Economic Issues Associated With Nutrient Management Policy

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FOREWORD

As public attention to nutrient contamination of U.S. waters intensifies, policymakers and resource managers alike will face some difficult choices with respect to reducing the discharge of nutrients to surface and ground water. While agricultural sources of excess nutrients have received widespread attention, other sources will receive increased scrutiny. With the reauthorization of the Clean Water Act and the next round of omnibus farm legislation pending, numerous alternatives for addressing nutrient pollution are being discussed. Extending the Coastal Zone Management Act language into the Clean Water Act is one. Broadening the scope of the conservation provisions included in the 1990 Food, Agriculture, Conservation and Trade Act is another.

In October, 1993, the Southern Natural Resource Economics Committee (SRIEG-10) held a workshop in Raleigh, North Carolina, to address economic issues associated with nutrient management policy. The workshop agenda included three specific components. The first part of the program consisted of an overview of nutrient management issues as they relate to the physical environment and to human health. Next, policy and program options at the federal and state levels were reviewed. The final section included a review of research into the effectiveness and economics of alternative nutrient management programs and technologies. This publication is a compilation of the papers presented at that workshop.

In the first paper Zublena addresses the question "how do excess nutrients find their way into the environment, specifically into water?" Second, Reckhow and Stow present an ecological perspective on the environmental impacts of excess nutrients in water. St. Clair explores the basis for human health concerns arising from the ingestion of water which contains high levels of nutrients, primarily nitrate.

Three papers addressed federal and state policy options. Zinn provides his insights into the likely direction of federal policy in terms of how nutrient issues will be handled. Perkinson discusses the approach taken by the state of Virginia in managing nutrient contamination problems in the Chesapeake Bay. Then, Lynch describes some of the problems which can arise when nutrient discharges, and the associated water quality concerns, violate political boundaries. He presents some lessons learned in the *State of Oklahoma vs. EPA* legal battle and describes how Oklahoma and Arkansas are attempting to overcome previous problems.

In the final group of papers, investigations into various nutrient management strategies and programs are reviewed. First, Rader describes the evolution and mechanics of a pollution credit trading program being implemented in North Carolina. Next, Jacobson, Hoag and Danielson present a theoretical evaluation of pollution credit trading and apply it to experiences in the North Carolina program. Bosch, Batie and Carpentier explore the benefits of targeting nutrient management programs based on physical attributes and management practices of agricultural operations. They evaluate the returns which might be expected from pursuing various levels of information for use in planning the targeting strategy. Finally, Boggess reviews the lessons learned from Florida's experiences with reducing nutrient runoff from dairies into Lake Okeechobee.

Appreciation is extended to the Farm Foundation for its financial support of the workshop and to the Southern Rural Development Center for its support of publication costs.

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NUTRIENTS IN THE ENVIRONMENT

HOW, WHY AND WHERE

By

Joe Zublena[1]

Nutrients in many forms are necessary for life and growth. But whose life and growth? Yours, mine, the neighborhood dog's? The answer is "yes" - for all living things including plants, insects, animals and aquatic life. What are the nutrients and where do they come from? And why are these essential elements sometimes unwelcome in the environment?

All living things exist because of their dependence on and success in finding a source of food. If we trace the food web, or chain, back far enough, it gets down to life-forms that could derive their "food" solely from nonliving or inorganic substances. These inorganic nutrients originated from the weathering and degradation of rocks over millions of years. When these organisms died and decomposed, the nutrients were released from their organic boundaries back to the environment. As life-forms become more complex through evolution, they generally feed at a higher level in the food chain where nutrients are more concentrated and easily attained. Keep in mind that nutrients, like all matter, cannot be created or destroyed. They can, however, change form.

Today, inorganic nutrients still play a major role as a source of nutrients for higher plant growth. Some of these nutrients are still available from the original weathered rocks on-site, whereas others may have been moved to the site by wind, water, animals or man. Movement in geologic time has been by wind, water, volcanic activity, and animals. Much of the movement happened during catastrophic events. Minerals and nutrients were not just scattered evenly but also deposited in large quantities in "pools" or depressions and streams. Humans have sought these mines and ore veins to provide an abundance of a specific mineral or nutrient. Many of these mines and deposits still provide the bulk of plant nutrients that are applied by man.

Sixteen nutrients are essential for higher plant growth. They are: carbon (C), hydrogen (H), oxygen (O), nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), sulfur (S), iron (Fe), manganese (Mn), boron (B), molybdenum (Mo), copper (Cu), zinc (Zn) and chloride (Cl). The essential soil nutrients are not needed in equal quantities by plants. Some are required in large amounts; these are called macronutrients. Others needed in very small quantities are called micronutrients. The macronutrients include N, P, K, Ca, Mg and S. The micronutrients include Zn, Cu, Fe, Mn, B, Mo and Cl. Ordinarily 94 to 99.5% of fresh plant tissue is made up of C, H and O, which come from air and water, and only 0.5 to 6% is provided from soil constituents. Yet, it is usually one or more of the 13 essential nutrients

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supplied in the soil that limit crop growth.

Soils in the southern United States are naturally very low in nutrients due to aeons of weathering that has "washed" nutrients out of the soil. Because of this, humans have had to supply additional nutrients to promote plant growth. In the early days of the United States, nutrients were added by cutting down growing vegetation, burning it and using the nutrients in the ash as a source of fertilizer. The problem with this technique was that the nutrient supply was fairly small and rapidly consumed by the newly planted crops. Soon the soil was once again infertile, and new areas had to be cleared and burned. Slash and burn is still practiced in many third world countries and is the primary source of tropical forest destruction.

Agriculture in the United States has progressed beyond slash and burn fertilization. Before the commercialization of mined and manufactured fertilizers, animal manures were the primary source of most nutrients. These sources, while providing most needed nutrients, were difficult and expensive to transport over long distances. Thus, most manures were used near the site of origination. Another difficulty with manure, that has not been solved, is that the concentration of nutrients is not always in the right proportions or quantities necessary to meet specific plant or soil test requirements. This means that when manure is applied to meet the needs of one specific nutrient like N, other nutrients are often over or under applied. Commercial fertilizers have solved many of these problems. They are more concentrated and thus more economical to transport. Likewise, they can be tailored to meet the specific fertility needs of the site, so under or over application can be avoided.

This general introduction was intended to acquaint you with the key nutrients and where they come from; to show that the major forces involved in nutrient transport are wind, water and volcanic activity; and to show that man and animals also play a role in nutrient transport. From this point on, the movement of agricultural nutrients from a management and water quality perspective will be discussed.

There is no doubt that human actions have influenced nutrient movement. Advances in fertilizer and nutrient management have turned our native infertile soils into some of the most productive in the world. The general philosophy of fertilizer management is to make sure that no nutrient is limiting. When nutrients were inexpensive, this idea often led to higher applications than were necessary for plant growth. The same was true for manure applications. Research for many of the nutrients shows that even though the concentration of nutrients in the soil was increasing, there were no negative effects on plant growth. For these nutrients, the analogy was made that storing nutrients in the soil was similar to storing your money in a bank. It was there when you needed it. This concept only applies to nutrients that do not easily move in soils. These include P, Fe, Cu, Zn, Mo, Mn, Mg, Ca and K. Keep in mind that the macronutrients are needed in greater quantities, so P and K were the nutrients most often applied in excess. At high K additions, some plants exhibit salt injury, and producers learned early not to store too much K in their soils. The same, however, is not true with P; no toxic effects have been reported.

From an environmental perspective, P is the "immobile" nutrient most often associated with surface water pollution. This usually occurs from water erosion of soil where the P is attached or complexed. This form of P is often referred to as particulate P and can be controlled with soil conservation practices such as terracing, reduced tillage, grassed waterways and vegetative field borders. Particulate P concentrations per unit of soil can be high or low depending on the amount accumulated in the soil. Standard fertilizer management today includes a soil test to determine the concentrations present. If amounts found are adequate for optimum plant growth, no additional P is recommended. Soil testing is a best management practice (BMP) that producers should use. Soil tests in the southern United States usually only analyze for immobile nutrients.

In addition to particulate P, recent research efforts are focusing on soluble P. Soluble P is not attached or complexed in soil and, as such, can move with surface waters, even when conservation practices stop all soil movement. Soluble P movement is promoted by excessive applications of P that saturate the soil's ability to fix and retain P. Soluble P is also more available for biological use and as such can impact surface waters to a much greater extent than more complex forms of P.

Phosphorus itself is not a toxin and does not directly pollute water. However, it is a required nutrient for algae growth, along with N and C, and when P, N and C are present in abundance, the algae take advantage of the surplus and over reproduce. These are the algae blooms that are often seen in summer. Most algae growth is not toxic. But when algae die, organisms in the water decompose them and use oxygen from the water in the process. If oxygen concentrations in the water are low to begin with, levels can be lowered further to a point that cannot sustain aquatic life. Fish kills are one result. Nutrients, in this scenario, are harmful to water quality.

Ground water is less vulnerable to nutrient contamination than surface water because of direct competition by plants and soil organisms. Contamination can occur when mobile nutrients are leached through the soil quicker than they can be utilized. Nutrients that can move through soil include N, S, B and Cl. Mobile nutrients are also susceptible to surface movement through soil erosion or soluble water flow like P. Certain forms of three of the four "mobile" nutrients are also considered health hazards if they exceed levels established by EPA for drinking water. The elements and limits are 10 ppm for nitrate-nitrogen (NO_3-N), and 250 ppm for sulfate-sulfur (SO_4-S) and chloride (Cl). Of these, N is the primary concern.

Nitrogen can be present in a solid, liquid or gaseous state. Most forms of N are of little concern for human health. The air we breathe is 78% N gas. Nitrogen is also found in the form of ammonium (NH_4^+) and nitrate (NO_3^-) in the soil. Clay particles and humus in the soil contain negative charges that serve as weak "magnetic" forces. Molecules with positive charges are held by these forces and are less likely to leach to ground water. This is why ammonium rarely leaches. While this "magnetism" is an excellent mechanism to retain nutrients in the soil, some microorganisms use these forms and change their molecular state. This is what happens to ammonium. Soil organisms convert it into nitrate, which is a negatively charged molecule. Since nitrate is negatively charged and soils have a negative charge, nitrate is repelled by the soil and readily leaches. Nitrate is the primary source of N found in drinking water.

Nitrate-N concentrations above 10 ppm pose a health risk to human infants less than six months old. The digestive tracts of these infants have a higher pH than adults and support different microorganisms. When nitrate enters an infant's digestive tract it is converted to nitrite, which is extremely reactive. The nitrite in turn reacts with the oxygen-carrying hemoglobin in the baby's blood stream and depletes it of oxygen. This reaction forms a new low-oxygen-carrying compound called *methemoglobin*. If enough methemoglobin forms, the baby begins to suffocate from a lack of oxygen. The disease is called methemoglobinemia or "blue baby syndrome" and is now quite rare in the United States.

Increased nitrate concentrations can occur in groundwater when rainfall percolating through the soil leaches the mobile nutrients. Nitrogen can leach through porous soils even when applied according to BMPs. Nitrate contaminated ground water occurs mostly under sandy-textured soils and at sites with shallow water tables. In most cases, N sources exceeding the recommended rates or improper management practices have been cited. High nitrates in well water have also been associated with poor well construction and improper site location.

Best management practices for N include: matching rates with site specific yield capabilities, applying at proper times to coincide with crop uptake needs, optimal placement for efficient use and prevention of off-site movement with appropriate conservation measures.

ECOLOGICAL IMPACTS OF EXCESS NUTRIENTS IN THE ENVIRONMENT: ISSUES, MANAGEMENT, AND DECISION MAKING

By

K.H. Reckhow and C. Stow[1]

Introduction

Early water quality management activities in the United States focused on control of point sources of pollution, such as municipal and industrial wastewater treatment plants. This emphasis existed largely because these sources were most visible, most controllable with existing technology, and most likely to yield improvements in water quality when controlled. Recently, concern has shifted toward nonpoint sources of water pollution, such as urban and agricultural runoff. It is now realized that comprehensive water quality management involves both point and nonpoint source control.

One of the most common water quality problems that results from point and nonpoint pollution is nutrient enrichment, or eutrophication. This problem is usually caused by excessive inputs of the nutrients phosphorus and nitrogen to a waterbody (stream, lake, or estuary) and the consequent growth of algae and aquatic plants. Other changes, such as decreased water clarity and shifts in fish species, may also occur. Effective management of eutrophication frequently involves both reduction of point source inputs of phosphorus and nitrogen and land management practices that diminish phosphorus and nitrogen runoff.

The purpose of this paper is to describe the ecological effects of, and management approaches for, nutrient enrichment of surface waterbodies. This discussion begins in the next section by defining excessive nutrient levels (nutrient enrichment) and describing changes in an aquatic ecosystem that accompany enrichment. Following that, sources of nutrients from the watershed and from human activities are identified. Then, methods of planning and analysis, principally nutrient loading and lake response models, for managing eutrophication in lakes and reservoirs are discussed. The paper concludes with the framing of the nutrient management question in a broad decision analytic context. While the emphasis of this paper is on lakes, virtually all of the material is also applicable to other surface water bodies (i.e., rivers and estuaries).

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Nutrient Enrichment

Eutrophication, or nutrient enrichment, is characterized by the concept of trophic state. The trophic state of a lake or surface waterbody is determined by a complex series of physical, chemical, and biological interactions (Wetzel 1983). Two apparently similar waterbodies may respond quite differently under similar sets of conditions. A single lake of large area, or with many dendritic arms, may exhibit varying trophic conditions in different segments. While the degree of response of any specific waterbody to environmental factors affecting trophic state may be quite individualized, these factors, and their interactions, can be generalized in ways useful for management purposes.

For lakes and other waterbodies, trophic state is a classification based on the rate of organic matter supply. Lakes with a low organic matter supply rate are referred to as oligotrophic; those with a high rate of organic matter supply are classified as either dystrophic or eutrophic, depending on the source of organic matter. Dystrophic lakes receive a high rate of allocthonous, or externally produced, organic matter. The organic matter in eutrophic lakes is primarily autocthonous, or internally produced. It is generally the eutrophic waterbodies, or the process of eutrophication, which is of management concern.

Eutrophication is viewed as a natural part of the aging process of a lake. In the general scenario, as a lake ages it gradually fills in with sediment. Productivity, the rate of production of organic matter by algae and macrophytes, increases in response to an increased loading of nutrients. Nutrient loading, the mass of nutrient received by the lake per unit area per time, increases as the lake fills in, and the lake area decreases.

Cultural eutrophication, eutrophication induced by human activities, is usually a response to increased nutrient loading resulting from an increased supply of nutrients. The term eutrophication is often loosely used to mean anthropogenic nutrient increases, though this is not strictly correct. Nitrogen and phosphorus, the nutrients of primary concern, may be introduced into a watershed (the area of land that provides drainage into the waterbody of interest) from a variety of human pursuits. Point source discharges, urban and agricultural runoff, deforestation, or other changes in land management may result in increased nutrient supplies to a lake.

Trophic state is influenced by many things in addition to nutrient loading. Sunlight is another important factor. Sunlight varies seasonally and with latitude. In north temperate waterbodies, eutrophication is usually associated with warm weather conditions when incident sunlight is strongest. In tropical and subtropical regions, problems associated with eutrophication may extend beyond the summer season.

The penetration of light into the water influences the productivity and hence the trophic state of a waterbody. Even in very clear lakes, light is attenuated as it passes through the water. The presence of dissolved substances and suspended particulate material further impedes the penetration of sunlight. As light strikes suspended sediment or phytoplankton it is absorbed or scattered, thus decreasing the depth to which it penetrates. The amount of scattering of light by suspended matter is measured as turbidity.

As a result of the decrease in available sunlight with depth, due to attenuation, productivity also decreases. The vertical zone near the lake surface, where photosynthesis occurs and productivity predominates over respiration, is called the trophogenic zone. The zone near the lake bottom where respiration occurs in excess of photosynthesis is the tropholytic zone. The theoretical depth at which the rate of photosynthesis and respiration are equal is called the compensation depth.

The attenuation of light with depth may also establish vertical mixing zones in a lake. As light is absorbed in the surface water, this water is warmed and becomes less dense. The deeper water, which receives less light, remains cooler and more dense than the surface water. If the density difference becomes great enough, the surface and bottom waters may become effectively isolated from one another. This is a very common seasonal phenomenon in temperate lakes and is referred to as summer stratification. The upper mixed layer is referred to as the epilimnion and the bottom layer is called the hypolimnion. Stratification can play an important role in the partitioning, and hence the availability, of nutrients in a waterbody.

Basin morphometry, or the physical dimensions of a lake, plays a role in determining lake productivity. Deep lakes tend to be less productive than shallow lakes. For a given nutrient loading rate, a deeper lake will have more dilution, and subsequently lower nutrient concentrations than a shallow lake. Also, the solar energy available for photosynthesis is less, per unit volume of water, in deeper lakes.

Residence time, or hydraulic detention time, is a measure of the average amount of time that water spends in a lake. Residence time is determined by the volume ratio of inflows to outflows and the lake volume. Lakes with longer residence times tend, generally, to be less eutrophic than those with shorter residence times. A longer residence time allows those processes which remove nutrients from the water to have more time to act, resulting in lower nutrient concentrations.

Clearly, the preceding ideas are all interrelated. Deeper lakes will tend to have longer residence times. Lakes of greater volume will tend to have longer residence times, and lower nutrient loading. However a shallow lake with a long residence time may be very eutrophic. A relatively deep lake with a short residence time may also be very productive. The shape of the lake basin is important in determining how the various physical features interact to influence trophic state.

Shape will also affect the spatial heterogeneity in a lake. Shallow areas near the lake edges may be more eutrophic than the deeper middle areas. Isolated branches may be more (or less) eutrophic than the main body of the lake.

This is not an exhaustive discussion of all factors affecting productivity in a lake or other waterbody. Temperature, micronutrients, water chemistry, toxins, and many other environmental features interact to determine trophic state. The characteristics mentioned, however, are those generally found to be of primary importance and are useful in predicting lake responses to a range of management scenarios.

Effects of Excess Nutrients on Human Activities

Eutrophication resulting from human activities in a watershed can result in the impairment of designated uses in a waterbody. Often the mismanagement that has brought about enhanced eutrophication has also resulted in other forms of water pollution, with effects that may be difficult to separate from one another.

Drinking water can be adversely affected by enhanced algal growth. Some genera of blue-green algae produce toxic compounds. Even after treatment, water may be left with an off-color and an unpleasant taste or odor. Silting of reservoirs is also accelerated by an abundance of algal productivity, resulting in decreased reservoir capacity. In addition, algal growth may cause fouling problems in water intakes, as well as in some industrial equipment.

A decline in commercial and recreational fisheries may be associated with increased eutrophication. Desirable species are often replaced by less desirable "trash" species in aquatic systems with advanced pollution problems.

Nuisance and aesthetic problems are the most often cited impairments associated with eutrophication. Scums and algal mats are unpleasant for many forms of water recreation. An abundance of rooted or floating vascular plants may obstruct navigation and inhibit boating as well as primary contact recreation. Increases in some types of obnoxious insects may also be associated with increasing eutrophication.

Besides the effects of eutrophication which may be considered undesirable because of direct impact on human activities, a number of more subtle ecological changes may occur. Significant change in the algal and macrophytic community of a waterbody may alter the other populations of lake biota as well. A noticeable change in the commercial and recreational fisheries is only a symptom of a changing biotic regime. Accompanying communities of [illegible], birds, and reptiles are likely to be modified, in response to the changing environment.

Physical and chemical changes may occur in a waterbody in response to eutrophication. [illegible] abundance of phytoplankton affects the absorption of sunlight. Light is less able [illegible] the water, resulting in a concentration of productivity near the lake surface. As less [illegible] able for heating at greater depths, warming may occur nearer the surface, resulting [illegible] epilimnion volume and increased hypolimnion volume.

pH in a waterbody can be affected by accelerated productivity. During the day, [illegible] producers are actively photosynthesizing, CO_2 is removed from the water. This [illegible] diurnal pH cycles with maximum pH levels in the 9 - 10 range.

[illegible] diurnal cycle of dissolved oxygen often occurs. During photosynthesis the [illegible] from algae and macrophytes may result in supersaturation of the water. At [illegible] respiration occurs, dissolved oxygen levels can drop to low levels. Extended [illegible]ast during a bloom can cause acute oxygen depletion, and possibly fishkills. [illegible] decay of large algal mats can exacerbate problems of dissolved oxygen

Eutrophication of a previously oligotrophic system may result in imposition of water use restrictions for a community. Ultimately this may mean economic losses via direct cost outlays, or opportunity losses. Additional treatment to alleviate taste and odor problems in drinking water supplies raises treatment costs. Chemical treatment of lakes to inhibit algal growth or macrophyte harvesting to maintain waterways require extra expenditures. Restrictions on swimming, boating, fishing, or other recreational or commercial activities may all mean direct or indirect revenue losses. In short, cultural eutrophication may impact the economic health of a community, as well as the ecological health of a waterbody.

Nutrient Sources

Nutrients and other pollutants of importance in lake management may originate from a variety of human activities. Industrial and municipal point source discharges are important nutrient sources for many aquatic systems. Nitrogen and phosphorus industrial byproducts are found in many effluents. Municipal discharges of treated or partially treated sewage contain large amounts of nitrogen and phosphorus.

Point sources are, in principal, relatively easy to control. Implementation of process controls in industrial facilities, phosphate detergent bans by municipalities, or end-of-pipe treatment can minimize the effects of discrete, identifiable pollution sources.

Land use practices, resulting in nonpoint source discharges of nitrogen and phosphorus, are more difficult to manage. Deforestation, or other vegetative removal, has been shown to cause increased nutrient loading in a watershed. Phosphorus in particular is strongly associated with soil particulate matter. The removal of vegetative cover promotes erosion and increases both the sediment and phosphorus load in proximal aquatic systems. Terrestrial vegetation contains a large nutrient pool in some ecosystems. The destruction of this vegetation can release the stored nutrients and make them available for leaching or runoff.

Agricultural practices can be important nonpoint sources of nutrients. Nitrogen and phosphorus fertilizers are applied in large quantity to many commercial croplands. Under poor management they may contribute to the nutrient load of a basin. Commercial feedlots and other intensive animal production facilities also produce wastes of high nutrient content, which often enter the watershed via surface or subsurface runoff.

In some waterbodies, internal nutrient sources may be important at some times of the year. In seasonally stratified lakes, nutrients can accumulate in the hypolimnion, as detrital matter precipitates from the epilimnion and decays. When the lake destratifies and mixes, the sudden flux of nutrients to the surface often results in an autumn algal bloom.

The bottom sediment of a waterbody may sometimes act as a nutrient source. Sediment interstitial water is generally very high in nitrogen and phosphorus compounds. Turbulent resuspension of the sediment or diffusion through the sediment-water interface may be mechanisms important in the nutrient supply of some lakes.

Additionally, in many lakes nitrogen is fixed from the atmosphere by blue-green algae. This means that some of the blue-green algal species are able to use atmospheric nitrogen as a nutritional source; clearly this can provide a competitive advantage under certain conditions.

While nitrogen and phosphorus are both important in determining the trophic state of an aquatic system, it is usually phosphorus that receives more attention in pollution abatement programs. There are two principal reasons for this. In most lakes, phosphorus is considered to be the limiting nutrient. That is, of the nutrients necessary for algal and macrophytic growth, there is usually a deficit of phosphorus. This may not be true in estuaries and is not always the case in lakes. In many instances, however, limiting the supply of phosphorus will alleviate the problems associated with eutrophication.

The other reason for focusing on phosphorus removal is that, for many effluents, it is easier to provide treatment for phosphorus removal than nitrogen removal. Phosphate detergent bans, coupled with additional treatment of municipal sewage, have proven effective for relieving the effects of eutrophication in some lake systems.

Eutrophication Management: Methods of Analysis

The study of eutrophication problems often involves an assessment of nutrient loading from the various land uses in the watershed. For existing land uses, direct measurement of nutrient loading is usually the best method for this purpose. When land uses do not yet exist and are projected for the future, then direct measurement is impossible; in that case, predictive modeling, typically using mechanistic simulation models or simple approaches involving nutrient export coefficients, is required. Each of these modeling approaches for estimation of nutrient loading has its uses and its limitations.

Mechanistic simulation models of nutrient runoff are based on the modeler's understanding of important processes; this understanding is translated into mathematical terms for the simulation model. Most of these models must also describe the rainfall-runoff process since the hydrologic cycle is important in nutrient transport. In addition, sediment erosion and sediment runoff play a key role in nutrient runoff. Phosphorus will readily bind to sediment particles, and thus sediment runoff and phosphorus runoff are often closely linked. Nitrogen, on the other hand, tends to remain soluble and is not so closely associated with sediments.

Among the more commonly used mechanistic nutrient runoff models are: (1) NPS (nonpoint source model; Donigian and Crawford 1976), ARM (agricultural runoff model; Donigian et al. 1977), and HSPF (hydrologic simulation program - fortran; Johanson et al. 1984) supported by USEPA, and (2) CREAMS (chemicals, runoff, and erosion from agricultural management systems; Knisel 1980) and AGNPS (agricultural nonpoint source pollution model; Young et al. 1987) supported by USDA. NPS, ARM, and particularly HSPF and CREAMS are large, detailed models that require a considerable amount of site-specific calibration data and modeler experience to run. These models can be used in principle to predict nutrient runoff from watersheds at short (single storm) and long (year) time frames associated with a variety of land use changes and practices (e.g., tillage practices). Unfortunately, the limited rigorous testing that has been undertaken for these models indicates that prediction errors often are quite large (particularly for short time frames).

AGNPS is a relatively simple mechanistic model that is attractive because it accounts for small scale spatial variability in a watershed. This allows linkage with GIS (geographic information systems) data bases and also supports watershed planning for the identification of locations in the watershed that are apt to be the major (and minor) contributors of nutrient runoff. Spatially- distributed models like AGNPS are likely to be emphasized in future research activities.

In addition to mechanistic models, the other frequently-used approach for nutrient runoff modeling is application of nutrient export coefficients. A nutrient export coefficient, or unit area nutrient load, is an estimate of the nutrient mass per unit time per unit area in watershed runoff. It is typically reported on an annual basis as a land-use-specific estimate (e.g., 1.0 g/m^2-yr of phosphorus from row crop agriculture). Information on nitrogen and phosphorus export coefficients is available in Reckhow et al. (1980), which contains lists of these coefficients, guidance on the selection of export coefficients, and discussion of monitoring designs for developing site-specific export coefficients. Annual export coefficients do not offer the temporal and mechanistic detail of the simulation models. On the other hand, they are simple to apply and understand, and uncertainty analysis is a relatively straightforward exercise.

Nutrient runoff modeling yields an estimate of nutrient input to a surface waterbody; to estimate how that waterbody responds to the nutrient input, a receiving waterbody model is needed. These models tend to be specific to the waterbody. That is, there are lake, river, and estuary models. Lake eutrophication models, which tend to be either simple statistical (i.e., regression) models or mechanistic simulation models, are by far the most common. As in the case of mechanistic nutrient runoff models, the mechanistic eutrophication models (see Chapra and Reckhow 1983) can provide substantial space, time, and ecologic detail, perhaps at the expense of prediction accuracy. The statistical models (see Reckhow and Chapra 1983), on the other hand, yield predictions of aggregate response, but may have lower prediction error in this modest task than do the large mechanistic models for their purposes. WASP4 (water analysis simulation program, version 4; Ambrose et al. 1988) is an example of a mechanistic lake/river/estuary eutrophication model, and the Vollenweider loading plot (see Reckhow and Chapra 1983) is an example of a statistical lake model.

Eutrophication Management: Planning and Decision Making

Too often in the past, conventional practice for eutrophication management has involved relatively little time allocated to identifying and agreeing upon program objectives. Typically, a few obvious objectives are chosen quickly, and most of the effort is then devoted to data gathering, scientific research, and analysis. For example, it has been common practice for a lake eutrophication management study to be focused on quantifying the relationship between nutrients (phosphorus and nitrogen) and algal density (e.g., chlorophyll concentration), because it can be completed in a reasonably satisfactory manner. Thus, attention is focused on a relatively well-studied expression, when a thoughtful consideration of objectives and attributes might have identified algal bloom stimuli as of greatest uncertainty in need of clarification. In this case, the result of inadequate attention to the objectives is an incomplete analysis or an analysis of the wrong problem.

As a better approach, decision analysis (Keeney and Raiffa 1976, von Winterfeldt and Edwards 1986) provides a logical structure for the analysis of complex decisions, such as the management of lake eutrophication. For decisions with multiple issues or multiple objectives, the problem is first decomposed into single objectives and attributes. These attributes are used to measure the degree to which an objective is achieved by a management option; attributes should be meaningful to the issue, measurable, predictable, comprehensive, and nonoverlapping. Identification of objectives and attributes leads to consensus with regard to the issues of concern to the decision maker(s). Subsequent analysis should focus on estimating the effects of various management actions on the levels of the attributes, perhaps using simulation models. Ideally, the analysis should quantify all uncertainties, which can be expressed probabilistically. A measure of value, utility, or net benefit must be assigned to each outcome, and attributes must be weighted for importance with respect to each other.

In principle, when the system state probabilities are combined with a utility function, the minimum-risk management strategy can be identified. In many instances, a final solution within the decision theoretic framework is not necessary, as the primary value is the insight and understanding provided by the analysis.

The criticism of much current practice relates to vague objectives such as "improve trophic state" or often ill-conceived objectives such as "reduce phosphorus loading below the 'dangerous' criterion on the Vollenweider loading plot so that the lake becomes mesotrophic." In most cases, these objectives do not pertain directly to use and enjoyment of the lake, unless water quality standards are explicitly expressed in terms of trophic state. However, as a rule, lake water quality standards either do not exist or concern chlorophyll a or dissolved oxygen concentrations. Meanwhile, use and enjoyment of the lake is apt to be more directly related to quantity and quality of fish populations, occurrence of floating algal mats, or extent of aquatic plant growths than to the publicly-vague term "trophic state."

An Example: Decision Analysis for Lake Okeechobee

Over the past several years, Lake Okeechobee in Florida has been one of the most studied lakes in the world. Farming, fishing, drinking water, aesthetics, and everglades restoration are among the issues which the South Florida Water Management District is considering in the development of a scientific basis for a management plan. As stated above, this effort should begin with a thorough consideration of management objectives and attributes, expressed in an *objectives hierarchy*.

For the management of eutrophication in Lake Okeechobee, an objectives hierarchy has been constructed and is presented in Figure 1. This begins with an all-encompassing objective at the top; a comprehensive set of issue-specific objectives is then derived that is consistent with the overall objective. Finally, attributes (identified by the arrowheads in the figure) that are meaningful, measurable, and can be predicted are derived for each specific objective.

Attributes provide the essential link between the program objectives or policy and the information needs. If decisions are to be made based on attribute levels, then the attributes must be meaningful to the decision maker. For example, chlorophyll a is not a meaningful attribute to decision makers, unless an ambient water quality standard for chlorophyll a exists. As conveyed in Figure 1, a meaningful attribute for nuisance algal growths is more apt to be "surface algal blooms" or "floating algal mats" or some other measure of direct concern to the public. While attributes like these are more difficult to understand scientifically and predict, they do reflect public value or utility, and thus they will be a measure by which the public assesses the success of a management program. The decision maker should translate all objectives into meaningful attributes like that above, and then present these attributes to scientists/engineers as indicative of the specific information needs for the problem under study.

At the same time, careful consideration must be given to selection and evaluation of management options. For the management of Lake Okeechobee with respect to eutrophication, Table 1 provides a list of some of the options that have been proposed. Across the top of the table are the attributes identified through the development of the objectives hierarchy in Figure 1.

The next step sounds straightforward but is extremely difficult to do well - fill in Table 1. The entries to the body of the table represent what each management option achieves for each attribute. Thus, for example, the table cell for the intersection of "herbicide treatment of aquatic weeds" (management action) with "species and space/time coverage of aquatic plants" (attribute) should contain an assessment (with uncertainty estimated) of the level of the attribute expected if the particular management strategy was implemented.

What will this assessment exercise achieve? At this point, given the previous work on Lake Okeechobee, a decision analytic assessment provides a framework to organize existing information with a focus on: (1) evaluation of management options, and (2) prediction of meaningful quantities. This helps to clarify: (1) what is known about quantities of interest, (2) where the critical uncertainties are, and (3) if it is wise to implement a management strategy now, or defer action and gather more information.

In many cases, decision analysis should not be looked to for the optimal management solution. Rather, the decision analytic *process* should help decision makers reach decision themselves by clarifying issues and focusing attention. For example, hard thinking about objectives and attributes leads to:

- key questions ("What features of algal growths really matter?")

- answers ("The time and space scale of surface algal blooms, particularly for blue-greens, is of greatest concern.")

- management options ("What can we do to reduce the time and space over which surface algal blooms occur?")

◆ assessment methods ("Can we use a mechanistic simulation model to predict the time/space extent of algal blooms, or are we better served with simple models and expert scientific judgment for interpretation?").

Assessment then leads to:

◆ predictions ("What is expected for space/time surface algal blooms if a specified set of land use restrictions is implemented?")

◆ uncertainties ("How uncertain are the predictions?")

◆ conclusions ("Do we know enough about impact to act now, or should more information be obtained?")

◆ decisions ("Given expected costs, environmental impacts, and uncertainties, should the specified set of land use restrictions be implemented?").

All of these tasks can be aided by the decision analytic framework.

Final Thoughts - A Parable for Eutrophication Management

Not very long ago in the Great Water Kingdom of the South, the people and the land flourished. Humans were masters over all they encountered - rivers that once flowed in tortuous, crooked paths were straightened, lakes were dammed to control great floods, and the swamps were drained to create a lush, sweet growth. People came from everywhere, first to visit and then to stay. Cities rose, initially in the sand and then in the swamp. Paradise was born.

But, as time passed, it became clear that all might not be well in the Great Kingdom. The people were restless. They were concerned that the green growth on the land was spreading to the Great Lake. The fish in the Great Lake, the birds, plants, and animals in the Great Swamp - all seemed to be affected. Had something happened? What? Should something be done? What?

The Queen assembled her advisors from throughout the Kingdom, and the people watched and listened.

"The farmers, the farmers!" exclaimed those dressed in green. "They enriched their land with phosphorus and nitrogen so that crops grow strong and tall and lush, and the cows produce great quantities of milk, but the phosphorus spreads and enriches the Great Lake as well."

"That's right," said the renowned scientist from the Halls of Knowledge. "We know that phosphorus stimulates growth on land and also in water."

Turning to the Queen, he continued. "Your majesty, we have identified the culprit, but we need to understand better its role in the Great Lake. What becomes of the phosphorus once it enters the Great Lake? Does it sink to the bottom? If it sinks to the bottom, will it rise into the water again? What effect does the wind have? We must understand the science!"

"But what science must you understand in order to manage the lake responsibly?" A barely audible voice was heard to ask from the audience. However, this question did not reach the Queen's ears, as the scientists from the Halls of Knowledge spoke eagerly in a babel of voices about the scientific knowledge to be gained, and the resources of the Kingdom that would be necessary to carry out this work.

And the people watched and listened.

Finally, a member of the audience could contain his impatience no longer.

"When will you ask us what we care about?" he inquired. "I can tell you this - we don't care about phosphorus; we just want the fish to catch, the water to drink, and the Great Lake again to be beautiful. Who cares about phosphorus?"

"But don't you see?" said one of the Queen's advisors. "Phosphorus enriches both the land and the water. If we understand the science concerning the phosphorus that enters the Great Lake from the farmlands, then we can make the lake well again."

"OK, that's what we want to hear," another in the audience said, "the lake will be well. So what can we expect the fishing to be like? And how will the lake look?"

"Whoa!" cautioned the Queen's advisor. "That's not in the science we propose to undertake. But don't worry, if we just reduce the phosphorus to a nice round number, all will be well!"

"No, the citizen is right," countered another member of the Queen's advisors. "We must have a scientific basis for the management decision, and here it is." The advisor directed everyone's attention to a graph, on which were drawn two parallel upward-sloping lines, one labeled "dangerous" and the other labeled "permissible."

"This graph presents 'Vollenweider's Phosphorus Loading Criterion,' " he said. "The graph reflects the knowledge of the great limnologist, Richard Vollenweider. After many years of study, Dr. Vollenweider proposed a simple relationship that could be used to determine when too much phosphorus was entering a lake. Dr. Vollenweider presented this relationship as a graph, and he labeled the transition to excessive phosphorus enrichment as 'dangerous.' Because of the farms, the Great Lake is now in the dangerous zone. Clearly, we ought to avoid dangerous conditions," he said with authority. "We must reduce the phosphorus inputs from the farms so that the Great Lake is no longer in a dangerous condition!" The advisors nodded in agreement.

"Then it's settled," said the Queen. "I will direct the Royal Treasurer to provide the necessary resources to the Halls of Knowledge in the name of good science, and we will immediately implement a plan to control phosphorus runoff from the farms so that the current dangerous situation is eliminated."

"But what does this mean for fishing, for swimming, for drinking, and for just plain enjoyment?" said a voice that was lost in the din of the excitement of the new scientific knowledge to be gained and the satisfaction in a management plan that seemed almost too simple to believe.

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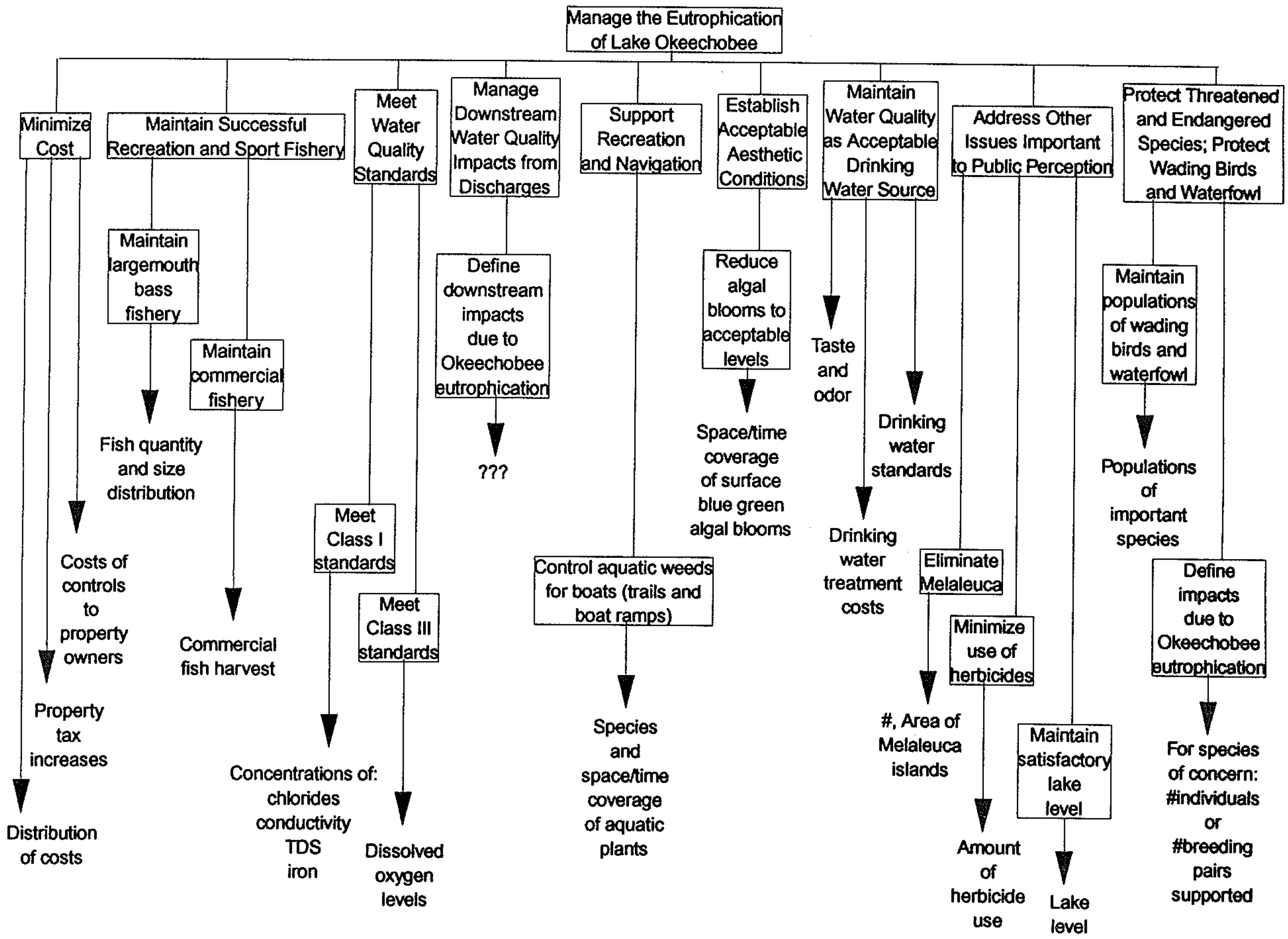
Figure 1. *Objectives hierarchy for the management of eutrophication in Lake Okeechobee.*



levels herbicide use Lake level supported

Table 1. *Management actions and attributes for the eutrophication of Lake Okeechobee.*

Management Options	Attributes (Measures of Effectiveness)																
	Distribution of Costs	Property Tax Increases	Costs of Controls to Property Owners	Fish Quantity and Size Distribution	Commercial Fish Harvest	Conc. of: Chlorides Conductivity TDS Iron	Dissolved Oxygen Levels	Species and Space/Time Coverage of Aquatic Plants	Space/Time Coverage of Surface Algal Blooms	Taste and Odor	Drinking Water Treatment Costs	Drinking Water Standards	#, Area of Melaleuca Islands	Amount of Herbicide Use	Lake Level	Populations of Important Species	For Species of Concern: #Individuals or #Breeding Pairs Supported
Restrict Land Use for Nutrient Load Controls																	
Lake Water Level Regulation																	
Mechanical Harvesting of Aquatic Weeds																	
Herbicide Treatment of Aquatic Weeds																	
Biological Controls (insects) for Aquatic Weeds																	
Prescribed Burning of Littoral Zone																	
Restrict Point Source Discharges																	
Implement Stormwater Management Programs																	
Biomanipulation for Algal Bloom Control																	
Reservoir Diversion																	

EXCESS NUTRIENT CONTAMINATION AND HUMAN HEALTH CONCERNS

By

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The first and overriding principle in the field of toxicology is that all substances are poisons; it is the dose that makes the poison. While it is usually very easy to understand that high levels of toxic chemicals can cause us harm, it is a bit less obvious that we can be poisoned by chemicals which have beneficial effects. Therefore, more is not always better, even when considering such "healthy" compounds as nutrients and vitamin tablets.

I have been asked to address the issue of potential adverse human health effects associated with excess nutrient consumption. I will focus on two nutrients intimately associated with agriculture, namely phosphorous and nitrate.

The main source of human phosphate poisoning does not result from the traditional use of phosphorous-containing fertilizers. The phosphorous contained in these formulations is converted to phosphate ions in the environment, and these bind tightly to sediments and soil particles and are therefore not typically available for human exposure in water, except in cases of extremes in pH. Instead, individuals involved in the manufacture of fertilizers, as well as those individuals manufacturing or frequently handling yellow phosphorous-containing rodenticides, are the humans at risk for phosphorous poisoning. These individuals may exhibit degenerative bone changes, especially of the jaw.

On the other hand, nitrate ion, another nutrient found in synthetic fertilizers as well as in manures, can readily contaminate water supplies. The ions are very water soluble and leach readily into ground water if applied in excess of the needs of a growing crop or to soil containing low amounts of organic matter. Septic systems and decaying organic matter are two other major sources of nitrate. Regardless of the source of the nitrate ion, the drinking water standard set for nitrate is 10 ppm (USEPA, 1987). Natural decay can contribute up to 3 ppm nitrate nitrogen to otherwise uncontaminated water supplies (Madison and Brunett, 1994), and limited studies across North Carolina suggest that the natural background level of nitrate nitrogen is generally at or below 1 ppm (Campbell, 1993).

Regardless of the source of nitrates, they undergo the exact same reactions in a biological system (Figure 1). Nitrates are converted in biological systems to nitrites. In humans, this conversion occurs in the saliva and in the stomach and is actually performed by

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microorganisms. Nitrites react readily with secondary amines to form a class of chemicals called nitrosamines; nitrosamines have long been associated with cancers of the brain and stomach.

Figure 1:

NO3 ——>	HO-N=O	+	R2N-H ——>	R2N-N=O+water
(nitrate)	(nitrite)		(secondary amine)	(nitrosamine)

While infants probably receive most of their nitrate intake from their water, diet is the greater contributor of nitrate in the adult human (USEPA, 1987). Nitrates can enter the food chain from sources other than contaminated water. Vegetables, particularly leafy greens, contain high levels of nitrates (Table 1). Cured meats are another major source of nitrate and/or nitrite in the human diet, with published values for the concentration of nitrates and nitrites in cured meats ranging from approximately 20-675 and 0-100 mg/kg respectively (Walker, 1990). Nitrates and nitrites have been used for many years to preserve foods; it is interesting to note that the incidence of stomach cancer (one of the most frequent sites of nitrite-induced cancer in laboratory animals) in humans in this country has steadily decreased as we have replaced preserved food with fresh foods stored under refrigeration (Public Health Service/NIH).

There are two main areas of concern with regard to nitrates and human health. In infants under the age of approximately 6 months, the composition and the contents of the stomach are considerably different that those of the adult. Infants tend to have a higher pH in their gastrointestinal tracts, producing a more favorable environment for the growth of microorganisms than in the older child or the adult (Amdur, Doull and Klaassen, 1991). These microorganisms convert nitrate into nitrite. In the infant, nitrite is absorbed from the gastrointestinal tract into the bloodstream, where it binds to the oxygen-carrying molecules of the blood (hemoglobin), forming a complex known as methemoglobin. Methemoglobin is brown in color, in contrast to the bright red color of hemoglobin which is carrying oxygen. Methemoglobin is incapable of carrying oxygen, so an infant suffering from methemoglobinemia develops a blue tinge to his skin and mucous membranes (Amdur, Doull and Klaassen, 1991); hence, the common (and much more pronounceable) name for this condition is "blue baby syndrome." This condition can be fatal in infants, but is reversible by the intravenous injection of a dye called methylene blue (Amdur, Doull and Klaassen). Humans over the age of 6 months are less at risk of this syndrome as the acidity of their gastrointestinal tract increases and the survival of the microorganisms which convert nitrate to nitrite markedly decreases.

The increase in gastrointestinal tract acidity results in dramatically reduced conversion of ingested nitrate to nitrite; infants convert 100% of ingested nitrate to nitrite, whereas adults convert only 10% to nitrite (USEPA, 1987).

Table 1. Average concentrations of nitrate in fresh vegetables[1]

Vegetable	mg/kg/Fresh Weight	mg Nitrate/4 oz serving
Turnip	9040[2]	1025
Melon	4932	560
Beets	3288[2]	373
Celery	3151	357
Rhubarb	2900	329
Radish	2600	295
Spinach	2470[2]	280
Lettuce	2330	264
Endive	1780	202
Parsley	1380	156
Kale	1096	124
Broccoli	1014	115
Cabbage	712	81
Leeks	700	79
Cauliflower	658	75
Carrot	274	31
Onion	235	26
Mushroom	219	25
Pepper (sweet)	165	19
Cucumber	151	17
Potato (white)	150	17
Turnip (root)	80	9
Tomato	80	9
Potato (sweet)	65[2]	7
Corn	62[2]	7
Okra	52	6
Green beans	46	5
Peas	40	4

[1] Condensed from Walker, 1990
[2] Also contains > 1 mg/kg nitrite

Nitrites produced in the stomach can have other serious consequences upon their reaction with secondary amines, which are ubiquitous in the proteins we eat. As noted above, nitrosamines are potent carcinogens. Of the over 120 nitrosamines tested in rodent bioassays (2 year cancer tests), over 75% have proven to be carcinogenic (Shank and Magee, 1981). In a study of humans with precancerous stomach lesions, there was a high degree of correlation between elevated stomach pH, nitrite content of the stomach, and precancerous lesions of the stomach (Chen et al., 1990). Nitrosamines can pass through the placenta, causing toxicity, birth defects, and/or cancer in the offspring (Ivankovik, 1979).

If nitrates are so prevalent in our diet and nitrosamines are such potent carcinogens, then why don't we all develop cancer related to the ingestion of these compounds? It is encouraging to note that an adequate intake of vitamins C and E, as well as the presence of some food preservatives which act as antioxidants (e.g. BHA, BHT), can inhibit the formation of nitrites from nitrates, thus preventing methemoglobinemia in infants and reducing the risk of cancer (Archer et al., 1975; Kamm et al., 1977). In fact, in nitrate/nitrite-cured meats (bacon, hot dogs, country ham), BHT and vitamin C must also be included as ingredients in these products to prevent nitrosamine formation (NC Primary Health Care Association). Other anti-cancer agents are present in fruits and vgetables; indole-3-carbinol, found in cruciferous vegetables, reduced the incidence of a nitrosamine-induced tumor by 50% in rats (Dragsted, Strube and Larsen, 1993).

In summary, overconsumption of nitrate ion by human infants and is probably the scenario of most concern with regard to human health effects of excess nutrient contamination. Minimizing water contamination due to nitrate by proper construction and maintenance of wells and septic, proper and timely application of fertilizers, and routine monitoring of private drinking water supplies will assure the protection of this sensitive population. On the other hand, because of genetic disorders which prevent recovery of adult methemoglobin and because of the historical link between heavy consumption of cured meats and stomach cancer, it also makes good sense to limit adult exposure to excess nitrate whereever possible. This does not, however, apply to reduction in the consumption of fresh fruits and vegetables, as well as an adequate protein intake. The benefits of eating fresh fruits, including vitamins, minerals, fiber, and anti-carcinogenic agents, far outweigh the potential health risks from the nitrates that they contain. A diet adequate in vitamins C and E has a marked protective effect against the cascade which begins the process of nitrosamine formation and the subsequent risk of nitrosamine-associated cancers.

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NUTRIENT MANAGEMENT:

THE CONGRESSIONAL SETTING

By

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Introduction

Yesterday, we talked about scientific aspects of nutrient management. Yesterday's discussions conveyed a strong sense of certainty about answers to many of the key scientific questions about nutrients in the environment and about technologies to prevent or reduce nutrient problems. This certainty is in sharp contrast to the various opinions being expressed in the current debate over alternative public policies to address nutrient management problems.

Nationally, the debate is no longer over the need to address nutrient management questions. Nonpoint pollution sources contribute to more than half of the remaining water quality problems, and nutrients are now the greatest cause of nonpoint pollution in lakes and ponds, and a significant problem in estuaries, according to information compiled by the Environmental Protection Agency. In addition, a national well survey conducted by EPA between 1988 and 1990 detected nitrates in a majority of the sample sites.

But two qualifications complicate the development of public policies to address what appears to be a straight-forward conclusion. First, the nutrient problem is not uniform across the landscape, for a wide variety of reasons that others can articulate better than I. Because of this heterogeneity, one solution will not effectively address all problems. For an environmental protection program, this implies a flexible approach, implemented below the Federal level.

Second, while there is little disagreement over the information that underlies the call for action, there is considerable disagreement over what approach to take to reach the desired solution, given current programs and problems, and the cultural setting. At the broadest national level, Water Quality 2000: A National Water Agenda for the 21st Century, and other reports prepared recently as fodder for the Clean Water Act reauthorization contain similar recommendations. These reports outline a similar litany of key aspects of needed policy, such as the roles at the Federal and State levels in agricultural pollution prevention, the division of labors among EPA, USDA, and other Federal agencies, and the need for nutrient management and planning at the farm level. In short, most of the questions today do not revolve around

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whether we have a problem and what should be our policy goals, but rather how do we effectively address the problem to achieve these goals.

The Political Context of the Nutrient Management Debate

The context of the nutrient management debate needs to be articulated if one is to understand the current deliberations. Many recent changes affect aspects of this issue. These changes need to be appreciated as they place the current debate in context. More changes are likely, so the context will continue to change as the policy process works toward a resolution. Many of the most important changes can be placed in two categories: institutions and issues.

Institutions

Federal institutions and their policies and programs are changing rapidly, with many proposals pending for still further change. While nutrient management has yet to get caught up in the change, it is likely to happen because relevant legislation will be debated soon, at a time when many of these changes could start to have a noticeable effect.

In Congress, the 103rd Congress is far different than the 102nd, with new members, new leaders, and turnover in key staff. More than 100 members are new to Congress this year. Necessarily, it takes time for them to learn how the institution operates and what they can accomplish within its rules and procedures. New and old members alike have new committee assignments, and they are learning about new issues and about how their constituents view these issues. Many of the key staff in the majority are moving to jobs in the new Administration, taking their knowledge and institutional memory off Capitol Hill.

All these changes tend to slow the legislative process. New committee chairs are inclined to hold hearings on the same topics that their predecessors addressed to refocus legislative proposals and debate to reflect their particular interests. The overall result is that little has moved through Congress, unless it has been a high priority for the new administration. At the end of the first session, the new administration has been given generally high marks for pushing through its legislative priorities, but subjects that were not on that priority list, including water quality legislation, have languished. While Senator Baucus promised to have clean water legislation out of the full committee by the end of the first session earlier this year, a bill has yet to emerge from Senator Graham's Clean Water Subcommittee.

Second, budget constraints are being more widely recognized and respected. The present budget agreement allows less wiggle room for funding programs, and given the debate at the end of the first session over further major reductions in programs and the Federal work force, it seems likely that flexibility will continue to shrink. "Frugal" is certainly in. Just remember the way funding for the supercollider was debated and vanquished -- it wasn't over whether the science was good or bad. Rather, even admitting that it was good and potentially valuable science, the debate was decided over the question of necessity versus extravagance. In the future the distinction between what is socially-desirable and what is affordable will continue to change, with many current programs being reduced or terminated because they are too expensive.

In the agriculture sector, the demise of the wool, mohair, and honey programs has been a very visible part of this debate. In these cases, the programs could not be justified against the double whammy of not only being viewed as outdated and socially irrelevant, but also as benefitting small constituencies in few congressional districts.

Third, a number of pending organizational changes in the executive and congressional branches could affect how the nutrient management question might be addressed. In the executive branch, these include USDA reorganization (including creation of a Farm Services Organization, and the degree to which the conservation effort is either within or separate from such an agency), elevation of EPA to the cabinet level, creation of an Office of Environmental Policy at both the White House and at USDA, and the possible termination of the Council on Environmental Quality. One key to how nutrient management will be addressed is how USDA is reorganized. USDA must be able to successfully claim it can do the job and counter the doubters from the environmental and consumer interest blocks who will say that environmental conditions must improve more rapidly and conclude that EPA and state environmental agencies should play a stronger role, relying on regulation in particularly difficult situations.

Congress is debating its own reorganization. Some of the changes under consideration would reduce the number of committees and subcommittees or alter committee responsibilities. How much change will occur in Congress is very unclear now, and with so little unity over direction by congressional leaders at this time, it seems likely that changes will be mostly tinkering around the edges.

Issues

Nutrient management issues revolve around the examination of several complex questions in the political arena. It is likely that none of these issues will be fully answered. But as they each slosh about in political and policy discussions, these debates will influence the outcome of the nutrient management debate.

The first question is "Should agriculture be treated as another business?" The major environmental laws of the 1970s differentiated and largely exempted agriculture from the rules that other economic sectors were required to meet. In the early 1980s, the agriculture sector's support reached a zenith with the farm credit crisis and images of poor farmers struggling to make ends meet. But since the mid-1980s, public perceptions have switched dramatically to viewing agriculture as a business, with concern over the intrusion of corporate America into agriculture, over the distribution of federal farm program payments to large farms, and over consumer health and welfare risks. Should this shift continue, with agriculture increasingly seen as another business, then nutrient management legislation will likely give individual producers less flexibility, place voluntary programs on a shortened leash, and make a greater number and type of activities subject to regulatory controls.

The second question is "Is the U.S. Department of Agriculture part of the issue or part of the solution?" A corollary is "will reorganization affect how USDA is viewed?" Some question whether USDA is up to the water quality challenge generally, and whether the voluntary approach based on incentives is always appropriate and successful, even when the worst problem sites can be identified as needing focused attention. More specifically, there are questions of how effectively SCS can work as both cooperator and regulator at the local level. Critics claim that USDA is ineffective as a regulator and point to the paultry assessment of penalties under the compliance programs. Supporters counter that USDA is effectively doing the job, that the penalties grow from year to year as enforcement increases, and that no programmatic backsliding has occurred in the face of heavy pressure from many traditional farm interests to ease up.

The third, closely related question is "How do you define success in water quality, and who will decide when it is enough?" Some environmentalists will argue that nothing less than the complete elimination of the nutrient problem is an acceptable goal. They say that the tools already largely exist, that there is no reason why they can not be applied across the landscape, and that continued deterioration of water resources at some locations may lead to virtually irreversible consequences.

Supporters of the current approaches and programs counter that a great deal of progress has been made in the past decade on many resource conservation fronts, far more than had been measured during preceding decades. They say that these changes are real measures of success and that, with more time, successes will continue to accumulate. They also say that there is every reason to anticipate that similar degrees of success can be anticipated in nutrient management. Also, they caution that the rules of the game should not be changed once again, confusing and disheartening public employees and farmers alike. They also say that the education and information activities needed to get these programs in place are already largely under way and, over time, are accomplishing much of what is needed. Policy makers will be asked to choose between giving agriculture more time with its "softer" solutions or treating the agriculture sector in a more rigorous fashion.

The final question is "How will the issues of water quality in agriculture, including nutrient management, be redefined by a broader redefinition of the way to frame environmental questions?" Policy debates over pesticides, endangered species, and other challenging topics for agriculture are elevating agriculture into broader debates; it is a new player in some of these. At the same time, the growing interest in alternative and broader management concepts, such as watershed management and ecosystem management, place agriculture within broader systems as well. These interests portend that the farm field will now be viewed as part of a larger system, and the farmer's fence line will lose its preeminence when considering many environmental and resource conservation issues. This transition occurred for soil erosion a decade ago, when environmentalists successfully documented that the costs of soil erosion off the farm were much higher than on the farm. These cost statistics were important in subsequent debates over compliance, the conservation reserve, and other proposals to control erosion. As the boundary created by the farmer's fence line becomes less concrete, producers are pushed into situations in which pressures for certain types of farm operation decisions will increasingly be brought by demands from outside their property.

In seeking answers to these questions, three solutions are currently receiving the most attention in national policy debates. One is the application of regulations. The logic that would tie the polluter pays concept into the watershed management concept is clear. This approach would use regulation to force more prudent management practices at those sites that are responsible for unacceptable levels of problems, and would not distinguish agriculture from other uses of the land and its resources. Opponents of regulations generally support either passive or active incentives. Passive incentives are available to anyone who is interested. Passive incentives include the traditional education/cost-sharing/technical assistance approach of agricultural programs. Active incentives are economic incentives that try to influence behavior by providing a marketplace that gives value or benefits to socially-desirable behavior. An example would be emissions trading under the acid rain provisions of the Clean Air Act. There is nothing in water pollution programs that is nearly so fully developed at this time.

A second set of solutions revolves around the application of more holistic solutions to the problems. Watershed planning and management is the one that receives the most attention in agricultural circles these days, but others such as ecosystem management are also being widely discussed. These concepts are supposed to tie what were formally external and separate considerations into an internally consistent framework, thus making resolution of individual problems more rational within a larger context. As was said earlier, the preeminence of the farm field boundary is being replaced by a boundary much further away, and individual farmers must consider the effects of their actions far from their property.

A third solution combines the holistic approach with efforts to maintain the importance of the farm boundary. Total resource planning puts all the conservation effort on a farm under one roof (really in one binder). But one can question what it buys beyond the sum of the parts (Can it be characterized as a land management equivalent of the air quality program's bubble concept?) Total resource planning would combine and coordinate water quality, drinking water, and groundwater protection provisions with other conservation efforts in a more consistent and efficient manner.

Congressional Interest in Nutrient Management

Congress is interested in nutrient management, if the introduction of relevant bills is any measure. But there is remarkably little that is new. Let me give you two measures of changing interest. First, CRS maintains an issue brief on groundwater issues, of which nutrient management is a key component. But last year, after several years, interest had died back to such a low level that we archived the issue brief.

Second, when legislation related to nutrient management has been considered, it was generally a "stand alone topic", addressed independent of other clearly related issues. Now it has been almost fully captured within the nonpoint pollution topic. This redefinition doesn't necessarily change the substance of the issue, but it does mean that it is less visible -- the best measure we have of that at the Congressional Research Service is the number and kinds of inquiries we get, and on this matter they appear to be way down this year when compared to recent years.

Regardless of the level of interest, the alternative approaches for dealing with nutrient management being discussed in Congress generally can be placed in one of three categories:

- Stay the course,
- Apply the CZMA approach, and
- Apply the watershed management concept.

The Clinton Administration supports preventing pollution using the watershed management concept -- an approach that will push corrective activities back onto farms (and other sources) as part of managing watersheds. Actions taken could be tied into finding excessive nitrogen in drinking water supply. Also, an increase in the regulation of drinking water could prompt States to increase well head protection which, in turn, means increased regulation of activities on adjoining surfaces. The Administration approach for the Clean Water Act supports the State designation of critical watersheds with planning and actions that would correct the worst problems in those watersheds. Similarly, its Safe Drinking Water Act proposal calls for source protection of ground and surface waters.

The Administration has coordinated its position with some of the pending legislation. Under Clean Water Act proposals, the 103rd Congress is considering major water quality proposals introduced by Senators Baucus and Chaffee (S. 1114), which will be the basis for subcommittee and full committee mark up early in 1994, and the Oberstar bill, (H.R. 2503). Also, Congressmen Studds (H.R. 2199) and Mineta (H.R. 2255) have introduced closely related bills which would impose taxes on pesticide and fertilizer manufacturers, among others, to help pay for local water pollution control projects. There are also a number of bills that focus on estuaries, coasts, or lakes that include nutrient components, such as a Mitchell bill (S. 1198). Finally, there are also a number of safe drinking water proposals that include pollution prevention and watershed management provisions, but few seem to be willing to hazard a guess as to where these might end up. Another potentially key bill was introduced just before this meeting by Senator Baucus (S. 1547, Safe Drinking Water Act Amendments) which would provide $20 million a year in grants of up to 50% to States to protect "water supply areas". While many bills have been introduced, almost no legislative activity has occurred, so it is still useful to look at the key provisions of these bills.

The Baucus/Chaffee bill would address nutrients in two ways. First, animal waste management facilities would be added to the list of projects that can be funded using construction grants. However, this bill also proposes that many other types of projects be added to the list. Because so many items are proposed to be added and because the available funding is limited, it is not clear how many of these facilities, if any, would actually be constructed using funding from this program. The second is coordination of Department of Agriculture programs with water quality activities. The bill would encourage the Department to take impaired waters into account in setting priorities. EPA would provide technical assistance to the Department. Also, watershed planning would be required, but the threshold appears to be less restrictive than in the Coastal Zone Management Act amendments of 1990.

The Oberstar bill promotes the watershed management approach for nonpoint pollution. It is somewhat more prescriptive than the Baucus/ Chaffee bill, but less prescriptive than legislation introduced in the 102nd Congress. The cornerstone of this approach is that States would have to update one fifth of their nonpoint plans each year. The plans would have to be more explicit than they are now and would have to give greater emphasis to nonpoint pollution.

In addition, both bills would require that a "cookbook" for management measures be produced, modeled after the guidebook that was produced to assist in implementing the Coastal Zone Management Act. This guide would include measures that take into account regional variations and varying degrees of problems. While there are no requirements that States would use this book, it is likely that it would provide important guidance as States work to implement their plans.

It is also important to note that the chairman of the primary committee of jurisdiction for water quality legislation in the House, the Public Works Committee, has yet to introduce a clean water reauthorization bill, although hearings have been held. This lack of a bill suggests that final congressional action may still be a long way off, even if the Senate is able to move its bill through the committee and the Senate floor early in 1994.

Nutrient management is also likely to be addressed in the 1990 farm bill. How it might be addressed, however, is difficult to predict. Congressional consensus on the appropriate action is not only unclear now, but also may well change before the middle of 1995, when the next farm bill provisions are likely to be considered. Just think back to 1985, when the Senate had rejected a proposal for a conservation reserve of less than a million acres a year earlier because of the cost, or to 1990, when the omnibus conservation bills introduced by Senators Fowler and Lugar in 1989 were believed by some to frame the issue, but turned out to encompass only a relatively small portion of what became a water and wetlands conservation title. Factors that could affect provisions in 1995 include new information on the role of agriculture in nutrient problems and on the overall extent of nutrient problems, and provisions in clean water legislation, among others.

In 1995, there will be many possibilities for programs that would affect nutrient management in the farm bill. One of the two that currently seems more promising is emphasis on the holistic concepts, discussed earlier, including total resource planning, watershed management, and ecosystem management. Applying any of these concepts to agriculture would mean that individual producers would now be treated as part of larger systems, and that conditions off the farm would be more important in determining what resource conservation efforts would be encouraged or required on the farm.

The second is some form of reward or stewardship payment program for conservation. Many specific proposals that have been mentioned take many forms and are generally termed either green ticket or environmental stewardship payments. These proposals call for stewardship incentives that might range from cost sharing to easements. Programs might include both incentive and implementation payments to encourage desirable behavior. They would be made available in locations that are especially vulnerable to environmental problems, rather than tied to raising specified crops. Thus, those currently outside traditional commodity programs would be eligible for this program. Availability would be made in exchange for environmental services, not to support farm income.

Possible Outcomes

Even while national legislation is pending, many efforts already are affecting nutrient management at the State and local levels. In particular, drinking water standards will influence a variety of land uses. But it is also important to remember that the Safe Drinking Water Act only affects wells serving 15 connections or 25 people. While there are over 200,000 regulated entities, few of them are farms. However, public well water quality may be affected by activities on farms. The interesting phenomenon is that, where well testing has led to the discovery of nitrate contamination problems, the response has been self regulation in some cases. Examples include Nebraska, where limits were placed on both the timing and volume of fertilizer application, and Minnesota.

Another growing interest centered at the State and local level is partnership building. Agricultural interests talk incessantly about partnerships. While I am not aware of major initiatives bearing nationally-visible fruit as yet, these efforts seem to be intended to create larger coalitions to work together to address resource conservation and environmental problems such as nutrient management. The notion of partnerships is especially appealing because many of the largest impediments to environmental quality on the farm may actually come from institutional barriers created by lack of communication, or different and conflicting mandates, rather than topical barriers. That being said, it is also true that claims about what partnerships accomplish often seem exaggerated, especially when they are tied to the process and with full participation, rather than with actual changes on the ground. While many interesting things are occurring at various locations scattered across the country, and fostered in some cases by an expanded partnership, less is happening at the national level.

A few predictions can be made about national policy addressing nutrient management and where it seems headed. First, legislation reauthorizing the Clean Water Act is likely be enacted during 1994, though no earlier than the summer. There is also some chance that it may get held up until the 104th Congress, in essence placing it in tandem with the farm bill. The pace at which clean water legislation moves will be directly related to the contents. The less controversial the provisions and the fewer subjects that are covered, the quicker it will move. One possibility is enactment of a "light" version that leaves many of the controversial issues to a later date. If this should happen, it is not clear how topics like nutrient management might be treated; are they either so lacking in controversy or so easy to resolve that compromise has already been reached and they will appear, or will this topic be placed on the side for later consideration so as to move the overall legislation? None of this is clear at this time.

There are likely to be both rewards and penalties for the agriculture community if nutrient management legislation is delayed. One thing delay may buy is more carefully crafted policy. For watershed management, policies which carefully fit the farmer into a broader network of all land owners and users will attract considerable interest. With time, there will be both more information and more experience to draw on -- the availability of this additional knowledge does not, of course, automatically mean that the program will be more effective.

Also, there are likely to be penalties associated with delay. One is that agriculture's political power base will continue to shrink. While the effect is not necessarily noticeable from year to year, over time, other interests are gaining a much stronger voice in policies and politics that affect agriculture. The resolution of issues is increasingly less sympathetic to agriculture. Also, if data show that the traditional agriculture approach of voluntary participation is not up to the challenge, then it will be in a weaker position to say that it is capable of addressing its problems.

The bottom line is that nutrient management has a visible place within the greater scheme of water quality and resource conservation policy issues. It will neither drive the resolution of these broader issues, nor is it viewed as being significant enough to stand alone. Therefore, resolution will occur within a myriad of other environmental topics that have a significant, and consequently complex agricultural component. That bigger picture is hard to define at this time.

STATE PROGRAMS FOR NUTRIENT MANAGEMENT:

THE VIRGINIA EXAMPLE

By

Russ Perkinson[1]

Introduction

Nutrient management can be defined as management strategies to match nutrient rates and timing of applications to correspond with crop uptake in order to minimize environmental impacts associated with nutrient use. The driving forces in the development of Virginia's nutrient management program were the protection of Virginia's ground and surface waters from nutrient pollution, and the nutrient reduction goals of the Chesapeake Bay restoration program.

The Chesapeake Bay is the largest and most productive estuary in North America. The Bay produces half of the blue crabs and one quarter of the oysters harvested in the United States. The main stem of the Bay is about 200 miles long and varies in width from 3 to 30 miles, with a surface area of 2,200 square miles. Nine major rivers, draining 64,000 square miles, empty into the Bay.

The Chesapeake Bay region has placed heavy emphasis on agricultural nutrient management as a method of reducing nitrogen and phosphorus loadings in the Bay. In 1982, the results of a six year study were published by EPA. The study was authorized by Congress, at a cost of $27 million, to identify the causes of declining water quality in the Chesapeake Bay. In the Bay, acreage of submerged bottom grasses, a vital habitat for many forms of Bay life, had declined sharply over the previous two decades. (Previously, vast acreages of submerged grasses have been documented back to colonial times.) The study concluded that three primary factors were contributing to the Bay's decline: excessive sediment loads, excessive levels of nitrogen and phosphorus, and toxic contaminants. It is interesting to note that, of the toxic compounds found in the study, agricultural pesticides were not found to be a significant factor in the decline of the Bay.

High nutrient levels in the Bay result in excessive growth of phytoplankton, the tiny plants which grow suspended in water. At low to moderate populations, the phytoplankton are beneficial to Bay life by providing a source of food to animal forms. At high populations, the phytoplankton growth will cloud the Bay's water, reducing the light transmission to the bottom grasses which reduces their vigor or results in death of the grass beds. As the phytoplankton die, the decomposition process will decrease dissolved oxygen in the water, which directly stresses higher forms of marine life.

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Nutrient reduction strategies must focus on both nitrogen and phosphorus levels in the Chesapeake Bay region. In the fresh water tributaries, phosphorus is generally the limiting factor to phytoplankton growth. However, in the salt water areas of the Bay, nitrogen is the limiting factor in the summer months when most phytoplankton growth occurs, while phosphorus may be the limiting factor in other seasons.

Evolution of Nutrient Management in the Bay Region

Following the six year EPA study, models were used to assist in the determination that a 40% reduction in controllable nutrient loads in the Bay would be necessary to return the Bay to an acceptable condition. In 1987, the Governors of Maryland, Pennsylvania, and Virginia, and the Mayor of Washington D.C. signed the second Chesapeake Bay Agreement which committed the jurisdictions to meet a 40% reduction in controllable nutrient loads.

Agriculture's contribution to the nutrient enrichment of the Bay is significant. As long as a significant agricultural industry has existed in the Bay watershed, it has contributed nutrients to the Bay's waters. However, as the agricultural industry became increasingly specialized and the population base in the area grew, nutrient inputs to the Bay increased. Farms once relied largely on farm-produced feed and nutrients. Nutrients were cycled on the farm through crops and livestock and back to crops. In contrast, many modern farms rely totally on imported feeds, particularly in the poultry and swine sectors. Thus producers are concerned about disposal of nutrients contained in manure, rather than efficient use of those nutrients.

Virginia's Program

Virginia's statewide nutrient management program was established in 1989. The Division of Soil and Water Conservation, which is in the Department of Conservation and Recreation, is the lead nonpoint source pollution management agency and operates the nutrient management program. The overall program goal is to assist farmers and others in managing agricultural and other fertilizers, animal manures, and sewage sludges and to prevent the misapplication, improper storage, discharge or other use of these products which may result in ground and surface water degradation caused by excess nitrogen and phosphorus from agricultural and urban turf nonpoint sources. In the Chesapeake Bay watershed, an additional goal is to reduce nutrient inputs to the Bay from agriculture by 40% by the year 2000 through nutrient management and soil conservation.

Nutrient management staffing includes a program manager and 10 field nutrient management specialists. The field specialists assist farmers in developing site specific nutrient management plans. The nutrient management specialists also educate and assist farmers in manure spreader calibrations, interpret results of manure tests, monitor nitrate levels in soils, conduct field plot demonstrations and field days on practices capable of improving water quality, conduct farmer educational meetings, and train personnel from industry and government in nutrient management principles. These specialists have developed over 1,000 nutrient management plans on 240,000 acres of cropland. Nitrogen and phosphate use reductions are estimated at 5.2 million pounds and 4.4 million pounds respectively from the planning activities.

An assistant program manager focuses on special projects. One such project has involved development of a strategy for pursuing nutrient management with the lawn care industry. The first step involved convincing the land grant university that water quality should be considered in the university's turf nutrient recommendations. After the rates and timing of application recommendations were revised, the Department began negotiating with major lawn service companies to sign a water quality improvement agreement whereby the companies agree to utilize rates approved by the Department. Retailers of turf fertilizers were also approached to promote the use of point-of-sale brochures which detail proper lawn fertilization practices.

Additional technical assistance is provided by 11 water quality specialists employed by conservation districts in the Virginia coastal plain covered by the Chesapeake Bay Preservation Act to write farm water quality plans having a nutrient management component. Extension agents and Soil Conservation Service employees also cooperate in nutrient management programming.

Nutrient Management Planning

A nutrient management plan is a written site specific plan indicating how the major plant nutrients (nitrogen, phosphorus and potassium) are to be managed annually for expected crop production and for the protection of water quality. Nutrient management planning involves several steps. First, field sites are evaluated to determine potential productivity of soil and any environmental features of importance. Soil surveys are used extensively during the process. Soil type is an important constraint since it is not changeable through management. Farmer yield history and experience with each site are also considered in determining realistic expected crop yields.

Soil test results are used for each field to evaluate the nutrient resources already available to the farmer. A combination of soil test results and soil productivity potential determines the crop nutrient needs for planning purposes. These nutrient needs may be modified during the growing season if subsequent site specific soil or tissue tests indicate an adjustment is necessary. Once crop needs are known, on-farm nutrient sources such as manures or legumes are considered first, followed by balancing crop needs with purchased fertilizers.

Field limitations based on production constraints or environmental concerns are considered before allocating nutrient sources to fields. Timing of manure and fertilizer applications is scheduled as closely as possible to the time of greatest crop nutrient needs. Timing of applications is most critical on environmentally sensitive sites such as those having highly permeable soils or karst topography.

Although the nutrient management planning process involves evaluating economic, agronomic and environmental aspects of nutrient use, planning in itself is of little benefit if sound environmental protection practices are not employed as components of a plan. A plan is only as good as the nutrient application practices such as site specific target yield determinations, efficient split applications of nitrogen, and proper timing of manure applications.

Program Delivery Options

Voluntary approaches to nutrient management are generally most effective where a win-win solution can be found relating to both agricultural production economics and environmental impacts. Use of the nitrate soil test for corn is a good example. Economics is an important factor which can help or hinder nutrient management acceptance in voluntary programs. In low value commodity crops, the cost of purchased fertilizer is significant. Thus economics will prevent extreme over-fertilization. However, in high value crops such as vegetables and nursery stock, the cost of fertilizer is insignificant in the industry cost structure. Economic considerations alone may lead to high nutrient losses to the environment in these cases.

With on-farm manures, economic impacts can be either positive or negative in influencing nutrient management decisions. On farms where considerable nutrients are purchased or cycled on the farm, nutrient management may have a positive impact. Farms which import large amounts of nutrients in feeds may not benefit economically from nutrient management since manure disposal can become a cost.

Incentives can be used to speed the adoption process for nutrient management practices, or to subsidize practices which may not be in the farmer's short term economic interest without the incentive. Subsidizing the cost of manure tests and providing cost share funding for rye cover crops to trap excess nitrates are examples of recurring incentives. Cost sharing on animal waste storage facilities or equipment tax credits are examples of one-time incentives which can provide lasting benefits.

Regulations may be necessary where problems are significant and voluntary approaches and incentives are not effective. They may also be an option in situations where economic pressures alone could cause disposal at the lowest cost possible, regardless of environmental impacts, as could be the case with land application of municipal sewage sludge. A third appropriate case for regulations involves sectors where the majority of industry is in compliance, but a few environmental "bad actors" still remain.

Virginia's nutrient management program employs a mix of delivery mechanisms. These include voluntary, incentive, and regulatory approaches.

Voluntary Efforts

Virginia has placed emphasis on voluntary practices which can lead to the greatest reductions of nutrients in ground and surface waters. Individual practices are promoted in voluntary nutrient management plans, educational meetings, field days, and various media. For example, manure testing is a valuable tool in selling farmers on nutrient management and insuring the agronomic and environmental accuracy of farm-specific nutrient management plans. Manure test results often impress farmers with the value of nutrients contained in manures as related to possible fertilizer savings and greatly increase the farmer's confidence in relying on manure as a nutrient source. The Department provides $30,000 in annual funding to operate the manure testing lab at Virginia Tech. Technical assistance to calibrate manure spreaders likewise provides a high water quality return, particularly when combined with manure sample analysis.

New technology such as nitrate soil testing for corn will play an increasing role in nutrient management. Nitrate soil testing can provide a confidence check for farmers relying on an organic source of nitrogen and can result in significant production cost savings. Innovative research on future nutrient management tools and techniques must stay at least three to five years ahead of current approaches to allow for continued progress.

Incentive Programs

Two primary incentive programs in Virginia include a state tax credit and cost share assistance. A state tax credit on nutrient management-related farm equipment was enacted in 1990. The state tax credit is 25% of the purchase price or $3,750, whichever is lower, on advanced technology farm equipment meeting certain minimum criteria. The equipment tax credit applies to:

- sprayers for pesticides and liquid fertilizers;
- pneumatic fertilizer applicators;
- monitors and flow regulators;
- manure application equipment; and
- tramline adaptors.

To claim the credit, the farmer must have in place a nutrient management plan approved by the local Soil and Water Conservation District.

A nutrient management plan must also be developed for farmers receiving state animal waste storage cost share assistance. Cost share programs also focus on nutrient control BMPs such as the use of rye winter cover crops to trap nitrogen.

Regulatory Approaches

Two state regulatory programs require nutrient management plans. Nutrient management plans are required by the Department of Environmental Quality prior to the issuance of Virginia Pollution Abatement Permits for livestock farms with liquid waste having more than 300 animal units in confinement and for smaller farms which constitute a water quality hazard. In addition, draft sewage sludge regulations are being developed which include many elements of a nutrient management plan.

The Chesapeake Bay Preservation Act requires nutrient management plans in the Virginia coastal zone for agricultural land designated by counties as being within the resource protection area. Some counties have designated the entire county as a resource protection area. In addition, 100 foot wide permanently vegetated buffers are required along streams or water bodies unless a soil and water quality conservation plan, containing a nutrient management component, is implemented for fields adjoining the buffer, in which case the buffer may be reduced to a minimum of 25 feet. All plans must be approved by the local Soil and Water Conservation District.

A training and certification program is being developed geared toward consultants, fertilizer dealers, sewage sludge representatives, and others who may develop nutrient management plans. The training and certification program will be a means of training individuals to write nutrient management plans required by various incentive and regulatory programs, and should enhance voluntary efforts as well.

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INTER-STATE NUTRIENT MANAGEMENT

By

Robert Lynch[1]

Introduction

Contrary to popular belief outside of the state, Oklahoma possesses extensive surface and ground water resources. Among the fifty states, Oklahoma ranks very high in the miles of shoreline surrounding reservoirs. This is made possible by the fact that all but one of the major rivers in the state has been impounded. The consequences of stream pollution are accentuated by this practice as pollutants discharged into a stream are very likely to wind up in a reservoir. For nutrient pollution this is especially important as the ecological effects of nutrients are more visually pronounced in reservoirs versus flowing waters.

The contribution of nutrients to the degradation of reservoirs, through accelerated eutrophication, is a common occurrence across the country and in Oklahoma. In a recently completed study of more than one hundred small (<400 hectares) reservoirs in Oklahoma, over fifty percent were found to be eutrophic or hypereutrophic. These reservoirs are in rural areas and do not receive point sources of pollution; therefore, nutrients contained in inflow waters must originate from nonpoint sources. Major reservoirs are experiencing similar degradation through nutrient pollution; however, many of these are subject to both point and nonpoint sources of pollution. The most well known case of reservoir eutrophication in Oklahoma is Tenkiller Ferry Reservoir.

Tenkiller Ferry Reservoir is located in northeastern Oklahoma and is fed primarily by the Illinois River. The drainage basin for the reservoir is approximately 234,820 hectares, with roughly half being in Arkansas. The reservoir is a very important recreational resource for Oklahoma; however, water quality has degraded rapidly over the past decade, primarily due to nutrient pollution. In past decades, the reservoir was renowned for its clarity and was extensively used by scuba divers; however, the turbidity is now such that only small areas near the dam can support this activity. Algal induced turbidity is very high in the upper reaches of the reservoir where an extensive anoxic zone has developed. Recreation in this part of the reservoir has been severely diminished by both loss of aesthetic properties and damage to the fisheries.

Eutrophication in Tenkiller Ferry Reservoir has received much public and private attention not only due to the degradation of the resource but also as a result of the inter-state sources of pollution responsible for the degradation. There has been considerable controversy concerning the source and magnitude of the nutrient contribution from the two states. This has fueled much debate as well as an unparalleled stirring of public interest in an environmental

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issue in Oklahoma. Unfortunately, the debate has often left the objective and/or scientific arena and become a political issue and a source of animosity between Arkansas and Oklahoma. One particular issue ended in a law suit which was finally settled by the U.S. Supreme Court. The purpose of this paper is to examine those issues which in the past have hampered inter-state cooperation in addressing environmental problems within the Illinois River Basin and to identify current methods for dealing with inter-state environmental issues.

Points of Controversy

Although there are several opinions regarding the source of the conflict between Oklahoma and Arkansas, the basis of the disagreement can be reduced to several 'points of controversy'. Based upon these five points, both an understanding of the source of the conflict and identification of methods for avoiding conflict in the future can be achieved.

Differences in the River Between States

The Illinois River is listed in Oklahoma standards as both a scenic river and an outstanding resource water. As such, it is one of only six in the state. The availability of free-flowing, relatively clear rivers in Oklahoma is very limited; therefore, Oklahoma views the Illinois River as especially important.

The portion of the river in Arkansas is relatively shallow, narrow, low gradient, and not particularly scenic, especially in light of the fact that Arkansas has an abundance of very scenic rivers. Many of these are within a short driving distance of the Illinois River. As a result, Arkansas does not view this river as particularly noteworthy or deserving of special protection.

In addition to differences in the river itself, the associated riparian corridor is much more diverse and scenic in Oklahoma and the river basin itself is less developed. Northwest Arkansas is relatively more developed than northeast Oklahoma and is undergoing the most dynamic growth in the entire state. The river basin in Oklahoma contains only one community greater that 5,000 population while in Arkansas there are three communities over 25,000 population.

Differences in the Perception of Water Quality

Water quality in the Illinois River is viewed differently by various interest groups; however, as a general rule, clarity is the most valued property of the river. From this viewpoint, the river seems to be in fairly good shape as it is relatively clear in most public access areas. There have been numerous allegations that significant decreases in river clarity have occurred over the last decade, although data collected during the past fifteen years do not support this conclusion. Much of the beauty of the river is associated with the physical nature of the riparian corridor such as bluffs, vegetation, and wildlife as well as the free-flowing nature of the river itself, all of which are properties that would continue to exist regardless of nutrient levels. There is a general agreement and ample data to indicate that the river contains very high levels of both phosphorous and nitrogen; however, there remains considerable disagreement over the extent of degradation that these levels of nutrients may be causing. It would appear that many of the effects of nutrient pollution are not readily apparent on the scale of public perception. In conclusion, although the public places a high value on water clarity and believes that this value has been lost to some degree, there is insufficient evidence to support this belief (at least over the past fifteen years), despite the fact that nutrient levels are high.

Areas of Impact

The applicability of downstream standards can be looked at from two angles. The first, which will be discussed in detail in following sections, concerns the legal limits of state standards. The second deals with establishing accepted zones of impact for nutrient loading.

Most water quality standards are based upon numerical criteria and apply at certain points below a specific discharge, usually the point where complete mixing of effluent and stream water has occurred. For toxic pollutants, this is the area of greatest concentration and the most likely location of toxic effects. If the values at that point exceed numerical criteria, then a water quality violation is said to occur. Nutrients are somewhat unique in that their effects will likely be most pronounced far below their point of entry into the stream and that those effects may not be visible or quantifiable unless the stream is impounded.

It is unanimously accepted that Tenkiller Ferry Reservoir is undergoing rapid eutrophication as a result of nutrient loading and is already severely degraded in it upper reaches. Although there are a number of both point and nonpoint sources of nutrients in the river it is certain that many of these are entering the system many miles above the lake. The question then is 'How far below a discharge can negative effects be tied to a discharger?'. The relative importance of different discharges can be determined through modelling, although using this method to identify the effects of specific dischargers is not yet common, especially when they are so numerous. The geographic separation between cause and effect as well as the number of nutrient sources creates a complex situation in the Illinois River Basin.

Origin of Nutrients

The location and source of nutrients remains a topic of controversy in spite of the numerous water quality studies which have been, and are being, conducted. From a technical viewpoint, the data indicate that Oklahoma and Arkansas contribute roughly equal amounts of nutrients to the river; however, the opinion of the public in each state is that the other state is the main culprit. For example, many in Oklahoma feel that, if the wastewater treatment facilities in Arkansas were eliminated, the river would be of adequate quality. Conversely, the reigning opinion in Arkansas is that confined animal operations and recreation in Oklahoma are the main problem.

There is also considerable disagreement concerning the relative magnitude of point versus nonpoint sources of nutrients. Not surprisingly, point source dischargers claim that nonpoint sources are the prime contributors while nonpoint source contributors blame point sources. This agreement appears to be common to both states. Based upon available data, it would appear that the sources are roughly equivalent in terms of their effect on the river.

There are four primary sources of nutrients within the basin: wastewater treatment facilities, confined and unconfined animal operations, plant nurseries, and on-site sewage disposal. Each of these sources produces significant quantities of nutrients; however, the portion of the nutrients produced by these operations which actually reach the river is unknown.

The amount of waste produced by some of these sources is listed in Table 1. Several important conclusions can be drawn from the data contained in this table. Most notable is that the waste produced by animal operations greatly exceeds that produced by humans within the basin. Additionally, it can be seen that the waste produced by cattle and poultry are approximately equivalent. This is a very important fact. Most of the focus of nutrient management efforts have focused on poultry waste with little attention being placed on the effects of cattle.

Table 1. Character of Waste Produced in the Illinois River Basin.

Type of Waste	Quantity (dry tons)	Nitrogen (lbs.)	Phosphorous (lbs.)
Human (urban)	2,134	258,477	25,845
Human (rural)	5,091	616,609	61,810
Poultry	92,728	10,305,741	3,087,623
Cattle	105,379	9,798,898	3,193,299

The poultry industry preceded cattle operations by several years. The soils in this area are generally poor and consist primarily of chert rubble which is unsuitable for pasture, row cropping, or other horticultural activities. Poultry operations produce a high quality waste that when applied to these soils can result in the development of excellent pasture (a colleague in Arkansas claims that pasture can be established on asphalt highway after poultry waste application). The development of pasture allows for the establishment of range cattle or dairy operations. These operations then produce additional quantities of waste. Within the Illinois River Basin in Oklahoma it can be calculated that the approximately 83,000 cattle produce as much phosphorous and nitrogen as the eleven million poultry. Since waste from un-confined cattle is essentially un-managed, its effect on water quality is almost entirely a function of the location of the cattle. Cattle spend much of their time, especially during hot weather, loitering near streams and their effects on stream systems can be readily observed by site visits.

Although poultry waste is managed to some extent, it has been shown that application rates greatly exceed plant uptake rates and that poultry waste production exceeds available land within the basin in Oklahoma. Another factor which compounds the problem is that waste is applied year-round while plant uptake is limited to the 210 day growing season. When calculated in terms of human equivalents, it can be shown that the waste produced by animals is roughly equal that produced by two million people. It should be noted that animal numbers within the basin in Arkansas are much higher than Oklahoma and that similar waste management problems exist; therefore, nonpoint sources likely cause a significant portion of nutrient loading in that area.

Two other observations which are noteworthy are that the amount of human waste produced in Oklahoma is relatively small compared to animal waste and that rural systems, primarily septic tanks, handle the majority of human waste in Oklahoma. Recent studies have shown that a significant proportion of these private systems are either inadequately designed, inadequately installed, or both. This could play an important role in nutrient loading within heavily populated sub-watersheds of the Illinois River Basin.

The primary conclusion that can be drawn from this data is that animal wastes have a much greater potential to cause nutrient loading than human waste in that much more is produced and that which is produced is essentially untreated and applied directly to the land. Much of the focus in the basin has been upon the easily identifiable municipal wastewater treatment facility discharges; however, it can be seen from this data that more attention should be placed upon nonpoint sources.

Differences in State Standards

Differences in water quality standards between Arkansas and Oklahoma were the original cause of the inter-state disagreement and are a continued source of controversy. Central to this issue is the enforceability of nutrient standards, which remains an important topic of discussion across the country. This dilemma is magnified by the fact that neither state has numerical standards for nutrients, a situation which is common throughout the United States.

Oklahoma assigns beneficial uses to each water body listed in state standards. Oklahoma recognizes aesthetic properties (intrinsic value) as beneficial uses, a recognition which contributes to the inter-state controversy for two reasons: 1) Arkansas does not recognize aesthetic properties as beneficial uses in state standards and, 2) this beneficial use is very hard to quantify as aesthetic values vary by individual and are generally non-quantifiable.

One of the primary reasons for placing aesthetic properties as beneficial uses is to address the nutrient issue and, in fact, Oklahoma's nutrient standard is based upon the degradation of this beneficial use. It is an accepted fact that if nutrients rise above a certain level, algae will grow to such an extent as to cause degradation of aesthetic properties. While this is almost invariably true, there remain some problems in taking this approach in evaluating the effects of nutrients. Again, the quandary of individual variation in the perception of aesthetic properties is encountered. A second, and more fundamental, problem is that in most running waters, the algae that respond to nutrient additions tend to be benthic organisms; therefore, their growth will seldom result in a change in water clarity or color, although stream bottom materials may be heavily coated by algal growth. Very few people will identify this growth as a degradation of aesthetic properties until it reaches a very advanced state. This discussion can be boiled down to the questions: 'Are nutrient additions stimulating algae to nuisance levels?' and 'How can this be measured (quantified)?'.

Although there are levels of algae in lakes, as measured by chlorophyll content per liter, above which it is recognized that undesirable effect may occur, there are not similar levels for benthic algae which are widely accepted. When this is added to the fact that few people even recognize benthic algae as algae, much less as a problem, it is difficult to imagine a scenario under which impairment of aesthetic properties, as measured by the growth of benthic algae, could be quantified for any enforcement action. Work is being done to establish impairment levels and in isolated geographic areas these levels are understood; however, this information has not been applied on a wide scale. Bearing these limitations in mind, another approach for determining nutrient impacts in streams must be explored.

Bioassessment of streams has been conducted for many decades to determine the health of biological communities and, indirectly, the quality of water. Until recently, these assessments were largely subjective in nature and rarely produced objective measures of stream health. Over the past few years, new assessment techniques have been developed which quantify impairments to biological communities as measured by relative differences between impaired and unimpaired streams. Bioassessments are based upon the theory that chemical disturbance (pollution) of aquatic systems will cause a decrease in species diversity through the establishment of conditions under which fewer organisms are adapted to live. Secondly, these few organisms will then become dominant and usually will occur in large numbers.

Nationally accepted assessment methods are available for fish and benthic macroinvertebrates and a few states (Kentucky, Montana, Oklahoma) have developed methods and criteria for benthic algae. These methods have proven very useful in determining the effects of non-toxic substances, such as nutrients, as well as assessing the multiplicative or synergistic effects of the many chemical compounds which may be present in concentrations below water quality standards. They have also been shown to be useful in separating out the effects of habitat limitation from water quality in the determination of the structure of biological communities. Most importantly from an enforcement standpoint, these assessments have also been shown to be legally defensible.

Bioassessment methods would appear to have excellent potential for use in assessing the effect of nutrient pollution; however, there remain some problems, primarily political, with their use. Perhaps the primary remaining disadvantage is the relative unpredictability of biological communities. Although it is possible to predict that communities will be impaired by the addition of nutrients, the degree of impairment is difficult to quantitatively determine, a priori. Additionally, it is not possible to accurately predict a quantifiable response to nutrient removal. The inability to quantifiably predict community response presents a significant hurdle as the majority of individuals who design nutrient control systems are engineers who expect predictable systems. Decision makers also expect to be given a quantified prediction of the effects of nutrient policy, especially when the expenditure of millions of dollars might be involved.

Based upon current bioassessment techniques, it is possible to identify stream impairment and to predict the general quality given different levels of nutrients. Most people accept damage to the fish community as evidence of pollution. Although fish are sensitive to many toxins, they are not, unfortunately (from the standpoint of identifying pollution), particularly sensitive to nutrient pollution, and those species which are sensitive are generally both small and non-game species, neither of which are characteristics highly valued by the public. In fact, nutrient additions up to a certain point will improve fish community productivity, especially many game fish species.

The structure of the benthic macroinvertebrate community is significantly altered under elevated nutrient levels. While the number of organisms may increase, species diversity is decreased, reflecting the less favorable conditions. In addition, particular species which are indicative of poor water quality will become dominant. Political acceptance of changes in the structure of benthic macroinvertebrate communities as evidence of pollution is not yet widespread. Requiring a municipal wastewater treatment plant to implement nutrient removal which would likely cost millions of dollars in order to protect 'bugs' would be a difficult sell. Benthic algae, particularly diatoms, also respond to nutrient additions by increasing their numbers, while at the same time undergoing a decrease in species diversity through the establishment of dominance by a few non-sensitive species. As with the benthic macroinvertebrates, it will be very difficult to sell damage to this community as evidence of pollution.

Despite these drawbacks and limitations, assessment of biological communities would appear to offer the best, and at current time the only, means of measuring nutrient impacts near their source. Given that the most noticeable effects of nutrients will likely be far downstream in impoundments and that it is practically impossible to tie those effects to a single source, we must either develop new measures of nutrient impairment or begin to employ those at hand. If we cannot provide methods to demonstrate that nutrients have a deleterious effect, then they are not, in fact, pollutants!

History of Inter-State Interactions

Accounts of the same historical event can be different, depending upon the viewpoint of the observer. Such is the case concerning the facts of the dispute between Arkansas and Oklahoma over nutrient discharge into the Illinois River.

In the late 1980's, the City of Fayetteville, Arkansas applied for a permit to discharge waste from a new municipal wastewater treatment facility into a tributary of the Illinois River. Oklahoma protested the issuance of the permit to USEPA; however, Arkansas proceeded to permit the discharge. Oklahoma pursued its case through legal channels and after a lengthy period the case wound up at the United States Supreme Court.

In its case, Oklahoma alleged that the discharge would result in a degradation of water quality within Oklahoma. Specifically, Oklahoma argued that the discharge would result in a violation of state nutrient standards, as measured by degradation of aesthetic properties. The court ruled that Oklahoma's downstream standards did not apply and that Oklahoma could not prove that the proposed discharge would degrade the quality of water in the Illinois River.

Several lessons were learned from this experience:

1. Oklahoma had an inadequate data base.

2. Impairment of aesthetic properties is difficult to prove, much less predict.

3. Nutrient standards should be based upon quantifiable measures.

4. Going to court only created animosity between the states and did not settle the issues. Even if Oklahoma had won on the applicability of downstream standard, the technical issue of quantifying nutrient impacts would have remained. In summary, if the court had found that Oklahoma's standards did apply, the discharge would probably have still been permitted since the state could not prove that it would degrade water in the Illinois River.

The consequence of the court's decision is an area of some concern. The fact that downstream standards do not apply creates an incentive for upstream states to place their discharges near borders so that the water quality in their state is not degraded. While it seems unlikely that any state or industry would take this approach, the court's decision establishes a principle of non-responsibility for upstream dischargers.

A second area of concern is more philosophical in nature but has important consequences for future actions. As previously discussed, the Illinois River contains very high levels of nutrients. One of the dilemmas that this situation presents is that increasing the levels of nutrients already present may not result in a noticeable or measurable difference, even in Tenkiller Ferry Reservoir, and would most certainly not result in drastic changes in the river. If this is the case, then it is hard to argue for setting nutrient controls on new discharges. This argument has been used in other areas and can be summarized in the question 'If the river is already polluted, then why should we have to conform to strict standards if we won't make it any worse?'. It is hard to argue against this stance from an economic standpoint and, given that it is very difficult to prove that a nutrient discharge will cause a problem, especially when high levels already exist in receiving waters, a more philosophical approach may be necessary.

This approach establishes the principle that increased loading of a problem pollutant(s) will not be allowed. For the Illinois River, this approach would establish a policy that, since we know nutrients are a problem, no additional discharges of nutrients will be allowed, even if it can't be proven that such a discharge would make things worse. It is essential that such a stance be taken, especially in regard to nutrients, until such time that the technical issues of measuring nutrient effects are codified into standards. This approach has been tried in limited areas through the development of Total Maximum Daily Loads (TMDL) which establish accepted loadings, reduce current loadings to those levels, and restrict additional loading. There has been some outcry that this is a no-growth policy; however, this is not necessarily the case. Through the use of such techniques as mitigation banking, trade-offs between remediation and new growth can be established which meet the goals of the TMDL in protecting the resource while allowing for responsible and accountable development.

Current Approaches For Addressing Inter-State Environmental Issues

Based upon the results of the court case and the unfortunate consequences that it generated, Arkansas and Oklahoma have jointly decided to take a different approach to settling environmental issues. The states have agreed to work together to solve problems with a general agreement to address potential inter-state problems in their early stages before conflict develops. On its own part, Oklahoma has taken the stance that legal action is the least desirable means of accomplishing its goals and will take those steps necessary to see that problems are faced in a non-confrontational manner.

The generic approach to addressing inter-state environmental issues has been through the establishment of inter-state task forces or working groups. These can be divided into two categories, political and technical. In the political arena, the two state governors appointed an inter-state environmental task force to address current and future problems and to provide the governors with yearly reports concerning relevant issues. This task force has been effective in identifying issues and in healing some of the ill feelings between the states; however, they have yet to have the opportunity to make decisions or reach conclusions on any contentious issue. This group is composed primarily of legislators and agency heads and deals with problems at the political level.

The other groups have been technical and issue-specific and have been staffed almost entirely with scientists and technicians. The goal of the technical working groups is to develop decision making and planning processes which are based upon science, not politics. As a means to reaching this goal, these committees have avoided attorney members. Although attorneys serve a useful role in developing and establishing inter-state policy, past experience has shown that discussions over nutrient issues are best conducted by those who have a technical knowledge of the subject. Examples of these working groups include committees to develop a process for setting a Total Daily Maximum Load for the Illinois River, development of a Comprehensive Basin Management Plan for the entire river basin, and the Siloam Springs study.

The potentially most contentious issue between the states at the current time concerns the discharge from the Siloam Springs Wastewater Treatment Facility (WWTF). A working group has been formed to address the concerns of both states and is rapidly moving towards making environmental decisions with major consequences. The Siloam Springs WWTF discharges into Sager Creek just inside the Arkansas border. The creek then flows into Oklahoma and merges with Flint Creek, which is a state scenic river and a tributary of the Illinois River. Available data show that this creek contains the highest levels of nutrients of any creek in the Oklahoma portion of the Illinois River Basin. Again, Oklahoma does not have data to prove that state standards are being violated by these levels of nutrients. The two states have agreed to view this through a number of parameters but most significantly they will be assessing degradation (impairment) through disruptions in biological communities. If the inter-state study group agrees that impairment is occurring as result of nutrient discharges, the Siloam Springs discharge permit will be re-examined to consider a requirement for nutrient removal.

As previously mentioned, the states have established a TMDL working group, which meets under the auspices of USEPA to determine the goals and procedures for developing a TMDL for the entire length of the river. This process has the greatest potential for addressing river problems as the levels of nutrients that the river and Tenkiller Ferry Reservoir can accommodate will be determined. After allowable loading is determined, allocation of loading to individual states and within states to point and nonpoint sources will be conducted. This will be a difficult process, both from a technical and political viewpoint; however, if agreement can be reached through this process, then considerable progress will have been made on inter-state cooperation and river protection.

The two states are also working to develop a joint river basin management plan, also under the auspices of USEPA. Each state is developing its own plan which will subsequently be merged into a single comprehensive basin management plan. The intention of this plan it to provide a useful framework for addressing river problems from a holistic viewpoint, rather than by the piecemeal fashion that has been done in the past. The development of a TMDL is central to the success of the basin management plan.

From these experiences it would appear that the most appropriate method for solving inter-state problems is through inter-state working groups. It is obvious that USEPA should play a major role in this process and should serve as an arbiter of disputes. Since the majority of funding for environmental programs comes through USEPA, this would appear to be an appropriate role. In regard to the financial role of USEPA, it should be pointed out that a very limited amount of funds has gone towards addressing nonpoint source issues. USEPA funding for wastewater treatment facilities in the basin has been in the tens of millions of dollars, while funding for nonpoint source controls has been in the hundreds of thousands of dollars. It is probably unrealistic to expect voluntary controls on nonpoint source pollutants to occur unless significant additional cost share funds are directed towards this source.

Conclusions

The issue of nutrient pollution presents many challenges to regulators, environmentalists, landowners, and point source producers. Nutrients have not traditionally been looked upon as priority pollutants; however, as the national focus has shifted from point source toxicity to general ecological health, an awareness of their importance has emerged. Despite this increased awareness, the development of accepted nutrient criteria has been limited. Assessment of nutrient effects through changes in biological communities would appear to have the greatest potential, although these methods are not widely accepted outside of the scientific community.

Conflict between Oklahoma and Arkansas over nutrient issues has not resulted in a satisfactory resolution of problems. There remains controversy over many points, both political and scientific. The relative contribution of nutrients from each state, the magnitude of point versus nonpoint sources, and the ability to prove nutrient-based impairment present technological hurdles to solving problems. The applicability of downstream state standards and the difference between the states with respect to the value of the river and the surrounding environment continue to be topics of political discussion. A number of inter-state efforts have been initiated to address environmental issues and these efforts would appear to have a significant potential for success. A strong effort is being made to base decision making on science rather than politics and both states appear to agree on this more objective approach for addressing specific issues.

NUTRIENT TRADING AS A MANAGEMENT OPTION:

THE TAR-PAMLICO EXPERIMENT

By

Douglas N. Rader[1]

Introduction

Past strategies for combatting nutrient pollution have proven largely ineffective. In particular, addressing the nutrient contribution from diffuse (nonpoint) sources, such as agriculture, is both technically challenging and politically difficult. As a result, the North Carolina Environmental Management Commission (EMC), the U.S. Environmental Protection Agency (EPA), and several governmental entities around the country have begun to explore a new approach to nutrient reduction, one that features "nutrient trading."

This approach is similar to other environmental trading programs that have received growing attention in recent years (Hahn and Hester, 1988; 1991). The most notable of these is the acid deposition program contained in Title IV of the Clean Air Act Amendments of 1990.[2]

Trading programs seek to achieve environmental goals in the most cost-effective manner by providing flexibility to the universe of polluters causing a problem. Such programs typically set an overall environmental goal -- the total quantity of a pollutant that will be allowed into a system over a specified time period --and then apportion that amount among the specific actors or sectors contributing to the problem. By allowing the players to trade such allocations among themselves, the program lets market forces produce a cost-effective outcome. Trading programs take advantage of the disparity in control costs between various activities and firms that generate the pollutant of concern, provide incentives for polluters to overcontrol their discharges or emissions, and recognize conservation and efficiency as acceptable alternatives to mandated technology (Tietenberg, 1985).

In December 1989, at the behest of a coalition of point-source dischargers and environmentalists, the EMC adopted a nutrient reduction strategy for the Tar-Pamlico River Basin that, for the first time in an estuarine context, features nutrient trading (EMC, 1989). Specifically, publicly owned treatment works (POTWs) in the basin, which traditionally have borne the brunt of nutrient control efforts, are allowed to achieve required reductions in nutrient loadings to the Tar-Pamlico by funding the implementation of agricultural best management practices (BMPs); the premise of the program is that such BMPs achieve substantially more cost-

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[2] Clean Air Act Amendments of 1990, P.L. 101-549 secs. 401-416, 104 Stat. 2584-631 (codified at 42 U.S.C. sec. 7651-7651o.

effective nutrient control than further advanced treatment at the plants themselves.

This paper first provides background on the nutrient problem nationally and in North Carolina. I then discuss the limitations and failings of traditional nutrient control strategies. Finally, I discuss the Tar-Pamlico program and raise a number of critical issues regarding the future of that program and nutrient trading generally.

Nutrient Pollution in the United States

Nutrient pollution is one of North Carolina and the nation's most serious and intractable water pollution problems. A wide array of sources -- including agriculture, sewage treatment plants, mining, industrial dischargers, and both stationary and mobile air emitters -- are responsible for the nutrient loadings to our lakes, rivers, and estuaries. This nutrient pollution, primarily in the form of nitrogen and phosphorus, fuels explosive algal growth and attendant deoxygenation, or eutrophication, of water bodies. The severe stress on aquatic ecosystems resulting from excess nutrient levels has been responsible for massive fish kills and implicated in epidemic fish and crustacean diseases observed in recent years.

Nationally, fully 49% of degraded lakes and reservoirs, 28% of degraded streams and rivers, and 48% of degraded coastal waters are overenriched with nutrients from a variety of sources. Only sediment pollutes more miles of streams. Agriculture is by far the most important source of nutrient pollution to freshwater aquatic systems in the United States, accounting for roughly 58% of lake pollution and 53% of stream and river pollution. In coastal waters, agriculture accounts for roughly 20% of nutrient pollution. Other important sources of nutrients to surface waters include municipal sewage treatment plants (about half of contaminated coastal waters and 15% of contaminated streams and rivers), and other nonpoint source pollution (mining, urban runoff, silviculture and construction) (USEPA, 1990).

Many of our nation's most important waters are degraded or threatened by nutrient enrichment. These include Chesapeake Bay, Long Island Sound, Buzzard's Bay, the Gulf of Mexico, the Gulf of Maine, Narragansett Bay, and the Everglades. A detailed discussion of the extent of nutrient-related deoxygenation in U.S. estuaries is given in Stanley (1985); Rabalais, Dagg and Boesch (1985) and Whitledge (1985).

Nutrient Pollution in North Carolina

Many of North Carolina's most important coastal rivers display serious signs of nutrient pollution. Algal blooms have caused oxygen depletion and kills of fish and other aquatic organisms in the Tar-Pamlico River Estuary, the Neuse River Estuary, the Chowan River, and the New River. The disappearance of rooted aquatic plants and the appearance of previously unknown fish and crab diseases have also been related at least in part to increasing nutrient pollution. Furthermore, the upstream, impounded portion of the Cape Fear River has also displayed severe eutrophication. All of these water bodies have been officially declared to be nutrient enriched, and assigned the supplemental classification of "Nutrient Sensitive Waters" by the EMC.

The North Carolina Division of Environmental Management (DEM) estimates that fully 82% of degraded estuarine waters suffer from algal blooms or oxygen depletion. Although nutrient pollution seems less severe in freshwater, accounting for only 5% of degraded lakes, many other lakes are considered by the state to be "threatened" by accelerating eutrophication, notably Falls of the Neuse Reservoir, and Jordan Lake. The large Piedmont lakes in the Catawba Basin also show signs of nutrient enrichment and eutrophication. Degradation of freshwater streams and rivers by nutrients was not expressly listed by DEM, being lumped into the "other" category, comprising 60% of degraded waters (DEM, 1990).

The principal causes of water pollution in North Carolina mirror national trends. Roughly 80% of degraded waters are degraded by nonpoint source pollution. Nutrient budgets drawn up for impaired coastal river basins identify nonpoint sources, principally agriculture, as being responsible for about 75% of nutrient pollution. (DEM, 1990). Technical assessments of nutrient limitation conducted in situ have revealed that both nitrogen and phosphorus are often limiting at different times of the year (Paerl, 1983; 1987).

Limitations of Traditional Nutrient Control Strategies

Past attempts to control nutrient pollution have been largely unsuccessful, mostly as a result of the inability of government agencies to deal with nonpoint source pollution in a direct, regulatory fashion (Thompson, 1989). As a result, nonpoint pollution control remains a largely voluntary effort in the United States. EPA's recently released nonpoint source strategy is based on passing the buck through the states to local units of government (USEPA, 1989). Many states, including North Carolina, have developed wholly voluntary cost sharing programs as their primary, if not exclusive, measure for addressing agricultural pollution. The North Carolina Agricultural Cost Share Program provides roughly $8 million per year to implement best management practices statewide, implemented through local conservation boards at a ratio of 3:1.

The traditional approach to nutrient control begins with identification of nutrient impaired waters. Identification of a nutrient-impaired water body normally occurs when excessive nutrients appear in the water, when excessive algal growth can be measured as a result of nutrient enrichment, or when dissolved oxygen levels are depressed as a result of algal or decomposer metabolism. Impairment is usually measured against some stipulated numerical water quality standard, either for nutrients, for algal pigments such as chlorophyll-a, or for dissolved oxygen (or some combination). For example, New Jersey has water quality standards for total phosphorus (0.05 mg/l in lakes; 0.10 mg/l in streams, unless it can be shown that phosphorus is not limiting), and for dissolved oxygen.[3] North Carolina relies on standards for chlorophyll-a and dissolved oxygen. In contrast, Florida has a narrative standard for nutrients: "In no case shall nutrient concentrations of a body of water be altered so as to cause an

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[3] N.J.A.C. 7:9-4.14(c)(6).

imbalance in natural populations of aquatic flora or fauna."[4]

Such identification is complicated by the fact that natural systems are heterogeneous, and normally contain varying nutrient loads which may prompt blooms of algae at some times and some places within the water body irrespective of anthropic inputs. Thus, concentrations of nutrients such as nitrogen or phosphorus or levels of chlorophyll-a may violate water quality standards even with minimal human input. Similarly, many lakes and estuaries are subject to stratification which may cause the development of bottom layers of water that are oxygen poor. Nutrient additions to such waters commonly cause an increase in the frequency, duration and spread of such anoxic or hypoxic conditions. Where baseline water quality data are sparse, or where heterogeneity is great, definitive determinations of nutrient overenrichment become technically difficult.

North Carolina guards against inappropriate classification by excluding apparent violations of dissolved oxygen standards when they can be shown to be a result of natural causes. Thus, "swamp waters, lake coves or backwaters and lake bottom waters may have lower values if caused by natural conditions,"[5] and "swamp water, poorly flushed tidally influenced streams or embayments, or estuarine bottom waters may have lower values if caused by natural conditions."[6]

In North Carolina, the identification process proceeds under the jurisdiction of the EMC as a supplemental water quality classification, authorized in 15 NCAC 2B. 0214. Waters that "are experiencing or are subject to excessive growths of microscopic or macroscopic vegetation" which the EMC finds "to substantially impair the use of the water for its best usage" may be classified Nutrient Sensitive Waters (NSW).[7]

Once a water body is identified as nutrient enriched, goals for reduction are commonly established. The mechanisms used to set such goals vary widely. In the Chesapeake Bay region, the goals for nutrient reduction of 40% for both nitrogen and phosphorus were adopted by governmental representatives from all states in the Chesapeake watershed (USEPA, 1992). Once such goals are established, then strategies to attain those goals must be designed and implemented.

In North Carolina, no specific goal-setting mechanism exists. The water quality rules which apply to waters classified NSW include a narrative standard of no discharge above

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4 Rules and Regulations of the State of Florida, Title 17, Chapter 17-3.091(20); see also id. 17-3.111(16) and 17-3.121(19).

5 15 NCAC 2B .0211(b)(3)(B).

6 15 NCAC 2B .0212(b)(3)(B).

7 15 NCAC 2B .0214(a).

background levels -- those levels upstream of any particular source.[8] This standard is to be implemented, unless such restrictions "would cause a serious economic hardship without equal or greater benefit to the public."[9] Goals generally result from a consideration of control strategies that are practicable instead of an a explicit consideration of the level of reduction necessary to achieve compliance with water quality standards. An exception to this general rule was the analysis conducted prior to the classification of the Chowan River as NSW, where specific numerical projections were made relating phosphorus loads to chlorophyll-a concentrations (DEM, 1982).

In practice, effluent limits set at upstream concentrations have never been adopted for any NSW-classified water body. Instead, alternative nutrient management strategies are developed by the DEM staff and adopted by the EMC. The strategies adopted for the upper Cape Fear Basin (Jordan Lake watershed) and Upper Neuse Basin (Falls Lake watershed) focused on removal of phosphorus from point sources, imposing a 2 mg/l effluent limit in lieu of the stricter "background" concentration approach, and encouraging nonpoint controls through voluntary participation of farmers in the North Carolina Agricultural Cost-Share Program (DEM, 1983). The Chowan River Basin NSW strategy, adopted in 1982 by North Carolina and in 1985 by Virginia, featured reduction goals of 35% basinwide for phosphorus, and 20% in the North Carolina portion of the basin for nitrogen. The reductions were to be met in North Carolina by a combination of point-source effluent limitations, conversion of discharges to land application, and funding of best management practices through both states agricultural cost-share programs (DEM, 1982; Virginia Water Control Board, 1985). A recent appraisal of the success of the program found that North Carolina had achieved a 29% reduction in phosphorus. Virginia has depended mostly upon nonpoint-source controls to achieve its reductions, and has lagged behind (DEM, 1990). In fact, the recent Nutrient Enriched Waters designation in Virginia did not include the Chowan Basin; point source controls are still not fully in place.

Other NSW classifications in North Carolina include the Lower Neuse and the New River. The Lower Neuse strategy was adopted in February 1988, and focuses on removal of phosphorus from point sources, requiring effluent limitations of 2 mg/l for all discharges above 50,000 gpd and all new or expanding dischargers. Compliance was expected in 1993. Nitrogen reductions were to be achieved through voluntary BMP's, and were expected to be minimal: 10% (Dodd, 1992). DEM began managing the New River as NSW in 1987 by action of Director Paul Wilms, applying phosphorus limits of 2 mg/l to all new, renewed or expanded NPDES permits with design flows greater than 50,000 gpd (DEM, 1988). The EMC recently formalized this classification.

One serious difficulty in adopting nutrient reduction goals and in designing nutrient management strategies is the relatively poor understanding of the complex natural systems that are the focus of such efforts, particularly estuaries. Costly efforts are underway in the Chesapeake Bay and Long Island Sound to develop models that will relate nutrient inputs to algal

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[8] 15 NCAC 2B .0214(e).

[9] 15 NCAC 2B .0214(f)(2).

growth and deoxygenation in bottom waters of both estuaries. Such models usually contain two principal functions, one to predict responses within the estuary itself and the other to route materials within the watersheds to assess the relative importance of nutrients delivered into different portions of the watershed. Clearly, one kilogram of nitrogen removed from a small headwater area is not equivalent to a kilogram removed directly from a problem reach of the estuary. Successfully predicting the effectiveness of a nutrient control program design depends upon understanding the system well enough to predict both how much less nutrients will be delivered to the problem area and what effect that reduction will have once achieved.

The history of nutrient control in the United States in many ways reflects the slow development of our scientific understanding of how aquatic systems function, but also our propensity to address discrete sources of pollution at the expense of nonpoint sources. Early technical work on eutrophication focused on the effects of phosphorus enrichment in freshwater lakes. Similarly, regulatory action based on the CWA focused initially on point sources of phosphorus pollution, with only peripheral consideration of nonpoint sources and no consideration of nitrogen controls. In fact, EPA funding policies actively discouraged implementation of nitrogen removal or multiple nutrient removal (STAC, 1986). For example, over $1 billion was spent in the 1970's to renovate the large sewage treatment plants on the Potomac River, including very stringent effluent limits for phosphorus, but not for nitrogen, which increased in loading by 7% during the decade (ICPRB, 1982). This investment resulted in significant improvements in water quality generally, but was not totally effective because nitrogen was not controlled. Many estuaries are now known to be either nitrogen limited or limited by both nitrogen and phosphorus (Paerl, 1989).

In some cases, severe nutrient enrichment problems have been dealt with successfully by such traditional means. Two cases in point are the Delaware River and Lake Washington. Both of these water bodies were enriched primarily as a result of direct discharges of nutrients from sewage treatment plants. In fact, anecdotal evidence suggests that the Delaware could be smelled by airline pilots at flying altitudes. Lake Washington was nothing short of an open sewer in the 1950's. In both cases, the commitment of public funds to improved sewage treatment led to dramatic recoveries of these important waters. Today, Lake Washington sparkles with life and is an important recreational resource for the region. The Delaware system still shows algal growth maxima at the freshwater/saltwater boundary, but nothing compared to former levels (Sharp, 1988).

Another facet of traditional nutrient control that has shown some success has been bans on the use of phosphate in many detergent applications. The results of North Carolina's phosphate bans are similar to the experience with other such initiatives: a reduction in both influent and effluent phosphorus concentrations of about one third (DiFiore, 1988).

The Nutrient Trading Alternative

In light of the limited progress made through traditional nutrient programs, a number of interested parties -- including regulators, dischargers, and environmental groups -- have begun to explore the alternative of nutrient trading (USEPA, 1992; Bartfield, in press). A trading program requires that a total nutrient loading limit be established for a water body (or for a category of sources discharging to a water body) and apportioned among relevant sources. Instead of mandating controls or numerical effluent limits for all sources within each category, trading programs allow sources within a category, or within the system as a whole, to trade allowances -- or, put conversely, reduction obligations -- among themselves. The goal is to let market forces find the most cost-effective mix of reductions that will achieve the water quality goal. By reducing total costs, a trading program can also make pursuit and achievement of the goal more viable politically.

Where a unit of a pollutant from one source in a system is not comparable to a unit from another source (e.g., upstream vs. downstream sources), or where a safety factor is required to ensure that the desired reduction is achieved, trading ratios may be used. Thus, a program could set a point/nonpoint ratio of, say 1:3, in which case three units of nonpoint source reduction would be necessary to obtain credit for one unit of point source load reduction.

While nutrient trading can be applied exclusively to point sources or nonpoint sources, the greatest promise appears to lie in the area of point/nonpoint trading. As a general matter, such "low-tech" practices as installing buffer strips or retention ponds, composting, reducing nutrient inputs, or restoring wetlands achieve far cheaper nutrient removal than advanced, high-cost wastewater treatment technologies. Under a trading program, point sources have an incentive to meet their regulatory obligations by helping to reduce nonpoint source discharges. Point/nonpoint trading programs also have the potential to solve the historical policy conundrum of nonpoint source control. By serving as the stimulus for the adoption of a basinwide strategy of the sort envisioned by existing federal water quality planning processes, such programs underscore the need for limits on total nonpoint, as well as point source, loadings. A fully operating trading program would include such limits, but could achieve them in a flexible, cost-effective, and politically acceptable way (rather than mandating uniform controls on all nonpoint sources).

EPA has identified the following necessary conditions for a successful point/nonpoint source trading program:

a. The water body must be identifiable as a watershed or segment;

b. There must be a combination of point sources and controllable nonpoint sources and each type of source must contribute a significant portion of the total pollutant load;

c. There must be a water quality goal for the watershed that necessitates action;

d. There must be accurate and sufficient data with which to establish targets and measure reductions;

e. Point sources, at a minimum, must meet technology-based discharge requirements as required by the Clean Water Act.

f. There must be significant load reductions for which the marginal cost of each pound reduced of nonpoint source controls (multiplied by the trading rate) is lower than for upgrading point source controls;

g. Point sources must be facing requirements to either upgrade facility treatment capabilities or trade for nonpoint reductions in order to meet water quality goals;

h. There must be an institutional structure to facilitate trading and monitor results; and

i. Sufficient and effective implementation mechanisms must be in place or enacted as part of the trading system -- including appropriate enforcement mechanisms (USEPA, 1992).

Prior to the EMC's adoption of the Tar-Pamlico strategy, only a few trading programs had been initiated in the water quality arena. These programs are described briefly in Levitas and Rader (1992) and in detail in USEPA (1992). For a variety of reasons, none of these programs has actually resulted in any trades.

The Tar-Pamlico Program

Given the history of nutrient-related problems in the Tar-Pamlico basin, in 1988 the Pamlico-Tar River Foundation (PTRF), a citizens group based in Washington, NC, petitioned the EMC to classify the basin as "Nutrient Sensitive Waters." In response to this petition, in April of 1989, DEM published a report and set of recommendations entitled "Tar-Pamlico River Basin: Nutrient Sensitive Waters Designation & Nutrient Management Strategy." In that report, DEM made the following principal recommendations:

(1) that the Tar-Pamlico Basin be classified NSW;

(2) that "[n]utrient reduction studies . . . be performed to better define what nutrient level reductions are required to alleviate nutrient-related water quality problems on a long-term basis;"

(3) that an initial nutrient goal be adopted of achieving "no increase in nitrogen or phosphorus inputs to the basin from point sources and a reduction in nitrogen and phosphorus inputs from nonpoint sources;"

(4) that the nonpoint source goal be achieved through expansion of the North Carolina Agricultural Cost Share Program in the upper portion of the basin and an emphasis on animal feeding operations and other major nitrogen sources;

(5) that the point source goal be achieved through the following series of actions:

a. new dischargers would be required to evaluate non-discharge alternatives as their primary option and implement such a system unless they demonstrate that doing so is technically or economically infeasible (in which case, those new dischargers larger than 0.1 MGD would have to meet effluent limits of 6 mg/l on total nitrogen and 2 mg/l on total phosphorus);

b. existing dischargers would not be required to meet the foregoing effluent limits except upon expansion which resulted in a design flow greater than 0.5 MGD. (Upon permit renewal, however, existing dischargers would be required to have a permit reopener that would allow for the inclusion of numerical nutrient limits in the future if, based on the recommended studies, such limits were determined to be necessary).

(6) that the strategy be re-examined periodically and modified as necessary.

These recommendations were the subject of a public comment and hearing process conducted by the EMC in the summer of 1989. The Environmental Defense Fund, PTRF, and other environmental groups were extremely dissatisfied with the proposed NSW implementation strategy. Our primary criticisms were that: (1) the strategy did not include a specific nutrient reduction target -- making it the first estuarine nutrient control program in the country not to feature such a numerical goal; (2) the strategy did not include a goal of reducing nutrient pollution from point sources; (3) a more aggressive, targeted approach to nonpoint sources was needed; and (4) substantial projected increases in atmospheric nutrient contribution were not considered and no effort was made to address atmospheric loading.

At the same time, point source dischargers in the basin -- primarily the publicly-owned treatment works were alarmed by the projected costs of complying with the proposed numerical effluent limitations. The dischargers understandably complained of the inequity of the proposed strategy, which seemed to affect one category of sources disproportionately, and not the most significant one at that. They also argued that a long-term nutrient strategy for the basin required a more complete understanding of the nutrient dynamics of the system (including the relative impacts of different nutrients, different source categories, and different loading points).

The environmentalists and dischargers therefore jointly petitioned the EMC for additional time to develop an alternative plan for nutrient reduction in the Tar-Pamlico basin. At its September 1989 meeting, the EMC approved the NSW designation but granted the petitioners' request, with the proviso that the hearing officer's recommendation would go into effect if an alternative strategy was not developed and adopted at the end of 90 days.

Over the next several months, the environmentalists, the dischargers, and DEM met regularly to work on an alternative strategy. Just before the 90-day deadline, all parties signed off on an alternative NSW implementation strategy to present to the EMC. This document, which the Commission unanimously approved at its December 1989 meeting contained the following principal features:

- To assist in the development of long-term nutrient reduction goals and strategies, an association of point-source dischargers (The Tar-Pamlico Basin Association) agreed to fund (at an estimated cost of $400,000) the preparation of an estuarine nutrient model for the basin.

- Pending completion of the model and the development of a long-term NSW plan, the dischargers would be required to achieve approximately the same level of nutrient reduction that would have resulted from implementation of the hearing officer's recommendation. DEM projected that between 1989 and 1995 Rocky Mount, Greenville, and Pinetops would expand their sewage treatment plants and therefore be subject to numerical nutrient limits under the original strategy. DEM estimated the application of such limits to those plants would result in a 191,000 kg reduction in annual nitrogen loading and a 15,000 kg reduction in phosphorus.

- In lieu of in-plant improvements, the dischargers could achieve these reductions by making payments into the cost-share program to help fund the installation of agricultural BMPs in the basin. Based on past experience, these payments were calculated at $56 per kilogram of nutrient which included a safety factor of 3 for cropland BMPs and 2 for animal BMPs. In addition, the dischargers agreed to contribute $150,000 toward the administrative costs of BMP implementation.

- The dischargers agreed to perform an engineering evaluation of their existing plants to identify cost-effective operational or minor capital improvements to reduce nutrient loads. Any reductions achieved as a result of such improvements would be credited toward the dischargers reduction goal.

- Existing dischargers expanding to greater than 0.5 MGD who were not part of the association would be required to meet the numerical nutrient limits or offset any additional nutrient loadings through BMP payments.

- New dischargers would be required to meet the numerical nutrient limits.

- A section 208 planning agency, including diverse interest groups, would be established to assist with long-term planning and nutrient management in the basin.

- DEM retained the authority to require additional nutrient removal where localized water quality problems arise.

- Finally, if the dischargers failed to fulfill their funding obligations under the alternative strategy, all existing dischargers with design flow greater than 0.1 MGD would have to comply with the numerical nutrient limits within five years.

Given the short time frame in which this strategy was developed, and the many complex issues involved, the parties soon identified a number of details of the plan that required further attention. Negotiations over the next two years produced a revised strategy document that the EMC approved in February of 1992. The most serious problem was that the interim nutrient goal had been expressed as level of reduction without an explicit statement of the relevant baseline. Moreover, to be implemented, the goal needed to be expressed as a total nutrient load for the Association that could be measured based on monitoring reports and offset through BMP payments.

All parties agreed that the interim nutrient goal should be one that reflected a 200,000 kg reduction in Association nutrient loads from the levels that would result if flow increased as projected and no new controls were required. However, the parties had rather different ideas about (1) the dischargers' projected 1994 flows; and (2) what the dischargers' nutrient concentrations would be with no new controls. Reaching agreement on these points proved to be a formidable challenge, but the revised strategy document does now provide for a series of stepped down annual nutrient loading limits for the Association. The revised strategy also includes a monitoring and reporting protocol, on the basis of which DEM can determine whether the Association will be required to make BMP payments for a given year. Because the point-source limit applies to the Association as a whole, the program builds in the potential for trading between point sources.

In addition, the engineering evaluation revealed that substantial nutrient reduction could be achieved at the POTWs at less expense than anticipated -- with the possible result that no BMP payments would be required. (Indeed, as a result of such improvements, the Association was 20% below its allocation for 1991.) Because all parties were interested in using the interim period to test the viability of point-nonpoint trading, the dischargers agreed to make a minimum payment of $500,000 into the BMP fund over three years. The revised strategy also clarifies that BMP credits shall have a useful life of ten years unless cost-share program contracts with the nonpoint sources provide for a longer period.

The nutrient trading program has the distinct advantage of requiring an integrated assessment of nutrient control opportunities. For the first time, BMP funds will be targeted on the most serious nonpoint problems, instead of allocated on an ad hoc basis, and installation and performance will be tracked. An effective targeting and tracking process will be crucial to the long-term success of this and any other nutrient trading program.

The net effect of the nutrient control program for the Tar-Pamlico will be to reduce total nutrient loads from the POTWs involved from a peak of almost 670,000 kg in 1989 to a maximum of 425,000 kg in 1994, a reduction of at least 36%. In addition, the mammoth retooling of the waste management system at Texasgulf, by far the basin's biggest discharger and the source of up to half the total basin's phosphorus, will achieve a reduction in phosphorus from all sources of close to 50%. Total nitrogen loading reductions will be much less, because the major sources of nitrogen loading, particularly agriculture, have no specific requirements to achieve reductions in Phase I. Nitrogen loads could even increase.

Recent developments show the effectiveness of the overall program, even in its interim phase. Minimum payments from the dischargers and funds from federal sources are being combined to pay for a very exciting integrated nonpoint source control program in the Chicod Creek basin, a 35,000 acre watershed channelized in the late 1970's/early 1980's by SCS. "Before" and "after" nutrient monitoring is being conducted by DEM, along with biological monitoring. These efforts will help assess the actual reductions accomplished by the implementation of BMP's in this critical portion of the basin.

The Future of Nutrient Trading in the Tar-Pamlico Basin

Nutrient trading represents the latest attempt at the integrated basinwide management of point and nonpoint source pollution that has evaded regulators for decades. By driving total program costs down, trading has the potential to make the development and attainment of water quality goals -- and the targeted effective control of nonpoint sources -- a reality. However, many unanswered questions remain, with regard to the Tar-Pamlico program and nutrient trading generally.

Several features of the Tar-Pamlico program bear further scrutiny. First, the interim goals for nutrient reduction are clearly inadequate over the long term. The timidity of the original DEM proposal is well-documented by the impact of the minor tune-ups at the plants. More rational nutrient reduction goals will be identified after the estuarine water quality models are completed and adopted as part of the revised strategy by 1995. (Hydroqual, the contractor for the estuarine water quality model, has now completed its work and will present the model for evaluation to DEM next week.) A back-of-the-envelope prediction for nitrogen reduction requirements is in the 30-50% range. Achieving such reductions will require a more comprehensive approach to sources of nitrogen not currently regulated, such as agricultural and atmospheric inputs. Similarly, the final targets and programs adopted for this basin must acknowledge the unique liabilities and opportunities associated with controlling nitrogen rather than phosphorus, and should focus on nitrate nitrogen elimination instead of transfer into the groundwater. The use of wetland restoration as a BMP, with concomitant increases in denitrification, presents a potent tool in this regard.

(The development of appropriate reduction targets is made more complicated by the recent discoveries by Dr. Joann Burkholder of a weird, toxic dinoflagellate which is apparently responsible for many of the huge fish kills in the Pamlico, and which reacts strongly to phosphorus concentration (APES, 1993).)

Second, it remains to be seen whether nonpoint controls driven by trading will provide an effective substitute for point-source controls. As EPA has observed:

> Nonpoint sources are typically spread out within a watershed and loadings are more diffuse and random than point sources, and are generally more dependent on the weather and topographical conditions. Further, there is a greater degree of uncertainty about the effectiveness of nonpoint source control, especially about the actual reductions achieved and the permanency of those reductions. Point source loadings, while more costly to reduce (in most circumstances), are more easily monitored and regulated, whereas nonpoint sources are far more difficult to monitor and are largely unregulated (USEPA, 1992).

In the case of the Tar-Pamlico program, monies generated by trading do not go directly to specific projects, but are channeled through a fund administered by DSWC as part of the cost-share program. The virtue of this approach is that it takes advantage of an existing administrative structure, but at the same time it prevents the development of an actual market in emissions credits or reduction obligations. In addition, the Association's responsibility for offsetting excess discharges ends with its payment to the BMP fund. The Division of Soil and Water Conservation maintains implementation and compliance authority for BMPs. This relationship requires pseudo-regulatory functions from a strictly non-regulatory agency, and provides no clear enforcement pathways should installation or operation of funded BMPs fail or maintenance lapse. Moreover, the opportunities for third party actions under this arrangement seem severely constrained.

Nevertheless, solutions to our most pressing water quality problems must be found. And those solutions must deal directly with pollution from agricultural and nonpoint sources. They must also be cost-effective in order to avoid squandering the limited resources that are available from the public for pollution control. We believe nutrient reduction trading provides an important and flexible tool that must be considered in the design of any program that fits these criteria.

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THE THEORY AND PRACTICE OF POLLUTION CREDIT TRADING IN WATER QUALITY MANAGEMENT

By

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Historically, the focus of water quality management has been on reducing point source effluent discharges. This approach has yielded some improvements in the nation's water quality but states continue to report to the U.S. Environmental Protection Agency (EPA) that significant portions of water bodies remain unfit for their designated uses. As point source contributions have decreased over the years, the relative share of nonpoint source contributions has increased making impairments from nonpoint sources such as sedimentation and nutrient enrichment more evident (USEPA, 1992a). Commensurate with the increased attention are calls for government intervention.

Policies to date have been dominated by point-source regulations. These regulations, based on pollution-mitigating technology, form the backbone of state and federal water quality management strategies for two reasons: (1) industrial and municipal point sources historically had been the worst and most obvious offenders of surface water quality in the 1970s, and (2) they were the easiest to address, because their loadings emerge from a discrete point such as an end of a pipe. Nonpoint source pollution is more difficult to manage because monitoring and enforcement are more complex when sources are diffuse (Cabe and Herriges, 1991; Segerson, 1988; Xepapadeas, 1991). It is harder to identify nonpoint polluters and to assess them for their marginal damages.

An alternative to regulating nonpoint sources is to create markets for trading pollution credits which permit polluters to allocate pollution allowances thus lowering monitoring and enforcement costs. In theory, pollution credit trading can help achieve the goal of controlling nonpoint source pollution in a flexible, cost effective manner. Nevertheless, in practice these programs have not performed well. Currently, four trading programs for trading water pollution credits have been implemented: two in Colorado, one in Wisconsin, and one in North Carolina. A fifth program is in the developmental stages in Colorado (USEPA, 1992b). To date, *no trades have been completed in any of these programs.*

In order to examine why trading activity has not lived up to theoretical expectations, we examine the theory of pollution credit trading and compare it to how the concept is applied in the Tar-Pamlico nutrient trading program in North Carolina. The Tar-

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Pamlico nutrient trading program is the most recently enacted trading program. It is unique because the point source dischargers involved are treated as a single unit with regards to nutrient loading limits, monitoring, and enforcement. This allows for the possibility of point/point source trading as well as point/nonpoint source trading. We begin with a general discussion about pollution trading theory as it relates to nonpoint sources in water. A detailed description of the Tar-Pamlico program follows which is organized to highlight the consistency of the program with theoretical constructs. We conclude with an assessment of the program, attempting to provide insight about why trades have not occurred and what could be done to encourage more trades. The assessment is based on observations from theory and the Tar-Pamlico trading program.

Pollution Credit Trading

General Theory

Pollution credit trading is a market-based alternative to traditional command-and-control regulations (Stavins and Whitehead, 1992; Hahn and Stavins, 1991). Command-and-control regulations tend to force all dischargers to adopt the same measures and practices for pollution control and thus bear identical shares of the pollution control burden regardless of their relative impacts. Unlike command-and-control policies, which seek to regulate the individual polluter, market-based policies focus on the overall level of pollution in a given area. In terms of policy, a market-based approach achieves the same aggregate level of control as might be set under a command-and-control approach but it permits the burden of pollution control to be shared more efficiently among dischargers. Market-based policies provide monetary incentives for the greatest reductions in pollution by the dischargers that can do so most cheaply. The result is that fewer total economic resources are used to achieve the same level of pollution control, or more pollution control is obtained for the same level of resources.

One type of market-based program is pollution credit trading or emission trading (Tietenberg, 1985). In essence, a regulator sets a ceiling on the amount of pollution allowed for a whole group of polluters within a "bubble" and issues permits to individual polluters within that bubble for their share of the total pollution allowed. Polluters can then buy or sell permits so that those who can clean up cheaply can do so and then make money by selling spare pollution credits to those for whom cleaning up would be more expensive. By allowing polluters to buy or sell pollution allocations among themselves, the program lets market forces produce a cost-effective outcome (Montgomery, 1972). Trading programs take advantage of the differences in pollution control costs between various polluters and provide incentive for polluters to "overcontrol" their discharges or emissions.

The theory of pollution credit trading is based on marginal cost analysis. An important proposition of pollution credit trading is that the costs of achieving a given reduction in discharges will be minimized only if the marginal costs of control are equalized across all dischargers. Following Tietenberg (1985), Figure 1 illustrates this point. Assume that there are two sources of the pollutant that face the same marginal cost of removing the pollutant and total pollution discharges must be reduced by 15 units. Figure 1 shows all possible allocations of the

15-unit reduction between the two sources. For example, the left-hand axis represents the allocation of the entire control responsibility to the second source while the right-hand axis represents the opposite situation--source one bears the entire control responsibility. All points in between represent different degrees of shared responsibility. The cost-effective allocation occurs when the marginal cost of reducing discharges is equal--when source one and two clean 7.5 units each. Any other allocation would result in higher total control costs. To see this, suppose before pollution credit trading, source one controlled 5 units and source two was assigned 10 units of pollution reduction. The sources would have an incentive to trade because source two's marginal cost (point A) is greater than source one's (point B). Source two could reduce its cost as long as it could buy credits from source one at a price lower than A. Source one would be better off if it could sell credits for a price greater than B. When the price of a pollution credit corresponds to level C, there are no further gains from trade. The shaded area represents the reduction in total cost possible if source one and two are allowed to trade pollution credits.

A very attractive feature of pollution credit trading is that control authorities can achieve cost-effective allocations despite their lack of knowledge about control costs. The control authority defines the total level of discharge reduction, leaving the ultimate choice about control responsibility up to the dischargers, who have better information on available control techniques and their associated costs than the control authority.

Pollution credit trading systems have been used on a limited basis in the U.S. Examples include the EPA's Emissions Trading Program (for air pollution management) and the nation-wide lead phasedown during the 1980s which allowed fuel refiners to "bank" and "trade" lead content savings (Dudek and Palmisano, 1988; Hahn and Hester, 1989). Experience with pollution credit trading is more limited as applied to water quality management.

Water Pollution Credit Trading Programs

Pollution credit trading in principle allocates reductions in pollutant loading across point and nonpoint sources within a particular watershed using least cost as the criterion. Point/nonpoint source trading is typically the trading option focused on academically and within a policy context (Letson, 1992; USEPA, 1992b). Nutrients (nitrogen and phosphorous) are most often the type of pollutant traded because (1) they are contributed by both point and nonpoint sources in significant amounts, (2) nutrient problems typically arise as a result of accumulated loadings, rather than from highly localized effects, and (3) there are a large number of difficult issues specific to trades involving toxic pollutants that have not been resolved. Point/nonpoint source trading has come to mean granting publicly owned treatment works and industrial point sources the option of bringing agricultural and urban nonpoint sources under control rather than simply requiring further controls at point sources. The regulator continues to focus on the more easily identified and managed point sources, but grants them more flexibility to pursue lower cost control options (Letson, 1992).

The Tar-Pamlico Nutrient Trading Program

Program Motivation

The Tar-Pamlico River Basin covers an area of 5,401 square miles in 16 counties in eastern North Carolina Piedmont, and inner and outer Coastal Plain regions (see Figure 2). It is the fourth largest river basin in the state. It contains 2,308 miles of freshwater streams and 148 active permitted surface water (fresh and salt water) dischargers. Seven dischargers are major municipals, twelve are minor municipals, two are major industrial, and 127 are nonmunicipal. Eighteen of these dischargers are categorized as major dischargers (with a flow of greater than 0.5 million gallons per day). Most of the major dischargers are publicly owned treatment works (municipal dischargers). The river basin also contains 634,400 acres of salt water.

Nonpoint source runoff from farming and forestry operations and point source dischargers, including a large phosphate mining operation near the terminus of the Pamlico River, are three major sources of nitrogen (N) and phosphorous (P) loadings in the Tar-Pamlico system. Point sources account for about 18% of the Tar-Pamlico's nitrogen load, while most of the rest comes from nonpoint sources. Agriculture and livestock, for example, contribute about 40% of the river's nitrogen, and forestry and air pollution each add another 20% (Pamlico-Tar River Foundation Newsletter, 1992).

In 1988, the Pamlico-Tar River Foundation, a citizens group based in Washington, NC, petitioned the Environmental Management Commission to classify the Tar-Pamlico River Basin as Nutrient Sensitive Waters (NSW). When a water body is designated as NSW, the Division of Environmental Management (DEM), part of the NC Department of Environment, Health and Natural Resources, must develop a special nutrient management plan for that water body--titled a NSW Implementation Strategy (the strategy). As part of the strategy, the DEM developed a set of recommendations. An initial nutrient goal of achieving no increase in N or P inputs to the basin from point sources and a reduction in N and P from nonpoint sources was recommended.

The recommendations of the DEM proved unsatisfactory to several environmental groups such as the North Carolina Environmental Defense Fund and the Pamlico-Tar River Foundation because neither a specific numeric nutrient reduction target nor a goal of reducing nutrient pollution from point sources was included (Levitas and Rader, 1993). Point source dischargers who had plans to expand their facilities by 1995 were concerned about the projected costs of complying with the proposed numerical effluent limitations. It was argued that meeting the proposed effluent limits would cause serious economic hardships to the public served by these dischargers because some facilities lacked the capability to remove N and P from their discharges and would face expensive upgrades[2]. The dischargers also pointed out that the

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2 Preliminary estimates indicated that $50-100 million would have to be spent on secondary and tertiary plant upgrades if each and every plant were to meet the nutrient limits at their current sizes (Green).

proposed strategy would affect point source dischargers much more than nonpoint source dischargers when nonpoint source loadings are more significant contributors to the nutrient problem (DEHNR). In light of these criticisms, the environmentalists, the dischargers, and the DEM developed an alternative plan for nutrient reduction in the Tar-Pamlico River Basin that contained a provision for nutrient credit trading. The alternative plan was approved by all parties in December, 1989, and revised in February, 1992 (Tar-Pamlico NSW Implementation Strategy, 1992).

The NSW Implementation Strategy

The strategy adopted by the DEM, the NC Environmental Defense Fund, the Pamlico-Tar River Foundation, and a coalition of dischargers called the Tar-Pamlico Basin Association (the Association) set up a two-phased plan to achieve basinwide nutrient reduction goals. During Phase I (1990 to 1994), the parties involved agreed to develop an estuarine computer model for the basin, conduct engineering evaluations of the wastewater treatment plants, and implement a nutrient reduction trading program. In Phase II (beginning 1995), a long-term nutrient reduction strategy will be implemented based upon the results of the estuarine computer model (scheduled for completion by December 1993).

The estuarine computer model is funded by the Association and overseen by the DEM. The purpose of the model is to better define the relationship between nutrient loadings and estuarine water quality. Specifically, the model will assess the relative importance of nutrients from point and nonpoint sources, sediments, and the atmosphere to algal growth and oxygen stress, recommend future nutrient target reductions, and track and target best management practices (BMPs) for reducing agricultural nonpoint source discharges.

The Association also funded the engineering evaluations conducted at individual member facilities. The evaluations were called for in order to optimize nutrient removal capabilities by point source dischargers. The evaluations (completed in 1991) indicated that most plants met the P limit originally proposed by the DEM, while N levels typically exceeded the originally proposed limits (USEPA, 1992b). By implementing operational and minor capital improvements recommended by the consultants, both sets of proposed nutrient limits were collectively achieved by the Association.

The Association has thirteen members--twelve publicly owned treatment works and one industry. Several major dischargers in the basin chose not to join the coalition, including the basin's nine other publicly owned treatment works and the large phosphate mining operation. Those dischargers generally felt that their plants either were or soon would be working well enough to meet the state's stricter nutrient limits. Thus, joining the Association seemed an unnecessary expense (Pamlico-Tar River Foundation Newsletter, 1992). Nonetheless, the aggregate discharge flow of the Association members represents 80% of the total permitted discharge within the river basin (Green, 1993). The Association determines its own operating rules and financial obligations of members. Program operating cost allocations to date have been a function of individual members' permitted flows, as a percentage of the Association's aggregate flow (Blount, 1993). No new members can be added to the Association during Phase

I. Association membership may be reopened to include other parties for participation in Phase II.

Nutrient Trading Details

Nutrient Reduction Goal -- The nutrient reduction goal of the strategy reflects the nutrient reduction level that would presumably have been achieved through the effluent limits originally proposed by the DEM. The proposed limits were calculated based on concentration limits and projected flow for the three facilities planning to expand before 1995 and would have resulted in a nutrient reduction of 180,000 kg/yr total N and 20,000 kg/yr total P. Thus, the Association's combined nutrient reduction goal for 1994 (the last year of Phase I of the strategy) was set at 200,000 kg/yr. Given the projected 1994 flows for Association members and assuming no nutrient reduction from pre-strategy concentrations, it was estimated that by the end of 1994 the Association's annual nutrient loading would reach approximately 625,000 kg/yr. The total nutrient reduction of 200,000 kg/yr is provided for through a series of stepped down annual nutrient loading limits for the Association, culminating in a nutrient load of 425,000 kg/yr in 1994[3]. The Association must provide annual reports to the DEM summarizing nutrient loadings and nutrient load reductions and the method of reduction for each member facility. Association members are held jointly responsible for achieving the annual nutrient loading allowance for the entire Association, in lieu of individual plant effluent restrictions. Within the nutrient loading allowance, members may allocate individual discharge levels among themselves. If the Association does not meet the nutrient loading allowance, it may purchase nutrient credits through the contribution of funds to the NC Agricultural Cost Share Program.

Agricultural Cost Share Program -- The Agricultural Cost Share Program is administered through the Division of Soil and Water Conservation, which is part of the NC Department of Environment, Health and Natural Resources and is the state's lead agency for agricultural nonpoint source pollution under the Clean Water Act Section 319. Participation by farmers in the Agricultural Cost Share Program is voluntary. The program pays 75% of the average cost to implement agricultural BMPs such as grassed waterways and livestock manure treatment lagoons and 50% of the cost of technical assistance to farmers. The maximum payment is $15,000 per year per farmer. The installation of BMPs is motivated by the farmer's desire to comply with federal programs and/or the desire to be a responsible steward of the land. Association funds will supplement state cost-share money already allocated to the Tar-Pamlico Basin.

The Association agreed to fund additional Division of Soil and Water Conservation personnel to assist in BMP review and identification. The purpose of these funds is to design and establish the nutrient-reduction trading system, including targeting and documenting existing BMPs in the basin and similar activities. The Division of Soil and Water Conservation will prioritize funding to BMPs that have the highest potential and efficiency for nutrient removal.

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3 The total nutrient loading levels for the Association are 525,000 kg/yr in 1991, 500,000 kg/yr in 1992, 475,000 kg/yr in 1993, and 425,000 kg/yr in 1994. Ninety percent of the loading level is the total N level and the remaining ten percent is the total P level.

Funds are allocated to local Soil and Water Conservation Districts.

If a farmer enters into a contract with the Soil and Water Conservation District and fails to maintain the BMP, the farmer is required to reimburse a prorated amount to the state. Monitoring of BMPs is accomplished through spot checks on a certain number each year (Hunt, 1993).

Price of a Pollution Credit -- In order to establish a point/nonpoint trading system, an appropriate trading ratio must be determined. The trading ratio is the amount of nonpoint source control that a point source discharger must undertake to create a credit for a given unit of point source discharge. The Tar-Pamlico trading program uses a 3:1 ratio for cropland BMPs and a 2:1 ratio for confined animal operations. The ratios were set at greater than 1:1 to provide a safety factor. Nonpoint source loadings are less predictable over time and space because they are more random and less reliably controlled than point sources. Accounting for these trading ratios, the Association must pay $56/kg of excess nutrient discharges (whether N or P) into the Agricultural Cost Share Program. This figure is based on the average nonpoint source control costs in watersheds in the Tar-Pamlico area plus administrative fees. All BMP pollution credits have a useful life of ten years unless cost share program contracts with the nonpoint sources provide for a longer period. If the nutrient reduction goal for the Association were met entirely through the funding of agricultural BMPs, it was estimated that $11.2 million would be needed for nonpoint source improvements (200,000 kg * $56/kg = $11.2 million).

Non-Association Dischargers -- Existing non-Association dischargers expanding to 0.5 million gallons per day or greater are subject to nutrient effluent permit limitations originally proposed by the DEM for expanding facilities--2 mg/l total P and 4 mg/l (summer) and 8 mg/l (winter) total N. Less stringent permit limitations may be obtained if offset by nutrient-reduction trading based upon a credit price of $62/kg/yr.

New dischargers will be required to evaluate non-discharge alternatives as their primary option, and implement a non-discharge system unless they can demonstrate that it is technically or economically infeasible. If implementation of a non-discharge system is not possible, new dischargers will be subject to the nutrient effluent permit limits proposed by the DEM. New dischargers will not be able to participate in the nutrient trading program.

Enforcement of Program Terms -- If the terms of the agreement are violated, all existing dischargers (Association and non-Association dischargers) must meet the nutrient effluent permit limits originally proposed by the DEM. If a localized water quality problem arises, the DEM may require individual point sources to remove nutrients. Also, if a discharger agrees to bring nutrient removal facilities into operation and the Association receives credit toward its allowable annual nutrient loading, but then the facility does not meet its projected nutrient removal level, the DEM may add nutrient limits to the facility's discharge permit. The Association must then pay for the projected pollution credits plus a penalty charge of 10%.

Program Costs -- The Association will spend approximately $1.13 million during Phase I of the strategy. The estuarine computer model was funded at a total of $400,000 and the engineering evaluations totaled $40,000. In order to ensure the availability of funds for the agricultural BMP implementation necessary to test the viability of point/nonpoint trading, the Association agreed to make minimum yearly payments to the Agricultural Cost Share Program. The sum of the minimum payments that must be made to the Agricultural Cost Share Program during Phase I is $500,000. The Association also provided a total of $150,000 to the Division of Soil and Water Conservation for additional staff members. Finally, legal and administrative fees to date total approximately $40,000 (Leyen, 1993).

Non-Association contributions have been important to the implementation of the program. The total amount of funds received from the EPA to date is $1.24 million with another $350,000 expected for FY93 (Harding, 1993). Added to the Association's contributions, the nutrient trading program has already cost approximately $2.72 million. Table 1 summarizes the program expenditures.

Program Assessment

Several factors that influence the success of the Tar-Pamlico program emerge from the above discussion. First, the transactions costs to make trades have been significantly reduced by the formation of a point-source polluter association. The Association, which is responsible for 80 percent of permitted discharges, can trade as a group with an established organization that represents farmers. Members of the Association have already paid the majority of the costs to make trades by partially supporting the farmer organization. The farmer organization incurs the costs to implement best management practices to offset the pollution credit to the Association. Lowering the transactions costs has a positive influence on pollution credit trading. In the first section below we discuss the potential advantages of this system which allows for point/point source trading within the Association, and point/nonpoint trades between the Association and the farm group. Transactions costs are ignored.

A second notable feature of this program is the implication of its administrative structure. Point source dischargers are made to pay for pollution control, but farmers are paid for pollution prevention. This arrangement brings about questions of equity, but more to the point for our purposes, it influences the costs and incentives for both farmers and the association, which is the topic of the second section below. In the third section, we discuss some of the limitations for trading that result from the program pricing rules. The price of trades is elevated in the Tar-Pamlico program by requiring the association to have a greater than a one-to-one trading ratio for farm to firm trades and by pricing trades at the average, rather than marginal, price of best management practices.

Potential Gains From Trading

The underlying cost structure of association member firms is not known. Nevertheless, insights can be gained about the potential for trades by comparing the program to the theory of pollution credit trading outlined earlier. Since the point source nutrient limits apply to the Association as a whole, the Tar-Pamlico nutrient trading program provides the opportunity for point/point source trading. This is advantageous because combining point/point and point/nonpoint trades results in a more cost-effective allocation than point/nonpoint trading alone.

A simple numerical example illustrates this point. Suppose there are only two point source dischargers and the marginal costs of reducing nutrient discharges are equal for all units of nutrient reduction. Also assume that one point source discharger, PS1, is responsible for paying for any required nutrient reduction and may trade with the other point source discharger, PS2. Figures 3 and 4 show that if PS1 is responsible for cleaning up 15 units of nutrient discharges, its most cost-effective method would be to buy half of the required units from PS2 and reduce the other half itself. At this allocation, marginal cost is equal and total cost is minimized. Total cost to PS1 would decrease from $150 if PS1 reduced all 15 units itself to $75 through trading with PS2[4].

Now suppose that PS1 is again responsible for paying for the discharge reduction and can trade with a nonpoint discharger. Assume the marginal cost for all units of discharge reduction by the nonpoint source (NPSa) is less than PS1's marginal cost (which is the commonly held view). In this example, the marginal cost of NPSa is half that of PS1. Referring to Figures 3 and 4, the cost-effective allocation would be for PS1 to clean up 5 units and NPSa to clean up 10. Total cost would be less than if PS1 traded with PS2 ($50 vs. $75). However, the total cost of reducing 15 units of discharge could be lower if PS1 were allowed to trade with both PS2 and NPSa. If pollution credit trading is conducted under marginal cost pricing, PS1 and PS2 would each clean up 3.75 units and NPSa would clean 7.5 units. Figure 5 depicts this solution. Total cost is driven down to $37.5.

Suppose that the marginal cost of nonpoint source nutrient cleanup is not less than the marginal cost of point source cleanup. Assuming the marginal cost of nonpoint source cleanup is twice that of the point sources, it can be shown that it is still better to allow trading between the point sources and the nonpoint source rather than restricting trades to between point sources. The marginal cost of nonpoint source cleanup is now NPSb on Figure 3. Total cost would be higher if PS1 traded only with NPSb instead of trading with PS2 alone ($100 vs. $75, see Figure 4) but allowing PS1 to trade with PS2 and NPSb would drive the total cost to $60 (see Figure 6). Therefore, the greater the number of trading participants, the more opportunities to trade, and the lower the total cost of a given number of units of discharge reduction. Table 2 summarizes the cost-effective allocations under the different trading scenarios.

Administrative Structure

The problem of establishing an appropriate market value for a pollution credit is not unique to nutrient trading programs. The EPA's Emission Trading Program has problems which

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[4] The total cost to PS1 is

$$\int_0^n f(x)dx + \int_n^L g(x)dx$$

where $f(x)$ is the marginal cost of PS1's cleanup, $g(x)$ is the marginal cost of the trading partner's cleanup, and PS1 reduces nutrient discharges by n units while the trading partner reduces discharges by $(L - n)$ units. In this example, $f(x)=g(x)=(4/3)x$ and $L=15$ units.

it is trying to eliminate by centralizing buying and selling interest in the new Chicago Board of Trade cash market and establishing a market for futures contracts and options on futures contracts in SO_2 emissions allowances (Rosenzweig and Villarreal, 1993). Exchanges such as the one the Chicago Board of Trade envisions for SO_2 are unrealistic for nutrient pollution credit trading due to the limited geographic size of a typical nutrient trading market. Instead, brokers could act on the behalf of buyers and sellers and issue lists of credits offered for sale, much as real estate brokers offer property for sale over multiple listing services[5]. However, nutrient trading programs differ fundamentally from SO_2 emission trading programs in the assignment of property rights to buyers and sellers. Different property rights are assigned to point source and nonpoint source dischargers. Point source dischargers are legally required to control nutrient discharges while farmers are not. A market with marginal cost pricing is more difficult to set up if the number of buyers and sellers are few and their property rights are different.

The uneven assignment of property rights can discourage trades in at least two ways. The first is a problem of enforcement. Pollution credit trading theory is based on self regulation among participants with equal responsibility. However, the Association is not involved in the implementation of nonpoint source controls beyond the point of providing nutrient reduction funds to the cost share program. The Association has no responsibility or authority to ensure that BMPs funded through trading are either implemented correctly or maintained. Furthermore, it does not have any input as to specific locations for BMP implementation within the basin. The Division of Soil and Water is responsible for targeting and implementing BMPs, and it relies heavily on local Soil and Water Conservation District officials to make inspections of BMP projects, and work with farmers to assure compliance. The arrangement relieves the Association from risk of noncompliance if BMPs are not successful in achieving nutrient load targets. However, individual members of the Association are at risk of the DEM instituting more stringent effluent limits on them, regardless of their participation in the Association and monetary contributions to the BMP fund (USEPA, 1992b).

The second way that the current system discourages trades is that farmers are not required to reduce pollution levels. Farmers may therefore be reluctant to accept trades since it could be construed as admission that they are polluters. In the absence of required pollution reductions, farmers would agree to participate only as sellers of pollution reductions where the costs of reducing their pollution are compensated with payments from point source members. The political attitudes toward farmers have been changing (for example, farmers are no longer exempt from OSHA rules and regulations) but voluntary behavior in response to positive incentives such as the cost-sharing program appears to be the mechanism that will prevail for years to come for achieving current national nonpoint source control objectives (Reichelderfer, 1990). Virtually all states rely on educational/voluntary approaches for addressing agricultural nonpoint source pollution control (Savage, 1985). Farmers would not want to speed up any movement away from their current relative advantage by drawing attention to themselves. Firms also may be reluctant to transfer control to outsiders if the costs of compliance are low since it

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5 Brokers already exist for trades in SO_2 credits (Reisch, 1992). Also, educational software has been developed to aid planners in making decisions about whether to trade or upgrade their facilities (Sheridan, 1992).

could convey sensitive information or focus attention on their pollution levels to the public. This is particularly problematic in the thin market for the Tar-Pamlico.

Finally, willingness to trade was highly dependent on the need to reduce pollution, which is a function of limits set by the North Carolina DEM. Since the Association was able to meet its loading allocation in 1991 and 1992, it was not necessary to allocate the loading allowance among member facilities, nor was the Association required to make excess loading payments. As a result of the operational and minor capital improvements implemented after the engineering evaluations, the Association was about 13% below its allocation for 1991 and 15% below in 1992. Despite increasing discharge volume, the Association is expected to meet the declining nutrient targets in 1993 and 1994 (Pamlico-Tar River Foundation Newsletter, 1992). Any point/point source trading activity could be completed at a marginal cost near zero because the nutrient loading allowance was not yet limiting (actual loadings were less than the allowed loadings). Until the marginal cost of point source reductions exceed the price of a nutrient credit, there is neither economic incentive nor need to conduct point/point or point/nonpoint source trades.

Program Pricing Rules

Marginal vs Average Cost Pricing -- One of the most important limitations of the Tar-Pamlico nutrient trading program is how the price of a nutrient pollution credit is determined. The $56/kg of nutrient is based on an average cost for the region. However, the key to achieving the most cost-effective distribution of nutrient reduction is marginal cost pricing. If the price of a pollution reduction credit is not based on marginal cost, any likely sequence of trades would produce a trading equilibrium which is not cost-effective (Tietenberg, 1985, p.90).

Figure 7 illustrates this point. Returning to the previous numerical example, suppose that the marginal cost to the Association of cleaning different levels of nutrients is represented by PS1 and the marginal cost to the nonpoint source is NPSa. Suppose the price of a nutrient credit is set at $5/kg (below the equilibrium level of $6.66). In this case, each source would clean up 7.5 units of nutrients. The nonpoint source would not be willing to sell more than 7.5 units because the marginal cost of doing so is greater than $5. Total cost to PS1 increases by an amount represented by area EAB. Area EAGF represents the amount of money no longer being transferred to NPSa from PS1. The loss in producer surplus to the farmer is represented by area ECDB. Suppose now that the price is set above the equilibrium level--say $8 (see Figure 8). The point source would clean 6 units and pay the nonpoint source to clean the remaining 9. Total cost to PS1 once again increases by an amount represented by are EAB. The farmer gains producer surplus (represented by area ABCD) but also loses some surplus (represented by area EBG). Area EGFH represents the amount of money no longer transferred from PS1 to NPSa.

Trading Ratio -- The average trading ratio used in the Tar-Pamlico nutrient trading program is 2.5:1. Setting the trading ratio at a level greater than 1:1 reduces the cost-effectiveness of nonpoint source controls from the perspective of point source dischargers because each credit becomes more expensive as the ratio increases. This discourages trading. Suppose the trading ratio were 4:1. Referring to Figure 3, if the marginal cost of nonpoint source cleanup is NPSa, the 4:1 trading ratio would increase the effective marginal cost of nonpoint source cleanup to

NPSb. Referring to Table 2, the point source dischargers would clean up more units of nutrients at a higher total cost than if NPSa determined the marginal cost of trading.

Conclusions

The objectives of federal water pollution control legislation have not been met in large part because of the contribution of nonpoint sources. Recent federal legislation has attempted to deal with the problem of nonpoint source contributions to water pollution but how nonpoint controls would be imposed and who would pay for them remains a subject of debate (Libby, 1985). One promising method of approaching nonpoint source control is to use market-forces such as nutrient credit trading programs.

The idea of using market forces to protect the environment has been supported by economists for the past 25 years. Only recently, however, has the broader policy community begun to regard market instruments favorably. In a new unreleased report from the US General Accounting Office, the EPA, States, and wastewater agencies are called upon to implement nutrient credit trading as a means of managing water quality (Carr, 1992). The EPA has identified 950 water bodies that could use trading for nutrient control (USEPA, 1992b). This increased attention on nutrient trading programs necessitates the careful study of existing nutrient trading programs such as the Tar-Pamlico program.

Dischargers have not made trades in any of the water pollution credit trading programs implemented so far. There are insufficient data to measure why trades are not being made. However, by comparing the implementation of the Tar-Pamlico program with the theoretical development of how markets should be implemented, we have been able to identify several factors that enhance or discourage trades. The program features many of the elements of an optimal design. Most notably, the program reduced transactions costs at the margin by creating a management unit to administer trades at a fixed rate. However, using the fixed rate eliminated the marginal cost benefits needed for efficient trading. In addition, trading costs were raised by the requirement that the Association buy more expected control than it needed, in order to account for uncertainty in BMP effectiveness.

Putting theory into practice requires compromise between the desire to promote cost-effectiveness on the one hand and the desire to have an administratively simple, yet politically acceptable program on the other. The Tar-Pamlico nutrient trading program has successfully instituted a market-based program that is administratively simple and politically acceptable and has taken important steps toward achieving the theoretical level of cost-effectiveness. Given the uneven assignment of property rights between point and nonpoint source dischargers and the resulting lack of a mechanism for marginal cost pricing, it is imperative that research continue on the monitoring and control of various types of BMPs. State implementation of trading programs is hampered by a lack of generally applicable models or data linking land use practices to water quality impacts. Improved information about the nutrient-removal capabilities of BMPs would also allow for a more cost-effective trading ratio.

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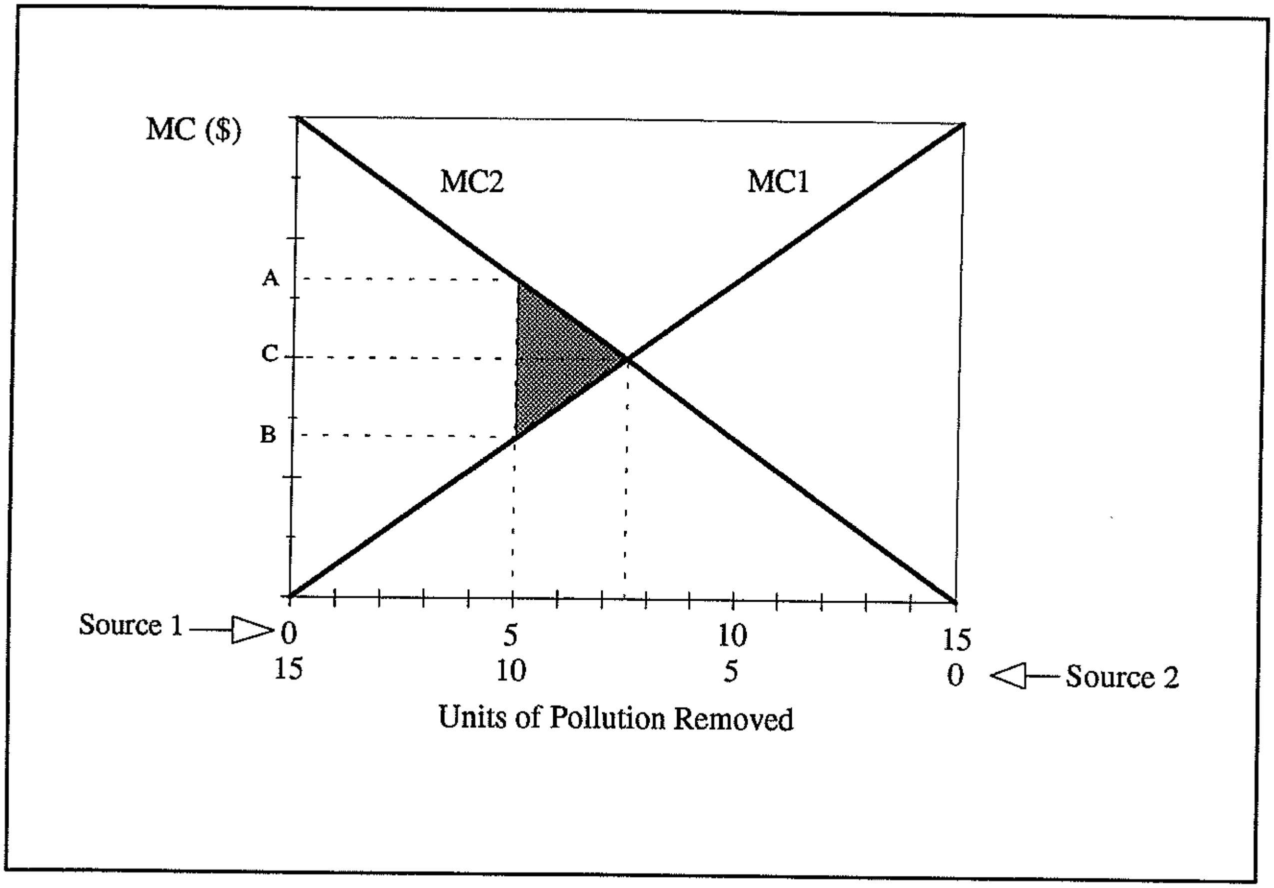

Figure 1. The graph shows the marginal cost of pollution removal for two pollution sources. The cost-effective allocation of a 15-unit pollution reduction is 7.5 units for each source--where the marginal cost of pollution removal is equal.

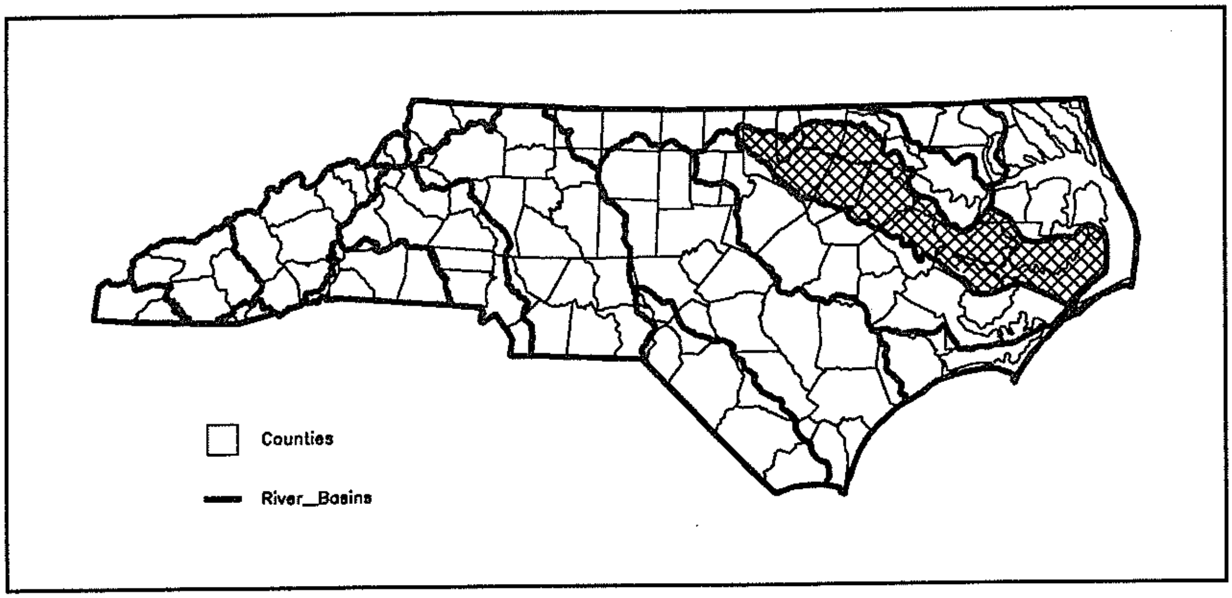

Figure 2. NC is divided into 17 major river basins. The Tar-Pamlico River Basin is shaded.

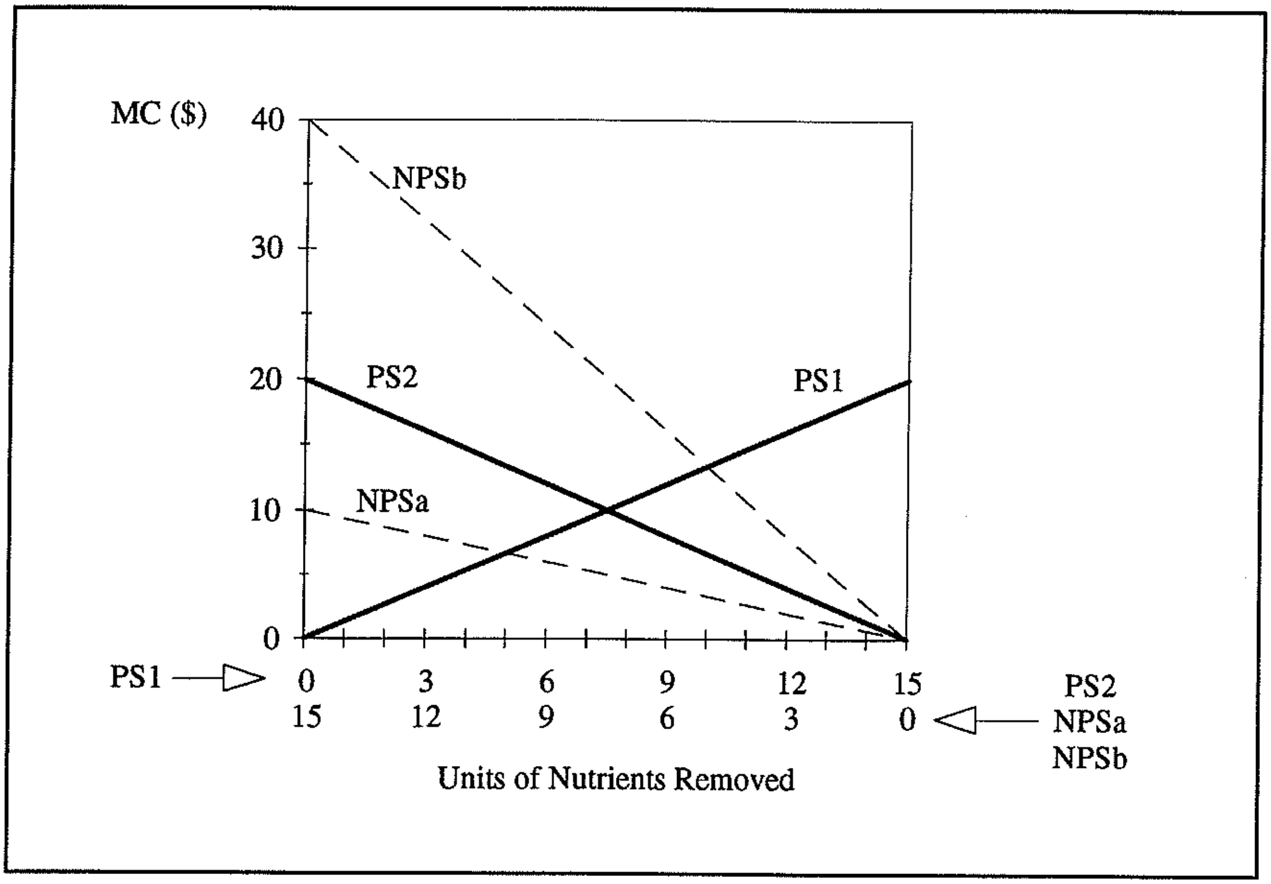

Figure 3. The graph shows the marginal cost of nutrient removal for two point sources (PS1 and PS2) and for two nonpoint source dischargers (NPSa and NPSb).

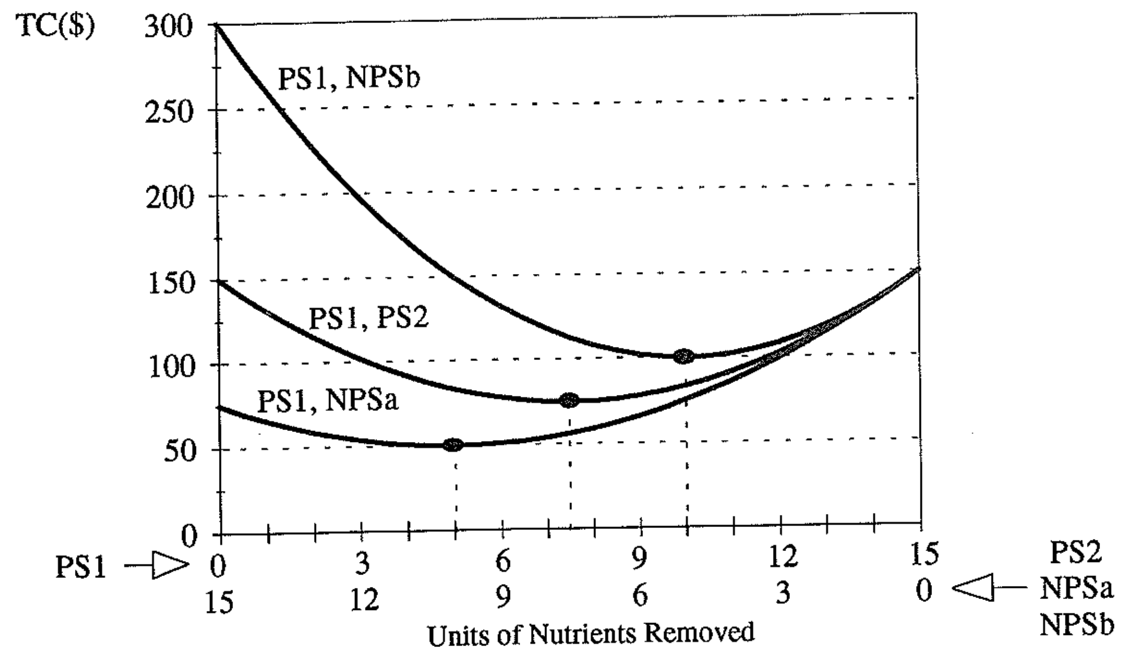

Figure 4. The graph shows the total cost to PS1 of removing different levels of nutrients itself and buying credits from NPSa, PS2, or NPSb to achieve a 15-unit reduction of nutrient discharge.

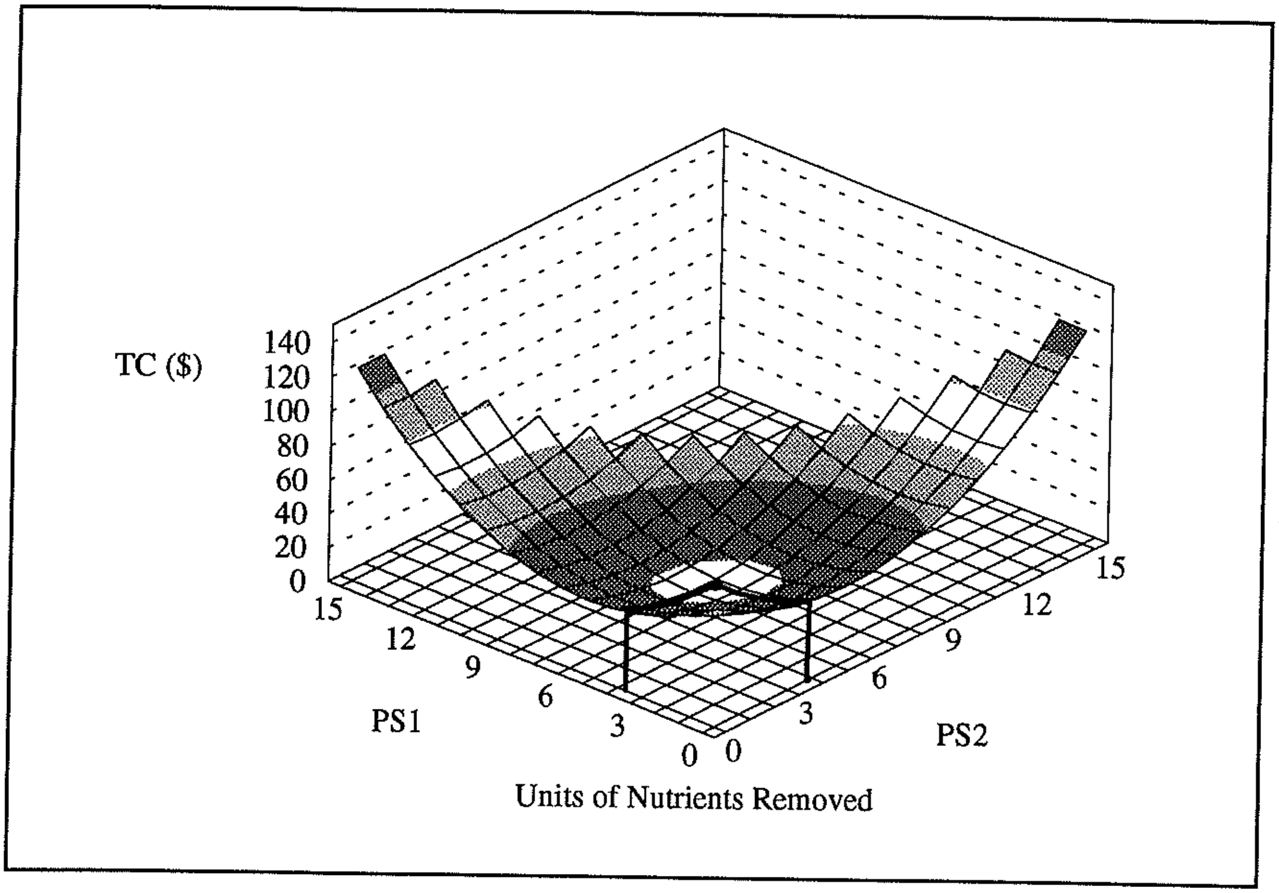

Figure 5. The graph shows the total cost to PS1 of different nutrient removal combinations between PS1, PS2, and NPSa. The x-axis represents the units of nutrients reduced by the point sources with the remaining units cleaned by NPSa. For example, if PS1 and PS2 clean up 3.75 units each, NPSa will clean the remaining 7.5 units and the total cost to PS1 will be $37.5 (as marked on the graph).

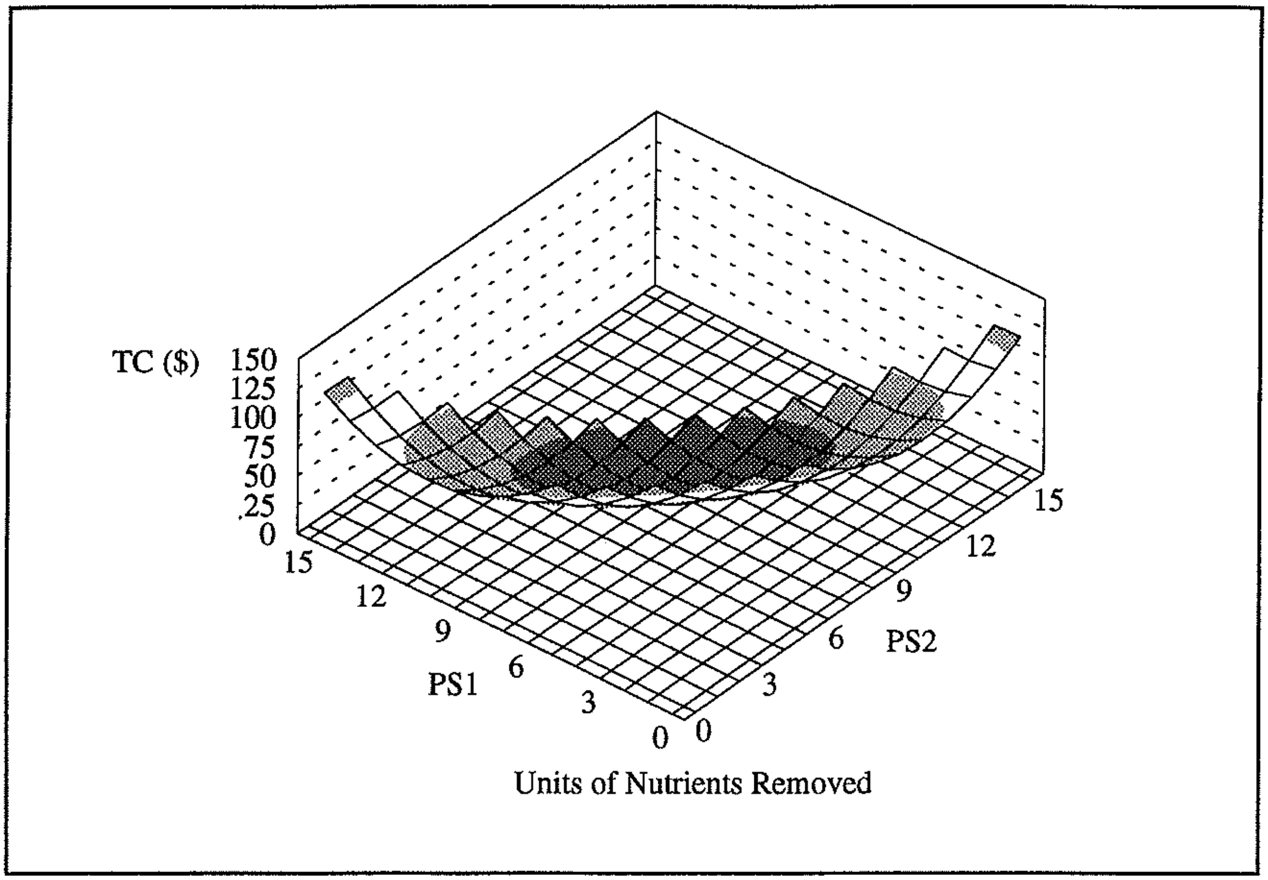

Figure 6. The graph shows the total cost to PS1 of different nutrient removal combinations between PS1, PS2, and NPSb. The x-axis represents the units of nutrients reduced by the point sources with the remaining units cleaned by NPSb.

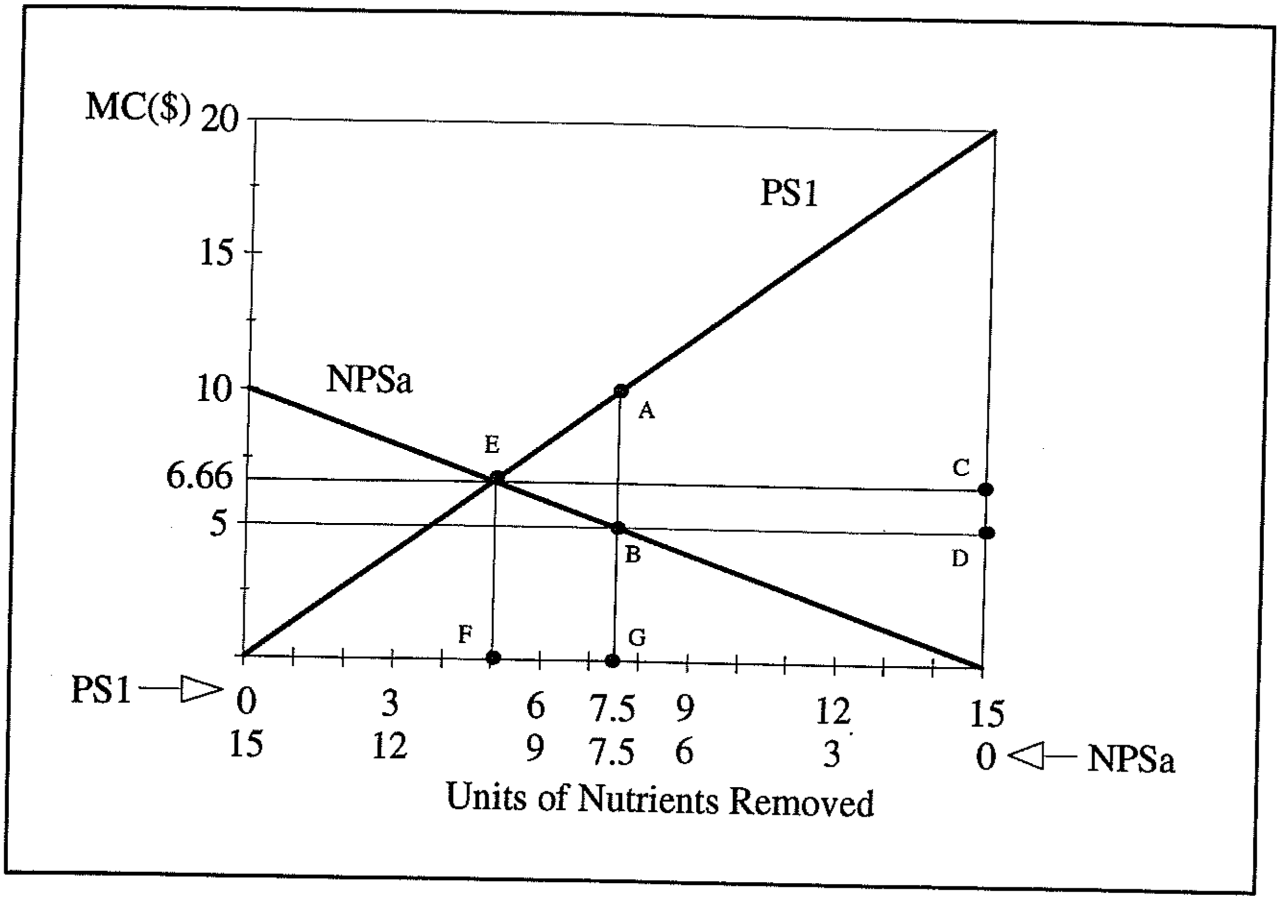

Figure 7. The graph shows that if the price of a nutrient credit is set below the equilibrium level, PS1 will clean up too many units while NPSa will clean up too few (point G) when compared to the cost-effective allocation (point F).

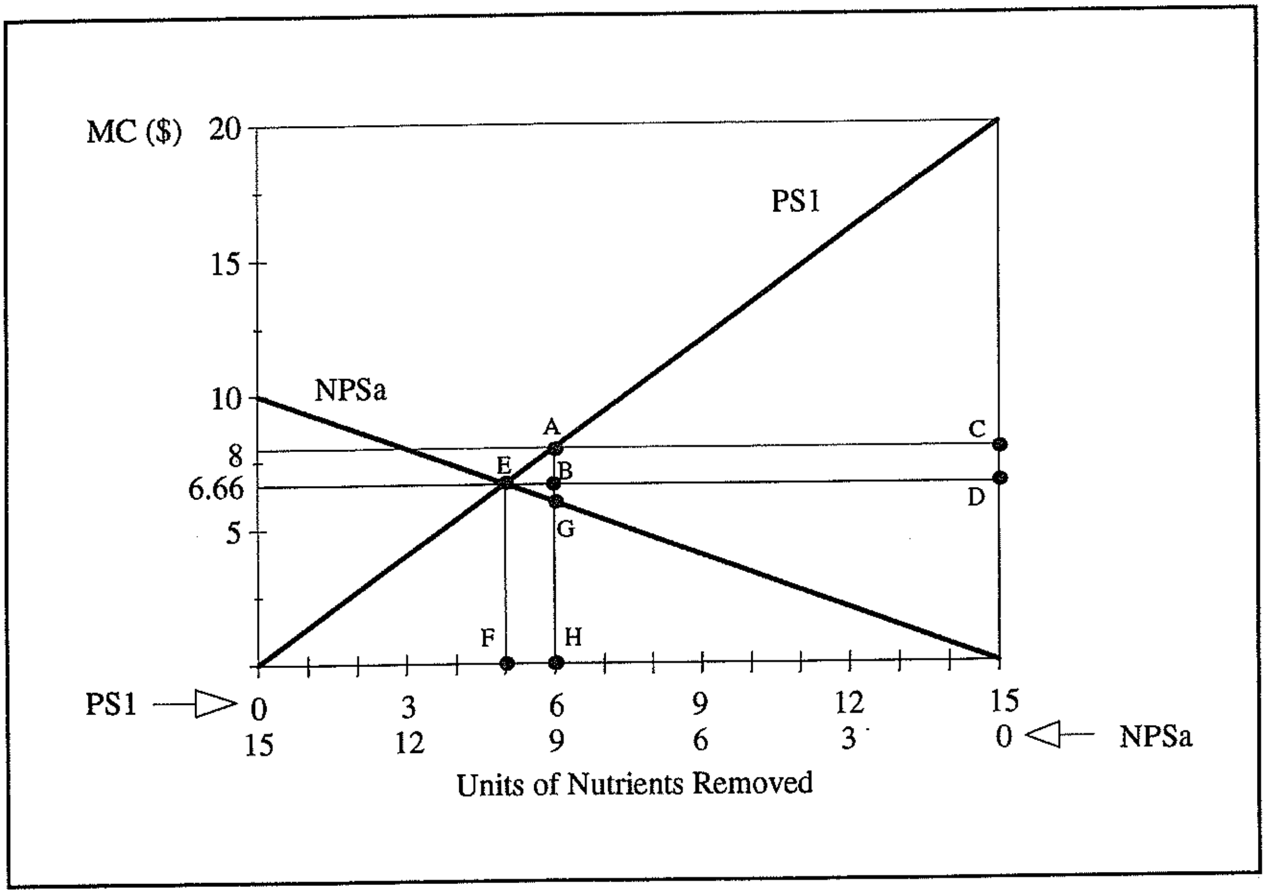

Figure 8. The graph shows that if the price of a nutrient credit is set above the equilibrium level, PS1 will clean up too many units while NPSa will clean up too few (point H) when compared to the cost-effective allocation (point F).

Table 1. Program Development and Implementation Expenditures

Source	Amount ($1,000)	Description
The Association	400	estuarine computer model development
	500	minimum ACSP contribution during Phase I
	150	DSWC for staff positions
	40	engineering evaluations
	40	legal and administrative fees (approximately)
EPA	220	year one development of computerized nutrient management framework (FY91, through CWA Section 104(b)(3) grant)
	120	program activities related to nonpoint source management in basin (FY91, through CWA Section 319)
	500	(FY92, through Section 104(b)(3) grant)
	400	initiation of pollution reduction strategies--most allocated toward agricultural BMP demonstration project (FY92)
	350	funds not yet received by DEM
Total	$2.72 million	

Abbreviations

ACSP: Agricultural Cost Share Program
BMP: best management practice
CWA: Clean Water Act
DEM: Division of Environmental Management
DSWC: Division of Soil and Water Conservation
EPA: Environmental Protection Agency

Table 2. Summary of trading options

Trading Participants	Total Cost($)	Marginal Cost($)	Units of Nutrients Removed By PS1	PS2	NPSa	NPSb
PS1	150	20	15	-	-	-
PS1,NPSb	100	13.33	10	-	-	5
PS1,PS2	75	10	7.5	7.5	-	-
PS1,PS2,NPSb	60	8	6	6	-	3
PS1,NPSa	50	6.66	5	-	10	-
PS1,PS2,NPSa	37.5	5	3.75	3.75	7.5	-

THE VALUE OF INFORMATION FOR TARGETING WATER QUALITY PROTECTION PROGRAMS WITHIN WATERSHEDS

By

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Introduction

Water quality protection efforts could be made more effective by collecting detailed information on each site's potential for water quality damage and targeting strategies at sites with the greatest potential for damage (Nonpoint Source Evaluation Panel, 1990). Interest in targeting began as a way of increasing cost-effectiveness of federal soil erosion control programs (Park and Sawyer, 1985). Targeting involved: 1) a shift from single-field, production-oriented plans to whole-farm, conservation-oriented plans; 2) an effort to identify critical watersheds where erosion control would be most cost effective; and 3) a policy in selected counties of offering higher rates of cost share for practices on fields where greater amounts of erosion reduction are likely to be obtained from the practice (Park and Sawyer, 1985).

In a case study of a highly erodible watershed, Park and Sawyer (1985) found that per-ton costs of erosion reduction were 34 percent lower than the national average implying greater cost effectiveness from targeting more highly erodible regions. They also found that cost effectiveness of erosion control could be greatly increased by offering more cost sharing for sites and practices with higher potential for erosion reduction.

Interest in targeting has expanded to considering the off-site benefits of reduction in agricultural pollution. Ribaudo (1986, 1989) showed that allocation of program funds based on estimates of off-site damages from erosion as well as on-site erosion rates would result in a much different regional allocation of soil erosion control efforts than when only on-site erosion was considered.

The development of computerized Geographic Information Systems (GISs) has facilitated targeting by making it easier to collect, store, and analyze large amounts of data characterizing land and water resources. Data describing the physical characteristics and farming practices of individual tracts of land can be stored within the GIS and used to identify environmentally fragile

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sites with high potential to pollute ground and/or surface water. While better information and targeting will enhance water quality protection, it is unknown if the benefits of acquiring full information exceed the costs.

It may be more cost effective to obtain only partial information on an agricultural area rather than incur the costs of full information. For example, physical information such as soil type, slope, and distance from surface water bodies may be relatively cheap to collect and maintain because these characteristics change very slowly. By contrast, information on farming practices such as type of enterprises, chemical and fertilizer inputs, and tillage practices may be costly to obtain and maintain. These practices change relatively rapidly and the information must be updated frequently.

The focus of this study is on the potential to use information on farm physical characteristics and farming practices in order to increase the cost effectiveness of water quality protection programs. A conceptual model is presented to demonstrate the potential costs and benefits of using either full or partial information on farm physical characteristics and practices in order to target water quality protection efforts. Procedures for estimating the value of information are described next followed by a discussion of information costs and factors affecting the value of collecting information for targeting. Finally a representative farm is analyzed to illustrate the likely benefits of collecting information on farm characteristics for targeting water quality programs.

Conceptual Model

In this study, the effectiveness of information and targeting is evaluated in the context of a performance-based standard, that is, where a limit is placed on allowable losses of one or more pollutants. Performance standards could be enforced by using models to estimate losses of the pollutant from farms with given sets of production practices, soils, and weather conditions (Harrington, Krupnick and Peskin, 1985; Abler and Shortle, 1991). Unlike direct regulations which place limits on types of production practices used, performance standards give farmers flexibility as to how the standard will be met thus reducing the on-farm cost. However, administrative and enforcement costs would be high because of the large amount of information needed to monitor compliance. In addition, the amount of allowable pollution may not be allocated among farms in a way that minimizes the costs of meeting the standard (Abler and Shortle, 1991). However, targeting based on information about farmers' compliance costs may reduce costs by identifying which farmers have the lowest costs of meeting standards.

A simple model of two farms is presented to demonstrate benefits of targeting a performance standard based on farm resource characteristics. Two farms, A and B, each produce a single crop, Y, with one variable input, X, according to the production functions A(X) and B(X), respectively. The farms have fixed and equal amounts of land which differ in productivity and pollution potential. Farm B has less productive land than A as indicated by a lower marginal productivity of X for any level of input use up to some upper bound at which the marginal product of X is zero. The upper bound may differ on each farm. The objective function for farm A is:

$$Max\ \pi_a = A(X)\cdot P - P_x\cdot X_a - FC_a \tag{1}$$

where FC_a represents fixed costs; P_x is the price per unit of X; and X_a denotes X usage on farm A.[2] Assuming it is profitable to use a positive amount of X, the farm maximizes its profit where the following first order condition is satisfied:

$$A'(X)\cdot P - P_x = 0 \tag{2}$$

The marginal value product of the input is set equal to its price. Analogous equations for the objective function and first order necessary conditions for optimization apply to farm B. The optimal level of X may differ between the two farms. Pollution resulting from production does not enter the first order condition because the farms do not bear the cost of pollution.

Now assume X pollution from each farm is regulated by allowing no more than m units of X loss per acre. Each farm is equal in cultivated area and each farm has a total allowable loss of M. The functions $L_a(X)$ and $L_b(X)$ describe X pollution as a function of application for the two farms. Farm profits are optimized subject to the pollution constraint as shown in the following equations:

$$Max\ \pi_{an} = A(X)\cdot P - P_x\cdot X_a - FC_a + \lambda(M - L_a(X_a)) \tag{3}$$

$$Max\ \pi_{bn} = B(X)\cdot P - P_x\cdot X_b - FC_b + \gamma(M - L_b(X_b)) \tag{4}$$

Assuming it is profitable to use a positive amount of X, optimal returns are achieved for farm A where

$$A'(X)\cdot P - P_x - \lambda\cdot L_a'(X_a) = 0 \tag{5}$$

and for farm B where

2 Land area is not specified as a constraint because there is only one production activity occurring on one type of land. The problem can be viewed as determining the optimal amount of X to apply to the total amount of land.

$$B'(X)\cdot P - P_x - \gamma\cdot L_b'(X_b) = 0 \tag{6}$$

π_{an} and π_{bn} represent profits to farms a and b, respectively, when no information about the farm is used to set the performance standard. Each farm maximizes returns by setting the amount of X applied at a level where the marginal value product of one unit of X equals its price plus the shadow price of one unit of X pollution multiplied by the marginal propensity of the application unit of X to pollute. If the focus is on surface water pollution, the value of L(X) is likely to be greatly affected by the location of the farm relative to surface water. Nutrient runoff from fields located far from streams or for which the intervening land provides a significant buffer is likely to be less damaging to surface water quality than runoff from other fields. Thus, if possible, loadings should be expressed as loadings to the edge of stream or to some downstream point rather than to the edge of field.

λ and γ represent the shadow prices of the pollution constraints on farms A and B, respectively. If the sum of net returns from all farms is to be maximized subject to a restriction on the total amount of X pollution, then the allowable X pollution should be allocated among the farms so that the imputed input prices (shadow prices of the IX pollution constraint) are equal (Henderson and Quandt, 1980). Because the production and pollution functions are different on the two farms, it is unlikely that these imputed input prices or shadow prices will be equal when each farm faces the same pollution constraint. Assume that when each farm is at its optimal production level with the pollution constraint, farm A has a higher marginal crop response to X and a lower potential to pollute due to the greater productivity of its land. Then $A'(X_a) > B'(X_b)$ and $L'_a(X_a) < L'_b(X_b)$, which would imply that the shadow price of the constraint is higher on farm A ($\lambda > \gamma$). Total profits on the two farms could be increased by relaxing the constraint on farm A and tightening it on farm B. Cost effectiveness of the performance standard could be increased by targeting it based on each farm's productivity and potential to pollute.

Net Value of Full Information

The net value of full information for targeting a performance standard is equal to the increased farm returns that can be achieved by all farms with the information compared to the returns achieved without information minus the increased costs of gathering the information. With the information, the returns earned by all farms are maximized where the shadow values imputed to the X pollution constraint are equal among farms. In the absence of information, the restriction is met by allowing each farm an equal amount of X pollution on a per-acre basis. If the farms are equal in area, the amount of the constraint, M, is equal for each farm. The net value of full information (V_f) in the example with two farms is:

$$V_f = \pi_{af}(M_{af}) + \pi_{bf}(M_{bf}) - \pi_{an}(M) - \pi_{bn}(M) - C_f \tag{7}$$

V_f may be positive or negative. M_{af} and M_{bf} represent the amounts of X pollution allocated to farms A and B, respectively, such that the shadow prices of the pollution constraints are equal and the total amount of X pollution equals 2M. π_{af} and π_{bf} represent profits to farms A and B, respectively, under full information, i.e., when the performance standard is set so that the shadow price of the standard is equal between the two farms. C_f, the cost of gathering full information, is likely to be high because the information needed to calculate each farm's costs of meeting the standard is extensive.

Net Value of Partial Information

Perhaps collection of partial information is a more cost effective basis for targeting. For example, selected variables such as predominant enterprise on the farm, soil type, or field slope may correlate strongly with the individual farm's marginal cost of meeting the standard. Possibly targeting the standard based on one or more of these individual characteristics will allow most of the benefits of targeting to be achieved at greatly reduced cost of gathering information. The net value of partial information is:

$$V_p = \pi_{ap}(M_{ap}) + \pi_{bp}(M_{bp}) - \pi_{an}(M) - \pi_{bn}(M) - C_p \qquad (8)$$

where π_{ap} and π_{bp} represent profits to each farm when the performance standard for each farm is based on partial information about each farm's characteristics; M_{ap} and M_{bp} represent the standards set for each farm based on partial information; and C_p is the cost of collecting partial information. If $V_p > V_f$, then the optimal strategy is to collect only partial information for use in the GIS.

Estimating the Value of Information

Estimating the Value of Full Information

In order to accurately estimate the value of full information, it is necessary that net returns to all farms be maximized subject to meeting pollution restrictions. An optimal allocation of X pollution for each farm must be determined in order to maximize net returns to all farms as a group. Conceptually the maximization can be done in the framework already presented by finding an allocation of pollution M to each farm that equates the shadow prices of limited X pollution between farms. In reality, the first order conditions shown above are likely to be difficult if not impossible to solve for any realistic farm situation with multiple enterprises and constraints.

A more useful approach is to use mathematical programming to maximize returns to each farm with a specified level of X pollution constraint. Programming results can be used to determine the optimal amount of X pollution to be allocated to each farm. For example, on farm A the nonlinear production and pollution functions could be replaced with alternative linear activities with different levels of X application, X pollution, and net returns above variable cost. Activities would be chosen so that they bracket the range of economically relevant relationships between X and yield. Assume for simplicity that on farm A two such activities are specified, Y_1 and Y_2, where each represents a different level of crop yield, X application, and X pollution.

Then the farm's objective function is specified as:

$$Max\ P_1 \cdot Y_1 + P_2 \cdot Y_2 \tag{9}$$

subject to:

$$l_1 \cdot Y_1 + l_2 \cdot Y_2 \le M \tag{10}$$

$$Y_1 + Y_2 \le Z \tag{11}$$

P_1 and P_2 represent the per acre net returns above variable costs (including X), l_1 and l_2 represent amounts of X pollution per acre of crop production, M is the total X pollution allowed to the farm, and Z is total area for planting the crop.

The program solution generates shadow prices of X pollution for a specified amount of pollution, M. As M is increased, the shadow price will tend to decline. By varying M, it is possible to generate a schedule relating the shadow price of the constraint on X pollution to the amount of pollution allowed. The maximum shadow price is obtained when M is 0. The minimum shadow price, 0, occurs when the constraint on X pollution is no longer binding. If such a schedule were derived for each farm, it would be possible to determine the allocation of X pollution to each farm that maximizes the value of the restricted amount of X pollution allowed to all farms.

For example, assume that for farm A the shadow price of M has a value of λ_1 when up to m_{a1} units of X pollution are allocated. Thereafter, the shadow price declines to λ_2 for up to m_{a2} units of X pollution. Further amounts of X pollution have a value of 0 indicating that the constraint becomes nonbinding. Similarly for farm B the shadow prices are γ_1 and γ_2 for amounts of pollution, m_{b1} and m_{b2}, respectively. Farm net returns for a total allowable pollution of M to be allocated to the two farms, is maximized as follows:

$$Max\ \lambda_1 \cdot a_1 + \lambda_2 \cdot a_2 + \gamma_1 \cdot b_1 + \gamma_2 \cdot b_2 \tag{12}$$

subject to:

$$a_1 \leq m_{a1} \tag{13}$$

$$a_2 \leq m_{a2} \tag{14}$$

$$b_1 \leq m_{b1} \tag{15}$$

$$b_2 \leq m_{b2} \tag{16}$$

$$a_1 + a_2 + b_1 + b_2 \leq \overline{M} \tag{17}$$

The total amount of X pollution to be allocated to farm A can be determined from the solution values of a_1 and a_2 and the allocation to farm B from b_1 and b_2. These values are then inserted as right hand side values for the X pollution constraint for each farm and the farm programming problem is solved again for each farm to determine the net returns to all farms with full information. Net returns to all farms when each is allocated an equal amount of X pollution on a per acre basis are then subtracted from the returns with full information. This difference minus the estimated cost of collecting the full information represents the net value of full information.

Estimating the Value of Partial Information

Selected farm characteristics such as predominant soil type or predominant farm enterprise may be significantly correlated with farm cost of complying with limitations on pollution or, equivalently, with the shadow price of the pollution constraint. These farm characteristics can be obtained with less effort than what is required to calculate each farm's shadow price and might serve as an effective basis for targeting. Statistical analysis will show which variable(s) is (are) most highly correlated with the shadow price. The performance standard could then be imposed on the subset of farms which are most strongly characterized by this (these) variable(s). Returns to all farms and total pollution with this strategy could be compared with returns and pollution with full information as well as with no information.

For example, if soil productivity were positively correlated with the shadow price of the pollution constraint, one could impose the pollution constraint on only farms with the least productive land. The number of farms included should be small enough to make targeting worthwhile and yet large enough to insure that a desired reduction in pollution for the watershed could be obtained. The effectiveness of partial information is evaluated by imposing the pollution constraint on the subset of farms and computing their net returns. Total net returns to all farms in the watershed (including those not affected by the restriction) are estimated with the programming models. Total returns with the restrictions are then compared with returns under no information when each farm faces the same performance standard. The increase in net returns with partial information minus the costs of collecting partial information is the estimated net value of partial information.

Information Collection Costs

Costs of information for targeting consist of identifying farm operators, collecting farm data from the operators, and coding, storing, and analyzing the data. Data collection and analysis might proceed in two phases. Phase one would involve sampling and analysis of a subset of the watershed farms to obtain in-depth farm information. The analysis would be done to estimate the pattern of nutrient and chemical use, factors affecting that use, and potential loadings from such use. The analysis would also seek to determine the variable(s) that might be used to target farms for water pollution control measures. The second phase of the effort would involve sampling the remaining farms in the watershed to determine the distribution of the key target variable(s) and to identify the subset of farms at which to target pollution control efforts.

Both phases of information collection and analysis would have to be repeated periodically because the distribution of agricultural land use as well as types of agricultural technologies in use change over time. The required frequency of such collection efforts depends on how rapidly agriculture within each watershed is changing. It is likely that collection would have to be done more often for full information systems compared to partial information systems. Full information systems rely on more characteristics of the farm meaning there are more variables subject to change compared to partial information.

Factors Affecting Information Value

Prior to deciding whether to obtain information to support targeting of water quality programs, the decisionmaker may wish to contemplate whether the returns from information are likely to justify the expenses of collection. The value of information is likely to increase with the amount of variation in the resource characteristic(s) that affect(s) the relationship between farm production activities and the type of pollution of concern. For example, if the decisionmaker is concerned with reducing delivery of phosphorus (P) to surface water, then the value of information on soil P levels among farms is likely to increase with variation among farms in initial soil P content. The value of information may also increase if farm physical characteristics and practices that increase pollution potential are correlated. For example, if farms with high soil P levels are located near surface water, then the returns from information for targeting these farms would be raised because of the correlation between these two pollution-enhancing factors.

Case Farm Analysis: Restrictions on Phosphorus Runoff

An illustration of the effects of selected farm characteristics on the costs of restricting sediment phosphorus (P) delivery is presented based on dairy farming in the Lower Susquehanna Watershed in Pennsylvania and Maryland. Farming activities in this area are important sources of phosphorus (P) and nitrogen (N) deliveries to streams and ultimately to the Chesapeake Bay. Information on farm characteristics and practices on randomly selected fields during 1989-1991 is available for a sample of over 500 farm operators in the watershed. This information was collected as part of the Area Studies Survey done jointly by the Economic Research Service and the National Agricultural Statistics Service. The sample of farms was generated by randomly selecting sites from the National Resources Inventory (NRI) (Soil Conservation Service) and identifying and interviewing operators of the farms containing the sites. Most of this information was collected for a specific field and includes types of crops planted, crop yields, rates of fertilizer and chemical application, tillage practices, and soil type. Whole farm information obtained included numbers and types of livestock, crop acreages, total farm acreage, and farm sales class.

A Representative Dairy Farm

The interaction between costs of restricting sediment P delivery and farm characteristics was evaluated for a representative dairy farm based on data from the Area Studies Survey. The average distance of the farm from surface water, average slope-length factor for the farm's cropland, and average initial soil P level were varied to determine how these parameters affect the costs of reducing sediment P delivery. Average distance from surface water and average slope of cropland are physical characteristics that are very slow to change barring some major structural development such as terracing. However, soil P level can be readily affected by farming practices. In particular, the number of animals per acre of cropland affects soil P. As animal densities increase, the rate of manure application per acre of cropland generally increases. At higher manure application rates, the P content of applied manure is likely to exceed crop removal. P is relatively immobile with the primary loss occurring with eroded soil;[3] thus higher manure applications imply higher soil P levels.

The representative dairy farm contains 283 acres of which 232 acres can be planted to harvested crops, 37 acres are usable for pasture only, and 14 acres are in other uses (primarily roadways, farmstead, and woodland). The farm has approximately 5,700 hours of operator and family labor available per year. Additional labor can be hired at $6.00 per hour. Potential crops include corn silage, corn grain, alfalfa, orchard grass hay, wheat, oats, soybeans, and pasture. With the exception of pasture and corn silage, all crops can be bought or sold. The farm contains facilities for 83 dairy cattle of which 46 are dairy cows and the remainder heifers and calves. Milk production averages 15,500 pounds per cow, the approximate average for herds of less than 50 cows (Ford, 1992). Cows can be produced on a corn silage ration, an alfalfa hay based ration, an alfalfa haylage ration, or a combination of corn silage and alfalfa hay. The farm has adequate manure storage to allow spreading once per year. Crop and livestock production costs, prices, and livestock rations are taken from Penn State Farm Enterprise Budgets. Crop yields are based on soil type as described in Serotkin. Additional description of the farm model is provided in Bosch (1993).

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[3] Some phosphorus may dissolve and be carried off in surface runoff.

Sediment P delivery

Sediment P delivery is calculated by determining the amount of sediment erosion and the proportion of sediment erosion that is potentially delivered to streams. Then the amount of P in delivered sediment is calculated.

Sediment erosion

Sediment erosion per acre of crop rotation r (SED_r) is calculated using the Universal Soil Loss Equation (USLE) (USDA, 1991; Wischmeier and Smith, 1978). The equation is:

$$SED_r = \sum_c \sum_t RFACTOR \cdot KFACTOR \cdot LSFACTOR \cdot CFACTOR_{rct} \cdot PFACTOR \cdot ROTAC_{rct} \qquad (18)$$

where LSFACTOR accounts for the slope length and steepness, RFACTOR accounts for rainfall runoff rate and amount, KFACTOR accounts for soil erodibility, and CFACTOR represents crop cover for a given crop c in a given rotation r with a given technology t. PFACTOR represents conservation practice which could include contouring, stripcropping, and terracing.

$ROTAC_{rct}$ is the proportion of the rth rotation acre made up by the cth crop and tth technology. Values of RFACTOR, KFACTOR, and LSFACTOR are obtained from the National Resource Inventory (NRI) record corresponding to each of the 515 sample points in the Area Studies data. Values for the CFACTOR and PFACTOR are assigned to specific crop rotation, tillage, and conservation practice combinations based on values suggested in the Pennsylvania Technical Guide (USDA, 1991).

Sediment delivery

Sediment that reaches the nearest stream is usually less than erosion within the field. The amount of delivered sediment that reaches the stream, DELIVRAT, as a proportion of total eroded sediment is calculated as (Shanholtz et al., 1990):

$$DELIVRAT = e^{-WTDCOVER \cdot TOTDIST \cdot SLOPEFN} \qquad (19)$$

e is the base of natural logarithms. WTDCOVER is the weighted average cover factor for the land along the flow path between the site and the nearest stream. WTDCOVER is computed as (Shanholtz and Zhang, 1988):

$$WTDCOVER = [(0.4233 \cdot CROPDIST) + (0.71 \cdot PASTDIST) + (1.1842 \cdot WOODDIST)]/(CROPDIST + PASTDIST + WOODDIST) \qquad (20)$$

CROPDIST, PASTDIST, and WOODDIST represent the distances along the flow path between the site and the stream made up of cropland, pasture, and woodland, respectively.

TOTDIST equals:

$$TOTDIST = CROPDIST + PASTDIST + WOODDIST \qquad (21)$$

If TOTDIST exceeds 1,312 feet, the minimum delivery ratios of 0.184, 0.055, and 0.009, respectively, for cropland, pastureland, and forest land are used (Shanholtz et al., 1990). If TOTDIST exceeds 1,312 feet and more than one type of land use occurs along the flow path, a weighted average of the minimum delivery ratios is calculated based on the proportion of each land use that occurs. SLOPEFN accounts for slope of the flow path between the site and the nearest stream. The equation for SLOPEFN is (Heatwole et al., 1987):

$$SLOPEFN = e^{-16.1 \cdot (SLOPTOSTR + 0.057)} + 0.60 \qquad (22)$$

where SLOPTOSTR is the average slope (decimal fraction) of land along the flow path from the site to the stream and e is the base of natural logarithms.

Sediment P loss

Equation (23) estimates the amount (pounds) of algae-available P (SEDALPHOS) per ton of delivered sediment:

$$SEDALPHOS = 0.002 \cdot ALGEPHOS \cdot PERAT \qquad (23)$$

0.002 is a conversion factor to convert units of mg per kg to lbs per ton. ALGEPHOS represents the mg of algae-available P per kg of soil and is estimated as (Wolf et al., 1985):

$$ALGEPHOS = 71.18 + (3.22 \cdot BRAYMGKG) \qquad (24)$$

where BRAYMGKG represents the Bray soil test estimate of P in mg per kg of soil.

PERAT is a dimensionless P enrichment ratio that varies inversely with clay content in the range between approximately 1.5 and 4.0. PERAT is included to account for the tendency of P to adhere to the finer more erodible clay and silt particles (Tim et al. 1992). PERAT is estimated as (Shanholtz and Zhang, 1988):

$$PERAT = 4.79 \cdot CLAY^{-0.29} \qquad (25)$$

where CLAY represents the average percent clay content of the surface soil layer.

Soil parameters

Table 1 shows the assumed initial values of three soil and location parameters used in the analysis: initial soil P level, length-slope factor, and distance from nearest surface water. The length-slope factors shown are the 50 and 75 percentile values for the 249 sample points corresponding to farms that are classified as dairy. Distances shown are the 25 and 50 percentile values for the 249 sample points. The initial P levels, the estimated amount of P in the top six inches of soil based on the Bray soil test, represent the 25, 50, and 75 percentile values for the 249 sample points where each sample point received the 1990/1991 mean P value reported for the county in which it was located (Wolf, 1993). A Chester soil type was assumed, which is a Group 1 productivity soil (Serotkin, 1993).

Land cover and slope along the flow path from the field to the nearest stream are based on the cover and slopes reported for all 515 sample points in the watershed. The slope used is the median slope for all points, six percent. Cover distribution is: cropland, 76 percent; pasture, 17 percent; and woodland, 7 percent.

Results

A baseline analysis of the farm was done with no constraints on sediment P delivery As shown in table 1, returns above variable costs do not vary with changes in physical characteristics. Soil productivity was assumed to remain the same regardless of slope-length or distance to water. The farm already had adequate P in manure to meet crop requirements; thus varying soil P did not affect commercial fertilizer requirements or returns. Sediment P delivery varied greatly with changes in each of the three physical characteristics. For example, soil P delivery increases by about 60 percent as the initial soil P increased from a low to high value and with slope and distance held constant. Soil P delivery increases by slightly under 60 percent when distance is increased from low to high with initial soil P and slope length held constant. Changes in the slope-length factor have a greater impact, with P deliveries increasing by over 100 percent when the factor increases from low to high with initial P and distance held constant.

When P deliveries were reduced by 20 percent, income above variable costs was reduced by about the same amount for all resource situations. Farms with lower P delivery potential had less of an absolute reduction to achieve; however, their sediment P delivery potential per acre was also less. The net result was that all situations required about the same adjustment of crop acreages and production techniques to achieve the required reduction in P. The adjustment involved shifting about 60 acres out of corn grain and alfalfa into permanent pasture. Cow numbers were not affected.

Shadow prices indicating the cost per additional pound of P restriction varied inversely with the initial sediment P delivery. The high soil P, high slope-length factor, and low distance to water had the highest initial P delivery and the lowest shadow price. The low soil P, low slope-length factor, and high distance to water had the lowest P delivery and highest shadow price. This inverse relationship occurs because sites with high potential to deliver sediment P can achieve larger reductions in P deliveries per acre of land converted from crops to pasture than can sites with lower potential to delivery sediment. Thus, the high potential P delivery sites have lower costs per unit of P reduction compared to the low potential P delivery sites.

The implication for information and GIS is that while physical characteristics (slope and distance from stream) are important, information on farming practices, represented here by the initial P content of the soil, is important also. With slope-length factor and distance to stream constant, shadow prices of sediment P delivery fell by about one-third as the initial soil P level increased from low to high. These results are preliminary; further analysis with different farm types and different soil resource situations is needed to confirm them.

Conclusions

Targeting may be a way to increase the cost effectiveness of water quality protection programs. Advancements in Geographic Information Systems hardware and software offer new potential for efficiently gathering information to support targeting of water quality protection programs. Nonetheless information for GISs may be expensive and time consuming to acquire and maintain; thus careful consideration of how much information to obtain is warranted. Conceptual measures of the values of full and partial information were presented based on the effectiveness of information in reducing the farm level costs of achieving water quality standards. Procedures for estimating the values of full and partial information were discussed.

Results from a case farm analysis were presented to determine the relative importance of distance of the site from a stream, slope within the field, and initial soil P level in determining costs of reducing sediment P deliveries. Results indicated that slope within the field is the most important determinant of the per-unit cost of reducing sediment P deliveries followed by distance from water bodies and initial sediment P concentration which are of equal importance.

Further farm level analysis is being done with data representing a range of farm types and sizes and soil physical characteristics from the Lower Susquehanna Watershed to evaluate the value of information on soil physical characteristics and farm practices in increasing the effectiveness of non-point water quality protection programs. Analysis will be conducted for both P and N runoff as well as N leaching. Research outcomes may have important implications for GIS design to support water quality protection programs.

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Table 1. Effects of sediment P delivery constraints on farm net income for varying field slopes, distances to surface water, and initial soil P levels

Initial soil P (lb/ac)	Slope length factor (LS)	Distance to water (ft)	Baseline		20 percent reduction in sediment P delivery	
			sediment P deliv. (lbs)	returns above variable costs	returns above variable costs	shadow price ($/lb)
102	0.821	480	165	120563	113660	218
102	0.821	850	106	120563	113752	340
102	1.716	480	345	120563	113670	104
102	1.716	850	221	120563	113680	163
110	0.821	480	174	120563	113694	207
110	0.821	850	111	120563	113584	323
110	1.716	480	363	120563	113640	99
110	1.716	850	233	120563	113707	155
199	0.821	480	262	120563	113685	137
199	0.821	850	168	120563	113724	214
199	1.716	480	547	120563	113653	66
199	1.716	850	350	120563	113630	103

A CASE STUDY OF NUTRIENT MANAGEMENT FOR FLORIDA DAIRIES

By

William G. Boggess[1]

Problem Statement

Lake Okeechobee is the second largest freshwater lake contained in the contiguous United States with a surface area of 730 square miles and a drainage area of more than 4600 square miles (SWIM, 1989). Located in south central Florida (Figure 1), the Lake is the direct water supply for five municipalities, provides backup supply for the lower east coast of Florida, and provides ecological, recreational and irrigation benefits to many users.[2] Lake Okeechobee is a shallow (i.e., average depth of 9 feet), highly productive, eutrophic lake which is in danger of becoming hypereutrophic (i.e., excessive nutrient concentrations leading to algae blooms, depletion of dissolved oxygen and fish kills).

The threat posed by nutrient enrichment of the Lake was first documented in a series of limnological studies in the 1970s (Joyner, 1971, Davis and Marshall, 1975, Federico et al., 1981). Federico et al. examined the trophic status of the Lake using a modified Vollenweider model which identified phosphorus as the limiting nutrient. The studies also determined that the Taylor Creek/Nubbin Slough (TC/NS) and Lower Kissimmee River (LKR) drainage basins contributed 30% and 20% of the phosphorus loads and 5% and 31% of the water inflows to Lake Okeechobee. Direct rainfall accounted for 39% of the water and 17% of the phosphorus.

Concurrent with Joyner's study, the Governor called together a Conference on Water Management in South Florida in September, 1971. One of the conclusions of the conference was that the condition of Lake Okeechobee, the heart of both water quantity and quality in south Florida, should be improved (Florida Department of Administration, 1976). The Governor's Conference was followed by a public hearing in 1972 sponsored by the Central and Southern Florida Flood Control District (renamed and rechartered as the South Florida Water Management

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2 In 1985-86 the combined recreational and commercial fishing industries generated $28.4 million in expenditures and sales (Bell, 1987). Bell also estimated that the lake's recreational user value was $8.3 million annually, or concerting this to an asset value, the Lake Okeechobee fishery resource was valued at nearly $100 million. The lake also provides irrigation water for the sugarcane industry which is estimated to provide 18,000 jobs and to generate $1.3 billion annually of economic activity in the state (Mulkey and Clouser, 1988).

District (SFWMD) in 1972)[3]. The results of this hearing, coupled with widespread public and governmental agency concern over the condition of Lake Okeechobee, prompted the Florida Legislature to establish and fund the Special Project to Prevent the Eutrophication of Lake Okeechobee in 1973. The final report published in 1976 identified the primary sources of phosphorus as high density dairy pastures and faulty dairy waste control systems. The report prioritized the TC/NS and LKR basins for implementation of phosphorus management plans.

More recent figures for the entire Lake Okeechobee Watershed confirm that agriculture is the dominant source of phosphorus entering the watershed (Fonyo et al., 1991). The largest sources of net phosphorus imports to the basin are improved dairy and beef cattle pastures (45.9% of the total), followed by sugar mills (14.9%), dairy barns (14.3%), sugarcane fields (13.5%), and truck crops (6.9%) (Table 1). Table 2 summarizes phosphorus imports into the Lake Okeechobee Watershed by material. Fertilizer constitutes 73.2% of the total, and dairy feeds account for 15.9%. Together, fertilizers and feed account for 93.5% of the annual imports of phosphorus and agricultural production is responsible for 98% of the net phosphorus imports to the watershed.

This case study examines what can be learned from Florida's 15 years of experience trying to control phosphorus runoff from dairies into Lake Okeechobee. Specific objectives are to: (1) provide a brief description of the natural system; (2) provide an overview and chronology of phosphorus management/control programs; (3) describe related monitoring programs and analysis; (4) outline the evolution of phosphorus control technologies and incentives for adoption; (5) examine the economic and environmental impacts of the various programs; and (6) derive lessons for similar problems.

Background

One hundred years ago, south Florida fresh water circulated in a slow, rain-driven cycle (40-65 inches per year) of meandering rivers and streams, shallow lakes, and wetlands including unique saw grass marshes. Starting at a chain of lakes south of Orlando, water flowed into the Kissimmee River. The Kissimmee meandered 103 miles south into Lake Okeechobee. During wet seasons, water spilled over the Lake's low southern rim, and flowed south across the everglades saw grass in a 50-mile wide sheet moving at a rate of approximately one hundred feet per day toward Florida Bay.

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3 The 1972 Florida Water Resources Act (Florida Statutes, Chapter 373) assigned the management of water rights to the State and created a system of five water management districts in the state based on hydrologic boundaries. The Act was based on the Model Water Code developed by Maloney et al. (1972). Each district is governed by a board of directors appointed by the Governor and has its own property taxing authority. The districts are charged with managing and protecting water resources. Although the districts have a great deal of autonomy in dealing with water resource issues, they are subject to legislative mandates and to state agencies with ultimate responsibility for water resource issues such as the Florida Department of Environmental Protection.

Table 1. Sources of annual net phosphorus imports to the Lake Okeechobee watershed.

Basin Activity	Annual Net P Import (tons/yr)	% Total Net P (%)
Nonpoint Source Activities		
Improved pasture	2736	45.8
Sugarcane	807	13.5
Truck crops	412	6.9
Other agricultural	106	1.8
Urban	75	1.3
Total Nonpoint Sources	4136	69.3
Point Source Activities		
Dairy	850	14.2
Sugar mills/refineries	907	15.2
Sewage treatment plants	74	1.2
Total Point Sources	1831	30.7
Total All Sources	5967	100.0

Source: Fonyo et al. 1991.

Table 2. Summary of imports of phosphorus-containing materials to the Lake Okeechobee watershed.

Material	P Import (tons/yr)	% of Total P
Fertilizer (P_2O_5)	5379	73.2
Feed supplements-beef	326	4.4
Feed-dairy	1168	15.9
Replacement heifers-dairy	16	0.2
Detergent-dairy	6	0.1
Sugarcane	304	3.1
Food and detergent-human consumption	145	2.0
Total Annual P Import	7344	100.0

Source: Fonyo et al. 1991

Modification of the natural freshwater system in south Florida began in the late 1800s as investors began developing the area. Over the next 100 years, a series of development, drainage, flood protection, and water supply programs resulted in the construction of 1400 miles of canals and levees. The most important project was the massive federally funded, flood-control and water supply project known as The Central and Southern Florida Flood Control Project authorized by Congress in 1948. Major modifications included: (1) the channelization of the Kissimmee river into a 56 mile-long, 300 foot-wide, 60-foot deep canal known as C-38; (2) construction of the 25 foot-high, Herbert Hoover Dike encircling Lake Okeechobee and providing control over most inflows to and all outflows from the Lake; and (3) creation of three water conservation areas south of Lake Okeechobee to store excess flood waters and to provide supplemental water supply. A series of canals, control structures and pumping stations are currently used to control freshwater movement south of Lake Okeechobee.

Agriculture first began to develop around Lake Okeechobee in the 1920s. Originally agriculture was limited by poor drainage and poor soils. Identification of micronutrient deficiencies in the Everglades Agricultural Area (EAA) led to a significant increase in production in the 1930s. Establishment of the sugar program in the 1960s led to a dramatic increase in sugarcane and winter vegetable acreage. During this period water quality problems first began to develop south of the Lake.

Agriculture north of the Lake consists primarily of dairy and beef cow/calf operations with limited acreage of citrus and vegetable production. Dairying, the most important agricultural industry, first began to develop in Okeechobee County in the early 1950s. Originally the south Florida dairy industry had been concentrated around Miami, but urban development after World War II forced them to move. The south Florida dairy industry, centered in Okeechobee County just north of Lake Okeechobee provides fresh milk for the large urban centers along Florida's lower east coast.

As a result of the Central and South Florida Flood Control Project, the major components of the natural drainage system can be controlled somewhat independently. Given this degree of independence and the differential nature of the water quality problems, current concerns over water quality in central and south Florida have manifested themselves as three separate efforts: (1) the Kissimmee River Restoration Project which aims to "restore" the natural meandering flow of the river through oxbows and wetlands (Loftin et al., 1990); (2) the Lake Okeechobee SWIM[4] plan which is designed to control nutrient loads in order to protect the Lake's vital water supply, recreational, and ecological benefits; and (3) the Everglades SWIM plan designed to address concerns about the quantity, temporal distribution and quality of water released from the Everglades Agricultural Area (EAA) south through the Water Conservation Areas (WCAs) into the Everglades National Park (SFWMD, 1990). This case study focuses on efforts aimed primarily at controlling nutrient loads to Lake Okeechobee which have culminated in the Lake

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[4] In 1987 the legislature passed the Surface Water Improvement and Management (SWIM) Act (Florida Statutes Chapter 373.451 - 373.4595). The Act dictates that the five water management districts in Florida design and implement SWIM plans for priority water bodies. The act also established a trust fund to provide financial support through FDER.

Okeechobee SWIM Plan.

Phosphorus Management/Control Programs

Based on the 1976 recommendations of the Special Project to Prevent the Eutrophication of Lake Okeechobee, initial nutrient control efforts focused on reducing phosphorus runoff from dairies in the TC/NS basins (Albers et al., 1991). The first program was a state funded project called the Taylor Creek Headwaters Program (TCHP) which began in 1978 with the objective of fencing cows from waterways and determining the impact on stream water quality. The project was limited in scope, confined to the headwaters of Taylor Creek.

In 1981, federal funds were obtained under the Rural Clean Waters Program (RCWP) to address water quality concerns in the entire TC/NS basin. The goal of the TC/NS RCWP was to reduce phosphorus concentrations in water flowing into Lake Okeechobee from the basin by 50% by 1992 (NWQEP, 1989). The objectives were to implement BMPs (e.g., fencing waterways, shade structures, filter strips) and evaluate the impact on basin water quality. The state-funded TCHP project was combined with the RCWP program to provide additional funds for the implementation of BMPs. The SFWMD was given the responsibility of monitoring water quality to determine the efficacy of BMPs for phosphorus reduction beginning in 1978 and continuing to date (Flaig and Ritter, 1989).

The Lower Kissimmee River RCWP was initiated in 1987 to reduce agricultural nonpoint source pollution in the LKR basin. The objective was to implement BMPs for each dairy to reduce loadings from animal waste and fertilizer. The goal was to reduce phosphorus loads from the Kissimmee River to the Lake by 43%.

In August, 1985, the Governor directed the secretary of FDER to direct a study of the conditions affecting Lake Okeechobee and to make recommendations for its protection and improvement. FDER formed the Lake Okeechobee Technical Advisory Committee (LOTAC I), which concluded that the phosphorus concentrations in the lake doubled between 1973 and 1984 and that the lake was losing its ability to assimilate phosphorus (LOTAC, 1986). LOTAC I produced a number of recommendations, including that detailed agricultural BMPs should be planned and implemented in the TC/NS and LKR basins, that would prohibit discharge of barn wash water and retain the runoff from high cow density areas for the 25-year, 24-hour storm. LOTAC I also recommended that a set of research and demonstration projects totaling approximately $8 million be conducted to examine fertilization practices, dairy ration formulation, chemical and biological treatment of barn wash water, and basic biogeochemical behavior of phosphorus in soil and water.

In August 1986 the Governor issued executive order 86-150 directing the secretary of FDER to implement the recommendations of LOTAC I with regulations to be in place by May 1987. The Florida Department of Agriculture and Consumer Services (DACS) was directed to complete a cost share program patterned after the TCHP by October 1986. FDER, working with SFWMD, SCS, and dairy representatives drafted the "Dairy Rule" (F.A.C. 17-6.330 through 17-6.337) which became effective June 1987. The rule specified that the dairies in the

TC/NS and LKR basins had to implement specified technologies to prevent the discharge of barn wash water and to retain the runoff from high intensity areas for the 25-year, 24-hour storm. A total of 49 dairies (approximately 45,000 cows) came under the jurisdiction of the Dairy Rule. DACS secured funds from the legislature to cost share the construction.

The dairy industry requested, and was granted, a dairy ceasing operations program for dairies that chose not to comply with the dairy rule. Dairymen were offered a payment of $602 per cow (approximately half of the money was provided by SFWMD and half by the State) in return for a deed restriction prohibiting the property from being used for a dairy or any other concentrated animal feeding operation[5]. The dairymen retained ownership of the cows and the property. A total of 18 dairies signed ceasing agreements which eliminated 14,039 cows from the basin.

The 1987 SWIM Act directed the SFWMD to protect the water quality of Lake Okeechobee (Chapter 373.451 - 373.4595, Florida Statutes). Based on limological studies, the SFWMD determined that the long term annual phosphorus load needed to be reduced to 397 tons in order to meet the SWIM Act's water quality goal. The SFWMD was required to develop a plan to meet this reduction by July 1992. The SFWMD developed an interim plan (SFWMD, 1989) consisting of research and regulatory initiatives. The regulatory component of the SWIM plan is to be accomplished primarily by the implementation of phosphorus performance standards. A performance standard of 0.18 mg per liter average annual, total phosphorus concentration was adopted for inflows to the lake. The standard was calculated by dividing the 397 ton target loading by the long-term water inflow to the lake. The 0.18 performance standard is applied to tributary discharges but not to runoff from individual properties. For dairies, the allowable discharge concentration for total phosphorus was set at 1.2 mg per liter based on calculations that the assimilative capacity of streams and wetlands would result in the 0.18 standard being met at the lake inflow structure. For improved pasture land uses, which include dairy heifer and beef cow-calf operations, the standard is 0.35 mg per liter. Other land uses are required to remain at their historical levels, with the exception that land uses currently below the 0.18 standard are permitted to come up to the standard.

Monitoring Program

A monitoring program was developed in 1989 to support the regulatory aspects of the "Works of the District" rule formulated under the Interim Lake Okeechobee SWIM plan. The objectives of the program are to evaluate the efficacy of BMPs, to provide background

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[5] The restriction includes the following specific language: "The property described herein is hereby and perpetually restricted to the extent it is prohibited from being used for a commercial dairy as defined in Rule 5D-1.01(49), FAC, or concentrated animal feeding operation as defined in Rule 17-6.330(1C), FAC. Landowner must obtain appropriate permits and show evidence of an approved conservation plan developed to USDA Soil Conservation Service standards with proper nutrient balance if agricultural operations are to be carried out on this property.

information for a surveillance monitoring program, and to provide on-going checks on compliance with the runoff concentration standards. Under the "Works of the District" rule, a total phosphorus concentration standard was selected over a phosphorus load standard due to ease of implementation and greater correlation to changing land use management (Flaig and Ritter, 1989). A load standard requires precise field measurements to calculate discharge at each site which is problematic for two reasons. First, accurate flow measurements are difficult to obtain for streams with a low gradient, poor access, and poor stream measurement sections. Second, nutrient loads from storm runoff are sensitive to hydrologic variation. This makes calculations of long term monthly or annual phosphorus loads dependent upon rainfall patterns and seasonal influences and complicates enforcement.

The concentration standard has been converted into a regulatory criteria to provide a workable, attainable standard requiring minimal data collection (Flaig and Ritter, 1989). The components of the off-site performance standard are: (1) a total phosphorus concentration standard not to be exceeded on an average annual basis; and (2) a maximum total phosphorus concentration not to be exceeded when fewer than six samples have been collected. These values are based on the 50% probability that the annual off-site phosphorus concentration limitation will be exceeded. The first criteria defines an average annual standard to evaluate long term behavior. The second criteria provides a means to identify a serious problem with a limited record of water quality samples. These criteria have been formulated into an administrative rule for permitting and enforcement (Rule 40E-61, F.A.C.).

Monitoring activities in the TC/NS and LKR basins consist of surface water sampling, rainfall measurement, stream stage and ground water stage measurement (Flaig and Ritter, 1988). Surface water grab samples are collected bi-weekly at all dairies and tributaries in both TC/NS and LKR. Samples are analyzed for total phosphorus. Similar samples are collected and analyzed for quality assurance and quality control. In addition, the dairies are required under the Dairy Rule to sample phosphorus concentrations in groundwater on a quarterly basis.

The costs of the monitoring program are a major concern in the implementation of the program. Water sample collection and analysis for total phosphorus range from $50 to $95 per sample. The cost increases where sample sites are difficult to reach, which is common with dairy discharge locations. Assuming two discharge locations, bi-weekly sampling, and a cost of $50 per sample, monitoring costs would exceed $2500 per year per dairy. The SFWMD is responsible for monitoring surface water discharges. The dairies are required by FDER to self-monitor ground water quality on a quarterly basis.

Phosphorus Management Technologies and Incentives

To be technically effective, phosphorus control practices have to physically change phosphorus flows through the production system. Phosphorus flows can be affected in four general zones in a production system: (1) phosphorus material imports (source reduction); (2) onsite treatment/storage; (3) phosphorus product exports (export enhancement); and (4) offsite treatment/storage. Control practices that operate in zones (1) and (3) may be classified as phosphorus use management practices, whereas those operating in zones (2) and (4) are phosphorus waste management practices. Practices operating in all four zones have been proposed and studied as options for controlling phosphorus runoff into Lake Okeechobee. To date, however, no offsite treatment/storage technologies have been implemented.

Dairies in the Lake Okeechobee basin are currently implementing the fourth generation of phosphorus management BMPs. The various phases of BMP implementation tended to overlap, making it difficult to quantify precisely the efficacy of the various stages of BMP implementation. A brief, chronological discussion of the four generations of technologies and incentives follows.

In the early 1970s the State and SCS encouraged the development of lagoon systems to capture milking barn wash water and to direct the effluent into seepage fields. The second generation of BMPs was associated with the TCHP program and consisted of pasture improvement and waterway protection to eliminate the direct loading of wastes (i.e., onsite storage). The TCHP program, initiated in 1978, was a small scale trial program limited to the headwaters of Taylor Creek which accounted for only 1% of the water, but 12% of the phosphorus entering Lake Okeechobee via S191 (Albers et al., 1991). The program was voluntary, with the State providing 100% cost sharing.

The TC/NS RCWP program was approved and funded in 1981. The primary goal of the TC/NS RCWP was to extend the scope of the TCHP by contracting with all twenty-four dairies in the drainage basin to implement pasture and waste management BMPs to reduce nutrient runoff. Beef cow/calf farms that had been extensively drained and lands within a quarter mile of waterways were also targeted. Specific BMPs implemented included fencing cattle from waterways, establishing vegetative filter strips along waterways, providing cattle crossings over streams and ditches, providing shade structures for cattle away from streams and waterways, and recycling barn wash water (Stanley et al., 1988). The program was voluntary (though backed by an implicit threat of regulation), with 75% federal cost sharing. The TCHP program was combined with the TC/NS RCWP in 1981 and the state funds were used to leverage the federal cost sharing.

The LKR RCWP began in 1987. Originally it was envisioned as an extension of the TC/NS RCWP with the primary focus being to improve pasture and nutrient management on dairy and beef cow/calf farms via voluntary participation with federal cost sharing. But in 1987 the State passed both the Dairy Rule and the SWIM Act which mandated implementation of technology standards by 1991 and performance standards by 1992. Faced with these new regulations, dairymen shifted their focus from low cost, pasture and nutrient management BMPs (second generation) towards more mechanical capture and removal methods (third generation) that would satisfy the technology standard specified in the dairy rule (RCWP, 1990). Thus, the incentive structure under the LKR RCWP has evolved from voluntary, with federal cost sharing, into a mandatory, technology based standard, with primarily state cost sharing.

The dairy rule represents the third generation of BMPs. Passed in June 1987, the dairy rule specified that all existing dairies were required to submit construction permit applications along with BMP designs by June 1989. (New dairies are subject to the SWIM runoff standards and are not eligible for state cost share.) Within 18 months of construction permit issuance, the

BMP construction must be completed and an operating permit obtained from FDER. In order to satisfy the technology standard, the dairy rule designs were required to: (1) collect all wastewater and runoff from barns and high intensity areas for a 25-year, 24-hour storm; (2) dispose of nutrients by approved methods, particularly land application by irrigation; (3) fence cattle from waterways; and (4) monitor water quality discharges to insure system adequacy. The dairy rule technologies formalize the earlier focus on onsite storage enhancement and expand the focus to include nutrient recycling and source reduction. In addition, onsite treatment systems (chemical and biological) have been implemented by a couple of dairies.

Typical dairy rule designs called for constructing perimeter ditches around the barns and high intensity areas[6] to collect all of the runoff from a 25-year, 24-hour storm. The runoff is processed through a two-stage lagoon system and then applied, via center pivot irrigation systems, to forage production sprayfields. The sprayfields are sized to insure that annual application rates of phosphorus do not exceed the uptake of the forage crop, generally 60 pounds per acre per year. In addition, the dairies must have sufficient land available for land spreading of solids collected.

The State initially planned to provide 75% cost sharing of construction costs under the dairy rule. But escalating construction costs from an initial cost share estimate of $250,000 to over $1,000,000 per dairy, resulted in a revised sliding scale for cost sharing. DACS provided cost sharing ranging from $233 to $433 per cow depending upon the size of the dairy, with the smaller dairies receiving the higher rate (Conner, 1989). This sliding scale reflects the significant construction cost economies of scale enjoyed by large dairies (i.e., 1500 cows) relative to small dairies (i.e. 350 cows) (Giesy, 1987). The net result is that cost sharing under the dairy rule averaged 67% of the estimated construction costs.

The companion dairy ceasing operations program provided an alternative economic incentive-based option to the technology based standard. A fixed payment of $602 per cow was offered based on political "fairness" or equity concerns.[7] There was no "market" or competitive bidding for the easements. One would hypothesize that smaller, less efficient dairies with particular location or drainage problems would be more apt to accept the easement option. Eighteen of the dairies opted for the easement. Sixteen of the dairies that chose to close were relatively small with an average herd size of just under 700 cows versus an 1050 cow average for the thirty dairies that chose to comply. In addition, one operator decided to close two large dairies (2900 cows) because he had inadequate land available for spreading of wastes.

Dairy operators were allowed to sign contracts for both the easement program and the dairy rule, knowing they would eventually have to choose between the two options. This

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6 High intensity areas are defined as areas of concentrated animal density generally associated with milking barns, feedlots, holding pens, travel lanes and contiguous milk herd pasture where the permanent vegetative cover is equal to or less than 80 percent.

7 The $602 figure reflects an approximate 75% cost share of an estimated cost of $800 per cow to move a dairy 500 miles.

practice allowed dairy operators to obtain knowledge of the specific phosphorus concentrations in runoff from their lands and an estimate of the cost of compliance prior to the final easement deadline. However, this practice led to considerable uncertainty and higher administrative costs for the FDACS, ASCS, SCS, and SFWMD.

Under the interim Lake Okeechobee SWIM plan, phosphorus concentration performance standards have been specified for the dairies and other land uses in the basin. Enforcement of the performance standards is just being initiated. Dairy operators in anticipation of enforcement of the runoff standards have made numerous modifications and management adjustments to reduce runoff concentrations. These include changing phosphorus content of fertilizer and feed, relocating cows, relocating incentives inside the high intensity area, moving fences, plugging, cleaning or reshaping ditches, etc. Several dairy operators chose to build confinement or semi-confinement dairy systems which, though more expensive, provide greater control over animal wastes. Two dairies have also implemented chemical and biological treatment systems.

The evolution of phosphorus control practices reflects a trend toward increasing collection and treatment of dairy wastes. The percentage of the dairy wastes being collected steadily increases from approximately 25% under the first generation of BMPs, to 65% under low-tech dairy rule designs, to 85% under the high-tech dairy rule designs, to essentially 100% under total confinement. In addition, the level and type of treatment of the wastes also increases from first generation simple lagoon/drain field to two-stage lagoon with controlled land application to, in a couple of cases, chemical or biological treatment. The net effect has been a steady conversion of a primarily nonpoint source to a point source.

Uncertainty due to lack of information about the extent and mechanics of the phosphorus runoff problem and about the efficacy of alternative control technologies led to a cautious, evolutionary application of control technologies. The evolution of incentives for participation reflected the same uncertainties. Economic and equity concerns dominated early programs, whereas efficacy and the certainty of effect have dominated more recent programs. As a result, incentives have slowly evolved from purely voluntary with 100% cost sharing, through voluntary with steadily decreasing cost sharing, to regulatory technology based standards with partial cost sharing, to performance based standards backed by substantial fines for noncompliance.

The specification of performance standards recognizes the advantages of allowing individual flexibility in responding to environmental regulations as well as the need to provide incentives for operators to manage their systems carefully. Management tends to be lax under pure technology standards once the technology has been installed. In this particular case, the government allowed dairy operators considerable latitude to design systems that not only met the requirements of the technology standard, but that took into consideration the unique aspects of each particular dairy.

With the exception of the dairy ceasing operation agreement and cost sharing, economic incentives have not been employed. In the early stages of the problem, concerns over equity and in finding a "fair" solution limited the use of economic incentives to cost sharing. In the later stages, economic incentives were generally considered to be too uncertain in their effectiveness and strict technology and performance standards were imposed instead.

Three additional types of economic incentives would appear to be feasible options. One would be to convert the dairy buyout or easement program from a fixed amount to a market or bid system that would reflect the differential costs of compliance and values of the dairying property right across dairies of different sizes, locations, and management capabilities. This approach would combine the desirable efficiency aspects of economic incentives with the high certainty of efficacy sought by environmentalists.

Secondly, since over 90% of the phosphorus entering the basin is accounted for in fertilizers and feeds (Table 2), an input tax would be relatively easy to implement and administer. But an input tax provides only indirect incentives to control emissions and thus as a sole approach would probably be an inefficient means of achieving the rather stringent water quality goals dictated in the SWIM plan. The tax does provide a relatively easy program to implement and administer, and it would provide a source of funds for companion cost sharing or abatement programs.

An emissions tax would provide more direct incentives for dairymen to control runoff. However, runoff loads are difficult to quantify and thus concentration standards and monitoring protocols have been developed for implementing the performance standards. Emissions taxes or tradeable emission permits could conceptually be based on the same concentration measurements (Segerson, 1988).

Summary of Costs and Impacts

Formal cost effectiveness calculations for the various programs or for the implementation of specific BMPs are complicated by several factors. First, the various programs and expenditures have been intertwined making it difficult to separate overall expenditures by program. Second, it is difficult to quantify the impact of specific changes in land use due to lags in effects, variations in rainfall, and overlapping practices. Third, it will take another year or two before the impacts can be measured. But it is possible to trace out the history, source and magnitude of expenditures to date and to examine overall changes in water quality trends.

Program Costs

An estimated $36 million has been spent over the past ten years on programs to control phosphorus runoff from agricultural lands north of Lake Okeechobee (Table 3). Various government sources provided approximately three-quarters of the total with farmers providing the balance. Expenditures for research, permitting, monitoring and enforcement are excluded from the government total. Likewise expenditures for roofed structures and for operation and maintenance of the BMPs are excluded from the farmers' total. The State provided $16.5 million (61%) of the $27 million government total; the SFWMD provided $7.7 million (28%); and the federal government provided $3.14 million (11%). The federal government, however, provided 82% of the government funding for the RCWP.

Table 3. Costs of programs for controlling phosphorus runoff from agricultural lands north of Lake Okeechobee.

	Source of Funds ($M)					
Programs	Federal	State	SFWMD	Total Government	Farmer	Total All Sources
RCWP No. 14	3.14	0.31	0.40	3.85	0.45	4.30[a]
Dairy Rule	----	11.89[b]	3.18[c]	15.07	5.76[d]	20.83
Dairy Buyout	----	4.31	4.14	8.45	2.78[e]	11.23
Total	3.14	16.51	7.72	27.38	8.99	36.37

Sources: Rural Clean Water Project No. 14, Annual Progress Reports 1988 (Stanley et al., 1988), 1989 (Conway et al., 1989) and 1990. Florida Department of Agriculture and Consumer Services.

[a] 2,567,598 ($2,132,321 government cost share and $435,277 farmer cost share) can be apportioned to the TC/NS RCWP prior to the Dairy Rule. The remaining $1,734,483 can be apportioned to the LKR RCWP - which has been implemented in conjunction with the Dairy Rule. (Figures are based on 1988, 1989 and 1990 RCWP No. 14 annual progress reports).

[b] Includes $2,259,881 that was administered through the RCWP.

[c] Does not include research or monitoring costs.

[d] Based on estimated total construction costs for eligible items. Cost of ineligible items such as roofed structures are excluded, as are operation and maintenance costs. Includes $553,002 of farmer cost share under the RCWP.

[e] Estimate based on 14,039 cows at $198 per cow (i.e., $800-602).

The breakdown of expenditures between the RCWP and the Dairy Rule are rather arbitrary since the two programs overlapped beginning in 1987. A total of $2.13 million was spent by the government and $435,277 by farmers under the TC/NS RCWP prior to the DairyRule. An additional $4.55 million has been expended under the auspices of the TC/NS-LKR RCWP since 1986. Much of this was spent in the LKR on practices required by the Dairy Rule which go far beyond the original RCWP goals of pasture and nutrient management. This shift in emphasis is reflected in the difference in the average cost of BMPs installed in the two basins. In TC/NS, 27,897 acres were served by BMPs at a total cost of $1.72 million or $61 per acre. In the LKR, 6,926 acres were served by BMPs at a total cost of $3.16 million or $456 per acre (RCWP, 1990).

The dairy rule and dairy ceasing operations programs have been funded without federal support. The state government provided the majority of the funding, although the SFWMD provided nearly half (49%) of the dairy ceasing operations expenditures (Table 3). Construction costs for the Dairy Rule plans range from $418 to $1086 per cow with an average cost of $659 per cow. Two dairies elected to construct total confinement barns at an approximate cost of $1200 per cow.

Economic Impacts of the Dairy Rule and the Ceasing Operations Programs

Based on a 1991 survey of the dairies that choose to comply with the Dairy Rule, along with detailed financial simulations, Boggess et al. (1991) estimated that complying with the Dairy Rule will cost dairy operators an average of $1179 per cow or $1.2 million per dairy, net of cost share received. Total expenditures on construction averaged $923 per cow, $691 for components mandated by the Dairy Rule and $232 for optional components. Cost share received averaged $355 per cow. The dairy operators' costs break down as follows: net out-of-pocket investments averaging $596,000 ($568 per cow), revenue losses during construction averaging $369,000 ($352 per cow), and projected present value of net income losses over the first five years of operation averaging $272,000 ($259 per cow). Amortized over the economic life of the system, these costs indicate that complying with the dairy rule amounts to a production cost handicap of $1.10 per hundredweight of milk.

In a companion study, Mulkey and Clouser (1992) estimated the economic impact on the regional economy associated with the closing of the dairies under the dairy ceasing operations program. They estimated that closing the dairies reduced milk production by over 200 million pounds resulting in reduced milk sales in the range of $30 - $34 million. As a result, annual losses to the regional economy include between $47.6 and $54.3 million in sales, between 465 and 531 full-time jobs, and between $9.0 and $10.2 million in earnings.

Impacts - Monitoring Data Analysis

The ecological health of Lake Okeechobee has been related to the total phosphorus (TP) concentration in the pelagic zone of the Lake (Federico, et al. 1981). Where the concentration is below 50 mg/m^3 the Lake is considered healthy. Since the early 1970s the concentration has been rising. In recent years the concentration has fluctuated dramatically from year to year. The in-lake phosphorus concentration shows little correlation with phosphorus loading. The poor correlation is due in part to fluctuating Lake stage and resuspension of bottom sediments which is common in shallow lakes. Although the long term health of the Lake is linked to the load, there is little year-to-year correlation between load and in-lake concentration.

There is also no clear pattern in the time series of annual loads for the tributaries TC/NS (S191) and LKR (S65E, S154)[8]. The calculated loads at the basin scale are extremely sensitive

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[8] T tests were performed comparing the mean loadings during the period 1973-1979 with mean loadings during the period 1980-1989. The mean loadings from TC/NS were 43 tons lower during the 80s (113.5 tons) than during the 70s (156.9 tons). But the

to runoff volume. In particular, storm events following long antecedent dry periods tend to produce large TP flushes. In this region where tropical storms and long dry seasons are typical, there is rarely an average year. Consequently it is difficult to relate changes in phosphorus load to changes in land management. Experience has shown that the TP concentration in runoff is a function of cow density and proximity to open water; runoff concentrations from lagoons range from 20 to 40 mg/l, while runoff from intensive pastures range from 2 to 5 mg/l and unimproved pasture runs less than 1 mg/l.

The long term trend[9] in total phosphorus concentration in tributary runoff is a useful metric for evaluating land use change. Analysis of time series of TP concentration for runoff from TC/NS (S191) for the period 1973-1991 identifies three distinct periods in the data. During the mid-1970s, cow numbers were increasing and water quality was steadily decreasing. This period corresponded with the Special Project Report in 1976 documenting that phosphorus was the limiting factor in the Lake and identifying the dairies as the primary source. During the late 1970s and early 1980s the "dairy phosphorus problem" began to receive a lot of attention resulting in the TCHP in 1978 and the TC/NS RCWP in 1981. Trend runoff concentrations of TP at S191 were essentially unchanged during this period. Under the TCHP and RCWP, BMPs began to be implemented in the TC/NS basin beginning in 1980 and the result has been a significant downward trend during the 1980s.

Median TP concentrations in runoff from TC/NS peaked at approximately 1.1 mg/l around 1980, since then they have declined by about 50% to between 0.5 and 0.6 mg/l. A similar decline in absolute terms is needed to reach the 0.18 mg/l standard that has been established by the SFWMD. Most of the decline to date can be attributed to second generation BMPs installed under the TC/NS RCWP. It is too early to assess whether the combined effects of the Dairy Rule and the ceasing operations agreement will be sufficient to reach the target concentration at S191.

Overall, the results from the monitoring program indicate that the BMPs have improved water quality, particularly in the TC/NS basin. It is clear that water quality can be improved by practices that enhance soil storage, reduce P imports, and reduce availability of P to surface water discharge. Runoff concentrations at many of the tributaries however, still exceed the 0.18 standard. More time is needed before the impact of the Dairy Rule and dairy ceasing operations programs can be completely assessed.

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coefficient of variation was 0.44 and the difference was significant at only a 15% confidence level. Changes in loadings from S154 and S65E were not significant at levels less than 25%. Interestingly, average total loadings from all basins other than S191, S154 and S65E were essentially unchanged from the 1970s (304.7 tons) to the 1980s (298.8 tons) if the impact of the Interim Action Plan (IAP) is ignored. The IAP, which limits backpumping of water into Lake Okeechobee from the Everglades Agricultural Area, was initiated in 1979 and has been credited with reducing average TP loadings to the Lake by 10 tons per year (SWIM, 1989).

9 The water quality concentration data are positively skewed, thus the trends are calculated using the seasonal medians rather than the means.

Implications

One of the most obvious implications of the Lake Okeechobee experience is that programs designed to solve complicated, nonpoint pollution problems often will be evolutionary in terms of their complexity, rather than revolutionary. The political process of dealing with the uncertainty and lack of information about the problem and alternative solutions, equity concerns (including property rights/takings issues), and administrative inflexibility once programs are put in place, all but guarantee a cautious, step-by-step approach. In the case of Lake Okeechobee, key components of the nonpoint programs have evolved in complexity over time including technologies, monitoring programs, and incentive mechanisms.

The evolution of technologies is in effect converting a primarily nonpoint source into a point source. Likewise, monitoring programs have evolved in purpose and design from an initial focus on problem assessment, to measuring efficacy of practices, and finally to providing a basis for implementing and determining compliance with performance standards. Finally, incentive mechanisms have evolved from purely voluntary with full cost sharing, to voluntary with partial cost sharing, to implied regulatory threats, to a technology based standard with cost sharing, to finally a performance based standard with no cost sharing. But the threat of potential regulation throughout the process stimulated high levels of "voluntary" participation.

The second major implication is that communication and cooperation are essential if complex nonpoint problems are to be solved. Participation in the program by the dairies was greatly assisted by clear documentation that phosphorus loads affected the health of the Lake, and that dairies were the primary source of the problem. A simple materials balance analysis of phosphorus flows in the watershed was particularly effective in clarifying the sources of the phosphorus. Likewise, although the SFWMD is often perceived as the "bad guy", the presence of monitoring and regulatory staff in Okeechobee County greatly improves communication and understanding, particularly since the requirements on the landowners continue to evolve. Finally, the TC/NS - LKR RCWP has experienced an unusual degree of cooperation between federal, state, district, and county governments as well as with the dairy operators, which has been critical to the success of the program.

The third major implication is that traditional textbook economic incentives (emission taxes) have not been used in the Lake Okeechobee programs and may not be viable alternatives for many nonpoint source problems due to the uncertainty of effect, political aversion, administrative inflexibility, and monitoring (measurement) problems (Anderson, et al., 1990, Baumol and Oates, 1988). A broader concept of economic efficiency that accounts for the reality of differential political and administrative costs associated with alternative incentive mechanisms needs to be encouraged. Traditional textbook comparisons of market incentives versus technology standards often understate the real flexibility producers are given to comply with standards and thus generally overstate the real advantages of economic incentives. This is particularly true if firms are homogeneous. Economic incentives have and can continue to play a role in the Lake Okeechobee situation. Input taxes can be used to raise revenues to offset the costs of abatement, cost sharing can be provided, property right easements can be purchased, and marketable permit systems may be feasible in some circumstances.

The fourth major implication is that before emissions can be taxed or emission permits traded, emissions must be measurable. Measuring nonpoint source loadings is difficult and expensive. For many nonpoint source pollutants, measurement is technically or economically infeasible. The SFWMD is currently developing procedures to monitor nonpoint concentrations to be used as the basis for assessing compliance with performance standards. It is important to recognize that the monitoring requirements for a tradeable permits program may be considerably more stringent than are needed to enforce performance standards.

The fifth major implication is that the combination of incentives, timely research and demonstration projects, and flexibility to respond has resulted in cost effective results. The formulation of performance standards in the Lake Okeechobee SWIM plan and the potential threat of enforcement were critical factors in stimulating the development of a market for composting dairy wastes as a soil amendment and in the reduction in phosphorus content of dairy feed rations. Unfortunately, the performance standards are coming on the heels of a technology standard which has already somewhat limited the flexibility of the dairies to respond.

The power of the market was also exhibited indirectly in the Lake Okeechobee dairy ceasing operations program as the higher cost dairies or dairies with higher discharge concentrations were selectively attracted to the program. The efficiency of the program might have been enhanced if a competitive bidding system had been employed.

The final implication is that nonpoint source problems are generally going to be addressed in a cost effectiveness context (Baumol and Oates, 1988) due to the difficulty of measuring benefits, uncertainty about key parameters of the problem, and the political preference for specifying specific targets (e.g., the 397 ton target for Lake Okeechobee). But cost effectiveness calculations are extremely complex in the case of most nonpoint source problems due to the evolutionary aspect of technologies and incentives, and the dynamics of the system including lags in effect and stochastic effects.

An accurate cost effectiveness assessment of the Lake Okeechobee nonpoint source programs is impossible at this point. Preliminary results are consistent with two common characteristics of pollution control programs. First, the marginal cost of reducing emissions increases exponentially. Preliminary results from the TC/NS RCWP indicate that roughly a 45% reduction (0.5 mg/l) in seasonal median trend phosphorus concentrations was achieved at a cost of approximately $100 per cow. In contrast, the Dairy Rule cost approximately ten times that per cow, and the hope is that the median trend concentrations will fall another 0.4 mg/l. Second, increasing the reliability of nonpoint source regulations (i.e., a concentration standard which must be met ninety percent of the time rather than on average) would drive up costs dramatically as evidenced by the peak concentrations in the monitoring data.

Current Directions

Recently, the Chesapeake Bay Nonpoint Source Evaluation Report (1990) recommended that efforts to clean up the Bay: (1) take a mass balance approach, (2) employ a systematic planning framework, (3) target problem areas, (4) use a mix of regulatory and nonregulatory mechanisms, and (5) shift from using the term BMPs to Best Management Systems (BMSs) to reflect a more comprehensive, systems approach. The University of Florida is currently working with the SFWMD to assist them in developing a final SWIM plan for Lake Okeechobee that is consistent with the majority of the Chesapeake Bay Report recommendations.

A geographic information system (GIS) based, decision support system was developed to assist District managers in evaluating alternative nonpoint source control plans. The system, dubbed LOADSS, takes a mass balance approach and provides a systematic planning framework for evaluating both pollution reduction and abatement practices. The GIS structure allows for spatial evaluation and targeting of phosphorus control practices. The purpose of LOADSS is to provide information on the cost effectiveness of alternative plans for achieving the 397 ton target. This information will be used along with evaluations of alternative incentive mechanisms to formulate the final Lake Okeechobee SWIM plan. The final plan will likely incorporate a combination of pollution reduction and abatement practices and a mix of incentive mechanisms.

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