CHESAPEAKE BAY RESEARCH CONFERENCE

Effects of Upland and Shoreline Activities on the Chesapeake Bay

Proceedings of the Chesapeake Bay Research Conference

EFFECTS OF UPLAND AND SHORELINE LAND USE ON THE CHESAPEAKE BAY

Edited

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Sponsored

by

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Preface

Chesapeake Bay, the largest estuary in the world, is extremely resourceful in seafood production, recreation, cargo shipping and navigation. The welfare of the bay affects more people than just those living on its shores. The healthy condition of the bay rests upon the control of shore erosion, wetland growth, point and nonpoint source pollution which are all related to upland and shoreline land uses.

The purpose of the conference is to provide a forum for an exchange of information among scientists and technical personnel concerned with the effects of the land use on the water quality and uses of the bay. Papers included in the proceedings cover three major areas of interest: (1) Rural, urban, and atmospheric nonpoint source pollution; toxic hazardous waste, wastewater, sludge, and other point source pollution. (2) The effects of land use on marina siting, shellfish, habitats, vegetated and nonvegetated wetlands, submerged aquatic vegetation, benthic systems, and freshwater and saline zones. (3) Economic, legal, institutional, social, managerial, and regulatory aspects of land use affecting the water quality of the bay.

Organization and planning for the conference has been the responsibility of the following individuals:

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BACKGROUND INFORMATION ON THE CHESAPEAKE BAY COMMISSION AND SELECTED CONCLUSIONS FROM "CHOICES FOR THE CHESAPEAKE: THE FIRST BIENNIAL REVIEW OF THE ACTION AGENDA"

by

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The Chesapeake Bay has been the focus of an enormous amount of discussion, debate and action over the past five years, and the Chesapeake Bay Commission has been intimately involved in, and frequently at the forefront of, these efforts. It is particularly encouraging to realize that support for the Chesapeake Bay restoration effort has come from virtually all segments of the extended Bay community. Legislators, scientists, government officials, industry and the general public have joined forces in a collective effort to identify and address the causes for the deterioration of this magnificent resource. When these programs and deliberations were initiated several years ago, few, if any, of those involved realized the enormity of the task being undertaken in efforts to restore the Bay. Nor was anyone fully aware of the momentum which would be generated or the degree to which public opinion would focus on and support these efforts. Programs to protect, restore and enhance the viability and productivity of the Chesapeake Bay have become clear and highly visible priorities at all levels of government throughout the entire drainage basin.

The Chesapeake Bay Commission represents a unique tri-state effort to confront the problems facing the Bay in a coordinated and cooperative fashion. The Commission was initially formed in 1980 by joint action of the Maryland and Virginia General Assemblies in recognition of the two states' common interests in the quality and uses of the resources of the Chesapeake Bay. In 1985, again by mutual legislative action, the Commonwealth of Pennsylvania was incorporated as a full and equal partner in the Chesapeake Bay Commission.

The original intent in establishing the Commission was to create an interstate legislative advisory commission to assist and guide the legislatures in identifying and confronting those Bay-related issues which might be best addressed through joint or cooperative actions of all states in the region rather than by the individual or independent actions of a single state. Many issues of concern in the Bay region transcend state boundaries and geo-political subdivisions, but the Bay, historically, has not been regarded or comprehensively examined as a single entity. The very creation of the Commission, then, was a recognition and acknowledgement of the fact that the Bay is a shared resource and that her stewardship is a shared responsibility. The addition of Pennsylvania to the Commission ensures that the three states which most directly and immediately contribute to, and benefit from, the water quality and productivity of the Chesapeake Bay will share a common forum in which they can exchange information, discuss problems of individual and/or collective concern, and, hopefully, develop and propose solutions to those problems. The significance of this action is that, for the first time, the three states are united in their efforts to develop legislative and regulatory programs to address the status of water quality and living resources throughout the entire Chesapeake Bay watershed.

A major impetus for the increased interest and effort directed toward the health of the nation's largest and most productive estuary, of course, was the completion in 1983 of the EPA's comprehensive and much-publicized Chesapeake Bay study. That study, completed after six years of intensive study and an expenditure of almost $30 million, identified serious local and Bay-wide problems such as nutrient enrichment, concentrations of toxic substances in the sediments and water column, oxygen depletion, and declines in the abundance of submerged aquatic vegetation and other living resources.

The release of these findings, and other factors, have combined to bring the plight of the Chesapeake Bay to the forefront of the public consciousness. Support has been galvanized from all quarters, including elected officials at all levels of government. Since late 1983, the states of Maryland, Virginia and Pennsylvania, as well as the District of Columbia and the federal government, have embarked upon an unprecedented cooperative and coordinated effort to restore the Chesapeake Bay. Major legislative, budgetary and regulatory initiatives have been enacted by all jurisdictions. These new and expanded programs have been in place now for almost two full years and, while no state can yet claim unqualified success, all do appear to be on the right track. All of the problems, of course, have not been solved, nor have all of the issues even been addressed. The Chesapeake Bay restoration program must be an ongoing long-term process and a number of years, or perhaps even decades, of perseverance will be required before the community is rewarded with improved water quality and healthier living resources. While it may be too early to realize any concrete physical improvements in water quality or living resource productivity, however, it is not too early to begin to review those programs which have been put into place, to analyze the level of resources which have been committed to those programs, and to determine which problem areas will require new programs and additional commitments.

For this reason, the Chesapeake Bay Commission committed itself in 1983 to periodically hosting a meeting to review the progress being made toward improving the health of the Bay. The first such review was conducted in September of 1985 in Baltimore, Maryland. This "First Biennial Review" was preceded by a series of Work Group meetings held throughout the summer and attended by legislators, agency officials, scientific experts and interested citizens from the three states and the District of Columbia. The purpose of these meetings was to discuss and assess the current status of, and recent progress in, the states' efforts in addressing four distinct problem areas: point sources of pollution, nonpoint sources of pollution, fisheries management and living resources, and land use and resource trends. The tentative conclusions and recommendations of the Work Groups were compiled and presented for discussion at the Biennial Review, where they were analyzed and prioritized by conference participants.

The deliberations and findings of the Nonpoint Sources and Land Use Work Groups are most germane to this gathering. The issues identified and discussed by these groups are among the most important which will confront the Bay community in the years to come, and they impact all areas of Bay life and activities. The Nonpoint Sources Work Group recognized that the states of Pennsylvania, Maryland, Virginia, and the District of Columbia have initiated significant efforts to reduce pollution resulting from nonpoint sources. The increased level of activity is manifested both in terms of new programs which have been enacted since 1984 and in the increased allocation of resources to existing programs. Each of the three states has initiated, as a principal element of its nonpoint source pollution control package, a program to encourage the implementation of best management practices on agricultural lands. Each has made cost share funds available to the agricultural community and has developed educational and outreach programs to inform and encourage farmers regarding the benefits of installing BMPs. The amounts of funding and manpower committed to these programs vary widely from state to state but, due to limited resources, no state has yet been able to fully meet the demands for assistance. At this date, only a small proportion of farmers in the watershed are receiving direct assistance, and additional sources of funding must be sought.

Each state has developed its own system for defining priority areas to which available funds should be targeted, and each has developed a different approach regarding the implementation and enforcement of the programs. While all of the existing programs are essentially voluntary in nature, there are differing degrees of regulatory support or "back-up" available to state agencies and state policies vary concerning the appropriateness and timing of state-imposed intervention or enforcement actions.

Though efforts to date have been concentrated principally in the area of encouraging best management practices on farms, problems attendant to stormwater management and adequate enforcement of existing erosion and sediment control laws have also been recognized. The participants at the Biennial Review identified the following areas as priority concerns in controlling nonpoint sources of pollution.

- Cost-share programs should be continued and expanded as necessary. States must recognize that these programs are long-term efforts which will require continuing support (administrative and technical, as well as financial) from the legislatures and executive agencies if they are to demonstrate positive results. Cost-share programs must be carefully tracked to ensure that resources are being utilized for optimal efficiency and effectiveness. States should also examine alternatives to federal and state cost share programs. Economic incentives such as tax credits and use-value assessments have been shown to be effective in many instances. Regulatory approaches, as well as private funding and funding from local government sources should also be considered.

- Continuing and increased participation on the part of the agricultural community, developers and the general public is absolutely essential to the success of these programs. Farmers, developers and the public at large must be made aware of the economic and environmental benefits which can result from their actions. Ongoing educational efforts are also important to ensure the awareness and support of all nonpoint source control programs directed to the public sector, urban and suburban, as well as rural. Financial resources must be made available to deliver these educational programs.

- It is important that adequate manpower and resources at the federal, state and local levels be appropriated to these programs in all jurisdictions and that authorized positions be filled as expeditiously as possible in order for programs to achieve their maximum potential within the shortest feasible time frame. It is particularly important that new programs be granted adequate opportunity and resources to demonstrate their capabilities and effectiveness.

- Effective enforcement of erosion and sediment control laws for non-agricultural activities has been identified as a problem in all areas. Sufficient resources must be appropriated if erosion and sediment control programs are to be effectively implemented and enforced.

Other issues identified through the Review process included the need to develop an effective monitoring program with the goal of identifying the impacts of best management practices as reflected in water quality changes. Another concern was the direction of efforts to identified priority areas. The states must continually review and evaluate their targeting strategies for both urban and agricultural nonpoint source control programs to ensure that the most beneficial and cost-effective results are obtained. Given the limited amount of available funding, it is essential that state efforts and resources be directed toward those areas and programs which will demonstrate the most positive results. Finally, the Nonpoint Source Work and Discussion Groups identified a need for continued research into the role of nitrogen and phosphorus in the Chesapeake Bay ecosystem. Particularly acute are the need for a better understanding of nitrogen pathways in the Bay and sediment-water interactions as they relate to nutrient release.

The Land Use and Resource Trends Work Group had perhaps the most difficult assignment since land use planning was not a specific focus of the Chesapeake Bay Initiatives. The Group was given the task of evaluating land uses and resource development trends in the Chesapeake Bay watershed, a task which proved to be somewhat more difficult than had originally been anticipated. Not only is the subject matter rather diverse and amorphous, but, in terms of a review or assessment of progress, there is very little in the way of baseline information which can be used in attempts to measure the advances, if any, which have been made over the past two years. The information needed to address this issue is not available from any central source, but, rather, when available at all, is widely dispersed among a number of private and governmental agencies in all three states.

The basic focus of their efforts, then, was to document population growth and development trends in the Bay region. The intent was not to imply that growth is necessarily bad or that it should in any way be discouraged since continued growth is essential to maintain the relative economic and social prosperity of the area. It must also be realized, however, that growth creates additional demands for living, working and shopping places, increases the need for sewage treatment plants and the loads which those plants must accommodate, and intensifies the need for additional public services such as highways, schools, fire and police protection, and water and sewer facilities, all of which represent increased costs and additional burdens for local governments. Development, when unplanned or unanticipated, frequently occurs in a haphazard fashion, often in those areas which are least capable of handling growth pressures and the subsequent demand for additional services. To the extent which these changes in growth and development patterns require the conversion of forestlands, croplands and pasturelands to other more intensive land uses, such as residential and commercial development, additional pressures are placed upon the fragile and already-stressed Chesapeake Bay ecosystem.

The Chesapeake Bay Commission, by establishing land use and management as a specific focus for one of the four work groups preparing for the Biennial Review, has defined this issue as one which is central to the restoration and protection of the Bay.

Much of the discussion of the work group focused on the ability and response of local governments in implementing the land use authorities granted them by state governments. Concern was centered on the issue of whether local governments were equipped and willing to protect state interests in water quality and natural resource protection. Development in the Chesapeake Bay watershed has been concentrated along the Harrisburg-Baltimore-Washington-Richmond-Norfolk urban corridor. Growth patterns over the past five years, however, have begun to show some elements of change. Rural areas such as Southern Maryland and the Northern Neck-Middle Peninsula areas of Virginia are experiencing rapidly increasing growth pressures. Jurisdictions such as these are frequently ill-equipped to handle the more sophisticated planning functions associated with rapidly developing urban areas.

Growth in the Chesapeake Bay area will continue to increase into the foreseeable future, leading to increased stress upon the Bay ecosystem. The impacts of growth and land development, however, can be mitigated and, to some extent, directed through existing comprehensive land use planning and zoning mechanisms. Enhancing, or even maintaining, the quality of the Bay while accommodating growth, however, will involve compromise.

While many localities have strong zoning ordinances and/or regulatory programs in place, enforcement is a problem in all jurisdictions. Without adequate enforcement, regulations are virtually meaningless. Individual zoning and permit decisions may gradually erode well laid and well-intentioned land use plans and policies.

Another issue raised was whether enabling legislation which authorizes local governments to plan and zone is sufficient to grant them authority to protect water quality and other resources. In Virginia, for instance, the authority of local governments to incorporate environmental criteria into their zoning regulations has been called into question through a number of court challenges. In almost all cases, the courts have invalidated innovative exercises of local zoning authority to control growth or to preserve environmental amenities.

Regulation, however, is not the only approach to directing land use trends or mitigating impacts. The influence of taxes and other economic incentives and disincentives affecting the quality and pattern of development has also been stressed. The potential role of the state government in serving as a role model for the mitigation of impacts through proper development techniques was emphasized. A central concern was that state government assume a more active leadership and oversight role in assisting local governments in the development and execution of effective land use policies.

The Land Use Work and Discussion Groups identified the following problem areas as immediate, or priority, issues which should be addressed in the area of land use and growth development control:

- The states must take a stronger role in the development of comprehensive land use plans and zoning regulations, particularly as they affect water quality and habitat preservation and protection, and where potential inter-jurisdictional conflicts exist. More specifically, the state's role should include: (1) a definition of state interests that transcend local goals, and (2) protection of those interests in the event that local governments fail to do so.

- Without adequate enforcement, local zoning regulations are meaningless. The federal government, states and localities must provide sufficient leadership, direction and resources to effectively enforce existing and proposed land use programs. Jurisdictions should be evaluated and ranked as to their effectiveness in administering water quality protection programs. Such evaluation should include the consistent and long-term collection of local land use data by an entity which could report annually to the legislatures.

- State governments should provide more active leadership and oversight as well as increased financial, technical, legal and policy support to local governments to assist them in reducing the adverse impacts of growth and development. States should also implement training programs for local planning officials.

- Stricter standards for sewage treatment, stormwater management, sediment control, and water conservation, as well as offsets in terms of installing best management practices on farmland and on redevelopment of urban areas will become increasingly necessary to maintain or enhance the quality of the Bay system as population growth and land development continue. Local governments must develop a commitment to effective land use practices and provide the funds, or fee structures, to support these programs.

Additional issues raised included the role of the state in protecting agricultural and forest lands. The preservation of agricultural and forested land, as well as all sensitive habitat areas within the Chesapeake Bay drainage basin, should be matters of explicit state policy and should not be left solely to the discretion of local governments. Innovative programs such as tax incentives and preservation easements should be examined and utilized to make it more economically attractive to reduce the amount of forest and agricultural land which is converted to more intensive uses. The need for accurate and current land use information was also identified. In order to have a yardstick to measure the impacts of growth and development and to assess the viability and effectiveness of state and local land use programs, it is essential that state and local governments, as well as the general public, have access to land use information which is frequently and consistently updated. There are monitoring programs in place for the waters and living resources of the Bay, but there is currently no system in place to assess changes in land use on a Baywide basis.

This paper, obviously, touches only briefly on those issues facing the Land Use and Resource Trends Work Group. The fundamental conclusion is that growth and development in the Bay watershed will continue to accelerate and that all jurisdictions in the region must have mechanisms in place to deal with these changes in a comprehensive, yet flexible, manner. This will require an increase in manpower and resources, as well as increased willpower, dedication and education.

It is clear from this Review process that many accomplishments have been gained since the original commitments were made in 1983. It is equally clear that many important issues have yet to be addressed and that the most difficult tasks may lie in the future. Programs which most obviously and visibly affect the water quality of the Chesapeake Bay have been adopted and put into place. Furthermore, the Chesapeake Executive Council has adopted a long-range Chesapeake Bay Restoration and Protection Plan. The plan establishes goals and objectives which have been mutually agreed upon by all jurisdictions, and therefore represents a major achievement in improving interstate and federal-state communication and cooperation.

What is needed at this juncture is a blue print for action which defines the level of necessary funding and specifies the individual projects which must be put into place this year, next year and the year after. In developing this blue print, there are several vital and fundamental issues which must be addressed and resolved:

1. For more than fifty years, institutional programs and structures have been in place to provide technical expertise to farmers to assist them in curbing soil erosion. New techniques and practices are being developed to deal specifically with nutrients rather than with soil erosion <u>per se</u>. The basic problem lies in convincing farmers to adopt these practices. It will probably never be possible to provide cost share assistance for the installation of best management practices on every acre of farmland which is critical to the Chesapeake Bay. Thus, all jurisdictions are confronted with the questions of

whether voluntary programs which encourage, but do not require, the installation of agricultural best management practices are sufficient to stem the flow of nutrients and sediment leaving the farmlands and entering the waterways, or must mandatory regulatory measures be implemented?

2. Many local governments have not made full use of the land use authority granted to them in zoning enabling legislation. In the face of increasing population pressures and attendant demands for the conversion of low intensity land use to residential and commercial development, can the Bay be restored in the absence of a mandate that land development be directed, and its impacts mitigated, according to state developed and imposed standards?

3. An overall nutrient control strategy for the Bay has not yet been developed. Only in the Susquehanna, the upper main stem of the Bay and in the Potomac River have phosphorus removal technology been utilized and, even in those areas, there are no established limits on the total load of phosphorus which can be discharged. Nitrogen removal is scheduled only for the Patuxent estuary. The time frame and appropriate locations for the removal of nutrients from point sources remain largely unanswered questions. Should permitted nutrient loads be established immediately, and subsequently reviewed and modified as knowledge and understanding improve, or should definitive action depend upon the results of additional modelling and research efforts?

4. The final, and perhaps most critical question, revolves around the appropriation and allocation of funds. While a massive amount of financial resources has been committed to the Bay restoration effort since 1984, it is clear now that these commitments are just a beginning. Success will depend on a desire and willingness to commit several times what has already been dedicated. Are the states willing to continue and increase levels of funding for the Bay initiatives? If so, where will the necessary resources be found?

The Chesapeake Bay Commission hopes that some, if not all, of these questions can be answered before the convening of the Second Biennial Review in 1987. Meanwhile, we hope and fully expect that the states, as well as the Chesapeake Bay Commission, will continue to honor the commitments which have been made to protecting and enhancing the resources of the Chesapeake Bay. With the anticipated expansion of existing efforts and the introduction of new programs in the future, there is every reason to believe that the states will manage to find the resources to finance and accomplish our agreed-upon goals, and that these cooperative efforts will enable us to restore this national treasure.

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A GIS APPROACH TO IDENTIFYING AG NON-POINT POLLUTION

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ABSTRACT

The use of a computerized Geographic Information System (GIS) was tested against a manual approach for identifying agricultural non-point source pollution of the Chesapeake Bay. The Universal Soil Loss Equation (USLE) with a delivery ratio function was used to combine soils, water bodies, elevation, and land use data to estimate potential sediment delivery to streams from agricultural land. The pilot study revealed that the computerized approach was more effective, less expensive, and holds greater potential for reuse than the manual approach. The products of this study will be used to target State cost-share funds for the implementation of Best Management Practices (BMPs).

INTRODUCTION

<u>Statement of the Problem</u>

In an era of great alarm over the degradation of the Chesapeake Bay and also a time of limited financial resources to fight such a decline, the ability to target resources where they can do the most good rather than merely spreading them around is especially crucial. This paper describes work undertaken at Virginia Polytechnic Institute and State University in Blacksburg for the Division of Soil and Water Conservation of the Virginia Department of Conservation and Historic Resources. It is the intention of this work to develop a cost-effective, accurate, and timely method of estimating sources of and potentials for agricultural non-point source pollution of the Chesapeake Bay so that farmers and landowners can be contacted and offered financial assistance for implementation of best management practices (BMPs).

The task was begun (by others) employing manual techniques, but it became apparent that this approach was not going to give satisfactory results within budgetary limitations. At this time, a team of agricultural engineers and a landscape planner was organized to consider whether a computerized geographic information systems approach would be effective in addressing the non-point source pollution problem. As in the manual approach, sediment loss was to be calculated by the Universal Soil Loss Equation (USLE) with a delivery ratio included to predict how much soil might make it to a body of water that fed into the bay or the bay itself.

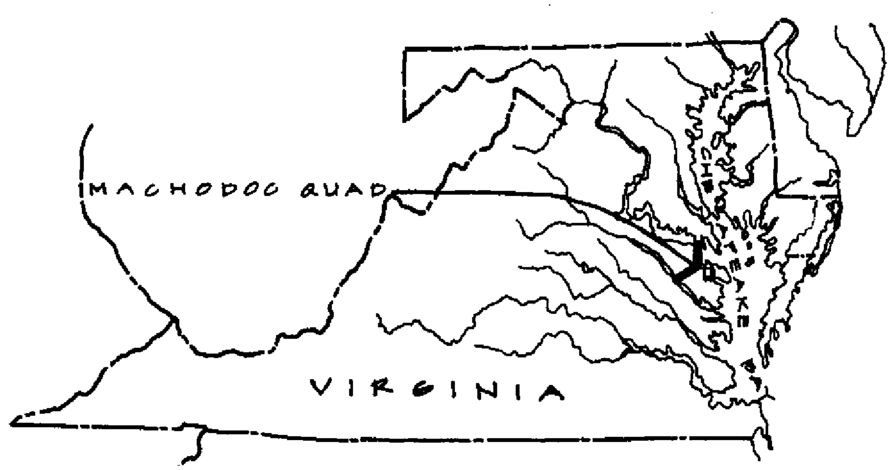

Figure 1. Location of the Machodoc Quadrangle, Westmoreland County, Virginia

To this end, the Machodoc USGS 7-1/2 minute quadrangle was selected as a the area for the pilot study. This covered an area of Virginia (see figure 1) that had been completed by the manual process and would provide a ready geographic comparison. The Virginia Tech team had previous experience in calculating agricultural non-point source pollution and in the creation and use of geographic information systems, but had never combined the two. GISs have been used for general land use planning at Virginia Tech, but not for the more precise kind of calculations that would be necessary for predicting sediment loading. It was predicted that a computerized GIS approach would have many advantages over a manual one. Some of these are obvious, such as the reuseablity and flexibility of a digital data base, the speed of analysis, and the ability to test alternative scenarios. What needed to be ascertained was whether this approach could give accurate results and save time and money over the manual strategy.

Survey of Previous Work

Equations like the Universal Soil Loss Equation (USLE) (equation 1) have been used in the United States since about 1940 to predict the average rate of soil erosion for given crops, management practices, soil types, rainfall patterns, and topography (Wischmeier and Smith, 1978). The equation is:

$$PSL = R \times K \times LS \times C \times P \qquad (1)$$

where PSL is the potential sediment loading, R is the rainfall factor, K is the soil erosivity factor, LS is the length slope factor, C is the cover factor, and P is the practice factor.

Added to this equation is a delivery ratio (DR) to provide an estimate of the amount of the eroded soil that might actually make it to a nearby stream (or other body of water). The equation used for this study (equation 2), then is a modified USLE:

$$PSL = R \times K \times LS \times C \times P \times DR \qquad (2)$$

The following equation is used for determining DR.

$$DR = 10(r/L) \qquad (3)$$

where:

L is overland flow length in feet and is measured as the shortest downslope distance to a water cell.
r is relief in feet and is the difference in elevation between an agricultural cell and the associated stream cell.

Spanner (1983) used a previously created GIS and the USLE in predicting soil erosion rates in Santa Cruz County, California. He found a savings of both time and money in using a computerized approach over conventional ground sampling techniques in examining large areas. Walsh (1985), in describing the Oklahoma Geographic Information Retrieval System (OGRIS), suggests its usefulness in calculating the USLE. In addition, he comments on the likelihood of refining the USLE in the future with computer capabilities.

None of these authors make use of a delivery ratio as part of the USLE, which makes the calculations what Hansen and Hellmund (1983) call "static," that is they do not consider spatial or temporal relationships, in this case how much eroding soil actually is deposited in a body of water.

APPROACH

The earlier manual pilot study of the Machodoc quadrangle made use of aerial photographs to locate agricultural land, USGS topographic maps to measure distances and determine differences in elevation, and the Soil Conservation Service soil survey for Westmoreland County to identify soil types and erosivity factors.

The Data Base

There are two major strategies for creating digital representations of paper maps: vector and raster. The vector (sometimes called polygon) approach summarizes maps as points and line segments and the values associated with these points and the areas surrounded by the line segments. Raster, the second approach (also called grid), summarizes all map data on the basis of a uniform grid which is superimposed over the map. Values are then assigned to each cell to represent the major type on the map or if some variable is present or not. A raster approach was selected for this project because of the availability of appropriate software at Virginia Tech and the strict time constraints.

Data encoding is often the most time consuming phase of creating a GIS data base. Encoding involves assigning to each cell a numeric value to represent a variable class, which can be an actual value or a legend value (e.g. a 10 for soil mapping unit 8B). Difficulty, however, often is encountered when two or more values for a variable exist within a cell. Several strategies, depending on the variable being encoded, can be used to overcome this problem. The centroid (center) value was used for elevation maps because of the continuous nature of the data. For soil type, the predominant soil type within each cell is assigned to the entire cell. These strategies were chosen because the cell size, 1 hectare (2.47 acres), is within the accuracy of the soil map (1:20,000) and the topographic map (1:24,000) used for this study.

The Process

The basic steps required to create the computerized data base, and calculate Potential Sediment Loading (PSL) rates for this pilot study are as follows:

1. Obtain USGS topographic and SCS county soil maps.
2. Create transparent grid (100 meter) overlays registered to the Universal Transverse Mercator coordinate system (UTM) for each paper map.
3. Delineate watersheds (not to exceed 50 sq. miles) on topo maps. Watersheds are used as areas to summarize PSL values.
4. Encode data (Soil mapping units from the county soil survey; elevation, land use, water bodies, and watersheds from the topographic map.
5. Derive maps with developed software. (Soil erosivity from soil mapping unit; slope—both average and maximum—from elevation; length-slope from slope; delivery ratio from elevation, water bodies, and land use.)
6. Create PSL map file from derived maps and constant rainfall factor (R = 250), constant cover (C =1.0 or 0.15), and constant practice (P = 1). PSL values are generated using maximum and weighted average cell slope, and with and without a delivery ratio, for a total of eight combinations.
7. Rank PSL data in descending order for agricultural land and categorize into class intervals of 20 percent each.
8. Tabulate Pollution Density Indexes for each watershed by dividing the total PSL for the watershed by its area.

Estimating Cell Slope

Cell slope is determined by weighting the slope between the cell and its eight neighbors with the following expression (equation 4). The steepest and longest slope receive the highest weight.

$$s_w = C_1 (S_o/4)^{0.5} + C_2 (S_e/4)^{0.5} \qquad (4)$$

Where:

S_o=Sum of squares of sloes of diagonal neighbors (odd numbered)

S_e=Sum of squares of slopes of adjacent neighbors (even numbered)

C_1, C_2 = Slope length weighting factors. C_1=0.43 C_2=0.57

S=Cell slope from the relationship $(E_c)/l_i$ with E_c as cell elevation, E_i as elevation of neighbor i, and l the cell length of neighbor i. (For a one hectare cell size: adjacent cells l=328 feet, diagonal cells l=464 feet.)

Calculating the Length-Slope Factor

The LS factor was computed for each cell from slope values. The following equation was used:

$$LS=((L/72.6)^m)(0.43 + 0.30S_w+0.043S_w^2)/6.661 \tag{5}$$

Where:

L is slope length in feet, in this case equivalent to the cell length.

$m = 0.2$ if $S_w \leq 1$ $m = 0.3$ if $1 < S_w \leq 3.5$ $m = 0.4$ if $3.5 < S_w \leq 4.5$
$m = 0.2$ if $S_w > 4.5$

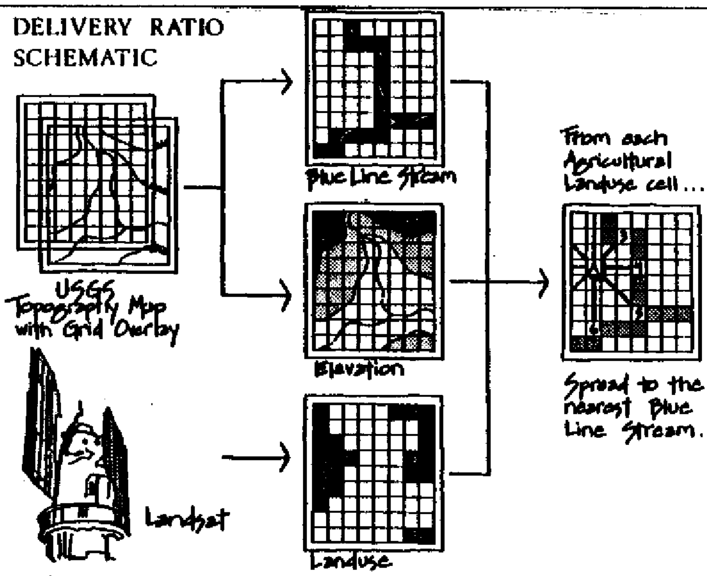

Figure 2. Calculating delivery ratio.

Calculating the Delivery Ratio

Of special interest is the calculation of the delivery ratio, illustrated in figure 2. The delivery ratio map file is derived from the elevation, water bodies and land use maps. The elevation map file provides the basis for determining the distance from a cell to the nearest downstream water body and the elevation between the two points. Delivery ratios are calculated only for agricultural land, therefore, the land use map file is needed to identify agricultural cells. The following algorithm is used to determine the nearest body of water.

1. Search the land use map for an agriculture cell.
2. From the agricultural cell move through the stream map in eight cardinal directions until a body of water or the edge of the study area is encountered.
3. Determine the overland flow distance to the nearest downhill stream and its elevation for the eight possible flow routes. The distance is measured in a straight line from the front edge of a cell to the centroid of a stream cell.
4. Determine the delivery ratio with equation 3.

Calculating PSL

The PSL map file was determined by multiplying together each map file: soil erosivity (K), length slope (LS), and delivery ratio (DR) on a cell by cell basis. Cover and rainfall erosivity factors were held constant in this pilot study.

SUMMARY OF RESULTS

The following eight scenarios were used in calculating the PSL rates.

a) LS factor based on maximum cell slope with calculated delivery ratio and cropland (C = 1).
b) LS factor based on average cell slope with calculated delivery ratio and cropland (C = 1).
c) Same as a) but without a delivery ratio.
d) Same as b) but without a delivery ratio.
e) Same as a) but with pastureland (C = 0.15).
f) Same as b) but with pastureland (C = 0.15).
g) Same as e) but without a delivery ratio.
h) Same as f) but without a delivery ratio.

The data then were ranked in descending order (highest to lowest) and placed in five groups of equal range by number of cells. Table 1 summarizes the grouped PSL values for each of these scenarios.

The relative trends of all methods are similar. A significant reduction in estimated PSL resulted when the cover factor for pasture was used, as was expected, because it is linearly related to other variables in the basic equation. Beside illustrating the effect of pasture cover on soil loss, the results also demonstrate the ability of the GIS algorithms to replicate manual procedures in evaluating alternative scenarios. The results for the two methods of computing slope differ as expected, that is, the PSL rates calculated using average cell slope are less than rates calculated using the maximum cell slope. The magnitude of soil loss for both the maximum slope and average slope scenarios, with no delivery ratio, indicates over-estimation of sediment loadings (some values in excess of 1000 tons per acre). The maximum slope method, however, highlights steep landscapes located along the edges of fields and usually nearer streams and, these areas usually have a higher potential pollutant problem. The average slope scenario appears to provide a better balance between "reasonable" estimates of PSL values and identifying specific PSL source areas. Additionally, this method helps smooth discontinuities between adjacent cells, therefore, minimizing distortion due to aggregating data to the cellular format. We recommend using the average cell slope for future studies.

PSL Scenario	A (highest 20%)	PSL (tons/acre) for five equal classes B	C	D	E (lowest 20%)
1. Maximum slope Cropland (C=1) Calculated DR	>47.5	11.7-47.5	4.5-11.7	1.7-4.5	0-1.7
2. Maximum slope Cropland (C=1) No DR	>81.0	32.8-81.0	18.6-32.8	11.5-18.6	0-11.5
3. Maximum slope Pasture (C=.15) Calculated DR	> 7.0	1.8-7.0	0.7-1.8	0.3-0.7	0-0.3
4. Maximum slope Pasture (C=.15) No DR	>12.1	4.9-12.1	2.8-4.9	1.7-2.8	0-1.7
5. Average slope Cropland (C=1) Calculated DR	>17.3	5.6-17.3	2.7-5.6	1.1-2.7	0-1.1
6. Average slope Cropland (C=1) No DR	>28.8	16.2-28.8	11.5-16.2	8.0-11.5	0-8.0
7. Average slope Pasture (C=.15) Calculated DR	> 2.6	0.8-2.6	0.4-0.8	0.2-0.4	0-0.2
8. Average slope Pasture (C=.15) No DR	> 4.3	2.4-4.3	1.7-4.3	1.2-1.7	0-1.2

Table 1 Ranking of PSL rates for different land cover, slope and delivery ratio scenarios.

Pollution density indexes, as previously defined, are listed by watershed in Table 2 for PSL rates computed by the average slope method. These data show that all cropland agriculture will result in potentially higher than desirable soil loss. Table 2 also contains comparable values for a pastureland scenario. As expected, this management practice has significantly reduced the sediment loading potential.

	No.	Total Acres	Agricultural Acres	Pollution Density Index	
				Cropland	Pasture
Watershed	1	686.7	197.6	16.44	2.45
	2	2,707.1	802.7	11.43	1.71
	3	308.7	111.2	1.59	0.24
	4	13,765.3	5,740.3	14.17	2.12
	5	4,606.5	1,501.8	30.14	4.52
	6	7,521.1	3,443.2	24.30	3.65
	7	44.5	32.1	2.25	0.33

Table 2 Pollution density indexes for pilot study area.

Precise comparisons between the manual and computer approaches are difficult to make because methodologies are significantly different. Specifically the size of the unit of calculation accounts for procedural differences in the two methods of calculating PSL. The "farm unit" used as the unit of calculation in the manual procedure can vary in size from a few acres to hundreds of acres, which results in considerable averaging of factors in the USLE. Using the GIS concept, the area is sub-divided into small (1 hectare), uniform cells and the USLE factors are determined for each cell. Several to many cells are used to define a given "farm unit" and with the GIS individual cells become the unit of calculation. In this sense, the GIS is a refined version of the manual procedure. It is much more efficient, however, in that data can be readily checked for errors and many scenarios can be routinely evaluated. Regardless of the refinements used, however, both methods should give similar rankings of PSL rates. If a "farm unit" was ranked by the more aggregated manual procedure as a problem area, then in general the approach with the GIS should indicate some cells within the "farm unit" as also being a potential pollutant source. It also would be expected (and proved to be the case) that some cells identified by the GIS method as having a high potential would be missed by the hand method due to the averaging effect of large units of calculation.

Comparisons with the hand-generated map from the earlier pilot study showed general agreement. (For comparisons of results see Shanholtz, et al, 1985.) With the GIS, targeting is more specific as expected, as the edge of fields near streams often received high rankings.

Several areas were selected for a more detailed evaluation of specific values calculated for each variable. The resulting data suggest excellent agreement in that no values appear significantly out of range. Because the GIS system is a refinement of the manually-derived procedure, (which results in considerable aggregation), the range of values for each variable calculated by the GIS would be expected to generally bracket comparable values obtained using a manual procedure. The delivery ratios, which were determined from the edge of a "farm unit" in the manual process, should be near the upper range or possibly slightly higher then those calculated by the GIS.

It is difficult to compare these results with data summarized over entire fields, but close inspection of the manually derived map shows areas that will provide the best comparisons. These data also suggest that the high PSL values generated for some cells will significantly effect regional averages. These cells are often at the steep edges of fields and are mostly, but not totally, agricultural—the steepest slopes usually being wooded. If the magnitude of soil loss over regions is very important, then some smoothing of the elevation data may be necessary to reduce the influence of these steep areas that are not in agriculture, but are in cells with agriculture.

CONCLUSIONS

This pilot project clearly illustrates the utility of a Geographic Information System in identifying the non-point source pollution potential of agricultural land in Virginia.
Checking the results of both approaches revealed that potential sediment loading factors were calculated as accurately by the GIS as by the manual techniques. The strongest attributes of the GIS approach are future usefulness based on ease of updating, flexibility for use with other similar projects, and cost-effectiveness in comparison with manual techniques.

Further refinements in the calculation of delivery ratio are needed. The approximation of how much soil actually reaches a stream was affected in this study only by distance and slope, whereas in reality the land cover separating agricultural land and water would certainly have an effect.

Due to the success of this pilot study, a larger (22 county) project is currently underway to identify areas of potential non-point source agricultural pollution in Eastern Virginia. The cell size for elevation was increased to 4 hectares (9.88 acres) as a cost-saving measure, after it was determined that this did not affect the accuracy of the calculations to any large degree. The other data continues to be gathered at the 1 hectare resolution. Landsat data is being processed by the Virginia Institute of Marine Science, thus automating the creation of land use maps.

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AGRICULTURAL NON-POINT SOURCE POLLUTION CLASSIFICATION

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ABSTRACT

The Environmental Protection Agency's (EPA) Environmental Photographic Interpretation Center (EPIC), through funding from EPA Region 3 and with the cooperation of the Virginia Soil and Water Conservation Commission (SWCC), developed and performed a pilot non-point source pollution classification program of the agricultural land in Westmoreland County, Virginia. The purpose of the program was to develop a methodology for targeting the application of cost share funds to implement Best Management Practices for soil and nutrient conservation.

The program utilized aerial photography to identify agricultural activity and to delineate field units. Each field unit was assigned a potential sediment load (PSL) value which was based on the field's pollution generating (PG) potential and its pollution delivery (PD) potential.

The PG potential was computed using three components of the Universal Soil Loss Equation (rainfall, soil erodibility, and slope-length) in a simplified form. The PD potential was computed using a simplified form of the equation created by Williams and Berndt (1972) to predict field edge to stream delivery of sediments.

There were 1,808 field units identified comprising 44,743 acres of agricultural land. Twenty-four (24) items of data relating to soils, and physiographic characteristics were recorded for each field unit. The fields were separated into five equal-sized groups based on their PSL value. Those fields falling into the category with the highest PSL's have the greatest potential to be non-point source problems. These rankings were then used to classify watersheds to identify those with the highest concentrations of potential problem areas.

BACKGROUND

The delivery of non-point source pollution from agricultural field sites has been cited as one of the major management problems facing the Chesapeake Bay today. The most notable problems associated with agricultural non-point source pollution are the introduction of agri-chemicals (herbicides and pesticides) and fertilizers (phosphorus and nitrogen), soil particles and high concentrations of organic matter from livestock and poultry operations. Common pathways for these contaminants are in surface runoff (either in solution, suspension, or partitioned to soil particles) or migration through infiltration.

The Universal Soil Loss Equation (USLE), developed in 1954 by the Science and Education Administration (Wischmeier and Smith, 1978), was one of the first models developed to predict field edge delivery of sediment from a particular field site. The individual variables which comprise the USLE are the basis of many watershed soil loss models. In addition, while the USLE is a valid predictor of sediment yield from a field site, it does not incorporate all the intricate interactions which affect the actual delivery of chemical pollutants from a field site to a watershed under study.

Since the USLE was developed, other models have been developed to study land use activities and their effects on water quality. Traditionally these have been based upon sediment delivery potential because it was assumed that many of the contaminants which had the greatest effect on water quality were partitioned to the soil particles being washed from field sites.

Current models to study water quality incorporate many of the complexities of the environment (infiltration, vegetation barriers, distance and slope to stream, seasonal changes, etc.) and the chemical nature of the contaminants with which they are concerned. These models typically are very large, complex, and require a long period of time for data collection before any results can be attained.

The intent of this study is to develop a methodology for studying local agricultural land and associated physical factors to produce a general indicator of areas which have the greatest potential for degrading associated water resources. Recognizing the limitations of the USLE and dynamic modeling techniques, this study seeks to combine portions of these two ideologies and produce a management tool that is easy to use and that can be put together in a short time frame so that regional resource managers can immediately assess problems located within their area of responsibility. The information provided will be current and will allow them to evaluate present and future management alternatives to meet water quality objectives.

PROJECT DEVELOPMENT

The Virginia Soil and Water Conservation Commission (SWCC) approached EPIC in December 1983 with a proposal to perform a detailed agricultural land use inventory that would eventually encompass the entire York and Rappahanock River Basins. Dr. Theo Dilaha, of the Virginia Polytechnic Institute and State University, was the primary consultant for the Virginia SWCC and EPIC in the development of the mathematical equations used in the study.

The study was to provide a perspective on potential non-point pollution sources associated with agricultural activities that would allow local Soil Conservation District Offices to decide how best to distribute their cost share funds. These funds are available to implement Best Management Practices (BMP's) to control soil erosion.

The final product would accomplish this by allowing local Soil Conservation Service (SCS) personnel to target farm units which have the greatest relative potential to be non-point source pollution problems. The funds for control measures would be distributed to those areas which have the greatest need.

Westmoreland County was selected as the pilot study area to develop the methodologies and procedures that would eventually be applied to the whole study area. The county is located in the area known as the "northern neck" of Virginia, which extends along the south side of the Potomac River before it empties into the Chesapeake Bay.

The determination of a potential sediment loading (PSL) factor for each field unit is based upon two sets of information. The first is the pollution generating (PG) factor. It is derived directly from the USLE and is a mathematical representation of the physical factors inherent in a given field unit which relate to its sediment production potential.

The factors addressed in computation of the PG factor are rainfall, soil erodibility, and slope-length. Soil management practices which are part of the USLE were not used. It was felt that this investigation should only reflect those conditions which resulted from a field's physical position and related natural conditions and not those created by man's intervention. This also helped to relieve any suspicions on the part of participating farmers that "Big Brother" was looking over their shoulder.

The second set of information used in determining the PSL factor is the delivery ratio (DR) factor. It is a mathematical representation of those factors relating to the physical conditions existing between the edge of a particular field unit and the receiving stream. The DR factor is modeled after that presented by Williams and Berndt (1972).

After all the PSL's had been determined, the Total Loading Potential (TLP) and a Pollution Density Index (PDI) were calculated for each watershed. The TLP is the product of the area of a field and its PSL. The summation of all TLPs for a watershed, divided by the area of the entire watershed, resulted in the PDI. The PDI ranks the individual watersheds according to the relative concentration of problem field sites located within each watershed.

Once all the parameters and equations were chosen, working strategies and methodologies were developed as work progressed in the early stages of the field site analysis in September 1984. Data collection was complete by the end of December 1984, while data processing, final editing and product delivery were complete in March of 1985.

STUDY AREA

The Virginia Agricultural Non-Point Source demonstration study was conducted in Westmoreland County, Virginia. Westmoreland County is approximately 252 square miles in size, with about 69% of the county lying in the Potomac River drainage basin and 31% in the Rappahannock River drainage basin.

The county was chosen for the demonstration study because it is one of the more intensely farmed counties in the Potomac and Rappahannock drainage basins. It also has a diverse topography ranging from low lying, flat central plain to steeper, hilly piedmont-type topography.

Soils in the county consist of seven general soil associations: 1. Lumbee-Leaf-Lenoir, 2. Nansemond-Tetotum-State, 3. Rumford-Kempsville-Emporia, 4. Montross-Ackwater, 5. Suffolk-Rumford, 6. Rumford-Kempsville-Turbeville and 7. Tetotum-Bojac-Pamunkey (USDA, 1981). All but Type 1 are well to moderately well drained soils. Soils tend to be loamy and sandy, and lie on low to high marine and fluvial terraces.

The total annual precipitation in Westmoreland County is 40 inches, with 55 percent of it occurring in April through September (USDA, 1981). The time period of peak rainfall corresponds with the growing season for most crops grown in the county. Thunderstorms occur on about 40 days per year, with the majority of them occurring during the summer months.

The major drainage basins that are either totally or partially included in Westmoreland County are Nomini Creek, Yeocomico River, Popes Creek, Mattox Creek, Cat Point Creek, Totusky Creek, Monroe Creek and Lower Machodoc Creek.

METHODOLOGY

The rankings of the agricultural field units were based upon a PSL factor that was determined by the following equation:

$$PSL = R \cdot K \cdot LS \cdot DR \qquad (1)$$

where: R is the rainfall factor, K is the average soil erodibility, LS is the slope-length factor, and DR is the delivery ratio. It should be noted that the PSL is only an indicator for potential sediment delivery that can be used for comparison against other field units, and not a predictive model for actual sediment loading.

An in-depth description of the methods and equations used in calculating the PSL follows.

Field Unit Identification

Color infrared aerial photography of Westmoreland County was obtained in the months of June, July and August 1984. The photography was acquired at an approximate scale of 1:24,000 to match the scale of the topographic base maps as closely as possible.

Duplicate color positive transparencies made from the original photography were analyzed on a standard Richards light table. The imagery analyst outlined each of the pasture, cropland and feedlot areas onto a mylar overlay to a 7.5-minute USGS topographic quadrangle.

A watershed map, which was created by delineating the watershed for each blue line stream shown on the topographic map, was placed beneath the field site map. Each of the field sites was then divided at those points where watershed boundaries fell within a field site. The stream itself also constituted a dividing line if it bisected a field site.

Field site size was maintained between 5 and 100 acres. A minimum of 5 acres was established because of the difficulty in analyzing such a relatively small area and the relatively minor impact of fields less than 5 acres in size. A maximum of approximately 100 acres was established because analysis and ranking of too large an area does not pinpoint the problem area sufficiently. If a field site still exceeded a manageable size at this point in the delineation procedure, further subdivision was done using the following criteria: (1) roads bisecting the field, (2) different agriculture practice or crop types, (3) fence lines visible on the imagery, and (4) tree lines. If no logical subdivision point could be identified, the field remained as it was.

Upon completion of field unit delineation, each field was assigned a field unit number. Field units within a watershed were numbered sequentially from 1 to n, from top left to bottom right of the quad sheet, and given a 1 or 2 letter prefix to identify the watershed (i.e., the 77th field in Nomini Creek was designated NC77).

Average Soil Erodibility Factor (K)

The average soil erodibility factor, or "K" factor, was determined for all field units using the United States Department of Agriculture Soil Conservation Service (USDA-SCS) soil survey photo maps. The 1:20,000 soil survey maps were reduced to 1:24,000 scale so they could be placed directly beneath the field unit map. The analyst was then able to record all the soil types (up to five) which occurred within each field and estimate the percent of cover of each soil type.

The individual "k" values for each soil type, obtained from the SCS manual, and the percent coverage of each, was then entered into the computer for calculation of the final "K" factor for each field site. The final "K" factor was determined using the formula:

$$K = k_i\ (\%_i) + k_{(i+1)}\ (\%_{(i+1)}) + \ldots + k_n\ (\%_n) \qquad (2)$$

where: K is the average soil erodibility for each field, k_i is the soil erodibility for each unique soil type, $\%_i$ is the percentage that soil type k_i occupies in a particular field, n is the number of soil types in a particular field up to a maximum of five.

Relief Measurements

Three measurements of topographic relief were made for each field site. These were the relief of the field site itself (highest point to the lowest); the relief from the point of transport at the edge of the field to the point of entry into the nearest blue line stream; and the relief from the point of entry into the blue line stream to the level of the major recieving water body (the hydraulic gradient).

The analyst determined the relief values to within 5 feet by utilizing the contours on the topographic map and interpolating for in-between values (contour intervals are 10 feet).

The field relief and the relief to the nearest blue line stream are used in the determination of the slope-length factor and the delivery ratio.

Length Measurements

Four measurements of length were collected for each field site. These were maximum slope-length, distance to nearest blue line stream, distance from the point of entry at the blue line stream to the major recieving water body (the hydraulic length), and the combined length of the contour lines that would exist at 25, 50 and 75 percent of the topographic relief of the field.

Measurement of the maximum slope-length was made from a line drawn by the analyst which represented the analyst's best estimate, based on topographic analysis, of the maxium distance a unit of water could flow over the field surface before encountering a point of deposition or confluence. A point of deposition is defined as a point along the pathway of surface water flow where a flattening of the angle of slope of the soil surface causes soil sediments suspended in the surface flow to settle out and collect. A point of confluence is defined as a point along the pathway of surface water flow where several different surface flows come together, resulting in the alteration of the sediment transport and erosional characteristics of the flow.

The contour length distance is the combined length of the contour lines that would exist at 25, 50 and 75 percent of the total field relief. The placement of these lines (drawn by the analyst in most cases) was based on the previously determined maximum relief of the field and the configuration of the existing contour lines from the topographic map.

The distance to the nearest stream was defined as the shortest distance water would travel after leaving the edge of the field en route to the nearest blue line stream. The analyst made this determination based on topographic analysis of the USGS map.

The maximum slope-length and the contour length are both used in the determination of the slope-length factor. The distance to the nearest stream is used in the computation of the delivery ratio. The utilization of these measurements will be discussed in the next section.

Determination of the hydraulic length, which was defined as the distance from the point where water entered a blue line stream to the mouth of that watershed where it entered a major drainage unit, was done by counting quarter-mile stream segments which were marked on a stream segmentation overlay created from the topographic maps. The major drainage units in this study were the Rappahanock and Potomac Rivers. The hydraulic length was not used in the determination of the PSL but was requested by the Virginia SWCC to be included in the final data set.

Area Measurements

The area of each individual field unit and the total area of each watershed were determined. Each field area was digitized on EPIC's Image Analysis System and then converted to acres and square feet by the computer.

The field area was used in determination of the slope-length factor for use in developing the PSL. The watershed area was used to calculate the Pollution Density Index (PDI). For those watersheds that extended into other counties, only the portion within the Westmoreland County boundary was used for the PDI calculation.

Determination of the PSL

As mentioned earlier the PSL is computed by equation (1). The rainfall

$$PSL = R \cdot K \cdot LS \cdot DR \qquad (1)$$

factor "R" was obtained from a USDA publication (USDA, 1978), and is a reflection of average rainfall recorded in Westmoreland County from many years of historical records. It is a constant for this study but will change in other geographic locations.

The average soil erodibility "K" was determined as previously described by weighting the erodibility values for each of the soils present on a site according to the proportion of the area they occupy.

The slope-length factor (LS) was the most difficult to compute. The first step was to compute the average slope for the field site. The method used is called the "contour-length method" from Williams and Berndt (1976). The equation for the average slope took the form of:

$$s = .25Z\ (C.25 + C.50 + C.75)/A \qquad (3)$$

where: C.25 is the length of relief contour at 25% of the field relief (ft); C.50 is the length of relief contour at 50% of the field relief (ft); C.75 is length of relief contour at 75% of the field relief (ft); and A is the total area of field (square ft).

The result of the average slope was then used to determine the LS factor, whose equation took the form of:

$$LS = (X/72.6)^m\ (0.43 + 0.3s + 0.0432s)\ /\ 6.613 \qquad (4)$$

where: X is the maximum slope length, s is the average field slope, and m is an exponential value dependent on the value of s. The LS equation was adapted from USDA (1978).

The delivery ratio (DR) was computed from the following equation:

$$DR = 10\ R/L \qquad (5)$$

where: R is the relief to the nearest blue line stream and L is the distance to the nearest blue line stream. This equation was adapted from Williams and Berndt (1972).

The maximum value for DR was restricted to "1." This was done because a value larger than 1 mathematically indicates that more than 100% of the available suspended materials was being delivered to the blue line stream.

The final PSL factor was then used to rank all the field sites as to their potential to be non-point source pollution problem areas.

The computation of the PDI value for each watershed was then computed by multiplying the PSL's for the fields contained in the watershed by the areal measurement of each corresponding field. The sum of all these values is called the Total Loading Potential (TLP) for each watershed.

$$TLP = \sum \text{[PSL (for each field "y" in watershed x)} \cdot \qquad (6)$$

$$\text{(area in acres of field "y")]}$$

The TLP is then divided by the total area (in acres) within the watershed to get the PDI.

$$PDI = TLP\ (\text{watershed } x) /\ \text{Total Area}\ (\text{watershed } x) \qquad (7)$$

<u>Computer Data Analysis</u>

All of the data collected was entered into EPIC's Calma Graphic Interactive Image Analysis System. The system is based on a Data General Eclipse S230 minicomputer.

Macros were developed to automatically process the data using the above equations to produce the final PSL. The only manual calculation required was the exponentiation in determining the LS factor because the system was unable to do exponentiation.

After all the PSL's had been generated, a communications tape containing all the raw data (24 pieces of data for each field unit) and the final PSL for each field was created. The tape was then transferred to the VAX 11/780 minicomputer at the Chesapeake Bay Program office in Annapolis, Maryland to complete the final statistical analysis.

A special FORTRAN package was developed for this transfer of information so that SAS statistical operations could be done on the data. The final ranking of the field sites was then performed by a SAS sorting operation, which sorted the field sites by their PSL's and created a hierarchical listing based upon their PSL value. Five equal-sized groupings were created. Those having the highest PSL values were put in Category "0," those with the lowest in Category "4." The computations to determine the PDI values for each of the watersheds were also performed on the VAX. Other basic statistics calculated were the mean value of the PSL's, standard deviation, variance, and maximum and minimum values.

Final data products included a printout of all fields sorted by ranking factor, quad sheet and watershed; a printout of 24 data elements for a field unit listed by watershed; and USGS 7.5-minute quadrangles with field unit and watershed overlays.

RESULTS

Analysis of Westmoreland County, including computer analysis of the data, took 5 months to complete. A total of 1,808 field units were analyzed in detail, comprising 44,743 acres of agricultural land. Of this, 42,529 acres (95.1%) were in cropland and 2,205 acres (4.9%) were in pasture. The average field unit size was 25 acres. The range of the PSL values for the field sites was 0.02 to 234. The mean value was 18.5 with a standard deviation of 22.94.

Only 26 feedlots were found in Westmoreland County in this study. They all appear to be either hog or cattle feeding operations. The small relative amount of pasture land in the county appears to correlate with the existence of few livestock feeding operations.

Final data products included a printout of all fields sorted by ranking factor, quad sheet and watershed; a printout of 24 data elements for a field unit listed by watershed; and USGS 7.5 minute quadrangles with field unit and watershed overlays.

The range of PDI values for the 49 watersheds was from 0.1 to 82.8, with a mean value of 8.5 and standard deviation of 14.06. The watersheds with the greatest PDI's were Little Meadow Run (82.8) and Bristol Mine Run (56.6). The percentage of agricultural activity within these two watersheds was 15 and 45%, respectively. Table 1 lists the ten watersheds with the highest PDI values.

Table 1: Ten watersheds in Westmoreland County study area with the highest PDI values.

Watershed	PDI	TLP	% Agric.	Area (acres)
Little Meadow Run	82.8	19,858.6	15	239.7
Bristol Mine Run	56.6	22,380.7	45	395.7
Mill Run	26.1	8,883.4	32	340.2
M	17.0	357.3	43	21.0
R	15.3	16,334.6	77	1,067.9
P	14.9	16,132.6	12	1,081.3
Y	14.9	2,677.6	68	179.4
Currioman Creek	12.6	22,096.6	15	1,756.3
D	12.3	2,270.5	18	184.5
Smarts Creek	11.5	6,190.2	41	538.8
X	26.4	11,718.2	37	580.5
S.E.	7.6	2,722.6	7	173.6

These watersheds have an average land area of only 580.5 acres, which is much less than the 3,150.3 acre average for the study area. The average percent agricultural land for the ten watersheds was 37%, which is slightly higher than the 34% calculated for all watersheds in the study area.

The data in Table 2, which lists the ten watersheds in the study area with the greatest percentage of agricultural land, suggests that the percentage of agricultural land alone is not the controlling factor resulting in high PDI values. The average percentage (67%) of agricultural

Table 2: Ten watersheds in Westmoreland County study area with the greatest percentage of land in agricultural land use.

Watershed	% Agric.	PDI	Area (acres)	TLP
A	86	6.3	67.5	422.7
Z	83	0.9	136.4	118.3
R	77	15.3	1,067.9	16,334.6
L	70	1.5	142.8	218.3
Y	68	14.9	179.4	2,677.6
K	63	0.4	30.3	12.8
S	59	1.9	37.3	71.7
N	55	3.3	120.2	392.6
Totusky Creek	54	6.5	2,254.5	14,657.1
Wilkerson Creek	55	5.0	155.9	778.7
X	67	5.6	419.2	3,568.4
S.E.	4	1.7	225.7	2,006.9

land for these ten watersheds is significantly higher than that of all the watersheds (34%), but their average PDI is only 5.6. This is below the average of 8.5 for the study area and far below the 26.4 average for the ten highest PDI values.

The relative size of a watershed also does not correlate well with the PDI value. Table 3 lists the ten watersheds in the study area with the greatest total acreage along with percent agricultural land and PDI values. It is interesting to note that the average PDI value for these watersheds (5.6) is the same as those with the greatest percent agricultural land. In addition, the average percent agricultural land

Table 3: Ten watersheds in Westmoreland County study area with the largest area in acres.

Watershed	Area (acres)	% Agric.	PDI	TLP
Nomini Creek	35,238.1	36	9.5	333,454.0
Cat Point Creek	20,402.6	27	8.8	179,392.2
Yeocomico River	13,908.9	31	5.0	69,747.2
Popes Creek	11,084.6	19	8.8	98,071.4
Mattox Creek	11,041.4	19	5.5	60,558.0
Lower Machodoc Creek	11,000.2	30	3.9	42,770.2
Peedee Creek	7,889.1	34	6.4	50,506.0
Monroe Creek	5,336.2	10	0.8	4,215.0
Potomac River	3,452.9	16	4.4	15,266.2
Bonum Creek	3,422.3	37	3.2	11,002.2
X	12,277.6	26	5.6	86,498.0
S.E.	30,26.1	3	0.9	31,877.5

in these large watersheds is only 26%, which is below the 34% average for the entire study area.

Table 4, which lists the ten watersheds with largest TLP, may be the most useful. The average area of these watersheds (11,609 acres) tended to be much larger than the average (3,150 acres) for the study

Table 4: Ten watersheds in Westmoreland County study area with the greatest TLP values.

Watershed	TLP	Area (acres)	% Agric.	PDI
Nomini Creek	333,454.0	35,238.1	36	9.5
Cat Point Creek	179,392.2	20,402.6	27	8.8
Popes Creek	98,071.4	11,084.6	19	8.8
Yeocomico River	69,747.2	13,908.9	31	5.0
Mattox Creek	60,558.0	11,041.4	19	5.5
Peedee Creek	50,506.0	7,889.1	34	6.4
Lower Machodoc Creek	42,770.2	11,000.2	30	3.9
Troy Creek	32,388.8	3,372.7	25	9.6
Bristol Mine Run	22,380.7	395.7	45	56.6
Currioman Creek	22,096.6	1,756.3	15	12.6
X	91,136.5	11,609.0	28	12.7
S.E.	30,710.6	3,242.8	3	5.0

area, while the percent of agricultural land (28%) was slightly less than the 34% for the entire study area. Most significantly, the average PDI (12.7) is well above the average for the study area (8.5). By using TLP to extract the problem watersheds, the watersheds with a high PDI value but a very small area are eliminated. Because of their small average area (x = 580.5 acres), the contribution of the ten watersheds with the highest PDI values as a non-point pollution source would probably be minimal.

CONCLUSIONS

Through the development of this project, much progress has been made towards developing a useful management tool for control of non-point source agricultural pollution. Further research is needed to determine which parameters are most useful and which methods of obtaining those parameters are most effective. In addition, the methodologies involved need to be streamlined.

Overall it is felt that most of the raw data collected is valid, and that it can serve as a useful tool for resource management decision making. The utilization of conservation tillage and other BMP's may have reduced the potential of some of those sites which had high PSL's. A rigorous field investigation was warranted in order to determine how best to interpret the data collected, but was not possible due to funding limitations. Thus, a focus of future studies should be to determine whether or not the study identifies known problem areas, and to determine if BMP's are in place at those sites with high PSL's.

One of the parameters which was measured and not considered in the computation of the PSL was the hydraulic length. This value is especially important in reflecting the potential for sediment or nutrient fallout prior to entering the primary receiving water body. It is felt that this value should be integrated into the evaluation process to create a more meaningful statistic to represent potential loadings.

The use of a Geographic Information System and digitized data bases would be the greatest asset in streamlining the methodology. This would expedite extraction and manipulation of specific data elements and would also provide the ability to study best-fit alternatives when weighting the individual parameters to determine which combination of factors most accurately reflects real world situations.

While the land use evaluations made with the aerial photography was a strong point of the study, the role and information provided by the aerial photos can be improved and enhanced. Only a small fraction of the data available from the photos was used. Many bits of information such as the existence of erosion gullies, barren areas, poor crop growth, tillage practices, and buffer zones could have been extracted. These factors should be incorporated into future studies.

Reevaluation of the methodologies used and the interpretation of the results is continuing at EPIC, Virginia Polytechnic Institute and State University and the Virginia SWCC. This technique will soon be available to aid decision makers in targeting their areas of greatest need and evaluating their distribution of resource management funds.

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Chesapeake Bay Research Conference
Williamsburg, Virginia, March 20-21, 1986

ACCURACY ANALYSIS OF SPATIAL DATA BASE USED IN ASSESSING POLLUTION

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ABSTRACT

A digital map data base was prepared for the Northern Neck Soil and Water Conservation District for identifying areas of high potential of sediment loading to the Chesapeake Bay. U.S. Soil Conservation Service soil maps, land use information and U.S.G.S. topographic maps were used to generate the digital maps.

A raster data base was obtained from the maps and areas of high nonpoint source pollution potential were predicted based on that data, using the universal soil loss equation with a delivery ratio. This paper examines the problem of choosing the size of the cell which is used in the calculations and the effect of that choice on the final model prediction.

In order to study the cell size effects, the data from a portion of the Northern Neck Soil and Water conservation District was first digitized into several different sized cells, then the model was used with the different sets of data. As the size of the cell was increased the detail of the parameter which was being used in the model was lost and the model's ability to detect small problem areas was lost. This loss of sensativity resulted in a smoothing of the data, and a prediction of pollution potential based on average values rather than the more precise values possible through a finer system of cells.

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RIPARIAN AREAS AND WATER MANAGEMENT TO CONTROL NONPOINT POLLUTANTS

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ABSTRACT

Drainage water coming from agricultural land is likely to have higher nutrient concentrations than drainage water from forested areas. This is true even when best management practices are being utilized on the land. Timbered riparian areas which are present between many agricultural fields and streams in the Atlantic Coastal Plain are very effective in removing nutrients and sediment from the drainage water. Improved drainage systems are necessary for agricultural production in many Coastal Plain soils. The improvement of drainage can result in an increase in nitrogen efflux in the drainage water. However controlled drainage in systems with good subsurface drainage can be utilized to minimize nitrogen losses in the drainage water.

INTRODUCTION

Because of rainfall in excess of evapotranspiration, water drains via surface or subsurface flow from all land on the Atlantic Coast. This water always contains some nitrogen, phosphorus and sediment. Even when present in low concentrations in drainage water, nitrogen and phosphorus can contribute to water quality problems in receiving steams and estuaries. For example, a forested watershed is generally considered to represent the minimum loss of nutrients to drainage water. Yet it has been estimated that 47% of both the nitrogen and phosphorus input to the Chowan River in North Carolina comes from forested areas. This river has experienced severe water quality problems in the past few years in the form of algae blooms caused by excess nutrients. Thus any increase in nutrient concentrations can be a potential problem for downsteam areas.

Our work has concentrated on the contribution of agriculture to nonpoint sources of nutrients and sediment. We first attempted to quantify the amounts of nutrients leaving cultivated fields and determining what factors controlled these losses. Recent efforts have focused upon developing methods to minimize contributions of nutrients to drainage water utilizing methods which are compatible with sustained high agricultural production. This paper summarizes some of these findings.

WATER QUALITY IN THE PIEDMONT

The prospects for improving water quality in the Chesapeake by improving the quality of water draining from agricultural areas in the Virginia Piedmont are not good. This conclusion is based on data from this area which indicate that drainage water from agricultural watersheds contains only slightly more nutrients than drainage water from predominantly forested areas (Table 1). These data are similar to data obtained in the North Carolina Piedmont.

Table 1. Nitrogen and phosphorus concentrations in the drainage water of three agricultural and four sivicultural Virginia Piedmont Watersheds. (Data from Humenik et al., 1980)

	Forested Watershed	Agricultural Watershed
% Forested	86.0	55.0
% Agricultural	13.0	43.0
Nitrate-Nitrogen (mg/L)	0.03	0.12
Total Nitrogen (mg/L)	1.43	1.50
Total Phosphorus (mg/L)	0.07	0.18

As will be discussed in more detail later for Coastal Plain drainage water, nutrient concentrations in drainage water at the agricultural fields' edge were probably much higher than concentrations observed in the streams. Another factor minimizing the importance of the Piedontagricultural area to water quality in the Bay is assimulative capacity of the streams between the two areas. However the use of Best Management Practices such as minimum tillage should be encouraged for upstream water quality.

WATER QUALITY IN THE COASTAL PLAIN

Effect of Riparian Areas

The potential contribution of agricultural drainage water to water quality problems in the Bay is much greater than the threat from the Piedmont. Several factors are responsible. One of the most significant factors is that the concentration of nitrogen and phosphorus in the drainage water from agricultural fields tends to be higher in the

Coastal Plain than in the Piedmont, although erosion is much less of a problem in the Coastal Plain. A much higher percentage of the land area in the Costal Plain has been converted to agricultural use with the potential for even greater conversion. Another factor is that the drainage water has considerably less distance to travel before reaching the Bay than that from the Piedmont and Mountains.

There are many factors which influence the magnitude of the loss of fertilizer nutrients in drainage water from the Coastal Plain agricultural areas. Among these are crops grown, fertilizer rate, soil type, drainage system design and management and other cultural practices. Losses typical of those measured in agricultural Coastal Plain watersheds are shown in Table 2. The nitrogen losses, in particular, are larger than those reported for the Piedmont. One very significant factor which should be noted is that only 48% of the most intensively cropped watershed is cultivated.

Table 2. Annual losses (three year average) of nitrogen and phosphorus in drainage water from two Coastal Plain watersheds in North Carolina. (Data from Jacobs and Gilliam, 1985)

Watershed	Size	% Cropped	Nitrate N	Total N	Total P
			------lb/ac/yr-------		
Well drained	3,000 acres	48	2.2	4.0	0.14
Poorly drained	17,000 acres	25	0.4	2.2	0.21

We also measured losses of nitrogen and phosphorus from representative fields within each watershed. The nutrient losses in drainage water at the fields' edge were much larger than the losses from the watershed. For example, in the well drained watershed, the average loss of nitrate-nitrogen in drainage water at the fields' edge was 29 lb/ac/yr. The loss of nitrate-nitrogen from the entire watershed was only 4.5 lb/cultivated acre. Thus there was a loss of approximately 25 lb/ac/yr of nitrate-nitrogen between the fields' edge and the exit from the watershed. This observation has strong implications for future resource management to protect the quality of water in our coastal water resources. Thus let's examine what happened to the 25 lbs of nitrogen which was lost from the drainage water.

The land use pattern in the watershed is very common in all Atlantic Coastal Plain areas. Agricultural land is present on the better drained uplands and forested areas are on the more poorly drained bottomlands. The drainage water from the cultivated fields must pass through the wooded areas via surface or subsurface flow to reach the steams which drain the area. We found that subsurface drainage water from a field lost essentially all of its nitrogen as it moved through a heavily vegetated riparian bufer area as narrow as 50 ft. This is illustrated by the data in Table 3 which were obtained by monitoring the shallow ground water as it flowed from agricultural fields into natural or improved drainage ways. The removal of nitrogen from subsurface drainage water in the riparian areas is almost totally a result of denitrification. Plant uptake accounts for only a small percentage of the removal.

Even with improved drainage some removal of nitrogen is seen as the ground water moves from the field into the surface waters (Table 3). The tile line effluent concentration of nitrate-nitrogen is lower than that in the shallow ground water drained by the tile line. Most of the

Table 3. Mean nitrate-nitrogen concentration across well transects from agricultural fields to natural stream and improved V-ditch. (Data from Jacobs and Gilliam,1985)

	Concentration mg/L
Natural Drainage	
Field	7.6 ± 1.5(st. dev.)
Field Edge	5.9 ± 3.7
Stream Edge	0.2 ± 0.5
Stream	0.9 ± 0.8
Improved V-Ditch + Tile	
Field	16.2 ± 3.3
Tile Line	10.2 ± 3.5
V-Ditch	6.5 ± 2.2

shallow ground water movement is restricted to lateral flow with little deep seepage loss of nitrogen from the field. The lower nitrate-nitrogen in the tile effluent is from denitrification near the entry point of the tile line. More denitrification occurs as the water flows into the ditch via subsurface flow below the tile line. This results in the water in the ditch having a lower nitrogen concentration than the water in the tile line or ground water. The lower nitrate concentration in the ditch could also result from presence of surface runoff in the ditch. However this field had very little surface runoff and concentration measurements were made after surface runoff had been flushed from the system.

We have also used 137-Cs to determine sediment deposition over the past 20-25 years in riparian areas within an agricultural watershed(Cooper et al.,1985). The amount of sediment which was deposited within the watershed was compared with the data of Humenik et al.(1983) who had measured the loss of sediment from the watershed in a 208 study. Approximately 88% of the sediment which had moved from the agricultural fields and upland forests during the 20 year period remained in the watershed. About 80% of the sediment deposited within the watershed was deposited in riparian areas above the floodplain swamp. Over 50% of the sediment was deposited within 100 meters of the exit location from the field. The total amount of sediment deposited within the watershed studied most intensively is given in Table 4.

Table 4. Sediment accumulation and distribution during the past 20-25 years in Cypress Creek watershed.(Data from Cooper et al., 1985.)

Location	Sed. Depth cm	Total Weight Mg	Sed. Distrib. %	Silt (%)	Clay (%)
Forest Edge	15-50	2800	19	19	6
First Order Stream	5-20	2800	19	57	11
Higher Order Streams	5-15	5900	40	45	15
Flood Plain Swamp	0-5	3300	22	38	24
Total		14800	100		

Table 5. Total phosphorus distribution in the 137-Cs sediments in Cypress Creek watershed. (Data from Cooper et al., 1985.)

Location	Total P kg	P Distribution %
Forest Edge	500	6
First Order Stream	1000	13
Higher Order Streams	3000	37
Flood Plain Swamp	3500	44
Total	8000	100

The total phosphorus present in the 137-Cs sediments is shown in Table 5. Phosphorus appears to be much more mobile through the watershed than sediment. In contrast to the sediment distribution, most of the total phosphorus has moved into the lower parts of the watershed. Over 40% of the total phosphorus in the sediments deposited in the last 20-25 years is present on the floodplain swamp. Although the amounts of total sediment deposited on the floodplain swamp is low, these sediments contain a much higher % of clay sized materials. The clay is largely responsible for the trapping of the phosphorus within the watershed. We estimated that about 50% of the phosphorus which had been removed in drainage water from agricultural fields in this watershed in the last 20-25 years remained in the forested areas of the watershed.

Effect of Agricultural Drainage System Design and Management

It is not always possible to pass agricultural drainage water over or through riparian areas. This is particularily true in the very flat land near the coast. Many of the soils are very poorly drained and require improved drainage systems for agricultural production. Design and management of the drainage system can influence the nutrient content of the drainage water as well as time distribution of the outflows from essentially all land where improved drainage is necessary for agricultural production. In this paper, drainage system design refers to whether a field is largely surface or subsurface drained as well as spacing and depth of improved subsurface drainage system. Controlled drainage refers to restricting the flow of subsurface drains by the use of some mechanical structure.

Th proportion of the drainage water which leaves agricultural fields via surface or subsurface drainage has a large influence upon the potential pollutants carried by the water (Baker and Johnson, 1976; Gilliam and Skaggs, 1985). Surface runoff carries more sediments, pesticides and phosphorus than subsurface flows. A higher proportion of subsurface flow is accompanied by a greater loss of nitrate-nitrogen and generally a greater loss of total nitrogen. The effects on nitrogen and phosphorus losses are illustrated by data in Table 6 from the Coastal Plain of North Carolina.

Table 6. Effect of type of drainage on nitrogen and phosphorus efflux from three similarly cropped soils in the North Carolina Coastal Plain.

Nutrient	Poor Subsurface	Intermediate	Good Subsurface
	Drainage Type		
	lb/ac/yr		
Nitrate-N	3.7	15.7	32.4
Total-N	13.6	20.0	42.1
Total-P	0.5	0.3	0.2

The three fields from which the data in Table 6 were collected were in a corn-soybean rotation and cultural practices were similar. The field with poor subsurface drainage contained ditches spaced approximately 300 feet apart, but the internal conductivity was so poor that most of the drainage water was removed via surface runoff. The intermediate field had a similar drainage system but this field had a sand layer present at a depth of three feet. This sand layer improved the drainage to the open ditches, but still was not as well drained as the field with two equally spaced drain tubes installed parallel to the open ditches. In the field with good subsurface drainage, nearly all drainage water reached the open collecter ditches via subsurface flow. The large effect that the type of drainage has upon nutrient outflows has significant implications for the design and management of drainage systems.

Approximately half of the drainage water from agricultural land in North Carolina and the Eastern U. S. occurs during the period of December through March. In many cropping systems, drainage during this period is not agriculturally critical, so drainage water can be managed to minimize nutrient outflows without influencing agricultural production. Our initial experiments on controlled drainage were designed to control water only during the winter, but we now know that controlled drainage throughout the year offers potential for increased agricultural production as well as providing environmental benefits.

In the poorly drained flat soils of the Lower Coastal Plain, flashboard risers in collection ditches have been used to control water tables in fields up to 100 acres in size. These poorly drained soils have sufficient organic matter in the top five feet to cause reducing conditions below the water table. Water passing through this zone on the way to an outlet has essentially all of the nitrate removed from it by denitrification and phosphorus by absorption.

Because of the higher water table maintained with controlled drainage, surface runoff will be increased. Since surface runoff contains a higher concentration of phosphorus than subsurface flow, an increase in phosphorus losses would be expected under controlled drainage. The data in Table 6 are a good indication of the potential effects of controlled drainage on nitrogen and phosphorus effluxes from a naturally poorly drained Lower Coastal Plain soil with a good subsurface drainage system installed on it. The good subsurface drainage represents the conditions which exist under no control and the poor subsurface drainage represents maximum control throughout the year. It would be expected that actual control conditions would be between these two extremes with regard to nitrogen and phosphorus losses to surface water.

Deal et al. (1985) used nutrient losses measured in several experiments under different types of drainage in conjunction with the water management model DRAINMOD (Skaggs, 1978) to predict nutrient losses for six soils for hypothetical field conditions. The soils modeled were poorly drained to very poorly drained and all had a high water table during much of the year unless improved drainage systems were installed. The values given in Table 7 are the average values computed for a 20 year period for two soils under one drainage system(good surface drainage-good subsurface drainage).

Table 7. Prediction of annual nutrient efflux from two naturally poorly drained soils under controlled and conventional water management considering seepage losses.

Soil	Conventional Drainage Nitrate N	Total N	Total P	Controlled Drainage Nitrate N	Total N	Total P
	lb/ac/yr					
Portsmouth	33.6	38.4	0.07	21.4	25.4	0.13
Wasda	26.1	31.1	0.17	17.7	22.5	0.25

Control of field drainage does result in a very significant decrease in nitrogen efflux from agricultural fields. Under the conditions simulated, the control did result in an increase in the phosphorus efflux. It should be emphasized that the above simulations are for fields with good subsurface drainage. Controlled drainage has little effect in fields where the predominant drainage is surface runoff. The above data are also for particular management conditions (see Deal et al., 1985 for details) and other management conditions will yield different results.

We have also investigated the use of controlled drainage in a channelized stream draining approximately 10,000 acres in a cooperative project with USDA-ARS. The control of flow in the stream resulted in an approximate decrease of 33% in the nitrate concentration in the water leaving the watershed and had no effect upon the phosphorus concentration.

Structures to control drainage do represent a cost to agricultural producers. However, the same structures which can be utilized to improve drainage water quality can also be used to increase water use efficiency and improve crop yields. The system also has the potential to be used for subirrigation by pumping water into the ditch and letting the drainage system distribute the water throughout the field. These combinations have proven so attractive to farmers in North Carolina that approximately 21 subirrigation systems and 100 controlled drainage systems (representing 25,000 acres) have been installed in the past 2-3 years. More than 100 new systems are currently in the process of being evaluated for potential installation. Futhermore, the North Carolina Legislature allocated 2.2 million dollars for 1985 to help cost/share implimentation of BMP's in "nurient sensitive watersheds". One designated BMP is structures to provide controlled agricultural drainage.

CONCLUSIONS

Agricultural drainage water is likely to always have a higher nutrient concentration than drainage water from well managed forests. However, there are several water management techniques available to minimize the movement of these nutrients into major bodies of water where water quality problems may occur. Riparian areas adjacent to fields serve as very effective filters and their utilization should be encouraged where practical. Another water management technique which has proved very practical where riparian areas cannot be used is controlled drainage. Agricultural water management practices can be used to reduce nutrients reaching nutrient sensitive waters but these practices are site specific.

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Agricultural Drainage And Water Quality Relationships In The Chesapeake Bay

by

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ABSTRACT

Surface water quality was monitored in agricultural watersheds with poorly drained soils in western Kent County Delaware for 33 months. Drainage ditches were being constructed on one of the watersheds monitored while another watershed had drainage ditches constructed approximately 10 years ago and a third watershed had no drainage construction.

Drainage construction increased turbidity, suspended solids, dissolved solids, total phosphorus and ortho phosphorus concentrations. Ammonia concentrations were highest in early spring and lowest in the winter. Once drainage construction was completed, concentrations of turbidity, dissolved solids, organic nitrogen, ortho phosphorus and total phosphorus decreased because of reduced stream bank erosion and stabilization of the stream channel.

INTRODUCTION

Parts of Kent and Sussex counties, Delaware, require drainage to have productive agricultural soils. Drainage has been installed on the Delmarva Peninsula since colonial days.

Drainage projects in Delaware have raised the question of what effect drainage has on water quality. Some people believe it has a detrimental effect on water quality, while others believe that flooding of poorly drained soils contributes to sheet erosion and poor water quality by soil particle flotation. In drained soils, particle flotation will not occur and water quality should improve. The only time water quality would be degraded is during the construction phase.

This paper reports on a project which was initiated in 1982 to evaluate the impact of agricultural drainage on water quality.

EXPERIMENTAL PROCEDURES

Three watersheds in western Kent County, which are in the Choptank River basin, were monitored in 1982 at Routes 269, 10 and 103. Drainage construciton was underway in the watershed at Route 269 in 1982. The watershed is located on Sangston Prong, which is a tributary of Gravelly Branch. The Route 10 watershed, located near Sandtown on the Sandtown Branch, has no drainage construction. The third watershed, located on a tributary on the Tappahanna Ditch on Route 103, had drainage ditches constructed around 1970.

Drainage construction was completed on the Route 269 watershed in early 1983. It was decided to move the monitoring station at Route 269 to a different watershed on Route 266 where construction was occurring on several of its tributaries. The watershed on Route 266 was selected for monitoring because drainage construction would continue until the project is completed. It is located on White Marsh Branch which is also a tributary of Gravelly Branch.

The size of the watersheds vary from 308 to 1257 ha (Table 1). Over 50 percent of the land use in each watershed is cropland. All of the cropland is planted either to corn or soybeans and no livestock or poultry are located in the watersheds. Over 75 percent of the soils on each watershed are classified as poorly drained. The major poorly drained soil types are Pocomoke sandy loam and Fallsington sandy loam. Watershed slopes are less than one percent.

Automatic water samplers were installed on the Routes 269, 10 and 103 watersheds in March of 1982. In October of 1983, the automatic sampler at Route 269 was moved to the Route 266 watershed. The samplers were used only to collect runoff samples and were triggered by an increase in the water level in the drainage ditch. Once the samplers were set in operation, 24 samples were collected at one-hour intervals, or if the water level in the drainage ditch decreased in less than 24 hours to below the elevation when the sampler began operation, the sampler automatically shut off before 24 samples were taken. The samplers were set so they would start taking samples when the water level had risen approximately 3.7 cm above baseflow conditions. Samples were collected from March to September in 1982 and from March, 1983 until July, 1985.

Grab samples were collected in polyethylene bottles on Routes 269, 10 and 103 watersheds at seven to fourteen day intervals from March to September 1982 and from March, 1983 until July, 1985. When the monitoring station at Route 269 was moved to Route 266 in October, 1983, grab sampling was continued on the Route 269 watershed. Also, grab sample collection was started at Routes 265A, 265B and 265C stations. All three of these locations are on sub-branches of the main Route 266 tributary. Routes 265B and 265C tributaries had drainage construction and clearing taking place in October 1983 and Route 265A had no drainage construction.

Table 1. Land Use of Watersheds

Watershed	Drainage	Area ha	Forest ha	Cropland ha	Urban ha
Route 269	Under Construction	545	115	430	0
Route 10	None	682	299	382	1
Route 103	Completed	308	93	210	5
Route 266	Under Construction	1257	270	977	10

All storm runoff and baseflow samples were analyzed for pH, ammonia, nitrate-nitrite nitrogen, organic nitrogen, ortho phosphorus, total phosphorus, suspended solids, turbidity and total dissolved solids by procedures outlined in Standard Methods (APHA, 1980).

RESULTS AND DISCUSSION

A total of 26 grab samples were taken from each watershed at Routes 269, 10 and 103 in 1982 and a total of 43 grab samples were taken in 1983. Also a total of 13 grab samples were taken at Route 266 and 11 samples were taken at Routes 265A, 265B and 265C from October to December in 1983. A total of 48 grab samples were taken in 1984 from all watersheds. Storm samples were collected with the automatic samplers in 1982, 1983, 1984 and 1985 at Routes 269, 10, 103 and 266. For each year a large number of storms were sampled. Some of the concentration data for the watersheds are summarized in Tables 2 to 8.

Table 2. Median Concentrations Of Chemical Parameters For Baseflow Samples For Route 269.

Date	No. of Samples	NH_3	NO_3	Org N	Ortho P (mg/L)	Total P	SS	TDS	Turbidity[a]
1982									
Spring	11	.14	1.60	0.85	.034	.13	23	181	9
Summer	12	<.05	3.67	.63	.016	.064	1	83	<2
1983									
Spring	13	<.05	3.63	.49	.027	.050	4	102	12
Summer	13	<.05	3.44	.28	<.010	.018	1	84	4
Fall	14	<.05	3.16	.16	.012	.013	5	113	5
1984									
Winter	10	<.05	3.65	.08	.016	.027	4	93	4
Spring	13	.06	3.55	.59	.014	.030	4	156	4
Summer	13	.11	3.44	.26	.023	.050	1	102	<2
Fall	12	.08	4.33	.31	.015	.042	1	116	<2

[a] Turbidity expressed as FTU.

Table 3. Median Concentrations Of Chemical Parameters For Baseflow Samples For Route 10.

Date	No. of Samples	NH_3	NO_3	Org N	Ortho P	Total P	SS	TDS	Turbidity[a]
					(mg/L)				
<u>1982</u>									
Spring	11	.10	1.09	.91	0.49	.10	8	111	9
Summer	12	<.05	1.76	.87	.11	.13	3	83	5
<u>1983</u>									
Spring	13	<.07	1.50	.64	.034	.079	7	92	19
Summer	13	<.05	1.60	.34	<.049	.056	2	87	8
Fall	14	<.08	1.25	.16	.049	.079	7	110	12
<u>1984</u>									
Winter	10	<.05	1.70	.16	.034	.045	6	92	13
Spring	13	.11	1.57	.79	.077	.081	8	109	15
Summer	13	.09	1.78	.19	.061	.097	1	89	6
Fall	12	.08	1.76	.32	.11	.14	3	108	8

[a] Turbidity expressed as FTU.

Table 4. Median Concentrations Of Chemical Parameters For Baseflow Samples For Route 266.

Date	No. of Samples	NH_3	NO_3	Org N	Ortho P	Total P	SS	TDS	Turbidity[a]
					(mg/L)				
<u>1982</u>									
Fall	11	<.05	1.42	.29	.038	.079	23	168	59
<u>1984</u>									
Winter	10	<.05	2.22	.10	.034	.045	12	91	21
Spring	13	<.05	2.11	.53	.049	.067	12	138	12
Summer	13	.11	1.59	.27	.042	.058	2	81	18
Fall	12	.08	1.95	.45	.042	.079	2	75	11

[a] Turbidity expressed as FTU.

Table 5. Median Concentrations Of Chemical Parameters For Baseflow Samples For Route 103.

Date	No. of Samples	NH_3	NO_3	Org N	Ortho P (mg/L)	Total P	SS	TDS	Turbidity[a]
<u>1982</u>									
Spring	11	.06	1.23	.64	.064	.073	11	115	9
Summer	12	<.05	.23	.51	.057	.11	9	101	7
<u>1983</u>									
Spring	13	<.05	1.60	.57	.049	.085	4	134	11
Summer	13	<.05	.09	.15	.019	.039	3	103	9
Fall	14	<.05	.22	.21	.023	.027	7	134	9
<u>1984</u>									
Winter	10	<.05	1.75	<.05	.041	.045	3	108	7
Spring	13	.12	1.15	.52	.034	.059	9	120	11
Summer	13	.07	.24	.38	.046	.076	3	106	14
Fall	12	.08	.25	.26	.046	.058	4	136	6

[a] Turbidity expressed as FTU.

Table 6. Median Concentrations Of Chemical Parameters For Baseflow Samples For Route 265A.

Date	No. of Samples	NH_3	NO_3	Org N	Ortho P (mg/L)	Total P	SS	TDS	Turbidity[a]
<u>1983</u>									
Fall	11	.13	2.56	.91	.16	.25	91	218	51
<u>1984</u>									
Winter	10	<.05	4.46	.10	.038	.067	2	101	9
Spring	13	<.05	3.29	.66	.086	.11	7	117	6
Summer	13	.13	1.86	.52	.080	.11	3	113	7
Fall	12	1.05	.29	.52	.14	.16	45	143	32

[a] Turbidity expressed as FTU.

Table 7. Median Concentrations Of Chemical Parameters For Baseflow Samples For Route 265B.

Date	No. of Samples	NH_3	NO_3	Org N	Ortho P	Total P	SS	TDS	Turbidity[a]
					(mg/L)				
1983									
Fall	11	.06	1.02	.48	.049	.097	73	183	51
1984									
Winter	10	<.05	1.33	<.05	.045	.057	10	120	25
Spring	13	.17	1.47	.47	.038	.12	18	143	16
Summer	13	.14	1.02	.26	.039	.058	3	76	15
Fall	12	.08	1.13	.23	.028	.046	2	62	4

[a] Turbidity expressed as FTU.

Table 8. Median Concentrations Of Chemical Parameters For Baseflow Samples For Route 265C.

Date	No. of Samples	NH_3	NO_3	Org N	Ortho P	Total P	SS	TDS	Turbidity[a]
					(mg/L)				
1983									
Fall	11	.11	.32	.16	.023	.045	33	135	49
1984									
Winter	10	<.05	.62	<.05	.027	.050	24	112	46
Spring	12	<.05	.38	.54	.035	.062	23	104	27
Summer	13	.17	.27	.37	.025	.046	3	72	18
Fall	12	.06	.41	.35	.015	.042	2	69	7

[a] Turbidity expressed as FTU.

Ammonia: Ammonia concentrations in baseflow were generally higher in the early spring and November and December than during the summer or early fall. Temperatures are greater during the summer and early fall, so nitrification rates should be higher. Ammonia concentrations were higher in runoff than baseflow. On the Route 269 watershed for the month of April, 1982 the average baseflow ammonia concentration was 0.10 mg/L while the mean ammonia concentrations for five storms ranged from 0.13 to 0.60 mg/L. There was no trend in the storm runoff data that related ammonia concentrations to the drainage characteristics of the watershed.

Nitrates: During the spring of 1982 the median nitrate concentration in the Route 269 watershed was 1.60 mg/L, but increased to 3.67 mg/L in the summer of 1982. Median nitrate concentrations have remained above 3.0 mg/L since the summer of 1982 on Route 269. In general baseflow nitrate concentrations have ranged from 2.0 to 4.0 mg/L on the Route 269 watershed from 1983 to 1985.

Baseflow nitrate concentrations were higher in the spring than during the summer and fall in the Route 103 watershed. Nitrate concentrations were lower in the Route 103 watershed than the Routes 269, 10 and 266 watersheds. In general, storm runoff nitrate concentrations were slightly higher than baseflow nitrate concentrations. Average nitrate concentrations for two storms in the Route 103 watershed for May, 1983, were 1.87 and 1.76 mg/L while the average baseflow nitrate concentrations for May was 1.03 mg/L.

It appears drainage construction in the Route 269 watershed has increased the nitrate concentrations. Route 266, 265B and 265C have not shown increase in nitrate concentrations since drainage construction has occurred. Route 265A had high nitrate concentrations in the fall of 1983 and winter and spring of 1984. Nitrate concentrations were greatly reduced in the summer and fall of 1984 on the Route 265A watershed. High nitrogen uptake rates by the excessive algae growth in the Route 265A may have caused the decrease in nitrate concentrations.

Nitrate concentrations were lower in the Route 103 watershed in the summer and fall than the other watersheds. Because of drainage, higher crop yields are occurring on the Route 103 watershed, which results in more nitrogen being taken up by crops and less leached to the groundwater and discharged as baseflow. During the spring, nitrate concentrations were higher in the Route 103 watershed than in the Route 10 watershed where no drainage construction has occurred. Nitrification will occur more rapidly on the Route 103 watershed because of drainage construction.

Organic Nitrogen: Organic nitrogen concentrations in baseflow in the spring of 1982 were somewhat higher on the Routes 269, and 10 watersheds than the Route 103 watershed. Median concentrations for the Route 269, 10 and 103 watersheds were 0.85, 0.91 and 0.65 mg/L. For the fall of 1983, the median baseflow organic nitrogen concentrations for Routes 269, 103, 10 and 266 watersheds were 0.16, 0.43, 0.26 and 0.29 mg/L, respectively. In 1983, the drainage construction was completed in the Route 269 watershed, organic nitrogen concentrations were similar to the Route 103 watershed. Organic nitrogen concentrations were not higher in the Route 266 watershed where drainage construction is occurring than the other watersheds. Organic nitrogen concentrations were higher during storm events than in baseflow. There appears to be no large difference between the watersheds in organic nitrogen concentrations for storm events. For the spring of 1983, average organic nitrogen concentrations in storm flow were 1.20, 1.26 and 1.40 mg/L for Routes 269, 103 and 10 watersheds, respectively.

<u>Ortho and Total Phosphorus</u>: Both ortho and total phosphorus storm flow concentrations were greater than baseflow concentrations. For April, 1983, the average baseflow ortho phosphorus concentration for the Route 103 watershed was 0.044 mg/L and the storm flow concentration was 0.093 mg/L. For total phosphorus, the baseflow concentration was 0.065 mg/L and the storm flow concentration was 0.16 mg/L. Phosphorus concentrations for the Route 269 watershed in 1983 were generally lower than phosphorus concentrations for the Routes 103 and 10 watersheds. Both ortho and total phosphorus concentrations in baseflow were lower in 1983 than 1982 on the Route 269 watershed. Very little drainage construction was occurring in 1983 on the Route 269 watershed. Route 266 watershed phosphorus concentrations were higher than Route 269 watershed concentrations during the fall of 1983 and in 1984. It appears during drainage construction both ortho and total phosphorus concentrations will increase but will decrease again after construction is completed. Some undrained watersheds may have higher phosphorus concentrations than drained watersheds. The Routes 10 and 265A watersheds had the highest ortho and total phosphorus concentrations.

<u>Turbidity, Suspended Solids and Dissolved Solids</u>: Spring baseflow suspended solids and dissolved solids concentrations were higher in the Route 269 watershed than the Routes 103 and 10 watersheds in 1982, but this trend did not occur in 1983. In the spring of 1982, baseflow suspended solids and dissolved solids concentrations were almost twice as high from the Route 269 watershed where drainage construction was occurring than the Routes 103 and 10 watersheds. In 1983, baseflow turbidity, suspended solids and dissolved solids concentrations on the Route 269 watershed were similar to concentrations on the Routes 103 and 10 watershed when no construction was taking place. Baseflow turbidity and suspended solids concentrations were higher for the Routes 266, 265A, 265B, and 265C watersheds than the other three watersheds for the period of October, 1983 to March, 1984. The high turbidity, suspended solids and dissolved solids concentrations on the Route 265A watershed that has no drainage construction occurring are probably caused by excessive algae growth. The flow velocity is very low in the Route 265A watershed and during parts of the year stagnant water is found in the ditch with extremely high algae concentrations. In general, turbidity, suspended solids and dissolved solids concentrations increased for all watersheds for storm flow. The increase in suspended solids concentrations would be related to the increased ortho and total phosphorus concentrations observed during

storm events, since the sediment would carry particulate phosphorus. In most cases, phosphorus concentrations in agricultural runoff will decrease if erosion rates are reduced.

SUMMARY AND CONCLUSIONS

Agricultural watersheds were monitored in western Kent County from March to September, 1982 and from March, 1983 to July, 1985 to evaluate the influence of agricultural drainage on water quality. Some of the conclusions that may be drawn from the monitoring are:

1. Drainage construction will increase turbidity, suspended solids and dissolved solids concentrations in baseflow.

2. Total and ortho phosphorus concentrations are increased by ongoing drainage construction.

3. Turbidity, suspended solids, dissolved solids, ortho phosphorus, and total phosphorus concentrations are higher in storm flow than baseflow.

4. Ammonia concentrations are highest in early spring and lowest in the winter.

5. Once drainage construction is completed, concentrations of turbidity, suspended solids, dissolved solids, organic nitrogen, ortho phosphorus and total phosphorus will decrease in a short period of time because of reduced stream bank erosion and stabilization of the stream channel.

REFERENCES

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Chesapeake Bay Research Conference
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WATERSHED/WATER QUALITY MONITORING FOR BMP EFFECTIVENESS EVALUATION

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ABSTRACT

A watershed/water quality monitoring project has been established in Westmoreland County, Virginia, as part of the Chesapeake Bay Program to study the effectiveness of Best Management Practices on nonpoint source pollution control. The watershed is 1470 hectares in size, with land use being approximately half agricultural and half forested. The upland areas with mild slopes are generally in row crops and low-land areas are steep and forested. The soils on the watershed are predominantly sandy and loamy well drained soils.

The watershed was selected for its large proportion of cropland (50%) and lack of point source pollution discharges that could affect water quality. A main monitoring station was established at the base of the watershed in the spring of 1985. A second monitoring station, which covers approximately 214 ha. of mainly agricultural lands, was also established on uplands of the same watershed. Beginning in 1986, a water quality improvement plan will be implemented by the Division of Soil and Water Conservation to increase usage of BMPs by farmers within the watershed. The overall goal of this project is to assess the short term and long term effects of intensive agricultural BMP implementation on water quality from small agricultural watersheds.

Each watershed was instrumented with an automatic water quality sampler and a stage recorder. The sampler was set to take discreet samples over the range of the hydrograph based on changes in the stage. Precipitation was monitored on the watersheds at seven different locations either by weighing type or tipping bucket rain gauges. In addition, wind speed and direction, air temperature, soil temperature, relative humidity and evaporation data is also being collected as part of this project.

The water quality samples are analyzed for both soluble and sediment bound organic -N, ammonium -N, nitrate -N, total -N, ortho -p and total -p. Nutrients and sediment are the primary water quality parameters of concern since they are suspected as being the primary causes of declining water quality in the Chesapeake Bay. The biological monitoring program involves the collection of monthly substrate samples for the determination of protozoan diversity. Surface runoff is also monitored for pesticides on a monthly basis and during selected storm runoff events by collecting grab samples. Groundwater is also sampled for both nutrient and pesticides analyses on a monthly basis.

Preliminary results of the biological monitoring program indicates that Nomini Creek is slightly eutrophic which is indicative of a nutrient enriched environment. Only a few pesticides have been found in the limited number of samples collected to date and all concentrations were within EPA recommended limits. This paper covers rainfall, runoff and water quality data collected since the spring of 1985 from this watershed.

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HSPF MODELING OF THE PATUXENT RIVER BASIN, MARYLAND

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ABSTRACT

The Patuxent River Nonpoint Source Pollution Study is a 7-year project to assemble a comprehensive data base describing the hydrometeorologic and water-quality characteristics of the 930 sq.mi. watershed and calibrate and verify a water-quality model of the entire drainage basin. A monitoring program is underway at 13 sites in the basin. This includes six stream sites to quantify nonpoint source pollutant loads from a representative mixture of land uses. Seven single land-use sites focus on agricultural practices, including those for corn, soybeans, small grain, tobacco, and dairy production. Approximately 12 storms are being sampled annually at each site and 12 base-flow samples are being collected annually at the stream sites. All water samples are analyzed for nitrogen and phosphorus species, organic carbon, and suspended solids. A data base management system is being used for time-series and water-quality data and a geographic information system is being used to assemble and manage land-use, soils, and basin-characteristics data.

INTRODUCTION

Background

The Patuxent River Nonpoint Source Pollution Study is a joint effort by the Maryland Office of Environmental Programs (OEP) and the U.S. Geological Survey (USGS) to develop a comprehensive nonpoint source water-quality data base and a watershed model to help plan water-quality programs in the Patuxent River basin (fig. 1). It is a unique effort that is intended to demonstrate the practicality and utility of a basinwide monitoring and modeling approach. The experience gained will be valuable in future efforts to study similar water-quality problems in other basins. Robert Summers and Gary Fisher are co-project chiefs for OEP and USGS, respectively.

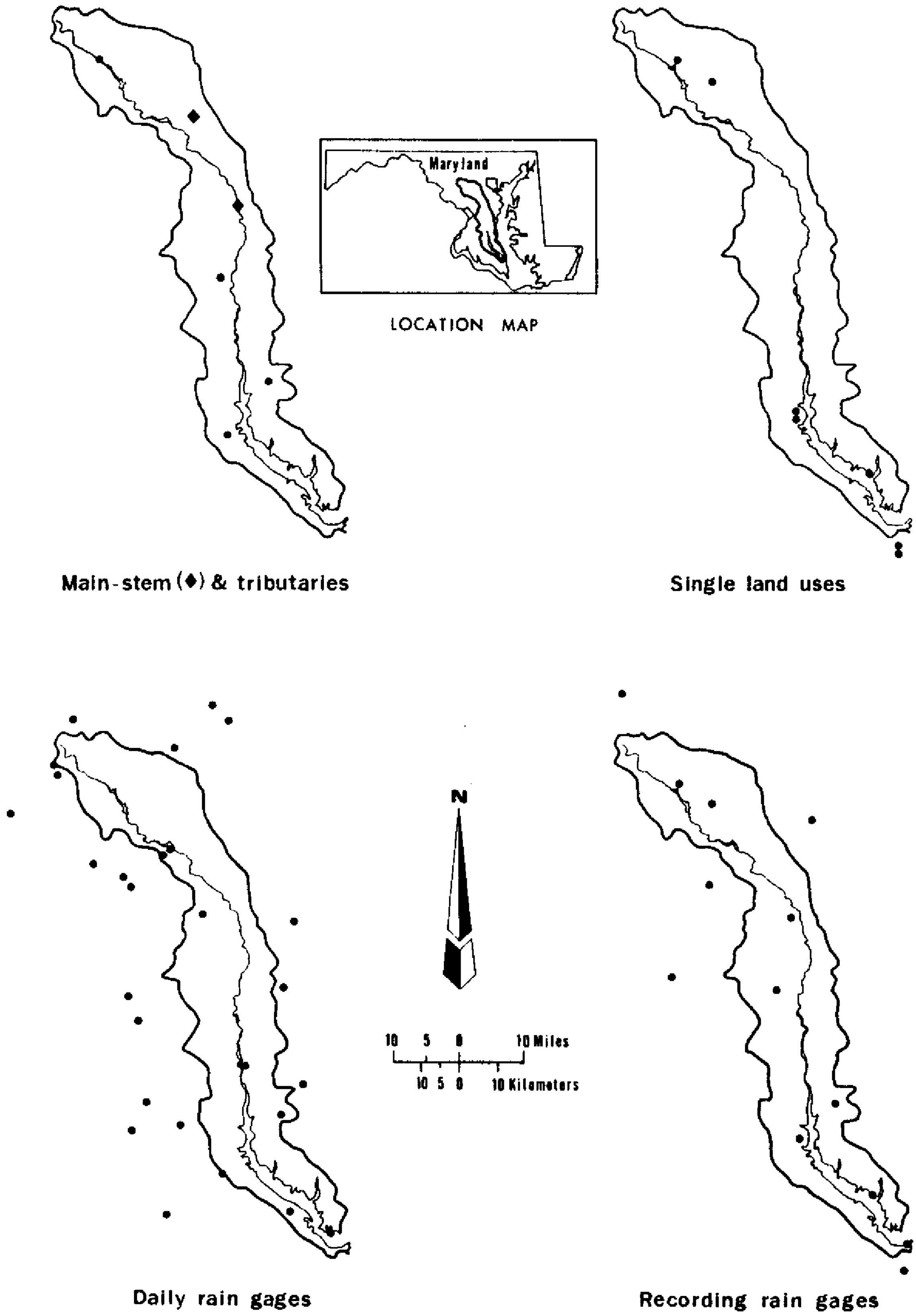

Figure 1. -- Maps of the Patuxent River basin showing data-collection network.

The Patuxent River is the largest tributary of the Chesapeake Bay that is contained entirely within Maryland and presents an ideal test case for the assortment of water-quality management alternatives that have been proposed for the Bay. Many of the management issues of concern for the Bay require assessment at a level of detail that would be cost-prohibitive, if addressed using a single, Bay-wide model. Thus, it is desirable that manageable sub-basins be identified for detailed study that can then serve as an example for further efforts in other parts of the Bay basin. Being under the jurisdiction of a single state, yet large enough to include a wide spectrum of the factors affecting water quality, the Patuxent basin provides an unique opportunity for detailed basin management research.

The monitoring and modeling program described here will be used to quantify nonpoint source pollutant loadings, identify critical regions of nonpoint source pollution in the Patuxent River basin, evaluate the effectiveness of various best management practices (BMP's) in minimizing water-quality impacts from nonpoint sources, and forecast the impacts upon water-quality of various land-use policies. A detailed set of water-quality objectives was developed by OEP in cooperation with the Patuxent Nonpoint Source Workgroup of the Patuxent River Commission. These objectives and the approach to be taken in addressing them in this project are described in the preliminary study plan (OEP, 1984).

This project is being closely coordinated with related work being carried out by OEP, USGS, the U.S. Environmental Protection Agency (EPA), the University of Maryland, and others. Related projects include the OEP/USGS Fall Line Monitoring Program, EPA/OEP Chesapeake Bay Monitoring Program, and OEP Estuarine Water-Quality Modeling. Projects by others include modeling of parts of the Patuxent River watershed by the Maryland-National Capitol Park and Planning Commission.

Study area

The Patuxent River drains an area of about 930 sq.mi. located in eight counties. It represents about 1.5 percent of the total Bay watershed and about 10 percent of that within the State. Land use in 1980 was about 53 percent forested, 41 percent agricultural, and 6 percent urban. However, there is considerable development pressure to accomodate the expansion of the Baltimore and Washington metropolitan areas.

The study area is located within two physiographic provinces -- the Piedmont Plateau and the Atlantic Coastal Plain. The southern part, within the Atlantic Coastal Plain, is characterized by gently rolling, dissected uplands and very flat, often marshy bottomlands. Soils are generally permeable, although drainage is impeded in the lowlands by high water tables. The Piedmont Plateau in the northern part is characterized by higher elevations, gently rolling hills, and deep, narrow stream valleys. Soils are generally well drained.

The climate is generally one of warm summers and mild winters. The coldest period is usually in late January and early February and the warmest is in the last half of July and early August. Monthly precipitation is distributed fairly uniformly throughout the year, and the average annual precipitation is about 43 in. Long-duration storms occur predominately during the cold season (December through March). Average precipitation intensities, however, are highest from June through September, whereas the lower intensity storms occur from December through April.

DATA COLLECTION

Elements of the study include the assembly and maintenance of a hydrometeorologic data base, water-quality data collection before and after the implementation of BMP's, and model calibration and verification.

The study will attempt to quantify loadings of nutrients and sediment from various sources to the Patuxent River estuary and to assess the influence of factors such as soils, topography, land use, and location upon these loadings. Point pollution sources are already being monitored, and will be included in the watershed modeling.

In order to quantify the nonpoint source loadings, 13 data-collection sites have been established (fig. 1). Two are located on the main-stem Patuxent River to study in-stream processes and to determine total loadings from large portions of the basin. Four sites have been established on smaller tributaries, chosen to be representative of different portions of the basin, to study the combined influence of more than one land use upon water quality in those areas. Seven sites collect data from single land uses to quantify the loadings from predominant agricultural land uses and to evaluate the effectiveness of BMP's.

The single land-use sites focus on agricultural practices. Although many studies have been made of nonpoint source loadings from agricultural areas, agricultural practices and regional hydrologic characteristics vary significantly, so that generalizations regarding agricultural loadings cannot easily be made. Previous studies of agricultural areas within the Patuxent River basin did not provide sufficient information for the formulation of a comprehensive watershed model for water-quality planning (Haith, Humenik, and Walter, 1983). Therefore, the emphasis in data collection for this study is upon nonpoint pollution from agricultural areas that are characteristic of the Patuxent River basin. Table 1 lists the characteristics of the single land-use sites being investigated.

Table 1. -- Characteristics of single land-use sites (1986 season).

Crop	Physiography	Treatment	Area (acres)	Soils	Slopes (percent)
Pasture (Dairy)	Piedmont	Severely eroded exercise area	8	Mt. Airy ch.l. (in channel) Glenville s.l. (on hillsides)	8-25 3-8
Corn	Piedmont	Minimum tillage continuous	9	Manor g.l. Manor l. Chester g.s.l. (50-10-40 mix)	8-15 15-25 3-8
Corn	Coastal Plain	Conventional tillage	16	Sassafras sa.l. Mattapex f.sa.l. (50-50 mix)	0-2 0-2
Corn	Coastal Plain	Minimum tillage	4	Sassafras sa.l.	2-5
Soybeans	Coastal Plain	Conventional tillage	8	Matapeake s.l. Woodstown f.sa.l. (50-50 mix)	0-2 0-2
Tobacco	Coastal Plain	Conventional tillage w/o rotation	5	Matapeake f.sa.l. Mattapex f.sa.l. Croom g.sa.l. (40-40-20 mix)	0-2 0-2 5-10
Tobacco	Coastal Plain	Conventional tillage w/rotation	5	Matapeake s.l. Othello s.l. Mattapex s.l. (20-40-40 mix)	0-2 0-2 0-2

Notes:
1. Area and soil mix values are approximate.
2. ch.=channery   l.=loam   g.=gravelly   sa.=sandy   f.=fine

A major feature of the single agricultural land use monitoring program is a comparison of runoff quality with and without BMP's. Two types of comparisons are being made. Side by side monitoring (for corn and tobacco on the Coastal Plain in Table 1) is where two very similar fields, that differ in the BMP (or lack of) applied, are compared over the same period of time. A disadvantage of this approach is that it is very difficult to find similar fields for side by side comparison. Sequential monitoring (all others in Table 1) is where the same field is monitored both with and without BMP's, in sequence. A major disadvantage of this approach is that it takes twice as long to acquire the same data and that during this time, significant variations in weather conditions may occur. Also, because of changing market conditions, it is difficult for a farmer to commit himself to a given crop for an extended period of time. Because of these problems, sequential monitoring is only being done on State-owned or otherwise very stable land where long-term commitments can be reasonably expected.

Other nonpoint source loadings must also be included in the watershed modeling. Loadings from forested areas are being estimated from results of previous studies in this and other basins. Loadings from many urban areas have been determined by the recently completed Nationwide Urban Runoff Program (U.S. Environmental Protection Agency, 1983), including urban areas which are typical of the development in the Patuxent River basin. These results are being evaluated to determine loadings from urban areas within the basin.

For the seven single land-use monitoring stations, sampling frequency is 12 storms per year. Storms are selected to represent seasonal variability among storms. There is no base flow at these sites. Flow-weighted composite samples are being taken for all water-quality parameters using automatic samplers. Flow is measured using a flow meter. Local rainfall is measured using a tipping-bucket rain gage. Bulk precipitation samples are also being collected to obtain an estimate of total atmospheric input of nutrients.

Twelve storms per year are also being sampled at the four tributary and two river main-stem sites. Flow-weighted composite samples are being taken, but due to to longer duration of the storm hydrograph for larger streams, it is sometimes necessary to take several composites per storm and subsequently calculate the total storm load. In addition, base-flow quality is being determined by taking samples with a frequency of approximately once per month. Flow is being monitored using a flow meter, with backup from a standard USGS stream gage.

Meteorologic data is being obtained from the National Weather Service network and is supplemented by rainfall measurements collected at the single land-use sites and 2 additional weather stations established for the project to measure rainfall, wind, evaporation, temperature, and solar radiation. Daily data are also obtained from an existing volunteer weather observer network, as available. Figure 1 shows the location of the meteorologic stations.

The river main-stem and the tributary monitoring sites are equipped with refrigerated sample storage facilities. At the single land-use sites, storage facilities are iced before or immediately after the initiation of sampling. Samples are retrieved within 24 hours and laboratory analysis is performed within 48 hours of the initiation of sampling.

Water-quality analyses are being done by a Maryland Department of Health and Mental Hygiene laboratory in Baltimore, Maryland. Suspended sediment determinations are being made at a USGS sediment laboratory in Harrisburg, Pennsylvania.

Data management

Data collected in this project will ultimately reside in both the USGS WATSTORE data base and the OEP Chesapeake Bay Program data base. These data will include meteorologic and discharge measurements and water-quality determinations. A third data base will be used during the modeling phase of the project. This data base will utilize features of ANNIE, a pre-processor program for modeling-data management and analysis. ANNIE uses the Watershed Data Management System (WDS), a modeling-data format developed cooperatively by USGS, EPA, and the Soil Conservation Service (Alan Lumb, personal communication).

Raw data are processed through two channels. Meteorologic and discharge data are processed using a Prime computer and standard USGS software. After processing and quality control, these data are stored in WATSTORE. Following quality control, the results of water-quality analysis are stored in the Chesapeake Bay Program data base, located on a VAX computer. Exchange of data between systems is accomplished by tape and by the use of identical microcomputer systems at both OEP and USGS project offices. ANNIE is used on the USGS Prime computer to input data into WDS and to prepare inputs for HSPF modeling on either the Prime or VAX systems. Both computers are used for modeling.

Geographic Information System

For this project, a Geographic Information System (GIS) is being assembled that will describe most of the physical and cultural characteristics of the Patuxent River basin. A GIS is a digitally-encoded representation of conventional maps. It contains the same information as any paper map -- land cover, roads, technical information, physical features, etc. — but in a computer storage device. The advantage of this digital data over the hard copy is that it is not a one-time product — it can be reconstituted in any way that the user requires. A map can be quickly produced with any color scheme, at any scale, and with any combination of physical features.

The use of a GIS in hydrologic modeling is especially beneficial. To model a watershed, data must be assembled that describe the physical makeup of the basin. The traditional approach to assembling these data has involved considerable manpower for mechanically overlaying and planimetering maps of the various data types required. Because these efforts were manpower intensive, variations in the watershed modeling scheme were extremely limited. However, with a GIS many modeling schemes can easily be tried.

Attributes of the GIS for the Patuxent River basin include land use, soils, slopes, streams, cultural features, and watershed boundaries. Using ARC-INFO, parameters will be developed from these data for various approaches to modeling the watershed.

MODELING

After an extensive review of modeling alternatives for the Patuxent basin, including consultation with noted experts in the field of nonpoint source pollution modeling (OEP, 1984), a consensus was reached to model the basin using the U.S. Environmental Protection Agency's Hydrological Simulation Program - Fortran (HSPF). The modeling, performed cooperatively by OEP and USGS professional personnel, is proceeding concurrently with the monitoring. Initially, basin segmentation and calibration of the HSPF hydrologic component is being performed using data from existing USGS and other data-collection programs. This work will be followed by calibration and verification of the HSPF water-quality component. The model will not be fully verified for conditions following BMP implementation until after the data have been collected. However, preliminary sensitivity analysis and evaluation of alternative BMP's and land use will be performed once the basin model is calibrated. This work will occur during the fourth year of the project.

To advise the modeling effort, the services of Anthony Donigian have been engaged. Mr. Donigian has had extensive involvement in the development of HSPF and its predecessors. Also, Alan Lumb of the USGS Office of Surface Water will serve as advisor to the project.

TIMETABLE

This project is a multi-phase effort spanning seven years. It includes one year for planning, monitoring site selection and installation, five years of data collection, and one year for preparation of a final report. Modeling will proceed concurrently, as shown on Figure 2, a general time line for the project.

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Figure 2. -- General time line for the Patuxent River Nonpoint Source Pollution Study.

TASK	Federal Fiscal Year (Oct.1 - Sept.30)							
	1984	1985	1986	1987	1988	1989	1990	1991
<u>Project Planning</u>								
Develop detailed study plan	**	****	*	*	*	*	*	
Review literature	**	****	**	*	*	*	*	*
Compile existing data base	**	****	*					
Select monitoring sites	*	***						
<u>Monitoring</u>								
Purchase equipment	*	**						
Install sampling sites		****	**		**			
Water-quality sampling: streams			****	****	****	****	****	
Water-quality sampling: fields			**	****	****	****	****	
<u>Modeling</u>								
Subdivide basin for modeling			**					
Calibrate hydrologic model			**	****	****			
Calibrate water-quality model				**	****	**		
Verify water-quality model						**	****	**
Run sensitivity analysis					****	****	****	***
Production runs								***
<u>Reports</u>								
Progress reports	*	*	*			*	*	
Data reports				*	*	*	*	*
Interpretive reports				*	**			***

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CHROMIUM CONCENTRATIONS IN CORN AND BARLEY GROWN ON SEWAGE SLUDGE-AMENDED SOILS[1]

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ABSTRACT

The effect of Cr on plant growth from sludge application to agricultural soils is at present speculative. Field investigations were conducted in 1984 and 1985 in the Coastal Plain, Piedmont, and Ridge and Valley regions of Virginia to evaluate Cr uptake by corn (Zea mays L.) and barley (Hordeum vulgare L.) plants grown in sludge-amended soil. The sludge used was aerobically digested from a wastewater treatment plant with major industrial inputs. Rates of sludge application were 0, 42, 84, 126, 168, and 210 dry mt ha^{-1}. These sludge rates correspond to Cr applications of 0, 1021, 2042, 3063, 4084, and 5105 kg ha^{-1}. The Cr concentrations in corn and barley tissue were low (<2.8 mg kg^{-1}) at all rates of Cr application on the soils under study.

When excessive levels of Cr are applied as sewage sludge and incorporated into agricultural soils at pH levels in the range of 5.5 to 7.0, Cr forms insoluble Cr hydroxide which is non-mobile in soil. At soil pH levels >7.0, however, trivalent Cr can be oxidized to hexavalent Cr, which exists in soil as the chromate or dichromate anion. Hexavalent Cr anions are mobile in soils, and therefore, are potential groundwater pollutants. To prevent plant uptake and leaching of hexavalent Cr, guidelines for land disposal of high Cr-sludges should account for the extremely low solubility of Cr hydroxide at pH levels in the range 5.5 to 7.0. Formation of insoluble Cr compounds in the pH range of 5.5 to 7.0 accounts for the low uptake of applied Cr by corn and barley in this investigation. It is neccessary to adjust soil pH from 5.5 to 7.0 to ensure formation of insoluble Cr compounds and, thereby, to prevent movement of Cr into groundwater which may result in

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subsequent pollution of the Chesapeake Bay.

REVIEW OF LITERATURE

Introduction:

An increase in land application of wastewater sludges has stimulated interest in the potential environmental pollution hazard of Cr-bearing sludges. Federal regulations have established that total Cr may not exceed 1 μmole L^{-1} in public water supplies (EPA, 1976, 1980). The World Health Organization (1973) estimated that intake of Cr by Americans varied from 5 to 100 μg day^{-1} and that assimilation of Cr(III) ingested by drinking public water is unlikely. Adult urinary loss of Cr is 5 to 10 μg day^{-1} and at least this much must be replaced to maintain balance. Scott (1972) concluded that Cr is important for glucose metabolism in animals and its activity is related to that of insulin. The hexavalent form of Cr [Cr(VI)], however, is an irritant and is corrosive to mucous membranes (National Academy Sciences, 1974).

Chromium ranges from trace concentrations to 250 μg g^{-1} as chromic oxide in soil. Concentrations of Cr are usually higher in soils derived from basalt or serpentine. High Cr levels exist in ultramafic igneous rocks, shales and clays, and in phosphorites. Chromium concentrations in phosphorites range from 30 to 3000 μg g^{-1}. Phosphorites are used as fertilizers and are a source of soil Cr contamination as are limestones (National Academy Sciences, 1974).

Chemistry of soil chromium:

Chromium is present in soil in either the trivalent [Cr(III)] or hexavalent [Cr(VI)] oxidation state. Two forms of Cr(III) may exist in soil, these include the trivalent Cr cation (Cr^{3+}) and the chromium oxide anion (CrO_2^-). Two forms of Cr(VI) that may exist in soils include the chromate and dichromate anions, CrO_4^{2-} and $Cr_2O_7^{2-}$, respectively (Reisenauer, 1982). Hexavalent Cr is toxic to plants, mobile in soils, and exists as a potential ground water pollutant (Bartlett and Kimble, 1976a; Bartlett and James, 1979; Shivas, 1980). It is important to consider the valence state in regard to the environmental implications of Cr-sludge application to soil. Since Cr(VI) is more toxic and mobile in soils than Cr(III), we must consider the possibility that Cr(III) might convert to Cr(VI) and vice versa. In the development of guidelines for the land disposal of high Cr wastes, it was assumed that the extremely low solubility of $Cr(OH)_3$ at pH levels >5.5 would prevent both plant uptake of Cr and downward movement of Cr into ground water (U.S. EPA, 1977).

Until 1976, chemical reactions of Cr in soils were largely speculative. Bartlett and Kimble (1976a) showed that, as solution pH was raised above 4, solubility of Cr(III) decreased. Mertz (1969) as cited by Bartlett and Kimble (1976a) assumed that the Cr precipitate consisted of macromolecules with Cr ions in six coordination complexes with water and hydroxyl groups. Trivalent Cr and Al chemistry are similar in soils. Both cations tend to be octahedrally coordinated. In addition, both Cr(III) and Al become anions in soil when the pH is greater than neutral. Research by Bartlett and Kimble (1976b) has shown that the pH-solubility curve of Cr(VI) in the presence of excess Al is quite similar in shape to that of phosphate with excess Al. Hexavalent Cr has been shown to become completely insoluble near pH 6 and to become soluble again above pH 8. It is probable that portions of Cr(VI) coprecipitate with Al in soils. All of the soils studied adsorbed Cr(VI) except one with a $CaCO_3$ horizon. The orthophosphate present in the soils competed for adsorption sites and prevented Cr(VI) adsorption.

Chromium studies in soils show that the presence of organic matter brings about spontaneous reduction of some Cr(VI) to Cr(III) even at pH levels above neutrality. Hexavalent Cr reduction does not occur in soils low in organic matter unless an energy source is provided (Bartlett and Kimble, 1976b). They developed the following equation to explain the reduction of Cr(VI) along with the oxidation of a hypothetical compound:

$$\begin{gathered} HOHC{=}R{=}CHOH + K_2Cr_2O_7 + 6HCl \rightleftarrows \\ KOOC{-}R{-}COOK + 2CrCl_3 + 5H_2O. \end{gathered} \qquad (1)$$

This equation was developed with the understanding that 3 meq of HCl were required to prevent a pH rise during the reduction of 1 mmole of Cr(VI). Bartlett and Kimble (1976b) showed that, with the addition of HCl three times the molar quantity of Cr(VI), the pH after reduction exactly equaled the pH of the system which received Cr(III) directly.

Additions of Cr(III) to soil has resulted in pH decreases. Soil pH decreased from 4.5 to 3.9 where Bartlett and Kimble (1976b) applied 10 µmole of $CrCl_3$ g^{-1}. Grove and Ellis (1980a) proposed that water-soluble Cr(III) compounds added to soils account for pH decreases and for reversion of resulting Cr compounds to the less soluble Cr_2O_3:

$$\begin{gathered} Cr^{3+} + 6H_2O \rightleftarrows Cr(H_2O)_6{}^{3+} \rightleftarrows \\ [Cr(OH)_x(H_2O)_{6-x}]^{3-x} + xH^+ \rightleftarrows \tfrac{1}{2}\, Cr_2O_3 \cdot zH_2O. \end{gathered} \qquad (2)$$

The CrO_3 [Cr(VI)] incorporated into soil hydrolyses rapidly to H_2CrO_4 and subsequent dissociation of H_2CrO_4 causes a temporary decrease in soil pH (Grove and Ellis, 1980a) as shown by equation (3):

$$CrO_3 + H_2O \rightleftarrows H_2CrO_4 \rightleftarrows 2H^+ + CrO_4{}^{2-} \xrightarrow[3e^-]{6H^+} Cr^{3+} + 4H_2O. \qquad (3)$$

Reduction of Cr(VI) to Cr(III) is dependent upon the availability of both protons and electrons. Hexavalent Cr reduction, therefore, proceeds more rapidly in acid than in alkaline soils (Cary et al., 1977). This reductive step will increase the soil pH slightly. The electron donor would be either soil organic matter or Mn (Bartlett and Kimble, 1976b; Bartlett and James, 1979). Bartlett and Kimble (1976b) have shown that Cr(VI) reduction is inhibited in the absence of organic matter. After Cr(VI) is reduced by the reactions of equation (3), the reduced Cr enters the reaction pathway of equation (2).

Although early research by Bartlett and James (1976a) indicated that the oxidation of Cr(III) to Cr(VI) did not occur in soil, it was later shown that a fresh moist field soil will oxidize substantial quantities of Cr(III) to Cr(VI). Bartlett and James (1979) indicated that, in the earlier research, Bartlett and James (1976a) used an air dried soil devoid of oxidized Mn. Bartlett and James (1979) have shown that the oxidation of Cr(III) to Cr(VI) in soils is the result of Mn reduction. Specifically, their research indicated that oxidation of Cr(III):

1. Did not occur in soils low in Mn,
2. Did not occur in acid soils if Mn was in the reduced form,
3. Was accompanied by an increase in reduced Mn, and
4. Occurred at nominal concentrations when oxidized Mn was converted to a reduced form by drying.

Bartlett and James (1979) reported that, of 20 possible couplings of Cr and Mn half reactions, 16 of these couplings would give spontaneous reactions for the formation of Cr(VI). Oxidation therefore appears to occur in soil in the presence of oxidized Mn, which serves as the electron acceptor.

Chromium (III) added to soils as a metal ion is rapidly adsorbed and/or hydrolyzed, and precipitated in the absence of soluble complexing ligands. The Cr(III) applied to soils as waste amendments, however, may remain soluble due to the addition of organic acids (James and Bartlett, 1983a). James and Bartlett (1983a) showed that citric acid, fulvic acids, and water soluble organic matter, prevented Cr(III) precipitation in solution above pH 5.5. One year after an application of 750 µmoles of Cr-citrate per 100 g of soil, soluble Cr (75 µmoles per 100 g) was still present in the soil regardless of pH. The disappearance of soluble Cr-citrate may involve the adsorption of the chelate as an anion or uncharged species by soils rich in iron oxide or kaolinite (Grove and Ellis, 1980b).

Compounds capable of chelating Cr(III) or reducing Cr(VI), such as citric acid, may be present in organic waste materials added to soils or may form during decomposition. The solubility of Cr may be affected by the interaction between oxidation-reduction and organic complexation in soils. The addition of an organic ligand such as citrate can increase Cr solubility [Cr(VI)] by facilitating Cr(III) oxidation. Chromium-citrate is more soluble in soil than $Cr(OH)_3$ and is oxidized more slowly over a longer period of time. This phenomena probably reflects the fact that the chelate is not rapidly precipitated by soils and that the addition of an organic ligand such as citrate can facilitate reduction of Cr(VI) (James and Bartlett, 1983b). The objective of this research was to determine if high chrome-sludges are an environmental hazard when used as soil amendments.

MATERIALS AND METHODS

Field experimentation:

Field experiments were conducted to evaluate corn (Zea mays L.) and barley (Hordeum vulgare) response to applications of an aerobically digested sludge from a wastewater treatment plant with high Cr influent. Rates of sludge application were 0, 42, 84, 126, 168, and 210 dry mt ha^{-1}. These sludge rates correspond to Cr applications of 0, 1021, 2042, 3063, 4084, and 5105 kg ha^{-1}.

The field experiments were conducted at three sites in the Coastal Plain, Piedmont, and Ridge and Valley physiographic regions of Virginia. The field experiments were located on Bojac loamy sand (coarse-loamy, mixed thermic Typic Hapludult; pH, 6.3; CEC, 5.4 cmol(+)kg^{-1}), on Davidson clay loam (clayey, kaolinitic, thermic Rhodic Paleudult; pH, 6.3; CEC, 12.5 cmol(+) kg^{-1}), and on Groseclose silt loam (clayey, mixed, mesic Typic Hapludult; pH 6.0; CEC, 9.3 cmol(+) kg^{-1}).

Field plots consisted of in situ lysimeters to limit the size of the field experiments required for sludge applications and to prevent movement of the sludge components from the experimental area. The lysimeters consisted of an isolated volume of soil, 2.3 by 1.5 m and 0.9-m deep. A randomized complete block design with four replicates was used at the three experimental sites.

Corn crop. The three experimental sites were planted to 'Pioneer 3192' field corn in the spring of both 1984 and 1985. Each spring, 200 kg N ha^{-1} was applied to the control treatment. After stands were established, seedlings were thinned to a population of 57,300 plants ha^{-1} (23,200 plants A^{-1}). Corn grain yields were determined at plant maturity in the fall and were adjusted to 15.5% moisture content.

Barley crop. Barley variety 'Henry' was planted in 18-cm rows at a seeding rate 134.5 kg ha^{-1} in the fall of 1984. Nitrogen was applied to the control treatment at the rates of 20 and 80 kg ha^{-1} in late fall and early spring. Barley silage yields were determined from the Bojac soil at Feekes' growth stage 7 to 9 and silage yields from the Davidson and Groseclose soils at Feekes' growth stage 10.3 to 10.5 (Large, 1954). Barley silage yields were adjusted to 65% moisture content.

Tissue analyses:

Ten corn earleaves were sampled at the early silk growth stage and corn grain was harvested from the entire plot at physiologic maturity. Barley plants were harvested 1" above the soil surface in the spring as previously described. The earleaf, grain, and barley samples were dried at 70°C for 72 h and ground to pass a 20-mesh (0.833 mm) sieve in preparation for Cr analyses. One-half gram subsamples of the ground corn and barley tissue, and grain were digested in a HNO_3-$HClO_4$ acid mixture prior to determination of Cr by flameless atomic absorption spectroscopy.

Statistical analyses:

Crop yield response and plant nutrient levels for the evenly spaced sludge application rates were analyzed by orthogonal polynomial and contrast comparison analyses.

RESULTS AND DISCUSSION

Studies of potential metal toxicities in soils can be divided into two categories: (1) the phytotoxicity from the application of highly soluble metal salts and (2) the impact on the food chain of heavy metals applied to agricultural soils as organic waste amendments. The second category is of primary environmental concern because of increased interest in land disposal of sludge. The criteria used for land application of sludge-borne metals has been described (U.S. EPA, 1983). These regulations were designed to limit the cumulative loading rates of Cd, Cu, Ni, Pb, and Zn in agricultural cropland to levels which would not adversely affect human health. Certain states have adopted more conservative regulations than outlined in federal guidelines. Sludges added to soils increase organic matter which improves soil properties and water holding capacity. Sludges may contain adequate quantities of macronutrients such as N and P for crop production. As long as the application rate does not exceed the crop N requirement, the potential for groundwater contamination of NO_3^- is no greater from sludge application than from commercial fertilizer application (Chaney, 1982). Unlike the cumulative loading rate guidelines for Cd, Cu, Ni, Pb, and Zn to agricultural cropland, the effect of Cr from sludge application to agricultural land is at present speculative.

In the first field experiments during 1984 corn grain yields were consistently higher on all three soils at the highest sludge application rate compared with the control inorganic N treatment (200 kg N ha^{-1}). Yields at the highest sludge application rate were 12,600, 10,810, and 13,220 kg ha^{-1} on Bojac loamy sand, Davidson clay loam, and Groseclose silt loam, respectively. Orthogonal polynomial comparison analysis indicated that corn grain yields increased linearly (P=0.01) as a function of the five evenly spaced sludge applications for the Bojac, Davidson, and Groseclose soils. The plants appeared healthy throughout the first growing season with no apparent visual toxicity symptoms.

Corn earleaf tissue, sampled at the early silk stage, had Cr levels which ranged from 402 to 945, 70 to 233, and 70 to 333 ng g^{-1} (ppb) for the Bojac, Davidson, and Groseclose, respectively. The Cr concentrations in corn earleaves, averaged over all treatments, were 647, 212, and 132 ng g^{-1} (ppb) for the Bojac, Groseclose, and Davidson soils, respectively. Based on contrast analyses, the Cr concentrations in the earleaves from the Bojac, a coarse textured well drained soil, was higher (P=0.01) than in the earleaves from the finer textured Groseclose, and Davidson soils. Chromium was nondetectable [<70 ng g^{-1} (ppb)] in the corn grain from all sludge application rates for the three soils.

Barley was planted in the fall of 1984 as a winter cover crop and harvested for silage in early spring 1985 prior to a second season of corn. Silage yields for the inorganic N control treatment (200 kg N ha^{-1}) were 9890, 9670, and 10,730 kg ha^{-1} compared to the highest sludge application rate (210 dry mt ha^{-1})

which yielded 8680, 9100, and 9670 kg ha^{-1} for the Bojac, Davidson, and Groseclose soils, respectively.

Based on contrast comparison analysis there was no difference in the Cr concentration in the silage for the no N control treatment compared with the five sludge application rates for each of the three soils. Chromium levels in the silage ranged from 1000 to 2748, 860 to 1890, and 560 to 1100 ng g^{-1} (ppb) for the Bojac, Davidson, and Groseclose soils, respectively. The Cr concentrations in the silage, averaged over all treatments, were 1602, 1342, and 878 ng g^{-1} (ppb) for the three soils, respectively.

Corn earleaf tissue, sampled at the early silk stage from the corn in 1985 had Cr levels which ranged from 620 to 1100, 220 to 450, and 320 to 660 ng g^{-1} (ppb) for the Bojac, Davidson, and Groseclose soils, respectively. The Cr concentrations in corn earleaves, averaged over all treatments were 877, 430, and 343 ng g^{-1} (ppb) for the Bojac, Groseclose, and Davidson soils, respectively. The Cr levels in the 1985 corn earleaves (as in 1984) from the Bojac soil, averaged over all treatments was higher, ($P=0.01$) than in the earleaves from the Groseclose and Davidson soils. Stollenwerk and Grove (1985a) showed that very little adsorption of Cr(VI) occurred in soil after removal of Fe oxide and hydroxide coatings with $Na_2S_2O_4$. They concluded that Cr(VI) was adsorbed by these coatings. In agreement with this, the Cr concentration in the earleaves of corn sampled in 1984 and 1985 from the Davidson soil, with a high Fe oxide and hydroxide content, was lower than from the Bojac and Groseclose soils. James and Bartlett (1983c) have shown that the adsorption reaction with $Fe(OH)_3$ removed 78% of applied Cr(VI) and that liming decreased exchangeable Cr(VI) by 71%. Present data indicate that the excessive Cr levels applied as an organic amendent to the three diverse Virginia soils was not detrimental to corn or barley production.

Investigations on the effects of Cr application have indicated that increases in Cr metal concentrations of plant tissue have occurred on some soils. Mortvedt and Giordano (1975) showed that the addition of Cr(VI) as Na_2CrO_7 at 320 µg g^{-1} resulted in Cr concentrations of 29 µg g^{-1} in corn tissue while the application of Cr(III) resulted in 2.8 µg g^{-1}. The application of 1360 µg g^{-1} of Cr in sewage sludge, however, resulted in 1.5 µg g^{-1} of Cr in the tissue with no decrease in grain yield. Cunningham et al. (1975) showed that Cr in corn tissue was higher from inorganic salt treatments than from Cr treatments from organic wastes. Chromium acetate applied at the rate of 700 µg g^{-1} increased tissue Cr levels from <3 µg g^{-1} to 46 µg g^{-1}, whereas the application of 697 µg g^{-1} of Cr in treated sludge increased tissue Cr concentration to only 8 µg g^{-1}. Kelling et al. (1977) reported that 110 kg ha^{-1} of Cr applied as sewage sludge did not increase plant tissue levels of either rye (*Secale cereale*) or sorghum-sudangrass (*Sorghum-sudanese*). In a comparitive yield study, Sykes et al. (1981) indicated that application 500 µg g^{-1} of Cr as tannery sewage sludge did not affect lettuce (*Lactuca sativa*) and radish (*Raphanus sativus*) yields and that $Cr(OH)_3$

rate of 500 $\mu g\ g^{-1}$ of Cr did not affect bean (Phaseolus spp.) yield. In contrast, there was a decrease in bean yield from application of 500 $\mu g\ g^{-1}$ as tannery sewage sludge.

Transformations of chemical species can occur in sludge-soil mixtures following incorporation of high chrome-sludge into soil. Plant available levels are controlled by chemical equilibrium processes. Added Cr to soils by sludge-amendments is chelated by or adsorbed to soil organic matter or Al, Fe, or Mn hydrous oxides. In the worst case scenario of excessive Cr application rates to soils, little organically amended Cr will enter the plant-food chain. Chromium is so strongly chelated in plant root cells that very little is translocated to crop shoots (Chaney, 1982). Chaney (1983) has reviewed chrome wastes from the tanning industry and pointed out that the U.S. EPA removed Cr tanning waste from the hazardous waste list in 1980 due to the presence of Cr(III) rather than the Cr(VI) form.

CONCLUSIONS

Soluble and toxic Cr(VI) formed in or added to soils may be removed by anion adsorption or precipitation or by reduction to low-solubility cationic forms. At soil pH levels >6.4, $HCrO_4^-$ dissociates to CrO_4^{2-}. Soil pH affects the form of Cr(VI) reacting with soil and the rate of reduction of Cr(VI) to Cr(III). Liming soils which contain Al and Fe sesquioxides and kaolinite decreases exchangeable Cr(VI) due to a decrease in positive charge on the soil colloids as pH increases above the minerals zero point charge. In aerobic soils easily oxidized organic compounds will act as reducing agents for Cr(VI).

Soil acidifying compounds and reducing agents, such as organic matter, can be incorporated in the soil to promote Cr(VI) reduction. After Cr(VI) reduction, limestone may be added to form more insoluble Cr compounds.

Long term implications of Cr waste amendments to agricultural soils on pollution of ground water are not completely understood. Even though experimental data indicate that adsorption reactions will minimize Cr as a source of ground water contamination, Cr desorption data are required to evaluate the full potential of Cr(VI) pollution in the Chesapeake Bay. It is recommended that the method outlined by Stollenwerk and Grove (1985b) be followed for monitoring levels of Cr(VI) in groundwater and in the Chesapeake Bay. Caution is required for current standard water analysis techniques can result in reduction of Cr(VI) to Cr(III) by sample acidification and by impurities such as NO_2 present in HNO_3. This reduction reaction of Cr(VI) could lead to erroneous values for the potential pollutant.

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WATER-BORNE NUTRIENTS IN THE POTOMAC RIVER SYSTEM

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ABSTRACT

Statistical analysis of data has led to the development of methods to compare the relative contributions from different portions of the Potomac River Basin of water-borne nitrogen delivered to the Potomac Estuary. The results of the analysis are useful in determining which upland areas (and therefore, soil types and land use practices) most heavily influence the amounts of nutrients reaching the Estuary and ultimately the Chesapeake Bay.

Standard linear regressions were derived for total nitrite/nitrate versus river flow. The National Stream Quality Accounting Network (NASQAN) water quality data were used for total nitrite-nitrate data, and the United States Geological Survey (USGS) daily averaged data were used for river flow. Nitrogen data were regressed against single station flow, and multivariate regressions against lagged upstream flows were performed for two sub-basins. Travel times from each of the upstream gages to Chain Bridge were determined by a routing model based on USGS time of travel studies. The multivariate regression results produced substantial improvement in the coefficient of determination ($\bar{r}2$) for both concentration and load.

The effect of land use on the way nitrogen reaches the river system is investigated by examining the relative proportions of flow from upland sub-basins. This analysis over time and space indicates how different areas impact the water quality of the River.

DESCRIPTION OF DATA

In this study, the feasibility of using multivariate regression analysis to predict nutrient loads and concentrations at the fall line of the Potomac River is assessed. Available nutrient observations at the fall line (Chain Bridge monitoring station) are insufficient input data for most water quality models. Various single-station regression techniques have been used in the past [NVPDC, 1983; HydroQual, 1982; Hydroscience, 1976] to reconstruct continuous records. Because of the detailed long term flow data available for upstream stations at major tributaries in the Potomac River basin, successful application of this additional information could greatly improve the current methodology for estimating nutrient data.

Initial examination of this proposed methodology was begun in a cooperative project with the Quality of Water Branch of the USGS. Daily flow data are used for the ten year period January 1, 1974 to December 31, 1983. Four upstream USGS flow gaging stations are chosen to represent the streamflow of the Potomac River and major tributaries above the fall line. Flow data from the USGS station at Little Falls near Washington, D.C. is used to represent Chain Bridge flow for the calculation of loads. The stations (see Table 1) account for 87% of the drainage area. Several methods are available to adjust for the missing area including the regression constant, a 'dummy' station, and area weighted adjustment of the Goose Creek flow.

Table 1. Drainage Area for Selected Flow Gaging Stations

Station	USGS #	Drainage Area sq. mi.	% of Total at L. Falls
Potomac River at Shepherdstown, WV	1618000	5936	51
Shenandoah River at Millville, WV	1636500	3040	26
Monocacy River at Jug Bridge, MD	1643000	817	7
Goose Creek near Leesburg, VA	1644000	332	3
Subtotal		10,125	87
Potomac River near Washington, D.C. (Little Falls)	1646500	11,570	100

The nutrient data used is from the National Stream Quality Accounting Network (NASQAN) station at Chain Bridge for the same 10 year time period as the flow data. Initially, six nutrient species are used in this analysis: dissolved and total ammonia, dissolved and total nitrite/nitrate, and dissolved and total phosphorus. Regression of nutrient concentration and instantaneous flow at Chain Bridge showed very poor adjusted coefficient of determination ($\bar{r}2$) ranging from 0 to 30%.

MULTIVARIATE ANALYSIS

Multivariate regression analysis of nutrient concentration with the four upstream flow stations showed substantial improvement in $\bar{r}2$. In order to refine the model, methodology was developed to determine travel time from each of the upstream gages to Chain Bridge for varying flow regimes. A routing model had been developed by Steiner, et al. [1984] based on USGS time of travel studies [Taylor et al. 1984] and was modified in order to estimate the time lag in the arrival of flow (nutrients) at Chain Bridge.

The new relationships between lagged flow data and nutrient concentration and loads are examined in greater detail for nitrite/nitrate for a 3 year subset of the initial data. Total nitrite/nitrate is chosen because the correlation with single station flow (at Little Falls) is low: $\bar{r}2$ = 10.5%, and it is more conservative than the volatile parameters such as ammonia (see Figure 1).

Figure 1. Little Falls Flow (cfs) vs. Total Nitrite/Nitrate (mg/l) at Chain Bridge

```
             -
             -
             -                    2
         100 +             *
             -                    *
    Flow     -
(000's cfs)  -             *  **  2
             -             *
          50 +                **  2**
             -                     **          *
             -             *      2*        *     *
             -        * 2 *  32   ** *            *
             -  * ***22246656 6* *24  * *3    *        *
           0 + **25454852*4**  * **   *
              +---------+---------+---------+---------+
             0.0       0.7       1.4       2.1       2.8

                        Total N/N (mg/l)
```

The starting data base consists of lagged flow in the sub-basins:

Potomac to Shepherdstown
Shenandoah River
Monocacy River
Goose Creek
Remaining local drainage to Chain Bridge (D.C.)

and sampling data of total nitrate/nitrite at Chain Bridge for three years beginning with 1979. Initially, linear regression analysis is used only to investigate the underlying relationships between the flow and nitrogen data. In addition, single harmonic Fourier analysis is applied to the nitrogen data in order to explain some of the variation.

Equations relating load and concentration of nitrogen at Chain Bridge to streamflow at various gaging sites are derived using multivariate linear regression. In the example described below two streamflow variables, representing upper basin flow and lower basin flow, are used. The upper basin is taken to be the Appalachian portion of the basin above Harpers Ferry. Upper basin flow is computed from the Potomac River gage at Shepherdstown and the Shenandoah River gage at Millville. The lower basin is taken to be the Piedmont portion of the basin. Lower basin flow is computed from the Monocacy River gage at Jug Bridge and the Goose Creek gage at Leesburg. Splitting the flow into upper and lower basin components yields better prediction results for nitrogen at Chain Bridge than by simply using Little Falls streamflow. This result can be interpreted as follows: splitting streamflow into Appalachian and Piedmont components allows the model to distinguish the effects of land-use, soil type, and geology on nitrogen inputs to the river. For example, high streamflow from the Monocacy River with its heavy agricultural land use, erodible soils, and close proximity to the Potomac estuary has a much greater impact on nutrient loads to the estuary than high streamflow from the forested South Branch Potomac River in West Virginia. More refined partitions of streamflow contributions might provide even better estimates of Chain Bridge nutrient loads than those obtained by upper basin -- lower basin differentiation.

The regression equation relating Chain Bridge nitrogen loads to upper basin flow and lower basin flow is the following:

$$NL = 1.0*FU + 1.8*FL \quad (1)$$

where:
NL = Chain Bridge nitrogen load in tons/day,
FU = Upper basin flow in cfs,
FL = Lower basin flow in cfs.

The adjusted coefficient of determination ($\bar{r}2$) is 90%. By comparison, the adjusted coefficient of determination for the regression of Chain Bridge nitrogen load versus Chain Bridge streamflow is 60%.

The regression equation relating Chain Bridge nitrogen concentration to upper basin and lower basin flow is the following:

$$NC = 1.6*XU + 3.4*XL \tag{2}$$

where:

NC = Chain Bridge nitrogen concentration in mg/l,
XU = fraction of total flow from upper basin,
XL = fraction of total flow from lower basin.

The adjusted coefficient of determination ($\bar{r}2$) is 23%. By contrast, the adjusted coefficient of determination obtained from regressing Chain Bridge nitrogen concentration against Chain Bridge flow is 10.5%.

Seasonality is an important feature of both nitrogen and streamflow data. To predict nitrogen load and concentration from streamflow, it is necessary to determine whether seasonality is also an important feature in the relationship between nitrogen and streamflow. The analysis indicates that the relationship between nitrogen concentration and streamflow has a strong seasonal component. This seasonality may reflect the change in flow regime during the course of a year. For example, Spring storm runoff from saturated land provides very different pathways for nitrogen to enter river channels than ground water contributions to river channels during Summer baseflow periods. Partitioning flow between ground water and storm flow sources is one approach that might improve estimates of nitrogen concentration at Chain Bridge. Refinement of the estimates of travel time of nutrients from upstream monitoring stations might also improve estimates of Chain Bridge nutrient loads and concentration.

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EVALUATION OF EFFECTIVENESS OF BMPs IN URBANIZING WATERSHEDS

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ABSTRACT

A desktop computer model has been developed to estimate pollutant loadings associated with stormwater runoff in urban areas and to evaluate the effectiveness of BMPs in reducing pollutant loadings. The model is a continuous simulation model that takes into consideration the pollutant buildup during dry days and removal during rainfall events. The model uses the Monte Carlo method to simulate the amounts of daily rainfall and the number of days between rainfall events. Runoff is calculated using the SCS TR-55 method. Pollutant washoff is calculated using first order washoff equations. Pollutant loadings simulated by the model are BOD, Total Phosporous (TP), Total Nitrogen (TN), and Total Suspended Solids (TSS). The effects of five types of BMPs in reducing pollutant loadings in the runoff can be evaluated.

INTRODUCTION

Stormwater from urbanizing areas is a major component of nonpoint source pollution. The proper management of urban stormwater runoff to reduce the amount of pollutants transported in the runoff requires the implementation of best management practices (BMPs). Since efforts to use BMPs to control pollutant loading in urban runoff are relatively new undertakings, the effectiveness of these controls has not been well defined. A desktop computer model to estimate the total amount of pollutant loadings contained in urban runoff and the amounts that may be removed by various BMPS has been developed.

The pollutant loading in urban runoff is based on the total mass of pollutant accumulation on the ground surface and the fraction which is removed during a storm event. This is a time dependent process, therefore a continuous model is required to simulate the process for a particular study period. The continuous model is able to account for the accumulation of pollutants on dry days and the removal of pollutants during storm runoff for continuous periods of study (i.e., on a monthly and/or yearly basis).

RUNOFF

Runoff is calculated using the SCS TR-55 curve number method where:

$$Q = (P - .2S)^2/(P + .8S) \qquad \text{If} \quad P > 0.2S \qquad (1)$$

$$Q = 0 \qquad \text{If} \quad P \leq 0.2S \qquad (2)$$

where:

$$I_a = 0.2S \qquad (3)$$

$$S = (1000/CN) - 10 \qquad (4)$$

It is assumed that the runoff from a rainfall event occurs within a 24 hour time period starting from the beginning of the rainfall. The model incorporates the new TR-55 Runoff Curve Number (RCN) Table with 78 land use catagories. Correction of the RCN for 5-Day Antecedent Moisture Conditions is incorporated in the model.

POLLUTANT ACCUMULATION AND WASHOFF RATES

In order to determine the pollutant loading in the runoff, it is necessary to determine the amount of pollutant accumulation on the ground at the start of a rainfall event and the amount that is washed off during the event. The amount of pollutant on the ground at the start of a rainfall event is a function of the number of days without rainfall prior to the event and the daily accumulation of pollutants on the ground surface. Field studies conducted in 1978 by the Northern Virginia Planning District Commission (NVPDC) and Virginia Polytechnic and State University (VPISU) have identified average annual dry weather accumulation rates (lbs/acre/day) for pervious and impervious fractions of urban and rural land uses. These rates are presented in a report by NVPDC (1979) and are used to estimate the pollutant accumulation on the ground at the beginning of a rainfall event as follows:

$$P_o = NDSLR * DWPAR * DA \qquad (5)$$

where:

P_o = Pollutant Accumulation on the Ground (lbs)
NDSLR = Number of Days Since Last Rainfall Event (days)

DWPAR = Dry Weather Pollutant Accumulation Rate (lbs/acre/day)
DA = Watershed Drainage Area (acres)

Pollutant washoff is calculated using the first order washoff equation that was developed for the USEPA Storm Water Management Model (1971) and subsequently tested by Sartor and Boyd (1972):

$$P_o - P = P_o * (1-e^{-br\Delta t}) \quad (6)$$

where:

P_o = Pollutant Accumulation on Ground at Beginning of Time Step (lbs)
P = Pollutant Remaining on Ground at End of Time Step (lbs)
$P_o - P$ = Pollutant Washoff During Time Step (lbs)
b = 4.6 For Impervious Surfaces
b = 1.4 For Pervious Surfaces
Δt = Time Step of Calculation (hour)
r = Rainfall Excess (runoff) During Time Step (in/hr)

This equation is based on the assumption that 0.5 in/hr of rainfall excess will washoff 90% of the pollutant accumulation on impervious surfaces. It is also assumed that the runoff rate for a particular time interval is constant. It is obvious that the pollutant washoff is a function of the rainfall excess for the time step.

Pollutant washoff is calculated for each rainfall event using a 1 hour time step for the 24 hour time period of the runoff. The rainfall disrtibution for a rainfall event is assumed to be equal to the SCS Type II dimensionless rainfall distribution. The runoff at the start and end of each time step is calculated using the SCS TR-55 method with a precipitation amount equal to the 24 hour rainfall amount multiplied by the ordinate of the Type II distribution curve at the start and end of each time step. The rainfall excess for the time interval is the difference between the runoff at the start and end of the time interval divided by the time step. The pollutant washoff is summed for the 24 hour period using the rainfall excess for each time step and the first order washoff equation.

Any pollutant remaining on the ground at the end of the rainfall event is added to the dry weather accumulation during the period of no rainfall prior to the next rainfall event.

MONTE CARLO SIMULATION

In order to model pollutant loadings in the storm runoff, the occurrence of rainfall events and the amount of rainfall per event must be known. Historical rainfall records are available from the National Weather Service. The interevent times of rainfall events and the amount of rainfall per event are random variables.

The Monte Carlo simulation procedure was selected to generate the occurrence of rainfall events and the amount of rainfall per event for future time periods. This method uses random numbers and the probability distribution of the historical data to simulate future system behavior.

The amount of rainfall per event is a continous distribution as the amount of rainfall can assume any value greater than zero. However, the time between the occurence of rainfall events is a discrete distribution as the number of days between rainfall events can only assume discrete, integer values.

Rainfall Simulation

To determine the historical distribution of the amount of rainfall per event, the historical data must be analyzed and a distribution that best fits the data selected. Alternately, the sample can be reconstituted by a suitable transformation such that the transformed sample follows a particular distribution. This method transforms the original data set, which has an unknown distribution, into a data set with a known distribution. The Monte Carlo simulation procedure is applied to the transformed data set to generate a random data set with the known distribution. The simulated data set is then converted back to the original disrtibution through the use of an inverse transformation. The result is a randomly generated future data set with the distribution of the original historical data set. The advantages of this approach are discussed by McCormick (1984).

The transformation that has been selected is the power transformation that was first proposed by Box and Cox (1964), which transforms the sample into a near normal distribution.

The power transformation is a family of transformations of the following type:

$$T_i = ((y_i)^{\lambda} - 1)/\lambda \qquad \text{When} \quad \lambda \neq 0 \tag{7}$$

$$T_i = \ln y_i \qquad \text{When} \quad \lambda = 0 \tag{8}$$

where:

T_i = Transformed Sample Value
Y_i = Original Sample Value
λ = Constant of Transformation

A different transformation occurs for each value of λ. The correct value for λ is the value that produces a transformed sample with a coefficient of skewness (C_s) of zero. The coefficient of skewness is defined as:

$$C_s = M_3 / (M_2)^{1.5} \tag{9}$$

where:

M_2 = Second Moment of the Sample About the Mean
M_3 = Third Moment of the Sample About the Mean

The basic procedure used in simulating the amount of rainfall per event is as follows:

1. The power transformation is used (with λ selected so that $C_s = 0$) to transform the original sample with unknown distribution into a normal distribution with mean, μ and standard deviation, S of the original sample.
2. Pairs of uniform random numbers (U_1, U_2) are generated and a standardized normal data set is calculated using an exact inverse method proposed by Box and Mueller (1958):

$$Z_i = (-2\ln U_1) * \cos 2\pi U_2 \tag{10}$$

$$Z_i = (-2\ln U_1) * \sin 2\pi U_2 \tag{11}$$

3. A normal data set is generated from the standardized normal data set with mean, μ and standard deviation, S of the original sample as follows:

$$X_i = (Z_i * S) + \mu \tag{12}$$

4. The simulated normal data set is converted back to the original sample distribution using the following inverse transformation:

$$P_i = ((\lambda * X_i) + 1)^{1/\lambda} \tag{13}$$

Days Between Rainfall Events Simulation

It is generally assumed that if the probability that an event will occur during a small time interval is very small, and if the occurrence of this event is independent of the occurrence of other events, then the time interval between the occurrence of events is exponentially distributed. However, the time between the occurrence of rainfall events is measured in days which can only assume discrete, integer values. Since the time between the occurrence of rainfall events is a discrete distribution, the power transformation method can not be used to simulate future interevent times since it requires a data set with a continous distribution. Therefore, a discrete distribution is needed for use in the Monte Carlo simulation method that closely approximates the exponential distribution.

The Poisson distribution was selected for use in the model to simulate the number of days between rainfall events. The Poisson distribution was selected because of its shape flexibility and its well known relationship with the exponential distribution. If it is assumed that the time between rainfall events during during a particular time period is exponentially distributed, then the number of events occurring per time period is Poisson distributed. The Poisson distribution is a discrete distribution with both mean and variance equal to λ. The parameter can have any positive value and need not be an integer.

To simulate the number of days between rainfall events the mean of the historical data is calculated. Poisson random data values are generated using the Monte Carlo simulation method with mean λ of the historical data set. Generation of Poisson distributed values is done using a method presented by Tocher (1963) that generates random uniform variates (r_i) on the interval from one to zero until one of the following relationships holds:

$$\prod_{i=0}^{x} r_i \geqslant e^{-\lambda} > \prod_{i=0}^{x+1} r_i \qquad (14)$$

EVALUATION OF BMPS

The cost-effectiveness of BMPs in reducing the pollutant loads in urban runoff is a function of the pollutant removal rate of the BMP and the cost of designing, constructing, and maintaining the structure. The following BMPs can be evaluated by the model:

1. Dry Pond Detention Basins
2. Wet Pond Detention Basins
3. Extended Wet Pond Detention Basins
4. Infiltration Trenches
5. Porous Pavement

Dry pond detention basins are defined as detention basins that provide for peak shaving of storm runoff but do not maintain a permanent storage pool during dry periods. Wet pond detention basins are defined as detention basins that provide for peak shaving of storm runoff and maintain a permantent storage pool during wet and dry periods. Extended wet pond detention basins are defined as detention basins that provide for peak shaving of storm runoff and extended detention times to improve pollutant removal and maintain a permanent storage pool during wet and dry periods.

The removal rate of pollutants by BMPs is dependant on the volume of runoff detained by the BMP, the pollutant concentrations in the runoff, and the removal efficiency of the BMP. The volume of runoff entering the BMP is calculated using the SCS TR-55 RCN method. The volume of storage required for infiltration trenches and porous pavements is calculated using the design procedure developed by the Maryland Department of Natural Resources (1984). The volume of storage required for detention basins is calculated using the approximate detention basin routing procedure in TR-55. Basins are designed to limit post development peak runoff rates to predevelopment levels for a selected design storm. Peak inflow rates to the basin are calculated using the SCS TR-55 tabular hydrograph method. The total pollutant load in the runoff that is stored in the BMP is calculated by multiplying the volume of runoff stored in the BMP by the pollutant concentration in the runoff.

Numerous values are reported in the literature for the removal rates of different BMPs. The removal rates selected for use in the model are taken from studies by the NVPDC (1979) and Schueler, et al (1985) and are summarized in Table 1.

To evaluate alternative BMP strategies, a cost analysis is performed. Costs used in the model were developed by Allison (1985). The costs include base construction costs (exclusive of land costs), contingency costs (design and administration of construction), and operation and maintenance costs.

Table 1 - Pollutant Removal Rates of BMPs

	Pollutant Removal (%)			
BMP Type	TSS	BOD	TP	TN
Detention Basins:				
Dry Pond	14	0	20	10
Wet Pond	55	22	66	28
Extended Wet Pond	91	42	42	27
Infiltration Trench	96	84	61	41
Porous Pavement	96	84	61	41

For each BMP evaluated the amount of pollutant removal and the cost of the facility is calculated. This allows for the evaluation of different locations and combinations of

BMPs in a drainage area easily and quickly.

APPLICATION

The model was used to evaluate the best arrangement of detention basins for the demonstration watershed shown in Figure 1. The watershed consists of 650 acres of woods and farm land that is to be developed into a single family residential and commercial area. The watershed is divided into 2 subdrainage basins of 400 and 250 acres each. The alternatives considered are: 1) development without use of BMPs, 2) the location of a wet pond detention basin at the outlet of each subdrainage area, and 3) the location of 1 large wet pond detention basin at the outlet of the watershed. The effectiveness of the alternatives is evaluated for a time period of 1 year. A total of 112 rainfall events occur during the study period with a total of 37.57 inches of precipitation. The predevelopment runoff is 2.28 inches (5,351,000 CF) and post development runoff is 3.92 inches (9,235,000 CF) without any BMPs.

The results of the evaluation are shown in Table 2. Alternative 1 results in a yearly pollutant load of 297,331 lbs. of TSS, 42,423 lbs. of BOD, 775 lbs. of TP, and 6,727 lbs. of TN. Alternative 2 consists of a 836,000 CF and a 738,000 CF wet pond detention basin at the outlet of Subareas 1 and 2, respectively. Alternative 3 consists of a 1,957,000 CF wet pond detention basin at the outlet of the watershed. As seen in Table 2 , Alternative 2 costs $206,415 and reduces yearly pollutant loads to 145,159 lbs. of TSS, 34,066 lbs. of BOD, 305 lbs. of TP, and 5,012 lbs. of TN. Alternative 3 costs $164,270 and reduces yearly pollutant loads to 133,342 lbs. of TSS, 32,327 lbs. of BOD, 261 lbs. of TP, and 4768 lbs. of TN. Based on these figures, Alternative 3 provides the greatest reduction in pollutant loads at the lowest cost.

The detention basins in this example are sized by the model to limit post development peak runoff for a 10 year design storm to predevelopment levels. However, it is seen from the results that this does not result in the reduction of post development total yearly pollutant loadings for all of the water quality parameters in the runoff to predevelopment levels. If further reduction in pollutant loadings of the water quality parameters is desired, additional BMP alternatives would need to be evaluated.

CONCLUDING REMARKS

The model can be used to evaluate the performance and cost-effectiveness of 5 alternative BMPs types at any location within a selected watershed for study periods of up to 1 year in length. Data required to run the model includes the

Table 2 - Results of Evaluation of Demonstration Watershed

Alternative	Size (1000 CF)	Cost	Total Storm Runoff (1000 CF)	Total Pollutant Load (lbs) TSS	BOD	TP	TN
PREDEVELOPMENT CONDITIONS (w/o BMPs):							
Subarea 1	--	--	4,734	126,697	19,562	258	2,686
Subarea 2	--	--	617	27,548	992	11	165
Totals	--	--	5,351	154,245	20,554	269	2,851
POST DEVELOPMENT CONDITIONS:							
Alternative 1 (w/o BMPs)							
Subarea 1	--	--	6,476	157,155	33,770	481	4,581
Subarea 2	--	--	2,759	140,176	8,653	294	2,146
Totals	--	--	9,235	297,331	42,423	775	6,727
Alternative 2 (w/BMPs)							
Subarea 1	836	$106,450	798.6	82,080	27,317	205	3,467
Subarea 2	738	$99,900	0.0	63,079	6,749	100	1,545
Totals	1,574	$206,350	798.6	145,159	34,066	305	5,012
Alternative 3 (w/BMPs)							
Watershed	1,957	$164,270	0.0	133,342	32,327	261	4,768

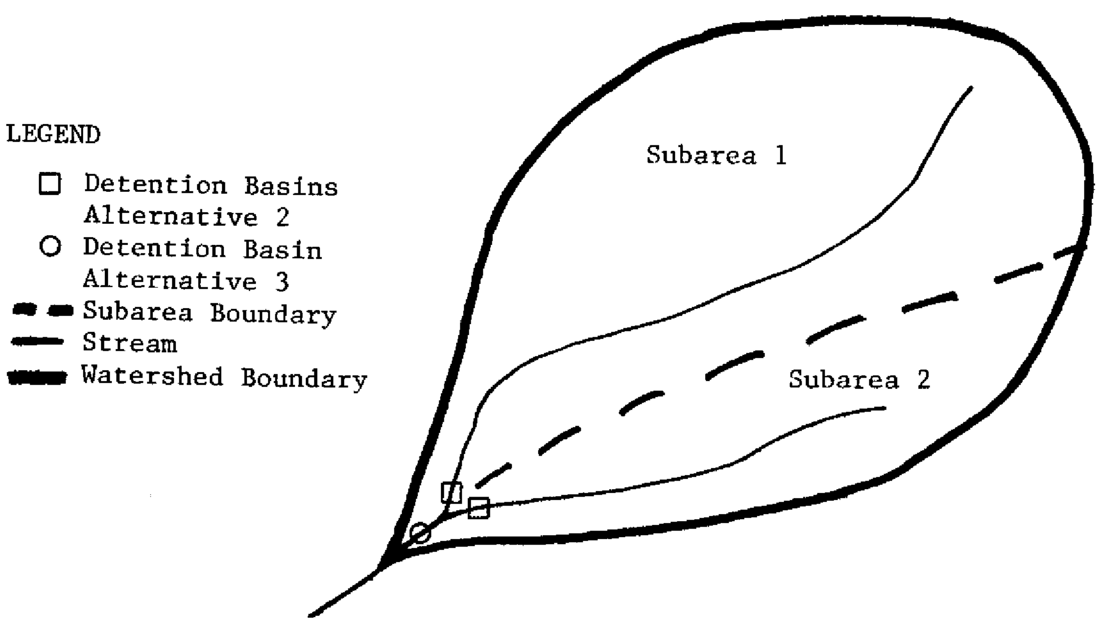

Figure 1 Demonstration Watershed

historical rainfall amounts and the number of days between rainfall events, and the physical characteristics of the watershed required by the SCS TR-55 method to calculate the runoff from the watershed.

Application of the model to a watershed is limited only by the physical limitations of the SCS TR-55 method. Subdrainage areas may not be larger then 2000 acres and each BMP site must be considered as a subdrainage area. The model has been run with 20 subdrainage areas and 112 rainfall events.

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OVERVIEW OF BMP'S FOR CONTROLLING ANIMAL WASTE

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ABSTRACT

Waste generated in livestock and poultry operations has potential to impact on surface and groundwater supplies as nonpoint source pollution. Animals produced in confinement buildings generate large quantities of wastes which are generally disposed of by land application. Often insufficient land is available to assimilate the manure nutrients at a crop utilization rate. Manure storages at the point of production (point source) also contribute to the nonpoint source problem, and after manure is land applied, it offers the threat of being washed or leached into receiving waters. Animals which are maintained wholely or partially on pasture also may contribute to nonpoint source pollution by directly depositing wastes in streams and lakes, or by creating denuded areas or disturbed crossings and stream banks, or other areas which are easily eroded and contribute to turbidity and sedimentation problems.

Best Management Practices (BMP's) are a practice, or combination of practices, which are deemed to be the most effective practicable means of preventing or reducing the amount of pollution generated by nonpoint sources to a level compatible with water quality goals. Most often these measures have been suggested as common sense approaches which many farm operators employed long before nonpoint source pollution became a concern. However, in recent years regulatory agencies, extension workers, other farm service personnel, and farmers have given BMP's increased emphasis as a possible means of reducing agricultural contributions to water quality problems.

This paper will present an overview of the major animal waste related BMP's normally practiced in livestock agriculture. A qualitative assessment will be given of the practicality and effectiveness of each practice, and potential problems and need for increased verification of cost effectiveness of some of the practices.

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ANIMAL WASTE LAGOONS AND GROUNDWATER QUALITY IMPACTS IN COASTAL PLAIN SOILS

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ABSTRACT

Groundwater quality at two sites around clay-lined animal waste lagoons has been monitored for three years. A swine waste lagoon located in an Evesboro loamy sand soil is having a severe impact on groundwater quality. Concentrations of NH_3-N above 100 mg/L have been measured in shallow wells around the lagoon. At the second site which has three lagoons and a settling basin concentrations of NH_3-N, NO_3-N, Cl and TDS are above background levels in some of the monitoring wells.

INTRODUCTION

Many anaerobic lagoons on the Delmarva Peninsula are clay lined and installed in either sandy loam or loamy sand soils with a high water-table. The Soil Conservation Service has been interested in determining if seepage is occurring from these clay lined animal waste lagoons.

Ritter and Chirnside (1983) found an unlined anaerobic lagoon for swine wastes, had some impact on groundwater quality. During the first year of operation, NO_3-N, NH_3-N and organic N concentrations increased in some of the monitoring wells but decreased to lower levels after the first year.

In a study of unlined lagoons in the Coastal Plain soils in Virginia, Ciravola et al (1979) found that two anaerobic swine lagoons caused minimum groundwater contamination. A third lagoon contaminated groundwater with Cl and NO_3-N in excess of drinking water standards. Sewell (1978) found NO_3-N and Cl concentrations increased rapidly in groundwater taken from wells 15 m from an unlined anaerobic dairy lagoon during the first six months of lagoon operation. Later the NO_3-N concentrations decreased to levels similar to those before the lagoon was loaded. Median NO_3-N concentrations of all the test wells were below 10 mg/L. The lagoon was located in an area with silt loam and sandy loam soils to a depth of 1 m and a quartz sand horizon at 1 to 4 m. Nordstedt et al (1971) found that NO_3-N concentrations were above background levels in the groundwater in wells at a depth of 2 m and a 15 m distance from a dairy lagoon in a clay soil that had been in operation for 8 months.

METHODS AND MATERIALS

In 1982, after visiting a number of farms with anaerobic lagoons, two sites where lagoons with clay liners have been in operation for a number of years were selected for monitoring groundwater quality. The first site has an anaerobic lagoon that is 4.6 m deep for a 500 head hog finishing unit with a flushing system. The lagoon has been in operation for approximately 5 years. The second site is located on a farm that raises beef and hogs and has a slaughter house. There are a total of three lagoons and settling basin on the site. One of the lagoons receives waste from the slaughter house and one receives waste from a hog feeding operation. Both of these anaerobic lagoons have been in operation for approximately six years. In the fall of 1982, the farmer constructed a new 200 head beef cattle feedlot with a flushing system for cleaning and alleys. The manure is flushed several times a day into the settling basin and the effluent overflows into an anaerobic lagoon. All three lagoons and settling basins are clay lined and approximately 1.8 m deep.

At the first site a total of six monitoring wells were installed. All wells were placed a distance of 7.6 m from the lagoon. One well was 12.2 m deep, while the other five wells were 4.6 m deep. At the second site a total of seven monitoring wells were installed. One well was 15 m deep and another well was 11.3 m deep. All other wells were 4.6 m deep.

Polyvinylchloride pipe of 30 mm diameter was used as well casing. All of the shallow wells and the deep wells had 1.5 m of PVC screen with 0.25 mm slot width. All wells were installed by the auger drilling method. The anular space between the well casing and wall of each hole was sealed with bentonite from the ground surface to 1.8 m below the ground surface.

The monitoring wells at both sites have been sampled every other month since October, 1982. Samples have been taken with a battery operated pump. All samples have been analyzed for NO_3-N, NH_3-N, Cl and total dissolved solids (TDS) by procedures outlined in Standard Methods (APHA, 1980).

The soil on Site #1 is classified as an Evesboro loamy sand. The soil in the top 3.5 m is a medium sand with traces of clay. Below a depth of 3.5 m to a depth of 6.0 m the soil is composed of medium sand with a moderate amount of clay.

Site #2 has Sassafras sandy loam, Fallsington sandy loam and Pocomoke sandy loam soils. Both the Pocomoke and Fallsington soils are classified as poorly drained. The soil varies from a fine to medium sand with a fair amount of clay in the top 1.8 m. From 1.8 to 6.0 m the soil varies from fine to coarse sand with traces of clay and gravel in some areas.

RESULTS AND DISCUSSION

Mean concentrations of NH_3-N, NO_3-N, Cl and TDS are presented in Tables 1 to 4.

The average NN_3-N concentrations of the monitoring wells at Site #1 ranged from 4.48 to 640 mg/L. The highest concentrations occurred in wells 1, 2, 5 and 6. All of these wells are in the direction of groundwater flow from the lagoon. It appears the exchange sites on the soil particles are saturated with NH_3-N and that NH_3-N is moving through the soil profile to the water-table aquifer. Until the farmer stopped feeding hogs in the spring of 1984, he would empty the lagoon twice a year. This would allow the clay liner to dry out and develop cracks, so excessive seepage would occur.

Table 1. Ammonia Concentrations of Monitoring Wells.

Well No.	Well Depth (m)	Mean	Standard Deviation (mg/L)	Range
Site #1				
1	4.6	166	135	21.3-418
2	4.6	504	354	33.5-1057
3	4.6	4.48	4.00	0.12-11.2
4	4.6	6.63	7.47	0.44-12.7
5	4.6	640	532	16.9-1497
6	12.2	56.0	46.6	2.02-157
Site #2				
1	15.3	0.13	0.16	<0.05-0.49
2	4.6	0.71	0.33	0.28-1.32
3	4.6	0.85	1.62	<0.05-4.81
4	4.6	12.2	13.5	0.38-44.0
5	11.3	1.98	1.92	0.08-6.78
6	4.6	0.26	0.39	<0.05-1.16
7	4.6	4.27	11.2	<0.05-39.6

Table 2. Nitrate Nitrogen Concentrations of Monitoring Wells.

Well No.	Well Depth (m)	Mean	Standard Deviation (mg/L)	Range
Site #1				
1	4.6	1.81	1.49	0.30-6.23
2	4.6	7.79	9.43	0.20-29.1
3	4.6	38.5	27.6	1.92-78.1
4	4.6	47.4	28.2	11.2-109
5	4.6	3.35	1.98	0.72-6.81
6	12.2	17.7	15.0	4.28-54.3
Site #2				
1	15.3	0.23	0.16	<0.05-0.65
2	4.6	11.9	6.95	4.00-27.7
3	4.6	4.73	7.45	0.11-29.1
4	4.6	3.05	6.16	<0.05-23.5
5	11.3	0.60	0.98	<0.05-3.18
6	4.6	1.84	0.86	0.12-3.15
7	4.6	0.40	0.43	<0.05-1.59

Table 3. Chloride Concentrations of Monitoring Wells.

Well No.	Well Depth (m)	Mean	Standard Deviation (mg/L)	Range
Site #1				
1	4.6	117	68	45-281
2	4.6	270	156	108-540
3	4.6	11	7.0	3.9-25
4	4.6	21	14	7.8-46
5	4.6	238	105	66-447
6	12.2	41	14	22-66
Site #2				
1	15.3	13	4.4	6.7-21
2	4.6	163	49	102-262
3	4.6	47	23	11-88
4	4.6	64	38	2.9-144
5	11.3	32	20	14-88
6	4.6	10	2.0	7.9-14
7	4.6	38	13	8.5-57

Table 4. Total Dissolved Solid Concentrations of Monitoring Wells.

Well No.	Well Depth (m)	Mean	Standard Deviation (mg/L)	Range
Site #1				
1	4.6	800	468	231-1806
2	4.6	2555	1168	697-4399
3	4.6	482	200	181-719
4	4.6	620	268	285-1043
5	4.6	1191	700	253-1856
6	12.2	349	63	260-451
Site #2				
1	15.3	148	85	97-266
2	4.6	683	380	383-1812
3	4.6	200	78	86-337
4	4.6	497	395	201-1126
5	11.3	186	72	87-301
6	4.6	198	217	39-571
7	4.6	324	296	120-1061

Also, since the lagoon is 15 m deep, seepage may occur through the bottom of the lagoon with enough hydraulic head. The lagoon has overflowed several times in the vicinity of well 6 which may be causing the high NH_3-N concentrations in well 6.

Wells 3 and 4 at site #1 have the lowest NH_3-N concentrations, but have the highest NO_3-N concentrations. There is less clay in the vicinity of wells 3 and 4 than the other monitoring wells, so more nitrification may be occurring. Wells 3 and 4 are located close to the lagoon but not in the direction of groundwater flow, so they should not have as serious water quality degradation as some of the other monitoring wells.

Chloride concentrations are lowest in wells 3 and 4 and highest in wells 1, 2 and 5 at site #1. The chloride concentrations follow the same pattern as the total of the NH_3-N and NO_3-N concentrations. Wells 3 and 4 have the lowest NH_3-N and NO_3-N concentrations.

All monitoring wells have high TDS concentrations at site #1. Wells 2 and 5 which have the highest NH_3-N concentrations also have the highest TDS concentrations.

There has been considerable variation in NH_3-N, NO_3-N, Cl and TDS concentrations in all monitoring wells at site #1 over the three years of monitoring. Concentrations of pollutants were high in all of the monitoring wells when groundwater monitoring was initiated.

At site #2 only three of the wells have average NH_3-N concentrations above 1.0 mg/L. Wells 4 and 5 are located in the vicinity of the slaughter house lagoon. Ammonia concentrations have only recently increased in well 7. This well is lcoated near the settling basin constructed in 1982. Only one of the monitoring wells at site #2 has an average NO_3-N concentration above 10 mg/L. Nitrate concentrations in well 2 have been higher than any of the other monitoring wells since the project started. Nitrate concentrations in well 4 have only recently started to increase with the highest concentration reaching 23.5 mg/L. Some of the monitoring wells are located in poorly drained soils, so some denitrification may be occurring.

Chloride concentrations are higher in wells 2 and 4 than the other wells. These two wells also have the highest TDS concentrations. Either NH_3-N or NO_3-N concentrations are also higher in these wells than the other monitoring wells. Both Cl and TDS concentrations increased in well 7 in 1984 from 1983.

The lagoons at site #2 have not had near as severe an impact on groundwater quality as the lagoon at site #1. None of the lagoons at site #2 are pumped below design operating levels.

CONCLUSIONS

1. Clay lined livestock waste lagoons may degrade groundwater quality in the Chesapeake Bay watershed.
2. Clay-lined livestock waste lagoons should not be pumped below design operating levels or they may cause severe groundwater contamination as a result of the clay drying out and cracking and causing excessive seepage.

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A COMPUTER MODEL FOR THE PLANNING AND DESIGN OF URBAN BMP's

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ABSTRACT

An easy-to-use desk top model has been developed for use in personal computers to simulate watershed response to a rainfall event and to estimate nonpoint source pollutant loadings associated with the storm event. The algorithms utilize the SCS method for calculating runoff hydrographs for a single storm event. The pollutant loading transported by the runoff is assumed to be proportional to the amount of pollutants remaining on the ground surface for any time interval. In addition, this model allows for the design, evaluation and cost effectiveness analysis of various best management practice (BMP) measures as tools to control pollutants transported by runoff.

INTRODUCTION

A significant element in the protection of the water quality of the Chesapeake Bay is the management of stormwater from urbanizing areas. Implementation of management programs involves substantial investment in best management practices (BMPs) for control of runoff and pollutants transported by runoff. This paper presents a methodology developed to compare various management strategies on the basis of their impact on runoff and pollutant load at the outlet from the watershed and on the total costs involved.

HYDROGRAPH DEVELOPMENT

Algorithms have been developed to generate a runoff hydrograph for a single storm event using the method developed by the U. S. Soil Conservation Service (SCS). The SCS method calculates runoff based on the following equations:

$$Q = \frac{(P - 0.2S)^2}{(P + 0.8S)} \qquad (1)$$

$$S = 1000/CN - 10 \tag{2}$$

where:

Q = runoff amount (inches)
P = 24-hour rainfall amount for design storm (inches)
S = potential abstraction (inches)
CN = runoff curve number

The runoff curve number reflects such design information as soil types, antecedent soil moisture conditions, and the type, vegetative cover, and hydrologic condition of up to 78 different land use possibilities as described by the user. Other required information includes the rainfall depth for the specific frequency storm and basin times of concentration and times of travel. Hydrographs for sub-basins as well as a composite watershed hydrograph can be generated using the tabular SCS method.

BMP DESIGN AND EVALUATION

Algorithms have been developed to allow for the design and/or evaluation of three basic types of BMP structures. These structures are detention ponds (dry ponds, wet ponds, and extended wet ponds), infiltration trenches, and porous pavement.

Detention Ponds

Routing the inflow hydrograph through a detention facility was accomplished using the storage-indication working curve method, given by equation (3):

$$I_1 + I_2 + \frac{2S_1}{\Delta t} - O_1 = \frac{2S_2}{\Delta t} + O_2 \tag{3}$$

where:

I_1, I_2 = inflow rates at times 1 and 2, respectively

O_1, O_2 = outflow rates at times 1 and 2, respectively

S_1, S_2 = storage volume at times 1 and 2, respectively

Δt = time interval

The outflow rating curve and storage vs. elevation data must be provided by the user. A user supplied rating curve eliminates the need for user modification of the software to handle different types of outlet structures. The algorithm is repeated until the pond provides sufficient reduction of peak inflow to meet the design requirements. Once a satisfactory design has been found, the detention facility is sized depending on what type of pond has been specified. The outflow hydrograph produced by the routing procedure is lagged according to the relationship in equation (4) then linearly combined with previously calculated hydrographs.

$$\text{Lag Time} = T_{\text{peak outflow}} - T_{\text{peak inflow}} + T_{\text{travel}} \tag{4}$$

Infiltration Structures

The following methodology applies to both types of infiltration structures examined in this study, i.e. infiltration trench and porous pavement. An infiltration trench is defined as a subsurface trench that is used to temporarily store runoff in a stone filled reservoir and exfiltrate the runoff through the surrounding soil media (Maryland Dept. of Nat. Resources, 1984). The surface of the trench consists of either a stone covered area or a grass covered area with an inlet. Porous pavement is defined as a low density, permeable asphalt surface in which water is rapidly transmitted to an aggregate reservoir subbase for storage (Maryland Dept. of Nat. Resources, 1984). Water then infiltrates into the surrounding soil media. The design for infiltration structures is based on controlling the increased runoff for a specific frequency storm event. The volume of water that must be stored in the stone subbase to accomplish this purpose is given by equation (5):

$$V_w = \Delta Q_u A_u + P\, A_s - fTA_s \tag{5}$$

where:

V_w = volume of water that must be stored in the stone reservoir

ΔQ_u = increased upland runoff volume in inches

A_u = area upstream of facility site that contributes runoff to the site

P = precipitation

A_s = surface area of structure

f = infiltration rate (in/hr) of soil surrounding stone reservoir

T = stone reservoir filling time (value of 2 hrs found for designs based on SCS type II storm)

Utilizing the relation given in equation (6) for the gross volume of the stone subbase,

$$V_s = d_s A_s \tag{6}$$

in combination with equation (5) yields the design equation (7):

$$d_s A_s V_r = \Delta Q_u A_u + PA_s - fTA_s \tag{7}$$

where:

V_s = gross volume of stone subbase

d_s = depth of stone subbase

V_r = void ratio of stone subbase

Equation (7) may be solved based on user supplied information such as geometric limitations, amount of runoff to be handled by the structure, and elevation of groundwater table.

Once the infiltration structure design has been finalized, the program calculates a hydrograph for the drainage area by the SCS method which reflects the impact of the BMP installation. The BMP site hydrograph is then linearly combined with previously calculated hydrographs for the watershed to obtain a composite hydrograph.

POLLUTOGRAPH GENERATION

Pollutographs can be produced by the model to represent dynamic pollutant loadings during the storm event. Four water quality parameters are examined: total N, total PO_4, BOD, and total suspended solids. Loadings are based on the total mass of pollutants which has accumulated on the ground surface during dry days and the fraction of this mass which is removed by the runoff. The total mass of accumulated pollutants prior to the storm event is a weighted value accounting for pollutant dry weather accumulation rates (pounds/acre/day) for both pervious and impervious areas for all land use types. In addition, pollutant concentrations can be entered instead of mass accumulation rates; a separate algorithm translates these values into mass of accumulated pollutants. The amount of pollutant washoff in any time interval is given by the relationship

$$\frac{dp}{dt} = -\ kP \tag{8}$$

which integrates to

$$P_o - P = P_o\ (1 - e^{-kt}) \tag{9}$$

where:

P_o = initial loading (pounds or mg)

P = mass remaining after time, t

k = constant

t = time

$P_o - P$ = mass washed away in time, t

The constant, k, a function of runoff, was determined by assuming that a uniform runoff of 0.5 in/hr will wash off 90% of the initial pollutant load in one hour (Wanielista, 1978). Thus, equation (8) becomes (Wanielista, 1980)

$$P_o - P = P_o(1 - e^{-4.6rt}) \quad \text{for impervious areas} \tag{10}$$

$$P_o - P = P_o(1 - e^{-1.4rt}) \quad \text{for pervious areas} \tag{11}$$

where:

r = rainfall excess (in/hr).

To account for the fact that the runoff rate is usually not constant for a storm event, the rainfall excess term in equations (10) and (11) is calculated over each time interval to obtain an average value as follows:

$$r_{avg} = \text{Delta } Q/\text{Delta } t \tag{12}$$

Average runoff rates are calculated for the runoff from pervious and impervious areas separately. Rainfall excess from pervious areas and impervious areas is found using the SCS runoff equation (Eq. 1) based on assigned curve numbers for each type of area. Type II storm rainfall distribution in 24 hours is adopted to find rainfall excess for a delta t with a typical value of one hour. Applying equations (10) and (11) with the averaged runoff rates to both types of pollutant loadings yields a total amount of pollutants washed off during a time period; concentration is obtained through the following relationship:

$$\text{Conc (mg/l)} = \text{Mass washed/Delta Q} \quad \text{for each delta t} \qquad (13)$$

Pollutographs are calculated using equation (13) for each of the four pollutants analyzed for each subarea in the watershed as well as composite pollutograph for each pollutant for the entire watershed.

COST ANALYSIS

A basis for evaluating alternative urban stormwater management investments was developed after extensive literature review (Allison, 1985). Three cost factors are considered in this analysis: base construction costs, contingency costs, and operation and maintenance costs. Table 1 presents equations developed to facilitate BMP evaluation (all costs in fourth quarter 1980 dollars).

TABLE 1 -- BMP COST EQUATIONS

BMP Type	Construction Costs (C)
Dry pond	$\$77.4 \times V^{0.51}$
Wet pond	$\$77.4 \times V^{0.51}$
Extended Wet Pond	$\$77.4 \times V^{0.51}$
Infiltration Trench	$\$62.12\ V^{0.51} + 0.30(D \times L) + 0.57V + 0.23(L \times W)$
Porous Pavement	$\$62.12\ V^{0.51} + 0.30(D \times L) + 0.57V + 1.75(L \times W)$
	Contingency Costs
Dry Pond	C x 0.25
Wet Pond	C x 0.25
Extended Wet Pond	C x 0.25
Infiltration Trench	C x 0.25
Porous Pavement	C x 0.35
	Annual Operation & Maintenace Costs
Dry Pond	C x 0.05
Wet Pond	C x 0.05
Extended Wet Pond	C x 0.05
Infiltration Trench	C x 0.03
Porous Pavement	$6.50/cu.yd.

where:

C = construction cost ($)
D = depth of stone sub-base (ft)
L = length of facility (ft)
W = width of facility (ft)
V = volume of facility (cu. ft.)

The figures given in Table 1 do not reflect the cost of the land. The volume range of the structures used to develop the relationships given for construction costs was 2000 - 800,000 cubic feet (Allison, 1985).

APPLICATIONS

For purposes of illustration, the software package was applied to a hypothetical 1000 acre watershed under the following conditions: (1) 5 year design storm with 24 hour rainfall depth of 8 inches; (2) 2 sub-basins within watershed: sub-basin 1 has agricultural land use; sub-basin 2 has commercial and residential land uses; (3) Pollutant loading rates applied were typical of the Northern Virginia area (Biggers, et al. 1980); (4) BMP pollutant removal rates were obtained from Hartigan, et al. (1980)

Four BMP facilities were designed and located within the hypothetical watershed. Figures 1 - 3 show the pre- and post-BMP hydrographs generated for the design conditions for sub-basin 1, sub-basin 2, and the entire watershed. The 0.35 million cubic foot extended detention pond installed in sub-basin 1 delays and reduces the peak runoff for that sub-basin, as can be seen in Figure 1. The combined effects of an infiltration trench, a porous pavement facility, and a 0.51 million cubic foot extended detention pond on the runoff produced in sub-basin 2 is illustrated by Figure 2. Figure 3 shows the combined effect of all four BMP structures on the hydrograph generated at outlet of the watershed.

In addition, the effectiveness of the BMP installations in controlling runoff quality can be seen by Figures 4 - 6. The amount of pollutant removal attained by the BMP structures can be seen in the pre- and post-BMP pollutographs generated for the individual sub-basins for the pollutant total suspended solids, shown in Figures 4 and 5. The combined pollutant removal achieved by all BMP structures installed in the watershed can be seen in Figure 6, which illustrates the pollutograph generated at the watershed outlet for this pollutant. Other types of pollutants may be analyzed in a similar manner.

Thus, the main application of this model is to demonstrate and evaluate the effectiveness of the BMP on runoff quantity and quality. This model is at a medium level of sophistication, meant to be used as a management tool for estimation of the location, size, and type of control measures needed to attain a given level of runoff and pollutant control at the outlet of the watershed. In other words, detailed simulation of pollutant source and buildup, washoff and transport/deposition is not performed. The graphical and/or tabular output facilitates comparison between pre - and

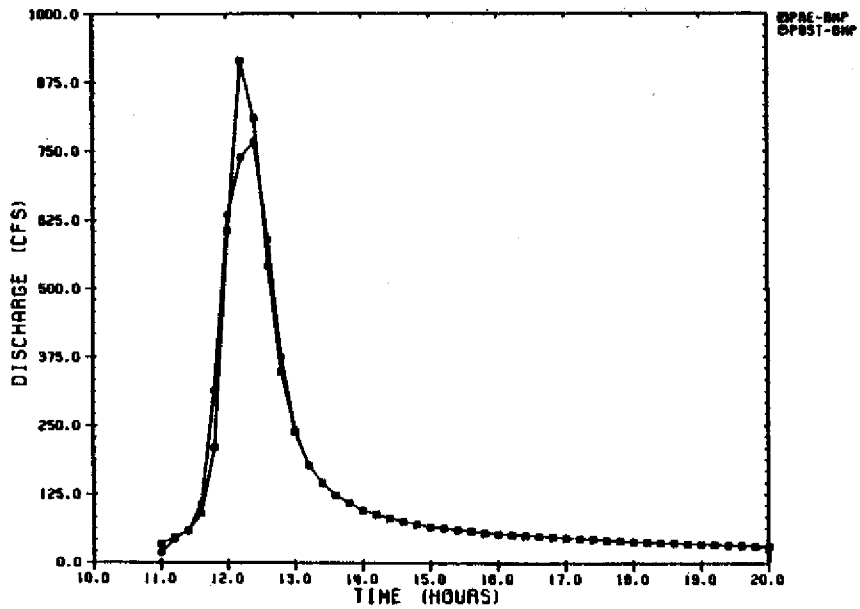

Figure 1. Subbasin 1 Hydrographs

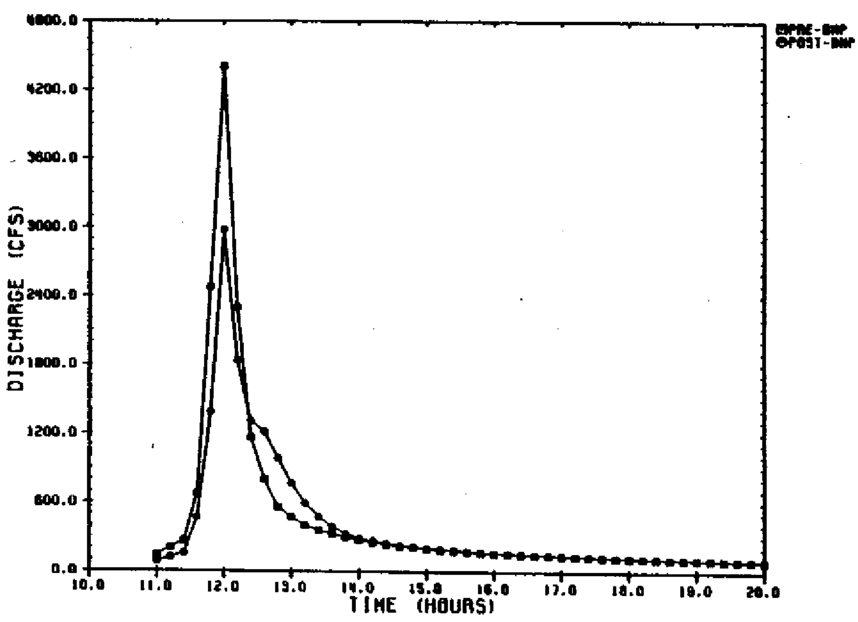

Figure 2. Subbasin 2 Hydrographs

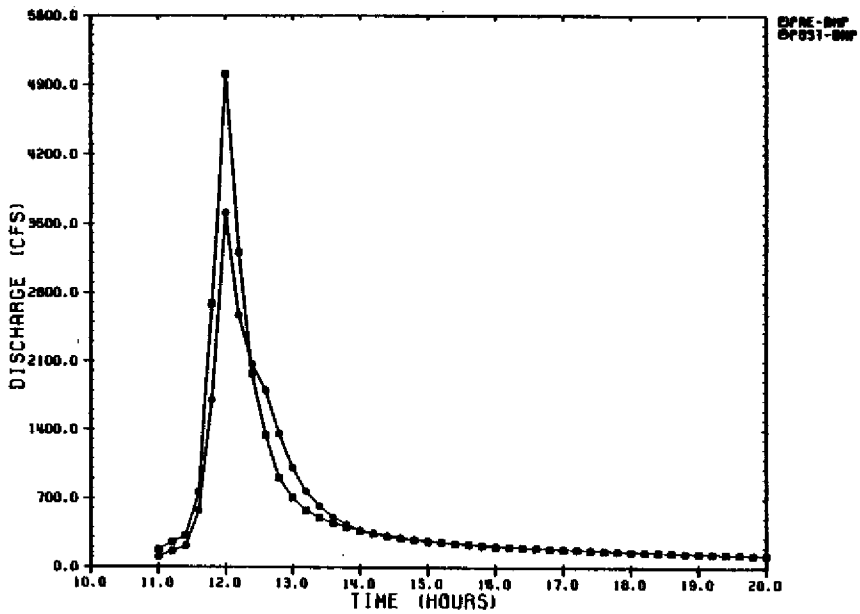

Figure 3. Composite Watershed Hydrographs

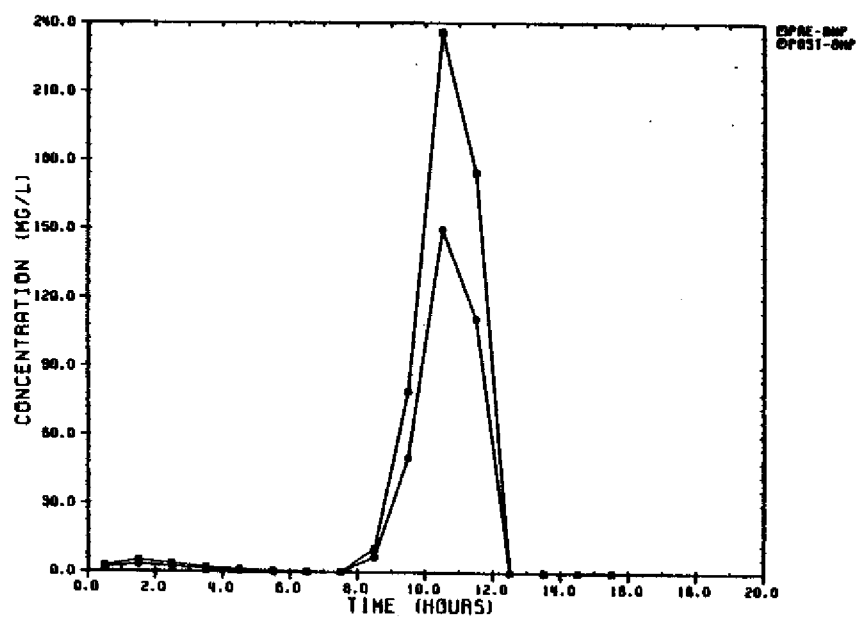

Figure 4. Subbasin 1 Pollutographs

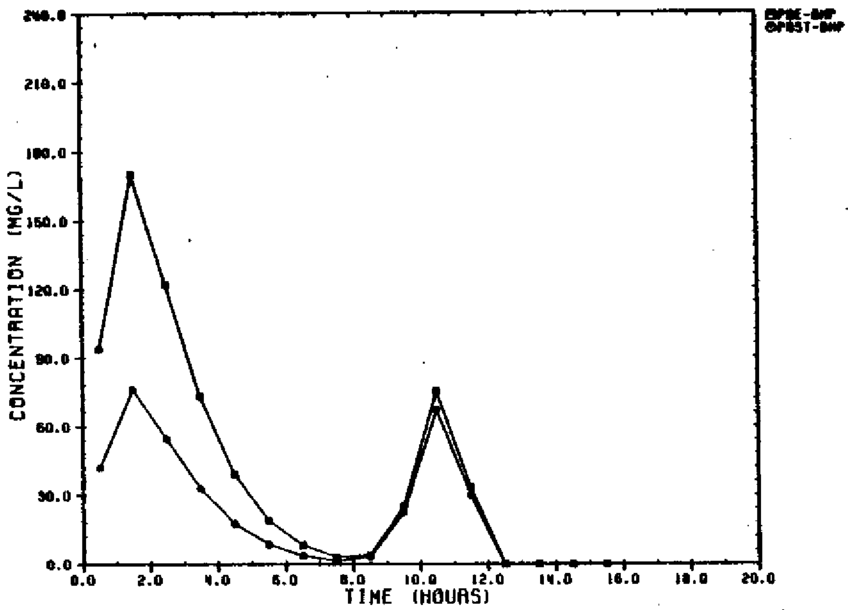

Figure 5. Subbasin 2 Pollutographs

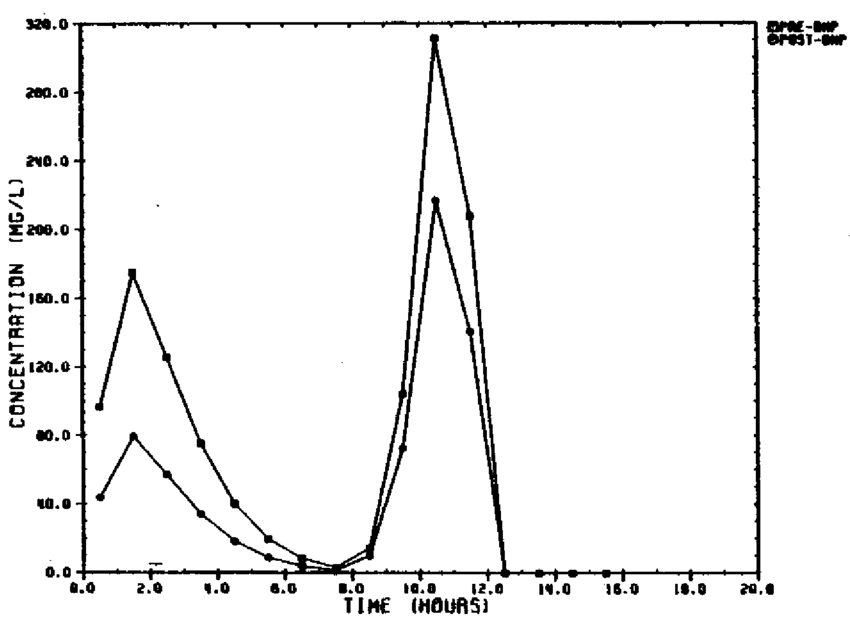

Figure 6. Composite Watershed Pollutographs

post-development conditions for planning purposes. In addition, this model provides an order of magnitude analysis of the cost effectiveness of alternate management strategies.

DISCUSSION

This model is limited to small watersheds ($<$ 2000 acres) as dictated by SCS TR-55. In addition, the SCS tabular method used in this model requires the time of concentration for each subarea to be less than 2 hours and the travel time to be less than 4 hours. Limitations of the version of BASIC used in programming prohibit the design and/or evaluation of large numbers of BMP structures in a large watershed.

In order to fully capture first flush effects on runoff quality from impervious areas, the time increment used in the washoff equations (Equations 10 and 11) should be made as small as possible. Limitations of the micro-computer and the version of BASIC used in this study restricted the time increments to one hour; better results can be obtained using a smaller time increment.

ACKNOWLEDGEMENT

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WASTEWATER DISPOSAL IN ESTUARY-CONNECTED GROUNDWATER

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ABSTRACT

Bermed infiltration ponds (BIP) are used for wastewater disposal in low coastal areas where there is a confined aquifer near the ground surface. That saturated zone is below mean low water of the Chesapeake Bay and receives water from the pond due to hydraulic head. The permeable zone (2-3 m thick) is bounded above by silty top soil and below by a silt-clay layer that is 3-4 m below ground surface. Water quality measurements in several ponds and surrounding ground-water are reported. For example, chloride concentrations in the groundwater were reduced by fresh water of the pond. Bacterial concentrations in the pond were higher than levels recommended for swimming. Fish sampling indicated satisfactory survival of bluegills. The ponds generally have 0.10 ha water surface and are located on residential lots larger than 1.6 ha. This research reports on one method to reduce contamination of shellfish waters by eliminating home sewage pollution.

INTRODUCTION

Along the shores of Chesapeake Bay where early American colonists settled, poorly drained soil for homesites was not a major problem until indoor plumbing was installed. Increasing house water use caused traditional septic systems, designed for better drained soils, to fail during high water table conditions of the winter season. Wastewater standing in yards and roadside ditches created health and odor problems. A collection sewer system was not an economically feasible alternative

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for the dispersed homes of farmers and watermen of the region. One disposal method found acceptable for this area has been the bermed infiltration pond (BIP). Generally, a single family BIP has about 0.10 hectare water surface, is 2-4 meters deep and receives effluent of a little less than secondary treatment quality. Pond water is pushed out into the confined groundwater by hydraulic head in the pond.

Historically, environmental health officials have had to rely on the Manual of Septic Tank Practice (USPHS, 1967) and state regulations based thereon. Information on alternative disposal methods was not dependable and adherence to regulation reduced the potential for inconsistency. Consequently, drainfield tiles were placed at the prescribed distance below ground surface independent of soil conditions. Increased understanding of saturated soil hydraulics suggested the need to design a disposal system for existing soil conditions. Since potable water was available at about 60 meters deep and near-surface groundwater at less than about 6 meters was poor quality, the feasibility of saturated soil disposal was considered. Water quality characteristics of the near-surface aquifer indicated high chloride concentrations and total dissolved solids greater than 1500 mg per liter (Lomax and Stevenson, 1982).

In addition to these measurable constituents, bad odors also make this aquifer unsuited for domestic wells. It was recognized that the Chesapeake Bay and its tributaries were probably connected with this groundwater layer. Geologically, the Eastern Shore of the Bay is a sedimentary river delta made of layers of clay, silt, and sand (Foss et al, 1978). The seasonally fluctuating water table varied from the ground surface in late winter to about 3 meters deep in the early fall. Most of the homesites are less than 3 meters above mean sea level.

A bermed infiltration pond was so named because of its character- istics. The soil removed during excavation is distributed around the perimeter to form a berm with a gradual slope and a height above original grade of 0.5 meter. Pond water is part of the near-surface aquifer and thus disposal occurs by infiltration into the groundwater. During periods of no wastewater input, the saturated soil connection keeps water in the pond. This connection is one reason that pond is more descriptive than "lagoon", since the latter is normally separated from the groundwater. And incidentally, the word pond has a pleasant connotation.

Installation of a BIP requires a specific soil profile (Figure 1). The top meter of the profile is a silt to clay type of soil with very low permeability. During periods of high water in the pond, this upper layer is intended to function as a top confining zone, preventing pond water from surfacing outside of the berm. Between one and three meters below ground surface, there should be a relatively permeable, saturated zone. Below the permeable zone is a tight silt or clay layer more than a meter thick that confines the near-surface aquifer and separates it from deeper groundwater. The seasonal water level fluctuation in this permeable zone showed

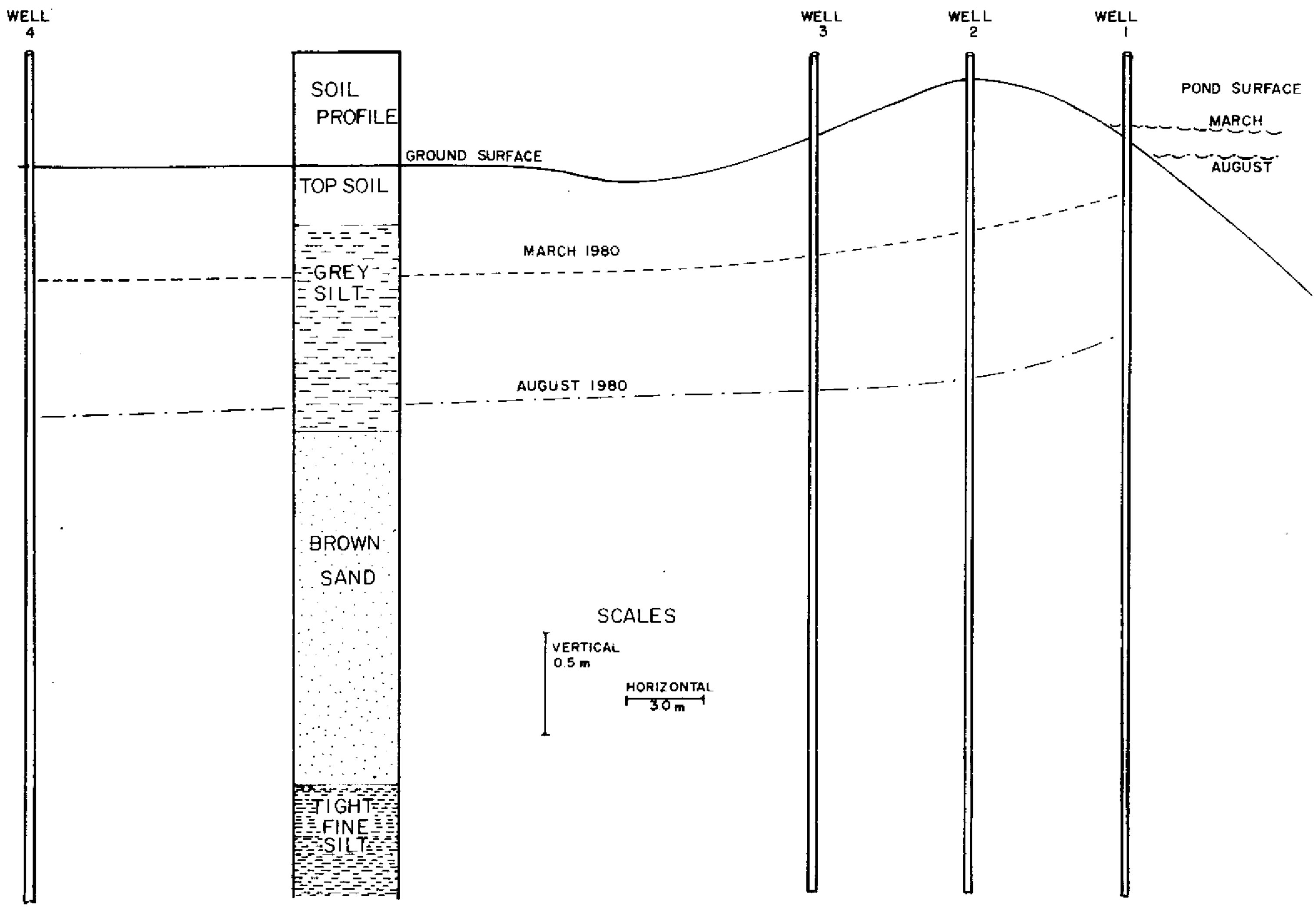

Figure 1. Selected piezometer profiles for well transect at school Bermed Infiltration Pond.

multiple gradient directions in one location for different seasons and rainfall (Christy, 1980). Changing gradient directions may mean that pond water is dispersed in more than one direction.

Literature related to pushing water into saturated soil is limited, with the work of Wood (1976) on deep well injections being the most pertinent in this context. Development of a water table mound has been described mathematically for an underground lake type of water table receiving percolating effluent (Fielding, 1981; Willis, 1976); however, little information was available on direct effluent disposal into a variable potential aquifer. Saturated flow theory would suggest that a water table mound would result from a wastewater disposal system in a near-surface aquifer. Actual measurements of a water table mound around a BIP were needed to support development of regulations.

The purpose of this study of bermed infiltration ponds was to evaluate their ability to dispose of wastewater and the subsequent environmental effect.

METHODS

Evaluation of bermed infiltration ponds included three aspects, hydraulic and water quality measurements and fish sampling. The hydraulic evaluation occurred at a BIP constructed in 1977 to serve an elementary school and hereafter referred to as the school pond. A transect of four piezometers was installed away from one end of the pond and water table elevations were measured to the nearest 3.2 mm (1/8 inch) once per month. The piezometer locations and soil profile are shown on Figure 1. An accumulating water meter recorded the domestic water use which was considered equivalent to the wastewater discharged to the 0.25 ha pond.

Water quality effects of a BIP were evaluated at the school pond and a residential pond. During the period 1977-1979, monthly samples for the school site were taken from the pumping chamber that followed the three step anaerobic treatment unit, from the pond, and from the four piezometer wells. Since 1979, samples were taken annually. Samples were transported on ice to the laboratory and analyzed for chloride, nitrate nitrogen, ortho-phosphate, total phosphorus, and biochemical oxygen demand (BOD). Nonfiltrable residue and pH were measured in the effluent and pond samples. Filtrable residue (dissolved solids) was measured in well samples. All procedures were those described in Standard Methods (APHA, 1975). Bacterial assays included total and fecal coliform tests using the most probable number (MPN) technique.

The water quality of an established BIP, constructed in 1971 to serve two residences, was evaluated from 1971 to 1981 (Coble, 1981). Samples taken from the residential pond were analyzed for chloride, nitrate nitrogen, total phosphorus, total solids, pH, and total and fecal coliforms. Chloride and nitrate concentrations and coliform numbers were measured in five surrounding sample wells, 15-270 meters from the pond.

Several other residential BIP's were sampled with a hand seine to check on the survival and reproduction of stocked fish. A one meter deep, 15 meter long seine was stretched across a corner of a pond, pulled in to shore and then spread on the berm. We identified and counted the fish before putting them back into the pond. This procedure was repeated three to five times at each site.

RESULTS AND DISCUSSION

Hydraulic Measurements at School Pond

The hydraulic evaluation of a BIP indicated that, as expected, the pond water level did rise above the surrounding water table. Figure 1 shows two measured water levels in the pond and piezometers at the time of normal seasonal extremes. Throughout the seasonal fluctuations in natural water table, the pond surface was always 0.5 m or more above the receiving ground water. The school pond dispersed 37,000 to 67,000 liters per month according to measured domestic water use. The hydraulic gradient shown on Figure 1 illustrates how a BIP works to dispose of wastewater in saturated soil. During part of the year, the pond surface reached Well #1 but the water level within the well was always lower. This observation was evidence of a tight well casing. At the school, the pond water level never exceeded the berm nor were wet spots found outside the berm. These measurements and observations suggested that the BIP was acceptable from a hydraulic perspective.

Water Quality Measurements

The water quality evaluation of BIP's showed that the receiving groundwater was not adversely effected by infiltration from the ponds. The water quality measurements for the residential BIP studied during 1974 and 1975 indicated differences between the pond water and surround groundwater. Table 1 shows the bacteria measurements and traditional sewage indicators chloride and nitrate. The minimal coliform counts and low nitrate concentrations surrounding the pond were encouraging and thus additional BIP's were built in the county.

Table 2 shows the results of monthly sampling at the school pond site. The treatment unit effluent was lower in BOD than would be expected from a residence. For example, Karikari et al (1975) reported an effluent BOD mean value of 134 mg oxygen per liter for the home system of their study, compared with 79 mg per liter from the school. Neither BOD nor dissolved oxygen were measured regularly in the school pond.

Inorganic chemicals such as nitrate and chlorides are the normal concern relative to groundwater. The chloride concentration in the treatment unit effluent was much lower than groundwater values and thus the pond had lower chlorides than background. This observation at the school pond confirmed earlier results from a residential BIP (Table 1). Nitrate nitrogen was lower in the pond than the treatment effluent which indicated dilution, photosynthesis or probably both occurred in the pond. Orthophosphate reduction in the pond may have been supporting evidence for photosynthetic activity.

Bacteriological results for the school pond indicated that swimming should be discouraged. There was significant reduction of both total and fecal coliforms relative to the treatment effluent, and the number of fecal bacteria measured in the pond (89 MPN/100 ml) was below the recommended water contact limit of 200 MPN/100 ml (Maryland Department of Health and Mental Hygiene). The absence of fecal coliforms in the well samples (#1-3) suggested that the bacteria were not travelling with the hydraulic gradient. The high average bacteria levels for Well #4 resulted from samples later in the study period and appeared to be from foreign material added to the well. The total coliform number for Well #1 did not show any trend as a function of time so that progressive movement of bacteria was not observed.

Since 1979, the most notable change in water quality has been an increase in bacteria concentration in the pond. Measurements of fecal coliform counts have ranged from 79 to 7900 MPN per 100 ml in the pond water samples. Nitrogen and phosphorus concentrations were similar in 1983-84 to those of 1977-1979. Nitrate concentration in the wells ranged up to 0.7 mg N per l. Total phosphorus in the wells averaged 0.70 mg P per l.

Fish Sampling

Sampling the ponds for fish showed that all but one pond of a total of eight sampled were healthy environments. In June, 1976, the variety of fauna found in seven ponds included bluegill, killifish, mosquito fish, and tadpole. One small pond had very little phytoplankton at sampling time and no rooted vegetation along the bank. This pond had no observed fauna and was classified unhealthy at that time. The surface area of this pond was smaller than is presently recommended for a BIP. Although not all ponds contained bluegill, as the homeowners may have wished, none of the ponds had mosquito larvae. Several of the same ponds were sampled again in November 1977. At this time, the same fish species were not observed at a given pond. The inconsistent sampling results suggested more intensive study was needed, but such an effort was not funded. A third survey of four ponds in July 1979 found fish in three ponds and extensive vegetation in the fourth. The size of bluegills in these ponds indicated consistent reproduction in two ponds and no recent young in the other. Although the fish communities of the BIP's may not be managed for optimum productivity, survival and growth of the bluegills were encouraging. Mosquito predation within the ponds was considered good. Homeowners were satisfied with their BIP as a wastewater disposal system.

TABLE 1. Water Quality Measurements for Residential Bermed Infiltration Pond and Surrounding Groundwater, 1974-1976.

Parameter	Pond	Groundwater (5 wells)
Total Coliform, mean MPN/100 ml	280	23
Fecal Coliform, mean MPN/100 ml	17	None Detected
Chloride, mg Cl/l		
mean	313	2133
range	240-590	42-5510
Nitrate Nitrogen, mg N/l		
mean	0.33	0.04
range	0.10-0.30	0.00-0.08

Regulatory Guidelines

As a result of the hydraulic, water quality, and fish evaluation efforts, BIP guidelines have now been established. A minimum lot size of 1.6 ha (4 acres) and the specific soil profile described above are necessary site requirements for a BIP along the Chesapeake Bay. Pond surface at original grade should be 0.10 ha (0.25 acres) or larger. Treatment of the wastewater entering the pond may be by 1) an aerobic unit, 2) a four chamber septic tank, or 3) a septic tank followed by sand filter (2000 gallon capacity total). Fish should be stocked in the pond for mosquito control.

Homeowner maintenance of a BIP and treatment unit is more than that required for a traditional septic system. These various management duties have not limited, thus far, the acceptance of the system by either homeowners or environmental health officials. Although the cost of the BIP exceeds that of a drainfield, the absence of wastewater puddles and smelly ditches makes the BIP a substantial improvement over failing septic systems.

TABLE 2. Water Quality Measurements for Bermed Infiltration Pond Receiving Anaerobic Effluent Mean Values, 1977-1979.

Parameter	Treatment Effluent	Pond	Well #1	Well #2	Well #3	Well #4
Total Coliform MPN/100 ml	1.18×10^6	902	254	ND*	ND	305
Fecal Coliform MPN/100 ml	$.238 \times 10^6$	89	ND	ND	ND	11
Chloride mg/Cl^-/1	93	402	560	4100	2847	1012
Nitrate Nitrogen mg N/1	3.2	0.5	1.0	4.0	1.0	0.6
Ortho-Phosphate mg P/1	5.63	0.04	0.15	0.06	0.05	0.02
Total Phosphorus mg P/1		0.10	0.35	0.18	0.08	0.13
Non-Filtrable Residue mg/1	45	35				
Filtrable Residue mg/1			1093	7682	5955	1230
pH	7.4	8.3				
Biochemical Oxygen Demand, mg_2/1	79					

*ND = None Detected

NOTE: Well locations are shown in Figure 1.

CONCLUSIONS

The bermed infiltration pond was evaluated for hydraulic characteristics in the near-surface groundwater and found to provide dependable disposal of effluent. Water quality effects on groundwater from a BIP were found to be minimal. Chloride and nitrate concentrations in the groundwater were reduced by the fresh water of the pond. Fish sampling indicated satisfactory survival of bluegills. Guidelines for the siting and design of BIP's have been established.

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for "Wastewater Disposal in Estuary-Connected Groundwater"

by K. M. Lomax and E. W. Asplen

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A STRONG WATER-CONSCIOUS ETHIC IN THE BAY REGION

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ABSTRACT

Colonial emphasis on the Bay as an important navigational resource surrounded by accessible, arable land gave way to a post-colonial era characterized by multiple resource conflicts. Entrenched special interests have recently been overcome to a significant extent as marked by the signing of the Chesapeake Bay Agreement of 1983. The recent federally-funded Bay research effort has made a significant contribution to the development of the current management program set into motion by that agreement. However, more critical factors may be the re-evaluation of the Bay as a regional amenity and the public perception of equity associated with Bay management decisions.

INTRODUCTION

As the third century of Chesapeake Bay management ends, it is important to reflect on the change in human values ascribed to the Bay. Taking this historical approach provides the necessary background for understanding the forces that led up to the signing of the Chesapeake Bay Agreement of 1983. The works of Middleton (1984), Reps (1972), and Capper and others (1983) shed light on the development of attitudes about the Bay.

Following an historical overview, a more detailed look at present-day values is presented with emphasis on social and economic trends. These trends have contributed to a change in both public and private sector attitudes about the importance of protecting and enhancing the Bay.

Finally, the roles that re-evaluation and equity have played in launching the current Bay management program are explored.

HISTORIC OVERVIEW

In 1607, the "seed of the British Empire" and ultimately the United States was successfully, though not without difficulty, established in the Virginia colony.[1] That seed was nurtured by two widely held views of the Bay at the time. First, through empirical knowledge gained by earlier sixteenth century explorers, the colonists knew the Bay to be a very navigable body of water and the land associated with the region to be especially accessible due to the physiography of the Bay and her many tributaries. Second, contemporary writers of the day fostered romantic notions about the Bay and contributed to an exalted view of her beauty and potential.[2]

The navigability of the Bay and accessibility of the surrounding land rapidly led to the establishment of an agrarian society that thrived economically as a result of trade with Great Britain. Trade with the mother country was dominated for many years by one commodity - tobacco. The success of tobacco as a stable in the Tidewater colonies was the result of its popularity as a medicinal and social herb in Britain; the desire of British rulers to develop colonial outposts that would help develop a favorable balance of trade; and the relative ease with which it could be cultivated in and transported from the colonies to Great Britain.[3]

During this period, "applied research" concerning the Bay was undertaken to support improved navigation. In fact, navigation was considered important knowledge among members of the educated classes in the Chesapeake region.[4] Captains of ships approaching the Bay from the Atlantic would frequently take soundings of the bottom while still out of sight of land. By noting both the depth and character of the sediments, a good mariner could determine his distance from land and could check his position relative to lines of latitude. Above Cape Charles and below Cape Henry (latitude 37°) the bottom is composed of hard, reddish sand. South of Cape Charles, a lead and tallow sounding would bring up mud mixed with sand and small oyster shells.[5] Thus, soundings were a useful way to check a ship's course while preparing to enter the Bay.

The Bay's many shallows and shoals proved a challenge to colonial mariners. However, it wasn't until 1731 that a concerted effort was made to chart Chesapeake bathymetry. This work, initiated by the Virginia Council, was interrupted.[6] Nevertheless, most likely as part of a privately financed venture, the first large-scale chart of Bay hydrography appeared in 1735. This chart is credited to Captain Walter Hoxton who was associated with a London tobacco firm.[7]

The value of the Bay for navigation and the nature of colonial trade with Great Britain profoundly affected patterns of land use in the Tidewater. The use of land, in turn, led to dramatic consequences for many of the Bay's tributaries.

The very navigability of the Bay has been directly linked to the failure of the colonial mercantile-planters to establish towns in the English tradition. The London Company tried repeatedly to encourage the formation of "compact and orderly" towns.[8] But the increasing tobacco market continued the trend of large private holdings, many with their own landings at the water's edge, well into the 1700's. It wasn't until the mid-1700's that towns finally took hold in the region. River towns such as Yorktown, Port Royal, Urbanna, Piscataway, Port Tobacco, and Joppa grew up around tobacco inspection houses after inspection acts were passed in 1730 and 1747 by Virginia and Maryland, respectively.[9]

The massive clearing of land and intensive cultivation of tobacco took its toll on the hydrology of many of the rivers. Erosion and siltation were greatly accelerated. Colonial regulatory actions attempted to address these problems more by treating the symptoms than by treating the causes. For example, in 1679 Virginia delegated to its Tidewater counties the power to "clear the rivers from loggs and trees, which may annoy and endanger boates, and sloops."[10] The dumping of ship's ballast was also subject to regulation.

Deforestation continued, however, extending into the Piedmont uplands. This led to even more dramatic impacts as a result of increased freshwater flows during storms. Destructive freshets reached their peak during colonial times in 1771 with a flood that submerged many of the river towns and ruined acres of fertile land with deposits of sand and stones."[11]

Eventually, many of the once-flourishing river towns were eventually cut off from vital access to the main Bay. In addition, tobacco cultivation had exhausted the soils in the older Tidewater lands, forcing a shift to grain crops and other commodities. After the Revolutionary War, tobacco production was still important but had shifted to the Piedmont region.[12]

In contrast to the Tidewater experience, the Piedmont region was settled by agriculturalists who were dependent upon overland transportation to get their goods to market. The Bay still provided the crucial navigational link between both the Piedmont and Tidewater regions and the major trade routes which had expanded to include not only Britain, but southern Europe, the West Indies, and the other American colonies as well. Due to the lack of direct access to the Bay, activity in the Piedmont became centered around towns located along the fall line. In this way, Baltimore, Upper Marlboro, Georgetown, Richmond, and Fredericksburg became thriving communities.[13]

In the towns located along the fall line, the water courses that eventually found their way to the Bay became valued for other than navigational reasons. The geologic coincidence that, in fact, rendered the upper portions of these waterways non-navigable, endowed them with energy that could be harnessed to drive mechanical devices. Thus, an agrarian society previously devoted to the production and exportation of raw agricultural commodities could become increasingly more engaged in the manufacture of refined goods such as milled flour and textiles. By the latter half of the eighteenth century, the focus of shipping had shifted to Baltimore in Maryland and to Norfolk in Virginia.

It wasn't until the post-colonial era that Chesapeake Bay became highly valued for her aquatic resources. Actually, the early colonists supplemented their English beef-oriented diet with seafood from the Bay's bountiful waters and engaged in fishing for sport from time to time. However, a conscious decision on the part of Great Britain to restrict salt importation to the Bay colonies in order to further stimulate tobacco production apparently succeeded in discouraging commercial fishing ventures. Without access to reasonably priced salt, fish could not be adequately preserved on a large scale.[14]

Following the Revolutionary War, the inhabitants of Maryland and Virginia became engaged in commercial seafood harvesting. In fact, the famous Compact of 1785 was signed by the two states to establish fishing rights for Virginians in Potomac waters (legally, part of Maryland) and to provide for the free passage of ships bound for Maryland through Virginia waters of the Bay. The compact was deemed to be such a successful compromise that it became a post-colonial model for settling interstate disputes.[15]

While the waters of the Bay were from the outset valued by European settlers for navigation and, somewhat later, for their abundance of seafood, these same waters, being brackish by nature, were not held in high regard as a drinking water source. Early colonists dug wells or, where available, used natural springs for supplying drinking water.[16] Combined with an ignorance of the germ theory of disease and the potential role of water as a disease carrier, this negative evaluation of much of the surface water associated with the Bay and her tributaries led to a general disregard for maintaining water quality. In fact, conscious decisions were made to use waterways as conveyances for waste - first domestic, then industrial in origin.

But beyond the estuary, fresh surface waters could be and were tapped as water supply sources. Nonetheless, conflicting uses of these waters led to the need for government regulation. In establishing the Baltimore Water Company in 1808, the Maryland legislature made provisions for fining anyone caught polluting the Jones Falls in the vicinity of the pumping house.[17]

Another public health issue related to surface water that received increased attention during the 1800's was the fear of epidemic diseases such as cholera, malaria, and yellow fever. These became associated with stagnant water in mill ponds, harbors, and wetlands. Laws and policies condemning mill ponds and encouraging the filling of wet, lowlands were popular in the latter part of the nineteenth century.[18]

By 1900, a continually expanding population in Maryland and Virginia had developed a pluralistic view of the value of Chesapeake Bay and her many tributaries. The use of surface waters for navigation; as sources of energy, drinking water and seafood; for swimming, aesthetic enjoyment, and ultimately, waste disposal created a resource competition that grew evermore difficult to mediate and control. Legal cases were frequently decided based on the perceived threat to human life. However, courts were much less likely to side with plaintiffs that could not clearly demonstrate a cause-and-effect link between alleged water pollution and human health and safety. Cases proving even less successful were those arguing that pollution or alteration of watercourses was adversely effecting aquatic life. In a precedent-setting Virginia case back in 1833 involving a stagnant mill pond, the court asserted its authority to "prevent the destruction of health and life," but distinguished between danger to health and danger to fish and navigation. The court ruled that it could allow experiments with the latter but not with the former.[19]

During the early 1900's, declines in finfish and oyster production brought accusations that pollution was being used as a "scapegoat" for over-fishing.[20] However, the growing need of an expanding population for sewage disposal contributed to the sense that pollution was indeed being allowed to outstrip the assimililative capacity of the Bay and her tributaries. In Maryland, the staunch oyster lobby championed the cause of improved sewage treatment. Thus did Baltimore become the first major city in the United States to adopt a wastewater treatment system. This accomplishment is historically noteworthy because it was really the oyster industry, rather than public health, that was being directly protected.[21]

The mid-1900's witnessed a raising of the national conscience about cumulative, long-term environmental impacts. By the 1960's, works like <u>Silent Spring</u> caught the attention of a public whose environmental values seemed to be lying dormant, waiting to be shaken by the reality of major degradation being relentlessly brought about by human action. Scientists spoke increasingly about the "systems approach" to analysis and values ascribed to the environment began to reflect the ecological concepts of population dynamics and resource limitation. In terms of the Bay, this period was marked by an emphasis on improving resource management.

Nationally known water quality management issues of the day tended to focus on freshwater bodies such as the Great Lakes, the Hudson River, and Lake Tahoe. Finally, in the late 1960's, attention was paid to estuarine pollution by special federal studies that included components devoted to the Chesapeake Bay. In 1968, the first Maryland Governor's Conference on Chesapeake Bay was held.[22]

The change in environmental perspectives on a national level which accelerated during the sixties and the many unanswerable technical questions that plagued Bay managers led to the massive commitment of $27 million in federal funds for scientific research on the Bay from

1975-1983. The signing of the now historic intergovernmental Chesapeake Bay Agreement in December of 1983 marked the beginning of a formal commitment on the part of Maryland, Virginia, Pennsylvania, the District of Columbia, and the federal government to launch a management program based on the findings and recommendations of the Bay research effort. Almost overnight, the Agreement seems to have achieved national and, to a more limited extent, even international recognition among environmental managers. The consensus embraced by the Agreement will no doubt be recorded as an historic achievement in much the way that the Compact of 1795 came to be recognized as a model for interstate compromise.

RE-EVALUATION OF THE IMPORTANCE OF CHESAPEAKE BAY

Years of conflicting resource use led to entrenched positions regarding the management of Chesapeake Bay. Commercial fisherman, sportsman, municipal managers, industrialists and environmentalists had each entered the arena of dispute concerning the Bay at one time or another in the recent past. How, then, was it possible to overcome many self-serving interests in order to achieve the degree of consensus over future Bay management represented in the Agreement of 1983? Three critical factors related to this question are presented below.

First, the federally-funded research effort caused scientific study of the Bay to be greatly accelerated. Furthermore, the objective of the effort was to attempt to use an ecological method of analysis in order to relate the findings from a series of independent studies. Though many technical questions still remain unanswered, the results of the Bay research effort have been expressed in a format that exceeds the degree of comprehensiveness of earlier studies.

The second factor relates to a frequent assertion that literature of today is dominated by scientific thought that is generally less expressive of values than in the past.[23] Emphasis on quantification of environmental impacts tends to lead to the question "will the proposed action have a significant effect?" rather than the more value-ladened question "will the proposed action contribute to the achievement of an expressed goal?" However, it may be argued that by avoiding explicit expression of values associated with the Bay, the results of the research effort and subsequent management program have become more widely embraceable. The study of itself does not overtly preclude or select any particular use of the Bay over any other use. Instead, representatives of various special interest groups have been able to rally around a set of findings that basically say "we are all part of the problem and therefore, we must all be part of the solution." Thus a coalition of diverse interests now agrees that it is in the broad public interest that the waters of the Bay must be made cleaner and her aquatic resources enhanced and protected.

Underlying this collective view is a series of important social and economic trends that have contributed to a re-evaluation of the importance of Chesapeake Bay. This re-evaluation is the third and perhaps most critical factor that has led to the success of the Chesapeake Bay Program. While the impact of each of the following statements could easily be the topic of a separate paper, they are summarized below as major examples to illustrate this point:

1. Continued population growth in the region is accompanied by rising employment, housing, recreational, and waste disposal needs, placing increased demands on land and water resources.
2. National declines in basic industry have focused greater regional attention on the need to provide employment alternatives.
3. The pursuit of high technology industries and growth in the service and trade sector are associated with greater emphasis on regional amenity values than is basic industry with its overriding need for raw materials and an efficient distribution network.
4. The decline in traditional port activity (especially in Baltimore) has led to the continued pursuit of funds for dredging shipping channels and upgrading port facilities, and has further contributed to the concern for employment alternatives.
5. Partly as a result of all of the above, major urban centers such as Baltimore and Norfolk are placing additional emphasis on the amenity value of their urban waterfronts.
6. A rapid growth in recreational boating on the Bay reflects not only increased population size, but the growing pursuit of leisure time activities associated with a post-industrial economy.
7. Changes in land ownership are having a marked effect on the more rural parts of the Bay region. For example, the sale of agricultural and forest land to foreign investors on Maryland's Eastern Shore appears to be motivated by relatively cheap land prices and rural amenity values.
8. The seafood industry continues to struggle with the problems of diminishing fisheries and the constant threat that a contamination event like the discharge of kepone to the James River or radioactivity from a nuclear power plant could ruin the healthful image of Chesapeake Bay products.
9. There is increased public recognition of economic losses that occur as a result of shoreline processes including serious localized erosion problems and the larger regional effects of a worldwide rise in sea level.

One result of a number of the above trends is that for the first time in history, the amenity value of Chesapeake Bay has become an influential factor affecting the regional economy. It is perhaps as important as the Bay's navigational, waste disposal, and fisheries resource values of both the past and present. Thus, the beauty and majesty of the Bay which once inspired the early European explorers has emerged as a powerful attribute that can affect locational decisions related to industry and commerce.

Several authors have linked the concept of amenities with economic development and population growth. Perloff and Wingo have suggested that amenity resources include a special juxtaposition of climate, land, coastline and water offering conditions of living which exert a strong pull on migrants from less pleasantly situated parts of the nation.[24] Gottmann has described migration trends in terms of two types of rapidly developing areas which he classifies as either "Riviera" or "megalopolitan."[25] The Riviera settlements (California and Floridian coasts in the U.S. and the French and Italian Rivieras abroad) developed primarily because of the amenity values inherent in the areas rather than the presence of an agglomeration of commercial and industrial functions.

Although the Bay region lacks extensive beaches or breathtaking views typically associated with areas of exceptional amenity value (i.e. the "Rivieras"), much of it is nevertheless endowed with a high degree of scenic value, recreational opportunity, and temperate climate - all within easy reach of the nation's capital and the surrounding megalopolitan area. These factors have in fact been emphasized repeatedly in public relations campaigns aimed at drawing new industry to the region. "Come to Chesapeake Bay, the land of pleasant living" has become a popular economic development theme.

The elevated importance of the Bay's amenity value has helped bring industry, government, public interest groups, and private citizens together in the pursuit of a management program that will enhance and maintain that value. An attitude of enlightened self-interest on the part of users of the Bay has prevailed, allowing a degree of increased environmental regulation and the commitment of large sums of public funds to advance the Bay cleanup effort.

RE-EVALUATION AND EQUITY AS CRITICAL FACTORS IN BAY MANAGEMENT

The purpose of the federal Chesapeake Bay research effort was to provide a thorough characterization of the physical, chemical and biological qualities of the Bay and to assess the extent to which the current condition of the Bay has been influenced by human activity. The first of these two objectives is a relatively straightforward one that can be achieved through intensive, systematic empirical study. However, the second objective is far more complicated and involves the application of more indirect methods of measurement. Laboratory experimentation, mathematical modeling, and interpolation of historic data are primary examples of such indirect methods that were employed by Bay researchers.

The outcome of the research effort was a description of the various components of the Bay that exceeded in thoroughness all previous studies. Analysis of historical data led to the establishment of generally accepted trends in resource productivity. However, the documentation of cause-and-effect relationships between human actions and Bay resource impacts is fraught with assumptions that proved to be extremely difficult to test or beyond the scope of the effort.

In spite of this limitation of the Bay study, federal and state officials were able to garner enough public support to embark on the extensive management program initiated in 1983. At the state level, Maryland enacted a particularly ambitious set of "Bay Initiatives" during its 1984 legislative session. At least three of these Initiatives are remarkable in that they succeeded in the face of considerable scientific and technical controversy. These include:

1. a commitment to remove nitrogen at selected sewage treatments on the Patuxent River which, as of this writing, has not been accepted by the U.S. Environmental Protection Agency as technically justifiable;
2. adoption of a phosphate detergent ban in spite of controversy over the issue of whether this action will really have a significant impact on Bay water quality; and
3. establishment of the Chesapeake Bay Critical Area - a 1,000-foot land use management zone surrounding Maryland's portion of the Bay and tributaries to the head of tide - which was created in spite of controversy over the degree of significance of land use impacts within this relatively narrow zone on Bay water quality and aquatic habitat.

What is most interesting is that these three initiatives are embedded in a broad program that was designed to reflect the comprehensive format of the Bay research findings. That the entire initiatives package became enacted, even with elements of considerable controversy, is an historic achievement.

In the Maryland case, it can be argued that the re-evaluation of the importance of Chesapeake Bay served as the basis for strong governmental action in the absence of conclusive answers to many of the highly technical questions relating to the significance of human impacts on the Bay system. In effect, this re-evaluation has induced a heightened state of "water consciousness" that must now be taken into account by both public and private sectors weighing the pros and cons of human activity. Within the public sector this has taken the form of increased scrutiny of the possible effects of such things as permitted wastewater discharges, proposed waterway projects, and large-scale development activity. Within the private sector, developers are still trying to maximize economic returns associated with Bay amenities while justifying the pollution minimization potential of specific development proposals.

In addition to this re-evaluation, the equity of the Maryland Bay Initiatives was an issue that played an equally important role in 1984. The age-old questions of "who pays?" and "who benefits?" were addressed by the initiatives in such a way that equity - or the appearance of equity - was achieved. Nearly everyone in the State can now be said to be making at least some contribution toward Bay cleanup. In other words, to some extent, "everyone pays" and "everyone benefits." Whether the State can continue to support its management program in the face of dwindling federal environmental funds will be the true test of the relative importance of the Bay as an amenity and resource provider as compared with its value for navigation and waste disposal.

CONCLUSION

The re-evaluation of the importance of Chesapeake Bay has led to the emergence of a strong sense of the Bay's amenity value. This value, which historically had not been an overt one influencing Bay laws or policy, appears to be held in common by an otherwise pluralistic community of Bay users. This re-evaluation along with the findings of the recently completed federal research effort have led to the adoption of a comprehensive Bay management program. In the face of unanswered technical questions, the success of this program has been highly influenced by the public's perception of its equity. In Maryland, this has been dramatically demonstrated by the adoption of a far-reaching set of Bay Initiatives. Unless fiscal resources become too constrained, it is likely that Chesapeake Bay will continue to evolve as a resource of great collective value to the public. The consequence of this, which is already evident, is a "water conscious" approach to private as well as public decision-making effecting the Bay and her surrounding watershed.

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KEY WORDS

Human values, re-evaluation, water-conscious ethic, equity.

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EXTINGUISHING LEGAL RIGHTS IN THE COMBAT OF NONPOINT SOURCE POLLUTION

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ABSTRACT

This paper considers alternative ways in which the government can effect farmers' property rights through incentives and restrictions. An analysis of government programs or potential programs in the framework of property rules, liability rules and alienability to control nonpoint source pollution. The paper considers the likelihood of success of legal challenges to coercive governmental efforts, in particular the programs which might be subject to challenge as a taking of farmers' property rights. Framing the appropriate argument for governmental intervention is a central core of the paper. Judicial interpretation of highest and best use of land has undergone a redefinition. How is this transformation likely to affect a Virginia court which has historically protected the right of the property owner from the intervention of government land use programs?

Since the early 1970's, the public view of the relationship between agriculture and the environment has been in a state of transition. Traditionally, the farmer was viewed as a manager of nature, extracting from the earth a bounty to support himself and the nation. However, as the environmental impacts of extracting that bounty have become more apparent, farmers' use of natural resources have become subject to more careful review and regulation. In essence, farmers' rights are also in a state of transition. "Rights to use land are being conditioned on effects that uses have on others, principally users of water and aquatic environments (Braden, 1982)."

The severity of agricultural nonpoint source pollution and the limited effectiveness of previous and current pollution control programs have caused conservationists, economists and policy makers to give serious consideration to alternative programs or policies to control the nonpoint pollution problem. These alternatives include changes which would alter the current allocation of property rights, as well as the rules under which those rights are protected and exchanged. This paper will present an overview of the agricultural nonpoint source pollution problem. Then the current control efforts and suggested alternatives will be reviewed. The assignment of property rights established by selected legal cases will be discussed in terms of environmental cases from Virginia and other states. Finally, the impact of changes in pollution control policies on property rights and possible legal implications will be discussed.

What is Nonpoint Source Pollution?

The Clean Water Act (CWA) recognizes two sources of water pollution: point and nonpoint. The federal system of effluent limitations and permits is directed at point sources, which are defined as "any discernible, confined and discrete conveyance...from which pollutants are or may be discharged" into navigable waters (U.S. Congress, 1977). Nonpoint source pollution is not defined in the CWA. The implication is that nonpoint source is that type of pollution which enters a stream and does not emanate from any "discernible, confined and discrete conveyance," i.e., nonpoint pollution is any pollution that does not enter from a point source. For its program planning purposes, the U.S. Environmental Protection Agency (EPA) refers to nonpoint source pollution as:

1. Generated by diffused land use activities, not identifiable activities.

2. Conveyed to waterways through natural processes such as storm runoff or ground water seepage, rather than deliberate, controlled discharge, and

3. Not susceptible to "end of pipe" treatment, but controllable by changes in land management or process practices.[1]

Generally recognized nonpoint sources include agricultural and silvicultural activities, mining, construction sites, roads and road maintenance, industrial sites and parking lots. Agriculture is by far the most significant contributor to nonpoint source pollution, however. Researchers have suggested that more than 50 percent of the sediment deposited in streams and lakes washes from agricultural land (Council on Environmental Quality, 1979). In addition, it has been estimated that agricultural sources contribute over 33 percent of the oxygen-demanding loads, 66 percent of the phosphorus and 75 percent of the nitrogen discharged into streams (U.S. Water Resources Council, 1978). Effluent limitations and a permit requirement were imposed on point sources of water pollution as a result of the CWA. However, nonpoint sources remained exempt from those controls. The following statement by EPA largely explains the exclusion of nonpoint source pollution from the permit system.

> The major characteristics of the pollution problem which is generated by runoff...is that the owner of the discharge point...has no control over the quantity of the flow or the nature and amounts of the pollutants picked up by the runoff. The amount of flow obviously is unpredictable because it results from the duration and intensity of the rainfall event, the topography, the type of groundcover, and the saturation point of the land due to any previous rainfall. Similar factors affect the type of pollutants which will be picked up by that runoff, including the type of farming practice employed, the rate and type of fertilizer application, and the conservation practices employed...[2]

Some agricultural activities, for example animal feed lots, are defined as point sources and regulated under the Clean Water Act. However, the regulations only cover concentrated animal feed lots and only if a large number of animals is contained in the operation.[3] "Concentrated animal feeding operations" are required to secure National Pollution Discharge Elimination (NPDES) permits. Permits were required for irrigation return flows in 1973,[4] but in 1977 Congress specifically provided that "the administrator (EPA) shall not require a permit under this section for discharges composed entirely of return flows from irrigated agriculture, nor shall the administrator directly or indirectly require any State to require such a permit."[5] Congress has also stated that

> normal farming, silviculture, and ranching activities such as plowing, seeding, cultivating, minor drainage, harvesting for the production of food, fiber and forest products, or soil and water conservation practices,... construction or maintenance of farm or stock ponds or irrigation ditches, or the maintenance of drainage ditches, (and)... construction or maintenance of farm roads or farm roads...[6]

cannot be regulated through the establishment of an effluent limitation or dredge and fill permit.

Control of Nonpoint Source Pollution

Control of nonpoint source pollution was first discussed in Section 208 of the Federal Water Pollution Control Act Amendments of 1972 (U.S. Congress, 1972). Section 208, as amended by the CWA, requires area-wide planning for waste treatment management by states and certain designated areas. Under this process, an area-wide plan is developed by a planning agency at the state level and is submitted to EPA for approval. Then a regional operating agency is designated to carry out the plan. These regional operating agencies may be either existing or newly created local, regional or state agencies or political subdivisions (March, Kramer, and Geyer, 1981).

The area-wide plan must specify a process to identify agriculture related nonpoint sources of pollution and methods to control these sources. Given the general inapplicability of permits and other forms of effluent limitations to nonpoint source pollution, an alternative control technique is required. The control technique authorized by the CWA and EPA is the implementation, by farmers, of "best management practices (BMPs)". BMPs have been defined by EPA as those methods, measures or practices to prevent or reduce water pollution which include but are not limited to structural and non-structural controls, and operation and maintenance procedures (March, Kramer, and Geyer, 1981). Agricultural BMPs include provisions for most soil erosion control practices as well as animal waste control facilities.

Most agricultural BMP programs are of a voluntary nature. The programs provide for education and information about agricultural nonpoint source pollution, its damages, and its controls. The programs also include a cost sharing strategy to assist farmers in installing often costly pollution control practices. Cost sharing of BMPs has also been instituted because of the off-farm nature of the benefits of BMP adoption. However, questions have been raised as to the effectiveness of the voluntary cost sharing program for pollution control. Despite the availability of cost sharing, there is little incentive for farmers to voluntarily adopt BMPs when they will not recognize, directly, the benefits of their investments. That cost sharing funds are limited further reduces the likelihood of widespread BMP adoption.

Along with the questions regarding the effectiveness of the current nonpoint control program have come suggestions for alternative programs. However, any move away from the current voluntary program will require some specification as to how pollution will be "measured" and what form control should take. Many states have focused their 208 programs on sediment control because, compared to agricultural chemical pollutants, more is known about the effects of land management practices on soil loss. The Universal Soil Loss Equation (USLE) provides clearly defined, scientific principles upon which to base program priorities and enforcement actions (Braden, 1982). Also, using the USLE, soil conservationists can make recommendations as to the most effective BMP or combination of BMPs for a specific problem.

One program change which has been suggested is the institution of a cross-compliance program. Under cross-compliance, farmers would be required to adopt a specified BMP program to be eligible for benefits from agricultural income-support and subsidy programs. The types of programs which might be involved include federal price support, loan subsidy and crop insurance programs. However, the effectiveness of a cross-compliance strategy would depend on the extent to which the other programs included were used by the farmers with serious erosion problems, the extent to which the threat to withhold the other programs would provide sufficient incentive for farmers to take action, and the extent to which farmers could effectively control their problems (Clark, Havercamp and Chapman, 1985).

Another alternative which has been suggested involves a soil loss tax. Under such a scheme, farmers would be charged on the basis of the amount of soil (and presumably associated contaminants) carried off their land. The tax would be set at a level that represented the costs of the pollution to society, and its size would vary between areas of the country and across different farms within an area to reflect the differences in the magnitude of soil loss and associated pollution problems (Clark, Havercamp, and Chapman, 1985). However, determining the efficient level of a tax would be difficult, as would monitoring the actual runoff of soil and contaminants.

Several regulatory approaches have been suggested which would require farmers, by law, to implement a specified BMP program or to meet a specified soil loss limit. Iowa has passed such a statute, under which its Soil and Water Conservation Districts may establish regulations to require erosion control. However, the state must provide financial assistance to persons forced by the regulations to control erosion (Braden, 1982). Requiring compliance without a guarantee of financial assistance is also a possibility, especially in cases where public health statutes are violated.

In addition to statutory law, common law theories such as nuisance and trespass may be called upon to address non-point source pollution. Under common law, a public nuisance is an unreasonable interference with a right of the general public. A private nuisance is an interference with another individual's use of or enjoyment of property. The notion of pollution is a public nuisance may, in some cases, arise directly from a constitutional provision.[7] Under the private nuisance theory, farmers have been sued for allowing cattle and hogs access to streams and for allowing livestock wastes to reach the streams or wash onto plaintiff's property.[8] Similar cases have involved chemicals and sedimentation.

The theory of trespass is another common law approach to pollution control. Trespass involves an intentional physical invasion of someone's exclusive use of his land (Prosser, 1971). Water transportation of dirt and animal carcasses have been held pollution trespass (Davidson, 19__).

Although the common law theories are operative, their enforcement in cases of nonpoint source water pollution is complicated by the requirements of proof and damages. Determining the exact source of pesticide or fertilizer contamination is most likely impossible. Similarly, an accounting of the damages from the nuisance or trespass of some parts per million of a chemical is, at best, a rough estimate. Therefore, the enforcement of common law may be less desirable and less effective than legislative intervention.

Whether nonpoint source pollution control programs continue to solicit voluntary cooperation, albeit in a modified approach, or whether some regulatory approach is adopted, it appears that changes in the current nonpoint source pollution control programs are forthcoming. Such a transition is seldom a smooth process. In the case of nonpoint source pollution, the transition is further frustrated by the corresponding problems of identification, definition and enforcement of changing property rights allocations.

A Discussion of Property Rights

In the framework developed by Calabresi and Melamed, a right ("entitlement") confers favor among individuals or groups making competing claims to an object or privilege (Calabresi and Melamed, 1972). In defining rights, two matters must be decided: a) initial ownership (allocation) of rights, and b) rules under which they may be exchanged. "Collective enforcement of both the initial allocation and the conditions for exchange is required if a legal system is to have meaning (Braden, 1982).

According to Calabresi and Melamed, the allocation decision must reflect accepted tenets of social relations, including distributional equity and judicial consistency. Within these constraints, asserts Braden,

> an entitlement should be made to the party best able to evaluate its social worth or, secondarily, to the party who can act most cheaply to correct errors in its initial allocation (i.e. evaluate and initiate exchange) (Braden, 1982).

As an interpretation of the Coase Theorem (Coase, 1960). where competing parties are equally able to evaluate an entitlement and initiate exchange, where transaction costs are absent, and where income effects are negligible, the equilibrium resulting from exchange is invariant with respect to the initial allocation of rights. However, in reality, income effects are often not negligible and competing parties are not equally equipped to trade. Also, in most cases, transactions are not costless.

Rules for exchange of rights must also be considered. Calabresi and Melamed noted three options: 1) property rules, 2) liability rules, and 3) inalienability. Braden's (1982) discussion of these options is perhaps most clear.

> With property rules, consent to an exchange must be given in advance by all parties at an acceptable price, like market transactions. Beyond the government's presence in setting and enforcing rules of fair exchange, neither the price nor the willingness to exchange is subject to direct government control...Under liability rules, prior consent to an exchange is not required, and prices are set by an objective third party, such as a court. Government involvement is increased here, relative to property rules, because affected individuals are not likely to agree to a rate of compensation without the threat of arbitration...In other situations, liability rules may be appropriate because many individuals with interests in a transactions are not efficiently represented in the market. Assessment and representation of those interests by an objective authority (government) may be cheaper than negotiating separately with each one... Finally, inalienability disallows exchange of specified rights under some or all circumstances. The state controls both transactions (by prohibitions) and prices (e.g. fines and prison terms).

Bromley (1978) considers these same rules but discusses them as the rules by which rights are protected, rather than exchanged. His framework is as follows. Given two parties, A and B, if:

1) A owns the entitlement--

When A is protected by a property rule, A may interfere with B and can only be stopped if B buys off A.

When A is protected by a liability rule, B may stop A from interfering but must compensate A.

2) B owns the entitlement--

When B is protected by a property rule, A may not interfere with B without B's consent.

When B is protected by a liability rule, A may interfere with B but must compensate B.

When B is protected by inalienability, A may not interfere with B under any circumstances and the stopping does not imply compensation.

Bromley (1978) also notes that transactions costs will be higher when entitlements are protected under property rules rather than liability rules because the property rule requires a prior agreement among the parties. He suggests that, as a result, interference will likely be greater under the latter.

Property Rights and Nonpoint Source Pollution Control

The allocation of property rights to water and its use is not explicitly defined. There is precedent to suggest that these rights are owned in most cases by the farmers. Indeed, under the current program of voluntary BMP implementation, it can be argued that farmers have been endowed with the rights to the "use" of water. With the current program, those rights are protected under a property rule, and may be "purchased" by the public if farmers choose to exchange their rights to create nonpoint source pollution for cost sharing assistance in controlling the pollution. In addition to cost-sharing programs administered by the Agricultural Stabilization and Conservation Service and state agencies, the tax credit provided by the Commonwealth of Virginia is an example of a law designed to allow the "purchase" of property rights by the public. The law provides that a credit against state taxes of 25 percent and all expenditures for the purchase and installation of conservation tillage equipment up to a total of $2,500.[9] Replacing these current programs with one of the alternatives discussed above would result in a change in the definition and/or allocation of property rights. In many cases, current holders of the rights are likely to oppose a change.

Adoption of a cross-compliance strategy would not entail a fundamental change in the allocation or protection of rights. Rather, such a program would redefine farmers' rights to use land and water resources and would likely reduce their value. Redefining what farmers have a right to do and receive changes the benefits associated with owning a certain entitlement. Making participation in federal income-support programs dependent upon pollution control activities would impose on farmers substantial costs, either in BMP adoption or loss of program benefits. Cross compliance was adopted by both House and Senate as a part of the 1985 Farm Bill.[10]

Imposition of a soil loss tax would involve a more significant change, as the resource use entitlement would be shifted from farmers to the public. Under such a tax, farmers would be allowed to interfere with the public right but would be required to compensate (pay tax to) the public. Thus, the public's entitlement would be protected under a liability rule. A reallocation of the entitlement away from farmers would likely be strongly opposed by the farmers, especially because of their sudden loss of a valuable right (right to loss soil) without any compensation for that loss. In fact, they would have to pay for the loss of soil.

An alternative which would leave the pollution entitlement in the hands of farmers but which would change the rule under which it is protected is one which would require compliance by farmers with pollution control regulations in return for guaranteed cost sharing assistance. This is the type of change which has been made in the soil conservation program in Iowa.[11] In this case, the farmer remains entitled but is protected by a liability rule. Such a change would be expected to reduce pollution significantly as compared to the current program.

A mandatory pollution control policy would not go unchallenged by farmers. However, it is likely that, given adequate cost sharing assistance, farmers would be forced to comply. Iowa's soil conservation statute was challenged, but in the case of Woodbury County Soil Conservation District V. Ortner (279 NW2d 276), the court concluded:

> ...the test is whether the collective benefits (to the public) outweigh the specific restraints imposed (on the individual)...It is important therefore to consider the nature of the public interest involved and the impact of the restrictions placed on defendants use of their land...It should take no extended discussion to demonstrate that agriculture is important to the welfare and prosperity of this state. It has been judicially recognized as our leading industry... The state has a vital interest in protecting its soil as the greatest of its natural resources, and it has a right to do so...While this (legislation) imposes an extra financial burden on defendants, it is one the state has a right to exact. The importance of soil conservation is best illustrated by the state's willingness to pay three-fourths of the cost...The remainder to be paid by defendants...is still substantial, but not unreasonably so. A law does not become unconstitutional because it works a hardship...[12]

It appears, then, that where a significant need for pollution control is combined with a willingness to compensate the farmer, a mandatory program could be legally enforced.

A mandatory program which required pollution control but did not provide cost sharing assistance (or provided limited cost sharing on a first come-first served basis, as with the current program) would reallocate the pollution entitlement from farmers to the public. The public's entitlement would be protected by a property rule. If Congress passed such a law, it could be upheld in a court of law, despite strong opposition from farmers. In the case of Virginia's State Water Control Board V. Train (559 F2d 921), The State Water Control Board filed suit to prevent Virginia municipalities which did not receive financial assistance from the government from having to comply with effluent limitations for publicly owned sewage treatment plants by the specified date. In its decision, the court held that:

> The bill which the House subsequently passed empowered EPA to extend the 1977 deadline for up to two years in cases where compliance is physically or legally impossible. But...it did not limit the applicability to those facilities receiving financial assistance. Moreover, even the provision authorizing case-by-case extension of the deadline was later deleted without comment by the Conference Committee. This clearly provides strong support for the conclusion that Congress meant for the July 1, 1977 deadline to be rigid and that it did not intend that sewage treatment plants not receiving timely federal grants should be exempt from that deadline.[13]

If public sentiment in favor of controlling nonpoint pollution is sufficiently strong, a shift in pollution rights away from farmers could be upheld, without requiring compensation of the farmers.

Such a reallocation of rights without compensation could be considered by farmers to be a taking. However, it is not clear that farmers could avoid compliance with pollution control standards based on this issue. In upholding the natural use of land as its highest and best use, a Wisconsin court concluded that restrictions on the use of land to maintain its natural use did not constitute a taking of property rights. In Just v. Marinette Co.,[14] Just filled part of his land. The filling violated an ordinance which required a special permit for filling, drainage and other changes in the use of swamp and marsh land. The court stated that the:

> real issue is whether the conservancy district provisions and the wetlands-filling restrictions are unconstitutional because they amount to a constructive taking of the Justs' land without compensation. Marinette county and the state of Wisconsin argue the restrictions of the conservancy district and wetlands provisions constitutes a proper exercise of police power of the state and do not so severely limit the use or depreciate the value of the land as to constitute a taking without compensation.

Maintaining a body of water in its natural state might well be considered a duty of adjacent landowners, because, as the court held:

> it is a conflict between the public interest in stopping the despoilation of natural resources, which our citizens until recently have taken as inevitable and for granted, and an owner's asserted right to use his property as he wishes. The protection of public rights may be accomplished by the exercise of the police power unless the damage to the property owner is too great and amounts to a confiscation. The securing or taking of a benefit not presently enjoyed by the public for its use is obtained by the government through its power of eminent domain... An owner of land has no absolute and unlimited right to use it for a purpose for which it was unsuited in its natural state and which injures the rights of others.
>
> This is not a case where an owner is prevented from using his land for natural and indigenous uses. The uses consistent with the nature of the land are allowed and other uses recognized and still others permitted by special permit.[15]

If highest and best use is defined for water as use in an unpolluted state, then this case raises questions as to farmers' rights to allow nonpoint source pollution to emanate from their land. For a discussion of Virginia decisions in the land use area, see BeVier and Brion (1981).

<u>Some Additional Considerations</u>

Farmers' dissatisfaction with the allocation changes associated with adopting a mandatory nonpoint source pollution control program would not be the only difficulty accompanying such a change. Enforcing water quality regulations is especially difficult for nonpoint pollution because extensive monitoring is required and complicated linkages between water quality and land use practices must be established. Each parcel of land has a unique potential for contributing to pollution. Enforcing a reallocation of rights away from farmers would be very expensive.

Transaction costs associated with alternative programs should also be considered. Leaving the entitlement in the hands of the farmers but changing from a property rule to a liability rule for protection could be expected to reduce transaction costs, as suggested by Bromley. Such a reduction might offset somewhat the costs associated with compensation of the farmers under the required cost sharing. If the rights are reallocated so that the public is entitled, the soil loss tax strategy would likely be preferable to the regulatory program in terms of lower transaction costs. This assumes legally acceptable measures of control. However, the allocation, as well as distribution, of the costs would be an additional issue with which the government would have to deal.

Any change from the current program will be costly. As Batie has noted, there are two kinds of costs associated with nonpoint source water pollution - the costs of doing something and the costs of doing nothing (Batie, 1985). Changes in property rights will involve administrative, enforcement and compensation costs. However, a continuation of the current voluntary approach will likely be a "slow boat on waters that remain polluted (Cook, 1985). As public concern over nonpoint source water pollution continues to mount, changes become more likely. However, such change must be effective, enforceable and ethical. As Braden (1982) has asserted, an acceptable program will be one which provides a mechanism by which the public interest can be served while providing fair compensation to owners whose rights are exchanged by fiat, retains the flexibility of individual ownership, and is consistent with the national ethic of maximum individual liberties consistent with the general welfare.

Endnotes

[1]41 Federal Register. 24,709, 24710 (1976).

[2]Federal Appellants' Memorandum on "Impossibility", Natural Resources Defense Council v. Costle, 568 F.2d. 1369 (1977).

[3]40 CFR Sec. 412.10 (1985).

[4]38 Fed. Reg. 1800 (July 5, 1973).

[5]33 U.S.C.A. Sec. 1342(1) (1985).

[6]33 U.S.C.A. Sec. 1344(t) (1985).

[7] See Rogers W., Handbook on Environmental Law 183-185 (1977). See E.g. New York Const. art XIV, Sec. 4, Mich. Const. art. 4, Sec. 52. Although Va. Const. art. XI, states that it is the policy of the Commonwealth to conserve, develop, and utilize its natural resources so that the people have clean air, pure water, and use and enjoyment of lands and waters, the provision is not self-executing nor does it create a duty in and of itself. Robb v. Shockoe Slip Found. ___Va.___, 324 S.E. 2d 674 (1985).

[8]Ibid. Sec. 8.03 notes 31-21.

[9]Va. Code. Ann. Sections 58.1-334 and 58.1-432. The tax credit may be carried forward for up to five years. This provision is in addition to current federal and tax law which would provide for Investment credit (I.R.C. Sec. 38) and cost recovery (I.R.C. Sec. 168) for equipment used in a trade or business. $3,500 of conservation tillage equipment can be written off against federal and state taxes the first year effective reducing the cost by 35 percent of purchase price.

10Idem.

[11]Iowa Code Sec. 467A (1985).

[12]Woodbury County Soil Conservation District v. Ortner, 279 N.W.2d. 276.

[13]State Water Control Board v. Train, 559 F.2d. 921.

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KEYWORDS

1. nonpoint source pollution
2. property rights
3. government programs
4. intervention
5. land use

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LANDSCAPE PLANNING AND THE BAY: A CASE STUDY

The 1984 and 1985 Virginia Tech
Fourth Year Landscape Architecture Students
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ABSTRACT

For the last two years a team of students in the Landscape Architecture Program at Virginia Tech has focused its attention on a Virginia County in the Chesapeake Bay watershed in efforts to understand the broad impacts of people and land use on the bay. They used a Geographic Information System (GIS) approach in identifying conflicts between existing and proposed land uses and the environment. Two future land use scenarios were developed and evaluated for the county, one scenario based on a projection of current growth patterns, the other based on the existing comprehensive plan. After comparing these scenarios for their potential impact on environmental quality, alternative development strategies were suggested to aid the county in growing in harmony with its natural environment, especially the bay.

Conclusions from the study which might be found useful in other bayside counties include:

- The usefulness of a computerized geographical information system.
- The importance of getting the public involved in the planning process, helping them to understand the effects of land activities on the bay.
- The vital necessity of revising current trends in land use development in an effort to save the bay and maintain more liveable communities.

PROBLEMS OF THE BAY

The Chesapeake Bay, the world's largest estuarine system with a watershed of some 64,000 square miles in six states, is affected extensively by the land surrounding it. Mismanagement of inland areas and of the bay itself have considerably reduced the productivity of the estaury since the 1950's. According to an Environmental Protection Agency study, this decline can be attributed to three major problems:

- Nutrient enrichment,
- Decline in submerged aquatic vegetation, and
- Accumulation of toxic substances.

These problems result from both upstream non-point source pollution (partly in the form of agricultural and urban runoff) and point source pollution (the direct influx of contaminants into the water).

Agricultural runoff is a primary cause of nutrient enrichment. Nitrogen and phosphorous fertilizers, especially from plowed fields, infiltrate the watertable or are carried off with eroding soil and from there quickly enter the the bay. Once in the bay, the nutrients cause accelerated plant growth in the form of surface algal blooms, which when they decay consume large quantities of dissolved oxygen, leaving the water unsuitable for many important life forms such as crabs and oysters.

The decline in submerged aquatic vegetation is directly related to sedimentation. Runoff from areas failing to use sedimentation and erosion control practices causes silt to accumulate on seagrasses. The seagrasses die, thus depriving the bay of an essential habitat for spawning fish, shellfish, and waterfowl food sources. Heavy metals, such as mercury, lead, and chromium, enter the bay from industrial point sources and remain there for many years. The damage to the bay from such point source pollution has made some seafood inedible.

Misuse of coastal wetlands is another pressing problem associated with the decline of the bay. Local residents often do not fully appreciate the vital role of wetlands. Most know that wetlands are valuable as habitat areas. However, their other functions are also important, and are often overlooked. Wetlands act as sponges during floods, sediment traps, toxic assimilators, sources in the food chain, and erosion controls.

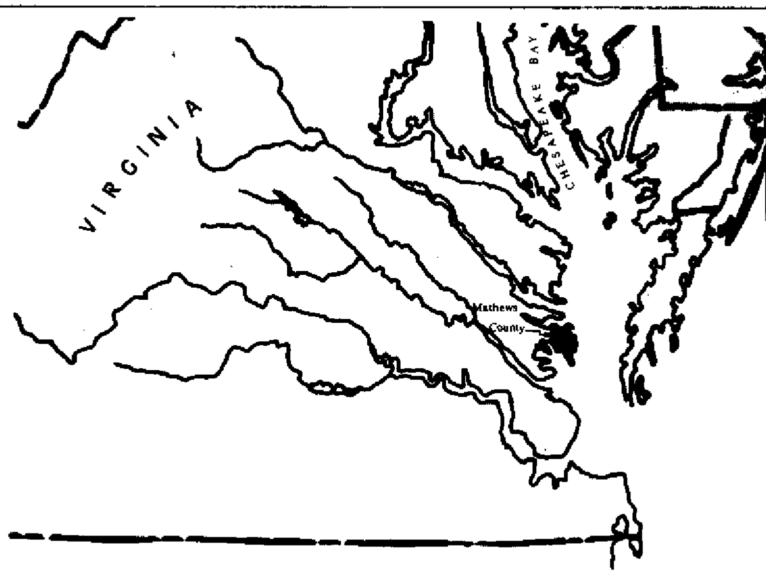

Figure 1. Mathews County, Virginia: its location on the Chesapeake Bay

APPROACH

The case study approach seeks to find generalities for wider application by creating solutions to specific situations. In this case by exploring a specific county on the Chesapeake Bay, solutions to land use problems there and in other parts of the bay drainage were sought. Specifically ideas for more environmentally sensitive land use planning were explored by studying a typical landscape, Mathews County. A thorough inventory of natural and cultural resources began with extensive interviews with a wide range of county officials and residents and the collecting of mapped data. Analysis of this data help refine the teams understanding of land use problems there. A Geographical Information System (GIS) was used as one mechanism for synthesizing some of the collected data into a spatial format. The GIS allowed such questions as the following to be answered:

- Where are there areas within the county environmentally suitable for development, such as housing or commerce?
- What are some of the environmental problems of areas within the county that look likely to be developed in the near future?
- Where are the areas within the county that should be considered for conservation because they contain resources of value to the whole community?
- Are there environmental problems facing areas targeted by the county's comprehensive plan as growth areas?

Prototypical designs were created which included sensitive ways of siting land uses in harmony with the environment. The suitability studies and the prototypical designs were presented and discussed with county residents. The following is a more detailed discussion of the process.

THE CASE STUDY

Mathews County, Virginia, has much in common with other Chesapeake Bay counties. In general, the county is extremely flat, which causes serious drainage problems, especially in association with areas of high water table and poorly percolating soils. For these and other reasons most of Mathews County is undeveloped and managed for forestry. Some of the characteristics of the county include:

Wetlands. Approximately 2,900 acres within Mathews County are wetlands. Wetlands face stiff development pressures because of their location at the shoreline—prime housing sites. They are important habitat areas for birds, crustaceans, and fish and help diminish the effects of flooding by absorbing and then slowly releasing floodwaters. Wetlands and sand dunes absorb the energy of incoming waves and thereby reduce shoreline erosion.

People. According to the 1980 census, the population of Mathews County is 7,995 people, all residing in a land area of 88.7 square miles. Estimates (Mathews County Comprehensive Plan, 1982) place the county's 1990 population at 9,354, an increase of 1,359 in ten years. Indeed, Mathews County has gained a reputation as a desirable retirement spot. In 1980, 35 percent of its people were 55 years or older compared to 18 percent statewide. Mathews County's biggest attraction to retirees and tourists is the Chesapeake Bay with its abundant opportunities for water-related recreation. Also valued is the historic, rural quality of the county.

Agriculture. At present, 18 percent of Mathews County is farm or pasture land, much of it plagued with a high water table. The total acreage of land devoted to farming continues to decline. Some farmers are using new management practices, such as no-till farming, to reduce topsoil loss and subsequent pollution of streams, wetlands, and eventually the bay.

Seafood. The county has a long history of gathering seafood from the Chesapeake's rich bounty, an activity which was a natural compliment to the area's now diminished shipbuilding industry. Today Chesapeake watermen find that declining catches and contamination are threats to the future of their livelihoods.

Forestry. Mathews County, environmentally limited as it is for so many human activities, is ideally suited for forestry. In fact a majority of county land (60%) is currently managed for forest products, and most of that by one corporation. Land characteristics that make agriculture and homebuilding difficult or expensive are tolerated by trees.

Light Industry. A small but varied group of light industries currently exists within the county, including flag embroidery, seafood processing (including crab picking houses) and rope-making.

History. The historical landmarks of the county offer cultural and educational benefits. In turn, the county earns money from the tourism these landmarks invite.

Recreation. Various types of recreation exist in the county, with the most plentiful opportunities being water-based (e.g., fishing, boating, and waterskiing). Several private marinas, public ramps and individual homeowners' docks exist along Mathews' shores. A severely eroded public beach and boat ramps are the major public access points to the bay. Popular land-oriented recreational activities include softball, tennis, basketball (all primarily for residents), and hunting and camping (privately available for residents and tourists).

IMPLICATIONS OF CURRENT LAND DEVELOPMENT TRENDS

What will happen to the county if land uses continue to develop as they have in the recent past? The trends would suggest the following:

- Residential development will occur primarily along the shore,
- Commercial development will occur primarily along major roads near population centers,
- Agriculture will decline,
- Forestry will stabilize, and
- Poor soil percolation rates will severely limit development (unless sewage treatment plants are built or alternative treatment techniques become common.)

Figure 2 shows a map of the county as it might be in the future if these trends continue.

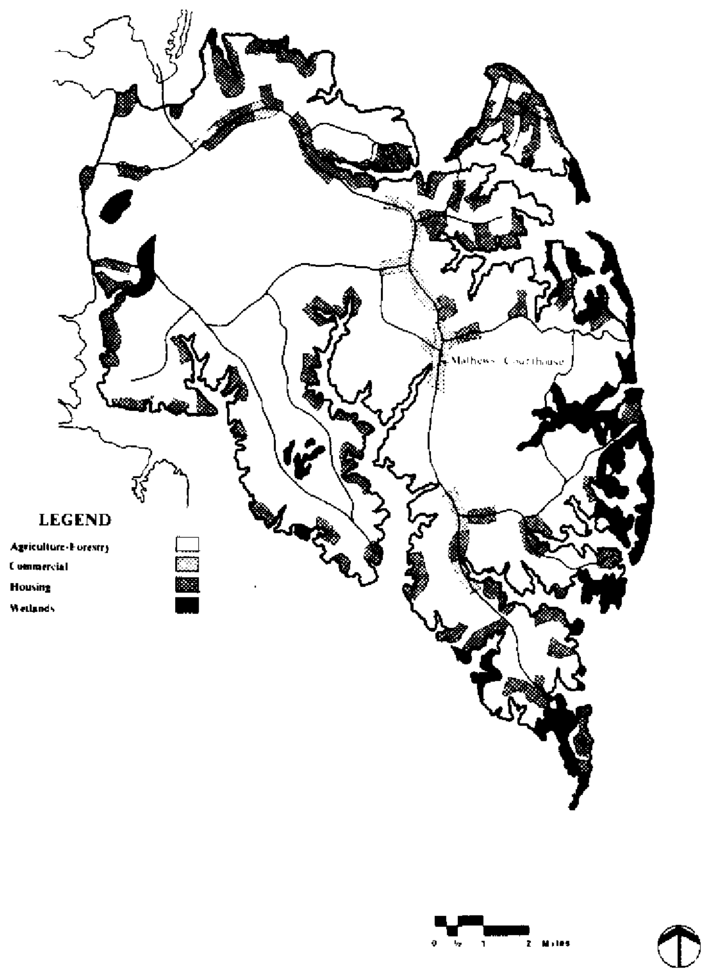

Figure 2. Potential future land use based on current trends.

Problems with such land use patterns include:

- the loss of wetlands to roads and housing and degradation of wetlands due to sedimentation from nearby construction and on-going residential activities;
- the vulnerability of most housing to flooding because of locations within the 100-year floodplain;
- the aesthetic degradation that shoreline development can cause for boaters and opposite-shore residents;
- the destruction of the rural quality by strip commercial development along highways 14 and 198, which presents motorists (residents and visitors) with an image of extensive parking lots and commerce; and
- the natural plant succession of formerly open fields as agriculture is abandoned and the loss of long views across farmed land; i.e., a lost rural image.

This suggests that while the county may presently be perceived as a very good place to live or vacation, continued development may result in the loss of some visual, environmental, and economic resources. These resources include the perceived rural quality of the county, the beauty of its shores, the multiple-functioning wetland areas, and the seafood industries.

What are the factors that contribute to these trends and cause subsequent problems? The most obvious for housing is the attraction of people to water. People, especially retirees who have chosen to move to a coastal community, often want to live where they can see the water. The soils of the county are arranged such that those most effective for septic systems are also at the water's edge. The conflict arises because the land at water's edge is the most dynamic of any in the county. Perhaps it is this energy of water tearing at land that is the attraction—and at the same time the danger for homebuilders.

Areas which are attractive for commerce from a business standpoint, say with high visibility, are not always areas, which if developed, serve the community's best interests. Mathews is a rural county, but strip commercial development along some of the major roads would change this perception or visual framework of the county.

To ensure a liveable future the community must articulate its needs and goals, be aware of the county's environmental constraints (poor soil percolation, e.g.) and opportunities (water-related activities), and take steps to reconcile these conflicting factors.

Creating and Using a Computerized Geographic Information System

Pertinent map data were gathered for Mathews County to locate vegetation types, roads, soils, buildings, wetlands, the areas served by sewage treatment system, and water bodies. Sources of data included United States Geologic Survey topographic maps, Soil Conservation Service soils maps, and Department of Housing and Urban Development Federal Insurance Administration flood hazard boundary maps. When necessary these sources were updated through interpretation of recent aerial photographs taken for the Virginia Highway Department. These maps were entered into the Geographical Inforamtion System (GIS) for analysis. One of the most obvious changes in the county since most of the maps were made was the shoreline configuration, especially New Point Comfort where acres of land have been claimed by the bay. The GIS was then used to map areas suitable and unsuitable for development.

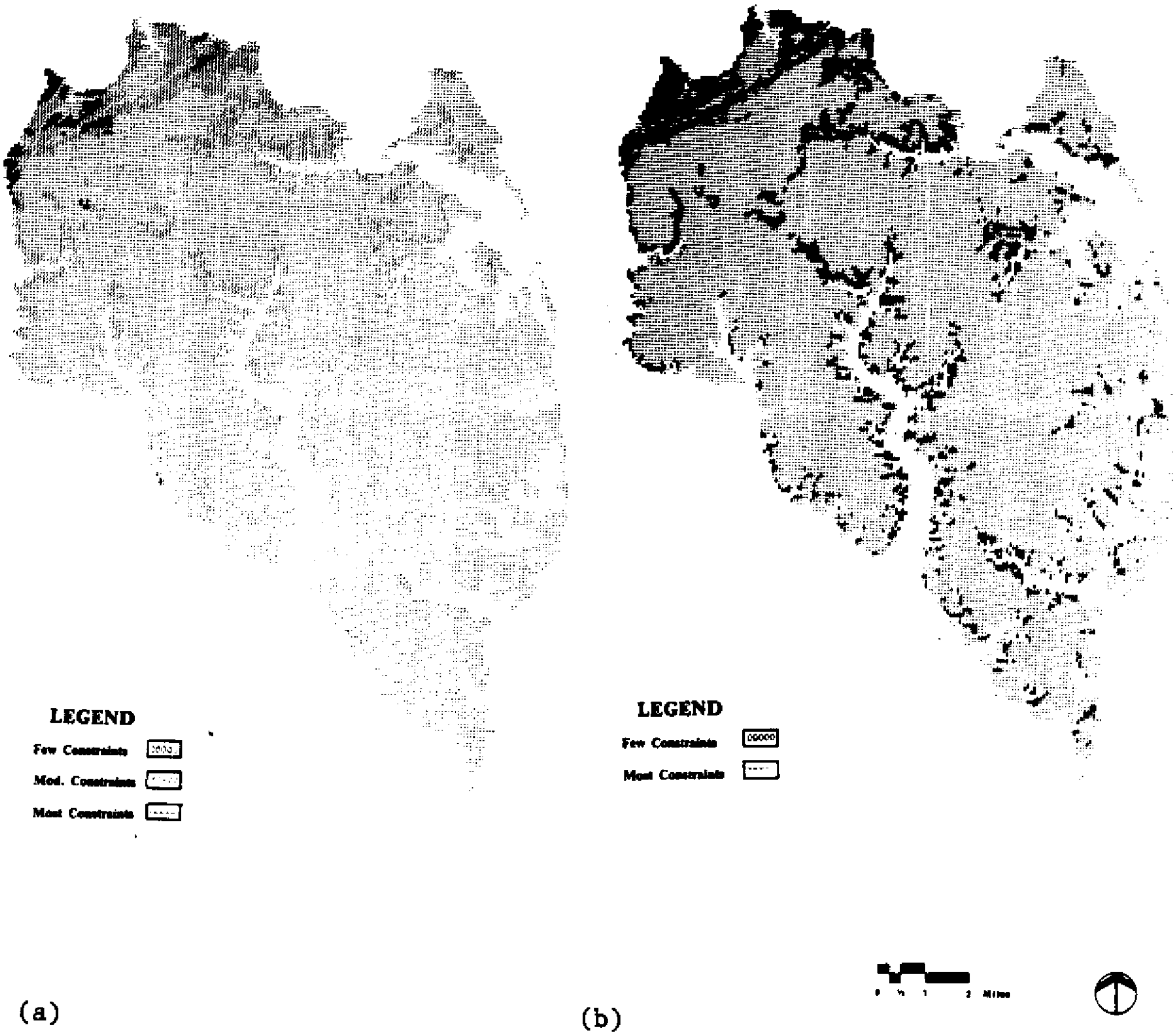

Figure 3 (a) Suitability for structures with septic systems, (b) Suitability for structures with sewers.

Land Use Suitability

To determine the capability for development within Mathews County (based on the land's physical characteristics but not on any overall market or economic factors) the legal and site construction considerations can be mapped. In this way, constraints for four different land use activities were determined. The land uses are: structures (houses, small commercial, small light industry, etc.), marinas, recreation (parks), and conservation. An important consideration is the increased capability of land to support various uses as technology overcomes former limitations. Mathews residents were frequently heard to remark that they felt the county was "pretty much developed out." While this may be true of some areas of the county today because of much land that doesn't percolate fast enough for a septic system, advances in technology may change this. It is only a question of time until some of these inventions become affordable and legally acceptable.

After considering applicable laws (if any), standard siting criteria, and the special conditions of Mathews County, appropriate computer maps were combined to determine potentially suitable areas for each individual land use type. Typical is the map shown in figure 3(a) suitability for small structures with septic systems. One of the greatest constraints to development is the very low capability of much of the county's soil to serve as a septic system whereby wastes can be purified. Constraints for structures were mapped twice for the county due the significant difference whether a treatment facility is available or not, (as in the case in the Courthouse area). The first map (figure 3(a)) shows suitability for structures with individual septic systems, the second (figure 3(b)) shows suitability for structures served by sewage treatment plants. The following base maps were used: the soils map was used to avoid areas with high shrink-swell potential and for the structures with septic systems, slow percolation rates; the water table map was used to avoid sites with a water table less than 24 inches below the surface, and finally land in the floodplain.

As the maps illustrate, with the exception of scattered points, all the areas least constrained for structures are in the northern end of the county out of the floodplain, where land is less flat and where soil percolation rates are better. This synthesis indicates that about 400 acres in the county have few constraints for structures. A portion of this land could be served by sewage treatment facilities although some sites would probably too distant for ready access.

FINDINGS

Mathews County has a rural character which its residents wish to preserve. This goal is desirable not only to Mathews residents but to the Chesapeake Bay as a whole since development along streams and river will increase the stress already placed on the Bay. Building in the floodplain also has serious financial and safety repercussions. Many lives have been lost due to flooding, and millions of dollars in disaster relief have been spent nationwide. Floodplain areas should be reserved for land uses that are least affected by flooding, such as recreation, forestry, farming, and conservation.

Future development either should be steered away from vulnerable lowlands or buildings should be designed to resist storms and flooding. Based on the land suitability analysis undertaken in this study, safe land exists within the county near the shore—along the Piankatank River, land that is not only less likely to flood and also has soils better suited for septic systems.

Currently only a small part of the entire county is suitable for development, a fact which provides the strongest argument for keeping the county rural. In the future, however, as technology develops alternate ways of dealing with waste treatment—affordable treatment not dependent on soil capabilities—that limitation may drop away and land formerly considered undevelopable may be developed. In the interim clustering of development will leave larger expanses of open space, but it will also provide economic justification for installing a sewage treatment plant that will prevent leaching of septic field sewage into the Bay.

An increased use of BMPs (best management practices) can reduce the impact that agriculture has on water quality. Since Mathews County desires a rural character, these practices are especially important because they will maintain the integrity of farming while lessening the negative influences that farming has on the Chesapeake Bay. Funding and education are available to county farmers who are interested in practices such as no-till and grass filter strips. These practices, along with the many others, can help make agriculture and the Chesapeake Bay more compatible.

CONCLUSIONS

Applicable to other localities in the bay drainage area from this study are its environmental analysis and use of a GIS for synthesizing data, evaluating alternative land use scenarios. The traditional reliance on the unsuitability of poorly percolating soils as a means to curb growth may have to be replaced by active decisionmaking on the part of the community as former limitations to development are overcome. A key part of any planning effort in any community is the recognition of unique resources that exist within that community. In Mathews it may be the recognition that an older population can add special character to the community and in fact become a social and economic focus.

The Chesapeake Bay is a fragile ecosystem that is under a great deal of stress. The land uses in the areas surrounding the bay have a tremendous influence on its survival. Without appropriate planning and management, increased nutrification, non-point source pollution, higher chlorine levels, and toxic wastes may further threaten shellfish, fish, waterfowl, and other bay dwellers.

The Chesapeake Bay in former times was seen as the great connection, It connected towns, cities and farms, serving as a ready highway. It still does connect them, even though we with our land-based modes of transportation think of it as a divider. Today the bay, as it drains 64,000 square miles of our country ties these areas together. Today more people are becoming aware that what they do upstream affects others and its the bay that is helping them realize this. It may be the problems of the bay that serve to unite these people, giving them a sense that caring for the environment is everyone's concern, and an active, complex, difficult concern at that.

ACKNOWLEDGEMENTS

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DREDGING AND DISPOSAL IN THE CHESAPEAKE BAY

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ABSTRACT

The management questions involved in dredging and open water dredged material disposal revolve around three basic issues: 1) stability of the dredged material within the defined disposal area, 2) altered resource value of bottoms affected by dredged material, and 3) mobilization of toxins from contaminated dredged material. These issues encompass all aspects of the physical and biological character of a project site. The actual evaluation of dredging and dredged material disposal has generally concentrated on acute impacts to bottom dwelling organisms, benthos, and or to water column characteristics. This emphasis has resulted in ample evidence that many disposal practices have at least short term detrimental effects. With few exceptions, these studies have failed to assess the relationships between the benthos and the dredged material as a new sediment habitat, or the consequences of altering the hydrodynamic regime, or the resource value of the benthos.

Unless toxics are involved the natural process of recolonization or recruitment will return benthos to the disposal area. The key questions then for effective management of dredging and disposal revolve around longer-term processes that will influence recolonization, these are natural sediment dynamics, hydrodynamics, and biogenic activity.

INTRODUCTION

Most studies dealing with dredging and disposal usually find an acute impact that is relatively short lived (McCauley et al. 1977, Rosenberg 1977, Flemer et al. 1968, Diaz and Boesch 1977). In areas where the impacts are severe long-term disturbance may result, but these types of dredging projects, at least in the literature, are not the norm (Kaplan et al. 1975, Rosenberg 1977). In most areas where disposal effects are long-lived toxics or other factors usually play a significant role in the disturbance (Saila et al. 1972). Disregarding the case where toxics are involved, since dredged material that is classified as toxic is not disposed of in open water, initiation of recolonization by whatever means is usually quick.

Communities are well on their way to some recovery point within days or months depending upon the particulars of the environment concerned.

With the reality that dredging and disposal will occur and there will be some acute impact, any dredged material management plan must look beyond the acute and evaluate the consequences of dredging activities in the long-term. Consideration for long-term alteration to important processes that regulate the value of the habitat being managed should be paramount. The value of a habitat being either a direct, such as protection, or indirect, such as trophic support, ecological support for fisheries species is usually the ultimate concern for environmental management (Lunz et al. 1978, Lunz and Kendall 1982).

A management approach to dredging activities in open water needs then to consider the long-term stability of dredged material within a disposal area and potential for alteration of the bottom resource value. In this paper we will present how the physical and biological components of the environment interact with dredging activities.

MECHANISM OF IMPACT AND RESPONSE

Most studies dealing with dredge material disposal, in open water, usually find an acute impact that is relatively short lived. Initiation of recolonization, by immigration or recruitment, is usually quick with communities well on their way to "recovery" within days or months depending upon the physical character of the environment and the season.

The mechanisms of the impact in all cases can be reduced to several common elements:

1 - Physical disruption of benthos by burial.

2 - Instability of new sediment surface and changes in mass properties cause problems in support and respiration.

3 - Dredge material retains its original geochemical composition after disposal resembling digenetically mature deep sediments. The increased elemental flux and oxidation reactions when these sediments are placed on the surface pose potential toxic and low dissolved oxygen stress to the benthos.

4 - Changes in particle sizes available to benthos and loss of food value. While dredge material may be high in organic content it all tends to be highly refractive and unavailable to benthic feeders.

The responses of the benthos to these elements can also be summarized:

1 - Reduction of individuals and species through death which leads to reduction in standing crop and resource value.

2 - Physiological stress induced in survivors by increased elemental fluxes and lowered dissolved oxygen.

3 - Rechanneling of energy to maintain feeding, respiration, and support.

With time the process of recolonization, either through immigration or larval recruitment, quickly puts the benthos back into a recovery phase. Time then is the common element that lessens the physical disruption of the dredged material and guides the recovery phase.

LONG-TERM PROCESSES

There are three long-term processes that are important in the context of dredging and dredging impacts on habitat value. They are natural sediment dynamics, hydrodynamics and biogenic activity. These processes are at work continually to shape the benthos and determine to a great degree the resource value of the bottom to fisheries species. Dredging and disposal then need to be considered in light of how they fit within these long-term processes to either impact or enhance the value of a bottom. With this in mind less emphasis need be placed on acute effects.

NATURAL SEDIMENT DYNAMICS AND HYDRODYNAMICS

At one time or another all these elements fit into the natural dynamics of sedimentation. The problems come when the scales of events are compared between natural and dredging processes. For example, natural turbidity is generated from resuspension of surface sediments whereas dredging turbidity comes from the suspension of deeper deposits. A schematic representation of natural sediment dynamics is presented in Figure 1. This cycle is at work over the entire subtidal environment with the rates of flux from one state to another dependent upon weather and tides. The benthos have evolved within this natural sediment cycle and are adapted to the particular disruptions encountered in various environments.

Dredging and disposal alter this sediment cycle, at a localized level. For a short period of time turbidity is caused by deep deposits, the original sediment surface is buried leaving the new surface composed of deep sediments. After disposal hydrodynamic forces quickly bring the disposal area under the influence of natural sediment dynamics. The dredged material quickly starts to lose its digenetically mature character and is immediately covered by thin layers of natural sediment.

It is the adaption of the benthos to the workings of natural sediment dynamics that allows acute impacts of dredging to be short lived. Since new dredged material resembles more a deeper deposit initial colonizers tend to be the opportunistic species because of their wider environmental tolerances and tendency to live within the very surface sediments.

BIOGENIC ACTIVITY AND SUCCESSIONAL STAGE

An important factor in the colonization of opportunists is the thin veneer of natural sediments that quickly covers the dredged material. Early colonizers tend to be closely associated with the water-sediment interface and either suspension or surface deposit feed. These initial recolonizers immediately start to modify the sediments through irrigation and reworking. A successional sequence is then initiated in the dredged material that leads toward development of "climax" community and substrates (climax being used to describe the benthos and sediments from a similar natural habitat that has been undisturbed.).

The path sediment succession takes is most predictable being dependent on very general categories of benthic organisms, from initial surface dwellers to later deep infauna. The succession of the benthos is less predictable, from the onset it is directed by the makeup of the sediment. As species set and grow larger the amount of biogenic activity increases. Both sediment and benthic succession are interdependent, one does out proceed far without the other. Sediment succession is very dependent on initial stages of benthic succession while later stages of benthic succession will be delayed until "climax" sediment succession is reached. The lag and interplay of these two successions may account for the disparities in recovery times noted among the studies of acute dredging impacts.

On dredged material or any defaunated natural bottom the rate of recovery of the benthos is mainly a function of the long-term stability of the system. Dredging creates a localized biological vacuum that disrupts communities. Initially more individuals can temporarily occupy the new habitat. With time species interact and turnover, and depending on the sediment quality of the dredged material and barring toxicants the resource value of the benthos returns to some level.

A PRACTICAL EXAMPLE

Plans have been developed over the last 10 years to deepen the main navigation channel up the Chesapeake Bay to Baltimore from 42 to 50 feet. In Virginia waters approximately 33 million cubic yards of sediment will be dredged and disposed of in two open water sites. The disposal plan and monitoring program were developed from interactions between the Baltimore and Norfolk District Corps of Engineers and the Commonwealth of Virginia. The monitoring plan while documenting the acute effects concentrates on the long-term impacts.

The basic management strategy in developing the disposal and monitoring plans were to minimize acute impacts and follow the resource value of the bottom for long-term changes that may be related to the disposal operation. With this in mind a baseline study was undertaken to assess existing conditions of the benthos and bottom sediments and estimate the magnitude of their spatial and seasonal variability (Diaz et al. 1985). The resource value of the benthos in trophic support of fisheries species was estimated using the Benthic Resources Assessment Technique developed by Lunz and Kendall (1982).

We found the composition of the benthic community and its final resource value to be controlled by sediment type, salinity, depth, and seasonal dissolved oxygen depression. For a given salinity the benthic resource value was higher in sediments having a mixture of sand and mud (silt-clay) relative to the sediments that are pure sand or clay or silt. In these areas of mixed sediments the biogenic structure of the sediment was well developed and the communities characteristic of mature successional stages. These areas supported a high biomass of benthos that was being utilized by fisheries species. Communities in pure sand or mud did have a resource value but it was lower than mixed sediments, with sand having a higher value than mud. Areas that were pure mud and stressed by low dissolved oxygen had the lowest resource value.

The total area of the Bay can be broken down into sand, mixed, and silt-clay habitats, as follows:

	Sand	Mixed	Silt-Clay
VA+MD	57%	25	18
VA	67	20	13

For the purpose of long-term management of the bottoms resource value it would then be most prudent to protect the areas of mixed sediments. The possibility also arises that resource value of silt-clay areas may be increased by the addition of sandy sediments, assuming other important factors such as dissolved oxygen or sediment stability are not problems.

The Virginia disposal sites identified for use in the Baltimore channel project involve 2% of the Virginia bottom with higher benthic resource value near Wolf Trap and 0.3% of the lower resource bottom near Rappahannock Shoals (Figure 2). One disposal site in each area will be used. At Wolf Trap the sites were very similar being mostly mixed sediments of high resource value. Neither site is significantly higher in value. At Rappahannock Shoals the primary site varied from pure mud to sand and also had low and high resource value habitats. The alternate site was uniformly muddy and had overall a moderate to low resource value. It would then seem, in the long run, most appropriate to use the alternate site for disposal. The possibility also exists that the sandier channel sediments will raise the value of the alternate site.

To minimize any of the long-term impacts of the channel deepening it would seem that at Wolf Trap the key is the rate of spread of the sediment after disposal. The communities present are not adapted to high rates of sediment accumulation. If the hydrodynamic regime spreads the material "slowly" then it is likely that the high resource value of the region will be preserved. On the other hand rapid movement of the dredged material out of the disposal area will likely cause depression in resource value.

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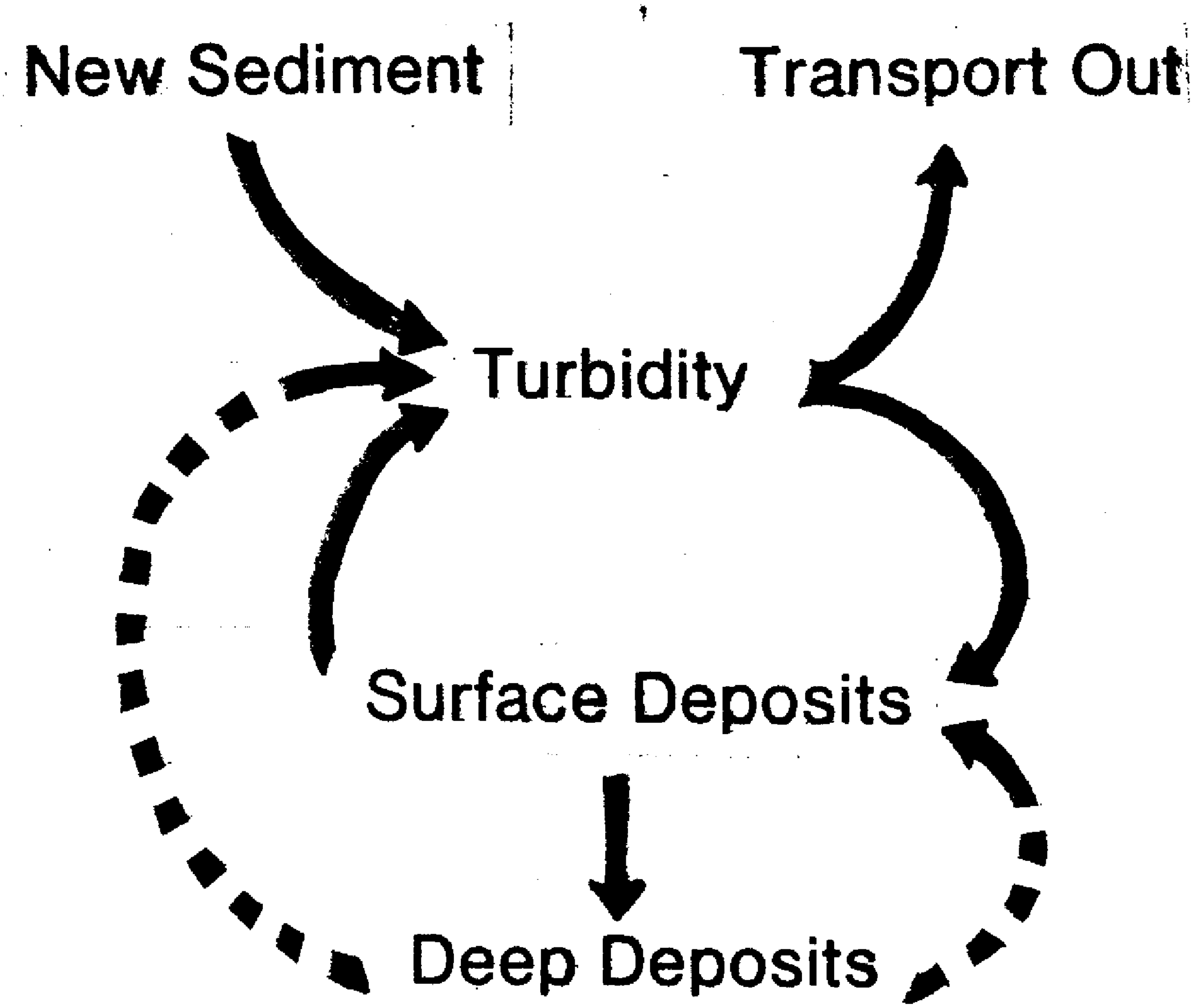

Figure 1. Schematic representation of natural sediment dynamics and how dredging and open water disposal affect these dynamics. Dredging affects are depicted by broken lines. Surface deposits are considered to be on the order of 15 cm in thickness.

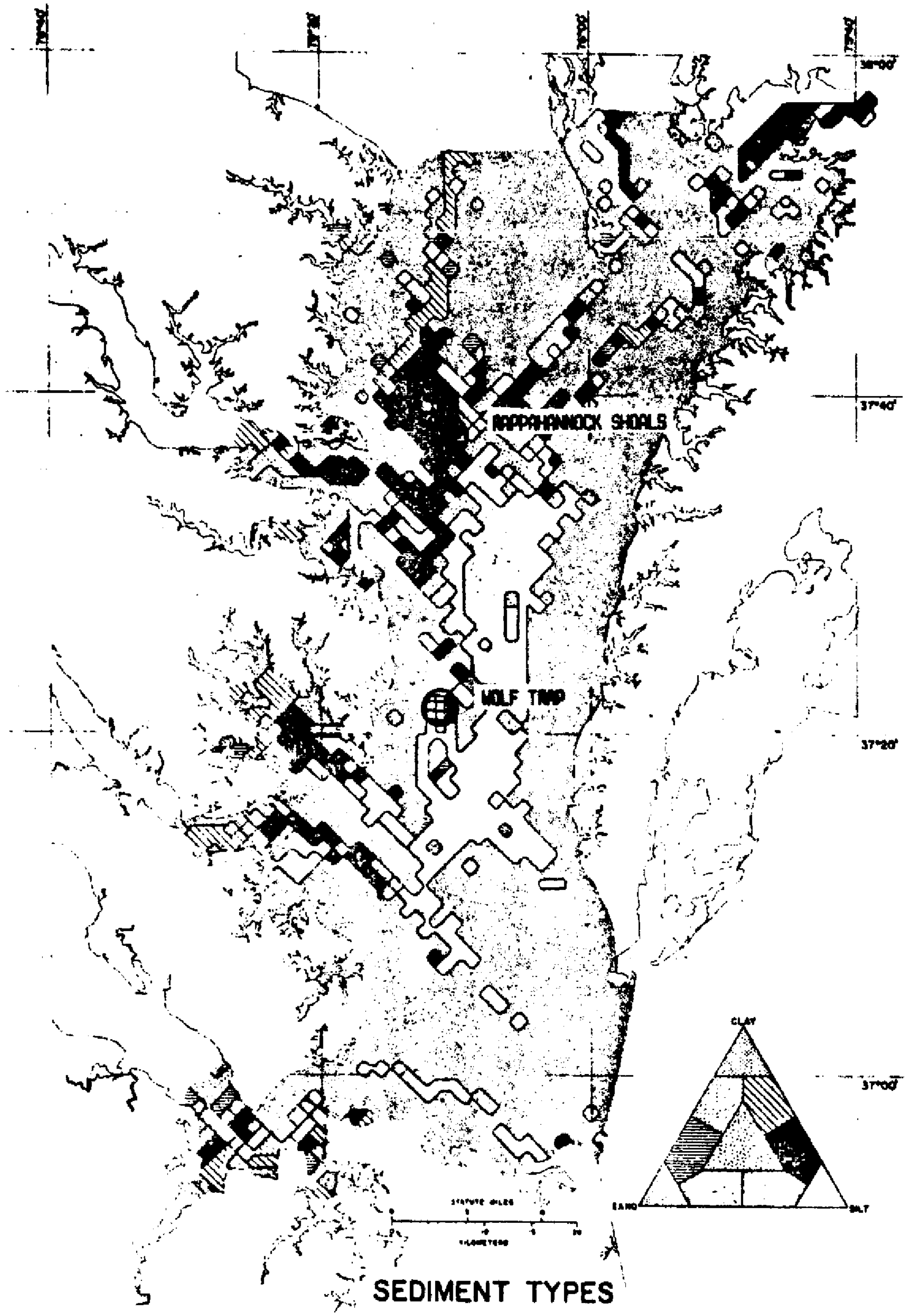

Figure 2. Map of sediment types in the lower Chesapeake Bay with approximate location of the Wolf Trap and Rappahannock Shoals disposal areas.

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MAN'S PHYSICAL EFFECTS ON THE ELIZABETH RIVER

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ABSTRACT

Man's ever increasing activities in the Elizabeth River, i.e. dredging, disposal of dredged material and waterfront development, have drastically altered the river floor, reshaped the shoreline and changed the circulation. Long-continued dredging of shipping channels, which is fostered by coal export, larger ships, and military needs, has moved 220 million cu yds of sediment since 1870. As a result channel depth has increased 1.8 fold, and maintenance dredging rates have doubled about every 35 years. Open water disposal released 40 million cu yds into Hampton Roads and lower Chesapeake Bay. Landfill buried tributary creeks, moved the waterfront into the river and reduced the river area by 27%. As a consequence of reduced area and greater channel depth, current velocity has diminished and near-bottom salinity likely increased. These conditions induce faster sedimentation that in turn, creates a need for greater maintenance dredging and hence, greater disposal. The dredge and fill cycle, therefore, is self-perpetuating. The long-term trends of channel deepening, enlargement, and landfill, are expected to continue in response to larger ships, military needs and projected sea-level rise.

INTRODUCTION

Ever since Europeans settled along the Elizabeth River in the 1600's, man's activities have reshaped the shoreline, altered the river floor, used and abused the river in different ways. Formerly the river had numerous tributary creeks, extensive marshlands and beaches fronting Hampton Roads. Today, after 100 years of accelerated development, many creeks and marshlands are buried and the beaches are replaced by bulkheads and shipping facilities. The objective of this study is to show that the present status of the river is a product of past activities. Not all large changes are of recent origin. Instead, they were small at first and have continued piecemeal over many decades. Since the changes are permanent and additive, the large-scale problems we face today are caused by cumulative effects of small changes over a long time. To gain evidence for this idea we addressed the question: How has the kind, the rate and magnitude, of change shifted with time in response to man's activities? Understanding the causes and effects allows us to predict what future changes are likely to occur.

INFORMATION SOURCES AND METHODS

Charts of the U. S. Coast and Geodetic Survey dated 1853, 1872-73, 1908 and 1982 provide information to determine shoreline and bathymetric changes after adjustment to a common vertical datum, and reduction or enlargement, to a common scale in a Map-O-Graph unit or by photography. Data on the amount and location of material dredged was derived from extensive files and annual reports of the U.S. Army Engineer District, Norfolk. These data consist of: (1) project records of federal, Corps controlled, anchorages and channels, and (2) permit records of non-Corps projects, both federal and private controlled, anchorages, berths and channels. The location of material dredged was obtained from Corps survey charts which were prepared from surveys before and after dredging as well as at other times. Dredging projects are categorized as: (1) "new work," which removes undisturbed old material to enlarge a channel, or to increase its controlling depth, and (2) "maintenance dredging," which removes material accumulated in a previously dredged channel. Figure 1 shows the distribution of dredged areas, i.e. Corps and non-Corps controlled.

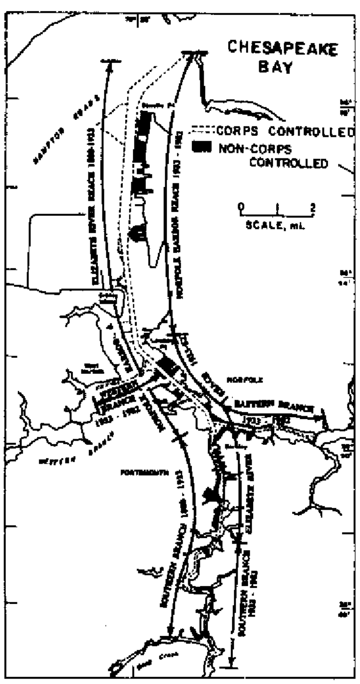


Figure 1. Distribution of dredged channels and areas, Corps and non-Corps controlled. Note, the extent and names of various channel reaches were redefined at various dates.

HISTORICAL BACKGROUND

The trends of dredging and filling have proceeded with growth of the cities of Norfolk and Portsmouth and the changing patterns of maritime trade, industry and military activities. Man's activities in the river evolved through four stages:

(1) Colonial agriculture and early port development, 1625-1785
(2) Port expansion, 1785-1880
(3) Large-Scale development, 1880-1955
(4) Modern development, 1955-1982

Settlement of the river began in the early 1600's as part of the plantation tobacco economy in the region. Norfolk, founded as a town in 1682, was just a local shipment point. There was no need for a port in the region because most planters shipped direct to Europe from their own wharves. By 1725, however, trade with the West Indies and Europe, and the interchange of goods in the North Carolina-Chesapeake Bay region, fostered port growth together with development of ship repair and shipbuilding facilities. The wharves were built toward the main channel at first, but as trade expanded in the 1700's, interwharf shores were filled for docks, thus moving the waterfront into the river. In 1802, 12 wharves existed at Norfolk near Town Point (Wertenbaker, 1962).

With the coming of steamboats about 1820, much trade by-passed Norfolk going to New York, Baltimore and Richmond, but this was partly offset by expanded trade through canals connecting the river with North Carolina and the Roanoke Valley. Construction of a naval shipyard at Portsmouth about 1812, generated much activity, including expansion of waterfront facilities along the Southern Branch of the Elizabeth River. As Norfolk and Portsmouth grew, expansion shifted into creeks and marshland which were used for disposal of refuse, ship ballast stones, construction debris and oyster shell. Small streams were converted into sewers and large creeks into canals.

When railways reached the river from southwest Virginia coal fields in 1880, and steamboats were improved to transport coal, large-scale development followed. By 1889 transhipment facilities were completed at Portsmouth, Berkley, Eastern Branch, Sewells Point, Pinner Point and West Norfolk. Pier slips were dug out and access channels dredged from the river to the slips. These facilities brought more ships and larger ships with deeper drafts than in earlier decades. More open water anchorages were required and with increasing ship draft, deeper channels were necessary to provide safe passage over shoals through the river and to the piers. This trend is still in progress today.

DREDGING TRENDS

Channel Depth and Size. The first effort to deepen the main shipping channel began in 1872 with dredging of entrance shoals off Sewells Point and off Town Point, Norfolk. By 1876 Congress authorized a comprehensive 25-foot channel for 10 miles from Sewells Point to

Norfolk and Portsmouth. This was the first stage in a series of dredging projects by which the shipping channel was deepened between 1880 and 1968. Figure 2 shows how the natural irregular profile of the channel floor was progressively smoothed and lowered by removing shoals and filling holes. Although each increment of dredging was relatively small, over 100 years the overall increase in depth is great, i.e. 1.8 times greater than the original average depth.

When channels are dredged deeper, they are also often lengthened, enlarged and straightened relative to their predredged condition. Whereas the first comprehensive project (1880-1889) extended 10 miles, today, dredged channels extend throughout the river and into certain tributary creeks, a network that penetrates landward 27 miles from the mouth. Today's channels, which are regularly dredged, occupy about 2.9 sq miles or 25% of the original river area. As evident in Figure 1, much larger proportions of the original river floor have been dredged in narrow reaches of Southern Branch than elsewhere. Furthermore, dredging has increased the fluid volume of the river from about 184 million cu yds in 1872 to 276 million cu yds in 1982, a 50% increase of its original volume.

Amount of Dredged Material. When the main channel was deepened to 21 and 25 feet between 1872 and 1911, repetitive maintenance dredging was infrequent and produced less than 0.5 million cu yds per year on the average. When the channel was deepened to 30 feet in 1906-1911, the rate of maintenance dredging during the next two decades increased more than two-fold (Fig. 3). Dredging rates reached a peak in 1940 when 6.6 million cu yds were removed from the Norfolk Harbor Reach as a World War II emergency effort. In addition, deepening the channel from 30 to 38 feet removed an additional 7 million cu yds and improving pier slips removed 2 million cu yds. During post-War years, maintenance dredging rates in Norfolk Harbor Reach fluctuated

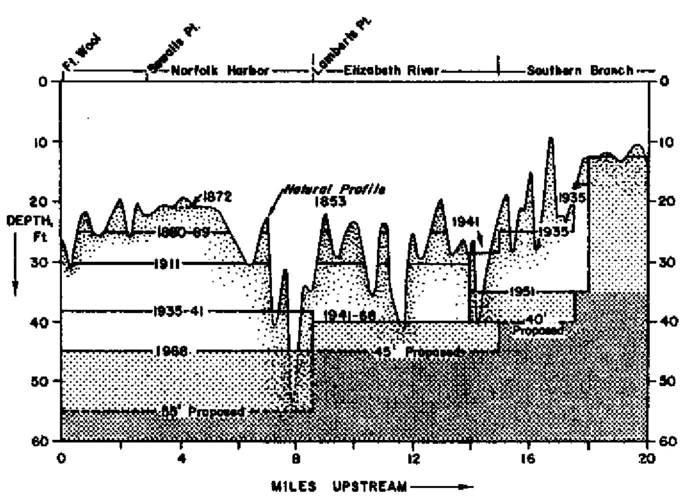
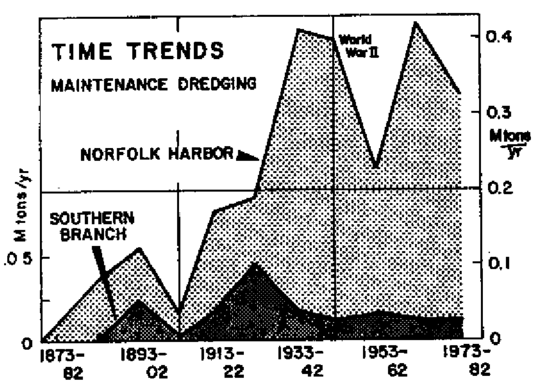


Figure 2. Change in the longitudinal channel profile along the Elizabeth River with time and stages of deepening between 1853 and 1968. Lowermost step-wise profile, dashed, is the controlling depth for a proposed project (U. S. Army, 1979).

from about 0.3 to 2.8 million cu yds annually. These variations reflect changes in sedimentation rate as well as available funds for dredging that fluctuate with economics and political pressures. Whereas dredging rates in the Norfolk Harbor Reach persisted at a substantial level between 1960-1980, rates in the Elizabeth River Reach and Southern Branch diminished slightly (Fig. 3).

When the dredging rates are averaged by decade over 100 years and considered as function of channel volume, a statistically significant correlation is disclosed (Fig. 4). As channel size, mainly depth and width, increased, maintenance dredging rates also increased. If channel size is increased by the proposed 5 to 10-foot deepening, the historical trends predict dredging rates in the subsequent decade will increase to an average of about 2.2 million cu yds per year, an increase of 50% compared to rates between 1963-1982. This trend implies that as channels are dug deeper and larger, faster rates of sedimentation are induced. In turn, this creates a need for dredging greater amounts of maintenance material as well as larger amounts to dispose of. Dredging, therefore, is self-perpetuating.

<u>Distribution of Dredged Material</u>. Sediment deposited in the river is not distributed uniformly on the shipping channel floor but mainly



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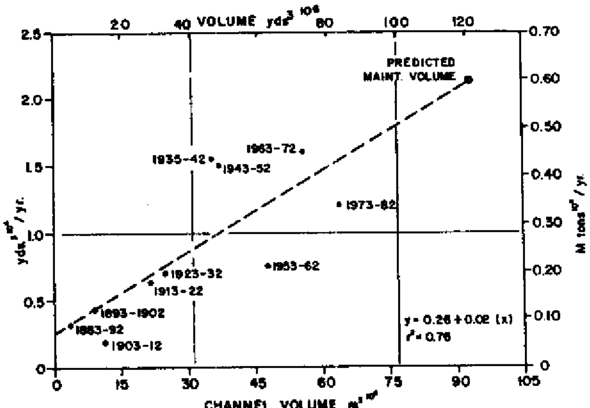

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Figure 3. Temporal trends of annual maintenance dredging rates averaged by decade for Norfolk Harbor reach (right scale) and Southern Branch (left scale) between 1873 and 1982. One megaton (M, or million ton) is equivalent to about 3.64 million cubic yards.

Figure 4. Maintenance dredging rate as a function of channel size (volume) between 1883 and 1982. Rates represent average annual rate by decade in Norfolk Harbor reach. Linear regression line, dashed, excludes anomalous data of World War II, 1935-1952.

accumulates in shoals along lower margins of the channel (Fig. 5A) as well as in pier slips and berths. These zones are favorable for sediment accumulation because they are less energetic than the central channel that is churned by ships or tidal currents. As shown in Figure 5B, the distribution of average annual amount of dredged material by volume, is greatest in the Norfolk Harbor reach. Landward from Lambert Point in the 40-foot channel, the dredged quantities drop abruptly and then decline further in the Southern Branch. A similar distribution is displayed for average dredged rates (Fig. 5C) and for sedimentation rates, which are estimated from the dredge rates. Review of historical distributions (e.g. Fig. 3) indicates that while maintenance dredging rates have changed drastically with an increase in channel size, the location of maximum dredged rates in the Norfolk Harbor Reach has remained essentially the same. This reach is close to a major supply of sediment that enters the river from Hampton Roads via landward flow through the lower salt layer. Alternately, it enters via the upper layer over shoals north of Craney Island.

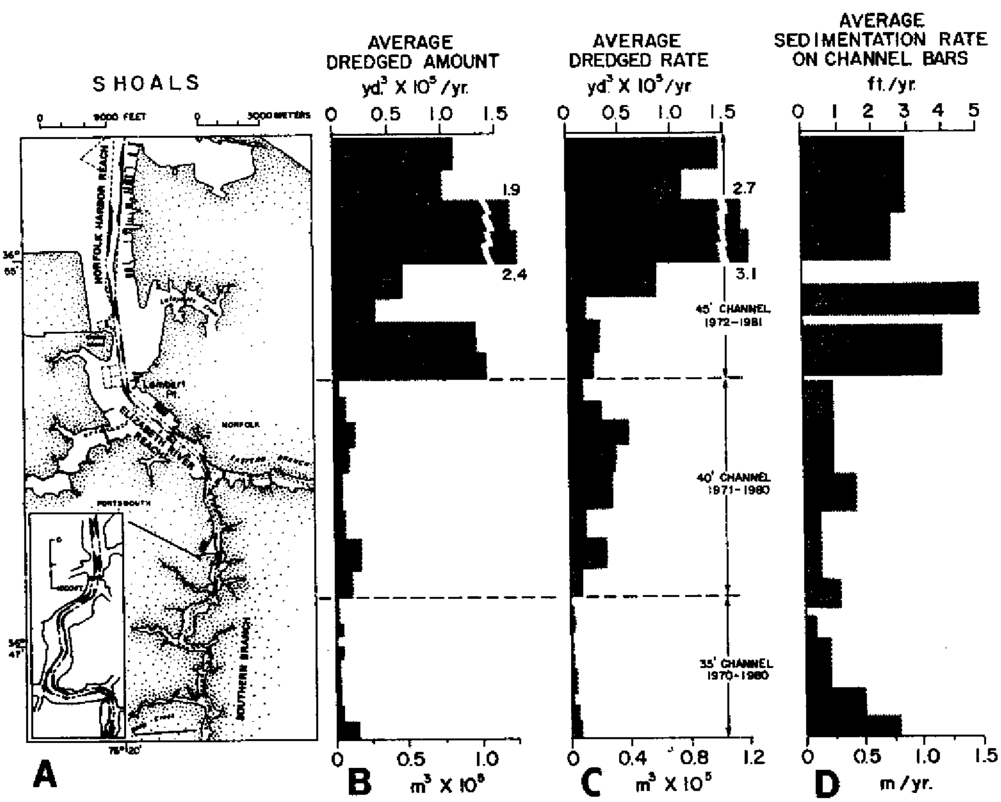

Figure 5. Distribution of maintenance dredged material in the Elizabeth River. (A) approximate location of shoals, black, (B) average dredged amount, by volume, (C) average annual dredged rate, and (D) average sedimentation rate on shoals in the channel which is derived from the dredging rates over a 10-year period. Data based on bathymetric changes compiled by Berger et al. (1985) and adjusted to agree with total "credited" dredged amounts of Corps of Engineers annual reports.

DISPOSAL TRENDS

Once dredging is performed, either new work or maintenance, it becomes necessary to dispose of enormous amounts of material. Where then, does it all go?

Open Water Disposal. From 1872 to 1889 dredged material from the main shipping channel reportedly was dumped near sites where it was dredged, i.e. deep holes of the river and on shoals around Craney Island and the Lafayette River mouth. Lacking space within the river, sites were subsequently moved about as its successors were filled up. Between 1893 and 1919 over 10 million cu yds were dumped outside the river in lower Chesapeake Bay, an area east of Fort Wool called the "Rip Raps" (Fig. 6). From 1918 to 1940 an estimated 7 million cu yds were dumped in a broad area off Lynhaven Bay. The Lynhaven site was restricted by amphibious training activities, therefore, dumping was moved toward Thimble Shoals channel in 1941-1942. This site was discontinued, however, because of adverse effects on the shipping channel. During World War II and until 1951, 20 million cu yds were dumped in lower Hampton Roads (Fig. 6), a deep water site where material is subject to redistribution by relatively fast currents. Recognizing the problems of open water disposal and need for long-term disposal capacity, the Corps of Engineers constructed in 1954-1957 a four square mile enclosed disposal basin located north of Craney Island (Fig. 6). Use of the Craney Island

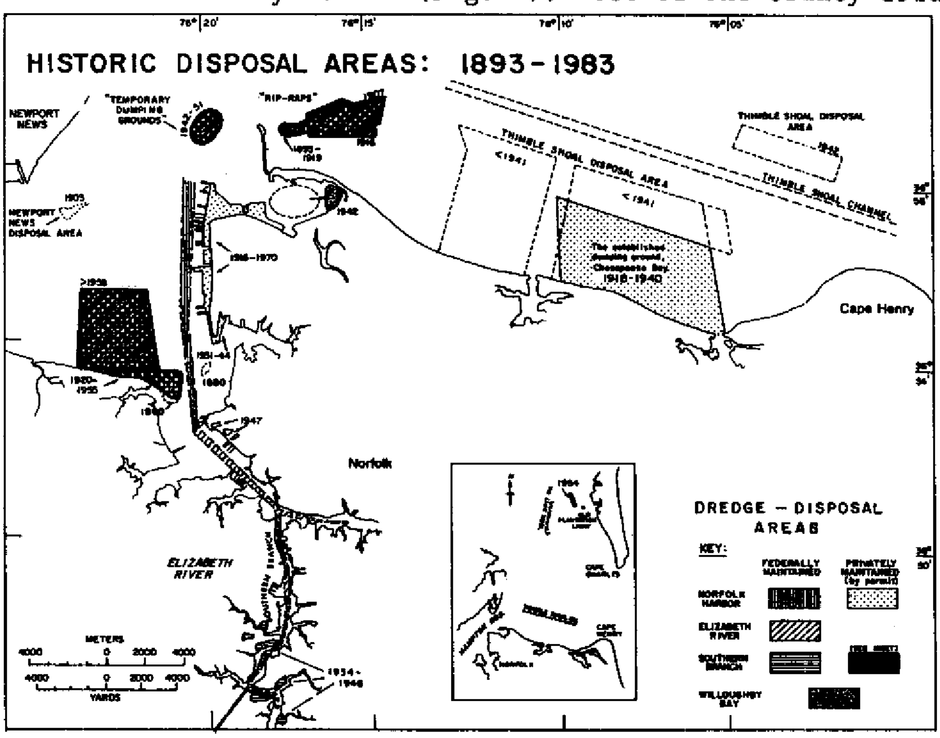

Figure 6. Historic disposal areas in open water areas and the Craney Island disposal basins. Dotted and hachured patterns key the disposal area to the channel from which the material was reportedly derived.

disposal area ended open water dumping of dredged material from the river, but 58 years of prior dumping left behind an estimated 40 million cu yds in lower Chesapeake Bay and Hampton Roads.

Landfill. In the 1870's the "Common Council" of Norfolk requested that dredged material from the river be used to fill lowlands within the city. Land reclamation was needed not only to improve drainage, and thus alleviate the menace of yellow fever, but to provide more space to expand the city. Filling began about 1878 in small creeks and in bordering marshlands along the waterfront at Norfolk and Portsmouth. By 1908 large areas were reclaimed for coal transhipment facilities at Pinner Point and Lambert Point as well as for the Jamestown exposition at Sewells Point (Figs. 7, 8). Accelerated activity during World War I created piers, docks and associated landfill for an army base near Tanner Point, for a naval base at Sewells Point, and for bulkheads and pier slips in Southern Branch near the Portsmouth Naval Ship Yard (Fig. 7). Intense naval activity during World War II produced large-scale changes including pier construction and land reclamation at Sewells Point, dredging and

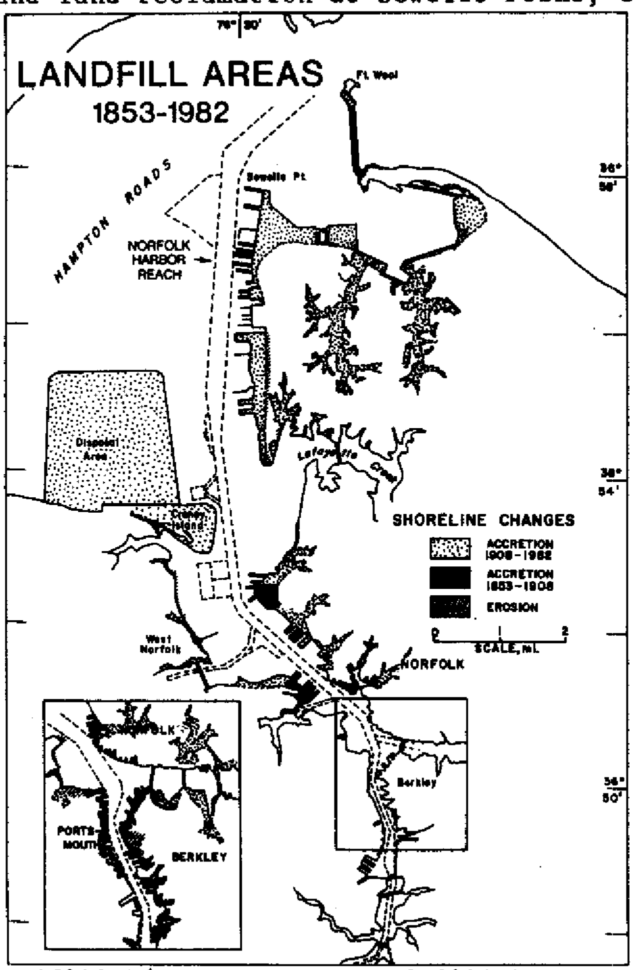

Figure 7. Landfill history. Zones of fill between 1853-1908 (black) and 1908-1982 (dotted). Data based on shoreline changes from old U.S.C. and G.S. charts. Main shipping channel, dashed.

filling on east side of Willoughby Bay and filling of two large creeks for an airfield. Additionally, much dredged material was disposed behind dikes on the original Craney Island and adjacent lowlands west of the island. After 1957 most dredged material was disposed in the Craney Island disposal basin. This basin extends the river mouth 2 miles seaward. When filled to capacity it will add another increment of reclaimed land to the river shoreline.

In summary, little is left of the original shoreline. Filling has proceeded piecemeal over 100 years, first here, then there. The general pattern is: (1) initial fill around the original urban hubs, (2) later, fill in seaward zones as Sewells Point, and then (3) centralized fill in the Craney Island disposal basin. The cumulative effect of filling over many years is to move the shoreline into the Elizabeth River, thus narrowing the river and reducing its surface area. Altogether about 27% of the original river area has been lost.

DREDGE AND DISPOSAL BUDGET

Although most changes produced by dredging and disposal occur in small increments, they are mainly permanent changes. Therefore, it remains to size up the amount of dredged material, determine the cumulative amount over 100 years and compare the amount with the disposal amount reported or estimated. Table 1 provides relevant data for the total cumulative amount in various categories.

Of the total dredged material, 220 million cu yds, 168 million cu yds is accounted for. The apparent deficit of disposal material, 53 million cu yds, or 24% of the total amount dredged may be caused by: (1) incomplete records showing where dredged material was disposed and how much, (2) lack of detailed bathymetric surveys of disposal sites, (3) incomplete surveys or measurements of landfill volume, (4) unknown contribution of other material, other than dredged material, to the measured landfill volume, (5) apparent "loss" of material by current dispersal, or by settlement and

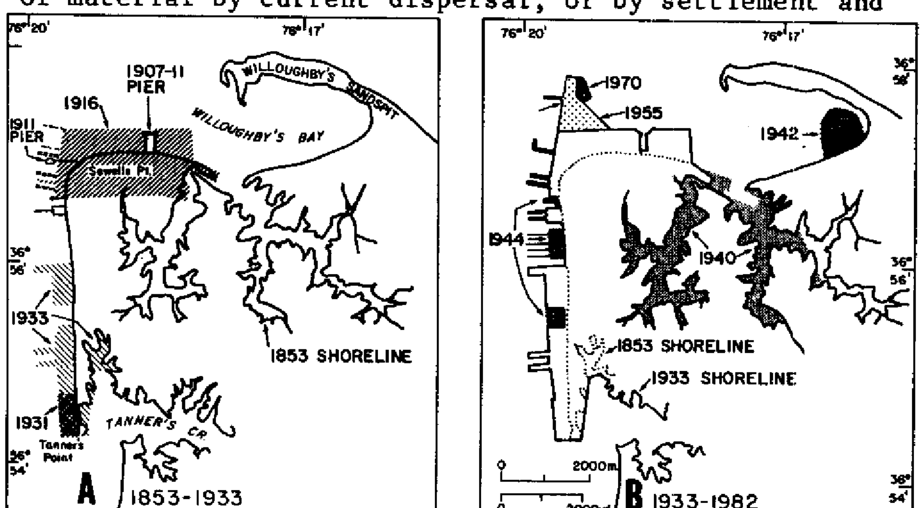

Figure 8. Evolution of landfill in the Sewells Point-Tanner Point area. Based on shoreline changes from old charts and reports. (A) 1853-1933, (B) 1933-1982.

sediment consolidation. By applying simulated consolidation rates for material in the Craney Island disposal basin (Palermo et al., 1981) to landfill, 15% of the deficit is accounted for.

The cumulative amount of dredged material (Table 1), reveals the enormous amount of sediment moved from a relatively small river. For comparison, the amount dredged from the Elizabeth between 1956-1982, which averages 3.7 million cu yds per year or about 1.1 million tons per year, is equivalent to the average 64% of the annual river input of sediment, 1.7 million tons per year, transported by the James River at Richmond. Dredging and disposal therefore, are major geologic processes.

DREDGING			DISPOSAL		
	VOLUME (YDS^3) 10^6	TOTAL %		VOLUME (YDS^3) 10^6	TOTAL %
FEDERAL (CORPS):			FEDERAL (CORPS):	63	37
MAINTENANCE	97	44	OPEN WATER (40)		
NEW WORK	52	24	OTHER (23)		
PERMIT, NON-CORPS			PERMIT, NON-CORPS	88	53
MAINTENANCE	45	20	CRANEY ISLAND (53)		
NEW WORK	26	12	OTHER (35)		
			UNDIFFEREN'TD:	15	10
TOTAL:	220		TOTAL:	166	

HYDRODYNAMIC EFFECTS

As landfill reduced the river surface area between 1853-1982, the intertidal volume of water, or tidal prism, also diminished, an estimated 24% of the original prism. Consequently, maximum tidal currents through the mouth are weakened, an estimated 17% of the natural velocity. Hydraulic model tests of a proposed 5 to 10-foot channel deepening (Richards and Morton, 1983), showed that maximum flood and ebb current velocities near the bottom, diminished at a majority of points, by 0.10 to 0.39 fps. Additionally, near-bottom salinity increased 0.5 to 4.0 ^{o}/oo and stratification intensified. Although the depth changes tested were relatively small, it is likely that the trend of lower velocity and higher salinity is part of a large long-term trend produced by successive increases of channel depth. The hydrodynamic effects enhance sediment trapping in the river and thus induce faster sedimentation as well as better retention of pollutants. Sedimentation rates in undredged tributary rivers of lower Chesapeake Bay are about 0.2 cm per year. By comparison rates in the Elizabeth are 15 to 195 cm per year, an increase of 10^2 to 10^3 fold (Table 2).

FUTURE TRENDS

The cycle of dredge-fill-sedimentation will continue. Dredging is the means by which maritime commerce and military activities are maintained. Without dredging the port could not remain competitive. The main site of sedimentation and dredging may be expected to remain the same. The site of disposal however, will change after the Craney Island disposal basin is filled to capacity in about 30 years. Future disposal activity may either continue to centralize the material adjacent to, or within, the river; alternately it may move the site seaward. As containerization of cargo reduces the need for dockside loading, old dock facilities will be rebuilt and modernized. Therefore, fill around old piers and docks will continue to move the waterfront into the river. Because containerization requires large open storage areas, reclamation will shift to lowlands landward of the river.

As man's activities are also changing global climate, the long-term rise of sea level will accelerate. Today's rates in Hampton Roads, which include subsidence, are about 9 inches per century. Hoffman et al. (1983) project global rates will increase 3 to 17 times during the 21st century and reach a level 2 to 12 feet higher than now by the year 2100. To offset flooding, existing and future waterfront areas on the Elizabeth will require higher elevation. Dredged material is a likely asset to provide the needed elevation. Rising sea level however, increases water depth. It remains to be seen if the accelerated rise of sea level will keep pace with or exceed the trend of larger and deeper draft ships and hence alleviate increasing dredging rates. Table 2 compares the physical changes, past, present and proposed.

ACKNOWLEDGEMENTS

This study was supported by grant R806002-02 of the U. S. Environmental Protection Agency, Chesapeake Bay Program. Thanks are due to Mr. Gene Whitehurst of the U. S. Army Engineer District, Norfolk, for assistance with Corps dredging data. Kay Stubblefield of VIMS drafted the figures. This is a VIMS contribution.

Table 2. Summary of physical changes in the Elizabeth River.

Feature	Past[1]	Present	Proposed[2]
Mean Channel Depth, ft.	19	35-45	40-55
Channel Length, mi.	10	27	27
Channel Width, ft.	200-400	250-1,500	250-1,500
Channel Volume, $10^6 ft^3$	900 $10^6 ft^3$	3,394	3,850
Maintenance Material, $10^6 yds^3$	0	3.7[3]	4.3
Sedimentation Rate, cm/yr	0.2	15-195	19-240
River Surface Area, $10^6 ft^2$	325	239	---

[1] Natural or originial condition, 1853-1880; [2] U.S. Army Engineer District, Norfolk, 1979; [3] Average, 1963-1982.

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IMPACTS OF ALUM SLUDGE ON TIDAL FRESHWATER STREAMS

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ABSTRACT

Alum sludge, generated in the processing of surface water for drinking water supplies, has traditionally been discharged into nearby streams in Virginia and elsewhere. Alum sludge contains aluminum in an insoluble and non-toxic form. There remains concern, however, that alum sludge may have a negative impact on receiving waters.

The distribution of aluminum-enriched sediments was determined as the change in aluminum:silicon ratio in three tidal freshwater streams receiving alum sludge discharges. Primary production, marsh community status, and benthic invertebrate community composition were determined along each stream. Primary production of phytoplankton and benthic algae was lower at an upstream station than at a downstream station in two streams. Marsh communities were judged to be typical for streams of this type. Benthic invertebrate communities increased in species richness and total abundance with increasing distance from the discharge point in all streams. The biological parameters were compared with aluminum:silicon ratio and salinity gradients and with biological communities along similar streams not receiving alum sludge. Changes in the biological community were concluded to be more closely related to salinity structure of the systems than to introduction of alum sludge.

INTRODUCTION

Alum flocculation has proven to be an effective method to purify surface waters for municipal drinking water supplies. Water treatment facilities must ultimately dispose of an alum sludge, a waste consisting of aluminum hydroxide plus various inorganic and organic particles. While not considered toxic (Burrows, 1977), alum sludge is bulky with a high water content. A long-standing method of disposal has been discharge into receiving streams downstream of the water treatment plants. Alum sludge remains in suspension for some time during which it is carried downstream before being incorporated into the bottom substrate.

In tidewater Virginia, the Cities of Norfolk and Newport News both utilize alum flocculation at several water treatment plants. The resultant sludge has for as many as 80 years been discharged into various subestuaries of the Chesapeake Bay system. In Norfolk, the Moores

Bridges plant discharges into Broad Creek, a tributary of the Elizabeth River Eastern Branch (Blair et al., 1983). In Newport News, the Harwoods Mill plant discharges into the Poquoson River, the Lee Hall plant into the Warwick River.

The focus of the present report is the ecological effects of alum sludge in the streams near the point of discharge. Ecological effects might include: 1) changes in primary production, 2) changes in the fringing marsh community, and 3) impacts on the benthic invertebrate community. We defined the distribution of alum sludge and evaluated the effect of the discharge in several components of the stream ecosystems; phytoplankton, benthic algae, marsh community development and benthic invertebrate community structure.

MATERIALS AND METHODS

A series of stations were sampled extending downstream from the discharge point in each of three streams receiving alum sludge, the Poquoson River, the Warwick River, and Broad Creek in tidewater Virginia (Figure 1). Stations extended three to five miles downstream in each of the streams to a point where the stream either widened and deepened dramatically or until it discharged into a larger stream. Station density was greater at upstream locations where changes in impacts were expected over short distances.

In March 1984, long (ca 1 m) sediment cores, 10 cm in diameter, were collected primarily to determine the distribution of alum. Several other geochemical and geological parameters were evaluated to define the general nature of the substrate; these have been discussed elsewhere (Diaz and Roberts 1985; Diaz et al. 1985; Roberts and Diaz, 1985). Enrichment of sediments with aluminum was determined as the Al:Si ratio. Samples from selected depths in each sediment core were analyzed for silicon and aluminum with a PGT Model 342 X-ray Dispersive Spectroscope coupled with a computer. The ratio of the elements was selected as the data output from the analytical system.

Primary production by the surface phytoplankton community was determined at two stations each in the Poquoson and Warwick River systems, one near the discharge point, the second at a downstream site . Production was estimated by a modification of the ^{14}C-uptake method (Strickland, 1960; Diaz et al., 1985). At the same time, primary production of the benthic algal community was determined using a light and dark chamber method (Rizzo, 1977). Phytoplankton samples were incubated in situ at the collection site at about 15 cm depth. Immediately after starting the phytoplankton incubation, the benthic algal incubation was initiated. After establishing the upstream station, the downstream station was sampled before returning to the upstream station to terminate both incubations. Productivity was measured in winter (March) and again in late spring (early June). The early sampling period approximately corresponded to the "spring bloom" of the estuarine phytoplankton community. The second sampling might better have been delayed until late July or August to observe extreme summer conditions, but this was not possible within the schedule of the contract funding this study.

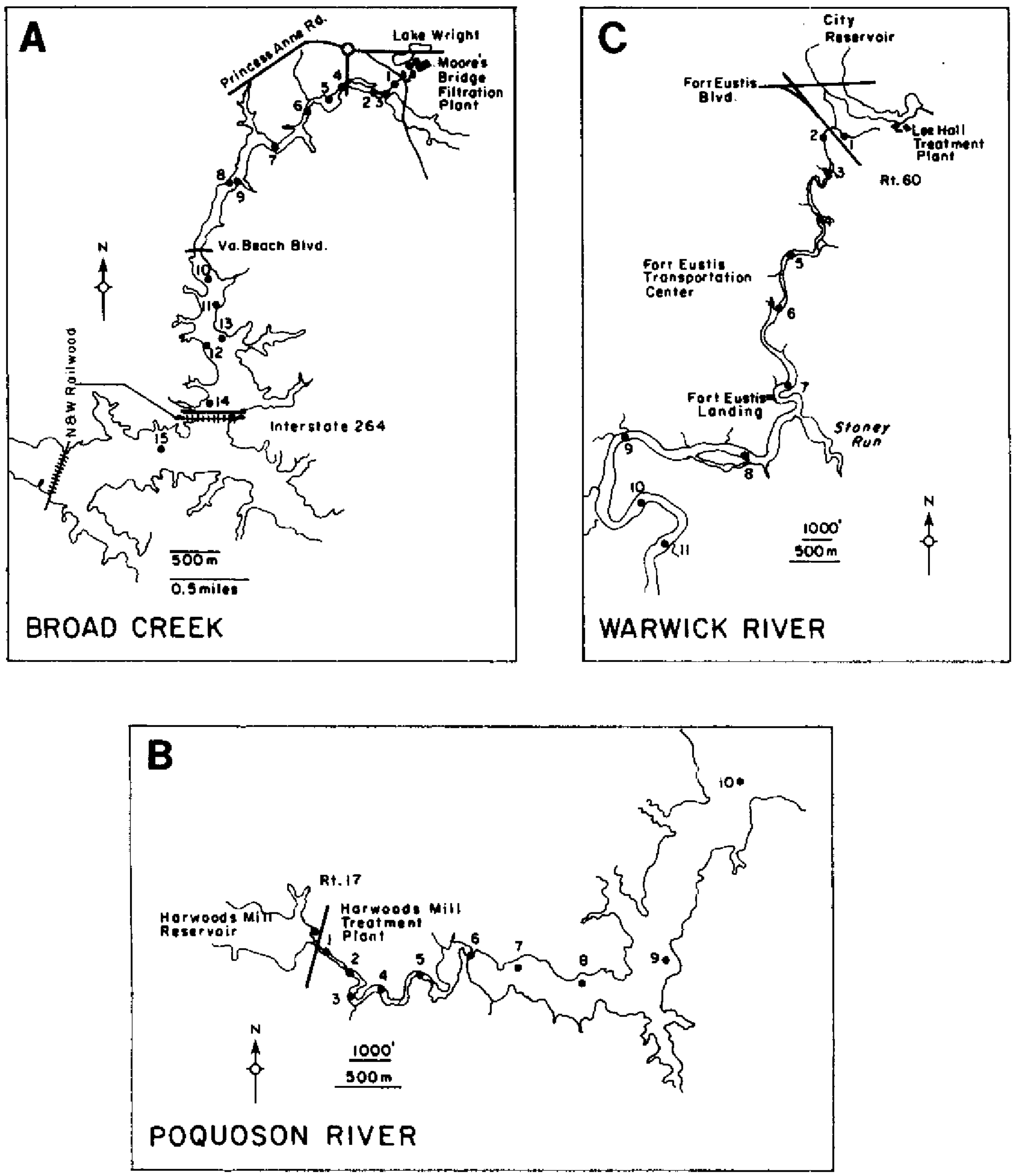


Figure 1 Station maps for the three estuaries studied. A) Broad Creek in Norfolk, VA; B) Poquoson River in York County, VA; C) Upper reach of Warwick River, Newport News, VA.

Chlorophyll concentration in the surface water at each station and the chlorophyll content of the sediment surface were determined fluorometrically after extracting samples in a DMSO:acetone:water:DEA mixture at room temperature in the dark for 24 hours (Burnison, 1971). Temperature, salinity and carbonate alkalinity were measured at each station during the productivity studies (Strickland and Parsons, 1972).

The marsh community fringing all three streams was examined visually. The species composition in each area was determined along with estimates of the areal coverage by definable community types. These observations were compared to historical data for each stream found in Moore (1977) and Silberhorn (1974).

The benthic communities in the three streams were sampled in late May/early June. Five to ten cores (5 cm diameter) were preserved and returned to the laboratory for analysis. Core samples were sieved to 250 um; the invertebrates were then hand picked and identified to the lowest taxon possible; most marine and freshwater macroinvertebrates were identified to species. Each species was enumerated individually in each core.

RESULTS AND DISCUSSION

Aluminum is a natural constituent of clay minerals, and therefore, one would expect some specific background concentration based on the type of clay mineral present. To define a background Al:Si ratio for the study sites in Virginia, the ratio for samples from an area in the York River not receiving alum sludge as well as pure samples of kaolinite and montmorillonite were analyzed. A ratio of 0.2 was accepted as representative of the probable background.

Aluminum was clearly enriched in surface sediments near the point of sludge discharge in all three streams (Figure 2). The proportion of aluminum in sediment samples generally declined with distance downstream and with depth in the sediment. Background Al:Si ratios were found in samples from a 1 m depth in the sediment cores from every station, with background levels generally being found at shallower depths as one proceeds downstream. This is the general type of distribution which one would expect with frequent, though discontinuous introduction. The Warwick River exhibited greater enrichment than either of the other streams, reflecting a greater input into the Warwick than other streams.

Phytoplankton productivity was measured before and during an alum sludge discharge event in the Poquoson River. In the Warwick River, a single measurement was made at each station since there was high turbidity associated with alum floc whenever the sites were visited regardless of the timing of discharge.

In the Poquoson River, phytoplankton productivity at Station 1 was extremely low (<2 mg $C/m^3/h$) on the day prior to discharge of the alum sludge both during March and June sampling. At both sampling times, the low productivity corresponded to a low standing crop of algae as indicated by chlorophyll a concentrations at or below 1.1 mg/m^3. On the day of sludge discharge in both March and June, productivity measured at

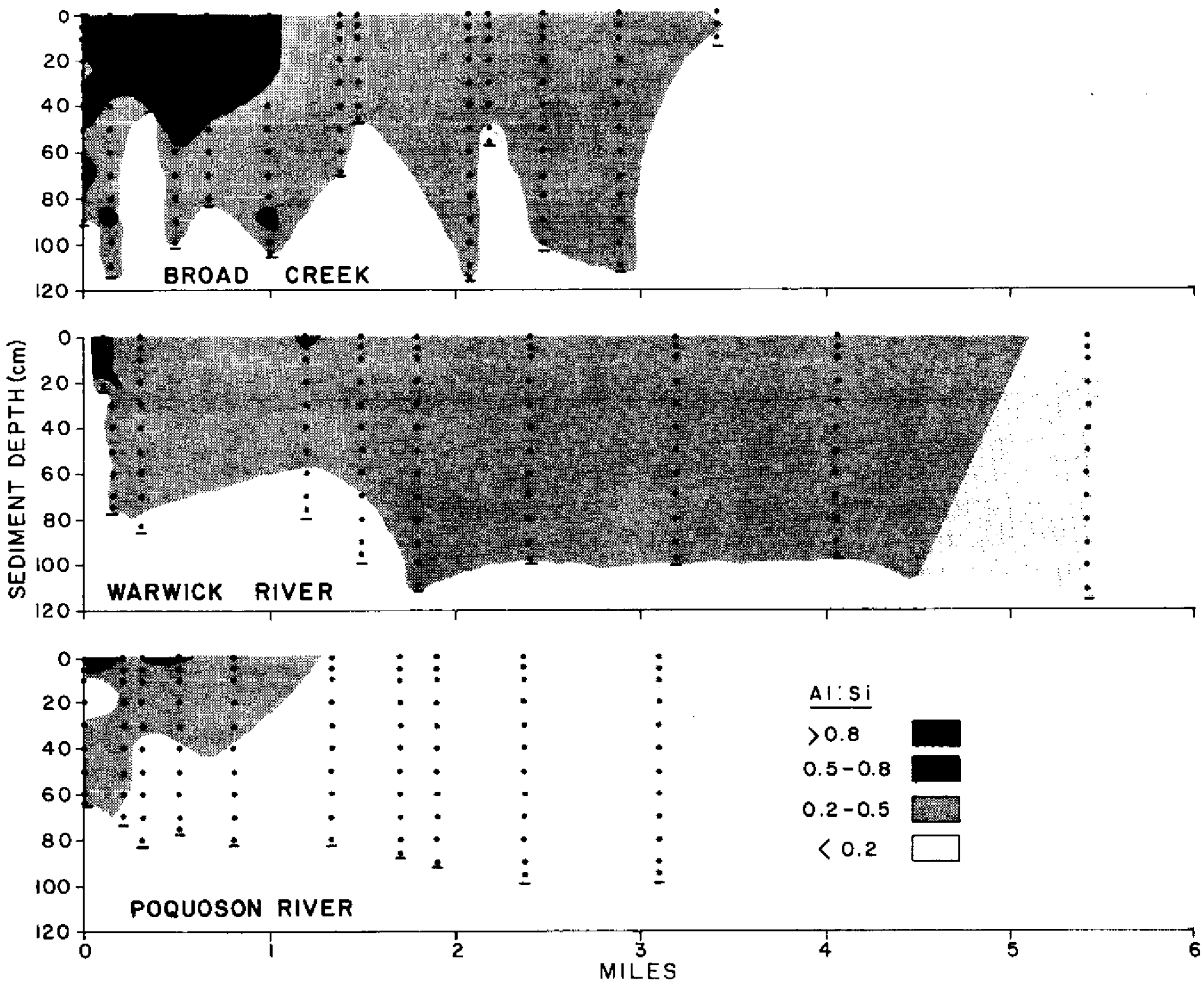


Figure 2 Graphical presentation of the distribution of aluminum enrichment in bottom sediments of the three estuaries studied.

Station 1 was increased 10-fold corresponding to a large increase in standing crop (4-fold in March and 57-fold in June). The dramatic increases in standing crop and corresponding increases in primary production assuredly reflect the presence of algal material in sludge derived originally from the reservoir.

At the upstream Warwick Station 3, primary production in March was low corresponding to a low standing crop. In June, primary production was essentially unchanged from that in March despite the nearly 10-fold increase in apparent standing crop. At the downstream station 6 in both rivers, production prior to the discharge was several times higher than at the upstream stations during both March and June. Standing crop of phytoplankton at the downstream Poquoson River station was not different before and after discharge.

Few productivity studies have been conducted specifically in habitats like the Poquoson and Warwick Rivers. Stross and Stottlemyer (1965) reported levels of phytoplankton primary production in the Patuxent River, MD. In March, phytoplankton productivity was similar at Patuxent Station 22 to that at Poquoson Station 1 and Warwick Station 3. In contrast, production at Warwick Station 6 was markedly higher than other freshwater stations. At mesohaline stations (Poquoson Station 6 and Patuxent Station 6), production was equal for the two rivers in March. In June, productivity at the two stations nearest the discharge, Poquoson Station 1 and Warwick Station 3, was low compared to Warwick Station 6 and Patuxent Station 22, where production was an order of magnitude higher. At the mesohaline stations, production was similar in the Poquoson and Patuxent Rivers.

Standing crop, expressed as chlorophyll _a_ concentration, was low at Poquoson Station 1 during both March and June. The freshwater stations in the Warwick and Patuxent Rivers were quite similar in standing crop. There was a dramatic increase in standing crop at Warwick Station 3 in June compared to March, but at both times, the standing crop was within the range observed in the Patuxent River. Standing crop at mesohaline stations in the Patuxent and Poquoson Rivers was similar in March and June.

The assimilation ratio (i.e. the ratio of production to standing crop expressed as chlorophyll _a_ concentration) at Poquoson Station 1 in March was low on the day preceeding a sludge discharge but within the expected range for this salinity regime (Flemer, 1970). In June, the assimilation ratio was strongly depressed at Poquoson Station 1 before the alum sludge discharge, and increased only slightly on the day of the sludge discharge despite the large increase in chlorophyll a concentration. The assimilation ratio for communities at Warwick Station 3 was within a normal range though low in March, but severely depressed in June.

The high sediment loads in the upper reaches of the Poquoson and the Warwick produce a severe, albeit not demonstrably toxic, effect on primary production. At best, primary production is restricted to the upper 5 to 10 centimeters of the water column. This condition gradually improves as one progresses downstream.

Gross production by benthic algae at the Poquoson River stations was extremely low in March, reflecting low light availability on both sampling days which were heavily overcast with occasional light to heavy rain. Despite the poor conditions, it is clear that gross production was increased at both stations on the day of sludge discharge; this may be an artifact of increased light availability. Similarly, gross production by the benthic algal community at both Warwick Stations 3 and 6 was low and net production was essentially zero, despite adequate light intensity. In June, gross production, respiration, and net production at Poquoson Stations 1 and 6 were nearly the same on the day preceeding the sludge discharge. No comparison is possible for the day of discharge because some material in the effluent (aluminum hydroxide?) interfered with the Winkler oxygen method. Thus there is no clear evidence of any impact of the alum sludge discharge on the benthic algal community as measured by primary production rate during June, although there was a difference in March.

Gross and net production of benthic algae in the Poquoson and Warwick rivers were similar at both sampling times. There did not appear to be any impact of the sludge discharge on the productivity of the benthic algal community at the downstream stations. Gross production of benthic microalgae is reported to range between 10 and 190 mg $C/m^2/h$ (Gallagher and Daiber, 1974). Gross production at Poquoson Station 1 and Warwick Station 3 was low, but within the range of values observed in other shallow subtidal mudflats adjacent to marshes. Gross production at both downstream stations was at the high end of previously reported range of production rates.

The concentrations of chlorophyll a (2-27 mg/m^2) in sediments at Poquoson Station 1 and Warwick Stations 3 and 6, all freshwater habitats, are low in comparison to data from a similar habitat in the James River (50 to 60 mg/m^2; Rizzo, personal communication). The observed concentrations fall at the low end of the range of chlorophyll concentrations reported for intertidal mud flats in brackish and saline water. Data for the mesohaline stations were comparable to data reported for similar subtidal habitats (Rizzo, 1977).

The tidal marsh communities observed in all three rivers were typical of streams of similar geomorphometry and salinity regime. None of the marshes had changed since marsh inventories were made a decade ago (Silberhorn, 1974; Moore, 1977, Silberhorn, unpublished survey).

The portions of the Poquoson and Warwick Rivers receiving alum discharges pass through marshes classified as brackish water mixed type 12. In the Warwick close to Route 60, the marsh vegetation grades into swamp/bottomland hardwood forest consisting of black gum, red maple, and sweet gum. These tree-dominated wetlands were judged to be only marginally tidal. Along the upper reach of Broad Creek, there are approximately 70 acres of predominantly type 12 tidal marsh.

In the Poquoson River, 26 taxa were collected at 10 stations. The most abundant taxa were amphipods, followed by polychaetes and oligochaetes. Faunal diversity was high with five amphipod, eight polychaete and two oligochaete species present. The fauna from the Poquoson River was composed almost entirely of marine species; only 5 of 26 taxa were of freshwater origin. The salinities at all stations were high enough to reduce the occurrence of freshwater species.

In the Warwick River, 13 benthic taxa were collected at 11 stations. The most abundant taxa were harpacticoid copepods, oligochaetes, and cladocerans. The species were freshwater forms with limited or no salt tolerance. In the reach of the Warwick River studied, the salinity at all stations was 0 °/oo.

In Broad Creek, 17 benthic taxa were collected at 15 stations. The fauna from Broad Creek was composed of salt-tolerant freshwater species and estuarine endemic species of marine origin. The most abundant taxa were oligochaete worms, followed by polychaete worms. The distribution of the fauna suggests that Stations 1, 2 and 3 are never exposed to salt water or exposed only for short intervals. From Station 4 to 10, an area where salinities are thought to fluctuate widely, the fauna included a mixture of freshwater and marine species, with oligochaetes numerically dominant. Stations 11 to 15 were characterized solely by estuarine species, suggesting that the salinity never drops below 5-8 °/oo in this reach of the creek.

A characteristic of fauna that inhabits transitional and fluctuating low salinity habitats is that they are very eurytopic and tolerant of extreme environmental conditions. These species may individually exhibit extremely high abundances when conditions favor. Freshwater and marine species which are more sensitive to fluctuations in their environment are excluded from these habitats.

The number and relative abundances of taxa were low at upstream stations in all three creeks, and with the exception of the Warwick River, increased as one progressed downstream. Given the nature of the habitats under consideration, the changes in number of taxa and relative abundance could result either from a deleterious effect of alum sludge or from the salinity distribution. In order to assess these alternative possibilities, the number of taxa and total abundance at each station in all three rivers were plotted against the Al:Si ratio at the sediment surface at each station and the salinity at the station (Fig. 3 and 4).

The number of species was low at all stations at which the salinity was below 15 °/oo, regardless of the Al:Si ratio. While no stations with Al:Si ratios greater than 0.6 had more than 3 species, several stations with Al:Si ratios less than 0.6 had 4 or less species present (Fig 3). The number of individuals was also generally low at salinities below 15 °/oo; a few stations in fresh or nearly fresh water had extremely high numbers of individuals, always of a single species (Fig 4). At several of these stations we observed high organic loading which favored a particular species; in some such cases, the fauna was reduced to that single species. This situation never occurred at stations with a high Al:Si ratio, but these stations were generally different in geomorphometry.

While species diversity and infaunal abundances were generally low, they were well within ranges reported for other low salinity fluctuation environments (Diaz, 1977; Tenore, 1972; Jordan, et al., 1976). While the evidence is not conclusive given the paucity of stations at which

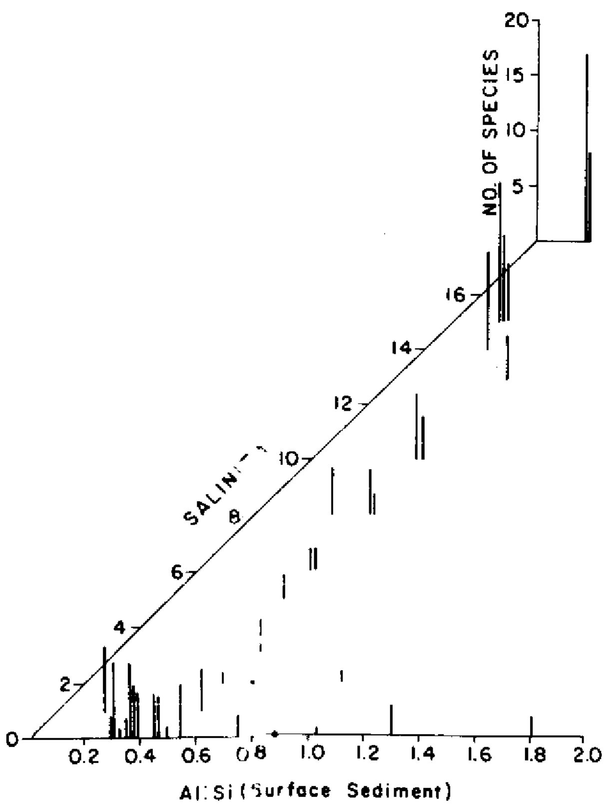
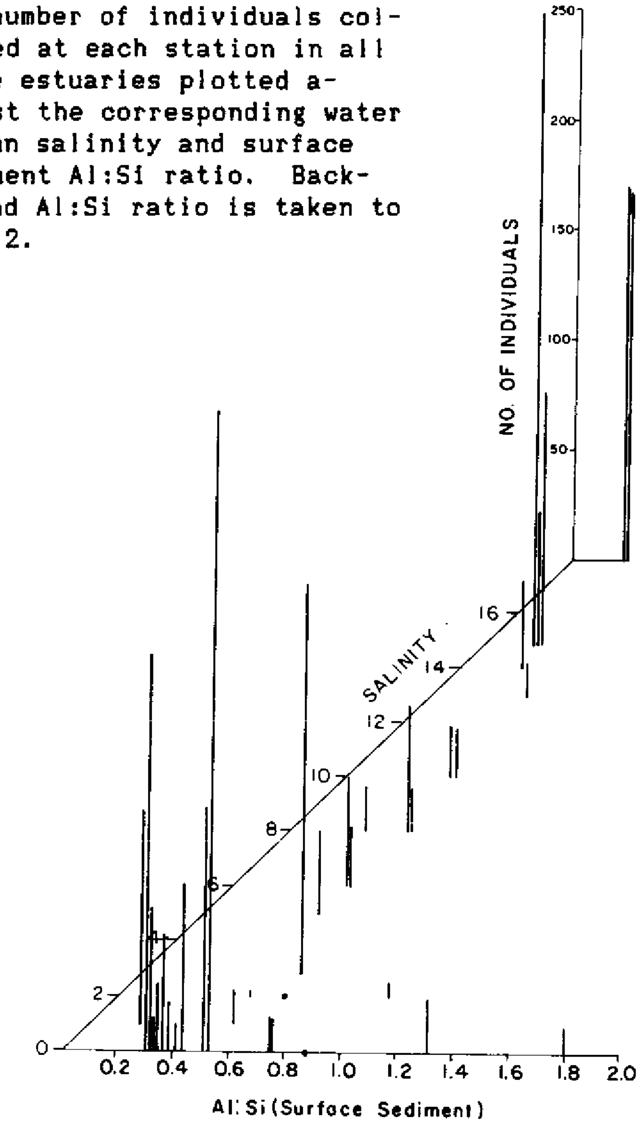


Figure 3 The number of species collected at each station in all three estuaries plotted against the corresponding water column salinity and surface sediment Al:Si ratio. Background Al:Si ratio is taken to be 0.2.

Figure 4 The number of individuals collected at each station in all three estuaries plotted against the corresponding water column salinity and surface sediment Al:Si ratio. Background Al:Si ratio is taken to be 0.2.



the Al:Si ratio at the sediment surface exceeded 0.5, there is little or no basis for alleging a severe negative impact of aluminum enrichment on the benthic invertebrate species diversity or abundance. In naturally physically stressed environments such as freshwater transition zones, detection of effects of additional stresses becomes nearly impossible (Roberts et al. 1975).

There was no evidence of an adverse effect from the alum sludge discharges except with regard to the primary producers. In this case, effects observed are attributed to increased suspended solids loads and light availability, and hence ultimately on the ability of the primary producers to form new biomass, rather than to any toxic effect. The effect on light availability probably exists nearly continuously in both the Warwick River and Broad Creek, and immediately following discharges in the Poquoson River. Marsh grass and benthic invertebrate communities show no evidence of any impact from alum sludge deposited on them.

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ORGANIC SEDIMENT TOXICITY IN THE LOWER CHESAPEAKE BAY

by

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ABSTRACT

A study was conducted to test the suitability of sediment from the Hampton Raods Harbor for open ocean disposal. Lethal and sublethal toxicity tests were used to statistically group geographic patterns associated with sediment concentrations of PNAH's in the port system. Results of 10-day solid phase lethal bioassays and 96 hour suspended solid phase lethal and sublethal experiments conducted with sediments collected from the navigational channels yielded a consensus "map" of the region. The use of three *a priori* geographic groups were shown to have unique PNAH characteristics in both the suspended solid and solid fractions. The "clean" group, suitable for ocean disposal, produced few biological effects. These stations contained sediments from Hampton Roads Harbor and the mainstem of the Elizabeth River, and generally had low levels of PNAH's. The "contaminated" group was associated with significant biological effects from the Southern Branch of the Elizabeth River. This region was represented by high levels of low molecular weight PNAH's. A third group from the upper reach of the Southern Branch produced moderate biological responses. The statistical "fingerprinting" of both biological and PNAH data produced a similar geographic map of sediment quality. Sediment PNAH's from the lower Chesapeake Bay appeared to have been introduced by distant anthropogenic sources while contaminant patterns within the harbor were related to more proximate sources: coal dust from loading piers, runoff from creosote factories, shipyard activities, industrial combustion, and major transportation routes.

INTRODUCTION

Industrialization of our coastal rivers and estuarine has created numerous sources of potentially toxic substances. Most point and non-point source contaminants ultimately reside in the sediments where they accumulate. One group of pollutants that are of particular concern is a class of compounds known as polynuclear aromatic hydrocarbons (PNAH's). They are long-lived toxins, many of which are mutagenic and/or carcinogenic at high levels.

The potential sources of PNAH compounds are numerous: fossil fuel products such as creosote, coal and incomplete combustion fuels (e.g. automobile exhausts, home heating, industrial smoke stacks, incinerators, etc.), among others (EPA, 1979). Recent sediment surveys from the Elizabeth River revealed high concentrations of PNAH's (i.e. high ppm range) in certain sections of the river (Alden and Hall, 1984; Alden et al., 1985). An experimental design was developed to "map" and characterize the distribution of PNAH's in the sediments of the major navigational channels of the river and to compare those to patterns of sediment quality in the Port of Hampton Roads and the lower Chesapeake Bay. The results from this and previous studies conducted by this laboratory should provide managers and regulators with data necessary to make sound policy decisions.

METHODS AND MATERIALS

The Elizabeth River is located at the southern end of the Chesapeake Bay. It joins the James River as part of the Port of Hampton Roads, Virginia, one of the worlds largest natural harbors. The Elizabeth River includes a mainstem extending from Sewell's Point to Town Point, and the Western, Eastern and Southern Branches. Station locations correspond to the river miles of the main navigational channels (Figure 1). Only samples collected from stations along the major navigational channels in Hampton Roads Harbor (D, E, F) and the Elizabeth River (G-S) are considered in this study. Samples from each station were from a transect:

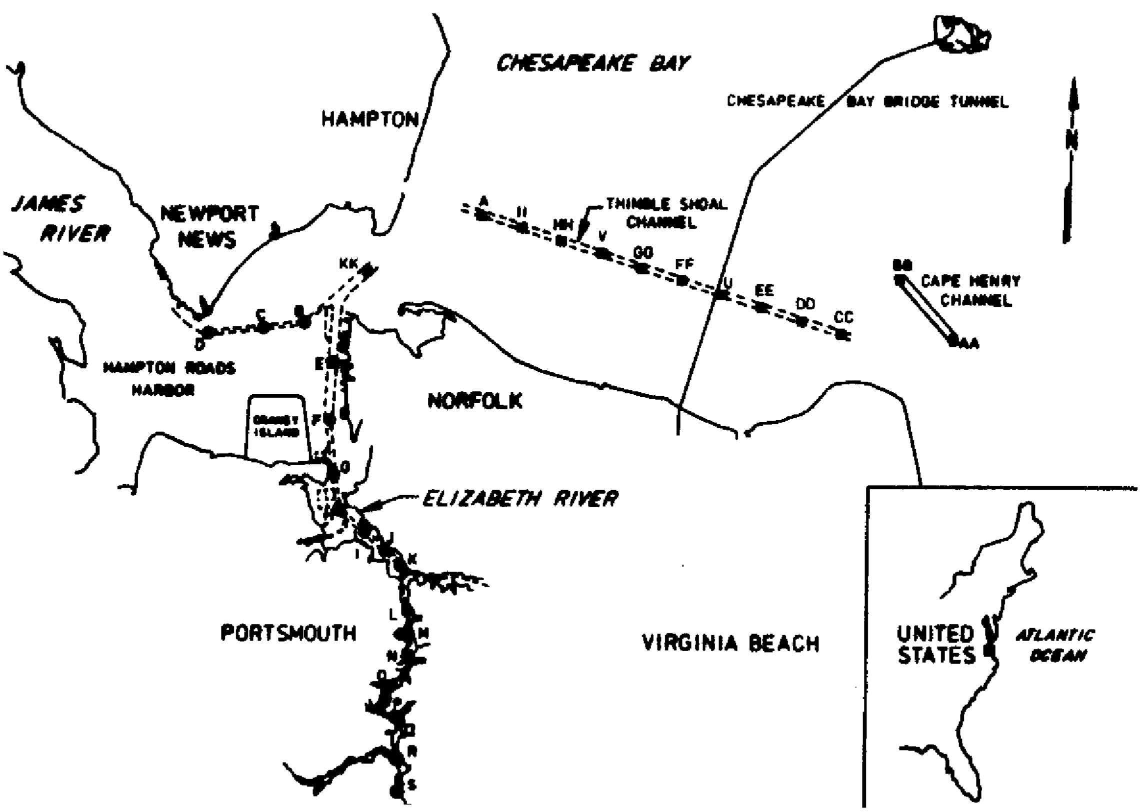


Figure 1. Map of station locations from the Lower Chesapeake Bay.

one mid-channel grab, and collections from depths that were midway up each channel wall (c.a. 6m in-depth). Sediment discriminant scores were plotted as triplets: the mid-channel sample data plotted between the western and eastern side samples, which were shown as top and bottom, respectively.

Lethal and sublethal bioassays employed have previously been described (Alden and Young, 1984; Alden et al., 1984, respectively) and follow the standard guidelines (Implementation Manual, EPA/COE, 1978). The grass shrimp, ***Palaemonetes pugio*** was the test species used in all of the tests. Respiration and osmoregulation studies were performed to evaluate sublethal effects during the 96-hour suspended solid experiments as described by Alden et al. (1984).

A discriminant analysis (Klecka, 1975) was employed to reduce the data into groups to summarize the major patterns. This classification analysis provided an objective means of determining whether the data from any given station was statistically more similar to the "clean" or "contaminated" groups (sediments from Stations N, N/O, and O) (Alden and Hall, 1984).

The term "total PNAH's" for this study represent the total of the 16 PNAH's listed by EPA as "priority pollutants." Carcinogenic PNAH's consisted of some of the stronger cancer-causing agents: Benzo(a)pyrene, Benzofluoranthenes and Indenopyrene. The pyrosynthetic PNAH's consisted of Phenanthrene, Fluoranthene, Pyrene, Benzo(a)anthracene, Chrysene, Benzofluoranthenes (B(b)Fl, B(k)Fl), Indenopyrene and Benzo(ghi)Perylene.

RESULTS

PNAH sediment analysis conducted in 1982 resulted in moderate to high concentrations at Stations H to R (Table 1). Total PNAH concentrations were greatest from Station M ($\bar{x}$ = 66$\pm$44 mg/kg) and decreased gradually upstream (Figure 2a). Values downstream were generally lower except at Station J which displayed the second highest sediment level. The carcinogenic and pyrosynthetic compounds exhibited similar trends, with highest concentrations at Stations J and N (Figure 2b - $\bar{x}$ = 21$\pm$10 mg/kg and 18$\pm$16 mg/kg; Figure 2c - $\bar{x}$ = 23$\pm$11 mg/kg and 20$\pm$16 mg/kg, respectively).

Maximum mortalities for solid phase experiments were detected for sediment experiments from Station K through O, with low to moderate values found throughout the remaining test sites (Figure 3). Mortalities represented by suspended solid bioassays were conducted concurrently with sublethal experiments in 1982 (Figure 4a). The histograms represent mean values (vertical bars are two standard errors; n-5). Shading indicates mortalities significantly greater (α=0.05) than controls of the experimental set. Sublethal data for respiration, hyporegulation and hyperregulation capacity from suspended solid experiments are arranged geographically (Figures 4b, c, & d, respectively). Sediment elutriates from Stations JK, N, N/O, O, Q & R resulted in significantly depressed respiration rates and reduced hyporegulation

Table 1. Mean PNAH values (standard errors in parentheses; n=3 transetc samples) for sediments from 1982 survey. Values are in μg/kg dry wt.

PNAH (Abbreviation)	Station (River Mile) D (NM)	E (4)	F (6)	G (8)	H (9)	I (10)	J (11)	K (12)	L (13)	M (14)	N (15)	O (16)	P (17)	Q (18)	R (19)	S (20)
Napthalene (N)	0	0	0	0	821 (473)	0	1,564 (1,564)	0	309 (309)	588 (294)	466 (308)	953 (953)	417 (212)	0	0	0
Acenaphthylene (Acy)	0	0	0	0	0	0	0	0	230 (230)	2,700 (2,700)	0	0	0	0	0	0
Acenaphthene (Acn)	0	0	0	0	2,509 (2,074)	0	425 (425)	0	438 (438)	591 (297)	255 (255)	1,186 (896)	115 (115)	0	0	0
Fluorene (F)	0	0	0	0	220 (220)	0	596 (596)	0	326 (42)	24,530 (24,205)	234 (234)	866 (866)	115 (115)	465 (465)	0	0
Phenanthrene (Ph)	0	0	32 (32)	0	798 (644)	73 (73)	1,358 (1,168)	0	674 (16)	688 (355)	731 (270)	5,001 (2,145)	527 (77)	0	0	0
Anthracene (A)	0	0	0	0	341 (183)	0	3,413 (3,413)	0	244 (244)	27,200 (26,284)	680 (526)	2,171 (410)	307 (207)	0	171 (171)	189 (189)
Pyrene (P)	0	0	0	495 (87)	2,577 (1,497)	340 (34)	5,179 (3,639)	660 (274)	1,330 (455)	1,075 (565)	2,098 (402)	1,470 (1,470)	1,972 (220)	846 (142)	733 (80)	714 (640)
Fluoranthene (Fl)	0	2,021 (2,021)	65 (65)	0	671 (525)	81 (81)	2,156 (1,218)	804 (654)	1,286 (52)	1,267 (687)	1,809 (331)	1,884 (1,984)	1,747 (576)	345 (345)	753 (227)	1,061 (729)
BenzoAnthracene (BaA)	0	0	0	0	423 (423)	0	1,991 (1,799)	1,406 (1,406)	737 (136)	1,553 (937)	735 (382)	620 (620)	1,292 (237)	505 (257)	283 (143)	1,313 (724)
Chrysene (Ch)	0	789 (202)	81 (81)	0	1,362 (798)	409 (210)	13,900 (8,617)	413 (291)	1,729 (31)	1,515 (1,515)	3,160 (1,256)	1,120 (1,120)	2,090 (1,160)	1,731 (551)	1,745 (271)	1,574 (1,574)
Benzo(ghi) Perylene (BPR)	0	0	0	0	0	0	1,079 (1,079)	0	0	0	0	0	548 (548)	0	0	0
Benzo(a)Pyrene (BaP)	0	277 (277)	0	0	1,312 (305)	366 (366)	16,486 (9,898)	628 (628)	731 (731)	2,291 (1,396)	362 (362)	872 (872)	3,324 (1,887)	2,783 (933)	2,674 (1,337)	1,652 (1,652)
Benzo(b)Fluoranthene (BbFl)	0	302 (302)	0	0	0	0	2,974 (2,974)	1,962 (1,962)	1,220 (446)	1,274 (655)	11,340 (10,860) (16,455)	1,010 (512)	2,570 (847)	2,075 (1,206)	1,839 (327)	121 (121)
Benzo(k)Fluoranthene (BkFl)	0	0	0	0	0	0	1,430 (1,430)	0	352 (352)	252 (252)	7,698 (7,698)	570 (570)	962 (580)	263 (263)	211 (211)	0
Indenopyrene (IP)	0	0	0	0	0	0	372 (372)	0	0	0	0	0	0	0	0	0

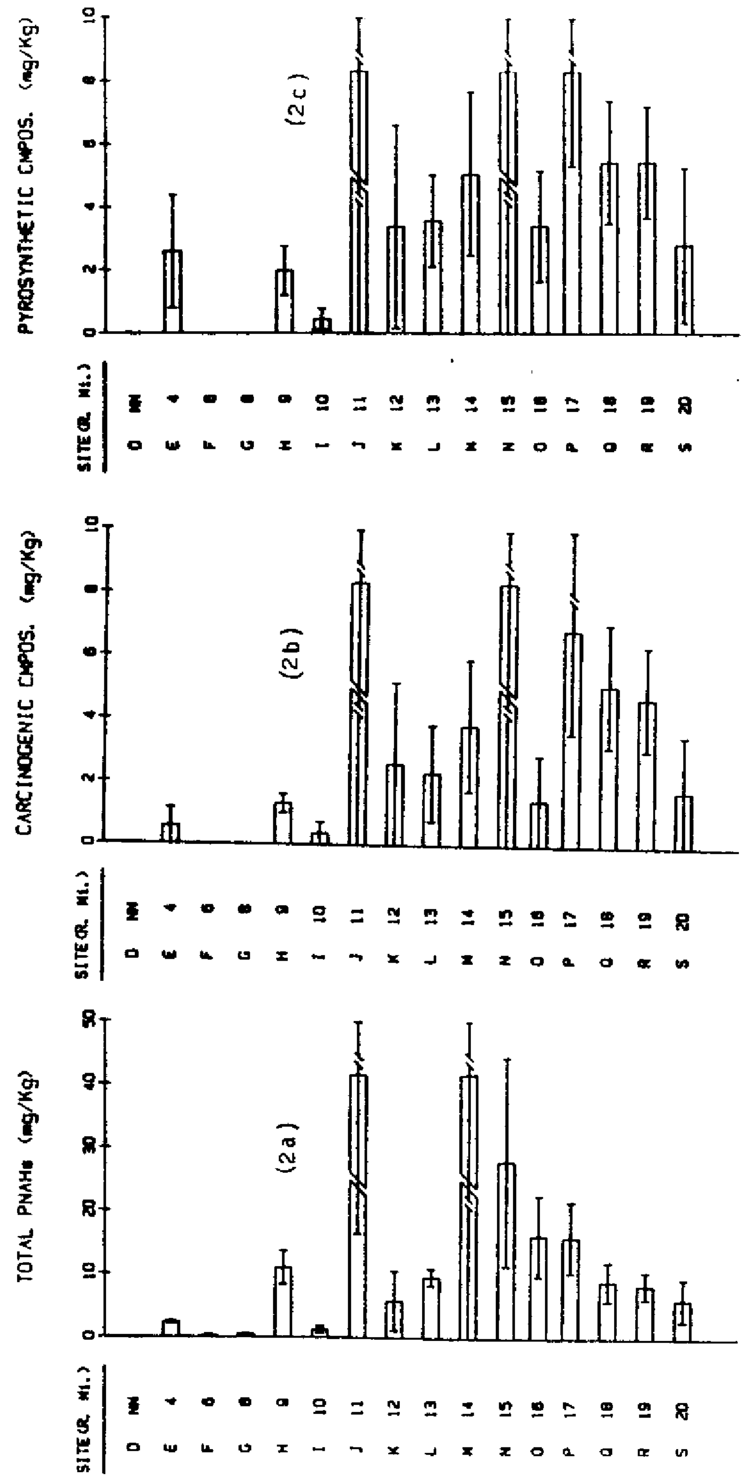
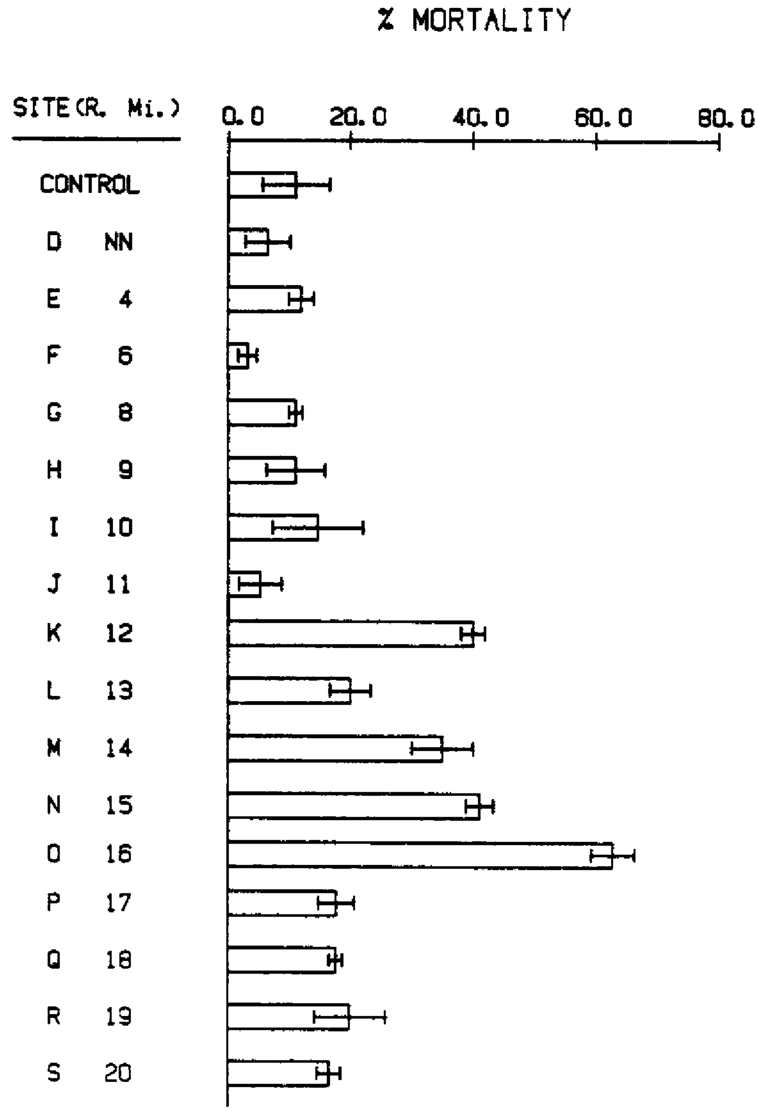


Figure 2. Composite PNAH variables for sediments (means and standard errors arranged geographically; n=3 samples per transect).

Figure 3. Mortality data from 10-day solid phase bioassays on ***Palaemonetes pugio***. Histograms represent mean values (vertical bars are two standard errors; n=5).



capacity. Hyperregulation was apparently unaffected by the experimental conditions tested.

Considering these patterns, three contiguous station groups of roughly equal size were selected for the a. priori groupings in the discriminant models for the PNAH analyses. They were: clean, nontoxic sediments -- Stations D, E, & F; contaminated sediments producing significant biological effects -- Stations N, N/O, & O; and intermediate conditions -- Stations Q, R, & S.

DISCUSSION

Potential contaminants from upland (terrestial) sources ultimately reside in the sediments of the surrounding estuaries and river bottoms.

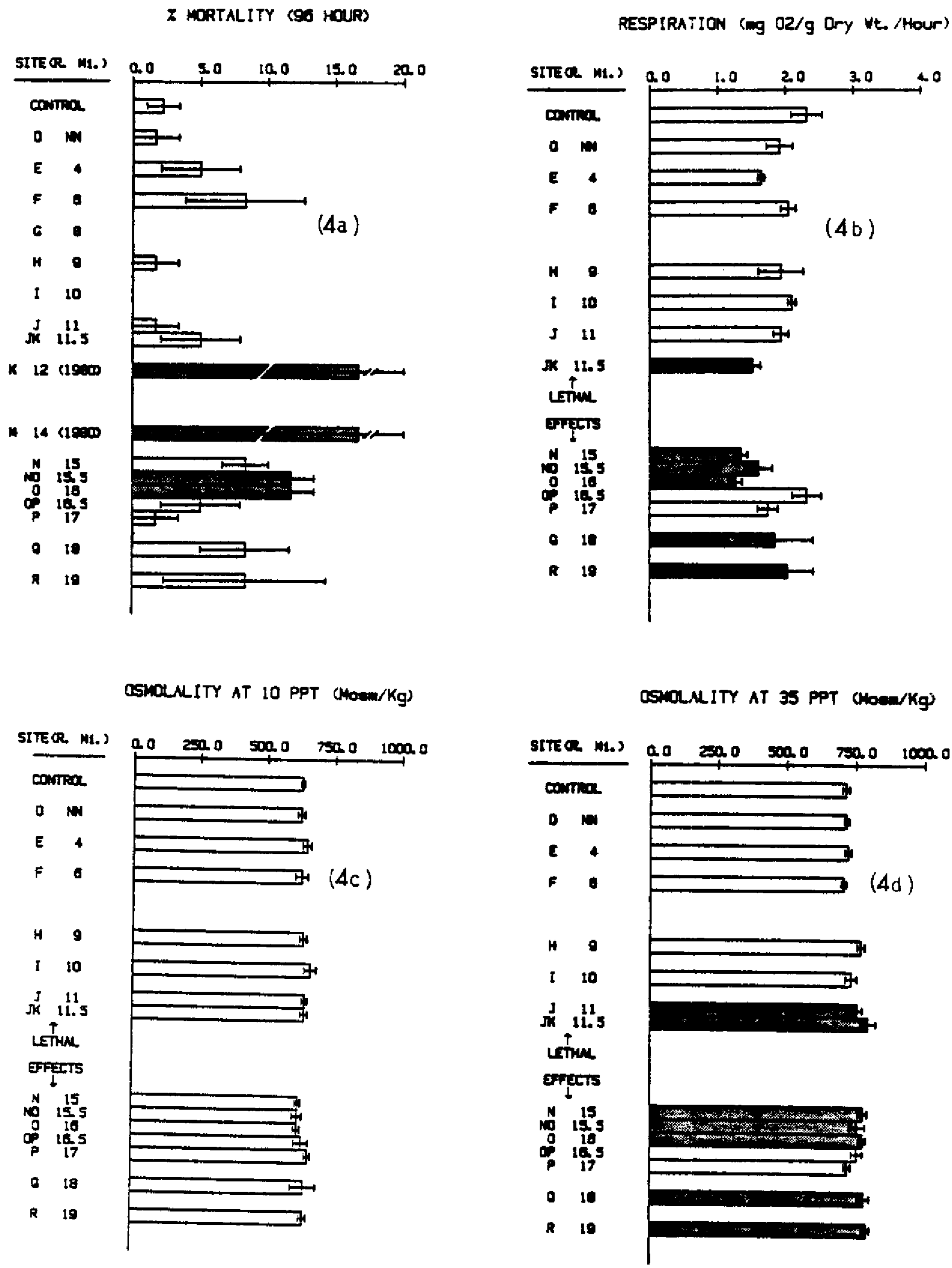


Figure 4. Biological data from suspended solid experiments on **Palaemonetes pugio**.

Quantitative and qualitative determinations of the PNAH's in sediments often reflect the characteristic nature of their point or non-point origins. The PNAH concentrations in this study varied substantially throughout the Port indicating multiple origins.

The maximum levels of PNAH's were associated with sections of the Southern Branch of the Elizabeth River where shipbuilding and repair operations were concentrated. In fact, the mean total PNAH for the most contaminated station (>60 ug/g dry wt) was several orders of magnitude higher than previously reported values (see LaFlamme and Hites, 1978; Pancirov et al., 1980; Murry et al., 1981; Poutanen et al., 1981; Readman et al., 1983; among others), and was only exceeded by three other studies on a world-wide comparison (Thompson and Eglinton, 1982). Lower molecular weight (2- and 3-ring) compounds associated with oils, fuels, paints and solvents appeared to be the potential sources of the observed values. Two creosote plants were located in the vicinity of Stations N & O. Qualitative estimates based on GC/MS analyses suggested that the PNAH's which were quantified may represent only 20% of the total amount of all aromatic compounds observed in this region.

Higher molecular weight PNAH's such as B(a)P and BFl were found upstream around Station P. Not only did the composition of PNAH's began to change, but levels declined. Sources of high temperature combustion from a local power plant and runoff from two major highways (Military Highway and I-64) appear to contribute the same combination of PNAH's in this region as reported elsewhere (Lake et al., 1979; John et al., 1979; Wakeham et al., 1980).

All PNAH's were relatively low in the Hampton Roads Harbor and most of the mainstem stations of the Elizabeth River. They were sporadically detected and were identified as the principal combustion product parent compounds (Fl, Pyre, BFl, B(a)P). The only clear exception was the sample taken in the vicinity of the coal piers (Station H) that compared to a similar pattern reported by John et al. (1979) for coal dust contamination.

Strong geographic patterns of biological effects were noted in the toxicity data. They exhibited the same pattern as the sediment determinations, with elevated mortalities and significant sublethal effects on grass shrimp exposed to the most industrialized regions of the river. Stations in Hampton Roads Harbor and the mainstem of the Elizabeth River produced little or no biological effects.

A "cause and effect" relationship cannot be made from the significant correlations found in this study. However, it is reasonable to assume that if the chemical patterns parallel the toxicity groupings, then some qualitative associations may be inferred between the degree of contamination and the biological responses. Geographic groupings with respect to the biological data proved to be unique in the chemical characteristics as well. A schematic map of the study area indicates the consensus of the biological and chemical models developed for this region of the Bay (Figure 5). The regions without shading are considered acceptable for ocean disposal. The heavily shaded region represents the zone where sediments should not be considered for ocean disposal. The cross-hatch regions represent transitional (single) and "intermediate" (double) groupings. The transition zone reflects the region where biological effects and/or PNAH concentrations are sporadically found. The "intermediate" zone requires reevaluation before sediments be approved for disposal activities.

Initial results indicate that most of the heavier compounds with low solubilities were found at concentrations greatly exceeding saturation level (EPA, 1980; May, 1980) and must be sediment-bound.

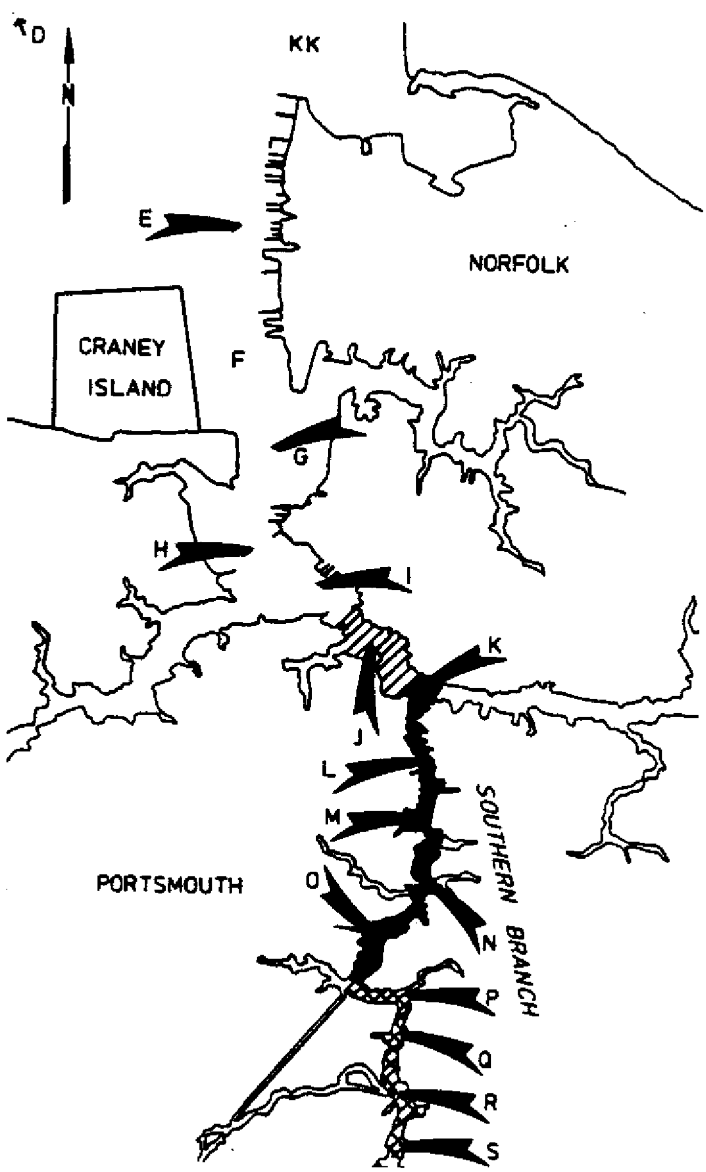


Figure 5. Consensus map of study area indicating overall geographic patterns of sediment quality.

This should limit their transport within the system and remain close to their source input. The low molecular weight PNAH's are the most soluble in water and pose the greatest toxicity to aquatic organisms. However, their distributional pattern remain within the more industrialized portions of the Southern Branch of the Elizabeth River. The other anthropogenic source of detectable PNAH contamination is the airborne transport of compounds associated with the incomplete combustion of fuels. These compounds are nearly cosmopolitan in distribution and display a low degree of toxicity, so are of less ecological concern to the disposal issue.

A major point of interest in the Port is the effect of maintenance dredging on the absolute PNAH concentrations found in the sediments. Results of the lethal bioassays suggest that maintenance dredging conducted in 1981 in the region between Stations N & P drastically reduced the toxicity of the sediments (Alden and Young, 1984). Subsequent PNAH determinations show that contamination in this region has been reduced. However, there is an apparent "re-invasion" of these pollutants since the 1982 sediment survey. This may be due, in part, to slumping of the banks following dredging operations and/or continued point and non-point source contamination in the region. Regardless of the particular source, the contaminants are present in high concentrations and remain a potential health hazard to both humans and aquatic organisms.

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Key Words:

Bioassays, Dredging, Organics, PNAH, Sediments

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TRACE ELEMENT CONTAMINATION FROM FLY ASH SITES NEAR CHISMAN CREEK VA.

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ABSTRACT

The safe disposal of fly ash from power plants remains a concern because of dwindling available disposal sites and potential or actual environmental consequences. During the period from 1957 to 1974, the Virginia Electric and Power Co. station at Yorktown used a mixture of coal and refinery coke for power generation. The fly ash and bottom ash byproducts were disposed of in borrow pits which drain into Chisman Creek, a small estuary near the York River. In 1980 a domestic well near the pits turned green and tests revealed high concentrations of V and Se in some wells. Subsequently, contaminated wells were capped and homes were connected to the municipal water supply.

In late 1981 with a small grant from the Virginia Environmental Endowment, we began a more detailed investigation of possible contamination from the pits. The sampling program included groundwater, surface water, estuarine water, flyash, soils near the pits, oysters and a variety of plants in the immediate vicinity. Most importantly, these samples were analyzed by PIXE (Proton-Induced X-Ray Emission), a sensitive and accurate multielemental technique which can simultaneously detect all elements from silicon to uranium without prior knowledge of the elements present.

Cores from monitoring wells drilled into one of the pits were analyzed to determine the spatial elemental composition within the fly ash and fly ash leachate. Analysis of shallow and deep well waters gave evidence for both vertical and lateral migration of leachate from the pit, although soil interactions apparently restrict groundwater concentrations outside the pits. Comparison of PIXE analyses of leaf tissues from woody upland, woody wetland, and and wetland monocot species with controls demonstrated accumulation of several elements, especially selenium and nickel. Accumulation of several elements in wetland plants, as well as elevated Ni, V, and As concentrations in surface sediments for the upper mile of the extuary together indicate that some mobilization of trace elements into Chisman Creek is still occuring. Nickel and vanadium are unusually abundant in this fly ash due to the use of refinery coke in the fuel. Quantitative estimates of contamination would require a larger sampling program and greater resources than we had available for this work. In this regard analysis of oyster samples was inconclusive due to the limited number of available samples and tidal fluctuations in the estuary.

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EXPERIMENTAL SECTION

Sample Collection and Preparation. Each fly ash, soil and estuarine sediment sample was kept in a plastic bag until dried in a specially prepared, trace element free convection oven at 60 ^{O}C for at least 12 hours. All samples were dry sieved through a 3 mm polypropylene screen to remove large pieces of organic matter or pebbles and homogenized before selecting an aliquot for sample digestion.

Separate portions of the dried, homogenized samples were subjected to room temperature leaching with 5N nitric acid (in the ratio of 10 ml to 5g dry weight) for 2 hours. Soil or fly-ash was separated from the 5N nitric acid leachate and subsequent 2% nitric acid rinses by centrifugation. The leachate and rinses were combined, doped with indium as an internal standard, and spotted on targets for PIXE analysis. The HNO_3 mild chemical leaching procedure was tested previously for marine sediments collected on the mid-Atlantic outer continental shelf (Harris, et al., 1977).

Sediment bottom grabs were collected along the entire Chisman Creek estuary and in Goose Creek. Sediment cores were taken in the upstream reaches of Chisman Creek. A similar location in Back Creek was selected to serve as a control site. Estuarine sediments were wet sieved with the aid of deionized water and dried at 60 ^{O}C for approximately 48 hours. 5N HNO_3 leachates were prepared as described above.

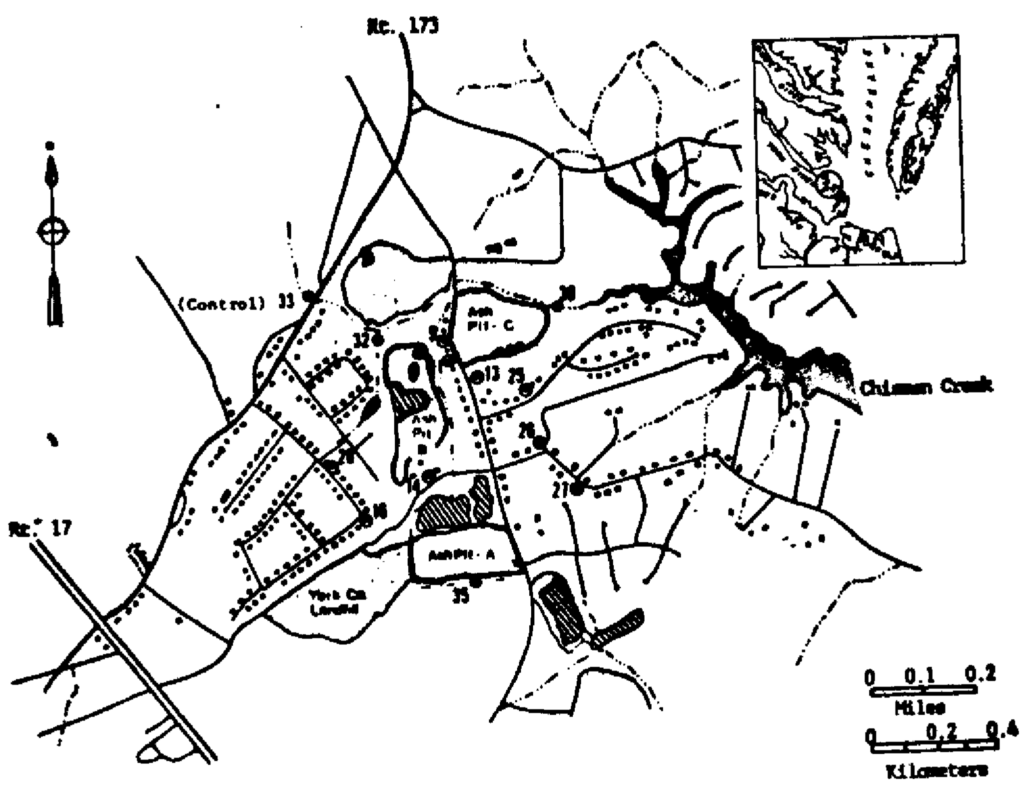

Figure 1. Fly ash disposal site near Chisman Creek in York County, VA

Water Samples

All water samples were collected in 1 liter polypropylene narrow-mouthed bottles, placed on ice and delivered to the laboratory within 4 hours of collection. The rapid and complete separation of particulates and dissolved fraction is essential to prevent any equilibration between the solution particles and container walls prior to analysis. pH readings were taken in the laboratory both before and after filtration through 0.4 nuclepore filters. The filtrate (dissolved fraction) was preserved using 0.05M HNO_3 until analysis (as recommended by the U.S. EPA (1979). The filtrate was doped with indium as an internal standard and subjected to atomic absorption, PIXE or fluoride analysis. For some samples with relatively low dissolved solids the filtered water samples were preconcentrated to improve detection limits using an all plastic closed system (Grant, 1983).

A total of 66 plant samples, consisting of 24 separate species were collected on February 12, May 5, July 21 and October 15 1982. With the exception of perennial species collected on February 12 (grasses, rushes, goldenrod, etc.) all samples were living viable tissue. Upland species were collected near ash pits A, B, and C (figure 1) while wetland species were collected to the North and East of pit C.

Available species at sites located within the pit itself were sampled where possible in proximity to sampling piezometers where ground water was monitored. Control samples of the same species were collected where feasible at several sites upstream of the pit and in local marsh systems. Woody species were sampled by the removal of above ground growth (healthy branches and leaves) while the smaller herbaceous species were taken whole wherever possible (roots, rhizomes, stems). The samples were placed in plastic storage bags and transported immediately to the laboratory.

Upon receipt at the laboratory, plant parts were dissected immediately to remove dead tissue and to isolate subsamples from some species for separate analyses. These subsamples were then placed in pre-cleaned polyethylene bags in preparation for washing.

The wash procedure employed was designed to effectively remove site contaminants and dust, while still retaining those endogenous trace elements within plant tissues. Plants were agitated at least five times with several portions of deionized H_2O and rinsed until visible evidence of particles was absent from the rinse water. After washing all samples were rinsed (5X) with deionized water and dried in a trace metal free oven at 60 ^{0}C; for an average of 8 days to constant weight, as recommended by NBS for SRM 1575 (Pine Needles). After drying, each sample was then ground to §100 mesh using a SPEX 8000 grinder mill with acrylic beads or a SPEX 8500 shatterbox, using an alumina ceramic puck and dish.

Analytical Procedures.

Prepared samples were analyzed for trace element content using either Proton Induced X-ray Emission (PIXE), Atomic Absorption Spectrophotometry (AA), or both. PIXE is particularly useful for large scale environmental studies because of its rapid analysis of a large number of elements simultaneously from one sample (Harris, et al., 1977). AA requires more analytical time, but provides a lower detection limit for some elements. The PIXE data were analyzed for elemental composition using the comprehensive computer program described by Buckle, et al. (1976).

Powdered targets for PIXE analysis were prepared using standard laboratory procedures. The ground material was pushed on to a pre-tared polycarbonate film through a 100 mesh nylon screen. The powder was weighed and encapsulated on the blank with polystyrene film solution. The result is a uniform distribution of the material over a measurable area. External standards used in the analysis included NBS 1575 Pine Needles and NBS 1571 Orchard Leaves. Powdered targets of these materials were prepared using the identical procedure. Quality control procedures in routine use were as follows:

(1) the use of highest purity reagents
(2) in-house generation of high purity acids and water equivalent to NBS specifications
(3) routine analysis of reagents used
(4) rigorous cleaning of all labware before use according to established protocols
(5) inclusion of procedural blanks with samples
(6) analyses of standard reference materials of known composition similar to the samples analyzed
(7) analyses of master mixes prepared from highest purity elements or compounds.

The pH of all water samples was measured in the laboratory within 2 hours of receipt using an Orion Model 801 pH meter with a Ross combination pH electrode (Model 815500). Fluoride measurements were made using an Orion Model 94-09 solid state fluoride electrode accoding to EPA method 340.2.

RESULTS AND DISCUSSION

In this study a variety of sample types was collected from different environments. The results of the analyses will be presented in a sequence more or less analagous to the presumed transfer of trace elements from the fly-ash through the environment: the composition of fly-ash and groundwater within the ash deposits, transport processes to the estuary, trace element accumulation in plants proximal to the disposal site, and possible contamination in estuarine sediments and oysters.

Fly ash and groundwater within the fly ash pits.

A relatively intensive sampling of one fly-ash deposit, Pit C, was conducted because this pit is very close to Chisman Creek and near the two homes where "green water" occurred. When wells were drilled to determine the stratigraphy, (Oakes, 1982) ash samples were collected from several depths. Soil samples from the Tabb formation and the underlying Yorktown formation were also collected.

It is important to remember that all of these trace elemental analyses were conducted on fly-ash which has weathered in the natural environment for a period of 10 to 20 years. Accordingly, a portion of the soluble elements probably has already leached from the fly-ash and some unknown amount of equilibration has already occurred between leached trace elements in the groundwater and soil particles in the surrounding area.

It is unlikely that more than a small fraction of the "total" trace elements in fly-ash would become available to the environment through chemical or biological action. Accordingly, a few chemical procedures for leaching trace elements from fly-ash were compared. Concentrations from the leaches generally decrease in the following order: total metals (aqua regia digest slurry), aqua regia metals (without undigested particles), 5N HNO_3 digest (aqueous mineral acid digest). It should be noted that the pH of groundwaters within the pit was as low as 3.5. Thus the concentrations of those elements which are soluble at that pH, nickel for example, are a significant fraction of the total, whereas the solubility of other elements, such as titanium, which form hydroxides or hydrous oxides above pH 3, are extremely low.

The most useful chemical indicator of environmental availability used in this study is believed to be the mild HNO_3 digestion at room

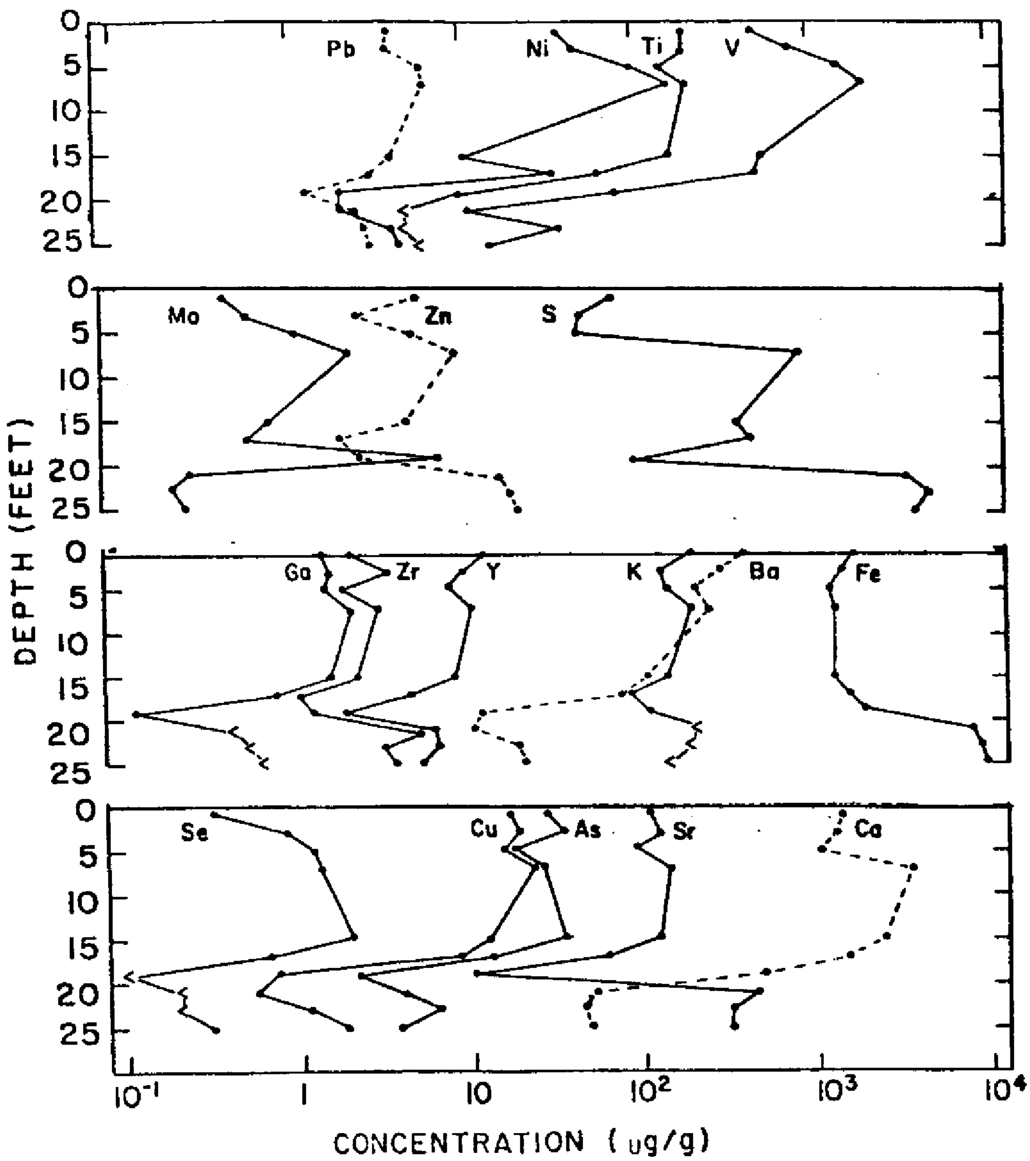

Figure 2. Trace element variation with depth in fly ash and soils from well 17D. PIXE results are from 5N HNO_3 leachates in ug/g (dry weight)

temperature (non-oxidizing conditions). The results of the acid digestions, expressed as percent of "total metal", were generally in the range of 40-90%. These percentages probably represent an upper bound on the portion of these elements which would be available to the environment under the natural conditions encountered. While the percentages for many elements are relatively small, it should be kept in mind that there is a very large volume of fly-ash in these pits.

The vertical distribution of trace elements within and below fly-ash Pit C is illustrated by Figure 2 for the drilling core from well 17. Physical examination of this vertical profile revealed fly ash down to a depth of approximately 18 feet, then approximately 2 feet of quartz sand remaining from the Tabb formation, followed by penetration into the Yorktown formation below. Arsenic, for example, can be seen at a concentration of approximately 50 ppm throughout the fly-ash, decreasing to 4 ppm in the Tabb formation and rising to a concentration of approximately 8 ppm in the underlying Yorktown formation. In general it can be seen that most elements are higher in concentration in the fly-ash than they are in the underlying Yorktown formation with notable exceptions of iron, zinc and sulphur. Molybdenum is apparently highest in the Tabb formation.

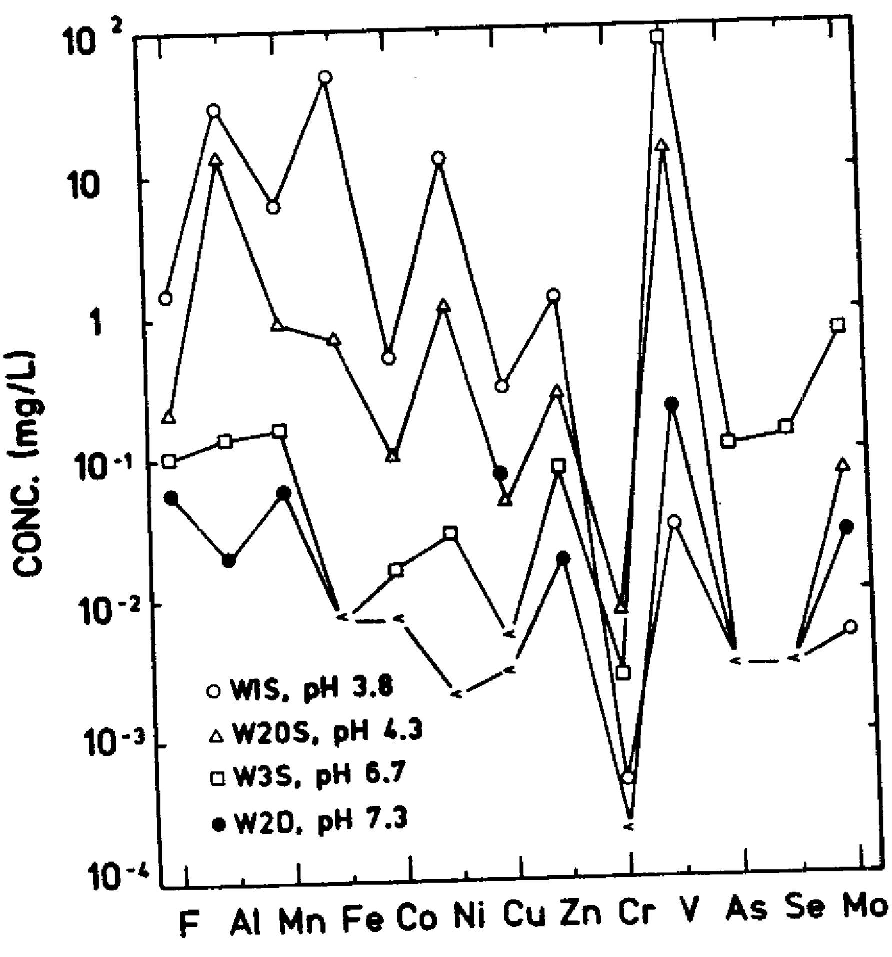

Figure 3. Dissolved trace element concentrations in groundwater residing in fly ash pit C.

Representative trace element data for water from monitoring wells in pit C are plotted in figure 3. The most highly contaminated water was found in wells 1S and 20S, near the bottom of the fly ash and to the downstream side of groundwater flow. Well 3S lies in the pit bottom on the upstream side and well 2D is adjacent to well 3S, but completely in the Yorktown formation underlying the fly ash. These analyses are included in a more complete spatial profile than possible from data reported previously (VA, SWCB, 1981), which shows that upon percolation through the fly ash of pit C, groundwater pH decreases to 3-4 while leaching trace elements before entering Chisman Creek.

Trace element transport to the estuary.

One goal of the groundwater monitoring was to determine if conditions varied seasonally. This was accomplished by repetitive sampling of most of the SWCB wells and some of the wells installed by Oakes (1982). Analysis of these data for most wells, within and outside the disposal sites, showed concentrations varied by less than a factor of 4 for the entire three year period. All of these samples were taken under quiescent conditions, at least several days after a storm event.

The primary purpose of the groundwater sampling was to ascertain the degree of contamination and the area affected. A necessary step towards that goal was the determination of groundwater characteristics within the fly ash deposit. Accordingly, initial groundwater samples were analyzed for a large number of elements. Outside fly ash pit C to the south (upstream), the concentrations of most elements in monitoring wells dropped rapidly to near detection limits, especially acid soluble ones such as Al, Fe, Mn, and Ni. In contrast, the concentrations of elements which can exist as anions near pH 7 (As, Se, V, Mo, etc.) persist at elevated concentrations, although at levels generally below Va Dept of Health limits.

Since there are many areas where exposed fly ash particles as well as dissolved contaminants may be dispersed further during storm events, surface water samples were analysed from a drainage ditch between pit A and new home sites and also from a drainage pond. Storm flow water (pH 7.0) had higher concentrations of most elements than controls which increased about an order of magnitude for most elements after 3 days standing (pH 6.4). Pond waters (pH 7.6) showed similar and often greater levels of contamination.

In order to assess the importance of particulates in storm runoff, PIXE analyses were conducted on suspended particulate matter from stream water, pond water and the drainage water in the construction ditch. All suspended particulates proved abundant in the major and trace elements relative to particulate matter in water from the control stream site. Especially enriched in storm water particulates were vanadium, manganese, nickel and arsenic. Comparing particulate trace to the dissolved fraction for the same water samples, it is evident that some trace elements, such as vanadium and nickel, are entering the estuary in both particulate and dissolved forms.

Minor and Trace Elements in Plant Tissues.

The concentrations of 28 selected trace elements were determined by PIXE from powder targets for 6 plant tissue controls. Excellent detection limits were obtained for several "difficult" elements to determine: arsenic, selenium, and molybdenum. Arsenic and selenium were generally below detection limits in control samples, but frequently elevated and precisely determined in plants grown on contaminated sites in neutral soils. Vanadium, chromium and cadmium, three environmentally significant elements present in the fly ash, were below PIXE detection limits in most plant samples. However, the detection limits for these elements can be improved by approximately two orders of magnitude after digestion of the plant tissues followed by graphite furnace atomic absorption analyses, and these determinations should be done in future studies.

The concentrations of 21 trace elements found above PIXE detection limits in the wetland monocot, Typha latifolia (cattail) are compared in Figure 4 to control values. Both nickel and selenium are markedly elevated in this aquatic plant located near well 1 in the path of ground water draining from fly ash pit C. Interestingly, the concentrations of several other elements appeared to go in opposing directions - comparing cattail shoots to the inflorescence portion of this particular plant - most notably, potassium, barium, manganese and iron. The use of a "<" symbol at a relative concentration of 1.0 denotes that the trace element in both sample and control was below detection limits. A "<" symbol at a relative concentration less than 1.0 means that the control sample alone was found to be above detection limits, and a ">" symbol above a relative concentration of 1.0 means that the contaminated sample only was found to be above detection limits.

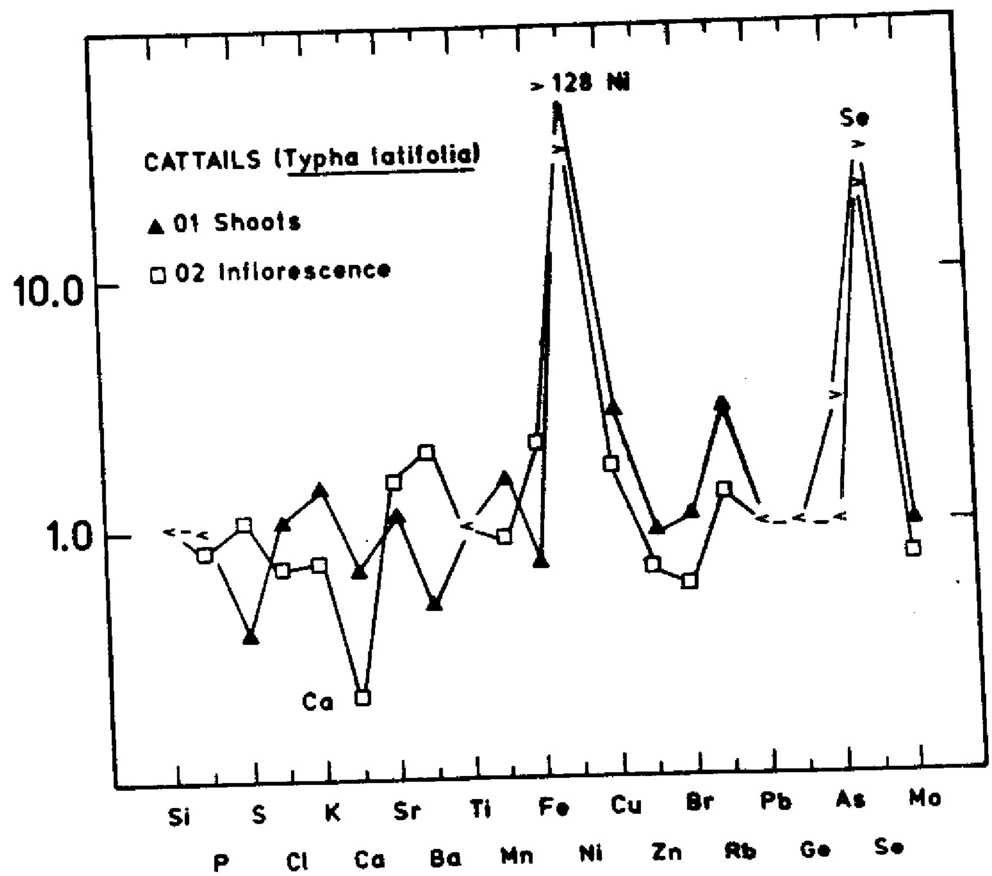

Figure 4. Trace elements in fly ash exposed cattails, relative to controls. PIXE analysis of powder targets.

The trace element composition of _Pinus virginiana_ tissues taken from two different sites in fly ash pit A (pH neutral) show that concentrations of strontium and barium, two elements abundant in the fly ash, are elevated at the expense of calcium. Moreover, the concentrations of arsenic, selenium and molybdenum -- three elements present in fly ash and soluble under pH neutral conditions -- are markedly elevated. Similarly, for _Pinus taeda_, the concentrations of strontium, germanium, arsenic, selenium, and molybdenum are elevated in plants growing on neutral fly ash but much lower when located near well 1 (acidic conditions). This is undoubtedly related to the chemical solubility of the elements under these conditions, as well as bioavailability to plants. The concentrations of manganese, nickel and copper, three acid-soluble elements, are higher in _P. taeda_ grown near well 1 than near well 4.

Trace elements for sweet gum tissues, also collected near wells 1 and 5, were compared to controls. Once again, for a plant grown on neutral fly ash, many trace elements are elevated in concentration, most conspicuously nickel and selenium. Interestingly, the concentrations of chlorine, manganese, bromine and lead are much lower in these tissues than in "controls".

As part of this preliminary survey, numerous other plant species were collected on or near fly ash pit sites, for which suitable controls of the same species were not available. For several of the plant species, the concentrations of Ni, As, Se, Rb, Sr, Mo, and Ba are elevated compared to other control plants of similar species. These elevated concentrations apparently reflect bioaccumulation by these species also from trace elements in the ground water.

A recent study of the mobility and bioavailability of elements in uranium mill tailings (Dreesen, et. al.) also reported the release of As, Mo, Se, and U from alkaline tailing leachates, which were readily assimilated by two western plant species, especially Mo and Se. Scanlon & Duggan (1979) have investigated trace element uptake in eight woody plant species from fly ash and reported that Ni and Se appeared to be especially available to plants. Our findings with well weathered fly ash appear consistent with these reports.

Trace elements in estuarine sediment and oysters.

Most trace elements are transported within a water system via particulate matter and, due to physical processes, are deposited as sediments. Thus they provide a record of prior conditions. Chisman Creek is a small (4 mile) subestuary fed by numerous small freshwater streams which are generally narrow (3 feet) and shallow (less than 2 feet). Goose Creek drains the northern portion of the basin. The main channel of the estuary is broad (0.5 mile) and river depths at Mean Low Water range from 12 feet at the mouth (confluence with the Poquoson River) to less than 3 feet adjacent to the ash disposal sites.

During the sampling period, seasonal runoff variations caused salinity in Chisman Creek to fluctuate; spring values ranged between 14-18 ppt throughout the main stem; in summer the creek was relatively homogeneous, and salinity was about 20-21 ppt. The pH in both Chisman and Back Creek estuaries ranged from 8 to 9. However, the upstream station in Chisman Creek, which was adjacent to an ash disposal site, was more acidic (pH of 6.4). Suspended solids ranged from 25-50 mg/l in both creeks.

Estuarine sediment samples were analyzed for 69 trace elements, many of which were below detection limits. Previous fly-ash studies in other estuaries have found arsenic, cadmium, chromium, lead, nickel, selenium and zinc at levels of 1 ppm or greater in fish tissues (Davison et al., 1974, Dreeson et al., 1977, Ray and Parker, 1977, Theis and Wirth, 1979). Because of the potential ecological and health concerns, the discussion of results has focused on these elements, as well as copper and vanadium.

Vanadium, arsenic and nickel exhibited a sharp increase in concentration with distance upstream. Of the 69 elements analyzed, vanadium in Chisman Creek sediment samples underwent the most dramatic increase. At the mouth of Chisman Creek, levels were 4.6 ppm. Four miles upstream, vanadium levels reached 541 ppm. Additionally, following a rain storm, stream sediments adjacent to Pit C (STR 30) contained vanadium concentrations of 605 ppm.

It is important to note that all sediment samples were passed through a 3mm mesh sieve to remove large particles and shells. However, no additional fractionation of the samples was done. Variations in particle size distributions undoubtedly affect the trace element distributions, and some portion of the observed decrease in concentration downstream is due to the increase in particle size that typically occurs in all estuaries (Luoma, 1983)

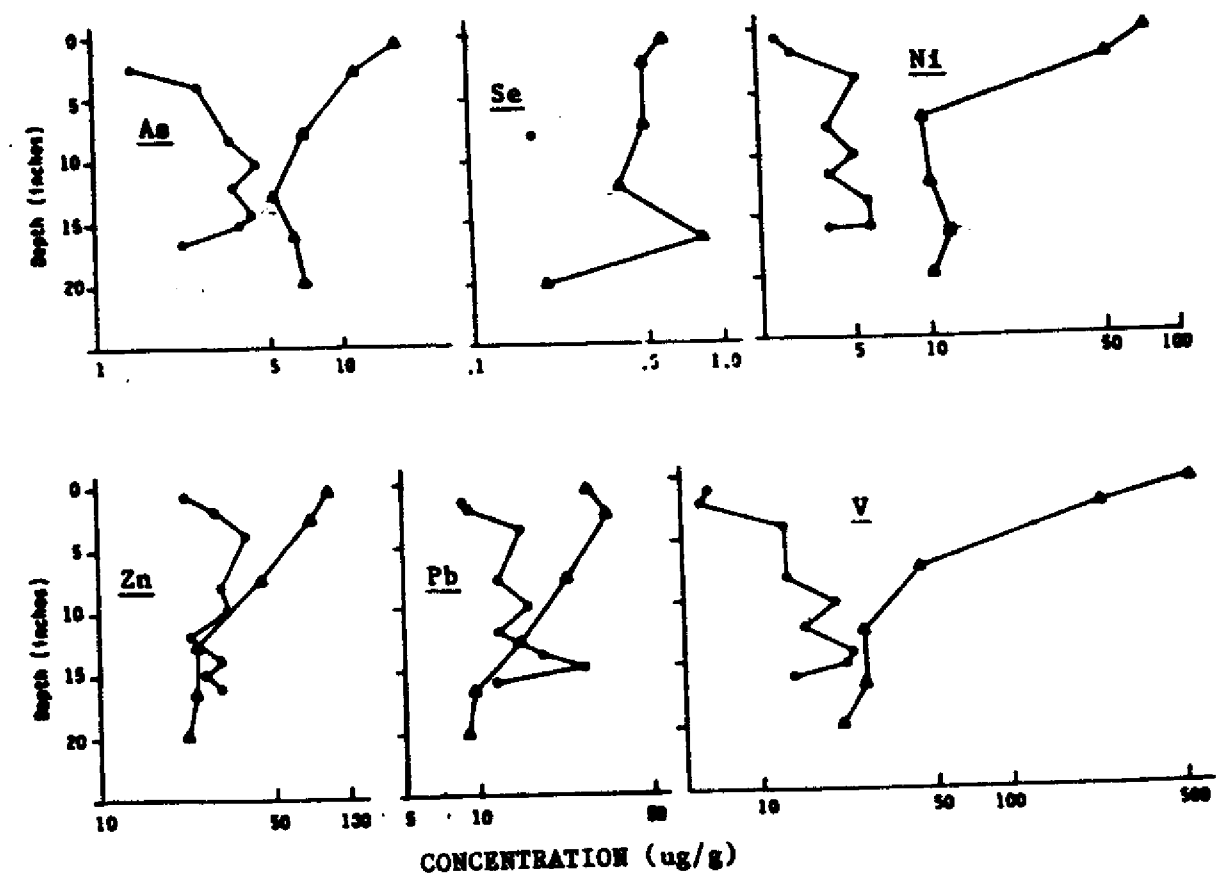

Figure 5. Trace elements in sediment cores from Chisman Creek ( ) and Back Creek ( ), the control site. PIXE analyses of 5N HNO_3 leaches.

Analyses for 20 inch sediment cores are given in figure 5. For the elements vanadium, selenium, zinc, lead, arsenic, nickel and copper, concentrations were in all cases lower in the control core taken in Back Creek than in the core from Chisman Creek. Cadmium was not measurable in either creek and chromium was detected only at a depth of 12.5 inches in Chisman Creek (6 ppm). Concentrations of the trace elements in Chisman Creek were generally highest in the top 6 inches of sediment and, except for zinc and lead, these values were an order of magnitude greater than those present in Back Creek.

Vanadium, nickel and arsenic, elements which appear to be reasonable indicators of fly-ash impacts in the estuary, were greatly elevated in the Chisman Creek core to a depth of 6 inches when compared to the control core. Concentrations became more constant at depths greater than 10 to 12 inches, which may reflect background levels for the estuary. It also should be noted that at depths greater than 15 inches (75 years), trace element concentrations were similar in both creeks. A physical inspection of sediment texture indicated that the Chisman Creek core sediment was quite homogeneous with depth. The presence of uniform texture suggests that the observed elevated trace element concentrations in the upper 6 inches reflect contamination rather than the influence of a gradient in sediment particle size.

Trace element levels in oysters are difficult to interpret, primarily because of the low abundance in this estuary. Vanadium appeared to be the only element that showed a distinct increase in concentration with distance upstream. No trend was evident for most elements, although concentrations of copper, zinc, cadmium, nickel, chromium and lead were highest in oysters from Goose Creek.

Summary. The composition of fly ash and groundwaters in and near waste disposal sites in York County, VA have been investigated. Ten years after closing this site, the groundwaters outside the pits contain substantial concentrations of many trace elements present in the ash. Continuing dispersal of many trace elements through leachates and particulates during storm events was demonstrated. PIXE analyses revealed accumulation of several trace elements in terrestrial and aquatic plants and the top layers of estuarine sediments in Chisman Creek.

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TRACE ELEMENT CONTAMINATION FROM FLY ASH SITES NEAR CHISMAN CREEK, VA.

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SUBSURFACE HYDROLOGY IN A MESOHALINE TIDAL MARSH

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ABSTRACT

The role of tidal marshes in coastal nutrient budgets has been the subject of numerous studies in estuarine ecology. Few studies, however, have assessed the importance of subsurface hydrology as a possible mechanism of chemical exchange between marshes and open water. This paper reports an initial characterization of the subsurface flow regime of a tidal marsh. The study site was found to possess stratigraphic layering and vertical variation in hydraulic conductivity. Time records of piezometric head suggest that, over a tidal cycle, a net horizontal export of water to the creek occurs. Evidence for the existence of vertical flow gradients during some time periods is also shown. Implications for modeling water exchange between marsh soils and adjacent estuaries are discussed.

INTRODUCTION

The exchange of materials and energy between tidal marshes and adjacent water bodies has been a subject of intensive research over the past several decades. As a result of this effort it appears that many marshes may import sediment, inorganic nutrients bound to sediments and heavy metals (Nixon, 1980). Speculation that salt marshes may act as long term sinks for materials encouraged some workers to monitor experimental loadings of nutrients and heavy metals on marshes to test whether the marshes could assimilate and store pollutants from the water column. Results were often considered encouraging (Valiela et al., 1973; Simpson et al., 1983). Other studies indicated that salt marshes export materials, especially in the case of particulate organic carbon and dissolved nutrients (Odum, 1979; Nixon, 1980). These results suggest that although marshes may accumulate materials over the long term they may still be relatively "leaky" systems for many substances over the course of seasons or years.

Chemical transport mechanisms between marshes and adjacent water bodies are hydrologically mediated. The mechanisms of chemical exchange by surface flow on tidal marshes have received some attention (Odum, 1979), yet few detailed hydrological studies have been conducted to characterize subsurface fluxes between marshes and open water (Hemond et al., 1984; Jordon and Correll, 1985). Yet, the high interstitial concentrations of dissolved nutrients in salt marsh soils suggest that even limited seepage across the soil boundary could have profound effects upon the capacity of the marshes to retain nutrients.

This study represents an initial step towards identifying the pathways and volumes of subsurface exchange between a mesohaline tidal marsh and the adjacent estuary.

METHODS

The study site is located at the mouth of a small, tidally flushed tributary (Carter Creek) of the York River, Va. Research was conducted on a transect across a narrow marsh (20m) without complex drainage channels. The marsh is vegetated primarily by *Spartina alterniflora* with a narrow zone of *Spartina patens* and *Distichlis spicata* occurring at the back of the marsh at the base of the upland slope. A profile of the marsh, including the underlying strata, was determined. To facilitate measurement of the height of the water table and pore pressure at depth, small diameter wells and piezometers were constructed from extruded acrylic pipe (3/8" I.D.). The tips of the piezometers were milled with densely drilled holes (diam = 2 mm) for a length of 10 cm and then capped; holes were drilled over most of the length of the well tubes. The wells and piezometers were installed next to the elevated catwalks at seven locations on a transect perpendicular to Carter Creek. Two wells and a nest of four piezometers (two at 25 cm, one each at 45 and 75 cm) were installed at each sampling location.

Once in place the piezometers were used to estimate hydraulic conductivity by the method of Luthin and Kirkham (1949). Pore pressure and water table heights were then monitored during four complete tidal cycles. Measurements were repeated in each instrument on approximately 30 minute intervals.

RESULTS

Coring at the study site revealed a multilayered system. The porous and highly organic surface soil is relatively thin (0.3 to 0.8 m) and overlies two mixtures of sand and finer materials (Fig. 1). At the back of the marsh (adjacent to the upland slope) the marsh is underlain by a yellowish sand with intervening layers of tan clay. This soil inclines steeply toward Carter Creek and is contiguous with the soil of the upland slope at the back of the marsh. Beneath the marsh-creek transition is a dark sand intermixed with fine mineral and humified organic material.

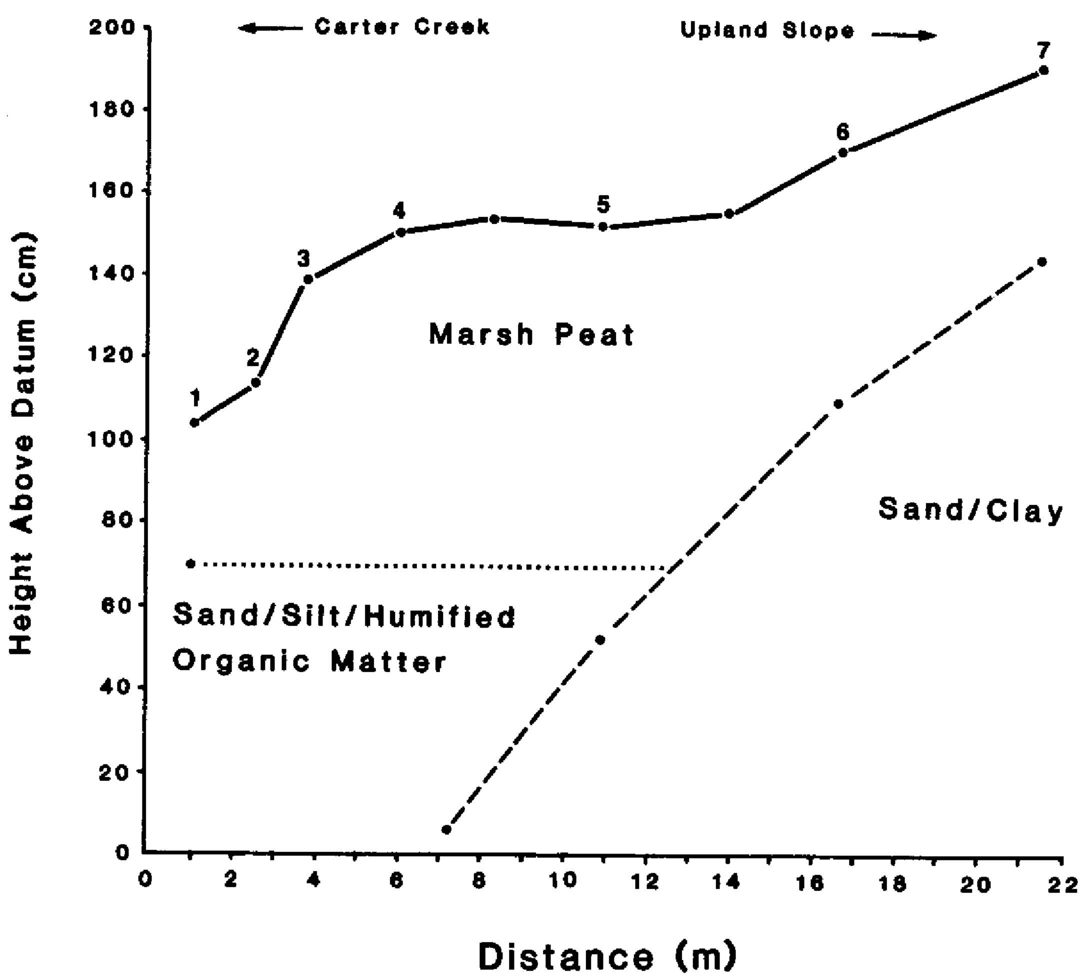

Figure 1. Topography and stratigraphy at the Carter Creek study site. The measurement locations are numbered.

Averaged over all of the transect locations and depths, the mean hydraulic conductivity (K) was 5.8×10^{-4} cm sec^{-1}, which compares closely with the conductivity of a very fine sand or a silt. The marsh peat alone had an average conductivity of 9.8×10^{-4} cm sec^{-1} which exceeded that of the underlying layers by more than an order of magnitude (Table 1).

TABLE 1. HYDRAULIC CONDUCTIVITY (cm sec^{-1}) OF MARSH PEAT AND UNDERLYING STRATA.

Location	n	$\bar{x}$	Range min	Range max
1) marsh peat	10	9.8×10^{-4}	3.0×10^{-4}	3.8×10^{-3}
2) sand/silt/ organic	5	8.7×10^{-5}	1.7×10^{-5}	2.0×10^{-4}
3) sand/clay	3	5.2×10^{-5}	2.1×10^{-5}	7.6×10^{-5}

Figure 2 shows a typical time record of hydraulic head from station 3, located at the top of the steepest part of the creekbank (Fig. 1). Between 8:30 A.M. and 2:30 P.M. the hydraulic gradient is toward the creek indicating subsurface flow is in that direction. As water drains from the marsh soil there is a slow downward movement of the water table. Higher values of hydraulic head at 75 cm suggests that upward seepage joins the predominantly horizontal flow out of the marsh. At 2:30 P.M. the level of the rising tide exceeded that of the water table at station 3. After that time the direction of flow reversed and the drained portion of the marsh began to refill with water seeping in laterally from the creek. About 3:00 P.M. the flooding water spread across the surface at site 3 causing vertical infiltration to begin (indicated by the vertical spread of hydraulic head values after 3:00 P.M.). By 4:00 P.M. the replacement of pore water was complete at site 3 as shown by the correspondance of hydraulic head at 25 cm with the level of the flooding tide.

DISCUSSION

Detailed hydrological investigations have been attempted only recently in tidal marshes (Hemond and Fifield, 1982; Knott et al., unpublished; Hemond et al., 1984). Investigations of the hydrological properties of marsh soils have revealed wide variation in hydraulic conductivity and horizontal layering with a tendency for conductivity to decrease with depth (Knott et al., unpublished). In addition, monitoring of pore pressure distributions in marshes over time have shown that vertical, as well as horizontal flow

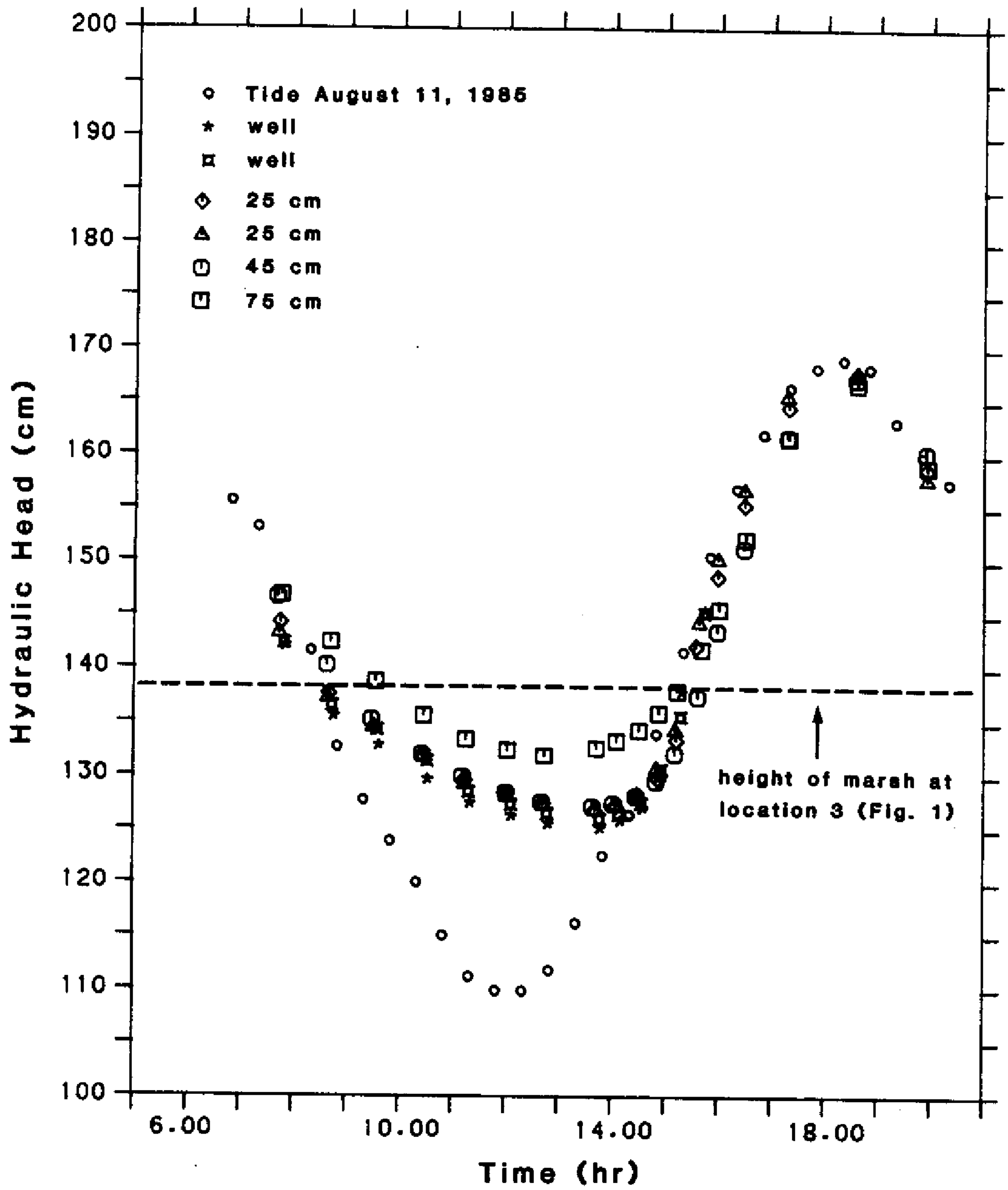

Figure 2. Time record of water table heights and piezometric head at three depths: Station 3, August 11, 1985.

appears to occur beneath marshes (Hemond and Burke, 1981; Hemond and Fifield, 1982). At this preliminary stage in our work, our findings have corroborated those above.

Rapid refilling of the drained portion of the marsh (1 hour compared to 6 hours to drain) also was noted in the study of Jordon and Correll (1985). We hypothesize that it occurs because the drained area of the marsh is refilled both with water from the rising creek and by continuing subsurface flow toward the creek from higher parts of the marsh. A net, horizontal export of water from the marsh soil to the creek over one tidal cycle is therefore implied. The net export of water through the creekbank is compensated for by vertical infiltration once the surface of the marsh becomes flooded.

The presence of layered soils and vertical flow components in tidal marshes indicate that simplified approaches to estimating subsurface flow (Jordon and Correll, 1985; Yelverton, unpublished) can probably be improved upon. We are preparing a two-dimensional, numerical model which can incorporate spatial variation in hydraulic conductivity and vertical flow gradients. This approach will provide insight into the relative importance of horizontal versus vertical flow pathways as well as provide more reliable estimates of the volumes of pore water exchange in tidal marshes.

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ECONOMIC AND INSTITUTIONAL PERSPECTIVE

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ABSTRACT

Economics can and must play a more prominent role in policy formulation and resource allocation for managing impact of land uses on the Chesapeake Bay. Cost-sharing for best management practices through the Virginia Nonpoint Program must be based on most efficient activities. Therefore, limited resources must be targeted to those activities which provide the greatest return in terms of improved water quality. A major issue which must receive greater attention is the economics of tradeoffs between control of point sources versus nonpoint sources of pollution.

INTRODUCTION

Economics is a study of the proper method of allocating scarce resources among competing ends--an allocation that achieves a stipulated optimizing or maximizing objective. (Ferguson, 1971, p.1) The primary concern in management of the Chesapeake Bay is resource allocation. Therefore economics can and must play a more important role in policy formulation and resource allocation for managing impact of land uses on the Chesapeake Bay.

From an institutional perspective, property rights become an important consideration in any economic analysis of pollution and related environmental matters. Dealing with property rights requires an interface between law and economics. Property must be considered in terms of rights, not concrete objects. Ownership must be considered as a set of rights to use property in certain ways (sell rights, prevent others from using, etc.) Property and ownership are created by, defined by, limited by society's system of law. (Dales, 1975, pp. 58-76) Economics, then, must consider the using of property rights within this system of law.

A special kind of property, common property, that which is owned and used in common, constitutes much of the natural resources of the Chesapeake Bay area. A few restrictions exist on use of common property (for instance, limit on rockfish harvest) but most of the resource is unrestricted with respect to use. This unrestricted situation can be characterized as a no-policy on use of common property. But, a no-policy always favors someone (swimmers vs polluters). One question the economist asks, what is the cost of enforcing some stated policy compared to the gain from adopting the policy? Economic concepts must be brought to bear on this type of policy design.

Economics cannot be used in a vacuum but must include social issues such as who has the right to continue to pollute and who has the right to other uses of the waters. For instance, what are the societal implications of assigning pollution rights in one area and water rights to fisherman in other areas? Likewise, technological efficiency as well as decisions to develop new technologies creates significant socio-economic consequences. For instance, technologies for control of nonpoint sources of pollution not only have a significant impact on cost of agricultural production, but most likely will change the structure of agriculture as it is known today--that is, type of crops and livestock produced in any given area and the nutritional value of those products will change.

The bottom line is that managing the environment must involve a process of allocating scarce resources. Choices must be made. The economic system provides a good framework for making those choices. But the system must be made to function within existing institutions and cultural arrangements to guide the allocation of resources within both the private and public sector. (Freeman et. al., 1983, pp. 64-79)

Private markets often fail because prices for environmental goods and services are not included in production and distribution decisions for many goods and services when self-interested individuals are not held responsible and take advantage of private benefits without bearing costs. Economists are fully aware of these market failures in which resources are not allocated to their desired use. But, let us also remember that just as inefficient markets exist so do we have inefficiency in the public sector as inefficient governments attempt to allocate resources. Certainly, we have the self-interest of politicians, bureaucrats and other special interest groups. Quite often in the public sector, efforts cost more and clean up less than if the system were well run and efficient. Economists must try to balance these imperfections. (Schultze, 1977, pp. 43-50)

Water pollution controls are costly. It is obvious that devoting more resources to pollution control means less resources to do other things which are valued by a society. This is the basis for the concept of opportunity cost. Society must compare what it receives from devoting resources to pollution control with what it gives up by taking resources from other uses. This concept is the heart of the benefit-cost analysis approach. Also, the concept of marginalism--small or incremental changes--should be used for establishing C/B type choices at the margin. The important question is whether the incremental increase in value gained from one use is worth the opportunity cost in incremental changes in other uses.

Many tools already exist for use in analyzing resource use decisions. For market type goods, use values as determined through prices in the market economy reflect scarcity and a willingness of consumers to pay for the good or service. Market goods estimating techniques can be fairly easily applied and used to establish resource use values. Examples include consumer surplus, net factor income and cost savings in production approaches.

The second category of goods and services is the nonmarket type--prices not determined by the market economy. While difficult, a body of methodology has been developed and is being used to estimate values for nonmarket goods and services. These techniques are being used to place value on uses of wetlands, controlling erosion, shellfish harvest, waste disposal and transportation services. Such approaches use property value differentials, household expenditures, repair of damaged materials and travel costs. These evaluation procedures estimate impact of changes in water quality as related to various measurements of value of people.

Many economic studies are being conducted involving both market and nonmarket goods as they relate to the effects of upland and shoreline land-use activities on the Chesapeake Bay. These studies involve the institutional aspects of management agencies, fisheries management, recreation, and marina operations just to name a few. But for convenience the remainder of this paper will focus on economic and institutional considerations of controlling nonpoint sources of pollution.

THE VIRGINIA NONPOINT PROGRAM

Virginia initiated a program in July 1984 to reduce agricultural and urban nonpoint source pollution in the Chesapeake Bay Drainage Basin in Virginia. The Virginia General Assembly approved a total of $975,000 for the first year. The U.S. Environmental Protection Agency contributed an additional $875,000 making the total funding $1,850,000 for FY 1984-85. This amount will be increased to $3,600,000 in FY 1985-86 with $1,425,000 coming from state sources and $2,175,000 coming from the Environmental Protection Agency. (Virginia Department of Conservation, 1985, p. 1).

Efficiency of BMPs

A primary economic concern is the efficiency and effectiveness of the urban and agricultural Best Management Practices (BMP) cost-sharing component of that program. The purpose of the BMP cost-share program is to provide direct financial assistance to farmers who implement BMPs that will result in a reduction in pollution potential. As more and more BMP's are implemented through cost-sharing it is intended that they will become standard practice so that the need for cost-sharing will eventually diminish. But just as government control and regulation are inefficient, from an economic efficiency perspective, cost-sharing subsidies can be shown to be inefficient as the best way to improve or maintain water quality. Thus, the question is how long and in what quantity the state should provide cost-share monies relative to other available alternatives. Alternatives include: tax credits, loans, charge systems, preferential assessments, insurance, use of taxes, etc. While additional cost-share funds may be needed at the outset of the program, the ultimate measure of progress from an institutional perspective could be eliminating the need for cost-share monies. Options other than cost-share may very well be more efficient in obtaining desired water quality goals.

Debates about whether compliance with controls on nonpoint sources should be mandatory or voluntary continue. Those opposed to voluntary controls contend that goals will not be met if we rely solely on financial incentives to control sources. Some legislators contend that the taxpayer is not obligated to pay for controls--a property rights question. Other legislators contend that it is inevitable that nonpoint source controls be mandatory in order to protect the investment and the advantages resulting from mandatory point source controls. (Environmental Reporter, 1983)

The optimal level of control of nonpoint sources must receive increased attention. Some suggest a cost-effectiveness ranking of policies and management practices for allocating scarce resources. That often used term, cost-effectiveness analysis, is defined as finding ways to accomplish a goal in the least expensive way possible. Or, it can mean squeezing the most environmental protection from a given budget. Although well-developed, this analysis has most often included only costs of resources used in the mix of technological control strategies. Some important components which have been overlooked from society's standpoint are the direct costs associated with a project (overhead administrative cost, enforcement costs, maintenance costs, costs of obtaining knowledge and costs of evaluation) and the opportunity cost of using scarce resources for a given project or at a given level of effort relative to other uses of those scarce resources.

Targeting of Limited Resources

Expenditures for BMPs in any area for any practice must be compared to the impact on water quality and uses of that water body. Limited available resources must be targeted to areas and/or BMP practices with the greatest potential for water quality improvement. Although availability of BMP funding is severely limited, the Virginia program does attempt to target available resources to priority areas and for priority practices. The Virginia plan contains a starting framework to provide site-specific information on cause-and-effect relationships. A tracking system is being used to provide information on reduction in pollutants that are transported to the Bay. The program utilizes the existing Soil Conservation Service (SCS) reporting system, aerial photography to identify potential problems, limited monitoring on some small watersheds, and some computer modeling. Information on the effectiveness of BMPs will be provided by built-in demonstration projects. But, an effective system must become a management system, and not just a report generator. Therefore, the system must be modified to include measures of change in off-site water quality impacts as well as measures of on-site activities. The program requires an evaluation of the direct impact of land activities on uses of the water. Site-specific criteria that take into account local conditions for determining use attainability is the key concept. That evaluation must be a foremost objective in measuring progress of any nonpoint source control program.

One method considered by the State was to prioritize use of funds and direct them to those areas with the highest potential for causing water quality problems by weighting the Universal Soil Loss Equation (USLE) loading factors by an indicator of distance from a stream and the elevation relative to the nearest stream. This estimate of cost per ton of sediment load reduction appears to be a considerable improvement over past targeting procedures. (Kramer, 1985, p. 8)

While the pollution source identification data base will aid the targeting concept, economic efficiency requires that both wetland and agricultural BMP activity be related to changes in water quality and the subsequent impact on uses of the affected water resources. One policy objective for any nonpoint program should be clear; that is, to assure the most effective use of limited resources available to the program in terms of water quality improvement, not on-site soil productivity.

Economists recognize that the interrelationships between use of nonpoint best management practices on land and the impact on water quality has received little attention. The ability to relate pollution loads to water concentrations is still imperfect and the scientific basis for definitive cause and effect is only now being established. Consequently, the question of how much pollution must be reduced to achieve a quality of water that can support a specific living resource or other level of use of the water resource in a site-specific area becomes the significant question for better localized nonpoint management decisions. The decentralized-decision making can use economic incentives and disincentives to minimize cost of obtaining given ends. Most often the installer of the BMPs is the best person to find the cheapest way. Economists can and must help ask the right questions and make known needs for specific data for the decision process.

The recent Chesapeake Bay study by the U.S. Environmental Protection Agency clearly demonstrates the relationship of nutrients to decline in submerged aquatic vegetation and the consequent loss of spawning and nursery grounds. (Mackiernan, 1984) The study also demonstrates the negative impact of toxic compounds on hatching and survival of fish resources. Impacts on other resource uses such as commercial shipping, domestic water uses, recreation, irrigation, etc., must also be considered and integrated into the economic analysis.

Nonpoint Versus Point Source

The comparison of nonpoint controls with point source controls is becoming a significant economic concern. An important measure of progress from a policy perspective will be how well benefit/cost measures of nonpoint controls are used to make comparisons with benefit/cost measures of point source controls. Some rather comprehensive work at Resources for the Future indicates that past measures to control pollution from cropland would have had greater effect in reducing phosphorus delivered to the nation's water than measures to control discharges from industrial and municipal point sources. (Crosson, 1983) Only through use of realistic cost/benefit values can appropriate decisions be made about tradeoffs on the most efficient combination of point and nonpoint controls for any given area. Consideration must be given to the economic feasibility of both point and nonpoint controls because cost savings result from using a combination of the two. While the tradeoff analysis has received only limited attention in bay management, the Commonwealth's program makes it possible to start evaluating the differences among point and nonpoint controls as well as between these controls and the resulting improvements in resource uses.

Tradeoff between control of point versus nonpoint sources for any given area is gaining increased attention at all levles of government. Jaksch of U.S. Environmental Protection Agency (EPA) office of Policy, Planning and Evaluation states that existing nonpoint sources can often be controlled far more economically than further controls of point sources. Also, nonpoint control could allow for future growth of point sources discharges where water quality would therefore not be threatened. (Environmental Reporter, 1984) Much of the debate at the state and federal level is based on this interrelationship between control of point and nonpoint sources. To date there has been limited progress in implemented nonpoint controls even though water quality improvements in some waters such as the Chesapeake Bay can be better obtained through control of nonpoint source pollutants.

The main barrier to controlling nonpoint source is not technological. It is the absence of an effective and acceptable institutional framework.

POLICY STUDY OF NONPOINT PROJECT IN CHESAPEAKE BAY

A recent study by researchers at Virginia Tech evaluated alternative public policies for encouraging the use of agricultural BMPs. The study focused on two coastal watersheds (the Nansemond and Chuckatuck watersheds) which drain into the Chesapeake Bay. It provides information about the potential effectiveness of public policy actions for reducing nonpoint source pollution. Policies examined included regulatory programs, soil loss taxes, and cost-sharing programs. A watershed model was constructed which allowed a unique opportunity for policy analysis. Unlike previous economic studies in other areas which generally focused on soil loss as a proxy for nonpoint source pollution, the model used in this study enabled a simultaneous analysis of policy impacts on soil loss and on nitrogen and phosphorus pollution. The overall purpose of the study was to analyze the economic relationships among agricultural production activities, pollution control policies, and generated pollution. (Kramer, et al., 1984)

Detailed study results are available in published form. In summary, the results indicate that a regulatory program can have differing effects on cropping patterns depending on which pollutant is targeted for reduction. A program with explicit water quality goals could have a different impact on land use practices than a program like the SCS Agricultural Conservation Program which has a primary goal of maintaining soil productivity. This analysis suggests that studies of nonpoint pollution control which use soil loss as a proxy for nutrient loadings may yield misleading results.

The regulatory approaches, while effective in reducing pollution, decrease net farm income in the watersheds. However, in no case do the reductions exceed four percent of base line income. Yet, even income declines of this magnitude could have severe impacts on farmers, particularly during periods of financial stress for agriculture. Of course, a regulatory program would be objectionable to many because of its interference with farmer decision making. Nor does it encourage greater pollution reduction for those farms which can abate pollution more cheaply than others. The regulatory approach would be difficult to implement since estimation of pollutant loadings on a farm by farm basis would be required. An alternative regulatory approach not considered in this study would be to require the use of specific BMPs rather than limit the amount of allowable pollutants.

Effluent taxes are another method of encouraging the adoption of BMPs. While this study examined a tax on soil loss, taxes on other pollutants could also be imposed. A $0.50 per ton tax has a modest effect on pollution generation in the model, primarily by encouraging a shift to no-till. With a $1.00 per ton tax, it becomes economical to avoid part of the tax by planting no-till corn, using a cover crop with peanuts, and installing grassed waterways. Like the regulatory program, a soil loss tax would be unpopular with farmers and difficult to enforce since soil loss would have to be determined for each farm.

The cost-share alternatives appear effective in reducing pollutant loadings and have the political advantage of raising net farm income. In this study, cost-shares greater than 50 percent had little effect on generated soil, nitrogen, or phosphorus, but did encourage use of animal waste BMPs on hog farms. Cost-shares in the model lead to increased farm income because farmers receive income both from agricultural production and from the government. Tradeoffs between farm income and pollution must be recognized. Cost-sharing and other policies can be used to encourage the implementation of nonpoint pollution controls.

Effective nonpoint management programs must utilize site-specific water-use impact analysis as well as on-site productivity analysis to determine most efficient levels of nonpoint control. Those programs must also utilize economic analysis to select optimal combinations of point and nonpoint controls for site-specific areas. Results of the policy study imply that a program with explicit water quality goals could have a different impact on land use practices than a program which has a primary goal of maintaining soil productivity. Tradeoffs between farm income and pollution must be recognized just as tradeoffs between control of point and nonpoint sources are needed for an effective and efficient program. Policies other than cost share can be effectively used to encourage the implementation of nonpoint pollution controls.

Extension of Study to Cross Compliance

McSweeney and Kramer expanded on the Nansemond Chuckatuck study to examine the potential effects of requiring farmers to adopt pollution and erosion control practices as a precondition to qualifying for price support and crop insurance programs. (McSweeney and Kramer, 1986) Such a requirement is referred to as cross-compliance.

As stated by McSweeney and Kramer, risk can play an important role in farmers decisions to use soil conservation measures and was a central concern of the study.

In studying the potential effects of a cross-compliance strategy, it is of particular interest to view the problem in a risk framework since income stabilization is a major goal of farm programs.

A representative farm model was constructed, in which major emphasis was placed on capturing the essence of the soil loss nonpoint control decision problem. A quadratic programming model was constructed for a representative 251 acre southeast Virginia crop farm. Included in the model were the four primary crops in the study area: corn, soybeans, winter wheat, and peanuts. A limited set of BMPs was included in the farm model. On the basis of recommendations by soil conservation personnel in the study area, the following set of BMPs was included: no-till cultivation, sod filter strips, no-till cultivation with sod filter strips, cover crops, sod filter strips in conjunction with a cover crop, and grade stabilization structures.

The model was used to examine the impacts of the two cross-compliance scenarios relative to the current incentives structure.

The first strategy would permit access to program benefits as long as a total farm soil loss limit was not exceeded. Thus, any combination of BMPs would be allowed. The second strategy would require protection of crop acreage by BMPs as a precondition to program enrollment or insurance purchase, regardless of soil loss. Both of these were compared to the current incentive system available to farmers: cost-shares with no strings attached.

A CURRENT STUDY

A study is just underway by Kerns, Kramer and Johnson which will focus on the relationship between on-site activities and on-site/off-site impacts including impacts on the local community.(Kerns, 1985) It will provide the information needed to convince land managers and local officials of the need for various levels of action. It will also provide a framework for making decisions on tradeoff between point and nonpoint controls.

With application of economic analytical techniques and decision tools, two important aspects of the overall water quality program will be greatly improved. First, several studies have demonstrated use of cost-effectiveness techniques for making tradeoffs for control among various sources of nonpoint pollutants. But realistic application demands that total costs as well as total benefits which result from controls must be included in the decision analysis. Second, pressures have been mounting for tradeoff decisions between control of nonpoint versus additional, incrementally-more-expensive point source controls. EPA has indicated a desire to apply some of these cost-minimization techniques to areas of the Chesapeake Bay.

An increasingly important function of the nonpoint program at the local level is going to be more coordination among local inputs (such as land assessment and taxation) and the effective use of available resources including public and private funding sources. Alternatives such as production and enterprise decisions, private investment, and local funding must be considered as an integral part of the on-site/off-site economic impact analysis as well as in the decision analysis for point/nonpoint tradeoff allocations.

Programming-modeling techniques which combine production activities, BMP practices, and water quality constraint measurements will be refined and modified for use in the analysis. Likewise, other existing techniques such as the Virginia Counties' Economic Impact Model and the Virginia Input-Output Model will be modified for evaluation of socio-economic impact analysis. A computerized model which focuses on the relationships among agricultural land use practices and nonpoint source water pollution will be used to analyze various alternatives.

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EMERGENCE OF INTEGRATIVE ENVIRONMENTAL MANAGEMENT

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ABSTRACT

The benefits of integrative environmental management are: cost-effectiveness, long-term protection of the resource, enhanced potential for multiple use, more rapid and effective restoration of damaged resources to usable condition, reduced conflicts over use. Major obstacles to achieving these goals exist in both the educational and management system that must be reduced or eliminated. It will then be possible to manage and protect the Chesapeake Bay as a system.

INTRODUCTION

"In human affairs, the willed future always prevails over the logical future." René Dubos

The value of integrative environmental management is so evident after even the most superficial examination that one wonders why we have such difficulty implementing this goal. The more obvious benefits are:

(a) cost-effectiveness,
(b) improved long-term protection of the resource,
(c) enhanced possibilities for effective multiple use,
(d) more rapid and effective restoration of damaged resources to usable condition,
(e) reduced expenditure of energy on conflicts over use and the possibility of redirection of these energies and funds to resource management.

Despite the many benefits and the fact that integrative environmental management is rarely directly opposed publicly, there are formidable obstacles to achieving this goal, as participants in this meeting fully realize. The benefits are a better program for successfully managing effects of upland and shoreline activities on the Chesapeake Bay and an improved perspective on how best to get cause/effect relationships between upland and shoreline activities and uses of the Bay resources. This should result in improved health in the Bay. I will focus first on some of the obstacles to achieving this goal.

Although integrative environmental management has been practiced for centuries by many "primitive cultures," it is often regarded as a new idea in technologically advanced societies. It is worth noting that the holistic approach required for integrative environmental management should not displace the reductionist approach now in vogue. Dramatic advances in technological development were made possible by the reductionist approach. A reductionist investigates a component of a system to determine how it functions. This development has been seen in the field of medicine where specialists look at only one portion of the human body or at a particular function or disease. Some notable failures resulting from over-specialization have resulted in the "newly emerging field" of holistic medicine. In biology, a geneticist working with fruit flies may have little interest in attending a seminar or paper presentation on chironomid taxonomy, even though both research areas involve insects. Each of these specialists almost certainly publishes in different journals, belongs to and attends meetings of different professional societies, and associates with their "own kind" in terms of research interests at large biological meetings. True, some people do not fit this mold, but most scientists, in this age of specialization where new developments are numerous and appear with great rapidity, can only spend significant amounts of time outside their area of specialization at great peril to their professional careers. Interestingly, new developments in biotechnology have forced some of the geneticists to consider the environmental impact of genetically-altered organisms, thus forcing them to take a more holistic view. However, the task of looking at the relationship of a component to the entire system (Fig. 1) of which it is a part is viewed as a dramatically new idea despite lip service in the academic community on the need for a systems view. Many at this meeting can remember vividly the resistance to Rachel Carson's new idea of looking at the effects of pesticides on non-target organisms as well as on the target organisms themselves. Had she not had

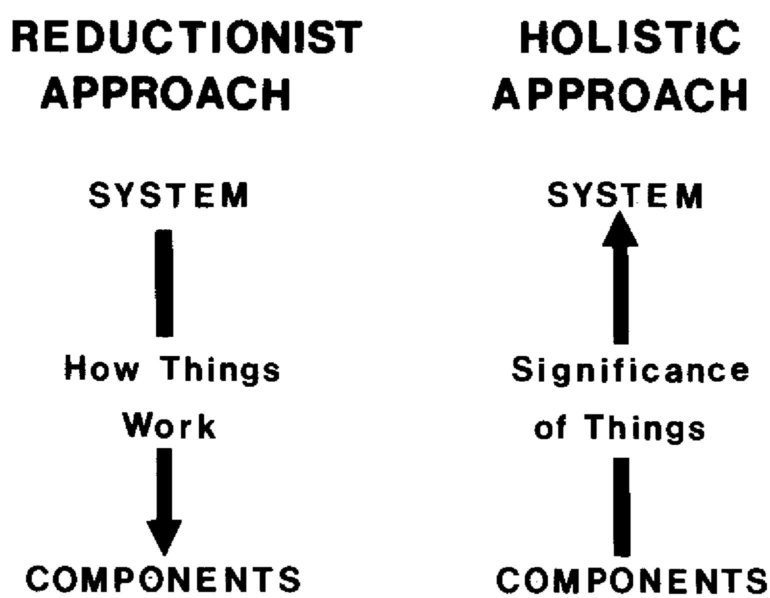

Fig. 1. A reductionist approach investigates a component of a system to determine how it functions, and the holistic approach looks at the relationship of a component to the system to determine its significance.

the skill to write the book The Silent Spring in a style understandable and appealing to the general public, who then forced scientists to take a holistic view, the resistance to the systems approach might have had delayed acceptance even longer than it did.

THE WILLED FUTURE

It is remarkable that one of the most persuasive cases for optimism about the environmental future of this planet could come from a man on his death bed (Dubos, 1982). However, Rene Dubos was an exceptional individual with a solid record in science, a flair for choosing words in a language that was not his native tongue, and an abiding and convincing belief that humans can change the course of events if their will is sufficiently strong. René Dubos was not a believer in utopias, but he was a firm believer in taking advantage of the beauty all around us. He wrote his last article entitled "A Celebration of Life" (Dubos, 1982) on a hospital bed in his 81st year and died shortly after on February 20, 1982. In the article, he wrote "I believe, as do many others, that industrial civilization will eventually collapse if we do not change our ways - but what a big *if* this is."

THE TYRANNY OF SMALL DECISIONS

Integrative environmental management very often fails because of what William Odum (1983) calls the tyranny of small decisions. These can be either organizational or environmental and, in both cases, can be made for perfectly good reasons. However, if the decisions are implemented they have secondary, often unintended, effects that are disastrous to the larger system. For example, some years ago in my own institution, staff in interdisciplinary centers could not be tenured or promoted in the centers but had to go through this process in a traditional department in one of the basic disciplines. It should not be surprising to anyone that the objectives of the center and the department were not identical and might not even overlap to a significant degree. Interdisciplinary centers at that time were not charged with teaching, and departments were. Centers were charged with getting large interdisciplinary grants, and departments were not. Although directors of centers are generally, though not entirely, senior people (usually full professors with tenure), assistant directors and participating staff are not. The job description of the assistant director is to assist the director in determining what capabilities and interests faculty members in different disciplines (in all administrative units in the university) have for working on interdisciplinary teams. This means spending time in many different departments on campus and, as a consequence, spending very little time in the particular department where the assistant director must be judged for tenure and promotion. Similarly, interdisciplinary center staff members do not have heavy teaching responsibilities and are generally funded through the research division and/or grants, and their position descriptions reflect this assignment. Nevertheless, the department that tenures and promotes them may insist on a certain base level of teaching, even though this is incompatible with the position description and the requirements of the position. In addition, success in interdisciplinary activities requires publication in journals and other outlets noted for interdisciplinary activities. These publications are often not given the same credit in a discipline-oriented department as are publications appearing in the journals dear to the hearts of the tenure and promotion committee. As a consequence, administrative staff and faculty upon which the center depends heavily for administering large and complex interdisciplinary grants may very well not be granted tenure and/or promotion by a personnel committee from a traditional department that uses a different set of criteria than those used for center operations. As a general rule, persons skilled in interdisciplinary work are highly marketable in both research-oriented academic institutions as well as consulting firms, industry, and the government. They are also ambitious and, thus, likely to leave an institution where their career advancement is blocked. They leave immediately to take a more promising position, leaving the director with the entire administration of one or more large interdisciplinary grants that cannot be possibly handled by one person. Although the higher university administrators are generally aware of these problems and sympathetic toward interdisciplinary activities, they are extremely reluctant to intervene in the department tenure and promotion decisions because they fear being accused of interfering with faculty governance. They are almost certainly correct in this judgment. Some universities have attempted to avoid this problem by forming interdisciplinary departments with what is considered to be a critical mass mixture of disciplines. This mixture is invariably both insufficient in quantity and in breadth because of the limited resources of the university and, more important, because each problem requires a different mix of disciplines. As a consequence, the array of problems that such a department can solve is limited by the disciplines available in the department rather than those available in the institution as a whole. Nevertheless, the latter option is far preferable to the earlier choice because the individuals are tenured and promoted within that system. Therefore, the system has greater stability and is more likely to complete a grant or contract without incident than the other system where centers are administratively distinct from traditional departments but where the traditional departments retain their perogatives, particularly in tenure and promotion.

The same type of "small decision" problems exist in dealing with environmental issues. One example will illustrate this point. A permit for industrial plant siting may come before a zoning board and a variety of other organizations but may not come before the group charged with maintaining water quality until it is too late or the wrong questions might be asked. For example, for a complex set of reasons, an industry may be located just above an impoundment when it could without much additional trouble or expense have been located below it. All of its waste treatment facilities represent the latest technology and, therefore, are exemplary in terms of the material being discharged. However, the impoundment traps nutrients and toxicants, and problems are more likely to occur than if the industry were located on the free-flowing river downstream of the impoundment. This has actually happened a number of times unnecessarily because the "small decision" that seemed so appropriate in the context of the limited series of perspectives was not when the entire system was examined. All the small decisions leading to such mistaken locations were in conformance with existing laws and regulations and were not strenuously opposed during any of the hearings.

Avoiding the tyranny of small decisions is not easy because the impact of each of the small decisions must be considered in the larger context of the system for which the decision is being made. Although the objective is clear, the implementation will take great effort.

PROBLEMS ASSOCIATED WITH LEVEL OF DETAIL IN PARTICIPATING DISCIPLINES

"It is the mark of an instructed mind to rest satisfied with the degree of precision which the nature of the subject permits and not to seek an exactness where only an approximation of the truth is possible." Aristotle

Most scientists and many engineers will make measurements to the fifth decimal place if they are capable of doing so. This often happens in interdisciplinary activities associated with integrative environmental management because the level of detail necessary to influence a management decision is not communicated. For example, if management is interested in determining that some environmental quality stays within the range of 6 to 8, making a measurement of 7.1234 is pointless. Decision analysis emphasizes that information has no value if it does not influence a decision. In addition, precision is of no value if the level of detail is inappropriate for the decision being made. This occurs when the difference is too minute and does not permit discrimination between or among options being selected or when differences between the options are so evident at a low level of detail that additional precision is inappropriate. Unfortunately, scientists and some engineers wish to demonstrate to colleagues in their own discipline that they know the latest methodology and are capable of making the most precise measurements that such knowledge permits. Also, the level of detail necessary is not stated, so investigators automatically do the best they can do.

Large interdisciplinary projects almost never have enough money to gather all the information that would be helpful in making a decision. As a consequence, each component (often discipline) is asked about the cost of generating a certain type of information. If the level of detail necessary for this particular decision is not communicated effectively (including how the information will be integrated with other information, the time when the information must arrive to influence the decision, and the number of replications necessary to ensure confidence in the data), the level of detail may be much greater than appropriate or the information base may be much larger than necessary. When money is lacking for gathering all the data on the shopping list, the different types of information are given priority ratings: some are usually funded at the full level, others at partial levels, and some not at all. Unfortunately, information in the first two categories may be more detailed than is necessary, and, thus, money is diverted from other sources of information that would have otherwise been funded. As a consequence, one of the primary tasks of effective integrative environmental management is communication about the level of detail necessary in each component for making an effective reliable decision.

Failure to generate the most appropriate data for the decision means that it will be less sound than it could have been. This will deter administrators and managers from using integrative environmental management in their next decision. Additionally, the project will not enhance the professional reputations of those involved as much as it might have.

HOW CAN ONE BEST GET CAUSE AND EFFECT RELATIONSHIPS BETWEEN UPLAND AND SHORELINE ACTIVITIES AND USES OF BAY RESOURCES?

It is a *sine qua non* that, in all biological systems as one progresses from subcellular through ecosystem, new properties become apparent at each higher level of organization that could not be studied adequately or perhaps even detected at lower levels of organization. Therefore, the evidence on which a decision is based should be appropriate to the level of organization at which decisions should be made. For the Chesapeake Bay, the key management decisions should be made at the ecosystem level. Clearly, the Chesapeake Bay is markedly influenced by the surrounding land mass, so the ecosystem should be viewed in its larger context. Some of the influences are beneficial, such as the dead leaves that furnish much of the energy and nutrient in headwater streams feeding into the Bay. Some are clearly detrimental, such as pesticides and other persistent toxic chemicals that drain from the land mass into the aquatic system. If the Chesapeake Bay drainage system is treated as a problem in quality control, then ecosystem boundaries might be established on the land mass from which water that eventually appears in the Bay is derived. Of course, this is a partially artificial demarcation because the land mass, the Bay itself, and its tributaries can be markedly influenced by events originating from outside this system, for example, acid rain, climatic changes due to global CO_2, and genetically-altered microorganisms transported into the system on airborne particulates. However, the establishment of a good management plan and a base line of information will eventually enable a fairly reliable estimate to be made on the effects of these stresses and their importance to the overall system. Some steps that can be taken to enhance the effectiveness of integrative environmental management follow.

1. Define the scope of the management study

Many environmental projects have been carried out without a clear understanding of the scope of the study. The scope should be determined by the purposes for which the information generated will be used. Therefore, the objectives must be clearly identified. Once the need for an environmental study is established, the first task should be to define the scope of study. Three general problem areas are especially critical at this initial stage: (1) to establish the specific objectives of the study, (2) to define the geographic area to be studied, and (3) to determine the level of study detail.

The overall objective of an environmental study should be to identify the critical or important environmental characteristic of the area so that this information can be used to make appropriate management decisions. Unfortunately, until recently, the inclusion of environmental evidence into regional management plans has been the exception rather than the rule. As a consequence, many environmental studies have been carried out as perfunctory exercises to fulfill regulatory requirements rather than to influence regional management decisions. This should definitely not be the case!

2. Establish geographic boundaries

Since the primary function of an environmental study is to provide the basic information with which to evaluate various alternative courses of action, both structural and non-structural, it is essential in water resources planning to determine the geographic boundaries of the study carefully. All too often, environmental studies are carried out on a project, site-specific basis with little or no overview of the total ecosystem involved. When this is the case, as is often true for perfunctory environmental assessments, weighing and evaluating alternatives adequately are not possible for making appropriate management decisions. Typically, the impetus for an environmental study is the declaration of a proposed action, such as the construction of a steam electric power plant, and is, therefore, likely to be highly site-specific. Although the primary destabilizing effects, if any, may well occur within the confines of a specific site, many primary and secondary effects may occur outside the site or within the site but originating from sources far distant from the point source discharge being studied. Because reasonable structural and non-structural alternative management strategies should involve the larger system, it is difficult to get the necessary information by piecing together a series of site-specific environmental studies however exemplary they may be. This is not to say that such information may not ultimately be useful, but site-specific studies cannot be relied upon heavily for managing large drainage basins.

3. Determine the appropriate level of detail

The decision regarding the depth of study required is also critical. This should be carefully thought out before the study begins and not on an *ad hoc* basis when the study is underway. Of course, modifications are usually necessary, but they should be systematic and orderly, as the entire study should be. Anyone who has been involved with environmental management realizes there is no "cookbook" standard for the level of information required to produce a suitable study. Likewise, those people responsible for environmental planning realize that a commitment of personnel, time, and money is required to obtain useful and timely environmental information that will influence the decision-making process. Naturally, the level of detail required should be governed by the overall objectives of the study and the anticipated use of the information rather than by an arbitrary and unrealistic cost limit or the disciplinary biases of the specialists.

GETTING THE BEST TALENT

Integrative environmental management is by definition "impure" because it involves persons from a number of "pure" disciplines. Most "pure" disciplines are largely artificial constructs given an illusory reality by the reductionist approach to problem solving. It is true that biologists work primarily with living things, but to consider them in isolation from their physical/chemical environment is sheer idiocy and, therefore, immediately involves other disciplines. One might think that physics and biology are quite distinct, yet physicists, such as David Gates (1980) of the University of Michigan, have produced some very interesting insights into animal behavior and plant survival by determining the energy exchange in the biosphere and developing predictive models as to what will happen to certain animals and/or plants under certain thermal regimes. To the man on the street or the average administrator, all biologists may look alike or all physicists may look alike, and many years of education and/or indoctrination are necessary to make the distinctions in subdiscipline that specialists love. The guardians of disciplinary purity have many strategems to enhance their activities. Among these are the power to approve or disapprove a candidate for the Ph.D., the ability to grant or deny tenure and promotion, and the ability to exclude articles as inappropriate from some of the most prestigious journals in the discipline. In my own institution, a land grant university with a mandate for teaching, research, and service and a motto "That we may serve," many faculty proudly identify themselves as pure or theoretical scientists in a particular discipline and may even add that they would not be tainted by doing applied work. Curiously, theoretical ecologists, who would consider a Ph.D. candidate's ignorance of the writings of Aldo Leopold unacceptable, often ignore his exortation of nearly a half century ago that "the time has come for science to busy itself with the earth itself." If one examines the titles of the papers in the most prestigious ecological journals during this same period, one can find a relatively small percentage of the articles dealing directly with restoration ecology, environmental toxicology, environmental risk analysis, and the like. Both environmental toxicology and restoration ecology require the development of predictive models and an understanding of ecosystem structure and function, all of which should be important to theoretical ecology. Yet the opportunities in these two fields have been largely overlooked and, in fact, studiously ignored in some cases by theoretical ecologists. Occasionally, a group will be convened to show how theoretical ecology may be used to solve environmental problems, but these efforts largely ignore the contributions and needs of the other disciplines, such as sanitary engineering, environmental chemistry, and other subgroups that have already made important contributions to the field of environmental management and without which it would be impossible to achieve success.

Since the very best people in any field are acutely conscious of all these factors and since the "approved" problems are as fascinating to them as the "unapproved" problems, they, not too surprisingly, generally choose the former. Those unable to compete for the increasingly scarce positions and research funding are displaced into the less desirable interdisciplinary activities, which they do reluctantly with the hope they may someday escape to pure science where their hearts lie. A few contrarians find ways of ignoring the system or minimizing its effects upon their careers and cheerfully engage in interdisciplinary activities. In academic institutions, these individuals are usually offered some degree of protection in the form of institutes and centers. A few renowned scientists turn to interdisciplinary activities late in their careers when their credentials in the pure and theoretical aspects of a discipline are so notable that they cannot be tarnished seriously by the new activity. Since the interdisciplinary fields depend strongly on hypotheses and concepts developed in the disciplines and the disciplines can validate these to a much greater extent than has hithertofore been possible in the applied fields, it is a pity that the relationship between them is not more harmonious.

Curiously, the situation is quite different in medicine (and certain other fields) where the commitment to human well being is universally recognized as a central issue. A cure for AIDS is unlikely to be patronizingly referred to as "applied medicine."

Many cultures have had science and philosophy flourish and then decline. Among the factors contributing to this decline seem to be the loss of touch with the world and society and becoming increasingly preoccupied with abstractions. Since the abstractions are only understood by a few, the economic base supporting academia is eroded, and the intellectual vigor diminished with great rapidity. I have often heard theoretical ecologists bemoan the fact that no one pays attention to their pronouncements, but this is probably because they are difficult to relate to what is occurring in other disciplines and to the needs of society as a whole. Integrative environmental management offers theoreticians in a variety of disciplines the opportunity for investigations that will validate many of their predictive models, enable the development of new models, and, most importantly, maintain communication with society regarding the value and importance of their activities to society as a whole. Getting the most promising scientists to engage in integrative resource management in the numbers that will be required will require some fundamental changes in academic institutions and the reward system in the various professions before any significant advance can be made.

I have used ecology frequently in this discussion because it purportedly examines all components of an ecological system, such as the Chesapeake Bay. Unfortunately, industrial and agricultural components are usually arbitarily excluded from the theoretical or pure papers as unnatural or applied, despite the enormous impact they have on natural systems. In a presidential address to the British Ecological Society, Anthony Bradshaw, a plant ecologist at the University of Liverpool, suggested that the abilty to restore a disturbed ecosystem was "the acid test" of understanding that system. This would place restoration ecology, generally thought of as applied or practical, at the theoretical center of the field because it would be a crucial test of the predictive models and paradigms of ecology. In this context, restoring a damaged ecosystem is the ultimate validation of theoretical ecology. The uncertainty of the outcome and the possibility of undermining many cherished beliefs of theoretical ecologists may be as much of a deterrent to entering the field of restoration ecology as the loss of "face" in doing applied work. Bradshaw has placed the challenge before British ecologists, who certainly cannot claim to have an abundance of pristine ecosystems for their research, and it will be interesting to see how they respond. The same challenge would be equally valid in the People's Republic of China where ecosystems have been altered for many thousands of years and even in the United States where major alterations have occurred only within the last few hundred years.

Welsh ecologist John Harper's recent essay entitled "Beyond Description" (1982) has as its central theme the fact that theoretical ecology has tended to be highly descriptive in nature and has so far made little progress as a rigorous experimental and predictive science. There are, of course, some notable exceptions to this, such as the works of the late Robert MacArthur. However, in general, evidence for the truth of the charge can be obtained by reading any of the issues of some of the prestigious journals in the field of ecology. Harper argues that many of the basic premises of ecology, such as the concept of species, are actually ecologically inadequate. The concept of species that ecologists customarily use in preparing an inventory of ecological communities is primarily a structural one developed by taxonomists interested in variations in the structural details of plants and animals rather than in the less physically conspicuous variations in physiological or ecological function. In short, Harper claims that the conceptual basis of ecology may be inadequate and will remain so as long as these weaknesses are not strikingly evident, as will be the case if ecological research remains primarily descriptive. Such weaknesses will become abundantly clear when the concepts and predictions are tested by experiments involving entire ecosystems. Harper admits that the complexity of ecological communities may discourage experiments designed to test specific, carefully formulated hypotheses but that one key to resolving this problem is by beginning with relatively simple perhaps artificial systems (underlining mine). In short, this is already being contemplated and carried out by the field of restoration ecology (e.g., Brooks et al., 1985). Bradshaw and Chadwick (1980) have argued that the restoration of a disturbed ecosystem is, in fact, a great intellectual challenge and an opportunity for ecologists.

I am convinced that large ecological systems, such as the Chesapeake Bay, cannot be adequately managed and protected unless there is better integration of information than now exists. This includes examining the system level attributes of the larger Chesapeake Bay ecosystem, including the adjacent land mass. Some profound changes in attitudes toward interdisciplinary study must occur in order for this effort to succeed.

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KEY WORDS

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resource allocation

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NONPOINT SOURCES AND THEIR IMPACT

by

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ABSTRACT

It is not possible to establish conclusively that application of the best management practices to the nonpoint sources will arrest the decline and significantly improve the quality of the Chesapeake Bay ecosystem. A modeling effort, which can take into account the distribution of nutrient, runoff and sediment contributing areas as well as zones of erosion and deposition appears to be needed to predict with a greater degree of accuracy potential nutrient losses to the Bay. Separate studies now comprising the Chesapeake Bay Program may benefit from a unifying geostatistical analysis of available data. The analysis should attempt to account for field observed spatial and temporal variability components and their effect on nutrient loads.

INTRODUCTION

Chesapeake--the great river in which fish with hard shell abound (Michener, 1978)--is rather young on a geological time scale. The most recent retreat of glaciers and melting of ice in late Pleistocene has resulted in a corresponding rise in sea level that submerged what used to be the Susquehanna River Valley along with many of its tributaries and created the Bay as we know it today. As time went on, shoreline erosion, sediment and nutrients transported to the Bay from surrounding areas by streams and rivers (Figure 1), along with wave and tidal action, have shaped a uniquely fertile and dynamic estuarine ecosystem with diverse biological communities, abundant crab and oyster beds and large schools of fish. The pristine beauty of the Bay, mild climate, and abundance of natural resources have long attracted man. The original settlers were gradually replaced by modern man, by his towns, cities and large industrialized areas. As forests and grass covered clearings gave way to concrete urban centers, suburban developments, and tracts of intensively cultivated farmland, changes began to appear. Milk, poultry and beef production increased. New factories were built, commercial fisheries were launched, and, large harbors were dredged. As exploitation of natural resources prevailed, the great estuarine ecosystem began to deteriorate. The sediment and nutrient contributions from nonpoint sources increased and significant contributions from point sources appeared. Delicate ecological balance which took thousands of years to establish was changed rapidly in less than 300 years by man.

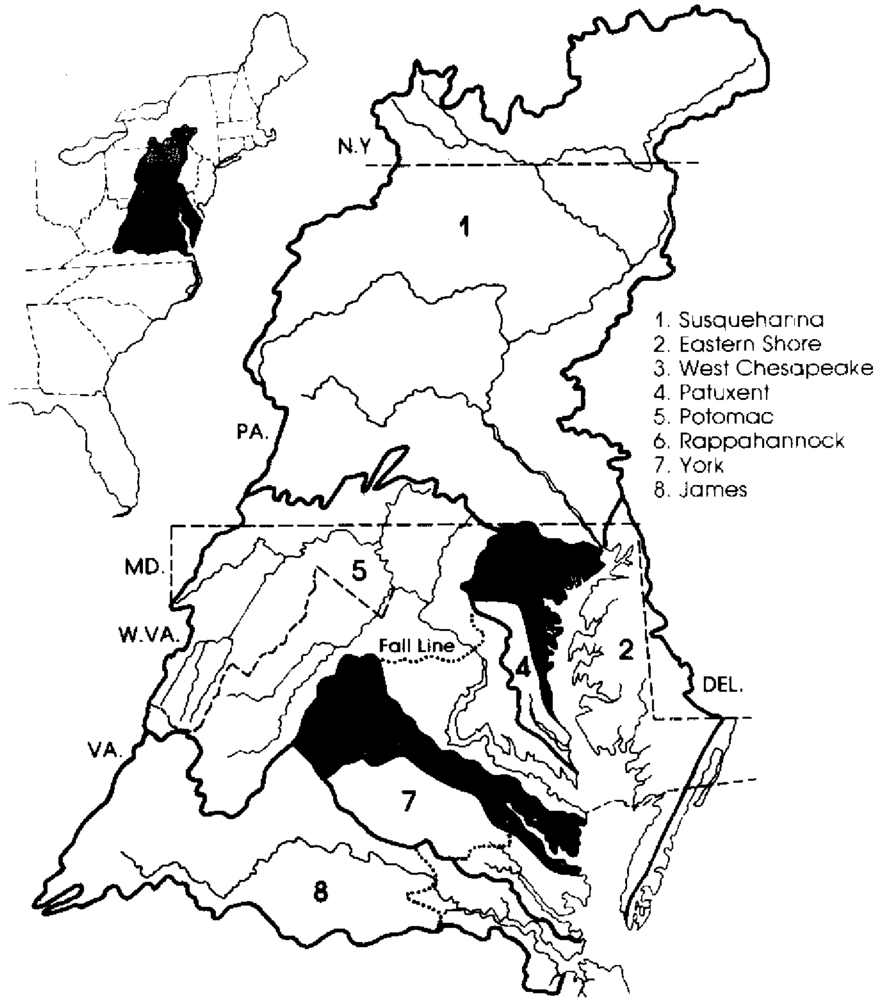

Figure 1. The Chesapeake Bay drainage basin (from EPA, 1983b, their Figure 2).

Much of the blame for deteriorating quality of the Bay is placed on nonpoint source pollution. In the context of the Chesapeake Bay, nonpoint sources are those diffuse usually agricultural, forest, or even urban sources that contribute sediment and nutrients (primarily N and P) to the Bay (Gillean and Macknis, 1983).

Declining commercial productivity and deteriorating crab, oyster, fish and submerged aquatic vegetation (SAV) habitats of the Bay have brought about public outcry and resulted in congressional funding of the USEPA Chesapeake Bay Program (CBP) in 1975. The technical part of the program was started in 1976 and its objective was to identify major environmental problems of the Bay. Concurrently a study on environmental management was initiated to assess available mechanisms and propose alternate control strategies. By choosing what appeared as the three most critical areas--nutrient enrichment, toxic substances and decline of submerged aquatic vegetation (SAV)---for an initial research thrust, the CBP study became committed to certain specific and not totally unexpected courses of action and conclusions.

Numerous research and survey reports have now been prepared. They contain the methodology, findings and recommendations of the CBP study, a result of nearly 40 research projects, grants and cooperative agreements between EPA and major scientific institutions in the Bay region. These technical studies have deepened the knowledge of the Bay and provided a basis for recommended management strategies. One of these strategies is an examination of control programs both present and future using a predictive nonpoint source (NPS) computer model (Hartigan et al., 1983). The model utilizes nutrient loading factors obtained by calibrating it on 11 small fields and watersheds ranging in size from 2 to 60 ha. These nutrient loading factors attempted to correlate N and P losses from uniform land use areas with runoff, with sediment transport based on the Universal Soil Loss Equation (Wischmeier and Smith, 1978), and with perceived nutrient loadings within study watersheds which were monitored for 1 to 1.5 years. The loading factors thus derived were incorporated into the basin (16,600,000 ha) model of the Bay area. There is a danger in attaching significance to any one year of data because of natural variability and the wisdom of this approach has been questioned (Schnabel and Gburek, 1985) on scientific grounds. Nevertheless, the CBP study pattern, primary conclusions and principal recommendations compare favorably with similar studies in other areas (i.e., Chesters et al., 1979; Sullivan et al., 1980a, 1980b).

Primary recommendations, based on available data from the CBP and on the simulation results with NPS model, include control of point and nonpoint pollution from urban and agricultural areas, monitoring, research, and a coordinated future Bay wide management program (EPA, 1983). To control point and nonpoint source pollution, basin wide implementation of effluent controls and varying levels of best management practices (BMP) are recommended. Nonpoint agricultural and urban sources and point industry sources are targeted for immediate remedial action under the assumption that such action will restore the quality of the Bay. Even though it is tempting to agree that control of effluent quality and implementation of the BMP will solve the Bay's problems, we should pause and reflect on the findings and then consider them in a broader context of what is currently known. Since it is not possible to cover every aspect of the comprehensive and impressive Chesapeake Bay study program I will primarily endeavor here to look at a few specific topics, hoping to add something to the understanding of environmental problems confronting the Chesapeake Bay region.

SOME UNRESOLVED QUESTIONS

Consideration of the Chesapeake Bay Program immediately raises these questions:

- Why does the rate of nutrient enrichment (EPA, 1983, p. 22) continue to increase despite decline in cropland, increase in forest and use of no-till and other conservation practices by a growing number of farmers (EPA, 1983, p. 15)?

- If nutrient loads are a leading cause of algal blooms and contribute to the decline of submerged aquatic vegetation (Mackiernan et al, 1983, p. 27-28), why is the peak bloom production (Taft, 1983, p. 126-133) out of phase with nutrient load maxima?
- To what extent should the regional impact of acid precipitation (Cowling, 1983) be evaluated?
- To what extent a more sophisticated modeling approach might be helpful? Should not the consideration of primary contributing areas, erosion and deposition mechanics, scale effects, as well as spatial and temporal variability be included in simulation of non-point source pollution (Hartigan et al., 1983)?
- How can point measurements and local estimates of soil characteristics be credibly extended to the whole drainage basin (Macknis et al., 1983a)?

NUTRIENT LOADING

Control Strategies

Between 1950 and 1980 land used for cropland and pasture in the Chesapeake Bay region has declined while forested and urban areas have increased (Mackiernan et al., 1983), concurrently more farmers begun to use no-till, or other SCS recommended conservation practices. Despite these trends, nutrient loadings of N and P to the Bay are on the rise. Table 1 indicates average amounts of different N and P forms contributed to the Bay by atmospheric, fluvial, point and benthic sources. If we assume that the increase in loadings cannot be ascribed solely to the growth of urban areas, the question naturally arises as to how effective will the use of BMP on cropland be in decreasing nutrient loads. This point is illustrated in Table 2 which shows simulated percent change in N and P inputs to the Bay, relative to existing conditions and subject to implementation of three control strategies in three representative river basins. Control strategies Level 2 and 3 pertain to nonpoint sources and control strategies TP1, TP2 and TN6 apply to point sources. While considerable reduction in nutrient loads is possible in James and Potomac River Basins which have large point source inputs, control of nonpoint pollution on Susquehanna even when using Level 2 and Level 3 BMP appears at best marginal particularly with respect to nitrogen. Therefore, it seems doubtful that nutrient loadings to the Bay will substantially decrease in the near future.

Based on large scale simulation results, Gianessi and Peskin (1981) and (Gianessi et al., 1981) have shown that even if point discharges are subject to secondary treatment and best control technology currently available, while nonpoint discharges of sediment and associated nutrients are reduced to zero, a substantial number (60 to 70 percent) of fluvial sources in a region such as the Chesapeake Bay may still equal or exceed baseline (1972) concentrations of N and P.

Table 1. Average annual nutrient and fluvial sediment input[1] to the Chesapeake Bay.

Constituent	Atmospheric Sources	Fluvial Sources	Point Sources	Benthic Sources	Total
	metric tons x 10^2				
Total Nitrogen-N	183	808	236	140	1373
Nitrite + Nitrate-N	66	506	78		650
Ammonia-N	40	41	123	146	351
Total Phosphate-P	7	47	49	34	137
Orthophosphate-P	2	15	31	34	81
Sediment		30,073			30,073

[1]From Smullen et al. (1982) their Table VIII.1(a), p. 232.

Table 2. Simulated effect of selected control strategies[1] on existing N and P loads carried to the Bay by Susquehanna, Potomac and James Rivers.

Control Strategies	Susquehanna		Potomac		James	
	P	N	P	N	P	N
	Percent change from existing (1982)					
a. Level 2	-16	- 1	- 4	- 1	- 1	0
b. TP2 + Level 2	-29	+18	-17	+ 2	-45	0
c. Level 2 + Level 3[2]	-22	- 5				
d. TP1, TN6	-17	+12	-22	-21	-55	-30

[1]Control strategies (Macknis et al., 1983b, Table 36, p. 167).

Level 2 = Conservation tillage, i.e., minimum or no-tillage, nonpoint sources.

Level 3 = Conservation practices, i.e., contour farming, strip-cropping, nonpoint sources.

TP2 = P-limitation (2 mg L^{-1}), point sources.

TP1,TN6 = P-limitation (1 mg L^{-1}) and N-limitation (6 mg L^{-1}), point sources.

[2]Level 2 in upper Susquehanna, Level 3 in Lower Susquehanna (Macknis et al., 1983b, Figure 2, p. 136).

Uncertainty as to what nutrient levels can be tolerated and what levels should be considered critical from the standpoint of excessive phytoplankton production and decline of SAV complicates the situation in the Bay region. While Voinov and Svirezhev (1984) have shown, using a modeling approach, that total amount of substances in a reservoir (i.e., phytoplankton + detritus + nutrients) is an important ecosystem eutrophication control parameter similar approach does not appear to have been tried in the Chesapeake Bay study.

Nurtrient Enrichment

Nutrient enrichment in the Bay is usually linked directly to algal blooms and is assumed to contribute to the decline in the SAV. However, maximum fluvial nutrient loads occur in the Winter and Spring (Table 3), while the peak of phytoplankton production is in the Summer (Smullen et al., 1982). Thus a direct relationship between phytoplankton productivity and nutrient concentration particularly nutrient concentration derived from nonpoint sources is difficult to demonstrate. Highest phytoplankton production in the Summer coincides with lowest concentrations of fluvially transported nutrients. Although computations of primary productivity of the Bay (Smullen et al., 1982) assume continuous nutrient recycling from detritus, and ready availability of N and P to phytoplankton, most nutrients (particularly P) are either attached to sediment particles or enter the Bay sediments rather quickly. Moreover, nutrient inputs into the Bay from atmospheric and fluvial sources vary greatly in space and time, while the most readily available forms (ammonia-N and orthophosphate-P) originate in the peak Summer months primarly from benthic and point sources (Table 3). Under these circumstances control of nonpoint inputs from Susquehanna may have little effect on peak algal blooms.

Table 3. Average nutrient loads of Ammonia-N, Orthophosphate-P, Nitrate and Nitrite-N, and Total Phosphate-P delivered to the Bay by atmospheric, fluvial, point and benthic sources during Winter, Summer and Fall (Smullen et al., 1982).

Source	Nitrogen				Phosphorus			
	Winter	Spring	Summer	Fall	Winter	Spring	Summer	Fall
	metric tons x 10^2							
	Ammonia-N				Orthophosphate-P			
Atmospheric	9	16	9	10	0.4	0.5	0.5	0.5
Fluvial	12	17	6	6	4	6	2	2
Point	30	31	31	31	8	8	8	8
Benthic	36	31	41	38	-	-	34	-
	Nitrate-Nitrite-N				Total-P			
Atmospheric	13	21	21	10	1	2	3	1
Fluvial	146	206	72	80	14	20	6	7
Point	19	20	20	20	12	12	12	12
Benthic	-	-	-	-	-	-	34	-

Phosphorus

Phosphorus occurs naturally in the Chesapeake Bay basin ecosystem, and there is a background concentration. It is an essential plant nutrient and is added to the system through a variety of activities both agricultural and other. In general, cropland contributes most (60 to 85%) of the fluvially transported phosphorus, with about 7/10 of it in adsorbed form on sediments and 3/10 in solution (ortho- and organic-P). Since the principal loss pathway from a water column is through sedimentation about 3/4 of the phosphorus is permanently lost in benthic deposits. Thus, of the phosphorus entering the Bay from cropland only 15 to 20% may be considered available, and of this amount on the average, only about 1/3 is water soluble and algae available. Hence, algae available P derived from cropland on a year round basis may be estimated at 5 to 7%, and on a summer alone basis at less than 1%. Unfortunately a large contribution from benthic sources under anaerobic Summer conditions (Table 3) appears to provide soluble P for phytoplankton use. Even that amount however is relatively minor compared to P seasonally recycled from organic forms. In attempting to control nonpoint source contributions of soluble P the situation appears analogous to controlling the weight of the ship by dieting the captain. Most phosphorus in the natural systems is neither soluble, nor plant available, nor algae available. It is associated with organic and mineral fractions of the soil, is only very slightly soluble,and even if present in desorbable form, attached to soil particles and subject to loss primarily through runoff and erosion. Since the soil phosphorus is very highly buffered, any withdrawals by plants, or other means, will be rapidly replenished by phosphorus conversion from less to more available forms. If indeed algal blooms result from excess P derived under anaerobic conditions from benthic sources in the Bay there seems little hope that benthic phosphorus pool will ever be sufficiently depleted to afford a measure of control. Thus any management practices aimed at controlling phosphorus loss at the field scale, while good in themselves from the standpoint of economics and agricultural productivity, are unlikely to affect the total phosphorus balance in the Bay to any great extent.

Nitrogen

To place nitrogen contribution to the Bay from nonpoint sources in proper perspective we need to remember, that most of the soil nitrogen (95%) is in an organic form derived from plant and animal tissue, while only about 5% exists as nonexchangeable NH_4^+ and small amounts of NH_4^+, NO_2^- and NO_3^- in soil solution (Smith, 1982). Large part of the organic N has not yet been fully identified, and there is much uncertainty in scientific community concerning the process of mineralization. While deamination does release NH_4^+ to soil solution there is an increasing evidence that some organic N becomes adsorbed on clay and is resistant to further microbial attack (Harter and Stotzky, 1971; Sóresen, 1972). The NH_4^+ in soil solution can be taken up by plants and microorganisms, volatilized under alkaline conditions as NH_3 and re-adsorbed on soil colloids. It can be slowly leached by percolating waters and increased by atmospheric inputs. Bulk of it is oxidized to NO_2^- and subsequently to NO_3^- by microbial nitrifiers. A resulting nitrate load in solution is highly mobile and portions of it may be leached, lost as gas by denitrification, or taken up by plants and microbes and eventually returned to the organic pool. Soil pH, soil porosity and moisture status as well as ambient temperature play a significant part in transformation and release of organic nitrogen as NH_4^+, NH_3, NO_2^-, NO_3^- or N to the environment. Specifically, which pathway is used depends to an extent on seasonal climate changes, timing of rain and runoff events as well as management practices employed. For example, winter application of untreated wastes (manure) to frozen ground may be potentially more damaging then if the manure was incorporated into the soil. Although nonpoint pollution of the Chesapeake Bay from nonpoint agricultural sources is primarily in the form of nitrite-nitrate N, it appears that phytoplankton will preferentially use ammonia, so long as threshold amounts of 1 to 1.5 μg at/L (18-27 μg/L) are present. Much of the spring nitrite-nitrate load entering the upper Bay from Susquehanna passes unused to the lower Bay and similar situations appear to exist in other tributary estuaries (Taft et al., 1978). However, even in the summer fluvial inputs of ammonia to the Bay by the three major tributaries are in excess of the threshold value. Smullen et al. (1982) for example, list the mean daily Summer nutrient loads. Using their data[1] ammonia load from Susquehanna would be on the order of 80 μg/L, values from Potomac are 47 μg/L and for James River are 38 μg/L. Accordingly, we will primarily concern ourselves here with the potential mechanisms of NH_4^+ input to the Bay from nonpoint sources.

Illite and vermiculite clay minerals strongly bind NH_4^+ in the interlayer of the 2:1 clay lattice (Mengel, 1985). Although sometimes referred to as fixed, this NH_4^+ fraction is nevertheless slowly available. Profile depletion patterns suggest that (Mengel, 1985) deeper soil layers release NH_4^+ while the amount of fixed NH_4^+ in the topmost layer remains essentially unchanged due to rapid resupply of depleted reserves from mineralization occurring at the surface. Since the surface layer is particularly prone to erosion during Spring and Summer, substantial quantities of NH_4^+-N may move off the field attached to sediment. Under natural conditions and favorable temperature the two forms of ammonia exist in equilibrium,

$$NH_3 + H_2O \rightleftarrows \overset{+}{NH_4} + OH \qquad (1)$$

controlled by the pH of solution (Loehr, 1984). At pH above 7 there would be increasingly more NH_3 in solution and availability to phytoplankton and loss of NH_3 by volatilization would be high, while at lower pH fixation in the NH_4^+ form is more likely. Under acid conditions and on soils with low cation exchange capacity containing vermiculite and illite clay minerals, fixation could become significant, and the proportion (F) that is in un-ionized form (NH_3) may be written (according to Loehr, 1984, p. 344) as,

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[1]Smullen et al. (1982), Table III, p. 179.

$$F = \frac{1}{1+10^{(pK_a - pH)}} \tag{2}$$

where pK_a is the negative logarithm of the ionization constant in equation (1). While pK_a varies as a function of temperature, equation (2) describes how the relative amount of NH_3 changes as the pH is raised.

Soils of the Chesapeake Bay region (Figure 2) consist primarily of Inceptisols (Dystrochrepts) in the Susquehanna River Basin and Ultisols (Hapludults) in the lower Susquehanna, Potomac and James drainages. Under natural conditions these soils are generally acid, low in exchangeable bases and subject to acid precipitation. Consequently, a general shift towards NH_4^+ may be expected and increasing amounts of NH_4^+ could become fixed on sediments and transported to the Bay by runoff and erosion. In the Bay however, a

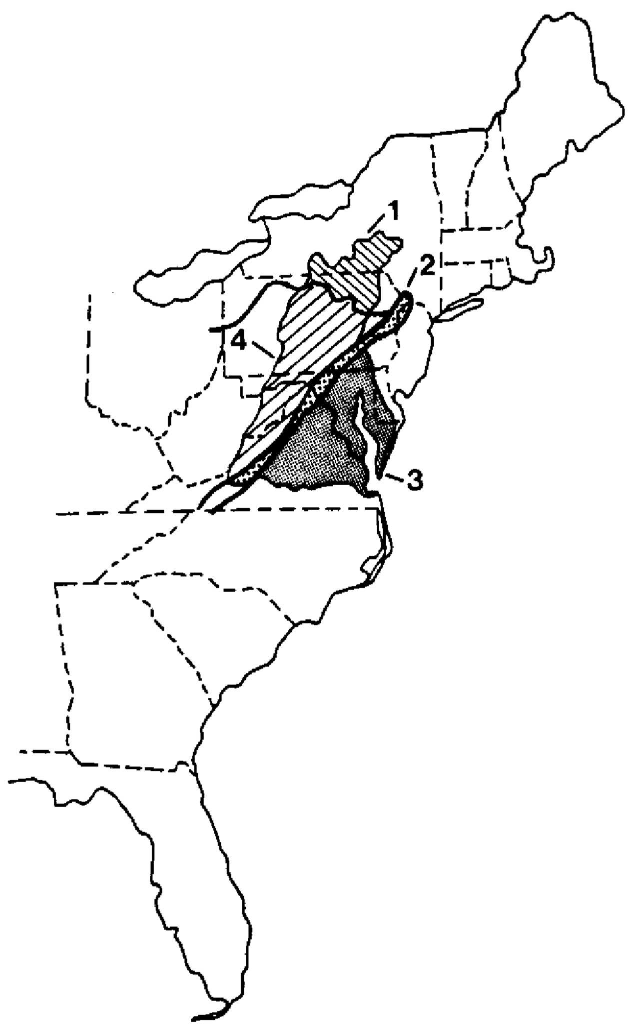

Figure 2. Soils of the Chesapeake Bay area.

pH shift towards neutral or alkaline may favor a release of NH_3, particularly during the warm summer months. Although cell membranes are relatively impermeable to NH_4^+, NH_3 can pass through quite easily (Loehr, 1985). It should therefore be the form that is preferentially algae available.

Typically soils used for crops are more highly buffered, limed, and fertilized than other areas so above effects might not be as pronounced. Best management practices primarily designed to control erosion may not always be successful, since the loss of topmost layer of soil containing most of the adsorbed NH_4^+ may amount to only a fraction of the prescribed tolerance level. Such losses would not be limited to cropland alone but would apply to other land uses wherever the mineralization of organic matter occurs, consequently the NH_4^+ contributing areas may cover much of the Chesapeake Bay drainage basin.

Primary Productivity

Amounts of N and P required to support primary biomass productivity of the Bay are shown on the first two lines of Table 4. The amounts on line 1 were calculated directly from Smullen et al. (1982) based on the primary productivity of 492 g C/m^2/yr and Redfield ratio of 106:16:1 (C:N:P). The values in brackets are based on the primary productivity of 200 g C/m^2/yr estimated for the Bay from Lieth (1975), Bunt (1975), and Likens (1975). In comparison the values given by Smullen et al. (1982) appear somewhat excessive. In general, productivity of swamps and marshes is on the order of 1500 g C/m^2/yr, that of lakes and rivers is approximately 200 g C/m^2/yr and that of marine ecosystems is between 50 and 300 g C/m^2/yr. It seems therefore unlikely that the primary productivity of the Bay could be as high as 492 g C/m^2/yr. As a consequence, contributions by recycling (bottom line in Table 4) appear excessive.

The concept of nutrient cycling has been applied to the Bay in an attempt to explain the apparent large primary productivity demand for N and P. According to this concept N and P are converted on different time scales, from inorganic to living organic forms and back again to inorganic nutrient pool. Short-term recycling is assumed to operate in water column and surface of sediments on a scale of minutes to weeks, while long-term recycling of nutrients from deep sediments is expected to operate on a scale of months to years (Taft, 1983). Thus the impact of individual nonpoint source nutrient inputs, particularly inputs from major flood events, is multiplied several-fold and its effects are felt for a long time. It is tempting to speculate how much of the organic-N transported to the Bay by streams and rivers, enters this recycling pool. Certainly much of the winter applied manure could be in this category as well as assortment of organic detritus derived from no-till, urban and forest areas.

Thus, even if the primary productivity of the Bay is closer to 200 g C/m^2/yr contributions by recycling (in brackets), would still be larger than those derived from nonpoint sources and implementation of controls on nonpoint loadings might not affect immediate outcome significantly. Under these circumstances the long-term response of

Table 4. Relationship between seasonal phytoplankton productivity and nutrient input into the Bay adapted from Smullen et al. (1982), p. 228-231.

	Total N				Total P			
	W	S	Su	F	W	S	Su	F
	metric tons x 10^2							
Required[1]	499 (208)	997 (416)	2254 (943)	1256 (524)	68 (28)	136 (57)	308 (128)	172 (71)
Atmospheric	28	73	55	28	1	2	3	1
Fluvial	233	327	114	127	14	19	6	7
Point	58	59	59	59	12	12	12	12
Benthic	36	28	41	38	-	-	34	-
In Water	83	83	103	95	2	2	23	2
Total[2]	434	570	370	342	30	37	78	24
Recycled	65 -	408 -	1883 (573)	914 (182)	38 -	99 (20)	230 (50)	149 (47)

[1]Required to support primary biomass productivity of phytoplankton, values in brackets are based on Lieth, 1975; Bunt, 1975; and Likens, 1975 estimates.

[2]Includes net flux at the mouth.

the Bay would need to be considered. Small changes in loading may be important. Recent calculations based on a model of anoxia (Officer et al., 1983) suggest that a 3 percent reduction in annual N load and 11 percent reduction in annual P load from Susquehanna River could remedy the anoxia problem in the Bay, and result in lower benthic contributions. It is doubtful that a simple goal of reducing average annual loads will alone solve the Bay's problems. More attention needs to be paid to individual contributions by large events and how best to control them.

Acid Precipitation

In recent years the acidity of precipitation has increased sharply over much of the Chesapeake Bay drainage basin. Large quantities of sulfur and nitrogen oxides that are emitted into the atmosphere by burning of fossil fuels and smelting of sulfide ores are converted into strong acids which dissociate completely in aqueous solutions and lower the pH of rain to 4.2 (typical winter) and 3.5 (typical summer) (Likens et al., 1979). Since the sharp decline in pH coincides approximately with the accelerated deterioration of the Bay, it is not unreasonable to assume that the two phenomena might be related.

We have suggested earlier the likelihood of an increased NH_4^+ fixation by clay minerals under more acid surface soil conditions. A subsequent shift in pH to alkaline when sediment carried by rivers enters the Bay may lead to release of increasing amounts of ammonia preferentially utilized by the phytoplankton. Because precipitation when it falls interacts initially with the shallow topmost layer of soils usually rich in organic matter impacts of acid rain could be widespread.

The effects of acid precipitation on soils in general are varied and very sensitive to changes in local conditions. Typically, on highly buffered, limed cropland they would be minimal. However, it could be argued that not all soils are adequately buffered and diverse impacts may apply to abandoned or poorly farmed cropland, to forests and to urban areas. One impact that may have long-term effects on the ecosystem as a whole concerns changes in productivity brought about by acid rain. Consequently, not only nutrient losses need to be monitored but also changes in the site productivity index (Loucks, 1984; Rogowski, 1985) should be evaluated. Such changes may offer an insight into Chesapeake Bay ecosystem productivity and expose potential long-term effects of impacts such as nutrient stripping and incipient aluminum toxicity within the Bay drainage.

Impacts of atmospheric deposition on poorly buffered surface waters are well documented for portions of Scandinavia and Northeastern U.S. Atmospheric deposition in marshy areas can account for as much as 95% of the NH_4-N and 83% of PO_4-P load over a body of water (Flora and Rosendahl, 1982), however the effects of this loading will not always be apparent in the chemistry of the surface waters when natural nutrient concentrations are low, uptake by plants is rapid, and organic immobilization in the sediments is significant. How well buffered are the marshy coastline areas of the Bay? Nutrient concentrations certainly may at times be low and sediment entrapment is frequently a factor.

At this time it is difficult to say exactly how much is the acid precipitation contributing to the Chesapeake Bay problems. Because of the widespread distribution of acid rain throughout Bay's drainage area the contribution potentially could be significant now, or become significant in the future. To date however, no studies relate the impact of acid precipitation to the decline in the Bay productivity.

CONTRIBUTING AREAS

Although true extent of nutrient loading from nonpoint sources and its relation to primary biomass productivity, anoxia and acid rain remain unresolved, there are other aspects of the problem that merit attention. This is particularly true where nonpoint sources and their interaction with the BMP is concerned. The best management practices (BMP) were designed primarily for onsite control of nutrient and sediment losses from agricultural fields and are generally subject to a soil loss tolerance (T) values applicable to specific soil type, cover conditions and area. Natural background levels of soil loss for the United States have been estimated to range from 0.7 to 1.6 metric tons/ha/yr (Wischmeier, 1976). In general, sediment yield from agricultural land uses is variable and ranges from 1 to 40 metric tons/ha/yr. Well managed rangeland approximates natural background, while on cropland degree of ground cover, tillage methods, and the extent of conservation practices will affect the amount of soil detachment and sediment yield under normal climatic conditions (Olness et al., 1975). Conservation practices are designed to be effective for 1, 10, 50 or even 100 year storm. However, even under best management conditions fields are subject to certain minimal soil loss levels which can change drastically for major floods or hurricane size events. It is these infrequent catastrophic occurrences that can supply large and potentially critical loads of nutrients (Alberts et al., 1978) to the Bay to be stored in benthic sources for future release by recycling.

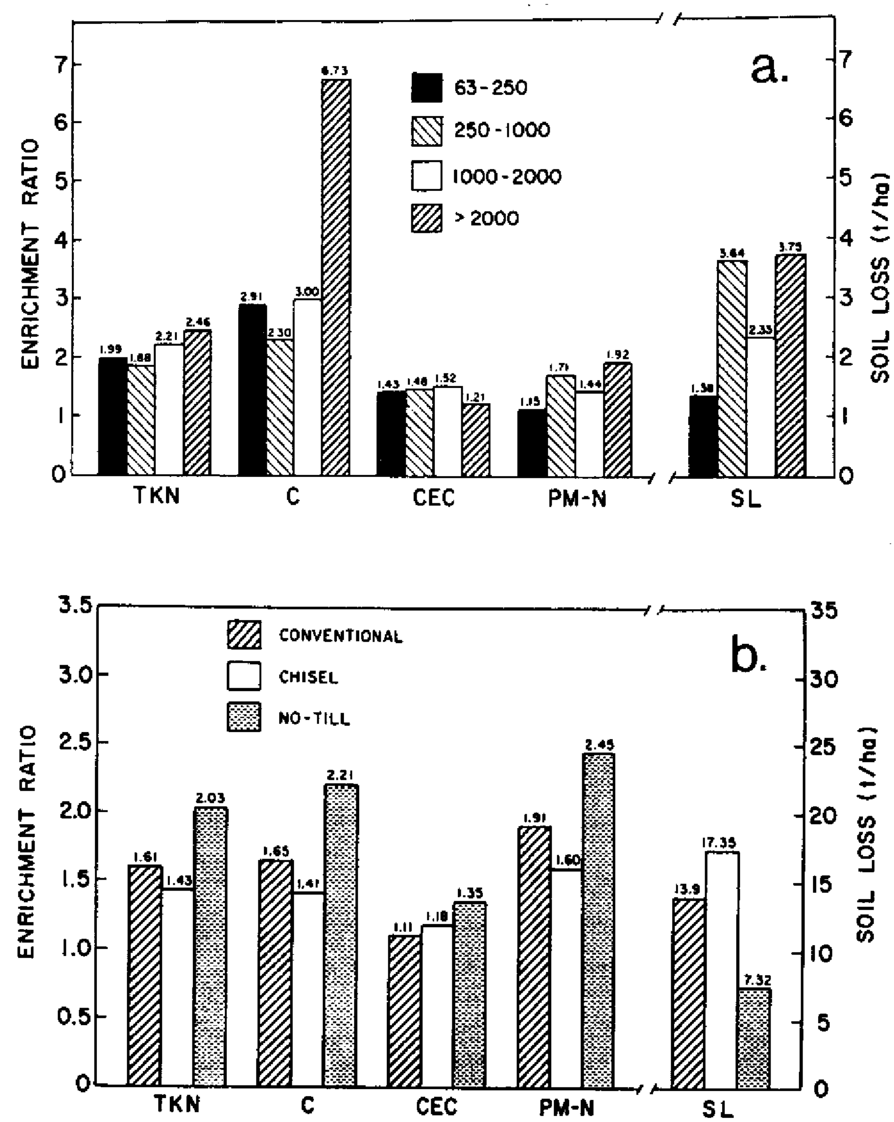

Figure 3. (a) Enrichment ratios for various nutrients (TKN = total Kjeldahl N, C = organic carbon, CEC = cation exchange capacity, PM-N = potentially mineralizable N and SL = soil loss) in four size classes of eroded aggregates from Lexington si.l. (from Young et al., 1985, their Figure 6); (b) enrichment ratios of various nutrients (TKN = total Kjeldahl N, C = organic carbon, CEC = cation exchange capacity, PM-N = potentially mineralizable N and SL = soil loss) for three tillage practices (row crops) on Russell si.l. (from Young et al., 1985, their Figure 6).

On a watershed scale, cycles of erosion and deposition and nutrient relocation with depth and over an area continue throughout the year subject to landscape topography, underlying geology, extent of cover as well as climatic and man-made constraints such as amount and rate of rainfall, timing of ground disturbance, antecedent moisture, or fertilizer application. Occasional major storms may flush out accumulated sediment and nutrients into streams draining the area. If the event is extensive, much stream trapped sediment and associated nutrients may reach a major tributary and enter the Bay. Simulations in use today rely primarily on "average" conditions and on management practices which are adjusted to normal tolerance levels at the field site. Consequently, such models cannot give a true picture of actual conditions and cannot predict cause-and-effect related inputs to the Bay area much less infer their timing. To do that we need to incorporate sediment and nutrient tracking models and optimizing techniques into computer simulations. Such models would then be capable of predicting not only the location of contributing areas on a watershed but also the circumstances (1, 10, 50 or 100 year storm) under which they may impact the Bay. A modest start in that direction has been made. Studies of erosion impact on agricultural productivity (Young et al., 1985) suggest that enrichment ratios vary between nutrients, as well as between sizes of eroded aggregates and particles found in the sediment (Figure 3a). Consequently, reductions in soil loss attributable to a given conservation practice will not necessarily result in comparable reduction in nutrient losses. For example, the same study showed that nutrient enrichment was greatest in no-till areas compared to chisel and conventional tillage fields even though the opposite was true for soil loss values (Figure 3b).

Landscape topography likewise plays a key role in nonpoint contributions to runoff and erosion. Perrens et al. (1984) suggest a likelihood of extensive zones of soil deposition on downslope sectors of concave profiles (Figure 4), while Zaskavsky and Rogowski (1969) have shown that these areas may also be the primary zones of streamline convergence and seepage (Figure 5) particularly if a shallow, less permeable layer underlies topsoil. Such zones have been referred to (Gburek and Pionke, 1983) as runoff contributing areas, and have also been delineated as zones where runoff is most likely to start (Khanbilvardi and Rogowski, 1984). Consequently, it seems likely that at least in the Chesapeake Bay region these also may be the primary zones which contribute to nutrient enrichment and sediment load in streams. Since these areas are often also the zones of seepage where shallow nutrient enriched (NO_3) groundwater emerges on the land surface, their identification and special management may therefore merit attention from the standpoint of nonpoint source pollution control.

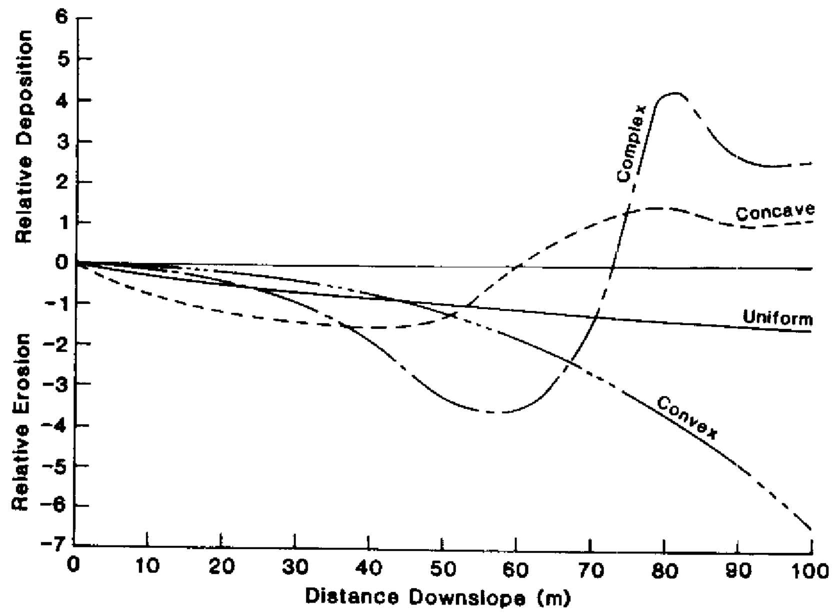

Figure 4. Distribution of simulated (CREAMS, USDA, 1980) erosion and deposition on different profile shapes relative to erosion computed with USLE (Wischmeier and Smith, 1978) for a uniform 100 m long, 5% slope (from Perrens et al., 1985, their Figure 2).

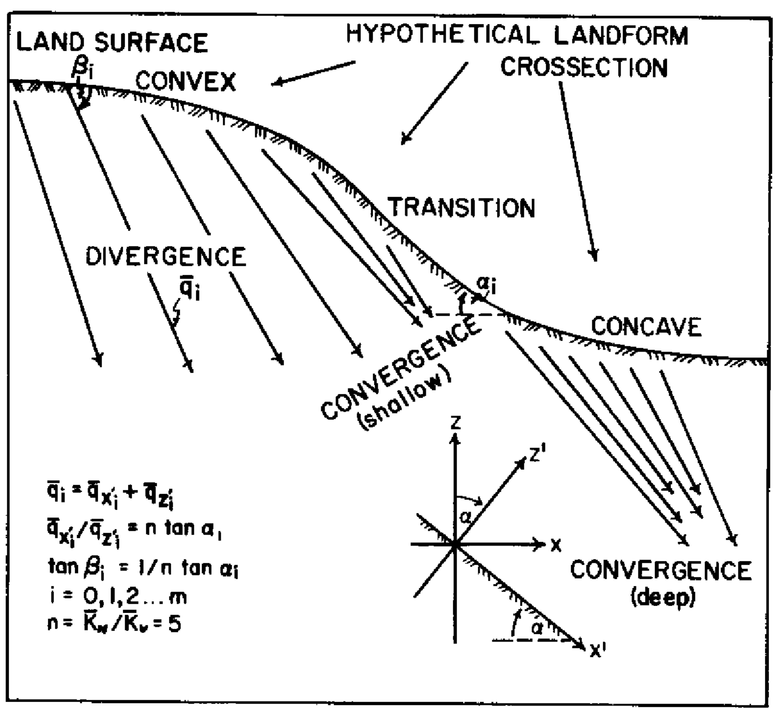

Figure 5. Infiltration stream lines for a hypothetical landform element (from Khanbilvardi and Rogowski, 1984, their Figure 3).

EROSION AND DEPOSITION

Predictive Simulation

According to the concept of partial or source area (Engman and Rogowski, 1974) rainfall and runoff are not uniformly distributed over an entire landscape but occur with different intensity in certain specific portions of the basin. Special significance of delineating the contributing areas associated with nutrient and sediment yield from agricultural watersheds was pointed out above. Nitrate fertilizer is water soluble (NO_3) and readily leached to groundwater or transported in surface runoff. Ammonium-N (NH_4^+) is not as readily leached and can be transported when adsorbed on soil particles (illitic clays and vermiculites). Since highly erodible fine clay soil particles are also generally higher in adsorbed N and P, while organic matter containing nitrogen and organic P is lighter and more erodible than soil, N and P content of stream carried load may be considerably higher (as much as 50% for N) than concentration present in watershed soils (Sweeten and Reddell, 1978). Consequently, accurate prediction of erosion and deposition zones on a watershed is a necessary first step in application of the BMP to cropland.

The Universal Soil Loss Equation (USLE) in conjunction with the sediment delivery ratio (SDR) is the most widely used method for estimating onsite erosion (Wischmeier and Smith, 1978). Although developed originally for use on croplands, it is currently used for planning conservation practices for a variety of land uses. Even though USLE is based on many years of data from experimental plots, it is essentially an empirical formula that cannot separate sheet from rill erosion, sediment transport, scour, or deposition.

Despite its shortcomings the USLE is the only lumped parameter model with sufficient application experience and its slope-length factor, "LS" has long been identified as one of the most important factors in estimating soil loss. Originally this factor was evaluated on 9 percent, 22, 44 and 66 m long uniform slopes. Therefore, the use of USLE on longer slopes, or in concave, or convex topography requires a correction. A rectangular or square plot at constant slope is ideal for application of the USLE, hence subdividing a site into rectangular or square subareas, assumed to be homogeneous and uniform, and computing erosion on each, has been an accepted simulation approach (Eli and Paulin, 1981). Since this type of slope is rare among natural watersheds, which are seldom homogeneous and uniform, a different approach needs to be tried.

Many slopes are irregular in configuration, and typically contain concave and convex sections. Erosion and deposition may thus occur at different points of the same slope (Meyer and Kramer, 1969). These factors must be accounted for by using an appropriate subdivision of the slope to improve definition of a subarea. Convergence of streamlines on concave slopes may create saturated flow conditions at the surface sooner than expected. This means that on concave parts of a slope, moisture in soil will build up faster and continuing infiltration will maintain the soil wetter for longer periods of time. It is a logical assumption that surface runoff will most likely be initiated at such points.

Erosion-Deposition Model (EDM)

One possible approach is to use an erosion-deposition model such as the EDM model developed recently (Khanbilvardi et al., 1983; Khanbilvardi and Rogowski, 1984; Khanbilvardi and Rogowski, 1985). EDM combines the USLE with a system of equations describing hydrologic and hydraulic processes of soil erosion. The input requirements of the model are easy to obtain, and nonlinear behavior within an agricultural watershed can also be simulated. In the model, a watershed is divided into a grid of square subareas, each represented by a node point in the center. To be consistent with the homogeneity assumption, the size of a subarea must be small enough so that all important parameter values within its boundaries can be assumed uniform. In related studies, using geostatistical techniques, we have concluded that estimation of erosion on 1 ha basis (100 x 100 m) will likely lead to optimum prediction capability (Rogowski et al., 1985).

The model computes, using the USLE, the amount of soil detached on each contributing subarea, delineates a pattern of flow channels (rills) and interrill areas, and quantifies the main subprocesses of erosion: (1) detachment by rainfall, (2) transport by rainfall, (3) detachment by rill flow, and (4) transport by rill flow. Detachment by rainfall and transport capacity of runoff are defined by appropriate equations. All soil detached from interrill areas is assumed to move laterally to the closest active rills. Flows in rills transport this soil, as well as rill sediment detached by scour, to the closest stream. Concentration of detached soil and sediment at any point of time, or space within a rill, is the lesser of the rill flow transport capacity, or the availability of particles for transport. Soil detached from interrill areas and sediment from scour constitute the "potential loss" component while sediment yield refers to soil and sediment actually transported out of the subarea, plot or watershed. The final output of the EDM includes such sediment yield values, as well as magnitudes of erosion and deposition in each subarea, and patterns of flow and sediment movement.

The estimated rainfall excess for each subarea is computed by considering distribution of antecedent moisture content and infiltration at each node using Philip's infiltration equation (Philip, 1957). Since the downward infiltration flux has a tendency to deviate from the vertical, the final flow direction will depend on the slope of land surface, its changes, and on the degree of profile anisotropy. Profile as a whole may be considered to be anisotropic if the combined hydraulic conductivity normal to the soil surface is smaller than the hydraulic conductivity parallel to the soil surface. In a sloping soil, combined force of gravity and pressure gradient will usually cause a deviation in infiltration streamlines from the vertical (Zaskavsky and Rogowski, 1969). Therefore, in addition to the flow component normal to the land surface, there exists a lateral flow component parallel to the land surface which contributes to runoff. To account for the shape of the landform, elevations at the corners of each subarea are recorded and compared with elevations at the center. The corrected shape values delineate not only the areas where runoff is likely to begin but also the primary seepage zones that affect runoff water quality (Pionke and Urban, 1985). The accuracy of this approach diminishes as the size of the area increases above 100 m^2 (10 x 10 m). Consequently, suspected sites may need to be evaluated on a grid finer than the 100 x 100 m basis (1 ha).

Eroded soil from contributing areas along with nutrients and sediment detached by flow in rills is routed downslope from node to node to the rill or stream outlet. For rill or stream channel scour to occur the flow shear stress must exceed the critical shear stress necessary for sediment transport. The particle or aggregate size distribution of eroding soil should be known or can be estimated. While the eroded soil is usually a mixture of both the primary particles and aggregates, their relative distribution is a function of soil properties, management practices, extent and type of cover as well as rainfall and runoff characteristics. If the median particle or aggregate size for a given soil or group of soils is known, it can be used in the model. If it is not known the mean particle size is approximated from Manning's roughness coefficient.

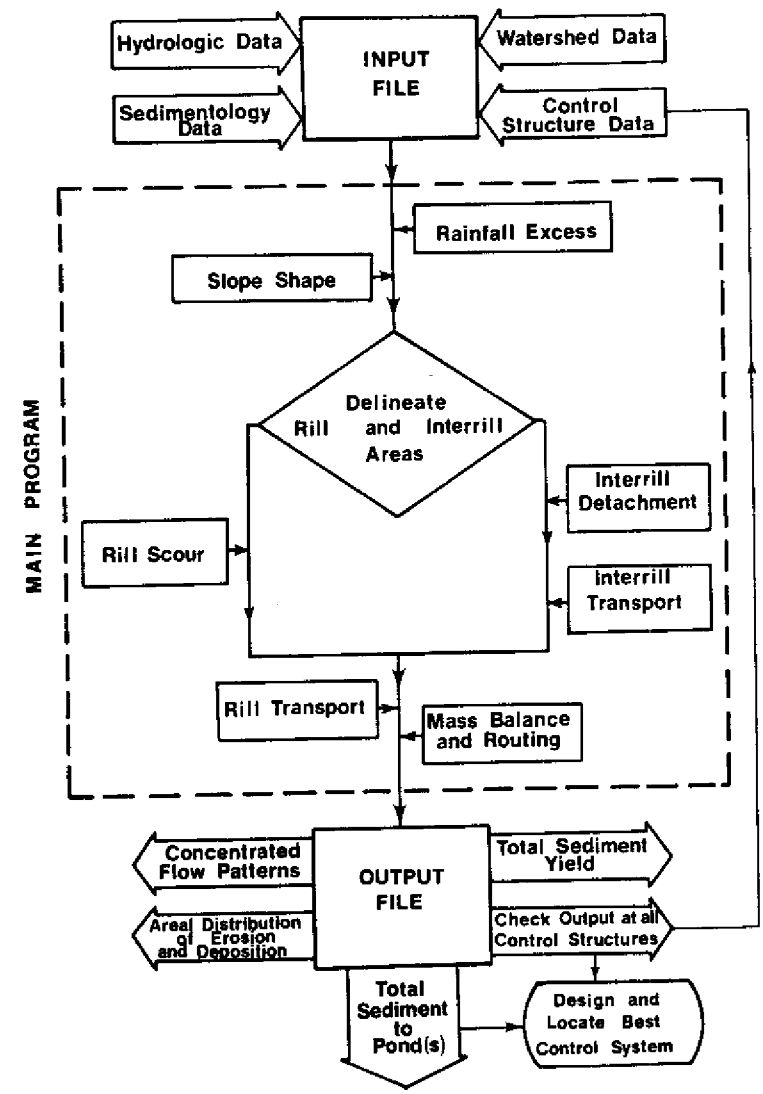

Figure 6. Flow chart of the Erosion-Deposition Model (Khanbilvardi and Rogowski, 1985).

Once the detachment components of the erosion process have been computed, standard routing procedure (Khanbilvardi et al., 1983) is used. Briefly, the model compares all principal slope directions, selects the optimum flow path at each rill node, computes the flow travel time, determines the potential flow rate and depth for each rill segment and computes the rill transport capacity. The model flow chart is shown in Figure 6 and the program is written in FORTRAN and BASIC for use on mainframe and microcomputers.

Practical Example

Just how such a model could be applied to nonpoint source pollution problems is illustrated in the example that follows.

Let us suppose that the area of interest is a 600 ha site. We can represent the site by a 20 x 30 node regular square grid of 100 x 100 m subareas (1 ha each). Let us further suppose Figure 7 gives the distribution of soils and Figure 8 shows the site topography.

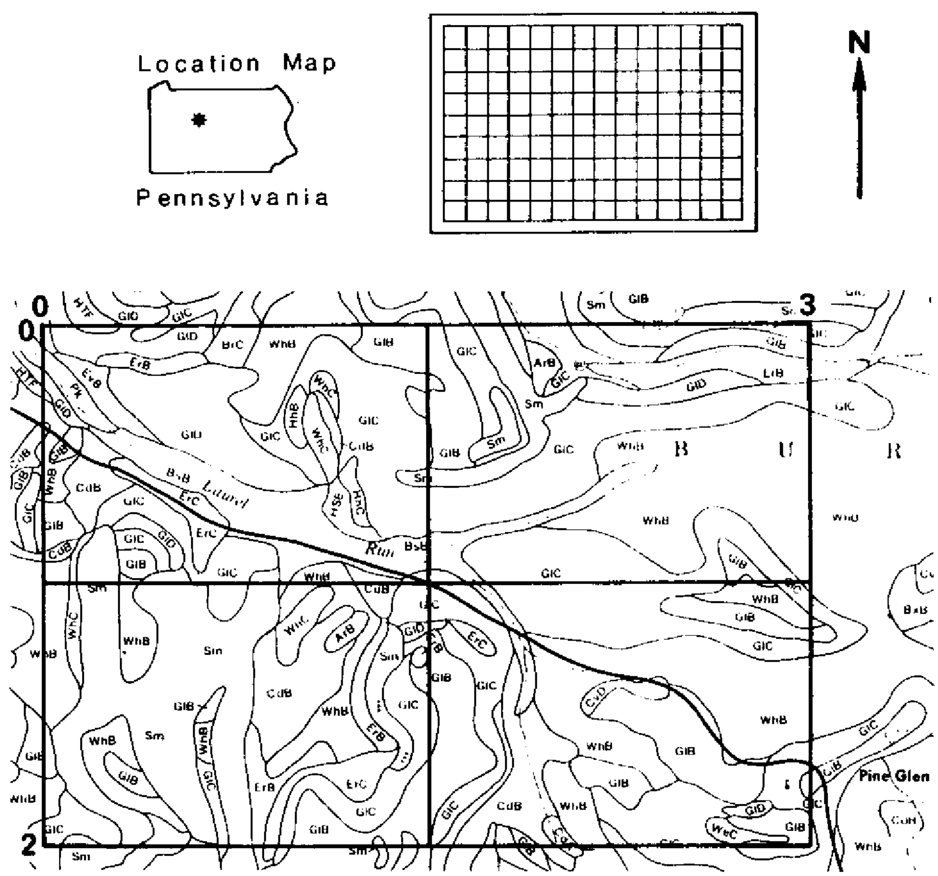

Figure 7. Location of the experimental 600 ha site, distribution of soil series (symbols) and a grid network used to record map data. Slope is given as the third letter of soil symbol: A, 0-3; B, 1-8; C, 5-16; and D, 10-30% (from Khanbilvardi and Rogowski, their Figure 3.

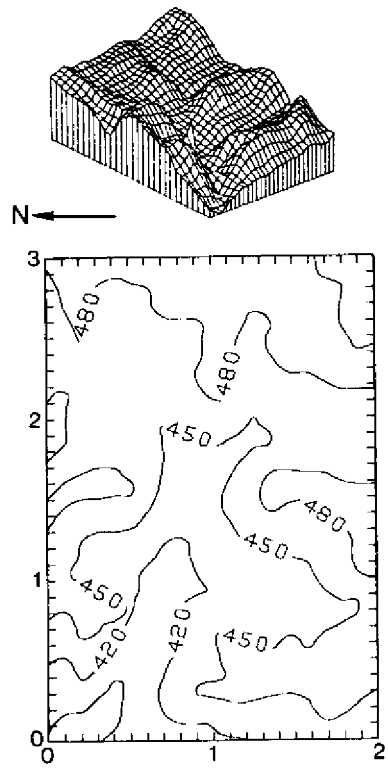

Figure 8. Topography of the 3 x 2 km experimental site, tic-marks designate 0.1 km distances (from Khanbilvardi and Rogowski, 1984, their Figure 4).

The EDM model is now executed for a 6 hr-10 yr storm event (75 mm). After determining the cumulative infiltration on each subarea, the program checks for rill sources and generates a pattern of rills terminating in outlets 1 to 4 at the site boundaries or in internal sinks a to g (Figure 9), and delineates contributing (shaded) interrill areas. Because of the high intensity and long duration of rainfall, the rills and contributing areas appear to cover most of the site.

Outlet 1 in Figure 9 corresponds to the outlet of a stream, which drains the area. Assuming that the whole area was impacted, would necessitate some kind of control structures on the stream suggested by a pattern of sinks a, b, c, d, e, f and g in Figure 9. These sinks can be thought of as potential locations of sediment detention ponds.

Using values of potential soil loss (USLE) at each node, the model then computes for all rills the total amount of soil available for transport in each part of each rill. Both the soil transported to the rill by sheet flow from the interrill areas and the soil detached in the rill by scour are considered. Should the nutrient contributions and distributions relative to particle size be known, the model could also be used to compute potential loading factors for each node. By comparing the transport capacity of rill flow with the total amount of soil available for transport the program determines

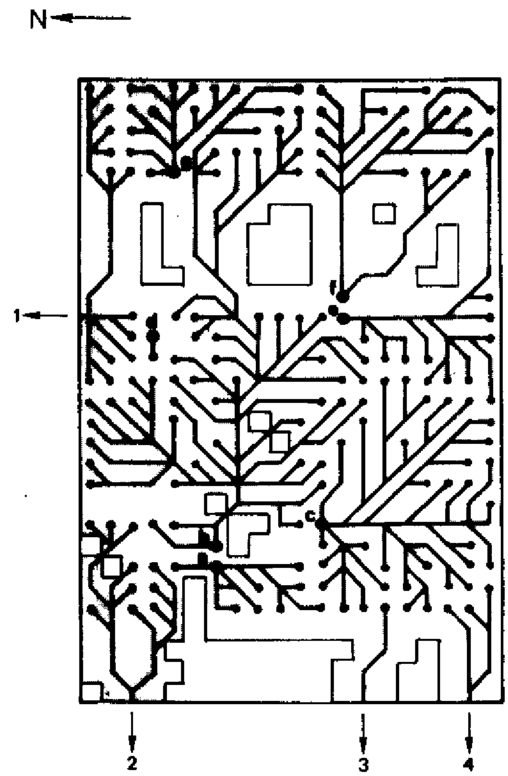

Figure 9. Distribution of rills, of contributing areas (grey) of sinks (a,b,g) and of outlets (1,2,4), at the 3 x 2 km experimental site (from Khanbilvardi and Rogowski, 1984, their Figure 9).

sediment yield or deposition at each node (Figure 10). Prediction of the associated nutrient yield at the node is more complicated since it would depend on the chemical forms as well as particle sizes and nutrient enrichment ratios (Young et al., 1985) of the eroding material.

When rills converge, or surface runoff enters a stream, the sediments in each flow compartment exchange chemicals with the mixing solution and indirectly with each other. Estimates of the postmixing solution concentration and mass of chemical retained by the suspended solids must be obtained to predict the mass of a chemical transported downstream from a confluence (Schnabel, 1985). Kunishi and Pionke (1985) have developed a computer model of P-sorption isotherm to describe partitioning of P between runoff and sediment derived from different source areas. Their work is based on the work of Taylor and Kunishi (1971) who extended Schofield (1985) and Beckett and White (1964) concepts to stream sediments and suspensions associated with large volumes of water. Both of these approaches (Pionke et al., 1985) could be incorporated into the EDM to provide node by node accounting of N and P loading and distribution.

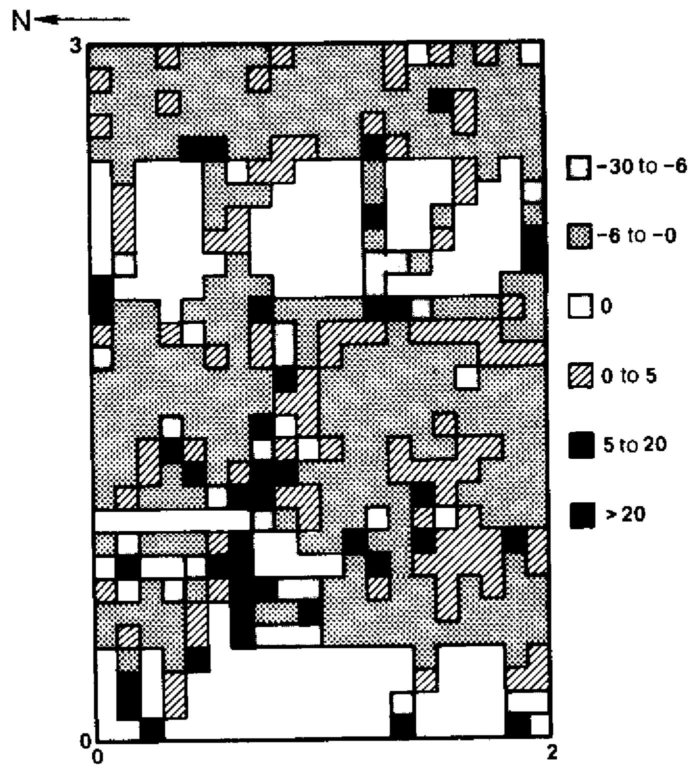

Figure 10. Distribution mosaic of sediment yield (negative) or deposition (positive) (mm) at the 3 x 2 km ex perimental site, blank areas signify no change (from Khanbilvardi and Rogowski, 1984, their Figure 10 a).

Management Choices

The significance of the eroding particle or aggregate size was pointed out earlier (Young et al., 1985). In the EDM model, particle size diameter enters into the computation of flow transport capacity and rill scour components both outright and through Shields Diagram. Dimensional analysis suggests that decreasing median particle size will increase the transport capacity relatively less than the rill scour. Assuming that soil at the site has aggregates ranging between 4 and 12 mm in diameter, Tables 5 and 6 show the influence of median particle size (at 6, 9 and 12 mm) on sediment yield and erosion rate at the seven principal sinks and four outlets indicated in Figure 9. Results indicate considerable variation between different drainages and a consistent increase in soil loss for smaller aggregates. Other data not listed shows that as particle size decreases rill scour component essentially stays the same, while interrill erosion component increases. The EDM sediment yield predictions thus appear to be in agreement with what happens in the field: larger sediment yield and probably larger nutrient loads would generally be expected on less aggregated soils. Increases in erosion rate as the median

Table 5. Soil loss[1] and erosion rates[2] for areas contributing sediment to outlets outside the watershed (from Khanbilvardi and Rogowski, 1984).

Aggregate size (mm)	Outlets: 1 = 40 ha Loss (T)	Rate (kg/m^2)	2 = 35 ha Loss (T)	Rate (kg/m^2)	3 = 15 ha Loss (T)	Rate (kg/m^2)	4 = 16 ha Loss (T)	Rate (kg/m^2)	Watershed = 106 ha Total Loss (T)	Rate (kg/m^2)	Rate[3] (kg/m^2)
6	920	2.30	478	1.35	280	1.87	634	3.96	2307	2.18	0.38
9	893	2.23	441	1.26	274	1.83	626	3.91	2234	2.11	0.37
12	863	4.11	411	1.17	267	1.78	595	3.72	2136	2.02	0.36

[1]Soil loss in metric tons.

[2]Rate of soil loss based on the size of contributing area.

[3]Rate of soil loss based on 600 ha watershed.

Table 6. Predicted soil loss (L) and erosion rate[1] (R) computed for three median partical sizes[2] on areas contributing sediment to sinks[3] inside the watershed (from Khanbilvardi and Rogowski, 1984).

Size	Sinks: a L	a R	b L	b R	c L	c R	d L	d R	e L	e R	f L	f R	g L	g R	Watershed L	Watershed R
6	803	3.35	867	0.49	1501	1.75	102	5.10	479	1.14	3086	3.91	363	1.65	7201	1.67
9	780	3.25	735	0.41	1393	1.62	102	5.10	438	1.07	3028	3.84	342	1.55	6818	1.58
12	754	3.14	615	0.35	1158	1.35	102	5.10	400	0.98	2400	3.04	321	1.46	5750	1.33

[1]Soil loss (L) in metric tons and erosion rate (R) in kg/m^2.

[2]Median particle size in mm.

[3]a = 24 ha, b = 177 ha, c = 86 ha, d = 2 ha, e = 41 ha, f = 79 ha, g = 22 ha, and watershed = 431 ha.

particle size decreases at least for this site do not appear to be linear and indications are that they are likely to plateau out for aggregate sizes between 1 and 2 mm.

In Table 5 respective sizes of sediment contributing areas associated with outlets are given, while Table 6 shows sediment yield to internal sinks (a through g). The areas contributing to sinks account for about 82% of the site. Looking at it another way the model shows that if proper structures were in place to contain the sediment in the sink areas, only 18% of the site is likely to be the source of sediment yield. Under these circumstances the soil loss rate based on a 600 ha area for a 9 mm median particle size would be (Table 5) 0.37 kg/m^2 (< 2 T/A). The analysis focuses attention and directs BMP to areas that are likely to be major contributors (c, f). The analysis also points out, considering associated N and P loadings, the pitfalls of blindly applying the BMP to a site as a whole. Under certain conditions (rain > 6 hr-10 hr storm) conservation control structures at some of the sinks (i.e., at b, c, or f) may not hold allowing potentially much larger and more severe loadings of the stream, despite the fact that overall design soil loss is well below tolerance level (< 2 T/A) required for most cropland.

It is not always wise to use nutrient loading functions based on soil content of N and P. Different size storms may carry away different size aggregates and impact the Bay altogether differently. While control at field or watershed scale using models such as the one discussed here appears within grasp overall impact of such controls on the Bay, particularly from the standpoint of nutrient loads, is less well understood. If indeed the anoxia model mentioned earlier is approximately correct, simulations presented above indicate that reductions suggested by Officer et al. (1983) to control anoxia (11% N and 3% P loads) will be either within experimental error or associated with climatic change, unless a more precise accounting of what goes on at the field scale is developed and implemented.

SCALE CONSIDERATIONS AND VARIABILITY

Scale Interrelationships

The CBP by its very nature is an amalgam of interdisciplinary studies conducted on different scales, ranging from point samples to basin wide estimates without any apparent structural fabric connecting the pieces. The question before us is therefore two-fold. First, what might this fabric be and second how are the different scales connected to one another. I will leave the first question till last and briefly consider the interrelationships of different scales.

Predicting behavior of ungaged watersheds is complicated because of heterogeneity in time and space with respect to weather variables, vegetative elements as well as actual soil properties.

Soil chemical and physical properties, required as inputs for many deterministic models, vary considerably within the soil series (e.g., Rogowski, 1971a; Nielsen et al., 1973; Sharma et al., 1983; Webster and Burgess, 1980; Wilding and Dress, 1983; Webster, 1985), and such variability has been shown to affect the response behavior of larger areas (e.g., Warrick et al., 1977b; Sharma and Luxmoore, 1979; Smith and Hebbert, 1979; Bresler et al., 1979). There are several aspects of soil variability which need to be explored from the standpoint of CBP. For example, what are the parameters which would be most indicative of hydrologic and chemical variability in the Chesapeake Bay Basin, how and where should these parameters be measured, how best to analyze such data and assess to what extent the variability affects response of the area in time.

In the last decade soil scientists have shown an increasing interest in statistical analysis of spatially distributed data, interpretation and extrapolation of results to larger areas as well as different scale considerations (Warrick and Nielsen, 1980; Peck, 1983; Sharma, 1983; Rogowski et al., 1985). It was soon realized that the main impediment to the development and application of quantitative physically-based models describing nonpoint source pollution is the lack of adequate characterization of field soil heterogeneity. While extensive studies of field variability continue much of the work is of the exploratory rather than applicative nature (Greminger et al., 1985; Kachanoski et al., 1985a, 1985b).

In a program such as CBP understanding of soil physical and chemical processes on a spatial scale is especially important to enable the prediction of the Bay response behavior. In practice CBP studies range in size from plots ($< 10^2$ m^2) to river basins ($> 10^7$ m^2). Ideally one would like to have a measure of pertinent chemical and hydrological properties and associated variability at each of these scales, but because of lack of appropriate techniques, measurements usually are made on small point samples collected from "representative" areas, or in situ at "representative" locations within component watersheds. Basic questions that arise pertain to the interpretation of "representative": how representative are the locations within a watershed, and how representative are the properties measured at a point for an area as a whole, and more specifically what does an average of these representative properties from representative locations mean.

At a scale of individual soil particle or pore, soil system is highly variable, however such variability is ignored by measuring properties on samples of much larger volume (> 50 cm^3). The variation of a property generally decreases with an increase in the volume of the sample and eventually reaches a relatively constant value. The smallest volume at which this can occur is defined as a representative elementary volume (REV). Usually smaller samples will have larger uncertainty in estimating population mean, while fewer samples of larger volume may be adequate (Sisson and Wierenga, 1981), provided we are sampling within the boundaries of a reasonably homogeneous soil unit. Just where these boundaries are is sometimes difficult to define as they may vary from property to property, and from area to area. There is a fundamental difference in the way the additive soil properties (e.g., volumetric water content and chemical properties) and the intensive properties (e.g., hydraulic conductivity) are derived from the physical behavior of individual pores. By increasing the volume of a sample, the variance for an additive property is likely to decrease at a much faster rate then that for an intensive property. This has been demonstrated for hydraulic conductivity by Babalola (1978) who found that variance on a 92 ha field was on the average only 1.5 times larger than on a 2.3 ha plot.

Practical Application

In practice, hydrological and chemical properties in an area such as Chesapeake Bay are measured on sample volumes chosen for convenience, or to accommodate the equipment, and these measurements are considered points of a continuum. However, most physically-based deterministic models assume that REV can be defined and is repeated regularly throughout the land area in question. Such assumption may be open to criticism.

Attempts have also been made to derive field distributions of a property from laboratory values. For example, hydraulic conductivity distributions have been computed from laboratory measured moisture characteristics (e.g., Childs and Collis-George, 1950; Millington and Quirk, 1959; Brooks and Corey, 1964; Green and Corey, 1971) and moisture characteristic and hydraulic conductivity distributions have been approximated from only a few field measurements (e.g., Brooks and Corey, 1964; Rogowski, 1971, 1972a). While such procedures enable rapid determination of approximate magnitude of field values of hydrologic parameters based on soil type, they shed little light on within soil series variability, and even less (Webster, 1985) has been done along these lines with regards to nutrients or sediment transport (Rogowski et al., 1985).

Soil Variability

Coefficient of variation (CV) is generally accepted as a relative measure of soil variability. Use of CV however, has limitations; for example, the variance of a parameter is usually found to increase with the increase in the size of area sampled (Beckett and Webster, 1971), and the relative magnitude of this increase may vary for different properties and under different conditions. For example, CV for exchangeable Mg may be between 0.05 and 0.15 for a pedon but between 0.37 and 1.21 for a mapping unit (Wilding and Dress, 1983). The magnitude of variability for soil properties on a few hectares or less is usually found to be moderately high (CV > 0.35) for exchangeable H, Ca and Mg organic matter, fine clay, soluble salts and water contents, as well as hydraulic conductivity; moderate (CV = 0.15-0.35) for total clay content; cation exchange capacity, base saturation, soil structure and low (CV < 0.15) for pH, bulk density, silt content and particle size distributions (Warrick and Nielsen, 1980; Wilding and Dress, 1983).

The variability of a property over an area can also be described by the probability density function (PDF), which contains information about the averages (mean, mode, median) and their moments, and these permit estimation of confidence limits. Identifying the appropriate PDF for a parameter under given conditions has important implications, for example, in computing the number of observations required for estimating mean with a specified degree of precision (Rogowski, 1972b; Sharma, 1983), and in determining the integrated response of an area (e.g., Sharma and Luxmoore, 1979; Warrick and Amoozegar-Fard, 1979).

The higher coefficients of variation in the PDF are usually associated with larger skewness. Properties exhibiting larger CV (> 0.40) are found to have a log-normal distribution, while those with lower CV (< 0.40) may be adequately fitted with a normal function (Rao et al., 1983). Transport coefficients such as hydraulic conductivity, diffusivity, sorptivity and some chemical properties, electrical conductivity, organic carbon, total N, are usually found to be log-normally distributed, while properties such as water content, bulk density, porosity, are generally normally distributed (Rogowski, 1972b; Warrick and Nielsen, 1980; Wilding and Dress, 1983). Distributions of Ca, Mg and C/N ratios are usually of the gamma type.

Soil spatial variability in a large area such as Chesapeake Bay can be systematic or random. Systematic variability is a change in soil properties as a function of landform, geomorphic elements, soil forming factors and soil management (Wilding and Dress, 1983). While in general, spatial variability will increase as the size of the area increases, often it will plateau out and maximum variability of individual properties may occur within a readily definable and at times small area. Since within this area soil variables can be considered continuous, it might be advantageous to know the projected size of the area.

In many practical situations, such as Chesapeake Bay study, the extent of soil variability could be determined initially for several easily-measured parameters, since it is more important to characterize the variability adequately by a procedure which allows many rapid approximate measurements of a property over the area, rather than expanding the same effort in getting only a few measurements of exact values. Subsequent use of geostatistical analysis could shed light on the behavior of other variables.

Geostatistics

Unlike the REV concept which assumes homogeneity, geostatistics considers so-called regionalized variables whose values are related in some way to their position and which demonstrate spatial dependence. Application of geostatistics to soil science is relatively recent (e.g., McBratney et al., 1981; Hajrasuliha et al., 1980; Rogowski, 1980; Webster, 1985; Sisson and Wierenga, 1981; Vieira et al., 1981; Gajem et al., 1981; Sharma et al., 1983), and its usefulness, limitations, and further development in relation to environmental problems is yet to be fully explored and appreciated.

Environmental scientists generally characterize soil variability by estimating an areal mean or interpolating between data points. To achieve this more elegantly Matheron's theory of regionalized variables (Matheron, 1971) may be used as a basis of analysis (Journel and Huijbregts, 1978). In such an approach soil unit is treated as a random function, and no underlying mathematical relationship between soil properties and their location within a unit (unlike PDF approach) is a priori assumed. The theory quantitatively evaluates the extent and nature of properly dependence with respect to their location in space. This dependence is described by a semivariogram.

Semivarioigram

A semivariogram $\gamma(h)$, expresses the spatial dependence of neighboring observations, measured as a function of a distance vector h. It is simply half the variance of the differences between observation points and can be written as,

$$\gamma(h) = \frac{1}{2N} \sum_{i=1}^{N} \left[Z(x_i) - Z(x_i+h) \right]^2 \tag{3}$$

where N is the number of pairs $[Z(x_i), Z(x_i+h)]$ at separation distance h. Computed semivariances are discrete estimates of a continuous function $\gamma(h)$ which describes average rate of change of γ with distance. A semivariogram describes the variance structure of the observations in an area, and these observations may follow any type of frequency distribution.

An idealized semivariogram is shown in Figure 11. Numerically the sill (C) in the limit corresponds to sample a priori variance, while the range (a) delineates a neighborhood where the variable is continuous. In practice, provided the area sampled is large enough, the sill may approximate population variance.

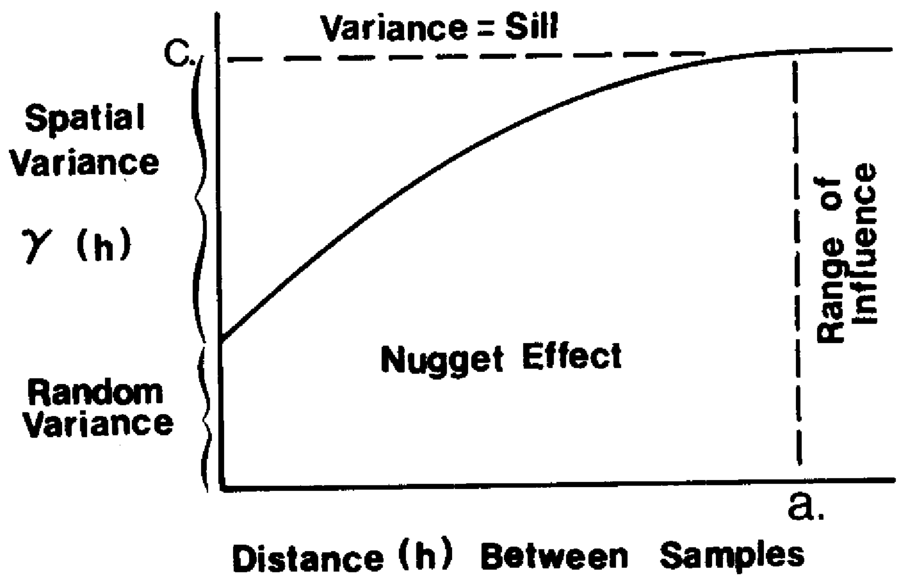

Figure 11. Idealized semivariogram for a soil property showing "sill" (C), "range" (a) and "nugget effect."

The presence of a sill means that a variable is partly random ($h > a$) and partly continuous ($h < a$). Quite frequently a semivariogram does not go through the origin, but at $h = 0$ has some finite value known as a "nugget effect." When this happens, we suspect an existence of spatial structure on a scale smaller than sampling interval h, which may have to be subsampled to obtain a point variogram that goes through the origin. Mottles and concretions in a soil profile (Burgess and Webster, 1980a,b), inclusions in soil series delineation, trees in a woodland, all would be expected to register a nugget effect. On the other hand, properties such as hydraulic conductivity and water content distribution, may have little or none of it (Russo and Bresler, 1981; Vieira et al., 1981).

Variogram estimates

We will briefly consider now how to estimate variograms over a large area such as watershed or drainage basin provided semivariograms for point sample are available. This aspect of geostatistics more closely pertains to Chesapeake Bay applications and considers the notion of sample size. Intuitively we know that if for example, soil moisture is measured "at a point" with a neutron probe the sample represents about a 30 to 50 cm diameter sphere. In the context of a field this can perhaps be considered a point sample.

Based then on what in our estimation are point samples and their semivariogram we could say something about the expected variability on a field, watershed or even Chesapeake Bay drainage basis using the "regularizatiion" procedure. Regularization may be thought of as an attempt to express field data on a gross basis of areas contrasted to point values. Let us suppose that soil, or ecological data fit one of the common models of point semivariogram. Then for samples of any length ℓ we have using the expression from Journel and Huijbregts (1978, p. 84) whenever h--distance between samples--is large compared to ℓ,

$$\gamma_{\ell}(h) = \gamma(h) - \bar{\gamma}(\ell,\ell), \; |h| \geq \ell \tag{4}$$

where $\bar{\gamma}(\ell,\ell)$ (Figure 12) describes the dimensions and spatial arrangement of sampling information. As ℓ increases $\bar{\gamma}(\ell,\ell)$ will also increase with decreasing $\gamma_{\ell}(h)$. In practice for a fixed ℓ, however, values of $\gamma_{\ell}(h)$ will depend on where the sampling points are. Thus estimating $\gamma_{\ell}(h)$ by two points that are far apart is more accurate than if they were close together. Values of $\bar{\gamma}(\ell,\ell)$ in one dimension (lumping of core data) can be readily evaluated (Clark, 1982, p. 48), for larger dimensions appropriate tables of auxiliary functions must be used.

Estimation and dispersion variance

The estimation variance (σ_E^2) tells us how point samples are related to field values, while the dispersion variance $D^2(v/V)$ describes the scatter of point sample values about the field mean. Expressions for calibration of estimation (σ_E^2) and dispersion (D^2) variance are similar in form to Equation (4). We rewrite $\bar{\gamma}(\ell,\ell)$ as $\bar{\gamma}(v,v)$, where v implies area or volume rather than ℓ which implies length. In a space continuum there may be, for example, an unknown field domain V, and a known sample domain v. An average semivariogram between all points of V is denoted as $\bar{\gamma}(V,V)$, an average semivariogram between all sample points v is denoted as $\bar{\gamma}(v,v)$, and an average semivariogram between all sample points and domain points is denoted as $\bar{\gamma}(v,V)$. The variance of estimation (or extension of results from sample domain v to field domain V) can then be written as,

$$\sigma_E^2(v,V) = 2\bar{\gamma}(V,v) - \bar{\gamma}(V,V) - \bar{\gamma}(v,v) \tag{5}$$

The estimation (or extension) variance (σ_E^2) gives an error of estimation of an unknown field domain V by information contained in sample domain v. Similarly, the variance of dispersion may be written as:

$$D^2(v/V) = \bar{\gamma}(V,V) - \bar{\gamma}(v,v) \tag{6}$$

where $D^2(v/V)$ describes the scatter of sample values v about the mean of V.

Sources of variability

Figure 12 shows an idealized point and composite (by length ℓ) semivariograms and illustrates graphically the relationships between them. The composite semivariogram $\gamma_{\ell}(h)$ will normally be lower

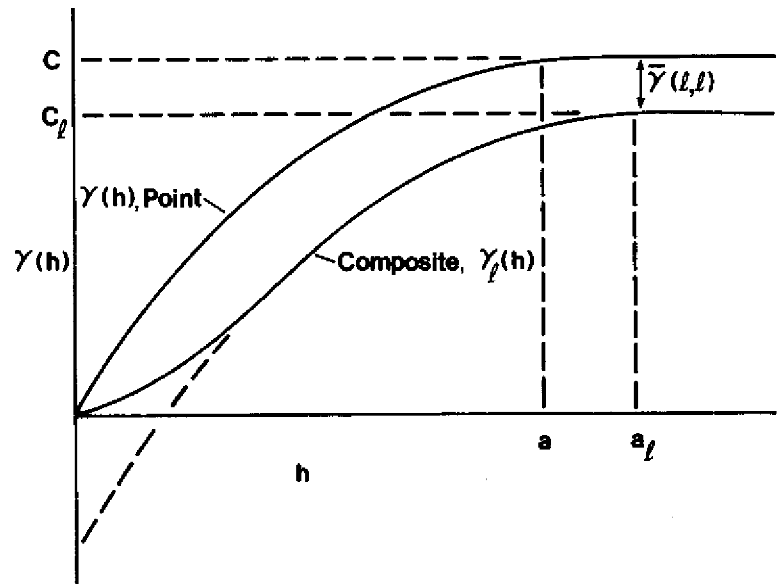

Figure 12. Idealized relationship between point and composite semivariograms.

with a larger range (a_ℓ) and a lower sill (C_ℓ) differing from the point semivariogram $\gamma(h)$ by $\bar{\gamma}(\ell,\ell)$. The point and composite semivariograms may be considered as a series of nested structures, although the true nested structures are not necessarily the realizations of point and composite semivariograms, but may arise as a result of soil forming factors operating on different scales (Burrough, 1983). While regularization relates to one another known increments of length, volume, or area, nested structures generally reflect different scales of variability found in the field. The variability associated with a distribution of a variable within a field may be due to many reasons (Journel and Huijbregts, 1978). Since soil is a complex system, its properties are not only a consequence of interaction of natural soil forming factors such as climate, parent material and topography with various physical, chemical and biological processes, but also result from man-induced land use and management schemes. The effects of these factors on soil properties is likely to be evident at varying spatial scales.

It is important to identify sources of variability and their corresponding spatial scale. For example, at a point $h = 0$ variability may be due to errors in sampling techniques and on a scale $h \leq 10$ mm differences in textural components of the soil matrix (sand, silt, and clay) may show up. For $h \leq 1$ m difference in layering and horizonation would appear, and for $h \leq 100$ m we would see differences due to series delineation while for $h \leq 100$ km differences in soil orders may become noticeable. Depending on a scale of measurement each one of these sources of variability would be expressed either as a nugget effect or as a nested structure. The composite semivariogram describing the variability would then be the sum of these nested structures, i.e.,

$$\gamma(h) = \gamma_0(h) + \gamma_1(h) + \gamma_2(h) \tag{7}$$

Alternately if our interest was at a particular scale of measurement, only the pertinent structures would be utilized in the semivariogram construction. The procedure described would allow targeting the controls and BMP to where they were most needed and yet would permit evaluation of impacts for the Bay area as a whole.

Geostatistics thus combines a good deal of theoretical sophistication with much flexibility in applying the theory to actual field phenomena. It may therefore provide a necessary fabric that can relate to one another diverse CBP studies conducted at different scales.

SUMMARY AND CONCLUSIONS

Even though nutrient loading of the Bay from diverse sources (including the nonpoint sources) is well documented it is not possible to establish conclusively that control of point sources and application of the best management practices to the nonpoint sources will per se arrest the decline and improve the quality of the Chesapeake Bay ecosystem. The reason for this is the uncertainty about the primary productivity of the Bay, the true extent of recycled nutrient inputs, and ongoing benthic contributions.

In general the program (CBP) appears to lack adequate resolution of the nutrient load associated with the sediment. Although erosion potential appears well defined throughout the area there is no predictive mechanism that can delineate contributing areas nutrients and sediment transported to the Bay. Likewise there appears to be no mechanism to demarcate primary zones of deposition. This makes the application of the best management practices both arbitrary and expensive. The contributions of sediment and nutrients are to a large extent dependent on the size of events likely to occur within the Chesapeake Bay Basin. Yet little information appears available on the critical minimum storm size for the Bay area and its relation to the best management practices being implemented.

Chesapeake Bay Program is a collection of separate studies carried out on different area and time scales. It needs a unifying analysis that can relate component parts to the whole. Geostatistical analysis of available data is suggested. Such a treatment would describe observed data variability and relate studies carried out on different scales and at different times to one another.

It is unlikely that suggestions offered here will solve the Bay's problems overnight. They hopefully offer a different perspective on this beautiful and bountiful area.

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A Comprehensive Watershed Model for Nonpoint Source Pollution Assessment[1]

by

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ABSTRACT

Nonpoint source pollution from cropland has been identified as the primary source of nitrogen and sediment, and a significant source of phosphorus in the Chesapeake Bay. These pollutants, whether from point or nonpoint sources, have been found to be the primary cause of declining water quality in the Bay. Numerous studies have indicated that, for many watersheds, a few critical areas are responsible for a disproportionate amount of the nutrient and sediment yield. Consequently, if pollution control activities are concentrated in these critical areas, then a far greater improvement in downstream water quality can be expected with limited funds.

In this paper, ANSWERS, a distributed parameter watershed model is used to evaluate the effect of no-till cultivation on sediment yield from an agricultural watershed. The model also is used to evaluate the impact of using no-till practices in critical areas within the watershed. In addition, a technique is presented for selecting a design storm for ANSWERS. The technique creates an n-year recurrence interval storm with a duration equal to the time of concentration of the watershed. The intensity pattern was simulated on a 10-minute interval using a first-order Markov model with a lognormal distribution. Also presented is a proposed phosphorus model to be incorporated into ANSWERS. The phosphorus model includes both soluble and sediment-bound components, and can be used to study the effects of Best Management Practices on phosphorus yields for the Chesapeake Bay Program.

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INTRODUCTION

Recent studies on the decline of the Chesapeake Bay have concluded that both point and nonpoint source pollution are responsible for water quality degradation in the Bay (USEPA, 1983). In recent years, significant progress has been made in developing technology for controlling point sources, while nonpoint sources of pollution have been relatively neglected.

Nonpoint source pollution is transported primarily by runoff from urban, agricultural and mining areas, and construction sites. Runoff carries sediment, organic matter, bacteria, pesticides, metals, nutrients, and other chemicals. Nutrients, primarily nitrogen and phosphorus, can be a major problem because they cause eutrophic algae growth. As the algae die and decay, they utilize dissolved oxygen, which reduces the oxygen available to living organisms. In addition, excess algae increases the turbidity of water and reduces the available sunlight to submerged aquatic vegetation, a valuable food source and breeding ground for aquatic organisms.

The Environmental Protection Agency (EPA) Chesapeake Bay study concluded that nitrogen and phosphorus are the primary pollutants responsible for declining water quality in the Bay (USEPA, 1983). The EPA Chesapeake Bay Watershed Model estimated that nonpoint sources were responsible for approximately 39 percent of the phosphorus and 67 percent of the nitrogen during an average year. Furthermore, cropland was estimated to be responsible for 27 and 60 percent of the phosphorus and nitrogen from nonpoint sources, respectively (USEPA, 1983). Cropland, therefore, is the primary source of nitrogen and a major source of phosphorus in the Bay.

Sediment as a nonpoint source pollutant causes many problems. Sediment increases the turbidity of water and its deposition can kill submerged aquatic vegetation, as well as reduce the storage volume of waterways. In addition, a large percentage of the phosphorus and pesticides that enter waterways are adsorbed onto soil particles. Thus, the processes of soil erosion and sediment transport play a significant role in the water quality decline.

A reduction in soil erosion from cropland would result in a significant decrease in the quantity of nutrients entering the Bay. One method of reducing soil erosion is through the use of Best Management Practices (BMPs). This has been the approach taken by national soil and water conservation programs, whose goal is maintaining or improving agricultural productivity. These programs now have an additional benefit, that of improving downstream water quality. Encouraging farmers to implement BMPs, however, has not always been successful. In Virginia, the traditional approach has been to provide technical assistance to farmers who request it, and to provide financial incentives to farmers who implement approved BMPs. Unfortunately, funds have always been limited. As a result, BMPs are not as widely used as they could be.

Numerous studies have indicated that, for many watersheds, a few critical areas are responsible for a disproportionate amount of the nutrient and sediment loadings to downstream waters. Consequently, if pollution control activities can be concentrated in the critical areas, then far greater improvements in downstream water quality can be expected with limited funds. A methodology for identifying potentially critical source areas is currently under development by the Departments of Agricultural Engineering and Landscape Architecture at Virginia Tech. The system, Virginia Geographical Information System

(VIRGIS), includes topography, soils, and land use data. It is being used to identify potentially critical source areas of nonpoint source pollution using the Universal Soil Loss Equation and simple sediment delivery algorithms. Once the potentially critical areas are identified, there is a need for a more precise technique to evaluate the sediment and nutrient yields from these critical areas and to evaluate the effectiveness of various BMPs in reducing these yields.

This paper demonstrates the use of ANSWERS (Areal Nonpoint Source Watershed Environmental Response Simulation) in evaluating the effectiveness of no-till practices in reducing the sediment yield from an agricultural area. ANSWERS is a distributed parameter watershed model which has already been extensively tested and verified on watersheds in Illinois, Indiana, Iowa, Ohio, Oklahoma, Texas, Virginia, Pennsylvania, and Ontario, Canada. In addition, a technique is presented for selecting a design storm to be used by ANSWERS. The technique creates a design storm of the desired duration, amount, intensity pattern and recurrence interval for a given watershed based upon the precipitation records of neighboring areas. Also presented is a proposed phosphorus model, which includes both soluble and sediment bound phosphorus components. The work presented in this paper represents the first step towards the development of a more complex version of ANSWERS for modeling the particle size distribution of sediment and phosphorus transport from agricultural areas.

METHODOLOGY

Model Description

ANSWERS is a distributed parameter, deterministic, watershed model developed for predicting the hydrologic response of watersheds to storms and the erosion and sediment response of watersheds to different agricultural management systems. The basic hydrologic model, developed by Huggins and Monke (1966), describes the processes of interception, infiltration, surface storage, interflow and surface runoff. The hydrologic model is described in more detail by Beasley (1977), Beasley et al. (1980), and Beasley and Huggins (1980). Beasley (1977) expanded the model to include erosion and sediment transport, tile drainage and channel flow. The current model also simulates parallel tile outlet terraces, sediment basins, grassed waterways, and field borders (Beasley and Huggins, 1981).

Dillaha (1981) developed an extended version of the sediment transport model, which is capable of simulating the transport of individual particle size classes in a sediment mixture during the overland flow process. These improvements allow ANSWERS to be used to study the transport of sediment bound pollutants, and to investigate the effects of changing watershed characteristics on fines enrichment.

The ANSWERS model requires that a watershed be divided into a grid of small square elements. Within each element, the hydrologic parameters (slope direction and magnitude, vegetation, soil type, surface condition, rainfall and management practices) are assumed to be uniform, and the hydrologic processes are treated as independent functions of the parameter characteristics. The degree of uniformity of these parameters is used to determine the limiting element size. The output from each element is routed to its downslope neighboring elements, and eventually is routed to the watershed outlet.

The greatest advantage of ANSWERS and other similar models is that they can be used to predict the response of watersheds to changes in conditions in small areas of the watershed. In conjunction with its erosion and sediment transport model, ANSWERS can identify the critical areas within the watershed that have high erosion rates and determine whether or not the soil losses from these areas contribute substantially to the total sediment yield of the watershed. The model is thus an excellent planning tool for quantitatively evaluating the advantages of various BMP scenarios.

Design Storm

The major driving force for any runoff related model is precipitation. Rainfall impact influences soil detachment and transport by adding to the turbulence of overland flow. The time distribution of precipitation also governs the extent and magnitude of surface runoff. The use of event oriented models like ANSWERS requires the careful selection of a design storm. Determining an appropriate rainfall distribution can be approached in several ways. One approach is to use the rainfall distribution from a series of actual storm events. Disadvantages of this approach include high computational costs and added complexity to the analysis by adding uncertainties concerning the recurrence interval for the storm events selected. ANSWERS can also use a generalized storm distribution as presented by Kent (1968). However, these distributions are for large areas and localized effects may be significant.

The approach taken in this study was to utilize a simulated intensity distribution. The storm duration was taken as the time of concentration of the watershed. This results in a storm with the highest possible intensities and peak runoff for a given recurrence interval. The time of concentration of the watershed, T_c (min), was calculated using:

$$T_c = 0.02\ L_c^{0.77}\, S_c^{-0.385} + \left[\frac{2.2\ n\ L_o}{\sqrt{S_o}}\right]^{0.467} \qquad [1]$$

where L_c is the maximum length of channel flow (m), L_o is the maximum length of overland flow (m), Sc and So are the channel and overland flow slopes (m/m) associated with L_c and L_o respectively, and n is Mannings roughness coefficient. The first part of Equation 1 (Schwab et al.,1981) was developed to describe channel flow, while the second part describes the travel time for overland flow (Kerby, 1959).

A rainfall amount for the chosen storm duration was obtained from United States Weather Bureau (USWB) Technical Paper (TP) No. 29 for the desired recurrence interval. In this study a 2-year recurrence interval storm, occurring during mid to late spring when cropland is most susceptible to erosion, was chosen for the design storm.

The next step was to develop an intensity pattern using a first-order Markov model with a lognormal distribution. A lognormal distribution can be expressed as (Ang and Tang, 1975):

$$f_X(x) = \frac{1}{\sqrt{2\pi}\ x\ \zeta}\exp\left[-1/2\left(\frac{\ln(x) - \lambda}{\zeta}\right)^2\right] \qquad x \geq 0 \qquad [2]$$

where λ is a scale parameter and ζ is a shape parameter. The rainfall intensities were fit to a lognormal distribution using Virginia break-point precipitation data presented by Shanholtz and Burford (1967). Storm events during the month(s) desired were then divided into 10-minute intervals and transformed by taking the cube root. A lag-one serial correlation was also required and was calculated as described in Haan (1977). Next, the transformed data was fit to a lognormal distribution as suggested by Haan (1977). The estimated lognormal distribution and a histogram of actual data for the Nomini Creek Watershed, which will be discussed later, are shown in Figure 1.

A first-order Markov model was used to simulate 10-minute rainfall intensities. The Markov model, as given by Haan (1977), preserves the mean, variance, skewness and first-order serial correlation of the original data. The model is based on a transformation $Y_i = \ln(X_i - \alpha)$, where the X's are the original data, and α is a constant. Y_{i+1} is simulated using the following equation given in Haan (1977):

$$Y_{i+1} = \mu_y + \rho_y(1)(Y_i - \mu_y) + t_{i+1}\ \sigma_y \sqrt{1 - \rho_y^2(1)} \qquad [3]$$

where μ_y, σ_y, and $\rho_y(1)$ are the mean, standard deviation and first-order serial correlation coefficient of the natural logarithms of the data, and t is a random number following a t-distribution. To preserve the statistical properties of the original data, the following equations must be solved (Haan, 1977):

$$\mu_x = \alpha + \exp(\sigma_y^2/2 + \mu_y) \qquad [4]$$

$$\sigma_x^2 = \exp\left[2(\sigma_y^2 + \mu_y)\right] - \exp(\sigma_y^2 + 2\mu_y) \qquad [5]$$

$$\gamma_x = \frac{\exp(3\sigma_y^2) - 3\exp(\sigma_y^2) + 2}{\left[\exp(\sigma_y^2) - 1\right]^{1.5}} \qquad [6]$$

$$\rho_x(1) = \frac{\exp(\sigma_y^2\ \rho_y(1)) - 1}{\exp(\sigma_y^2) - 1} \qquad [7]$$

In these equations; μ_x, σ_x^2, γ_x , and $\rho_x(1)$ are the mean, variance, coefficient of skew and the first-order serial correlation coefficient of the original data, respectively. The values of μ_y ,σ_y, $\rho_y(1)$, and α are estimated using Equations 4 thru 7, and are then used in Equation 3 to generate a value of Y_{i+1}. The value of X_{i+1} is then calculated using:

$$X_{i+1} = \exp(Y_{i+1}) + \alpha \qquad [8]$$

Use of the Markov model requires an initial 10-minute intensity, I_{10} , to initiate a storm sequence. The initial 10-minute intensity was found to be a function of both total rainfall amount,A, in inches, and storm duration, D, in hours. An appropriate model was determined using least-squares multiple regression and was found to be:

$$\ln(I_{10}) = -1.11\ln(D) + 0.895\ln(A) \qquad [9]$$

This relationship was determined for the Nomini Creek Watershed from 112 storm events for the months of April, May and June. Random values for I_{10} were then simulated with the following equation:

$$I_{10} = \exp[-1.11\ln(D) + 0.895\ln(A) + s\ \ N(0,1)] \qquad [10]$$

where s is the estimated residual standard deviation of the natural logarithm of I_{10} and $N(0,1)$ is a random value generated from a standard normal distribution (Haan, 1977).

The procedure for generating a design storm can be summarized as follows:

1. Determine the time of concentration of the watershed using Equation 1.
2. From USWB TP-29 find a rainfall amount for an n-year recurrence interval for a duration equal to the time of concentration of the watershed.
3. Simulate a first 10-minute intensity using Equation 10.
4. Use Equation 3 to simulate the number of 10-minute intensities equal to the storm duration.
5. Multiply each simulated 10-minute intensity by the original storm amount divided by the simulated storm amount.

Step 5 is necessary in order to obtain the original storm amount. The multiplication of the intensities by a constant preserves the variance, coefficient of skew, and first-order serial correlation of the simulated data. A histogram of the simulated and actual 10-minute intensities is shown in Figure 2.

DEMONSTRATION WATERSHED

Watershed Description

The Nomini Creek watershed used to demonstrate the use of ANSWERS is located in Westmoreland County, Virginia. This watershed was selected because the Virginia Division of Soil and Water Conservation had identified it as a potentially critical source of nonpoint pollution. The watershed will be monitored over the next 10 years to assess the long-term water quality impacts of BMP implementation. The watershed contains approximately 1500 hectares (3700 acres), with land use being approximately half agriculture and half forested. The upland areas with mild slopes are generally in row crops. Steep areas and the lowlands are forested. The watershed is characterized by well drained sandy and loamy soils.

Scenario Descriptions

The watershed was partitioned into one hectare elements, as shown in Figure 3. The arrows in each element in Figure 3 indicate the slope direction or direction of flow for that element. The darker arrows designate channel elements. Soils, topographic and land use data required by the ANSWERS model were obtained from soil surveys and 1:24000 scale United States Geological Survey topographic maps. All other parameters required by ANSWERS were estimated using procedures discussed by Beasley and Huggins (1981). The closest available precipitation data to fit the intensity distribution was from the Pony Mountain Branch watershed in Culpeper County, Virginia, which is located approximately 70 miles Northwest of the Nomini Creek watershed. For this study, the time of concentration of the watershed was found to be 130 minutes using Equation 1. Using USWB TP-29, the rainfall amount for a 2-year recurrence interval storm with a 130-minute duration was found to be 53 mm (2.1 in.). For this example, an initial 10-minute storm intensity of 9.5 mm/hr (0.37 in/hr) was simulated using Equation 10. The first-order Markov model was then used to simulate the intensity pattern given in Figure 4.

Simulations were conducted using version 4.840815 of ANSWERS for six early spring scenarios. For each of the scenarios the antecedent soil moisture content was assumed to be 75 percent of saturation. All forested land was assumed to be in good managed condition, and the agricultural soils were assumed to be classified as having good productivity. The agricultural land was assumed to be planted in either corn or soybeans and both crops were considered to be in the seed bed cropstage. In this area corn is usually planted from the beginning to middle April and emerges in early May. Soybeans are planted towards the end of April and have not emerged by the beginning of May.

As a control, two runs were made in which conventional tillage corn and soybeans were each simulated. With conventional tillage, the fields were assumed to be turn plowed in the fall, with Spring disking and harrowing prior to planting. The cropland was assumed to follow a simple two-year corn and soybean rotation.

The next two scenarios were for no-till corn and soybeans. Again, a corn and soybean rotation was used with the no-till crops being planted in the previous crop's residue. Soybeans were assumed to be planted in 5000 kg/ha (4500 lb/ac) of residue corn stubble, while the corn was planted in soybean residue.

The last two scenarios were predominantly conventional tillage but select critical areas were no-tilled. Critical areas were arbitrarily defined as those cells having erosion rates greater than 30,000 kg/ha (14 tons/ac) during the first two conventional tillage scenarios for the given design storm. Individual critical elements were located, and the entire field containing the critical elements were changed to no-till. Field boundaries were defined from aerial photos and a field survey of the watershed.

Results and Discussion

The simulated design storm used in this study is presented in Figure 4. The design storm closely approximates observed rainfall distributions in the study area. The synthetic design storm was then used with the six scenarios described previously.

Table 1 is a summary of the BMP scenarios investigated. As shown, implementation of no-till practices on cropland in the watershed reduced predicted soil loss by 92 and 99 percent for the no-till corn and soybean scenarios, respectively. When no-till practices were used only on those fields containing areas with erosion rates greater than 30,000 kg/ha (18 percent of the total cropland), soil losses were reduced by 26 and 29 percent for the critical area no-till corn and soybean scenarios, respectively. In 1985, cost share funds for using no-till practices were $37.00 per hectare. Using this rate, the cost of reducing sediment loss in the no-till corn scenario was $25.80 per metric ton of soil loss reduced, and $16.50 per metric ton of soil loss reduced if only the critical fields were no-tilled. The scenarios involving soybeans predicted similar cost reductions.

Since cost sharing monies for off-site water quality benefits are limited, it is essential that they be applied in the most economical manner if off-site benefits are to be maximized. If the results from the Nomini Creek watershed example could be extrapolated to neighboring areas, and cost sharing funds were applied only to the most critical fields, then the cost effectiveness of the cost sharing program could be improved 36 to 39 percent in terms of sediment loss. Using the soybean simulations as an example, and assuming that $100,000 in cost sharing funds were available, 2530 additional tons of soil loss could be prevented if cost sharing funds were applied only to critical areas rather than to all fields.

Table 1. Tillage Scenario Cost and Soil Loss Comparisons.

SCENARIO		1 SOIL LOSS (kg)	2 PERCENT REDUCTION	3 COST ($)	4 SEDIMENT REDUCTION COST ($/ton)
CROP TYPE	TILLAGE PRACTICE				
Corn	Conventional	1570	--	--	--
	No-till	130	92	27900	25.80
	Critical Area	1160	26	5000	16.50
Soybean	Conventional	1520	--	--	--
	No-till	20	99	27900	24.70
	Critical Area	1080	29	5000	15.20

[1]Average soil loss/hectare of cropland.
[2]Percent reduction in soil loss compared to conventional tillage scenarios for the same crop.
[3]State cost sharing monies which would have been received for using no-till practices using the 1985 rate of $37.00/ha.
[4]Cost per metric ton of soil loss reduced.

It is important to note that these projections were based upon the results of a single design storm. If cost per ton of soil loss prevented were determined on an annual basis for all the storms that might be expected to occur through out the year, then the cost per ton would be reduced significantly because of the greater annual soil loss. It is also important to note that these projections were based upon a critical area being defined as a field containing an element with an erosion rate greater than 30,000 kg/ha (14 tons/acre) for the given design storm. Selection of different erosion rates for critical area identification could change these projections significantly.

PROPOSED PHOSPHORUS MODEL

A model for describing the transport of phosphorus in surface runoff is important for assessing the impact of BMPs on phosphorus yields. To model both the sediment-bound and soluble components of phosphorus transport it is necessary to model the particle size distribution of the eroded sediment since phosphorus is primarily sediment-bound and associated with the smaller and most easily transported sediment particles.

A block diagram of the proposed phosphorus model is shown in Figure 5. When the rainfall rate exceeds the infiltration and surface storage requirements, surface runoff begins and soil may be detached by overland flow and raindrop impact. During this turbulent mixing process, soluble phosphorus is desorbed into the surface runoff, and sediment-bound phosphorus is transported with the eroded sediment particles. Sediment may also settle out during the surface runoff process when there is a transport capacity deficit, which results in the deposition of the sediment-bound phosphorus. Within the surface runoff, the soluble and sediment-bound phosphorus are always in a dynamic equilibrium as the runoff makes its way to the watershed outlet, and to receiving waterways. To mathematically describe the transport of phosphorus in surface runoff, it is useful to first discuss the forms and availability of phosphorus within agricultural watersheds.

Forms and Availability of Phosphorus

Phosphorus existing in the soil and surface waters can be classified as particulate, or sediment-bound, and dissolved. Schaller and Bailey (1983) categorized sediment-bound phosphorus as:

1. Adsorbed: labile and exchangeable phosphorus.
2. Organic: various forms including phytins, phospholipids.
3. Precipitates formed from the reaction of phosphates with Ca, Fe, Al, and other cations.
4. Minerals: amorphous, short-range order and crystalline minerals with Ca, Fe, Al, and other cations.

Dissolved phosphorus exists as orthophosphate, inorganic polyphosphates, or as organic phosphorous compounds, while total phosphorus is the sum of sediment-bound and dissolved forms. Approximately two-thirds of the phosphorus occurring in the soil is inorganic (Shaller and Bailey, 1983), but the actual percentage is constantly changing due to the varying microbial decomposition of plant residue and other organic compounds within the soil system. Organic phosphorus is decomposed by microorganisms and mineralized to inorganic phosphate ions, which are available to the plant. Conversely, bacteria can immobilize these phosphate ions by converting them back to organic phosphorus.

The form of phosphorus entering surface waters is very important in determining the quantity of available phosphorus for aquatic vegetation. Soluble inorganic phosphorus is readily available while sediment-bound phosphorus is generally considered unavailable for algae growth in aquatic systems. Soluble phosphorus is transported by surface runoff and insoluble phosphorus is adsorbed to the soil particles and is transported with the eroded soil. During the transport process, there is a dynamic equilibrium between the soluble and sediment-bound phases of phosphorus. For example, a high concentration of soluble phosphorus and a low concentration of sediment-bound phosphorus, can result in the adsorption of the soluble phosphorus by the sediment. Conversely, under certain conditions sediment-bound phosphorus can be desorbed into solution.

A portion of the phosphorus in the soil is bound to the soil particles and is not readily plant available. To increase agricultural productivity, commercial fertilizers are often applied to the soil to increase the plant available phosphorus. However, when the fertilizer comes in contact with the soil, it is quickly converted into less available forms which are adsorbed to the soil particles. The soil pH governs the forms to which phosphorus is converted. In acidic soils phosphates are converted to iron and aluminum phosphates, and in alkaline soils calcium phosphates are formed (Novotny and Chesters, 1981). The method of fertilizer application, surface application or subsurface injection, and the type of fertilizer, liquid or solid, also influences the rate at which the phosphorus is converted. Other factors include tillage practice, temperature, vegetation, soil moisture, and soil type. Because of the high affinity of the soil for phosphorus, the downward movement of phosphorus in the soil profile is very slow. Thus, phosphorus is usually not a cause of groundwater contamination.

Vegetation as a Source of Phosphorus

A possible source of phosphorus in surface runoff is the leaching of phosphorus from live plant material and decaying plant residue. Most of the past work has been on the leaching of phosphorus from decaying plant material (Timmons, et al., 1970; White, 1973). Comparatively little work has been done on the leaching of phosphorus from live plants (McDowell, et al., 1980). Several studies have found that the amount of soluble-inorganic phosphorus in the plant leachate increased with plant age (Gosz, et al., 1973; McDowell, et al. 1980; Sharpley, 1981). Sharpley (1981) found that soil-water stress also increased the soluble-inorganic phosphorus in the plant leachate. The type of vegetation also effects the amounts of phosphorus leached (Burwell, et al., 1974; Gburek and Heald, 1974). However, more work is needed to quantify the amount and rate of desorption of phosphorus from various crops at during different growth stages before this process can be incorporated into the ANSWERS model.

Modeling Phosphorus Desorption From the Soil Surface

The desorption of sediment-bound phosphorus is important in the modeling of phosphorus desorption from the soil surface into surface runoff. By assuming that phosphorous desorption from the soil into surface runoff is diffusion controlled, Sharpley, et al. (1981b) developed a desorption equation of the form:

$$P_d = K\ P_o\ t^{\alpha}\ WS^{\beta} \tag{11}$$

where P_d is the cumulative phosphorus desorbed in grams of phosphorus per gram of soil, P_o is the initial amount of desorable Phosphorus in micrograms per g of soil, t is contact time in minutes, WS is the water to soil ratio in cubic centimeters per gram, and K, α, and β are empirical constants. The parameters K, α, and β in Equation 11 are dependant on the soil characteristics, and can be estimated experimentally in the lab. Sharpley (1983) developed general expressions for estimating these parameters using 60 different soils from across the United States, which correlate the parameters to the clay and organic carbon content of the soil. The expressions are given as:

$$K_L = 1.422\,(\text{percent clay/organic C})^{-0.829} \tag{12}$$

$$K_B = 0.630\,(\text{percent clay/organic C})^{-0.689} \tag{13}$$

$$\alpha = 0.815\,(\text{percent clay/organic C})^{-0.540} \tag{14}$$

$$\beta = 0.141\,(\text{percent clay/organic C})^{+0.429} \tag{15}$$

where K_L is the coefficient K corresponding to labile phosphorus status of the soil measured using isotrophic dilution with ^{32}P (Sharpley, 1983), and K_B is the coefficient K corresponding to Bray-I available soil phosphorus status, measured using procedures from Bray and Kurtz (1945).

Sharpley, et al. (1981a) and Ahuja, et al. (1982) developed an expression to describe the rate of phosphorous desorption by taking the derivative of Equation 11 with respect to time, which yields:

$$\frac{dP_d}{dt} = \alpha\ K\ P_o\ t^{\alpha - 1} \times WS^{\beta} \qquad [16]$$

and letting,

$$\frac{dP_d}{dt} = \frac{C\ I}{EDI\ \rho_b} \qquad [17]$$

where dP_d/dt is the time rate of change of phosphorus desorbed in micrograms of phosphorus per gram of soil per sec, C is the concentration of soluble phosphorus in runoff in micrograms of phosphorus per cubic centimeter, EDI is the effective depth of interaction in centimeters, I is the rainfall intensity in centimeters per second, and ρ_b is the soil bulk density in grams per cubic centimeter. Solving for the phosphorous concentration in solution yields:

$$C = \frac{\alpha\ K\ EDI\ \rho_b\ P_o\ t^{\alpha - 1}\ WS^{\beta}}{I} \qquad [18]$$

The desorption of phosphorus from the soil surface into surface runoff is initiated by turbulent mixing caused by raindrop impact and overland flow. In Equation 18 the EDI represents the thin layer of soil that interacts with rainfall to release soluble phosphorus into solution. Ahuja, et al (1981b) used ^{32}P, a relatively immobile tracer, to determine the depth of interaction for phosphorous desorption. They found that the EDI increased with time, and concluded that the EDI was more dependant on the storm duration, than on the soil type. In a similar study using a bromide tracer, Ahuja and Lehman (1983) found that the contribution of chemicals released into surface runoff decreased exponentially with soil depth. Sharpley (1985) found that the EDI increased exponentially with increasing slope, increased linearly with increasing rainfall intensity, and found that these increases were independent of soil type. Sharpley (1985) and Sharpley, et al. (1981a) found that the degree of soil aggregation also effected the EDI, as well as the magnitude of the effect of rainfall intensity and slope on the EDI. Sharpley (1985) found that the EDI was not related to the degree of aggregation when wheat straw was incorporated into the soil, and found that as the percent cover increased the EDI decreased. Ahuja (1982) also found that soil cover decreased the EDI.

Ahuja and Lehman (1983) hypothesized that the transport mechanism of phosphorus to surface runoff is a turbulent diffusion process caused by rainfall impact. This mechanism implies that as the hydrologic conductivity of the soil increases, the depth of phosphorus contribution from the soil increases, along with the total amount transferred. In addition, as the canopy and ground cover increase, the amount of phosphorus transferred decreases. A general expression for estimating the EDI was developed by Sharpley (1985) over a wide range of rainfall and management practices, and is given as:

$$\ln(EDI) = -3.130 + 0.071\,(\text{soil aggregation}) + 0.576\,\ln(\text{soil loss}) \qquad [19]$$

On a practical note, when modeling the desorption process the EDI is usually assumed constant. As Sharpley (1985) points out, a constant EDI is a simplification of a complex physio-chemical process, and will not exist over an entire watershed under normal conditions. However, for many applications a constant EDI must still be used until more complex quantitative expressions for the EDI are developed.

Modeling Sediment-bound Phosphorus

Eroded soil usually contains a higher proportion of clay and fines than the original soil mass. This selective erosion of fines occurs because the fine soil particles are eroded and transported more readily than course particles. In addition, larger particles tend to be deposited first, due to their higher settling velocity. As a result, the eroded soil usually has a higher concentration of nutrients, due to the higher ion exchange capacity of clays and fines. This nutrient enrichment can be expressed as an enrichment ratio, which is the concentration of the nutrient in the eroded material, divided by the concentration of the nutrient in the original soil mass. The loading of sediment-bound phosphorus can be expressed as (Frere, et al., 1980):

$$P_s = ER \ P_t \ SED \qquad [20]$$

where P_s is the sediment-bound phosphorus transported by surface runoff in grams, ER is the phosphorus enrichment ratio, P_t is the total phosphorus content of the soil surface in grams of phosphorus per gram of soil, and SED is the sediment transported by surface runoff in grams. The phosphorus enrichment ratio can be estimated by:

$$ER = \frac{SSA_{sed}}{SSA_{soil}} \qquad [21]$$

where SSA_{sed} and SSA_{soil} are the specific surface areas of the eroded and original soils in square meters per gram, respectively. The specific surface area is the surface area of the soil particles divided by the mass of the soil particles.

Modeling Phosphorus Adsorption/Desorption in Surface Runoff

The desorption of phosphorus from soil occurs in two distinct phases. The first desorption phase is very fast, taking minutes to hours, and the second desorption phase is slow, taking days to months. However, in many cases the adsorption of phosphorus by soil can be assumed to be instantaneous. An equation commonly used to describe the equilibrium conditions of phosphorus reactions is the Langmuir isotherm.

The Langmuir equation was initially derived to describe the adsorption of gases by solids (Langmuir, 1918). The equation is valid for a monomolecular layer and assumes a constant energy of adsorption, which is independent of surface coverage. The equation also assumes no interaction between adsorbate molecules, and that a maximum adsorption exists when the reactive adsorbent surface of the monomolecular layer is filled. The Lamgmuir equation is often written in the form (Tchobanoglous and Schroeder, 1985):

$$\frac{x}{m} = \frac{Q_o \ b \ C_e}{1 + b \ C_e} \qquad [22]$$

where x is the mass of material adsorbed (adsorbate) on the solid phase in grams, m is the mass of solid (adsorbent) on which adsorption occurs in grams, C_e is the equilibrium concentration of adsorbate in grams per cubic meter, Q_o is the adsorption maximum in grams of phosphorus per g of soil, and b is a constant related to binding strength of adsorbent in cubic meters per gram. The coefficients Q_o and b may be estimated experimentally, or estimated from the general relationships developed by Ryden, et al. (1972):

$$Q_o = -3.5 + 10.7\,(\text{percent clay}) + 49.5\,(\text{percent organic C}) \qquad [23]$$

$$b = 0.061 + 169832 \times 10^{-pH} + 0.027\,(\text{percent clay}) + 0.76\,(\text{percent organic C}) \qquad [24]$$

The major advantage of the Langmuir equation is that the equation has an adsorption maximum, and can thus be used to describe the adsorption capacity of soil for phosphorus. However, the Langmuir equation assumes a constant energy of adsorption with increasing surface coverage, which is not likely to occur in nature. However, according to Bohn, et al. (1979), in some cases as the reaction sites are filled the energy of adsorption decreases, and the interaction of the adsorbent molecules increases. These two effects tend to cancel each other, which results in an approximately constant energy of adsorption. On a practical basis, when the Langmuir equation applies, it is limited to the range of the experimental data.

When modeling the phosphorus transport on a watershed scale, the erosion process and desorption of soluble phosphorus into surface runoff will need to be modeled separately. Thus, there is a need for a nonequilibrium expression for modeling the soluble and sediment-bound phosphorus concentrations in surface runoff. One possible approach is to modify the Langmuir equation as follows:

$$\frac{P_o + \Delta P}{SED\ SSA_{sed}} = \frac{Q_o\, b\, (C_o\, V - \Delta P)}{1 + b\, (C_o\, V - \Delta P)} \qquad [25]$$

where P_o is the initial sediment-bound phosphorus in grams, ΔP is the amount of phosphorus adsorbed/desorbed in grams, C_o is the initial soluble phosphorus in the surface runoff in grams per cubic meters, and V is the volume of flow in cubic meters. Equation 25 represents a pseudo mass balance, where ΔP represents the mass transfer of phosphorus from the liquid to the solid state. Given the initial soluble and sediment-bound phosphorus concentrations, the solution of Equation 25 can be obtained by solving for ΔP. Once ΔP is known, the final phosphorus concentrations can be calculated as:

$$P_f = P_o + \Delta P \qquad [26]$$

$$C_f = \frac{C_o\, V - \Delta P}{V} \qquad [27]$$

where P_f is the final sediment-bound phosphorus in grams, and C_f is the final soluble phosphorus concentration in grams per cubic meter.

SUMMARY AND CONCLUSIONS

The use of the ANSWERS model was demonstrated for allocating cost sharing monies to achieve maximum off-site benefits. Use of the model was found to improve the effectiveness of allocating cost sharing monies 36 to 39 percent for the example presented. A procedure was developed for generating a synthetic design storm of variable intensity for use in hydrologic models requiring variable rainfall intensity data as an input. The procedure synthesizes a design storm of the required recurrence interval and duration with selected statistical characteristics similar to those of the natural precipitation records from which it was produced.

Also presented are the equations necessary for the development of a phosphorus transport model which can be incorporated into the ANSWERS model. The phosphorus model accounts for the desorption of phosphorus from the soil into surface runoff, the transport of sediment-bound phosphorus, and the adsorption/desorption process between the soluble and sediment-bound phases in surface runoff. Also included are brief discussions on the forms and availability of phosphorus, vegetation as a source of phosphorus in surface runoff, and the development of the phosphorus transport equations.

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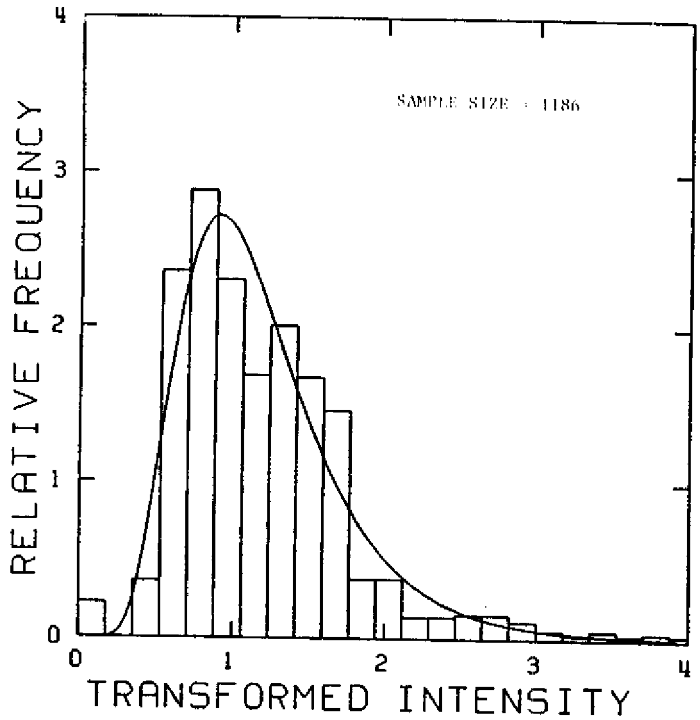

Figure 1. Histogram of May rainfall intensity transformed by the cube root for 10-minute intervals, with a lognormal density function fitted to the data.

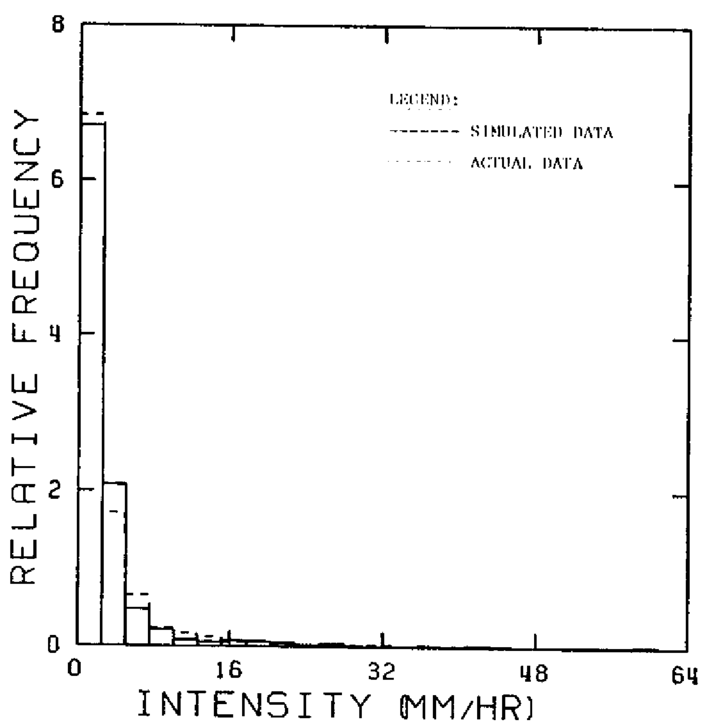

Figure 2. Histogram of simulated and actual 10-minute intensities for the month of May.

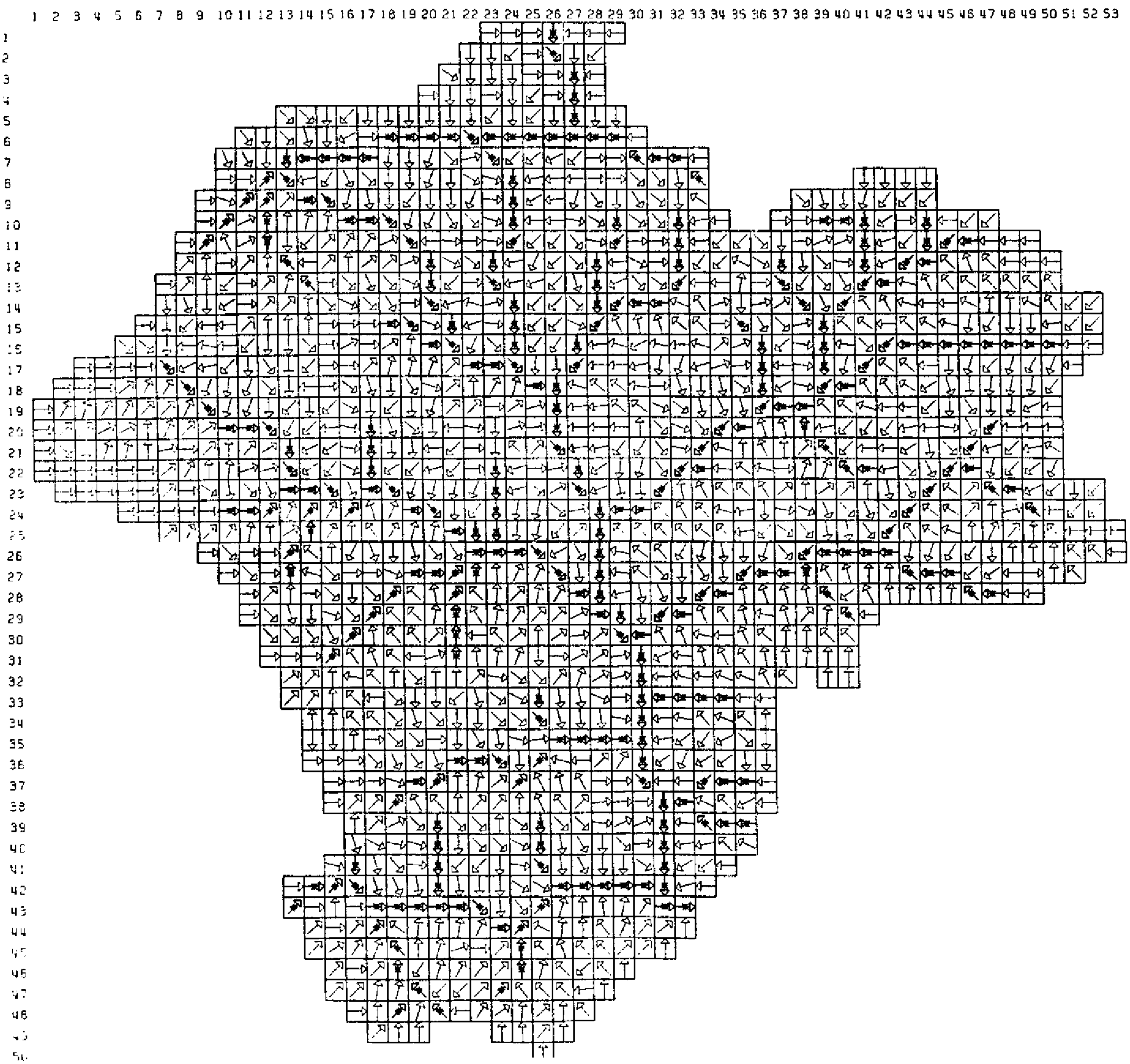

Figure 3. Westmoreland County watershed partitioned into one-hectare grids with flow directions and channel elements.

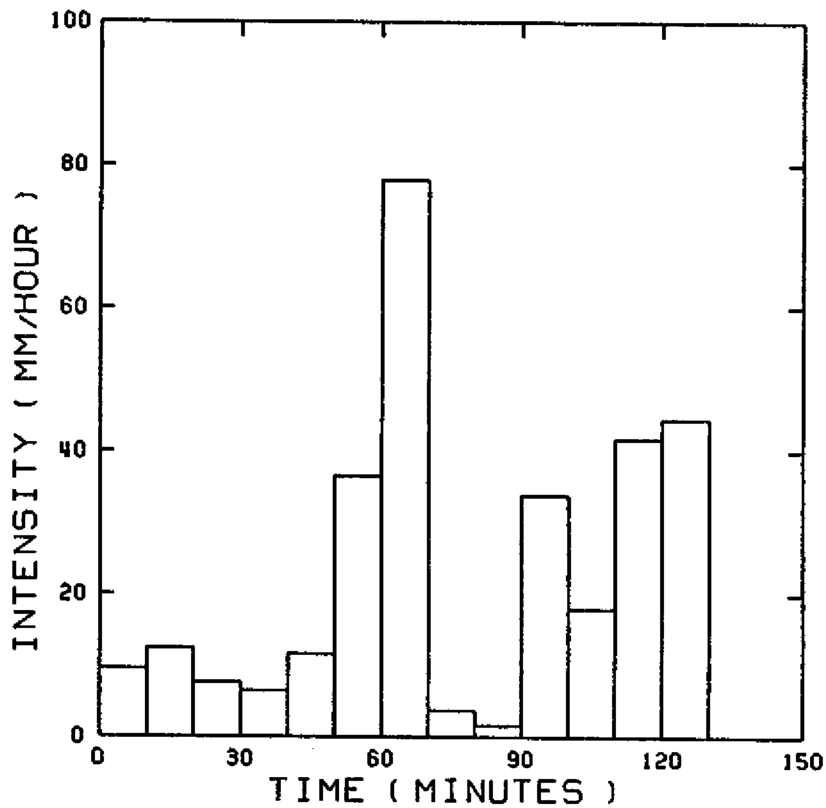

Figure 4. Simulated intensity pattern.

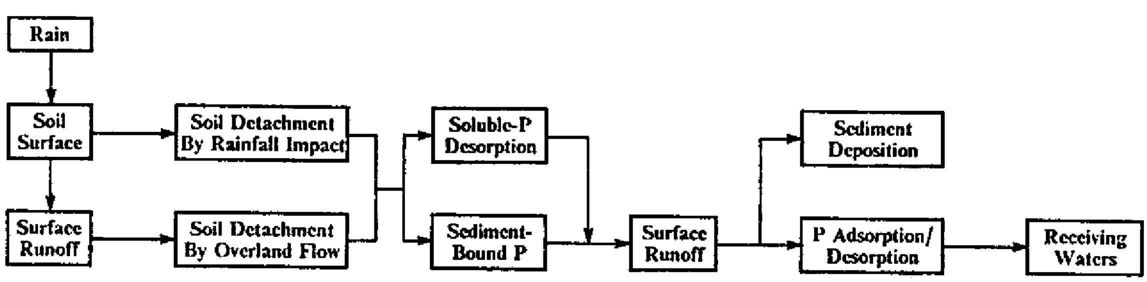

Figure 5. Block diagram of phosphorus transport process.

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OVERVIEW OF BMPs FOR CONTROLLING ANIMAL WASTES

by
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ABSTRACT

Waste generated in livestock and poultry operations has potential to impact on surface and groundwater supplies as point and nonpoint source pollution. Animals produced in confinement buildings generate large quantities of wastes which are generally disposed of by land application. Often insufficient land is available to assimilate the manure nutrients at a crop utilization rate. Manure storages at the point of production (point source) also contribute to the nonpoint source problem, and after manure is land applied, it offers the threat of being washed or leached into receiving waters. Animals which are maintained wholely or partially on pasture also may contribute to nonpoint source pollution by directly depositing wastes in streams and lakes, or by creating denuded areas or disturbed crossings and stream banks, or other areas which are easily eroded and contribute to turbidity and sedimentation problems.

Best Management Practices (BMPs) are a practice, or combination of practices, which are deemed to be the most effective practicable means of preventing or reducing the amount of pollution generated from nonpoint sources to a level compatible with water quality goals. Most often these measures have been suggested as common-sense approaches which many farm operators employed long before nonpoint source pollution became a concern. However, in recent years regulatory agencies, extension workers, other farm service personnel, and farmers have given BMPs increased emphasis as a possible means of reducing agricultural contributions to water quality problems. This paper will present an overview of the major related animal waste BMPs normally practiced in livestock agriculture.

INTRODUCTION

Development and management of animal waste handling systems were important concerns of Extension and other public service workers in the United States during the 1970's. Many livestock and poultry producers were forced by state and federal agencies to confront mounting pollution hazards from manure and other wastes generated by their operations. A new and active U. S. Federal Environmental Protection Agency (EPA) promulgated the National Pollution Discharge Elimination System (NPDES) which helped focus programs of many state pollution control agencies on concerns about pollution from agricultural sources. Educational emphasis, for the most part, tended to be placed on development of control and management structures and/or systems for manure disposal. Although application of manure to the land was usually a part of such systems, it was generally considered to be tied to *disposal* rates rather than *utilization*.

There is general recognition that agriculture is placing considerable demands on our underground and surface waters, especially through nutrient, organic matter, and sediment loadings. However, surface runoff and groundwater seepage may also contribute to pathogen, chemical, drug, heavy metal, and pesticide loadings. A few simple calculations, and some knowledge about how many farmers manage their crop fertilization programs, indicate that manure management in many instances may have a significant impact on this problem. Methods, referred to as Best Management Practices (BMPs), have been promoted among farmers and others to aid in controlling point and nonpoint source pollution from agricultural sources. These practices are voluntary, common-sense measures which, in many instances, are already being employed by farmers.

Water pollution from animal production occurs from both point and nonpoint sources. Point source pollution is traceable to a well-defined source such as a feedlot or waste disposal lagoon. More often, though, pollution from livestock can best be categorized as nonpoint source simply because authorities don't know about the location of all livestock operations. So, water pollution from animal production is usually classified as nonpoint source pollution, and is the result of storm runoff and groundwater seepage from ill-defined areas.

BMPs: SOLVING NONPOINT POLLUTION PROBLEMS

Federal and state programs have encouraged livestock and poultry producers to voluntarily implement BMPs. A broad range of BMPs have been suggested including management, structural, and agronomic practices which have potential to control or abate nonpoint source pollution. Sometimes these practices are based on research which has conclusively established their effectiveness. Often BMPs are common-sense measures which many producers already employ to reduce erosion, and projected reduction in pollution levels is difficult to quantify. It is generally accepted, however, that BMPs should maintain or improve productivity while controlling nonpoint pollution if properly installed and maintained.

Assistance is available to help producers choose combinations of BMPs to suit specific farming operations. Extension agents, and specialists from state land-grant universities have traditionally provided educational advice and information to help farmers evaluate pollution potential from their individual operations, and to select BMPs and other measures for customizing their waste management and farm water quality plan. Technical assistance for the planning, design, and implementation of BMPs is available from local USDA Soil Conservation Service and through local Soil and Water Conservation Districts (SWCDs). More recently, SWCDs have begun to provide personnel who may also provide technical advice and assistance. Cost sharing for waste management practices has long been provided through local Agricultural Stabilization and Conservation Service (ASCS) offices to encourage installation of pollution control measures on farms. More recently, additional funds have been provided from state and federal sources through

SWCDs for those areas having direct impact on the Chesapeake Bay and Chowan River Basin.

BMPs for Pasture Management:

Pasture land used for open grazing must be well managed to maintain a dense, healthy grass cover to prevent erosion and runoff. Increased animal traffic around feeding and watering facilities makes these areas particularly susceptible to erosion, so they should be located well away from streams, drainageways, and other water bodies. Portable shades, hay feeding racks and similar facilities should be periodically moved about the pasture area to prevent overgrazing and denuding of any area. Pastures should be rotated as a management tool for maintaining good vegetative cover.

In operations where streams flow through pastures, stream bank erosion and waste deposition can become a problem. This is especially true when heavy traffic patterns develop such as where cattle leave feedlot and barn sites, and must pass through a stream while moving to pasture areas. Cross-over culverts or bridges combined with fencing may be the only means of alleviating these problems. Where heavily grazed areas or feedlots are present, fencing may be necessary to keep livestock out of an area that is easily eroded, such as a fragile stream bank or a steep slope, or to extend the life of a farm pond. Proximity of public water supplies or shellfish areas will require special attention to control of runoff from livestock production areas, and management of fields which receive applications of manure.

High Animal Density Areas:

Barns, feedlots, and other areas where animal density is great must be given special attention. While such areas can be more technically termed "point sources", they also represent heavy concentrations of wastes with great potential for pollution. They are also much easier to identify and devise control measures for than many nonpoint source problems.

Uncontaminated water from land above high density facilities should be intercepted and carried around livestock facilities to reduce the need to collect and dispose of polluted runoff. In other words, the important principle is to KEEP CLEAN WATER CLEAN. But even with diversion of offsite runoff, considerable runoff from animal lots can occur. Where lot or facility runoff may occur, collection channels and retention basins offer a method of collecting the polluted runoff, settling heavy solids, and holding liquids until field disposal by irrigation may occur.

Open confinement lots often are improperly located near streams and drainageways. One means of reducing pollution potential of these facilities is to develop properly designed grass filter strips which, when properly maintained between these areas and receiving water bodies, will trap sediment and other pollutants. Design procedures have been developed for filter strips, but their use is no guarantee that pollution problems will be elimimated from facilities established at a poor site. An ideal location for any high density production area is well away from streams, lakes, or other water bodies, at or near the top of a slope, and out of floodplains. If possible, open lots should be laid out with the narrow dimension in the direction of the greatest slope. This reduces the effect of slope length on erosion potential.

Although not a direct factor in water pollution, proximity to neighbors, access to facilities, and prevailing wind direction should be considered in development of new livestock housing, and in the selection and location of waste handling and disposal facilities. Otherwise, nuisance problems may develop.

BMPs for Confinement Facilities:

Buildings for environmentally controlled production of livestock and poultry should include a well conceived plan for handling and disposal of waste products. Waste problems can usually be eliminated if adequate manure collection, storage and treatment facilities are provided and maintained. As above, buildings and waste facilities should be located out of floodplains and away from streams and lakes, with diversions for upslope "clean" water. In Virginia, waste facilities should be properly designed and constructed to provide adequate storage and treatment capacity. All plans for construction and operation of animal waste handling facilities must be approved by the State Water Control Board; other states with watersheds draining to the Chesapeake Bay have somewhat different permitting procedures.

Liquid manure handling and storage systems have become popular in confinement production facilities. Advantages of liquid systems include flexible storage capacity, convenience, and preservation of nutrients. Liquid storage structures are usually either steel or concrete "silo-type" or clay-lined earthen basins. All liquid storage and/or treatment systems should include diversions for protection from flooding or filling with uncontaminated runoff. However, where contaminated runoff from lots and holding areas may occur, facilities should be planned for the needed extra storage in addition to manure produced. Plans should also be included for storage of milking parlor wastewater. Where manure pits or open manure collection gutters will be flushed, recycled water from lagoon systems should be used in order to reduce demand on fresh water supplies and to reduce the total volume of waste generated. Operators should periodically inspect all waste management components and facilities, and perform needed maintenance in order to prevent waste spills.

Where "solid" manure handling and storage is used, runoff from waste collection, handling, and storage areas should be intercepted and directed towards a holding basin or lagoon for later land disposal. Where possible, above-ground storage areas should be covered by a roof and should include diversions to keep clean runoff from entering the storage area. If the storage is not roofed, special attention must be given to intercepting and storing polluted runoff. Roof gutters can help collect precipitation and channel it, unpolluted, out of manured areas. Storage areas should be located for year-round access to permit spreading when field and weather conditions permit.

Livestock and poultry producers at times are faced with disposal of dead animals and birds. Carcasses must be disposed of in a manner which will prevent the spread of disease organisms. Acceptable methods include burying and sealed disposal pits (if soil and groundwater conditions are suitable), grinding and composting, rendering, and incineration.

BMPs for Animal Waste Utilization:

The spreading of livestock and poultry manure on land has been a time-honored, convenient disposal method that benefits the soil system. Overall, U. S. agriculture uses more than 10 million tons of chemical fertilizer nitrogen annually. Manures can provide about 45% of this amount, or about 28% after allowing for storage and handling losses; the magnitude of losses depends on the method of handling and management involved. The problem in recent times has been the concentration of increasing numbers of animals and poultry on small parcels of land, so that nutrients returned to land (often in addition to commercial fertilizers applied) far exceed crop requirements. In such cases , surface runoff and groundwater pollution problems develop. Proper handling of manures in such cases becomes a challenge to return manure nutrients to the land at a rate balanced with the ability of plants to utilize them. Otherwise, alternative ways must be found to convert the manure to other usable and environmentally acceptable forms.

The popularity of large, liquid manure storage structures was mentioned above. Pollution officials, however, have begun to express concern that construction of "bigger and better" waste storage facilities may not be helping to reduce nutrient loadings on area streams and groundwaters. They believe that, in many cases, manure is accumulated in large quantities, and then spread on fields and pastures, often at times when the ground is frozen or when conditions are optimum for either leaching or runoff. Therefore, the farmer assistance programs to encourage construction of storage facilities may not be addressing a major problem---excessive and untimely nutrient applications to crops and pastures, which then impacts upon groundwater and streamflow, ending ultimately in the Chesapeake Bay.

Stories continue to appear in newspapers and magazines about cases of farm water supplies in the Midwest, and elsewhere, exceeding drinking water standards for nitrate. In 1984, spot studies in Virginia's Shenandoah Valley showed that five percent of the wells tested on one representative small creek tributary were registering nitrate levels above the public health drinking water standard (10 ppm). Fifty percent of the wells tested were over half the standard. Considering the intensive agricultural use of the area, both crop and livestock/poultry, it is likely that the situation will worsen.

Challenge for the Future- Optimum Utilization of Animal and Poultry Wastes:

Because of the lower costs usually associated with land application and the nutrient benefits derived by crops from manure, this method of utilization will continue to be the mainstay of effective and safe manure disposal on most farms. There are a number of key objectives, however, which must be taken into account in developing a manure management plan. These include:

1. Number of animal units (1000 lbs live weight), land area available, and type of soil available for application-- Determines the frequency and rate of application.
2. Characteristics of the manure-- This is influenced by the specie of animal/bird, as well as type of storage/handling and management used. These factors, in turn, determine the amount of plant available nutrients, especially nitrogen, in the manure for determining the application rate.
3. Topography of application area-- Slope of the application area and position relative to farm ponds, streams, and drainage ditches; impacts on the potential nutrient loss and pollution hazard.
4. Type of crops and rotation-- Rate, time, and method of application will relate to the types of crops to be grown, cover crops, and crop rotations used.
5. Climatic conditions-- Wind conditions, temperature, form of precipitation, impending weather, antecedent soil moisture, and winter application on frozen ground will affect nutrient loss and potential water pollution.

Nitrogen is often the factor which limits the amount of manure which should be applied to pastures and cropland. If levels greater than required by the crop or pasture are applied, nutrients may enter surface water either by direct runoff or subsurface interflow, or may leach into the groundwater. In order to supply the nitrogen needs of the crop, while minimizing the above hazard to surface and groundwater, manure should be applied at rates not exceeding crop requirements. This will depend on the type of crop, soil type, and yield goals established. Such information is usually available in guides available from agronomy departments at most land-grant universities and local Extension offices.

A number of excellent discussions and worksheets are available for computing safe levels of manure utilization (Klausner, et al., 1985; Midwest Plan Service, 1985; White, et al., 1980; USDA/EPA, 1979). However, one usually finds there are slight differences in each of the methods because of the assumptions made. This results in wide differences in calculated field nutrient loading rates. Some of the factors related to these differences are:

1. Differences in estimating manure nutrient content based on animal species, method of management, and storage;
2. Effect of season of the year (mean temperature, precipitation);
3. Problems with determining organic nitrogen mineralization rate.

Probably the best approach is to obtain the services of a testing lab to determine representative nutrient levels for manure from the particular operation prior to computing the land application rate. If tests remain fairly stable for several years, it may be possible to skip testing every other year, but testing should never be abandoned altogether.

Manure is also a good source of phosphorus and potassium. Although manure applications are generally based on nitrogen, phosphorus is sometimes used as a basis, with additional nitrogen added in chemical fertilizer form to make up the difference required by the crop. Excess application of phosphorus is usually not a problem because of the tremendous capacity of the soil to absorb and hold it. However, if the land has received over-applications of phosphorus for many years, test levels may be in the medium to high range, and it might be wise to shift to a phosphorus basis for determining manure application rates.

Similarly, potassium is tightly bound by the soil, and over-application is not usually a problem. When potassium levels reach 5 % in the feed of ruminants, however, tetany problems may be encountered. In such cases, a potassium basis might be used to calculate manure application rates. Where soil potassium may be high, or tetany may present a risk, plant tissue analysis should be made to determine the extent of the problem.

Manure should be plowed or disked in as soon as possible to minimize nitrogen loss and to begin release of nutrients for plant use. Since most losses occur within the first 24 hours after application, disking, injecting, chiseling, or knifing manure into the soil will minimize odors and nutrient losses to the air and to runoff. Nitrogen losses, through ammonia volatilization, are lower during humid or cold days than during dry, warm, windy days. So, ammonia losses are generally greater during the Spring and Summer. Poultry manures are highly alkaline, so ammonia losses are greater than from other manures; this makes quick soil incorporation especially important in these cases.

Manure should be applied as near the planting date as possible so more nutrients will be available to plants, especially in areas of high rainfall and with soils prone to nitrate leaching or denitrification. But lowered germination and reduced seedling growth could occur if planting is done immediately after heavy manure applications, especially when high ammonium or inorganic salts may be present, as with poultry manures. Fall-Winter applications may mean 25-50 percent nitrogen losses from leaching and denitrification, but the extra time before planting allows microorganisms more time to decompose organics and release nutrients.

Course-textured soils generally decompose manure faster that fine-textured soils because of better air and water movement. Fine-textured soils, however, tend to retain more nutrients in the upper soil profile. Soil physical properties also affect waste application rates; fine-textured soils, because of lower water infiltration rates, require limited waste application rates to avoid runoff problems. And, while coarse-textured soils will accept higher liquid waste application rates without runoff, their lower exchange rates require lower, more frequent application rates throughout the growing season to reduce nitrate leaching. Nitrification inhibitors are available, and have been tested with some liquid manures to reduce leaching losses, especially when used with fall applications.

One problem often observed in the field is that farmers are reluctant to give full credit for nitrogen from manure when planning fertilizer needs. This is also true for legume crops such as alfalfa. As a result, more nitrogen is often applied than the crop can utilize, resulting in leaching losses of nitrogen to groundwater and stream baseflow. More effort and innovation is needed in helping farmers to properly utilize this resource, thereby saving them money, while reducing water pollution.

PRODUCTION OF BYPRODUCTS

Biogas (Methane):

The technology needed to generate biogas (approximately 60% methane, 40% carbon dioxide) has been know for over 100 years. Inexpensive and abundant petroleum energy has hindered adoption of this energy source in the U. S. A good general discussion of biogas systems and their design is provided by Parsons (1984). Advantages of anaerobic digestion are:

1. Production of biogas with significant value as a fuel (approximately 600 btu per cu.ft.);
2. Considerable reduction of odor potential of digested effluent and sludge;
3. Conservation of manure nutrients; nitrogen in more available form;
4. Rodents and flies not attracted to digested effluent;
5. Sludge easily dewatered for other uses.

There are, however, some major disadvantages:

1. Equipment is large, expensive, and semi-experimental;
2. High standards of maintenance and management required;
3. Process is sensitive to temperature, pH, loading rates, and changes in input materials;
4. Few cost-effective uses have been found for biogas;
5. Digester systems will not significantly reduce the volume of manure to handle, although solids will be reduced;
6. Digested material may still be a pollutant.

Anaerobic digester systems continue to interest producers and others, and will move onto the American farm as energy costs increase and economic payback improves. As an indicator of the energy yield possible from a laying hen operation, one pound of manure should yield from 1.3-1.7 cu.ft. of biogas per day. Therefore, a 100,000 bird flock could potentially produce 36,000 cu.ft. of biogas per day, an equivalent of 21,600,000 btu's of energy (235 gallons of propane). However, a layer operation would not likely need that much heat energy, and the gas is difficult to store. Electrical cogeneration has been suggested as one of the best ways of using the gas produced. For constant output (24 hours per day), this operation could support a 68 hp generator set producing 45 kwhr, or 8-hr operation of a 203 hp set producing 135 kwhr.

Composting:

Composting is a means of biologically stabilizing organic material to a relatively stable humus-like material. Both anaerobic and aerobic composting processes can be used, but modern composting usually involves aerobic systems; air is introduced either through windrowing and turning, or forced aeration. Composting has the following benefits:

1. Stabilization of putrescible organic matter;
2. Destruction of pathogens and seeds;
3. Conservation of nutrients and resistant organic material in the raw material;
4. Production of a uniform, sterile, relatively dry, odor free end product;
5. Production of a soil conditioner and fertilizer;
6. Production of a material more suitable for transport to other distant disposal sites.

Disadvantages include:

1. Expense for equipment and operation;
2. Loss of nitrogen (ammonia), especially when Carbon:Nitrogen ratios are below 20:1;
3. Over-aeration may retard or stop microbial activity.

A number of commercial systems are available for composting manure. Many of these have worked conceptually, but long-term mechanical difficulties have developed. Feasibility for a large number of producers is unlikely; there is a price associated with compost processing. Some proponents suggest that a market could be developed for poultry compost. Even though the product is an excellent humus/fertilizer, it will often be competing with other products in the gardener/homeowner market which are not as valuable, but cost less to produce and market. Unless buyer recognition for its superior qualities can be developed, it is unlikely that large quantities of manure can be marketed in this form.

Refeeding:

In areas where cattle and sheep feeding operations coexist with poultry enterprises, there is limited opportunity for recycling manure as ruminant feed. Proven processes have been developed for direct incorporation into diets, and for ensiling manure with crop residues. Manure with high copper content may contribute to copper toxicity when fed to sheep, so care must be taken in ration formulation. In cases where refeeding may be practiced, manure may be more valuable as a feed ingredient than as a source of crop nutrients.

SUMMARY

Waste generated in livestock and poultry operations has potential as both point and nonpoint source water pollution. Land application of manure will continue to be the mainstay of agricultural waste management, but improved management of nutrients from organic, plant, and commercial fertilizer sources is needed. Best Management Practices offer the most practicable means of preventing or reducing the amount of pollution generated from animal sources to a level compatible with water quality goals. A broad range of BMPs have been employed including management, structural, and agronomic practices which have potential to control or abate pollution from animal and poultry sources. Sometimes these measures are based on conclusive research which has established their effectiveness. Often, however, BMPs are common-sense measures which many producers have employed for years to reduce erosion, and reduction in pollution levels is difficult to project or quantify. It is generally accepted, however, that BMPs maintain or improve productivity while controlling nonpoint source pollution.

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MARINA SITINGS FROM THE SCIENTIFIC ADVISOR'S VIEWPOINT

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ABSTRACT

The institutional framework of regulatory agencies involved with management of marina development is multi-layered and, at the local level, variable. The variety of agency purviews are not seen to be well coordinated so as to ensure consistent thorough reviews of all development plans. An absence of comprehensive information for impact assessment further complicates the process. Since the information deficit is not amenable to solution, better interagency coordination and establishment of long range management objectives are proposed as ameliorative steps toward resolution of the existing management dilemma.

INTRODUCTION

Shoreline development projects frequently raise a wide variety of concerns which run the gamut from environmental to economic. In the Commonwealth of Virginia, review, assessment and regulation of proposed projects can involve a diverse and variable group of special interest groups, advisory groups and regulatory agencies. One type of project which routinely involves almost all the concerns and groups is marina development. As such, these projects provide interesting case studies of the current efforts in Virginia to manage shoreline development and the estuarine resources of the Commonwealth.

Management of marina development places a premium on establishment of long range management objectives and short term impact assessments. This is a result of the large number of regulatory agencies involved and

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The opinions expressed in this paper are solely those of the author and in no way represent an official position of the Virginia Institute of Marine Science.

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the conflict marina development presents between development and preservation goals. It is the purpose of this paper to: 1) briefly review the institutional framework and impact assessment related to management of marina development, and 2) assess their respective effects on the management process. The review of these topics is based on the author's personal experiences with the process. The intent is to highlight some of the problem areas and suggest some possible modifications.

INSTITUTIONAL FRAMEWORK

Regulation of marina development involves a diverse group of agencies arrayed over three levels of government (i.e. local, state and federal). These agencies operate from unique perspectives with overlapping, although not necessarily coordinated, purviews. The regulatory agencies are in turn, supported to varying degrees by a variety of advisory agencies. The multiplicity of involved parties occasionally results in very thorough reviews of proposals and occasionally results in disjunct, discordant reviews. The dichotomy seems to result from the lack of a protocol for coordination of all agency reviews.

At the local level, a proposal for marina development will be reviewed by the Wetlands Board, the building inspector, the local health department and the local zoning board, if one exists. The purview of the building inspector and local health department are relatively well defined and limited. The design of structures and upland site development are regulated by the building inspector. The health department regulates potable water supply and sewage disposal. The Wetlands Boards purview is less well defined. In its most narrow construction, it covers any activities in the intertidal zone, which is specifically defined based on the presence/absence of vegetation. Broader construction of the purview has allowed boards to consider and/or regulate development in either waterways or on land, which may impact the intertidal resources. The local zoning board possesses the most extensive purview in terms of area and activities. One difference between zoning boards and other local agencies is the clear charge to the zoning board to consider the appropriateness of individual projects based on surrounding land uses, either existing or planned. Typically, there is no coordination among local regulatory agencies as far as management objectives are concerned. Particularly in rural areas coordination which does occur is frequently serendipitous, in the form of individuals who serve in multiple roles.

At the state level, proposals for marina development can initiate reviews by the Virginia Marine Resources Commission (VMRC), the Health Department and its Bureau of Shellfish Sanitation, and the State Water Control Board (SWCB). The common aspect of the review conducted by each of these agencies is a concern for water quality and its impacts on marine/estuarine resources and/or human health. Coordination among these agencies at the level of individual projects exists in the form of timing of permit issuance but does not typically involve in depth consultations or concerted efforts to share information or expertise. The state agencies effectively operate independently, regulating development in an effort to achieve agency-specific objectives.

At the federal level review and regulation of marina development is operationally vested in the United States Army Corps of Engineers (Corps). The Corps is supported in its review by advice from the Environmental Protection Agency, National Marine Fisheries Service and the U.S. Fish and Wildlife Service. The Corps review focuses on environmental effects and navigation considerations. Within these areas of concern its purview effectively overlaps that of both the local and state agencies (especially wetlands boards and the VMRC). In Virginia there is a formal procedure for coordination of project reviews by the four federal agencies and the state agencies involved. The process is limited, however, to case-by-case sharing of information and does not extend to thorough coordination of federal and state management objectives. Local agencies do not typically participate in the federal/state review or coordination.

IMPACT ASSESSMENT

The potential impacts associated with marina development can be broadly categorized as economic, social or ecological. This classification is arbitrary and only for purposes of this paper. The importance attached to these different types of potential impacts is variable, dependent on both the setting and local interests. Not all potential impacts are reviewed for each proposed project. Again, local interests generally influence the scope of the assessment.

Each type of potential impact encompasses a number of specific items. Assessments of the extent of impact are generally proffered by a number of interested parties and can be either quantitative or qualitative in nature. Table 1 summarizes the types of potential impacts, the sources of assessments usually relied upon and the type of assessment. The general types and their relative importance are reviewed below. (For more detailed treatment refer to the Coastal Marina Assessment Handbook prepared by the U.S. Environmental Protection Agency, 1985.)

Economic Impacts

Economic impacts are generally amenable to quantification, although in cases of cash flow the assessments are usually educated estimates. Typically economic impacts assume their greatest importance in management decisions when either the local government is interested in promoting economic development or adjacent property owners are concerned about diminished property values. In the absence of these interests, economic impacts do not typically receive overt considerations.

Social Impacts

Adjacent property owners and local interest groups usually ensure social impacts are at least aired in the resource management forum. With few exceptions, assessment of social impacts is not amenable to quantification. Regulatory agencies at all levels typically receive at least some information on social impacts but none of them possess a formal protocol for evaluating these impacts and weighing them against other considerations. The author's personal experience indicates that social impacts are of greatest importance at local and state levels. Both the local wetlands board and the VMRC conduct public hearings at which evaluation of social impacts (generally offered by those affected) constitute a significant part of the testimony received for consideration. The role these evaluations play in the final decisions can be highly variable, but (due to the absence of formal protocols for weighing) not amenable to quantification. (See Davos, 1977 and Dee et al., 1973 for treatment of weighing in management decision processes.)

Ecological Impacts

In general, ecological impacts are the focus of most of the deliberation and decision rationale in regulation of marina development. Indeed, the author's personal observation has been that parties with essentially economic or social concerns for a proposed project will frequently attempt to portray those concerns in an ecological format in order to achieve the greatest impact on regulatory agencies. If anything, this places a premium on the development of accurate assessments of potential environmental impacts. Unfortunately, in most cases, such assessments remain beyond the abilities of scientific advisors. The body of quantitative information currently available for assessment of marina impacts simply does not permit either site specific or general evaluation of potential impacts. (See Brandsma et al., 1973 and Nixon et al., 1973 as examples of the detailed studies necessary for impact assessment.)

For purposes of discussion, ecological impacts can be divided between direct impacts and secondary impacts. Direct impacts are those resulting from the physical construction of the project. Secondary impacts are those resulting from the operation or use of the facility. From the perspective of a scientific advisor, appreciation of the distinction between these two types of impacts is essential. When giving an *a priori* evaluation of ecological impacts it is much easier to speak with certainty about direct impacts than it is to imply certitude about secondary impacts.

Data which might support predictions of secondary impacts associated with marina development simply do not exist (see Raytheon Co., 1978 for review). Among researchers addressing the question, the current consensus seems to be that: 1) evaluations must be site specific, and 2) the necessary correlations between the physical parameters of a site and potential ecological impacts (particularly water quality) simply do not exist. Indeed, such correlations may never be sufficiently refined to support site specific management decisions regarding secondary impacts due to the inherent uncertainty associated with human behavior. It will be difficult enough to understand the relationships between local bathymetry, current patterns and pollutant transport mechanisms for prediction of potential zones of pollution impact, without also having to assess the probability an individual boat owner will choose to bypass his marine sanitation device or spill a can of gasoline.

DISCUSSION

Given the multiplicity of regulatory agencies and the limitations of impact assessment associated with marina development, what are the consequences for the resource management process? In Virginia, the author's experience indicates at least two immediately apparent consequences. First, decision rationales are not notably consistent. Second decisions are rendered on a case-by-case basis.

Inconsistency among decision rationales takes two forms. First, on a formal level, not all management decisions are supported by recorded statements of rationale. This does not mean the decisions themselves are necessarily inconsistent, but it does frustrate attempts to document the decision process and develop an "agency profile" (a detailed understanding of agency procedures and objectives). Second, rationale statements when they do exist, are not typically complete in the sense of summarizing all factors considered and indicating how the considerations were balanced in reaching the management decision. Again, this frustrates analysis of the decision process and development of "agency profiles."

Failure to produce consistent decision rationales arises from several causes and cannot be blamed entirely on the individual management agencies. The lack of formal protocols for documenting rationales is something individual agencies could correct. However, effective utilization of any such protocol presupposes that: 1) agency management objectives are well defined and clearly understood; 2) the relative importance of the multiple considerations in marina development have been determined; and 3) the information to support rationale development is consistently available. The first two of these items might be effectively resolved (see McAllister, 1980 and Westman, 1985), but the third item, as indicted in the impact assessment section of this paper, is frequently beyond the control of resource managers.

The benefits of possessing an "agency profile" are several. First, a clear record of an agency's concerns, weighing of factors and decision protocol provides a basis for coordination among agencies. Second, documentation of past decisions generates an "institutional memory" which in turn facilitates consistency in the face of personnel turnover. Finally, clear understanding of management goals allows prior planning by developers, enabling them to avoid the expenses of proposing undesirable or unacceptable projects.

The second basic consequence of the current institutional framework and information supply for marina management has been the rendering of decisions on a case-by-case basis. On the surface this may seem desirable since it implies individual projects find approval or disapproval based solely on their merits. A further implication, however, is that efforts to manage natural resources on a regional scale (i.e. any scale larger than the individual project) are compromised. Since it is not possible to effectively evaluate the incremental impacts associated with one more marina or twenty additional slips on a given body of water, decision rationales based on cumulative impact assessment are not typical. The necessity to render defensible decisions, frequently constrains managers to consider only the most certain consequences of marina development. The result is a frustration of a widely held philosophy of resource management which calls for conservation of a resource until the impacts associated with development are fully understood.

The benefits of resource management on a regional scale seem particularly pertinent in the case of marina development. Since it is not possible to know precisely when "enough is enough," managers are currently caught in the continuing dilemma of trying to balance the need for development with the need to preserve "natural" systems. With no relief from this quandry imminent, an approach which provides some accommodation for both objectives seems appropriate. At least two methods are possible. Both require establishment of management goals in advance of any individual regulatory decisions. One method is establishment of conservation or preservation zones from which some forms of development are excluded. The other method is to spread development out in an effort to minimize potential cumulative impacts (see Anne Arumdel County, 1980 for example). In both approaches the resource is effectively "zoned" on a large spatial scale. The benefits include the enhanced probability of preserving "natural" characteristics of a system or parts of a system and provision of the opportunity for developers to operate within known guidelines.

CONCLUSION

Proposals for marina development have posed and will continue to pose difficult problems for resource managers in the Commonwealth of Virginia. The number and variety of regulatory agencies potentially involved in review of such proposals can make the process complicated and does not ensure consistently thorough or coordinated decisions. Additionally, the efforts of these agencies to perform their roles are confounded to varying degrees by a lack of comprehensive information on which to base decisions. The two basic consequences of these situations are inconsistent decision rationales and a restriction of reviews to case-by-case evaluations.

The effective management of marine/estuarine resources would seem to require a comprehensive overview of all facets of the managed system, encompassing economic, social and ecological considerations. Because the purview of the regulatory agencies are not uniformly comprehensive, effective coordination is essential for meaningful attainment of goals. This coordination is not evident at present. While some of the agencies do interact regularly, there is not obvious unification of efforts. Achievement of such coordination should not be impossible since, ostensibly, all the regulatory agencies share similar or related goals.

A step in the process of developing a consistent and thorough management effort would be documentation of decision rationales by each agency. This can form the basis for analysis of similarities and dissimilarities among agency objectives. In some cases it may also help better define those objectives. In turn, the availability of specific information on the decision process of each agency may provide an opportunity for reduction in duplication of reviews and a chance to ensure consistently thorough review of all proposals.

The efficacy of management of marina development on a case-by-case basis is questionable in view of the shortcomings of impact assessment. The current process of management provides at least the potential for piecemeal despoliation of the resource. Promulgation and implementation of a regional management plan may provide the opportunity for preservation of some benefits of undegraded systems and simultaneously aid developers in effective planning.

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TABLE 1

Potential Impacts Associated with Marina Development

	Assessment	
	Type	Source*
Economic Impacts		
cash flow associated with construction	quantitative	L,D
cash flow associated with operation	quantitative	L,D
alteration in local tax base	quantitative	L
alteration of surrounding property values	quantitative	L
Social Impacts		
aesthetics	qualitative	I
alteration of neighborhood "character"	qualitative	I
alteration of waterway use patterns	qualitative	D,I
Ecological Impacts		
direct impacts		
destruction of intertidal wetlands	quantitative	D,A
destruction of subtidal bottoms	quantitative	D,A
disruption of subtidal benthic communities	qualitative	A
alteration of local water quality	qualitative	A
alteration of riparian lands	quantitative	D,A
secondary impacts		
alteration of water quality	qualitative	A
condemnation of local shellfish beds	quantitative	S
modification of circulation	qualitative	A
increase in wake induced erosion	qualitative	A

* L=local government
S=state agency
D=developer
I=interest group
A=scientific advisor