Development of A Hybrid Fuzzy-Stochastic Modeling Approach for Examining the Environmental Performance of Surface Flow Constructed Wetland

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ABSTRCT

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Ling Peng Xiao

Storm water is considered as a significant source of contaminants to receiving rivers and the constructed wetland has been used to treat storm water before the discharge. In this study, a hybrid fuzzy-stochastic modeling approach is developed to examine the wetland treatment efficiency, to analyze the environmental impact associated with the wetland effluents into the receiving water, and to quantify system uncertainties. The proposed approach first incorporates a water quality model to simulate storm water flow going through the wetland and the fate and transport of nutrients in the wetland. A Monte Carlo modeling method is next developed to extend the water quality model, providing a stochastic simulation of the concentration distribution of nutrients in the wetland effluents. It is intended for the analysis of probabilistic environmental risks associated with wetland effluents on the receiving waters. The fuzzy membership functions are further used to quantify the variability or suitability of regional surface water guidelines, which is incorporated into the Monte Carlo modeling framework to identify the integrated risks from the discharge on the river.

The developed modeling approach has been applied to the Kennedale wetland, a storm water treatment system, in the city of Edmonton, Canada. Before the environmental risk assessment, the HEC-RAS (Hydrologic Engineering Centers River Analysis System) model and the QUAL2K (River and Stream Water Quality) model are applied to simulate the flow and nutrients removal efficiency in the wetland. According to the simulation results from the HEC-RAS model and the QUAL2K model, the removal efficiencies of TN (Total Nitrogen) by the wetland are 25.64% and 13.59%, respectively. The removal efficiencies of TP (Total Phosphorus) are 50% and 50.91%, respectively. The differences between the HEC-RAS simulation results and on-site field data are 0.05% for TN and 6.1% for TP. The differences between the QUAL2K simulation results and on-site field data are 13.99% for TN and 4.35% for TP based on this study. The water quality simulation results from the two models are both acceptable compared to the monitoring data. It is seen that the HEC-RAS model has better performance on modeling this field case, and is integrated with the environmental risk assessment process. Consequently, the results of the integrated risk assessment referring to different guidelines in the North America show that the concentrations of TN at the wetland discharge port have a high possibility of violating the TN guidelines in both Alberta, Canada and the US EPA (Environmental Protection Agency). Similarly, the concentrations of TP at the wetland discharge port have a high possibility to violate the Canadian and US TP guideline during this study period. Therefore, the nutrients in storm water discharges from the Kennedale wetland may have a great risk to adversely affect the receiving river (North Saskatchewan River) at the time of this study. The analysis results of nutrient guidelines have supported the management of decision making process, and the study results indicate that the developed hybrid fuzzy-stochastic modeling approach is a useful tool for the practical managing of wetland systems and the impact of the wetland discharges on the receiving waters.

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Table of Contents

List of Figures

List of Tables

List of Symbols

A	Flow area for subdivision (m^2) or cross sectional area (m^2)
A_c	Cross-sectional area (m^2)
A_s	The surface area (m^2)
a, b, α and β	Empirical coefficients that are determined from velocity-discharge and stage-discharge rating curves, respectively
a_1, a_2	Velocity weighting coefficients
α_1	Fraction of algal biomass that is nitrogen (mgN / mgA)
α_2	Fraction of algal biomass that is phosphorus (mgP / mgA)
B	Width (m)
β_2^*	Rate constant: oxidation of nitrite to nitrate (1/day)
β_4^*	Rate constant: oxidation of OrgP to PO_4 (1/day)
C	Expansion or contraction loss coefficient or concentration (mg/L)

C_s	A local environmental criterion (mg/L)
dc_i / dt	Changing rate of the concentration of an element with respect to time
d	Average channel depth (m)
$E_i^{'}$	The bulk dispersion coefficient between elements i and $i+1$ (m^3/d)
F_1	Fraction of algal uptake from ammonium pool (unitless)
f_{opp}	The fraction of the phytoplankton internal phosphorus that is in organic form
f_L(L)	The probability density function
g	Gravitational acceleration (m^2/s)
H	Flow depth (m)
h_e	Energy head loss (m)
h_{ce}	Contraction and expansion losses (m)
K	Conveyance for subdivision

KNR	First order nitrification inhabitation coefficient (L/mgO)
L	Discharge weighted reach length (m) or a random contaminant's concentration (mg/L)
L_{lob}, L_{ch}, L_{rob}	Cross section reach lengths specified for flow in the left overbank, main channel, and right overbank, respectively (m)
N (σ_x, μ_x)	A normal distribution function of σ_x and μ_x
n	Manning's roughness coefficient for subdivision
σ_1^*	Algal setting rate (m/day)
σ_5^*	Settling rate: organic phosphorus (1/day)
σ_x	The standard deviation of x (1/day^2)
P_{ab}	The coefficient of the preferences for ammonium as a nitrogen source for bottom algae
ρ^*	Algal local respiration rate (1/day)
Q	Flow rate (m^3/s)

Q_i	The outflow from element i to element $i+1$ (m^3/d)
$Q_{out,i}$	The total outflow from element due to point and nonpoint withdrawals (m^3/d)
$\overline{Q}_{lob}+\overline{Q}_{ch}+\overline{Q}_{rob}$	Arithmetic average of the flows between sections for the left overbank, main channel, and right overbank, respectively (m^3/s)
q_{Pb}	The bottom algae cell quotas of phosphorus (mgP/mgA)
q_{Pp}	The phytoplankton cell quotas of phosphorus (mgP/mgA)
ϕ	Water temperature (℃) or concentration (kg/ m^3)
R	Hydraulic radius for subdivision (m) or the risk level quantified as the probability of system failure
S	Friction slope or sources and sinks (kg/s or mg/L·s)
S_i	Sources and sinks of the constituent due to reactions and mass transfer mechanisms (g/m^{3}d or mg/ m^{3}d)
$\bar{S}_f$	Representative friction slope between two sections

t	Time (s)
Γ	User-defined dispersion coefficient (m^2/s)
U	Mean velocity (m/s)
u	Face velocity or average linear velocity (m/s)
u^*	Shear velocity (m/s)
μ	Local growth rate for algae (1/day)
μ_x	The mean value of x
V	Volume of the water quality cell (m^3)
V_1, V_2	Average velocities (m/s)
V_i	Volume of i^{th} element (m^3)
v	Average flow velocity (m/s)
W_i	The external loading of the constituent to element i (g/d or mg/d)
w	Average channel width (m)

x	A key parameter or distance (m)
Δx	The length of the element (m)
Y_1, Y_2	Depth of water at cross sections (m)
y	Average channel depth (m)
Z_1, Z_2	Elevation of the main channel inverts (m)

CHAPTER 1 INTRODUCTION

1.1 Overview

City storm and urban runoff generate large quantities of storm water. This storm water may contain a large number of contaminants, including BOD (Biochemical Oxygen Demand), SS (Suspended Solid), nutrients, heavy metals, de-icing salts, hydrocarbons and fecal coliforms, etc., which, when discharges into rivers, may have significant impacts on their ecosystems. Generally, Storm water is transferred by storm water pipes either directly to watercourses or into sustainable (urban) drainage systems such as ponds or wetlands to be treated (Scholz, 2011). In this study, storm water is treated by a surface flow constructed wetland and finally discharged into the river.

To aid in assessing the environmental risks of wastewater discharges, a few studies about numerical model application have been reported. For instance, Meinhold et al. (1996) applied the Monte Carlo simulation to evaluate the risks of radium and lead for human health in wastewater. Riddle et al. (2001) used a random walk model to calculate the distributions of dispersed oil concentration from wastewater discharges in the North Sea. Additionally, Dunn et al. (2014) applied a near-field and a far-field dispersion model in combination with a hydrodynamic model to simulate the transport and fate of treated wastewater which would be discharged into the Geographe Bay.

In many risk assessment studies, local environment guidelines or standards were applied as evaluation criteria. However, a few of those guidelines or standards are overly conservative or not strict enough (Li et al., 2008; Chen et al., 2010). An observation is that the guidelines for one water quality parameter vary widely from place to place. For instance, the surface water quality guidelines for TN is 2.2 mg/L in the Netherlands, but for the USEPA Ecoregion I guideline, it is only 0.31 mg/L (Neeteson, 2000; USEPA, 2002). Therefore, the variability of those guidelines can be further addressed.

The uncertainties inherent to the evaluation criteria, such as the contaminants' physical, chemical and toxic characteristics, and media conditions, etc., cannot be expressed as probability distributions as uncertainties of randomness do (Darbra et al., 2008). On the other hand, fuzzy logic method is widely used to quantify uncertainties related to incomplete or imprecise characteristics, such as the uncertainties inherent to evaluation criteria. It can generate acceptable quantitative results (Chen et al., 2010). For instance, fuzzy membership functions can be used to quantify the suitability associated with evaluation criteria (Chen et al., 2003).

Additionally, different types of uncertainties need to be considered when the environmental risk assessment is undertaken. The fuzzy logic approach or stochastic modeling method applied alone is not sufficient in many cases, therefore, hybrid fuzzy-stochastic modeling approaches can be further studied and developed (Chen et al., 2003).

1.2 Research Objectives

The objectives of this study are as follows:

(1) To develop a hybrid fuzzy-stochastic modeling approach for analyzing a field-scale wetland. The developed modeling approach could be used to examine the wetland treatment efficiency, to analyze the environmental impact associated with the wetland effluents into the receiving water, and to quantify system uncertainties.

(2) To incorporate and examine two water quality analysis models (the HEC-RAS model and the QUAL2K model) through the fuzzy-stochastic modeling approach for simulating storm water flow going through a field-scale surface flow constructed wetland and the transport and fate of nutrients in the wetland.

(3) To apply the developed fuzzy-stochastic modeling approach to the Kennedale wetland in the city of Edmonton, on quantifying the wetland performance, water quality parameter changes and the risks of the wetland effluents on the receiving river water under wetland design scenarios.

1.3 Organization of the Thesis

The thesis is organized in the following seven chapters:

Chapter 1 presents a general introduction about storm water discharge, the previous studies about wastewater risk assessment and problems related to them, as well as the research objectives.

Chapter 2 introduces literature review about surface flow wetland modeling studies and environmental risk assessment studies.

Chapter 3 presents the detailed methodology of the integrated hybrid fuzzy-stochastic risk assessment system.

Chapter 4 describes the information about the field case. It also gives the required input data for applying the two water quality models on the field-scale wetland case and the water quality simulation results from the two models, as well as the water quality parameter changes in wetland design scenarios predicted by the customized water quality modeling component.

Chapter 5 presents the integrated environmental assessment results from the hybrid fuzzy-stochastic approach.

Chapter 6 gives the discussions about the possible reasons of differences between the simulation results and the monitoring data, the comparison of flow simulation between two water quality models, and causes of the removal efficiency decrease.

Chapter 7 concludes the results of this thesis, presents a list of contributions and suggestions of future studies.

CHAPTER 2 LITERATURE REVIEW

2.1 Wetland Modeling Studies

This thesis focuses on the surface flow constructed wetland, so the following literature reviews are all associated with surface flow wetland modeling studies.

2.1.1 Surface flow wetland modeling

Smith (1980) defines wetlands as “a half-way world between terrestrial and aquatic ecosystem and exhibit some of the characteristics of each”. Wetlands have a few functions, such as water storage and flood mitigation, and wildlife habitat, one important of which is to increase water quality by using natural energy (sunlight), natural vegetation, without requiring the power for aerating or mixing, large amounts of human labor or chemical additions (Lin et al., 2002; Mihelcic and Zimmerman, 2010).

The constructed wetland which mimics a natural wetland has been increasingly used for the treatment of different types of wastewaters (Kotti et al., 2013). It is an alternative to traditional wastewater treatment systems, which is considered as efficient and cost-effective (Jou et al., 2012). The constructed wetland is more flexible than traditional wastewater treatment systems with respect to the geometric condition. It can be designed and built based on the geographic situation of the potential construction site (Alvarez-Cobelas et al., 2001). Classified by flow mode, constructed wetlands usually have three types: surface flow constructed wetlands, subsurface flow constructed wetlands and vertical flow constructed wetlands (Mihelcic and Zimmerman, 2010). The field case studied in this thesis is a surface flow constructed wetland. The surface flow constructed wetlands, also called free water surface constructed wetlands, are similar to natural open-water wetlands in appearance and treatment mechanisms as wastewater flows on the surface of substrates. Figure 2.1 describes a schematic diagram of surface flow constructed wetland. Most of the organic contaminants in wastewater are removed by the biofilm generated by the stems and trunks of vegetation which grow underwater (Ye, 2011).

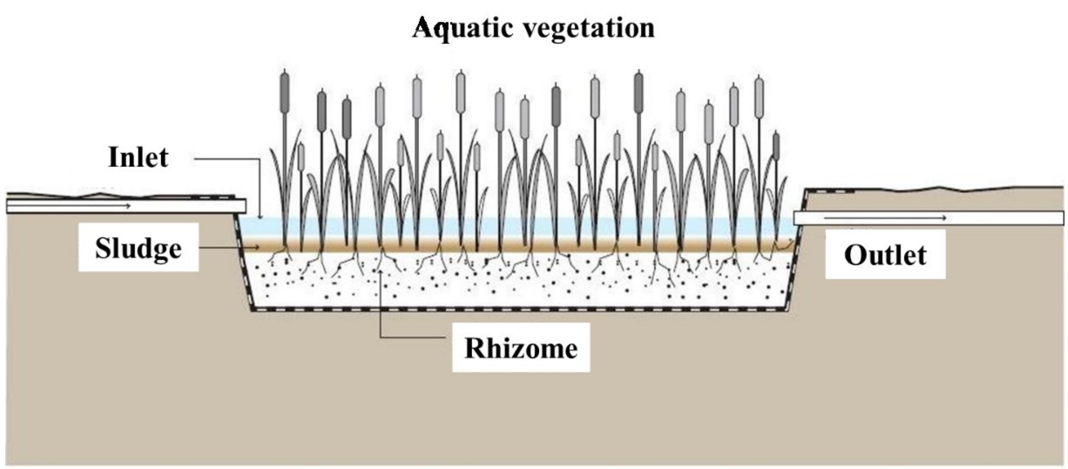

Figure 2.1. Schematic diagram of surface flow constructed wetland (Adapted from Ye, 2011)

As the application of surface flow constructed wetlands on treating wastewaters has increased, so has the development of modeling the processes in the wetland system, aiming to simulate the transport and fate of contaminants and to predict the removal performance (Kotti et al., 2013). Various categories of models with different complexity have been developed. Table 2.1 summarizes a few modeling studies of surface flow constructed wetland. Some of the wetland models are relatively simple first-order, K-C* or regression models (Rousseau et al., 2004). For instance, Jou et al. (2008) applied a first-order biokinetic model to simulate the removal performances of BOD and nitrogenous biochemical oxygen demand (NBOD) by a surface flow constructed wetland. From their studying results, the observed data of removal efficiencies fell within ranges of the simulated BOD and NBOD reductions. The limitations of this first-order biokinetic model are evident. It is too simple, and maybe cannot obtain accurate simulation results for complex field cases (Jou et al., 2008).

Tuncsiper et al. (2006) used a first-order plug flow model and a multiple regression model to estimate the removal performances of nitrogenous pollutants by a surface flow constructed wetland. They reported that the regression model provided better predictions of effluent concentrations than the first-order plug flow model. The first-order plug flow model estimated slightly higher or lower values than the observed data when compared with the multiple regression model. On the other hand, as the basic kinetic models, such as first-order models and regression models, depend on analyzing influent, effluent concentrations and hydraulic residence time only without simulating dynamic processes, the removal efficiencies of wetland systems maybe cannot be predicted accurately due to the complexity of wetland systems (Tuncsiper et al., 2006).

Some other of the wetland models are more complex hydrodynamic or system dynamic models (Langergraber et al., 2009). For instance, Jou et al. (2012) simulated a surface flow constructed wetland using the QUAL2K model to analyze the BOD removal efficiencies and manage the wastewater renovation system. The QUAL2K model was developed by US EPA for simulating the transport and fate of stream contaminants. Sixteen water quality parameters could be combined to analyze the water quality in this model. The simulation results indicated that the removal efficiencies for BOD were 81-82% which was close to the observed data from the field case (72-84%). The QUAL2K model is a one-dimensional model, which assumes that the flow is steady. It does not have independent flow simulation module also. So it maybe cannot conduct accurate flow simulation (Jou et al., 2012).

Chavan and Dennett (2008) designed and used a wetland water quality model (WWQM) to evaluate nitrogen, phosphorus and sediments retention from a surface flow constructed wetland system. The WWQM model included four submodels: hydrological, nitrogen, phosphorus and TSS (Total Suspended Solid). It is reported that the WWQM simulation results of nutrient and sediments retention were reasonable and agreed with the observed data from the field case. With required input data from the field case, this model can be a useful tool for better understanding nutrients and sediments removal processes, and designing the wetland system. The model assumes that the flow is in the steady state. It maybe cannot obtain reasonable flow simulation for complex flow cases (Chavan and Dennett, 2008).

Naz et al. (2009) modeled a surface flow constructed wetland using an artificial neural network (ANN) modeling approach for simulating the removal performances of the wetland, and predicting the future planning of wastewater treatment system. It is reported that the ANN model provided a reasonable match between the observed data and the predicted concentrations of total COD (Chemical Oxygen Demand), soluble COD and total BOD in the effluents of the constructed wetland. Though the ANN model is very useful and widely applied for wetland modeling studies, large quantities of observed data are needed for model application and validation. The model application process may be time-consuming.

Table 2.1. Modeling studies of surface flow constructed wetland

Study description	Results	Limitations	Reference
A first-order biokinetic model was applied to simulate the removal performances of BOD and NBOD.	The observed removal efficiencies fell within ranges of the simulated results	Too simple; cannot obtain accurate results for complex cases	Jou et al., 2008
A first-order plug flow model and a multiple regression model were applied to estimate removal performances of nitrogenous pollutants.	The regression model provided better predictions of effluent concentrations. The first-order plug flow model estimated slightly higher or lower values than the observed data.	The basic kinetic models cannot simulate dynamic processes	Tuncsiper et al., 2006
The QUAL2K model was applied to manage the wastewater renovation system.	The removal efficiencies for BOD were 81-82% which was close to the observed data (72-84%).	1D model; steady flow assumption; no independent flow simulation module	Jou et al., 2012
The WWQM model was used to evaluate nitrogen, phosphorus, and sediments retention.	The simulation results agreed with the observed data.	Steady flow assumption	Chavan and Dennett, 2008
An ANN model was used to simulate the removal performances of a constructed wetland.	The model provided a reasonable match between the observed and the predicted data of pollutants in the effluents.	large quantities of data are needed; time-consuming	Naz et al., 2009

The QUAL2K model is applied for simulating a long and narrow surface flow constructed wetland in one study, which is mentioned above (Jou et al., 2012). In that study, as the QUAL2K model is a stream water quality model initially, the long and narrow surface flow wetland is considered as a stream. Reasonable simulation results are obtained from that study. The field case studied in this thesis is also a long and narrow surface flow constructed wetland, so the QUAL2K model can be tried and applied in this study.

The HEC-RAS model has never been applied for simulating water quality of constructed wetlands in previous wetland modeling studies, which is a stream water quality model. An important reason may be that the function of water quality analysis has been released in a new version of HEC-RAS model (Version 4.1) since 2010 (Brunner et al., 2010a). However, the current version of HEC-RAS model has similar water quality analysis mechanisms to the QUAL2K model. Furthermore, it has independent flow simulation module which may conduct better flow simulation than the QUAL2K model. Therefore, the HEC-RAS model can be tried and applied in this study.

2.1.2 The HEC-RAS model

The hydraulic flow of the channel is simulated before water quality analysis in the HEC-RAS model. The HEC-RAS hydraulic analysis is executed for building the geometry of different types of channels. A component of the model, steady flow water surface profile computations, is used for calculating water surface profiles for steady gradually varied flow. The computational procedure is based on the solution of a one-dimensional energy equation. Energy losses are assessed by friction and contraction/expansion (Brunner, 2010b).

Another component of the model, water quality analysis, is used to perform stream water quality analysis. An advection-dispersion module is included in this version of HEC–RAS. Transport and fate of a few water quality constituents is now available in the HEC-RAS model. These water quality constituents can be analyzed: Dissolved Nitrogen (NO_3-N, NO_2-N, NH_4-N, and Org-N); Dissolved Phosphorus (PO_4-P and Org-P); Algae; Dissolved Oxygen (DO); and Carbonaceous Biological Oxygen Demand (CBOD) (Brunner et al., 2010a).

2.1.2.1 Water surface profiles calculation

(1) Equations for basic profile calculations

By resolving an energy equation with a repetitive mechanism, water surface profiles are calculated from one cross section to the next. The energy equation is as follows (Brunner, 2010b):

$$Z_2 + Y_2 + \frac{a_2 V_2^2}{2g} = Z_1 + Y_1 + \frac{a_1 V_1^2}{2g} + h_e \tag{2.1}$$

where Z_1, Z_2 = elevation of the main channel inverts (m); Y_1, Y_2 = depth of water at cross sections (m); V_1, V_2 = average velocities (m/s); a_1, a_2 = velocity weighting coefficients; g = gravitational acceleration (m^2/s) and h_e = energy head loss (m).

The energy head loss (h_e) between two cross sections include friction losses and contraction or expansion losses. The equation for the energy head loss is as follow (Brunner, 2010b):

$$h_e = L\bar{S}_f + C\left|\frac{a_2 V_2^2}{2g} - \frac{a_1 V_1^2}{2g}\right| \tag{2.2a}$$

where L = discharge weighted reach length (m); $\bar{S}_f$ = representative friction slope between two sections and C = expansion or contraction loss coefficient.

The distance weighted reach length, L, is calculated as:

$$L = \frac{L_{lob}\overline{Q}_{lob} + L_{ch}\overline{Q}_{ch} + L_{rob}\overline{Q}_{rob}}{\overline{Q}_{lob} + \overline{Q}_{ch} + \overline{Q}_{rob}} \tag{2.2b}$$

where L_{lob}, L_{ch}, L_{rob} = cross section reach lengths specified for flow in the left overbank, main channel, and right overbank, respectively (m) and $\overline{Q}_{lob} + \overline{Q}_{ch} + \overline{Q}_{rob}$ = arithmetic average of the flows between sections for the left overbank, main channel, and right overbank, respectively (m^3/s).

(2) Cross section subdivision for conveyance calculations

Conveyance is calculated from the following Manning's equation (Brunner, 2010b):

$$Q = KS_f^{1/2} \tag{2.3a}$$

$$K = \frac{1.486}{n} AR^{2/3} \tag{2.3b}$$

where K = conveyance for subdivision; n = Manning's roughness coefficient for subdivision; A = flow area for subdivision (m^2) and R = hydraulic radius for subdivision (m).

(3) Evaluation of the mean kinetic energy head

The mean energy is computed by a flow weighted energy from the three subsections of a cross section for a given water surface elevation. To calculate the mean kinetic energy, it is necessary to obtain the velocity head weighting coefficient “a”, “a” is calculated as follows (Brunner, 2010b):

$$a\frac{\overline{V}^2}{2g} = \frac{Q_1\frac{V_1^2}{2g} + Q_2\frac{V_2^2}{2g}}{Q_1 + Q_2} \tag{2.4a}$$

In general:

$$a = \frac{\left[Q_1V_1^2 + Q_2V_2^2 + \ldots + Q_NV_N^2\right]}{Q\overline{V}^2} \tag{2.4b}$$

(4) Friction loss evaluation

Friction loss is calculated in HEC-RAS by the product of $\overline{S}_f$ and L (Equation 2.2a), where $\overline{S}_f$ is the representative friction slope for a reach, and L is defined by Equation 2.2b. The friction slope at each cross section is computed from Manning’s equation as follows (Brunner, 2010b):

$$S_f = \left(\frac{Q}{K}\right)^2 \tag{2.5}$$

(5) Contraction and expansion loss evaluation

Contraction and expansion losses are evaluated by the following equation (Brunner, 2010b):

$$h_{ce} = C\left|\frac{\alpha_1 V_1^2}{2g} - \frac{\alpha_2 V_2^2}{2g}\right| \tag{2.6}$$

where C = the contraction or expansion coefficient.

2.1.2.2 Water quality calculation

(1) Advection dispersion equation

An advection-dispersion module included in the HEC–RAS model is implemented for water quality analysis. The transport and fate of contaminants are calculated in HEC-RAS by the following advection-dispersion equation (Brunner et al., 2010a):

$$\frac{\partial}{\partial t}(\mathrm{V}\phi) = -\frac{\partial}{\partial x}(\mathrm{Q}\phi)\Delta\mathrm{x} + \frac{\partial}{\partial x}(\Gamma\mathrm{A}\frac{\partial\phi}{\partial x})\Delta\mathrm{x} \pm \mathrm{S} \tag{2.7}$$

where V = volume of the water quality cell (m^3); ϕ = water temperature (℃) or concentration (kg/ m^3); Q = flow (m^3/s); Γ = user-defined dispersion coefficient (m^2/s); A = cross sectional area (m^2) and S = sources and sinks (kg/s).

(2) Source and sink equations

a) Algae

Algal growth and respiration affects the algal concentration, nutrient concentrations and dissolved oxygen. The single internal source of algal biomass (A) is algal growth. Two sinks are simulated: algal respiration and settling. Sources and sinks of algae are computed as (Brunner et al., 2010a):

$$A_{source/\sin k} = A\mu - A\rho^* - \frac{\sigma_1^*}{d} A \tag{2.8}$$

where ρ^* = algal local respiration rate (1/day); σ_1^* = algal setting rate (m/day); d = average channel depth (m) and μ = local growth rate for algae (1/day);

b) Nitrate nitrogen (NO_3)

The only internal source of nitrate nitrogen is oxidation of nitrite (NO_2) to nitrate (NO_3). The only modeled sink is algal uptake. Sources and sinks for the nitrate are (Brunner et al., 2010a):

$$NO_{3source/\sin k} = \beta_2^*(1-\exp^{-KNR \cdot DOX})\,\mathrm{NO}_2 - (1-\mathrm{F}_1)\alpha_1 \mu A \tag{2.9}$$

where β_2^* = rate constant: oxidation of nitrite to nitrate (1/day); KNR = first order nitrification inhabitation coefficient (L/mgO); α_1 = fraction of algal biomass that is nitrogen (mgN/mgA); F_1 = fraction of algal uptake from ammonium pool (unitless) and μ = local growth rate for algae (1/day).

c) Organic phosphorus (OrgP)

The only internal source of organic phosphorus (OrgP) is algal respiration. Internal sinks for OrgP are decay of OrgP to form orthophosphate (PO_4), and settling to the bed. Sources and sinks for the organic phosphorus are (Brunner et al., 2010a):

$$OrgP_{source/\sin k} = \alpha_2 \rho^* A - \beta_4^* OrgP - \sigma_5^* OrgP \tag{2.9}$$

where β_4^* = rate constant: oxidation of OrgP to PO_4 (1/day); σ_5^* = settling rate: organic phosphorus (1/day); ρ^* = algal local respiration rate (1/day) and α_2 = fraction of algal biomass that is phosphorus (mgP / mgA).

2.1.3 The QUAL2K model

The QUAL2K model is often used to analyze water quality for rivers or streams. QUAL2K uses Excel as the graphical user interface. It can conduct one-dimensional hydraulic calculations for steady flow simulation. Sixteen water quality parameters can be combined to analyze the water quality in this model (Chapra, 2008).

2.1.3.1 Segmentation and hydraulics

(1) Segmentation

The flow simulation in the QUAL2K model represents a river or stream as a series of reaches. These representative reaches of a stream have constant hydraulic characteristics (e.g., slope, bottom width, etc.). As shown in Figure 2.2, the reaches are numbered starting from the headwater of the river's main stem (Chapra, 2008).

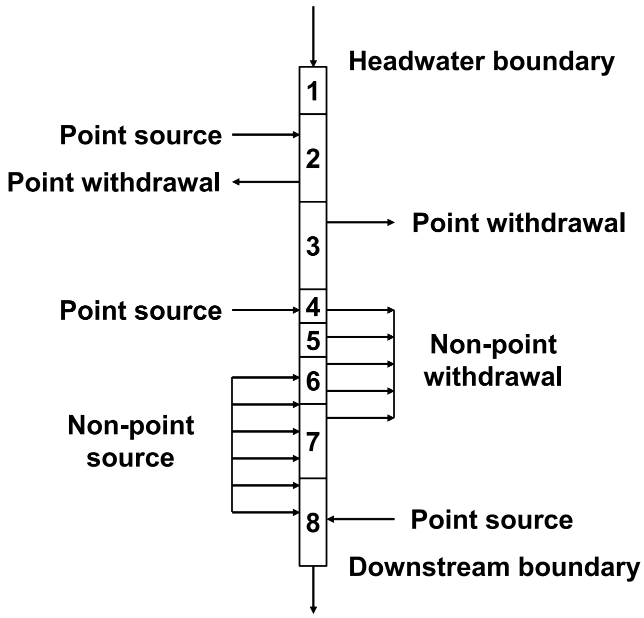

Figure 2.2. Segmentation scheme for a stream (Chapra, 2008)

(2) Hydraulic characteristics

For flow simulation, the following power equations are used to compute the mean velocity and depth (Chapra, 2008).

$$U = aQ^{b} \tag{2.10a}$$

$$H = \alpha Q^{\beta} \tag{2.10b}$$

where a, b, α and β are empirical coefficients that are determined from velocity-discharge and stage-discharge rating curves, respectively; Q represents flow rate (m^3/s), U represents mean velocity (m/s) and H represents flow depth (m).

The cross-sectional area and width of flow can be determined by the following equations (Chapra, 2008):

$$A_c = \frac{Q}{U} \tag{2.11a}$$

$$B = \frac{A_c}{H} \tag{2.11b}$$

where A_c represents cross-sectional area (m^2) and B represents width (m).

The surface area and volume of one element in a reach are calculated by the following equations (Chapra, 2008):

$$A_s = B\Delta x \tag{2.12a}$$

$$V = BH\Delta x \tag{2.12b}$$

where A_s represents the surface area (m^2); V represents volume (m^3) and Δx represents the length of the element (m).

2.1.3.2 Water quality calculation

(1) Contaminant transport and fate equation

The transport and fate of one contaminant are calculated by the following equation for water quality analysis in QUAL2K (Jou et al., 2012):

$$\frac{dc_i}{dt} = \frac{Q_{i-1}}{V_i} c_{i-1} - \frac{Q_i}{V_i} c_i - \frac{Q_{out,i}}{V_i} c_i + \frac{E'_{i-1}}{V_i}(c_{i-1} - c_i) + \frac{E'_i}{V_i}(c_{i+1} - c_i) + \frac{W_i}{V_i} + S_i \qquad (2.13)$$

where dc_i / dt = Changing rate of the concentration of an element with respect to time; Q_i = the outflow from element i to element $i+1$ (m^3/d); $Q_{out,i}$ = the total outflow from element due to point and nonpoint withdrawals (m^3/d); V_i = volume of i^{th} element (m^3); E'_i = the bulk dispersion coefficient between elements i and $i+1$ (m^3/d); W_i = the external loading of the constituent to element i (g/d or mg/d) and S_i = sources and sinks of the constituent due to reactions and mass transfer mechanisms (g/m^3d or mg/ m^3d).

(2) Source and sink equations

a) Nitrate nitrogen

The source of nitrate nitrogen is nitrification of ammonia. The sinks of it are denitrification and plant uptake (Chapra, 2008):

$$S_{ni} = \text{Nitrif} - \text{Denitr} - (1 - P_{ab}) \frac{\text{BotAlgUptakeN}}{H} \tag{2.14}$$

where P_{ab} = the coefficient of the preferences for ammonium as a nitrogen source for bottom algae and H = water depth (m).

b) Organic phosphorus

The sources of organic phosphorus are plant death and excretion. The sinks of it are hydrolysis and settling (Chapra, 2008).

$$S_{po} = f_{opp} q_{Pp} \frac{\text{PhytoDeath}}{H} + f_{opb} q_{Pb} \frac{\text{BotAlgDeath}}{H} - \text{OPHydr} - \text{OPSettl} \tag{2.15a}$$

where f_{opp} = the fraction of the phytoplankton internal phosphorus that is in organic form; q_{Pp} = the phytoplankton cell quotas of phosphorus (mgP/mgA) and q_{Pb} = the bottom algae cell quotas of phosphorus (mgP/mgA).

2.2 Uncertainties in Environmental Risk Assessment

Environmental risk assessment is essential to any wastewater management processes for minimizing the effects of wastewater discharges. As field case conditions and required input data for environmental modeling tend to be vague or imprecise; therefore, uncertainty exists in any environmental risk assessment studies (Darbra et al., 2008). Generally, uncertainties in risk assessment may have two types: randomness and incompleteness. There are also two main according ways to deal with these uncertainties: stochastic simulation method and fuzzy logic method (Qin and Huang, 2009).

2.2.1 Stochastic simulation method

Stochastic simulation method uses probability functions to analyze the randomness in environmental modeling parameters. Among various stochastic methods, the Monte Carlo simulation method has been widely used (Darbra et al., 2008). The Monte Carlo simulation method uses random sampling to study properties of system parameters which behave randomness. Monte Carlo methods are mainly used in three different issues: optimization, numerical integration and generation of samples from a probability distribution. In environmental risk assessment studies, the function of sample generation from a probability distribution is used widely, and the Monte Carlo simulation method are frequently applied in environmental risk assessment studies (Lemieux, 2008). For instance, Schuhmacher et al. (2001) applied Monte Carlo simulation method to analyze the uncertainty related to an environmental risk assessment due to organic toxic chemicals for the residents living around a municipal solid waste incinerator. By coupling CHARM (Chemical Hazard Assessment and Risk Management) model and the Monte Carlo simulation, Mukhtasor et al. (2004) assessed wastewater toxic risks produced by an offshore platform.

The stochastic simulation approach mainly be used when sufficient information is available for estimating the probability distributions of uncertain parameters (Darbra et al., 2008). If the type of uncertainties such as uncertainties of incompleteness cannot be expressed as probability distributions as uncertainties of randomness do, the fuzzy logic method can be considered.

2.2.2 Fuzzy logic method

Fuzzy logic method uses membership functions and linguistic parameters to express incompleteness in environmental issues (Qin and Huang, 2009). Fuzzy logic method can deal with the situation of "partial truth" to quantify uncertainties. It can define a "degree of membership" for parameters by membership functions. The membership functions take one of only two values: 0 (representing complete non-membership) and 1 (representing complete membership). Values between 0 and 1 are used to represent partial membership and the membership level (Darbra et al., 2008). Fuzzy logic method has been used for quantifying incompleteness uncertainty in environmental risk assessment. For instance, Li et al. (2008) applied fuzzy membership functions to quantify the uncertainties related to air quality standards including uncertain human exposure pathways and exposure dynamics, etc.

2.3 Summary

Each wetland simulation model has its advantages and limitations. The choice of a model is determined by the model's complexity, and the requirements and expectations of a particular wetland field case. Some wetland simulation models are very complex and required considerable amount of on-site field data to adequately simulate contaminant removal processes. However, various limitations, such as lack of time or funding, and complexity of field case, may preclude the acquirability of many model input data (Chavan and Dennett, 2008). Therefore, the effective wetland simulation models can be further developed. The application examples of wetland model onto field-scale wetlands can be further studied and there are not many wetland model application examples in previous model studies.

Furthermore, the uncertainty analysis methods including stochastic simulation and fuzzy logic approach have been rarely applied to field-scale wetlands before. They were not previously used to improve wetland simulation models. Among those environmental risk assessment studies which quantified system uncertainties, they tended to apply stochastic simulation approach alone to analyze the uncertainties of randomness in their systems. They did not quantify uncertainties of incompleteness such as variation inherent to the evaluation criteria (Chen et al., 2010). Actually, various types of uncertainties need to be considered when the environmental risk assessment is undertaken. The stochastic modeling method or fuzzy logic approach applied alone is not sufficient in many risk assessment studying cases, therefore, hybrid fuzzy-stochastic modeling approaches and their wetland field case applications can be further studied (Chen et al., 2003).

CHAPTER 3 METHODOLOGY

3.1 Methodology Overview

A hybrid fuzzy-stochastic modeling approach is proposed and developed in this thesis. It mainly includes two modules: water quality modeling method and integrated risk assessment system. Figure 3.1 shows a flow chart about the first part framework of the methodology: water quality modeling method. The field case studied in this thesis is a long and narrow surface flow constructed wetland that can be considered as a stream for modeling. Two representative water quality models (the HEC-RAS model and the QUAL2K model) are chosen to conduct the modeling work for the same field case after the literature review about wetland modeling studies. Two groups of simulation results can be obtained. Then, the simulation results from the two models are compared with the on-site field water quality data to determine if they are acceptable or not and which model performs better on modeling this field case. After the better model is determined and validated, it is applied to predict the water quality parameter change for several wetland design scenarios, and integrated with the environmental risk assessment process.

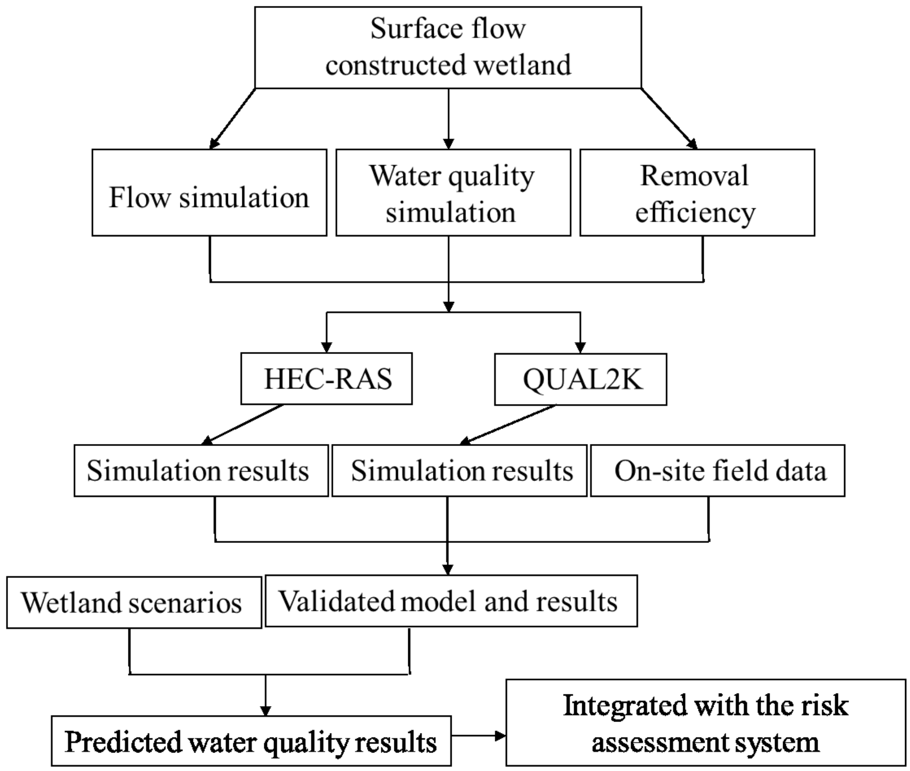

Figure 3.1. The framework of water quality modeling method

In the second part methodology of this thesis, an integrated environmental risk assessment approach is proposed. The Monte Carlo modeling method is developed to extend the water quality model to provide a stochastic simulation for quantifying the uncertainties of the system and the risks of the discharges on the receiving river. Particularly, the fuzzy membership functions are established for assessing the water quality guidelines of contaminants in different regions, which is incorporated into the Monte Carlo modeling framework to identify the integrated risks from the discharges on the receiving river. With the integrated process as shown in Figure 3.2 and 3.5, the hybrid fuzzy-stochastic modeling approach is used as a risk assessment tool for the assessment and management of wastewater discharges into river ecosystems.

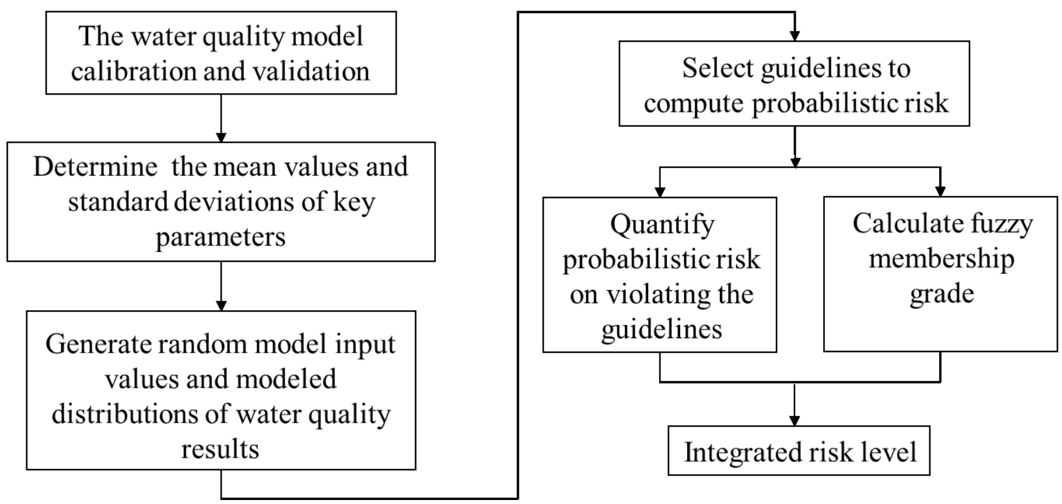

Figure 3.2. The framework of the integrated risk assessment system

3.2 Water Quality Analysis Models

The HEC-RAS model and the QUAL2K model are both applied for water quality analysis modeling in this study. The water quality analysis is implemented in HEC-RAS by the following advection-dispersion equation (Brunner et al., 2010a):

$$\frac{\partial}{\partial t}(\mathrm{V}\,\phi) = -\frac{\partial}{\partial x}(\mathrm{Q}\,\phi)\Delta\,\mathrm{x} + \frac{\partial}{\partial x}(\Gamma\,\mathrm{A}\frac{\partial \phi}{\partial x})\Delta\,\mathrm{x} \pm \mathrm{S} \tag{3.1}$$

where V = volume of the water quality cell (m^3); ϕ = water temperature (℃) or concentration (kg/ m^3); Q = flow (m^3/s); Γ = user-defined dispersion coefficient (m^2/s); A = cross sectional area (m^2) and S = sources and sinks (kg/s).

The water quality analysis is implemented in QUAL2K by the following equation (Jou et al., 2012):

$$\frac{dc_i}{dt} = \frac{Q_{i-1}}{V_i}c_{i-1} - \frac{Q_i}{V_i}c_i - \frac{Q_{out,i}}{V_i}c_i + \frac{E'_{i-1}}{V_i}\left(c_{i-1} - c_i\right) + \frac{E'_i}{V_i}\left(c_{i+1} - c_i\right) + \frac{W_i}{V_i} + S_i \tag{3.2}$$

where dc_i / dt = Changing rate of the concentration of an element with respect to time; Q_i = the outflow from element i to element $i+1$ (m^3/d); $Q_{out,i}$ = the total outflow from element due to point and nonpoint withdrawals (m^3/d); V_i = volume of i^{th} element (m^3); E'_i = the bulk dispersion coefficient between elements i and $i+1$ (m^3/d); W_i = the external loading of the constituent to element i (g/d or mg/d) and S_i = sources and sinks of the constituent due to reactions and mass transfer mechanisms (g/m^{3}d or mg/ m^{3}d).

With the comparison between the two models' simulation results and observed field case data, which one performs better for this field case could be determined. The better model is integrated with the environmental risk assessment process.

3.3 Hybrid Fuzzy-Stochastic Risk Assessment Approach

3.3.1 Monte Carlo method for quantifying system uncertainty

The uncertainties of both model and data need to be considered when the environmental risk assessment is undertaken (Chen et al., 2010). The Monte Carlo modeling method can be developed to extend the water quality model to quantify system uncertainties, and the modeling results are used for risk quantification in the next step.

The randomness of a key parameter from the water quality model can be described by a probability distribution using the Monte Carlo modeling method. Then, random values in the probability distribution are inputted into the water quality model, and a distribution of water quality results can be obtained (Qin et al., 2009). For instance, in the HEC-RAS model, β_2^* (rate constant: oxidation of nitrite to nitrate) is determined as a sensitive and key input variable, and it is a random parameter with a certain range. A normal distribution can be used to express its uncertainty and randomness (Riddle et al., 2001; Chen et al., 2010). Random values of the variable generated by the Monte Carlo modeling method are based on related references and observed mean value obtained in this field case. The normal generators can be expressed as follows:

$$x = N(\sigma_x, \mu_x) \tag{3.3}$$

where x represents a key parameter; N (σ_x, μ_x) represents a normal distribution function of σ_x and μ_x; σ_x is the standard deviation of x and μ_x is the mean value of x.

The random values of each parameter are inputted into the water quality model after their normal distributions are generated, and then the distributions of the water quality results can be obtained.

3.3.2 Probabilistic risk assessment

After the distributions of the water quality simulation results are obtained from the water quality model and the Monte Carlo modeling method, the environmental risk level is quantified by the equation as follows (Chen et al., 1998):

$$R = P(L > C_s) = \int_{C_s}^{\infty} f_L(L)\,dL \tag{3.4}$$

where R is the risk level quantified as the probability of system failure; $f_L(\mathrm{L})$ is the probability density function; L is a random contaminant's concentration (mg/L) and C_s is a local environmental criterion (mg/L).

3.3.3 Construction of fuzzy membership functions for evaluation criteria

A few water quality guidelines are often overly conservative or not strict enough as mentioned above; therefore, the practicability of those guidelines can be further addressed. In particular, the fuzzy membership functions provide a method to fulfill this task. In this study, TN and TP are used as indicators (City of Edmonton drainage services, 2012) and triangle membership functions are to be formulated by the analysis of the adverse environmental impacts from nutrients on the receiving waters.

3.3.3.1 Fuzzy membership functions for TN

Table 3.1 shows that the surface water quality guidelines for TN in different countries and regions are varied. The surface water quality guideline for TN in the

Alberta, Canada is 1.0 mg/L (AENV, 1999). For determining nutrients criteria in different areas, the United States is divided into 14 distinct eco-regions, and each eco-region has its own guideline. Three representative guidelines from them are analyzed in this study. In eco-region I (Willamette and Central Valleys), eco-region V (South Central Cultivated Great Plains) and eco-region VI (Corn Belt and Northern Great Plains), the guidelines for TN are 0.31 mg/L, 0.88 mg/L and 2.18 mg/L, respectively (USEPA, 2002). The guideline for TN in the Netherlands is the least strict (2.2 mg/L) compared to the guidelines mentioned above (Neeteson, 2000).

Table 3.1. The surface water quality guidelines for TN in different countries and regions

Origin		TN concentration (mg/L)	References
Alberta, Canada		1.0	AENV, 1999
United States	USEPA Ecoregion I	0.31	USEPA, 2002
	USEPA Ecoregion V	0.88	
	USEPA Ecoregion VI	2.18	
Netherlands		2.2	Neeteson, 2000

To establish the membership functions for the fuzzy evaluation criteria on TN, the ratio (TN/NO_3) = 2.6 and (TN/NO_2) = 14.7 are obtained and used in this study according to Watercenter (2013) and the observed data from the field case (City of Edmonton drainage services, 2012). Establishment of the membership functions for the fuzzy evaluation criteria on TN is conducted in three steps:

(1) Determination of the maximum tolerable TN concentration (C_{max})

Nitrite (NO_2), an important component in TN, is toxic in water. It is reported that 1 mg/L for Nitrite-Nitrogen is determined as the maximum contaminant level in drinking water by the U.S. federal government (CNA Environmental, 2005). When the concentration of NO_2-N is 1 mg/L, TN is around 14 mg/L according to the ratio (TN/NO_2 =14.7) mentioned above (Watercenter, 2013; City of Edmonton drainage services, 2012).

Nitrate (NO_3), another essential component in TN, is also toxic when it has a high concentration in water. It has potential risk to result in the methemoglobinemia in infants and has toxic effects on livestock. Several laboratory studies indicated that 10 mg NO_3-N/L nitrate in water had adverse effects on sensitive aquatic animals (Camargo and Alonso, 2006). Similarly, 10 mg/L for Nitrate-Nitrogen is determined as the maximum contaminant level in drinking water by the U.S. federal government (CNA Environmental, 2005). Furthermore, Douda (2010) reported that NO_3-N concentrations in surface waters could actually higher than 25 mg/L due to nitrogen pollution, but NO_3-N concentrations were only up to 2.0-2.3 mg/L in localities with undisturbed populations of river mussel. Camargo et al. (2005) proposed that 2 mg NO_3-N/L should be the maximum concentration in surface water for the protection of sensitive aquatic animals. Based on the ratio (TN/NO_3 = 2.6), TN should be lower than 5 mg/L.

Thus, when the concentrations of TN are higher than 5 mg/L in surface water, they might have an adverse impact on sensitive aquatic animals. Conservatively, the TN of 5 mg/L is chosen as a completely unsuitable level, and the maximum tolerable TN concentration should be determined as 5 mg/L (C_{max} = 5 mg/L) with a suitability grade of 0 in the membership functions. Therefore, when the values of C (Concentration) are 5 mg/L or even higher, the membership grade μ $(C_{max}) = 0$. TN concentrations lower than 5 mg/L in surface water would be more suitable to be used as the guideline or standard.

(2) Determination of the most suitable TN level ($C_{optimal}$)

Camargo and Alonso (2006) estimated that the adequate water quality criteria for NO_2-N should be between 0.08 and 0.35 mg NO_2-N/L for protecting sensitive aquatic animals. The safe range of TN should be 1.18 - 5.15 mg/L based on the ratio ($TN/NO_2 = 14.7$).

The field case studied in this thesis is located in Alberta, Canada. The surface water quality guideline for TN in Alberta, Canada is 1.0 mg/L (AENV, 1999). From the studies of Chambers et al. (2011), the IPS (Ideal Performance Standards) for TN in the Southern Alberta, Canada is 0.98 mg/L. Furthermore, Camargo and Alonso (2006) reported that the levels of TN lower than 0.5-1.0 mg TN/L could prevent aquatic ecosystems from developing acidification and eutrophication. TN at these low levels could also protect aquatic animals from the toxicity of inorganic nitrogenous compounds.

Therefore, the most suitable TN level $C_{optimal}$ is determined as 0.5 mg/L based on the above analysis, with the membership grade $\mu(C_{optimal}) = 1$ (Camargo and Alonso, 2006).

(3) Determination of the minimum possible TN concentration (C_{min})

For the minimum possible TN concentration, the extreme situation is considered as $C_{min} = 0$ mg/L, which is impractical and cannot be implemented as a standard. In the membership functions, C_{min}= 0 mg/L is assigned with the membership grade μ (C_{min}) = 0.

In summary, the membership functions of fuzzy evaluation criteria quantifying the "suitability" of TN guidelines are obtained based on the above analysis (Figure 3.3):

$$\mu(C_s) = 2\,C_s, \text{ when } 0 \leqslant C_s \leqslant 0.5 \tag{3.5a}$$

$$\mu(C_s) = 1.11\text{-}0.22\,C_s, \text{ when } 0.5 < C_s \leqslant 5 \tag{3.5b}$$

where C_s is a variable denoting a regulated guideline or standard value (mg/L).

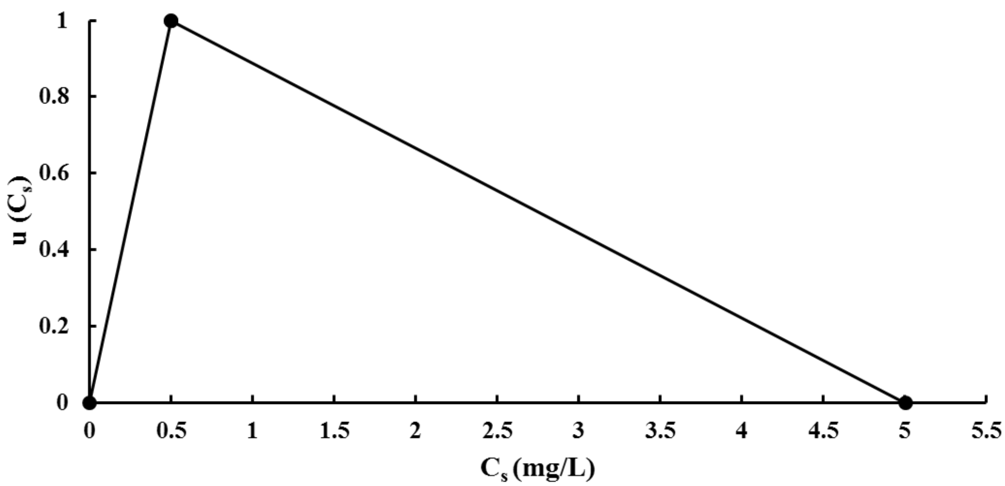

Figure 3.3. Fuzzy membership functions of evaluation criteria for TN

3.3.3.2 Fuzzy membership functions for TP

Table 3.2 shows that the surface water quality guidelines for TP in different countries and regions are varied. The surface water quality guideline for TP in Alberta, Canada is 0.05 mg/L (AENV, 1999). For TP criteria in U.S., the guidelines in eco-region I (Willamette and Central Valleys), eco-region II (Western Forested Mountains) and eco-region VI (Corn Belt And Northern Great Plains) are 0.047 mg/L, 0.01 mg/L and 0.076 mg/L, respectively (USEPA, 2002). The guideline for TP in Netherlands is the least strict (0.15 mg/L) compared to the guidelines mentioned above (Neeteson, 2000).

Table 3.2. The surface water quality guidelines for TP in different countries and regions

Origin		TP concentration (mg/L)	References
Alberta, Canada		0.05	AENV, 1999
United States	USEPA Ecoregion I	0.047	USEPA, 2002
	USEPA Ecoregion II	0.01	
	USEPA Ecoregion VI	0.076	
Netherlands		0.15	Neeteson, 2000

In a typical natural water body, the concentration of TP is approximately 23 mg/m^3. Among the TP, SRP (Soluble reactive phosphorus), OrgP (Organic soluble phosphorus) and particulate phosphorus are around 3 mg/m^3, 14 mg/m^3 and 6 mg/m^3, respectively (Kutty, 1987). To establish the membership functions for the fuzzy evaluation criteria on TP, the ratio TP/SRP = 23/3 and TP/OrgP = 23/14 are used in this study. Establishment of the membership functions for the fuzzy evaluation criteria on TP is conducted in three steps:

(1) Determination of the maximum tolerable TP concentration (C_{max})

Mainstone et al. (2002) reported that when the concentration of SRP, an important component in TP, was higher than 10 ug/L in rivers, it might have adverse effects on the growth of individual plant species (algae and higher plants). When SRP is greater than 10 ug/L, TP would be higher than 0.077 mg/L based on the ratio TP/SRP (Kutty, 1987).

In the studies of Mainstone et al. (2002) on TP, it was reported that the growth rates and standing crop of riverine algal communities could be affected with a great potential when the concentrations of TP reached up to no less than 0.2-0.3 mg/L in water. Additionally, it was obvious that the risk level of TP changed most rapidly when TP concentration increased from the natural level to around 0.2-0.3 mg/L.

Conservatively, the TP of 0.2 mg/L is chosen as a completely unsuitable level, and the maximum tolerable TP concentration should be determined as 0.2 mg/L (C_{max} = 0.2 mg/L) with a suitability grade of 0 in the membership functions. Therefore, when the values of C are 0.2 mg/L or even higher, the membership grade $\mu(C_{max}) = 0$. TP concentrations lower than 0.2 mg/L in surface water would be more suitable to be used as the guideline or standard.

(2) Determination of the most suitable TP level ($C_{optimal}$)

It was reported by Mulholland and Hill (1997) that the natural riverine concentrations of SRP were less than 10 ug/L. Similarly, Mainstone et al (2002) also reported that the natural level of SRP would be below 10 ug/L. Accordingly (TP/SRP = 23/3), TP would be below 0.077 mg/L (Kutty, 1987).

The recommended US EPA criteria for TP in different aggregate nutrient ecoregions for rivers and streams are from 0.01mg/L to 0.076 mg/L (USEPA. 2002). The surface water quality guideline for TP in the Alberta, Canada, where the study case is located, is 0.05 mg/L (AENV, 1999). Mainstone et al. (2008) reported that target concentrations of TP, which were included in "Common Standards" guidance for both SSSI (Sites of Special Scientific Interest) and SAC (Special Areas of Conservation) rivers and lakes by the UK nature conservation agencies in 2004, were between 0.01 to 0.05 mg/L. Additionally, Mainstone et al (2002) estimated that 0.03 mg/L of TP was the mean natural concentration in all case studies. This value was an indication of ecologically desirable background load.

Therefore, the most suitable TP level is determined as $C_{optimal} = 0.03$ mg/L, the membership grade $\mu(C_{optimal}) = 1$.

(3) Determination of the minimum possible TP concentration (C_{min})

For the minimum possible TP concentration, the extreme situation is considered as $C_{min} = 0$ mg/L, which is impractical and cannot be implemented as a standard. In the membership functions, $C_{min} = 0$ mg/L is assigned with the membership grade $\mu(C_{min}) = 0$.

In summary, the membership functions of fuzzy evaluation criteria quantifying the "suitability" of TP guidelines are obtained based on the above analysis (Figure 3.4):

$$\mu(C_s) = 33.33\, C_s, \text{ when } 0 \leq C_s \leq 0.03 \quad (3.6a)$$

$$\mu(C_s) = 1.18 - 5.88\, C_s, \text{ when } 0.03 < C_s \leq 0.2 \quad (3.6b)$$

where C_s is a variable denoting a regulated guideline or standard value (mg/L).

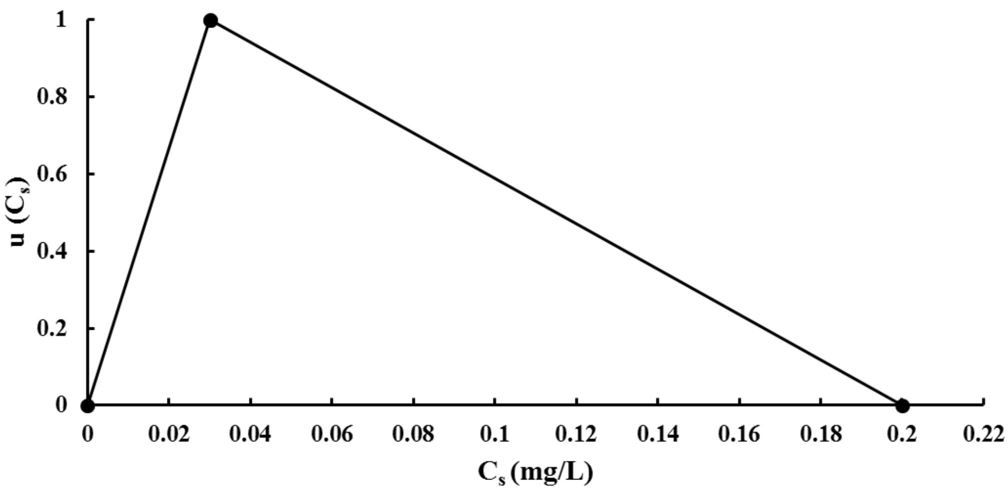

Figure 3.4. Fuzzy membership functions of evaluation criteria for TP

3.4 Integrated Risk Assessment System

Uncertainties exist in two aspects: the uncertainties in the modeling system and the variation of environmental guidelines. The first aspect of uncertainties could be quantified by the customized water quality model with the Monte Carlo simulation method, and the second aspect of uncertainties could be analyzed by the fuzzy membership functions.

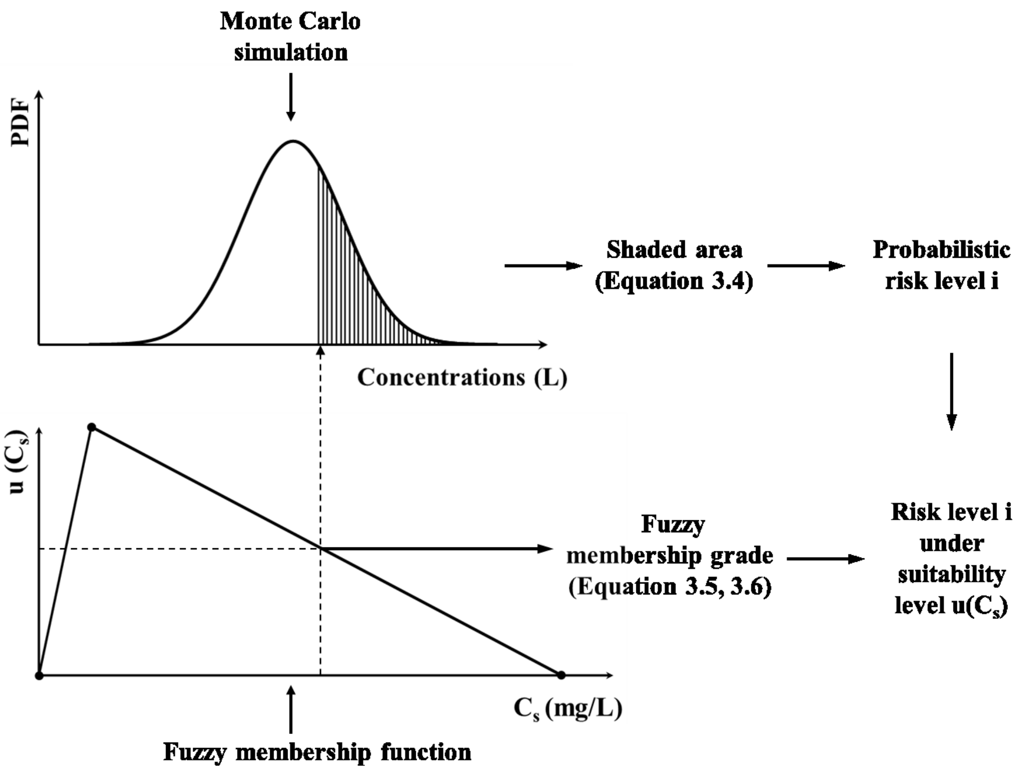

Figure 3.5. Integrated risk assessment using the hybrid fuzzy-stochastic modeling approach (Adapted from Chen et al., 2010)

Figure 3.2 presents the framework of the integrated risk assessment system. The follows show the steps to quantify the risk level through the proposed integrated risk assessment system:

(1) Based on the water quality simulation results, the Monte Carlo simulation generates distributions of contaminant concentrations as shown by the PDF (Probability Density Function) curve in Figure 3.5.

(2) If a criterion C_s is used as the water quality guideline or standard, the shaded area represents the possibility of the contaminant concentrations violating the guideline or standard. The risk level (R) of violating this guideline or standard can be calculated by Equation 3.4. For instance, let $C_s = 1$ mg/L, which is the surface water guideline for TN in the Alberta, Canada (AENV, 1999), then the risk level R can be calculated.

(3) The suitability of using a particular guideline (e.g. the surface water guideline 1mg/L for TN in Alberta, Canada) can be quantified based on Equation 3.5 and Figure 3.3. The result indicates that the practicability of this guideline is 0.89 out of 1. Finally, the integrated risk level under this guideline can be obtained as shown in Figure 3.5.

CHAPTER 4 MODELING AND ASSESSMENT FOR THE KENNEDALE WETLAND CASE

4.1 Field Case Description

The field case studied in this thesis is the Kennedale end-of-pipe constructed wetland. It is located in the city of Edmonton, Canada. This wetland is designed to treat about 70% of the storm water from the Kennedale storm basin which contains a significant percentage of storm water from the Edmonton's storm system. The field case is a long and narrow surface flow constructed wetland that can be considered as a stream for modeling. Figure 4.1 describes the locations of Edmonton, Kennedale basin and Kennedale wetland (City of Edmonton, 2012).

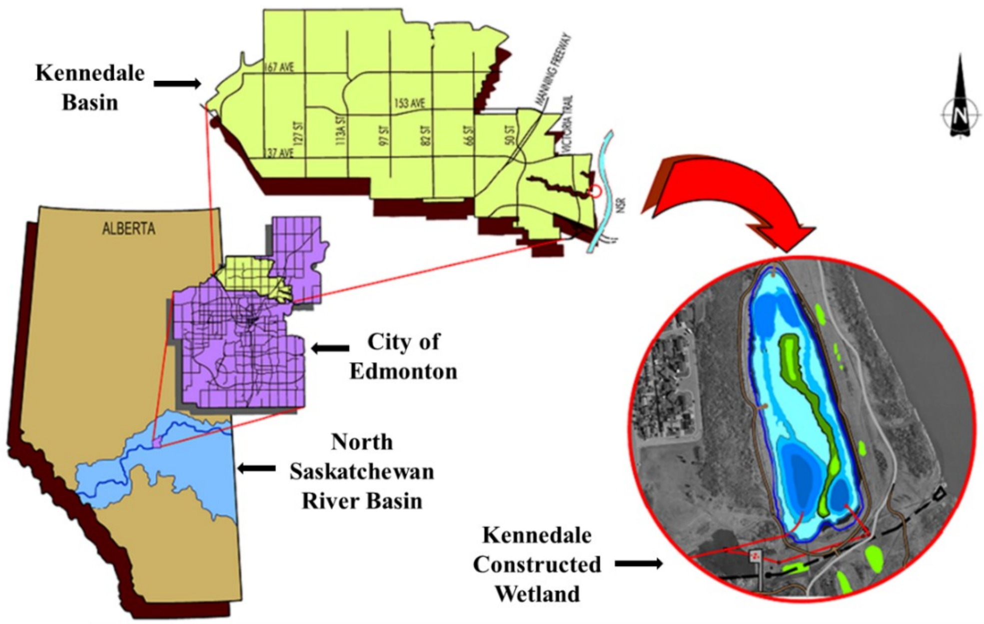

Figure 4.1. The locations of Edmonton, Kennedale basin and Kennedale wetland (Adapted from City of Edmonton, 2012)

The storm water from the Kennedale storm basin is delivered into the Kennedale wetland when its flow rates are no more than 0.5 m^3/s. When the storm water flow is higher than 0.5 m^3/s in rainfall periods, a proportion of storm water will be diverted into another storm water treatment system units to make sure that the flow delivered into the wetland can keep no more than 0.5 m^3/s (City of Edmonton, 2012).

The schematic diagram of Kennedale wetland is shown in Figure 4.2. The storm water, which is diverted into the wetland, first enters into the forebay. After flowing through the low flow channels, deep marshes and deep pools, it finally arrives at the micropool. During the whole journey, the majority of contaminants in storm water are removed by a series of physical and biochemical processes. Storm water cleaned by the Kennedale wetland will be discharged into the North Saskatchewan River.

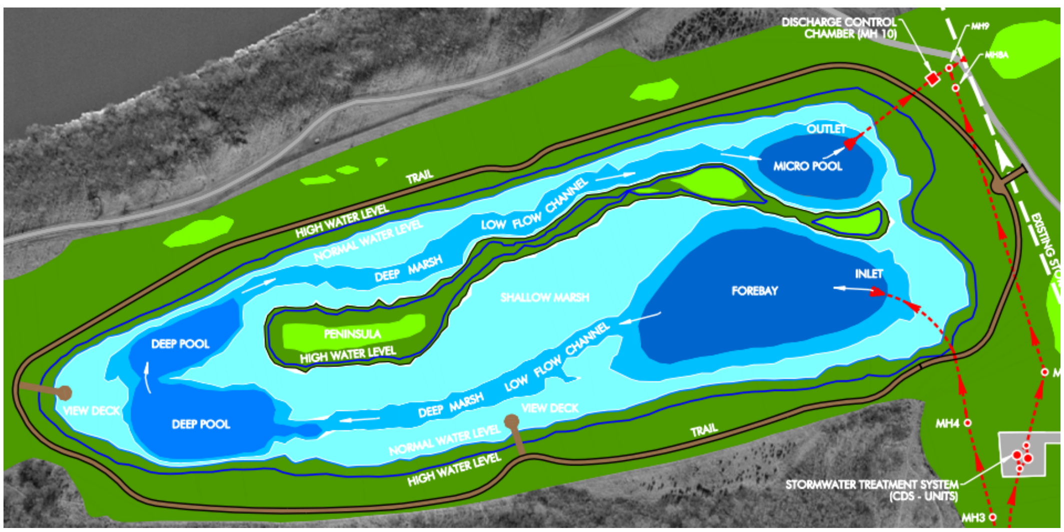

Figure 4.2. Schematic diagram of Kennedale wetland (City of Edmonton, 2012)

To assess removal performance of the Kennedale wetland and potential impacts of storm water discharges from the Kennedale wetland on the North Saskatchewan River, TN and TP are studied and used as indicators (City of Edmonton drainage services, 2012). The transport and fate of nutrients in the Kennedale wetland are simulated with the water quality modeling component, and the integrated risk levels are quantified by the hybrid fuzzy-stochastic risk assessment approach.

4.2 Water Quality Analysis Results

4.2.1 HEC-RAS simulation

4.2.1.1 Key input data

(1) Geometric and hydraulic data

a) The stream system schematic

The stream system schematic is developed by drawing and connecting the various reaches of the system within the geometric data editor. Figure 4.3 shows that the flow is drawn in the geometric data editor of the HEC-RAS system. The channel is built by many cross sections.

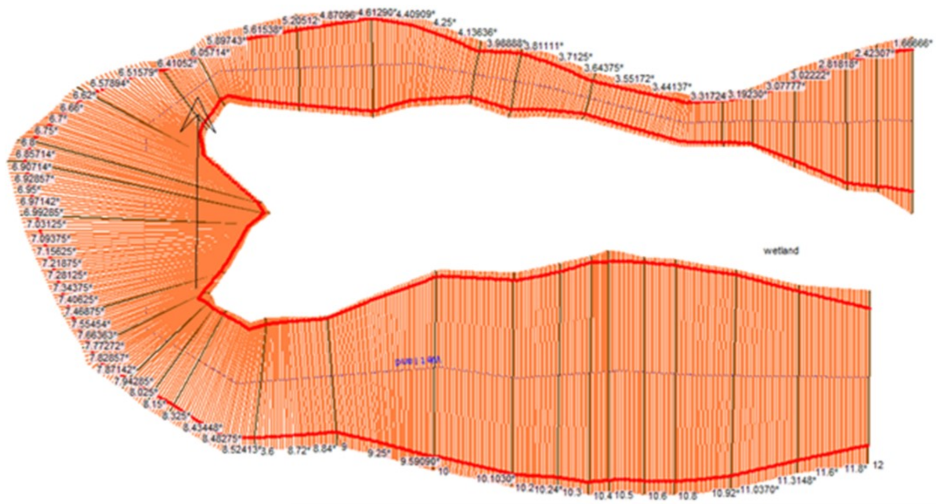

Figure 4.3. The stream system schematic built in the HEC-RAS model

b) Cross section data

Cross section data are required to be inputted at representative locations throughout a stream reach and at locations where changes occur in discharge, slope, shape, or roughness. Each cross section is described by entering the station and elevation data (X-Y data) from left to right (Brunner et al., 2010a). It also can be seen in Figure 4.3 that many cross sections were set for building the stream channel. For instance, Figure 4.4 shows the required coordinates for describing one cross section.

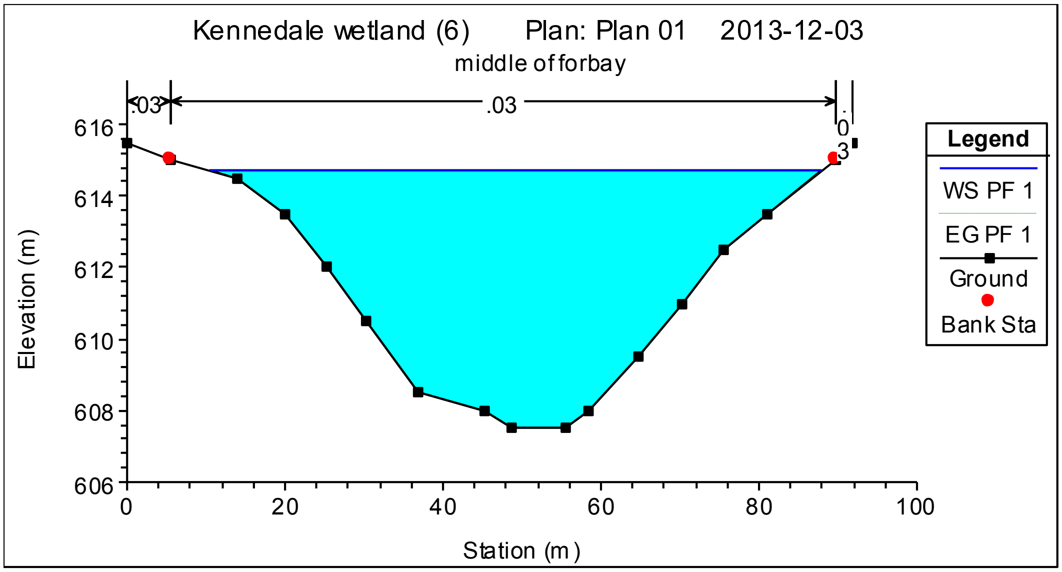

Figure 4.4. The required coordinates for describing one cross section

c) Other key input data

Table 4.1 shows other key geometric and hydraulic input data. They are obtained from the field case and the related reference (City of Edmonton drainage services, 2012; Brunner et al., 2010a).

Table 4.1. Key geometric and hydraulic input data

Flow rate (m^3/s)	Water surface elevation (m)	Manning's n	Contraction and expansion coefficients
0.14	614.7	0.03	0.1 and 0.3

(2) Water quality data

Table 4.2. Key water quality input data

Parameters	Boundary conditions	Initial conditions
Water temperature	12 ℃	12 ℃
Algae	2 mg/L	4 mg/L
DO	5 mg/L	8 mg/L
CBOD	30 mg/L	15 mg/L
NH_4	0.158 mg/L	0.054 mg/L
NO_2	0.175 mg/L	0.068 mg/L
NO_3	0.995 mg/L	0.385 mg/L
OrgN (Dissolved organic nitrogen)	0.55 mg/L	0.191 mg/L
OrgP (Dissolved organic phosphorus)	0.2 mg/L	0.014 mg/L
PO_4	0.043 mg/L	0.003 mg/L

In the HEC-RAS model, boundary conditions and initial conditions represent the water quality conditions in the wetland influent and the background water quality conditions, respectively. Table 4.2 shows the input data for water quality parameters. The water quality input data are based on the final report of 2011 Kennedale and Pylypow wetland performance monitoring project (City of Edmonton drainage services, 2012).

4.2.1.2 Flow simulation result

With all required input data for flow simulation, the X-Y-Z perspective plot of flow in the Kennedale wetland is constructed using HEC-RAS as shown in Figure 4.5. All water quality simulations are based on this flow simulation result.

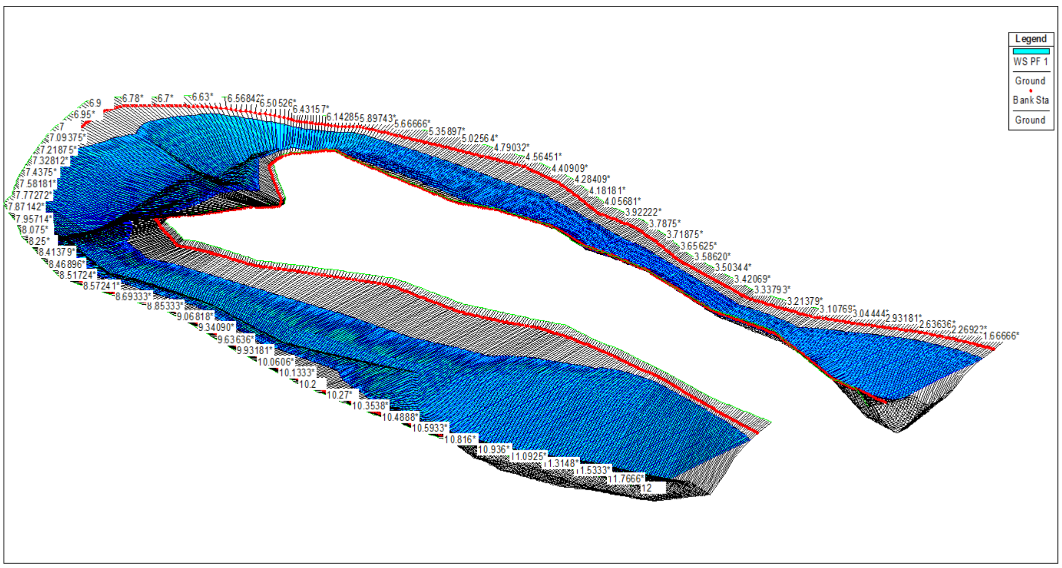

Figure 4.5. X-Y-Z perspective plot of the flow

4.2.1.3 Water quality simulation results for TN and TP

TN and TP cannot be simulated in the HEC-RAS model directly, but NO_3 and OrgP can be modeled. So the simulation results of NO_3 and OrgP are obtained from HEC-RAS first, then the ratio TN/NO_3 and TP/OrgP are used to get the simulation results for TN and TP in this study. Based on the studies from related references and the observed data from the field case, the ratio $TN/NO_3 = 2.6$ and $TP/OrgP = 23/14$ are obtained and used in this study (Kutty, 1987; City of Edmonton drainage services, 2012; Watercenter, 2013).

Table 4.3. The simulation results for nutrients from HEC-RAS

	Inlet (mg/L)	Outlet (mg/L)	Removal
Simulation results of NO_3	0.995	0.740	25.63%
Simulation results of TN	2.878	2.14	25.64%
Simulation results of OrgP	0.2	0.1	50%
Simulation results of TP	0.328	0.164	50%

The water quality simulation results for nutrients can be obtained from HEC-RAS based on the simulated flow and water quality input data. Table 4.3 shows the simulation results for TN and TP. The simulation result for TN from HEC-RAS shows that 25.64% of TN is removed after treated by the wetland. As for TP, 50% of it is removed.

4.2.2 QUAL2K simulation

4.2.2.1 Key input data

The headwater and reach information are inputted in the QUAL2K model for the flow simulation. The mean velocity of the flow is determined as 0.0147 m/s. The water surface elevation is 614.7 m, and the depth of headwater is 2.7 m (City of Edmonton drainage services, 2012). The whole channel of the wetland is divided into seven reaches, as required in QUAL2K, the input reach data are shown in Table 4.4 (City of Edmonton drainage services, 2012). The water quality data inputted in QUAL2K are almost the same as in HEC-RAS as the two models are applied to simulate a same field case.

Table 4.4. The input data for building reaches

Reach	Description	Upstream location (km)	Upstream elevation (m)	Mean depth (m)
1 (headwater)	From inlet to the middle of forebay	0.724	612	4.95
2	From the middle of forebay to the end of low flow channel	0.667	607.5	3.95
3	Deep marsh	0.542	614	0.95
4	Deep pool 1	0.426	613.5	0.95
5	Deep pool 2	0.356	614	0.7

Table 4.4. The input data for building reaches (To be continued)

Reach	Description	Upstream location (km)	Upstream elevation (m)	Mean depth (m)
6	From the end of deep pool 2 to the start of micro pool	0.311	614	1.45
7	Micro pool	0.049	612.5	1.95

4.2.2.2 Water quality simulation results for TN and TP

As the simulation results of HEC-RAS and QUAL2K would be compared, NO_3 and OrgP are simulated in QUAL2K also. The simulation results for TN and TP are gotten based on the ratios $TN/NO_3 = 2.6$ and $TP/OrgP = 23/14$ (Kutty, 1987; City of Edmonton drainage services, 2012; Watercenter, 2013). Table 4.5 shows the simulation results for TN and TP from QUAL2K.

Table 4.5. The simulation results for nutrients from QUAL2K

	Inlet (mg/L)	Outlet (mg/L)	Removal
Simulation results of NO_3	0.995	0.863	13.27%
Simulation results of TN	2.878	2.487	13.59%
Simulation results of OrgP	0.2	0.098	51%
Simulation results of TP	0.328	0.161	50.91%

According to the simulation result for TN from QUAL2K, the removal efficiency is 13.59%, which is different from the simulation result in HEC-RAS (25.64% for TN). The removal efficiencies are similar from the simulation results between HEC-RAS and QUAL2K for TP, which are 50% and 50.91%, respectively. Both the models' simulation results would be compared with the on-site field data to determine which model gives better performance for this field case.

4.2.3 Comparison between two models' results and on-site field data

The water quality simulation results from the two models are compared with the on-site field data. The on-site field data shown in Table 4.6 are the mean values of dozens of observed water quality data obtained between May and July in 2011 in the Kennedale wetland by regular water quality measurement (City of Edmonton drainage services, 2012). Table 4.6 shows the comparison between the simulation results and the on-site field data.

Table 4.6. The comparison between simulation results from the two models and the on-site field data

	HEC-RAS		QUAL2K		On-site field data (mg/L)
	Result (mg/L)	Difference	Result (mg/L)	Difference	
TN at outlet	2.14	0.05%	2.487	13.99%	2.139
TP at outlet	0.164	6.1%	0.161	4.35%	0.154

It can be seen that the differences (or errors) between the HEC-RAS simulation results and on-site field data are 0.05% for TN and 6.1% for TP. The differences between the QUAL2K results and on-site field data are 13.99% for TN and 4.35% for TP. The possible reasons for these differences are: 1) during the modeling study, a few input data cannot be obtained directly and have to be assumed according to related references (e.g. the value of Manning's n), so lack of accurate input data may be one reason; 2) The flow simulated in the water quality models are in steady state, but the flow state in the field case is more complicated;

3) The constructed wetland is a complex system with various uncertainties. However, the modeling system uncertainties can be further examined through a fuzzy-stochastic approach.

Both two models are applicable for simulating this field case as their differences are acceptable compared to the on-site field data (Jia and Culver, 2006). Based on the simulation results and the analysis of the two modeling systems, the HEC-RAS model is determined as the better model for modeling this field case. The validated water quality modeling component would be integrated with the environmental risk assessment process and applied to simulate and predict the water quality parameter changes for several design scenarios in this field case.

4.2.4 The design scenarios simulated by customized water quality modeling component

The flow rate is inputted as 0.14 m^3/s in the previous simulation work as it is the typical mean daily flow rate measured in the Kennedale wetland (City of Edmonton drainage services, 2012). However, the flow rates are not always 0.14 m^3/s in different days and seasons. In large rainfall periods, when the flow rates of storm water are higher than 0.5 m^3/s, a proportion of storm water will be diverted into another storm water treatment system units in order to keep the flow rates in the wetland as no more than 0.5 m^3/s. Actually, it is common that the flow rate reaches to 0.5 m^3/s in large rainfall days according to the on-site field data (City of Edmonton drainage services, 2012). Therefore, it is reasonable to assume scenarios with the flow rate of 0.5 m^3/s. Three scenarios are designed and simulated by the customized water quality modeling component to predict the water quality changes in these situations.

4.2.4.1 Scenario description

Scenario 1: in large rainfall periods, the amount of contaminants at the wetland inlet are assumed the same, but the flow rate reaches to 0.5 m^3/s due to the rainfall. In this scenario, the flow rate increases from 0.14 m^3/s (Q_0) to 0.5 m^3/s (Q_1), so the flow rate now is 3.57 times larger than the previous one, and the contaminants are all diluted. To make sure the amount of contaminants at the wetland inlet are the same, the concentrations of contaminants at the inlet in this scenario (C_1) should be 0.28 of the previous contaminant's concentration (C_0). In sum, in scenario 1, $Q_1 = 0.5\ m^3/s = 3.57\ Q_0$; $C_1 = 0.28\ C_0$. Table 4.7 shows the concentrations of different water quality parameters inputted in the customized water quality model for scenario 1 and the original simulation case.

Table 4.7. The reported initial concentrations and the definition concentrations of scenario 1

(mg/L)	Algae	CBOD	OrgN	NH_4	NO_2	NO_3	OrgP	PO_4
C_0*	2	30	0.55	0.158	0.175	0.995	0.2	0.043
$C_1 = 0.28\ C_0$ (Scenario 1)	0.56	8.4	0.154	0.044	0.049	0.279	0.056	0.012

(* The reported initial concentrations are from City of Edmonton drainage services, 2012)

Scenario 2: the flow rate Q_2 is designed as 0.5 m^3/s also, but the designed amount of contaminants is increasing due to the industrial development in the city of Edmonton in the future. Therefore, even the flow rate increases from 0.14 m^3/s (Q_0) to 0.5 m^3/s (Q_2), the concentrations of contaminants at the wetland inlet in this scenario (C_2) are assumed as the same as the original ones (C_0). In sum, $Q_2 = 0.5\ m^3/s = 3.57\ Q_0$; $C_2 = C_0$.

Scenario 3: this scenario is designed as a control sample to examine the stability and consistency of the model results. In sum, $Q_3 = Q_0 = 0.14\ m^3/s$; $C_3 = 0.28\ C_0$.

4.2.4.2 The simulation results of the design scenarios

The water quality simulation results are obtained from the water quality modeling component for the three design scenarios. Table 4.8 and 4.9 show the simulation result comparisons for TN and TP between the three scenarios and the original one.

Table 4.8. The simulation result comparison for TN between the three scenarios and original one

	Inlet (mg/L)	Outlet (mg/L)	Removal
Scenario 1 ($Q_1 = 0.5$ m^3/s; $C_1 = 0.28\ C_0$)	0.806	0.763	5.33%
Scenario 2 ($Q_2 = 0.5$ m^3/s; $C_2 = C_0$)	2.878	2.716	5.63%
Scenario 3 ($Q_3 = 0.14$ m^3/s; $C_3 = 0.28\ C_0$)	0.806	0.581	27.92%
Original simulation ($Q = 0.14$ m^3/s; $C = C_0$)	2.878	2.14	25.64%
Surface water quality guideline for TN in Alberta, Canada (AENV, 1999)	1.0 mg/L		

Table 4.9. The simulation result comparison for TP between the three scenarios and original one

	Inlet (mg/L)	Outlet (mg/L)	Removal
Scenario 1 ($Q_1 = 0.5$ m^3/s; $C_1 = 0.28$ C_0)	0.092	0.074	19.57%
Scenario 2 ($Q_2 = 0.5$ m^3/s; $C_2 = C_0$)	0.328	0.262	20.12%
Scenario 3 ($Q_3 = 0.14$ m^3/s; $C_3 = 0.28$ C_0)	0.092	0.051	44.57%
Original simulation ($Q = 0.14$ m^3/s; $C = C_0$)	0.328	0.164	50%
Surface water quality guideline for TP in Alberta, Canada (AENV, 1999)	0.05 mg/L		

In scenario 1, the amount of contaminants are assumed the same as the original ones, but the flow rate reaches to 0.5 m^3/s from 0.14 m^3/s due to the large rainfall, the concentrations of contaminants in this scenario are diluted. Therefore, the concentrations of TN and TP at the wetland discharge port are only 0.763 mg/L and 0.074 mg/L, respectively. Comparing the water quality results in this scenario to the surface water quality guideline for nutrients in Alberta, Canada, the nutrients in storm water, which would be discharged into the North Saskatchewan River, may not generate a potential of adverse impact on the river in this scenario.

In scenario 2, the amount of contaminants are increased due to the industrial development, even in large rainfall periods, the concentrations of contaminants are assumed the same as the original ones. The concentrations of TN and TP at the wetland discharge port in this scenario are 2.716 mg/L and 0.262 mg/L, respectively. Comparing them to the surface water quality guideline, it can be seen that the concentration of TN at the outlet is 2.7 times larger than it in the guideline, and the concentration of TP is 5.2 times larger. Therefore, the discharge may generate a great potential of adverse impact on the receiving river in this scenario.

Clearly, based on the simulation results for all the three design scenarios and the original case, it can be seen that in the large rainfall periods, when the flow rate reaches to 0.5 m^3/s from 0.14 m^3/s, the TN removal efficiencies in the wetland decrease from approximately 26% to around 5.5%, no matter how much the concentrations of contaminants are at the wetland inlet. Similarly, the TP removal efficiencies decrease from approximately 45% to around 20%. Firstly, these decreasing trends of removal efficiency show the reasonable stability and consistency of the simulation results from the water quality modeling component. Secondly, these results indicate that increasing flow rate could obviously affect removal efficiency during water quality modeling. By contrast, changing contaminant concentrations alone does not affect removal efficiency.

CHAPTER 5 INTEGRATED ASSESSMENT RESULTS FROM THE HYBRID FUZZY-STOCHASTIC APPROACH

5.1 Monte Carlo Simulation Results

5.1.1 Simulation results for TN

5.1.1.1 The mean value and standard deviation

In the HEC-RAS model, β_2^* (rate constant: oxidation of nitrite to nitrate) is identified as a sensitive and key input variable, and it is a random parameter with a certain range (Zison et al., 1978). The mean value (μ) of the parameter β_2^* is determined as 0.35 (1/day) based on the comparison between the simulation results and observed data for TN, Seven values of the parameter were reported by Zison et al. (1978), they are 0.2; 0.25; 0.3; 0.35; 0.4; 0.45 and 0.5 (1/day). Thus, the standard deviation σ can be calculated, σ = 0.108 (1/day^2).

5.1.1.2 Simulation results

A normal distribution of random values for parameter β_2^* can be generated with its mean value and standard deviation. Then, the concentration distribution of TN at the wetland outlet can be obtained. Figure 5.1 illustrates the distribution of

TN concentrations at the wetland outlet. According to the Monte Carlo modeling results for TN, 98% of the TN concentrations at the wetland outlet are in the range of [2.076, 2.192] (mg/L). Those results would be used for risk quantification in the next step.

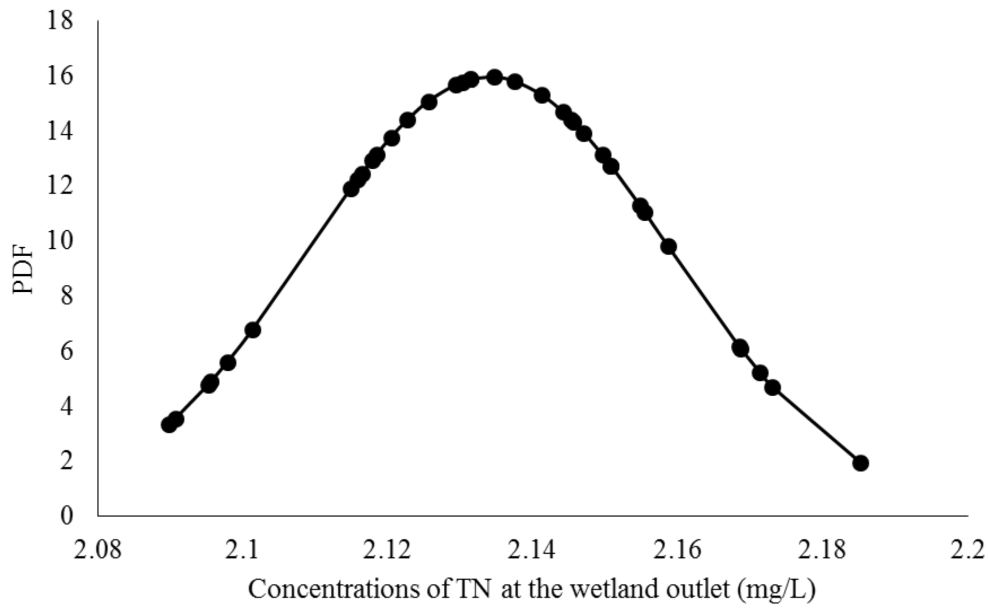

Figure 5.1. The distribution of TN concentrations at the wetland outlet

5.1.2 Simulation results for TP

5.1.2.1 The mean value and standard deviation

In the HEC-RAS model, β_4^* (rate constant: oxidation of OrgP to PO_4) is identified as a sensitive and key input variable also, and it is a random parameter with a certain range (Zison et al., 1978). The mean value (μ) of the parameter β_4^* is determined as 0.6 (1/day) based on the comparison between the simulation results and observed data for TP, Six values of the parameter were reported by Zison et al. (1978), they are 0.45; 0.5; 0.55; 0.6; 0.65 and 0.7 (1/day). Thus, the standard deviation σ can be calculated, $\sigma = 0.097$ (1/day^2).

5.1.2.2 Simulation results

A normal distribution of random values for parameter β_4^* can be generated with its mean value and standard deviation. Then, the concentration distribution of TP at the wetland outlet can be obtained. Figure 5.2 illustrates the distribution of TP concentrations at the wetland outlet. According to the Monte Carlo modeling results for TP, 98% of the TP concentrations at the wetland outlet are in the range of [0.16, 0.2] (mg/L). Those results would be used for risk quantification in the next step.

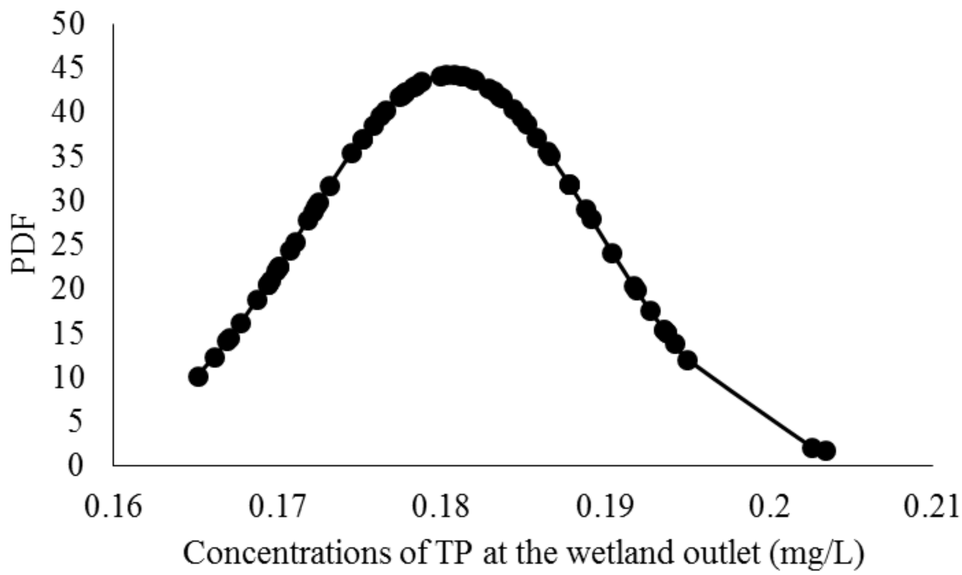

Figure 5.2. The distribution of TP concentrations at the wetland outlet

5.2 Hybrid Fuzzy-Stochastic Risk Assessment Results

5.2.1 Integrated risk assessment results for TN

Table 5.1 summarizes the integrated risk assessment results for TN. The guideline of Alberta, Canada for TN has the membership grade of 0.89 indicating the guideline's high applicability. Under this guideline (TN = 1.0 mg/L), the integrated risk level is 1 which means the guideline has a 100% possibility of being violated during this study period. Similarly, the guideline of US EPA in Ecoregion V for TN has the membership grade of 0.92 indicating the guideline's high applicability. Under this guideline (TN = 0.88 mg/L), the integrated risk level is also 1.

Table 5.1. The integrated risk assessment results for TN

Origin		Regulated value for TN (mg/L)	Integrated risk level	
			Stochastic risk	Membership grade
Alberta, Canada		1.0	1	0.89
United States	USEPA Ecoregion I	0.31	1	0.62
	USEPA Ecoregion V	0.88	1	0.92
	USEPA Ecoregion VI	2.18	0.033	0.63
Netherlands		2.2	0.004	0.626

It can be noticed that the guideline of US EPA in Ecoregion VI for TN has the membership grade of 0.63 indicating the less effective suitability. Under this guideline (TN = 2.18 mg/L), the integrated risk level is 0.033 which means that the guideline only has a 3.3% possibility of being violated. Similarly, the guideline for

TN in the Netherlands has the membership grade of 0.626 indicating that the guideline's suitability is similar to the guideline of US EPA in Ecoregion VI. Under this guideline (TN = 2.2 mg/L), the integrated risk level is 0.004 which means that the guideline only has a 0.4% possibility of being violated during this study period. Therefore, using different guidelines can lead to entirely different risk level results, which indicates the significance of analyzing the practicability of local guidelines.

5.2.2 Integrated risk assessment results for TP

Table 5.2 summarizes the integrated risk assessment results for TP. The guidelines of Alberta, Canada and US EPA in Ecoregion I for TP have the membership grade of 0.89 and 0.9 indicating the guidelines' high applicability. The guidelines of US EPA in Ecoregion V and Ecoregion VI for TP have the membership grade of 0.79 and 0.73 indicating the less effective applicability. By contrast, the guideline for TP in the Netherlands has the membership grade of only 0.3 indicating the guideline's low applicability. Though the guidelines for TP are different in these countries and regions, the integrated risk levels are all 1, which indicates that the TP concentrations in the wetland effluents are very high, all the guidelines have the 100% possibility of being violated during this study period.

Table 5.2. The integrated risk assessment results for TP

Origin		Regulated value for TP (mg/L)	Integrated risk level	
			Stochastic risk	Membership grade
Alberta, Canada		0.05	1	0.89
United States	USEPA Ecoregion I	0.047	1	0.9
	USEPA Ecoregion V	0.067	1	0.79
	USEPA Ecoregion VI	0.076	1	0.73
Netherlands		0.15	1	0.3

CHAPTER 6 DISCUSSIONS

In this chapter, the possible reasons of differences between the simulation results and the monitoring data, the comparison of flow simulation between two water quality models, and causes of the removal efficiency decrease are further discussed in below.

(1) According to the comparison between the simulation results from the HEC-RAS model and the observed data from the Kennedale wetland, the differences (or errors) between the simulation results and observed data for TN and TP are 0.05% and 6.1%, respectively. These differences and related uncertainties are attributed to:

a) When a model is chosen to simulate water quality for a field case, various input data are required, including geographical data, meteorological data, hydraulic data and water quality data, etc. It is common in environmental modeling studies that not all the required input data can be obtained from the study case. A few of them have to be assumed according to references related to similar field cases. For example, the Manning's n is an important parameter for the channel modeling. The value of Manning's n is highly variable and determined by the factors such as channel surface roughness, vegetation, channel irregularities, size and shape of the channel and suspended material, etc. Among these factors, the most important factors for determining the value of Manning's n for a particular field case are the type and size of materials of the channel and the channel shape (Brunner, 2010b). As the soil in the bottom of the Kennedale wetland is clay. Thus, 0.03 was chosen as the Manning's n value in this case (Brunner, 2010b). Additionally, the uncertainties of essential parameters can be further examined through a fuzzy-stochastic modeling method.

The acquirability and accuracy of input data are essential for the model choice and simulation performance. The reasons of differences between simulation results and on-site field data are various, lack of accurate input data could be one of them. For example, in this study, the flow simulated in the HEC-RAS model is more reasonable than in the QUAL2K model. The HEC-RAS model is determined as the better one for this field case, one of the reasons may be that more precise input data can be obtained for water quality simulation in HEC-RAS than in QUAL2K.

b) The HEC-RAS model can be applied for simulating the contaminant removal performances in the Kennedale wetland system based on the comparison between the observed data and the simulation results. However, the variables (such as flow velocity) analyzed in the HEC-RAS model, only change in one direction along the channel (Robinson, 2012). Furthermore, the flow modeled in HEC-RAS is a steady gradually varied flow, where the velocity and depth vary along the channel path but are time-independent (Brunner, 2010b). In the field-scale study of this research, the flow in the Kennedale wetland is more complicated.

c) A constructed wetland is a complex system associated with a number of uncertainties. The mechanisms of contaminant removal in a constructed wetland are comprehensive processes including various physical and biochemical reactions. According to the observed water quality data from the Kennedale wetland (City of Edmonton drainage services, 2012), the water quality data and contaminant removal efficiencies vary widely in different seasons and days. Therefore, differences between the simulation results from a model and the observed data from a real field case are acceptable when they are under a certain level (Jia and Culver, 2006).

(2) Although the HEC-RAS model and the QUAL2K model can both be applied for simulating this field case, the HEC-RAS model is considered as the better one which is not only due to its simulation results, but also because of its modeling system. The HEC-RAS model has an independent modeling module for simulating flow. The simulated flow must be built in the HEC-RAS model before

the water quality analysis. To build the flow in the model, many cross sections of the channel should be constructed, as shown in Figure 4.3. Dozens of station and elevation data (X-Y data) from left to right must be inputted for determining each one cross section as illustrated in Figure 4.4. The Figure 4.5 also describes the X-Y-Z perspective plot of flow in the Kennedale wetland constructed using the HEC-RAS model.

On the other hand, the flow calculations in the QUAL2K model are easier as shown in equations from equation (2.10) to (2.12). Simple power equations are applied to compute the mean velocity and depth of flow (equation 2.10), and then the cross-sectional area and width of the channel can be approximately calculated (equation 2.11). Comparing to the flow calculation in the QUAL2K model, the flow simulation in the HEC-RAS model is more representative based on its consideration of cross section geometry of the wetland. It may be an important reason that water quality simulation results from HEC-RAS are better than the results from QUAL2K.

(3) The increase of flow rate can obviously affect removal efficiency based on the simulation results of the design scenarios, but changing contaminant concentrations alone does not. The mechanism of the transport and fate of

contaminants in the HEC-RAS model may reflect the reasons, and is based on the following equation (Adapted from Brunner, 2010b):

$$\partial C / \partial t = -v \partial C / \partial x + \partial (\Gamma \partial C / \partial x) / \partial x \pm S \tag{6.1}$$

where C = concentration (mg/L); v = average flow velocity (m/s); Γ = dispersion coefficient (m^2/s); t = time (s); x = distance (m) and S = sources and sinks (mg/L· s).

It can be seen from equation (6.1) that the contaminant removal efficiency is related to three processes: advection; dispersion; and sources and sinks (S). As the sources and sinks (S) process mainly changes with the variation of contaminant concentration, the advection and dispersion processes may be affected due to the flow rate increase.

Based on the flow simulation results, when flow rate = 0.14 m^3/s, the flow velocity = 0.0147 m/s, and when flow rate = 0.5 m^3/s, the flow velocity = 0.0115 m/s. So, increasing flow rate can slightly decrease the flow velocity. Based on the equation (6.1), the advection process would be reduced also, and then, the removal efficiencies could be decreased.

During the water quality simulation process, the dispersion coefficient (Γ) is calculated by the HEC-RAS model itself based on hydraulic variables. The equations for calculating dispersion coefficient (Γ) in HEC-RAS are as follows (Brunner, 2010b):

$$\Gamma = \frac{u^2 w^2}{y u^*} \tag{6.2a}$$

where Γ = dispersion coefficient (m^2/s); u = face velocity or average linear velocity (m/s); w = average channel width (m); y = average channel depth (m) and u^* = shear velocity (m/s).

$$u^* = \sqrt{gdS} \tag{6.2b}$$

where u^* = shear velocity (m/s); g= gravitational constant (9.81m/s^2); d= average channel depth (m) and S= friction slope.

Firstly, the increase of flow rate can slightly decrease the flow velocity as mentioned above, the face velocity (or average linear velocity, u) would also decrease. Secondly, when the flow rate reaches to 0.5 m^3/s from 0.14 m^3/s in large rainfall periods, the water surface level in the wetland increases accordingly, then the channel depth would increase. Lastly, because of the channel depth increase, the shear velocity (u^*) would also increase according to equation (6.2b). In sum, due to the integrated change which the face velocity (u) decreases, and the shear velocity (u^*) and channel depth both increase, the dispersion coefficient (Γ) would decrease obviously based on equation (6.2a), then the dispersion process would reduce also. Therefore, the reduction of both advection and dispersion processes would lead to the worse removal efficiency.

CHAPTER 7 CONCLUSIONS

7.1 Summary of the Research

A hybrid fuzzy-stochastic modeling approach has been proposed in this thesis to analyze the environmental impact related to storm water discharges from the constructed wetland to a river, and to quantify uncertainties in the modeling system and the related water quality guidelines. Combining the Monte Carlo modeling method with the water quality model, a stochastic simulation is conducted to generate concentration distributions of nutrients in the wetland effluents. This stochastic simulation serves as the basis for the risk assessment. The triangle fuzzy membership functions are established to reflect the environmental impacts of nutrients and the suitability of nutrient guidelines, which is incorporated into the Monte Carlo modeling framework to identify the integrated risks from the discharge on the river. The quantification of risks on receiving river resulting from nutrients in storm water discharges is then implemented based on the hybrid fuzzy-stochastic analysis results.

Two water quality models HEC-RAS and QUAL2K are incorporated and examined through the developed fuzzy-stochastic modeling approach to simulate the field-scale wetland and to analyze the removal efficiency of contaminants by the wetland. The simulation work is applied to the Kennedale wetland located in the city of Edmonton, Canada. Reasonable results are obtained. According to the simulation results from the HEC-RAS model and the QUAL2K model, the removal efficiencies of TN by the wetland are 25.64% and 13.59%, respectively. The removal efficiencies of TP are 50% and 50.91%, respectively. The differences between the HEC-RAS simulation results and on-site field data are 0.05% for TN and 6.1% for TP, and the differences between the QUAL2K results and on-site field data are 13.99% for TN and 4.35% for TP based on this study. The water quality simulation results from the two models are acceptable, and the two models can both be applied for simulating this field case. With the analysis of the two modeling systems and the comparison between their water quality simulation results, the HEC-RAS model is determined as better than the QUAL2K model on modeling this field case. The validated and customized HEC-RAS model is integrated with the environmental risk assessment process and applied for analyzing and predicting water quality changes in three design scenarios. The simulation results of the design scenarios indicate the reasonable stability and consistency of the simulation results from the developed modeling approach.

According to the integrated risk assessment results, the concentrations of TN in the wetland effluents in this field case have the 100% possibility to violate the guidelines of Alberta, Canada and US EPA in Ecoregion V which are both highly suitable. Similarly, the concentrations of TP in the wetland effluents have the 100% possibility to violate the guideline of Alberta, Canada and US EPA in Ecoregion I during this study period. Therefore, the nutrients in storm water discharges from the Kennedale wetland in this field case may have a great potential to adversely affect the receiving river (North Saskatchewan River) at the time of this study.

7.2 Contributions of the Research

Based on the study mentioned above, the contributions of this thesis are summarized as follows:

(1) A hybrid fuzzy-stochastic modeling approach has been developed to examine the wetland treatment efficiency, to analyze the environmental impact associated with the wetland effluents into the receiving water, and to quantify system uncertainties. It combines and extends a water quality model to systematically analyze the wetland system. The integrated risk assessment results indicate that the nutrients in storm water discharges from the Kennedale wetland in this field case may have a great potential to adversely affect the receiving river (North Saskatchewan River) at the time of this study.

(2) The developed fuzzy-stochastic modeling approach, which includes a customized water quality model and a risk assessment model, has been applied to the Kennedale wetland in the city of Edmonton. The full-scale validation indicates that the developed modeling approach is useful for the practical managing of wetland systems and the impact of the wetland discharges on the receiving waters.

(3) Representative water quality models (the HEC-RAS model and the QUAL2K model) are systematically assessed for simulating the transport and fate of nutrients in a surface flow constructed wetland and analyzing the nutrient removal performance by the wetland, which are rarely applied for modeling field-scale wetlands in previous modeling studies. Thus, more modeling tools are available to help the design and operation of constructed wetland systems. The validated and customized water quality model is applied for analyzing and predicting water quality parameter changes in three design scenarios, and integrated with the environmental risk assessment process. It can also be used to predict water quality of the Kennedale wetland in the future. This study is the first such study in Canada and can serve as a reference for modeling studies of similar long and narrow wetlands or streams in the future.

(4) The developed fuzzy-stochastic approach could not only quantify different system uncertainties, but also examine the variation of guidelines or standards. The uncertainty analysis methods including stochastic simulation approach and fuzzy logic approach are rarely applied to field-scale wetlands before. The analysis results of nutrients guidelines have supported the management of decision-making process.

7.3 Future Studies

The possible future studies which can improve the study in this thesis are presented as follows:

(1) In this research, only two key model parameters (β_2^* and β_4^*) are studied and quantified their randomness and uncertainty. More key model parameters and the relation between different key model parameters could be further studied in order to better assess system uncertainties.

(2) Uncertainties associated with water quality guidelines and other components are comprehensive, thus, other fuzzy logic approaches, for instance, nonlinear L-R membership functions, could be used to better address other aspects (e.g. nonlinearity) of uncertainties with the help of more on-site field data.

(3) Unsteady flow simulation module, or other 2D or 3D numerical models could be applied to conduct better simulation on flow and transport and fate of contaminants due to the complexity of constructed wetland system.

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