State of the art for animal wastewater treatment in constructed wetlands

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Abstract Although confined animal production generates enormous per-unit-area quantities of waste, wastewater from dairy and swine operations has been successfully treated in constructed wetlands. However, solids removal prior to wetland treatment is essential for long-term functionality. Plants are an integral part of wetlands; cattails and bulrushes are commonly used in constructed wetlands for nutrient uptake, surface area, and oxygen transport to sediment. Improved oxidation and nitrification may also be obtained by the use of the open water of marsh-pond-marsh designed wetlands. Wetlands normally have sufficient denitrifying population to produce enzymes, carbon to provide microbial energy, and anaerobic conditions to promote denitrification. However, the anaerobic conditions of wetland sediments limit the rate of nitrification. Thus, denitrification of animal wastewaters in wetlands is generally nitrate-limited. Wetlands are also helpful in reducing pathogen microorganisms. On the other hand, phosphorus removal is somewhat limited by the anaerobic conditions of wetlands. Therefore, when very high mass removals of nitrogen and phosphorus are required, pre- or in-wetland procedures that promote oxidation are needed to increase treatment efficiency. Such procedures offer potential for enhanced constructed wetland treatment of animal wastewater.
Keywords Dairy; denitrification; nitrification; phosphorus; redox; swine

Introduction

Animal production is a major component of agriculture in the USA. It is vital for both food stability and economic health. However, there are increasingly more environmental problems associated with the present-day scale of animal production. These problems include nuisance odors, pathogens, concentrated wastewater, inadequate land treatment sites, residential encroachment, and new regulations. Increasingly, large-scale animal production occurs in confinement where enormous per-unit-area quantities of waste are generated. Additionally, industry expansion and relocation have introduced these significant waste treatment issues to non-traditional areas. For example, much of the Florida dairy industry moved from Lake Okeechobee to the Suwanee River Region. New large dairies have moved into Texas and New Mexico during the 1990s. Similarly, swine production grew from 2 to 12 million pigs in North Carolina during the 1990s.

Currently, most enterprises apply both solid and liquid waste to land for terminal treatment. This traditional method of waste management was not only used for centuries; it was essential for food production since it was the primary source of cropland fertilization. Nonetheless, application of liquid animal waste to land has unique problems, such as, high solids content, high nutrient concentrations, and limited pumping distances. Regulators and the public are demanding improved alternatives. One of these alternatives is constructed wetlands, which are generally perceived to be a technology that is relatively affordable and operationally simple. Wetlands have been used successfully for advanced treatment of municipal and residential wastewaters in the USA and around the world for over three decades (Kadlec and Knight, 1996). Compared to conventional systems, they have less construction, operation, and energy costs plus more flexibility in pollutant loading. They are also flexible in soil specificity; constructed wetlands can be built on aerated upland

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soils, and hydric soil conditions will develop when the soils are flooded. These hydric conditions will then support aquatic plant life and wetland processes. Currently, there are livestock producers in at least 26 states across the USA using constructed wetlands to treat animal wastewaters. However, there are limited data for the treatment of animal wastewater in constructed wetlands.

Two types of wetlands (subsurface and free-water-surface systems) are typically used (Hammer, 1989). Gumbricht (1993) categorized free-water-surface systems into free-floating-plant ponds, submersed-plant ponds, and constructed wetlands with emergent plants. Payne and Knight (1997) as well as others considered wetlands with surface-flow emergent plants to be the only likely candidate for wide scale adoption. Subsurface systems are subject to clogging and limited oxygen (O_2) diffusion, and floating aquatic systems are more affected by pests and cold temperatures. Additionally, these wetlands are more expensive to construct and operate than surface-flow wetlands. This paper will focus on free-water-surface-flow systems – their components and performance.

Wetland components

Plants

Extensive work on plant material selection has been done by the Soil Conservation Service (Marburger, 1992). The selection of appropriate plants for constructed wetlands depends on the functional requirements of the system. The most commonly used plant genera in constructed wetlands for animal waste treatment are *Scirpus*, *Typha*, and *Juncus*. Generally, a polyculture of submersed, floating, and emergent plants occupies the wetlands. Additionally, the plant community is not static. It changes with conditions of plant community health, wetland operation, and weather.

Oxygen transport

Wetland plants transport O_2 from leaves and stems to roots (Armstrong, 1964), providing an oxidized microenvironment in the anaerobic root zone. The juxtaposition of aerobic and anaerobic zones at the root-water-soil interface is critical to the treatment of wastewater (Good and Patrick, 1987). The efficient use of wetlands for wastewater treatment depends on the O_2 transport capacity of the plant-root system and O_2 diffusion across the free soil-water interface. Diffusion rates of O_2 can be lower than 0.12 g m^{-2} h^{-1} in anaerobic soil (Stolzy and Flühler, 1978), while O_2 transport and diffusion through wetland plants range from 0.02 to 1.2 g m^{-2} h^{-1} (Kadlec and Knight, 1996). The higher values of O_2 transport could be very important in the nitrification of ammonia to nitrate, a process that requires 4.33 g O_2 per gram of ammonia-N (NH_3-N) to nitrate-N. Different plant species have different capacities for O_2 transport. For instance, bulrushes have higher rates of O_2 transport and more oxidized sediments than cattails (Reddy *et al.*, 1989; Szögi *et al.*, 1994). Oxygen availability is also affected by the O_2 demand of the wetland. This of course could be a significant problem for concentrated animal wastewaters. Oxygen concentration in wetland waters will vary with the season of the year and the time of day. With cooler water temperature, O_2 saturation is greater and the O_2 demand is smaller. Oxygen also varies diurnally with photosynthesis during the light and dark periods. This is particularly true in constructed wetlands in open water areas that produce O_2 via submersed macrophytes and phytoplankton. These open water areas can be used for animal wastewater treatment, and they can be designed into the system. For instance, the marsh-pond-marsh system as described by Hammer (1989) takes advantage of this O_2 production from submersed macrophytes and phytoplankton. Such systems have been used for swine wastewater treatment in Mississippi by Cathcart *et al.* (1994) and at NCA&T State University, Greensboro, NC, by Reddy *et al.* (2000). Additionally, the depth of water in the wetland can affect denitrification. Hunt *et al.* (2000) reported the wetlands were generally nitrate limited for denitrification and that denitrification enzyme potential decreased as water depth increased in wetlands used for swine wastewater treatment.

Carbon removal

Constructed wetlands will not completely remove carbon (C) because plant litter and plant/root exudate continually adds C to the system (Hunt *et al.*, 1994). Yet, low levels of soluble C are not a problem because it is necessary for anaerobic respiration and denitrification. Furthermore, carbon removal is generally not a land-limited aspect of animal waste treatment. In fact, removal of large quantities of C from wastewater is a strongpoint of land treatment systems; and dewatered waste materials, particularly if composted, can be transported and spread to available land. The main consideration for wetlands is avoidance of large application of suspended solids, which will fill the wetland and degrade its treatment functionality. Hunt and Vanotti (1999) discussed both passive and high tech methods of solids removal from animal waste.

Phosphorus removal

Phosphorus (P) removal is via sedimentation, plant uptake, organic matter accumulation, immobilization, and soil sorption. Phosphorus is present in soils and sediments in organic and inorganic forms. The relative proportions of organic and inorganic P vary widely. Organic P may comprise a substantial reservoir because the litter-sediment processes control the long-term P removal capability of wetland ecosystems (Faulkner and Richardson, 1989). Inorganic P is retained by calcite, clay minerals, organometallic complexes, and Fe and Al oxides and hydroxides (Parfitt, 1978; Gale *et al.*, 1994). Numerous investigators have found that oxalate-extractable iron is associated with P adsorption (Syers *et al.*, 1973). This fraction comprises the poorly crystalline iron oxi-hydroxides that become highly soluble under prevalent-reduced conditions. This increased P solubility explains why wetlands treated with wastewater can become P-saturated and export excessive quantities of phosphate in a few years (Richardson, 1985). A rapid decline of P-removal efficiency (from 99 to 78% in one year) for constructed wetlands that treated swine wastewater was observed by Szögi *et al.* (1994). This rapid decline was probably related to the high content of poorly crystalline iron oxi-hydroxides of the wetland soil, strong soil reduction, and high load of P.

Nitrogen removal

Nitrogen (N) is removed from wastewater through processes including filtration, sedimentation, uptake by plants and microorganisms, adsorption, nitrification-denitrification, and volatilization. Organic N can be initially removed via filtration and sedimentation, but it will be mineralized and released over time as NH_3-N. Ammonia-N in the form of ammonium (NH_4-N) ion can be absorbed either by wetland plants through roots or by anaerobic microorganisms and converted back to organic-N or immobilized as an exchangeable ion in soil. Szögi *et al.* (2000) reported that NH_4-N pore water profiles peaked just below the sediment-water interface (0 to 5-cm) and decreased with depth at all sites in surface flow wetlands used for swine treatment. It was postulated that the NH_4-N peak levels in the 0-5 cm layer were related to plant uptake, soil adsorption, microbial assimilation and mineralization of sediment organic matter.

Gaseous losses of N through nitrification-denitrification can be very large; they are generally the most significant N-removal mechanisms for natural and constructed wetlands (Bowden, 1987; Faulkner and Richardson, 1989). Under anaerobic conditions, NH_4-N would normally build up to excessive levels. However, O_2 diffusion from the atmosphere to the overlying floodwater and O_2 transport by plants to the rooting zone can form oxidized microsites. The gradient between high concentrations of NH_4-N in the reduced soil and low concentrations in the oxidized microsites and layers causes diffusion of NH_4-N into these oxidized microsites and layers where nitrification can occur (Patrick and Reddy, 1976; Reddy and Graetz, 1988). Nitrification requires pH values above 5, aerobic conditions, and autotrophic nitrifying bacteria. Rapid nitrification can occur at oxidized interfaces of the root or liquid surface or when the wetland is periodically dry. Nitrate-N can then diffuse into the anoxic zone where it is denitrified to dinitrogen gas (N_2).

Additionally, ammonia can be lost through volatilization under alkaline pH conditions. This mechanism was initially thought to be of little consequence since constructed wetlands for animal wastewater treatment are generally < 8 in pH. However, the high ammonia concentrations in the wastewater have caused concern that volatilization may be a significant factor even at neutral pH. The very high rates of ammonia loss and low dissolved O_2 values add to this concern because there may not be enough oxygen to account for the complete nitrification-denitrification of the lost ammonia. Large volatilization is, of course, undesirable because NH_3-N can be absorbed by the surrounding ecosystems (i.e., cropland, pastureland, and wooded zones) and cause ecosystems shifts. Pre-wetland nitrification may be necessary to fully exploit the denitrification potential of constructed wetlands, which could be over 50 kg ha^{-1} day^{-1} (Hunt *et al.*, 1999). Ammonia is generally a large portion of the total nitrogen content, and concentration > 200 mg L^{-1} may cause significant plant growth problems. Thus, there are plant health aspects that also encourage pre-wetland nitrification. However, our initial measurements indicate NH_3 volatilization is not a major factor.

Design

Constructed wetlands should be considered only as a component of a total animal wastewater treatment system. At a minimum, wastewater treatment for solids removal is needed ahead of the wetlands. Payne and Knight (1997) discussed the various design approaches in animal wastewater wetlands. The main methods are the NRCS guidelines, Reed *et al.* (1995), and Kadlec and Knight (1996). Stone *et al.* (2000) evaluated the design approaches in relation to the performance of swine wastewater wetlands. They found the design procedures reasonably accurate for nitrogen but phosphorus removal was overestimated.

Case studies of dairy and swine wastewater treatment

The results of several studies on constructed wetland treatment of dairy and swine wastewater are presented in Table 1 (Cathcart *et al.*, 1994; Skarda *et al.*, 1994; Hunt *et al.*, 1995; Cooper and Testa, 1997; Hermans and Pries, 1997; McCaskey and Hannah, 1997; Moore and Niswander, 1997; Reaves and DuBowy, 1997; Reddy *et al.*, 2000). All parameters, except fecal coliform, exhibited large variabilities between sites. For example, BOD_5 ranged from 53 to 93%, TKN ranged from 37 to 86%, and TP ranged from 42 to 83% removal. The treatment systems in Kosciusko Co., IN, and in Alabama showed the best performances with > 90% removal of total solids and BOD, > 80% removal of N, and > 75% removal of P.

The focus of our research in Duplin Co., NC, was to determine wetland treatment efficiency of swine wastewater and define redox conditions, denitrification potentials, and agronomic cropping potentials of constructed wetlands used for swine wastewater treatment (Hunt *et al.*, 1999; Szögi *et al.*, 2000). Three sets of two, 3.6 by 36 m wetland cells were constructed in Duplin Co., NC, in 1992; they contained either natural wetland plants or water-tolerant agronomic plants. Nitrogen loading rates of 3 to 25 kg ha^{-1} day^{-1} were used (Table 2). Mass N removal ranged from 81 to 94%. Phosphorus removal was much lower as the rate of application increased, < 35%. Redox conditions were highly anaerobic in the soils of all wetland cells in summer. Hunt *et al.* (1994) reported that rush (*Juncus effusus*) and bulrushes (*Scirpus americanus*, *Scirpus cyperinus*, and *Scirpus validus*) were not greatly different from bur-reed (*Sparganium americanum*) and cattails (*Typha angustifolia* and *Typha latifolia*) in effective treatment of swine wastewater. However, the bulrushes had more oxidized sediment than did the cattails. The higher O_2 transport rates of the bulrush allowed mildly oxidized soil conditions in the winter. Denitrification enzyme assay indicated that the wetland soils were nitrate-limited for denitrification. Szögi *et al.* (2000) also reported that saturation culture soybean and flooded rice satisfactorily treated swine wastewater in a constructed wetland, and the seed harvest removed significant amounts of nutrients as grain. The agronomic plants treated swine effluent very similarly to natural wetland plants when loading rates were < 10 kg ha^{-1} d^{-1} of N.

Ammonia volatilization was also measured at the Duplin Co. site with an open chamber device. These tests indicated that ammonia volatilization was occurring. From average hourly rates, it was estimated that 7 to 17% of the nitrogen load to the wetlands was removed through NH_3 volatilization. Because tests were conducted at only one wetland site, we need additional data before we can make a definitive conclusion on ammonia volatilization, but these results indicated that NH_3 volatilization was not responsible for removing the majority of nitrogen from the swine wastewater. This suggests that either oxygen diffusion and hence nitrification-denitrification is underestimated or there is another mechanism for ammonia loss that is being overlooked. Two novel nitrogen pathways may account for discrepancies in the data. The first has been labeled anaerobic ammonia oxidation. It is described by the equation:

Table 1 Operational reductions in dairy and swine wastewater parameters using constructed wetlands (Cathcart et al., 1994; Skarda et al., 1994; Hunt et al., 1995; Cooper and Testa, 1997; Hermans and Pries, 1997; McCaskey and Hannah, 1997; Moore and Niswander, 1997; Reaves and DuBowy, 1997; Reddy et al., 2000)

Site	BOD_5	COD	TSS	NH_3-N	TKN	TP	Coliform
	% reduction in concentration						
Dairy							
Kosciusko Co., In	93	–	94	89	86	83	–
Oregon State University	61	47	73	54	57	66	94
DeSoto Co., MS (11°C)	68	50	59	68	–	42	89
DeSoto Co., MS (22°C)	84	77	70	81	–	63	97
Mercer Co., KY	66	–	88	87	37	59	–
Essex, Ontario	66	–	66	80	–	69	99
Swine							
MS (Marsh-Pond-Marsh)	54	–	69	71	–	44	–
NCA&TSU, NC (Marsh-Pond-Marsh)	53	–	68	60	51	44	–
AL (continuous wetland)	90	–	89	85	83	76	–

Table 2 Mass removal of N in constructed wetlands, Duplin Co., NC (June '93–Nov. '97)

Nitrogen Load	System	
	Rush/bulrush	Cattails/bur-reed
kg ha^{-1} day^{-1}	Mass Removal, %	
3	94	94
8	88	86
15	85	81
25	90	84

% Mass Removal = % mass reduction of N (NH_3-N + NO_3-N) in the effluent with respect to the nutrient mass inflow of all wetland cells in summer. Hunt *et al.* (1994) reported that rush (*Juncus effusus*) and bulrushes (*Scirpus americanus*, *Scirpus cyperinus*, and *Scirpus validus*) were not greatly different from bur-reed (*Sparganium americanum*) and cattails (*Typha angustifolia* and *Typha latifolia*) in effective treatment of swine wastewater. However, the bulrushes had more oxidized sediment than did the cattails. The higher O_2 transport rates of the bulrush allowed mildly oxidized soil conditions in the winter. Denitrification enzyme assay indicated that the wetland soils were nitrate-limited for denitrification. Szögi *et al.* (2000) also reported that saturation culture soybean and flooded rice satisfactorily treated swine wastewater in a constructed wetland, and the seed harvest removed significant amounts of nutrients as grain. The agronomic plants treated swine effluent very similarly to natural wetland plants when loading rates were < 10 kg ha^{-1} d^{-1} of N.

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25	90	84

% Mass Removal = % mass reduction of N (NH_3-N + NO_3-N) in the effluent with respect to the nutrient mass inflow

$$NH_4^+ + NO_2^- \rightarrow N_2 + 2H_2O$$

Recent research indicates that this process is performed by ammonia oxidizing bacteria and a newly discovered bacterium (Jetten *et al.*, 1999). To convert ammonia to N_2 through this pathway requires only half of the oxygen needed to convert ammonia to N_2 by conventional nitrification-denitrification. It also may be possible by some as yet unidentified process for ammonia to be converted to N_2 under completely anoxic conditions. This possibility is supported by recent data on N_2 production from animal waste lagoons (Lowry Harper, USDA-ARS, Athens, GA, personal communication).

The combined results of these studies on wastewater treatment suggest that constructed wetlands are excellent for mass removal of N. However, at the high loading rates necessary for substantive mass removal, constructed wetlands do not produce an effluent acceptable for discharge. Thus, subsequent land application is necessary. Croplands, vegetative strips, and woodlands are viable options for final treatments. Because terminal land application does not require a polished effluent, it is an approach that fits well with the capacities of constructed wetlands. Furthermore, pre- and post-wetland treatments will allow wetland adaptations to fit unique and changing water quality requirements.

References

Armstrong, W. (1964). Oxygen diffusion from the roots of some British bog plants. *Nature*, **204**, 801–802.

Bowden, W.B. (1987). The biogeochemistry of nitrogen in freshwater wetlands. *Biogeochemistry*, **4**, 313–348.

Cathcart, T.P., Hammer, D.A. and Triyono, S. (1994). Performance of a constructed wetland-vegetated strip system used for swine waste treatment. In: *Constructed Wetlands for Animal Waste Management*, P.J. DuBowy and R.P. Reaves (eds.), Purdue Research Foundation, West Lafayette, IN, pp. 9–22.

Cooper, C.M. and Testa, S., III. (1997). A constructed bulrush wetland for treatment of cattle waste. In: *Constructed Wetlands for Animal Waste*, E.P.A. Special Publication, Gulf of Mexico Program – Nutrient Enrichment Committee, Payne Engineering and CH2M Hill, pp. II.14–II.24.

Faulkner, S.P. and Richardson, C.J. (1989). Physical and chemical characteristics of freshwater wetland soils. In: *Constructed Wetlands for Wastewater Treatment – Municipal, Industrial and Agricultural*, D.A. Hammer (ed.), Lewis Publishers, Chelsea, MI, pp. 41–72.

Gale, P.M., Reddy, K.R. and Graetz, D.A. (1994). Phosphorus retention by wetland soils used for treated wastewater disposal. *J. Environ. Qual.*, **23**, 370–377.

Good, B.J. and Patrick, W.H., Jr. (1987). Root-water-sediment interface processes. In: *Aquatic Plants for Water Treatment and Resource Recovery*, K.R. Reddy and W.H. Smith (eds.), Magnolia Publishing, Orlando, FL, pp. 359–370.

Gumbricht, T. (1993). Nutrient removal processes in freshwater submersed macrophyte systems. *Ecol. Eng.*, **2**, 1–30.

Hammer, D.A. (ed.) (1989). *Constructed Wetlands for Wastewater Treatment – Municipal, Industrial, and Agricultural*. Lewis Publishers, Chelsea, MI.

Hermans, P. and Pries, J. (1997). Essex treatment wetland, Essex, Ontario, Canada. In: *Constructed Wetlands for Animal Waste Treatment*, E.P.A. Special Publication, Gulf of Mexico Program – Nutrient Enrichment Committee, Payne Engineering and CH2M Hill, pp. II.25–II.29.

Hunt, P.G. and Vanotti, M.B. (1999). Animal residual treatment and soil and water resource management. *Animal Residuals Management Conf. Proc.* (CD-ROM), 10 pp.

Hunt, P.G., Szögi, A.A., Humenik, F.J., Rice, J.M. and Stone, K.C. (1994). Swine wastewater treatment by constructed wetlands in the southeastern U.S. In: *Constructed Wetlands for Animal Waste Management*, P.J. DuBowy and R.P. Reaves (eds.), Purdue Research Foundation, West Lafayette, IN, pp. 144–154.

Hunt, P.G., Thom, W.O., Szogi, A.A. and Humenik, F.J. (1995). State of the art for animal wastewater treatment in constructed wetlands. *Proc. 7th Intl. Symp. on Agricultural and Food Processing Wastes*, pp. 53–65.

Hunt, P.G., Szogi, A.A., Humenik, F.J. and Rice, J.M. (1999). Treatment of animal wastewater in construction wetlands. *8th Intl. Conf. on the FAO ESCORENA Network on Recycling of Agricultural, Municipal and Industrial Residues in Agriculture Proc.* (Rennes, France), pp. 305–313.

Hunt, P.G., Szogi, A.A., Humenik, F.J., Reddy, G.B., Poach, M.E., Sadler, E.J. and Stone, K.C. (2000). Treatment of swine wastewater in wetlands with natural and agronomic plants. *9th Workshop of FAO ESCORENA Network on Recycling of Agricultural, Municipal and Industrial Residues in Agriculture Proc.* (Gargnano, Italy). (accepted for publication June 20, 2000).

Jetten, M.S.M, Strous, M., Schoonen, K.T., Schalk, J., van Dongen, L.G.J.M., van de Graaf, A.A., Logemann, S., Muyzer, G., van Loosdrecht, M.C.M. and Kuenen, J.G. (1999). The anaerobic oxidation of ammonium [Review]. *FEMS Microbiology Reviews*, **22**(5), 421–437.

Kadlec, R.H. and Knight, R.L. (1996). *Treatment Wetlands*. Lewis Publishers, Boca Raton, FL.

Marburger, J.E. (1992). Wetland plants. Plant materials technology needs and development for wetland enhancement, restoration, and creation in cool temperate regions of the United States, Terrene Institute, Washington, DC.

McCaskey, T.A. and Hannah, T.C. (1997). Performance of a full scale constructed wetland treating swine lagoon effluent in northern Alabama. In: *Constructed Wetlands for Animal Waste Treatment*, E.P.A. Special Publication, Gulf of Mexico Program – Nutrient Enrichment Committee, Payne Engineering and CH2M Hill, pp. II.5–II.8.

Moore, J.A. and Niswander, S.F. (1997). Oregon State University dairy wetland In: *Constructed Wetlands for Animal Waste Treatment*, E.P.A. Special Publication, Gulf of Mexico Program – Nutrient Enrichment Committee, Payne Engineering and CH2M Hill, pp. II.30–II.33.

Parfitt, R.L. (1978). Anion adsorption by soils and soil materials. *Adv. Agron.*, **30**, 1–50.

Patrick, W.H., Jr. and Reddy, K.R. (1976). Nitrification-denitrification reactions in flooded soils and sediments: Dependence on oxygen supply and ammonium diffusion. *J. Environ. Qual.*, **5**, 469–472.

Payne, V.W.E. and Knight, R.L. (1997) Constructed wetlands for treating animal wastes – Section I: Performance, design, and operation. In: *Constructed Wetlands for Animal Waste Treatment*, E.P.A. Special Publication, Gulf of Mexico Program – Nutrient Enrichment Committee, Payne Engineering and CH2M Hill.

Reaves, R.P. and DuBowy, P.J. (1997). Tom Brothers' dairy constructed wetland. In: *Constructed Wetlands for Animal Waste Treatment*, E.P.A. Special Publication, Gulf of Mexico Program – Nutrient Enrichment Committee, Payne Engineering and CH2M Hill, pp. II.9–II.13.

Reddy, K.R. and Graetz, D.A. (1988). Carbon and nitrogen dynamics in wetland soils. In: *The Ecology and Management of Wetlands*, D.D. Hook *et al.* (eds.), Vol. I, Timber Press Portland, OR, pp. 307–318.

Reddy, K.R., D'Angelo, E.M. and DeBusk, T.A. (1989). Oxygen transport through aquatic macrophytes: The role in wastewater treatment. *J. Environ. Qual.*, **19**, 261–267.

Reddy, G.B., Hunt, P.G., Phillips, R., Stone, K.C. and Grubbs, A. (2000). Treatment of swine wastewater in marsh-pond-marsh constructed wetlands. *7th Intl. Conf. on Wetland Systems for Water Pollution Control*, 8 pp.

Reed, S.C., Crites, R.W. and Middlebrooks, E.J. (1995). *Natural Systems for Waste Management and Treatment*. 2nd edn, McGraw-Hill, NY.

Richardson, C.J. (1985). Mechanisms controlling phosphorus retention capacity in freshwater wetlands. *Science*, **228**, 1424–1427.

Skarda, S.M., Moore, J.A., Niswander, S.F. and Gamroth, M.J. (1994). Preliminary results of wetland for treatment of dairy farm wastewater. In: *Constructed Wetlands for Animal Waste Management*, Purdue Research Foundation, West Lafayette, IN, .J. DuBowy and R.P. Reaves (eds.), pp. 34–42.

Stolzy, L.H. and Flühler, H. (1978). Measurement and prediction of anaerobiosis in soils. In: *Nitrogen in the Environment*, D.R. Nielsen and J.G. McDonald (eds.), Vol. 1, Academic Press, New York, pp. 363–426.

Stone, K.C., Hunt, P.G., Szogi, A.A., Humenik, F.J. and Rice, J.M. (2000). Constructed wetland design and performance for swine lagoon waste water treatment. ASAE Paper #00-4148, 11 pp.

Syers, J.K., Harris, R.F. and Armstrong, D.E. (1973). Phosphate chemistry in lake sediments. *J. Environ. Qual.*, **2**, 1–14.

Szögi, A.A., Hunt, P.G., Humenik, F.J., Stone, K.C., Rice, J.M. and Sadler, E.J. (1994). Seasonal dynamics of nutrients and physico-chemical conditions in a constructed wetland for swine wastewater treatment. ASAE Paper #94-2602.

Szögi, A.A., Hunt, P.G. and Humenik, F.J. (2000). Treatment of swine wastewater using a saturated-soil-culture soybean and flooded rice system. *Trans. ASAE*, **43**(2), 327–335.