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Spatiotemporal water quality variability in a highly loaded surface flow wastewater treatment wetland

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Abstract

This study evaluates spatiotemporal relationships between water quality parameters (WQPs), nutrients, suspended solids, and biochemical oxygen demand (BOD) concentrations within an engineered wastewater treatment wetland system in the Georgia Piedmont, USA. We explored factors related to treatment efficiency within a heavily loaded 630-m^2 surface flow wetland system over a 2-yr period. Relationships between temperature, dissolved oxygen (DO), and oxidation-reduction potential (ORP) were observed; relationships were also seen between these WQPs and nutrient concentrations. Because temperature, DO, and ORP affect nitrogen (N) cycling rates, seasonal trends in N forms were evident in the system. Organic N and inorganic/organic phosphorus concentrations correlated with solids concentrations in the vegetated system without exhibiting seasonal trends. Surface water within the vegetated section generally exhibited anoxic conditions, leading to removal of nitrate-N within the system; however, limited mineralization and nitrification occurred, which greatly limited overall N removal. Plant selection and lack of maintenance likely led to high solids and BOD contributions to treatment wetland surface water, which varied substantially between and along monitored transects. Because so few studies have investigated treatment dynamics within treatment wetland cells, focusing solely on influent/effluent characterization, radical spatiotemporal variability may be the norm as opposed to the commonly accepted assumptions of relatively uniform pollutant degradation across treatment wetland cells. This spatiotemporal variability in WQPs underscores the dynamic nature of treatment wetlands and the need for routine maintenance, including sludge removal and plant harvesting.

1 | INTRODUCTION

Engineered wetlands for municipal wastewater treatment are an increasingly common practice worldwide (Kadlec & Wallace, 2009). Such wetlands treat wastewater by (a) filtration and sedimentation of solids, (b) decomposition of organic matter, (c) microbial nitrification of ammonium

Abbreviations: BOD, biochemical oxygen demand; DO, dissolved oxygen; NVS, non-volatile solids; ORP, oxidation-reduction potential; TKN, total Kjeldahl nitrogen; TN, total nitrogen; TP, total phosphorus; TS, total solids; TSS, total suspended solids; VS, volatile solids; WQP, water quality parameter.

(NH_4^+), (d) denitrification of nitrate (NO_3^-), (e) adsorption of phosphorus (P) to sediments, and (f) plant uptake of nutrients (Kadlec & Wallace, 2009). To date, most studies have only monitored influents and effluents to assess system effectiveness (Knight et al., 2000; Song et al., 2006), potentially masking heterogeneity within cells and concomitant opportunities for improved design, modeling, and maintenance. First-order plug flow models may be largely inadequate for constructed treatment wetland design and performance prediction (Gargallo et al., 2017; Kadlec, 2000; Kadlec & Wallace, 2009), although models have been augmented with residence time distribution analyses (Carleton, 2002; Carleton & Montas, 2007; Holland et al, 2004; Werner & Kadlec, 2000), pollutant speciation (Kadlec, 2003), and treatment cell design and layout (Wang & Jawitz, 2006), leading to various wetland modeling approaches and more incorporation of hydraulic efficiency and biochemical/ecological processes in these engineered systems. Toward this aim, relationships between water quality parameters (WQPs) and nutrient dynamics within wetland systems must be better understood.

Complex biological processes, including plant–soil–microbial interactions, are important for pollutant removal within wetlands (Kadlec & Wallace, 2009). Spatial variability of plants and organic material accumulation in wetlands create system complexity (Gargallo et al., 2017; Kadlec & Wallace, 2009; Strosnider et al., 2017). This system complexity, and thus the prediction of the ability to remove nutrients, solids, and organic matter, are not well understood (Gargallo et al., 2017; Kadlec & Wallace, 2009). Furthermore, the spatial heterogeneity of vegetation type and density affecting wastewater flow dynamics may compromise treatment efficiency (Lightbody et al., 2008). Spatial and seasonal variability in organic material deposition/accumulation in treatment wetland areas may lead to short-circuiting flow, a decrease in retention time, and subsequent decrease in treatment performance over time. Such unexplained spatial and temporal variability of these dynamic living systems poses challenges for treatment wetland design, operation, and maintenance for optimal performance. Few studies exist regarding treatment wetland operation or maintenance, and the most pertinent encountered in the literature pertain to stormwater wetlands (e.g., Al-Rubaei et al., 2016; Merriman & Hunt, 2014), and federal design guidelines for wastewater treatment wetlands have not been updated in over 20 yr (USEPA, 2000).

The objectives of this study were (a) to determine the extent to which spatiotemporal differences in water quality existed within a highly loaded surface-flow wastewater treatment wetland cell in the Georgia Piedmont, USA, and (b) to better understand those differences via relationships to other water quality parameters. The overarching goal is to move toward improved design guidance, operations, and maintenance practices for surface-flow wastewater treatment wetlands in similar climatic regions.

Core Ideas

- Ambient water quality parameters in wastewater treatment wetlands are highly variable.
- Nutrient removal can be limited by redox conditions that limit necessary transformations.
- Spatiotemporal heterogeneity and complexity affect system performance.
- Results underscore the importance of sludge/vegetation management for long-term performance.

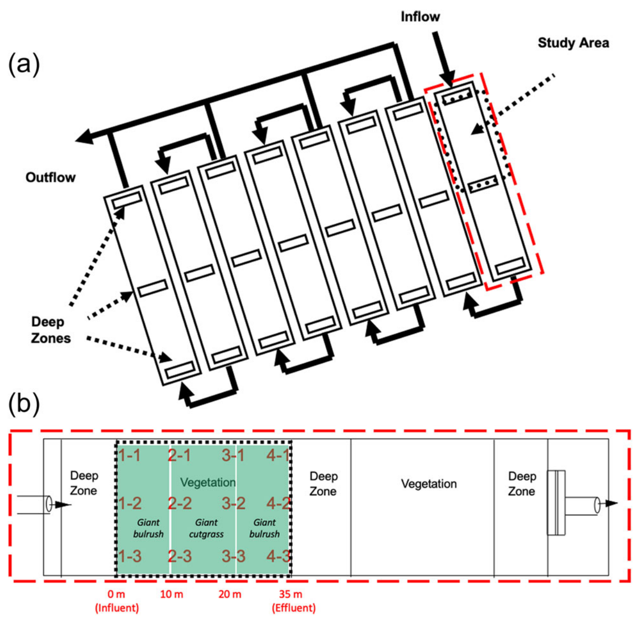


FIGURE 1 (a) Constructed wetland treatment system layout. Focal study cell is outlined in red. (b) Locations and sections of monthly grab sample collection in green (not to scale). Distances of cross-sections from beginning of vegetative zone (study area), respective to the flow gradient, are 0 m (influent) and 10, 20, and 35 m (effluent). The vegetative zone begins 5 m from the inflow pipe after an open-water deep zone. Planted sections of giant bulrush and giant cutgrass are shown

2 | MATERIALS AND METHODS

2.1 | Site description

The Tignall Water Reclamation Facility (Wilkes County, GA; 33°52′00″ N, 82°44′30″ W; Figure 1a) was constructed in 1993 as a duckweed (*Lemna* spp.) system and a partial-mix aeration pond operating in parallel prior to inflow to a secondary effluent surface-flow treatment wetland system (Hitchcock & Smith, 2008). The latter surface-flow treatment wetland system is the focus of this study. During the study period, municipal wastewater was received and treated from ∼400 homes and several light manufacturing facilities. The system received an average flow rate of 31,800 gal d^{-1} (120 m^3 d^{-1}). Supplemental Table S1 provides operating flows and concentrations associated with inflows and outflows from the entire system. Permitted National Pollutant Discharge Elimination System limits for wastewater discharge from the facility were 10 mg L^{-1} for biochemical oxygen demand (BOD), 20 mg L^{-1} for total suspended solids (TSS), and 4 mg L^{-1} for ammoniacal-nitrogen (NH_3–N + $NH_4{}^+$–N). The wetland system had been in operation for 6 yr at the time of study, consistently meeting National Pollutant Discharge Elimination System limits. The system has eight cells, each with approximately 122 m by 18 m bottom surface area; 131 m by 21 m top surface area; and 1.2 m depth, with an operating depth typically between 0.08 and 0.30 m, allowing 0.9 m maximum freeboard. The first wetland cell was chosen as the focus for this study because it experienced relatively consistent flow-through conditions and elevated pollutant loading in a well-vegetated setting. This focal cell was planted with giant bulrush [*Schoenoplectus californicus* (C.A. Mey.) Palla] and giant cutgrass [*Zizaniopsis miliacea* (Michx.) Döll & Asch.]. In summer months, rather than discharge, wastewater was recirculated through the entire system until the system reached capacity to take advantage of evapotranspiration and increased residence time.

2.2 | Sample collection and preparation

From May 1999 to April 2001, water samples were taken from 12 locations in the first wetland zone (35 m long by 18 m wide) every 4–6 wk (Figure 1b). These locations were selected based on the distribution of planted vegetation and wastewater flow. Transect 2 is downstream of *S. californicus* plantings, Transect 3 is downstream of *Z. miliacea* plantings, and Transect 4 is downstream of a second stand of *S. californicus*. Samples were collected just below the water surface but above the sediment to minimize suspending solids. Some low water depths (<5 cm) due to sludge accumulation required taking several small samples and combining them into a larger container. Therefore, vertical sampling within the vegetative zone above the sediment layer was not possible for this study. Grab samples were immediately placed on ice for transport and storage. Samples were analyzed for BOD and solids concentrations (APHA, 2012). A second set of samples was collected for nutrient analyses at the same locations and preserved in 2 ml concentrated H_2SO_4 per liter of wastewater sample.

2.3 | Surface water measurements and analyses

The following WQPs were measured at the location of each grab sample within the vegetative zone: temperature (˚C), specific conductivity (μS cm^{-1}), oxidation-reduction potential (ORP, mV), pH, turbidity (NTU), and dissolved oxygen (DO) concentration (mg L^{-1}). These parameters were measured using a calibrated hand-held YSI 6820 sonde and YSI 610-D meter. Ammonium-nitrogen ($NH_4{}^+$–N), nitrite-N ($NO_2{}^-$–N), nitrate-N ($NO_3{}^-$–N), and orthophosphate ($PO_4{}^{3-}$) concentrations, with detection limits of 0.0075, 0.014, 0.014, and 0.01, respectively, determined using USEPA methods 350.2, 353.2, and 365.1 (USEPA, 1983), were analyzed from the acid-preserved samples using a TRAACS 2000 automated wet chemistry system. For further nutrient concentration determination, water samples were digested in a $CuSO_4$ and sulfuric acid solution prior to total Kjeldahl N (TKN) and total P (TP) analyses using TRAACS, with detection limits of 0.04 and 0.014 mg L^{-1}, respectively, determined using USEPA methods 351.3 and 365.4 (USEPA, 1983). Quality assurance and quality control included correlation coefficient checks for known standards ($r^2 > .99$), dilution checks (% difference 5%), and duplicate control cup checks throughout each run (% difference <5%). If any of these controls was not met, the samples were reanalyzed. Total N (TN) was calculated as the sum of TKN and $NO_3{}^-$–N + $NO_2{}^-$–N. Analyses for total solids (TS), volatile solids (VS), and TSS were performed using standard methods (APHA, 2012). Nonvolatile solids (NVS) were calculated by subtracting VS from TS. The BOD analyses were also performed using standard methods for water and wastewater analyses (APHA, 2012).

2.4 | Statistical analyses

All data analyses were conducted using JMP statistical software (SAS Institute). Due to non-normality and heteroscedasticity of data, the nonparametric Wilcoxon and Kruskal–Wallis tests were used to determine if sampling location (Transects 1, influent; Transects 2, 3, and 4, effluent) and date of sample collection had significant effects ($\alpha = .05$) on WQPs and on nutrient, BOD, and solids concentrations. The Steel–Dwass test was used to separate treatment medians when significant differences were found ($\alpha = .05$). Spearman's correlation analyses were also conducted between WQPs and nutrient, BOD, and solids concentrations.

2.5 | Treatment performance

Percent nutrient, solids, and BOD removals (Equation 1) and area removal rate (ARR; Equation 2), defined as the mass of constituent removed per square meter of the vegetative zone per day (g m^{-2} d^{-1}), were calculated to characterize removal of nutrients, solids, and BOD within the vegetative zone of the wetland cell (Hsueh et al., 2014; Kadlec & Wallace, 2009):

$$\% \text{ Removal} = \left(C_\mathrm{i} - C_\mathrm{e}\right)/C_\mathrm{i}{}^{*}\ 100 \tag{1}$$

$$\text{ARR} = Q/A * \left(C_\mathrm{i} - C_\mathrm{e}\right) \tag{2}$$

where C_i is the average influent concentration (mg L^{-1}) along Transect 1, C_e is the average effluent concentration along Transect 4, Q is the average flow rate (m^3 d^{-1}), and A is the area (m^2) of the vegetative zone. The results only represent effectiveness across the monitored vegetative zone (Figure 1). Constituent removal percentages were calculated using mean inflow and outflow concentrations for sample locations at transects along the beginning of the vegetative zone (1-1, 1-2, 1-3) and end zone (4-1, 4-2, 4-3).

3 | RESULTS AND DISCUSSION

3.1 | Surface water quality parameters

Throughout the sampling period (May 1999 to April 2001), the temperature ranged from 5.9 to 28.4 ˚C, ORP ranged from −320 to 251 mV, and DO ranged from 0.4 to 9.4 mg L^{-1} (Figure 2). Monitoring various water quality parameters over space and time provided information about the environmental conditions under which the treatment wetland system operated as well as relationships between these WQPs. The correlation between temperature and DO (Supplemental Table S2) indicated a strong inverse relationship ($\rho = -.56$), as did the correlation between temperature and ORP ($\rho = -.74$). As expected, the correlation between DO and ORP was high ($\rho = .70$). Dissolved oxygen, ORP, and pH were significantly lower during the warm season (April–September) than during the cool season (October–March), whereas temperature and specific conductivity were significantly higher during the warm season ($p < .05$). In addition to seasonal trends in water quality parameters, there were significant trends ($p < .05$) observed between sampling locations each month (Supplemental Table S3). When considering influent and effluent values only, temperature significantly decreased across the vegetative zone in 8 of the 17 mo sampled and significantly increased during one sampling event. Effluent pH significantly decreased as compared with influent values for six sampling events, which could be attributed to the release of protons during nitrification and production of organic acids via macrobiotic and microbiotic processes, although these processes were not measured explicitly in this study.

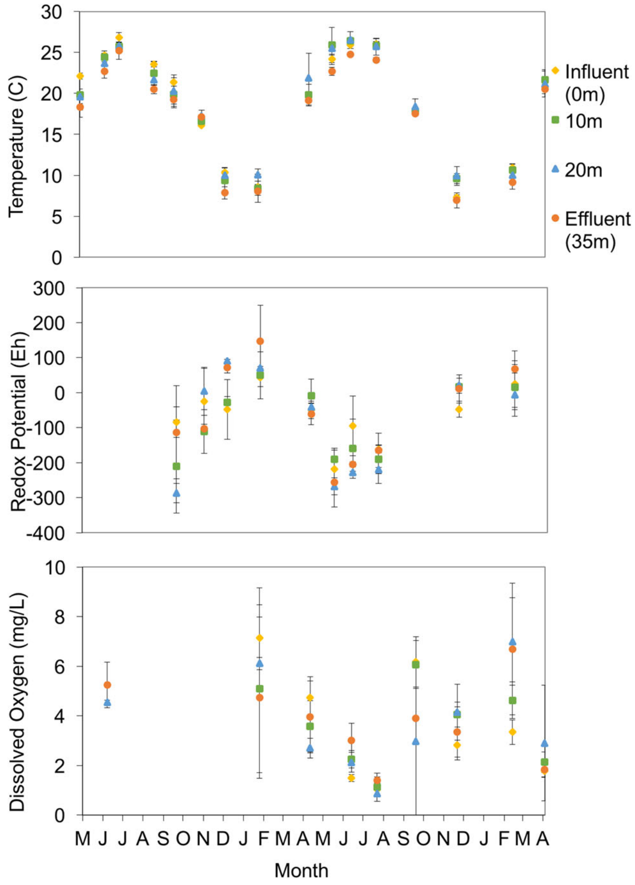


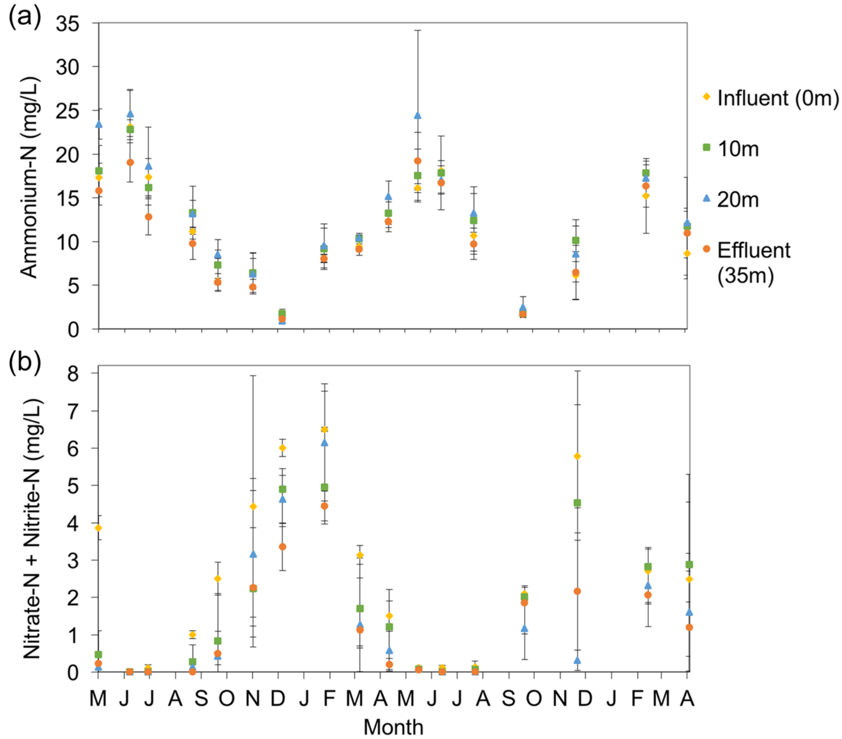
FIGURE 2 Monthly mean values for temperature, redox potential, and dissolved oxygen collected at the time of sampling, for each sampling transect, showing seasonal patterns. Standard deviations are given by error bars

3.2 | Nutrient and solids concentrations

For inorganic N constituents, $NH_4{}^{+}$–N concentrations decreased, whereas $NO_3{}^{-}$–N + $NO_2{}^{-}$–N concentrations were higher during the cooler winter months (Figure 3a and b).

The relationships between $NH_4{}^{+}$–N and $NO_3{}^{-}$–N + $NO_2{}^{-}$–N indicate a seasonal variability in nitrification and denitrification rates, which are influenced by DO concentrations and ORP. Lower concentrations of $NH_4{}^{+}$–N during colder temperatures, likely due to increased aerobic microbial activity, have been documented in comparable wetland treatment systems (Dzakpasu et al., 2015; Hsueh et al., 2014). The observed seasonal trend could be due to the ability of wastewater to retain higher DO concentrations in colder months, thus encouraging nitrification. Also, cooler temperatures and a shorter daylength in winter months may decrease duckweed (prevalent throughout the system) productivity and microbial activity, resulting in less biomass decomposition, less respiration, and less DO consumption, thereby providing more DO for nitrification to occur (Benjawan & Koottatep, 2007).

FIGURE 3 Monthly mean concentrations for (a) ammonium-N and (b) nitrate/nitrate-N for each sampling transect. Standard deviations are given by error bars



When comparing Transects 1–4, significantly higher concentrations of TKN and TN were observed in the center of the vegetative zone as compared with influent or effluent values, whereas NO_3^-–N concentrations decreased across the vegetative zone, with Transect 3 and effluent concentrations significantly lower than influent concentrations (Figure 4). However, NO_3^-–N and TN concentrations were extremely spatially variable (often orders of magnitude different even along the same transect during a single sampling event) as well as temporally variable (even from one sampling event to the next) (Figure 5).

The types of microorganisms present in the system are important to the treatment efficiency of the system as well as the nutrient fluxes that occur. Nitrifiers typically occur in higher ORP (Eh > +100 mV), whereas denitrifiers are typically active in lower ORP (−300 mv < Eh < +100 mV) (Howard-Williams, 1985; Reddy & Graetz, 1988; Vymazal, 1995). Of the 115 ORP data points collected during the sampling period, only four ORP readings were over 100 mV (collected in February 2000 and 2001), indicating that throughout the study period denitrification, rather than nitrification, was more favorable in the water column. Most of the N throughout the study period, both spatially and temporally, was in the form of organic N and NH_4^+–N, with very little NO_3^-–N + NO_2^-–N observed (Figure 6). Total Kjeldahl N concentrations, as a combined measure of NH_4^+–N and organic N, exhibited trends similar to NH_4^+–N over the 2-yr sampling period: median TKN concentrations were typically lower in colder months than in warmer months. An effective treatment wetland should have a favorable balance of nitrification and denitrification in order to convert NH_4^+–N to NO_3^-–N + NO_2^-–N to N_2, thus removing N from the wastewater (Kadlec & Wallace, 2009). This N removal from aquatic systems requires the occurrence of both aerobic and anoxic zones. Passive aeration could serve to create aerobic zones in areas naturally prone to anoxic conditions. Effluent recycling and dynamic effluent control (e.g., stockpiling nitrified waters for subsequent denitrification) could also be applied to enhance overall N removal.

Comparing the nutrients to solids by Spearman correlations overall (Supplemental Table S2), TKN, TN, organic N, PO_4^{3-}, and TP had the highest correlations ($p < .0001$) with TS concentrations (.82, .81, .81, .77, and .74, respectively), considered to be due to the high organic (volatile) fraction of the solids (Supplemental Figure S2a and b). The organic fractions associated with suspended solids contributed significantly to existing nutrient concentrations ($p < .0001$), with ρ values >.70 for VS compared with TKN, TN, organic N, PO_4^{3-}, and TP. Phosphorus tends to bind to sediment particles (Bhomia & Reddy, 2018; DiLuca et al., 2017), which may explain the correlation between PO_4^{3-} and solids concentrations (averaging ρ = .74 for comparison with all solids measurements). Paudel et al. (2010) observed P transport and storage within a treatment wetland system to be primarily controlled by soil TP content and preferential flow paths. The observed spatiotemporal heterogeneity of TP and PO_4^{3-} in the water column, and generally elevated values in the center of the cell (Figure 5), point to potential P remobilization from accumulated sludge enhanced by anoxia (e.g., Palmer-Felgate et al., 2011; Paudel et al., 2010).

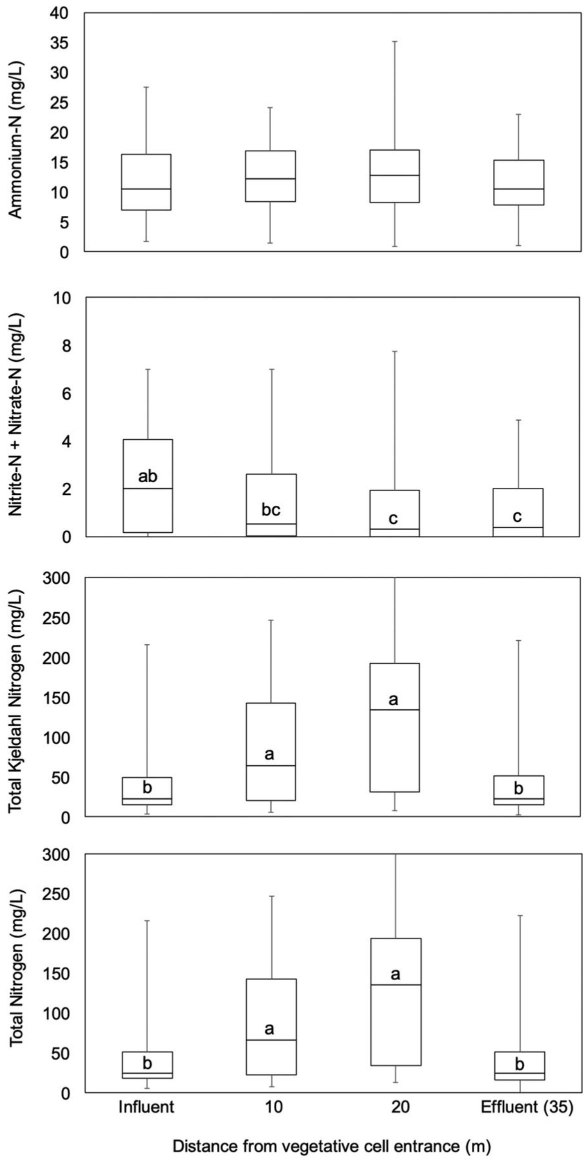


FIGURE 4 Box plots of ammonium-N, nitrate/nitrate-N, total Kjeldahl N, and total N concentration data in terms of spatial distribution through the vegetative zone for all months (Hitchcock & Smith, 2008). Different letters indicate statistically significant differences ($p < .05$)

Statistical analyses indicated a significant positive correlation between NH_4^+–N and solids concentrations, with correlations averaging .42 for all solids measurement comparisons. Results also indicated a significant negative correlation between NO_3^-–N + NO_2^-–N concentrations and solids concentrations, with correlation coefficients averaging −.45 for comparison with all solids measurements. The high organic content of solids in the center of the vegetative zone (Figure 7b), caused by the buildup of decaying plant biomass, likely depleted oxygen and other terminal electron acceptors (i.e., NO_3^-–N + NO_2^-–N) while resulting in higher NH_4^+–N concentrations.

The relationship between solids concentrations and other WQPs was determined to be significant, with the highest inverse relationships existing between ORP and TS concentrations (ρ = −.48). This relationship indicates that the organic nature of these solids contributes to anoxic conditions, encouraging denitrification to occur at organic-rich locations. In vegetated areas where flow slowed, dead plant biomass from emergent vegetation and duckweed as well as microbial biomass and other solids resulted in visually observed sludge accumulation, which subsequently led to higher VS and TP concentrations in the vegetative zone (Figures 7b and 8b). Elevated TP may have been due to anoxia-driven P remobilization (Palmer-Felgate et al., 2011). DiLuca et al. (2017) observed a similar relationship between aggregations of macrophyte litter and elevated P concentrations in a treatment wetland. Also, plant productivity, decomposition, and organic export typically vary between different types of aquatic vegetation, especially in macrophytic freshwater wetland systems (Mitsch & Gosselink, 2000).

Giant cutgrass [*Zizaniopsis miliacea* (Michx.) Döll & Asch.] tends to grow and senesce very quickly, with nodes more likely to become detached than rooted in substrate (Fox & Haller, 2000), substantially contributing to dead organic material that reaches the water surface. In contrast, giant bulrush [*Schoenoplectus californicus* (C.A. Mey.) Palla] tends to grow more slowly and remain standing once dead, contributing less dead organic material to the water column (Mallison & Thompson, 2010). The cutgrass species was positioned in the center section of the vegetative zone on which this study focuses. An elevated dead organic material contribution of *Z. miliacea* to solids concentrations, especially VS, was evident in this area of the wetland. Magri et al. (2016) found higher VS and TS concentrations associated with a constructed wetland for sludge dewatering planted with *Z. miliacea* as compared with identically sized constructed wetlands planted with other aquatic plant species. Birch and Cooley (1982) determined that *Z. miliacea* retains much of its previous-year production of aerial biomass and, upon subjection to tidal water movements, can shed its previous-year standing dead biomass. Hu et al. (2010) have also documented the role that similar plant species, such as Manchurian wildrice [*Zizania latifolia* (Griseb.) Turcz. ex Stapf], have in the spatiotemporal distribution of ORP in their study of plant diversity and thus effecting wetland treatment performance; macrophytes that die back very quickly and easily shed organic material (e.g., cutgrass) may be best suited (a) at the downstream edge of planted sections and (b) downstream of macrophytes that tend to grow more slowly and remain standing once dead (e.g., bulrush), potentially leading to less accumulation of sludge (dead plant material) and essentially becoming "trapped" within the planted section. Sediment accumulation as sludge can adversely affect flow through the wetland system, short-circuiting

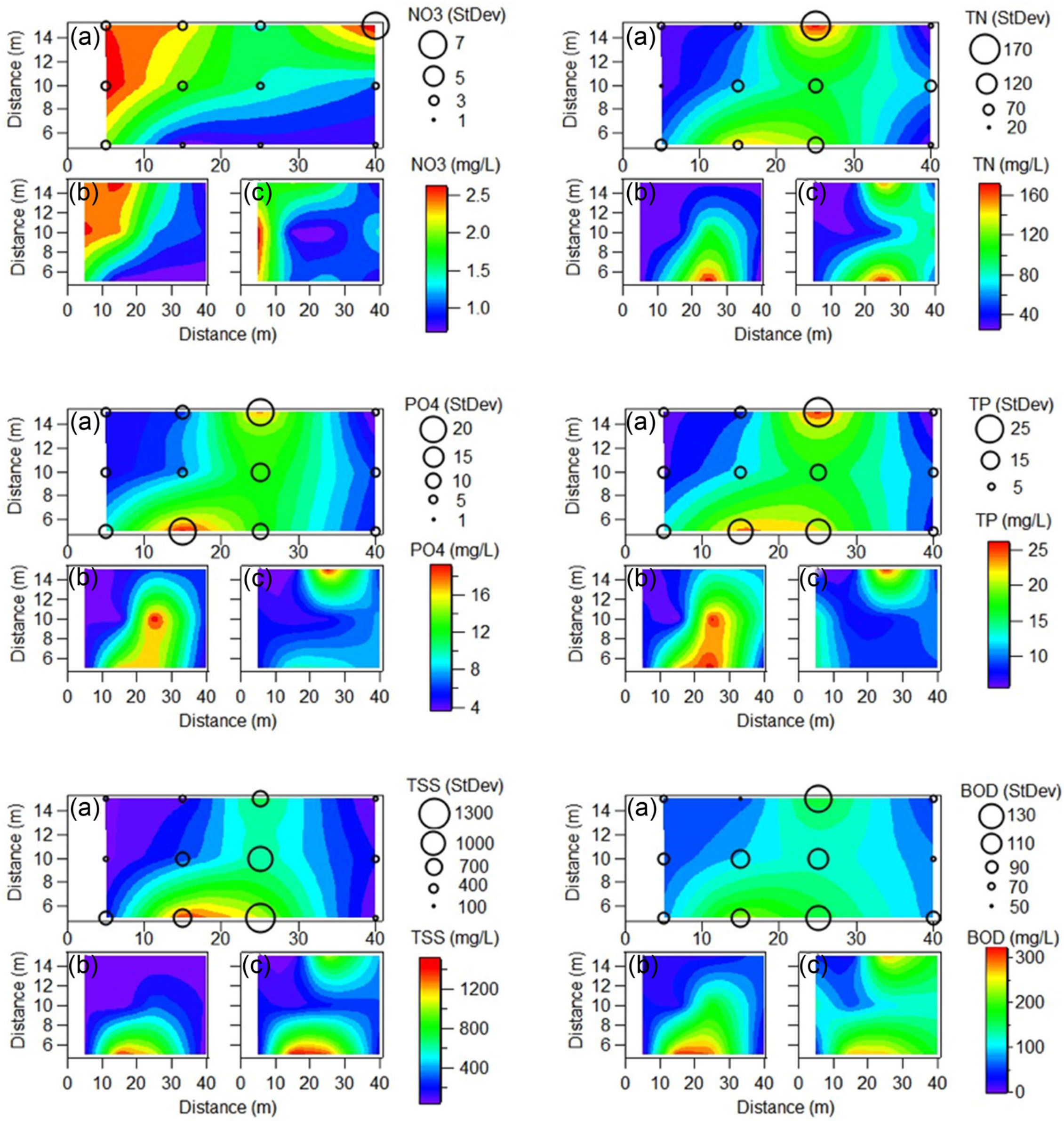


FIGURE 5 Heat maps for nitrate-N, total N (TN), orthophosphate, total P (TP), total suspended solids (TSS), and biochemical oxygen demand (BOD) concentrations (circles indicate sampling location), with mean and standard deviation concentrations during the entire sampling period (A) and snapshots from example representative consecutive sampling events (21 Apr. 2000 and 26 May 2000 for B and C, respectively) within the first vegetated section of the study area. General flow direction is from left to right across the width (y-axis) of the vegetated section (Figure 1). Elevated standard deviations and differences of over an order of magnitude between consecutive sampling dates are common throughout these data, demonstrating broad spatiotemporal variability across all the water quality parameters tracked

treatment processes and reducing hydraulic retention time less than that as designed and thus decreasing performance. Paudel et al. (2010) found that flow resistance was a primary model input that affected P removal and treatment wetland performance. Additionally, the high abundance of duckweed (*Lemna* spp.) throughout the system increased organic material and sediment contributions over time.

3.3 | Treatment performance

Treatment performance in the vegetative zone was evaluated by month in terms of percent constituent removal and areal removal rate for the entire zone (Supplemental Table S4). Ammoniacal-N concentrations were significantly reduced in only 1 of 17 mo sampled (December 1999) and significantly increased for the May 2000 sampling event.

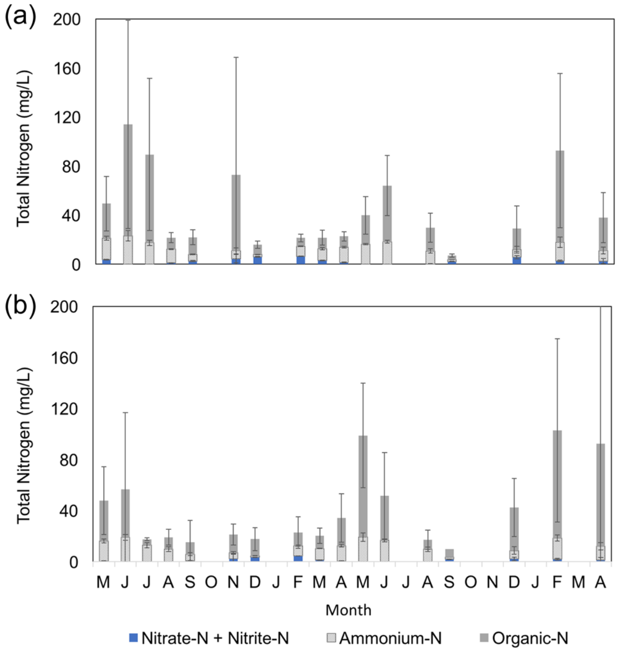


FIGURE 6 (a) Influent and (b) effluent total N (including NO_2/NO_3 and total Kjeldahl N) mean concentrations in vegetative zone over a 2-yr period. Standard deviations are given by error bars

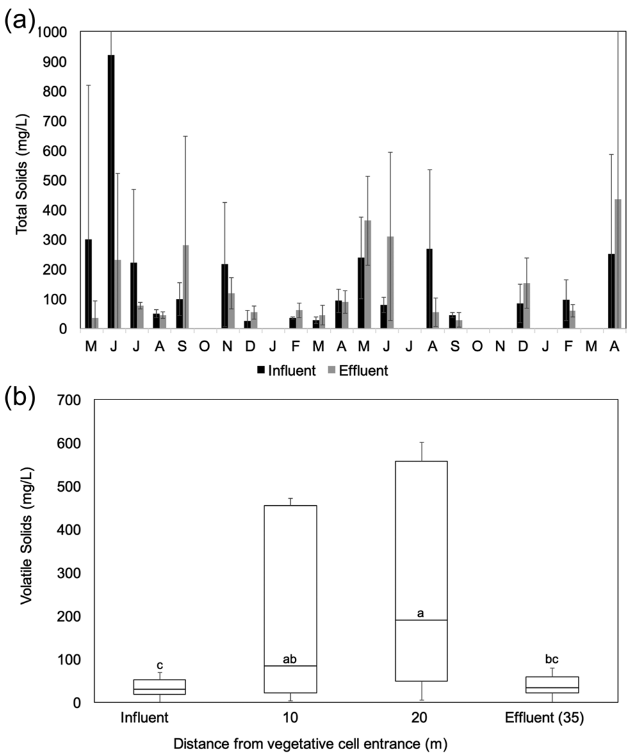


FIGURE 7 (a) Monthly mean concentrations of influent and effluent total solids in the vegetative zone over a 2-yr period and (b) box plots of volatile solids concentration data in terms of spatial distribution through the vegetative zone for all months. Standard deviations are given by error bars (Hitchcock & Smith, 2008). Different letters indicate statistically significant differences ($p < .05$)

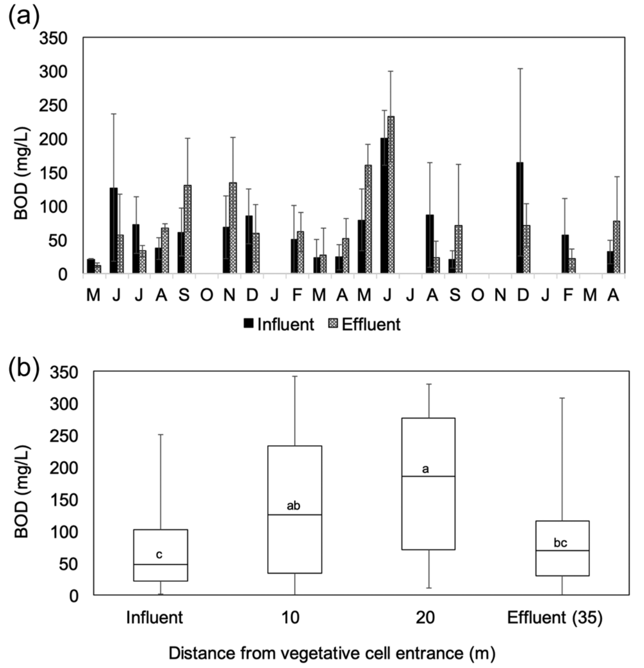


FIGURE 8 (a) Monthly mean concentrations of influent and effluent biochemical oxygen demand (BOD) in vegetative zone over a 2-yr period and (b) box plots of BOD concentration data in terms of spatial distribution through the vegetative zone for all months. Standard deviations are given by error bars

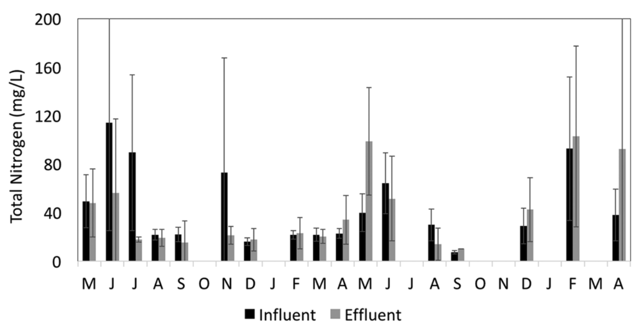


FIGURE 9 Mean total N concentrations at the beginning and end of the vegetative zone over the length of the study. Standard deviations are given by error bars

Nitrate/nitrite-N concentrations were significantly reduced for 10 of 17 mo sampled, ranging from 32 to 100% removal and from 0.35 to 0.01 to 0.69 g m^{-2} removed per day (Figure 9; Supplemental Table S4). Orthophosphate and organic P concentrations were not significantly reduced in any months but did increase for the April and May 2000 sampling events (Figure 10). This phenomenon is not unexpected because this was within the first zone of a highly loaded system continuously operated for 6 yr, accumulating solids and associated P (mostly as available P). As shown in Figure 5, nutrient, solids, and BOD concentrations were extremely spatially variable along the same transect (during a single sampling event) as well as

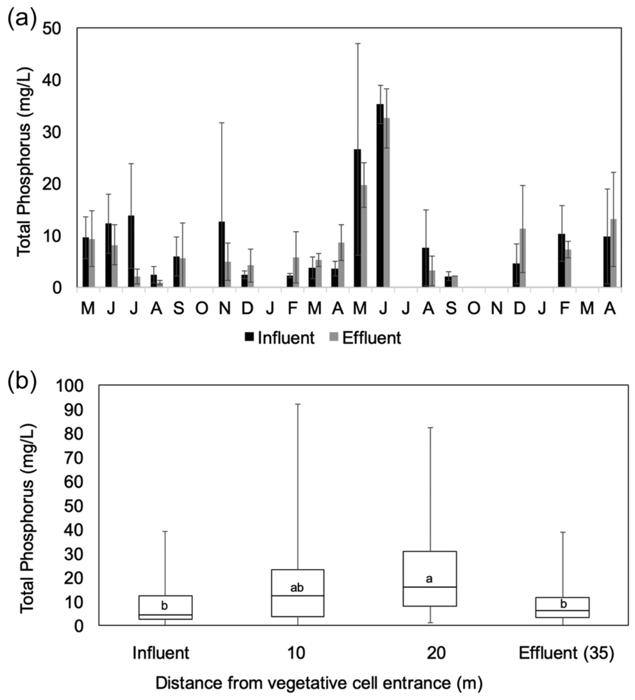


FIGURE 10 (a) Monthly mean concentrations of influent and effluent total P (TP) in the vegetative zone over a 2-yr period and (b) box plots of TP concentration data in terms of spatial distribution through the vegetative zone for all months. Standard deviations are given by error bars. Different letters indicate statistically significant differences ($p < .05$)

temporally variable (even from one sampling event to the next). Such variability represents non-ideal hydraulic plug flow conditions and increased potential for short-circuiting, which likely affected treatment dynamics and highlights the importance of routine maintenance efforts, including sludge removal, grading, and plant harvesting (Gorgoglione & Torretta, 2018).

Treatment wetland P removal often decreases over time, especially for heavily loaded systems (Bhomia & Reddy, 2018; DiLuca et al., 2017; Fink & Mitsch, 2004; Kadlec & Wallace, 2009; Wu et al., 2017). Sorption sites in the substrate were likely saturated after 6 yr of heavy loading, rendering plant uptake as the remaining major available removal factor, as has been noted by Wu et al. (2017) in another treatment wetland study. Total Kjeldahl N, organic N, and TN concentrations were significantly reduced in July 1999 but significantly increased for May 2000 at the time of sampling. Anoxic conditions throughout the study period likely favored denitrification over nitrification, so TN removal was very likely limited by nitrification. These conditions could explain the lack of significant NH_4^+–N removal and extremely high reduction of NO_3^-–N + NO_2^-–N, which accounted for very little TN in the study area. Incorporation of passive aeration or unsaturated vertical-flow elements within constructed wetlands may allow for greater rates of nitrification to occur (due to increased dissolved oxygen concentrations), and incorporation of ammonium adsorbent materials could serve to further remove ammonium from the water column (Kizito et al., 2017; Xiong et al., 2011; Yadav et al., 2018).

Total solids, NVS, and VS concentrations were not significantly reduced throughout the sampling period (Figure 7a). Total solids, VS, and TSS concentrations significantly increased during the February 2000 sampling event (Supplemental Table S4). Total suspended solids concentrations were significantly reduced for only one sampling event (May 1999). Biochemical oxygen demand concentrations were significantly reduced in only 1 of 17 mo sampled (May 1999) and significantly increased during the August 1999 sampling event. When comparing differences in concentrations between all four transects of the vegetative zone for all sampling events, some constituents significantly increased in the middle of the zone before decreasing at the end of the zone back down to near influent levels (Figures 7b, 8b, and 9b). In general, TSS samples collected from the middle two transects of the vegetative zone showed increases, sometimes over an order of magnitude, higher than influent and effluent concentrations. Although BOD generally did not significantly increase or decrease when comparing influent and effluent concentrations (Figure 10a), BOD did significantly increase in the middle portion of the vegetative zone (Figure 10b), where greater dead biomass (sludge) was visibly present.

4 | CONCLUSIONS

Results from this study demonstrated how system design, plant selection, and maintenance can significantly affect buildup of organic materials, redox conditions, and removal capacity of nutrients and BOD within the vegetative zone of a surface flow wastewater treatment wetland. Results also revealed the spatially heterogeneous nature of nutrient, BOD, and solid concentrations within the vegetative zone. Transects across the direction of flow, representing theoretically identical hydraulic retention times, and therefore theoretically identical water quality transformations, displayed radically different values. Because this is one of few studies that have focused within a treatment wetland cell, rather than influent/effluent alone, it is likely that highly variable spatiotemporal signatures are common within these systems. Better understanding of spatial and temporal dynamics allow for implementation of improved design strategies (e.g., improved plant selection/positioning, flow distribution/redistribution), construction quality (e.g., grading within tight tolerances), as well as operational and maintenance strategies (e.g., recirculation, regrading of surfaces, harvesting, replanting, sludge removal). Further, understanding the dynamics within constructed wetland systems can aid in the development of operations and maintenance strategies that could improve treatment efficiency and increase system longevity.

This spatial and temporal heterogeneity reveal a system far from ideal plug-flow and, therefore, treatment conditions. Operation and maintenance efforts should focus on establishing and retaining plug-flow conditions. De-sludging, regrading surfaces, harvesting, and replanting are interventions that ought to be considered at higher frequencies than many treatment wetland operators have traditionally applied them. Special care should be taken with flow collection and redistribution system (the deep zones in the case of the cell in this study) design and maintenance.

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AUTHOR CONTRIBUTIONS

Daniel Hitchcock: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Project administration; Validation; Visualization; Writing – original draft. Natasha Bell: Data curation; Formal analysis; Investigation; Validation; Visualization; Writing – review & editing. William Strosnider: Data curation; Formal analysis; Validation; Visualization; Writing – review & editing. Matt C. Smith: Funding acquisition; Resources; Supervision; Writing – review & editing.

CONFLICT OF INTEREST

There are no conflicts of interest associated with this study and/or published work.

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