STATE OF THE ART FOR ANIMAL WASTEWATER TREATMENT IN CONSTRUCTED WETLANDS

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ABSTRACT

Confined animal production generates enormous per-unit-area quantities of waste, and wastewater from dairy and swine operations has been successfully treated in constructed wetlands. However, solids removal prior to wetlands treatment is essential for long-term functionality. When wetlands were combined with grass filter strips, an application of swine wastewater containing 14 kg ha^{-1} day^{-1} of N was treated to over 95% N removal. Plants are an integral part of wetlands, and cattails and bulrushes are commonly used in constructed wetlands. However, bulrushes transport more O_2 to sediment. Improved oxidation and nitrification may also be obtained by the use of the open water strips of marsh/pond/marsh designed wetlands. Wetlands normally have sufficient denitrifying population to produce enzymes, C 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. Phosphorus removal is also somewhat limited by the anaerobic conditions of wetlands. Therefore, when very high mass removal of N and P is required, pre- or in-wetland procedures that promote oxidation are needed to increase treatment efficiency. Such procedures offer the greatest potential for improved treatment capacity for constructed wetland treatment of animal wastewater.

INTRODUCTION

Over one-sixth of the $175 billion agricultural economy consists of animal production. Increasingly, large-scale animal production occurs in confinement where enormous per-unit-area quantities of waste are generated. Therefore, to remain viable, animal production enterprises must have functional and sustainable waste management systems. These systems must process both liquid and solid waste for dairy and swine enterprises. Currently, most enterprises apply both solid and liquid waste to land for terminal treatment. Application of liquid animal waste to land has unique problems, such as nuisance odor, high solids content, high nutrient concentrations, and limited pumping distances. In addition to these technical problems, new regulations, residential encroachment, and increased animal numbers often cause the available land treatment sites to be inadequate. Thus, producers continually search for animal wastewater treatment systems that are more efficient but also less expensive, labor intensive, or land consumptive.

Constructed wetlands have received attention in recent years as a method of animal wastewater treatment. Wetlands have been used successfully for advanced treatment of municipal and residential wastewaters in the United States and around the world for over three decades (Watson et al., 1989). Compared to conventional systems, they have less construction, operation, and energy costs but more flexibility in pollutant loading. They are also flexible in soil specificity; constructed wetlands can be built on aerated upland soils, and the 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

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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-flow) are typically used (Hammer, 1989). Subsurface systems are subject to clogging and limited O_2 diffusion. Free-water-surface-flow systems are more suitable for animal wastewater treatment. This paper will focus on free-water-surface-flow systems. Gumbricht (1993) categorized free-water-surface systems into free-floating-plant ponds, submersed-plant ponds, and constructed wetlands with emergent plants.

WETLAND FUNCTIONALITY

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 requirement of the system. It has been demonstrated that free-floating plants can be very effective in the treatment of raw sewage, as well as primary and secondary treated effluent (Reddy and DeBusk, 1985; Reddy et al., 1989). Submersed plants are less suited for treatment of raw sewage and have been used for tertiary treatment and polishing municipal effluents (Gumbricht, 1993). Emergent plants are usually used as a component in constructed wetlands for wastewater treatment. The roots and rhizomes of the plants play an important role in the nutrient removal process. They provide surfaces for bacterial growth, filtration of solids, nutrient uptake (Nichols, 1983), and O_2 to anoxic soil environments to promote nitrification-denitrification (Armstrong, 1964; Reddy et al., 1989). The most commonly used genera in constructed wetlands for animal waste treatment are *Scirpus*, *Typha*, and *Juncus*. 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. They 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.

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 its 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 100 to 400 g m^{-2} h^{-1} (Reddy et al., 1989). Different plant species have different capacity 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. Wastewaters with extremely high biochemical oxygen demands (BOD) will generally exceed the capacity of the wetland to supply O_2 and will consequently limit treatment. Such wastewaters may require dilution, aerobic pre-treatment, or very low loading rates. 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 submergent macrophyte 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 submergent macrophytes and phytoplankton. Such a system has been used for swine wastewater treatment in Mississippi by Cathcart et al. (1994). Oxygen can also be applied to wastewater in pre- and post-treatments such as overland flow. In overland flow, the water depth is kept very thin, and O_2 can diffuse across a short distance from the atmosphere to the active microbial site on the soil and plant surfaces (Hunt and Lee, 1976). This allows effective development of very thin aerobic and anaerobic zones.

Carbon Removal

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. However, C removal is an essential aspect of N removal in wetlands because high C levels increase the O_2 demand and promote the growth of heterotrophic organisms. Low O_2 levels will decrease nitrification because it is carried out aerobically by autotrophic bacteria, and the increased heterotrophs will outcompete the autotrophs for surface area on which nitrification could take place. Constructed wetlands will not completely remove C because plant litter and plant/root exudate continually add C to the system (Hunt et al., 1994). Yet, low levels of soluble C are not a problem because soluble C is necessary for anaerobic respiration and denitrification.

Substantial amounts of C can be removed from wastewater by solid-liquid separation. Several methods of dewatering have been developed. The USDA-Agriculture Research Service, Dairy Research Unit at Madison, Wisconsin, developed a dewatering press for dairy wastewater that has promise; and the FAN separator (FAN Engineering USA, Inc.[2], Columbus, OH) is a commercially available device that is being used to reduce the solids load to lagoons. Solids removal can also be enhanced by increased residence time in a settling basin, and both flocculation and precipitation can be increased by polymer additions. Separated solids can be spread directly on the land or composted. Reduction of the C load that remains in the wastewater can be accomplished in an anaerobic lagoon. If further C reduction is needed, the wastewater can be treated in a facultative lagoon where wind aeration or pumped air can supply O_2 for accelerated decomposition and nitrification. After the reduction of the C level, wastewaters can then be run through an alternating sand filter, media filtration system, or across an overland flow treatment site.

Nitrogen Removal

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 can take several possible pathways. Ammonia-N can be lost through volatilization under alkaline pH conditions, which often occurs in open water areas where algal growth can consume large amounts of CO_2 (Mikkelsen et al., 1978; Reddy and Graetz, 1981). It might even be possible to manage wastewaters to promote NH_3-N loss. However, NH_3-N can be absorbed by the surrounding ecosystems (i.e., cropland, pastureland, and wooded zones), and continual emissions of large amounts of NH_3-N may cause undesirable shifts in these ecosystems. 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. Under anaerobic conditions, NH_4-N would normally build up to excessive levels. However, both O_2 diffusion from the atmosphere

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[2] Mention of a trade-mark, proprietary product, or vendor is for information only and does not constitute a guarantee or warranty of the product by the U.S. Department of Agriculture and does not imply its approval to the exclusion of other products or vendors that may also be suitable.

overlying floodwater and O_2 transport by plants to the rooting zone can form an oxidized upper zone in the soil. The gradient between high concentrations of NH_4-N in the reduced soil and low concentrations in the oxidized layer causes an upward diffusion of NH_4-N in the oxidized zone, where transformation to NO_3-N occurs (nitrification) by bacteria (Patrick and Reddy, 1976; Reddy and Graetz, 1988). Nitrification requires pH values above 5, aerobic conditions, and autotrophic nitrifying bacteria. Rapid nitrification can occur when the wetland is periodically dry from lack of wastewater application or very high evapotranspiration. Nitrate-N is a very mobile ion and can be rapidly assimilated by plants and microbes or lost through surface and ground water flow. However, under wetland conditions, it diffuses from the aerobic soil layer to the anaerobic zone and undergoes microbial reduction (Reddy et al., 1980). Denitrification is a microbial respiration process that reduces NO_3-N to nitrous oxide and molecular N. It requires denitrifying microbes, anaerobic conditions, and C as an energy source. Gaseous losses of N can be very large; they are generally the most significant N-removal mechanism for natural and constructed wetlands (Bowden, 1987; Faulkner and Richardson, 1989).

Phosphorus Removal

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.

Design

Constructed wetlands should be considered only as a component of a total animal wastewater treatment system. Wastewater treatment for solids removal is needed ahead of the wetlands. Treatment options include mechanical solids separation, stack pads with leachate collection for storage of animal wastes, or collection of the total animal waste stream in lagoons for solids settling and treatment (USDA-SCS, 1992b). Solids removal will ensure that the wetlands are not so loaded with C that they are totally anaerobic and incapable of supporting the interactive functioning of aerobic and anaerobic processes such as nitrification and denitrification. Problems with NH_3-N concentration and the potential toxicity to wetland plants can be reduced by management of rainwater and lot runoff as well as by dilution with fresh-water-recycled effluent.

Since the US-EPA does not allow direct stream discharge of animal wastewater, wetland effluent must be reused or applied to land. Wastewaters are generally 1) irrigated directly to surrounding land (i.e., vegetative filter strips, cropland, or woodlands) or 2) further treated in receiving ponds or lagoons before being discharged, reused as wash water, or recycled to the wetland.

Several elements must be considered in designing the wetlands component of an animal waste treatment system. They include 1) sources of wastewater (animal type and number, lot runoff, rainfall collection, milkhouse, stack pad leachate, etc.); 2) wastewater content (BOD, TSS, N, P, etc.); 3) hydraulic flow that affects BOD reduction, pollutant transport rate, and odor; 4) seepage and evapotranspiration losses; 5) suitable finishing stage for final treatment; and 6) total land area needed for all components of the entire treatment system.

Discussions of design and operation of constructed wetlands for municipal wastewater can be found in Findlater et al. (1990), Watson et al. (1989), Hammer (1989), Reed and Brown (1992), and Reed (1993). The only substantive guide for constructed wetlands for animal wastewater treatment has been published by the Natural Resource Conservation Service (USDA-SCS, 1991). The NRCS lists objectives for wetlands effluent that has BOD and TSS of < 30 mg L^{-1}, NH_3 + NH_4 < 10 mg L^{-1}, and hydraulic residence time of 12 days. However, these effluent characteristics may not be as important as mass load reduction, particularly N and P. Excessive mass loading of these nutrients can cause overloading and limited life of the terminal land application area.

Actual design and operation of constructed wetlands will depend upon the quantity and quality of specific wastewater loads, and different sources produce wastewater with very different

Table 1. Characteristics of Dairy Wastewaters from Varying Sources.

Parameter	Stack pad leachate	Paved barnyard runoff	Runoff,* leachate and milkhouse	Milkhouse
	mg L^{-1}			
BOD	2050	----	13,500	150 - 810
TSS	1120	360 - 1940	----	310 - 2175
VSS	----	220 - 1640	----	115 - 1430
TKN	960	95 - 235	710	55 - 247
NH_3-N	402	20 - 95	38	15 - 63
TP	136	220 - 650	80	11 - 32
SP	68	10 - 50	----	6 - 19

* Baldwin and Davenport (1994).

Table 2. Swine Waste Characterization—Anaerobic Lagoon; Feedlot Runoff (USDA-SCS, 1992a).

Component	Units	Anaerobic lagoon Supernatant	Anaerobic lagoon Sludge	Feedlot runoff* Runoff water	Feedlot runoff* Settling basin sludge
Moisture	%	99.75	92.40	98.50	88.8
TS	% w.b.	0.25	7.60	1.50	11.2
VS	mg/L x 10^3	1.2	46		41.0**
COD	"	1.2	65		
BOD	"	.4			
TKN	"	.35	3	.91**	2.5**
NH_4-N	"	.22	.8	.54**	2.0**
TP	"	.08	2.7	.17**	1.0**
K	"	.38	7.6	.50**	4.5**
C:N ratio	"	2	8		

* Semi-humid climate (approx. 760-mm annual rainfall); annual sludge removal.
** g day^{-1} Kg^{-1} of pig weight.

characteristics as illustrated in Tables 1 and 2. However, any design can be rendered nonfunctional by operator neglect. Operation within design specification and judicious monitoring of the overall functioning of the wetlands are essential.

CASE STUDIES OF DAIRY WASTEWATER TREATMENT

The results of three studies on constructed wetland treatment of dairy wastewater are presented in Table 3. In Lagrange Co., Indiana, constructed wetlands were used to treat wastewater from a 70-milk-cow herd (Reaves et al., 1994). Barnyard runoff, milkhouse wastewater, and manure leachate passed through a settling pad prior to application to the wetland cells. The system consisted of three, surface water, wetland cells operated in parallel. Each cell was 6.1 by 61 m with a bottom slope of 1%. Cell bottoms were lined with bentonite, and hydric topsoil was added. The predominate vegetation was cattails (*Typha latifolia*), smartweed (*Polygonum* spp.), and reed canary grass (*Phalaris arundinacea*). Treatment efficiencies were sufficient to be helpful in reducing the nonpoint load toward Lake Appleman, which was downslope of the dairy. However, the wetlands fell short of their potential because of excessive solids loading. A 10-cm-thick layer of solids accumulated in the initial third of each wetland cell during the first year of operation. These results point out the necessity of diligence in solids removal operations as well as design for solids removal prior to wetland application.

Oregon State University dairy farm had six wetland cells that were operated in parallel (Skarda et al., 1994). Cells were 4.6 x 29 m with 30-cm water depth. The dairy used a recycling flush system with solids removed, wastewater was stored in a large tank prior to entering the wetlands, and the wetlands discharge was collected in a storage pond for pumping back to the storage tank. Pollutants entering the wetlands were maximized at 100 mg L^{-1} for NH_3-N and 1500 mg L^{-1} for total solids, and BOD had a maximum daily load of 74 kg ha^{-1}. These data were taken during the start-up phase, and efficiencies were not quite as high as for the Lagrange Co. system.

Table 3. Operational Reductions in Dairy Wastewater Parameters with Constructed Wetlands for Three Systems (adapted from DuBowy and Reaves, 1994; Cooper et al., 1992b).

Parameter	Lagrange Co., IN	Oregon State Univ.	DeSoto Co., MS (Cells 1&3) Start-up	DeSoto Co., MS (Cells 1&3) Stable
	% reduction			
BOD_5	79	61	58	75
COD	--	47	--	--
TS	--	49	23	26
TSS	72	73	50	64
TDS	36	--	12	12
TKN	64	57	--	--
NH_3-N	64	54	73	90
NO_3-N	62	75	--	--
TP	74	66	58	61
SP	63	63	57	63
DO	--	97	41	49
EC	--	24	--	--

A DeSoto Co., Mississippi, dairy farm had three 6.1- x 24.4-m cells operating in parallel that received wastewater from a 41.1- x 51.8-m primary lagoon (Cooper et al., 1992b). Total waste volume entering the lagoon was 10.3 m^3 day^{-1}. It included barnyard runoff from a 352-m^2 cattle holding lot, rainfall from building roofs, and milkhouse wastewater. Cells 1 and 3 were single cells, but cell 2 was divided into four equal compartments. During start-up, most nutrient removal efficiencies were lower than after the cells reached a stable phase. This could be related to the more complete integration of microbial activity with macrophytes in the wetland. Primary treatment and dilution resulted in a lower influent BOD concentration and in wetland-treated effluent BOD < 30 mg L^{-1}. Reductions of 54% for BOD, 92% for total coliforms, and 60% for NH_3-N occurred in the first quarter of cell 2. Dissolved O_2 dropped from 3.2 to 1.3 mg L^{-1} in the first cell, but it rose slightly to 2.5 mg L^{-1} in the effluent. Both total and soluble P were reduced by greater than 66%, but the reductions were more equally distributed throughout the cells. This suggests that the total capacity of the wetland's chemical and physical processes of adsorption, precipitation, and sedimentation were required and that P removal may decrease with time. Total solids and dissolved solids were not removed effectively by any of the systems, but total solids can be removed by subsequent land application.

Thom and coworkers (unpublished data) evaluated constructed wetlands on a Mercer Co., Kentucky, dairy farm. It had two 9.1- x 24.4-m cells in series; they were planted with cattails (*Typha angustifolia*). The wastewater originated from lot runoff, milking facilities, and leachate from a covered manure stack pad. The liquids were collected in a settling basin for solids reduction and discharged by gravity to the first wetland cell. Table 4 lists parameter means for the settling basin effluent as well as the influent and effluent from the first wetland cell during the first 12 months of operation. The settling basin removed a large amount of TKN, NH_3-N, volatile solids, and total solids; but it did not greatly reduce the BOD. The first wetland cell further reduced the contaminant concentration except for TKN. There was no discharge from the second cell during this period. However, there was an at-grade rock-filled trench to allow discharge to flow over a fescue (*Festuca arundinacea*) vegetative filter strip (18.3 m x 53.4 m) as a final nutrient removal component. Thus, this system has functioned very well for mass removal of contaminants.

CASE STUDIES OF SWINE WASTEWATER TREATMENT

In Mississippi, Cathcart et al. (1994) studied a marsh/pond/marsh constructed wetland system for the treatment of swine wastewater. It contained two, parallel 0.04-ha constructed wetlands in

Table 4. Water Quality Parameters in the Settling Basin and First Cell of a Wetland Receiving Dairy Wastewater, Mercer Co., KY.

Parameter	Settling Basin	Influent	Effluent	Reduction
	----------	mg/L	----------	%
DO	0.5	0.6	0.8	--
BOD	465	452	158	66
TSS	3516	1132	408	88
VSS	2085	898	357	83
TP	113.8	71.6	47.1	59
SP	60.5	26.5	15.0	75
TKN	197.0	107.5	123.8	37
NH_3-N	78.8	32.8	10.3	87

series with two, 0.04-ha vegetative strips. Their wetland cells were 33 m long with less than a 1% slope. The shallow ends were planted with cattail (*Typha latifolia*, L.) and water chestnut (*Trapa nutans*, L.). The pond, a 15-m section in the middle of the wetland, was 23 cm deeper than the ends and unplanted. The combination of depth and turbidity has restricted emergent plant encroachment in the pond section. The wastewater flowed from a facultative lagoon that primarily treated wastewater from a farrowing house. Wastewater had an NH_3-N concentration of about 110 mg L^{-1}. The post-wetland vegetative strips were 46 m long and contained grasses, weeds, and woody shrubs. The hydraulic loading rate was 1.3 cm day^{-1}. The BOD, NH_3-N, and total o-PO4-P were loaded at 6.1, 14.3, and 7.8 kg ha^{-1} day^{-1}, respectively.

The oxidative influence of the pond section was demonstrated by data taken between April and July 1993. The mean and range for O_2 concentrations of wastewater at the influent, pond section, and effluent were 3.4 (0.2-15.0), 9.1 (0.2-19.0), and 5.1 (0.5-15.3) mg L^{-1}, respectively. This oxidative component was most likely a very important aspect of the high N removal capacity of these wetland cells. Hunt (unpublished data, 1994) found that wetland cells used for treatment of swine were nitrate-limited for denitrification. Thus, O_2 to support nitrification is very important for the treatment capacity. The treatment efficiency of the wetlands and the grass filter strips for concentration and mass are presented in Table 5. The NH_3-N and *o*-PO_4-P loading rates were very high. Yet, the wetlands gave 71% and 44% mass removal for NH_3-N and *o*-PO_4-P, respectively; and the combined mass treatment efficiency was > 95% for all parameters. This was extremely good wastewater treatment. Two hundred and fifty days at this loading rate would remove 3.5 Mg N ha^{-1} yr^{-1}. This removal rate is about 10 times greater than that expected for forage grasses.

Hubbard et al. (1994) investigated the use of overland flow in vegetative strips composed of varying lengths of grass and trees for the treatment of swine wastewater by overland flow. However, the permeability of their soils produced a combination of overland and lateral subsurface flow. Their work showed the substantial nutrient assimilative capacities of the grass filter strips and wooded riparian zones. Overland flow has long been known to be an effective method of treating cannery wastewater and municipal wastewaters, and its processes are somewhat similar to those of wetlands (Hunt and Lee, 1976; Peters et al., 1981) . Thus, overland flow strips may be very good pre- or post-wetland components of functional and sustainable treatment systems.

In Alabama, McCaskey et al. (1994) were concerned with the BOD-based loading rate of 67 kg of BOD ha^{-1} day^{-1} because this guideline was developed for municipal wastewater, which has a relatively low BOD:NH_3-N ratio (McCaskey et al., 1994; USDA-SCS, 1991 and 1992a,b). They hypothesized that swine wastewater loading rates would be limited by NH_3-N, and they conducted

Table 5. Reductions of Waste Components on Both a Concentration and Mass Basis (reproduced by permission from Cathcart et al., 1994).

	Wetland		Vegetated Strip*		Overall	
	Conc.	Mass	Conc.	Mass	Conc.	Mass
	%					
BOD_5	51	54	54	92	76	96
NH_3-N	66	71	83	97	94	99
T-PO_4-P	39	44	53	91	71	95
SS	63	69	35	91	77	97

* Vegetated strip reductions are based upon effluent from the constructed wetlands. These values are considered upper estimates because they are based on grab samples and may have missed discharges related to storm events.

experiments with three loading rates in three, two-stage wetland cells. Wastewater was generated from a 500-pig yr^{-1} farrow-to-finish facility. It was initially treated in two-stage lagoons and diluted to obtain an NH_3-N concentration of about 95 mg L^{-1}. Hydraulic loading rates were altered to obtain 11.0, 4.6, and 2.3 kg BOD ha^{-1} day^{-1}. The overall treatment efficiency of the wetland was affected by loading rate with the lowest rate giving over 94% removal for all parameters. However, this loading rate of N (2.6 kg ha^{-1} day^{-1}) was only 2 to 4 times greater than that used with forage grass. At the highest loading rate, NH_3-N and total P were applied at 11.5 and 8.0 kg ha^{-1}, respectively. The 89% and 79% mass removals of N and P, respectively, at the highest loading rate were very encouraging. They are in general agreement with those obtained in Mississippi, and addition of a vegetative strip or some other final polishing step would likely give very high mass removals for all rates.

The focus of our research in North Carolina 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., 1994; Szõgi et al., 1994). 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 and 10 kg ha^{-1} day^{-1} were used in the first and second years, respectively. Mass removal averaged 90% for N and 80% for P. Above-ground dry matter production for rush/bulrushes and bur-reed/cattails was 12 and 33 Mg ha^{-1}, respectively. Flooded rice yield was 4.5 Mg ha^{-1}, and soybean cultivar Young that was grown in saturation culture yielded 2.8 Mg ha^{-1} (Nathanson et al., 1984; Cooper et al., 1992a). Redox conditions were highly anaerobic in the soils of all wetland cells in summer. 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. Phosphorus mass removal decreased substantially in the second year.

The combined results of these studies on swine wastewater treatment suggest that constructed wetlands are excellent for mass removal of N and P. 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. Crop lands, vegetative strips, and woodlands are viable options for the final treatments. Terminal land application does not require discharge permits and monitoring of discharge water quality. Therefore, it is an approach that fits well with the capacities of constructed wetlands. The mass removal of N and P can likely be increased by pre- and post-wetland treatment of wastewater, and research on several of these possibilities is currently under way.

SUMMARY

Animal waste treatment is a significant agricultural and environmental problem that is growing rapidly as a result of expanded, confined animal production. Large numbers of production facilities in watersheds and river basins necessitate functional and sustainable treatment of wastewaters. Wetlands have been used successfully for municipal wastewater treatment, and they have potential for the treatment of animal wastewater.

However, the strategies for successful wetland treatment of municipal wastewater and animal wastewaters are different. In municipal wastewater treatment, the systems are operated to meet stream discharge requirements, and strict monitoring of discharge water is required. Currently, direct discharge of animal wastewater is not allowed. Thus, reduction of contaminant mass is the desirable and achievable goal for constructed wetland treatment of animal wastewater. Constructed wetlands can transform and assimilate large quantities of C, N, and P from wastewater. Hence, they dramatically reduce the mass per unit area of contaminant that must be applied to the terminal land treatment site.

Animal wastes are generally high in solids, and solids removal prior to wetlands is essential for their long-term functionality. Constructed wetlands have successfully reduced N and P from dairy and swine wastewater. When they were combined with grass filter strips, an application rate of 14 kg ha^{-1} day^{-1} of N was treated with over 95% removal. Cattails and bulrushes have been the most commonly used wetland plants. Both plants assimilate large quantities of nutrients, but bulrushes transport more O_2 and produce more oxidized sediment. Open water strips have been used in marsh/pond/marsh designs for improved oxidation and nitrification. Denitrification of swine wetlands is generally nitrate-limited; wetlands commonly have sufficient denitrifying population to produce enzymes, C to provide microbial energy, and anaerobic conditions to promote denitrification. However, the reduced condition of wetland sediments limits the rate of nitrification. Any pre- or in-wetland procedures that promote oxidation and nitrification will increase denitrification dramatically. Such procedures offer the greatest potential for improved treatment capacity for constructed wetland treatment of animal wastewater.

Phosphorus removal is somewhat limited by the P adsorption characteristics of the wetland soil and plant litter layers as well as the reduced condition of wetlands. However, addition of iron or aluminum to the wetlands may have potential for improving treatment efficiency. Additionally, treated effluent can be passed through filter media with iron-fortified peat or other P-removing media before discharge.

Ultimately, the necessary treatments will depend upon the amount of land available for wastewater discharge. Where land is very limited and the operator is concerned with the longevity of the terminal land treatment site, high percentages of the wastewater nutrient load need to be removed in the wetland. Here it is likely that pre- and post-wetland treatments will be necessary because C must be removed, NH_3-N must be nitrified, and P must be removed; and these processes are too varied and complex to proceed optimally in the same simple wetland. However, if treatment steps and wetland cells are properly sequenced, wetlands have the potential for very high levels of mass removal.

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