

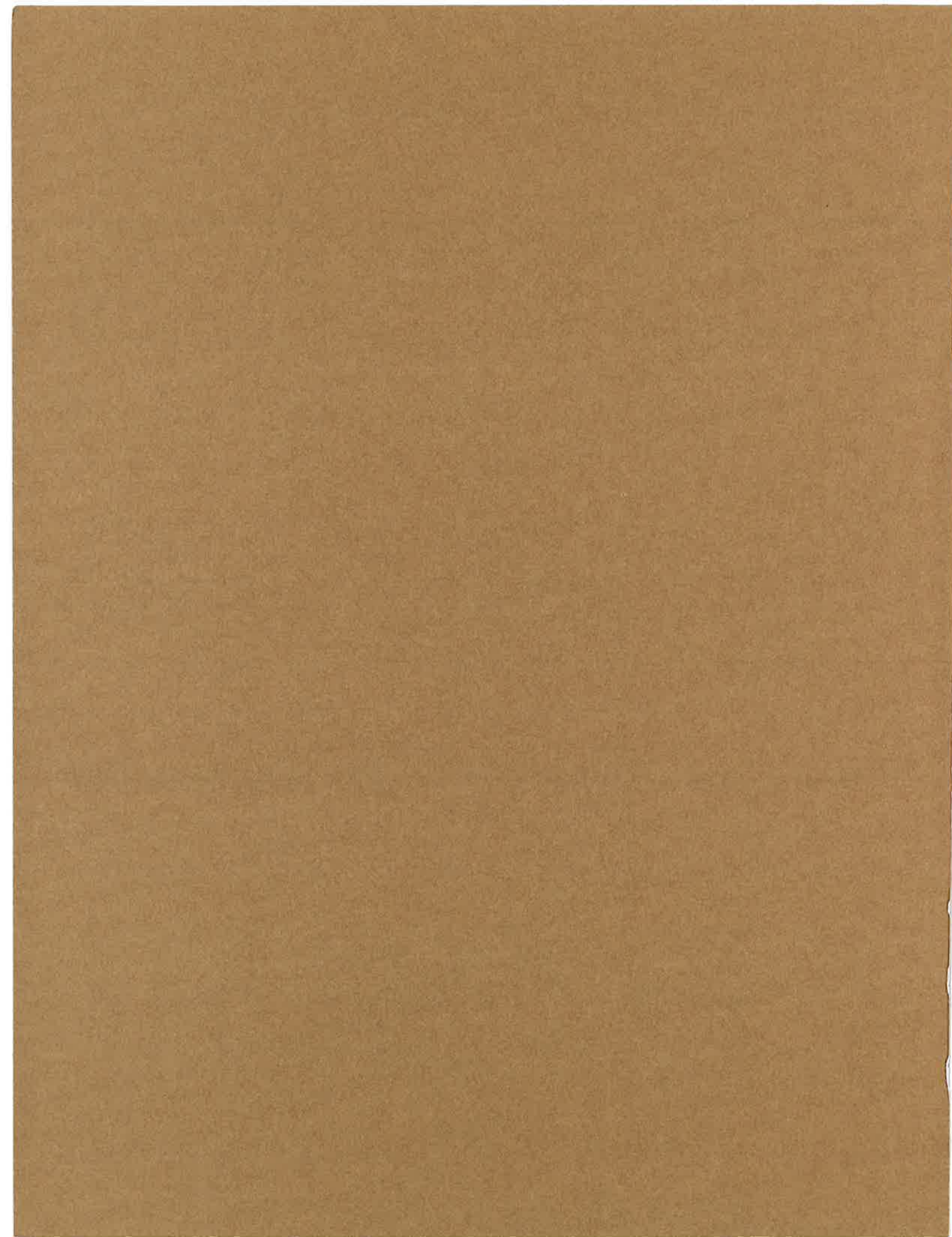
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Perspectives on the Chesapeake Bay, 1990

Advances in Estuarine Sciences



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Perspectives on the Chesapeake Bay, 1990

Advances in Estuarine Sciences

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Preface

The Chesapeake Bay is an extremely complex and variable estuarine ecosystem, influenced by diverse factors. Within the Bay's natural boundaries, a spectrum of aquatic environments—ranging from freshwater to nearly full-strength seawater—supports diverse organisms and allows many chemical reactions to take place. Characterized by complexities in circulation patterns, nutrient cycles, and food webs, the Chesapeake Bay is a unique and highly productive natural system.

Historically, the Chesapeake Bay has demonstrated a remarkable resilience to many natural or man-made perturbations. Unusual events such as hurricanes, droughts, and seasonal temperature extremes have caused imbalances, but the Bay has gradually recovered its former state of dynamic equilibrium. Similarly, the Bay remained relatively unchanged over several centuries of urbanization, shipping, and fishing.

Yet today the Chesapeake Bay appears to be a fragile ecosystem increasingly vulnerable to the relentless encroachment of man. In fact, most of the problems currently perceived as causing declines in the health of the Bay have a common denominator—people. Man has directly influenced the estuary by adding his wastes and by withdrawing resources from the Bay and its tributaries. In addition, people have acted indirectly by changing the character of the land, water, and air that surround and interact with the Bay. In short, man is altering the hydrological and ecological continuum of the Chesapeake Bay watershed. Today we recognize ecosystem thresholds beyond which resilience or assimilative capacity can be exceeded resulting in such perceptible changes as low dissolved oxygen concentrations, increasingly turbid waters, or lowered abundances of fish, shellfish, and other organisms.

On a more subtle level, many researchers point to changes in the pathways of carbon and energy flow through the Bay food web. Although increased amounts of nutrients such as phosphorus and nitrogen have stimulated greater production of phytoplankton, it appears that the carbon energy resulting from photosynthesis is not yielding greater quantities of useful metazoans such as finfish and shellfish. Indeed, it appears that the collective effects of water quality changes, habitat losses, recruitment failure, and fishing mortalities have shifted carbon energy away from the economically productive metazoan food web and into the trophic “dead end” of microbial production. By remineralizing excess carbon production in the microbial food web, the ecosystem consumes precious oxygen and subsequently loses habitat for the more useful metazoan species.

These kinds of Bay-wide impacts result from massive inputs of nutrients and other chemicals coming from sewage treatment plants or industrial operations (referred to as point sources) or from the stormwater runoff of rural or urban land (called non-point sources). These natural materials normally recycle in the environment among plants and animals, or among land, air, and water. But the large human population in the Bay watershed has disrupted the balance of the recycling process and has led to severe problems in some regions of the Chesapeake Bay.

Another type of problem confronting the Bay comes from toxic compounds—man-made products created by industrial activity, or naturally-occurring chemicals that are concentrated to levels far exceeding the trace quantities normally found in the environment. Toxic materials tend to be concentrated in regions of the Bay close to manufacturing industries or waste disposal sites. Problems caused by toxic compounds are

difficult to predict or understand because of their extremely complex chemical properties. However, these compounds can cause serious human and environmental health hazards when they enter the Bay.

The complex interactions between pollutants from point and nonpoint sources, toxic compounds, and ecosystem change are further exacerbated by the diverse cause-and-effect sequences occurring throughout the Bay watershed. For example, land use changes in Pennsylvania could begin a sequence of physical, chemical, and biological events that much later produce oxygen deficiencies in Maryland deep water. Thus, the observed impact is separated from its cause in both time and space.

These complexities underscore not only the need for research, but also the importance of presenting research findings to environmental decision-makers. Management options can be complex and can require years of sustained effort before yielding significant improvements. Clearly, the Bay system does not necessarily respond to instant management fixes, nor does it hold to boldly declared target dates for restoration milestones. Why? Because we simply don't know all the answers.

Even with our considerable and growing knowledge base for the Chesapeake system, there remain many uncertainties and gaps in information. We do not fully understand all that is wrong with the Bay, or how to repair the system most effectively, or how long and how intensive our restoration efforts must be. Wise management and regulatory decision-making depend on our ability to clarify these uncertainties. And to a large extent, the immediate value of Chesapeake Bay research will be judged by the contribution it makes to addressing management needs.

Regardless of our degree of commitment to restoring the Bay, we must recognize that our state, federal, and local agencies work with limited fiscal resources. We must assess the effectiveness of ongoing management programs and continue to develop and implement new and better strategies. Chesapeake Bay research will play a key role in the success of these plans and subsequently the success of our entire restoration effort. Research results promise to help resource managers understand how pollutants cycle through and affect the Bay's ecosystem. Moreover, research findings can allow managers to focus the array of pollution abatement programs and direct limited financial resources more effectively.

It is for these reasons that the Chesapeake Research Consortium publishes the *Perspectives* series on selected Chesapeake Bay research topics. These volumes provide the research community with a valuable mechanism for incorporating new research findings, understanding, and scientific consensus into management actions for restoring the Bay system. The first *Perspectives* volume formed the basis for the 1988 Chesapeake Bay research conference, as this volume will for the 1990 conference. By publishing these synthesis papers and organizing conferences that bring together the scientific and management communities, CRC works to generate and disseminate the scientific information that is critical for wise management decisions and a truly effective restoration of the Chesapeake Bay.

—JOSEPH A. MIHURSKY
—MICHAEL HAIRE

Perspectives on the Chesapeake Bay, 1990: Introduction

As the Chesapeake Bay Program resolves some of the issues before it, other problems come to the forefront for consideration. The 1983 Bay Agreement identified a small number of critical issues to be addressed. The selection of these issues was based on a consensus among citizens, resource managers, and the scientific and technical community. These groups agreed, first, that these problems were important, and, second, that we knew enough about them to develop successful solutions.

As time has passed, the different parts of the complex interstate, state-federal, state-local, public-private, and legislative-executive entities that comprise the Chesapeake Bay Program have coalesced into an increasingly effective and efficient apparatus for dealing with the various parts of the problem. It has become apparent that the solutions to many of the problems articulated in 1983 Agreement and its successor, the 1987 Bay Agreement [1], are not going to be as simple as was hoped in 1983. The basic consensus as to the importance of the original problems still holds, but some newly-identified problems demand solutions and require integration into the Program.

The Chesapeake Bay Program is moving in uncharted waters. No other environmental management effort on this scale has ever been attempted in a system as complex as the Chesapeake. The effort is complicated by the diversity of approaches and the interrelationship of various program activities. The Comprehensive Research Plan approved by the Chesapeake Bay Program Executive Council in July 1988 [1] recognizes the need for continuing interdisciplinary studies of the Bay's estuarine system, subsystems, and watershed from both basic and applied perspectives. The Research Plan clearly recognizes that the Bay Program cannot "stay tied up at pierside waiting for all of the answers to all of the questions before setting sail".

This publication presents four reviews of scientific and technical topics relevant to activities of the Chesapeake Bay Program. Like the reviews in the first publication in this series [2], these topics have broad implications beyond the immediate scope of the disciplines involved and promise to make contributions to other areas of research and management.

For example, modeling has been used in various ways in the Chesapeake Bay Program since its inception as a research program funded by the Environmental

Protection Agency. The term modeling, however, is frequently misused in discussing the ways in which the Bay's problems should be solved. The distinction between conceptual and simulation models is not universally recognized, and the relationship of ecosystem models to water quality, hydrodynamic, and population (fisheries) models is unclear even to many participants in the Bay program. The contribution by Wetzel and Hopkins, *COASTAL ECOSYSTEM MODELS AND THE CHESAPEAKE BAY PROGRAM: PHILOSOPHY, BACKGROUND, AND STATUS*, clarifies misconceptions about the use and nonuse of ecosystem models within the Chesapeake Bay program. For scientists and managers outside the field of ecosystem modeling, it also offers a sense of where the effort in the Chesapeake Bay stands in relation to modeling projects in other coastal systems.

Efforts to understand problems of living resources and their response to pollutants initially focused on the water column and submerged aquatic vegetation habitats. However, as we have refined our understanding of nutrient and sediment processes, particularly as they relate to storage and mobilization in the sediments, it has become apparent that we cannot ignore the role of the benthos. Diaz and Schaffner, in *THE FUNCTIONAL ROLE OF ESTUARINE BENTHOS*, provide a timely and current review of this subject.

A major strategy of the Bay restoration and protection involves controlling nonpoint sources of pollution by applying best management practices (BMPs) to any land-based activity that might have a deleterious effect on the Bay's aquatic habitat or living resources. Dillaha, in *ROLE OF BEST MANAGEMENT PRACTICES IN RESTORING THE HEALTH OF THE CHESAPEAKE BAY: ASSESSMENTS OF EFFECTIVENESS* reviews the science behind the assessment of present and developing BMP technology and outlines the strengths and weaknesses of some current practices.

As the Chesapeake Bay program begins to address issues related to toxics, difficult decisions will have to be made, often without the luxury of complete knowledge of either the problem or the effectiveness of the proposed solutions. Cairns and Orvos outline a framework for these choices in their paper, *DEVELOPING AN ECOLOGICAL RISK ASSESSMENT STRATEGY FOR THE CHESAPEAKE BAY*. This term "risk assessment" is one which should become familiar to all who are interested in a restored Chesapeake Bay.

The following summaries touch only briefly upon the material developed by the authors. Readers of this executive summary are urged to read the full papers. The authors were requested to write their reviews for a broad audience including scientists and managers who are not familiar with the intricacies of the specific disciplines covered in the reviews.

Ecosystem Modeling and the Chesapeake Bay Program

The Chesapeake Bay Program is not using the full potential of ecosystem modeling in the planning and implementation of restoration and protection strategies. This under-utilization is not unique to the Chesapeake Bay. Few attempts have been made to fully incorporate ecosystem models in the management of other coastal regions. Some relatively successful conceptual models of coastal systems and subsystems, including several for the Chesapeake Bay, have been diagramed and developed. A few of these conceptual models have been taken through the steps of computer simulation validation and sensitivity analysis.

The most successful application of ecosystem models has been in the planning and guidance of research programs. Ecologists can confidently use conceptual and simulation models to identify data weaknesses and gaps and to form testable hypotheses about specific systems. However, managers have not embraced some of the more successful "management-oriented" ecosystem models, possibly because these models were not directly commissioned by managers themselves.

Four cases of management-oriented, ecosystem-level conceptual and simulation modeling provide a good description of the interaction between the Chesapeake Bay Program and the developing state of ecosystem modeling in the region. At the beginning of the Bay Program's research phase, the U.S. Environmental Protection Agency and the U.S. Fish and Wildlife Service funded the development of a set of conceptual models that eventually consisted of a whole-system model with subsystem models of emergent wetlands, submerged aquatic vegetation, plankton, benthos, and fish populations structured by feeding types. At the time these models were developed, they were used to focus discussions on general directions for the program, but were not developed further into simulation models or used in the planning of specific research or management activities. (The model developer's departure from the Bay area may have contributed to the neglect of the models' further development.)

The other three cases of ecosystem modeling discussed by Wetzel and Hopkins involve modeling conducted by established members of the Bay research community who have been involved with the federal- or state- supported research and management activities of the Chesapeake Bay Program. Kemp and his colleagues at the University of Maryland (Horn Point Environmental Laboratory) used a hierarchical structure in developing a simulation model consisting of six submodels—autotrophs, epibiota, plankton and water, benthos and sediment, mobile invertebrates, and nekton—which interact through common compartments. This set of models was used extensively to investigate the decline of submerged aquatic vegetation. Contemporaneously with these modeling efforts, Wetzel and his colleagues from the College of William and Mary (Virginia Institute of Marine Science) developed conceptual and simulation ecosystem models dealing with the dominant seagrass species, eelgrass (*Zostera marina*) and widgeon grass (*Ruppia maritima*). Both of these modeling efforts concluded that the principal explanation for the decline of the Bay's submerged aquatic vegetation was a reduction in light intensity caused by increased turbidity largely due to excess nutrients and suspended sediment.

The final ecosystem modeling effort reviewed is the network analysis approach currently being used by Ulanowicz and his colleagues at the University of Maryland's Chesapeake Biological Laboratory. Wulff and Ulanowicz have used the network approach to compare the Chesapeake Bay with the Baltic Sea ecosystem. Many consider this approach to have great promise for future work in the Chesapeake.

The major emphasis in the Chesapeake Bay has been on linking water quality-hydrodynamic models with models of fisheries populations. Ecosystem modeling has not been supported to the same extent. Wetzel and Hopkins point out that the existence of a state-of-the-art, three-dimensional, time-variable hydrodynamic-water quality model may provide the Bay scientific community with the detail necessary to link physical, water quality, and ecological processes within a single modeling framework. They suggest that coupled ecosystem hydrodynamic-water quality models will be required to address the large-scale management decisions facing the Chesapeake Bay Program.

The expertise necessary to move on to this next generation of management-oriented ecosystem modeling already exists within the Chesapeake Bay research community. What appears to be lacking is a commitment to use ecosystem models to support management decisions.

Functional Role of Chesapeake Bay Benthos

The water column, plankton, and dissolved nutrients (and the interactions among them) have occupied most of the attention of the scientists and managers of the Bay Program. Initially, the Bay bottom was treated merely as a sink or sometimes as a possible source of nutrients entering the water column. The importance of the benthic system was brought sharply to the attention of Bay strategists when the initial sediment submodel in the water quality model failed totally to respond to large-scale changes in nutrient reductions. A crash research program developed sediment nutrient flux algorithms sufficiently sophisticated to handle the proposed scenarios for the 1991 reevaluation of nutrient strategy.

Diaz and Schaffner make a strong case that any management strategies beyond the present iteration cannot ignore the functional role of the benthic organisms in the Bay. The general structure and distribution of the benthic community in the Bay are fairly well known, and the effects of many geological, physical, and chemical variables are reasonably well understood. However, the influence of functional characteristics of the benthos, such as feeding rates and methods, on sediment modification are not sufficiently well known for them to be incorporated into the technical models being used to evaluate management strategies.

With the benthic environment serving as the major storage compartment for almost all of the materials that enter the Bay, any management strategy dealing with excess nutrients and toxic additions must come to grips with sediment-organism interactions. Many benthic species make good indicators of past and current environmental conditions because of their close association with sediment through much of their life history. Not only do benthic organisms serve as integrators of environmental conditions, but they also can generate site-specific sediment conditions. Benthic organisms directly affect the transport of nutrients, pollutants, and oxygen across the sediment-water interface in both directions.

In addition to their role in mixing sediments, many benthic organisms are also capable of increasing sedimentation. Filter feeders can generate large quantities of biodeposits. These biodeposits can alter the size distribution of sediments and benthic community composition through the ability of filter-feeding benthos to selectively remove material from the water column. The chemical characteristics of deposits resulting from benthic activity may also differ from those of naturally settling particles, because the digestive processes of the benthic organisms can both chemically alter clay mineral structure and add large

amounts of organic material to the deposits. Colonies of benthic organisms also have been demonstrated to trap sediments and increase sedimentation rates in the shallow waters of the Chesapeake Bay.

The benthos are also a key component in the overall energy flow within the estuarine ecosystem. In the shallow Chesapeake system, there is a high probability that much of the phytoplankton and microheterotroph production moves into the benthos. Secondary production in the benthos, in turn, supports many of the fishes and larger invertebrates that are harvested for human consumption. Diaz and Schaffner estimate that the Bay benthos produce about 194,000 metric tons of carbon each year, or about seven times the production required to support the maximum combination of fishery yields taken from the Bay.

The importance of benthic function to the overall Bay ecosystem highlights the need to increase our knowledge of the benthos if we are to make wise choices in the next generation of management issues. Fortunately, there is extensive literature on the benthos of the Chesapeake Bay, and much of what we know of benthic processes in estuarine systems has been learned in the Chesapeake. Diaz and Schaffner present a series of key questions about the benthos that are central to Chesapeake Bay management.

Questions related to interactions between benthic populations and benthic habitats include the investigation of organism-environment relationships that can predict sources and pathways of energy flow within and among habitats, and the possibility that the spatial arrangement of various benthic habitats may play an important role in benthic function.

A second category of questions relates to the role of benthos in the overall trophic structure of the estuary and includes more precise definitions of the linkage of primary organic sources with secondary production and the linkage of fishery yields with benthic production.

Fundamental issues related to the interaction of the benthos with the materials of direct concern to the Chesapeake Bay Program include benthic uptake of toxic materials and resulting influence on the fate of these compounds; the magnitude of control that benthic organisms exert on diagenetic processes and sediment dynamics; and the relative importance of biological and physical processes controlling diagenetic processes and nutrient and toxic dynamics.

Final questions deal with the effects of long-term climatic changes on benthic function; the presence of any long-term periodic cycles of benthic population; and the role of episodic events such as large storms or dredge material disposal in restructuring benthic habitats and communities.

Assessment of Best Management Practices in the Chesapeake Bay Watershed

"To BMP or not to BMP, that is the question." Unfortunately the Bay Program managers dealing with nonpoint source pollution in the Chesapeake Bay watershed do not have the luxury of soliloquy. Nonpoint source pollution is a major contributor to the water quality problems throughout the Chesapeake watershed. In some basins, it is estimated that nonpoint pollution far exceeds point source pollution. Consequently the restoration and protection of Chesapeake Bay water quality cannot be accomplished without significant reduction of nonpoint source pollution. The only feasible approach known is the use of best management practices, or BMPs.

What are BMPs? How do they work? How do we know they work? These three simple questions do not have simple answers. Dillaha, in his discussion of the use and effectiveness of BMPs, describes the many uncertainties in the choice of specific BMPs for specific sites and situations, and points out that the techniques available for resolving these uncertainties are not easily applied. Most BMPs are designed to reduce the pollutant load in surface water. Recent studies, however, indicate that some of these practices, although very effective in reducing surface runoff, may increase problems of groundwater quality.

BMPs such as conservation tillage, contouring, terracing, and planting of cover crops are erosion control practices that reduce the pollutant carrier mass. The effectiveness of these BMPs in reducing the total amount of sediment loss is offset to some extent because the sediment that does erode is enriched with fine-grained material characterized by a higher adsorptive capacity. Conservation tillage, the most widely used BMP, has been very effective in reducing erosion in most applications. The acceptability of conservation tillage in Maryland and Virginia is enhanced by higher crop yields and lower production costs relative to conventional tillage.

BMPs such as vegetative filter strips have been tried extensively in the Bay watershed, with mixed results. In hilly regions, vegetative filter strips do not appear to be very effective. In flatter terrains, this practice is more effective, but only as long as the strips are properly maintained to retain sediment-trapping capability.

Evaluating BMPs is one of the most difficult tasks facing water quality managers. Each site selected for BMP evaluation is unique. Detailed monitoring or measurement of the effectiveness of a BMP at one site may have little or no relevance to the effectiveness of the same BMP at a different site. Edge-of-field effects

have been characterized for a number of BMPs, but those measurements are not easily translated into impacts on quality of the receiving water. Even if monitoring programs could be designed to properly evaluate the effectiveness of specific BMPs, the problem of response time of the system must be considered. It has been estimated that the time required to detect water quality improvements from specific projects may exceed a decade.

To overcome the limitations of monitoring to evaluate BMP effectiveness, a number of models have been developed for nonpoint source problems. Nonpoint source models usually focus on the creation of pollutants and transport of these pollutants across the land surface to receiving waters or through the soil to groundwater. Screening models are relatively simple models used to identify problems within a watershed or basin or to make some preliminary qualitative evaluations of alternatives. These models have shown that a few critical areas in a watershed were disproportionately responsible for pollutant loadings.

More complex models, hydrological assessment models, have been developed to assess current conditions or evaluate alternative management strategies. These models have been developed for review of both field-scale and watershed-scale assessments. An inherent problem with many of the models is the lack of data for appropriate calibration or verification in specific basins, watersheds, or other sites. Dillaha suggests that BMP benefits probably can be best addressed through use of properly selected simulation models that account for site-specific conditions.

A number of critical research questions related to BMP effectiveness must be answered if the full potential of nonpoint source pollutant reduction is to be realized in the Chesapeake Bay watershed. We must obtain a better understanding of the processes transporting pollutants between fields and streams. In this context, we must determine satisfactory equations for describing sediment transport in shallow overland flow and develop nitrogen-accounting models that better simulate nitrogen transformations and the availability of nitrogen species in the various microclimates of the Bay watershed.

If less material is applied to fields, less material is available for stream or groundwater loading. Development of reliable tests for plant-available nitrogen and refinement of methods for applying agricultural chemicals below the surface with minimum soil disturbance will allow farmers to maintain yields while reducing the amounts of chemicals available for transport to streams.

Development, testing, and verification of better models cannot be accomplished without better data

bases. Long-term intensive monitoring programs should be established on a number of watersheds within the Bay region to provide these data bases. While these data bases are being developed, modeling efforts should concentrate on physically based, deterministic, distributed-parameter models that can be used without site-specific calibration.

Comparison of alternative models must be improved through the development of standard methods to evaluate predictive results from the different models. In addition, targeting or screening models should be improved, particularly as they relate to cost-effectiveness of alternative nonpoint source control programs. The use of geographic information systems is considered an area for special emphasis.

Certain BMPs are becoming almost standard practice in agriculture. Specific studies of the design, installation, and maintenance of these practices should receive priority consideration. Candidates for this specific focus are vegetated filter strips (particularly as regards maintenance), wet ponds enhanced with wetlands (particularly as regards long-term nutrient removal), conservation tillage (particularly in regard to long-range impact on groundwater), and alternative animal waste containment and treatment facilities (particularly with regard to nutrient removal and disinfection).

An Ecological Risk Assessment Strategy for the Chesapeake Bay

Consider the person who wants to do something in the Chesapeake Bay watershed that will have what he or she considers to be a very minimal and acceptable impact on the Bay's resources. A number of individuals oppose the activities, because they believe there is a possibility of some adverse impact on the Bay, and they are willing to tolerate no adverse impact of any kind. How is the situation to be resolved?

Almost all activities of the human species will cause some perturbations of nature. It is not possible to predict completely and accurately what the impact of most activities will be. There is a risk involved in making any decision. All of us use risk assessment in our day-to-day activities; we just don't follow formal protocols in doing so. Therefore, environmental risk assessment should not be considered a novel or strange approach to the resolution of the Bay's problems.

Cairns and Orvos point out that protocols for assessing ecological and human health risk are developed sufficiently to assess the risk from a particular event at a specific site. However, comparable procedures have not been developed for conducting risk assessment of activities that may occur at one point in

the watershed with adverse environmental or ecological processes elsewhere in the watershed.

A key element in the Cairns and Orvos strategy for developing a Bay-wide risk assessment program is the selection of appropriate endpoints (effects). This selection process must involve scientists, regulators, municipalities, business interests, agricultural interests, environmental interests, and any other relevant group or entity involved in Chesapeake Bay matters. General agreement about endpoints may not be easy to attain, and consensus will probably change with time. Without agreement on endpoints, however, it will not be possible to develop a defensible and acceptable predictive capability.

Once endpoints have been selected, evaluation of suspected hazards in specific systems or parts of systems can proceed in a logical framework. The traditional risk assessment approaches, qualitative vs. quantitative, reductionist vs. holistic, and top-down vs. bottom-up, which have been used in different settings, must be combined for comprehensive risk assessment in a system as complex as the Chesapeake Bay watershed.

Direct validation of some of the predictive capability of risk assessment models (particularly those models that deal with the effects of specific compounds on specific species) is possible in the laboratory. However, the more complex models, particularly those developed to assess risk at the ecosystem level, cannot be directly validated, primarily because of the potential for damage in field validation experiments. A surrogate for planned experiments may be found in the inadvertent discharges of hazardous materials or spills (ecoaccidents). Unfortunately from a research perspective, the initial reaction to ecoaccidents is to clean them up or to attempt to mitigate the immediate damage, with little time or effort spent in trying to determine the immediate ecological effects of the accident. Cairns and Orvos clearly state the need for research and regulatory groups to plan ecoaccident studies well before any accident occurs.

Since the effectiveness of risk assessment is intimately tied to our understanding of the Chesapeake Bay ecosystem and the effects of specific materials or activities on that ecosystem, gaps in our understanding (or gaps in technology available to help us gain the necessary understanding) must be filled. Necessary research directly applicable to risk assessment includes: the development of environmentally realistic tests of acute and chronic toxicity; the incorporation of present and improved toxicity tests into models developed for microcosm and mesocosm systems; and the increased development and use of biomarkers to assess effects. In addition to these specific research needs, a number

of more general areas need study: energy flow dynamics in the Bay watershed; the resiliency of some estuarine systems; ecological population theory in relation to risk assessment; the capabilities of wetlands areas to remove toxics; the utility of created wetlands as replacements for natural wetlands; and the importance of groundwater both as a contributor (nonpoint pollution) and as a resolver (through water purification processes functioning in aquifers) of the Bay's water quality problems.

Risk management and risk assessment must be inseparable management and scientific partners. Risk assessment is quintessentially a scientific activity; risk management, as the synthesis of technical, socioeconomic, and political considerations, is primarily a management function. However, neither risk assessors nor risk managers can perform their proper function without interaction with the other group. Analyzing and characterizing risk is a scientific pursuit. Deciding whether a risk is acceptable is a question for society as whole.

Conclusion

For convenience and manageability, the day-to-day activities of the Chesapeake Bay Program are coordinated through committees and workgroups such as Living Resources, Modeling, Toxics, Stock Assessment, etc. In reality, however, these areas are not isolated from each other, and the best decisions consider and integrate the deliberations of all these groups. In an analogous fashion, scientific and technical studies undertaken in a specific area often generate information relevant to many other activities.

We believe this to be the situation in the four areas that are reviewed in this report.

The management community involved in the Bay Program interacts on an almost continuous basis. This interaction is critical to the success of a program that deals across political boundaries with a natural system as complex as the Chesapeake Bay and its watershed. In the Bay region, as elsewhere, there is less regular communication between researchers in different disciplines or between much of the scientific and technical community and the day-to-day managers.

Projects such as this document try to improve communications—both within the scientific and technical community, and between that community and those responsible for management of the Bay's environment and resources. We hope that the material contained in these papers stimulates the incorporation of new and appropriate ideas into the Chesapeake Bay Program and contributes to the ultimate success of the most ambitious estuarine management program ever attempted.

—MAURICE P. LYNCH

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Coastal Ecosystem Models and the Chesapeake Bay Program: Philosophy, Background, and Status

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Models and Modeling in Ecosystem Research

Models are abstractions or simplifications of systems. As such they have become useful tools in scientific analysis of natural systems because the complexity of natural systems is too overwhelming to be perceived intuitively [14]. Ecosystem models are defined operationally as models conceptualized at the hierarchical level of community organization and above. An ecosystem model should incorporate sufficient functional attributes of the real system to mimic some aspects of its behavior, but obviously such models cannot and should not encompass all attributes. Deterministic, numerical ecosystem models synthesize large amounts of information on individual parts of systems and objectively explore hypothesized relations between components. Isolation of successively smaller parts of systems for detailed measurement and study under controlled laboratory conditions may not reveal the role of the part in the larger system. It is in this area that ecosystem modeling is particularly valuable. Perhaps through the integration of reductionist data, models can demonstrate emergent properties of the larger, whole system.

Simulation modeling of ecosystems has progressed rapidly over the past 15 years. Advances in computer hardware and software have played an obvious part in

this progress; a less obvious factor is the maturing of the modeler and model end-user. There has been increased demand for integrative syntheses of large data bases collected from multidisciplinary research programs, and the use of models has expanded into other areas as well. Models are now commonly used to plan and guide research programs from the outset—to guide program development, to identify data weaknesses and gaps, to evaluate management-oriented alternatives, and, perhaps most important, to formulate testable hypotheses about a system's structure and function.

Numerical models of marine processes have been in existence since the late 1930s. Early attempts to examine population phenomena were little changed from the basic growth equations of Lotka [28] and Volterra [44], in which population size was a function of constant birth and death rates. Later workers [11, 34-36] incorporated terms for phytoplankton division rate, sinking, respiration, zooplankton grazing, light fields, nutrient limitation, and vertical turbulence. Only in 1949 did Riley incorporate phytoplankton and zooplankton equations into a system of equations with feedback control. Application of these early models was seriously limited by the lack of adequate computational equipment. Not until the introduction of analog and digital computers in the 1960s was there full development of dynamic, feedback-controlled models of ecological processes.

During this time modelers interested in ecosystem analysis and those interested in population dynamics

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diverged in approach. Even without computer capability it was possible to model the behavior of two populations over time as long as the equations remained in the simple Lotka-Volterra form. However, in order to simulate and analyze the controls on processes, such as primary production, it was necessary that models include complex equations representing (at least hypothetically) biological reality. Without advanced hardware and software for numerical analysis, such detailed equations could be solved only for steady-state conditions.

Management agencies must often make decisions based on the consequences of specific actions in complex dynamic ecosystems. While intuition frequently fails to provide the correct prediction, so does population modeling. A population model that describes the response of an organism under controlled and limited conditions may fail in the larger context where an organism may be affected indirectly through feedback interactions. By contrast, ecosystem modeling, because it incorporates abiotic controls and feedback interactions, often can provide reliable predictions to assist managers in assessing environmental impacts.

Construction and Evaluation of Ecosystem Models

Any modeling effort should begin with a clearly defined set of objectives and questions. Once model objectives are described, the general course of events in developing an ecosystem model is: (1) construction of a conceptual model, (2) representation as a diagrammatic model, (3) formulation of interaction equations, (4) computer simulation, (5) calibration, sensitivity analysis, and validation, (6) revision and reformulation if necessary, and (7) generation of new hypotheses and/or predictions, if appropriate [14]. Often this sequence of events repeats continuously as new data are collected and the questions the model is to address evolve.

Before a model can be used to generate hypotheses, evaluate internal dynamics and controls, or make predictions, it must be validated. The procedures and criteria used to validate an ecosystem-level simulation model are not fully developed, but they generally involve the comparison of model-generated data with field-derived data. This comparison must use data other than those used in model construction (with which a good fit would be expected because of the interdependence of the two data sets). Therefore many modeling programs include additional independent data collection, which allows statistical tests of correspondence between the independent and model data sets. When it is too expensive to initiate large data-collection efforts for validation purposes, sensitivity analyses can reveal which parameters most affect model output.

Additional data collection can then be focused on those critical or sensitive components.

Sensitivity analysis is further useful in identifying parameters or processes causing interesting model behaviors (output). Such analyses frequently reveal that indirect feedbacks and second-order interactions of no apparent importance are in fact important. When models identify processes previously thought unimportant or when results do not adequately follow observed field data, alternate hypotheses can be suggested to account for the predicted behavior. Based on these alternate hypotheses, new research can be proposed to test predictions using a combination of model simulation analyses, field studies, and laboratory experiments.

Ecosystem Models of Coastal Systems Outside the Chesapeake Bay Region

Several estuarine ecosystem-level models from outside the Chesapeake Bay region were chosen to represent the best of both management- and research-directed efforts. The modeling efforts described below were all developed for specific purposes beyond merely summarizing large and complex data sets. They illustrate the contribution of modeling in guiding research programs and assisting managers with decision-making. The first three models described are clearly management-oriented, while the last three are concerned primarily with basic research.

There have been few efforts to fully incorporate ecosystem-level models in the management process. Although many models have been conceptualized and diagrammed, few have been simulated and even fewer have been maintained as ongoing efforts. We have chosen management-oriented models to represent each of these stages of development: (1) conceptual/diagrammatic, (2) one-time simulation, and (3) ongoing simulation and expansion of effort. Each of the research-oriented models was conceptualized early in a research program and has evolved and become more complex as the programs developed.

The first of these modeling efforts consists of a group of both conceptual and simulation models [5, 12, 27] of the Mississippi River deltaic plain region in Louisiana. The models are management-oriented and attempt to integrate information from several hierarchical levels, ranging from the salt marsh to the entire southern Mississippi River drainage basin, including urban centers. Next in the management-oriented series are models developed and simulated by Hopkinson and Day [19, 20] for two purposes: (1) to evaluate effects of urbanization on nutrient runoff to a coastal swamp forest and (2) to evaluate drainage options for promoting water runoff from agricultural areas while at the same time decreasing eutrophication of receiving

waters. Last, modeling efforts of Costanza et al. [5-7] at the spatial scale of the landscape are discussed.

For the examples in the research-oriented category, three long-term and evolving research-directed modeling programs deal with tidal marsh and estuarine ecosystems [26, 45, 51, 52]. Two of the three have many similarities but have distinctly different philosophies of model control. These models have been integral in the planning, development, and analysis of long-term ecological research programs.

Management-Oriented Models

Mississippi River deltaic plain conceptual models.

The U.S. Fish and Wildlife Service solicited a series of conceptual models of the Mississippi River deltaic plain region [2, 5] for the purpose of summarizing existing data in a form useful for scientists and coastal managers. The deltaic region consists of the broad, topographically flat portions of Mississippi and Louisiana that encompass the largest active delta system in North America. The region's dynamic nature, high biological productivity, and intense level of economic activity (including fisheries, transportation, and minerals extraction) have combined to create enormous problems for resource management. The models integrate information on such diverse topics as ecology, hydrology, climatology, and economics and are the product of several decades of research by the Center for Wetland Resources at Louisiana State University.

Conceptual models were constructed in a hierarchical manner in order to (1) organize existing information on relevant temporal and spatial scales, (2) organize environmental and management problems within this framework, and (3) target areas in need of additional research. The region was abstracted hierarchically into three spatial scales: the entire region, hydrologic units, habitats. The conceptual models describe the overall compartmental and flow structure, interactions, and forcing functions considered most important.

At the highest spatial scale of the region, the most important forcing functions were riverine inputs of water and sediments, and interactions with the Gulf of Mexico and the atmosphere. The primary issues addressed at this scale were wetland loss, natural switching and direct diversion of Mississippi River waters within the coastal zone, and water and air quality and chemical waste disposal.

At the intermediate spatial scale of hydrologic units, which encompass discrete coastal watersheds, the primary issues addressed by the conceptual modeling were (1) the role of wetlands in fishery production, (2) the effects of hydrologic modifications (e.g., canal construction, spoil and levee construction) on ecosystem function, (3) eutrophication and toxic substance

influences on ecosystem function, and (4) salt water intrusion and its effect on community structure, land loss, and municipal water supply.

The smallest scale of model conceptualization was at the habitat level. Habitats were identified as contiguous zones having similar vegetational composition (salt, brackish, and freshwater marshes; swamps; aquatic zones; and uplands). Questions of interest included (1) human-induced stresses on habitats, such as eutrophication and water impoundment, and (2) estimation of the rates of ecological production from each habitat and its value to the overall economy of the deltaic region.

The U.S. Fish and Wildlife Service must frequently render opinions concerning the effects specific activities have on the vitality of habitats and species populations in coastal regions. In order to facilitate the decision-making process and to guide resource management and coastal planning, the agency has had a number of research teams around the United States prepare ecological characterization studies. These studies describe the important components and processes of selected ecosystems and provide an understanding of their relationships through synthesis and integration of extant physical, biological, and socioeconomic information. The rationale is that only through an understanding of ecosystem function will it be possible to effectively manage natural resources and prudently guide development; and ecosystem models offer a means to organize such diverse information. The hierarchical modeling approach exemplified by the deltaic region study has proved successful in conceptualizing and qualitatively elucidating the complex relationships operating at various levels of ecological organization by minimizing problems associated with differences in scale and duration between physical and biological events. This approach has aided decision-makers in establishing cause-and-effect relations and in systematically evaluating the effects of specific activities.

Urban runoff and receiving waters eutrophication models.

Hopkinson and Day [19, 20] constructed a pair of complementary models of a coastal swamp forest to investigate relationships among urbanization, hydrology, nutrient loading to water bodies, and wetland eutrophication. One model investigated the interactions between changing land use in uplands, and storm water and nutrient runoff to adjacent coastal swamps. The second model addressed the effects of sheet flow vs. channeled stream flow through swamp forests on (1) rates of stormwater runoff from the uplands, (2) swamp productivity and nutrient dynamics, and (3) eutrophication of receiving water bodies.

The area modeled was the des Allemands swamp forest ecosystem located in the headwaters of a large

coastal basin bordered by two distributaries of the Mississippi River and the Gulf of Mexico. Natural levees along the distributaries direct runoff water into the interior wetlands and from the headwaters to the Gulf of Mexico. Several studies conducted prior to the modeling effort indicated that urbanization and increased agricultural production on the surrounding uplands were causing fundamental ecological changes in the swamp forest ecosystem. It had been documented that lakes that were once clear and coffee-colored, with abundant game fish populations, had become characterized by frequent algal blooms, periodic fish kills, and "trash" fish populations. Other studies found strong correlations between the density of drainage canals in the swamp, eutrophication of lakes, and decreasing productivity of impounded swamp forests. The Louisiana Office of State Planning predicted in 1975 that the uplands surrounding the swamp ecosystem would experience substantial development in the following 20 years as a result of secondary growth associated with the construction of an offshore oil terminal [19, 20]. The modeling study was a first attempt to integrate several studies that had been conducted in and around the swamp forest ecosystem.

The objectives of the modeling were: (1) to quantitatively predict present and future rates of nutrient and water runoff from the natural levee uplands as a result of changing land use during urbanization, (2) to ascertain the effects of channeled canals and their associated spoil banks on water flow through and drainage in the swamp forest, (3) to evaluate the feasibility of routing upland runoff directly through backswamp areas rather than through drainage canals, and (4) to determine the effectiveness of directing runoff as sheet flow through backswamps in reducing the nutrient load to receiving bodies of water.

Hopkinson and Day used a hydrologic model developed by the Environmental Protection Agency (EPA) to form the nucleus of a larger model that could address all these issues. The EPA Storm Water Management Model is a comprehensive mathematical model capable of simulating urban storm water runoff and the receiving effects for lakes and rivers; it is divided into four major blocks. Hopkinson and Day used portions of the RUNOFF block to simulate runoff from uplands and the RECEIVE block to simulate swamp and lake hydrology and nutrient dynamics.

Most values for equation parameters were obtained from the literature rather than measured in the field. Model validation did not include rigorous field comparisons; rather, sensitivity analyses identified those parameters that most affected model output. The authors found that the parameters that most affected model results were those in which they had the greatest

confidence. Hopkinson and Day strongly suggested that future scientific effort should be directed toward validation of the models with special attention directed to parameters most critical to model results.

Simulation results showed that runoff volumes and nutrient loading to the swamp forest would increase greatly by 1995 if projected changes in land use came about. It was predicted that nutrient runoff to the swamp would increase substantially as a result of the increased runoff. Simulations of swamp hydrodynamics showed that spoil banks retard water exchange between backswamps and streams, causing prolonged ponding. It was found that discharge of upland runoff could be increased by removing spoil banks and introducing water directly to backswamps rather than to drainage canals. Important secondary benefits of backswamp introduction of runoff water would be a net decrease in nutrient loading to receiving lakes and an increase in swamp productivity.

Perhaps because these models were not solicited by management agencies, none of the alternatives suggested by these modeling studies was ever implemented or further evaluated.

Spatial modeling of wetland dynamics. Most ecosystem-level models are designed to predict compartmental dynamics at a single point, often in homogeneous space. Costanza, Sklar, and White [6] have extended this approach to modeling spatial dynamics by arranging a spatial array of point ecosystem models and connecting them with fluxes of water, nutrients, and sediments. The approach sacrifices generality for greater realism and precision. The payoffs are significant: the model can realistically simulate major changes in land-cover patterns across large geographic regions resulting from various-site specific management alternatives as well as natural changes.

The impetus for developing spatial models was a request by the U.S. Fish and Wildlife Service to evaluate the Corps of Engineers' plan to extend a levee along the east bank of the Atchafalaya River that would restrict water and sediment flow into the Terrebonne marshes of coastal Louisiana [7]. The proposed action represented a unique opportunity to study landscape dynamics and develop models for this spatial scale of resolution. Also, the Atchafalaya landscape is changing rapidly enough to provide the necessary data to test basic assumptions and hypotheses concerning landscape development.

The spatial simulation model—the Coastal Ecological Landscape Spatial Simulation Model (CELSS)—consists of 2,479 interconnected "cells," each representing 1 km² [38]. Each cell in the model contains a dynamic, nonlinear simulation model. Variables

include water volume and flow, sediment, nutrients (nitrogen), salt concentration, and plant biomass and productivity. The model produces weekly maps of all the state variables and habitat types. The balance between sediment deposition and erosion is particularly critical to habitat succession and the productivity of the area. Habitat succession occurs in a cell in the model when physical conditions change sufficiently that one habitat type becomes more appropriate than another.

Primary inputs for the model are: (1) detailed, digitized maps for several past years; (2) historical maps of canal and levee construction; (3) weekly records for climatic variables; and (4) water level and salinity in the Gulf of Mexico. In addition field measurements of plant biomass, production, and nutrient uptake are required. The model was initialized for 1956 conditions. It was calibrated by starting with the 1956 conditions and simulating the changes in the area by weekly time steps to 1978, for which another ecosystem type map was available. The simulated and real maps were statistically compared for degree of fit, and parameters were adjusted iteratively within predetermined ranges of uncertainty to maximize the fit [4]. A by-product of this approach was a fairly elaborate sensitivity analysis of the model's response to changes in the parameters, which gave the modelers considerable insight into its dynamic behavior. The model was validated by comparing its predictions for 1983 with the real 1983 data set. These data were not used in the calibration phase.

The model has great potential for application to issues concerning wetland deterioration. It has been used to investigate natural wetland loss as related to sediment supply, marsh productivity, water flow, and sea-level rise (natural or induced by "greenhouse" warming), and human-induced wetland loss and gain as caused by Mississippi River levees, canals, dredged material placement, marsh impoundments, barrier island stabilization, and controlled diversions of the Mississippi River.

The simulation of long-term habitat changes in the coastal marshes of the Atchafalaya River has demonstrated that ecological and physical processes can be coupled and realistically modeled. The results of the model indicated that the current trend of continued habitat loss will lead to severe wetland degradation. The modelers have shown that when spatial processes and cumulative impacts are considered together, the effects are greatly magnified.

Costanza et al. (personal communication) are currently running the model to the year 2033 for several different scenarios of interest to coastal managers. They are also developing spatial models for two different types of coastal ecosystems: (1) the Patuxent

River watershed in the mid-Chesapeake Bay region, and (2) the North Inlet marshes and watershed in South Carolina. The results of these efforts will provide (1) increased understanding of the processes controlling changes in landscapes, (2) principles for adjusting spatial and temporal scales to optimize predictability in models, and (3) new methods for examining the goodness-of-fit between predictions and data that are appropriate for spatial ecological modeling and that require a degree of spatial pattern recognition [40].

Research-Oriented Models

Sapelo Island salt marsh models. The Sapelo Island salt marsh ecosystem has been the focus of an ecosystem-level modeling effort since the early 1970s. A review of the development of the modeling program illustrates not only the philosophical and applied nature of the continuing studies but also the inherent evolutionary pattern of most large-scale modeling programs. In addition to serving as an analytical tool [47], the model has generated testable hypotheses relative to the principal components, fluxes, and controls on carbon dynamics in ecosystems dominated by *Spartina alterniflora* [51-54].

The research program at the University of Georgia Marine Institute includes modeling as an integral part of the overall effort. Field research at Sapelo Island dates back to the early 1950s. The aim of the initial modeling effort was to summarize and integrate the results of these studies and to identify profitable avenues for future research. Specific objectives of each modeling effort became more sophisticated as basic knowledge of ecosystem structure and function improved.

The salt marsh model is composed of 14 abiotic and biotic compartments that represent the principal components exchanging carbon among salt marsh-tidal creek sediments, water, and the atmosphere. The model explicitly identifies above- and below-ground biomass and production of *Spartina* and incorporates an anaerobic detrital community in the sediments. The model is process-oriented but has good resolution at the population level of ecosystem organization.

The model's unique mathematical structure is based on a proposed minimum set of laws governing the growth and interactions of populations as they form communities and ultimately ecosystems [49, 50]. External or environmental forcing functions (e.g., light, temperature, sea level, and tides) were not modeled explicitly; rather, specific rate coefficients for biotic exchanges were varied seasonally. Revisions of the model included algorithms for simulating carbon export due to tidal exchange that were driven internally. Use of non-linear feedback-controlled equations in the model partially dictated the type of data collected in

field studies. The mathematical derivation of the feedback functions represents a hypothetical statement regarding control. Because all parameters defining the feedback functions are theoretically measurable, the controls are experimentally testable. The adoption of this approach allowed the modeling effort and the experimental research effort to be closely coupled from the initiation of the program.

Over its history of almost two decades, the salt marsh model has evolved through seven versions. Although the major goal of modeling within the overall research program has not varied, each version was modified to address specific questions raised in earlier model simulations and to incorporate data from new experimental work and field observations. The first version of the model was implemented specifically (1) to identify where information and data were missing relative to the conceptual model and to aid in guiding research, (2) to identify parameters and/or controls that governed model behavior, and (3) to qualitatively predict the role of the salt marsh ecosystem as a potential source or sink of carbon for contiguous systems and to identify which components were primarily responsible for the observed behavior. Version 3 was undertaken to evaluate the importance of tidal export in the carbon economy of the marsh system. The sixth version was directed toward increasing the resolution of the mechanisms governing tidal export, as well as revising those parameter values for which new data were available. Based on model simulation results of version 6, the modelers proposed three alternate hypotheses that could account for carbon found in excess in the earlier version: (1) bedload transport in tidal creeks, (2) erosion from marsh surfaces after intense rain storms, and (3) biological vectors of movement (i.e., feeding and migration of large animals). A new version has been constructed and simulated, the results of which formed the thematic focus of a research program currently funded by the National Science Foundation (Wiegert and Wiebe, personal communication).

This modeling effort illustrates the variety of roles ecosystem modeling can and often does play in the organization, direction, and analysis of ecosystem-level research. The models have provided insight into the ecology of coastal wetlands that otherwise would have come much later or not at all. New generations of the salt marsh model will include information and material flows for both carbon and nitrogen; to accomplish this a second generation of simulation models will be needed, with complete revision of compartment and flow structures.

North and Parker River ecosystem models. In a collaborative field and modeling effort, the University of

New Hampshire Complex Systems Research Center and the Marine Biological Laboratory Ecosystems Center have been investigating nutrient dynamics in tidal freshwater marshes and rivers in coastal Massachusetts. The movement of conservative elements and the processing of nitrogen and dissolved organic carbon were studied in relation to the movement of water in fringing river marshes. A hydrodynamic model was developed to integrate field and laboratory data. The one-dimensional water-flow model had the capability of accounting for material transport and incorporating material decay coefficients; thus it could describe the behavior of both conservative and nonconservative constituents.

Modeling was an integral part of the North River research program during the 1970s. Initially, conceptual models were developed to organize and synthesize literature information and to form a coherent plan for field and laboratory research. Only after a period of ambitious field work did the need arise for development of a simulation model. The initial objectives were to (1) calculate material budgets for the river systems, (2) predict the spatial and temporal distribution of constituents throughout the estuary, (3) determine the nature and magnitude of resident biotic transformations of nitrogen and dissolved organic carbon, (4) determine the influence of water movement alone on the distribution of nitrogen and carbon within the river, and (5) evaluate the importance of biotic uptake in controlling the distribution of nitrogen and carbon in the river.

Vorosmarty et al. [45] used a detailed one-dimensional tide propagation model in which elevation area and bottom friction coefficients are specified for each finite element. The model was calibrated and assumed validated when the simulated distribution of salinity closely matched field observations for a tidal cycle. In the North River, field observations had indicated that inorganic nitrogen concentrations consistently decreased with movement downstream. The calibrated hydrodynamics model predicted that the decrease in nutrient concentration could be accounted for largely by water movement alone (i.e., dilution). After incorporation of biological transformations the model indicated that benthos and marshes were responsible for 66% and 33% of total uptake, respectively. The calibrated North River model obviated the necessity for extremely time-consuming in-situ sampling. Not only did the model provide important information on the mass flux of nitrogen within the North River system, but it also provided information on the relative importance of dilution and biotic processing on nitrogen dynamics and predicted the spatial and temporal distribution of constituents throughout the river.

It is unfortunate that a stable funding source was not available for continuing research and modeling efforts

in the North River. Considerable field data had been collected on nitrogen transformations within marsh sediments, nitrogen nutrition of marsh macrophytes, and detrital decomposition in the marsh. These data were never incorporated into the simulation modeling effort. Fortunately, the model was general enough that it was easily transferred to the Parker River when funding became available in the 1980s for research on this system. For the Parker River ecosystem, the principal question addressed by the model is the cause of a convex downstream pattern in dissolved organic carbon concentration. For both systems, the modeling efforts have been useful in estimating material budgets.

Narragansett Bay ecosystem model. Kremer and Nixon [26] developed a deterministic numerical ecosystem model of Narragansett Bay to guide research on the Narragansett Bay ecosystem and to aid managers in decisions concerning power plant sitings, the potential of oil spill impacts, and estuarine eutrophication [29]. The model provided the means for synthesizing more than 20 years of data on individual parts of the system. To some extent the emergent properties of a complex system such as Narragansett Bay could be simulated in a mechanistic fashion by combining data on individual parts of the system.

Narragansett Bay is a phytoplankton-based ecosystem in coastal Rhode Island. Here, as opposed to the ecosystems of Sapelo Island, North River, and Parker River, attached algae and emergent macrophytes (marshes) are of reduced ecological importance. Therefore, emphasis was placed on representing the major elements of the plankton and the nutrient cycle of the system.

Among the techniques used to organize existing data were tables, graphs, material budgets, and flow diagrams. Conceptual models that initially contained high levels of biological complexity were reduced to seven major compartments for digital simulation analysis. The simulation model represented flows of mass and energy, as well as mechanisms controlling the interactions among principal components. Hydrodynamic processes were represented in the ecosystem model by incorporation of a numerical hydrodynamic model of the Bay developed at the University of Rhode Island. A crude relation between tide height and net transfer of chemical and biological concentrations was used to interconnect eight spatial regions of the Bay, which were assumed to be structurally similar in a biological sense and were vertically and horizontally homogeneous.

Considerable attention was given to ecologically realistic representation of the major factors controlling phytoplankton and zooplankton population dynamics. A maximum rate was postulated as a function of one

factor, and the effects of other factors were included as dimensionless fractions that reduced the maximum rate. For example, phytoplankton growth was based on temperature (which varied daily and seasonally), nutrients (including nitrogen, phosphorus, and silica), and light (which varied by season, day, self-shading, water column depth, and cloud cover). All influenced net growth of phytoplankton as shown by empirically verifiable data.

Model validation was accomplished in two ways. First, model output was compared with field data collected independently on phytoplankton, zooplankton, and nutrient concentrations at 13 stations in the bay concurrent with the modeling effort. Second, model analyses were conducted to determine the sensitivity of simulation output to specific parameters in the model. These analyses suggested areas where additional research was needed.

The simulation results provided valuable information that would have been prohibitively expensive if not impossible to collect in a field program. For example, at the outset of the research program the relative importance of nutrient limitation and grazing in controlling phytoplankton abundance was unknown. When the model was run with zooplankton removed from the system, the basic bimodal pattern of phytoplankton abundance was still observed. The major differences with zooplankton removed were a slight increase in the spring phytoplankton bloom, substantially higher biomass of phytoplankton in summer, and a lack of marked oscillations. Interactions related to nutrients, benthic grazing, and tidal flushing accounted for the gradual decline in biomass after blooms. Other questions evaluated with the model were the importance of zooplankton excretion and benthic nutrient fluxes in supplying phytoplankton nutrient requirements, and the importance of predation pressure on zooplankton.

Kremer and Nixon [26] strongly cautioned the scientific and management communities against direct application of this or other ecosystem models to problems of environmental management. The Narragansett Bay model was not designed to respond to such large-scale perturbations as large increases in organic inputs, significant changes in salinity distribution, or anoxic bottom waters. However, the inherent constraints are less important when the model is used to explore responses of the present system to relatively minor changes in parameters and processes that are specifically included in its formulation. Examples of relevant questions that the Narragansett modelers felt could be evaluated included (1) the effect of phytoplankton and zooplankton mortality that might result from power plant entrainment and (2) the effects of

nutrient inputs when the volume of tertiary sewage is halved or doubled.

Present Ecological Problems in the Chesapeake Bay Region

With the initiation of the Chesapeake Bay Program over a decade ago, an intensive, Bay-wide effort was begun to identify, research, and develop management plans for critical living and non-living resources of the Bay. Five areas have received much of the attention of both scientists and managers: (1) losses and declines in emergent and submerged aquatic vegetation [17, 31, 32, 22, 46, 48], (2) deterioration in water quality caused by increased loading of nutrients (primarily excess nitrogen and phosphorus) to the principal tributary estuaries and the Bay proper [3, 8, 15, 21], (3) apparent increases in the areal extent and duration of bottom-water hypoxia and anoxia [9, 30, 37, 39, 41], (4) sources, sinks, and fates of organic and inorganic contaminants as they affect natural resources and human health [16], and (5) losses and declines in recreational and commercial fisheries [1, 25].

Use of ecosystem models to address these problems in the Chesapeake Bay has been limited. We use the term "ecosystem model," as discussed in the previous sections, to include only those conceptual or simulation models that include the principal components, pathways, and controls for the flow of energy or cycling of materials and nutrients. By definition these include hierarchical models at or above the community level of ecological organization. In the following section, we review some recent ecosystem models applied to the Chesapeake Bay relative to these environmental issues and discuss the role they have played in systems conceptualization, research planning and direction, and application to management issues.

Chesapeake Bay Ecosystem Models

As for other coastal systems, ecological modeling based on principal components and/or major subsystems in the Chesapeake Bay is a fairly new tool for systems analysis. The most intense ecological modeling activity in the Chesapeake Bay has occurred since the initiation of the Chesapeake Bay Program in 1978. However, development and progress in Chesapeake Bay ecosystem modeling has been slower than for similar programs in other areas. Ecosystem modeling has been secondary to other funded activities and has not received the support necessary to maintain a significant and sustained level of effort. In comparison, a great deal of effort and support has been devoted to the development and application of numerical simulation

models for coupled hydrodynamic-water quality models [10, 18]. Here we report on four ecosystem-level conceptual and simulation models that have been directed at Chesapeake Bay research and management. They are (1) the conceptual models developed by Green [13] at the initiation of the Bay Program, (2) the simulation modeling efforts of Kemp et al. [22, 23] that deal with both living resources (primarily submerged aquatics) and the potential socioeconomic impacts of various management strategies for upper Chesapeake Bay systems, (3) the whole-system and principal-component models for submerged aquatics (eelgrass) in the lower Bay reported by Wetzel and Neckles [48], and (4) the whole-ecosystem model recently reported by Wulff and Ulanowicz [55]. These models cover the range from conceptual to management application.

Conceptual Modeling of the Chesapeake Bay Ecosystem

At the beginning of the Chesapeake Bay Program, the U.S. Environmental Protection Agency and the U.S. Fish and Wildlife Service funded the development of conceptual models to depict the principal living resources, the interactions among them, and those external and internal factors influencing systems behavior. The resulting models indicated the principal carbon and nutrient pathways and proposed conceptual models for the principal subsystem components of the Bay ecosystem.

After discussions with Bay scientists and a review of the literature, Green [13] proposed six conceptual models for the Bay ecosystem: a whole-system model and five subsystem models representing (1) emergent wetlands, (2) seagrasses or submerged aquatic vegetation, (3) plankton, (4) benthos, and (5) trophic dynamics of fish aggregated by feeding type. Green's whole-ecosystem model proposed 41 components or state variables divided among wetlands, water column, benthos, and the environment. The subsystem models were composed of compartments representing the principal living resources and nutrients. Figure 1 shows the 41 principal components assigned to each of the subsystem models, which when coupled make up the conceptual Chesapeake Bay model.

The extent to which these conceptual models were used by Bay scientists, program officers, or management personnel is unknown. At the time of their development and reporting, the models served to focus discussion. To our knowledge none of the models has been translated into a simulation version or been employed in directing research activities. However, these models' primary purpose was to focus attention on the Bay as an ecosystem, and in this they were in part successful.

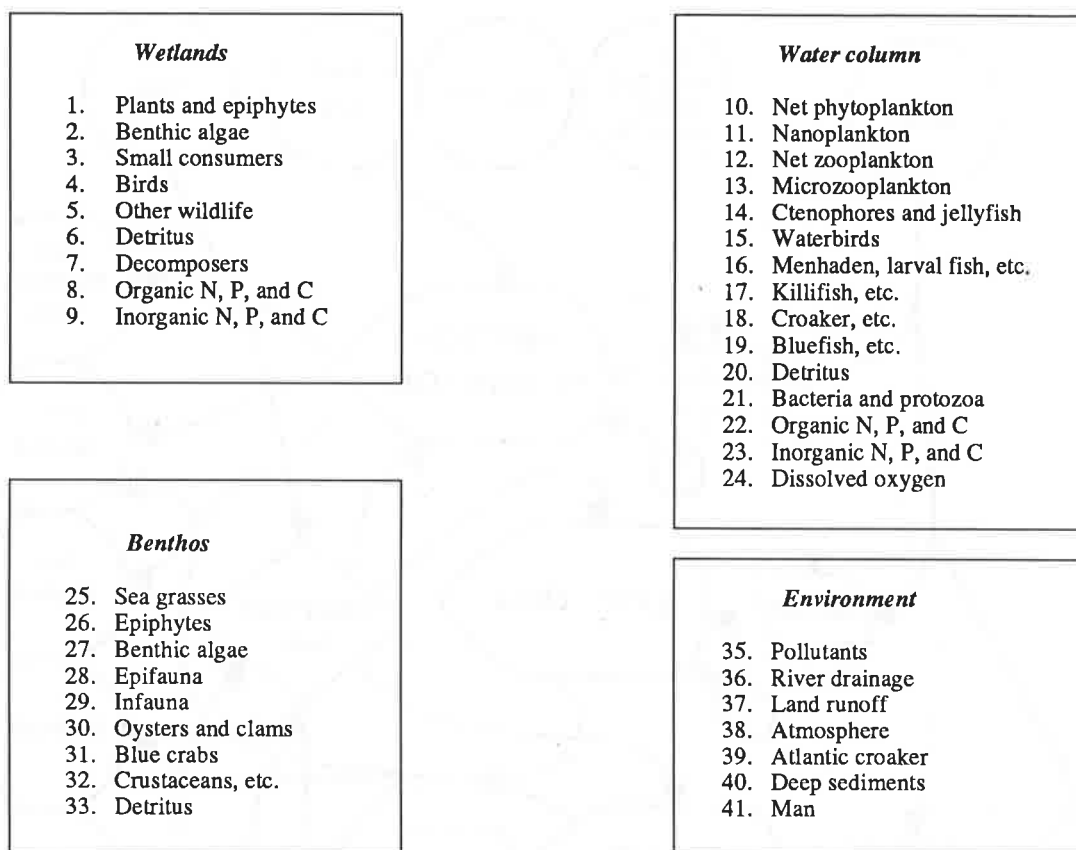


Figure 1. Compartmental structures (41) and submodels (4) included in the conceptual ecosystem model of the Chesapeake Bay [13].

Hierarchical Modeling of Complex Systems

A particularly interesting approach to ecosystem modeling was adopted by University of Maryland scientists in their Bay Program research on submerged aquatics. Kemp et al. [22] have summarized their initial efforts. The hierarchical structure of their model is composed of six principal submodels: autotrophs, epibiota, plankton and water, benthos and sediments, mobile invertebrates, and nekton. Each submodel is developed as a simulation version and the interaction among submodels is via common compartments. The output from one submodel serves as input to one or more other submodels. The interesting aspect of this approach is that the modeler per se becomes part of the interactive modeling process by controlling the output and connectivity among the submodels. The rationale behind the approach is that each submodel is tractable and serves as an analog to the experimental research program.

Simulation studies with the autotroph and nekton submodels were used to evaluate aquatic primary production by the principal autotrophic components

(phytoplankton, submerged macrophytes, macrophyte epiflora, and benthic algae), the principal factors influencing the timing and magnitude of primary production, and the potential effects of changes in primary production on the nekton (resident, predatory, and schooling fish) (Figures 2 and 3). The models were validated by comparison of predicted standing stocks (biomass and/or numbers) with field-derived estimates. In general, the model predictions agreed with field estimates, so the models were further used to investigate causal relations for explicitly modeled parameters (light, sediments, nutrients, and herbicides). The results suggested that the primary control on aquatic macrophyte biomass and productivity was submarine light as influenced by nutrient concentrations and suspended sediment loads. Herbicides, which had been considered initially as a potential cause for the loss of submerged aquatics throughout the Bay, had little effect because their ambient concentrations were low [23]. On the basis of these modeling results, supported by both direct observation and experimentation, Kemp and colleagues suggested that management concerns should

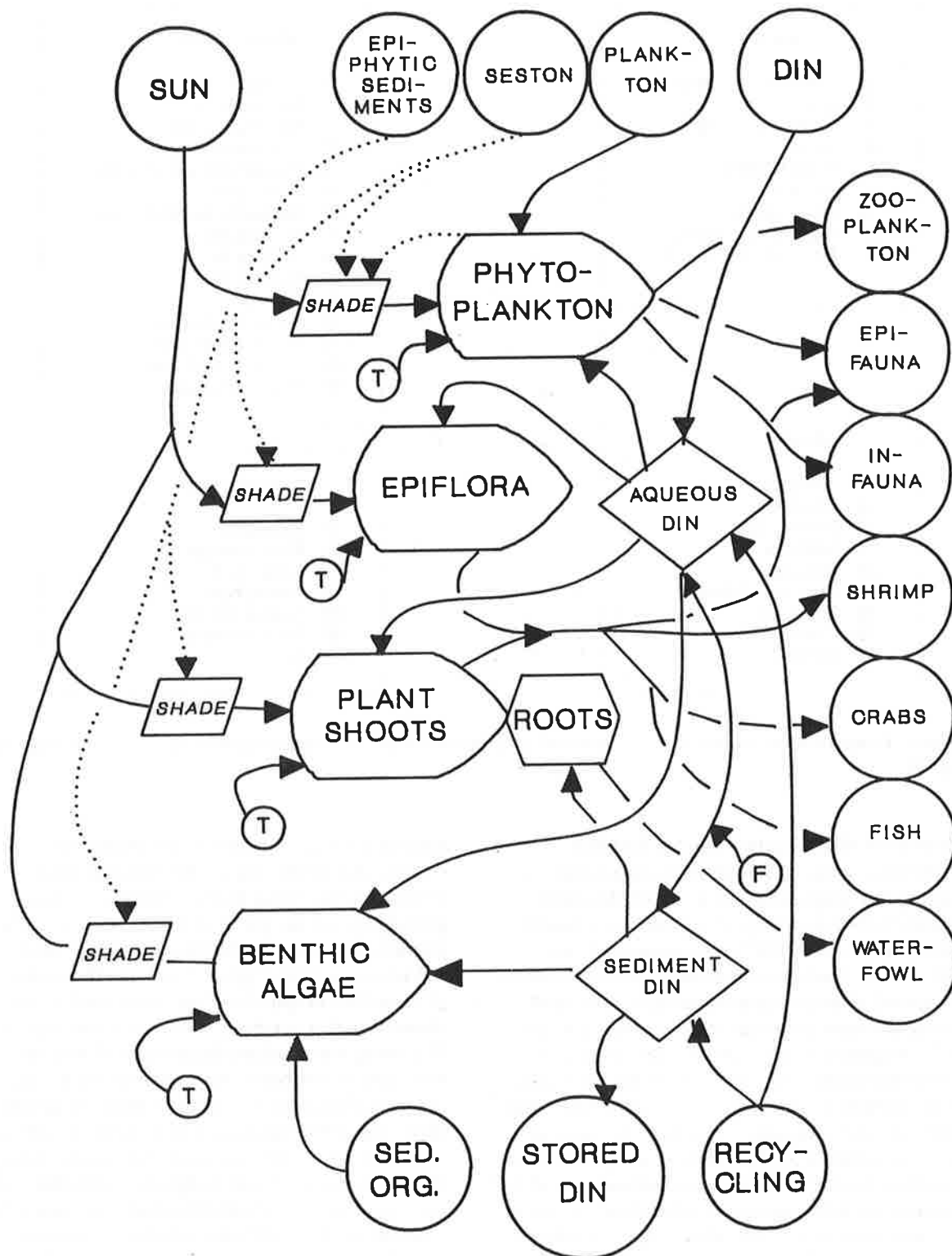


Figure 2. The compartmental structure and flow diagram used to simulate the autotroph submodel for studies of submerged aquatic vegetation in the upper Chesapeake Bay. For clarity of the diagram, the flows indicating losses from compartments due to respiration, excretion, or mortality have been omitted (redrawn from Kemp et al. [23]). T = temperature; F = turbulence; DIN = dissolved inorganic nitrogen; sed. org. = sediment organics.

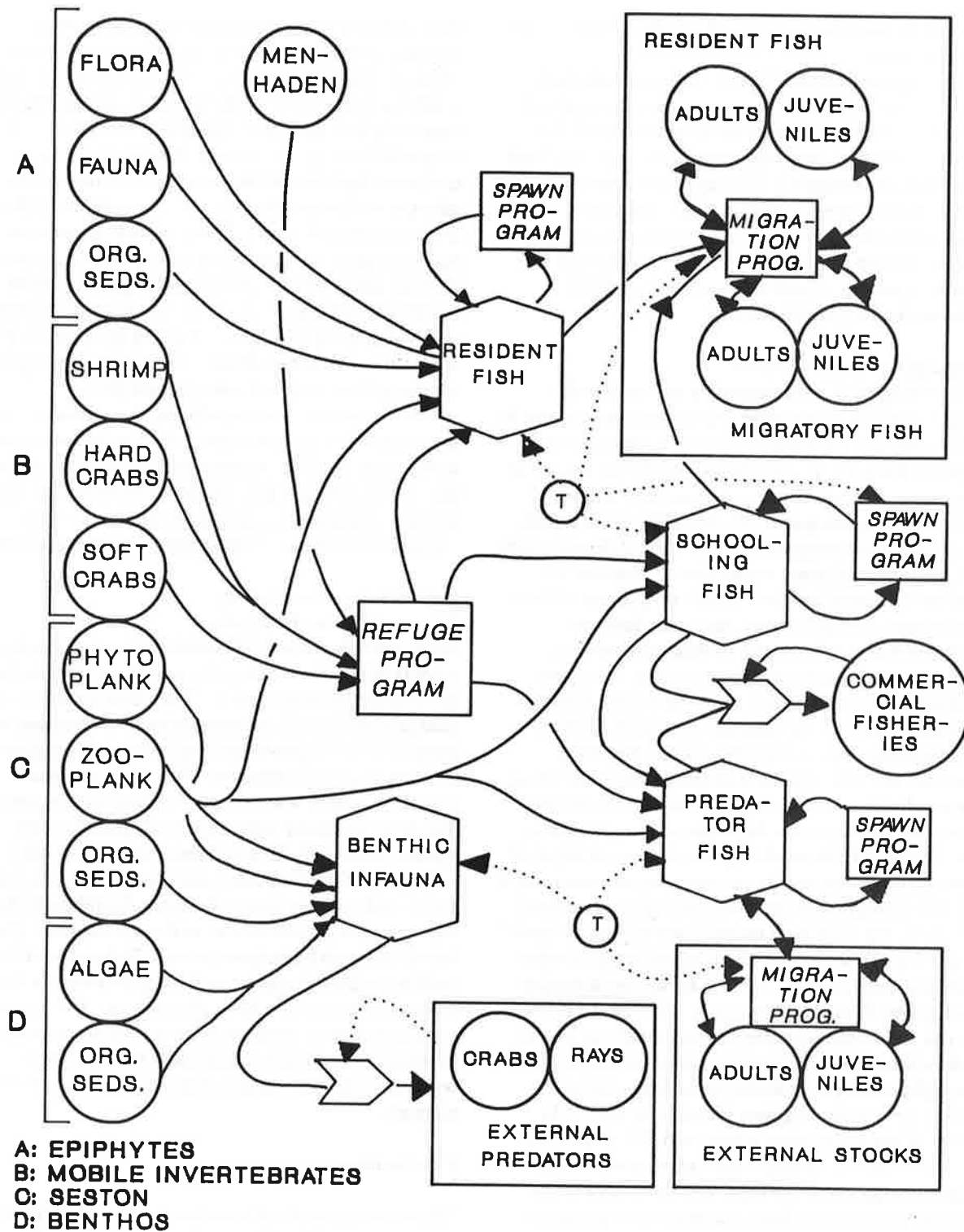


Figure 3. The compartmental structure and flow diagram used to simulate the nekton submodel for studies of submerged aquatic vegetation in the upper Chesapeake Bay. As in Figure 2, some flows have been omitted for clarity (redrawn from Kemp et al. [23]). T = temperature; org. sed. = organic sediments.

focus on controlling nutrient and sediment inputs to the Bay ecosystem.

The approach exemplified by the Maryland studies illustrates the necessary strong coupling that must exist between conceptual-simulation ecosystem modeling, empirical research, and management to insure that both ecological and socioeconomic needs are considered within the ecosystems context. The framework proposed by Kemp et al. [22] should be considered a first-generation "model" for guiding the efforts of Bay scientists, managers, and users to resolve current problems in the Chesapeake Bay.

Seagrass Ecosystem Models

While Kemp et al. were beginning studies on submerged aquatics in the upper Bay, comparable research with ecosystem modeling as a major component was begun in the lower Bay on the two dominant species of seagrasses, *Zostera marina* (eelgrass) and *Ruppia maritima* (widgeon grass) [46, 48]. The initial modeling efforts in the program were directed at development of a realistic conceptual model that (1) depicted the principal structural and functional components of these ecosystems, (2) summarized the extant data and identified where information or data were lacking, (3) could, on mathematical translation and computer simulation, simulate carbon flow and trophic interactions that implicitly included the effects of altered or variable submarine light environments, dissolved inorganic nutrient regimes, and the effects of declining seagrass habitat, and (4) would guide a multi-disciplinary research effort over its projected five-year duration. Results of this modeling activity are not reported in the literature because its primary purpose was to be a research management tool [Wetzel, unpublished data].

Continuing work with the seagrass ecosystem model for the lower Chesapeake Bay resulted in the development of a detailed subsystem model for the dominant macrophyte, *Z. marina* (Figure 4). Driving the development and implementation of this model was a clear need to better address the relationships among submarine light, macrophyte photosynthesis, epiphytic fouling, and epiphyte-grazer interactions [42]. The results of previous research demonstrated empirically the close coupling among these factors and principal system components, and suggested that rather small changes in submarine light available for macrophyte photosynthesis might have great impact on community stability and might explain the dramatic declines in seagrass distribution and abundance observed in the lower Bay [32, 46].

On the basis of simulation studies with this model, Wetzel and Neckles [48] concluded that submarine

light intensity was the principal factor governing eelgrass community stability and long-term survival (10-year simulation studies). Factors that affect light available to the macrophyte include suspended particles (primarily fine inorganic sediments) and the degree of epiphytic fouling. As a result of these modeling exercises, further empirical studies have been undertaken to understand the principal controls on epiphyte growth (light and nutrients) and the role of resident epiphytic grazers. The model in its present version can evaluate the effects of altered light regimes as influenced by suspended particles on the depth distribution and productivity of eelgrass. Current revisions to the model are intended to address water quality alterations resulting from proposed management plans.

From a management standpoint, these models and their simulation results support the conclusions offered by Kemp et al. [23]: Nutrient and sediment inputs to the subestuaries and Bay mainstem must be reduced if existing submerged aquatic macrophytes are to be conserved or enhanced either naturally or artificially.

Network Analysis and the Chesapeake Bay Ecosystem

Wulff and Ulanowicz [55] developed ecosystem-level models of the Chesapeake Bay and Baltic Sea for comparative purposes (Figure 5). They developed a common compartmental and interactive flow structure and addressed ecosystem-level characteristics by applying network analysis techniques. The result of these modeling studies was the rather surprising conclusion that the Chesapeake appears to be operating under greater stress than the Baltic Sea ecosystem. The interesting aspect of this work is not the actual conceptualization and construction of the network (although these are important considerations in the development phase) but the analysis techniques per se. This approach (i.e., network analysis) appears to have great promise for the analysis of ecosystem-level phenomena. The potential for application of these ecosystem analysis techniques to management issues, particularly for issues that address whole ecosystems, requires further study and support.

Conclusions

These ecosystem-level simulation models for coastal ecosystems and the Chesapeake Bay illustrate the varied roles for modeling at this level of hierarchical organization. Models have provided conceptual schemes for viewing the system as a whole, summarized large data sets to identify both sensitive processes and data gaps, guided research into profitable areas for

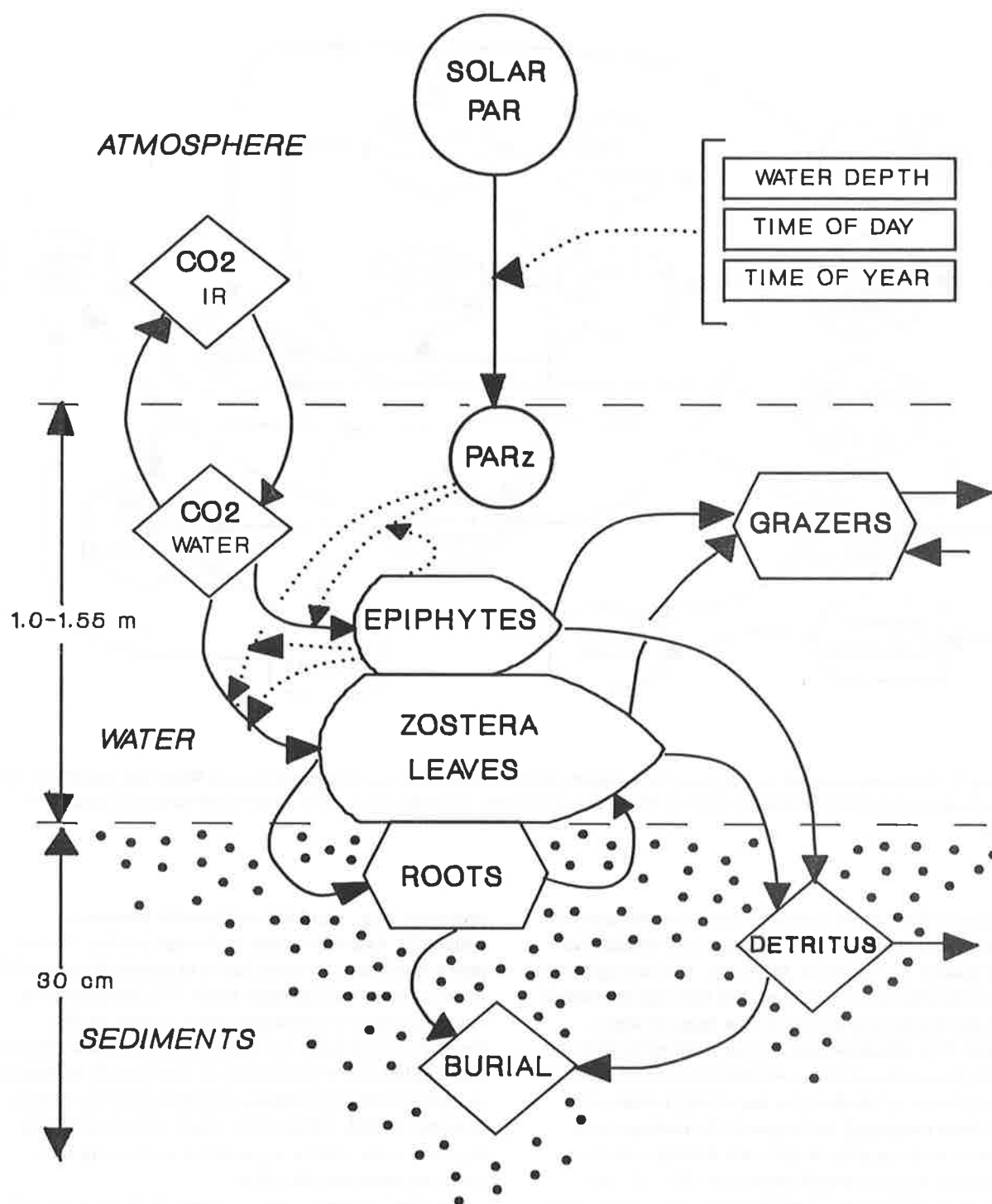


Figure 4. The compartmental structure and flow diagram used to simulate *Zostera marina* (eelgrass) photosynthesis and growth in the lower Chesapeake Bay under various conditions of environmental stress. Solid lines represent material transfers and dashed lines represent negative feedback controls (from Wetzel and Neckles [48]). PAR_z = photosynthetically active radiation at depth z .

tool in research and management will help insure that the functioning of the whole system or principal subsystems is not compromised as a result of actions directed at lower levels of ecological organization.

We suggest that the best way to approach this effort is by dedicating resources, both financial and scientific, for ecological modeling of Chesapeake Bay living resources. The technology and scientific expertise exists today within the Bay community. The level of effort and commitment of funds should be comparable to that of the present hydrodynamic-water quality modeling efforts. Effective, long-term management of Chesapeake Bay living resources depends to a large extent on this commitment and coupling with physical-water quality models. Development of these "hybrid" models will better address both fundamental and management-oriented questions related to living resources and environmental quality.

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The Functional Role of Estuarine Benthos

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Introduction

The benthic environs of the Chesapeake Bay range from intertidal flats to deep channels. Each habitat within this range is characterized by a myriad of organisms and processes, the majority of which are cryptic and not easily observed or understood. Yet it has been suggested that most physical, chemical, geological, and biological processes in estuaries are regulated or modified by interactions with the benthic system [84]. An increasing body of evidence demonstrates that benthic organisms are involved in basic functional processes influencing energy transfer, nutrient dynamics, and cycling and fate of toxicants. This review evaluates functional relationships of benthic organisms in this complex system. We limit our coverage to subtidal areas; reviews for vegetated and intertidal habitats are presented elsewhere [91, 264, 282].

Specific questions regarding the functional importance of subtidal benthos that we will address are:

- Given present information, can we identify major benthic habitats in Chesapeake Bay? Can we predict organism abundance, biomass, and functional group composition and scales of variability in these parameters on the basis of known animal-habitat relationships?
- For distinct habitat types what are the characteristic mechanisms and rates of benthic processes that affect function? Can we predict the importance of biotic processes relative to physical processes in nutrient dynamics and in the movement of sediments or their associated toxicants?

- What factors control benthic production? How does benthic production contribute to the overall energy budget for the Bay?

Animal-Habitat Relationships

Early studies focusing on the ecology of macrobenthic organisms ($>500\ \mu\text{m}$) in the Chesapeake Bay estuarine system identified strong patterns of association between individual organism abundance or community structure and gradients of physical-chemical parameters [33-37, 80, 81, 87, 103, 108, 138, 152, 165, 168, 178, 203, 211, 245, 252, 271, 280, 315, 339, 343, 350, 352]. These studies demonstrated that salinity is the major factor governing organism distribution and patterns of species diversity in the Chesapeake Bay, as it is in estuaries worldwide [83, 270, 360]. At moderate to high salinities within the estuary, patterns of organism distribution, abundance, and diversity are further correlated with sediment type [34, 81, 210]. Distribution and abundance patterns for dominant benthic macrofaunal invertebrates in the Chesapeake Bay estuarine system reflect these relationships (Table 1).

Other major physical gradients within the estuary also influence the distribution of benthic communities and abundances of component organisms. Oxygen availability in the water column exerts an important influence on benthic community structure and function, both directly by affecting benthic organisms' metabolic processes and indirectly by affecting water column processes, particularly in the upper Chesapeake Bay [137, 151, 166, 221, 226, 260]. Energy gradients and frequency of bottom disturbance are important in limiting species distribution patterns in the lower Bay [210, 311, 313] and also influence how biological and physical processes interact in the benthos [314, 315]. Organism response to energy gradients seems especially important in determining where large infaunal

This review is contribution no. 1595 from the College of William and Mary, Virginia Institute of Marine Science.

Table 1. Dominant macrofaunal groups documented in major benthic habitats of the Chesapeake Bay.

Taxon	TF A B	OL A B	LM A B	HM-M A B	HM-MX A B	HM-S A B	P-M A B	P-MX A B	P-S A B	PE-MX A B	PE-S A B
<i>Oligochaeta</i> (esp. <i>Linodrilus</i> spp.)	*	+									
Chironimidae (In)	+										
<i>Corbicula fluminea</i> (B)	+										
<i>Rangia cuneata</i> (B)		*	-	+							
<i>Leptocheirus plumulosus</i> (Am)		+									
<i>Marenzelleria</i> (<i>Scolecoides</i>) <i>viridis</i> (A)		+	+			+					
<i>Gammarus</i> spp. (Am)	-	-									
<i>Sireblosipio benedicti</i> (A)			+	+	+	+					
<i>Heteromastus filiformis</i> (A)			+	-	-	-					
<i>Tubificoides</i> spp. (A)	-	+	+	+	+	+					
<i>Nereis succinea</i> (A)	-	+	+	+	+	+					
<i>Cyathura</i> spp. (Is)	-	+	+	+	+	+					
<i>Macoma balthica</i> (B)	-	+	+	+	+	+					
<i>Mya arenaria</i> (B)	-	+	+	+	+	+					
<i>Gemma gemma</i> (B)	-	+	+	+	+	+					
<i>Crassostrea virginica</i> (B)	-	-	-	-	-	-					
<i>Leucon americanus</i> (C)			+	+	+	+					
<i>Mulinia lateralis</i> (B)	-		-	-	-	-					
<i>Paraprionospio pinnata</i> (A)		-	-	+	+	+					
<i>Mediomastus ambiseta</i> (A)				+	+	+					
<i>Glycinde solitaria</i> (A)				-	-	-					
<i>Glycera</i> spp. (A)				-	-	-					
<i>Pseudeurythoe paucibranchiata</i> (A)				-	-	-					
<i>Acteocina canaliculata</i> (G)				-	-	-					
<i>Pectinaria gouldii</i> (A)				-	-	-					
<i>Loimia medusa</i> (A)				-	-	-					
<i>Amphelisca</i> spp. (Am)				-	-	-					
<i>Molgula manhattensis</i> (Ur)				-	-	-					
<i>Sigambra tentaculata</i> (A)				-	-	-					
<i>Nephtys</i> spp. (A)				-	-	-					
<i>Macrorynchene zonalis</i> (A)				-	-	-					
<i>Blattaria heteroseta</i> (A)				-	-	-					
<i>Notomastus latericeus</i> (A)				-	-	-					
<i>Chaetopterus variopedatus</i> (A)				-	-	-					
<i>Clymenella torquata</i> (A)				-	-	-					
<i>Phoronis</i> spp. (Ph)				-	-	-					
<i>Mercenaria mercenaria</i> (B)				-	-	-					
<i>Tellina agilis</i> (B)				-	-	-					
<i>Ensis directus</i> (B)				-	-	-					
<i>Anastigos caperatus</i> (A)				-	-	-					
Haustoriidae (Am)				-	-	-					
<i>Spioptanus bombyx</i> (A)				-	-	-					
<i>Mytilus edulis</i> (B)				-	-	-					

For notes see page 27.

species and associated biogenic structuring of sediments will be found in the estuary [311, 312, 315]. Gradients in allochthonous and autochthonous carbon input are likely to be important in structuring communities [269], but have generally not been studied directly in the Chesapeake Bay estuarine system [185].

Estuaries are characteristically variable, and the scales of spatial and temporal variability are important for understanding and predicting benthic community structure. It seems reasonable to expect that elucidating these relationships will help to identify patterns of community function as well. Studies of temporal variability in estuarine benthic communities have demonstrated strong seasonal trends as well as variability on shorter and longer time scales [37, 70, 86, 152, 163, 166, 268, 359]. Many non-seasonal trends are difficult to explain [37, 86, 359]; others can be attributed to changes in physical conditions, especially salinity and oxygen [163, 166, 325].

Characterizing major estuarine habitat types on the basis of a full range of physical variables facilitates understanding of animal-habitat relationships and benthic response to important physical variables both within and among habitat types [311, 315]. This approach seems especially important in the Chesapeake Bay, where many physical factors are highly correlated and spurious correlations between biotic and physical processes are likely to be overlooked. Given the existing biological and physical data base for the Chesapeake Bay, we present our view of major benthic habitat characteristics in Table 2.

Effects of Benthic Organisms on Physical Dynamics and Chemical Processes of Estuarine Sediments

The influence of benthic organisms on estuarine function, particularly with respect to parameters and processes that affect toxicant and nutrient dynamics,

depends in part on the physical-chemical behavior of these reactants. Some substances remain as solutes in the water mass, while others have affinities for particles [263]. Toxicants tend to bind with small particles that are rich in organics [263, 308]. Therefore, we center our discussion on current knowledge of organism-sediment-fluid interactions, with particular reference to the Chesapeake Bay. Mechanisms and rates of organism interactions with the sedimentary environment are first summarized so that the relative magnitudes of processes are clear and so that predictions can be made regarding the likely importance of different processes throughout the estuary. Effects on sediment physical dynamics, chemical diagenesis, and nutrient exchange across the interface are reviewed. A final section considers how these basic interactions are likely to influence toxicant transport and fate within the Chesapeake Bay and other estuarine or shallow coastal systems.

Functional Characteristics of Chesapeake Bay Benthos

Biogenic alteration of sediment and pore-water characteristics, transport, and mixing is primarily a function of two things: organism densities, and their ways of living and moving within sedimentary deposits. Yet most benthic environments encompass a myriad of processes and interactions that reflect the inevitable variety of living habits exhibited by resident fauna. Classification into broader functional categories can be useful for identifying how larger groups of organisms influence benthic processes [114, 205, 235, 287, 288]. Although simple classification schemes may not allow adequate prediction of organism effects on some processes, especially sediment transport [179, 256], the approach greatly facilitates estimates of the magnitudes of different processes, thereby lending insight into the potential outcome of many interactions [127, 140].

Feeding methods and rates are among the most important variables influencing sediment modification by benthic organisms. Feeding methods determine where benthic organisms obtain particles or other food items (i.e., the water column, surface, or subsurface). Rates of feeding determine, in part, the rates at which organisms move and mix sediments. Living positions and motility patterns determine the potential limit of influence by a given organism within a sedimentary deposit. Motility is also strongly correlated with the types of structures benthic organisms build or create. The types of structures organisms produce (e.g. tubes, fecal pellets, tracks) determine, in part, which benthic processes they are likely to influence. Information on functional characteristics, including feeding type,

Notes for Table 1 (facing page). Habitat abbreviations: for salinities, TF = tidal freshwater, OL = oligohaline, LM = low mesohaline, HM = high mesohaline, P = polyhaline, PE = poly-euhaline; for sediments, M = mud, MX = mixed, S = sand. A = no. individuals/m², categories are: (-) <100, (+) 100 - 1000, (*) >1000; B = ash-free dry weight/m², categories are: (-) < 1, (+) 1-20, (*) >20. Taxonomic categories are: A = Annelida, Am = Amphipoda, B = Bivalvia, C = Cumacea, G = Gastropoda, In = Insecta, Ph = Phoronida, Is = Isopoda, Ur = Urochordata. Data compiled from [80, 88, 158, 168, 187, 210, 252, 280, 311, 312].

† Locally abundant only

Table 2. Characteristics of major benthic habitats in the Chesapeake Bay estuarine system.

Habitat type	Physical characteristics	Macrobenthic community characteristics	Macrofauna density	Macrofauna biomass
Tidal freshwater				
Shoals	Shallow depths Mud to sand sediments Wave- and tide-dominated High turbidity (allochthonous carbon) Low to moderate light penetration	Stenohaline, otherwise eurytopic fauna Deposit and suspension feeders Moderate diversity	Low	Bivalves high Others low
Channels	Intermediate depths Mud to sand sediments Fluid mud possible Tide-dominated High turbidity (allochthonous carbon) No light penetration Occasional low oxygen	Stenohaline, otherwise eurytopic fauna Deposit and suspension feeders Moderate diversity	Low	Bivalves high Others low
Oligohaline				
Shoals	Shallow depths Mud to sand sediments Wave- and tide-dominated High deposition (allochthonous carbon) Low to moderate light penetration	Euryhaline, eurytopic fauna Deposit and suspension feeders Low diversity	Low to high	Bivalves high Others low
Channels	Moderate depths Mud sediments Fluid mud possible Tide-dominated High deposition (allochthonous carbon) No light penetration Occasional low oxygen	Euryhaline, eurytopic fauna Deposit and suspension feeders Low diversity	Low to high	Bivalves high Others low
Mesohaline				
Shoals	Shallow depths Sand sediments Wave- and tide-dominated Low to moderate turbidity Moderate light penetration Occasional low oxygen	Euryhaline, eurytopic fauna All feeding types Moderate diversity*	Moderate to high*	Bivalves high Others moderate
Channels	Intermediate to deep depths Mud sediments Fluid mud possible Tide-dominated High turbidity No light penetration Seasonal low oxygen	Euryhaline, eurytopic fauna All feeding types Moderate diversity*	Moderate to high*	Bivalves high* Others moderate*
Polyhaline				
Shoals	Shallow depths Sand sediments Wave- and tide-dominated Low turbidity High light penetration	Stenohaline, eury- to stenotopic fauna All feeding types Moderate diversity	Low to moderate	Low to moderate

Basin	Intermediate depths Silt and fine sand sediments Tide-dominated Low turbidity Seasonal light penetration Occasional low oxygen	Stenohaline, eury- to stenotopic fauna All feeding types High diversity	Moderate	Moderate to high
Channels	Moderate to deep depths Mud to sand sediments Tide-dominated Moderate turbidity No light penetration Occasional low oxygen	Stenohaline, eury- to stenotopic fauna All feeding types Moderate to high diversity	Moderate	Low to high
Poly-euhaline				
Shoals	Shallow depths Sand sediments Wave- and tide-dominated Low turbidity High light penetration	Stenohaline, stenotopic fauna All feeding types Moderate to high diversity	Low to moderate	Low to moderate
Basin	Intermediate depths Silt and fine sand sediments Tide-dominated Low turbidity Seasonal light penetration	Stenotopic fauna All feeding types High diversity	Moderate	Moderate to high
Channels	Moderate to deep depths Mud to sand sediments Tide-dominated Low turbidity Seasonal light penetration	Stenotopic fauna All feeding types High diversity	Moderate	Low to high

* Except when low oxygen conditions prevail

mobility mode, defecation mode, and type of dwelling produced are summarized along with depth range and sediment reworking rates for common Chesapeake Bay organisms in Table 3.

Oxygen distribution into the sediments is also strongly influenced by functional characteristics of resident fauna. In most benthic environments, oxygen penetration into the sediment by molecular diffusion is limited to a few millimeters; transport into deeper layers is strongly mediated by macrofaunal ventilation [283, 284]. Organisms ventilate sediments by feeding, respiring, and burrowing [200, 227, 236]. Ventilation rates reported for marine and estuarine benthic species vary considerably. Mangum [224] reported low ventilation rates (<4 ml/hr) for maldanid polychaetes, but rates as high as 480 ml/hr have been reported for the suspension-feeding polychaete *Chaetopterus varipodatus* [74]. Others have reported rates for polychaetes ranging from 1.5 ml/hr to about 180 ml/hr, with most rates ranging between about 50 and 180 ml/hr [8, 11, 74, 75, 197, 227]. Measured rates for bivalves range from about 5-90 ml/hr for *Mulinia lateralis* [324] up to 344 ± 32 ml/hr for *Mya arenaria* [225]. Low rates reported for amphipods [118] contrast with a wide

range of rates reported for burrowing decapods (29-3,204 ml/hr) [101, 136, 191].

Patterns of ventilation exhibited by tube and burrow dwellers are generally complex, but all species studied are apparently intermittent irrigators [8, 74, 101, 117, 136, 224]. This means that nearly all tube and burrow structures are likely to undergo periodic hypoxia, which may be important for the dynamics of some important biogeochemical processes such as the coupling of nitrification and denitrification (see below).

Through excretion benthic organisms add reactive materials to the sediment that may alter diagenetic processes. Nearly all invertebrates excrete urine that contains sugars and amino acids as well as ammonia [64, 354]. Average ammonium excretion rates for small infaunal invertebrates range from <0.01 $\mu\text{mol/hr}$ for the small amphipod *Corophium volutator* to >1.0 $\mu\text{mol/hr}$ for the polychaete *Nereis virens* [45, 157, 191, 198, 202, 364]. Mucous exopolymers are produced in great quantities by bacteria and many other benthic organisms including macrobenthos [69, 162, 179, 298]. Mucopolysaccharides and associated compounds alter the "stickiness" of sediments and can be a source of reactive organic substances. Production of these sub-

Table 3. Functional characteristics and sediment reworking rates due to feeding and burrowing for common macrobenthic organisms of the Chesapeake Bay. Taxonomic groups are as indicated in Table 1.

Taxon	Feeding type	Mobility	Defecation mode	Dwelling structure produced	Depth range (cm)*	Sediment reworking rate (mg dry wt. of individual)	Notes	Source(s)
<i>Oligochaeta</i> (A) (especially <i>Limnodrilus</i> spp.)	HD	LM	S,P	B	0-20	0.7-1.2* (0.5) 15.1-28.4* (0.5)	At 3.5°C At 21.7°C	[16, 45, 235, 314, 338]
Chironomidae (In)	SD,SF	LM		T	0-10			[235]
<i>Corbicula fluminea</i> (B)	SF	M	S		0-2	14.8 (1.0 gram wet wt.) 24.9 (31.0)	For <i>C. japonica</i>	[248, 275]
<i>Rangia cuneata</i> (B)	SF	LM	S,P		0-5			[53, 329, 334, Schaffner unpub.]
<i>Leptocheirus plumulosus</i> (Am)	SF,SD	S		T	0-10			[160, 281]
<i>Maranzellaria</i> (<i>Scolecopeloides viridis</i>) (A)	SD,SF	M?	S,R	B	0-20	58.9-177.2 (juveniles)* 98.5-207.4 (adults)	At 20-25°C	[79, 160]
<i>Gammarus</i> spp. (Am)		M	S,R		0+			[39, 192]
<i>Streblospio benedicti</i> (A)	SD,SF	S,LM	S,R	T	0-10	11.7-18.7*		[79, 160, 195]
<i>Heteromastus filiformis</i> (A)	HD	LM	S,P	B,V	0-35	262.5-700*	Yearly range	[52, 160, 195, 355]
<i>Tubificoides</i> spp. (A)	HD	LM	S,P	B	0-25	0.3-1.4* (0.2) 20.7-31.9 (0.6) 0.8-3.5 (0.6)	At 3.5°C** At 14.0°C At 21.7°C	[16, 160, 338]
<i>Nereis succinea</i> (A)	SD,O	LM	S,P	B	0-30	3.9 (0.9) 20.2 (5.8) 103.7 (38.1)	At 15°C	[55, 109, 160, 192, 311]
<i>Cyathura polita</i> (Is)	C?	M			0-15	2080 [§]	Displacement	[160, 247, 281]
<i>Lepidactylus dysticulus</i> (Am)	SF	M			0-3			[85]
<i>Macoma balthica</i> (B)	SD,SF	S	S,P	B	0-30	2.5, 27.0 (2.2) 4.5, 49.9 (5.1) 6.5, 61.0 (8.8) 9.1, 77.3 (14.3)	At 6°C, feces only, total	[10, 44, 49, 160, 192, 282, 315]
<i>Mya arenaria</i> (B)	SF	S	I,U	B	0-25	27 (1820) 23 (990)		[47, 145, 160, 192]
<i>Gemma gemma</i> (B)	SF	LM	S		0-2			[43, 247, 321]
<i>Crassostrea virginica</i> (B)	SF	S	I,RI		0+	140.0 (480) 188.5 (1410) 222.9 (1140)		[145, 147, 192]

<i>Leucon americanus</i> (C)	SD	M			0-1,0+		[Schaffner unpub.]
<i>Mulinia lateralis</i> (B)	SF	M	S,R		<u>0-5</u>		[192, 280, 311, 329]
<i>Paraprionospio pinnata</i> (A)	SD,SF	LM	S,R	B	<u>0-25</u>	39.6-51.9* (juveniles) 90.9-232.8 (adults)	[79, 195, 217, 280, 311]
<i>Mediomastus ambiseta</i> (A)	HD	LM	S,P	T	0-6		[195, 311]
<i>Glycinde solitaria</i> (A)	C	M			0-6		[195, 311]
<i>Glycera</i> spp. (A)	C, BD	M	R	BG	0-15		[192, 195, 311]
<i>Pseudeuruthoe paucibranchiata</i> (A)	C?, BD	M?			0-24		[195, 311]
<i>Acteocina canaliculata</i> (G)	C	M	S,P		0-2	2080 [§]	[192, 247, 311]
<i>Pectinaria gouldii</i> (A)	HD	LM	S,U,P	T,V?	0-8	6000 (80)	[126, 142, 195, 311, 355]
<i>Loimia medusa</i> (A)	SD	S	I,U	T	<u>0-15</u>	5100 (900)	[142, 149, 195, 286, 311]
<i>Ampelisca</i> spp. (Am)	SD	S	S,R	T	<u>0-5</u>		[241, 311]
<i>Molgula manhattensis</i> (Ur)	SF	S	S,R		0+	80 (240) 40 (110)	[145, 192, 311]
<i>Sigambra tentaculata</i> (A)	C?, BD	M?			0-30		[109, 195, 311]
<i>Nephtys</i> spp. (A)	C, SD	M	R		0-10		[62, 109, 192, 195, 306, 307, 311, 337]
<i>Macroclymene zonalis</i> (A)	HD	S	S,C	T,V	0-20		[224, 286, 311]
<i>Bhawania heteroseta</i> (A)	C?	M			0-10		[109, 311]
<i>Notomastus latericeus</i> (A)	BD	LM	B		0-15		[109, 311]
<i>Chaetopterus variopedatus</i> (A)	SF	S	S,P	T	<u>0-15</u>		[311]
<i>Clymenella torquata</i> (A)	HD	S	S,U	T,V	0-30	1200 (47)	[95, 142, 224, 286, 289, 311]
<i>Phoronis</i> spp. (Ph)	SF	S	S	T	<u>0-20</u>		[304]
<i>Mercenaria mercenaria</i> (B)	SF	LM	S,R		<u>0-15</u>		[192, 329]
<i>Tellina agilis</i> (B)	SF, SD	M	R		<u>0-10</u>		[192, 329, Schaffner unpub.]
<i>Ensis directus</i> (B)	SF	M	S,R		<u>0-20+</u>		[192, 329, Schaffner unpub.]
<i>Haustoriidae</i> spp. (Am)	SF	M			0-10	277-9900	[171, Schaffner unpub.]
<i>Spiophanes bombyx</i> (A)	SD, SF	S	S	T	<u>0-15</u>	4.3-14.2 [#]	[79]
<i>Mytilus edulis</i> (B)	SF	S	S, RI		0+		[192, 311]

For notes see page 32.

stances is of potential importance for physical sediment dynamics and chemical processes [4, 162, 179, 256, 298]. Macrobenthic tube and burrow linings are typically composed of mucopolysaccharides, which may be rich in sulfides or phosphates [8, 202, 246, 366].

Effects of Benthic Organisms on Physical Sediment Dynamics

The effects of benthic organisms on physical sediment dynamics are well documented for both marine and freshwater environments [179, 193, 205, 235, 238, 287, 288]. It has been shown that organism activities may significantly influence rates of sediment deposition, types of particulate materials deposited, sediment stratigraphy, mass properties, and transport probabilities.

Faunal effects on deposition processes. Enhanced deposition of particulate materials due to the feeding activities of benthic suspension feeders (including most major taxonomic groups) is a common and probably important process promoting benthic-pelagic coupling in estuarine and shallow coastal systems [97, 98, 121, 145-148, 275, 276, 299, 346]. Chesapeake Bay oysters on 0.405 ha of bottom may produce up to 981 kg dry weight of biodeposits weekly [145]. Calculations suggest that suspension feeders can daily filter $\geq 100\%$ of the volume of water in shallow-water habitats, including parts of the Chesapeake Bay [63, 78, 125, 168].

The physical characteristics of particles transferred from the water column to the benthos by suspension feeders are different from those exhibited by naturally sedimenting particles [276, 299]. Digestive processes may alter clay mineral structure [15, 276], and this may

influence particle-toxicant relationships [293]. When particles retained by four species of suspension feeders were studied by Haven and Morales-Alamo [148], 82-93% by volume were smaller than 4 μm ; 95% were smaller than 9 μm . Fecal pellets produced by suspension feeders are typically in the size range of sand particles [192, 276] and have settling velocities much higher than those calculated for fine sediment particles [146, 233, 275, 276, 331]. Such biodeposition processes may lead to the formation of organic-rich muds in areas where muds would not be predicted to settle on the basis of hydrodynamic regimes [276].

Benthic organisms may enhance sediment deposition by other mechanisms as well. Lynch and Harrison [219] showed that a dense colony of the small tube-building amphipod, *Ampelisca abdita*, enhanced sediment deposition at a shallow subtidal (3.6 m) site in the York River, Virginia, presumably by sediment trapping. Sediment trapping by tube-building organisms may increase the deposition of fine sediments in areas where they would not typically be deposited [22].

Bioturbation: effects on sediment mixing and mass properties.

Effects of benthic organisms on sediment mixing vary as a function of physical processes (especially lateral advection and sediment accumulation) and biological feeding, burrowing, and tube-building processes [180, 243]. Interactions between these processes are stochastic and can have complex effects on particle distributions [180, 300].

Diffusional models of sediment mixing processes [230] predict that when D_b (the particle bioturbation coefficient) is high relative to the sedimentation rate, bioturbation homogenizes near-surface sediments [135] and obscures the effects of other processes on stratigraphic signals [316]. When the sedimentation rate is high relative to bioturbation, organisms have little effect on particle redistribution [262]. Schaffner et al. [315] suggested that even relatively rapid bioturbation, limited to a very narrow mixing zone at the sediment surface, had little effect on mixing in high-sedimentation areas of the James River.

Non-random bioadvective mixing by head-down deposit feeders may move sediment large distances quickly [292]. In the absence of diffusional mixing, marker peaks (introduced at the sediment surface, as a toxicant might be added) subjected to advective mixing move down within the sediment with little or no broadening [292]. Diffusional and advective mixing act together to disperse materials introduced at the sediment surface uniformly throughout feeding depths [300, 301]. Rapid turnover and homogenization of the sediment column to feeding depths typifies areas dominated by head-down deposit feeders [94, 224, 286, 314].

Notes for Table 3 (previous pages). Feeding types are: BD = subsurface deposit, HD = head down subsurface deposit, SD = surface deposit, SF = suspension, C = carnivore, O = omnivore. Mobility modes are: S = sedentary, LM = limited mobility, M = mobile. Defecation modes are: S = material deposited on surface, I = injected into water column, P = pellet, ovoid or ellipsoid, R = rod, RI = ribbon, C = coil, U = unconsolidated. Dwellings produced are: B = burrow, T = tube, V = void. Sediment reworking rates are expressed as mg dry weight sediment individual⁻¹ day⁻¹, values reported for egestion (feces and pseudofeces, if present) unless otherwise indicated.

* Underlining indicates that species feeds at the sediment surface; o+ indicates depth range above sediment surface.

+ Calculated from data presented by author.

* First value is rate without suspended particles; second is rate with suspended particles, originally reported as mg individual⁻¹ h⁻¹ (D. Dauer, personal communication)

§ Assuming 1 cm³ = 2.08 grams dry wt.

** For *Peloscolex multisetosus*.

++ For *Amphitrite ornata*.

Feeding, burrowing, and tube-building may have differing effects on particle distributions. Selective ingestion of fine particles, incorporation of these particles into fecal pellets, and subsequent deposition of fecal pellets at the sediment surface will tend to keep fine particles in an active surface mixed layer [180, 338]. Burrowing may produce the opposite results by selectively "pumping" coarse sediment fractions towards the surface [239]. The activities of some deposit feeders result in graded bedding, typically with a coarse lag layer forming below feeding depths [289, 300, 338]. Most species studied also exhibit some selectivity for finer particles in tube construction [8, 132, 202, 318].

Alterations in particle characteristics and mass properties result from bioturbation activities. Selective ingestion of fine or organically encrusted particles by many deposit feeders means that fecal pellets are often organically enriched relative to ambient sediments [213, 214]. Fecal pellet production increases particle size by packaging small silt- and clay-sized particles into sand-sized particles. Schaffner et al. [315] noted that pelletization of surface sediments (0-5 cm) resulting from the feeding activities of *Macoma balthica* was nearly complete for a site in the James River during the early summer. Fecal pellets and remnants of fecal pellets are an important component of Chesapeake Bay sediments [148, 192, 195, 252]. In muddy sediments, feeding and burrowing activities tend to increase sediment water content and decrease compactness [205, 290].

Erosion and sediment transport. The complexities of processes governing organism effects on sediment erosion and transport have been discussed by Jumars and Nowell [179]. Benthic organisms affect erodibility and sediment transport via alteration of: (1) fluid momentum impinging on the bed [102, 319], (2) particle exposure to the flow [256], (3) adhesion among particles [128-130, 164], and (4) particle momentum [174, 179, 256, 302]. Because the net effects of a species on erosion and transport are highly dependent on flow conditions, bed configuration, and community composition, general predictions of stabilization vs. destabilization cannot be made given existing data [179, 216]. Interactions are likely to be especially complex in estuaries since both physical and biological processes are characteristically dynamic.

Effects of Benthic Organisms on Chemical Diagenesis and Nutrient Flux across the Sediment-Water Interface

Benthic metazoans influence sediment diagenesis and nutrient flux both directly (by modifying sediment diffusion geometry, mixing sediments, pumping fluids,

and adding reactive substances such as mucus) and indirectly (by influencing microbial activities and growth rates). Basic metabolic processes contribute to the consumption of oxygen, the degradation of organic matter, and the release of carbon dioxide and limiting nutrients.

Structural effects introduced by macrobenthos strongly influence sediment diffusion geometries by altering the pathways along which fluids flow and solutes are exchanged [4, 6]. At typical natural densities, burrow construction by populations of single macroinfaunal species can increase oxic sediment volume by at least 30-150% [175, 198]. Ventilation enhances the daily exchange rate of solutes across the sediment-water interface in shallow marine and freshwater environments [104, 114, 232]. Changes in the sediment fabric may also enhance exchange by increasing sediment porosity and decreasing tortuosity [73, 114, 234]; these changes may enhance passive sediment ventilation [349]. Physical processes may enhance ventilation of abandoned biogenic structures [5]. Active tube and burrow ventilation by macroinfaunal invertebrates appears to be an important process governing oxic sediment volume and the distribution of infaunal organisms in the Chesapeake Bay [92, 311, 312].

Sediment ventilation by macrobenthic organisms increases the apparent diffusion within the sediments over that observed by molecular diffusion [2]. Empirically determined "apparent diffusion coefficients" range from about 10^{-5} to 10^{-4} $\text{cm}^2\text{sec}^{-1}$, i.e., 10-100 times higher than molecular diffusion coefficients for bulk sediments [4]. Aller's "average diffusion geometry" model, which accounts for the three-dimensionality of macrofaunal effects on diagenetic reactions, shows that: (1) pore-water solute concentration is a function of infaunal size and abundance, (2) reaction rate kinetics determine reactant response to animal activities, and (3) vertical profiles of solutes depend on depth variation in reaction rates as well as faunal attributes. In general, burrow creation and irrigation by macrofauna prevent solute buildup away from the sediment surface. Given organisms may, however, have varying effects on solute flux as a function of both organism characteristics (e.g., density or size) and the kinetics and depth dependence of reaction rates.

Macrobenthos may directly influence the concentration of reactants and the environment in which they are transformed. When coupled with lateral advection, biogenic sediment reworking will tend to increase inventories of organic carbon and other reactants in areas of high biological activity [4, 7]. The combined effects of sediment mixing and ventilation tend to move the redoxcline down into the sediment (or out from burrow and tube walls) by moving electron acceptors

down into the sediment [4, 114]. Therefore, macrobenthos effectively increase the depth at which more energetic diagenetic reactions can occur.

The distribution and rates of microbial activities within the sediment are generally determined by the availability of electron acceptors for respiration and the supply of metabolizable organic substances [56]. Infauna stimulate rates of microbially mediated organic decomposition and remineralization. Feeding by consumers increases the surface area of organic detritus available for microbial growth, rearranges particles and microniches or reexposes new surfaces, and may maintain microbial populations in a high-productivity growth phase [10, 12, 46, 68, 110, 112, 141, 144, 181, 215, 337]. Ventilation and mixing decrease metabolite buildup and increase electron acceptor supply [29, 198, 201, 202]. Organisms also add reactive substances [8, 298] or increase subduction or capture of reactive organic matter [4, 7].

Examples: effects on carbon, phosphorus and nitrogen cycling. Biodeposition by infaunal and epifaunal suspension feeders in the Chesapeake Bay and other coastal systems enhances organic matter deposition [98, 145, 183, 248]. Partly because of microbial stimulation, carbon mineralization rates are also enhanced in the presence of infaunal organisms [14, 46, 98, 113, 114, 201]. Two routes of remineralization—through metabolic processes and through production of animal tissues (see Benthic Energy Flow)—are the dominant pathways controlling carbon flux out of active surface sediments. Most benthic communities apparently lose relatively little carbon via burial [9, 28, 344].

Benthic organisms are likely to influence the distribution of phosphorus in sediments by enhancing deposition of organic and inorganic phosphorus, altering the position of the redoxcline, and redistributing particulates relative to the redoxcline. Studies on direct infaunal effects on phosphorus behavior and the importance of these processes for exchange in estuarine and marine habitats are generally lacking. In freshwater habitats, chironomids either enhance the release of phosphorus as a positive function of organism abundance and temperature [122, 123] or have no apparent effect on phosphorus flux [231]. Bioturbation by freshwater tubificids apparently slows the release of phosphorus during subsequent periods of anoxia because a ferric hydroxide-orthophosphate complex is prevented from forming [114]. Phosphate may be trapped in the burrow walls of marine and estuarine macroinfaunal organisms because of the formation of insoluble mixed iron-manganese complexes and phosphate sorption [3, 139, 190, 198]. Flux across burrow walls shows sensitivity to concentrations in the water column, as would

be expected given a functioning phosphate buffer system [198]. Bacterial processes may also be important in regulating net flux [99]. Animal excretion of phosphorus may be far greater than measured fluxes, a possibility suggesting that interacting physical or biological processes must limit flux [3, 40, 99, 198, 255].

Benthic nitrogen recycling involves numerous transformations mediated by both microbes and larger metazoans. Important mechanisms by which benthic organisms influence nitrogen transformations and flux are the same as those influencing the degradation of organic matter. These processes have been reviewed in detail [2, 4, 6, 114, 155, 200].

For a wide variety of marine, estuarine, and freshwater habitats, macroinfaunal invertebrates have been found to increase efflux of ammonium or total dissolved inorganic nitrogen and influx of oxygen across the sediment-water interface [2, 14, 96, 156, 157, 186, 199, 200, 231, 364]. Animals and associated burrows generally account for a high percentage of bulk ammonium or total inorganic nitrogen flux (17-90% and 34-80%, respectively), but these values are highly dependent on sediment organic content, nitrification activity, and animal species and density [200, 364]. Flux of ammonium across the sediment-water interface is influenced by depth-dependent reactions. Either uptake by benthic microalgae or processes causing rapid oxidation at the sediment-water interface can inhibit release from the sediment [10, 156]. Excretion of ammonium by macrofauna is important and can account for 20-60% of ammonium flux from the sediments in marine and freshwater habitats [29, 202, 237, 248, 364]. Temperature strongly influences aerobic respiration rates and therefore rates of ammonium flux [253, 254]. Variable uptake and release rates have been reported for nitrate flux across the sediment-water interface [54, 220]; rates may be sensitive to water column concentrations [198].

High values for potential nitrification are associated with tube and burrow structures and fecal pellets produced by macrofauna [29, 157, 202]. Macrobenthos increase total denitrification and are likely to increase the ratio of denitrification to total nitrification [6]. In some cases, macroinfauna stimulate nitrification-denitrification coupling, which results in loss of inorganic nitrogen [106, 200]. Tight coupling of nitrification-denitrification processes in coastal and estuarine sediments has been attributed to macrofaunal burrow construction and sediment pelletization, both of which provide increased juxtaposition of anoxic and oxic microenvironments [176, 200, 282, 310].

Effects on Toxicants

Interactions between toxicants and benthic organisms

are relatively poorly studied [285]. Our overall understanding of how benthic organisms influence the transport and fate of toxicants is generally limited by the fact that toxicants themselves are a diverse group, characterized by complex chemical reactions. Chemical and physical factors affecting input, distribution, and availability of metals and organics in estuaries have been summarized by Olsen et al. [263], Fowler [119], Rice and Whitlow [293, 294], and Sanders and Riedel [308]. As outlined by Rice and Whitlow [293], benthic organisms affect the distribution and behavior of metals in the environment in five ways: (1) by mechanical bioturbation, they alter the quantity of reactive surface area and redistribute particles relative to the redoxcline; (2) they increase the depth of the redoxcline and depress the zone of iron and manganese reduction, such that transition metals associated with iron and manganese or sulfides have different distributions; (3) they influence microbial processes that affect speciation and cycling; (4) by direct ingestion and egestion, they alter the chemical environment; and (5) they act as reservoirs. Similar relationships are likely to apply to relationships between organisms and organic toxicants.

Accumulation of toxicants in sediments is often enhanced by biotic processes, especially suspension feeding [119]. Metals and organic toxicants may be enriched in fecal pellets and burrows of benthic organisms [1, 8, 38, 170]. Bioturbation activities are likely to increase the inventories of particulate-sorbed toxicants as they increase the inventories of organic material [4, 7]. The creation of areas of intense biogeochemical activity by infaunal organisms influences the flux and behavior of metals. Some metals are mobilized under anoxic conditions, especially in areas of intense decomposition along outer walls of some burrows [8]. Following mobilization, iron, manganese, and zinc are concentrated by precipitation on oxidized burrow walls; metals may also be scavenged from the ventilation stream [6, 8].

Bioturbation has been reported to increase the flux of metals from the sediment [31, 161, 173, 194, 296, 297]. Oscillations in pore-water concentrations related to the combined effects of microbial activity and bioturbation may enhance metal flux across the sediment-water interface [161]. Bioturbation also appears to enhance the transport of hydrophobic organic pollutants out of sediments [182]. The presence of macrofauna may enhance microbial degradation of organic toxicants [26]. Changes in faunal density or behavior as a function of toxicity [48, 57] will likely also influence rates of cycling. Distribution of toxicants in sediments is also influenced by bioaccumulation, trophic transfer, biodegradation, and migration, as discussed by Swartz and Lee [330] and Fowler [119].

Understanding organism effects on toxicant distribution and behavior is complicated by the apparent importance of density-dependent processes [293-295]. In general, dense populations of macrofauna reduce the levels of potential toxicants in both the environment and the individuals comprising the population [293]. Interactions between biological and physical processes are also important. For example, rates of metal flux across the interface may be enhanced by coupling physical particle resuspension and bioturbation [309].

Relative importance of biological vs. physical processes. The relative importance of biologically mediated processes in different regions of the estuary varies as a function of the relative intensities of biological and physical processes and non-linear interactions [314, 315]. Studies of animal-sediment interactions conducted as part of the EPA-sponsored Chesapeake Bay Program [252, 280, 342] suggested that, on an areal basis, biotic sediment mixing processes dominate physical processes throughout much of the main-stem Chesapeake Bay bottom. Based on patterns of primary sediment structure, bioturbation intensity, and types of biogenic structures preserved in sediment cores from the mainstem bay and the Virginia tributaries, Schaffner et al. [314, 315] were able to identify distinct patterns in bioturbation processes. They found that time-averaged sediment structuring and mixing in low-salinity areas (i.e., tidal freshwater to low mesohaline) are dominated by physical processes but that bioturbation may be important in areas of very low energy and low sediment accumulation. Bioturbation predominates at high salinities (poly- to euhaline) even where tidal currents are moderately strong and sediments may accumulate, but is less important where physical reworking due to oceanic or wind waves is intense. Within major habitat types, bioturbation intensity was found to vary as a function of salinity, reflecting gradients in faunal characteristics and abundance. Bioturbation is severely limited in Bay areas characterized by anoxia [280, 281, 314].

Trends in other biologically mediated processes are also likely. For example, in low-salinity habitats (tidal freshwater to mesohaline) benthic feeding processes are likely to greatly enhance the deposition of carbon, nutrients, and other substances from the water column. In the lower polyhaline estuary this process may be an important mechanism for benthic-pelagic coupling in areas populated by the polychaete annelid *Chaetopterus variopedatus* and associated epifaunal communities [312]. Bioadvective sediment mixing is likely throughout the estuary, given the distribution of head-down feeders. The depth of sediment column influenced by this process will increase with increasing salinity, as

dominant head-down feeders progress from small oligochaetes and capitellid polychaetes in low-salinity areas to larger maldanid polychaetes in high-salinity areas. Biologically enhanced sediment ventilation will predominate where sediment disturbance and frequency of oxygen stress are low (i.e., intermediate depths or protected shallow areas in low-salinity habitats, and intermediate and greater depths at high salinities), since these areas generally support the greatest densities of large or deep-dwelling infaunal species. Reactions mediated by available oxic-anoxic interface area should be enhanced in areas of high fecal pellet production (especially where suspension and interface feeders exhibit high biomass [see Benthic Energy Flow section]) and high sediment ventilation (areas dominated by *Macoma* spp. and large tube-builders). Seasonality is likely for nearly all biologically-mediated processes because organism feeding, burrowing, and respiring activities are strongly driven by temperature. Sublethal oxygen stress, limited to summer months for the Chesapeake Bay system, will influence the behavior of some benthic organisms and their importance in some processes.

Benthic Energy Flow

Concept and Importance

In general, the annual income and output of energy for an ecosystem are in balance. What we are most interested in are the energetic transformations that occur among and within portions of an ecosystem during the year. Benthic habitats (or the benthic boundary layer [362]) are conspicuous sites for the focusing and transforming of biological energy and are an integral part of ecosystem function. Lindeman [207] was one of the first to consider this overall flow and balance of matter in an energetic sense: if all the components of an ecosystem could be expressed in common units of energy, then the functioning of the system could be more easily understood. Thermodynamics is then the common denominator that defines the manner in which energy can be transformed and describes the ecological usefulness of different forms of energy [27, 356]. Benthic secondary production is part of the larger scheme of energy movement through an ecosystem; it is best thought of as a process within an ecosystem and not as a distinct entity (see systems and network models presented elsewhere [24, 27, 363]).

Productivity of an ecosystem refers to its capacity to produce organic matter either autotrophically (primary production) or heterotrophically (secondary production). By definition secondary production for a period of time is considered to be the total of growth increments of all individuals existing at both the start and the end of the period, the growth of newly born individ-

uals, and the biomass of individuals that do not survive to be part of the final population biomass [358]. Secondary production includes yield (in the sense of harvest); loss to predators and decomposers, which represent mortality; and growth and reproduction, which represent increase in biomass of a population. Typically secondary production calculations do not consider metabolic processes and represent net production.

Measurements of biomass, standing stock, or standing crop, while important in comparing immediately available energy, are quite inadequate for predicting rates of predator cropping, yield, or growth. To understand energy flow and production we need to know the rate of organic matter elaboration (bioenergetics) and the factors (biotic and abiotic) that control the fate of biomass produced (Table 4). Trophic interactions are the main pathways by which energy is moved through an ecosystem [258]. Measuring the production of all trophic levels should elucidate much of dynamics of an ecosystem. The development of our understanding of these dynamics has consisted of clarification of concepts of production and factors controlling food uptake, assimilation, and metabolism.

Estimation of benthic secondary production can be approached from several directions. It can be estimated directly either from patterns of population growth and mortality [71, 100] or from physiology [365]. It can be estimated indirectly from life history characteristics (life span) [303], size at maturity [133] or maximum size [25], or from turnover ratios (production/biomass) [317]. Because of the diversity of benthic species and their complex life histories it is impossible to apply a single method for estimation of secondary production. Difficulties in quantitatively estimating population size and following the life histories of organisms, along with the large amount of time and labor needed to process data, have limited the number of secondary production studies in all estuarine systems.

Organic Matter Budgets

The process of benthic secondary production is driven by two things: the sources of organic matter, which provide energy for the diverse secondary producers; and the fate of biomass produced, which either is consumed by predators or decomposers, or survives as standing stock. In the Chesapeake Bay, as with many estuarine systems (see summaries in Boynton et al. [41]) overall carbon dynamics are dominated by phytoplankton production in both the water column and benthos [124, 320, 345]. Not all of the phytoplankton production directly reaches the benthos. Bacterioplankton form a microbial loop [357] that internally cycles a portion of organic carbon production in the water column [223, 320]. The shallowness of the Chesapeake

Table 4. Factors that affect secondary production (P), with the general direction of effect.

Factor	General effect	Source
Abiotic factors		
Temperature	Higher temperature = higher P	[177, 365]
Salinity	Higher salinity = higher P (to a point)	[242]
Sediment type	Mixed sediments = higher P	[18, 32, 242, 273]
Exposure	Semi-exposed areas = higher P	[273]
Tidal elevation	Lower elevation = higher P	[159]
Depth	Intermediate depths = higher P	[278]
Water quality	Increased nutrients = higher P (to a point)	[140]
Toxics	Excess nutrients or more pollutants = lower P	[21, 105, 269]
Habitat complexity (vegetation)	More structure = higher P	[140]
Biotic factors		
Predation	Tendency to increase P	[60, 140, 150]
Competition	Tendency to increase P*	[265, 266]
	Competition for space reduces P	[273]
Amensalism	Tendency to decrease P*	[287]
Quality of food	Higher labial C = higher P	[208, 336]
Quantity of food	More food = higher P	[206, 365]
Successional stage	Pioneering stage = higher P	[257]
	Equilibrium stage = lower P	[291]
Recruitment success	Higher recruitment = higher P	[189, 242]
Life history		
Life span	Shorter life = higher P	[267, 303]
Age	Younger age = higher P	[250]
Size	Smaller body = higher P	[25, 111, 133]
Voltinism	More generations = higher P	[353]

* Theoretical effect

Bay, however, closely links the water column with the bottom so that production (phytoplankton and microheterotroph) in the water column has a high probability of being transferred to the benthos through turbulent mixing and subsequent suspension- or filter-feeding activities of organisms [63, 65, 77, 152, 223, 249], or through direct sedimentation [82, 143, 320, 327]. A substantial portion of deposited organic carbon may pass through the "small food web" made up of microheterotrophs and meiofauna [204].

In spring, when phytoplankton production and biomass peak [41, 115, 322], about 50% of the carbon transported to the benthos is from phytoplankton [40]. The other 50% of the carbon comes from allochthonous and other autochthonous sources [41, 42]. In summer, there is more consumption of phytoplankton production in the water column by planktivorous fish [24], zooplankton [154], and microheterotrophs [20, 66], with about 20-40% of the carbon delivered to the benthos coming from phytoplankton [41, 320].

Other sources of dissolved and particulate organic carbon to the benthos include autochthonous production from vascular plants, seagrasses, benthic microalgae

and macroalgae, and allochthonous additions from land runoff and sewage [342]. The contribution of these sources to the total carbon budget of the Bay varies seasonally and spatially [28]. Delivery of these carbon sources to the benthos is patchy and reflects proximity to sources [153, 272]. This is particularly true for sewage inputs [87, 342]. In tidal freshwater, oligohaline, and lower mesohaline habitats, particulate organic carbon from upland drainage is a major portion of the total supply; however, quantitatively little is known about its utilization [28, 116]. Processing and cycling of autochthonous and allochthonous production is done almost entirely in the benthos through detritus food webs [23, 61, 228, 244, 335].

Trophic Transfer

Secondary production by benthic organisms is the major pathway by which organic carbon is recycled out of the sediment and eventually out of the Chesapeake Bay system. The majority of fishes that utilize the Chesapeake Bay are involved in organic carbon export. As transients, they are only within the Bay for a portion of their life cycle [19, 209, 323]. Benthic secondary

production may also be a means by which eutrophication is directly regulated through the accumulation of carbon and nutrients in living biomass [184, 249, 259, 345]. Much of the obvious productivity in the Bay, in terms of fishery yield, is directly linked to the secondary production of the benthos through feeding [30, 168, 208, 209, 229, 347].

There is, however, no simple connection between benthic secondary production and fishery species. Not all benthic production is utilized by or available to fishery species [172, 218, 242]. In addition to the stochastic element of predation, which allows for a certain level of prey survival, estuarine organisms avoid predators by quick escape responses, by burrowing below the sediment surface, and to a lesser extent because of large size. Benthic standing stock biomass at any given time then represents surviving prey. We can consider the ratio between annual production and biomass (P/B ratio), the inverse of which constitutes an estimate of the percentage of production that goes into maintaining the standing stock. This is not considered to be energy respired but energy that is needed to produce biomass. While average biomass is a static measure of energy over the course of a year, a portion of the biomass is replaced by new biomass produced during the year. Approximately 20-50% of the total benthic secondary production in estuarine systems is carried over from year to year as standing stock biomass [23, 168]. The percentage biomass carryover tends to be higher in "mature" communities that exhibit successional advanced characteristics [257, 360, 361], because in these communities organisms live longer and are larger, which increases biomass.

A portion of the secondary production is also cycled within the benthos by infaunal predators [13, 58, 240, 348]. Many infaunal species are predacious (particularly important are nemerteans, many polychaetes, and some amphipods and gastropods) and influence the energetics of other infaunal species by preying on adult, juvenile, or larval stages [13, 67, 261]. The production of infaunal predators is then potentially available to epifaunal predators and may actually be more available than nonpredacious infauna, because of the free-burrowing and surface-searching habits associated with a predacious life history [13]. Based on abundance the ratio of predacious to nonpredacious infauna can be as high as 0.25 in sand, 1.38 in mixed sediments, and 0.12 in mud [13]. The importance of infaunal predators to benthic energetics seems to vary with sediment type. In Port Hacking, Australia, Rainer [278] found that in sand infaunal predators accounted for 8-40% of the total benthic secondary production, in mixed sediments they accounted for 9-12%, and in mud there were no infaunal predators. On the basis of the ratio of predacious to

nonpredacious infauna, one would expect similar trends in predator productivity in the Chesapeake Bay [167].

Estimates of the energy transfer (expressed as a ratio of prey consumed/total prey production) needed to sustain epibenthic predators (fish and crabs) are 12-17% of the annual infaunal production for poly-euhaline shallow sandy habitats (Gullmar Fjord, Sweden) [107], about 20% for the Ythan Estuary [23], 4-75% for shallow muddy-sand habitats (Skagerak-Kattegat, Sweden) [242], 30-50% for Georges Bank [326], and 39-67% for low mesohaline muddy habitats (Patuxent River) [168]. These estimates reflect inter-habitat and inter-ecosystem differences in the magnitude of secondary production utilized by predators; these differences are functions of predation pressure, recruitment-settlement success, and food supply [17, 242]. During years when any combination of these factors is favorable, consumption efficiency declines. In the Skagerak-Kattegat, Moller et al. [242] found that when benthic recruitment was above average, consumption efficiency of epibenthic predators declined to 4-10%. During years of average infaunal recruitment, consumption efficiency of epibenthic predators was 51-75%. The average consumption efficiency for six consecutive years in the Skagerak-Kattegat habitats was 43%. While these estimates of utilization span the 20-25% value classically thought to represent energy transfer to fish species [196], the higher values are likely most representative of total energy transferred to all epibenthic predators. In the Chesapeake Bay, if on average 10% of the annual secondary production is consumed within the benthos and 30% survives as standing stock or is not available to epibenthic predators, then 60% is left for potential consumption by epibenthic predators (fish and crabs).

If fisheries landing statistics for the Bay are assumed to be at least representative estimates of minimum fisheries species production, and the production or ecological efficiency (fish prey consumption/fish production ratio) of the fishes is assumed to be about 17-22% [134, 222], then the total consumption of benthic invertebrates can be estimated. About 21,400-27,500 metric tons of organic carbon, or 267,500-343,750 metric tons wet weight, of benthic organisms are needed each year as fish food to sustain demersal fishery yields (Table 5). This magnitude of secondary production is easily supported by benthic organisms and represents a total Bay average of 1.9-2.4 gC·m⁻²·yr⁻¹ or 23.4-30.1 g wet weight·m⁻²·yr⁻¹, with 11,427 km² taken as the area of the Bay and tributaries [72]. In addition to serving as prey several benthic herbivores are directly harvested and contribute about 950 metric tons of organic carbon annually (Table 4). This slightly increases overall benthic productivity to about 2.0-2.5 gC·m⁻²·yr⁻¹.

Table 5. Summarized fisheries landing statistics (metric tons wet weight), with estimates of benthic biomass (metric tons organic carbon) needed to support this yield of major bottom-feeding species, or to support maximum sustainable yield (MSY). Production or ecological efficiency (fish prey consumption/fish production ratio) of the fish and crabs is assumed to be about 17-22 % [134, 222]. Wet weight was converted to organic carbon by: 1 g wet wt = 0.08 g C [353].

Species	Annual recreational average, 1960-1979 [328]	Annual commercial average, 1970-1977 [188]	MSY [340, 341]	Benthic biomass needed for average year	Benthic biomass needed for MSY
Benthic predators					
Blue crab		28,000		10,200 - 13,200	
			29,500		10,700 - 13,900
Maryland	14,200*			5,200 - 6,700	
Virginia	7,100+			2,600 - 3,300	
Total				18,000 - 23,200	
Spot		1,000		360 - 470	
			1,400		500 - 660
	3,500			1,300 - 1,600	
Total				1,660 - 2,070	
Croaker		900		320 - 420	
	1,800			650 - 850	
Total				970 - 1,270	
White perch		500		180 - 230	
			1,400		500 - 660
	1,430			520 - 670	
Total				700 - 900	
Flounder		130		50 - 60	
			1,400		500 - 660
Total				50 - 60	
Grand total commercial and recreational				21,400 - 27,500	
Grand total commercial				11,100 - 14,400	
Grand total recreational				10,300 - 13,100	
Grand total maximum sustainable yield				12,200 - 15,900	
Benthic herbivores*					
Oyster		10,300			
			13,600		
Hard clam		400			
Soft clam		1,200			
			27,300		
Grand total commercial				950	
Grand total maximum sustainable yield				3,270	

* Averaged landings, 1983-1984 (Maryland DNR 1989)

+ Assumed to be 50% of Maryland landings.

* Fisheries yield is assumed to be a minimum estimate of annual production.

Table 6. Predicted macrobenthic secondary productivity of major subtidal benthic habitats in the Chesapeake Bay. (Only areas of the mainstem, James, and Potomac Rivers were used because detailed sediment data were not available for other tributaries.) Data are summarized from [80, 90, 93, 168, 169, 172, 187]. See Table 2 for habitat characteristics.

Major habitat	Area (km ²)	Mean annual productivity (SE), g C/m ² /yr		Total habitat productivity (SE), metric tons C/yr	
Tidal freshwater				18,552	
Mud	455	1.8	(0.5)	819	(228)
Sand	102	145.5	(143.7)*	14,841	(14,657)
Mixed	10	289.2	(246.5)	2,892	(2,465)
Oligohaline				9,853	
Mud	496	14.4	(3.4)	7,142	(1,686)
Sand	59	18.0	(3.6)*	1,062	(212)
Mixed	76	21.7	(5.1)	1,649	(388)
Low mesohaline				10,066	
Mud	393	14.4	(4.1)	5,659	(1,611)
Sand	98	41.0	(0.4)	4,018	(39)
Mixed	36	10.8	(1.0)	389	(36)
High mesohaline				31,266	
Mud	1,525	8.1	(1.3)	12,352	(1,982)
Sand	1,388	8.8	(2.8)	12,214	(3,886)
Mixed	268	25.0	(4.1)	6,700	(1,099)
Polyhaline				69,016	
Mud	509	9.0	(2.5)	4,581	(1,272)
Sand	1,764	32.0	(11.2)	56,448	(19,757)
Mixed	512	15.6	(3.6)	7,987	(1,843)
Euhaline				7,753	
Mud	148	17.2	(11.4)+	2,546	(1,687)
Sand	768	5.7	(0.9)	4,378	(691)
Mixed	29	28.6	(1.1)	829	(32)
Total		8,636		146,506	
Total Chesapeake Bay system	11,427			193,854*	

* Average of mud and mixed sediments production.

+ Average of sand and mixed sediments production.

* Assumes remaining tributaries have similar proportion of habitats.

Total Bay Secondary Production

The strengths of organismal-environmental relationships are such that it is possible to approximate and predict macrobenthic secondary production, and hence potential trophic energy, for any given habitat (Table 6). While the error associated with these broad habitat predictions is large, the predictions do provide a relative starting point for considering the magnitude of energy that flows through the macrobenthos. On a unit area basis the highest amount of macrobenthic organic carbon (potential trophic energy) is in the tidal freshwater habitats (32.7 metric tons C·km²·yr⁻¹) and the least

in the euhaline habitats (8.2 metric tons C·km²·yr⁻¹).

The bulk of the tidal freshwater production is due to the introduced clam, *Corbicula fluminea*, while production of the euhaline habitat is spread more evenly over hundreds of species [80, 90, 93, 172, 187]. Production within the other habitats is usually dominated by several species with many others contributing. An exception is oligohaline habitats, which are also dominated by an introduced clam, *Rangia cuneata*.

On a total area basis, macrobenthic production is highest in polyhaline habitats (69,016 metric tons C/yr) and lowest in the euhaline habitats (7,753 metric tons

C·yr⁻¹). The bulk of the Bay's macrobenthic production, about 70%, occurs in high mesohaline and polyhaline habitats (Table 6). Overall, a total of about 194,000 metric tons of organic carbon is produced each year. This is about seven times the production needed to support the maximum combination of fishery yields (27,500 metric tons C). Maximum sustainable yield models for key commercial species require an even smaller portion of the total macrobenthic production (Table 5). While unharvested commercial and non-commercial demersal fish consume an unknown portion of benthic production, it does not seem that Allen's paradox (more predator production than prey production [27, 212]) is at work in the Chesapeake Bay.

Importance of Habitat

Secondary production is not spatially uniform. The magnitude of productivity (both primary and secondary) is regulated through the complexities of physical and biotic interactions that make up a habitat (Table 2). These interactions affect the basic aspects of an organism and population that are critical to the magnitude of production (growth, individual weight, life span, fecundity, and reproductive success). It is therefore difficult to determine the effect of any one factor on production. For example, the influence of temperature on production is not the same as its influence on metabolism. In metabolism studies there is usually a doubling of metabolism for every 10°C rise in temperature (Q_{10}). With secondary production the situation is more complicated because environmental interactions are superimposed on the fundamentally physiological production process. *Acartia clausi* maturation rates in the Black Sea increased by a factor of 3.7 between 8°C and 21°C, but specific production increased by 4.5 over the same temperature range [365].

Not much detail is known about subtidal softbottom habitat-production relationships within any ecosystem, because of the complexity of interactions and the limited number of production studies. In general, production tends to be highest in spatially complex shallow habitats (such as seagrass beds [120, 247, 273, 278] and oyster reefs [76]) and lowest in spatially uniform deep habitats (such as muddy channels [51] and isolated basins [278]). This habitat-related gradation in production is an important aspect of ecosystem energetics that must be considered in accounting for the trophic importance of the benthos and in assessing the relative value of habitats. Averaged estimates of total macrobenthic secondary production, based on direct calculations of biomass and P/B ratios from a variety of sources [80, 89, 90, 93, 168, 172, 187] by major subtidal unvegetated softbottom habitats in the Chesapeake Bay clearly demonstrate the variation in

production between habitats (Table 6).

The most productive habitats tend to have mixed and sandy sediments, with muddy habitats showing the lowest production. Mean levels (\pm SE) of production in mud, sand, and mixed sediments, reported from a number of Bay studies (Table 6), are 10.7 ± 1.4 , 14.9 ± 3.5 , and 58.9 ± 35.5 gC·m⁻²·yr⁻¹, respectively. Similar patterns of sediment-associated secondary production were found by Pihl [273] at 22 sites along the coast of Sweden and by Bodin et al. [32] in the Bay of Douarnenez, France. The effect of salinity is superimposed on sediment type, with highest production in oligohaline and low-mesohaline muds, low-mesohaline and polyhaline sands, and tidal freshwater and high-mesohaline mixed sediments.

Periodic low dissolved oxygen (hypoxia, <2 ppm O₂) and anoxia (0 ppm O₂) in bottom waters may be a key factor in affecting secondary productivity, depending on the concentration of oxygen and the length of time a habitat is exposed. Habitats that are exposed to extensive periods of anoxia have low annual production [168, 278, 279]. The amount of productivity in these habitats is a function of how quickly benthos can recruit and grow during periods of "normal" oxygen concentrations [251]. In a near-anoxic basin (Port Hacking, Australia) productivity was lower than at a nearby station that experienced only hypoxia, by a factor of almost 16 [278, 279]. Chesapeake Bay areas that experience anoxia do not seem to have as high a factor differentiating productivity of habitats. While some areas known to be affected by anoxia do have lower productivity, the trend is not consistent. The inconsistency is likely due to a combination of duration of exposure to anoxia and rapid recovery of benthos [166]. Secondary production in habitats that are known to experience only hypoxia is of the same magnitude as in habitats that always have normal oxygen concentrations.

Annual P/B or turnover ratios express the number of times that the biomass changes (turns over) for the year studied relative to total biomass produced. When these ratios are calculated for major taxonomic groups by habitat, they can show trends in the overall contribution of taxonomic groups to production. In the Chesapeake Bay, polychaetes are most productive at mesohaline salinities in mud and mixed sediments. Mollusks are most productive in high-mesohaline muds and polyhaline sands. Crustaceans have higher production in mixed-sediment oligohaline and sandy low-mesohaline habitats. Nemerteans are most productive in mesohaline sands. Overall, crustaceans have the highest average annual P/B ratios (5.7 ± 1.3) followed by polychaetes (4.9 ± 1.4), nemerteans (4.3 ± 1.6), and mollusks (2.9 ± 1.6). These trends in P/B ratios reflect the life histories of the organisms (Table 4) [353], and

are similar to those found in other systems. However, the overall magnitude of the P/B ratios for Chesapeake Bay is higher than in other systems [303, 351], reflecting optimal conditions for high secondary production—temperate climate, large pool of organic matter, and, just as important, a large population of predators to keep benthic populations in a high-growth phase.

Comparison with Other Systems and Trends

The weighted average macrobenthic production for the Chesapeake Bay is about $17 \text{ gC}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$, with a total habitat range of about $2\text{--}289 \text{ gC}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$. This average production compares well with data from other major estuarine systems. However, the high range for the Chesapeake Bay is higher than other systems. The majority of estuarine and coastal production studies were conducted in polyhaline and euhaline sandy habitats, with a variety of methods. The range of macrobenthic production in these subtidal unvegetated habitats is about $1\text{--}146 \text{ gC}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$. The range for similar habitats in Chesapeake Bay is about $6\text{--}32 \text{ gC}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$.

Long-term variation in benthic energy flow results from the benthos interacting with temporal (natural and anthropogenic) trends that affect the secondary production process (Table 2). Understanding of these interactions is insufficient to relate their importance to ecosystem function. Few community production studies have addressed more than single-year time periods and those that have done so have reported different trends. Over a four-year period at a muddy site off the Northumberland coast, Buchanan et al. [50] found that benthic population densities more than doubled, but there was essentially no change in benthic production. The total yearly production was differently partitioned between species of differing P/B ratio each year. In shallow sandy habitats of the Skagerrak-Kattegat, Møller et al. [242] found median productivity over a six-year period was about $10.7 \text{ gC}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$, with one-year productivity reaching $145.6 \text{ gC}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$ after a very successful recruitment of benthos. After the Grevelingen estuary was closed from the sea, the productivity of its shallow sand habitats increased about twofold over a four-year period [361]. The increase in productivity was stepped: productivity in the first two years was about $27 \text{ gC}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$, and in the second two years, about $53 \text{ gC}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$. The changes in the Grevelingen could be explained largely by succession patterns for developing ecosystems and the physical dynamics of the habitat.

Summary

In this review we have described the important ties of the benthic subsystem to the entire Chesapeake Bay ecosystem. In general, benthic community structure and the distribution of benthic organisms in the

Chesapeake Bay estuarine system are strongly correlated with salinity and sediment type and are further influenced by patterns of dissolved oxygen and other sources of physical variability. Long-term trends in abundance and biomass can often be explained by coincident changes in the environment.

Most functional groups are well represented throughout the estuary. Suspension-feeding bivalves (e.g., *Corbicula fluminea* and *Macoma* spp.) dominate faunal biomass in low-salinity areas. In higher-salinity areas other suspension-feeding taxa (especially the polychaete *Chaetopterus variopedatus* and tunicate *Molgula manhattensis*) are important biomass contributors, along with suspension-feeding bivalves (e.g., *Ensis directus* and *Mytilus edulis*) and a variety of other feeding types.

Although the mechanisms and rates by which benthic organisms influence their environment are known for some of the organisms (or closely related species) found in the Chesapeake Bay, we still lack a general understanding of basic feeding processes, behavior, and living habits for many benthic species. In addition, the effects of these processes on nutrient flux, remineralization, and especially toxicant transport and fate are very poorly understood. We do not have sufficient information to predict the importance of temporal variability in these processes, although some insight into the net results of physical and biological interactions in the sedimentary environment has been gained from the stratigraphic record. At some times, benthic organisms appear to serve as the primary pumps or conduits for the bidirectional flow of nutrients, pollutants, and other elements across the sediment-water interface or benthic boundary layer. They are also clearly a key link in supporting the high yield of commercial and recreational fisheries (fish and crabs) by converting primary organic matter into forage biomass. Some benthic species (clams and oysters) are also directly harvested.

The benthic environment functions as the major storage compartment for virtually all materials that enter the Bay, from pollutants to primary organic matter. The close association of benthic organisms with the sediment and their life histories make them good integrators of past and current environmental conditions, and thus ideal sensors of pollutant impacts. While local problems from pollution are easily recognized in the benthos, the subtle broad-scale effects on benthic function are unknown. This lack of understanding of system-level effects is due to the inherent variability of estuarine systems and our lack of knowledge about factors that control benthic function. Such knowledge is essential if we are ever to distinguish the effects of pollutants in a holistic context or to enhance fishery yields.

Management Implications for the Benthos and Benthic Resources

The 1987 Chesapeake Bay Agreement has underscored the need for sound management of all Bay resources [59]. If this task is to be accomplished, the entire Bay must be perceived as a single functioning ecosystem. Effective management of living resources and their habitats requires knowledge of what affects these resources, both positively and negatively, and how they function within the ecosystem. Because the benthic subsystem (the entire benthic boundary layer) is the final focus for pollutants and is integrally connected to every other Bay subsystem, reliable detection and interpretation of habitat conditions requires adequate understanding of benthic function.

Any management plan for Chesapeake Bay living resources needs to start with the recognition of the Bay as a single ecosystem. From this perspective a detailed plan or model can be produced that will allow for identification of key flows and controls. Preliminary attempts to model the Chesapeake Bay confirm how complex and interconnected the system is [24, 131, 363]. This level of complexity requires that managers more clearly focus their attention, consider ecosystem-level implications, and set limits to management goals. This response would improve communication with scientists, who are asked to provide data necessary to determine whether management strategies have been or would be successful.

This review of benthic function points to a lack of data within the benthic compartment coupling benthos to other Bay components in terms of energy flow and other material cycling. This lack of data is not specific to the Chesapeake Bay but extends to many estuarine and coastal ecosystems. Although comprehensive data on specific aspects of benthic coupling (usually energy flow) are available for relatively simple systems [305, 332, 363], major problems in the management of living resources arise because of lack of data.

The more important research needs for Chesapeake Bay that must be addressed in the near future to allow sound management of living resources are:

- Are there organism-environment relationships that can predict energetic characteristics (sources and pathways of flow) within and among habitats?
- Does the spatial arrangement of various benthic habitats (i.e., bare sand, mud, marshes, seagrass beds, oyster bars) play an important role in benthic function?
- How closely coupled are the primary organic carbon sources (autochthonous and allochthonous) and secondary production?
- How closely coupled are fishery yields and benthic production?
- What magnitude of control is exerted by the full range of benthic organisms (microfauna to fish) on diagenetic processes and nutrient dynamics?
- What are the relationships between toxicants and benthic organisms that control uptake and fate?
- What is the relative importance of biological vs. physical processes in controlling diagenetic processes and nutrient and toxicant dynamics?
- How do long-term changes in climate affect benthic function, and are there any long-term periods?
- What role do episodic events (e.g., large storms, dredge material disposal) play in restructuring benthic habitats?

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Role of Best Management Practices in Restoring the Health of the Chesapeake Bay: Assessments of Effectiveness

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Introduction

Nonpoint source (NPS) pollution is a significant source of water quality problems in the Chesapeake Bay. Pollution control programs are attempting to reduce NPS pollutant generation with best management practices (BMPs), which are intended to minimize the negative environmental consequences of land-use activities while maintaining the productivity of the land.

The goal of this article is to discuss the role of BMPs in NPS pollution control and to describe methods of BMP assessment. Specific objectives are:

- Discuss the factors and/or processes that affect BMP effectiveness.
- Describe BMPs commonly used in the Chesapeake Bay region.
- Discuss why BMP effectiveness varies from site to site.
- Describe the principal methods—monitoring and modeling—used for BMP assessment.
- Identify research needed to improve the effective utilization of BMPs for NPS pollution control.

Factors Influencing BMP Effectiveness

Pollutant loss from land is a function of the concentration of the pollutant in its carrier (water or sediment) and the mass of the carrier. Consequently, pollutant losses can be reduced by decreasing either concentrations or carrier mass. Unfortunately, reducing carrier mass often increases concentrations, and pollutant reductions may not be as great as expected [2]. The principal transport mechanism of a pollutant is often a function of the degree to which the pollutant adsorbs to soil. Adsorption is a function of the chemical properties of the soil and chemical pollutant. Soils high in clay and organic matter have high adsorptive

capacity, while sandy soils have low adsorptive capacity. If the adsorptive capacity of a soil for a chemical can be assessed, the likely transport mode of the chemical and a BMP appropriate for its control can be determined [2].

Of equal importance with pollutant loss from the land surface is pollutant transport from source areas to downslope waterbodies. During this process, pollutant transport is reduced by deposition, sorption, and chemical transformations. Coarser sediment is trapped by terraces, filter strips, riparian zones, ponds, and other BMPs that reduce flow velocities and sediment transport capacity. Although BMPs may reduce sediment yield, reductions in adsorbed chemical losses often are much lower, because sediment-bound chemicals are preferentially adsorbed to fine sediment, which is less susceptible to deposition than coarser sediment.

The second mechanism reducing downslope pollutant transport is sorption. Dissolved chemicals in surface or subsurface flow can be removed by adsorption to the land surface, vegetation, or suspended sediment.

The third process affecting pollutant losses during transport is chemical and biological transformation. This process is usually not significant in surface runoff in upland areas because transport time from the source area to upland streams is too short. Significant transformations may occur, however, if the transport time from upland streams to receiving waters is long. Pollutants transported via subsurface drainage, which have a relatively long transport time, are susceptible to transformations both in the root zone and in groundwater. Mineralization of organic nitrogen and denitrification of nitrate in riparian zones [63] and degradation of soluble pesticides are important transformations.

Description and Comparison of Common BMPs

Currently, only a few BMPs are both technically feasible and socially and economically acceptable [2]. In selecting BMPs to control a particular pollutant, it is useful to determine how the pollutant is being transported to receiving waters. A BMP that interferes with this process can then be identified. With this in mind, BMPs can be classified as reducing pollutant carrier mass, pollutant concentrations, or pollutant delivery from the source area to receiving waters.

Important BMPs that reduce carrier mass include conservation tillage, contouring, terraces, and cover crops. These BMPs are erosion control practices and consequently are effective in reducing sediment-bound pollutants. Conservation tillage and other BMPs that reduce sediment loss often enrich eroded sediment with soil fines. Since fines have a higher adsorptive capacity, they transport proportionally more chemicals than coarser particles, and sediment loss from conservation tillage systems is often enriched with sediment-bound chemicals. These practices also increase infiltration and may increase groundwater contamination.

BMPs intended to reduce pollutant concentrations involve the method, rate, timing, and formulation of chemical applications. The concentration of a chemical at the land surface (and therefore the amount available for loss in surface runoff) may be reduced by removing the chemical, incorporating the chemical, reducing application rates, or applying the chemical when it is less susceptible to loss. To control subsurface losses, the mass of chemical in the soil must be reduced by lower rates of application, better timing, or use of formulations less susceptible to leaching. Examples of BMPs intended to reduce pollutant concentrations in surface and subsurface flow include integrated pest management, improved manure and fertilizer management, street sweeping, and waste oil collection and recycling.

Structural practices that reduce pollutant delivery include terraces, filter strips, ponds, sediment detention basins, and infiltration trenches. These BMPs improve surface water quality by increasing sedimentation and infiltration, but they may increase groundwater contamination.

Whatever the BMP, it is imperative to remember that BMP effectiveness is extremely site-specific. If a BMP is effective at one site, there is no guarantee that it will be equally effective at another site. The effectiveness of BMPs is controlled by characteristics of the site's soil, topography, cropping, and climatology.

Since no two sites have identical characteristics, BMP effectiveness will vary from site to site. Consequently, the system of BMPs selected to control NPS pollution problems at any particular site must be designed to address the unique characteristics of that site. The following sections contain brief descriptions of BMPs commonly used for NPS pollution control in the Chesapeake Bay region. An assessment of the relative effectiveness of these BMPs for NPS pollution control is given in Table 1. Additional information on BMPs used for NPS pollution control in the Chesapeake Bay basin has been reviewed by the Chesapeake Bay Liaison Office [114].

BMPs that Reduce Carrier Mass

Conservation tillage. Conservation tillage is the fastest-growing practice in the history of U.S. agriculture [22]. Currently over half the cropland in Virginia and Maryland is in conservation tillage, because in this region it results in higher crop yields and lower production costs than conventional tillage. It is also presumed to be one of the best available BMPs for controlling NPS pollution. Conservation tillage is defined as any tillage or planting system that leaves at least 30% of the soil surface covered with crop residue after planting. Major types of conservation tillage include no-till, ridge-till, strip-till, and mulch-till. Conservation tillage affects pollutant transport in surface runoff by decreasing soil erosion and surface runoff, increasing infiltration, and reducing incorporation of agricultural chemicals.

Surface residues associated with conservation tillage reduce soil erosion by protecting the soil from flowing water and raindrop splash. If raindrops do not hit the soil surface, soil particles are not detached as easily from the soil mass and erosion is greatly reduced. Erosion has been found to be approximately halved for every 9-16% increase in residue cover [5]. This finding implies that conservation tillage should reduce erosion by 75-90% compared with conventional tillage. Conservation tillage systems also increase infiltration and reduce average annual runoff volumes by about 25% compared with conventional tillage [2], but these effects are highly variable. Reduced runoff would be expected to reduce the transport of dissolved and sediment-bound chemicals. Unfortunately, concentrations of dissolved and sediment-bound chemicals in surface runoff often increase with conservation tillage and may offset a large part of the reduction in runoff volume.

There is considerable uncertainty as to the effect of conservation tillage on the movement of pesticides, nitrates, and other dissolved chemicals to groundwater. One of the chief unknowns is the effect of

Table 1. Best management practices and structural practices for nonpoint source pollution control.

Practice	Type	Effectiveness in reducing losses		
		Sediment	Nutrients	Pesticides
Agricultural				
Conservation tillage*	BMP	Excellent	Mixed	Mixed
Winter cover crops	BMP	Good	Good	Fair
Critical area planting	BMP	Excellent	Fair	Fair
Vegetative filter strip*	BMP	Mixed	Mixed	Unknown
Contouring*	BMP	Good	Good	Good
Terracing*	BMP	Excellent	Good	Good
Strip cropping	BMP	Good	Good	Good
Ponds*	Structural	Good	Fair	Fair
Pasture management	BMP	Good	Good	-
Nutrient management	BMP	-	Excellent	-
Integrated pest management	BMP	-	-	Excellent
Urban				
Street sweeping	BMP	Good	Good	-
Infiltration trenches	Structural	Good	Good	Good
Porous pavement*	Structural	Good	Good	-
Sediment barriers	Structural	Good	-	-
Sediment detention basin*	Structural	Good	Fair	-
Mulching and temporary cover	BMP	Good	-	-

* Practices that can potentially increase groundwater quality problems.

preferential flow through soil macropores on chemical movement through the vadose zone. Preferential flow allows surface water to flow through macropores in the soil and bypass the biologically active region of the root zone responsible for most biological and chemical transformations. With preferential flow, dissolved chemicals move 2-20 times faster than would be predicted by conventional Darcy flow theory [12]. In addition, preferential flow may allow supposedly immobile pollutants to reach groundwater. Research indicates that conservation tillage greatly increases preferential flow because without tillage, macropores resulting from earthworm tunnels and decayed root channels are not destroyed [115, 6].

Most studies have found that pesticide loss is less with conservation tillage than with conventional tillage. Generally, 15-40% more pesticides are applied with conservation than conventional tillage [57, 23], but this increase is usually offset by reduced sediment yields and runoff volumes. Pesticide loss reductions are smaller than sediment and runoff volume reductions, however, because the pesticides concentrate on the soil surface with conservation

tillage, resulting in higher concentrations of pesticides in sediment and surface runoff.

The most significant factors affecting nutrient transport with conservation tillage are the placement, timing, and rates of fertilizer applications. The primary goal of conservation tillage is to minimize disturbance of surface residues. From an agronomic and water quality viewpoint, however, it is desirable to incorporate or place fertilizers close to plant roots and away from the soil surface where they are subject to loss in surface runoff. Unfortunately, these two goals may be in conflict because typical fertilizer incorporation practices also incorporate residue.

When fertilizers are broadcast and not incorporated, they concentrate near the soil surface and are susceptible to surface loss. Conservation tillage has been reported to increase nutrient concentrations at the soil surface by 600% [2]. In a five-year study comparing conventional and no-till corn-soybean rotations, concentrations of phosphorus in the upper 5 cm of the soil profile were 67% higher with no-till [41]. Similar results are expected with nitrogen except that increased infiltration with conservation

tillage may leach nitrate lower into the soil profile. Concentration of nutrients near the soil surface with conservation tillage has two consequences: higher concentrations of nutrients in eroded sediment, and higher concentrations of dissolved nutrients in surface runoff. For example, in the corn-soybean rotation study discussed above, sediment-associated phosphorus loss would decrease with no-till only if the 67% increase in soil phosphorus concentrations were offset by a 67% reduction in soil loss. Dissolved nutrient concentrations in runoff are directly proportional to nutrient levels at the soil surface [4]. Thus losses of dissolved nutrients with conservation tillage will not decrease relative to conventional tillage unless the increased concentrations are offset by larger reductions in runoff volume.

Vegetative filter strips. Vegetative filter strips (VFSs) are bands of planted or indigenous vegetation, situated between pollutant source areas and receiving waters, which remove sediment and other pollutants from surface runoff. VFSs are a type of structural practice designed to remove pollutants from field effluents, rather than reduce pollutant generation within the field. The major pollutant removal mechanisms associated with VFSs involve changes in flow hydraulics that enhance infiltration, deposition, filtration, adsorption, and absorption of pollutants. Currently, there are no standards or accepted methods for VFS design, and many VFSs are installed in areas where they are ineffective for pollutant reduction [35].

Research at the University of Kentucky on VFSs found that sediment-trapping efficiencies were high as long as flow was shallow and the VFSs were not inundated with sediment, but trapping efficiency decreased dramatically at higher runoff rates that inundated the media [7, 56, 57]. Several short-term experimental studies have reported on the effectiveness of VFSs in reducing concentrations of sediment, nutrient, bacteria, and organics in agricultural runoff [29-31, 34, 35, 40, 78, 87, 122, 123]. These studies have reported that over the short term and with shallow flow, VFSs are very effective for sediment and sediment-bound pollutant removal, with trapping efficiencies exceeding 50%. Dissolved pollutants such as nitrate and orthophosphorus, however, are not removed as effectively, and several studies reported that effluents from VFS often had higher concentrations of dissolved nutrients than the influent [34, 78]. This result was attributed to the release of previously trapped sediment-bound nutrients that were converted to soluble forms. VFS plots with concentrated flow, similar to that expected under field conditions, were reported to be 20-50% less effective than shallow-

flow plots for pollutant removal [34].

VFS performance in the field was evaluated by observing VFSs on 18 farms in Virginia [29]. Performance was reported to fall into two categories depending upon site topography. In hilly areas, VFSs were judged to be ineffective for pollutant removal because most drainage concentrated in natural drainageways within the fields before reaching the VFSs. Flow across these VFSs during the larger runoff-producing storms (the most significant in terms of water quality) was primarily concentrated, and the VFSs were locally inundated and ineffective. This assessment was confirmed by the fact that little sediment accumulated in most of the VFSs observed.

In flatter regions, VFSs appeared to be more effective. Slopes were more uniform, and larger portions of stormwater runoff entered the VFSs as shallow flow. This observation was supported by significant sediment accumulations in many of the coastal-plain VFSs. Several 1- to 3-year-old VFSs had trapped so much sediment that they were higher than the fields they were protecting. In these cases, runoff flowed parallel to the VFS until it reached a low point, where it crossed the VFS as concentrated flow. These VFSs needed maintenance to regain their sediment-trapping ability, but the landowners, with no economic incentive, were unlikely to perform the maintenance.

Recently, researchers have begun investigating the effectiveness of riparian zones in removing pollutants from cropland runoff. Riparian zones have been reported to trap 84-90% of the sediment [18] and 50% of the phosphorus [17] leaving cultivated fields. Several models have been developed for VFS design and evaluation. The Kentucky filter strip model is an event-based model for designing VFSs with respect to sediment removal [7, 58, 59]. The model was evaluated using data from experimental field plots in Mississippi for multiple storm events, and predictions were in good agreement with observed sediment discharge values [60]. Several researchers [46, 118] have evaluated VFS effectiveness for erosion control using the CREAMS model [66]. CREAMS, like the Kentucky model, does not consider the long-term effectiveness of VFSs because it cannot account for sediment accumulations within a VFS. Consequently, CREAMS would be expected to overestimate long-term sediment trapping. The model also cannot account for concentrated flow effects.

An event-based VFS model, GRAPH [72], was developed to simulate nutrient transport in VFSs. The model is based on the GRASSF version of the Kentucky VFS model [56]. GRAPH considers the effects of advection, adsorption/desorption processes,

and changes in sediment size distribution on phosphorus transport. GRAPH was verified with data from VFS field plots and model predictions, and observed phosphorus transport in VFSs compared favorably.

Grassed waterways. Grassed waterways are structural practices designed to prevent gully formation by transporting surface runoff and sediment downslope with as little accumulation of sediment as possible. If gully erosion is a significant problem, grassed waterways are an effective practice for their control. During smaller runoff-producing storms, flow in grassed waterways is shallow and they may function like VFSs. No quantitative studies have been reported, however, which investigated the effectiveness of grassed waterways for pollutant reduction [84]. One study reported that grassed waterways that trapped sediment quickly became inundated with sediment and were rendered ineffective [71].

Contouring. Contour farming has been found to reduce soil loss 0-85% on an average annual basis, depending on the field slope, the height of ridge/furrow systems, and local rainfall characteristics [105]. Reductions in sediment loss and runoff volumes of $\geq 50\%$ are typical in areas with mild slopes [2]. Contouring is most effective on permeable soils with mild slopes [84]. With conventional tillage or ridge-till contouring, surface runoff collects in furrows and there is little runoff and soil loss from the field unless the furrows overtop. If the furrows overtop and fail, water from one furrow flows downslope to the next furrow, causing it to fail; and the process repeating down the slope causes severe rill or gully erosion. Soil loss under these circumstances is as severe as that which would occur without contouring. Because of this problem, contouring may not be suitable for every site.

Terraces. Terraces are very effective in reducing NPS pollution in surface runoff; level terraces were reported to reduce soil loss by 94-95%, nutrient losses by 56-92%, and runoff by 73-88% [2]. These reductions are achieved by storing water temporarily and allowing sediment to deposit and water to infiltrate. Consequently, terraces would be expected to enhance the potential for the movement of dissolved pollutants to groundwater. Under some circumstances, graded terraces can improve field drainage and reduce pollutant loadings to groundwater. As with contouring, severe erosion can result if terraces are overtopped. Properly designed and maintained terraces will last for 10 to 20 years. Terraces are expensive to install, and more cost-effective pollutant reduction can often be accomplished with conservation tillage and contouring [84].

Close-grown crops and winter cover crops. Close-grown crops such as small grains and grasses are very effective in reducing erosion, with $\geq 70\%$ reductions in soil loss and 11-96% reductions in runoff volume compared with conventional tillage [2]. Winter cover crops are effective for erosion and runoff control because they provide a protective canopy and root system to hold the soil in place over the winter. By spring, when fields in conventional tillage are bare and most major pollution-producing storms occur, cover crops are well developed and the soil is protected from erosion. Cover crops also reduce surface runoff by minimizing surface sealing and by increasing infiltration and the resistance to overland flow. Non-legume cover crops have been reported to decrease nitrate loadings to groundwater due to plant uptake over the winter and subsequent release to crops during the late spring and summer [110]. Legume cover crops were also reported to tie up soil nitrogen over the winter and provide nitrogen for subsequent crops, reducing fertilizer requirements [84].

Protective cover. Protective vegetative cover and mulching of bare soil in urban areas is similar to conservation tillage in agricultural areas. Covering exposed soil surfaces with vegetation, straw, gravel, wood chips, and other available mulches protects the soil surface from raindrop impact, increases soil roughness and infiltration, and reduces soil erosion. Vegetative cover and mulching are more effective for NPS pollution control in urban areas than other urban BMPs because they reduce losses of fine sediment, clay and silt particles, which most other urban BMPs cannot control. Straw mulch on 12% and 15% slopes applied at rates of 5 and 10 metric tons/ha, respectively, reduced erosion by 75-80% [88]. Mulching is only a temporary solution, however, and every effort should be made to convert bare soil to a permanent protective cover as soon as possible during the construction process for permanent erosion control.

BMPs that Reduce Pollutant Concentrations

Nutrient and pesticide management. Improved management of agricultural chemicals and animal wastes can greatly reduce NPS pollution. Often, the formulation of a chemical may influence its susceptibility to loss. For example, some formulations are highly soluble but others are only slightly soluble. Use of a less soluble form would be desirable if groundwater contamination were a problem, whereas a more soluble form might be desirable if erosion were high and surface runoff were the pollutant problem. For example, in runoff events immediately after surface application of fertilizers, urea (a highly

soluble form of nitrogen) was found to be less susceptible to loss in surface runoff than ammonium nitrate [79]. Presumably, urea's higher solubility and lower adsorptivity allowed it to infiltrate into the soil profile where it was less susceptible to loss in surface runoff. The ammonium nitrate, however, adsorbed to soil particles, could not move down into the soil, and was lost with eroded sediment. If runoff had occurred several days after fertilizer application, rather than immediately, these results could have been reversed. The ammonium nitrate would probably have been converted to soluble nitrate, a form that would have leached down into the soil profile, whereas the urea might have been converted to ammonium, a form less susceptible to leaching. These differences demonstrate the importance of the timing of chemical applications with respect to runoff events.

Nitrification inhibitors can be used to block the conversion of ammonia (which is plant-available but not readily leachable) to nitrate, which is highly mobile. In cooler regions of the U.S. these inhibitors have been found to be effective in reducing ammonia volatilization as well as nitrate loadings to groundwater after applications of nitrogen fertilizer and animal wastes [49]. Nitrification inhibitors have not been reported to be very effective in the temperate Chesapeake Bay region. Slow-release nitrogen fertilizers were reported to reduce nitrogen losses by as much as 40-85% [83].

Losses of dissolved nutrients and pesticides are highly correlated with the timing of chemical applications. Losses increase as the time between application and the first runoff-producing event decreases. The best way to minimize these losses is to apply only the amount of chemical needed by the crop, close to the time it is needed by the crop. For example, fall fertilization is undesirable if crops will not be planted until spring because the fertilizer may be lost before the crops can use it. Similarly, manure applications are not recommended in the winter because the nutrients will not be utilized until spring. For nitrogen, split applications are highly recommended because nitrogen is very mobile. With this method, nutrients meeting a portion of the crop's needs are applied at planting and the rest are applied later in the growing season when a better estimate of crop nutrient needs can be made. This practice can greatly reduce fertilizer applications while maintaining crop yields. Split application of nitrogen can reduce nitrogen losses by up to 30% in comparison with single applications [83]. Another alternative is to plant a cover crop in the fall to scavenge residual nitrate in the root zone over the winter. The cover

crop is then incorporated in the spring to release nitrogen for the spring crop. This reduces nitrate leaching in the fall and winter when most nitrate leaching occurs in temperate climates [12]. Soil testing and application of the minimum amount of fertilizer recommended to meet plant needs is the most important BMP for nutrient management [83].

Placement of chemicals is also important. Furrow-band applications were found to reduce losses of carbofuran in surface runoff by about 50% compared with surface broadcasting [14]. Herbicide application under corn residue had little effect on herbicide losses in surface runoff compared with application to the residue itself [4]. Chemical incorporation can also reduce chemical losses. Subsurface application reduced picloram concentrations in surface runoff by 72% compared with surface application [13]. Herbicide application followed by disking reduced losses by approximately two-thirds compared with surface application.

Similar trends have been found with nutrients. Disking reduced losses of soluble fertilizers by 50-75% compared with surface application, and plow-down incorporation reduced losses almost to those of unfertilized control plots [2]. Similarly, point injection of fertilizer to a depth of 5 cm did not result in additional fertilizer loss compared with unfertilized control plots. A study of band incorporation of phosphorus fertilizer found that there were no significant differences in dissolved phosphorus concentrations in runoff from conventional, no-till, and conservation tillage plots [81]. Soluble phosphorus losses were found to be correlated with runoff volume reductions. The method of chemical incorporation is also affected by the type of crop residue. Shallow tillage with knives or disks may be acceptable with corn residue, but a single disking for ammonia application with soybean residue may significantly reduce surface cover and increase soil erosion [5]. If subsurface application equipment is unavailable or unacceptable from an erosion standpoint, dribble banding of liquid and solid fertilizers is recommended to minimize nutrient losses in surface runoff [80]. Animal wastes should either be applied via liquid injection or be incorporated immediately after surface application to minimize volatilization and losses in surface runoff [82].

Morrison [80] gives an excellent review of machinery for improved fertilizer application with conservation tillage. Slot injectors for liquid and dry fertilizers are described which greatly increase the efficiency of fertilizer use and minimize losses in surface runoff. Coulter/nozzle, v-wheel-and-sweep, high-pressure nozzle slot injectors, and injectors on

paraplow blades are described along with an innovative spoked-wheel point injector [3].

Integrated pest management. Integrated pest management (IPM) can be defined as the use of management practices for pest control that eliminate unnecessary applications of pesticides and replace pesticide use with biological and cultural controls whenever possible. A basic premise of IPM from a farmer's viewpoint is that pesticides should never be applied unless the cost of not applying pesticides (in terms of reduced crop yield and quality) exceeds the cost of applying pesticides. This is a radical departure from conventional farming, where pesticides are often applied whether they are needed or not as a routine prophylactic practice. Important IPM practices include: biological controls, regulatory procedures (training of applicators), cultural methods, pest-resistant crops and livestock, scouting by IPM specialists, and genetic manipulation of crops to increase their resistance or of pests to reduce their viability. Significant reductions in pesticide usage have been achieved in most IPM programs without decreasing agricultural profitability. The most significant factor retarding adoption of IPM is lack of education of potential users. An overview of IPM principles and practices is given by the Council of Agricultural Science and Technology [19].

In one study, 80% of New York apple producers were reported to use some IPM practices [68]. Producers that employed comprehensive IPM practices used 30%, 47%, and 10% less insecticides, miticides, and fungicides, respectively, and saved an average of \$95.80/ha-yr over an 11-year period without significantly affecting fruit quality. The study also found that IPM users were younger and better educated than non-users. Another New York study found that increased use of IPM with onions led to a 32% reduction in pesticide use [42]. In Indonesia, IPM techniques reduced pesticide usage by 60% and increased rice yields by 25% [12]; apparently, the amount of pesticides required to control the pesticide-resistant pests was so high that the pesticides had a toxic effect on the rice crop. Evidence that education can reduce pesticide use is also supported by data from Nebraska suggesting that more-educated or better-trained pesticide users have fewer application errors than less-educated users [53]. Additional information on BMPs for controlling pesticide losses is reported by the National Water Quality Evaluation Project [85].

Street-cleaning. Street cleaning and collection of leaves and grass clippings can have a significant

impact on storm-water quality [44]. Street cleaning once or twice a day was required to reduce losses of sediment and heavy metals [93]. Typical street-sweeping programs clean streets only once or twice per month with an efficiency of approximately 50%, but efficiency decreases as average particle size decreases. Street-sweeping alone is usually ineffective in reducing urban pollutant losses to acceptable levels [88]. A significant portion of the organic matter in storm-water runoff in urban areas is due to leaves and grass clippings deposited on street surfaces. A rigorous program of grass and leaf removal can significantly reduce these losses [88].

Chemical use control and hazardous waste

collection. Regulatory or voluntary programs such as waste oil recycling and elimination of road salts can reduce urban water quality problems. Use of road salts is estimated to result in \$3 billion in environmental damage in the United States each year [44]. Replacement of road salts with hydrophobic de-icers has been found to reduce these problems [69]. One liter of waste oil can contaminate a million liters of clean water. Problems with waste oil and other hazardous wastes can be reduced if convenient collection and recycling centers are provided.

Structural Practices

Sediment traps and barriers. Sediment traps are small temporary structures (straw bales, stone or pre-fabricated check dams, silt fences, excavated ditches, and small pits) used during construction projects to trap coarse sediment particles by reducing flow velocities and promoting deposition. Their effectiveness for removing dissolved pollutants, fine sediment, and pollutants associated with fine sediment is minimal [88]. Sediment traps and barriers require frequent maintenance to maintain their effectiveness. They should be inspected after every runoff event, and accumulated sediment should be removed whenever it exceeds 50% of the device's sediment storage capacity.

Sediment detention basins and ponds. Sediment detention basins are large structures designed to reduce peak runoff rates and to remove a certain percentage of the sediment in stormwater runoff. There are three general types of detention ponds: dry ponds, wet ponds, and extended wet ponds [70]. Dry ponds are not very important in reducing pollutant losses because of their short detention times. Their primary purpose is to reduce peak runoff rates, but they will also trap coarse sediments if cleaned out regularly. Wet ponds have a permanent pool level,

and during runoff events additional water is temporarily stored to reduce peak runoff rates. Wet ponds have longer detention times than dry ponds and are more effective in removing sediment and sediment-bound pollutants. Sediment removal efficiencies of wet ponds are typically 50-75% [51]. Detention ponds are most effective in watersheds with coarse soils and least effective in watersheds with fine soils. Extended wet ponds have sufficiently long detention times to allow all but the most colloidal sediment to settle. These ponds are very effective for the removal of sediment and sediment-bound pollutants, but they have limited impact on dissolved pollutants unless they have very long detention times that allow sufficient time for biological assimilation. Sediment detention basins may need to be cleaned as often as twice per year to maintain their efficiency.

Infiltration trenches. Infiltration trenches are subsurface trenches filled with coarse aggregate that are used to store surface runoff until it can infiltrate into the soil. They are particularly effective in intercepting the first flush of runoff from impervious areas. This interception is important because the first flush usually has the highest pollutant concentrations and may be responsible for most of the pollutant load from a storm. The surfaces of infiltration trenches are usually covered with grass, stone, or other porous material to allow water to enter the trench rapidly. All sediments and sediment-bound pollutants entering infiltration trenches are trapped, but dissolved pollutants may be transported to groundwater. Infiltration trenches require a high degree of maintenance because the walls of the trenches tend to clog with fine sediment [88].

Porous pavement. Porous pavement is low-density permeable asphalt with a thick aggregate reservoir base course for water storage [70]. Precipitation entering porous pavement can be stored until it infiltrates into the soil, or it can be released slowly to reduce peak runoff rates. An investigation of porous pavement in New York found that it reduced peak runoff rates by as much as 83% [44]. Porous pavement has frequently been reported to have problems with clogging. This problem indicates that porous pavement is effective in trapping sediment and would also be effective for removal of sediment-bound pollutants until it clogs. Monthly cleaning with conventional vacuum trucks is recommended to minimize clogging. If porous pavement systems are designed for infiltration of all of the influent, they could become a source of groundwater contamination.

Cost-Effectiveness of BMPs

The cost-effectiveness of different BMPs for reducing sediment loadings to Lake Springfield in Illinois was found to vary greatly [102]. The costs of conservation tillage, structural BMPs (sediment basins, terraces, grass waterways, etc.), and lake dredging were \$1.05, \$6.73, and \$5.16 per metric ton of sediment; thus conservation tillage was by far the most economic practice. The structural practices were not cost-effective because they were not effective during the infrequent larger storms that were responsible for most of the sediment loadings to the lake. The Rural Clean Water Program (RCWP) at Rock Creek had similar findings. Structural practices were found to be very effective for sediment reduction (55% reduction), but their initial capital costs were high and their annual maintenance costs were estimated to range from \$22-37/ha (for sediment retention structures) to \$49-\$119/ha (for improved irrigation structures) [75]. Because of these high costs and fears that the structures would not be maintained after cost-sharing ended, the project started encouraging conservation tillage and discovered that sediment losses were reduced up to 90%. In addition, productivity generally increased with conservation tillage. Farmers switching to conservation tillage saved an average of \$82/ha/yr. Thus conservation tillage was a better practice than conventional tillage agronomically as well as environmentally.

Methods Used to Evaluate BMPs

With the increased use of BMPs for NPS pollution control, there is an obvious need for methods of assessing BMP effectiveness. Proper management of any system requires reasonable estimates of the expected impacts of alternatives being considered. This is particularly true with NPS pollution control, as planners face the often conflicting goals of minimizing pollution and maximizing economic return to the landowner. An effective plan can be developed only from good data. Estimates of BMP effectiveness are essential for (1) selecting the most appropriate BMP for a particular problem and site; (2) estimating the benefits of BMP implementation; (3) ranking BMP alternatives in terms of cost-effectiveness; and (4) determining an optimum BMP program based upon program objectives. Program objectives include both pollution control objectives, such as meeting desired water quality goals, and financial objectives, such as maximizing the cost-effectiveness of cost-share monies, minimizing costs to the landowner, maintaining long-term productivity, or maximizing return

to the landowner while meeting water quality objectives [107].

Approaches

Two approaches have been used to evaluate BMP effectiveness: monitoring and modeling. Water quality monitoring can be defined as any effort to obtain an understanding of the physical, chemical, and biological characteristics of water via statistical sampling. Monitoring is the most direct way to assess BMP effectiveness, but it has several significant drawbacks. First, there is tremendous variability in the soil, land use, topography, and weather factors influencing BMP effectiveness. Consequently, a BMP may be highly effective for pollutant control at one site but ineffective at another site. Second, monitoring is time-consuming and expensive. If the particular BMPs selected for a site are found to be ineffective, considerable time and resources will have been wasted and it may take many more years to develop an effective program.

The second approach to BMP assessment uses mathematical models. Physically-based models attempt to describe the physical, chemical, and biological processes affecting the natural system being modeled. This allows consideration of site-specific soil, land use, topographic, and weather factors that are critical to BMP effectiveness. Models of another class, empirical models, are less suitable for BMP evaluation because they do not describe the physical processes controlling the system being simulated. They are simply statistically derived equations and should not be used beyond the range of data used in their development. Consequently they are not good for simulating the site-specific nature of BMPs [2].

Monitoring Approaches

The level of monitoring required to evaluate BMP effectiveness should be a function of the objectives of the monitoring program. If the only objective is to determine whether BMP implementation is effective in reducing pollutant losses from a watershed, then a weekly grab-sampling program at the watershed outlet, with streamflow monitoring, may be adequate. At the other extreme, if the data are to be used for model development and evaluation, then monitoring may have to include: weather parameters; land use changes; chemical applications, rates, and timing; chemical levels in soils; cultivation, planting, and harvesting methods and dates; and streamflow. In addition, edge-of-field monitoring may be necessary to assess the effectiveness of specific BMPs in reducing in-field pollutant losses, and monitoring of ephemeral drainageways may be necessary to assess pollutant transport

from fields to streams. A major problem with edge-of-field monitoring is that it measures what is leaving the field and not what is reaching downslope waterbodies. Unless losses between the field and receiving waters are accounted for, pollutant loadings and concentrations may be greatly overestimated.

The study objectives will also determine the size of the area to be studied and whether natural or simulated rainfall will be used. Monitoring studies on smaller areas (<0.5 ha) and short-term studies can often benefit from the use of simulated rainfall that allows control of the timing, amount, and intensity of the rain. Using simulated rainfall can reduce both the time and cost of data collection by allowing BMPs to be evaluated during critical crop-growth stages and chemical application periods.

One of the earliest and most popular approaches for evaluating BMPs involves the use of paired plots or watersheds. With this approach, two or more areas with similar size, soils, topography, and weather are treated the same and monitored during a pre-BMP phase for two years or more. An empirical statement can then be made about the behaviors of the watersheds relative to each other. Then, after BMPs are implemented in one or more (but not all) of the watersheds, monitored watershed responses are compared with those predicted by the regression equations to determine if the BMPs have had an effect.

Replicated plots and watersheds are another traditional method for evaluating BMPs. With this method, each treatment is replicated two or more times and all the plots are monitored simultaneously. Replicated plots are usually small because it is often difficult to find large areas with the desired uniform soils, slope, and cropping history and because replicated plot studies require the analysis of very large numbers of samples.

Another approach involves the use of upstream and downstream monitoring. With this method, a pair of monitoring stations are established upstream and downstream of a subwatershed in which BMPs have been installed. The upstream and downstream concentrations are then compared to assess the impact of BMP implementation. For this method to succeed, it is critical that the flow volume from the drainage area above the BMP be similar to the flow volume of the BMP drainage area. Otherwise, the upstream flow will dilute the discharge from the BMP area so much that there will be no detectable differences in the concentrations.

Spooner et al. [107, 109] give an excellent overview of monitoring system designs for evaluating the effectiveness of agricultural NPS pollution control programs. They point out that monitoring systems

often lack a sound experimental design, with the result that data collection efforts and results are inconclusive. Four approaches for design and analysis of monitoring systems are presented: (1) before-and-after time trend analysis uncorrected for meteorological variables, (2) before-and-after time trend analysis corrected for meteorological variables, (3) monitoring upstream and downstream of a NPS implementation area, and (4) paired watershed designs. They suggest appropriate hypotheses and statistical techniques for each experimental design. The paired watershed design is recommended as the best monitoring method for documenting water quality improvements in the shortest period of time. If the quantity of NPS pollution prior to implementing BMPs is of interest, the upstream-downstream approach is recommended. The before-and-after design is reported to be the easiest design to follow, but it is not recommended because it may be difficult to attribute water quality improvements to BMP installation. The time required to detect the desired change will be a function of the noise in the system and how large a change is actually being made.

A generalized framework for integrating agricultural NPS water quality problem identification, data collection, data management, and project assessment has been developed [67]. It suggests logical and efficient ways to combine data evaluation and assessment with problem definition and data collection. This report also provides comprehensive lists of the types of data that must be collected to address different NPS pollution problems.

Another study developed a five-step conceptual model to optimize the design of NPS pollution control programs and monitoring systems [94]: (1) definition of monitoring objectives to guide data collection efforts; (2) choice of level of detail required for monitoring system alternatives; (3) watershed analysis to identify critical areas for BMP implementation and monitoring; (4) development of a monitoring program to detect and verify statistically the sources of NPS pollution; and (5) prioritization of monitoring tasks with reference to program objectives. Whitfield [119] also provides a good review of monitoring project designs.

Guidelines for measuring water quality impacts of NPS control projects were presented by a USEPA-USDA interagency task force [96]. The procedures presented are minimum recommended techniques for detecting water quality changes due to BMP implementation in waters impaired by NPS pollution. Specific evaluation alternatives are presented for streams, lakes, and groundwater. The report notes that improvements in water quality impaired by

eutrophication and biological degradation may not be measurable within 3- to 5-year project periods because natural systems are highly variable and are slow to show response to subtle changes.

A symposium on the design of water quality information systems was held in Fort Collins, Colorado in 1989 to assess the quality and utility of water quality data and to emphasize a systems approach to water quality monitoring design [106]. A key point made at the conference was that collecting data is easy compared with analyzing it and that monitoring projects must have quantifiable information goals identified at their outset that will result in quantitative information. Proposed statistical methods need to be evaluated in light of expected data limitations such as serial correlations, seasonality, missing data, non-detections, and multiple observations in one sampling period. The sampling frequencies, variables monitored, and monitoring locations can then be determined from the information goals and supporting statistical analysis. The conference also stressed that data collection is worthless without good information objectives, decision technology, quality assurance, and data management. Quality assurance was reported to be the most critical component of data collection.

The same general techniques described for monitoring of surface water quality also apply to groundwater monitoring. Groundwater monitoring is more complex, however, because it is more difficult to measure and allow for covariates such as flow rates, recharge area, dilution from the recharge area outside the study area, loadings to groundwater, and the quality of upstream or upgradient water. It is also difficult to track the movement of pollutants through the soil profile to groundwater. In addition, groundwater is often very slow to respond to reductions in pollutant loadings. Depending on the aquifer and the pollutant of concern, it may take anywhere from a month to centuries for groundwater quality to improve. On the other hand, because pollutant concentrations in groundwater are much less variable than those in surface waters, smaller changes in pollutant concentrations are necessary to demonstrate significant impacts of BMPs on groundwater quality. Numerous publications give more details on groundwater and groundwater monitoring [20, 43, 65, 113].

Rural Clean Water Program Monitoring Projects

The 10-year Rural Clean Water Program (RCWP) was initiated in 1980 to demonstrate how agricultural NPS pollution could be controlled. The program is administered by the USDA and USEPA. Each project had

four principal activities: cost-sharing of BMPs, technical assistance to farmers, educational programs for farmers, and water quality monitoring. Cost-sharing of up to 75% of the cost of practices with a maximum of \$50,000 per farm was allowed. BMP cost and effectiveness in improving water quality was found to be highly site-specific. For example, manure storage resulted in substantial water quality improvements in the Vermont RCWP where manure could be used to replace commercial fertilizer, but there were few water quality benefits from manure storage in the Pennsylvania RCWP because so much manure was available that even if no commercial fertilizer was applied, there was still gross over-application of nutrients [92].

A review of the first seven years of the RCWP reported that targeting of BMPs was "the key to NPS pollution control" and essential for cost-effective water quality improvements [76]. Considerations used to target resources to critical areas contributing the most to water quality impairment usually included erosion rates, proximity to streams, and livestock densities. Provision of technical services such as nutrient analysis of manure, plant tissue, and soil (Pennsylvania and Virginia RCWPs) was very effective in inducing farmers to improve nutrient management practices. Several projects demonstrated that conservation tillage and management of nutrients and pesticides are the most cost-effective BMPs for NPS pollution control.

There appears to be a direct correlation between watershed size and the time required to detect improvements in water quality [76]. Water resource systems that are managed closely or that flush quickly show the fastest improvements. Consequently, irrigation ditches and small streams would be expected to respond much more quickly than rivers, lakes, and estuaries. In the first six to seven years of the RCWP, none of the RCWPs achieved significant

water quality improvements in lakes [76]. Estimated response times required to achieve measurable improvements in water quality after BMP implementation are given in Table 2.

The RCWP monitoring project budgets for the five comprehensive monitoring projects have ranged from \$700,000 to \$2,000,000 over a 10-year period [76]. Most of these projects are not evaluating specific BMPs but rather evaluate overall watershed response. Of the first four RCWPs to show significant water quality improvement, three were not comprehensive monitoring projects but had well-designed grab-sampling programs. This result demonstrates that appropriate grab sampling can document water quality improvements in 5-10 years. To detect these changes, however, will require a 20-40% change in geometric mean pollutant concentrations because of the random variability in aquatic systems. It was suggested that information gained from the RCWP may not establish cause-and-effect relationships between BMPs and water quality, nor can the success of one project necessarily indicate the likely success of another project because of the site-specific nature of BMPs [76].

The five RCWP projects with comprehensive monitoring programs were studied to assess their cost-effectiveness with respect to water quality improvements [121]. The results of these five projects suggested the following considerations for improving the efficiency of future agricultural NPS programs:

- Target funds and projects only towards water-bodies with water quality problems causing substantial economic damage.
- The relative costs and effectiveness of BMPs can greatly influence project effectiveness. Use BMPs appropriate for the area, and target them to the areas responsible for the water quality problems.
- The effectiveness of a particular BMP can vary

Table 2. Estimated number of years required to achieve and detect water quality improvements.

Water resource system	Response time (years)	Time for significant response (years)
Irrigation canal	0 - 1	3 - 8
Stream	1 - 5	5 - 13
Estuary	0 - 5	5 - 12
Lake	2 - 10	6 - 14
Groundwater aquifer	Unknown	Unknown

Adapted from [76].

greatly from one site to the next because of differences in soils, cropping, topography, and other factors.

- Some projects may not be economically justifiable unless on-site benefits such as long-term productivity are considered in addition to off-site water quality benefits.

Below are additional details on three of the RCWPs. Details are sketchy on most of the RCWP projects because little has been published.

St. Albans RCWP was initiated in 1981 in the 13,500-ha St. Albans Bay watershed in Vermont [16]. Water quality impairment in the bay was attributed to excessive nutrient loss from cropland and dairies. An extensive monitoring network was established to measure the impacts of BMP implementation. There were four types of monitoring: (1) grab sampling at bay stations, (2) instream sampling of four tributaries and the St. Albans wastewater treatment plant with automatic samplers, (3) a pair of small watersheds with automatic samplers and stage recorders to evaluate improved manure management practices, and (4) grab sampling at four other stream locations at 20-day intervals to monitor additional subwatersheds. Flow was measured continuously at each station, and three recording raingages monitored precipitation. Four years of monitoring was insufficient to evaluate water quality trends in the bay and its tributaries. Only in the paired watersheds was it possible to document significant improvements in water quality.

Rock Creek RCWP was initiated in 1980 to investigate the effectiveness of structural BMPs such as sediment detention basins, vegetative filter strips, and improved irrigation structures in reducing water quality impairment caused by irrigation return flows [86]. The project's monitoring program was successful in demonstrating improvements in water quality. The BMPs resulted in significant reductions in sediment, total phosphorus, orthophosphorus, and total Kjeldahl nitrogen. Reductions in inorganic nitrogen losses were not conclusively demonstrated. This RCWP was somewhat unusual in that it was conducted in an arid region (20-25 cm of annual precipitation) where almost all runoff was produced by surface irrigation.

The original purpose of the Rock Creek RCWP was to show that water quality would respond to structural improvements to the irrigation system: lined channels, gated pipe, vegetative filter strips, sediment basins, and buried pipe runoff control systems. These structural BMPs were successful in the short term, but

the project officials concluded that they would be unsuccessful in the long run because these practices require a high degree of maintenance, which would not continue after cost-sharing ended. Economic analysis suggested that it would be more effective to use less costly field management practices such as irrigation water management and conservation tillage, which are designed to keep soil and nutrients in the field.

Nansemond River-Chuckatuck Creek RCWP. Results of the first five years of the 10-year Nansemond-Chuckatuck RCWP in southeast Virginia are described by Fisher [45]. Over \$2 million in cost-share funds have been contracted, and significant reductions in agriculturally generated pollutants from cropland and livestock have been reported. Agricultural BMPs being implemented include: permanent vegetative cover; animal waste control systems; diversions; grazing land protection systems; waterway systems; cropland protective systems; critical area plantings (vegetative filter strips); conservation tillage; stream protection; sediment detention, erosion, and water control structures; fertilizer management; and nutrient management. The project reports that it has achieved substantial water quality improvements, but states that these improvements may be masked by declining water quality due to rapid urbanization in the area.

Statistical Design Criteria for Effective Monitoring

Numerous researchers have investigated the role of statistical design in the development and assessment of monitoring programs [11, 61, 95, 108, 109, 116, 117]. One study reviewed methods for determining statistically significant changes in water quality due to NPS control programs and presented a method for determining the magnitude of water quality changes needed to document significant differences over time [108]. Adjustments were presented that could reduce the estimate of variability and decrease the water quality change required for statistical significance. These adjustments included accounting for changes in precipitation and runoff, changing sampling frequency, increasing the monitoring period, and performing statistical trend analysis. For the data from the Rock Creek (Idaho) RCWP, a 30-60% reduction in the unadjusted geometric mean concentration was required for documenting a significant change in water quality. However, if adjustments were made to reduce the estimate of variability, the required change fell to 20-40%. In a similar study, it was found that NPS phosphorus loadings to Lake Erie would have to be reduced by about 35% before the changes could be detected. If the loads were adjusted

for changes in discharge, the required change would drop to 20%. At present rates of adoption of conservation tillage and associated decreases in phosphorus loadings, a 35% change would require more than 25 years to detect, but the 20% change would require only eight years [95]. Obviously, in this case a monitoring design that incorporates flow measurements would be desirable. Monitoring at the Taylor Creek-Nubbin Slough RCWP was reviewed to determine what levels of detected changes in total and orthophosphorus concentrations were required to indicate real changes in water quality. The minimum level (after adjustment for covariates such as precipitation and seasonality) ranged from 10% to 59% over nine years [109].

Limitations of Monitoring

Data obtained in many monitoring studies are often so compromised in one or more respects that they are of little value [27]. Routine water quality monitoring programs have generated mountains of data in the past, but little data analysis has been done and reported. Monitoring often has a poorly defined purpose and results in a "data-rich but information-poor" syndrome [117]. To overcome these problems, study designers must address not only the what, where, when, and how of sampling, but also the why. These are the critical questions of quality assurance. Thus it is essential that a quality assurance plan be prepared for all monitoring projects to insure that data supply the needed information [33]. The major problem with monitoring is that it is so demanding that often it becomes a goal in itself and its original purpose is forgotten [27].

Modeling Approaches

Nonpoint source models used to evaluate the effectiveness of BMPs for NPS pollution control range in complexity from simple empirical models like the USLE [120] to complex physically-based watershed-scale models like ANSWERS [9]. Some models have also been developed for evaluating specific BMPs such as filter strips [56, 72]. NPS models generally concentrate on the generation of pollutants and their transport across the land surface to waterways or through the soil profile to groundwater. Water quality models, on the other hand, are more concerned with the transport and fate of pollutants during concentrated flow in streams, rivers, lakes, and other large waterbodies. Because of these differences, NPS models are often called loading models, while water quality models are called receiving water models. Only a few models, such as HSPF [64] and SWAM

[25] (see following section), combine both loading and receiving water aspects into one overall model.

Types of Models

NPS pollution models can generally be classified as either screening or hydrologic assessment models [89]. Screening models are usually relatively simple and are intended to identify problem areas within large drainage basins or to make preliminary qualitative evaluations of BMP alternatives. One of the most important uses of screening models is the identification of potentially critical sources of NPS pollution within watersheds. Numerous studies have indicated that, for many watersheds, a few critical areas are responsible for a disproportionate amount of the pollutant yield. Consequently, concentration of pollution control activities in these critical areas can maximize the improvement in downstream water quality achievable with limited funds [111]. Because of the simple nature of screening models, their predictions are expected to be accurate only within an order of magnitude or so. Examples of NPS screening models include the USLE, GAMES [77], and VirGIS [101] (see following section).

Hydrologic assessment models are much more complex than screening models and are intended for assessing current conditions or alternative management scenarios. The predictions of good hydrologic assessment models should be within a factor of 2 of observed values if model parameters are measured on site or if the model is calibrated. Otherwise, predictions should be within an order of magnitude of observed values. Hydrologic assessment models can also be subdivided into field-scale and watershed-scale models. Field-scale models attempt to describe hydrologic processes within a single field or land resource unit with uniform soils, cropping, topography, and weather. They do not attempt to describe pollutant transport and fate beyond the boundaries of the field. Examples of field-scale hydrologic models include: CREAMS [66]; CNS and CPM [54]; GLEAMS [73]; NTRM [100]; and PRZM [15] (see following section).

Watershed-scale hydrologic assessment models attempt to describe pollutant transport in the field and between fields and receiving waters. Some are event-oriented (single storm predictions) while others are continuous simulation models. Several can be used to identify critical sources of NPS pollution in watersheds and to target BMPs. Most can be used to evaluate the cost-effectiveness of alternative BMP implementation scenarios. Important NPS watershed-scale hydrologic models include: AGNPS [125], ANSWERS [9], ARM [38], HSPF [64], NPS [36],

STORM [112], SWAM [25], SWMM [62], and WEPP [50] (see following section).

Available NPS Models

Available NPS hydrologic assessment models are shown in Table 3, with indications as to what parameters are simulated.

Chemicals Runoff and Erosion from Agricultural Management Systems (CREAMS) [66] is a physically-based, field-scale model developed for comparing pollutant loads from alternate management practices. CREAMS does not require parameter calibration with observed data, but calibration improves its accuracy [74]. One of the most attractive features of CREAMS is its comprehensive user's manual, which documents the model's development and facilitates parameter selection. The model has been tested in many areas of the world and is the state-of-the-art field-scale model for BMP assessment. Although it is intended for use as a continuous simulation model, it can also be used as an event-oriented model. The model estimates runoff volume, peak runoff, infiltration, evapotranspiration, soil moisture, percolation, sediment yield, particle-size distribution of eroded sediment, and losses of dissolved and adsorbed nitrogen, phosphorus, and pesticides in surface runoff and percolate. The primary limitation of the model is that as a field-scale model, it is limited to areas with uniform soils and cropping and does not consider pollutant transport to receiving waters. CREAMS has also been found to underestimate runoff volumes. It is more accurate in representing average annual runoff volumes than daily or monthly runoff volumes [103]. It does not work well in cold climates [16].

Cornell Nutrient Simulation (CNS) and Cornell Pesticide Simulation (CPS) Models [54] are field-scale models developed to predict nutrient and pesticide losses from agricultural fields. Both models use the SCS curve number equations to predict surface runoff (CPS also uses the Green-Ampt [52] infiltration equation) and a modification of the Universal Soil Loss Equation (USLE) to predict soil erosion [90]. Some problems were encountered in comparing the CNS model simulations and observed data in Georgia and New York [89]. The CPS pesticides model considers pesticide degradation, volatilization, percolation through the root zone, and loss in surface runoff.

Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) Model [73] uses the basic foundation of CREAMS and adds compo-

nents to simulate the movement of water and chemicals within the crop root zone. At present, GLEAMS simulates subsurface movement only of pesticides, but a nitrogen model is being developed. GLEAMS does not consider movement between the root zone and the water table. GLEAMS divides the root zone into 3 to 12 layers, and pesticide transport within the root zone is by advection. Diffusion and volatilization are not considered. Pesticide application can be partitioned between the soil and foliage and can be incorporated to any depth. Pesticide degradation rates can vary by soil zone. GLEAMS can simultaneously model the transport of 10 chemicals and their degradation products, and multiple applications of pesticides are allowed each year. In an independent evaluation, GLEAMS was found to predict peak pesticide concentrations within an order of magnitude and often within a factor of 2 or 3 of observed values [104].

Nitrogen-Tillage-Residue Management (NTRM) Model [98, 99, 100] is a field-scale, continuous simulation model developed to evaluate existing and proposed soil management practices with respect to erosion, soil fertility, tillage, crop yield, crop residues, and irrigation. The model simulates carbon and nitrogen transformations including nitrification, denitrification, mineralization, immobilization, urea hydrolysis, and non-symbiotic nitrogen fixation as a function of soil moisture and temperature, with use of zero- and first-order process equations. The model is more sophisticated than other field-scale models in describing the physical processes affecting transport and transformations, but computing requirements are high.

Pesticide Root Zone Model (PRZM) [15] is a field-scale, continuous simulation model developed by the USEPA to simulate the effects of agricultural management practices on pesticide fate and transport. Runoff is predicted by the SCS curve number equation; soil loss, by a modification of the USLE. The model simulates the vadose zone from the soil surface to groundwater. The vadose zone is divided into several layers with various properties and degradation rates. Pesticide processes considered include advective and dispersive flux, sorption, degradation in the soil, and plant uptake. Volatilization is not considered. Applications can be partitioned between foliage and the soil surface. Surface applications can be incorporated by tillage. The model permits only one application of pesticides per year. In a study in south Georgia, PRZM was used as a screening model (no calibration), and predictions were similar to those of GLEAMS. Most predictions of peak pesticide concentrations were within a factor of 2 or 3 of

observed values, and almost all were within an order of magnitude [104].

Agricultural Nonpoint Source Pollution (AGNPS)

Model [125] is an event-based, watershed-scale model developed to simulate runoff, sediment, chemical oxygen demand, and nutrient transport in surface runoff from ungaged agricultural watersheds. Subsurface transport processes are not considered at present, but a groundwater loading version of the model is planned. Nutrients considered include nitrogen and phosphorus. The model operates on a square cell basis, which facilitates data base creation. All model inputs are defined on the cell level. Pollutants are routed from the source cell through intervening cells

to the watershed outlet. Model output may be viewed at any cell, a capability that allows identification of critical source areas and evaluation of targeting alternatives. Runoff volume is simulated with use of the SCS curve number method and the peak runoff rate equation used in CREAMS. Erosion and sediment transport are calculated with modified forms of the USLE and Bagnold's stream power equation [1]. Nutrient yield in the sediment-bound phase is calculated as a function of the nutrient content of the field soil, the sediment yield, and an enrichment ratio that is a function of soil texture and sediment yield. Soluble nutrient loss is a simple function of the soil nutrient level and an extraction coefficient. The model considers only losses of total nitrogen and phosphorus

Table 3. Principal models used for nonpoint source pollution assessment.

Model*	Time scale	Watershed characterization	Groundwater loading	Parameters simulated*
Field-scale				
CREAMS	Continuous	-	Yes	S,N,P
CNS	Continuous	-	Yes	S,N
CPM	Continuous	-	Yes	S,P
GLEAMS	Continuous	-	Yes	S,N,P
NTRM	Continuous	-	Yes	S,P
PRZM	Continuous	-	Yes	S,P
WEPP	Continuous	-	Yes	S
Watershed-scale				
AGNPS	Event	Distributed	No	S,N,B,COD
ANSWERS	Event	Distributed	No	S,N
ARM	Continuous	Lumped	Yes	S,N,P
HSPF	Continuous	Lumped	Yes	S,N,P,B,DO,O
NPS	Continuous	Lumped	Yes	S,N,DO
STORM	Continuous	Lumped	No	S,N,BOD
SWAM	Continuous	Distributed	Yes	S,N,P
SWMM	Continuous	Distributed	Yes	S,N,B,COD,O
WEPP	Continuous	Distributed	No	S

*CREAMS = Chemicals Runoff and Erosion from Agricultural Management Systems; CNS = Cornell Nutrient Simulation; CPM = Cornell Pesticide Simulation; GLEAMS = Groundwater Loading Effects of Agricultural Management System; NTRM = Nitrogen-Tillage-Residue Management; PRZM = Pesticide Root Zone Model; WEPP = Water Erosion Prediction Project Model; AGNPS = Agricultural Nonpoint Source Pollution; ANSWERS = Areal Nonpoint Source Watershed Environment Response Simulation; ARM = Agricultural Runoff Management; HSPF = Hydrologic Simulation Program-Fortran; NPS = Nonpoint Source Pollution Loading; STORM = Storage, Treatment, and Overflow Model; SWAM = Small Watershed Model; SWMM = Storm Water Management Model; WEPP = Water Erosion Prediction Project Model.

* S = sediment, N = nutrients, P = pesticides, COD = chemical oxygen demand, B = bacteria, DO = dissolved oxygen, BOD = biochemical oxygen demand, O = others.

and does not consider nutrient transformations. The model also allows for inputs from feedlots, sewage treatment plants, and other point sources. Data file creation is time-consuming, since 22 parameters must be specified for each cell; however, a model user's guide is available [124] and the CREAMS handbook [66] is a useful source of parameter values. The model has been tested for runoff estimations on 20 watersheds in the north central United States. Peak runoff rates were approximately 1.6% less than observed values with a coefficient of determination of 0.81. Sediment yield predictions compared well with observed sediment yields from two watersheds in Iowa and Nebraska [125]. The nutrient model has not been adequately tested yet, but limited testing on Minnesota watersheds indicated that the model provides realistic estimations of nutrient concentrations in runoff [125]. The model's effectiveness in predicting total runoff volume was not reported, so it is difficult to assess the ability of the model to predict total yields.

Areal Nonpoint Source Watershed Environment Response Simulation (ANSWERS) Model [9] is an event-oriented, watershed-scale model developed to describe the impact of existing and proposed agricultural management practices on water quality in ungaged watersheds. Recent versions of ANSWERS include an extended sediment detachment/transport model allowing prediction of sediment yield and concentrations for mixed particle-size distributions [28], a phosphorus transport model [111], and a nitrogen transport model [32]. ANSWERS subdivides the watershed into a uniform grid of square cells. Land use, slopes, soils, and management practices are assumed to be uniform within each cell. Typical cell sizes range from 0.4 to 4 ha, with smaller cells providing more accurate simulations. Eight to 10 parameter values must be provided for each cell. The extended sediment model uses a modification of Yalin's equation similar to that in CREAMS [47].

A non-equilibrium desorption equation is used to account for the desorption of soluble phosphorus from the soil surface to surface runoff. Sediment-bound phosphorus is modeled as a function of the specific surface area of the eroded sediment. The equilibrium between soluble and sediment-bound phosphorus is modeled with a Langmuir isotherm [111]. The nitrogen transport version of the model simulates nitrogen transformations of applied fertilizer and soil nitrogen between the time of fertilizer applications and runoff events. Soluble nitrogen transport in surface runoff is modeled with the assumption of complete mixing of the soil surface and surface runoff. Sediment-bound nitrogen is modeled as a

function of the clay content of transported sediment. The nutrient transport versions of ANSWERS received preliminary verification using water quality data collected from rainfall simulator plot studies. Model predictions of dissolved orthophosphorus, nitrate, ammonia, and sediment-bound total nitrogen and phosphorus were generally within a factor of 3 of observed values. The phosphorus transport version has been used to demonstrate how targeting of BMPs can be used to increase the cost-effectiveness of cost-share monies [111]. Targeting of cost-share funds to critical areas in the Nomini Creek, Virginia, watershed (10% of the cropland area) was shown to cut the cost of reducing phosphorus losses by approximately 80%.

In an independent evaluation of ANSWERS, the Wisconsin Department of Planning found that ANSWERS was inaccurate and impractical for their land use planning purposes [8]. A review of the Department's report, however, shows that ANSWERS was not designed for the Department's intended application. This mismatch demonstrates a common modeling problem; attempts to make the model fit the situation rather than finding a model suitable for the situation are seldom successful.

Agricultural Runoff Management (ARM) Model [38] is a continuous simulation model developed to estimate runoff, sediment, nutrient, and pesticide loadings to surface waters from surface and subsurface flow. The model is an overland flow version of the Stanford Watershed Model [21]. Small watersheds of 200 to 500 ha in size can be simulated. Land use, cropping, and management practices are assumed to be uniform throughout the watershed, so it is not possible to identify critical source areas or evaluate targeting strategies. ARM is poorly suited to NPS planning on ungaged watersheds because it requires long-term historical runoff and water quality records for calibration. Data requirements are extensive and data are difficult to obtain in most cases because many parameters have little physical significance. Calibration, testing, and verification are suggested for each application of the model [24].

Hydrologic Simulation Program-Fortran (HSPF) Model [64] is an improved version of the ARM model and is probably the most extensively used NPS model [26]. HSPF is a continuous, watershed-scale model developed to simulate the movement of dissolved oxygen, organic matter, temperature, pesticides, nutrients, salts, bacteria, sediment, pH, and plankton from the land surface through streams, reservoirs, and groundwater. Both point and NPS inputs can be simulated. This capability allows comparisons

between the relative magnitudes of point and NPS pollution during water quality planning. HSPF is better than ARM at simulating watershed diversity because it allows the watershed to be subdivided into land segments with relatively uniform meteorologic characteristics, soils, crops, and management practices. Runoff from the land segments drains to channel reaches with uniform hydrologic properties and to larger receiving waters if they exist. It is difficult to include many land segments in the model because increasing their numbers greatly increases requirements for calibration and input data. HSPF is a large model that requires several years of historical hydrologic records for calibration, extensive data bases, and large computer resources. A publication is available to help with selecting model parameters for simulating agricultural BMPs [39]. The calibration required to run the model and its lumped parameter approach make it difficult to evaluate changing watershed conditions caused by BMP implementation. The model is calibrated to existing conditions, and modifying parameters for future conditions is difficult [88]. HSPF has received much less independent verification than other NPS models and should be used with caution. Formal training is recommended before attempting to use HSPF [24].

Nonpoint Source Pollution Loading (NPS) Model [36] is an earlier version of the ARM model developed to estimate nutrient losses in surface runoff from urban and agricultural areas. Like ARM and HSPF, NPS requires historical hydrologic records for calibration. In addition to nutrients, the model simulates runoff, sediment, water temperature, and dissolved oxygen. Data requirements are not as extensive as those of ARM, and the model can simulate runoff from up to five conceptual land segments in a single run. The model was reported to adequately simulate total nitrogen and phosphorus loadings and concentrations from agricultural watersheds where nutrients were primarily sediment-bound [37]. Where runoff is low and pollutants are primarily in dissolved forms, the model does not predict well because it does not consider dissolved pollutant transport. All pollutant losses other than sediment are estimated by multiplying sediment losses by a potency factor. Thus the model would probably be poor in evaluating BMPs like conservation tillage that greatly reduce sediment yields but often increase pollutant concentrations. Because of these shortcomings, the model has limited value for evaluating BMPs [24]. A modified version of NPS was used as the NPS pollution loading submodel in the Chesapeake Bay Program's Chesapeake Basin Model [55]. The model

received a limited amount of calibration on 11 test watersheds within the basin. Few details on the calibration and testing of the model have been reported, and it is unclear whether the model was appropriate or well-calibrated.

Storage, Treatment, and Overflow Model (STORM) [112] is either an event-based or continuous simulation model developed for urban storm-water management. The program is intended for simulating the quantity and quality of runoff from small, primarily urban, watersheds, but rural areas can also be simulated. Modeled parameters include total and volatile suspended solids, biochemical oxygen demand, total nitrogen, and orthophosphorus. As with NPS, the water quality parameters are assumed to be related to suspended solids. The model does not route surface runoff, and because of the questionable use of the rational method for estimating runoff, runoff volumes can be highly inaccurate even with calibration [88]. Soil loss from pervious areas is estimated by using the USLE and user-supplied delivery ratios. Wash-off from impervious areas is estimated by using a form of the Sartor wash-off equation [97]. The model considers storage and treatment of storm-water and can consider urban BMPs such as sediment detention basins and ponds (using a trap efficiency parameter) and street sweeping.

Small Watershed Model (SWAM) [25] is a continuous simulation, watershed-scale model currently being developed to simulate the response of small agricultural watersheds (less than 2500 ha) to land management practices. SWAM is essentially a watershed version of CREAMS and can represent fields, channel segments, ponds or small reservoirs, and groundwater flow. Like AGNPS and ANSWERS, SWAM uses a grid of square cells to represent overland flow source areas. Runoff from the source areas is routed to downslope receiving waters. The model has an internal weather generator to simulate weather if historical time series are not available. Water quality processes considered include nutrient cycling and transport; decay of plant residue; pesticide application, degradation, and transport; and soil heat flux. Management practices that can be simulated include irrigation, tillage, fertilizer and pesticide application, terraces, buffer strips, animal grazing, crop rotations, and contour farming [25]. Validation studies that have been reported deal only with individual components of the model, and it is unclear whether the entire model has ever been assembled. The developers of SWAM report that validation and testing of the complete model is just beginning [25].

Storm Water Management Model (SWMM) [62] is the most sophisticated and widely used model developed for urban storm-water management. SWMM is a continuous simulation, watershed-scale (5 to 2000 ha) model that simulates runoff quantity and quality from pervious and impervious areas, erosion, scour, sediment transport, dry weather flow and pollutant routing in sewers, storm-water storage and treatment, and receiving water quality. SWMM divides a watershed into small homogeneous subcatchments (a maximum of 200) and routes runoff from these catchments to the drainage system. SWMM simulates wash-off from impervious surfaces in the same manner as STORM. Loadings of pollutants other than sediment are generated from sediment yields using user-supplied potency factors. The program is large and requires extensive input data, but calibration is not required. A comprehensive user's manual is available [62], and the USEPA supports regular user's conferences and workshops on the model. The model's use for simulating NPS pollution processes and problems was reported to be limited [88].

Water Erosion Prediction Project Model (WEPP) [48, 50] is one of the most significant developments that will affect NPS pollution control efforts in the future. This model is being developed by the U.S. Department of Agriculture to replace the USLE. The USLE was developed over 20 years ago and has been an integral part of virtually all NPS and erosion-control planning efforts. The WEPP model is intended to correct some of the deficiencies of the USLE, such as poor estimation of erosion with contouring and steep slopes. In addition, WEPP will be able to predict soil losses for individual storms using readily available input data. WEPP will be computer-based and will contain soil, crop, and weather databases to facilitate model use. The model will simulate the effects of climate, soils, topography, and cropping-management conditions on erosion, deposition, and sediment transport. The model will consist of three basic versions: (1) representative overland flow profile, (2) watershed, and (3) grid. The initial version of the WEPP technology was delivered during the summer of 1989 and the final model version intended for public use is expected to be available in 1992.

Geographic Information Systems (GISs) are data bases/models that can be used for a wide variety of land-use planning purposes. For NPS pollution control planning, they have been shown to be very effective in targeting and prioritizing NPS pollution control resources. Virginia's program for agricultural

NPS pollution control is built around the Virginia Geographic Information System (VirGIS) [101]. Other states and the U.S. Environmental Protection Agency in the Chesapeake Bay region have developed similar GISs for their NPS programs. VirGIS contains seven layers of base data: soils, land use, surface water, elevations, watershed boundaries, political boundaries, and locations of livestock facilities. The base data are stored in a cellular digital map form. Cell sizes range from 1/9 to 1 ha depending upon the data type. Additional data layers derived from the base data layers include cell slope, slope length, length-slope factor, erodibility factor, soil loss tolerance factor, delivery ratio, water quality index, and erosion index. VirGIS data are currently being used in the Virginia NPS program to identify and prioritize agricultural land areas needing improved NPS management. Once areas have been prioritized, cost-share and technical resources are targeted to improve the cost-effectiveness of the program. VirGIS data has also been used to identify highly erodible lands for the Soil Conservation Service. VirGIS currently contains over 36 million cells, or an area of 4 million ha. The system is expected to expand until the entire Chesapeake Bay drainage in Virginia is mapped.

Advantages and Limitations of Modeling

Modeling was found to be an important element of the RCWP program [76]. Models used for targeting and BMP assessment in the RCWP included: CREAMS [66], AGNPS [124], ANSWERS [10], and the USLE [120]. RCWP modeling experiences suggested that all models must be carefully calibrated for site-specific conditions even if it is claimed that a model requires no calibration. For example, CREAMS was found to be inaccurate in the northern climate of the Vermont RCWP, but minor modifications of the model greatly improved its accuracy and utility there [16]. A method for determining whether model predictions are within a prescribed factor of true values was developed and demonstrated on PRZM [91].

An important obstacle in using models for BMP assessment is that the greatest modeling needs are in rural ungaged watersheds. Lumped parameter models requiring calibration such as ARM, HSPF, and NPS are of limited value since there are few if any historical data available for calibration. Models such as AGNPS, ANSWERS, and SWAM that do not require calibration, however, require extensive amounts of information on watershed characteristics, which may or may not be readily available. In general, physically-based, deterministic models are better able to simulate the effects of BMPs and are therefore

recommended for evaluation of BMP effectiveness. At best, however, they and all other NPS models are accurate only to within a factor of 2 or 3, and their predictions should be used with full consideration of these uncertainties.

Summary and Implications

Considerable progress has been made in quantifying edge-of-field pollutant losses for most NPS pollutants for site-specific conditions. In addition, a theoretical understanding of many of the processes affecting pollutant transport has been developed. This knowledge has allowed the development of a variety of mathematical models for simulating pollutant fate and transport. Unfortunately, many of the models are highly site-specific, and the data required for use of the models are not readily available. Also, because most models are research-oriented, very few (if any) are capable of accurately simulating more than one or two different BMPs; thus they are not generally useful for planning purposes.

Conservation tillage is probably the most universally accepted BMP for reducing agricultural NPS pollution in surface runoff. Conservation tillage and other BMPs, however, are unlikely to achieve significant reductions in nutrient and pesticide delivery to waterways unless nutrient and pesticide levels in surface soils can be reduced. Surface application of fertilizers is the most popular method of fertilization for conservation tillage, but it is inappropriate from both agronomic and water quality viewpoints. New chemical application methods are needed that incorporate agricultural chemicals into the soil with minimal disturbance of surface residue. The effects of conservation tillage on pollutant loadings to groundwater is still open to considerable debate. Since conservation tillage increases infiltration and preferential flow, it is reasonable to assume that pollutant loadings to groundwater will also increase.

The effectiveness of BMPs for NPS pollution control was repeatedly reported to be highly site-specific. Variability in site conditions (soils, land use, topography, and weather) makes it difficult to show statistically significant differences between management practices. Consequently, it is difficult to use information obtained from RCWP and other monitoring projects to establish direct cause-and-effect relationships between BMPs and water quality, and the success of one project does not indicate the likely success of another project. Field- or plot-scale and paired watershed-scale studies in areas with uniform site conditions are often successful in finding statistically significant differences in BMPs, but

transferring these results to areas with different site conditions is questionable. Watershed-scale monitoring programs require a minimum of 5 years to detect water quality improvements. Because of time and economic constraints and because of the difficulty of extrapolating experimental field data from one area to another, evaluation of future BMP benefits can probably best be accomplished with properly selected simulation models that account for the effects of site-specific soil, crop, topographic, and weather conditions.

Research Needs for Effective Utilization of BMPs

Research needs to improve the effectiveness of BMPs for NPS pollution control include:

- Improved understanding of the processes transporting pollutants between fields and streams. Particular emphasis should be placed on the role of filter strips and riparian zones.
- Development of satisfactory equations for describing sediment transport in shallow overland flow. Currently-used equations were developed for channel flow conditions, which differ significantly from shallow overland flow conditions.
- Development of a nitrogen accounting model for temperate humid regions to allow better simulations of nitrogen transformations and availability of various nitrogen species.
- Development of a reliable test for plant-available nitrogen to reduce over-application of nitrogen fertilizers.
- Development of methods for applying agricultural chemicals below the soil surface while minimizing disturbance of surface residue.
- Development of a VFS design model that considers the effects of concentrated flow and long-term sediment and nutrient accumulation in VFSs.
- Building of comprehensive data bases for model development, testing, and verification. A series of intensively monitored (long-term) watersheds should be established that collect detailed data on weather; land use; cropping; chemical applications; runoff rates, quantity, and quality; and soil chemistry and pollutant transport. Without these data, models will continue to be evaluated with inadequate data bases, and modeling limitations will be blamed on data rather than on the models themselves.
- Concentration of model development efforts for BMP assessment on physically-based, determin-

istic, distributed-parameter models. These models are designed for use without calibration and have significant theoretical advantages over other types of models in evaluating BMP effectiveness.

- Development of standard methods to determine whether model predictions fall within a prescribed range of true values. Without this, it is difficult to compare alternative models or determine the adequacy of model predictions.
- Improved screening/targeting models to identify potentially critical sources of NPS pollution to

improve the cost-effectiveness of NPS pollution control programs. Particular emphasis should be placed on the development of geographic information systems for NPS management.

- Evaluation of the long-term nutrient removal capabilities of wet ponds enhanced with wetlands.
- Improved understanding of the long-term effects of conservation tillage practices, particularly no-till, on groundwater quality.
- Research into the effectiveness of alternative animal waste containment and treatment facilities on nutrient removal and disinfection.

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