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CONSTRUCTED WETLANDS

Long-Term Trends in Nitrogen Removal by an Aridland Constructed Treatment Wetland

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Abstract

Cities are increasingly pursuing more sustainable and resilient infrastructure. The increased use of Urban Ecological Infrastructure (UEI), including constructed treatment wetlands (CTW), may be particularly important for aridland cities with scarce water resources. In this paper we document eight years of nitrogen (N) dynamics in an aridland CTW in Phoenix, Arizona, USA, where N removal must be balanced by the trade-off of atmospheric water losses. We have documented a "biological tide", wherein transpiration-driven water loss is actively replaced by a slow movement of surface water into the marsh from adjacent open water areas. Our analysis combined long-term water budget data with nitrogen budgets for the vegetated marsh and the entire CTW system. The objective was to demonstrate how the biological tide enhanced N uptake in this aridland CTW. We attributed roughly 50% of the annual N uptake by the vegetated marsh to new water entering via the biological tide. Thus, while it seems counter-intuitive to design aridland CTWs to optimize transpirational water losses, our data suggested that careful design of the plant community and spatial configuration of vegetated marsh versus open water may enhance both the biological tide and N removal efficiency.

Keywords Constructed treatment wetlands · Urban sustainability · Wastewater treatment · Nitrogen budget · Water budget · Transpiration

Introduction

Urban living has become an increasingly common trend over the past two centuries. In fact, the proportion of human populations living in cities around the globe has increased from 10% to over 50% since 1900 and is projected to reach 80% by 2050 (Grimm et al. 2008). The increased density of urban systems offers opportunities for enhanced efficiency of infrastructure, but it also presents greater challenges for resource management. This is particularly true of water management in aridland cities. In the last century, cities have transformed into "sanitary cities" that rely on highly centralized, engineered, and expensive infrastructure designed to keep inhabitants healthy (Melosi 2000; Grove 2009). While this infrastructure can meet short-term demands, it often imposes large systemic inertias that hinder a city's ability to pursue novel or transformative new solutions to growing problems (Childers et al. 2014). To address these challenges, many cities are turning to Urban Ecological Infrastructure (UEI, *sensu* Childers et al. 2019) for more sustainable and resilient solutions. Constructed treatment wetlands (CTW) are a prime example of UEI (Greenway 2005); they provide cities with a relatively low cost and low maintenance solution to urban wastewater and water reclamation challenges (Wallace and Knight 2006; Kadlec and Wallace 2009; Nivala et al. 2013; Childers et al. 2015).

Constructed treatment wetlands are effective options for treating municipal wastewater effluent because of simple technologies and relatively low construction costs complemented by minimal active management (Wallace and Knight 2006; Nivala et al. 2013). Surface-flow treatment wetlands, for instance, are designed to remove various forms of pollution and excess nutrients from effluent (Kadlec and Wallace 2009; Fonder and Headley 2013). This type of CTW typically features a mix of open water areas, emergent vegetation, and waterlogged soils (Fonder and Headley 2013), with designs that are often specific to local or regional characteristics such as water quality regulations, site-specific conditions, and climate (Fonder and Headley 2010; Tanner et al. 2012).

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Water is a limiting resource in aridland cities, and many are using water reclamation strategies to meet current and future water demands (Greenway 2005). For example, in Phoenix, AZ, USA, virtually all municipal effluent is reused in some way (Metson et al. 2012). This includes the use of CTWs as a sustainable solution to treat effluent. However, using CTWs in arid or semi-arid climates may be challenging because of large water losses to the atmosphere in settings where water reuse is important (Green et al. 2006). This climatic trade-off is an issue in Phoenix. In Arizona, CTWs are designed and managed to remove nitrogen (N), which is the limiting nutrient in Arizona surface waters (Grimm and Fisher 1986) and is thus the main macronutrient regulated by law. For this reason, N has been the focus of our long-term CTW research.

Since summer 2011, the Wetland Ecosystems Ecology Lab at ASU has been quantifying the nitrogen and water budgets at the Tres Rios CTW, which is operated by the City of Phoenix Water Services Department and is part of the largest wastewater treatment plant in the city (Sanchez et al. 2016; Weller et al. 2016; Bois et al. 2017). In this system, we have found that large amounts of water are lost via transpiration – as much as 20–25% of the water overlying the vegetated marsh daily – and we have documented the horizontal advection of surface water into the marsh from adjacent open water areas to replace this transpirational loss (Bois et al. 2017). We have named this phenomenon the "biological tide" (Sanchez et al. 2016) due to its similarity to water movement in coastal wetlands in response to astronomical tides.

The objective of this paper was to provide quantitative evidence for how the biological tide is affecting N uptake by the vegetated marshes in the Tres Rios aridland CTW. In their recent review of 31 CTW systems from around the world that were designed to treat wastewater for reuse, Tao et al. (2017) included three from Arizona; our Tres Rios system was one of them. The discharge from the Tres Rios CTW enters the Salt River where it flows downstream for roughly 10 km to the Lower Buckeye Diversion dam. At this point, all flow is diverted from the river channel and used to irrigate agricultural fields. It is worth noting that this is the only stretch of the Salt River in the Phoenix Metro Area where the river is perennial. The data from Tao et al. (2017) allowed us to compare the efficacy of the Tres Rios CTW against a number of similar systems.

This research is a significant contribution to aridland CTW literature because it demonstrates through empirical evidence that the dry, hot climate in Phoenix enhances N removal at Tres Rios. While the Tres Rios CTW, and other similar systems, may present a direct trade-off between greater N removal and atmospheric water losses, it seems clear that the advantage of the former makes the latter acceptable. Furthermore, the use of CTW UEI saved the City of Phoenix from needing to make very costly investments in a technological solution to the N removal challenge (by Water Services Department estimates, an engineering solution would have cost up to 50-fold more). Additionally, the insights gained by this study will improve CTW design and management strategies, thus increasing wetland performance.

Methods

Site Description

This study took place at the Tres Rios CTW in Phoenix, AZ, USA, which is part of the city's largest wastewater treatment plant. Construction was completed in 2010, and Wetland Ecosystems Ecology Lab research began in 2011 with a focus on the largest wetland treatment cell that is composed of 21 ha of vegetated marsh and 21 ha of open water with several small islands (Fig. 1). In this study we differentiated between two systems: the entire 42 ha system and the 21 ha vegetated marsh subsystem. The CTW receives 95,000 to 270,000 m^3 d^{-1} of effluent, depending on the time of year. Open water depths are 1.5–2.0 m, while vegetated marsh depths average roughly 25 cm; these values rarely varied due to management practices. The wetlands were originally planted with seven native plant species: *Typha latifolia*, *Typha domingensis*, *Schoenoplectus acutus*, *Schoenoplectus americanus*, *Schoenoplectus californicus*, *Schoenoplectus maritimus*, and *Schoenoplectus tabernaemontani* (Weller et al. 2016). The growing season for the wetland vegetation is defined as March through September, while the cooler season is October through February. The 42 ha CTW has a designed hydraulic residence time of four days, which means that much of the water entering the Tres Rios system leaves without ever contacting the vegetated marsh. Steidl et al. (2019) recommended longer residence times to increase nitrogen retention in surface flow CTW systems and Akratos and Tsihrintzis (2007) reported that an eight day retention time was sufficient for subsurface flow CTW systems located in hot climates.

Fig. 1 Aerial image of the Tres Rios Constructed Treatment Wetland in Phoenix, AZ. Arrows represent the inflow and outflow, and numbers and white lines show the location of the marsh transects where water quality samples were taken

Water Budget Calculations

We quantified transpiration at intervals of two months using a LICOR LI-6400 (LI-COR, Lincoln, Nebraska, USA) handheld infrared gas analyzer that measured water vapor flux from the leaves or stems of plant species groups in 50 cm intervals from the water surface to the tip of the plant. Following Weller et al. (2016), the plant species groups were *T. latifolia* + *T. domingensis*, *S. californicus*, *S. acutus* + *S. tabernaemontani, and S. americanus* when present. The infrared gas analyzer measurements captured critical ambient atmospheric conditions such as photosynthetically active radiation (PAR), air temperature, and relative humidity (see Sanchez et al. 2016 for more details).

We scaled leaf-level transpiration rates in space, or into whole-system transpiration volumes, using our estimates of species-specific live macrophyte biomass (per Weller et al. 2016). Hourly meteorological data from the on-site meteorological station maintained by the City of Phoenix were used to scale the leaf-level transpiration rates in time and to estimate hourly transpiration (per Sanchez et al. 2016). These transpirational water losses were consistently highest in July, showing a strong seasonal pattern when solar radiation, air temperature, and plant biomass were at annual maxima (Sanchez et al. 2016; Weller et al. 2016). Plant-mediated surface water movement (the biological tide) was also strongest during the hot summer months (Bois et al. 2017). We calculated the whole-system water budget as (inflow + precipitation) – (outflow + transpirational loss + open water evaporation; see Sanchez et al. 2016 for more details). By integrating these water budget data with the N budget, we were able to discern the role of the biological tide in enhancing N removal by the vegetated marsh.

Water Quality

We utilized a dual-gradient experimental design in this study. The first gradient focused on the entire 42 ha system from inflow to outflow (blue arrows in Fig. 1), and the second gradient focused on the 21 ha vegetated marsh with 10 evenly distributed transects that ran from the open water interface to the shore (white lines in Fig. 1; see Weller et al. 2016 for more details). We collected triplicate surface-water grab samples at the inflow and outflow and at the open water interface and shore ends of three of the 10 marsh transects (numbers shown in Fig. 1) and also measured water temperature, pH, and conductivity at all sampling locations. All samples were collected in acid-washed 1 L Nalgene bottles and stored on ice until processing and analysis at the lab. Unfiltered samples for inorganic N analysis (nitrite = NO_2^-, nitrate = NO_3^-, and ammonium = NH_4^+) were centrifuged to remove particulates and analyzed on a Lachat Quick Chem 8000 Flow Injection Analyzer (detection limit 0.85 μg NO_3^- N L^{-1} and 3.01 μg NH_4^+ N L^{-1}). To calculate N flux at the whole-system and marsh scales, N concentrations were multiplied by their respective water fluxes. The whole-system calculations comprised inflow and outflow water volumes, whereas the marsh calculations included the average water volume overlying the marsh, accounting for underwater plant stem volume, plus additional water entering via the biological tide.

Additional calculations included whole-system areal removal rate (ARR) and concentration removal efficiency (CRE; Palmer et al. 2009). The ARR was calculated as N removal per square meter each month (g N m^{-2} mo^{-1}), and the CRE was calculated as the percent (%) of N reduction:

$$ARR = [q \times (N_{in} - N_{out})/1000]/a$$
$$CRE = [(N_{in} - N_{out})/N_{in}] \times 100$$

where q = hydraulic loading rates (L mo^{-1}), N = inlet and outlet concentrations of NO_2^-, NO_3^-, and NH_4^+ (mg L^{-1}), and a = area (m^2).

A Plant-Mediated Biological Tide

The biological tide is a unique phenomenon involving plant-mediated surface water movement in marshes of the Tres Rios CTW, deriving its name from active horizontal advection from open-water areas similar to that seen in coastal wetland tides (Bois et al. 2017). It is the lateral movement of water into the vegetated marshes that brings additional N with it, thus improving the N removal efficiency of the marsh. This phenomenon is particularly strong in summer months, when plant productivity, air temperature, and solar radiation are high while relative humidity is very low. In order to calculate the biological tide's impact on N removal by the marsh, we calculated the water volume lost to transpiration each month based on an average 25 cm water depth and after accounting for the volume occupied by underwater plant stems (see Sanchez et al. 2016 for more details). Nitrogen uptake calculations were performed separately for the biological tide, marsh, and whole-system. We compared the biological tide contribution to both whole-system N uptake values and marsh N uptake values to demonstrate its relative role in nitrogen removal at both scales and during two seasons (the growing season = March – September, and the cool season = October – February):

$$\text{Biological Tide} = v_t\,(N_{water} - N_{shore})$$
$$\text{Total Marsh} = \text{Biological Tide} + v_m\,(N_{water} - N_{shore})$$
$$\text{Whole-System} = (q_{in} \text{ x } N_{in}) - (q_{out} \text{ x } N_{out})$$

where v_t = transpiration volume (L), v_m = marsh water volume (L), N = respective nitrogen concentrations (mg L^{-1}), and q = respective inflow and outflow volumes (L mo^{-1}).

We considered these estimates to be conservative for a number of reasons: 1) nighttime transpiration data were not collected due to site access restrictions; 2) transpiration values were not calculated when hourly meteorological station data were missing or unreliable; and 3) volumes of water displaced by thatched dead vegetation on the marsh surface could not be accounted for, thus our estimates of total water volume overlying the vegetated marsh were overestimates.

Results and Discussion

We estimated the impact of the biological tide on nitrogen removal by calculating and comparing the water and N budgets. First, we calculated plant biomass and transpiration to determine the significance of vegetation-driven water losses to the water budget. Constructing the N budget provided insight into the relationship between these transpirative water losses and N removal. Our comparison of these budgets revealed a tradeoff between high transpiration rates and the biological tide's enhancement of N uptake throughout the treatment wetland. Thus, future CTW designs in aridland climates may consider strategies to achieve greater amounts of transpirative water losses in order to enhance overall N removal for water treatment.

Biomass and Transpiration

We calculated daily biomass and transpiration values since July 2011, and both show consistent seasonal trends with the highest values in the hot summer months and the lowest values during the cooler winter months. There was a strong relationship between live plant biomass and transpiration losses from the marsh – the latter of which drives the biological tide and its transport of additional N into the marsh (Fig. 2; Weller et al. 2016, Bois et al. 2017). The close coherence between the biomass and transpiration data makes sense, as we scaled leaf-specific transpiration measurements to the entire marsh using our estimates of live plant biomass.

The hot and dry climate is the clear driver of the biological tide phenomenon in the Tres Rios CTW. The plant-mediated water losses that we documented in these marshes were comparable to transpiration rates reported for other wetlands. *Typha* spp. transpired at a peak rate of 23 mm d^{-1} in a Mediterranean-climate CTW (Pedescoll et al. 2013). Tuttulomondo et al. (2016) reported peak summer daily transpiration rates by *Typha* species of about 35 mm d^{-1} for a demonstration CTW in Sicily, Italy – a rate similar to the 30–35 mm d^{-1} we have reported for Tres Rios (Sanchez et al. 2016). However, transpiration rates were noticeably lower in the Florida Everglades for the same *Typha* species (Koch and Rawlik 1993). While south Florida is hot, particularly in the summer, the climate is also quite humid, and relative humidity is an important control on transpiration rates. Eichelmann et al. (2018) reported eddy flux-based evapotranspiration losses from three restored wetlands in California's Sacramento-San Joaquin River delta of 996–1140 mm yr^{-1}, or roughly 2.7 mm d^{-1} in this cooler Mediterranean climate.

Water Budget

Daily inflow and outflow totals have been measured by the City of Phoenix since January 2012, when flow gauges began collecting data. Despite the minimal amount of precipitation in this dry climate (the long-term average annual precipitation is 20 cm), rain inputs have always been included in our water budget estimates (Sanchez et al. 2016). The other components of the budget were internal water losses via transpiration and open water evaporation (for details, see Sanchez et al. 2016, Bois et al. 2017). Notably, the CTW was constructed over a clay lining, so losses to groundwater are considered to be negligible. Transpiration and total internal water losses

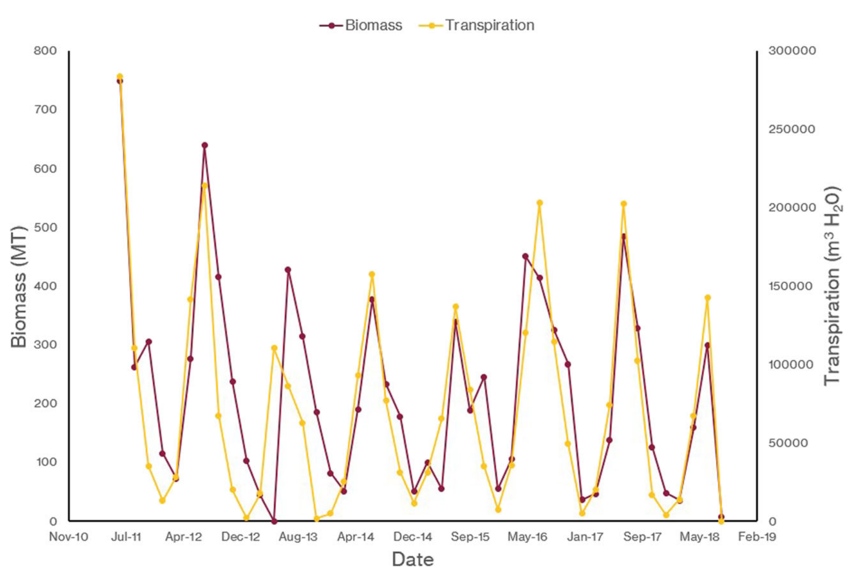

Fig. 2 Total plant biomass of the 21 ha of marsh and monthly total transpirative water losses from the marsh

followed the same seasonal trends as biomass and were highest during the warmest months (Fig. 3). Due to inaccurate inflow data prior to July 2013, we could not estimate the whole-system water budget until after this date. We estimated that, from July 2013 to July 2018, approximately 35% of the average annual internal water loss was due to transpiration, increasing to 44–66% during the growing season. Open-water evaporation accounted for the remaining internal water losses.

Sanchez et al. (2016) found that, on average, 70% of the whole-system water budget deficit was represented by transpiration + evaporation, with deficit defined as the difference between outflow and inflow. They attributed the remaining 30% to estimation errors. This is significantly higher than transpiration contributions to water budgets of wetlands located in cooler or more humid places. For example, evapotranspiration accounted for 13% of the water budget at a CTW in the Netherlands (Meuleman et al. 2003), 10% for a CTW in Venice, Italy (Favero et al. 2007), and only 3% at a CTW in Missouri, USA (Kadlec et al. 2010). Wu et al. (2013) found the highest N removal rates during summer months by a CTW in northern China, when air temperature and transpiration rates were maximal. Furthermore, large transpirational water losses are also drawing oxygenated, N-rich water into the soils and likely enhancing coupled nitrification-denitrification (sensu Martin et al. 2003). Thus, Arizona's arid climate and the biological tide it drives means that greater volumes of water and N are making contact with vegetated marsh and its biogeochemically active soils, where most N uptake and processing is taking place.

Nitrogen Budget

We estimated the N budget for the entire 42 ha system and separately for the 21 ha vegetated marsh to differentiate their dynamics while acknowledging that coupled nitrification-denitrification in the wetland soils is probably responsible for most N processing at Tres Rios (Kadlec 2008). Management schemes that periodically lower water levels to aerate wetland soils have been shown to make CTW systems more efficient at N removal because this couples nitrification-denitrification processes in the soils (Pires et al. 2020). But the vertical movement of oxygenated, N-rich water into marsh soils that is driven by high rates of transpiration also enhances this biogeochemical coupling (Martin et al. 2003).

We calculated whole-system N concentration differences for every other month (July 2011 to January 2018) using the inflow and the outflow data. Nitrite (NO_2^-) concentrations in the inflow water ranged from 0.084 to 2.7 mg L^{-1} and outflow concentrations ranged from 0.062 to 3.2 mg L^{-1} (Fig. 4a). From these values we estimated an average whole-system NO_2^- ARR of 0.16 ± 0.27 (SE) g N m^{-2} mo^{-1}, or 63 ± 110 kg N mo^{-1} for the 42 ha system (Fig. 5a). Nitrate (NO_3^-) concentrations were higher and showed greater temporal variability, with inflow concentrations ranging from 1.5 to 7.5 mg L^{-1} and outflow concentrations from 1.0 to 6.8 mg L^{-1} (Fig. 4b). The average whole-system NO_3^- ARR was 5.4 ± 2.6 (SE) g N m^{-2} mo^{-1}, or 2200 ± 1000 kg N mo^{-1} for the entire system (Fig. 5b). Concentrations of ammonium (NH_4^+) in the inflow water ranged from 0.33 to 2.1 mg L^{-1} while outflow concentrations ranged from 0.056 to 1.7 mg L^{-1} (Fig. 4c), and we estimated an average whole-system NH_4^+ ARR of 3.3 ± 0.57 (SE) g N m^{-2} mo^{-1}, equating to 1400 ± 240 kg N mo^{-1} for the 42 ha system (Fig. 5c).

We estimated N flux within the vegetated marsh by analyzing the concentration differences between the open water interface and the shore. The NO_2^- concentrations at the open water interface ranged from 0.066 to 3.4 mg L^{-1} while the concentrations near the shore ranged from 0.0033 to 0.28 mg L^{-1} (Fig. 6a). From these values we estimated an

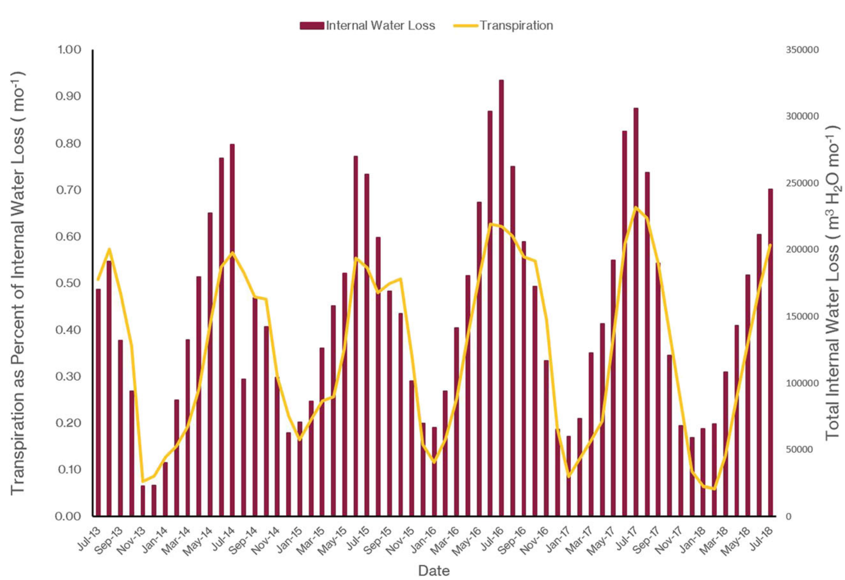

Fig. 3 Transpiration as a percentage of total, or internal, water loss to demonstrate the magnitude of transpirative water losses in the Tres Rios CTW

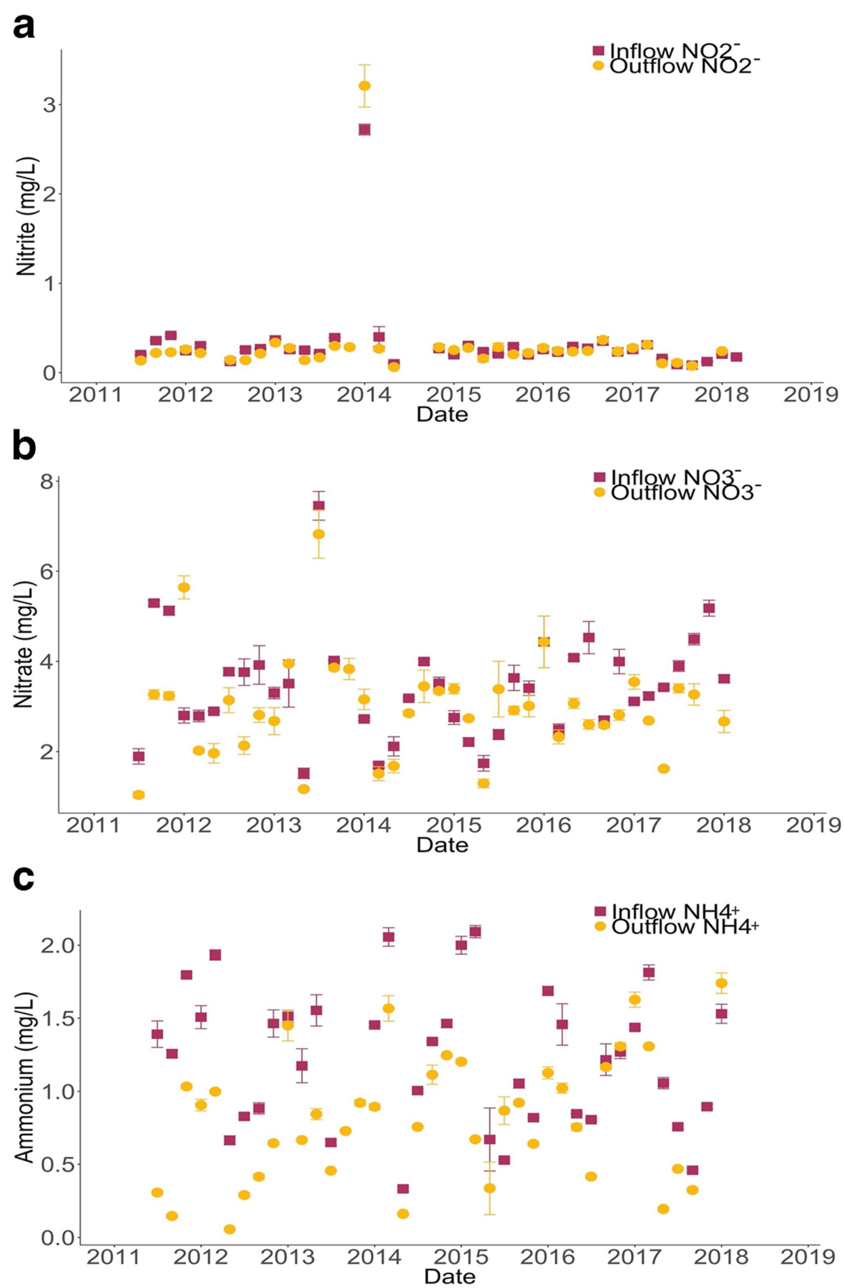

Fig. 4 Concentrations of nitrogen as it enters (inflow) and leaves (outflow) the treatment wetland at the whole-system scale for (top) NO_2^-, (middle) NO_3^-, and (bottom) NH_4^+

average NO_2^- ARR of 0.13 ± 0.03 (SE) g N m^{-2} mo^{-1}, or 28 ± 6.3 kg N mo^{-1} by the 21 ha marsh (Fig. 7a). As with the whole-system data, NO_3^- concentrations at the open water interface showed the highest temporal variability, ranging from 0.59 to 8.2 mg L^{-1} (Fig. 6b) while concentrations near the shore ranged 0.00 to 1.7 mg L^{-1}, resulting in an average NO_3^- ARR of 1.6 ± .16 (SE) g N m^{-2} mo^{-1} or 340 ± 33 (SE) kg N mo^{-1} in the 21 ha marsh (Fig. 7b). Concentrations of NH_4^+ at the open water interface ranged from 0.26 to 1.8 mg L^{-1} while shore concentrations ranged from 0.039 to 2.4 mg L^{-1} (Fig. 6c). These equated to an average NH_4^+ ARR of 0.28 ± 0.076 (SE) g N m^{-2} mo^{-1}, or 60.0 ± 16.0 kg N mo^{-1} at the 21 ha marsh scale (Fig. 7c).

Comparing our N uptake estimates to other CTWs located in similar climates was challenging because the literature on aridland CTW dynamics is not deep. Tao et al. (2017) reported annual average inflow NH_4^+ and $NO_3^- + NO_2^-$ concentrations, for 2014, of 1.19 and 1.7 mg L^{-1}, respectively, and average outflow concentrations of <0.8 and 2.1 mg L^{-1} NH_4^+ and $NO_3^- + NO_2^-$, respectively. These values suggest that the Tres Rios CTW was a sink for NH_4^+ and a source of $NO_3^- + NO_2^-$ in 2014. These findings were not consistent with our longer-term data which showed consistently lower outflow fluxes of all three constituents out of the system relative to inflow fluxes and consistently positive ARR (Fig. 5). One likely reason for this discrepancy is that the volume of

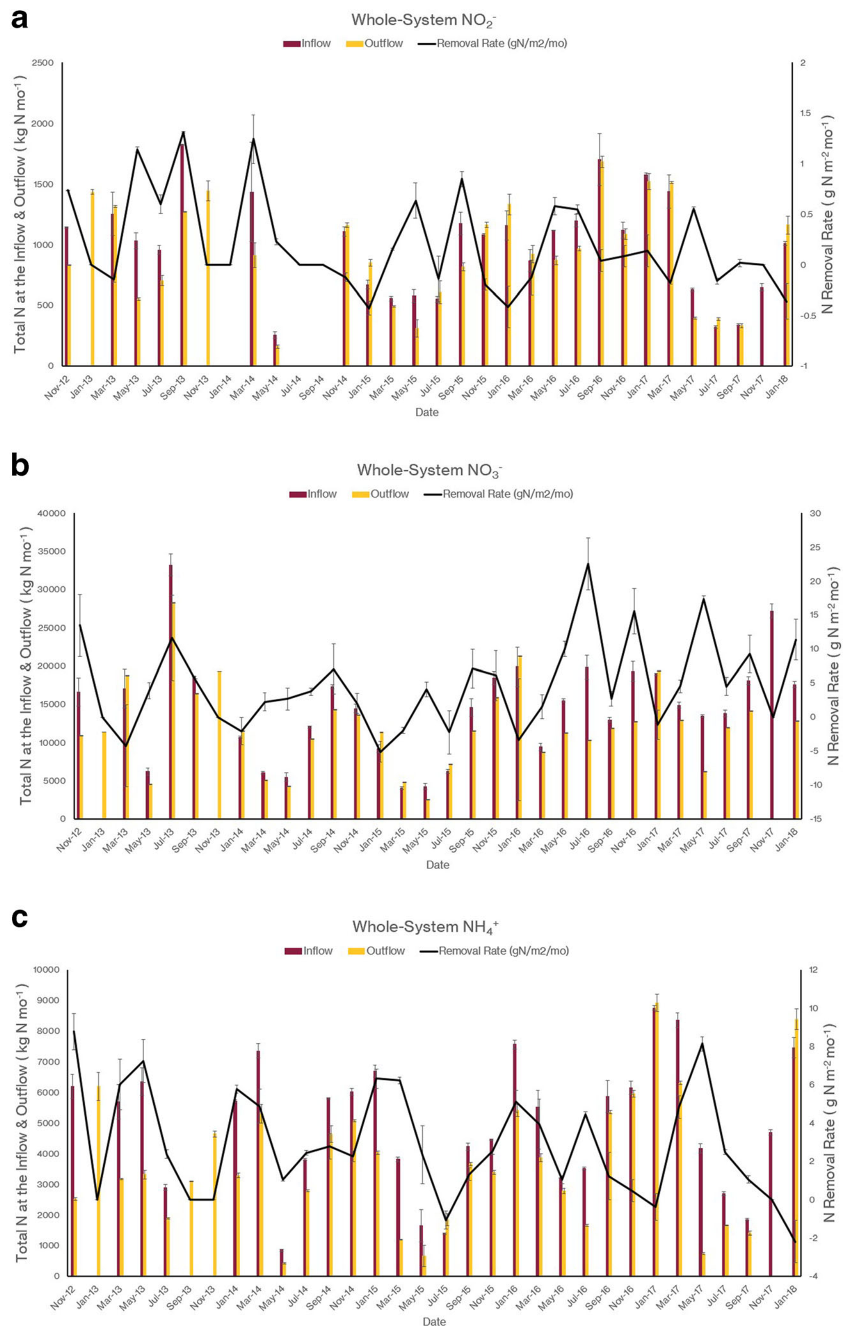

Fig. 5 Nitrogen flux into (inflow) and out of (outflow) the CTW and the N removal rates (ARR) at the whole-system scale for (top) NO_2^-, (middle) NO_3^-, and (bottom) NH_4^+

water leaving the system was always considerably lower than the inflow volume, because of internal losses via transpiration and evaporation. Thus, inflow and outflow concentration data do not tell the full story of whole-system fluxes.

A semi-arid CTW located near the Santa Ana River, California, USA, included six 0.13 ha macrocosms that were designed to remove NO_3^- from wastewater effluent (Bachand and Horne 1999). Most of the N removal by these small CTWs was attributed to uptake by *Typha* spp. (Bachand and Horne 1999), which is similar to the Tres Rios CTW (Weller et al. 2016). Inflows to the Santa Ana River macrocosms had average NO_3^- and NH_4^+ concentrations of 9.27 mg N L^{-1} and 0.16 mg N L^{-1} with average concentration decreases of 0.90 mg N L^{-1} and -0.05 mg N L^{-1}, respectively (Table 1). In comparison, N removal rates by the Tres Rios CTW were, on average, 2.49% and 75.7% more efficient at the 42 ha whole-system scale and 65.3% and 90.1% more efficient within the 21 ha marsh for NO_3^- and NH_4^+, respectively, in spite of the two systems having somewhat similar climates and plant species (Table 1).

The type and diversity of plant species in a CTW are likely one important driver of CTW system effectiveness. In their review of N removal by CTW systems, (Lee et al. 2009) reported that while monospecific stands of vegetation

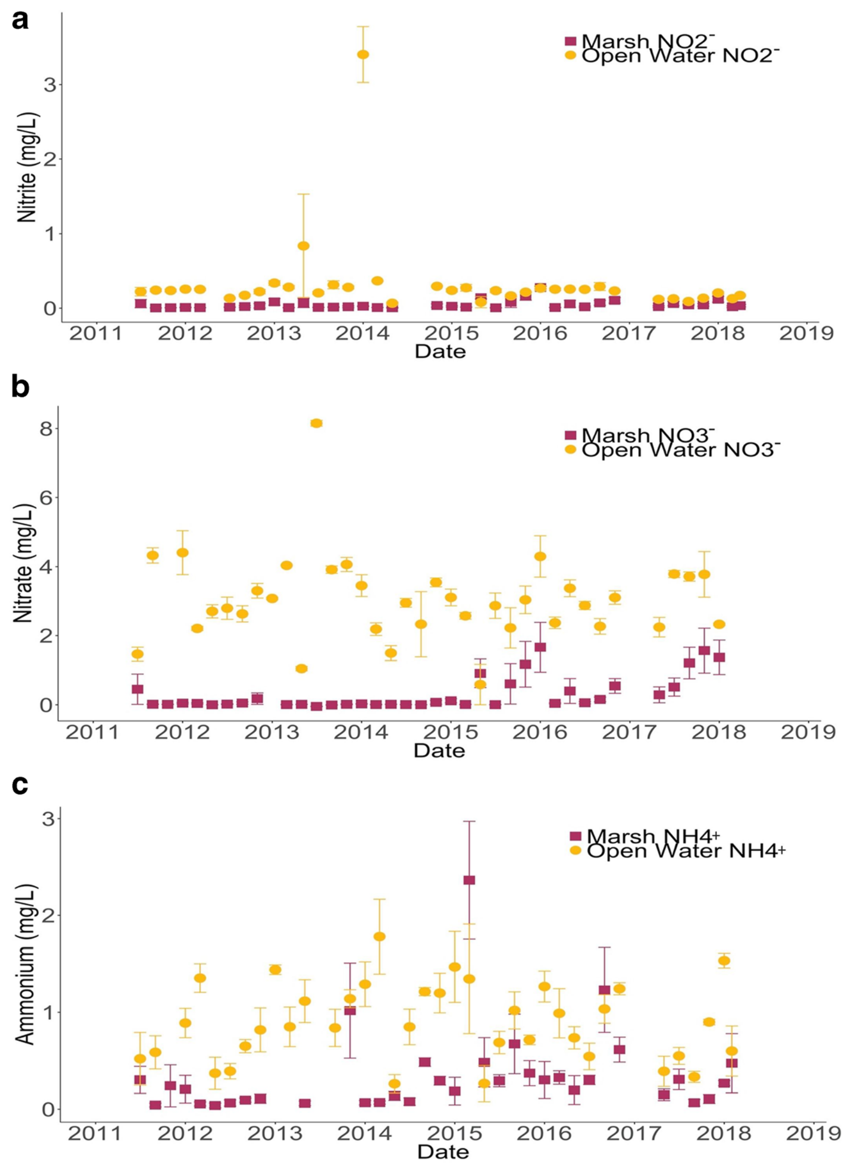

Fig. 6 Concentrations of nitrogen at the open water interface (open water) and near the shore (marsh), for the marsh subsystem, for (top) NO_2^-, (middle) NO_3^-, and (bottom) NH_4^+

achieved optimal N removal rates, species diversity made the plant community adaptable to its surroundings and more resilient to disturbances. Barbera et al. (2009) reported NO_3^- removal efficiencies for two different experimental CTWs in Sicily, Italy – one vegetated by *Phragmites australis* and the other by an assortment of plant species that spontaneously colonized the system. They found an average removal rate of 87% by the monotypic system and a 70% removal rate by the species-diverse system. In contrast, Weller et al. (2016) proposed that a more diverse macrophyte community led to higher N removal at the Tres Rios CTW. In their analysis, they found that *Typha* spp. accounted for around 70% of the total plant biomass, with *S. acutus* + *S. tabernaemontani*, *S. americanus*, and *S. californicus* filling out the plant community (Weller et al. 2016). In July 2011, these respective plant species showed average transpiration rates of 30 ± 0.8, 16 ± 0.4, 9.6 ± 0.02, and 0.6 ± 0.3 mm H_2O day^{-1} (Sanchez et al. 2016). Accordingly, the proportion of total N uptake during the growing season attributed to these respective species or species groups was 70%, 18%, 10%, and 2% (Weller et al. 2016). These findings suggested to Weller et al. (2016) that an ideal aridland CTW design should be designed for a diverse plant community while also strategically emphasizing a dominance of plant species with higher transpiration rates in order to maximize the biological tide and thus the system's overall N removal efficiency. In order to test this recommendation, we estimated the amount of N removed by these biological tide effects and present these data in the next section.

Contribution of the Biological Tide to N Uptake

Our N budget results confirmed that the biological tide is responsible for enhanced N removal by the Tres Rios CTW. In most months, a substantial fraction or even the majority of

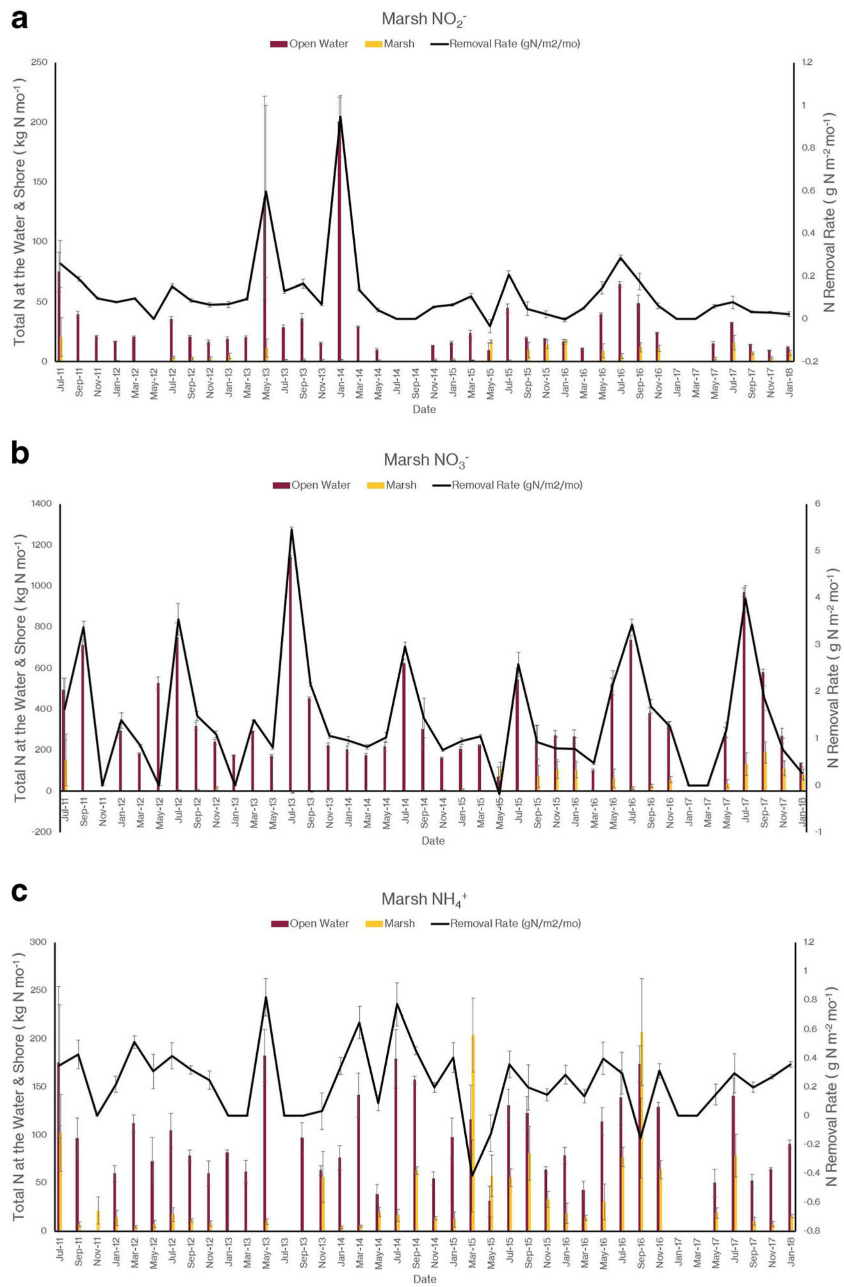

Fig. 7 Nitrogen flux at the open water interface (open water) and near the shore (marsh), and the N removal rates (ARR) for the marsh subsystem, for (top) NO_2^-, (middle) NO_3^-, and (bottom) NH_4^+

Table 1 Comparison of data from a semi-arid CTW macrocosm in Santa Ana CA USA and from Tres Rios, including average dissolved inorganic nitrogen (N) concentrations, average decrease in N concentrations from inflow to outflow, and concentration removal efficiencies (CRE)

Analyte	Measurement	Santa Ana River CA Macrocosms *0.13 ha (×6)*	Tres Rios (whole-system) *42 ha*	Tres Rios (marsh only) *21 ha*
NO_2^-	Avg. Conc.	N/A	0.32 mg N L^{-1}	0.32 mg N L^{-1}
	Avg. Decr.	N/A	0.018 mg N L^{-1}	0.28 mg N L^{-1}
	CRE	N/A	*9.55%*	*74.8%*
NO_3^-	Avg. Conc.	9.27 mg N L^{-1}	3.5 mg N L^{-1}	3.1 mg N L^{-1}
	Avg. Decr.	0.90 mg N L^{-1}	0.47 mg N L^{-1}	2.7 mg N L^{-1}
	CRE	*9.71%*	*12.2%*	*85.4%*
NH_4^+	Avg. Conc.	0.16 mg N L^{-1}	1.2 mg N L^{-1}	0.90 mg N L^{-1}
	Avg. Decr.	−0.05 mg N L^{-1}	0.43 mg N L^{-1}	0.55 mg N L^{-1}
	CRE	*−31.3%*	*34.0%*	*58.8%*

N being sequestered by the marsh was N being sourced by biological tidal advection of new water (Fig. 8). The advection of new surface water and N into the marsh, driven by this unique phenomenon, increased the amount of N coming into contact with wetland vegetation and soils. Because virtually all N in the water overlying the marsh was sequestered (Fig. 7), this increased N supply into the marsh enhanced N uptake by the entire Tres Rios CTW, with the most dramatic effects seen during the hot, dry summer months (Fig. 8).

We compared N removal by the 42 ha Tres Rios CTW to several other CTW systems. The first was a smaller 9.3 ha subtropical CTW in Taiwan (Hsueh et al. 2014). The average concentration removal efficiencies (CRE) for ammonium by this system were consistently around 85% throughout the year (Hsueh et al. 2014) while the Tres Rios marsh CRE averaged about 59% NH_4^+ removal efficiency and the whole-system CRE was roughly 34% (Table 1). The most notable difference between the two wetland systems was the seasonal variation in N uptake. At Tres Rios, the system's highest areal removal rates were consistently highest in warmer months and the lowest rates were during the cooler season (Figs. 5 and 7). By contrast, values for NH_4^+ CRE differed by only 1% in

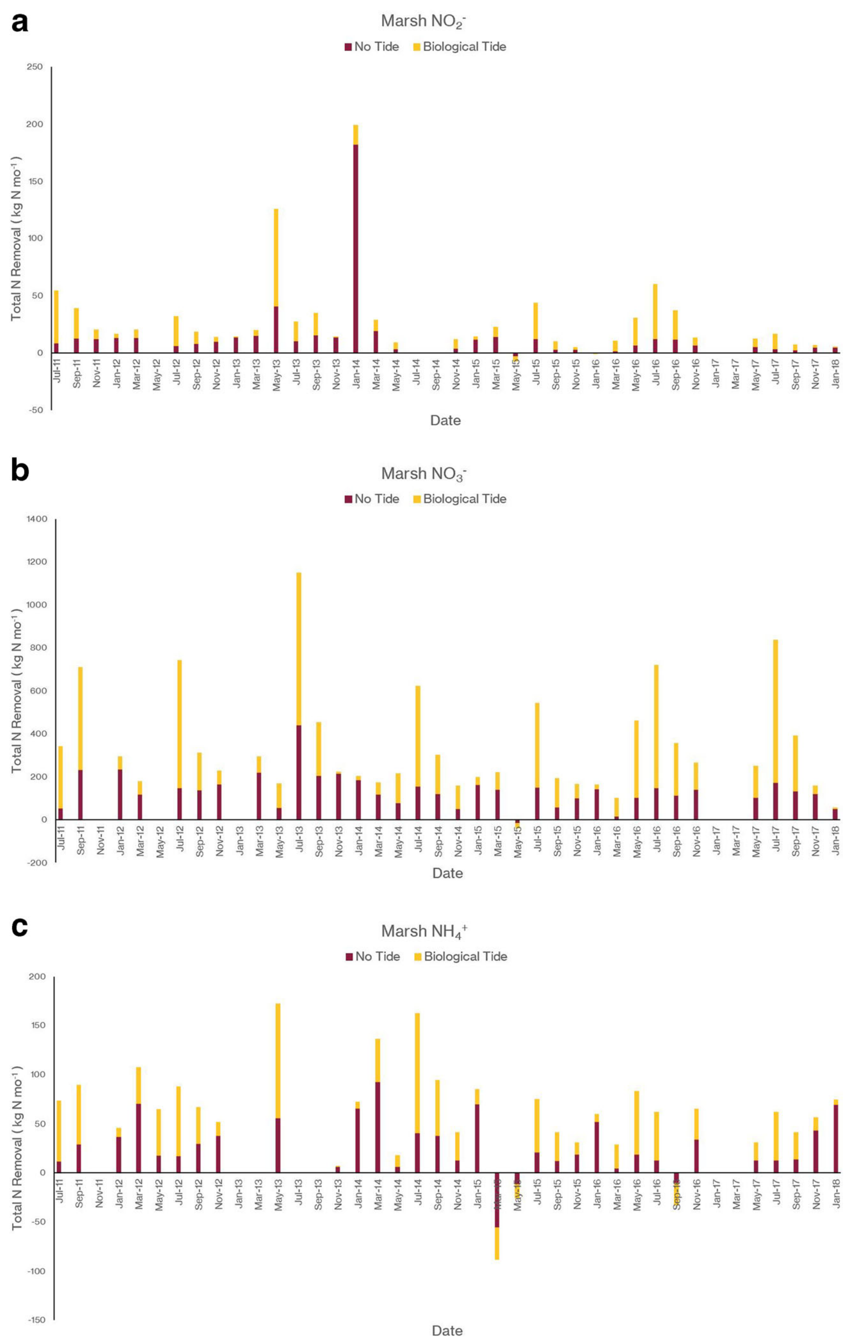

Fig. 8 The contribution of the biological tide to total nitrogen sequestration by the 21 ha vegetated marsh for (top) NO_2^-, (middle) NO_3^-, and (bottom) NH_4^+

Table 2 Comparison of seasonal differences in Areal Removal Rates (ARR) between the whole-system and marsh subsystem at the Tres Rios CTW, where Cool Season = October–February, Growing Season = March–September, and Year-round = all months

Analyte	Season	Whole-System ARR *42 ha* [g N m^{-2} mo^{-1}]	Marsh ARR *21 ha* [g N m^{-2} mo^{-1}]
$NO2^-$	Cool Season	- 0.41 ± 0.40	0.088 ± .014
	Year-round	0.16 ± 0.27	0.13 ± .030
	Growing Season	0.43 ± 0.22	0.15 ± .038
$NO3^-$	Cool Season	4.7 ± 3.6	0.89 ± 0.091
	Year-round	5.4 ± 2.6	1.6 ± 0.16
	Growing Season	5.6 ± 2.1	1.9 ± 0.18
$NH4^+$	Cool Season	3.6 ± 0.55	0.19 ± 0.015
	Year-round	3.3 ± 0.57	0.28 ± 0.076
	Growing Season	3.2 ± 0.58	0.32 ± 0.10

the Taiwan CTW among the seasons (Hsueh et al. 2014). Our second comparison was with a pilot-scale CTW near the Sannogawa River in Japan that was designed to remove N from agricultural runoff and domestic wastewater before it entered Lake Kasumigaura (Sheng and Hosomi 2008). They sampled this system from May through October over two years and reported a mean total N loading rate of 93.3 g m^{-2} and an average removal rate of 30.6 g m^{-2}, for a CRE of 32.8% (Sheng and Hosomi 2008). During the growing season at Tres Rios, by comparison, the CRE for NO_2^-, NO_3^-, and NH_4^+ were 15.7%, 17.3%, and 39.9% at the whole-system scale and 77.7%, 87.4%, and 52.8% within the marsh, respectively. Although the biological tide was strongest during hotter seasons, it had a positive effect on N removal by the marsh regardless of the time of year (Table 2).

The presence of the biological tide at Tres Rios increased year-round N uptake in both the entire 42 ha CTW and in the 21 ha marsh subsystem for all dissolved inorganic nitrogen species. Nitrogen removal rates at the 42 ha whole-system scale were enhanced because of the biological tide by 15.3–35.7% for NO_2^-, 2.82–12.1% for NO_3^-, and 0.984–2.37% for NH_4^+ (Table 3). More significantly, however, was the biological tide's effect on N removal by the 21 ha marsh, where we estimated the increases in N uptake efficiency to range from 28.9–126% for NO_2^-, 35.5–184% for NO_3^-, and 42.4–133% for NH_4^+ (Table 3). For all three constituents, the greatest increase in N uptake efficiencies occurred during the growing season, when plant productivity, transpiration, sunlight, and air temperatures were highest while relative humidity was very low. These results confirm that the biological tide is making the Tres Rios CTW more efficient at N uptake than its counterparts in cooler or more mesic climates.

Table 3 Enhanced N removal due to the biological tide (% improvement) for each dissolved inorganic nitrogen species across the seasons at the whole-system and marsh subsystem scales in the Tres Rios CTW, where Cool Season = October–February, Growing Season = March–September, and Year-round = all months

Analyte	Season	Whole-System ARR *42 ha* [%]	Marsh ARR *21 ha* [%]
$NO2^-$	Cool Season	15.3	28.9
	Year-round	35.7	96.3
	Growing Season	18.3	126
$NO3^-$	Cool Season	2.82	35.5
	Year-round	9.54	145
	Growing Season	12.1	184
$NH4^+$	Cool Season	.984	42.4
	Year-round	2.04	110
	Growing Season	2.37	133

Conclusions

The innovation of this study was our ability to quantify the contribution of the transpiration-driven biological tide to N uptake by the Tres Rios CTW. Our data confirmed that this unique phenomenon increased the amount of N entering the marsh where it was largely sequestered. Our water budget calculations allowed us to differentiate N in water overlying the marsh from N in water being advected into the marsh to replace transpirational losses. This differentiation allowed us to 1) estimate that total annual N removal by the 21 ha vegetated marsh was more than double with the biological tide compared to without it and 2) to demonstrate that this enhanced efficiency was greatest during the growing season. The biological tide also substantively contributed to N removal by the entire 42 ha CTW system in all seasons. This is an unintended, but clearly positive, consequence of building this CTW in a hot, dry climate.

Based on these results, there are many new avenues of research and considerations in design focused on taking full advantage of, and optimizing for, the biological tide's effect on N removal efficiency by CTWs. We recommend that CTW designers and managers carefully consider the types of vegetation to use in CTWs in order to optimize system performance. Specifically, we recommend selecting wetland plant species combinations that will maximize transpirational water losses, acknowledging the counter-intuitive trade-off that this recommendation has for water loss in aridland settings where water availability is often a significant challenge. A spatially-articulate hydrodynamic model that includes biogeochemical processes, plant productivity, and transpirational water losses could be used to optimize the design and spatial configuration of marsh versus open water in CTWs. Enhancing our knowledge base about how aridland CTWs function will further promote the use of these effective and efficient UEI treatment systems, allowing dry-climate cities to move towards more sustainable and resilient futures as they address complex challenges with limited water resources and growing populations.

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References

Akratos C, Tsihrintzis V (2007) Effect of temperature, HRT, vegetation and porous media on removal efficiency of pilot-scale horizontal subsurface flow constructed wetlands. Ecological Engineering 29(2):173–119. https://doi.org/10.1016/j.ecoleng.2006.06.013

Bachand PA, Horne AJ (1999) Denitrification in constructed free-water surface wetlands: II. Effects of vegetation and temperature. Ecological Engineering 14(1–2):17–32. https://doi.org/10.1016/S0925-8574(99)00017-8

Barbera AC, Cirelli GL, Cavallaro V, Di Silvestro I, Pacifici P, Castiglione V, Toscano A, Milani M (2009) Growth and biomass production of different plant species in two different constructed wetland systems in Sicily. Desalinization 246:129–136

Bois P, Childers DL, Corlouer T, Laurent J, Massicot A, Sanchez CA, Wanko A (2017) Confirming a plant-mediated "biological tide" in an aridland constructed treatment wetland. Ecosphere 8(3):1–16. https://doi.org/10.1002/ecs2.1756

Childers D, Sanchez C, Weller N (2018) Long-term monitoring and research of the ecology of the Tres Rios constructed treatment wetland, Phoenix, AZ, ongoing since 2011 ver 2. Environmental Data Initiative. https://doi.org/10.6073/pasta/3eb1f02c8db033f63a144a6f9d778fa7

Childers DL, Bois P, Hartnett HE, Mcphearson T, Metson GS, Sanchez CA (2019) Urban ecological infrastructure: an inclusive concept for the non-built urban environment. Elementa: Science of the Anthropocene 7(1):46. https://doi.org/10.1525/elementa.385

Childers DL, Cadenasso ML, Grove JM, Marshall V, McGrath B, Pickett STA (2015) An ecology for cities: a transformational nexus of design and ecology to advance climate change resilience and urban sustainability. Sustainability 7(4):3774–3791. https://doi.org/10.3390/su7043774

Childers DL, Pickett STA, Grove JM, Ogden L, Whitmer A (2014) Advancing urban sustainability theory and action: challenges and opportunities. Landscape and Urban Planning 125:320–328. https://doi.org/10.1016/j.landurbplan.2014.01.022

Eichelmann E, Hemesa KS, Knox SH, Oikawa PY, Chamberlain SD, Sturtevant C, Verfailliea J, Baldocchia DD (2018) The effect of land cover type and structure on evapotranspiration from agricultural and wetland sites in the Sacramento–San Joaquin River Delta, California. Agriculture and Forest Meteorology 256-257:179–195

Favero L, Mattiuzzo E, Franco D (2007) Practical results of a water budget estimation for a constructed wetland. Wetlands 27(2):230–239. https://doi.org/10.1672/0277-5212(2007)27[230:PROAWB]2.0.CO;2

Fonder N, Headley T (2010) Systematic classification, nomenclature, and reporting for constructed treatment wetlands. In: Vymazal J (ed) Water and nutrient Management in Natural and Constructed Wetlands. Springer, Dordrecht, pp 191–219

Fonder N, Headley T (2013) The taxonomy of treatment wetlands: a proposed classification and nomenclature system. Ecological Engineering 51:203–211. https://doi.org/10.1016/j.ecoleng.2012.12.011

Green M, Shaul N, Belaivski M, Sabbah I, Ghattas B, Tarre S (2006) Minimizing land requirement and evaporation in small wastewater treatment systems. Ecological Engineering 26:266–271

Greenway M (2005) The role of constructed wetlands in secondary effluent treatment and water reuse in subtropical and arid Australia. Ecological Engineering 25(5):501–509. https://doi.org/10.1016/j.ecoleng.2005.07.008

Grimm NB, Faeth SH, Golubiewski NE, Redman CL, Wu J, Bai X, Briggs JM (2008) Global change and the ecology of cities. Science 319(5864):756–760. https://doi.org/10.1126/science.1150195

Grimm NB, Fisher SG (1986) Nitrogen limitation potential of Arizona streams and rivers. Arizona-Nevada Academy of Science 21(1):31–44

Grove JM (2009) Cities: managing densely settled social-ecological systems. In: Folke C, Kofinas GP, Chapin FS III (eds) Principles of ecosystem stewardship: resilience-based natural resource management in a changing world. Springer, New York, pp 281–294

Hsueh M, Yang L, Hsieh L, Lin H (2014) Nitrogen removal along the treatment cells of a free-water surface constructed wetland in subtropical Taiwan. Ecological Engineering 73:579–587. https://doi.org/10.1016/j.ecoleng.2014.09.100

Kadlec RH (2008) The effects of wetland vegetation and morphology on nitrogen processing. Ecological Engineering 33(2):126–141. https://doi.org/10.1016/j.ecoleng.2008.02.012

Kadlec RH, Wallace S (2009) Treatment wetlands, 2nd edn. CRC Press, New York

Kadlec RH, Cuvellier C, Stober T (2010) Performance of the Columbia, Missouri, treatment wetland. Ecological Engineering 36(5):672–684. https://doi.org/10.1016/j.ecoleng.2009.12.009

Koch MS, Rawlik PS (1993) Transpiration and stomatal conductance of two wetland macrophytes (*Cladium jamaicense* and *Typha domingensis*). American Journal of Botony 80(10):1146–1154. https://doi.org/10.1002/j.1537-2197.1993.tb15346.x

Lee C, Fletcher T, Sun G (2009) Nitrogen removal in constructed wetland systems. Engineering in Life Sciences 9(1):11–22. https://doi.org/10.1002/elsc.200800049

Martin J, Hofherr E, Quigley MF (2003) Effects of *Typha latifolia* transpiration and harvesting on nitrate concentrations in surface water of wetland microcosms. Wetlands 23(4):835–844

Melosi MV (2000) The sanitary city: environmental services in urban America from colonial times to the present. Johns Hopkins University Press, Baltimore

Metson G, Hale R, Iwaniec D, Cook E, Corman J, Galletti C, Childers DL (2012) Phosphorus in Phoenix: a budget and spatial representation of phosphorus in an urban ecosystem. Ecological Applications 22(2):705–721. https://doi.org/10.1890/11-0865.1

Meuleman AFM, van Logtestigin R, Rijs GBJ, Verhoeven JTA (2003) Water and mass budgets of a vertical-flow constructed wetland used for wastewater treatment. Ecological Engineering 20(1):31–44. https://doi.org/10.1016/S0925-8574(03)00002-8

Nivala J, Headley T, Wallace S, Bernhard K, Brix H, van Afferden M, Müller RA (2013) Comparative analysis of constructed wetlands: the design and construction of the ecotechnology research facility in Langenreichenbach, Germany. Ecological Engineering 61:527–543. https://doi.org/10.1016/j.ecoleng.2013.01.035

Palmer H, Beutel M, Gebremariam S (2009) High rates of ammonia removal in experimental oxygen-activated nitrification wetland mesocosms. Journal of Environmental Engineering 135(10):972–979. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000053

Pedescoll A, Sidrach-Cardona R, Sánchez JC, Bécares E (2013) Evapotranspiration affecting redox conditions in horizontal constructed wetlands under Mediterranean climate: influence of plant species. Ecological Engineering 58:335–343. https://doi.org/10.1016/j.ecoleng.2013.07.007

Pires I, Da Costa G, Queluz J, Garcia M (2020) Effect of hydraulic retention time on chemical oxygen demand and total nitrogen removal in intermittently aerated constructed wetlands. Revista Ambiente & Água 15(3):1–11. https://doi.org/10.4136/ambi-agua.2504

Sanchez CA, Childers DL, Turnbull L, Upham R, Weller NA (2016) Aridland constructed treatment wetlands II: plant mediation of surface hydrology enhances nitrogen removal. Ecological Engineering 97:658–665. https://doi.org/10.1016/j.ecoleng.2016.01.002

Sheng Z, Hosomi M (2008) Nitrogen transformations and balance in a constructed wetland for nutrient-polluted river water treatment using forage rice in Japan. Ecological Engineering 32(2):147–155. https://doi.org/10.1016/j.ecoleng.2007.10.004

Steidl J, Kalettka T, Bauwe A (2019) Nitrogen retention efficiency of a surface-flow constructed wetland receiving tile drainage water: a case study from North-Eastern Germany. Agriculture, Ecosystems and Environment 283:106577. https://doi.org/10.1016/j.agee.2019.106577

Tanner CC, Sukias JPS, Headley TR, Yates CR, Stott R (2012) Constructed wetlands and denitrifying bioreactors on-site and decentralized water treatment: comparison of five alternative configurations. Ecological Engineering 42:112–123. https://doi.org/10.1016/j.ecoleng.2012.01.022

Tao W, Sauba K, Fattah KP, Smith JR (2017) Designing constructed wetlands for reclamation of pretreated wastewater and stormwater. Review of Environmental Science & Technology 16:37–57

Tuttulomondo T, Leto C, La Bella S, Leone R, Virga G, Licata M (2016) Water balance and pollutant removal efficiency when considering evapotranspiration in a pilot-scale horizontal subsurface flow constructed wetland in Western Sicily (Italy). Ecological Engineering 87:295–304

Wallace SD, Knight RL (2006) Small-scale constructed wetland treatment systems: feasibility, design criteria, and O&M requirements. Water Environment Research Foundation (WERF), Alexandria

Weller NA, Childers DL, Turnbull L, Upham RF (2016) Aridland constructed treatment wetlands I: macrophyte productivity, community composition, and nitrogen uptake. Ecological Engineering 97:649–657. https://doi.org/10.1016/j.ecoleng.2015.05.044

Wu H, Zhang J, Wei R, Liang S, Li C, Xie H (2013) Nitrogen transformations and balance in constructed wetlands for slightly polluted river water treatment using different macrophytes. Environmental Science and Pollution Research 20(1):443–451. https://doi.org/10.1007/s11356-012-0996-8

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