IMPLEMENTATION OF THREE-DIMENSIONAL EUTROPHICATION MODEL OF BUBBLY CREEK, CHICAGO, ILLINOIS

BY

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THESIS

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The Chicago Area Waterways System (CAWS) is a highly branched complex consisting of natural rivers and artificial canals. Known for its flow reversal project done in the early 20th century, the CAWS now links Lake Michigan with the Mississippi watershed, and the water generally flows westward, away from the lake. This is to prevent highly polluted water from entering the lake during storm events which could contaminate Chicago’s main drinking water source. It has been found that during extreme weather events, the water level in the Chicago River can be higher than the water level in the lake, and to prevent flooding in the city, the water has to be allowed to flow back to the lake. This could cause contamination of the drinking water source and the close down of beach areas in the summer months.

One of the main sources of contaminants in the CAWS is Bubbly Creek, a highly contaminated channel that joins Chicago River’s South Branch near Ashland Avenue. It served as the disposal area for Chicago’s largest stockyard, Union Stockyard during the late 19th century, and tons of animal remains were deposited onto the river bed everyday. The decomposition of those organic substances has depleted the dissolved oxygen in the water column, and annihilated most aquatic lives. Other issues such as stagnant flow conditions during dry weather season and contaminants brought in by highly polluted flow discharged from Racine Avenue Pumping Station (RAPS) exacerbate the degradation of the ecosystem. Extensive studies have been done in the area to study the transport and fate of the contaminants, but a detailed three-dimensional eutrophication model for Bubbly Creek has never been established due to constraints in computational power. With the development of high speed computers and advancement in parallel programming, computational power has increased dramatically, and is no longer a barrier for the development of more complex three-dimensional models. Thus, in this research, a three-dimensional eutrophication model based on the EPA (En-
Environmental Protection Agency) approved three-dimensional model EFDC (Environmental Fluid Dynamics Code) is developed to study the flow and water quality characteristics in Bubbly Creek for the period of May 2009. The validation of the simulation results is done with the help of measured data provided by USGS (U.S. Geological Survey) and MWRDGC (Metropolitan Water Reclamation District of Greater Chicago). It has been found that the water quality in Bubbly Creek is predominantly controlled by combined sewer overflows (CSO) from RAPS. The dissolved oxygen (DO) level in Bubbly Creek could reach near saturation when pumping activity occurs, and would rapidly decrease to near zero after the pumping activity has stopped. The contaminants brought in by the discharge from RAPS not only impair the local ecosystem in Bubbly Creek, but also travel all the way to the downstream boundary at Stickney and deteriorate the DO concentration along the way.
ACKNOWLEDGMENTS

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Secondly, I want to express my sincere appreciation to Dr. Zhenduo Zhu, who is an expert in EFDC, for his patient guidance for the model setup, and for his valuable suggestions that led to the completion of this thesis.

I would also like to thank all the members of our group for their friendship and encouragement. They have made my journey here in Urbana-Champaign a wonderful and unforgettable experience.

Last but not least, I want to thank my parents in China for their selfless support and love. They are the first ones who have faith in me, and encouraged me to embark on this journey to pursue my dreams.
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CHAPTER 1

INTRODUCTION

1.1 Motivation

Being the third most populated city in the United States, the City of Chicago, with a population of nearly 3 million, has always been facing a lot of challenges when it comes to water related issues. Whether it is providing enough drainage to prevent flooding during heavy storm events, or just maintaining clean and safe drinking water sources, the tasks are strenuous, and many well-known civil engineering feats have been accomplished to try to solve those issues. The Chicago River flow reversal project for example, is one of such notable achievements accomplished by the early 20th century.

![Figure 1.1: Chicago River Before and After Diversion](image)

During the late 19th century, the City of Chicago was undergoing rapid modernization with the population reaching half a million by the end of the 1880s. Factories, breweries and slaughter houses were built along the Chicago River, which looked like Figure 1.1(left) at that time, to provide jobs, drinks and
food for the new metropolis. Waste that was dumped into the river either deposited to the bottom, or was carried away by the flow to Lake Michigan, which was the main source of drinking water supply for the city [12]. In 1885, an extreme storm event caused so many contaminants to be carried into Lake Michigan that the water became almost undrinkable[8]. In response to this threat, the Illinois General Assembly created the Chicago Sanitary District, now the Metropolitan Water Reclamation District of Greater Chicago (MWRDGC), to replace the Illinois-Michigan Canal with the Chicago Sanitary and Ship Canal (CSSC), and reverse the flow so that it would be conveyed from Lake Michigan into the South Branch of the Chicago River, and to the Mississippi watershed[15]. Figure 1.1(right) shows what current Chicago River looks like.

Even with the water flowing the opposite way, during large storm events, water level in the Chicago River could be higher than the water level in the lake. To prevent flooding in the city, MWRDGC sometimes have to let the water flow back to the lake. It was observed that by allowing water to enter Lake Michigan during large storm events, contaminants resuspended from the bottom of certain reaches of the river, such as Bubbly Creek, could also enter the lake, causing contamination to the drinking water source and the closing down of beach areas.

In the study done by Quijano et al[11], a three-dimensional EFDC-WASP water quality model was developed to simulate the flow reversal phenomenon for the entire Chicago River, including the CSSC, North Shore Channel (NSC), and the Calumet Sag Channel (CSC). The model is capable of providing satisfactory simulation results for most part of the model domain, but falls short when it comes to matching the measurement data near Bubbly Creek, which is one of the major historical sources of contaminants in the Chicago River. Therefore, a more detailed, small domain three-dimensional water quality model focusing on Bubbly Creek area is desired to better understand the water quality dynamics in this region. In this research, a complete three-dimensional eutrophication model, built upon the simplified BOD-DO model developed by Dr. Sumit Sinha[13], will be used to study the water quality characteristics in the Bubbly Creek area during the month of May 2009.
1.2 Current Issues: About Bubbly Creek

Running entirely within the city of Chicago, Bubbly Creek, also known as the South Fork of South Branch of the Chicago River, is a 2.2km long channel which starts at the Racine Avenue Pumping Station (RAPS) near 38th Street, and ends at Chicago River’s South Branch near S Ashland Ave. It is a rather small stream with width varying between 40 to 65 meters, and depth ranging from 2 meters near the RAPS to 5 meters near its mouth at the turning basin.

Figure 1.2: Location of Bubbly Creek

In the late 19th century, Bubbly Creek served as the primary dumping ground for the Union Stockyards, Chicago’s largest stockyard during that time. It gained its name because the animal leftovers remained in the bed would decompose and create methane bubbles. Even to this date, bubbles still
constantly float to the surface of the river during warm weather periods. This man-made degradation to the sediment condition has lead to the depletion of dissolved oxygen in the water column, and the extinction of most aquatic lives. Other problems such as stagnant flow conditions during dry weather period, and excessive amount of combined sewer overflow from RAPS make the restoration of Bubbly Creek’s ecosystem a challenging task.

1.3 Objective and Thesis Outline

The objective of this research is to develop a working three-dimensional eutrophication model that is capable of simulating the water quality dynamics in the water column of the model domain. Both the hydrodynamic and the water quality results will be validated with the help of measured data.

The chapters in this thesis are organized as follows:

- Chapter 2 discusses the theories and equations used in the hydrodynamic and water quality model.
- Chapter 3 presents the development of the hydrodynamic model, and the validation and discussion of simulation results.
- Chapter 4 presents the development of the water quality model, and the validation and discussion of simulation results.
- Chapter 5 summarizes the results, discusses the limitations of the current model, and provides suggestions for future works.
CHAPTER 2

EFDC MODEL OVERVIEW

2.1 Hydrodynamic Model

The Environmental Fluid Dynamics Code (EFDC) is an EPA (Environmental Protection Agency) approved multifunctional surface water modeling system, which includes hydrodynamic, sediment-contaminant, and eutrophication components. Originally developed at the Virginia Institute of Marine Science (VIMS) and School of Marine Science of The College of William and Mary by Dr. John M. Hamrick, EFDC has evolved over the past few decades to become one of the most widely used and technically defensible hydrodynamic models in the world.

EFDC can simulate water and water quality constituent transport in geometrically and dynamically complex water bodies, such as rivers, estuaries, lakes, and coastal seas. The governing equations used in EFDC for ambient environmental flows begins with the vertically hydrostatic, boundary layer form of the turbulent equations of motion for an incompressible fluid. The hydrostatic assumption is generally a preferred modeling assumption when the modeled water bodies have horizontal length scales orders of magnitudes larger than their vertical length scales. This is a valid assumption in the case of the Chicago River. In addition, it will be much more convenient to formulate the equations if the horizontal coordinates are curvilinear and orthogonal. Therefore, a time variable mapping transformation is required to provide a uniform resolution in the vertical direction, aligned with the gravitational vector and bounded by bottom bathymetry and a free surface. The mapping function used is given by,

\[ z = \frac{z^* + h}{\zeta + h} \]  

(2.1)
The $z^*$ denotes the original physical vertical coordinates, while $-h$ and $\zeta$ represent the physical vertical coordinates of the bottom bathymetry and the free surface respectively. Studies done by Vinokur [20], Blumberg and Mellor [2] and Hamrick [7] present the details of the transformation. After applying the mapping function, and utilizing the Boussinesq approximation (density differences are ignored except where they appear in terms multiplied by gravity), the desired form of the governing equations are obtained. Momentum and continuity equations, as well as the transport equations for salinity and temperature are presented as follows:

\begin{align*}
\partial_t (mHu) + \partial_x (m_y Hu) + \partial_y (m_x Hv) + \partial_z (mw) \\
- (mf + v\partial_x m_y - u\partial_y m_x) Hv = -m_y H \partial_x (g\zeta + p) \\
- m_y (\partial_x h - z\partial_x H) \partial_z p + \partial_z (mH^{-1} A_v \partial_z u) + Q_u
\end{align*} \tag{2.2}

\begin{align*}
\partial_t (mHv) + \partial_x (m_y Hu) + \partial_y (m_x Hv) + \partial_z (mw) \\
+ (mf + v\partial_x m_y - u\partial_y m_x) Hu = -m_x H \partial_y (g\zeta + p) \\
- m_x (\partial_y h - z\partial_y H) \partial_z p + \partial_z (mH^{-1} A_v \partial_z v) + Q_v
\end{align*} \tag{2.3}

\begin{align*}
\partial_z p = -gH (\rho - \rho_o) \rho_o^{-1} = -gHb
\end{align*} \tag{2.4}

\begin{align*}
\partial_t (m\zeta) + \partial_x (m_y Hu) + \partial_y (m_x Hv) + \partial_z (mw) = 0
\end{align*} \tag{2.5}

\begin{align*}
\partial_t (m\zeta) + \partial_x (m_y H \int_0^1 u dz) + \partial_y (m_x H \int_0^1 v dz) = 0
\end{align*} \tag{2.6}

\begin{align*}
\rho = \rho(S, T)
\end{align*} \tag{2.7}

\begin{align*}
\partial_t (mHS) + \partial_x (m_y HuS) + \partial_y (m_x HvS) \\
+ \partial_z (mwS) = \partial_z (mH^{-1} A_v \partial_z S) + Q_S
\end{align*} \tag{2.8}
\[ \partial_t(mHT) + \partial_x(m_yHuT) + \partial_y(m_z HvT) + \partial_z(mwT) = \partial_z(mH^{-1}A_b \partial_z T) + Q_T \]  \hspace{1cm} (2.9)

The u and v terms in equations from Eq. (2.2) to Eq. (2.9) are the horizontal velocity components in the curvilinear orthogonal coordinates x and y; \( m_x \) and \( m_y \) are the square roots of the diagonal components of the metric tensor; \( m = m_x m_y \) is the Jacobian or square root of the metric tensor determinant. After applying the mapping function in the vertical direction, the vertical velocity \( w \) in the stretched dimensionless vertical coordinate \( z \), is related to the physical dimensioned velocity \( w^* \) by Eq. (2.10).

\[ w = w^* - z(\partial_t \zeta + um_x^{-1} \partial_x \zeta + vm_y^{-1} \partial_y \zeta) + (1 - z)(um_x^{-1} \partial_x h + vm_y^{-1} \partial_y h) \]  \hspace{1cm} (2.10)

The total depth \( H \), is computed as the sum of the depth below the undisturbed physical vertical coordinate origin, \( h \), and the free surface elevation above that, \( \zeta \). The pressure \( p \) is the ratio of physical pressure in excess of the reference density hydrostatic pressure to the reference density, and is given by this form: \( p = \rho_o g H (1 - z)/\rho_o \). In the momentum equations Eq. (2.2) and Eq. (2.3), \( f \) and \( A_v \) are the Coriolis parameter and the vertical turbulent or eddy viscosity respectively, and \( Q_u \) and \( Q_v \) are momentum source and sink terms. The buoyancy term \( b \) in Eq. (2.4) is, as defined in the partial differential equation, the ratio of the deviation of density from its reference value to the reference density. The density, \( \rho \) is a function of temperature, \( T \), and salinity, \( S \), as shown in Eq. (2.7). Eq. (2.6) is produced by integrating Eq. (2.5) with respect to \( z \) from bottom \( (z=0) \) to top \( (z=1) \) using the kinematic boundary condition of \( w=0 \) at both locations. Similar to the momentum equations, \( Q_s \) and \( Q_T \) in Eq. (2.8) and Eq. (2.9) represent the source and sink terms for salinity and temperature respectively, and \( A_b \) represents the vertical turbulent diffusivity. The vertical turbulent viscosity, \( A_v \), and the vertical turbulent diffusivity, \( A_b \), are provided by the second moment turbulence closure model developed by Mellor and Yamada [9], and modified by Galperin et al. [5]. The model relates the vertical turbulent viscosity and diffusivity to the turbulent intensity, \( q \), turbulent length scale, \( l \), and
Richardson number $R_q$ in the following manner:

$$A_v = \phi_v ql = 0.4(1 + 36R_q)^{-1}(1 + 6R_q)^{-1}(1 + 8R_q)ql \quad (2.11)$$

$$A_v = \phi_b ql = 0.5(1 + 36R_q)^{-1}ql \quad (2.12)$$

$$R_q = \frac{gH\partial \zeta}{q^2 \frac{l^2}{H^2}} \quad (2.13)$$

The so-called stability functions $\phi_v$ and $\phi_b$ in Eq. (2.11) and Eq. (2.12) account for reduced or enhanced vertical mixing or transport in stable and unstable vertically density stratified environments, respectively. Table 2.1 summarizes the definitions of all variables mentioned above.

Table 2.1: Summary of variables

<table>
<thead>
<tr>
<th>Variables</th>
<th>Definition</th>
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<tbody>
<tr>
<td>$u, v$</td>
<td>Horizontal velocity components in the curvilinear orthogonal coordinates $x$ and $y$</td>
</tr>
<tr>
<td>$w$</td>
<td>Vertical velocity in the stretched dimensionless vertical coordinate $z$</td>
</tr>
<tr>
<td>$w^*$</td>
<td>Physical dimensioned velocity</td>
</tr>
<tr>
<td>$m_x, m_y$</td>
<td>Square roots of the diagonal components of the metric tensor</td>
</tr>
<tr>
<td>$m$</td>
<td>Square root of the metric tensor determinant</td>
</tr>
<tr>
<td>$H$</td>
<td>Total depth</td>
</tr>
<tr>
<td>$h$</td>
<td>Depth below the undisturbed physical vertical coordinate origin</td>
</tr>
<tr>
<td>$\zeta$</td>
<td>Free surface elevation above the undisturbed physical vertical coordinate origin</td>
</tr>
<tr>
<td>$p$</td>
<td>Pressure</td>
</tr>
</tbody>
</table>
Table 2.1 (cont.)

<table>
<thead>
<tr>
<th>Symbol</th>
<th>Description</th>
</tr>
</thead>
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<tr>
<td>$\rho, \rho_0$</td>
<td>Density and reference density</td>
</tr>
<tr>
<td>$f$</td>
<td>Coriolis parameter</td>
</tr>
<tr>
<td>$A_v$</td>
<td>Vertical eddy viscosity</td>
</tr>
<tr>
<td>$Q_u, Q_v$</td>
<td>Momentum source and sink terms</td>
</tr>
<tr>
<td>$b$</td>
<td>Buoyancy</td>
</tr>
<tr>
<td>$S, T$</td>
<td>Salinity and Temperature</td>
</tr>
<tr>
<td>$Q_s, Q_T$</td>
<td>Salinity source/sink terms and Temperature source/sink terms</td>
</tr>
<tr>
<td>$A_b$</td>
<td>Vertical turbulent diffusivity</td>
</tr>
<tr>
<td>$A_v$</td>
<td>Vertical turbulent viscosity</td>
</tr>
<tr>
<td>$q$</td>
<td>Turbulent intensity</td>
</tr>
<tr>
<td>$l$</td>
<td>Turbulent length scale</td>
</tr>
<tr>
<td>$R_q$</td>
<td>Richardson number</td>
</tr>
<tr>
<td>$\phi_v, \phi_b$</td>
<td>Stability functions</td>
</tr>
</tbody>
</table>
2.2 Water Quality Model

Dissolved oxygen (DO) is the key component that sustains various life forms in the water, and is therefore considered the health indicator of estuarine systems. The EFDC water quality model centers around this issue, and aims to predict the primary production and consumption of DO with a very detailed set of model state variables shown in Table 2.2. The three variables, namely water temperature, salinity and total suspended solids required for the computation of the 21 state variables stated above, are computed and provided by the EFDC hydrodynamic model, and the interactions among the state variables are illustrated in Figure 2.1. The kinetic processes presented in the EFDC water quality model are mostly derived from the Chesapeake Bay three-dimensional water quality model, CE-QUAL-ICM by Cerco and Cole [3]. Unlike other water quality models, such as Water Quality Analysis Simulation Program (WASP), which uses biochemical oxygen demand (BOD) to represent oxygen demanding organic material, EFDC water quality model uses organic carbon instead. Due to limited measurement data availability during the modeling period, the following state variables will not be simulated in this project: (2) Diatom Algae, (3) Green Algae, (16) Particulate Biogenic Silica, (17) Dissolved Available Silica, (18) Chemical Oxygen Demand and (20) Total Active Metal. The kinetic sources and sinks, as well as the external loads for each simulated state variable will be described in this section.

Table 2.2: EFDC Water Quality Model State Variables

| (1) Cyanobacteria          | (12) Labile Particulate Organic Nitrogen |
| (2) Diatom Algae           | (13) Dissolved Organic Nitrogen          |
| (3) Green Algae            | (14) Ammonia Nitrogen                    |
| (4) Refractory Particulate Organic Carbon | (15) Nitrate Nitrogen                  |
| (5) Labile Particulate Organic Carbon | (16) Particulate Biogenic Silica       |
| (6) Dissolved Organic Carbon | (17) Dissolved Available Silica          |
| (7) Refractory Particulate Organic Carbon | (18) Chemical Oxygen Demand |
| (8) Labile Particulate Organic Phosphorus | (19) Dissolved Oxygen                   |
| (9) Dissolved Organic Phosphorus | (20) Total Active Metal                |
| (10) Total Phosphate       | (21) Fecal Coliform Bacteria             |
| (11) Refractory Particulate Organic Nitrogen | |

Eq. (2.14) is the governing mass-balance equation for each of the water qual-
Figure 2.1: Schematic diagram for the EFDC water column water quality model

* TSS from hydrodynamic model
ity state variables.

\[
\begin{align*}
\partial_t C + \partial_x (uC) + \partial_y (vC) + \partial_z (wC) &= S_C \\
+\partial_x (K_x \partial_x C) + \partial_y (K_y \partial_y C) + \partial_z (K_z \partial_z C)
\end{align*}
\]  
(2.14)

where

\( C \) = concentration of a water quality state variable
\( u, v, w \) = velocity components in the x-, y-, and z-directions, respectively
\( K_x, K_y, K_z \) = turbulent diffusivities in the x-, y-, and z-directions, respectively
\( S_C \) = internal and external sources and sinks per unit volume

The advective and diffusive transport terms for state variables are accounted for by the last three terms on the left-hand side (LHS) and right-hand side (RHS) of Eq. (2.14) respectively. The first term on the RHS of Eq. (2.14) represents the kinetic processes and external loads for each of the state variables. When solving the governing mass-balance equations, the kinetic terms are decoupled from the physical transport terms, and the equation for kinetic processes alone is given by Eq. (2.15):

\[
\partial_t C = S_C
\]  
(2.15)

which can then be split into reactive and internal source/sink terms given by Eq. (2.16).

\[
\partial_t C = K \cdot C + R
\]  
(2.16)

where

\( K \) = kinetic rate (time\(^{-1}\))
\( R \) = source/sink term (mass volume\(^{-1}\)time\(^{-1}\))

2.2.1 Algae

Algae plays a very important role in this model. It generates oxygen through photosynthesis process, and consumes oxygen by producing organic carbon via metabolism and predation. Three types of algae can be simulated by the EFDC water quality model, namely cyanobacteria (blue-green algae), di-
atoms and green algae, and they are denoted by the subscript $x$ in Eq. (2.17). In this project, only cyanobacteria will be simulated. Processes included in the algae model are

- growth (production)
- basal metabolism
- predation
- settling
- external loads

These processes are described in Eq. (2.17):

$$\frac{\partial}{\partial t}B_x = (P_x - BM_x - PR_x)B_x + \partial_z(WS_x \cdot B_x) + \frac{WB_x}{V} \tag{2.17}$$

where

$B_x =$ algal biomass of algal group $x$ (g C m$^{-3}$)
$t =$ time (day)
$P_x =$ production rate of algal group $x$ (day$^{-1}$)
$BM_x =$ basal metabolism rate of algal group $x$ (day$^{-1}$)
$PR_x =$ predation rate of algal group $x$ (day$^{-1}$)
$WS_x =$ settling velocity of algal group $x$ (m day$^{-1}$)
$WB_x =$ external loads of algal group $x$ (g C day$^{-1}$)
$V =$ cell volume (m$^3$)

The production of the algae is affected by their maximum growth rate, availability of nutrient (ammonia, nitrate and phosphorus), light intensity, water temperature and salinity. The effects of these processes are considered multiplicative as shown in Eq. (2.18). Algal metabolism consists of two parts: respiration and excretion, and algal matters are returned to dissolved organic and inorganic pools in the environment after the metabolism process. It is also generally considered to be an exponentially increasing function of temperature as indicated in Eq. (2.19). Algal predation is shown in Eq. (2.20) assumes a constant predation rate. The difference between predation and basal metabolism is that the end products of the predation process is mainly particulate form of organic and inorganic matter.
\[ P_x = PM_x \cdot f_1(N) \cdot f_2(I) \cdot f_3(T) \cdot f_4(S) \] (2.18)

\( PM_x \) = maximum growth rate under optimal conditions for algal group x (day\(^{-1}\))
\( f_1(N) \) = effect of suboptimal nutrient concentration (0 \( \leq \) \( f_1 \) \( \leq \) 1)
\( f_2(I) \) = effect of suboptimal light intensity (0 \( \leq \) \( f_2 \) \( \leq \) 1)
\( f_3(T) \) = effect of suboptimal temperature (0 \( \leq \) \( f_3 \) \( \leq \) 1)
\( f_4(S) \) = effect of salinity on cyanobacteria growth (0 \( \leq \) \( f_4 \) \( \leq \) 1)

\[ BM_x = BMR_x \cdot \exp(KTB_x[T - TR_x]) \] (2.19)

\( BMR_x \) = basal metabolism rate at TR\(_x\) for algal group x (day\(^{-1}\))
\( KTB_x \) = effect of temperature on metabolism for algal group x (\( ^\circ C^{-1} \))
\( TR_x \) = reference temperature for basal metabolism for algal group x (\( ^\circ C \))

\[ PR_x = PRR_x \cdot \exp(KTB_x[T - TR_x]) \] (2.20)

\( PRR_x \) = predation rate TR\(_x\) for algal group x (day\(^{-1}\))

2.2.2 Organic Carbon

The current model has three organic carbon pools: refractory particulate, labile particulate, and dissolve.

**Particulate Organic Carbon**

The difference between labile and refractory particulate is the decomposition rate. Labile particulate organic carbon decompose in days to weeks while refractory particulate organic carbon takes longer-than-weeks to decompose. The sources and sinks for the labile and refractory particulate organic carbon are:

- algal predation
- dissolution to dissolved organic carbon
settling

- external loads

Eq. (2.21) and Eq. (2.22) are the governing equations for refractory and labile particulate organic carbon.

\[
\begin{align*}
\partial_t RPOC &= \sum_{x=c,d,g,m} FCRP \cdot PR_x \cdot B_x - K_{RPOC} \cdot RPOC \\
&\quad + \partial_z (WS_{RP} \cdot RPOC) + \frac{WRPOC}{V} \\
\partial_t LPOC &= \sum_{x=c,d,g,m} FCLP \cdot PR_x \cdot B_x - K_{LPOC} \cdot LPOC \\
&\quad + \partial_z (WS_{LP} \cdot LPOC) + \frac{WLPOC}{V}
\end{align*}
\] (2.21)

\[
\begin{align*}
RPOC &= \text{concentration of refractory particulate organic carbon (g C m}^{-3}\text{)} \\
LPOC &= \text{concentration of labile particulate organic carbon (g C m}^{-3}\text{)} \\
FCRP &= \text{fraction of predated carbon produced as refractory particulate organic carbon} \\
FCLP &= \text{fraction of predated carbon produced as labile particulate organic carbon} \\
K_{RPOC} &= \text{dissolution rate of refractory particulate organic carbon (day}^{-1}\text{)} \\
K_{LPOC} &= \text{dissolution rate of labile particulate organic carbon (day}^{-1}\text{)} \\
WS_{RP} &= \text{settling velocity of refractory particulate organic carbon (m day}^{-1}\text{)} \\
WS_{LP} &= \text{settling velocity of labile particulate organic carbon (m day}^{-1}\text{)} \\
WRPOC &= \text{external loads of refractory particulate organic carbon (g C day}^{-1}\text{)} \\
WLPOC &= \text{external loads of labile particulate organic carbon (g C day}^{-1}\text{)}
\]

**Dissolved Organic Carbon**

Sources and sinks for the dissolved organic carbon model are:

- algal excretion and predation
- dissolution from refractory and labile particulate organic carbon
- heterotrophic respiration of dissolved organic carbon (decomposition)
• denitrification
• external loads

The kinetic equation for these processes is described in Eq. (2.23).

$$\partial_t DOC = \sum_{x=c,d,g,m} \left( \left[ FCD_x + (1 - FCD_x) \frac{KHR_x}{KHR_x + DO} \right] + FCDP \cdot PR_x \right) \cdot B_x + K_{RPOC} \cdot RPOC + K_{LPOC} \cdot LPOC - K_{HR} \cdot DOC - Denit \cdot DOC + \frac{WDOC}{V}$$  \hspace{1cm} (2.23)

$DOC =$ concentration of dissolved organic carbon (g C m$^{-3}$)
$FCD_x =$ fraction of basal metabolism exuded as DOC at infinite DO concentration
$KHR_x =$ half-saturation constant of DO for algal DOC excretion (g O$_2$m$^{-3}$)
$DO =$ dissolved oxygen concentration (g O$_2$m$^{-3}$)
$FCDP =$ fraction of predated carbon produced as dissolved organic carbon
$K_{HR} =$ heterotrophic respiration rate of dissolved organic carbon (day$^{-1}$)
$Denit =$ denitrification rate (day$^{-1}$)
$WDOC =$ external loads of dissolved organic carbon (g C day$^{-1}$)

2.2.3 Phosphorus

Three organic forms of phosphorus (refractory particulate, labile particulate, and dissolved) and one inorganic form of phosphorus (total phosphate) are included in the phosphorus model.

Particulate Organic Phosphorus

The source/sink terms included for the refractory and labile particulate organic phosphorus model are:

• algal basal metabolism and predation
• hydrolysis to dissolved organic phosphorus
• settling
• external loads
The kinetic equations for refractory and labile particulate organic phosphorus are:

\[
\partial_t RPOP = \sum_{x=c,d,g,m} (FPR_x \cdot BM_x + FPRP \cdot PR_x) APC \cdot B_x - K_{RPOP} \cdot RPOP \\
\quad \cdot RPOP + \partial_z (WS_{RP} \cdot RPOP) + \frac{WRPOP}{V} \tag{2.24}
\]

\[
\partial_t LPOP = \sum_{x=c,d,g,m} (FPL_x \cdot BM_x + FPLP \cdot PR_x) APC \cdot B_x - K_{LPOP} \cdot LPOP \\
\quad \cdot LPOP + \partial_z (WS_{LP} \cdot LPOP) + \frac{WLPOP}{V} \tag{2.25}
\]

\begin{align*}
RPOP &= \text{concentration of refractory particulate organic phosphorus (g P m}^{-3}) \\
LPOP &= \text{concentration of labile particulate organic phosphorus (g P m}^{-3}) \\
FPR_x &= \text{fraction of metabolized phosphorus produced as RPOP} \\
FPL_x &= \text{fraction of metabolized phosphorus produced as LPOP} \\
FPRP &= \text{fraction of predated phosphorus produced as RPOP} \\
FPLP &= \text{fraction of predated phosphorus produced as LPOP} \\
APC &= \text{mean algal phosphorus-to-carbon ratio for all algal groups (g P per g C)} \\
K_{RPOP} &= \text{hydrolysis rate of refractory particulate organic phosphorus (day}^{-1}) \\
K_{LPOP} &= \text{hydrolysis rate of labile particulate organic phosphorus (day}^{-1}) \\
WRPOP &= \text{external loads of refractory particulate organic phosphorus (g P day}^{-1}) \\
WLPOP &= \text{external loads of labile particulate organic phosphorus (g P day}^{-1})
\end{align*}

**Dissolved Organic Phosphorus**

Sources and sinks for dissolved organic phosphorus includes:

- algal basal metabolism and predation
- hydrolysis from refractory and labile particulate organic phosphorus
- mineralization to phosphate phosphorus
- external loads
The kinetic equation for dissolved organic phosphorus is:

\[
\frac{\partial DOP}{\partial t} = \sum_{x=c,d,g,m} (FDP_x \cdot BM_x + FPDP \cdot PR_x)APC \cdot B_x + K_{RPOP} \\
\cdot RPOP + K_{LPOP} \cdot LPOP - K_{DOP} \cdot DOP + \frac{WDOP}{V} \tag{2.26}
\]

\(DOP\) = concentration of dissolved organic phosphorus \((g \text{ P m}^{-3})\)
\(FPD_x\) = fraction of metabolized phosphorus produced as dissolved organic phosphorus
\(FPDP\) = fraction of predated phosphorus produced as dissolved organic phosphorus
\(K_{DOP}\) = mineralization rate of dissolved organic phosphorus \((\text{day}^{-1})\)
\(WDOP\) = external loads of dissolved organic phosphorus \((g \text{ P day}^{-1})\)

**Total Phosphate**

The total phosphate model includes dissolved and sorbed phosphate, and the sources and sinks terms are:

- algal basal metabolism, predation, and uptake
- mineralization from dissolved organic phosphorus
- settling of sorbed phosphate
- sediment-water exchange of dissolved phosphate for the bottom layer only
- external loads

The kinetic equation for total phosphate is:

\[
\frac{\partial PO_4}{\partial t} = \sum_{x=c,d,g,m} (FPI_x \cdot BM_x + FPIP \cdot PR_x - P_x)APC \cdot B_x + K_{DOP} \\
\cdot DOP + \frac{\partial}{\partial z}(WS_{TSS} \cdot PO_4) + \frac{BFPO_4d}{\Delta z} + \frac{WPO_4t}{V} \tag{2.27}
\]
\[ PO4t = \text{total phosphate (g P m}^{-3}\text{)} \]
\[ PO4d = \text{dissolved phosphate (g P m}^{-3}\text{)} \]
\[ PO4p = \text{particulate phosphate (g P m}^{-3}\text{)} \]
\[ FPI_x = \text{fraction of metabolized phosphorus produced as inorganic phosphorus} \]
\[ FPIP = \text{fraction of predated phosphorus produced as inorganic phosphorus} \]
\[ WSTS = \text{settling velocity of suspended solid (m day}^{-1}\text{)} \]
\[ BFPO4d = \text{sediment-water exchange flux of phosphate (g P m}^{-2}\text{day}^{-1}\text{)} \]
\[ WPO4t = \text{external loads of total phosphate (g P day}^{-1}\text{)} \]

### 2.2.4 Nitrogen

Three organic forms of nitrogen (refractory particulate, labile particulate, and dissolved) and two inorganic forms of nitrogen (ammonium and nitrate) are included in the phosphorus model.

**Particulate Organic Nitrogen**

The source/sink terms included for the refractory and labile particulate organic nitrogen model are:

- algal basal metabolism and predation
- hydrolysis to dissolved organic nitrogen
- settling
- external loads

The kinetic equations for refractory and labile particulate organic nitrogen are:

\[
\frac{\partial}{\partial t} \text{RPON} = \sum_{x=c,d,g,m} (FN_{Rx} \cdot BM_x + FN_{RP} \cdot PR_x) \cdot ANC_x \cdot B_x - K_{RPON} \\
\quad \cdot \text{RPON} + \partial_z (WS_{RP} \cdot \text{RPON}) + \frac{WR_{PON}}{V} \tag{2.28}
\]

\[
\frac{\partial}{\partial t} \text{LPON} = \sum_{x=c,d,g,m} (FN_{Lx} \cdot BM_x + FN_{LP} \cdot PR_x) \cdot ANC_x \cdot B_x - K_{LPON} \\
\quad \cdot \text{LPON} + \partial_z (WS_{LP} \cdot \text{LPON}) + \frac{WL_{PON}}{V} \tag{2.29}
\]
\[ \text{RPON} = \text{concentration of refractory particulate organic nitrogen (g N m}^{-3}) \]
\[ \text{LPON} = \text{concentration of labile particulate organic nitrogen (g N m}^{-3}) \]
\[ F_{NR_x} = \text{fraction of metabolized nitrogen produced as RPON} \]
\[ F_{NL_x} = \text{fraction of metabolized nitrogen produced as LPON} \]
\[ F_{NR_P} = \text{fraction of predated nitrogen produced as RPON} \]
\[ F_{NL_P} = \text{fraction of predated nitrogen produced as LPON} \]
\[ \text{ANC}_x = \text{nitrogen-to-carbon ratio in algal group x (g N per g C)} \]
\[ K_{RPON} = \text{hydrolysis rate of refractory particulate organic nitrogen (day}^{-1}) \]
\[ K_{LPON} = \text{hydrolysis rate of labile particulate organic nitrogen (day}^{-1}) \]
\[ W_{RPON} = \text{external loads of refractory particulate organic nitrogen (g P day}^{-1}) \]
\[ W_{LPON} = \text{external loads of labile particulate organic nitrogen (g P day}^{-1}) \]

**Dissolved Organic Nitrogen**

Sources and sinks for dissolved organic nitrogen includes:

- algal basal metabolism and predation
- hydrolysis from refractory and labile particulate organic nitrogen
- mineralization to ammonium

The kinetic equation for dissolved organic nitrogen is:

\[
\partial_t DON = \sum_{x=c,d,g,m} (F_{ND_x} \cdot BM_x + F_{NDP} \cdot PR_x) \text{ANC}_x \cdot B_x + K_{RPON} \cdot RPON + K_{LPON} \cdot LPON - K_{DON} \cdot DON + \frac{W_{DON}}{V} \quad (2.30)
\]

\[ DON = \text{concentration of dissolved organic nitrogen (g N m}^{-3}) \]
\[ F_{ND_x} = \text{fraction of metabolized nitrogen produced as dissolved organic nitrogen} \]
\[ F_{NDP} = \text{fraction of predated nitrogen produced as dissolved organic nitrogen} \]
\[ K_{DON} = \text{mineralization rate of dissolved organic nitrogen (day}^{-1}) \]
\[ W_{DON} = \text{external loads of dissolved organic nitrogen (g N day}^{-1}) \]

**Ammonium Nitrogen**

Sources and sinks for ammonia nitrogen includes:
• algal basal metabolism, predation, and uptake
• mineralization from dissolved organic nitrogen
• nitrification to nitrate
• sediment-water exchange for the bottom layer only
• external loads

The kinetic equation for ammonia nitrogen is:

$$\partial_t NH_4 = \sum_{x=c,d,g,m} (FNI_x \cdot BM_x + FNIP \cdot PR_x - PN_x \cdot P_x)ANC_x \cdot B_x$$

$$+ K_{DON} \cdot DON - Nit \cdot NH_4 + \frac{BF_{NH_4}}{\Delta z} + \frac{WNH_4}{V} \quad (2.31)$$

$FNI_x$ = fraction of metabolized nitrogen produced as inorganic nitrogen
$FNIP$ = fraction of predated nitrogen produced as inorganic nitrogen
$PN_x$ = preference for ammonium uptake by algal group x ($0 \leq PN_x \leq 1$)
$Nit$ = nitrification rate (day$^{-1}$)
$BF_{NH_4}$ = sediment-water exchange flux of ammonium (g N m$^{-2}$ day$^{-1}$)
$WNH_4$ = external loads of ammonium (g N day$^{-1}$)

**Nitrate Nitrogen**

Sources and sinks for nitrate nitrogen includes:

• algal uptake
• nitrification from nitrate
• denitrification to nitrogen gas
• sediment-water exchange for the bottom layer only
• external loads

The kinetic equation for nitrate nitrogen is:

$$\partial_t NO_3 = - \sum_{x=c,d,g,m} (1 - PN_x)P_x \cdot ANC_x \cdot B_x + Nit \cdot NH_4 + \frac{BF_{NO_3}}{\Delta z}$$

$$- ANDC \cdot Denit \cdot DOC + \frac{WN_{NO_3}}{V} \quad (2.32)$$
\( ANDC = \) mass of nitrate nitrogen reduced per mass of dissolved organic carbon oxidized
\( BFNO3 = \) sediment-water exchange flux of nitrate (g N m\(^{-2}\) day\(^{-1}\))
\( WNO3 = \) external loads of nitrate (g N day\(^{-1}\))

### 2.2.5 Dissolved Oxygen

Sources and sinks for dissolved oxygen in the water column includes:

- algal photosynthesis and respiration
- nitrification
- heterotrophic respiration of dissolved organic carbon
- oxidation of chemical oxygen demand
- surface reaeration for the surface layer only
- sediment oxygen demand for the bottom layer only
- external loads

The kinetic equation for nitrate nitrogen is:

\[
\partial_t DO = \sum_{x=c,d,g,m} \left( (1.3 - 0.3 \cdot PN_x)P_x - (1 - FCD_x) \frac{DO}{KH_{Rx} + DO} BM_x \right)
\cdot AOCR \cdot B_x - AONT \cdot Nit \cdot NH4 - AOCR \cdot K_{HR} \cdot DOC
\frac{DO}{K_{HCDO} + DO} KCOD \cdot COD + K_r (DO_s - DO) + \frac{SOD}{\Delta z}
+ \frac{WDO}{V} \tag{2.33}
\]

\( AONT = \) mass of dissolved oxygen consumed per unit mass of ammonium nitrogen nitrified
\( ANCR = \) dissolved oxygen-to-carbon ratio in respiration
\( K_r = \) reaeration coefficient applied to the surface layer (day\(^{-1}\))
\( DO_s = \) saturated concentration of dissolved oxygen (g O\(_2\) m\(^{-3}\))
\( SOD = \) sediment oxygen demand (g O\(_2\) m\(^{-2}\) day\(^{-1}\)) applied to the bottom layer
\( WDO = \) external loads of dissolved oxygen (g O\(_2\) day\(^{-1}\))
The effect of algae, respiration of carbon, and nitrification on dissolved oxygen has been explained in the previous sections, so the remainder of this section will explain the effect of surface reaeration on dissolved oxygen.

**Effect of Surface Reaeration on Dissolved Oxygen**

The reaeration rate of dissolved oxygen at the air-water interface is proportional to the oxygen gradient across the interface, which is the difference between the saturated DO concentration and the current DO concentration. The saturated concentration of DO decreases as temperature and salinity increase, and is described in Eq. (2.34)[6]

\[
DO_s = 14.5532 - 0.38217 \cdot T + 5.4258 \times 10^{-3} \cdot T^2
- CL \cdot \left(1.665 \times 10^{-4} - 5.886 \times 10^{-6} \cdot T + 9.796 \times 10^{-8} \cdot T^2\right)
\] (2.34)

\(CL\) = chloride concentration (mg/L)=S/1.80655

The reaeration coefficient includes the effect of turbulence generated by bottom friction (O’Connor and Dobbins [10]) and that by surface wind stress (Banks and Herrera [1]):

\[
K_r = \left(\frac{K_{ro}}{\sqrt{\frac{u_{eq}}{h_{eq}}} + W_{rea}}\right) \cdot \frac{1}{\Delta z} \cdot KT_{r}^{-20} \] (2.35)

where

- \(K_{ro}\) = proportionality constant =3.933 in MKS unit
- \(u_{eq}\) = weighted velocity over cross-section (m sec\(^{-1}\))
- \(h_{eq}\) = weighted depth over cross-section (m)
- \(B_o\) = width at the free surface (m)
- \(W_{rea}\) = wind-induced reaeration (m day\(^{-1}\)) shown in Eq. (2.36)
- \(KT_r\) = constant for temperature adjustment of DO reaeration rate

\[
W_{rea} = 0.728 U_{w}^{1/2} - 0.317 U_{w} + 0.0372 U_{w}^2
\] (2.36)

\(U_w\) = wind speed (m sec\(^{-1}\)) at height of 10 m above surface
CHAPTER 3
HYDRODYNAMIC SIMULATION

3.1 Domain description and boundary conditions

The domain mesh used in this simulation is developed based on the Full Chicago River Domain mesh created by Sumit Sinha [14]. The current domain starts from Grand Avenue and ends at Laramie Avenue just upstream of the Stickney Water Reclamation Plant (SWRP). The two United States Geological Survey (USGS) gauging stations, Station No. 0536118 at Grand Avenue and Station No. 05536140 at Stickney provide stage and discharge measurements which make them ideal boundary conditions for this simulation. The daily discharge coming into the domain at Grand Avenue, and the gauge height measurements at Stickney between May 1st and May 31st are presented in Figure 3.1 and Figure 3.2 respectively[17]. The recorded discharge at Grand Avenue serves as one of the system inflow boundaries, while the gauge heights at Stickney serves as an open boundary condition at the end of the domain. The measurements shown in Figure 3.2 are measured based on the Chicago City Datum (CCD), which is 579 feet (176.7 m) above the mean sea level[16].

The other two boundaries of this domain are the Racine Avenue Pumping Station (RAPS), and the Chicago River Controlling Works (CRCW), which are marked in Figure 3.3. In May 2009, there were four storm events that triggered pumping activities in RAPS as shown in Figure 3.4. During those four events, RAPS served as the other inflow boundary of the system. For the rest of the simulation period, it stayed as a closed boundary condition. The CRCW on the other hand had no activities during this period of time, and hence was set as a no flow closed boundary condition.
Figure 3.1: Daily Discharge at Grand Avenue between May 1\textsuperscript{st} and May 31\textsuperscript{st}

Figure 3.2: Gauge height at Stickney between May 1\textsuperscript{st} and May 31\textsuperscript{st}
Figure 3.3: Bubbly Creek Model Domain

Figure 3.4: Pumping activities at RAPS during May 2009
3.2 Grid generation

The grid generation in the current model utilizes the same set of bathymetry data used in the previous study done by Sumit Sinha [13], and the data is presented in Figure 3.5. Based on the given bathymetry, a depth-averaged grid with sigma coordinates in the vertical direction was generated using mesh generation software Gridgen (www.pointwise.com), and plotted in Figure 3.6 to Figure 3.8. The entire mesh consists of 5961 cells in horizontal and 8 layers in vertical with average grid size of about 30 meters along the direction of the flow and 10 meters across. All 8 layers are equally distributed in the lateral direction.

Figure 3.5: Bathymetry data for the Chicago River

3.3 Hydrodynamic simulation results and validation

Initialization of the model

The validation of the hydrodynamic simulation results is achieved using the stage height measurements measured at Columbus Drive on Main Stem of the Chicago River near CRCW (USGS Station No. 05536123), and at Grand Av-
Figure 3.6: 3D grid of the entire domain

Figure 3.7: 3D grid of Main Stem
Figure 3.8: 3D grid of Bubbly Creek

enue (USGS Station No. 05536118). The comparison between the observed data and the simulated results are shown in Figure 3.9 and Figure 3.10. It is clear that the match between the observed data and the simulated data is very good throughout the entire simulation period. However, both stations are located quite upstream of the simulation domain, and are relatively far away from Bubbly Creek, which is our main area of concern. It would be great if there could be continuous stage measurement data in or close to Bubbly Creek to validate the simulated flow results, especially during a storm event when pumping activities are present.

In general, when storm events are not present, the flow velocity in the model domain is quite slow, at around 0.1 to 0.2 m/s on average. Figure 3.11 shows the flow velocity in the simulation domain 80 hours after the beginning of the simulation when no rainfall was present. It is noteworthy that two sections in this domain, Bubbly Creek and Main Stem, have extremely low flow velocities compared to the rest of the channel due to their closed boundary conditions at one end. The near stagnant stream flow would usually cause water quality issues such as depletion of DO in the water due to insufficient reaeration induced by mixing. While low DO concentration isn’t an issue in the Main Stem, most likely due to its busy boat traffic which created enough
Figure 3.9: Stage comparison at Columbus Drive

Figure 3.10: Stage comparison at Grand Avenue
Figure 3.11: Flow velocity (m/s) in the model domain 80 hours after the beginning of simulation

Figure 3.12: Flow velocity (m/s) in Bubbly Creek 140 hours after the beginning of simulation
turbulence induced reaeration, and the regular opening and closing of locks to exchange water with Lake Michigan, it is one of the, if not the most concerning problems in the Bubbly Creek. Figure 3.12 shows the flow velocity in Bubbly Creek 140 hours after the beginning of the simulation, right after the first storm event has taken place. It is clear that the pumping activities at RAPS have increased the flow velocity in Bubbly Creek from near zero to almost 0.65 m/s. Figure 3.13 shows the flow velocity in Bubbly Creek 145 hours after the beginning of the simulation, right after the first storm event has ended. The flow velocity has fallen back to near zero again.

Flow reversal phenomenon in the Chicago River during storm event found in the water quality study done by Quijano et al[11] is not present in this simulation because the gates at CRCW are closed throughout the simulation period.
Sumit Sinha [13] has done a very detailed 3D water quality analysis for the Bubbly Creek using a simplified DO-BOD model in EFDC. As the name suggests, his model had only 2 water quality constituents, dissolved oxygen and biochemical oxygen demand. The purpose of such simplification, as he stated in his paper, is because “the calibration of (the complete) model will be a time consuming and a daunting exercise” and to avoid “problems with the availability of data”. The simplification also reduces the simulation run time, and hence makes long term simulation viable.

Although the simplified model has such advantages and provides satisfactory simulation results, it does not tell the whole story of what is really going on in the system. The simulated DO results might be close to what was measured in the field, but a lot of it has to do with the calibration of water quality parameters which we do not know. For example, one of the most critical water quality parameters in the model is the sediment oxygen demand (SOD). In Sinha’s paper, the SOD in Bubbly Creek was fixed at a constant level of 3.3 g/m$^2$/day, which is significantly lower compared to the SOD values, ranging from 6.7 g/m$^2$/day to 12.1 g/m$^2$/day, measured by Waterman et al [21]. Such discrepancy should not be taken lightly, and the simplified model is unable to provide a justifiable explanation for this disparity. Therefore, a more complete water quality model is required to understand the entire processes.

The current model utilizes the EFDC water quality model which comes with 21 state variables as shown in Table 2.2. Due to the scope of this work and limited availability of measured data, the following variables are not included in this research: (2) Diatom Algae, (3) Green Algae, (16) Particulate Biogenic Silica, (17) Dissolved Available Silica, (18) Chemical Oxygen Demand (COD), and (20) Total Active Metal (TAM). The above mentioned variables are ei-
ther present at very low concentrations and have very low impact on the entire system and therefore not measured, such as silica, COD and TAM, or grouped into other terms such as algae. Raw water quality data measured by MWRDGC provides phytoplankton related measurements in terms of chlorophyll-a concentration. Since the raw data does not distinguish between different types of phytoplankton, assumptions have to be made. All phytoplankton are assumed to be cyanobacteria with chlorophyll to carbon conversion ratio (mg Chl: mg C) equal to 30\cite{19}. Other measured variables such as total organic carbon and total organic phosphorus, are divided into particulate and dissolved form based on the ratio of total suspended solids concentration versus the total dissolved solids concentration.

Before diving into model initialization and calibration, many water quality parameters in EFDC need to be setup. After collecting data from about 30 papers related to EFDC water quality modeling for rivers and lakes from all over the world, a table summarizing appropriate ranges for all of the parameters was created with the help of Zhenduo Zhu. Table 4.1 shows some of the important parameter values used in this model. Water quality parameter values within the appropriate ranges are selected as initial input parameters, and are subject to change if results are not satisfactory.

### 4.1 Water Quality Model Initialization

The MWRDGC has provided monthly water quality measurements at several stations along the Chicago River. Figure 4.1 shows the location of the stations within the model domain. Each station has at least one water quality measurement between April to May 2009. A script was written to find the closest water quality measurement station to every single cell in the model, and use the most recent water quality data available at that station as that cell’s initial condition. For stations with measurements in both April and May, linearly interpolated data will be used as the initial condition. Figure 4.2 shows some example initial conditions used in this model (all units are in mg/L).
<table>
<thead>
<tr>
<th>Parameter</th>
<th>Range</th>
<th>Value used in current model</th>
<th>Unit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ratio of anoxic to oxic respiration</td>
<td>0.5</td>
<td>0.5</td>
<td></td>
</tr>
<tr>
<td>Nitrogen-to-carbon ratio of algae</td>
<td>0.13 - 0.175</td>
<td>0.167</td>
<td>g N/g C</td>
</tr>
<tr>
<td>Dissolved oxygen-to-carbon ratio</td>
<td>2.67</td>
<td>2.67</td>
<td>g O2/g C</td>
</tr>
<tr>
<td>Mass dissolved oxygen consumed per mass ammonium nitrified</td>
<td>4.33</td>
<td>4.33</td>
<td>g O2/g N</td>
</tr>
<tr>
<td>Basal metabolism rate for cyanobacteria</td>
<td>0.01 - 0.1</td>
<td>0.01</td>
<td>1/day</td>
</tr>
<tr>
<td>Algal carbon-to-chlorophyll ratio</td>
<td>0.03 - 0.09</td>
<td>0.03</td>
<td>g C/mg Chl</td>
</tr>
<tr>
<td>Depth of maximum algal production</td>
<td>0.25 - 1</td>
<td>1</td>
<td>m</td>
</tr>
<tr>
<td>Fraction of dissolved organic carbon produced by algal metabolism</td>
<td>0 - 0.2</td>
<td>0.15</td>
<td></td>
</tr>
<tr>
<td>Fraction of labile particulate carbon produced by algal metabolism</td>
<td>0 - 0.8</td>
<td>0.65</td>
<td></td>
</tr>
<tr>
<td>Fraction of refractory particulate carbon produced by algal metabolism</td>
<td>0 - 0.5</td>
<td>0.2</td>
<td></td>
</tr>
<tr>
<td>Fraction of dissolve organic carbon produced by algal predation</td>
<td>0 - 0.6</td>
<td>0.2</td>
<td></td>
</tr>
<tr>
<td>Fraction of labile carbon produced by algal predation</td>
<td>0.12 - 0.7</td>
<td>0.5</td>
<td></td>
</tr>
<tr>
<td>Fraction of refractory carbon produced by algal predation</td>
<td>0.2 - 0.35</td>
<td>0.3</td>
<td></td>
</tr>
<tr>
<td>Dissolved organic carbon respiration rate</td>
<td>0.01 - 0.34</td>
<td>0.02</td>
<td>1/day</td>
</tr>
<tr>
<td>Dissolved organic nitrogen mineralization rate</td>
<td>0.01 - 2</td>
<td>0.08</td>
<td>1/day</td>
</tr>
<tr>
<td>Dissolved organic phosphorus mineralization rate</td>
<td>0.005 - 0.2</td>
<td>0.1</td>
<td>1/day</td>
</tr>
<tr>
<td>Background light extinction coefficient</td>
<td>0.05 - 1.4</td>
<td>0.4</td>
<td>1/m</td>
</tr>
<tr>
<td>Nitrogen half-saturation for cyanobacteria</td>
<td>0.01 - 0.2</td>
<td>0.01</td>
<td>mg/L</td>
</tr>
<tr>
<td>Half-saturation concentration of nitrate required for denitrification</td>
<td>0.1 - 1</td>
<td>0.1</td>
<td>g N/m3</td>
</tr>
<tr>
<td>Half-saturation concentration of DO required for nitrification</td>
<td>0.1 - 3</td>
<td>1</td>
<td>mg O2/L</td>
</tr>
<tr>
<td>Half-saturation concentration of NH4 required for nitrification</td>
<td>0.1 - 1</td>
<td>1</td>
<td>mg N/L</td>
</tr>
<tr>
<td>Half-saturation concentration of DO required for oxic respiration</td>
<td>0.5 - 2</td>
<td>0.5</td>
<td>g O2/m3</td>
</tr>
<tr>
<td>Phosphorus half-saturation concentration for cyanobacteria</td>
<td>0.001 - 0.005</td>
<td>0.001</td>
<td>mg P/L</td>
</tr>
<tr>
<td>Labile particulate carbon dissolution rate</td>
<td>0.01 - 0.13</td>
<td>0.13</td>
<td>1/day</td>
</tr>
<tr>
<td>Refractory particulate carbon dissolution rate</td>
<td>0 - 0.01</td>
<td>0.01</td>
<td>1/day</td>
</tr>
<tr>
<td>Minimum hydrolysis rate of labile particulate organic phosphorus</td>
<td>0.002 - 0.24</td>
<td>0.2</td>
<td>1/day</td>
</tr>
<tr>
<td>Minimum hydrolysis rate of refractory particulate organic phosphorus</td>
<td>0 - 0.01</td>
<td>0.01</td>
<td>1/day</td>
</tr>
<tr>
<td>Minimum hydrolysis rate of labile particulate organic nitrogen</td>
<td>0.01 - 0.12</td>
<td>0.08</td>
<td>1/day</td>
</tr>
<tr>
<td>Minimum hydrolysis rate of refractory particulate organic nitrogen</td>
<td>0.001 - 0.008</td>
<td>0.001</td>
<td>1/day</td>
</tr>
<tr>
<td>Effect of temperature on basal metabolism of algae</td>
<td>0.032 - 0.069</td>
<td>0.069</td>
<td></td>
</tr>
<tr>
<td>Maximum nitrification rate at optimal temperature</td>
<td>0.05 - 0.5</td>
<td>0.35</td>
<td>g N/m3/day</td>
</tr>
<tr>
<td>Settling velocity for cyanobacteria</td>
<td>0.01 - 0.8</td>
<td>0.25</td>
<td>m/day</td>
</tr>
<tr>
<td>Settling velocity for organic particle</td>
<td>0.03 - 1</td>
<td>0.2</td>
<td>m/day</td>
</tr>
</tbody>
</table>
Figure 4.1: MWRDGC Stations with monthly water quality measurements

Figure 4.2: Example Water Quality Initial Conditions for May 1st 2009
4.2 Water Quality Model Boundary Conditions

There are two water quality inflow boundaries, one located at Grand Avenue which is the starting point of the model domain, and the other one is at RAPS. The water quality data used for the Grand Avenue boundary is from WW46 Station, located just downstream of Grand Avenue as shown in Figure 4.1. Since only three measurements are available at Kinzie station in the months of April, May and June respectively, daily input data are linearly interpolated based on these three measurements. The input loads are then evenly distributed across the width and depth of the cross-section. There are no measurement data available for RAPS during or near the simulation period. Therefore, mean historical values measured at RAPS are used as shown in Table 4.2, and the concentrations remain constant throughout the simulation period [11].

<table>
<thead>
<tr>
<th>NH3 (mg/l)</th>
<th>NO3 (mg/l)</th>
<th>DON (mg/l)</th>
<th>IP (mg/l)</th>
<th>DOP (mg/l)</th>
<th>CBOD (mg/l)</th>
<th>DO (mg/l)</th>
<th>PHY (mg/l)</th>
<th>FC (cfu/100ml)</th>
</tr>
</thead>
<tbody>
<tr>
<td>RAPS 2.69</td>
<td>0.62</td>
<td>8.16</td>
<td>1.25</td>
<td>1.25</td>
<td>71.67</td>
<td>4.5</td>
<td>0</td>
<td>160.000</td>
</tr>
</tbody>
</table>

Table 4.2: Mean historic concentrations measured at Racine Ave Pumping Station

Notice the carbon related concentration is measured in terms of CBOD. This is converted to organic carbon by dividing the CBOD concentration by the oxygen-to-carbon mass ratio (=2.67 g O_2/g C)[4]. It is also assumed that half of the organic carbon input is in dissolved form and half in particulate form because the high discharge velocity would usually scour organic carbon residue inside the pipes. The SOD in the Bubbly Creek is set to equal to 8 g/m^2/day, which falls within the range provided by Waterman et al[21], and the SOD in the main channel is set equal to 1 g/m^2/day.

4.3 Calibration Parameter

With most of the water quality measurement data available only once per month, it would be impossible to calibrate the parameters based on the sparse sets of data points available within the 1-month-long simulation period. Luckily, continuous DO measurements are available at measuring sta-
tions indicated in Figure 4.3. Therefore the goal of this calibration process is to try to match the simulated DO results with the continuous DO measurements available. The simulation results will be presented in the next section.

Before starting the calibration process, it is worth mentioning that there are over a hundred water quality parameters that could be calibrated, and changing those parameters one by one would be almost an impossible task to complete. Therefore, it is necessary to pinpoint key parameters that have the most significant impact on the DO concentration and perform a primary calibrations before carrying on with more detailed calibration processes in the future. By looking at Eq. (2.33) which is also shown below, one could easily see that the sink terms for DO are algae metabolism, nitrification of ammonia, DOC respiration, COD and SOD. Since COD is not present in the current model, it got moved out of the equation.
\[ \frac{\partial_t DO}{x=c,d,g,m} = \sum \left( (1.3 - 0.3 \cdot PN_x)P_x - (1 - FCD_x) \frac{DO}{KHR_x + DO}BM_x \right) \]
\[ \cdot \frac{DO}{KH_{CDO} + DO} KCO \cdot COD + K_r (DO_s - DO) + \frac{SOD}{\Delta z} \]
\[ + \frac{WDO}{V} \]

Ammonia’s concentration in the river is much lower compared to that of organic carbon or phytoplankton, usually about an order of a magnitude. Even though it would support the growth of phytoplankton, and in turn affect DO through phytoplankton’s metabolism, its direct impact on DO is relatively low and hence would not be considered in the primary calibration. The three major sink terms left are the phytoplankton, DOC, and SOD. The primary calibration would focus on parameters related to these three variables.

4.4 DO Results

The simulated DO results are exported from the model at locations same to where the MWRDGC measurement stations are located. The DO results comparison in the model domain are shown in the following figures.

![Figure 4.4: DO Comparison at 36th St for May 2009](image-url)
Figure 4.5: DO Comparison at I55 for May 2009

Figure 4.6: DO Comparison at Clark St for May 2009
Figure 4.7: DO Comparison at Kinzie St for May 2009

Figure 4.8: DO Comparison at Loomis St for May 2009
At first glance at Figure 4.4, DO simulation seem to match with the measurement data pretty well. When the storm events started, DO level rose immediately due to loads coming from the pumping discharge, and when the storm had passed, it fell back quickly due to high sediment oxygen demand and high level of organic carbon in the water column.

However, during the period of 22\textsuperscript{nd} to 26\textsuperscript{th} of May, the simulation results do not seem to capture the sudden increase in DO concentrations. Upon further investigation of the data, the measurements during this period seemed questionable. First, by looking at the RAPS discharge data shown in Figure 3.4, the pumping activity only started on the 26\textsuperscript{th} day. Before that, there were no pumping activities at all. The DO concentration in the Bubbly Creek is usually close to 0 when there are no pumping activities, and the discharge coming from RAPS is the only source of DO that’s large enough to raise the DO concentration in the creek. Second, by looking at the DO measurements at the nearest measuring station in Bubbly Creek, Station I55 in Figure 4.5, it is clear that although there was a peak on the 21\textsuperscript{st} day, the DO level declined steadily to zero afterwards instead of increasing as shown in Figure 4.4. The peak is more likely to be caused by the interaction with the main channel due to Station I55’s close proximity to the turning basin. Figure 4.10 shows the DO concentration and Figure 4.11 shows the DOC and LPOC concentrations in Bubbly Creek on 22\textsuperscript{nd} May. It is clear that
the organic carbon concentration is lower closer to the turning basin because of diffusion due to larger concentration gradient between the two locations. Since organic carbon is a key sink term for DO, this causes the DO level at I55 to be higher. As a result, this brings the question of what caused this discrepancy between the simulation results and the measurement data. Is it due to unrecorded flow that had changed the DO concentration or is it due to measurement error? Looking at the historical DO concentrations at 36th St shown in Figure 4.12, it is not difficult to see that there are many absurdly high DO measurements in the range of 15 to 27 mg/L. These values are way above the common saturated DO concentration in rivers and lakes. Therefore, it would be reasonable to question the accuracy of the measurement data. One possible cause for the erroneous measurements is probably the “bubbling nature” of Bubbly Creek. As the organic matter in the sediment continuous to decompose, methane and hydrogen sulfide gases are released. When these bubbles come in contact with the measuring probe, they might cause the readings to go wild. However, this is just a hypothesis. In order to understand the true reasons behind this discrepancy, the best way would be to deploy more measuring probes in the area and compare the results among different probes. Despite this mismatch between the simulation and field data, the rest of the results seem to fit pretty well.

Results from I55 also look satisfactory, capturing most of the peaks during storm events pretty well. The under-prediction of DO during no flow period was probably due to the fact that the SOD was set to be quite high. As mentioned earlier, the study done by Waterman et al[21] suggests the appropriate SOD value to be from 6.7 to 12.1 g/m²/day. However, the underlying assumption he made in this study was that unlimited amount of oxygen was provided so the organic matter could fully decompose. In the case of a storm event, when large amount of almost saturated discharge enters the system, this is not a bad assumption because DO is not the limiting factor. During no flow period however, this assumption would no longer be valid. The limited amount of DO would reduce the amount of SOD, and DO would not decrease at such a high rate. This is a limitation of the current EFDC model because it could only take SOD as constant values or a time series. In order to dynamically calculate SOD based on other water quality parameters, implementation of sediment diagenesis model is required, but that would be
out of the scope of this thesis.

Simulation results at Clark Street in the Main Stem do not seem to match with the measurements. This is expected because in the numerical model, the
Main Stem behaved somewhat like the Bubbly Creek during the simulation period. Because the gates at CRCW were closed throughout the simulation period, the Main Stem was isolated from the main channel with close to zero flow velocity as shown in Figure 4.13. The stagnant flow condition would drastically reduce the reaeration rate according to Eq. (2.35). In real life however, the DO concentration in the Main Stem is greatly influenced by human activities. Due to busy boat traffic throughout the day, the surface of Main Stem is actually very well mixed. This mixing will greatly increase the DO concentration in the water, and since the measuring probe is located only 1 meter below the water surface, it is not surprising to see near saturated DO measurements in this area. In addition, MWRDGC would also implement discretionary-diversion during summer months, to improve the water quality condition in Chicago River by bringing in cleaner water from Lake Michigan. If the operation data could be made available, this would greatly improve the simulation results in Main Stem.

Kinzie street is located very close to the starting point of the model, and the entry boundary condition uses the continuous DO data from the Kinzie Station and other water quality measurements from WW46. Due to its close proximity to the boundary condition, the simulation results at Kinzie are expected to fit the measurement data very well, and it is indeed the case here.
Figure 4.13: Flow velocity in Main Stem 80, 160, 240 and 320 hrs after the start of simulation
as shown in Figure 4.7. There are however, occasional slight disagreements between the observed data and the simulated results. The main cause for this is probably due to the lack of continuous measurements for other water quality variables. As mentioned in Section 4.2, water quality measurements were taken once a month, and the input boundary condition for the numerical model was linearly interpolated based on the only 2 or 3 measurement data available. This does not capture the fluctuation of various water quality variables in the water column, and would definitely impact the DO results at Kinzie to some extent. Luckily, the overall reaction rate between water quality variables are much lower compare to their advection rate, and hence the general trend of DO could still be captured very well.

Figure 4.8 and Figure 4.9 show the DO results at Loomis St Station and Cicero Ave Station. The agreement at Loomis looks very nice, and the general trend of variation is captured by the model. Results at Cicero Ave are acceptable. The model is able to capture the average values of DO fluctuation, but is not able to match the detailed ups and downs. This is understandable considering that the input data available are at a very low resolution. A more detailed, maybe daily water quality measurement might improve model’s ability to capture the DO trend. The discrepancy could also be partly due to the influence from SWRP right downstream from where the data was collected. A larger model domain including the SWRP might help to visualize the impact from SWRP’s effluent.

Overall, the current EFDC model is capable of capturing the characteristics of DO in most part of the domain pretty well. The simulated DO results have a better match with the measured data in most part of the channel compared to Sinha’s simplified BOD-DO model, especially in the Bubbly Creek. Although this comes at the cost of longer computational time, the true advantage of the complete eutrophication model is that it could simulate other water quality state variables, and provide the user a more thorough understanding of what is going on in the water. Of course such advantage is based on the assumption that the model could indeed simulate other variables with reasonable accuracy. The simulation results for the other variables will be presented in the next section.
4.5 Water Quality Results at Monitoring Stations

The simulated results for other water quality state variables are also exported from the model at locations same to where the water quality monitoring stations are located. The water quality results for different state variables at each station are shown in the following figures. Although only one measurement point is available for each station for the entire simulation period, the purpose of this comparison is to check whether model results are within an acceptable range from the measurement data. A much longer simulation period, at least one year, is required to provide more detailed information about the accuracy of the results simulated by the numerical model.

STATION I55

Simulation results shown in Figure 4.14 to Figure 4.19 at Station I55 are generally quite good and are not far off from the measurement data. The model is underestimating phytoplankton concentration by half, and this is partially caused by the discharge coming from RAPS. There are four sudden reductions in phytoplankton concentration in Figure 4.14 and they correspond to the four storm events happened in May. Due to the lack of available water quality data in the discharge coming from RAPS, historical mean values were used as fixed concentrations for the effluent. The phytoplankton concentration in the effluent was assumed to be 0 as indicated in Table 4.2. Therefore, during storm events, phytoplankton that was present in the water will be washed out, and cause the underestimation. The lower level of phytoplankton will lead to lesser nutrient uptake, and this is reflected in the overestimation of ammonia and total phosphorus in Figure 4.15 and Figure 4.19.

CLARK ST

Results from Clark St shown from Figure 4.20 to Figure 4.25 are less satisfactory. The model overall overestimate almost all state variables except for phytoplankton, but the results are still within the same order of magnitude as the measurement data. As mentioned in the previous section, MWRDGC will implement discretionary-diversion during summer months. The pristine water coming from the lake might lead to lower measurement values.
Figure 4.14: Phytoplankton results at Station I55

Figure 4.15: Ammonia results at Station I55
Figure 4.16: Nitrate results at Station I55

Figure 4.17: Organic Nitrogen results at Station I55
Figure 4.18: Total Organic Carbon results at Station I55

Figure 4.19: Total Phosphorous results at Station I55
Figure 4.20: Phytoplankton results at Clark St

Figure 4.21: Ammonia results at Clark St
Figure 4.22: Nitrate results at Clark St

Figure 4.23: Organic Nitrogen results at Clark St
Figure 4.24: Total Organic Carbon results at Clark St

Figure 4.25: Total Phosphorous results at Clark St
KINZIE ST

Results at Kinzie are expected to match with the measurement data very well because Kinzie is close to the inflow boundary. Figure 4.26 to Figure 4.31 show that the fit is indeed very nice.

LOOMIS ST

Results at Loomis St are also very satisfactory as shown from Figure 4.32 to Figure 4.37. It is worth noting that the results at Loomis follow a similar pattern to the results at Kinzie. It seems that RAPS’ discharge was unable to extended its impact to Loomis St during this simulation period. Similar with the previous stations, the model is underestimating phytoplankton.

CICERO AVE

Results at Cicero Ave are also very decent falling within the same order of magnitude as shown from Figure 4.38 to Figure 4.43. It can be clearly seen that the impacts from RAPS’ discharge were carried all the way to the end of the modeling domain. The first three storm events have lower peaks compared to the last one, because the discharge from RAPS was the highest
Figure 4.27: Ammonia results at Kinzie St

Figure 4.28: Nitrate results at Kinzie St
Figure 4.29: Organic Nitrogen results at Kinzie St

Figure 4.30: Total Organic Carbon results at Kinzie St
Figure 4.31: Total Phosphorous results at Kinzie St

Figure 4.32: Phytoplankton results at Loomis St
Figure 4.33: Ammonia results at Loomis St

Figure 4.34: Nitrate results at Loomis St
Figure 4.35: Organic Nitrogen results at Loomis St

Figure 4.36: Total Organic Carbon results at Loomis St
for the last storm event, and the dilution was less significant.

**SUMMARY**

Overall, the model results at each monitoring station appear to be well within the acceptable range of the measurement data. It should be noticed however, that the current model is underestimating phytoplankton concentrations at all stations. The cause of this is very likely due to certain assumptions made in the current model. Due to time constraint and the scope of this study, some simplification of the model has to be made. One assumption that might have the greatest impact on the dynamics of phytoplankton would be the assumption of constant temperature and solar radiation. The current model assumes that algae will receive constant sunlight for half of the time. During “daytime”, growth will be dominating, and during “nighttime”, growth will stop and respiration will dominate. This trend can be observed from the diurnal fluctuation pattern found in Figure 4.26. For summer months, this assumption would underestimate the amount of sunlight received by the algae. According to U.S. Naval Observatory[18], the daylight length in May 2009 for Chicago ranges from 14 hrs to 15 hrs, which is much longer than the assumed daylight time. Furthermore, solar radiation is not constant through-
Figure 4.38: Phytoplankton results at Cicero Ave

Figure 4.39: Ammonia results at Cicero Ave
Figure 4.40: Nitrate results at Cicero Ave

Figure 4.41: Organic Nitrogen results at Cicero Ave
Figure 4.42: Total Organic Carbon results at Cicero Ave

Figure 4.43: Total Phosphorous results at Cicero Ave
out daytime, but greatest during the middle of the day. In order to capture
the variation in solar radiation, the thermal dynamics model for EFDC will
need to be incorporated.

4.6 Water Quality Results for the Entire Domain

Another advantage of the complete eutrophication model is that it is ca-
39 pable of tracking the extend of environmental impact to the entire system
due to each storm event. Figure 4.44 to Figure 4.48 capture the movement
of various water quality state variables in the system before and after the
first storm event which occurred on the 140th hr after the start of the sim-
ulation. It is clear that pollutants coming out of the creek will be carried
away and reach the downstream end of the domain in about 50 hours. The
high concentration pollutants leaving the creek will be rapidly diluted and
dispersed in the main channel to form a relatively lower concentrated plume.
Immediately after the storm event has ended, the pollutants still present in
the creek will remain there and consume all the DO that’s left, while the
plume keeps traveling downstream. From Figure 4.48, it is not difficult to
observe that the reduction in DO due to pollutants can extend all the way
to the end of the model domain, and the DO concentration at Stickney has
been lowered by 1 to 2 mg/L. Keep in mind that this is not a large storm
event. More severe storm events, such as the September 2008 event modeled
by Quijano[11], will bring more pollutants to the Chicago River, and the
flow reversal due to the opening of the gates at CRCW will bring all the
pollutants to Lake Michigan.
Figure 4.44: DOC concentration in the domain before and after the first storm event
Figure 4.45: LPOC concentration in the domain before and after the first storm event
Figure 4.46: Phytoplankton concentration in the domain before and after the first storm event
Figure 4.47: Ammonia concentration in the domain before and after the first storm event
Figure 4.48: DO concentration in the domain before and after the first storm event
5.1 Summary

In this project, a high resolution, three-dimensional eutrophication model for the Chicago Area Waterways System (CAWS) is developed to simulate and capture the interactions between various water quality state variables inside the system. The model domain covers the reach of waterways starting from Grand Avenue on the north, and ending at Laramie Avenue on the south. Bubbly Creek and the Main Stem are also included in the model domain. The model is based on the Environmental Fluid Dynamics Code (EFDC) developed by Dr. John M. Hamrick, and is considered an improvement upon the simplified BOD-DO model developed by Dr. Sumit Sinha. The hydrodynamics simulation results are calibrated and validated with the help of the discharge and stage measurement data provided by the U.S. Geological Survey gauging stations at Grand Avenue and Stickney. The water quality simulation results are calibrated and validated with the help of the continuous DO measurement data provided by the Metropolitan Water Reclamation District of Greater Chicago (MWRDGC).

In general, the model is capable of capturing DO characteristics with great accuracy in most part of the domain, and the other water quality results are also very close to, or within the same order of magnitude with the measurement results. Simulation results in Bubbly Creek, the area of interest of this model, match very nicely with the measurement data, and clearly demonstrate the CSO-dominant nature of the water quality in the stream. Results in Main Stem are less satisfactory due to unknown human activities. Time series plots of the results in the entire domain show the movement of contaminants coming out of RAPS during storm events, and the pollutants
can travel all the way to the downstream boundary at Stickney and cause reduction of DO in the water column. After the storm event has passed, highly concentrated pollutants still present in Bubbly Creek will be stuck there, and consume any DO left in the water column.

The ability offered by this model to simulate and capture environmental impacts coming from Bubbly Creek during storm events will be very useful in determining what precautions to take when storm events hit. The model could also be helpful in deciding what remediation strategies to use to help restore the water quality condition in Bubbly Creek. Compared with the simplified BOD-DO model, the eutrophication model offers the advantage of simulating the transport and fate of algae, which is one of the key factors that influence the DO dynamics in water column. In addition, other water quality state variables simulated by the eutrophication model could provide users the ability to assess Chicago River’s contribution to the algal bloom at Gulf of Mexico.

5.2 Model Limitations and Future Work

The ability to simulate more water quality constituents comes at the cost of much higher computational time required. In the current model, with 5961 cells in horizontal and 8 layers in vertical, the total computational time required to run a month-long simulation with 4 processors running simultaneously is about 24 hours. The simplified BOD-DO model with similar mesh size would only take 13 to 14 hours of computational time with 4 processors running at the same time[13]. This is an important trade-off that needs to be considered if long term simulations are desired.

In addition to the longer computational time, it is also much more difficult to calibrate the three-dimensional eutrophication model since so many variables are inter-connected. This study has only calibrated the key parameters that have the greatest impact on the whole system based on the formulas used in the numerical model, but the calibration process has already become the most time consuming part of the entire project.
As mentioned in the paper, due to constraints in time and availability of data, some assumptions and simplifications have to be made. A dry weather scenario is chosen to neglect the need to consider CSO inflows into the system, and both the thermal dynamics model and the sediment transport model are not incorporated here. For this particular simulation event, the impact caused by the omission of these two models is not very significant, because the temperature fluctuation within one month is not very large, and the sediment transport in the Chicago River is small during dry weather conditions. However, for a long-term simulation of at least one year, these two models should be considered to capture the seasonal variation in water quality and the impacts from sediments in extreme weather events. For future work, these two models could be included, and a long-term simulation is also desired to help improving the calibration of the eutrophication model and understanding of the seasonality of the water quality state variables.
REFERENCES


