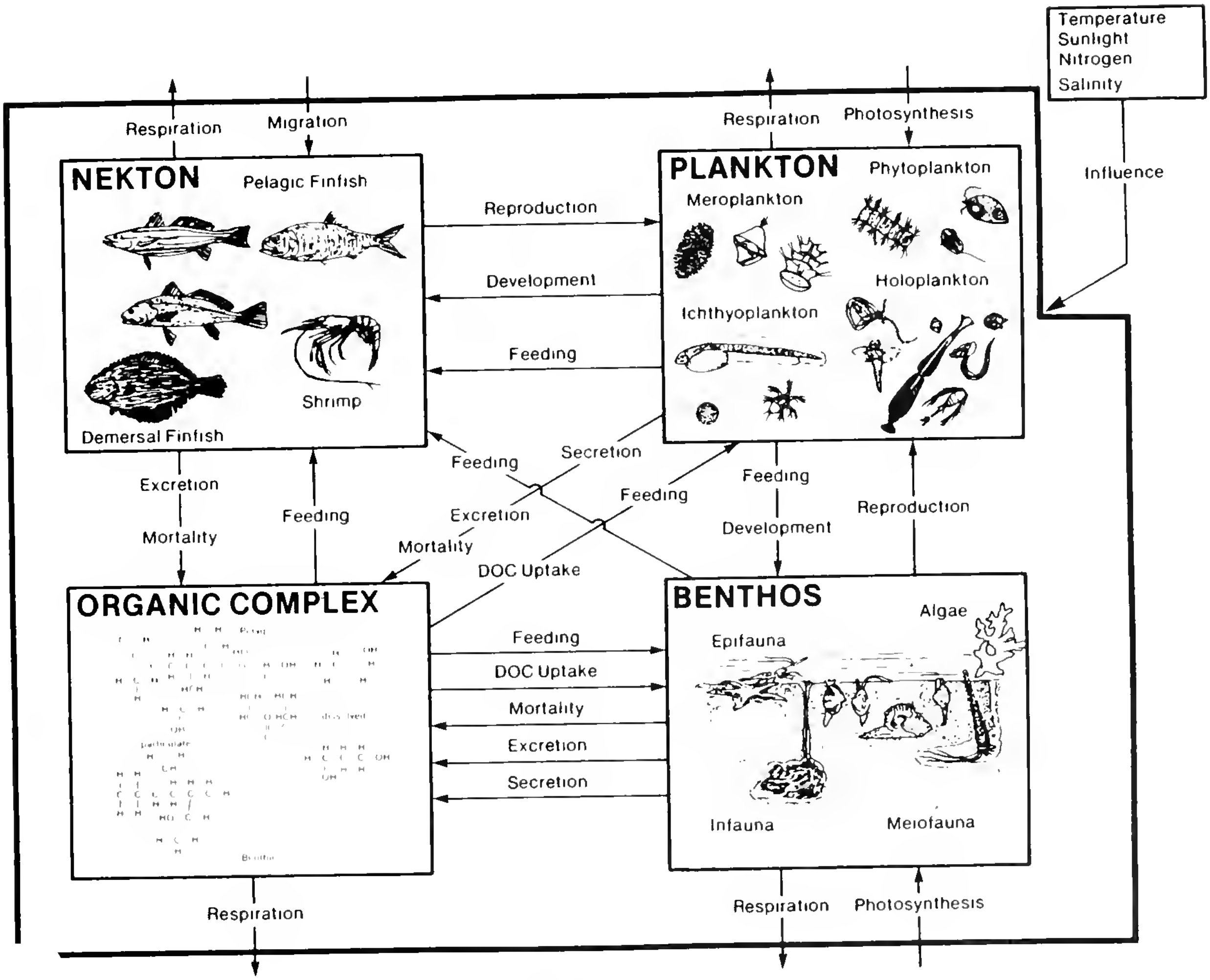
Marine Ecosystem Modeling

Proceedings From a Workshop Held April 6-8, 1982



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Proceedings From a Workshop Held April 6-8, 1982 Frederick, Md.

Kenneth W. Turgeon, Editor

Washington, D.C.
August 1983

U.S. DEPARTMENT OF COMMERCE
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FOREWORD

Marine ecosystem modeling has evolved significantly since Fleming (1939)[1] formulated and published his phytoplankton population dynamics model for the English Channel. Aside from the variation in modeling approaches and techniques, the change has been most apparent in two areas: (1) model complexity and resolution and (2) model application. Model development in the first area is a natural outgrowth of computer technology and advances in marine ecology. The second area, model application, is a direct consequence of the national environmental awareness of the late 1960s and the ensuing environmental legislation of the early 1970s. A new category of models has been developed--predictive models for application to environmental impact assessment and natural resource management.

Management of marine resources, especially living resources, within a framework of sound environmental and economic practices and principles is difficult and complex. Vast amounts of multidisciplinary data must be collected, analyzed, interpreted, assimilated, and forged into a reasonable and workable management plan. These plans should (1) prevent irreversible or unacceptable degradation of the environment, (2) deter overexploitation of valuable resources, (3) minimize multiple use conflicts, and (4) provide alternative management strategies designed to protect the environment. The decisionmaking process is complicated by the need to develop plans before the environment is insulted.

Resource managers could easily be overwhelmed by the volume and diversity of data and information necessary to today's environmental planning process. They need tools which will integrate existing knowledge into useful frames of reference. Additionally, they need tools which will allow them to assess in advance possible consequences of the various management options. Ecosystem modeling is such a tool. Obviously it is not the only tool, but it is one which can be of great assistance to the decisionmaker. Wise use of ecosystem modeling requires awareness of the limitations and capabilities of modeling. The participants in this workshop have provided, and, I hope will continue to provide much needed guidance and insight.

John V. Byrne
Administrator
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[1] Fleming, R. H. 1939. The control of diatom population by grazing. J. Cons. Perm. Expl. Mer. 14:210-227.

PREFACE

This workshop, held on April 6-8, 1982 in Frederick, Maryland, was convened and sponsored by the Environmental Data and Information Service (EDIS)[1] of NOAA. EDIS has long been interested and involved in the application of quantitative mathematical modeling to problems of resource management and assessment of environmental and man-induced impacts on valuable national resources. Its Center for Environmental Assessment Services (CEAS)[2] regularly uses climatic, oceanographic, and ecological models as scientific tools to provide decision assistance to managers of critical national resources, especially food and marine resources. Quite often these models are applied in a predictive or forecasting mode to enable managers to consider the consequences of various environmental scenarios, devise alternative management strategies, and develop contingency plans. Like most users of scientific tools, we are constantly on the lookout for ways to improve those tools, expand their applications, and ensure that they are well-suited to their intended use.

The major objective of this workshop was to bring together research scientists, model builders, model users, resource economists, resource managers, and administrators to discuss and assess the current and future role of ecosystem modeling as a useful and practical tool in marine environmental impact assessment and in the development and implementation of management strategies and policies for conservation and wise utilization of maine resources.

Participants were invited from government, academia, research institutions, and private industry. The composite blend of diverse scientific and management/administrative expertise from both public and private sectors allowed for an open and critical dialogue on the issues which were addressed and, perhaps more importantly, ensured that the workshop findings would represent a balanced perspective as opposed to a limited or biased view. The findings reported in these proceedings may have been significantly different had only single guild of professionals (e.g., modelers or resource managers) been involved in the deliberations.

The workshop was divided into three sessions: (1) invited papers, (2) working panels, and (3) a plenary session. The invited papers were presented on the first day and provided a valuable information base on a broad spectrum of marine ecosystem modeling topics directly pertinent to the workshop objectives. Four working panels composed of 12 to 13 panelists each convened on the morning of the second day. It is worth mentioning that the panels also represented a diverse mix of professional background and expertise; no panel was dominated by experts in the topic area. This "within panel blending" was in keeping with our desire to provide a balanced consensus of viewpoints and opinions. Each panel addressed a general area of marine ecosystem modeling relevant to resource

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[1] EDIS was merged with the National Earth Satellite Service (NESS) as part of the NOAA reorganization which was effective in December 1982. The new service is the National Environmental Satellite, Data, and Information Service (NESDIS).

[2] CEAS is now the Assessment and Information Services Center (AISC).

management and environmental impact assessment. These areas of discussion are (1) Panel A - Ecosystem Modeling as a Fisheries Management Tool; (2) Panel B - Ecosystem Modeling as an Environmental Management Tool; (3) Panel C - Integration/Linkage of Biological and Physical Ecosystem Models; and (4) Panel D - Integrating Ecosystem Modeling into a Socio-Economic Framework. Although the panels were requested to consider a given list of several issues or questions as it may pertain to their general topic area, the panels were free to follow their own agendas for debate and discussion. The key charge to the panels was that their deliberations should be focused towards the enhancement of ecosystem modeling as a tool for marine resource management and impact assessment planning and policy. The plenary session met on the morning of the third day. The individual panel reports were presented and opened to discussion by the audience.

We (EDIS) are fully cognizant that the panels were not able to comprehensively address their topic areas in only one day. As a matter of fact, any one of the topic areas, especially that of Panel D, could by itself have been the basis for a several days, multipanel workshop of its own. The results of this workshop certainly are not the definitive treatise on marine ecosystem modeling. However, the participants were enthusiastic and diligent in tackling their tasks. Because of their tireless efforts, these proceedings should provide useful information for future assessment, development, and application of ecosystem modeling in a marine resource management context.

Kenneth W. Turgeon
Workshop Chairman

ACKNOWLEDGMENTS

The able and willing assistance of the many people who were involved in the planning and convening of this workshop and in the preparation and publication of the workshop proceedings is gratefully acknowledged. While it is not possible to cite all these persons individually, there are several whose efforts were instrumental to the successful completion of the overall project. Commander Leonard "Slim" Pickens of the NOAA Corps provided much needed help in the early planning and preparation stages. Ms. Cindy Marston did an admirable job in providing logistics and coordination support. Dr. Peter Grose provided editorial assistance as well as being involved in the planning and preparation phases. Ms. Lora Birch and Ms. Marcia Collie accomplished the arduous task of transcribing the workshop manuscripts into a standardized type and format. Mr. Jack Ellis handled all the publication arrangements. Lastly, special thanks are sincerely extended to Dr. Robert Reeves for his outstanding and valuable assistance throughout all phases of the undertaking.

SUMMARY

The consensus of the workshop was that mathematical ecosystem modeling has proven to be a useful and practical scientific tool for addressing marine environmental assessment and resource management concerns. Modeling is a valuable mechanism for quantifying and integrating multidisciplinary data into a meaningful information base which would otherwise have been unattainable. Models allow decisionmakers to evaluate a priori the consequences of various actions and management strategies. Viewed from this perspective, virtually all modeling efforts can be of some value to decisionmakers provided they are timely and cost effective. However, the continued role of ecosystem modeling in a management/assessment context will require modelers to develop new methodologies which will make modeling more relevant to the specific information needs of managers, decisionmakers, and other users of modeling results. Two key areas of special importance are (1) restructuring traditional modeling approaches to incorporate methodologies suitable for addressing social and economic consequences of environmental impacts and resource management strategies, and (2) developing appropriate stochastic methodologies which will provide users with probability of occurrence, risk assessment information.

The workshop cautioned that ecosystem modeling results should never be the sole or even the primary basis for a management/assessment decision. Models, regardless of their apparent complexity and exhaustiveness, are simplified mathematical representations of natural systems and, as such, they are imprecise mimics and predictors of reality. It is unreasonable to expect that any model will ever provide definitive results free of uncertainty to complex and often multi-objective management/assessment issues. While these points may seem trivial and obvious, they are crucial to the success of any modeling project. Management expectations of modeling efforts must be tempered by recognition and acceptance of the many factors which limit the predictive power of models. Models are tools which provide insight to environmental/ecological relationships; they do not provide answers per se, and they are not ends in themselves.

The workshop was unanimous in its appraisal that the concept of holistic, site-adaptable and problem-adaptable generic ecosystem models was neither a prudent nor feasible management goal to pursue either now or in the foreseeable future. The necessity to incorporate numerous untested assumptions and theoretical relationships into the overly complex structure of such universal application models would completely erode any potential predictive effectiveness and preclude any usefulness to real world management/assessment needs. Rather, the general feeling was that ecosystem models are best suited to their intended purposes when tailored to meet the site-specific and problem-specific nature of the management/assessment concern. Model complexity should be limited to that which is necessary to provide the required information.

A broad spectrum of diverse modeling approaches and techniques is available to the modeler. The panels reviewed these in generalized terms and found that each has its advantages and disadvantages and must be assessed in light of the particular situation and application. In any case, the suitability of a given approach/technique to a given problem must be determined not by personal preference but by objective criteria such as (1) the quality and resolution of data available to construct the model, (2) the program objectives of the

management/assessment issue, (3) the resources available to develop, implement, and operate the model, and (4) the capability of the various modeling methodologies to provide the required information within known and acceptable confidence limits.

The workshop perceived a definite need for modelers and managers to collaborate and work closely together through all phases of the modeling project as well as the overall program to assure that modeling efforts are responsive to the program objectives and pertinent to the information needs of the decisionmakers. Modeling proceeds through an iterative process which continually redefines the conceptual and functional structure of the model. Similarly, program objectives may be redefined and refocused in light of new data. Modelers and managers must be continuously aware of each other's sphere of limitations and options. Modeling activities which take place in isolation from the other aspects of the overall management/assessment program will lead to misunderstandings and misconceptions about what information is required of the model and what information can be provided by the model. As a result, the model's utility will be less-than-satisfactory. Thus, it is imperative that modelers and managers interact every step of the way <u>including</u> model operation and interpretation of the results. This interaction should involve others associated with the program (e.g., field scientists, economists, demographers) and proceed through well-defined stages (see Panel B's report for a detailed treatment).

Lack of critical data and knowledge of basic ecological processes and interactions including pollutant impacts and synergisms are major impediments to the overall effectiveness of ecosystem modeling as a management and decision-making tool. The participants concurred that the quality of the data base on which a model is built is a major factor influencing the confidence limits of the modeling results. It was agreed that the value of ecosystem modeling as a marine resource management and impact assessment tool would be greatly enhanced if crucial data gaps were filled. For example, pre- and post-recruitment multispecies ecosystem models have virtually no current application to fisheries management concerns because production, mortality, and dispersal processes (both biotic and abiotic) which affect egg, larval, and juvenile survival are poorly known. A quantitative understanding of these processes is mandatory before fisheries models as resource management tools can progress beyond the more traditional single species, population dynamics type of models currently in use.

With regard to environmental quality ecosystem modeling, key research priorities which were identified are (1) fluxes and processes of material exchange between estuaries and coastal waters, sediments and water column, and sediments and organisms, and (2) pollutant effects on organismal physiology, reproduction, and survival for different life stages. Participants expressed the viewpoint that ecosystem models used as research tools to test and formulate hypotheses of causal relationships and to identify data needs are of as much value to long-term management and assessment objectives as are predictive models. Modelers, managers, and administrators should not lose sight of this important aspect of ecosystem modeling.

In addition to ecological data needs, there was general recognition that algorithms which will translate ecosystem modeling results and environmental quality criteria into economic and societal terms for use in cost-benefit and risk/damage assessment analyses must be developed. The priority for this is high since established standards for the marine environment do not exist.

Finally, to state that a model is no better than the data available to construct and quantify it is a much cited truism, but there is more to enhancing confidence in ecosystem models than just improving the quantity and quality of the data. Ecosystem modeling has evolved in an unstructured fashion, and model development is not guided by formal criteria. As a result, ecosystem modeling as a scientific discipline is not firmly grounded in strong theory. Quite often mathematical functions and time-space aggregations of variables are chosen more by tradition than by rigorous scientific principles. Modelers should give high priority to the development of sound theoretical principles for modeling and the establishment of formal criteria for model formulation and evaluation. There was a general feeling that an initial step in this much needed direction would be to devise and implement testing procedures for comparing, validating, and clarifying the properties of several existing models and modeling methodologies. The knowledge gained from these activities would be of tremendous value to the future development and application of ecosystem modeling as a marine environmental assessment and resource management tool.

NEEDS FOR MODELING IN MARINE POLLUTION ASSESSMENT

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ABSTRACT

Marine pollution assessment will more effectively serve the needs of environmental management decisions if the assessment and decision processes are integrated through a single systems analytic approach. In such an approach the alternatives for marine pollution management and the values placed on potential outcomes by the decisionmaker (or by the public affected by the decisions) focus the information needs of the assessment. The assessment-decision analytic process provides the basis for research-monitoring programs designed to validate the predictions used in reaching the decisions. Iterative application of this integrated assessment approach will lead to successive refinements in both environmental understanding and the effectiveness of environmental policy, especially if the analyses are carefully and objectively documented at each step.

The modeling needs for this integrated process encompass the areas of deterministic and simulative models of environmental processes and effects, probabilistic or risk assessment models, and models for decision analysis, including models of value or utility and models for time preference and risk aversion. These needs are discussed briefly, and a generic set of information requirements is presented for major marine pollution issues. This set of needs requires the formulation, validation, and documentation of predictive modeling techniques, which should be iteratively applied and refined in a management decision context to be most effective.

INTRODUCTION

To identify the types of modeling efforts needed for marine pollution assessment, or more generally for environmental assessment, first requires an understanding of just what is meant by environmental assessment. Why do we do environmental impact assessment or marine pollution assessment?

Environmental impact assessment has been defined as "an activity designed to identify, predict, interpret and communicate information about the impact on man's health and well-being, of proposed human actions" (Munn 1975). By way of contrast, Holling (1978) has urged that the environmental assessment concept and practice be incorporated into an overall process of adaptive environmental management and policy design, in which environmental, economic, and social understanding are integrated and applied at the beginning of the policy design process, and that understanding is systematically improved both as part of the design process and after implementation of the policy. Although these statements carry strong implications of environmental assessment for decisionmaking, the interface between assessment and decisions is poorly defined and frequently ineffective (Holling 1978, Frenkiel and Goodall 1978).

Environmental impact assessment is frequently practiced in a strongly reactive mode of operation, in response either to perceived or suspected environmental degradation, or to planned human activities or developments with environmental implications. Neither of these reactive approaches is closely coupled to the decision or planning process that led to or motivated the assessment.

Reaction to perceived degradation provides for assessing the present and past condition or status of selected environmental components, to determine whether, why, and how much change has occurred. The approach is largely descriptive, although it may improve general understanding of ecosystem responses to stress and may identify problems that should be dealt with in the future. One example of such an assessment was the large-scale investigation to determine the effects of oil drilling and production in Louisiana coastal waters (Ward et al. 1979).

Reaction to planned activities is somewhat more anticipatory in that the present status of the environment is analyzed to determine whether the planned activities are likely to create or cause problems. This approach may lead to the identification of stipulations intended to alleviate the potential problems. The Outer Continental Environmental Assessment Program for oil and gas development in Alaska (Engelmann 1979), for example, has contributed to the design of stipulations on the sale and operations of lease tracts in the Arctic (Weller et al. 1979).

Historically, environmental assessment programs have contributed greatly to our ability to describe present or past states of the environment and considerably less to our understanding of how various ecosystem components interact and respond to man-induced perturbations to the system. Present assessment practice has been criticized as both ineffective and wasteful (Ward 1978; Holling 1978), primarily because of the reactive and descriptive character of the studies and the lack of coupling with the decision processes which the assessments purportedly support.

If one accepts as a basic premise that a primary intent or expectation of the environmental assessment is to guide the formulation of policy or to aid decisionmaking through the identification of the optimal course of action from among a selection of proposed alternatives, then there are several implications for the environmental assessment itself. First, it is essential that much greater emphasis be placed on predicting the possible future states of the environment that may result from adoption of those policies and alternatives. Furthermore, the environmental research required for such an assessment should be strongly focused by the concerns of the decision or policy problem. Early recognition of the elements of a rational decision process can help considerably in focusing the environmental assessment on those ecological processes and consequences that are most important and useful to the waste-management decisions that may ultimately influence or modify the magnitude or extent of marine pollution.

A further implication of the foregoing premise is that the classes of modeling efforts required for the assessment are not restricted to the environmentally oriented models, such as ecosystem and population dynamics, circulation and trajectories, etc. These types of effort must be supplemented by probabilistic modeling or risk assessment (Kates 1978) and value modeling, both of which have long been recognized as important components of decision-analysis (Howard 1966, 1968; North 1968). In this paper, I review the implications of orienting marine pollution assessment strongly toward environmental management and decisionmaking and identify some of the most important modeling needs from a decision-oriented perspective.

A DECISION ORIENTATION FOR MARINE POLLUTION ASSESSMENT

In figure 1, environmental assessment is depicted as an integral part of a rational decision process. The figure explicitly relates the past and proposed activities of man to the potential environmental consequences of those activities and to the processes of analysis, decision, and regulation through which we manage or control the activities. The starting point for the overall process is problem perception, which arises from resource-use conflicts between ocean pollution and other ocean uses. These perceived conflicts may result from proposed human activities, or from the consequences of past human activities. The decision process (steps 2-5 in fig. 1) is initiated in response to the perceived problem.

The first step in the decision process is to identify the alternatives (2) which may be taken to reduce or eliminate the problem. For each of these alternatives, then, potential outcomes must be identified and quantified (3). Since some outcomes will be conditional upon the occurrence of uncertain or stochastic events or processes, the likelihood of these events must be established and incorporated into a probability of occurrence for each of the outcomes. The probable outcomes should be quantified in objective terms, as nearly free of implicit values as possible. To each of the probable outcomes, the ultimate decisionmaker(s) must ascribe some measure of value (4) that allows comparison of overall costs and benefits associated with each probable outcome and each alternative. Once the alternatives have been so evaluated, then selection of the optimal alternative (5) is straightforward. The reasonable or rational decisionmaker will select that alternative with the lowest expected overall cost (or the greatest expected overall benefit).

Once a particular decision alternative is implemented, of course, the actual outcome (6) depends upon those numerous stochastic processes and events that were (hopefully) identified and used in the assessment or prediction step (3). This actual outcome may be perceived as a new problem and may initiate another, subsequent decision process. It must be emphasized that a good decision does not always produce a good outcome. A good decision is one based on a logical treatment of the information, values, and preferences of the decisionmaker(s); whereas a good outcome is one that is favorably regarded by the decisionmaker(s) (Matheson and Howard 1968). Stochastic processes or events may in fact lead to low-risk, but highly unfavorable, outcomes. It becomes essential, therefore, that all available and relevant information is considered and used appropriately in the assessment or prediction step (3), and that the values or preferences of the decisionmaker(s) are accurately reflected in step (4) for each of the possible outcomes identified and quantified in step (3).

The steps shown in figure 1 constitute a basic, generic environmental assessment and pollution management approach which can be systematically applied, documented, tested, and improved as new management options come under consideration. The traditional elements of environmental assessment are mostly contained in step (3) of the process: predictively defining and quantifying potential pollution-related outcomes for the marine environment. It must be recognized, however, that reliable prediction, even in a probabilistic sense, of many environmental characteristics is beyond the present state-of-the-art. It is important to recognize the entire decision and assessment process, therefore,

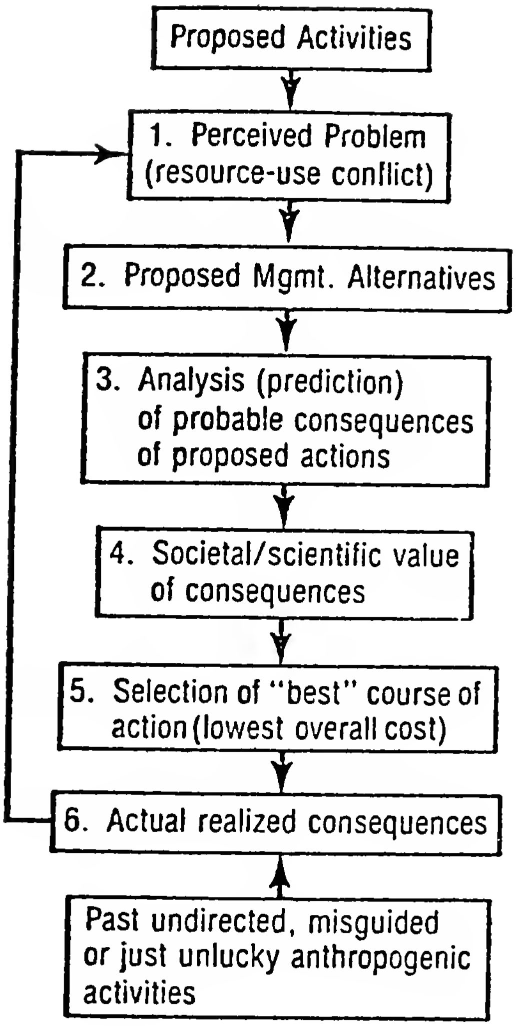

Figure 1. Major elements of the decision process related to marine pollution assessment and management.

as an iterative and adaptive process (Holling 1978), in which best-effort predictive assessment is performed to verify the adequacy of the predictions and the appropriateness of the policy.

Explicit recognition of the decision process is helpful in anticipating the management alternatives that may be considered and the values that may be used in decisionmaking. These in turn aid in the identification of research programs and information products needed most in the decision process. Formalization of the decision process thus provides a structure for the assessment program:

a) Prior identification of resource-use conflicts, potential management alternatives, and values related to outcomes guides the specification of environmental characteristics and processes most in need of research, which should be oriented toward reliable measurement and prediction of outcomes.

b) Identification of the specific outcomes relevant to the decision process guides the specification of the monitoring (long-term research) programs that are required to test and validate the predictions used to identify the optimal pollution control strategy.

It should be emphasized that together the environmental assessment, waste disposal strategy or policy, and research-monitoring design constitute an iterative experimental approach. In this evolving and adaptive approach, a particular pollution management strategy is selected and implemented as the optimal alternative, based on the best available technical information and the applicable values at the time. This pollution management strategy or policy leads to a large-scale and long-term waste management program. Stipulations in the program policy pre-establish the limits of allowable environmental change, and the disposal practices are designed so as not to exceed those limits. The sampling design and analytical requirements of the research-monitoring program are specified (ahead of time) to determine the changes that actually occur. If the pre-established limits are unexpectedly approached, then both the predictive methodology and the disposal practices can be appropriately modified. In a sense, the waste management program constitutes a large-scale environmental experiment, in which the expected limits of environmental change are hypothesized (predicted) for certain stipulated disposal conditions, and a research-monitoring program is implemented to validate the hypothesis and to provide a basis for modifying the stipulations in the event the predictions were in error.

It is essential that more concerted effort be placed on management-oriented marine environmental assessment and on formulation of optimal marine disposal strategies and policies so that valid judgments and selections may be made among atmospheric, aquatic, and land-based disposal alternatives. Historically, environmental assessment programs have contributed to the overall understanding of fate and effects of pollutants in the environment. Future improvements in this understanding can contribute much more directly and effectively to pollution management if the decision process is explicitly recognized and the decision analysis techniques are applied.

GENERIC INFORMATION NEEDS FOR MARINE POLLUTION MANAGEMENT DECISIONS

In the context of the marine pollution management decision process, there is a great extent of commonality among most marine pollution issues and the information needs required to address them. For each of the major issues related to marine pollution, the types of management alternatives are few (table 1), although in any particular situation, of course, these alternatives will be expressed in terms of the specific engineering practices, disposal location, regulatory stipulations, etc., most applicable to the contaminants or insults of that situation. Similarly, the marine resource values potentially at risk from the resultant pollution are essentially the same for all issues (tables 2 and 3), even though details of use patterns, habitat characteristics, and species compositions may vary greatly from one situation to the next. Regardless of these differences, however, both the measures of outcome for the resources at risk (table 2) and the mechanisms of pollution impact (table 3) will be fairly consistent from one situation to the next. The overall assessment of marine pollution management alternatives requires that the expected value be computed for each of the probable outcomes (expressed in the measures of table 2) that are predicted for each of the alternatives under consideration. This generic prediction problem involves two major (and extremely complex) tasks:

1) <u>Predicting the Outcomes</u>, i.e., predicting the magnitudes, spatial scales, and durations of change in the measures of outcome (table 3) for each class of resource values for each identified management alternative. This predictive effort must consider all the relevant mechanisms of impact (table 3) and all the relevant stochastic events which may cause different outcomes.

2) <u>Evaluating the Outcomes</u>, i.e., establishing the actual or relative value of each outcome as a function of the measures in table 3, and the spatial scales and duration of impact. These marine resource values must be considered along with the other socio-economic values associated with each management option, so that the optimal alternative can be identified. These valuation approaches are likely to differ from region to region, depending upon the experience, preferences, and circumstances of the decisionmaker and the population affected by the decision. Thus, optimal management alternatives will be different for similar problems at different times and places.

Efforts to date on resource evaluation have met with little success, especially for nonmarket values, such as long-term global human habitat, ecological or wilderness esthetic values, and birds and endangered species (Mattson 1979). By contrast, valuation of commercial fish and shellfish, including discounting approaches for future yields (Clark 1976), is reasonably well established; and approaches for valuing human health (Howard 1979, North and Merkhofer 1976, Kates 1978) and recreational values (Wilman 1978, Bonnieux et al. 1980) are proceeding on several fronts related to other decision problems. Matheson and Howard (1968) note that "though these questions of evaluation may be difficult, logic demands that they be approached directly in monetary terms if monetary resources are to be allocated." The Willingness-to-Pay approach is one way of approaching the monetary value of outcomes and has the virtue of simplicity compared with utility assignments, which may be very complex indeed

Table 1. Classes of management alternatives available for marine pollution problems.

- Source Control
  - discharge quantity
  - discharge location
  - discharge methodology
    - engineering characteristics
    - discharge timing
    - discharge rate
- Waste Pretreatment (discharge quality)
- Restrict Use Patterns (Zoning)

Table 2. Values at risk from marine pollution and potential measures of outcome.

- Human Health
  - total deaths/yr. for exposed population
  - avg. shortening of life span
  - avg. number sick days/yr.
- Fish/Shellfish (Comm. and Sports)
  - stock size
  - annual marketable production
  - reproductive potential
  - disease incidence
  - age-size composition/mortality
  - contaminant levels
  - fraction of historical/available range occupied
- Birds and Endangered Species
  - population size
  - reproductive potential
  - age-size composition/mortality
  - disease incidence
  - contaminant levels
  - fraction of historical/available range occupies
- Recreational-Aesthetic Values
  - number "floatables"/mile
  - sport fisherman·days/yr.
  - scuba diver·days/yr.
  - coastal motel-tourism receipts
- Ecological-Aesthetic (Wilderness) Value
  - species composition
  - diversity
  - heterotrophy/feeding type index
- Long-Term Global Human Habitat
  - primary production
  - atmospheric CO_2/O_2
  - toxicant concentrations

Table 3. Values at risk from marine pollution and mechanisms of pollution impact requiring predictive assessment.

- Human Health
  - contaminated seafood
  - water contact

- Fish/Shellfish (Comm. and Sports)
  - bioaccumulation of toxicants
  - habitat disruption
  - habitat loss
  - food loss

- Birds and Endangered Species
  - bioaccumulation of toxicants
  - habitat disruption
  - habitat loss
  - food loss

- Recreational-Aesthetic Values
  - floatable wastes
  - turbidity

- Ecological-Aesthetic (Wilderness)
  - bioaccumulation of toxicants
  - habitat disruption
  - habitat loss
  - food loss
  - (ecosystem structure-function)

- Long-Term Global Human Habitat
  - primary production loss

where multiple attributes with widely different value bases are involved (Keeney and Raiffa 1976).

Guided by the identification of management alternatives and resource values, one can formulate the general assessment steps required to achieve the prediction of probable outcomes (fig. 1, step 3). These steps are outlined below in the (approximate) sequence that they must be performed for each identified management alternative:

1. Characterize source compositions and magnitudes for key contaminants. The source characteristics (contaminant composition, concentration, location, flux, etc.) represent a series of decision variables for each source, i.e., factors which generally can be controlled through the decision itself.

2. Predict distributions of key contaminants in and on the water column and in sediments as a function of source characteristics and time.

    a. Predict transport
    b. Predict transformations

3. Predict rates and extents of habitat disruption and modification for key organisms as a function of source characteristics and time.

4. Predict bioaccumulation rate and levels for key contaminants in key organisms as a function of exposure regime.

5. Predict changes in population size and production of key species based on bioaccumulation of contaminants and habitat disruption and modification.

6. Predict the outcomes, in terms of human health measures, as a function of contaminant (and pathogen) concentrations in key organisms and the general distribution of contaminants.

7. Predict the outcomes, in terms of measures of recreational values, as a function of distribution of contaminants, habitat disruption and modification, and population size/production of key species.

8. Predict the outcomes, in terms of Ecological or Wilderness values and Global Human Habitat values, as a function of habitat disruption and modification and population size/production of key species.

9. Design and implement research or monitoring strategies specifically related to testing and validating the predictions in 2-8.

It is obvious that few of the nine objectives outlined above represent simple, currently available scientific technology. Each of the predictive capabilities called for in objectives 2-8 requires sophisticated modeling approaches that clearly involve many additional variables left implicit in the outline. The prediction of outcomes is depicted in figure 2 to illustrate some of these other information needs and the relationships to quantifying changes in values. Whereas steps 1 and 9 above may be unique and under the control of

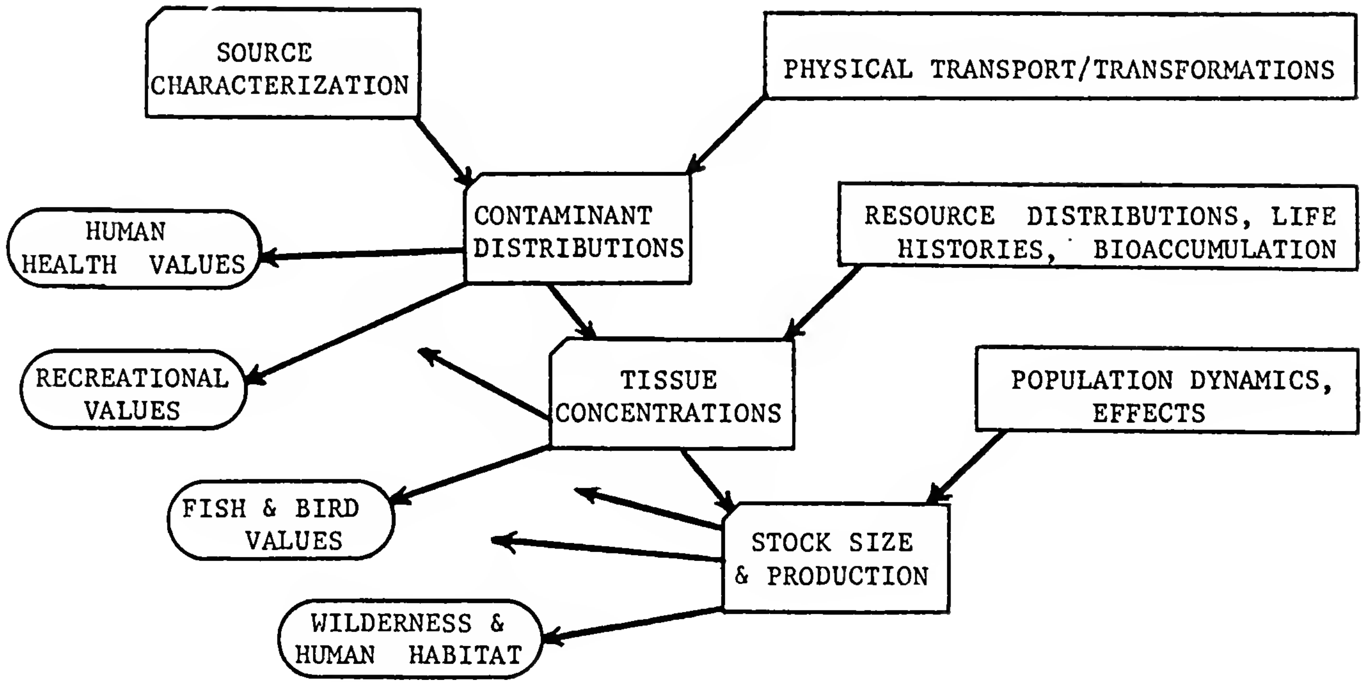

Figure 2. Predicted outcomes. Relationships among different classes of environmental information required for expressing outcomes oriented towards resource values.

the decisionmaker for any particular developing pollution situation, steps 2-8 represent generic needs for information and predictive methodologies that will be very similar in most cases. These generic needs are schematicized in figure 2. A major intended implication of the outlined objectives is that research and information products not contributing directly to these objectives are likely to be of little direct value to the decision process leading to the selection of optimal waste management alternatives. The principal processes and relationships that must be modeled to support the successive predictions outlined in figure 2 are discussed below.

The source characteristics, which are sometimes controllable decision variables, serve as input data for trajectory and dispersion models. These models should include subroutines that account for physical and chemical transformations and transport of different phases. Relevant processes to be modeled may include: biodegradation, adsorption-desorption, sedimentation, bedload movement, bioturbation, etc. The resultant time and space scales for physical contaminant distributions can be coupled with information on demography and susceptibilities to generate the measures of outcome for human health and recreational values. Coupled with information on resource distribution and life histories, the physical distribution of contaminants leads through bioaccumulation or uptake models to predicted tissue concentrations in key organisms. These tissue concentrations are used in computing exposure for humans and other predators, like birds and mammals. Coupled with information on effects, especially on direct mortality or reproductive success, the bioaccumulation information provides input for population dynamics models for key species, which lead to predictions of stock size and production. Many of the measures of outcome in table 2 can be predicted or estimated directly through these computational or modeling steps. The specific modeling requirements must be driven, however, by the measures of outcome, which must in turn be convertible to measures of value. Thus, all-encompassing ecosystem modeling is probably not required, and many potential input variables may be suppressed after appropriate sensitivity analysis. When time scales of concern exceed the predictive capacity of existing models, the decisionmaker must rely more heavily on associated research-monitoring programs to test the adequacy of the predictions.

The one class of resource values for which comprehensive ecosystem modeling may be most applicable is wilderness values. In this instance, however, the valuation approaches are not at all defined, and the suggested measures of outcome may ultimately prove to be of little use. In this case, greatest initial emphasis should be placed on development of acceptable and useful valuation approaches.

Most of the needs outlined above and in figure 2 have been recognized for some time as essential to understanding and predicting pollutant fates and effects on marine ecosystems (Wolfe and Rice 1972, Wolfe 1975, Warlen et al. 1977). The particular point emphasized here is that these predictions can be much more highly focused on the problem at hand by explicit consideration of the management alternatives and the specific values at risk. Even though full and complete marine ecological understanding is never achieved, decisions impacting the marine ecosystem will continue to be made. These decisions can be strengthened by careful documentation of a structured decision-oriented analysis that considers alternatives, values, and uncertainties. Formal techniques for such analysis have become well established in operations research and systems engineering (Howard et al. 1977). In the context of environmental assessment, decision analysis assumes the additional dimension of iterative and adaptive application to an evolving problem (pollution) as our basic understanding of ecosystem function grows. This added dimension derives from the stability and resiliency of ecosystems which allow us to research the validity of our understanding and predictions after instituting the policy that resulted from those same predictions. In this adaptive environmental assessment and management approach (Holling 1978), however, the basic decision analysis remains the same. Modeling needs that arise from the decision analysis itself, as opposed to fundamental cause-effect relationships in the marine ecosystem, are outlined in the next section.

MODELING REQUIRED FOR DECISION ANALYSIS OF POLLUTION MANAGEMENT PROBLEMS

Decision analysis is merely a structured procedure for analyzing the merits of various alternatives in a decision. Decision analysis helps to ensure that essential steps have been consciously considered in the decisionmaking process and also facilitates explicit documentation of the form and content of those considerations. For decision problems as complex as marine pollution management, in which tradeoffs must be balanced among economic costs, human health values, commercial and sports fishery production, recreational and wilderness values, and global human habitat values, over potentially large scales of space and time, it is important to document the analysis as completely as possible to provide both a logical justification for the chosen policy or alternative and a basis for improved decision making in the future.

Careful, complete, and objective documentation of the decision process is particularly important in matters of public policy, such as marine pollution, where numerous different factions, with different perceptions of the problem and different values on the potential outcomes, may be affected by the decision. The documented analysis serves to inform all concerned on the specific considerations that entered the decision and provides a basis for iterative objective improvement in successive applications of the analysis.

Matheson and Howard (1968) have succinctly outlined a comprehensive approach for the analysis of decisions and the identification of optimal alternatives, which is summarized in the following section. The sequence of steps involved in the analysis (fig. 3) involves the creation and manipulation of a hierarchy of models (fig. 4) through which the relative merits of each available alternative are assessed. These steps are consistent with the elements of a rational decision process discussed earlier (fig. 1).

The *deterministic phase* of the decision analysis focuses on the development and sensitivity analysis of the structural and value models used in predicting outcomes and values for each identified alternative. For assessment of marine pollution decisions, the structural model would address the environmental processes outlined in table 3 following the structure suggested on pages 13-14 and in figure 2 to generate a set of outcome variables (table 2) for each alternative. The value model is designed to assign each outcome a value, by translating each member of the set of outcome variables (table 2), as defined and quantified in time and space, into an appropriate measure of value. A time-preference model is also created to translate values that occur over a timestream

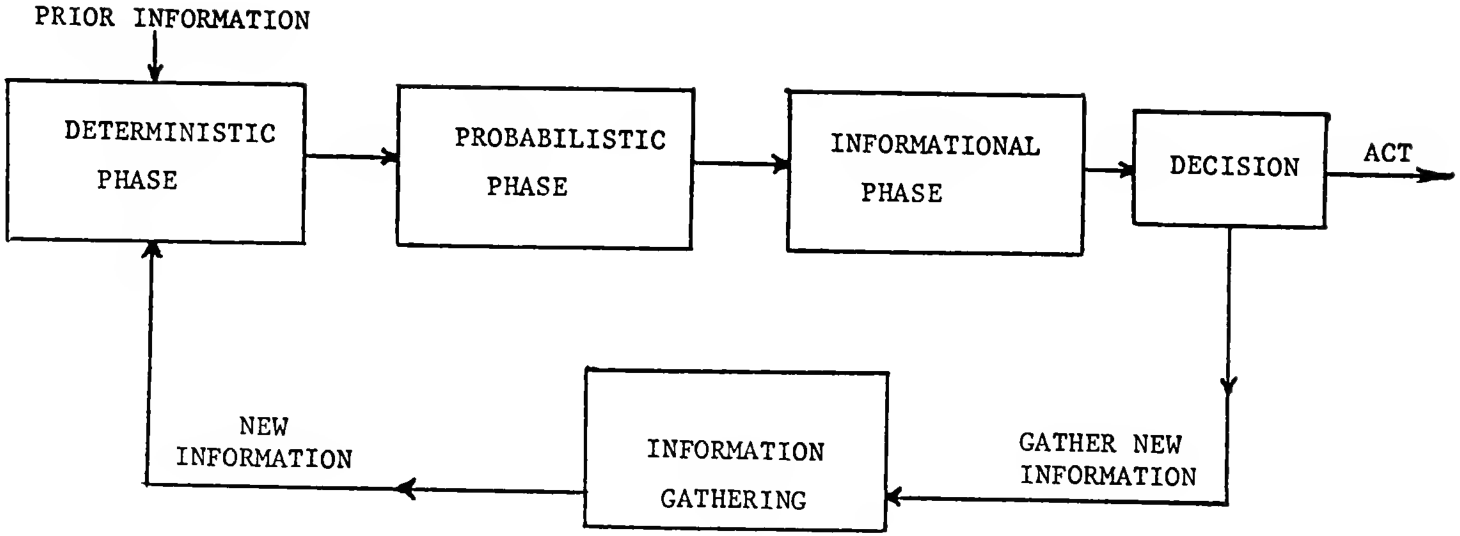

Figure 3. The phases of decision analysis. (From Matheson and Howard 1968. Used with permission.)

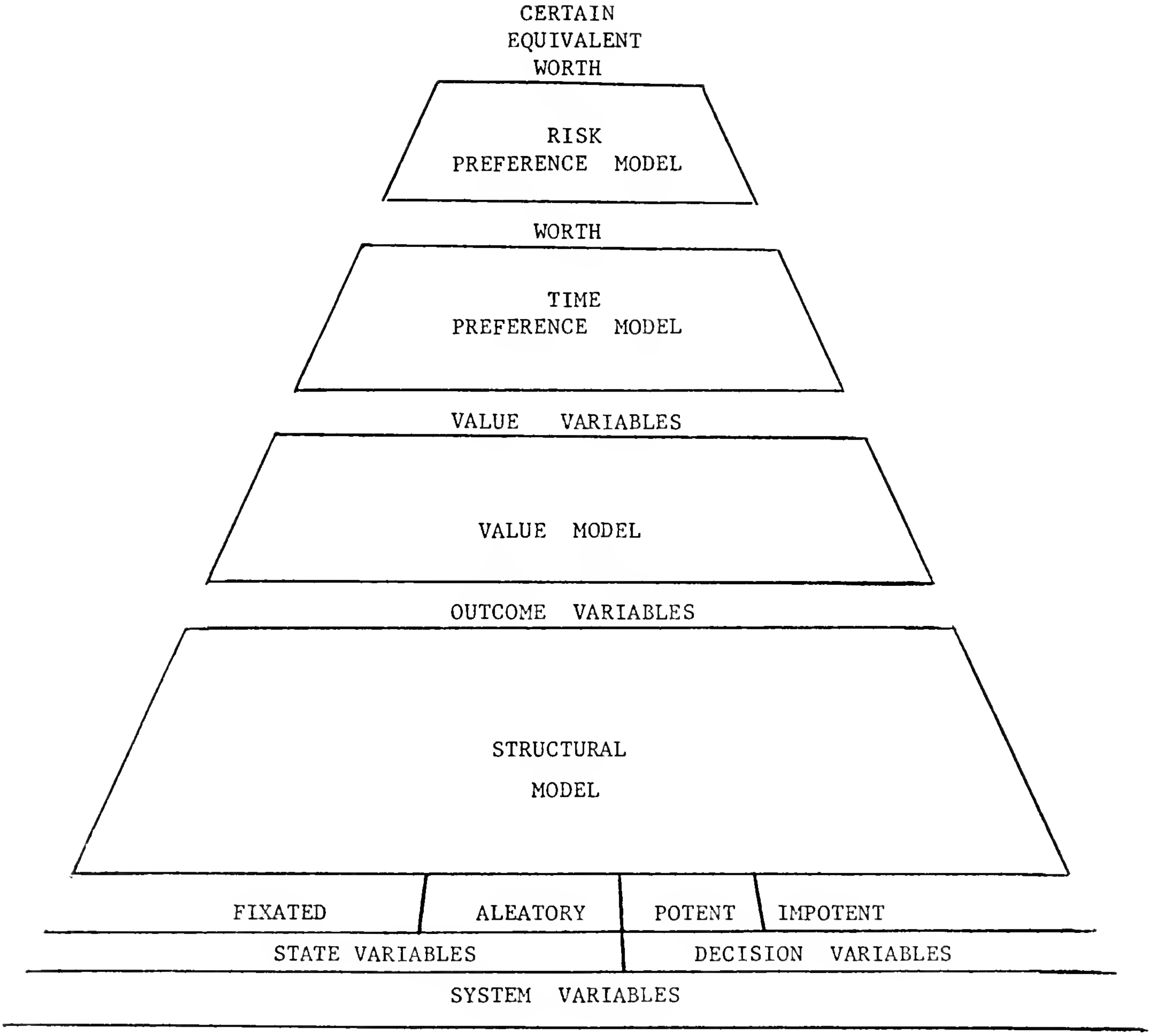

Figure 4. Hierarchy of models needed in decision analysis. (From Matheson and Howard 1968. Used with permission.)

in the future into present net value or worth. For economic values this time-preference model is used to account for discount rates and inflation. The objectives of the deterministic phase are: 1) to determine whether particular alternatives are deterministically dominated, i.e., they always have a lower worth than some other alternative, regardless of what values are selected for the state variables; and 2) to determine which of the decision variables (table 1) and the state, or environmental, variables are most influential in affecting the worth of each alternative.

The probabilistic phase of the decision analysis focuses on resolving the uncertainty in value or worth of any alternative. This uncertainty arises from those (aleatory) state variables to which the worth of the alternatives is found during the deterministic phase to be most sensitive. Probability distributions must be assigned to these aleatory variables over their potential ranges of values. The structural model is expanded to include the probabilistic values for the aleatory variables and the expected outcomes, expected values, and expected worth are then calculated by the structural model, value model, and time-preference model, respectively, based on the assigned probabilities. A probabilistic sensitivity analysis then reveals whether one or more alternatives are stochastically dominated, i.e., whether a particular alternative always has a lower probability of achieving any specified worth than another alternative, over the entire range of potential worth. Stochastically dominated alternatives may be dismissed from further consideration because they consistently have lower probable worth and are logically excluded. If certain alternatives prove not to be stochastically dominated, then there is a logical risk in selection of any alternative, and risk preference (Matheson and Howard 1968) must be modeled for the decisionmaker(s) or his surrogate. The risk-preference model translates the expected worth under uncertainty into a certain worth equivalent. The certain worth equivalent may be viewed as the smallest offered certain worth that would induce a risk-averse decisionmaker to prefer another alternative in place of one with uncertain outcomes and therefore an uncertain worth. At this stage of the analysis, the optimal alternative, the one with the highest certain equivalent worth, has been identified.

The informational phase of the decision analysis focuses on determining how much one should pay to upgrade the quality of information available for the analysis prior to making the final decision. In effect the nature of the information to be gained from a particular experimental program is anticipated, with an associated cost, and the decision analysis models are then restructured about that information, with its new (expected) attendant uncertainties. The deterministic and probabilistic phases of the analysis are repeated for this "new" information, and the optimal alternative is again identified with its new certain equivalent worth. The value of the information is exactly that cost of the experimental program which would make the certain equivalent worth of the decision with the improved information just equal to that of the decision without the information. This process is repeated for each of the aleatory variables, separately and in combination, for which improved information might be obtained. Where the value of the information exceeds the cost of the experimental program (including the costs of the delay in the decision), then the decision to gather new information is supported (fig. 3) and the analysis should be reiterated on the basis of the improved information. When further information gathering is projected to be overly costly, then the decisionmaker is ready to act.

Computerized modeling techniques have been developed for structuring and analyzing decisions. For example QUICKTREE© is a set of APL functions designed to assist in evaluating decision trees with up to 1000 trajectories (alternative-outcome combinations). This program was developed by the Decision Analysis Group of SRI International, Menlo Park, California.[1]

SUMMARY CONCLUSIONS

1. Marine pollution assessment would serve environmental management decision needs much more effectively if the assessment and decision processes were integrated through a systems analysis of waste disposal management decision problems.

2. In such a decision-analytical approach, the objectives of the environmental assessment are focused by the available alternatives and the values of the decisionmaker, and the modeling needs can be successively refined through sensitivity analysis oriented toward those limited objectives.

3. Generic consideration of marine pollution problems suggests major environmental modeling needs in the following areas:

    A. Transport and Transformations of Pollutants

        1. dispersion of dissolved and suspended materials
        2. bed load transport
        3. adsorption dynamics
        4. bioturbation
        5. evaporation-dissolution
        6. physical-chemical transformations (state changes, chemical reactions, decomposition, etc.).

    B. Bioaccumulation and Effects

        1. uptake-depuration kinetics

    C. Population Dynamics of Key Species

    D. Ecosystem Modeling

        1. nutrient-planktonic dynamics
        2. stability-resiliency-recovery from stress.

4. Decision-analytic modeling and systems approaches are currently available and are applicable to marine pollution management problems. Effective application requires close interaction among environmental scientists and modelers, decisionmakers, and systems or decision analysts.

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[1] Mention of tradenames does not imply endorsement by NOAA or the Department of Commerce.

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COMMENTS ON MODELING FROM THE STANDPOINT OF A RESEARCH ADMINISTRATOR

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EDIS asked me to comment on modeling from the standpoint of the research administrator. The bulk of the effort in the Fisheries Center is essentially ecological. We have a particular focus on the resources utilized by man. For each of our very large ecosystems we really have only one model and this model often drives our entire program. Each Fishery Research Center is concerned with one or two very large ecosystems of varying complexity and size. These VLE's, as we call them, don't have hard boundaries, of course. For us, modeling is a continual process--a way of life not, unfortunately, marked by continued, steady progress, but rather a process that leads to questions as well as answers and tends to vary in intensity through time.

One continuing problem that we have is that of deciding how much centralized effort to put into modeling at any one point in time. In general each scientist models his own specific element of the work. There is a need, however, for the integrative modeling effort. There is a marked tendency to centralize this effort. Unfortunately, all too often such centralization leads to the development of one general model driven by a single dominant personality. While it may seem more efficient to proceed this way, it can lead to serious difficulties because of the limited grasp of the complexity of an ecosystem by any one individual and by the pulsating and varied nature of opportunity to make advances. It is better, in my opinion, to proceed somewhat more slowly by rotating the involvement of individuals dealing with these more general models. This has the benefit of varying the effort commensurate with the opportunity to make specific advances and does not result in a model frozen in the image of one individual.

Change is the name of our game. The situation that we are dealing with is dynamic and, to some degree, intractable to deal with because of the logistics of sampling the ocean. Almost all of the decisions that are made with respect to resources events are based on our ability to measure change, not absolutes. Our fundamental problem is one of specifying rates.

The first version of our model, possibly it is the first ocean model, was published by George L. Clark in 1948 in a paper entitled "The Dynamics of a Marine Ecosystem."[1] The present model is about the fifth generation of this model. It has improved with each succeeding year as more and more data accumulates and interactions become clearer. At the present time it is relatively robust at the two ends--that is, we know a great amount about the initial energy inputs and primary production, and the outputs, that is, the fish and other resource populations. The internal anatomy is still unclear although even here rapid advances are being made.

The general robustness of the model, even in its third generation, was sufficient to convince us and others that the only solution to the problem created by the massive foreign fishing effort off our shores was to impose an overall biomass quota. We had enough information, for example, to convince others that we had a sausage machine that produced at a reasonably well defined rate, although we couldn't specify precisely the type of sausage that might be produced at any one time. Such a biomass limit was imposed, an impressive 'first' in the history of ocean management.

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[1] Ecological Monographs, vol. 16, no. 4, pp. 321-335.

Many of the changes that are taking place today in the resource populations are not always predictable, but at least many can be rationalized on the basis of the model. It has become very clear, for example, that predation on the smaller size of fishes has a great deal to do with the ultimate size of adult populations. It is also becoming obvious that disease plays a far more significant role than previously acknowledged.

In any event, in the Northeast Fisheries Center we have one overall guiding model but many variants, subsets and aspects of this model. There are three general needs which may be categorized as follows:

1. A critically important need is that for a relatively transparent version in popular science terms. This version is one basis for communicating to our constituents. These are usually static versions of the model and contain information about the processes and rates in terms that every man can understand. An example (fig. 1) of one such version--it speaks in terms of dollars rather than grams of carbon per meter squared per year--was developed to illustrate energy flow for an audience of non-biologists.

2. The second need is to clarify the dynamics of the ecosystem and ultimately to predict resources events. This version of the model is characteristically scientific, opaque, and cumbersome to deal with unless you are an expert. The dynamic model required in this instance has a minimum of 50 or more state variables. This number of variables is sufficient to overload most computer main frames and accordingly it is less dynamic than we would like, but it exists and it is used.

3. From my point of view the most significant version of the model is that one we use for communicating with our bosses. It is transparent, perhaps deceptively so, but it does lead to the development of other kinds of information that are necessary.

Our simple form of the model can be stated as follows (fig. 2):
Change = Fishing (F) + Predation (P) + Disease (D) + Environment (E).
Let's have some definitions:

Fishing (F). This component refers to all the activities associated with population assessment. It includes survey cruises, age and growth studies, analysis of landings data, and a great deal of analysis.

Predation (P). The feeding of one species on another. We now can demonstrate the likelihood that predations can and do structure the fish segment of the ecosystem. The function is felt by man 2 or more years after the fact since much of this predation takes place on the very young fish.

Disease (D). Disease is an obvious "player" in the ecosystem. It's full importance has only recently begun to be recognized. Fortunately, disease phenomena are also relatively easy to deal with in the predictive mode once the principles of their involvement are understood. Disease generally has the same "future" significance that predation has. It manifests itself early and usually on younger fish. Its effects peak several years after the initial "insult." Disease is often directly linked to physical environmental change, both natural and man-caused.

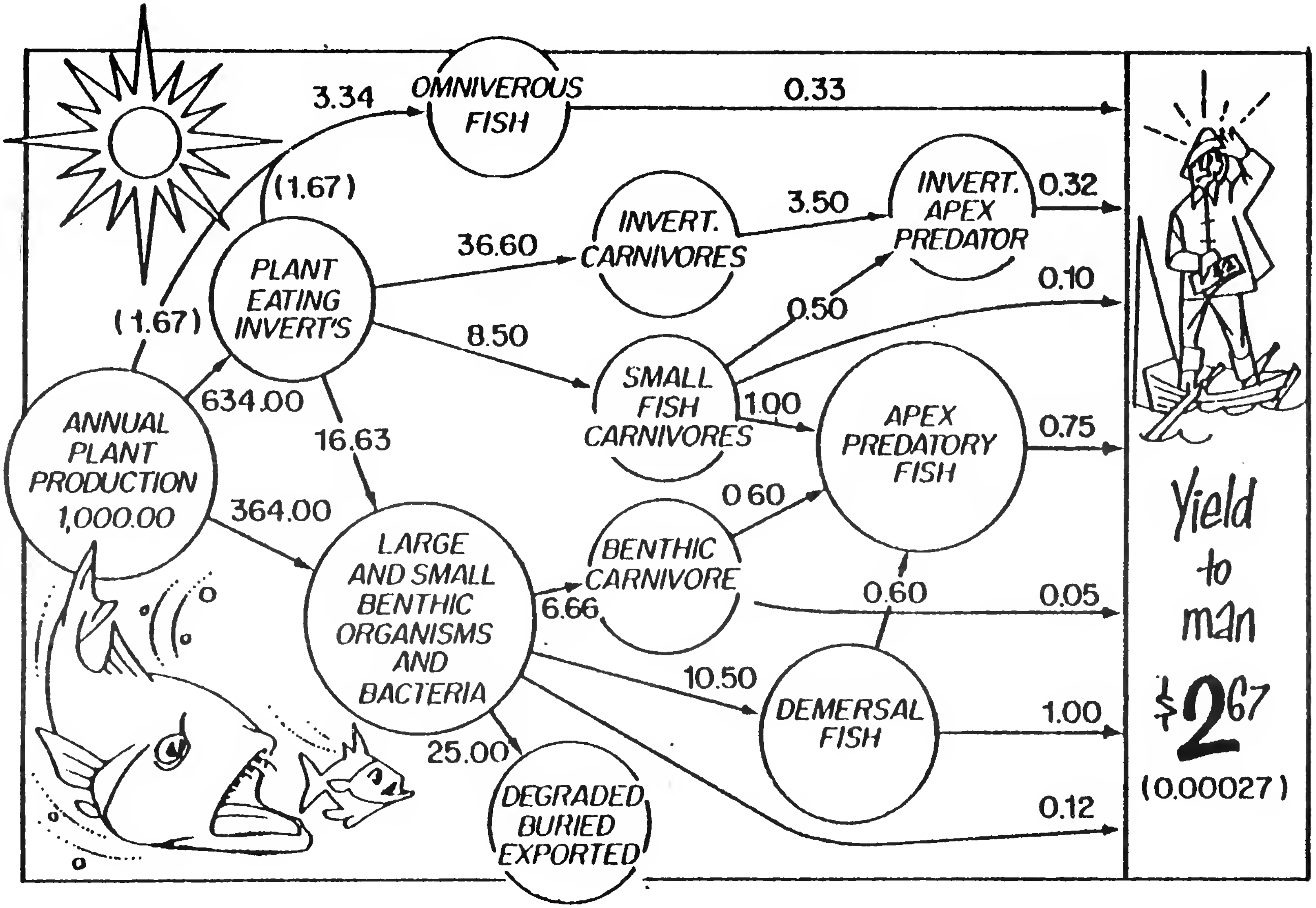

Figure 1. A simplified energy flow diagram for the region off the New England coast. Flows are in dollars.

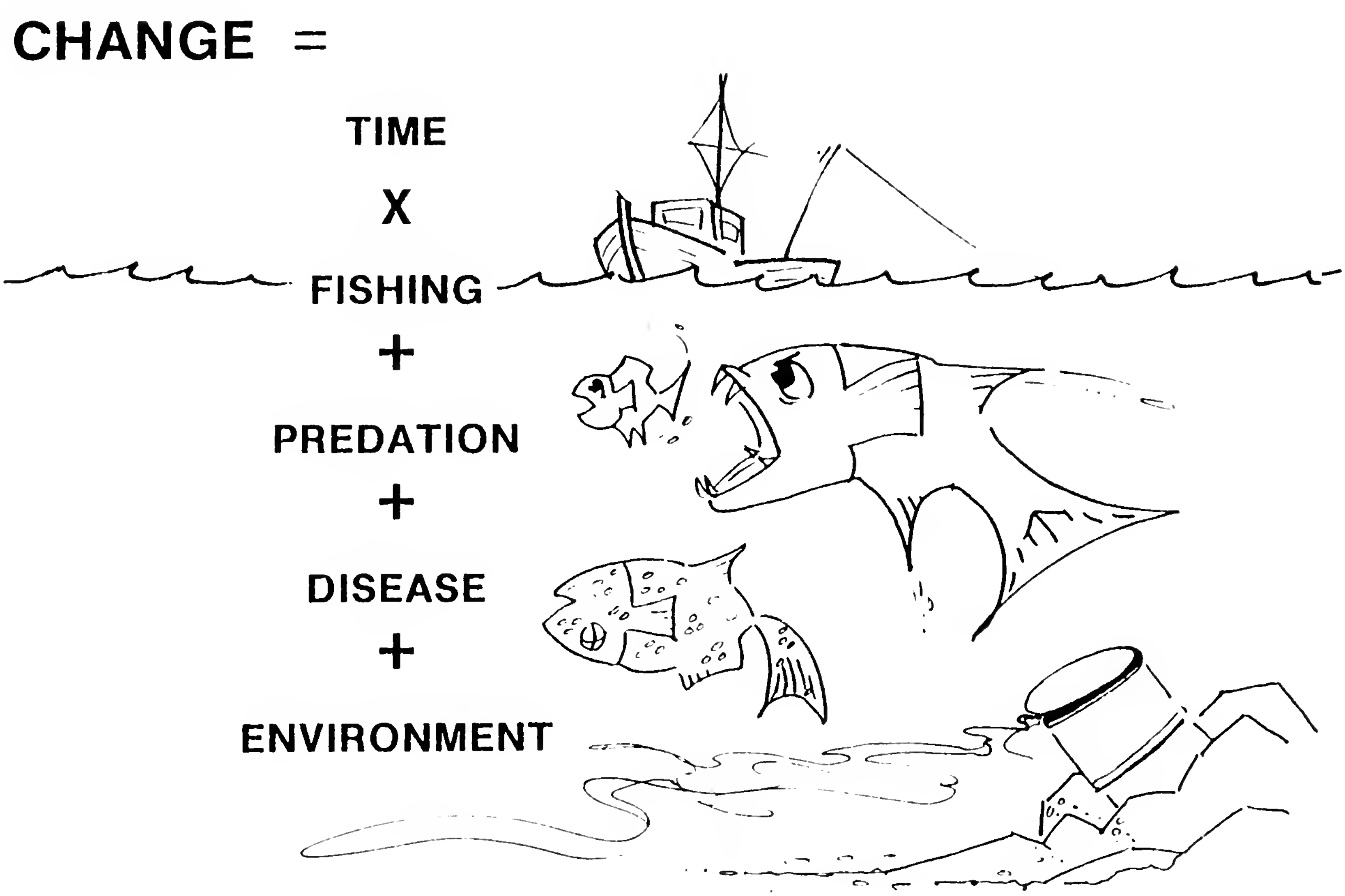

Figure 2. Simplistic representation of a dynamic fisheries model.

Environment (E). The physical environment component refers to the flux of temperature, salinity, and nutrients in the water. The environmental effects are usually subtle, often difficult to predict, and always difficult to sample. This element is subdivided into two aspects--the natural environment and the man-created environment (e.g., pollution).

For a moment let's look at each of these variables and examine how they can be used to evaluate our program and its usefulness to the country.

We are often asked by our bosses, "How much is enough?" In figure 3 each of these "system" variables (F, P, D, and E) has been evaluated by the appropriate staff members of the Center. These are performance functions and they are their estimates of the value of information provided by these experts in answering questions asked of them. Program cost is displayed on the X axis and the relative value of information on the Y axis. Perfect information, of course, is represented by 1 and is not attainable. The dots on the curves represent our estimate of our present position. In a way you can call this the "expert's comfort (or discomfort) index." You will note in the performance function for F (Fishing), the experts estimate their present ability to provide information as approximately 0.46. It is costing us approximately $6 million to provide this information. The F function has been broken down into two curves. The lower curve is concerned with the data collected from the fishermen themselves. In terms of predicting change, it very quickly flattens out. The reason for this is obvious. Fishermen go to those areas where the fish congregate and the data collected from them is accordingly strongly biased when used to estimate population size. This is the same bias that a census taker would experience were he to be confined solely to basing his estimates on the population of our country on New Year's Eve in Times Square. Catch data has some real value up to a point, but beyond that point it tells you virtually nothing. In order to significantly improve the amount of information from such data, more than the expenditure of money would be involved. One would virtually have to mandate nearly perfect data from each fishing vessel and from a sociological point of view that is a virtual impossibility. The other curve is based on data that activities such as research vessel surveys would provide, the more appropriate and unbiased census type data which is necessary to make a prediction.

In the case of predation and disease research, the costs for significant information are not as overwhelming as they are for developing appropriate F information. In no small part this is because much of the data required can be, and is, collected during cruises carried out for other purposes. Further, such data is used more abstractly - in prediction models - where the principles are as important as the facts. The environmental curves have been subdivided into two components, that which is natural and that which is man-caused. The value of information available today is estimated at about 0.3, mostly because of our inability to deal with the flux of the environment in real time using such sampling tools as vessels. It is in this curve that one sees the opportunity for a small investment in modern technology that would greatly increase the value of information that can be provided. This technology exists - it is called remote sensing. The man-caused environmental changes potentially can reach a higher level of information value than can any other category, simply because the man-caused environmental effects tend to persist for a long time, as for example DDT, which can now be found throughout the world.

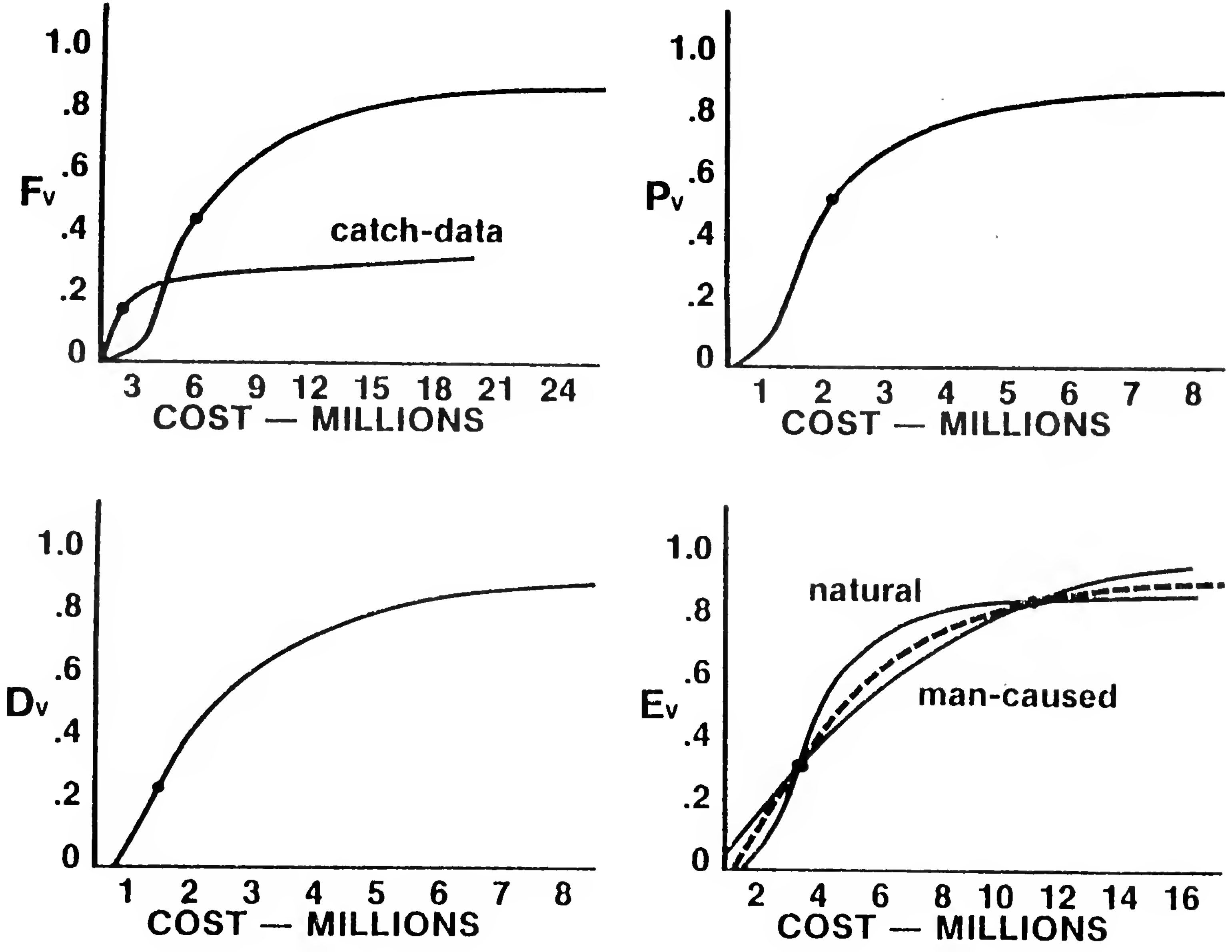

Figure 3. Value of information as a function of cost in dollars. See text for explanation.

This is only part of the story, however.

In figure 4 the relative value of information with respect to time is illustrated. It depends very much on the time horizon of concern how valuable the information from each of these system variables may be. The effects of predation and disease, vis-a-vis the fisherman, both tend to peak 3 to 4 years ahead. In the case of environment, it becomes very important as you look further ahead, and finally the environment concerns dominate when one is looking ahead as much as 10 years. The highest value for the natural environmental information at the zero time horizon, for example, is related to the fact that this information is useful to those pursuing, for example, tunas, or any other resource which tends to congregate along ocean fronts near the surface. Because one of our overriding scientific questions is that of the significance of temporal and spatial variability of environmental parameters, knowing what is going on at the time the research vessels are on the scene is also important. Again that new technology - remote sensing - pops up as a significant and cost effective tool to be invoked as soon as possible.

To return now to the more complicated segment of this figure, that referring to fishing, there are three curves. If one wishes to maximize "now," or to put it into other words, discount the future, then fishing data is very important. If, however, one wishes to maximize both revenue and stability (longer term parameters) from existing populations of fish, then the curve is far different. Since most of the resources have moderately long lives in the fishery, a haddock year class, for example, can last several years, then other information becomes important, as for example, predation and disease. Since recruitment to populations of fishes is controlled by many variables, at the present time it is almost impossible to predict recruitment much more than 3 or 4 years ahead. This curve thus tapers off quickly after 6 years. If one desires to maximize ecosystem revenue, the curve falls somewhere in between these last 2 or at about 5 years. Going back to the sausage machine analogy, such activity would require directing the fleet to harvest certain species as opposed to others, with the consequence of somewhat less ability to pass judgments on fishing data alone, but required far more reliance on knowledge of the recent history and future consequences of predation and disease.

Figure 5 is simply a single graph combining all the information in figure 4. The most desirable mix of programs is thus indicated in terms of the relative value of information against the time horizon. When one puts this information into the computer and evaluates the overall information return, given various budgetary levels, a new series of curves is generated. These are presented in figure 6.

To generate figure 6 we assumed that we had 1.5 full-time research vessels available and a capacity to use satellite derived products. Actually we have the full time use of two vessels and cannot as yet effectively use remotely sensed data. It should be noted that these figures do not precisely reflect our actual budget situation. The computer was asked to evaluate the relative information return from each of the four system variables, given that each time horizon was equally important. We are incidentally, at about the 1X level.

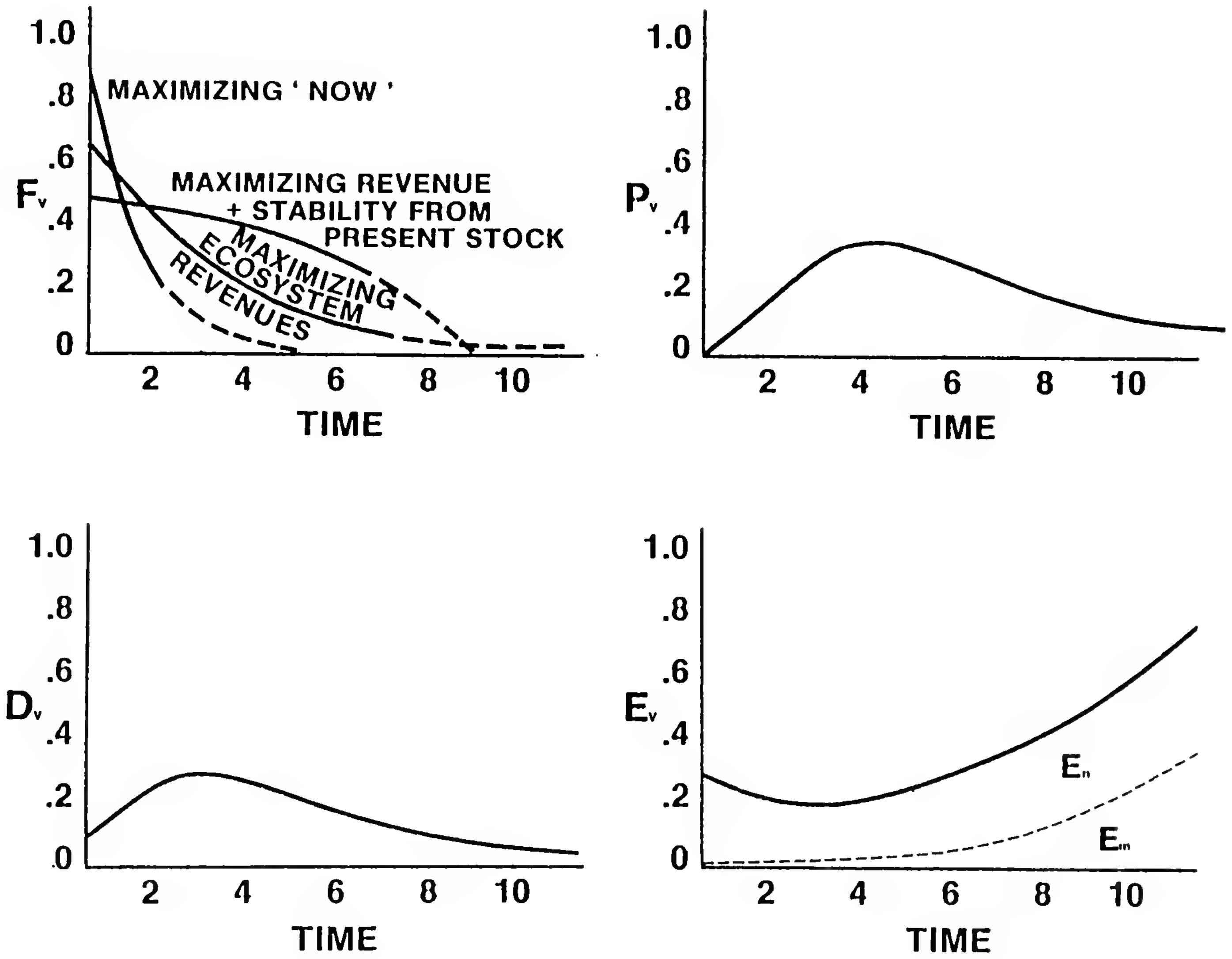

Figure 4. Relative value of information as a function of time in years. See text for explanation.

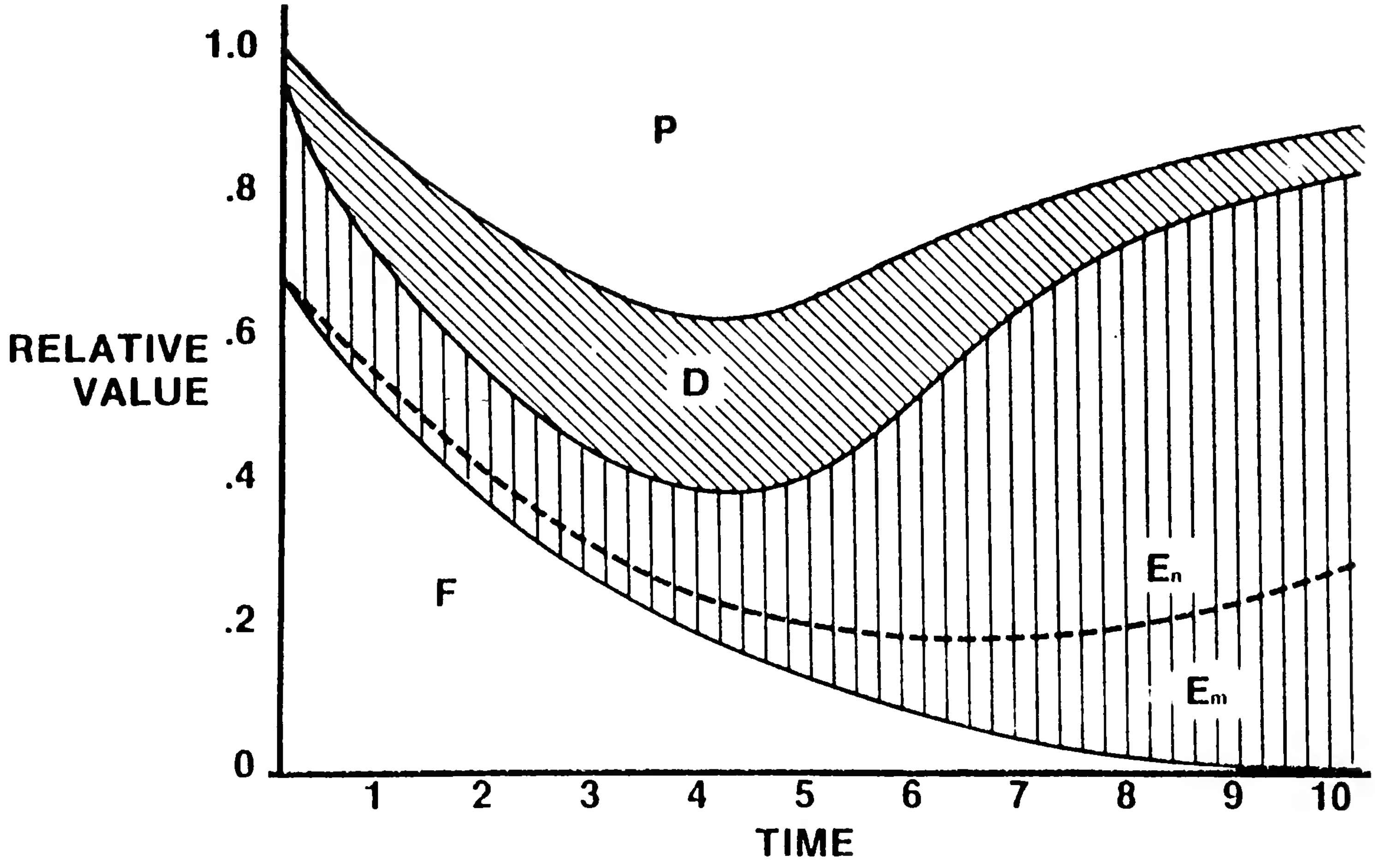

Figure 5. Composite graph of the information presented in fig. 4; relative value of information as a function of time.

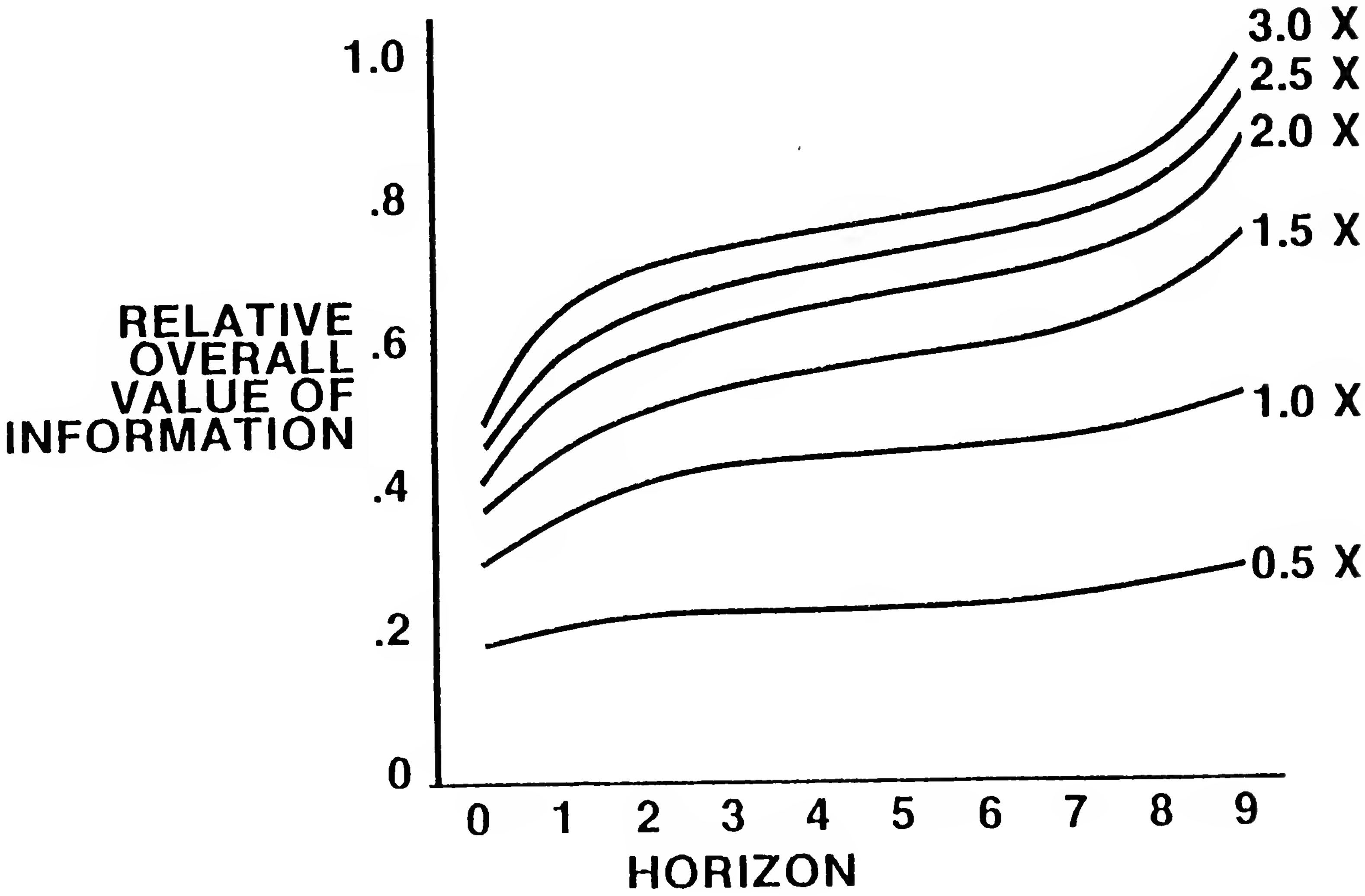

Figure 6. Relative overall value of information as a function of time for several levels of funding. Current level of funding is 1.0 X.

Several things come out immediately - none of which are counterintuitive. First, given present technology, there is a point at which further increments in the budget do not provide significant increases in the amount of new information. Second, it is difficult to increase the value of real time information (0 to 2-year horizon) without an inordinate expenditure of funds. Third, the time horizon from 3 to 7 years is an area within which it is difficult to modify the programs to achieve significant specific increases in information. Fourth, the distant horizon tends to accrue benefits from programs in the middle range and of itself does not need to be addressed in explicit terms. This is probably just as well since it is at this horizon that the environmental factors begin to dominate and it is here also that there is little capability at present to effectively predict what will happen 10 years from now anyway insofar as environmental change is concerned.

Going beyond this level of generalization would not be particularly helpful. One can play other games with this system and get slightly different results. Almost without exception, when these games are played, one discovers very little that isn't obvious in terms of just plain common sense. Nonetheless, from the standpoint of the administrator, whether the administrator is the Center Director, the Director of a Service, or the Director of NOAA, one can now come back to the "how much is enough" question.

Let's go back to figure 1 and the performance function for fishing based on survey cruises. The administrator can make the judgment that from his point of view the value of the information being provided is not 0.46 but 0.6. He, after all, is the one who makes decisions and his judgment of the value of information is important and in the long run should probably be one of the driving forces in program development. This has the effect of transforming the X axis. The performance function itself hasn't changed, but the expenditure to achieve that level of information is now addressed on a different basis. Information with the value of 0.6 on fishing costs $6 million as far as he is concerned. This is his "comfort index." Now one can have a constructive dialogue.

This performance function model is but an extension of our Northeast Fisheries Center model for the ecosystem. For all practical purposes, it is the same model. It has simply been modified to serve another purpose. Much of the information to be presented at this symposium will deal with models at the other end of the spectrum. These activities are important. They are, however, critical to the development of these other versions. The version I have presented here deals with broader problems, as for example, budgeting, program development, and evaluating areas both for reduction or enhancement. Such integrative and semantically different models have the virtue of allowing the research administrator to get into the system and evaluate program needs in terms of his perception of our reason for being, on a communication bridge where both the expert and the administrator can meet as peers.

MARINE ECOSYSTEM MODELING: THE POTENTIAL FOR APPLICATION IN BENEFIT MEASUREMENT

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Washington, D.C.

INTRODUCTION

The use of benefit-cost analysis is one of the foundation stones on which the Reagan administration is building its effort at deregulation. Executive Order 12291 of February 1981 formalizes this position. In the environmental area, the U.S. Environmental Protection Agency (U.S. EPA), is currently working as hard as its reduced budget and manpower will allow to produce the benefit estimates required by the Office of Management and Budget (OMB) for such diverse actions as the steel industry effluent limitation guidelines and review of the national ambient air quality standard for ozone.[1]

There are difficult economic problems involved in almost any environmental benefit estimation exercise. These include such matters of technical (and political) dispute as how to value future benefits (and costs); how to value human morbidity and mortality; how to value aesthetic effects (such as better visibility or cleaner looking water); and how to account for values that may be held by nonusers of the environmental assets in question. What is not always recognized either by economists or by regulatory agency staff is that our ability to make defensible benefit estimates may also be constrained by lack of comprehensive knowledge of existing conditions of the ambient environment and by our inability to predict how those conditions will change with implementation of the proposed program or regulation.[2]

The key place in benefit analysis of ecological knowledge and, especially, predictive ability can be illustrated by consideration of recreational (sport) fishing as a source of water pollution control benefits. A brief summary of a completed project aimed at estimating the benefits of water pollution control accruing via freshwater recreational fishing will illustrate the method and its requirements. Then, discussion of the obstacles to application of the method to marine recreational fishing will bring us face to face with the subject of this workshop, marine ecosystem modeling. Our assessment of the state of this art from the point of view of benefit measurement is a rather gloomy one. It is far from obvious that a defensible measure of marine recreational fishing benefits can be made on the basis of existing knowledge and modeling ability. This view may provoke comment, correction, derision, or despair, but we emphasize that it represents a serious effort to identify applicable data and models. If we are wrong in our assessment, a possible lesson might be the need for better communications between the marine ecomodelers and environmental economists.

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[1] Perhaps "ironic" is the kindest way of describing the conflict between adding difficult analytical requirements, benefit-cost analysis in particular, and reducing available analytical resources. Indeed, the notion that the application of benefit-cost analysis will lead to less regulation is a comment on preconceptions about results rather than a logically defensible tactic for achieving "deregulation."

[2] In this paper, we shall ignore the very serious problems created for benefit-cost analysis by the focus of Executive Order 12291 on regulations, given the fact that most environmental regulations are written for industries rather than for environmentally meaningful units such as river basins or airsheds.

One final introductory note is in order. The question of marine recreational fishing benefits is far from a trivial one. Indeed, in the most careful and widely circulated summary of existing water pollution control benefit studies, marine recreational fishing looms very large, accounting for 24 percent of total annual benefits from the application of best available treatment, BAT (Freeman 1979). This makes it the largest single source of benefits, larger even than the sum of all diversion uses and three times larger than freshwater sport fishing. This last comparison alone would be enough to call into question the estimate of marine recreational fishing involved. Our knowledge of levels of participation and of values attributed to days of each kind of fishing makes it very unlikely that the result of changes in water quality could stand in this relation. Improving our confidence in these benefit estimates could be very important to the future of the water pollution control program.

METHODOLOGY: A FRESHWATER STUDY AND THE ROLE OF ECOLOGICAL MODELING

The method we use for estimating benefits of water pollution control accruing via sport fishing rests on the projection of changes in recreational fishing availability or quality as links between the results of pollution control policy and the decisions about fishing made by individuals. Thus, pollution discharges are projected to be reduced because of policy implementation. Those reductions are translated into increased availability and attractiveness of recreational fishing resources; and the availability and attractiveness of these resources is taken into account by individuals in their decisions whether and how much to engage in recreational fishing.

In our freshwater study (Vaughn and Russell 1981) availability was measured by surface acres, by state, of natural water bodies offering recreational fishing opportunity; that is, supporting some fish population. Quality within this total availability was indicated by dominant type of fish population supported, because it seemed that recreational fishing activities aimed at different species types might well involve both different decision relations for the individuals involved and different values of the experiences. The key observation here for freshwater fishing is that changes in the species distribution of the fish population brought about by improvements in such indicators as dissolved oxygen, suspended solids, and toxic chemical loads are themselves classified as improvements by anglers. Thus, said very simply, clean water means "game" fish (such as trout or bass), dirty water means "rough" fish (such as carp, drum, or buffalo fish), and game fish are generally preferred to rough fish by anglers. Thus, pollution control tends to increase the amount of water yielding higher quality fishing relative to that yielding lower quality.

Thus, our fundamental hypothesis is that discharge reductions ultimately lead to increased numbers of fishing days per year.[3] Because those days are valued at an amount greater than the sum of private (travel, tackle, etc.) and public (fisheries management) costs of producing them, they represent part of the benefit of the policy. A secondary hypothesis is that there will on average

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[3] We assumed that if such population shifts could take place they would take place, speeded along by state fish and game managers who share and reflect the commonly accepted view of angler preferences (Vaughn and Russell 1981).

be an upgrading in the quality of the fishing days as measured by the species sought. Higher quality days are valued more than lower quality days. This method does not provide an estimate of the benefit due to added enjoyment of each fishing day attributable to higher water quality per se (e.g., to greater clarity or better smell).

The details of the economic justification of our central hypothesis are rather arcane and unlikely to interest any but other economists. Even more arcane is the justification for using the change in participation (days) times the average of fishermen's prepollution-control values of their fishing days (distinguished by species sought) as the measure of program benefits. And our interests in this paper lie elsewhere in any case. Therefore, let us concentrate on the knowledge necessary to use the method, accepting that it is defensible. (The skeptical reader is invited to read Vaughn and Russell 1981 before making a decision on this point.)

The linkages we should be able to understand and model (whether formally or informally, in order to apply the method outlined above for obtaining national benefit estimates) may be summarized as follows:

- We should be able to predict how policy implementation will affect pollution discharges by location, quantity, and pollutant type across the entire nation.
- We should be able to predict how the pre- and post-policy discharge levels affect ambient water quality (or how ambient quality changes as discharges change) in terms not only of such familiar indicators as dissolved oxygen (DO), but in terms of supportable fish population types (or other measures relevant to participation decisions).
- We should be able to predict how increases in total amounts of water supporting recreational fishing and shifts in the composition of that water toward more highly valued fish species affect numbers of anglers and the amount of time they spend fishing.
- We should be able to put values on fishing activity of various kinds (that is, for practical purposes, on days spent fishing for various species).

This list suggests some of the formidable difficulties in the way of producing national benefit estimates for even this one subcategory. Just the first linkage, from law to changes in discharges, while not necessarily involving subtle economic or biological questions, does require massive amounts of information for every affected water body in the country. The last two linkages are the explicitly economic matters, and we shall ignore them in this paper. Instead, we shall concentrate on the second linkage, that from changes in pollution discharges to changes in one or another measure of water (or ecosystem) quality directly relevant to the sport fishing participation decisions of individuals.

Pollution Discharges and Water Quality Prediction in the Freshwater Study

We had available at Resources for the Future (RFF) a very large and complete inventory of point sources of residuals discharges to U.S. waterways. This included some data on baseline values of ambient water quality in terms of total suspended solids (TSS), temperature, and pH. Especially for the work reported here we gathered data on characteristics of U.S. waters by state in terms of ability to support any fish populations at all, and dominant fish populations supported, prior to implementation of the Fish and Wildlife Protection and Conservation Act and the Clean Water Act (FWPCA and CWA).

Baseline Discharge Inventory

Estimates of pollutant discharges from point source sectors were made for 1972 levels of control. The basic approach for most point source sectors was to assemble a plant inventory showing county location and plant output for all plants, by industry, in 1972. EPA-developed waste coefficients, based on survey data from actual plants, were then used to estimate total waste generated at the plant in 1972 assuming no end-of-pipe pollution control. The actual end-of-pipe wastewater treatment equipment in place in 1972 was then determined for each plant wherever possible. Standard removal efficiencies associated with this equipment were assumed and application of these efficiencies to estimated raw loads produced estimates of actual discharges for 1972. The basic source of data used in this project was a set of technical reports known as Development Documents, prepared by the Effluent Guidelines Division of the U.S. EPA.

Estimates of annual pollutant discharges from municipal sewage plants by county were derived from data on Biological Oxygen Demand (BOD) and TSS discharges from 24,209 individual municipal sewage treatment plants surveyed in the 1974 EPA's "Needs Survey" (U.S. EPA 1975).

The pollutants carried in nonpoint source discharges were also estimated for all nonpoint source sectors at the county level of detail. For certain sectors (e.g., irrigation section flow, urban runoff, stream bank erosion, and acid mine drainage), we relied on the analyses of other investigators and accepted their estimates of annual pollutant discharge. For a second set of sectors (e.g., ship ballast cleaning, accidental oil spills, and recreational boating), national estimates of discharges were prorated to counties using detailed information on the location and extent of each sector's activities. For a third set of sectors (e.g., sediment from construction and mining), national estimates of discharge were prorated to counties according to the county's share of employment in these activities weighted by an estimate of runoff per acre.

A set of county-by-county estimates of sediment loss and sediment-related pollutant discharge from nonirrigated cropland, woodland, pastureland, and rangeland were derived from estimates of gross soil erosion made by the U.S. Department of Agriculture (USDA). The USDA estimates were made by applying the Universal Soil Loss Equation (USLE) to about 200,000 field sample points as part of the 1977 National Resource Inventory. We then estimated sediment delivery ratios and ratios for pollutants attached to sediment for 156 geographically homogeneous regions and assumed them to be constant for all counties within each region.

The county-by-county discharge estimates are linked to a detailed water quality network model. This network model consists of 304 rivers, 175 lakes and reservoirs, and 37 bays. A total of 1,051 nodal points have been established along the rivers included in the model. See the schematic in figure 1. Each county in the United States is assigned to at least one node. (In certain cases more than one node was assigned if it was clear that the county's drainage went into more than one waterbody in the network.) The nodal points and water bodies are linked together in the network model in a logical downstream fashion.

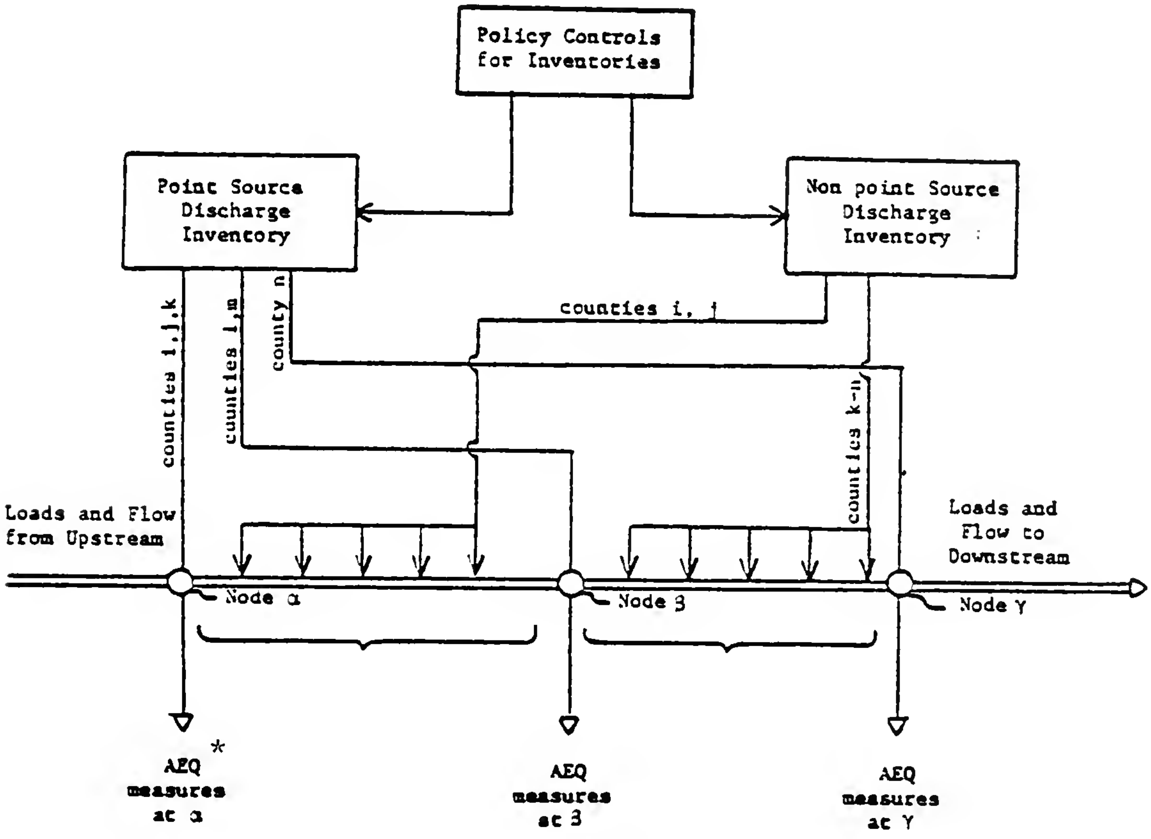

Figure 1.--RFF Water Quality Network Model: Schematic Structure for One Stream
*AEQ = Ambient Environmental Quality

The network model contains estimates for each river nodal point for the following hydrodynamic parameters:

Streamflow
Altitude
Width
Cross section area
Velocity
Surface water temperature

These estimates are drawn from published and unpublished data maintained by the USGS.

Using the above data, pollution transport functions convert injections of pollutants at the nodal points to concentrations of pollutants at and between the nodes. Initial injections of pollutants at a node are mixed with quantities of pollutants arriving from upstream nodes; the mixture is diluted by the flow at the node; and the mixture is transported to the next node while on the way certain constituents are removed by natural processes. Dissolved oxygen concentrations are estimated with Streeter-Phelps relationships. (The decay rates for all the modeled pollutants were provided by Ken Young of GKY and Associates, Inc. A detailed discussion of the parameters and equations may be found in Gianessi, Peskin, and Young 1981.) It should be noted that the model assumes that point source pollutants are inserted at specific nodal points. However, agricultural nonpoint pollutants are assumed evenly inserted along the reach between the nodal points.

Translating Ambient Water Quality into Dominant Fish Populations (Fishery Types)

Predictions of values of classical water quality parameters still leave us one large step from the desired goal of a water-quality-related measure connected directly to recreational fishing. In making that step we commissioned a survey of the fishery biology and management literature on which was based a set of rules (an informal model) that appeared to capture whatever consensus existed on the water quality conditions, in terms of DO, pH, TSS, and water temperature, necessary for the survival and reproduction of fish populations of various types. (Recall the assumption that managers will make sure that if upgrading is possible it will occur.)

These rules for determining type of fishing supportable are summarized as logical if-then statements in table 1. From the logical statements see that if dissolved oxygen is very low or temperature very high the water will be unsuitable for recreational fishing. At higher DO levels, the fishery type depends on both temperature and suspended solids loadings. But, because temperature is usually a function of natural factors, and suspended solids loadings are exogeneously specified in our water quality model and do not change in response to FWPCA/CWA implementation, policy-related water quality improvement as reflected in changing species class is a function of changing dissolved oxygen concentrations.[4] The improvement chains are shown schematically in table 2.

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[4] Other, similarly rough, rules for translating physical chemical water quality parameters into measures of fishing quality may be found elsewhere. For example, U.S. EPA in "The Red Book" (1976) asserts that "a minimum concentration of dissolved oxygen to maintain good fish population is 5.0 mg/l. The criterion for salmonid spawning beds is a minimum of 5.0 mg/l in the interstitial water of the gravel." (p. 123) Alabaster (1969) summarizes the then current state of knowledge about differential sensitivity of fish species to pollution and concludes:

> It seems that there is already sufficient information from laboratory and field observations to conclude that coarse fish are generally somewhat less sensitive to pollution than trout, though differences between species are not always the same for all poisons.

More recently, the English Water Research Centre summarized and commented on the development of European water quality standards designed to protect fish life (with some species differentiation)(Water Research Centre 1979). This report emphasized that the continuing gaps in our knowledge of fish response to pollution made guidelines more desirable than inflexible requirements.

Table 1.--Logical Statements Translating Ambient Water Quality Measures into Three Mutually Exclusive Fishery Groups According to Water Quality

Rule No.			Water Class
1.	If pH is above 10.0 then	:	No Fishery Supportable
2.	If pH is below 5.0 then	:	No Fishery Supportable
3.	If DO is below 2.0 mg/1 then	:	No Fishery Supportable
4.	If Temp. is above 35°C then	:	No Fishery Supportable
5.	If summer streamflow is zero then	:	No Fishery Supportable
6.	If DO is between 2.0 mg/1 and 3.0 mg/1 then	:	Rough Fish
7.	If DO is above 3.0 mg/1 and Temp. if above 32°C then	:	Rough Fish
8.	If DO is above 3.0 mg/1 and TSS is greater than 100 mg/1 then	:	Rough Fish
9.	If DO is greater than 3.0 mg/1, Temp. is between 18°C, and TSS is below 100 mg/1 then	:	Warmwater Gamefish/ Panfish
10.	If DO is greater than 5.0 mg/1, Temp. is less than 18°C, and TSS is less than 100 mg/1 then	:	Coldwater Gamefish

Key:

DO: 90-day average summer dissolved oxygen concentration
Temp.: 90-day average summer temperature
TSS: Annual average total suspended solids concentration

Table 2.--Fishery Types as Determined by Water Quality Using Nielsen's Rules

Water Quality Improvements Traced to Policy	Exogenously Determined Water Quality Conditions			
	Cold Water Temperature $\leq$ 18°C		Warm Water Temperature Between 18°C and 32°C	
	Low Solids Water TSS $\leq$ 100 mg/l	High Solids Water TSS $\geq$ 100 mg/l	Low Solids Water TSS $\leq$ 100 mg/l	High Solids Water TSS $\geq$ 100 mg/l
Dissolved Oxygen > 5.0 mg/l	Coldwater Gamefish			
> 3.0 mg/l			Warmwater gamefish/ panfish	
> 2.0 mg/l	Roughfish	Roughfish	Roughfish	Roughfish
< 2.0 mg/l	Unfishable [a]	Unfishable [a]	Unfishable [a]	Unfishable [a]

[a] Other unfishable conditions include pH greater than 10 or less than 5; temperature greater than 32°C; summer flow zero.

PROBLEMS IN APPLICATION TO MARINE RECREATIONAL FISHING

Application of the above method to the case of marine recreational fishing requires that we find some index or measure of water or ecosystem quality that can be used to link changes in pollution discharges to changes in individuals' participation decisions. Beyond this, we must have data on pre-policy levels of quality and on the related level of participation. Further, it must be possible to value in a defensible way pre-policy and post-policy participation. Here we can continue to concentrate on the linkage requirement, since this brings us to the core of the paper.

First, let us note the several conditions that such a linking index or measure must satisfy.

1. The index, be it dominant species type or whatever, must be linked backward to measures of ambient water quality, which in turn are linked to levels of pollution discharge.

2. There must be an a priori reason to expect participation decisions to reflect values of the index.

3. Data must be available to test the hypothesis from 2 and in the process to product projection equations for participation as a function of the index.

In moving from freshwater to saltwater recreational fishing, several possibilities for this key index present themselves. An obvious candidate is dominant species, as in freshwater, and there exists some evidence that this effect can be significant, at least for well-defined bodies of water such as enclosed bays and estuaries. Resident populations within semi-enclosed marine waters with serious pollution problems apparently are dominated by such fish species as bay sardines and toadfish. Marine gamefish such as bluefish and striped bass do not seem to frequent dirty areas, nor do they use contaminated estuaries for spawning or nursery areas. but because of the tremendous available dilution and the mobility of major marine species, it seems unlikely that any significant fraction of the nearshore ocean would be found to be dominated by low quality species (e.g., U.S. Department of the Interior 1970, Sindermann 1975, McErlean et al. 1972).

A second possibility for the index is some measure of fish population (numbers, or average size, for example) as a proxy for prospective success in angling (bag). This was the basis of the benefit estimation procedure used by Bell and Canterbery in their major study (1976). There seem to be four problems with this measure, however. First, the appropriate variable, standing as a proxy for prospective angler bag, may be a complicated, but is certainly an unknown, combination of the several possible measures; number of fish, average size; and number of very large fish. Second, the measurement and prediction of any indicator of bag raises a difficult question of simultaneity, for participation and bag are probably connected through the sustainable yield function for a species (or group of species) as well as through the participation equation reflecting human decisions. Third, with so many important species highly migratory in habit, tracing any pollution effects on a size variable would require tracing migration routes and understanding effects that may involve only one stage of a fish's life cycle. Fourth, even for resident fishes, the effect of some kinds of pollution, especially that involving nutrients and organic carbon, may be ambiguous. (See, for example, Chittenden 1971, 1976; Roberts et al. 1975; Wise 1974; McHugh 1972, 1975; Hedgpeth 1966; Pararas-Carayannis 1973; Carlisle 1969; Brown and Beck 1972; Bascom et al. 1979; Soule and Oguri 1979; National Marine Fisheries Service 1972; Butler et al. 1972; Smith 1973.)

A third possibility might be thought of as an amalgam of the first two--some index of ecosystem health, reflecting species types present and abundance, and indicating the overall quality of the available experience. (See, for example, Hillman et al. 1977, McErlean et al. 1972, Bader et al. 1970, Bechtel and Copeland 1970, Haedrich 1975, Livingstone 1975.)

Practical application of any of these possible indexes requires that we be able to make the backward (to discharges) and forward (to angling decisions) links. An initial reconnaissance of the literature on pollution effects on marine fishes has convinced us that the backward link will at least be very difficult. With the exception of laboratory studies of acute toxicity using a variety of elements and compounds and many different fishes, there seems to be remarkably little systematic quantitative understanding of how pollution of natural saltwater effects the availability of fishes of different species, or of how it affects population numbers and size distributions.

In table 3 we summarize the findings of our survey to emphasize for the reader the paucity of pollution-related information (that on suspended solids) relative to that on temperature and salinity tolerances. Further, the information we were able to find on dissolved oxygen tolerances--the single most important pollution-related water quality measure for our freshwater work--is confined to a few studies, too varied in methods and results to permit meaningful comparisons.[5]

Thus, unless further searching can improve our data base in this key area, it appears we cannot make the backward link using models of pollution discharge dispersion, dilution, and transformation as the bases for predicting fish availability or quality measures. This will be especially unfortunate because we do anticipate having available at RFF a coastal county discharge inventory, complete for the lower 48 states, and reflecting point and nonpoint sources, including the inflow of pollution loads in freshwater streams.[6] It will be possible to project how these loads will change with implementation of the clean water legislation of 1972 and 1977.

In terms of the link forward to angling decisions, the species type, prospective bag, and ecosystem effects are all defensible as influences on participation decisions. However, to use any of these indices it will be necessary that it match the available data on participation. That is, differential levels of participation across measurement units must be tied to differential values of the chosen index(es). For example, to our knowledge the best available data on participation in saltwater angling identifies individuals no more finely than by state of residence.[7] In order to test the effect of an

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[5] We found DO references for Atlantic cod (Wise 1961, and Davis 1975); American eel (Usui 1974); Atlantic herring (Dorfman and Westman 1970); white perch (Dorfman and Westman 1970); shad (Chittenden 1973, Hoff et al. 1966); striped bass (Chittenden 1971, Dorfman and Westman 1970); and tuna (Dizon 1977).

[6] The inventory is being developed as part of the Strategic Assessment Project of the Office of Resource Coordination and Assessment of the National Oceanic and Atmospheric Administration. This project also involves the generation of maps showing areas of critical importance in the life cycles of numerous coastal fishes (fin and shell) and it may be possible that by combining inventory and maps we can produce a useful mixed (quantitative and qualitative) model. See Ehler et al. n.d.

[7] This is the nature of the data from the 1975 National Survey of Hunting, Fishing, and Wildlife-Associated Recreation, NSHFWR (U.S. Department of the Interior n.d.).

index value on participation we would require measures of pre-policy index values appropriate to each state. For example:

- If we had species availability data, we might use the analog to our freshwater measures, state acres per capita of water dominated by particular species types.

- For abundance, we might use average state saltwater bags.

- For ecosystem "health" we might use a state index of marine water quality, perhaps one created out of a combination of species dominance and abundance measures.

In any event, the basic problem is finding a measure for which values are available for all coastal states...no trivial requirement. One route to such knowledge that might be explored is via state fish and game departments, coastal zone commissions, sea grant universities, and other state and local institutions. This would involve a mail survey supplemented as necessary with phone calls and even personal visits. Our experience with a similar study in the freshwater context was that a significant fraction of the actual respondents (staff biologists) will quarrel with the terms of reference and even with the purpose of the overall project and only grudgingly provide their own versions of local knowledge.

CONCLUSION

In short we cannot be entirely sanguine about the prospects for successful completion of a study in the marine context analogous to our study of the benefits of water pollution control accruing via freshwater recreational fishing. The major reason for this very cautious view is the apparent lack of an index of water or ecosystem quality about which we know enough to provide model links backward to pollution discharges and forward to angler perceptions and tastes. Important questions for this workshop might well be:

- To what extent can existing marine ecosystem models provide predictions in terms relevant to pollution control policy assessment--whether strictly benefit estimation or something not involving translation to dollar terms?

- If our rather negative answer to this question for marine recreational fishing holds across other benefit categories, such as swimming and commercial fishing, should it matter to NOAA? To the modelers at this workshop?

- If it does matter, what might be done to improve the situation? Is it entirely a matter of long-run research, or are there short run improvements that could be made by, for example, clever linking of existing models and isolated research results?

Table 3.--Some Preliminary Information on Species Tolerance of Three Environmental Parameters

Temperature (°C)	Salinity (parts per thousand)	Suspended Solids[a]
Low Temperature Range	Tolerant of Low Salinity (20)	Extremely Sensitive ($LC_{50} \leq 1000$ppm)
0-15 cod, hake (squirrel), pollock 0-20 herring, smelt, hake (white), winter flounder 5-20 hake (silver), searobin (north), Atlantic mackerel, cunner 5-25 striped bass, scup	perch, shad, windowpane, weakfish, tancod, herring, flounder (winter, summer), scup, puffer, mackerel (king), cod, croaker, drum (red, black), eel, crevalle jack, ladyfish, grey mullet, pinfish, smelt, spot, striped bass, seatrout (sand, spotted), salmon	herring (menhaden)
		Sensitive ($1000 < LC_{50} \leq 10,000$ ppm)
Medium Temperature Range	Narrow Salinity Range ($10 \leq S \leq 40$)	white perch striped bass croaker weakfish
10-30 black sea bass, kingfish, American eel, summer flounder, perch, bluefish 10-35 puffer, spot >10 shad, seatrout (spotted) 13-30 American eel, red porgy <30 flounder (summer)	pigfish, spadefish, kingfish, bonito, grunt, barracuda, cunner, mackerel (Spanish), searobin (striped), tuna, seatrout (silver)	
		Tolerant ($10,000 < LC_{50} < 100,000$ppm)
		spot cunner
High Temperature Range	Tolerant of High Salinity (> 55)	
>15 seatrout (sand), tuna, little tunny, mullet (grey), grouper >18 bonito, snapper (grey) >20 mackerel (king, Spanish) dolphin <32 crevalle jack, white perch 20-35 barracuda <38 sea catfish	croaker, drum (red, black), eel, flounder (summer), crevalle jack, ladyfish, grey mullet, pinfish, smelt, spot, seatrout (spotted)	Extremely Tolerant ($LC_{50} > 100,000$pp) toadfish

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[a] LC_{50} is concentration of solids at which 50 percent of sample died with 24 hours exposure. Thus, these concentrations refer only to short term acute effects. For longer run toleration, concentrations would have to be much lower, and placement of fish in classes might or might not change.

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USE AND LIMITS OF ECOSYSTEM MODELS: A GREAT LAKES PERSPECTIVE

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INTRODUCTION

During the past 10 years the Great Lakes Basin has been an exciting proving ground for developing, testing, and applying numerical, aquatic ecosystem models. Several research efforts developed during the early 1970's have been brought to bear on issues of overenrichment and contamination of these magnificant inland seas. A certain amount of maturity has evolved in the modeling field from the experience. Where models were originally touted as mystical algorithms with the ability to make decisions for managers, they are now in better perspective and have become useful aids for the decisionmakers (e.g., PTSTF, 1980). The applications and explored limits of models of the Great Lakes have important implications, not only for analysis of limnological systems, but for marine ecosystems as well.

Probably the most significant insight gained from our experience has been to not rely on a single model or even a single class of models for analysis of the Lakes. The collective efforts on the Great Lakes have included models from the simplest, empirical correlations between nutrient levels and in-lake water quality to more theoretical (and less verified) models simulating dynamic, time-dependent mechanisms of biological, chemical, and physical process interactions. Heidke's (1979) summary of the modeling efforts on the Great Lakes lists as many as 50 different water quality models that have been calibrated and/or verified for one or more of the Great Lakes.

Rather than repeat or augment that report, I will present herein results from several studies carried out at NOAA's Great Lakes Environmental Research Laboratory that demonstrate uses and limits of ecosystem models. I will describe briefly an ecological model developed and calibrated for data amassed during the International Field Year for the Great Lakes (IFYGL) on Lake Ontario and illustrate how its use has provided a better understanding of the structure and function of that lake. I will then describe results from studies that explored the limits to this and similar models by assessing, first, a consequence of nonunique coefficient estimates in light of field measurements and, second, error propagation from model inputs to predictions.

THE IFYGL MODEL

The ecological model (Scavia 1980a) simulates phytoplankton, zooplankton, cycles of phosphorus, nitrogen, silicon, and carbon, and oxygen balance; and calculations of carbonate equilibrium (fig. 1). Each model compartment is described by a differential equation representing biological and chemical processes. For example, the phytoplankton equation includes terms for gross primary production, respiration, excretion, grazing, and sinking; and the zooplankton equation includes terms for grazing assimilation, respiration, excretion, defecation, and predation.

Gross phytoplankton production is modeled as a single-step process that assumes, for the time scale of the model, that rates of uptake and growth are in equilibrium. The process is modeled with a temperature-dependent maximum growth rate times a reduction factor for light and nutrient limitation. Potential light limitation is modeled after Steele (1965) and potential nutrient limitation is expressed as a saturating function for each nutrient. The

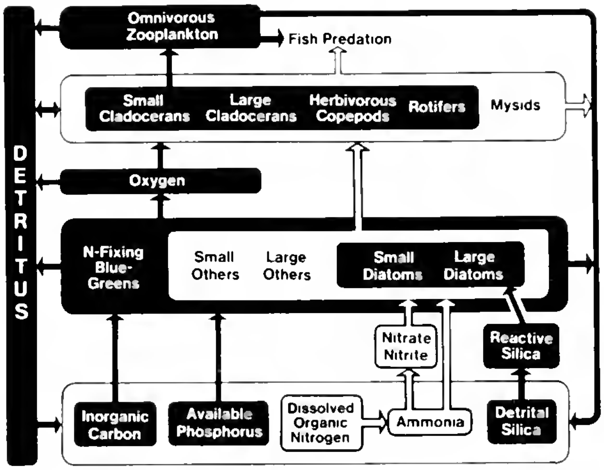

Figure 1. Components of the ecological portion of the model. From Scavia (1979).

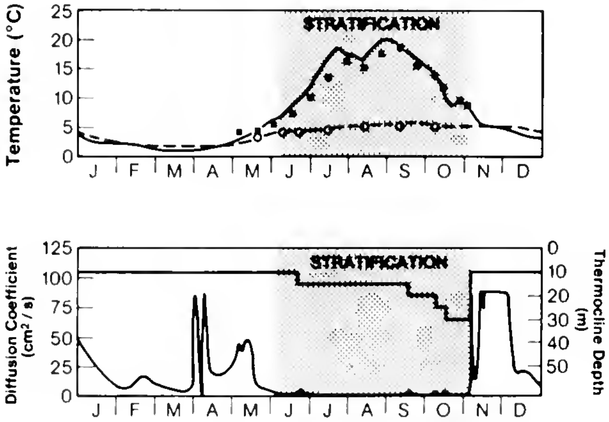

Figure 2. Simulated and observed lakewide averaged temperatures. (Top): ■ - epilimnion, O - hypolimnion. Simulated thermocline depth and diffusion coefficients (bottom). From Scavia (1979).

threshold formulation is used to determine which nutrient or light is limiting. Phytoplankton respiration is considered to be the sum of a low maintenance rate plus a term proportional to production and temperature. To maintain constant nutrient stoichiometry within the plankton, nutrient excretion is proportional to respired carbon.

Zooplankton grazing is a temperature-dependent saturation expression based on total food supply. The expression includes a minimum threshold for feeding as well as size-selective resource partitioning. Feeding and assimilation efficiencies are food-specific constants. Food ingested but not assimilated is egested as detritus. Respiration is a temperature-dependent process that is the sum of a low maintenance rate and a term proportional to the feeding rate. Nutrient excretion is proportional to respiration.

Transformation rates between detritus, dissolved organic nitrogen, ammonia, nitrite plus nitrate, available phosphorus, and available silicon pools all are temperature dependent and first order. Sinking rates of phytoplankton and detritus are size and density dependent.

The purpose of the model is to synthesize information on process kinetics into equations describing the rate of change of ecosystem components as functions of the components themselves, of environmental variables, and of group-specific coefficients. Process equations posed by experimentalists over the past several years are used to describe specific biological and chemical processes. For some of the processes more than one theory or hypothesis has been espoused by researchers as a control or regulator. Use of a particular process equation in the overall framework is one way to test that theory or hypothesis in the context of the larger system. As such, the model is merely a tool allowing a quantitative evaluation of isolated process theories in a whole-system context.

The model considers the lake to be homogeneous horizontally and to be segmented vertically into two layers representing lakewide averaged epilimnion and hypolimnion. A 1-dimensional, 18-layer heat diffusion model calibrated to temperature profiles measured in Lake Ontario is used to calculate variable depth of the thermocline and average epilimnion and hypolimnion temperatures. Values of the diffusion (or exchange) coefficient between the two layers are then calculated from temperature changes in the two layers and from the extent of thermocline displacement (fig. 2).

The ecological model was calibrated with data collected during the International Field Year for the Great Lakes (IFYGL) from March to November 1972. Documentation of seasonal cycles of IFYGL data and model output, as well as detailed discussions of model equations, coefficients, and sensitivity analyses, are presented elsewhere (Scavia 1980a).

Although the model includes detailed process equations describing the food web and nutrient cycles in Lake Ontario it is still a crude representation of reality. Therefore, before this model was used to examine lake-scale phytoplankton production and phosphorus cycling, its adequacy as a representation of the seasonal dynamics of relevant lakewide averaged properties in Lake Ontario was tested. This was done by comparing measured and simulated properties. The comparisons are best for those properties measured with least uncertainty (i.e., chemical properties) and worst for those properties difficult to measure (i.e., phytoplankton and zooplankton carbon concentrations); however, these comparisons result in general agreement between measured and simulated properties (figs. 3-5). This demonstrates the model's ability to simulate the seasonal dynamics of phytoplankton, zooplankton, and major nutrients; however, model output should also be compared to measurements of process rates to see how well the model simulates the internal dynamics of the system. In the discussions that follow, both simulated and measured process rates are used to ensure that internal model dynamics are consistent with observations. I use process rates that were measured during IFYGL or measured in Lake Ontario during other years. Also, where certain processes were not observed in Lake Ontario, information from other field and laboratory studies are used for general comparisons. Simulated and measured rates were also in general agreement, which allows one to use the model to speculate about the relative importance of these processes. This emphasis on process rate comparisons is discussed further below in the context of nonunique parameter estimates.

ANALYSIS OF LAKE ONTARIO

This model was exercised within two physical frameworks. First, to explore the controls of phytoplankton production on a lake-scale, seasonal basis, output from the horizontally averaged, two-layer model described above was analyzed with particular emphasis on cycling of phosphorus, the primary limiting nutrient (Scavia 1979). Second, to explore development and maintenance of strong offshore nutrient and plankton gradients, output from a two-dimensional, coupled ecological-physical transport model was analyzed for the spring-summer transition period (Scavia and Bennett 1980).

Lake-scale, seasonal dynamics - Simulated processes controlling seasonal dynamics of one of the modeled phytoplankton groups are shown in figure 6. Stippled areas on each graph indicate net rate of change of population biomass. In winter and early spring, this algal group, as well as the others, is controlle primarily by the balance between gross primary production and two physical processes, sinking and vertical mixing. During this time, phytoplankton gross production is limited mainly by the availability of light which is controlled both by the amount of incoming solar radiation and the depth to which the phytoplankton are mixed. Mixing between the two model layers becomes quite intense in early spring (fig. 2), indicating that the mixing depth is the depth of the entire water column. Thus, loss to dark, deeper waters prevents substantial increases in algal biomass.

Riley (1942, 1946, 1963) and Riley et al. (1949) discussed the importance of the relationship between depth of the sunlit zone and mixing depth. They suggested that phytoplankton realize their productivity only after the thermocline begins to develop because it is only after this time that losses to the dark, nonproductive strata are reduced. Significant increases in algal biomass occur only after midspring, when surface waters in Lake Ontario begin to warm and the lake begins to stratify vertically (fig. 6). At this time (early June), phytoplankton populations increase rapidly and concentrations of nutrients they assimilate begin to decrease. The concentrations of nutrients decrease because they also become relatively isolated from the nutrient-rich lower strata.

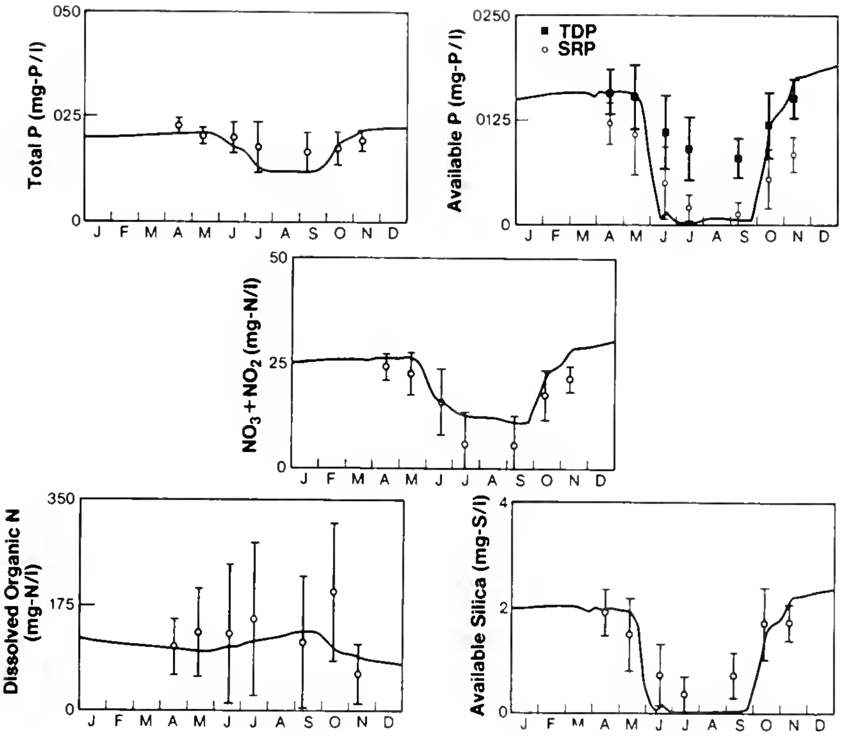

Figure 3. Seasonal dynamics of data and model output for epilimnion (0-20 m) of Lake Ontario during 1972. (a) Total P; (b) filterable P; (c) nitrate and nitrite; (d) dissolved organic N; (3) soluble reactive Si. Data are lake-wide mean ± 1 S.D. From Scavia (1980a).

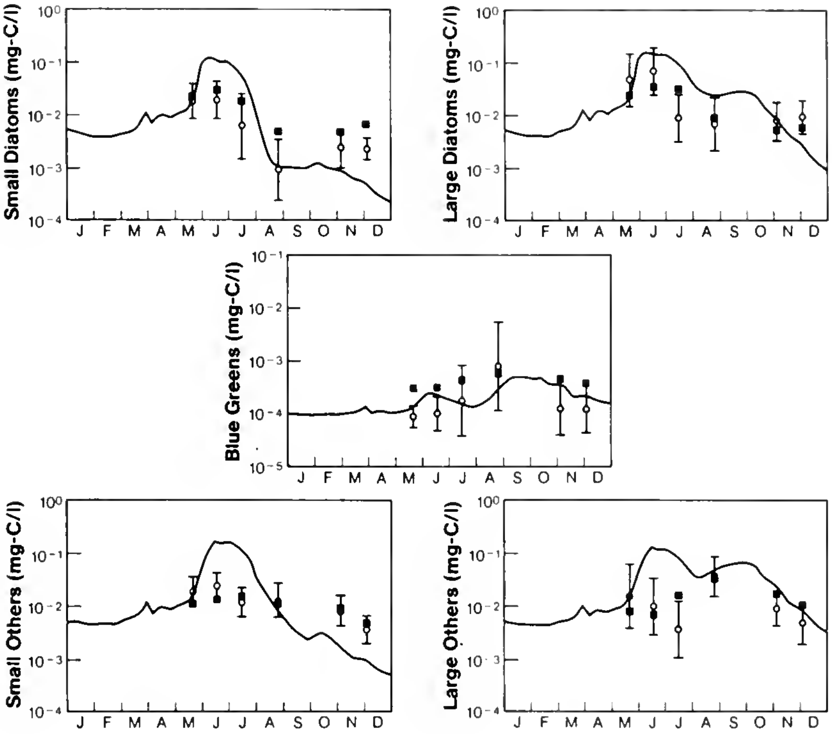

Figure 4. Seasonal dynamics of data and model output from epilimnion (0-20 m) of Lake Ontario during 1972. (a) Small diatoms; (b) large diatoms; (c) blue-greens; (d) small others; (e) large others. Φ - lake-wide mean ± 1 S.D. for surface samples and ■ - lake-wide mean for depth-integrated profiles. From Scavia (1980a).

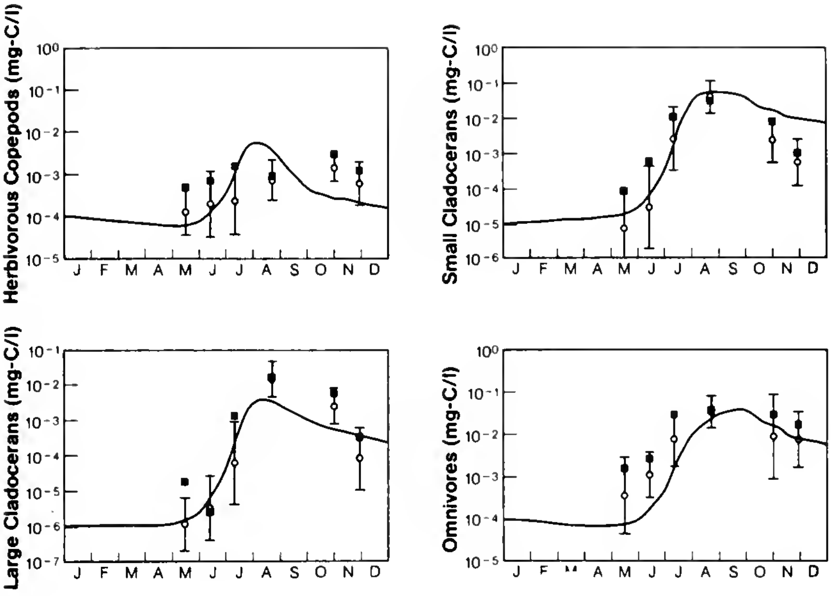

Figure 5. Seasonal dynamics of data and model output for epilimnion (0-20 m) of Lake Ontario during 1972. (a) Herbivorous copepods; (b) small cladocerans; (c) large cladocerans; (d) omnivores. Symbols as in Figure 4. From Scavia (1980a).

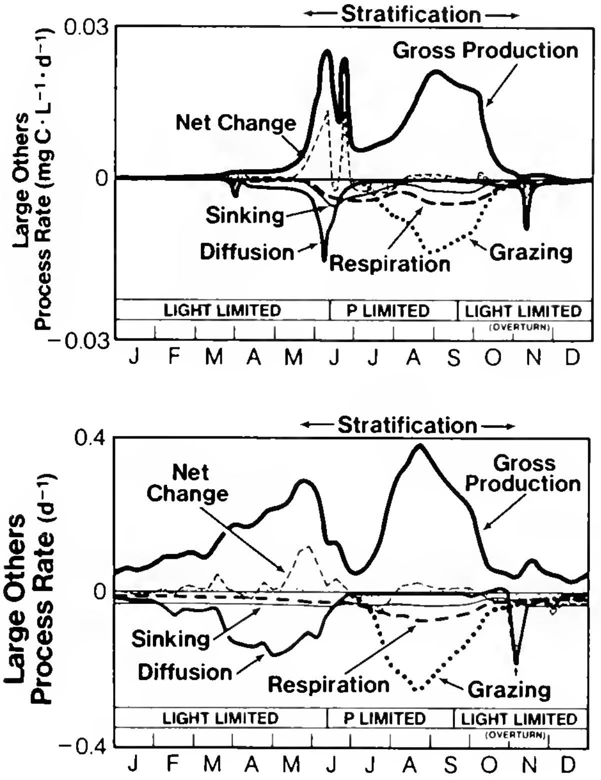

Figure 6. Rate plots indicating simulated controls of epilimnion phytoplankton dynamics in Lake Ontario during 1972. Large Others represent nondiatoms greater than 20 μm. Top - Rates as d^{-1}, bottom - rates as mg $C \cdot L^{-1} \cdot d^{-1}$. Strippled area represents net growth rate. From Scavia (1979).

Diatoms, dominated by species with temperature optima in the lower range of Lake Ontario temperatures (Stoermer and Ladewski 1978), increase their productivity sooner than the other taxa. Consequently, diatoms assimilate available silicon and become limited by that nutrient before other groups deplete the available phosphorus pool and become phosphorus limited. Late in summer, phosphorus becomes limiting for diatoms as well. Phytoplankton production is limited by nutrients (silicon and phosphorus) from this time until the end of September. This has also been suggested previously, based on mass balance considerations (Stadelmann and Fraser 1974) and on algal assays (Sridharan and Lee 1977).

During the same period (late summer), grazing stress by zooplankton becomes most intense (fig. 6). This timing of simulated grazing pressure reflects the general seasonal pattern of crustacean zooplankton biomass (fig. 5). All of the dominant species in Lake Ontario produced major biomass peaks during July or August in 1972 (McNaught et al. 1975). Also, for a previous year in Lake Ontario, Glooschenko et al. (1972) measured and compared the relative abundances of chlorophyll *a* and pheopigments and suggested that a high correlation between average percent pheopigment (relative to total chlorophyll *a*) and zooplankton abundance was probably a result of zooplankton grazing. On a lakewide average basis, they found highest values for the percent of total chlorophyll *a* as pheopigments to occur in August-October. This further substantiates the simulation results in figure 6.

In late September the thermocline deepens (fig. 2) and nutrient-rich, hypolimnetic water is mixed with epilimnetic water. Because of this increase in nutrient concentrations and the simultaneous increase in mixing depth, algae again become limited by light. In early November the lake overturns and becomes vertically homogeneous and phytoplankton concentrations begin to approach winter values. Of course, this is a simplification of the three-dimensional effects discussed by Simons (1976); however in a one-dimensional model, all advective and dispersive processes are parameterized as vertical mixing.

Although it is clear from the above analysis that nutrient limitation does not solely control phytoplankton dynamics, the role of nutrients, especially phosphorus and silicon, is certainly critical during the period of stratification. Phytoplankton dominant during summer months in Lake Ontario are limited primarily by phosphorus, as demonstrated in simulations discussed above and as shown by recent experimental work (Sridharan and Lee 1977). Therefore, to understand better the control of phytoplankton dynamics in Lake Ontario, one must investigate processes influencing the cycling of phosphorus.

Figure 7 illustrates the seasonal changes in simulated concentration of available phosphorus and rate of gross primary production in the epilimnion. As discussed above, after spring, phytoplankton become limited by nutrients and thus the production rate decreases sharply. It is interesting to note, however, that although production decreased considerably, it did not approach low winter values and, in fact, after the initial drop, it increased gradually. This sustained production proceeds at the same time that available phosphorus concentrations, both actual and simulated (fig. 3, 7a), are extremely low. One might expect that, with phosphorus concentrations this low ($<1\mu g$ P/L) and sustained phytoplankton production, phosphorus assimilation by algae would rapidly drive the concentration of phosphorus to virtually zero and thus limit severely further phytoplankton productivity. However, this is not the case. It appears that supplies of phosphorus during this time period are sufficient to balance phytoplankton assimilation. Thus the importance of internal cycling of phosphorus (Golterman 1973, Rigler 1973) cannot be ignored. The sum of rates of detritus remineralization, phytoplankton and zooplankton excretion, and diffusion input from the hypolimnion is approximately equal to the rate of assimilation by phytoplankton. This analysis (fig. 7) suggests that decomposer input is about one-fourth the excretion input from algae and zooplankton during summer stratification and that phosphorus input from the lower waters is important only before and after stratification.

To examine more closely relationships among various processes in this conceptual phosphorus cycle, I constructed a phosphorus flow diagram (fig. 8) from model output averaged over the period July-September and compared the results to available information. Sizes of the five phosphorus compartments are representative of Lake Ontario for this time period (figs. 3-5). Rate of conversion of available phosphorus to particulate phosphorus (Stadelmann and Fraser 1974) and fluxes of phosphorus across the thermocline (Stadelmann and Fraser 1974, Burns and Pashley 1974) are also representative. Zooplankton grazing rates represent 59 to 53 percent of the animals' body phosphorus per day for omnivores and herbivores, respectively. This is consistent with general dry weight rations for small crustacean zooplankton (Parsons and Takahaski 1973). Zooplankton excretion rates represent 14 and 11 percent of body phosphorus per day for omnivores and herbivores, respectively. These rates agree with excretion rates summarized by Ganf and Blazka (1974) if one assumes for these animals a nitrogen to phosphorus body weight ratio of approximately 11 (cf. Parsons and Takahaski 1973). Quantitative information on other processes is not available and therefore, in those cases, model information alone will be used.

Figure 8 shows that it would take less than 1 day in the summer epilimnion for phytoplankton to deplete the available phosphorus pool if there were no recycling and that external sources and hypolimnetic sources alone could not meet this algal phosphorus demand. In fact, this analysis indicates that 86 percent of the assimilated phosphorus is recycled within the epilimnion. Stadelmann and Fraser (1974) estimate this recycling to be 87 to 93 percent based on mass balances for the upper 20 m at a single station during the same time period.

Rigler (1973) suggested that excretion by zooplankton and direct release from ultraplankton were equally important. Results of the present analysis suggest that, for this five-compartment conceptualization, zooplankton excretion and direct release by phytoplankton are approximately equal and are the most important processes supplying phosphorus to the available pool. The rate of remineralization of detrital phosphorus is somewhat slower. This rate represents regeneration of approximately 1.7 percent of the detrital phosphorus per day, which is within the range measured by DePinto and Verhoff (1977).

The role of zooplankton in the phosphorus cycle must be emphasized. While the zooplankton has an obvious role in applying pressure to reduce algal concentrations (fig. 6), it also appears to play a dual role in recycling phosphorus. Not only does the zooplankton input directly to the available nutrient pools through excretion, but it also serves as a supplier of detrital material (feces),

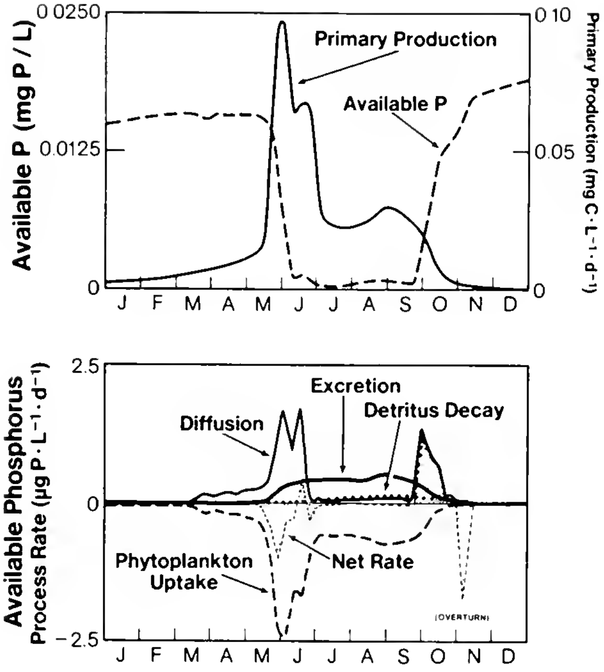

Figure 7. Top: Simulated concentration of available phosphorus and rate of gross primary production in the epilimnion. Bottom: Rate plots indicating control of available phosphorus dynamics. Stippled area represents net rate of change. From Scavia (1979).

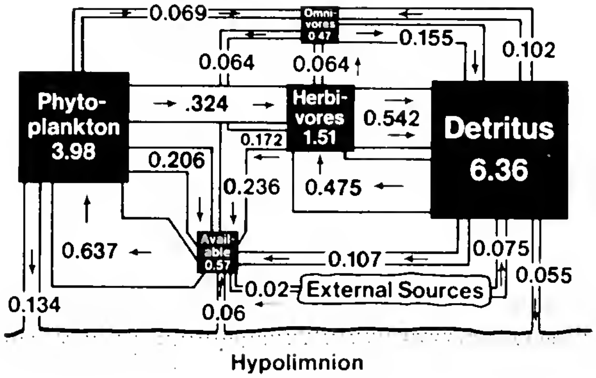

Figure 8. Phosphorus flow diagram. Concentration in boxes are in µg $P \cdot L^{-1} \cdot d^{-1}$. These values are averaged over the period of July-September for the top 15 m. Phytoplankton, zooplankton, and detritus are evaluated in the model in terms of carbon and are converted to phosphorus for this figure by assuming a $C_{100}P_1$ atomic ratio. From Scavia (1979).

which eventually adds to the available nutrient supply. Thus, it may well be that the zooplankton is principally responsible for high recycling rates estimated by Stadelmann and Fraser in Lake Ontario.

Control of spatial gradients - While the above analyses provide information regarding the major controls of biological production in Lake Ontario, they were based on intentionally crude physical segmentation. The two-layer representation reduced the complexity of physical interactions and thus allowed focus on biological and chemical interactions. However, there can often be vast differences between lakewide averaged conditions and nearshore conditions, especially during the spring-summer transition period. Because it is nearshore conditions that most often affect people, one must be concerned with spatial variations.

The spatial distribution of biological and chemical properties in the Great Lakes is determined by variations of water depth, sunlight, and temperature, by the loctions of rivers, and by currents. Separating these physical factors from in situ transformation is a difficult problem, a problem which numerical simulation can help solve.

During the transition period between spring and summer both vertical mixing and large-scale circulation are important. Also, the temperature and current patterns of Lake Ontario are relatively two-dimensional (i.e., small longshore gradients); therefore, variations along the long axis of the lake are negligible. We (Scavia and Bennett 1980) simulated flow, temperature, and biological and chemical processes for a north-south transect of the lake. Our approach was as follows.

Temperature calculations of a hydrodynamic model (Bennett 1971, 1974) were compared to observations, and the model was adjusted until computed and observed temperature contours were in general agreement. We then repeated the calculations, this time including the chemical and biological processes from the ecological model decribed above (Scavia et al. 1976, Scavia 1980a), and compared them to observations.

During the spring transition period a combination of strong heating and low wind speeds causes the thermocline to form. Because the lake begins the spring colder than 4°C (the temperature of maximum water density), this process starts in shallow water. Thus the lake is divided into two hydrodynamic regions -- a deep region where the water is less than 4°C and where surface heating causes vertical mixing, and a shallow region near the coast with temperature greater than 4°C, which may stratify.

In figure 9, simulated temperature is compared with observations. The model correctly simulates this general spring temperature pattern and the depth of the thermocline. In addition, thermocline extension further off the north shore than the south is also reproduced.

Wind and buoyancy combine to cause simple but interesting flow patterns. The wind tends to drive a one-cell pattern, with upwelling at shore to the left of the wind and downwelling near the opposite shore. Heating drives a two-cell pattern, with warm water rising near both shores and colder water sinking in the deep region.

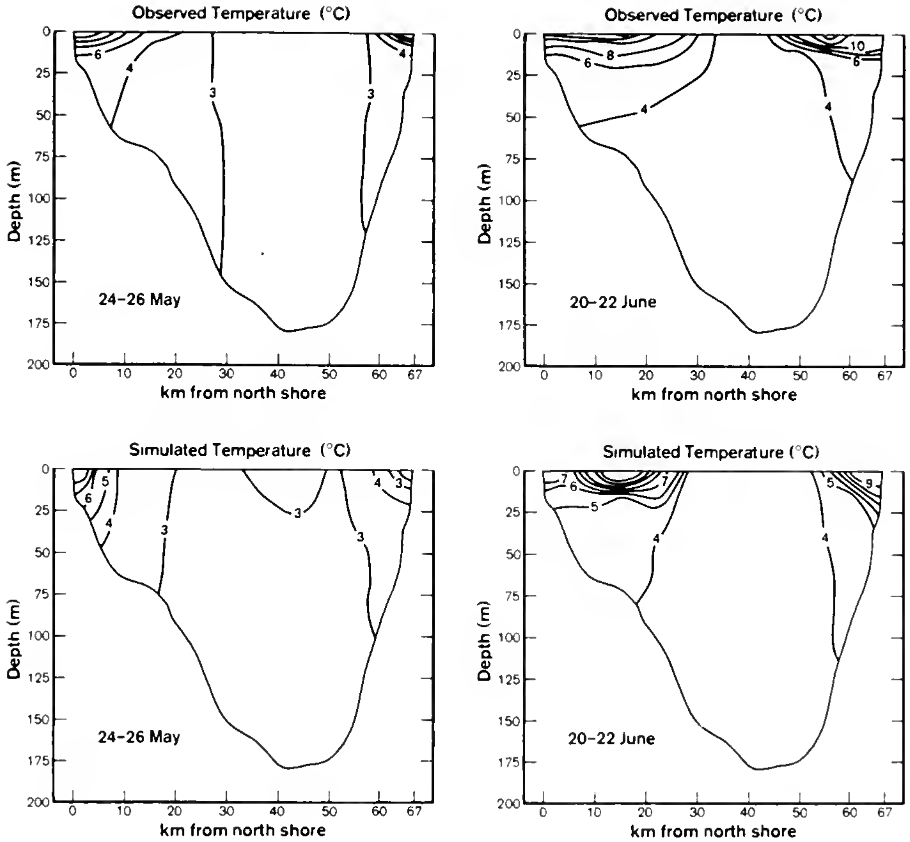

Figure 9. Comparison of observed (upper) and calculated (lower) temperatures (°C) corresponding to May and June cruises. From Scavia and Bennett (1980).

Figure 10 shows mean circulation for three 24-day periods and the mean for the entire 72-day simulation. During the first two 24-day periods the wind came predominantly from the east; because the Coriolis force deflects the water to the right, water near the surface flows north and deep water flows south in compensation. Upwelling near the south shore and downwelling near the north shore closes this circulation. During the last 24-day period the wind and circulation reverse. For the whole 72-day period the wind's effects tend to cancel and circulation looks like the thermally driven pattern.

The simple picture of a combination of wind and buoyancy effects should be considered an average circulation pattern. At any given time flow is dominated by wind, and it is only because the thermally driven flow is more persistent that it is as important as the wind-driven flow.

In general, nutrient concentrations had slight offshore gradients and were homogeneous vertically in early April. These data were used to initiate the simulation. By the end of May, distinct offshore gradients had developed; nutrients, especially phosphorus and silica, were severely depleted in regions within 10 km of both shores (fig. 11). Also, at this time, no strong vertical gradients were obvious in either the model output or the data. At the end of the period of simulation (corresponding to the June 20-22 cruise) the symmetry of the north and south shore contours was lost. The region of nutrient depletion along the north increased to greater than 25 km and vertical stratification was not as strong there. The spatial and temporal progression of the region of nutrient depletion is demonstrated for phosphorus in figure 11; inorganic nitrogen and soluble reactive silica have similar patterns. The comparison between observed total dissolved phosphorus (TDP) and modeled available phosphorus (AP) is made because previous simulations of phosphorus cycling in Lake Ontario (Scavia 1979) indicate that in spring AP is approximated best by TDP, due, presumably, to production of easily hydrolyzed phosphorus compounds during the previous winter.

Biomass parameters (chlorophyll *a* and particulate organic carbon) in April were relatively homogeneous vertically, with higher values nearshore. Patterns in May generally showed offshore gradients and little vertical structure (fig. 12), except for evidence of nearshore subsurface chlorophyll peaks, which the model did not reproduce. The simulations indicated that between May and June cruises, upwelling moved the higher nearshore concentrations offshore, creating a lens of high biomass about 15 km from the north shore. (See figs. 9 and 10.) By the June cruise, increased nearshore production apparently created higher biomass again close to shore. A similar structure was produced along the south shore with the exception that, like the nutrient contours, the biomass contours were constrained closer to shore by the wind-driven flow.

Given this model as an adequate representation of major dynamics in Lake Ontario, we performed numerical experiments to find out which of the physical mechanisms was most important. In these experiments we ran two simulations and compared the results to the original calculations discussed above (henceforth, the normal case). In the first simulation, we eliminated mass transport by advection and diffusion. In the second simulation, only vertical diffusion was included. In all cases in situ biological and chemical processes and sinking were included and temperature distributions were kept the same as in the normal case. The results of these experiments are summarized in figure 13 for available phosphorus and particulate organic carbon.

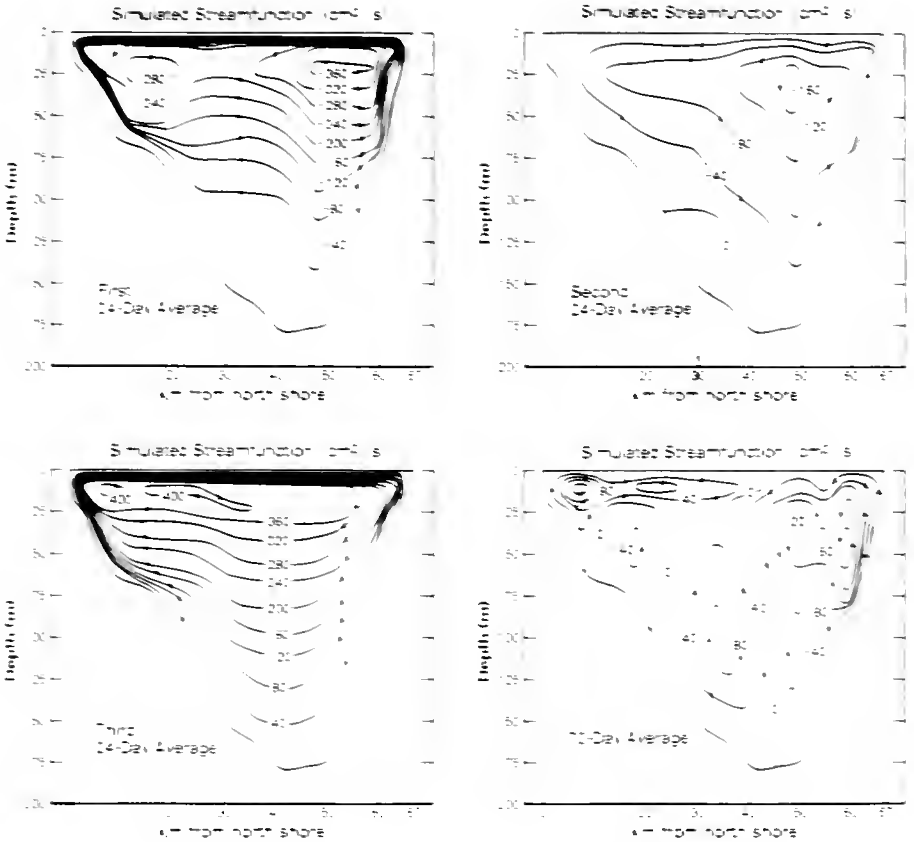

Figure 11. Contours of calculated stream functions (cm^2 s) averaged over three 24-d periods and over the entire period of simulation. From Scavia and Bennett (1980).

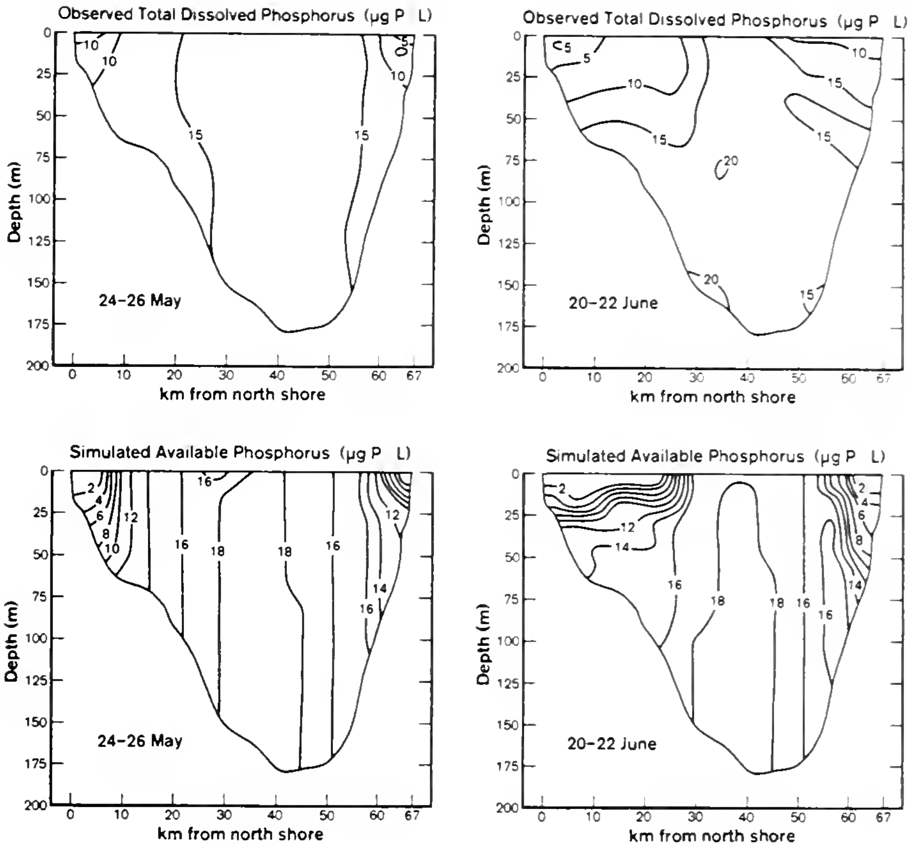

Figure 11. Comparison of observed (upper) and calculated (lower) concentrations of phosphorus (µg P/L) corresponding to May and June cruises. Observed phosphorus is total dissolved phosphorus and calculated phosphorus is that considered available for phytoplankton growth. From Scavia and Bennett (1980).

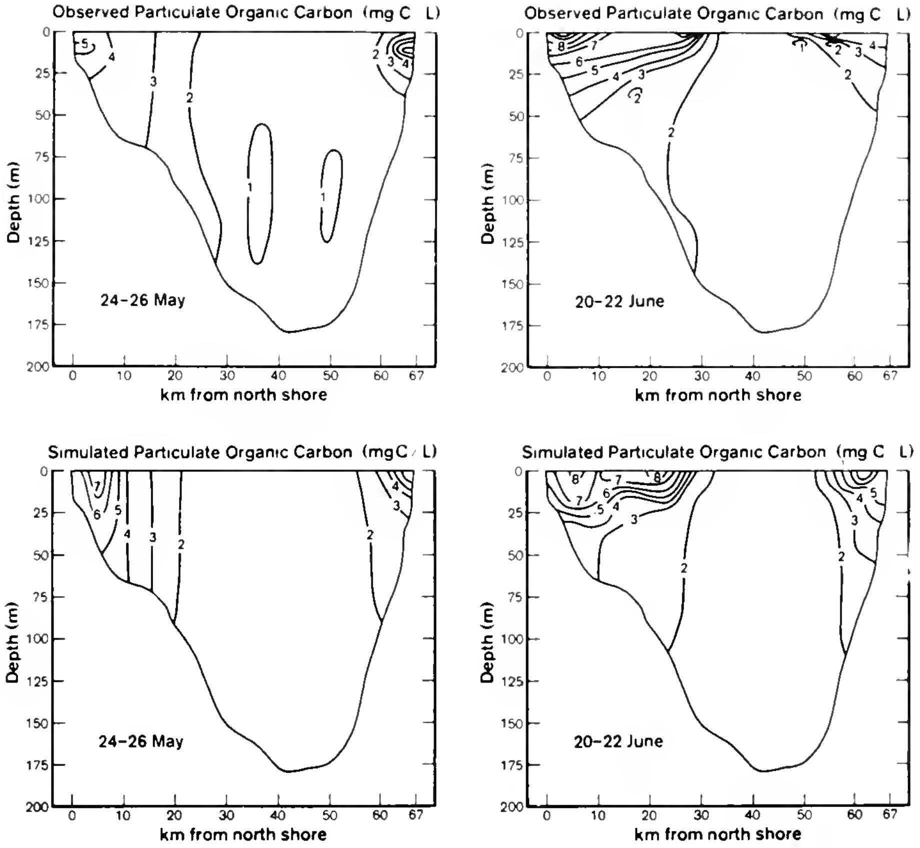

Figure 12. Comparison of observed (upper) and calculated (lower) concentrations of particulate organic carbon (mg C/L) corresponding to May and June cruises. Calculated carbon is the sum of simulated phytoplankton, zooplankton, and detrital carbon. From Scavia and Bennett (1980).

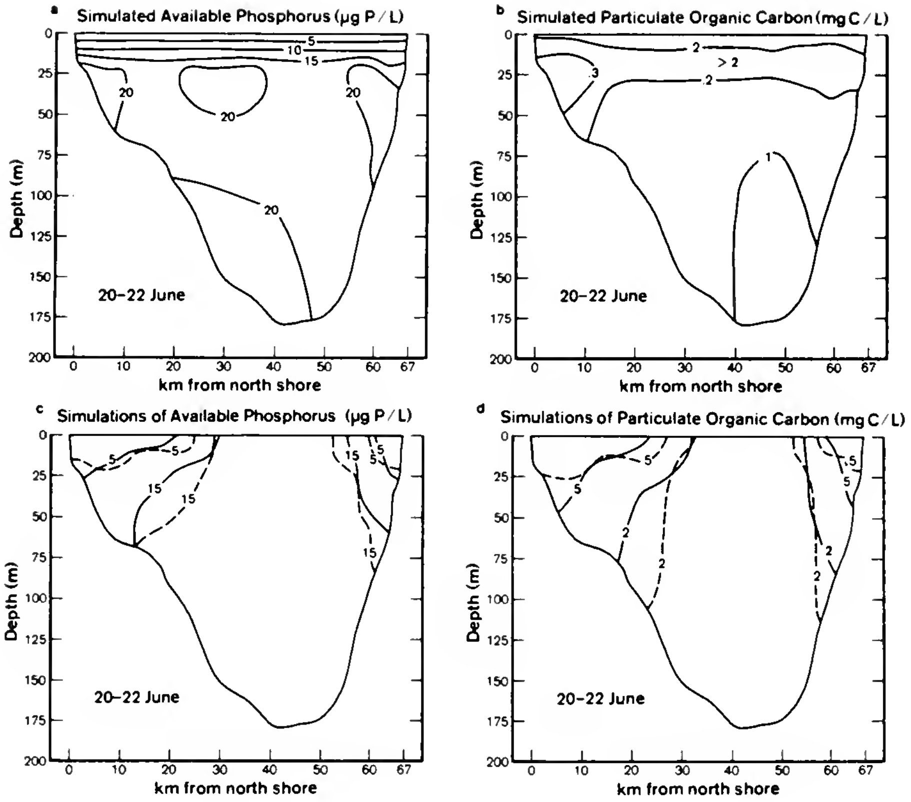

Figure 13. Contours of model output demonstrating the relative effects of physical and biological processes on the distributions of available phosphorus (µg P/L) and particulate organic carbon (mg C/L). a,b - contours of model output generated with no advection or diffusion included; c,d - broken lines represent normal case (i.e. all processes included); solid lines represent simulation without vertical or horizontal advection. From Scavia and Bennett (1980).

The simulation with no advection or diffusion showed that phytoplankton production in the cold offshore waters, although reduced when compared to inshore waters, is sufficient to deplete nutrients from the entire surface layer within the 72-day simulation (fig. 13a). Therefore, temperature-controlled, in situ production alone was not sufficient to produce the persistent offshore gradients.

The second simulation illustrates that vertical mixing, together with in situ production, is sufficient to reproduce the observed biological and chemical patterns. The major effect of vertical and horizontal advection is to smooth the nutrient gradients through increased mixing caused by repeated reversals in flow direction. The same is true for plankton along the relatively dilute, cold-water boundary (fig. 13d).

It appears that the distribution of chemical and biological properties in the vicinity of the 4°C isotherm is controlled primarily by the interaction of in situ processes and the differences in vertical mixing on either side of the isotherm. Shoreward, the water mass is weakly stratified vertically. This reduces the mixed depth and allows increased biomass production and subsequent nutrient depletion (Sverdrup 1953, Stadelmann et al. 1974). Lakeward, deep vertical mixing keeps a significant portion of the phytoplankton removed from the sunlit surface layers and therefore inhibits their growth.

The region near the 4°C isotherm has been referred to as the "thermal bar." Many notions regarding this region have suggested, as the name implies, that elevated biomass concentrations shoreward of the isotherm are caused by some barrier to their transport offshore. The above analysis demonstrates that in fact no such barrier exists. Biomass, concentrated in surface waters shoreward, are simply diluted vertically throughout the water column when transported offshore. In this fashion, the presence of this convergence zone near the 4°C isotherm in fact enhances offshore transport.

LIMITS OF ECOLOGICAL MODELS

Compensating errors - Ecosystem models that are faithful to extant theories relating various processes of nature tend to become complicated, nonlinear collections of equations. Often, verification of these more mechanistic models is not possible by usual techniques because it is difficult to obtain complete and independent data sets. This is because sampling all of the properties simulated in more mechanistic models is difficult and expensive (e.g., zooplankton biomass). Even when such data sets are available and these models have been "verified" by usual techniques, one is left with serious questions concerning model reliability because these generally nonlinear models have increased degrees of freedom.

Increased degrees of freedom, in this context, means that more than one set of coefficient values will satisfy the usual tests for calibration and verification. The basis for increased degrees of freedom is the cyclic nature of mechanistic models. Since these models generally simulate ecosystem cycles, one would not expect material to accumulate excessively in one particular component but rather to flow among all of the components. Then, because of principles of mass conservation, one could expect that, if flow rates were increased or decreased proportionately, state variable concentrations would not be affected significantly (at least not within the variability usually inherent in field verification data). The following example demonstrates one way in which compensating errors at the process level can lead to erroneous conclusions regarding system controls.

After initial calibration to measurements of state variables for the model described above (fig. 1), simulated process rates were compared to measurements. For this comparison, a summer-averaged (July-Sept.) phosphorus flow diagram was constructed as described above from aggregated model output. Flow (or transfer) rates were then compared to measurements and calculations from Lake Ontario and to other, more theoretical estimates. Many of the simulated process rates (fig. 14a) were very low (as much as 3-7 times lower) compared to measured values. Most serious discrepancies in transfers were among available phosphorus, phytoplankton, and zooplankton. I recalibrated the model keeping process rates in mind and most coefficient values still within acceptable ranges. The new calibration, shown in figure 14b, is the same as that discussed above (fig. 8). Here, state variables are close to the originally calibrated values and can still be considered calibrated; however, process rates are much higher and, in fact, much closer to observed values (Scavia 1979).

This example demonstrates that if the model were calibrated only against state variables and then used to examine control of phosphorus cycling, then the relative importance of certain processes would be overestimated by almost an order of magnitude. For example, regeneration of available phosphorus from detritus P is relatively more important in figure 14a than in figure 14b and the relative importance of external loads and of transport into and out of the epilimnion is exaggerated in figure 14b.

Because of increased degrees of freedom and the usual lack of long-term verification data, mechanistic models need verification tests beyond the standard tests used for state variable simulation. Two general types of verification can be useful additions to the usual tests (Scavia 1980b): 1) comparison of aggregated output from the mechanistic model with output from simpler models and empirical correlations that have been verified or proven to be generally applicable and 2) a comparison of simulated process rates with rates measured in the field or in the laboratory to determine if the model's internal dynamics are consistent with measured and theoretical dynamics.

Effects of uncertain inputs: Numerical models have become relatively common tools in lake management. In many cases, they have also become useful for suggesting research needs, synthesizing extant information, and analyzing aquatic ecosystems in ways that are not tractable through field and laboratory studies alone. Models used most often in both contexts have similar attributes; they are generally time-dependent, often nonlinear, ordinary differential equation models based on parameterized physiological processes and mass conservation.

These models, whether from the management or the research milieu, have another common thread: they are generally deterministic. That is, although it is often recognized that model initial conditions, parameters, and forcing functions have stochastic components, they are seldom accounted for. Moving beyond acknowledgment of variances of these elements to assessment of their effect is important because these stochastic properties affect the confidence

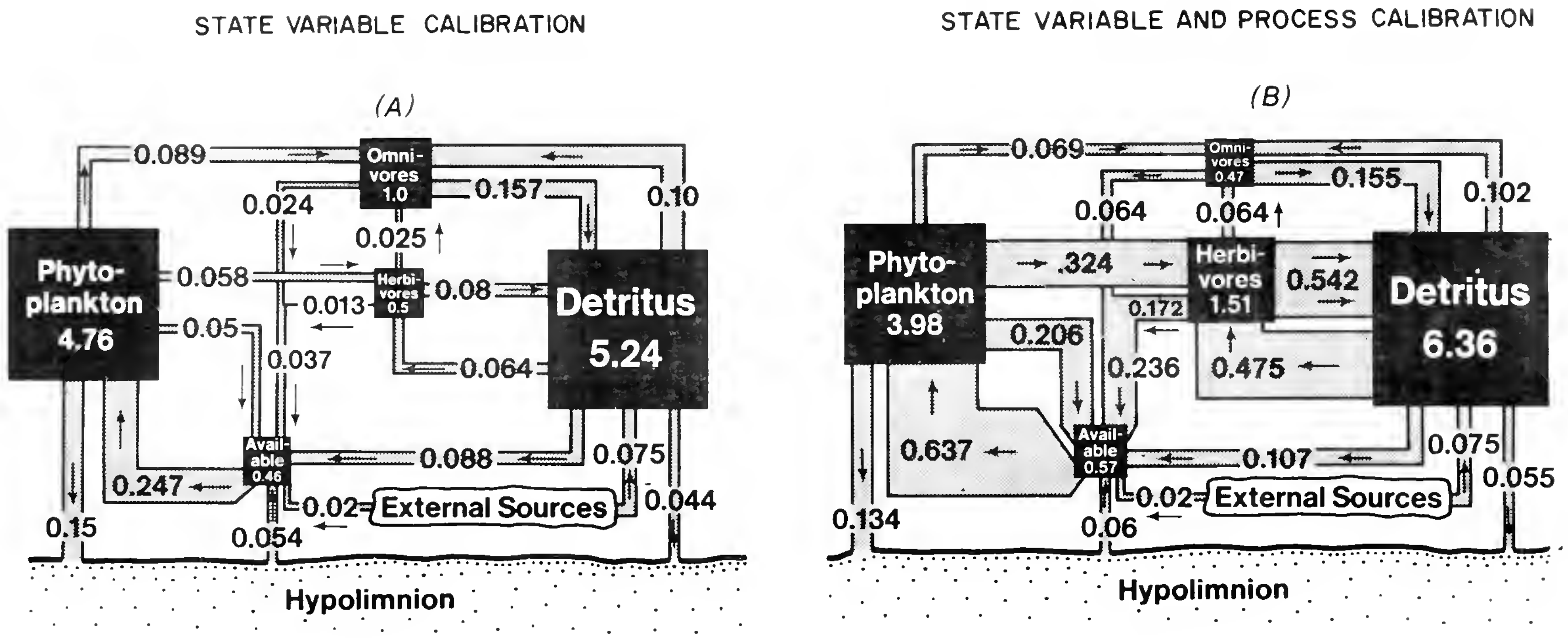

Figure 14. Phosphorus Flow Diagram. From Scavia (1980b).

that can be placed in the model output; that is, confidence is generally inversely related to variance.

Analysis of this variability is important in a management context to establish error bounds on predictions. Output from these deterministic models often influences decisions affecting many thousands of people socially and economically (e.g., Vallentyne and Thomas 1978); yet quantitative confidence limits are lacking for these models (Thomann and Barnwell 1980). In particular, only qualitative evaluations of calibration and verification results have been carried out to date, and experience with even these tests is limited. Because eutrophication models are crude representations of highly variable, stochastic systems, ignoring such important attributes often results in naive confidence or unwarranted disbelief in the models' solutions. For these models to become more generally accepted and effectively used, they must be placed in their proper perspective. Evaluating the effects of input (forcing function and parameter) variance on model output provides some of the needed perspective.

Analysis of model variability is also important in research contexts where a model's ability to simulate must be evaluated prior to investigation of specific system properties and recognition of actual system variability is important. Output from these models is often used to assess the relative importance of various system compartments or processes and thus to focus additional effort on key problems. Prior to using a model in this context, it is important to evaluate its ability to function as a synthesizer or interpolator. Traditionally, this evaluation is done by comparing model and measurement trajectories, with no quantitative assessment of model or measurement variability. As discussed above and in Scavia (1980b) comparison of modeled and measured state variables alone is not sufficient for this purpose. Calculation of variance associated with state variables and of correlations among state variables and parameters will assist in evaluation of these models for use in research contexts.

We (Scavia et al. 1981a, b) have used Monte Carlo and first-order variance propagation analyses to explore impacts of uncertain parameters, loads, and initial conditions on a relatively simple model of plankton seasonal dynamics in Saginaw Bay, Lake Huron. In these analyses, we use estimates of natural variability of the input properties as sources of uncertainty. For Saginaw Bay, natural variability far outweighs uncertainty introduced by inaccurate measurements.

Treating input errors in that way does not strictly estimate error associated with the ability of the model to predict. To do this, one certainly must examine errors introduced by the equations themselves and perform the analysis over the time frame of the prediction, as has been done for some empirical and simpler lake models (e.g., Reckhow 1979). However, because variance due to measurement errors is small compared to natural variability in this system, these variance estimates measure at least their contribution to prediction variances.

Using the input statistics and first-order variance propagation, state-variable variance estimates were made for an annual simulation based on variances of the initial conditions and given parameters. Resulting variances are represented as model output plus or minus its standard deviation in figure 15.

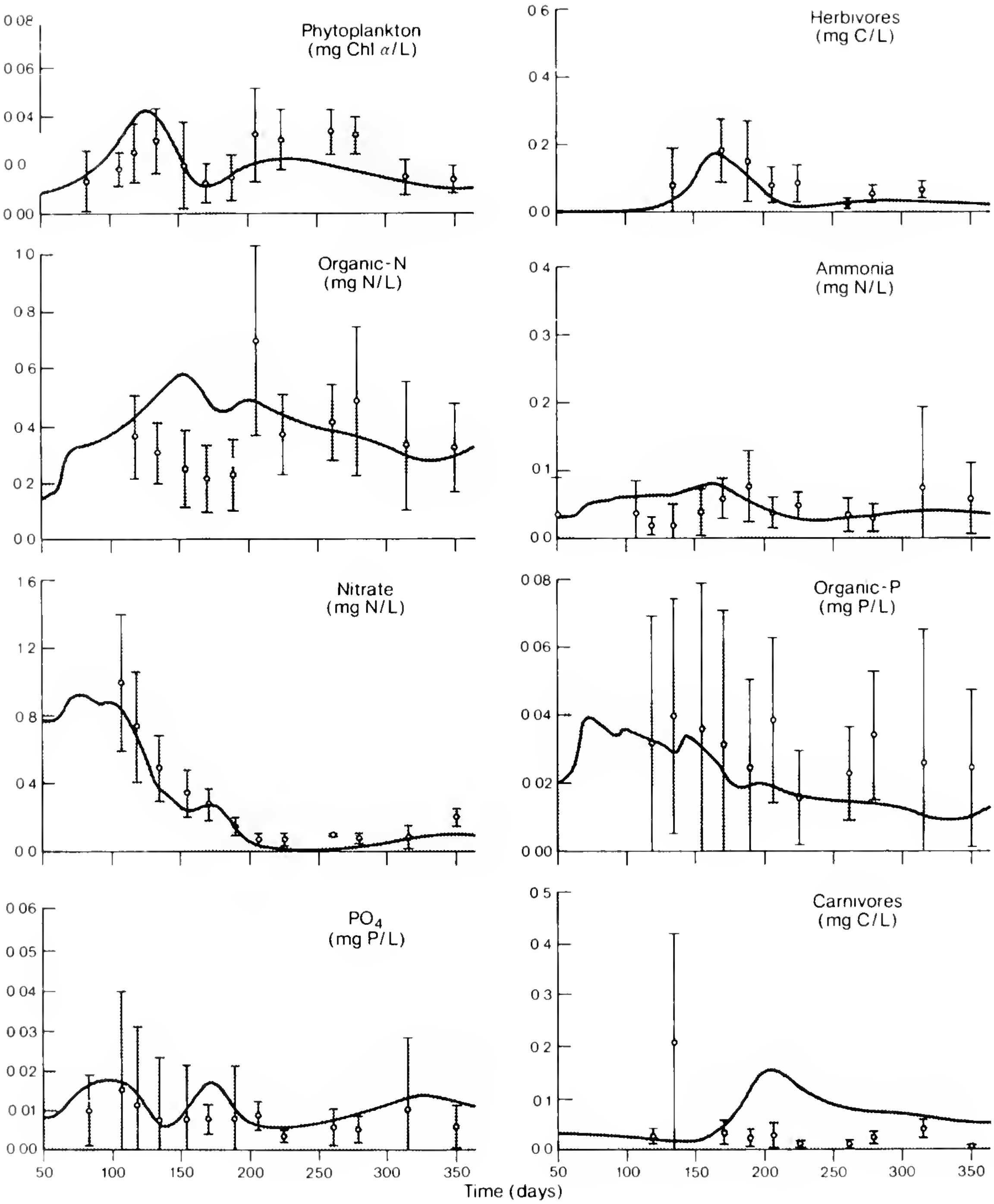

Figure 15. Plots of eight state variable trajectories (smooth curve) from Saginaw Bay model. Model error estimates are represented as ±1 standard deviation bands (shaded) from first-order analysis. Data (circles with error bars) are represented as baywide mean plus or minus standard deviation of all samples. From Scavia et. al. (1981a).

Peaks in variance estimates occurred at times when state variables were changing fastest. Maximum coefficients of variation (CV=standard deviation divided by state variable value) ranged between 148 and 722 percent, however, the average CV during the summer ranged between 33 and 407 percent. While these values are large, they are in many cases comparable to natural variability within the bay (table 1, fig. 15). However, because we have included only a subset of potentially important error sources and because we expect longer term prediction errors to be larger than those estimated here, it is of interest to determine the most significant sources of variability in this model. From the standpoint of model variance the relative effects can be demonstrated easily. In the simulations discussed below, initial condition, parameter, load, and mixing parameter variances were each used singly or in simple combinations.

Assuming perfect knowledge of initial conditions (i.e., initial condition errors=0) reduced maximum output variances only slightly. Conversely, assuming uncertain initial conditions and perfect knowledge of parameter values resulted in much lower errors. Thus, parameter variance contributes far more than initial-conditions variance. (See first 3 lines of table 2.) Variance associated with loadings contributed little, even when compared to the low initial-condition contribution (line 4, table 2). None of the CV increased more than 20 percent when loading variances were included. In fact, only ammonia-nitrogen (NH_3-N) and nitrate-nitrogen (NO_3-N) CV increased more than a few percent. Including uncertainty (CV=10%) in a mixing parameter describing transport between bay and lake also had little effect (line 5, table 2). In fact, even when its assumed variance was doubled, no state-variable maximum CV increased more than 1 percent. These results are consistent with more detailed analyses performed on an ecologically simpler, two-segment model (Scavia 1980c).

Input loading variance estimates represent time variability only. It is well known that estimating loads from highly variable, episodic inputs is difficult. To examine the potential influence of these inadequacies, two more cases were run, two and ten times the load variance, respectively. These runs assume that loading standard deviations, other than due to temporal variability, are equal to that due to temporal variability and equal to three times that variability, respectively. Dilution effects of the inner bay (volume 10^{10} m^3) somewhat mitigated even this variability when compared to variance propagated from initial-condition and parameter sources (lines 6 and 7, table 2). The largest effects were seen in the CV for NH_3-N and NO_3-N; increasing load variances by a factor of ten, an extreme case, resulted in doubling their model-output standard deviations.

These tests of relative effects of different variance sources on propagated variances for a 1-year simulation indicated that parameters were by far the most significant contributors. The effects of initial-condition variance were quickly surpassed by the effects of parameter variance during the simulation, and only when very large loading measurement errors are assumed do load variances contribute significantly. We did not examine results of errors propagated over longer than the 1-year time frame. If we were examining long-term prediction errors, the effects of uncertain load predictions (not measurements) would have to be considered. This would certainly increase the variance contribution of loading estimates.

Because the parameter errors had the largest impact on model output errors, we (Scavia et al. 1981a) made use of the propagated covariance matrix to identify

TABLE 1. Maximum and Mean of Summer Coefficients of Variation (Percent) Calculated by the First-Order Analysis From Uncertain Initial Conditions and Parameters Compared to Coefficients of Variation From Measured Variables.

Variable	Maximum	Summer Mean*	Observations†
Phytoplankton	593	78	52
Herbivores	772	206	65
Organic N	148	33	48
Ammonia	201	155	92
Nitrate-nitrite	550	407	40
Organic P	163	48	96
PO_4	552	186	115
Carnivores	707	266	67

* Summer: July-September

† Calculated coefficient of variation of spatially averaged values from all sampling dates

TABLE 2. Estimated Maximum Coefficients of Variation (Percent) with Different Variance Sources.

Variance Sources	Phyto-plankton	Herbi-vores	Organic N	NH_3	NO_3	Organic P	PO_4	Carni-vores
Initial condition only	38	81	24	46	123	22	58	58
Parameters only	592	768	148	196	536	163	548	704
Parameters and initial conditions	593	772	148	201	550	163	552	707
Initial conditions and loads	38	81	25	55	148	26	60	59
Initial conditions, loads, and mixing parameter	38	81	25	55	151	26	61	60
Initial conditions, loads (×2), and mixing parameters	39	82	26	63	172	28	63	61
Initial conditions, loads (×10), and mixing parameter	41	84	33	106	291	45	81	69

which parameters were most important in terms of both model sensitivity and model errors. Identification of those parameters and associated processes suggest areas requiring further research.

One final aspect of model uncertainty becomes apparent when viewing the distributional properties of model output generated from Monte Carlo simulations (Scavia et al. 1981b). In this analysis the following procedure was used. The model equations were solved repeatedly. Each model execution was performed with initial conditions and parameter values selected randomly from their individual distributions. From these repeated model executions, state variable means, variances, and other statistics were calculated at 4-week intervals throughout the period of simulation. The analysis was terminated after 1000 simulations, at which time state variable means and variances were converging.

Histograms generated from the 1000 simulations for selected state variables at different points in time during the annual cycle are shown in figure 16. It is interesting to note that even though the initial conditions and parameter values are drawn from relatively smooth, symmetric distributions, the resulting model output distributions can be dramatically asymmetric and polymodal.

Several implications are suggested by these distributions. Those that are spread out suggest that variability in control parameters (initial conditions and coefficients) has a dramatic effect. That is, a fairly uniform output distribution suggests many possible model outcomes are equally likely as control parameters vary within their confidence ranges. Distributions that are dramatically narrow indicate relative insensitivity of that state variable to the uncertainty in control parameters. Bimodal or polymodal distributions suggest that even though control parameter values have probabilities of occurring that vary smoothly through their distributions, the model produces state-variable values that jump from one category of high probability to another with very few outcomes occurring in between. Controls of such threshold behavior both in models and in nature are not well understood, but careful attention must be paid to the possibility of it occurring.

SUMMARY AND CONCLUSIONS

I have outlined how a particular ecosystem model has been used to better understand the structure and dynamics of Lake Ontario. This is but one example of how integrated modeling and experimental science has advanced our ability to understand and perhaps simulate and predict dynamics of the Great Lakes. This and similar models generally represent collections of process relationships developed through independent empirical studies and as such they merely test those relationships in the context of the whole system. That is, the models test our ability to simulate algal dynamics, for example, by balancing rates of gain and loss calculated from expressions developed independent of the whole system. While certain relationships among processes may be testable, it usually becomes intractable to study such relationships in nature. This is particularly true in systems like the Great Lakes and marine waters where physical processes can be important. In those cases, a model that is firmly based on independent, empirically tested constructs and exercised within the framework of carefully designed field observations may be the only means to improve our understanding of, and thus our ability to predict, responses of the aquatic ecosystem to altering stresses.

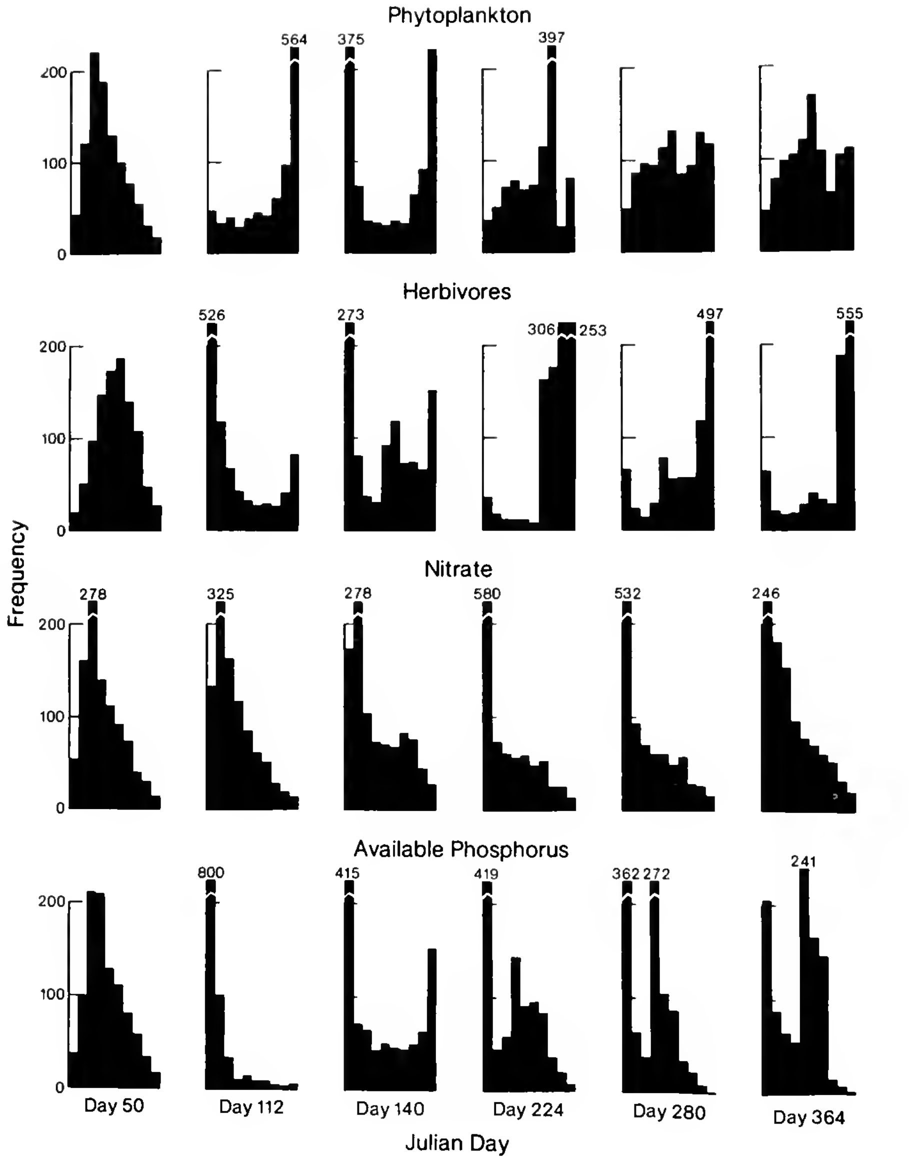

Figure 16. Frequency distributions of Monte Carlo output for four state variables at selected time slices during annual simulation. Day 50 distributions are initial distributions. 1000 cases were used. From Scavia et al. (1981b).

Because these models are based generally on mechanistic relationships rather than on strict statistical criteria, they can certainly produce reasonable simulations using sets of different coefficient values. While each set of values may result in indistinguishable state variable predictions, they will not likely all give accurate representation of the processes that contribute to those state variable trajectories. Errors in those process rate simulations may lead to erroneous conclusions concerning critical controls of ecosystem dynamics. It is thus important to measure simultaneously both state variables and process in situ when developing or testing these models. Although many field estimates of processes are difficult to obtain, measurement of even some will provide checkpoints of the simulation of internal dynamics. It is less likely that compensating errors in process simulations will remain if the model adequately reproduces both measured states variables and certain processes.

Finally, it must be remembered that nature is not predictable in a strictly deterministic fashion given the state of our understanding of its laws. Many events occur on time and space scales smaller than we can attempt to model. These events will add stochastic properties to our measurements. Recognition of these processes and their effects on model prediction confidence must be addressed in any model of nature.

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OIL SPILL MODELING FOR OCS ENVIRONMENTAL ASSESSMENT AND DECISIONMAKING

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ABSTRACT

The Outer Continental Shelf (OCS) Lands Act, as amended, requires the Secretary of the Interior to make publicly owned oil and gas resources available to help meet our Nation's energy needs. Within the Department of the Interior, the Bureau of Land Management (BLM) carries out the leasing process; the Minerals Management Service supervises leases once they have been sold (until early 1982 this function was performed by USGS). Since OCS leasing involves actions which may impact the environment, the leasing process is subject to the National Environmental Policy Act (NEPA). NEPA in turn requires that BLM produce environmental statements (ES) for each scheduled lease sale. A significant portion of each ES deals with accidental spills related to production and transportation of offshore oil.

Oil spill risk analysis modeling is performed in DOI jointly by BLM and USGS. Input data are provided by BLM; the modeling work itself is performed and formally reported by USGS; and the model results are used by BLM. The modeling work is undertaken from a cumulative perspective and includes spill simulations from existing transportation routes and, where applicable, from existing Federal leases.

Model results are analyzed for use by decisionmakers on matters relating to lease tract deletion and transportation alternatives and lease stipulations.

The modeling work is predictive (for the next two to three decades) and couched in probabilistic terms, to account for the numerous uncertainties existing at the prelease sale stage. Thus, the model is unlike deterministic or real time models.

INTRODUCTION

1. Background

The Outer Continental Shelf Lands Act of 1953 charges the Department of the Interior (DOI) with carrying out a national offshore oil and gas leasing program. Within the DOI, the Bureau of Land Management (BLM) is responsible for leasing offshore resources and the Minerals Management Service is responsible for supervising offshore leases. (Lease supervision and enforcement activities were transferred from the U.S. Geological Survey (USGS) to the Minerals Management Service in early 1982.)

Since its first offshore lease sale in 1954, BLM has sold more than 4,000 leases for a total in bonus bids of $36 billion; lease rental and production royalties have exceeded $10 billion (BLM 1981). Through 1980 a total of 5.4 billion barrels of crude oil has been produced (USGS 1980).

The Outer Continental Shelf (OCS) program is carried out in compliance with numerous laws. A compilation of laws governing resources and resource management on the OCS is presented in DOI (1981). The National Environmental Policy Act of 1969 and the OCS Lands Acts Amendments of 1978 provide for the environmental modeling work reported on in this paper.

The Federal OCS includes all areas seaward of a line drawn 3 geographical mi from the shoreline. Off Texas and western Florida the line is drawn 3 marine leagues from the shoreline. Legislation defining the OCS is contained in the Submerged Lands Act of 1953 and the Outer Continental Shelf Lands Act of 1953. Seaward limits to America's OCS are either not defined or are in negotiation with adjacent nations. OCS lease sales have been held within all regions of the OCS except for portions of the Bering and Chukchi Seas and areas with water depths greater than 2 to 3 km.

Oil spills are a major environmental issue identified by parties associated with OCS leasing (Federal and State governments, industry, conservation groups, and the general public). Expressions of concern and requests for information for use in analysis have led to the development of the DOI oil spill risk analysis model. Model outputs address three topics: the likelihood of spill occurrence, likely pathways that spills might follow, and the risks that spills will occur and will contact various resources. The model treats uncertainties inherent with OCS related spills as well as spills from other sources. The model serves as a quantitative framework for synthesizing enormous amounts of environmental information. Model outputs are intended for use by environmental analysts and program decisionmakers.

2. Scope

Oil spill risk analysis modeling is carried out as a joint effort of BLM and USGS. Because of this and the wide collection of users of the modeling results, the work may be considered as a DOI project. Input data to the model are provided by BLM. These data includes all environmental information as well as definition of leases to be offered and transportation senerios to be used if oil is found. USGS provides estimates of oil and gas resources within the lease sale area and carries out all of the model computations. Reports of model output are provided to BLM for use and analysis in writing environmental statements and drafting DOI decision documents. Decision documents are finalized within the Department.

Input data reflect the best available information for each of the OCS lease sale areas. The existing information base is supplemented through studies sponsored by BLM's Offshore Environmental Studies Program. The Studies Program began in the mid-seventies and continues to date. Approximately half of the BLM sponsored studies have been conducted off Alaska. Data on ocean circulation, local winds, and locations of various marine resources comprise the bulk of model inputs.

The spatial extent of each model accounts for spills originating from production sites as well as along transportation routes. Spill trajectories are analyzed for up to 30 days. For example, a recent model for the area off the southeastern coast extended from Miami to Norfolk and from the shore to well over the Blake Plateau.

The oil spill risk model is predictive in that it treats future events. The future, extending 2 to 3 decades, is defined by the estimated time to complete production from an offshore lease. Because the model deals with uncertain events and environmental circumstances, it is couched in probabilistic terms. This treatment of uncertainty distinguishes the DOI model from deterministic oil spill models, such as those developed and run by the National Oceanic and Atmospheric Administration and the U.S. Coast Guard for response to real time spills (Wallops Workshop 1980).

OIL SPILL RISK MODEL

1. Overview

The DOI oil spill risk model is one of the largest environmental models in current use; larger models are used by the National Weather Service. The overall workings of the model are discussed by Smith et al. (1982). Detailed documentation of the model is presented by Lanfear and Nakassis (1980) and Lanfear and Samuels (1981). A brief overview of the model with examples of lease sale applications is presented by Lanfear et al. (1970). Model runs for each OCS lease sale, beginning in 1976, are described and results presented in the USGS Open File Report series; recent examples, for Southern California and the Gulf of Mexico, are presented by Samuels et al. (1981a) and LaBelle and Samuels (1981), respectively.

For a specific application the region to be modeled is defined based upon the locations of potential spill sites (production and transportation), locations of potentially vulnerable resources, and knowledge of winds and currents which characterize the region. Typically the modeled region includes 600 to 800 nmi of coastline and extends seaward about 400 nmi.

The model is structured on a grid (480 x 480). All spatial information are digitized and portrayed on the model grid. Computer programs are incorporated to allow input data to be on virtually any map projection and map scale. Digitized inputs include the shoreline, ocean currents, and locations of biological resources. Hypothetical oil spills are launched from potential spill sites and advected within the model grid by wind and current. Spill contacts with various "targets" are recorded along with the time between spill launch and contact. Spills are launched throughout the year and in sufficient numbers to establish statistical significance to the contact probabilities.

Contact probabilities derived in this fashion are conditional in that spill occurrence is assumed. The conditional probabilities are then combined with the likelihood of spill occurrence to yield the final (joint) probabilities that spills will occur and will contact specific targets.

2. Salient Points

The oil spill risk model is well documented in the published literature. In addition, as noted above, there are published reports describing each sale specific model run. Thus, I attempt here to summarize selected topics. The topics were chosen to reflect the capabilities of the model. The topics discussed below are intended to also represent the collection of issues most frequently raised by the public who interact with the OCS Program.

The model deals with oil only on the ocean surface. Spills, represented by points (hypothetical center of mass of a surface slick), are advected by surface currents and winds. The surface currents are climatological (generally average monthly) and do not contain an accounting of local wind effects (geostrophic flow). Local wind-induced drift is computed to be 3 1/2 percent of the local wind speed with a 20° clockwise rotation from the wind direction. Following the work of Samuels et al. (1981b), recent model runs incorporate a variable deflection angle which is an exponential function of wind speed. The oil advection algorithm is taken as the vector sum of the surface current vector and local wind-induced drift. Seasonal portrayals of surface currents vary spatially as dictated by available data; examples are contained in Samuels et al. (1982a). Consequences of using various hypothesized versions of surface currents have been examined (Lanfear and Amstutz 1981).

Local winds are sampled at 3-h intervals, in Monte Carlo fashion, from seasonal wind transition matrices. The matrices are constructed from time series observations, generally measured at coastal stations. The 41 x 41 matrices represent eight directions with five speed classes each, and the calm condition. Winds are thus treated as a first-order Markov process. Wind zones are assigned over the modeled area to dictate where each station time series applies. Wind zone definition is determined through comparison of wind roses at sea, derived from ship observations, with those constructed from the coastal station time series.

Spill advection continues until the spill contacts land, encounters a model boundary, or remains at sea for more than 30 days. Spills are launched throughout the year (500 per season or 2,000 per year) from each launch site; thus the Monte Carlo sampling error does not exceed 2 to 3 percent. Launch sites may be: single points, to simulate platform locations; along lines, to simulate pipelines and tanker routes; or collections of uniformly distributed points, to simulate several platforms in a small area.

Biological resources (commercial fishing areas, whale migration routes, sea otter ranges, pelagic sea bird feeding grounds, etc.) are represented spatially for times (months) they are considered sensitive to oil spills. Shorelines may be subdivided by type or use (e.g., rocky shores, salt marsh, high-intensity use beaches) and by land segments. Land segments may be designed to be of equal length (typically 20 to 30 nmi) and also by arbitrary criteria such as to be coincident with political subdivisions.

Conditional probabilities are tabulated annually by launch site identification and target name, using transit intervals of 3, 10, and 30 days. These conditional probabilities (numerically determined by winds, currents, locations, and temporal sensitivities of resources and locations of spill sites) portray risks from oil spills without consideration of the likelihood of oil spill occurrence. Conditional probabilities, though useful to decisionmakers, are beyond their control.

Determination of future spill incidence is a complex task. The DOI oil spill risk model projects future spill incidence upon past experience using volume of oil as an exposure variable. Predicted probability distributions for spill incidence are calculated separately for platforms, pipelines, and tankers. Spill occurrence is calculated for size categories equal to and greater than 1,000 barrels and 10,000 barrels. The predictive procedure used in the model was initially developed by Devanney and Stewart (1974) using Bayesian techniques. Additional analyses have addressed the data bases and alternative exposure variables (Stewart 1976, Stewart and Kennedy 1978). Devanney and Stewart (1974) estimated the number of spills using the negative binomial distribution. Since the exposure variable (volume of oil) associated with an OCS sale is less than the historical exposure, the Poisson distribution serves as an excellent approximation to the negative binomial distribution (Smith et al. 1982). The Poisson distribution is defined by its single parameter, the expected number of spills.

The oil spill data bases used in the DOI model were initially assembled by Devanney and Stewart (1974) and Stewart (1975, 1976). OCS production records are maintained by USGS, as are records of platform and pipeline spills. Oil spill incidences are currently under review by the Futures Group (1982) under contract with BLM.

There have been 10 OCS platform spills greater than or equal to 1,000 barrels since 1964; eight of these occurred prior to 1974. A critical analysis of these platform spills and production since 1964 has quantified a decrease in platform spill rate (Nakassis 1982). Pipeline and tanker spill rates are also under review by personnel in BLM and USGS.

Spill incidence rates used in the DOI model receive considerable criticism from numerous groups. Generally the criticisms suggest that the spill rates are either too high or too low; that the rates are not applicable to some OCS regions (recall that all of America's OCS production to date has been from the western and central Gulf of Mexico and southern California); and that there must be better exposure variables. Those of us responsible for the oil spill model have adopted the following policy:

1) the past OCS experience is the best available quantitative basis from which to predict future events;

2) the number of spill incidences is of record and will be made available to those who desire testing their hypotheses regarding spill incidence;

3) if alternative exposure variables can be demonstrated to be more precise and of more utility to decisionmakers they will be incorporated;

4) it is reasonable to use a length of past record comparable with the length of forecast (this, incidently, is necessary if we are to examine the record for changing rates);

5) as for regional applicability, the spill rates used in the model have been examined against the records of experience in Cook Inlet (production from State leases) and in Prohdue Bay (onshore production) - the experiences in these two locations are not different, in a statistically significant sense, than our experiences in the Gulf of Mexico; and

6) lastly, we are able to define the circumstances (required production without spills) needed to achieve a difference from our past experience.

The spill incidence record on America's OCS is in fact remarkedly low. Although we cannot quantify specific reasons, the consequences are a tribute to all from our society who participate in offshore oil and gas activities.

The final stage of oil spll risk analysis combines the conditional probabilities and spill likelihood to yield final (joint) probabilities--the probabilities that spills will occur and will contact various resources.

The calculation of joint probabilities involves use of hydrocarbon resources expected to be produced and transported. The volumes used in the model are provided by USGS and are the mean economically recoverable resources.

Joint probabilities are calculated as follows:

(a) Form matrix [c], where elements $c_{i,j}$ are the conditional probabilities that a spill from site j will contact target i.

(b) Form matrix [s], where elements $s_{j,k}$ are the expected number of spills from site j due to unit volume produced at site k.

(c) Form matrix $[u] \underline{=} [c] \times [s]$, where elements $u_{i,j}$ are the expected number of spills occurring and contacting target i due to production of a unit volume of oil at site k.

(d) Form vector [v], where element v(k) represents the volume of oil expected to be produced at site k.

(e) Form vector $[L] \underline{=} [u] \times [v]$, where element l(i) is the expected number of contacts to target i.

(f) Using the Poisson distribution discussed earlier and the vector [L], the probability of exactly n contacts to target i may be calculated by: $P(n,i) = [l^n(i) \exp(-l(i))]/n!$

The joint probability that one or more spills will occur and will contact target i is obtained by summing $P(n,i)$ over $n > 0$. More detailed discussion of the above procedure is contained in recent lease sale specific oil spill reports published by USGS, such as Samuels et al. (1982b). The steps outlined above illustrate the combining of OCS production and transportation spills. This is an important factor--one cannot decide to produce at a specific site without concomitant consideration of oil transport.

USE OF MODEL RESULTS

1. Applications

Model results are presented and analyzed in Environmental Statements (ES) prepared in BLM's OCS Field Offices. The content and analyses of an ES are defined by the National Environmental Policy Act and its implementing regulations. In compliance with these, the model addresses each proposed lease sale from three perspectives. First, a portrayal of the proposed sale by itself, as though there were no other activities taking place in the sale area. Second, a portrayal of the sale area environment as it would be without the proposed sale. Third, the cumulative circumstances, a portrayal of the sale area environment as it would be if the lease sale were held and all other activities continued. Sources of accidental spills considered in the model framework for the cumulative circumstance include: production and transportation of petroleum hydrocarbons into and within the area.

For each of these portrayals the analysts must consider the proposal in its entirety as well as various alternative sale configurations. Oil volumes, used as exposure variables in the oil spill risk model, are also used in sale specific economic analyses thereby affording a common basis for cost/benefit analysis.

Analytical attempts have been made to maximize oil production while minimizing oil spill risk (Smith et al. 1979). Although this can be readily accomplished, numerical solution would presumably require assignment of weights to the targets. Without weights the targets would be treated with equivalence, which implies that spill contacts yield equivalent consequences. Weights can be assigned in terms of acceptable numbers of contacts. It seems reasonable to assume that decisionmakers make such judgments during their personal deliberations.

At the final stages of a proposed OCS sale process, the model results are incorporated into decision documents. Decision documents are used by agencies within DOI to make recommendations, concerning the proposed sale and its alternatives, to the Secretary.

2. Complications

The oil volumes used in the model analysis are mean estimates of economically recoverable amounts. In analyzing sale alternatives (tract deletions for example), one often encounters circumstances where volume dependencies (for economic and geologic reasons) exist. The analysis thus goes well beyond simply deleting an element from the volume vector.

Quite often we observe that oil spills from transportation routes present greater threat to resources than spills from production sites. This is expected based upon proximity. The DOI is committed to write an ES in frontier areas which addresses the development stage of the offshore leasing process. (Frontier areas are those in which there has been no previous production; the development stage follows leasing but precedes actual production.) A development ES will be greatly enhanced by the capabilities of the oil spill risk model to evaluate alternative transportation routes.

Model outputs consist of thousands of numbers. The majority of the probabilities is negligible; nevertheless, analysis of the remaining values is a difficult and complex task. Not all of the values can be analyzed. All of the probabilities are presented in the ES, however, to enable readers to apply their own judgments and analyses.

The model deals with contacts; impacts are quite another matter. Impacts from oil spills are very much dependent upon how and where a spill occurs (underwater blowout versus blowout into the atmosphere; pipeline failure at sea versus tanker spill near shore, etc.). Impacts are also very much dependent upon the physical-chemical properties of the spilled oil. These properties are, in almost all cases, unknown at the time of a lease sale analysis. Oil properties will be far better understood at the time a development ES is written. Impact assessment requires knowledge of oil properties not only at the location of the spill but at the time and location of contact with sensitive resources. Thus, impact analysts desire quantification of oil weathering. Examination of oil weathering studies reveals that time is the single most important variable. Study of weathering algorithms reveals near linear dependencies on time. As a first approximation then, the model retains measure of the time between spill occurrence and target contact. The times for which conditional and joint probabilities are accumulated (3, 10, and 30 days) were chosen for their use as implicit measures of oil weathering--as well as for matters relating to containment and clean-up.

Spill sizes range over seven to eight orders of magnitude. In general, given such a range, the average size has little meaning. Observed OCS platform spills have the least range in sizes, with pipelines second; tanker spills account for the widest range in sizes. The OCS platform spills (1964 to present) greater than or equal to 1,000 barrels range from 1,500 to 77,000 barrels; the average is approximately 19,000 barrels.

3. The Future

Efforts have been underway for some time now to apply three-dimensional ocean circulation models on the OCS. Results have been substantial: to our understanding of coastal circulation, to our abilities to quantify stochastic elements of the circulation, and to supply circulation data beneath the sea surface.

Meteorological studies have successfully established quantitative means to construct at-sea time series winds from time series measured along the coast. Several years of direct measurement of time series winds are being acquired with anchored meteorological buoys.

Studies of the distribution of sea ice, its properties, and oil/ice interaction are yielding results applicable to oil spill risk analyses.

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OIL SPILL-FISHERY IMPACT ASSESSMENT MODELING:

APPLICATION TO GEORGES BANK

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ABSTRACT

An oil spill fishery impact assessment model composed of an oil spill fates model, a continental shelf hydrodynamics model, an ichthyoplankton transport and fate model, and a fishery population model has been applied to the Georges Bank--Gulf of Maine region to assess the probable impact of oil spills on several important commercial fisheries. The model addresses direct impacts of oil on a commercial fishery through hydrocarbon-induced egg and larval mortality. This early life stage mortality is estimated by mapping the dynamic spatial/temporal intersection of the surface and subsurface oil concentrations resulting from the spill, and juxtaposing dynamic maps of the developing eggs and larvae. Ichthyoplankton entering an area with hydrocarbon concentrations in excess of a specified threshold are assumed lost. Model output is given in terms of differential catch, comparing the non-impacted and the hydrocarbon impacted fisheries.

Simulations of the impacts of monthly oil well blowouts at a site in the North Atlantic Outer Continental Shelf (OCS) lease area have been completed for Atlantic herring and Atlantic cod. Results of these case studies clearly show the importance of spill timing and location, spatial and temporal spawning patterns, and details in the hydrodynamic transport field as critical factors in determining spill impact. Model system sensitivity studies to incremental losses of a given year class and the influence of the Georges Bank gyre on impact are also presented.

INTRODUCTION

One of the major concerns associated with Outer Continental Shelf (OCS) hydrocarbon exploration and development, is the release of oil into the marine environment, and the resulting short- and long-term effects on the ecosystem. Of particular concern in productive fishing areas such as Georges Bank, is the impact of spills on the higher trophic levels, and more specifically on commercially important fish species. Realistic assessments of the impact of spills on these species are essential if the fish and mineral resources of these shelf regions are to be managed rationally.

An objective assessment methodology is clearly needed to support a rational resource management policy. The methodology selected should be able to (1) reliably quantify impacts, (2) appropriately represent the space and time scales of the pollutant events of interest, (3) take advantage of existing environmental data and therefore minimize the need for additional data, (4) be formulated within the framework of management needs, (5) address a well-defined ecological unit, (6) limit the number of empirical formulations and, (7) be transferrable to other geographical areas and species. These requirements are intended to assure that the approach to impact assessment is credible, useful, and affordable.

Scientists addressing biological impact assessment problems have developed three major model methodologies, characterized here as the statistical, the indicator species, and the ecosystem dynamics approaches.

The statistical approach is the oldest, most common, and potentially the simplest form of impact estimated. The basis of the approach is the derivation of an empirical transfer function relating some ecosystem metric to impact magnitude. Typical metric constructs include matrices, perhaps with weighting functions (Cantor 1977), species diversity indices, and correlation procedures (Pielou 1977). Field sampling and monitoring programs, coupled with a variety of statistical methods, represent the standard format for statistical environmental impact analyses. Major drawbacks of the approach arise from its sensitivity to the metric selected, and the high degree of natural variability usually reflected in the underlying data. This latter problem results in a common inability to answer the question, "When is an impact (i.e., a change in the system metric) significant?" In addition, questions of which variables to measure, over what time span, and at what sampling intervals and locations are non-trivial, but must be answered. Because of the high degree of agglomeration inherent in statistical methods, the dynamic complexities of the "real" system become obscured. Thus, little insight into the processes controlling system response is gained, and the ability to address a variety of realistic management options is usually lost.

At the other end of the complexity spectrum lies the full ecosystem modeling approach. In its purest form, this methodology incorporates all our knowledge of environmental functioning at the process level. In practical applications some agglomeration at the biomass level (Laevastu and Larkins 1981) or at the level of numbers (Anderson and Ursin 1977; Reed and Balchen 1982) is necessary to achieve completion of a project within prescribed temporal, economic and computational constraints. In the past, this agglomeration commonly led to a "box model" representation, in which species or species types and their interactions were represented through a set of highly parameterized differential equations (e.g., Chen 1975; Kelly 1975). Increasing accessibility of powerful computational facilities has made feasible more detailed, process-specific representations. Such detail is attractive because of increased "realism". The construction of such an ecosystem model is based on some form of conservation laws, including appropriate source, sink, and interaction processes, to relate ecologically critical components.

The more detailed and "realistic" these models are, the more variables and parameters they require. Such efforts are therefore sometimes characterized as "data hungry", and may require relatively arbitrary assignment of values to many parameters. Conclusions drawn from simulation outputs may therefore be subject to considerable uncertainty. The data necessary to support this modeling approach, in terms of input as well as validation, is rarely available at present.

On the positive side, an ecosystem model with appropriate spatial, temporal, and biological resolution makes better use of available data than simple statistical analyses, in that information inherent in the data is conserved in the model. Spatial species dynamics, for example, are well documented in most fisheries data, but traditional fish population models (e.g., Beverton and Holt 1954) have essentially neglected this aspect. More recent ecosystems models for fisheries management (Laevastu and Larkins 1981; Reed and Balchen 1982) have demonstrated the importance of spatial dimensions in arriving at an understanding of the governing processes.

These process-explicit ecosystem models with spatial representation are also useful for investigating the impacts of various alternative management strategies. Improved data gathering technologies (e.g., satellite remote sensing), combined with improved computer capabilities will render these complex models more credible, giving them wider acceptance, and they will probably become the preferred management tool of the future.

The fact remains that rational decisions must be made now, in the 1980s, concerning relatively complicated and important resource management alternatives. The indicator species, process-explicit modeling approach to environmental impact assessment provides a workable compromise between the statistical and full ecosystem approaches. Single species indicator models have seen extensive use in assessing the impact of power plant location and operation in riverine, estuarine, and coastal areas on commercial and sport fisheries (Lawler 1972; Hess et al. 1975; Van Winkle 1977; Spaulding and Isaji 1979). Narrowing the scope of an ecosystem model to address one important species, retains process-explicit capabilities (e.g., physical transport, migration, or feeding), increases validation potential through parameter identifiability (Gentil and Blake 1981), and therefore model credibility, while reducing time, expense, and data demands in development. In short, a single species modeling approach, augmented by a judicious choice of processes for inclusion in the model, represents a utilitarian compromise between the two other alternatives discussed above, and provides a useful methodology for estimating environmental impacts and investigating management alternatives.

The single species modeling approach has been used to estimate the impact of oil spills on the Georges Bank cod fishery (Reed 1980; Reed et al. 1981). This impact assessment methodology is currently being extended to additional species (Spaulding, Saila et al. 1981; 1982). A summary of some recent results from this work follows.

OVERVIEW OF OIL SPILL FISHERY IMPACT ASSESSMENT MODEL SYSTEM

The oil spill fishery impact assessment model system addresses first order direct impacts of oil on a commercial fishery through hydrocarbon-induced egg and larval mortality. The model system components follow the approach given by Reed (1980), Reed and Spaulding (1978, 1979), and Reed et al. 1981 for the hydrodynamic, ichthyoplankton transport and fates, and oil spill fates components, and the approach given by Lorda and Saila (1980) and Lorda et al. (1982) for the fish population dynamics component. The simulated processes within an relationships among these four models are shown in figure 1.

In operation, the ichthyoplankton transport and fates model, using a toxicity threshold assumption, output from the oil fates model on the distribution of spilled oil, and a definition of the spatial and temporal spawning patterns of the species of interest, estimates the oil-induced mortality of eggs and larvae. Surviving eggs and larvae undergo transport and mortality as usual, except that density-independent mortality is increased in proportion to the number lost due to the oil spill. An impact analysis is achieved by running the fishery model to determine the perturbations caused by the spill event on the baseline equilibrium catch.

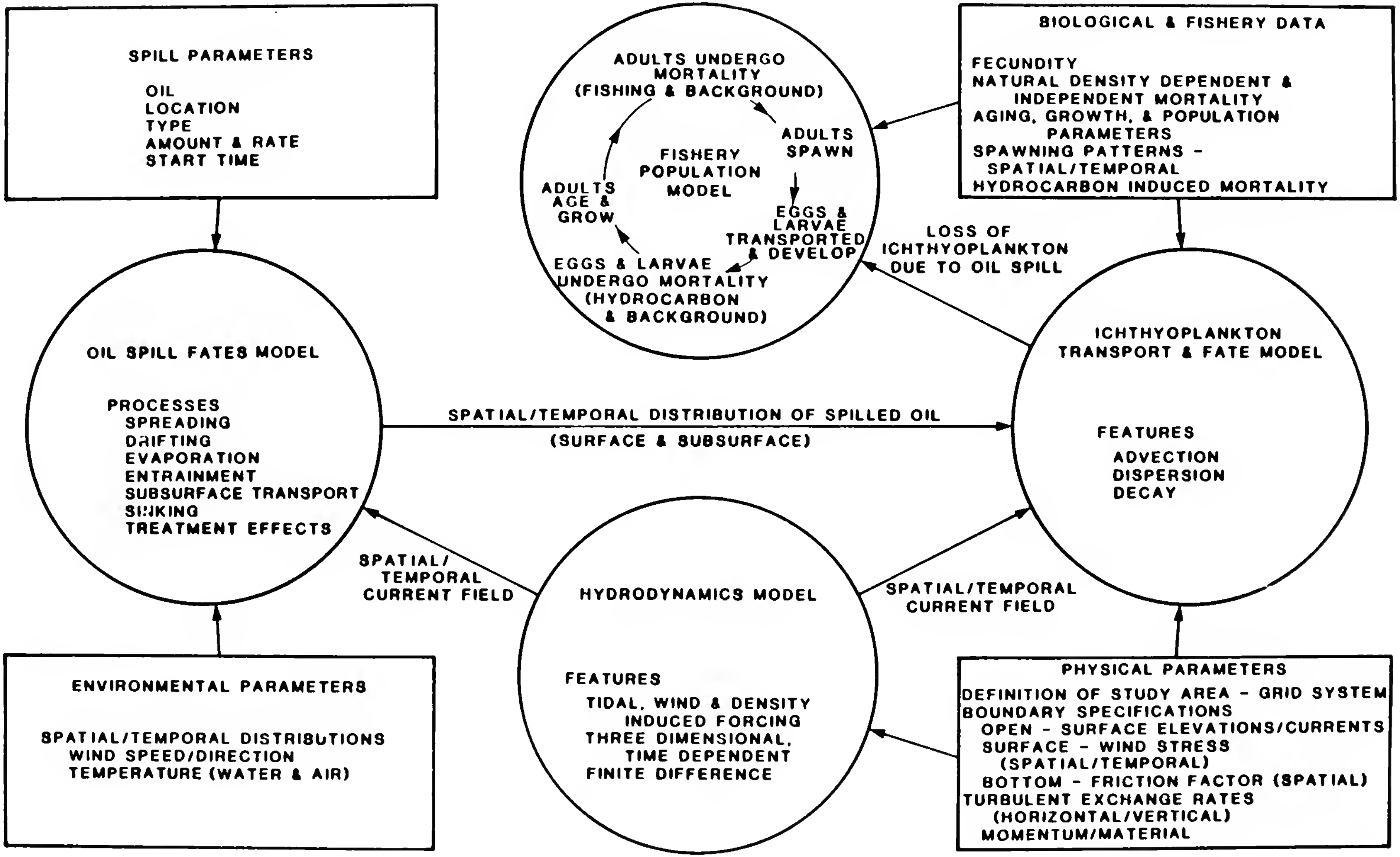


Figure 1. Model system component interaction schematic. Circles indicate models. Boxes define input data.

It should be noted that these model predictions are the relative sizes (i.e., ratios) of the oil-impacted and non-impacted catches. These ratios provide a measure of the oil spill impact expressed as percent catch deviations from some non-impacted average catch or expected "normal" catch. Given the actual size of this baseline catch, the predicted percent catch deviations can either be converted into simple economic terms by assigning a dollar value to the particular species under study, or be used in sophisticated econometric models to project the onshore impacts of an offshore oil spill. No economic component is presently included in the model system. The projected annual relative sizes of the oil impacted and non-impacted catches can be used to assess the short-term impact in the fishery, while the cumulative percent catch deviations over any chosen number of years may be used as an estimate of the long-term impact.

For more detailed descriptions of the model system and individual submodels, the reader is referred to Reed (1980), Reed et al. (1981, 1982), and Spaulding, Saila et al. (1981, 1982), Lorda et al. (1982), Anderson and Spaulding et al. (1982).

APPLICATION OF MODEL SYSTEM TO GEORGES BANK STUDY AREA: SELECTION OF CASES

Selecting the appropriate oil spill test cases to perform is extremely complicated given the number of variables in the many model components, and the complex interconnections among the various submodels. Rather than attempt to determine through trial and error which spills would have the greatest impact on a given fishery or on the combined fishery, it was decided to select several oil spill scenarios that might be typical for the area and the proposed oil exploration activities. This selection process was performed after a review of the literature on likely spill locations and sizes (Danenberger 1977; Devanney and Stewart 1974; Moore et al. 1973).

To investigate the sensitivity of the model system a series of simulations has been performed varying spill location, spill size, oil type, and spill timing. An analysis of these results suggests that spill timing is one of the most critical parameters in determining impact.

To study the influence of spill timing on impact, a series of independent monthly blowout spill simulations was performed. Each discharged 68 million gallons of Norwegian Statfjord Crude over 30 days. Table 1 shows a detailed list of the oil spill fates input data for these monthly spill cases. Spill site location, the existing OCS Sale No. 42 lease sites, and the proposed OCS Sale No. 52 sites are shown in figure 2.

Three major criteria have been used in the selection of candidate fish species: (1) species which rank high in terms of commercial value and volume, (2) species which have been reasonably well studied and for which biological information is sufficient for building population dynamics models, and (3) species for which data concerning early life history stages are available for Georges Bank. Based on these considerations, Atlantic cod, _Gadus morhua_ and Atlantic herring, _Clupea harengus_ will be discussed here.

Table 1. Oil spill fates model input data.

Spill Location:	(1) North site 68° 12 min W longitude 40° 37 min N longitude
Thermocline depth:	10 meters
Wind:	Each month (Nantucket Weather Station) 1972-73, 3 hour intervals (Spills commence on the first day of each month)
Temperature	Winter - 7.5° C Spring - 5° C Summer - 14° C Fall - 10° C
Currents:	Derived from Bumpus et al. (1965) Seasonal drifter data plus wind driven slab flow
Oil:	
Density	0.8338 GM/CM**3
Interfacial tension	30.0 Dynes/CM**2
Kinematic Viscosity	7.95 Centistoke
Oil fraction (Statfjord Norway crude)	% by weight
Paraffin C6 - C12	15
Paraffin C13 - C22	13
Cycloparaffin C6 - C12	14
Cycloparaffin C13 - C22	12
Aromatic C6 - C11	7
Aromatic C12 - C18	3.5
Naptheno-Aromatic C9 - C25	3.5
Residual	32
Spill size and release rate:	Well blowout 68,000,000 gallon release Over 30 days
Total spill simulation time:	90 days or until oil leaves study area
Time step (Oil spill model):	3 hours
Horizontal dispersion rate:	10 M**2/Sec

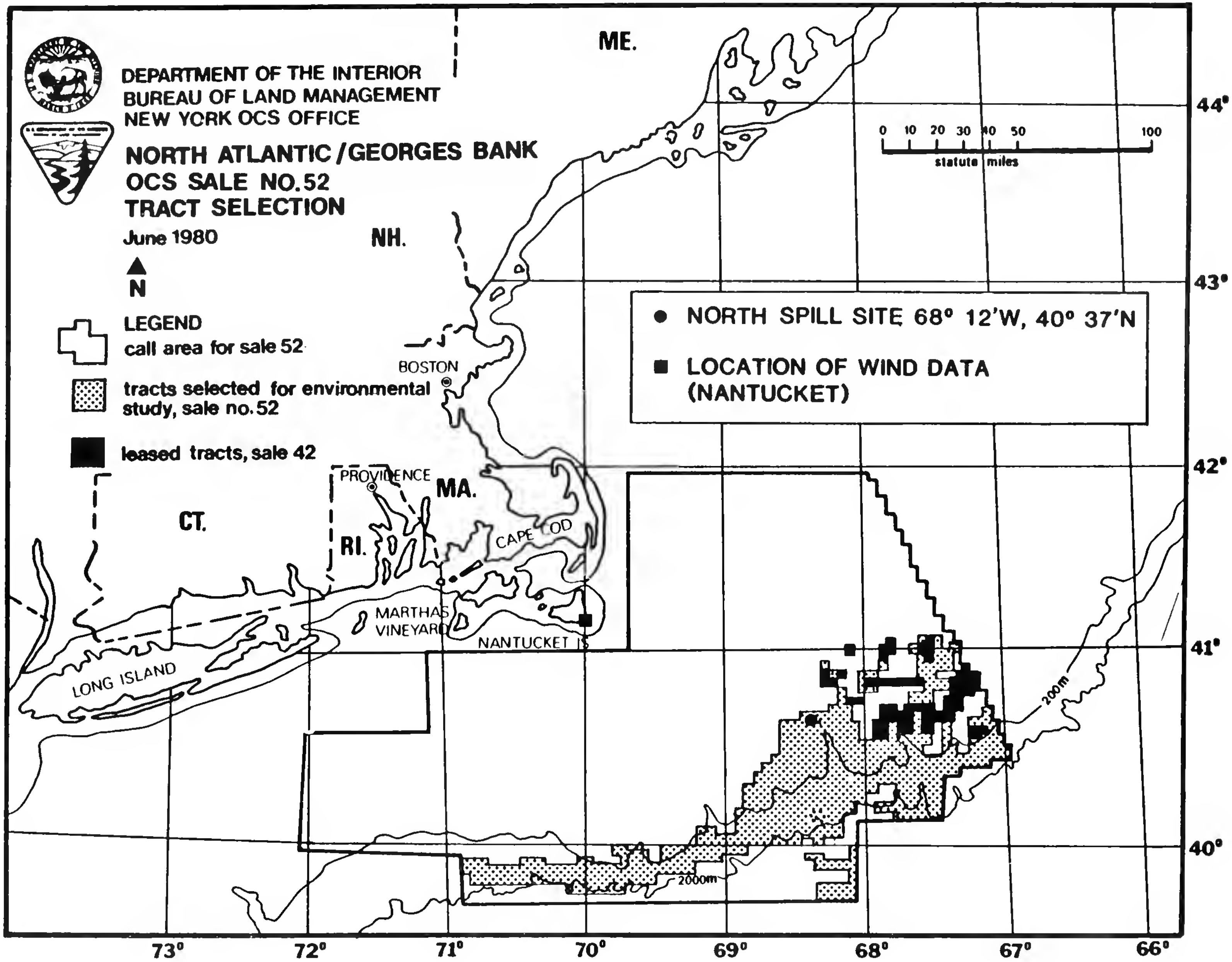


Figure 2. Study area detail showing the location of the spill site and the OCS lease tracts for Sales 42 and 52.

SUMMARY OF RESULTS

Oil Spill Fates Model

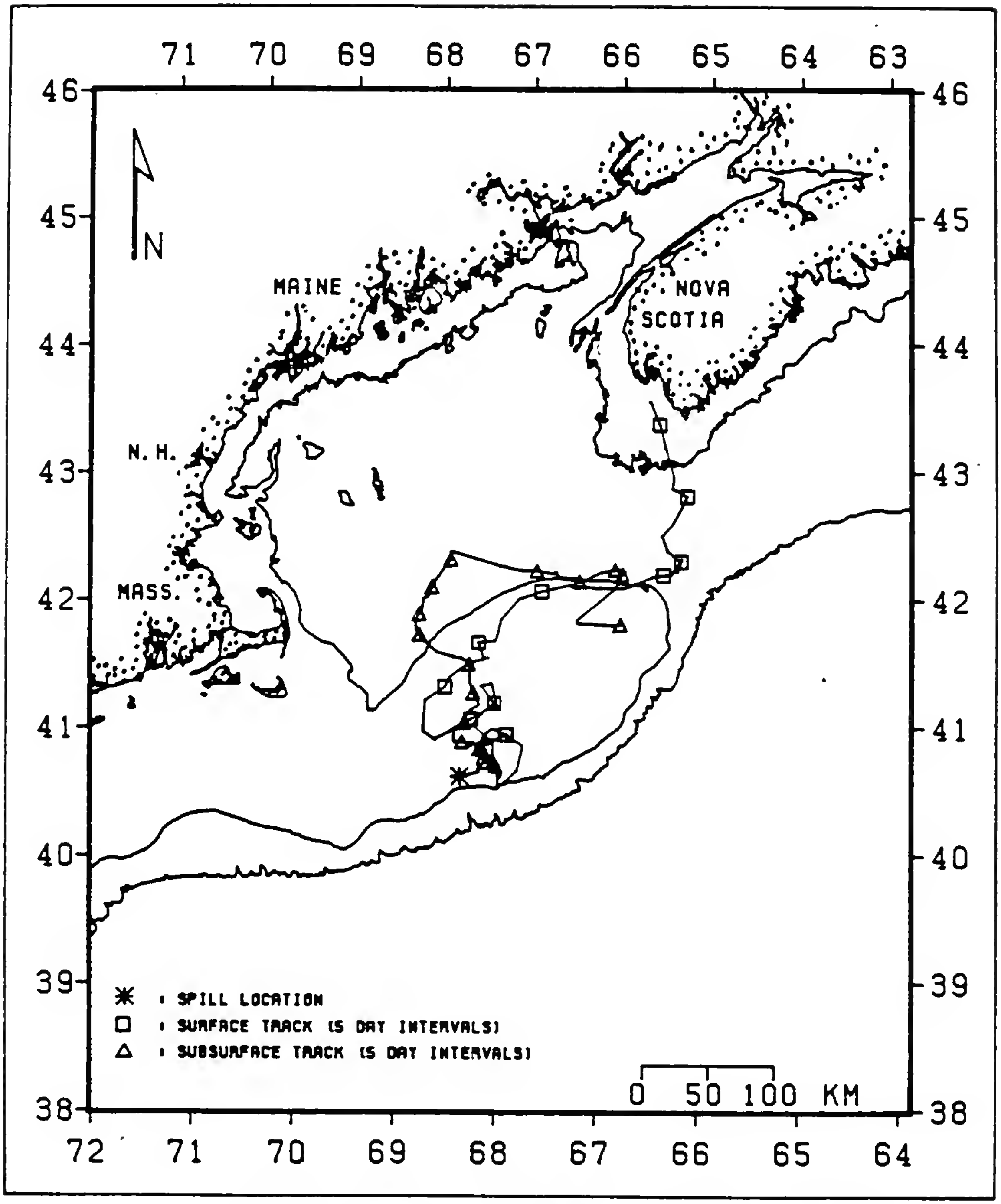
The oil spill fates model was used to simulate the 12 monthly spill cases noted in table 1. Figures 3 and 4 show the trajectories of the surface slick and subsurface hydrocarbon concentrations with time, noted in 5 day increments from the start of the spill event, for the May and December spills respectively. The other 10 monthly cases are documented in Spaulding, Saila et al. (1982). The centers of mass of the surface slick and the 50 part per billion (ppb) subsurface oil concentration contour are used to define the trajectories. An outline of the coast and the 100 m and 1,000 m bathymetric contour lines have been included to assist the reader in orienting the spill location to both land and important shelf and basin bathymetry. Noted at the bottom of each figure is detailed information on key spill parameters.

An analysis of the 12 simulations shows differences in the response of the surface and subsurface oil to the combined effects of the wind-induced and long-term residual flow patterns drawn from a summary of drifter studies (Bumpus and Lauzier, 1965). The surface spillets readily respond to wind forcing, while the subsurface oil is more markedly influenced by the long-term advective field. In overview, the subsurface oil trajectories are toward the southwest and west in the fall and winter months, and become strongly influenced by the Georges Bank gyre in the spring and summer. The trajectories of the surface spillets are more convoluted than the subsurface trajectories because of their strong response to the passage of weather events. Generally, surface spillets on Georges Bank are transported to the southwest or southeast during the fall and winter and toward the northeast from spring through early summer, following the seasonal mean wind conditions. Stronger winter winds carry the surface oil away from the spill site more rapidly than do the lower velocity late spring and summer winds.

It is interesting to note that when the wind speeds become more moderate, such as in August, the surface spillet trajectories are affected by the transient wind and residual flow in about equal magnitude. This observation suggests that knowledge of the offshore residual current patterns is critical in determining the trajectory of spilled oil in moderate weather, and indicates the need for a proper hydrodynamic modeling effort to produce improved impact estimates.

Table 2 presents the final mass balance in percent for each monthly spill simulation, with the environment partitioned into atmosphere, sea surface, and subsurface (water column) components. The first two columns show the mass and volume of oil spilled. The last column, labeled "outside domain", indicates that this percentage of the oil has left the study area by exiting through one of the model boundaries.

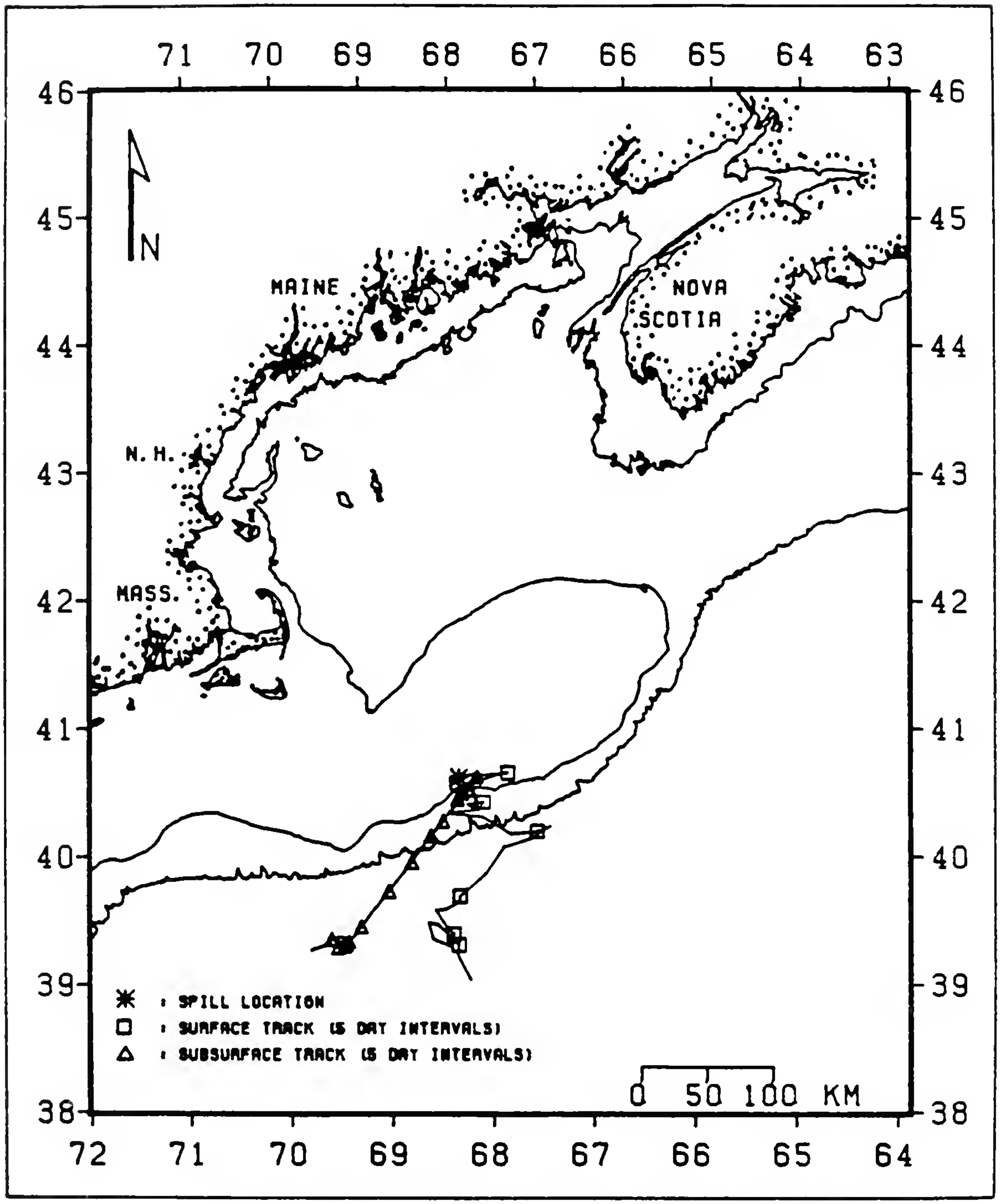
A review of the monthly oil spill final mass balances shows that 36.5 ± 0.5 percent is evaporated, 6 ± 3 percent is in the water column and the remaining 57 ± 3 percent of the spilled oil is found on the sea surface. Based on previous studies of mass balance for other oil spills (Spaulding, Saila, et al. 1981) it is clear that the major structure of the final mass balance relationships is determined by the oil type, with environmental factors such as wind and temperature exhibiting only secondary roles.



MOVEMENT OF SPILLED OIL FOR 90 DAYS AFTER THE RELEASE

SPILL PARAMETERS
OIL- STATFJORD NORWAY CRUDE
SITE- 68 DEG 12 MIN W. 40 DEG 37 MIN N
TYPE- WELL BLOWOUT
AMOUNT- 68 MILLION GAL OVER 30 DAYS
START- MAY . JULIAN DAY 121

Figure 3. Trajectories of the surface and subsurface hydrocarbon centers of mass for the May spill.



MOVEMENT OF SPILLED OIL FOR 70 DAYS AFTER THE RELEASE

SPILL PARAMETERS
OIL- STATFJORD NORWAY CRUDE
SITE- 68 DEG 12 MIN W. 40 DEG 37 MIN N
TYPE- WELL BLOWOUT
AMOUNT- 68 MILLION GAL OVER 30 DAYS
START- DECEMBER, JULIAN DAY 335

Figure 4. Trajectories of the surface and subsurface hydrocarbon centers of mass for the December spill.

Table 2. Final oil spill mass balances for the monthly simulations.

		MASS BALANCE (%) ***				ENVIRONMENTAL DATA	
	SPILL DATE*	ATMOSPHERE	SURFACE	SUBSURFACE	OUTSIDE DOMAIN	TEMPERATURE C°	WIND SPEED (m/sec)
WINTER	JANUARY (001)**	36.66	54.7	8.47	0.01	8.4	6.42
	FEBRUARY (032)	37.17	53.45	7.49	1.67	7.5	6.70
	MARCH (060)	37.15	53.8	6.7	1.98	6.7	6.10
SPRING	APRIL (091)	35.89	56.27	7.47	0.00	5.8	6.08
	MAY (121)	35.78	58.68	5.31	0.00	5.	4.93
	JUNE (152)	35.93	58.91	4.91	0.00	8.	4.67
SUMMER	JULY (182)	36.43	58.44	4.44	0.31	11.	4.48
	AUGUST (213)	37.02	58.77	2.77	1.14	14.	4.14
	SEPTEMBER (244)	36.57	57.85	5.24	0.03	12.6	4.73
FALL	OCTOBER (274)	36.19	58.29	5.33	0.00	11.3	4.83
	NOVEMBER (305)	36.33	56.98	6.5	0.00	10.0	5.37
	DECEMBER (335)	36.14	57.17	6.55	0.00	9.2	5.39

* SPILL PARAMETERS

OIL: STATFJORD NORWAY CRUDE
SITE: 68° 12' W, 40° 37' N
TYPE: WELL BLOWOUT
AMOUNT: 68 MILLION GALLONS (214,800 METRIC TONS)
OVER 30 DAYS
DATE: SPILL STARTED ON FIRST DAY OF MONTH
SIMULATION PERIOD: 90 DAYS

** JULIAN DAY OF SPILL START

*** THESE MASS BALANCE ESTIMATES ARE PROVIDED BY THE OIL SPILL FATES MODEL. THEY, IN GENERAL, DO NOT SUM TO 100% DUE TO ROUNDOFF ERRORS IN BOOKKEEPING FOR THE OIL MASS.

A comparison of the amount of oil in the water column with the mean wind speed over the first 30 days of the simulation period, for each spill event, shows a strong correlation. The amount of oil in the water column under a spillet increases approximately as the square of the mean wind speed (Audunson et al. 1979). The windier winter months show double the oil mass in the water column compared to the quiet summer months. It is also seen that the source for this additional subsurface oil is the sea surface. Water temperature has no effect on the oil mass entrainment as presently formulated, and therefore no correlation between subsurface oil and water or air temperature is noted.

The slight variations in the amount of oil found in the atmosphere represent the combined effects of temperature and wind with higher temperatures and faster winds giving increased evaporative losses. The summer spills appear to be dominated by the temperature effect, while the winter spill mass balances are largely governed by the wind.

Fishery model

Figures 5 and 6 show the temporal distribution of spawning activity as simulated for herring and cod, and the reductions of incoming year classes due to each of the 12 spills investigated. These figures demonstrate the critical importance of spill timing relative to the temporal distributions of spawning. The results shown are based on egg and larval oil-induced mortalities estimated with the ichthyoplankton transport and fate model. The ensuing year class reductions were then incorporated into the adult fish population model, a non-linear, non-spatial matrix formulation (Lorda and Saila 1980), to project variations in catch.

It is clear also that the size of the spawning grounds and their locations relative to the spill origin (fig. 7 and 8) play a major role in determining the extent of the cohort reduction in each case. The two cod spawning grounds (fig. 7) are well defined, and the spill site is located on the edge of the largest one (on Georges Bank). The result is that the largest cod cohort reduction (77.5%) occurs when the spill starting time (day 60) matches the peak of the combined Georges Bank and Nantucket Shoals spawning. This indicates a concentration of the oil-induced mortality in the earlier life stages (eggs and yolk-sac) before significant larval dispersion occurs. Unlike cod, the herring spawn over a large area of Georges Bank (fig. 8). This initially higher dispersion of the herring eggs results in consistently lower oil-induced mortalities. The largest oil impact for the herring (17.6% cohort reduction) does not match its spawning peak as in the case of cod, but occurs about 45 days later. Although a mismatch of 12 days can be explained by the initially demersal herring eggs (yolk-sac is the first pelagic stage), the remaining 30 days of difference between the spawning peak and the largest oil-induced mortality seem to indicate that the spreading of the spill must proceed for that number of days before the largest possible number of herring larvae can be oiled at a lethal concentration level.

The effective reductions of the initial cohort sizes caused by the oil were implemented in the fishery model by specifying equivalent one-time reductions in the probability of survival through year-0. The effects of this reduced cohort survival on the expected catches over the 50 years following the occurrence of the oil spill are summarized in tables 3 and 4 for the herring and cod fisheries,

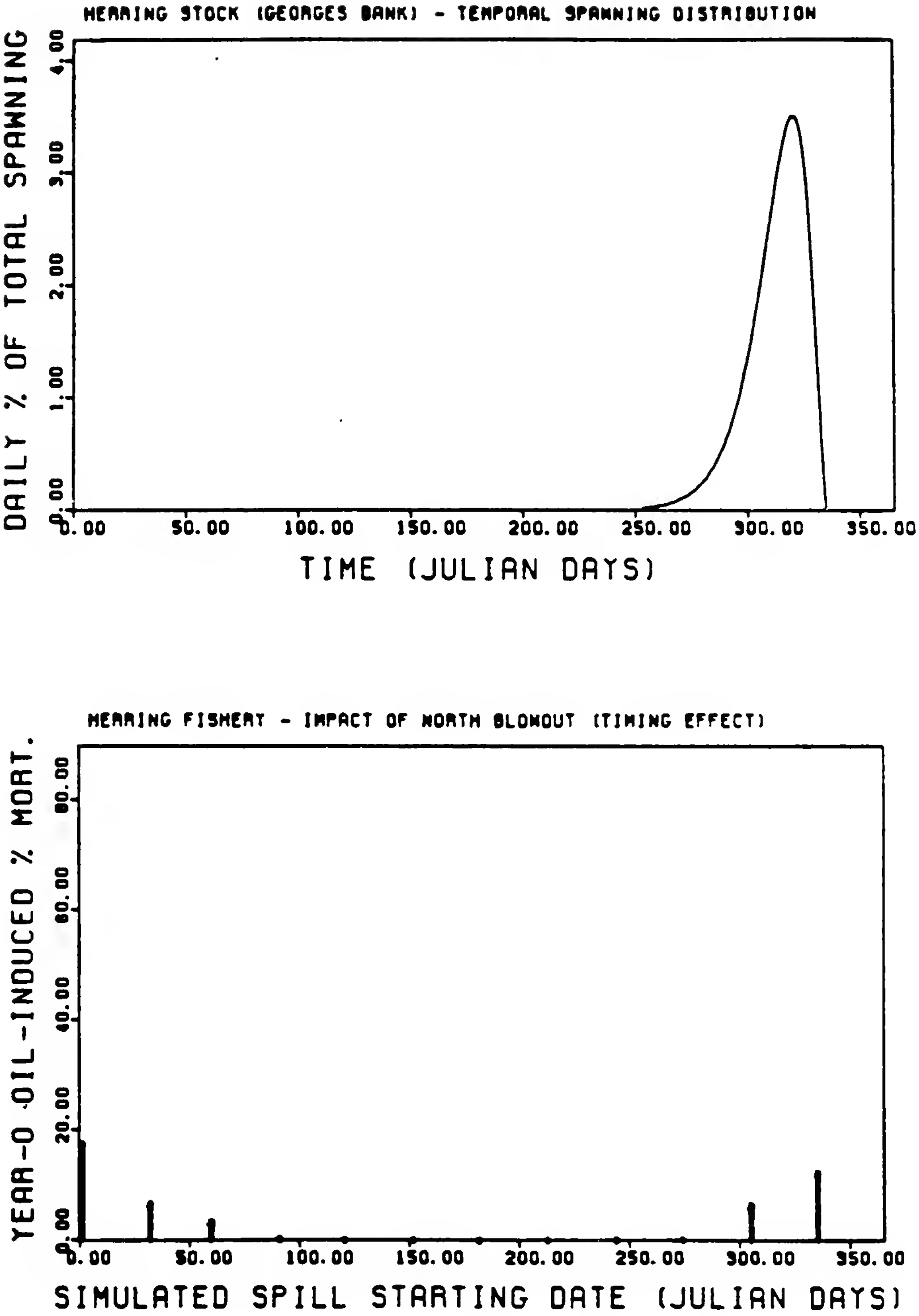
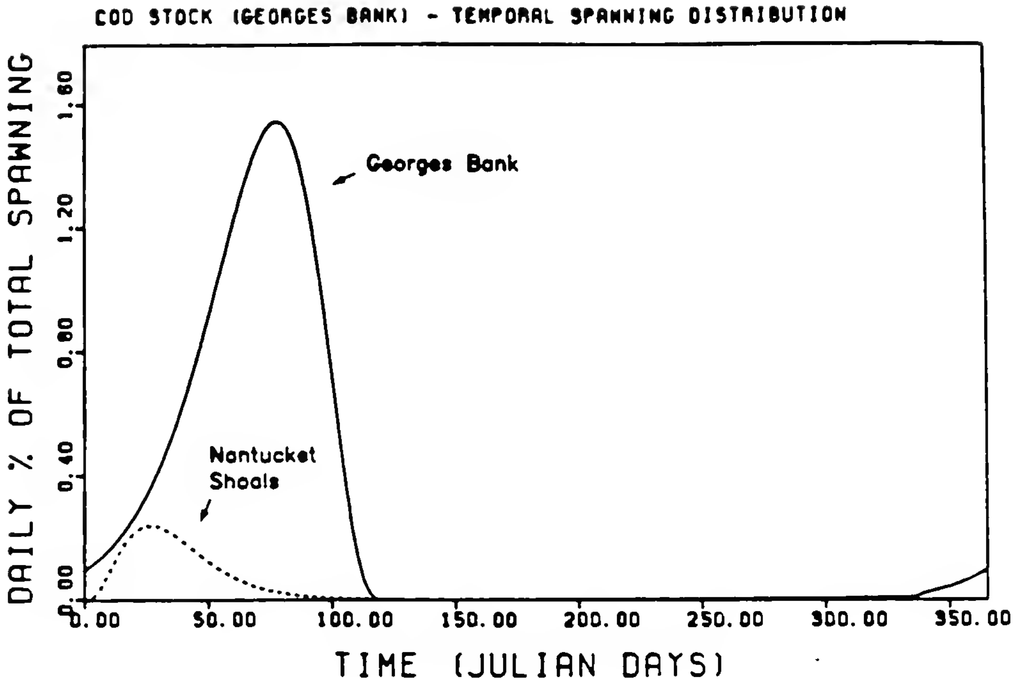


Figure 5. Herring population. Temporal spawning distribution and oil-induced mortalities in Year-0 class for the twelve monthly blowout spills.



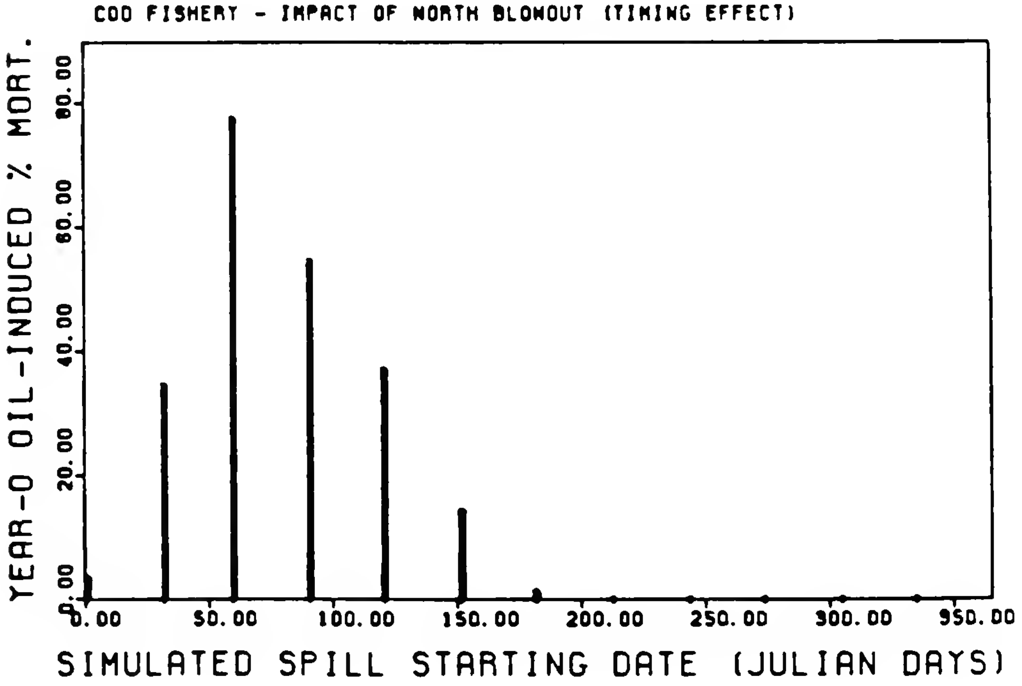


Figure 6. Cod population. Temporal spawning distribution and oil-induced mortalities in Year-0 for twelve monthly blowout spills.

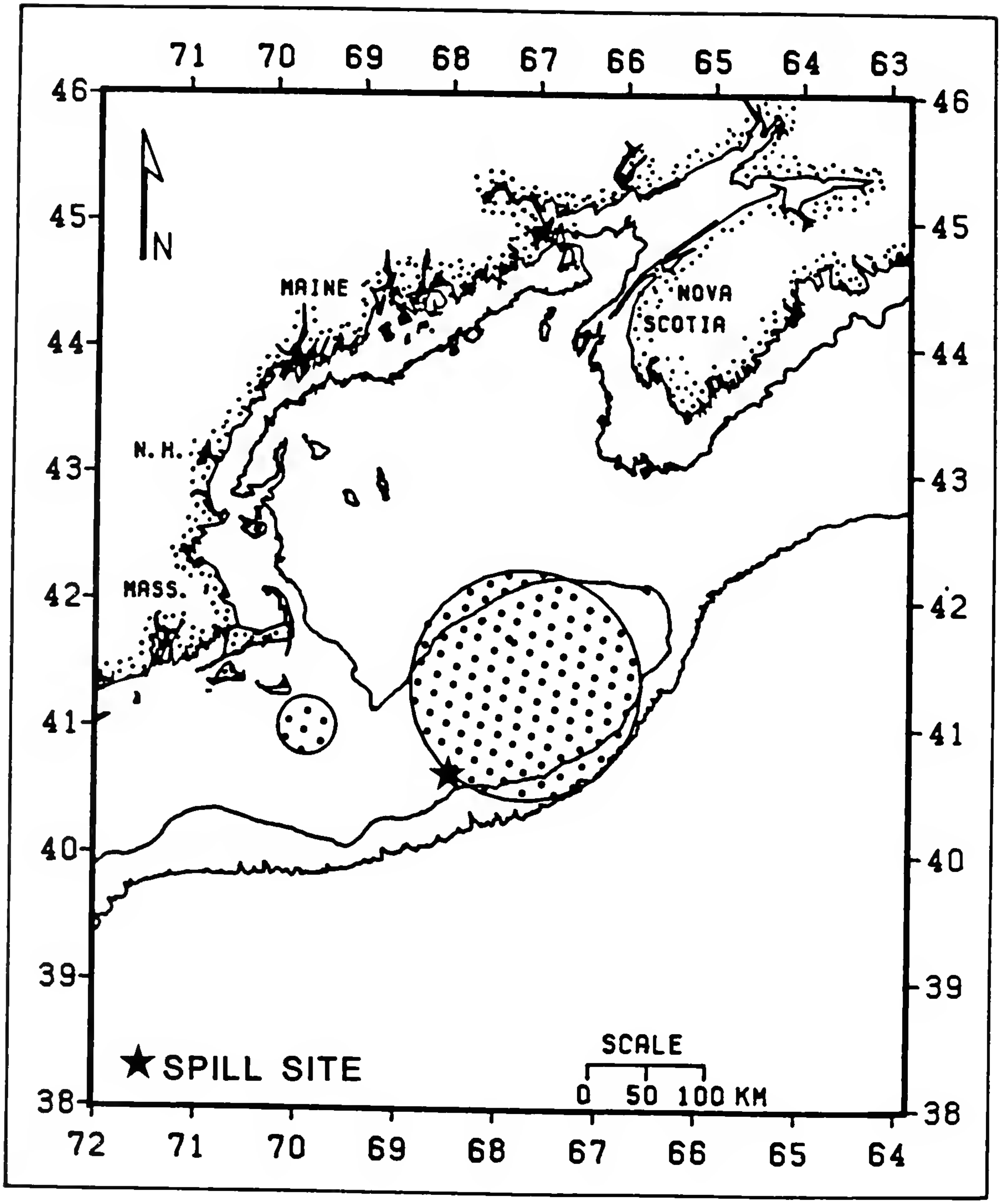


Figure 7. Spawning locations for Atlantic cod.

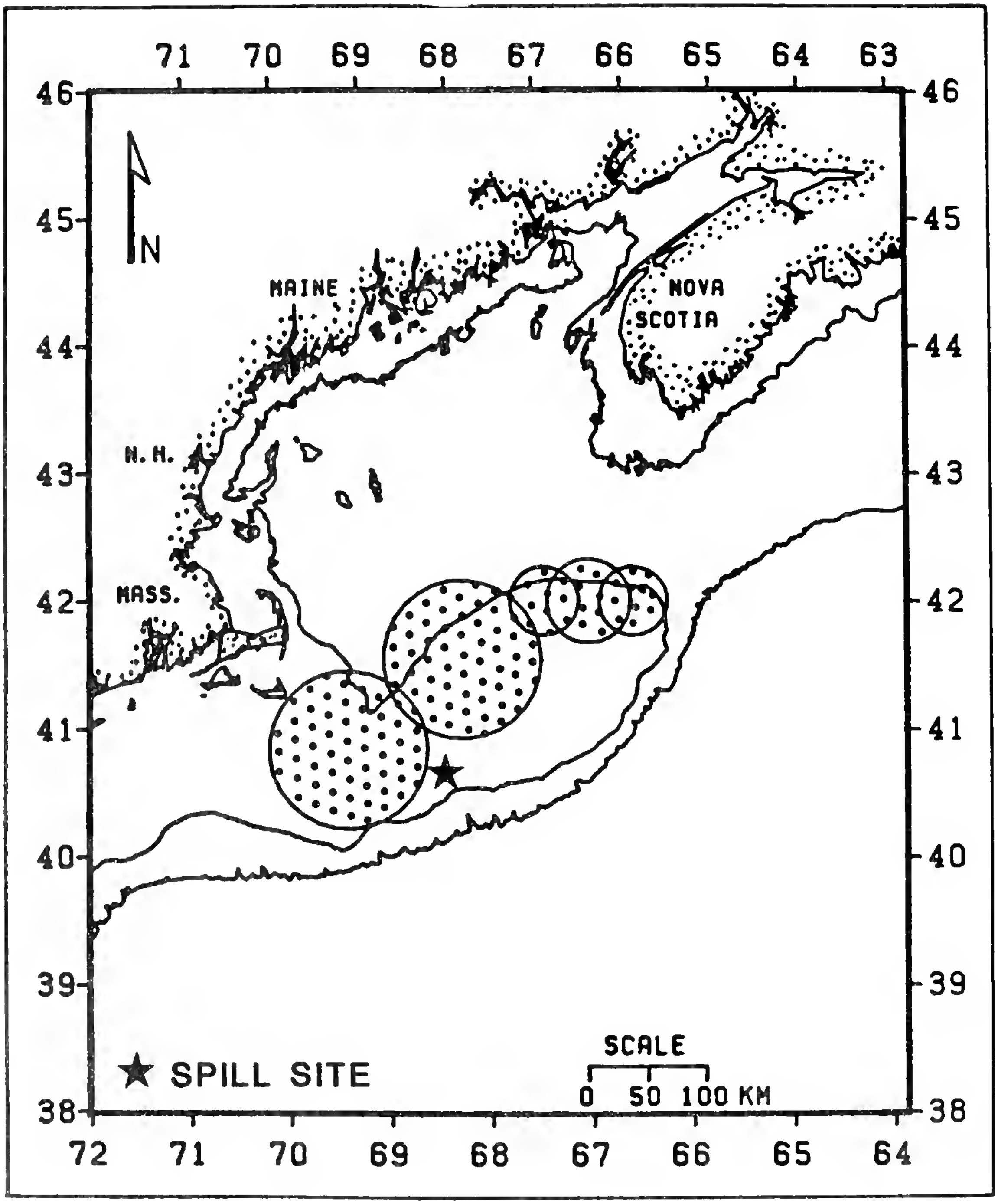


Figure 8. Spawning locations for Atlantic herring.

respectively. These tables show the values of three selected model output variables for each of the 12 spills simulated. The interpretation of these output variables, largest annual catch reduction A, largest cumulative reduction B, and final long-term loss C, should be clear from the headings, and from figures 9 and 10. Variable C, the sum of the catch losses, or ultimate loss to the fishery for each spill, is discussed below. The January (Julian day 1) oil spill for herring and the March (Julian day 60) spill for cod are used for this discussion since they resulted in the largest percent cohort reductions in each of the two species.

The behavior of the three fishery model variables for these two cases over the 50 years following the spill is shown in figure 9 for the herring and figure 10 for the cod. It is clear from these plots that the cod compensatory mechanism overcompensates for the added oil-induced mortality, while the herring population, with a weaker compensatory mechanism, is only able to return to its initial equilibrium size. Although A, the largest one-year catch loss, would be similar for comparable oil-induced mortalities in both species, B and C, the largest cumulative catch loss over 50 years and the ultimate loss, respectively, are quite different. In the case of the herring, both B and C are virtually the same variable because the largest cumulative loss always corresponds to the last year in the series of catch projections. For the cod fishery, B is maximum in the 7th year after the spill, decreasing thereafter and converging to an ultimate loss C, which is less than half of the maximum value of B (fig. 10). It should be noted that C, the ultimate loss to the fishery, of roughly 21 percent of a single year's equilibrium catch in the case of cod for the March spill, cannot be shown in figure 10 except as the value to which B ultimately converges. The difference in the dynamics of the impacted herring and cod fisheries results from the different formulations of compensatory mortality used to model these two species. It is interesting to note that for the most damaging time of the blowout, the ultimate catch loss for both fisheries is roughly 21 percent of the one-year equilibrium catch, despite the fact that the actual oil-induced ichthyoplankton mortality for the herring was only 17.6 percent versus 77.5 percent for the cod.

MODEL SYSTEM SENSITIVITY STUDIES

When applying a system of models as complex as that outlined here, the interactions between the submodels and the sensitivity of model system predictions to various assumptions and parameterizations are critical to determining model validity, and in understanding what processes are most important in controlling system behavior. Sensitivity studies to percent loss of a year class and the effect of the Georges Bank gyre on impact predictions are discussed below. These two topics have been selected for presentation since they are the key to answering some common questions related to the impact of oil spills on commercial fisheries for the Georges Bank Region: What if a spill eliminates an entire year class? Does the Georges Bank gyre trap pollutants and cause increased impacts?

Additional sensitivity studies on toxicity threshold levels, compensatory mortality regimes, and all major fishery model parameters are documented in Reed et al. (1979, 1981), and Spaulding, Saila et al. (1981, 1982) and are currently an area of active investigation.

Table 3. Herring fishery (Georges Bank): North Blowout spill impact simulations. Summary of simulation runs for different spill starting dates.

MODEL Output Variables:			(A)		(B)		(C)
JULIAN DATE	Percent OIL-MORT	TIME STEPS	CATCH $ LOSSES Largest	YR.	CUMULATIVE CATCH $ LOSSES Largest	YR.	FINAL SUM CATCH $ LOSSES
0001	17.61	50	3.99	4th	21.30	50th	21.289
0032	6.69	50	1.51	4th	8.07	50th	8.063
0060	3.34	50	0.76	4th	4.03	50th	4.023
0091	0.14	50	0.03	4th	0.17	50th	0.167
0121	0.03	50	0.01	4th	0.03	50th	0.034
0152	0.00	--	0.00	---	0.00	----	0.000
0182	0.00	--	0.00	---	0.00	----	0.000
0213	0.02	50	0.01	4th	0.03	50th	0.027
0244	0.04	50	0.01	4th	0.05	50th	0.048
0274	0.07	50	0.02	4th	0.09	50th	0.087
0305	6.40	50	1.45	4th	7.73	50th	7.721
0335	12.33	50	2.79	4th	14.89	50th	14.884

Table 4. Cod fishery (Georges Bank): North Blowout spill impact simulations. Summary of simulation runs for different spill starting dates.

MODEL Output Variables:			(A)		(B)		(C)
JULIAN DATE	Percent OIL-MORT	TIME STEPS	CATCH $ LOSSES Largest	YR.	CUMULATIVE CATCH $ LOSSES Largest	YR.	FINAL SUM CATCH $ LOSSES
0001	3.92	50	0.68	4th	2.82	7th	1.342
0032	34.50	50	5.99	4th	24.80	7th	10.744
0060	77.52	50	13.46	4th	55.68	7th	20.733
0091	54.62	50	9.49	4th	39.25	7th	15.889
0121	37.09	50	6.44	4th	26.67	7th	11.454
0152	14.24	50	2.47	4th	10.24	7th	4.725
0182	1.10	50	0.19	4th	0.79	7th	0.380
0213	0.00	--	0.00	---	0.00	---	0.000
0244	0.00	--	0.00	---	0.00	---	0.000
0274	0.00	--	0.00	---	0.00	---	0.000
0305	0.00	--	0.00	---	0.00	---	0.000
0335	0.11	50	0.02	4th	0.08	7th	0.037

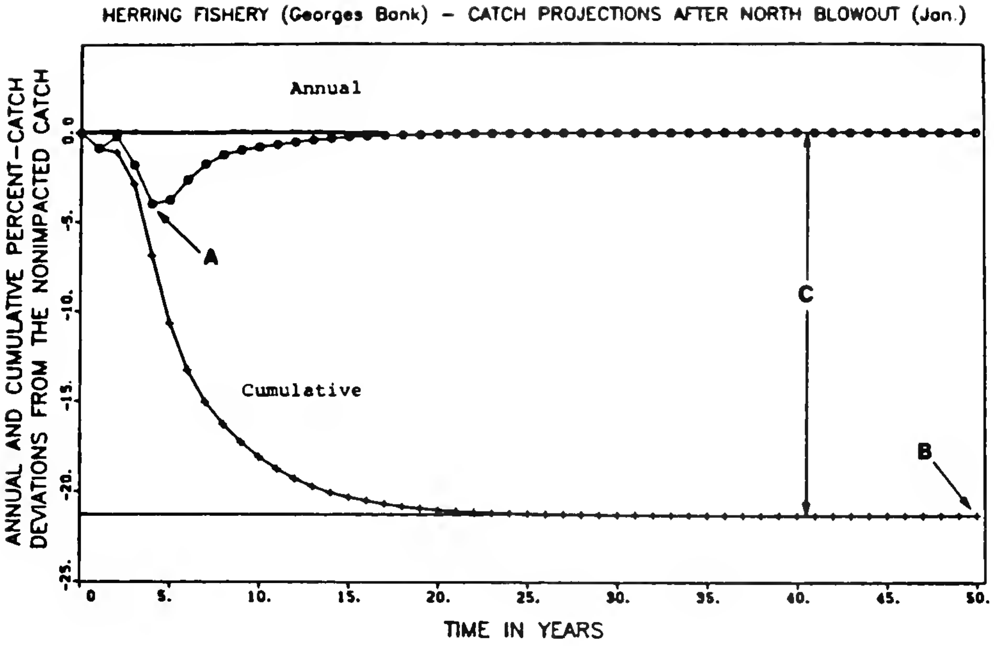


Figure 9. Impacted herring fishery. Catch projections after the January (Day 1) North Blowout spill simulation.

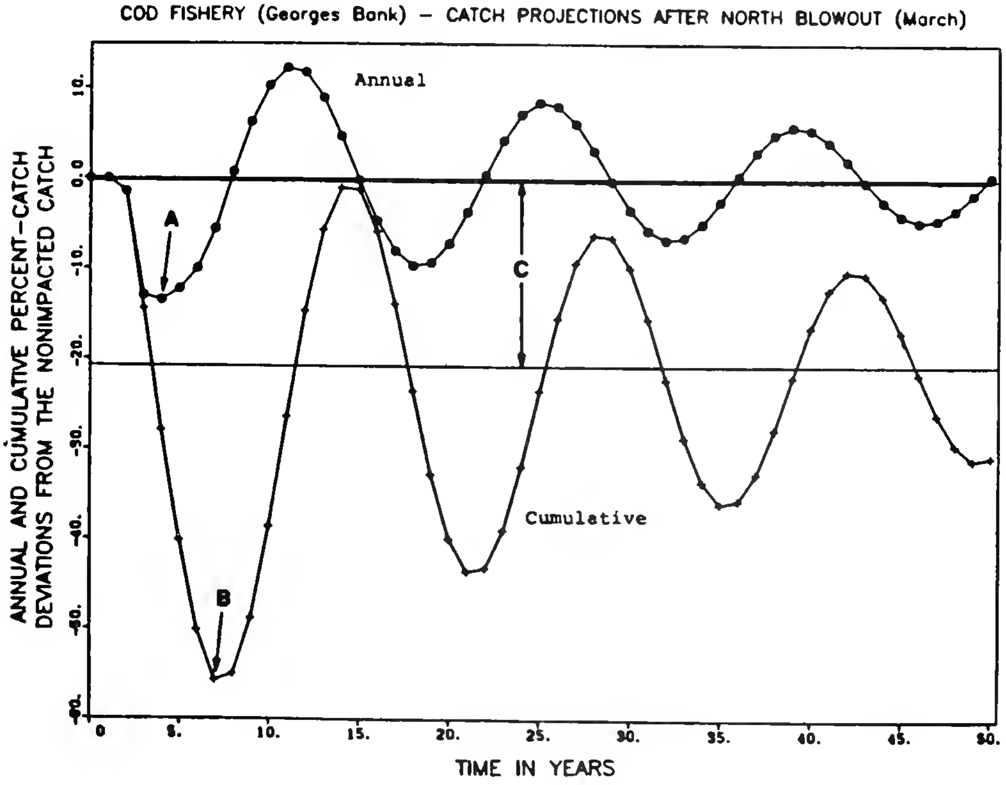


Figure 10. Impacted cod fishery. Catch projections after the March (Day 60) North Blowout spill simulation.

Sensitivity to percent loss of a year class

The sensitivity of the two selected species to percent loss of a year class has been investigated by projecting the age-specific density vector of each equilibrium population for 50 years after simulating 10 percent incremental reductions in the probability of survival of the first cohort through year-0. The results of these simulations for the herring and cod fisheries are shown in tables 5 and 6, respectively.

The interest of these simulations comes from the fact that within the limitations of the fishery model itself they cover the full range of impacts resulting from any oil spill event that might occur on Georges Bank regardless of size, type, timing, and duration, as long as two or more successive cohorts of the same species are not impacted. It is interesting to note that even the elimination of an entire year class (i.e., 100% cohort reduction) results in rather moderate losses to the fishery in the case of cod. On the other hand, the loss of an entire year class in the herring population results in ultimate losses to the fishery equivalent to more than a one-year equilibrium catch (table 5).

These differences in sensitivities of the cod and herring are the result of the different population structures and fecundities of each species, together with the different compensatory formulations used to model them. The strongest compensation occurs in the cod, which typically follows a dome-shaped (Ricker 1958) form of stock and recruitment relationship. The herring compensatory ratio is about 25 percent less than for the cod. In addition, the herring compensatory mortality follows the Beverton and Holt (1954) form of stock recruit relationship, which weakens even further the ability of this species to compensate for year class reductions. The net result is that the herring stock demonstrates less dynamic response to large cohort reductions than the cod stock.

Finally, the largest one-year catch loss (model output variable A), for the two species can be understood by examining the age structure of the total catch. It is clear that the value of A cannot be larger than the largest percent contribution of a single age-class to the total catch. For instance, A is 17.4 percent for the cod for the loss of an entire year class, which is the contribution of the four year olds to the total catch. Thus, a fishery based on a long-lived species such as cod, in which the catch is typically composed of many year classes, is much less susceptible to large catch losses from single event, acute oil spills. A fishery based on the current herring stocks on the other hand, will tend to rely upon only one or two age groups to supply most of the catch, so that recruitment fluctuations caused by oil spills (or variations in natural mortality) will be more strongly reflected in the catch on a percentage basis.

Effect of Georges Bank gyre on impact predictions

An issue which has consistently been raised at the public hearings on the lease sales of tracts off the New England coast concerns the possible formation of an anticyclonic gyre on Georges Bank. There is a variety of oceanographic data supportive of such a gyre during the spring and early summer (Bigelow 1927; Bumpus and Lauzier 1965; EG&G 1979). The existence of such a feature makes sense biologically in that ichthyoplankton would be retained in the Bank's

Table 5. Herring fishery (Georges Bank): Sensitivity to percent loss of a year-class. Summary of simulation runs with 10 percent incremental losses.

MODEL Output Variables:			**(A)**		**(B)**		**(C)**
					CUMULATIVE		**FINAL SUM**
SIMUL No.	**Percent OIL-MORT**	**TIME STEPS**	**CATCH $ LOSSES Largest**	**YR.**	**CATCH $ LOSSES Largest**	**YR.**	**CATCH $ LOSSES**
1	10.00	50	2.26	4th	12.08	50th	12.067
2	20.00	50	4.53	4th	24.21	50th	24.194
3	30.00	50	6.79	4th	36.41	50th	36.382
4	40.00	50	9.06	4th	48.67	50th	48.636
5	50.00	50	11.32	4th	61.00	50th	60.959
6	60.00	50	13.59	4th	73.41	50th	73.356
7	70.00	50	15.85	4th	85.89	50th	85.830
8	80.00	50	18.11	4th	98.46	50th	98.387
9	90.00	50	20.38	4th	111.12	50th	111.031
10	100.00	50	22.64	4th	123.86	50th	123.768

Table 6. Cod fishery (Georges Bank): Sensitivity to percent loss of a year-class. Summary of simulation runs with 10 percent incremental losses.

MODEL Output Variables:			**(A)**		**(B)**		**(C)**
					CUMULATIVE		**FINAL SUM**
SIMUL No.	**Percent OIL-MORT**	**TIME STEPS**	**CATCH $ LOSSES Largest**	**YR.**	**CATCH $ LOSSES Largest**	**YR.**	**CATCH $ LOSSES**
1	10.00	50	1.74	4th	7.19	7th	3.361
2	20.00	50	3.47	4th	14.39	7th	6.522
3	30.00	50	5.21	4th	21.57	7th	9.480
4	40.00	50	6.95	4th	28.76	7th	12.234
5	50.00	50	8.68	4th	35.94	7th	14.782
6	60.00	50	10.42	4th	43.11	7th	17.124
7	70.00	50	12.16	4th	50.29	7th	19.261
8	80.00	50	13.89	4th	57.46	7th	21.193
9	90.00	50	15.63	4th	64.63	7th	22.925
10	100.00	50	17.37	4th	71.79	7th	24.461

plankton-rich water, explaining the intense productivity of the area (EG&G 1981). The problem is that an oil spill might be similarly retained, and its impact on the ichthyoplankton thereby increased.

The oil spill - fishery impact assessment model system has been used to investigate this question for winter (Julian day 32) and spring (Julian day 121) spills. The residual advective field underlying these two test simulations is the summer data set shown in figure 11 derived from charts compiled by Bumpus and Lauzier (1965). This residual current field, which was held constant through the two simulations, was selected because it showed the strongest gyre-like configuration over the banks. Random walk diffusion and the definition of the wind driven hydrodynamics remain as in prior simulations.

The impacts on cod of the 68 million gallon north blowout spill scenario under this altered current pattern are summarized in table 7. Not surprisingly, the replacement of the winter residual current field, which is essentially unidirectional to the south and southwest, with the strong summer gyre results in greatly increased impacts. The impacts for spills on days 32 and 121 are nearly the same, subject to this constant residual advective field. This is due firstly to the fact that dispersion of oil into the water column is relatively complete after a few days. The effect of wind on the subsurface distribution is therefore limited primarily to this initial period. Secondly, the two spills occur on either side of the cod spawning peak near Julian day 90 (fig. 6). Because the day 32 subsurface oil remains in the general vicinity, it affects the eggs spawned during this peak time. Conversely, in the case of an oil spill occurring on day 121, the eggs stay in the area longer, and therefore remain subject to the toxic action of hydrocarbons released after the time of maximum spawning activity. These observations simply reflect the expected tendency of a gyre-like formation to retain transported constituents.

Although this model experiment is relatively artificial, it demonstrates the importance of the inclusion of subsurface representations in oil spill impact prediction work, and strongly underscores the importance of determining the correct structure of the advective field for correct estimation of impact magnitudes.

CONCLUSIONS

It is clear from the above discussion that spill timing, spatial and temporal spawning distributions, and population dynamics of the species of concern are critical factors in determining the impact of spill events on the Georges Bank herring and cod fisheries. In the case of cod, compensatory mechanisms within the fishery, combined with the existence of several year classes of mature fish, will tend to attenuate or reduce large losses in one year class caused by the proximinity of spawning site to spill location. For herring, diffuse spawning locations protect the population from localized oil spill pollution events, although internal population dynamics result in relatively large catch losses in the long term due to small numbers of year classes.

Sensitivity studies on percent loss of a year class further display the importance of compensation in determining the impacts of a pollutant event on a fishery. A fishery such as cod, with relatively strong compensatory behavior,

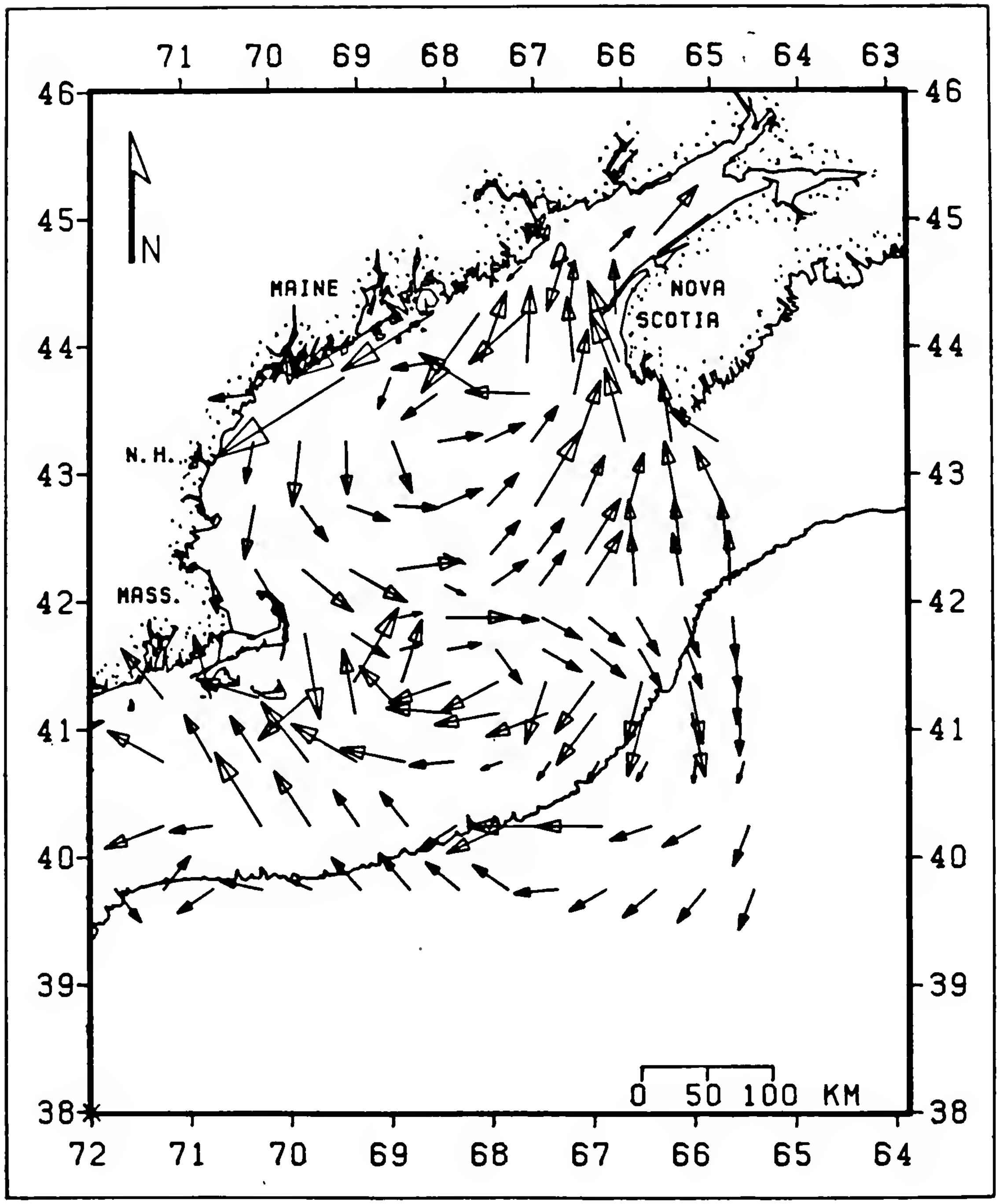


Figure 11. Current field used for gyre transport investigations.

Table 7. Effect of gyre on impact predictions for Georges Bank cod eggs and larvae. Entries are percent of one year's ichthyoplankton oiled.

		SPILL DAY	
		32 (WINTER)	121 (SPRING)
RESIDUAL ADVECTIVE FIELD	NORMAL DYNAMICS	34.5%	36.9%
	GYRE	60.1%	60.5%

can undergo a single pollutant event with relatively small losses in projected catch. Herring, on the other hand, with its weak compensation shows a much greater sensitivity to losses of new recruits.

The presence and duration of the Georges Bank circulation gyre, as represented by the simple parameterization used here, are important factors in determining impact magnitudes. Significantly higher egg and larval mortality are observed if the gyre is present, because this circulation feature increases the exposure time of spawned products to toxic levels of the pollutant. This simple simulation demonstrates that the circulation features of this area must be carefully considered in order to make realistic assessments of impacts.

While significant progress has been made through this modeling approach in understanding the impact of oil spills on a commercial fishery, we have only taken the first step. The most important lesson of the research to date is that realistic impact assessment procedures need to take an integrated view of the environment. The interrelationships among the components of the physical and biological systems under study must be correctly represented within the model to provide a sound basis for rational resource management decisions. As with all productive modeling studies the present work has helped focus on the changes necessary to improve the credibility and accuracy of the model system. Efforts are currently in progress to further validate and upgrade each component of the impact assessment methodology.

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A SIMULATION MODELING FRAMEWORK FOR ECOLOGICAL RESEARCH IN COMPLEX SYSTEMS: THE CASE OF SUBMERGED VEGETATION IN UPPER CHESAPEAKE BAY[1]

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ABSTRACT

This paper provides a broad overview of a system of simulation models developed in conjunction with empirical research programs to study the declining abundance of submerged aquatic vegetation (SAV) in Chesapeake Bay and the effects of this decline on ecological and socio-economic processes. A hierarchical organization of models and submodels allowed the simplification needed for tractability while maintaining sufficient detail for examining mechanisms of ecological interaction. Two models of the SAV ecological subsystem are presented. First, the Autotroph Model was used to investigate consequences of shifting competition for light and nutrients among 4 groups of primary producers (SAV, phytoplankton, epiphytes and benthic microalgae). This model, which has been calibrated and verified against independent data sets, was used to extrapolate from controlled experiments to consider effects of nutrient enrichment. The second of these, the Nekton Model, was developed to test possible effects of declining SAV on the trophic structure and relative abundance of 3 fish groups. The model's design utilizes certain elements of traditional fish population models within the generic structure of an ecosystem model. An SAV resource management model was developed by aggregating the details of these and other ecological submodels and is linked to a suite of simulation models which relate human activities to estuarine processes and societal values in the Bay region. We argue here that this management modeling framework allows results of scientific research to be integrated into political and socio-economic networks toward balanced uses of those estuarine resources related to SAV.

INTRODUCTION

Estuaries such as Chesapeake Bay are complex and dynamic ecological systems which benefit human societies in many ways. On one hand, these coastal ecosystems provide a bountiful source of fisheries production and diverse recreational opportunities. On the other hand, natural biogeochemical processes within these systems are capable of transforming many wastes emanating from human activities into useful components of regional and global cycles. In some cases low levels of waste inputs (such as nutrients and organics) can, in fact, enhance estuarine productivity. However, in many of these environments waste loading rates are such that they detract significantly from the estuary's value as a source of fisheries and recreation. Hence, a serious problem evolves wherein legitimate but competitive uses of the natural resource are in direct conflict with one another.

In the last two decades, Chesapeake Bay has undergone some documented changes. One such change has been the drastic decline of submerged aquatic vegetation (SAV) which once dominated littoral zones throughout the estuary. Coincident with this loss of aquatic plants, there have been significant changes in water quality (including increased levels of turbidity, nutrients and agricultural herbicides), as well as declines and shifts as in various fisheries (Boynton et al., 1979). Stevenson and Confer (1978) postulated that many of these alterations in water quality are attributable to increased waste loadings to the Bay from both sewage outfalls and diffuse sources, and that such deteriorating conditions have led to the loss of ecosystems associated with these submerged plants. Moreover, it was hypothesized that the decline of SAV has contributed to detrimental changes in fisheries production (Boynton et al., 1979).

Hence, this loss of SAV communities represents a convenient case study for examining the emerging problem of managing and balancing the conflicting uses of estuarine resources. The purpose of this paper is to describe a simulation modeling framework which has served to organize, focus and elaborate a broad empirical research program for investigating this problem.

RESEARCH ORGANIZATION AND DESIGN

Peception of Problem

In figure 1 we have illustrated our perception of the important interactions related to the decline of SAV, including (1) factors contributing to the decline, (2) ecological consequences of the decline, and (3) socio-economic ramifications of this ecosystem modification (Boynton et al., 1981). In this cartoon SAV are shown to act as natural nutrient sinks and sediment traps, both processes having economic analogs in terms of sewage treatment plants and channel dredging operations, respectively. Furthermore, SAV communities are suggested to be important sources of food and habitat promoting growth of fish, shellfish, and waterfowl stocks which are harvested in commercial and recreational endeavors. Various watershed activities are shown to influence estuarine water quality (nutrient, sediment, and herbicide additions) via direct discharges and runoff which are, in turn, regulated by rainfall and other factors. Throughout this cycle some economic enterprises (e.g., agriculture) may detrimentally influence SAV while others (e.g., fishing and dredging) are affected by plant losses. While this presentation may be useful as an overview of the basic relationships involved in the problem, it does not indicate the nature of such relationships. Hence, we need a more explicit framework within which mechanistic connections are embodied.

We recognized in this research project a rare opportunity to address several scientific hypotheses of theoretical and empirical interest within a broad context of resource management questions. However, to do so effectively it was necessary to use a scheme whereby the complexity of this problem could be dealt with in an organized, piece-wise simplified fashion. We, therefore, developed a hierarchical approach for the overall research program which enabled us to integrate highly controlled experiments (testing mechanistic hypotheses) together with descriptive field measurements (characterizing the structure and function of these SAV ecosystems). This allowed us to combine a spectrum of research methods and scales of interest into a unified effort. We have discussed the relative merits and philosophical underpinnings of this scheme elsewhere at length (Kemp et al., 1980).

A variety of conceptual and simulation models were utilized to integrate this research program. We reasoned that models could facilitate the coupling of experimental findings on "causality" (i.e., influence) with the inherently holistic perspective of descriptive in situ observations. Furthermore, simulation models could be used to confer generality upon specific results at either end of the controllability-realism spectrum (Kemp et al., 1980). This would be done by constructing, calibrating, and verifying models with data from a variety of systems. Thus, we concluded that simulation models could be used to examine the possibility that altered water quality conditions contributed to the decline of SAV in various regions of Chesapeake Bay. Such models would help to interpolate and extrapolate the results of experimentally inferred

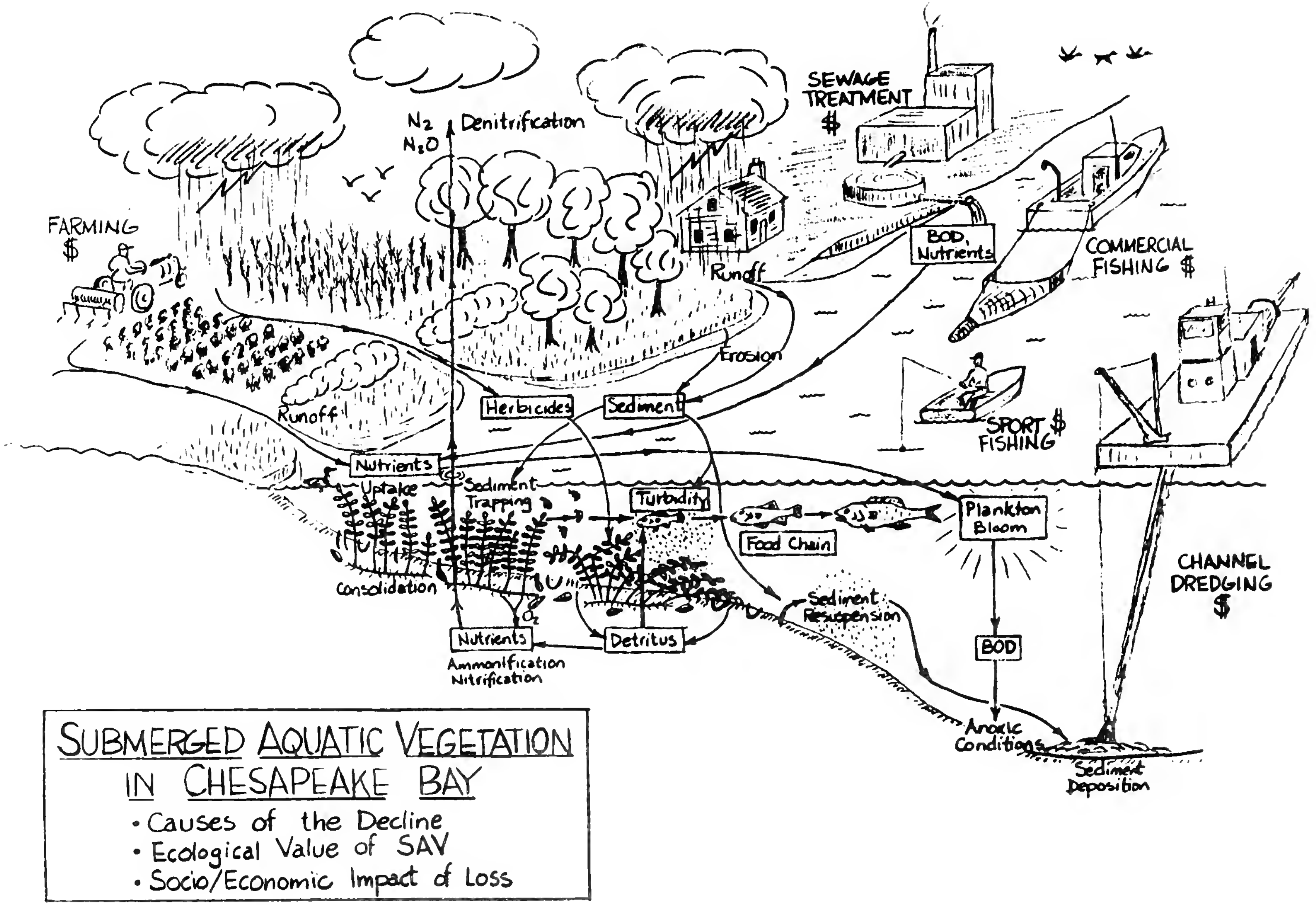

Figure 1. Conceptual schematic of depicting submerged aquatic vegetation in relation to human activities and ecological production along with other resource values.

relations for any combination of water quality factors observed (past or present) in nature.

Simulation Modeling Structure

We employed two distinctly different strategies for simulation modeling which were central to thee overall SAV research program. One strategy was directed primarily toward understanding the dynamic behavior of the seagrass ecosystem including energy flux, predator-prey interactions, nutrient cycling and trophic structure. As before (in the broad design of our research program), we utilized a hierarchical perception to decompose a detailed SAV ecosystem model into a cluster of subsystem models. This allowed us to maintain sufficient ecological detail against the limits of conceptual and computational tractability. The other approach in our modeling program emphasized the role of these plant communities in a larger context of the entire estuarine system including socio-economic considerations. Here, we developed an aggregated version of the SAV ecosystem model (i.e., combined submodels) and placed it into a sequence of cascading connections of influence, which lead from human uses of the estuary for waste disposal, through the SAV ecosystems, to human uses of the estuary as a source of fisheries harvest and other recreational activities. In this paper we describe the structure and the logic behind this dual modeling framework, and we provide a few selected results from these models to indicate briefly the breadth of research questions which were addressed.

SAV ECOSYSTEM MODEL

Ecosystem Modeling Framework

The initial step in developing a simulation model of the SAV ecosystem involved identification of the level of aggregation and essential state variables. Obviously, there are certain misleading consequences of reduced dimensionality such as artifically conferred stability (e.g., Schaffer 1981). However, we have taken cognizance of population time-constants (Goodall 1974, Schaffer 1981), as well as life histories, trophic relations and habitats (Boling et al., 1975) in defining aggregated biological state variables. We have reduced the number of chemical variables (e.g., plant nutrients) by recognizing basic principles of chemical kinetics whereby biochemical rates are determined by a single rate-limiting step or substrate (e.g., Brezonik 1972). In all we defined 37 state variables to be included in this model. There are a few published examples of analytical or simulation models for seagrasses or other submerged macrophytes (Titus et al., 1975, Belyaev et al., 1977, Short 1980, Weber et al., 1981, Verhagen and Nienhuis 1983, Adams et al., 1979). However, all but one of these dealt with plant production only, and none contained more than 8 state variables. It was decided that this many (37) variables in one model would produce a virtually unmanagable system of equations, particularly given the necessary high degree of connectivity.

A hierarchical scheme of six subsystem models was used to define the SAV ecosystem (fig. 2). Other modelers have similarly utilized hierarchical approaches (Goodall 1974, Overton 1975, McIntire and Colby 1978), and various methods have been suggested for interconnecting subsystem models. We elected to simulate subsystem models independently and then to use outputs of each as

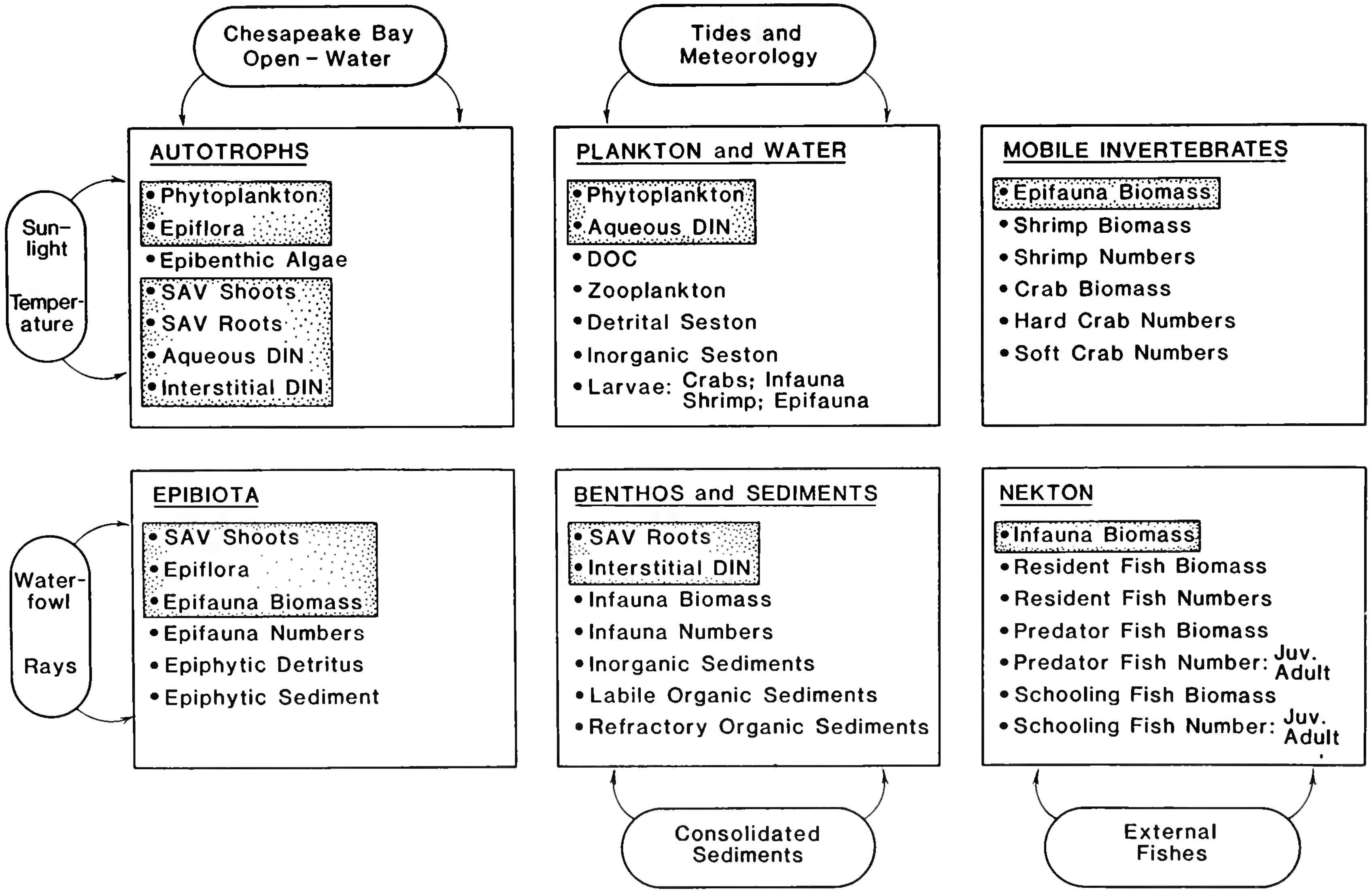

Figure 2. Definition of six ecosystem submodels in terms of state variables used in each, and external forces acting on them. Variables listed in shaded boxes reappear in one or more other submodels.

inputs to the others. This procedure is necessarily iterative, where the modeler serves as an interfacing mechanism. While it can be tedious, this approach has the flexibility to allow the modeler's intuition to function freely. Theoretically, if each subsystem model is well-calibrated, the interconnections among them would match.

Subsystems were defined so as to maximize internal interactions and minimize connections with external variables (Simon 1973). The resulting Subsystems are (fig. 2): (1) the Autotrophs which compete for light and nutrients, (2) the Epibiota which inhabit leaf surfaces of the dominant autotroph (SAV), (3) the Water, with its suspended and dissolved substances, (4) the Benthos and the sediments supporting them, (5) the Large Mobile Invertebrates, and (6) the Nekton which feed on production from other subsystems. The sum of the state variables contained in all 6 subsystem models is 45; however, 8 of these occur in more than one subsystem. This redundancy of variables means that the state spaces overlap, and it further insures consistency in the overall behavior of the SAV ecosystem model and its subsystem simulations. It is apparent in figure 2 that the number of common variables (in shaded boxes) decrease away from the Autotrophs, suggesting a reduction in the number of direct interactions among variables at higher trophic levels.

These models were designed to represent a unit area of water and sediment in an SAV ecosystem with spatial averaging implied. Both carbon (C) and nitrogen (N) are modeled in this scheme, where N is conserved within the model during all transactions while C is transformed (with CO_2 making the difference) as needed according to prescribed C:N ratios for all biological state variables. Flows of both C (and associated free energy) and N are crucial to the behavior of this ecosystem. However, to include both with completely conserved materials would require nearly twice the number of variables. Other chemical factors such as oxygen and phosphorus are assumed to be nonlimiting to the ecosystem's behavior, and those are thus omitted. Several previous modeling studies have explicitly considered both C and N (e.g., Walsh 1975 a,b, Kremer and Nixon 1977, Hopkinson and Day 1977, Najarian and Taft 1981). However, most ecosystem models have been confined to tracing the flows of either carbon (energy) or nutrients but not both (Najarian and Harleman 1977, Wetzel and Wiegert 1983).

The mathematical structure of this model uses nonlinear, first-order differential equations simulated by finite difference techniques. There is one equation for each state variable, and each term in an equation represents an interaction between variables. In the following two sections of this paper, we report some salient aspects of two of these subsystem models, the Autotrophs and the Nekton. These subsystems are at opposite ends of the ecological trophic chain, one (Autotrophs) being more externally regulated (by Sunlight, nutrient inputs, etc.), while nekton dynamics result more directly from production at lower trophic levels.

The Autotroph Subsystem Model

A major objective in developing the Autotroph subsystem model was to examine the consequence of changing patterns of turbidity, nutrients and grazing on the competitive balance among the primary producers in an SAV community. This model is depicted in figure 3, where phytoplankton, epiflora, SAV and benthic micro-algae all compete for limited availabilities of light and nutrients. Competition for light is direct via shading, while competition

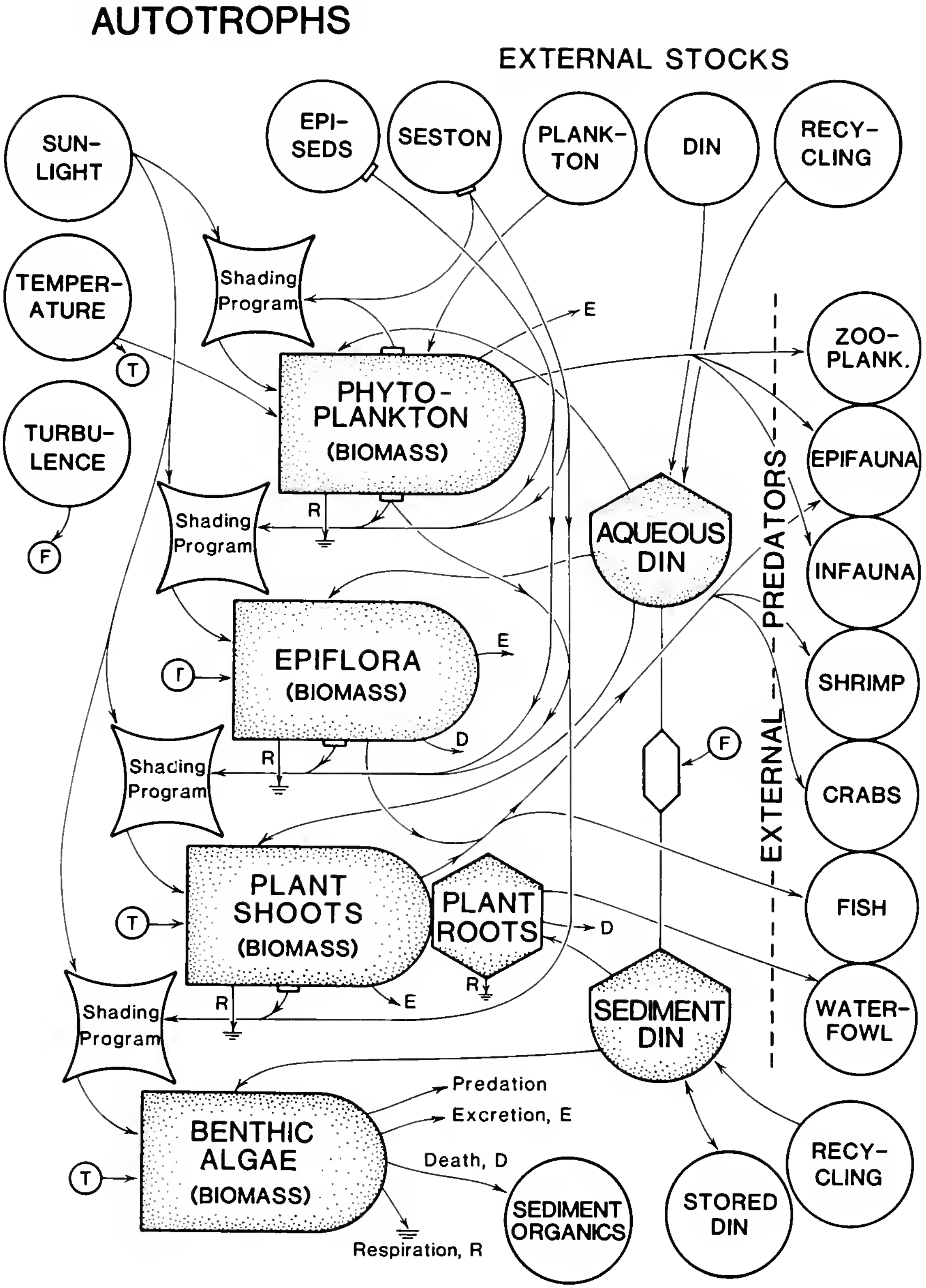

Figure 3. Autotroph ecosystem submodel presented in terms of state variables (shaded symbols) and interactions (lines with arrows) among variables and with external forcing functions (circles). Symbols are based on Odum (1971).

occurs for two sources of dissolved nutrients through periodic depletion of supplies, and only the rooted vascular plants have direct access to both nutrient sources. The 7 state variables here are connected to numerous external factors, both those in another subsystem and those entirely external to the SAV community.

The nature of mathematical formulations used can be illustrated with the primary production term in the SAV growth equation:

$$P = [C/N][ATTEN][LKIN][TEMP][NKIN][LAI]. \tag{1}$$

Here, SAV production (P) is a multiplicative function of 6 auxilliary variables: [C/N], the nitrogen-to-carbon conversion; [ATTEN], the light attenuation relation; [LKIN], the photosynthesis-irradiance function; [TEMP], the temperature kinetics; [NKIN], the nitrogen uptake relation; and [LAI], an index of leaf area representing the ability to absorb photons. Light attenuation follows a simple Beers-Lambert relation with various materials contributing to the effect (e.g., Parsons et al., 1979):

$$I_z = I_o e^{-kz} \tag{2}$$

where I_z and I_o are light levels at depth, z, and at water surface, respectively. The attenuation coefficient k is taken as the sum of individual k's for seston, epiphytic material and SAV leaves, where each k is a linear function of the amount of material per m^2 with the overall intercept attributable to dissolved substances and the water itself. The photosynthesis-irradiance relation is approximated by a rectangular hyperbola (Parsons et al., 1979):

$$P = P_m \left[\frac{I_z}{K_i + I_z}\right], \tag{3}$$

where P_m is the maximum photosynthesis possible, and K_i is the light level at 0.5 P_m. Data for all of the light relations were obtained from experiments in our laboratory (Kemp et al., 1981). The temperature (T) function used is a simple Arrhenius relation,

$$TEMP = e^{-(K_t/T)}. \tag{4}$$

Values for K_t were obtained from the literature for related species (Titus and Adams 1979, Barko and Smart 1981). A higher order equation (Johnson et al., 1974) which accounts for temperature stress via protein denaturation at elevated T was used in some versions of the model.

Little information was available concerning the appropriate algebraic expression for describing SAV nitrogen uptake (V) from two sources (water column and sediment pore-water). We chose a formulation analogous to the Michaelis-Menten relation, and assuming a single maximum uptake rate ($V_m = f(P_m)$) but differing half-saturation constants

$$V = V_m \left[\frac{N_a+k^*N_b}{K_s+(N_a+k^*N_b)}\right] , \qquad (5)$$

where N_a and N_b are aqueous and benthic nitrogen concentrations of dissolved inorganic nitrogen (mostly NH_4^+), and K_s is the half-saturation constant for uptake of N_a and (K_s/k^*) is the half-saturation for N_b. Again, these coefficients were calculated from our own experimental data, primarily for the SAV species, Potamogeton perfoliatus (Kemp et al., 1981). Similar expressions were used to describe light, nutrient and temperature interactions in primary production of other autotrophic groups.

The basic behavior of this model is illustrated in the calibration output (fig. 4). The close correspondence between model and field data is also apparent here. For clarity the variances associated with these data are not given. However, the model trace is generally well within the 95 percent confidence interval for field observations. Subsequently, the veracity of this model was compared to a second independent data set, and again good agreement was obtained between model and measurements (Kemp et al., 1983b). The effects of nutrient additions to this model system were also very similar to those observed in large experimental ponds, and the model was used to extrapolate results from these systems to actual estuarine conditions (Kemp et al., 1983 a,b). It is interesting to note the slight asynchrony of peak summer abundance for these 4 components, indicating some temporal separation of niches to minimize competition toward system homeostasis (Lewis 1980).

The Nekton Subsystem Model

The hypothesis to be investigated with the Nekton model was that changes in SAV abundance would influence total fish abundance and would shift the balance among various trophic and habitat fish groups. This model is important in the overall simulation framework because nekton provide a crucial feedback control for the other ecosystem submodels (fig. 2) and because its output provides a principal linkage to management concerns.

The general organization of the Nekton submodel is described in figure 5, where categories of fish (including total biomass, and adult or juvenile numerical abundance for each) compete for various food items, an important one of which, benthic infauna, is explicitly included in this model. Other food sources are external to the model, and most are variables in other subsystems. The nekton system here is defined by 9 state variables in 4 categories. There are 3 variables within 2 fish groups, and 2 within the third, "Resident Fish". There is some direct predator-prey interaction among the 3 fish groups; however, competition for limited foods also represents an indirect mechanism of interaction. Model fish groups are connected to external fish populations, with immigration and emigration controlled by temperature cues and density dependent factors.

The 3 categories of fish are functional classifications defined on the basis of similar habitat, trophic relations and life histories. The ecological units were developed as a compromise allowing aggregation but retaining some of the mechanistic relationships which characterize populations in nature (namely where they live, what they eat, and how and when they reproduce). This condensation

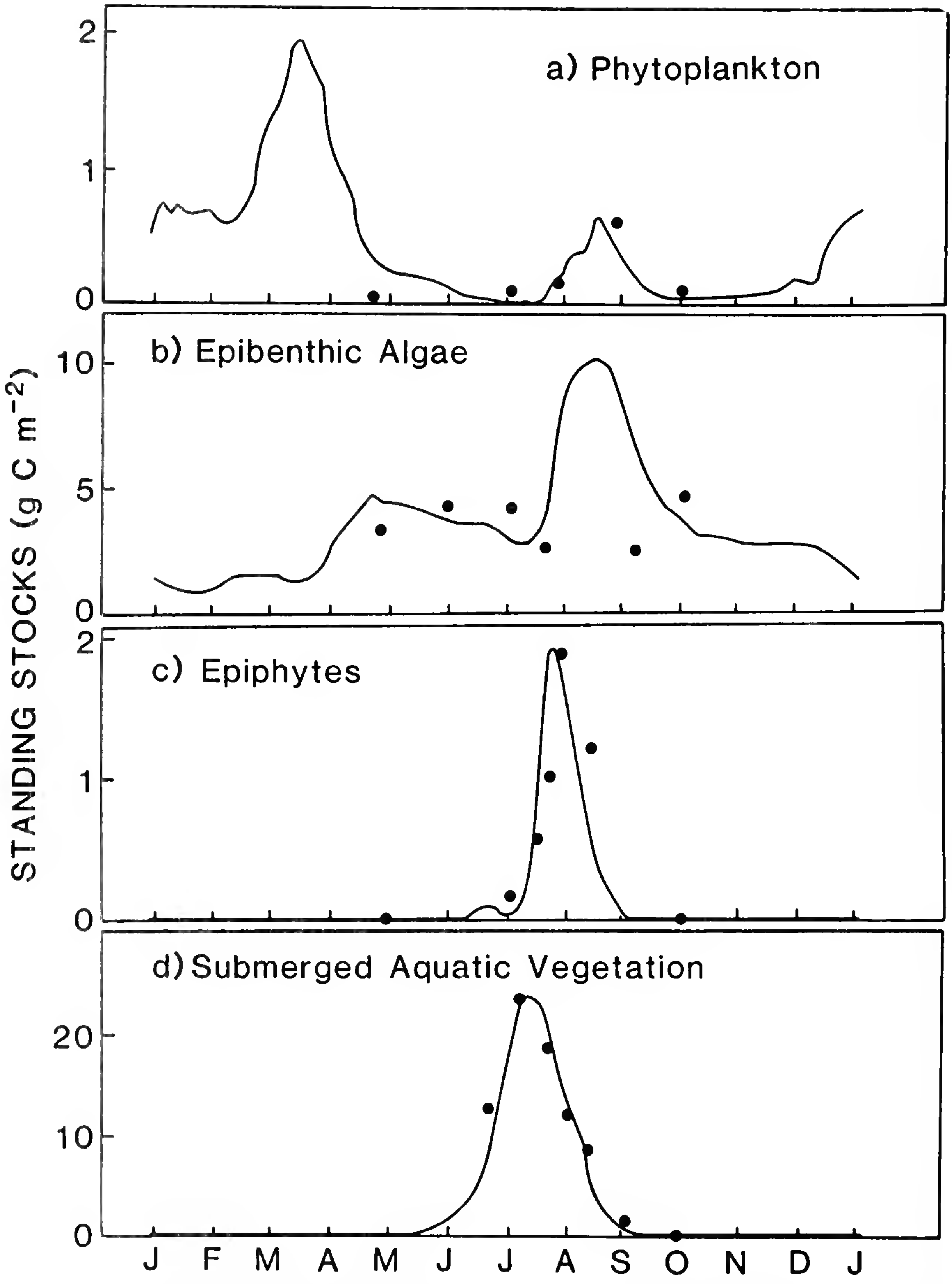

Figure 4. Calibration simulations from autotroph ecosystem submodel (fig. 3) for standing stocks of a) phytoplankton, b) epibenthic algae, c) epiphytes, and d) submerged aquatic vegetation. Data are taken from Kemp et al. (1981).

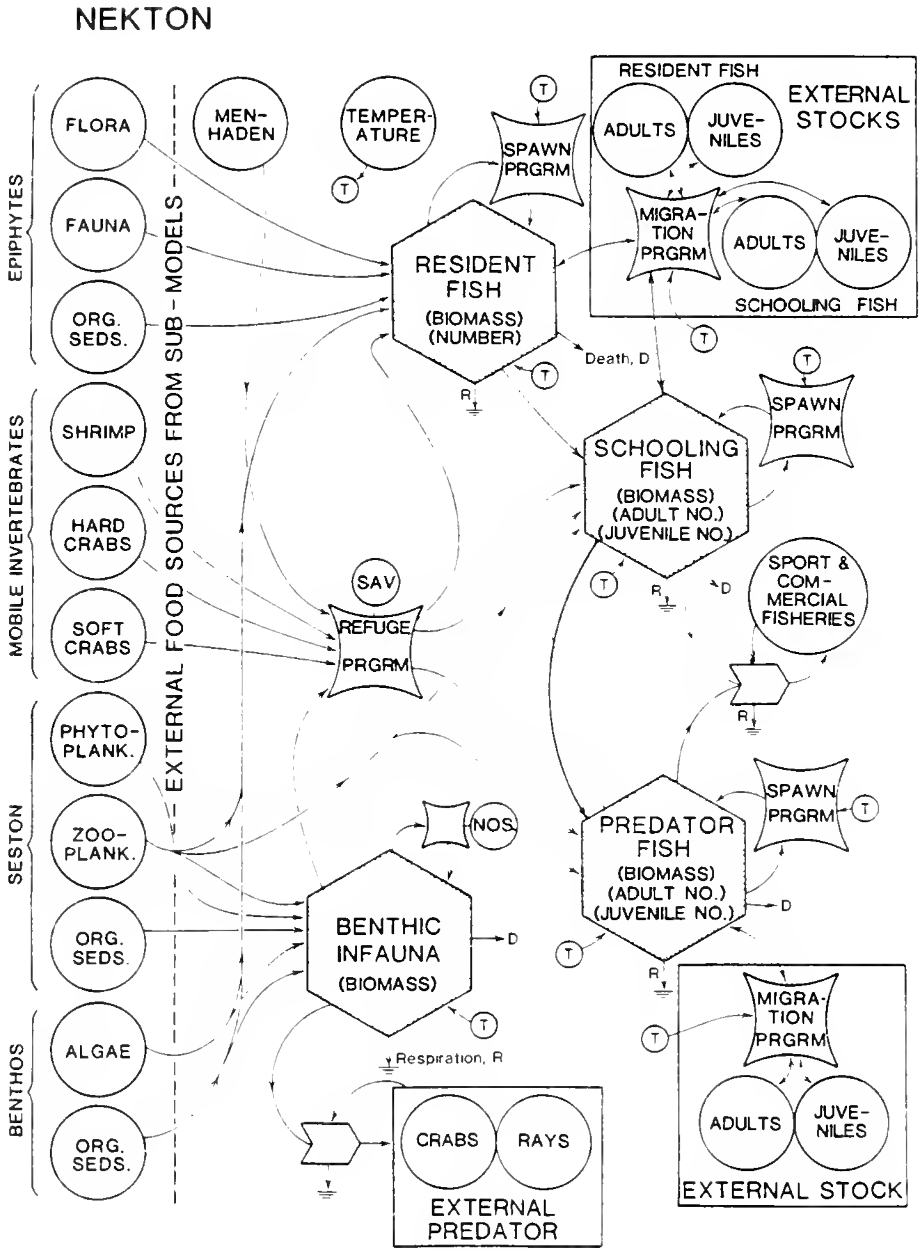

Figure 5. Nekton ecosystem submodel presented in terms of state variables (shaded symbols) and interactions (lines with arrows) among variables and with external forcing functions (circles). Symbols are based on Odum (1971).

process is a necessary abstraction of ecological modeling, representing an attempt to balance among criteria of realism, precision and generality (Levins 1966). These fish groups are what Boling et al., (1975) referred to as "para-species", defined consistent with modeling objectives. It is fortunate that the fish assemblages in Chesapeake Bay's brackish SAV ecosystems are of relatively low species diversity. In fact, 80-95 percent of the fish biomass in each of the 3 categories (defined above) is comprised by 1-3 principal species (Lubbers et al., 1981) with similar functional characteristics. The major "Resident Fish" are Fundulus spp., Lucania parva and Apeltes quadracus; the most important "Schooling Fish" are Anchoa mitchilli and Menidia spp.; "Predatory Fish" are dominated by Pomatomis saltatrix and Morone americana.

The elements of nekton life cycles are included in the model by utilizing special subroutines for spawning, recruitment and migration. Recruitment from juvenile to adult age (size) classes is also represented in the model structure for Schooling and Predatory Fish in terms of juvenile and adult numerical abundance. Thus, issues of stock-recruitment and density-dependence can be treated in the model, albeit at a coarse-grained level. The use of numbers and biomass as distinct, but coupled, state variables allows considerable flexibility and structural condensation while maintaining realistic model behavior. This approach, which was used by Steele (1974) for zooplankton in his model of the North Sea pelagic ecosystem, provides a means for tracking both energy flow (as biomass) and population information (as numbers). Predator-prey relations are often best described in terms of numerical abundance, while metabolic processes are more a function of biomass. Traditional population models consider numbers only (in separate age groups), while most ecosystem models utilize biomass only. This model attempts to combine the strengths of both.

The mathematical form of equations used in the model can be illustrated in terms of Schooling Fish biomass and adult numbers. The temporal rate-of-change for biomass ($\dot{Q}_{35}$) is

$$\dot{Q}_{35} = \text{assimilation} - \text{predation mortality} - \text{fishing mortality} - \text{spawning effort} - \text{respiration} + \text{immigration} - \text{emigration}, \qquad (6)$$

while for adult numbers ($\dot{Q}_{36}$) the rate-of-change is

$$\dot{Q}_{36} = \text{recruitment from juveniles} - \text{predation mortality} - \text{fishing mortality} + \text{immigration} - \text{emigration}. \qquad (7)$$

Overall, the terms in the biomass equation utilize an interplay between variables of biomass and number, where, for example, assimilation (a fixed fraction of consumption) and mortality involve biomass and numbers for both prey and predator, while the respiration term involves only biomass (Q_{35}). The terms in Eq. 7 are (with the exception of recruitment) derived from those in Eq. 6, with the reciprocal of average size used to convert from biomass units to numbers.

The formulation for predation utilizes concepts of threshold densities (e.g., Wiegert 1975), size-selective feeding (e.g., Brooks and Dodson 1965), other criteria of selectivity (e.g., Ivlev 1961) and refuge provided by SAV structure (Heck and Orth 1980). Predation is taken as the product of the predator activity (PRED) times prey availability (PREY). In the case of adult Schooling Fish,

$$\text{PRED} = k_1\ Q_{35}[\log(L_1 + k_2(Q_{35}/Q_{36})]\ [\exp k_3\ T] \qquad (8)$$

where T is temperature, L_1 is related to minimal feeding rate for small organisms, (Q_{35}/Q_{36}) is average size of predator, and k's are empirical coefficients. Similar expressions are used for predation by juveniles but different prey items are involved. Thus, both juvenile and adult feeding contribute to biomass (Q_{35}), allowing for ontogenic changes in diets (e.g., Carr and Adams 1973).

Prey availability is defined as the product of prey biomas (Q_b), a polynominal function of average prey size, $f(Q_b/Q_n)$, and a prey refuge function created by SAV,

$$\text{PREY} = k_4\ Q_b[f(Q_b/Q_n)]\ [L_2 + \exp(-k_5(Q_p-L_3))] \qquad (9)$$

where L_2 is the maximum refuge offered, L_3 is the lower threshold of plant biomass (Q_p) for incipient refuge effect, and where the availability function cannot exceed unity. The polynominal function of average prey size exhibits a broad central region (20-180% of mean prey size) with reduced availability when prey become very small or very large. Other details of model formulation are described in Kemp et al., (1981).

The general behavior of this model is indicated in the calibration simulation presented in figure 6. Simulated time-course of benthic infaunal biomass follows field observations reasonably close, both in magnitude and timing, although the model shows a slower winter-spring growth in the community than the data would indicate. At this preliminary stage of model development, we can only say that model output is in the right order-of-magnitude, and that certain temporal trends such as abundance of Resident and juvenile Schooling Fish are reasonably consistent with data. Seasonal patterns of biomass are generally skewed too far into the autumn, probably due to problems in the emigration subroutines. Ultimately, it is hoped that this model will help us to understand the way in which competitive shifts among the autotrophic groups influence the relative balance in fish abundance among the 3 groups (fig. 5) which are well down the trophic chain from those primary producers. Model simulations can be used to distinguish the relative importance of habitat (e.g., predatory refuge) versus primary food production in leading to these effects, while such a distinction could not easily be made through field experimentation.

RESOURCE MANAGEMENT MODELING

Management Modeling Framework

Parallel to the detailed ecosystem modeling, we developed a system of resource management models for focusing on the multiple interactions of human activities with resource ecosystems. In general this modeling effort was designed to assist in utilizing scientific knowledge towards balanced and productive management of Chesapeake Bay resources. In contrast to the detailed ecosystem models, this research was intended to assess both the relative importance of factors contributing to the decline in SAV abundance, and the consequences of this decline (in terms of such factors as fish production). The modeling

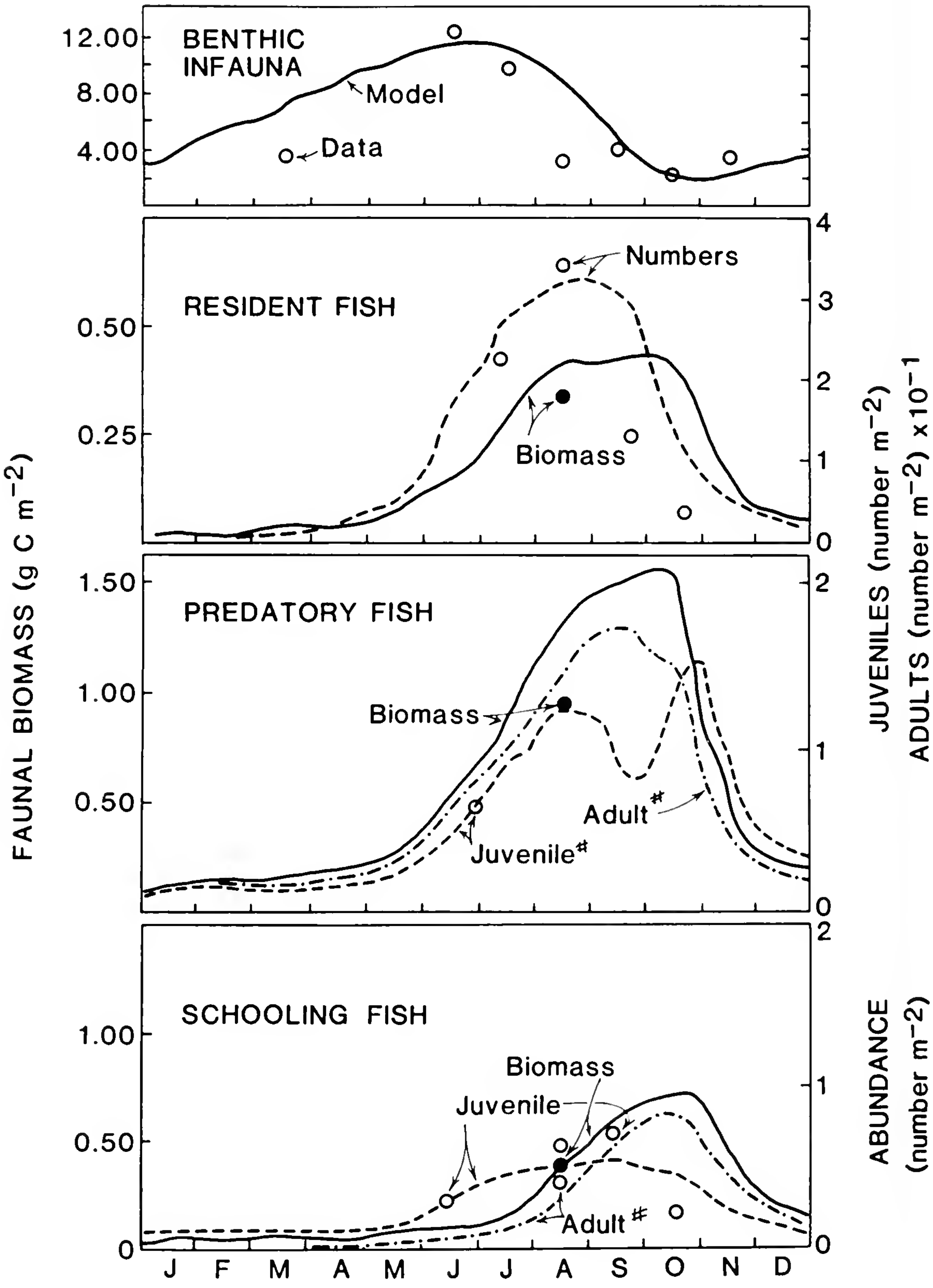

Figure 6. Calibration baseline simulation from nekton ecosystem submodel (fig. 5) for biomass and/or numerical abundance of a) benthic macro-infauna, b) predatory fish, c) schooling fish, and d) resident fish. Data are taken from Kemp et al. (1981).

framework explicitly establishes the interactions between SAV ecosystems and human economic systems. Direct and indirect effects of alternative management scenarios were assessed in terms of economic values and ecological processes. Finally, this framework provided a heuristic format for understanding some principles of resource management.

This scheme is illustrated in figure 7 as a cluster of interconnected models representing the influence of human activities, as modified by physical forces (e.g., rain, sunlight, tides), on SAV ecosystem dynamics which in turn affect resources valued by society. Briefly, meteorological conditions coupled with agricultural practices are shown as inputs to the Watershed Runoff Model (Holton and Yaramanglou 1979) which links the Universal Soil Loss Equation to hydrologic and chemical process models, thereby routing water, nutrients, sediments and herbicides from fields to estuary. A simplified 2-layer "box model" of Estuarine Circulation, based largely on continuity at steady-state (Officer 1980), receives agricultural runoff and sewage nutrient wastes and transports water and materials through the estuary providing an ambient water quality field to which SAV are exposed. These materials, along with direct agricultural runoff, provide inputs to the SAV Ecosystem Management Model (SEMM), the details of which will be discussed in the next section. Outputs from SEMM, including fisheries and recreational activities are input functions to the Resource Economics Model which estimates equivalent economic values associated with these features (Kahn 1981, Boynton et al., 1981).

Depicted on the left side of figure 7, the marginal costs and benefits associated with various economic activities (Land Development, Agricultural Production, and Sewage Treatment) are calculated (Boynton et al., 1981). Also associated with these economic processes are direct or indirect effects on waste loading to the estuary. Resource values are combined with costs and benefits of watershed activities to establish viable Resource and Waste Control Options which managers and citizens consider towards developing resource policies. In general, connections between submodels are undirectional, with feedback occurring only indirectly through the management decision process. For example, materials enter the estuary from the watershed, while the estuary, per se, has little direct influence on watershed activities. In this scheme the modeler serves as the interface between connected submodels, and piece-wise simulations can be performed with no loss of information since there are no direct feedbacks. In other words, the output information from simulations in one submodel are used by the modeler to define input conditions for the next sub-model in the sequence.

SAV Ecosystem Management Model

At the focal point of this resource management framework is the SAV Ecosystem Management Model (SEMM). The SEMM was designed to emphasize interactions between SAV ecosystems and human systems (fig. 8), specifically water quality effects on SAV production and abundance, and the habitat and food-chain factors whereby SAV enhances fish production. The structure of SEMM aggregates much of the complexity which had been emphasized in the SAV ecosystem submodels (e.g., Autotroph and Nekton Models in fig. 3 and 5). Our intention here was to preserve sufficient detail in ecological function so that relevant interactions with socio-economic activities could also be included without losing conceptual and computational control. Sensitivity analyses performed for the ecosystem submodels provided some guidance on strategies of aggregation, wherein crucial

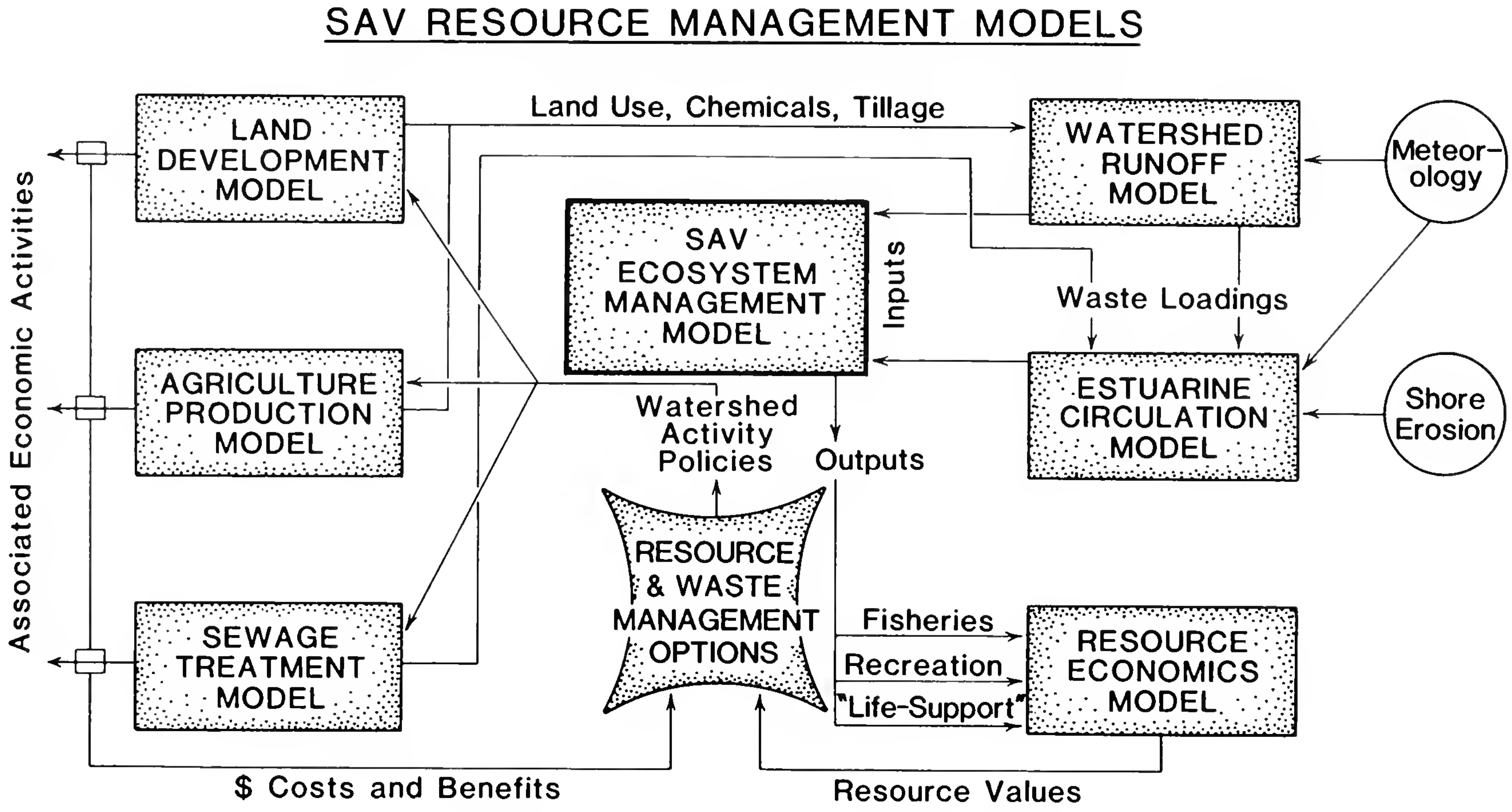

Figure 7. Schematic diagram of interrelations among various modeling programs organized toward developing overall management strategies for submerged aquatic vegetation in Chesapeake Bay.

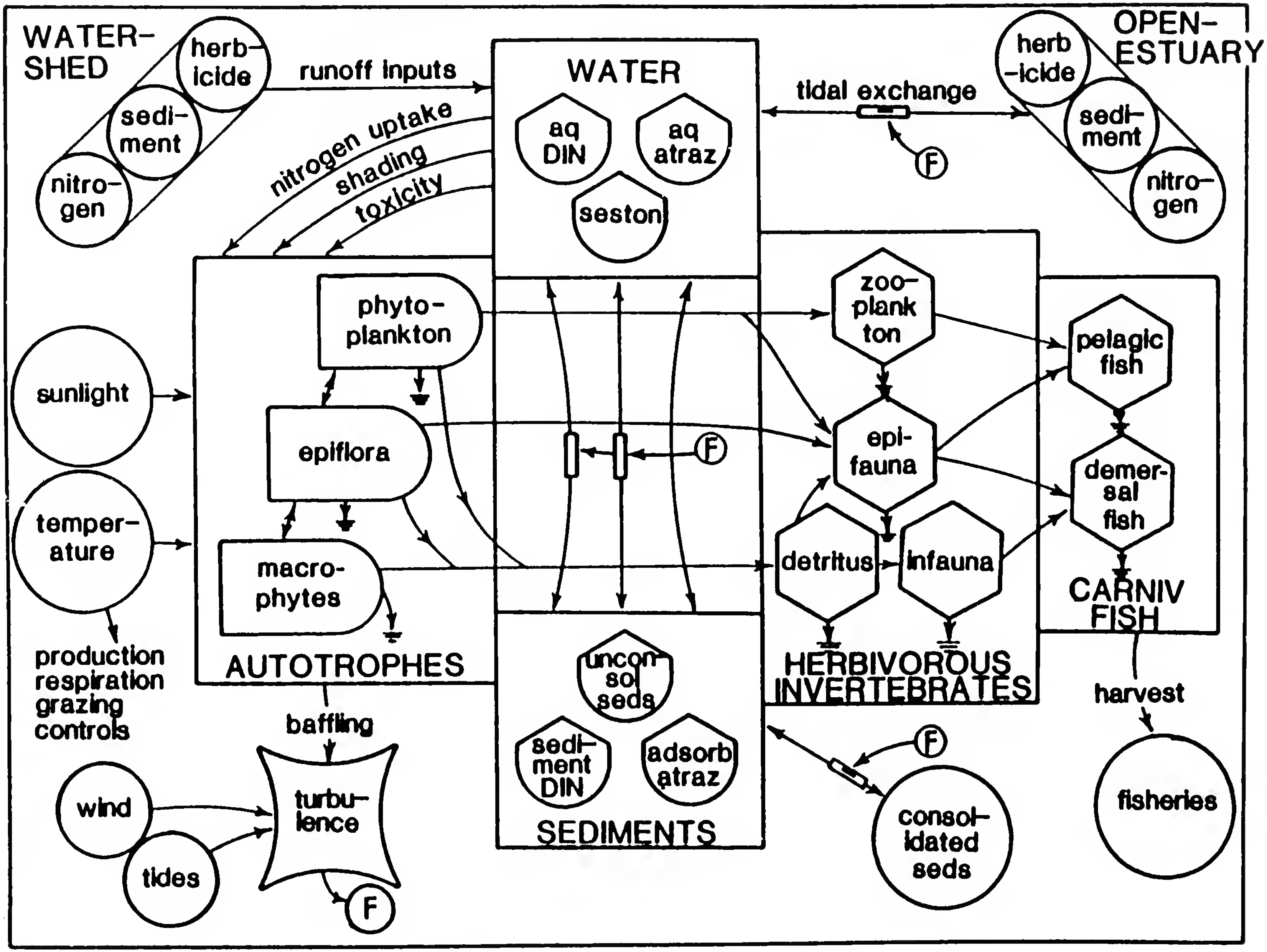

Figure 8. "Submerged Aquatic Vegetation Ecosystem Management Model" presented in terms of state variables (shaded symbols) and generalized interactions (lines with arrows) among variables and with external forcing functions (circles). Symbols are based on Odum (1971).

variables and pathways were preserved, and less sensitive factors were either omitted or combined.

The general structure of SEMM, illustrated in figure 8, is comprised of 15 state variables organized into 5 groups: 1) the Autotrophs or photosynthetic organisms all competing for limited light and nutrient resources; 2) the Sediments and their associated chemistry; 3) the Water with its dissolved nutrients and herbicides, as well as suspended sediments (seston); 4) the Herbivorous Invertebrates at the lower end of the food-chain; and 5) the Carnivorous Fish at the top of the food-chain. These state variables are driven by 11 seasonally varying external forcing functions. SEMM includes 2 new state variables (aqueous and adsorbed herbicide, atrazine) not occurring in the ecological submodels but included here because of the potential importance in resource management. The differential equations which formalize this model are essentially similar to those used in the detailed submodels. However, they tend to be less mechanistic and more linear in form. This is consistent with the concept of increasing linearity of systems with increasing degree of aggregation (e.g., Patten 1975; Odum 1983).

Multiple simulation experiments with SEMM allowed us to consider the relative effects of herbicide, sediment, nutrient loading on SAV production (fig. 9). Here, growth of SAV exhibits little response to changes in herbicide loading from the watershed. Sediment inputs produce a more dramatic effect on SAV; however, it appears that much of the total estuarine sediment loading is derived from natural processes such as shore erosion and is therefore less managable. Nutrient (and in particular nitrogen) loading at low levels causes an enhanced SAV growth, whereas reduced SAV production results from inputs greater than estimated 1960 rates. The reduction in SAV photosynthesis at high nitrogen levels results from enhanced growth of planktonic and epiphytic algae, which effectively reduce light available to SAV.

Management Options and Socio-Economic Tradeoffs

Results of SEMM simulations were integrated into a larger analytical framework in which the socio-economic consequences of several management options were evaluated using the framework presented in figure 7. Preliminary estimates of economic and ecological trade-offs between agriculture and fisheries were made for selected management scenarios. Agricultural costs and estuarine benefits are summarized in table 1 for three such scenarios: 1) Reversion back to 100 percent "conventional" tillage from the current 50/50 split with minimum tillage; 2) 25 percent reduction in fertilization rates; and 3) 10 percent reduction of area in cultivation. We have estimated costs and benefits in economic ("surplus value") terms. In table 1 economic benefits and costs ranged from $20,000 to $900,000 and benefit:cost ratios were always substantially less than 1.0. The impact on agriculture was at least 3 times greater than on the estuarine resources, but given the fact that these analyses are very preliminary approximations, management options with ratios approaching 0.3 should probably be further considered for insights and implications. The relative impact of each cost or benefit on agricultural and fisheries sectors of the regional economy was also calculated as the percentage of "gross sectorial product" for the respective sector. It can be seen that, while the gross economic effects are greater for agriculture than for fisheries, the relative impacts on the regional sectorial economy are generally equal or greater for fisheries (table 1). This latter perspective suggests that while there might be short-term

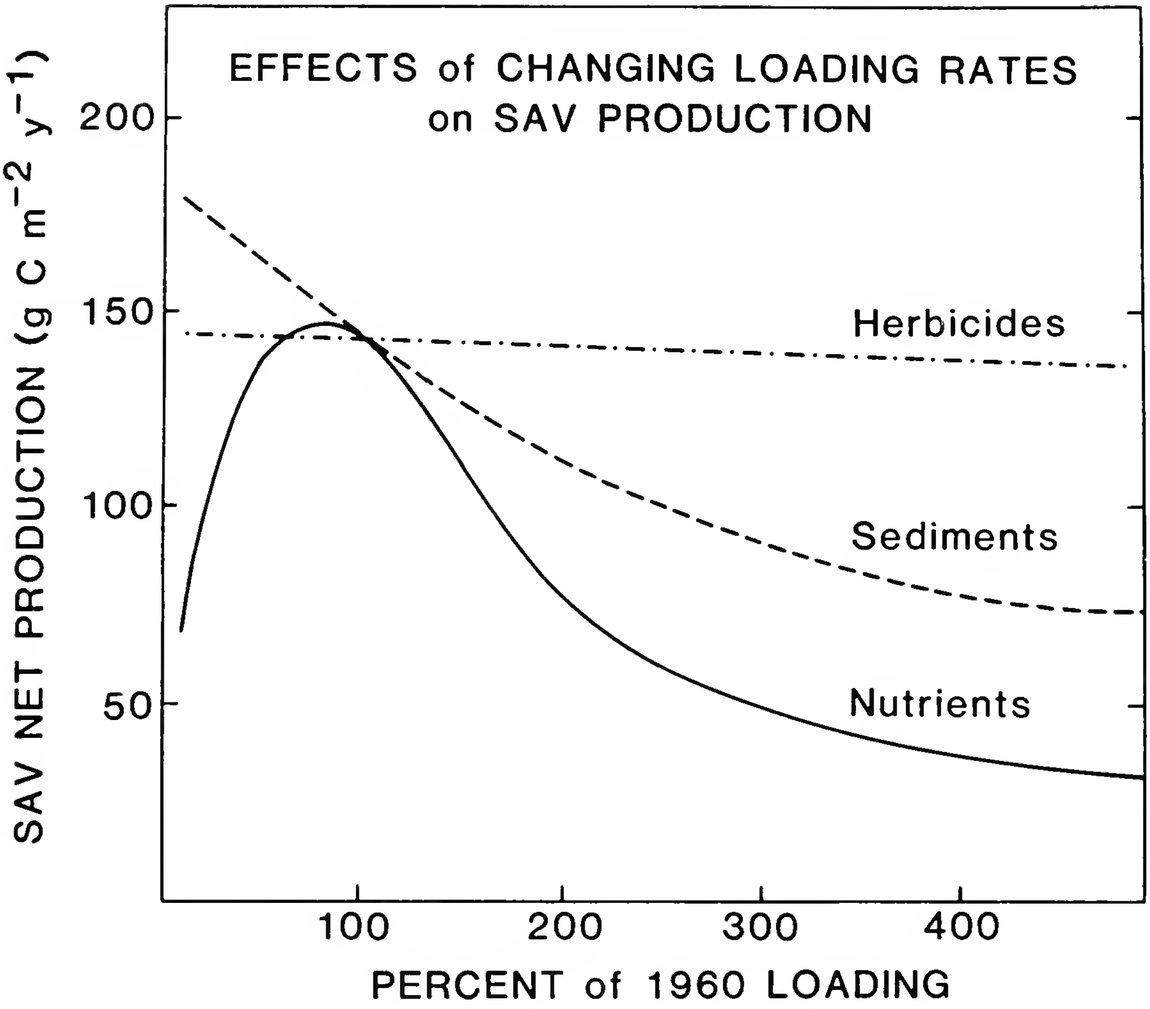

Figure 9. Summary of the effects of changing inputs of a) herbicides, b) sediments, and c) nutrients on annual net production of submerged aquatic vegetation (SAV). Lines represent the distillation of several simulation scenarios from SAV Ecosystem Management Model (fig. 8).

TABLE 1. Trade-offs between agricultural and estuarine resource values for alternative scenarios of watershed management.

ITEM	MANAGEMENT SCENARIOS*		
	100% CONVENTIONAL TILLAGE	25% REDUCTION OF FERTILIZER USE	10% REDUCTION AGRICULTURAL LAND
Agricultural Costs†			
•Cost (1978 US $)	$-30,000	$-900,000	$-200,000
•Relative Impact# (% Gross Sectorial Product)	0.6	18.6	4.1
Fisheries Benefits**			
•Benefit (1978 US $)	$-20,000	$+100,000	$+ 50,000
•Relative Impact# (% Gross Sectorial Product)	3.1	15.4	7.7
Benefit:Cost Ratio	0††	0.11	0.28

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[*] Costs (-) and benefits (+) are given both in terms of 1978 USA dollars, based on surplus values arguments. Numbers refer to estimated impact in Choptank River/Estuary Region, and details of calculations are provided in Boynton et al. (1981).

[†] This represents the loss in agricultural productivity resulting from these alternative watershed management scenarios.

[#] The relative impact of each cost or benefit to the agriculture and fisheries (respectively) sectors of the regional economy was calculated as the percentage of "gross sectorial product" (GSP).

[**] This represents the gain in commercial and recreational fish production resulting from these alternative watershed management scenarios.

[††] Costs were incurred to both sectors under this scenario so that the benefit:cost ratio is zero.

advantage (because of the benefits accrued to the farmer) to allow agricultural wastes to diminish the value of estuarine resources, in the long run it may be considered unfair for the smaller, estuarine-based activities to bear the burden of associated costs.

MODELING IN SCIENCE AND RESOURCE MANAGEMENT

Models in Scientific Research

The role of ecological modeling in scientific research has been much debated over the last decade or so (e.g., Watt 1975, Wiegert 1975). Some of the often mentioned utilities of modeling in environmental research include 1) organizing research objectives and methods (also identifying missing information or poorly understood relationships and formalizing scientific hypotheses), 2) interpolating and extrapolating from a given data base, and 3) testing sensitivities of model variables in relation to their real-world counterparts. Pielou (1981) has recently reviewed and critiqued these and other aspects of ecological modeling, concluding that many of the various uses of models are reasonable but often overstated.

In the course of our SAV research program, we have attempted to utilize models toward a number of these objectives. While the ultimate success in meeting such objectives will necessarily remain unclear, several more pragmatic benefits have emerged from the modeling effort. For example, the conceptual exercise associated with model development has provided a means for a productive dialogue among diverse research specialists to integrate varied information. As such, a model can serve as a format for discussion. Furthermore, models at the conceptual level can help to bridge the dichotomy between descriptive and experimental research objectives. Within the conceptual framework of ecological models the linkage between mechanistic processes and overall ecological structure can be made explicit.

Models in Resource Management

While ecological models have been used effectively in various resource management programs (e.g., Jeffers 1973, Cooper 1975), logistic and personnel problems can hinder this relationship (Mar 1974). We suggest in schematic form (fig. 10) that balanced use of natural resources, such as Chesapeake Bay, might best be achieved through interaction among resource managers, scientists and users, with ecological models serving as an interface between these diverse groups. In this view, management agencies develop alternative strategies for resource allocation (1, numbers refer to figures) by interacting with user groups and by following traditional management practices. While it is generally difficult to project the potential impact of new management scenarios (2) on natural resources or on regional socio-economic interests, managers attempt to select policies (3) which lead to balanced resource uses (10). Various user groups utilize the Bay in response to needs and desires (7), some of which conflict with one another (8), resulting in an identifiable management problem (9). Scientific groups study the Bay to catalogue and understand the dynamic behavior of the ecosystem and its component parts (4). Such research generates scientific data (5), which in itself is of some use to resource managers, but which finds more utility as it emerges into a conceptual framework (models). Such models can be formalized into a simulation modeling scheme (6), which can be used to forecast potential impacts of various management options. In addition,

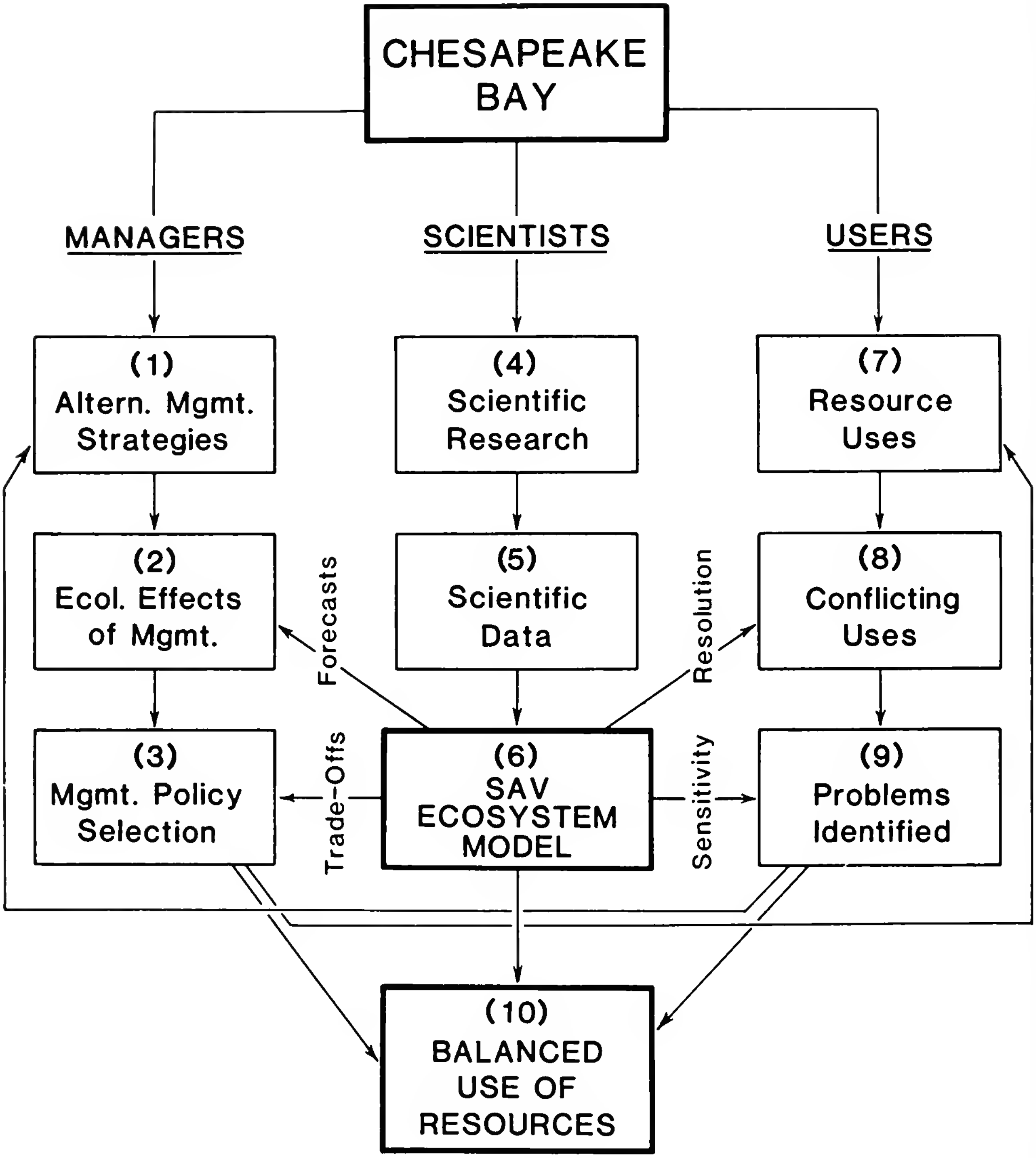

Figure 10. Conceptual flow chart illustrating potential interactions among resource managers, estuarine scientists, and resource users relating to submerged aquatic vegetation (SAV) in Chesapeake Bay. In this hypothetical framework ecosystem modeling plays a pivotal role integrating the activities of all three groups toward balanced uses of SAV and other natural resources.

these models can aid the manager in understanding the explicit and implicit trade-offs involved in each scenario towards judicious selection of an appropriate management policy. The model or suite of models can also be used to aid in identification of conflicting resource uses, thus facilitating the development of a program of balanced resource use.

While this scheme suggests modeling to be a central element in unifying various aspects of resource management, there are limitations which must be recognized as well. In some instances, a mathematical simulation modeling approach, with attendent costs and investments of time, may not be required. Simple calculations may be adequate in some instances, while scientific intuition (which in itself represents a qualitative model) may be sufficient in others. Great care must be taken in developing modeling programs appropriate to the resource questions being addressed. For example, models are sometimes developed at the wrong spatial and temporal scales, or models may be so complex that they tend to obfuscate rather than clarify and objectify resource management options. Nonetheless, if such potential problems are avoided, models can play a very useful role in resource management, particularly if they are viewed as one of many tools which can be utilized in this area of research.

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OPTIMAL EXPLOITATION, BY MUSSEL RAFTS, OF THE RIA DE AROSA, SPAIN: PREDICTIONS OF A FIRST-GENERATION MODEL

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INTRODUCTION

The Ria de Arosa is an estuary on the northwest Atlantic coast of Spain which has been intensively exploited by raft culture of mussels (Mytilus edulis) since the late 1940's (Tenore et al. in press). The Ria Arosa has an area of 250 km^2 with an average depth of 19 m. The yield of mussels averages 86,000 Tm (wet weight) yr^{-1} from approximately 2,000 rafts (Perez and Roman 1979). Each of the rafts is about 19 m square and supports 700-800 ropes, some 8-9 m long. The mussels and associated organisms attach and grow on these ropes. An additional 200 oyster rafts produce currently about 2,500 Tm yr^{-1} (wet weight) of Ostrea edulis.

For the past several years the Ria de Arosa has been the site of an intensive ecological study, performed by a joint team of Spanish and American scientists. The objectives were: 1) to document the major processes occurring in the estuary, 2) to ascertain the factors responsible for the high productivity, 3) to compare the effect of the mussel rafts in Ria de Arosa with a relatively unmodified estuary immediately to the north (Ria de Muros), and 4) to obtain the data necessary to construct a simulation model capable of predicting the optimal number of mussel rafts.

Two years ago we began building a multicompartment ecosystem simulation model of the Ria de Arosa, one which employed realistic (= definable) parameters and functional feedback relationships which included provision for threshold effects (see Wiegert et al. 1975; Wiegert et al. 1981). Thus the model is 'explanatory' in the sense of postulating mechanistic causes for the observed sets of field data. The model is a set of hypotheses, arrived at by induction and utilizing wherever possible ecological, physiological and physical data. The predictions by these (model) hypotheses can be tested by the independent seasonal standing crop measurements from the estuary.

This kind of model, although required if any advance in theory is desired, is far too demanding of data and time to be preferred when only simple prediction is required. Unfortunately, in the present instance, no other course was possible because the data requisite for construction of a multiple-regression type "predictive" model were not available. There is only one estuary being heavily exploited. Furthermore, experimental manipulation of the Ria de Arosa in the sense of increasing or decreasing numbers of rafts and observing the results is not possible because the rafts are individual family enterprises.

The conserved or bookkeeping unit in this first-generation model is elemental N, because this is a major factor limiting primary production in these Spanish estuaries. The overall high productivity of the Ria de Arosa is maintained by periodic upwelling which results in a large volume of N-rich water being moved into the estuary (Tenore et al. in press). The structure of this model plus associated submodels and details of many simulation experiments performed with them will be published elsewhere (Penas and Wiegert, in preparation). Here we present only a brief report of the salient features of this first-generation model and the effect of varying the number of rafts in the estuary.

METHODS

This initial nitrogen flux model AROSAN1 is constructed as a set of nonlinear, discontinuous differential equations representing the fluxes. For all fluxes into a biotic compartment, (e.g., ingestion) feedback control is effected by means of functions which provide for representing 1) a maximum rate of ingestion of N, 2) a refuge of available N, 3) an upper satiation level for available N, 4) a maximum density or carrying capacity for the feeding compartment (this is based solely on space, assuming available N levels are optimal), and 5) a lower response threshold which specifies that density at which negative effects of crowding are first manifested.

In linear fluxes (such as excretion, detritus production, nonpredatory mortality plus egestion, and fluxes between abiotic state variables) simple donor-dependent equations have been used. In the case of the fluxes representing exploitation by man, we have assumed, that the fishing effort (and the harvest) is directly proportional to the density of a resource. Thus a linear donor-dependent equation is reasonable. At least this is so in mussel and oyster fisheries although it is arguable in fishes and crustaceans.

Input of inorganic nitrogen by upwelling is represented by a rate of intrusion and is given a different value for each of the four seasons. Thus, upwelling is averaged over seasonal (3-mo) periods and is not simulated as separate events. This is a serious constraint in this initial model although in general its effect is to exaggerate the potential shellfish production and thus reinforce the conclusions about the potential effect of additional rafts.

The resource feedback terms included in all nonlinear fluxes are, in most cases, of the linear form given in Wiegert (1974). Only for microbial action on the three particulate organic nitrogen state variables did we employ the resource feedback based on the ratio of microbial biomass to substrate, given by Christian and Wetzel (1978). Crowding feedback functions are linear in all cases.

The Program AROSAN1. The FORTRAN program AROSAN1 simulates the dynamics of this system through time. Numerical integration is done with a simple Euler routine and the iteration interval is 0.1 day. By means of an arithmetic IF statement, in one out of 37 iteration loops, the standing stocks of the 22 state variables are stored in a data array. At the end of the simulation, this data array is transferred to an external data file to be plotted. This array then contains the variation of standing stocks and fluxes by means of 99 values through the year of simulation.

THE MODEL

Twenty-two state variables, five abiotic and 17 biotic, were defined (table 1) based on their position in the trophic web of the ecosystem. In some cases, the state variables represent taxonomic units rather than trophic ones. This is so in epibenthic fauna (Echinoderms, Crustaceans, Fishes), where trophic differentiation is quite troublesome, for most important groups are omnivorous and, in some cases, the data were available only for taxonomic units. Two species cultured on rafts (mussel and oyster) are represented as separate state variables, due to their high biomass, their importance in the trophic web, and our interest in simulating the variations of their yields, and the effects

Table 1. List of state variables in model AROSANI, Ria de Arosa.

1. *INORGANIC NITROGEN. Includes the three main compounds taken up by primary producers: ammonia, nitrite and nitrate.
2. PHYTOPLANKTON.
3. MACROPHYTES. Microalgae in intertidal and subtidal rocky substrates, and mussel rafts.
4. BENTHIC MICROFLORA.
5. MUSSEL. *Mytilus edulis*, cultured on rafts.
6. OYSTER. *Ostrea edulis*, cultured on rafts.
7. OTHER BIVALVES. Mainly clams (*Tapes decusata*) and cockles (*Cardium edule*) in the intertidal and subtidal sandy flats.
8. ZOOPLANKTON.
9. PELAGIC FISH. Dominated by the zooplankton-feeder, *Aterina presbiter*.
10. HERBIVORE RAFT EPIFAUNA. Filter-feeders living (and competing) with mussels on the ropes, mainly a sea cucumber (*Cucumaria normanii*) and tunicates (*Ciona intestinalis* and *Phallusia mamillata*).
11. CARNIVORE RAFT EPIFAUNA. Dominated by an asteroid, *Asterias rubens*.
12. DETRITIVORE RAFT EPIFAUNA. Dominated by a small decapod, *Pisidia longicornis* and a lugworm, *Arenicola ecaudata*.
13. MICROBES. Heterotrophic forms (both aerobic and anaerobic).
14. MEIOFAUNA. Nematodes, and minor groups.
15. INFAUNA. Mainly Polychaetes.
16. EPIBENTHIC ECHINODERMS.
17. EPIBENTHIC CRUSTACEANS.
18. DEMERSAL FISHES.
19. *D.O.M. Organic nitrogen in dissolved organic matter.
20. *P.O.M. Organic nitrogen in suspended particulate organic matter.
21. *DETRITUS ON MUSSEL ROPES. Organic matter in the detritus (mainly from mussel feces) on the ropes.
22. *ORGANIC MATTER IN SEDIMENTS. Organic nitrogen to an average depth of 16 cm of "active" (nitrogen cycling) sediment.

*Indicates an abiotic variable

of their cultivation on the whole ecosystem.

The standing stocks are expressed in grams of nitrogen m^{-2}, as an average for the whole system.

Besides the state variables listed on the next page, some other nitrogen pools are considered only as sources or sinks. These are oceanic water (source of inorganic nitrogen, sink for dissolved and particulate organic nitrogen), dissolved N_2 (source/sink by fixation/denitrification), and man (sink by fishing).

The flow diagram (fig. 1) shows the main pathways considered in nitrogen transfer and transformation. Fluxes are expressed in grams nitrogen m^{-2} day^{-1}. Most excretion and detritus production (egestion plus nonpredatory mortality) fluxes are omitted to make the diagram clearer. Because of their quantitative importance, fluxes of excretion and detritus production by mussels have been retained in the diagram.

The main pathways of nitrogen transfer are: the inorganic N rate of supply (by upwelling, mineralization, and mussel excretion), inorganic N uptake by seaweeds and phytoplankton, and secondary production by zooplankton and mussels. This secondary production has been strongly affected by extensive mussel culture. Formerly, phytoplankton production gave rise to a relatively long and complex pelagic food chain. This has been "short circuited" by the mussels. This fact, along with the other natural short circuit, that of the seaweed production (larger than that of phytoplankton and very lightly grazed) produced a shortening of the initial food chain to one with a large production of detritus. From this arise several questions that the model was constructed to answer.

1) Is the system capable of recycling this much detritus or will it accumulate in the sediments?

2) Can the system support a much larger mussel and oyster culture taking into account the food available and question 1? The important factors affecting the answers to these two questions are, besides nitrogen itself:

1) Light limits production in winter; its influence is simulated by seasonal changes in the maximum rates of uptake by the photosynthetic state variables.

2) Oxygen concentration affects sediment transfers by limiting production by meiofauna and infauna. These bioturbators enhance microbial colonization of organic matter and, thus, organic nitrogen recycling.

3) The saturation threshold of microbial uptake of organic nitrogen in sediments is low because once a certain level of O_2 depletion is achieved (in relation to organic matter supply), the realized rate of uptake of organic nitrogen by microbes cannot increase, substrate availability notwithstanding.

4) The carrying capacity of mussels determines the number of mussel rafts.

Because most of the data available on the ecology of the Ria de Arosa were in units of biomass or carbon, conversion factors have had to be used. For animals, Vinogradov's (1953) average of 7.6 percent nitrogen has been used. In the case of phytoplankton, two different C/N ratios were used: 6.0 for blooming phytoplankton, and 10.0 for slow-growing phytoplankton (estimated from Donaghay et al. 1977).

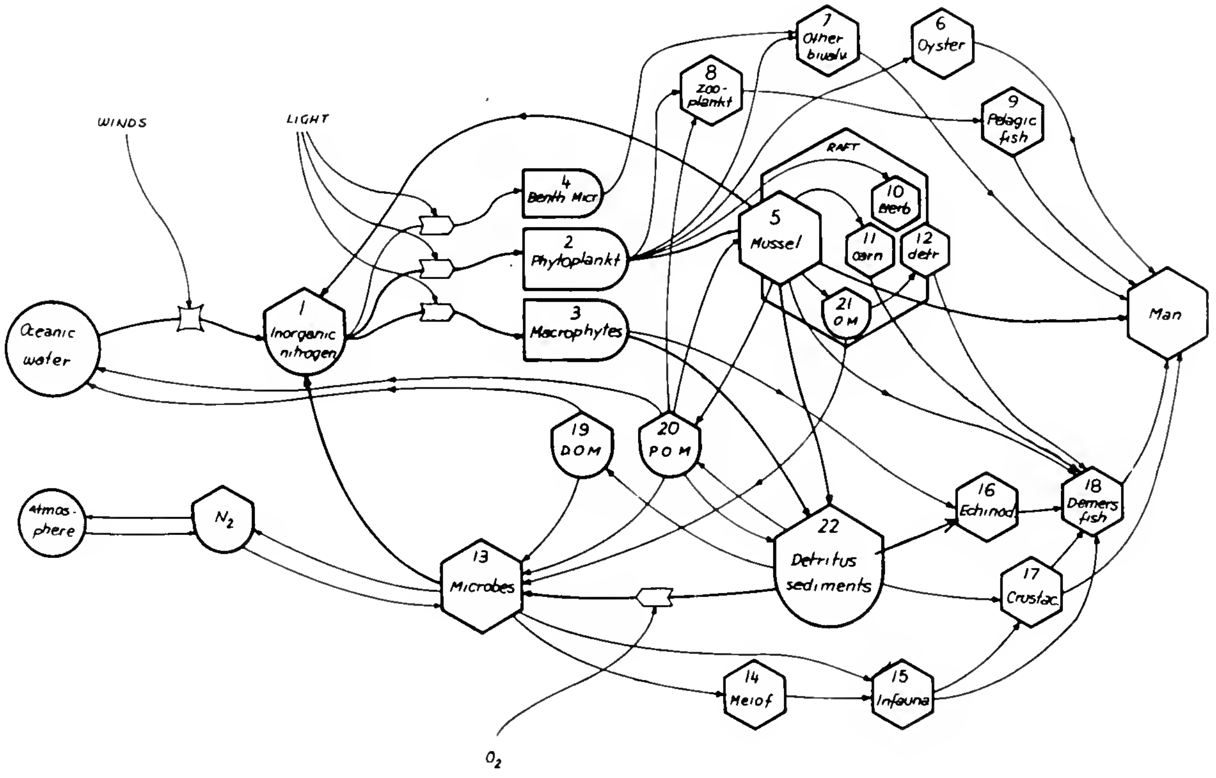

Figure 1. Nitrogen flow diagram of the Ria de Arosa and the ecosystems.

RESULTS AND DISCUSSION

Simulations with AROSAN1 produced the following results when the carrying capabity for mussels was set to equal the current number (2,000) of rafts.

Phytoplankton (fig. 2) and zooplankton (fig. 3) show standing stock values and an annual pattern of variation in agreement with data reported by Tenore and Gonzales (1975). The simulated primary production by macrophytes (fig. 4) tracks the annual pattern found by Fuentes (unpublished). Increasing the number of mussel rafts depresses the standing stock of phytoplankton, and because of both direct ingestion and reduction of their food, that of zooplankton as well. Macrophytes, because they grow on the ropes of mussels, increase directly with increase in number of rafts. We emphasize that the model was constructed from experimental and literature data on rates. Thus the measured standing stocks cited above are independent of the construction of the model and are available as a first test of the models' predictive ability.

Varying the carrying capacity threshold for mussels in order to simulate a change in number of rafts produces a decreasing yield of mussels per raft as the raft number increased (fig. 5). This is due to the reduction in standing stock of phytoplankton and zooplankton. Mussel production is predicted to reach an asymptote of approximately 105,000 Tm (wet weight) yr^{-1}.

The increased standing stock of mussels does, however, have an impact on the recycling of N between the sediments and the water. The model predicts (fig. 6) an accumulation of N in the sediments as the number of rafts increases. This is a consequence of the assumption that the saturation threshold for the microbial uptake of the substrate is readily reached and surpassed by the tremendous amount of detritus deposed by the increasing numbers of mussels. The value of that threshold also takes into account the influence of bioturbation by meiofauna and infauna that are strongly limited by the oxygen depletion accompanying the increased rain of detritus onto the bottom.

The model also predicts a pronounced negative impact of mussels on the productivity of oysters. This seems to be due to a difference in the refuge response thresholds for these two species with respect to a shared food. This is an intriguing and wholly unexpected result from the model. If it is supported by further field and modeling work, it could form the basis for some very interesting simulations involving the socio-economics of oyster and mussel production and pricing and lead to a better monetary return from this valuable marine resource, the productive Ria de Arosa. This is the focus of our continuing work with this model.

CONCLUSIONS

A crude first-generation ecosystem model of the Ria de Arosa simulates annual patterns for the major biotic components that are in agreement with field data not used in the construction of the model. The predicted response to an increase of mussel rafts over the current 2,000 or so in the estuary was:

1) a decreased return of mussels per raft

2) an increased and accelerating retention of N in the sediments

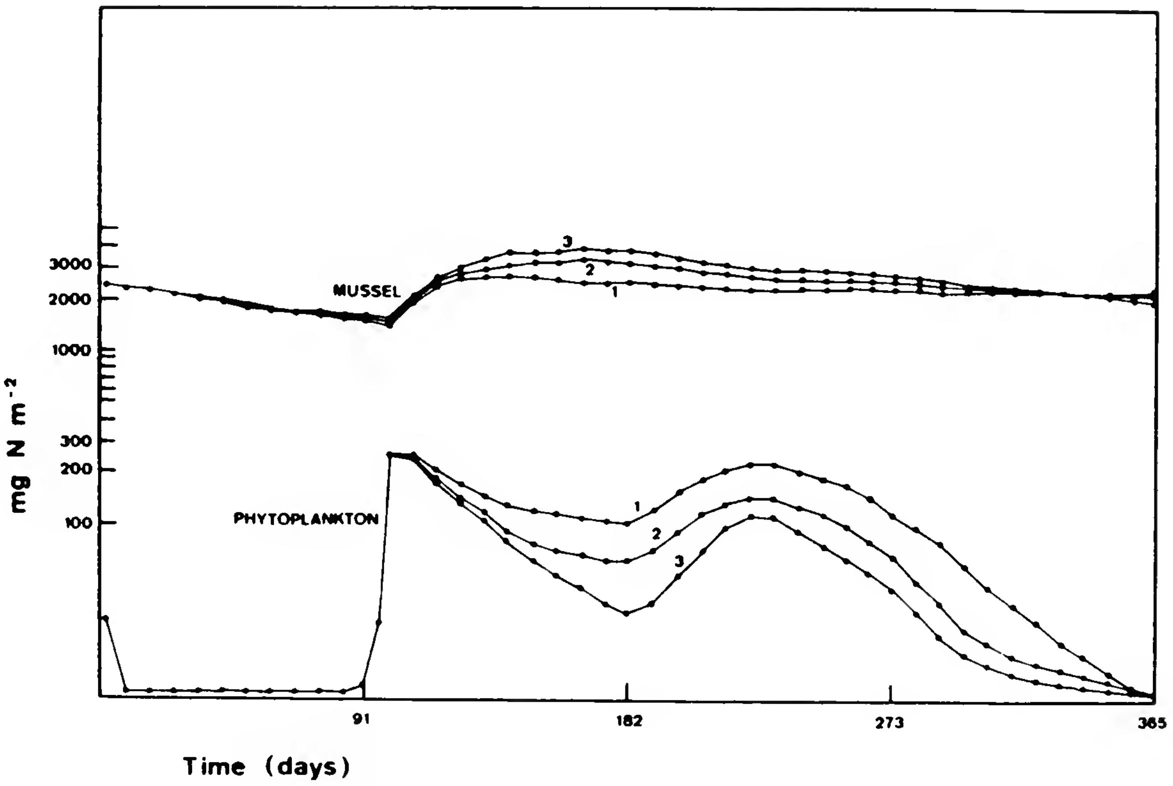

Figure 2. Variation in standing stocks of phytoplankton and mussel through one year of simulation. Curves #1 correspond to the current situation (around 2,000 mussel rafts). Curves #2 & 3 correspond to increases to 2,600 and 3,200 rafts respectively.

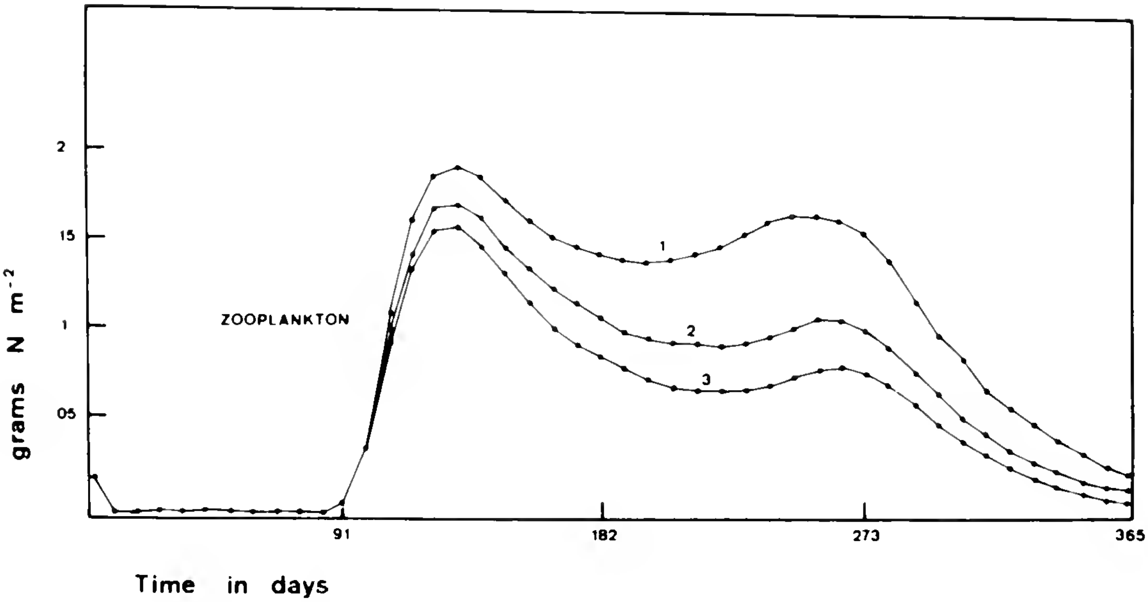

Figure 3. Variation in standing stock of zooplankton through one year of simulation. Curve #1, 2,000 rafts; curves #2 & 3, 2,600 and 3,200 rafts respectively.

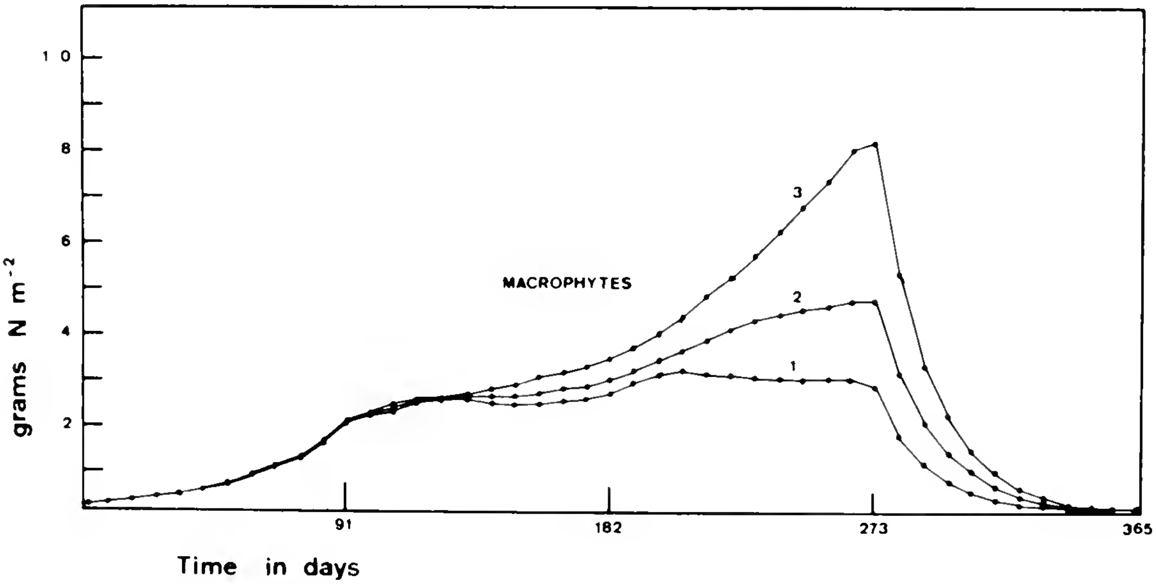

Figure 4. Variation in standing stock of macrophytes through one year of simulation. Curve #1, 2,000 rafts; curves #2 & 3, 2,600 and 3,200 rafts respectively.

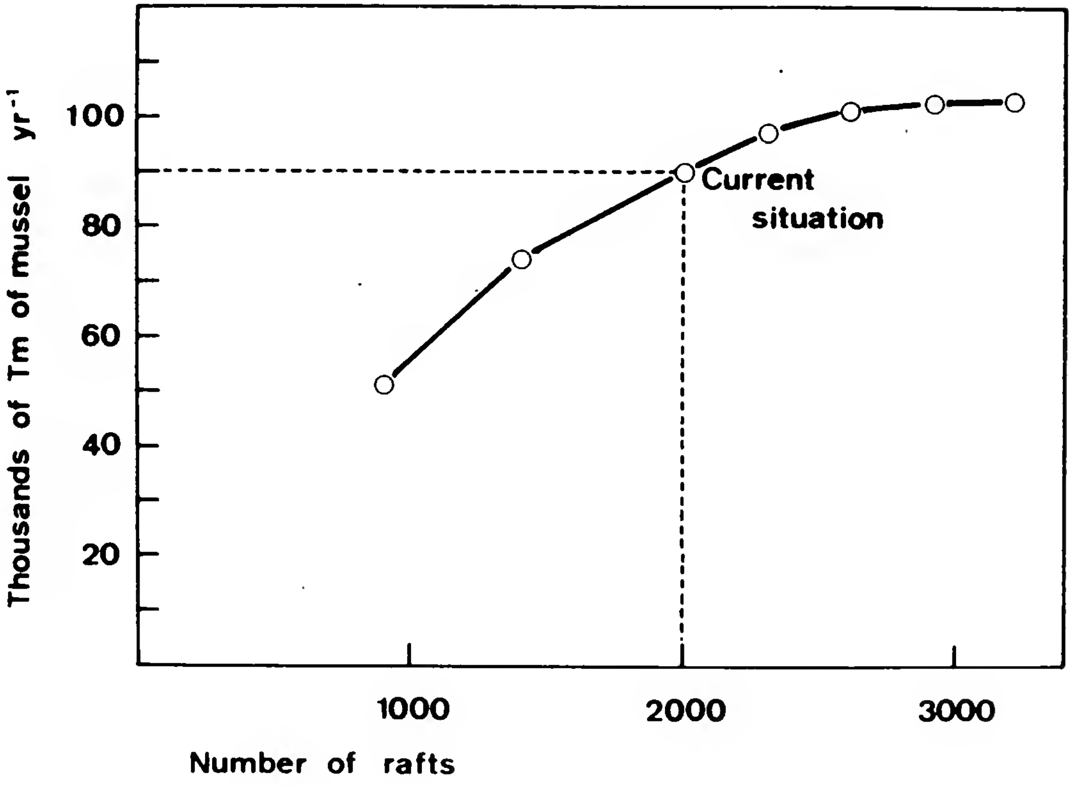

Figure 5. Annual yield of mussel (*Mytilus edulis*) (in Tm of wet weight) for a varying number of rafts.

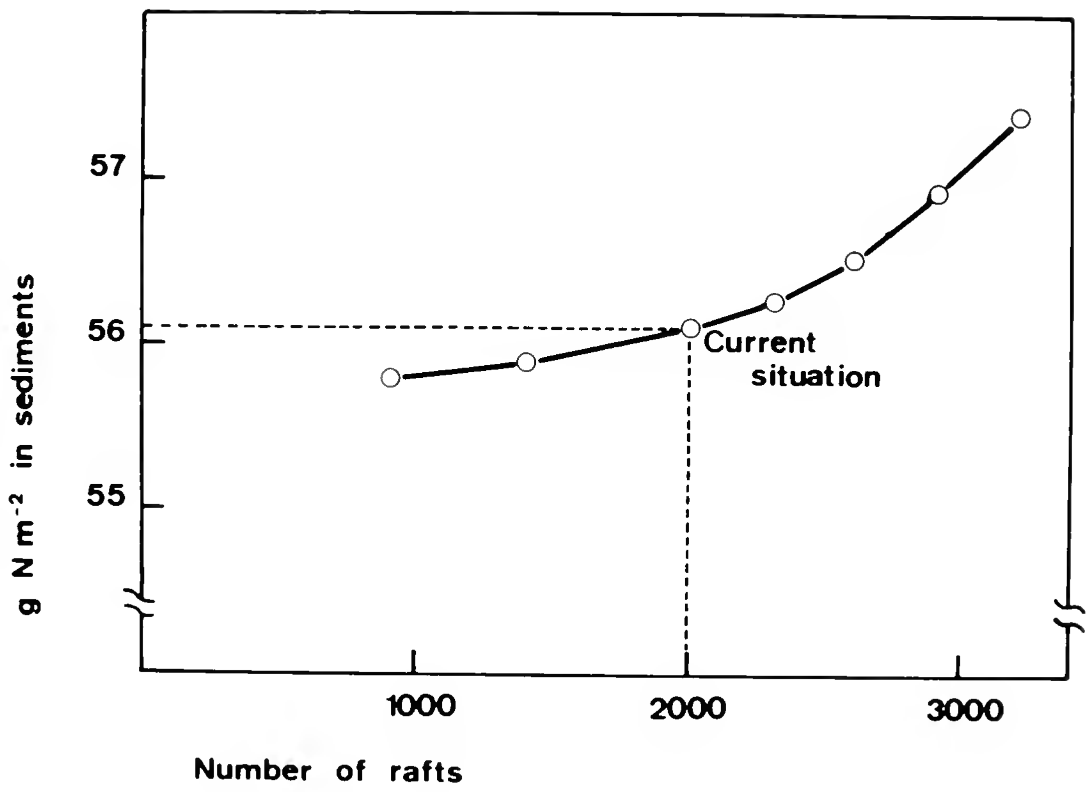

Figure 6. Particulate organic nitrogen in sediments. Standing stock after one year (with an equal initial condition) for a varying number of mussel rafts.

3) a further decrease in the production of oysters per existing oyster raft.

This first model was necessarily somewhat simplistic but raises some interesting questions and makes some defensible predictions about mussel production and raft number. In view of the great economic importance of this estuary, the Ria de Arosa, and the virtual impossibility of decreasing the number of rafts, once it is permitted to rise from the present value, we see little justification for any decision to change the status quo, at least until all the predictive modeling has been completed.

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ATLANTIC MARSH-ESTUARINE NEARSHORE DETRITAL SYSTEM (AMENDS) MODEL

(abstract)

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ABSTRACT

Our purpose in contructing this model is to summarize fishery and ecological data; and by analyzing known and assumed trophic interactions, to determine the role of yield species in the flow of organic matter through the system. This effort aids in identifying fundamental ecological processes controlling system production, and thus helps plan research to: (1) relate habitat type to fishery production and (2) compare actual to potential fishery production. The ultimate goal of this approach is to simulate how the system responds to changes in trophic spectrum of the harvest.

In developing the model we use three kinds of information: (1) commercial and recreational fishery yield, (2) primary production, and (3) trophic relationships (diet and conversion efficiency). The first two are available on a regional basis and help determine the geographic boundaries of the system. We consider only those species which are harvested close to shore and are supported by food produced there. The boundaries encompass an ecologically homogeneous area from Long Island to Georgia and out to 8 km.

Our approach to structuring the system is to work down the food chain from fishery yields to primary production, rather than the usual approach of trying to balance inputs and outputs at each step up the chain.

In this way we derived an estimate of material needed to drive the yield. Trophic level of yield species is expressed as the inverse of their production cost, where cost is amount of primary production needed to produce one unit of organism. Estimates of natural mortality were used to check the validity of assumed trophic relationships.

Figure 1 is a simplified picture of the system. It shows inputs and outputs averaged over a 20-yr period. All units are kilograms of dry organic matter. Much detail has been omitted in order to emphasize several major characteristics of the system.

Vascular plants (primarily marsh grasses) account for nearly as much primary production as algae. Much of the primary production enters the food chain through the bacterial-detrital complex. The fishery yield has a low trophic structure; that is, most of the yield is near the base of this (tilted) pyramid of increasing ecological cost. The width of the bars is proportional to the yields, and the placement of the bars indicates the relative trophic level. The yield of piscivorous fish is a relatively low proportion of the total (7%) and could be supported by natural mortality of menhaden.

Some of the preliminary conclusions from model analysis are:

1) The total yield (143,000 MT dry weight) is significant. It equals 20 percent of the North Sea yield.

2) The yield per square meter is high. It equals about 3 gm dry wt/m^2 which is about twice the North Sea.

3) The high yield per square meter results from:

   a. high primary production (500 gm dry wt/m^2, which is about two times the North Sea) and

   b. short, efficient food chains.

4) Construction of the simplest food chains consistent with the available data indicate that the yield populations require 20-40 percent of total primary production for their support.

5) A high proportion of primary production is utilized directly as detritus particles with little loss of energy through intermediate microbial steps.

6) The most important information gaps which limit our understanding of the system are:

   a. cost of producing forage food fish and

   b. percentage of energy lost in detrital decomposition.

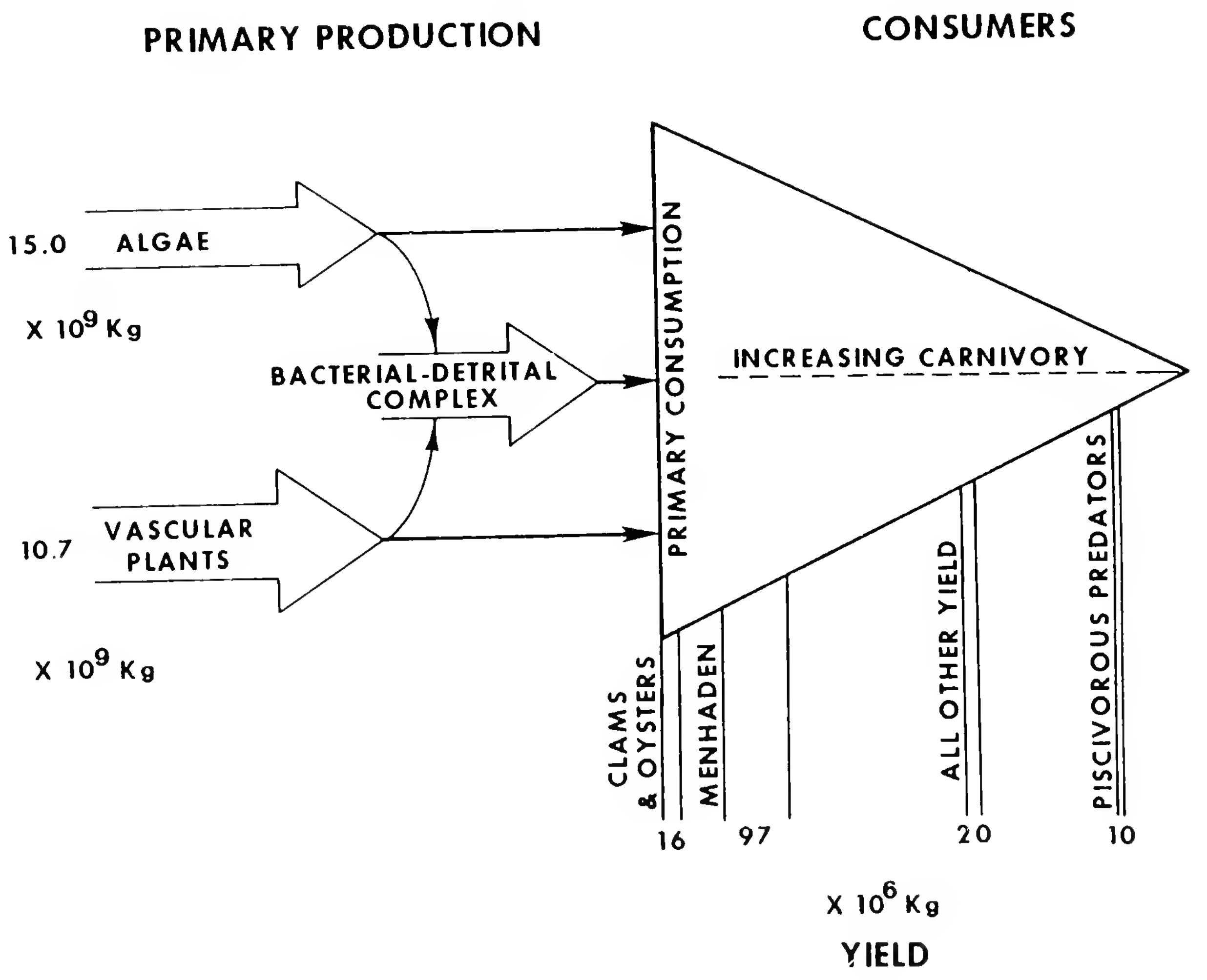

Figure 1. Simplified conceptual model of the Atlantic marsh-estuarine nearshore detrital system.

A SIMULATION MODEL OF A NEAR-SHORE MARINE ECOSYSTEM OF THE NORTH-CENTRAL GULF OF MEXICO

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ABSTRACT

Possible interactions between shrimp and bottomfish were incorporated into a theoretical model of the near-shore ecosystem of the north-central Gulf of Mexico. The model was used to simulate the changes in the standing stock and harvests of shrimp that might be side effects of reducing the unwanted fish caught in shrimp trawls by either of two methods: (1) decreasing the proportion of the bycatch that is discarded (but keeping the total quantity caught the same) and (2) reducing the total quantity caught (but discarding the same proportion of the bycatch). Possible direct and indirect influences of each change in harvesting procedure on food availability to shrimp, predation on shrimp, and competition with shrimp were explored. Although discards are a minor portion of the dead organic material used by shrimp for food, decreasing the discard rate by utilizing the bycatch decreased the availability of food to shrimp and its competitors in the model system. Using shrimp trawls with reduced catch efficiency for fish relative to that for shrimp resulted in increased total availability of food for shrimp but increased competition from bottomfish for that food. The increased supply of the common food ultimately outweighed increased competition from bottomfish, and the shrimp standing stock and shrimp yield recovered in spite of a higher standing stock of bottomfish. Predation on shrimp by bottomfish was a relatively minor influence compared to the other two effects. Changes in standing stocks not directly connected to shrimp were as important in determining the response of shrimp stocks to different strategies for reducing discards as changes in standing stocks that interacted directly with shrimp stocks as prey, predators, or competitors.

INTRODUCTION

A simulation model of the near-shore marine ecosystem of the north-central Gulf of Mexico is under development at the Southeast Fisheries Center of the National Marine Fisheries Service, Miami, Fla. The modeling effort is being used to investigate the dynamics of interactions among economically important species such as shrimp (*Penaeus* spp.) and menhaden (*Brevoortia patronus*) that are harvested from this highly productive area, which has been called "the fertile crescent" (Gunter 1963). An immediate need of fishery managers is to know the possible effects on shrimp of reducing the discarded bycatch of the shrimp fishery. Fishing operations that use relatively nonselective gear such as bottom trawls usually catch species other than those at which they direct their effort. Incidentally caught, or "bycatch," species are sometimes sold, but they are often dumped overboard when the amount is extremely large or when their value is low relative to that of the target species; such is the case in the Gulf of Mexico shrimp fishery. The fish discarded by the shrimp fishery are almost always killed by the trawl or by handling on deck and are dead when returned to the sea.

The total ex-vessel value of shrimp is greater than that of any other U.S. fishery species. More than half of the U.S. shrimp harvest comes from the northern Gulf of Mexico. The weight of "discards" averages approximately 14 times the weight of the shrimp landed and amounts to about 400,000 metric tons in the offshore area from Perdido Bay, Fla., to Point Au Fer, La., which has been delineated as the "primary area" of concentration of these species for purposes of making resource surveys. Atlantic croaker (*Micropogonias undulatus*), a sciaenid fish, makes up over 50 percent of the bycatch by weight. Five other species, three of which are sciaenids, make up about another 30 percent (GMFMC 1980). A directed fishery harvests about 50,000 metric tons of these species from the north-central Gulf of Mexico. They are used in the manufacture of pet food and oriental fish paste (surimi), as well as being marketed fresh. Catch rates and total landings of the bottomfish fleet have declined in recent years, and some state fishery biologists and participants in the groundfish industry feel that the decline is due to competition from an increasing number of shrimp trawlers.

Fishery managers charged with "optimizing yield" from our marine resources need to know whether the discards are a necessary consequence of harvesting shrimp, an economically valuable fishery product, or represent a waste that should be eliminated, preferably without economic hardship to the shrimping industry. A key question is whether killing fish in the shrimping operation and throwing them back to sea may actually enhance the yield of the target species by (1) reducing predation or competition, (2) providing a supplemental food source, or (3) increasing nutrient regeneration and thereby stimulating primary productivity, leading to a greater availability of shrimp food. If the present way of handling discards enhances shrimp production, then reducing discards could affect shrimp harvests detrimentally.

There are two possible ways to reduce discards without reducing the level of shrimping effort. One way is to catch fewer fish by using shrimp trawls with a reduced efficiency for catching fish relative to that for catching shrimp. This might increase the standing stock of living fish in the system and decrease the standing stock of dead fish returned to the system. Another way to reduce discards is to land a larger proportion of the bycatch. This would reduce the quantity of dead fish returned to the system without increasing the standing stock of living fish. These alternative approaches to reducing discards might affect shrimp harvests differently because one approach increases the standing stock of living fish and the other does not. The effects depend on interspecies dynamics and nutrient cycling in the system.

The ecosystem model integrates qualitative and quantitative information about the system into a mathematical characterization that simulates interspecies dynamics and nutrient cycling. Some simulations, using preliminary data, were performed to determine the possible effect on shrimp of alternative strategies to reduce discards. Modeling results indicate that the shrimp harvest could be affected by either method of reducing discards. If shrimp trawls to reduce fish catchability relative to that of shrimp were used, however, then an initial decline in the shrimp harvest might be followed by a recovery to present levels or slightly higher. In the simulation, the extent to which bottomfish selected against shrimp in favor of alternative prey affected the recovery of shrimp stocks. Indirect interactions among species were as important as direct interactions in determining initial and long-term responses of shrimp stocks to different strategies.

Model results may be dependent upon certain parameters in the model that determine the rate of nitrogen remineralization by living bottomfish. They may also depend on the Michaelis-Menten half-saturation constant, which determines the rate of phytoplankton production as influenced by the concentration of nitrogen in the water.

It is too early in the development of the model to state with any assurance that model results apply to the real world. The model must undergo further testing and evaluation before specific management recommendations can be made on the basis of simulations. At this point the modeling effort suggests that changes in fishing pressure can alter the balance between species in both competitive and prey-predator interactions. Work with the model suggests that standing stocks that are large relative to other standing stocks in the system can be remarkably stable in the face of heavy fishing pressure. At the same time, fishing pressure on such stocks can influence other stocks, even those that are neither prey nor predator of the harvested species. The model demonstrated that animals at higher trophic levels, such as marine mammals, can have large impacts on lower trophic groups, such as zooplankton. Both the stability and instability exhibited within the model system in the face of fishing pressure were due to the complex interconnections of system structure.

In this paper I describe the mathematical structure of the model and present and discuss results of simulations. A more detailed description of the north-central Gulf of Mexico near-shore marine ecosystem and the conceptual basis for the model design were presented in a previous paper (Browder 1981).

MODEL STRUCTURE

The model is diagramed in the energy flow language of H.T. Odum (1982) in figure 1. It is a food web of 12 compartments connected by flows of energy in the form of organic matter and by flows of nutrients. The compartments represent inorganic nitrogen; standing stocks of phytoplankton and animals in eight trophic groups; and organic material of two types, high nitrogen and low nitrogen. The model receives three inputs: solar radiation (as gross primary productivity), inorganic nitrogen, and low-nitrogen organic material, the latter two of which come from river inflow. The types of exchanges between compartments of the model are nutrient uptake, phytoplankton rain to the bottom, feeding by animals, release of unassimilated organic material by animals, and the release of mineralized nitrogen through the decomposition of organic material and in the elimination of metabolic waste products from animals. Energy leaves the system as carbon dioxide (through the respiration of plants, animals, and decomposers) and as harvests.

The basic mathematical structure of the model is a set of 12 differential equations:

$$\dot{Q}_1 = J_1 + \Sigma N_i - L - P_{1,2}, \qquad Q_1\text{: INORGANIC NITROGEN}$$

$$\dot{Q}_2 = J_2 - P_{2,3} - \Sigma P_{2,j} - R_2, \qquad Q_2\text{: PHYTOPLANKTON}$$

$$\dot{Q}_3 = J_3 + P_{2,3} + \Sigma F_j - \Sigma P_{3,j}, \qquad Q_3\text{: LOW-NITROGEN ORGANIC}$$

$$\dot{Q}_4 = F_5 + (B - P_{4,12}) - \Sigma P_{4,j}, \qquad Q_4\text{: HIGH-NITROGEN ORGANIC}$$

$$\dot{Q}_5 = \Sigma P'_{i,5} - \Sigma P_{5,j} - R_5, \qquad Q_5\text{: ZOOPLANKTON}$$

$$\dot{Q}_6 = \Sigma P'_{i,6} - \Sigma P_{6,j} - R_6 - H_6, \qquad Q_6\text{: PELAGIC FISH}$$

$$\dot{Q}_7 = \Sigma P'_{i,7} - \Sigma P_{7,j} - R_7, \qquad Q_7\text{: BENTHOS}$$

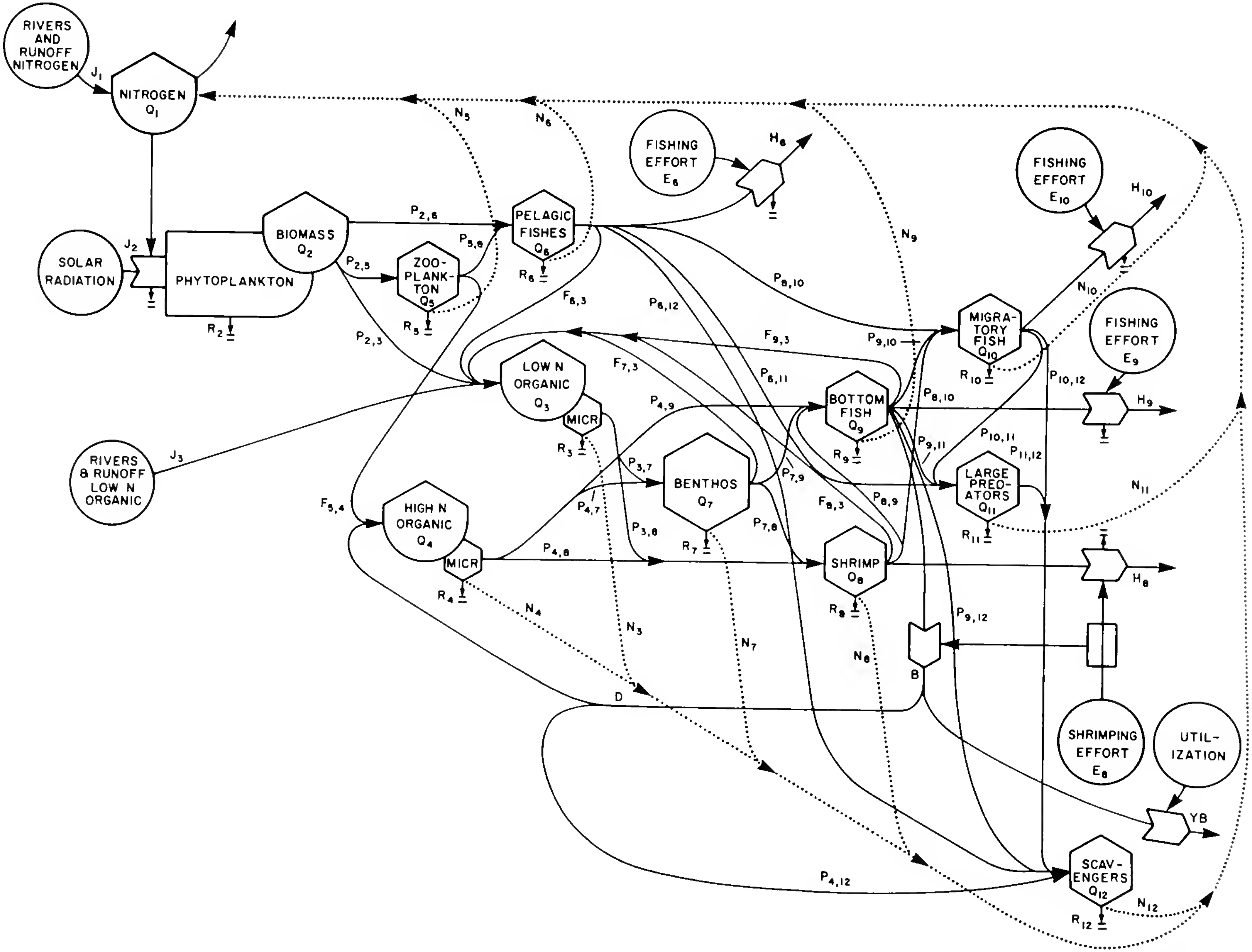

Figure 1. Diagram of northcentral Gulf of Mexico nearshore marine fishery ecosystem. (Symbols are from the energy circuit language of H. T. Odum 1982).

$Q_8 = \Sigma P'_{i,8} - \Sigma P_{8,j} - R_8 - H_8,$ Q_9: SHRIMP

$\dot{Q}_9 = \Sigma P'_{i,9} - \Sigma P_{9,j} - R_9 - H_9 - B,$ Q_9: GROUNDFISH

$\dot{Q}_{10} = \Sigma P'_{i,10} - \Sigma P_{10,j} - R_{10} - H_{10},$ Q_{10}: MIGRATORY PELAGICS

$\dot{Q}_{11} = \Sigma P'_{i,11} - \Sigma P_{11,j} - R_{11},$ Q_{11}: MARINE MAMMALS

$\dot{Q}_{12} = \Sigma P'_{i,12} + cB - \Sigma P_{12,i} - R_{12},$ Q_{12}: LARGE SCAVENGERS

where

i is a number from 2 to 12 representing food compartments,

j is a number from 5 to 12 representing predator compartments,

J_k are flows into the system from outside,

N_j are nitrogen release and remineralization rates,

L is the rate of loss of inorganic nitrogen in currents,

$P_{i,j}$ are flows of energy between compartments, most of which represent feeding,

$P'_{i,j}$ are the assimilated portion of food intake,

F_j are flows of unassimilated food to organic compartments from animal compartments,

B is the bycatch rate,

R_j are respiration rates,

H_j are harvest rates.

Flows indicated in the equations but not represented by connecting lines in the model diagram can be assumed to be null flows.

J_1 and J_3 are constants representing nitrogen and low-nitrogen organic matter flows into the system. J_2, gross primary productivity (GPP), is a variable that is a product of S_2, GPP at saturation, multiplied by a Michaelis-Menten equation, with nitrogen as the substrate (controlling factor):

$$J_2 = S_2(Q_1/Km_1 + Q_1).$$

$P_{1,2}$ is the rate of uptake of organic nitrogen by phytoplankton:

$$P_{1,2} = n_2(J_2 - R_2),$$

where n_2 is the nitrogen concentration in phytoplankton.

L, the loss of nitrogen from the system, and $P_{2,3}$, phytoplankton rain, are simple functions of standing stock:

$$L = c_1Q_1 \text{ and}$$

$$P_{2,3} = c_{i,3}Q_2.$$

Rates of predation are donor and recipient controlled and governed by the simple equation

$$P_{i,j} = c_{i,j}W_{i,j}Q_iQ_j,$$

where

Q_j is predator standing stock,

Q_i is prey standing stock,

$W_{i,j}$ is the selectivity factor of j for each i, and

$c_{i,j}$ is a rate-coefficient determined by balancing the model for steady-state conditions by a method to be described.

The material assimilated by the predator is determined by the equation

$$P'_{i,j} = P_{i,j}A_{i,j},$$

where $A_{i,j}$ is a matrix of assimilation coefficients, ranging from 0 to 1.0. The assimilation coefficient is set at 0 in cases where there is no exchange between compartments.

Unassimilated food, which is deposited in the low-nitrogen organic compartment, is a function of the rate of feeding on each prey and the assimilation coefficient specific to that prey:

$$F_j = \Sigma(P_{i,j}(1-A_{i,j})).$$

Respiration and harvest rates are simple functions of standing stock:

$$R_j = r_jQ_j,$$

$$H_j = e_jQ_j,$$

where r_j is the respiration rate-coefficient specific to each compartment and e_j is the harvesting rate-coefficient specific to each compartment. The coefficient e is equivalent to the fishing mortality (F) of mathematical fishery biology (Ricker 1975), which has two components, effort (f) and catchability (q).

Discard rate is a function of the rate of shrimping effort (e_8), bottomfish standing stock (Q_9), and the ratio of bottomfish catchability to shrimp catchability (b):

$$B = be_8Q_9.$$

Remineralization is treated as a simple function of respiration rate:

$$N_j = n_jR_j,$$

where n_j is the ratio of nitrogen released in excrement (or bacterial excretions, in the case of the detritus compartments) to carbon dioxide released in respiration from each organic compartment except phytoplankton.

FLOW-BALANCING PROCEDURE

Flows and rate-coefficients were calculated by an iterative top-down flow-balancing procedure that was based on three assumptions:

1) The system at present is in steady state (over the long term, standing stocks are neither growing or declining).

2) Animals with several food sources feed nonselectively on these sources in a proportion that is equal to the proportion of the standing stocks of these food sources in the environment.

3) Selectivity can be approximated by differential "weighting" of two or more feeding flows to the predator.

In the steady-state situation, the inflows to a compartment equal the outflows. Total inflows to a compartment can, therefore, be determined if the total outflows are known. The assumption that relative feeding rates on alternative food sources are equal to relative standing stocks of these food sources allows the inflows to a compartment to be apportioned among food sources. If selectivity is considered important, the apportionment can be weighted accordingly. The equations for calculating flows and setting rate-coefficients by this procedure are as follows:

$$X_j = \Sigma(Q_jA_{i,j}) / \Sigma P_{j,i},$$

$$P_{i,j} = X_jQ_iW_{i,j},$$

$$P'_{i,j} = A_{i,j}P_{i,j}$$

$$c_{i,j} = (X_jW_{i,j})/Q_j.$$

Flow balancing must start at the top of the food web with animal groups that experience no significant predation or where the predation level can be estimated independently. (In this particular case I started the flow-balancing calculation with the large scavenger compartment.) The essential inputs for determining flows and setting rate-coefficients by the flow balancing method are (1) standing stocks for all compartments, (2) respiration rate-coefficients for all animal compartments, (3) either respiration rate-coefficients or rates of all inflows from outside the system for all food-chain-base compartments (i.e., phytoplankton and organic materials), (4) assimilation coefficients for each food of each predator, and (5) selectivity weighting factors for alternative prey of each predator. If assimilation rates are not known and cannot be even roughly estimated or if selectivity is not thought to be important, $A_{i,j}$ and $W_{i,j}$ can be eliminated from the above equations.

Calculation of initial feedbacks of unassimilated food to organic material compartments and of mineralized nitrogen to the nitrogen compartment are included in this iterative procedure. A computerized routine calculates inflows and outflows of nitrogen at steady state in order to set the constants relating nitrogen excretion rates to respiration rates. In this routine, the inflow of nitrogen to each compartment is determined by multiplying the rate of feeding on each food source at steady state by the nitrogen concentration of that food source. The quantity of nitrogen used in growth and predation is determined by multiplying growth and predation rates by body nitrogen concentration. The remainder is apportioned between elimination in excrement (N) and elimination in fecal material (NF). (Excrement is the elimination of metabolized body wastes, usually in the form of urine. Fecal material consists of ingested material that was not assimilated.) Constants relating nitrogen mineralization rates to respiration rates are then calculated for use in the model.

In ecosystem modeling, the usual method for calculating the flows of nitrogen in excrement and fecal material is to make the nitrogen concentrations in excrement (carbon-dioxide-equivalent sugar) and fecal material equal to the nitrogen concentration of the organism. Calculations based on values from a laboratory study by Darnell and Wissing (1975) indicate that nitrogen concentrations in excrement or fecal material are not equal to the nitrogen concentration of the organism or even to the weighted nitrogen concentration of its food sources. Furthermore, if such relationships are assumed, nitrogen inflows will not equal nitrogen outflows to each compartment when organic flows are in steady state, if nitrogen concentrations of the various compartments differ. The method used in quantification of this model should be more accurate than that ordinarily used and should assure more realistic results.

The necessary input information for the nitrogen initialization routine is (1) body nitrogen concentrations for all compartments (except high-nitrogen organic, which had to be calculated in the routine) and (2) ratios of nitrogen released in excrement to nitrogen released in feces from the animal compartments. Nitrogen concentration of the high-nitrogen organic compartment was a variable of the model that depended upon the mix of zooplankton fecal pellets and fish discards in the compartment.

Computer programs for the top-down iterative flow-balancing procedure and the model were written in Microsoft BASIC and were run on an Ithaca Intersystems Z80-based microcomputer. The Euler method of numerical integration was used in the simulations. The iteration interval for the simulation was 0.1 day.

QUANTIFICATION

The initial conditions, rate-coefficients, and other constants that were required to run the model are shown in table 1. The values used for each of these parameters and the calculations and references that were the bases for

Table 1. Stock initial conditions for input coefficients flow model.

Flows from outside Sources	Value	Calculation	Information Source
J1 Inorganic Nitrogen	112	$(2000 \text{ mg } NO_3\text{-N/m}^3)^a$ $(1.9 \times 10^9 \text{m}^3\text{/day})^b$ $/(3.42 \times 10^{10} \text{m}^2)^c$	a. Concentration for Mississippi River only (Sackett 1972). b. Freshwater inflow from Miss. & Atchafalaya Rivers (Sackett 1972). c. An arbitrary area of bottom estimated to be under immediate influence of Mississippi River discharge. Extends from Pt. Au Fer, LA to Perdido Bay, FL Area inside 93 meters (50 fathoms)(Patella 1975).
J2 Gross primary productivity	2,711	GPP x Q2 = (NPP + R)Q2 $=(1.642^a + 0.768^b)1{,}125$	a. Sklar (1976). b. Ryther and Guillard (1962).
J3 Low-nitrogen organic matter	3,037	$\{[(15 \times 10^4 \text{mgC/m}^2\cdot\text{yr})^a$ $(1.367 \times 10^{10} \text{m}^2)^b$ $/(6.069 \times 10^{10} \text{m}^2)^c]$ $+ [(20\text{mgC/l})^d(6.9 \times$ $10^{14} \text{l/yr})^e/(3.42$ $\times 10^{10} \text{m}^2)^f]$ 2.5qDOM/gC} /360 days	a. Estimated annual carbon export from Barataria Bay per square meter of inshore water (Happ *et al*. 1977). b. Area Louisiana inshore water (Perret *et al*. 1971). c. Area Louisiana offshore water to 93 meters (50 fathoms. d. Concentration DOC in Mississippi River water (96% drops out within 25 km of discharge) (G. Harvey, NOAA/AOML/Miami, pers. comm.). e. Annual volume discharge Mississippi and Atchafalaya Rivers. f. Area inside 93 meters (50 fathoms) from Pt. Au Fer, LA to Perdido Bay, FL (Patella 1975).

Table 1. Continued 2.

Stock Initial Conditions mg DOM/m^2)	Value	Calculation	Information Source
Q1 Nitrogen^a	1,260	$(126 mg/m^3)^a (10 m^3/m^2)^b$	a. Mg inorganic nitrogen NO_3, NO_2, & $NH_3 - N$ (Sklar 1976). b. Assumed depth of photic zone.
Q2 Phytoplankton	1,125		
Q3 Low N-organic	218,850	$(10 mg/g)^a (7295 g/m^2 \cdot cm)^b$ $(3 cm)^c$	a. Concentration of organic material (d.w.) in shallow water Mississippi Delta marine sediment (Hausknecht 1980). b. Weight of inorganic fraction per centimeter thick square meter area, calculated by the method of Bennett and Lambert (1971) and assuming a specific gravity for clay sediment of 2.70 (Bennett pers. comm.). c. Depth of sediment in which more than 80% of meiofauna occurs (Tietjen 1969).
Q4 High N organic	33.39		a. Assumed, based on estimated daily inflow of zooplankton fecal pellets and fish discards.
Q5 Zooplankton	145.9	$(2432 ind/m^3)^a$ $(6 \times 10^{-3} mgDOM/ind)^b$ $(10 m^3/m^2)^c$	a. Reitsema (1980). b. Conover (1959). c. Assumed photic zone.
Q6 Pelagic fish (menhaden)	2,966	$(4.5 \times 10^{14} mg)^a$ $(2)^b (0.2)^c /$ $(6.069 \times 10^{10} m^2)^d$	a. Approximate LA menhaden landings - 1975 NMFS (1978). b. Assumed ratio coastal herrings/menhaden. c. Dry wt/wet wt (Parsons et al. 1977). d. LA. bottom area inside 93 meters (50 fathoms) (Patella 1975). Note: Annual fishing mortality of 1.0 assumed.
Q7 Benthos	8,000		Parker et al. (1980).

Table 1. Continued 3.

Stock	Initial Conditions mg DOM/m^2)	Value	Calculation		Information Source
Q8	Shrimp	79.42	$(2.41 \times 10^{13} mg)^a (0.2)^b / (6.069 \times 10^{10} m^2)^c$	a. b. c.	Approximate LA landings 1975 (NMFS 1978). Dry wt/wet wt. LA bottom area inside 93 meters (50 fathoms) (Patella 1975).
Q9	Bottomfish	2,011	$(430 \times 10^{12} mg)^a (0.80)^b (0.2)^c / (3.42 \times 10^{10} m^2)^d$	a. b. c. d.	Estimated groundfish catch incl. discards in area from Pt. Au Fer. LA to Perdido Bay, FL (GMFMC 1980). Assumed fishing mortality. Dry wt/wet wt (Parsons *et al.* 1977). Area inside 93 meters (50 fathoms) from Pt. Au Fer. LA to Perdido Bay, FL (Patella 1975).
Q10	Migratory fish (mackerels)	32.4	$[(2.27 \times 10^{12} mg)^a / (0.41)^b + (4.5 \times 10^{12} mg)^c / (0.2756)^d (0.67)^e] (0.2)^f (2)^g / (1.81 \times 10^{11} m^2)^h$	a. b. c. d. e. f. g. h.	U.S. king mackerel landings 1975. King mackerel fishing mortality estimated on basis of FMP (GMSAFMC 1980). U.S. Spanish mackerel landings 1975 (NMFS 1978). Spanish mackerel fishing mortality estimated on basis of FMP (GMSAFMC 1980). Estimated fraction of total stocks that spend time in Gulf. Dry wt/wet wt. Factor to estimate total migratory biomass from mackerel biomass. Gulf bottom area inside 93 meters (50 fathoms) from 26°N lat. in western Gulf (Brownsville) to 28°N lat. in Gulf (Tampa).
Q11	Large predators (dolphin)	6.6	$(0.44 \times 10^{-6} dolphin/m^2)^a (150 \times 10^6 mg/dolphin)^b (0.2)^c (0.5)^d$	a. b. c. d.	Density of dolphin at LA coastal sites. Leatherwood *et al.* (1978). Estimated weighted average weight of bottlenose dolphin. Dry wt/wet wt (Parsons *et al.* 1977). Correction factor to account for better than average densities in sampling area.
Q12	Large scavengers (sharks)	24.29	$(24.29 mg/m^2)^a (0.2)(5)^b$	a. b.	Estimated average biomass for Gulf (L. Rivas. Unpubl.). Assumed concentration in near-shore area.

Table 1. Continued 4.

Respiration Rates (mg DOM/m^2.day)		Value	Information Source
R2	Phytoplankton	0.768 x Q2	Ryther and Guillard (1962) for 4μ, eurythermal diatiom (*Cyclotella nana*, 3H).
R3	Low nitrogen organic	0.01872 x Q3	Set in flow-balancing procedure to approximate value for zooplankton fecal pellet decomposition calculated from Johannes and Satomi (1966) (0.1834).
R4	High nitrogen organic	(0.2071) x Q4	Calculated in flow-balancing procedure.
R5	Zooplankton	0.32 x Q5	Conover (1959) for 6 g D.W. copepod.
R6	Pelagic fish (menhaden)	0.020 x Q6	Hettler (1976) for 226 g W.W. menhaden.
R7	Benthos	0.081 x Q7	Pamatmat (1980) 1 g W.W. (assumed 20x anaerobic rate).
R8	Commercial shrimp	0.0414 x Q8	Bishop *et al.* (1980) for 6.7 g brown (assumed 12 hrs activity.
R9	Bottomfish	0.018 x Q9	Hoss (1974) for 100 g pinfish.
R10	Migratory pelagics (mackerel)	0.006 x Q10	Brill (1979) for 1 kg skipjack tuna.
R11	Large predators (dolphin)	1.38 x Q11	Irving *et al.* (1941) for 150 kg bottlenose dolphin.
R12	Large scavengers (sharks)	0.0068 x Q12	Brett and Blackburn (1978) for 900 g spiny dogfish.
Harvest rate coefficients (9 DOM/m^2.day)			
E1	Pelagic fish (menhaden)	0.01389 x Q6	NMFS (1978)
E2	Shrimp	0.002778 x Q8	NMFS (1978)
E3	Bottomfish bycatch	1.023 x E2 x Q9	GMFMC (1980)
E4	Bottomfish directed	0.0004061 x Q9	NMFS (1978)
E5	Migratory pelagics	0.0000451 x Q10	NMFS (1978)

Table 1. Continued 5.

Nitrogen Flows and Controls ($mgN/m^2 \cdot day$ or mgN/m^2)		Value	Body N/DOM	Information Source
K_{m1}	Michaelis Menten coefficient for effect on nitrate nitrogen concention on phytoplankton production	120		Based on a half-saturation constant of 1.5 M/l for Asterionella japonica, a coastal diatom, determined by laboratory experiments by Eppley and Thomas (1969).
L	Inorganic nitrogen lost to the system in currents	238		Assumed to be proportional to inorganic nitrogen concentration.
P1,2	Nitrogen uptake by phytoplankton	111		Assumed to be proportional to growth, grazing loss, and sinking rates in phytoplankton or the difference between gross primary production and respiration, multiplied by the nitrogen concentration in phytoplankton.
				Nitrogen remineralized is assumed to be proportional to respiration of animals (and of the microbes decomposing organic material). Values were estimated from body N/DOM ratios by calculating steady state nitrogen flows corresponding to steady state organic matter (DOM) flows. In the calculations, nitrogen not used in growth or predation was divided between excrement and feces in the proportions 0.8 and 0.2, respectively, in all animals except zooplankton. For zooplankton, the apportionment was 0.2 for excrement and 0.8 for feces to reflect the high-nitrogen concentration of zooplankton fecal pellets suggested by the literature. References below are for body N/DOM ratios.
	Phytoplankton (Q2)		.06	Strickland (1960) for mixed taxa.
N3	Low N organic (Q3)	0.05 X R3		Estimate for river detritus.
N4	High N organic (Q4)	0.0756 x R4		
N5	Zooplankton (Q5)	0.0165 x R5	.073	Parsons et al. (1977) for zooplankton.

Table 1. Continued 6.

Nitrogen Flows and Controls ($mgN/m^2 \cdot day$ or mgN/m^2)		Value	Body N/DOM	Information Source
N6	Pelagic fish (Q6)	0.0453 x R6	0.0924	Sidwell 1981 for menhaden.
N7	Benthos (Q7)	0.0353 x R7	0.1202	Darnell and Wissing (1975).
N8	Shrimp (Q8)	0.0959 x R8	0.1083	Sidwell (1981) for commercial penaeid shrimp.
N9	Bottomfish (Q9)	0.1159 x R9	0.1297	Darnell and Wissing (1975) for pinfish.
N10	Migratory fish (Q10)	0.1023 x R10	0.1250	Estimated from a value for mackerel from Sidwell (1981) for muscle tissue only.
N11	Marine mammals (Q11)	0.1096 x R11	0.1308	Sidwell (1981) for bottlenose dolphin muscle.
N12	Large scavengers (Q12)	0.1098 x R12	0.1445	Sidwell (1981) for *Sphyrna blochii* muscle.

Table 1. Continued 7.

	Assimilation Efficiencies (Proportions)	Value	Information Source
A2,5	Phytoplankton (Q2) to zooplankton (Q5)	0.7	Parsons et al. (1977) for copepods.
A2,6	Phytoplankton (Q2) to pelagic fish (Q6)	0.42	Assumed.
A3,7	Low N organic (Q3) to benthos (Q7)	0.5	Assumed.
A3,8	Low N organic (Q3) to shrimp (Q8)	0.2	Jones (1973) for brown shrimp.
A4,7	High N organic (Q4) to benthos (Q7)	0.9	Assumed.
A4,8	High N organic (Q4) to shrimp (Q8)	0.4	Jones (1973) for brown shrimp.
A4,9	High N organic (Q4) to bottomfish (Q9)	0.86	Darnell and Wissing (1975) for polychaetes consumed by pinfish.
A5,6	Zooplankton (Q5) to pelagic fish (Q6)	0.6	Assumed.
A6,10	Pelagic fish (Q6) to migratory fish (Q10)	0.8	Assumed.
A6,11	Pelagic fish (Q6) to large predators (Q11)	0.8	Assumed.
A6,12	Pelagic fish (Q6) to large scavengers (Q12)	0.8	Assumed.
A7,8	Benthos (Q7) to shrimp (Q8)	0.4	Jones (1973).
A7,9	Benthos (Q7) to bottomfish (Q9)	0.7	Assumed.
A8,9	Shrimp (Q8) to bottomfish (Q9)	0.86	Darnell and Wissing (1975) for pinfish.
A8,10	Shrimp (Q8) to migratory fish (Q10)	0.7	Assumed.
A9,10	Bottomfish (Q9) to migratory fish (Q10)	1.0	Assumed.
A9,11	Bottomfish (Q9) to large predators (Q11)	1.0	Assumed.
A9,12	Bottomfish (Q9) to large scavengers (Q12)	1.0	Assumed.

Table 1. Continued 8.

	Assimilation Efficiences (Proportions)	Value	Information Source
A10,11	Migratory fish (Q10) to large predators (Q11)	0.7	Assumed.
A10,11	Migratory fish (Q10) to large scavengers (Q12)	1.0	Assumed.
A11,12	Large predators (Q11) to large scavengers (Q12)	1.0	Assumed.
	Selectivity Weighting Factors		
W3,7	Low-N organics (Q3) to benthos (Q7)	0.008	Assumed, adjusted downward from .01 to set high-N respiration rate coefficient.
W3,8	Low-N organics (Q3) to shrimp (Q8)	0.001	Assumed.
W8,9	Shrimp (Q8) to bottomfish (Q9)	0.01	Assumed.
W6,12	Pelagic fish (Q6) to large scavengers (Q12)	0.1	Assumed.
W9,12	Bottomfish (living)(Q9) to large scavengers (Q12)	0.1	Assumed.
W10,12	Migratory pelagics (Q10) to large scavengers (Q12)	0.1	Assumed.
W11,12	Marine mammals (Q11) to large scavengers (Q12)	0.001	Assumed.
Other W	All other	1.0	Assumed.
Area for Calculating Biomass (square meters)			
Louisiana coastal inside 50 fathoms		$6,069 \times 10^{10}$	Patella (1975).
"Primary area" of bottomfish fishery Management Plan (Point Au Fer, LA to Perdido Bay, FL inside 93 meters (50 fathoms))		3.42×10^{10}	Patella (1975).
U.S Gulf coastal area N of 28° N off Florida coast and N of 26° N off Texas coast		18.1×10^{10}	Patella (1975).

Table 1. Continued 9.

Assimilation Efficiences (Proportions)	Value	Information Source
Conversions for Standardizing Units		
O_2 to dry organic matter (DOM)	x 1	Parsons *et al*. (1977).
Carbon (C) to DOM	x 2.5	Parsons *et al*. (1977).
Chlorophyll to carbon	x 25	Parsons *et al*. (1977).
Wet weight to dry weight	x 0.2	Parsons *et al*. (1977).

NOTE: Superscripts in equations refer to information sources, indicated by matching letters these values are also shown in table 1. The model was quantified for "average" conditions, and seasonality was ignored; thus model forcing functions were constants. Values in the literature and fisheries statistical data were used for the quantification.

Standing stock estimates are specific to the area offshore the Mississippi Delta. Fishery standing stocks were estimated from fishery landings data for 1975, using the relationship:

$$N = C/F,$$

$$Q = N(D/M),$$

where N is population number, C is landings, F is instantaneous fishing mortality, D is average dry weight of individuals, and M is the area inside 93 m (50 fathoms) covered by the landings, or, in the case of migrating species, area covered in migrations. The F values used in these calculations were crude approximations and may be sources of error.

Pelagic forage fish standing stock was based on menhaden catches for Louisiana and doubled to take into account other forage fish species in the area. The shrimp standing stock was also based on Louisiana landings. Bottomfish standing stock was based on landings and estimated bycatch of croaker and other bottomfish species in the area from Point Au Fer, La., to Perdido Bay, Fla. The model is quantified for the area from the eastern Louisiana border to Point Au Fer, with respect to the above three groups.

The standing stock of migratory pelagics was based on king and Spanish mackerel catches for the entire U.S. Gulf coast. The mackerels are not fished commercially to any extent in the north-central Gulf, but those stocks fished commercially elsewhere probably spend about half the year in the northern Gulf. By dividing estimated total Gulf standing stocks by total area of the Gulf, I obtained an estimate for an "annual average" standing stock of mackerels in the model area. This value was doubled to provide an estimate of coastal pelagics standing stock, which includes other species in this same trophic group for which there are no commercial landings on which to base standing-stock estimates.

Animal respiration rate-coefficients were specific to each compartment of the model and were based on values for a representative species in each compartment or a related or similar type species. Calculations of nitrogen flows are based on body nitrogen concentration of a representative species in each compartment or a related or similar type of species. For zooplankton, the N/NF ratio used (in the initialization routine) for setting nitrogen release rates relative to excretion rates was 0.3 to 0.7. The routine calculated a nitrogen concentration ratio in fecal pellets of 0.06455 at steady state (compared to a value of 0.0448 calculated from Johannes and Satomi (1966)). For bottomfish, the N/NF ratio employed was 0.99 to 0.01 (compared to a value of 0.9942 to 0.0058 from Darnell and Wissing (1975) for pinfish, *Lagodon rhomboides*). Specific nitrogen information of this type was not available for the other animal compartments. The production of feces by the three higher trophic compartments (migratory pelagics, marine mammals, and large scavengers) was considered to be insignificant in the model, and all nitrogen loss was designated to excrement. For the other animal compartments, the N/NF ratio was 0.8 to 0.2.

Phytoplankton standing stock (estimated from chlorophyll data) and net primary productivity values were from offshore Mississippi Delta studies. The Michaelis-Menten half-saturation constant relating phytoplankton production to nitrogen concentration in the water came from a laboratory study with a coastal diatom, using nitrate as the substrate. Phytoplankton respiration was estimated from generalized information and is not specific to the area or the species found there.

Some assimilation coefficients were available from the literature, but most were approximated, based on qualitative information. Selectivity weighting factors of alternative food sources for benthic organisms, shrimp, bottomfish, and large scavengers were weighted differently to reflect qualitative information from the literature regarding feeding selectivity. High-nitrogen organic material was weighted higher than low-nitrogen organic material for both benthos and shrimp, assuming a preference by these groups for the high-nitrogen material. The weighting factor for feeding by benthos on low-nitrogen organic material was adjusted downward slightly to set the respiration rate-coefficient for high-nitrogen organic material (calculated in the flow balancing procedure) close to the value of the decomposition rate-coefficient for fecal pellets that was calculated from Johannes and Satomi (1966); the model value was 0.2071, as compared to the literature value of 0.1834. Benthos was weighted equal to high-nitrogen organic for feeding by shrimp. Shrimp received a low weighting relative to alternative prey of bottomfish to reflect the infrequent occurrence of commercial penaeid shrimp in the stomachs of bottomfish that has been noted by Sheridan et al. (1981). This would tend to minimize the effect that predation by bottomfish could have on shrimp standing stock in the model. Discards were weighted slightly more heavily than live fish in the food flow to large scavengers, reflecting the fact that sharks and others in this group have a scavenging tendency, although they also feed on live animals. Marine mammals were weighted lower than live fish in the feeding flow to scavengers because the size of adult marine mammals equals or exceeds that of large scavengers, and adults are capable of protecting both themselves and juveniles. Weighting factors were estimated on a $\log_{10}$ scale, as a first approximation. Simulation tests indicated that weighting factors of this magnitude were necessary for differences between simulations caused by differences in weighting factors to be detected on the scale of the simulation graphs.

SIMULATION TESTS

Using the model, tests were made of the effect of alternative strategies of handling bycatch on fishery yields and standing stocks. These strategies were as follows:

1. Present trawling and discarding practices.

2. Half of bycatch not discarded (utilized).

3. Shrimp trawl efficiency for catching fish cut in half.

4. Shrimp trawl efficiency for catching fish halved and directed fishing effort for bottomfish doubled.

In making these tests, I assumed that the basic structure of the system (i.e., the number of compartments and the links between compartments) did not change as an adjustment to new conditions. In each test, no rate-coefficient or other constant in the model program was changed other than the one (or in test #4, two) changed to simulate the test condition.

A sensitivity test was run to determine how changes in the selectivity weighting factor of bottomfish for shrimp affected the behavior of the model under alternative strategies of handling bycatch and how this might affect the conclusions drawn from simulation results. A test was also made of the influence on model results of the relative standing stocks of bottomfish and the other groups. In a further test, marine mammals were made to feed on bottomfish discards as well as living bottomfish, a feeding flow that was not included in the original model.

In the initialization routine, model coefficients were set at steady-state to represent the present practice of handling discards. Under present practices, all standing stocks were constant throughout the 5-year period of the simulation (fig. 2).

Figures 3 through 5 are simulation graphs of the management tests. These graphs track the standing stock of each compartment for the 5 years following imposition of new conditions. Under test conditions, standing stocks were initially the same as under steady-state conditions, but changed in response to the particular conditions imposed, moving toward a new steady-state (or, in some cases, toward a stable oscillation). Shrimp stocks declined and leveled at a new steady-state approximately 28 percent lower than the former one when half of the bycatch was utilized rather than being returned to the system (fig. 3). When discards were reduced through the use of shrimp trawls with half the efficiency for catching fish, shrimp stocks declined initially but rebounded to present (steady-state) levels (fig. 4). When the combination of shrimp trawls half as efficient in catching fish and a doubling of directed effort at bottomfish was tested, shrimp standing stock declined by 10 percent (fig. 5). When shrimp standing stocks declined, shrimp harvests declined, as indicated in table 2.

Other standing stocks also responded to the tested changes in harvesting practices. Marine mammals, zooplankton, and high-nitrogen organic material increased when shrimp trawls that reduced fish catchability relative to shrimp catchability were used, but decreased when the bycatch was utilized rather than discarded. Contrarily, stocks of pelagic fish and migratory pelagic fish decreased when shrimp trawls that reduced fish catchability relative to that of shrimp were used but increased when the bycatch was utilized. Standing stock of bottomfish and phytoplankton and the quantity of nitrogen in the water also changed under different harvesting conditions, but these changes were so small relative to the total quantity that they could not be seen on the graphs. An idea of the effect of test conditions on these larger, more stable compartments can be obtained by comparing total annual respiration of these compartments during the different test simulations (table 3). Respiration is directly proportional to standing stock and thus total respiration is an index of the level of standing stock over a period of time.

In the first sensitivity test, weighting factors were changed to indicate less selectivity of bottomfish against shrimp relative to alternative prey. The change in weighting factors increased the predation rate of bottomfish on

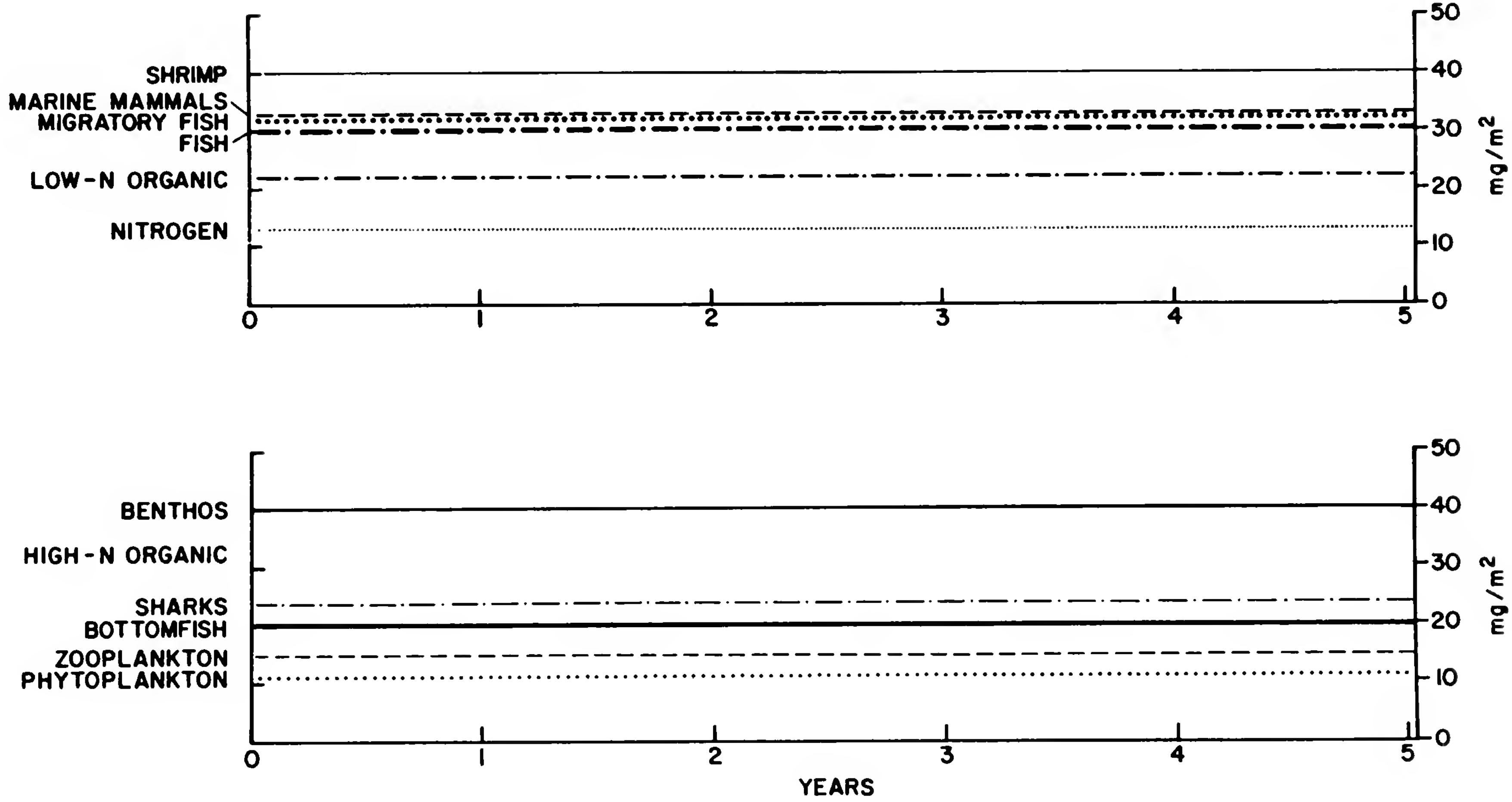

Figure 2. Plotted simulation of standing stocks of model system under present conditions of handling bycatch (assuming model system is in steady-state under these conditions) (test #1). NOTE: Numbers on the horizontal axis represent biweeks (15-day periods). There are 24 biweeks in a model year. The simulation spans 120 biweeks, or five years. The standing stock of each compartment is scaled to fit the ordinate. Scaling factors are as follows: nitrogen, 0.01; phytoplankton, 0.01; low-nitrogen organic material, 0.0001; high-nitrogen organic material, 1; zooplankton, 0.1; pelagic fish, 0.01; benthos, 0.005; shrimp, 0.5; bottomfish, 0.01; migratory fish, 1; marine mammals (large predators), 5; sharks (large scavengers), 1.

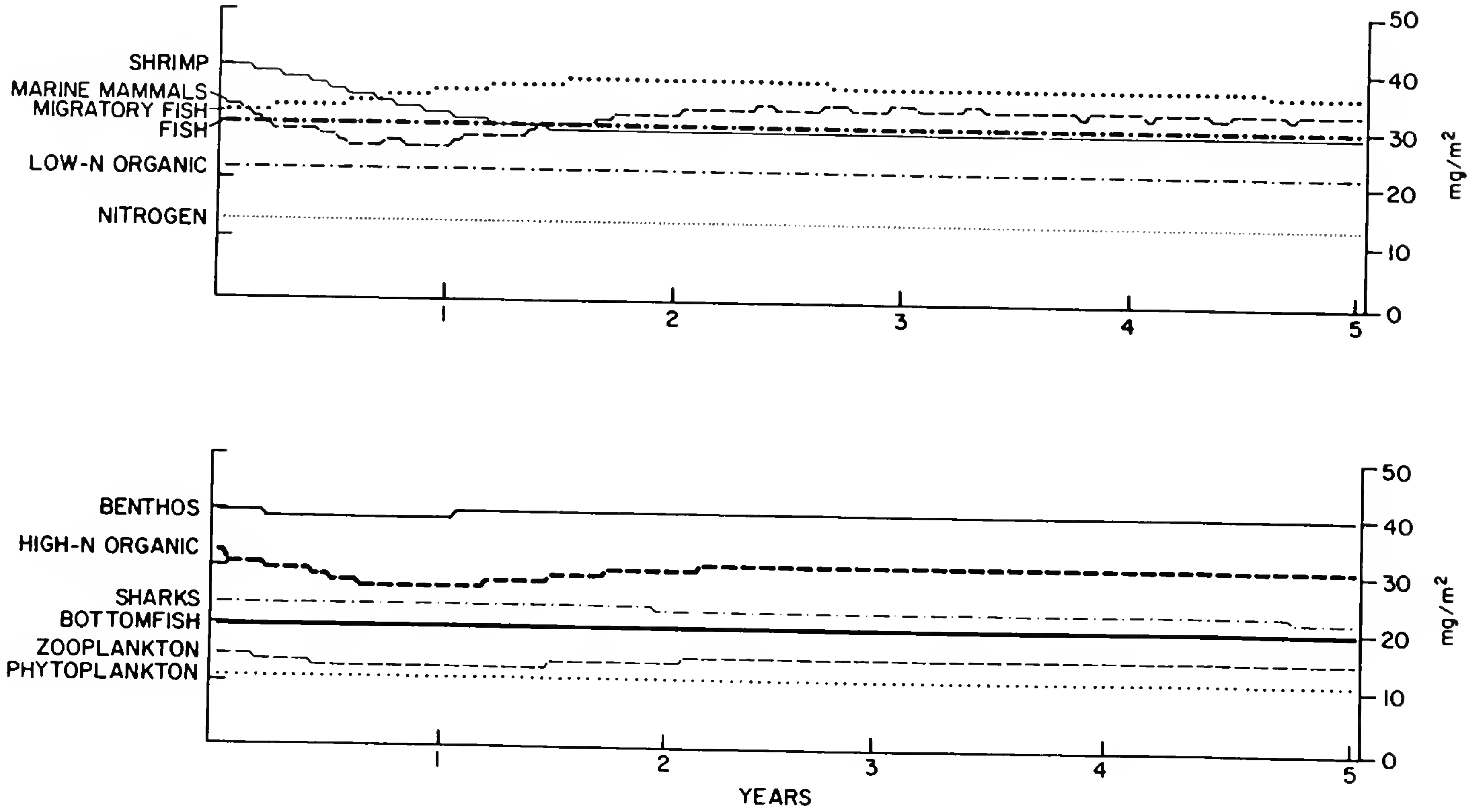

Figure 3. Plotted simulation of standing stocks of model system with one half of the bycatch utilized (test #2). NOTE: Numbers on the horizontal axis represent biweeks (15-day periods). There are 24 biweeks in a model year. The simulation spans 120 biweeks, or five years. Numbers on the ordinate represent standing stock. The standing stock of each compartment is scaled to fit the ordinate. Scaling factors are as follows: nitrogen, 0.01; phytoplankton, 0.01; low-nitrogen organic material, 0.0001; high-nitrogen organic material, 1; zooplankton, 0.1; pelagic fish, 0.01; benthos, 0.005; shrimp, 0.5; bottomfish, 0.01; migratory fish, 1; marine mammals (large predators), 5; sharks (large scavengers), 1.

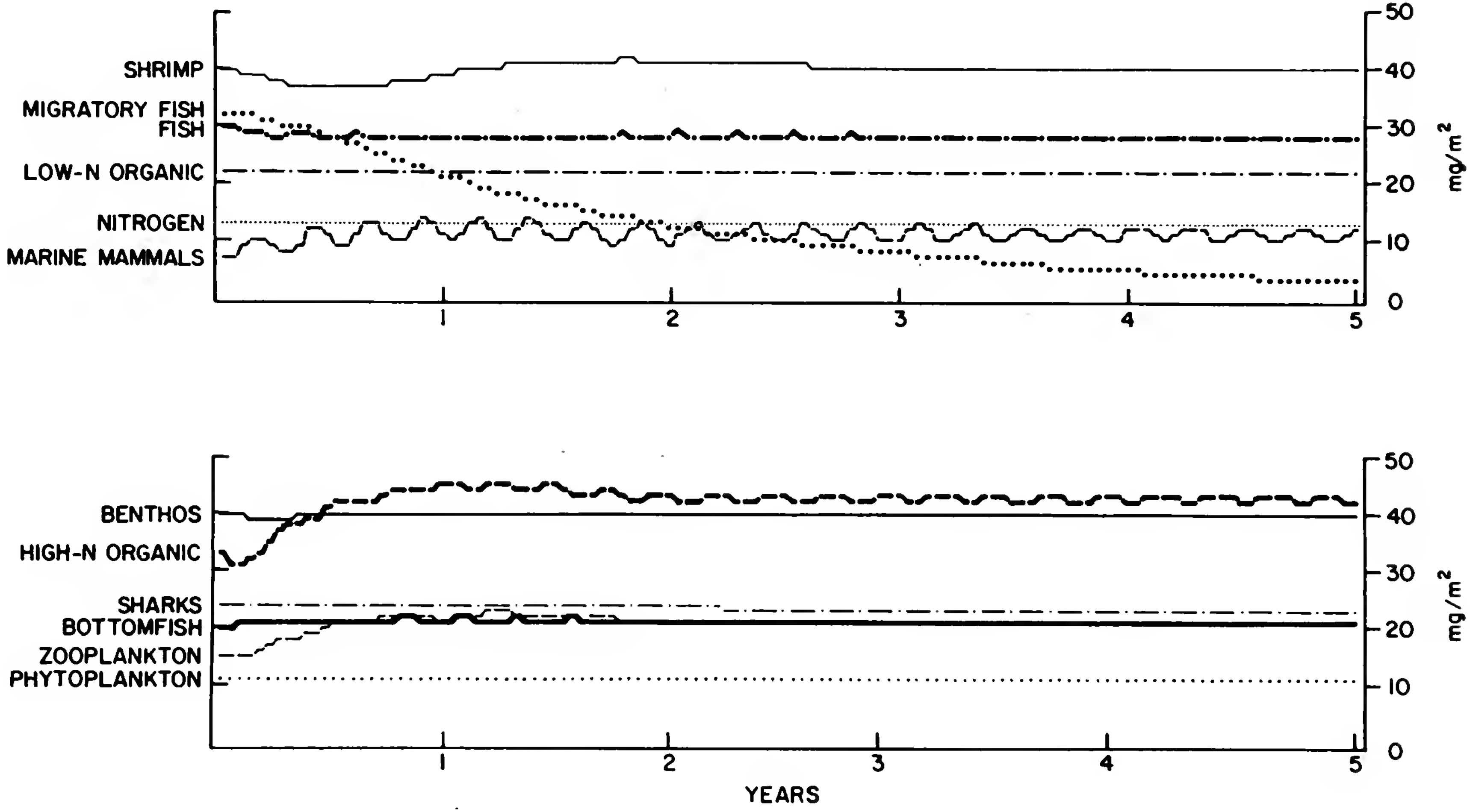

Figure 4. Plotted simulation of standing stocks of model system with the fish catch efficiency of shrimp trawls reduced by one half (test #3). NOTE: Numbers on the horizontal axis represent biweeks (15-day periods). There are 24 biweeks in a model year. The simulation spans 120 biweeks, or five years. Numbers on the ordinate represent standing stock. The standing stock of each compartment is scaled to fit the ordinate. Scaling factors are as follows: nitrogen, 0.01; phytoplankton, 0.01; low-nitrogen organic material, 0.0001; high-nitrogen organic material, 1; zooplankton, 0.1; pelagic fish, 0.01; benthos, 0.005; shrimp, 0.5; bottomfish, 0.01; migratory fish, 1; marine mammals (large predators), 1; sharks (large scavengers), 1.

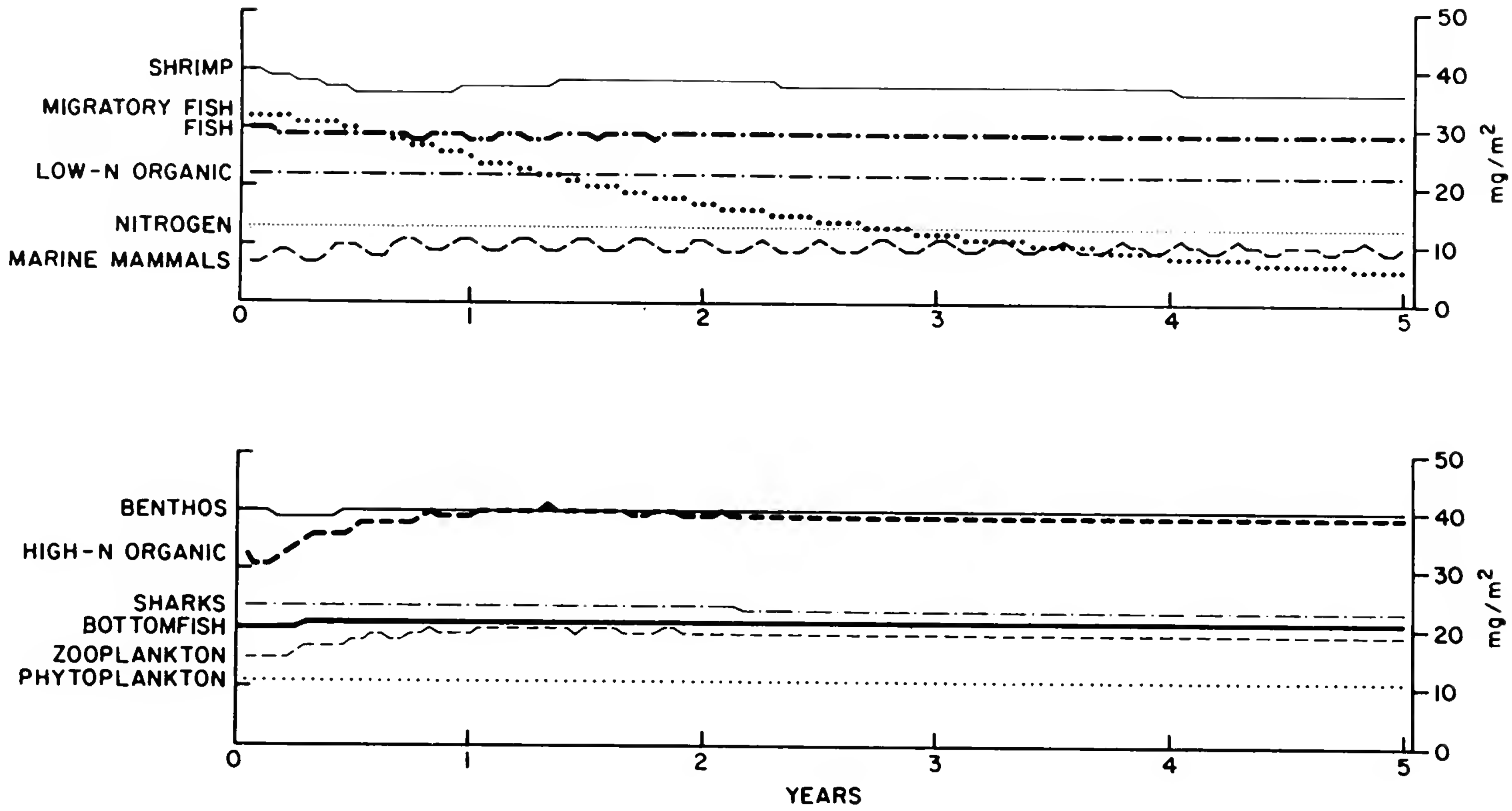

Figure 5. Plotted simulation of standing stocks of model system with fish catch efficiency of shrimp trawls reduced by one half and effort of directed groundfish fleet doubled (test #4). NOTE: Numbers on the horizontal axis represent biweeks (15-day periods). There are 24 biweeks in a model year. The simulation spans 120 biweeks, or five years. Numbers on the ordinate represent standing stock. The standing stock of each compartment is scaled to fit the ordinate. Scaling factors are as follows: nitrogen, 0.01; phytoplankton, 0.01; low-nitrogen organic material, 0.0001; high-nitrogen organic material, 1; zooplankton, 0.1; pelagic fish, 0.01; benthos, 0.005; shrimp, 0.5; bottomfish, 0.01; migratory fish, 1; marine mammals (large predators), 1; sharks (large scavengers), 1.

Table 2. Simulated annual harvests over five-year period under tested harvesting strategies.

	Present Strategy	Selective Gear[a]	Utilize Bycatch[b]	Selective Gear[a] Increase Fishing[c]
Year 1				
Pelagic fish	1,483	1,420	1,499	1,442
Shrimp	79.41	75.68	71.67	74.33
Bottomfish	292.4	308.3	288.3	605.7
Migratory fish	7.52	6.37	7.84	6.76
Bycatch	2,047	1,079	2,018	1,060
Discards	2,046	1,079	1,009	1,060
Total landings	1,862	1,810	2,875	2,128
Year 2				
Pelagic fish	1,483	1,409	1,489	1,431
Shrimp	79.41	81.96	59.08	75.25
Bottomfish	292.4	312.1	290.6	612.7
Migratory fish	7.52	3.73	8.68	4.72
Bycatch	2,047	1,092	2,034	1,072
Discards	2,046	1,092	1,017	1,072
Total landings	1,862	1,807	2,864	2,124
Year 3				
Pelagic fish	1,483	1,417	1,479	1,436
Shrimp	79.41	81.20	57.74	74.61
Bottomfish	292.4	311.3	292.5	612.2
Migratory fish	7.52	2.251	8.74	3.30
Bycatch	2,047	1,090	2,048	1,071
Discards	2,046	1,089	1,023	1,071
Total landings	1,862	1,812	2,861	2,126
Year 4				
Pelagic fish	1,483	1,415	1,481	1,435
Shrimp	79.41	79.92	58.27	73.34
Bottomfish	292.4	312.2	292.2	613.8
Migratory fish	7.52	1.38	8.58	2.32
Bycatch	2,047	1,093	2,046	1,074
Discards	2,046	1,092	1,023	1,074
Total landings	1,862	1,808	2,863	2,124
Year 5				
Pelagic fish	1,483	1,415	1,481	1,435
Shrimp	79.41	79.37	58.12	72.67
Bottomfish	292.4	312.5	292.2	614.6
Migratory fish	7.52	.84	8.49	1.6
Bycatch	2,047	1,094	2,046	1,076
Discards	2,046	1,093	1,023	1,075
Total landings	1,862	1,807	2,863	2,124

[a] Reducing catchability of shrimp trawls for groundfish by 50 percent.

[b] Utilizing 50 percent of groundfish bycatch.

[c] Increasing directed fishing effort on groundfish by 50 percent.

Table 3. Total annual respiration for organic compartments under different harvesting strategies.

Compartment	Present Strategy	Selective Gear[a]	Utilize Bycatch[b]	Selective Gear[a] Increase Fishing[c]
Year 1				
Phytoplankton	$3.11\mathrm{x}10^4$	$3.08\mathrm{x}10^4$	$3.09\mathrm{x}10^4$	$3.12\mathrm{x}10^4$
Low-N Organic	$1.47\mathrm{x}10^6$	$1.47\mathrm{x}10^4$	$1.47\mathrm{x}10^4$	$1.47\mathrm{x}10^4$
High-N Organic	$2.53\mathrm{x}10^3$	$3.00\mathrm{x}10^4$	$2.75\mathrm{x}10^4$	$2.18\mathrm{x}10^4$
Zooplankton	$1.68\mathrm{x}10^4$	$2.24\mathrm{x}10^4$	$2.03\mathrm{x}10^4$	$1.56\mathrm{x}10^4$
Pelagic Fish	$2.14\mathrm{x}10^4$	$2.05\mathrm{x}10^4$	$2.08\mathrm{x}10^4$	$2.14\mathrm{x}10^4$
Benthos	$2.33\mathrm{x}10^5$	$2.33\mathrm{x}10^5$	$2.32\mathrm{x}10^5$	$2.32\mathrm{x}10^5$
Shrimp	$1.18\mathrm{x}10^3$	$1.13\mathrm{x}10^3$	$1.11\mathrm{x}10^3$	$0.88\mathrm{x}10^3$
Bottomfish	$1.30\mathrm{x}10^4$	$1.37\mathrm{x}10^4$	$1.34\mathrm{x}10^4$	$1.29\mathrm{x}10^4$
Migratory Fish	$7.00\mathrm{x}10^1$	$5.93\mathrm{x}10^1$	$6.29\mathrm{x}10^1$	$8.07\mathrm{x}10^1$
Marine Mammals	$3.28\mathrm{x}10^3$	$5.21\mathrm{x}10^3$	$4.54\mathrm{x}10^3$	$2.99\mathrm{x}10^3$
Large Scavengers	$5.96\mathrm{x}10^1$	$5.93\mathrm{x}10^1$	$5.93\mathrm{x}10^1$	$5.82\mathrm{x}10^1$
Year 2				
Phytoplankton	$3.11\mathrm{x}10^4$	$3.07\mathrm{x}10^4$	$3.08\mathrm{x}10^4$	$3.11\mathrm{x}10^4$
Low-N Organic	$1.47\mathrm{x}10^6$	$1.47\mathrm{x}10^4$	$1.47\mathrm{x}10^4$	$1.47\mathrm{x}10^4$
High-N Organic	$2.53\mathrm{x}10^3$	$3.32\mathrm{x}10^4$	$3.02\mathrm{x}10^4$	$2.33\mathrm{x}10^4$
Zooplankton	$1.68\mathrm{x}10^4$	$2.52\mathrm{x}10^4$	$2.26\mathrm{x}10^4$	$1.70\mathrm{x}10^4$
Pelagic Fish	$2.14\mathrm{x}10^4$	$2.03\mathrm{x}10^4$	$2.06\mathrm{x}10^4$	$2.13\mathrm{x}10^4$
Benthos	$2.33\mathrm{x}10^5$	$2.34\mathrm{x}10^5$	$2.33\mathrm{x}10^5$	$2.34\mathrm{x}10^5$
Shrimp	$1.18\mathrm{x}10^3$	$1.22\mathrm{x}10^3$	$1.12\mathrm{x}10^3$	$0.86\mathrm{x}10^3$
Bottomfish	$1.30\mathrm{x}10^4$	$1.38\mathrm{x}10^4$	$1.36\mathrm{x}10^4$	$1.30\mathrm{x}10^4$
Migratory Fish	$7.00\mathrm{x}10^1$	$3.47\mathrm{x}10^1$	$4.39\mathrm{x}10^1$	$8.13\mathrm{x}10^1$
Marine Mammals	$3.28\mathrm{x}10^3$	$5.69\mathrm{x}10^3$	$4.98\mathrm{x}10^3$	$3.31\mathrm{x}10^3$
Large Scavengers	$5.96\mathrm{x}10^1$	$5.85\mathrm{x}10^1$	$5.84\mathrm{x}10^1$	$5.73\mathrm{x}10^1$

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[a] Reducing catchability of shrimp trawls for groundfish by 50 percent.

[b] Utilizing 50 percent of groundfish bycatch.

[c] Increasing directed fishing effort on groundfish by 50 percent.

shrimp by a factor of 100. Simulation results indicated that shrimp standing stock and the shrimp harvest were more sensitive to the reduction in bottomfish fishing mortality rate caused by trawls with a reduced efficiency for catching fish when predator selectivity against shrimp was eliminated. When there was no selectivity against shrimp by bottomfish (weighting factors set at 1 and predation rate increased by a factor of 100), the shrimp stock did not fully recover in the 5-year period of the simulation (fig. 6).

In a second sensitivity test, the initial standing stock of bottomfish was reduced by one-half, which changed the relationship of standing stock of this compartment to that of the others. This change made little difference to the response of the model system to management tests, except that, under conditions that induced expansion of bottomfish standing stock, the expansion occurred more rapidly. For the third sensitivity test, a feeding link was established between discards and marine mammals. Discards were weighted equally with other mammal food sources. This change made almost no appreciable change in simulation results, except that the oscillation in marine mammals that typified the management-test simulations was eliminated from the simulation of the effect of utilizing half the bycatch.

The simulation plots revealed oscillations in many of the standing stocks under non-steady-state conditions. These oscillations were intrinsic to the model system and not due to oscillating system inputs, because all inputs to the model system were constants. The frequency of oscillation of the marine mammal standing stock (fig. 3) is obviously an artifact of the model and cannot reflect the real world, because marine mammals have slow maturation rates and long gestation periods and the standing stock of marine mammals obviously would not fluctuate several times a year. Fluctuations such as those seen in other standing stocks of the model system are plausible, although, in the real world, seasonal variations in solar radiation and river discharge would superimpose seasonal cycles on any intrinsic cycles of the system.

DISCUSSION

To help understand the basis for the changes observed in the plots, calculations were made of total annual inflows and outflows to several of the compartments under the test conditions. These annual totals are shown in tables 4 through 12.

Model simulation results indicate that reducing discards by either proposed strategy could have some detrimental effect on shrimp harvests initially, but readjustments in the system will allow the shrimp stock and shrimp harvest to recover within 2 years under the option of using trawls that reduced fish catchability relative to that of shrimp, if predation pressure of bottomfish on shrimp is low (selectivity weighting factor against shrimp of 0.01 or less). The simulations of the model suggest that bottomfish harvesting strategies could influence shrimp by more mechanisms than those considered in the question originally posed in the introduction. Some of the influences observed in the simulations were counter to what was expected.

Although nitrogen remineralization was decreased when the discard rate was reduced through bycatch utilization, the rate of nitrogen remineralization was greatest when bottomfish fishing mortality was reduced by one-half through the

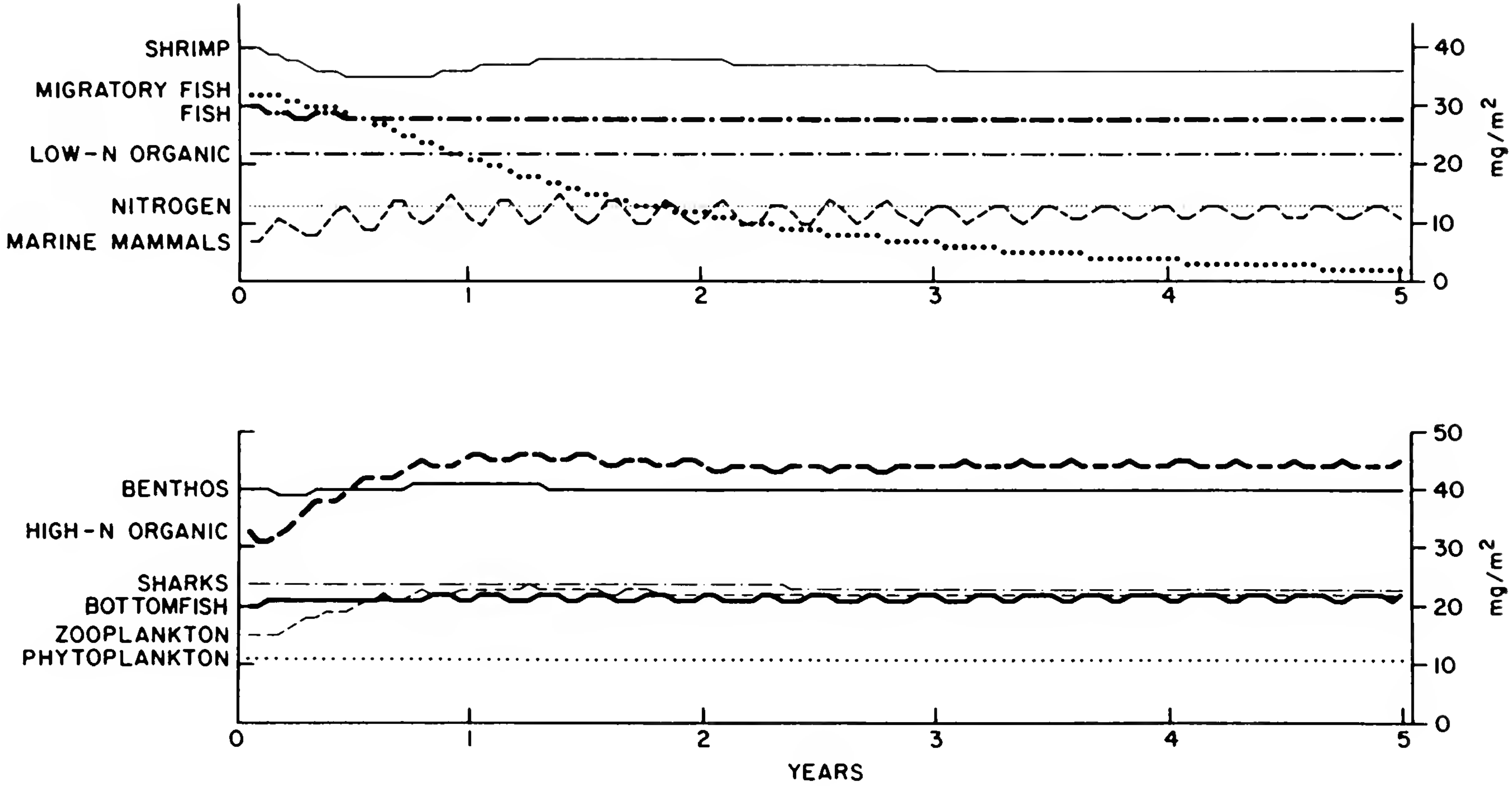

Figure 6. Plotted simulation of standing stocks of model system with fish catch efficiency of shrimp trawls reduced by one half and no selectivity of groundfish against shrimp (weighting factor increased by 100). NOTE: Numbers on the horizontal axis represent biweeks (15-day periods). There are 24 biweeks in a model year. The simulation spans 120 biweeks, or five years. Numbers on the ordinate represent standing stock. The standing stock of each compartment is scaled to fit the ordinate. Scaling factors are as follows: nitrogen, 0.01; phytoplankton, 0.01; low-nitrogen organic material, 0.0001; high-nitrogen organic material, 1; zooplankton, 0.1; pelagic fish, 0.01; benthos, 0.005; shrimp, 0.5; bottomfish, 0.01; migratory fish, 1; marine mammals (large predators), 1; sharks (large scavengers), 1.

Table 4. First-year inorganic nitrogen budget.[a]

	Management Strategies		
	Present	Selective Gear	Utilize Bycatch
INPUTS			
Imported (J(1))	453,600	453,600	453,600
Recycled (N(3-12))	85,491	85,662	85,399
OUTPUTS			
Exported (FLN)	85,919	85,920	85,878
Uptaken (P(1,2))	39,895	40,065	39,842

[a] Units are milligrams nitrogen per square meter per year.

Table 5. First-year energy budget for phytoplankton.[a]

	Management Strategies		
	Present	Selective Gear	Utilize Bycatch
INPUTS			
Gross primary production (J(2))	975,993	975,983	975,719
OUTPUTS			
Grazing by zooplankton (P(2,5))	33,372	44,081	30,637
Grazing by pelagic fish (P(2,6))	50,527	47,951	51,168
Phytoplankton rain (P(2,3))	581,028	575,739	582,218
Respiration (R(2))	311,051	308,211	311,676

[a] Units are milligrams dry organic matter per square meter per year.

Table 6. First-year energy budget for high-nitrogen organic compartment.[a]

	Management Strategies		
	Present	Selective Gear	Utilize Bycatch
INPUTS			
Fecal pellet rain (F(5))	10,012	13,224	9,191
Discarded fish (D)	2,046	1,079	1,009
OUTPUTS			
Ingestion by benthos (P(4,7))	9,436	11,179	7,964
Ingestion by shrimp (P(4,8))	12.99	14.65	10.07
Ingestion by groundfish (P(4,9))	81.47	102.0	68.78
Respiration (R(4))	2,527	2,995	2,163
COMPETITION RATIOS			
P(4,8)/P(4,7)	.001377	.001310	.001260
P(4,8)/P(4,9)	.1594	.1436	.1464

[a] Units are milligrams dry weight per square meter per year.

Table 7. First-year energy budget for zooplankton compartment.[a]

	Management Strategies		
	Present	Selective Gear	Utilize Bycatch
INPUTS			
Ingestion of phytoplankton (P(2,5))	33,372	44,081	30,637
OUTPUTS			
Predation by pelagic fish (P(5,6))	6,553	8,353	6,066
Respiration (R(5))	16,808	22,421	15,400
Egestion (fecal pellet production) (F(5))	10,012	13,224	9,191
RATIOS			
P(2,5)/R(5)	1.985	1.966	1.989
P(5,6)/R(5)	.3899	.3726	.3939
P(2,5)/P(5,6)	5.093	5.277	5.051

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[a] Units are milligrams dry weight per square meter per year.

Table 8. First-year energy budget for pelagic forage fish compartment.[a]

	Management Strategies		
	Present	Selective Gear	Utilize Bycatch
INPUTS			
Ingestion of phytoplankton (P(2,5))	50,527	47,951	51,168
Ingestion of zooplankton (P(5,6))	6,553	8,353	6,066
OUTPUTS			
Harvesting (H(6))	1,483	1,420	1,499
Predation by migratory pelagics (P(6,10))	68.35	55,54	71.95
Predation by marine mammals (P(6,11))	2,208	3,351	1,900
Predation by large scavengers (P(6,12))	39.64	37.76	39.71
Respiration (R(6))	21,355	20,450	21,581
Egestion (F(6))	31,929	31,153	32,104

[a] Units are milligrams dry weight per square meter per year.

Table 9. First-year energy budget for marine mammals compartment.[a]

	Management Strategies		
	Present	Selective Gear	Utilize Bycatch
INPUTS			
Ingestion of pelagic fish (P(6,11))	2,208	3,351	1,900
Ingestion of groundfish (P(9,11))	1,489	2,499	1,251
Ingestion of migratory pelagics (P(10,11))	24.12	31.96	21.35
OUTPUTS			
Predation by large scavengers (P(11,12))	.00082	.00139	.000745
Respiration (R(11))	3,279	5,209	2,792

---

[a] Units are milligrams dry weight per square meter per year.

Table 10. First-year energy budget for benthic compartment.[a]

	Management Strategies		
	Present	Selective Gear	Utilize Bycatch
INPUTS			
Ingestion of low-N organic (P(3,7))	494,859	492,884	488,868
Ingestion of high-N organic (P(4,7)	9,436	11,179	7,964
OUTPUTS			
Predation by shrimp (P(7,8))	3,112	2,960	2,768
Predation by groundfish (P(7,9))	19,523	20,557	18,974
Respiration (R(7))	233,272	232,904	229,962
Egestion (F(7))	248,356	247,560	245,231
COMPETITION RATIO			
P(7,8)/P(7,9)	.1594	.1440	.1459

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[a] Units are milligrams dry weight per square meter per year.

Table 11. First-year energy budget for shrimp compartment.[a]

	Management Strategies		
	Present	Selective Gear	Utilize Bycatch
INPUTS			
Ingestion of low-N organic (P(3,8))	85.12	80.93	76.98
Ingestion of high-N organic (P(4,8))	12.99	14.65	10.07
Ingestion of benthos (P(7,8))	3,112	2,960	2,768
OUTPUTS			
Harvesting (H(8))	79.41	75.68	71.67
Predation by groundfish (P(8,9))	1.938	1.947	1.725
Predation by migratory pelagics (P(8,10))	1.830	1.478	1.716
Respiration (R(8))	1,184	1,128	1,068
Egestion (F(8))	1,943	1,850	1,729

---

[a] Units are milligrams dry weight per square meter per year.

Table 12. Second-year energy budget for shrimp compartment.[a]

	Management Strategies		
	Present	Selective Gear	Utilize Bycatch
INPUTS			
Ingestion of low-N organic (P(3,8))	85.12	87.37	63.42
Ingestion of high-N organic (P(4,8))	12.99	17.60	8.32
Ingestion of benthos (P(7,8))	3,112	3,215	2,303
OUTPUTS			
Harvesting (H(8))	79.41	81.96	14.89
Predation by groundfish (P(8,9))	1.938	2.135	1.433
Predation by migratory pelagics (P(8,10))	1.830	.9343	1.570
Respiration (R(8))	1,184	1,222	880.6
Egestion (F(8))	1,943	2,010	1,437

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[a] Units are milligrams dry weight per square meter per year.

use of shrimp trawls with a reduced catch efficiency for fish, despite a reduction in the rate that dead fish were returned to the system to close the nutrient cycle (table 4). Annual gross primary productivity was slightly decreased when half the bycatch was utilized, decreasing the quantity of dead fish returned to the model system. However, annual gross primary productivity did not increase when trawls that reduced fish catchability relative to that of shrimp were used to decrease discards, despite the fact that nutrient regeneration rates were highest under this condition (table 5). Perhaps this was because saturation concentrations for nitrogen with respect to phytoplankton photosynthesis were more frequently exceeded under the condition of half the shrimp trawl efficiency for catching fish than under the present strategy for handling discards.

The direct effect of discard rate on the standing stock of high-nitrogen organic material was small, and the rate of zooplankton fecal pellet deposition was so great by comparison that it overshadowed discarding as a source of high-nitrogen organic material (table 6). Although decreasing shrimp trawl efficiency for catching fish resulted in a decrease in the rate of deposition of dead fish, this was greatly outweighed by an increase in zooplankton fecal pellet production that was an indirect result of decreasing shrimp trawl efficiency for catching fish. Both the rate of inflows and rate of outflows to the zooplankton compartment were influenced by the discard rate and the bottomfish fishing mortality rate (table 7). Predation of pelagic forage fish on zooplankton was decreased when the standing stock of pelagic forage fish was depressed by predation of marine mammals, which appears to have been due to the greater abundance of bottomfish, which are also their prey (table 9). The rate of fecal pellet production increased when zooplankton standing stock increased.

Although predation by bottomfish did exert some detrimental influence on shrimp stocks and shrimp harvests, this influence was minor when compared to the influence of competition for common food between shrimp and bottomfish and between shrimp and benthos (tables 10 and 11). An increased availability of food for shrimp outweighed the increased competition for that food in the second year (table 12). How the shrimp stock in the model system reacted to a change in bottomfish standing stock was determined by the balance between production of fecal pellets, which provide food for shrimp, benthos, and bottomfish, and the pressure of competition for food with both benthos and bottomfish.

Initially, when the fishing mortality of bottomfish was reduced with shrimp trawls with half the efficiency for catching fish, shrimp standing stock and the shrimp harvest reacted negatively. However, shrimp standing stock and the shrimp harvest recovered when the supply of fecal pellets increased to the point where an increased food supply overcame increased competition for food from the larger standing stock of bottomfish.

Under any management strategy, dead fish formed a small component of the high-nitrogen organic compartment, relative to zooplankton fecal pellets, which represented 82.5 percent of compartment standing stock at steady-state. Furthermore, the nitrogen released from this compartment in decomposition was infinitesimal compared to that released in the decomposition of low-nitrogen organic material, and it was even small compared to that released in animal excrement. Nevertheless, utilizing the bycatch, which decreased the rate of discarding without increasing the standing stock of living bottomfish, did affect shrimp stocks and shrimp harvests in the model system. Conclusions from the simulations differ in this detail from conclusions of Browder (1981) and

Sheridan et al. (1981). The conclusion made in the 1981 paper was that shrimp production would not be affected by decreasing the discard rate. This conclusion was based only on relative rates of nitrogen remineralization from different sources, and food flows to shrimp from different sources. In the simulations, the response of phytoplankton production to changes in water nitrogen concentration was dependent upon the Michaelis-Menten half-saturation constant, relative to the concentration of nitrogen in the water, rather than on the relative rates of release of nitrogen from the various compartments. The simulation exercise demonstrated that it is possible for a change in a nitrogen inflow that is very small relative to other flows to affect model results. The model, however, does not include nitrogen fixation and denitrification. These flows are difficult to quantify. If they are large in this system relative to the nitrogen flows included in the model, they may influence the way that variations in other nitrogen flows affect the system.

Model results were sensitive to the weighting factor determining the intensity of bottomfish predation on shrimp. Model results appeared somewhat sensitive to the apportionment of nitrogen between the excrement and fecal material of bottomfish and possibly also of zooplankton and other animals. Results are undoubtedly also sensitive to the Michaelis-Menten half-saturation constant that relates phytoplankton production to the concentration of nitrogen in the water. Although values for these parameters that were used in the model had some basis in the literature, none were specific to the north-central Gulf of Mexico. No specific sensitivity tests have yet been run on these parameters. The impact of variations in these values on model results and conclusions has not been fully explored.

Do model results apply to the real-world system? In the development of an ecosystem model, there are at least four levels of procedure at which variations can affect model results: (1) assumptions of the conceptual model, (2) translation of the conceptual model into mathematical equations, (3) model quantification, and (4) model computerization. Ecosystem models are difficult to validate, but there are a number of things that can be done to improve confidence in an ecosystem model and its results. One is to test the validity of the assumptions independently of the model and to assess the dependence of the model on these assumptions. A second is to test the sensitivity of the model to variation in initial values, rate-coefficients, and other constants, such as those singled out above (bottomfish selectivity weighting factor for shrimp relative to alternative prey, apportionment of waste nitrogen between feces and excrement, and Michaelis-Menten half-saturation constant relating gross primary productivity to water nitrogen concentration). Quantification of a model requires many rough estimations. It is important to know whether any of these have major effects on model results, not only to evaluate the present reliability of the model but also to determine what further work is needed to improve the model's reliability. Sensitivity testing is done by varying values one at a time and comparing simulation results. Sensitivity test results can lead to a more intensive search of existing literature, which may yield improved estimates, or to field or laboratory studies designed to obtain the critical information that is needed to adequately answer the question.

Sensitivity testing is needed not only for model input values but also for model structure. The model is, of necessity, a simplification. Does the addition of structural detail significantly alter system results? If so, this needs to be known. Finally, errors in program coding can cause responses that cannot possibly reflect the real-world situation. Working with a model over a period of time increases the likelihood that errors such as these will be found.

A model is a simplistic interpretation of the system. Whether or not this model reflects the response of the real system to the conditions we are testing remains to be seen. It is certain, however, that even a simple system such as this 12-compartment model has complex and unanticipated reactions to harvesting pressure and its variation, due to multiple indirect effects operating along several pathways. It is therefore likely that the response of the real system to harvesting pressure and its variation is even more complicated and difficult to predict. This is the important message for management of the modeling effort at this point.

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MATHEMATICAL MODEL OF OXYGEN DEPLETION IN THE NEW YORK BIGHT: AN ANALYSIS OF PHYSICAL, BIOLOGICAL, AND CHEMICAL FACTORS IN 1975 AND 1976

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INTRODUCTION

The increasing use of the sea for urban waste disposal requires an understanding of the subtle biological impacts of nutrients, organic matter, and other pollutants discharged into coastal waters. Significant water quality problems frequently result from the transient response of natural waters to some perturbation such as changes in circulation patterns, climatic conditions, nutrient enrichment, organic carbon loading, or phytoplankton species composition. In particular, an extensive bloom of the dinoflagellate Ceratium tripos developed throughout the Middle Atlantic Bight continental shelf from January through July 1976. From July through September 1976 bottom water oxygen was also progressively depleted to less than 2 ml $0_2\ell^{-1}$ over a large area (8600 km^2) of the New Jersey coastal region from the 20 m to 60 m isobath (fig. 1). Presence of this anoxic region resulted in the formation of hydrogen sulfide and mass mortalities of demersal fishes and shellfish (Swanson and Sindermann 1979). The occurrence of the C. tripos bloom, and the onset of anoxic conditions, suggested that the abundance of C. tripos had imposed an unusually large oxygen demand on the subpycnocline layer off the New Jersey coast and was causally related to the anoxic episode (Malone et al. 1979; Falkowski and Howe 1976).

A marine ecosystem model, designed to evaluate the relative significance of natural physical and biological processes and anthropogenic waste inputs on oxygen depletion in the New York Bight, described the continental shelf of the Bight as a two-layer system separated by the pycnocline (Stoddard 1983). The model equations specified the interactions of carbon, oxygen, and nitrogen in an analysis of oxygen depletion, nutrient dynamics, and phytoplankton distribution during the stratified summer season.

A comparison of a year of high bottom oxygen content (1975) and the large-scale anoxic episode during 1976 is the problem setting for the analysis. Previous studies of the 1976 anoxic episode (Falkowski et al. 1980; Swanson and Sindermann 1979; Falkowski and Howe 1976) examined the influence of urban waste inputs, and climatological, physical, chemical, and biological forcing on oxygen depletion in the New York Bight. The model quantitatively incorporates the hypotheses presented in these studies coupled with a quasi-time-dependent circulation sub-model (Han et al. 1980) to assess the relative significance of the various forcing terms on the development of anoxia.

RESULTS

Verification of the model included a synthesis of physical, chemical, and biological data collected by Brookhaven National Laboratory and other institutions during the MESA New York Bight Project. In general, the model is capable of simulating most of the ecological behavior of the Bight where the calculations are in reasonable agreement with observations during a year of high bottom oxygen (1975) (fig. 2), and during the anoxic episode in 1976 (fig. 3). The significance of transport processes on water quality distributions in the Bight are clearly demonstrated in an analysis of: the component sources and sinks of nitrogen, Ceratium, and dissolved oxygen for the 1976 verification case; and onshore subpycnocline penetration of high nitrate across the shelf during June - October for the 1975 verification case. The model also clearly demonstrates the influence of the flow field reversal on the accumulation of Ceratium and other particulate substances during June - July 1976 (fig. 4).

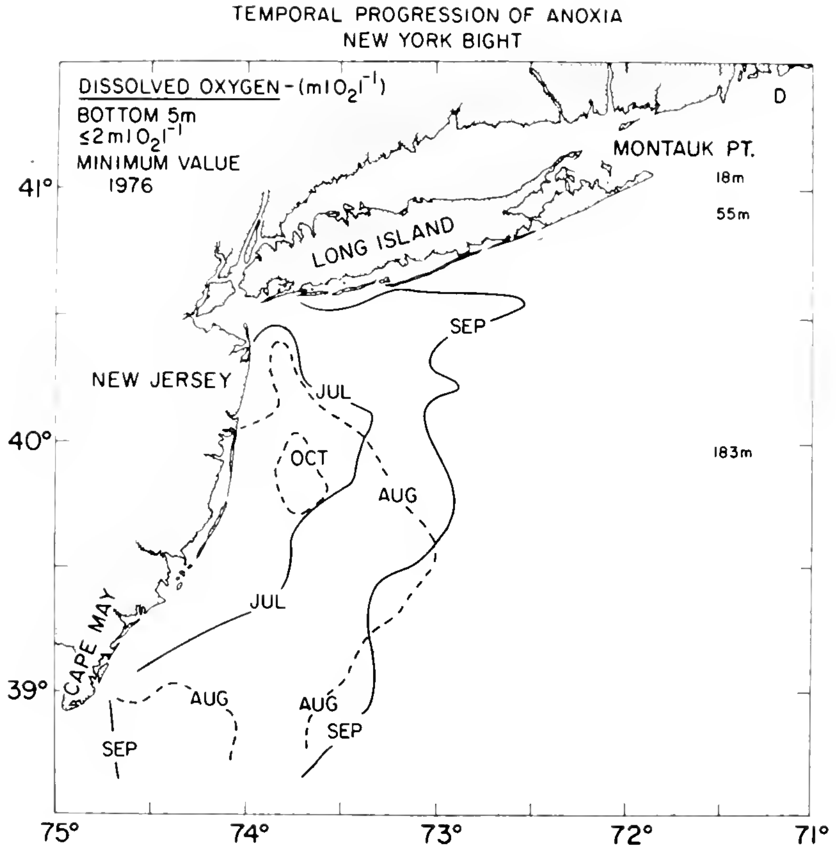

Figure 1. Temporal progression of anoxia in the New York Bight: July-October 1976

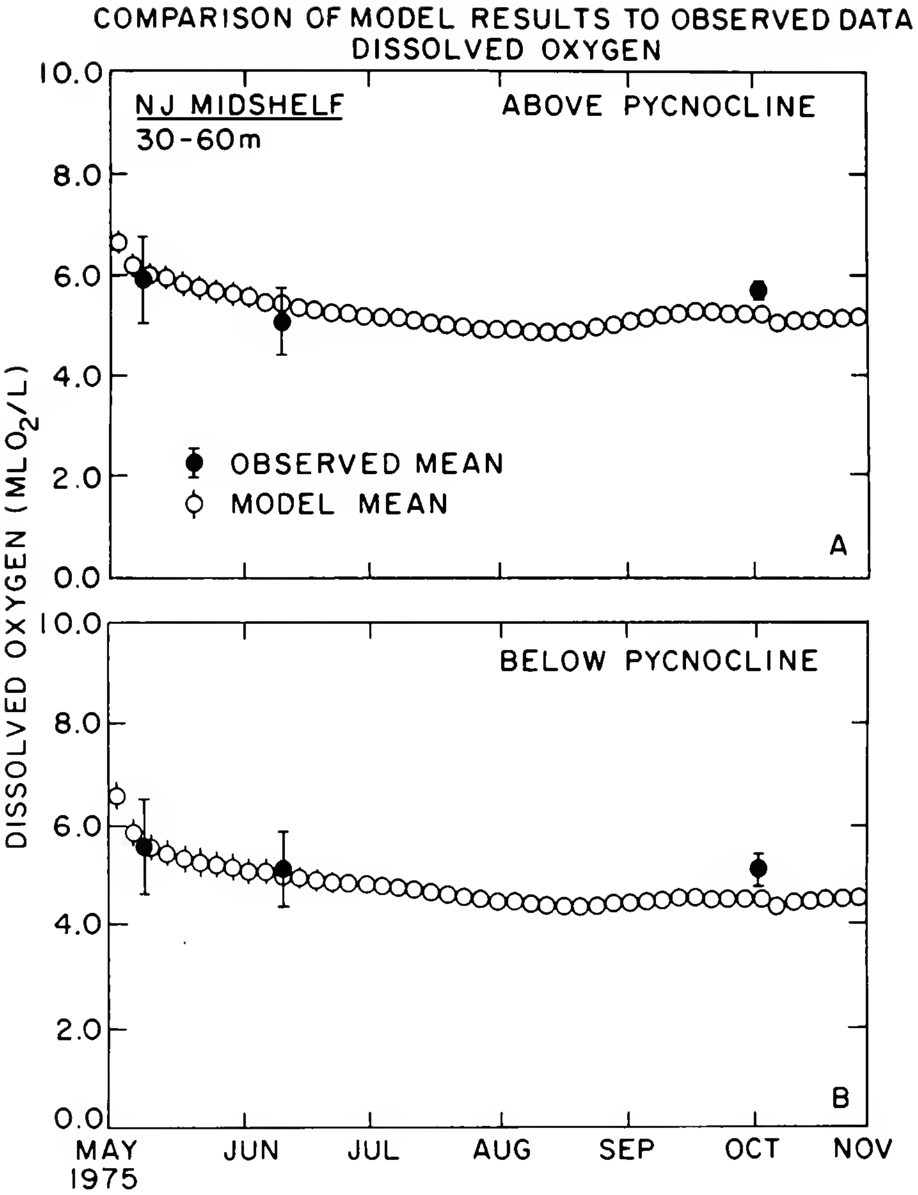

Figure 2. Temporal comparison of calculated and observed dissolved oxygen for the New Jersey midshelf: 1975.

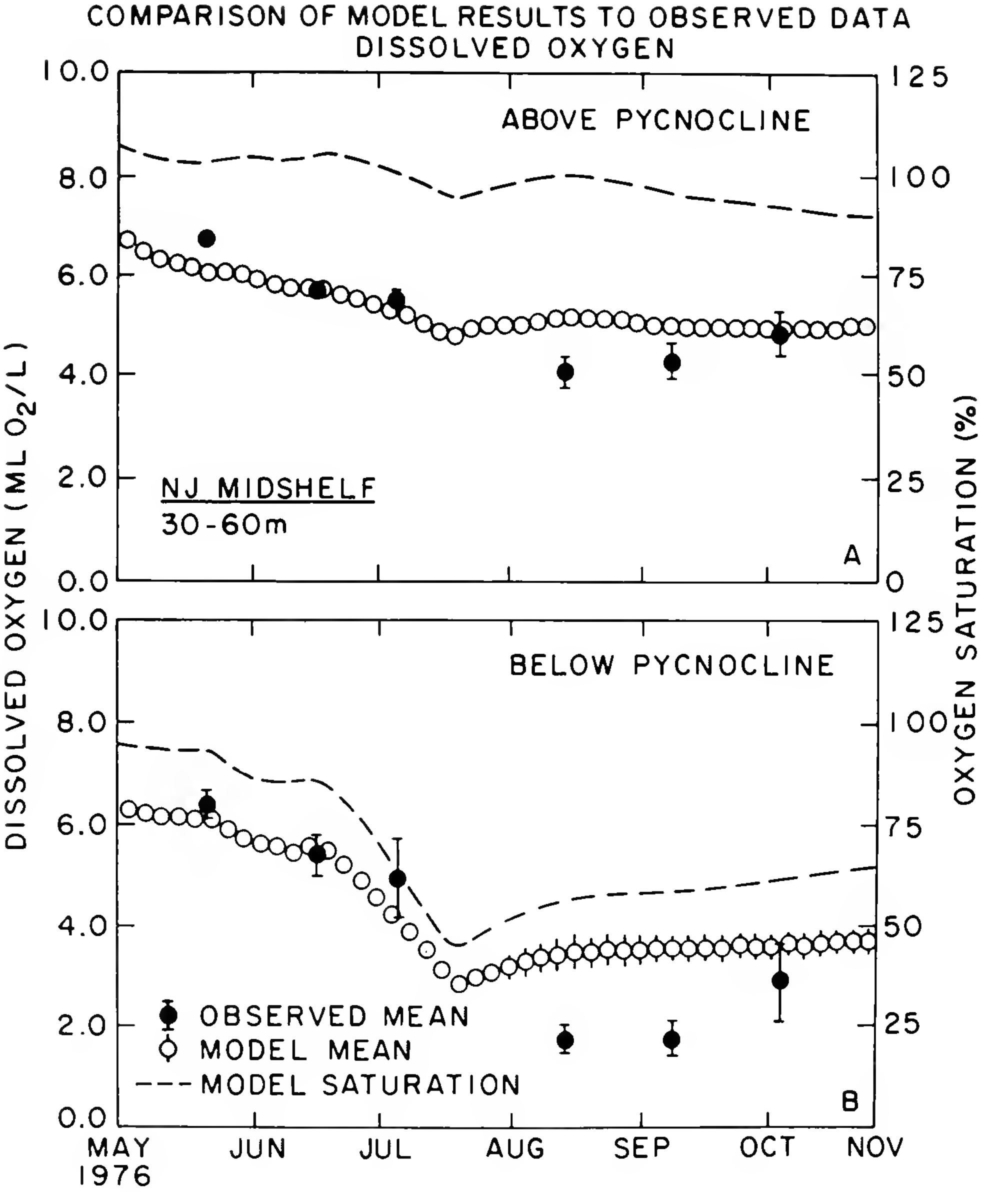

Figure 3. Temporal comparison of calculated and observed dissolved oxygen for the New Jersey midshelf: 1976.

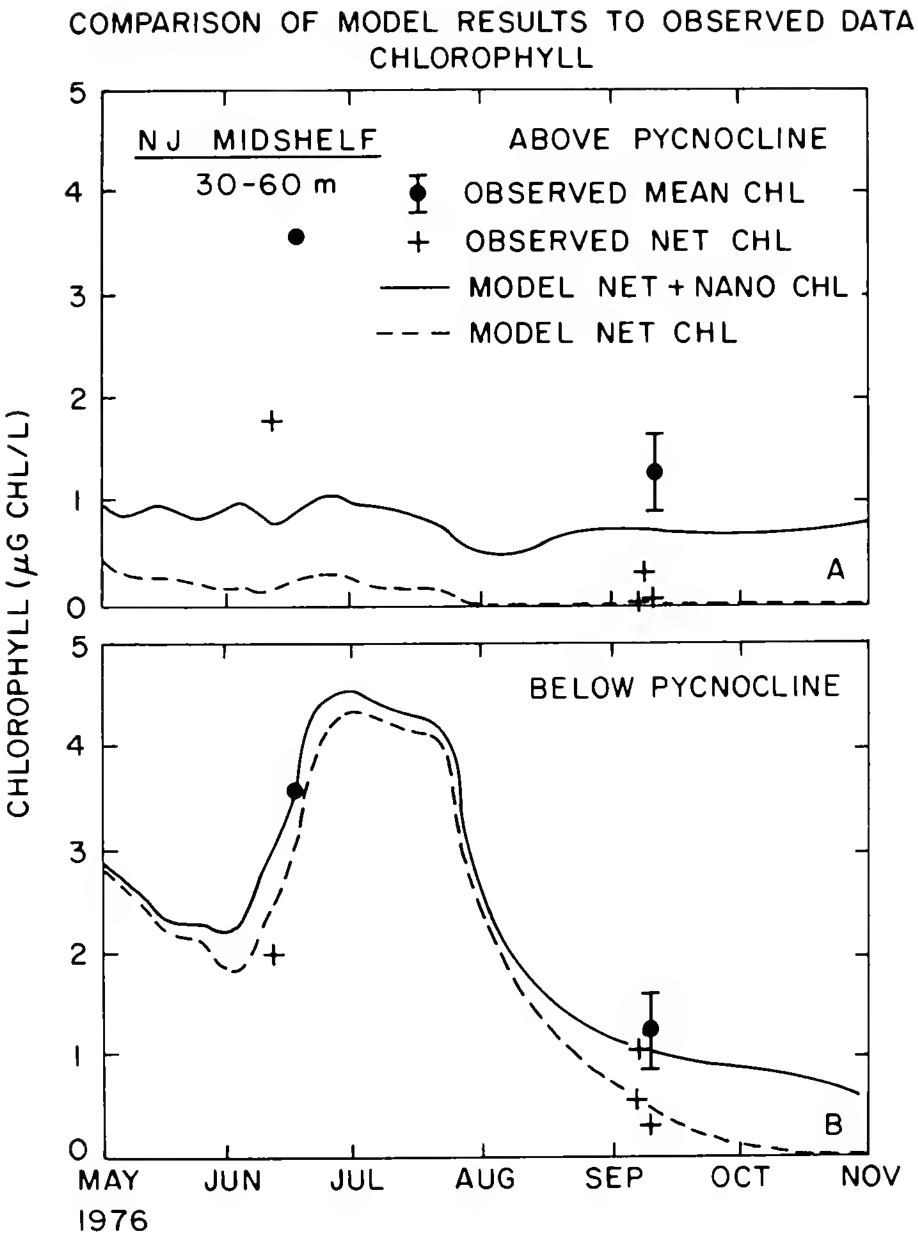

Figure 4. Temporal comparison of calculated and observed chlorophyll for the New Jersey midshelf: 1976.

The occurrence of anoxia in 1976 resulted from temporal sequence of climatological, physical, and biological processes that were significantly different in 1976 in comparison to 1975, or other years of record. Analysis of the observed and computed components of the midshelf oxygen budget (fig. 5) demonstrates the interaction of physical transport mechanisms and biological/ chemical processes on the cross-shelf and seasonal distribution of dissolved oxygen in 1976. The dominant physical and biological factors influencing the observed spatial and temporal distributions of oxygen included:

(1) Circulation regime over the New Jersey shelf with upwelling predominant from May-July and downwelling occurring in August.

(2) Cross-shelf vertical distribution of viable <u>Ceratium</u> populations and the balance between photosynthetic oxygen production and respiratory oxygen demands, i.e., compensation depth, in relation to the anomalously deep pycnocline during May-August.

(3) Decline of the <u>Ceratium</u> bloom and decomposition of the additional organic carbon load within the water column and on the seabed resulted in the onset, and progression, of anoxia over the New Jersey shelf during July-September.

(4) Increased wind mixing, surface cooling, and erosion of the pycnocline resulted in replenishment of oxygen during late September-October above, and below, the pycnocline with the isothermal vertical dispersion coefficient (10-15 cm^2sec^{-1}) estimated from the observed data and a one-dimensional, vertical calculation for the New Jersey Midshelf.

Oxidation of particulate organic carbon derived from sewage effluents represented a negligible component (1%) of water column oxygen consumption over the New Jersey midshelf in 1976. Even within the Apex region, where the ocean dump sites are located and sewage carbon concentrations are higher, the oxygen demand from sewage-derived materials represented a minor component (4%) of oxygen consumption below the pycnocline. This suggests that the major impact of carbonaceous waste discharges is limited to the vicinity (30 km) of the discharge location, i.e., a relatively localized process. The results of the oxygen budget analysis demonstrate the necessity of including a kinetic process in the model that accounts for the conversion of phytoplankton biomass to nonliving organic detritus in the water column (O'Connor et al. 1981). Incorporation of such a mechanism would then account for the distribution of particulate organic carbon observed in August-September 1976 with mineralization of the observed biomass accounting for the oxygen depletion rate observed during July-September 1976 over the midshelf area off New Jersey.

MODEL PROJECTION OF APEX WATER QUALITY

The validated model was used to evaluate the impact of a ten-fold increase in present urban carbon and nitrogen loading (Mueller et al. 1976) on eutrophication and oxygen depletion in the Apex (Segar and Berberian 1976; Garside et al. 1976). Steady-state August distributions are used to examine the water quality response in the Apex to present and increased anthropogenic loading. Substantial increases in sewage sludge carbon diminished light penetration near the ocean dumpsites and reduced primary productivity. In the outer Apex, increased nitrogen abundance increased primary productivity slightly over an

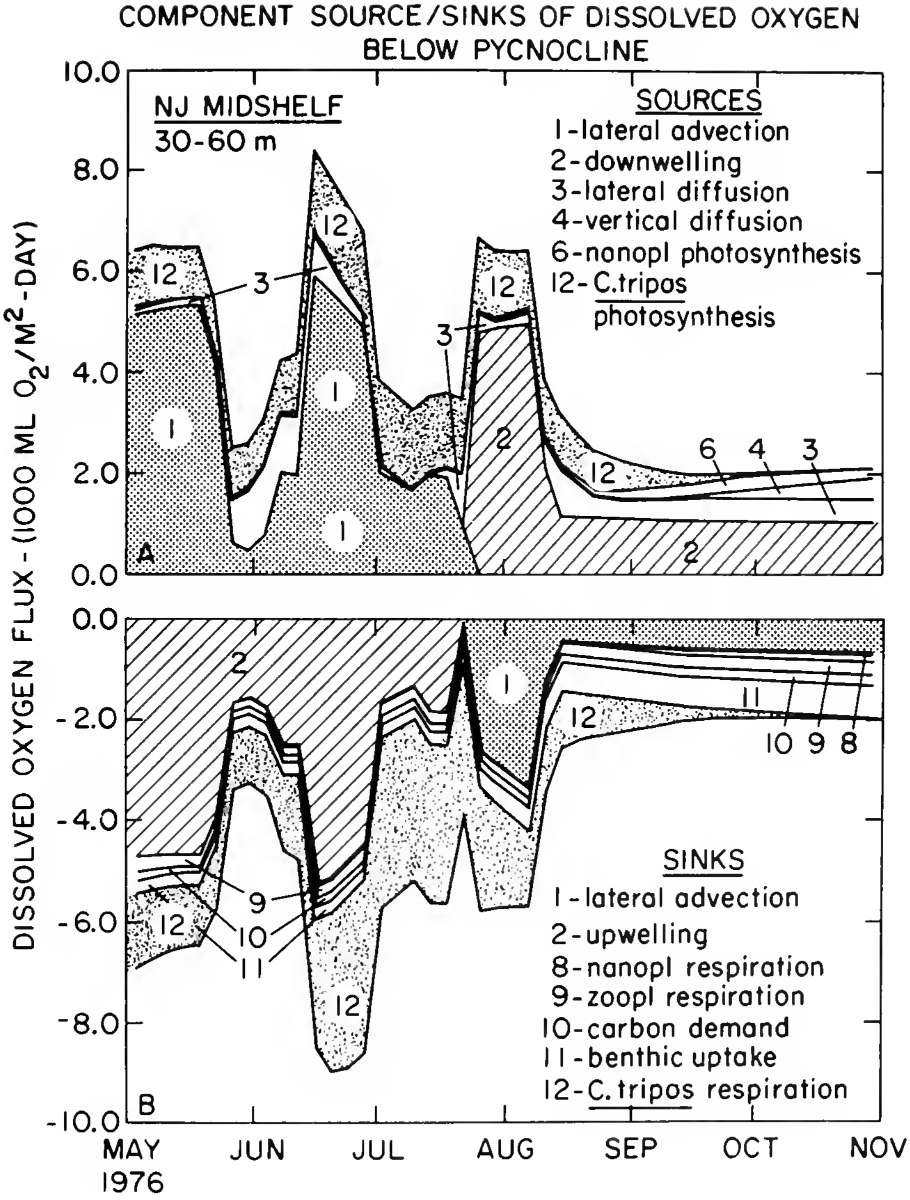

Figure 5. Component sources and sinks of dissolved oxygen below the pycnocline for the New Jersey midshelf: 1976.

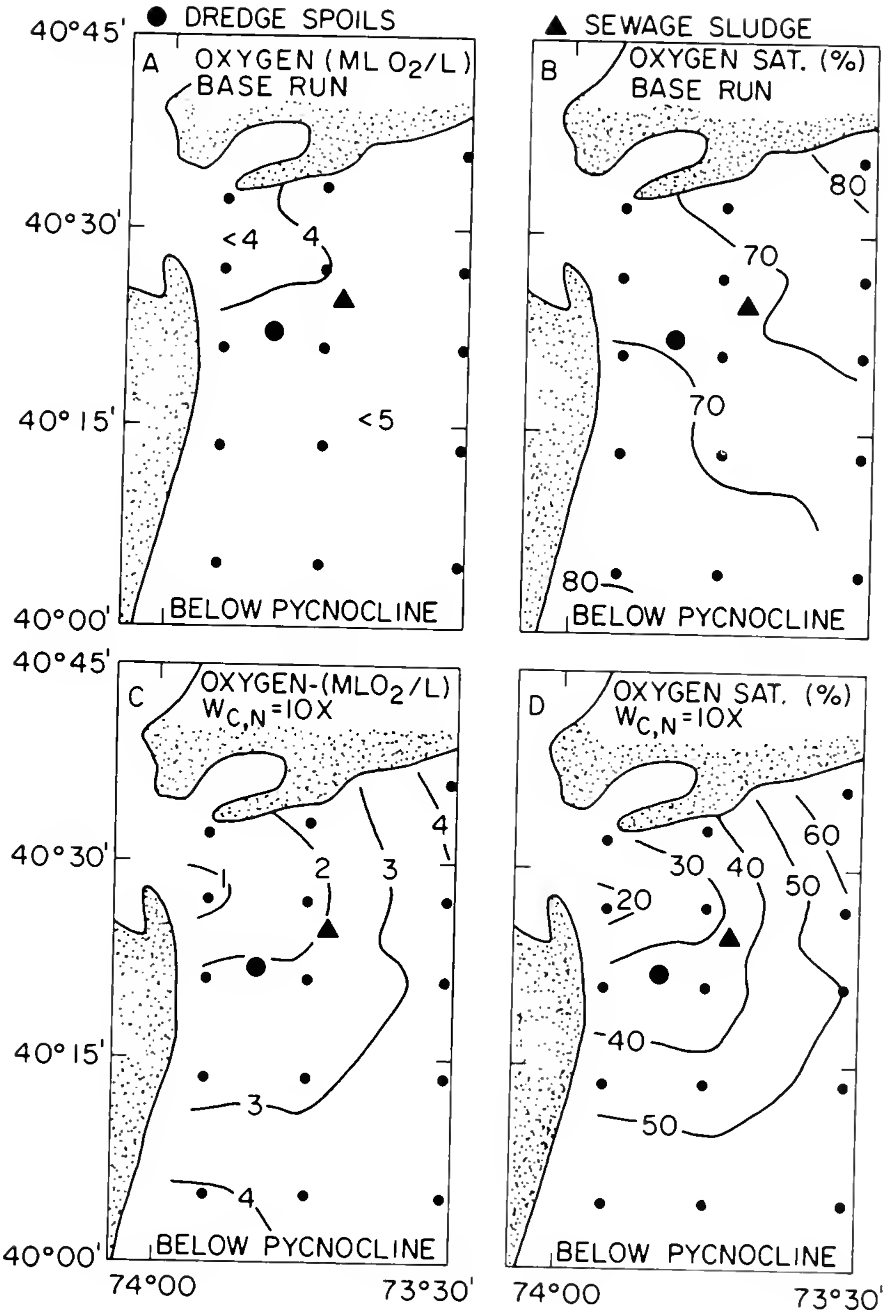

Figure 6. Spatial comparison of dissolved oxygen and oxygen saturation below the pycnocline for the base run and 10X projection: August.

area proportional to the ten-fold increase in waste discharges (Garside et al. 1976). Within the inner Apex, the increased oxygen demand from the additional urban carbon loading resulted in anoxia below the pycnocline (fig. 6) and a reduction in surface layer oxygen to 60-70 percent of saturation.

The similarity between simulated detrital carbon content in the inner Apex (50-100 gm C m $^{-2}$) and the estimated POC content of the *Ceratium* bloom within the Apex (80 gm C m^{-2}) and the New Jersey midshelf (25-125 gm C m^{-2}) in June 1976 suggests that the model response of anoxia to the additional urban carbon loading is a realistic calculation of carbon-oxygen dynamics within the Apex. These results, consistent with preliminary estimates of the assimilative capacity (37 gm C m^{-2}) of the New York Bight (Goldberg 1979), suggest that a critical detrital carbon content of the water column is 50-100 gm C^{-2} for the New York Bight.

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Panel A - Ecosystem Modeling as a Fisheries Management Tool

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I. INTRODUCTION

The focus of the Panel was on major fishery management concerns and the role of numerical models in providing the information necessary to meet these concerns or needs. First, the Panel outlined fishery management needs and the general kinds of models currently used to meet those needs. The utility of these models was discussed, and an attempt was made to assign priorities to future research in terms of potential for improvement in predictive capability. No attempt was made to prepare a complete list of all the variations of models--just the general types of models.

Next, the Panel discussed the role of ecosystem models in fishery management. There was general recognition by the Panel that full-scale mechanistic ecosystem models (i.e., involving quantitative and dynamic linkages among all the principal physical and biological processes which control organic production in the aquatic environment) are not being widely used for fishery management. This is because quantitative knowledge of the natural ecological mechanisms which control fish production is still very limited; in particular, the factors which control survival in the first year of life of fishes are not well understood, and this is the life stage when mortality is both high and variable. The Panel considered the critical role of the recruitment process in driving fishery fluctuations and discussed the importance of modeling in helping to clarify this process.

II. FISHERY MANAGEMENT NEEDS VERSUS MODELS IN CURRENT USE

The products of ecological models in current use which meet basic needs of fishery managers can be put into three general categories: 1) short-term projections of fishable stock, 2) long-term harvest strategies, and 3) environmental quality. A brief outline was constructed of the various types of models relative to these categories.

Short-Term Stock Projections

The basic tool of the fishery manager is still the traditional single species population dynamics model which can yield good forecasts of fishable stock size providing that reasonably accurate data on current stock abundance and age structure are available, and assuming that the recruiting age class is a small proportion of the average harvest. The theory underlying these models is well developed, and the models are being utilized at or near their full potential wherever adequate fishery data bases are available. The limitations of these models are that they do not take account of species or environmental interactions and they are of limited value where annual recruitment represents a large fraction of potential harvest.

Pre-recruit (i.e., juvenile fish) estimates of abundance often are correlated with year-class strength and thus can provide useful forecasts of recruitment. Recruitment estimates together with the traditional population models add a further improvement to short-term stock projections. These approaches are also being utilized at or near full potential given sufficient time series of data for establishing a regression. Improvements in sampling procedures can sometimes help, but the general priority for research in this area is low. The lead time achieved with pre-recruit abundance estimates is usually only a few years, and precision of estimates is seldom high.

Single species and multispecies Virtual Population Assessment (VPA) methods are very useful for estimating parameters needed for traditional population dynamics models. The VPA is a calculation procedure more than a model, but it provides good "hindcasting" ability in terms of mortality parameters and stock sizes in previous years, and it is still under development. VPA calculations do not provide forecasts, and estimates of the most recent year or two are subject to the greatest errors due to uncertainties of fishing mortality rates applicable in the most recent years. On the other hand, multispecies VPA methods do provide a method for estimating natural mortality of pre-recruit stages caused by predation. Further advances along these lines are clearly feasible given the data on abundance of pre-recruits and predation rates, and these results would have useful applications to both short-term forecasts and long-term harvest strategies.

Since the major natural fluctuations in fish populations result chiefly from variation in recruitment, a top priority for improving predictions of future population abundances is the development of models incorporating definitive insight into the physical and biological mechanisms controlling the recruitment process. It is conceivable that empirical relationships might be derived from a long time series of recruitment and general environmental data (e.g., temperature, timing of spawning), but it is unlikely these will ever be adequate for anything but gross predictions (e.g., recruitment always poor after abnormally cold spawning seasons). More likely, extensive field and laboratory studies will be needed to clarify the relative importance of mortality processes including food supply, predation, physical transport, and disease. At present, models of these processes do not have much predictive ability because our understanding of the mortality processes is still largely qualitative (even the timing of first year mortality is seldom known let alone the causes). Recruitment process models offer the largest potential improvement in predictive capability, but the logistic problems are enormous for unraveling such complex large-scale phenomena as the growth, dispersal, and survival of larvae even for one species. A number of conceptual models of the process are available, but empirical measurements for testing the validity of the models are extremely rare.

Long-Term Harvest Strategies

Short-term predictions of abundance are the bases of short-term decisions, e.g., what should the catch be for next year? The specific management decisions are based on a combination of predicted abundances and long-term exploitation strategies which, in turn, are based on balancing tradeoffs between harvesting now or later and between target population sizes and target species.

Recruitment is also the critical issue in the development of models in support of long-term harvesting strategies. Here we are more concerned with long-term trends in abundance (long-term average abundance) rather than annual fluctuations, and the average relationship between spawning stock size and recruitment is the feature of major concern. Traditional stock/recruitment models (e.g., Ricker, Beverton, and Holt) are not adequate because the parameters essential to long-term management strategies are not estimated with adequate precision. New approaches are needed which include stochastic elements, i.e., assessment of the risk of transition from the condition where recruitment is sufficient to support fishing to a condition where probability of collapse of the fishery is high. The new paradigm for stock/recruitment models probably should be based on a world-wide analysis of the empirical behavior of various classes of exploited fish populations. At the same time, experimental studies on mortality mechanisms are needed on these same classes of fish populations because changes in the environment can alter the relative importance of certain mechanisms, thereby confounding the relationship between stock size and recruitment.

Another type of model in current, if limited, use for evaluating long-term strategies is the surplus production model. This approach has been applied to both single species and multispecies situations. The predictive capability of this model is low because it does not explicitly take account of the recruitment process, nor does it incorporate any definitive environmental linkages. In essence, all the complex physical and biological processes controlling fish production are assumed to follow a certain pattern, but the model provides no basis for testing the validity of the assumptions.

Once the stock/recruitment problem has been clarified, multispecies fisheries models incorporating recruitment effects will have significantly improved predictive value for charting long-term management strategy. For the time being, multispecies extensions of traditional single species models can be used to help evaluate alternative strategies. For example, multispecies yield-per-recruit analysis (post-recruit stages only) is feasible given the same data required for a single species VPA. With this approach, tradeoffs between exploitation of different species can be evaluated to a significant degree in advance of solutions to the multispecies stock/recruitment problem. The most recent multispecies VPA models are also designed to incorporate pre-recruit stages as well. However, predation rates and absolute abundances of pre-recruit stages are necessary to model this aspect, and these data are seldom available in any fishery.

Biomass balance models and holistic ecosystem models have also been used to help develop guidance for management of multispecies fisheries. However, these approaches make assumptions about energy transfers and linkages which are not firmly grounded in our quantitative understanding of the actual control mechanisms. Hence, their predictive power is unknown and unverifiable.

Two other approaches are also used to help formulate long-term strategy: 1) statistical models of yield (or population abundance) and long-term environmental trends and 2) gross energy budgets. Use of environmental trend-fishery yield correlations requires a long consistent time series of data, and is fraught with risk as experience has shown time after time. Energy budgets are another way of helping analyze long-term strategies, particularly through estimation of limits to total fish yield in an ecosystem. However, they are either heavily dependent on theoretical transfer efficiencies between trophic levels, or they require a very comprehensive data base on actual production (not yield) and food consumption rates of all major biological components of an ecosystem. Such data bases, particularly the predator-prey interactions, are not available for full scale ecosystems.

Environmental Quality

Quality of the environment is, in the long run, the most important aspect of ecosystem-fishery management since it controls carrying capacity. Effects of overfishing generally are reversible and usually can be mitigated within a relatively short time simply by reducing fishing pressure. In contrast, effects of a degraded environment may not be reversible, or only after a long recovery period, assuming cause of degradation is removed.

Numerous environmental impact models have been constructed to deal with short-term effects of oil spills, power plant discharges, and other pollution such as acid wastes, sewage, etc. In all cases, these models have utility assuming worst case estimates for target organisms. However, their predictive power is generally poor because the cumulative synergistic effects on the ecosystem are not known, and even for target species (e.g., fish), the effects are only local and often do not apply to the entire population. Thus, it is usually impossible to estimate the real significance of a pollution impact, particularly for continental shelf populations. Detailed knowledge of hydrodynamics as well as biological effects are required for quantitative modeling in these cases, and input data are seldom adequate for ocean environments.

The same general models are applicable in the case of chronic pollution, but the problem of estimating biological effects is even more difficult than in the short-term case because of the enormous expansion of the relevant scales of time, if not also space. Nevertheless, since we are dealing with chemical effects on organisms, biological effects of pollutants can be measured in laboratory experiments for a limited but controlled set of environmental conditions and then extrapolated to the natural environment using the model and ambient concentrations of the pollutant in the animals and environment. Direct verification of pollution effects on organisms at the population level in the natural environment is, however, extremely difficult.

Summary of Models

A brief outline of the various types of models was prepared by the Panel and is summarized in table 1. General research priorities were assigned according to expected payoff in terms of improved predictive capability taking into account cost-benefit considerations for various kinds of research. In this case, research implies a new initiative, i.e., developing new methodology (model) or obtaining new ecological information.

Table 1.--Summary of results/discussions of Panel A - Ecosystem Modeling as a Fisheries Management Tool

MANAGEMENT CONCERNS	MODEL TYPES	CURRENT APPLICATION	PRIORITY FOR RESEARCH	COMMENTS
Short Term Prediction: fishable stock projection	1) Traditional fish population dynamics models	YES	LOW	These models are developed to their full potential. Little improvement to be expected except where data bases are grossly inadequate.
	2) Pre-recruit	YES	LOW	Regression relationships generally straightforward but distribution/availability of juveniles to sampling gear usually results in high cost and/or low precision.
	3) Multispecies VPA	YES	MEDIUM	Not really useful for prediction. A method of assessing recruitment.
	4) Recruitment	LIMITED	HIGH	Production, mortality, and dispersal processes between spawning and recruitment poorly known. Both empirical and mechanistic relationships need to be determined between these processes and environmental variables.
Long Term Harvest strategies	5) Stock/recruitment (empirical & mechanistic)	LIMITED	HIGH	See 4) above. Mechanistic relationships are of special high priority because of possible long-term changes beyond current ranges of observed data.
	6) Surplus production (single & multi-species)	YES	LOW	Based on unverified assumptions.
	7) Multispecies* (post recruitment only)	LIMITED	MEDIUM	Good for utilization strategies, but limited in predictive capability without pre-recruit knowledge.

Table 1.--Continued.

MANAGEMENT CONCERNS	MODEL TYPES	CURRENT APPLICATION	PRIORITY FOR RESEARCH	COMMENTS
Risk assessment	8) Multispecies* (pre- & post-recruitment)	NONE	HIGH	See 4) and 5) above.
	9) Environmental trends	LIMITED	MEDIUM	Important to monitor environmental conditions for long-term trends and large anomalies. However, predictions based on time series are risky without understanding of control mechanisms.
	10) Static energy budget	YES	MEDIUM	Useful for first approximations to production limits if no empirical time series available. Needed for preliminary investigation when no other type of model is available or applicable.
Environmental Quality	11) Pollution	YES	MEDIUM	Pollution effects potentially are greatest importance, particularly pollutant interactions and synergistic effects. However, this concern is rated medium because a relatively higher level of funding exists and more information is available compared with those other items listed.

* Multispecies includes energy budget formulations, biomass balance models, and holistic ecosystem models.

III. ECOSYSTEM MODELS AS A FISHERY MANAGEMENT TOOL

There have been few attempts so far to apply full-scale ecosystem models to fishery management problems. Full-scale means a quantitative understanding of the dynamic processes controlling production rates of all the major biological components including the linkages to the physical driving forces. Perhaps the Andersen-Ursin model for the North Sea comes closest in this regard. However, this model falls short of real predictive or even diagnostic power because we do not have a quantitative understanding of the mechanisms controlling the recruitment process in fish. We cannot link survival of young fish quantitatively and mechanistically to either primary or secondary (zooplankton) production nor to the circulation dynamics which drive the whole process of organic production in the sea. In fact, we are just beginning to measure predation mortality on pre-recruit fish by recruited sizes. Determining the relative importance of this predation, which is applied largely to post-larval stages, requires that we also measure larval mortality rates, a prodigious logistic problem. There was a clear consensus in Panel A that more fundamental understanding of this recruitment process is required before we can expect ecosystem models to achieve predictive capability really useful for short-term fishery management needs. The possibilities are perhaps somewhat better for helping evaluate long-term strategies with ecosystem models, but even here the major priority is for better quantitative understanding of ecological processes rather than development of more and better ecosystem models. There are already plenty of models available. Definitive input data is the critical need.

In spite of these problems the Panel felt that there should be a continuing effort toward the application of ecosystem models as a mechanistic framework within which to help clarify the nature of processes controlling fish production. For example, given qualitative but size-specific data on predator-prey interactions in the natural environment and experimental estimates of energy requirements for normal growth, it may be feasible to explore the robustness or reasonableness of alternative hypotheses on pre-recruit mortality processes taking into account time dependent aspects and physical dynamics affecting larval fish food production. Furthermore, multispecies models, involving predator-prey interactions properly scaled by at least an empirically based energy budget, may provide some insight into the potential for affecting long-term average biomass and species composition of finfish populations through alternative harvest strategies. Ecosystem models may also be helpful in resolving resource allocation problems (e.g., recreational vs. commercial fishery disputes over the relative impacts of the respective fisheries on resources).

Since an appropriate model depends critically on the available data as well as the specific questions posed, the development of generic ecosystem models would appear to be of limited value (except in very broad terms). On the other hand, comparisons between different models using the same data base might be helpful in some cases for clarifying properties of the models. It would be perhaps more useful to apply a given model to two generally comparable ecosystems which occur in different locations and which have independent measures of the critical quantities (biomass, production, consumption) and mechanisms (growth, mortality).

On the critical problem of the recruitment process, empirical models have clearly been useful for developing hypotheses, but understanding the mortality mechanisms and evaluating them with dynamic models will be necessary for testing those hypotheses. The known mortality mechanisms of pre-recruit fish and their stochastic nature implies that recruitment predictions of high accuracy are not likely for any reasonable cost. Therefore, fishery managers will need to utilize strategies which, on the average, will optimize benefits as opposed to looking for precise predictions on an annual basis. One possible area of "high payoff" research for modest cost might be an in-depth review of patterns of stock recruitment data for major species groups on a global scale as a basis for a first approximation of relative potential risk of fishery collapse.

The panel discussed, in general, the relative merits of dynamic versus static models in fishery applications but came to no firm conclusions other than each approach had advantages and disadvantages depending on the available data base. For a given data base, the question was posed as to the relative effectiveness of dynamic versus static models for uncovering counterintiutive characteristics of complex systems. Here again, comparison of results among different ecosystems of generally comparable structure seems to be a worthwhile and prudent approach.

Panel B - Ecosystem Modeling as an Environmental Management Tool

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I. INTRODUCTION

The Panel identified six subtopics under the general heading "Ecosystem Modeling as an Environmental Management Tool" for discussion. These subtopics are: 1) state-of-the-art with respect to management application; 2) technical basis of modeling; 3) interactions among managers, modelers, and research scientists; 4) transferability of models; 5) translation of modeling results to environmental quality criteria; and, 6) use of existing data bases.

The Panel formed subpanels, each of which was tasked with addressing one of the specific topic areas. A Panel consensus was formed for each specific topic, and a priority list of recommended high return research activities which would enhance the usefulness and effectiveness of ecosystem modeling as a resource management tool was generated.

II. STATE-OF-THE-ART WITH RESPECT TO MANAGEMENT APPLICATIONS

Overview

Ecosystem modeling is an attempt to describe mathematically the structure and function of ecosystems. This mathematical representation of natural systems is perhaps the most difficult task in ecology since it requires an understanding of the physical, chemical, and biological variables, processes, and interactions that give rise to ecosystem states. The science of ecology is hampered by several inherent problems that make mathematical modeling of ecosystems or ecosystem components difficult. Ecology is an immature science and as such it lacks the theoretical richness of the physical and chemical sciences. It has few, if any, theories that allow precise predictions of organismic or ecosystem state as a function of abiotic and biotic forcings. For example, physics has its force and energy laws that allow rather precise estimates of at least average physical states, and chemists can calculate reaction kinetics and byproducts. In contrast, ecologists have not yet been able to tie their science to proven mechanistic or empirical relationships of cause and effect.

Another inherent difficulty that ecological modeling must contend with is the plethora and interaction of biotic and abiotic variables that control ecosystem state. Physical variables such as temperature, radiation, and turbulence are driving variables that control ecosystem structure and function. Chemical variables ranging from nutrients to toxics can interact to alter ecosystem state and response. Biotic components such as species present, population densities, competition, and behavioral patterns combine to provide the ecosystem modeler with a complex matrix of interacting variables which is almost impossible to untangle. This problem becomes more complex when one tries to integrate the synergies between all three classes of variables. It is true that the wise scientist tries to minimize extraneous variables emphasizing only those factors subjectively believed to be "important." However, he or she must enter into that exercise with the knowledge that the underlying assumptions and conceptual formulations may be incompletely defined.

Finally, ecological data which are crucial to ecosystem modeling needs are often lacking. Perhaps most important is the lack of basic physiological data for many species. For example, verifiable toxicological data are available for only a handful of chemicals and species. The data that exist are for single compounds tested under limited environmental or experimental conditions; knowledge of synergistic effects of toxics and toxic effects over a wide range of environmental conditions is virtually nonexistent. Important ecosystem rate kinetics are unknown as are fluxes, routes, and reservoirs of many elements and compounds in ecosystems.

These constraints in theory, complexity, and data are facts that ecosystem modelers must face before they promote their potential products to resource managers and decisionmakers. Engineers can predict safe loading levels for bridges, and the public expects ecologists to be able to predict safe loading levels for pollutants in ecosystems. Ecologists and resource managers must be eminently cognizant of the constraints to ecosystem modeling and the limitations to model applications and predictions.

State-of-the-Art

The term "ecosystem model" encompasses a very broad range of approaches, each with strengths and weaknesses, each with different data requirements for development and evaluation, and thus each relevant to different aspects of management and decisionmaking. None of these approaches, however, can now or in the foreseeable future provide results that are conclusive and reliable enough to be the sole basis for any significant management decision. Further, modelers do not believe that model output should ever be the sole or primary basis for a management decision. The state-of-the-art of ecosystem modeling is such that it can produce information that is germane and useful to management, but such information must always be closely integrated with many other forms of information input: expert opinions, direct experiments, and socio-economic, health, and political deliberations.

It is important to recognize quantitative models of all types as extensions of the conceptual perceptions of how ecosystems function. As such, models are never more than tests of necessarily incomplete hypotheses. As these hypothesis evolve with our experience, modeling inherently becomes an iterative intellectual process. In this context, it is important to recognize the necessity of including the modeler and the modeling process throughout the decisionmaking process or, conversely, including the decisionmaker throughout the modeling process (see Section III).

While this may seem to be a serious limitation to modeling, it is an essential perspective to take when attempting to specify the state-of-the-art of modeling and how models can provide useful input to management. The extent and quality of scientific information about different topics varies greatly. Similarly, there is a spectrum of approaches to ecosystem modeling, each with its specific data requirements and each capable of contributing information at different degrees of resolution to management decisions. Modelers vary widely in their interests and convictions about what model types are best. Perhaps a more appropriate view is to see the various modeling approaches as members of a spectrum of strategies. Relatively simple empirical models are at one end of the spectrum while highly detailed mechanistic models incorporating many functional forms of complex cause and effect interactions are at the other end. Throughout this broad spectrum there is the consideration of deterministic versus stochastic modeling techniques. The deterministic approach is the simpler of the two since a unique set of outputs, free of variability, results from each unique set of input data. Stochastic approaches, on the other hand, incorporate variability (uncertainty) in the model structure often resulting in nonunique outputs which are viewed as probabilities of occurrence within the natural variability of the system being modeled.

The remainder of this section is a brief summary of a general framework within which the various approaches may be viewed.

Empirical Models

A number of purely statistical techniques exist for describing apparent relationships among or between empirical observations. In the simplest case, linear regression can relate a single dependent variable (Y) to an independent variable (X). Multiple linear regression relates the range of observed Y-values to more than one X or independent variable. Polynomial regressions are more complex, and assume that the variation in Y is related as some nonlinear function of one or more X variables.

The statistical methods are well known and readily available on most computers. Thus, these methods are relatively quick and inexpensive to implement if the data are available. The most serious limitation is that such regression methods do not necessarily reveal anything about causal relationships, though it is often tempting to suspect they do.

Mechanistic Models

At the other extreme of modeling strategies is the family of models that attempt to represent the functional mechanisms that regulate the processes taking place in the natural ecosystem. These models are based upon massive amounts of data, often at the level of physiological responses of organisms to important features of their environment. Such data are almost never available for all or most of the species in any given system, and assumptions must be made about specific features unique to each locale. The predictions of these mechanistic models are usually compared to data of a fundamentally different kind: synoptic observations of the standing stocks of the state variables through time. Thus, this modeling approach requires the greatest amount of data at both the physiological and ecosystem level and incorporates a large number of assumptions. The strength, however, is that these models potentially can "predict" the responses of many components of the ecosystem to new or extreme conditions, even when such conditions historically have never existed. These predictions, though they may be important, are based on an explicit series of identifiable statements about the underlying causal mechanisms. When used correctly, with continuous appreciation of the assumptions, these models may be the only quantitative method for evaluating certain environmental scenarios.

Intermediate Strategies

In between the extremes of empirical and mechanistic models falls a broad range of approaches, many combining, to various degrees, elements of both. Mechanisms can be specified, but with various theoretical perspectives. For example, many approaches attempt to relate changes in certain state variables to forcing factors in the environment using simple or complex statistical expressions. Thus, linear or nonlinear terms may be used to define the dynamic interactions among the compartments at the ecosystem level. In contrast to the mechanistic strategy, the data required for these formulations of processes are not physiological, but are the synoptic observations of the state variables and forcing functions. Thus, the data requirements are more limited and generally site specific, but again causal mechanisms are less fundamentally represented.

Deterministic and Stochastic Modeling Approaches

Output of deterministic models can be seriously misleading if the modeled environment is characterized by variable or uncertain conditions. This is particularly true when both controllable and uncontrollable variables have strong stochastic components. Also, estimates of these future mean properties are often not useful to managers. Quite often managers require probabilities of certain events occurring. For example, the frequency of occurrence and duration of dissolved oxygen (DO) excursions below some acceptable limit could be more important than mean DO concentrations.

Certain stochastic environmental properties are quantifiable statistically (e.g., current roses, light and temperature variability, and nutrient loadings). We have the means in hand (e.g., Monte Carlo, 1st order variance propagation analyses, random differential equations) to address the effects of this natural variability on model predictions. All models used in predictive or diagnostic modes should take advantage of these or similar methods to account for, or at least address, quantifiable uncertainty.

Deterministic models of lumped properties (e.g., functional groups of phytoplankton) use parameter values, initial conditions, etc., that presumably represent typical species in that group. However, in nature functional groups are often led by strong dominants whose dynamics may not represent those of the total group. One might expect, though, that the dynamics of individuals within the group (e.g., their kinetic coefficients) may follow some distribution (not necessarily symmetrical) centered on those of the "typical" component. These conditions may be amenable to Monte Carlo analysis. A model solution is obtained many times, each time using kinetic coefficients drawn randomly from some distribution centered on the typical coefficient values. This analysis will produce output that has estimates of both the mean response and spread of responses. While the central tendency may represent the response of the typical (dominant) component, the mean response, which can often be different from the central tendency response, may better represent the total functional group. A simple deterministic solution will mimic the typical response only. This particular type of analysis addresses both the uncertainty introduced by model aggregation as well as that introduced by imprecise measurements of kinetic coefficients. Methods other than Monte Carlo can be used to introduce stochasticity in ways which are based on different ecological assumptions. For example, where Monte Carlo assumes parameters are uncertain but constant throughout the simulation, 1st Order Variance Propagation Analysis and other methods based on random differential equations assume instantaneous probability distributions for parameters during each simulation. These methods can help place in perspective the potential errors (both real and artificial) introduced by measurement uncertainty, conceptual aggregations, etc.

Including potential variability in ambient properties (e.g., currents, light) and in aggregation effects (e.g., within functional group variability) in a Monte Carlo analysis or other method, for example, is one way to produce distributional output that can be summarized as percent occurrence and duration of certain desirable or undesirable events. This becomes more usable information relating to problems like that of dissolved oxygen as mentioned above.

Although methods of uncertainty analysis are well documented and have had extensive application in physical sciences, they have had only limited applications to environmental/ecological problems. More work needs to be done to assess the relative utility of various approaches to uncertainty analysis in ecosystem models.

Thus, each approach in the range of strategies included in the general term "ecosystem modeling" has identifiable strengths and weaknesses. None is "best" or "worst," but each may contribute useful input to the management process given different questions being asked, different available data, different personnel, and different constraints in time and money. Generally, the simple models will be easiest to understand by the manager inexperienced in modeling, and may well be the most effective in terms of cost and time. However, their appeal is deceptive; if the proposed perturbations are beyond the prior range experienced by the system, predictions will be very tentative at best. In these cases, especially, detailed mechanistic models may provide the only numerical tool for evaluating various scenarios.

Given the spectrum of modeling strategies and their respective assumptions, it is clear that it is desirable to apply more than one modeling approach. When the predictions of various models spanning this range of resolution converge, managers and scientists may have more confidence in the results of the modeling experiment.

Another viable though expensive alternative is what has been called "living models." Large scale macrocosms have recently reached a level of sophistication to be useful in management. A specific example is the Marine Ecosystem Research Center (MERC) in Rhode Island. This facility has met two critical criteria: the experimental systems can be shown to 1) replicate each other, and 2) mimic changes in the natural system within acceptable limits. Real experiments can be designed on these living models which have a capability to respond in ways no numerical model can ever approach.

II. TECHNICAL BASIS OF MODELING

Modelers frequently state that no model can be better than the data on which it is based. However, there is more to establishing confidence in a model than assuring the quality of the data set, although this is an important aspect. Although ecosystem modeling is a highly technical activity, it mainly proceeds by unwritten procedures. There do not exist, as yet, formal rules for formulating models. The process generally occurs in stages--problem definition, model conceptualization, formulation, calibration, validation, output to user--but the ecological community has given very little attention to developing the technical basis of each of these steps. Examples are:

1. Problem definition. Mathematicians recognize poorly specified versus well specified problems. What criteria can be developed to assure that modeling activities occur in response to well specified problems? How can normal procedures for testing hypotheses be engaged here?

2. Model conceptualization. Models are supposed to be analogies which organize relevant considerations about a problem into a consistent interactive set. Usually, some scheme of functional taxonomy is constructed in the process. How may formalisms such as set, hierarchy, and category theories contribute to this process? What do such theories tell us about consequences of aggregation or disaggregation for function preservation?

Structural characteristics of a model influence model behavior. There is considerable latitude in design of the basic structure of the model (i.e., number of compartments, connections, feedbacks, modes of input, etc.) and equal latitude in the mathematical interpretation of this structure. Little work has been done that systematically explores the effect of model structure on results. An appreciation of robustness of results to different model configurations is needed to evaluate the reliability of a model in representing the ecosystem. This knowledge is particularly important with ecosystem models because they are so difficult to evaluate.

3. Model formulation. Can formal criteria be developed to aid in the choice of alternative mathematical functions? Are rules of parsimony available, e.g., at what point would decisions be made to select dynamic over static, nonlinear over linear, spatial over nonspatial models? In general, the criteria for such decisions have been very soft in the past. For example, quite often an empirical, curve-fitting function is chosen because of tradition. The criteria for such choices should be amenable to formalization.

Once a model is mathematically formulated, it should not be carelessly used in computer simulations. The twin mathematical properties of existence and uniqueness are almost never investigated for ecological models. It may be possible to generate numerical solutions of equations for which no analytical solutions exist. Ecological models are usually mathematically complex, and careful attention should be given to their mathematical properties before they are acceptable for environmental management purposes.

4. Model calibration. Adequate data to implement ecosystem models almost never are available. Is it possible to develop mathematically sound procedures for generating missing data comparable to the missing data procedures which are

used in statistics? How can a model be used to set priorities for supplemental data gathering for calibration purposes?

5. Model validation. A definitive theory of this critical phase in modeling procedures has not yet been written. There are tacit philosophical and logical considerations which underlie present validation practice. These need to be drawn together in a formal validation theory. Probably no topic under "Technical Basis of Modeling" is more important than establishing exact rationales, procedures, and criteria for validation, verification, corroboration, etc.

6. Output to User. Just as a model is a conceptual and mathematical analogy of the real world, so its output for useful purposes should be subject to feedback considerations which relate it to the original problem definition. The problem of simplifying model output into user-relevant information should be amenable to the same kinds of formal considerations as model conceptualization itself.

The surest way to increase confidence in models and modeling is to have them firmly grounded in sound theory. Ecological modeling now is an art; it needs to become a formalized exact science.

Recommendations

1. The theory of modeling in all areas should be considered high priority research. There is a dearth of such theory in ecological modeling, and improved theory is certain to increase reliability and usability of models.

2. The number of possible calibration solutions to a model can be reduced if the model is required to represent adequately process-rate and state-variable measurements. This will not eliminate problems of nonuniqueness. However, we may be forced to use such models; and calibration against both types of data substantially reduces the degrees of freedom.

3. Studies of applicability of stochastic modeling procedures are encouraged. Model uncertainty or confidence levels should accompany any model projection made.

III. INTERACTION AMONG MANAGERS, MODELERS, AND FIELD SCIENTISTS

The success of any modeling effort for resource management depends heavily upon clear, continuous communication between the modelers and the managers who will use the model results. This fact is generally accepted, but too often lines of communication are severed before the management decision is made. We feel the dialogue should continue up to and beyond the decision and lead to refinements in the model and in the data collecting scheme. By extending communication, both the short-term decision-oriented management goals and the long-term ecosystem-oriented scientific goals will be realized.

Several studies could be cited to illustrate these points. Moreover, formats for the interaction of managers, modelers, and field scientists are available (e.g., Adaptive Environmental Assessment Technique). It is beyond the scope of this report to treat the subject exhaustively. Instead, we suggest that such an interaction should contain, as a minimum, the following basic elements:

1. Identify Management Need

    a. Types of Management Activities

        - Immediate problem or crisis (impact assessment)
        - Establish best alternatives (risk assessment)
        - Planning and long-term development
        - Resource management

    b. Evaluate Modeling as a Problem Solving Tool

        - Modeling may not be appropriate or applicable in all cases

2. Develop Model

    a. Workshop for Managers, Modelers, and Research Scientists

        - Clearly and specifically define the management need, problem to be solved, and questions to be asked
        - Define terms to prevent misunderstanding
        - Agree on relevant and tractable areas to approach through modeling
        - Include the citizens' perception of the problem at the outset
        - Establish the range of scenarios to be tested while a model is in the early stages of development
        - Recognize the benefits of both the decision-oriented management goal (short term) and the ecosystem-oriented scientific goal (long term) of increased understanding of ecosystem processes
        - Assess the need to extrapolate beyond existing data

    b. Modelers Produce Array of Suitable Models

        - Decide on approach (probability, worst case, etc.)
        - Design a conceptual model around the defined problem
        - Determine temporal and spatial requirements
        - Determine precision of the answers accounting for natural variability versus model uncertainty
        - Define data requirements versus availability--what historical data and new data are needed?

- o Review results from similar natural or experimental ecosystems
- o Agree on suitable ecosystem structure for the model
- o Set standards for documenting model development and output format

c. Select Model to be Used

- o Reconfirm data requirements based on model structure
- o Determine nature and extent of proper operation of the model
- o Assure accuracy of model code
- o Compare model behavior with natural system behavior
- o Plan type and duration of research to be continued after decision is made and implemented

3. Model Output Interpretation

a. Realize that Models Yield Information, Not Answers

- o Interpretation of output is usually required
- o Dialogue continues during interpretation
- o Results should be presented in probabilistic terms if possible
- o Citizen involvement still desirable

b. Evaluate Management Strategies in Light of Model Results

c. Management Decision is Made

4. Post Decision

a. Modelers, Scientists, and Citizens Informed of Decision

b. Citizen Reaction is Reviewed

c. Model's "Track Record" is Established by Comparing Model Results with Ecosystem Response

- o Data collection is continued through post-decision phase
- o Model is refined as necessary to improve its performance and future usefulness for decisionmaking

IV. TRANSFERABILITY OF ECOSYSTEM MODELS

It is recognized that, for several reasons, it is desirable to have models that are usable for a variety of different sites and environmental conditions. These reasons include reduced development costs and increased confidence in the robustness of model outputs. However, it must be strongly emphasized that modeling is strongest when designed and tuned to answer quite specific questions for particular configurations of biotic and abiotic assemblages. As a minimum, the structure and assumptions built into each model must be re-examined each time it is used to ensure that they are still appropriate. While processes and interrelationships may be translocated, the specific rates and state variables may not be. This would be particularly true where species composition or genera are different at the new modeled site. Blind reuse of existing models invites results which are at best inappropriate and at worst seriously erroneous. To prevent this, model users (managers) should avoid translocation of models without the direct involvement of scientists and model builders to re-verify the model design and implementation. In short, if participants take seriously the hypothetical and experimental nature of models and the need for continuous communication among modelers and management, attempts to apply existing models to different ecosystems or even to different problems in the same system will be done only with careful evaluation of the model's strengths and limitations.

V. TRANSLATION FROM ECOSYSTEM MODELING RESULTS TO ENVIRONMENTAL QUALITY CRITERIA

Ecosystem models can be powerful environmental assessment tools. In theory they allow evaluation of more than one quality criterion at the same time and allow indirect as well as direct effects to be determined. Considering their potential, ecosystem models can be more fully utilized in environmental assessment.

One problem is that ecosystem models often produce results that managers have difficulty using. The methodology for making ecosystem models more meaningful to environmental assessment requires further development. Different types of modeling results applicable to environmental assessment needs must be identified and their relevance explained to managers so that they will understand them and can determine what will be useful to them. A difficult, controversial, but desirable goal is to translate ecosystem modeling results into terms that can be used in cost-benefit analysis.

Many complications will be encountered in any approach to quantify costs and benefits. Some costs and benefits vary according to user, and optimum balance between costs and benefits may not be the same from all perspectives. For instance, increasing the primary productivity of a water body may be desirable from a fisheries perspective but undesirable from the viewpoint of recreational users of a water resource manager. Some costs and benefits are not associated with any particular user group but rather with general quality of life; although these are probably the most "valuable" to society, they are the most difficult to quantify. A further complication of cost-benefit calculations is that nonlinear relationships clearly exist.

Quantifying environmental quality criteria in terms of economic value is a problem not just for ecosystem modeling but for environmental assessment in general. Only a few researchers in the country have attempted to address this question. Two examples are E.P. Odum and H.T. Odum (1973) and Gosselink, E.P. Odum, and Pope (1973). The Odums suggested that natural systems provide "ecological life support" services for man and his economies and that the amount of energy captured by the system in photosynthesis (gross primary production) is a reasonable index of the magnitude of a system's life support function and, therefore, its value. Gosselink, E.P. Odum, and Pope put a dollar value on tidal marsh land by multiplying annual primary production (in this case, net primary production), in energy terms, by the ratio of the Gross National Product (GNP) to the National Energy Consumption Index (NECI).

Determining economic value in terms of saleable products such as commercial fish is a deceptively easy way to present ecosystem modeling results. It is deceptive because it leads to an undervaluation; saleable products are not the only products and services of ecosystems that are supportive of the society and the economy.

Various techniques have been used to put a dollar value on gross primary production. The rationale is that gross primary production represents the amount of energy from the sun being captured by the system. However, evaluation on the basis of this rationale is not always appropriate. Furthermore, evaluation on this basis alone must be incomplete because differences in gross primary productivity do not reflect all environmental quality differences in an ecosystem experiencing different types and levels of impacts.

Our recommendations on the development of methodology concentrate on the following:

1. Identification of simulatable ecosystem properties and products that can serve as useful environmental quality criteria.

2. Categorization of these properties and products according to relevance to the needs of various user groups or for their global utility (i.e., quality of life).

3. Quantification of these properties and products in terms of environmental quality criteria.

4. Translation of environmental quality criteria into terms of economic value.

While it is not the role of the ecosystem modeler to decide what user groups to favor or what environmental quality criteria to optimize, ecosystem modelers must be involved in the translation process. Conversely, the modelers and research scientists must be informed by managers as to what objective criteria are most important to them in making a decision. Scientific methods, perhaps including sociological and demographic surveys, etc., may then be used to connect in some rigorous way these two perspectives.

VI. USE OF EXSTING DATA RESOURCES FOR ECOSYSTEM MODELING

A voluminous amount of environmental and ecological data has been collected over the past decades. The potential usefulness of these data to ecosystem modelers, researchers, resource managers, and others is obvious. Unfortunately, these historical and scientifically valuable data sets are scattered among individuals, universities, research centers, industries, and local, state, and Federal government agencies. For all practical purposes, these data sets are inaccessible to those who have to complete a task or project within the realistic constraints of time, money, and personnel. As a result, many ecological issues are probably addressed incompletely, and environmental decisions are made based on only a small fraction of the total data which is directly pertinent to the issue.

As a first step towards rectifying this situation, an annotated list of existing data sets could be compiled and regularly updated. Although this listing would never be 100% complete, it would be a valuable starting point for initially identifying relevant data sets and their residence. As a start, many of the larger and more extensive data records residing with government agencies, research centers, and universities could be catalogued.

Eventually the mechanisms should be developed to incorporate some of these larger data sets into one data bank that performs the statistical and associative functions which data-base management systems commonly perform. This system should be "user friendly," accessible, and have a user-feedback mechanism so that the data sets can be evaluated and updated.

VII. HIGH RETURN, HIGH PRIORITY RESEARCH ACTIVITIES

-Highest Priority-

1. Model validation tests

Conduct comparisons of several (minimum of three to five) ecosystem models to one or more Marine Ecosystem Research Center experiments.[1] MERC is also a model in the sense that it is similar to macrocosms of Narragansett Bay. Three or more ecosystem modeling projects of approximately $50,000 each would enable the testing of modeling methodology to see if the models can predict the macrocosm outcome and how well they do it.

Another approach to model validation is to compare model output and environmental data for several before and after case studies for areas where good data sets exist. Examples might be power plant sitings, sewage outfalls, etc. Results will be directed to questions of how well ecosystem models predict either mean ecosystem state and/or how much variability occurs in the ecosystem.

2. Environmental quality criteria and cost-benefit analysis

Further development of methodology, which should include:

a. identification of simulatable ecosystem properties and products that can serve as useful environmental quality criteria.

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[1] The Marine Ecosystem Research Center of the University of Rhode Island is offered here as an example; it is the program which the panelists are familiar with. Similar programs at other institutions would also be suitable for verification, calibration, and testing of model structure and formulation.

b. categorization of these properties and products according to relevance to the needs of various user groups or for their global utility in terms of quality of life.

c. quantification of these properties and products in terms of environmental quality criteria.

d. translation of environmental quality criteria into terms of economic value.

-Lower Priority Projects-

(information and review paper projects)

1. Case studies

Case studies of ecosystem models which have been utilized for resource management. These should, at a minimum, consist of a collection of examples, an annotated bibliography, a critical analysis of how the model was formulated, and what the results were.

2. Data sources

An annotated list of data sources of possible use to ecosystem modelers including useful information about type of data, form of data, quality of data, etc.

3. Model examination

Theoretical investigation of the properties of ecosystem models and how they are affected by their structure (e.g., linear vs. nonlinear equations). Examination of the program language used and the structure and documentation of the code. Examine how and to what extent model behavior is influenced by structure and language.

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Panel C - Integration/Linkage of Biological and Physical Ecosystem Models

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I. INTRODUCTION

Many biological and physical ecosystem models have been developed in recent years to help marine resource managers assess the hazards and benefits of diverse activities which impact the marine environment. These models, for the most part, have been either species, site, or problem specific. Nonetheless, their usefulness has sparked hope that they can somehow be integrated or linked to form a "generic" model capable of treating a variety of ecological communities, geographical locations, and impact assessment concerns. The Panel considered how realistic this hope is in light of real world constraints on modeling development and application, the state of the art, and our level of understanding of ecological processes and interactions.

After much discussion, the panelists agreed that development of a universal application generic model or suite of generic models is neither a practical nor prudent management goal today. The scope of a model is severely limited by the funds, time, data, and expertise available to the modeling effort. These constraints are not expected to ease much in the foreseeable future. As a consequence, models must remain tailored to specific species, sites, and problems, and for a good fit, the species, sites, and problems must be precisely defined. To do this, the manager and modeler must continually work together to:

(1) set reasonable limits on the funds and time available to make the proper decision;

(2) locate, interpolate, and extrapolate data needed to make the correct decisions;

(3) identify alternative management strategies and modeling approaches; and

(4) identify critical information gaps that must be filled by research specialists.

These conclusions emerged from the Panel's consideration of the six questions posed by the workshop convenor (EDIS). The Panel's responses to each question are presented in greater detail in the following pages.

II. APPLICABILITY OF PREDICTIVE (FORECASTING) MODELS

The utility of a model to a manager depends on the amount of confidence that can be assigned to its predictions or forecasts. This "level of confidence" is influenced by many factors which interact synergistically to enhance or diminish the predictive capability (utility) of a model. Basic among these factors are the nature of the management/assessment objective, the data base, and the degree of ecological complexity which must be modeled to minimally address the resource management and impact assessment concerns. In general, models addressing problems that involve few, noninteracting species and processes work best.

The properties and dynamics of seawater are far better understood than those of marine populations, communities, or ecosystems. Thus, the more a problem is dominated by deterministic physical processes, the better it can be modeled. Examples are nearshore spill and plume trajectories and settling of dredging spoils. Conversely, the more a problem is dominated by complex and interactive biological processes, the less easily it can be modeled. Uncertainties about the biological structure, function, and interrelationships of a population, community, or ecosystem usually overwhelm all other factors that might limit forecasting ability. Also, as the time and space scales and resolution of the management problem expand, the predictive capability diminishes. Again, this most strongly affects the biological models. For example, the trajectories of nearshore spills and plumes can be predicted with reasonable accuracy for 1 to 2 weeks into the future. However, as the temporal and spatial boundaries of the problem expand, weather-related chance events and uncertainties about the physiology of species, the life histories of populations, the structure and function of communities, interactions among communities, and rates of biotic energy and material flows become overwhelming. As a result, short-term (day, week) acute impacts such as mortalities in habitats contacted by high concentrations of toxic materials can be modeled with much greater predictive success/confidence than can long-term (month, year) chronic impacts such as physiological and genetic damage caused by constant low or high toxicant concentrations.

Process dynamics (e.g., primary production, grazing, nitrification) and single species population dynamics can be modeled with greater confidence than community dynamics. Empirical processes and population models (e.g., fisheries stock models) that essentially interpolate data have consistently yielded reliable predictions as long as interpolation limits are not exceeded; they quickly break down outside of those limits. Mechanistic processes models (e.g., uptake and grazing kinetics) not only can interpolate data, but they can also, at least in theory, extrapolate beyond the ranges of experience. A critical factor to the predictive success of both approaches is the data record upon which the conceptual and mathematical models are based. The least squares, best fit methodology of the empirical models requires an extensive data base of field observations; a length of record of 20 yr is often the preferred minimum. In addition, as the empirical relationships deviate further from linearity there is a concomitant need for greater temporal and spatial resolution in the data base. Because of their cause and effect approach, mechanistic models are not as dependent on lengthy records of field observations and field data resolution as are the empirical models. Mechanistic models can derive/formulate causal relationships from laboratory experiments. However, a degree of uncertainty is always introduced when laboratory-derived relationships are applied to the natural ecosystem. This uncertainty becomes magnified when the range of laboratory test conditions is narrower than the range of natural conditions and when forcing functions and rate coefficients are derived singly but applied synergistically. It should also be pointed out that when empirical and mechanistic models are connected to represent complex ecological processes and interactions the predictive capability of the integrated/linked models rapidly diminishes.

III. CONSIDERATION OF HOLISTIC AND LIMITED COMPONENTS ECOSYSTEM MODELS

Holistic Ecosystem Models

The development of generic holistic ecosystem models that apply to many environmental problems and habitats does not appear to be practical or prudent. The detailed structures of a model depend strongly on the problem being addressed and the habitat of concern. A model with a versatile structure for widespread problematic, geographic, and habitat application would be impossibly complex. For instance, to model problems that are dominated by physical transport processes (e.g., oil spill trajectories), detailed representations of hydrodynamic processes would be required. Nutrient enrichment problems would need detailed descriptions of bacterial and planktonic communities in addition to the hydrodynamics. The movement of toxic substances through biotic networks requires detailed descriptions of macrophyte, invertebrate, and vertebrate communities and interactions. A model designed to treat spills, eutrophication, and bioaccumulation of toxics in even one limited geographical area would require field and laboratory studies several orders of magnitude more costly than is warranted by the economics of the problem(s) being addressed. Hence, it would not be cost effective to attempt to develop even a single-site holistic model let alone a site-adaptable and problem-adaptable generic model.

Limited Components Ecosystem Models

Most resource management problems can be addressed more cost effectively by considering only that portion (structure and function defined in time and space) of an ecosystem which is directly involved in the management problem/concern/question. For example, until an oil spill contacts a species of interest, transport concerns dominate the problem. Once the species has been contacted, biological concerns dominate. In this case, transport and single-species models would be far more appropriate than a comprehensive ecosystem model. For fisheries problems where factors such as temperature, food, weather events, and the locomotive ability of the species dominate over physical transport processes, a population or trophic dynamics model would be adequate to meet the management concerns.

IV. PLANNING AND DESIGN OF ECOSYSTEM ANALYSIS AND ASSESSMENT PROGRAMS

Definition and Execution of Program Objectives

If it is to be effective, an ecosystem analysis and assessment program needs a clearly stated objective and plan of execution. One of the early steps in the development of the program is to recognize and formulate an acceptable trade-off between what information the manager would like to have to make an "absolutely perfect" decision and what information can be generated within the limits of the allocated resources (i.e., time, money, and personnel) to enable the manager to make a "best" decision. Making this trade-off work involves an iterative exchange of expertise and expectations between modelers and managers. It is critical that the manager and modeler work together from the outset of the project and continue to collaborate during all stages of execution. This will assure that the objectives and modeling approach are pertinent to the decision and achievable.

Besides assisting in the definition of program objectives, a modeler can, in this initial phase of the program, assemble germane information in a framework that pinpoints the modeling tasks that must be undertaken to meet the program objectives. Once these tasks have been identified, time and funding limits set by management along with the modeling needs will determine the sequence of project activities and staffing requirements. The manager and modeler may have to re-examine and renegotiate the program objectives.

As field and laboratory specialists become involved in later stages of the program, new and pertinent information will be generated. This additional information, when compared with the objectives and plan of execution, may indicate that another round of reformulation is needed.

All parties (managers, modelers, and specialists) should jointly participate in the development, calibration, and application of the model. They should also participate in the interpretation of the results, especially with regard to their relation to the management problem and program objectives.

V. ADVANTAGES AND DISADVANTAGES OF VARIOUS MODELING APPROACHES AND TECHNIQUES

Ecosystem modeling can be broadly defined as the application of mathematics to quantitatively simulate ecosystem response to changing environmental conditions. There are many different approaches to modeling ecosystem behavior, and each has its advantages and disadvantages. Table 1 summarizes the Panel's consensus on the advantages and disadvantages of various modeling approaches and techniques as they relate to resource management and impact assessment needs.

VI. AREAS OF HIGH PRIORITY, HIGH RETURN RESEARCH FOR ECOSYSTEM MODELING

The following areas have been identified as high priority research which would significantly enhance the value of ecosystem modeling as a marine resource management and impact assessment tool. These research areas are not directed towards ecosystem modeling per se. Rather, they define crucial gaps in our understanding of basic ecological processes and interactions.

1) The key question in fisheries is the stock recruitment relationship which can be addressed by determining what factors affect egg, larval, and juvenile survival. Basic questions which need to be addressed are:

    a) What are the early life histories of the species of concern?; and,

    b) What is the influence of predation as a mechanism for controlling stock size?

Table 1.--Summary of advantages and disadvantages of various modeling approaches and techniques.

Approach-Technique	Advantages	Disadvantages
Empirical (Statistical)	Relatively simple to construct.	Assumes that the same dynamics and structure are always operating.
	Gives rapid response to management questions.	Cannot extrapolate beyond the historical observed field data.
Mechanistic (Causal)	Can potentially define all the important processes and interactions.	Increased complexity over the empirical approach.
	Can be used to extrapolate beyond the range of observed field data.	Formulation and assumptions can predefine results.
	Amenable to experimental and conceptual testing.	Causal relationships which drive the system must be known and quantifiable.
Static (Time Invariant)	Relatively simple to construct.	Potential for inappropriate aggregation of time dependence.
	System response easily analyzed.	Gives no information about system behavior with time.
	First order estimates.	Cannot be time-stepped with high level of confidence.
	Organizes information on key system components.	
	Can be useful in focusing data collection efforts.	
Dynamic (Time Variant)	Gives better approximation to real processes.	Output is often overwhelming and sorting the signal from the noise is difficult.
	Estimates impacts on different time scales.	Requires increased computations over static method.
	Shows potential interaction between temporal forcing variables.	Relatively difficult to construct and quantify.
		Relatively large data requirements.

Table 1.--Cont'd.

Approach-Technique	Advantages	Disadvantages
Partial Ecosystem (Limited Components)	Forces modeler to focus on study objectives.	May miss important interactive mechanisms. Increased reliance on assumptions of system functions.
Holistic Ecosystem	Potential to represent full system complexity.	Data requirements exhaustive.
	Excellent research tool.	Many transformations and functional relationships poorly understood.
	Full range of impact studies possible.	Full parameter sensitivity study often impossible.
		Little value in most management scenarios.

2) There is a need to determine the individual and synergistic impacts of pollutants, habitat destruction, and natural processes on egg and early larval mortality of fish and shellfish in our most impacted estuaries.

3) We need to determine the extent to which various taxa (from bacteria to marine mammals) can adapt or acclimate to contaminant stress before suffering reproductive problems.

4) The physical and biological coupling between estuarine and coastal waters needs to be described with particular emphasis placed on the flux of contaminants and particulate matter.

5) The concentration levels or dosages of contaminants or mixtures of contaminants which cause reproductive stress in the early life stages of fish, shellfish, benthic fauna, and plankton need to be investigated and defined.

6) The uptake pathways and rates, biochemical processing, and depuration pathways and rates of contaminants by marine biota need to be determined.

7) The exchange of material between the water column and the sediments needs to be better defined.

8) Sediment-organism interactions need to be better quantified with emphasis on the fluxes of pollutants.

9) Definition of vertical exchange processes in the presence of density gradients and tide and wind forcing remain key areas of weakness in hydrodynamic modeling.

VII. WEAKNESSES AND STRENGTHS IN ECOSYSTEM MODELING

Critical Weaknesses of Ecosystem Modeling

1) Modeled representations of many biotic and abiotic interactions are not ecologically strong nor fully understood.

2) In general, models are not subjected to a rigorous and iterative process of repeated applications to verify and refine model structure (e.g., variables, rates, processes).

3) The functional form of critical process definitions are often ill-derived (e.g., Q10 functions for temperature forcing on biotic process rates).

4) There is a high degree of uncertainty about appropriate levels of space and time aggregation, particularly relating to pollutant-ecosystem interactions.

5) Levels of uncertainty in overall model predictions are high (i.e., confidence levels are low).

6) Defining the bounds (what is to be included and how) of the modeled system for a given management application is difficult and often arbitrary.

7) Models seldom incorporate changes in structure caused by either natural variability, man-induced impact, or both.

8) Mathematical characteristics of the commonly used procedures (particularly for nonlinear systems of equations) are not well documented.

9) There is a definite lack of diversity of approaches to model formulation and mathematics.

10) Life histories of many critical species or species groups are not well known and cannot be modeled adequately.

Strong Aspects of Ecosystem Modeling

1) Models are excellent organizers of ecological information.

2) Models are a formal and objective vehicle for balancing differing points of view.

3) Models have the potential for answering the "what if" questions *a priori*.

4) Models have the potential ability to focus research studies and field efforts, the results of which can be used to upgrade model predictive ability.

Panel D - Integrating Ecosystem Modeling Into a Socio-Economic Framework

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Panelists: Daniel Basta, Chairman
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Richard Wetzel
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Panel D assessed ecosystem modeling as a tool for marine resource management decisions in the context of integrating and translating modeling structure and results into a viable framework for addressing socio-economic concerns and values. Our deliberations centered on the kinds of questions that might be asked within this context and how they relate to various environmental impact activities such as ocean dumping, ocean mining, coast line development, pollutant and contaminant discharges, and oil and gas exploration of the Outer Continental Shelf.

The Panel noted that ecosystem modeling approaches and applications are not directly amenable to addressing most socio-economic concerns associated with marine resource management and environmental impact decisions and policies. Ecosystem modeling efforts most often concentrate on assessing either the distribution of inputs or the effect of inputs on ecological variables of no direct socio-economic value. Specifically, traditional ecosystem models usually relate, either empirically or mechanistically and at various levels of complexity, the impact of various inputs to the system such as pollutant discharges on various system variables such as standing stocks of phytoplankton or zooplankton. Often pollutant/contaminant inputs are related to dispersal patterns and environmental concentrations.

To truly model ecosystem alterations within a meaningful socio-economic context, the traditional modeling approaches must be restructured to incorporate methodologies suitable for addressing the social and economic consequences of ecological impacts. While this observation may seem obvious, the development and incorporation of these integrative methodologies are nontrivial problems. The translation of quantified modeling results into the socio-economic realm of cost-benefit ratios, quality of life standards, commercial and recreational values, legalities, esthetics, short-term versus long-term advantages and disadvantages, etc., is neither a simple process nor an easy task. It is, however, one that should be attempted with more detail and regularity than has been done in the past.

To make ecosystem models more useful within a socio-economic context, modelers should consult with anthropologists and economists to determine what modeling outputs can be effectively valued. Questions concerning the most efficient way to extract resources and the quantification of economic losses resulting from ecological impacts thus become very relevant for modelers to consider. The socio-economic viewpoint permits a critical examination of the economic value of specific components of the ecosystem. In many cases ecosystems and ecosystem components are valued differently by different people for different reasons, uses, or purposes. Often the perceived values are based on both qualitative and quantitative criteria. Thus, the modeler must be willing to incorporate qualitative factors into the conceptual model and quantify them in the mathematical model. This need to recognize and quantify social and economic factors within the ecosystem model requires that the modeling effort be structured from a total system perspective of which the ecology/biology is only one portion of the input and output data/information bases.

Delineation and definition of the spatial and temporal characteristics of an ecosystem model must be compatible with and relevant to the decisionmaking needs of "political" and resource management units. This point cannot be stressed enough. Spatial characteristics must be pertinent to considerations of boundaries of jurisdiction and "area of impact" concerns. Similarly, temporal characteristics must be pertinent to the timing with which relevant decisions will be made. Ecosystem models structured on mesoscale magnitudes will not be applicable to small scale management needs.

Ecosystem models are often quite complex, and the addition of variables inherent to other systems might make models more complex and perhaps unmanageable. The Panel stressed and gave considerable emphasis to the notion that modelers need to tailor model complexity to the specific problem at hand. This should be accomplished through a systematic examination of all the variables and a subsequent elimination of those that seem inappropriate. If this is not done, inappropriate variables may be given undue attention at too great a cost, not only in time, effort, and money, but also in terms of the decisionmaking value and usefulness of the model's output data. Tailoring model complexity to the problem at hand will require close consultation between modelers and managers (model users) before the model is developed.

The Panel expressed a concern for the amount of effort that should be expended on ecosystem models - are they worthwhile exercises, and how much should be expended on their development and implementation? It is difficult to answer this question in abstract without a particular well-defined problem or set of socio-economic questions. The Panel concurred that any modeling effort will have some value to decisionmakers, given that:

1) the needs of the decisionmakers have been anticipated;

2) the model is relevant to the needs of the decisionmaker;

3) the model has some predictive value;

4) the model focuses on critical decision outcomes;

5) the model gives results with an acceptable level of variability; and,

6) the modeling effort in the context of the above criteria is cost effective.

The Panel thus stressed the need to explore each of these criteria when contemplating a modeling effort. The Panel further stressed the point that these criteria should be examined very early in the planning stages of the modeling effort. Model development should not progress beyond the conceptualization stage until the above criteria have been considered in detail and met.

Another viewpoint expressed by the Panel concerns the modeling effort. This viewpoint focused on the importance of ecosystem modeling as a research tool and framework for formulating problems, e.g., in evaluating logic, information flow, and structuring the development of important questions which would not otherwise evolve. Planning and development and/or application of an ecosystem model will always require that a screening or preliminary assessment by conducted by both "expert modelers" and "decisionmakers."

The Panel expressed concern that there was a serious lack of information and data for many ecosystem modeling applications, particularly when these applications were oriented to specific societal problems. The Panel feels that this serious deficiency stems from conceptual gaps in biology and socioeconomics; if the utility of the modeling process is to be improved, these conceptual gaps will need to be identified and rectified. The Panel further noted that unsuccessful ecosystem modeling attempts sometimes resulted not from the model itself, but from the quality and applicability of of the data base used to conceptualize and quantify the model. Thus, if a model is structured to be highly dependent on the quality and applicability of the input information, the modeling results will be no better than the data base. Inadequate input information will always yield less-than-satisfactory modeling results under these circumstances. On the other hand, the Panel recognized the need for some models to be based upon inadequate information/data bases so that these models could be useful in specifying data needs and design of sampling programs and studies to elucidate unknown or poorly understood ecological mechanisms and processes. A careful consideration reveals this to be another difference between the types of models and their intended purposes (as suggested above). If predictive models are to be useful, they must be designed for application to real world, resource management decisions and must be based on good, scientifically defensible data. Research models, on the other hand, may be designed on inadequate data and theoretical constructs since one of their key values is to delineate the information needs that must be addressed in the next round of data generation and investigation.

In concluding its deliberations, the Panel felt that there is a strong potential for ecosystem modeling applications within the context of a socioeconomic framework. Modeling could provide administrators and decisionmakers with a prediction of the outcomes of their decisions, albeit this would have more practical value if presented as probabilities of occurrence. As such, these models would improve the decisionmaking process. However, the value that the Panel saw in the ecosystem modeling approach was not without qualification. In a practical sense, models need to be more relevant to the needs of decisionmakers. This will occur, not through simply better communication between modelers and decisionmakers, but through explicit plans for modelers and decisionmakers to interact throughout the course of the modeling exercise. The objective is for the decisionmaker and his or her staff to obtain a general understanding of the operation and critical elements of the model, its strengths and weaknesses in a given application, and its accuracy and precision. Of particular importance in this regard is the need for modelers to develop models which clearly address the original questions and concerns; the model must be responsive to its original purpose. Ultimate utility of a model to a client/user most likely will depend directly on the quality of effort spent in problem definition and the scoping of objectives, resources, constraints, and strategies. Toward this end there may be a role for a modeler/practitioner in the procurement phase comparable to that of an architect/engineer for a construction project. In terms of scope, the Panel felt that if models were to address socio-economic questions and concerns, it was critical that the models contain socio-economic components interacting with the biological components. This would require enhanced communication between biologists/ecologists, social scientists, and economists.

QUESTIONS POSED TO THE PANELS FOR THEIR CONSIDERATION

Six general questions were put forth for possible consideration by the panels as they related to the panel topic:

1) For what portion of the marine biome and for what ecosystem components can we use predictive (forecasting) models? What are the general limits or confines of predictability? What are the spatial and temporal limits or restrictions?

2) Site specific models have a high degree of "realism" and perhaps precision, but they usually are not transferable to other geographical areas or problems. Is it feasible/practical to develop a "generic" model which is adaptable to other areas and problems? Are certain model components translocatable? How do we relate tradeoffs between resolution, precision, and transferability? Specificity versus generality?

3) At which level and to what extent should modelers be involved in the up-front planning and designing of ecosystem analysis/assessment programs? Who should they interface with? What should their input be?

4) What are the advantages and disadvantages of empirical versus mechanistic models, static versus dynamic models, and holistic versus limited focus (i.e., single species or several species) models? How do we relate the modeling approach to the management question, the available data base, and the available modeling resources?

5) Can the panels identify and give priority to several areas of "high return" research which would enhance ecosystem modeling and which can be accomplished for a relatively low financial investment (e.g., $50,000)?

6) In a very broad sense can the panels identify critical weaknesses and strengths of ecosystem modeling as a resource management and environmental impact assessment tool?

Each of the four panels addressed, to a greater or lesser extent, several or all of the six questions. Panel C chose to systematically address all six questions. Panel B formulated its own list of issue questions and found that, in substance, it was nearly identical to the above list. Panels A and D addressed the questions as they surfaced in their deliberations and debated them to a point that was necessary to arrive at a panel consensus.

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