

Oxygen transfer in marsh-pond-marsh constructed wetlands treating swine wastewater

KYOUNG S. RO¹, PATRICK G. HUNT¹, MELVIN H. JOHNSON¹, TERRY A. MATHENY¹,
DEAN FORBES² and GUDIGOPURAM B. REDDY²

¹USDA-ARS Coastal Plains Soil, Water & Plant Research Center, Florence, South Carolina, USA

²North Carolina A&T State University, Department of Natural Resources, Greensboro, North Carolina, US

Oxygen transfer efficiencies of various components of the marsh-pond-marsh (M-P-M) and marsh-floating bed-marsh (M-FB-M) wetlands treating swine wastewater were determined by performing oxygen mass balance around the wetlands. Biological oxygen demand (BOD) and total nitrogen (TN) loading and escaping rates from each wetland were used to calculate carbonaceous and nitrogenous oxygen demands. Ammonia emissions were measured using a wind tunnel. Oxygen transfer efficiencies of the aerated ponds were estimated by conducting the ASCE standard oxygen transfer test in a tank using the same aeration device. Covering pond water surface with the floating bed slightly decreased oxygen transfer efficiency. The diffused membrane aeration (26.7 kg O₂ ha⁻¹ d⁻¹) of M-P-M was surprisingly not as effective as plant aeration in the marsh (38.9 to 42.0 kg O₂ ha⁻¹ d⁻¹). This unusually low oxygen transfer efficiency of the diffused aeration was attributed to its low submergence depth of 0.8 m compared to typical depth of 4.5 m. The wetlands consisting entirely of marsh removed similar amounts of C and N without investing additional equipment and energy costs of aerating ponds in the middle of wetlands.

Keywords: Swine wastewater, ammonia emission, oxygen balance, constructed wetlands.

Introduction

Municipal and livestock wastewater can be treated cost effectively and passively with constructed wetlands in which physico-chemical and biological processes actively remove potential environmental contaminants such as suspended solids, oxygen demanding material, and nutrients.^[1,2] Instead of having continuous marsh wetlands, some wetland systems consists of a combination of marsh and aerated ponds to promote nitrification and denitrification.^[3–5] Poach et al.^[4] used marsh-pond-marsh (M-P-M) constructed wetlands to treat wastewater from swine anaerobic lagoons. These M-P-M removed 43, 41, and 44% of total suspended solids (TSS), chemical oxygen demand (COD), and total nitrogen (TN), respectively. The M-P-M consistently removed 37 to 51% of TN even under highly varying nitrogen loads ranging from 7 to 40 kg-N ha⁻¹ d⁻¹. However 23 to 36% of TN applied to the wetlands was removed via undesirable NH₃ volatilization process.^[5]

To reduce NH₃ volatilization rate from the aerated ponds, Reddy et al.^[6] covered the aerated ponds with a

floating cover made of recycled closed-cell foam on which bulrush was planted. These floating covers dramatically reduced NH₃ volatilization from the aerated ponds while maintaining N removal capacity of the marsh-floating bed-marsh (M-FB-M) similar to that of M-P-M. In addition, M-FB-M also promotes the use of recycled closed foam materials as a floating bed material. Unless recycled, these non-biodegradable closed foam wastes would exacerbate the limited land filling space for solid wastes disposal.

It was enigmatic how M-FB-M was able to maintain TN removal efficiency similar to M-P-M without emitting ammonia to the atmosphere. This could be possible if the aeration system under the floating bed introduced significantly more oxygen than open pond and/or the plants on the floating bed provided a significant amount of oxygen to stimulate nitrification process. In order to gain insight on the aeration efficiency of these wetlands, a series of oxygen transfer and ammonia volatilization experiments were conducted. The objectives of this study were to (i) independently determine oxygen transfer efficiencies of the aeration system in a tank simulating open- and covered-pond conditions and (ii) perform oxygen mass balance of these wetland systems treating swine anaerobic lagoon wastewater and estimate the plant aeration fluxes of the marsh and the floating bed.

Address correspondence to Kyoung S. Ro, USDA-ARS Coastal Plains Soil, Water & Plant Research Center, 2611 W. Lucas St. Florence, SC 29501; E-mail: Kyoung.Ro@ars.usda.gov
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Materials and methods

Constructed wetlands

The experiments were conducted at the swine research unit on the North Carolina A&T State University Farm using two M-P-M and two M-FB-M constructed wetlands. In addition to the four wetland cells, two more wetland cells were also operated at the site that consisted entirely of marsh (M). The performance of M was only used for assessing the accuracy of marsh aeration flux estimated from oxygen balance of M-P-M. Other than M, each wetland consisted of two marsh sections at both influent and effluent ends and a central pond either open or covered with floating wetlands (Fig. 1). The central ponds were continuously aerated at an airflow rate of 70 LPM using 4 EDI Flex Air™T-Series membrane diffusers (Environmental Fabrics Inc, Gaston, SC). The marsh sections were planted with broadleaf cattail (*Typha Latifolia* L.) and American bulrush [*Schoenoplectus americanus* (Pers.) Volkart ex Schinz and R. Keller]. For the M-FB-M, the central aerated pond was covered with a floating bed of recycled closed-cell foam compressed into plane forms and covered with non-woven fabric (Environmental Fabric Inc., Gaston, SC). Bulrush obtained from the marsh sections of wetland cells and the cuttings of Giant Bulrush (*Scirpus californicus*) were planted into a 15-cm peat soil layer placed on top of the floating bed. Detailed

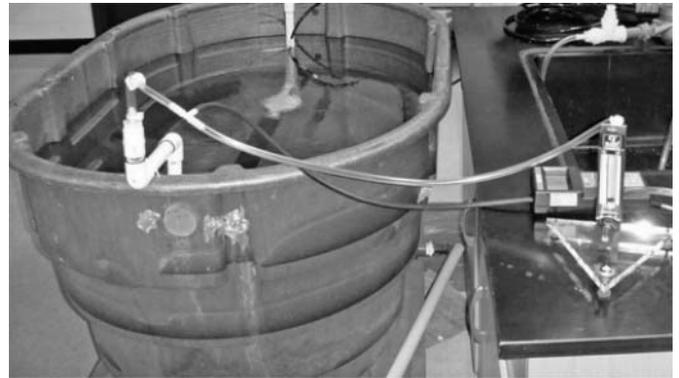


Fig. 2. Clean water aeration testing setup.

description of these constructed wetlands can be found in Reddy et al.^[3,6] and Poach et al.^[4]

Oxygen transfer experiments

Twenty six sets of clean water oxygen transfer tests were conducted in order to measure oxygen transfer efficiency of the EDI Flex Air™T-Series membrane diffuser. These tests were conducted in a tank measuring $1.4 \times 1.0 \times 0.6$ m (Fig. 2). The $6.3 \text{ cm} \times 0.6 \text{ m}$ membrane tube diffuser was mounted horizontally above the tank floor and was submerged 0.5 m. About 0.4 m^3 of distilled water was poured into the tank. The clean water tests were conducted in accordance with the ASCE Standard method^[7] using a multi-probe dissolved oxygen (DO) meter (556 MPS, YSI, Yellow Springs, Ohio). At each run, 10–30% excess theoretical oxygen demand of sodium sulfite (Na_2SO_3) was added to the tank as slurry. Cobalt chloride was added as a catalyst. After initial vigorous mixing of the water, dissolved oxygen (DO) concentration of the test water became zero or near zero. Three different aeration rates (i.e., 15, 23, and 30 LPM) were tested. As the aeration started, DO concentration increased with time. One-minute interval DO concentrations along with water temperature were monitored until the saturation DO concentration was achieved. These time-series DO concentration data were fitted to Equation 1, which was derived from oxygen mass balance of the tank water.

$$C = C_{\infty} - (C_{\infty} - C_0) \exp(-K_{La}t) \quad (1)$$

where

C = time-series DO concentration (kg m^{-3}),
 C_{∞} = saturation DO concentration (kg m^{-3}),
 C_0 = initial DO concentration (kg m^{-3}),
 K_{La} = overall volumetric mass transfer coefficient (hr^{-1}),
 t = time (hr).

As recommended by the ASCE Standard Method, three parameters C_0 , C_{∞} , and K_{La} were estimated simultaneously via non-linear regression analysis of time-series DO

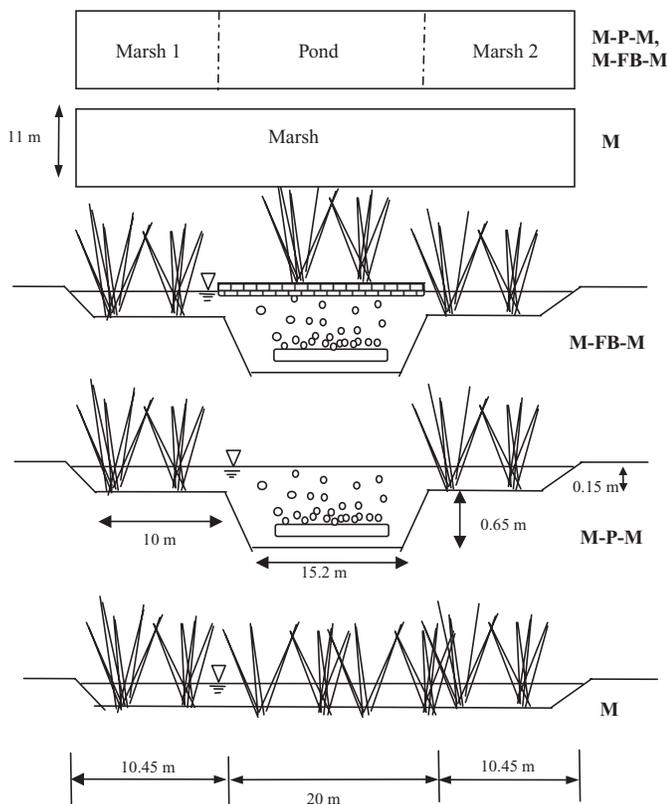


Fig. 1. M-P-M, M-FB-M, and M constructed wetlands treating anaerobic swine lagoon water.

concentration data using GeoPad Prism (GeoPad Software Inc., CA). The ASCE standard also defines several transfer parameters such as standard oxygen transfer rate (SOTR, $\text{kg O}_2 \text{ hr}^{-1}$) and standard oxygen transfer efficiency (SOTE, % O_2 absorbed per unit of O_2 supplied). These standard parameters can be used to estimate actual oxygen transfer rates in the field conditions. Standard conditions are defined as 20°C , 1 atm barometric pressure, and zero DO concentration. The values of K_{La} were normalized to 20°C using Equation 2.^[8]

$$K_{La,20} = K_{La,T}\theta^{20-T} \quad (2)$$

where

$K_{La,20} = K_{La}$ at 20°C (hr^{-1}),
 $K_{La,T} = K_{La}$ at T (hr^{-1}),
 $T =$ water temperature ($^\circ\text{C}$),
 $\theta =$ temperature coefficient (typically 1.024).

Standard oxygen transfer rate (SOTR, $\text{kg O}_2 \text{ hr}^{-1}$) represents the theoretically maximum amount of oxygen that can be transferred into 20°C water containing no dissolved oxygen. SOTR was calculated as:

$$SOTR = K_{La,20} C_{\infty,20} V \quad (3)$$

where

$C_{\infty,20} =$ saturation DO concentration at 20°C (kg m^{-3}),
 $V =$ volume of water (m^3)

Standard oxygen transfer efficiency (SOTE) was then calculated as:

$$SOTE = \frac{SOTR}{W_{O_2}} \times 100 \quad (4)$$

where

$W_{O_2} =$ mass rate of oxygen supplied to water ($\text{kg O}_2 \text{ hr}^{-1}$)

Results and discussion

BOD₅, TN removals and NH₃ emission

During the two field campaigns in May and June 2005, a total of 16 NH_3 volatilization measurements were made with the wind tunnel from the pond sections of M-P-M and M-FB-M. As shown in Table 1, NH_3 volatilization rate from the central pond section was dramatically reduced from 3.1 ± 4.2 to $0.3 \pm 0.3 \text{ kg NH}_3\text{-N ha}^{-1}\text{d}^{-1}$ for M-P-M and M-FB-M, respectively. During the same period, M-FB-M removed 96% and 93%, M-P-M removed 92% and 97%, and M removed 92% and 94% of incoming BOD_5 and TN, respectively. These BOD_5 , TN, and NH_3 volatilization rates were used to perform oxygen balance of the wetlands.

Table 1. BOD_5 , TN, and NH_3 volatilization rates during May and June, 2005.

		BOD_5 ($\text{kg ha}^{-1}\text{d}^{-1}$)	TN ($\text{kg-N ha}^{-1}\text{d}^{-1}$)
M-P-M	In	6.3 ± 2.4	9.1 ± 2.3
	Out	0.5 ± 0.6	0.3 ± 0.4
	Pond NH_3 Volatilization Rate ($\text{kg-N ha}^{-1}\text{d}^{-1}$)	3.1 ± 4.2	
M-FB-M	In	6.5 ± 2.4	9.3 ± 2.3
	Out	0.3 ± 0.2	0.7 ± 0.5
	Pond NH_3 Volatilization Rate ($\text{kg-N ha}^{-1}\text{d}^{-1}$)	0.3 ± 0.3	
M	In	6.6 ± 2.5	9.4 ± 2.4
	Out	0.5 ± 0.5	0.6 ± 0.5

Oxygen transfer by diffused aeration

As shown in Figure 3, the DO concentration profile data during aeration tests fitted well with Equation 1; most R^2 's were >0.99 . Once K_{La} and C_{∞} were determined from each aeration test run, SOTR and SOTE were calculated using Equations 2, 3, and 4. Standard oxygen transfer efficiency (SOTE) of the EDI Flex AirTMT-Series membrane diffuser was determined at air flow rates of 15, 23, and 30 L min^{-1} . The air flow rate did not significantly influence the values of SOTE. However, covering the water surface with the FB material decreased the value of SOTE from $3.8 \pm 0.8\%$ for open water to $2.9 \pm 0.4\%$ for covered water ($p = 0.0005$) as shown in Figure 4. These values are much smaller than typical SOTEs of 15–19% for membrane diffusers^[8]. However, these typical diffusers' SOTEs are tested at a water depth of 4.6 m, while our tank water depth was only 0.5 m. The manufacturer's performance curve shows SOTE value of 26% at 4.6 m submergence and 70 LPM. The lowest depth shown on the performance curve is 1.5 m and the SOTE for 70 LPM is 10%. Because of this low submergence depth, the membrane diffuser would not be efficient in transferring oxygen from the aeration. In determining actual oxygen

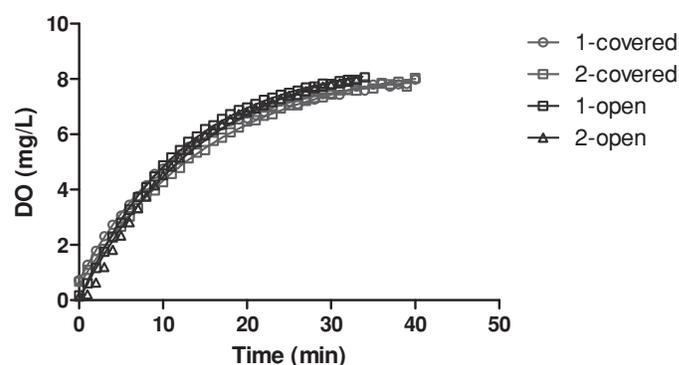


Fig. 3. DO profiles during aeration test.

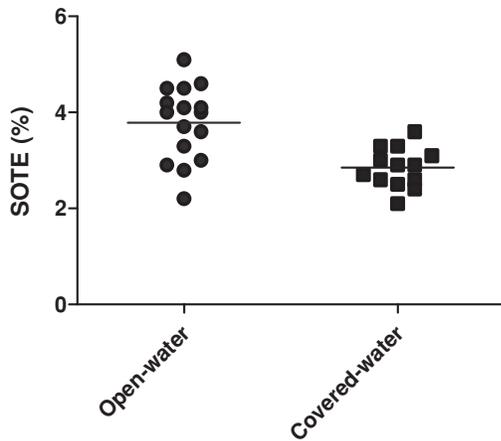


Fig. 4. SOTE of the EDI Flex AirTMT-Series membrane diffuser conducted in open water and covered water.

transfer rate in the pond with submergence depth of about 0.8m, we assumed the same SOTE determined from clean water tank depth of 0.5 m.

Using Equation 4 with SOTE of 3.8 or 2.9% and the mass rate of 27.8 kg O₂ d⁻¹ (i.e., W_{O₂}) delivered by the field aeration rate of 70 LPM, SOTR values were calculated; SOTR values were 1.1 and 0.8 kg O₂ d⁻¹ for open water and covered water, respectively. These SOTR values represent the theoretical maximum oxygen transfer rate at zero DO concentration at standard condition. The actual amount of oxygen transferred into water depends on DO concentration, temperature, mixing intensity, and tank geometry. The actual oxygen transfer rate under field conditions can be estimated by^[8]:

$$AOTR = SOTR \left(\frac{\beta C_{\infty, T} - C}{C_{\infty, 20}} \right) (1.024^{T-20}) \alpha \quad (5)$$

where

AOTR = actual oxygen transfer rate (kg O₂ d⁻¹),

C_{∞, T} = clean water saturation DO concentration at field T (kg O₂ m⁻³),

C = operating field DO concentration (kg O₂ m⁻³),

$\alpha = K_{La}$ correction factor = $\frac{K_{La, wastewater}}{K_{La, cleanwater}}$, typically 0.4 to 0.8,

$\beta = C_{\infty}$ correction factor, typically 0.95 to 0.98.

The average water temperature during the period of NH₃ volatilization study was 24°C. Average DO concentrations for the pond sections of M-P-M and M-FB-M were 2.4 and 0.9 g m⁻³, respectively. The alpha factor generally decreases with increase in water depth above a submerged diffuser; SOTE increases with water depth^[9]. Alpha factor for their lowest water depth of 1.5 m ranged from ca. 0.65 to 0.9. Whereas the water depth of about 0.8 m above the EDI Flex AirTMT-Series membrane diffuser in the ponds of both M-P-M and M-FB-M was lower than 1.5 m, the maximum value (i.e., 0.8) of typical alpha factor range (i.e.,

0.4 to 0.8) was used to estimate actual oxygen transfer rate in those ponds. A mid value of 0.97 for beta was also used to calculate the actual oxygen transfer rate. Using above conditions, the actual oxygen transfer rate delivered by aerating the open pond of M-P-M (0.59 kg O₂ d⁻¹ or 26.7 kg O₂ ha⁻¹ d⁻¹) was slightly more than that of M-FB-M (0.57 kg O₂ d⁻¹ or 25.8 kg O₂ ha⁻¹ d⁻¹). In both systems, diffused aeration transferred only about 2% of 27.8 kg O₂/d pumped into the pond water. This result also showed that the enhanced N removal from the M-FB-M could not be attributed to its enhanced aeration efficiency. In order to analyze aeration efficiencies of each component of these wetlands, we performed oxygen mass balance around the wetlands.

Oxygen balance of wetlands

Assuming a pseudo-steady state was achieved, the oxygen mass balances around the two wetland systems were performed as:

$$\begin{aligned} UOD_{MPM} + NOD_{MPM} &= AOTR_{MPM} + M_{O_2, surf} \\ &+ M_{O_2, marsh} \text{ for } M - P - M \end{aligned} \quad (6)$$

$$\begin{aligned} UOD_{MFBM} + NOD_{MFBM} &= AOTR_{MFBM} + M_{O_2, FB} \\ &+ M_{O_2, marsh} \text{ for } M - FB - M \end{aligned} \quad (7)$$

where

UOD_{MPM, MFBM} = ultimate carbonaceous oxygen demands of M-P-M or

M-FB-M (kg O₂ d⁻¹)

NOD_{MPM, MFBM} = nitrogenous oxygen demands of M-P-M or M-FB-M (kg O₂ d⁻¹)

AOTR_{MPM, MFBM} = actual O₂ transfer rates of M-P-M or M-FB-M (kg O₂ d⁻¹)

M_{O_{2, marsh}} = O₂ transferred into the two marsh sections of M-P-M or M-FB-M (kg O₂ d⁻¹)

M_{O_{2, FB}} = O₂ transferred through the floating bed of M-FB-B (kg O₂ d⁻¹)

M_{O_{2, surf}} = surficial oxygen transfer rate (kg O₂ d⁻¹)

Equations 6 and 7 neglected the difference in mass rates of oxygen into and out of wetland systems via water flow because its magnitude would be less than 1% of diffused aeration.

Surficial oxygen transfer

Oxygen transferred through the water-air interface of the open ponds of M-P-M was estimated using the new unified transfer coefficient equation^[10]. This equation was developed from the literature database published for the last 50 years. Using the new unified equation, the amount of surficial oxygen transferred into the open-pond water surface

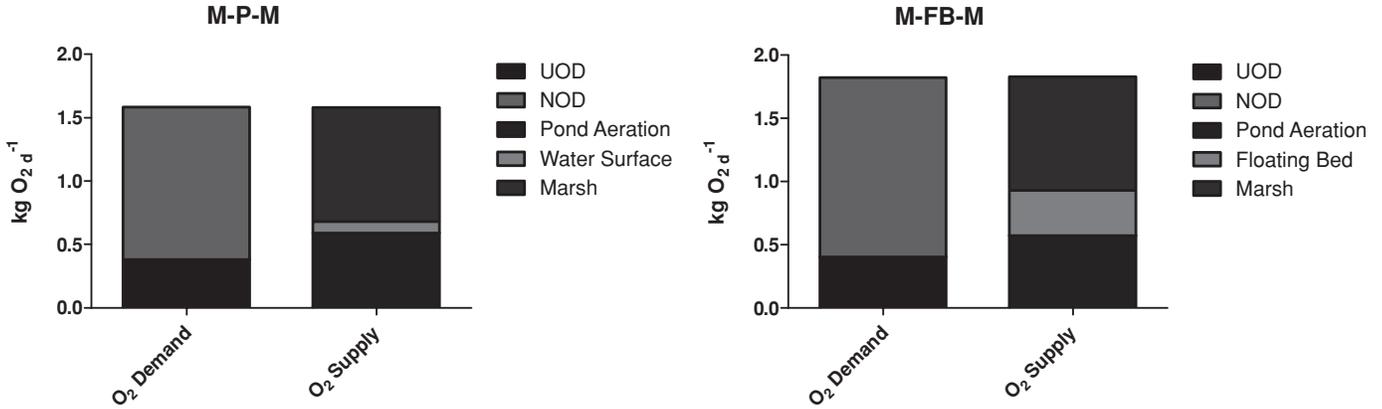


Fig. 5. Oxygen requirements and supply components of wetland systems.

was estimated as:

$$M_{O_2, surf} = 86,400 \cdot A \left[170.6 \cdot Sc^{-1/2} U_{10}^{1.81} \left(\frac{\rho_a}{\rho_w} \right)^{1/2} \right] \cdot 2.78 \times 10^{-6} (C_{\infty, T} - C) \quad (8)$$

Where

- A = pond water surface area of M-P-M (m^2)
- Sc = Schumidt number for oxygen (ν/D)
- ρ_a = air density ($kg\ m^{-3}$)
- ρ_w = water density ($kg\ m^{-3}$)
- U_{10} = wind speed at a reference height of 10 m ($m\ s^{-1}$)
- ν = kinematic viscosity of water ($m^2\ s^{-1}$)
- D = molecular diffusivity of oxygen ($m^2\ s^{-1}$)

Using Equation 8 with the average 10-m wind speed of 1.0 m/s at the study site and field conditions during the study period, the surficial oxygen transfer into the M-P-M pond was estimated to be 0.09 $kg\ O_2\ d^{-1}$ (or 2.1 $kg\ O_2\ ha^{-1}\ d^{-1}$). Compared to the oxygen transferred by the diffused aeration of the M-P-M, the surficial oxygen transfer rate represents only about 8% of the diffused aeration oxygen transfer rate.

Carbonaceous oxygen demand

The ultimate oxygen demand for the wetlands came from oxidizing carbonaceous material. The carbonaceous oxygen demand was estimated from the BOD_5 removal rates. It is commonly assumed that BOD_5 or 5-day BOD of wastewater represents about 68% of the ultimate carbonaceous oxygen demand.^[8]

$$UOD = (M_{BOD_5, in} - M_{BOD_5, out}) / 0.68 \quad (9)$$

Where

- $M_{BOD_5, in}$ = BOD_5 loading rate ($kg\ O_2\ d^{-1}$),
- $M_{BOD_5, out}$ = BOD_5 escaping rate ($kg\ O_2\ d^{-1}$).

The carbonaceous oxygen demands during the study period were 0.38 and 0.40 $kg\ O_2\ d^{-1}$ for M-P-M and M-FB-M, respectively.

Nitrogenous oxygen demand

The nitrogenous oxygen demands were estimated by using the known stoichiometric relationship of biochemical N pathway responsible for biological N removal. The biological N removal pathway in wastewater treatment plants and lagoons involves a two-step process, nitrification and denitrification.^[8,11] The oxygen requirement for removing 1 kg of TN from this pathway is about 4.2 $kg\ O_2$. Assuming all TN removed from wetland cells other than volatilized portions had been subjected to this nitrification-denitrification, the oxygen requirement for TN removal is estimated as:

$$NOD = 4.2 (M_{TN, in} - M_{TN, out} - M_{NH_3, vol}) \quad (10)$$

where

- $M_{TN, in}$ = TN loading rate ($kg-N\ d^{-1}$),
- $M_{TN, out}$ = TN escaping rate ($kg-N\ d^{-1}$),
- $M_{NH_3, vol}$ = NH_3 volatilization rate ($kg-N\ d^{-1}$).

The nitrogenous oxygen demands during the study period were 1.2 and 1.4 $kg\ O_2\ d^{-1}$ for M-P-M and M-FM-M, respectively. It is interesting that about 3 times more oxygen were required to removed N than organic C in these wetlands.

Oxygen transfer into marsh sections

Whereas we did not measure the ammonia volatilization rates from the marsh sections of the wetlands during the study period, the NH_3 volatilization correlation developed by Poach et al.^[5] was used. Based on TN loading rate, the NH_3 volatilization rate from the two marsh sections was estimated to be 0.03 $kg-N\ d^{-1}$. This NH_3 volatilization rate was about one half of that from open pond of M-P-M, but three times that from the covered pond of M-FB-M.

Using Equation 6, $M_{O_2,marsh}$ was then estimated based on oxygen balance around M-P-M. The value of oxygen flux into marsh was $38.9 \text{ kg O}_2 \text{ ha}^{-1} \text{ d}^{-1}$ (or $3.9 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$). This value of plant aeration flux was comparable to those of subsurface gravel bed wetlands with cattails and bulrush reported in Kadlec and Knight^[2]. In order to assess the accuracy of the plant aeration flux of M-P-M marsh sections, the plant aeration flux of the two M wetland cells during the same period were calculated in similar manner. The M provided $42.0 \text{ kg O}_2 \text{ ha}^{-1} \text{ d}^{-1}$, which was close to that estimated from M-P-M O_2 balance (i.e., $38.9 \text{ kg O}_2 \text{ ha}^{-1} \text{ d}^{-1}$). It was surprising that the marsh transferred more oxygen than the membrane diffused aeration per unit surface area. This low efficiency of diffused aeration resulted from the fact that the submerged depth of 0.8 m was too shallow for adequate contact time for oxygen transfer to occur while bubbles rose up the water column.

Oxygen transfer through floating bed

Assuming the same marsh O_2 transfer rate for the marsh sections of M-FB-M, the oxygen transfer through the float bed ($M_{O_2,FB}$) was estimated from Equation 7. In order to remove TN biologically with a small NH_3 volatilization rate, $0.36 \text{ kg O}_2 \text{ d}^{-1}$ (or $16.6 \text{ kg O}_2 \text{ ha}^{-1} \text{ d}^{-1}$) in addition to $0.57 \text{ kg O}_2 \text{ d}^{-1}$ from the diffused aeration must be transferred into the M-FB-M through the floating bed as shown in Figure 5. It is about one half of that transferred by marsh sections per unit area.

Conclusions

Oxygen transfer efficiencies of different components of M-P-M and M-FB-M were analyzed by performing oxygen mass balance around the wetlands. Separate aeration tests using the same diffused membrane aerator in a clean water tank showed that the efficiency of the diffused aeration would be 5 or 6 time less than that reported in the literature due to its shallow submergence depth. The theoretical maximum transfer efficiency at 20°C (i.e., SOTE) would be only about 3%, compared 15–20% for deeper submergence depth. Covering water surface with the floating bed

slightly decreased oxygen transfer efficiency. Plant aeration flux in the marsh sections of the wetland was actually higher than that of the diffused membrane aeration. It is therefore concluded that the aeration of the pond sections was not as effective as marsh. The marsh aeration flux calculated from M was comparable to that obtained from M-P-M. The aerated open-pond was not effective in enhancing BOD and TN removal capacities compared to M-FB-M or M, while it promoted unwanted NH_3 volatilization. The wetland cells consisting entirely of marsh were as effective as M-FB-M or M-P-M with small NH_3 volatilization.

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