WETLANDS, Vol. 20, No. 1, March 2000, pp. 48–61

THE EFFECTS OF VARIED HYDRAULIC AND NUTRIENT LOADING RATES ON WATER QUALITY AND HYDROLOGIC DISTRIBUTIONS IN A NATURAL FORESTED TREATMENT WETLAND

Theodore Blahnik[1] and John Day, Jr.[2]
[1] *Ensafe, Inc.*
4545 Fuller Drive, Suite 326
Irving, Texas, USA 75038

[2] *Lousiana State University*
Department of Oceanography and Coastal Sciences and Coastal Ecology Institute
Baton Rouge, Louisiana, USA 70803

Abstract: Hydrology and water quality distributions in a Louisiana forested waste-water treatment wetland were studied under four different hydraulic loading rates (HLR). Pond discharge, surface-water elevations, and fluorescent dye travel times were recorded to assess surface-water hydrology, and water samples were collected for nitrate, ammonium, phosphate, and suspended solids analyses. Wetted surface area increased with pond discharge rate, and 58 to 66 percent of surface-water flow was concentrated in shallow channels covering only 10 to 12 percent of the total study area. Water residence times were much longer (0.9 to 1.1 days) than minimum dye travel times (2 to 3 hours) through the 4-hectare study area. Relative to study area influent concentrations, study area outflow concentrations of nitrate and total and organic suspended solids were lower, ammonium was higher, and phosphate was generally unchanged. However, there was an increase in concentrations of nitrate, ammonium, and phosphate within 50 m of the study area inflow location. Ammonium and phosphate did decrease from these peak concentrations. Net nitrate production was observed within 50 m of the pond outfall and was probably due to nitrification. Net nitrate removal was observed beyond this distance and ranged up to 0.10 $g \cdot m^{-2} \cdot d^{-1}$ probably due primarily to denitrification. In general, nitrate removal rates increased linearly with changes in nitrate loading rates. Results show that nutrient distributions are linked to hydrology. Higher pond discharge rates created more treatment surface area, and higher constituent loading rates produced higher removal rates. Therefore, discharge rates could be manipulated, and physical control structures could be installed to increase wetted surface area and increase removal efficiency within the wetland. Higher loading rates could then be processed without requiring significant increases in treatment area.

Key Words: wetland, waste-water treatment, hydrology, nitrate, phosphate, ammonium, total suspended solids

INTRODUCTION

The ability of wetlands to absorb or transform nutrients and suspended solids in wastewater is well established (Godfrey et al. 1985, Hammer 1989, Cooper and Findlater 1990, Breaux 1992, Knight 1994, Hiley 1995, Kadlec and Knight 1996). The rate at which water quality changes occur has also been studied in a number of wetland types, climates, and latitudes (WPCF 1990). While many of these rates of change are generally acknowledged to be dependent on influent concentrations and water flux (Knight et al. 1987, WPCF 1990, NADB 1993), studies have focused primarily on rates averaged over entire receiving wetland areas and not the distribution of rates within the wetland.

Hydrologic conditions can directly affect the chemical and physical processes governing nutrient and suspended solids dynamics within wetlands (Mitsch and Gosselink 1993). Because flow through wetlands is unequally distributed due to different degrees of channelization and microtopography (Hammer and Kadlec 1986, Kadlec 1994), flow distributions and the factors affecting them need to be examined in order to assess water quality dynamics within a wetland.

A freshwater forested wetland in southern Louisiana was selected for this study based on long-term use of the area for tertiary treatment of municipal waste-water from the city of Breaux Bridge. The area of the wetland adjacent to the city's waste-water oxidation ponds has been receiving treated municipal effluent for nearly 50 years. During periods of low rainfall, the hydrology of the area near the oxidation pond discharge is dominated by waste-water flow Previous studies (Day et al. 1994, Hesse 1994, Hesse et al. 1998) have shown that this wetland is capable of removing substantial amounts of nutrients and suspended material from the waste-water in relatively short distances (<300 m) from the influent point without causing significant ecological damage and while also enhancing processes such as tree growth and sediment accretion.

The objectives of this study were to assess variations in hydrologic and water quality distributions under varying waste-water flows, to correlate physical and chemical processes that may lead to these variations, and to examine implications of these results as they apply to future waste-water discharge at the site. In order to accomplish these objectives, we applied waste-water at three different flow rates over a six-week period and measured changes in hydrologic conditions and water quality distributions.

DESCRIPTION OF AREA

The receiving wetland is a cypress-tupelo and bottomland hardwood swamp located near the city of Breaux Bridge, Louisiana, USA in St. Martinville parish. The wetland is approximately 1295 hectares and is bounded on the north, west, and east by natural levee uplands of Bayou Teche and the Vermillion River and on the south by Evangeline and Ruth Canals (Figure 1). The City of Breaux Bridge has discharged municipal waste-water effluent to the northeast part of the wetland for nearly 50 years. The city presently operates three oxidation ponds with a mean daily flow of 0.01 to 0.02 m^3s^{-1}. Pond discharge can enter the swamp through any of five discharge pipes located along the western levee of the ponds. Before this study, all discharge entered the swamp through the southernmost pipe about 500 m south and downstream of the immediate study area. A more extensive description of this wetland can be found in Day et al. (1994) and Hesse et al. (1998).

Bottomland hardwoods dominate near the pond levees, but there is a rapid transition to a cypress-tupelo dominated forest. Dominant tree species near the pond levees include willow (*Salix nigra* Marsh.) and tupelo (*Nyssa aquatica* L.). In the cypress-tupelo dominated forest, dominant species include cypress (*Taxodium distichum* (L.) Rich.), swamp maple (*Acer rubrum* L.), and tupelo. The understory consists of large areas of little or no undergrowth with smaller areas of thick vegetation where larger trees have fallen, leaving gaps in the forest canopy. Shallow soils consist of coarse organics near the surface grading downward to primarily organic clays within a few centimeters.

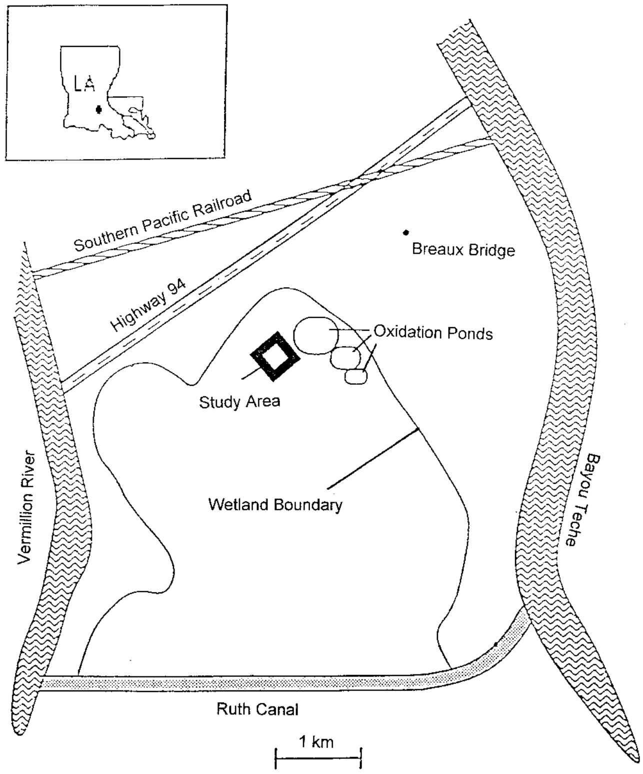

Figure 1. Vicinity map showing the study area location and important local land features.

The study focused on a 4-hectare plot located adjacent to the northernmost discharge point of the ponds (Figure 2). The study area slopes slightly to the south (0.5 m/km). Small variations in relief of about 10 cm in much of the area are related to low-lying channel areas, surface-water storage depressions, or higher vegetated areas. A 50-cm-high mounded feature along the study area's eastern boundary near the pond levee resulted from spoil placement during pond levee construction.

The hydrology of the study area is influenced by rainfall, backwater flooding, levees and spoil banks, inflow from the city oxidation ponds, relatively impermeable subsurface soils, and a complex network of shallow, natural channels and storage areas. There were almost no areas of standing water immediately before the study period. Backwater flooding from the Vermillion River occurs primarily during the fall, winter, and early spring and can increase study area water depths up to 1 m near the pond levees. Oxidation pond levees, an old canal, and canal spoil banks divert upland runoff around the study area.

MATERIALS AND METHODS

Site Preparation

Thirty-eight staff gages (Figure 2) and a continuous recording water-level gage were installed to measure

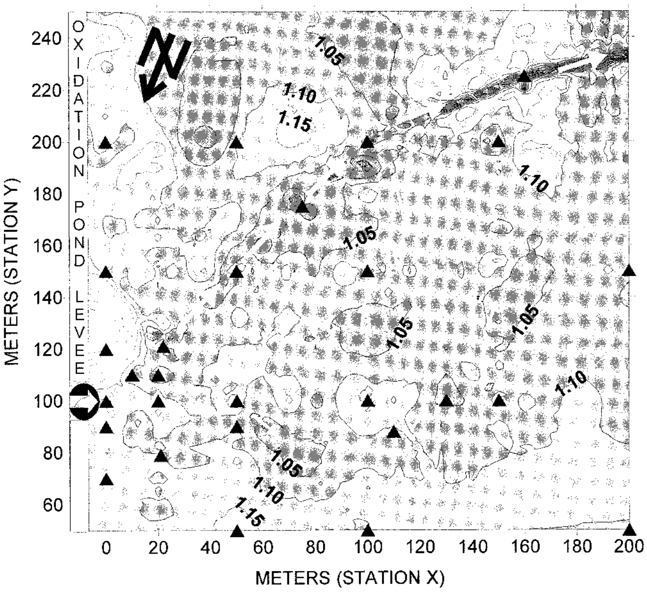

Figure 2. Study area topography (0.05 m contours, msl) and site features. Darker shaded areas indicative of low-lying channel areas. Staff gage locations (shown as triangles) were referenced by Station [Y + X]. The circled arrow represents the oxidation discharge pipe location. The large and small arrows represent the primary and secondary flow pathways, respectively, through the study area. The area of higher elevation just south of the discharge pipe is a spoil mound created during pond levee construction.

changes in surface-water elevation across the study area. Locations were concentrated near the area of discharge, at survey-grid intersections, and in channel areas. The staff gages were made from 1.5-m sections of PVC pipe graded every 0.25 cm. The staff gages were pushed about 60 cm into the ground to minimize the potential for vertical movement. The continuous recording gage could measure changes in surface and subsurface-water elevations at a 1:12 cm scale within a 1 m range. Precipitation was monitored using both an on-site, manual-reading rain gage and daily rainfall data collected at the Southern Regional Climate Center (SRCC) Breaux Bridge 4-S gaging station located approximately 4 km south of the study area.

To establish vertical and horizontal controls for ground and staff gage locations, a topographic survey estimated elevations to the nearest 0.3 cm relative to the top of the study area discharge pipe (1.844 m) using a laser level. An initial 300 × 300 m survey grid was laid out on lines spaced 100 m apart, and surface elevations were recorded every 10 m. Additional survey lines with elevations read every 1 m were then laid out in the 200 × 200 m area nearest the discharge pipe, with lines parallel to the grid spaced 50 m apart and lines diagonal to the grid running between parallel line intersections.

Table 1. Study period durations, pond discharge rates, and number of water quality sample locations. Field sampling was performed on the last day of each study period.

Study Period Dates	Discharge Rate ($m^3 \cdot d^{-1}$)	No. Water Quality Stations
Jun 05–June 11	503 ± 65	10
June 12–June 18	901 ± 13	10
June 19–July 01	1164 ± 114	11
July 02–July 10	901 ± 25	7

Field Sampling and Laboratory Analyses

The field investigation was divided into four discrete, 6- to 13-day study periods (Table 1) distinguished by a relatively constant waste-water loading rate applied to the study area. Each study period included one sampling event that included staff gage readings, surface-water sample collection, and a fluorescent dye tracer study. No sampling events occurred within 2 days after any daily rainfall event >0.5 cm to help prevent dilution by storm water.

Staff gage readings and water samples were collected concurrently. Water quality samples were collected at seven to eleven locations (Table 1) over a period of about three hours, primarily in channel areas near staff gage locations. Additional samples were collected outside the study area to provide reference or ‘‘background’’ concentrations.

After completing water sampling, gage readings, and flow estimation, a fluorescent dye tracer study was carried out to assess direction and relative velocity of flows through the wetland. Small amounts (<10 ml) of Rhodamine fluorescent dye were added to the effluent inflow at the pond discharge pipe. The leading edge of the dye plume was then mapped and timed as it moved through the wetland. Dye was added to the leading edge when the plume became faint or indiscernible from surrounding waters. The plume was followed until it either left the study area or entered a stagnant area where diffusion rather than advection appeared to dominate plume movement.

Water samples for nutrient, chloride, and total suspended solids analyses were collected in acid-washed bottles, placed on ice, and transported back to the laboratory. In the laboratory, portions of the samples were filtered through 0.7-micron glass-fiber filters and frozen for analysis of nitrate+nitrite (USEPA 1979, Method 353.2), ammonium (USEPA 1979, Method 350.1), and phosphate (USEPA 1979, Method 365.2). Remaining sample volumes were analyzed for total suspended solids (APHA 1992, 2540D), particulate organic matter (APHA 1992, 2540E), and chloride content.

Water Balance Calculations

Pond effluent estimates and Southern Regional Climate Center precipitation and temperature data were used to calculate a water balance (Thornthwaite 1957) in mm/day for the study area where:

$$HLR_{eff} + G_{in} + P + F_i = E + O_s + G_{out} + S_{wb} \quad (1)$$

where: HLR_{eff} = hydraulic loading rate to study area (pond discharge rate/wetted surface area); G_{in} = ground-water inflow to surface water; P = Precipitation; F_i = Backwater flood inflow from the Vermillion River; E = Potential Evapotranspiration; O_s = Surface-water outflow; G_{out} = Surface-water seepage into ground-water table; and S_{wb} = Surface-water and bank storage.

Ground-water inflow to surface water and surface-water seepage into ground water were considered negligible due to the presence of relatively impermeable underlying soils (Terzaghi and Peck 1967, Breaux 1992). No backwater flooding was observed during the study period. Surface-water and bank storage was insignificant based on observed changes in surface-water elevations. Therefore, equation (1) reduces to:

$$HLR_{eff} + P = E + O_s \quad (2)$$

Analysis of Flow

Quantitative analysis of flow within the wetland was needed to estimate nutrient transport through the wetland. Due to the presence of channels, vegetation, hummocks, and other obstructions, total flow (Q_t) through any cross-section of the study area was divided into a number (N) of separate sub-sections of flow (q_i) so that:

$$Q_t = \sum_{i=1}^{N} q_i \quad (3)$$

This relationship allowed the use of the Manning equation to calculate the relative conveyance (K) of flow through separate submerged cross-sections (A_{xi}) along water-surface contours of equal gradient (Henderson 1966) where:

$$q_1/K_1 = q_2/K_2 = q_3/K_3 = \ldots q_N/K_N \quad (4)$$

and

$$K_i = [A_{xi}\ R^{2/3}]/n_i \quad (5)$$

Because cross-sectional widths were orders of magnitude greater than depths, the hydraulic radius (R) was set equal to the average water depth for each subsection. Surface-water gradients and depths for each event were estimated from contours generated from topographic survey and staff gage data using SURFER V6.04 (Golden Software 1997). Hydraulic roughness

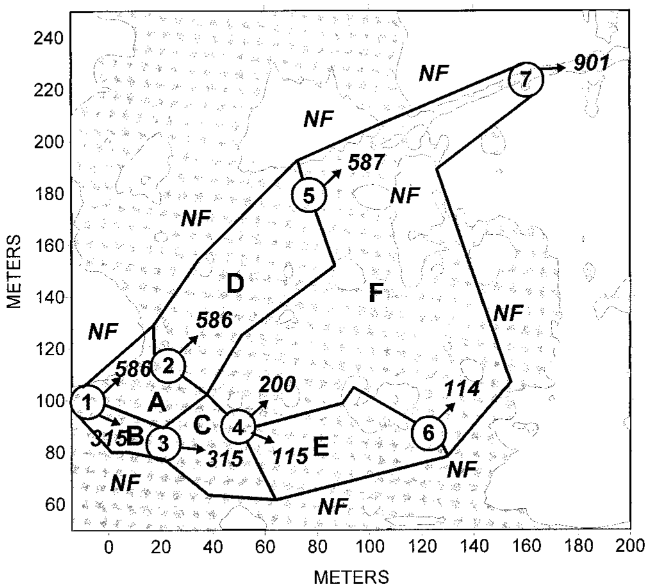

Figure 3. Example treatment zone distribution used in calculating material flux rates within the study area. Example shown is for July 10th. Each lettered polygon represents a separate treatment zone. Each numbered circle indicates a water quality sampling location. Shading represents wetted surface area. 'NF' boundaries indicate dry or stagnate areas of no observable flow.

coefficients (n) were estimated using techniques presented by Cowan (1956). Tracer study results were used to omit storage and other areas of no observable flow from consideration in conveyance calculations.

The water residence time is normally expressed in units of days and is the ratio of the volume of water in the wetland divided by the volumetric flow rate of water through the wetland. Pond effluent-water residence times (RT) for the study area were calculated using observed pond effluent rates (Q_{pond}) and wetted volume (V) estimates generated from water surface and topographic elevations where:

$$RT = V/Q_{pond} \quad (6)$$

Material Flux

Material flux ($g \cdot m^{-2} \cdot d^{-1}$) refers to the rate of change in material as it moves into and out of a given area. For each of the four study periods, the study area was broken into multiple treatment zones based on representative sampling locations, tracer study results, and the water elevation gradients. Figure 3 shows a typical layout of treatment zones within the study area.

Flux was calculated for each zone using water quality sampling locations for representative inflow and outflow concentrations and conveyance calculation results for representative inflow and outflow rates ($m^3 \cdot d^{-1}$). Most zones included a singular inflow and singular outflow reference point, so that percent change in concentration and material flux through these zones were equal to:

$$\% \text{ Change} = \Delta C/C_{in} \quad (7)$$

and

$$\text{Flux} = \Delta C \cdot Q_z/A_z \quad (8)$$

where: $\Delta C = C_{in} - C_{out}$ = influent concentration ($g \cdot m^{-3}$)—effluent concentration ($g \cdot m^{-3}$); Flux = material flux through treatment zone, positive and negative values indicate net removal and production, respectively ($g \cdot m^{-2} \cdot d^{-1}$) Q_z = flow through treatment zone ($m^3 d^{-1}$); A_z = area of treatment zone (m^2)

Each event also included treatment zones with three separate inflows. For these zones (e.g., Figure 3, Zone F), percent change in concentration was weighted based on the contribution of each inflow to the total flow through the zone and the concentration specific to each inflow where:

$$\% \text{ Change} = \frac{(Q_1 \cdot \Delta C1/C1_{in} + Q_2 \cdot \Delta C2/C2_{in} + Q_3 \cdot \Delta C3/C3_{in})}{Q_1 + Q_2 + Q_3} \quad (9)$$

Flux through multiple-inflow zones was calculated by summing the unit flux rates for each inflow so that:

$$\text{Flux} = Q1_z \cdot \Delta C1/A1_z + Q2_z \cdot \Delta C2/A2_z + Q3_z \cdot \Delta C3/A3_z \quad (10)$$

Areas (Ai_z) in Eq. 9 represent only the portion of the treatment zone over which the accompanying flow (Qi_z) passed. Note that for all multiple-inflow zones, there was always a singular representative outflow point so that:

$$C1_{out} = C2_{out} = C3_{out} \quad (11)$$

Statistical Analyses

Correlations run on data generated during this study were made using simple linear regression techniques. Statistical results were generated using Statistical Analytical Software (SAS). The level of significance (*p*-value) corresponds to the rejection area in a two-tailed test. The coefficient of determination (r^2) is also given for each correlation.

RESULTS

Water Balance

For the period between June 01 and July 10, 1996, water balance results show that pond effluent was diluted very little by rainfall or other water sources and was relatively conserved through the study area. When compared to rainfall (434 mm) and estimated potential evapotranspiration (292 mm), pond effluent (1235 mm) generated 90 percent of the 1377 mm surplus for the entire study period. Water balance calculations showed 24 percent of study area surface-water could be removed by evapotranspiration within the study area. Assuming some rainfall and storm water storage are available for evapotranspiration, surface flow was between 76 and 100 percent conserved through the study area.

Surface-water Gradients and Depths

Pond effluent pooled in the area near the discharge pipe and moved most readily through the study area in two channel systems. A primary channel system began south of the discharge pipe and generally increased in depth and width to its exit point from the study area. The primary channel was generally 3 to 6 m wide and 10 to 15 cm lower than the surrounding topography. The remaining outflow followed a less-defined secondary channel system north of the discharge pipe, moving slowly through a much larger area until discharging at several points back into the primary channel. Secondary channels were generally 1 to 4 m wide and 5 to 10 cm lower than surrounding topography. Surface-water elevation gradients were generally constant for each study period. For each event, surface water moved radially away from the discharge point about 30 m before forming a more uniform southerly gradient (0.8 to 1.2 m/km). Maximum surface-water depths were generally <25 cm and were found near the middle of primary channels. In general, surface-water areas less than 5 cm deep were stagnant.

Hydraulic Loading Rates and Water Residence Times

RTs ranged from 0.9 to 1.2 days, and simple linear regression results showed decreases in RT with increases in HLR ($r^2 = 0.82$; $P = 0.09$). HLRs and RTs adjusted by using only channel areas and volumes (estimated as areas with >5 cm surface-water depth) were slightly less than unadjusted RTs and showed stronger correlation ($r^2 = 0.98$; $P = 0.01$) with respect to the HLR.

Tracer Study Results

Dye tracer results showed similar pathways, and travel times correlated well with changes in pond effluent rates. Tracer study results (Figure 4) showed flow paths quickly diverging about 15 m from the discharge pipe. In general, the dye plume followed the deeper, channelized areas where vegetation was sparse

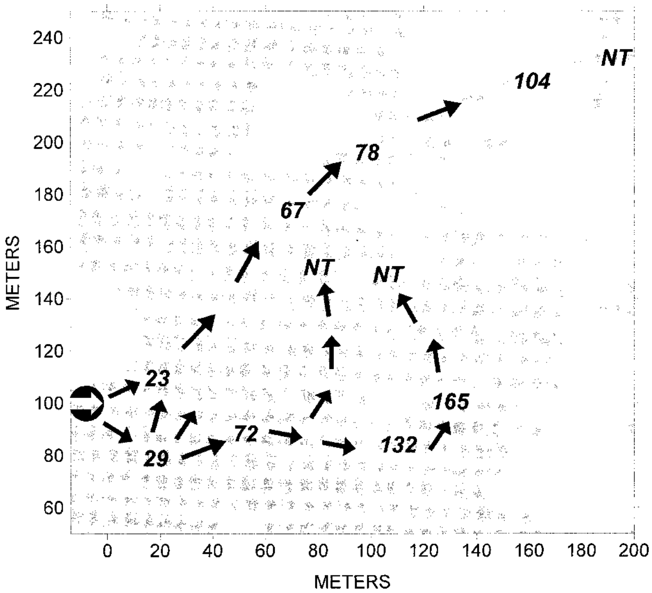

Figure 4. July 10 tracer study results with arrows showing typical primary dye flow paths. Numbers show minimum elapsed travel times in minutes from the pond discharge pipe. Travel times were proportionately shorter or longer for study periods of higher or lower discharge, respectively. 'NT' indicates the location where diffusion, rather than advection, begins to dominate the movement of the leading edge of the dye plume.

or flow was not slowed by fallen trees or other debris. Linear regressions showed minimum dye travel times to similar reference points (Table 2) decreased as pond discharge rates increased.

Flow Distribution

Estimated roughness coefficients ranged from 0.1775 in shallowest areas of flow (0 to 5 cm) to 0.0475 in the deepest channel areas (>15 cm). Their reflection of higher resistance to flow in shallow areas corresponds to field observations where slowest velocities were seen in the shallow areas outside of the channels.

The majority of flow moved through the primary channel area. Conveyance calculations showed 58 to 66 percent of the surface-water flow moved across about 15 to 25 percent of the total wetted surface area and only 10 to 12 percent of the total study area. The July 10 conveyance map (Figure 5) shows typical flow distributions.

Water Quality Concentration Gradients

Water quality concentration maps (Figures 6A-E) show distributions of total suspended solids (TSS), particulate organic matter (POM), ammonium, phosphate, and nitrate. Percent concentration decreases from the peak observed concentration to the study area outflow are in Table 3. TSS and POM both generally decreased in concentration from inflow to outflow. However, concentration peaks were observed 20 to 30 m downstream from the pond discharge pipe for TSS on July 01 and for POM on June 11. Ammonium and

Table 2. Mean pond effluent rates (Q_{pond}) and minimum dye travel times in minutes to similar reference points within the study area. Station I.D. numbers refer to [Y + X] grid shown on Figure 2. r^2 and p-val refer to regression results of minimum travel times vs. Q_{pond}.

		Minimum Elapsed Travel Time (minutes) from Discharge Pipe (Station 100 + −15) to:			
	Q_{pond}	Station	Station	Station	Station
June 11	502	44	39	111	NA
June 18	901	18	26	68	106
July 01	1164	10	22	56	88
July 10	901	29	23	67	104
	r^2	0.91	0.88	0.94	0.99
	p-val	0.05	0.06	0.06	0.03

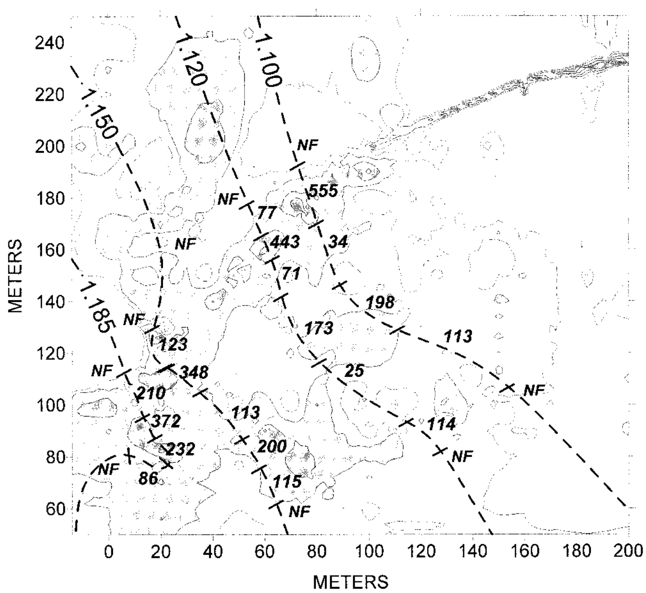

Figure 5. Example conveyance map showing the relative distribution of total cross-sectional flow through the study area. Dashed lines are water surface elevation contours in meters (msl). Labeled values indicate the amount of flow ($m^3 \cdot d^{-1}$) across the labeled portion of each dashed contour. Shading represents water depths in 5 cm intervals. 'NF' (noflow) indicates the terminal end of each conveyance cross-section based on zero water depth or tracer study results. Results show that the deeper channel areas convey the majority of the study area flow.

phosphate concentrations generally peaked 50 to 75 m downstream from the discharge pipe. Concentration peaks were followed by steady drops in ammonium and phosphate concentrations to the study area outflow. Nitrate concentration peaks occurred at the discharge pipe on June 18 and July 10 and 20 to 30 m downstream of the discharge pipe on June 11 and July 01. Peaks were generally followed by drops in concentration, although concentrations increased near the study area outflow on July 01.

Material Flux Rates

Percent removal vs. loading rate plots (Figures 7A–7E) indicate that removal efficiencies are generally dependent on loading rates or proximity to the discharge pipe. In areas of net removal (>0% removal), removal efficiencies decreased with increases in loading for all compounds except nitrate. TSS and POM plots show a more gradual drop in removal efficiency with increases in load than do ammonium and phosphate, which generally decreased exponentially with increases in load. Nitrate removal efficiency was generally greater than 75% at loading rates above 0.2 $g \cdot m^{-2} \cdot d^{-1}$. Net production (<0% removal) occurred in areas where either loading rates were generally near natural background levels or the treatment zone was located near (<50 m) the discharge pipe.

Linear regressions of net nitrate removal rates vs. nitrate loading rates (Figure 8) from areas showing nitrate decreases indicate that removal rates increase with increases in loading rates. Unadjusted data indicate that the wetland is capable of removing nitrate at a rate over 45% that of the applied loading rate. An outlier in the bottom right portion of Figure 8 represents data taken near the pond discharge pipe where high dissolved oxygen levels likely inhibited denitrification. Omitting the outlier, removal rates increase to 80% of the applied loading rate.

Ammonium flux vs. POM flux plots (Figure 9) show that the majority of ammonium production occurred concurrent with POM removal. Moreover, the majority of ammonium production occurred in areas less than 50 m from the discharge pipe where POM concentrations were high.

DISCUSSION AND CONCLUSIONS

Hydrology

Water balance results from this study indicated that between 76 and 100 percent of the total pond discharge flow was conserved through the study area. This agrees with Hey et al. (1994), who found that 81 to 99% of the total flow was conserved in constructed treatment wetlands with similar underlying soils and similar climates. Estimated roughness coefficients in and out of channel areas fall within the range of published values (Arcement and Schneider 1989, Hall and Freeman 1994).

Dye travel pathways through the study area indicated that large wetted areas of the wetland were effectively stagnant and acted primarily as no-flow storage areas. Such areas showed little mixing with dye in nearby flowing areas and are similar to the gross areal inefficiencies found in other natural wetland treatment systems (Chen and Wang 1994, Kadlec 1994). Observed minimum dye travel times through the study area were an order of magnitude less than volume-based water residence time estimates. This disparity is similar to the wide range of mixing and residence time distributions that are found in other wetland treatment systems (Kadlec 1994).

Flow distributions in the study area showed strong preferential pathways with the majority of flow moving across only 10 to 12 percent of the study area. Such non-uniform distributions are common in natural wetlands (Hammer and Kadlec 1986, Kadlec and Knight 1996) and can significantly influence water quality distributions and removal efficiencies (Kadlec 1994). Although water residence time is not likely equal to actual mean residence time in the wetland (Werner and Kadlec 1996), it is often used as an initial estimator of residence time where water residence time (RT) estimates the area needed in either designing constructed wetlands or planning discharges to existing ones. Our results indicate that under non-flooded conditions, better estimates of RT can be made based on hydraulic loading rates calculated using only channel areas and volumes as opposed to total wetted surface area.

Water Quality Concentration Distributions

Water quality distributions show a general decrease in suspended solid and inorganic nutrient concentrations through the study area (Figures 6A–6E). Observed percent removals (Table 3) from peak observed concentration to the study area outflow ranged from 32 to 95% for nitrate, 13 to 37% for ammonium, 3 to 50% for phosphate, 48 to 91% for TSS, and 60 to 84% for POM and compare well with other wetland tertiary treatment systems (Knight et al. 1987, WPCF 1990, Hiley 1995).

Phosphorus dynamics in wetlands are controlled by redox potential, pH, soil minerals, and the amount of native soil P (Faulkner and Richardson 1989). Most phosphorus retained in wetlands is in soils where a number of precipitation and sorption reactions are important. Phosphorus retention in wetlands is strongly

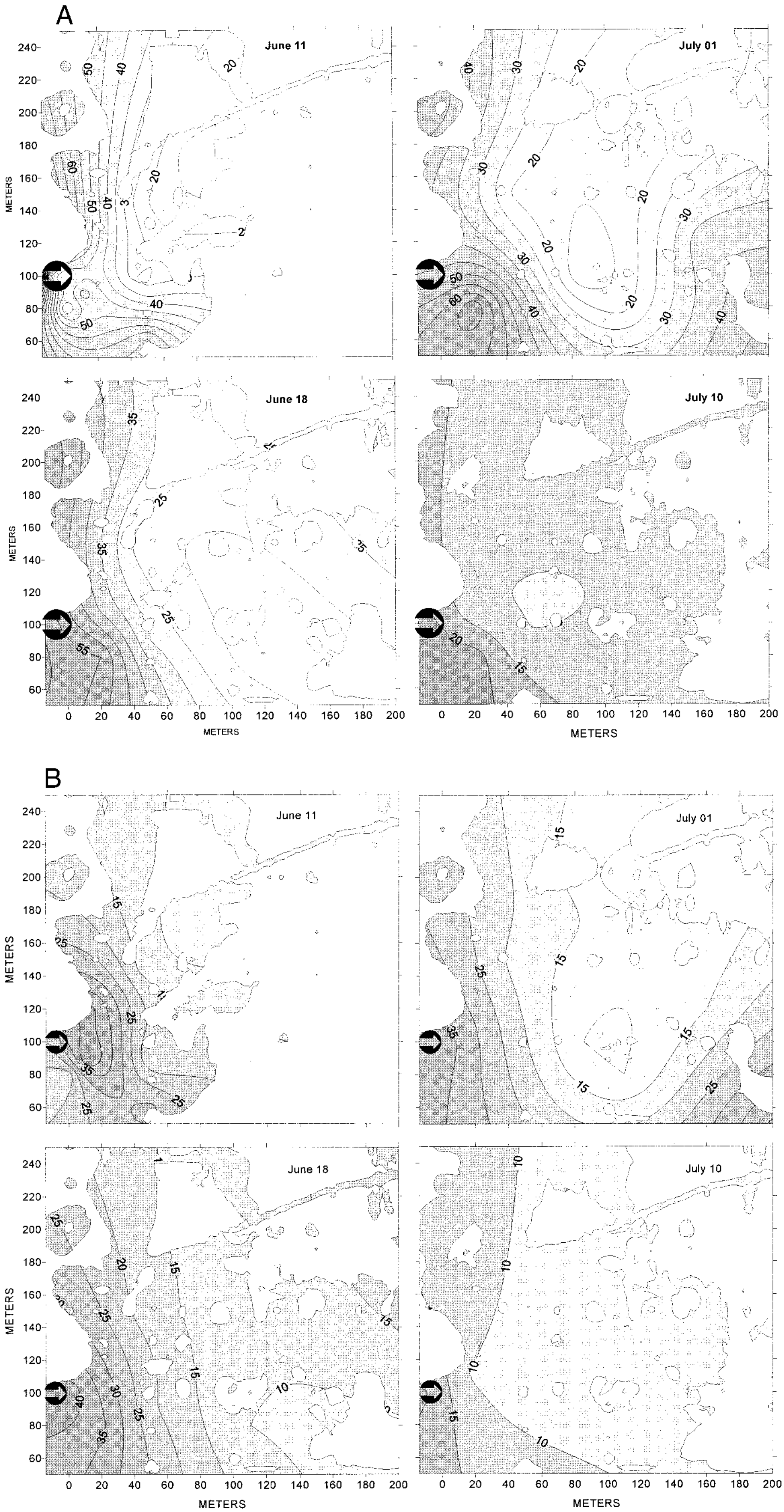

Figure 6. A. Total Suspended Solids (TSS) concentration contours (mg/L) for each sampling event. B. Particulate Organic Matter (POM) concentration contours (mg/L) for each sampling event.

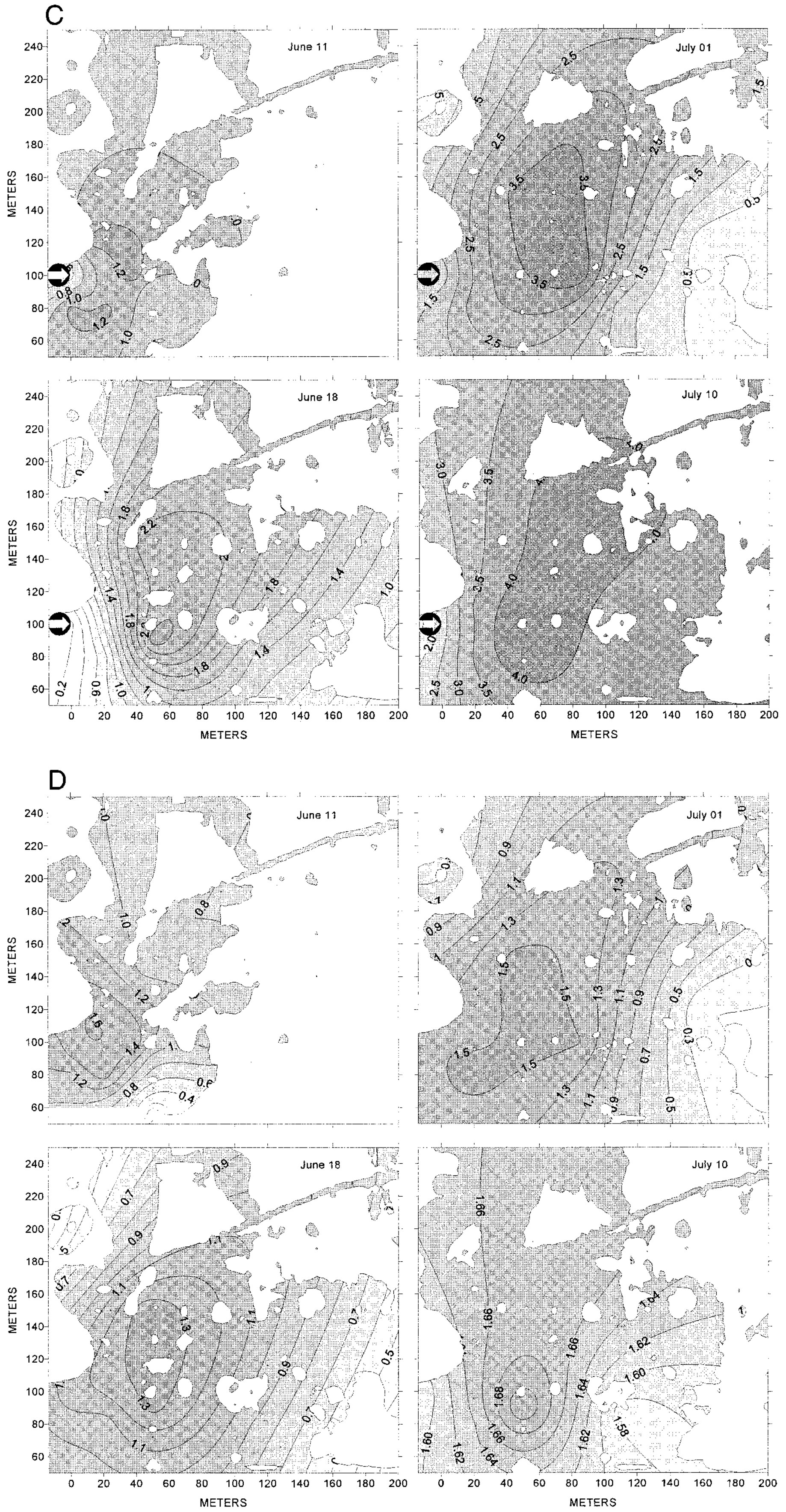

Figure 6. Continued. C. Ammonium concentration contours (mg/L) for each sampling event. D. Phosphate concentration contours (mg/L) for each sampling event.

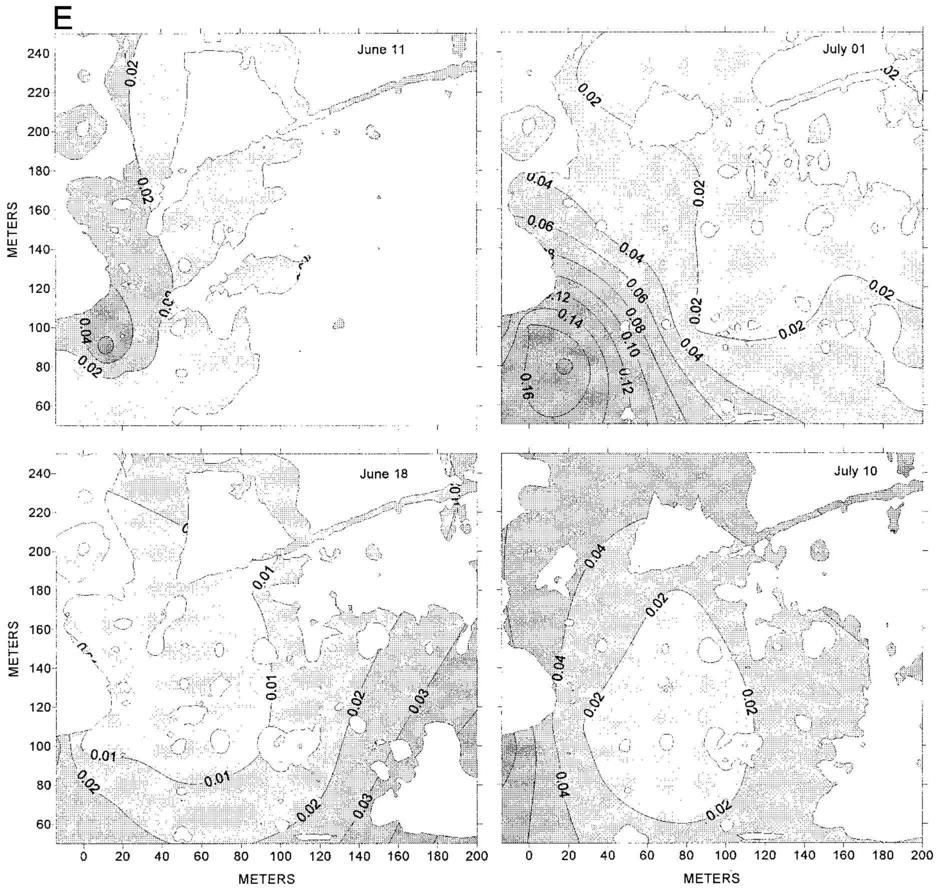

Figure 6. Continued. E. Nitrate concentration contours (mg/L) for each sampling event.

dependent on loading rates. Phosphorous retention efficiency is typically 65 to 95% at loading rates less than 5 g$\cdot$ m$^{-2}\cdot$ yr^{-1} (Faulkner and Richardson 1989).

Nitrate and ammonium distributions reflect typical sequential nitrogen cycling processes (Reddy et al. 1980), and their spatial distributions within the study area imply certain processes are linked to proximity to the oxidation pond discharge pipe. Similar to our results were those reported by Garside et al. (1976), where nitrate concentrations increased after discharge from an oxidation pond and were greatest immediately downstream from the discharge pipe. This concentration peak near the discharge pipe was likely due to nitrification resulting from a high availability of ammonium in the pond effluent and to increased oxygenation from mixing during discharge (Atlas and Bartha 1981). Decreases in nitrate (Figure 6E) further downstream from the discharge pipe were likely largely due to denitrification. As distance from the discharge pipe increased and POM concentrations decreased, dissolved oxygen decreased and the system approached anoxia. Under anoxic conditions, nitrate is used more as an electron acceptor than as a nutrient source (Patrick 1982), and denitrifying organisms are favored.

Denitrification is limited by organic carbon availability, redox potential, pH, temperature, and nitrate concentrations (Patrick 1982, Faulkner and Richardson 1989, Gearheart 1992), so multiple factors may be limiting nitrate removal processes in the water column and sediments. For example, low nitrate removal efficiencies (<50%, Figure 7E) generally coincide with either low nitrate loading rates (<0.01 g m^{-2} d^{-1}), locations near (<50 m) the pond discharge pipe where oxygen from mixing during discharge inhibits denitrification (and enhances nitrification), or low POM concentrations found in areas far from the discharge pipe. However, where nitrate loads are higher (>0.02 g m^{-2} d^{-1}) and further (>50 m) from the discharge pipe, nitrate removal efficiency was generally above 75% for the range of observed loading rates.

Table 3. Percent concentration removal from peak concentration location to study area outflow.

Date	NO_3	NH_4	PO_4	TSS	POM
June 11	85%	23%	50%	91%	84%
June 18	72%	13%	27%	53%	60%
July 01	95%	37%	25%	77%	73%
July 10	32%	15%	3%	48%	73%

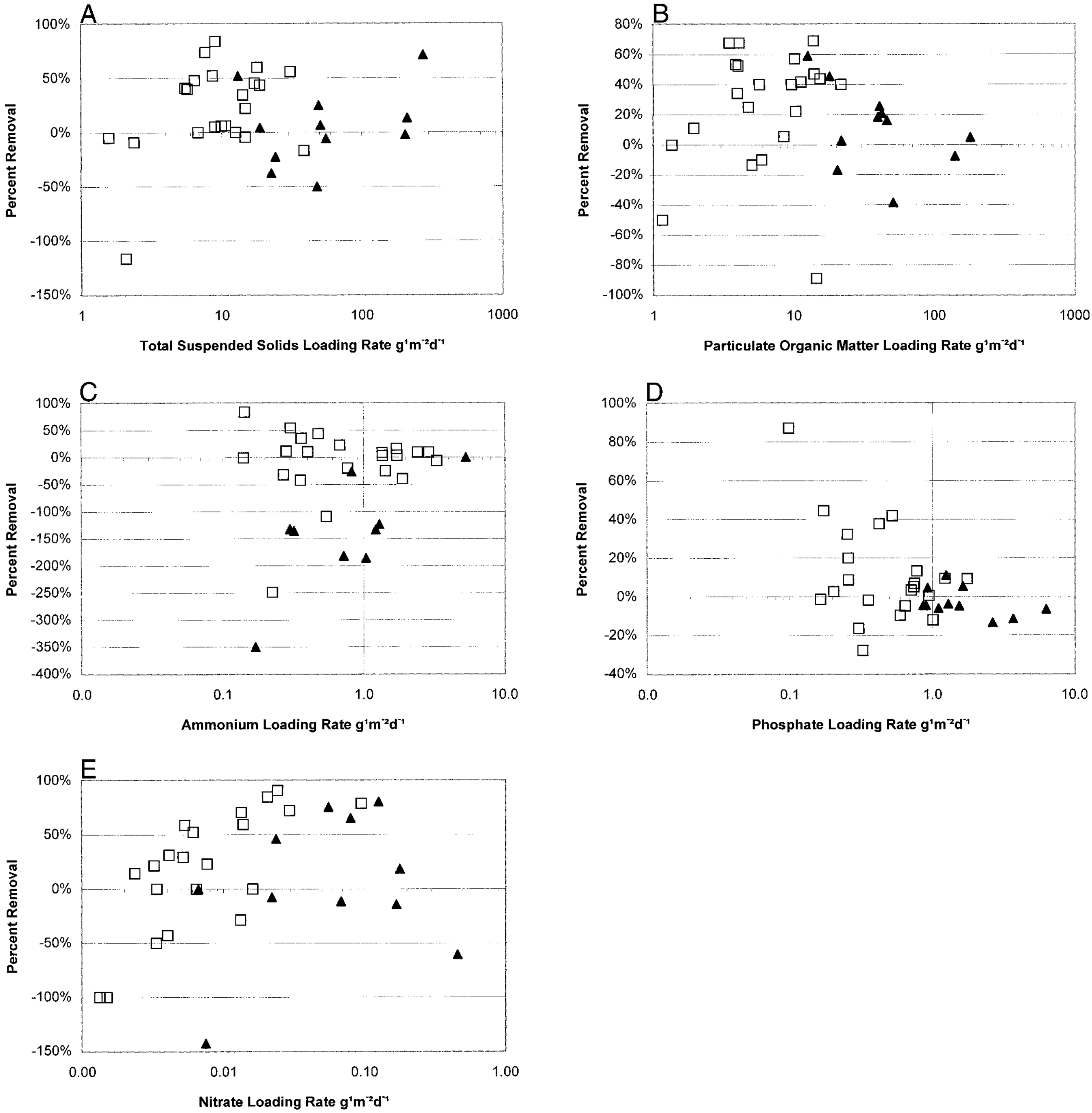

Figure 7. A. Percent Removal of Total Suspended Solids vs. Total Suspended Solids Loading Rates ($g \cdot m^{-2} \cdot d^{-1}$). B. Percent Removal of Particulate Organic Matter vs. Particulate Organic Matter Loading Rates ($g \cdot m^{-2} \cdot d^{-1}$). C. Percent Removal of Ammonium vs. Ammonium Loading Rates ($g \cdot m^{-2} \cdot d^{-1}$). D. Percent Removal of Phosphate vs. Phosphate Loading Rates ($g \cdot m^{-2} \cdot d^{-1}$). E. Percent Removal of Nitrate vs. Nitrate Loading Rates ($g \cdot m^{-2} \cdot d^{-1}$). Squares and triangles represent areas greater than and less than 50 meters from the discharge pipe, respectively.

Similar to other treatment wetlands (Tiedje 1988, Zhang 1995), ammonium concentrations increased immediately downstream (Figure 6C) from the discharge pipe, which is likely due to remineralization from oxidation of organic matter. Remineralization of ammonium by dying plants and microbes probably account for the observed increases (Kemp 1989). Tiedje (1988) proposed that in the carbon-rich environment near a pond discharge pipe, facultative microorganisms adapted to both fermentative metabolism and dissim-

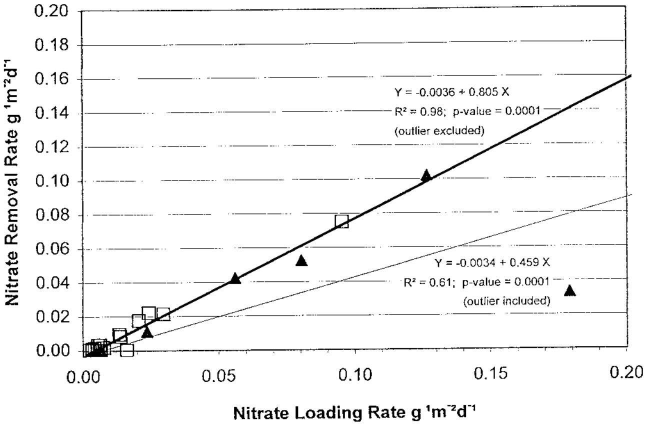

Figure 8. Net Nitrate Removal Rates ($g \cdot m^{-2} \cdot d^{-1}$) vs. Nitrate Loading Rates ($g \cdot m^{-2} \cdot d^{-1}$). The outlier in the lower right corner of the figure is near the pond discharge pipe where net nitrate production was typically observed.

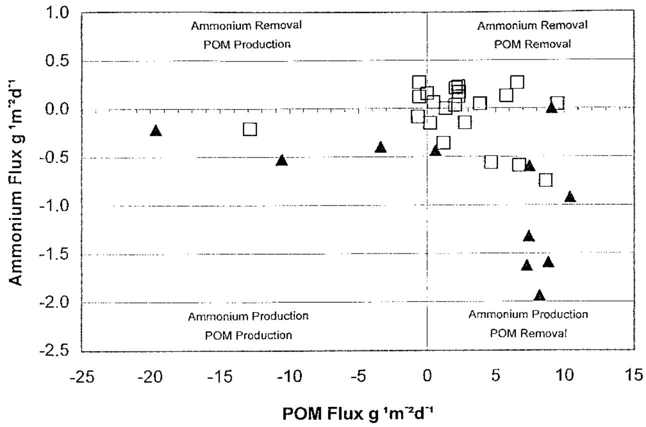

Figure 9. Ammonium Flux ($g \cdot m^{-2} \cdot d^{-1}$) vs. Particulate Organic Matter Flux ($g \cdot m^{-2} \cdot d^{-1}$). Positive and negative indicate net removal (+) or net production (−) respectively. Squares and triangles represent areas greater than and less than 50 meters from the discharge pipe, respectively.

ilatory pathways to ammonium are favored. After reaching a peak concentration shortly downstream from the discharge pipe, ammonium concentrations generally decreased with increases in retention time and distance from the peak, agreeing with patterns seen at natural and constructed treatment wetlands (Kadlec and Knight 1996) where ammonium concentrations showed exponential decreases relative to increases in residence time and travel distance.

In general, suspended solid and nutrient concentrations decreased more rapidly outside of the primary channel areas. These greater rates of decrease can be attributed to longer residence times per unit area due to decreased velocities and greater surface area to water volume ratios inherent to shallower depths.

Water Quality Removal Efficiencies and Flux Relationships

The decrease in removal efficiency with increases in loading rates shown in phosphate and ammonium percent removal vs. loading rate plots (Figures 7C and 7D) were similar to total P and N removal efficiencies in studies by Richardson and Nichols (1985) and Faulkner and Richardson (1989). Our results are more variable, however, than those presented in these two studies. Both the Richardson and Nichols and the Faulkner and Richardson studies present results for overall treatment efficiency for various wetland systems. Our data include points in and out of channel areas and at varying distance from a discharge pipe. Thus, even though there is a general decrease in efficiency with increasing loading, our results show the kind of variability that can occur within a single treatment wetland.

When compared to loading rates at similar sites (WPCF 1990, Kadlec and Knight 1996), average loading rates (Table 4) to the overall study area were much lower for nitrate and ammonium, similar for phosphate, and greater for TSS. Calculated loading rates based on the wetted area were higher, but comparisons with other studies should be based on total area since most studies calculate loading rate to the total wetland. The maximum loading rate to a single cell was much higher than the average loading rate to the total wetland entire. As expected in most point discharge wetland treatment systems, these maximum loading rates shown in Table 4 were observed in the areas nearest the effluent discharge point. By comparison, the av-

Table 4. Loading Rates ($g^{-2} \cdot d^{-1}$) for nitrate, ammonium, phosphate, and total suspended solids (TSS) calculated for the entire 4.0 ha study area, for the wetted area alone, for the treatment cell with the maximum observed loading rate, and comparisons to other studies averages calculated for their entire receiving wetland.

Parameter	Total Area[a] Avg.	Wetted Area[b] Avg.	Treatment Cell[c]	Other Studies Avg.
Nitrate	0.0023	0.0035	0.46	0.14[d]
Ammonium	0.072	0.11	5.31	0.22[d]
Phosphate	0.034	0.057	6.32	0.031[e]
TSS	1.4	2.5	274.5	0.24[e]

[a] Total study area was based on peak observed concentration for each event averaged over four hectares and included hummocks and other large non-submerged areas.
[b] Wetted area estimates included only submerged portions of the total study area.
[c] Maximum observed loading rate to a single treatment cell during the study.
[d] Kadlec and Knight (1996).
[e] WPCF (1990).

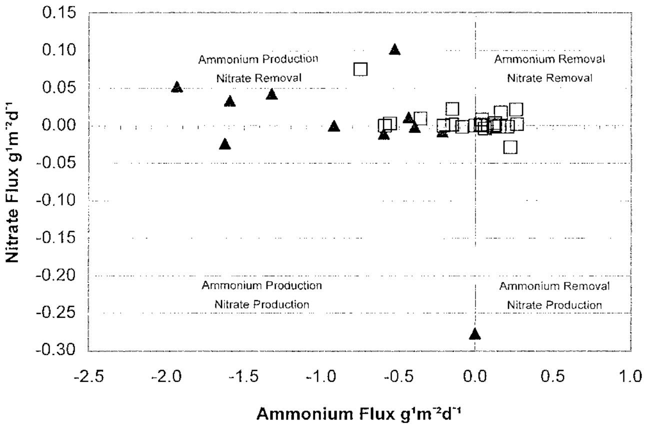

Figure 10. Nitrate Flux ($g \cdot m^{-2} \cdot d^{-1}$) vs. Ammonium Flux ($g \cdot m^{-2} \cdot d^{-1}$). Positive and negative indicate net removal (+) or net production (−) respectively. Squares and triangles represent areas greater than and less than 50 meters from the discharge pipe, respectively.

erage loading rate to the entire 1295 ha forested wetland at Breaux Bridge is about 0.0051 $g \cdot m^{-2} \cdot d^{-1}$ nitrogen and 0.0026 $g \cdot m^{-2} \cdot d^{-1}$ phosphorus. These data indicate that there is a relatively high assimilation of the effluent at Breaux Bridge within the first 200 m of the discharge point.

Because nitrate removal rates can be highly correlated to loading rates (Kadlec 1988), comparisons were made by event. Similar to other studies (Knight et al. 1987, Gale et al. 1994, Phipps and Crumpton 1994), nitrate removal rates increased linearly with respect to loading rate (Figure 8) in areas of net nitrate removal.

Algae present in oxidation pond discharge represent a nutrient-rich, suspendable solid available for transport and capable of affecting nutrient distributions across the wetland (Kadlec 1988). Figure 9 shows that most zones with net ammonium production coincided with zones with net POM reduction. Moreover, Figures 6B and 6A show that ammonium concentrations remained high in the presence of high concentrations of POM and generally did not begin to decrease until POM concentrations dropped below 10 to 15 mg/L. This implies that decomposition and mineralization of POM played a primary role in ammonium production in the wetland, and ammonium levels will not decrease as long as high concentrations of POM remain. Heavy shading in the study area inhibits algal growth and allows for net decreases in POM to occur. However, this interaction may be more important in less shaded wetlands where algae persist for longer distances and inhibit net decreases in ammonium.

Comparisons of nitrate flux rates to ammonium flux rates (Figure 10) show that some of the highest values of ammonium production (<-1.0 $g \cdot m^{-2} \cdot d^{-1}$) coincided with the highest observed nitrate removal rates (>0.02 $g \cdot m^{-2} \cdot d^{-1}$). This implies that nitrate removal processes may also include dissimilatory reduction to ammonium. As much as 34% of nitrate transformation in treatment wetlands has been attributed to ammonium production (Cooke 1994).

Implications for Waste-Water Management

The results indicate that the 4-ha study area and the surrounding 1295-ha Cypriere Perdue wetland are capable of treating much greater hydraulic and nutrient loads than at present, and that a site-specific monitoring and discharge plan that included minor pretreatment or alterations in hydrologic features could be designed to accommodate any changes in discharge.

Because nitrate removal rates appear to increase linearly with nitrate loading rates and ammonium removal efficiency decreases with increases in loading rates, conversion of total nitrogen to nitrate prior to discharge would likely increase total N removal. Pretreatment oxidation options are relatively inexpensive and could include simple modifications such as discharging pond effluent over a broad weir or spillway as opposed to the present pipe-discharge system.

If a reduction in total wetland area required to treat the waste-water became necessary, channels near the discharge pipe could be easily dammed with earth or other bulky material to force water out of the channels. This action would increase wetland surface contact area, evapotranspiration, and retention time within the wetland, all of which would lead to increased constituent removal rates.

If total treatment area is not a limitation, additional discharge could be applied that would only affect the distance within the wetland required to assimilate the additional load. Because the oxidation ponds are located near the upper end of the Cypriere Perdue wetland, straight line travel distance to the nearest downstream receiving body of water is about 4 km. Not accounting for channel meander and increased average width of flow expected from additional discharge, which would further increase wetland treatment efficiency, discharge rates could be increased an order of magnitude, and water quality concentrations would still likely attenuate to natural levels within 2 km of the pond discharge point.

Under any discharge program, careful attention should be paid to channel locations for monitoring of surface-water quality. Because of their potential for material transport far faster than estimated water residence times, channel areas are the most likely portion of a treatment wetland to convey water to receiving bodies prior to meeting water quality standards for discharge.

LITERATURE CITED

American Public Health Association (APHA) 1992. Standard Methods for the Examination of Water and Wastewater. 18th Edition. APHA. Washington, DC, USA.

Arcement, G. J. and V. R. Schneider. 1989. Guide for selecting Manning's Roughness Coefficients for Natural Channels and Flood Plains Manning's n. United States Geological Survey Water Supply Paper 2339.

Atlas, R. and R. Bartha. 1981. Microbial Ecology. Addison-Wesley Publishing Company, Reading, UK.

Breaux, A. 1992. The use of hydrologically altered wetlands to treat wastewater in coastal Louisiana. Ph.D. Dissertation. Louisiana State University, Baton Rouge, LA, USA.

Chen, S. and G. T. Wang. 1994. A theoretical model for simulating BOD5 removal in constructed wetlands. Proceedings from the Water Environment Federation Conference 8:25–35.

Cooke, J. G. 1994. Nutrient transformations in a natural wetland receiving sewage effluent and the implications for waste treatment. Water Science and Technology 29:209–217.

Cooper, P. F. and B. C. Findlater (eds.). 1990. Constructed Wetlands in Water Pollution Control. Pergammon Press. Oxford, UK.

Cowan, W. L. 1956. Estimating hydraulic roughness coefficients. Agricultural Engineering 37:473–475.

Day, J. W., Jr., A. M. Breaux, S. Feagley, P. Kemp, and C. Courville. 1994. Use attainability analysis of long-term wastewater discharge on the Cypriere Perdue Forested Wetland at Breaux Bridge, LA. City of Breaux Bridge, LA, USA.

Faulkner, S. and C. Richardson. 1989. Physical and chemical characteristics of freshwater wetland soils. p. 41–72. In D. Hammer (ed.) Constructed Wetlands for Wastewater Treatment. Lewis Publishers, Chelsea, MI, USA.

Gale, P. M., K. R. Reddy, and D. A. Graetz. 1994. Phosphorous retention by wetland soils used for treated wastewater disposal. Journal of Environmental Quality 23:370–377.

Garside, C., T. C. Malone, O. A. Roels, and B. A. Sharfstein. 1976. An evaluation of sewage derived nutrients and their influence on the Hudson estuary in New York light. Estuarine Coastal Marine Science 4:281–289.

Gearheart, R. 1992. Use of constructed wetlands to treat domestic wastewater, City of Arcata, California. Water Science and Technology 26:1625–1637.

Godfrey, P. J., E. R. Kaynor, S. Pelczarski, and J. Benforado (eds.). 1985. Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. Van Nostrand Reinhold, New York, NY, USA

Golden Software. 1997. SURFER V.6.04 Golden Software, Inc. Golden, CO, USA.

Hammer, D. E. and R. H. Kadlec. 1986. A model for wetland surface-water dynamics. Water Resources Research 22:1951–1958.

Hammer, D. (ed.). 1989. Constructed Wetlands for Wastewater Treatment. Lewis Publishers, Chelsea, MI, USA.

Henderson, F. M. 1966. Open Channel Flow. Prentice-Hall, Inc. Englewood Cliffs, NJ, USA.

Hall, B. R. and G. Freeman. 1994. Study of hydraulic roughness in wetland vegetation takes new look at Manning's n. U.S. Army Corps of Engineers Waterways Experiment Station, Wetlands Research Program Bulletin 4:1–4.

Hesse, I. 1994. Dendroecological determination of municipal wastewater effects on *Taxodium distichum.* Ph.D. Dissertation. Louisiana State University, Baton Rouge, LA, USA.

Hesse, I., J. Day, and T. Doyle. 1998. Long-term growth enhancement of bald cypress (Taxodium distichum) from municipal wastewater application. Environmental Management 22:119–127.

Hey, D., K. Barrett, and C. Biegen. 1994. The hydrology of four experimental constructed marshes. Ecological Engineering 3:319–343.

Hiley, P. D. 1995. The reality of sewage treatment using wetlands. Water Science Technology 32:329–338.

Kadlec, R. H. 1988. Modeling nutrient behavior in wetlands. Ecological Modeling 40:37–66

Kadlec, R. H. 1994. Detention and mixing in free water wetlands. Ecological Engineering 3:1–36.

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

Kemp, W. M. 1989. Estuarine Chemistry. p. 79–143. In J. W. Day, Jr., C. A. S. Hall, W. M. Kemp, and A. Yanez-Arancibia (eds.) Estuarine Ecology. John Wiley & Sons, Inc. New York, NY, USA.

Knight, R. L., T. W. McKim, and H. R. Kohl, 1987. Performance of a natural wetland treatment system for wastewater management. Journal of the Water Pollution Control Federation 59:746–754.

Knight, R. L. 1994. Treatment wetlands database now available. Water and Environment Technology 6(2):31–33.

Mitsch, W. J. and J. G. Gosselink. 1993. Wetlands. Van Nostrand Reinhold, New York, NY, USA.

NADB (North American Treatment Wetland Database). 1993. Electronic database created by R. Knight, R. Ruble, R. Kadlec, and S. Reed for the USEPA.

Patrick, W. H., Jr. 1982. Nitrogen Transformation in submerged soils. p. 449–465. In F. J. Stevenson (ed.) Nitrogen in Agricultural Soils. Soil Science Society of America, Madison, WI, USA.

Phipps, R. G. and W. G. Crumpton. 1994. Factors affecting nitrogen loss in experimental wetlands with different hydrologic loads. Ecological Engineering 3:399–408.

Reddy, K. R., P. D. Sacco, and D. A. Graetz. 1980. Nitrate reduction in an organic soil-water system. Journal of Environmental Quality 9:283–288.

Richardson, C. J. and D. S. Nichols. 1985. Ecological analysis of wastewater management criteria in wetland ecosystems. In P. J. Godfrey (ed.) Ecological Considerations in Wetland Treatment of Municipal Wastewaters. Van Nostrand Reinhold, New York, NY, USA.

Terzaghi, K. and R. B. Peck. 1967. Soil Mechanics in Engineering Practice, 2nd ed. John Wiley and Sons, Inc., New York, NY, USA.

Thornthwaite, C. W., 1957. Instructions and tables for computing potential evapotranspiration and the water balance. Laboratory of Climatology, Centerton, NJ, USA.

Tiedje, J. M. 1988. Ecology of denitrification and dissimilatory nitrate reduction to ammonium. p. 179–244. In A. J. B. Zehnder (ed.) Biology of Anaerobic Microorganisms. Wiley-Interscience, New York, NY, USA

U. S. Environmental Protection Agency (USEPA). 1979. Methods for Chemical Analysis of Water and Wastes. United States Environmental Protection Agency, Washington, DC, USA, EPA 600/04-79/020.

Werner, T. and R. H. Kadlec. 1996. Application of residence time distributions to storm water treatment systems. Ecological Engineering 7:213–234.

WPCF (Water Pollution Control Federation). 1990. Natural Systems for Wastewater Treatment Manual of Practice FD-16. Water Pollution Control Federation, Alexandria, VA, USA.

Zhang, X. 1995. Use of a natural swamp for wastewater treatment. Ph.D. Dissertation. Louisiana State University, Baton Rouge, LA, USA.

Manuscript received 26 January 1998; revisions received 28 September 1998, 26 February 1999, and 20 July 1999; accepted 26 October 1999.