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NUMERICAL SIMULATION OF SEDIMENT RELATED PROCESSES IN WATER QUALITY MODEL

Xiaobo Chao¹, Yafei Jia², F. Douglas Shields, Jr.³ and Charles M. Cooper⁴

ABSTRACT

Sediment is a major nonpoint-source pollutant, and the exchange of materials between water and sediment is an important component of the lake eutrophication process. Suspended sediment increases water surface reflectivity and light attenuation in the water column. Nutrients can be absorbed to sediment particles and desorb from sediment to the water. In addition, nutrients can also be released from bed sediment. In this study, a water quality model, CCHE3D_WQ, was applied to simulate the concentrations of phytoplankton and nutrients in a shallow, natural lake with special emphasis on sediment-related processes. A formula was generated from field measurements to calculate the light attenuation coefficient using the concentration of chlorophyll and suspended sediment. The concentrations of adsorbed and dissolved nutrients due to adsorption-desorption were calculated using two formulas derived based on the Langmuir Equation. The release rates of nutrients from the lake bed were calculated by considering the effects of the concentration gradient across the water-sediment interface, pH, temperature, dissolved oxygen concentration, and flow conditions. Model algorithms describing the adsorption and desorption of nutrients from sediment particles as well as their release from bed sediment were tested using laboratory experimental data. Model simulation results show that there are strong interactions between sediment-related processes and nutrient concentrations.

1. INTRODUCTION

It is well known that the sediment-related nonpoint source pollution is mainly caused by agricultural activities. This nonpoint source pollution causes serious impacts to natural aquatic ecosystems. Sediment has been identified as one of the leading nonpoint source pollutants in the United States. It is estimated that 60% of the 3 billion tons of sediment per year deposited in U.S. waterways originates from agricultural lands, and these sediments are associated with large amounts of contaminants such as nutrients and pesticides (Rebich and Knight, 2001).

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In many cases, sediment-related processes drive water quality. For example, suspended sediment (SS) reduces light intensity, which is one of the most important impact factors on the growth of phytoplankton. Nutrients and pesticides adsorb to sediment particles and also desorb from sediment to the water. In addition, nutrients adsorbed to deposited organic and inorganic sediments may be released back into the water column.

Studying water quality problems in a natural environment with numerical models is a very effective approach. As a result, many numerical models for simulating water quality processes have been developed and the studies of water quality problems with numerical models are increasing rapidly (Cercio and Cole 1995, Gin et al. 2001, Jin et al. 1998, Wool et al. 2001). Some well-established three dimensional models, such as WASP6 (Wool et al. 2001), CE-QUAL-ICM (Cercio and Cole 1995), Delft3D-WAQ (Delft Hydraulics 2003), MIKE3_WQ [Danish Hydraulic Institute (DHI) 2005], ECOM/RCA (Hydroqual 2004), etc., have been used to simulate water quality variables in river, lake, estuary and ocean environments. These models generally cover basic physical, chemical and biological processes of aquatic ecosystems. However, only a few are capable of simulating the effects of sediment on the water quality. The processes of adsorption-desorption and bed release of nutrients from sediment were considered in WASP6, CE-QUAL-ICM, and Delft3D-WAQ; however, the effect of SS concentration on the growth of phytoplankton was not simulated in these models. In those models, the release rate of nutrients from the bed sediment was determined using functions of the concentration gradient across the water-sediment interface; however, the effects of pH and dissolved oxygen concentration on the release rate were not considered. In WASP6 and CE-QUAL-ICM, the adsorption-desorption of nutrients by sediment were described using a simple linear isotherm. In view of the limited understanding of the sediment-associated water quality processes, the accuracy of the simulations using the aforementioned models has room for improvement.

Three major sediment-related processes were presented to consider the effects of sediment on the water quality processes, including: the effect of sediment on the light intensity for the growth of phytoplankton, the adsorption-desorption of nutrients by sediment and the release rate of nutrients from the bed sediment layer. A water quality model was developed based on these sediment-related processes and applied to a real-life problem to validate its capability to simulate the concentrations of phytoplankton and nutrients in Deep Hollow Lake, a small oxbow lake in Mississippi Alluvial Plain.

2. SEDIMENT-RELATED PROCESSES IN WATER QUALITY MODEL

2.1 The Effect of Sediment on the Growth of Phytoplankton

2.1.1 The growth of phytoplankton

Phytoplankton, the group of planktonic plants that float in waters, plays a central role in the carbon and nutrient cycles that comprise the model ecosystem. The conceptual framework for phytoplankton kinetics is mainly based on the models WASP6 (Wool et al. 2001) and CE-QUAL-ICM (Cercio and Cole 1995). Total chlorophyll is used as a simple measure of phytoplankton biomass. The effective source term for phytoplankton can be calculated by

$$\sum S_{chl} = (G_p - D_p - P_{set})C_{chl} \quad (1)$$

in which $\sum S_{chl}$ is the effective source term for phytoplankton; G_p is the growth rate of phytoplankton (day^{-1}); D_p is the death rate of phytoplankton (day^{-1}); P_{set} is the effective phytoplankton settling rate (day^{-1}); and C_{chl} is the concentration of chlorophyll (mg/l).

The growth rate of phytoplankton is determined by the availability of nutrients, the intensity of light, and the ambient temperature. The effects of each factor are considered to be multiplicative:

$$G_p = P_{mx} f_N f_I f_T \quad (2)$$

in which P_{mx} is the maximum phytoplankton growth rate (day^{-1}); and f_N , f_I and f_T are the limitations due to nutrient availability, light intensity, and temperature, respectively.

The light limitation factor f_I is obtained by integrating the Steele equation over depth and time (Chapra 1997):

$$f_I = \frac{2.72 f_d}{K_e \Delta z} \left[\exp\left(-\frac{I_0}{I_m} e^{-K_e(zd+\Delta z)}\right) - \exp\left(-\frac{I_0}{I_m} e^{K_e \cdot zd}\right) \right] \quad (3)$$

in which f_d is the fractional daylight; Δz is the model segment (spatial element) thickness (m); zd is the distance from the water surface to the top level of a computational element in the water (m); I_0 is the daily light intensity at the water surface (ly/day); I_m is the saturation light intensity of phytoplankton (ly/day). K_e is the total light attenuation coefficient, and it is determined by the effects of water, chlorophyll and SS, and can be expressed by (Stefan et al. 1983):

$$K_e = K_0 + K_{chl} + K_{ss} \quad (4)$$

where K_0 is the background light attenuation (m^{-1}); K_{chl} is the attenuation by chlorophyll (m^{-1}); K_{ss} is the attenuation by SS (m^{-1}). Eq. (13) below provides an empirical formula to calculate K_e for our example case site based on C_{chl} and SS concentration s (mg/l).

The light attenuation coefficient K_e in Eq. (4) is an important parameter for phytoplankton growth. Wool et al. (2001) proposed a formula for K_e based on the sum of the background light attenuation, K_0 , and the phytoplankton self-shading attenuation, but ignoring the attenuation by SS:

$$K_e = K_0 + 0.0088 C_{chl} + 0.054 C_{chl}^{0.67} \quad (5)$$

In fact, the SS increases both the surface reflectivity and light attenuation in the water. A modified form of Eq. (4) was developed using work by Stefan et al. (1983) that showed attenuation by SS can be given by:

$$K_{ss} = \gamma s \quad (6)$$

where s is the concentration of SS; γ is a coefficient. Coupling Eqs. (5) and (6), Eq. (4) can be written as:

$$K_e = K_0 + 0.0088 C_{chl} + 0.054 C_{chl}^{0.67} + \gamma s \quad (7)$$

γ can be obtained based on field measurements.

2.1.2 The effect of sediment on the light attenuation coefficient

It has been observed that in the water column, the extinction of light is proportional to the light at any water depth:

$$\frac{dI}{dz} = -K_e I \quad (8)$$

in which I is the light intensity; z is the water elevation.

At the water surface, $z = 0$ and $I = I_0$, so

$$I = I_0 e^{(-K_e z)} \quad (9)$$

and

$$\frac{I}{I_0} = e^{(-K_e z)} \quad \text{or} \quad \ln\left(\frac{I}{I_0}\right) = -K_e z \quad (10)$$

Based on the measured I , I_0 and z , K_e can be obtained by data fitting.

To determine the parameters γ in Eq (7) and the measured light attenuation coefficient, about 20 sets of field measurements were conducted in Deep Hollow Lake to measure the light intensity, and concentrations of SS and chlorophyll.

Light intensities were measured using a LICOR LI-250 light meter and a spherical quantum radiation sensor which measures photon flux from all directions underwater. This measurement was called Photosynthetic Photon Flux Fluence Rate (PPFFR). Units are in micromol per second per meter squared per microamp. Data was collected at Deep Hollow Lake approximately every two weeks (weather permitting). Light radiation was measured during conditions of unobstructed sunlight (no clouds) at three sites (DH1, DH2, and DH3 stations in Figure 8) between the hours of 10:00 AM and 2:00 PM. At each station light intensities were measured in air, at water surface and approximately every 10 cm until the unit touched the lake bottom. In addition, the concentrations of SS, chlorophyll were also measured.

Fig. 1 shows the measured light intensity under the water. It can be observed light intensity decrease quickly with depth in the water column due to the effects of water, sediment and phytoplankton. The light can penetrate about 1/3 of the total water depth. Fig. 2 shows the relations between $\ln(I/I_0)$ and water elevation z . It is obviously there is a linear relations between $\ln(I/I_0)$ and z . The slope of the line is the measured light attenuation coefficient.

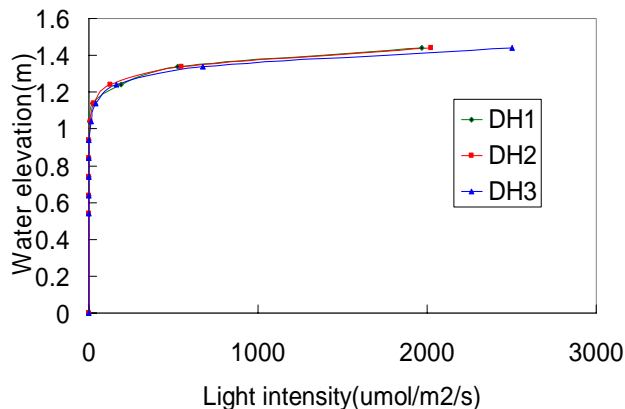


Fig. 1. Measured light intensity in water

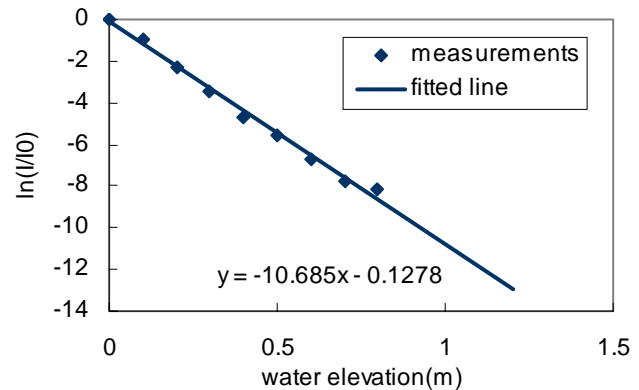


Fig. 2. $\ln(I/I_0)$ versus water elevation z

The Eq. (7) can also be expressed as

$$K_e - (K_0 + 0.0088C_{chl} + 0.054C_{chl}^{0.67}) = \gamma s \quad (11)$$

set $f(K_e) = K_e - (K_0 + 0.0088C_{chl} + 0.054C_{chl}^{0.67})$, so

$$f(K_e) = K_e - (K_0 + 0.0088C_{chl} + 0.054C_{chl}^{0.67}) = \gamma s \quad (12)$$

In Deep Hollow Lake K_0 is taken as 1.2. Based on about 20 sets of measured data, including light attenuation coefficient, concentrations of SS and chlorophyll, the relationship between $f(K_e)$ and s was obtained. As shown is Fig. 3, a regression line was generated from these data with an r^2 value of 0.86. The slope of the regression line, γ , was equal to 0.0452 which was close to the value of 0.043 obtained by Stefan et al. (1983) for a nearby lake. So Eq.(7) can be expressed as:

$$K_e = 1.2 + 0.0088C_{chl} + 0.054C_{chl}^{0.67} + 0.0452s \quad (13)$$

Eq.(13) was adopted to calculate to light attenuation coefficient in Deep Hollow Lake.

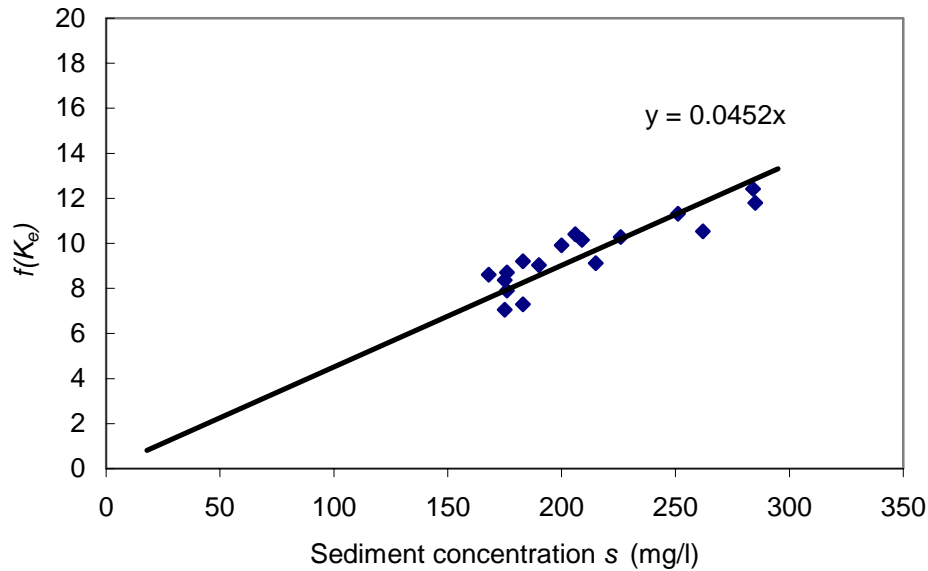


Fig. 3. Relationship between $f(k_e)$ and sediment concentration s

2.2 Processes of Adsorption-desorption of Nutrients by Sediment

2.2.1 Mathematical equations

There are adsorption-desorption interactions between dissolved nutrients and SS in the water column. Adsorption is a process in which dissolved nutrients become associated with SS particles. Desorption is the opposite process and thus refers to the release of adsorbed nutrients from SS.

Because the reaction rates for adsorption-desorption are much faster than that for the biological kinetics, an equilibrium assumption can be made (Wool et al. 2001). It is assumed that the

interaction of the dissolved and particulate phases reach equilibrium instantaneously in response to nutrient and sediment inputs so as to redistribute nutrients between dissolved and solid-phase compartments. Thus, the processes of adsorption-desorption are assumed to reach equilibrium at each time step of the numerical simulation.

In some models, the adsorption-desorption of nutrients by sediment is described by a linear isotherm, and the ratio of particulate and dissolved nutrient concentration is assumed as a constant (Ishikawa and Nishimura 1989; Wool et al. 2001). However, most experimental results show the Langmuir equilibrium isotherm is a better representation of the relations between the dissolved and particulate nutrient concentrations (Fox et al. 1989; Appan and Wang 2000; Bubba et al. 2003). In this study, the Langmuir equation was adopted to calculate the adsorption and desorption rate. The equilibrium adsorption content (Q) can be expressed as:

$$Q = \frac{Q_m KC_d}{1 + KC_d} \quad (14)$$

where C_d is the dissolved nutrient concentration after the adsorption reaches equilibrium; Q_m is the maximum adsorption capacity; and K is the ratio of adsorption and desorption rate coefficients. It is assumed that the volume of the nutrient/water/sediment mixture solution is V_0 , which is a constant before and after adsorption. C_0 is the initial nutrient concentration at each time step in the solution, and s is the sediment concentration. Since the initial amount of nutrient in the solution is same as when the adsorption reaches equilibrium, it can be expressed as:

$$C_0V_0 = C_dV_0 + sV_0Q \quad \text{or} \quad C_d = C_0 - sQ \quad (15)$$

so

$$C_d = C_0 - C_p \quad (16)$$

where the particulate concentration C_p can be written as

$$C_p = sQ \quad (17)$$

By substituting Eqs.(15) and (16) into Eq.(14) and simplifying, it can be shown that:

$$C_p = \frac{1}{2} \left[\left(C_0 + \frac{1}{K} + sQ_m \right) - \sqrt{\left(C_0 + \frac{1}{K} - sQ_m \right)^2 + \frac{4sQ_m}{K}} \right] \quad (18)$$

$$C_d = \frac{1}{2} \left[\left(C_0 - \frac{1}{K} - sQ_m \right) + \sqrt{\left(\frac{1}{K} + C_0 - sQ_m \right)^2 + \frac{4sQ_m}{K}} \right] \quad (19)$$

Eqs. (18) and (19) are used to calculate the concentration of particulate and dissolved nutrients due to adsorption-desorption. These concentrations are determined by the initial concentration of nutrients C_0 ; the adsorption constants K and Q_m ; and the SS concentration s . It can be seen that the ratio of C_p and C_d is not a constant.

2.2.2 Comparison with experimental case

A laboratory experiment was designed to study the adsorption capacities of ortho-phosphorus by 13 Danish sands (Bubba et al. 2003). The experimental results show that the concentration of ortho-phosphorus in the liquid phase and the adsorption content can be described using Langmuir equilibrium equation. The maximum adsorption capacity Q_m and the Langmuir adsorption constant K were obtained from these results. Data was also presented showing the concentrations of particulate and dissolved phosphate under different initial concentrations at equilibrium adsorption.

The concentrations of dissolved and particulate ortho-phosphorus due to adsorption-desorption can be obtained using Eqs. (18) and (19) based on the total initial concentration C_0 , maximum adsorption capacity Q_m , Langmuir adsorption constant K , and SS concentration s . Based on the experimental conditions, Eqs.(18) and (19) were used to calculate the concentrations of particulate and dissolved ortho-phosphorus at initial concentrations of 2.5, 5, 10, 20, and 40 mg/l. Two of the Bubba et al. (2003) data sets (Birkesig and Vestergard) were used for comparison. As shown in Fig.4 and Fig.5, the results obtained from Eqs. (18) and (19) are generally in good agreement with the measurements.

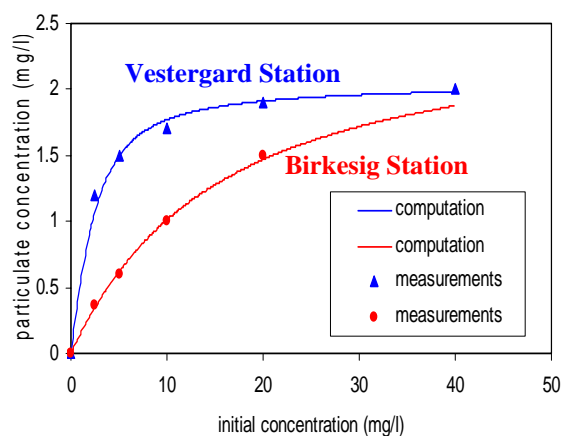


Fig. 4. The concentration of particulate phosphate versus different initial concentrations at equilibrium

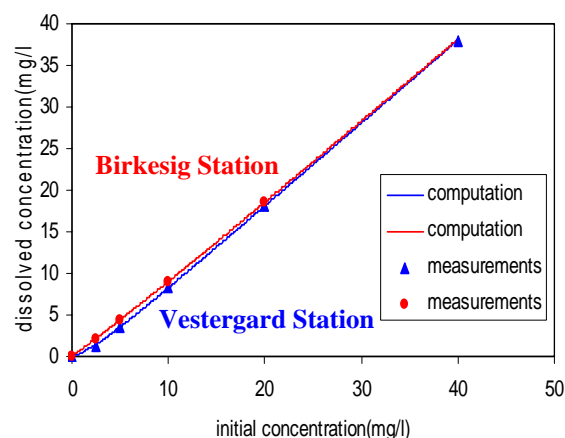


Fig. 5. The concentration of dissolved phosphate versus different initial concentrations at equilibrium

2.2.3 Comparison between the Langmuir equation and linear Assumption

To simulate the process of adsorption and desorption of phosphate by sediment, some numerical models are based on the “Linear Assumption”—they assume a constant ratio of the concentration of particulate to dissolved phosphorus (e.g., Ishikawa and Nishimura 1989; Wool et al. 2001). Some evidence suggests that the particulate fraction of phosphate (*particulate concentration / total concentration*) ranges from 0.01 to 0.33 (Wool et al. 2001), implying a range for the particulate to dissolved ratio between 0.01 and 0.5. However, experimental results such as those presented in experimental case 2.2.2 support the use of the Langmuir equilibrium equation. Fig. 6 shows the concentrations of dissolved and particulate phosphate at equilibrium for the experimental case. The Langmuir equation and Linear Assumption were also plotted for comparison. It can be observed that the Langmuir equation is generally in good agreement with the full range of the data, but the Linear Assumption appears to be valid only when the concentration of dissolved phosphate is less than 2.5 mg/l, producing large errors outside this range.

It is obvious that the Langmuir equation is more accurate to express the relations of phosphate in liquid and solid phase at equilibrium. The maximum adsorption capacity and the Langmuir adsorption constant can be obtained based on the experimental measurements. The Linear Assumption is simpler and the constant particulate fraction is only available for the low phosphate concentration. Since the particulate fraction keeps constant during the whole simulation period, it

may cause some errors for the various concentrations of SS. Due to the wide ranges of particulate fractions, some additional measurements have to be conducted to obtain the values of fractions.

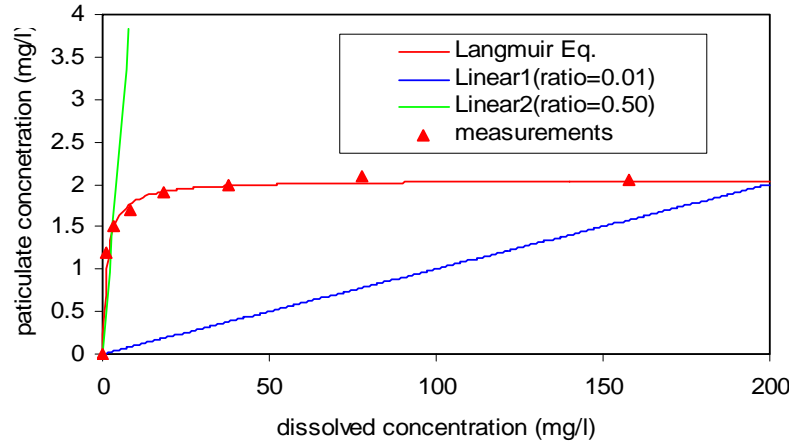


Fig. 6. Comparison of phosphate concentration obtained by Langmuir Equation and Linear Assumption

2.3 Release of Dissolved Nutrients from Bed

2.3.1 Mathematical equations

Bed release, or diffusion from the bed into the water column, is an important source of inorganic and organic nutrients. The release rate of nutrients from the bed sediment layer is affected not only by the concentration gradient across the water-sediment interface, it is also affected by pH, temperature and dissolved oxygen concentration. Based on Romero (2003), the bed release rate S_{diff} can be expressed as:

$$S_{diff} = \theta_{sed}^{T-20} S_c \left(\frac{K_{dos}}{K_{dos} + DO} + \frac{|pH - 7|}{K_{pHS} + |pH - 7|} \right) \quad (20)$$

where S_c is the diffusive flux of nutrients ($mg/m^2/day$); K_{dos} (mg/l) and K_{pHS} are the values that regulate the release of nutrient according to the dissolved oxygen (DO) and pH in the bottom layer of the water column of depth Δz_b (m); θ_{sed} is the temperature coefficient. The diffusive flux S_c can be calculated using Fick's first law which expresses that the flux is directly proportional to the concentration gradient and the porosity of sediment (Loeff et al. 1984; Moore et al. 1998):

$$S_c = -\phi D_m \frac{dC}{dz} \approx \frac{\phi D_m}{\Delta z_b} (C_b - C_w) = k(C_b - C_w) \quad (21)$$

where D_m is the molecular diffusivity (m^2/day); ϕ is the porosity of sediment; Δz_b is the diffusive sub-layer thickness near the bed (m); k is the diffusive exchange coefficient at water-sediment interface (m/day); C_w and C_b are the concentration of nutrients in water and water-sediment interface, respectively.

2.3.2 Comparison with experimental data

A laboratory experiment was conducted in the USGS Oregon District Laboratory (Fisher and Wood 2004) to study the release rate of phosphate from bed sediment with different pH levels in the overlying water. During the first four days, some sediment was added to remove the original phosphate in water column. Then the concentration of phosphate in water column was measured continuously for 50 days while phosphate was released from the bottom sediment. As shown in Fig. 7, there were three phases in this experiment with pH levels of 8.1, 9.2 and 10.3, in each phase, respectively. In general the concentration of phosphate in water column was increased due to the bed release. With the increase of water column concentration, bed release rate decreased. The measured release rates of phosphate were 1.4, 1.0, and 0.7 $\text{mg}/\text{m}^2/\text{day}$ for the three phases, and the bed release source terms S_{diff} were 0.4, 0.3 and 0.3 $\mu\text{g}/\text{l}/\text{day}$, respectively. Based on the measured S_{diff} , the concentration of phosphate in water column was simulated using the presented numerical model. As shown in Fig. 7, the numerical results are generally in good agreement with the measurements.

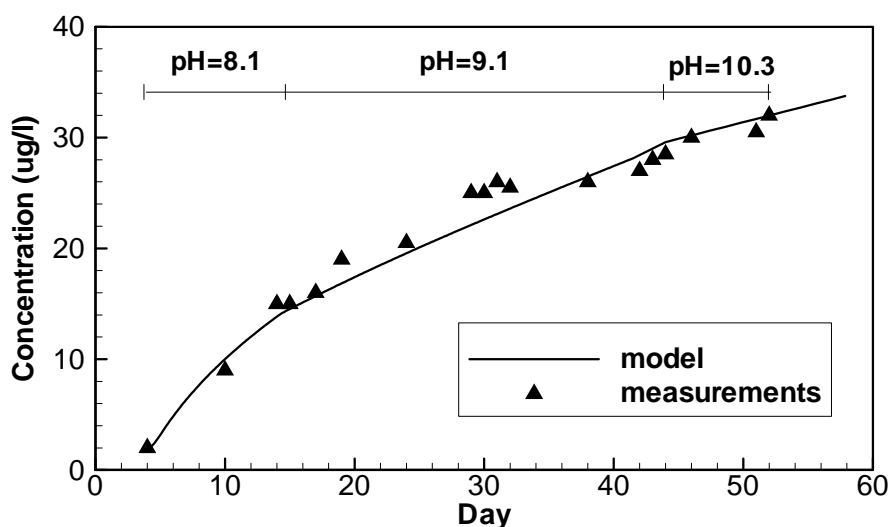


Fig. 7. Concentration of phosphate in water due to bed release

3. APPLICATION TO DEEP HOLLOW LAKE

A water quality model, CCHE3D_WQ, was applied to simulate the concentrations of phytoplankton and nutrients in Deep Hollow Lake with emphasis on sediment-related processes.

3.1 Study Area

The Mississippi River Alluvial Plain is one of the most intensively farmed agricultural areas of the United States. The quality of surface water resources in this area are particularly vulnerable due to excessive sediment, nutrients, and pesticides transported from agricultural fields. The Mississippi Delta Management System Evaluation Area (MDMSEA) project is part of a national program designed to evaluate the impact of agricultural production on water quality and to develop best management practices (BMPs) to minimize adverse effects. Deep Hollow Lake, a small oxbow lake located in Leflore County, Mississippi, was selected as a study site for MDMSEA.

Deep Hollow Lake receives runoff from a two square kilometer watershed which is heavily cultivated. Fig.8 shows the contours of bathymetry of Deep Hollow Lake. It has morphology of a typical old oxbow lake, with a length of about 1 km and a width of about 100 m. Lake water depth ranges from 0.5 m to 2.6 m, with greatest depth in the middle. Weekly or biweekly samples of SS, nutrients, chlorophyll, bacteria, and other selected water quality variables were collected at Stations DH1, DH2 and DH3. Two of the major inflows were located at the Stations UL1 and UL2 and monitored for water quality and quantity by the U.S. Geological Survey (Rebich and Knight, 2001). The inlet boundaries of water quality constituents were calculated based on those field measurements.

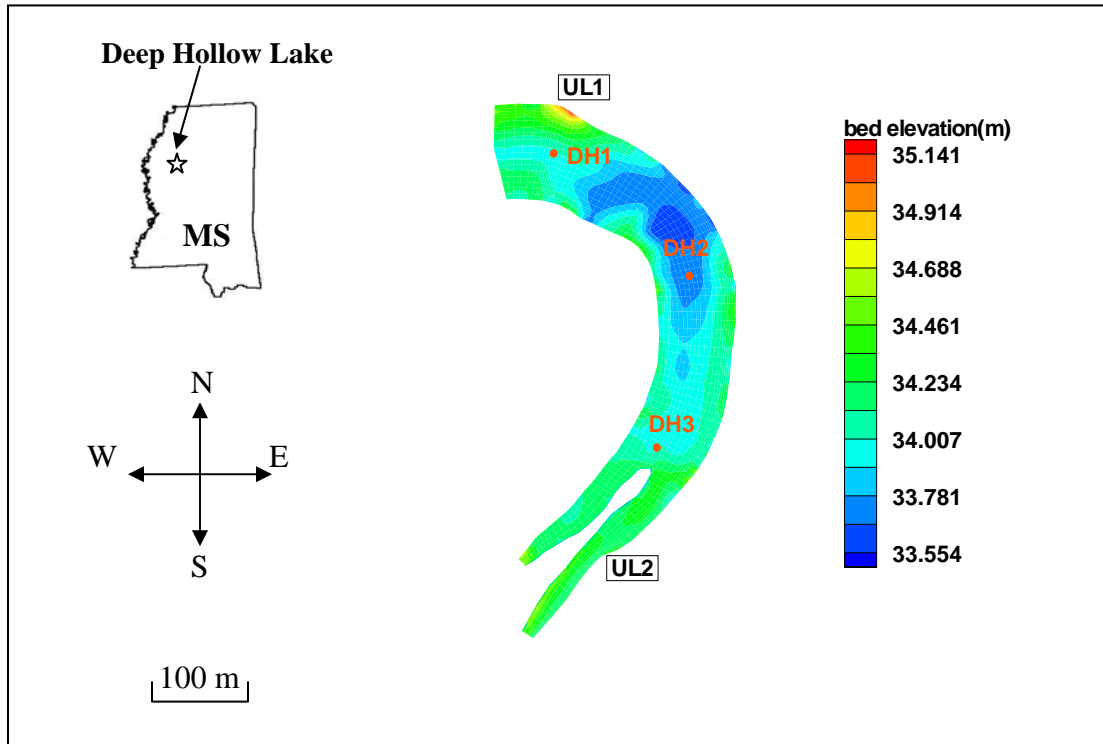


Fig. 8. Deep Hollow Lake

The nutrient levels in Deep Hollow Lake are mainly dependent on the fertilizer loadings in the surrounding farmland and the quantity of runoff. Field measurements also show that the concentrations of nitrate and ammonia in the lake are very low, while the concentration of phosphorus is relatively high in comparison with other areas of USA. SS concentrations are relatively high, exceeding published levels known to adversely impact fish growth and health (Rebich and Knight 2001). Field data from stations DH1, DH2 and DH3 were used for model calibration and validation. In general, the data show that water quality conditions at the three stations are similar. Only data from Station DH1 are presented here.

Based on bathymetric data, the computational domain was discretized into a structured finite element mesh. In the horizontal plane, the irregular computational domain was represented by a 95×20 mesh. In the vertical direction, the domain was divided into 8 levels with finer spacing near the bed.

3.2 Model Calibration and Validation

CCHE3D_WQ was calibrated using biweekly field data and analysis of lake water samples obtained between April to June, 1999. For calibration runs, the velocity field induced by wind and runoff were obtained from runs of CCHE3D. Water quality model parameters were adjusted repeatedly to obtain a reasonable reproduction of the field data. Some of the model parameters were obtained directly from special experiments and field measurements, as described below and by others (Portielje and Lijklema 1993; DiToro 2001; Wool et al. 2001; Hipsey et al. 2003; Bubba et al. 2003). All adopted parameter values are in the range reported in the literature (Wool et al. 2001, Cerco and Cole 1995).

The light attenuation coefficient K_e was calculated using Eq.(13) by considering the effects of water, chlorophyll and SS. The adsorption and desorption of ammonium nitrogen, nitrate and phosphate by sediment was simulated by the Langmuir equation. For Deep Hollow Lake, the concentrations of ammonium and nitrate were very low, so the adsorption and desorption of ammonium and nitrate from sediment were expected to be insignificant and were not incorporated. At equilibrium, the concentration of dissolved and particulate phosphate due to adsorption-desorption was calculated using Eqs. (18) and (19). Dissolved inorganic and organic nutrients may also be released to the water column from bed sediments. In the water quality model, the bed release rate of ammonium, nitrate, organic nitrogen, phosphate, and organic phosphorus were calculated using Eq. (20).

The model calibration results for Station DH1 are shown in Figs. 9 and 10. In general, the model provided reasonable reproduction of patterns and acceptable magnitudes for water quality constituents. The mean values of the model results are generally in good agreement with the field observations. However, the ability of the model to reproduce temporal variations in field measurements was not as good as the mean value reproduction. These differences may arise due to the fact that measurements occurred weekly while the time step for the simulation was 1 hour. In addition, the water quality processes in the lake system are likely more complicated than those reproduced in the simulation model.

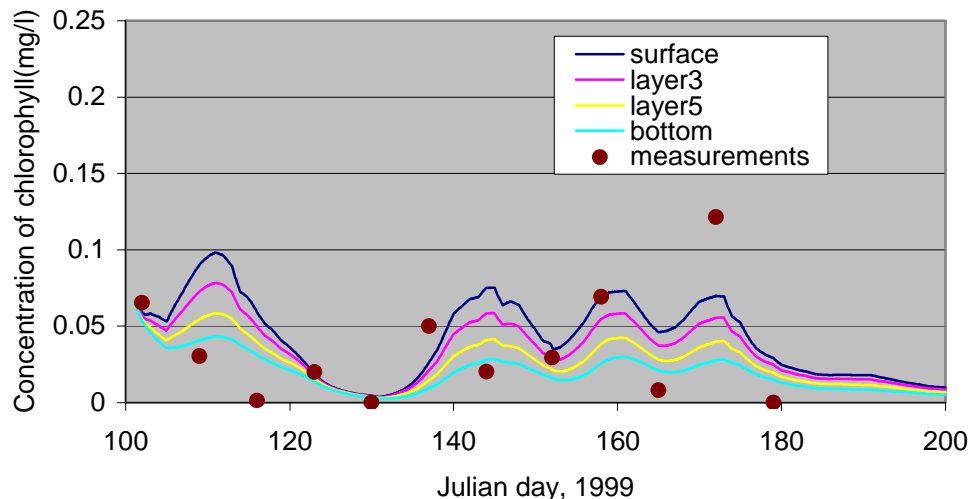


Fig. 9 The concentration of chlorophyll at Station DH1(calibration)

The period from September to December 1999 was chosen for model validation. Computed flow currents, and measured boundary conditions, weather data and SS concentrations were used to predict the concentrations of water quality constituents. Parameter values in the water quality model were those calibrated values using the data of April – June 1999. Fig. 11 shows the simulated and observed concentrations of chlorophyll. As for the calibration run, trends and quantities of

concentration of chlorophyll obtained from the numerical model were generally in agreement with the observations.

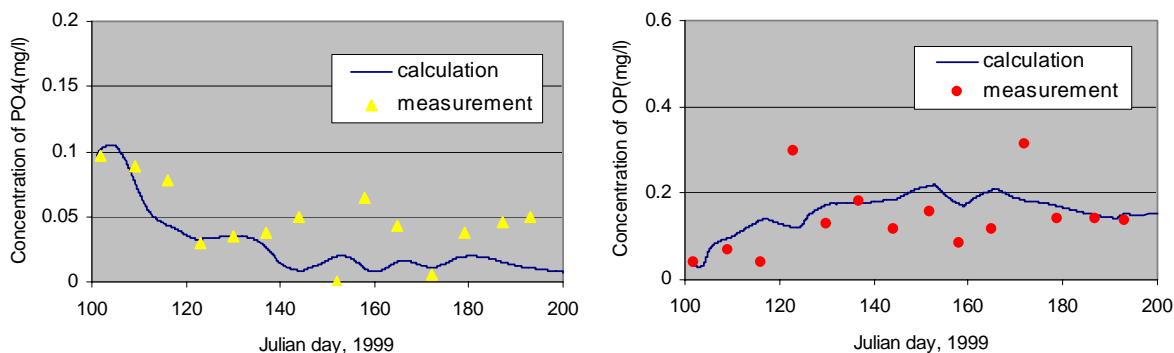


Fig. 10 The concentration of phosphorus at Station DH1(calibration)

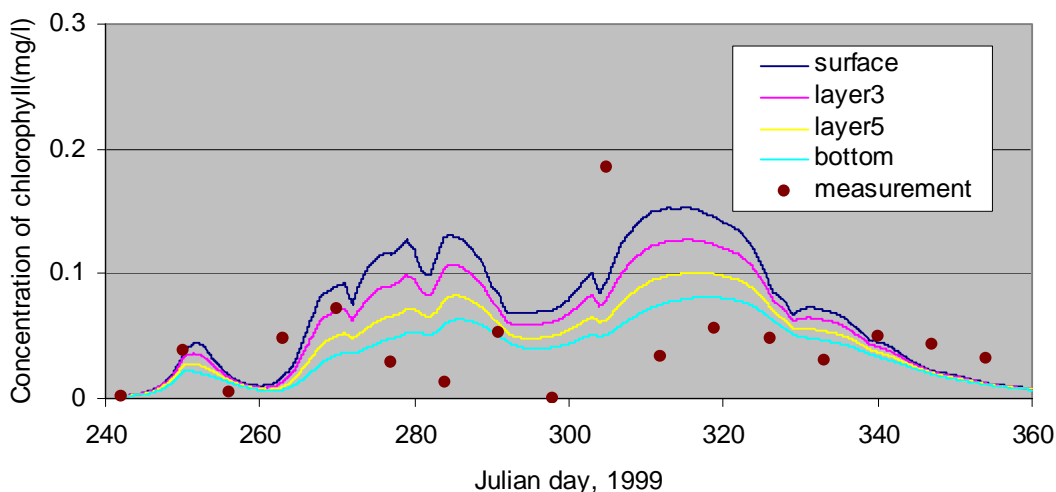


Fig. 11. The concentration of chlorophyll at Station DH1(validation)

Fig. 12 shows the simulated and observed concentrations of ortho-phosphorus and total organic phosphorus, respectively. Without considering the sediment-associated processes of adsorption, desorption and bed release, the model overestimated ortho-phosphorus concentration and underestimated organic phosphorus. After considering those processes, the root mean square error (RMSE) of ortho-phosphorus concentration was reduced from 0.029 to 0.016 mg/l, and for organic phosphorus was reduced from 0.051 to 0.037 mg/l.

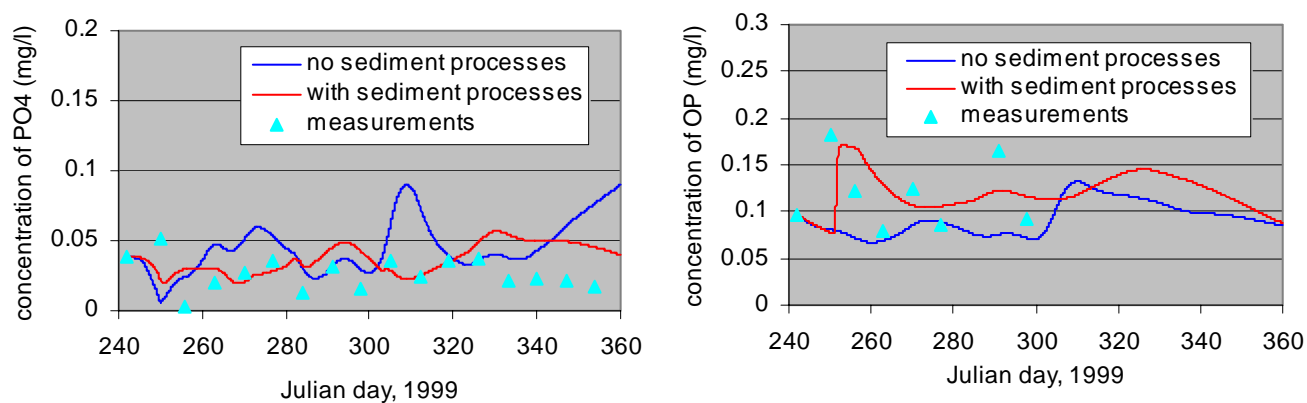


Fig. 12. The concentration of phosphorus at Station DH1(validation)

3.3 Scenario Study

Field data show that major water quality problems in Deep Hollow Lake in the late 1990s were caused by excessive sediment loads carried by runoff from surrounding cultivated lands (Rebich and Knight, 2001). In order to improve the water quality of the lake, various best management practices (BMPs) were employed to reduce sediment loads on the farm lands. After the reduction of lake sediment concentration, there were significant increases in Secchi depth and chlorophyll concentration, and fish populations responded positively (Rebich and Knight, 2001). The proposed water quality model was evaluated for sensitivity of SS to the chlorophyll concentration in the lake.

To show the sensitivity of primary production to SS, a series of hypothetical lake SS concentrations were input to the model, while the flow conditions and nutrient loadings at inlet boundary were kept the same. The interactions between SS and nutrients in the lake were considered in the model simulation. Fig. 13 shows the results. As expected, the concentration of chlorophyll is inversely related to sediment concentration. When the SS concentration is tripled, the simulated lake primary production practically ceased. Fig. 14 shows the sensitivity of temporal mean chlorophyll concentration to SS for the entire simulation period. The average base chlorophyll concentration was 0.059 mg/l. When lake SS was reduced by 50%, simulated mean chlorophyll concentration increased about 40%. When lake SS was doubled, the chlorophyll concentration fell to 37% of the base condition. This tendency agrees with observations before and after best management practices were installed in the Deep Hollow Lake watershed (Rebich and Knight, 2001).

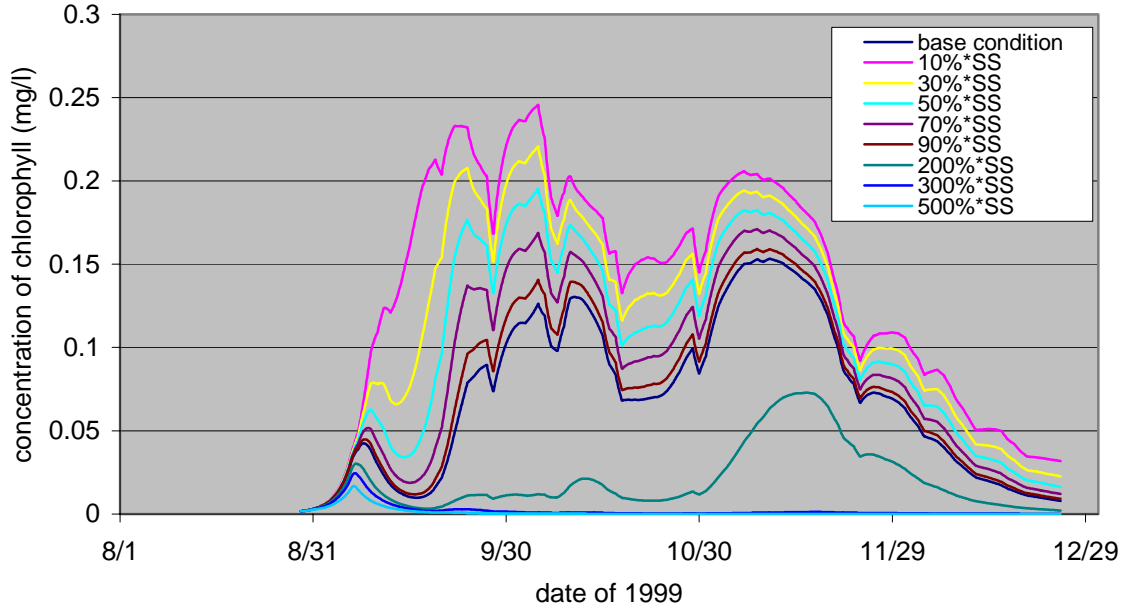


Fig. 13. The time series of chlorophyll concentration with different SS level (Aug.-Dec., 1999)

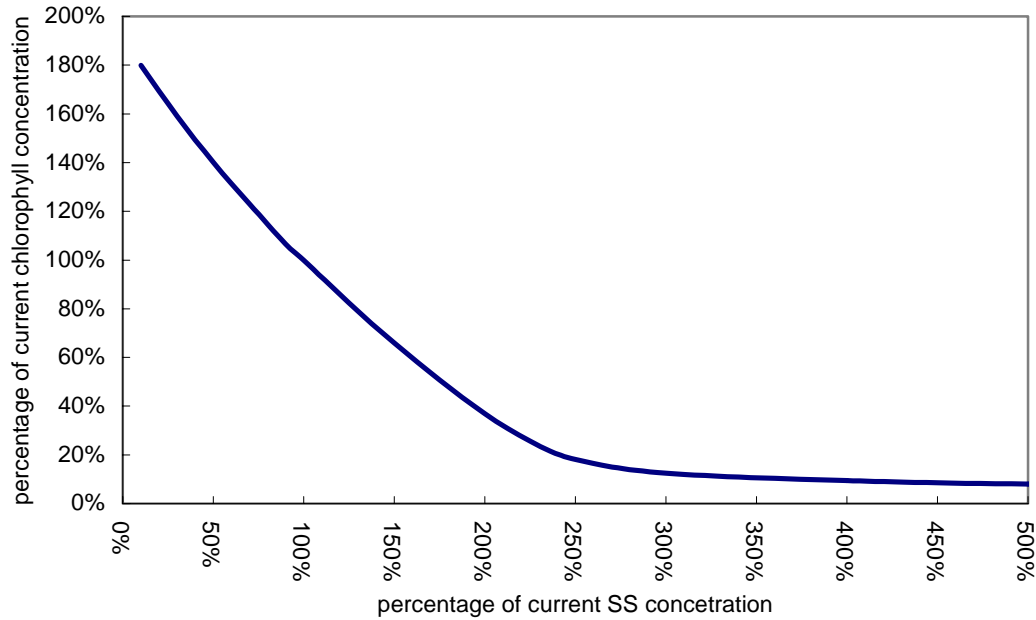


Fig.14. The effect of reduction/increasing of SS on the chlorophyll concentration

4. CONCLUSIONS

Sediment-related processes in water quality model were studied. A formula was obtained from field measurements to calculate the effect of sediment on the light attenuation coefficient. Based on the Langmuir equation, two algebraic equations (18) and (19) were derived to calculate the concentrations of particulate and dissolved nutrient at equilibrium due to adsorption-desorption. Eq. (20) was obtained to calculate the release rate from bed sediment by considering the effects of the concentration gradient across the water-sediment interface, pH, temperature and dissolved oxygen concentration. The measurements and numerical results show that the Langmuir equation is more reasonable than the simple Linear Assumption to express the relationship between dissolved and particulate nutrients.

A three-dimensional water quality model was applied to simulate the concentration of water quality constituents in shallow lakes where sediment-related processes are important. In this model the effects of sediment on the growth of phytoplankton, and the processes of adsorption-desorption and bed release of nutrients from sediment were simulated. The model was successfully applied to the study of the concentration of constituents important to the water quality of the Deep Hollow Lake. Realistic trends and magnitudes of nutrient and phytoplankton concentrations obtained from the numerical model generally agreed with field observations.

The model was then used to conduct analyses of the sensitivity of lake chlorophyll concentration to concentrations of SS. Lake primary productivity is mainly limited by SS concentration, which limits light penetration. The effects of sediment-related processes are quite important in water quality processes.

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