Calibration and Verification of a Tidal Prism Eutrophication Model for the Lynnhaven Bay (U.S.A)

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A tidal prism eutrophication model, an one-dimensional intertidal model, is developed to study water quality conditions at small coastal basins and tidal creeks. The model simulates the physical transport processes using the concept of tidal flushing. The concept is simple and straightforward, and thus is ideal for small coastal basins with complex geometry. The model, having twenty-four state variables in the water column, simulates salinity, temperature, dissolved oxygen, three algal groups, and the cycles of carbon, nitrogen, phosphorus and silica. The model is applied to the Lynnhaven Bay, a small coastal basin of Chesapeake Bay in U.S.A. The model is calibrated using the field data collected in 1994, and then is verified using the independently collected data in 1980. The model overall gives a good reproduction of the field data, partly owing to the data collected from the field surveys specifically designed for the model application. This paper presents the procedure, and the results of the model calibration and verification.

Key words : eutrophication model, tidal prism model, small coastal basins, calibration, verification

Introduction

The coastal basins connect the land masses to the larger water body or coastal sea, and thus constitute the pathway of nutrients and sediments. The application of a sophisticated model, such as a three-dimensional model, to small coastal basins and tidal creeks is often unfeasible or impractical due to their limited size and relatively shallow depths. The tidal prism model, an one-dimensional intertidal model, simulates the physical transport processes in terms of the concept of tidal flushing (Ketchum, 1951; Kuo & Neilson, 1988). Because of its simple and straightforward nature, a tidal prism model is ideal for small coastal basins with complex geometry including those with a high degree of branching. Tidal prism models have been successfully applied to study water quality conditions at several small coastal basins (e.g., Ho et al., 1977; Kuo & Hyer, 1979; Cerco & Kuo, 1981).

To provide a tool for water quality management of small coastal basins and tidal creeks, Kuo & Park (1995a) has developed a tidal prism eutrophication model. The model is applied to the Lynnhaven Bay, a small coastal basin of Chesapeake Bay in U.S.A. The model is calibrated and verified using extensive field data collected in 1980 and 1994, and the results of the model application are briefly presented in Park & Kuo (1996a). This paper gives a detailed presentation of the procedure, and the results of the model calibration and verification for the Lynnhaven Bay.

Model Description

In the present tidal prism eutrophication model, the change of mass in a model segment over one tidal cycle, Δm, may be expressed as:

$$\Delta m = [\text{mass in}] - [\text{mass out}] + [\text{external sources}] + [\text{kinetics}] \quad (1)$$

where the term [external sources] includes point and nonpoint source inputs over one tidal cycle and the term [kinetics] represents the biogeochemical kinetic processes, which may cause an increase or a decrease of a particular substance within a segment. The first two terms in the right-hand side of Eq. 1 represent mass transport by the water movement and are combined to be referred to as "physical transport processes", and the last term [kinetics] is referred to as "kinetic processes" in this paper. The formulations of the physical transport and the kinetic processes, and their solution schemes are briefly described below. A complete documentation of the present model can be found in Kuo & Park (1994).

1. Physical Transport Processes

The tidal prism model, an one-dimensional intertidal model, simulates the longitudinal distribution of dissolved and particulate constituents at slack-before-ebb, therefore it is more applicable to an elongated coastal embayments or tidal creeks. The rise and fall of the tide at the mouth of a tidal basin cause an exchange of water masses through the entrance. It results in temporary storage of seawater and freshwater in the basin during flood tides, and drainage of these waters during ebb tides. Since water brought into the basin on flood tides mixes with the water inside, a portion of the pollutant mass in the basin is flushed out on ebb tides. This flushing mechanism due to the rise and fall of the tide is called tidal flushing (Ketchum, 1951; Kuo & Neilson, 1988).

The model of transport by tidal flushing is based on the division of the prototype water body into segments, each of which is considered to be completely mixed at high tide. The length of each segment is defined by the tidal excursion, the average distance a water particle travels on the flood tide, because it is the maximum length over which complete mixing can be achieved. The present model simulates the physical transport using the concept of tidal flushing for the main channel, the primary branches (those from the main channel) and the secondary branches (those from the primary branches). The model treats the secondary branches as storage areas, which exchange the water masses with the primary branches as tide rises and falls. Modeling of the shallow shoaling regions in estuaries and tidal rivers by treating them as storage areas is described in Kuo & Park (1995b).

2. Kinetic Processes

The tidal prism eutrophication model in Kuo & Park (1995a) has twenty-four state variables in the water column (Table 1), and Fig. 1 illustrates the kinetic processes included in the water column of the model. The kinetic processes included in the model for each of the three algal groups are growth, basal metabolism (respiration and excretion), predation, and settling. For dissolved oxygen (DO), the model simulates the effects of algal photosynthesis and respiration, nitrification, heterotrophic respiration, oxidation of chemical oxygen demand (COD), surface reaeration, and sediment oxygen demand (SOD). The model simulates cycling of carbon, phosphorus, nitrogen and silica. Each of the organic carbon, phosphorus and nitrogen is divided into particulate and dissolved components, and the particulate component is further divided into refractory and labile fractions depending on the time scales of decay. The model does not simulate inorganic carbon. The model has one state variable for inorganic phosphorus, total phosphate (PO4t), which consists of dissolved (PO4d) and particulate (PO4p) fractions. The model has two state variables for inorganic nitrogen, ammonium (NH4) and nitrite+nitrate (NO23) nitrogen.

Table 1. Model state variables in the water column

1) salinity (S)	2) temperature (T)
3~5) three groups of algae (B_c, B_d and B_g)	6) refractory particulate organic C (RPOC)
7) labile particulate organic C (LPOC)	8) dissolved organic C (DOC)
9) refractory particulate organic P (RPOP)	10) labile particulate organic P (LPOP)
11) dissolved organic P (DOP)	12) total phosphate (PO4t)
13) refractory particulate organic N (RPON)	14) labile particulate organic N (LPON)
15) dissolved organic N (DON)	16) ammonium N (NH4)
17) nitrite+nitrate N (NO23)	18) particulate biogenic silica (SU)
19) available silica (SA)	20) dissolved oxygen (DO)
21) chemical oxygen demand (COD)	22) total suspended solid (TSS)
23) total active metal* (TAM)	24) fecal coliform bacteria (FCB)

* Total active metal may not be modeled by using total suspended solid as sorption site for total phosphate and available silica.

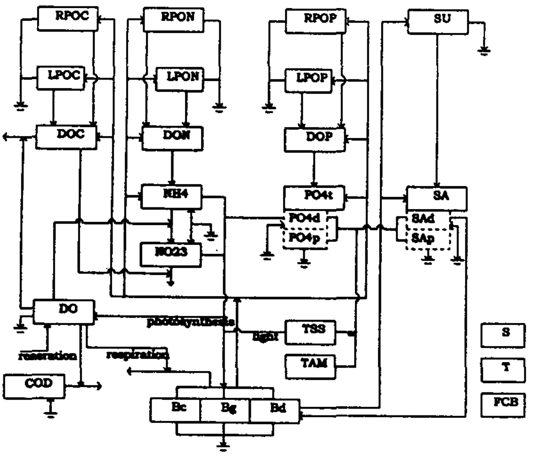

Fig. 1. Kinetic interactions among the twenty-four water column state variables (see Table 1 for the full non-abbreviated names of model state variables).

The model has two state variables for silica cycle, particulate biogenic silica and available silica (SA). The model simulates the sorption-desorption of PO4t and SA using the equilibrium partitioning with the sorption site quantified with either total suspended solid (TSS) or total active metal (TAM). The kinetic formulations used in the Chesapeake Bay three-dimensional water quality model (Cerco & Cole, 1994a and 1994b) are modified and used in the present model.

The sediment process model developed by DiToro & Fitzpatrick (1993) was used for modeling of the Chesapeake Bay mainstem and major tributaries. This sediment process model is slightly modified and incorporated into the present model. The model has twenty-seven state variables in the sediment. The sediment process model, upon receiving the particulate organic matter deposited from the overlying water column, simulates their diagenesis and the resulting fluxes of inorganic substances and SOD back to the water column. Since the sediment process model is not activated for the model application to be presented in this paper, which will be discussed later, further description of the sediment process model will not be included in this paper.

3. Solution Scheme

The present model uses an innovative solution scheme, which involves decoupling of physical transport processes and kinetic processes in the governing mass-balance equations (Eq. 1). The decoupled equations are solved by employing multi-step computation, in which the kinetic processes are applied for the first half tidal cycle, followed by the physical transport for a full tidal cycle, and the kinetic processes are applied again for the remaining second half tidal cycle. To avoid a truncation error, which is inherent to the finite difference solution of a differential equation, the kinetic equations are linearized, mostly for the Monod type expressions, and then solved analytically. The solution scheme, which results in a simple, efficient and yet accurate computational procedure, is described in Park & Kuo (1996b).

Model Calibration

The tidal prism eutrophication model is applied to the Lynnhaven Bay, U.S.A (Fig. 2). Since 1975, the DEQ (Department of Environmental Quality, Virginia) has been monitoring bimonthly the water quality conditions in the Lynnhaven Bay. The locations of the nine sampling stations (L1 to L7, LBC and TC) are shown in Fig. 2. To produce a more complete data set for the model calibration, VIMS (Virginia Institute of Marine Science) conducted supplementary field surveys at the same

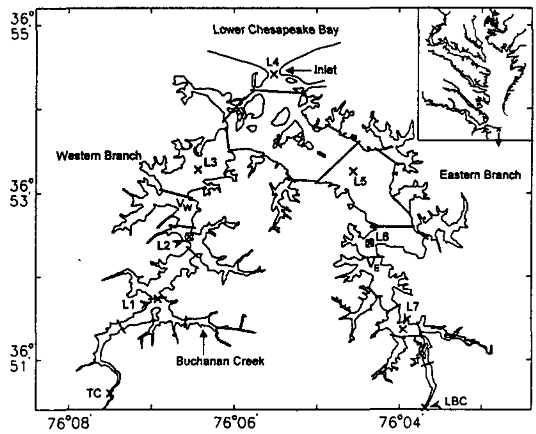

Fig. 2. The Lynnhaven Bay showing the model transects (solid lines) and the station locations (× = 1994 longitudinal surveys, □ = 1994 intensive survey, and V_E and V_W are the station locations for the 1980 verification results presented in Figures 5 and 6, respectively). The insert shows lower Chesapeake Bay and the location of the Lynnhaven Bay.

nine stations as the DEQ monitoring program. The 1994 VIMS surveys included four longitudinal surveys and one 25-hour intensive survey in June-December 1994. The data set consisting of the 1994 VIMS survey data and the DEQ monitoring data in 1994, which are presented in detail in Park et al. (1995a), is used for the model calibration. The water quality parameters, measured in 1994 VIMS surveys and the DEQ monitoring program, and used for the model calibration, are listed in Table 2. The calibration run simulates the period from June 21 to December 7, 1994.

Since only the chlorophyll measurements are available to quantify the total algal biomass, only one algal type in the model is simulated to represent the total algal biomass. Because diatoms are not explicitly modeled, the silica cycle in the model is not activated. TSS is simulated to quantify the sorption site for phosphate, and thus TAM is not modeled. The simulation period of six months is short relative to the time scales of the sediment processes, which makes the initial conditions quite important in sediment modeling. The optimum results of running the sediment process model with the quantity and quality of the available sediment data would be to calibrate the model such that it generates the sediment fluxes required to reproduce the observed parameters in the water column. The same stage is attained by specifying the required benthic fluxes, which are obtained from the previous works at the Lynnhaven Bay (Ho et al, 1977; Kuo et al., 1982) and through the model calibration.

Table 2. Water quality parameters measured at the nine stations in the 1994 VIMS surveys (Fig. 2) and used for the model calibration

salinity	temperature
secchi disk depth	
chlorophyll-a/phaeophytin	
total particulate carbon	dissolved organic carbon
total particulate phosphorus	total dissolved phosphorus
particulate (sorbed) inorganic phosphorus	dissolved phosphate
total particulate nitrogen	total dissolved nitrogen
ammonium nitrogen	nitrite+nitrate nitrogen
dissolved oxygen	chemical oxygen demand
total suspendid solid	
fecal coliform bacteria	

1. Preparation of Input Data for Calibration

The entire modeling domain is divided into twenty-three segments. The method of segmentation of a water body and the determination of the segment lengths are described in Kuo & Park (1994). The geometric data (high tide volume, tidal prism and mean depth at each segment) from Kuo et al. (1982) are used for the present model application. A sinusoidal curve that is fitted to approximate the observations is used to specify the spatially uniform temperature as a function of time.

The present model application simulates the sorption-desorption of PO4t using the equilibrium partitioning with the sorption site quantified with TSS:

$$PO4p=\frac{K_{PO4p}\cdot TSS}{1+K_{PO4p}\cdot TSS}PO4t \text{ and } PO4d=PO4t-PO4p \quad (2)$$

where K_{PO4p} is an empirical sorption coefficient relating phosphate sorption to TSS concentration. Sorption coefficient is calculated using Eq. 2 with the measured TSS, PO4d and PO4p from the 1994 VIMS surveys. The values range from 0.014 to 0.306 per g m^{-3} with the overall average of 0.082 per g m^{-3} and the standard deviation of 0.063 per g m^{-3}. A median value, $K_{PO4p}=0.066$ per g m^{-3}, is employed in the present study.

The present model requires specification of the ratios of nitrogen-to-carbon (ANC) and phosphorus-to-carbon (APC) in algae. The composition of particulate organic matter from the surface water may serve as an index of the algal stoichiometry, especially in shallow systems such as the Lynnhaven Bay. The 1994 VIMS survey data show that the ratios of particulate nitrogen-to-carbon as a function of dissolved inorganic nitrogen concentration (NH4+NO23) generally follow the constant Redfield algal nitrogen stoichiometry, ANC=0.167 g N per g C, which is used in the present study. Using the observations from the surface water in Chesapeake Bay, Cerco & Cole (1994b) showed that the ratios of particulate phosphorus-to-carbon decrease with decreasing PO4d, which may be approximated by an empirical formulation:

$$APC=(CP_{prm1}+CP_{prm2}\cdot\exp[-CP_{prm3}\cdot PO4d])^{-1} \quad (3)$$

where the three empirical constants, CP_{prm1}, CP_{prm2} and CP_{prm3}, may be obtained by fitting the measured APC (the ratios of particulate phosphorus-to-carbon) against the measured PO4d. The 1994 VIMS survey data are used to obtain $CP_{prm1}=41.1$ g C per g P, $CP_{prm2}=40$ g C per g P and $CP_{prm3}=200$ per g P m^{-3}, which are employed in the present study.

The present model requires, to simulate the algal growth, daily solar radiation intensity and fractional daylength. Hourly measurements of solar radiation intensity at VIMS (Gloucester Point, Virginia), located 30 km to the north of the Lynnhaven Bay, are used to estimate daily mean light intensity and fractional daylength. Algal growth in the present model also requires an input of light extinction coefficient. The total light extinction coefficient (Kess) is assumed to consist of three parts: background light extinction (Ke_b), light extinction due to suspended sediments (Ke_{TSS}) and that due to algae (Ke_{CHL}):

$$Kess = Ke_b + Ke_{TSS} \cdot TSS + Ke_{CHL} \cdot CHL \qquad (4)$$

where CHL is the chlorophyll-a concentration (mg CHL m^{-3}). The three empirical coefficients are determined through multiple regression of the observed data of Kess, TSS and CHL from the 1994 VIMS surveys: $Ke_b=0.735$ m^{-1}, $Ke_{TSS}=0.018\,m^{-1}$ per g m^{-3} and $Ke_{CHL}=0.06\,m^{-1}$ per mg CHL m^{-3}. The Kess used in multiple regression is calculated from the observed secchi disk depth (SD) using Kess=C/SD where SD is in meters and the constant C ranges from 1 to 2. The value of C=1.44, which has been observed in turbid coastal waters (Walker, 1982), is employed in the present study.

The data from the 1994 VIMS survey on 6/21 are used to estimate the initial conditions. Since not all the model state variables were measured from all model segments, approximations based on the previous studies (e.g., Cerco & Cole, 1994b) are used to estimate the initial conditions for each state variable at each model segment. The field data at the inlet (station L4 in Fig. 2) from the 1994 VIMS surveys on 6/21, 8/23, 10/4 and 12/7 are linearly interpolated to estimate the open boundary conditions. The present model is configured such that it does not require explicit specification of the upriver boundary conditions. Rather, the flux through the upriver boundary is defined to be zero, with the upriver contributions incorporated through nonpoint source discharges and loads.

Nonpoint source discharge rates and inputs of carbon, phosphorus and nitrogen are estimated using the outputs from the U.S. Army Corps of Engineers' STORM model (Abbott, 1977). Using the 1994 conditions (hourly rainfall data at the Norfolk Airport, located 5 km to the east of the Lynnhaven Bay, and the land use patterns for the drainage basin from the Hampton Roads Planning District Commission), the STORM model calculates quantity and quality of runoff for 1994. Since the STORM model does not output for all the state variables for the carbon, phosphorus and nitrogen cycles at all model segments, approximations based on the previous studies (e.g., Ho et al., 1977; Kuo et al., 1982; Cerco & Cole, 1994b) are used to estimate the nonpoint source discharges and loads for each state variable at each model segment. The DO concentrations in nonpoint source discharges are taken to be 80% of the saturated values, which are calculated using the temperature data from the two most upriver stations in the DEQ monitoring program (stations LBC and TC in Fig. 2). Finally, it is assumed that there is no nonpoint source input of salinity, algae, and COD. There was no point source input into the Lynnhaven Bay in 1994.

2. Calibration Results

The present model has one calibration parameter for the physical transport processes, the returning ratio α representing the fraction of water volume that leaves a segment at falling tide and returns from the adjoining downriver segment at the following rising tide (Kuo & Park, 1994). Since salinity is solely the result of physical transport processes, the returning ratio is calibrated using salinity data, and a value of 0.3 is obtained. The next step in calibration is to simulate the concentration fields of nonconservative state variables. The calibration procedure is started with the model run using the parameter values determined from the Chesapeake Bay water quality modeling study (Cerco & Cole, 1994b). The parameter values are then adjusted until the model results agree, to some satisfactory degree, with the observations. In this trial and error calibration, the first target usually is to reproduce the observed algal biomass, since it affects most of the other state variables (Fig. 1).

The results from the previous modeling studies in the Lynnhaven Bay (Ho et al., 1977; Kuo et al., 1982) serve as guidelines for the modification of some parameters from those used in the Chesapeake Bay modeling study. The calibration parameters related to the kinetic processes are in accordance with the results from the previous modeling studies in the Lynnhaven Bay (Ho et al., 1977; Kuo et al., 1982), and are within the ranges of the reported literature values. The values of all the calibration parameters can be found in Park et al. (1995b).

In the model calibration, the model results are compared with the observations from the nine stations (Fig. 2) in the 1994 surveys. Some examples are presented in Figures 3 and 4, which respectively show the time-series comparison at the station L6 in the Eastern Branch and at the station L2 in the Western Branch. Comparisons are presented for eight parameters: salinity, DO, CHL, TSS, total organic carbon (TOC=RPOC+LPOC+DOC+B where B=organic carbon in algal biomass), dissolved organic carbon (DOC), total phosphorus (TP=RPOP+LPOP+DOP+PO4t+APC · B) and total nitrogen (TN=RPON+LPON+DON+NH4+NO23+ANC · B).

Excellent agreement in salinity between the model results and the field data in Figures 3 and 4 reflects the credibility of the physical transport in the model. In Figures 3 and 4, TOC contains the organic fraction (both particulate and dissolved) and the fraction in the algal biomass, and TP (or TN) contains the organic fraction (both particulate and dissolved), inorganic fraction, and the fraction in the algal biomass. A model's capability of reproducing the total matter, such as TOC, TP and TN, is a crucial measure of its credibility. Good reproduction of the amount of the total matter in a model reflects that the model has reasonable external loads including point and nonpoint source inputs, and benthic fluxes. Only when a model contains the same amount of the total matter present in the prototype, the model is able to try to reproduce each of state variables that comprise the total matter. The present model overall gives a good reproduction of the field data (Figures 3 and 4), which is partly owing to the data collected from the 1994 VIMS surveys. These surveys, specifically designed for the model calibration, allow us to estimate several important

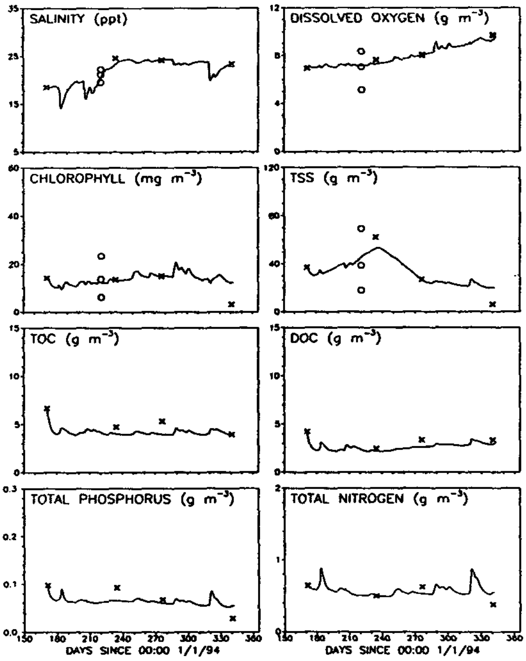

Fig. 3. Time-series comparison of the calibration results at the station L6 in the Eastern Branch (Fig. 2): solid line=model results, × =longitudinal survey data and ○=daily mean and range from intensive survey.

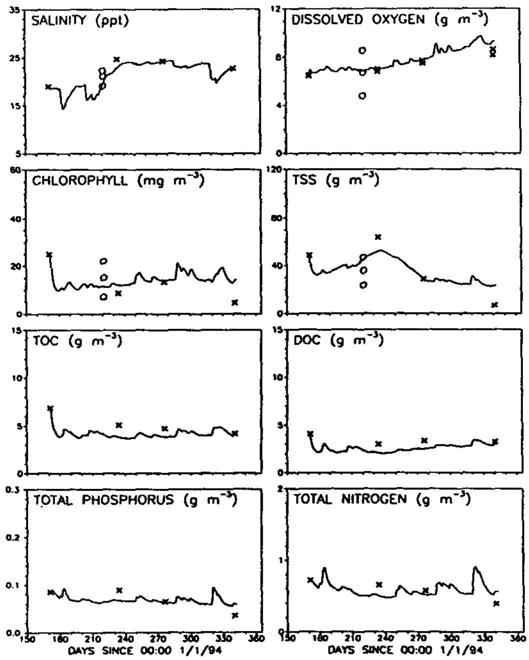

Fig. 4. Time-series comparison of the calibration results at the station L2 in the Western Branch (Fig. 2): solid line=model results, × =longitudinal survey data and ○=daily mean and range from intensive survey.

water quality parameters such as algal stoichiometry, sorption coefficient, light extinction coefficients, etc. The estimation of these important parameters from the observations makes the model calibration and verification far less subjective.

Model Verification

After calibration, an independent data set is required to verify the model calibration. The tidal prism eutrophication model, described in Kuo & Park (1994), is evolved from the one in Kuo & Neilson (1988) that has eight water column state variables. For the application of the original version, a comprehensive series of field surveys was conducted by VIMS at the Lynnhaven Bay in April-October 1980. The 1980 VIMS surveys included nine slackwater surveys and one 26-hour intensive survey at the fourteen sampling stations. The data set consisting of the 1980 VIMS survey data and the DEQ monitoring data in 1980, which are presented in Park et al. (1995a), is used for the model verification. The water quality parameters, measured in the 1980 VIMS surveys and used for the model verification, are listed in Table 3. The verification run simulates the period from April 28 to October 9, 1980.

As in calibration, only one algal component in the model is simulated to represent the total algal biomass, and thus the silica cycle is not activated. TSS is simulated to quantify the sorption site for phosphate, and thus TAM is not modeled. The sediment process model is not activated. Rather, the sediment fluxes determined in the calibration run are employed. All input parameters, which are used in the calibration run, are employed in the verification run, except those that may vary temporally. These parameters include water temperature, solar radiation intensity, fractional daylength, initial conditions, boundary conditions, and point and nonpoint source inputs.

Table 3. Water quality parameters measured at the fourteen stations in the 1980 VIMS surveys and used for the model verification

salinity	temperature
secchi disk depth	
chlorophyll-a/phaeophytin	
carbonaceous biochemical oxygen demand (5-day and ultimate)	
total phosphorus	dissolved phosphate
total Kjeldahl nitrogen	ammonium nitrogen
nitrite + nitrate nitrogen	
dissolved oxygen	
fecal coliform bacteria	

1. Preparation of Input Data for Verification

A sinusoidal curve that is fitted to approximate the observations is used to specify the spatially uniform temperature as a function of time. Hourly measurements of solar radiation intensity at the Norfolk Airport, are used to estimate daily mean light intensity and fractional daylength. The data from the 1980 VIMS slackwater survey on 4/28 are used to estimate the initial conditions. The same approximations employed in the calibration are used to estimate the initial conditions for each state variable at each model segment. The field data at the inlet from the 1980 VIMS surveys on 4/28, 5/27, 6/11, 7/8, 8/12, 9/8, 9/26~27 (intensive survey), 9/29, 10/1 and 10/7 are linearly interpolated to estimate the open boundary conditions.

As in calibration, nonpoint source inputs are estimated using the STORM model outputs for 1980. Using the 1980 rainfall data and the land use patterns for the drainage basin, the STORM model calculates quantity and quality of runoff for 1980. The same approximations employed in the calibration are used to estimate the initial conditions for each state variable at each model segment. The DO concentrations in nonpoint source discharges are taken to be 80% of the saturated values, which are calculated using the temperature data from the two most upriver stations in the DEQ monitoring program (stations LBC and TC in Fig. 2). There was a point source input into the Lynnhaven Bay in 1980 from the Birchwood Gardens sewage treatment plant (STP), which no longer exists. Its outfall was located at the Buchanan Creek, a small tributary to the Western Branch (Fig. 2). The characteristics of the point source inputs from this STP were determined with extensive measurements and used for the modeling study in Kuo et al. (1982). These characteristics are used to specify

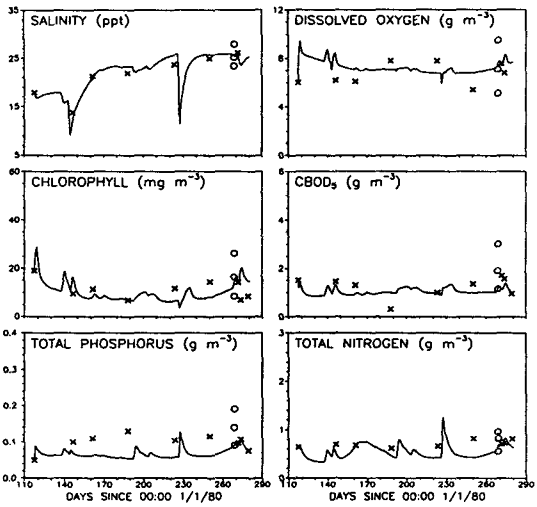

Fig. 5. Time-series comparison of the verification results at the station V_E in the Eastern Branch (Fig. 2): solid line=model results, × =slackwater survey data and ○=daily mean and range from intensive survey.

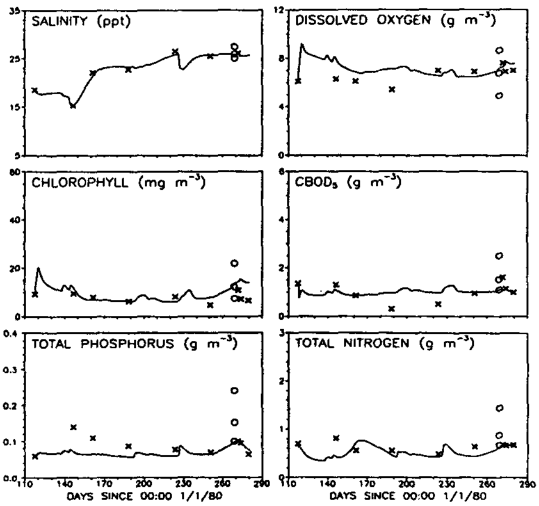

Fig. 6. Time-series comparison of the verification results at the station V_W in the Western Branch (Fig. 2): solid line=model results, × =slackwater survey data and ○=daily mean and range from intensive survey.

temporally constant point source inputs for the 1980 verification run.

2. Verification Results

In the model verification, the model results are compared with the observations from the fourteen stations in the 1980 surveys. Some examples are presented in Figures 5 and 6, which respectively show the time-series comparison at the station V_E in the Eastern Branch and at the station V_W in the Western Branch. Comparisons are presented for six parameters: salinity, DO, CHL, 5-day carbonaceous biochemical oxygen demand ($CBOD_5$), TP and TN. The $CBOD_5$ in the present model may be estimated from heterotrophic respiration of DOC, COD and algal respiration (Park et al., 1995b).

Excellent agreement exists in salinity between the model results and the field data, again confirming the credibility of the physical transport in the model. The model overall gives a reasonable reproduction of the field observations. The model however slightly underestimates PO4d, which results in the slight underestimation of TP in Figures 5 and 6. The channel dredging, which has been conducted many times between 1980 and 1994 for navigational purpose, would have changed the characteristics of the bottom sediment considerably. Hence, the benthic flux of PO4d may not be the same in 1980 and 1994. Although the verification results in Figures 5 and 6 are not as good as the calibration results in Figures 3 and 4, the verification results appear quite satisfactory and thus validate the model calibration.

Summary

Figures 3 to 6 provide a qualitative comparison between the model results and the field data in the model calibration and verification. Quantitative assessments of model accuracy are desirable to render the evaluation of models less subjective. Quantitative assessments for the model calibration and verification are attempted with the three quantitative measures of errors, which are listed in Table 4 for salinity, DO, CHL, TSS, TOC, $CBOD_5$, TP and TN. The mean error (ME), which indicates whether the model overestimates or underestimates the data on the average, is defined as:

$$ME = \frac{1}{N} \sum_{i=1}^{N} (P_i - O_i) \qquad (5)$$

where P_i and O_i are the corresponding model result

Table 4. Statistical summary of model calibration and verification: the mean error (ME), the mean absolute error (MAE), the relative error (RE) and the number of observations (N).

Variables	ME	MAE	RE (%)	N
salinity	0.146	1.08	4.8	90
DO	0.412	0.809	11.4	109
chlorophyll-a	− 0.971	5.32	40.2	112
TSS	−20.2	25.7	41.1	44
TOC	− 1.75	1.75	35.5	18
$CBOD_5$	− 0.495	0.848	53.0	92
TP	− 0.029	0.045	37.0	111
TN	− 0.091	0.186	25.9	111

and observation, respectively and N is the number of observations. The mean absolute error (MAE), defined as:

$$MAE = \frac{1}{N} \sum |P_i - O_i| \tag{6}$$

is a measure of the absolute deviation of the model results from the data on the average, with zero MAE being ideal. The relative error (RE) is defined as the ratio of the MAE to the mean of the data:

$$RE = \frac{\sum |P_i - O_i|}{\sum |O_i|} \tag{7}$$

and indicates the magnitude of the MAE relative to the data on the average.

This paper presents the procedure and the results of calibration and verification of the tidal prism eutrophication model described in Kuo & Park (1994) using the field data presented in Park et al. (1995a). The results of the model application are briefly presented in Park & Kuo (1996a), and a more complete description of the model calibration and verification is given in Park et al. (1995b). With so many state variables included, the present model requires vast amounts of field data for its application, as described in this paper. In the application of a water quality model, the model's capability to reproduce the total matter, such as total carbon, total phosphorus and total nitrogen, is a crucial measure of its credibility. Only when a model contains the same amount of the total matter present in the prototype, the model is able to reproduce each of state variables that comprise the total matter. In order to be able to reproduce the amount of the total matter accurately, the model should have reasonable external loads including point and nonpoint source inputs, and benthic fluxes.

The present model overall reproduces the observations very well, which is partly owing to the data collected from the 1994 VIMS surveys. These surveys were specifically designed to collect the data for the model application, which enables us to estimate several important water quality parameters from the field data, including algal stoichiometry, sorption coefficient, light extinction coefficients, etc. Reasonably calibrated and verified water quality models may serve as a powerful tool for water quality management.

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