

The Role of Models in Addressing Coastal Eutrophication

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Summary

Eutrophication models have become increasingly important tools for scientists and managers attempting to determine the best and most cost-effective way to assess and control the flow of nutrients into estuaries and coastal waters. However, use of models has been controversial, in part because there is not yet agreement in the scientific community over what criteria should be used to judge models used for management decisions. In this chapter we discuss ways in which eutrophication models could be better evaluated for use in a management context. Rather than being judged on their ability to match current conditions, we advocate robust tests of how these models perform in response to perturbations in nutrient inputs or other stressors.

We note that there have only been a few attempts to systematically compare different eutrophication models with common data sets, something that is now being widely employed for models being developed for use in other management contexts. Mesocosms, and other large-scale experiments, could also be used more effectively to test model responses to changes in conditions. Finally, tracer studies are now becoming a powerful tool for model development and testing.

Introduction

Coastal areas throughout the world receive large amounts of nutrients from sewage, fertilizer, animal wastes, and atmospheric deposition (NRC 2000). These nutrients stimulate primary production that can lead to the excessive production of organic material, or eutrophication (Nixon 1995). Excessive nutrient enrichment has had detrimental effects on many coastal ecosystems, including depletion of oxygen in bottom waters, blooms of nuisance or toxic algae, decreased water clarity, losses of finfish and shellfish, and the loss of submerged

aquatic vegetation. The problem is widespread: for example, symptoms of eutrophication are present in nearly one third of the estuaries in the United States (Bricker et al. 1999).

There has been a concerted effort in many areas to prevent the eutrophication of clean coastal and estuarine systems and to reverse the eutrophication of highly impacted ones through the management of nutrient inputs. As part of this effort, models have become increasingly important tools for scientists and managers attempting to determine the best and most cost-effective way to assess and control the flow of nutrients into estuaries and coastal waters (e.g., Boesch et al. 2001). In this chapter, we will briefly discuss some of the successes and problems with two classes of models that have seen widespread use for the management of eutrophication in the coastal zone. The first are models that are used to calculate nutrient inputs from the watershed. The second are numerical "water quality models" used to calculate nutrient, carbon, and oxygen dynamics in the water column and sediments. The goal of this paper is to explore how such models could be developed, tested, and used in a management context more effectively.

Nutrient-Input Models

A variety of approaches have been used to assess nutrient loading from watersheds, and several papers have reviewed the major models available (Deliman et al. 1999; NRC 2000; Valiela et al. 2002). The field has advanced rapidly and many models now use a hybrid approach to calculating nutrient inputs. The models, however, tend to fall into three categories: (1) complex simulation models, which have been developed to predict both hydrologic and water quality processes in the watershed under dynamic or steady-state conditions (e.g., HSPF; Bicknell et al. 1993); (2) "spreadsheet models," which use land-use data to generate export coefficients or mean concentrations to calculate loads (Valiela et al. 1997); or (3) statistical models, which use empirical data to develop relationships between characteristics in the watershed and nutrient export (e.g., SPARROW; Preston et al. 1998; Caraco and Cole 1999). There are many examples of models of each type being successfully applied in a variety of watershed types, but in nearly all cases there needs to be some adjustment to the parameters when the models are applied to a different location.

Nutrient-input models are judged based upon how well they can predict observed nutrient concentrations in streams or groundwater entering coastal systems. If total nutrient loading were the only objective, this would be sufficient, but if mitigation efforts are to be undertaken, it is probably equally, if not more important, to know the specific source of nutrient loading. For example, some models developed for the northeastern United States may have greatly underestimated the importance of nitrogen from precipitation and erroneously assumed a very high nitrogen export from undisturbed forests (NRC 2000). Another

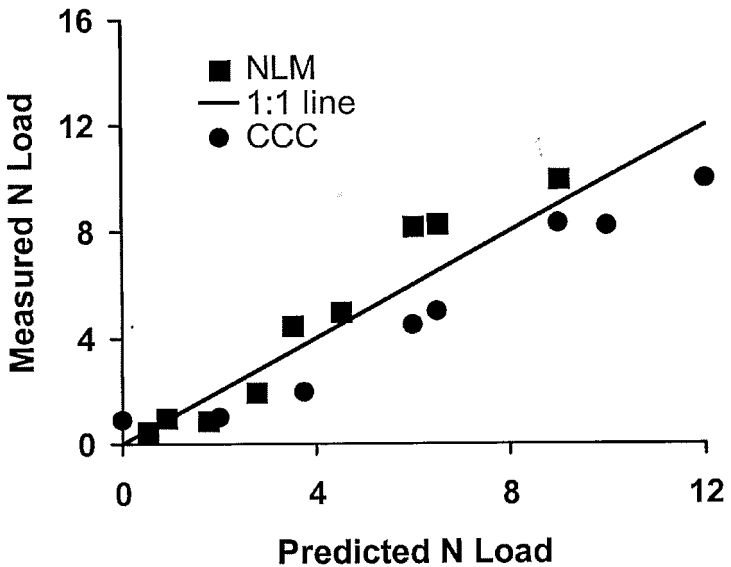


Figure 18.1. A comparison of the predicted versus the measured N loading to 9 sub-estuaries using 2 different models of N inputs from watersheds (Valiela et al. 2002).

important consideration is to determine whether the model accurately reflects how nutrient inputs to the receiving waters change with changes within the watershed.

Applying models that were developed using different structures and assumptions to the same watershed and comparing how they partition nitrogen sources is one way to examine the robustness of the models' conclusions. Valiela et al. (2002) compared a number of different models used to estimate land-derived nitrogen inputs to a series of nine small sub-estuaries in Waquoit Bay having very different population densities. The predicted nitrogen loads were compared to nitrogen loads that had been calculated from nitrogen concentrations measured in groundwater and recharge estimates. Some of the models were developed for use in these watersheds, which consist of unconsolidated sands and glacial till, while others were developed in other watersheds.

Process, spreadsheet, and statistical models were included in the comparison. In general many of the models were judged to be quite "responsive": that is, they predicted a change in loading that correlated well with the observed loading change. In contrast, there was a very large variability in the predictions of the percentage of nitrogen coming from different sources. For example, both the Waquoit Bay nitrogen-loading model (NLM; Valiela et al. 1997) and Cape Cod Commission model (CCC; Eichner and Cambareri 1992) are quite accurate in their prediction of nitrogen loading (Figure 18.1) and show excellent respon-

siveness, but they differ in how they predict the N is derived. The NLM predicts that 47% of the N inputs are from wastewater, 32% come from the atmosphere, and 21% is attributable to fertilizer. In contrast, the CCC predicts that 83% comes from wastewater while only 1% comes from the atmosphere. Overall, the range of predictions for wastewater-derived N varied from 13 to 83% (Valiela et al. 2002). From a management point of view, this large range in the estimate of the percentage of N coming from a controllable source creates grave problems when attempting to develop a strategy of nitrogen mitigation.

Linked Hydrodynamic-Water Quality Models

Because of the dynamic nature of estuarine and coastal ecosystems, there is great interest in simulation models that include transport and stratification. It is now common, and nearly expected, that models of the coastal ecosystems will include a sophisticated treatment of the hydrodynamics of the system. One of the most common types of eutrophication models, which has been widely applied by managers, is the coupled hydrodynamic-water quality model. Water quality models calculate parameters such as dissolved oxygen, chlorophyll *a*, and particulate carbon on hourly or daily time scales using nutrients, light, and temperature as drivers.

Water quality models differ in the complexity of the hydrodynamics schemes employed as well as the kinetic expressions used to describe water quality variables (reviewed in NRC 2000). Many are derived from a common model or contain common components. For example, the CE-QUAL-ICM model being used in Chesapeake Bay (Cero and Cole 1993; Cero 2000) and the ECOM/*EM model applied to Massachusetts Bay and the New York Apex (HydroQual 1991, 1995) both contain a common sediment diagenesis submodel (DiToro and FitzPatrick 1993) and many key features of water column dynamics. These models, as do most models designed to examine eutrophication problems (e.g., Humborg et al. 2000), share common features in that they use light, temperature, and nutrient concentrations and a kinetic framework to calculate phytoplankton production. Phytoplankton may be broken down into several functional groups. In the Massachusetts Bay model, phytoplankton were broken down into a winter group (diatoms) and a summer group that differed in their silica requirements. The Chesapeake Bay model has three groups: diatoms, cyanobacteria, and green algae. Production is lost through respiration (modeled as a function of temperature) and predation. Most eutrophication models do not model higher trophic levels but treat predation as a loss term that is a function of temperature and that may also be a function of phytoplankton type. Lost production enters one of several possible dissolved or particulate detrital pools that differ in their decomposition rates.

Current eutrophication models have several limitations (NRC 2000). In general, the models are much more successful in predicting the seasonally averaged

conditions in the estuary than in predicting the highs and lows that are often of greatest concern to management. The inability to model toxic and nuisance algae is another serious limitation. All of the nutrient-based models have been criticized for their very limited ability to deal with higher trophic levels. In spite of these limitations the models have seen widespread use, and eutrophication models have been successful in helping guide management decisions in a number of locations.

Case Studies: Massachusetts Bay and Chesapeake Bay

A linked hydrodynamic–water quality model was used to predict water quality in the Boston Harbor–Massachusetts Bay region given a variety of locations for the municipal wastewater outfall (HydroQual 1995). The modeling effort was supported by an extensive monitoring effort. Simulations were run with both the existing harbor surface wastewater discharge and with a proposed location offshore into deeper (30–35m) water in Massachusetts Bay. The simulations showed lower chlorophyll conditions in both the harbor and the bay for most of the year after relocation. This was because the model predicted that with the harbor discharge most of the nutrients were already reaching the bay. With relocation to the bay, nutrients from the outfall remain below the photic zone for much of the year, leading to somewhat lower overall production than before. Mass-balance calculations, stable isotopic results, and simple box models of N processes in the harbor (Giblin et al. 1997; Kelly 1999) also indicated that there was a significant export of N from the harbor into the bay with the harbor outfall and provided corroboration of the model predictions of current conditions.

Other scenarios were conducted in which the model was used to determine the importance of atmospheric deposition and secondary treatment on chlorophyll and oxygen levels. Finally, the model was used to calculate the percentage of the nitrogen coming from anthropogenic sources compared to oceanic sources. The model calculated that more than 90% of the nutrients entering the Massachusetts Bay system came from oceanic sources, indicating the small role that anthropogenic nutrients may currently play in controlling productivity in this system as a whole.

Overall, the model results were useful in both management and public forums. The strong effort to corroborate some portions of the model predictions by independent means was very helpful in gaining support for the use of the model. Eventually the outfall was moved.

The original impetus to develop a linked hydrodynamic–water quality model of Chesapeake Bay was to better understand contemporary and historical trends in bottom-water anoxia (Cero 2000). After initially developing a two-dimensional vertically averaged model, the decision to proceed with a full three-dimensional model was made, and the model was completed in 1984 (HydroQual 1987). The initial model did not include sufficient detail in sediment processes, so the model was coupled with a sediment diagenesis model

(DiToro and FitzPatrick 1993) and a watershed model. Subsequently an atmospheric model was added. The model has been used extensively for scenario testing (Cero 2000), especially for examining how the reduction of nutrient loads will affect bottom-water anoxia. However, use of this model for managing nutrient inputs to Chesapeake Bay has been controversial, in part because there is not yet agreement in the scientific community over what criteria should be applied for judging models used in management decisions (Blankenship 2000).

Differences between Water Quality Models and Other Food Chain Models

The treatment of zooplankton is the one area where the current generation of coastal and estuarine eutrophication models differs significantly from pelagic food chain models being developed by the oceanic research community. While the impetus for many of the oceanic models was research, models are now also being developed with management issues in mind. The oceanic models have always placed a much greater emphasis on including higher trophic levels as specific components in the model, either in relatively simple formulations that use a single zooplankton compartment (e.g., Guillaud et al. 2000) or by simulating complex pelagic food webs that include microzooplankton and bacteria (Tett and Wilson 2000). Originally, pelagic food chain models, unlike the coastal eutrophication models, did not have a benthic component, but more recent models of large systems, including that of the North Sea, are attempting to incorporate pelagic and benthic processes as well as multilevel food-chain dynamics (e.g., Baretta et al. 1995).

The Future of Models Used to Address Coastal Eutrophication

Current water quality models, which generally focus on algal biomass and carbon and oxygen dynamics, do not address all the issues that are of concern. The loss of specific habitats, such as submerged aquatic vegetation, is particularly important in many shallow systems. Habitat models for these ecosystems are currently under development, and managers have begun to use them to test restoration goals (Wetzel 1996). These models are also being coupled to water quality models.

Higher trophic levels offer a great challenge. Attempts to use models to analyze the affects of nutrients on higher trophic levels have focused largely on single species or single functional groups and have had limited successes. Eutrophication models coupled to even fairly simple food web models have yet to become fully functional. Recently, progress has been made coupling bioenergetic models of higher trophic levels with spatially explicit models of physical and biological parameters such as oxygen, salinity, temperature, and prey density (Dermers 2000).

Testing and Using Eutrophication Models for Management: Issues and Approaches

As models grow in size and complexity, there are more terms that can be "tuned" to fit the available data. When the number of model parameters exceeds the number of data input parameters it is possible to develop a model that can fit the observed data with more than one parameter set (Aber 1997; Chapter 11). Consequently, model predictions will vary depending upon the parameter set chosen to fit the data (Beck 1987). Oreskes et al. (1994; Chapter 2) have suggested that models cannot be validated or verified but only confirmed by demonstrated agreement between observation and prediction and that such confirmation is only partial. One solution to this problem is to be sure that managers recognize the uncertainty in the models and use models as tools for exploring questions rather than for generating answers (Deegan et al. 2001; Chapters 7 and 16). While this would reduce the misuse of models, it still leaves open the question as to what sorts of criteria should be used to judge the performance of a model when the goal is investigating possible future management actions (Chapter 7).

We suggest that an important criterion for management models is that the model response to the stressor of interest, such as a change in nutrient loading, be confirmed in some manner outside of the calibration data set. Inter-comparisons of models across a range of systems that receive different level of nutrient inputs is one way to test such models. Experimental systems, such as mesocosms, offer another. A third method, which is only now being widely applied, is the use of stable nitrogen isotopes as tracers of processes to provide independent confirmation of nitrogen flows within models.

Intercomparisons

Multisite intercomparisons of complex models are expensive and difficult to perform but can prove to be exceedingly valuable in demonstrating the state of the science (Chapter 12). Managers will have more confidence using models in areas where there is substantial agreement among models. Areas where there are substantial differences highlight research priorities. Both the terrestrial-ecosystem-modeling community (e.g., Kicklighter et al. 1999) and the global-climate-modeling community have benefited from intercomparisons. Policy makers in the global-change arena now commonly assess the impact of policy options using climate predictions generated from several different general circulation models.

Comparisons of nutrient-input models, such as that carried out by Valiela et al. (2002) should become more widespread. Intercomparisons of water quality models is more problematic because the biological models are now coupled to complex hydrodynamic models, and the hydrodynamic models themselves differ between systems. However, a first step would be to test only the water quality portion of the model using the available site-specific hydrodynamic model.

There are now sufficient systems where there are both good hydrodynamic models and a wealth of monitoring data where an intersite, intermodel comparison could be undertaken. Ideally the comparison would include models with a variety of structures and complexity (Chapter 12).

Experimental Systems

Experimental systems offer another way to test the biological portion of eutrophication models in isolation from the hydrodynamic component. A modified version of the eutrophication model used in the Chesapeake, Long Island Sound, and Massachusetts Bays programs, was used to model more than two years of data from a series of mesocosm experiments run at the University of Rhode Island Marine Ecosystem Research Laboratory (MERL). The mesocosm experiments were done to examine the effects of increased nutrient inputs to coastal waters with loadings calculated to mimic from 1 to 32 times the current area loading to Narragansett Bay (Nixon et al. 1986). The model reproduced water-column oxygen quite well and captured the overall seasonal cycles of chlorophyll *a* in all treatments. However, the model tended to overpredict the average chlorophyll *a* levels while missing the extreme high and low values (NRC 2000). This result was similar to that observed when the model was applied to natural settings and suggested that the biological component of the model, rather than the hydrodynamic component, was responsible for the model response curve being smoothed relative to nature. The sediment portion of the model was able to match the observed distribution in sediment oxygen demand quite well, but the modeled fluxes differed from the data for silica and nitrate fluxes (DiToro 2001). As a consequence, the model did not reproduce some aspects of the water-column nutrient dynamics. This type of comparison of model to data over a range of nutrient values helped demonstrate the responsiveness of the model to changes in nutrient inputs. Direct comparisons of model output to a range of conditions can give managers a much better feel for the strengths and limitations of the model.

Experimental systems can also be used to distinguish limitations in model structure from problems with parameter values. Vallino (2000) compared a simple food web model (Figure 18.2) to data from four experimental pelagic mesocosms: control (bag A); a large single addition of dissolved organic carbon (DOC) to reach a concentration of 300 μM (bag B); daily additions of dissolved inorganic nutrients to reach target concentrations of nitrogen (5 μM), phosphorus (0.5 μM), and silica (7 μM), (bag C); and a one time addition of DOC with daily nutrient additions (bag D). The experiment was run for 21 days and a host of variables were measured daily.

The model was designed to focus on the importance of organic matter production and consumption and to specifically examine the potential role of DOC in altering the competition between bacteria and phytoplankton for nutrients. Unlike most eutrophication models, bacteria and zooplankton (heterotrophs) were specifically modeled. The model contained 29 parameters governing the

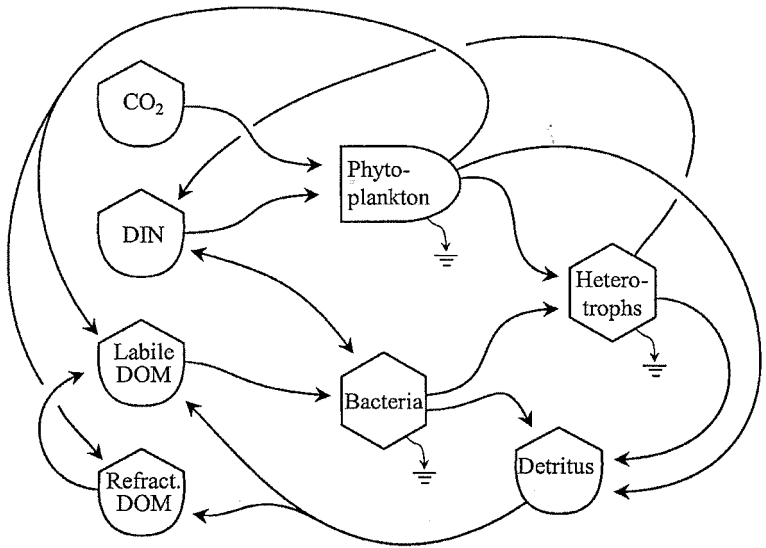


Figure 18.2. The structure of the foodweb model developed by Vallino (2000) and tested in several mesocosms.

growth kinetics of the organisms and decomposition of the organic matter pool, and 10 parameters were used to specify the initial conditions of the state variables. Three of the parameters were fixed and the other 36 were set using data assimilation to determine the optimum values. The parameters were fit to the data from the nutrient + DOC bag (bag D) and then applied to all other treatments.

The optimized parameter set was able to do an excellent job simulating conditions in bag D as evidenced by the correspondence between modeled and measured concentrations of chlorophyll *a*, DOC, and DIN (Figure 18.3). Unlike most eutrophication models, the model captured the full range of primary production (not shown) and chlorophyll *a* values although the experiment was run for only 21 days. When the same calibration was applied to the DOM-only bag (bag B) the correspondence between the model and the data was quite good (Figure 18.3). The model's fit to the control bag (A) may or may not be considered adequate depending upon the application. The model completely fails to capture the chlorophyll *a*, DIN and DOC dynamics observed in the nutrient-alone bag (C) (Figure 18.3).

To examine the effect of using a specific treatment on parameterizing the model, the model was reparameterized using data from the nutrient-only treatment (bag C). Again the model was able to do an excellent job mimicking the behavior of the bag from which it was calibrated (Figure 18.4). The fit to the

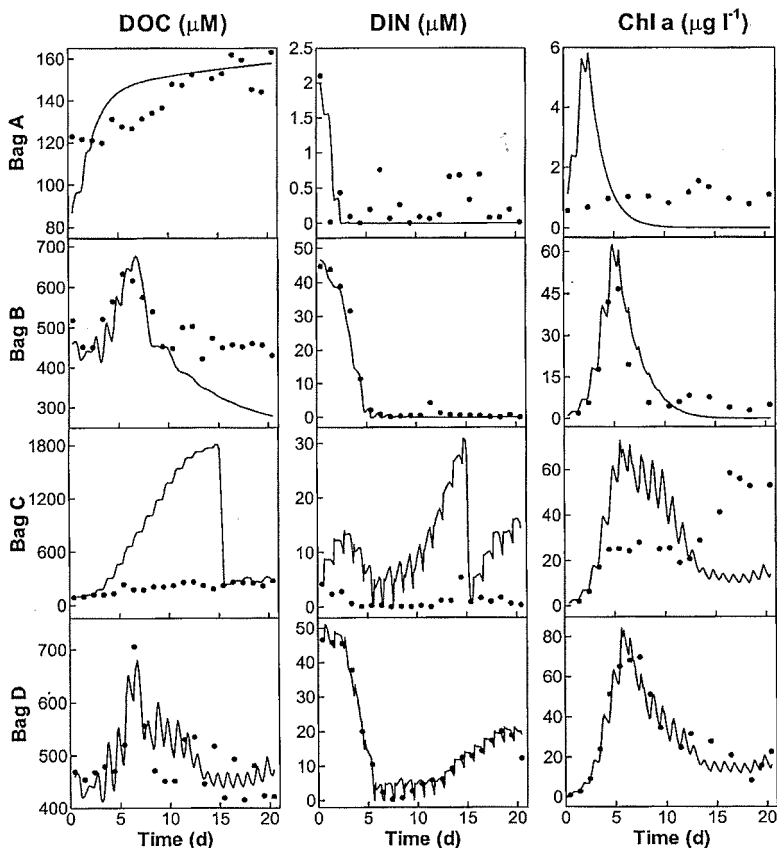


Figure 18.3. The simulated and observed data from four mesocosms, Bag A (control), Bag B (DOM added), Bag C (inorganic nutrients added daily), and Bag D (DOM added and daily additions of inorganic nutrients). All parameters were fit only using the data from Bag D. Lines indicate model output while symbols show data.

control bag (A) is poor, but the model did capture the low chlorophyll *a* values of this treatment relative to the nutrient addition bags (Figure 18.4). Where the model completely failed is in simulation of either treatment when DOC was added (bags B and D; Figure 18.4).

Why does the model fail to capture the dynamics of all of the bags using a single parameter set? When parameterized individually, the model could reproduce specific treatments with good accuracy and precision. This suggests that the problem is not parameter uncertainty but rather that the structure of the model is not appropriate for modeling across a range of treatments that include

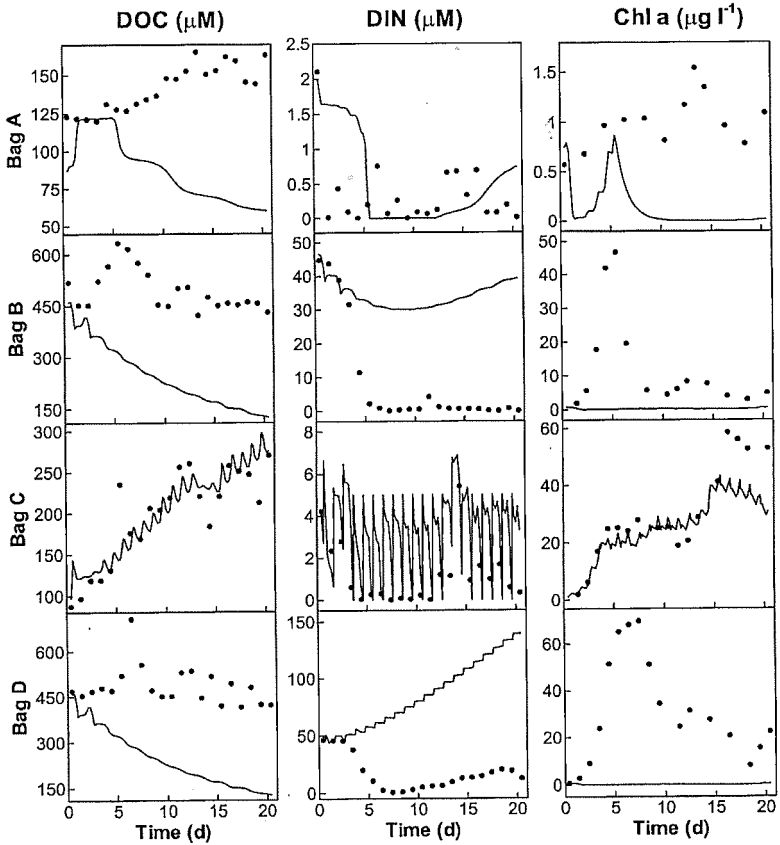


Figure 18.4. The simulated and observed data from four mesocosms, Bag A (control), Bag B (DOM added), Bag C (inorganic nutrients added daily) and Bag D (DOM added and daily additions of inorganic nutrients). All parameters were fit only using the data from Bag C. Lines indicate model output while symbols show data.

both high DOC and low DOC values (Vallino 2000). Therefore, we conclude that this model is inadequate as a prognostic tool. This type of insight would not be possible without comparing the model to different treatments.

Experimental systems can provide the range of conditions needed to vigorously evaluate the responsiveness of eutrophication models. They will not replace field data or cross-system comparisons, but their utility should not be underestimated.

Tracers

Stable isotopic tracers are now being used in a variety of ways to develop and evaluate models. In some cases, there is a strong natural isotopic signal that can be exploited. For example, different models used in calculating nutrient loading to the Waquoit Bay sub-estuaries calculate very different source strengths even when the total N-loading predictions are similar (Valiela et al. 2002). Studies have shown that nitrogen from on-site septic systems is considerably heavier than that derived from atmospheric or fertilizer sources. The $\delta^{15}\text{N}$ signal of dissolved N in the groundwater entering the sub-estuaries allows for an independent estimate of the wastewater source strength (McClelland and Valiela 1998) that then can be used to further evaluate the predictions of the different models.

Although there are places when natural isotopic signals can be exploited as a natural tracer with success, there are many places where they cannot. However, the lower cost of isotopes has now made it possible to carry out tracer additions to natural systems. An extensive comparison of the N processing of streams and estuaries using tracer studies revealed a great deal about N processing in these systems (Peterson et al. 2001) and led to the development of better models of nutrient dynamics in streams. A nitrogen isotope tracer addition to forests has helped improve our ability to model N cycling in forests (Currie et al. 1999).

Model Parameterization

Rigorous model–data comparisons are the key to creating confidence in the utility of the model in the management community, yet in many areas there are disagreements over the form that this comparison should take (Chapter 12). One frequent recommendation is that models should include realistic, empirically based parameterizations that would tie the model to field and laboratory observations (Aber 1997; NRC 2000). The counterargument is that the model is a simplification of the system (Chapter 2). Compartments and flows may represent aggregations that have no exact equivalent in the real world and as such, cannot be individually calibrated from laboratory data (Vallino 2000; Chapter 13). Indeed, Wallach and Genard (1998) have shown that a better fit to observations can be obtained when parameter values are allowed to exceed their expected ranges in highly aggregated models. Therefore calibrating the model using data-assimilation techniques helps remove the parameter uncertainty that cannot be removed through laboratory experiments or observations. Additional information about the sensitivity of the parameters in the model and codependence of parameters can be investigated through these techniques (discussed in detail in Vallino 2000).

An analysis of the parameter values calculated using data assimilation techniques under different experimental treatments can also lead to insights about the model and help identify areas where the model structure is not adequate, or

Table 18.1. A comparison of the values of some parameters determined using data from either the DOM + DIN addition experiment (Bag D) or the DIN only addition (Bag C) for the model shown in Figure 18.2 (Vallino 2000). See text for details.

<i>Parameter</i>	<i>Bag D</i>	<i>Bag C</i>
Maximum specific DOC uptake rate by bacteria	49.9 d ⁻¹	39.2 d ⁻¹
Half-saturation constant of DOC consumption by bacteria	48.8 μM	24.6 μM
Mortality of bacteria	48.4 d ⁻¹	49.3 d ⁻¹
Decomposition of detritus	49.6 d ⁻¹	50.0 d ⁻¹
Maximum growth efficiency of heterotrophs	0.151	0.866
Half-saturation constant of H feeding on A and B	200 μM	0.31 μM

where compartment aggregation leads to an inappropriate representation of the system. For example in the food web model (Figure 18.2) the best fit for the model to the DOC + DIN treatment resulted in growth kinetics for bacteria that would be considered extreme and unrealistically high decomposition rates for detritus (Table 18.1).

When the model was recalibrated to the DIN-alone bag, growth parameters were still high and the values showed little change from the initial calibration (Table 18.1). This analysis suggests that the bacteria and detritus compartments are actually composed of multiple compartments and that when aggregated as single compartments, a high turnover is necessary to mimic the system. Had the model successfully been able to mimic all nutrient and DOC treatments with a single calibration, we suggest that it should not have been rejected as a potential tool for exploring management options simply because of the parameter values, which by virtue of their aggregation may in fact not be comparable to laboratory derived values.

In contrast, there were large changes in the parameters between the model fits to bag C versus bag D governing zooplankton growth kinetics (Table 18.1). In bag C, which did not receive DOM additions, zooplankton growth parameters favor a more efficient growth behavior. This is perhaps understandable, since the microbial loop, which can support higher trophic levels, is less important in bag C. When the parameters from the fit to bag C are used to describe the growth in bag D, the zooplankton are able to readily graze down the phytoplankton when given the additional support provided by the microbial loop, which in turn is supported by the increase in DOM availability. In essence, the model underestimates the importance of the microbial loop when fit to bag C

and overemphasizes it in bag D. This indicates that the model structure governing the microbial loop is not well articulated in the current model, hence the lack of model robustness between treatments.

Conclusions

Modelers and managers are well aware that models are simplifications of the systems they are designed to mimic. However, it may not always be obvious where the multiple processes are hidden within the model structure. A calibrated model may appear to represent a single explicit process firmly based upon empirical data, and yet when performing in a calibrated model, this process may actually represent a quite complex set of processes that are lumped together and no longer completely represent the observations. In this case, the fit between model output and observations may be satisfactory only under a limited range of conditions.

Successful use of models in a management context does not require that they perfectly mimic nature (Chapter 2), yet models are often evaluated by how well they match current observations over seasonal or annual timescales. In fact, the ability of a model to match current conditions may be a poor predictor of how the model will respond to changes in conditions outside the range of current observations. If the goal is to understand the response of the system to stressors, a different criterion for the testing and evaluation of eutrophication models may be warranted.

We need to find robust ways to test how models perform in response to perturbations and to evaluate how realistic the response appears. To date there has been little attempt to systematically compare different models with common data sets. Such an exercise, especially if carried out using data from a range of coastal ecosystems, would improve our understanding of the strengths and weaknesses of current eutrophication models. The coastal-eutrophication-modeling community could also make much better use of mesocosms, and other large-scale experiments to test model response to changes in conditions. Tracer studies offer another way of testing models and are only now being done on a widespread basis. A tighter connection between the experimentalists and the modeling community in designing and carrying out more of these types of experiments could facilitate the development of both better models and an improved understanding of how systems respond to perturbation.

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