# Remediation of Nitrogen and Phosphorus from Nursery Runoff during the Spring via Free Water Surface Constructed Wetlands[1]

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## Abstract

Agricultural operations face increasing pressure to remediate runoff to reduce deterioration of surface water quality. Some nursery operations use free water surface constructed wetland systems (CWSs) to remediate nutrient-rich runoff. Our objectives were twofold, first to examine the impact of two hydraulic retention times (HRT, 3.5 and 5.5 day) on CWS performance, and second to determine if increased nutrient loading from internal CWS and nursery sources during the spring contributed to nutrient export in excess of regulatory limits. We quantified nutrient loading and removal efficiency in a free water surface CWS from late winter through late spring over three years and monitored various water quality parameters. Total nitrogen in runoff was reduced from 20.6 ± 2.8 mg·liter$^{-1}$ (ppm) to 4.1 ± 1.3 mg·liter$^{-1}$ (ppm) nitrogen after CWS treatment. Phosphorus dynamics in the CWS were more variable and unlike nitrogen dynamics were not consistently influenced by water temperature and hydraulic loading rate. Phosphorus concentrations were reduced from 1.7 ± 0.8 mg·liter$^{-1}$ (ppm) $PO_4$-P in influent to 1.2 ± 0.6 mg·liter$^{-1}$ (ppm) $PO_4$-P in CWS effluent, but substantial variability existed among years in both phosphorus loading and removal rates. The CWS was able to efficiently remediate nitrogen even under high spring loading rates.

**Index words:** internal loading; nonpoint source pollution; agrichemicals; nutrient monitoring; nutrient removal efficiency.

### Significance to the Nursery Industry

Nurseries use large quantities of water and fertilizers during the growing season to produce marketable crops. With these water and fertilizer applications, there is potential for nutrient enrichment of nearby surface and ground waters. Many nurseries are implementing water and nutrient management plans to save money and more effectively manage inputs and runoff. Constructed wetland systems (CWSs) are one effective method for treating nutrient rich runoff. However, in the spring, there is potential for increased nutrient loading from internal CWS sources as well as via fertilizer moving in runoff from nursery production areas. We intensively monitored nutrient assimilation by a two-stage free water surface flow CWS in South Georgia during the late winter through spring period over three years. The CWS received runoff from 48.6 ha (120 A) of production. We found high assimilation rates for nitrogen throughout the CWS. Phosphorus removal was variable and additional research is needed to develop economically viable secondary treatment systems for consistent phosphorus removal. Efficient phosphorus assimilation correlated with the spring flush of vegetative plant growth in the CWS, while nitrogen assimilation correlated with water temperature, hydraulic loading rate, and retention time. Free water surface CWSs are effective for remediating nitrogen from nursery runoff.

### Introduction

Eutrophication is a natural process that occurs with nutrient additions (mainly nitrogen and phosphorus) into surface waters. Residential, urban, and agricultural land uses contribute to nonpoint source nutrient-rich runoff. Local, state, and federal environmental agencies are under pressure to limit nutrient discharges from identifiable nonpoint source contributors to further protect and improve water quality (28). Nurseries use greater quantities of agrichemicals per acre in a given year than traditional row-crop or orchard operations because of the plant density per acre, along with artificial substrate used in containers, large number of species grown, and rapid plant turnover. Nursery operations that do not capture and recycle runoff can be significant contributors to agricultural nonpoint source contamination. Currently, California, Florida, Maryland, Oregon, and Texas have adopted regulations mandating runoff capture (1, 2, 22). Further regulation is likely in other states as efforts continue to protect and maintain the quality of surface and ground water resources. In fact in January 2009, the US Environmental Protection Agency in concert with the Florida Department of Environmental Protection proposed an expedited schedule for establishing numeric nutrient criteria limiting N and P pollution in Florida lakes, rivers, streams, springs, and canals (32). These numeric standards may be finalized for lakes and streams (fresh water sources) by October 2010 and would be based on ecological health and balance rather than toxicity thresholds and would be enforced on a watershed basis, with little differentiation between point and nonpoint source contributors.

Strategic installation of free water surface and horizontal subsurface-flow constructed wetland systems (CWSs) to assimilate municipal and agricultural nutrient loads is an increasingly popular remediation technique in Australia, Israel, Europe, South Africa, and the United States (10, 11, 24,

[1]Received for publication March 4, 2010; in revised form June 24, 2010. Financial support of this project through the Floriculture and Nursery Research Initiative for Environmental and Resource Management Practices and Strategies, USDA Agriculture Research Service, Ft. Pierce, FL, is gratefully acknowledged. Technical Contribution No. 5813 of the Clemson University Experiment Station.

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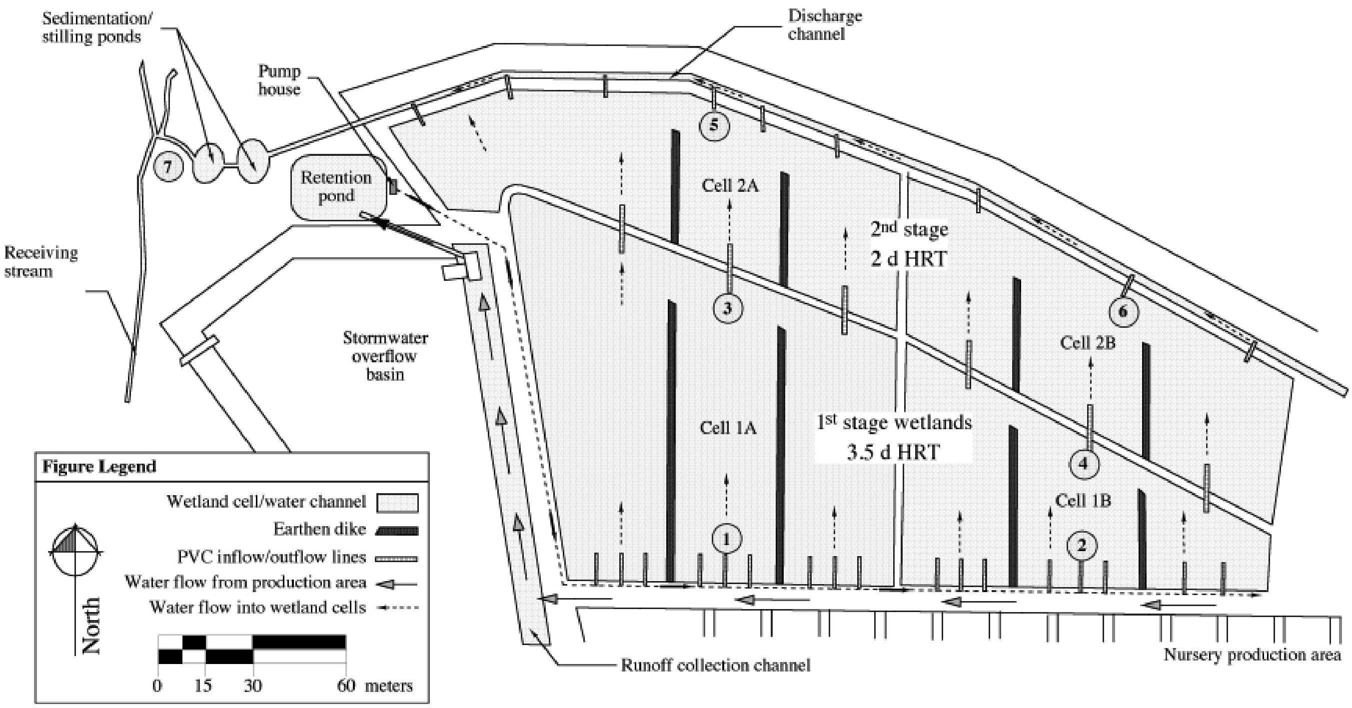


**Fig. 1. Diagram of a free water surface constructed wetland system (3.77 ha) captures runoff from 48.6 ha nursery production area. Sampling location and description are numbered as follows: influent (sites 1 and 2), 3.5 d HRT (stage 1 outflow, sites 3 and 4), 5.5 d HRT (stage 2 outflow, sites 5 and 6), and effluent (site 7).**

35). Ease of maintenance and the low cost of establishment are two main factors contributing to their increased usage in place of traditional wastewater treatment systems. Many studies have shown that surface (2, 13, 21, 34) and subsurface (2, 10, 11, 15, 16, 29, 34) CWSs effectively remove some N, but the quantity and form of N removed depends on a variety of factors. Seasonal and environmental effects such as pH, temperature, dissolved organic carbon, dissolved oxygen, day length, and plant growth cycles impact nutrient processing in wetlands (3, 12, 26, 27).

Internal nutrient loading is seasonal and driven by nutrient dissolution from decaying plant material and release of phosphorus from sediments, as dissolved oxygen levels and pH values fluctuate, and may result in increased nutrient export from a treatment CWS during the spring (5–7, 23, 26). The above factors, compounded with the lack of cohesive design criteria based on quantitative research and limited characterization of internal wetland loading processes, result in CWSs that do not function optimally.

Container nurseries typically increase fertilization for early spring plant growth when the danger of freeze/frost damage is at a minimum. Therefore, there is potential for increased internal nutrient loading and fertilization to occur concurrently in CWSs used by nurseries to remediate runoff. This could potentially lead to nitrogen (N) export from a CWS that exceeds the federal drinking water quality limit of 10 mg·liter$^{-1}$ $NO_3$-N. Although phosphorus (P) is not yet regulated, the US EPA has suggested limits for freshwater total phosphate for streams and lakes or reservoirs of 0.05 and 0.025 mg·liter$^{-1}$ (ppm) respectively, and total phosphorus limits of 0.10 mg·liter$^{-1}$ (30). The objectives of this study were to characterize nutrient removal efficiency in a two-stage, free water surface CWS from late winter through late spring and to quantify the influence of hydraulic retention time on nutrient removal efficiency.

## Materials and Methods

*Study site.* The study site was a 3.8 ha (9.4 A), two-stage, vegetated, free water surface CWS that received excess runoff from a 48.6 ha (120 A) horticultural production area in Cairo, GA. This CWS was installed in 1997 to proactively reduce agrichemical runoff from nursery property. Runoff from production beds drains into a 0.46 km (0.29 mile) flow-control channel that drains into a 0.4 ha (1 A) retention pond. The runoff is then pumped from the retention pond into cells 1a and 1b in the first-stage of the CWS (Fig. 1). The first-stage has an average depth of 76.2 cm (2.5 ft) and a theoretical hydraulic retention time (HRT) of 3.5 days. Water flows into the first stage through 6.4 cm (2.5 in) polyvinyl chloride (PVC) pipes, and inflow rates range from 250 to 350 liters·min$^{-1}$ dependent upon daily pump flow rate. Two earthen dikes extend to within approximately 9.1 to 12.2 m (10.0–13.3 yd) from the end of cells 1a and 1b, dividing each cell into three sections in an open channel, which is 121.9 cm (48.0 in) deep, and then is gravity fed through 15.2 cm (6 in) PVC drain pipes into the second stage of the CWS.

The second stage of the CWS is also divided into three sections and joined at the discharge end for 6.1 to 9.1 m (6.7 to 10.0 yd) to provide a mixing zone before gravity-fed release into the discharge channel. The second-stage is on average 20.3 cm (8 in) deep, and has a theoretical 2 day (d) HRT. Both stages combined result in a theoretical 5.5 d HRT. The first stage will be referred to hereafter as the 3.5 d HRT treatment and the first and second stage in series combination will be referred to as the 5.5 d HRT treatment. Water then flows through the discharge channel and through two stilling ponds, where the flow rate slows to permit settling of remaining suspended sediments. Suspended sediments are known to facilitate nutrient and pesticide transport off-site if suspended in the water column (17, 18). Water exits the stilling ponds and ultimately leaves the property by merging with a nearby stream. Nutrient removal efficiency will be discussed with regard to 3.5 d HRT, 5.5 d HRT, and effluent.

Daily water flow into the wetland varied over the three years of the study. Daily pump volumes from the retention pond, receiving runoff from production areas, into the first stage of the constructed wetland were recorded, and flow rates were measured and calibrated early in the spring each year of this study. In 2003, average daily pump volume was 2.4 million liters·day$^{-1}$ (0.6 million gal·day$^{-1}$), and the average flow rate per inlet pipe in the CWS was 132.5 liters·min$^{-1}$ (35.0 gal·min$^{-1}$). Daily sampling did not occur in 2004 due to lack of nursery personnel for sampling, thus average pump volume and flow rate were not reported. Sampling resumed in 2005, daily pump volume was variable due to pump problems and ranged from 1.7 to 4.0 million liters·day$^{-1}$ (0.5 to 1.1 million gal·day$^{-1}$), when the pump was in operation, and average flow rate was 96.7 liters·min$^{-1}$ (25.5 gal·min$^{-1}$). In 2006, daily pump volume was 2.8 million liters·day$^{-1}$ (0.7 million gal·day$^{-1}$), and the average flow rate was 99.5 liters·min$^{-1}$ (26.3 gal·min$^{-1}$).

Vegetation in the first-stage of the wetland is dominated by giant bulrush (*Schoenoplectus californicus* (C. A. Mey.) Palla), alligator weed (*Alternanthera philoxeroides* (Mart.) Griseb.), water meal (*Wolffia brasiliensis* Weddell), common duckweed (*Lemna valdiviana* Phil.), maidencane grass (*Panicum hemitomon* Schult.), pickerelweed (*Pontederia cordata* L.), floating pennywort (*Hydrocotyle ranunculoides* L.f.), and common cattail (*Typha latifolia* L.). Second-stage vegetation is dominated by common duckweed, floating pennywort, water pennywort (*Hydrocotyle umbellate* L.), maidencane grass, common cattail, pickerelweed, and broadleaf arrowhead (*Sagittaria latifolia* Willd.).

*Water quality monitoring.* Water samples and temperatures were taken three times daily (8:30 AM, 12:00 PM, and 4:30 PM) from March 10 to May 16, 2003, at sampling sites 1, 3, 5, and 7 in the CWS (Fig. 1). Sampling did not occur in 2004 due to change in personnel at the nursery. Mid-day sampling temperatures and nutrient content changed very little when compared with morning and evening samples in 2003; thus, sampling frequency was reduced to twice daily to reduce nursery time commitment. In 2005, water samples and temperatures were taken twice daily (~8:30 AM and 4:30 PM) from March 2 to May 14. Sampling sites were expanded to include three additional sites 2, 4, and 6 (Fig. 1). Based on two years of multiple samples per day, we determined a single daily sample was sufficient to characterize daily nutrient flux. Thus, in 2006 samples were taken only once daily, from March 8 to May 17, and sampling sites were the same as those sampled in 2005. Water temperatures were monitored at sites 1 and 5 in 2003 and 2 and 6 during 2005 and 2006. Thermometer depth was approximately one-third the depth of the cell; at sites 1 and 2 average depth was 30.5 cm (12 in), while at sites 5 and 6 average depth was 10.2 cm (4 in).

Water samples were grab samples representing a particular time and location. Samples for anion analyses were filtered through 0.45 μm polytetrafluoroethylene (PTFE) membrane filters into 1.5 mL ion chromatography (IC) vials and stored at 4C (39.2F) until analysis with a Dionex AS10 IC with AS50 auto-sampler (Dionex Corp., Sunnyvale, CA). In 2003, 2005, and 2006 samples were analyzed for nitrite ($NO_2$), nitrate ($NO_3$), phosphate ($PO_4$), and sulfate ($SO_4$). In 2005 and 2006 samples were also analyzed for ammonia using an Orion Ammonia Electrode 95-12 (Thermo Electron Corp., Beverly, MA), as well as non-purgeable organic carbon (NPOC) and total N (TN) using a Shimadzu TOC-$V_{CPH}$ total organic carbon analyzer with TNM-1 TN measuring unit (Shimadzu Scientific Instruments, Kyoto, Japan). Non-purgeable organic carbon is the portion of total (dissolved) organic carbon remaining in acidified samples after the inorganic (carbonate, bicarbonate, and dissolved carbon dioxide or carbon from non-living sources) and purgeable (hydrocarbons, ketones, aldehydes, and halogenated hydrocarbons) fractions have been removed, by sparging carbon-free (ultra-pure) air through the sample for a set time period. The carbon remaining (both labile and refractory) may serve as the electron/energy source for denitrification (20, 35). We will refer to non-purgeable organic carbon as total organic carbon (TOC) throughout the rest of the article.

Statistical analyses. Constructed wetland water samples were analyzed for total nitrogen (N = $NO_2$ + $NO_3$ + $NH_3$), soluble reactive phosphorus (P, $PO_4$), TOC, hydraulic loading rate (liters·day$^{-1}$·m$^{-2}$), temperature, and $SO_4$. Constructed wetland nutrient loading rates (g·m$^{-2}$·day$^{-1}$) were calculated and used to estimate the total mass of N and P entering the wetland (nursery load), and to determine how much of that N and P was assimilated or transformed in the CWS. If nutrient concentrations exported from the 5.5 d HRT and effluent treatments exceeded the nutrient concentration exported from the 3.5 d HRT treatment, nutrients were considered to be from internal CWS sources. Nutrient removal efficiencies (REs), on a percentage basis, were calculated for each CWS stage with regard to influent concentrations (Equation 1).

$$\text{Nutrient RE} = \left(1 - \frac{\text{3.5 d HRT, 5.5 d HRT, or Effluent (mg/L)}}{\text{Influent (mg/L)}}\right) \times 100 \quad (1)$$

Three sample moving-average trend lines were used to characterize the sampling data (Excel; Microsoft Corp., Redmond, WA).

## Results and Discussion

*Nitrogen removal efficiency.* Total N ($NO_2$-N, $NO_3$-N, and $NH_3$-N) loading into the free water surface CWS varied from year to year (Fig. 2A). Nitrogen removal efficiency (NRE) in 2003 averaged over 88.9 ± 1.9%, 99.5 ± 0.1%, and 98.4 ± 0.3% for the 3.5 d HRT, 5.5 d HRT, and effluent treatments respectively (Fig. 3A-1). In 2005, average loading was considerably higher than in 2003 (Fig. 2A, Table 1), and the average nitrogen removal efficiency of the 3.5 d HRT, 5.5 d HRT, and effluent treatments fluctuated from 60 to 99% over the three-month sampling period (Fig. 3B-1). However, during 2005, from March 30 to April 9, N loading was suspended because of pump breakdown (Fig. 3B-3), resulting in no water flow into the CWS. Subsequent NRE over all HRTs averaged 91.2 ± 1.6% and export concentrations were low (0.20 ± 0.05 mg·liter$^{-1}$ total N, Fig. 3B-2) due to decreased N loading. Shortly after flow resumed into the CWS, NRE was 52.0 ± 4.7% for the 3.5 d HRT stage and averaged 69.1 ± 2.5% for the 5.5 d HRT and effluent treatments (Fig. 3B-1).

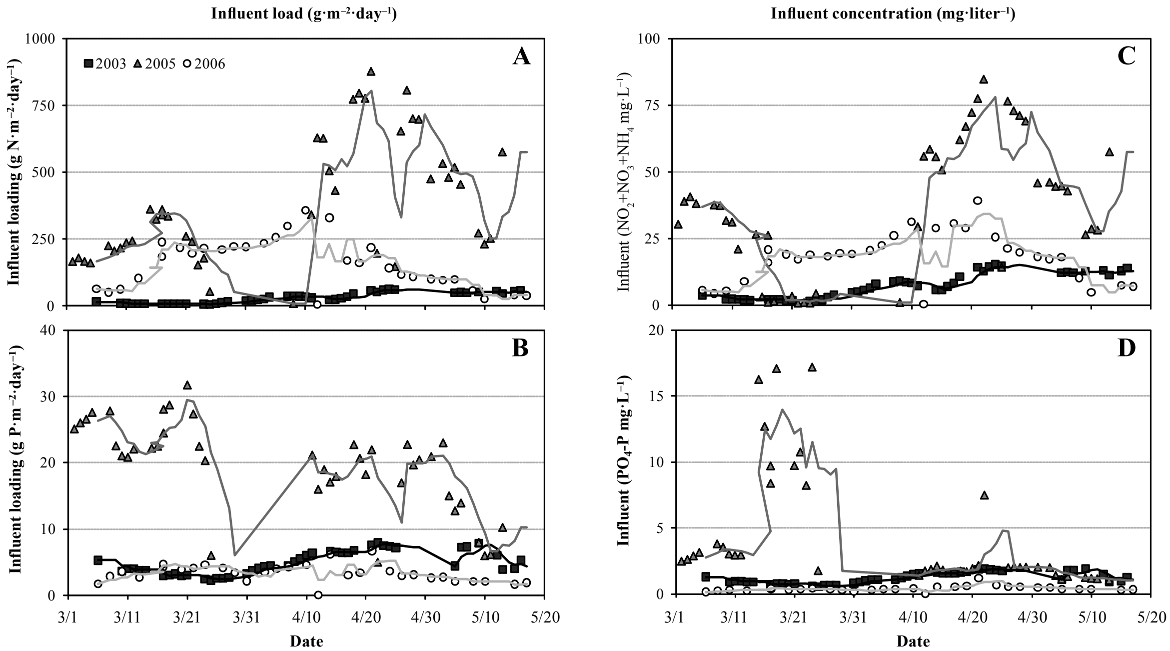


**Fig. 2. Total[1] nitrogen and soluble reactive phosphorus concentration (mg·liter$^{-1}$) and loading (g·m$^{-2}$·day$^{-1}$) from nursery production area into a free water surface constructed wetland system over three years[2] of spring sampling.**

[1]Nitrogen sources include nitrite ($NO_2$-N) and nitrate ($NO_3$-N) for all three years and ammonia ($NH_3$-N) for 2005 and 2006.
[2]Fitted lines are three sample moving average trendlines, no sampling occurred in 2004

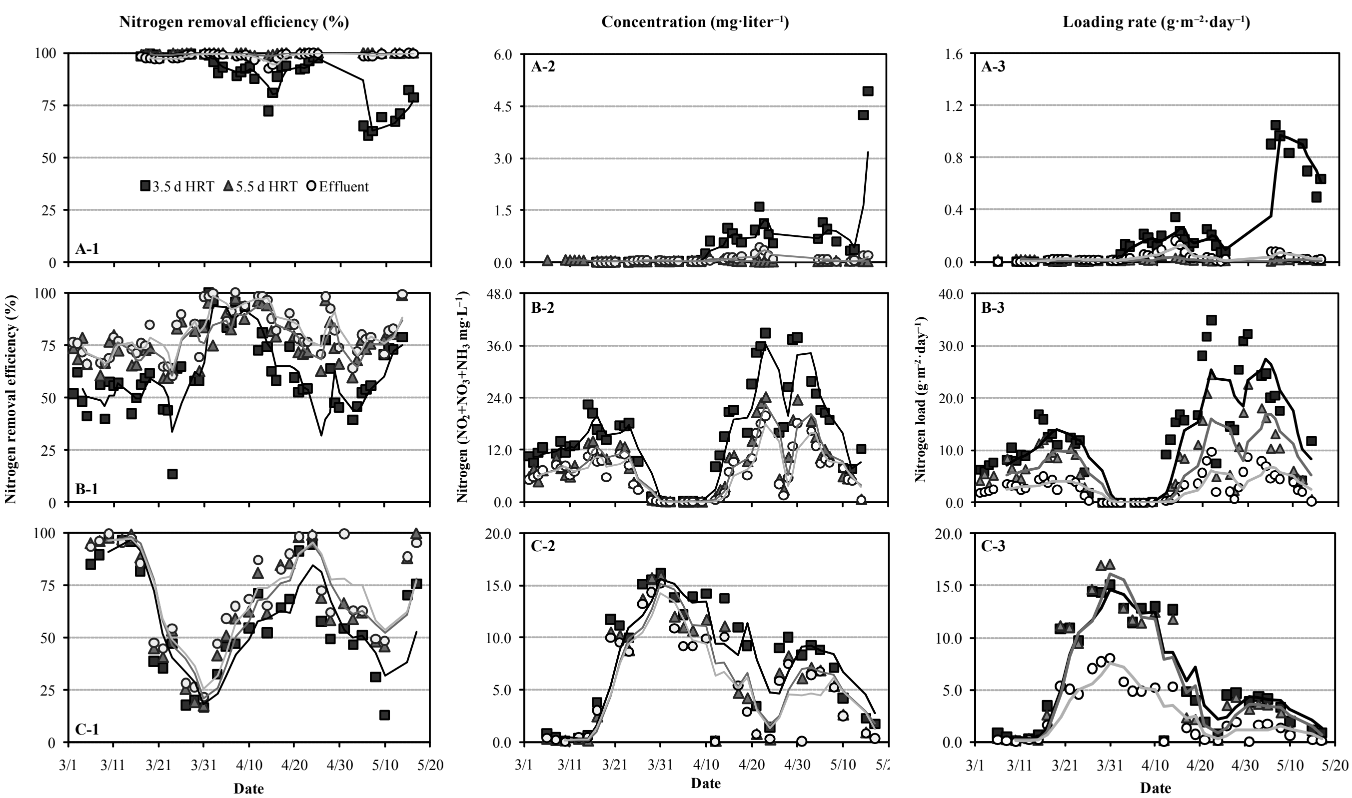


**Fig. 3. Total[1] nitrogen removal efficiency in and export from a free water surface constructed wetland system treating[2] runoff from a nursery production area over three years[3] of spring regeneration sampling in 2003 (A), 2005 (B), and 2006 (C).**

[1]Nitrogen sources include nitrite ($NO_2$-N) and nitrate ($NO_3$-N) for all three years and ammonia ($NH_3$-N) for 2005 and 2006.
[2]Treatment levels: 3.5 d HRT (3.5 day hydraulic retention time), 5.5 d HRT, and effluent.
[3]Fitted lines are three sample moving average trendlines, no sampling occurred in 2004.

**Table 1. Average[z] daily nitrogen[y] loading[x], assimilation, and export as measured in a two-stage free water surface constructed wetland treating[w] runoff from nursery production area runoff.**

| | | | Nitrogen ($mg \cdot liter^{-1}$) | | | Nitrogen ($g \cdot m^{-2} \cdot day^{-1}$) | | |
|---|---|---|---|---|---|---|---|---|
| | | | 2003 | 2005 | 2006 | 2003 | 2005 | 2006 |
| Nursery loading | Influent | March | 2.46 ± 0.26 | 25.3 ± 2.42 | 14.5 ± 1.84 | 10.1 ± 1.06 | 232.0 ± 19.0 | 166.0 ± 21.0 |
| | | April | 9.35 ± 0.78 | 44.2 ± 6.90 | 24.6 ± 2.73 | 38.4 ± 3.19 | 561.0 ± 65.9 | 200.0 ± 29.5 |
| | | May | 12.6 ± 0.27 | 40.5 ± 3.50 | 11.9 ± 2.20 | 51.8 ± 1.13 | 422.0 ± 44.2 | 65.9 ± 12.2 |
| Measured | 3.5 day HRT | March | 0.02 ± 0.00 | 12.2 ± 1.26 | 7.15 ± 1.93 | 0.01 ± 0.00 | 9.69 ± 0.85 | 6.77 ± 1.82 |
| | | April | 0.72 ± 0.08 | 17.3 ± 3.10 | 9.34 ± 1.45 | 0.15 ± 0.02 | 19.05 ± 2.56 | 6.95 ± 1.49 |
| | | May | 3.81 ± 0.31 | 16.5 ± 2.63 | 5.95 ± 1.20 | 0.81 ± 0.07 | 14.60 ± 2.89 | 2.78 ± 0.55 |
| | 5.5 day HRT | March | 0.02 ± 0.00 | 7.48 ± 0.88 | 6.51 ± 1.90 | 0.01 ± 0.00 | 6.66 ± 0.70 | 6.96 ± 2.03 |
| | | April | 0.04 ± 0.01 | 8.61 ± 1.91 | 6.76 ± 1.35 | 0.01 ± 0.00 | 9.95 ± 1.97 | 6.05 ± 1.56 |
| | | May | 0.03 ± 0.00 | 9.29 ± 1.83 | 4.16 ± 1.09 | 0.01 ± 0.00 | 9.25 ± 1.94 | 2.12 ± 0.54 |
| | Effluent | March | 0.04 ± 0.00 | 6.68 ± 0.74 | 6.28 ± 1.77 | 0.02 ± 0.00 | 2.96 ± 0.28 | 3.34 ± 0.94 |
| | | April | 0.12 ± 0.03 | 6.29 ± 1.51 | 6.00 ± 1.18 | 0.05 ± 0.01 | 3.57 ± 0.78 | 2.65 ± 0.67 |
| | | May | 0.11 ± 0.03 | 8.39 ± 1.55 | 3.17 ± 1.10 | 0.04 ± 0.01 | 4.18 ± 0.80 | 0.79 ± 0.28 |
| Internal loading | 3.5 day HRT | March | — | — | — | — | — | — |
| | | April | — | — | — | — | — | — |
| | | May | — | — | — | — | — | — |
| | 5.5 day HRT | March | — | — | — | — | — | 0.19 ± 0.21 |
| | | April | — | — | — | — | — | — |
| | | May | — | — | — | — | — | — |
| | Effluent | March | 0.03 ± 0.00 | — | — | 0.01 ± 0.00 | — | — |
| | | April | 0.09 ± 0.02 | — | — | 0.03 ± 0.01 | — | — |
| | | May | 0.08 ± 0.02 | — | — | 0.03 ± 0.01 | — | — |
| Assimilation | 3.5 day HRT | March | 2.44 ± 0.25 | 13.1 ± 1.16 | 7.36 ± 0.00 | 10.1 ± 1.06 | 222.0 ± 18.2 | 159.0 ± 19.1 |
| | | April | 8.63 ± 0.69 | 26.9 ± 3.80 | 15.29 ± 1.28 | 38.2 ± 3.17 | 542.0 ± 63.4 | 193.0 ± 28.0 |
| | | May | 8.78 ± 0.00 | 24.0 ± 0.87 | 5.95 ± 0.99 | 50.9 ± 1.06 | 407.0 ± 41.3 | 63.0 ± 11.6 |
| | 5.5 day HRT | March | 0.01 ± 0.00 | 4.74 ± 0.39 | 0.64 ± 0.03 | 0.00 ± 0.00 | 3.04 ± 0.15 | — |
| | | April | 0.68 ± 0.07 | 8.68 ± 1.19 | 2.58 ± 0.10 | 0.14 ± 0.01 | 9.10 ± 0.58 | 0.90 ± 0.00 |
| | | May | 3.79 ± 0.31 | 7.23 ± 0.79 | 1.79 ± 0.12 | 0.80 ± 0.06 | 5.34 ± 0.95 | 0.66 ± 0.01 |
| | Effluent | March | — | 0.80 ± 0.13 | 0.24 ± 0.13 | — | 3.70 ± 0.42 | 3.62 ± 1.09 |
| | | April | — | 2.32 ± 0.40 | 0.76 ± 0.17 | — | 6.38 ± 1.19 | 3.40 ± 0.89 |
| | | May | — | 0.90 ± 0.28 | 0.99 ± 0.00 | — | 5.07 ± 1.13 | 1.33 ± 0.26 |

[z]Average monthly value ± standard error of the mean.

[y]Total nitrogen measured as $NO_2$-N + $NO_3$-N + $NH_3$-N.

[x]Dash (—) represents no detected internal loading or assimilation, respectively.

[w]Treatment levels: 3.5 d HRT (3.5 day hydraulic retention time), 5.5 d HRT, and effluent.

This dramatic decrease in NRE resulted in sporadic N export (12.2 ± 1.7 $mg \cdot liter^{-1}$ $NO_3$-N) from the CWS, above the federally mandated limit of 10.0 $mg \cdot liter^{-1}$ (ppm) for $NO_3$-N from April 20 to May 2, 2005. Though monthly average effluent total nitrogen concentrations were 6.29 ± 1.51 and 8.39 ± 1.55 $mg \cdot liter^{-1}$ (ppm) for April and May respectively (Table 1, measured effluent 2005), well below the federal limit.

This brief period of relatively high N export, after 11 days with no flow into the CWS, highlights the importance of maintaining adequate flow rates into CWSs. Various N removal mechanisms are prevalent in CWS, denitrification accounts for 64 to 99% of $NO_3$-N removed, whereas plant assimilation, microbiota uptake, and dissimilatory reduction to $NH_4$-N account for 1–36% of measured $NO_3$-N removal (13). During the 11 day period of our study with no water or nutrient loading into the CWS, available $NO_3$-N declined to < 1.0 $mg \cdot liter^{-1}$, a minimum value identified by the US EPA (31) for maintaining denitrification activity. When denitrification potential declines or denitrifying bacteria acclimate to low (< 1.0 $mg \cdot liter^{-1}$) $NO_3$-N concentrations, up to 13 days may be required for denitrifying communities to attain complete denitrifying potential (25). We observed that 10 days were required for NRE efficiencies to return to pre-pump failure levels after flow into the CWS was reinstated. Denitrification was likely the major source of $NO_3$-N removal in the 3.5 d HRT treatment, whereas plant assimilation, microbiota uptake, or dissimilatory reduction to $NH_4$-N may have become more important means of nitrogen removal in the 5.5 d HRT and effluent treatments because $NO_3$-N concentrations were much lower (Fig. 3B-2).

During 2006, average loading increased for March and April sampling, but loading in May declined from both the 2003 and 2005 levels (Fig. 3C-3). This N loading decline can be attributed to decreased daily pump volume. Average May pump volume for 2003 and 2005 were 2.4 and 3.3 million liters (0.6 and 0.9 million gal) per day respectively, while

**Table 2. Average[z] daily phosphorus loading[y], assimilation, and export as measured in a two-stage free water surface constructed wetland treating[x] runoff from nursery production area runoff.**

| | | | Soluble reactive phosphate ($mg \cdot liter^{-1}$) | | | Soluble reactive phosphate ($g \cdot m^{-2} \cdot day^{-1}$) | | |
|---|---|---|---|---|---|---|---|---|
| | | | 2003 | 2005 | 2006 | 2003 | 2005 | 2006 |
| Nursery Loading | Influent | March | 0.69 ± 0.02 | 4.32 ± 0.76 | 0.31 ± 0.03 | 2.83 ± 0.08 | 23.9 ± 1.23 | 3.53 ± 0.29 |
| | | April | 1.47 ± 0.07 | 5.07 ± 1.12 | 0.51 ± 0.08 | 6.03 ± 0.30 | 32.2 ± 10.3 | 3.79 ± 0.49 |
| | | May | 1.40 ± 0.14 | 1.37 ± 0.12 | 0.39 ± 0.03 | 5.76 ± 0.57 | 12.9 ± 2.03 | 2.16 ± 0.15 |
| Measured | 3.5 day HRT | March | 1.02 ± 0.03 | 3.67 ± 0.13 | 0.22 ± 0.01 | 0.22 ± 0.01 | 2.18 ± 0.10 | 0.19 ± 0.01 |
| | | April | 1.25 ± 0.09 | 2.48 ± 0.45 | 0.22 ± 0.02 | 0.27 ± 0.02 | 1.24 ± 0.24 | 0.14 ± 0.02 |
| | | May | 1.89 ± 0.08 | 1.33 ± 0.03 | 0.33 ± 0.02 | 0.40 ± 0.02 | 1.30 ± 0.09 | 0.15 ± 0.01 |
| | 5.5 day HRT | March | 1.07 ± 0.04 | 3.70 ± 0.10 | 0.23 ± 0.02 | 0.40 ± 0.01 | 2.63 ± 0.12 | 0.25 ± 0.02 |
| | | April | 1.22 ± 0.08 | 1.98 ± 0.37 | 0.16 ± 0.02 | 0.45 ± 0.03 | 1.09 ± 0.22 | 0.12 ± 0.02 |
| | | May | 1.72 ± 0.07 | 1.11 ± 0.04 | 0.28 ± 0.02 | 0.65 ± 0.02 | 1.30 ± 0.12 | 0.14 ± 0.01 |
| | Effluent | March | 1.01 ± 0.04 | 3.58 ± 0.12 | 0.21 ± 0.02 | 0.39 ± 0.02 | 1.26 ± 0.06 | 0.11 ± 0.01 |
| | | April | 1.24 ± 0.08 | 1.70 ± 0.33 | 0.15 ± 0.02 | 0.47 ± 0.03 | 0.47 ± 0.08 | 0.05 ± 0.01 |
| | | May | 1.88 ± 0.05 | 0.88 ± 0.02 | 0.23 ± 0.04 | 0.71 ± 0.02 | 0.43 ± 0.02 | 0.06 ± 0.01 |
| Internal loading | 3.5 day HRT | March | 0.33 ± 0.01 | — | — | — | — | — |
| | | April | — | — | — | — | — | — |
| | | May | 0.49 ± 0.00 | — | — | — | — | — |
| | 5.5 day HRT | March | 0.05 ± 0.01 | 0.03 ± 0.00 | 0.01 ± 0.01 | 0.18 ± 0.01 | 0.45 ± 0.03 | 0.05 ± 0.01 |
| | | April | — | — | — | 0.18 ± 0.01 | — | — |
| | | May | — | — | — | 0.25 ± 0.00 | — | — |
| | Effluent | March | — | — | — | — | — | — |
| | | April | 0.03 ± 0.01 | — | — | 0.02 ± 0.00 | — | — |
| | | May | 0.16 ± 0.00 | — | — | 0.06 ± 0.00 | — | — |
| Assimilation | 3.5 day HRT | March | — | 0.64 ± 0.63 | 0.09 ± 0.01 | 2.61 ± 0.08 | 21.7 ± 1.13 | 3.33 ± 0.28 |
| | | April | 0.21 ± 0.00 | 2.59 ± 0.68 | 0.29 ± 0.06 | 5.76 ± 0.28 | 30.9 ± 10.1 | 3.65 ± 0.47 |
| | | May | — | 0.04 ± 0.09 | 0.06 ± 0.01 | 5.35 ± 0.55 | 11.6 ± 1.94 | 2.01 ± 0.14 |
| | 5.5 day HRT | March | — | — | — | — | — | — |
| | | April | 0.04 ± 0.01 | 0.50 ± 0.07 | 0.06 ± 0.01 | — | 0.15 ± 0.01 | 0.02 ± 0.00 |
| | | May | 0.17 ± 0.01 | 0.22 ± 0.00 | 0.05 ± 0.00 | — | — | 0.01 ± 0.00 |
| | Effluent | March | 0.05 ± 0.00 | 0.12 ± 0.00 | 0.02 ± 0.00 | 0.01 ± 0.00 | 1.38 ± 0.06 | 0.13 ± 0.01 |
| | | April | — | 0.28 ± 0.05 | 0.01 ± 0.00 | — | 0.62 ± 0.14 | 0.07 ± 0.01 |
| | | May | — | 0.23 ± 0.02 | 0.04 ± 0.00 | — | 0.87 ± 0.11 | 0.08 ± 0.00 |

[z]Average monthly value ± standard error

[y]Dash (—) represents no detected internal loading or assimilation, respectively.

[x]Treatment levels: 3.5 d HRT (3.5 day hydraulic retention time), 5.5 d HRT, and effluent.

daily pump volumes in May of 2006 averaged 1.7 million liters (0.5 million gal) per day. This decrease in daily pump volume coincided with decreased rainfall during the 2006 season; during the 2003 and 2005 sampling period total rainfall amounts were 31.8 and 49.5 cm (12.5 and 19.5 in) respectively, while in 2006 rainfall accumulation over the sampling period was only 17.0 cm (6.7 in). Because of this dry period, more water was recirculated by the nursery for irrigation purposes rather than released for treatment in the CWS. Nitrogen removal efficiency for 2006 fluctuated greatly over the three sampling months, with the greatest decrease in efficiency starting March 21 and peaking around March 31 (Fig. 3C-1).

Various environmental factors can influence NRE and contribute to decreased denitrification activity including increased water column dissolved oxygen concentrations, lack of adequate TOC to serve as an electron donor for denitrification, and plant species-specific effects (4, 8). We did not measure dissolved oxygen concentrations during this experiment. However, we did monitor dissolved oxygen on a monthly basis during another study on-site and recorded an increase from 1.0 ± 0.2 to 3.5 ± 0.6 $mg \cdot liter^{-1}$ dissolved oxygen in the 5.5 d HRT stage effluent from December through April in 2003, 2005, and 2006 (data not shown), so we can speculate that higher dissolved oxygen concentrations may have partially inhibited denitrification during this study. Higher dissolved oxygen concentrations may have increased dissimilatory reduction of $NO_3$-N to $NH_4$-N, but measured $NH_4$-N concentrations averaged 0.15 ± 0.03 $mg \cdot liter^{-1}$ for the 2006 sampling period and these measurements do not indicate dissimilatory reduction as a major mechanism of $NO_3$-N removal. Our TOC measurements (data not shown) indicated that TOC was not limiting denitrification. The CWS is established with many plant species, so species-specific effects may have influenced denitrification.

Even during periods of decreased NRE (efficiency < 80%), regardless of sampling year, the CWS functioned adequately to reduce average N export concentrations well below current water quality criteria limits (Table 1, measured effluent concentrations). This finding supports promoting CWS use for treatment of nutrient-rich runoff. However, we also observed that maintaining an adequate flow rate into the CWS was necessary to maintain healthy bacterial communities that contribute the majority of NRE.

*Phosphorus removal efficiency.* Soluble reactive phosphorus loading and removal dynamics in the CWS were very different from N loading and removal trends. Internal wetland loading processes contributed much of the P load exported from the CWS for the first few weeks of March 2003, 2005 and 2006 (Table 2, Fig. 4A-3, 4B-3, 4C-3). These results were consistent with the reports of other researchers where organic phosphorus from decaying plant material contributed to internal loading processes during the winter season along with phosphorus release from sediments under changing aerobic and anaerobic conditions (5, 14). Conditions that favor P release from sediment also occur if P concentrations entering a treatment stage are lower than the equilibrium P concentration of the sediment, and can result in net P export from a treatment stage rather than net P assimilation (19). Internal P loading from the 5.5 d HRT treatment is especially prevalent during March 2003 (0.05 ± 0.01 mg·liter$^{-1}$ $PO_4$-P), 2005 (0.03 ± 0.01 mg·liter$^{-1}$ $PO_4$-P), and 2006 (0.01 ± 0.01 mg·liter$^{-1}$ $PO_4$-P, Table 2). When P removal efficiency is > 0%, P assimilation is greater than P loading; if P removal efficiency is < 0%, internal loading processes (e.g. plant decay, bioturbation, sediment suspension, desorption, etc.) likely contributed to P generation and export (Fig. 4A-1, 4B-1, 4C-1).

During 2003, except for a 3 week period during April when plant growth was most active, P was exported from the CWS for the majority of the sampling season (Fig. 4A-1). Internal P loading occurred in the 5.5 d HRT and effluent stages, resulting in P export from the CWS (Table 2, Internal loading). We hypothesize that P assimilation occurred in the 3.5 d HRT stage in 2003 because active plant growth sequestered more P into plant tissues than was released from internal loading processes (Table 2, Internal loading). Other researchers have also noted that the majority of P removal occurred during periods of active plant growth (9). However, as plant species attained their mature size and growth, P removal efficiency (PRE) decreased below 0% and internal loading processes began to dominate P cycling in the CWS.

Soluble reactive phosphorus loading and export in 2005 and 2006 followed a different pattern than in 2003, with the 5.5 d HRT and effluent stages exporting less P than the 3.5 d HRT (Fig. 4B-1, 3 and 4C-1, 3). In 2005, the majority of P loaded was assimilated in the treatment stages, resulting in an average PRE of 53.0 ± 0.8% (Fig. 4B-1). This positive PRE in all CWS stages may be attributed to higher loading rates (Table 2, nursery loading influent), resulting in sorption processes and biological uptake dominating P cycling. In 2006, positive PRE began earlier in the treatment season (March 17) and continued through last sampling date in May, with average efficiency of approximately 51.2 ± 1.1% (Fig. 4C-1). The PRE values of the 5.5 d HRT and effluent treatments



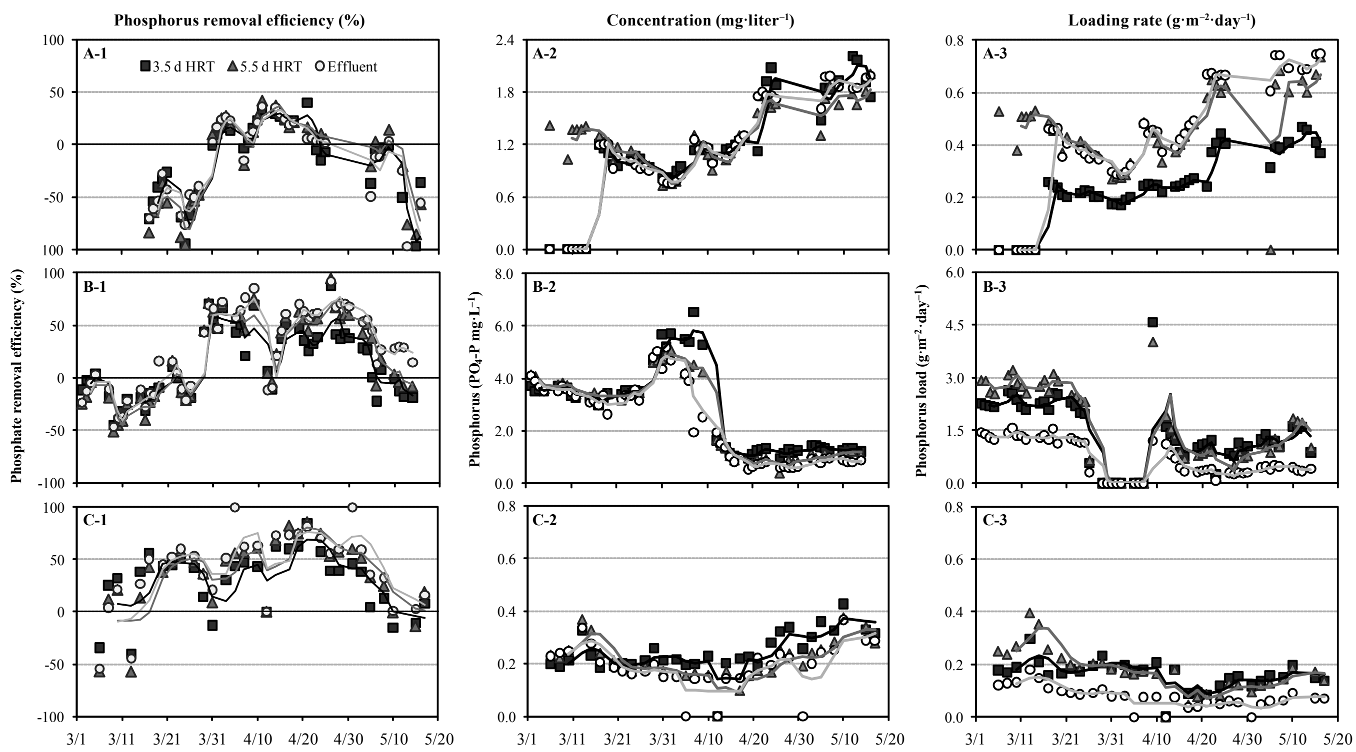


**Fig. 4. Soluble reactive phosphorus removal efficiency[1] in and export from a free water surface constructed wetland system treating[2] runoff from a nursery production area over three years[3] of spring sampling in 2003 (A), 2005 (B), and 2006 (C).**

[1]Solid line at 0 represents division between net assimilation and export from the HRT at which removal efficiency was calculated.

[2]Treatment levels: 3.5 d HRT (3.5 day hydraulic retention time), 5.5 d HRT, and effluent.

[3]Fitted lines are three sample moving average trendlines, no sampling occurred in 2004.

were generally higher than that for the 3.5 d HRT treatment. However, at times the 5.5 d HRT treatment exported more P than the 3.5 d HRT stage; this may be attributable to the sorption equilibrium status of the sediment. We hypothesize that the highest PREs correlated with periods of sustained plant uptake during the spring, but we cannot confirm this supposition because we did not harvest plant tissue during this experiment.

In summary, nutrient loading dynamics in this CWS were not only influenced by nursery production runoff, since it was the primary source of nutrients loaded into the CWS, but also by internal cycling processes, which influenced nutrient concentrations (primarily P) exiting the CWS. Soluble reactive P dynamics in the wetland were more variable than N dynamics. For optimum NRE, we observed that consistent water flow into the CWS were necessary. In most instances the 3.5 d HRT was adequate for removing N to concentrations below federal limits, but during 2005 when N loading rates were highest and pump problems occurred, treatment through the 5.5 d HRT stage was necessary to reduce N concentrations to acceptable levels.

Phosphorus removal efficiency was dependent upon plant growth over the spring and sediment equilibrium status. To optimize a CWS for P removal additional action is necessary, many researchers have studied this issue and recommend a horizontal subsurface flow cell lined with some P sorbing material (33). The free water surface CWS in this study did not consistently reduce P concentrations in effluent to federally recommended levels (0.1 mg·liter$^{-1}$ total P), but was able to assimilate P from both nursery and internal wetland sources during phases of active plant growth, which is also the time when the majority of nursery loading will occur. This CWS was able to consistently decrease N to < 7 mg·liter$^{-1}$ even when removal efficiency was below 80%. We found nursery concerns regarding internal nutrient loading of N during the spring from decaying detrital material to be unfounded because the CWS was able to remediate N from runoff water even when conditions were suboptimal.

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