CONSTRUCTED WETLANDS FOR ANIMAL WASTE WATER TREATMENT

P. G. Hunt, M. B. Vanotti, A. A. Szögi
USDA-ARS Coastal Plains Soil, Water
and Plant Research Center
Florence, SC

F. J. Humenik, and J. M. Rice
North Carolina State University
Biological and Agricultural Engineering Department
Raleigh, NC

ABSTRACT

Confined animal production generates enormous per-unit-area quantities of waste. Waste-water from dairy and swine operations has been successfully treated in constructed wetlands. Plants are an integral part of wetlands. Cattails and bulrushes are commonly used in constructed wetlands for their capacity to transport oxygen to the 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 microbial populations, carbon sources, and anaerobic conditions to promote denitrification processes. However, the anaerobic conditions of wetland sediments limit the rate of nitrification. Thus, denitrification of animal waste waters in wetlands is generally nitrate-limited. Phosphorus (P) removal is also somewhat limited by the anaerobic conditions of wetlands. Therefore, when very high mass removal of nitrogen (N) and P is required, pre- or in-wetland procedures that promote oxidation are needed to increase treatment efficiency. An overland flow treatment removed 59% of ammonia-N applied in swine waste water at a 50-kg ha^{-1} d^{-1} rate, and 10% was recovered as nitrate-N. A media filter treatment transformed up to 32% of the inflow total N into nitrate when waste water was recycled four times. Immobilization of nitrifying microorganisms in polymer pellets resulted in faster nitrification rates and smaller reactors. The rate of nitrification using these pellets was 600 g of ammonia-N m^{-3} d^{1}. Solids removal prior to wetlands treatment is also essential for long-term functionality. When wetlands were combined with grass filter strips, an application of swine waste water containing 14 kg ha^{-1} day^{-1} of ammonia-N was treated to over 95% removal. Polyacrylamide is effective for separating nutrients before the waste enters typical lagoon treatment. Low rates (25-100 mg L^{-1}) removed 80% of suspended solids, organic N, and organic P from swine flushing effluents.

INTRODUCTION

Animal waste treatment is a significant agricultural and environmental challenge that will need additional options as a result of expanded, confined animal production. Large numbers of production facilities in watersheds and river basins necessitate functional and sustainable treatment of waste water. Wetlands have been used successfully for municipal waste water treatment. New evidence shows that they have potential for the treatment of animal waste water.

Strategies for successful wetland treatment of animal waste water are different from municipal waste water treatment. In municipal waste water treatment, the systems are operated to meet stream discharge requirements. Currently, direct discharge of animal waste water is not allowed. Constructed wetlands can transform and assimilate large quantities of carbon (C), nitrogen (N), and phosphorus (P) from waste water. Hence, they can reduce the amount of nutrients that must be applied to the terminal land treatment site.

Ultimately, the necessary treatments will depend upon the amount of land available for terminal management. Where land is limited, high percentages of the waste water nutrient loads need to be removed in the wetland. Here it is likely that pre- and post-wetland treatments will be necessary because C and P must be removed, and ammonia N (NH_3-N) must be nitrified. If treatment steps and wetland cells are properly sequenced, wetlands have the potential for high levels of nutrient mass removal.

WETLAND FUNCTIONALITY

Two types of wetlands (subsurface and free-water-surface-flow) are typically used (Hammer, 1989). Subsurface systems are subject to clogging and limited oxygen (O_2) diffusion. Free-water-surface-flow systems are more suitable for animal waste water treatment. Gumbricht (1993) categorized free-water-surface systems into free-floating-plant ponds, submersed-plant ponds, and constructed wetlands with emergent plants.

Plants. Emergent plants are usually used as a component in constructed wetlands for waste water 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). Wetland plants transport O_2 from leaves and stems to roots (Armstrong, 1964), providing an oxidized microenvironment in the anaerobic root zone. The efficient use of wetlands for waste water treatment depends on the O_2 transport capacity of the plant-root system and its diffusion across the free soil-water interface. Oxygen availability is also affected by the O_2 demand of the wetland. Waste waters 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 waste waters may require dilution, aerobic pre-treatment, or low loading rates.

Organic Carbon. Carbon removal is an essential aspect of N removal in wetlands because high C levels increase the BOD and promote the growth of heterotrophic organisms. Low O_2 levels will decrease nitrification because it is carried out aerobically by autotrophic bacteria. 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 and nutrients can be removed from waste water by solid-liquid separation using dewatering presses and screens. Solids removal can also be enhanced by increased residence time in a settling basin. Both flocculation and precipitation can be increased by polyacrylamine (PAM) addition. Separated solids can be spread directly on the land or composted. Reduction of the C load that remains in the waste water can be accomplished in an anaerobic lagoon. If further C reduction is needed, the waste water 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, waste waters can be run through an alternating sand filter-media filtration system or across an overland flow treatment site to obtain nitrification.

Nitrogen. 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 carbon dioxide (CO_2) (Mikkelsen et al., 1978; Reddy and Graetz, 1981). It might even be possible to manage waste waters to promote NH_3-N loss by volatilization. However, NH_3-N can be absorbed by the surrounding ecosystems (i.e., surface waters, cropland, pasture land, 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 or by anaerobic microorganisms and converted back to biomass 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 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 nitrate by bacteria (or nitrification) occurs (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 waste water application or very high evapotranspiration. Under wetland conditions, nitrate diffuses from the aerobic soil layer to the anaerobic zone and undergoes microbial reduction to nitrous oxide (incomplete denitrification) and molecular N (complete nitrification) (Reddy et al., 1980). Denitrification 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. Organic P may comprise a substantial reservoir in wetland soils 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 iron and aluminum 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 oxides that become highly soluble under prevalent-reduced conditions. This increased P solubility explains why wetlands treated with waste water can become rapidly P-saturated and export excessive quantities of P in a few years (Richardson, 1985).

PREVIOUS STUDIES OF DAIRY WASTE WATER TREATMENT

Constructed wetlands in Lagrange Co., Indiana, were used to treat waste water from a 70-milk-cow herd (Reaves et al., 1994). Barnyard runoff, milkhouse waste water, 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 x 61 m with a bottom slope of 1% with a clay liner. The predominant vegetation was cattails (*Typha latifolia*), smartweed (*Polygonum* spp.), and reed canary grass (*Phalaris arundinacea*). The wetland treatments reduced nutrient loads of nearby Lake Appleman (Table 1), but excessive solids loading limited their potential. A 10-cm-thick layer of solids accumulated in the initial third of each wetland cell during the first year of operation. These results emphasize the need for solids removal pre-treatment.

The constructed wetland at the Oregon State University dairy farm (Skarda et al., 1994) had six 4.6- x 29-m wetland cells operating in parallel with a 30-cm water depth. The dairy used a recycling flush system with solids removed, waste water 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. Maximum levels of NH_3-N and total solids (TS) entering the wetlands were 100 mg L^{-1} and 1.5 g L^{-1}, respectively. Maximum BOD load was 74 kg ha^{-1} d^{-1}.

Table 1

Performance of four constructed wetlands systems for dairy waste water treatment (adapted from DuBowy and Reaves, 1994; Cooper et al., 1992; Hunt et al., 1995b).

Parameter	Indiana	Oregon	Mississippi	Kentucky
	---------- % reduction	----------	----------	----------
BOD_5	79	61	75	66
TSS	72	73	64	88
TKN	64	57		37
NH_3-N	64	54	90	87
TP	74	66	61	59

The constructed wetland at the DeSoto Co., Mississippi dairy farm (Cooper et al., 1992) had three 6.1- x 24.4-m wetland cells operating in parallel that received waste water from a 41.1- x 51.8-m primary lagoon. Total waste entering the lagoon was 10.3 m^3 day^{-1} and included barnyard runoff from a 352-m^2 cattle holding lot, rainfall from building roofs, and milkhouse waste water. Data in Table 1 (Mississippi) shows average performance of cells 1 and 3. Primary treatment and dilution resulted in a lower influent BOD, and a wetland-treated effluent BOD < 30 mg L^{-1}.

The constructed wetland at the Mercer Co., Kentucky dairy farm (Hunt et al., 1995b) had two 9.1- x 24.4-m cells in series that were planted with cattails. The waste water 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 wetland. Influent waste water characteristics (mg L^{-1}) were: BOD= 452, total suspended solids (TSS)= 1132, TP= 72, TKN= 108, and NH_3-N= 33. The settling basin removed about half of the volatile solids and total nitrogen but it did not reduce the BOD. Data in Table 1 shows performance of the first wetland during the first year. A rock-filled trench allowed discharge to flow over a fescue vegetative filter strip (18.3 m x 53.4 m) as a final nutrient removal component. This system has functioned well for nutrient mass removal.

PREVIOUS STUDIES OF SWINE WASTE WATER TREATMENT

In Mississippi, Cathcart et al. (1994) studied a marsh/pond/marsh constructed wetland system for the treatment of swine waste water. It contained two, parallel 0.04-ha constructed wetlands in 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 was unplanted. The combination of depth and turbidity has restricted emergent plant encroachment in the pond section. The waste water flowed from a facultative lagoon that primarily treated waste water from a farrowing house. Waste water 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 orthophosphate (o-PO_4-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 waste water 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 et al. (1995a) 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 2. 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. If this level of removal can be sustained over a 250-day period, such a system would remove 3.5 Mg N ha^{-1} yr^{-1}. This removal rate is about 10 times greater than that expected for forage grasses.

Table 2
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.

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 waste water by overland flow. 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 waste water and municipal waste waters, and its processes are somewhat similar to those of wetlands (Hunt and Lee, 1976). 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 waste water, which has a relatively low BOD:NH_3-N ratio. They hypothesized that swine waste water loading rates would be limited by NH_3-N, and they conducted experiments with three loading rates in three, two-stage wetland cells. Waste water 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.

SWINE WASTE WATER TREATMENTS AT KENANSVILLE

Site Characteristics

The research site is located in Duplin Co., NC. It has a nursery operation of 2600 pigs (average weight = 13 kg) that uses a flushing system recirculating lagoon liquid to keep the house clean and a single-stage lagoon for primary treatment. The average liquid volume of the lagoon is 4100 m^3 and the average total N inflow is 16.8 kg N d^{-1}. On a mass basis, 83% of N entering the lagoon is lost by ammonia volatilization or denitrification, 13% remains in the lagoon effluent, and 4% remains in the settled sludge (Szögi et al., 1996). Typically, the lagoon effluent contains 365 mg L^{-1} of TKN, mostly (> 95%) as ammonia, 93 mg L^{-1} TP, 1.9 g kg^{-1} TS, and 740 mg L^{-1} COD.

Constructed Wetland

The focus of our research was to determine wetland treatment efficiency of swine waste water and to define redox conditions, denitrification potentials, and agronomic cropping potentials of constructed wetlands used for swine waste water treatment (Hunt, et al., 1994; Szögi et al., 1994). Three sets of two 3.6- by 36-m wetland cells were constructed adjacent to the lagoon in 1992; they contained either natural wetland plants or water-tolerant agronomic plants (Rice and soybean). Nitrogen loading rates of 3 and 10 kg ha^{-1} day^{-1} were used in the first and second years, respectively, by mixing the lagoon effluent with fresh water before it was applied to the wetland. The low rate was established to investigate if stream discharge requirements could be achieved with just a lagoon-wetland system. Nutrient removal efficiency was similar in both rush/bulrushes and bur-reed/cattails plant systems (Table 3). Average mass removal of N was 94% at the loading rate of 3 kg N ha^{-1} day^{-1}, but insufficient for stream discharge requirements. Nitrogen removal decreased to 68% at the higher rate of 10 kg N ha^{-1} day^{-1}. Phosphorus mass removal efficiencies ranged from 40 to 100% at P loading rates < 1 kg P ha^{-1} day^{-1} and varied from 20 to 80% when P loading rates were 1 to 4 kg ha^{-1} day^{-1} (data not shown). 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 grown in saturated culture yielded 2.8 Mg ha^{-1}. Redox conditions were highly anaerobic in the soils of all wetland cells in summer, but 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 (Hunt et al., 1995a).

Table 3
Nitrogen loading rates and mass removal efficiencies for the constructed wetlands, Duplin Co., NC (June 1993 - January 1995). Data from Szögi et al. (1995).

Nitrogen Loading rate†	System	Mass Removal ‡ %
3 kg ha^{-1} d^{-1}	Rush/bulrush	94
	Cattails/bur-reed	94
10 kg ha^{-1} d^{-1}	Rush/bulrush	63
	Cattails/bur-reed	73

† Expressed as TN.
‡ % Mass Removal = % mass reduction of N (TN = NH_3-N + NO_3-N) in the effluent with respect to the nutrient mass inflow.

These results suggest that constructed wetlands are excellent for mass removal of N from swine waste water. However, 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 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 capacity of mass N removal by wetlands can likely be increased by pre-wetland treatment of waste water such as overland flow, media filtration, or the use of encapsulated nitrifier technology.

Overland Flow

In overland flow, nitrification occurs when a thin film of water is in close contact with the nitrifying population at the soil surface. It also offers the advantage of partial denitrification of NO_3-N in the underlying saturated soil layer (Hunt and Lee, 1976). An overland flow treatment consisting of a 4-×20-m plot with 2% slope was constructed next to the lagoon in summer 1995 (Szögi et al., 1996). The sides and bottom of the overland flow plot were lined with plastic and covered with 20 to 30 cm of sandy loam soil. The plot was planted to a mixture of fescue, coastal bermudagrass, and reed canary grass. Waste water was pumped from the lagoon to a storage tank, and applied as a fine spray with fixed sprinkler heads over the first 6 m of the overland flow plot. Waste water was applied for eight hours each day, five days a week, and surface and subsurface flow were collected at the end of the plot. The overland flow system had a hydraulic loading rate of 2.5 cm d^{-1}, which resulted in high rates of ammonia application (54 kg NH_3-N ha^{-1} d^{-1}). The high loading rate was necessary to obtain measurable surface flow at the end of the plot, but this practice is not uncommon in overland flow systems (Hegg and Turner, 1983). Data in Table 4 show that the overland flow treatment was effective in reducing the concentration of TSS, COD, TKN, and NH_3-N in both the surface and subsurface effluents. The increase in nitrate-N concentration in both effluents is an indication that nitrification occurred. The total ammonia-N removal efficiency was 59%, and 10% of the total ammonia N application was recovered in the effluent as nitrate-N. Humenik et al. (1975) reported a TN efficiency of 35% for swine lagoon waste-water treated on 17-m overland flow plots with a hydraulic load of 1.8 cm d^{-1}. The total mass P removal efficiencies varied from 40 to 80% at loading rates as high as 38 kg P ha^{-1} d^{-1}.

Table 4
Characteristics of the influent waste water and overland flow effluent separated in surface and subsurface flow (mean ± one standard error from August to December 1995).

Constituent	Unit	Influent Waste Water	Surface Effluent	Subsurface Effluent
Total Suspended Solids	g L^{-1}	0.25	0.21	0.11
Chemical Oxygen Demand	mg L^{-1}	446 ± 28	372 ± 33	207 ± 19
Total Kjeldahl Nitrogen	mg L^{-1}	250 ± 12	128 ± 16	78 ± 8
Ammonia Nitrogen	mg L^{-1}	198 ± 5	110 ± 6	73 ± 5
Nitrate Nitrogen	mg L^{-1}	0.2	24 ± 3	30 ± 3
pH		8.4	8.3	8.2

Media Filter

Recirculating media filters or trickling filters are popular among small waste generators, especially where soil conditions are not suitable for subsurface disposal systems (Rubin et al., 1994). To avoid the risk of clogging by lagoon liquid, our filtration unit used marl gravel instead of typical sand media. This modification enhanced natural aeration, and also provided rapid vertical flow rates and some phosphorus sorption. The unit consisted of a 1.8-m-diameter×0.9-m-height tank filled with the marl gravel and placed inside a second tank collecting the effluent for recirculation. Lagoon waste water was applied as a fine spray on the surface at hydraulic loading rates of 75 L m^{-2} h^{-1} and TN loading rates of 147 g m^{-2}.

Table 5
Characteristics of the influent waste water and media filter treated effluent (mean ± one standard error from August 16-23, 1995).

Constituent	Unit	Influent Waste Water	One cycle Effluent	Four cycles Effluent†
Total Suspended Solids	$g\ L^{-1}$	0.52 ± 0.11	0.26 ± 0.02	0.15 ± 0.02
COD	$mg\ L^{-1}$	869 ± 117	432 ± 35	403 ± 40
Total Kjeldahl Nitrogen	$mg\ L^{-1}$	327 ± 16	257 ± 14	180 ± 9
Ammonia Nitrogen	$mg\ L^{-1}$	236 ± 19	182 ± 14	147 ± 7
Nitrate Nitrogen	$mg\ L^{-1}$	6 ± 2	40 ± 5	80 ± 4
pH		8.6	8.5	8.4

† A total volume of 1100 L of lagoon liquid was passed once through the media filter (one cycle) and then recirculated three more times (four cycles).

About 50% of TSS and COD were removed from waste water with just one cycle, and losses of TN were 11% with one cycle and 22% with four cycles (Table 5). With four cycles, 24 to 32% of the influent TKN was converted to NO_3-N. Our results with lagoon swine waste water are in agreement with the expected performance of home sewage treatment units (Hines and Favreau, 1975; Mote et al., 1991) showing that media filters can provide excellent carbon and suspended solids removal as well as a high degree of nitrification.

Nitrification Enhancement with Encapsulated Bacteria

A possible new technology for efficient nitrogen treatment is the use of encapsulated nitrifying microorganisms (Vanotti and Hunt, 1996a). This technology is widely used in drug manufacturing and food processing. Its application to municipal waste water treatment has been recently developed in Japan. With wastes rich in carbonaceous materials, such as swine waste water, the nitrifying bacteria compete poorly with heterotrophic microorganisms. Nitrifiers need lower carbon, oxygen, a surface area, and a growth phase before sufficient numbers are present for effective nitrification of swine waste water. These conditions can be provided by immobilization. The nitrifiers are entrapped in pellets (cubes) made of polyethylen glycol (PEG) or polyvinyl alcohol (PVA) polymers that are permeable to ammonia, oxygen and CO_2 needed by these microorganisms, resulting in faster nitrification rates and smaller reactors. This is an important consideration for farming systems because aeration cost can be a limiting factor.

Nitrifying pellets were produced by a PVA-freezing method. An active culture of acclimated swine waste water nitrifying bacteria was first prepared using sludge seed from the overland flow treatment. The cells were successfully immobilized in 3-5 mm PVA polymer cubes. Using aeration only in batch experiments, nitrification of lagoon liquid started at 10 days and 69% of NH_4-N was lost by ammonia volatilization. In contrast, the waste water NH_4-N was completely oxidized in one day with the addition of nitrifying pellets and no NH_3 was lost. Ammonia removal potential of nitrifying pellets were evaluated under continuous flow treatment (Vanotti and Hunt, 1996b). Pellets were added at 15.3% (w/v) [17.5% (v/v)] pellet to total tank volume ratio. Ammonia loading rates were gradually increased from 194 to a maximum of 1287 mg NH_4-N per L of aeration tank per day (1.27 to 8.40 mg NH_4 -N g-pellet $^{-1}$ d^{-1}, respectively). Loading rates were changed by decreasing the hydraulic residence time (HRT) from 28 h to 4 h. Ammonia removal efficiencies of more than 90% were obtained with ammonia loading rates of 2.73 mg N g-pellet^{-1} d^{-1} and HRT of 12 h [influent NH_4-N=218 ppm, the influent (nitrate+nitrite)-N=0 ppm; effluent NH_4-N=27 ppm, effluent (nitrate+nitrite)-N=199 ppm]. Removal efficiencies decreased to about 47% at the highest rate of 8.4 mg N g-pellet^{-1} d^{-1} (HRT = 4 h). All ammonia-N removed was quantitatively converted into nitrate and nitrite forms. Nitrate was predominant (> 85%) at HRT higher than 12 h, while equal amounts of nitrate and nitrite were produced at the 4 h HRT (highest load). The rate of nitrification of swine waste water obtained with HRT of 4 h was 3.94 mg of NH_4-N g-pellet^{-1} d^{-1} (0.6 kg NH_4-N m^{-3} d^{-1}). On the other hand, an operation with 1000 pigs (avg. pig weight = 120 lb) needs to dispose of about 3.2 kg N per day after lagoon treatment. Thus, a small reactor with 5.3 m^3 volume (~ 1.5 x 2.0 x 1.8 m) is required to nitrify this effluent.

Solids and Nutrient Separation Using Pam

Liquid/solids separation before the manure solids enter the lagoon provides new alternatives for manure management. The high nutrient concentration and low moisture of the separated solid fraction make this fraction a valuable material. It can be temporarily stored, processed for refeeding or transported to nutrient deficient cropland. The total nutrient load to be treated is drastically reduced, thereby increasing the life of existing lagoons. When combined with more sophisticated process designs, it can allow the use of advanced treatment methods that may eventually replace the anaerobic lagoon. But separation of suspended solids and associated nutrients from flushed waste waters using mechanical separators is very inefficient (5 to 20%) without some type of chemical coagulation to bind together the small particles of solids into larger clumps.

Our evaluation of polyacrylamide (PAM) flocculants (Vanotti et al., 1996) showed that these compounds are effective for flocculating suspended solids and separating nutrients from flushing effluents. In this work, we tested 30 PAMs having a wide range of charge type and density. We found that cationic PAMs with low charge density are the best for swine waste water applications. Removal efficiencies of about 80% of total suspended solids, organic N and organic P were obtained with PAM rates as low as 25 ppm for waste waters having a total solids concentration of 3200 ppm, and PAM rates of 100 ppm for waste waters with 6800 ppm of total solids. It is anticipated that the removal of nutrients associated with the solids will be of great benefit to swine producers whose lagoons are of nominal size and who have limited acreage available for land application of the lagoon liquids.

ACKNOWLEDGMENTS

This research has been funded by USEPA Project No. CR823808-01-0 'Evaluation of Alternative Constructed Wetland Systems for Swine Waste water Treatment'; USDA-CSREES Project No. 95-34214-2392 'Management Practices to Reduce Nonpoint Source Pollution on a Watershed Basis'; and NC Herrings Marsh Run Water Quality Demonstration Project No. 90-EWQD-1-9504.

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

Cooper, C.M., S. Testa, III, and S.S. Knight. 1992. Evaluation of ARS and SCS constructed wetland/animal waste treatment project at Hernando, Mississippi. ARS-USDA, National Sedimentation Laboratory, Technology Application Project Report No. 17. 28 p.

DuBowy, P.J., and R.P. Reaves (eds.) 1994. Constructed Wetlands for Animal Waste water Management - Workshop Proceedings. Purdue University, West Lafayette, IN. 188 p.

Faulkner, S.P., and C. J. Richardson. 1989. Physical and chemical characteristics of freshwater wetland soils. Pp. 41-72. In D.A. Hammer (ed.) Constructed Wetlands for Waste water Treatment - Municipal, Industrial and Agricultural. Lewis Publishers.

Gale, P.M., K.R. Reddy, and D.A. Graetz. 1994. Phosphorus retention by wetland soils used for treated waste water disposal. J. Environ. Qual., 23:370-377.

Gumbricht, T. 1993. Nutrient removal processes in freshwater submersed macrophyte systems. Ecol. Eng., 2:1-30.

Hammer, D.A. (ed.) 1989. Constructed Wetlands for Waste Water Treatment - Municipal, Industrial, and Agricultural. Lewis Publishers, Chelsea, MI. 831 p.

Hegg, R.O., and A.K. Turner. 1983. Overland flow as a method of treatment for animal wastes - A review. Agricultural Wastes 8:167-184.

Hines, M., and R.E. Favreau. 1975. Recirculating sand filter: an alternative to traditional sewage absorption systems. Pp. 130-136. In Proceedings of the National Home Sewage Disposal Symposium. ASAE Pub. Proc.-175. St. Joseph, MI.

Hubbard, R.K., G.L. Newton, J.G. Davis, R. Dove, R. Lowrance, and G. Vellidis. 1994. Grass-riparian zone buffer systems for filtering swine lagoon waste. Pp. 124-143. In P.J. DuBowy and R.P. Reaves (eds.) Constructed Wetlands for Animal Waste Management. Purdue Research Foundation, West Lafayette, IN.

Humenik, F.J., R.E. Sneed, M.R. Overcash, J.C. Barker, and G.D. Wetherill. 1975. Total waste management for a large swine production facility. Pp. 68-71. In Proceedings of the Third International Symposium on Livestock Wastes. ASAE, St. Joseph, MI.

Hunt, P.G., and C.R. Lee. 1976. Land treatment of waste water by overland flow for improved water quality. Pp.151-160. In J. Tourbier and R.W. Pierson, Jr. (eds.) Biological Control of Water Pollution. University of Pennsylvania Press, PA.

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

Hunt, P.G., T.A. Matheny, A.A.Szögi, and F. J. Humenik. 1995a. Denitrification in constructed wetlands for swine waste water treatment. Pp. 329. Agronomy Abstracts.

Hunt, P. G., W. O. Thom, A. A. Szögi, and F. J. Humenik. 1995b. State of the art for animal waste water treatment in constructed wetlands. Pp. 53-65. In C. C. Ross (ed.) Proceedings of the Seventh International Symposium on Agricultural and Food Processing Wastes (ISAFPW95). June 18-20, Chicago, IL. ASAE, St. Joseph, MI.

McCaskey, T.A., S.N. Britt, T.C. Hannah, J.T. Eason, V.W.E. Payne, and D.A. Hammer. 1994. Treatment of swine lagoon effluent by constructed wetlands operated at three loading rates. Pp. 23-33. In P.J. DuBowy and R.P. Reaves (eds.) Constructed Wetlands for Animal Waste Management. Purdue Research Foundation, West Lafayette, IN.

Mikkelsen, D.S., S.K. De Datta, and W.N. Obcemea. 1978. Ammonia volatilization losses from flooded rice soils. Soil Sci. Soc. Am. J., 42:725-730.

Mote, C.R., J.W. Kleene, and J.S. Allen. 1991. On-site waste water renovation utilizing a partially saturated sand filter for nitrogen removal and lawn sprinkler for effluent discharge. Pp. 173-181. In Proceedings of Sixth On-Site Waste Water Treatment Conference. ASAE Pub. 10-91. St. Joseph, MI.

Nichols, D.S. 1983. Capacity of natural wetlands to remove nutrients from waste water. J. Water Pollut. Contr. Fed., 55:495-505.

Parfitt, R.L. 1978. Anion adsorption by soils and soil materials. Adv. Agron., 30:1-50.

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

Reaves, R.P., P.J. DuBowy, and B.K. Miller. 1994. Performance of a constructed wetland for dairy waste treatment in Lagrange County, Indiana. Pp. 43-52. In P.J. DuBowy and R.P. Reaves (eds.) Constructed Wetlands for Animal Waste Management. Purdue Research Foundation, West Lafayette, IN.

Reddy, K.R., E.M. D'Angelo, and T.A. DeBusk. 1989. Oxygen transport through aquatic macrophytes: The role in waste water treatment. J. Environ. Qual., 19:261-267.

Reddy, K.R., and D.A. Graetz. 1981. Use of shallow reservoirs and flooded soil systems for waste water treatment: nitrogen and phosphorus transformations. J. Environ. Qual., 10:113-119.

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

Reddy, K.R., W.H. Patrick, Jr., and R.E. Phillips. 1980. Evaluation of selected processes controlling nitrogen loss in a flooded soil. Soil Sci. Soc. Am. J., 44:1241-1246.

Richardson, C.J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science, 228:1424-1427.

Rubin, A.R., S. Greene, T. Sinclair, and A. Jantrania. 1994. Performance evaluation of drip disposal system for residential treatment. Pp. 467-474. In Proceedings of the Seventh International Symposium on Individual and Small Community Sewage Systems. ASAE Publication 18-94. St. Joseph, MI.

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

Syers, J.K., R.F. Harris, and D.E. Armstrong. 1973. Phosphate chemistry in lake sediments. J. Environ. Qual., 2:1-14.

Szögi, A.A., P.G. Hunt, F.J. Humenik, S.W. Broome, and J.M. Rice. 1996. Assessment of constructed wetlands for mass removal of nitrogen from swine waste water. In Proceedings of 2nd. National Workshop on the Use of Constructed Wetlands for Animal Waste Management. May 15-18, 1996. Forth Worth, TX. Texas A&M University.

Szögi, A.A., P.G. Hunt, F.J. Humenik, K.C. Stone, and J.M. Rice. 1995. Constructed wetlands for swine waste water treatment in the eastern Coastal Plain, USA. Pp. 675-682. In K.L. Campbell (ed.) Versatility of Wetlands in the Agricultural Landscape. ASAE Publication 70IP0995, St. Joseph, MI.

Szögi A.A., P.G. Hunt, F.J. Humenik, J.C. Stone, J.M. Rice, and E.J. Sadler. 1994. Seasonal dynamics of nutrients and physico-chemical conditions in a constructed wetland for swine waste water treatment. Paper No. 942602 presented at the International ASAE Winter Meeting, December 13-16, 1994. ASAE, St. Joseph, MI.

Vanotti, M.B., and P.G. Hunt. 1996a. The use of polymers for nitrogen removal in swine waste water: PAM and encapsulated nitrifier technologies. Pp. 116-120. Proc. Solutions: A technical conference on water quality. March 19-21, NC State Univ., Raleigh, NC.

Vanotti, M.B., and P.G. Hunt. 1996b. Enhanced nitrification of swine waste water with immobilized nitrifiers. P. 34. In Agronomy Abstracts. ASA. Madison, WI.

Vanotti, M.B., P.G. Hunt, and A.A. Szogi. 1996. Solids and nutrient removal from swine waste water using polyacrylamides. ASAE Paper 96-4052. Phoenix, AZ. 10 p.