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EUTROPHICATION MODELING FOR ESTUARINE WATER QUALITY MANAGEMENT

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Abstract: This paper presents an overview of estuarine eutrophication modeling for water quality management. The current modeling approach is first presented, followed by a discussion of key model components in estuarine models. Field data are shown to play a key role in supporting model development and calibration. The issue of linking different model components is also discussed. Finally, select case studies of modeling estuaries and tidal rivers are presented to demonstrate recent applications.

Key words: Eutrophication modeling; Estuaries; Dissolved oxygen

1. INTRODUCTION

Eutrophication has been one of the major water quality problems in estuaries and coastal waters in many parts of the world. In recent years, water quality and ecosystem models have been used extensively to assist decision-makers in developing control strategies for estuarine water quality management. Eutrophication models are now able to perform sophisticated simulations of water quality kinetics and ecosystem processes in the water column and sediments for multi-year runs. These complicated models require a significant amount of field data to support the model development and calibration analyses. This paper presents an overview of estuarine eutrophication modeling and its applications to a variety of water quality problems in decision making.

2. MODELING APPROACH

Two types of model configurations exist in hydrodynamic and water quality modeling of estuaries and coastal waters: finite-element and finite-difference models. While hydrodynamic models using finite-element method had their prime years in 1970's, they have gradually given way to finite-difference models for surface water modeling during the past two decades. The current trend in hydrodynamic and water quality modeling for coastal waters is using finite-difference or finite volume scheme. Three of the most widely used model code packages in the U.S.: EFDC (Environmental Fluid Dynamics Code), Army Corps of Engineers' CE-QUAL-ICM, and HydroQual's ECOSED are all finite-difference models.

In a recent communication with Dr. Earl Hayter of EPA's Environmental Research Lab in Athens, GA, it was clear that the finite-difference models are much more favored by the surface water modeling profession over the finite-element models. One of the main reasons for the switch is the computational effort, with finite-difference models holding almost a 10 to 1 edge in run time over finite-element models in many recent applications. During the past three years, we have been collaborating with our NATO allies on modeling nutrients in rivers and estuaries in the Baltic States and Eastern Europe, the finite-difference/finite-volume platform has been adopted and EFDC is the choice of the framework for our work.

More specifically, Hamrick (2002) has configured the EFDC (a finite-difference model) code to San Francisco Bay and bench marked with RMA-2 (a finite-element model) for the

Bay in a 2-D grid system. The EFDC configuration had 1,032 cells while the RMA-2 model had 3,786 cells. When run times were scaled by the number of cells and elements, EFDC configuration's run time per day per cell was about one eighth of RMA's run time per day per element!

In the modeling study of Florida Bay, a more advanced version, RMA-10 was used to simulate the hydrodynamics, which was linked with a finite-difference water quality model (Cerco et al., 2000). The technical problems of incorporating RMA-10 with a finite-difference water quality model were many folds. First, the RMA-10 model was never fully calibrated for real-time (at least one-year) simulations of Florida Bay. Second, mass conservation around a point (i.e., an element) cannot be transformed accurately into mass conservation for a finite-volume. As a result, interpolation and thereby, errors, were introduced into the water quality model calculations. Third, the RMA-10 grid is significantly finer than the water quality model grid, requiring spatial collapse of the hydrodynamic results, a daunting task particularly for coastal waters under tidal influence. A crucial task in linking a hydrodynamic model with a water quality model is to compare the salinity results from both models with the field data. Such a task, a key step of assuring correct mass transport in the subsequent water quality calculations, was not performed because the RMA-10 model was never calibrated with the field data.

The preferred approach today is the choice of the following:

a. Use the finite-difference method for hydrodynamic and water quality simulations with the same spatial grid.

b. Use the finite-difference method for hydrodynamics and finite-volume configuration for water quality simulations also with the same spatial grid system. This option would require a linkage procedure.

3. COMPONENTS OF A WATER QUALITY MODEL

A eutrophication-modeling package usually consists of the following basic components:

1. a watershed model
2. a hydrodynamic model
3. a water column kinetics model
4. a benthic diagenesis model

In many applications, a sediment transport model is included in the hydrodynamic module as well. A recent example of a comprehensive modeling package is the one developed jointly U.S. EPA and Army Corps of Engineers for the Chesapeake Bay (Cerco, 2001). It has all four components as listed above. In addition, an airshed model was also developed to simulate long-range transport of nitrate from mobile sources in the Midwest region of U.S.

4. MODEL LINKAGE

Over a decade ago, spatial collapsing of hydrodynamic grids to match the water quality model grids was a common practice in estuarine and coastal water modeling due to computational power constraints (Lung and Hwang, 1989). Today's computer hardware and software offer much greater computation power than 10 years ago, making spatial collapsing unnecessary in practice. For example, both hydrodynamic and water quality models in the Chesapeake Bay model (Cerco, 2001) share the same spatial grid. The hydrodynamic model uses a 3-D curvilinear grid with a finite-difference scheme while the water quality model is configured with a finite-volume scheme, fully matching the hydrodynamic grid. The finite-volume scheme essentially converts a 3-D computation into three 1-D computations, thereby significantly reducing the computation effort (Thomann and Mueller, 1987). Using the same spatial grid for the finite-difference and finite-volume schemes eliminates any linkage problems. More importantly, mass conservation, which poses a problem in finite-element methods, is maintained in finite-difference/finite-volume schemes.

Another advantage of this scheme is that the results of the calibrated hydrodynamic model may be saved and set aside to drive the water quality model runs. Thus, many water quality model runs can be made without running the hydrodynamic model each time, thereby calibrating the water quality model in an efficient manner.

5. CASE STUDIES

5.1 THE PATUXENT ESTUARY

The Patuxent Estuary, located in the state of Maryland, is a tributary of the Chesapeake Bay. Two major water quality problems consistently observed in the estuary are: low dissolved oxygen (DO) concentrations in the bottom waters near Broomes Island during summer months and high phytoplankton chlorophyll *a* levels in the upper estuary around Nottingham in spring. The goals of this study were 1) to develop and calibrate a water quality model to capture these seasonal phenomena along the estuary and 2) to use the model to predict potential changes in water quality under land use scenarios that would increase or decrease nonpoint nutrient loads.

There is a significant amount of ambient water quality data available on nutrient, algal biomass, and DO concentrations collected under the Chesapeake Bay monitoring program during the past two decades (http://www.chesapeakebay.net/wquality.htm). To save the computational effort, the modeling approach is to simulate the hydrodynamics and water quality in a single model run with the same temporal and spatial resolutions. The Army Corps of Engineer's CE-QUAL-W2, called W2 model (Cole and Wells, 2000), is suitable for this approach. The modeling grid for the Patuxent Estuary consists of 163 longitudinal segments at the surface with a length of 613 m. Each segment is then divided into 1-m layers in the water column. The entire water column is then configured with a total of 1993 segments. The hydrodynamic model is first calibrated with the 1997 – 1998 data, running from August 1,1997 to July 31, 1998 (Lung and Bai, 2003).

A crucial test of the model calibration is comparing the model-calculated vertical DO concentration profiles against data in the water column. DO concentrations measured from the Broomes Island sampling station, LE1.1 were selected for such a comparison because of the hypoxic bottom conditions during the summer months. Figure 1 displays the snapshot DO profiles from August 1997 to July 1998 vs. data. In general, the model results match the data well over this one-year period.

5.2 THE ANACOSTIA RIVER

Combined sewer overflow (CSO) related water quality problem has long been an issue in the water quality management of the Anacostia River in Washington, DC (Fig. 2), where the depression of dissolved oxygen levels was consistently observed following storms in the summer months. A predictive model was developed to quantify the effects of the reduced CSO loads on the sediment oxygen demand (SOD) as well as nutrient fluxes from the sediment. The model was used to determine the changes in SOD and ammonia flux following the implementation of potential CSO control measures by incorporating sediment diagenesis kinetics from Di Toro et al. (1990). The combination of measured fluxes, sediment production rates (Fig. 2), and nutrient concentrations in the sediment was used to support this modeling effort.

The resulting model is capable of calculating the internal fluxes across the sediment-water interface on a real-time basis for multiyear model runs (Lung, 2001). The model calculated carbon and nitrogen diagenesis rates in the sediment as well as SOD, methane, and nitrogen flux rates across the sediment-water interface (Lung, 2001). Note that methane gas is produced when its aqueous concentration exceeds its solubility in the pore water of the sediment. The diagenesis rates, SOD, and nutrient fluxes strongly depend on the seasonal pattern of water temperature in the system, reaching maximum levels during the summer months.

5.3 THE POCOMOKE ESTUARY

The Pocomoke Estuary is located on the eastern shore of the Chesapeake Bay in Maryland. A water quality model was developed for a nutrient total maximum daily load (TMDL) study. The modeling package included two modeling components: a hydrodynamic module based on the W2 model (Cole and Wells, 2000) and a eutrophication model based on the WASP/EUTRO5 model (Ambrose et al., 1993). In addition, a sediment diagenesis model based on the Di Toro et al. (1990) framework was also incorporated into the EUTRO5 code to simulate sediment-water interactions for SOD and nutrient fluxes.

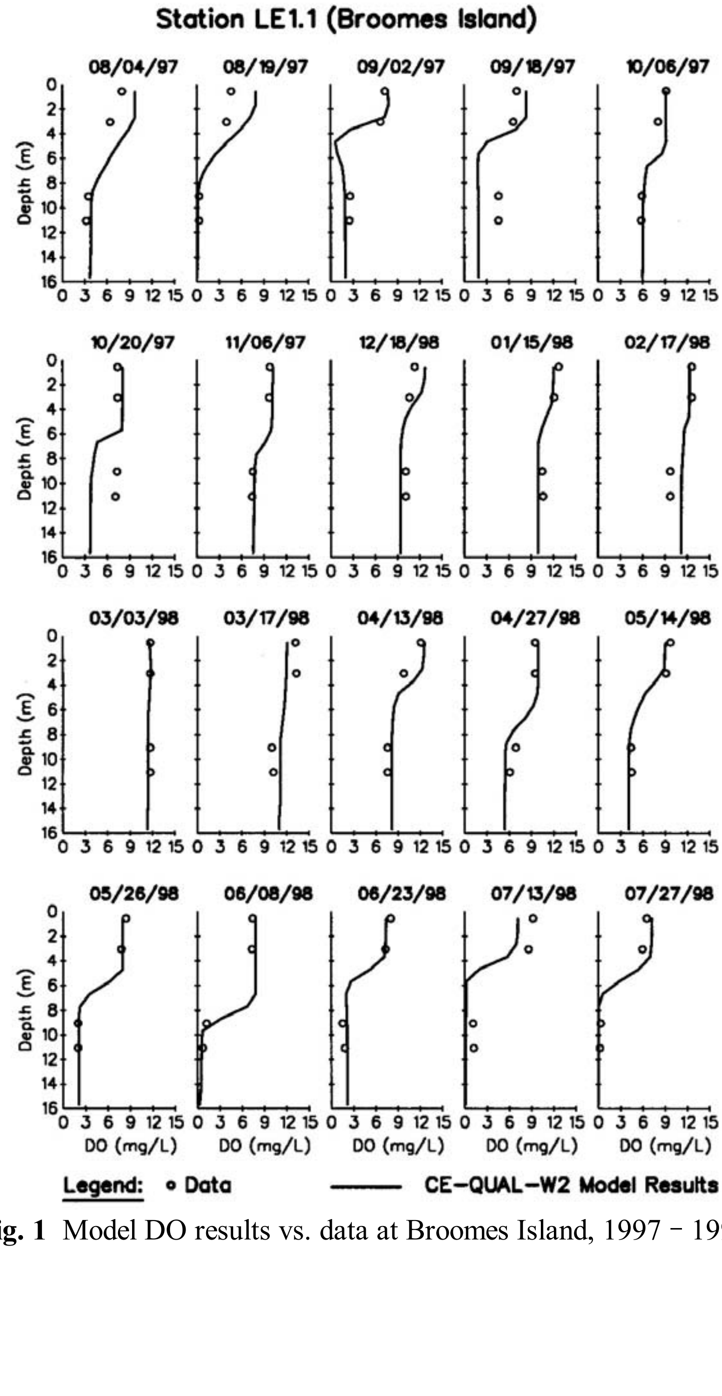

Fig. 1 Model DO results vs. data at Broomes Island, 1997 – 1998

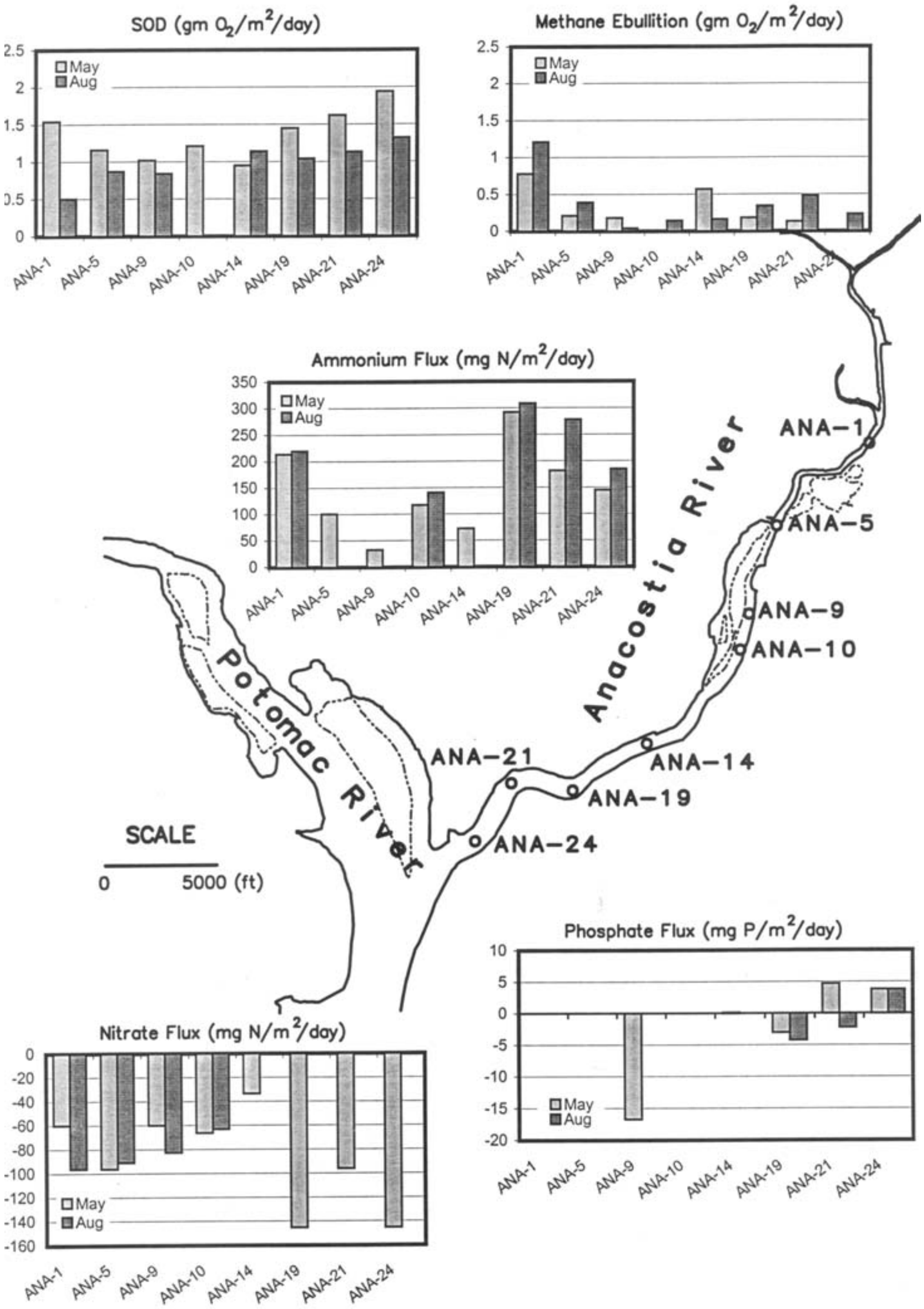

Fig. 2 Sediment Oxygen Demand and Nutrient Flux Rates in Anacostia River (from Lung, 2001)

It is essential that mass transport calculations from the WASP model match the results from the W2 model prior to the full simulations of water quality in the WASP model. Fig. 3 presents salinity results from both models vs. field data at station POK0037 for 1998, demonstrating the successful linkage of the two models.

6. CONCLUDING REMARKS

One unique aspect of water quality modeling is the importance of data. To illustrate the significance of data, once could compare water quality modeling with hydrodynamic modeling. A hydrodynamic model has at least five fundamental equations: three momentum equations, one continuity equation, and the equation of state. In a discipline founded centuries ago, these equations constitute the fundamental elements of a hydrodynamic model.

On the other hand, a water quality model is based on a single equation: mass balance. Beyond that, there are varying degrees of empiricism in the water quality model, that is, various ways of using data to formulate the water column kinetics. Due to its empirical nature, adequate, correct data are needed in model development, configuration, calibration, and validation.

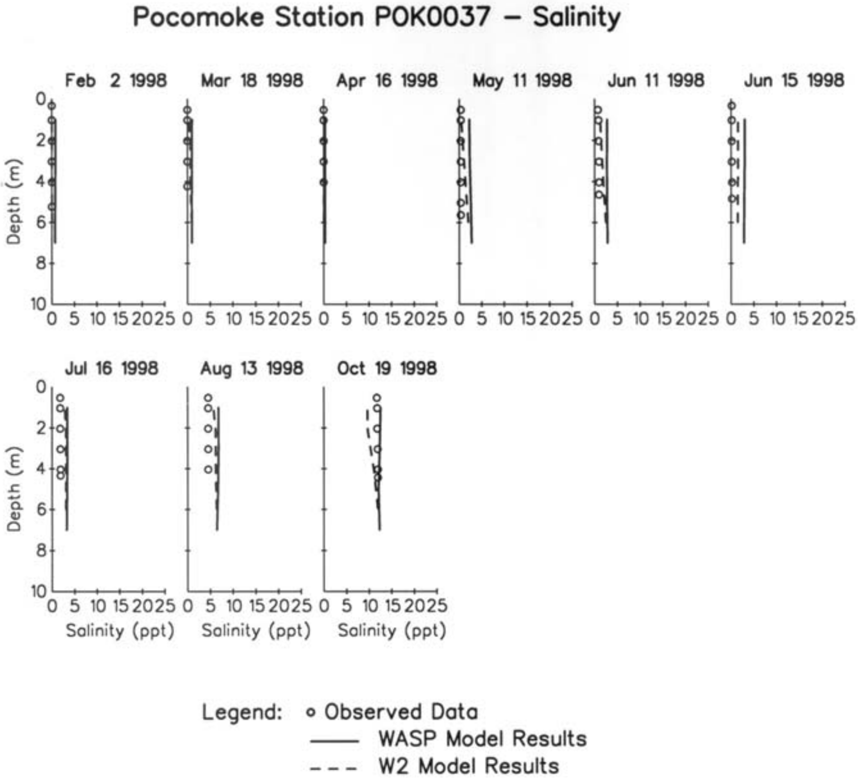

Fig. 3 Results of Linking the CE-QUAL-W2 and WASP Models of the Pocomoke Estuary

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