Modeling Approach on Studying the Effect of Water Level Fluctuation on Belowground Oxygen Dynamics in Wetland Mesocosms

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Abstract

Direct modeling approaches on evaluating the effect of water level fluctuation on belowground oxygen dynamic is scarce. Therefore, a simultaneous effort on experiment and modeling was carried out to develop, validate and evaluate the effect of water level fluctuation on belowground oxygen dynamics and concentration.

Eighteen number of vertical flow wetland mesocosms were designed to have four different systems as static water level systems with plant (SWLP) and without plant-control (SWLC) and fluctuating water level systems with plant (FWLP) and control-without plant (FWLC). They were fed continuously with the synthetic wastewater through an inlet at the bottom and discharged at an outlet just above the substrate surface. In FWL systems, wastewater was drained out every 3.5 days via a port at 50 cm, below the substrate. The experiment was conducted for 240 days and dissolved oxygen (DO) concentrations were monitored at five depths of the substrate, weekly. Numerical models were developed for each system to simulate oxygen diffusion and its related processes, during cycles of each for 3.5 days with four different hydraulic regimes of draining, rising, filling and steady states. The simulation study was carried out for the same experimental period.

The model results were compatible with the experimental values at all the cases until the later stage of the experimental period where the temperature at green house went below 15°C, during the winter season. The experimental results showed that the oxygen concentration in fluctuating water level systems (FWL) was 5-32% higher than that in static water level systems (SWL), with a considerable difference particularly at middle layers (27% - 32% higher). Model results on oxygen concentration at five depths of substrate generally agreed with experimental observation and revealed that six-day interval of water level fluctuation created higher efficiency on ammonium and organic carbon removal in terms of oxygen consumption comparing to 2.0 and 3.5day frequencies. Consequently, the experiment could prove the effectiveness of modeling DO variation as an effective technique in deciding the optimized systems for water level fluctuation.

Keywords: Aeration, Biodegradation, Diffusion, Mass Balance, Nitrification

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Introduction

Availability of oxygen is a vital factor on deciding the fate of pollutants in Constructed Wetland (CW) systems. The major treatment processes in CWs were recognized as sedimentation, filtration, sorption, microbial decomposition, nitrification, and denitrification (Wynn and Liehr, 2001). Most of these are related to the existence of oxygen. At the waterlogged conditions, oxygen is usually supplied into sediment by means of inflow, diffusion of atmospheric oxygen from the air-water interface, and root oxygen release (Armstrong et al.,1994a). Release of root oxygen varies with temperature, photoperiod (Gelda and Effler, 2002), substrate structure (Brix, 1990), and wastewater characteristics (Broecker et al., 1980; Thomann and Mueller, 1987).

Wetland plants are morphologically adapted to grow in the flooded conditions with the assist of comparatively higher internal air space for oxygen transportation from atmosphere to roots and rhizomes (Armstrong et al.,1994a). The oxygen released from roots creates oxidizing conditions in the anoxic substrate and stimulates aerobic decomposition of organic matter and growth of nitrifying bacteria (Brix , 1993). However, the oxygen transferred from atmosphere into deep stratum is assumed very small. Therefore, it could be speculated that the performance of wetland treatment can be increased by enhanced aeration. Water level fluctuation has been studied since long time as an artificial technique for aeration (Chabbi et al., 2000).

Many models have been developed so far to simulate oxygen processes in wetland systems, including oxygen transportation via plants (Armstrong et al.,1992), microbial consumption (Wynn and Liehr, 2001), mass transfer through the air-water interface (Molder et al., 2005), physical processes as diffusion, and convection (McGechan et al., 2005). However, it is rather difficult to find out models those simulate the oxygen transportation mechanisms during water level fluctuation. Subsequently, this study was focused on evaluating the oxygen concentration at static and fluctuating water levels with and without plants. The objectives of the study were to; (a) evaluate oxygen concentration at different depths under draining and flooding conditions in vertical flow wetland mesocosms and, (b) develop numerical models to simulate oxygen consumption and transformation mechanisms in bed media.

Material and Methods

Greenhouse experiment

The substrate layer of the mesocosms, constructed with PVC columns of 20 cm in diameter and 60 cm in height, were divided into two compartments. The bottom layer of 7 cm was amended with medium sized gravel and the upper layer was with fine sand to form a substrate layer of 50 cm in depth. Each column was arranged to have an inlet and an outlet of 4mm inner diameter, near the bottom and just above the sand layer respectively in order to facilitate a vertical flow.

Typha orientalis were collected from Minumatanbo wetland area (35° 51′ N and 139° 39′ E), Saitama, Japan in the middle of April 2007 and acclimatized in the greenhouse conditions for two weeks. Then the plants were uprooted, washed with tap water and cleaned off debris and dead parts. Uniformed sized young *T. orientalis* were planted in fourteen wetland columns, while four kept unplanted as controls. These systems were first fed only with tap water for the two weeks as an establishment period for plants and then with the synthetic wastewater (Table 1) at a constant head to maintain a uniform inflow rate for a total experimental run of 240 days continuously.

Table 1. Composition of influent synthetic wastewater

Parameter	Concentration (gL^{-1})
Potassium nitrate	0.07
Ammonium chloride	0.08
Urea	0.056
Glucose	0.0325

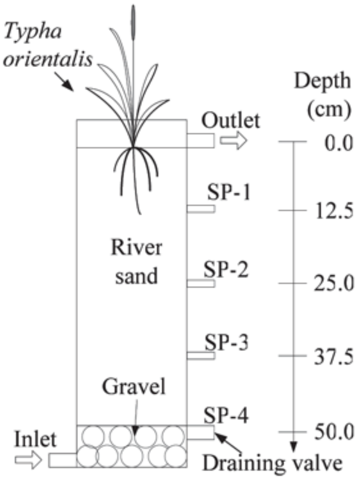


Figure 1. Column arrangement

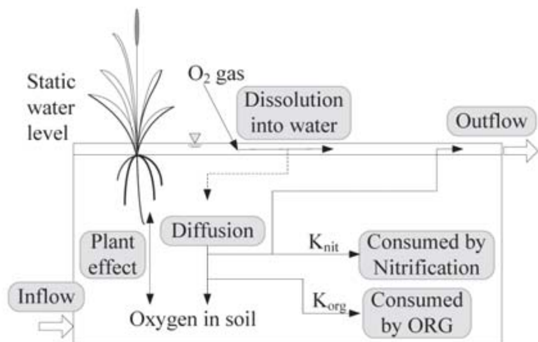
Four sampling ports (SPs) were arranged at 12.5, 25.0, 27.5, and 50 cm depths below the sand surface in each column (Figure 1). Sampling was done weekly at six sampling points as inlet, outlet and four ports in order to measure DO and evaluate the biological oxygen demand.

Model development

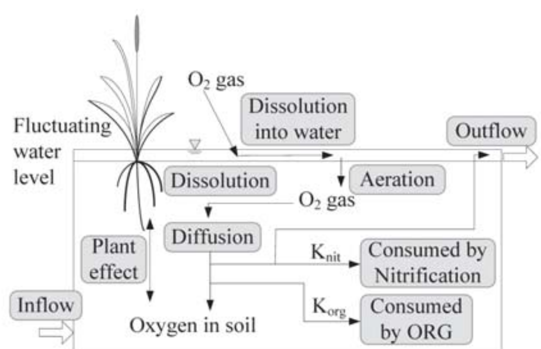
Four distinguish models were developed to simulate the depth wise variation of oxygen concentration in SWLC, SWLP, FWLC and FWLP systems. The models were based on the theory of mass balance (Jorgensen and Bendoricchio, 2001), which could be simply expressed as:

Accumulati on= input − output □ reaction (1)

The physical and biochemical processes included in the models were aeration, diffusion, gaseous oxygen dissolution, nitrification, organic degradation, and plant effects i.e. respiration and oxygen release (Figure 2). The models for FWL conditions were consisted of four hydraulic regimes as draining, raising, filling and steady state for a period of 1 h, 24 h, 7 h and 52 h respectively.



(a) Static water level (SWL) conditions



(b) Fluctuating water level (FWL) conditions

Figure 2. Physical and biochemical unit processes used in modeling

Aeration at air – water interface

Atmospheric oxygen with higher partial pressure tends to dissolve in unsaturated water as explained in Henry's law (Hillel, 1998). This mechanism plays a vital role in regulating the concentrations of various constituents (Thomann and Mueller, 1987). Ambient temperature could be the governing factor on deciding the amount of saturation level (Kadlec and Knight, 1996). Therefore, the effect of temperature T (°C) on aeration can be simulated with the following equation.

$$[O_2]_{aeration} = 14.652 - 0.41022\,T + 0.007991T^2 - 0.00007777\,T^3 \quad (2)$$

Oxygen diffusion

It is presumed that the diffusion is the key mechanism to transfer oxygen from surface layer to underneath substrate. However, the diffusion coefficient is much lower in water filled pores than that in gas filled pores (10^{-4}) (Hillel, 1998). The effective diffusion coefficient in the liquid phase of soil is also less than that in bulk water since the liquid phase occupies only a fraction of the soil volume, and the pore passages of soil are tortuous. Therefore, the actual path length of diffusion becomes significantly greater than the apparent straight-line distance (Hillel, 1998). Oxygen diffusion ($[O_2]_{diffusion}$) could be expressed according to the Fick's first law (Aachib et al., 2004) as following:

$$[O_2]_{diffusion} = -\,D_o dc/dx \quad (3)$$

where D_o is the diffusion coefficient in an environment of either bulk water or gaseous phase or liquid phase, and dc/dx is the solute's effective concentration gradient.

Dissolution of gaseous oxygen in pore water

The equilibrium of DO concentration in soil water increases with the pressure while decreases with temperature (Hillel, 1998) and could be calculated as:

$$[O_2]_{dissulution} = -k_{dis} D_o dc/dx \quad (4)$$

where k_{dis} is solubility coefficient of gaseous oxygen in water.

Biochemical reactions

Microbiological community in soil capitalizes the DO for the biological degradation of organic compounds, which is much faster than in anaerobic condition (Vymazal et al., 1998; Noorvee et al., 2007; Ouellet-Plamondon et al., 2006). The amount of oxygen consumed in this processes ($[O_2]_{org}$) could be speculated to follow up the first – order kinetic (Reddy and Patrick, 1984) as:

$$[O_2]_{org} = k_{org}[O_2]_{soil} \quad (5)$$

In addition, oxygen involves in nitrification process, decreasing ammonia concentration in soil water, while producing nitrate nitrogen. The oxygen concentration consumed in nitrification process ($[O_2]_{nit}$) also could be depicted with the first – order kinetic (Reddy and Patrick, 1984)as:

$$[O_2]_{nit} = k_{nit}[O_2]_{soil} \quad (6)$$

where ($[O_2]_{soil}$) is oxygen concentration in soil water, k_{nit} and k_{org} are nitrification and decomposition coefficients respectively and can be calculated at temperature T (°C) (Reddy and Patrick, 1984), using the following general equation :

$$k = k_{20}\theta_{20}^{(T-20)} \tag{7}$$

Oxygen respiration and release by plants

Wetland plant roots were observed to have both oxygen release and uptake abilities. As the soils in wetland are water-saturated, the rhizomes and roots of wetland plants are modified to consume the oxygen transported from living leaf sheaths, culms nodes and broken culms (above ground portions) for respiration (Brix et al.,1992). Respiration of aquatic plants is influenced by several environmental parameters, including temperature, DO concentration (Dawson et al., 1981; Sorrell and Dromgoole, 1989), tissue nutrient concentration, and light (Auer and Canale, 1982).

Oxygen is released through root system in order to maintain aerobic conditions for oxidizing already reduced compounds in the rhizosphere (Pezeshki, 2001). Young lateral roots are determined as the major source of oxygen release and therefore greater numbers of them may enhance the oxygen release (Armstrong and Armstrong, 1990).

In the models, the amount of plant affected oxygen concentration ($[O_2]_{plant-effect}$) was incorporated using the disparity between the planted and controlled or unplanted mesocosms.

Governing equations

The mass balance equations considering the above unit processes on oxygen consumption and transformation processes of SWLC, SWLP, FWLC and FWLP systems are given as:

For SWLC system:

$$\begin{aligned}\frac{dO_2(i)}{dt} = {} & \frac{Q_{in}}{V}[O_2]_{in} + [O_2]_{(i)\,aeration} \\ & \pm [O_2]_{(i)\,diffusion} - k_{org}[O_2]_{(i)soil} \\ & - k_{nit}[O_2]_{(i)soil} - \frac{Q_{out}}{V}[O_2]_{(i)out}\end{aligned} \tag{8}$$

For SWLP system:

$$\begin{aligned}\frac{dO_2(i)}{dt} = {} & \frac{Q_{in}}{V}[O_2]_{in} + [O_2]_{(i)areation} \\ & \pm [O_2]_{(i)\,diffusion} + [O_2]_{(i)\,plant-effect} \\ & - k_{org}[O_2]_{(i)soil} - k_{nit}[O_2]_{(i)soil} \\ & - \frac{Q_{out}}{V}[O_2]_{(i)out}\end{aligned} \tag{9}$$

For FWLC system:

$$\begin{aligned}\frac{dO_2(i)}{dt} = {} & \frac{Q_{in}}{V}[O_2]_{in} + [O_2]_{(i)aeration} \\ & + [O_2]_{(i)\,dissoluion} \pm [O_2]_{(i)\,diffusion} \\ & - k_{org}[O_2]_{(i)soil} - k_{nit}[O_2]_{(i)soil} \\ & - \frac{Q_{out}}{V}[O_2]_{(i)out}\end{aligned} \tag{10}$$

For FWLP system:

$$\begin{aligned}\frac{dO_2(i)}{dt} = {} & \frac{Q_{in}}{V}[O_2]_{in} + [O_2]_{(i)\,aeration} \\ & + [O_2]_{(i)\,dissolution} \pm [O_2]_{(i)\,diffusion} \\ & + [O_2]_{(i)\,plant-effect} - k_{org}[O_2]_{(i)soil}\end{aligned} \tag{11}$$

The daily changes of oxygen concentrations at various layers as inflow zone, middle, upper and water surface layers were calculated using differential equations. Parameters used in the models are depicted in Table 2. Model calibration was done with the corresponding observed data obtained during the greenhouse experiment.

Table 2. Parameters used in the model

Model parameters	Notation	Unit	Value
Nitrification rate	K_{nit20}	day^{-1}	0.05(Ouellet-Plamondon et al., 2006)
Arrhenius constant for nitrification	θ_{nit}	-	1.07(Ouellet-Plamondon et al., 2006)
Organic carbon disintegration	K_{org20}	day^{-1}	0.5[e]
Arrhenius constant for disintegration	θ_{org}	-	1.07(Ouellet-Plamondon et al., 2006)
Solubility coefficient	K_{dis}	-	0.03 (Hillel, 1998)
Oxygen diffusion coefficient in water	D_w	$dm^2\ day^{-1}$	0.019 (Kadlec and Knight, 1996)
Oxygen diffusion coefficient in saturated sand	D_{sat}	$dm^2\ day^{-1}$	$2.4\ x10^{-6}$[c]
Oxygen diffusion coefficient in unsaturated sand	D_{unsat}	$dm^2\ day^{-1}$	$2.4\ x10^{-8}$[c]
Oxygen concentration of inflow	$[O_2]_{inflow}$	mgL^{-1}	6.0 [ed]
Inflow rate	Q_{in}	$dm^3\ day^{-1}$	2.88 [ed]
Outflow rate	Q_{out}	$dm^3\ day^{-1}$	2.88 [ed]
Soil water volume	V	$dm^3 Layer^{-1}$	0.078 [ed]

Reference: c-calibrated, ed-experimental data, e-estimated

Results and Discussion

Comparison of model and experimental results on soil water DO concentration

DO concentration in pure water depends mainly on temperature. It is high at low water temperature and decreased gradually with increasing temperature (Kadlec and Knight, 1996). Calculated results in static water level systems (with and without plant) showed the daily oscillation of DO by temperature, clearly. The daily fluctuation of soil water oxygen concentration in summer was significantly lower than in winter period. On the contrary, experimental data showed non-significant change in the DO values, especially in the winter period. Therefore, it could be observed a difference between the calculated and experimental results at the later part of the experiment (Figure 3). The value used as the thermal coefficient (Arrhenius coefficient) in the simulation might not be accurate enough to incorporate the temperature effect during the lower temperature (10-15 °C) in winter period. In order to overcome this disparity between model predictions and the experimental values, it is amenable to quantify the Arrhenius coefficient at different temperatures for organic carbon degradation and nitrification.

In fluctuating water level systems (with and without plant), wastewater was drained every 3.5 days for 1 hour and allowed 1.5 days to be filled up the mesocosms. During this period, there was no outflow and no data on DO concentration. When comparing the model results of FWLC and FWLP (Figure 4), DO concentration of FWLP depicted more fluctuations than that of FWLC, due to the plant effect. Wetland plant roots impact on the porosity of media, resulting greater oxygen aeration and dissolution. Moreover, since the water level was altered at each fluctuation period, oxygen from air could be aerated into the substrate, influencing on oxygen release and uptake of plant roots, and finally impacting on oxygen concentration of pore water in bed media.

Evaluation on water level fluctuation impact on DO concentration

A distinctive difference in DO concentration between SWL and FWL conditions was encountered at the middle layers of each mesocosms (Table 3). During the fluctuating conditions, oxygen from air could be diffused into the substrate, after the draining phase. The penetrated oxygen was allowed to dissolve during the filling phase of wastewater inside pore space of substrate. It was not being existed in SWL conditions, because of the prevention of surface water layer and partial pressure in stratum (Brix et al.,1992).

Table 3. Comparison of mean DO concentrations at steady (SWL) and fluctuation (FWL) conditions

Depth (cm)	SWL (mgL^{-1})	FWL (mgL^{-1})	Increment %
0.0	7.25	7.63	5.2
-12.5	1.66	2.18	31.3
-25.0	1.53	2.00	30.7
-37.5	2.02	2.58	27.7
-50.0	2.82	3.02	7.1

In CW systems, oxygen concentration decreases due to the degradation of organic compounds, nitrification, consumption for chemical processes in sediment, and root respiration. In vertical flow conditions, oxygen concentration is increased with the flow direction. After a sufficient travel distance, the oxygen concentration may become depleted in soil water at a point where reaeration from external sources can begin to restore the DO in wetland background. This depletion and recovery phenomenon might have created the lowest DO concentration at the middle of the depth profile (Kadlec and Knight, 1996).

Effects of hydraulic characteristics on oxygen demand for nitrification and biodegradation

The oxygen demand for the nitrification and biodegradation was obtained at different scenarios with varying inflow DO levels (4.5 - 6.5 mgL^{-1}), inflow rates (0.06 - 0.12 Lh^{-1}) and fluctuation frequencies (2, 3.5 and 6 day). The variations in inflow rate and influent DO concentration within the above ranges did not show any substantial impact on the oxygen consumption due to nitrification and biodegradation processes. Therefore, the values of oxygen consumption due to nitrification and biodegradation were remained constant as 0.33 mgL^{-1} and 3.32 mgL^{-1} respectively at all conditions. However, the fluctuation has shown a higher consumption at the six-day interval, which might lead to improve the treatment capacity of CW (Table 4).

Table 4. Effect of water level fluctuation frequency on biochemical oxygen consumption (mgL^{-1})

Process	2 days	3.5 days	6 days
Nitrification	0.24	0.33	0.41
Decomposition	2.4	3.32	4.12

It would be worthwhile to emphasize here that the model has been formulated to simulate only the oxic conditions. When the fluctuation frequency decreases, the percentage time of the rising phase within a single cycle also was reduced. Consequently, the possibility for forming anoxic or anaerobic conditions was increased, but the model was itself not sufficient enough to evaluate this rather complex situation. Therefore, this model may not be accurate enough to evaluate the conditions beyond the lower fluctuation frequency than the studied period (6 days).

Sensitivity analysis

The results of the sensitivity analysis could reveal that the temperature [T] and the rate constant for organic carbon degradation [K_{org20}] are the most sensitive parameters among the eight parameters studied for the different four models (Table 5).

Table 5. Percentage variations of the most sensitive parameters in the analysis

Parameter	Variation (%)			
	(+10)	(-10)	(+50)	(-50)
Temperature [T]	4.7	-5.1	19.2	-30.0
Biodegradation coefficient [K_{org20}]	-3.0	3.2	-13.9	19.3

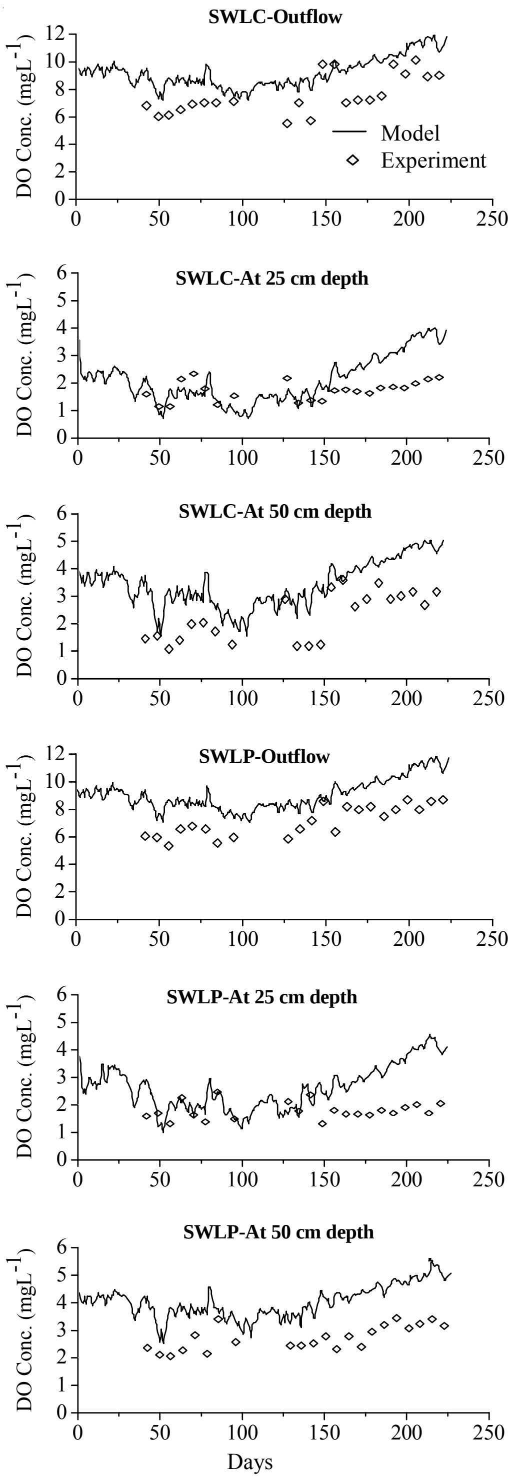


Figure 3. Variation of oxygen concentration in the mesocosms of static water level conditions with out plant-controlled (SWLC) and with plant (SWLP)

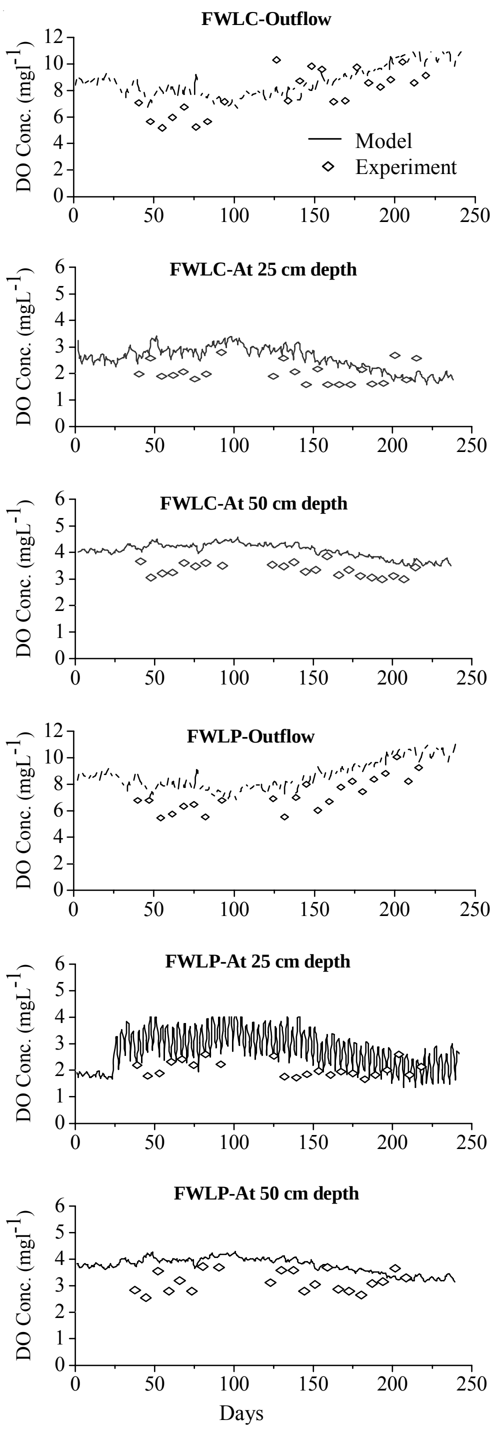


Figure 4. Variation of oxygen concentration in the mesocosms of fluctuating water level conditions with out plant-controlled (FWLC) and with plant (FWLP)

Due to the fact that the lesser possibility for occurring a relatively lrage experimental errors in temperature measurements, it could be presumed that the coefficient for biodegradation could be the most sensitive parameter with an error margin of 20% at the 1.5 times of its measured values.

Conclusions

The results of the modeling study revealed that the ammonium nitrogen and organic carbon removal efficiencies could be improved due to the exaggerated aeration activity with water level fluctuation in constructed wetlands. Consequently, the modeling approach could be an efficient technique on optimizing water level fluctuation systems in the design of well-functioning constructed wetlands.

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References

Aachib, M., Mbonimpa, M. and Aubertin, M. 2004. Measurement and prediction of the oxygen diffusion coefficient in unsaturated media, with applications to soil covers. **Water, Air and Soil Pollution**, 156: 163-193.

Armstrong, J. and Armstrong, W. 1990. Pathways and Mechanisms of Oxygen Transport in Phragmites australis. Constructed Wetlands in Water Pollution Control 11. **Pergamon Press**, Oxford.

Armstrong, J., Armstrong, W. and Beckett, P. M. 1992. Phragmites australis: Venturi- and humidity-induced pressure flows enhance rhizome aeration and rhizosphere oxidation. **New Phytologist**. 120: 197-207.

Armstrong, W., Brandle, R. and Jackson, M. B. 1994a. Mechanisms of flood tolerance in plants. **Acta Bot. Neerl**, 43: 307 - 358.

Auer, M. T. and Canale, R. P. 1982. Ecological studies and mathematical modeling of Cladophora in Lake Huron: III. The dependent of growth rates on internal phosphorus pool size. **Journal of Great Lakes Research**. 8: 93-99.

Brix, H., 1990. Gas exchange through the soil-atmosphere interface and through dead culms of Phragmites australis in a constructed reed bed receiving domestic sewage. **Water Resource.** 24; 259-266.

Brix, H., Sorrell, B. K. and Orr, P. T. 1992. Internal pressurization and convective gas flow in some emergent fresh water macrophytes. **Limnology and Oceanography**. 37: 1420 - 1433.

Brix H. 1993. Macrophyte-mediated oxygen transfer in wetlands: Transport mechanisms and rates. Constructed wetlands for water quality improvement. **CRC Press**, Inc. USA.

Broecker, W. S., Peng, T.H., Mathieu, G., Hesslein, R. H. and Torgersen, T. 1980. Gas exchange rate measurements in natural systems. **Radiocarbon.** 22: 676-683.

Chabbi, A., McKee, K. L. and Mendelssohn, I. A. 2000. Fate of oxygen losses from Typha domingensis (Typhaceae) and Cladium jamaicense (Cyperaceae) and consequences for root metabolism. **Am. J. Botany**. 87: 1081 - 1990.

Dawson, F. H., Westlake, D. F. and Williams, G. I. 1981. An automatic system to study the responses of respiration and photosynthesis by submerged macrophytes to environmental variables. **Hydrobiologia**. 77: 277-285.

Gelda, R. K. and Effler, S. W. 2002. Estimating oxygen exchange across the air-water interface of a hypereutrophic lake. **Hydrobiologia**. 487: 243-254.

Hillel, D., 1998. Environmental Soil Physics. **Academic Press**, California, USA

Jorgensen, S. E. and Bendoricchio, G. 2001. **Fundamentals of Ecological Modeling.** Elsevier Publishers.

Kadlec, R. H. and Knight, R. L. 1996. **Treatment of wetlands**. Lewis Publishers, Boca Raton, Florida.

McGechan, M. B., Moir, S. E., Castle, K. and Smit, I. P. J. 2005. Modeling oxygen transport in Reedbed-constructed wetland purification system for dilute effluents. **Biosystems Engineering.** 91 (2): 191 - 200.

Molder, E., Mashirin, A. and Tenno, T. 2005. Measurement of the oxygen mass transfer through air-water interface. **Environmental Science and Pollution Research**. 12 (2): 66-70.

Noorvee, A. E., P□ldvere and Mander, . 2007. The effect of pre-aeration on the purification processes in the long-term performance of a horizontal subsurface flow constructed wetland. **Science of the Total Environment**. 380: 229-236.

Ouellet-Plamondon, C., Chazaren, F., Comeau, Y. and Brisson, J. 2006. Artificial aeration to increase pollutant removal efficiency of constructed wetlands in cold climate. **Ecological engineering**. 2 7: 258-264

Pezeshki, S. R. 2001. Wetland plant responses to soil flooding. **Environmental and Experimental** Botany, 46: 299 - 312.

Reddy, K.R. and Patrick, W.H. 1984. Nitrogen transformation and loss in flood soil and sediments. **CRC Critical review in Environmental Control**. 13: 273-309.

Sorrell, B. K. and Dromgoole, F. I. 1989. Oxygen diffusion and dark respiration in aquatic macrophytes. **Plant Cell Environment**. 12: 293-299

Thomann, R. V. and Mueller, J. A. 1987. **Principles of Surface Water Quality Modeling and Control**. Harper & Row Publishers, NY.

Vymazal, J., Brix, H., Cooper, P. F., Haberl, R., Perfler, R. and Laber, J. 1998. Removal mechanisms and types of constructed wetlands. **Constructed Wetlands for Wastewater Treatment in Europe**. Leiden: Backhuys Publishers.

Wynn, T. M. and Liehr, S. K. 2001. Development of a constructed subsurface-flow wetland simulation model. **Ecological Engineering.** 16: 519 - 536.