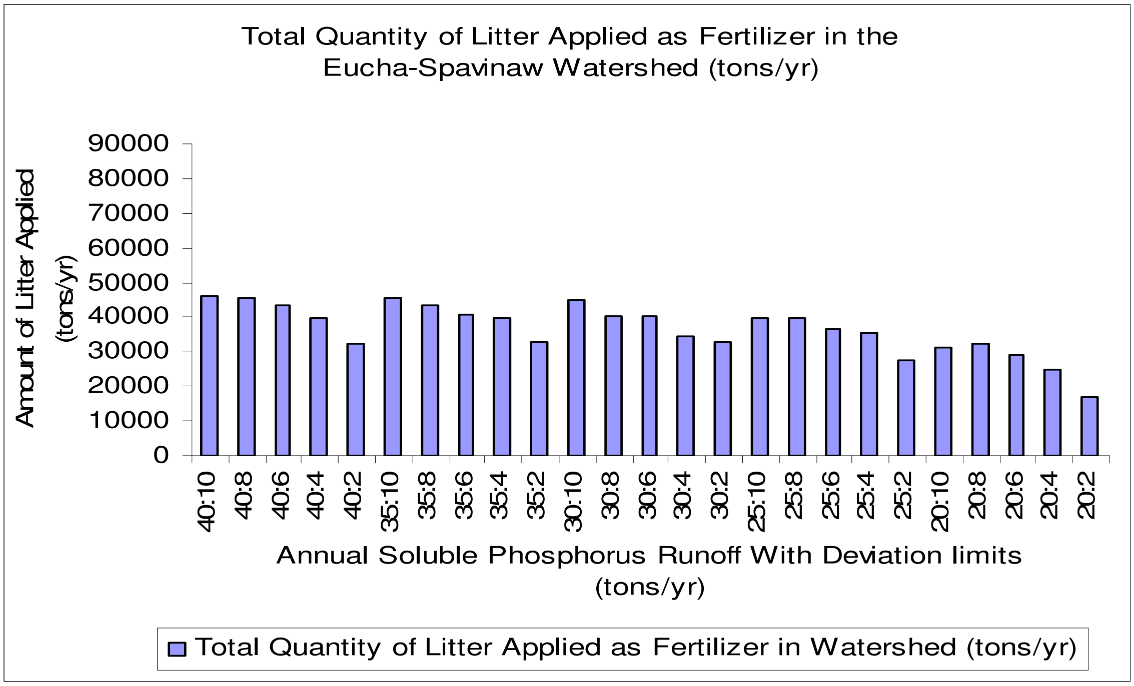


Figure 40 Quantity of Poultry Litter Applied as Fertilizer in the Watershed (tons/yr).

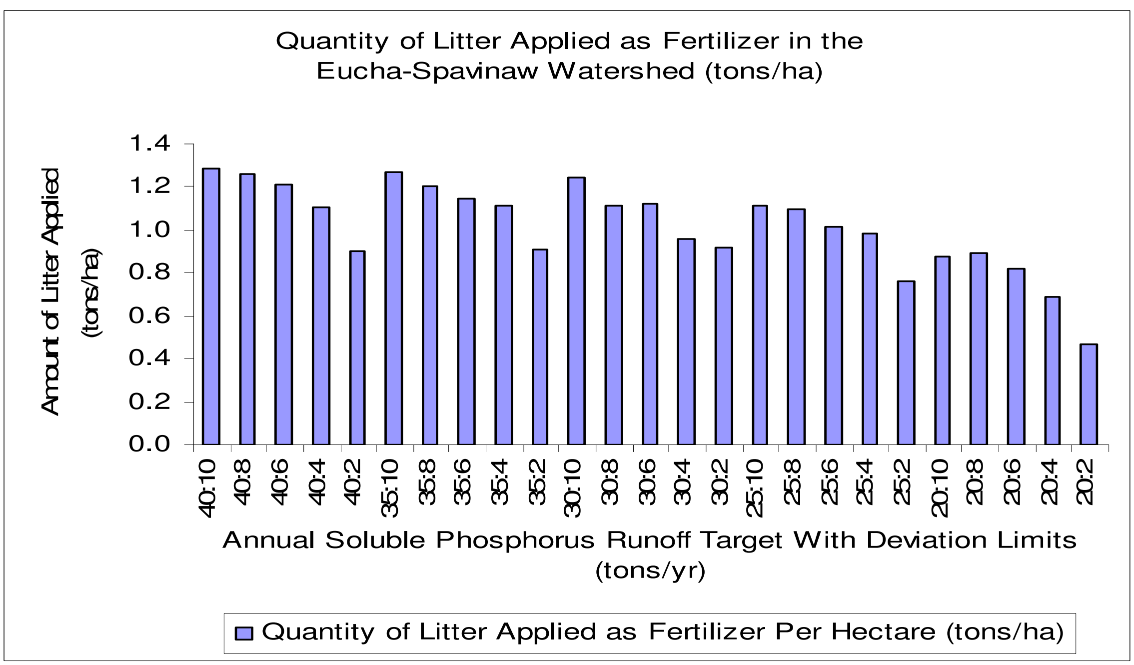


Figure 41 Quantity of Poultry Litter Applied as Fertilizer in the Watershed (tons/ha)

Optimal Poultry Litter Shipments to Litter-to-Energy Power Plant

Figure 42 below shows the effect of alternative annual soluble phosphorus runoff targets and phosphorus deviation limits above target on optimal litter shipments from chicken farm centroids in the watershed to the possible litter-to-energy processing plant with and without upper phosphorus deviation limits above target. As the allowable total annual soluble phosphorus loading for the entire watershed was reduced from 40 to 20 tons per year, the optimal amount of poultry litter shipped to the litter-to-energy processing plant (located at Jay, Oklahoma) rose depending on the tolerance or allowed deviation above the specified average total phosphorus target. As the mean annual soluble phosphorus loading target was reduced from 40 to 20 tons per year, the optimal amount of poultry litter shipped to the litter-to-energy processing plant increased from about 43,000 to 72,000 tons per year.

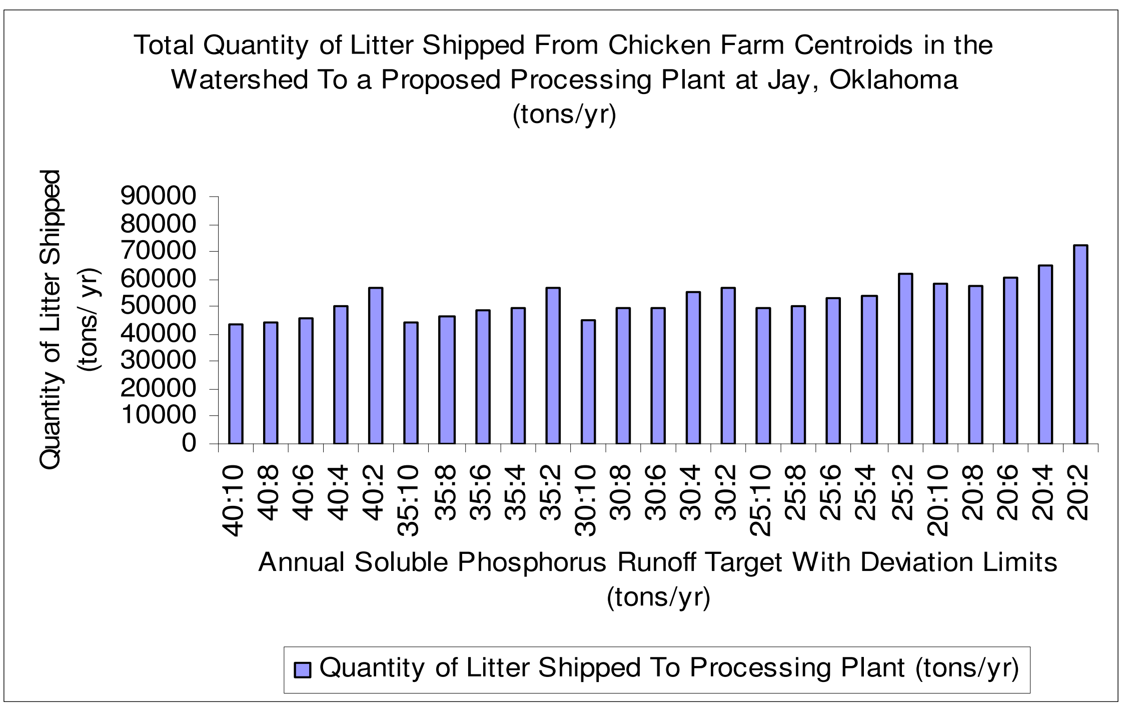
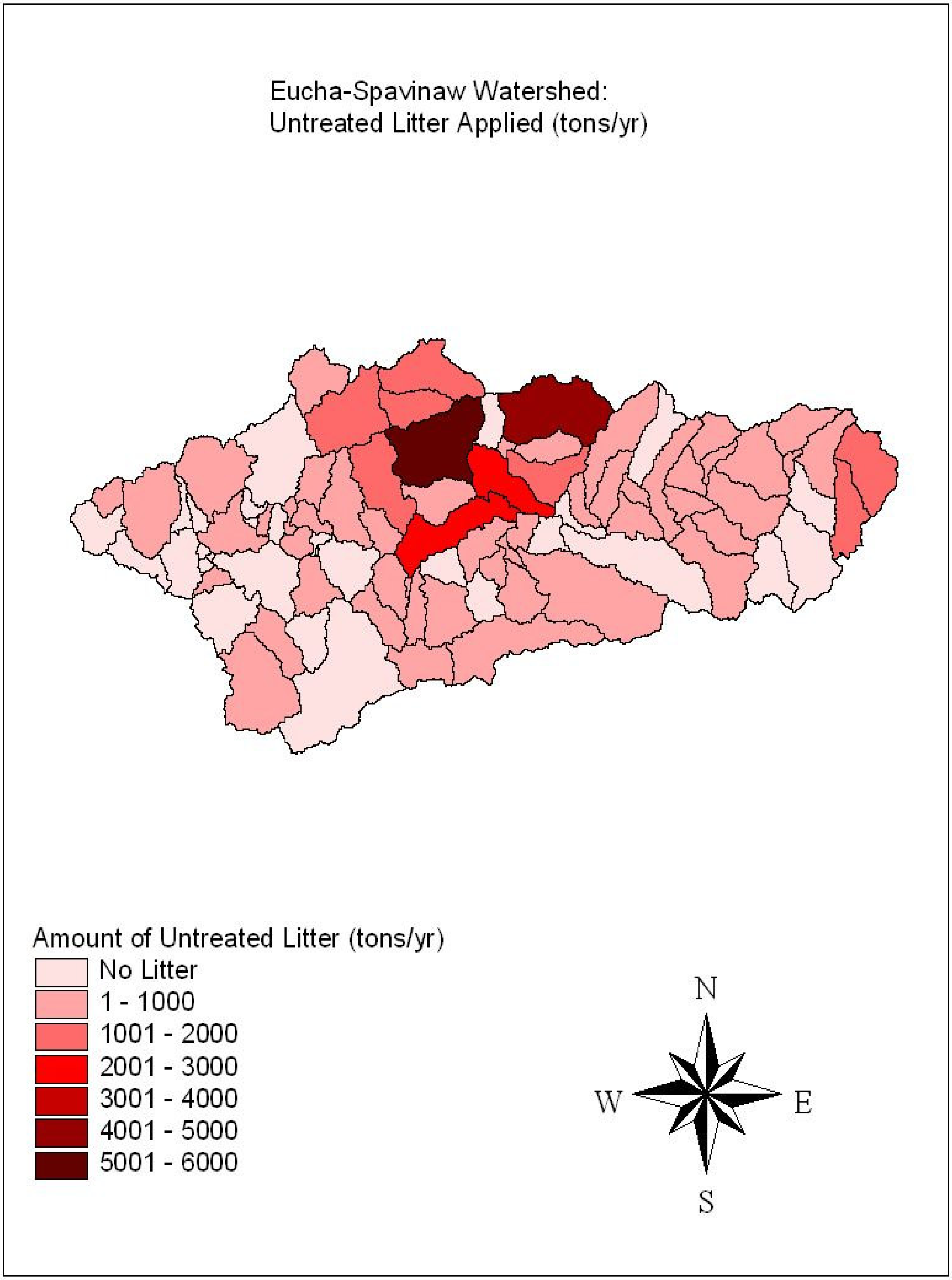


Figure 42 Quantity of Litter Shipped From Chicken Farm Centroids to Energy Plant.

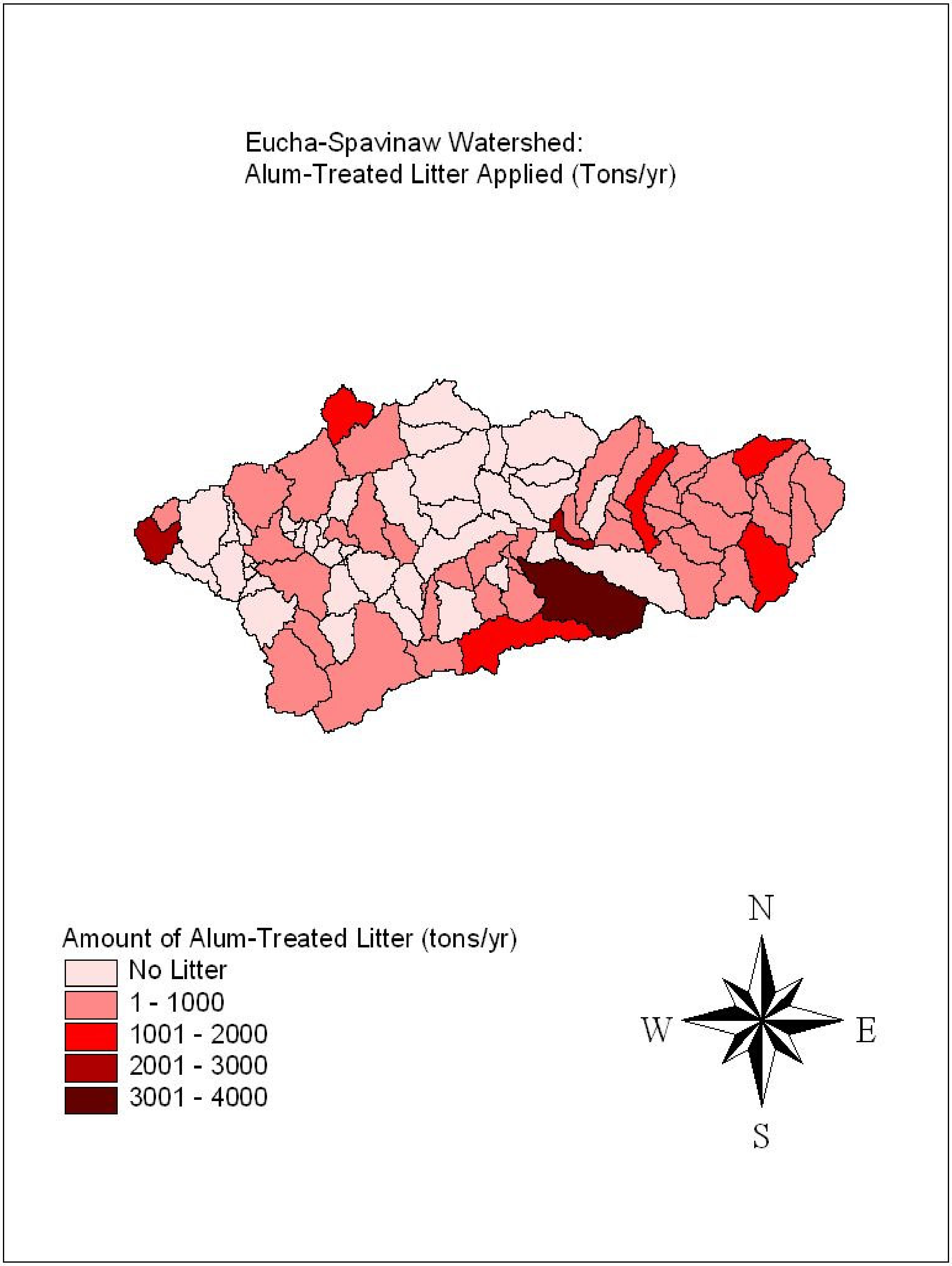
The imposition of an upper limit on phosphorus deviation of not more than 4 tons per year above the phosphorus loading target of 20 tons per year for the watershed resulted in further increase of the optimal amount of poultry litter shipped to the processing plant. The lower the maximum allowable phosphorus runoff deviation above a specified target the larger the increase in the estimated amount of poultry litter shipped to the litter- to-energy processing plant. However, the lower the maximum allowable total annual soluble phosphorus runoff target for the watershed the lesser the increase in the amount of poultry litter shipped to the litter-to-energy processing plant resulting from the imposition of upper phosphorus runoff deviations above the specified target.

Figure 43 to Figure 50 below show the optimal allocation of poultry litter applied on pastures by subbasin as the total soluble phosphorus runoff limit was reduced from the base phosphorus load (40 Mg per year with maximum allowable phosphorus deviation above limit of 10 Mg/yr) to 30Mg, 25 Mg, and 20Mg/yr with maximum allowable phosphorus deviations above limit set to 6Mg, 6Mg, and 4Mg per year, respectively. These phosphorus loads were selected because they represented mean phosphorus load targets for the watershed at which there was approximately a 20-25 percent chance of exceeding the mean phosphorus target in any particular year. As can be seen from Figure 43 to Figure 50, different subbasins responded differently to the alternative soluble phosphorus runoff standards that were imposed. For instance, Figure 43 and Figure 44 show that some subbasins (subbasins 50, 51, 52, 75, 76, 77, 85, 86, 87, and 88) in the watershed did not apply poultry litter on pastures when the phosphorus runoff was limited to 40 tons per year. A total of 37,220 tons of non-Alum-treated poultry litter was applied as fertilizer on pastures. Twenty-one of the 90 subbasins used non-Alum-treated poultry litter only (See Appendix for more details on the responses of individual subbasins to various phosphorus loss targets). Sixty-five percent of the untreated poultry litter was shared between 11 subbasins. Subbasins 21 and 5 used about 14 and 11 percent of untreated poultry litter, respectively, while subbasins 8 and 63 applied about 6 percent each. Subbasins 1 and 13 used approximately 5 percent of the untreated poultry litter each. A total of 25,707 tons of Alum-treated poultry litter was applied on pastures as fertilizer. Fifty-eight percent of the Alum-treated litter was shared between 8 subbasins, with subbasins 54, 39, and 32 receiving about 15 percent, 10 percent, and 9 percent of the total supply of Alum-treated litter, respectively. Subbasins 72 and 44 received approximately 7 percent and 5 percent of Alum-treated litter respectively, while subbasins 3, 11, and 26 used about 4 percent each.



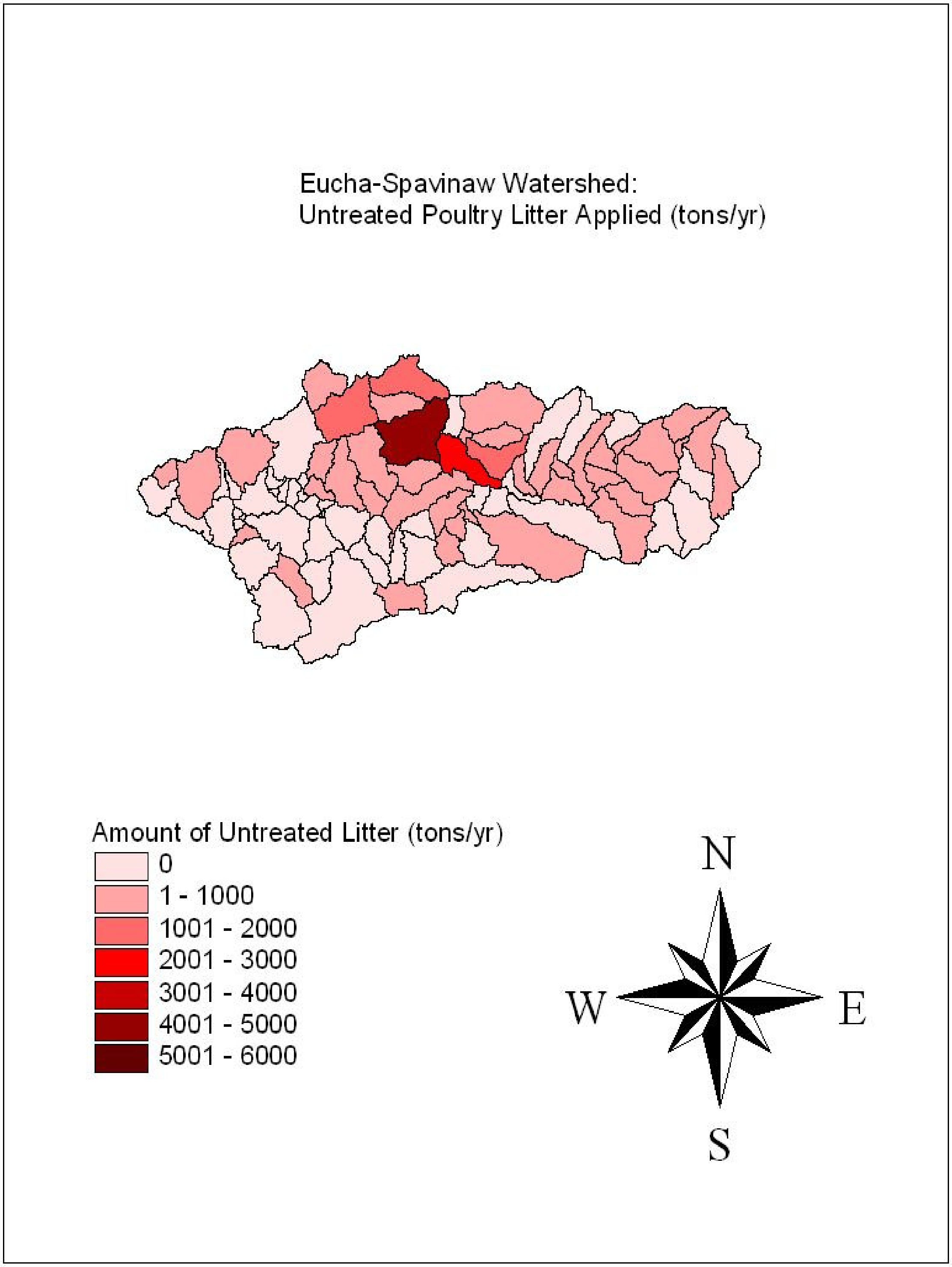
Source: Adapted from Storm et al. (2002).

Figure 43 Amount of Untreated Litter Required at P. Runoff Limit of 40 Mg and Deviation Above Mean Limit of not more than 10 Mg per year.



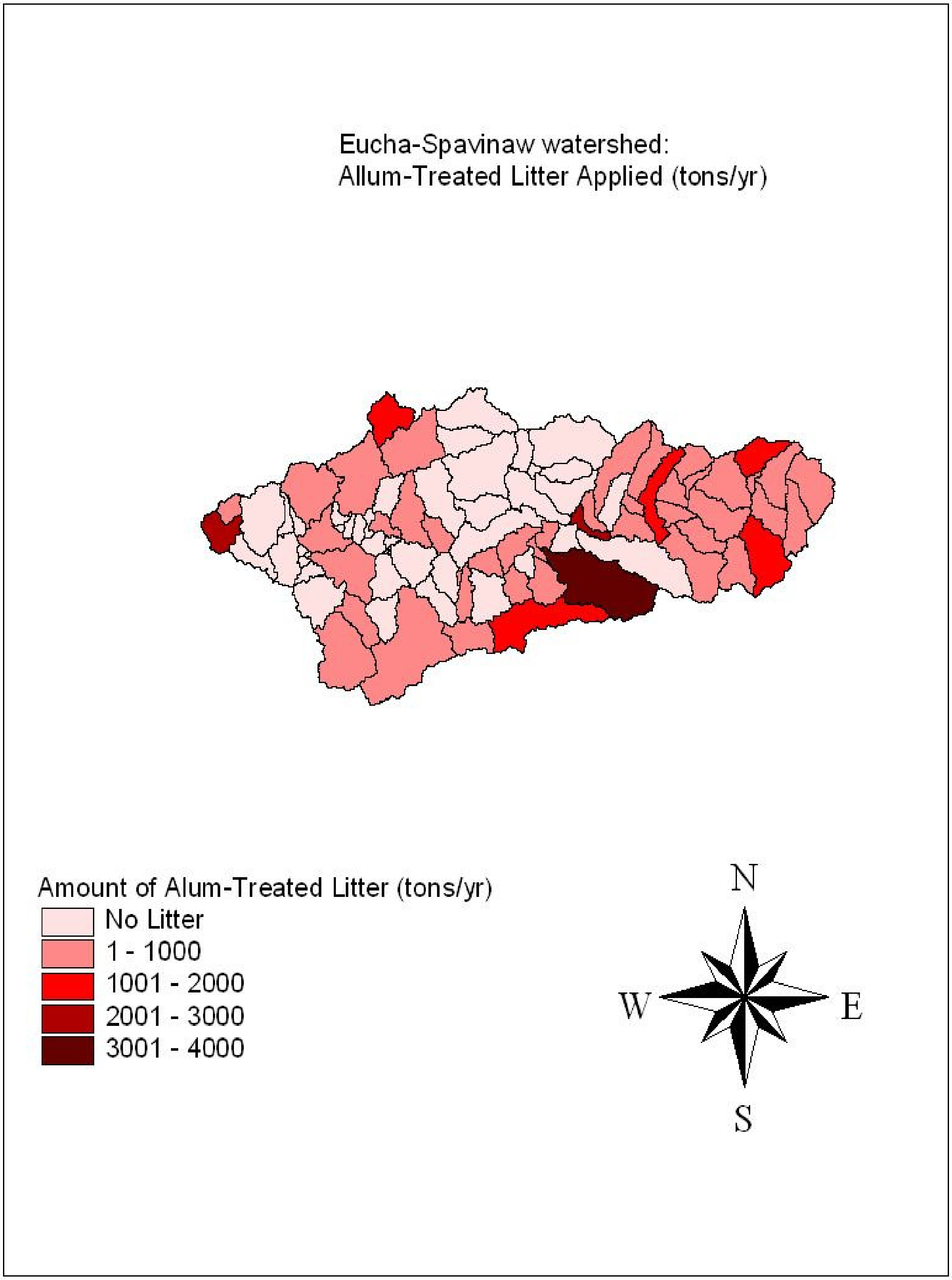
Source: Adapted from Storm et al. (2002).

Figure 44 Amount of Alum-Treated Litter Required at P. Runoff Limit of 40 Mg and Deviation Above Mean Limit of not more than 10 Mg per year.



Source: Adapted from Storm et al. (2002).

Figure 45 Amount of Untreated Litter Required at P. Runoff Limit of 30 Mg and Deviation Above Mean Limit of not more than 6 Mg per year.

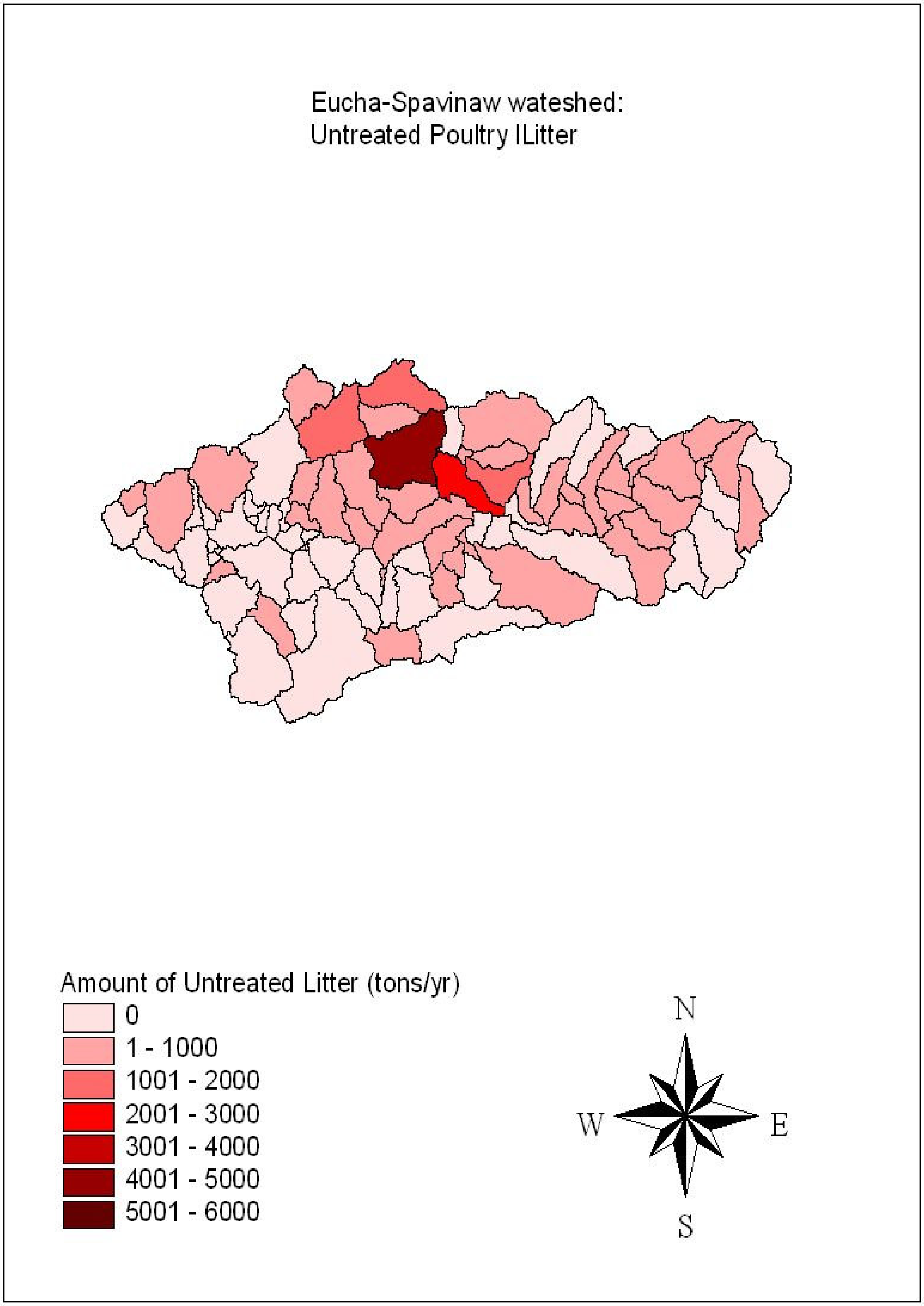


Source: Adapted from Storm et al. (2002).

Figure 46 Amount of Alum-Treated Litter Required at P. Runoff Limit of 30 Mg and Deviation Above Mean Limit of not more than 6 Mg per year.

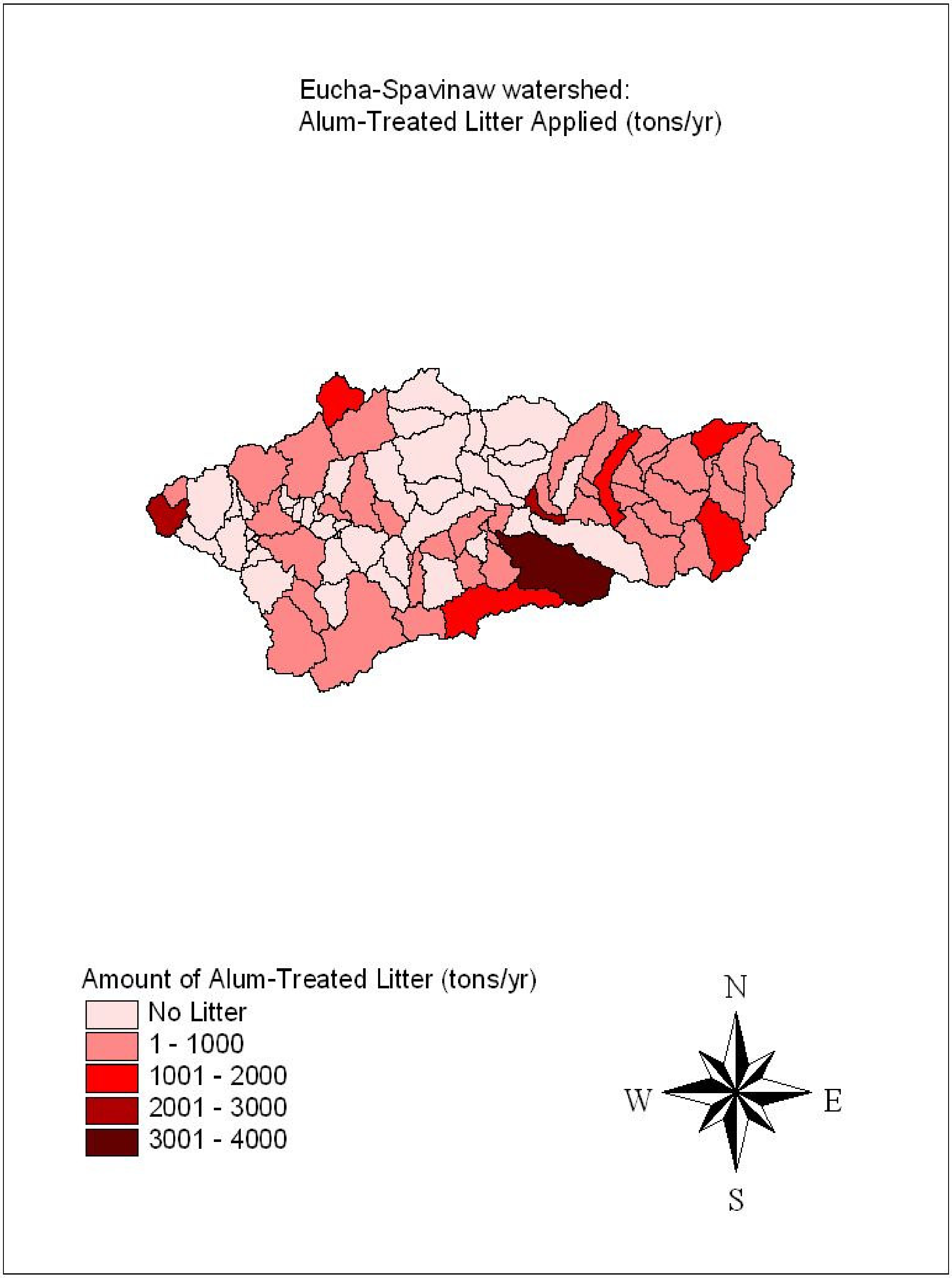
The amount of non-Alum-treated litter declined to 21,415 tons per year while that of Alum-treated poultry litter rose to 53,415 tons per year when the mean soluble phosphorus load was limited to 30 Mg / yr with a maximum allowable phosphorus deviation above limit of 6 Mg/yr as shown by Figure 45 and Figure 46 above. The distribution pattern remained relatively the same.

When the mean soluble phosphorus runoff limit was reduced to 25 Mg per year and the phosphorus deviation above limit reduced to 6 Mg per year, the amount of non-Alum-treated litter was further cut down to about 15,000 tons per year while the amount of Alum-treated poultry litter used rose significantly to about 65,000 tons per year (See Figure 47 and Figure 48 below. More details are provided in the Appendix.). Sixty-three percent of the untreated poultry litter was shared between 5 subbasins. Subbasins 21 used 31 percent of untreated litter, followed by subbasins 8 and 4 with a share of about 10 percent each. Subbasins 9 and 29 used approximately 7 percent and 6 percent of untreated poultry litter, respectively. 29 percent of the Alum-treated litter was shared between 5 subbasins, with subbasins 54, 13, and 5 receiving about 6 percent of Alum-treated litter each. Subbasins 40 and 72 applied about 5 percent each. Another 21 percent of Alum-treated litter was shared between 7 subbasins (subbasins 1, 14, 21, 29, 33, 67 and 71), each receiving about 3 percent. An additional 20 percent of Alum-treated litter was shared almost equally between 10 other subbasins. The rest of the remaining Alum-treated litter was used on pastures across the watershed in relatively smaller amounts per subbasin.



Source: Adapted from Storm et al. (2002).

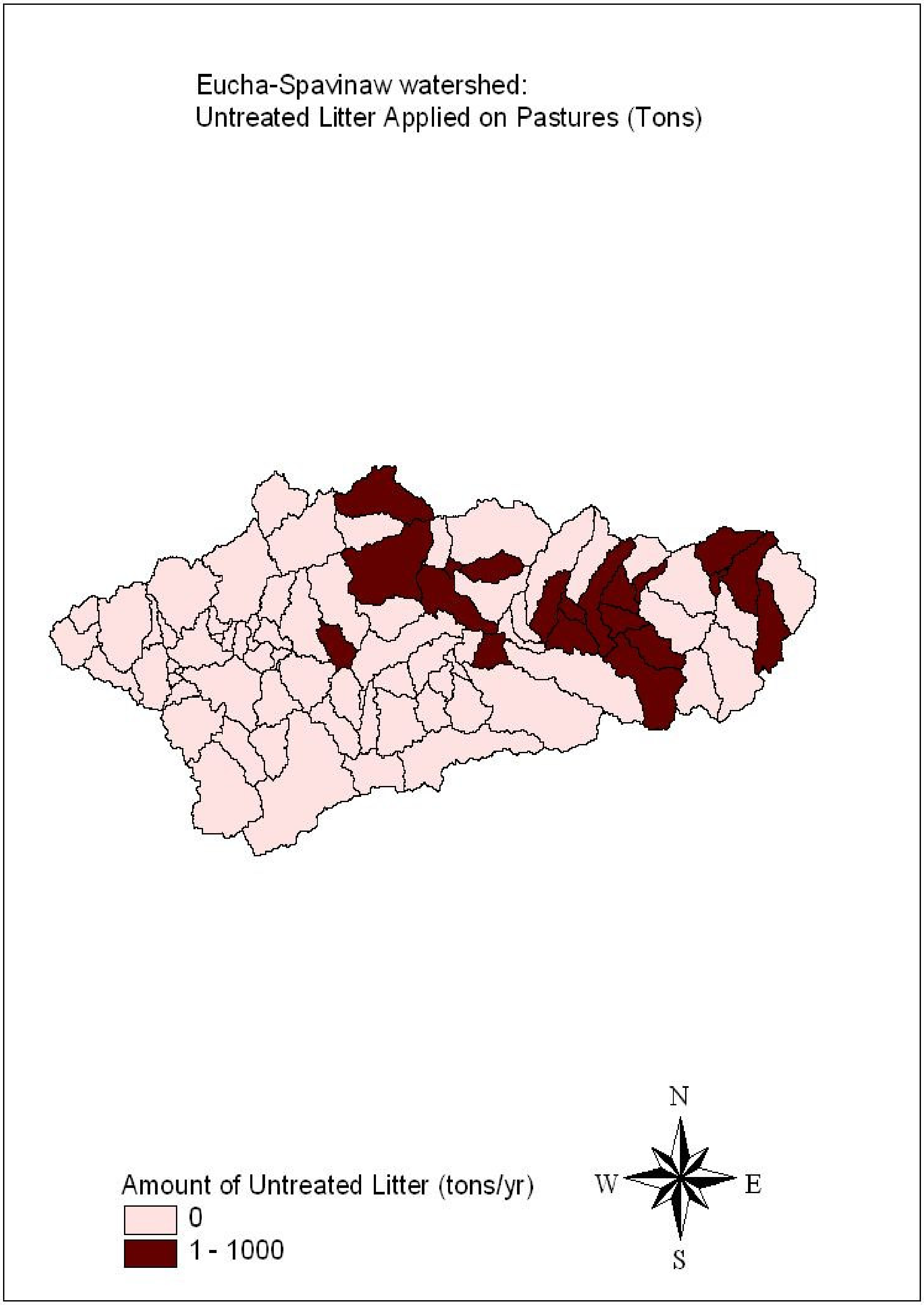
Figure 47 Amount of Untreated Litter Required at P. Runoff Limit of 25 Mg and Deviation Above Mean Limit of not more than 6 Mg per year.



Source: Adapted from Storm et al. (2002).

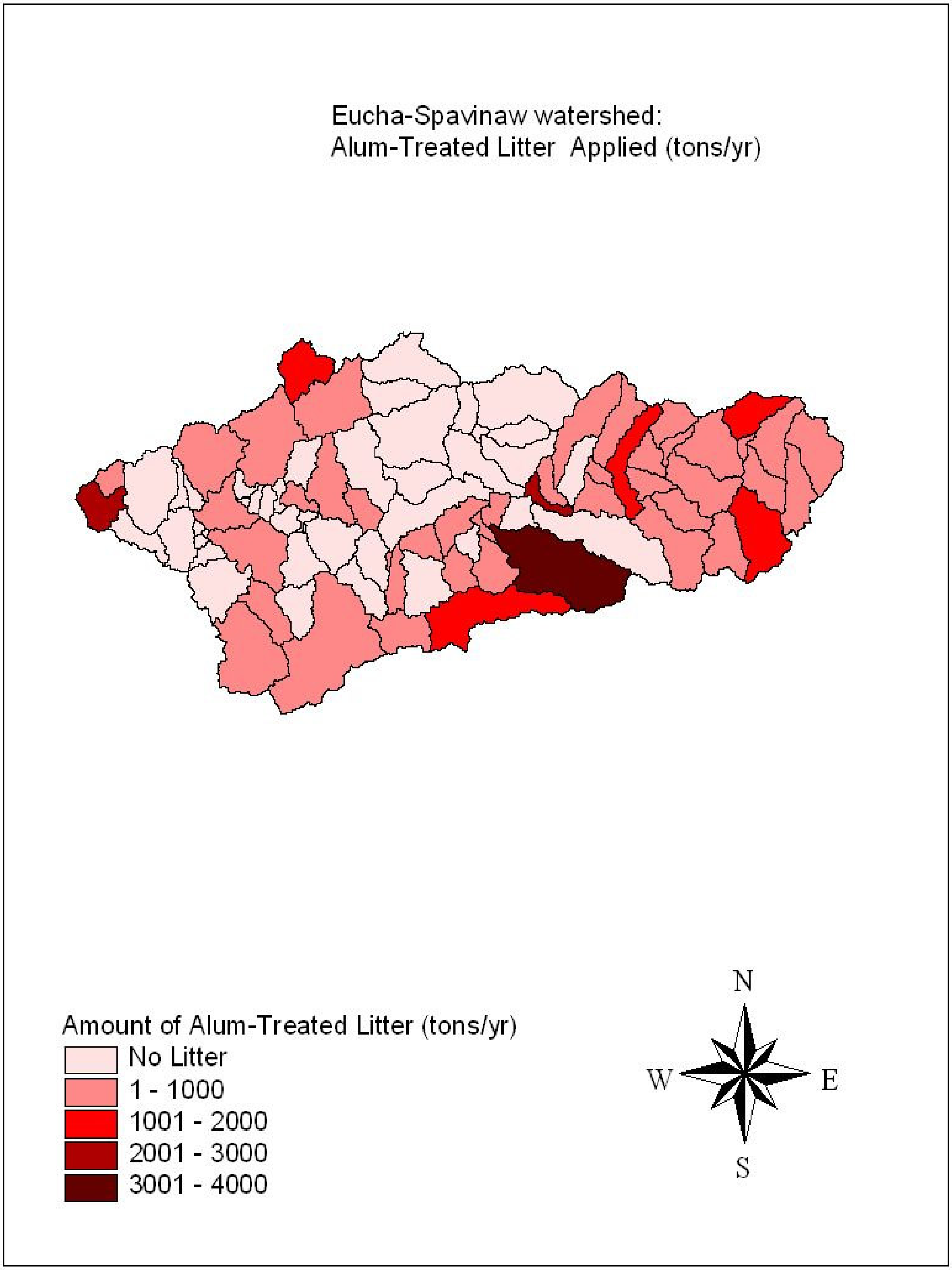
Figure 48 Amount of Alum-Treated Litter Required at P. Runoff Limit of 25 Mg and Deviation Above Mean Limit of not more than 6 Mg per year.

The amount of non-Alum-treated litter further declined to about 780 tons per year when the mean soluble phosphorus load was further limited to 20 Mg / yr with the maximum allowable phosphorus deviation above limit reduced to 4 Mg/yr. The quantity of Alum-treated poultry litter used increased significantly to approximately 86,000 tons per year (See Figure 49 and Figure 50. More details provided in the Appendix). . The distribution pattern for Alum-treated litter remained relatively the same. In the case of non-Alum-treated poultry litter, 29 percent was allocated to subbasin 14. Another 22 percent was allocated to subbasins 23 (8 percent) and 24 (14 percent), while Subbasin 1 used about 7 percent of the non-Alum-treated litter. Subbasins 21 and 37 received about 5 percent each. An additional 20 percent was shared almost equally between subbasins 8, 12, 15, and 26. The remaining 17 percent of non-Alum treated litter was shared thinly amongst the remaining few subbasins in the watershed.



Source: Adapted from Storm et al. (2002).

Figure 49 Amount of Untreated Litter Required at P. Runoff Limit of 20 Mg and Deviation Above Mean Limit of not more than 4 Mg per year.



Source: Adapted from Storm et al. (2002).

Figure 50 Amount of Untreated Litter Required at P. Runoff Limit of 20 Mg and Deviation Above Mean Limit of not more than 4 Mg per year.

Table 26 Aluminum Sulphate (Alum) Required by Optimal Management Practice at Various Mean Soluble Phosphorus Load Limits and Deviations Above Limit (tons/yr).

Mean Soluble Phosphorus Loss Limit (Mg/yr)	Deviation Above P. loss Limit (Mg/yr)	Optimal Management Practice and Management Variables							
		61	66	76	81	86	91	96	101
		1.8* LL0	1.8* LM0	3.5* ML0	3.5* MM0	3.5* MH0	5.3* HL0	5.3* HM0	5.3* HH0
		Amount of Aluminum Sulphate (Alum) Used (tons/yr)							
40	10	0	0	0	0	0	0	7,319	793
40	8	0	0	0	0	0	0	8,084	775
40	6	0	0	0	0	0	152	10,989	813
40	4	0	0	0	418	0	565	11,527	695
40	2	0	9	112	3,758	17	10,121	10,121	1,633
35	10	0	0	0	11	0	0	9,958	572
35	8	0	0	0	11	0	0	9,958	572
35	6	0	0	0	0	0	152	12,510	775
35	4	0	0	0	467	3	565	13,591	447
35	2	23	36	109	5,073	129	607	10,192	388
30	10	0	0	0	12	0	0	11,681	1,318
30	8	0	0	0	12	0	0	11,681	1,318
30	6	0	0	0	11	0	152	13,409	1,513
30	4	0	0	40	471	10	505	14,249	1,400
30	2	56	230	256	6,960	577	109	7,840	52
25	10	0	0	0	232	417	152	15,221	962
25	8	0	0	0	232	417	152	15,221	962
25	6	0	0	0	232	417	152	15,221	962
25	4	0	0	207	1,178	94	505	15,320	875
25	2	31	1,147	47	8,943	160	0	3,393	0
20	10	1	0	57	2,970	272	312	14,285	121
20	8	1	0	57	2,970	272	312	14,285	121
20	6	1	0	57	2,970	272	312	14,285	121
20	4	1	9	40	3,335	546	615	13,568	287
20	2	0	3,318	0	5,727	0	0	2,216	0

** Abbreviations used: Decimal number =Alum-Treated Poultry Litter Applied (1.8, 3.5, and 5.3 Mg/ha); First letter = Minimum Biomass (L=1.1, M=1.6,H=2.0 Mg/ha; Second letter = Stocking Rate (L=.63, M=1.0, H=1.26 Animal Units/ha); and the last number = Commercial Nitrogen Applied (kg/ha).*

Table 26 above shows that only eight management practices that involved Alum-treated litter were optimal (that is, management 61, 66, 76, 81, 86, 91, 96 and 101). These management practices represent the use of at least 1.8 Mg/ha of Alum-treated poultry litter. Generally, more Alum-treated litter was used as the mean annual soluble phosphorus load limit and deviations above the limit were reduced. Management practices 61, 66, and 76 used relatively less Alum-treated litter compared to other management practices in the optimal solution set. These management practices represent the use of 3.5Mg/ha or less of Alum-treated litter, low to medium biomass maintained during grazing (1600kg/ha or less) and low to medium stocking rates (1.0 AU/ha or less). Management practices 81, 91, 96 and 101 used more Alum-treated litter compared to other management practices in the optimal solution set. However, management practice 96 consistently used more Alum-treated litter at all levels of mean soluble phosphorus load limits investigated. This management practice represents the use of Alum-treated litter at the rate of 5.3 Mg/ha, the highest biomass maintained during grazing (2,000 kg/ha) and a medium stocking rate of 1.0 animal unit per hectare.

Total Annual Phosphorus Runoff from Pastures Under Alternative Mean Annual Soluble Phosphorus Runoff and Deviations Above Mean Phosphorus

Figure 51 below shows that phosphorus pollution in the watershed can be reduced to levels below the set annual soluble phosphorus runoff when the alum-treated poultry litter option is considered. Significant reductions in phosphorus runoff were achieved by varying expected phosphorus deviation above target at each soluble phosphorus level without reducing the annual soluble phosphorus runoff target. As the soluble phosphorus load limit was reduced from 40 to 20 tons per year, predicted phosphorus runoff from pastures declined from 40 to 12.5 tons per year. Soluble phosphorus runoff levels well below the expected annual phosphorus runoff target were obtained by varying only the phosphorus deviation limits above the specified target. Soluble phosphorus runoff levels from all soil types in the watershed significantly declined when alum-treated litter was used on pastures. Tonti and Nixa still produced the least amount of soluble phosphorus runoff whereas levels from Doniphan and Clarksville soils remained relatively higher.

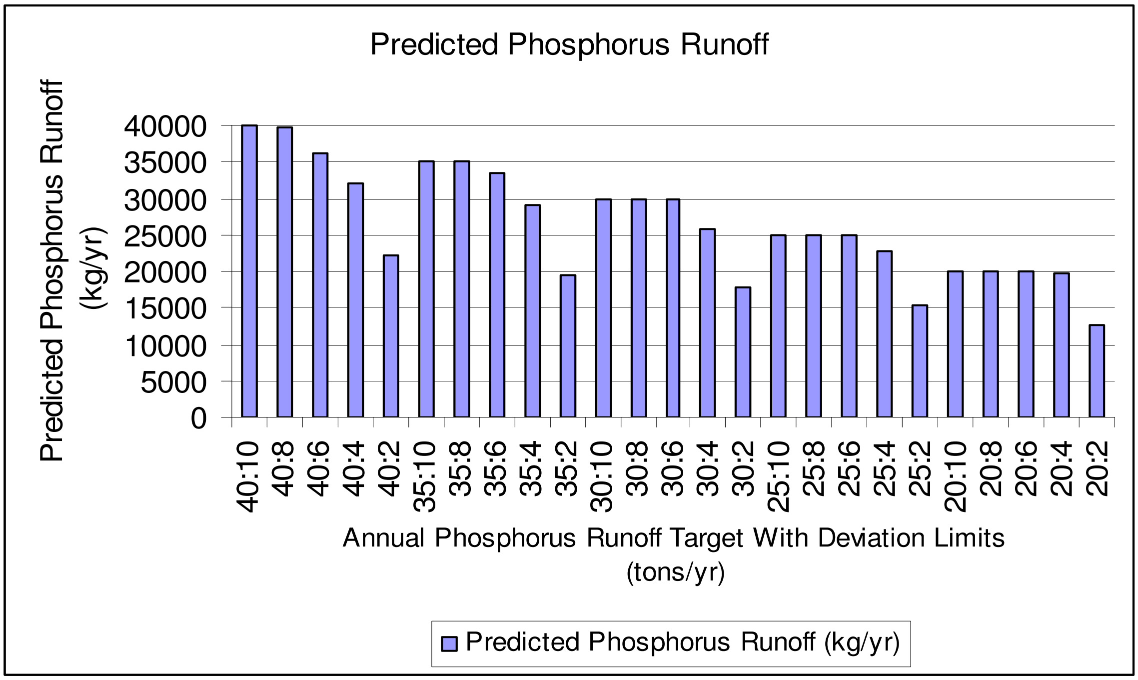


Figure 51 Predicted Annual Soluble Phosphorus Runoff from Pastures.

Figure 52 below shows the area-weighted average phosphorus runoff from pastures which declined from about 1.1 kg/ha given a phosphorus loss target of 40Mg per year to approximately 0.35 kg/ha when the phosphorus loss was limited to 20Mg per year.

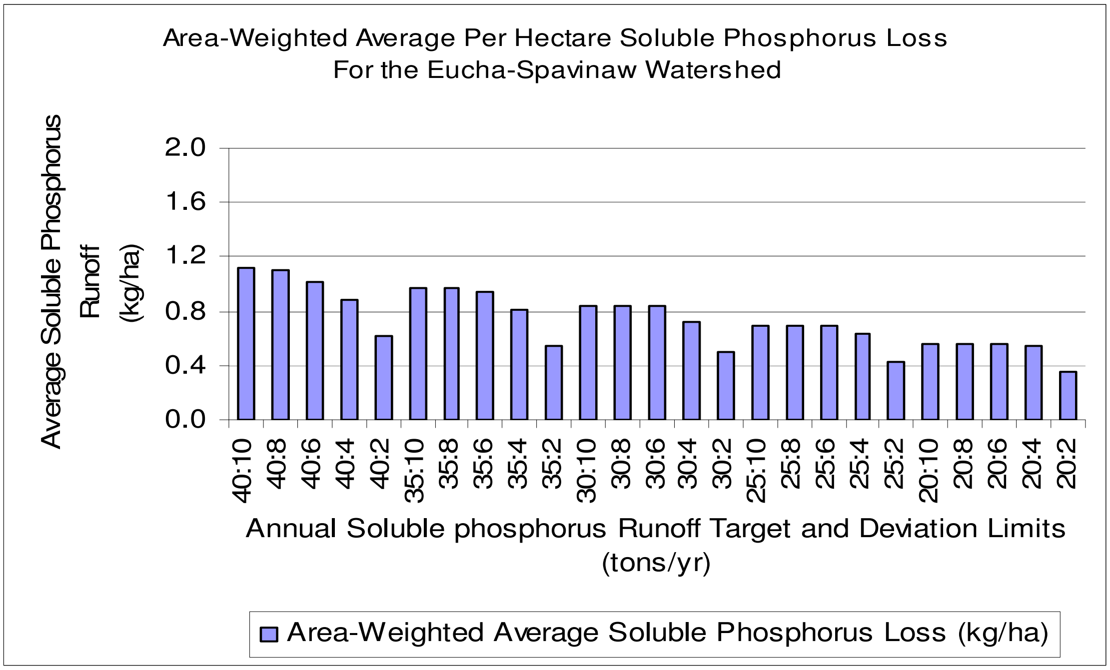


Figure 52 Weighted Average Soluble Phosphorus Loss From Pastures.

Optimal Amount of Phosphorus Loss For Major Soil Types

Figure 53 below shows the effect of alternative soluble phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of phosphorus runoff per hectare for the Clarksville soil in the Eucha-Spavinaw watershed. The estimated quantity of soluble phosphorus runoff per hectare for the Clarksville soil remained unchanged as the annual soluble phosphorus runoff target was reduced from 40 to 30 tons per year. However, the amount of phosphorus loss declined sharply at annual soluble phosphorus runoff targets lower than 30 tons per year. The amount of phosphorus runoff was estimated at 0.36 kg/ha when the annual soluble phosphorus runoff was limited to 20 tons per year given a maximum allowable mean phosphorus deviation above target of 2 tons per year. The imposition of phosphorus deviation limits above target yielded further and larger reductions when the phosphorus runoff was limited to 30 tons per year and above. The levels of soluble phosphorus runoff for this soil were significantly lower when Alum-treated poultry litter was allowed compared to levels obtained when farmers used untreated litter.

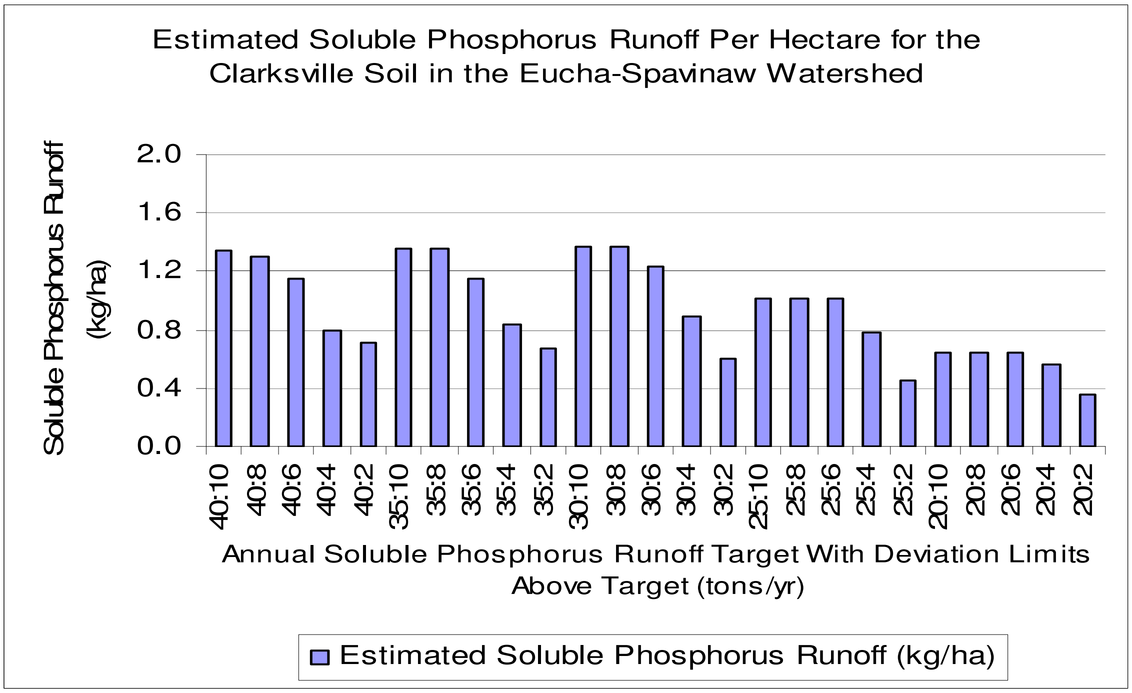


Figure 53 Estimated Soluble Phosphorus Runoff Per Hectare for the Clarkville Soil.

Figure 54 below shows the effect of alternative soluble phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of soluble phosphorus runoff per hectare for Nixa soil in the Eucha-Spavinaw watershed. The estimated quantity of soluble phosphorus runoff per hectare for the Nixa soil declined rapidly from about 1.3 kg/ha to 0.7 kg/ha as the annual soluble phosphorus runoff target was reduced from 40 to 30 tons per year and then remained relatively unchanged at lower limits of soluble phosphorus runoff.

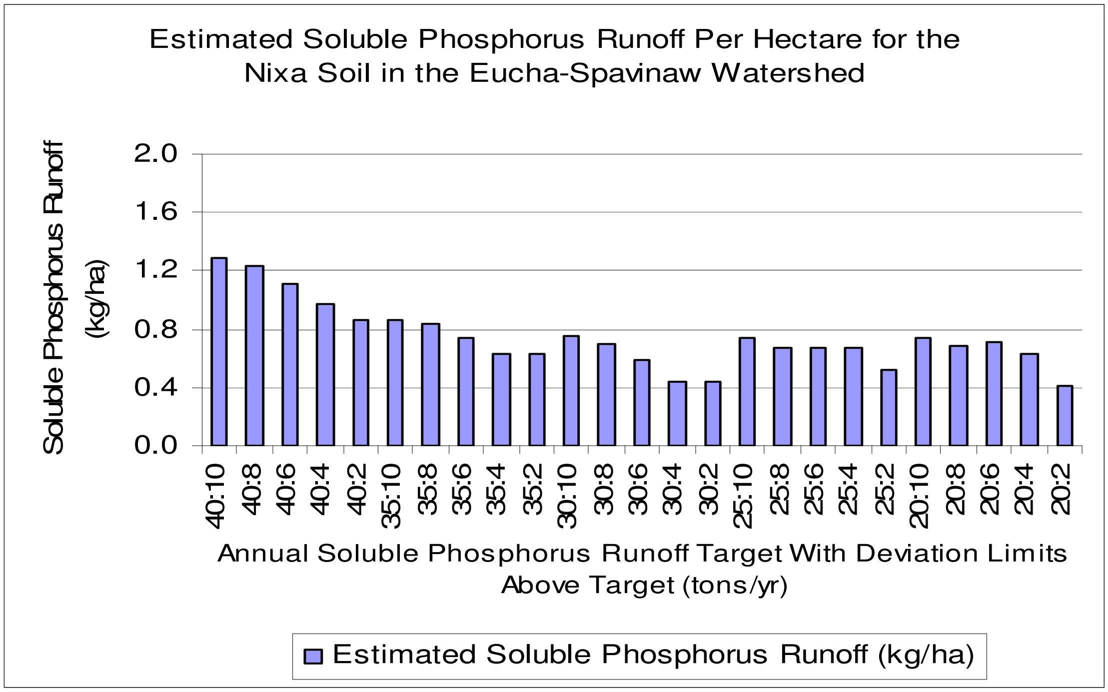


Figure 54 Estimated Soluble Phosphorus Runoff Per Hectare for the Nixa Soil.

The imposition of phosphorus deviation limits above target yielded further and larger reductions when the soluble phosphorus runoff was limited to 30 tons per year and above. The levels of soluble phosphorus runoff for this soil were significantly lower when Alum-treated poultry litter was allowed compared to levels obtained when farmers used untreated litter. However, soluble phosphorus runoff levels for the Nixa soil remained relatively lower than those of the Clarksville soil as the total soluble phosphorus limit for the watershed was reduced from 40 to 25 tons per year.

Figure 55 below shows the effect of alternative soluble phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of soluble phosphorus runoff per hectare for the Newtonia soil in the Eucha-Spavinaw watershed. The estimated quantity of soluble phosphorus runoff per hectare for the Newtonia soil declined steadily from about 0.76 kg/ha to 0.65 kg/ha as the annual soluble phosphorus runoff target was reduced from 40 to 30 tons per year and then declined rapidly at lower limits of phosphorus runoff till it reached 0.15 kg/ha when soluble phosphorus runoff was limited to 20 tons per year. The imposition of phosphorus deviation limits above target yielded further and larger reductions when the soluble phosphorus runoff was limited to 30 tons per year and above. The levels of phosphorus runoff for this soil were also significantly lower when Alum-treated poultry litter was allowed compared to levels obtained when farmers used untreated litter. The levels of soluble phosphorus runoff per hectare for the Newtonia soil were significantly lower than those attained in both Nixa and Clarksville soils at lower soluble phosphorus runoff limits.

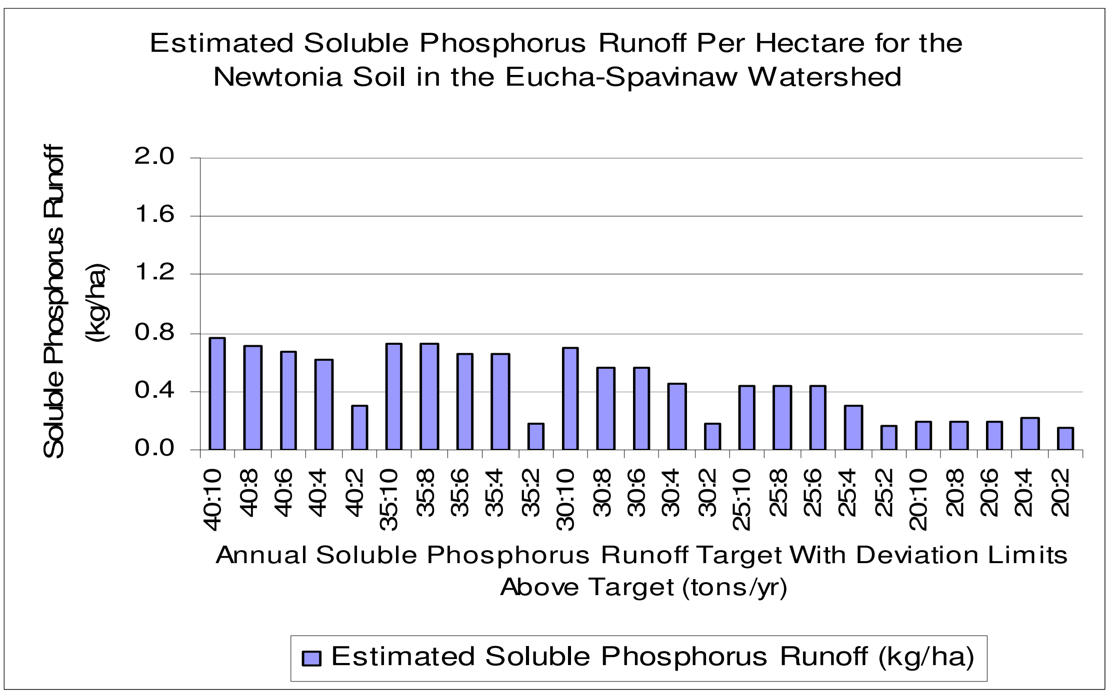


Figure 55 Estimated Soluble Phosphorus Runoff Per Hectare for the Newtonia Soil.

Figure 56 below shows the effect of alternative soluble phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of soluble phosphorus runoff per hectare for the Tonti soil in the Eucha-Spavinaw watershed. The estimated quantity of phosphorus runoff per hectare for the Tonti soil declined steadily from about 0.48 kg/ha to 0.13 kg/ha as the annual soluble phosphorus runoff target was reduced from 40 to 30 tons per year and then remained relatively unchanged at lower limits of phosphorus runoff. However, the amount of soluble phosphorus runoff per hectare for the Tonti soil remained lower than levels attained in the other soils at phosphorus runoff limits of 25 tons per year and above. When the total soluble phosphorus runoff for the watershed was limited to 20 tons per year, the amount of soluble phosphorus runoff per hectare from the Tonti soil was twice that of the Newtonia soil, and about half of the amounts attained in both the Nixa and Clarksville soils. Nonetheless, the levels of soluble phosphorus runoff for the Tonti soil were also significantly lower when Alum-treated poultry litter was allowed compared to levels obtained when farmers used untreated litter as crop fertilizer.

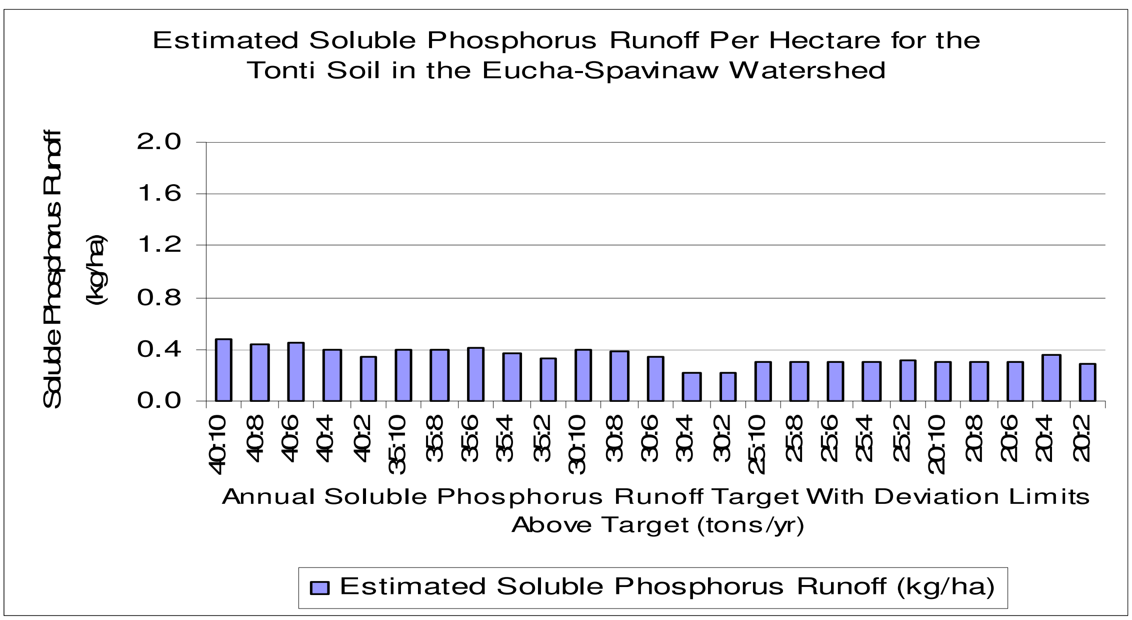


Figure 56 Estimated Soluble Phosphorus Runoff Per Hectare for the Tonti Soil.

Figure 57 below shows the effect of alternative soluble phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of phosphorus runoff per hectare for the Captina soil in the Eucha-Spavinaw watershed. The estimated quantity of soluble phosphorus runoff per hectare for the Captina soil declined rapidly from about 1.0 kg/ha to 0.70 kg/ha as the annual soluble phosphorus runoff target was reduced from 40 to 30 tons per year and then declined steadily at lower limits of soluble phosphorus runoff till it reached 0.50 kg/ha when soluble phosphorus runoff was limited to 20 tons per year. The imposition of phosphorus deviation limits above target yielded further and larger reductions when the soluble phosphorus runoff was limited to 30 tons per year and above. The Captina soil produced amounts of soluble phosphorus runoff per hectare higher than those for the Tonti soil at all soluble phosphorus runoff limits. The levels of phosphorus runoff for the Captina soil were also significantly lower when Alum-treated poultry litter was allowed compared to levels obtained when farmers applied untreated poultry litter on the pastures.

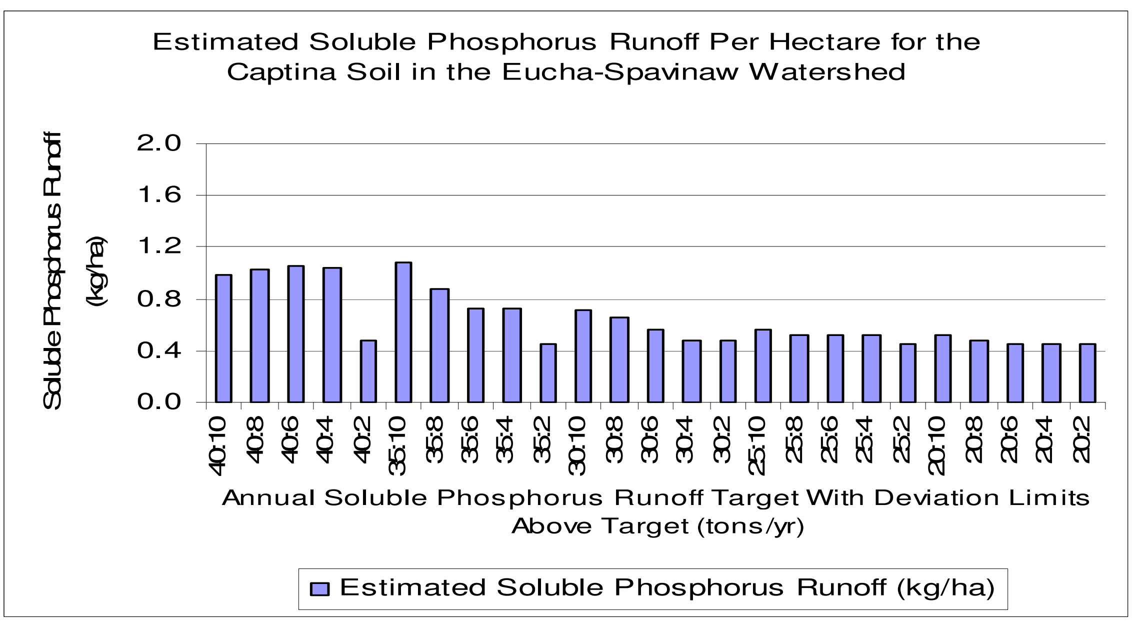


Figure 57 Estimated Soluble Phosphorus Runoff Per Hectare for the Captina Soil.

Figure 58 below shows the effect of alternative soluble phosphorus runoff targets and phosphorus runoff deviation limits above target on estimated amount of soluble phosphorus runoff per hectare for the Doniphan soil in the Eucha-Spavinaw watershed.

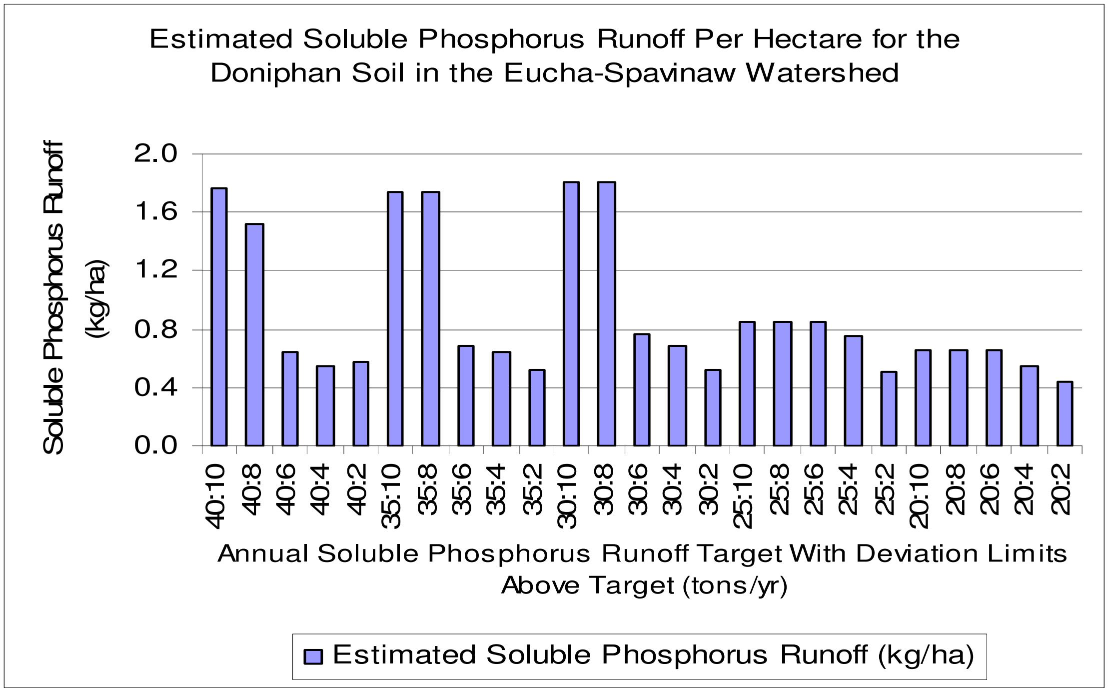


Figure 58 Estimated Soluble Phosphorus Runoff Per Hectare for the Doniphan Soil.

The Doniphan soil had a response pattern to changes in mean phosphorus runoff limits similar to that observed in the case of the Clarksville soil. The estimated quantity of soluble phosphorus runoff per hectare for the Doniphan soil remained unchanged as the annual soluble phosphorus runoff target was reduced from 40 to 30 tons per year. However, the amount of phosphorus loss declined sharply at annual soluble phosphorus runoff targets lower than 30 tons per year. The amount of phosphorus runoff was estimated at 0.44 kilograms per hectare when the annual soluble phosphorus runoff was limited to 20 tons per year given a maximum allowable mean phosphorus deviation above target of 2 tons per year. This soil produced the largest amount of soluble phosphorus runoff per hectare at all total soluble phosphorus runoff limits for the watershed when compared to other major soils. The imposition of phosphorus deviation limits above target yielded further and larger reductions when the soluble phosphorus runoff was limited to 30 tons per year and above. As observed in other major soils, the Doniphan soil also exhibited phosphorus runoff levels that were significantly lower when Alum-treated poultry litter was allowed than in the case where farmers used untreated poultry litter.

Total Agricultural Income from Grazing

Figure 59 below indicates that the total annual producer income from pasture management systems in the solution set when the annual soluble phosphorus runoff was limited to 40 tons per year was estimated at about $2.7 million. A 25 percent reduction in the soluble phosphorus runoff limit lowered producer income to about $1.7 million. A further reduction of the soluble phosphorus limit to 20 tons per year yielded an annual producer income from grazing of about $700,000.

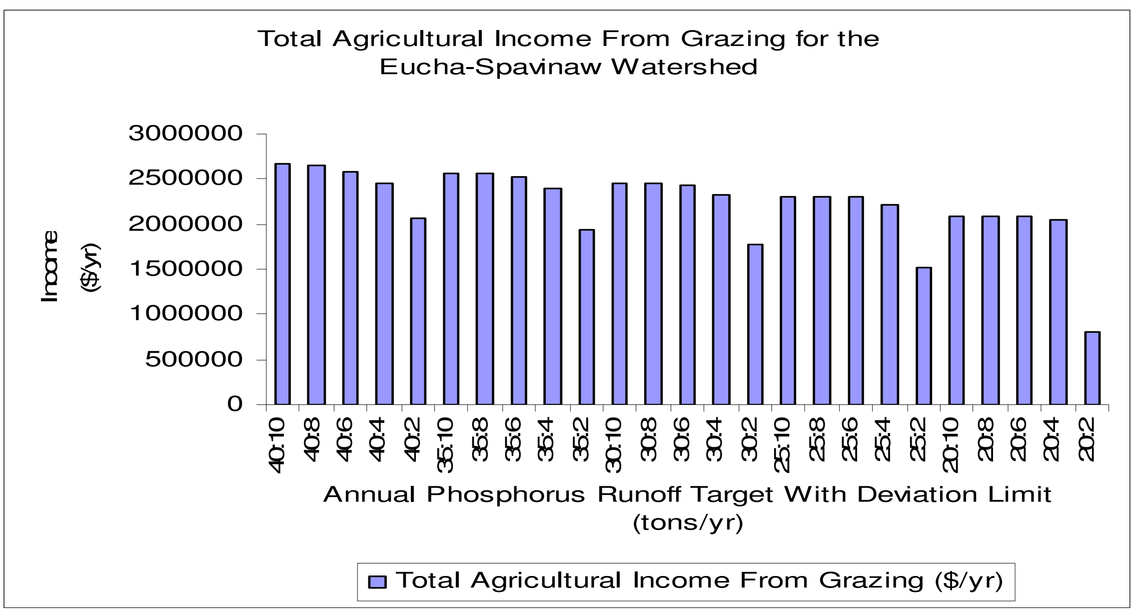


Figure 59 Estimated Total Producer Income from Grazing in the Watershed.

Total Abatement Costs with Alum-Treated Litter for the Watershed

Figure 60 below shows the respective reductions in agricultural income from grazing at each soluble phosphorus runoff target and deviation limit. The reductions in producer income were estimated as the difference in the value of the objective function (representing total agricultural net returns from grazing in the watershed) at the current allowable average phosphorus loading (assumed to be 40 metric tons per year) and the value of the objective function at each of the alternative annual soluble phosphorus loading targets (that is, at 35, 30, 25, and 20 tons per year) and a specified phosphorus deviation limit above a given phosphorus loading target. These reductions in producer income represent estimated total phosphorus pollution abatement costs for the watershed. We assumed that at the base or current phosphorus loading of 40 tons per year, there are no abatement costs incurred in the watershed. Figure 60 shows that estimated total abatement costs increased at an increasing rate as the total annual soluble phosphorus runoff limit for the Eucha-Spavinaw watershed was reduced from 40 to 20 tons per year. Total abatement costs rose from zero given a soluble phosphorus runoff limit of 40 tons per year to about $800,000 per year when the soluble phosphorus limit was reduced to 30 tons per year with an allowable phosphorus deviation limit above target of not more than 2 tons per year. Total abatement costs more than doubled to an amount of about $1.8 million per year when the soluble phosphorus runoff limit was further reduced to 20 tons per year given the same phosphorus deviation limit above target.

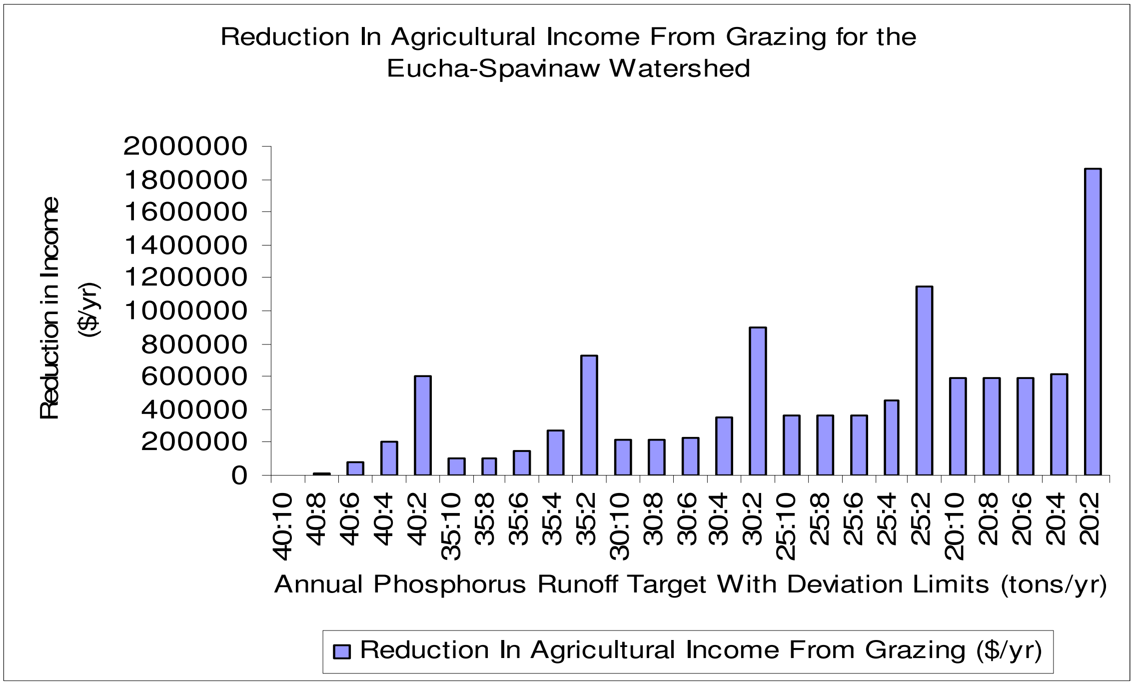


Figure 60 Estimated Total Phosphorus Pollution Abatement Costs

Marginal Phosphorus Pollution Abatement Costs for the Watershed

Figure 61 below presents the estimated cost of abating an additional ton of soluble phosphorus pollution per year in the watershed. Marginal abatement costs are shown to increase at an increasing rate as the annual soluble phosphorus runoff target and deviation limits are reduced. As can be seen from figure 60 below, reducing the annual soluble phosphorus runoff limit from 40 tons to 20 tons per year increased marginal abatement costs from $19.00 to $59.00 per ton given an allowable phosphorus deviation limit above target of 10 tons per year. When the allowable phosphorus deviation was limited to 2 tons per year, marginal abatement costs rose drastically from $390 to $3,872 per ton as the total annual soluble phosphorus runoff limit for the watershed was reduced from 40 tons to 20 tons per year.

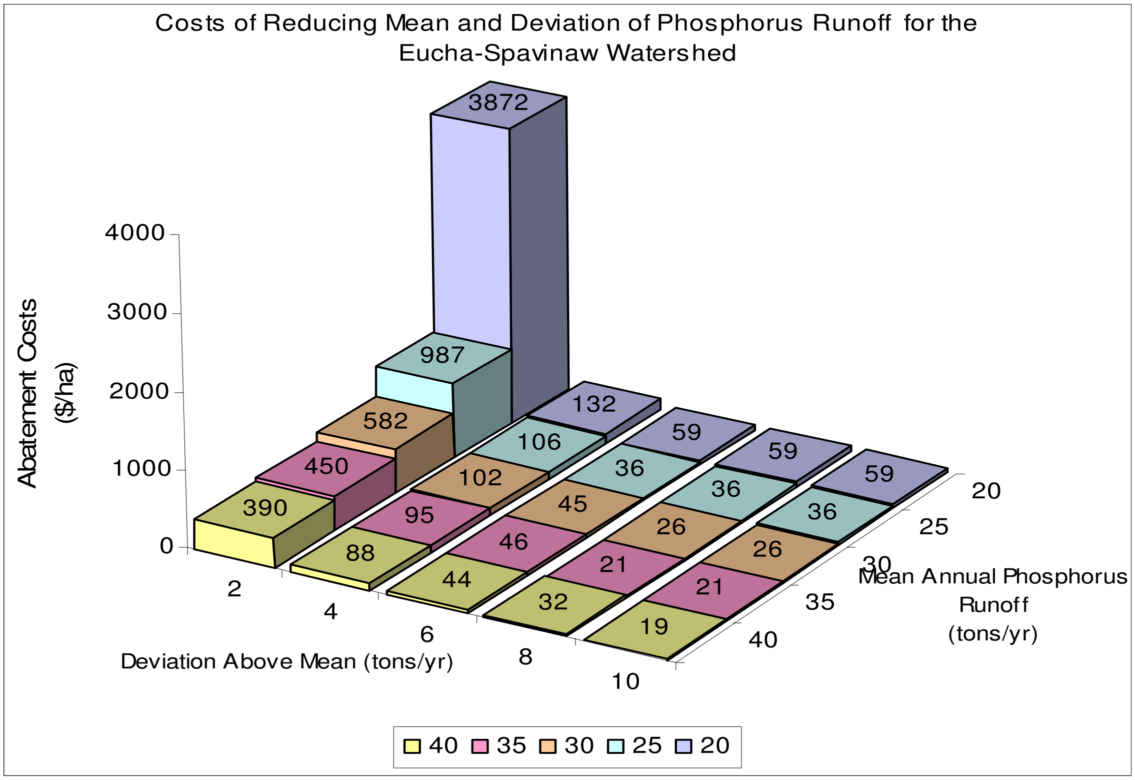
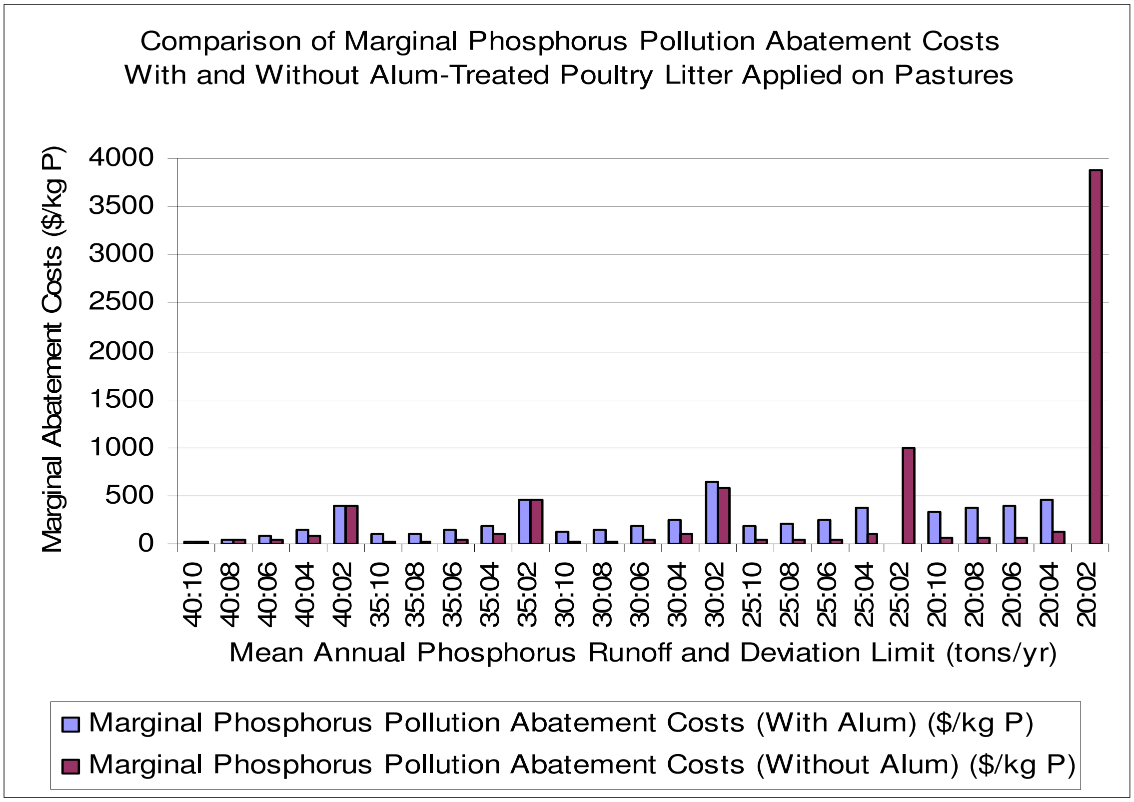


Figure 61 Estimated Marginal Phosphorus Pollution Abatement Costs

Marginal phosphorus pollution abatement costs with and without Alum-treated poultry litter are compared in Figure 62. Generally the marginal abatement costs increased as the mean annual phosphorus loss was restricted from 40 Mg to 20 Mg and phosphorus deviations above the mean lowered from 10 Mg to 2 Mg per year. However, the marginal abatement costs rose sharply when the mean phosphorus loss was limited to levels lower than 25 Mg per year with or without the Alum-treatment. Marginal abatement costs with Alum-treated litter were relatively higher compared to the without Alum-treatment scenario.



Note: In the case of Alum-treated litter, no abatement costs were recorded at mean phosphorus losses of 20 and 25Mg /yr given mean phosphorus deviation limit above target of 2Mg/yr. The solutions were infeasible.

Figure 62 Marginal Abatement Costs With and Without Alum-Treated Litter.

Efficient Phosphorus Pollution Control Policies for the Watershed

Based on the efficient allocation of pollution and the cost-effectiveness equimarginal principles discussed in earlier sections (see Figure 9 and Figure 10), reductions in phosphorus emissions in the Eucha-Spavinaw can be achieved through the use of economic instruments. These economic instruments can either be used as a complement to or as a substitute for direct regulation practices in the watershed to provide incentives to reduce those activities that emit excessive phosphorus loads into the environment. The Environmental Protection Agency (EPA) may (1) issue tradable phosphorus pollution permits, (2) impose per-unit charge or tax on phosphorus emissions, (3) offer a per-unit subsidy on each unit of phosphorus the polluter abates, and (4) impose a charge on phosphorus emissions and offer pollution control subsidy to polluters. The following sections provide more insight on each of these market-based pollution control options.

Emissions Standard / Legal Limit on Phosphorus Emissions

As illustrated in Figure 9, we noted that an efficient allocation of phosphorus pollution is that at which the marginal cost of control is equal to the marginal damage cost caused by the pollution for each source in the watershed. State or federal agencies might achieve that efficient allocation of phosphorus pollution in the Eucha-Spavinaw watershed by imposing a legal limit on the amount of phosphorus pollution allowed by each source. The total quantity of phosphorus pollution allowed on such permits would be limited to Z* on Figure 8 (the level of pollution at which the marginal abatement cost equals marginal damage cost). The establishment of a system of discharge licenses and controlling the allowable amounts of pollution on the permits could limit phosphorus pollution emissions in the watershed. Based on the mean annual phosphorus load targets used in this study, if the state or federal agency wanted a 50 % reduction in phosphorus emissions from the current level assumed to be 40 Mg per year, the total quantity of phosphorus pollution allowed on such permits would be limited to 20Mg per year. Polluters could be allowed to trade these permits with each other. The tradable permits or quota regime allows for the reallocation of the right to pollute among polluters (Coase, 1960). This trading arrangement would create an incentive to achieve reductions in phosphorus emissions below the legal requirements, enabling a producer to expand, or to sell the resultant phosphorus pollution credits to other polluters needing them in the Eucha-Spavinaw watershed.

<u>Per-Unit Phosphorus Emissions Tax or Per-Unit Pollution Control Subsidy</u>

Environmental degradation caused by excessive phosphorus pollution is an externality cost borne by society. The private polluters in the watershed do not take such a cost into account when making their production decisions. As such, they tend to oversupply the product or overuse the input with such a negative externality like poultry litter. The state or federal agency might achieve an efficient allocation of phosphorus pollution by imposing a tax or charge of a specified amount per unit of phosphorus discharged into the environment by each polluter in the Eucha-Spavinaw watershed. As illustrated in Figure 9 and Figure 10, the emission charge would be set equal to T*, the tax rate at which marginal abatement costs across polluters are the same and equal to marginal damage cost (That is, a charge where $MC_1 = MC_2 = MDC$). The total payment any polluter would make is determined by multiplying the amount of phosphorus pollution emitted times the per-unit effluent fee or tax. This polluter-pays principle would make each emitter to internalize the marginal damage caused by each unit of phosphorus emitted. By imposing the same emission tax on all emitters in the watershed, it is expected that all profit-maximizing polluters would respond to this internalized pollution cost by reducing emissions to a point where the marginal abatement cost is equal to the effluent fee or emission tax, T*. When that happens, the resulting phosphorus reduction allocation will be consistent with minimizing total phosphorus pollution control costs for the watershed.

An alternative approach to an emission charge would be to pay producers for each unit of phosphorus pollution they abated. As illustrated in Figure 9 and Figure 10, the payment rate or subsidy would be set equal to T*, the subsidy rate at which marginal abatement costs across polluters are the same and equal to marginal damage cost (That is, a control subsidy where $MC_1 = MC_2 = MDC$) (see Figure 9 and Figure 10) .

In this study, the shadow prices on the phosphorus constraint (obtained from the optimization model) provided a guideline for setting the tax rate at each of the mean annual phosphorus loads investigated. The per-unit emission tax was set equal to the shadow price. Table 27 shows the amount of tax revenue that will likely result from implementing the per-unit emission tax policy in the Eucha-Spavinaw watershed. Table 27 also shows the total amount of subsidy that will likely result from implementing the per-unit control subsidy policy in the Eucha-Spavinaw watershed. The total amount any producer could be paid was determined by multiplying the amount of phosphorus pollution abated times the per-unit control subsidy. Again, the per-unit subsidy rate was set equal to the shadow price on the phosphorus constraint at each of the mean annual phosphorus loads investigated.

Table 27 Tax Revenue / Subsidy Payments for Various Mean Phosphorus Load Limits.

Mean P. Loss Limit (Mg/yr)	Mean P. Deviation Limit (Mg/yr)	Actual Mean P. Loss (Mg/yr)	Actual P. Loss Abated (Kg)	Shadow Price (Tax Rate) ($/Kg)	P. Emissions Tax Revenue ($)	Pollution Control Subsidy ($)
40	10	40.0	0	18.54	741,400	0
40	8	39.7	273	31.69	1,258,798	8,642
40	6	36.2	3,764	43.84	1,588,610	165,030
40	4	31.9	8,057	88.37	2,822,838	711,962
40	2	22.1	17,876	390.43	8,637,865	6,979,495
35	10	35.0	5,000	21.25	743,890	106,270
35	8	35.0	5,000	21.25	743,890	106,270
35	6	33.6	6,444	45.84	1,538,200	295,400
35	4	29.0	10,990	95.37	2,766,829	1,048,131
35	2	19.5	20,501	449.65	8,767,753	9,218,327
30	10	30.0	10,000	25.86	775,890	258,630
30	8	30.0	10,000	25.86	775,890	258,630
30	6	30.0	10,000	43.25	1,297,470	432,490
30	4	25.8	14,178	101.88	2,630,678	1,444,442
30	2	17.7	22,298	582.45	10,310,374	12,987,786
25	10	25.0	15,000	35.51	887,750	532,650
25	8	25.0	15,000	35.51	887,750	532,650
25	6	25.0	15,000	35.51	887,750	532,650
25	4	22.9	17,149	106.24	2,427,764	1,821,876
25	2	15.3	24,687	987.17	15,116,519	24,370,121
20	10	20.0	20,000	58.54	1,170,800	1,170,800
20	8	20.0	20,000	58.54	1,170,800	1,170,800
20	6	20.0	20,000	58.54	1,170,800	1,170,800
20	4	19.6	20,386	132.14	2,591,800	2,693,760
20	2	12.7	27,324	3872.22	49,085,642	105,803,118

CHAPTER VI

SUMMARY AND CONCLUSIONS

Summary of the Procedures and Results

The Eucha-Spavinaw watershed, shared by Oklahoma and Arkansas, has been troubled by water pollution for years. Eutrophication of Lakes Eucha and Spavinaw is attributed to high phosphorus loading resulting largely from amount and history of land application of litter produced by an intensive poultry industry in the area. The purpose of this study was to determine litter and pasture management practices to reduce total phosphorus runoff from agricultural non-point sources in the watershed to meet various possible annual total phosphorus limits within a specified margin of safety at minimum social cost. Ambient-based approaches coupled with policy instruments such as taxes and subsidies have been shown to achieve socially efficient outcomes. USDA programs and policies to reduce non-point source agricultural nutrient runoff have relied upon voluntary technology-based approaches whereas USEPA programs tend to focus on trading technology rather than pollutants or loadings. The ability of these programs to meet the goals of water quality improvement is debatable. Mathematical programming and SWAT are useful tools in determining the most efficient methods of reducing nutrient loading in watersheds. A series of multi-year simulations were conducted where alternative management practices were tested in each HRU of the watershed. The BMPs (unique to each HRU) that maximized total producer income while meeting various phosphorus load reductions within a specified margin of safety were then selected for implementation in the watershed.

The objective of this study was to integrate GIS-based biophysical simulation modeling with a spatial mathematical programming model to identify pasture management practices suitable for specific sites that maximize net agricultural income for the Eucha-Spavinaw watershed while meeting maximum average annual phosphorus loads entering Lakes Eucha and Spavinaw within specified margins of safety. A GIS data base containing topography, hydrology, soils, and land use and crop histories was created and used as basic sources of input parameters for the SWAT modeling. The Eucha-Spavinaw watershed was subdivided into 90 subbasins based on topography and hydrology in the area, and further subdivided into 2416 hydraulic response units (HRUs) according to major soil type and land use in each subbasin. SWAT simulated sediment, crop yields, and nutrient yields at the watershed and each subbasin outlet. The simulations were performed on current and alternative pasture management practices. SWAT outputs allowed for geographical and temporal examination and comparison of sediment, crop yield, nutrient discharges, and potential for nutrient contamination of surface and groundwater within a given management practice and across different pasture management systems.

A set of 105 feasible grazing management practices was simulated and tested in each HRU in the watershed. These were constituted from different combinations of Alum-treated and non-Alum treated poultry litter and elemental nitrogen application rates, maximum biomass maintained for grazing, and stocking rates on pasture land. The SWAT output was input into a Target MOTAD risk programming model that selected the best management practice for each HRU in the watershed to maximize agricultural income from grazing while meeting maximum average annual phosphorus loads entering Lakes Eucha and Spavinaw within specified margins of safety. The model allowed for a possibility of transporting poultry litter from chicken farms to phosphorus deficient sub-basins within the watershed and to a litter-to-energy processing plant. Thus, the economic model optimally allocated best management practice(s) to non-point sources in each HRU and determined optimal quantities of litter transported between subbasins and optimal quantity of litter transported from each subbasin to the processing plant.

Conclusions

Several conclusions may be drawn from the findings of this study. First, pasture management systems using poultry litter as fertilizer generate potential nitrate and phosphorus contamination for the surface and ground water in the Eucha-Spavinaw watershed. Excessive land application of litter, phosphorus runoff, and water quality issues in the Eucha-Spavinaw watershed can be addressed in a modeling framework that takes into account environmental and economic aspects in the area. An integrated environmental-economic modeling approach, that combines the use of the SWAT model and mathematical programming can be used to assess the impact of current and alternative farming practices on water quality in the Eucha-Spavinaw watershed. The integrated biophysical-hydrologic-economic–modeling framework developed for this dissertation research reflected the major hydrologic and economic processes related to poultry litter supply, grazing management systems and phosphorus runoff in the Eucha-Spavinaw watershed. This decision-support tool could be used to assist policymakers in their strategic phosphorus loss reduction and water quality improvement decisions and in setting realistic and efficient Total Maximum Daily Loads for the watershed.

Second, the environmental-economic optimization model assigned various site-specific pasture management systems and litter allocations on the basis of relevant environmental and economic factors in that part of the watershed. There was no single management practice that dominated in all parts of the watershed. While it is straight forward to analyze the effects of using a single BMP in all HRUs, our findings suggest that meeting the TMDL for an entire Eucha-Spavinaw watershed at least-cost means that the best pasture management practice for each HRU must be individually chosen. This can be accomplished in a mathematical programming model that permits a choice of BMP unique to each HRU using the necessary coefficients which are derived from results obtained from the SWAT simulation model. The SWAT simulation model is not an optimization model. Optimization is required to determine least cost combinations of pasture management practices for each HRU to meet the TMDL for the Eucha-Spavinaw watershed at least-cost to society. The environmental-economic optimization model used in this study showed that least-cost abatement policies may differ significantly from uniformly applied command-and-control policies and be much less costly than the imposition of uniform restrictions across all HRUs in the watershed. The shadow prices obtained from the agricultural land area constraint of the optimization model provided relevant economic and environmental information that can be used to select HRUs and target specific areas in the watershed for inclusion in a phosphorus pollution reduction program. The shadow prices on the phosphorus constraint in the optimization model provided a guideline as to how pollution control policies could be implemented using economic instruments such as tradable phosphorus emission permits, per-unit phosphorus emissions tax or charge, and per-unit phosphorus control subsidies.

Third, optimal poultry litter application rates and phosphorus runoff varied from one soil to another within the Eucha-Spavinaw watershed. The econometric model determined that not all soil types in the Eucha-Spavinaw watershed contributed uniformly to the phosphorus runoff problem in the area. The optimization model however indicated that the least-cost way to reduce phosphorus loss in the watershed required equating marginal abatement costs across HRUs. A cost effective and efficient phosphorus runoff reduction program may comprise of producers controlling non-uniform amounts of phosphorus emissions depending on their marginal costs of controlling pollution. Producers with lower marginal phosphorus pollution abatement costs in the watershed will control large amounts of phosphorus pollution than others.

Soils such as Britwater, Razort, Clarksville, Captina, Secesh and Healing contributed significantly higher amounts of phosphorus than the Waben soil. An additional hectare of pasture on any of these soils increased phosphorus runoff by 1.5 kg per hectare on average. The phosphorus runoff problem worsened when pastures on these soils were heavily grazed at stocking rates exceeding 1.00 AU/ha and the plant biomass maintained during grazing was lower than 1600 kg/ha. The other soils that appeared not to generate significant levels of phosphorus runoff received higher optimal litter application rates compared to the set of soils specified above. Though reduction of litter application rates on pastures reduced total phosphorus runoff for the watershed, complete elimination of all fertilizer was found to actually increase total phosphorus loss on some soils because of increased erosion and sediment bound phosphorus owing to reduced plant biomass on the field. This implied that farmers must supplement litter nitrogen with commercial nitrogen on the pastures. It may be more cost effective to develop phosphorus reduction programs that target specific soil types within the Eucha-Spavinaw watershed rather than continue with the current uniform policy of limiting litter application rates strictly by soil test phosphorus. Based on considerable variation of litter application rates and phosphorus runoff by soil type, an approach that targets areas in the watershed for phosphorus loss reduction by focusing only on quantities of poultry litter used as fertilizer and estimated phosphorus runoff may not necessarily be effective in the Eucha-Spavinaw watershed.

Fourth, implementation of environment-friendly grazing management systems (those that represented less use of untreated poultry litter, maintaining high levels of plant biomass during grazing, and low stocking rates on pastures) played a major role in reducing phosphorus loss in the Eucha-Spavinaw watershed. However, it should be noted that in our simulation, the pastures were modeled as grazing units. The livestock removed phosphorus with the grass during grazing and then returned a sizeable amount of that phosphorus back onto the soil surface when they deposited manure on the field. Thus, grazing operations removed phosphorus from the deeper soil layers onto the soil surface where it was most likely to be washed off during storm events. Cattle also contributed to increased runoff by trampling on the plants and compacting the top soil. However, the imposition of restrictions on maximum allowable phosphorus loss for the watershed allowed for a choice of the best management practice for each location that resulted in overall phosphorus loss reduction and maximized agricultural income from grazing for the entire watershed.

Fifth, reduction of litter application rates on pastures resulted in producers applying more commercial nitrogen to maintain higher biomass pastures. When the mean phosphorus loss was restricted to 20 Mg per year, nearly all litter nitrogen was replaced by commercial nitrogen. Large increases in the use of elemental nitrogen to replace poultry litter (and reduce phosphorus runoff) increased nitrogen loss and potential nitrate contamination of surface and ground water.

Sixth, the use of alum-treated poultry litter appeared to be a very cost effective phosphorus runoff reduction strategy even at high annual phosphorus loss limits for the watershed. As the mean phosphorus loss limits were reduced, the pasture management practices that were adopted included those that encouraged (1) the use of alum-treated litter to meet the nitrogen requirement for the crop, (2) lowering stocking rates on the pastures (to maintain 1 AU/ha or less) and (3) retaining higher levels of plant biomass during grazing (at 1600kg/ha or above). However, the use of alum-treated litter would reduce phosphorus runoff in the short term. It is a temporary solution because continued use of alum-treated litter would lead to a long-term build-up of phosphorus in the soil. The increased phosphorus load over time would eventually result in increased levels of phosphorus runoff.

Seventh, the possible litter-to-energy plant received lesser amounts of poultry litter when producers had an incentive to use alum-treated poultry litter as fertilizer. The amount of poultry litter shipped to the litter-to-energy plant at Jay, Oklahoma increased at a slower rate (compared to the untreated poultry litter option) as the mean annual phosphorus load limit was reduced. These results suggested that the possible litter-to-energy processing plant at Jay, Oklahoma might not be a viable option on its own merit. There was also indication of potential poultry litter trading within the watershed as the mean phosphorus load was reduced. The general direction of litter shipment within the watershed was westward towards phosphorus-deficient subbasins. However, when the alum-treatment option was removed from the model, the litter-to-energy power plant located at Jay, Oklahoma became a more cost effective method of reducing both the level and the variability of phosphorus runoff as pollution limits for the Eucha-Spavinaw watershed were reduced from 40 to 20 tons per year.

Lastly, simulated annual phosphorus loss amounts from the watershed exceeded the target mean phosphorus load by larger deviations when it was set at 40,000 kg per year than when it was set at lower levels. This implied that compliance with the recommended phosphorus load reductions improved as the phosphorus loss target for the watershed was reduced. However, small reductions in deviation above target could be achieved without reducing the mean annual phosphorus load for the watershed. Reduction of larger deviations would require reducing the phosphorus loss target for the watershed. Significant reductions in total phosphorus runoff were achieved by varying phosphorus deviation limits above target without changing the mean phosphorus load when the Alum-treated poultry litter option was incorporated in the optimization model.

Policy Implications

The results of this study would aide in devising a conservation / phosphorus abatement program that could be implemented by State water quality agencies in the Eucha-Spavinaw watershed to maximize total producer income while total phosphorus emissions are held below a specified target to improve water quality in the basin. The findings of this study suggest that it is possible to reduce the total annual phosphorus load from non-point sources in the Eucha-Spavinaw watershed from 40 tons to 20 tons per year through regulation coupled with other necessary phosphorus pollution reduction strategies. The total annual phosphorus load for the Eucha-Spavinaw watershed could be regulated not to exceed 20 tons per year. This regulation would have to be supported by adoption of use of Alum-treated poultry litter, maintaining lower stocking rates (1.00 AU/ha and lower) to prevent overgrazing and maintaining higher biomass on pastures (at least 1600kg/ha) during grazing for cover to prevent erosion and phosphorus runoff. However, the economic incentive to voluntarily adopt these improved pasture management practices might be minimal unless producers are compensated for adoption of these environment-improving measures. Policies that encourage pasture management improvements such as reducing litter application rates, use of Alum-treated litter, appropriate nitrogen fertilization, high biomass maintained on pasture during grazing and maintaining low stocking rates on pastures would significantly reduce phosphorus loading into Lakes Eucha and Spavinaw. Such policies and other relevant agricultural pollution abatement programs need to be based on site-specific conditions including soil type for them to significantly contribute to reduction of phosphorus loading in the watershed.

The results of this study indirectly demonstrated that uniformly applied command-and-control policies such as the current policy to apply poultry manure based on soil test phosphorus were environmentally and economically inefficient in reducing the total phosphorus loading for the Eucha-Spavinaw watershed. This was shown by the fact that none of the solutions had uniform application of litter nor did the least-cost solution have uniform levels of phosphorus loss. Targeted phosphorus TMDLs for soils such as Tonti and Nixa would be very effective and efficient for phosphorus loading reduction in the Eucha-Spavinaw watershed. Providing for non-uniform litter application rates based on soil type and predicted amounts of phosphorus runoff would help meet the phosphorus loss target for the watershed at least cost to society.

This dissertation research developed and applied a comprehensive decision-support tool, an integrated biophysical-hydrologic – economic watershed model, with the ability to reflect the dynamic interactions of essential biophysical, hydrologic, agronomic, and economic components and to explore both the economic and environmental consequences of a wide variety of management practices and policy choices for the

Eucha-Spavinaw watershed. It is hoped that this model will assist water quality program managers in different locations in the watershed in choosing appropriate poultry litter and grazing management practices and policymakers in choosing appropriate phosphorus loss reduction programs and policies. It is strongly recommended that optimization models be integrated with GIS-based biophysical-hydrologic simulation models and made an integral part of the TMDL development process to come up with more realistic and efficient conservation and pollution abatement programs for the watersheds.

This study employed a transportation matrix in which poultry litter was being hauled from chicken farm centroids to subbasin centroids, including the possible litter-to-energy power plant located at Jay, Oklahoma. Results showed that if producers were to adopt the use of Alum-treated poultry litter, the desired phosphorus runoff standard will be met and thus there will be less poultry litter shipped to the litter-to-energy power plant from the watershed. Policies that encourage the use of Alum-treated poultry litter and subsidize transportation of litter within the watershed have a great potential to reduce total phosphorus loss in the watershed. It should be noted, however, that the use of alum-treated litter is a short term solution to phosphorus runoff. Continued use of alum-treated litter will result in the build-up of phosphorus in the soil and lead to increase in phosphorus runoff in the long-term.

Furthermore, when there was no restriction imposed on the maximum allowable phosphorus load for the Eucha-Spavinaw watershed, the proposed litter-to-energy power plant did not appear profitable on its own merit. Producers did not have an economic incentive to haul poultry litter to the litter-to-energy power plant due to high transportation costs. However, the proposed power plant became a more cost effective method of reducing both the level and the variability of phosphorus runoff as total annual phosphorus runoff limits for the watershed was reduced. Therefore, if policy makers could come up with policies that provide an enabling environment for profitable operation of the litter-to-energy power plant and provide economic incentives for hauling poultry litter than use it as fertilizer on pastures would drastically reduce the amount of phosphorus runoff in the Eucha-Spavinaw watershed. The litter-to-energy power plant could be established and operated as a cooperative and entrusted with the responsibility to pick up poultry litter from the respective chicken farm centroids (collection points) established in the watershed as modeled in this study. Producers would haul poultry litter to the power plant if stricter limits were imposed on total phosphorus load from pastures in the Eucha-Spavinaw watershed.

Limitations and Directions for Further Study

Agricultural pollution comes from both point and non-point sources in the watershed. This study assumed that point sources in the Eucha-Spavinaw watershed are achieving 100 percent phosphorus loss abatement and thus focused on agricultural non-point sources in the watershed. This may not be the case in real world practice. Furthermore, this study does not measure environmental damages and loss of recreational values resulting from sediment, phosphorus runoff and eutrophication of the Lakes Eucha and Spavinaw. The analysis in this study assumed constant land use and that poultry litter produced in the watershed was either applied on pastures as fertilizer or shipped to the possible litter-to-energy power plant located at Jay, Oklahoma. We did not consider shipment of poultry litter to destinations out of the watershed, complete non-use of current pasture land, possible changes to crop and range land in the watershed. Neither did we consider haying operations and the effects of supplementary feeding (e.g with winter hay) on total soil phosphorus and levels of phosphorus runoff in this study. Further analysis should amend these shortcomings.

The SWAT model is considered a very reliable modeling tool. However, its inherent uncertainty of parameter estimates is a major limitation in this study. This is because SWAT, as a biophysical simulation model, is a system of equations that represent a simplification of real world processes. There is lack of or incomplete knowledge on some of the variables involved (for instance, incomplete knowledge about the fate and transport of poultry manure with various handling systems and environments and interactions of phosphorus pollution with soil, water, and aquatic ecosystems) that increases uncertainty about the variables used in the model.

Several studies have demonstrated that market-based mechanisms can effectively and efficiently achieve the desired environmental pollution standard. It will be an essential exercise to explore the feasibility and associated transaction costs of establishing a phosphorus pollution trading program for the Eucha-Spavinaw watershed. Pollution trading programs can help achieve efficient targeting of pollution reduction measures.

Further studies could look at more alternative litter uses and other best management practices to control soil and nutrient loss in the watershed and how costs of reducing pollution vary spatially. Overall pollution control cost for the Eucha-Spavinaw watershed could be greatly reduced by identifying those areas with low cost of pollution control and designing appropriate pollution reduction programs.

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APPENDIX

Table A1 Amount of Poultry Litter Applied on Pastures by Subbasin at Selected Mean Annual Soluble Phosphorus Levels (40Mg, 30Mg, 25Mg, and 20Mg/yr and Deviations Above Limit (10Mg, 6Mg and 4Mg /yr).

	Mean Annual Soluble Phosphorus Levels With Associated Deviations Above Limits							
	40:10		30:6		25:6		20:4	
	Amount of Poultry Litter Used (tons/yr)							
Sub-basin	Non-Treated	Alum-Treated	Non-Treated	Alum-Treated	Non-Treated	Alum-Treated	Non-Treated	Alum-Treated
1	1,892	0	1,581	412	248	2,176	52	3,116
2	1,479	0	776	931	550	1,230	0	2,018
3	257	1,040	231	1,075	218	1,171	0	1,099
4	1,423	308	1,423	912	1,369	949	0	1,089
5	4,037	0	960	4,009	0	4,050	0	6,812
6	246	0	289	0	313	0	26	323
7	6	0	6	0	6	0	0	0
8	2,101	0	2,101	0	1,551	0	34	744
9	1,573	0	1,573	0	1,020	1,075	0	1,426
10	769	0	769	0	320	812	7	1,235
11	8	1,077	8	1,077	8	1,178	8	1,178
12	46	80	31	93	31	93	31	93
13	1,770	991	0	2,970	0	4,062	0	4,062
14	1,382	387	169	1,457	169	1,663	223	1,364
15	315	273	304	283	304	877	28	1,121
16	0	137	0	137	0	137	0	137
17	19	435	0	452	0	598	0	1,466
18	146	201	91	252	89	429	24	505
19	142	203	139	207	139	207	0	288
20	513	925	271	1,139	271	1,139	0	1,378
21	5,052	0	4,960	121	4,628	1,669	39	2,837
22	395	0	395	0	0	523	0	523
23	364	146	164	322	161	810	61	902
24	858	436	676	597	107	1,127	107	1,127
25	0	665	0	678	0	678	0	688
26	141	1,032	85	1,082	85	1,702	33	1,952
27	226	108	43	269	36	276	4	317
28	22	770	22	770	22	1,108	22	1,180
29	1,373	0	859	1,536	817	1,707	0	1,723
30	560	21	554	502	507	701	13	810

Table A1 (Continued)

	Mean Annual Soluble Phosphorus Levels With Associated Deviations Above Limits							
	40:10		30:6		25:6		20:4	
	Amount of Poultry Litter Used (tons/yr)							
Sub-basin	Non-Treated	Alum-Treated	Non-Treated	Alum-Treated	Non-Treated	Alum-Treated	Non-Treated	Alum-Treated
31	432	0	432	14	134	408	0	398
32	0	2,320	0	0	27	0	0	0
33	338	379	0	2,320	0	2,320	0	2,083
34	123	0	264	444	264	666	4	895
35	375	0	123	0	26	0	0	23
36	874	37	375	326	120	326	0	437
37	644	58	194	636	104	716	42	771
38	5	178	429	247	13	615	20	927
39	0	2,651	0	190	0	190	0	187
40	0	0	0	3,068	0	3,068	0	2,729
41	0	0	0	3	0	3	0	3
42	29	0	0	0	0	0	0	16
43	0	0	0	10	0	10	0	48
44	0	1,331	0	1,331	0	1,331	0	1,727
45	0	0	0	0	0	131	0	94
46	0	146	0	146	0	146	0	626
47	71	0	0	94	0	94	0	63
48	264	0	0	265	0	265	0	233
49	102	122	0	122	0	122	0	213
50	0	0	0	0	0	0	0	0
51	0	0	0	0	0	0	0	598
52	0	0	0	0	0	0	0	0
53	55	346	0	478	0	578	0	538
54	48	3,728	71	4,066	71	4,066	0	3,092
55	0	0	0	0	0	0	0	0
56	0	0	0	0	0	0	0	0
57	112	34	97	141	97	170	0	281
58	29	613	0	742	0	585	0	1,475
59	188	0	137	237	137	236	0	362
60	0	125	122	125	402	143	0	112

Table A1 (Continued)

	Mean Annual Soluble Phosphorus Levels With Associated Deviations Above Limits							
	40:10		30:6		25:6		20:4	
	Amount of Poultry Litter Used (tons/yr)							
Sub-basin	Non-Treated	Alum-Treated	Non-Treated	Alum-Treated	Non-Treated	Alum-Treated	Non-Treated	Alum-Treated
61	0	75	0	659	0	659	0	749
62	284	0	0	844	0	1,094	0	1,411
63	2,070	0	220	253	213	536	0	2,585
64	166	257	0	1,004	0	1,004	0	919
65	59	0	0	539	0	590	0	853
66	54	0	0	366	0	381	0	502
67	0	696	0	2,343	0	2,248	0	2,214
68	0	0	0	73	0	73	0	73
69	618	165	21	165	0	397	0	763
70	858	189	0	885	0	629	0	2,172
71	98	298	98	596	98	2,007	0	1,993
72	580	1,796	0	5,510	0	3,389	0	4,818
73	526	6	103	6	7	101	0	663
74	0	336	0	336	0	336	0	1,623
75	0	0	0	312	0	547	0	433
76	0	0	0	0	0	0	0	33
77	0	0	0	1,177	0	333	0	1,256
78	83	0	0	311	0	421	0	624
79	133	116	63	179	2	306	2	306
80	0	0	0	328	0	634	0	656
81	681	82	7	803	244	803	0	918
82	131	195	117	207	0	329	0	306
83	21	36	6	36	0	45	0	58
84	34	39	34	39	11	70	0	56
85	0	0	0	0	0	0	0	86
86	0	0	0	2	0	2	0	2
87	0	0	0	0	0	8	0	28
88	0	0	0	0	0	0	0	7
89	19	0	19	33	8	47	0	78
90	0	118	0	118	0	118	0	90

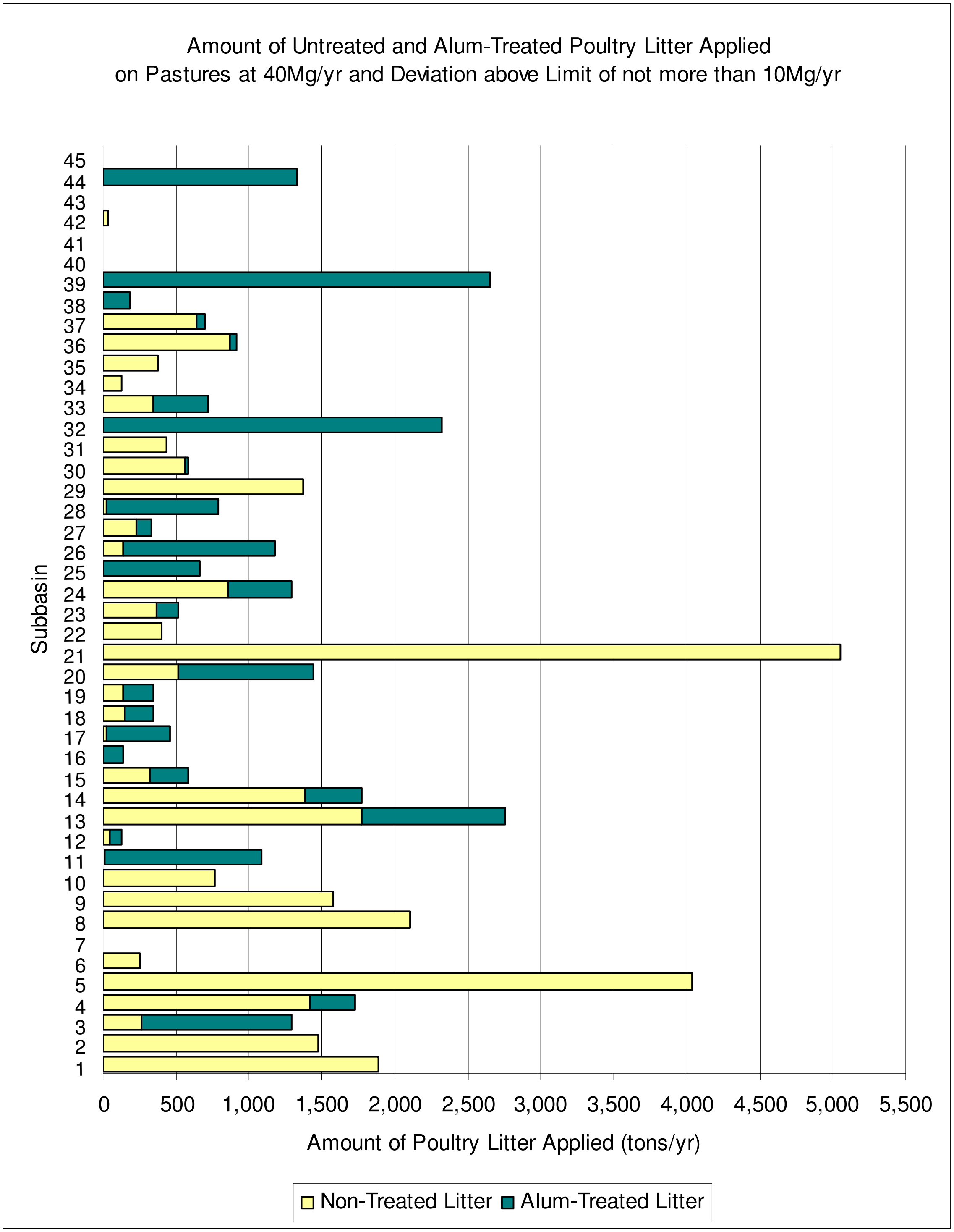

Figure A1. Amount of Untreated and Alum-Treated Poultry Litter Applied on Pastures At 40Mg/yr and Deviation Above limit of not more than 10Mg per year.

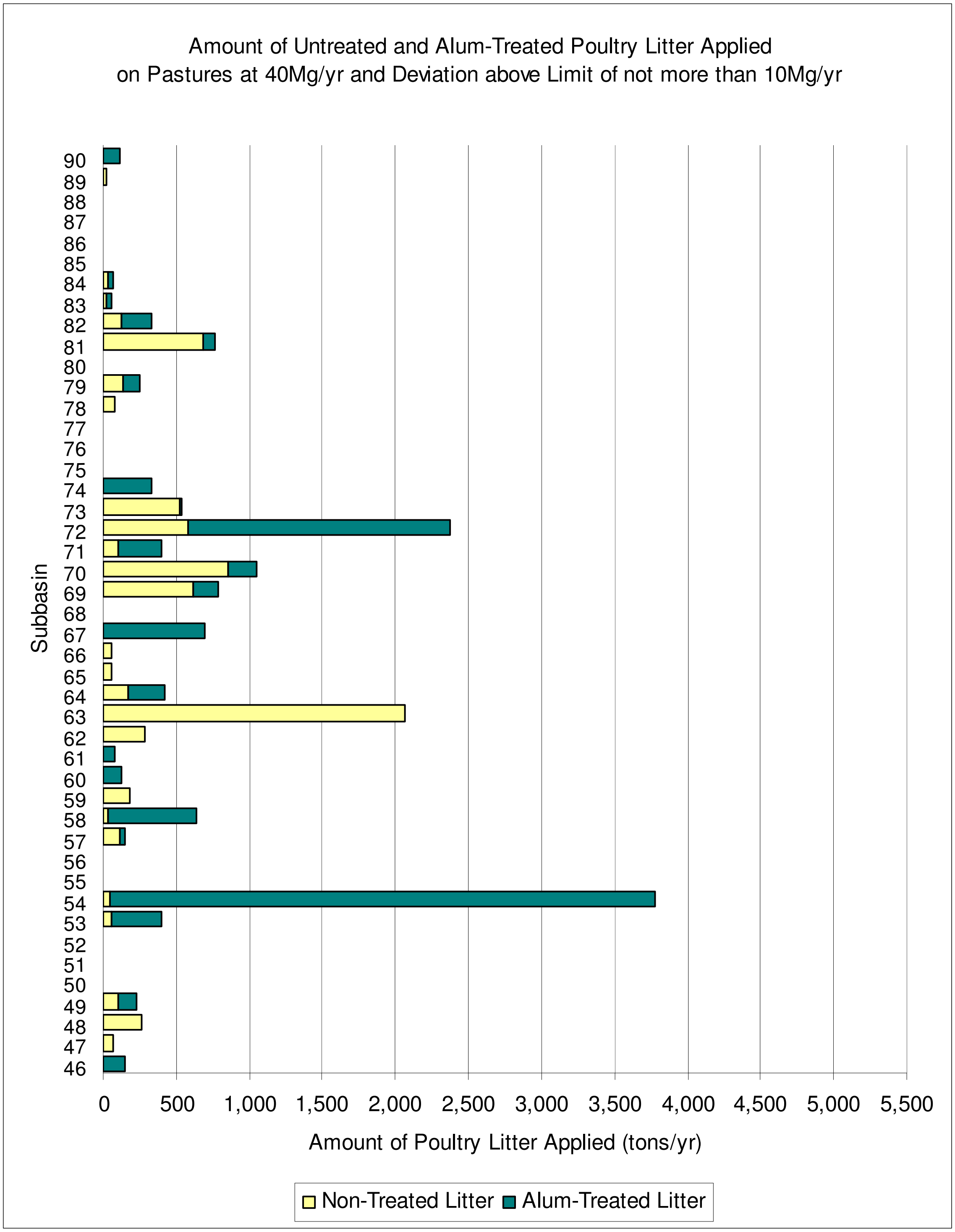

Figure A1. Amount of Untreated and Alum-Treated Poultry Litter Applied on Pastures At 40Mg/yr and Deviation Above limit of not more than 10Mg per year.

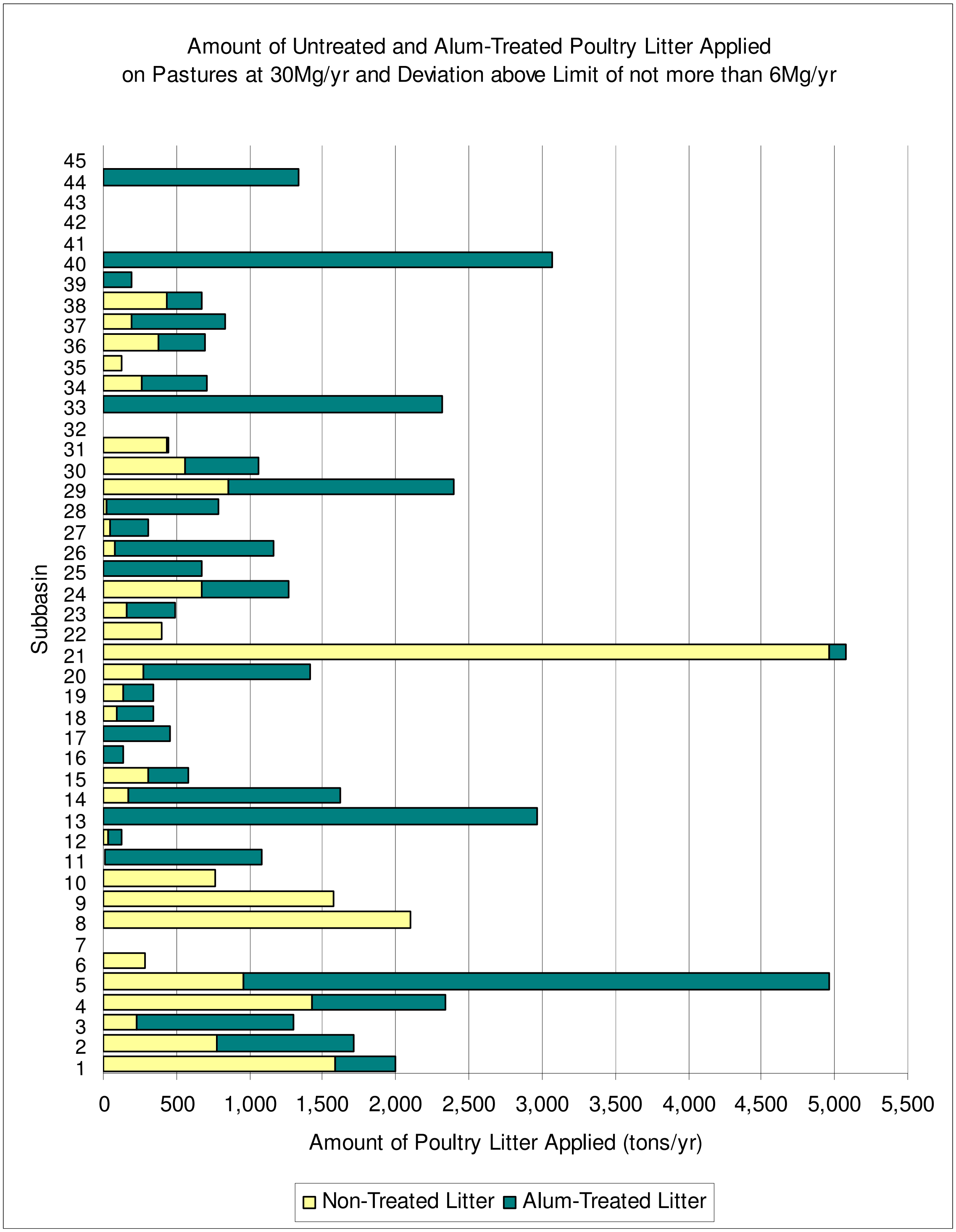

Figure A2. Amount of Untreated and Alum-Treated Poultry Litter Applied on Pastures At 30Mg/yr and Deviation Above limit of not more than 6Mg per year.

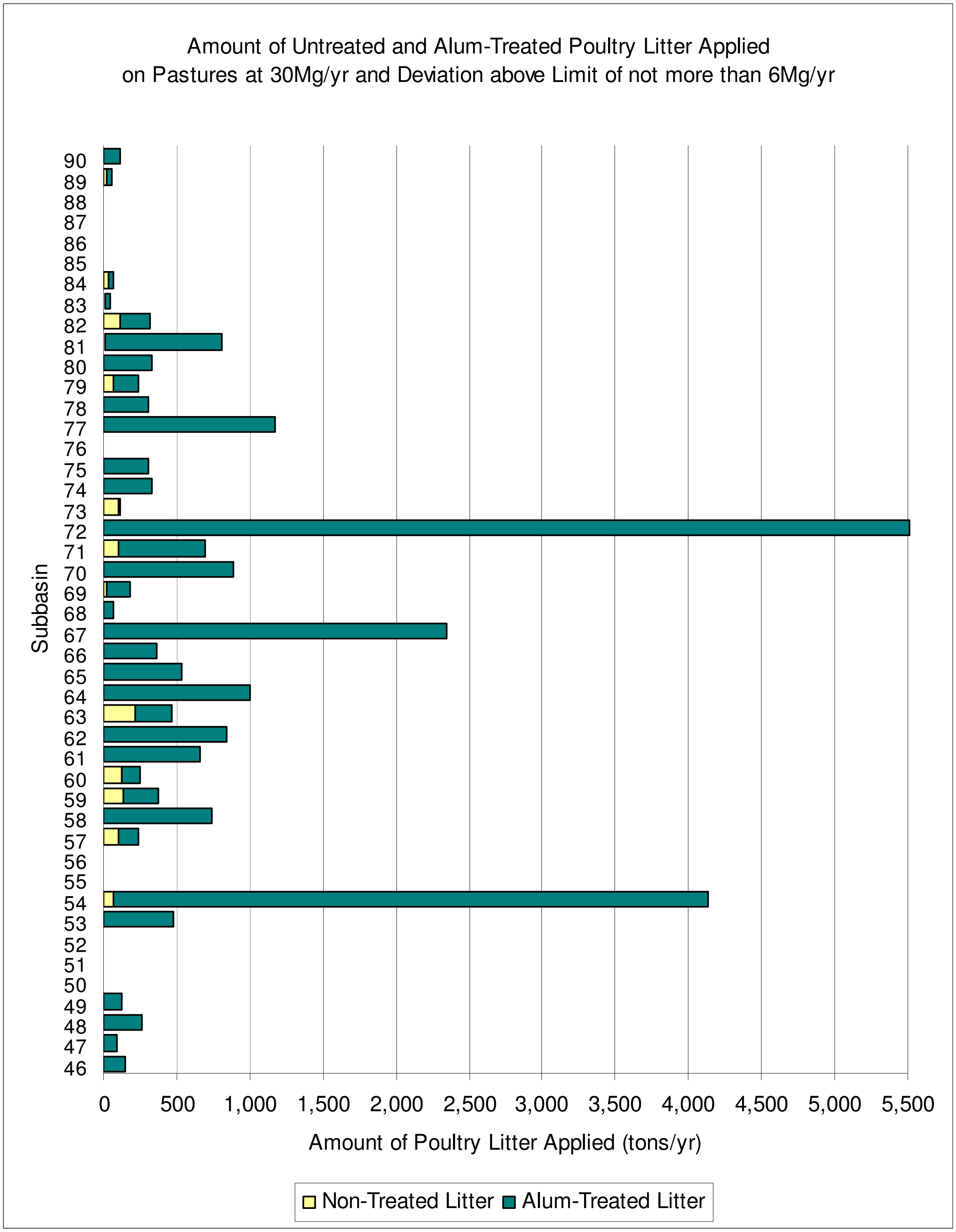

Figure A2. Amount of Untreated and Alum-Treated Poultry Litter Applied on Pastures At 30Mg/yr and Deviation Above limit of not more than 6Mg per year.

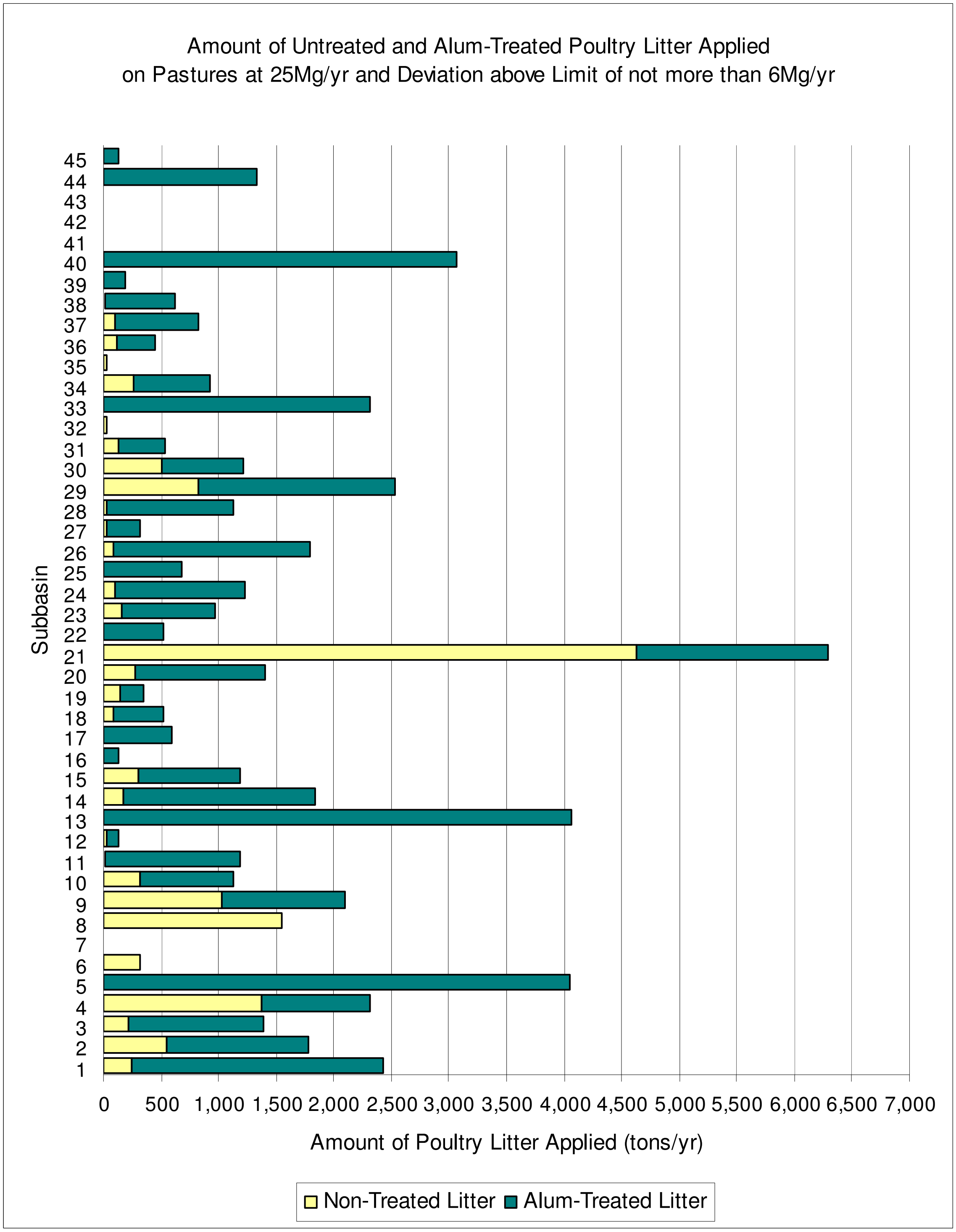

Figure A3. Amount of Untreated and Alum-Treated Poultry Litter Applied on Pastures At 25Mg/yr and Deviation Above limit of not more than 6Mg per year.

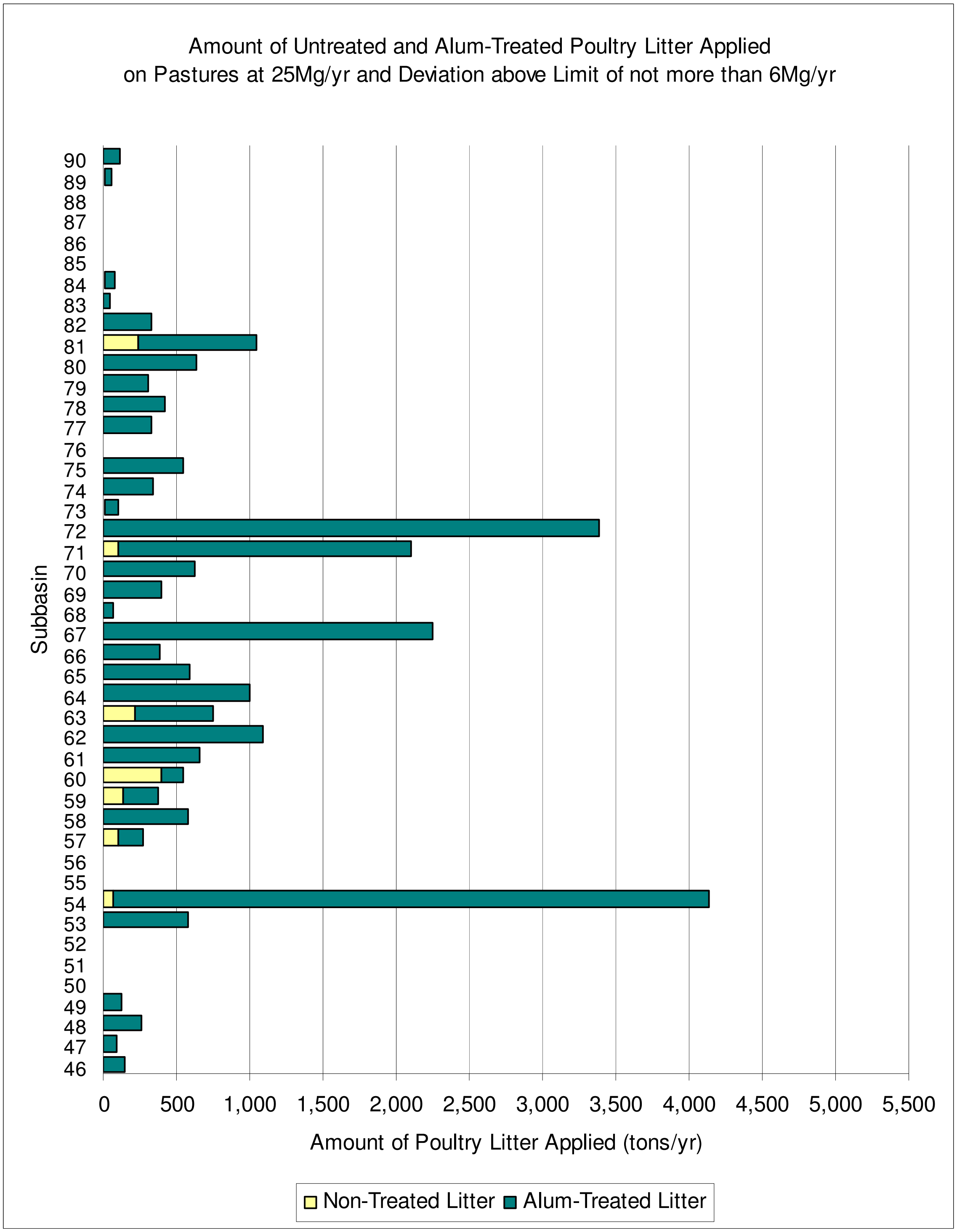

Figure A3. Amount of Untreated and Alum-Treated Poultry Litter Applied on Pastures At 25Mg/yr and Deviation Above limit of not more than 6Mg per year.

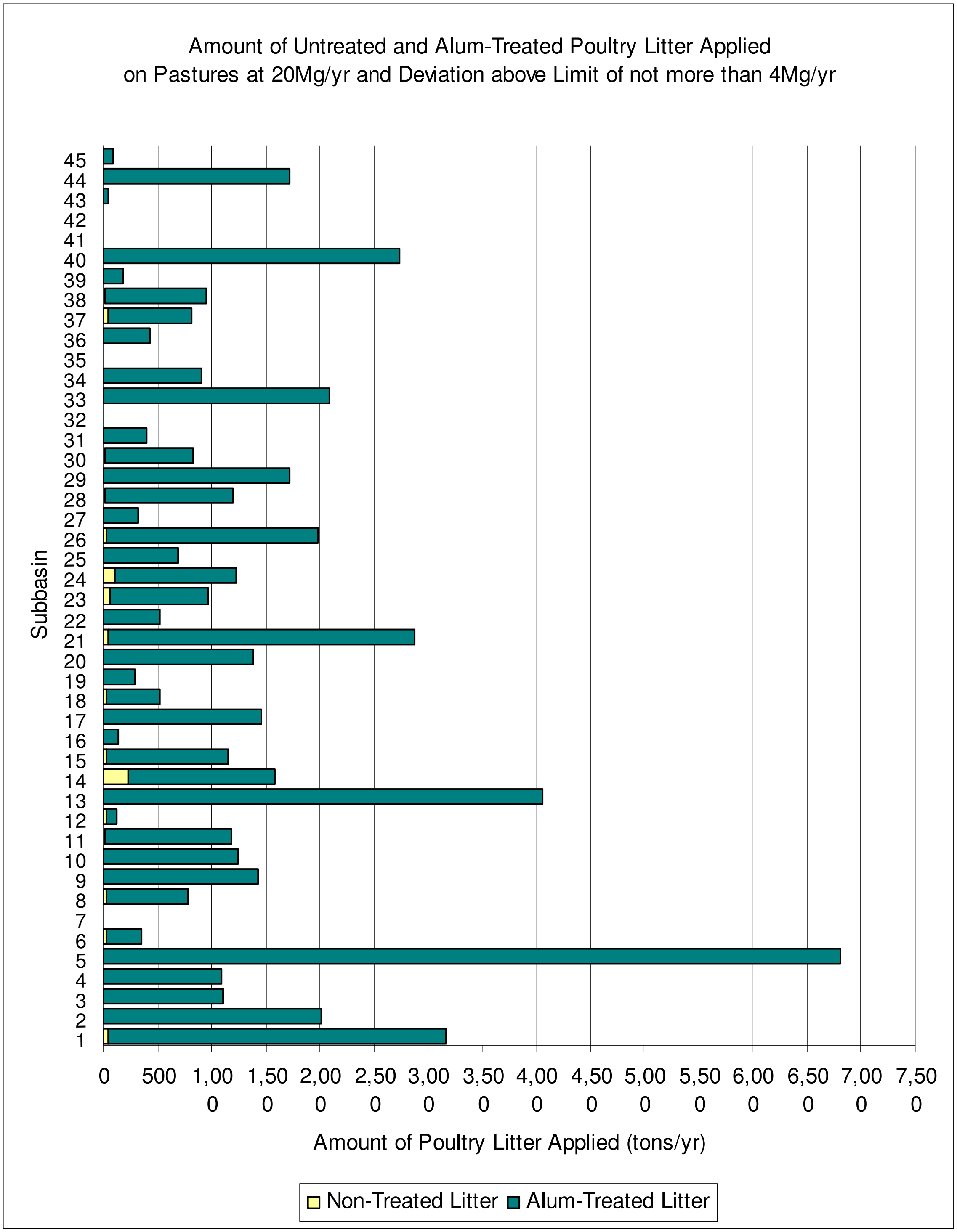

Figure A4. Amount of Untreated and Alum-Treated Poultry Litter Applied on Pastures At 20Mg/yr and Deviation Above limit of not more than 4Mg per year.

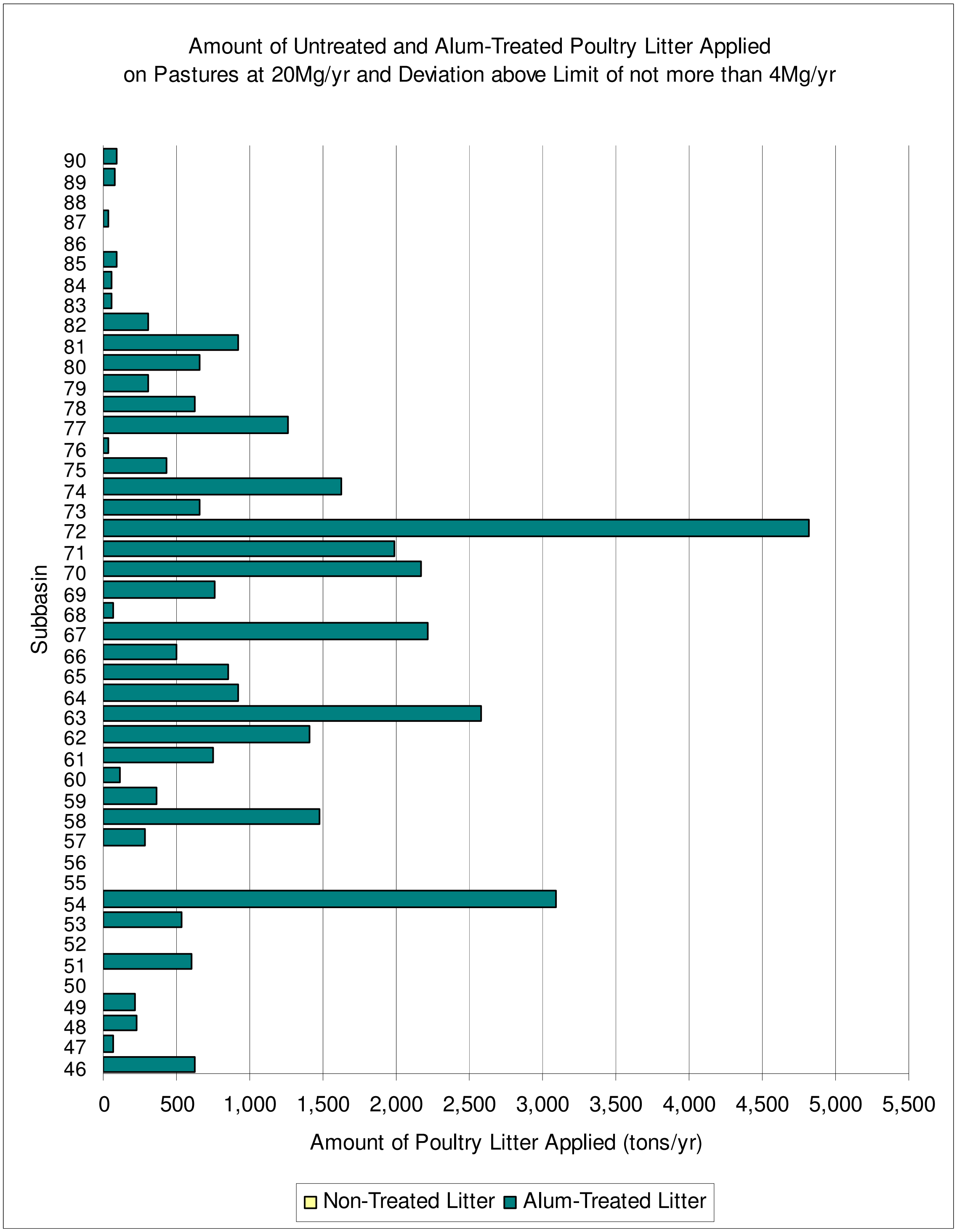

Figure A4. Amount of Untreated and Alum-Treated Poultry Litter Applied on Pastures At 20Mg/yr and Deviation Above limit of not more than 4Mg per year.

VITA

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Major Field: Agricultural Economics

Scope and Method of Study: This study develops and applies a comprehensive decision-support tool, an integrated biophysical-hydrologic – economic watershed model to determine the least cost mix, location, and magnitude of grazing management practices to meet various phosphorus TMDLs for the Eucha-Spavinaw watershed in Oklahoma within specified margins of safety. The GIS - based Soil Water Assessment Tool (SWAT) was calibrated and used to estimate sediment and nutrient loading under alternative management practices comprising of different combinations of litter application rates, commercial nitrogen, maintaining minimum biomass during grazing, and stocking densities. SWAT -generated site-specific coefficients were used in a Target MOTAD programming model to select a management practice for each site in the watershed.

Findings and Conclusions: Results indicate there is no single management practice that dominates in all parts of the watershed and that optimal poultry litter application rates can vary from one soil type to another within the watershed. That is a mix of management practices applied at different locations in the watershed are required to achieve any level of phosphorus abatement from the watershed at least cost. This implies that it may be more cost effective to develop phosphorus reduction programs that target specific soil types within the watershed rather than continue with the current uniform policy of limiting litter application rates strictly by soil test phosphorus. Complete elimination of all fertilizer was found to actually increase total phosphorus loss on some soils because of increased erosion and sediment bound phosphorus. Large increases in the use of commercial nitrogen to replace poultry litter and reduce phosphorus runoff do increase nitrogen loss. As either average annual limit on nutrient loss or the average deviations above the limit are reduced more of the litter is converted to energy or hauled from the watershed. The litter-to-energy plant does not appear profitable on its own merit but becomes a more cost effective method of reducing both the level and the variability of phosphorus runoff as pollution limits are reduced.

Approved by Advisor: Dr. Arthur Stoecker