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Impacts of Agricultural Activities on Water Quality in Oxbow Lakes in the Mississippi Delta

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ABSTRACT: In the Mississippi Delta, agricultural activity is a major source of nonpoint source (NPS) pollutants. Sediment, nutrients and pesticides have been considered as priority NPS pollutants and greatly affect the water quality in this area. The impacts of agricultural activities on water quality in oxbow lakes in this area were assessed based on field measurements and numerical model. The long term measured data was used to analyze the concentration distribution of sediment in the lakes pre- and post- implementation of BMPs installed in the watersheds. Three major sediment-associated water quality processes were studied, including the effect of sediment on the growth of phytoplankton, the adsorption-desorption of nutrients by sediment and the release of nutrients from bed sediment layer. The newly improved and validated 3D water quality model was used to simulate the water quality constituents with considering more realistic 3D mixing in the oxbow lakes by wind shear on water surface. As a result, the simulated concentration of chlorophyll and nutrients were generally in better agreement with field observations. This study confirms that the water quality of the lakes was sensitive to suspended sediment concentrations. Of equal importance, the capability of simulating various physical and chemical rate processes, such as growth, death, settling, degradation, etc. in the water have been added in the computational model. Therefore, when watershed water quality is concerned, one has to consider factors such as sediment loadings and agriculture activities. And, the effectiveness of the BMP can be assessed by newly improved 3D water quality computational model.

Keywords: *Water quality, Agricultural activity, Numerical model, Oxbow lake, Mississippi Delta*

1 INTRODUCTION

The United States has more than 330 million acres of agricultural land that produce an abundant supply of food and other products. However, agricultural activities can degrade water quality if appropriate management is not implemented. The most recent National Water Quality Inventory reported that the agricultural nonpoint source (NPS) pollution is an important factor influencing quality of surface waters, as well as a contributor to groundwater contamination. To address NPS pollution problems and reduce discharging pollutants into surface waters in agricultural areas, best management practices (BMPs) have been installed in many watersheds.

The Mississippi Delta is one of the most intensively farmed agricultural areas of the United States. However, agricultural activities have been identified as a major source of NPS pollution, which greatly affect the water quality in surface water and ground water. To evaluate the efficiency of BMPs in reducing chemical runoff, several oxbow lakes in this area were selected for assessments under various ARS Projects, including the "Mississippi Delta Management Systems Evaluation Area (MDMSEA)" and "Conservation Effects Assessment Project (CEAP)" (Locke 2004, Locke et al. 2008). All of these watersheds fall within the Lower Mississippi River Basin, a Long-term Agroecosystem Research (LTAR) area.

In the present paper, the impacts of agricultural activities on water quality in natural oxbow lakes in the Mississippi delta were assessed based on field observations conducted by scientists from the USDA-ARS National Sedimentation Laboratory, Water Quality and Ecology Research Unit (WQERU). The field data were assessed using a water quality model developed by the National Center for Computational

Hydroscience and Engineering, University of Mississippi. The simulated water quality constituents were generally in good agreement with field observations. The water quality model was also applied to conduct analyses of the sensitivity of chlorophyll concentration to the suspended sediment concentration and nutrient loadings in the lake. It was determined that the lake primary productivity was mainly limited by the suspended sediment concentration, while it was somewhat less sensitive to concentrations of nitrogen and phosphorus.

2 STUDY AREA AND FIELD MEASUREMENTS

2.1 *Mississippi Delta*

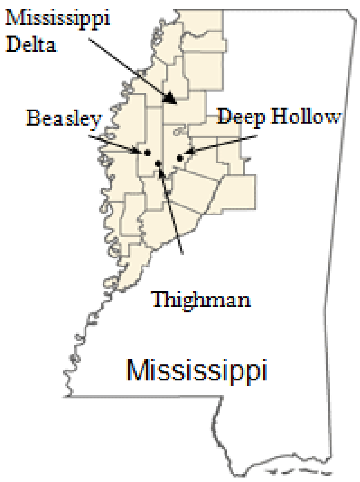


Figure 1. Mississippi Delta.

The Mississippi Delta is an alluvial plain located in the northwest portion of the State of Mississippi, and lies between the Mississippi River and Yazoo River. The Mississippi Delta is one of the largest contiguous agricultural areas in the United States with an area of about 5 million acres. The near level topography is well suited for large-scale mechanized agriculture, and the region is agronomically very productive. Major agricultural commodities include soybean, cotton, rice, corn, and catfish.

The rich, deep alluvial soils of the Mississippi Delta are highly erodible, and despite the flat topography, significant quantities of NPS sediment-associated pollution from agricultural activities are discharged into the water bodies, which can greatly influence water quality and aquatic biota (Rebich and Knight, 2001).

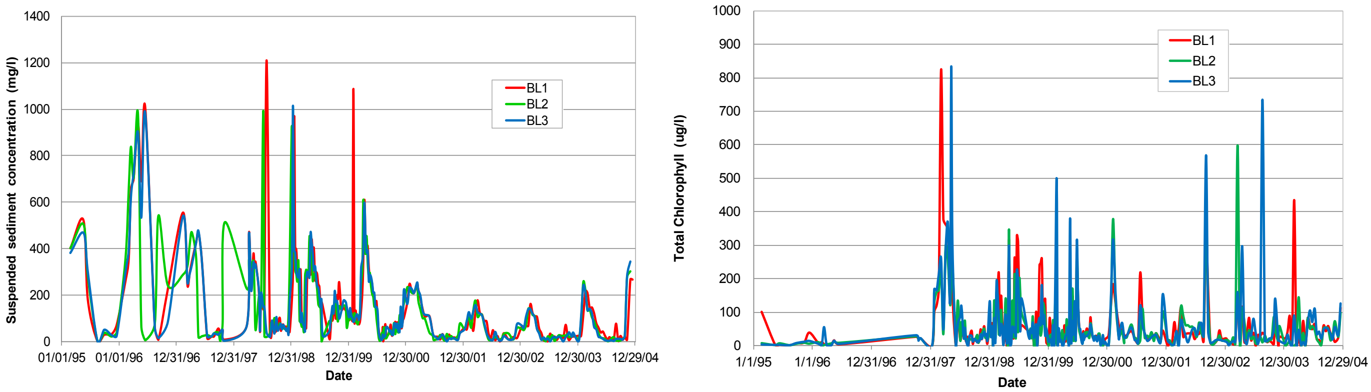
The Mississippi Delta Management Systems Evaluation Area project was initiated in 1995 to assess the effects of BMPs in oxbow lake watersheds (Locke, 2004). Three oxbow lakes were selected as study watersheds for this project: Beasley Lake, Deep Hollow Lake, and Thighman Lake (Fig.1). Oxbow lakes are a significant feature in the Mississippi Delta landscape, and the premise of the project was that the majority of runoff and associated contaminants would be captured in the lake. Therefore, lake water quality was a focus of integrated assessment of BMPs. BMPs were established in a hierarchy among the three lakes, where Thighman Lake watershed was a control (no BMPs implemented), Beasley Lake watershed only had structural BMPs installed, and both structural and agronomic BMPs were implemented in Deep Hollow Lake watershed. Although project scientists did not install BMPs in Thighman Lake watershed, farmers within the watershed implemented conservation management practices. Detailed information concerning the MD-MSEA study can be found in Locke (2004).

2.2 *Water quality measurements*

The MD-MSEA oxbow lakes were monitored by collecting water samples on a weekly or biweekly basis. After collection, samples were transported on ice to the USDA-ARS National Sedimentation Laboratory, Oxford MS, for processing and analysis. Samples were analyzed for suspended sediment (SS), nitrogen, phosphorus, chlorophyll, bacteria, pesticides. Generally, nitrate and ammonia concentrations in the lakes were relatively low, while phosphorus concentrations were relatively high (Knight et al., 2013). Suspended sediment concentrations were relatively high, exceeding the published levels known to adversely impact fish growth and health (Rebich and Knight, 2001). The water quality of the lakes was sensitive to suspended sediment concentrations because photosynthetic activity was limited by elevated turbidity levels following runoff events.

Fig. 2 shows the total concentrations of SS and chlorophyll of Beasley Lake pre- and post BMP implementations from 1995 to 2004. After implementation of BMPs in 1995-96, the mean total SS concentration was reduced from 360 mg/l to 130 mg/l, or about 65%; while the averaged total chlorophyll was increased from 10 µg/l to 60 µg/l, or about 5 times, which was more suitable for aquatic organisms. For the other two lakes, measured concentrations of SS and chlorophyll had a similar trend.

The measured data indicate that BMPs installed in the three lake watersheds, including those BMPs implemented by farmers in the control Thighman Lake watershed, reduced sediment loadings in the oxbow lakes. As a result, secchi depth visibility of the lakes increased, total phosphorus decreased, and chlorophyll concentration also increased. Since the nitrate and ammonia in the lakes were relatively low, the BMPs had little discernible effects on their concentrations. After the reduction of lake sediment concentrations, it was observed that the fish populations responded positively (Knight et al., 2013).



suspended sediment *Total chlorophyll*

Figure 2. The measured SS and chlorophyll concentrations in Beasley Lake.

3 MODEL DESCRIPTIONS

Numerical modeling is an effective and efficient approach for studying water quality constituents in surface water bodies. A three-dimensional water quality model was developed based on CCHE3D hydrodynamic model (Jia et al. 2013), and applied to predict the distributions of nutrient, phytoplankton, dissolved oxygen, pathogen, etc., in natural lakes affected by agricultural management. The model considers processes in the water column, the bed sediment layer, and the exchange between those layers. Several biochemical processes were selected for modeling: the phytoplankton kinetics, nitrogen cycle, phosphorus cycle, DO balance, pathogen processes. The conceptual framework for the eutrophication kinetics in the water column was mainly based on the WASP6 model (Wool et al. 2001). Eight state variables were involved in the interacting systems: ammonia nitrogen (NH3), nitrate nitrogen (NO3), phosphate (PO4), phytoplankton (PHYTO), carbonaceous biochemical oxygen demand (CBOD), DO, organic nitrogen (ON), and organic phosphorus (OP). To consider the impact of agricultural activities on water quality, three major sediment-associated water quality processes were simulated, including the effect of sediment on light penetration, the adsorption-desorption of nutrients by sediment and the release of nutrients from bed sediment (Chao et al. 2010). In addition, the fate and transport of pathogens were also included in the developed water quality model.

3.1 *Governing Equations*

In the water column, each one of the water quality constituents can be expressed by the following mass transport equation:

$$\frac{\partial C_i}{\partial t}+\frac{\partial (UC_i)}{\partial x}+\frac{\partial (VC_i)}{\partial y}+\frac{\partial (WC_i)}{\partial z}=\frac{\partial}{\partial x}(D_x\frac{\partial C_i}{\partial x})+\frac{\partial}{\partial y}(D_y\frac{\partial C_i}{\partial y})+\frac{\partial}{\partial z}(D_z\frac{\partial C_i}{\partial z})+\sum S_i \quad (1)$$

in which U, V, W are the water velocity components in x, y and z directions, respectively; C_i is the concentration of the ith water quality constituent; D_x, D_y and D_z are the diffusion coefficients in x, y and z directions, respectively; ΣS_i is the effective source term, which includes the kinetic transformation rate, external loads and sinks for ith water quality constituent.

The water quality model was decoupled with CCHE3D hydrodynamic model (Jia et al 2013). The CCHE3D model is based on the finite element method. A second-order upwinding scheme was adopted to eliminate oscillations due to advection. The velocity correction method was applied to solve the pressure and enforce mass conservation. The system of algebraic equations was solved using the Strongly Implicit Procedure (SIP). Based on the flow fields computed by CCHE3D, the concentration distribution of water quality constituents can be obtained by solving mass transport equations (1) numerically.

3.2 *Sediment-Associated Water Quality Processes*

3.2.1 *The Effect of Sediment on the Growth of Phytoplankton*

Phytoplankton plays a central role in the carbon and nutrient cycles that comprise the model ecosystem. Total chlorophyll is used as a simple measurement of phytoplankton biomass. The effective source term for phytoplankton is determined by the growth rate, death rate, and settling rate of phytoplankton. The growth rate is calculated by

$$G_p = P_{mx} f_N f_I f_T \tag{2}$$

in which P_{mx} is the maximum growth rate; and f_N, f_I and f_T are the limitations due to nutrient availability, light intensity, and temperature, respectively. The nutrient limitation factor f_N is obtained based on Michaelis-Menten equation and Liebig's Law of the minimum. The temperature limitation factor f_T is calculated by the formula proposed by Cerco and Cole (1995). The light limitation factor f_I is obtained by integrating the Steele equation over depth and time:

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

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 suspended sediment(SS), and can be expressed by (Stefan et al. 1983):

$$K_e = K_0 + K_{chl} + K_{ss} \tag{4}$$

where K_0 is the light attenuation by pure water (m^{-1}); K_{chl} is the attenuation by chlorophyll (m^{-1}); K_{ss} is the attenuation by suspended sediment (m^{-1}). Based on the research conducted by Stefan et al. (1983) and Wool et al. (2001), Eq. (4) can be given as:

$$K_e = K_0 + 0.0088 C_{chl} + 0.054 C_{chl}^{0.67} + \gamma s \tag{5}$$

where s is the concentration of suspended sediment; γ is a coefficient. The parameters of K_0 and γ can be obtained based on field measurements.

3.2.2 *Processes of Adsorption-Desorption of Nutrients by SS*

Adsorption and desorption are important processes between dissolved nutrients and SS in the water column. In water quality processes, 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). Many experimental results show the Langmuir equilibrium isotherm is a better representation of the relations between the dissolved and particulate nutrient concentrations (Bubba et al. 2003). The equilibrium adsorption content (Q) can be expressed as:

$$Q = \frac{Q_m K C_d}{1 + K C_d} \tag{6}$$

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. Based on Eq.(6), the particulate concentration C_p and dissolved nutrient concentration C_d can be calculated by

$$C_p = \frac{1}{2}\left[\left(C_0 + \frac{1}{K} + sQ_m\right) - \sqrt{\left(C_0 + \frac{1}{K} - sQ_m\right)^2 + \frac{4sQ_m}{K}}\right] \tag{7}$$

$$C_d = \frac{1}{2}\left[\left(C_0 - \frac{1}{K} - sQ_m\right) + \sqrt{\left(\frac{1}{K} + C_0 - sQ_m\right)^2 + \frac{4sQ_m}{K}}\right] \tag{8}$$

In the water quality model, the particulate concentration C_p, dissolved nutrient concentration C_d and total concentration ($C=C_p+C_d$) can be obtained numerically.

3.2.3 *Release of Dissolved Nutrients from Bed Sediment*

Bed release is an important source of inorganic and organic nutrients in the water column. In many models, the release rate of nutrients from bed sediment is determined based on the concentration gradient across the water-sediment interface. In fact, the bed release rate is also affected by pH, temperature and dissolved oxygen concentration. Based Romero (2003), the bed release rate can be expressed as:

$$S_{diff} = \theta_{sed}^{T-20} S_c \left(\frac{K_{dos}}{K_{dos} + DO} + \frac{|pH - 7|}{K_{pHs} + |pH - 7|} \right) \tag{9}$$

where S_{diff} is the bed release rate; S_c is the diffusive flux of nutrients; K_{dos} 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; 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 (Chao et al. 2006).

3.3 *Fate and Transport Processes of Pathogens*

Pathogens are disease causing micro-organisms which include various types of bacteria, viruses, protozoa, and other organisms. Pathogens are often found in contaminated water, frequently as a result of fecal matter from sewage discharges, leaking septic tanks, and runoff from animal feedlots. To analyze and identify the individual pathogen are usually time consuming and expensive, the indicator organism are frequently used to represent the pathogens. There are four commonly used indicators: total coliform, fecal coliform, Escherichia coli (E. coli), and enterococci. So the concentration of indicator organisms can be used to evaluate the water quality criteria for pathogen in a water body.

The concentration of indicator organisms can be solved using the mass transport equation (1). The effective source term for pathogen indicators can be calculated by:

$$\sum S_{path} = S_{load} + S_{sed} + S_{bed-w} + (G_{pt} - D_{pt}) C_{path} \tag{10}$$

in which ΣS_{path} is the effective source term for pathogen indicator; S_{load} is the source due to the external loading; S_{sed} is the source due to sediment erosion and deposition; $S_{bed\text{-}w}$ is the source due to bed release; G_{pt} is the growth rate; D_{pt} is the loss rate; and C_{path} is the concentration of pathogen indicator.

S_{sed} can be calculated by

$$S_{sed} = \max(E_b - D_b, 0) \frac{C_{path,b}}{1 - p'} + \min(E_b - D_b, 0) \left(\frac{f_p}{c_v} \right) C_{path} \tag{11}$$

in which $C_{path,b}$ is the pathogen concentration at bottom; E_b and D_b are the sediment erosion and deposition rates; p' is the porosity of bed sediment; f_p is the particulate pathogen fraction; and c_v is the volumetric concentration of sediment in water column.

The growth rate is determined by the temperature and nutrient availability, and can be calculated by:

$$G_{pt} = P_{ptmx} f_{pN} f_{pT} \tag{12}$$

where P_{tmx} is the maximum growth rate of pathogen; and f_{pN} and f_{pT} are the limitations due to nutrient availability and temperature, respectively. The nutrient limitation factor f_N is obtained based on Michaelis-Menten equation. The temperature limitation factor f_T is calculated by the formula proposed by Ross et al.(2003):

$$f_{pT} = \left\{ c_{T1}(T - T_{\min}) \left[1 - \exp(c_{T2}(T - T_{\max})) \right] \right\}^2 \tag{13}$$

where c_{T1} and c_{T2} are temperature parameters; T is the temperature in water column; T_{min} and T_{max} are minimum and maximum temperatures for pathogen to grow.

The pathogen loss is generally determined by the natural mortality, salinity, sunlight, predation, etc. The total loss rate can be calculated by:

$$D_{pt} = (0.8 + 0.02 S_a) 1.07^{T-20} + \frac{\alpha I_0}{K_e H} \left[1 - \exp(-K_e H) \right] \tag{14}$$

where S_a is the water salinity; H is the water depth; α is the proportionality constant and is generally equal to one.

4 APPLICATION TO AN OXBOW LAKE IN MISSISSIPPI DELTA

Deep Hollow Lake, one of ARS study lakes located in Mississippi Delta, was applied to demonstrate the capabilities of the developed water quality model. This lake receives runoff from a two square kilometer watershed which was heavily cultivated. Wind was an important driving force for the water movement within the lake (Fig.3). 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.

4.1 *Effect of SS on the Light Attenuation Coefficient in the Lake*

The light attenuation coefficient K_e in Eq.(4) is an important parameter for the growth of phytoplankton. Eq.(5) shows it is affected by water, concentrations of chlorophyll and SS. It has been observed that in the water column, the extinction of light is proportional to the light at any water depth:

$$\frac{dI}{dz} = -K_e I \tag{15}$$

in which I is the light intensity; z is the distance to water surface. At the water surface, $z = 0$ and $I=I_0$, so Eq.(15) can also be converted to:

$$\ln\left(\frac{I}{I_0}\right) = -K_e z \tag{16}$$

So K_e can be obtained by linear fitting based on the measured I, I_0 and z. To determine measured light attenuation coefficient K_e in Eq.(15), and the parameters K_0 and γ in Eq (5), 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 is called Photosynthetic Photon Flux Fluence Rate (PPFFR). Units are in micromole per second per square meter. Data was collected at Deep Hollow Lake approximately every two weeks from January to March, 2004 (weather permitted). Light radiation was measured during conditions of unobstructed sunlight at three sites: DH1, DH2, and DH3 Stations 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 in the water body until the unit touched the lake bottom. In addition, the concentrations of SS and chlorophyll at the same sites were also measured. Based on the field data, a regression line, with the slope γ and intersection K_0 equal to 0.0452 and 1.2, respectively, was obtained (Fig. 4). So Eq. (5) can be expressed as

$$K_e = 1.2 + 0.0088C_{chl} + 0.054C_{chl}^{0.67} + 0.0452s \tag{17}$$

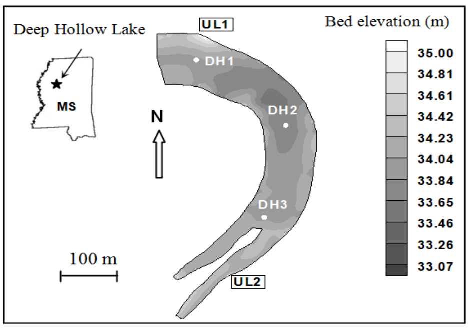


Figure 3. Deep Hollow Lake.

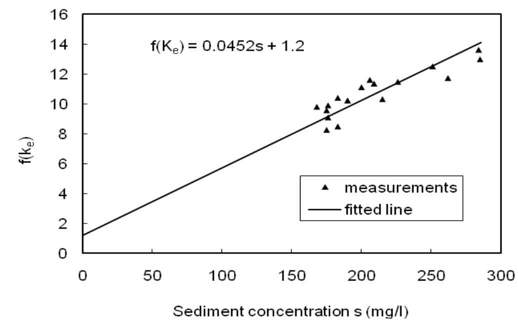
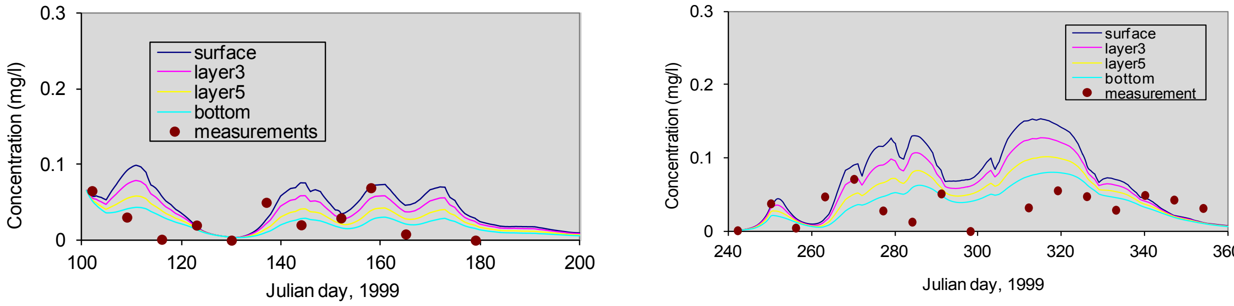


Figure 4. Relationship between $f(k_e)$ and sediment concentrations (based on Eq. 5, $f(K_e) = K_e - (0.0088C_{chl} + 0.054C_{chl}^{0.67}) = \gamma s + K_0$).

4.2 *Model Calibration and Validation*

The water quality model 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, and by others (Wool et al. 2001, Bubba et al. 2003). Fig.5a shows the simulated and observed chlorophyll concentration at Station DH1. 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 processes considered in the numerical model.

The period from September to December 1999 was chosen for model validation. Parameter values in the water quality model were same as those calibrated values. Fig. 5b shows the simulated and observed concentrations of chlorophyll. For the validation run, trends and quantities of concentration of chlorophyll obtained from the numerical model were generally in agreement with the observations.

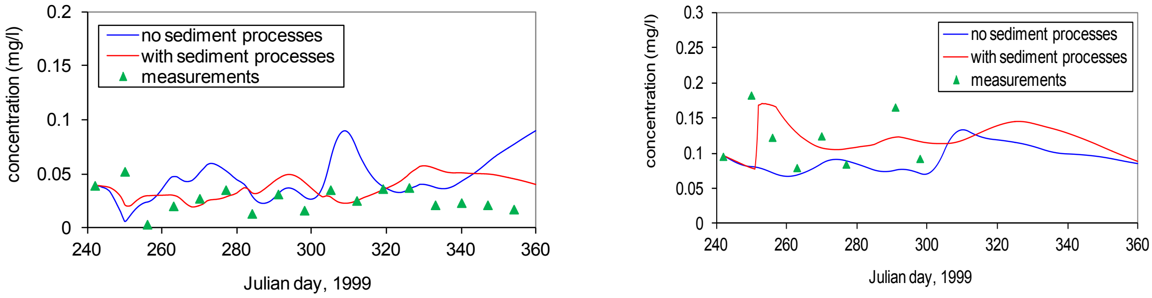


(b) Validation *(a) Calibration*

Figure 5. The concentration of chlorophyll at Station DH1.

4.3 *Effects of Sediment-Associated Processes on Water Quality captions*

The sediment-associated processes play important roles in water quality interaction system. To understand the effects of sediment in this system, the numerical model was applied to simulate the water quality constituents under two different conditions: considering the sediment-associated processes, and without considering those processes. Figs. 6a and 6b show 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, or reduced 45%; for organic phosphorus RMSE was reduced from 0.051 to 0.037 mg/l, or reduced 28%.



(a) ortho-phosphorus *(b) organic phosphorus*

Figure 6. The concentration of phosphorus at Station DH1.

5 DISCUSSIONS

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. In order to improve the water quality of the lake, various best management practices (BMPs), such as edge-of field BMPs and agronomic 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.

To show the sensitivity of chlorophyll concentration to SS, a series of hypothetical lake SS concentrations were input to the model, while the flow conditions and nutrient loadings at inlet boundaries were kept the same. The interactions between SS and nutrients in the lake were considered in the model simulation. The calibration and validation periods (1999) were selected for sensitivity study, and the current conditions of SS, nutrients, and chlorophyll were set as base conditions. Fig. 7 and Fig. 8 shows the measured concentrations of SS and chlorophyll at Station DH1 of Deep Hollow Lake in 1999. It was assumed the SS levels were varied from 10% to 300% of the base condition (Fig. 7), and the model was applied to simulate the responses of chlorophyll concentrations in the lake. As expected, the concentration of chlorophyll is inversely related to the sediment concentration.

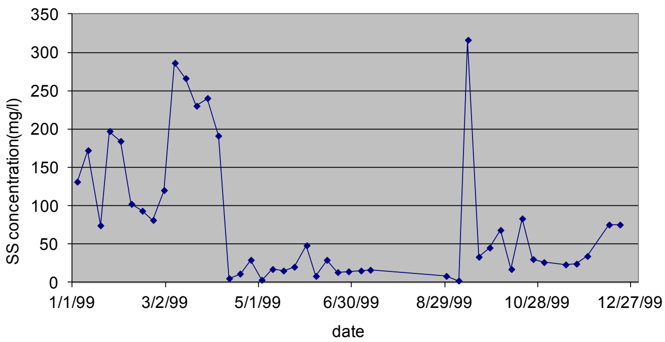


Figure 7. Measured concentration of SS in the lake.

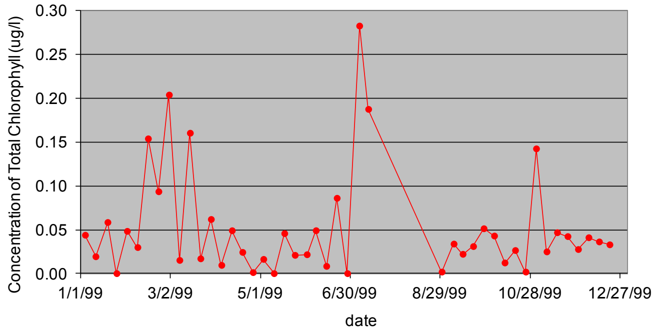


Figure 8. Measured total chlorophyll concentration in the lake.

Fig. 9 shows the sensitivity of temporal mean chlorophyll concentration to SS for the simulation period at DH1 Station. In this figure, the square represents the base condition (SS=100%, Chlorophyll=100%). When lake SS was reduced by 50%, simulated mean chlorophyll concentration increased about 40%.

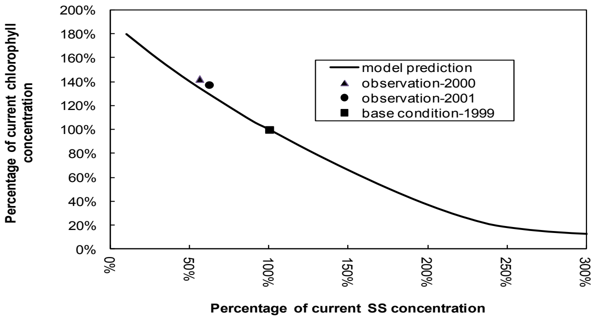
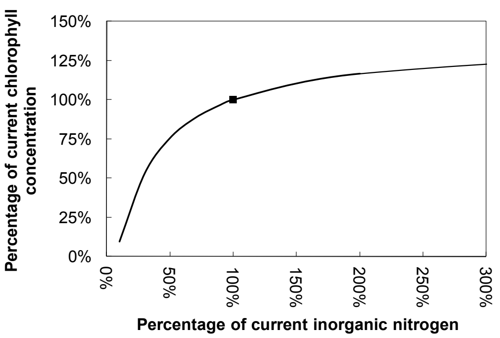


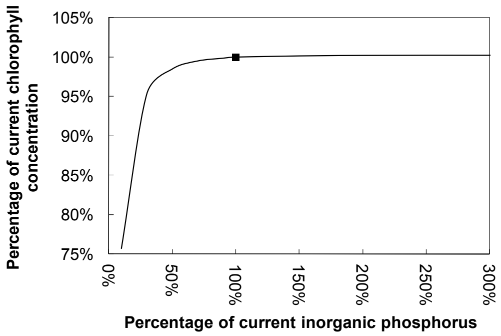
Figure 9. Sensitivity of temporal mean chlorophyll concentration to temporal mean SS.

When lake SS was doubled, the chlorophyll concentration fell to 37% of the base condition. Field observations of SS and chlorophyll in the years 1999, 2000 and 2001 were used to evaluate the model sensitivity analysis. Measured mean concentrations of SS and chlorophyll under the base condition (Year 1999) were 81 mg/l and 0.05 μg/l, respectively. Mean concentrations of SS in 2000 and 2001 were reduced to 56% and 62% of base condition, and the chlorophyll concentrations increased to 142% and 137% of the base condition, respectively (Fig.9). This tendency agreed with the numerical predictions shown in Fig. 9.

To analyze the effects of nutrient loadings on chlorophyll concentration, scenarios were generated by reducing and increasing the observed concentrations of dissolved inorganic nitrogen and inorganic phosphorus. The calibration and validation periods (1999) were selected for sensitivity study. The measured concentrations of inorganic nitrogen and inorganic phosphorus were set as based conditions. It was assumed the nutrient loadings were varied from 10% to 300% of the base conditions, and the model was applied to simulate the chlorophyll concentration in the lake. In the simulation, the concentration of SS was same as the base condition. The influences of inorganic nitrogen and inorganic phosphorus on the chlorophyll were simulated separately. Reducing nutrient loads produced lower chlorophyll concentrations, as expected. Fig. 10a and Fig. 10b show the sensitivities of temporal mean concentration of chlorophylls under the base conditions and reduction/increasing loads for inorganic nitrogen and inorganic phosphorus, respectively. In these figures, the squares represent the base conditions (nutrient=100%, Chlorophyll=100%). According to model simulations, reducing the lake inorganic nitrogen concentration by 50% would reduce the average chlorophyll concentration by approximately 24%. When the concentration of inorganic nitrogen was doubled, the chlorophyll concentration increased about 16.5%. For inorganic phosphorus, reducing the concentration by 50% in the lake would reduce average chlorophyll concentration by approximately 2%. When the concentration of inorganic phosphorus was doubled, the average concentration of chlorophyll increased less than 0.5%. This analysis indicates that the concentration of chlorophyll is much more sensitive to the inorganic nitrogen than that to inorganic phosphorus in Deep Hollow Lake. It may be the reason that the phosphorus concentration in the lake has reached the saturation level for the growth of phytoplankton.



a. Inorganic nitrogen



b. Inorganic phosphorus

Figure 10. The effect of reduction/increasing of nutrient loadings on the chlorophyll concentration.

6 CONCLUSIONS

The impacts of agricultural activities on water quality in oxbow lakes in the Mississippi Delta were studied based on field measurements and numerical modeling. The long term measured data showed the sediment concentration in the lakes reduced significantly after the implementation of BMPs installed in the watersheds.

Three major sediment-associated water quality processes were studied. A formula was obtained based on field measurements to calculate the light attenuation coefficient by considering the effects of SS and chlorophyll. Therefore, the growth rate of phytoplankton was reasonably predicted. The concentrations of particulate and dissolved nutrients due to adsorption-desorption were calculated using nonlinear formulas derived based on the Langmuir Equation. This approach is more realistic to model the process of adsorption-desorption than the linear approach. The release rates of nutrients from the bed sediment were calculated not only by considering the effects of the concentration gradient across the water-sediment interface, the effects of pH, temperature, and dissolved oxygen concentration were also taken into account. These sediment-associated water quality processes were included in the CCHE3D_WQ model to simulate water quality constituents in an oxbow lake in the Mississippi Delta. The trends and magnitudes of nutrient and chlorophyll concentrations obtained from the numerical model generally agreed with field observations. The measurements and computational results show that there are strong interactions between sediment and water quality constituents.

The water quality model was also applied to conduct analyses of the influences of SS concentration and nutrient loadings on the chlorophyll concentration. It was found that the lake chlorophyll is mainly limited by the SS concentration, while it is somewhat less sensitive to concentrations of nutrients.

Since water quality constituents in the oxbow lakes in Mississippi Delta are mainly influenced by agricultural activities, the model presented here is designed to assess their impacts. In many cases, the water flows and sediment concentrations in those lakes may have obvious 3D distributions, the developed CCHE3D_WQ model will be able to predict water quality distributions horizontally and vertically. The simulated results will provide more realistic information to assess the impacts of agricultural activities on water quality in the water bodies. Assessments may support the decisions for selection of agricultural conservation measures, adoption of planning and management policy, and operations to protect the quality of water, ecology and environment in areas adjacent to shallow lakes in agricultural watersheds.

The accuracy and efficiency of a water quality model are highly dependent on simulated physical-chemical processes, boundary conditions and computational technologies. To significantly improve water quality modeling, it is necessary to conduct additional researches to further understand the aquatic physical-chemical process, to improve model inputs on watershed information and agri-management, and to update computing methods for integrated watershed-surface water modeling system.

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