COMPARATIVE STUDIES OF VEGETATED AND NON–VEGETATED SUBMERGED–FLOW WETLANDS TREATING PRIMARY LAGOON EFFLUENT

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ABSTRACT. The submerged–flow (SF) wetland concept offers high organics and solids removal at relatively low cost for construction, operation, and maintenance. In this comparative study, vegetated and non–vegetated SF wetlands were investigated for their ability to treat primary lagoon effluent. The experimental design was comprised of three vegetated and three non–vegetated SF wetland beds (3 m x 1 m x 0.5 m) operated in a semi–continuous–flow mode, which were fed every 12 hours with a design 5–day residence time. The wetlands were packed with 19–mm diameter trap rock and planted with bulrush (Scirpus validus). The BOD5 and TSS removals throughout the two–year monitoring period were high (up to 98%) for both vegetated and non–vegetated wetlands and tended to follow seasonal variations. Vegetation slightly enhanced the reduction of biodegradable organic matter and suspended solids within the SF environment, although this enhancement was statistically insignificant (P > 0.05). The annual average mass removal for ammonia nitrogen in vegetated wetland beds was 3.3 kg ha–1 d–1 (up to 95%). The nitrification process in vegetated wetland beds was significantly more pronounced (P < 0.05) than non–vegetated beds. The lack of measurable dissolved oxygen in the non–vegetated wetlands likely restricted the nitrification process. The dissolved phosphorus reduction varied from month to month depending on seasonal variations of plant growth, ranging from 27% to 100% in vegetated wetland beds and from no removal to 66% in non–vegetated beds. The removal of dissolved phosphorus in vegetated beds was significantly higher than in non–vegetated wetland beds (P < 0.05). Based on the results of this study, vegetation significantly contributed to the reduction of nutrients, specifically ammonia nitrogen and dissolved phosphorus, from SF wetlands, but not for BOD5 and TSS.

Keywords. Biokinetics, Nitrification, Removal efficiency, Submerged–flow wetlands, Wetland hydraulics.

Constructed wetlands (surface and submerged–flow) are natural wastewater treatment systems that offer cost–effective treatment potential for small to moderately sized communities where suitable land is available (Reed et al., 1995). Submerged–flow wetlands (SF) are comparatively shallow (typically less than 0.6 m) constructed gravel beds vegetated with dense stands of water–tolerant plants such as cattails and bulrushes. Wastewater is allowed to flow through the length of the gravel beds, where the treatment is achieved. SF wetlands are effective as wastewater treatment processes for a number of reasons. Bacterial growth attached to the wetland gravel media and submerged roots is essential for soluble and colloidal biochemical oxygen demand (BOD) reduction and other biologically driven processes. The quiescent water conditions found in the SF media are conducive to removal of solids constituents from a wastewater. Based on the results from numerous research and demonstration SF projects using domestic wastewater, it appears that this process is capable of producing effluents with 5–day biochemical oxygen demand (BOD5), total suspended solids (TSS), and total nitrogen (TKN–N) concentrations as low as 5, 5, and 1 mg L–1, respectively. The removal of other wastewater contaminants such as phosphorus, heavy metals, refractory organics, and pathogens is dependant on site and wastewater–specific factors (Tchobanoglous and Schroeder, 1985).

With respect to system performance, SF wetland systems appear to be able to remove BOD5 at a rate of 110 kg ha–1 d–1 or more during the warmer seasons and when the amount of bacterial support structure provided by aquatic plants in the form of roots and stems is greatest (Tchobanoglous and Schroeder, 1985). During the cooler seasons, the BOD removal performance of SF wetland systems decreases because of the slowing of the metabolic activity of both the bacteria and the plants that support and in part sustain the bacterial growth. The removal of settleable organics is rapid due to the deposition and filtration, and close to 50% of the applied BOD is removed within the first third of the bed (Reed et al., 1995). The remaining BOD, in colloidal and dissolved forms, is reduced through microbial processes of the biofilm in the remaining two thirds of the bed. The BOD removal processes in the SF media are thought to be aerobic only near the surface and at microsites around the roots, and the remainder of the system is anaerobic decomposition (Tchobanoglous and Schroeder, 1985). Many of the biochemical reactions responsible for the reduction of organic matter in SF wetland systems appear to follow the plug–flow,
first–order kinetics of attached–growth biological reactors (Burgoon et al., 1995; Reed et al., 1995; WPCF, 1990; Watson et al., 1989; US EPA, 1988, 1993). Reed and Brown (1995) found that BOD removal was very effective in SF systems with relatively short hydraulic retention times (HRT) and that BOD removal exhibited a linear relationship with organic loading up to 140 kg ha$^{-1}$ d$^{-1}$.

Since the BOD removal is more efficient in aerobic systems, the effectiveness of SF wetland systems may be limited. In vegetated beds, plants are a primary source of oxygen for aerobic microbes (Moorhead and Reddy, 1988). The oxygen transport rate varies greatly for different aquatic plants, ranging from 1.2 to 3.49 g O$_2$ (kg dry root mass)$^{-1}$ h$^{-1}$. Steinberg and Comrood (1994) monitored the root zone oxidation state over a period of 87 days for alpine rush (Juncus alpinus), canarygrass (Phalaris arundinacea), and cattail (Typha latifolia) growing in gravel–nutrient solution cultures. The dissolved oxygen (DO) concentration in the root zone of cattail and canarygrass was <1 mg L$^{-1}$, while in alpine rush it ranged from 0 to 2 mg L$^{-1}$. They concluded that there was a plant species effect on the oxidation state of the root zone. The diffusion of DO from the root system to the surrounding root zone creates aerobic microsites that support aerobic bacteria and processes (Brix, 1993, 1997).

A major function performed by wetland systems is the reduction of suspended solids. These reductions are the result of a complicated set of internal processes, including the production of transportable solids by the wetland biota. Removal of TSS in SF wetland systems is effective and rapid. As evaluated by Kadlec and Knight (1996), regardless of temperatures, the effluent levels of TSS from different SF wetlands consistently remained below 20 mg L$^{-1}$ even though the influent TSS concentrations varied from 75 to 250 mg L$^{-1}$. Based on the studies of operational SF wetlands, Reed et al. (1995) reported that essentially all the solids removal occurred in the initial 10% to 20% of the bed length. The deposition of solids within the SF wetland media produces the potential for bed clogging, especially near the inlet end of the system (Kadlec and Knight, 1996; Watson et al., 1989). This can lead to a reduction in hydraulic conductivity and surface overflow, which affects the performance of SF wetlands.

Nitrogen removal can be very effective in constructed SF wetlands, up to 45 kg N ha$^{-1}$ d$^{-1}$, especially where systems experience consistently warmer temperatures (Reed et al., 1995). Nitrogen in the SF environment exists in inorganic (ammonia, nitrate, nitrite, and nitrogen gas) and organic (urea, amino acids, etc.) forms. The primary mechanisms of nitrogen removal in SF wetlands are ammonification (biological breakdown of organic nitrogen compound into ammonia), nitrification, denitrification, plant uptake, and volatilization (Sievers, 1997; Kadlec and Knight, 1996). The nitrification process involves the microbially driven aerobic conversion of ammonia to nitrate (Kadlec and Knight, 1996; Metcalf and Eddy, 1991). The denitrification process biologically transforms nitrate into nitrogen gas in anoxic conditions in the presence of organic matter. The volatilization process is characterized by the stripping of the ammonia at higher pH (generally >7.5). In the SF environment, carbonaceous organics are metabolized faster than nitrogenous compounds, which can easily deplete oxygen. The oxygen transfer into the substrate is limited due to the reduced interface with the atmosphere. Hence, plants are the primary source of oxygen. The lack of oxygen within the substrate markedly reduces the removal of ammonia from the wastewater (Kadlec and Knight, 1996).

Phosphorus removal in SF wetlands occurs from adsorption, absorption, complexation, microbial and plant uptake, and precipitation (Kadlec and Knight, 1996; Watson et al., 1989). An SF wetland system may show high phosphorus removal during the first year or two of operation due to adsorption by active contact sites of the media and plant uptake during the early vigorous growth and expansion of vegetation cover; however, removal efficiencies generally decrease with time (Reed et al., 1995). Plant uptake of phosphorus by bulrush in SF wetlands has been found to range from 0.14 to 0.27 kg P ha$^{-1}$ d$^{-1}$ (Kadlec and Knight, 1996). In submerged–flow wetland systems, phosphorus interconverts among soluble, precipitate, and complexed forms, depending on environmental conditions.

In order for SF wetland systems to function properly, both hydraulic and biokinetic requirements should be satisfied. Most cases of overland flow are the result of improper hydraulic design (Reed and Brown, 1995). Entry of solids may extend only 10% to 20% of the wetland length yet may dominate the hydraulic regime of the system (Watson et al., 1989). This may necessitate system designs with lower length/width ratios. Removal rates for SF wetlands are reported to be higher than for surface–flow wetlands because the plant–rooting media in the former provides more solid entrapment sites and specific surface area for biofilm (Reed et al., 1995). Inadequate treatment of domestic wastewater using conventional treatment technologies, such as adsorption fields and lagoons in small, non–sewered communities, has contributed to non–point–source pollution of surface waters and contaminated ground water supplies (Kadlec and Knight, 1996; Choate et al., 1993; Steiner and Combs, 1993). These conventional on–site wastewater treatment systems require sufficient land for effective removal results. A small constructed wetland system is believed to be one treatment alternative for locations where conventional septic systems are ineffective due to poor soil percolation, high groundwater, or karst geology.

The specific objectives of this study were: (1) to assess the comparative performance of vegetated and non–vegetated SF wetlands to treat septic tank effluent, (2) to evaluate the role of plants in the performance enhancement, and (3) to examine the influence of length/width ratio on treatment processes.

**Materials and Methods**

The experimental setup consisted of three vegetated and three non–vegetated rectangular galvanized steel beds, each 3 m long, 1 m wide, and 0.5 m deep and packed with 1.9 cm diameter trap rock. The average drainable porosity of the beds was 38% (figs. 1a, 1b, and 1c). Bulrush (Scirpus validus) was selected as the rooting plant in the vegetated beds because of its ability to root relatively deeply in the rock media. The wetland beds were installed 25 cm above the ground to facilitate hydraulic head measurement. Each wetland bed was fitted with a 4–cm perforated PVC distribution lateral at the inlet end and a collection lateral at the outlet end, placed 5 cm above the bottom of the bed (fig. 1c). All wetland beds were fed from the bottom section...
of the gravel media. The SF wetland system was operated in a semi–continuous–flow mode, which was fed every 12 hours and designed to maintain a system retention time of 5 days.

The SF wetland beds were installed at the Lake Capri wastewater lagoon system near Columbia, Missouri, which serves a subdivision with 50 homes. Each house is equipped with a septic tank, and its effluent is discharged into a network of pipes leading to the treatment lagoon. The wetland beds were fed with septic tank effluent drawn in equal amounts from the primary lagoon cell and sewer manhole. The composite wastewater had an average waste strength of 140 mg L$^{-1}$ BOD$_5$. The hydraulic loading rate applied to each wetland bed was 2.25 cm d$^{-1}$. These rates on average were lower than typical because of water fluctuations that occurred in the septic tank effluent discharge and in the lagoon cells, which caused dilution of the wastewater.

The wastewater was pumped to an overhead tank mounted at 2 m height and uniformly distributed to each wetland bed through a network of manifolds (4 cm diameter PVC pipes) and supply pipes (2.5 cm diameter PVC pipes). Manifold valves were used to calibrate equal flow distribution. The beds were housed in a greenhouse with adequate heating to prevent complete freezing of the beds and to eliminate dilution from rainfall. The ends of the greenhouse were opened from April to November, and shade cloth covered the unit during most of those months. For heating, two finned
where described by (Watson et al., 1989):

Before weighing. For the analysis of kinetic parameters in SF/C0083/C0083 on the standard methods (APHA, 1992):

Laboratory for the following water quality parameters based were sampled monthly. Water samples were analyzed in the laboratory for the following water quality parameters based on the standard methods (APHA, 1992):

- 5-day biochemical oxygen demand (BOD5) using the 5-day incubation method as described in Method 5210.
- Total and volatile solids (TS, TVS) and total and volatile suspended solids (TSS, TVSS) using gravimetric procedures as described in Method 2540.
- Ammonia (NH3–N) using the ammonia–selective electrode method with known addition.
- Nitrate (NO3–N), nitrite (NO2–N), and dissolved phosphorus (PO4–P) using the ionic chromatography procedure as described in Method 4500. Total Kjeldahl nitrogen, organic nitrogen, and total phosphorus were not measured in this study.
- Dissolved oxygen (DO), pH, and electrical conductivity (EC) using selective electrode methods.

For the analysis of horizontal pollutant removal, water samples were taken using a hand vacuum pump from the longitudinal sampling ports (transects), spaced at approximately 0.6 m intervals along the bed length. The sampling ports consisted of wells enclosed with 5–mm wire mesh retaining walls and fitted with a removable cylindrical media cage (12.5 cm diameter and 30 cm height) used for studying solids depositions within the wetland media (fig. 1d). Dissolved oxygen profile was sampled in October 1999 and June 2000 for each bed by collecting water samples from the horizontal sampling ports at two different levels (upper and lower sections).

Following plant dormancy, all wetland vegetation was harvested and dry biomass measured from ten equally spaced sections along the bed length to determine spatial distribution of plant growth. Plants were cut just above the rock surface and placed in convective drying ovens at 65°C for two days before weighing. For the analysis of kinetic parameters in SF wetlands, the following first–order equation was used (Kadlec and Knight, 1996):

\[
\ln \left( \frac{C_o}{C_t} \right) = -k_T t
\]

\[
t = \frac{nh}{q}
\]

where

- \(C_o\) = influent concentration (mg L\(^{-1}\))
- \(C_t\) = effluent concentration (mg L\(^{-1}\))
- \(k_T\) = volumetric reaction rate constant (d\(^{-1}\))
- \(h\) = wetland depth (m)
- \(t\) = hydraulic detention time (days)
- \(q\) = hydraulic loading rate (m s\(^{-1}\)).

The reaction rate constant is temperature dependent and described by (Watson et al., 1989):

\[
k_T = k_{20} (1.06)^{T-20}
\]

where \(k_{20}\) = reaction rate constant at 20°C, and \(T\) = temperature in °C.

Analysis of variance (ANOVA) was used to test the statistical significance (5% significance level), as quantified by probability (P–value), of the null hypothesis that the means of two or more samples of vegetated and non–vegetated wetlands are equal.

**RESULTS AND DISCUSSION**

The results of the SF wetland system performance are presented for 1998 and 1999. The monthly values are plotted against time for these system parameters.

**BOD5 REMOVAL**

The BOD5 removal rates throughout the two–year monitoring period were high for both vegetated and non–vegetated wetlands and tended to follow seasonal variations, e.g., higher removals in the summer and fall, and lower removals in the winter and early spring (fig. 2). Construction of this system was completed by July 1997. Monitoring began in 1998, after plants had developed some initial vegetative growth. The annual average influent BOD5 concentrations for 1998 and 1999 were 126 and 141 mg L\(^{-1}\), corresponding to average loadings of 27 and 30 kg ha\(^{-1}\) d\(^{-1}\) for 1998 and 1999, respectively. The average effluent BOD5 values from vegetated and non–vegetated beds for the same years were 22 and 20 mg L\(^{-1}\), and 28 and 26 mg L\(^{-1}\), respectively, which are below the Missouri standard average of 30 mg L\(^{-1}\).

Since the BOD5 removal process is dependent on temperature, removal efficiency rates appeared to follow seasonal variations (fig. 3). These rates were generally lower for mass removals in non–vegetated wetlands than in vegetated wetland beds. As seen in figure 3, in 1998 the mass BOD5 removal efficiency rates varied from 44% (January) to 98% (September) for vegetated beds and from 43% (January) to 92% (September) for non–vegetated beds. In 1999, however, those rates ranged from 66% (February) to 97% (September) for vegetated beds and from 57% (January) to 96% (August) for non–vegetated beds. Sampling results indicated that the vegetated beds achieved slightly better removal rates, and the rates were enhanced over time as the beds matured. However, the effect of vegetation was statistically insignificant (P > 0.05).

The transect results indicated a rapid horizontal declining trend of BOD5 over the length of the bed for both vegetated and non–vegetated wetlands (table 1). It was observed that more than 65% of biodegradable organic content was removed within the first one–third of the bed length of both the vegetated and non–vegetated wetlands. The longitudinal declining trend in the BOD5 concentrations was very similar in both the vegetated and non–vegetated beds. This would suggest that plants have little impact on the overall BOD5 removal efficiency of an SF wetland system.

Experimental results indicated that the BOD5 removal rates varied seasonally. These rates were generally lower for mass removals in non–vegetated wetlands than in vegetated wetland beds. In 1998, the minimum reaction rates were calculated for January (0.12 d\(^{-1}\) for vegetated beds and 0.11 d\(^{-1}\) for non–vegetated beds), while the maximum rate occurred in August for vegetated beds (0.83 d\(^{-1}\) and in
September for non–vegetated wetland beds (0.51 d⁻¹). In the second year of operation (1999), the lowest BOD₅ removal rate in vegetated beds was 0.25 d⁻¹ (February), while the highest value was 0.70 d⁻¹ (September). The average reaction rates improved as the wetlands matured. Based on 1999 results, the yearly average rates were 0.65 and 0.55 d⁻¹ for vegetative and non–vegetative beds, respectively.

Longitudinal dissolved oxygen (DO) measurements were made in October 1999 (with nearly dormant plants) and in June 2000 (with active plant growth) after more than two years of plant growth. The average water temperatures of the wetlands were 12°C and 23°C during the October and June sampling events, respectively. The measurements indicated that dissolved oxygen levels within the subsurface for both
the vegetated and non–vegetated wetland beds were low and varied from 0.1 to 0.5 mg L⁻¹. The influent DO was measured at 3.3 mg L⁻¹ in the October sampling and at 1.75 mg L⁻¹ in the June sampling. However, the DO level in the water was reduced to an average of 0.25 mg L⁻¹ in the October sampling and 0.16 mg L⁻¹ in the June sampling within the first 20% of the bed, indicating that there was little plant oxygen transfer into the wetland water during the active plant growth. Similar results were observed in experimental SF wetlands studied by Sievers (1997). There was no visible difference in measured DO levels along the bed length between the vegetated and non–vegetated beds (P > 0.05). This would imply that plant roots contribute little to the aeration process. However, it is reasonable to assume that plant roots oxygenate microsites close to the plant roots, making the DO available for rapid microbial uptake in these areas, but there is insufficient surplus oxygen to oxygenate the rest of the bed. In addition to contributing to the root–zone oxidative processes, the plant oxygen transfer is also offset by root respiration (Brix, 1993, 1997; Kadlec and Knight, 1996).

Bulrushes planted in the vegetated wetland beds had active growth from May through October. After two growing years, the plant roots were found to penetrate the entire substrate column (i.e., 27.5 cm), although the density of root fibers at the bottom layer of the wetland media was much less than in the upper section of the bed. The above–ground biomass of emergent plants also varied considerably spatially, higher in the inlet zone with growth tapering towards the outlet zone. The average biomass of emergent plants was higher in 1999 than in 1998, indicating continuous spatial growth and increased plant density. In 1998, the emergent plant biomass along the bed length ranged from 0.89 to 3.53 kg m⁻² on a dry weight basis, averaging about 2.36 kg m⁻². In 1999, the emergent plant biomass was 3.13 kg m⁻² at the inlet zone and 1.69 kg m⁻² at the outlet zone of the vegetated wetland beds. Based on the experimental data, plant roots and rhizomes contributed 3.25 kg m⁻² of dry biomass, as determined towards the end of 1999 operating year. Assuming nitrogen and phosphorus contents of bulrush biomass to be 2% and 0.15%, respectively, of the dry weight, as suggested by Kadlec and Knight (1996), the average nutrient values in the harvested emergent biomass were 532 kg N ha⁻¹ and 39.9 kg P ha⁻¹ in the 1999 growing year, when full plant maturity was considered attained in the wetland beds. These rates resulted in estimated 1.45 kg N ha⁻¹ d⁻¹ of nitrogen and 0.11 kg P ha⁻¹ d⁻¹ of phosphorus removal from the plant harvest during the 1999 growing season.

Table 1. Transect removal of BOD₅ and TSS along the bed length (sampling done in October 1999).

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<tr>
<th>Relative Longitudinal Distance (%)</th>
<th>Percent Reduction along the Bed Length</th>
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<tr>
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<td>BOD₅ (%)</td>
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Figure 4 shows that the influent TSS concentrations were extremely variable throughout the system operation. The maximum and minimum values for both the vegetated and non–vegetated wetland beds were 124 mg L⁻¹ in July 1999 and 29 mg L⁻¹ in November 1999, respectively. The highest concentrations occurred in the summer months from June through September. This was attributed to the summer algae blooms in the primary lagoon. It was observed that the TSS removal patterns and rates were similar for both the vegetated and non–vegetated beds, with no significant difference due to vegetation (P > 0.05). The TSS removals were high for both vegetated and non–vegetated wetlands and tended to follow seasonal variations, e.g., higher removals in the summer and fall, and lower removals in the winter and early spring. The TSS mass removal efficiency rates varied seasonally, with mean values for vegetative beds of 66% (σ = ± 29%) in 1998 and 73% (σ = ± 16%) in 1999. Mean values for non–vegetative beds were 68% (σ = ± 31%) in 1998 and 75% (σ = ± 16%) in 1999. The minimum rate for vegetated beds was 22% in February of 1998, when the wetland was new, and the maximum was 98% in September and October of 1998. For non–vegetated beds, the minimum removal rate (0%) occurred in February of 1998, and the maximum (96%) occurred in July and August of 1999. Results obtained from the transect analysis indicated that there was a rapid decline in TSS down the bed for both vegetated and non–vegetated wetlands, with more than 60% of TSS removed in the first half of the bed length (table 1).

Increased total solids concentrations were observed in the effluent of vegetated wetlands as compared to non–vegetated wetlands, primarily because of evapotranspiration (ET). Since the wetland system was operated in a semi–continuous mode, there was increased water loss due to plant transpiration, concentrating the salts in the wetland water. Kadlec and Knight (1996) and Kadlec (1989) reported that with batch feeding, evapotranspiration forms a significant fraction of the hydraulic loading, which could be as high as 10 mm d⁻¹ water loss for vegetated SF wetlands. When the system experiences water loss from the plant–rooting media, the outflow rates become smaller than the input hydraulic loading rates, concentrating solid constituents in the effluent. Kadlec and Knight (1996) suggested that the water mass losses due to surface evaporation from SF wetlands were negligible compared to plant transpiration rates. Considering unplanted wetlands as a control with minimal ET, the apparent plant transpiration losses were estimated for vegetated beds on the basis of increased electrical conductivity (fig. 5), which accounted for 20% of the total water mass loading in the first growing season and 58% in the second growing season.

### Nitrogen Removal

The ammonia concentrations in the influent varied greatly from month to month with a general, yet poorly defined, trend...
attributable to the changing season (fig. 6). That is, relatively higher concentrations of ammonia were observed in warmer spring and summer months than in colder winter months. This is best explained by the fact that the biological activity was higher in the primary lagoon (50% source of wastewater for this study) in warmer months, which facilitated the accelerated breakdown of organic nitrogen into ammonia. In addition, as algae died and decomposed into simpler compounds, it created an added source of nitrogen in the lagoon (usually observed in late spring and summer). However, frequent fluctuations of the water level in the primary lagoon may have caused the sudden surges of ammonia observed in some months (e.g., January and December of 1998). The mass loadings of ammonia nitrogen
into the SF system varied from 2.7 to 6.6 kg N ha\(^{-1}\) d\(^{-1}\), with the annual average loading of 4.1 and 4.8 kg N ha\(^{-1}\) d\(^{-1}\) for 1998 and 1999, respectively.

The ammonia nitrogen reductions followed seasonal variations in both the vegetated and non-vegetated wetland beds. The ammonia reduction rates in non-vegetated beds were significantly smaller than in vegetated beds (P < 0.05). As observed, the removal efficiency of NH\(_3\)-N appeared to increase slightly with the bed maturity. With the volatilization of ammonia nitrogen being minimal in these wetland beds, the small ammonia reductions observed in the non-vegetated beds were probably attributed to nitrification. The
lack of measurable dissolved oxygen in the wetlands likely restricted the nitrification process. In the subsurface reaction, nitrifying bacteria have to compete for oxygen with the heterotrophic bacteria responsible for breaking down organics. It is also possible that the retention time of 5 days was not sufficient for the nitrification process. In this case, effective ammonia removal would require either extended HRT (6 to 8 days) or a supplemental oxygen source for nitrification, as suggested by Drizo et al. (1997) and Reed et al. (1988).

Results indicated that nitrate concentrations in effluent from vegetated beds were very low, ranging from 0 to 0.32 mg N L^{-1} for the entire operation period (fig. 7). There were, however, slightly higher levels of nitrate in non–vegetated wetland beds with mean values of 0.6 mg L^{-1}–N in 1998 and 0.3 mg L^{-1}–N in 1999. As shown in figure 7, the nitrate values were virtually zero in the vegetated SF wetlands during the growing season. The low effluent nitrate concentrations were likely a result of denitrification in anaerobic zones within the wetland and plant uptake. The average annual mass removal for ammonia nitrogen in vegetated wetland beds was 3.3 kg ha^{-1} d^{-1} (σ = ± 2.6 kg ha^{-1} d^{-1}), which amounted to 75% removal. Removal rates in the non–vegetated beds varied from month to month (σ = ± 33%) with no apparent seasonal trend. Results indicated that the nitrate concentrations along the bed length showed no consistent pattern.

**Phosphorus Removal**

Annual average influent mass loading rate for dissolved phosphorus was 2.1 kg P ha^{-1} d^{-1} in 1998 and 1.8 kg P ha^{-1} d^{-1} in 1999, respectively. The influent phosphorus was generally greater in summer months due to greater biological activity in the primary lagoon, which broke down phosphorus–containing organic compounds into dissolved phosphorus (fig. 8). In addition, dead algae served as an additional phosphorus source in the lagoon water. Frequent fluctuation of water level in the primary lagoon (as much as 0.5 m) may have caused the sudden surge of influent phosphorus observed in some months of the season (e.g., September 1998 and July 1999).

The effluent phosphorus concentrations were reduced annually to an average of 1.7 (81% removal, σ = ± 26%) and 0.35 kg P ha^{-1} d^{-1} (10% removal, σ = ± 35%) in vegetated and non–vegetated wetland beds, respectively. From the performance results (fig. 8), it is clear that the removal of dissolved phosphorus from vegetated wetlands was significantly higher (P < 0.05) than from non–vegetated wetlands. There was very little removal of dissolved phosphorus in non–vegetated beds, with net internal production in some months. The internal production of dissolved phosphorus observed in non–vegetated wetlands was associated with the release from the mineralization of organic phosphorus previously trapped in the wetland. Figure 8 indicates that during the growing season (i.e., from May through October), the dissolved phosphorus mass removal rates in vegetated wetlands were almost 100%.

The sustained reduction of dissolved phosphorus in vegetated beds was likely caused by a luxury microbial and plant uptake (Metcalf and Eddy, 1991), acidic precipitation and complexation by humic substances, and sorption. Because the reduction rates in unplanted wetland beds were very small, it is also possible that the enhanced oxygenation by plants in the root zone may activate sorption sites on the biological attached growth. Plant roots are an important part of the biomass and comprise a significant fraction of the active phosphorus storage. The bottom layer of the substrate matrix below the root zone is relatively inactive in transforming phosphorus (Kadlec and Knight, 1996). Inability of plant
roots to spread densely and deeply into the subsurface substrate, coupled with extremely low oxygen levels as were determined in this study, likely restricted the uptake of phosphorus in the bottom layers.

**SUMMARY AND CONCLUSIONS**

Although these SF wetlands were built above ground, the results obtained in this study should approximate below–ground SF systems because of the use of shade cloth to limit summer temperatures and bed heaters to prevent winter freezing. In general, below–ground systems would be more buffered from temperature variations, and the exclusion of freezing. In general, below–ground systems would be more advantageous for constructing vegetated SF systems because of the use of shade cloth to limit direct rainfall would limit dilution. Major observations made in this study were:

- BOD removals throughout the two–year operating period were high for both vegetated and non–vegetated wetlands and tended to follow seasonal variations, e.g., higher removals in the summer and fall, and lower removals in the winter and early spring.
- Vegetation slightly enhanced the reduction of biodegradable organic matter in the SF environment when compared to non–vegetated wetlands, although the improvement was statistically insignificant.
- TSS removals were high for both vegetated and non–vegetated wetlands with no clear advantage for vegetated wetlands. The TSS mass removal efficiency rates tended to vary seasonally, with mean values of 66% ($\sigma = \pm 29\%$) in 1998 and 73% ($\sigma = \pm 16\%$) in 1999 for vegetated beds. For non–vegetated beds, the mean was 68% ($\sigma = \pm 31\%$) in 1998 and 75% ($\sigma = \pm 16\%$) in 1999.
- The average annual mass removal of ammonia nitrogen in vegetated wetland beds was 3.3 kg ha$^{-1}$ d$^{-1}$ ($\sigma = \pm 2.6$ kg ha$^{-1}$ d$^{-1}$), which amounted to 75% removal. The ammonia reduction during the growing season was as high as 100%. Removal rates in the non–vegetated wetland beds varied from month to month ($\sigma = \pm 33\%$) with no apparent seasonal trend. The ammonia removal process in vegetated wetland beds was much more pronounced than in non–vegetated beds. Since the nitrate levels in the effluent and along the bed length were very low, the nitrate produced as a result of nitrification was most likely denitrified into nitrogen gas or taken up by plants in the vegetated wetland beds.
- The dissolved phosphorus reduction in vegetated beds was greater than in non–vegetated beds, with differences varying depending on seasonal plant growth. The dissolved phosphorus reduction averaged 81% ($\sigma = \pm 26\%$) in vegetated wetland beds and 10% ($\sigma = \pm 35\%$) in non–vegetated beds. The sustained reduction of dissolved phosphorus in vegetated beds was likely caused by a luxury microbial and plant uptake, acidic precipitation and complexation by humic substances, and sorption.
- Results from this two–year, short–term study indicate that where ammonia nitrogen and dissolved phosphorus must be reduced, vegetated SF wetlands are preferred to non–vegetated SF wetlands. Where BOD$_5$ and TSS removal are major concerns, there is no significant difference between vegetative and non–vegetative wetlands.
- Based on the results from this two–year, short–term study, the recommended BOD$_5$ removal kinetic rates ($k_2$) for treating primary lagoon effluent using vegetated and non–vegetated wetlands are 0.65 and 0.55 d$^{-1}$, respectively. Most BOD$_5$ and TSS removal in the wetland media extended to only the first half of the wetland. And whenever there was ammonia and soluble phosphorus removal, primarily in the warmer months, it rapidly declined along the bed length. In addition, nitrate levels were very low throughout the wetland bed. Assuming that the hydraulic design of the system is adequate, these removal patterns may suggest that system designs with length/width ratios as low as 2 or 3:1 for SF wetland systems be used to treat primary effluent. For a similarly designed wetland surface area and detention time, the resulting larger cross–sectional flow area and shorter bed length should distribute more solids more uniformly over the bed length, reducing the possibility of premature inlet clogging.

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**REFERENCES**


