WASTEWATER TREATMENT USING VEGETATED SUBMERGED BED CONSTRUCTED WETLANDS FOR NITROGEN REMOVAL

By

GILBERT K. CABOUTLOELOE

2006

WASTEWATER TREATMENT USING VEGETATED SUBMERGED BED CONSTRUCTED
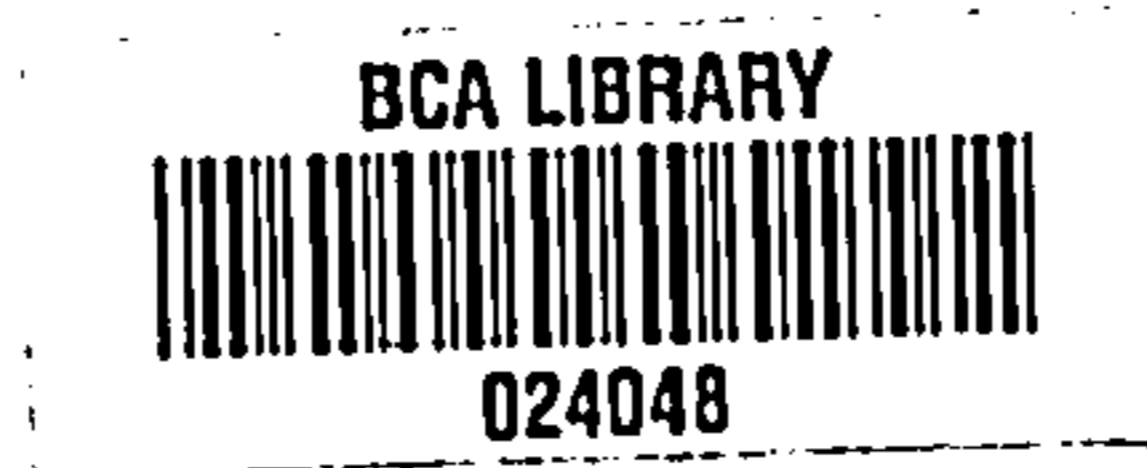
WETLANDS FOR NITROGEN REMOVAL

By

GILBERT K. GABOUTLOELOE

A dissertation submitted in partial fulfillment of

The requirements for the degree of

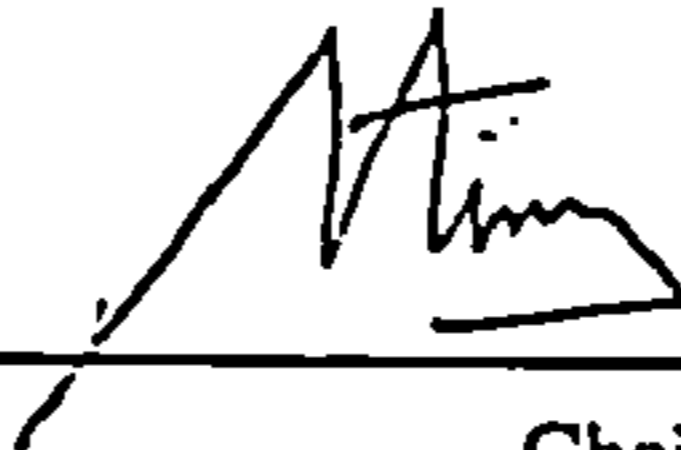DOCTOR OF PHILOSOPHY IN ENGINEERING SCIENCE

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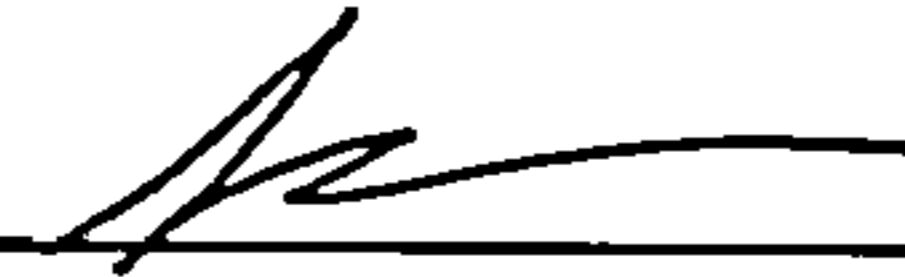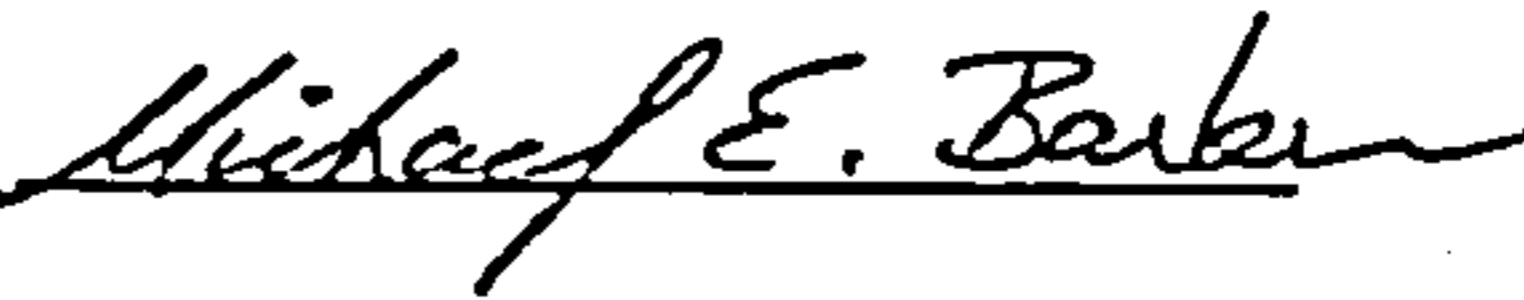Department of Biological Systems Engineering

DECEMBER 2006

To the Faculty of Washington State University

The members of the committee appointed to examine the dissertation of GILBERT K. GABOUTLOELOE find it satisfactory and recommend that it be accepted.

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Chair

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Michael E. Barber

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WASTEWATER TREATMENT USING VEGETATED SUBMERGED BED CONSTRUCTED WETLANDS FOR NITROGEN REMOVAL

Abstract

By Gilbert K. Gaboutloeloe, Ph.D.
Washington State University
December 2006

Chair: Claudio O. Stöckle

Vegetated submerged bed (VSB) constructed wetlands are engineered, low operation and maintenance cost wastewater and stormwater treatment technologies primarily designed for small facilities. The objectives of this dissertation were to evaluate the processes responsible for spatial distribution of nitrogen in VSB constructed wetlands in order to improve the nitrogen removal performance of this technology and present a model on direct atmospheric oxygen-transfer into VSB constructed wetlands for rational system design. A design criterion for enhancing nitrogen removal in VSB formulated from reported studies was also suggested.

Results from the nitrogen spatial distribution study conducted using a laboratory scale VSB constructed wetland demonstrated that there was vertical variation in nitrogen content in the water column of the VSB wetland system. The distributions of ammonium and nitrate nitrogen suggested that there was higher nitrification rate near the air/water interface than deeper into the water column, contrary to the assumption in plug flow and completely mixed tank reactor models currently used for modeling VSB wetlands. The distributions may be attributed to direct

atmospheric oxygen diffusion and/or plant supply in upper level and longer retention time associated with flow resistance in the deeper zone caused by high root density.

The direct atmospheric oxygen transfer accountable for nitrification in VSB was modeled using the modified two-film theory. The model satisfactorily predicted the vertical dissolved oxygen (DO) profiles when compared observed data. Oxygen fluxes predicted from this model ranged from 0.027 to 0.32 g m^{-2} day^{-1}. These are on the lower end of the range reported in literature and therefore can be used for conservative system design.

Several emerging technologies for the enhancement of VSB aeration for improved nitrification were reviewed. The technologies reviewed demonstrated promising potential for nitrification and denitrification. However the net accumulation of nitrate nitrogen generated from the nitrification process was found to be the most limiting factor. This resulted from the depletion of organic carbon necessary for denitrification during aerobic respiration phase of the system. A design criterion to alleviate the accumulation of nitrate nitrogen was suggested. Further development of these technologies to alleviate the nitrate nitrogen accumulation is recommended.

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MODELING OXYGEN TRANSFER INTO VEGETATED SUBMERGED BED CONSTRUCTED WETLAND.

Dedication

Daughter of the soil, born and raised in a round mud and cow dung hut plays cheerfully though barefooted in the dusty streets of Mafikeng a South African city dubbed the city of goodwill. Her father by the way, a visionary, educator, historian and an agriculturalist of repute decided together with his family to emigrate for greener pastures in the late 1950's, destination, the then Bechuanaland Protectorate. Upon arrival the sign board discouragingly read "Welcome to Bechuanaland Protectorate Beware of Dust" referring to the dust emanating from the unpaved roads of the then British Protectorate and the present day Botswana. Despite all the harsh conditions as summarized by the welcome sign board the family settled in Kanye village and in the later half of the 1960's a son was born to the daughter of the soil and that was just me.

I dedicate this dissertation to my mother.

CHAPTER ONE

INTRODUCTION

Constructed wetlands are engineered wastewater and stormwater treatment technologies designed primarily for small facilities requiring low operation and maintenance costs. Nitrogen, phosphorus, total suspended solids and carbon are the principal contaminants of municipal and agricultural wastewater usually treated in constructed wetlands. Carbon and nitrogen are removed through bacteria mediated oxidation-reduction processes that usually limit the treatment performance of these systems. Carbon removal in these systems is usually adequate but in most cases nitrogen removal is not sufficient to meet discharge limits (Zhu and Sikora, 1995; US EPA, 2000; Vymazal, 2005; Behrends et al., 2006).

Wastewater carbon and nitrogen are treated in two basic types of constructed wetlands namely the free water surface (FWS) wetland and subsurface flow (SSF) wetland also know as vegetated submerged bed (VSB) wetland (US EPA, 1993; 2000). VSB wetlands are documented to perform better than the FWS wetlands in carbon and nitrogen removal from wastewater (US EPA, 1993). As a more promising technology the performance enhancement of VSB wetland is evaluated in this study. Nitrogen removal by constructed wetlands from various forms of wastewater including municipal and agricultural wastewater mitigates eutrophication of open water bodies. Ammonia nitrogen removal rates are often low because the amount of dissolved oxygen (DO) available for bacterial oxidation of ammonium is usually limited (Hammer and Knight, 1994; US EPA, 2000; Wu et al., 2001). The limited amount of DO available for

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ammonia removal characteristic of most wastewater is exacerbated by competition for DO between the heterotrophic bacteria responsible for oxidation of organic carbon and the autotrophic bacteria that nitrify ammonium nitrogen inherent in most wastewater types (Sharma and Ahlert, 1977; Grady and Lim, 1980). Because of low DO in wetlands, nitrification is considered a rate limiting step for nitrogen removal and therefore this process drives the design of constructed wetlands. Other less important temporary documented pathways of nitrogen removal in constructed wetlands are plant uptake and sorption of ammonium onto wetland medium (Sikora et al., 1995; Zhu and Sikora, 1995).

Design criteria and analysis methods for constructed wetlands have evolved from relationships derived from a technical database quantifying nutrient removal (NATWD, 1993) to advanced models (Mashauri and Kayombo, 2000; Wynn and Liehr, 2001; Kincanon and McAnally. 2004; Langergraber and Simunek, 2005). The early design approach was based upon empirical equations derived only from the relationship between input and output nutrient levels with no consideration for intrasystem processes responsible for the interaction between processes that affect nitrogen removal (Martin and Reddy, 1997). Recent advanced models are useful tools for explaining the complexity of hydraulics in porous medium coupled with many processes involved in contaminant removal (Marsili-Libelli and Checchi, 2005). However, model complexity does not produce proportional insight into the factors affecting contaminant removal due to the difficulty of estimating a large number of parameters inherent in contaminant removal process (Rousseau et al., 2004; Marsili-Libelli and Checchi, 2005). There is the need to conduct studies on constructed wetlands to improve understanding of the system internal factors affecting contaminant removal processes and enhance predictive ability of models.

The availability of DO in VSB is enhanced by the transfer of oxygen from the atmosphere. There are two main pathways responsible for the transfer of atmospheric oxygen into VSB namely direct atmospheric diffusion and plants roots release (Kadlec and Knight, 1996). The atmospheric oxygen diffusion is the dominant pathway (Brix, 1990 and Wu et al., 2001). However, existing models on contaminant removal in VSB do not adequately account for the atmospheric oxygen diffusion and the associated contaminant removal (Mashauri and Kayombo, 2000; Wynn and Liehr, 2001; Langergraber and Simunek, 2005; Marsili-Libelli and Checchi, 2005). It is therefore essential to conduct studies to quantify and model oxygen flux into VSB to account for contaminant removal supported by available DO.

The goal of this dissertation is to evaluate the mechanistic processes in VSB constructed wetlands for rational system design with specific emphasis on spatial nitrogen distribution, modeling of atmospheric oxygen diffusion and assessment of enhanced passive aeration in VSB constructed wetlands. In the second chapter of this dissertation, spatial ammonium nitrogen distribution as a function of pH, chemical oxygen demand, dissolved oxygen and water temperature is investigated in order to better understand physical and biological processes and their impact on nitrogen removal. The third chapter presents an atmospheric oxygen diffusion model for VSB constructed wetlands. is presented. The fourth chapter reviews the feasibility of performance enhancement of subsurface constructed wetland through enhanced passive aeration and proposes a design criterion. The overall result of this work is a collection of investigations intended to improve the performance of VSB wetlands.

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CHAPTER TWO

INTRA-SYSTEM SPATIAL VARIATIONS OF AMMONIUM AND NITRATE NITROGEN IN HORIZONTAL FLOW VEGETATED SUBMERGED BED CONSTRUCTED WETLANDS

ABSTRACT

The spatial distributions of ammonia and nitrate nitrogen, dissolved oxygen (DO), chemical oxygen demand (COD), temperature, and pH and the associated variation in treatment performance within a horizontal flow vegetated-submerged bed (HF VSB) constructed wetland were investigated. Synthetic wastewater consisting of sucrose and ammonium chloride as carbon and nitrogen sources, respectively, was used. The evaluations were made at three cross sections along the longitudinal axis of the wetland with respect to the inlet. Water samples were collected from six depths below the air/water interface at each cross section.

Analysis of covariance of the data for the first two cross-sections closest to the inlet indicated that the shallowest depths exhibited significantly ($p < 0.05$) lower ammonia nitrogen and higher nitrate nitrogen concentrations compared to the other depths. There was no statistical difference of ammonia- and nitrate-nitrogen concentrations between depths at the third cross-section. However there was slightly lower ammonia nitrogen level at the last cross-section. For the two cross-sections, the distributions in ammonia and nitrate nitrogen concentrations were caused by higher ammonia nitrogen oxidation near the air/water interface due to atmospheric oxygen diffusion and longer retention time resulting from flow resistance caused by high root density. Reduction deeper down in the water column due to atmospheric oxygen diffusion was much smaller. Results also indicated that there were higher relative nitrification and COD removals at

the last two deepest depths compared to the middle two depths due to longer retention time. The longer retention time at the two deepest depths was due to low flow velocity induced by preferential flow near the middle zone which in turn was promoted by head difference between the inlet and outlet of the wetland. The DO distribution can be attributed to oxygen utilization by both heterotrophs and nitrifiers upon diffusion into the system near the air/water interface with little or no oxygen diffusing to the deeper layers. The most active part of the VSB system responsible for nitrogen and organic matter removal is shallow depths near the air water interface. Therefore the design of these systems should be based on area required for adequate atmospheric oxygen transfer.

INTRODUCTION

Cost-effective removal of ammonia and nitrate from wastewater discharge continues to be a challenge particularly for small treatment systems. Constructed wetlands are often suggested as a possible low cost treatment alternative to lagoons and sand filters. Sustainable nitrogen removal in horizontal flow vegetated submerged bed (HF VSB) constructed wetlands, as identified by most treatment wetland studies, is a two step process consisting of nitrification by autotrophic bacteria and respiratory denitrification by either anaerobic or facultative heterotrophic bacteria. Other potential removal mechanisms such as plant uptake, sediment adsorption and ammonia volatilization are generally able to account for only a fraction of nitrogen removal (Kadlec and Knight, 1996). Nitrification, which is an oxygen requiring process, is believed to be the primary rate-limiting step in nitrogen removal because HF VSB environments are predominantly anaerobic (Tanner et al., 2002). The limited availability of dissolved oxygen for nitrification in these organic-rich environments is attributed to rapid consumption of oxygen by fast-growing

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heterotrophic bacteria, which out compete slow-growing nitrifiers (Reddy and Patrick, 1984). Apart from limited oxygen initially dissolved within the influent wastewater, the main sources of oxygen in HF VSB wetlands are diffusion from the atmosphere through the water surface and through actively growing plant shoots into root-zone. However, the latter is subject to question as reported in several studies (Stein and Hook, 2005; Wu et al., 2001; Brix and Schierup, 1990).

Ammonia nitrogen removal in HF VSB wetlands is believed to be accomplished through nitrification in the aerobic layer near the air-water interface resulting from atmospheric oxygen diffusion (Wu et al., 2001). The ammonium nitrified in this aerobic layer results in a concentration gradient that causes ammonium in the anaerobic layer to diffuse upward (Patrick and Reddy, 1976; and Hammer and Knight, 1994). In an HF VSB study by Garcia et al. (2003), it was concluded that diffusion and plant supply mechanisms involved in water oxygenation that occur mainly at the top of the gravel beds were the main cause of the decrease in oxidation-reduction potential with depth. Other parameters such as COD and ammonia nitrogen increased. However, in several other HF VSB studies (Bowmer, 1987; Breen and Chick, 1995; and Rash and Lier, 1999), it was demonstrated through tracer tests that there was preferential bottom flow caused by higher root density induced flow resistance in the upper zones. This flow regime resulted in longer retention times and corresponding higher contaminant removal in the upper zones. Contrary to these findings, the current HF VSB design criterion on depth is such that the media does not have to be deeper than the rooting depth (Crites and Tchobanoglous, 1998; Kadlec and Knight 1996), thus attributing oxygen availability to plant aeration flux and not direct atmospheric oxygen diffusion.

The gradients of dissolved oxygen should also be investigated to better understand intra-system ammonia removal dynamics in HF VSB (Wu et al., 2001) and to reconcile the two aforementioned contrasting findings regarding the causes of ammonia gradients in VSB. To improve nitrogen removal design and thus performance of constructed wetlands it is essential to investigate the intra-system environment effects on ammonia and nitrate nitrogen removal. The objective of this study is to investigate variations of ammonia and nitrate nitrogen as functions of dissolved oxygen, COD, temperature, and pH across the length and depth of VSB treatment wetland.

MATERIALS AND METHODS

Wetland system description

A constant flow-through laboratory scale HF VSB treatment wetland constructed inside a greenhouse at Washington State University (Pullman, WA) was used in this study. As illustrated in Figures 2.1 and 2.2, the dimensions of the VSB system were 0.91 m wide, 1.22 m deep and 3.66 m long. The wetland was lined with PVC. The wetland was filled with approximately 0.005 m diameter gravel to a depth of 0.55 m from the bottom of the wetland. The pore fraction of the gravel was 0.22. The inlet and outlet consisted of perforated PVC pipes to distribute and collect the influent and effluent flow uniformly over the widths of the entry and exit zones respectively. The outlet was located at the bottom of the medium and was connected to an adjustable-level outlet pipe to maintain the water level at 0.1 m below the gravel surface resulting in a water depth of 0.45 m from the bottom of the wetland. Sample access tubes were installed at three cross-sections located at longitudinal distances 0.64 m, 1.89 m and 3.14 m from the wetland inlet as illustrated in Figure 2.2. At each cross-section, six 5 mm inner diameter neoprene tubes were

installed vertically to access sampling depths 0.05 m, 0.1 m, 0.15 m, 0.20 m, 0.25 m and 0.30 m below the air water interface. Locally harvested cattails (*Typha latifolia*) adapted to saturated soil conditions were planted in the wetland as shown in Figure 2.3.

To provide steady-state flow conditions for the measurement of ammonium nitrogen and dissolved oxygen gradients, a peristaltic pump was used to supply synthetic wastewater influent from a 906 L storage and mixing tank at approximately 1.9 ml/sec flow rate to achieve a 2 day hydraulic retention time.
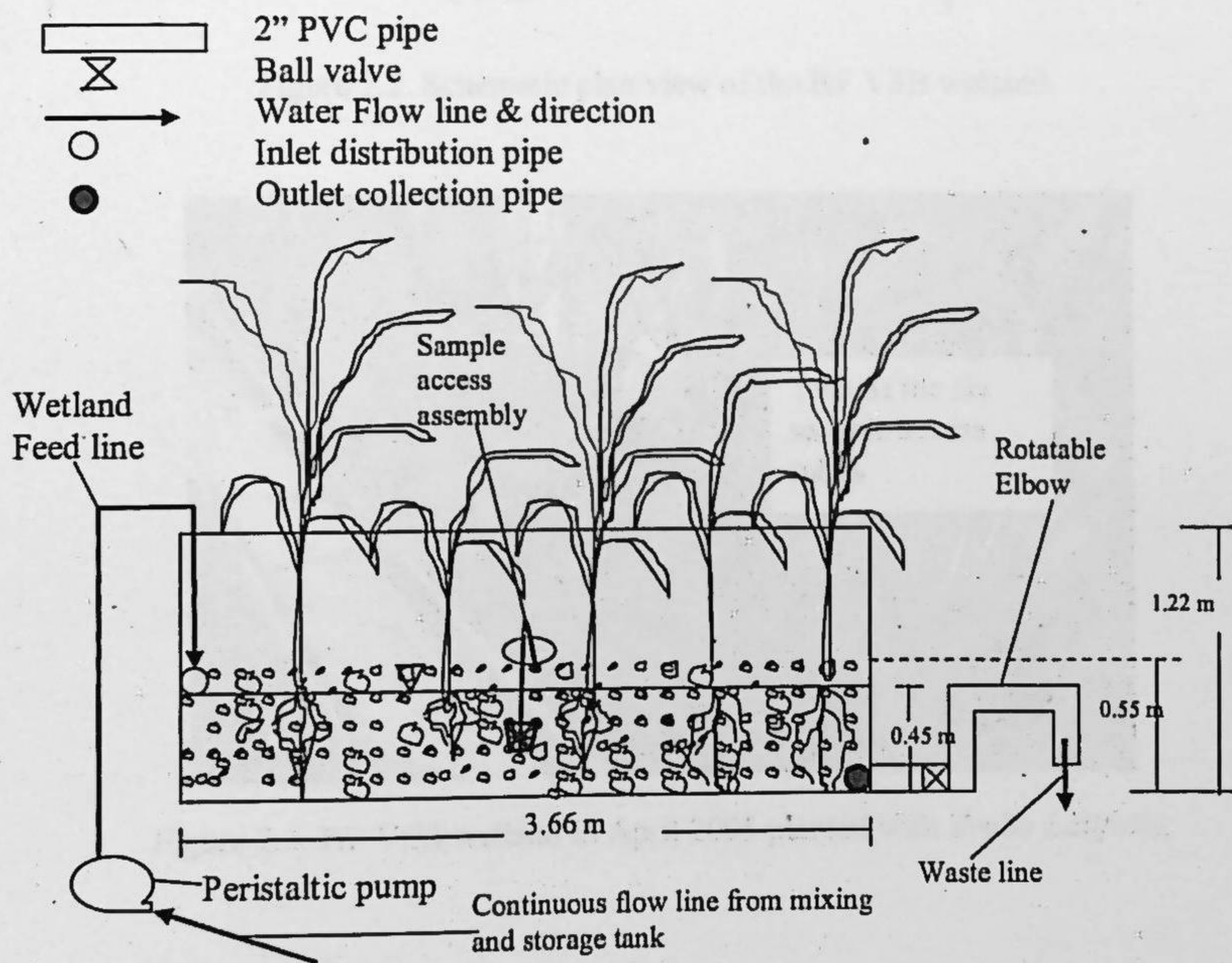
Figure 2.1. Schematic cross-sectional view of HF VSB wetland.
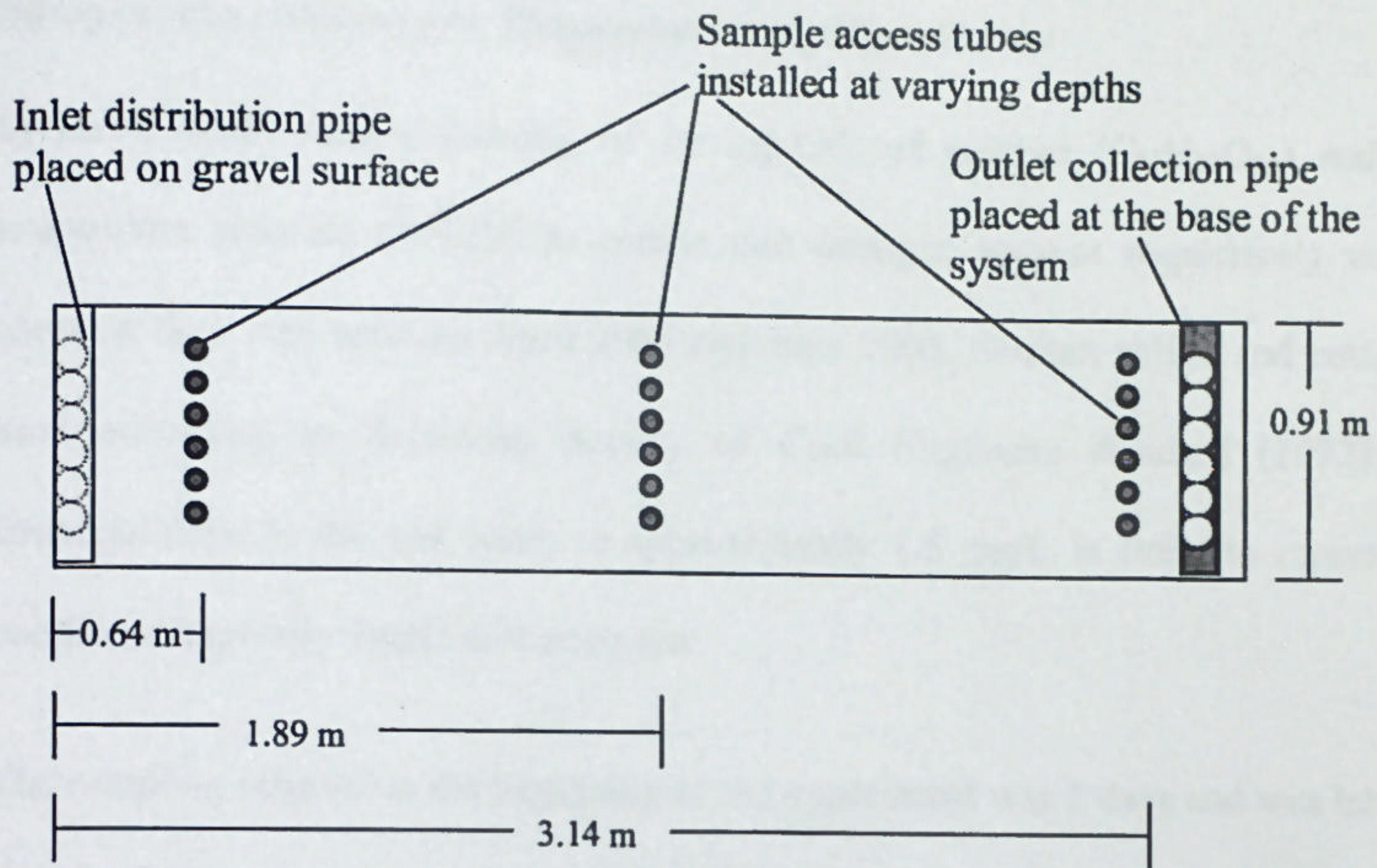
Figure 2.2. Schematic plan view of the HF VSB wetland.
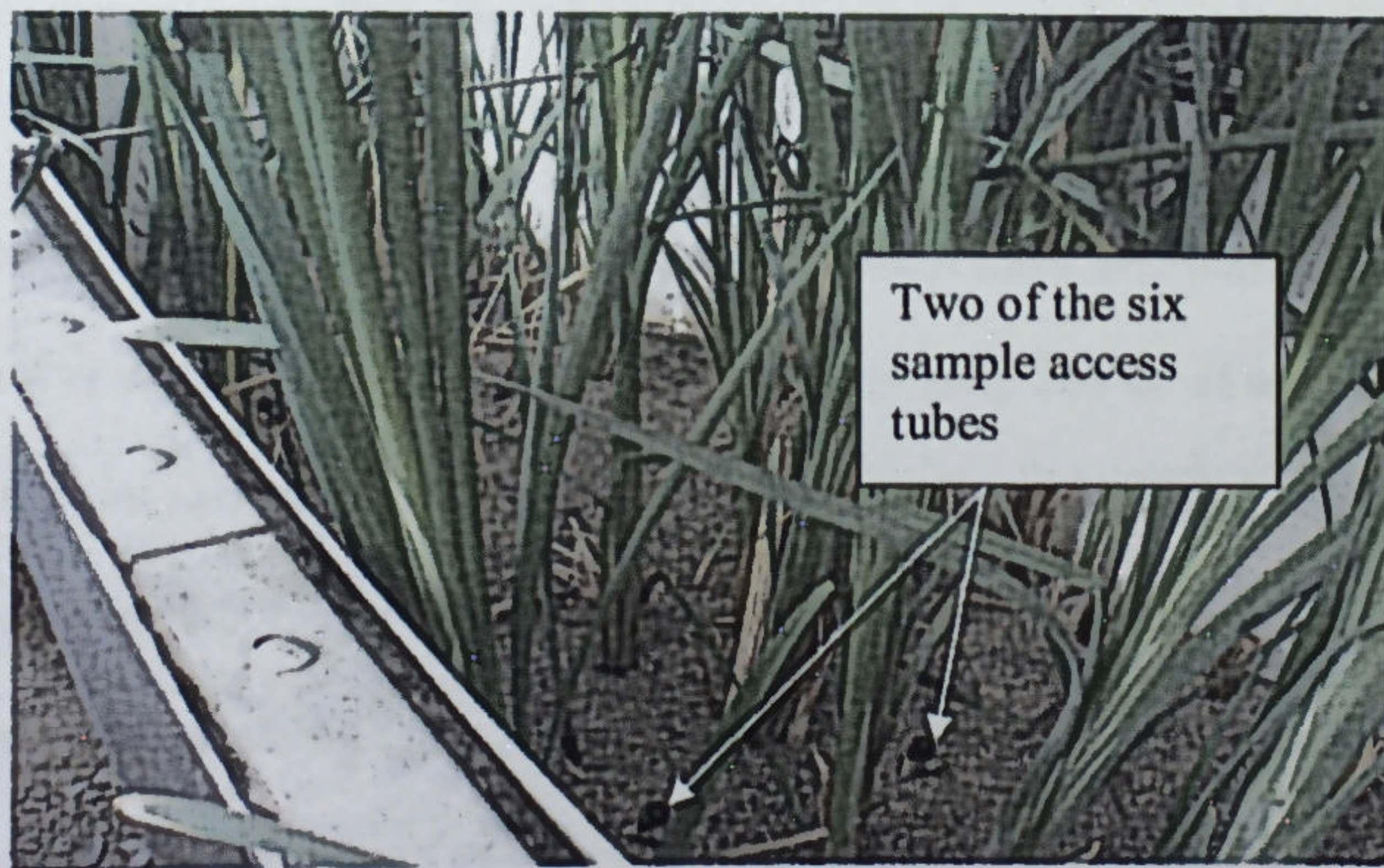
Figure 2.3. HF VSB wetland in April 2005 planted with *Typha Latifolia*.

Nitrogen, Dissolved oxygen, Temperature and pH profile test

Synthetic wastewater consisting of 30 mg-C/L of sucrose ($C_{12}H_{22}O_{11}$) and 50 mg-N/L of ammonium chloride (NH_4Cl) as carbon and nitrogen sources respectively was supplied at a constant flow rate between April 2005 and June 2005. Sodium sulfite and cobalt chloride were used according to American Society of Civil Engineers standard (1992) to reduce DO concentrations in the test water to approximately 1.5 mg/L in order to represent the low DO conditions typically found in wastewater.

The sampling interval at the beginning of the experiment was 2 days and was later increased to 4 then to 8 days to accommodate gradual changes in ammonia nitrogen concentration profiles. It took 50 days from commencement of sampling for the system to reach steady state in terms of ammonia nitrogen concentration after which the sampling interval was adjusted back to 4 days.

A sampling kit consisting of a vacuum hand pump, conical flask, and plastic tubing as illustrated in Figure 2.4 was used to extract the samples. Approximately 200 ml of sample was collected at each sampling point and 60 ml of sample was stored in an ice chest before being transferred to an US EPA approved laboratory where the samples were stored frozen awaiting analysis as recommended by *Standard Methods* APHA (1995). The remaining 140 ml sample was then used for on site measurement of dissolved oxygen and temperature by Extech® DO meter (accuracy: ± 0.4 mg/L and ± 0.8 °C, respectively). The stored samples were analyzed for ammonia and nitrate nitrogen by US EPA colorimetric automated phenate and colorimetric automated cadmium reduction methods 350.1 and 353.2, respectively (US EPA, 1983). The stored samples were also used for analysis of COD by closed reflux colorimetric standard method 5220 D (*Standard*

Methods APHA, 1995) and measurement of pH by Fisher Scientific accumet portable AP5® pH meter (accuracy ± 0.01 pH).
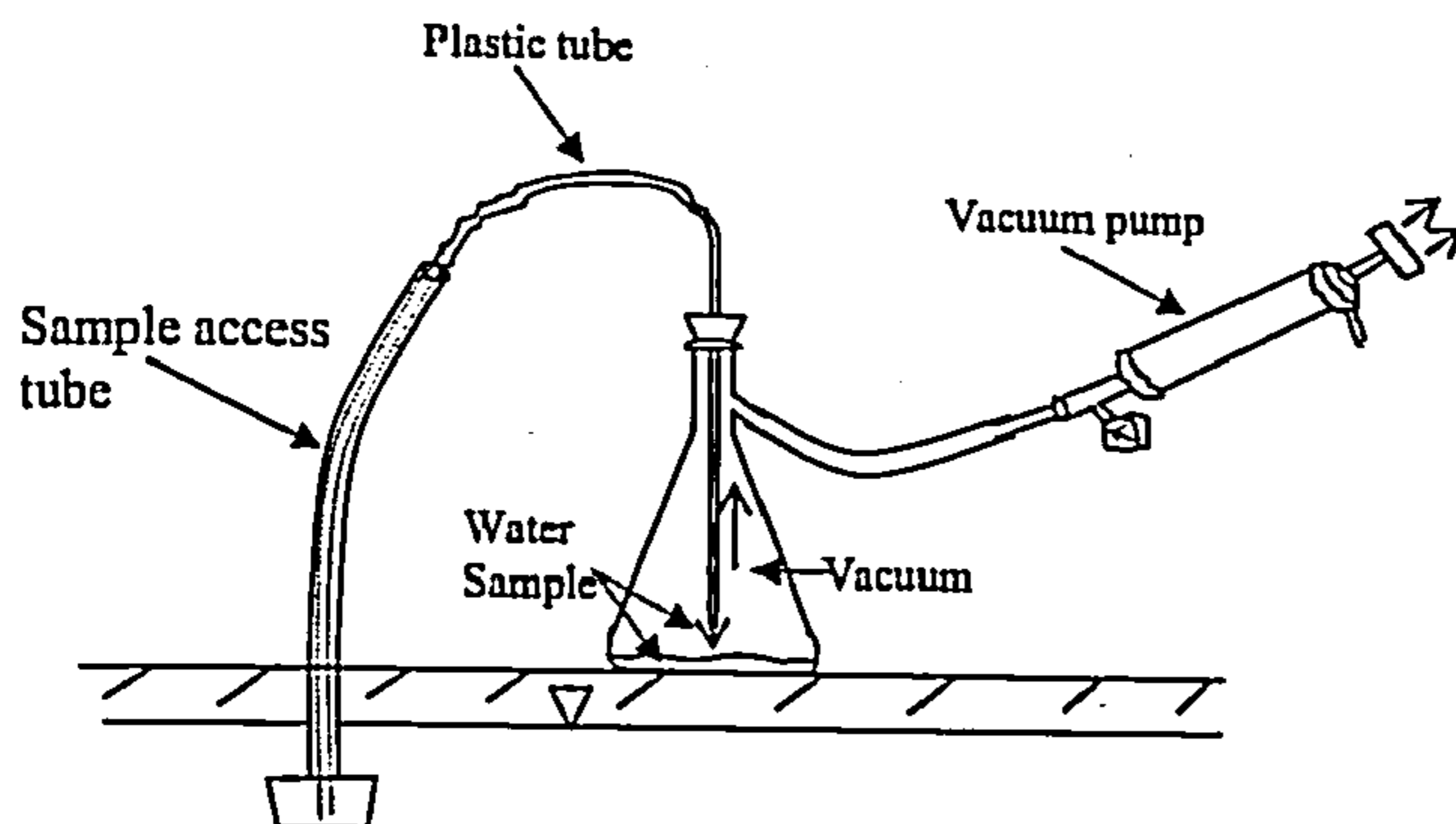
Figure 2.4. Schematic sample extraction kit assembly.

Statistical analysis

Analysis of covariance (ANCOVA) was used to determine if ammonia and nitrate nitrogen, COD, DO, pH and temperature varied significantly between the six depths. It is essential that the slopes of the variables' response as a function of depth are equal so that comparison between any two depths through time can be meaningful. This would imply that time does not affect the responses differently between depths and therefore eliminates time as a possible cause of variance. A significant interaction between time and depth means the slopes of depths versus time are not the same and thus nullifying comparisons. The first step of the NACOVA procedure was then to determine the equality of the slopes of response versus time for the different depths for all variables. The second step was to determine if there was significant difference between responses at different depths. For the significant differences the Least Squared Means (LSM)

multiple comparison procedure was used to determine between which depths the differences existed (Dean and Voss, 1999).

RESULTS AND DISCUSSION

Ammonia and Nitrate nitrogen spatial variations

The overall system performance in ammonia nitrogen removal as illustrated in Figure 2.5 show that the system reached a steady state flow condition after 50 days from the beginning of sampling. The increase in ammonium concentration prior to reaching steady state condition is accounted for by adsorption to the media until saturation. The slight fluctuation in ammonia nitrogen concentration after steady state is reached can be attributed to adsorption and desorption cycles of ammonium nitrogen on gravel and roots surfaces as demonstrated in a similar study (Nakano, 1999).The nitrogen spatial distribution is therefore analyzed from data collected after steady state condition was reached.

The spatial distribution results from this study showed the vertical distribution of both ammonia and nitrate nitrogen can be categorized into top, middle, and bottom zones. As demonstrated in Figure 2.6, the two shallowest depths tend to best represent trends near the wetland surface and the middle two represent middle zone and the remaining deeper depths represent the bottom zone. The top zone experienced higher decrease in ammonia nitrogen and higher consequent nitrate nitrogen levels compared to the other two zones. Ammonia nitrogen level in the bottom zone is lower than in the middle zone.

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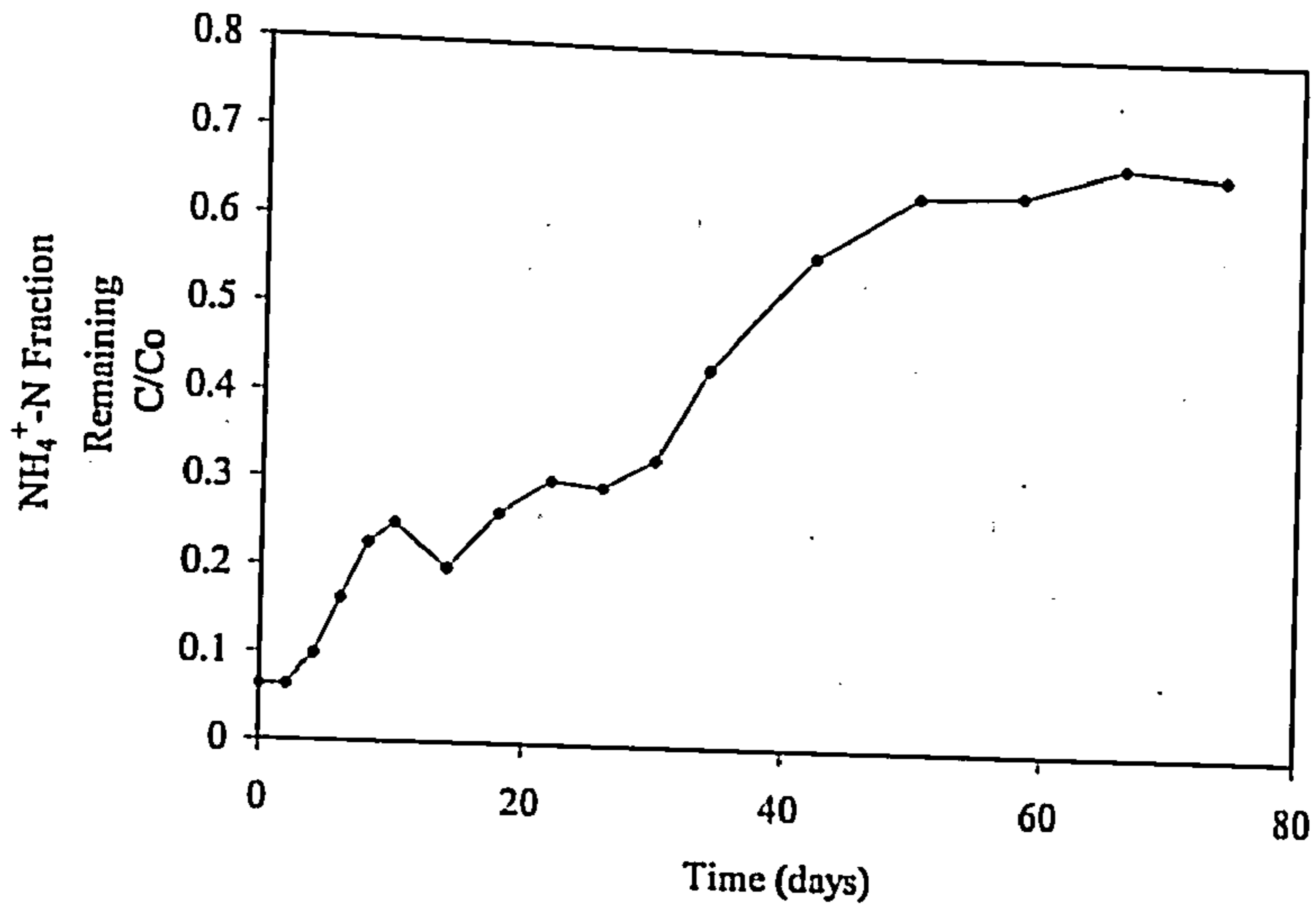
Figure 2.5. VSB break through curve illustrating duration to steady state

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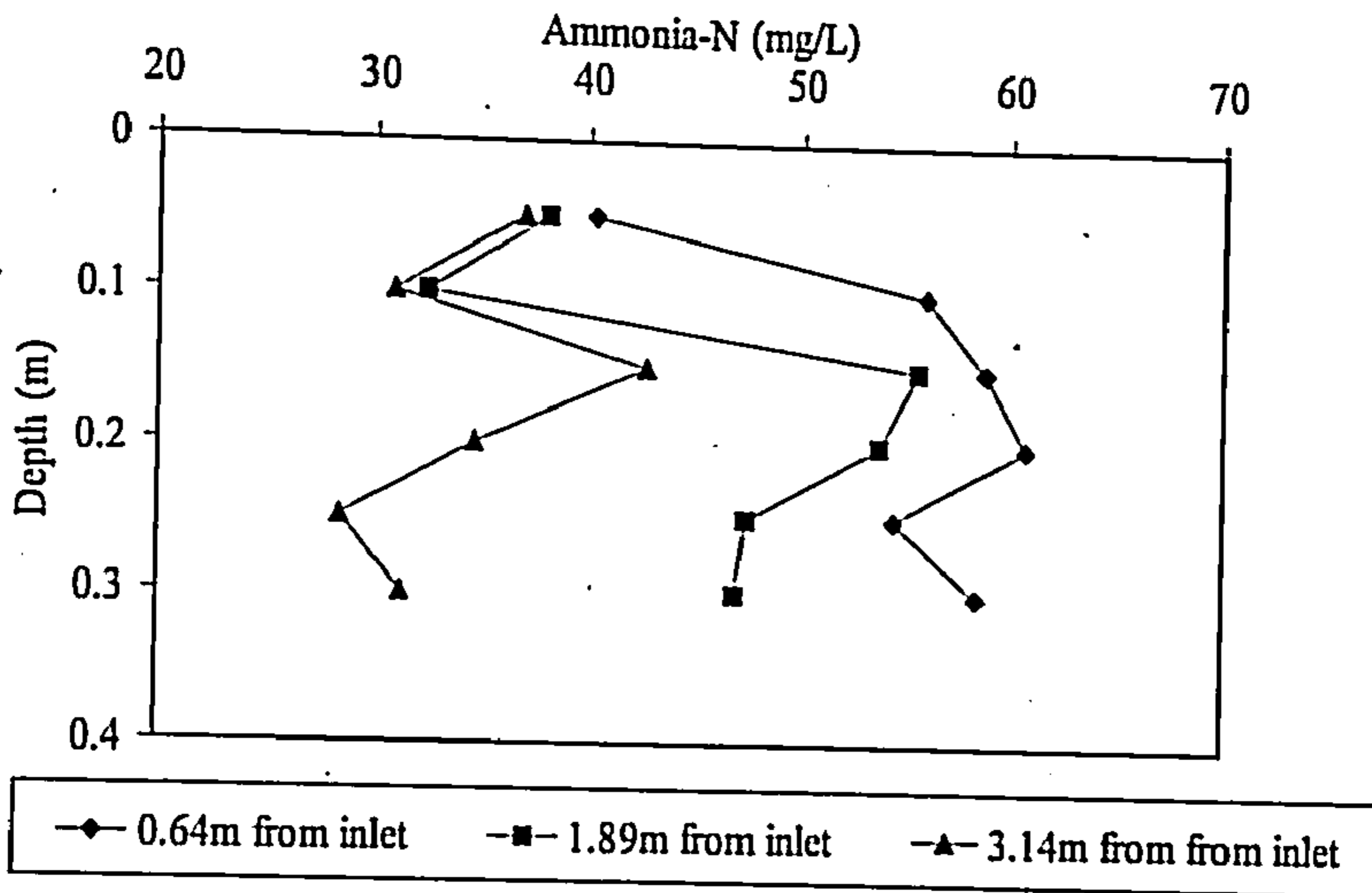
Figure 2.6. VSB pore water profiles of Ammonium-N concentration at the three sampling sites along the longitudinal axis
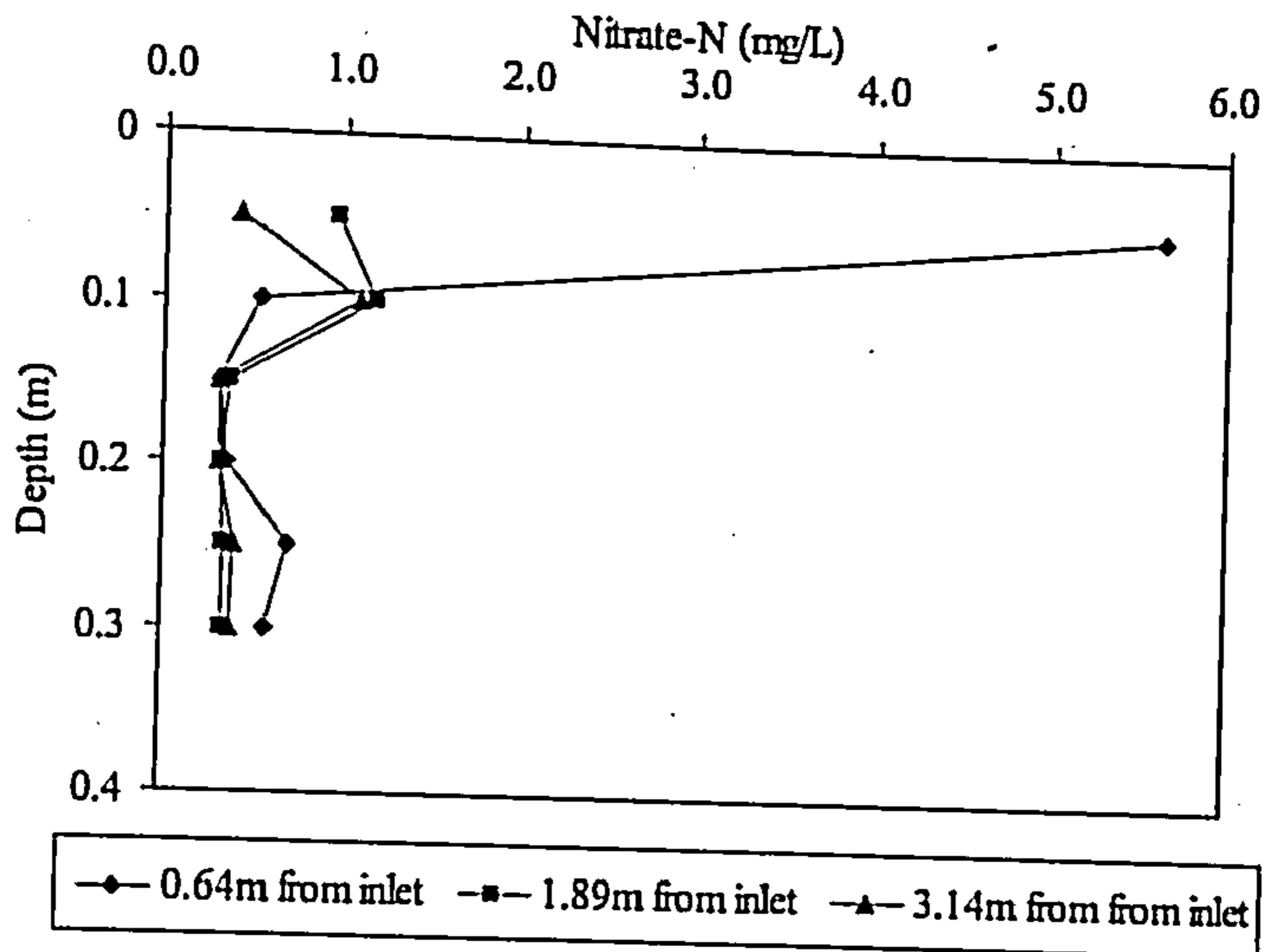
Figure 2.7. VSB pore water profiles of Nitrate-N concentration at the three sampling sites along the longitudinal axis

In addition to the described general nitrogen distribution trends, the three cross sections displayed statistically different levels of non-uniformity in vertical ammonia- and nitrate- N concentration profiles as shown in Tables 2.1 and 2.2. The values shown in the tables reflect the average of samples analyzed after steady state was reached. Values with the same superscript letters under each cross section and each depth are not significantly different at alpha level of 0.05. The mean concentration for 0.05 m depth, the shallowest depth, at the cross section located 0.64 m from inlet, was the only observation that was statistically lower for ammonia-N and higher for nitrate-N (p < 0.05) than mean concentration observed for all other depths at the same

cross section. All of the other remaining depths experienced statistically insignificant mean concentration differences ($p > 0.05$) amongst each other. The vertical distribution of ammonia-N and nitrate-N at the middle cross section experienced a slightly different trend than the first cross section. At this cross section, the top zone experienced statistically lower ammonia-N and higher nitrate-N levels than the other two zones. The results for the third cross section demonstrated no significant differences between depths for both ammonia-N and nitrate-N levels.

From the above description of the vertical distribution of nitrogen, it is clear that the top zone experienced the highest decrease in ammonia nitrogen followed by the bottom zone and lastly the middle zone. The top zone nitrate-N distribution corresponds to that of ammonia-N indicating that ammonia-N trends can be explained by nitrification. There are several studies on similar systems that explain through tracer tests, the distribution as seen between top and middle zones (Bowmer, 1987; Breen and Chick, 1995; Garcia et al., 2003; Headley et al., 2005). These will be discussed in more detail later in this section. There are no reported studies on nitrogen distribution demonstrating the results for the bottom zone similar to those found in this study.

Table 2.1. Mean Ammonia-Nitrogen concentration at the six sampling depths along the length of the pilot vegetated submerged bed constructed wetland.

Sample depth, m	Ammonia-N, mg/L (Standard deviation)		
	Cross Section Distance from inlet; m		
	0.64	1.89	3.14
0.05	40.3[a] (2.8)	38.1[ac] (2.6)	37.0[aig] (3.0)
0.10	56.0[b] (2.8)	32.4[c] (2.6)	30.9[cf] (3.0)
0.15	58.8[b] (2.8)	55.6[bd] (2.6)	42.7[g] (3.0)
0.20	60.7[b] (2.8)	53.7[de] (2.6)	34.7[fg] (3.0)
0.25	54.4[b] (2.8)	47.4[e] (2.6)	28.4[f] (3.0)
0.30	58.4[b] (2.8)	46.9[e] (2.6)	31.3[f] (3.0)

Table 2.2. Mean Nitrate-Nitrogen concentration at the six sampling depths along the length of the pilot vegetated submerged bed constructed wetland.

Sample depth, m	Nitrate-N, mg/L (Standard deviation)		
	Cross Section Distance from inlet; m		
	0.64	1.89	3.14
0.05	5.60[a] (0.79)	0.95[c] (0.18)	0.41[cb] (0.12)
0.10	0.53[bf] (0.79)	1.17[cf] (0.18)	1.08[cf] (0.12)
0.15	0.31[b] (0.79)	0.36[b] (0.18)	0.31[b] (0.12)
0.20	0.37[b] (0.79)	0.32[b] (0.18)	0.31[b] (0.12)
0.25	0.70[b] (0.79)	0.34[b] (0.18)	0.39[b] (0.12)
0.30	0.57[b] (0.79)	0.34[b] (0.18)	0.38[b] (0.12)

Compared to the middle zone, there is an indication of higher nitrification rate in the bottom zone demonstrated by the distribution of ammonia-N between the two zones. If the interstitial water velocity were the same between the two zones, higher nitrification rate would be expected in the middle zone than the bottom zone because the middle zone is closest to both the higher root density and the air/water interface which are responsible for wetland oxygen supply. Since the opposite trend is observed, other possible causes of this phenomenon need to be explored.

One alternative possibility is that a residence time difference exists between the two zones. For higher nitrification to occur in the bottom zone than in the middle zone the residence time in the bottom zone has to be longer. This implies lower interstitial water velocity in the bottom zone which is probably caused by the head difference between the system inlet and outlet.

As illustrated in Figure 2.2 the inlet pipe is placed on the gravel surface while the outlet pipe is placed at the base of the wetland. Due to this configuration the inlet and outlet experience highest and lowest hydraulic heads respectively. Therefore water would tend to flow faster in a zone above than below the diagonal line between the inlet and outlet in response to hydraulic head difference between the two sites. This effect results in lower interstitial water velocity near the bottom zone than middle zone of the wetland thus leading to higher residence time in the bottom zone.

The differences in nitrogen levels between the top and middle zones can be explained in part through examination of similar studies listed in Tables 2.3 and 2.4. In Table 2.3, the factors considered in comparing other studies with this study include system dimensions; sampling design (investigated distances from the inlet along the system length and sampled depths); gravel size; porosity; and nominal interstitial water velocity. It should be noted that the plants grown in this study were cattails compared to common reed for Garcia et al. (2003) and soft bulrush for the other studies. The nitrogen vertical distribution and tracer test study results listed in Table 2.4 were evaluated to explain the cause of observed distributions in this study. The top, middle, and bottom depths listed in Table 2.4 correspond to the two or three study depths listed in Table 2.3. Superscript 1 in Table 2.4 represents no difference in tracer residence time between depths while

superscripts 2 and 3 represent bottom and mid zones preferential flow as reflected by shorter tracer residence time respectively. Nominal hydraulic residence time (nHRT) in Table 2.4 is calculated from theoretical interstitial water flow velocity and distance of the monitored cross section from the inlet. The relative magnitude of the physical features of this study and literature data were considered when making comparisons because differences between the study scales can have an impact on the distribution of nitrogen. The investigated distances from the inlet and corresponding interstitial water velocity dictate residence time and thus the concentrations of ammonia and nitrate nitrogen. Therefore actual and fractional distances expressed as nHRT are used for comparisons between the studies.

There was a consistent increase in ammonia nitrogen with depth between the top and middle zones in this study and the same trend was observed in studies (2) and (3) (Table 2.4). Study (2), despite being operated under higher nominal interstitial flow velocity and influent concentration conditions and monitored at longer actual and fractional distances than in the current study, demonstrated systematically lower concentrations at the top compared to the middle zone for all monitored intermediate distances. In study (2) tracer test results showed no evidence of vertical preferential water flow at the first two intermediate distances from the inlet but bottom preferential flow was demonstrated at the last intermediate distance. The last three intermediate distances from the inlet in study (3) demonstrate nitrogen distribution and water flow patterns similar to study (2). Therefore the vertical nitrogen distribution pattern in studies (2) and (3) do not correspond to vertical water flow pattern because along the wetland length nitrogen distribution pattern is the same whether there is vertical preferential water flow or not. Lack of evidence of preferential flow effect on nitrogen distribution demonstrated in studies (2) and (3)

lead to the conclusion that a process such as oxygenation near the air/water interface by direct atmospheric oxygen diffusion was probably accountable for increase in nitrogen level with depth. The bottom preferential flow at the second intermediate distance from the inlet in study (4) was attributed to a decrease in root density with depth but there was no nitrogen gradient to demonstrate the effect of this flow pattern on nitrogen removal. The root density distribution as illustrated in Figure 2.8 suggests that the same conclusion as in study (4) could be drawn for the current study.

Figure 2.8. Root density distribution in the HF VSB wetland.

In the current study the top 20 cm from air/gravel interface experienced high root density and below this depth the root density was relatively very low as demonstrated in Figure 2.8. This root density distribution could lead to lower interstitial flow velocity near the air/water interface than deeper down the water column thus promoting dispersion. Dispersion was also demonstrated by tracer tests in another study conducted in the VSB by Nakano (1999).

Table 2.3. Physical features of the current study and literature studies.

Source[§]	System Dimensions Length × width × water depth (m)	Investigated distances from inlet (m)	Investigated depths below air/water interface (cm)	Water level below gravel surface (m)	Gravel size (mm) and type	Gravel Porosity (%)	Flow velocity (m/day)
(1)	3.66 × 0.91 × 0.45	0.64, 1.89, 3.14	5, 10, 15, 20, 25, 30	0.1	5 basaltic	22	1.8
(2)	10.49 × 5.24 × 0.5	2.62, 5.2, 7.86	0, 25, 45	0.2	10 granitic	39-40	2.4
(3)	5.5× 1.6 × 0.9	0.92, 1.83, 2.75, 3.67, 4.58	17, 50, 83	0.1	10 basaltic	40	0.6
(4)	50× 2 × 0.4	17, 34	0, 30	0.1	20-30	n/a	n/a
(5)	50× 2 × 0.4	12.5, 25, 37.5	0, 30	0.1	5-10 crushed quartz	n/a	7.1

[§] (1) The current study; (2) Garcia et al., 2003; (3) Headley et al., 2005; (4) Bowmer, 1987; (5) Breen and Chick, 1995; 1999.

Table 2.4. Ammonia Nitrogen gradients retrieved from literature.

Source[§]	Fractional distance from the inlet expressed as nHRT (hrs)	Inflow Ammonia Nitrogen (mg/L)	VSB ammonia-N concentration as function of depth (mg/L)			Tracer Flow pattern
			Top	Middle	Bottom	
(1)	8.39	47.4	37.2	48.2	44.5	n/a
	24.8		15.6	35.2	25.4	n/a
	41.2		10.7	15.7	9.0	n/a
(2)	26.2	61.5	39.2	48.8	51.2	No preferential flow [1]
	52		33.6	44.4	50.6	Bottom flow [2]
	78.6		30.6	41.0	48.0	
(3)	36.8	36.8	33.2	31.9	33.8	n/a
	73.2		32.7	30.2	33.2	Mid-depth flow [3]
	110		28.4	31.8	32.4	Mid-depth flow
	146.8		26.0	27.9	31.0	No preferential flow
	183.2		17.8	22.1	26.4	No preferential flow
(4)	17[*]	105.5	107.9	n/a	107.3	No preferential flow
	34[*]		105.7	n/a	106.8	Bottom flow
(5)	42.2	57.3	53.6	n/a	54.9	No preferential flow
	84.5		39.2	n/a	49.6	Bottom flow
	126.8		41.6	n/a	54.6	Bottom flow

[§] (1) The current study; (2) Garcia et al., 2003; (3) Headley et al., 2005; (4) Bowmer, 1987; (5) Breen and Chick, 1995.

[*] Absolute distances from inlet. No sufficient information to calculate nHRT.

Study (5), however, shows a different trend from studies (2) and (3) in that there was less difference between the top and bottom zone nitrogen levels that corresponds to no evidence of preferential flow at the first intermediate distance from the inlet. At the last two monitored distances furthest from the inlet in study (5), there were higher ammonia nitrogen levels at the bottom than at the top. This trend seems to correspond to

preferential bottom flow. The difference in ammonia nitrogen distribution pattern demonstrated by study (5) compared to studies (2) and (3) can be accounted for by differences in interstitial water velocity as shown in Table 2.3. Interstitial flow velocity for study (5) was approximately three and twelve fold larger than in studies (2) and (3) respectively. This large difference in velocity could have led to higher influence of velocity on nitrogen distribution in study (5) whereby higher nitrogen levels in the bottom zone can be attributed to relatively shorter residence time compared to top zone.

Due to the complexity of the factors dictating performance between these systems, the cause of the ammonia nitrogen gradients in the current study was inconclusive even though there is strong reason to believe that the system behaves more like studies (2) and (3) than other studies. The size of systems and water velocity in studies (2) and (3) are the most similar to that of the current study. The major differences between the current study and studies (2) and (3) as noted from Table 2.3 were the gravel size and the associated porosity. The gravel used in studies (2) and (3) are twice the size of gravel used the current study. This difference in gravel size and porosity could be the cause of the different nitrogen distribution between the middle and bottom zones in the current study compared to studies (2) and (3). This suggests that in the current study, the nitrogen distribution between the top and middle zone was most likely due to oxygenation process near the air/water interface whereas between the middle and bottom zones it was due to head difference between the inlet and outlet.

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Due to the distributions of COD shown in Table 2.5 and Figure 2.9, the vertical profile was categorized into top, middle and bottom zones the same way as categorized under nitrogen discussions. The values shown in the table reflect the average of eight samples. Values with the same superscript letters under each cross section in the table are not significantly different at α level of 0.05. The values with the same superscript letters in parenthesis represent no significant difference in COD concentration between cross sections at a given depth. In the cross section closest to the inlet, COD increased with depth until 20 cm depth which represents part of middle zone where COD started to decrease with depth as shown in Table 2.5 and Figure 2.9. The other two remaining cross sections did not demonstrate significant differences in COD concentration between the depths as shown in Table 2.5. However, as demonstrated in Figure 2.9, the COD distribution pattern at these cross sections is similar to the cross section closest to the inlet. These COD trends suggest that there could be higher resistance to water flow in the bottom zone relative to the middle zone leading to longer residence time at the former and thus higher COD removal. This observation reinforces the suggestion made under the nitrogen distribution discussion in that, the difference in head between inlet and outlet could be held accountable for the discrepancy in COD distribution between middle and bottom zones. The COD levels observed in the top zone of all cross sections could be a result of rapid consumption of carbon by aerobic heterotrophic bacteria in response to oxygenation by atmospheric oxygen diffusion near the air/water interface or a reflection of resistance to flow due to higher root density. This trend corresponded to ammonia and nitrate nitrogen gradients discussed earlier. This result is also in agreement with a VSB

study by Garcia et al. (2003) who attributed the increase in COD and ammonia nitrogen concentration to observed decrease in oxidation-reduction potential as depth increased. The apparent decrease in COD concentration difference between 20 cm depth and other depths along the length of the VSB as demonstrated in Table 2.5 and Figure 2.9 can be attributed to higher COD consumption near the middle zone than top and bottom zones. Diffusion of nitrate nitrogen from the top zone into the middle zone as a result of nitrate nitrogen concentration gradient as illustrated in Figure 2.7 could be a cause of increase in COD consumption along the system length at the middle zone. The COD distribution therefore mainly serves to reinforce the trends demonstrated by ammonia and nitrate nitrogen and the associated reasons for higher contaminant removal in the bottom zone of the system relative to the middle zone.
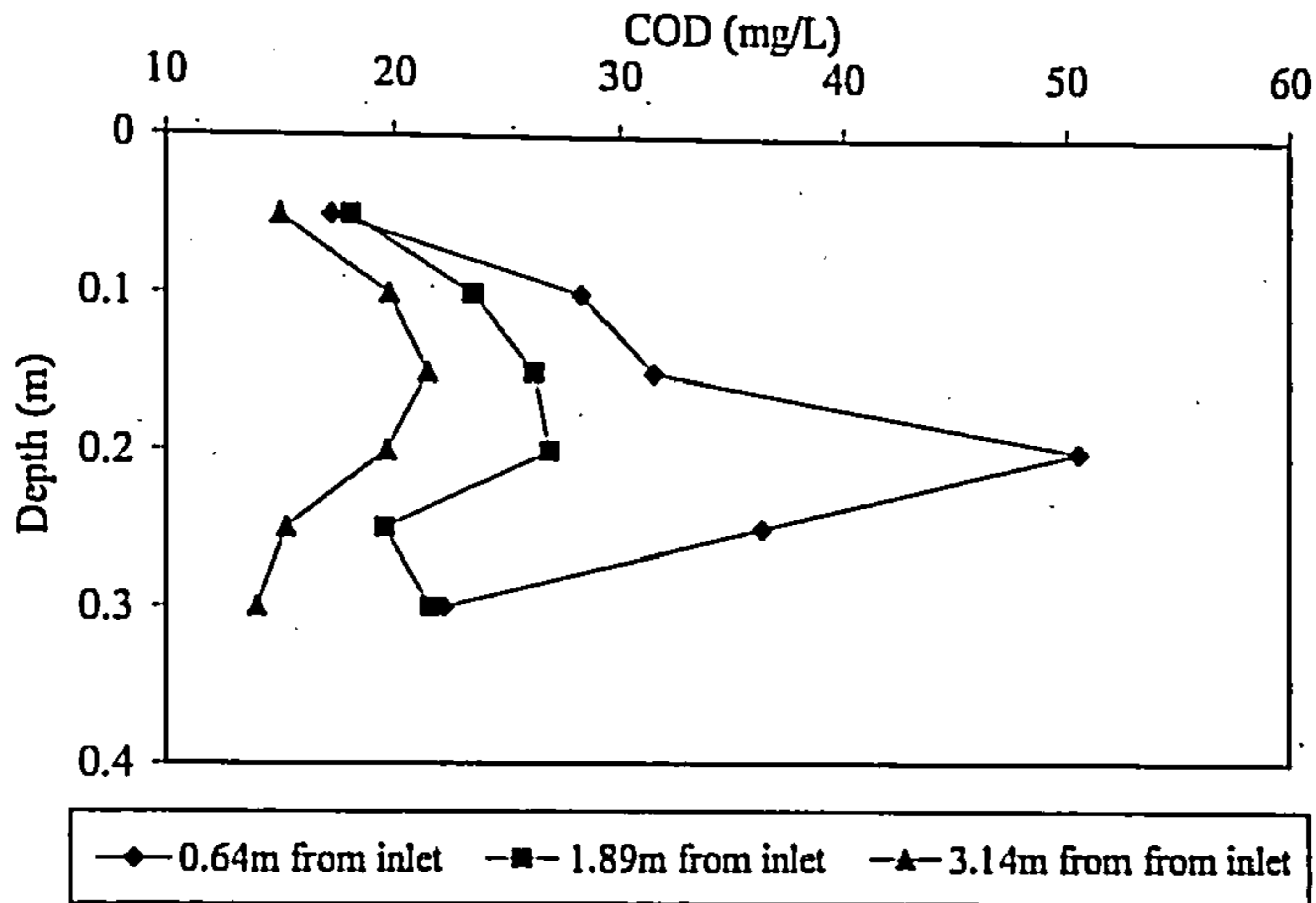
Figure 2.9. VSB pore water vertical profiles of chemical oxygen demand (COD) at the three investigated sites.

Table 2.5. Mean COD concentration at the six sampling depths along the length of the pilot vegetated submerged bed constructed wetland.

Sample depth, m	COD concentration, mg/L (Standard deviation)		
	Cross Section Distance from inlet, m		
	0.64	1.89	3.14
0.05	17.3 [a (a)](5.6)	18.1[a (a)] (3.6)	15.0 [a (a)] (2.9)
0.10	28.3 [a (a)] (5.6)	23.5[a (a)] (3.6)	19.8 [a (a)] (2.9)
0.15	31.5 [a (a)] (5.6)	26.2 [a (a)] (3.6)	21.5 [a (a)] (2.9)
0.20	50.5 [b (a)] (5.6)	26.9 [a (b)](3.6)	19.68 [a (b)](2.9)
0.25	36.3[b (a)] (5.6)	19.6 [a (a)] (3.6)	15.28 [a (a)] (2.9)
0.30	22.2[a (a)] (5.6)	21.6 [a (a)] (3.6)	14.0 [a (a)] (2.9)

Oxygen and pH variations

The values shown in Table 2.6 reflect the average of fifteen samples per depth. Values with the same superscript letters under each cross section in Table 2.6 are not significantly different at α level of 0.05. The values in the table with the same superscript letters in parenthesis represent no significant difference in DO concentration between cross sections at a given depth. The cross section closest to the inlet demonstrated some marginal differences in DO between the depths indicating that the near surface interstitial water was slightly more aerobic than deeper down the profile (Table 2.6 and Figure 2.10). The profiles for the second and third cross sections from the inlet did not demonstrate any difference in DO between the depths. This lack of significant difference in dissolved oxygen is probably caused by oxygen utilization for nitrification upon diffusion into the system at the air/water interface with very little excess oxygen penetrating to deeper depths. In support of the above postulate, DO decreased with increasing distance from the inlet at the top and middle zones as demonstrated by the first and second cross sections in Table 2.6 and Figure 2.10 indicating that there is no adequate oxygen replenishment along the system length to satisfy contaminant oxygen demand and subsequent saturation concentration of about 9 mg/L.

Wetland water chemistry and biology are affected by pH. Many treatment bacteria are not able to exist outside the range $4.0 < pH < 9.5$ (Kadlec and Knight, 1996). Nitrification is pH sensitive and rates decline significantly at pH values below 6.8. Optimal nitrification occurs in the 7.5 to 8.0 pH range. Denitrifiers operate best in the range $6.5 < pH < 7.5$ (Metcalf and Eddy, 2003). The pH distribution is also important for determining the

significance of nitrogen loss through volatilization. Ammonia nitrogen exists in water as either ammonium ions (NH_4^+) or un-ionized ammonia (NH_3), depending on pH of the water. At levels below pH 7 ammonium ions predominate and at pH > 7 ammonia gas is produced from ammonium ions and increase with increasing pH through the following Equations 2.1 and 2.2 (Metcalf and Eddy, 2003)

$$NH_3, \% = \frac{[NH_3] \times 100}{[NH_3] + [NH_4^+]} = \frac{100}{1 + [NH_4^+]/[NH_3]} = \frac{100}{1 + [H^+]/K_a} \tag{2.1}$$

Where [NH$_3$], [NH$_4^+$] and [H$^+$] are ammonia gas, ammonium and hydrogen ion concentrations respectively and K_a is acid ionization (dissociation) constant.

$$K_a = \frac{[NH_3][H^+]}{[NH_4^+]} \tag{2.2}$$

Un-ionized ammonia is relatively volatile and can be removed from solution to the atmosphere by diffusion through the water column upward to the surface and through mass transfer from the water surface to the atmosphere (Kadlec and Knight, 1996).

Table 2.6. Intra-system DO concentration at the six sampling depths along the length of

the VSB wetland

Sample depth, m	DO concentrations, mg/L (Standard deviation)		
	Cross Section Distance from inlet, m		
	0.64	1.89	3.14
0.05	$2.4^{a\,(a)}$ (0.05)	$2.1^{e\,(b)}$ (0.06)	$2.2^{f\,(b)}$ (0.05)
0.10	$2.3^{ab\,(a)}$ (0.05)	$2.2^{e\,(b)}$ (0.06)	$2.2^{f\,(b)}$ (0.05)
0.15	$2.3^{bc\,(a)}$ (0.05)	$2.1^{e\,(b)}$ (0.06)	$2.1^{f\,(b)}$ (0.05)
0.20	$2.2^{bc\,(a)}$ (0.05)	$2.1^{e\,(a)}$ (0.06)	$2.0^{f\,(b)}$ (0.05)
0.25	$2.1^{cd\,(a)}$ (0.05)	$2.1^{e\,(a)}$ (0.06)	$2.0^{f\,(a)}$ (0.05)
0.30	$2.1^{d\,(a)}$ (0.05)	$2.1^{e\,(a)}$ (0.06)	$2.1^{f\,(a)}$ (0.05)
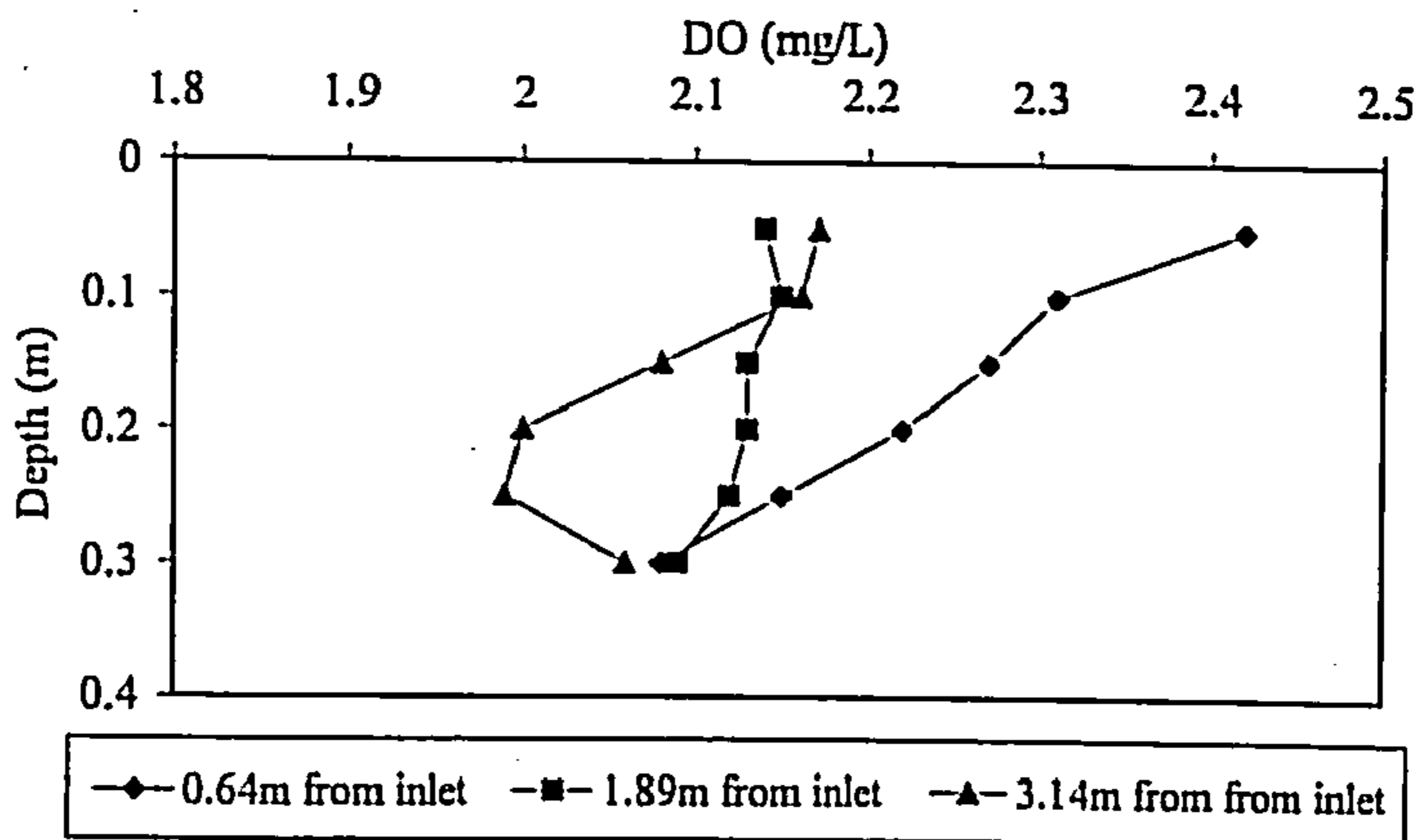
Figure 2.10. VSB pore water vertical profiles of dissolved oxygen (DO).

The pH values in this study were around 7.9±0.07. Table 2.7 contains the average values as a function of depth based on fifteen samples per depth. Values with the same superscript letters under each cross section and depth represent no significant pH difference at α level of 0.05. The pH conditions are within the optimal pH range for

nitrification and close to optimal conditions for denitrification. Under these pH conditions ammonium ions and ammonia gas account for approximately 95% and 5% of total ammonia nitrogen respectively. The system ammonia nitrogen distribution discussions are therefore justified to be on the basis of ammonium ions as the dominant ammonia species. There is no significant pH difference ($p > 0.05$) between the depths as shown in Table 2.7 for all the cross sections and therefore pH does not account for nitrogen concentration variation between the depths.

Table 2.7. Intra-system pH at the six sampling depths along the length of the VSB constructed wetland.

Sample depth, m	Average pH values (Standard deviation)		
	Cross Section Distance from inlet, m		
	0.64	1.89	3.14
0.05	7.8 [a] (0.06)	7.9 [a] (0.07)	8.0 [a] (0.08)
0.10	8.0 [a] (0.06)	7.9 [a] (0.07)	8.0 [a] (0.08)
0.15	8.1 [a] (0.06)	7.9 [a] (0.07)	7.9 [a] (0.08)
0.20	8.0 [a] (0.06)	7.8 [a] (0.07)	7.9 [a] (0.08)
0.25	7.9 [a] (0.06)	7.9 [a] (0.07)	8.1 [a] (0.08)
0.30	7.8 [a] (0.06)	7.9 [a] (0.07)	7.8 [a] (0.08)

Temperature variations

Some biochemical processes, notably the microbially mediated nitrogen processes are temperature sensitive (Kadlec and Knight, 1996). Therefore assessment of spatial variation in nitrogen treatment performance cannot be complete without investigation of spatial temperature distribution. The values with the same superscript letters under each cross section and depth in Table 2.8 represent no significant water temperature difference

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at α level of 0.05. The water temperature ranged from 18.3 to 20.8 °C with an average of 19.7± 0.1 °C. There were no significant differences (p > 0.05) in average water temperature either between different depths or between cross sections as shown in Table 2.8.

Table 2.8. Intra-system temperature at the six sampling depths along the length of the pilot vegetated submerged bed constructed wetland.

Sample depth, m	Water Temperature, °C (Standard deviation)		
	Cross Section Distance from inlet, m		
	0.64	1.89	3.14
0.05	20.0 [a] (0.1)	19.4 [a] (0.1)	19.4 [a] (0.1)
0.10	20.0 [a] (0.1)	19.7 [a] (0.1)	19.5 [a] (0.1)
0.15	19.8 [a] (0.1)	19.7 [a] (0.1)	.19.6 [a] (0.1)
0.20	19.8 [a] (0.1)	19.7 [a] (0.1)	19.7 [a] (0.1)
0.25	19.9 [a] (0.1)	19.7 [a] (0.1)	19.5 [a] (0.1)
0.30	19.7 [a] (0.1)	19.7 [a] (0.1)	19.5 [a] (0.1)

These findings are in agreement with studies by Brix (1990) and Kadlec (2001), who reported little temperature variation with depth in submerged beds. Temperature therefore, did not account for nitrogen variation between depths and cross sections.

CONCLUSIONS AND RECOMMENDATIONS

The VSB wetland ammonia- and nitrate-nitrogen, COD intra-system distribution results of the current study indicated that there were higher nitrification and COD removal near the surface than deeper down in the water column. The DO distribution results show that DO is consumed upon diffusion into the water column near air/water interface with no or

little oxygen diffusing to deeper depths. This DO distribution suggest that the level of nitrification as reflected by the detected levels of both ammonium and nitrate nitrogen near the surface could have been achieved through nitrification near the surface and diffusion of ammonium nitrogen into this zone from deeper down the water column in response to concentration gradient. The distributions of nitrogen, COD and DO are a result of predominantly direct atmospheric oxygen diffusion and longer retention time due to lower porosity caused by higher root density. Therefore the design of these systems should be based on the area required for atmospheric oxygen transfer. Nitrogen, COD and DO vertical distribution gradients in the current study are incompatible with the assumed vertical uniformity in the plug flow model. The results of this study indicate that there is higher relative nitrification and COD removal at the bottom than the middle zones most probably due to longer retention time caused by hydraulic head difference between the inlet and the outlet.

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CHAPTER THREE

MODELING OXYGEN TRANSFER INTO VEGETATED SUBMERGED BED CONSTRUCTED WETLAND.

ABSTRACT

Modeling the transfer of oxygen into constructed wetlands is essential for understanding BOD$_5$ and nitrogen removal mechanisms and for improvement of system design. A mechanistic model for atmospheric oxygen transport into vegetated submerged bed constructed wetlands was developed based on the two-layer model theory of a gas-liquid interface and associated molecular mass transport equations. Data from batch submerged bed constructed wetland microcosms were used to parameterize the model. The model satisfactorily predicted the vertical dissolved oxygen (DO) profiles. The model DO profile predictions were insensitive to changes in the unsaturated zone diffusion coefficient. This was due to the fact that resistance to oxygen flow in the unsaturated zone is negligible relative to the resistance in the saturated zone. The model DO profile predictions were however sensitive to the diffusion coefficient inside the saturated zone as well as to the oxygen consumption rate constant. However the model atmospheric oxygen flux was sensitive to changes in both the unsaturated and saturated zones. The predicted oxygen fluxes range from 0.027 to 0.32 g m^{-2} day^{-1} and are on the lower end of the range 0.077 to 7.92 g m^{-2} day^{-1} reported in literature.

INTRODUCTION

The use of Vegetated Submerged Bed (VSB) constructed wetlands for treatment of municipal waste streams continues to expand as more emphasis is placed on economically viable technologies for nutrient removal. The cost effectiveness of VSB constructed wetlands is driven by the level of treatment necessary to meet specific discharge water quality objectives. Although properly designed VSB constructed wetlands adequately remove organic matter from typical waste streams, nitrogen removal in these systems is generally limited (US EPA, 2000a; US EPA, 2000b; Vymazal, 2005). Oxygen (O_2) availability is one of the most limiting factors in determining contaminant removal by bacteria mediated processes in VSBs (Marsili-Libelli and Checchi, 2005), with typical values of dissolved oxygen (DO) usually below 1.0 mg/L (US EPA, 2000a). Ammonium nitrogen and organic carbon (OC) are the two principal components of oxygen demand. Nitrogen exerts nearly 90% of the oxygen demand using about 4.3 mg/L O_2 for each mg/L of ammonium nitrogen while organic carbon utilizes approximately 0.44 mg/L O_2 per mg/L of OC (Metcalf and Eddy, 2003). Therefore, in an oxygen limited microbial mediated process, nitrification is considered to be the rate limiting step for nitrogen removal since respiratory denitrification, the second and final step of the nitrogen removal process, is best suited for an anaerobic environment coupled with an adequate carbon supply prevalent in most wastewaters (US EPA, 1993; Tanner et al. 2002; Vymazal, 2005). Thus oxygen availability is the key to overall VSB wetland performance and modeling the supply of O_2 delivery is crucial for wetland design.

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To develop an oxygen transfer model it is essential to explore atmospheric oxygen transfer pathways into VSB wetlands. Two main pathways of oxygen transfer from the atmosphere into the wetland are documented: 1) direct atmospheric oxygen diffusion and 2) plant root oxygen release (Kadlec and Knight, 1996). There are limited research findings concerning the quantitative magnitude of direct atmospheric oxygen diffusion into VSB wetlands, essentially consisting of only three studies (Brix and Schierup, 1990; Nakano, 1999; Wu et al., 2001). The reported direct atmospheric oxygen diffusion fluxes range from 0.077 to 7.92 O_2 g/m^2/day.

There are several reports on plant root oxygen release because this pathway was originally believed to exclusively account for oxygen supplies in wetlands. Early reports suggested that plants were responsible for enhanced constructed wetland performance (US EPA, 1988, US EPA, 1993). The reported plant root release fluxes range from 0 to 28.6 O_2 g/m^2/day (Kadlec and Knight, 1996; Wu et al. 2001). This wide range of fluxes can be mainly attributed to differences in measurement methods (Bedford et al., 1991; Sorrell and Armstrong, 1994), plant type, root type, root density, the root internal oxygen concentration, oxygen demand of the surrounding medium and the permeability of the root walls (Sorrell and Armstrong, 1994). However further research indicated poor oxidation of reduced nitrogen compounds (Wood, 1995; Kadlec and Knight, 1996; Brix, 1997; Whitney et al., 2003) thus suggesting that early studies overestimated the oxygen transfer by wetland plants (Stein and Hook, 2005). Stein and Hook (2005), summarizing their series of greenhouse and laboratory studies (Allen et al., 2002; Hook et al., 2003; Stein et al., 2003 and Riley et al., 2005), concluded that plant root respiration rates varied

in response to the seasonal dormancy of wetland plant species and that lower temperature permitted increased oxygen transfer via the root zone. They further concluded that the potential for plants to enhance aerobic treatment processes is more limited during periods of active plant growth and associated higher temperatures when plant root respiration is highest. The contribution of plant root oxygen release during this critical time is minimal leaving direct atmospheric oxygen diffusion a probable dominant oxygen supply pathway.

Despite the apparent importance of direct atmospheric oxygen supply existing models on contaminant removal in VSB wetlands either do not adequately account for oxygen flux through this pathway (Schwager and Boll, 1997; Langergraber and Simunek, 2005) or assume that oxygen supply through this pathway is negligible with the dominant pathway being plant roots release (Kayombo et al., 2000; Wynn and Lier, 2001; Marsilli-Libelli and Checchi, 2005).The goal of this work is to develop a model that include direct atmospheric oxygen transfer into subsurface constructed wetlands thereby improving our ability to accurately design wetlands based on the required level of wetland performance.

MATERIALS AND METHODS

Model development

Overview

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Contrary to earlier assumptions, the contribution of aeration via root system in VSB wetlands is limited and direct diffusion of atmospheric oxygen is the dominant oxygen transfer pathway (Brix and Schierup, 1990; Wu et al., 2001; Stein and Hook, 2005). Various formulations have been used to describe the air-water gas exchange process, but the two most common are the "stagnant two-film" model (Lewis and Whitman, 1924) and the "surface renewal" model (Dankwerts, 1951).

In the first approach, an instantaneous equilibrium is implied at the air-water interface, and concentration gradients are assumed in the two stagnant thin boundary layers residing at each side of the interface. In the second approach, bulk "parcels" of water and air are constantly delivered to the interface as a consequence of local turbulence in both fluids. Upon arrival at the interface, an instantaneous interfacial equilibrium is assumed between liquid and gaseous parcels and diffusion proceeds inside these small fluid elements during their residence time at the vicinity of the surface. In this case, the model parameters include the molecular diffusion coefficients in both phases and the contact time.

Since the transport of O_2 through the layer system is a one dimensional molecular diffusion process in both cases, the one dimensional form of Fick's First Law (with z as the vertical direction), as illustrated in Figure 3.1 and shown in Equation 3.1, is used for the computation of flux (Liss and Slater, 1974).
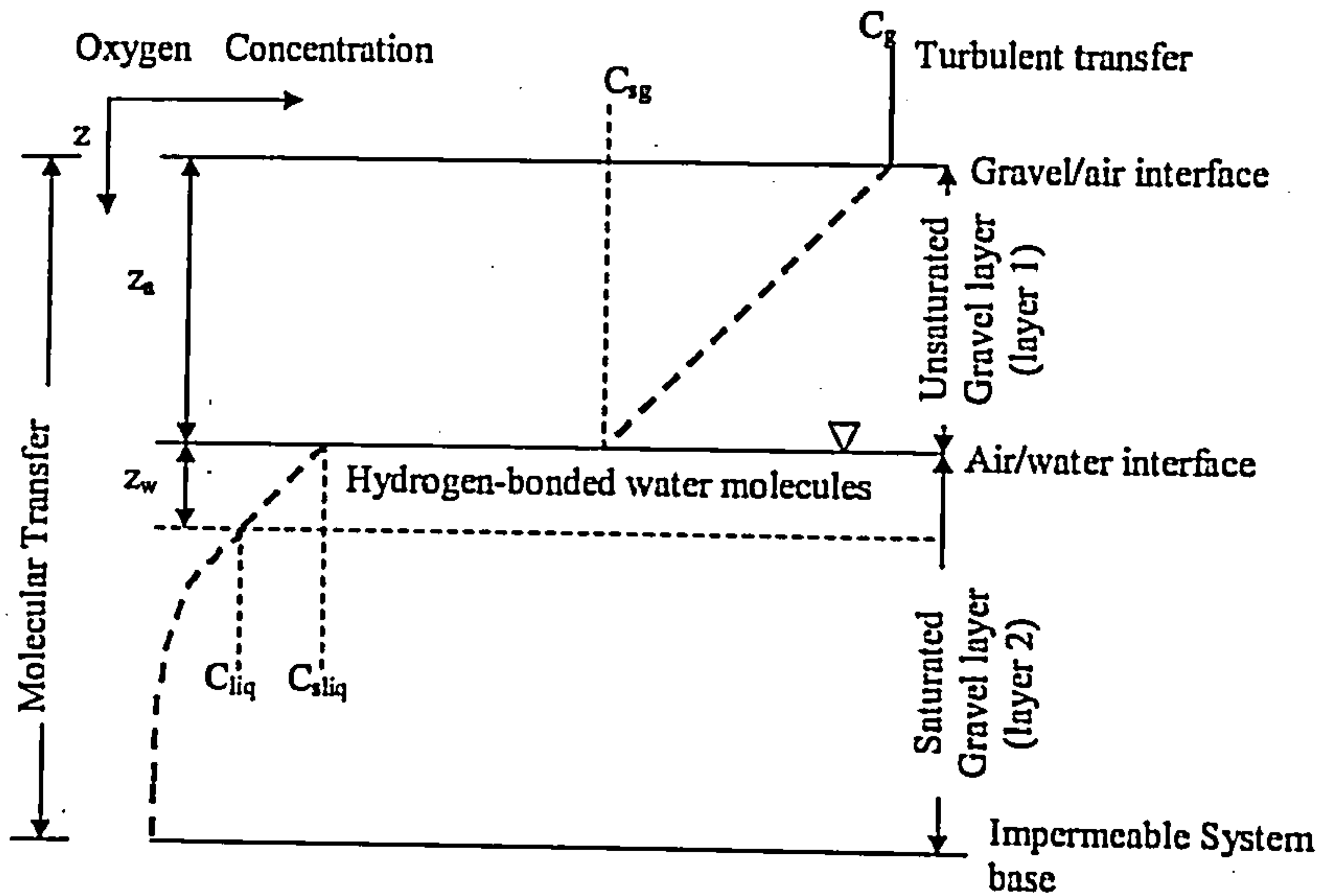
Figure 3.1. Modified Stagnant Two-Layer model of a gas-liquid interface (Lewis and

Whitman, 1924) for Vegetated Submerged Bed.

The terms C_g and C_{sg} represent oxygen concentrations [g m^{-3}] in the bulk air and in the unsaturated gravel layer adjacent to the air/water interface, respectively, while the terms C_{liq} and C_{sliq} represent oxygen concentrations adjacent to the hydrogen-bonded water film/bulk water interface, and in the saturated layer adjacent to the air/water interface in that order. The term z represent the distance [m] from the gravel/air interface into the reactor and z_a and z_w represent the thickness of the usaturated zone [m] and the hydrogen bonded molecules layer [μm].

$$F = -D \frac{\partial C}{\partial z} \qquad (3.1)$$

Where F is the oxygen flux through the layer [g m^{-2} day^{-1}], D is the coefficient of molecular diffusion of oxygen in layer material [m^2 day^{-1}], C is the oxygen concentration [g m^{-3}] and z is the vertical distance [m].

The more commonly used oxygen flux expression at an air-water interface of the two layer boundary system is given by:

$$F = k\Delta C \tag{3.2}$$

where F is as defined before, ΔC is the concentration difference across a particular layer, and k is the corresponding exchange constant commonly known as mass transfer coefficient [m day^{-1}]. Dobbins (1956) presented some calculations where he defined transfer across an unbroken film based on the two-film theory. This author derived an expression for k by solving Fick's Second Law and demonstrated that k could be reasonably approximated with only a very small error by:

$$k = \frac{D}{\Delta z} \tag{3.3}$$

Where Δz is the thickness of the layer across which oxygen flux is determined. When applying Equation 3.2 to the two-layer situation shown in Figure 3.1, and assuming that the transport of gas across the interface is a steady state process, it follows that:

$$F = k_g \left(C_g - C_{sg} \right) = k_{liq} \left(C_{sliq} - C_{liq} \right) \tag{3.4}$$

where k_g and k_{liq} are the mass transfer coefficients for the gas and liquid phases, respectively. Since oxygen transfer obeys Henry's Law, then:

$$C_{sg} = HC_{sliq} \tag{3.5}$$

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where H is the Henry's Law constant defined as the ratio of the equilibrium concentration in gas phase [g m^{-3} air] and the equilibrium concentration of dissolved oxygen in liquid phase [g m^{-3} water] (Metcalf and Eddy, 2003).

Even though the abrupt discontinuity near the interface, shown in Figure 3.1, is physically unrealistic, the film model is useful in visualizing the processes at the interface and for simplifying the theoretical calculations of gas exchange rates (Liss and Slater, 1974). Furthermore, predictions based on film approach often show little or no difference from those derived from more complex models (Danckwerts, 1970).

The "stagnant two film" model was initially developed for turbulent flow conditions in both the bulk gas and the liquid phases (Lewis and Whitman, 1924). A study by Pedersen (1998) suggested a modification of the two-film theory to account for oxygen transport across the air/water interface under quiescent conditions in both the air and the water layers. In Pedersen's (1998) study, pure oxygen or air bubbles maintained at a gas nozzle were submerged in a water-bath at 50 mm depth. Vertically upward water flow towards the bubble was induced by a water nozzle at various flow velocities while measuring the relative water flow velocity using a Laser Doppler Anemometer (LDA). The LDA velocity data was used to determine the mass transfer coefficient k and film thickness. A threshold velocity of 0.05mm s^{-1} was set to determine the boundary between flow zone and stagnant zone and thus stagnant film thickness. The main advantage of LDA is that it does not disturb the flow field. Results of Pedersen's study indicated that there is a hydrogen-bonded structure of water molecules at and near the air/water interface. Under

stagnant conditions, this structure has a thickness of 16μm and a mass transfer coefficient of 132μm/s. When the bubble is injected with the upward water flow, the outer part of the structure creates a laminar motion in the direction parallel to the interface. Increasing flow leads to an increase in the laminar layer thickness and a corresponding decrease in the thickness of the stagnant layer maintaining a constant thickness of the sum of the two layers up to a velocity of approximately 24 mm s^{-1} beyond which the overall layer thickness starts to decrease. During the flow velocity interval 0 to ~24 mm s^{-1} the measured k also remains constant. The comparison of measured k values versus calculated values from Equation 3.3, in this study, strongly indicated that the equation is valid thus reinforcing findings by Dobbins (1956). The interstitial water flow velocity in constructed wetlands is within the above velocity range. Therefore the modified two-film model is used in the current study for flow conditions in VSB constructed wetlands.

System configuration and process governing equations

The treatment wetland model consists of oxygen transfer and utilization processes in two distinct zones; namely the unsaturated gravel zone and the saturated gravel zone, as illustrated in Figure 3.2. The system is assumed to be impermeable at the bottom of the wetland (depth $z = L$) where L is the total wetland depth. Furthermore, the air/water interface is at $z = z_1$, the depth $z = 0$ to $z = L$ is filled with gravel, and the gravel/air interface is at $z = 0$.
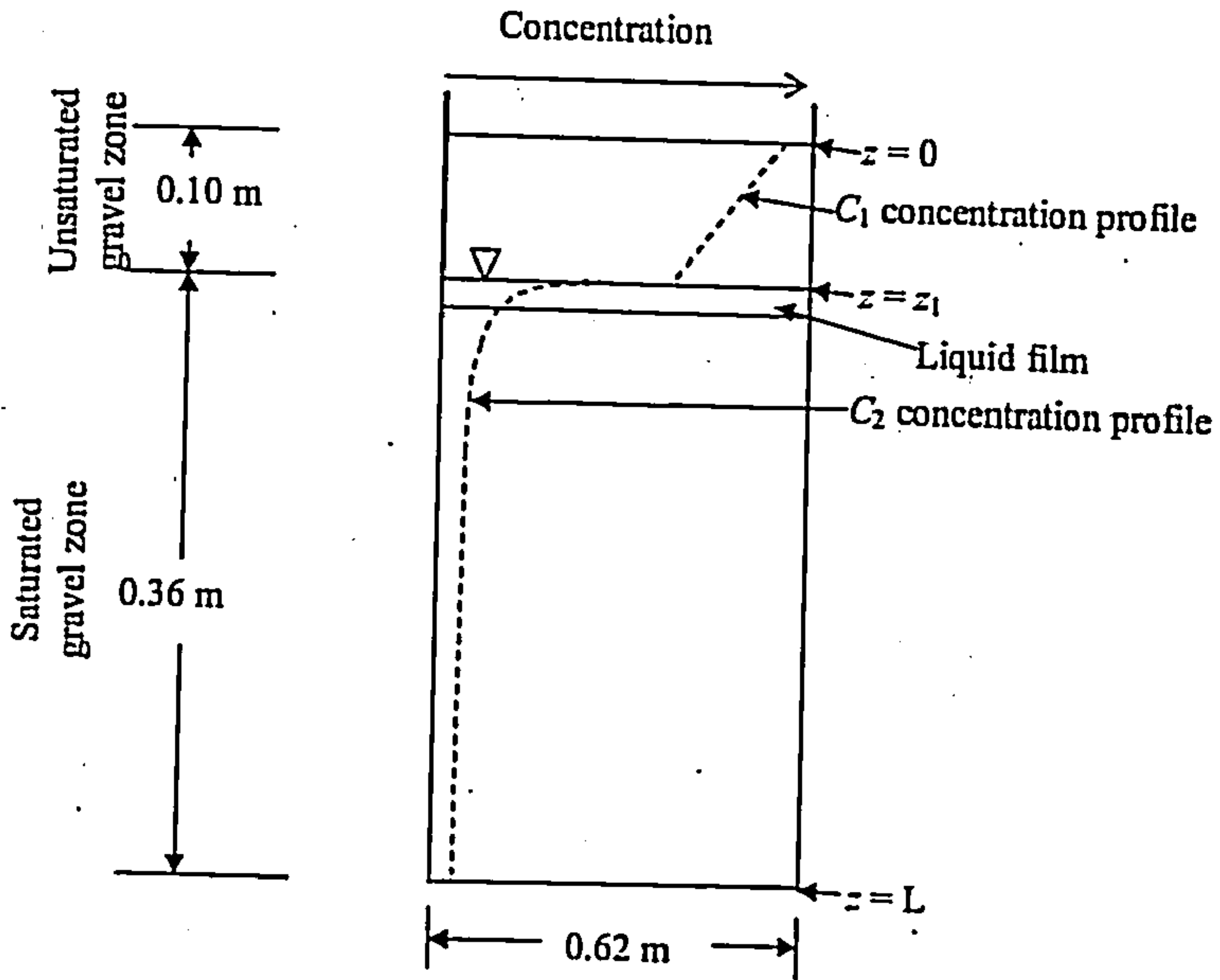
Figure 3.2. Schematic batch reactor configurations.

According to the law of conservation of mass, the change in the amount of oxygen in the volumetric unit of gravel pore space per unit of time for one dimensional flow is equal to change in flux minus the consumption of oxygen in the gravel under steady state conditions. This expression for the unsaturated zone is shown in Equation 3.6.

$$\frac{\partial C_1}{\partial t} = \frac{\partial F}{\partial z} - k_1 C_1 = D_1 \frac{\partial^2 C_1}{\partial z^2} - k_1 C_1 = 0, \quad 0 < z < z_1 \tag{3.6}$$

where C_1 is oxygen concentration [g m^{-3}], D_1 is diffusion coefficient [m^2.day^{-1}] and k_1 is volumetric first order biochemical oxygen utilization rate constant [day^{-1}] in the

unsaturated zone denoted by subscript 1. It is assumed that oxygen consumption in the unsaturated zone is negligible relative to the amount diffusing into the system from the atmosphere so k_1 is assumed to be zero in Equation 3.6. In addition, the steady state oxygen concentration at the air/gravel interface shown in Equation 3.7 is specified as the upper boundary (Kowalik, 1985).

$$C(t, 0) = C_0 \qquad (3.7)$$

where C_0 is atmospheric oxygen concentration [g m^{-3}] and t is time [day].

A similar expression for the saturated zone is shown in Equation 3.8.

$$\frac{\partial C_2}{\partial t} = \frac{\partial F}{\partial z} - k_2 C_2 = D_2 \frac{\partial^2 C_2}{\partial z^2} - k_2 C_2 = 0, \quad z_1 < z < L \qquad (3.8)$$

Where C_2 is oxygen concentration [g m^{-3}], D_2 is diffusion coefficient [m^2 day^{-1}] and k_2 is volumetric first order biochemical oxygen utilization rate constant [day^{-1}], in the saturated zone denoted by subscript 2. The term L is the total height of the reactor.

Equation 3.9 is the required boundary condition used in Equation 3.8 to ensure there is no flux out through the bottom of the reactor.

$$\frac{\partial C_2}{\partial z} = 0, \quad z = L \qquad (3.9)$$

The influence of temperature on the reaction rate is modeled via an Arrhenius relationship shown in Equation 3.10 (Mashauri and Kayombo, 2002; Rousseau et al., 2004).

$$k_2 = k_{2(20)}\theta^{(T-20)}$$

(3.10)

where k_2 and $k_{2(20)}$ are the rate of oxygen utilization [day^{-1}] in the saturated zone at the operating water temperature (T) and at a temperature of 20 ^{0}C and θ is the dimensionless temperature coefficient for oxidation which is estimated at 1.06 from most studies (Reed and Brown, 1995; Tanner et al., 1995).

The concentration profile in the unsaturated zone is obtained by solving Equation 3.6 for C_1 to obtain a linear relationship of the form expressed in Equation 3.11 (Dillon, personal communication, 2005; Cussler, 1997).

$$C_1(z) = az + b \qquad 0 < z < z_1$$

(3.11)

Where a is an oxygen concentration gradient in the unsaturated zone [g/m^3/m] and b is a constant of interception [g/m^3].

The constant b can be found by solving Equation 3.11 for $z = 0$ and applying boundary condition in Equation 3.7 to obtain Equation 3.12.

$$C_1(z) = az + C_0 \qquad 0 < z < z_1$$

(3.12)

The general solution for the saturated zone is derived from Equations 3.8 and 3.9. In Equation 3.8, we seek a function that describes oxygen mass transfer in water with the property that the second derivative of the function is the same as the function itself. This condition is necessary because close examination of Equation 3.8 shows that the second derivative of the oxygen concentration profile is equated to the function of the oxygen concentration profile. The exponential functions $y(t) = e^t$ and $y(t) = e^{-t}$ and their constant multiples satisfy this condition. Any sum of the exponential function solutions are also a

solution (Boyce and DiPrima, 2001). Therefore, an exponential function was used to find a solution for the saturated zone. The oxygen concentration function can therefore be expressed in the form:

$$C_2(z) = e^{rz} \tag{3.13}$$

Where r is the oxygen mass transfer coefficient [m^{-1}].

Substituting e^{rz} from Equation 3.13 for C_2 in Equation 3.8 results in Equation 3.14 which gives the solution for the mass transfer coefficient r:

$$\frac{\partial C_2}{\partial t} = D_2 r^2 e^{rz} - k_2 e^{rz} = D_2 r^2 - k_2 = 0$$

therefore

$$r^2 = \frac{k_2}{D_2} \rightarrow r = \pm\sqrt{\frac{k_2}{D_2}} \rightarrow r_-, r_+ \tag{3.14}$$

From Equations 3.13 and 3.14, the steady state solution for oxygen concentration profile is:

$$C_2(z) = Ae^{rz} + Be^{-rz} \tag{3.15}$$

where A and B are constants. The derivative of equation 3.14 with respect to z is:

$$\frac{\partial C_2(z)}{\partial z} = r\left[Ae^{rz} - Be^{-rz}\right] \tag{3.16}$$

Applying a boundary condition in Equation 3.9 on Equation 3.16 yields the solution of Equation 3.15 with substitution of constant A for B as follows:

$$r\left[Ae^{rL} - Be^{-rL}\right] = 0$$

therefore

$$Ae^{rL} = Be^{-rL} \rightarrow B = Ae^{2rL} \tag{3.17}$$

resulting in

$$C_2(z) = A\left[e^{rz} + e^{2rL}e^{-rz}\right]$$

In this system configuration, the two boundary conditions at the air/water interface derived from Equations 3.4 and 3.5 are as follows:

$$C_1(z_1) = H C_2(z_1) \tag{3.18}$$

$$D_1 \frac{\partial C_1(z_1)}{\partial z} = D_2 \frac{\partial C_2(z_1)}{\partial z} \tag{3.19}$$

where H is Henry's Law constant as described earlier.

Literature values for the diffusion coefficients D_1 and D_2, as shown in Table 3.1, were used in this model. General solutions for oxygen concentration profiles in the unsaturated and saturated zones depicted in Equations 3.11 and 3.16 and coupled with boundary conditions in Equations 3.17 and 3.18, were then solved.

A quasi-Newton numerical algorithm in the Microsoft Excel solver package was used to optimize the predicted first order oxygen utilization rate constant against observed data through the residual error analysis. Details of observed data collection are discussed in a later section. Normalized root mean square error (NRMSE) as shown in Equation 3.20 was used as the optimization criterion (Loague and Green, 1991).

$$NRMSE = \sqrt{\left[\sum_{i=1}^{n}(P_i - O_i)^2 \Big/ n\right]} \times \frac{100}{\overline{O}} \qquad (3.20)$$

where P_i is predicted values, O_i is observed values, n is number of observed values and $\overline{O}$ is the mean of the observed values.

Direct atmospheric oxygen flux into wetlands was then predicted using Equations 3.2 and 3.3 for k_2 values representative of field scale VSB constructed wetland conditions extracted from the literature.

Sensitivity analysis

A sensitivity analysis was conducted by varying the diffusion coefficients in the saturated and unsaturated zones within a range representative of the different conditions as extracted from literature. When one diffusion coefficient was varied the other was held constant at a value equivalent to free water or air condition extracted from literature.

Oxygen profile determination experiment

Five wetland batch reactors of dimensions 0.62 m width × 0.62 m length × 0.46 m depth were constructed in a greenhouse on the Pullman campus of Washington State University. Four of the five wetland reactors were sub-surface flow (SSF) batch reactors vegetated with locally harvested cattails (*Typha latifolia*) and the fifth was the same as the other four but without vegetation. Two of the four vegetated SSF reactors contained

gravel with a grain size of 32 mm, while the other two contained gravel size of 5 mm. The remaining unvegetated batch reactor contained 32 mm gravel and was used as control.

Two of the four vegetated reactors, one for each gravel size, were fed with a deoxygenated solution of 10 mg/L NH_4-N concentration, representing secondary municipal wastewater. The NH_4-N was prepared from NH_4CL. These reactors were seeded with bacteria laden effluent extracted from fish ponds. The remaining two vegetated reactors were fed with deoxygenated tap water only. The control reactor was also fed using only deoxygenated tap water. All wetlands were made anaerobic by the addition of sodium sulfite and cobalt chloride to the feed wastewater according to the ASCE Standard (1992). The DO of the tap water was measured prior to commencement of experiments and DO was removed to below 0.4 mg/L. The configuration of the tanks was as shown in Figure 3.3.
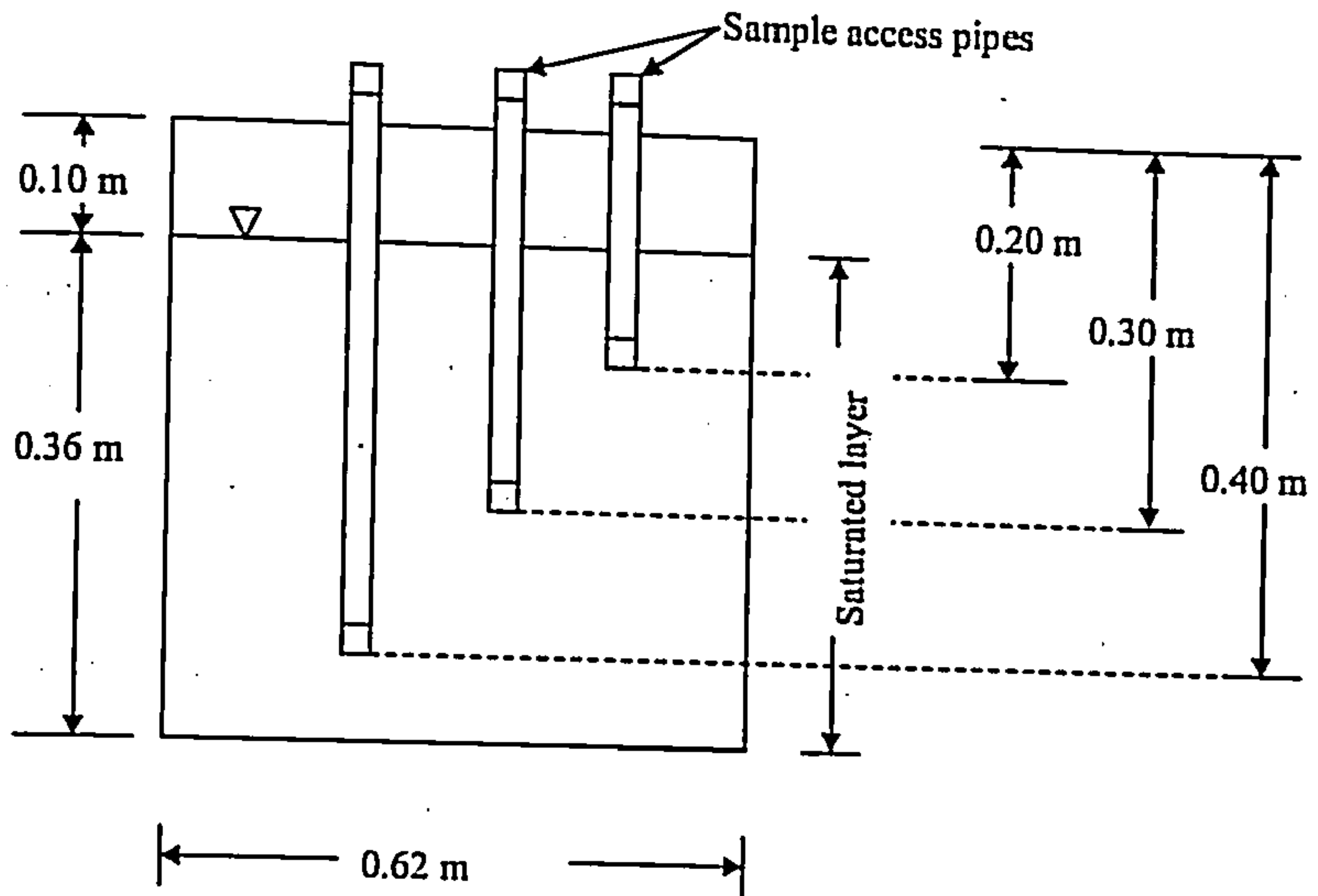
Figure 3.3. Schematic representation of the experiment setup

Sample access pipes were installed at depths 20, 30 and 40 cm below the gravel/air interface. An Extech® DO meter was used to measure the dissolved oxygen of the water at the three different depths. At each sampling point, water was extracted to draw in a representative sample at the depth of sampling point. The drawn water was temporarily stored in an air tight container and later put back into the access pipe after DO measurements were made at each sampling point. The pipes were purged with nitrogen gas after each measurement and pipe caps were used to close the top end of the pipes. Data was collected using daily time steps for seven days.

RESULTS AND DISCUSSIONS

Model parameterization

Diffusion coefficients (D) suitable for the systems under investigation were not experimentally determined. Therefore, studies on gas diffusion in porous media such as soil were assessed and compared to the conditions of the current study system (Smith, 1980; Sallam et. al, 1984; Kowalik, 1985; Erickson and Tyler, 2000). Literature data shown in Table 3.1 cover a range of conditions that encompass the current study system conditions.

The diffusion coefficient is a function of several porous media parameters including water condition. One important factor that influences D is the amount of large pores in the media; increasing the amount of large pores leads to an increase in D. This is the case because small pores increase tortuosity thus increasing resistance to mass transport. The amount of large pores in a porous media is a reflection of the size and shape of the porous media particles (Erickson and Tyler, 2000). Based on this reasoning, gravel sized particles (>2 mm) would have a greater D than sand sized particles (0.05-2 mm). Free water and air as well as waterlogged soils were evaluated to set the upper and lower oxygen diffusion limits for the system under investigation. As depicted in Table 3.1, the oxygen diffusion coefficient in air is four orders of magnitude higher than in water regardless of the media under which diffusion takes place. There are slight differences in the Table 3.1 values reported by different authors however, within each media type, all diffusion coefficients are within the same order of magnitude. Diffusion coefficients in

dry and waterlogged soils are generally one order of magnitude less than in free air and water respectively.

Table 3.1. Literature values of oxygen molecular diffusion in air and water under different situations.

Media	O$_2$ Diffusion Coefficient (m^2/day)	Source
Free water	*1.73×10^{-4}	Campbell and Norman (1998)
Free air	1.73	
Waterlogged soil	*2.16×10^{-4}	Kowalik (1985)
Free water	*1.76×10^{-4}	Kadlec and Knight (1996)
Free water	*1.81×10^{-4}	Crites and Tchobanoglous (1998)
Free water	*1.61×10^{-4}	Metcalf and Eddy (2003)
Free air	1.65	Smith (1980)
Very dry soil	6.6×10^{-1}	
Waterlogged soil	6.6×10^{-5}	
Free water	*2.05×10^{-4}	Millington (1955)
Waterlogged soil	*3.2×10^{-5}	

*Measured at standard temperature and pressure (101.3 kPa, 293.16 K)

Model Predictions

Variance between model and observed data ranged from as high as 15% to as low as 5% RMSE as shown in Table 3.2. There was a better model fit for the 5 mm gravel reactors than the 32 mm gravel reactors. Ammonia dosed reactors experienced better model fit than tap water dosed reactors. The oxygen utilization rate constants for the 5 mm gravel reactors were slightly lower than those for the 32 mm gravel reactors. The oxygen utilization rate constant for the experiment control is one order of magnitude less than in all other reactors as shown in Table 3.2. This difference was expected since the

experiment control reactor gravel was washed to minimize oxygen demand prior to commencement of the experiment.

Table 3.2. Optimized decay constants and error margins for five microcosm batch reactors.

Water type	Reactor Type				
Gravel size (mm)	Tap[a]	Tap[a]	Ammonia[a]	Ammonia[a]	Control*
	32	5	32	5	32
Reaction rate constant (day⁻¹)	0.0085	0.0076	0.0084	0.0079	0.00082
NRMSE (%)	15	11	9	5	5

[a] Vegetated batch reactors

* Tap water without vegetation

In general, the model overestimates O_2 concentrations for shallow depths and underestimates O_2 concentrations for deeper depths in all reactors as shown in Figures 3.4 to 3.7. The best fit was attained at the 0.3 m depth. The best model fit in terms of error margin and graphical fit was for the 5 mm ammonia dosed reactor as illustrated in Figure 3.6 and Table 3.2.
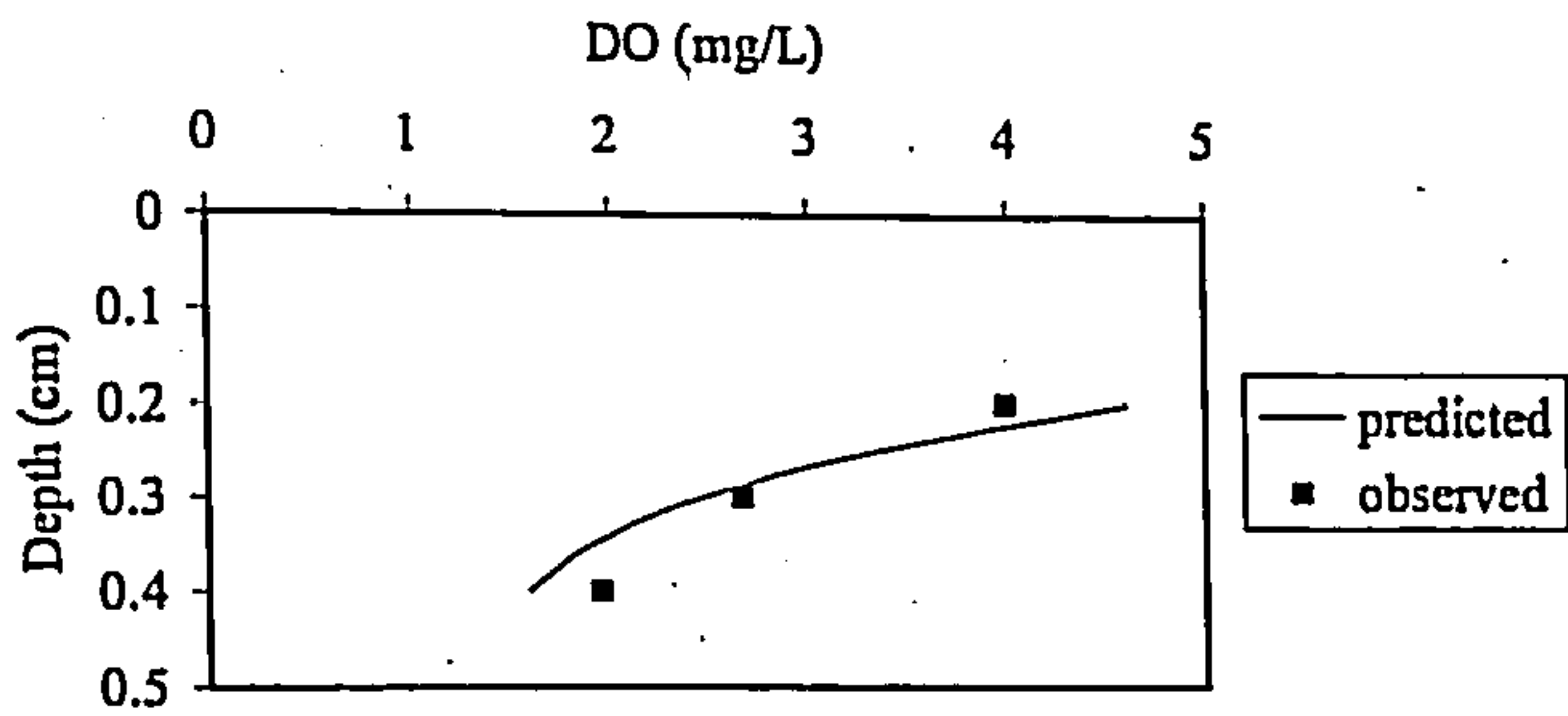
Figure 3.4. Dissolved oxygen profile in a 32 mm diameter gravel vegetated subsurface batch microcosm dosed with tap water.
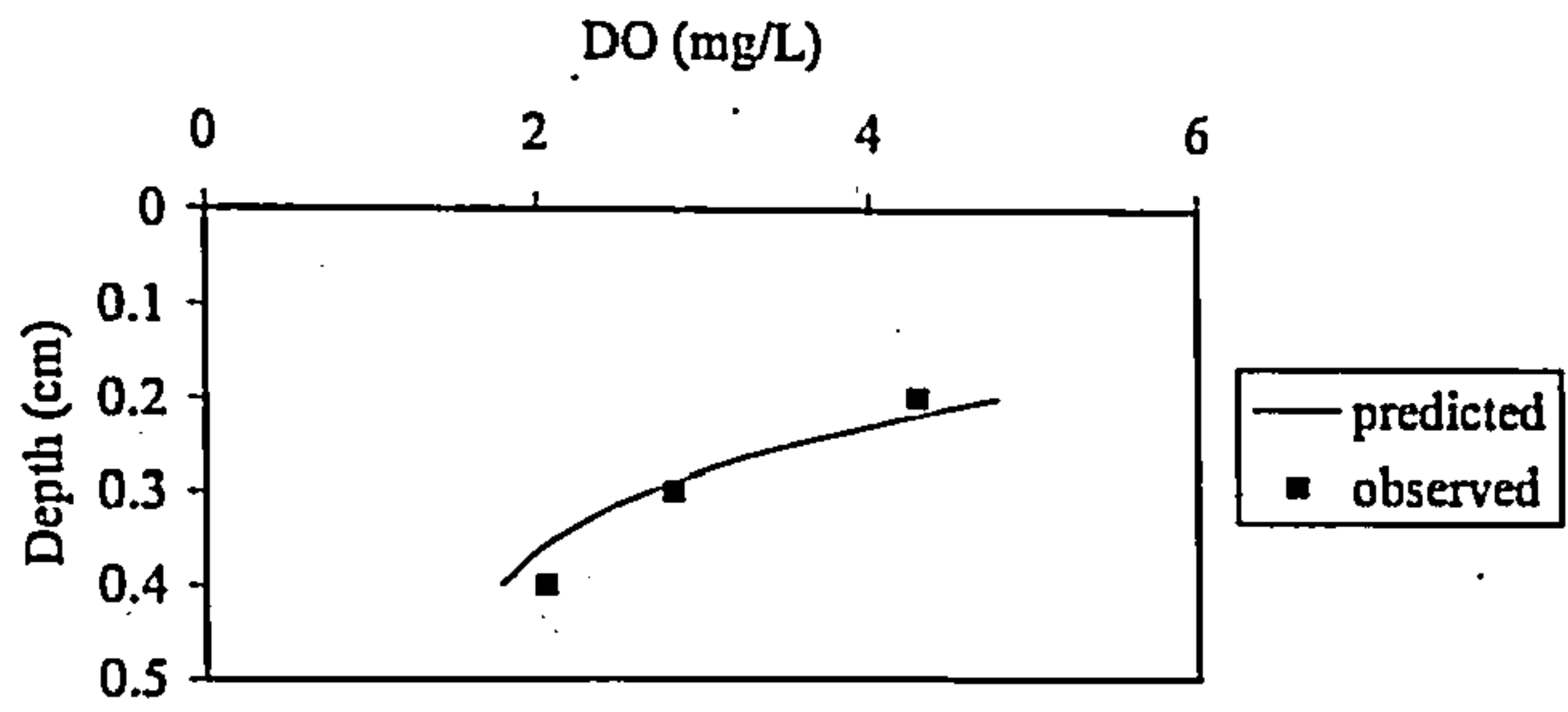
Figure 3.5. Dissolved oxygen profile in a 5 mm diameter gravel vegetated subsurface batch microcosm dosed with tap water.
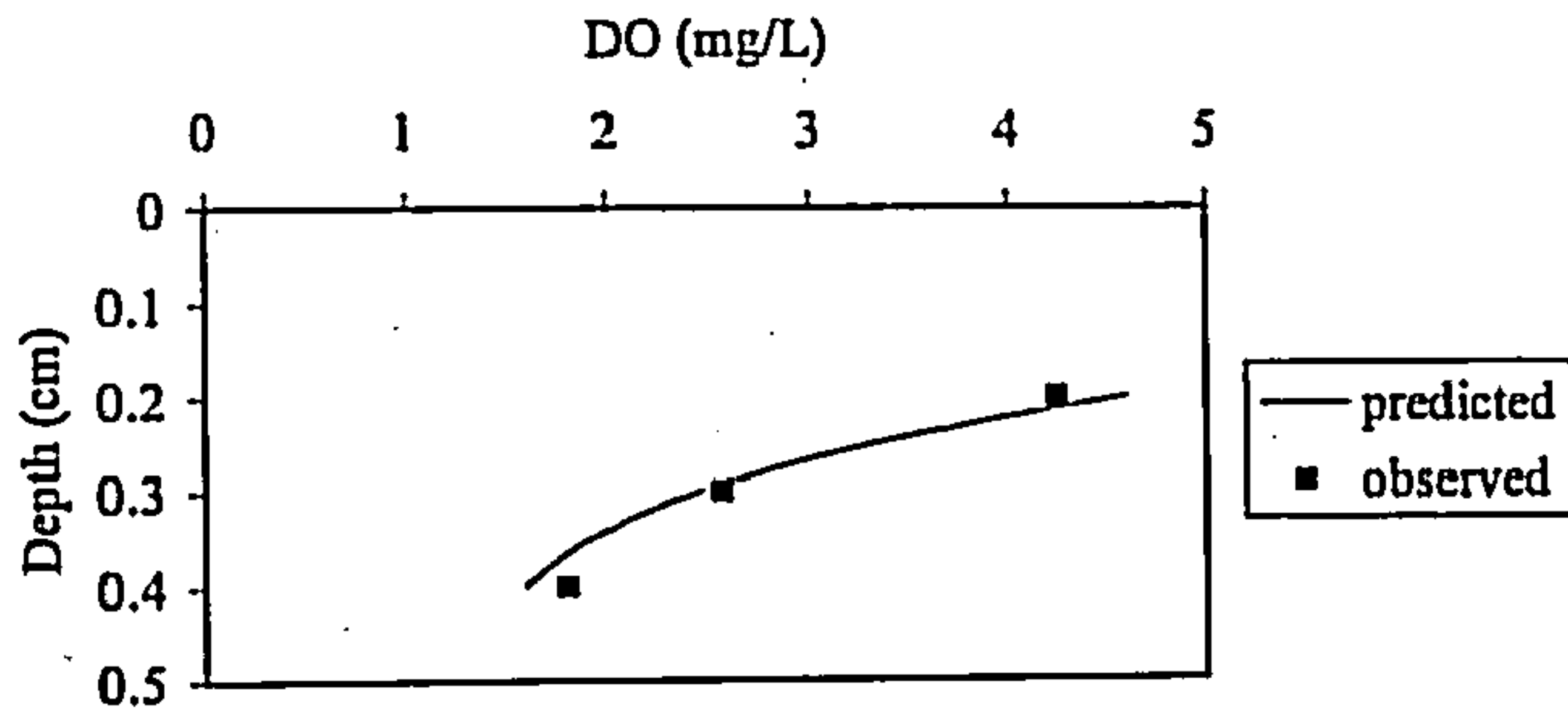
Figure 3.6. Dissolved oxygen profile in a 32 mm diameter gravel vegetated subsurface batch microcosm dosed with 10mg-N/L ammonia solution.
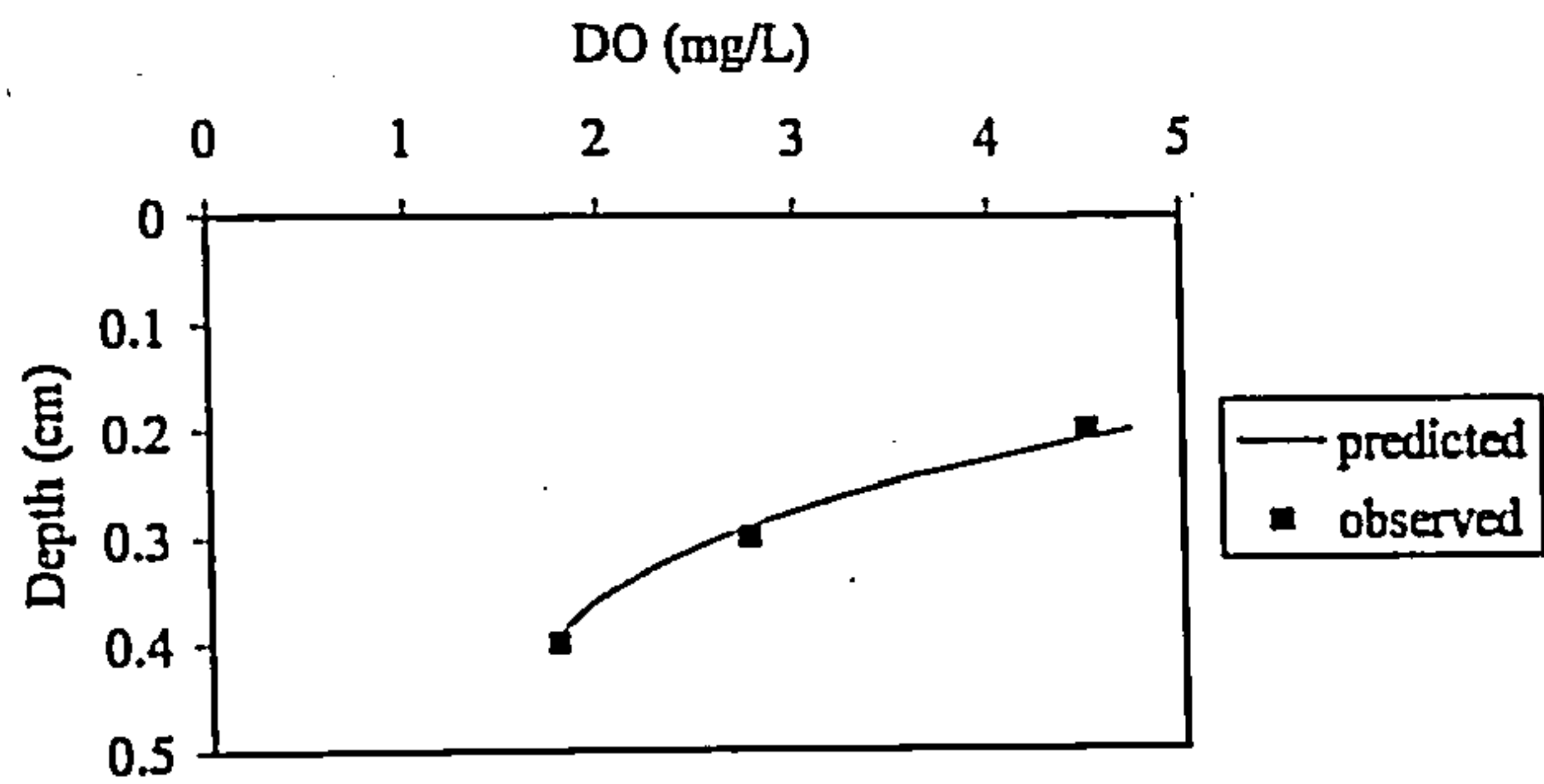
Figure 3.7. Dissolved oxygen profile in a 5 mm diameter gravel vegetated subsurface batch microcosm dosed with 10mg-N/L ammonia solution.

The predicted oxygen fluxes in this study as shown in Table 3.3 range from 0.027 to 0.32 $g\ O_2\ /m^2/day$ and are on the lower end of the range 0.077 to 7.92 $g\ O_2\ /m^2/day$ values reported in literature (Brix and Schierup, 1990; Nakano, 1999; and Wu et al., 2001). This range is more in agreement to the 0.077 $g\ O_2/m^2/day$ flux reported by Nakano (1999).

Table 3.3. Predicted direct atmospheric oxygen flux for different first order reaction rate constants extracted from literature.

Predicted O_2 Flux (g $O_2/m^2/d$)	Literature k (d^{-1})	Source
0.018	0.008	The current study
0.12-0.14	0.8-1.1	Crites, 1994
0.06	0.22	Tanner et al. 1995
0.14	1.104	Reed and Brown, 1995
0.17	1.84	Wood, 1995
0.15	1.35	Wood, 1995
0.12	0.86	Wood, 1995
0.07-0.32	0.3-6.11	Kadlec and Knight, 1996
0.027-0.14	0.1-1.2	Mashauri and Kayombo, 2002

From predicted (Table 3.3) and reported measured oxygen fluxes, it is inconclusive whether the model can accurately represent direct atmospheric oxygen fluxes in VSB constructed wetlands. However since the modeled fluxes concur with the lower end of the reported range it can be safely used as conservative design tool provided the reported fluxes were accurately measured and representative of different conditions.

Model sensitivity analysis was conducted to test the effect of changing model parameters on predicted oxygen fluxes as shown in Table 3.4. Changing the unsaturated zone diffusion coefficient over the range of values between free air and dry soil shown in Table 3.1 caused an approximate 63 % increase in oxygen flux. The model is also sensitive to changes in the diffusion coefficient of the saturated zone such that a change in the diffusion coefficient between free water and waterlogged soil leads to about a 53 % decrease in oxygen flux. The uncertainty in model parameters can lead to erroneous prediction of fluxes. However the results of the sensitivity analysis show that regardless of the accuracy of the assumed diffusion coefficient, predicted oxygen fluxes are maintained at the lower range of the reported values. Therefore the model can be used for conservative system design.

Table 3.4. Dissolved oxygen flux response to changes in diffusion coefficients and reaction rate constants.

	Flux (g O_2/m^2/d)			
k (d^{-1})	Unsaturated zone oxygen Diffusion Coefficients (m^2/d)		Saturated zone oxygen Diffusion Coefficients (m^2/d)	
	1.73	0.66	0.000173	0.000066
0.22	0.06	0.098	0.06	0.028
1.84	0.17	0.28	0.17	0.082
1.35	0.15	0.24	0.15	0.07
0.86	0.12	0.19	0.12	0.056
0.3-6.1	0.07-0.32	0.11-0.52	0.07-0.32	0.033-0.15
0.1-1.2	0.04-0.14	0.07-0.23	0.027-0.14	0.019-0.066

CONCLUSIONS AND RECOMMENDATIONS

A mechanistic model for atmospheric oxygen transport into VSB constructed wetlands was developed based on the two-layer model theory of a gas-liquid interface and molecular mass transport equations. The model satisfactorily predicts oxygen distribution in constructed wetland. The model predictive capability of oxygen flux into constructed wetland is inconclusive since it represents low fluxes contrary to limited literature values. The predicted fluxes therefore may be suitable for conservative system design. The accuracy of diffusion coefficients within practical limits does not have an impact on improving fluxes to represent the upper range of the reported values. There is a need to increase the data base for measured direct atmospheric oxygen diffusion to test model validity.

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CHAPTER FOUR

A REVIEW OF VEGETATED SUBMERGED BED CONSTRUCTED WETLANDS PERFORMANCE FOR NITROGEN REMOVAL

ABSTRACT

Removal of nitrogen (N) in constructed wetlands is rate limited by anoxic/anaerobic conditions. Nitrogen removal in wetlands is achieved through two pathways: (i) N cycling and (ii) storage. N cycling is a permanent removal pathway. The N cycling stages of importance for nitrogen removal in wetlands are ammonification, nitrification and denitrification. Storage is achieved by assimilation into plant and microbial biomass and adsorption. Storage is a temporary process since wetlands have a finite storage capacity and the stored N can be mineralized back into solution or undergo desorption. There has been an increasing interest in the development of technologies to alleviate nitrogen removal limitation in constructed wetlands by ensuring prevalence of conditions enhancing N cycling.

The purpose of this study is to review emerging technologies of vegetated submerged bed (VSB) constructed wetland systems aimed at improving nitrogen removal in wetlands through rational system design. The design and performance of these systems are reviewed. The oxygen transfer capacity and nitrogen removal mechanisms on system performance are evaluated. Constructed wetland most commonly consist of vertical flow (VF) and horizontal flow (HF) beds combinations, and HF beds with pulsing water level where VF and HF are aimed at nitrification and denitrification respectively. Nitrate

nitrogen accumulation is the most limiting factor in typical VF based systems and future research to address removal of nitrate nitrogen is highly recommended. Nitrogen treatment design criterion is presented.

INTRODUCTION

Vegetated submerged bed (VSB) constructed wetlands are low cost engineered wastewater treatment systems (Kadlec and Knight, 1996). There are two major types of VSB systems depending on flow regime. The most common systems have been designed with horizontal subsurface flow (HF) over the past 30 years (Cooper, 1999) but vertical flow (VF) systems are gaining popularity at present (Vymazal, 2005). It has been widely reported that the performance of both systems has been satisfactory where the standard level of treatment only requires removal of the 5-day biological oxygen demand (BOD_5) and total suspended solids (TSS) (US EPA, 1988; 1993; Cooper, 2001, Vymazal, 2005). However, there is a growing interest to achieve effluents low in nitrogen (Cooper, 1999). The removal of inorganic nitrogen is of major concern because of its toxicity to infants, fish and other aquatic animals and added oxygen demand (US EPA, 1988; 1993; Myrold, 1998). VF systems have been found to be more effective in removing nitrogen than conventional HF systems (Cooper, 2001; 2005).

Nitrogen in wastewater is present in particulate and dissolved organic and inorganic forms; the relative proportions of which depends on the type of waste and level of pretreatment (Jamieson, 2003). The primary inorganic forms of nitrogen entering

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wetlands are ionized and unionized ammonia (NH_4^+/NH_3) and nitrate (NO_3^-) nitrogen (Kadlec and Knight, 1996). The organic forms of nitrogen present in wetland are proteins, urea, uric acid, amines, purine, and pyrimides (Jamieson, 2003). The removal of particulate nitrogen is effected through settling and burial within the sediment while dissolved inorganic forms are removed by storage and through N cycling. Storage is only a temporary process because wetlands have a finite capacity and N assimilated to plant and microbial biomass or adsorbed to sediment can mineralize back into solution or undergo desorption. Removal of N through N cycling in wetlands is more permanent. Various forms of N in N cycling are converted into gaseous forms that are expelled into the atmosphere as nitrogen gas (N_2) or nitrous oxide (N_2O) in a two step process that involves nitrification and denitrification. Nitrification is an oxygen driven process which involves the transformation of NH_4^+ to nitrite (NO_2^-) then nitrate (NO_3^-). This process is substantially reduced at dissolved oxygen levels less than 1-2 mg/L in the water column (Hammer and Knight, 1994; Whitney et al., 2003). Denitrification is responsible for completing nitrogen removal by further transformation of NO_3^- to gaseous forms N_2O or N_2 which in turn volatilize out of the system.

The motivation for developing interest in VF and other variants of conventional HF wetlands is the desire to achieve fully nitrified effluents through sufficient oxygen transfer, an achievement not realized through use of the predominantly anaerobic HF systems (Cooper, 1999). In light of the concerns surrounding nitrogen treatment effectiveness in conventional VSB designs, new technologies have been and are still being developed to improve system performance while maintaining a cost competitive

edge over conventional mechanical wastewater treatment systems. In this study, oxygen transfer rates, nitrogen removal performance, and operation and maintenance costs between the different variants of VSB based treatment systems are reviewed for the formulation of design criteria. The effectiveness of alternating aerobic and anaerobic conditions in different treatment systems is also reviewed to formulate a proposal for optimizing total inorganic nitrogen removal. I believe that if properly designed alternating VF beds with conventional HF beds in series such that alternate aerobic and anaerobic conditions prevail for progressive removal of nitrate nitrogen in HF as it continues to be produced at VF stages will better regulate BOD removal. This approach will ensure that exogenous BOD contributes towards supporting denitrification in HF beds to improve overall nitrogen removal.

VEGETATED BED CONSTRUCTED WETLANDS

Background

There are two types of constructed wetlands that share many characteristics but are distinguished by the location of the hydraulic grade line; namely the free water surface (FWS) (also known as surface flow) and vegetated submerged bed (VSB) (also known as subsurface flow) wetlands. FWS resemble natural wetlands in appearance because they are populated with aquatic plants rooted in a soil layer on the bottom of the bed and hydraulic grade line exposed to the atmosphere. VSB do not resemble natural wetlands because they have no standing water, that is, their hydraulic grade line is submerged.

They contain a bed of media such as crushed rock, small stones, gravel, sand or soil, sea shells and plastic media which have been planted with aquatic plants (US EPA, 2000a; 1993; White, 1995). The common plants used in constructed wetlands are cattail (*Typha* spp), reeds (*Phragmites* spp), and Bulrush (Scirpus spp) because they are adopted to saturated conditions typical of constructed wetlands.

VSB wetlands have several advantages and disadvantages when compared to FWS wetlands and activated sludge systems. Some advantages of interest in this study are that VSBs: 1) provide effective treatment in a passive manner and minimize mechanical equipment, energy, and skilled operator attention; 2) are very effective and reliable for removal of BOD, TSS, and metals; 4) minimize odors, mosquitoes and similar insect vector infestations and contact by children and pets (US EPA, 1993, 2000b); and 5) provide greater available surface area for bacteria growth and subsequent treatment than FWS so the treatment responses may be faster and more sustainable. VSBs can therefore be smaller in land area than FWS systems designed for the same wastewater conditions (US EPA 1993).

The disadvantages of VSB systems are that they: 1) require large land area compared to conventional mechanical treatment processes; 2) removal of BOD, COD and nitrogen are partially temporary processes because there is cyclic release from system flora and fauna adding to system load; 3) water is anoxic thus limiting the potential for nitrification processes; 4) although VSB are smaller than FWS for the removal of most constituent contaminants, the high cost of the gravel media in the VSB systems can result in higher

construction costs (US EPA, 2000b); and 5) removal of nitrogen to low levels is possible but requires longer detention times (US EPA, 2000b). Increasing the wetland size and detention time compensates for this limitation, but this may not be cost effective (USEPA, 2000a, 2000b). Therefore there has been research interest in further development of VSB to mainly improve system aeration for the removal of nitrogen.

Oxygen transfer

There are two main documented pathways of oxygen transfer between the wetland water column and the atmosphere; plant roots oxygen release and direct atmospheric oxygen diffusion (Kadlec and Knight, 1996). Several attempts have been made to quantify oxygen transfer by the two pathways as shown in Table 4.1.

Table 4.1. Literature Oxygen transfer rates into VSB

Author	O_2 transfer rate $(g/m^2/day)$		Plants	Plant Root release determination Method
	Plant root release	Atmospheric diffusion		
Brix (1990)	0.02	3.78	*P australis*	Mass balance (respiration-culm flux)
Caffrey and Kemp (1991)	3.1-4.6	N/A	*P perfoliatus* *Z marina*	O_2-free solution
Kemp and Murray (1986)	0.41-0.53	N/A	*P perfoliatus*	O_2-free solution
Gries et al. (1990)	1.56-3.1	N/A	*P australis*	O_2-free solution
Nakano et al. (1999)	0.016	0.077	*T. latifolia*	Ti (III) buffer
Wu et al. (2000)	0.6 0.04	N/A N/A	*S pectinata* *T. latifolia*	Ti (III) buffer Ti (III) buffer
Wu et al. (2001)	0.02	6.01-7.92	*T. latifolia*	Ti (III) buffer

The quantification of plant-root oxygen release into VSB has remained a subject of debate (Bedford et al. 1991; Sorrel and Armstrong, 1994; Brix, 1997). As shown in Table 4.1, the amount of oxygen released vary between plant species and the methods of determining the oxygen release and that accounts for the differences in the reported plant root oxygen release rates. The titanium (III) buffer solution method has recently been chosen as the preferred method of quantifying plant root oxygen release after Sorrell and Armstrong (1994) compared this method with the oxygen-free methods. The titanium (III) buffer solution scavenge oxygen as it is released preventing it from being reabsorbed and this process is similar to natural sediments response to the release of plant root oxygen. Data from titanium (III) buffer solution method show that the amount of oxygen expected to be released by the plants is generally nominal in most VSB systems for all wetland plants as demonstrated in Table 4.1. Wu et al. (2000) observed that the amount of oxygen required to account for the observed nitrification indicated that release by both *T. latifolia* and *S pectinata* was very limited. Therefore it was concluded that the high nitrogen removal rate could have been a result of gravel sorption coupled with nitrification at the air-water interface. Two other studies by Brix (1990) and Wu et al. (2001) indicated that atmospheric oxygen diffusion into VSB was one order of magnitude higher than most reported plant root oxygen release. However passive oxygen transfer by both direct atmospheric oxygen diffusion and plant root release are inadequate compared to the amounts required for nitrogen removal in conventional VSB (US EPA, 2000a). However there is an indication that direct atmospheric oxygen diffusion provides a promising potential for oxygenation of constructed wetlands and research to explore this potential is necessary. Emerging technologies should therefore be formulated to improve

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oxygenation of VSB systems through the promotion of direct atmospheric oxygen transfer.

TECHNOLOGIES BASED ON PRINCIPLES OF CONVENTIONAL VSB

In the last ten years a new trend of research focusing on conventional VSB derived technologies aimed at enhancing oxygenation to achieve higher rates of nitrogen removal has emerged. These technologies comprise vertical flow VSB, horizontal VSB with pulsing water level and hybrids of horizontal and vertical flow VSB wetlands (Cooper, 1999, 2001; Vymazal, 2005, Behrends, 2006). In the current study, these technologies are reviewed on the basis of the mode of oxygenation enhancement; nitrogen and carbon removal performance, capital, operation and maintenance costs to formulate design criteria.

Horizontal Flow VSB with pulsing water levels

VSB technologies based on fluctuation of the water levels at varying frequencies in horizontal flow wetlands have been developed since the mid 1990s (Behrends, 1999). The oxygenation process was enhanced through automated pumping of wastewater back and forth between two adjacent wetland bed cells such that each cell was partially drained and filled on recurrent cycles. The drain stage of this reciprocating (ReCip) water treatment technology allows for atmospheric oxygen to diffuse into the biofilms attached to both plant roots and wetland substrate to promote aerobic processes as atmospheric air replaces water in the void spaces. The fill stage promotes anaerobic processes since these systems are poor at oxygenating the bulk water column. Alternating these aerobic and

anaerobic environments was expected to promote both nitrifying and denitrifying bacteria populations such that no particular bacterial consortia dominated the biofilm.

Several studies based on this principle with similar trends of ammonia-N; nitrate-N; and carbon removal performance were conducted (Behrends, 1996; Behrends, 2005 and Behrends et al., 2006). The most recent of these studies with the influent wastewater strength equivalent to an alternative technology reviewed later in this chapter is used in the current study to evaluate the treatment potential of this technology (Behrends, 2006).

In the current study, long term treatment performance of the integrated wetland treatment systems (IWTS) consisting of ReCip coupled with conventional constructed wetland beds in series was monitored for a period of 28 months. The first system consisted of surface flow (SF) constructed wetland followed by a ReCip bed and the second consisted of a ReCip bed followed by a subsurface flow (SSF) wetland. These systems treated clarified wastewater from recirculating aquaculture system.

Table 4.2. Performance data for two Integrated Wetland Treatment System (IWTS) where reciprocating bed is operated in series with conventional HF beds (Behrends et al., 2006).

| Parameters | IWTS I | | | IWTS II | | |
	Influent Conc. (mg/L)	SF Effluent (mg/L)	ReCip Effluent (mg/L)	Influent Conc. (mg/L)	ReCip Effluent (mg/L)	SSF Effluent (mg/L)
COD	172	107	28	182	46	24
TAN	50.9	45.9	3.8	51.4	6.0	3.4
TKN	72.2	58.2	3.9	66.3	10.4	3.4
NO_3^--N	0.16	0.65	27.91	0.16	8.79	3.4
DO	0.7	1.4	1.7	1.0	1.4	1.0

Results in Table 4.2 demonstrate a typical treatment performance of ReCip. The removal of total nitrogen was promising as demonstrated in Table 4.2. Despite the low level of dissolved oxygen (DO) in the bulk water, the removal of both total nitrogen and COD was adequate. This observation demonstrate that the sufficient transfer of oxygen into biofilms during the drain stage of the reciprocation process lead to the oxidation of carbon and nitrogen without carbon inhibitory effects on ammonia oxidation typical of conventional VSB treating low DO wastewater. In conventional VSB it has been observed that COD/N ratio >= 1 together with low DO caused the inhibition of nitrogen oxidation by carbon because the heterotrophic bacteria responsible for carbon oxidation grows faster and out compete autotrophic bacteria that oxidize ammonia nitrogen for limited DO (Hanaki et al., 1990; Williams et al., 1994 and Nakano, 1999). Therefore in the current study the authors' claim of enhanced oxygen transfer into the biofilms instead of bulk water for providing alternate aerobic and anaerobic conditions was justifiable as evidenced by adequate removal of both nitrogen and carbon at low water column DO. With respect to nitrate nitrogen removal, the authors of this work concluded that the efficient removal of suspended solids and associated organic carbon in the surface flow (SF) wetland of IWTS 1 deprived the downstream reciprocating bed of organic carbon necessary for nitrate nitrogen removal. Furthermore this conclusion was appropriate because COD removal of 79 mg/L accounted for 37 mg/L of carbon utilized in removing 15 mg/L nitrate nitrogen at a rate of 2.47 mg/L organic carbon per 1 mg/L of nitrate nitrogen (Kadlec and Knight, 1996; Lin, 2001). The remaining 42 mg/L carbon can be accounted for by aerobic respiration. The carbon utilized for nitrate nitrogen removal was

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approximately half of the total carbon removed. This was an indication that aerobic environment intended for the promotion of nitrification has a negative effect of depleting organic carbon necessary for subsequent denitrification.

Moreover, as demonstrated in IWTS II system (Table 4.2), when the influent COD concentration was higher there was better removal of generated nitrate nitrogen at a similar influent nitrogen loading rate as in IWTS I. Therefore a higher COD/N ratio in system IWTS II led to better removal of nitrate-N. Therefore wastewater COD/N ratio can be used to predict expected system performance. However there is need to conduct studies on varying levels of COD/N ratios to find threshold values appropriate for effective total nitrogen removal.

It can also be concluded that under low influent COD/N ratio there is a need to design systems that can slow down carbon oxidation under aerobic environment by providing longer retention time for anaerobic than aerobic environment. This can be achieved through a configuration similar to IWTS II in Table 4.2 where nitrate nitrogen generated from a reciprocating bed can be removed by a predominantly anaerobic SSF or SF bed. An alternative to explore is the use of the ReCip bed together with a reservoir of smaller footprint than a constructed wetland to provide longer retention time in an anaerobic environment for the removal of nitrate nitrogen.

When selecting a low cost wastewater and stormwater treatment technology the treatment performance assessment must be done together with cost effectiveness evaluation of the

technology. In light of this consideration, when compared to a hybrid technology that utilizes a similar principle of operation discussed later in this chapter, the ReCip technology has higher operation and maintenance cost due to pumps necessary for the reciprocation process between contiguous bed cells. However, this technology offers an alternative to hybrid systems that require more land for the same level of treatment.

Vertical and horizontal flow VSB hybrids

Under this type of system, there are combinations of two types of VSB beds utilized to achieve enhanced total nitrogen removal through nitrification and denitrification. The vertical flow (VF) VSB beds are predominantly aerobic and used for nitrification while conventional horizontal flow VSB beds are mainly anaerobic and therefore suitable for denitrification. These two types of VSB are used in series and various combinations. In a VF bed, water is distributed by pipes on the surface of the gravel and allowed to trickle down by gravity without saturating the substrate. The water is collected at the base of the bed.

Case studies of this type of technology are used to review the performance of this system. Data in Table 4.3 show the different VF and HF hybrid systems' BOD_5 and nitrogen removal performance and oxygen transfer rates. Under each system the bed types are placed in series in the order they are arranged in Table 4.3. The hybrid systems are categorized according to the sequence of the VF and HF stages in series. Systems (1) to (4) in Table 4.3 represent performance of hybrid systems where VF beds precede HF beds and systems (5) and (6) represent systems where HF beds precede VF beds. System (7) represents a system where HF beds are alternated with a VF bed. The arrangement of

the bed types in the different hybrid systems dictates the system carbon and nitrogen removal performance. VF beds support predominantly aerobic autotrophic and heterotrophic bacteria mediated oxidation of nitrogen and carbon in that order while HF beds mainly supports anaerobic heterotrophic oxidation and reduction of carbon and nitrate nitrogen respectively.

In systems (1) to (4) despite differences in system hydraulic loading rates (HLR) and influent carbon and nitrogen concentration, the treatment performance patterns in terms of BOD_5 and nitrogen removal are similar between systems in that there is satisfactory removal of BOD_5 while nitrification is high but incomplete. A BOD_5 and nitrogen mass balance of system (1) is analyzed to assess the treatment trends and corresponding implications on system performance demonstrated by hybrid systems (1) to (4). In this system the removal of BOD_5 did not inhibit nitrification as demonstrated by decrease in NH_4^+-N concentration from 50.5 mg/L to 14 mg/L through the VF stages. This is an indication of highly efficient wastewater oxygenation by the VF beds compared to HF beds. In the two VF stages the increase in NO_3^--N and the corresponding decrease of ammonium-N demonstrate nitrification in these two stages even though this was not enough to achieve full nitrification. This might be caused by undersized VF beds and inadequate retention time. In addition to this mass balance analysis, an assessment of the VF stages of systems (3) and (4) where system (4) performs relatively better in nitrification and BOD_5 removal than system (3) by virtue of being larger in surface area and thus lower HLR suggests that bed sizing at this stage of treatment has an important effect on the magnitude of nitrification and BOD_5 removal achieved. These are crucial

observations that suggest systems can be designed to regulate aerobic processes through VF bed sizing for rationed carbon removal between aerobic and anaerobic stages of a hybrid system.

The NO_3^--N produced in VF beds of systems (1) to (4) was partially removed by HF beds. This observation show that there was insufficient exogenous carbon left in the system to support complete denitrification in the predominantly anaerobic HF stages. In system (1) for example NO_3^--N decreased from 22.5 mg/L to 7.2 mg/L, a decrease in concentration of 15.3 mg/L despite only 7 mg/L BOD_5 decrease through the HF stages. Since a concentration of 2.47 mg/L BOD_5 is required to remove 1 mg/L of NO_3^--N therefore a BOD_5 reduction of 7 mg/L only accounted for 2.8 mg/L of NO_3^--N removal. The remaining 12.5 mg/L NO_3^--N decrease through the HF stages can be accounted for by denitrification supported by endogenous carbon supply and longer retention time. This added advantage of endogenous carbon supply can be further augmented by exogenous carbon supply to complete NO_3^--N removal. This observation shows that there is a need to provide a system design where alternate aerobic and anaerobic environments are regulated for rationed removal of exogenous carbon between the two environments to improve NO_3^--N removal and thus overall nitrogen removal efficiency.

Systems (5) and (6) further demonstrate that even though the level of nitrification and BOD_5 removal achieved was satisfactory there was a net accumulation of NO_3^--N through this type of system. The accumulated NO_3^--N in systems (5) and (6) in terms of

proportion of system influent NH_4^+-N concentration to effluent NO_3^--N concentration was relatively higher than in system type represented by systems (1) to (4). This observation is evidence of the need to use an HF bed to remove NO_3^--N generated in VF bed stage of a hybrid system. System (7) was the most balanced of the seven evaluated systems in terms of nitrification and denitrification. System (7) demonstrated higher nitrification and satisfactory BOD_5 removal while maintaining lower levels of NO_3^--N. However this system despite being operated at a similar magnitude of influent carbon and nitrogen concentration compared to the other six systems was relatively lightly loaded at 6.2 cm/day HLR compared to up to between 9-75 cm/day and this could have partially accounted for this system's better performance but also showed the promising treatment potential of alternating VF beds with HF beds in overall nitrogen removal.

It can be concluded that the nitrogen and carbon removal performance trends in this technology was similar to that of ReCip system and that the most limiting factor of the two systems was nitrate nitrogen accumulation mainly caused by the depletion of carbon during the aerobic phase of the two systems. Therefore the main deficiency of these technologies that require more future attention is the optimization of nitrate nitrogen removal. This can be achieved by rational partitioning of exogenous organic carbon oxidation between aerobic and anaerobic environments.

Table 4.3. Performance data for hybrid systems consisting of different combinations of VF and HF beds in series.

Source[§] System No.	System configuration	Surface Area (m^2)	HLR (cm day^{-1})	Wastewater constituent concentration (g m^{-3})						OTR (g m^{-2} day^{-1})
				Influent BOD$_5$	Effluent BOD$_5$	Influent NH_4^+	Effluent NH_4^+	Influent NO_3^-	Effluent NO_3^-	
(1)	VF	8	n/a	285	57	50.5	29.2	1.7	10.2	n/a
	VF	5	n/a	57	14	29.2	14	10.2	22.5	n/a
	HF	8	n/a	14	15	14	15.4	22.5	10.0	n/a
	HF	20	n/a	15	7	15.4	11.1	10.0	7.2	n/a
(2)	VF	16	75.0	269	171	45	28	0.5	21.3	32
	VF	30	40.0	171	43	28	16	21.3	17.2	28
	HF	60	20.0	43	27	16	15	17.2	12.3	1.64
(3)	VF	6.3	31.7	150	40	86	65	n/a	30	67
	HF	60	3.0	40	35	65	40	30	8	3.22
	HF	40	5.0	35	20	40	25	8	9	3.23
(4)	VF	25	8.0	105	20	70	25	n/a	n/a	n/a
	HF	110	1.8	20	10	25	15	n/a	n/a	n/a
(5)	HF	140	7.9	67	25	32	27	0.2	0.4	5
	VF	121	9.1	25	2	27	0.1	0.4	27	29
(6)	HF	456	3.1	423	53.5	18	14.7	1.3	2.5	11.6
	VF	30	46.7	53.5	40.5	14.7	2.9	2.5	9.8	27
(7)	HF	1050	1.8	273	140.2	67.1	51.2	0.32	0.36	2.87
	VF	300	6.2	140.2	48.5	51.2	15.9	0.36	5.14	10
	HF	500	3.7	48.5	20.4	15.9	5.9	5.14	2.21	1.60

[§] (1) Cooper, 2001; (2) O'Hogain, 2003; (3) Mæhlum and Stålnacke,1999; (4) Mæhlum and Stålnacke,1999; (5) Laber et. al., 2003; (6) Brix et al., 2003 (7) Obarska and Gajewska; 2003. [ʃ] Unavailable data.

Vertical and horizontal flow VSB hybrid design proposal

This type of technology offers lower operation and maintenance cost of the two reviewed system types and yet it is the less developed in terms of optimizing treatment potential. As a first step towards enhancing the potential of the VF and HF based treatment technology, an oxygen transfer rate based design together with necessary future research work are suggested.

Oxygen transfer rates (OTR) for VF stages of the systems are estimated from Equation 4.1 formulated by Cooper (1999). The calculated OTR is an estimate because it does not take into consideration the BOD removal by settlement and filtration; NH_4^+-N lost to plant uptake and to the air through ammonia volatilization; and NH_4^+-N gained through ammonification of influent organic nitrogen. However BOD_5 adjustments for oxygen recovered during denitrification are estimated from the nitrogen mass balance between NO_3^--N and NH_4^+-N at a rate of 2.47 mg BOD_5 utilized per 1 mg NO_3^--N removed. The objective of estimating OTR is to find a typical OTR for design purposes.

$$OTR = \frac{Flow \times \left[(BOD_5\ in - BOD_5\ out) + 4.3 \left(NH_4^+ - N\ in - NH_4^+ - N\ out \right) \right]}{total\ surface\ area\ of\ beds} \quad (4.1)$$

The oxygen transfer rates of the VF beds as shown in Table 4.3 are for most systems around 30 g m^{-2} day^{-1} and are independent of the HLR a feature that suggests this transfer rate is representative of VF beds. Other similar studies also reported an oxygen transfer rate averaging 30 g m^{-2} day^{-1} (Brix et al., 2002; Weedon, 2003; Cooper and Cooper,

2005) The high transfer rate of 67 g m^{-2} day^{-1} in system (3) is most likely an over estimation caused by BOD$_5$ and/or NH$_4^+$-N removal by other pathways other than oxidation. The relatively low transfer rate of 10 g m^{-2} day^{-1} in system (7) is achieved under relatively low HLR however it is inconclusive as to whether HLR caused this low relative rate of oxygen transfer rate. It can therefore be concluded that a design oxygen transfer rate of 30 g m^{-2} day^{-1} is representative of VF stages of the hybrid system.

The design criterion presented in the current study is based on the need to achieve effluents low in both ammonium-N and total oxidized nitrogen (TON). The total surface area (TSA) of VF beds required for nitrification can be determined by rearranging Equation 4.1 and setting the BOD$_5$ and ammonium-N discharge limits as shown in Equation 4.2. This area should be adjusted for the expected nitrification and BOD$_5$ removal in HF beds of a hybrid system.

$$TSA\,of\,VF\,beds = \frac{Flow \times \left[(BOD_5\,in - BOD_5\,limit) + 4.3(NH_4^+ - N\,in - NH_4^+ - N\,limit)\right]}{OTR} \quad (4.2)$$

The recommended OTR as discussed before for VF beds is 30g m^{-2} day^{-1}. This first step of the design ensures that there is sufficient oxidation potential in the system. The second step is to divide the total VF bed surface area into a number of smaller beds to attain the desired HLR for rationed carbon removal while achieving incremental nitrification as show in Equation 4.3.

$$Individual\ VF\ bed\ surface\ area = \frac{flow}{HLR}$$

$$No\ of\ beds = \frac{TSA\ of\ VF\ beds}{Individual\ VF\ bed\ surface\ area} \qquad (4.3)$$

There is no clear relationship between HLR and BOD/NH_4^+-N removal ratio in the reported studies but hydraulic loading rate in the region of 70 cm/day seem to result in a satisfactory nitrogen and low carbon removal. This loading rate is subject to debate since it is based on only one study (study 2), therefore there is need to conduct studies to find an HLR range for suitable stepwise BOD/NH_4^+-N removal ratio for varying waste strengths. The third and final step is to decide on the system configuration. In this step a small HF bed should precede the first VF bed for total suspended solids (TSS) removal since VF beds are reported to be prone to clogging by suspended solids (Platzer and Mauch 1997; Langergraber et al. 2003; Kay and Kunst 2005; Cooper, 2005). The subsequent HF and VF beds should be alternated.

CONCLUSIONS AND RECOMMENDATIONS

VSB system with pulsing water level demonstrated promising nitrogen removal potential through effective provision of aerobic environment for the adequate removal of ammonium nitrogen. The system also provided anaerobic environment for partial removal of nitrate nitrogen. The most limiting treatment factor in this type of system is the net accumulation of nitrate nitrogen. The accumulation of nitrate nitrogen is caused by the depletion of organic carbon during aerobic respiration phase leaving limited

carbon to support denitrification in anaerobic environment. Therefore future work aimed at alleviating this limitation is recommended. The other design limitation associated with this type of system is the need to automate the pulsing process resulting in higher costs and increased complexity of operation which may serve as a deterrent to potential users. Therefore there is need to investigate ways to formulate less complex and lower cost pulsing water level systems to compete with VF-HF VSB hybrid wastewater treatment systems.

VF-HF VSB hybrid system demonstrated promising potential in achieving required levels of nitrification but limited utility in terms of nitrate nitrogen removal. When VF beds are placed before HF beds in series they deplete organic carbon necessary for denitrification in HF beds. However HF beds demonstrated promising potential for internal carbon supply to enhance denitrification even though nitrate removal is not complete. When HF beds precede VF beds the HF beds effectively remove TSS and this effectively alleviates clogging of VF beds. However there is higher accumulation of nitrate-N detected at the effluent of systems where VF beds are used as final treatment beds. Therefore there is need to investigate an arrangement where HF and VF beds are alternated with HF being the first and final beds in this configuration. The first HF removes TSS to protect VF beds from clogging while the final HF would polish off nitrate-N generated in the final VF bed.

The presented VF-HF hybrid wastewater treatment system design based on the performance of existing systems is the first step towards achieving rational design criteria

for carbon and nitrogen removal in constructed wetlands. There is however the need to conduct studies to provide additional data for informed selection of hydraulic loading rates as a function of BOD_5 and ammonium nitrogen influent concentration at VF stages of the system. There is also the need to study the oxidation potential of the HF beds and use this information in the refinement of the design criteria of the hybrid system by properly adjusting for the total oxidation required at VF stages of the system.

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