CBP/TRS41/90
April 1990

Perspectives on the Chesapeake Bay, 1990

Advances in Estuarine Sciences

Chesapeake Bay Program

Printed on Recycled Paper

Perspectives on the Chesapeake Bay, 1990

Advances in Estuarine Sciences

Chesapeake Research Consortium
P.O. Box 1120
Gloucester Point, Virginia 23062

Compiled and edited by
Michael Haire and Elizabeth C. Krome

April 1990

Printed by the United States Environmental Protection Agency
for the
Chesapeake Bay Program

Printed on recycled paper

ACKNOWLEDGEMENTS

Members of the Steering Committee selected topics, authors, and reviewers for this document. The reviewers contributed helpful and thorough comments on the draft chapters. Marilyn Lewis of the VIMS Library performed literature searches for the authors. Susan Myers assisted in reference work, Cynthia Corlett made helpful editorial suggestions, and Alan Krome helped in proofreading. Pamela Owens gave capable and dedicated assistance in word-processing and production. Gail Mackiernan, Richard Batiuk, Maurice Lynch, and Robert Summers contributed knowledgeable guidance on specific chapters. Throughout the project Richard Reynolds and Steven Nelson provided advice and support.

STEERING COMMITTEE

Mr. Michael Haire
Dept. of the Environmental
Projects Program
2500 Broening Highway
Baltimore, Maryland 21224

Dr. Joseph Mihursky
Chesapeake Research Consortium
1 Williams Street
Solomons, Maryland 20688

Mr. Wayne Sullivan
3600 Edinger Road
Richmond, Virginia 23234

Dr. Charles C. Van Sickle
South East Forest Experiment Station
200 Weaver Blvd.
Asheville, North Carolina 22802

Ms. Gail Mackiernan
Maryland Sea Grant College
H. J. Patterson Hall
University of Maryland
College Park, Maryland 20742

Ms. Laura Lower
Council on the Environment
Ninth Street Office Building
Richmond, Virginia 23219

Dr. Maurice P. Lynch
Virginia Institute of Marine Science
Gloucester Point, Virginia 23062

Dr. L. Eugene Cronin
12 Mayo Avenue
Bay Ridge
Annapolis, Maryland 21403

Dr. Thomas Osborn
Chesapeake Bay Institute
The Rotunda
711 West 40th Street, Suite 340
Baltimore, Maryland 21221

TABLE OF CONTENTS

DISCLAIMER

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 Hopkinson 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 Hopkinson 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 ecosytem 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 neccessary 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

References

1. 1987 Chesapeake Bay Agreement. In: Comprehensive research plan, an agreement commitment report from the Chesapeake Executive Council. Annapolis, MD: Environmental Protection Agency; 1988.
2. Lynch, M.P.; Krome, E.C., eds. Perspectives on the Chesapeake Bay: recent advances in estuarine sciences. Gloucester Point, VA: Chesapeake Research Consortium; 1987; CRC pub. no. 127; CBP/TRS 16/87.

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

Richard L. Wetzel
College of William and Mary
School of Marine Science
Virginia Institute of Marine Science
Gloucester Point, Virginia 23062

Charles S. Hopkinson, Jr.
Ecosystems Center
Marine Biological Laboratory
Woods Hole, Massachusetts 02543

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

---

This review is contribution no. 1591 from the School of Marine Science and the Virginia Institute of Marine Science, College of William and Mary.

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 diagramed, 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 km^2 [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.

Wetlands

1. Plants and epiphytes
2. Benthic algae
3. Small consumers
4. Birds
5. Other wildlife
6. Detritus
7. Decomposers
8. Organic N, P, and C
9. Inorganic N, P, and C

Water column

10. Net phytoplankton
11. Nanoplankton
12. Net zooplankton
13. Microzooplankton
14. Ctenophores and jellyfish
15. Waterbirds
16. Menhaden, larval fish, etc.
17. Killifish, etc.
18. Croaker, etc.
19. Bluefish, etc.
20. Detritus
21. Bacteria and protozoa
22. Organic N, P, and C
23. Inorganic N, P, and C
24. Dissolved oxygen

Benthos

25. Sea grasses
26. Epiphytes
27. Benthic algae
28. Epifauna
29. Infauna
30. Oysters and clams
31. Blue crabs
32. Crustaceans, etc.
33. Detritus

Environment

35. Pollutants
36. River drainage
37. Land runoff
38. Atmosphere
39. Atlantic croaker
40. Deep sediments
41. Man

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

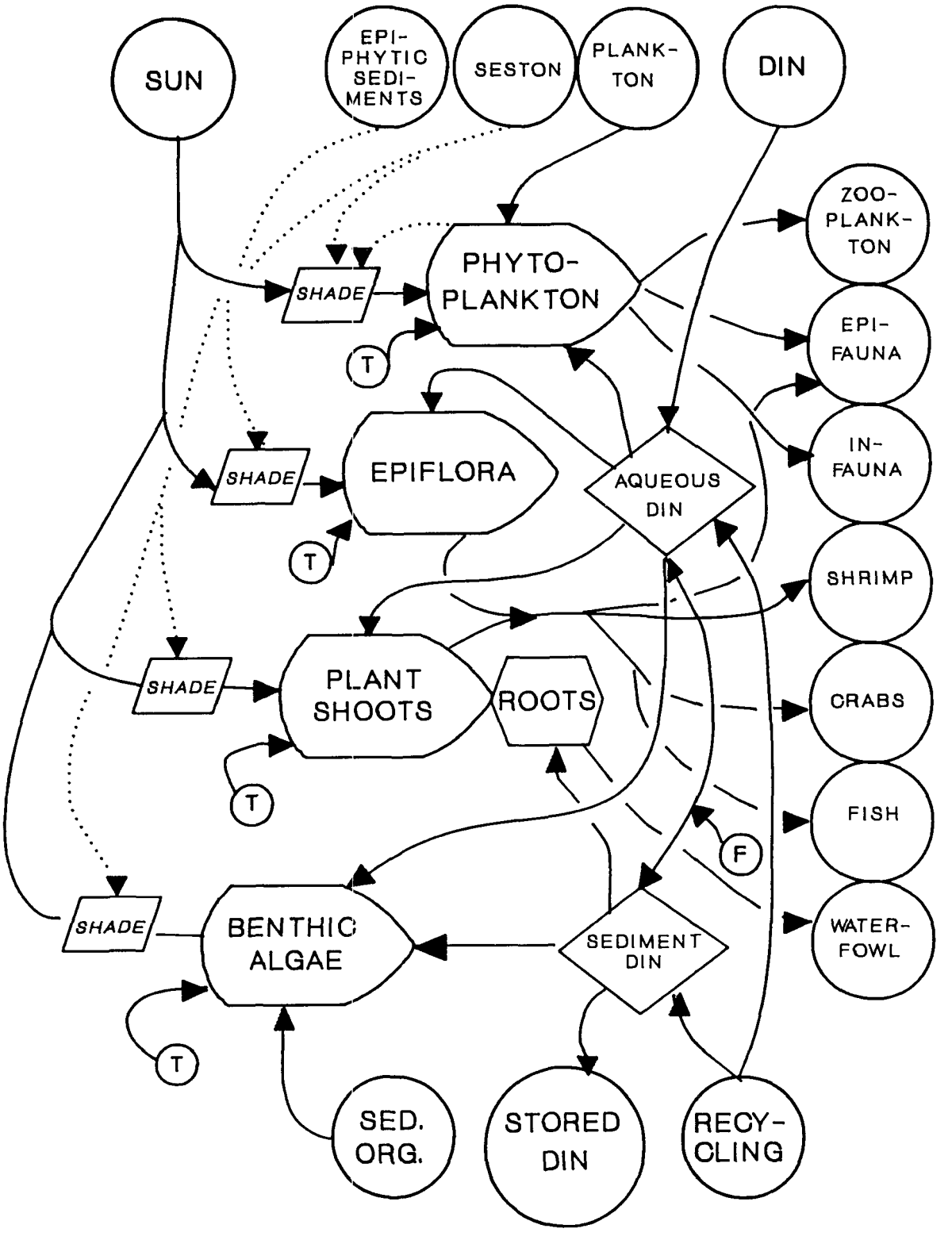

Figure 2. The compartmental structure and flow diagram used to simulate the autotroph submodel for studies of submerged aquatic vegetation in the upper Chesapeake Bay. For clarity of the diagram, the flows indicating losses from compartments due to respiration, excretion, or mortality have been omitted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

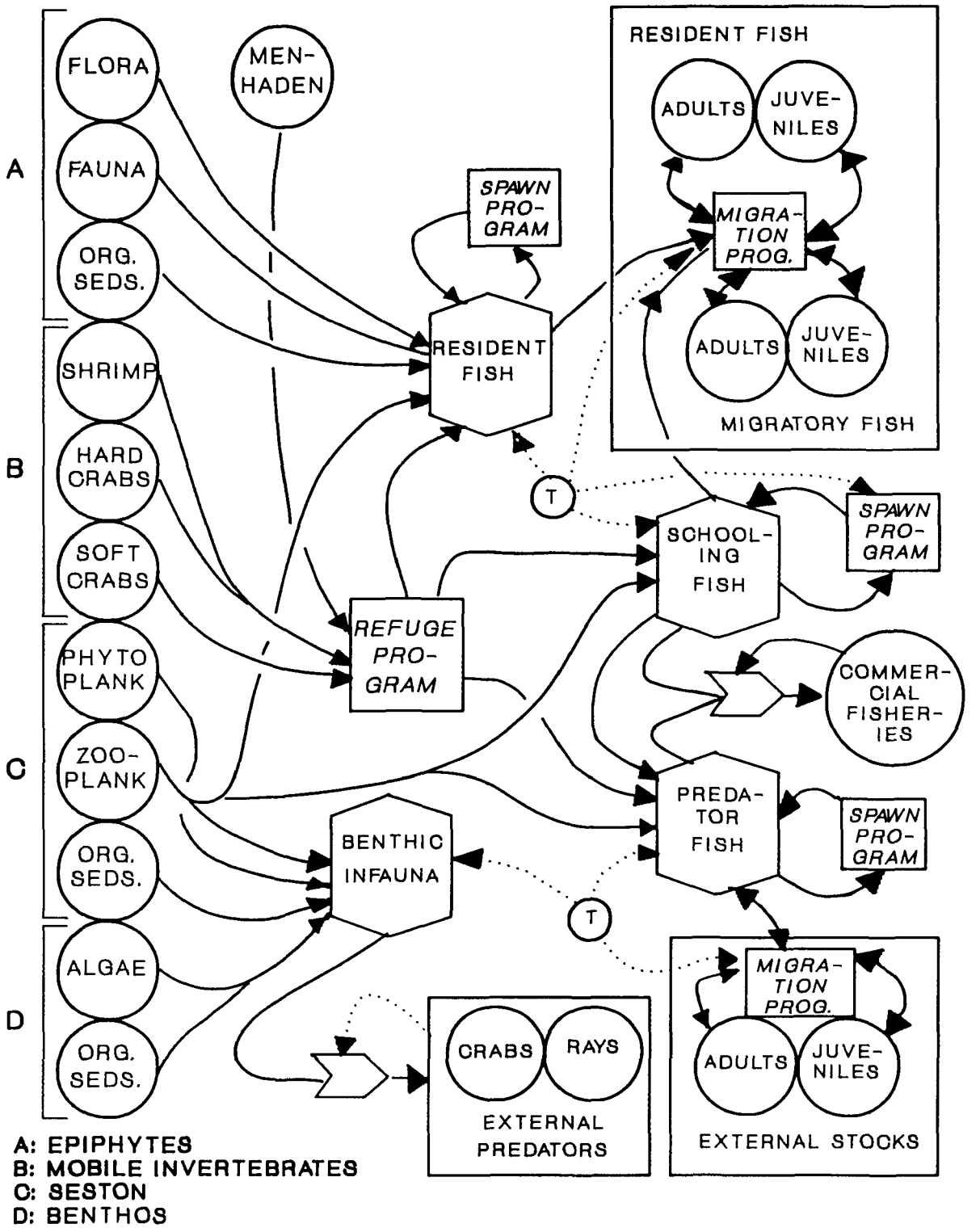

Figure 3. The compartmental structure and flow diagram used to simulate the nekton submodel for studies of submerged aquatic vegetation in the upper Chesapeake Bay. As in Figure 2, some flows have been omitted for clarity (redrawn from Kemp et al. [23]). T = temperature; org. seds. = organic sediments.

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

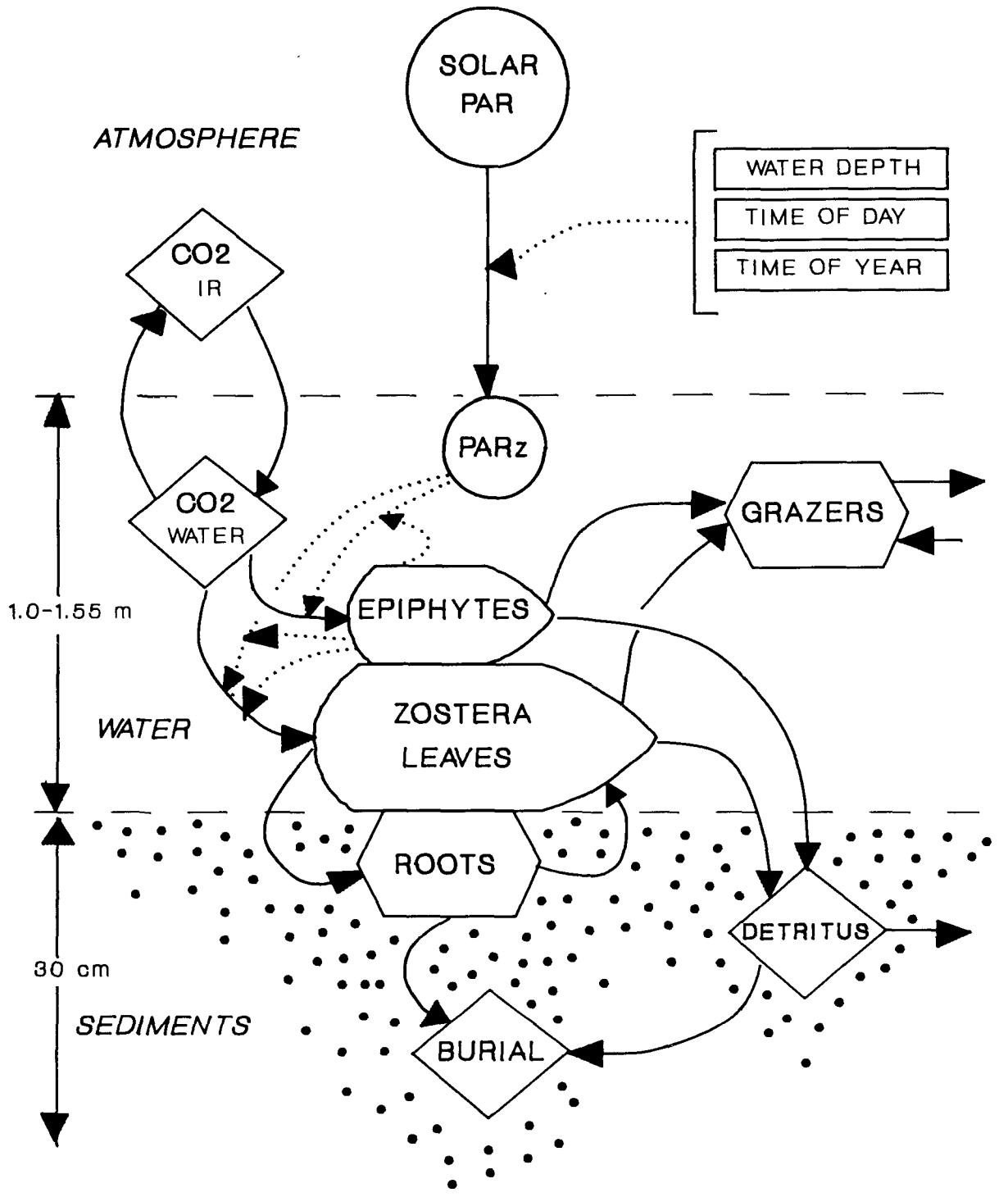

Figure 4. The compartmental structure and flow diagram used to simulate *Zostera marina* (eelgrass) photosynthesis and growth in the lower Chesapeake Bay under various conditions of environmental stress. Solid lines represent material transfers and dashed lines represent negative feedback controls (from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

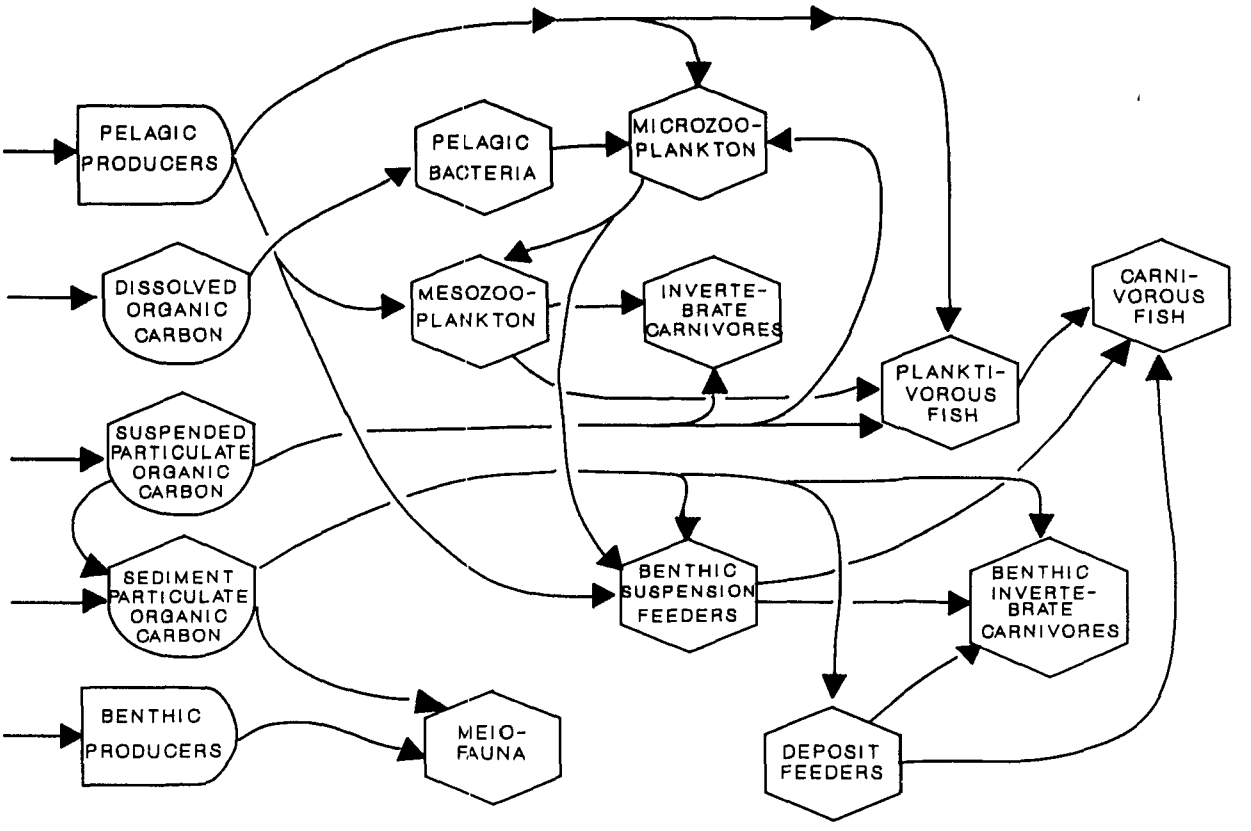

Figure 5. The compartmental structure and flow diagram for the Chesapeake Bay ecosystem used by Wulff and Ulanowicz [55] for network analysis and comparison with the Baltic Sea ecosystem. As in figures 2 and 3, some flows have been omitted for clarity.

addressing both basic scientific questions and contemporary management needs, and provided results that have guided management decisions. It is also apparent that ecosystem-level modeling and simulation analysis for Chesapeake Bay resources has been an underutilized tool whose potential is far from being realized.

We have not considered in this review two other major classes of models that are widely known and have been employed far longer in the management process: water quality models and fisheries models. Both have been reviewed elsewhere [43]. By our definition, these models are not ecosystem-level tools of analysis. However, the current efforts to develop and validate a three-dimensional, time-variable hydrodynamic-water quality model for the Chesapeake Bay [18] may provide the detail necessary for coupling physical, water quality, and ecological processes within a modeling framework. These models will incorporate much greater detail relative to biologically controlled processes (e.g., plankton and benthic processes) coupled to hydrodynamics and water quality. It may prove feasible in the near future to couple these models with large-scale ecosystem models by incorporating output of the hydrodynamic-water quality models as forcing/control input for ecosystem simulation studies, and similarly by using results of mechanistic ecosystem models to control feedback information to the hydrodynamic model. Ultimately, coupled models of this type and scale will be required for addressing large-scale management decisions.

For the present, ecosystem models for submerged aquatic vegetation, particularly those developed for the upper Bay, best exemplify modeling strengths and potentials in basic ecological research coupled with management needs for the Bay. Management of this largest of U.S. estuaries is a complex issue, and conflicting uses and demands on its resources will undoubtedly increase. Employing ecosystem-level models as a

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.

Acknowledgements

Developing and writing this review required the time and assistance of many colleagues and friends. Particular among these were Marilyn Lewis of the VIMS Library, who conducted an online literature search, and Rick Reynolds of CRC, who suffered through delays and weak excuses. Our special thanks are extended to Hilary Neckles, who provided us many helpful comments and editorial changes, and to Bruce Neilson and Carl Hershner for reviews of the draft manuscript. Also, thanks to three anonymous reviewers for suggestions and corrections that have been incorporated. To our modeling colleagues Bob Costanza, John Day, Michael Kemp, Scott Nixon, Charlie Vorosmarty, and Dick Wiegert, thanks for the conversations (debates) over the years that have helped focus the ideas and conclusions presented here.

References

1. Austin, H.M. Chesapeake Bay fisheries: an overview. In: Majumdar, S.K., Hall, L.W.; Austin, H.M., eds. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: The Pennsylvania Academy of Science; 1987:33-53.
2. Bahr, L.; Costanza, R.; Day, J.; Bayley, S.; Neill, C.; Leibowitz, S; Fruci, J. Ecological characterization of the Mississippi deltaic plain region: a narrative with management recommendations. Washington, D.C.: U. S. Fish and Wildlife Service, Division of Biological Services; 1983. 189 p.; FWS/OBS-82/69.
3. Correll, D.L. Nutrients in Chesapeake Bay. In: Majumdar, S.K.; Hall, L.W.; Austin, H.M., eds. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: The Pennsylvania Academy of Science; 1987:289-320.
4. Costanza, R. Model goodness of fit: a multiple resolution procedure. Ecological Modeling 47:199-215; 1989.
5. Costanza, R.; Bahr, L.; Neill, C.; Leibowitz, S.; Fruci, J. The Mississippi deltaic plain region study: an application of ecological models to the analysis and management of a complex coastal region. In: Lauenroth, W.; Skogerboe, G.; Flug, M., eds. Analysis of ecological systems: state-of-the-art in ecological modeling. Amsterdam: Elsevier; 1983:669-671.
6. Costanza, R; Sklar, F.; White, M. Modeling landscape dynamics in the Atchafalaya/Terrebonne marshes of coastal Louisiana. Bioscience 40(2):91-107; 1990.
7. Costanza, R.; Sklar, F.; Day, J. Using the Coastal Ecological Landscape Simulation Model (CELSS) for wetland management. In: Marani, A., ed. Advances in environmental modeling. Amsterdam: Elsevier; 1988:443-453.
8. D'Elia, C.F. Nutrient enrichment of Chesapeake Bay: an historical perspective. In: U.S. Environmental Protection Agency, Chesapeake Bay Program technical studies: a synthesis. Washington, D.C.: 1982:45-102.
9. D'Elia, C.F.; Webb, K.L.; Wetzel, R.L. Time-varying hydrodynamics and water quality in an estuary. In: Neilson, B.J.; Cronin, L.E., eds. Estuaries and nutrients. Clifton, N.J.: Humana Press; 1981:597-606.
10. Fitzpatrick, J.J.; Blumberg, A.F.; O'Connor, D.J.; Mulligan, T.J. Development of a coupled hydrodynamic/water quality model for assessing eutrophication and anoxia within Chesapeake Bay. In: Lynch, M.P.; McDonald, K.L., eds. Estuarine and coastal management—tools of the trade, proceedings of the 10th conference of the Coastal Society. Bethesda, MD: The Coastal Society; 1987:99-109.
11. Fleming, R.H. The control of diatom populations by grazing. J. Cons. Perm. Expl. Mer. 14:210-227; 1939.
12. Fruci, J.; Costanza, R.; Leibowitz, S. Quantifying the interdependence between material and energy flows in ecosystems. In: Lauenroth, W.; Skogerboe, G.; Flug, M., eds. Analysis of ecological systems: state-of-the-art in ecological modeling. Amsterdam: Elsevier; 1983:241-269.
13. Green, K.A. A conceptual ecological model for Chesapeake Bay. Washington, D.C.: U.S. Fish and Wildlife Service, Division of Biological Services; 1978. 22 p.; FWS/OBS-78/69.

14. Hall, C.A.S.; Day, J., eds. Ecosystem modeling in theory and practice: an introduction with case histories. New York: John Wiley & Sons; 1977. 684 p.
15. Heinle, D.R.; D'Elia, C.F.; Taft, J.L.; Wilson, J.S.; Cole-Jones, M.; Caplins, A.B.; Cronin, L.E. Historical review of water quality and climatic data from Chesapeake Bay with emphasis on effects of enrichment. Annapolis, MD: Chesapeake Research Consortium, Inc.; 1980; publ. no. 84.
16. Helz, G.R.; Huggett, R.J. Contaminants in Chesapeake Bay: the regional perspective. In: Majumdar, S.K.; Hall, L.W.; Austin, H.M., eds. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: The Pennsylvania Academy of Science; 1987:270-297.
17. Hershner, C.; Wetzel, R.L. Submerged and emergent aquatic vegetation of the Chesapeake Bay. In: Majumdar, S.K.; Hall, L.W.; Austin, H.M., eds. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: The Pennsylvania Academy of Science; 1987:116-133.
18. Hydroqual, Inc. Sediment processes and sediment modeling workshop. Proceedings of workshop no. 2; 1987 December 2-3. Mahwah, NJ: Hydroqual, Inc.; 1988.
19. Hopkinson, C.S.; Day, J.W. The relationship between development and storm water and nutrient runoff. Environ. Mgmt. 4:315-324; 1980.
20. Hopkinson, C.S.; Day, J.W. Modeling hydrology and eutrophication in a Louisiana swamp forest ecosystem. Environ. Mgmt. 4:325-335; 1980.
21. Jaworski, N.A. Sources of nutrients and the scale of eutrophication problems in estuaries. In: Neilson, B.J.; Cronin, L.E., eds. Estuaries and nutrients. Clifton, NJ: Humana Press, Inc.; 1981:83-110.
22. Kemp, W.M.; Twilley, R.R.; Stevenson, J.C.; Boynton, W.R.; Mean, J.C. The decline of submerged vascular plants in upper Chesapeake Bay: summary of results concerning possible causes. Mar. Tech. Soc. J. 17:78-89; 1983.
23. Kemp, W.M.; Boynton, W.R.; Hermann, A.J. A simulation modeling framework for ecological research in complex systems: the case of submerged vegetation in upper Chesapeake Bay. In: Turgeon, K.W., ed. Washington, D.C.: Marine Ecosystems Modeling, U.S. Department of Commerce, NOAA; 1983:131-158.
24. Kemp, W.M.; Boynton, W.R. Ecosystem modelling for understanding and managing natural estuarine resources associated with submerged vascular plants in upper Chesapeake Bay. Draft final report. Annapolis: Maryland Dept. Nat. Resources, Coastal Resources Division; 1983.
25. Kirkley, J.E. A socio-economic overview of the Chesapeake Bay fisheries. In: Majumdar, S.K.; Hall, L.W.; Austin, H.M., eds. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: The Pennsylvania Academy of Science; 1987:54-62.
26. Kremer, J.; Nixon, S. A coastal marine ecosystem, simulation and analysis. New York: Springer-Verlag; 1978.
27. Leibowitz, S.; Costanza, R. A preliminary input-output model of salt marshes in the Mississippi deltaic plain region. In: Lauenroth, W.; Skogerboe, G.; Flug, M., eds. Analysis of ecological systems: state-of-the-art in ecological modeling. Amsterdam: Elsevier; 1983:771-780.
28. Lotka, A.J. Elements of physical biology. Baltimore, MD: Williams and Wilkins; 1925. (Elements of mathematical biology) New York: Dover Press Reprint; 1956.
29 Nixon, S.; Kremer, J. Narragansett Bay—the development of a composite simulation model for a New England estuary. In: Hall, C.; Day, J., eds. Ecosystem modeling in theory and practice: an introduction with case histories. New York: John Wiley & Sons; 1977:621-673.
30. Officer, C.B.; Biggs, R.B.; Taft, J.L.; Cronin, L.E.; Tyler, M.A.; Boynton, W.R. Chesapeake Bay anoxia: origin, development and significance. Science 223:22-27; 1984.
31. Orth R.J.; Moore, K.A. Submerged aquatic vegetation of the Chesapeake Bay: past, present and future. In: Transactions of the 46th North American wildlife and natural resources conference. Washington, D.C.: Wildlife Management Institute; 1981:271-283.
32. Orth, R.J.; Moore, K.A. Chesapeake Bay: an unprecedented decline in submerged aquatic vegetation. Science 222:51-53; 1983.
33. Riley, G.A. Factors controlling phytoplankton populations on Georges Bank. J. Mar. Res 6:54-73; 1946.
34. Riley, G.A. A theoretical analysis of the zooplankton population of Georges Bank. J. Mar. Res. 6:104-113; 1947.
35. Riley, G.A.; Bumpus, D. Phytoplankton-zooplankton relationships on Georges Bank. J. Mar. Res. 6:33-47; 1946.
36. Riley, G.A.; Stommel, H; Bumpus, D. Quantitative ecology of the plankton of the Western North Atlantic. Bull. Bing. Oceanogr. Coll. 12:1-169; 1949.
37. Seliger, H.H.; Boggs, J.A.; Biggley, W.H. Catastrophic anoxia in Chesapeake Bay in 1984. Science 228:70-73; 1985.
38. Sklar, F.; Costanza, R.; Day, J. Dynamic spatial simulation modeling of coastal wetland habitat succession. Ecological Modeling 29:261-281; 1985.
39. Taft, J.L.; Taylor, W.R.; Hartwig, E.D.; Loftus, R.

Seasonal oxygen depletion in Chesapeake Bay. Estuaries 3:242-247; 1980.

40. Turner, M.G.; Costanza, R.; Sklar, F. Methods to evaluate the performance of spatial simulation models. Ecological Modeling 48:1-18; 1989.
41. Tuttle, J.H.; Jonas, R.B.; Malone, T.C. Origin, development and significance of Chesapeake Bay anoxia. In: Majumdar, S.K.; Hall, L.W.; Austin, H.M., eds. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: The Pennsylvania Academy of Science; 1987:442-472.
42. van Montfrans, J.; Wetzel, R.L.; Orth, R.J. Epiphyte-grazer relationships in seagrass meadows: consequences for seagrass growth and production. Estuaries 7(4A):289-309; 1984.
43. Vaughan, D.S.; Yoshiyama, R.M.; Breck, J.E.; DeAngelis, D.L. Review and analysis of existing modeling approaches for assessing population-level effects of multiple stresses on fish and shellfish. Oak Ridge, TN: Oak Ridge National Laboratory, Environmental Sciences Div.; 1982; publ. no. 1979.
44. Volterra, V. Variations and fluctuation of the number of individuals in animal species living together. In: Chapman, R., ed. Animal ecology. New York: McGraw-Hill; 1926:409-448.
45. Vorosmarty, C.; Moore, B.; Bowden, W.; Hobbie, J.; Peterson, B.; Morris, J. The transport and processing of nitrogen in a tidal, freshwater marsh and river ecosystem: modeling the roles of water movement and biotic activity in determining water quality. In: Lauenroth, W.; Skogerboe, G.; Flug, M., eds. Analysis of ecological systems: state-of-the-art in ecological modeling. Amsterdam: Elsevier; 1983:689-698.
46. Wetzel, R.L.; Penhale, P.A. Production ecology of seagrasses in the lower Chesapeake Bay. Mar. Tech. Soc. J. 17:22-31; 1983.
47. Wetzel, R.L.; Wiegert, R.G. Ecosystem simulation models: tools for the investigation and analysis of nitrogen dynamics in coastal and marine waters. In: Carpenter, E.J.; Capone, D.G., eds. Nitrogen in the marine environment. New York: Academic Press; 1983:869-892.
48. Wetzel, R.L.; Neckles, H.A. A model of *Zostera marina* L. photo-synthesis and growth: simulated effects of selected physical-chemical variables and biological interactions. Aquatic Bot. 26:307-323; 1986.
49. Wiegert, R.G. A general ecological model and its use in simulating algal-fly energetics in a thermal spring community. In: Geier, P.; Clark, L.; Anderson, D.; Nix, H., eds. Insects: studies in population management, vol. 1. Occasional Papers, Ecological Society of Australia; 1973:85-102.
50. Wiegert, R.G. Simulation modeling of the algal-fly components of a thermal ecosystem; effects of spatial heterogeneity, time delays and model condensation. In: Patten, B., ed. Systems analysis and simulation in ecology, vol. 2. New York: Academic Press; 1975:157-181.
51. Wiegert, R.G.; Christian, R.; Gallagher, J.; Hall, J.; Jones, R.; Wetzel, R. A preliminary ecosystem model of coastal Georgia *Spartina* marsh. In: Cronin, L., ed. Estuarine research, vol. 1. New York: Academic Press; 1975:583-601.
52. Wiegert, R.G.; Wetzel, R.L. Simulation experiments with a 14-compartment salt marsh model. In: Dame, R., ed. Marsh-estuarine systems simulation. Columbia: Univ. of South Carolina Press; 1979:7-39.
53. Wiegert, R.G.; Wetzel, R.L.; Vetter, E.F. Aerobe: An expanded ecosystem model of carbon transformations and transport in a Georgia salt marsh. 1984 ISEM Meeting, North American Chapter, Colorado State University, Fort Collins, Colorado 5-9 August 1984 (Abstract). 1984.
54. Wiegert, R.G.; Christian, R.; Wetzel, R. A model view of the marsh. In: Pomeroy, L.; Wiegert, R., eds. The ecology of a salt marsh. New York: Springer-Verlag; 1981:183-218.
55. Wulff, F.; Ulanowicz, R.E. A comparative anatomy of the Baltic Sea and Chesapeake Bay ecosystems. In Wulff, F.; Field, J.G.; Mann, K.H., eds. Network analysis in marine ecology: methods and applications. Lecture notes in coastal and estuarine studies. New York: Springer-Verlag; 1989:232-256.

The Functional Role of Estuarine Benthos

Robert J. Diaz and Linda C. Schaffner
Virginia Institute of Marine Science
College of William and Mary
Gloucester Point, Virginia 23062

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?

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

Animal-Habitat Relationships

Early studies focusing on the ecology of macrobenthic organisms (>500 μm) 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

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

Taxon	TF		OL		LM		HM-M		HM-MX		HM-S		P-M		P-MX		P-S		PE-MX		PE-S	
	A	B	A	B	A	B	A	B	A	B	A	B	A	B	A	B	A	B	A	B	A	B
Oligochaeta (esp. *Limnodrilus* spp.)	*	-	+	-																		
Chironimidae (In)	+	-																				
Corbicula fluminea (B)	+	-																				
Rangia cuneata (B)			+	*																		
Leptocheirus plumulosus (Am)			+	-	*	-	+	-														
Marenzellaria (Scolecolepides) viridis (A)			+	+	+	-					+	-										
Gammarus spp. (Am)	-	-	-	-	+	-																
Streblospio benedicti (A)					+	-	+	-	+	-	+	-										
Heteromastus filiformis (A)					+	-	-	-	-	-	+	-										
Tubificoides spp. (A)	-	-	+	-	+	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-
Nereis succinea (A)			-	-	+	-	+	-	+	-	+	-	-	-	-	-						
Cyathura spp. (Is)			-	-	+	-	-	-	-	-	-	-										
Macoma balthica (B)			-	-	+	+	+	*	+	-	+	*	-	-	-	-						
Mya arenaria (B)					+	-	-	-	-	-	+	-										
Gemma gemma (B)					-	-	+	-	-	-	*	-										
Crassostrea virginica †(B)					-	-	-	-	-	-	-	-										
Leucon americanus (C)							+	-	+	-			-	-								
Mulinia lateralis (B)			-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-
Paraprionospio pinnata (A)					-	-	+	-	+	-	-	-	+	-	+	-	-	-	+	-		
Mediomastus ambiseta (A)							+	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-
Glycinde solitaria (A)									-	-	+	-			-	-						
Glycera spp. (A)							-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Pseudeurythoe paucibranchiata (A)							-	-	+	-	-	-	+	-	+	-	+	-	-	-	-	-
Acteocina canaliculata (G)							-	-	-	-	-	-	-	-	+	-	+	-	-	-	-	-
Pectinaria gouldii (A)							-	-	-	-	-	-	+	-	+	-	+	-	-	-	-	-
Loimia medusa (A)							-	-					-	-	-	-	-	-	-	-	-	-
Ampelisca spp. (Am)									-	-			-	-	-	-	+	-	-	-	-	-
Molgula manhattensis (Ur)							-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Sigambra tentaculata (A)													+	-	+	-	-	-	+	-	-	-
Nephtys spp. (A)													-	-	+	-	-	-	+	-	-	-
Macroclymene zonalis (A)													-	-	+	+	-	-	+	+	-	-
Bhawania heteroseta (A)													-	-		+	-	-	+	-	-	-
Notomastus latericeus (A)													-	-	+	-	-	-	-	-	-	-
Chaetopterus variopedatus (A)													-	-	-	+	-	-	-	-	-	-
Clymenella torquata (A)													-	-	-	+	-	-	+	+	-	-
Phoronis spp. (Ph)													-	-	-	-	+	-	-	-	-	-
Mercenaria mercenaria †(B)													-	-	-	-	-	-	-	-	-	-
Tellina agilis (B)																	-	-	+	-	+	-
Ensis directus (B)															-	-	+	-	-	-	+	+
Amastigos caperatus (A)																			-	-	+	-
Haustoriidae (Am)																					+	-
Spiophanes bombyx (A)																	-	-	-	-	+	-
Mytilus edulis †(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/m^2, categories are: (-) <100, (+) 100 - 1000, (*) >1000; B = ash-free dry weight/m^2, 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 variopedatus* [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 μmol/hr for the small amphipod *Corophium volutator* to >1.0 μmol/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)
Oligochaeta (A) (especially *Limnodrilus* 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]
Corbicula fluminea (B)	SF	M	S		0-2	14.8 (1.0 gram wet wt.) 24.9 (31.0)	For *C. japonica*	[248, 275]
Rangia cuneata (B)	SF	LM	S,P		0-5			[53, 329, 334, Schaffner unpub.]
Leptocheirus plumulosus (Am)	SF,SD	S		T	0-10			[160, 281]
Maranzellaria (*Scolecolepides*) *viridis* (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]
Gammarus spp. (Am)		M	S,R		0+			[39, 192]
Streblospio benedicti (A)	SD,SF	S,LM	S,R	T	0-10	11.7-18.7*		[79, 160, 195]
Heteromastus filiformis (A)	HD	LM	S,P	B,V	0-35	262.5-700⁺	Yearly range	[52, 160, 195, 355]
Tubificoides 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]
Nereis succinea (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, 195, 311]
Cyathura polita (Is)	C?	M			0-15	2080§	Displacement	[160, 247, 281]
Lepidactylus dysticulus (Am)	SF	M			0-3			[85]
Macoma balthica (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]
Mya arenaria (B)	SF	S	I,U	B	0-25	27 (1820) 23 (990)		[47, 145, 160, 192]
Gemma gemma (B)	SF	LM	S		0-2			[43, 247, 321]
Crassostrea virginica (B)	SF	S	I,RI		0⁺	140.0 (480) 188.5 (1410) 222.9 (1140)		[145, 147, 192]

Leucon americanus (C)	SD	M			0-1,0+			[Schaffner unpub.]
Mulinia lateralis (B)	SF	M	S,R		0-5			[192, 280, 311, 329]
Paraprionospio pinnata (A)	SD,SF	LM	S,R	B	0-25	39.6-51.9* (juveniles) 90.9-232.8 (adults)		[79, 195, 217, 280, 311]
Mediomastus ambiseta (A)	HD	LM	S,P	T	0-6			[195, 311]
Glycinde solitaria (A)	C	M			0-6			[195, 311]
Glycera spp. (A)	C, BD	M	R	BG	0-15			[192, 195, 311]
Pseudeurythoe paucibranchiata (A)	C?,BD	M?			0-24			[195, 311]
Acteocina canaliculata (G)	C	M	S,P		0-2	2080§	Displacement	[192, 247, 311]
Pectinaria gouldii (A)	HD	LM	S,U,P	T,V?	0-8	6000 (80)	At 20 °C	[126, 142, 195, 311, 355]
Loimia medusa (A)	SD	S	I,U	T	0-15	5100 (900)	At 17 °C++	[142, 149, 195, 286, 311]
Ampelisca spp. (Am)	SD	S	S,R	T	0-5			[241, 311]
Molgula manhattensis (Ur)	SF	S	S,R		0+	80 (240) 40 (110)		[145, 192, 311]
Sigambra tentaculata (A)	C?,BD	M?			0-30			[109, 195, 311]
Nephtys spp. (A)	C,SD	M	R		0-10			[62, 109, 192, 195, 306, 307, 311, 337]
Macroclymene zonalis (A)	HD	S	S,C	T,V	0-20			[224, 286, 311]
Bhawania heteroseta (A)	C?	M			0-10			[109, 311]
Notomastus latericeus (A)	BD	LM	B		0-15			[109, 311]
Chaetopterus variopedatus (A)	SF	S	S,P	T	0-15			[311]
Clymenella torquata (A)	HD	S	S,U	T,V	0-30	1200 (47)	At 11 °C	[95, 142, 224, 286, 289, 311]
Phoronis spp. (Ph)	SF	S	S	T	0-20			[304]
Mercenaria mercenaria (B)	SF	LM	S,R		0-15			[192, 329]
Tellina agilis (B)	SF,SD	M	R		0-10			[192, 329, Schaffner unpub.]
Ensis directus (B)	SF	M	S,R		0-20+			[192, 329, Schaffner unpub.]
Haustoriidae spp. (Am)	SF	M			0-10	277-9900	Displacement	[171, Schaffner unpub.]
Spiophanes bombyx (A)	SD,SF	S	S	T	0-15	4.3-14.2*		[79]
Mytilus edulis (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 ≥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 biodiffusion 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 cm^3 = 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} $cm^2 sec^{-1}$, 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 phophorus, 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 denitrifcation 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 individuals, 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 successionally 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 $m^2 \cdot yr^{-1}$, with 11,427 km^2 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 $gC \cdot m^2 \cdot yr^{-1}$.

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 $C \cdot km^2 \cdot yr^{-1}$) and the least in the euhaline habitats (8.2 metric tons $C \cdot km^2 \cdot yr^{-1}$). 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

$C \cdot yr^{-1}$). 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 (± 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 $gC \cdot m^2 \cdot yr^{-1}$, 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 O_2) and anoxia (0 ppm O_2) 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 organims (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 $gC \cdot m^2 \cdot yr^{-1}$, with a total habitat range of about 2-289 $gC \cdot m^2 \cdot yr^{-1}$. 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 1-146 $gC \cdot m^2 \cdot yr^{-1}$. The range for similar habitats in Chesapeake Bay is about 6-32 $gC \cdot m^2 \cdot yr^{-1}$.

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 Skagerak-Kattegat, Moller et al. [242] found median productivity over a six-year period was about 10.7 $gC \cdot m^2 \cdot yr^{-1}$, with one-year productivity reaching 145.6 $gC \cdot m^2 \cdot yr^{-1}$ 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 $gC \cdot m^2 \cdot yr^{-1}$, and in the second two years, about 53 $gC \cdot m^2 \cdot yr^{-1}$. 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?

References

1. Abu-Hilal, A.; Badran, M.; de Vaugelas, J. Distribution of trace elements in *Callichirus laurae* burrows and nearby sediments in the Gulf of Aqaba, Jordan (Red Sea). Mar. Env. Res. 25:233-248; 1988.
2. Aller, R.C. The effects of animal-sediment interactions on geochemical processes near the sediment-water interface. In: Estuarine interactions. New York: Academic Press; 1978:157-172.
3. Aller, R.C. Quantifying solute distributions in the bioturbated zone of marine sediments by defining an average microenvironment. Geochim. Cosmochim. Acta 44:1955-1965; 1980.
4. Aller, R.C. The effects of macrobenthos on chemical properties of marine sediments and overlying water. In: McCall, P.L.; Tevesz, M.J.S. eds. Animal-sediment relations. New York: Plenum Press; 1982:53-102.
5. Aller, R.C. The importance of relict burrow structures and burrow irrigation in controlling sedimentary solute distributions. Geochim. Cosmochim. Acta 48:1929-1934; 1984.
6. Aller, R.C. Benthic fauna and biogeochemical processes in marine sediments: the role of burrow

structures. In: Blackburn, T.H.; Sorensen, J., eds. Nitrogen cycling in coastal marine environments. New York: John Wiley & Sons; 1988:302-338.

7. Aller, R.C.; Cochran, J.K. Th-234/U-238 disequilibrium in near-shore sediment: particle reworking and diagenetic time scales. Earth Plant. Sci. Let. 29:37-50; 1976.
8. Aller, R.C.; Yingst, J.Y. Biogeochemistry of tube-dwellings: a study of the sedentary polychaete *Amphitrite ornata* (Leidy). J. Mar. Res. 36 201-254; 1978.
9. Aller, R.C.; Yingst, J.Y. Relationships between microbial distributions and the anaerobic decomposition of organic matter in surface sediments of Long Island Sound, USA. Mar. Biol. 56:29-42; 1980.
10. Aller, R.C.; Yingst, J.Y. Effects of the marine deposit-feeders *Heteromastus filiformis* (Polychaeta), *Macoma balthica* (Bivalvia) and *Tellina texana* (Bivalvia) on averaged sedimentary solute transport, reaction rates and microbial distributions. J. Mar. Res. 43:615-645; 1985.
11. Aller, R.C.; Yingst, J.Y.; Ullman, W.J. Comparative biogeochemistry of water in intertidal *Onuphis* (Polychaeta) and *Upogebia* (Crustacea) burrows: temporal patterns and causes. J. Mar. Res. 41:571-604; 1983.
12. Alongi, D.M.; Tietjen, J.H. Population growth and trophic interactions among freeliving marine nematodes. In: Tenore, K.R.; Coull, B.C., eds. Marine benthic dynamics. Columbia, S.C.: Univ. of South Carolina Press; 1980:151-167.
13. Ambrose, J. Role of predatory infauna in structuring marine soft-bottom communities. Mar. Ecol. Prog. Ser. 17:109-115; 1984.
14. Andersen, F.O.; Kristensen, E. The influence of macrofauna on estuarine benthic community metabolism: a microcosm study. Mar. Biol. 99:591-603; 1988.
15. Anderson, A.E.; Jonas, E.C.; Odum, H.T. Alteration of clay minerals by digestive processes of marine organisms. Science 127:190-191; 1958.
16. Appleby, A.G.; Brinkhurst, R.O. Defecation rate of three tubificid oligochaetes found in the sediment of Toronto Harbour, Ontario. J. Fish. Res. Bd. Canada 27:1971-1982; 1970.
17. Arntz, W.E. Predation by demersal fish and its impact on the dynamics of macrobenthos. In: Tenore, K.R.; Coull, B.C., eds. Marine benthic dynamics. Columbia: Univ. of South Carolina Press, 1980:121-150.
18. Arntz, W.E.; Brunswig, D. Studies on structure and dynamics of macrobenthos in the western Baltic carried out by the joint research program "interaction sea-sea bottom" (SFB 95-Kiel). In: Persoone, G.; Jaspers, E., eds. Proceedings of 10th European symposium on marine biology. Bredene, Belgium: Universa Press; 1976:17-42.
19. Austin, H.M. Chesapeake Bay fisheries—an overview. In: Majumdar, S.K.; Hall, L.W.; Austin, H.M. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: The Pennsylvania Academy of Science; 1987:33-53.
20. Azam, F.; Fenchel, T.; Field, J.G.; Gray, J.S.; Meyer-Reil, L.A.; Thingstad, F. The ecological role of water-column microbes in the sea. Mar. Ecol. Prog. Ser. 10:257-263; 1983.
21. Bagheri, E.A.; McLusky, D. S. The oxygen consumption of *Tubificoides benedeni* (Udekem) in relation to temperature, and its application to production biology. J. Exp. Mar. Biol. Ecol. 78:187-197;1984.
22. Bailey-Brock, J.H. Sediment trapping by chaetopterid polychaetes on a Hawaiian fringing reef. J. Mar. Res. 37:643-656; 1979.
23. Baird, D.; Milne, H. Energy flow in the Ythan Estuary, Aberdeenshire, Scotland. Estuar. Coast. Shelf Sci. 13:455-472; 1981.
24. Baird, D.; Ulanowicz, R.E. The seasonal dynamics of the Chesapeake Bay ecosystem. Ecol. Monogs. 59:329-364; 1989.
25. Banse, K.; Mosher, S. Adult body mass and annual production/biomass relationships of field populations. Ecol. Monogr. 50:353-379; 1980.
26. Bauer, J.E.; Kerr, R.P.; Bautista, M.F.; Decker, C.J.; Capone, D.G. Stimulation of microbial activities and polycyclic hydrocarbon degradation in marine sediments inhabited by *Capitella capitata*. Mar. Environ. Res. 25:63-84; 1988.
27. Benke, A.C.; Hall, C.A.S.; Hawkins, C.P.; Lowe-McConnell, R.H.; Stanford, J.A.; Suberkropp, K.; Ward, J.V. Bioenergetic considerations in the analysis of stream ecosystems. J. N. Am. Benthol. Soc. 7:410-432; 1988.
28. Biggs, R.B.; Flemer, D.A. The flux of particulate carbon in an estuary. Mar. Biol. 12:11-17; 1972.
29. Blackburn, T.H.; Henriksen, K. Nitrogen cycling in different types of sediments from Danish waters. Limnol. Oceanogr. 28:477-493; 1983.
30. Blundon, J.A.; Kennedy, V.S. Refuges for infaunal bivalves from blue crab, *Callinectes sapidus* (Rathbun). Predation in Chesapeake Bay. J. Exp. Mar. Biol. Ecol. 65:67-81; 1982.
31. Boddington, M.J.; Miller, D.R.; DeGreitas, S.W. The effects of benthic invertebrates on the clearance of mercury from contaminated sediments. Ecotoxicol. Environ. Saf. 3:236-244; 1979.
32. Bodin, P.; Boucher, D.; Guillou, J.; Guillou, M. The trophic system of the benthic communities in the Bay of Douarnenez (Brittany). Proceedings of 19th European symposium on marine biology; 1985:361-370.
33. Boesch, D.F. Distribution and structure of benthic

communities in a gradient estuary. Williamsburg, VA: College of William and Mary; 1971; 120 p. Dissertation.

34. Boesch, D.F. Classification and community structure of macrobenthos in the Hampton Roads area, Virginia. Mar. Biol. 21:226-244; 1973.
35. Boesch, P.F. A new look at the zonation of benthos along the estuarine gradient. In: Coull, B.C., ed. Ecology of marine benthos. Columbia: University of South Carolina Press; 1977:245-266.
36. Boesch, D.F.; Diaz, R.J.; Virnstein, R.W. Effects of tropical storm Agnes on soft-bottom macrobenthic communities of the James and York estuaries and the lower Chesapeake Bay. Ches. Sci. 17:246-259; 1976a.
37. Boesch, D.F.; Wass, M.L.; Virnstein, R.W. The dynamics of estuarine benthic communities. In: Wiley, M., ed. Estuarine processes. New York: Academic Press; 1976b:177-196.
38. Boothe, P.N.; Knauer, G.A. The possible importance of fecal material in the biological amplification of trace and heavy metals. Limnol. Oceanogr. 17:270-274; 1972.
39. Bousfield, E.L. Shallow-water gammaridean Amphipoda of New England. Ithaca, NY: Cornell University Press; 1973. 312 p.
40. Boynton, W.R.; Kemp, W.M. Nutrient regeneration and oxygen consumption by sediments along an estuarine salinity gradient. Mar. Ecol. Prog. Ser. 23:45-55; 1985.
41. Boynton, W.R.; Kemp, W.M.; Keefe, C.W. A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. In: Kennedy, V.S., ed. Estuarine comparisons. New York: Academic Press; 1982:69-90.
42. Boynton, W.R.; Kemp, W.M.; Garber, J.H.; Barnes, J.M. Report to: Maryland Office of Environmental Programs. Chesapeake Bay Water Quality Monitoring Program. Ecosystems Processes Component. July 1985-May 1986. Solomons, MD:UMCEES, CBL, Univ. of Maryland; 1986. CBL 86-56 1-52.
43. Bradley, W.H.; Cooke, P. Living and ancient populations of the clam *Gemma gemma* in a Maine coast tidal flat. U.S. Fish Wildl. Serv. Fish. Bull. 58:305-334; 1959.
44. Brafield, A.E.; Newell, G.E. The behavior of *Macoma balthica* (L.) J. Mar. Biol. Ass. U.K. 41:81-87; 1961.
45. Bray, R.N.; Miller, A.C.; Johnson, S.; Krause, P.R.; Robertson, D.L.; Westcott, A.M. Ammonium excretion by macroinvertebrates and fishes on a subtidal rocky reef in southern California. Mar. Biol. 100:21-30; 1988.
46. Briggs, K.B.; Tenore, K.R.; Hanson, R.B. The role of microfauna in detrital utilization by the polychaete *Nereis succinea* (Frey and Leukart). J. Exp. Mar. Biol. Ecol. 36:225-234; 1979.
47. Brousseau, D.J. Population dynamics of the soft-shell clam *Mya arenaria*. Mar. Biol. 50:63-71; 1978.
48. Bryan, G.W.; Gibbs, P.E.; Hummerstone, L.G.; Burt, G.R. Copper, zinc and organotin as long-term factors governing the distribution of organisms in the Fal estuary in southwest England. Estuaries 10:208-219; 1987.
49. Bubnova, N.P. The nutrition of the detritus-feeding mollusks *Macoma balthica* (L.) and *Portlandia arctica* (Gray) and their influence on bottom sediments. Oceanology 12:899-905; 1972.
50. Buchanan, J.B.; Kingston, P.F.; Sheader, M. Long-term population trends of the benthic macrofauna in the offshore mud of the Northumberland coast. J. Mar. Biol. Assoc. U. K. 54:785-795; 1974.
51. Buchanan, J.B.; Warwick, R.M. An estimate of benthic macrofaunal production in the offshore mud of the Northumberland coast. J. Mar. Biol. Assoc. U. K. 54:197-222; 1974.
52. Cadee, G.C. Sediment reworking by the polychaete *Heteromastus filiformis* on a tidal flat in the Dutch Wadden Sea. Neth. J. Sea Res. 13:441-456; 1979.
53. Cain, T.D. Reproduction and recruitment of the brackish water clam *Rangia cuneata* in the James River, Virginia. U.S. Natl. Mar. Fish. Serv. Fish. Bull. 73:412-430; 1975.
54. Callender, E.; Hammond, D.E. Nutrient exchange across the sediment-water interface in the Potomac River estuary. Estuar. Coast. Shelf Sci. 15:395-413; 1982.
55. Cammen, L.M.; Rublee, P.A.; Hobbie, J.E. The significance of microbial carbon in the nutrition of the polychaete *Nereis succinea* and other aquatic deposit feeders. UNC Sea Grant Publication UNC-SG-78-12; 1978. 84 p.
56. Capone, D.G.; Kiene, R.P. Comparison of microbial dynamics in marine and freshwater sediments: contrasts in anaerobic carbon catabolism. Limnol. Oceanogr. 33:725-749; 1988.
57. Capuzzo, J.M.; Sasner, J.J. The effect of chromium on filtration rates and metabolic activity of *Mytilus edulis* L. and *Mya arenaria* L. Physiological responses of marine biota to pollutants. 1977:225-237.
58. Cederwell, H. Annual macrofauna production of a soft bottom in the northern Baltic proper. In: Keegan, B.F.; Ceidigh, P.O.; Boaden, P.J.S., eds. Biology of benthic organsims. Oxford, England: Pergamon Press; 1977:155-164.
59. 1987 Chesapeake Bay Agreement. In: Comprehensive research plan, an agreement commitment report from the Chesapeake Executive Council. Annapolis, MD: Environmental Protection Agency; 1988.
60. Chesney, E.J.J. Laboratory studies of the effect of

predation on production and the production:biomass ratio of the opportunistic polychaete *Capitella capitata* (type I). Mar. Biol. 87:307-312; 1985.

61. Christian, R.R.; Wetzel, R.L. Interaction between substrate, microbes, and consumers of *Spartina* "detritus" in estuaries. In: Wiley, M., ed. Estuarine interactions. New York: Academic Press; 1978.
62. Clifton, T.R. Heavy mineral concentration at the bottom of polychaete traces in sandy sediment. J. Sed. Petrol. 54:151-153; 1984.
63. Cloern, J.E. Does the benthos control phytoplankton biomass in South San Francisco Bay? Mar. Ecol. Prog. Ser. 9:191-202; 1982.
64. Cockcroft, A.C.; Webb, P.; Wooldridge, T. Nitrogen regeneration by two surf zone mysids, *Mesopodopsis slabberi* and *Gastorsaccus psammodytes*. Mar. Biol. 99:75-82; 1988.
65. Cohen, R.R.H.; Dresler, P.V.; Phillips, E.J.P.; Cory, R.L. The effect of the Asiatic clam, *Corbicula fluminea*, on phytoplankton of the Potomac River, Maryland. Limnol. Oceanogr. 29:170-180; 1984.
66. Cole, J.J.; Findlay, S.; Pace, M.L. Bacterial production in fresh and saltwater ecosystems: a cross-system overview. Mar. Ecol. Prog. Ser. 43:1-10; 1988.
67. Commito, J.A. effects of *Lunatia heros* predation on the population dynamics of *Mya arenaria* and *Macoma balthica* in Maine, USA. Mar. Biol. 69:187-193; 1982.
68. Connor, M.S.; Edgar, R.K. Selective grazing by the mud snail *Ilyanassa obsoleta*. Oecologia 53:271-275; 1982.
69. Costerton, J.W.; Geesey, G.C.; Cheng, K.J. How bacteria stick. Scient. Am. 238:86-95; 1978.
70. Coull, B.C. The use of long-term biological data to generate testable hypotheses. Estuaries 8:84-92; 1985.
71. Crisp, D.J. Energy flow measurements. In: Holme, N.A.; McIntyre, A.D., eds. IBP Handbook 16. Oxford, England: Blackwell Scientific Publishers; 1971:197-279.
72. Cronin, W.B. Volumetric, areal, and tidal statistics of the Chesapeake Bay estuary and its tributaries. Shady Side, MD: Chesapeake Bay Institute, The Johns Hopkins University; 1971; spec. rpt. 20, ref. 71-2 135.
73. Cullen, D.J. Bioturbation of superficial marine sediments by interstitial meiobenthos. Nature 242:323-324; 1973.
74. Dales, R.P. Respiration and energy metabolism in annelids. In: Florkin, M.; Scheer, B.T., eds. Chemical zoology. New York: Academic Press; 1969:93-109.
75. Dales, R.P.; Mangum, C.P.; Tichy, J.C. Effects of changes in oxygen and carbon dioxide concentrations on ventilation rhythms in onuphid polychaetes. J. Mar. Biol. Assoc. U. K. 50:365-380; 1970.
76. Dame, R.F. Energy flow in an intertidal oyster population. Estuar. Coast. Mar. Sci. 4:243-253; 1976.
77. Dame, R.F.; Zingmark, R.G.; Haskin, E. Oyster reefs as processors of estuarine materials. J. Exp. Mar. Biol. Ecol. 83:239-247; 521-526; 1984.
78. Dame, R.; Zingmark, R.; Nelson, D. Filter feeding coupling between the estuarine water column and benthic subsystems. In: Kennedy, V.S., ed. Estuarine perspectives. New York: Academic Press; 1980:521-526.
79. Dauer, D.M.; Maybury, C.A.; Ewing, R.M. Feeding behavior and general ecology of several spionid polychaetes from the Chesapeake Bay. J. Exp. Mar. Biol. Ecol. 54:21-38; 1981.
80. Dauer, D.M.; Ewing, R.M.; Ranasinghe, J.A. Macrobenthic communities of the lower Chesapeake Bay, Chesapeake Bay program, March 1985 - June 1987. Rpt. to Virginia Water Cont. Bd. Norfolk: Old Dominion Univ. Dept. Biol. Sci.; 1988; 310 p.
81. Dauer, D.M.; Stokes, T.L.; Barker, H.R.; Ewing, R.M.; Sourbeer, J.W. Macrobenthic communities of the lower Chesapeake Bay. IV. Bay-wide transects and the inner continental shelf. Benthic studies of the lower Chesapeake Bay 7. Int. Revue ges. Hydrobiol. 69:1-22; 1984.
82. Davies, J.M.; Payne, R. Supply of organic matter to the sediment in the northern North Sea during a spring phytoplankton bloom. Mar. Biol. 78:315-324; 1984.
83. Day, J.H. Estuarine ecology with particular reference to southern Africa. Rotterdam: A. A. Balkema; 1981.
84. Day, J.W.; Hall, C.A.S.; Kemp, W.M.; Yanez-Arancibia, A. Estuarine ecology. New York: John Wiley & Sons; 1989.
85. Dexter, D.M. Distribution and niche diversity of haustoriid amphipods in North Carolina. Ches. Sci. 8:187-192; 1967.
86. Diaz, R.J. Short term dynamics of the dominant annelids in a polyhaline temperate estuary. Hydrobiologia 115:153-158; 1984.
87. Diaz, R.J. Pollution and tidal benthic communities of the James River Estuary, Virginia. Hydrobiologia 180:195-211; 1989.
88. Diaz, R.S. The effects of pollution on benthic communities of the tidal James River, Virginia. Charlottesville: University of Virginia; 1977; 149 p. Dissertation.
89. Diaz, R.J.; Boesch, D.F. Environmental effects of the James River sewage treatment plant outfall construction: soft bottom macrobenthos. Rpt. to Hampton Roads Sanitation District, Hampton, Virginia 33; 1976.
90. Diaz, R.J.; Boesch, D.F.; Hauer, J.L.; Stone, C.A.; Munson, K. Aquatic biology-benthos. Habitat development field investigations, Windmill Pt. marsh development site, James River, Virginia. Appendix D: Environmental impacts of marsh development with dredged material: botany, soils, aquatic biology, and

wildlife. U.S. Army Engineer Waterways Experimental Station. 1978; Tech. rpt. d-77-23:18-54.

91. Diaz, R.J.; Orth, R.J.; Markwith, G.; Rizzo, W.; Wetzel, R.; Storey, K. Examination of tidal flats: a review of identified values. U. S. Dept. Transportation, Fed. Highway Admin.; 1982; 45 p.; FHWA/RD80/182.
92. Diaz, R.J.; Schaffner, L.C. Comparison of sediment landscapes in Chesapeake Bay as seen by surface and profile imaging. In: Lynch, M.P.; Krome, E.C., eds. Understanding the estuary: advances in Chesapeake Bay research. Gloucester Point, VA: Chesapeake Research Consortium: 1988:222-240; pub. 129.
93. Diaz, R.J.; Schaffner, L.C. Benthic community dynamics relative to open-water disposal of dredged material at Rappahannock Shoals. Interim Rpt. to U.S. Army Corps Engnr., Baltimore Dist.; 1989; 80 p.
94. Dobbs, F.C. Monitoring defecation activity of infaunal deposit feeders. Mar. Ecol. Prog. Ser. 12:47-50; 1983.
95. Dobbs, F.C.; Whitlatch, R.B. Aspects of deposit-feeding by the polychaete *Clymenella torquata.* Ophelia 21:159-166; 1982.
96. Doering, P.H. On the contribution of the benthos to pelagic production. J. Mar. Res. 47:371-383; 1989.
97. Doering, P.H.; Oviatt, C.A. Application of filtration rate models to field populations of bivalves: an assessment using experimental mesocosms. Mar. Ecol. Prog. Ser. 31:265-275; 1986.
98. Doering, P.H.; Oviatt, C.A.; Kelly, J.R. The effects of the filter-feeding clam *Mercenaria mercenaria* on carbon cycling in experimental marine mesocosms. J. Mar. Res. 44:839-861; 1986.
99. Doering, P.H.; Kelly, J.R.; Oviatt, C.A.; Sowers, T. Effect of the hard clam *Mercenaria mercenaria* on benthic fluxes of inorganic nutrients and gases. Mar. Biol. 94:377-383; 1987.
100. Downing, J.A.; Rigler, F.H. A manual on methods for the assessment of secondary productivity in fresh waters. IBP Handbook No. 17, 2nd ed. Oxford, England: Blackwell Scientific Publications; 1984.
101. Dworschak, P.C. The pumping rates of the burrowing shrimp *Upogebia pusilla* (Petagna) (Decapoda:Thalassinidea). J. Exp. Mar. Biol. Ecol. 52:25-35; 1981.
102. Eckman, J.E.; Nowell, A.R.M.; Jumars, P.A. Sediment destabilization by animal tubes. J. Mar. Res. 39:361-374; 1981.
103. Ecological Analysts. An environmental assessment and ecological survey of the aquatic biota, C. P. Crane power plant, 1978-1980. Rpt. to Baltimore Gas and Electric, Baltimore, Maryland. Sparks, MD: Ecological Analysts; 1981.
104. Edwards, R.W.; Rolley, H.L.J. Oxygen consumption of river muds. J. Ecol. 53:1-19; 1965.
105. Elliott, M.; McLusky, D.S. Invertebrate production ecology in relation to estuarine management. In: Halcrow, W.; Wilson, J.G., eds. Estuarine management and quality assessment. New York: Plenum Press; 1984.
106. Enoksson, V.; Samuelsson, M.O. Nitrification and dissimilatory ammonium production and their effects on nitrogen flux over the sediment-water interface in bioturbated coastal sediments. Mar. Ecol. Prog. Ser. 36:181-189; 1987.
107. Evans, S. Production, predation and food niche segregation in a marine shallow soft-bottom community. Mar. Ecol. Prog. Ser. 10:147-157; 1983.
108. Ewing, R.M.; Dauer, D.M. Macrobenthic communities of the lower Chesapeake Bay I. Old Plantation Creek, Kings Creek, Cherrystone Inlet and the adjacent offshore area. Benthic studies of the lower Chesapeake Bay 4. Int. Revue ges. Hydrobiol. 67:777-791; 1982.
109. Fauchald, K.; Jumars, P.A. The diet of worms. Oceanogr. Mar. Biol. Ann. Rev. 17:193-284; 1979.
110. Fenchel, T. Studies on the decomposition of turtle grass *Thalassia testudinum.* Limnol. Oceanogr. 15:14-20; 1970.
111. Fenchel, T. Intrinsic rate of natural increase: the relationship with body size. Oecologia 14:317-326; 1974.
112. Fenchel, T.M.; Jorgensen, B.B. Detritus food chains of aquatic ecosystems: the role of bacteria. In: Alexander, M., ed. Advances in microbial ecology. New York: Plenum Press; 1977:1-58.
113. Findlay, S.E.G.; Tenore, K.R. Effect of a free-living marine nematode (*Diplolaimella chitwoodi*) on detrital carbon mineralization. Mar. Ecol. Prog. Ser. 8:161-166; 1982.
114. Fisher, J.B. Effects of macrobenthos on the chemical diagenesis of freshwater sediments. In: McCall, P.L.; Tevesz, M.J.S., eds. Animal-sediment relations. New York: Plenum Press; 1982:177-220.
115. Flemer, D.A. Primary production in the Chesapeake Bay. Ches. Sci. 11:117-129; 1970.
116. Flemer, D.A.; Biggs, R.B. Particulate carbon:nitrogen relations in Northern Chesapeake Bay. J. Fish. Bd. Can. 28:911-918; 1971.
117. Foster-Smith, R.L. An analysis of water flow in tube-living animals. J. Exp. Mar. Biol. Ecol. 34:73-95; 1978.
118. Foster-Smith, R.L.; Shillaker, R.O. Tube irrigation by *Lembos websteri* Bate and *Corophium bonnelli* Milne Edwards (Crustacea:Amphipoda). J. Exp. Mar. Biol. Ecol. 26:289-296; 1977.
119. Fowler, S.W. Biological transfer and transport processes. In: Kullenberg, G., ed. Pollutant transfer and transport in the sea, Volume II. Boca Raton, FL: CRC Press; 1982:1-65.
120. Fredette, T.J.; Diaz, R.J. Secondary production of

Gammarus mucronatus Say (Amphipoda: Gammaridae) in warm temperate estuarine habitats, York River, Virginia. J. Crustacean Biol. 6:729-741; 1986.

121. Frithsen, J.B.; Doering, P.H. Active enhancement of particle removal from the water column by tentaculate benthic polychaetes. Ophelia 25:169-182; 1986.
122. Gallepp, G.W. Chironomid influence on phosphorus release in sediment-water microcosms. Ecology 60:458-465; 1979.
123. Gallepp, G.W., Kitchell, J.F.; Bartell, S.M. Phosphorus release from lake sediments as affected by chironomids. Verh. Int. Verein. Limnol. 20:458-465; 1978.
124. Garber, J.H. Benthic-pelagic coupling in the Chesapeake Bay. In: Lynch, M.P.; Krome, E.C., eds. Perspectives in the Chesapeake Bay: advances in estuarine sciences. Gloucester Point, VA: Chesapeake Research Consortium; 1987:11-34; pub. no. 127.
125. Gerritsen, J. Benthic filter feeders in Chesapeake Bay: production and biological control of water quality. Appendix A. Long-term benthic monitoring and assessment program for the Maryland portion of Chesapeake Bay: interpretive report. Report to Maryland Department of Natural Resources by VERSAR Inc., ESM Operations; 1988: 18 p.
126. Gordon, D.C. The effects of the deposit feeding polychaete *Pectinaria gouldii* on the intertidal sediments of Barnstable Harbor. Limnol. Oceanogr. 11:327-332; 1966.
127. Grant, J. The relative magnitude of biological and physical sediment reworking in an intertidal sediment. J. Mar. Res. 41:673-689; 1983.
128. Grant, J.; Gust, G. Prediction of coastal sediment stability from photopigment content of mats of purple sulphur bacteria. Nature 330:244-246; 1987.
129. Grant, J.; Bathmann, U.V.; Mills, E.L. The interaction between benthic diatom films and sediment transport. Estuar. Coast. Shelf Sci. 23:225-238; 1986.
130. Grant, W.D.; Boyer, L.F.; Sanford, L.P. The effects of bioturbation on the initiation of motion in intertidal sands. J. Mar. Res. 40:659-677; 1982.
131. Green, K.A. A conceptual ecological model for Chesapeake Bay. U. S. Fish and Wildlife Service, Biological Services Program FWS/OBS-78/69:22; 1978.
132. Gremare, A. Feeding, tube-building and particle-size selection in the terebellid polychaete *Eupolymnia nebulosa*. Mar. Biol. 97:243-252; 1988.
133. Greze, V.N. Production in animal populations. In: Kinne, O., ed. Marine ecology, vol. IV. New York: John Wiley & Sons; 1978:89-114.
134. Grossline, M.D.; Langton, R.W.; Sissenwine, M.P. Recent fluctuations in pelagic fish stocks of the Northwest Atlantic, Georges Bank region in relation to species interactions. Rapp. P. -v. Reun. Cons. Int. Explor. Mer 177:374-404. 1980.
135. Guinasso, N.L.; Schink, D.R. Quantitative estimates of biological mixing rates in abyssal sediments. J. Geophys. Res. 80:3032-3043; 1975.
136. Gust, G.; Harrison, J.T. Biological pumps at the sediment-water interface: mechanistic evaluation of the alpheid shrimp *Alpheus mackayi* and its irrigation pattern. Mar. Biol. 64:71-78; 1981.
137. Haas, L.W. The effect of the spring-neap tidal cycle on the vertical salinity structure of the James, York and Rappahannock Rivers, Virginia, U.S.A. Estuar. Coastal Mar. Sci. 5:485-496; 1977.
138. Haire, M.S. An investigation of the macrobenthic communities in the vicinity of the Morgantown power plant. Towson, MD: Towson State Univ.; 1978. Thesis.
139. Hale, S.S. The role of benthic communities in the nitrogen and phosphorus cycles of an estuary. Mineral Cycling in Southeastern Ecosystems 291-308; 1975.
140. Hall, D.J.; Cooper, W.E.; Werner, E.E. An experimental approach to the production dynamics and structure of freshwater animal communities. Limnol. Oceanogr. 15:839-928; 1970.
141. Hargrave, B.T. The utilization of benthic microflora by *Hyalella azteca* (Amphipoda). J. Animal Ecol. 39:427-437; 1970.
142. Hargrave, B.T. Prediction of egestion by the deposit-feeding amphipod *Hyallella azteca*. Oikos 23:116-124; 1972.
143. Hargrave, B.T. Factors affecting the flow of organic matter to sediments in a marine bay. In: Tenore, K.R.; Coull, B.C., eds. Marine benthic dynamics. Columbia: Univ. of South Carolina Press; 1980:243-263.
144. Harrison, P.G.; Mann, K.H. Detritus formation from eelgrass (*Zostera marina* L.): the relative effects of fragmentation, leaching and decay. Limnol. Oceanogr. 20:924-934; 1975.
145. Haven, D.S.; Morales-Alamo, R. Aspects of biodeposition by oysters and other invertebrate filter-feeders. Limnol. Oceanogr. 11:487-498; 1966.
146. Haven, D.S.; Morales-Alamo, R. Occurrence and transport of faecal pellets in suspension in a tidal estuary. Sediment. Geol. 2:141-151; 1968.
147. Haven, D.S.; Morales-Alamo, R. Filtration of particles from suspension by the American oyster *Crassostrea virginica*. Biol. Bull. 139:248-264; 1970.
148. Haven, D.S.; Morales-Alamo, R. Biodeposition as a factor in sedimentation of fine suspended solids in estuaries. Geol. Soc. Am. Mem. 133:121-130; 1972.
149. Haven, D.S.; Morales-Alamo, R.; Kraueter, J.N. Sediment mixing by invertebrates as shown by Kr-85. Gloucester Point, VA: Virginia Institute of Marine Science; 1981; spec. sci. report no. 109; 29 p.

150. Hayne, D.W.; Ball, R.C. Benthic productivity as influenced by fish predation. Limnol. Oceanogr. 1:162-175; 1956.
151. Hayward, D.; Haas, L.W.; Boon, J.D.; Webb, K.L.; Friedland, K.D. Empirical models of stratification variation in the York River estuary, Virginia, U.S.A. In: Bowman, H.J.; Yentsch, C.M.; Peterson, W.T., eds. Tidal mixing and plankton dynamics. Lecture notes on coastal and estuarine studies, vol. 17. Berlin, Springer-Verlag; 1986:346-367.
152. Heck, K.L., Jr. Benthos. In: Heck, K.L, Jr., ed. Lecture notes on coastal and estuarine studies: ecological studies in the middle reach of Chesapeake Bay, Calvert Cliffs. Berlin, Springer-Verlag; 1987: 97-110.
153. Heinle, D.R.; Flemer, D.A. Flows of materials between poorly flooded tidal marshes and an estuary. Mar. Biol. 35:359-373; 1976.
154. Heinle, D.R.; Flemer, D.A. Carbon requirements of a population of the estuarine copepod *Eurytemora affinis*. Mar. Biol. 31:235-247; 1975.
155. Henriksen, K.; Kemp, W.M. Nitrification in estuarine and coastal sediments. In: Blackburn, T.H.; Sorensen, J., eds. Nitrogen cycling in coastal marine environments. New York: John Wiley & Sons; 1988:207-249.
156. Henriksen, K.; Hansen, J.I.; Blackburn, T.H. The influence of benthic infauna on exchange rates of inorganic nitrogen between sediment and water. Ophelia Suppl. 1:249-256; 1980.
157. Henriksen, K.; Rasmussen, M.B.; Jensen, A. Effect of bioturbation on microbial nitrogen transformations in the sediment and fluxes of ammonium and nitrate to the overlying water. Ecol. Bull. 35:193-205; 1983.
158. Hinde, P. Macrobenthic community structure in the vicinity of an oil refinery: a consideration of sampling design and statistical technique. Williamsburg, Virginia: College of William and Mary; 1981; 180 p. Thesis.
159. Hibbert, C.T. Biomass and production of a bivalve community on an intertidal mud-flat. J. Exp. Mar. Biol. Ecol. 25:249-261; 1976.
160. Hines, A.H.; Comtois, K.L. Vertical distribution of infauna in sediments of a subestuary of central Chesapeake Bay. Estuaries 8:296-304; 1985.
161. Hines, M.E.; Orem, W.H.; Lyons, W.B.; Jones, G.E. Microbial activity and bioturbation-induced oscillations in pore water chemistry of estuarine sediments in spring. Nature 299:433-435; 1982.
162. Hobbie, J.E.; Lee, C. Microbial production of extracellular material: importance in benthic ecology. In: Tenore, K.R.; Coull, B.C., eds. Marine benthic dynamics. Columbia: Univ. of South Carolina Press; 1980:341-346.
163. Holland, A.F. Long-term variation of macrobenthos in a mesohaline region of Chesapeake Bay. Estuaries 8:93-113; 1985.
164. Holland, A.F.; Zingmark, R.G.; Dean, J.M. Quantitative evidence concerning the stabilization of sediments by marine diatoms. Mar. Biol. 27:191-196; 1974.
165. Holland, A.F., Mountford; N.K.; Mihursky, J.A. Temporal variation in upper bay mesohaline benthic communities: I. The 9-m mud habitat. Ches. Sci. 18:370-378; 1977.
166. Holland, A.F.; Shaughnessy, A.T.; Hiegel, M.H. Long-term variation in mesohaline Chesapeake Bay macrobenthos: spatial and temporal patterns. Estuaries 10:227-245; 1987.
167. Holland, A.F.; Mountford, N.K.; Hiegel, M.H.; Kaumeyer, K.R.; Mihursky, J.A. Influence of predation on infaunal abundance in upper Chesapeake Bay, USA. Mar. Biol. 57:221-235; 1980.
168. Holland, A.F.; Shaughnessy, A.T.; Scott, L.C.; Dickens, V. A.; Gerritsen, J.; Ranasinghe, J.A. Long-term benthic monitoring and assessment program for the Maryland portion of Chesapeake Bay: Interpretive report. Annapolis, MD: Maryland Dept. Nat. Resources, Power Plant Res. Prog.; 1988a.
169. Holland, A.F.; Shaughnessy, A.T.; Scott, L.C.; Dickens, V.A.; Ranasinghe, J.A.; Summers, J.K. Progress report: long-term benthic monitoring and assessment program for the Maryland portion of Chesapeake Bay (July 1986-October 1987). Rpt. to Maryland Power Plant Res. Prog. and Maryland Dept. Environ. Columbia, MD: Versar Inc.; 1988b; Rpt. No. PPRP-LTB/EST-88-11988.
170. Howard, S.; Little, D.I. Effect of infaunal burrow structure on oil penetration into sediments. Proceedings of 1987 oil spill conference, 10th biennial. USCG, API, EPA; 1987:427-431.
171. Howard, J.D.; Elders, C.A. Burrowing patterns of haustoriid amphipods from Sapelo Island, Georgia. In: Crimes, J.P.; Harper, J.C., eds. Trace fossils. Liverpool, England: Harper; 1970.
172. Huggett, D.V. A trophic resource analysis of dominant benthic macrofauna of the lower Chesapeake Bay. Williamsburg, VA: College of William and Mary; 1987; 86 p. Thesis.
173. Hunt, C.D.; Smith, D.L. Remobilization of metals from polluted marine sediments. Can. J. Fish. Aquat. Sci. 40(Suppl. 2):132-142; 1983.
174. Hylleberg, J.; Galucci, V.F. Selectivity in feeding by the deposit-feeding bivalve *Macoma nasuta*. Mar. Biol. 32:167-178; 1975.
175. Hylleberg, J.; Henriksen, K. The central role of bioturbation in sediment mineralization and element recycling. Ophelia Suppl. 1:1-16; 1980.
176. Jenkins, M.C.; Kemp, W.M. The coupling of nitrification and denitrification in two estuarine sediments. Limnol. Oceanogr. 29:609-619; 1984.

177. Johnson, M.G.; Brinkhurst, R.O. Production of benthic macroinvertebrates of Bay of Quinte and Lake Ontario. J. Fish. Res. Bd. Can. 28:1699-1714; 1971.
178. Jordan, R.A.; Sutton, C.E. Oligohaline benthic invertebrate communities at two Chesapeake Bay power plants. Estuaries 7:192-212; 1984.
179. Jumars, P.A.; Nowell, A.R.M. Effects of benthos on sediment transport: difficulties with functional grouping. Cont. Shelf Res. 3:115-130; 1984.
180. Jumars, P.A.; Nowell, A.R.M.; Self, R.F.L. A simple model of flow-sediment-organism interaction. Mar. Geol. 42:155-172; 1981.
181. Juniper, S.K. Stimulation of bacterial activity by a deposit feeder in two New Zealand intertidal inlets. Bull. Mar. Sci. 31:691-701; 1981.
182. Karickhoff, S.W.; Morris, K.R. Impact of tubificid oligochaetes on pollutant transport in bottom sediments. Environ. Sci. Tech. 19:51-56; 1985
183. Kautsky, N.; Evans, E. Role of biodeposition by *Mytilus edulis* in the circulation of matter and nutrients in a Baltic coastal ecosystem. Mar. Ecol. Prog. Ser. 38:210-212; 1987.
184. Kemp, W.M. Benthic-pelagic coupling: potential importance for fisheries in Chesapeake Bay. In: Houde, E.D., ed. Long-range research needs for Chesapeake Bay living resources, Workshop Rpt., Port Deposit, Maryland. CEES tech. ser. no. TS61-87; 1987.
185. Kemp, W.M.; Boynton, W.R. External and internal factors regulating metabolic rates of an estuarine benthic community. Oecologia 51:19-27; 1981.
186. Kikuchi, E. Contribution of the polychaete *Neanthes japonica* (Izuka) to the oxygen uptake and carbon dioxide production of an intertidal mudflat of the Nanakita River estuary, Japan. J. Exp. Mar. Biol. Ecol. 97:81-93; 1986.
187. Kimball, S.M.; Schaffner, L.C.; Hobbs, C.H. Geotechnical and benthic evaluation of sand resources in the lower Chesapeake Bay, Report 1: Thimble Shoal. Final Report to City of Hampton, Virginia and Council on the Environment; VA Coastal Resources Management Program; 1989; 46 p.
188. Kirkley, J.E. A socio-economic overview of the Chesapeake Bay fisheries. In: Majumdar, S.K.; Hall, L.W.; Austin, H.M. eds. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: The Pennsylvania Academy of Science; 1987:54-62.
189. Klein-Breteler, W.C.M. Food consumption, growth and energy metabolism of juvenile shore crabs, *Carcinus maenas*. Neth. J. Sea Res. 9:255-272; 1975.
190. Klump, J.V.; Martens, C.S. Biogeochemical cycling in an organic rich coastal marine basin. II. Nutrient sediment-water exchange processes. Geochim. Cosmochim. Acta 45:101-121; 1981.
191. Koike, I.; Mukai, H. Oxygen and inorganic nitrogen exchange in burrows of the shrimps *Callianassa japonica* and *Upogebia major*. Mar. Ecol. Prog. Ser. 12:185-190; 1983.
192. Kraeuter, J.; Haven, D.S. Fecal pellets of common invertebrates of lower York River and lower Chesapeake Bay, Virginia. Ches. Sci. 11:159-173; 1970.
193. Krantzberg, G. The influence of bioturbation on physical, chemical and biological parameters in aquatic environments: a review. Environ. Poll. (Series A) 39:99-122; 1985.
194. Krantzberg, G.; Stokes, P.M. Benthic macroinvertebrates modify copper and zinc partitioning in freshwater-sediment microcosms. Can. J. Fish. Aquat. Sci. 42:1465-1473; 1985.
195. Kravitz, M.J. Niche relations among polychaetes of a meso-polyhaline assemblage of the York River, Virginia. Williamsburg, VA: The College of William and Mary; 1983. Thesis. 113 p.
196. Krebs, C.J. Ecology: the experimental analysis of distribution and abundance. 2nd edition. London, England; Harper and Row; 1978:549-557.
197. Kristensen, E. Ventilation and oxygen uptake by three species of *Nereis* (Annelida:Polychaeta). II. Effects of temperature and salinity changes. Mar. Ecol. Prog. Ser. 12:299-306; 1983.
198. Kristensen, E. Effect of natural concentrations on nutrient exchange between a polychaete burrow in estuarine sediment and the overlying water. J. Exp. Mar. Biol. Ecol. 75:171-190; 1984.
199. Kristensen, E. Oxygen and inorganic nitrogen exchange in a *Nereis virens* (Polychaeta) bioturbated sediment-water system. J. Coast. Res. 1:109-116; 1985.
200. Kristensen, E. Benthic fauna and biogeochemical processes in marine sediments: microbial activities and fluxes. In: Blackburn, T.H.; Sorensen, J., eds. Nitrogen cycling in coastal marine environments. New York: John Wiley & Sons; 1988:275-299.
201. Kristensen, E.; Blackburn, T.H. The fate of organic carbon and nitrogen in experimental marine sediment systems: influence of bioturbation and anoxia. J. Mar. Res. 45:231-257; 1987.
202. Kristensen, E.; Jensen, M.K.; Andersen, T.K. The impact of polychaete (*Nereis virens* Sars) burrows on nitrification and nitrate reduction in estuarine sediments. J. Exp. Mar. Biol. Ecol. 85:75-91; 1985.
203. Krueger, F.E. Benthic organisms study. Rpt. to Potomac Electric Power Co. Philadelphia: Academy of Natural Sciences; 1979.
204. Kuipers, B.R.; deWilde, P.A.J.W.; Creutzberg, F. Energy flow in a tidal flat ecosystem. Mar. Ecol. Prog. Ser. 5:215-221; 1981.
205. Lee, H.; Swartz, R.C. Biological processes affecting

the distribution of pollutants in marine sediments. Part II. Biodeposition and bioturbation. In: Baker, R.A., ed. Contaminants and sediments, vol. 2. Michigan: Ann Arbor Science; 1980:555-606.

206. Levinton, J.S.; Stewart, S. The effects of sediment organics, detrital input, and temperature on demography, production, and body size of a deposit feeder. Mar. Ecol. Prog. Ser. 49:259-266; 1988.
207. Lindeman, R.L. The trophic-dynamic aspect of ecology. Ecology 23:399-418. 1942.
208. Lipcius, R.N.; Hines, A.H. Variable functional responses of a marine predator in dissimilar homogeneous microhabitats. Ecology 67:1361-1371; 1986.
209. Lippson, A.J.; Haire, M.S.; Holland, A.F.; Jacobs, F.; Jensen, J.; Moran-Johnson, R.L.; Polgar, T.T.; Richkus, W.A. Environmental atlas of the Potomac Estuary. Maryland Dept. Nat. Resources, Power Plant Res. Prog., Annapolis, Maryland; 1979.
210. Llanso, R.J. Macrobenthic community structure at the entrance to the Chesapeake Bay. Williamsburg, VA: The College of William and Mary: 1985. 119 p. Thesis
211. Loi, T.N.; Wilson, B.J. Macroinfaunal structure and effects of thermal discharges in a mesohaline habitat of Chesapeake Bay, near a nuclear power plant. Mar. Biol. 55:3-16; 1979.
212. Lopez, G.R. Comparative ecology of the macrofauna of freshwater and marine muds. Limnol. Oceanogr. 33:946-962; 1988.
213. Lopez, G.R.; Levinton, J.S. The availability of microorganisms attached to sediment particles as food for *Hydrobia ventrosa* (Montagu) (Gastropoda:Prosobranchia). Oecologia 32:263-275; 1978.
214. Lopez, G.R.; Levinton, J.S. Ecology of deposit-feeding animals in marine sediments. Quart. Rev. Biol. 62:235-260; 1987.
215. Lopez, G.R.; Levinton, J.S.; Slobodkin, L.B. The effect of grazing by the detritivore *Orchestria grillus* on *Spartina* litter and its associated microbial community. Oecologia 30:111-127; 1977.
216. Luckenbach, M.W. Sediment stability around animal tubes: the roles of hydrodynamic processes and biotic activity. Limnol. Oceanogr. 31:779-787; 1986.
217. Luckenbach, M.W.; Huggett, D.V.; Zobrist, E.C. Sediment transport, biotic modifications and selection of grain size in a surface deposit-feeder. Estuaries 11:134-139; 1988.
218. Lunz, J.D.; Kendall, D.R. Benthic resources assessment technique, a method for quantifying the effects of benthic community changes on fish resources. Oceans '82, Mar. Poll. Session, Washington, D.C. 1982: 1021-1027.
219. Lynch, M.P.; Harrison, W. Sedimentation caused by a tube-building amphipod. J. Sediment. Petrol. 40:434-435; 1970.
220. Lyons, W.B.; Loder, T.C.; Murray, S.M. Nutrient pore water chemistry, Great Bay, New Hampshire: benthic fluxes. Estuaries 5:230-233; 1982.
221. Mackiernan, G.B., ed. Dissolved oxygen in the Chesapeake Bay, processes and effects. Proc. Sem. Hypoxic and Related Processes in Chesapeake Bay UM-SG-TS-87-03:177; 1987.
222. MacKinnon, J.C. Analysis of energy flow in an unexploited marine flatfish population. J. Fish. Res. Bd. Can. 30:1717-1728; 1973.
223. Malone, T.C.; Kemp, W.M.; Ducklow, H.W.; Boynton, W.R.; Tuttle, J.H.; Jonas, R.B. Lateral variation in the production and fate of phytoplankton in a partially stratified estuary. Mar. Ecol. Prog. Ser. 32:149-160; 1986.
224. Mangum, C.P. Activity patterns in metabolism and ecology of polychaetes. Comp. Biochem. Physiol. 11:239-256; 1964.
225. Mangum, C.P.; Burnett, L.E. The extraction of oxygen by estuarine invertebrates. In: Vernberg, F.J., ed. Physiological ecology of estuarine organisms. Columbia: Univ. of South Carolina Press; 1975:147-163.
226. Mangum, C.P.; Van Winkle, W. Responses of aquatic invertebrates to declining oxygen conditions. Am. Zool. 13:529-541; 1973.
227. Mangum, C.P.; Santos, S.L.; Rhodes, W.R. Distribution and feeding in the onuphid polychaete, *Diopatra cuprea* (Bosc). Mar. Biol. 2:33-40; 1968.
228. Mann, K.H. Production and use of detritus in various freshwater, estuarine, and coastal marine ecosystems. Limnol. Oceanogr. 33:910-930; 1988.
229. Markle, D.E.; Grant, G.C. The summer food habits of young-of-the year striped bass in three Virginia rivers. Ches. Sci. 11:50-54; 1970.
230. Matisoff, G. Mathematical models of bioturbation. In: McCall, P.L.; Tevesz, M.J.S., eds. Animal-sediment relations. New York: Plenum Press; 1982:289-330.
231. Matisoff, G.; Fisher, J.B.; Matis, S. Effects of benthic macroinvertebrates on the exchange of solutes between sediments and freshwater. Hydrobiologia 122:19-33; 1985.
232. McCaffrey, R.J.; Myers, A.C.; Davey, E.; Morrison, G.; Bender, M.; Luedtke, N.; Cullen, D.; Froelich, P.; Klinkhammer, G. The relation between pore water chemistry and benthic fluxes of nutrients and manganese in Narragansett Bay, Rhode Island. Limnol. Oceanogr. 25:31-44; 1980.
233. McCall, P.L. The effects of deposit feeding oligochaetes on particle size and settling velocity of Lake Erie sediments. J. Sediment. Petrol. 49:813-818; 1979.
234. McCall, P.L.; Fisher, J.B. Effects of tubificid oligochaetes on physical and chemical properties of Lake Erie sediments. In: Brinkhurst, R.O.; Cook, D.G., eds.

Aquatic oligochaete biology. New York: Plenum Press; 1980:253-317.

235. McCall, P.L.; Tevesz, M.J.S. The effects of benthos on physical properties of freshwater sediments. In: McCall, P.L.; Tevesz, M.J.S., eds. Animal-sediment relations. New York: Plenum Press; 1982:105-176.
236. McCall, P.L.; Tevesz, M.J.S.; Schwelgien, S.F. Sediment mixing by *Lampsilis radiata siliquoidea* (Mollusca) from western Lake Erie. J. Great Lakes Res. 5:105-111; 1979.
237. McCall, P.L.; Matisoff, G.; Tevesz, M.J.S. The effects of a unionid bivalve on the physical, chemical and microbial properties of cohesive sediments from Lake Erie. Am. J. Sci. 1986.
238. McCave, I.N. The benthic boundary layer. New York: Plenum Press; 1974.
239. McCave, I.N. Biological pumping upwards of the coarse fraction of deep-sea sediments. J. Sediment. Petrol. 58:148-158; 1988.
240. McDermott, J.J. Observations on the food and feeding behavior of estuarine nemertean worms belonging to the order Hoplonemertea. Biol. Bull. 150:57-68; 1976.
241. Mills, E.L. The biology of an ampeliscid amphipod crustacean sibling species pair. J. Fish. Res. Bd. Canada. 24:305-355; 1967.
242. Moller, P.; Pihl, L.; Rosenberg, R. Benthic faunal energy flow and biological interaction in some shallow marine soft bottom habitats. Mar. Ecol. Prog. Ser. 27:109-121; 1985.
243. Moore, D.G.; Scruton, P.C. Minor internal structures of some recent unconsolidated sediments. Bull. Am. Assoc. Petrol. Geol. 41:2723-2751; 1957.
244. Mountford, K. Aspects of the ecology of a small estuarine embayment. Mar. Biol. 61:53-67; 1980.
245. Mountford, N.K.; Holland, A.F.; Mihursky. J.A. Identification and description of macrobenthic communities in the Calvert Cliffs region of the Chesapeake Bay. Ches. Sci. 14:360-369; 1977.
246. Muzii, E.D. A coating membrane in the calcareous tube of serpulids. Publ. Staz. Zool. Napoli 36:135-137; 1968.
247. Myers, A.C. Sediment processing in a marine subtidal sandy bottom community: I. Physical aspects. J. Mar. Res. 35:609-632; 1977.
248. Nakamura, M.; Yamamuro, M.; Ishikawa, M.; Nishimura, H. Role of the bivalve *Corbicula japonica* in the nitrogen cycle in a mesohaline lagoon. Mar. Biol. 99:369-374; 1988.
249. Newell, R. Ecological changes in Chesapeake Bay: are they the result of overharvesting the American oyster, *Crassostrea virginica*? In: Lynch, M.P.; Krome, E.C., eds. Understanding the estuary: advances in Chesapeake Bay research; proceedings of a conference; Baltimore, Maryland. Solomons, MD: Chesapeake Res. Consor.; 1988:536-546.
250. Nichols, F.H. Dynamics and energetics of three deposit-feeding benthic invertebrate populations in Puget Sound, Washington. Ecol. Monogr. 45:57-82; 1975.
251. Nichols, F.H. Dynamics and production of *Pectinaria koreni* (Malmgren) in Kiel Bay, West Germany. In: Keegan, B.F.; Ceidigh, P.O.; Boaden, P.J.S., eds. Biology of benthic organisms. Oxford, England: Pergamon Press; 1977:453-463.
252. Nilsen, K.N.; Diaz, R.J.; Schaffner, L.C.; Boesch, D.F.; Bertelsen, R.; Kravitz, M. The biogenic structure of lower Chesapeake Bay sediments. Final Report to US-EPA. 1982; Chesapeake Bay Program R805981-01-0.
253. Nixon, S.W. Remineralization and nutrient cycling in coastal marine ecosystems. In: Neilson, B.J.; Cronin, L.E., eds. Estuaries and nutrients. New Jersey: Humana Press; 1981b:111-138.
254. Nixon, S.W.; Oviatt, C.A.; Hale, S.S. Nitrogen regeneration and the metabolism of coastal marine bottom communities. In: Anderson, J.M.; MacFayden, A., eds. The role of terrestrial and aquatic organisms in decomposition processes. London, England: Blackwell Scientific; 1976:269-283.
255. Nixon, S.W.; Kelly, J.R.; Furnas, B.N.; Oviatt, C.A. Phosphorus regeneration and the metabolism of coastal marine bottom communities. In: Tenore, K.R.; Coull, B.C., eds. Marine benthic dynamics. Columbia: Univ. of South Carolina Press; 1980:219-242.
256. Nowell, A.R.M.; Jumars, P.A.; Eckman, J.E. Effects of biological activity on the entrainment of marine sediments. Mar. Geol. 42:133-153; 1981.
257. Odum, E.P. The strategy of ecosystem development. Science 164:262-270; 1969.
258. Odum, H.T. Systems ecology. New York: John Wiley & Sons; 1983.
259. Officer, C.B.; Smayda, T.J.; Mann, R. Benthic filter-feeding: a natural eutrophication control. Mar. Ecol. Prog. Ser. 9:203-210; 1982.
260. Officer, C.B.; Biggs, R.B.; Taft, J.L.; Cronin, L.E.; Tyler, M.A.; Boynton, W.R. Chesapeake Bay anoxia. Origin, development and significance. Science 223:22-27; 1984.
261. Oliver, J.S.; Oakden, J.M.; Slattery, P.N. Phoxocephalid amphipod crustaceans as predators on larvae and juveniles in marine soft-bottom communities. Mar. Ecol. Prog. Ser. 7:179-184; 1982.
262. Olsen, C.R.; Simpson, H.J.; Peng, T.H.; Bopp, R.F.; Trier, R.M. Sediment mixing and accumulation rate effects on radionuclide depth profiles in Hudson estuary sediments. J. Geophys. Res. 86:11020-11028; 1981.
263. Olsen, C.R.; Cutshall, N.H.; Larsen, I.L. Pollutant-particle associations and dynamics in coastal marine

environments: a review. Mar. Chem. 11:501-533; 1982.

264. Orth, R.J.; Heck, K.L.; Diaz, R.J. Littoral and intertidal systems in the Mid-Atlantic coast of the United States. Ecosystems of the world. New York: Elsevier Scientific Publishing Company (in press).
265. Paine, R.T. Food web complexity and species diversity. Am. Natur. 100:65-75; 1966.
266. Paine, R.T. A note on trophic complexity and community stability. Am. Natur. 103:91-93; 1969.
267. Parsons, T.R.; Takahashi, M.; Hargrave, B. Biological oceanographic processes, 2nd ed. New York: Pergamon Press; 1977.
268. Pearson, T.H.; Barnett, P.R.O. Long-term changes in benthic populations in some West European coastal areas. Estuaries 10:220-226; 1987.
269. Pearson, T.H.; Rosenberg, R. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Ann. Rev. 16:229-311; 1978.
270. Perkins, E.J. The biology of estuaries and coastal waters. London: Academic Press; 1974.
271. Pfitzenmyer, H.T. Benthos. A biological study of Baltimore harbor. Center for Environmental and Estuarine Studies, Univ. of Maryland. Spec. rpt. no. 6; 1975.
272. Pickral, J.C.; Odum, W.E. Benthic detritus in a saltmarsh tidal creek. In: Wiley, M., ed. Estuarine processes. New York: Academic Press; 1976:280-292.
273. Pihl, L. Exposure, vegetation and sediment as primary factors for mobile epibenthic faunal community structure and production in shallow marine soft bottom areas. Neth. J. Sea Res. 20:75-83; 1986.
274. Pihl, L. Food selection and consumption of mobile epibenthic fauna in shallow marine areas. Mar. Ecol. Prog. Ser. 22:169-179; 1985.
275. Prokopovich, N.P. Deposition of clastic sediments by clams. J. Sediment. Petrol. 39:891-901; 1969.
276. Pryor, W.A. Biogenic sedimentation and alteration of argillaceous sediments in shallow marine environments. Geol. Sco. Am. Bull. 86:1244-1254; 1975.
277. Rainer, S.F. Trophic structure and production in the macrobenthos of a temperate Australian estuary. Estuar. Coast. Shelf Sci. 15:423-441; 1982.
278. Rainer, S.F. Polychaeta as a representative taxon: their role in function and secondary production of benthic communities in three estuaries. In: Hutchings, P.A., ed. Proceedings of the 1st international polychaete conference, Sydney. The Linnean Society of New South Wales; 1984:370-382.
279. Rainer, S.F.; Fitzhardinge, R.C. Benthic communities in an estuary with periodic deoxygenation. Australian J. Mar. Freshwat. Res. 32:227-243; 1981.
280. Reinharz, E.; O'Connell, A.E. Animal-sediment relationships of the upper and central Chesapeake Bay. Final Report to USEPA; 1981; 71 p.
281. Reinharz, E.; Nilsen, K.J.; Boesch, D.F.; Bertelsen, R.; O'Connell, A.E. A radiographic examination of physical and biogenic sedimentary structures in Chesapeake Bay. Maryland Geological Survey; 1982; 58 p. Rpt. investig. no. 36.
282. Reise, K. Tidal flat ecology. New York: Springer-Verlag; 1985.
283. Revsbech, N.P.; Jorgensen, B.B.; Blackburn, T.H. Oxygen in the sea bottom measured with a microelectrode. Science 207:1355-1356; 1980a.
284. Revsbech, N.P.; Sorensen, J.; Blackburn, T.H. Distribution of oxygen in marine sediments measured with microelectrodes. Limnol. Oceanogr. 25:403-411; 1980b.
285. Reynoldson, T.B. Interactions between sediment contaminants and benthic organisms. Hydrobiologia 149:53-66; 1987.
286. Rhoads, D.C. Biogenic reworking of intertidal and subtidal sediments in Barnstable Harbor and Buzzards Bay, Massachusetts. J. Geol. 75:461-476; 1967.
287. Rhoads, D.C. Organism-sediment relations on the muddy sea floor. Oceanogr. Mar. Biol. Ann. Rev. 12:263-300; 1974.
288. Rhoads, D.C.; Boyer, L.F. The effects of marine benthos on physical properties of sediments: a successional perspective. In: McCall, P.L.; Tevesz, M.J.S., eds. Animal-sediment relations. New York: Plenum Press; 1982:3-52.
289. Rhoads, D.C.; Stanley, D.J. Biogenic graded bedding. J. Sediment. Petrol. 35:956-963; 1965.
290. Rhoads, D.C.; Young, D.K. The influence of deposit-feeding organisms on sediment stability and community trophic structure. J. Mar. Res. 28:151-178; 1970.
291. Rhoads, D.C.; McCall, P.L.; Yingst, J.Y. Disturbance and production on the estuarine seafloor. Scient. Am. 66:577-586; 1978.
292. Rice, D.L. Early diagenesis in bioadvective sediments: relationships between diagenesis of beryllium-7, sediment reworking rates and the abundance of conveyor-belt deposit-feeders. J. Mar. Res. 44:149-184; 1986.
293. Rice, D.L.; Whitlow, S.I. Early diagenesis of transition metals: a study of metal partitioning between macrofaunal populations and shallow sediments. The fate and effects of pollutants, Proceedings of a Maryland Seagrant Symposium; 1985a:21-31. Maryland Seagrant Publication UM-SG-TS-85-02.
294. Rice, D.L.; Whitlow, S.I. Diagenesis of transition metals in bioadvective marine sediments. Proceedings of the international conference on heavy metals in the environment, Athens, Greece. 1985b.
295. Rice, D.L.; Tenore, K.R.; Windom, H.L. The effect of detritus ration on metal transfer to deposit-feeding benthos. Mar. Ecol. Prog. Ser. 5:135-140; 1981.

296. Riedel, G.F.; Sanders, J.G.; Osman, R.W. The role of three species of benthic invertebrates in the transport of arsenic from contaminated estuarine sediment. J. Exp. Mar. Biol. Ecol. (in press).
297. Riedel, G.F.; Sanders, J.G.; Osman, R.W. The effect of biological and physical disturbances on the transport of arsenic from contaminated sediments. Est. Coast.Shelf Sci. 25:693-706; 1987.
298. Riemann, F.; Schrage, M. The mucus-trap hypothesis on feeding of aquatic nematodes and implications for biodegradation and sediment texture. Oecologia 34:75-88; 1978.
299. Risk, M.J.; Moffat, J.S. Sedimentological significance of fecal pellets of *Macoma balthica* in the Minas Basin, Bay of Fundy. J. Sediment. Petrol. 47:1425-1436; 1977.
300. Robbins, J.A. A model for particle-selective transport of tracers in sediments with conveyor belt deposit feeders. J. Geophys. Res. 91:8542-8558; 1986.
301. Robbins, J.A.; McCall, P.L.; Fisher, J.B.; Krezoski, J.R. Effect of deposit feeders on migration of Cs-137 in lake sediments. Earth Plant. Sci. Let. 42:277-287; 1979.
302. Roberts, H.H.; Wiseman, W.J. Lagoon sediment transport: the significant effect of *Callianassa* bioturbation. Proceedings of the 4th International Coral Reef Symposium; Manila; 1981:459-465.
303. Robertson, A.I. The relationship between annual production:biomass ratios and life spans for marine macrobenthos. Oecologia 40:193-202; 1979.
304. Ronan, T.E. Food-resources and the influence of spatial pattern on feeding in the phoronid *Phoronopsis viridis*. Biol. Bull. 154:472-484; 1978.
305. Rosenberg, R.; Olsson, I.; Olundh, E. Energy flow model of an oxygen-deficient estuary on the Swedish West coast. Mar. Biol. 42:99-107; 1977.
306. Sanders, H.L. Oceanography of Long Island Sound 1952-1954. X. The biology of marine bottom communities. Bull. Bingham Oceanogr. Coll. 15:345-414; 1956.
307. Sanders, H.L. Benthic studies in Buzzards Bay. III. The structure of soft-bottom communities. Limnol. Oceanogr. 5:138-153; 1960.
308. Sanders, J.G.; Riedel, G.F. Chemical and physical processes influencing bioavailability of toxics in estuaries. In: Lynch, M.P.; Krome, E.C., eds. Perspectives on the Chesapeake Bay: recent advances in estuarine science. Solomons, MD: Chesapeake Research Consortium; 1987:87-106; CRC Pub. No. 127.
309. Santschi, P.H.; Amdurer, M.; Adler, D.; O'Hara, P.; Li, Y.H.; Doering, P. Relative mobility of radioactive trace elements across the sediment-water interface in the MERL model ecosystems of Narragansett Bay. J. Mar. Res. 45:1007-1048; 1987.
310. Sayama, M.; Kurihara, Y. Relationship between burrowing activity of the polychaetous annelid, *Neanthes japonica* (Izuka) and nitrification-denitrification processes in the sediments. J. Exp. Mar. Biol. Ecol. 72:223-241; 1983.
311. Schaffner, L.C. Ecology of the benthos of the lower Chesapeake Bay. Williamsburg, VA: The College of William and Mary; 1987. 150 p. Dissertation.
312. Schaffner, L.C. Patterns of association and species diversity: evidence for positive interactions in an estuarine macrobenthic community. Mar. Ecol. Prog. Ser. (in press).
313. Schaffner, L.C.; Diaz, R.J. Distribution and abundance of overwintering blue crabs, *Callinectes sapidus*, in the lower Chesapeake Bay. Estuaries 11:68-72; 1988.
314. Schaffner, L.C.; Diaz, R.J.; Byrne, R.J. Processes affecting recent estuarine stratigraphy. Coastal Sediments '87, Waterways Div., ASCE, New Orleans; 1987:584-599.
315. Schaffner, L.C.; Diaz, R.J.; Olsen, C.R.; Larsen, I.L. Faunal characteristics and sediment accumulation processes in the James River estuary, Virginia. Estuar. Coast. Shelf Sci. 25:211-226; 1987.
316. Schiffelbein, P. Effect of benthic mixing on the information content of deep-sea stratigraphic signals. Nature 311:651-652; 1984.
317. Schwinghamer, P.; Hargrave, B. Partitioning of production and respiration among size groups of organisms in an intertidal benthic community. Mar. Ecol. Prog. Ser. 31:131-142; 1986.
318. Self, R.F.L.; Jumars, P.A. Cross-phyletic patterns of particle selection by deposit feeders. J. Mar. Res. 46:119-143; 1988.
319. Self, R.F.L.; Nowell, A.R.M.; Jumars, P.A. Factors controlling critical shears for deposition and erosion of individual grains. Mar. Geol. 86:181-199; 1989.
320. Sellner, K.G. Phytoplankton in the Chesapeake Bay: Role in carbon, oxygen and nutrient dynamics. In: Majumdar, S.K.; Hall, L.W.; Austin, H.M., eds. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: The Pennsylvania Academy of Science; 1987:134-157.
321. Sellmer, G.P. Functional morphology and ecological life history of the gem clam, *Gemma gemma* (Eulamellibranchia: Veneridae). Malacologia 5:137-223; 1967.
322. Sellner, K.G.; Kachur, M.E. Phytoplankton: distribution, production and integrators of environmental conditions. In: Heck, K.L., Jr., ed. Lecture notes on coastal and estuarine studies: ecological studies in the middle reach of Chesapeake Bay, Calvert Cliffs. Berlin: Springer-Verlag; 1987:12-37.
323. Setzler-Hamilton, E.M. Utilization of Chesapeake Bay

by early life history stages of fishes. In: Majumdar, S.K.; Hall, L.W.; Austin, H.M., eds. Contaminant problems and management of living Chesapeake Bay resources. Philadelphia: The Pennsylvania Academy of Science; 1987:63-93.

324. Shumway, S.E. Factors affecting oxygen consumption in the coot clam *Mulinia lateralis* (Say). Ophelia 22:143-171; 1983.
325. Sikora, W.B.; Kjerfve, B. Factors influencing the salinity regime of Lake Pontchartrain, Louisiana, a shallow coastal lagoon: analysis of a long-term data set. Estuaries 8:170-180; 1985.
326. Sissenwine, M.P.; Cohen, E.B.; Grosslein, M.D. Structure of the Georges Bank ecosystem. Rapp. P. -v. Reun. Cons. Int. Explor. Mer 183:243-254; 1984.
327. Smetacek, V.; von Brockel, K.; Zeitzschel, B.; Zenk, W. Sedimentation of particulate matter during a phytoplankton spring bloom in relation to the hydrographical regime. Mar. Biol. 47:211-226; 1978.
328. Speir, H.J. Status of some finfish stocks in the Chesapeake Bay. Water Air Soil Poll. 35:49-62; 1987.
329. Stanley, S.M. Relation of shell from to life habits of the Bivalvia (Mollusca). Geol. Soc. Am. Mem. 125:1-296; 1970.
330. Swartz, R.C.; Lee, H. Biological processes affecting the distribution of pollutants in marine sediments. Part 1. Accumulation, trophic transfer, biodegradation and migration. Contaminants and sediments, vol. 2:533-553; 1980.
331. Taghon, G.L.; Nowell, A.R.M.; Jumars, P.A. Transport and breakdown of fecal pellets: biological and sedimentological consequences. Limnol. Oceanogr. 29:64-72; 1984.
332. Teal, J.M. Community metabolism in a temperate cold spring. Ecol. Monogr. 27:283-302; 1957.
333. Tenore, K.R. Growth of *Capitella capitata* on various levels of detritus derived from different sources. Limnol. Oceanogr. 22:936-941; 1977.
334. Tenore, K.R.; Horton, D.B.; Duke, T.W. Effects of bottom substrate on the brackish water bivalve *Rangia cuneata*. Ches. Sci. 9:238-248; 1968.
335. Tenore, K.R.; Rice, D.L. A review of trophic factors affecting secondary production of deposit-feeders. In: Tenore, K.R.; Coull, B.C., eds. Marine benthic dynamics. Columbia: Univ. of South Carolina Press; 1980:325-340.
336. Tenore, K.R.; Hanson, R.B.; McClain, J.; Maccubbin, A.E.; Hodson, R.E. Changes in composition and nutritional value to a benthic deposit feeder of decomposing detritus pools. Bull. Mar. Sci. 35:299-311; 1984.
337. Tenore, K.R.; Tietjen, J.H.; Lee, J.J. Effect of meiofauna on incorporation of aged eelgrass, *Zostera marina*, detritus by the polychaete *Nephthys incisa*. J. Fish. Res. Bd. Can. 34:563-567; 1977.
338. Tevesz, M.J.S.; Soster, F.M.; McCall, P.L. The effects of size-selective feeding by oligochaetes on the physical properties of river sediments. J. Sediment. Petrol. 50:561-568; 1980.
339. Tortellotte, G.H.; Dauer, D.M. Macrobenthic communities of the lower Chesapeake Bay. II. Lynnhaven Roads, Lynnhaven Bay, Broad Bay and Linkhorn Bay. Int. Rev. ges. Hydrobiol. 68:59-72; 1983.
340. U. S. Army Corps of Engineers. Chesapeake Bay future conditions report. Vol. 3. Baltimore, MD: Baltimore District Corps of Engineers; 1978.
341. U. S. Army Corps of Engineers. Chesapeake Bay future conditions report. Vol 9. Baltimore, MD: Baltimore District Corps of Engineers; 1978.
342. U. S. Environmental Protection Agency. Chesapeake Bay Program technical studies: a synthesis. Washington, D.C.: USEPA; 1982: 635 p.
343. Virginia Institute of Marine Science. A study of the effects of dredging and dredge spoil disposal on the marine environment. Spec. scient. rpt. no. 8. Gloucester Pt., VA: Virginia Institute of Marine Science, College of William and Mary; 1967.
344. Valiela, I. Marine ecological processes. New York: Springer-Verlag; 1984.
345. Verity, P.G. Factors driving changes in the pelagic trophic structure of estuaries with implications for the Chesapeake Bay. In: Lynch, M.P.; Krome, E.C., eds. Perspectives on the Chesapeake Bay: advances in estuarine sciences. Solomons, MD: Chesapeake Research Consortium; 1987: 35-56; Pub. No. 127.
346. Verwey, J. On the ecology of distribution of cockle and mussel in the Dutch Wadden Sea, their role in sedimentation and the source of their food supply, with a short review of the feeding behavior of bivalve molluscs. Arch. Nederl. Zool. 10:172-239; 1952.
347. Virnstein, R.W. The importance of predation of crabs and fishes on benthic infauna in Chesapeake Bay. Ecology 58:1199-1217; 1977.
348. Virnstein, R.W. Predation on estuarine infauna: response patterns of component species. Estuaries 2:69-86; 1979.
349. Vogel, S.; Bretz, W.L. Interfacial organisms: passive ventilation in the velocity gradients near surfaces. Science 175:210; 1972.
350. Warinner, J.E.; Brehmer, M.L. The effects of thermal effluents on marine organisms. Inter. J. Air Water Poll. 10:277-289; 1966.
351. Warwick, R.M. Population dynamics and secondary production of benthos. In: Tenore, K.R.; Coull, B.C., eds. Marine benthic dynamics. Columbia: Univ. of South Carolina Press; 1980:1-24.
352. Wass, E. A check list of the biota of lower Chesapeake Bay and the Virginian Sea. Gloucester Pt., VA: Virginia Institute of Marine Science: 1972; 290 p.;

Spec. scient. rpt. no. 65.

353. Waters, T.F. Secondary production in inland waters. Adv. Ecol. Res. 14:91-164; 1977.

354. Webb, K.L.; DuPaul, W.D.; D'Elia, C.F. Biomass and nutrient flux measurements on *Holothuria atra* populations on windward reef flats at Enewetak, Marshall Islands. Proceedings 3rd international coral reef symposium, Univ. of Miami, Florida; 1977:409-415.

355. Whitlatch, R.B. Patterns of resource utilization and coexistence in marine intertidal deposit-feeding communities. J. Mar. Res. 38:743-765; 1980.

356. Wiegert, R.G. The past, present and future of ecological energetics. In: Pomeroy, L.R.; Alberts, J.J., eds. Concepts of ecosystem ecology. New York: Springer-Verlag; 1988:29-55.

357. Williams, P.J. Incorporation of microheterotrophic processes into the classical paradigm of the plankton. Kieler Meeresforsch 5:1-28; 1981.

358. Wineberg, G.C. Methods for the estimation of production of aquatic animals. (Russia), New York: Academic Press; 1971.

359. Wolfe, D.A.; Champ, M.A.; Flemer, D.A.; Mearns, A.J. Long-term biological data sets: their role in research, monitoring, and management of estuarine and coastal marine systems. Estuaries 10:181-193; 1987.

360. Wolff, W.J. Estuarine benthos. In: Ketchum, B.H., ed. Estuaries and enclosed seas, ecosystems of the world 26. New York: Elsevier Scientific Publishing Company; 1983:151-182.

361. Wolff, W.J.; Sandee, A.J.J.; deWolf, L. The development of a benthic ecosystem. Hydrobiologia 52:107-115; 1977.

362. Wright, D.L.; Prior, D.B.; Hobbs, C.H.; Byrne, R.J.; Boon, J.D.; Schaffner, L.C.; Green, M.O. Spatial variability of bottom types in the lower Chesapeake Bay and adjoining estuaries and inner shelf. Estuar. Coast. Shelf Sci. 24:765-784; 1987.

363. Wulff, F.; Ulanowicz, R.E. A comparative anatomy of the Baltic Sea and Chesapeake Bay ecosystems. In: Wulff, F.; Field, J.G.; Mann, K.H., eds. Network analysis in marine ecology. Methods and applications. New York: Springer-Verlag; 1989.

364. Yamada, H.; Kayama, M. Liberation of nitrogenous compounds from bottom sediments and effect of bioturbation by small bivalve, *Theora lata* (Hinds). Estuar. Coast. Shelf Sci. 24:539-555; 1987.

365. Zaika, V.E. Specific production of aquatic invertebrates. New York: John Wiley & Sons; 1973.

366. Zola, H. Sugar phosphate polymers in polychaete tubes and in mineralized animal tissues. Comp. Biochem. Physiol. 21:179-183; 1967.

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

Theo A. Dillaha
Virginia Polytechnic Institute and Scientific University
Department of Agricultural Engineering
Blacksburg, Virginia 24061

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 ≥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 runof; 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 ≥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 prefabricated 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, nondetections, 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 components 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.

References

1. Bagnold, R.A. An approach to the sediment transport problem from general physics. U.S. Geological Survey; 1966; professional paper no. 422-J. Available from: U.S. Geological Survey, Reston, VA.
2. Baker, J.L.; Johnson, H.P. Evaluating the effectiveness of BMPs from field studies. In: Schaller, F.W.; Bailey, G.W., eds. Agricultural management and water quality. Ames: Iowa State University Press; 1983:281-304.
3. Baker, J.L.; Calvin, T.S.; Marley, S.J; Dawelbeit, M. Improved fertilizer management with a point-injector applicator. 1985; ASAE paper no. 85-1516. Available from: Am. Soc. Agric. Engrs., St. Joseph, MI.
4. Baker, J.L.; Laflin, J.M. Effects of corn residue and fertilizer management on soluble nutrient losses. Trans. ASAE 25(2):344-348; 1982.
5. Baker, J.L.; Laflin, J.M. Water quality consequences of conservation tillage. J. Soil Water Conserv. 38(3):186-193; 1983.
6. Baker, J.L. Hydrologic effects of conservation tillage and their relative importance to water quality. In: Logan, T.J.; Davidson, J.M.; Baker, J.L.; Overcash, M.R., eds. Effects of conservation tillage on groundwater quality: nitrates and pesticides. Chelsea, MI: Lewis Publishers, Inc.; 1987:113-124.
7. Barfield, B. J.; Tollner, E.W.; Hayes, J.C. Filtration of sediment by simulated vegetation, I. Steady-state flow with homogeneous sediment. Trans. ASAE 22(3):540-545,558; 1979.
8. Baun, K.; Bohn, M.; Bannerman, R.; Konrad, J. Application of the ANSWERS model in a nonpoint source program. Madison, WI: Wisc. Dept. Natural Resources, Nonpoint Source and Land Management Section; 1986.
9. Beasley, D.B.; Huggins, L.F. ANSWERS - users manual. Chicago, IL: U.S. Environmental Protection Agency, Great Lakes Program Office; 1982; 53 p.; EPA-905/9-82-001.
10. Beasley, D.B. Applying the ANSWERS model to assess the impacts of conservation tillage on sediment and phosphorus yields to Lake Erie: final report of the modeling component—Tri-State Tillage Project. Chicago, IL: U.S. Environmental Protection Agency, Region 5; 1987; EPA-905/2-87-003.
11. Berryman, D.; Bobee, B.; Cluis, D.; Haemmerli, J. Nonparametric tests for trend detection in water quality time series. Water Res. Bull. 24(3):545-556; 1988.
12. Bouwer, H. Agriculture and groundwater quality. Civil Engineering 59(7):60-63; 1989.
13. Bovey, R.W.; Richardson, C.; Burnett, E.; Merkle, M.G.; Meyer, R.E. Loss of spray and pelleted picloram in surface runoff water. J. Environ. Qual. 7:178-180; 1978.
14. Caro, J.H.; Freeman, H.P.; Glotfelty, D.W.; Turner, B.J.; Edwards, W.M. Dissipation of soil incorporated carbofuran in the field. J. Agric. Food Chem. 21:1010-1015; 1973.
15. Carsel, R.F.; Smith, C.N.; Mulkey, L.A.; Dean, J.D.; Jowise, P.P. Users manual for the pesticide root zone model (PRZM): Release 1. Athens, GA: U.S. Environmental Protection Agency; 1984; EPA-600/3-84-109.
16. Clausen, J.C. The St. Albans Bay watershed RCWP: a case study of monitoring and assessment. Perspectives on nonpoint source pollution: proceedings of a national conference; 1985 May 19-22; Kansas City, MO. Washington, DC: U.S. Environmental Protection Agency; 1985:21-24; EPA-440/5-85-001.
17. Cooper, J.R.; Gilliam, J.W. Phosphorus redistribution from cultivated fields to riparian areas. Soil Sci. Soc. Am. J. 51:1600-1604; 1987.
18. Cooper, J.R.; Gilliam, J.W.; Daniels, R.B.; Robarge, W.P. Riparian areas as filters for agricultural sediment. Soil Sci. Soc. Am. J. 51:416-420; 1987.
19. Council of Agricultural Science and Technology.

Integrated pest management. Ames, IA; 1982; 105 p. Report no. 93.

20. Council of Agricultural Science and Technology. Agriculture and groundwater quality. Ames, IA; 1985; 62 p. Report no. 103.
21. Crawford, N.H.; Linsley, R.K. 1966. Digital simulation in hydrology: the Stanford Watershed Model IV. Palo Alto, CA: Dept. of Civil Engineering, Stanford University; 1966; tech. report no. 39.
22. Crosson, P. Conservation tillage and conventional tillage: a comparative assessment. Ankeny, IA: Soil Conservation Society of America; 1981.
23. Crosson, P.; Henthorn, M.; Duffy, M. The economics of conservation tillage. In: Sprague, M.A.; Triplett, G.B., eds. No-tillage and surface tillage agriculture. New York: John Wiley & Sons; 1982:409-436.
24. Crowder, B.M. Issues in water quality modeling of agricultural management practices: an economic perspective. Proceedings of the symposium on monitoring, modeling, and mediating water quality; 1987 May; Minneapolis, MN. Minneapolis: American Water Resources Association; 1987:313-327.
25. DeCoursey, D.G.; Seely, E.H. Validation tests of the small watershed model (SWAM). Modeling agricultural, forest, and rangeland hydrology: proceedings of the 1988 internatiional symposium; 1988 Dec. 12-13; Chicago, IL. St. Joseph, MI: Am. Soc. Agric. Engrs.; 1988:307-313.
26. DeCoursey, D.G. Mathematical models for nonpoint water pollution control. J. Soil Water Conserv. 40(5):408-413; 1985.
27. D'Elia, C.F.; Sanders, J.G.; Capone, D.G. Analytical chemistry for environmental sciences: a question of confidence. Environ. Sci. Technol. 23(7):768-774; 1989.
28. Dillaha, T.A.; Beasley, D.B. Sediment transport from disturbed upland watersheds. Trans. ASAE 26(6):1766-1772,1777; 1983.
29. Dillaha, T.A.; Sherrard, J.H.; Lee, D. Long-term effectiveness and maintenance of vegetative filter strips. Blacksburg, VA: Virginia Polytechnic Institute and State University, Virginia Water Resources Research Center; 1986; 55 p. VPI-VWRRC-BULL 153.
30. Dillaha, T.A.; Reneau, R.B.; Mostaghimi, S.; Shanholtz, V.O.; Magette, W.L. Evaluating nutrient and sediment losses from agricultural lands: vegetative filter strips. Annapolis, MD: U.S. Environmental Protection Agency, Chesapeake Bay Liaison Office; 1987; 93 p. CBP/TRS 4/87.
31. Dillaha, T.A.; Sherrard, J.H.; Lee, D.; Mostaghimi, S.; Shanholtz, V.O. Evaluation of vegetative filter strips as a best management practice for feedlots. J. Water Poll. Cont. Fed. 60:1231-1238; 1988.
32. Dillaha, T.A.; Heatwole, C.D.; Bennett, M.R.; Mostaghimi, S.; Shanholtz, V.O.; Ross, B.B. Water quality modeling for nonpoint source pollution control planning: nutrient transport. Blacksburg, VA: Virginia Division of Soil and Water Conservation; Virginia Polytechnic Institute and State University, Dept. of Agricultural Engineering; 1988; 117 p. Report no. SW-88-02.
33. Dillaha, T.A.; Mostaghimi, S.; Eddleton, C.D.; McClellan, P.W. Quality control/quality assurance for water quality monitoring. 1988. Available from: Am. Soc. Agric. Engrs., St. Joseph, MI; ASAE paper no. 88-2648.
34. Dillaha, T.A.; Reneau, R.B.; Mostaghimi, S.; Lee, D. Vegetative filter strips for agricultural nonpoint source pollution control. Trans. ASAE 32(2):491-496; 1989.
35. Dillaha, T.A. Water quality impacts of vegetative filter strips. 1989; ASAE paper no. 89-2043. Available from: Am. Soc. Agric. Engrs., St. Joseph, MI.
36. Donigian, A.S.; Crawford, N.H. Nonpoint pollution from the land surface. Athens, GA: U.S. Environmental Protection Agency; 1976; EPA-600/3-76-083.
37. Donigian, A.S.; Crawford, N.H. Simulation of nutrient loadings in surface runoff with the NPS model. Athens, GA: U.S. Environmental Protection Agency; 1977; EPA-600/3-77-065.
38. Donigian, A.S.; Davis, H.H. User's manual for agricultural runoff management (ARM) model. Athens, GA: U.S. Environmental Protection Agency; 1978; EPA-600/3-78-080.
39. Donigian, A.S.; Baker, J.L.; Haith, D.A. HSPF parameter adjustments to evaluate the effects of agricultural best management practices. Athens, GA: U.S. Environmental Protection Agency; 1983; EPA-600/3-83-066.
40. Doyle, R.C.; Stanton, G.C.; Wolf, D.C. Effectiveness of forest and grass buffer filters in improving the water quality of manure polluted runoff. 1977; ASAE paper no. 77-2501. Available from: Am. Soc. Agric. Engrs., St. Joseph, MI.
41. Erbach, D.C. Tillage for continuous corn and corn-soybean rotation. Trans. ASAE 25(4):906-918,922; 1982.
42. ES&T currents: technology. Environ. Sci. Technol. 23(7):758; 1989.
43. Everett, L.G. Groundwater monitoring: guidelines and methodology for developing and implementing a groundwater quality monitoring program. Schenectady, NY: Genium Publishing Co.; 1984.
44. Field, R. Urban runoff: pollution sources, control, and treatment. Water Res. Bull. 21(2):197-206; 1985.
45. Fisher, P.E. Cleaning up Nansemond-Chuckatuck: a

threatened success story. In: Critical issues in Tidewater Virginia. Blacksburg, VA: Virginia Polytechnic Institute and State University, Virginia Water Resources Research Center; 1988:17-18.

46. Flanagan, D.C.; Neibling, W.H.; Foster, G.R.; Hurt, J.P. Application of CREAMS in filter strip design. 1986: ASAE paper no. 86-2043. Available from: Am. Soc. Agric. Engrs., St. Joseph, MI.
47. Foster, G.R.; Lane, L.J.; Nowlin, J.D.; Laflen, J.M.; Young, R.A. A model to estimate sediment from field sized areas. In: Knisel, W.G., ed. CREAMS: a field-scale model for chemical, runoff, and erosion from agricultural management systems. Washington, DC: U.S. Dept. of Agriculture, Science and Education Administration; 1980:36-64; conservation research report no. 26.
48. Foster, G.R.; Lane, L.J.; Schertz; D.L.; Nordin, J.O.; Wingate, G.D. User requirements - USDA water erosion prediction project (WEPP). 1987; ASAE paper no. 87-2539. Available from: Am. Soc. Agric. Engrs., St. Joseph, MI.
49. Frey, W.W.; Blevins, R.L.; Murdock, L.W.; Wells, L.L.; Ellis, J.H. Effectiveness of nitrapyrin with surface applied fertilizer nitrogen in no-tillage corn. Agronomy J. 73:891-895; 1981.
50. Gilley, J.E.; Lane, L.J.; Laflen, J.M.; Nicks, A.D.; Rawls, W.J. USDA water erosion prediction project: new generation erosion prediction technology. Modeling agricultural, forest, and rangeland hydrology: proceedings of the 1988 international symposium; 1988 Dec. 12-13; Chicago, IL. St. Joseph, MI: Am. Soc. Agric. Engrs.; 1988:260-263.
51. Goldman, S.J.; Jackson, K.; Bursztynsky, T.A. Erosion and sediment control handbook. New York: McGraw Hill Book Co.; 1986:8.1-8.7.
52. Green, W.H.; Ampt, G.A. Studies in soil physics: I. The flow of air and water through soils. J. Agric. Sci. 4:1-24; 1911.
53. Grisso, R.D.; Hewett, E.J.; Dickey, E.C.; Schneider, R.D.; Nelson, E.W. Calibration accuracy of pesticide application equipment. Appl. Engr. Agric. 4(4):310-315; 1988.
54. Haith, D.A.; Loehr, R.C. Effectiveness of soil and water conservation practices for pollution control. Athens, GA: U.S. Environmental Protection Agency; 1982; EPA-600/3-82-024.
55. Hartigan, J.P.; Southerland, E.; Bonucelli, H.; Canvacas, A.; Friedman, J.; Quasebarth, T.; Roffe, K.; Scott, T.; White, J. Chesapeake Bay Basin Model final report. Annandale, VA: Northern Virginia Planning District Commission; 1983.
56. Hayes, J.C. Evaluation of design procedures for vegetal filtration of sediment in flowing water. Lexington: Univ. of Kentucky; 1979. Dissertation.
57. Hayes, W.A. Minimum tillage farming. Brookfield, WI: No-tillage Farmer, Inc.; 1982; 202 p.
58. Hayes, J.C.; Barfield, B.J.; Barnhisel, R.I. Filtration of sediment by simulated vegetation II. Unsteady flow with non-homogeneous sediment. Trans. ASAE 22(5):1063-1067; 1979.
59. Hayes, J.C.; Barfield, B.J.; Barnhisel, R.I. The use of grass filters for sediment control in strip mine drainage: Volume III. Empirical evaluation of procedures and verification. Lexington: Univ. of Kentucky, Inst. for Mining and Minerals Research; 1982; IMMR82/070.
60. Hayes, J.C.; Hairston, J.E. Modeling the long-term effectiveness of vegetative filters on on-site sediment controls. 1983; ASAE paper no. 83-2081. Available from: Am. Soc. Agric. Engrs., St. Joseph, MI.
61. Hipel, K.W. Nonparametric approaches to environmental impact assessment. Water Res. Bull. 24(3):487-492; 1988.
62. Huber, W.C.; Heaney, J.P.; Nix, S.J.; Dickinson, R.E.; Polmann, D.J. Storm water management model user's manual: version III. Gainesville: Univ. of Florida, Dept. of Environmental Engineering Sciences. 1983:505.
63. Jacobs, T.C.; Gilliam, J.W. Riparian losses of nitrate from agricultural drainage water. J. Environ. Qual. 14(4):472-478; 1985.
64. Johanson, R.C.; Imhoff, J.C.; Davis, H.H.; Kittle, J.L.; Donigian, A.S. User's manual for hydrologic simulation program-Fortran (HSPF): Release 7.0. Athens, GA: U.S. Environmental Protection Agency; 1981.
65. Keith, L.H., ed. Principles of environmental sampling. Washington, DC: American Chemical Society; 1988; 458 p.
66. Knisel, W.G., ed. CREAMS: a field-scale model for chemical, runoff, and erosion from agricultural management systems. Washington, DC: U.S. Dept. of Agriculture, Science and Education Administration; 1980; conservation research report no. 26.
67. Koehler, F.A.; Johnson, D.D.; Kreglow, J.M.; Dressing, S.A.; Maas, R.P.; Humenik, F.J.; Christensen, L.; Snyder, W.K. Conceptual framework for assessing agricultural nonpoint source projects. Raleigh: North Carolina State University, Biological and Agricultural Engineering Dept.; 1981; 60 p.
68. Kovach, J.; Tette, J.P. A survey of the use of IPM by New York apple producers. Agriculture, Ecosystems and the Environment 20:101-108; 1988.
69. Krukar, M.; Cook, J.C. Optimization and testing of highway materials to mitigate ice adhesion. Washington, DC: U.S. Environmental Protection Agency; 1978; EPA-600/2-78-035.
70. Kuo, C.Y.; Cave, K.A.; Loganathan, G.V. Planning

of urban best management practices. Water Res. Bull. 24(1):125-132; 1988.

71. Lake, J.; Morrison, J. Environmental impact of land use on water quality: final report on the Black Creek Project. Chicago, IL: U.S. Environmental Protection Agency, Region 5; 1977; EPA-905/9-77-007-B.
72. Lee, D.; Dillaha, T.A.; Sherrard, J.H. Modeling phosphorus transport in grass buffer strips. J. Environ. Engr. Div. 115(2):408-426; 1989.
73. Leonard, R.A.; Knisel, W.G.; Still, D.A. GLEAMS: groundwater loading effects of agricultural management systems. Trans. ASAE 30(5):1403-1418; 1987.
74. Leonard, R.A.; Knisel, W.G. Selection and application of models for nonpoint source pollution and resource conservation. Modeling agricultural, forest, and rangeland hydrology; proceedings of the 1988 international symposium; 1988 Dec. 12-13; Chicago, IL. St. Joseph, MI: Am. Soc. Agric. Engrs.; 1988:213-229.
75. Little, C.E. Geology and survival on the Snake River Plain. J. Soil Water Conserv. 44(2):127-129; 1989.
76. Maas, R.P.; Birchford, S.L.; Smolen, M.P.; Spooner, J. Agricultural nonpoint source control: experiences from the rural clean water program. J. Lake Reservoir Mgmt. 4(1):51-56; 1988.
77. Madramootoo, C.A.; Laperriere, L.; Barrington, S.F. Modeling runoff and sediment yields from Quebec watersheds. Modeling agricultural, forest, and rangeland hydrology; proceedings of the 1988 international symposium; 1988 Dec. 12-13; Chicago, IL. St. Joseph, MI: Am. Soc. Agric. Engrs; 1988:264-270.
78. Magette, W.L.; Brinsfield, R.B.; Palmer, R.E.; Wood, J.D. Nutrient and sediment removal by vegetated filter strips. Trans. ASAE 32(2):663-667; 1989.
79. Moe, P.G.; Mannering, J.V.; Johnson, C.B. A comparison of nitrogen losses from urea and ammonium nitrate in surface runoff water. Soil Sci. 105(6):428-433; 1968..
80. Morrison, J.E. Farm machinery development for no-tillage agriculture. Proceedings of the southern regional no-tillage conf.; 1986 June 18 (Southern Regional Series Bull. 319). Lexington: Univ. of Kentucky; 1986:16-32.
81. Mueller, D.H.; Daniel, T.C.; Lowery, B.; Andraski, B. The effect of conservation tillage on the quality of the runoff water. 1982; ASAE paper no. 82-2022. Available from: Am. Soc. Agric. Engrs., St. Joseph, MI.
82. National Water Quality Evaluation Project. State-of-the-art review of best management practices for nonpoint source pollution control: I. Animal waste. Raleigh: North Carolina Agricultural Extension Service, North Carolina State University Water Quality Group; 1982; 67 p.
83. National Water Quality Evaluation Project. State-of-the-art review of best management practices for nonpoint source pollution control: II. Commercial fertilizer. Raleigh: North Carolina Agricultural Extension Service, North Carolina State University Water Quality Group; 1982; 55 p.
84. National Water Quality Evaluation Project. State-of-the-art review of best management practices for nonpoint source pollution control: III. Sediment. Raleigh: North Carolina Agricultural Extension Service, North Carolina State University Water Quality Group; 1982; 49 p.
85. National Water Quality Evaluation Project. State-of-the-art review of best management practices for nonpoint source pollution control: IV. Pesticides. Raleigh: North Carolina Agricultural Extension Service, North Carolina State University Water Quality Group; 1984; 45 p.
86. Neubeiser, M.J. Rock Creek rural clean water program: the experiment continues. Perspectives on nonpoint source pollution: proceedings of a national conference; 1985 May 19-22; Kansas City, MO. Washington, DC: U.S. Environmental Protection Agency; 1985:391-396. Available from Lake Management Society, Merrifield, VA. EPA-440/5-85-001.
87. Norman, D.A.; Edwards, W.M.; Owens, L.B. Design criteria for grass filter areas. 1978; ASAE paper no. 78-2573. Available from: Am. Soc. Agric. Engrs., St. Joseph, MI.
88. Novotny, V.; Chesters, G. Handbook of nonpoint source pollution: sources and management. New York: Van Nostrand Reinhold Co.; 1981.
89. Novotny, V. A review of hydrologic and water quality models used for simulation of agricultural pollution. In: Giorgini, A.; Zingales, F., eds. Agricultural nonpoint source pollution: model selection and application. New York: Elsevier Publishers; 1986:9-35.
90. Onstad, C.A.; Foster, G.R. Erosion modeling on a watershed. Trans. ASAE 18:288-292; 1975.
91. Parrish, R.S.; Smith, C.N. A method for testing whether model predictions fall within a prescribed factor of true values, with an application to pesticide loading. Athens, GA: U.S. Environmental Protection Agency, Environmental Research Laboratory; 1988.
92. Piper, S.; Young, C.E.; Magleby, R. Benefit and cost insights from the Rural Clean Water Program. J. Soil Water Conserv. 44(3):203-208; 1989.
93. Pitt, R.E. Best management practices: implementation. Washington, DC: U.S. Environmental Protection Agency; 1979; EPA-905/9-81-002.
94. Reinelt, R.E.; Horner, R.H.; Mar, B.W. Nonpoint

source pollution monitoring program design. J. Water Res. Planning Mgmt. 114(3):335-352; 1988.

95. Richards, R.P. Estimating the extent of reduction needed to statistically demonstrate reduced phosphorus loading to Lake Erie. J. Great Lakes Res. 11(2):110-116; 1985.
96. Rittall, W.; Shuyler, L.R. Guidelines for evaluation of agricultural nonpoint source water quality projects. Raleigh: North Carolina State Univ., Biological and Agricultural Engineering Dept.; 1988; 59 p.
97. Sartor, J.D.; Boyd, G.B.; Agardy, F.J. Water pollution aspects of street surface contaminants. J. Water Poll. Cont. Fed. 46:458-667; 1974.
98. Shaffer, M.J.; Larson, W.E. Nitrogen-tillage-residue management (NTRM) model: technical documentation. St. Paul, MN: U.S. Dept. of Agriculture, Agricultural Research Service; 1982.
99. Shaffer, M.J.; Pierce, F.J. Nitrogen-tillage-residue management (NTRM) model: user's manual. St. Paul, MN: U.S. Dept. of Agriculture, Agricultural Research Service; 1985.
100. Shaffer, M.J. Simulation model for soil erosion-productivity relationships. J. Environ. Qual. 14(1):144-150; 1985.
101. Shanholtz, V.O.; Flagg, J.M.; Mostaghimi, S.; Heatwole, C.D. Agricultural pollution potential database for the Peanut Soil and Water Conservation District: interim report. Blacksburg, VA: Virginia Polytechnic Institute and State Univ., Dept. of Agricultural Engineering; 1989; 40 p. ISSL-89-5.
102. Skelly, T.M.; Hinsman, W.J. Cost-sharing soil conservation practices: economic pros and cons from a lake management perspective. J. Lake Reservoir Mgmt. 4(2):237-242; 1988.
103. Smith, M.C.; Williams, J.R. Simulation of surface water hydrology. In: Knisel, W.G., ed. CREAMS: a field-scale model for chemical, runoff, and erosion from agricultural management systems. Washington, DC: U.S. Dept. of Agriculture, Science and Education Administration; 1980:13-35; conservation research report no. 26.
104. Smith, M.C.; Campbell, K.L.; Bottcher, A.B.; Thomas, D.L. Field testing and comparison of the PRZM and CREAMS models. 1989; ASAE paper no. 89-2072. Available from: Am. Soc. Agric. Engrs., St. Joseph, MI.
105. Soil Conservation Service. Conservation P factor values for the revised universal soil loss equation. In: Revised universal soil loss equation manual (draft). 1989. Available from: USDA, Agricultural Research Service, National Soil Erosion Research Laboratory, West Lafayette, IN.
106. Spooner, J. Conference report: International symposium on the design of water quality information systems. In: National water quality evaluation project notes; no. 39, July 15, 1989. Available from: North Carolina Agricultural Research Service, North Carolina State University Water Quality Group, Raleigh, NC; 1989.
107. Spooner, J.; Maas, R.P.; Dressing, S.A.; Smolen, M.D.; Humenik, F.J. Appropriate designs for documenting water quality improvements from agricultural NPS pollution control. Perspectives on nonpoint source pollution: proceedings of a national conference; 1985 May 19-22; Kansas City, MO. Washington, DC: U.S. Environmental Protection Agency; 1985; EPA-440/5-85-001. Available from Lake Management Society, Merrifield, VA.
108. Spooner, J.; Jamieson, C.J.; Maas, R.P.; Smolen, M.D. Determining statistically significant changes in water pollutant concentrations. J. Lake Reservoir Mgmt. 3:195-201; 1987.
109. Spooner, J.; Brichford, S.L.; Maas, R.P.; Smolen, M.D.; Dickey, D.D.; Ritter, G.; Flagg, E. Determining the statistical sensitivity of the water quality monitoring program in the Taylor Creek-Nubbin Slough, Florida project. J. Lake Reservoir Mgmt. 4(2):113-124; 1988.
110. Stewart, B.A.; Woolhiser, D.A.; Wischmeier, J.H.; Caro, J.H.; Frere, M.H. Control of water pollution from cropland, vol. II. Washington, D.C.: U.S. Environmental Protection Agency, 1976; EPA-600/2-75-026b.
111. Storm, D.E.; Dillaha, T.A.; Mostaghimi, S.; Shanholtz, V.O. Modeling phosphorus transport in surface runoff. Trans. ASAE 31(1):117-127; 1988.
112. U.S. Army Corps of Engineers. Urban storm water runoff-STORM. Davis, CA: Hydraulic Engineering Center; 1975.
113. U.S. Environmental Protection Agency. RCRA ground-water monitoring technical enforcement document. Washington, DC: Environmental Protection Agency; 1986.
114. U.S. Environmental Protection Agency, Region 3. Chesapeake Bay Nonpoint Source Programs. Annapolis, MD: Chesapeake Bay Liaison Office; 1988; 122 p.; 516-002/80044.
115. Wagenet, R.J. Processes influencing pesticide loss with water under conservation tillage. In: Logan, T.J.; Davidson, J.M.; Baker, J.L.; Overcash, M.R., eds. Effects of conservation tillage on groundwater quality: nitrates and pesticides. Chelsea, MI: Lewis Publishers, Inc.; 1987:189-204.
116. Ward, R.C.; Loftis, J.C. Establishing statistical design criteria for water quality monitoring systems: review and synthesis. Water Res. Bull. 22(5):759-768; 1986.
117. Ward, R.C.; Loftis, J.C.; McBride, G.B. The "data-

rich but information poor" syndrome in water quality monitoring. Environ. Mgmt. 10(3):291-297; 1986.

118. Williams, R.D.; Nicks, A.D. Using CREAMS to simulate filter strip effectiveness in erosion control. J. Soil Water Conserv. 43(1):108-112; 1988.
119. Whitfield, P.H. Goals and data collection for water quality monitoring. Water Res. Bull. 24(4):775-780; 1988.
120. Wischmeier, W.H.; Smith, D.D. Predicting rainfall erosion losses - a guide to conservation planning. Washington, DC: U.S. Department of Agriculture, Science and Education Administration; 1978; 58 p. Agriculture handbook no. 537.
121. Young, C.E.; Magleby, R. Agricultural pollution control: implications from the Rural Clean Water Program. Water Res. Bull. 23(4):701-707; 1987.
122. Young, R.A.; Huntrods, T.; Anderson, W. Effectiveness of vegetative buffer strips in controlling pollution from feedlot runoff. J. Environ. Qual. 9:483-487; 1980.
123. Young, R.A.; Otterby, M.A.; Roos, A. An evaluation system to rate feedlot pollution potential. Washington, DC: U.S. Dept. of Agriculture, Agricultural Research Service; 1982; ARM-NC-17.
124. Young, R.A.; Onstad, C.A.; Bosch, D.D.; Anderson, W.P. AGNPS, agricultural non-point-source pollution model: a watershed analysis tool. Washington, DC: U.S. Dept. of Agriculture Agricultural Research Service; 1987; 80 p. Conservation research report 35.
125. Young, R.A.; Onstad, C.A.; Bosch, D.D.; Anderson, W.P. AGNPS: a nonpoint source pollution model for evaluating agricultural watersheds. J. Soil Water Conserv. 44(2):168-173; 1989.

Developing an Ecological Risk Assessment Strategy for the Chesapeake Bay

John Cairns, Jr., and David R. Orvos
University Center for Environmental and Hazardous Materials Studies
Virginia Polytechnic Institute and State University
Blacksburg, Virginia 24061

Introduction

The world's population growth, along with increased individual expectations for improved quality of life, have vastly increased the pressures upon natural resources. As a consequence, those resources, including the Chesapeake Bay, may eventually be unsuitable for use. An estimated population growth of 20% [83] by the year 2020 in and around the Bay will place additional burdens on this watershed, which is simultaneously used for a variety of purposes, with frequent conflict or competition between uses [34].

Protocols for assessing ecological and human health risk have recently been developed to the point that an accurate assessment of risk from a particular event in a local area can often be made. However, development of protocols for use on larger, regional ecosystems has not proceeded as rapidly. Effective management of regions such as the Chesapeake Bay cannot be carried out in the current fragmented fashion. Relevant research directives, rational risk assessment, and integrated resource management are essential. The following discussion reviews the components and strategies of risk assessment for regional areas, examines strategy development for the Bay, identifies areas needing additional research, and analyzes the importance of risk assessment to management practices.

Describing risk assessment methodologies requires definition of some terminology. *Stress* is used in this discussion to describe any factor that may cause damage to an ecosystem, even though in the past stress has been viewed as an individual physiological reaction. *Risk* is defined here as the probability of harm from a natural or anthropogenic stress in the environment [16]. Prior to 1977, potential environmental damage was assessed by considering only effects [75]. The coupling of effects assessment with exposure gave rise to *exposure assessment*, a process that has found its way into many federal regulations [75]. *Environmental risk assessment* is still a developing field and has been defined in different ways. A National Research Council Committee [63] defines environmental risk assessment as "the characterization of the potential adverse health effects of human exposure to environmental hazards." An Environmental Protection Agency (EPA) review of assessment methods [35] defines ecological risk assessment as any "assessment related to actual or potential ecological effects resulting from human activities."

Risk assessment should be a scientific endeavor, depending on scientific data and evaluations that provide information to scientists as well as the public, commercial, and regulatory sectors [75]. *Risk management*, however, as the process of determining how to deal with the risk, by definition includes scientific, political, and socioeconomic facets [16, 63]. Although adverse environmental effects upon human activities are obviously important, this review concentrates solely on ecological risk assessment using the EPA definition, with the understanding that both human and non-human activities affect the Bay's integrity [10].

While the objectives of particular ecological risk assessments may vary, any such assessment should (1) evaluate actual or potential risk from an environmental impact, (2) determine the probability that the impact may, in fact, adversely affect the environment, and (3) predict potential risk prior to the actual impact. These goals are feasible when the stressor and its affected area are well defined, but may be more difficult to achieve when the area is larger and the stressor is combined with other natural and anthropogenic factors, as is the case with the Chesapeake Bay.

The concept of localized risk assessment, the qualitative and/or quantitative evaluation of actual or potential harm to the environment, has been well documented and refined in recent years [8, 15, 33, 37-39, 42, 73]. The use of environmental impact assessments to predict and assess environmental and human

health risks has been mandated by federal, state, and local statutes for some time. These procedures, although they may be open to subjective bias because of legal ambiguity, have been useful for predicting localized impact from specific sources.

By contrast, the success of regional risk assessment has not been convincingly demonstrated. Such assessments examine risk to an entire region, such as the Chesapeake Bay, rather than just a particular locality, such as the area immediately around an industrial plant. Few studies have adequately addressed the regional concept, and additional research to examine what factors are important in regional assessment as well as the uncertainty involved is warranted [53, 75]. A good approach to regional risk assessment to date is offered by Hunsaker et al. [46]. The authors portray regional risk assessment as having two distinct phases: (1) a definition phase in which source terms (qualitative and quantitative descriptions of the disturbance source), endpoints, and a reference environment (geographic location and temporal period of the region at risk) are defined, and (2) a solution phase that combines exposure and effects to produce a risk probability. The interdependence of source terms, endpoints, and reference environment is emphasized, as is the importance of using functionally defined regions. For example, high ozone levels in the Adirondack region of New York caused subsequent insect outbreaks that affected water quality and wildlife habitat. A regional risk assessment resulted in the conclusion that high levels of ozone did have a regional effect, particularly on landscape pattern.

Quantitative risk assessment, or the assignment of a probability to a risk [41, 63], is difficult even in ideal situations because of the inherent variability of both the environment and the testing procedures used to evaluate the hazard. This task is further complicated when the region affected is large and diverse. An excellent review of several quantitative methods is found in Barnthouse et al. [4].

Conceptual Review of Risk Assessment

Hazard and risk assessment methodologies have been refined primarily because their use is required by federal, state, and local statutes. These statutes are reviewed elsewhere [2, 16, 54].

Risk assessment traditionally has several components [63]: (1) hazard identification, (2) dose-response assessment (usually for a particular substance), (3) exposure assessment, and (4) risk characterization. Hazard identification is the determination of whether a particular substance is either a human or ecological hazard [16]. It must be used with hazard control and assessment to provide effective hazard management (Figure 1). While identifying a hazard may seem simple, legal statutes may require a preponderance of scientific data to support the labeling of a substance as a hazard [71].

The dose-response assessment is one of the fundamental principles of aquatic toxicology [55, 70]. Characterizing the relationship between the applied dose of a substance and the response allows projections of what response a particular dose may elicit. This step will be difficult to apply to sources of pollution that contain many substances.

Exposure assessment is an important component of the risk assessment process since exposure must be accurately known before risk from a particular impact can be predicted or modeled. Exposure is the substance concentration that is in immediate contact with an organism [36]. Exposure is not the same as absorbed dose, a measurement that requires additional physical and physiological data. In its complete form, exposure assessment examines the route, duration, and magnitude of the exposure as well as describing the types of organisms, populations, and habitats that are exposed [63]. Recently, the term receptor characterization has been used to describe the process of determining which biota are subject to risk at a species, community, population, or ecosystem level [44].

Risk assessment combines hazard identification, dose-response assessment, and exposure assessment to produce an estimate of the impact a particular substance has when administered at a particular concentration. These components may be well understood for specific releases of well-characterized substances in a thoroughly studied ecosystem, but may be more ambiguous in a regional risk assessment. Even in localized areas, uncertainty plays a major role; this is amplified in regional areas such as the Bay.

Endpoint Selection

The risk assessment process involves many judgments, such as determining which impacts are important to assess, defining what is to be protected, and deciding how to measure impact on those parameters selected as important. The process of ascertaining which impacts are important may be of research interest only, but more likely will be of regulatory concern as well. Impacts applicable to the Chesapeake Bay include sediment loading, nutrient loading, and low-level toxic chemicals in water, sediment, and adjacent wetlands [34, 56, 82].

Defining what is to be protected is an important aspect of the strategy process. This subjective process incorporates political and socioeconomic factors, and priorities may change with time and administrations. Yet, without a definition of what resources to protect, subsequent development of strategy, testing procedures,

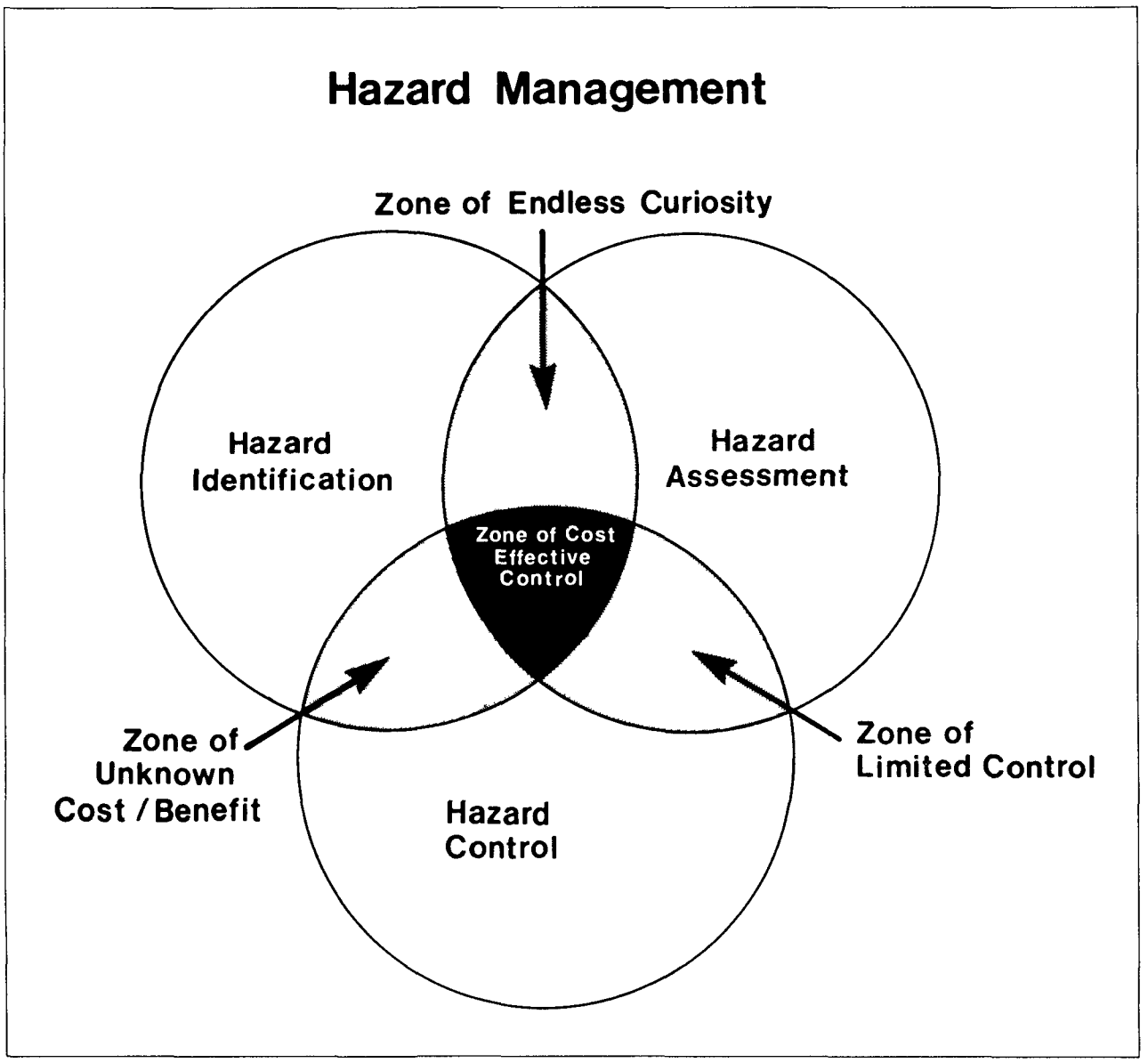

Figure 1. Components and interrelationships of hazard management.

and model development may be worthless. Although prioritizing resources is beyond the scope of this review, considerations should include protection of species diversity and ecosystem functioning as well as preservation of submerged aquatic vegetation (SAV), aquatic biota habitat, and adjacent wetlands.

Once areas of importance are defined, then endpoints for measuring the effect of stresses upon these important areas may be selected. Regardless of what endpoints are chosen, they should possess common characteristics (Table 1). Several groups of endpoints have been proposed, including assessment and measurement endpoints [46] and chronic and acute endpoints [33]. Generalized agreement about endpoints must be reached by all parties concerned about Bay risk assessment, including regulators, municipalities, researchers, and environmental groups. While such a consensus may be difficult to achieve, we must strive to agree on what is and what is not important to monitor in the Bay before risk assessment strategies can be further developed. Relevant endpoints for the Bay are discussed in a subsequent section (Endpoint Selection Criteria).

Information Uses and Transfer

Once defined, risks must then be communicated to the

Table 1. Ideal endpoint characteristics for Chesapeake Bay risk assessment.

Characteristic	Example
Socioeconomic relevance	Finfish yield
Biological relevance	Wetlands acreage/distribution
Susceptibility to hazard	Distribution of submerged aquatic vegetation
Unambiguous operational definition	Percentage loss of oyster production
Predictive capability	LC_{50} of zinc for particular species under standard conditions

managers, regulators, and politicians who will establish legal mandates [1]. Although characterizing and analyzing risk is a scientific pursuit, deciding whether that risk is acceptable to society is not [22]. Such decisions are made by politicians and managers using cost-benefit analyses and integrated management with a highly subjective, nonquantifiable component. While these individuals use scientific data, they also incorporate various political and socioeconomic components. Scientists in the past have often failed to realize this and have not entered the decision-making process; however, this process is a vital component of risk management. Scientists must ensure that scientific data are properly used by public officials. This is especially true when regulators call for additional data and data-reviews in order to sway a decision. Many examples exist of additional studies that have been funded for the purpose of delaying a controversial decision. Unfortunately, the Chesapeake Bay cannot wait for additional multitudes of data to be collected and analyzed before fundamental decisions are made concerning its ultimate fate. Reviews on decision-making processes and their associated socioeconomic and political components have been published elsewhere [63].

Types of Risk Assessment Methods

Even though many risk assessment methods exist [4, 35, 37, 52, 58], most assess potential for risk or perceived risk in a particular ecosystem. Methods can be grouped in several ways: qualitative vs. quantitative, top-down vs. bottom-up, or reductionist vs. holistic methods [16, 35]. Of the methods reviewed in the EPA [35] report, most have one of three objectives: (1) to set priorities, (2) to support the establishment of guidelines or standards, or (3) to serve as an input to risk management decision-making. Specific federal risk assessment methods have been in use for some time and vary depending upon the intended use. Designed for use with specific statutes as previously discussed, they are numerous; they have been reviewed elsewhere [35].

Reductionist ecological approaches to assessment compartmentalize the myriad ecological processes that may be affected into understandable units [16, 72]. The effect of the hazard upon these ecological compartments is then evaluated. Such approaches do not adequately evaluate synergistic effects and may be low in environmental realism. A relevant example for the Bay would be the study of nutrient addition. Compartmentalizing the problem allows the effects of sewage treatment plants, agriculture, and industry to be studied independently of each other; but the total effect of all of these sources upon an entire region may be ignored. Holistic approaches consider the ecosystem, with all of its interactions, as an entity for evaluation; however, environmental uncertainty may be unacceptable, and deciding what parameters should be evaluated is indeed difficult. The entire Bay cannot yet be studied to understand the complexities of interactions; therefore, a combination of these procedures must be utilized.

Another differentiation is between top-down and bottom-up methods [35]. Top-down methods evaluate structural and functional changes at both the community and ecosystem level directly and require ecosystem data; however, few data exist for actually predicting the effects of chemicals on such ecosystem properties [18, 35]. EPA concludes that top-down methods are useful for setting priorities but not for quantitative risk assessment. Bottom-up methods approximate community-level effects using models primarily based on laboratory data from population and individual responses, such as mortality [35]. These methods address ecosystem processes, especially transfer processes, but require large amounts of specific chemical and site data; therefore, they are most useful in supporting decisions for

site-specific locations. Another criticism [35] of bottom-up methods is that there is no accepted definition of "ecosystem health."

Local risk assessment strategies, such as bottom-up methods [35], often utilize large amounts of site-specific data for predictive and modeling capabilities. Therefore, applying them to regional settings, such as the Bay, will be difficult because of the expense of data collection. Although extensive data bases already exist for the Bay [47], they may not be useful if the chosen endpoint was not represented in the initial data collection. However, risk assessments of regional ecosystems may adopt endpoints such as landscape cover, community composition, biotic diversity, population shifts, and primary productivity. More uncertainty is likely in regional assessments than in small, localized assessments because current methodologies, geared primarily toward acute effects, may not be applicable to a large watershed that also contains a multitude of chronic and nonpoint pollution sources.

Quantitative risk analysis methods — i.e., those that quantify the probability of risk as well as the degree of uncertainty — include analysis of extrapolation error, fault tree analysis, ecosystem uncertainty analysis, analytic hierarchy method, and the quotient method. The ecological theory required in such system analysis is still in its infancy [35]. Research and refinement of these techniques must be continued since ultimately they will provide us with a quantified estimate of Bay risk and its associated uncertainty as well as giving regulators the capability of predicting risk. All of these quantitative methods, as well as their applications, advantages, and limitations, are described in an excellent review by Barnthouse et al. [4].

Another factor for consideration in strategy development is the extremes of acute vs. chronic releases of stressful inputs. Acute spills are infrequent, arouse negative public opinion [43], and may result in subsequent legislation, such as that following the Bhopal accident and the Alaskan oil spill. Chronic releases, often far more damaging, are less likely to attract public attention, funding, and ensuing regulation. Politicians and funding agencies must be made cognizant of these acute vs. chronic conflicts so that adequate attention and funding are directed to chronic input problems, including sediment influx, toxic chemicals, and nutrients. Both of these extremes require creation and/or modification of risk assessment schemes and different management approaches for their resolution.

Uncertainty Assessment

Estimates of uncertainty in risk assessment may be large and confusing. Several reviews exist [35, 38, 42, 46], so only a brief discussion will be presented here.

Table 2. Tiered system for determining hazard effects at different levels of biological organization.

Tier 1 - Screening or range finding tests
Tier 2 - Predictive tests
Tier 3 - Validating Tests
Tier 4 - Monitoring

From [23].

Uncertainty is inherent in the risk assessment process. Application or safety factors are often used in assessment approaches to deal with this uncertainty; these are sometimes, but not always, based on scientific data. Some other techniques used to include uncertainty in the actual assessment and modeling are statistical confidence limits, Monte Carlo simulation, sensitivity analysis, and field validation [35].

Significant uncertainty exists when laboratory bioassays are extrapolated to actual environmental effects, as is necessary in bottom-up schemes [20, 35]. Also, uncertainty is present in top-down approaches when ecosystem parameters are examined directly with ill-suited endpoints [35]. In conclusion, the EPA report states: "As models become more complex and representative of real-world processes, other difficulties arise." It also states: "The more realistic the model, the less likely it is that adequate ... data are available." Therefore, endpoints must be relevant to the Bay, testing procedures must be well documented, and experimental design and statistical analysis procedures must be adequate. It is recommended that a tiered system of tests (Table 2) be adopted and that prescriptive legislation be more ecosystem-specific [23]. Potential and real errors must be quantified and their effect upon risk assessment ascertained.

Ultimately, improved predictive models will be developed so that potential risk may be quantified. It is imperative that such models consider spatial and temporal scaling in the strategy development process [50] since both of these will influence any strategy developed for the Bay. For example, toxics and other environmental stressors may have a cumulative or aggregate impact through time that is not readily evident over a short period [24]. One of the serious problems in estimating risk on the basis of short-term toxicity tests on a site-by-site basis is the possibility of missing either cumulative impacts or aggregate impacts in the system as a whole. As a consequence, feedback of information taken directly from the Bay is essential to control error, at least until the predictive models of risk become more useful than they now are. In short, we need to continue and even improve the Bay monitoring program.

Table 3. Evaluation of recovery characteristics of selected ecosystems as rated by a panel of experts.

Ecosystems	Information presently available	Potential for man-influenced restoration
Water ecosystems		
Freshwater flowing	4	6
Freshwater impounded	5	7
Estuary	3	4
Marsh	6	8
Shoreline	7	2
Terrestrial ecosystems		
Arctic	8	5
Temperate	10	10
Tropic	2	1

Note: A rating of 10 represents the most information or highest potential; a rating of 1, the least information or lowest potential. From [27].

Field Validation Considerations

Many models and assessments are never field-validated [20, 26], and others may have poor predictive capability because of confounding environmental variables. Since the objective is to prevent damage to ecosystems, most predictive risk models are based on evidence in the literature, laboratory experiments, or extrapolations from accidents in somewhat similar ecosystems. Direct validation of these predictive models in the Bay itself is rarely possible because of the damage such a study would cause; therefore, the use of ecoaccidents to validate predictive models should provide information not readily gathered in any other way. Regulatory and research groups must plan ecoaccident studies now so that when an accident occurs, the logistics of studying it have already been addressed. An exchange of both planning and post-accident information with other groups facing similar problems can advance the development of a more robust array of predictive models. Although accidental releases may be useful in validating models that use toxic substances, it is urged that validation of models using sediment influx and nutrient addition should be considered now. Use of existing research facilities on the Bay as well as development of additional ones in remote areas would contribute to our understanding of how these inputs affect Bay function on a broader scale. We urge that models developed for microcosm and mesocosm use be tested in the Bay.

Risk to the Chesapeake Bay ecosystem might be divided into two components: (1) risk due to displacement or degradation of normal ecological conditions and (2) risk of an inordinate recovery time from such displacement. Indeed, estuaries have among the slowest recovery characteristics (4 on a scale of 10) of water ecosystems, while marshes and freshwater systems rank higher (Table 3). However, in some instances, recovery might be quite rapid from certain types of displacements in estuaries since estuarine organisms are more resilient. Fisher [40] demonstrated that estuarine phytoplankton were more resistant to chemical stress than, for example, organisms in open ocean, because they were more tolerant of variable salinity and temperature. In this case, the risk would be less than if severe displacement were accompanied by an extended recovery time. Both of these components can be estimated from information generated outside the Chesapeake Bay ecosystem itself but can be rigorously validated only by information obtained directly from the Bay.

Pertinent Ecological Theory

Examining the various forms and components of risk assessment requires a brief review of a few ecological principles. Obviously, a thorough examination of ecology and other disciplines essential to risk assessment is beyond the scope of this review, and readers are urged to consult other references [6, 37, 38, 72].

The biogeochemical cycling of nutrients is one of the most crucial elements for consideration, especially

for the Bay. Primary production in aquatic ecosystems is chiefly limited by nutrients, and any alteration may have significant ramifications [72, 82]. Increased nutrient inputs into the Bay, primarily nitrogen and phosphorus, have resulted in phytoplankton proliferation to such an extent that light penetration to SAV is reduced. This eutrophication appears to be the main element in SAV community changes since 1930 [32] and must be incorporated into a risk assessment scheme.

Organisms in the Bay may take up toxic materials either directly from the water or indirectly by ingesting food. Bioaccumulation, the process by which substances are taken up by aquatic organisms, is often confused with bioconcentration and biomagnification. Bioconcentration is a process of net accumulation of a substance directly from the water into the organism. Biomagnification, a consequence of bioaccumulation and bioconcentration, results in a net increase in tissue toxicant concentrations as the chemical passes through trophic levels [70]. Although the amount of a particular toxicant or other stress may be accurately known, the actual amount to which organisms are exposed is often not. The bioavailability of a substance is dependent upon many factors, including temperature, pH, hardness, and salinity [19, 37, 77]. These factors are important because they serve as modifiers of the toxic response and, therefore, must be considered in risk evaluation and resolution.

Ecological population theory describes relationships in populations, including birth, growth, density, regulation, and death. Such factors are important because determining toxicant effects on populations is far more feasible and realistic than determining effects on individuals. Barnthouse et al. [6] related population theory to ecological risk assessment and concluded that neither population nor ecosystem theory alone provides suitable models that could predict long-term impacts from the release of hazardous chemicals. However, such theory must be considered in research and assessment design and will become even more important in the risk evaluation and regulatory process as present methods are refined and new ones developed.

Assimilative capacity is the ability of an ecosystem to receive an impact without significant alteration in structural and/or functional parameters of the indigenous community [12, 13]. For most deleterious materials, there appear to be three ecological response zones: (1) one in which no adverse dose-related effects are noted, (2) one in which a change in concentration is accompanied by a change in response, and (3) one in which the system is incapable of further sublethal response [13]. Thus, exceeding this assimilative capacity will produce biologically significant environmental impact. Assimilative capacity depends upon many factors, including prior ecosystem stresses, nature and duration of stresses (using the definition of stress described earlier), system inertia and resiliency, and residual effects [12]. Critics of this concept (e.g., Campbell [30]) assert that some alteration, detectable or not, follows the introduction of a substance, man-made or not. Cairns [17] defends the concept with a thorough examination of both the strengths and weaknesses of assimilative capacity. Whether or not ecosystem assimilative capacity in fact exists, society is acting as though it does [13]; at present levels of population and individual expectations, it is difficult politically or technologically to act otherwise. Assimilative capacity may be measurable in the Bay with use of a scoring system that evaluates the "health" of an area and the stress to which it is being subjected [12]. This assimilative capacity theory may explain why many Bay problems have seemingly become more serious in recent years as the estuary encounters more toxic substances, nutrients, sediments, and wetlands degradation while lacking the ecological resources to compensate for them.

Biological integrity is defined as the maintenance of community structure and function of a particular locale [14]. It is often used as an all-encompassing term but may be quantified for some biota, such as fish, in a particular area [49]. Biological integrity is closely linked with two other concepts: species diversity and richness, and assimilative capacity. If the assimilative capacity of an area is exceeded, for example by a toxicant, biological integrity may be impaired as species richness and diversity decline. Although it is difficult to relate assimilative capacity to a regional watershed like the Chesapeake Bay because of the natural uncertainty in observations, it is unfortunate that more attention is not given to this crucial issue in determining ecological risk to the Chesapeake Bay and other major ecosystems.

Ecological Risk Assessment Strategy for the Chesapeake Bay

The main factors that became apparent as this discussion was being prepared were the complexity, diversity, and economic importance of the Chesapeake Bay. No one ideal scheme can be developed and applied to formulate a risk assessment scheme for this region. While dozens of plans could be considered, a framework of four plans could effectively address the following areas: (1) chronic point source impacts, (2) nonpoint source discharges, (3) cumulative and aggregate impacts, and (4) acute accidental releases of hazardous substances. A skeleton plan is shown

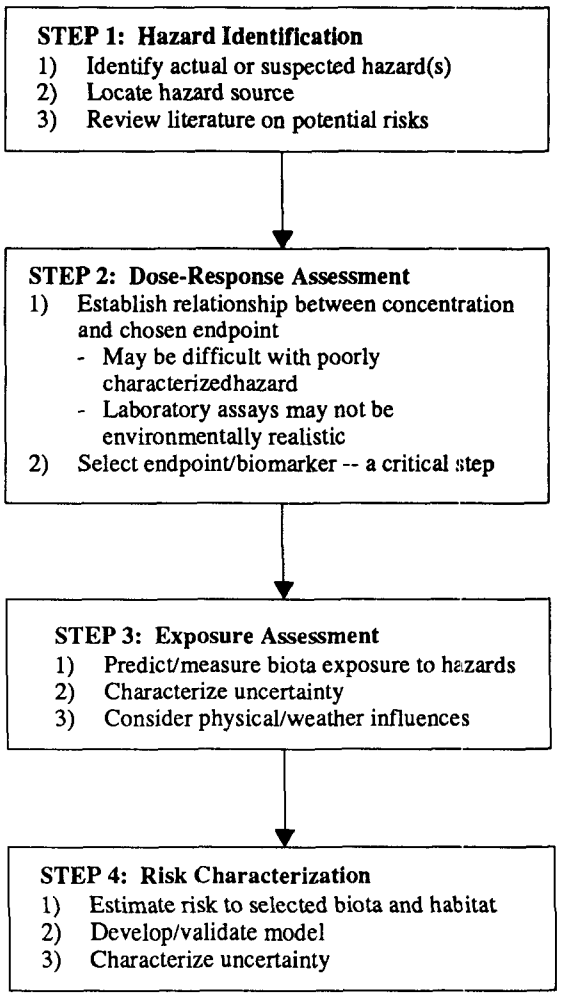

Figure 2. A skeleton risk assessment strategy for the Chesapeake Bay. This strategy will have to be modified for specific objectives as noted in the text. Dose-response assessment, step 2, may not be feasible for many regional assessments.

in Figure 2. While the strategy presented includes all of these, refinements will clearly be needed as new data and technology become available.

Estuarine Considerations

Many characteristics of the Bay contribute to the difficulty of developing a risk assessment strategy for it. Estuaries are often dominated by physical forces [54] and offer the greatest diversity in water composition [34, 64, 82]. Other unique features of the Bay also contribute to difficulties, including depth and width, salinity variation, flow characteristics, and nutrient inputs. Excellent reviews of these topics can be found elsewhere [64], but some major points are presented here. Although the depth of the Bay may reach 171 feet, the average depth is about 26 feet [82]. At this depth, the Bay is far more vulnerable to the effects of wind than bodies of deeper water, resulting in greater temperature fluctuations, higher waves, and decreased settling of suspended solids [64]. The Bay's deepest regions are in channels in its midstream. Width varies greatly, from 4 miles to 30 miles, and salinity also varies depending upon the season, precipitation, tidal cycle, and tributary flow [82]. The Bay drains a watershed area of some 64,000 square miles that is very diverse: 7% urban, 36% agriculture, 3% wetlands, and 54% forests. These features and diversity contribute to the Bay's environmental problems [10, 34, 82].

Factors adversely affecting the Bay that must be considered in the assessment process include sediment influx and transfer, nutrient input, wetlands degradation, and low-level toxicants [9, 10, 82]. The main difficulty presently facing the Bay is an increase in its primary production rate, already acknowledged to be among the highest of estuarine systems [82]. The presence of these stresses and their subsequent effects upon SAV and other components of the food chain have caused losses in habitat for the Bay, its tributaries, and adjacent wetlands.

Other large bodies of water, such as the Hudson River, the Great Lakes, and the Baltic Sea, have environmental problems resembling those of the Bay. Limburg et al. [54] examined the Hudson River, an estuarine system in New York, that has been used for power generation and suffered from toxic chemical release and dredging for highway construction. These activities affect that estuary in different and often unpredictable ways. Limburg and colleagues note that, of all the biota monitored in the Hudson, only fish seemed to be important to the public. They also stress the need for ecosystem testing using many species, along with careful assessments of the benthos, submerged aquatic vegetation, and the adjacent wetlands. These criteria apply to the Chesapeake Bay as well.

The Great Lakes also suffer from many of the same problems as the Bay, but to a large extent these have been overcome by international cooperation and intensive study. The environmental problems of the Lakes have been addressed by the International Joint Commission on the Great Lakes [28]. Problems are defined and solved by remedial action plans for the many regions that comprise the Lakes. Questions posed by the Science Advisory Board of the International Joint Commission for risk assessment, adapted for the Bay, are in Table 4. As is the case for the Bay, regional risk assessment for the Lakes has gaps in endpoint selection and uncertainty. A recent report from the National Wildlife Federation (NWF) [65] has stressed the problems and uncertainties of Lake environmental concerns. The report concluded that

"the problem of toxic pollution of Lake Michigan sport fish is more serious than has previously been reported." The report states that eating 11 meals of 30-inch lake trout in a lifetime exposes an individual to a 1-in-10,000 chance of developing cancer. The authors state the uncertainty of their prediction and the report appears not to have been peer-reviewed, but it does raise a point that has its parallel in the Bay, namely that human activities may come at a high cost.

Initial Strategy Development

Strategy development must define what impacts are to be examined, identify what resources are at risk as well as their relative values, determine what endpoints are scientifically and socially acceptable, and stipulate which testing procedures will be used to ascertain and predict environmental impact.

As previously noted, the Bay offers a complex environment for regional risk assessment. If such a regional assessment is to be successful, then several impacts must be studied concurrently. While the relative importance of these impacts is subject to debate, we have chosen the following for inclusion in this scheme: excess nutrients; sediment influx and transfer; and low-level toxicants, including herbicides and pesticides. These stresses have adversely affected SAV, finfish and shellfish, habitat, and water quality. Adverse effects on these resources have already resulted from eutrophication, primary production increases, changes in species distribution, habitat loss, and overharvesting. To quantify these impacts and ascertain their risk, proper endpoint selection is crucial.

Endpoint Selection Criteria

Obviously, the choice of endpoints and biomarkers must be relevant to the environment being assessed [5, 51, 74, 79] but will be under the influence of regulators, politicians, and other groups. Stress will have varied impacts on different ecosystems [7], but the majority of state-of-the-art biological tests for hazard assessment use single species as stress indicators. However, there are questions about the adequacy of this approach when such tests are used as predictive tools [18, 51].

The characteristics of the ideal endpoints for the Bay are described in Table 1. Potential structural endpoints for localized use in the Bay include species diversity, richness, range, recruitment, biomass, mortality, trophic structure, and fecundity. Extreme care must be used when selecting species for examination since spatial distribution, stress susceptibility, and economic or ecological relevance must be considered [54, 79, 81]. Biota such as SAV, oysters and other shellfish, plankton, benthic communities, and finfish (especially gamefish) should be used in the Bay and adjacent wetlands. Fish have often been used because of their economic and recreational importance [54], even though there are other organisms of comparable ecological importance. While examining the effects on fish of chronic exposure to toxics, Suter et al. [79] found the most sensitive effect was a reduction in fecundity, rather than effects on early life stages as is now being proposed by some regulators. Monitoring of birds associated with adjacent wetlands has been suggested for the Great Lakes and should serve well for the Bay also [3]. Bandurski suggests that amateur birdwatching groups can assist in the studies. Other structural endpoints, such as rates of loss for various wetland communities, are also in need of assessment.

While functional responses, such as enzyme inhibition, productivity, and nutrient cycling, have been somewhat useful in acute, localized assessments [29], they may not be especially useful in regional risk assessment since no consistent endpoints have been developed [80] and variability is a problem. Potential functional endpoints of use in the Bay include microbial and serum enzyme activities, behavioral assessments, net production, substrate utilization, and literally dozens of others that are reviewed elsewhere [25, 59, 60, 62, 69, 81].

Endpoints are not available that delineate selected parameter impacts when two or more stresses are present. While exceptions exist, such as metal accumulation in particular organs, organismal lesions, some enzymes, etc., these are difficult to apply in regional assessments. However, biomarker development should eventually provide the technology to improve the understanding of stress effects and delineation.

Large data bases exist for the Bay [57] that may aid in endpoint selection. While our choices of regional endpoints are not exhaustive, they should promote discussion. Our recommendations include assessing primary production via satellite [68]; using computerized geographical data bases to predict effects from agricultural and commercial activity in adjacent terrestrial areas [45]; monitoring SAV [10, 76]; determining wetland-associated bird populations; using fish communities as bioindicators [48]; delineating adjacent wetlands acreage to finer scales; determining the range and associated habitat quality for important species, such as shellfish, finfish, and SAV; and continuing to monitor water quality. Other methods, such as infrared monitoring and use of DNA, antibodies, and other biomarkers, may eventually be applicable to the Bay [36, 38, 61, 66], as will the use of computer-based risk assessment models [67].

Development of Risk Assessment Criteria

Once we have agreed on which resources are important

Table 4. Questions to consider in developing an environmental risk assessment strategy for the Chesapeake Bay.

1. How does the Chesapeake Bay compare to other regions with regard to toxics in the environment?
2. What is known about the fate and persistence of toxic chemicals in the Chesapeake Bay?
3. What are the most notable barriers that have prevented an ecosystem perspective in managing toxics in the Chespeake Bay?
4. To what degree can the toxic impacts observed in fish and other wildlife be used as an "early warning system" to protect human health and the ecosystem as a whole?
5. What general categories of toxics are of most concern in the Chesapeake Bay, and what are the relative toxicities of these substances at various levels of biological organization (i.e., populations, communities, ecosystem)?
6. Are there substantive differences between the levels of potential toxicants measured in the environment, e.g., what is the significance of 0.01 mg zinc, 0.1 mg, 1.0 mg, 10 mg, etc.?
7. Do we really have the methodologies and data bases to evaluate realistically the risk encountered by target populations through multiple and cumulative exposures in the Chesapeake Bay?
8. What are the effects of prolonged ingestion of fish and water containing trace levels of toxic chemicals on humans and other species?
9. Are there any examples of known injury to human health from toxic contaminants in the Chesapeake Bay?
10. What research methods are available to quantify the different patterns of toxic exposure risk and to identify potential interactive effects from combined chemical insults to the Chesapeake Bay ecosystem?
11. What are appropriate endpoints for monitoring ecosystem integrity and/or biological integrity?
12. What are the demographics of human populations consuming fish, shellfish, and other wildlife in the Chesapeake Bay ecosystem?
13. How does one identify critical subpopulations subject to the effect of toxic exposure under the assumption of known average populations?
14. How does one convert reactive interest in toxic chemicals (i.e., "not in my backyard") into proactive preventive efforts?
15. What are the benefits in cost, including concealed costs, in ignoring long-term burdens to society resulting from ecological damage for the sake of short-term gains with respect to economic exploitation of use resources?
16. How much are people willing to sacrifice (i.e., how much money will they pay) for good environmental quality and the prospect of long-term sustainable use?
17. Are present statutory frameworks reasonable and effective in protecting the Chesapeake Bay in view of the large data requirements and the frequent impossibility of meeting these requirements?
18. Are existing institutional frameworks adequate for development and appropriate interpretation of toxicity data for the Chesapeake Bay and for the management of biological, physical, and social dimensions of toxic risks?

19. Are there major differences between how risk is communicated by regulatory agencies and how it is perceived by the general public?

20. How can we do a better job of communicating risk, particularly in view of the "mixed messages" that the public gets from inconsistencies in guidelines and regulations?

21. How can we do a better job of lessening risks associated with contaminants in the environment; and how can preventive strategies be put in place that have as a basis a presumption of harm to the aquatic environment and to human health?

22. How can we learn to live with a system in which reduction of risk to even acceptable levels appears economically, technically, and politically unobtainable?

23. What implications for risk management are there in considering people (especially local populations) as parts of the impacted ecosystem?

24. How can we develop a better citizen understanding of ecological effects in order to encourage responsible individual behavior and generate political support for legislative action?

Source: John Cairns, Jr., is a member of the Great Lakes Science Advisory Board of the International Joint Commission and received permission from Dr. Peter C. Boyer, Secretary, to modify information from a report as it applies to the Chesapeake Bay.

---

to protect and have selected endpoints on which our decisions are to be based, we must quantify the amounts of sediment, nutrients, and toxic substances that our resources are being exposed to and determine how much exposure we will allow in the future. We must also seek to attain the policy of no net loss of wetland acreage and function. Fortunately, the Bay already has an extensive monitoring program. Ideally, a federal agency, such as EPA, should oversee the development and implementation of risk assessment so that interstate control can be maintained and political interference minimized.

Establishing acceptable limits for the various impacts that the Bay faces is not the purpose of this review. Currently, because of fiscal limitations, this is often done by using structure-activity relationships, single-species toxicity tests, and literature reviews. If we are to protect and restore the Bay, we must direct additional funds to agencies that will then be responsible for determining cause-and-effect relationships based upon scientific data from realistic experimental scenarios, including an increased use of sediment in toxicity tests. This will be an expensive, formidable task.

Once a model assessment scheme has been developed, and selected important resources and relevant endpoints have been be agreed upon, then monitoring of the Bay should allow assessment of its overall health and a prediction of future risk. Large amounts of data that are not assimilated into a comprehensive framework followed by definitive decisions and actions are useless. Such a coordinated framework must include components that will address data compilation, criteria formulation, and technology development. Obviously, new technology needs to be developed. Scientists, not politicians or managers, must decide what constitutes a healthy Bay; it will then be up to the politicians and managers to decide if such a goal is attainable based upon the collection of scientific, socioeconomic, and political considerations. The authors differ from others in believing scientists must be a part of the decision process so that the scientific component is not minimized in the final decision, as is now often the case.

Present procedures for regulating inputs into the Bay and assessing risk are primarily administered by the States of Maryland, Virginia, and Pennsylvania, and the District of Columbia under mandates from federal and state law and with the assistance of the EPA Chesapeake Bay Program [10, 11]. These management procedures have been criticized, especially those concerning future growth and development [83]. If the needs of the Bay, and not just the individual jurisdictions, are to be identified and addressed, then all parties must reevaluate their motives and goals.

Research Directives

If we are to go from a damage assessment philosophy to a risk assessment strategy for the Bay, crucial gaps in available technology must be identified and addressed. Currently, we are not able to delineate the effects of a particular stress when it is found in conjunction with several others; we are unsure of the long-term ramifications of low-level toxicant concentrations; we are ignorant of how toxicity exposure is affected by Bay sediments and biota; we need additional studies of the factors controlling bioavailability; we need to examine groundwater as a source of nutrients and pollutants to the Bay; and we need to know if replacement wetlands perform the same functions as natural wetlands.

We suggest that a long-term research plan be developed that identifies weaknesses in present knowledge [78] and prescribes what research is needed to correct those deficiencies. The following is presented as a listing of research needs over the next several years.

- Development of environmentally realistic toxicity tests, both acute and chronic, to deal with toxics and herbicides. Whether these take the form of single or multispecies tests, provisions for varied trophic levels and sediment inclusion must be developed.
- Determination of the feasibility of using present or improved toxicity tests as predictive models in microcosm or mesocosm systems.
- Increased development and use of biomarkers [35, 38, 44]. The utility of these tests appears promising but needs additional exploration and development.
- Further quantification of the energy flow dynamics in the Bay to increase understanding of the Bay watershed as an entity in itself.
- A better understanding of why some ecosystems are relatively resilient to impacts while others are severely affected.
- Refinements in and adaptation of ecological population theory as it pertains to risk assessment. The Oak Ridge National Laboratory is at the forefront of this area and should continue to be included in Bay ecological assessment development.
- Increased research into the quantity and quality of adjacent wetland areas, their capabilities in toxicant removal and as species habitat, as well as the utility of created wetlands as replacements for those destroyed by human activities.
- An enhanced understanding of the importance of groundwater in both contributing to (nonpoint pollution) and resolving (water purification in aquifers) some of the Bay's problems.

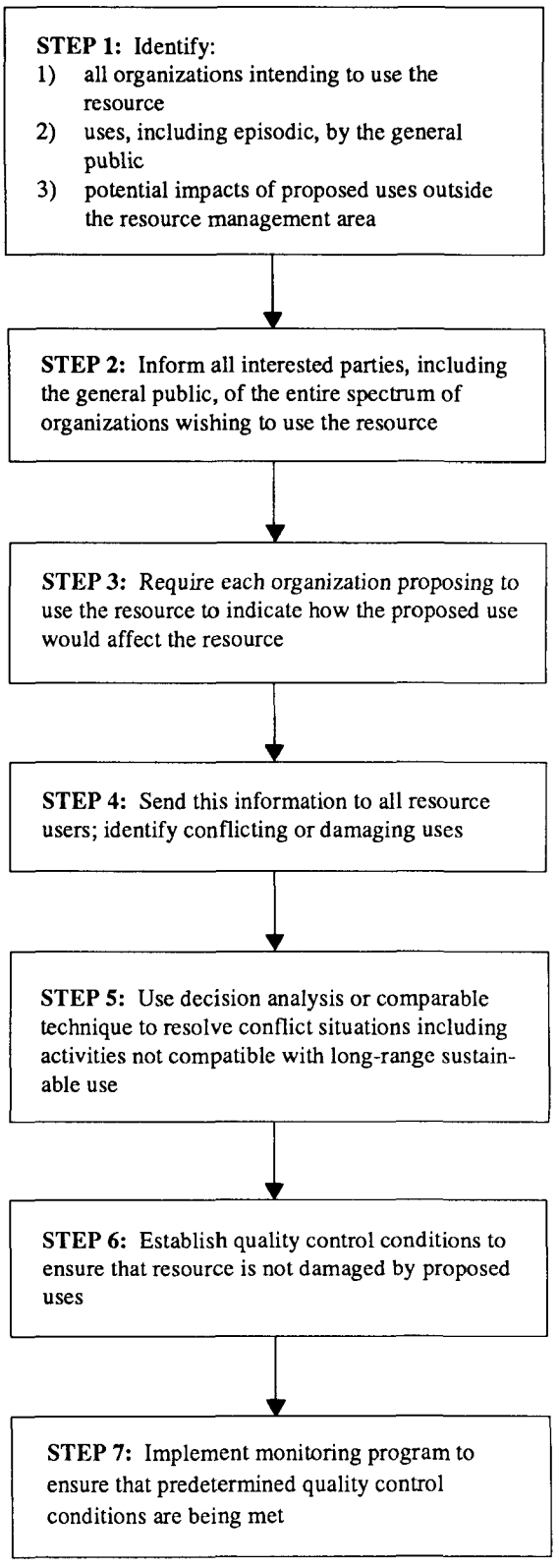

Figure 3. An integrated resource management protocol for the Chesapeake Bay. Such a protocol will allow an integrated approach to environmental problems facing the Bay.

Integrated Resource Management

If we are to succeed at predictive risk assessment, we must use integrated approaches incorporating all of these techniques, as well as those yet to be developed [21, 31].

Inevitably, management decisions involve an interaction of science, politics, and economics. In fact, risk assessment reflects a tension between the two basic objectives of regulation: factual accuracy and result orientation [71].

Integrated resource management has many benefits, including cost-effectiveness, long-term resource protection, enhanced potential for multiple use of resources, more rapid and effective restoration of damaged resources, and a reduction of conflicts between the various sectors involved in decision-making [20]. Integrated resource management should follow a stepwise process (Figure 3) with the primary objective being the protection of the ecological integrity of the Chesapeake Bay so that it will be suitable for sustained use in a variety of ways. To do this, adequate assessment of realized and potential risks must be executed. Legislators need to be provided with specific environmental quality objectives rather than general goals.

Summary

To develop an ideal risk assessment scheme for the Bay will require additional research and development of technologies, as described, that will allow hazard and exposure assessment beyond what is presently available. However, with available technology, we can establish a framework for regional Bay risk assessment. This will require collaboration and compromise. We must move from damage assessment to risk assessment so we can strive to predict the fate of the Bay. The Bay is a complex system, but by identifying the problems facing the Bay (habitat loss, species distribution shifts, changes in energy flow dynamics, nutrient enrichment, low-level toxics, wetlands loss, overharvesting), deciding how to measure effects (endpoints), establishing what are "acceptable" effects (criteria), and using risk assessment methodologies to forecast how selected resources will be affected, we can begin to control the ultimate environmental fate of the Bay.

Central issues in risk assessment remain: whether risk is significant, who is responsible for proving that significance, how to eliminate tension between component groups in the risk assessment process, and how much risk is acceptable [71]. Additional problems also remain for Bay risk assessment: no agreement as to what a healthy ecosystem is, minimal consensus as to what constitutes a relevant endpoint, the need to identify and quantify uncertainty, and a lack of methodologies applicable to a large watershed. Finally, large gaps exist in our best available technology, and relevant research must be initiated to address these.

It is hoped that this review will increase the awareness of the reader and stimulate discussion as to the methods and limitations of risk assessment. While we lack technology to achieve "ideal" risk assessment, we must not wait until that technology is developed to proceed. We must crawl before we walk, and walk before we can run. With regard to regional ecological risk assessment for the Chesapeake Bay, we are just starting to stand. Only through direction, cooperation, and compromise will the "ideal" ecological assessment be attainable.

Acknowledgements

The authors are indebted to Rich Batiuk, Barbara Niederlehner, Robert Atkinson, and two anonymous reviewers for their editorial suggestions. The cooperation of Rick Reynolds of CRC is also appreciated, as is the editorial assistance of Darla Donald.

References

1. Allen, F.W.; Hinckley, D. Comparing communication of ecological and health risks. Abstracts of the Tenth Annual Meeting of the Society of Environmental Toxicology and Chemistry; 82:81, 1989.
2. Anderson, F.R.; Mandelker, D.R.; Tarlock, A.D. Environmental protection: law and policy. Boston, MA: Little, Brown & Co.; 1984.
3. Bandurski, B.L. Biodancing: the 1-2-3 of Great Lakes monitoring. In: Caldwell, L.K., ed. Perspectives on ecosystem management for the Great Lakes. Albany, NY: State University of New York Press; 1988:191-241.
4. Barnthouse, L.W.; DeAngelis, D.L.; Gardner, R.H.; O'Neill, R.V.; Powers, C.D.; Suter, G.W., II; Vaughn, D.S. Methodology for environmental risk analysis. Oak Ridge, TN: Oak Ridge National Laboratory; 1982; ORNL/TM-8167.
5. Barnthouse, L.W.; Suter, G.W. Risk assessment ecology. Mech. Eng.; Nov. 1984.
6. Barnthouse, L.W.; O'Neill, R.V.; Bartell, S.M.; Suter, G.W. Population and ecosystem theory in ecological risk assessment. In: Poston, T.M.; Purdy, R., eds. Aquatic toxicology and environmental fate: 9th vol., ASTM STP 921; Philadelphia, PA: American Society

for Testing and Materials; 1986:82-96.

7. Barrett, G.W.; Rosenberg, R., eds. Stress effects on natural ecosystems. Chichester, England: John Wiley & Sons; 1981.
8. Bella, D.A. Impact assessment and future generations. Coastal Zone Manag. J. 4:249-267; 1978.
9. Biggs, R.B.; DeMoss, T.B.; Carter, M.M.; Beasley, E.L. Susceptibility of U.S. estuaries to pollution. Rev. Aquat. Sci. 1:189-207; 1989.
10. Bonner, P.A. The evolution of the Chesapeake Bay Program. Environ. Profess. 10:110-120; 1988.
11. Bostater, C.R.; Pepino, R.V.; Roland, J.V. Monitoring, analysis, and control of toxic and hazardous materials in the Chesapeake Bay. In: Eaton, J.G.; Parrish, P.R.; Hendricks, A.G., eds. Aquatic toxicology, ASTM STP 707; Philadelphia, PA: American Society for Testing and Materials; 1980:142-154.
12. Cairns, J., Jr. Estimating the assimilative capacity of water ecosystems. In: Sharma, R.K.; Buffington, J.D.; McFadden, J.T., eds. The biological significance of environmental impacts. Springfield, VA: National Technical Information Service; 1976:173-189.
13. Cairns, J., Jr. Aquatic ecosystem assimilative capacity. Fisheries 2(2):5; 1977.
14. Cairns, J., Jr. Quantification of biological integrity. In: Ballentine, R.K.; Guarraia, L.J., eds. The integrity of water. Washington, D.C.: US EPA, Office of Water and Hazardous Materials; 1977. Stock no. 055-001-01068-1.
15. Cairns, J., Jr. Hazard evaluation. Fisheries 3(2):2-4; 1978.
16. Cairns, J., Jr. Estimating hazard. BioScience 30:101-107; 1980.
17. Cairns, J., Jr. Discussion of: A critique of assimilative capacity. J. Water Poll. Control Fed. 53(11):1653-1655; 1981.
18. Cairns, J., Jr. Are single species toxicity tests alone adequate for estimating environmental hazard? Hydrobiologia 100:47-57; 1983.
19. Cairns, J., Jr. Factors moderating toxicity in surface waters. In: Wilson, J., ed. The fate of toxics in surface and ground waters. Philadelphia, PA: Proceedings of the 2nd national water conference, Acad. Nat. Sci. Phil.; 1984:49-64.
20. Cairns, J., Jr. What is meant by validation of predictions based on laboratory toxicity tests? Hydrobiologia 137:271-278; 1986.
21. Cairns, J., Jr. Integrated resource management: the challenge of the next ten years. In: Adams, W.J.; Chapman, G.A.; Landis, W.G., eds. Aquatic toxicology and hazard assessment: 10th vol., ASTM STP 971; Phil., PA: American Society for Testing and Materials; 1988:559-566.
22. Cairns, J., Jr. Politics, economics, science—going beyond disciplinary boundaries to protect aquatic ecosystems. In: Evans, M.S., ed. Toxic contaminants and ecosystem health: a Great Lakes focus. New York: John Wiley & Sons; 1988:1-16.
23. Cairns, J., Jr. Predicting, validating, and monitoring effects of toxics on large lake ecosystems. In: Schmidtke, N.W., ed. Toxic contamination in large lakes, vol. II. Chelsea, MI: Lewis Publishers; 1988:291-316.
24. Cairns, J., Jr. Gauging the cumulative effects of development activities on complex ecosystems. In: Gosselink, J.G.; Lee, L.C.; Muir, T., eds. Ecological processes and cumulative impacts: illustrated by bottomland hardwood wetland ecosystems. Chelsea, MI: Lewis Publishers, Inc.; 1990.
25. Cairns, J., Jr.; Pratt, J.R., eds. Functional testing of aquatic biota for estimating hazards of chemicals; ASTM STP 988; Phil., PA: American Society for Testing and Materials; 1988.
26. Cairns, J., Jr.; Orvos, D.R. Ecological consequence assessment: predicting effects of hazardous substances upon aquatic ecosystems using ecological engineering. In: Mitsch, W.J.; Jorgensen, S.E., eds. Ecological engineering: an introduction to ecotechnology. New York: John Wiley & Sons; 1989:409-442.
27. Cairns, J., Jr.; Dickson, K.L.; Herricks, E.E. The recovery and restoration of damaged ecosystems, a challenge for action: symposium analysis. In: Cairns, J., Jr.; Dickson, K.L.; Herricks, E.E., eds. Recovery and restoration of damaged ecosystems. Charlottesville, VA: University Press of Virginia; 1977:522-525.
28. Caldwell, L.K., ed. Perspectives on ecosystem management for the Great Lakes. Albany, NY: State University of New York Press; 1988.
29. Calow, P.; Maltby, L. Review of methods for assessing the impact of episodic pollution. Medmenham, England: WRC Environment; 1987.
30. Campbell, I.C. A critique of assimilative capacity. J. Water Poll. Control Fed. 53(5):604-607; 1981.
31. Chinchilli, J.E. Chesapeake Bay restoration program: is an integrated approach possible? In: Water-policy issues related to the Chesapeake Bay. Blacksburg, VA: VA Water Resources Research Ctr.; 1989:6-9.
32. Davis, F.W. Historical changes in submerged macrophyte communities of upper Chesapeake Bay. Ecology 66(3):981-993; 1985.
33. Dickson, K.L.; Rodgers, J.H. Assessing the hazards of effluents in the aquatic environment. In: Bergman, H.L.; Kimerle, R.A.; Maki, A.W., eds. Environmental hazard assessment of effluents. New York: Pergamon Press; 1986:209-227.
34. EPA. Chesapeake Bay: introduction to an ecosystem. Washington, D.C.: US EPA; 1987.

35. EPA. Review of ecological risk assessment methods, Fairfax, VA: ICF, Inc.; 1988; USEPA/230-10-88-041.
36. EPA. Appendix B: Strategies for exposure assessment research, report of the subcommittee of ecological effects. Washington, D.C.: US EPA; 1988; SAB-EC-88-040C.
37. EPA. Risk assessment guidance for Superfund, volume II, environmental evaluation manual. Washington, D.C.: US EPA; 1989; EPA/540/1-89/001.
38. EPA. Ecological assessment of hazardous waste sites: a laboratory and field reference. Washington, D.C.: US EPA; 1989; EPA/600/3-89/013.
39. Erickson, P.A. Environmental impact assessment: principles and applications. New York: Academic Press; 1979.
40. Fisher, N.S. On the differential sensitivity of estuarine and open-ocean diatoms to exotic chemical stress. Am. Nat. 111:871-895; 1977.
41. Gaylor, D.W. Quantitative risk estimation. In: Cothern, C.R.; Mehlman, M.A.; Marcus, W.L., eds. Risk assessment and risk management of industrial and environmental chemicals. Princeton, NJ: Princeton Scientific; 1988:23-43.
42. Grima, A.P.; Timmerman, P.; Fowle, C.D.; Byer, P. Risk management and EIA: research needs and opportunities. Quebec, Canada: Canadian Environmental Assessment Research Council; 1986.
43. Hann, R.W., Jr.; Cairns, J., Jr. Episodic exposures. In: National Academy of Sciences, compilers. Principles for evaluating chemicals in the environment. Washington, D.C.: National Academy Press; 1975:263-268.
44. Hartwell, I. Population risk assessments based on toxicity testing. In: Chesapeake Bay ambient toxicity assessment workshop report. Annapolis, MD: Environmental Protection Agency, Chesapeake Bay Program (in press); technical series report.
45. Hellmund, P.A. Landscape planning and the Bay: a case study. In: Kuo, C.Y.; Younos, T.M., eds. Effects of upland and shoreline land use on the Chesapeake Bay. Williamsburg, VA: Ches. Bay Research Conf.; 1986:149-157.
46. Hunsaker, C.T.; Graham, R.L.; Suter, G.W.; O'Neill, R.V.; Jackson, B.L.; Barnthouse, L.W. Regional ecological risk assessment: theory and demonstration. Oak Ridge, TN: Oak Ridge National Laboratory; 1989; ORNL/TM-11128.
47. Jacobs, D.; Haberman, D.; Smith, D.; Swartz, D.; Sigel, E.; Adams, M., compilers. Chesapeake Bay environmental data directory. College Park, MD: Univ. of MD; 1987.
48. Karr, J.R.; Fausch, K.D.; Angermeier, P.L.; Yant, P.R.; Schlosser, I.J. Assessing biological integrity in running waters: a method and its rationale. Champaign, IL: Illinois Natural History Survey, special publication 5; 1986.
49. Karr, J.R.; Yant, P.R.; Fausch, K.D.; Schlosser, I.J. Spatial and temporal variability of the index of biotic integrity in three midwestern streams. Trans. Am. Fish. Soc. 116:1-11; 1987.
50. Kemp, W.M.; Lewis, M.R.; Cunningham, J.J.; Stevenson, J.C.; Boynton, W.R. Microcosms, macrophytes, and hierarchies: environmental research in the Chesapeake Bay. In: Giesy, J.P., ed. Microcosms in ecological research. Springfield, VA: National Technical Information Service; 1980:911-936.
51. Kimball, K.D.; Levin, S.A. Limitations of laboratory bioassays: the need for ecosystem-level testing. BioScience 35:165-171; 1985.
52. Kimerle, R.A.; Adams, W.J.; Grothe, D.R. A tiered approach to aquatic safety assessment of effluents. In: Bergman, H.L.; Kimerle, R.A.; Maki, A.W., eds. Environmental hazard assessment of effluents. New York: Pergamon Press; 1986:247-264.
53. Levenson, J.B.; Stearns, F.W. Application of diversity to regional ecological assessment: a review with recommendations. Argonne, IL: Argonne National Laboratory; 1980; ANL/AA-21.
54. Limburg, K.E.; Levin, S.A.; Harwell, C.C. Ecology and estuarine impact assessment: lessons learned from the Hudson River (USA) and other estuarine experiences. J. Environ. Mgmt. 22:255-280; 1986.
55. Lloyd, R. The use of concentration-response relationship in assessing acute fish toxicity data. In: Dickson, K.L.; Maki, A.W.; Cairns, J., Jr., eds. Analyzing the hazard evaluation process. Washington, D.C.: American Fisheries Society; 1979:58-61.
56. Mackiernan, G.B. Sources and impacts of nutrients in the Chesapeake Bay. In: Coale, C.W.; Marshall, J.P.; Kerns, W.R., eds. Land use and the Chesapeake Bay. Blacksburg, VA: Virginia Polytechnic Institute; 1985:1-20.
57. Magnien, R.; Siegfried, R.; Summers, R. Chemical and physical monitoring of the Chesapeake Bay mainstem and tributaries. In: Lynch, M.P.; Krome, E.C., eds. State of the Chesapeake Bay, second annual monitoring report. Gloucester Point, VA: Chesapeake Research Consortium; 1987:3-19.
58. Maki, A.W. An analysis of decision criteria in environmental hazard evaluation programs. In: Dickson, K.L.; Maki, A.W.; Cairns, J., Jr., eds. Analyzing the hazard evaluation process. Washington, D.C.: American Fisheries Society; 1979:83-100.
59. Mehrle, P.M.; Mayer, F.L. Biochemistry/physiology. In: Rand, G.M.; Petrocelli, S.R., eds. Fundamentals of aquatic toxicology. Washington, D.C.: Hemisphere Publishing; 1985:264-282.
60. Meyers, T.R.; Hendricks, J.D. Histopathology. In: Rand, G.M.; Petrocelli, S.R., eds. Fundamentals of

aquatic toxicology. Washington, D.C.: Hemisphere Publishing; 1985:283-331.

61. Mlot, C. Global risk assessment. BioScience 39:428-430;1989.
62. Moore, M.N.; Lowe, D.M.; Livingstone, D.R.; Dixon, D.R. Molecular and cellular indices of pollutant effects and their use in environmental impact assessment. Wat. Sci. Tech. 18:223-232; 1986.
63. National Research Council. Risk assessment in the federal government: managing the process. Washington, D.C.: National Academy Press; 1983.
64. National Research Council. Fundamental research on estuaries: the importance of an interdisciplinary approach. Washington, D.C.: National Academy Press; 1983.
65. National Wildlife Federation. Lake Michigan sport fish: should you eat your catch? Ann Arbor: Great Lakes Natural Resource Ctr.; 1989; MSD03.
66. NOAA. 2nd interagency workshop on in-situ water quality sensing: biological sensors. Rockville, MD: NOAA; 1980.
67. Onishi, Y.; Olsen, A.R.; Parkhurst, M.A.; Whelan, G. Computer-based environmental exposure and risk assessment methodology for hazardous materials. J. Haz. Mat. 10:389-417; 1985.
68. Perry, M.J. Assessing marine primary production from space. BioScience 36:461-467; 1986.
69. Rand, G.M. Behavior. In: Rand, G.M.; Petrocelli, S.R., eds. Fundamentals of aquatic toxicology. Washington, D.C.: Hemisphere Publishing; 1985:221-263.
70. Rand, G.M.; Petrocelli, S.R., eds. Fundamentals of aquatic toxicology. Washington, D.C.: Hemisphere Publishing; 1985.
71. Ricci, P.F.; Molton, L.S. Risk and benefit in environmental law. Science 214:1096-1100; 1981.
72. Ricklefs, R.E. Ecology. New York: Chiron Press; 1979.
73. Rosenberg, D.M.; Resh, V.H.; Balling, S.S.; Barnby, M.A.; Collins, J.N.; Durbin, D.V.; Flynn, T.S.; Hart, D.D.; Lamberti, G.A.; McElravy, E.P.; Wood, J.R.; Blank, T.E.; Schultz, D.M.; Marrin, D.L.; Price, D.G. Recent trends in environmental impact assessment. Can. J. Fish. Aquat. Sci. 38:591-624; 1981.
74. Schaeffer, D.J.; Herricks, E.E.; Kerster, H.W. Ecosystem health: I. Measuring ecosystem health. Environ. Management 12(4):445-455; 1988.
75. Society of Environmental Toxicology and Chemistry, compilers. Research priorities in environmental risk assessment, workshop report. Washington, D.C.: Society of Environmental Toxicology and Chemistry; 1987.
76. Simons, J.D.; Orth, R.J. Distribution and abundance of submerged aquatic vegetation. In: Lynch, M.P.; Krome, E.C., eds. State of the Chesapeake Bay, second annual monitoring report. Gloucester Point, VA: Chesapeake Research Consortium; 1987:145-151.
77. Sprague, J.B. Factors that modify toxicity. In: Rand, G.M.; Petrocelli, S.R., eds. Fundamentals of aquatic toxicology. Washington, D.C.: Hemisphere Publishing; 1985:124-163.
78. Strachan, W.M.J. Research needs in support of the assessment process. In: Schmidtke, N.W., ed. Toxic contamination in large lakes, vol. I. Chelsea, MI: Lewis Publishers; 1988:343-359.
79. Suter, G.W.; Rosen, A.E.; Linder, E.; Parkhurst, D.F. Endpoints for responses of fish to chronic exposures. Environ. Tox. Chem. 6:793-809; 1987.
80. Suter, G.W.; Barnthouse, L.W. Population risk assessments based upon toxicity testing. In: Chesapeake Bay ambient toxicity assessment workshop report. Annapolis, MD: Environmental Protection Agency, Chesapeake Bay Program (in press); technical series report.
81. Swartz, R.C. Biological criteria of environmental change in the Chesapeake Bay. Ches. Sci. 13:S17-S41; 1972.
82. Wright, D.A.; Phillips, D.J.H. Chesapeake and San Francisco Bays: a study in contrasts and parallels. Marine Pollution Bull. 19(9):405-413; 1988.
83. Year 2020 Panel. Population growth and development in the Chesapeake Bay watershed to the year 2020. Chesapeake Executive Panel; 1988.