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Long term water quality improvements associated with the restored Prairie Creek wetlands in Ohio's Grand Lake St. Marys Watershed

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Wetland restoration has emerged in recent years as an essential strategy for improving water quality to mitigate harmful algal blooms and the associated decline in water quality, especially across the Midwest. As a consequence, there is a need for more long-term monitoring datasets to better understand the nutrient and sediment processing potential of these systems over time. In this study, surface water samples were collected and analyzed, and water volumes were tracked weekly from the inlet and outlet of a pump driven, flow through wetland along Prairie Creek in the Grand Lake St. Marys watershed over an 8-year period (2017–2024) in order to estimate yearly and seasonal nutrient load reductions. During this time, the wetland processed 4.44 million m^3, roughly 5.7%, of the annual Prairie Creek flows, wherein it showed promising overall concentration reductions between stream and wetland for TP (56%), SRP (81%), NO_3–N (60%), and TSS (10%). A Bayesian nonlinear model was used to describe within dataset variation among seasons and years highlighting the kind of ranges these systems can exhibit in water quality improvements. The results from this study show significant nutrient and sediment load reductions can be achieved using restored wetlands as a mitigation tool. Furthermore, these data contribute to our understanding of long-term efficiency, within year seasonal changes, as well as how management strategies can help restored wetlands realize their potential as tools to improve water quality.

Keywords Wetland restoration, Nutrient load reductions, Bayesian wetland modeling, Watershed conservation, Eutrophication

Wetlands were once a major component of the natural habitat throughout the continental United States, with precolonial (~ mid to late 1700s) estimates equating to approximately 11% of land area coverage[1]. These habitats have largely been lost (over 50%) over the past few centuries, primarily as a result of drainage for agriculture and urbanization. Losses are most striking in states like Ohio where 90% of the historical wetland area is gone, reducing the total wetland area to 1.8% by 1980[1]. While wetland acreage continues to be lost across the United States, albeit at less total area compared with historical numbers[2], there are some watersheds where this trend has started to reverse as a result of water quality initiatives. Wetland ecosystems play a vital role in nutrient reduction through a series of physical, chemical, and biological processes, while also offering a myriad of other benefits such as public recreational use, animal/plant habitat, groundwater recharge points, and storm/flood storage areas[3]. As a result of these increasingly valuable ecosystem services, wetland restoration projects are on the rise, especially in the Midwest, where eutrophication and resulting harmful algal blooms (HABs) have been increasing over the past few decades resulting in annual damage estimates in the billions of dollars as a consequence of diminished property values, decreased recreation opportunities (e.g., fishing revenue, fewer beach goers, etc.), and increased municipal water treatment costs[4].

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Wetland restoration efforts for the explicit purpose of improving ecosystem health date globally to the late 1800s with some of the original intents including reforestation and improving waterfowl habitat. In the United States, since the Clean Water Act was enacted in 1972, restored wetlands have been incentivized and even mandated to protect the nation's aquatic resources and improve water quality[5]. Wetlands have been restored using a variety of approaches ranging from simple reflooding of previously drained or separated areas originally classified as wetlands, enhancing wetland features in existing systems, to engineering new systems with particular features ranging in complexity to provide services such as surface water quality improvements, biodiversity enhancements, wastewater treatment, and flood water retention[6,7]. In the Grand Lake St. Marys Watershed (GLSM; western Ohio), the focus of this study, wetland restoration has become especially important since the passing of the Distressed Watershed Ruling in 2011[8]. With over 400 ha of total wetland area restored (and counting) throughout the watershed, these systems have become a vital tool for reducing excess nutrient runoff in its tributaries before loading into the titular lake[9]. Historical land area estimates suggest that around 1/3 of the entire GLSM watershed was once wetland habitat, however, this number fell by the early 2000s to less than 0.5%[10]. Recent restorations provide a level of gains in GLSM that should be built upon. On a statewide scale, wetland restoration efforts can be best seen through the lens of the recent H_2Ohio Program, a statewide water quality initiative started in 2019 to reduce nutrient runoff that has (among other things) helped facilitate the creation, enhancement, and/or mitigation of over 200 wetland projects statewide to date[11].

Wetlands improve surface water quality thus reducing eutrophication of water bodies by retention of nitrogen, phosphorus, and sediment through a combination of mechanisms ranging from sorption into soil, microbial cycling, plant uptake, and simple settling[12]. Recent meta-analyses by Land et al.[13] indicate median total phosphorus (TP) and nitrogen (TN) concentration reductions of 46% TP (CI 37–55%) and 37% TN (CI 29–44%) associated with wetland treated surface waters. These numbers are similar to those calculated by Ury et al.[14], Chavan et al.[15], and Spieles & Mitsch[16]. However, despite the research that has gone into restored wetlands and water quality, there is still a need for additional long-term monitoring datasets that include both consistent nutrients and hydrology data. Most published studies are comparatively short (1 or 2 years) and/or exhibit challenging hydrology (diffuse or difficult to measure inflows and outflows), precluding the specific kind of dataset building needed to track long-term source or sink dynamics necessary for scaling up and managing these practices as part of nutrient reduction strategies.

To evaluate functionality of a wetland for nutrient and sediment processing, long-term datasets with accurate hydrology information (e.g. inflow and outflow rates/totals) and high resolution nutrient data from both source waters and wetland areas spanning various weather conditions as well as natural habitat succession are needed. Short-term studies or studies from sites with complex hydrology (as is the case with most wetlands), do not easily allow for 'practical' projection estimates of wetland nutrient processing (i.e. retention) function. The reasons here are largely a result of varied weather patterns representing extreme dry/wet or 'normal' years, gradual succession patterns inherent with vegetation regime changes, eventual buildup of sediment over time, or diffuse and difficult to assess inflow/outflow points that can be hard to trace back to nutrient processing that are challenging to represent in shorter-term datasets[17]. As an additional layer of complexity, surface water data, particularly from systems like wetlands, even from the simplest wetland with the most regular and reliable dataset, are often challenging to analyze statistically. This is a function of the high temporal and spatial variation present in water samples as nutrient/sediment processing within a given system can change concentrations quickly and lead to a prevalence of non-normal data distributions oftentimes driven by extreme highs and below detection limit values. Moreover, the data themselves are often correlated leading to problems with multicollinearity among variables as well as autocorrelation in time and space, potentially masking cause and effect, making emergent trends difficult to detect[18–20]. Reducing complexity in hydrology and nutrient data as a consequence of site selection, site plan, and study design is what makes the wetland of this study a good candidate for long-term analysis as it is a relatively simple input-out model with nearly a decade of high resolution hydrologic and nutrient data for wetlands that have experienced a variety of weather and natural processes including vegetation changes and sediment buildup over time.

Objectives

The objectives of this study were to describe the long-term surface water nutrient and sediment reduction efficacy of a restored wetland in the Grand Lake St. Marys Watershed. Moreover, this study sought to characterize these load reductions in the context of overarching nutrient mitigation strategies for the watershed.

Materials and methods

Grand Lake St. Marys Watershed

Grand Lake St. Marys is a freshwater lake located in western Ohio (Fig. 1) that serves as a recreational area, wildlife habitat, and public drinking water source for the city of Celina (~ 10,885 people as of 2023). The 5,200-ha lake was originally built 1837–1845 as one of several feeder reservoirs to maintain water levels in the Miami & Erie Canal. The Grand Lake St. Marys Watershed, which drains to the lake, spans ~ 23,795 ha of primarily agricultural row crop fields with a few small urban areas (e.g. City of Celina; Fig. 1). The watershed includes eight main tributaries/subwatersheds varying in size: North Shore, Coldwater Creek, Grassy Creek, Monroe Creek, Beaver Creek, Prairie Creek, Big Chickasaw Creek, and Little Chickasaw Creek. Historically, the tributaries and lake were among the highest in nutrient concentrations as well as cyanobacterial algal biomass/toxin levels when compared to other watersheds in nationwide surveys[21,22]. For example, in Water Year 2009, surface water TP concentrations in Chickasaw Creek exceeded OEPA TMDL recommendations of 0.1 mg P/L on 80% of the sampling days, NO_3–N concentrations of 1.0 mg N/L 70% of the sampling days, and in-lake microcystin concentrations of 1 ppb 100% of the sampling days[23]. These findings led directly to the 2011 Distressed Watershed Ruling and a series of conservation programs and initiatives in the watershed, including wetland restoration

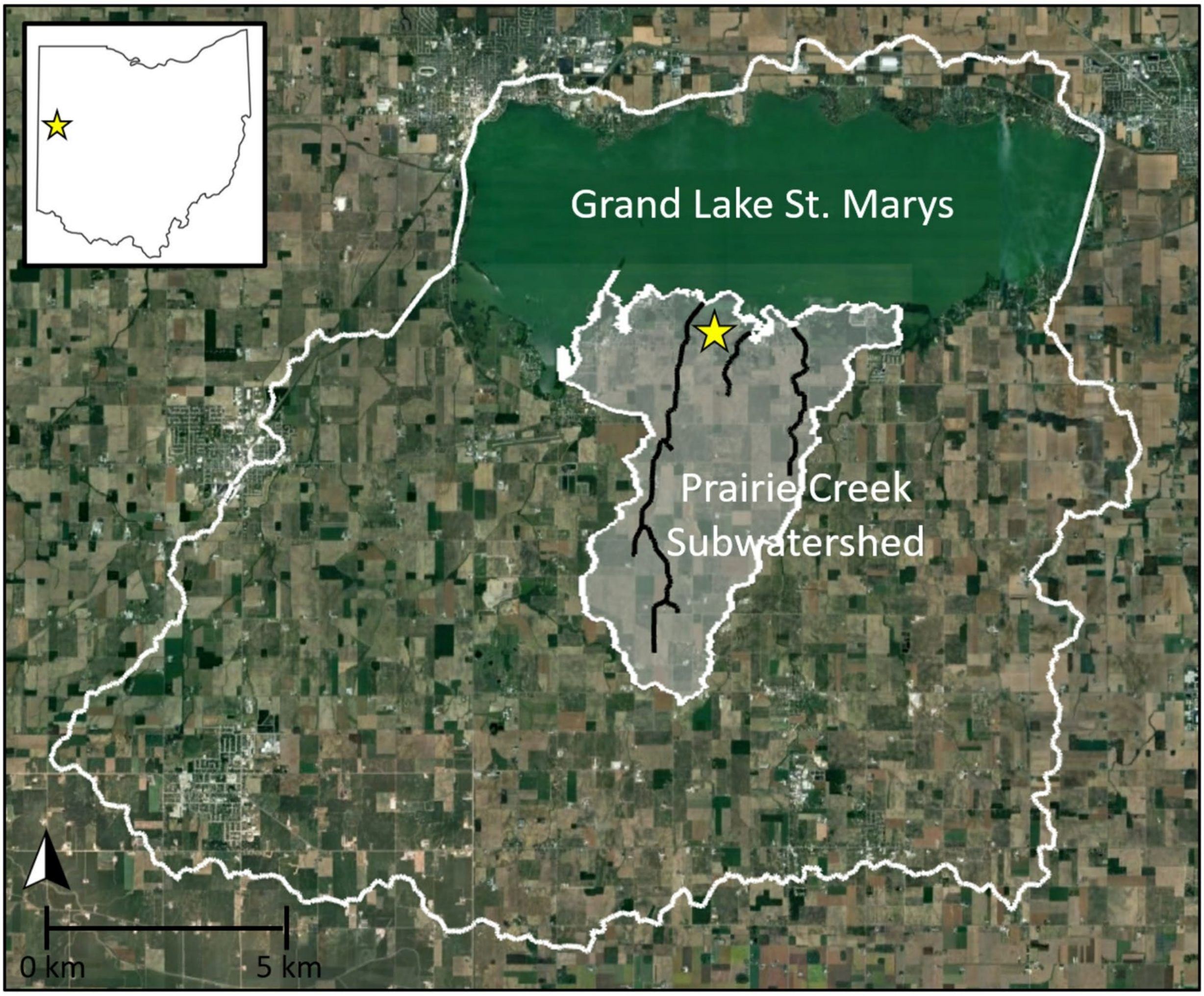

Fig. 1. Watershed map of the study area where the GLSM watershed is outlined in white, the Prairie Creek subwatershed is outlined and shaded in white, the Prairie Creek tributaries are outlined in black, and the Prairie Creek wetlands are designated by a star. Satellite imagery from Google Earth Pro[24].

projects[8,9]. The focus of this paper is on Prairie Creek restored wetland, the first of many restored native habitats in the watershed.

Site description

The Prairie Creek Watershed drains 3,106 ha via a network of small feeder ditches, creeks, surface swales, and field tiles that flow into Prairie Creek itself, which spans about five stream miles with three other small first-order tributaries (ditches) flowing into the lake (Fig. 1). The Prairie Creek Wetlands are situated south of Grand Lake St. Marys on previously drained wetland area that had been farmed (corn/soybean row crop) for nearly a century until restoration.

Restored in two phases, with the West side occurring in 2012 and the East side occurring in 2014, the Prairie Creek Wetlands are made up of two separate 'treatment trains', Prairie Creek East and Prairie Creek West (Fig. 2). In total, the entire site is ~ 32 ha with about 8 ha of this area classified as wetland habitat (remainder is restored prairie). Land for these wetlands was purchased in two phases, with the West side purchased in 2010 for $576,100 (16.1 ha) and the East side in 2012 for $880,000 (16.2 ha), through funding provided by the Clean Ohio Green Space Conservation Program. Funding from the EPA 319 Program (~$726,000 between 2010 and 2013) administered under the Clean Water Act in addition to ~$689,000 from local donations was used to fund design, construction, and restoration of the site. After completion of construction, both wetlands were drill seeded with a mix of emergent wetland plants (Cardno JFNew, cardnonativeplantnursery.com) consisting of native sedges, rushes, grasses, and forbs.

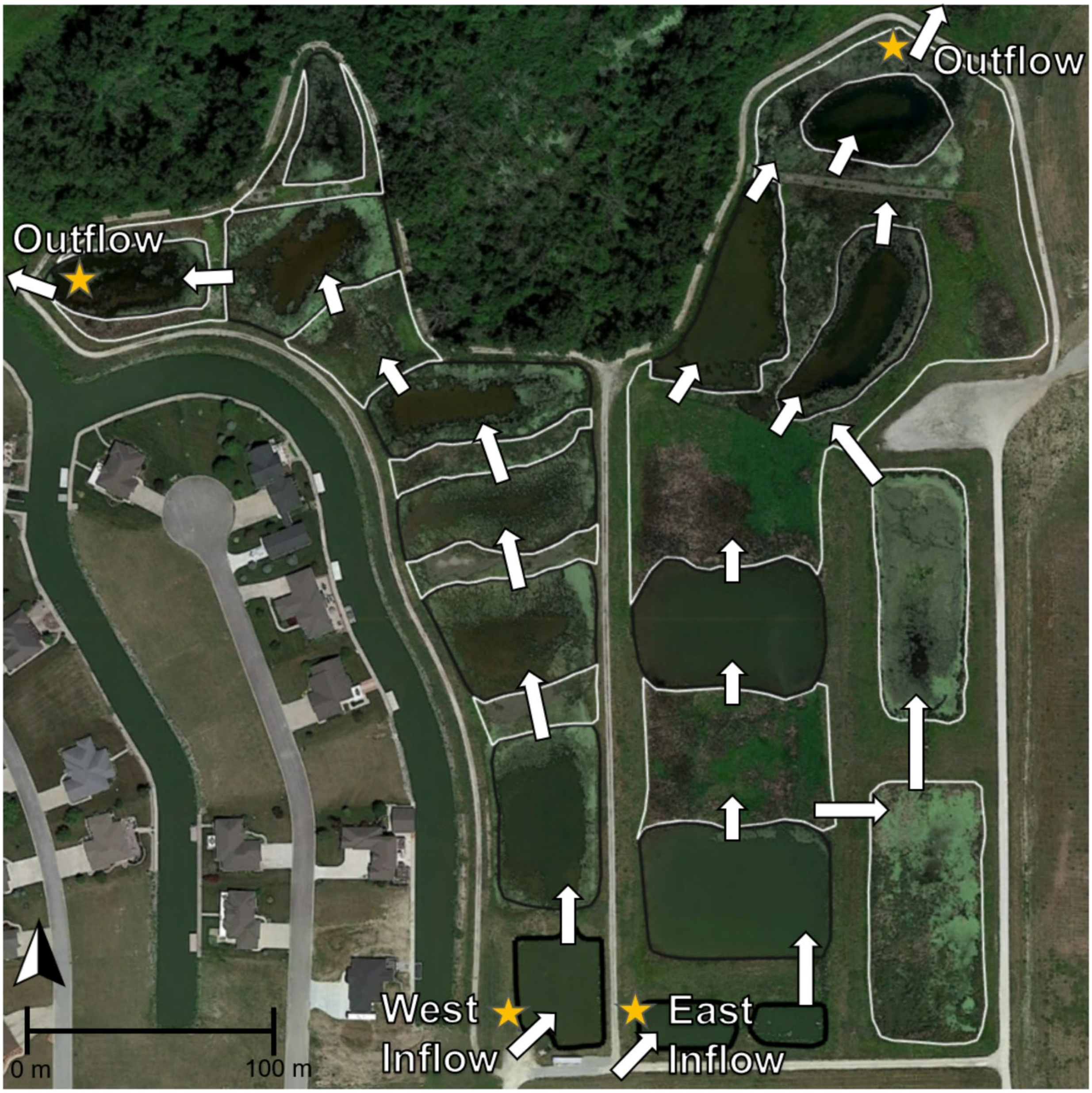

Fig. 2. Map outlining the flow path of the Prairie Creek Wetland where flow direction is shown by arrows and stars designate surface water sampling locations. Heavy black outlines indicate settling pool, light gray outlines indicate shallow marsh, and white outlines indicate vegetated flat areas. Satellite imagery from Google Earth Pro[25].

Hydrology

Water from Prairie Creek enters both wetlands from a 30HP Gorman Rupp pump connected to Prairie Creek. When water is not actively pumped, the stream continues to flow unabated into Grand Lake St. Marys. Water that is pumped into the wetland, first flows through a respective series of 'deep' water settling pools (~ 1.2–1.8 m) followed by shallow pools (~ 0.5 m) and vegetated marsh flats (~ 0.2 m) until exiting the site (Fig. 2). Given site elevations and location, water can only enter the site by pumped inflow (e.g., not from overland drainage or stream overflows). Water volume in both wetlands is controlled by a water level control structure (WLCS, Agri Drain) on the outflow pools. Water level control structures are structures that sit at the outflow path and provide the end user the ability to raise or lower the water level needed to flow through them using a combination of plastic 'boards or logs', effectively providing a level of control over the total volume of the system. Prairie Creek East holds a total volume of 9,947 m^3 with a 17.8-cm stoplog and 13,888 m^3 with 35.6-cm of stoplogs set at the outflow WLCS. Prairie Creek West holds a total volume of 8,967 m^3 with a 17.8-cm stoplog and 13,285 m^3 with 35.6-cm of stoplogs at the outflow WLCS. Using an average inflow rate for PC East and West combined ~ 1,874 m^3/day each assuming the typical 17.8-cm stoplog in place at each WLCS, average residence times are 5.3 and

4.8 days, respectively. Once water exits the wetland it outflows through a shallow vegetated marsh flat and into a small forested wetland before flowing into Grand Lake St. Marys.

Stream flows in Prairie Creek were tracked using long-term monitoring data from a nearby USGS station on Chickasaw Creek (#402913084285400) selected as a reference due to being in the GLSM watershed, close proximity, nearly identical land use and topography, and similar drainage area. Flows from the Chickasaw Creek USGS Station, drainage area equivalent to 4,823 ha, were multiplied by 0.644 to calculate stream flows for Prairie Creek at the wetland inflow, which has a drainage area equivalent to 3,106 ac, and then averaged to determine a daily stream flow. Wetland inflow volumes were tracked for both the East and West wetlands by using a depth over flat weir equation:

$$Q_{in} = \left(3.247 \times\ L \times\ H^{1.48} - \frac{0.566L^{1.9}}{1+2L^{1.87}}H^{1.9}\right) * 448.8$$

where Q_{in} is discharge in gallons per minute (GPM), L is the length of the flat weir in feet, and H is the depth of water over the flat weir in feet. Note that water inflows into each side of the wetland through a small concrete basin that spills over a level grade flat weir and into the respective settling pools. Using inflow and outflow nutrient and sediment concentrations, inflow volumes were adjusted weekly to obtain the highest pump rate while still removing nutrients and sediment. Outflow volume in the WLCS was measured similarly to inflow using a depth over flat weir equation:

$$Q_{out} = 3.19924 \times\ (W - 0.74H)H^{1.48}$$

where Q_{out} is discharge in GPM, W is width of the top weir in inches, and H is depth of water over the flat weir in inches (www.agridrain.com). Depths of water over the inflow and outflow weirs are taken weekly via point measurements and assumed to be constant over the week.

Management timeline

Since construction, routine management of the site has included annual pump maintenance and activity to promote active water movement, invasive plant species (e.g. *Phragmites*) spot spraying when needed, rotenone treatments ~ every 3 years to reduce Common Carp (*Cyprinus carpio*) numbers which have been linked to extensive vegetation damage[26], as well as public walking path upkeep around the site. In addition to these kinds of routine maintenance, both settling pools were dug out late 2021 to reset capacity for the start of the spring 2022 flow season.

Water quality analyses

Starting in June 2017 (dataset analyzed herein spans 2017–2024), surface water samples were collected from the inflow (Prairie Creek) and outflow (Wetland) in clean 1-L Nalgene bottles by overflowing and capping after rinsing with sample water. Note that water samples were collected on a weekly basis year round from the inflow source (Prairie Creek) and on a weekly basis from the outflow (Wetland) when water was actively flowing (pumped) through the system. Water at this site is pumped (i.e. flows) typically beginning in early/mid Spring and ending in mid Fall. Herein, seasons are defined as Spring (March 15–June 14), Summer (June 15–September 14), Fall (September 15–December 14), and Winter (December 15–March 14). Seasons were defined in this way to align with the agricultural rules and typical activity calendar coinciding with pre plant ground preparation, crop growing cycle, harvest time, and winter nutrient application ban periods followed in the watershed[8]. Upon water collection, samples were stored in a cooler on ice while in the field. Upon arrival back to the lab, a portion of the raw sample was filtered using a 0.45-µm nylon luer-lock filter. Filtered water samples were analyzed for soluble reactive phosphorus (SRP) using an ascorbic acid method (HACH TNT 843). Raw water samples were analyzed for total phosphorus (TP) using an ascorbic acid method, nitrate–N (NO_3–N) using a dimethylphenol method (HACH TNT 835), and total suspended solids (TSS) using a gravimetric method (Standards Method 2540). For TSS, 100 mL of raw water was filtered using 1.5-µm glass fiber filters. Minimum detection limits for analyses were 0.01 (phosphorus tests as P), 0.1 (nitrogen tests as N), and 0.5 (sediment tests) mg/L. All nutrient samples were analyzed using a HACH DR3900 Spectrophotometer and TSS filters were weighed using an analytical balance. Field duplicates and analytical blanks were processed alongside all weekly samples for QA/QC.

Data analysis

Nutrient loading was calculated as a function of inflow or outflow volume multiplied by nutrient concentrations. Given the number of water samples we were able to feasibly obtain over the course of the study we calculated wetland load on a weekly basis by multiplying the single nutrient or sediment concentration value measured in a given week by the total measured flows through the wetland for that week. These weekly loads, calculated by applying the weekly nutrient/sediment concentration measurement to flow volumes, were then summed and described by season and year. To estimate effect sizes, log response ratios (ln R) were calculated as a function of the natural log of load out divided by load in. This statistic (ln R) takes into account that while concentration reductions between inflow and outflow are important and certainly contribute to load reductions, they also factor in volumes of water that can differ between inflow and outflow. This statistic also facilitates a high-level comparison among nutrient species and sediment as to which is best being removed by the wetland.

Nutrient concentration reduction was estimated using percent decrease from input nutrient concentrations, C_{in}, and output nutrient concentrations C_{out}. Input and output concentrations below detectable limits were set to half of the detection limits.

$$\%decrease = \frac{C_{in} - C_{out}}{C_{in}}$$

Annual variation in wetland efficiency was modeled using a nonlinear model[27] that describes the input-output relationship. This model describes the overall efficiency of the constructed wetland by comparing the concentration of nutrients in the output and the concentration of the nutrients in the input.

$$C_{out} = \alpha \, C_{in}^{\beta} q^{\gamma}$$

where C_{out} is the output concentration, C_{in} is the input concentration, q is the hydraulic loading rate (m/d), and α, β and γ are regression coefficients. The hydraulic loading rate is calculated as

$$q = \frac{Q_{in}}{A}$$

where Q_{in} is the inflow (m^3/d) and A is the wetland surface area (m^2). The nonlinear regression predicting C_{out} was transformed by taking the natural log of both sides to linearize the equation. This step was needed to improve model convergence given the presence of values below the detectable limit (see below). Additionally, α and β were modeled with a random effect for year to capture variability in reduction efficiency across years. The new equation was

$$ln\left(C_{out,\,i}\right) = ln\left(\alpha_{\,j[i]}\right) + \beta_{\,j[i]} ln\left(C_{in,\,i}\right) + \gamma \, ln\left(q\right) + \epsilon_i$$

where $\alpha_{\,j[i]}$ is the intercept for year *j*; $\beta_{\,j[i]}$ is the slope for year j; and ϵ_i is random normal error with mean = 0 and standard deviation = $\sigma_{\,C}$. Random effects for each coefficient were:

$$\alpha_{\,j} \sim Normal\left(\mu_{\,\alpha}, \sigma_{\,\alpha}\right)$$
$$\beta_{\,j} \sim Normal\left(\mu_{\,\beta}, \sigma_{\,\beta}\right)$$

where $\mu_{\,\alpha}$ is the overall mean intercept; $\mu_{\,\beta}$ is the overall mean slope; and $\sigma_{\,\alpha}$ and $\sigma_{\,\beta}$ are standard deviation for the intercept and slope.

The data set contains input and output concentrations below the method detection limit. Ignoring observations below the detection limit can result in overestimations of the mean and underestimate the standard deviation[28]. Common substitution methods such as setting the value to the detection limit or somewhere between zero and the detection limit also produce bias in the mean and standard deviation[28]. Below method detection limits were imputed to reduce bias on regression coefficients (Table 1). Input concentrations that were below method detection limits were estimated using a censored normal distribution based on the mean and standard deviation of the overall concentration observed[29]. The censored location was set to the method detection limit of each nutrient. Output concentrations that were below method detection limits were automatically imputed within the

Nutrient	***N***	N_{bl}	$\%_{bl}$
Prairie Creek East			
NO_3–N - Input	210	1	0.5
NO_3–N - Output	210	0	0
TP - Input	210	0	0
TP - Output	210	2	1.0
SRP - Input	210	33	15.7
SRP - Output	210	165	78.6
TSS - Input	210	0	0
TSS - Output	210	0	0
Prairie Creek West			
NO_3–N - Input	195	1	0.5
NO_3–N - Output	195	0	0
TP - Input	195	0	0
TP - Output	195	2	1.0
SRP - Input	195	43	22.1
SRP - Output	195	115	59.0
TSS - Input	195	0	0
TSS - Output	195	0	0

Table 1. Total number of observations (N) taken on a weekly basis over the 8 year study period when the wetland was actively flowing, number below detection limit (N_{bl}), and percentage of observations below detection limits ($\%_{bl}$) for the Prairie Creek East and Prairie Creek West wetlands.

model based on the set of predictors (e.g., C_{in} and q) during the observation that was below method detection limits and coefficients[30].

Noninformative prior probabilities were used for all model parameters. The overall mean intercept (μ_{α}) and slope (μ_{β}) were given normally distributed prior probability distributions with mean = 0 and standard deviation = 2. Standard deviation for the year random effects for the intercept (σ_{α}) and slope (σ_{β}) were given half-Cauchy prior probability distribution with mean = 0 and standard deviation = 2. The standard deviation of the normally distributed error term was given a half-Cauchy prior probability distribution with a mean = 0 and standard deviation = 2. Three concurrent Markov Chain Monte Carlo (MCMC) chains were used for each model. Each chain consisted of 3000 iterations, with the first 1500 iterations discarded to remove the burn-in steps. The full posterior probability distribution consisted of 4500 total iterations. Traceplots and split R-hat for each estimated parameter were used to assess convergence. Convergence was achieved when no trend was observed in the traceplots, and the split R-hat was less than 1.1 for all estimated parameters. All models were fit in R 4.5.0[31] using Stan[32] and the rStan package[33].

Model predicted outflow concentrations were plotted as a function of inflow concentrations to visualize nutrient reduction efficiency. Lines that are higher indicate lower removal efficiency (i.e., for a given inflow concentration, a higher outflow concentration indicates lower efficiency). The slope of the line represents how quickly output concentrations are expected to increase as inflow concentration increases. A slope close to 0 would indicate inflow concentrations are not related to outflow concentrations and thus the interpretation would be the wetland is able to remove nearly all the nutrients that are coming in.

The input-output model used in this study is simpler than other wetland models (e.g., Artificial Neural Networks[34]). However, the constructed wetland here is designed such that it has a single input and a single output that is consistently monitored. Thus, simplifying the mechanistic relationships needed to be modeled. Yet, complicating factors outside of wetland design can still be handled relatively easily by fitting the model using Bayesian methods. For example, missing data is common due to random equipment failure (i.e., missing at random) or missing due to values falling below detection limits. Both of these scenarios are easily handled when they occur in the input or output datasets. Thus, our general approach here incorporates all sources of uncertainty and produces a full time series regardless of if the measured quantity was below detection limit or missing at random. Further, the output produced incorporates all known uncertainty in the data and results in posterior distributions of parameters that can easily be applied for scenario planning (e.g., extreme high loads from certain inflow events) both at this site or beyond in similar wetlands.

Results and discussion

Between 2017 and 2024, Prairie Creek East and West wetlands reduced nutrient loads by 204, 574, 7,096, and 42,011 kg compared to 212, 527, 7,195, and 34,792 kg of SRP, TP, NO_3–N, and TSS, respectively (Tables 2 and 3). Hydrologically, these two sides (East and West) captured ~ 2.3 million m^3 and ~ 2.1 million m^3, respectively, of the total ~ 77 million m^3 that flowed through Prairie Creek over this time, equating to ~ 5.7% of the annual stream flows (Table 4; Fig. 3). The greatest single year volume treatments came in 2019 for Prairie Creek East (470,780 m^3) and 2017 for Prairie Creek West (566,630 m^3). The greatest single year percentage of stream treated was in 2023 at 8.4% for East and 5.1% for West, partially due to lower flows in Prairie Creek.

PC East

The average annual percent concentration reduction between inflow and outflow was 60.1%, 64.0%, 82.0%, and 18.4% for NO_3–N, TP, SRP, and TSS, respectively (Table 2; Fig. 4). Year to year variation was present in the dataset with concentration reduction efficiencies (%) ranging from 47.0 (2020) to 71.0 (2019) for NO_3–N, 32.0 (2021) to 86.9 (2017) for TP, 12.0 (2023) to 93.3 (2024) for SRP, and −116 (2021) to 77.3 (2022) for TSS. Within year variation was also present with average seasonal concentration (note that winter is not included below due to no pumping/water flow during this season) reduction efficiencies (%) across time ranging from 54.0 (Spring) to 78.4 (Fall) for NO_3–N, 56.5 (Spring) to 71.1 (Fall) for TP, 65.1 (Fall) to 89.8 (Summer) for SRP, and 5.6 (Summer) to 33.6 (Fall) for TSS. Overall, log response ratios (ln R) showed that all nutrient species as well as sediment were negative, indicating that a greater inflow load compared to outflow was occurring. Annual cross comparison of log response ratios found that PC East was most effective at reducing SRP loads followed by TP and NO_3–N with TSS being the least in overall reductions (Table 2).

All nutrient statistical models for PC East converged with split R-hat being less than 1.10 for all parameters. The number of observations below detection limits were too high to fit the model of SRP and thus only observed data without observations below detection limits were used. Models for other nutrients included imputed values below the detection limit. Removal efficiency is described by vertical location of lines on Figs. 5 and 6. Lines that are more positive on the y-axis indicate higher output for a given input and thus, lower removal efficiency. The slope suggests whether removal is related to input concentrations, thus a slope of 0 suggests the wetland is removing additional amounts of nutrients as the input increases. The removal efficiency of NO_3–N and SRP was consistent throughout the years (Fig. 5A and B). Removal efficiency of TP was variable across years but the output concentrations were generally below −1.6 on the natural log scale (0.208 mg/l) for all inflow concentrations (Fig. 5C). Output concentrations were highest in 2020 and 2021 and lowest in 2017. Removal efficiency of TP was highest in 2017 and lowest in 2020 and 2021. The slope for all years was small, with 2021 being near 0, indicating the wetland is removing the maximum amount of TP that it is capable of and increased concentrations of TP are not being removed. Similarly, there was significant variability in removal efficiency across years but no strong relationship between inflow concentrations of TSS and outflow concentrations of TSS (Fig. 5D). TSS removal efficiency was lowest in 2021 and highest in 2022.

Season	Parameter	SRP (PO_4^{3-}–P)	TP (PO_4^{3-}–P)	NO_3–N	TSS
Spring	Average stream inflow concentration (mg/L)	0.069	0.274	8.524	41.7
	Average outflow concentration (mg/L)	0.016	0.119	3.924	31.9
	Concentration reduction (%)	77.5	56.5	54.0	23.6
	Total load in (kg)	7.99	214	6,987	32,774
	Total load out (kg)	1.18	66.2	2,451	17,116
	Load reduction (%)	85.2	69.1	64.9	47.8
	Mass removed (kg)	48	148	4,536	15,658
Summer	Average stream inflow concentration (mg/L)	0.111	0.323	2.097	31.6
	Average outflow concentration (mg/L)	0.011	0.112	0.779	29.9
	Concentration reduction (%)	89.8	65.2	62.1	5.6
	Total load in (kg)	141	419	2,695	41,916
	Total load out (kg)	10.6	108.8	808	24,491
	Load reduction (%)	92.5	74.0	70.0	41.6
	Mass removed (kg)	130	310	1,887	17,425
Fall	Average stream inflow concentration (mg/L)	0.089	0.331	3.490	37.7
	Average outflow concentration (mg/L)	0.031	0.096	0.755	25.0
	Concentration reduction (%)	65.1	71.1	78.4	33.6
	Total load in (kg)	34.3	148	885	17,097
	Total load out (kg)	8.31	32.6	213	8,168
	Load reduction (%)	75.8	78.0	75.9	52.2
	Mass removed (kg)	26	116	672	8,929
Winter	Average stream concentration (mg/L)	0.096	0.270	8.806	32.8
	Total load in (kg)	0	0	0	0
	Total load out (kg)	0	0	0	0
Annual	Average stream inflow concentration (mg/L)	0.093	0.309	4.433	36.1
	Average outflow concentration (mg/L)	0.017	0.111	1.767	29.5
	Concentration reduction (%)	82.0	64.0	60.1	18.4
	Loading rate ($g/m^2/yr$)	9.1	30.7	414	3,600
	Total load in (kg)	231	781	10,567	91,787
	Total load out (kg)	27	208	3,472	49,775
	Load reduction (%)	88.2	73.4	67.1	45.8
	Mass removed (kg)	204	574	7,096	42,011
	Mass removed ($g/m^2/yr$)	1.00	2.81	34.8	206
	Ln (R) ~ ratio of load out/load in	−2.14	−1.33	−1.11	−0.61

Table 2. Surface water nutrient and sediment data seasonally for Prairie Creek East from 2017 to 2024.

PC West

The average annual percent concentration reduction between inflow and outflow for PC West was 58.9%, 46.3%, 80.1%, and 1.2% for NO_3–N, TP, SRP, and TSS, respectively (Table 3; Fig. 4). Year to year variation was present in the dataset with concentration reduction efficiencies (%) ranging from 32.9 (2020) to 86.1 (2024) for NO_3–N, 25.7 (2021) to 66.5 (2017) for TP, 60.2 (2022) to 87.3 (2017) for SRP, and −74.8 (2021) to 63.7 (2023) for TSS. Within year variation was also present with average seasonal concentration reduction efficiencies (%) across time ranging from 33.4 (Fall) to 68.3 (Spring) for NO_3–N, 41.6 (Spring) to 58.0 (Fall) for TP, 72.7 (Spring) to 84.2 (Summer) for SRP, and −22.9 (Summer) to 37.9 (Fall) for TSS. Overall, log response ratios (ln R) showed that all nutrient species as well as sediment were negative, indicating that a greater inflow load compared to outflow was occurring. Similar to PC East, cross comparison found that PC West was also most effective at reducing SRP loads followed by NO_3–N and TP with TSS exhibiting the least overall reductions (Table 3).

All Prairie Creek West nutrient statistical models converged with split R-hat being less than 1.10 for all parameters. The number of observations below detection limits were also too high to fit the model of SRP in Prairie Creek West and thus only observed data without imputation were used. Nitrate removal efficiency was highly variable across years (Fig. 6A). Specifically, efficiency in 2018, 2023, and 2024 had the lowest removal efficiency at low nitrate inflow concentrations but highest at high nitrate inflow concentrations during the same years. SRP removal efficiency varied little across years (Fig. 6B). Overall, SRP removal efficiency was lowest in 2022 and 2023 and highest in 2017 and 2020. TP removal efficiency varied across years and inflow concentrations (Fig. 6C). Removal efficiency of TP was lowest in 2018, 2020, and 2021 and highest in 2023 and 2024 at low inflow concentrations. In contrast, all years had similar removal efficiency of TP at high inflow concentrations. Removal efficiency of TSS did not change across inflow concentrations (Fig. 6D) but was variable across years. TSS removal efficiency was lowest in 2018, 2020, and 2021 and highest in 2023 and 2024.

Season	Parameter	SRP (PO_4^{3-}–*P*)	TP (PO_4^{3-}–*P*)	NO_3–*N*	TSS
Spring	Average stream inflow concentration (mg/L)	0.072	0.285	8.375	40.2
	Average outflow concentration (mg/L)	0.020	0.166	2.652	37.1
	Concentration reduction (%)	72.7	41.6	68.3	7.6
	Total load in (kg)	41.4	163	5,352	24,599
	Total load out (kg)	8.05	95.1	1,683	21,515
	Load reduction (%)	80.6	41.5	68.6	12.5
	Mass removed (kg)	33	68	3,669	3,083
Summer	Average stream inflow concentration (mg/L)	0.108	0.320	2.128	32.3
	Average outflow concentration (mg/L)	0.017	0.181	0.947	39.7
	Concentration reduction (%)	84.2	43.5	54.0	−22.9
	Total load in (kg)	149	478	4,214	48,636
	Total load out (kg)	17.3	166	1,691	30,641
	Load reduction (%)	88.3	65.3	59.9	37.0
	Mass removed (kg)	131	312	2,523	17,996
Fall	Average stream inflow concentration (mg/L)	0.090	0.330	3.650	38.2
	Average outflow concentration (mg/L)	0.021	0.138	2.429	23.7
	Concentration reduction (%)	76.6	58.0	33.4	37.9
	Total load in (kg)	53.3	191	1,727	20,915
	Total load out (kg)	5.90	43.5	725	7202
	Load reduction (%)	88.9	77.2	58.0	65.6
	Mass removed (kg)	47	147	1,002	13,713
Winter	Average stream concentration (mg/L)	0.096	0.270	8.806	32.8
	Total load in (kg)	0	0	0	0
	Total load out (kg)	0	0	0	0
Annual	Average stream inflow concentration (mg/L)	0.094	0.312	4.275	35.9
	Average outflow concentration (mg/L)	0.019	0.167	1.756	35.5
	Concentration reduction (%)	80.1	46.3	58.9	1.2
	Loading rate ($g/m^2/yr$)	5.5	18.7	254	2,115
	Total load in (kg)	243	831	11,293	94,150
	Total load out (kg)	31	305	4,098	59,358
	Load reduction (%)	87.1	63.4	63.7	37.0
	Mass removed (kg)	212	527	7,195	34,792
	Mass removed ($g/m^2/yr$)	0.60	1.48	20.2	98
	Ln (R) ~ ratio of load out/load in	−2.05	−1.00	−1.01	−0.46

Table 3. Surface water nutrient and sediment data seasonally for Prairie Creek West from 2017 to 2024.

Literature comparison

Nutrient reduction findings for the Prairie Creek East and West Wetlands can be contextualized on a site-specific scale with several recently published load reductions associated with other wetlands. Land et al.[13] and Ury et al.[14] found average TP load retention efficiencies of 44% (median 49%) and 42% (median 38%), respectively. Prairie Creek East and West wetlands exhibited average TP load reduction efficiencies of 73% (median 79%) and 63% (median 68%), respectively, both greater than the average and median load reductions of the wetlands included in these meta-analyses. Overall, rates of TP retention in Prairie Creek East and West Wetlands were found to be 2.81 $g/m^2/yr$ and 1.48 $g/m^2/yr$, respectively. These values are comparable to the median TP load retention rate of 2.0 $g/m^2/yr$ found by Ury et al.[14] and average TP load retention rate of 4 $g/m^2/yr$ found by Land et al.[13]. Soluble reactive phosphorus retention efficiencies of 88% for Prairie Creek East and 87% for Prairie Creek West Wetlands are much greater than the average and median orthophosphate retention efficiencies of 48% and 47% found by Ury et al.[14]. The Land et al.[13] meta-analysis also included data on TN load reductions in which average TN load retention efficiencies were 39% (median 36%). Although TN was not measured for this study, a significant and often majority (75+% much of the time) of the total nitrogen in the watershed is NO_3–N, allowing for a comparison to the meta-analysis[8]. Nitrate load reduction efficiencies for Prairie Creek East and West Wetlands were found to be 67% (median 70%) and 64% (median 61%), respectively. Another study on constructed 'treatment train style' wetlands treating river water found nitrite + nitrate (NOx) nutrient load retention efficiencies of 74% and 59%[15], also comparable to percentages found in this study. It should be noted again though, that since load is dependent on concentration and volume it is possible that a wetland could have a high concentration reduction efficiency with a small total load reduced due to low volumes of water captured. On the other hand, load reductions can also be high even if a concentration difference between inflow and outflow is small given a large enough reduction in the volume of water. Since volume treated is tied to wetland design

Season	Parameter	Prairie Creek East	Prairie Creek West
Spring	Average daily wetland inflow volume (m^3)	1,713	1,772
	Total wetland inflow volume (m^3)	803,203	694,494
	Total wetland outflow volume (m^3)	636,400	560,497
	Stream total (m^3)	29,654,984	29,654,984
	Stream treated (%)	2.71	2.34
Summer	Average daily wetland inflow volume (m^3)	1,854	2,208
	Total wetland inflow volume (m^3)	1,297,794	1,468,164
	Total wetland outflow volume (m^3)	911,969.8	1,104,786
	Stream total (m^3)	9,293,071	9,293,071
	Stream treated (%)	14.0	15.80
Fall	Average daily wetland inflow volume (m^3)	1,474	1,888
	Total wetland inflow volume (m^3)	443,765	541,752
	Total wetland outflow volume (m^3)	263,416	321,358
	Stream total (m^3)	9,320,260	9,320,260
	Stream treated (%)	4.8	0.02
Winter	Average daily wetland inflow volume (m^3)	0	0
	Total wetland inflow volume (m^3)	0	0
	Total wetland outflow volume (m^3)	0	0
	Stream total (m^3)	29,053,771	29,053,771
	Stream treated (%)	0	0
Annual	Average daily wetland inflow volume (m^3)	1,731	2,012
	Total wetland inflow volume (m^3)	2,303,610	2,137,780
	Total wetland outflow volume (m^3)	1,688,234	1,644,030
	Stream total (m^3)	77,322,085	77,322,085
	Stream treated (%)	3.0	2.76

Table 4. Hydrology data for each wetland seasonally and over the entire period from 2017 to 2024.

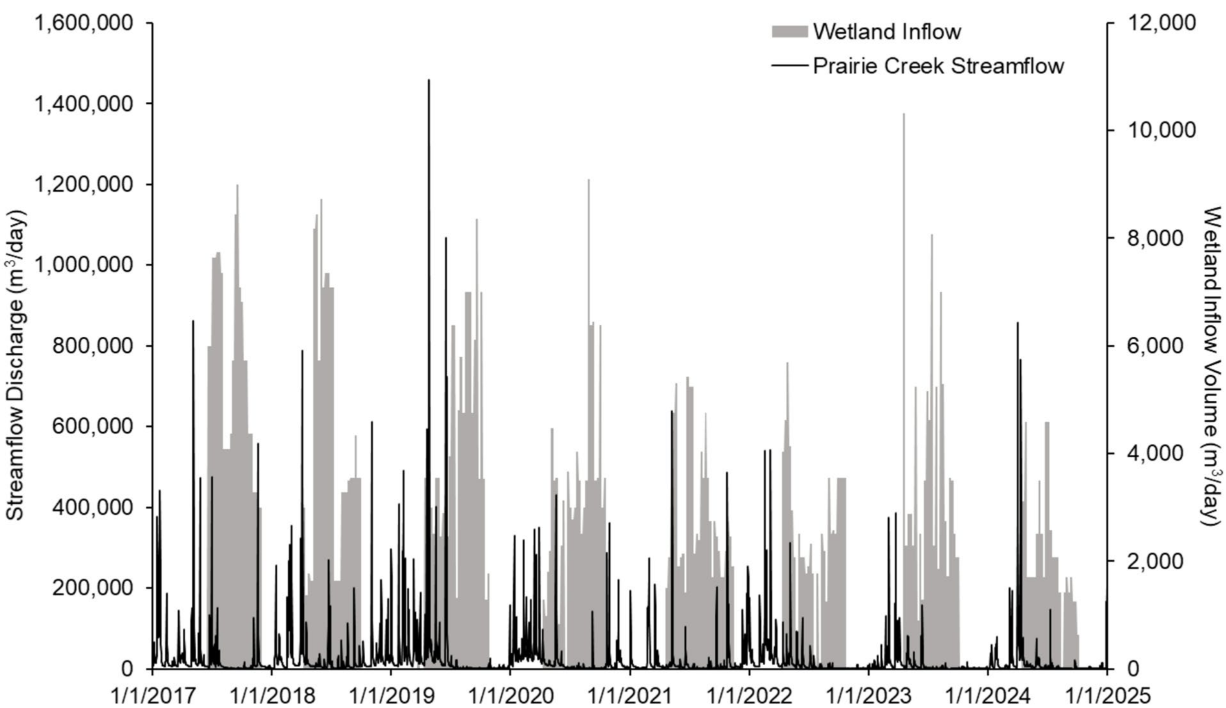

Fig. 3. Graph of average daily streamflow discharge for Prairie Creek (left axis) and total wetland inflow volume of Prairie Creek East and West combined (right axis).

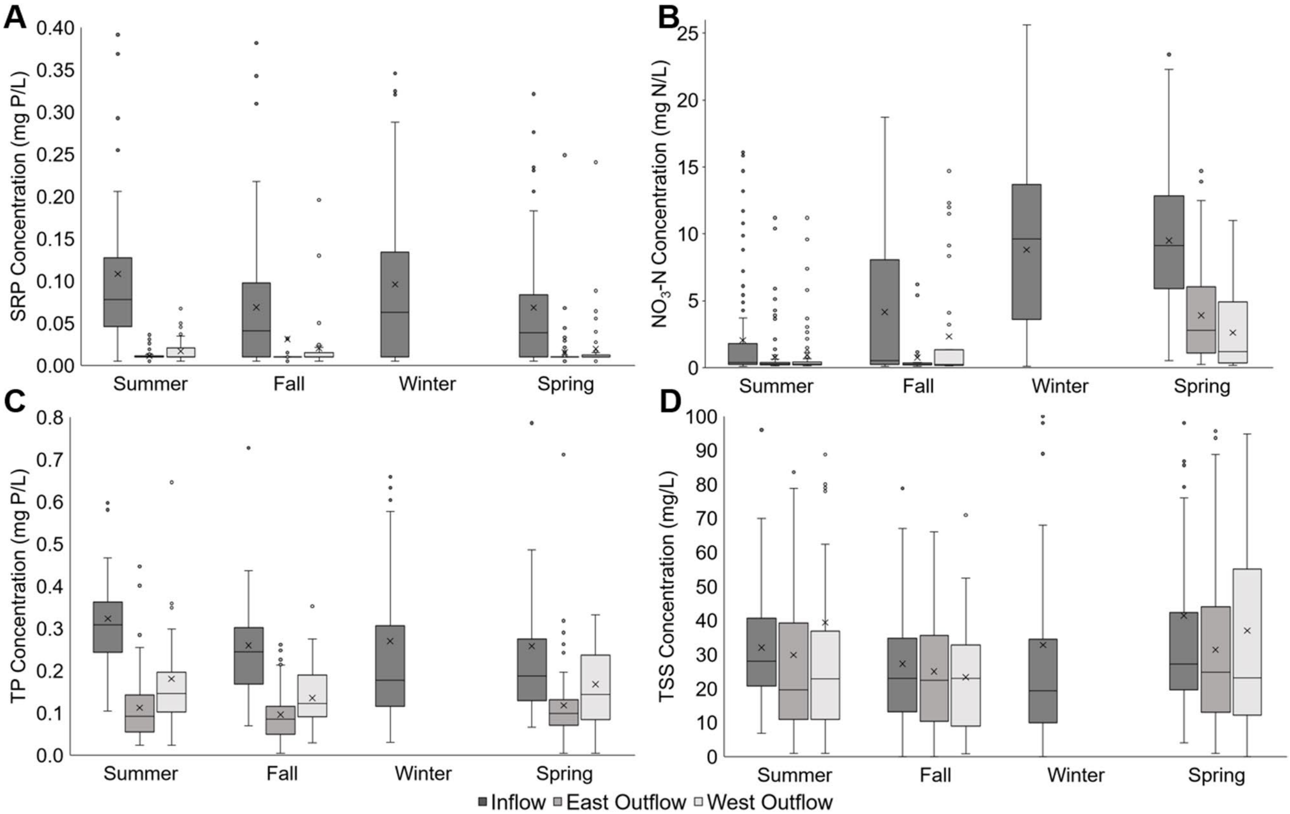

Fig. 4. Boxplots indicating median values and variation in nutrient and sediment concentrations in surface water by season for Prairie Creek, East Outflow, and West Outflow.

features in reconstructed wetlands, including how and what volume of water enters the wetland at any given time (in addition to other factors such as vegetation type and removal rates, and sediment nutrient accumulation or release), it becomes increasingly important for designs that seek to maximize conservation efficacy of nutrient reduction focused surface water wetlands[35,36]. Extending this, it is also important that long term monitoring data be maintained to document these nutrient changes as a result of restored wetlands. Complicating this is the need for long term monitoring to accurately capture a variety of flow events, which often requires a degree of high temporal resolution sampling that can be challenging to maintain for a variety of logistical and funding constraints.

Scaling up

Nutrient reduction findings for the Prairie Creek East and West Wetlands can also be contextualized on a watershed scale comparing load reductions with stream flow totals as well as runoff goals in GLSM. The Prairie Creek site is situated on ~ 32 ha with ~ 8 of these being classified as 'wetland' habitat with the whole site equating to roughly 1% of the total Prairie Creek Watershed area. Despite being situated on only 1% of the watershed area, this site 'treats' the equivalent of an average of 5.75% of the total annual flows through the watershed. It is not difficult to take the percent annual stream flow treated coupled with average concentration reductions for various nutrient species or sediment and use it to scale up to various load reduction targets. For example, if there was a watershed goal to reduce TP loading in the Prairie Creek watershed by 40%, only using wetlands like the study site, given that the two phases of this wetland average out to about a 55% concentration reduction annually across years and seasons–one would need to treat the equivalent of about 73% of the annual stream flows, meaning that a total of 12 or 13 of these sites would be needed (~ 388 ha, or 12.5% of the total watershed area) to achieve this goal. The GLSM watershed does have a series of nine element plans and a TMDL report on file providing some guidance as to total phosphorus/nitrogen/sediment load reduction goals. However, given the ongoing conservation efforts in the watershed, these estimates should be revisited prior to formally setting a watershed wide total load goal.

On a larger level, this scaling process could be used to project these results onto nutrient reduction load total goals for other systems that do have more specific reduction targets. For example, the Ohio Phosphorus Task Force recommends that the Lake Erie Watershed work towards a 40% spring phosphorus loading reduction goal to address HABs and other water quality issues[37]. This reduction goal for Lake Erie references 2008 spring loading levels from the Maumee River (roughly equivalent to ~ 573,000 kg P of the TP load (124,000 kg of which is the dissolved reactive phosphorus (DRP) load) to be reduced every year between March 1 and July 31), a particularly wet year that embodies the kind of problematic runoff conditions driving water quality issues[37].

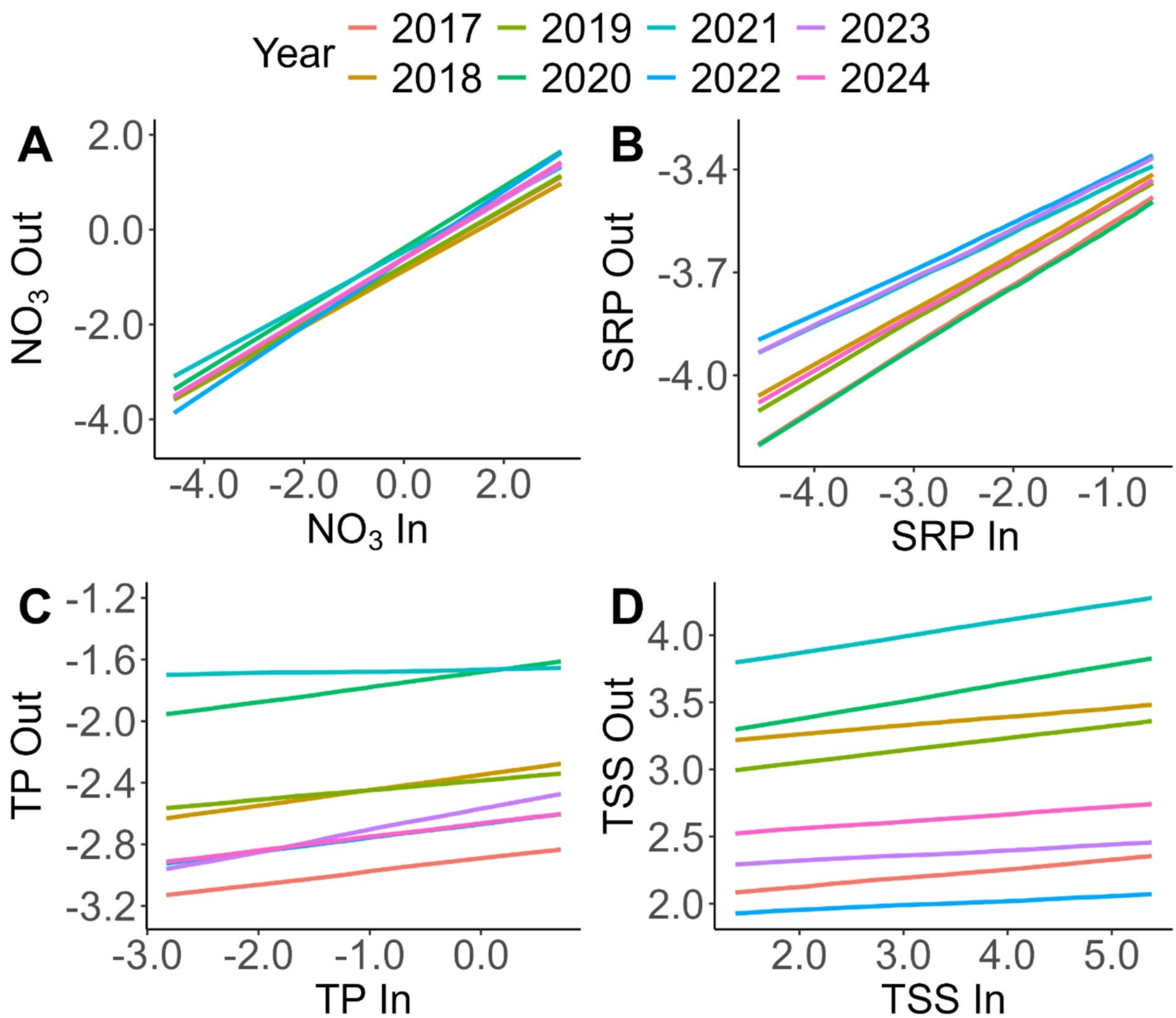

Fig. 5. Input-output model predictions of the relationship between Prairie Creek East wetland input concentrations (x-axis) and output concentrations (y-axis). Solid lines represent medians from the posterior predicted probability distribution. All nutrients are natural log transformed of the original values in mg/L.

Data from this study, indicate that on average Prairie Creek East and West combine to reduce TP by an average of ~ 66 kg y^{-1}, with the 'best year' in this study showing reductions of 112 kg between March 1 and July 31. Looking at data from the most efficient year suggests that a total of ~ 5,100 of these sites (~ 165,112 site ha with 41,278 classified as wetland) would be needed, totaling between 1.9% (based on total site wetland area) to 7.7% (based on total site area) of the entire 2,153,834-ha Maumee River watershed area to achieve this goal in the western basin of Lake Erie if no other conservation measures were considered. If the goal was only to drop SRP by the suggested 124,000 kg, leaving the remaining particulate phosphorus to another watershed practice (e.g. cover cropping, riparian strips, etc.), given that Prairie Creek East and West demonstrated a 'best spring year' of 68 kg SRP reduction (average 31 kg/yr) only a total of ~ 2000 of these sites (~ 64,750 site ha with 16,187 wetland ha) would be needed to achieve the DRP target load. Interestingly, these land area estimates are not too far off from the proposed 3–7% minimum watershed area that should be classified as wetland proposed by Mitsch and Gosselink[38] in their discussion of wetland areas needed for watersheds to better realize a myriad of ecosystem services, including water quality improvements. These estimates can help inform larger scope conservation programs, such as the H2Ohio program, in estimates of land area needed to meet water quality goals. This study can also help provide some sense of the kind of maintenance and upkeep that this level of conservation could require—reinforcing the idea that restored wetlands are not a kind of 'set up and forget' practice and that some level of active management, engineering design, and budget is likely required, especially if sediment removal to 'reset' systems is to be part of an ongoing schedule.

Ongoing improvements

Bringing these study results back to the site level, while certainly positive overall from a nutrient and sediment perspective—there is room for improvement. Specific site level improvements should target increasing retention

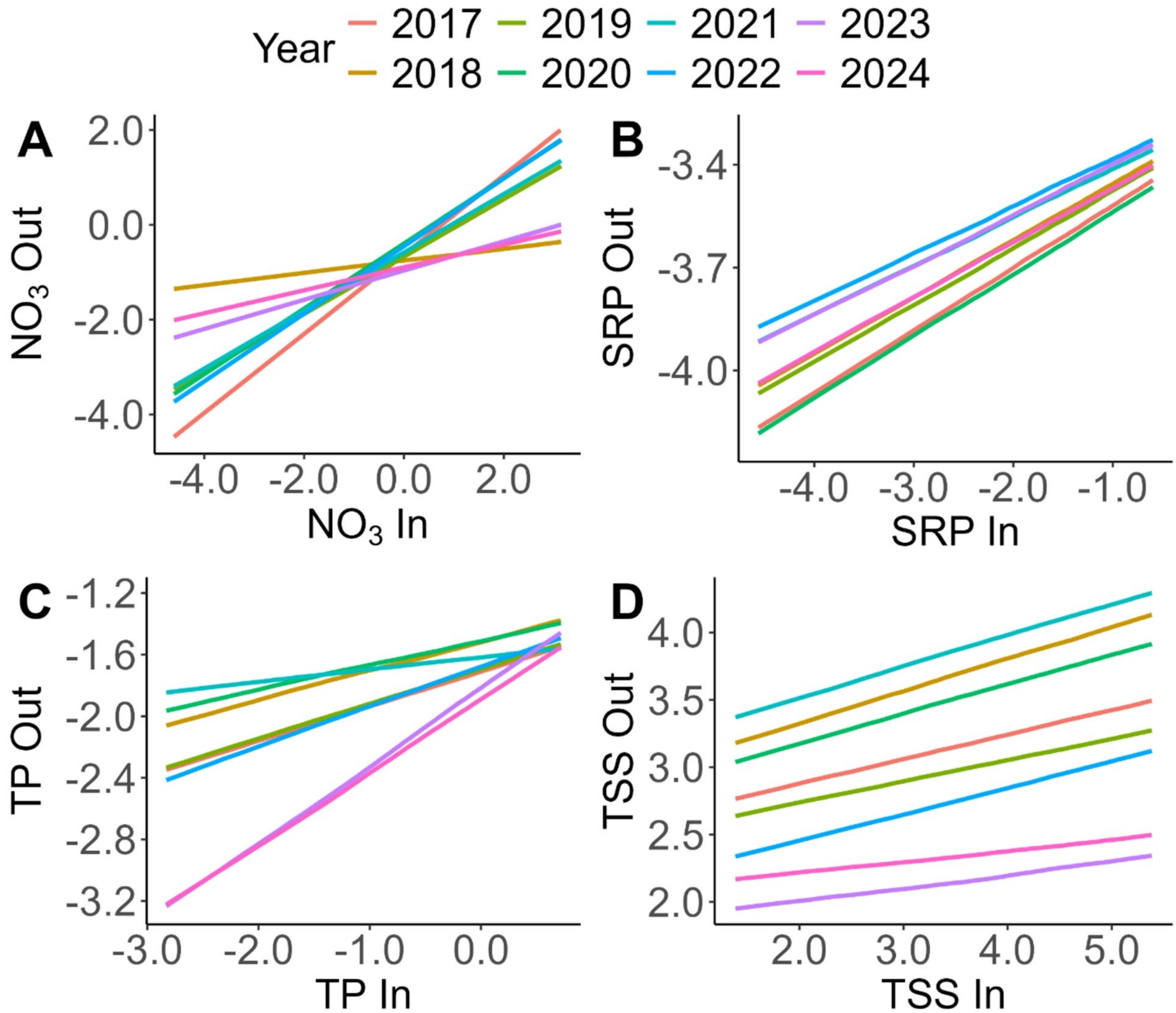

Fig. 6. Input-output model predictions of the relationship between Prairie Creek West wetland input concentrations (x-axis) and output concentrations (y-axis). Solid lines represent medians from the posterior predicted probability distribution. All nutrients are natural log transformed of the original values in mg/L.

capacity of particulates, which exhibited the weakest of the natural log response ratio effect sizes in the study (though they still indicated the wetland was a sink). In fact, TSS concentration data showed similar values or even slightly higher outflow than inflow for much of late 2020 and 2021 after years of acting as a sediment sink with similar inflow loading rates. This was interpreted as evidence that the settling pools had filled in to some degree. Once confirmed with observations, settling pools were dug out late 2021 to 'reset' in time for the start of the 2022 season which showed a return to lower TSS outflow concentrations and load reductions. This recent effort was shown in the increased removal efficiency of TSS in Prairie Creek East and West in years 2022–2024 as noted by lines at the bottom of the graph in Figs. 5 and 6. Unfortunately, the settling pools of Prairie Creek East and West are not that large or deep even when newly excavated so the flip from sink to source will likely reoccur. Similar pump style wetlands in the region which have larger surface area and deeper settling basins on the front design end have been shown to have much higher particulate retention, likely due to increased settling pool residence time and inflow energy reduction which can allow for suspended particulates to 'fall out' of solution[9]. One possible solution for Prairie Creek Wetlands would be to expand and join the East and West settling pools at the inflow point. This would involve reconfiguration of some walking trails and logistics at the site but greatly expand surface area, which would in turn provide a way to dig the pool deeper as side slope practicality requires a larger kind of surface area to angle slopes down to the desired depth which could hopefully improve water quality for years to come.

Conclusion

Restored wetlands like those at Prairie Creek can serve as tools to reduce nutrient loading as confirmed by results of this study. Long-term wetland monitoring datasets like those from the GLSM region that are nearly a decade old now allow researchers and managers to detect trends in nutrient loading of both streams as well as wetlands over time. Future work should dive deeper into disentangling the ecological mechanisms of these nutrient and sediment reductions—aspects such as succession in these habitats as change in soils or vegetation inevitably occurs or identifying and quantifying different zones of the wetland and what effect they have provide valuable information. Doing so will help inform management decisions related to site maintenance or design of future restoration projects dealing with area needed or placement localities preferred to maximize watershed benefits (e.g. residence times, volumes, layouts, etc.), which can in turn support realistic timelines of water quality improvements for conservation planning. Looking ahead, it is essential that monitoring be continued at this and other reconstructed wetlands to help inform future changes and understanding of these ecosystems, particularly as they are increasingly utilized as critical pieces of large-scale water quality improvement plans.

Data availability

Datasets generated during the current study are available from the corresponding author upon reasonable request.

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Author contributions

All authors contributed to the study conception and design. Material preparation, data collection and analysis were performed by Stephen J. Jacquemin, Jason Doll, Morgan Grunden, Haley Hoehn, and Theresa Dirksen. The first draft of the manuscript was written by Stephen J. Jacquemin, Jason Doll, Morgan Grunden, and Haley Hoehn, and all authors commented on the manuscript. All authors read and approved the submitted manuscript.

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Declarations

Competing interests

The authors declare no competing interests.

Additional information

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