Reprinted from the *Journal of Environmental Quality*
Volume 29, no. 6, Nov.-Dec. 2000.
677 South Segoe Rd., Madison, WI 53711 USA

A Water Chemistry Assessment of Wastewater Remediation in a Natural Swamp

Xiaowei Zhang, Sam E. Feagley,* John W. Day, William H. Conner, Irene D. Hesse, John M. Rybczyk, and Wayne H. Hudnall

ABSTRACT

Various aspects of water chemistry of a natural forested wetland were studied in order to determine the possibility of using the wetland for advanced wastewater treatment in Louisiana and to evaluate the wastewater effect on swamp water quality. The study was carried out by comparing treatment and control cypress–tupelo forests separated by a bottomland hardwood forest ridge. The treatment area (231 ha) received secondarily treated municipal wastewater at a rate of 6.3×10^6 L d^{-1}. The calculated hydraulic detention time of the wastewater was 120 d. Electrical conductivity (EC), pH, and concentrations of 5-d biological oxygen demand (BOD_5), dissolved oxygen (DO), solids, nutrients, and trace metals were monitored. Mean concentrations for the wastewater were 14.6 mg L^{-1} for total N and 2.5 mg L^{-1} for total P. The dominant form of N in the wastewater was NO_3–N. The swamp system attenuated the NO_3–N by 100%, total Kjeldahl nitrogen (TKN) by 69%, and total P by 66%. It appears that tertiary wastewater treatment was achieved due to the nutrient attenuations. Based upon our findings, we predict that the high N attenuation efficiency would enable the swamp to work well if the N loading rate were doubled. However, P removal was dependent on loading rate, hydraulic retention time, and temperature. The swamp was more efficient in treating wastewater during warm seasons than cool seasons. During the monitoring period, trace metals were not significantly increased in the swamp water because of very low concentrations in the wastewater.

THE application of secondary treatment plant wastewater to wetlands can reduce problems of direct discharge into surface waters. Studies have shown that wetlands chemically, physically, and biologically remove high amounts of pollutants and nutrients from input effluent (Godfrey et al., 1985; Reddy and Smith, 1987; USEPA, 1994; Kadlec and Knight, 1996). The efficiency of removal is determined by loading rate, retention time, and interaction of soil, water, vegetation, and microorganisms (Richardson and Nichols, 1985; Faulkner and Richardson, 1989; Breaux and Day, 1994). In Louisiana, natural wetlands are legally limited to receiving only secondary effluent, and only after approval on a case by case basis (Breaux and Day, 1994). More case studies should be undertaken to assess the overall feasibility of natural wetlands to function as a tertiary treatment process.

X. Zhang, Detroit Water and Sewerage Dep., Detroit, MI 48201; S. Feagley, Soil and Crop Sciences Dep., Texas A&M Univ., College Station, TX 77843-2474; J. Day, Dep. of Oceanography and Coastal Sciences, Louisiana State Univ., Baton Rouge, LA 70803; W. Conner, Baruch Inst. of Coastal Ecol. and Forest Science, Clemson Univ., Box 596, Georgetown, SC 29442; I. Hesse, Forest Sciences, Colorado State Univ., Fort Collins, CO 80523; J. Rybczyk, Biological and Environmental Sciences, California Univ.. Box 45, California, PA 15419; and W. Hudnall, Agronomy Dep., LSU Agricultural Center, Baton Rouge, LA 70803. This work was performed at the Louisiana State Univ. Joint contribution from the Oceanography and Coastal Sciences Department and Agronomy Department. Received 30 Sept. 1999. *Corresponding author (s-feagley@tamu.edu).

Published in J. Environ. Qual. 29:1960–1968 (2000).

The ability of forested wetlands to chemically, physically, and biologically remove pollutants, sediments, and nutrients from water has long been recognized. Many long-term studies have been conducted on wetland systems receiving wastewater effluent in the southeastern USA (Wharton, 1970; Kitchens et al., 1975; Boyt, 1976; Nessel, 1978; Shih et al., 1979; Tuschall, 1981; Dierberg and Brezonik, 1984; Ewel and Odum, 1984; Nessel and Bayley, 1984). One of the unique features of wetlands is their role in biogeochemical processes in ecotones. This feature is based on the ability of wetlands to act as a source, sink, and/or transformer of N, P, and certain heavy metals (Nixon and Lee, 1986). Since wastewater typically contains high concentrations of nutrients, even after secondary treatment, discharge into adjacent waters can cause eutrophication. Therefore, interest has increased in the use of natural wetlands as a simple and energy-efficient means of removing nutrients from wastewater and improving water quality (Nichols, 1983).

The City of Thibodaux, LA operates a secondary wastewater treatment plant that has the capacity of treating 15.1×10^6 L d^{-1} average daily flow and 22.7×10^6 L d^{-1} peak flow. The treatment system includes an aerated lagoon and a high-rate trickling filter. The present system has an average effluent of 13 mg L^{-1} BOD_5 and 30 mg L^{-1} total suspended solids (TSS) (Day et al., 1994). Fecal coliforms are being controlled with an ultraviolet radiation system. Until March 1992, the plant discharged effluent directly into the Terrebonne–Lafourch Drainage Canal (Fig. 1). The canal was water quality limited due to high nutrients, organic matter, pesticides, total dissolved solids (TDS), etc. (Louisiana Department of Environmental Quality, 1994). Advanced wastewater treatment had to be used in order to meet the effluent discharge criteria. As part of the treatment process, the city has been discharging effluent to the Pointe-au-Chene Swamp since March 1992. The discharge was allowed through a grant in accordance with Louisiana Department of Environmental Quality (LADEQ) with the City of Thibodaux. The monitoring program within the grant was designed to accomplish five objectives: (i) to meet the monitoring requirements outlined by LADEQ; (ii) to determine the fate of selected toxins, nutrients, and priority pollutants; (iii) to determine the effect of the discharge on the structure and productivity of the forest community; (iv) to determine the ability of the wetland system to assimilate N, P, C, and other elements; and (v) to assess the potential for wastewater to restore the sediment accretion balance in this subsiding system. The objective of this study, as a part of the overall project, was to determine the

Abbreviations: AWT, advanced waste treatment; BOD_5, five-day biological oxygen demand; DO, dissolved oxygen; EC, electrical conductivity; TDS, total dissolved solids; TKN, total Kjeldahl nitrogen; TSS, total suspended solids.

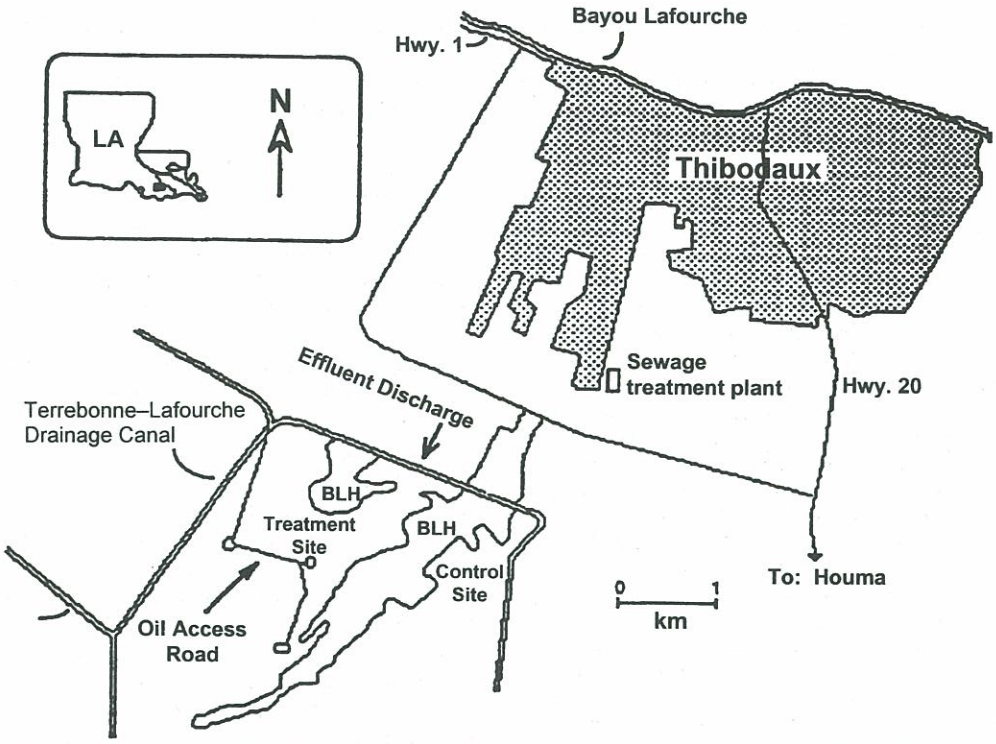

Fig. 1. Map of Thibodaux, LA, showing the location of the treated and control sites within Pointe-au-Chene Swamp. BLH = bottomland hardwood ridges (Day et al., 1994).

efficiency of the swamp in improving water quality and the effect of the effluent on the swamp water chemistry. Therefore, a 2-yr intensive water quality monitoring study was carried out to compare water quality data from a swamp treated with secondary effluent and a control swamp separated by a bottomland hardwood area.

Site Description

The Pointe-au-Chene Swamp lies on the backslope of the abandoned Bayou Lafourche distributary approximately 10 km southwest of Thibodaux, LA (Fig. 1). The study site consists of two almost continuously flooded forested wetlands within a 1425-ha hydrologically restricted basin separated by a bottomland hardwood ridge. The only significant water input into the swamp prior to the addition of effluent was via precipitation (Conner and Day, 1989). The ridge site, running from northeast to southwest (mean elevation = 1.16 m above mean sea level) is approximately 300 m wide and is vegetated primarily with oaks (*Quercus nigra* L. and *Q. texana* Buckley), sweetgum (*Liquidambar styraciflua* L.), American elm (*Ulmus americana* L.), palmetto [*Sabal minor* (Jacq.) Pers.], and boxelder (*Acer negundo* L.). The forested wetlands on either side of the ridge (mean elevation = 0.76 m above mean sea level) are dominated by ash (*Fraxinus pennsylvanica* Marshall), black willow (*Salix nigra* Marshall), baldcypress [*Taxodium distichum* (L.) Rich.], water tupelo (*Nyssa aquatica* L.), red maple (*Acer rubrum* L.), and palmetto (Conner and Day, 1989). This swamp is a typical coastal wetland due to its confinement and its classification as a cypress–tupelo swamp. Therefore, analysis of the data obtained from this swamp can be applied with a degree of confidence to similar ecosystems (Day and Farber, 1990).

The soil series within the study area are Fausse clay (very-fine, smectitic, nonacid, hyperthermic Vertic Endoaquepts) for the flooded sites and Sharkey clay (very-fine, smectitic, thermic Chromic Epiaquerts) for the ridge (USDA Soil Conservation Service, 1984). The soils contain 60 to 80% clay in the B horizon that effectively restricts ground water flow (Conner and Day, 1989).

The subtropical climate of the region is determined largely by its latitude (29°) and close proximity to the Gulf of Mexico. The mean annual air temperature is 20.6°C, ranging from 13.0°C in January to 27.5°C in July. Mean annual precipitation is 1670 mm yr^{-1}, but it has ranged from 790 mm in 1962 to 2220 mm in 1940 (Conner and Day, 1989).

Since March 1992, the 231-ha forested wetland site on the west side of the ridge has received secondarily treated municipal wastewater, which is pumped about 2.5 km from the treatment plant. Wastewater is discharged from 40 pipes located on a 610-m spoil bank that serves as the northern boundary of the site (Fig. 1). The effluent flows down a concrete apron 10 m in length, through an open area 40 m wide (Pond) into the forested swamp. It then flows southward, between the ridge on the east and an oil access road on the west and exits at a point where these two features nearly meet. Water exits the confined wetland into a larger swamp before it finally empties into the Terrebonne–Lafourche Drainage Canal. The combination of ridge, spoil bank, and access road hydrologically isolates the treatment swamp from the rest of the 1425-ha basin.

The swamp on the eastern side of the ridge is referred to as the control site and is not influenced by the effluent (Day, et al., 1994).

MATERIALS AND METHODS

A series of permanent study plots were established in March 1992 within the treated and control sites for water samplings. From March 1992 through March 1994, triplicate water samples were collected on a monthly basis at Inlet, Pond, T25, T50, T100, C25, C50, and C100. The Inlet sampling station is the location of the effluent discharge pipes. The Pond sampling station is the open area between the discharge pipes and the forested swamp. T refers to the treatment area, C refers to the control area, and the numerical values refer to distance in meters from the forest edge. Triplicate water samples were also collected at the T1600 station approximately bimonthly. The T1600 station was referred to as the Output sampling station for this study. Additionally, transects, with permanent water sampling stations marked every 200 m from the inlet to the 1600-m station, were established within the treated site. Duplicate water samples were collected along these transects in December 1992 and in January, June, and October 1994.

Water Quality Analyses

Water samples were analyzed for DO, temperature, pH, EC, TDS, BOD_5, TSS, and dissolved elements. Temperature and DO were measured in the field using an Orion Model 820 DO meter (Orion, Boston, MA). Measurements of pH, EC, and TDS were made in the field using a Corning (Corning, NY) Checkmate M90 portable microprocessor based pH and conductivity meters. Samples for dissolved elements were filtered using 0.45-μm millipore filters before analyses. Concentrations of NO_3^-, NO_2^-, SO_4^{2-}, PO_4^{3-}, and Cl^- were determined by ion chromatography (IC) with a Dionex (Sunnyvale, CA) Model 2010i IC. Concentrations of K, Ca, Na, Mg, Fe, Mn, Ni, Al, Cd, Cu, As, B, Pb, Zn, P, S, Cr, and Co were determined by an inductively coupled plasma (ICP) quantometer (Applied Research Laboratories [Beverly, MA] Model 34000). Ammonium concentrations were analyzed by the colorimetric, automated phenate method (USEPA, 1979; #350.1). Total Kjeldahl N was measured by the colorimetric, semiautomated block digestor AAII method (USEPA, 1979; #351.2). For BOD_5 analyses, initial DO was measured within 24 h after sampling and final DO was measured after 5 d of incubation at 20°C (Clesceri et al., 1989). For TSS, samples (50 to 200 mL) were filtered through predried and preweighed glass microfiber filters. Filters were dried at 105°C for 24 h and weighed to calculate the amount of TSS in the water (Clesceri et al., 1989).

Data Analysis

For all water quality parameters, a repeated measures analysis was performed using a General Linear Models procedure with factors as location and time (SAS Institute, 1988). The null hypothesis was that there were no differences in measured parameters at each sampling location and at each time of the sampling. Tests of hypotheses were performed with a level of significance $p > 0.05$. Duncan's multiple range tests were conducted for the main effects. The least squares means test was performed for pairwise comparison when significant two-way interactions were found.

RESULTS AND DISCUSSION

Water Temperature and Hydrology

The mean water temperature during the study period was 22.1°C, ranging from 12.5°C in December 1993 to 30.3°C in July 1992. The mean monthly precipitation was 150 mm, ranging from 28 mm for October 1992 to 320 mm for November 1992 (National Climatic Data Center, 1992–1994). Standing water was present at all times throughout the study period. Water levels in the control and treated sites exhibited similar patterns and changed in response to monthly precipitation and evapotranspiration (Day et al., 1994). Water level in the control site averaged 31.7 ± 12.3 cm above the forest floor. In the treated site, water levels averaged 32.7 ± 8.9 cm (Rybczyk et al., 1995).

The wastewater flow rate from the treatment plant was measured continuously by personnel of the treatment plant. Mean flow rate from March 1992 to March 1994 was $6.3 \times 10^6 \pm 1.41 \times 10^6$ L d^{-1}. Flow was very slow in the swamp and could not be visually observed. The theoretical calculated hydraulic detention time, calculated by a method described by Tchobanoglous and Schroeder (1987), was 120 d (Zhang, 1995).

Water Chemistry Assessment at Control, Inlet, and Output

Average concentrations of water quality parameters monitored at the effluent inlet, output (T1600), and control (averaged C25, C50, and C100 data) swamp water are shown in Table 1. The pH values in the entire study area remained near neutral throughout the year. This is consistent with data presented by McNamara (1978), which showed that these types of anaerobic forested wetlands are well buffered.

Secondary effluent contained organic material, N, P, dissolved salts, and trace elements. The effluent was highly nitrified, with NO_3–N being the dominant form of N. Soluble PO_4–P accounted for about 77% of the total P and SO_4–S accounted for about 82% of the total S. The difference between total and dissolved inorganic forms is an estimation of organically bound forms for P and S. Therefore, P and S in this wastewater were mainly inorganic forms. The concentrations of most trace elements were generally very low, often less than the detection limits of the inductively coupled plasma quantometer. Due to these low concentrations of trace elements, it is impossible to make any conclusions regarding their fates in the swamp water during the study period.

The control swamp water contained very low concentrations of both N and P. The average N to P ratio was 2.3, which is lower than that found for the effluent inlet (N to P ratio = 5.1). Soluble PO_4–P accounted for about 35% of the total P and soluble SO_4–S accounted for about 43% of the total S. Phosphorus and S, therefore, were mainly present in particulate and dissolved organic forms in the control swamp water. Comparisons of water quality parameters between the effluent of the treated swamp and the water of the control swamp indicate that discharging effluent into this swamp resulted in increased nutrient concentrations.

After wastewater was discharged into the swamp, the concentrations of many water quality parameters at the output station were significantly reduced compared with the concentrations in the effluent at the inlet station. It is important to note that the 1600-m station was the

Table 1. Mean concentrations of water quality parameters measured in the control, inlet, and output stations from March 1992 to March 1994.

Parameter†	Control	Inlet	Output	Attenuation‡
	mean ± SD			%
pH	7.60 ± 0.52	7.00 ± 0.42	7.00 ± 0.48	
EC (dS/cm)	113 ± 21b*	677 ± 69	427 ± 164a	37
	mg L^{-1}			
DO	3.3 ± 1.9a	2.7 ± 1.3	1.5 ± 1.5b	44
BOD_5	6.6 ± 7.6a	16.7 ± 7.9	7.61 ± 5.52a	54
NO_3–N	<0.05	8.7 ± 2.0	<0.05	100
NO_2–N	<0.3	1.0 ± 0.2	<0.3	100
TKN	0.8 ± 0.3a	2.9 ± 1.5	0.9 ± 0.5a	69
PO_4–P	0.1 ± 0.1b	1.90 ± 0.51	0.60 ± 0.50a	68
Total P	0.34 ± 0.28b	2.46 ± 0.70	0.85 ± 0.66a	66
SO_4–S	0.3 ± 0.4b	0.56 ± 3.18	3.87 ± 2.41a	60
Total S	0.78 ± 0.47b	11.8 ± 2.9	5.29 ± 3.29a	55
K	3.90 ± 1.51b	8.02 ± 1.76	5.66 ± 2.43a	29
Na	5.48 ± 1.80b	54.6 ± 11.9	36.2 ± 18.7a	34
Ca	9.78 ± 2.83b	57.4 ± 7.3	38.2 ± 16.0a	33
Mg	3.54 ± 0.92b	17.2 ± 2.3	12.0 ± 4.8a	30
Cl	9.48 ± 5.37b	56.6 ± 11.5	37.7 ± 18.0a	33
B	<0.0137b	0.13 ± 0.05	0.03 ± 0.03a	81
TDS	56 ± 11b	344 ± 35	215 ± 84a	38
TSS	97.2 ± 100.0a	19.2 ± 17.7	91.2 ± 125.6a	−375
Fe	1.21 ± 0.67a	0.49 ± 0.48	0.50 ± 0.49b	−2
Mn	0.22 ± 0.14a	0.29 ± 0.22	0.32 ± 0.23a	−10
Al	0.14 ± 0.14a	0.12 ± 0.14	0.06 ± 0.09b	50
Cu	<0.004	0.006 ± 0.007	<0.004	100
Zn	0.013 ± 0.010a	0.046 ± 0.025	0.006 ± 0.005b	87
Cr, Pb, Co, Ni, As, Cd	<DL§			

* Means for the control and output concentrations of each parameter are compared by *t*-test. Values showing the same letters are not significantly different at $p > 0.05$.
† EC, electrical conductivity; DO, dissolved oxygen; BOD_5, five-day biological oxygen demand; TKN, total Kjeldahl nitrogen; TDS, total dissolved solids; TSS, total suspended solids.
‡ (1 - output/input) × 100.
§ Detection limits (mg L^{-1}): Cr (0.008), Pb (0.09), Co (0.02), Ni (0.02), As (0.06), and Cd (0.005).

boundary for our study site. Actually, the wastewater continues through an additional 1194 ha of swamp land before it finally empties into the Terrebonne–Lafourche Drainage Canal. We are assuming that better water quality will be achieved before the water enters the canal.

High attenuation rates of N and P indicate that the swamp acted as a sink for the nutrients. No information about the advanced waste treatment (AWT) standards was available from the Louisiana Department of Environmental Quality. In Florida, the AWT standard for total N and total P for tertiary effluent is 3 and 1 mg L^{-1}, respectively. The mean total N and P at the output station (Table 1) were all less than the Florida AWT standards, indicating that tertiary treatment was achieved.

Soluble ions such as K, Ca, Na, Mg, and Cl had similar attenuation rates. These similar rates are mainly due to dilution with water in the swamp (Table 1). Chloride is a conservative constituent because it is not adsorbed by soil clay and does not take part in major biological processes. It is a required element by plants, but it is rarely reported as deficient; thus it is normally in sufficient quantities for normal plant growth. If the attenuation rate of the selected ion is higher than the attenuation rate of Cl, the other element's attenuation rate indicates that the processes involved are more than dilution. Similar attenuation rates of these elements indicate that the in situ supplies of K, Ca, Na, and Mg in the swamp were adequate for plant growth. Additional quantities of these elements from the wastewater were not apparently used as plant nutrients.

The TSS of the wastewater was increased by the swamp from an average of 19.2 mg L^{-1} at the inlet to 91.2 mg L^{-1} at the T1600 station. The average TSS at the T1600 station was within the same range as that found in the control, thus, there was no change in that water quality parameter. The increase in TSS in the wetlands is probably due to biological activity, such as the decomposition of vegetative growth stimulated by the effluent (Rybczyk, 1997).

At the output station, the NO_3–N concentration was below the detection limit (<0.05 mg L^{-1}) most of the time (Fig. 2), indicating that the swamp system can remove NO_3–N effectively. The NO_3–N was taken up by growing plants, microbially immobilized as organic N, or removed by microbial denitrification. However, the concentration of total P varied during the study period (Fig. 3). Phosphorus was removed by plant uptake, microbial assimilation, and soil adsorption. The amount of P present at the output station depended on the amount of P measured at the inlet and hydraulic retention time. The hydraulic retention time was approximately 120 d as calculated using a method discussed by Tchobanoglous and Schroeder (1987). Thus, a time lag between inlet and output peaks was expected. In May 1992 and April and June 1993, three high P peaks were recorded for the effluent (total P > 3.0 mg L^{-1}) at the inlet station. A high P concentration was measured at the output station for the following sampling date (2 mo later) except for June 1993. This indicates that it took about 60 d for the wastewater to move from the inlet to the T1600 sampling station. In June 1993, however, P concentration at T1600 did not correspond to the high P concentration measured from the inlet in April 1993. This was because of the high rainfall during June 1993 (National Climatic Data Center, 1993). West and Feag-

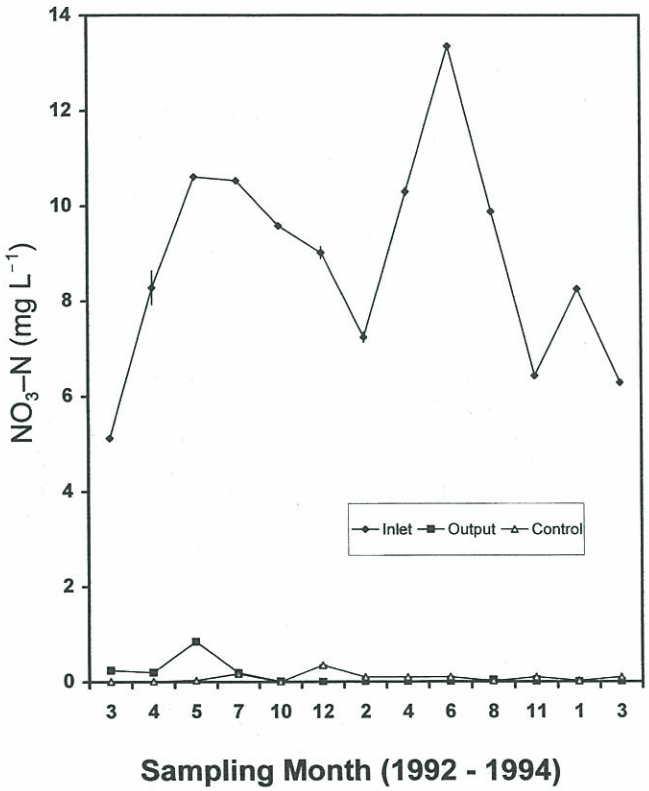

Fig. 2. Concentrations of NO_3–N measured at the inlet, output, and control stations.

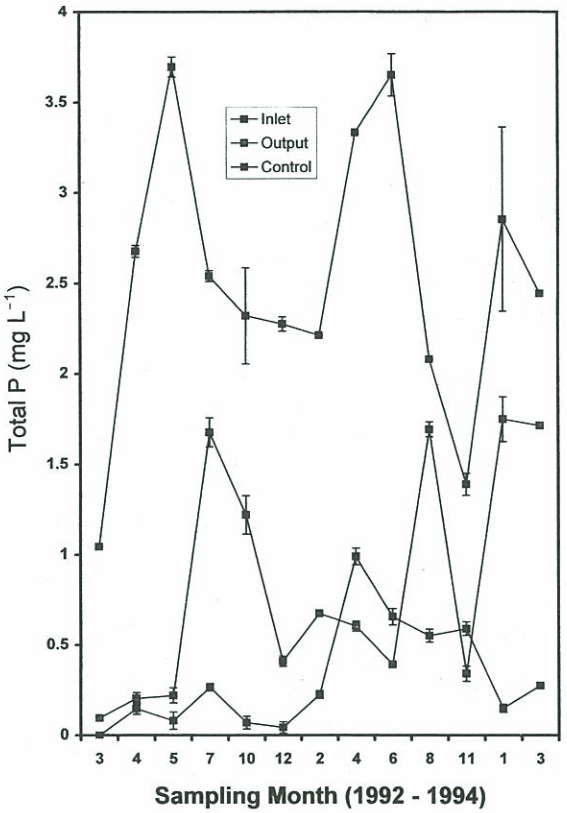

Fig. 3. Concentrations of total P measured at the inlet, output, and control stations.

ley (1995) reported that P concentration in Louisiana's precipitation was generally below detection limit ($P < 0.1$ mg L^{-1}). Thus, P in the swamp water could be diluted by the rain. It was also noted that the amount of P measured at T1600 actually increased with time when compared with the springs of 1992, 1993, and 1994. This indicates a microbial regeneration of P and/or a progressive saturation of adsorption sites. Although the swamp system was P deficient, P could not be assimilated by plants or microbes efficiently due to a very low N to P ratio caused by high N attenuation rate (almost 100%). The system was thus an N-limited system.

Comparing water quality between the output and the control, the concentrations of BOD_5, NO_3–N, TKN, Cu, and Zn at the output station were similar to those found for the control swamp water (Table 1). There was some attenuation of the soluble ions (Na, Ca, Mg, Cl, and SO_4–S) by the swamp system, but it did not reduce them to the control site concentrations for most of the sampling dates. The concentrations of soluble inorganic P and S observed at the output station accounted for about 70% of the total P and 65% of the total S, which were higher than the percentages found for the control swamp water. In the control swamp water, soluble inorganic PO_4–P accounted for 35% of the total P and SO_4–S accounted for 43% of the total S. The addition of the wastewater into the swamp, which shifted the swamp water from mainly organic forms of P and S to the inorganic forms, increased the potential of these elements to be taken up by plants or adsorbed by soil. Addition of wastewater, therefore, should stimulate nutrient cycling in the swamp.

Detectable concentrations of Zn and Cu were found in the effluent water at the inlet station. However, their concentrations were less than the detection limit (Zn, 0.001 mg L^{-1}; Cu, 0.004 mg L^{-1}) and less than the limit of quantitation (LOQ) (Jarvis, 1992) at the output station.

Transect Study in the Treated Swamp

Water Chemistry Changes Within the First 100 m from the Inlet

Table 2 lists of the mean concentrations of selected parameters measured within 100 m of the wastewater inlet. From the inlet to the pond, significant decreases in concentrations were mainly due to physical dilution when the swamp water and the wastewater mixed. After the pond station, concentrations of total P, BOD_5, K, and Ca were not significantly decreased at the T100 station. However, 63% of NO_3–N in the effluent was attenuated between the inlet and T100.

To study the effect of physical dilution, Cl was taken as a reference. The concentration changes of Cl along the flow pathway were assumed as a dilution effect by the swamp water. If dilution is the only process involved in the concentration changes of element A, the concentration changes of element A should be proportional to the concentration changes of Cl. This can be described as:

$$Cl_0/Cl_1 = A_0/A_1 \quad [1]$$

therefore,

$$A_1 = A_0/Cl_0 \times Cl_1 \quad [2]$$

Table 2. Water quality changes within the first 100-m distance from the inlet (n = 24).

Distance	No_3–N	Total P	Total S	B	BOD_5†	DO‡	TDS§	K	Na	Cl	Ca	Mg
	mg/L											
Inlet	8.4a*	2.49a	11.67a	0.13b	16.7a	2.57b	349a	8.07b	54.54a	56.66a	57.39b	17.23a
Pond	7.3b	2.24b	11.40b	0.13b	12.7b	6.42a	338b	8.02b	52.94b	55.21b	56.65cd	17.07b
25 m	5.0c	2.22b	10.97c	0.14a	11.1b	1.54c	338b	8.41a	51.81c	53.96c	58.16a	17.32a
50 m	4.0d	2.22b	10.86c	0.15a	11.4b	1.62c	326c	8.16b	51.42d	53.83c	57.12bc	17.03b
100 m	3.1e	2.21b	10.51d	0.15a	10.8b	1.13d	317d	8.00b	50.63e	52.96d	56.33d	16.81c
Reduction (%)	63	11	10	none	35	56	9	none	7	7	2	2

* Means within columns for the same parameter with the same letter are not significantly different at $p > 0.05$.
† 5-d biological oxygen demand.
‡ Dissolved oxygen.
§ Total dissolved solids.

where A_1 is the diluted concentration of element A at a distance away from the inlet, A_0 is the concentration of element A measured in the inlet, Cl_0 is the Cl concentration measured in the inlet, and Cl_1 is the Cl concentration measured at this distance.

Based on this assumption, the diluted concentrations of NO_3–N and PO_4–P at each sampling station were calculated by the concentration ratio of that parameter and Cl for the inlet multiplied by the concentration of Cl at that station. Figures 4 and 5 show the calculated dilution effects for NO_3–N and PO_4–P from the inlet to T100 and the measured NO_3–N and PO_4–P concentrations. The NO_3–N concentrations measured for the swamp water were much lower than that calculated by dilution factor (Table 2). This indicates that dilution was not the only process involved in NO_3–N concentration decrease. Other processes were responsible for NO_3–N removal. These results suggest that the swamp system can assimilate N within a relatively short distance. However, P reduction within the first 100 m was mainly due to dilution.

The highest DO value was found for the pond station (Table 2). This station is an open water area with less cover than within the forested swamp. The higher DO in the pond was probably due to higher photosynthesis (due to more sunlight), or attenuation due to (i) increased turbulence due to the process by which the effluent was delivered to the swamp (cascading down a ridged piece of concrete) and (ii) the exposure of the pond to wind, which often produced waves. The advantages of the pond to water quality improvement were that it provided more O_2 for microorganisms to decompose organic matter and enhanced sequential nitrification of NH_4 followed by denitrification. The DO decreased within the forested area.

Low DOs were found in the forested area during the summer months. Concentrations were higher in the winter months, when there was less biological demand. Available evidence indicates that naturally dystrophic

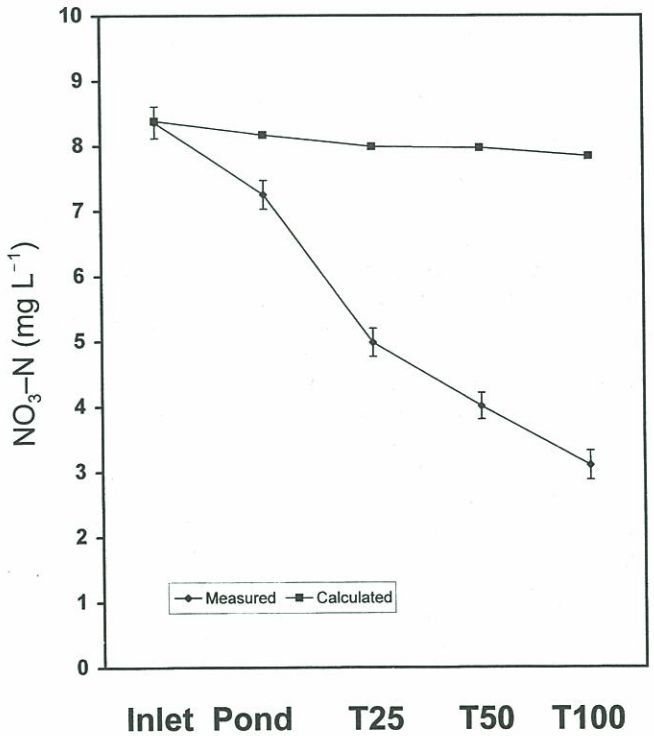

Fig. 4. Mean concentrations of NO_3–N, measured at the sampling stations in the treated site and calculated by the dilution factor using Cl (n = 24).

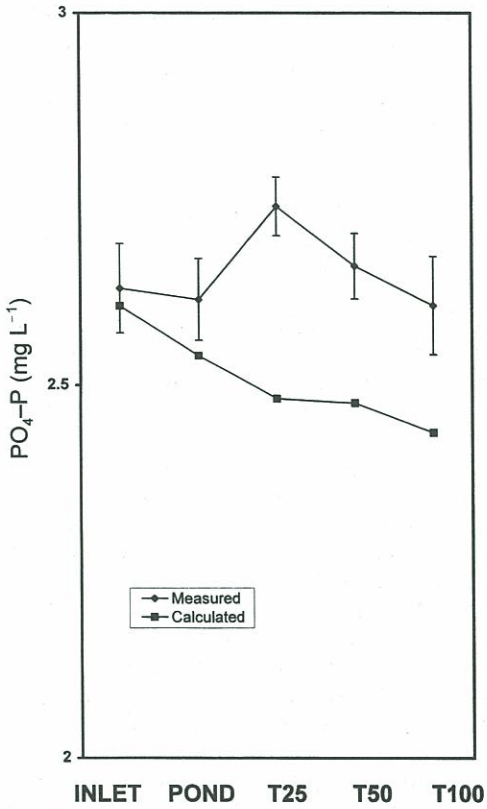

Fig. 5. Mean concentrations of PO_4–P, measured at the sampling stations in the treated site and calculated by the dilution factor using Cl (n = 24).

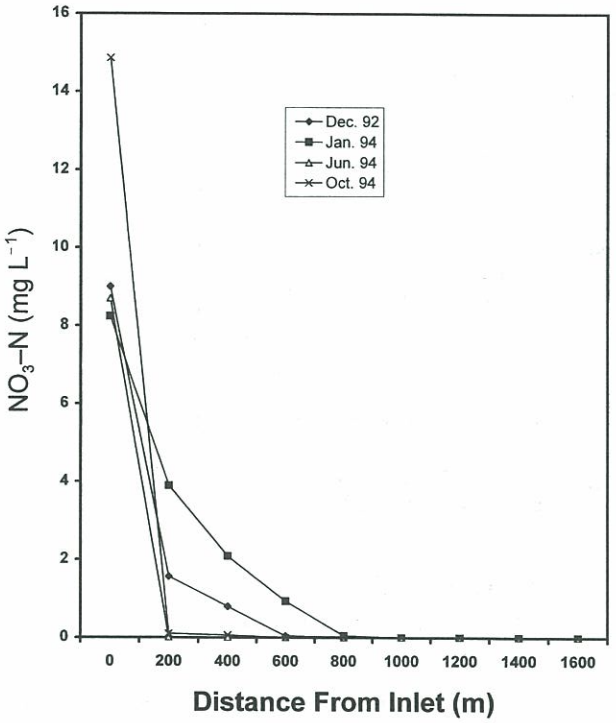

Fig. 6. Mean concentrations of NO_3–N, measured along the transect in December 1992 and January, June, and October 1994.

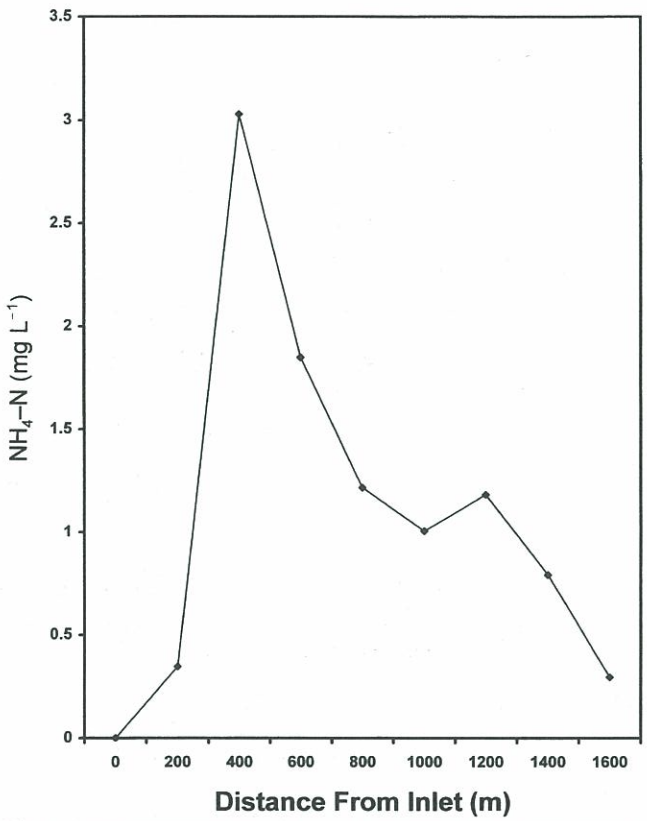

Fig. 7. Mean concentrations of NH_4–N, measured along the transect in June 1994.

waters are common in Louisiana wetlands and water bodies and streams surrounded by extensive wetlands. Dystrophic waters are characterized by high levels of humic organic matter derived from the watershed and low DO levels (Hutchinson, 1957; Miles and Brezonik, 1981). In the Atchafalaya Basin, Bryan et al. (1976) reported that a DO value of 3.0 mg L^{-1} was common in many streams of the basin and that a large area of swamp had DO concentrations less than 1.8 mg L^{-1}. They also reported a seasonality of DO, with highest values occurring in the winter and lowest in the summer. Similar patterns were reported for the swamp forests upstream of Lac des Allemands in the Barataria Basin. Dissolved oxygen values in streams draining the swamp were often lower than 5.0 mg L^{-1} (Butler, 1975), and values less than 2.0 mg L^{-1} were common in swamp waters for much of the year (McNamara, 1978; Kemp and Day, 1984). Only in winter did DO levels increase above 5.0 mg L^{-1} (Kemp and Day, 1984). We conclude that a low DO is a natural and common occurrence in the study area and continued low DO is to be expected with the application of effluent.

About one-third of the BOD_5 introduced with the wastewater was attenuated by the time the water left the pond (Table 2). Changes in the following 1600 m to the output station (Table 1) attenuated the BOD_5 to 54%. Thus, hydraulic retention time did not appear to play as important a role as dilution.

Water Chemistry Changes 1600 m from the Inlet

The concentrations of TDS, NO_3–N, P, S, B, K, Na, Ca, Mg, and Cl decreased with distance (Tables 1 and 2). This indicates that portions of each of these were assimilated into the wetland system. Nitrogen decreased dramatically with distance along the transect (Fig. 6). The NO_3–N decrease was also temperature dependent. During warm seasons (June and October 1994), NO_3–N was totally removed by the time the water reached 200 m. During winter, however (December 1992 and January 1994), NO_3–N was not removed completely until 800 m, about half-way from the inlet to the output station. The NH_4–N concentration was relatively low and consistent for the transect (Fig. 7). The NH_4–N concentration increased within the swamp compared with the inlet. This suggests that a biological re-release process was involved (Day et al., 1989; Faulkner and Richardson, 1989).

The strong reduction of NO_3–N is probably due to denitrification. Denitrification is an important mechanism related to the ability of wetlands to act as a nutrient sink. However, adequate assessments of the importance of the process to the use of natural wetlands for tertiary wastewater treatment are limited (Nixon and Lee, 1986). Nevertheless, studies have suggested that although denitrification occurs readily in wetland sediments, the process is limited by the supply of NO_3–N produced via nitrification (Nichols, 1983; Dierberg and Brezonik, 1984). Since NO_3–N is the major form of inorganic N in the effluent, denitrification should not be limited. Both Crozier et al. (1996) and Boustany et al. (1997) have shown that there are high rates of denitrification in the treatment area. High denitrification rates in marshes of Barataria Bay, Louisiana, have been reported by Smith and Delaune (1983). A number of studies suggest that wetlands can be efficient N sinks

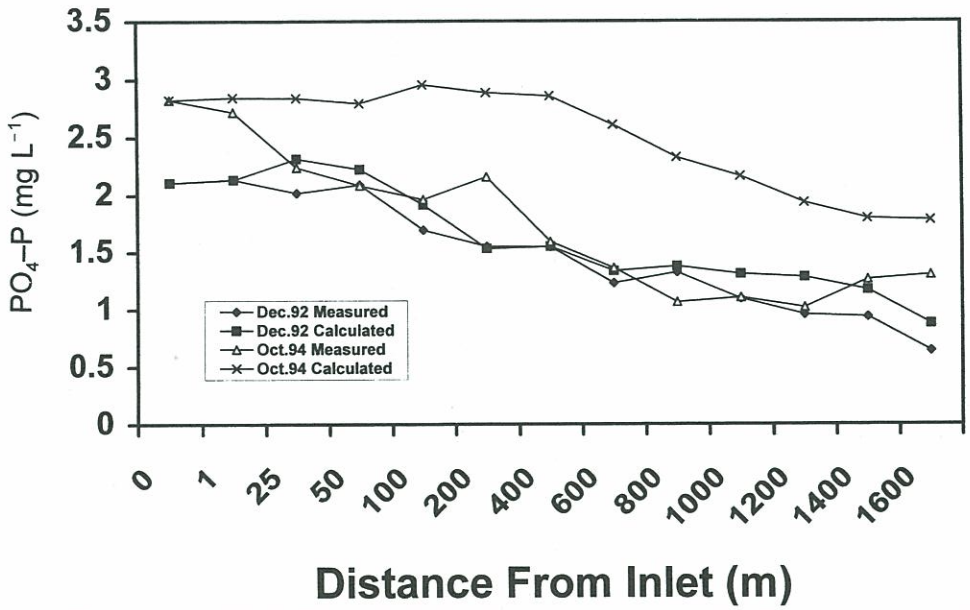

Fig. 8. Concentrations of PO_4–P, measured in the field and calculated by the dilution factor using Cl along the transect in the treated site in December 1992 and October 1994.

and that at NO_3–N loading rates between 10 and 20 g m^{-2} yr^{-1}, more than 90% can be removed (Kadlec and Tilton, 1979; Dierberg and Brezonik, 1985; Lane et al., 1999).

In December 1992, the measured PO_4–P concentrations along the transects had similar trends to that calculated by the dilution effect (Fig. 8). This indicated that physical dilution was the main process involved during December. However, the difference between the measured and the calculated PO_4–P concentrations was greater during October 1994. This indicated that other processes were involved in P removal at this time. The mean water temperature was 13.3°C for December 1992 and 26.8°C for October 1994. These data indicate that dilution was the major process for P decreasing along the transect when the mean temperature was low. Other processes such as plant uptake, microbial immobilization, and increased soil adsorption played very important roles for P removal in the warm season. For October 1994, if dilution was the only process, as we assumed based on the calculated concentration, equilibrium conditions would have occurred within the first 400 m. However, comparisons with concentrations measured in the field indicate that after 2 yr of wastewater application, the swamp still could remove P from the wastewater efficiently when temperatures were adequate to support plant growth.

CONCLUSIONS

The swamp system reduced 100% of the NO_3–N, 69% of the TKN, and 66% of the total P in the secondarily treated municipal wastewater. At 1600 m away from the wastewater inlet, mean concentrations were 0.9 mg L^{-1} for total N and 0.85 mg L^{-1} for total P, whereas the AWT standards in the tertiary effluent are 3 mg L^{-1} for total N and 1 mg L^{-1} for total P. Therefore, tertiary treatment was achieved. Processes other than physical dilution by the natural swamp water such as denitrification, plant uptake, microbial immobilization, and soil adsorption were proved to be actively involved in the wastewater remediation (Zhang, 1995).

The addition of the wastewater into this natural swamp, shifting the swamp water nutrients from mainly organic forms to inorganic forms, should stimulate nutrient cycling in the swamp. There was no potential detrimental loading of trace metals into the swamp water because of very low concentrations in the wastewater and similar concentrations at the output station compared with the control. Since "swamp water" is really neither "fresh water" nor "marine water", and supports different types of aquatic life, the elemental concentrations set by USEPA may not apply. The concentrations of BOD_5, NO_3–N, TKN, B, Cu, and Zn at 1600 m from the inlet were similar to the control concentrations. Soluble ions such as Ca, Mg, Na, and Cl were diluted by the swamp water, but the concentrations were not attenuated to the control levels. The attenuation efficiency of P was dependent on the inlet concentration and swamp water temperature. Due to the added P, the potential for eutrophication may be enhanced. However, during the study period, using odor as an indicator for eutrophication, there were no noticeable changes in the odor of the treated swamp water compared with the control swamp water. The swamp attenuated nutrients from the wastewater efficiently when temperatures were above 20°C and plants were actively growing.

Water quality in the treated site was improved within 100 m from the inlet with NO_3–N being reduced the most. Complete attenuation of NO_3–N was achieved within 800 m of the inlet. We postulate that the swamp system could attenuate N even if the N loading rate was doubled. Phosphorus was significantly decreased during the growing season. Based on the present loading rate, most other soluble ions were significantly diluted or attenuated by the system within the confines of this study. It appears that tertiary water treatment can be

achieved without detrimental effects to the swamp. The added nutrients should help revitalize this declining swamp.

ACKNOWLEDGMENTS

The study was partially supported by a grant from the City of Thibodaux, LA. Additional funding was provided by the Louisiana Dep. of Environ. Qual., USEPA, and the Louisiana Water Resour. Res. Inst. Dr. Glenn Garson helped with the statistical analyses.

REFERENCES

Breaux, A.M., and J.W. Day, Jr. 1994. Policy considerations for wetland wastewater treatment in the coastal zone: A case study for Louisiana. Coastal Manage. 22:285–307.

Butler, T.J. 1975. Aquatic metabolism and nutrient flux in a southern Louisiana swamp and lake system. M.S. thesis. Louisiana State Univ., Baton Rouge.

Clesceri, L.S., A.E. Greenberg, and R.R. Trussell. 1989. Standard methods for the examination of water and wastewater. American Public Health Association, Washington, DC.

Conner, W.H., and J.W. Day, Jr. 1989. A use attainability analysis of wetlands for receiving treated municipal and small industry wastewater: A feasibility study using baseline data from Thibodaux, LA. Center for Wetland Resources, Louisiana State Univ., Baton Rouge, LA.

Crozier, C.R., J.M. Rybczyk, and W.H. Patrick, Jr. 1996. Spatial gradients of dissolved nitrate and nitrous oxide in a wetland forest receiving treated sewage effluent. p. 65–68. *In* K. Flynn (ed.) Proc. South. Forested Wetlands Ecol. Manage. Conf., Georgetown, SC.

Day, J.W., Jr., and S.C. Farber. 1990. A feasibility analysis of the use of Louisiana wetlands for wastewater treatment. Fiscal Year 1990 Research Program. Louisiana Water Resour. Res. Inst., Louisiana State Univ., Baton Rouge.

Day, J.W., Jr., S.E. Feagley, I.D. Hesse, J.M. Rybczyk, and X. Zhang. 1994. The use of swamp forests near Thibodaux, Louisiana for application of treated municipal wastewater: Monitoring the effects of the discharge. DEQ Water Permit 2690 LA 0032948. Final Report to the City of Thibodaux, Louisiana and the Louisiana Department of Environmental Quality. Coastal Ecology Inst., Louisiana State Univ., Baton Rouge.

Day, J.W., Jr., C.A.S. Hall, W.M. Kemp, and A. Yanez-Arancibia. 1989. Estuarine chemistry. p. 79–143. *In* Estuarine ecology. John Wiley & Sons, New York.

Dierberg, F.E., and P.L. Brezonik. 1984. Nitrogen and phosphorus mass balances in a dome receiving wastewater. p. 112–118. *In* K.C. Ewel and H.T. Odum (ed.) Cypress swamps. Univ. Presses, Gainesville, FL.

Dierberg, F.E., and P.L. Brezonik. 1985. Nitrogen and phosphorus removal by cypress swamp sediments. Water Air Soil Pollut. 24: 207–213.

Ewel, K.C., and H.T. Odum. 1984. Cypress swamps. Univ. Presses, Gainesville, FL.

Faulkner, S.P., and C.J. Richardson. 1989. Physical and chemical characteristics of freshwater wetland soils. p. 41–72. *In* D.A. Hammer (ed.) Constructed wetlands for wastewater treatment: Municipal, industrial, and agricultural. Lewis Publ., Chelsea, MI.

Godfrey, P.J., E.R. Kaynor, S. Pelczarski, and J. Benforado (ed.). 1985. Ecological considerations in wetlands treatment of municipal wastewaters. Van Nostrand Reinhold, New York.

Hutchinson, G.E. 1957. A treatise on limnology. Wiley-Interscience, New York.

Jarvis, I. 1992. Sample preparation for ICP–MS. p. 172–224. *In* K.E. Jarvis et al. (ed.) Handbook of inductively coupled plasma mass spectrometry. Chapman and Hall, New York.

Kadlec, R.H., and R.L. Knight. 1996. Treatment wetlands. Lewis Publ., Boca Raton, FL.

Kadlec, R.H., and D.L. Tilton. 1979. The use of fresh water wetlands as a tertiary wastewater treatment alternative. Crit. Rev. Environ. Control 9:185–212.

Kemp, G.P., and J.W. Day. 1984. Nutrient dynamics in a Louisiana swamp receiving agricultural runoff. p. 286–293. *In* K.C. Ewel and H.T. Odum (ed.) Cypress swamps. Univ. Presses, Gainesville, FL.

Kitchens, W.M., Jr., J.M. Dean, L.H. Stevenson, and J.M. Cooper. 1975. The Santee Swamp as a nutrient sink. p. 349–366. *In* F.G. Howell et al. (ed.) Mineral cycling in southeastern ecosystems. ERDA Symp. Ser. 740513. U.S. Geol. Survey, Washington, DC.

Lane, R.R., J.W. Day, and B. Thibodeaux. 1999. Water quality analysis of a freshwater diversion at Caernarvon, Louisiana. Estuaries 22:327–336.

Louisiana Department of Environmental Quality. 1994. Water quality inventory, State of Louisiana water quality management plan. Vol. 5, Part B. Louisiana Dep. of Environ. Quality, Water Quality Management Division, Baton Rouge.

McNamara, S.J. 1978. Metabolism measurements of a flooded soil community in a Louisiana swamp forest. M.S. thesis. Louisiana State Univ., Baton Rouge.

Miles, C.J., and P.L. Brezonik. 1981. Oxygen consumption in humic-colored waters by a photochemical ferrous–ferric catalytic cycle. Environ. Sci. Technol. 15:1089–1095.

National Climatic Data Center. 1992–1994. Climatological data: Louisiana. Vol. 97–99. NCDC, Asheville, NC.

Nessel, J.K. 1978. Distribution and dynamics of organic matter and phosphorus in a sewage enriched cypress swamp. M.S. thesis. Univ. Florida, Gainesville.

Nessel, J.K., and S.E. Bayley. 1984. Distribution and dynamics of organic matter and phosphorus in a sewage enriched cypress swamp. p. 262–278. *In* K.C. Ewel and H.T. Odum (ed.) Cypress swamps. Univ. Presses, Gainesville, FL.

Nichols, D.S. 1983. Capacity of natural wetlands to remove nutrients from wastewater. Water Pollut. Control Fed. J. 55:495–505.

Nixon, S.W., and V. Lee. 1986. Wetlands and water quality. Wetlands Res. Prog. Tech. Rep. Y-86-2. U.S. Army Eng. Waterway Exp. Stn., Vicksburg, MS.

Reddy, K.R., and W.H. Smith (ed.). 1987. Aquatic plants for water treatment and resource recovery. Magnolia Publ., Orlando, FL.

Richardson, C.J., and D.S. Nichols. 1985. Ecological analysis of wastewater management criteria in wetland ecosystems. p. 351–391. *In* P.J. Godfrey (ed.) Ecological considerations in wetlands treatment of municipal wastewaters. Van Nostrand Reinhold, New York.

Rybczyk, J.M. 1997. The use of secondarily treated wastewater effluent for forested wetland restoration in a subsiding coastal zone. Ph.D. diss. Louisiana State Univ., Baton Rouge.

Rybczyk, J.M., X.W. Zhang, J.W. Day, Jr., I. Hesse, and S. Feagley. 1995. The impact of Hurricane Andrew on tree mortality, litterfall, nutrient flux, and water quality in a Louisiana coastal swamp forest. J. Coastal Res. Special Issue 21:340–353.

SAS Institute. 1988. SAS user's guide. Ver. 6.03. SAS Inst., Cary, NC.

Shih, S.F., A.C. Federical, J.F. Milleson, and M. Rosen. 1979. Sampling programs for evaluating upland marsh to improve water quality. Trans. ASAE 22:828–833.

Smith, C.J., and R.D. Delaune. 1983. Nitrogen loss from freshwater and saline estuarine sediments. J. Environ. Qual. 12:514–518.

Tchobanoglous, G., and E.D. Schroeder. 1987. Water quality. Addison–Wesley Publ., Reading, MA.

Tuschall, J.R., Jr. 1981. Heavy metal complexation with naturally occurring organic ligands in a wetland ecosystem. Ph.D. diss. Univ. of Florida, Gainesville.

USDA Soil Conservation Service. 1984. Soil survey of Lafourche Parish, Louisiana. U.S. Gov. Print. Office, Washington, DC.

USEPA. 1979. Methods for chemical analysis of water and wastes. EPA-600/4/-79-020. National Environmental Center, Cincinnati, OH.

USEPA. 1994. North American wetlands for water quality treatment database. USEPA, Office of Research and Development, Cincinnati, OH.

West, L.M., and S.E. Feagley. 1995. The chemical composition of atmospheric deposition collected from six Louisiana sites from 1983 to 1992. Atmos. Environ. 29(11):1211–1217.

Wharton, C.H. 1970. The southern river swamp: A multiple use environment. Bureau of Business and Economic Res., School of Business Admin., Georgia State Univ., Atlanta.

Zhang, X. 1995. Use of natural swamp for wastewater treatment. Ph.D. diss. Louisiana State Univ., Baton Rouge.