

Modeling biogeochemical cycles in Chesapeake Bay with a coupled physical–biological model

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Abstract

In this paper we describe the development and validation of a relatively simple biogeochemical model of Chesapeake Bay. This model consists of a 3-dimensional, prognostic hydrodynamic model that is coupled to an NPZD-type open ocean ecosystem model, which has been modified by adding additional compartments and parameterizations of biogeochemical processes that are important in estuarine systems. These modifications include an empirical optical model for predicting the diffuse attenuation coefficient K_d , compartments for representing oxygen and suspended sediment concentrations, and parameterizations of phosphorus limitation, denitrification, and seasonal changes in ecosystem structure and temperature effects. To show the overall performance of the coupled physical–biological model, the modeled dissolved inorganic nitrogen, phytoplankton, dissolved oxygen, total suspended solids and light attenuation coefficient in 1995 (a dry year) and 1996 (a very wet year) are examined and compared with observations obtained from the Chesapeake Bay Program. We demonstrate that this relatively simple model is capable of producing the general distribution of each field (both the mean and variability) in the main stem of the Bay. And the model is robust enough to generate reasonable results under both wet and dry conditions. Some significant discrepancies are also observed, such as overestimation of phytoplankton concentrations in shoal regions and overestimation of oxygen concentrations in deep channels, which reveal some deficiencies in the model formulation. Some potential improvements and remedies are suggested. Sensitivity studies on selected parameters are also reported.

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1. Introduction

In recent decades, coupled physical–biological models have been widely applied to the marine environment to simulate both physical and biogeochemical processes and study the interactions between them, especially the effect of physical factors on biological communities. The complexity of the physical models ranges from box (Li et al., 2000) and 1-D models (e.g., Marra and Ho, 1993; Doney et al., 1996; Hood

et al., 2001) to fully 3-D hydrodynamic models (e.g., Skogen et al., 1995; Lima and Doney, 2004). The biological models range from simple NPZ (nutrient, phytoplankton, zooplankton) (e.g., McClain et al., 1996) or NPZD (nutrient, phytoplankton, zooplankton, detritus) models (e.g., Doney et al., 1996; Oschlies and Garcon, 1999; Hood et al., 2003) to multi-nutrient, multi-species and size-structured ecosystem models (e.g., Moore et al., 2002, 2004; Lima and Doney, 2004). When such models are applied to estuarine and coastal waters they can provide a means of assessing the potential impacts of local management strategies and hence provide useful information to decision-makers.

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Chesapeake Bay is the largest and most productive estuary in the United States. The system is relatively shallow with a mean depth about 8 m. A deep channel running north–south more or less along the western side of the main stem dominates the bathymetry in the middle reaches of the Bay. In the lower reach, the deep channel becomes somewhat shallower and ill-defined. Similar to other estuarine systems (e.g., Lapointe and Clark, 1992; Pitkanen et al., 1993; Nagy et al., 2002), Chesapeake Bay has been suffering from degradation of water quality as a result of increased environmental stresses (Carpenter et al., 1969; Malone, 1992). Eutrophication in Chesapeake Bay has caused serious economic, aesthetic and ecological problems: harmful algae blooms (Bowers et al., 2000), loss of submerged aquatic vegetation (SAV) (Orth and Moore, 1983), hypoxia and anoxia at deep waters in summer (Cooper and Brush, 1991; Kemp et al., 1992; Hagy et al., 2004), among other things. And increased load of suspended solids directly reduces water clarity and when they deposit at the bottom they can have a detrimental impact on benthic organisms and production (Miller et al., 2002; Airoidi, 2003). Efforts have been made to reduce the N and P inputs from point and non-point sources and land-based sediment runoff with the goal of restoring the Bay to conditions observed in the early 1950s (Chesapeake Bay Agreement 1983, 1987, 2000). Numerical models have been used as a key analytic tool to provide guidelines in setting goals of nutrient and sediment reduction to achieve water quality standards (Cercio and Cole, 1994; Cercio and Noel, 2004; <http://www.chesapeakebay.net/wqcrestoring.htm>).

The Chesapeake Bay Program has developed a modeling system that is a state-of-the-art package of models that has been expanded and refined over more than a decade through the combined efforts of both scientists and managers. In an effort to model the complexity of the real world this package includes an airshed model (Regional Acid Deposition Model (RADM)) (Chang et al., 1990; Dennis, 1996), a watershed model (Hydrological Simulation Program-Fortran (HSPF)) (Bicknell et al., 1996; Greene and Linker, 1998), a 3-D curvilinear hydrodynamic model developed at U.S. Waterways Experiment Station (WES-CH3D) (Sheng, 1986; Johnson et al., 1991; Hood et al., 1999; Xu et al., 2002) and a water quality model (CE-QUAL-ICM) (Cercio and Cole, 1994; Cercio and Noel, 2004) coupled with a sediment (Di Toro and Fitzpatrick, 1993) and living resources (including SAV and benthos) model (Wetzel and Neckles, 1986; Madden and Kemp, 1996). This modeling system, which has been developed explicitly for management applications, is extremely complicated. The water quality model in this package alone has 24 state variables including two physical variables (temperature and salinity), multiple algal groups, two zooplankton groups, and multiple forms of nitrogen, phosphorus, carbon and silica. And there has been tendency toward including more and more complexity in an effort to simulate all of the potentially relevant biogeochemical processes. However, recent studies have demonstrated that more complexity in an ecosystem model does not necessarily improve model performance (Denman, 2003; Hood et al., 2003; Friedrichs et al.,

2006). Friedrichs et al. (2006) have shown that a simple NPZD model can reproduce as much of the observed biogeochemical variability as more complicated models in an open ocean system, and that more complex model formulations can lead to reduced predictive ability if they are not adequately constrained with data. In addition, simple models have many advantages in terms of identifying the most important processes and parameters that drive the observed variability. It is not clear, however, whether or not these results, which were derived from an open ocean model inter-comparison, are applicable in a complex estuarine system like Chesapeake Bay.

Nitrogen, silica and iron are the major limiting nutrients in the open ocean. In estuaries, iron is not likely to be an important limiting nutrient as a result of the close proximity of terrestrial Fe sources. Rather, in estuaries nitrogen, silica and phosphorus limit phytoplankton growth, with the latter becoming particularly important during periods of high freshwater runoff (Fisher et al., 1992). Variations in freshwater flow can therefore lead to seasonal and regional shifts in these limiting factors (e.g., D'Elia et al., 1986; Fisher et al., 1992).

Benthic processes play a far more important role in biogeochemical cycling in estuarine systems because of the closer proximity of the bottom (Boynton and Kemp, 1985; Seitzinger, 1988; Boynton et al., 1995). Sediments can be either a sink or source of nutrients. Sediment regeneration of phosphate and ammonium can provide a significant portion of phytoplankton N and P requirement (Fisher et al., 1982; Boynton and Kemp, 1985; Malone et al., 1988). The coupled nitrification and denitrification process in sediments represents an important pathway for removing nitrogen from the system (Boynton et al., 1995). These benthic influences are particularly important in coastal plain estuaries like Chesapeake Bay which are very shallow.

Another important difference between open ocean and estuarine systems is the influence of suspended sediments on light transmission in the water column. High sediment loads in estuaries, which are also associated with periods of high freshwater flow, can lead to very rapid light attenuation in estuarine waters which limits primary production (see Xu et al., 2005 and references therein).

Seasonal and interannual variability in river flow into the Chesapeake Bay is extremely large, with annual flow varying between about 20 and $60 \times 10^9 \text{ m}^3 \text{ yr}^{-1}$ (Harding, 1994). Because nutrient (and sediment) loads vary in direct proportion to flow, so does stimulation of phytoplankton growth, resulting in large seasonal and interannual variations in primary production (ranging from ~ 200 to $600 \text{ gC m}^{-2} \text{ yr}^{-1}$, Harding et al., 2002) and oxygen demand. Variations in river flow also impact sediment load/light and stratification which, in turn, controls the resupply of oxygen to bottom waters and regenerated nutrient in deep water to the euphotic zone.

The species composition of phytoplankton in Chesapeake Bay also shifts seasonally. The classic view is that in spring the diatom bloom accounts for the annual biomass peak but in summer flagellates and dinoflagellates make up most of the phytoplankton population. Accordingly, the food web of

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