Biogeochemistry (2025) 168:60
https://doi.org/10.1007/s10533-025-01250-7

Impact of river reconnection for coastal restoration on nitrate reduction in brackish marsh soils and bay-bottom sediments in coastal Louisiana, USA

Mercedes M. Pinzón · John R. White

Received: 13 May 2024 / Accepted: 27 June 2025 / Published online: 22 July 2025

Abstract Wetlands provide important ecosystem services, including improving surface water quality through nutrient removal. Louisiana has experienced ~ 4800 km^2 of coastal wetland loss between 1932 and 2016 due to high relative sea level rise and reduced sediment from the Mississippi River due to levees. The 2023 LA Coastal Master Plan aims to restore Louisiana's degraded coastline through restoration projects, including sediment diversions or river reconnection. The Mid-Barataria Sediment Diversion Project will reconnect the river sediment-laden water with the coastal wetlands of Barataria Basin to nourish degrading marshes. However, the diversion will also deliver substantial nitrate (NO_3^-) to the basin, potentially negatively impacting water quality. We quantified NO_3^- reduction rates at these high (2 mg/L) and low (0.5 mg/L) water column concentrations for marsh and submerged estuarine sediments using intact cores and a laboratory incubation. An additional treatment where 2 cm of mineral river sediment was placed over the organic marsh soil as a future, post-diversion scenario to simulate sediment deposition on the marsh once the river is reconnected. We hypothesized that NO_3^- reduction rates would decrease once mineral sediment is deposited on the organic marsh soil. For an aerobic water column, nitrate reduction rates for the vegetated marsh, post-diversion marsh, submerged eroded marsh, and estuarine sediment zones were 71.1 ± 2.7, 27.8 ± 4.5, 19.7 ± 1.2, and 13.0 ± 0.75 mg N m^{-2} d^{-1}, respectively. Thus, the post-diversion marsh NO_3^- reduction rate decreased by ~ 60% compared to the current vegetated marsh. However, we predict the newly deposited sediment will increase NO_3^- removal by 1.17× in the eroded marsh and estuarine sediment zones, which are always flooded and will receive river sediment. The marsh is only flooded 31–48% of the time, lessening the impact of the reduction. These findings can improve predictive water quality models used to assess nutrient loading and fate more accurately across the basin under the river reconnection scenario and inform other deltaic regions as freshwater flows are restored to coastal systems globally.

Keywords Wetland · Biogeochemistry · Nitrogen cycle · Coastal restoration

---

Responsible Editor: James B. Shanley

M. M. Pinzón · J. R. White (✉)
Department of Oceanography and Coastal Sciences, Louisiana State University, Baton Rouge, LA, USA
e-mail: jrwhite@lsu.edu

M. M. Pinzón
e-mail: mercedes.pinzon@ucf.edu

Present Address:
M. M. Pinzón
Department of Biology, University of Central Florida, Orlando, FL, USA

Introduction

Wetlands represent strategic ecosystems of great importance in linking the natural and built environments, the economy, and human communities. Wetlands serve a critical role because they regulate global biogeochemical cycles and can provide abundant ecosystem functions and services. Coastal wetlands along the Gulf of Mexico support 30% of the United States' commercial fish production and contain 50% of the country's oil and gas refinery capacity (Mendelssohn et al. 2012). Louisiana wetlands account for 40% of the national estuarine and coastal wetlands in the contiguous United States; however, between 1932 and 2016, there was a land loss of ~ 4,877 km^2 (CouvilLion et al. 2017). Thus, the Louisiana coast is experiencing ~ 80% of the nation's total coastal wetland loss (Wood et al. 2017).

The Louisiana wetland loss issue is due to a myriad of factors, and include the combined effect of regional coastal subsidence (~ 10 mm yr $^{-1}$) (Morton et al. 2005), eustatic sea-level rise (~ 3.2 mm yr^{-1}; Church et al. 2013), and shoreline edge erosion (DeLaune & White 2012; Sapkota & White 2021). Coastal Louisiana has one of the highest relative sea level rise rates (RSLR), exceeding 13 mm yr^{-1} (Jankowski et al. 2017). It has been demonstrated that when RSLR is compared with Louisiana marsh vertical accretion, only 65% of total coastal wetlands in southern Louisiana may likely keep pace with rising sea levels in the future (Jankowski et al. 2017). Thus, Louisiana could lose an additional 5,800–10,000 km^2 of coastal wetlands over the next 50 years if restoration approaches are not adopted to combat rising sea levels (Peyronnin et al. 2017). Anthropogenic activities have also contributed to this rapid coastal wetland loss. Extensive channeling and closure of river distributaries through levee construction have increased salinities and extensive dams along the Mississippi River have resulted in a 50% decrease in sediment delivery and freshwater inflow to the Louisiana coastline (Blum & Roberts 2009). This sediment starvation is also compounded by many other factors, such as widespread hydrological disturbance from oil and gas extraction and barrier island loss due to erosion (White et al. 2019).

The 2023 Louisiana Comprehensive Master Plan is a US $50 billion coastal restoration plan designed by the Louisiana Coastal Protection and Restoration Authority (CPRA) to rebuild and preserve up to ~ 25,000 km^2 of the Mississippi River deltaic plain. The coastal master plan outlines several restoration approaches to provide the necessary habitats to support the ecosystems while reducing the risk of flooding in the region's coastal communities. In particular, sediment diversions will be openings built within the levees of the Mississippi River that will be operated to allow freshwater, nutrients, and sediments to flow into the coastal basins at high river stages, reconnecting the river with the riparian coastal wetlands and bays to increase resilience to sea level rise and storms (Peyronnin et al. 2017).

Sediment diversions, however, will transport significant nutrient loads, primarily nitrate (NO_3^-), from the river to the vegetated and open-water habitats in the coastal basins. Nitrate concentration in the river ranges from 0.5 to 3.0 mg NO_3-N L^{-1} and generally increases from early spring to late summer due to wastewater input and agriculture runoff within the watershed (Mitsch et al. 2005; Pellerin et al. 2014; Tuffilaro et al. 2024). The introduction of NO_3^- could lead to increased algal blooms (Jung et al. 2023).

However, coastal wetlands are a natural sink for nutrients and have great potential to remove nitrogenous oxides (e.g., nitrate: NO_3^-; nitrite: NO_2^-) from surface waters as nitrate diffuses into sediments/soils, is converted to N_2 gas through microbial respiration and then released into the atmosphere (Burgin & Hamilton 2007; Reddy and DeLaune 2008). This facultative microbial mediated process is called direct denitrification (i.e., $NO_3^- \rightarrow N_2$) and is the main pathway for NO_3^- removal from coastal wetland ecosystems in Louisiana, ranging from 89 and 95% (Reinhardt et al. 2006; Vaccare et al 2019). Nitrate removal in coastal wetlands can also be performed, to a much less extent, by an alternative, conservatory reductive pathways such as dissimilatory nitrate reduction to ammonia (DNRA) which has been shown to be minimal in these coastal marshes (Bowes et al. 2022; Upreti et al. 2021). For example, VanZomeren et al. (2012) found that by adding labeled ^{15}N-NO_3 to LA *Spartina patens* marsh, 91% was lost to denitrification and less than 7% of the added N remained in the soil, which included the microbial biomass. Vaccare et al. (2019) spiked nitrate into sealed, anaerobic serum bottles containing Barataria Bay marsh soil and sediments and recovered through mass balance, an average of 93% of the N recovered through denitrification. Upreti et al. (2022), using the N_2: Ar method found that DNRA was < 10% of total NO_3^- reduction. In addition, plant uptake has been shown to assimilate up to 30% of the surface water nitrate (VanZomeren et al 2012). The nutrient mitigation function of wetlands intercepting the river water before reaching the Gulf of Mexico could help to alleviate coastal eutrophication and reduce annual coastal hypoxia (Hurst et al. 2016; Mitsch et al. 2005; Rabalais et al. 2002).

Barataria Basin has a coastal erosion rate of ~ 41 $km^2\ yr^{-1}$ (Wood et al. 2017). Barataria Basin marshes are more vulnerable to coastal erosion due to greater exposure to wind-driven waves, saltwater intrusion, and a range of subsidence rates dependent upon location (Byrnes et al. 2019). The Mid-Barataria Sediment Diversion is projected to nourish and maintain existing wetlands and create ~ 54 km^2 of new land over 50 years (CPRA 2017). Previous studies have estimated that high river discharge pulsing can contribute to high burial rates by depositing up to 2–5 cm of sediment over the organic marsh (Bevington et al. 2017; Shaw et al. 2018). However, beyond the introduction of sediments, the high nutrient-laden water load from the Mississippi River into Barataria Basin poses two important questions: 1) To what extent does the ecosystem have the capacity for NO_3^- reduction and 2) how will NO_3^- reduction change in the future after the organic-rich vegetated marsh soil becomes covered with mineral riverine sediments? The process of denitrification has many regulators, such as organic matter availability, oxygen content, nitrate concentration, facultative denitrifying microbes, and temperature (Smith & Tiedje 1979). Thus, the goal of this study was to evaluate the NO_3^- reduction potential of soils/sediments with different soil OM content and microbial activity in Barataria Basin. We hypothesized that organic matter limitation in mineral-rich coastal wetlands would reduce the NO_3^- reduction potential. Specific objectives included to (1) determine the relative importance of different soils/sediments; vegetated marsh, eroded, submerged marsh and submerged estuarine sediments nitrate reduction rates(2) assess the temporal change in nitrate reduction once the organic marsh substrate is covered with riverine mineral sediments as a future diversion scenario; and (3) evaluate the magnitude of nitrate reduction rates associated with high (2.0 mg N L^{-1}) and low (0.5 mg N L^{-1}) nitrate concentration to capture the spatial range of concentration that can be expected across the basin from the diversion.

Material and methods

Study site description

Barataria Basin is a shallow, micro-tidal, bar-built estuary in southeastern Louisiana, 40 miles south of New Orleans, Louisiana. The basin covers approximately 6,000 km^2, encompassing both wetlands and open water areas, with an average bay depth of ~ 2 m (Das et al. 2012). The basin is a wave-dominated estuary that experiences ~ 30 cm diurnal lunar tides (Georgiou et al. 2005). However, most of the water level variability is wind-driven during cold fronts in winter, and in combination with shallow depths, results in the basin having a well-oxygenated surface water column (Day et al. 2021). Barataria Basin surface water salinities range from near 0 ppt in the northern region of the basin near Lac Des Allemands to 25 ppt in the southern region close to the barrier island inlets to the Gulf of Mexico (Li et al. 2011). The basin is undergoing a deltaic degrading stage, with an average wetland loss rate of ~ 13.3 $km^2\ yr^1$ due primarily to marsh edge erosion (Sapkota & White 2021), sediment starvation from large-scale flood control levees along the Mississippi River, and extensive channeling causing salinity intrusion into fresh marsh regions (Wang et al. 2017). The study area is in the northeastern portion of Barataria Basin, to the west of the Lower Mississippi River (29°29′54.8"N 89°55′06.2"W) (Fig. 1A). The study site is located in the project nutrient-influenced area of the currently under construction Mid-Barataria Sediment Diversion. The marsh site is brackish with a mean surface water salinity of 9 ppt and a mean surface water temperature of 23.3 °C (CPRA, n.d.). The wetland site (i.e., interior marsh) was dominated by mesohaline wiregrass such as *Spartina sp* (51.5%) and *Juncus Roemerianus* (23.5%) (LDNR 2008). Due to the low slope landscape and small tidal range, similar marsh characteristics (Vaccare et al. 2019) extend for long distances compared to other coastal areas with greater slope and higher tidal regimes, where vegetation community changes occur over shorter horizontal distances. The vast majority of the area

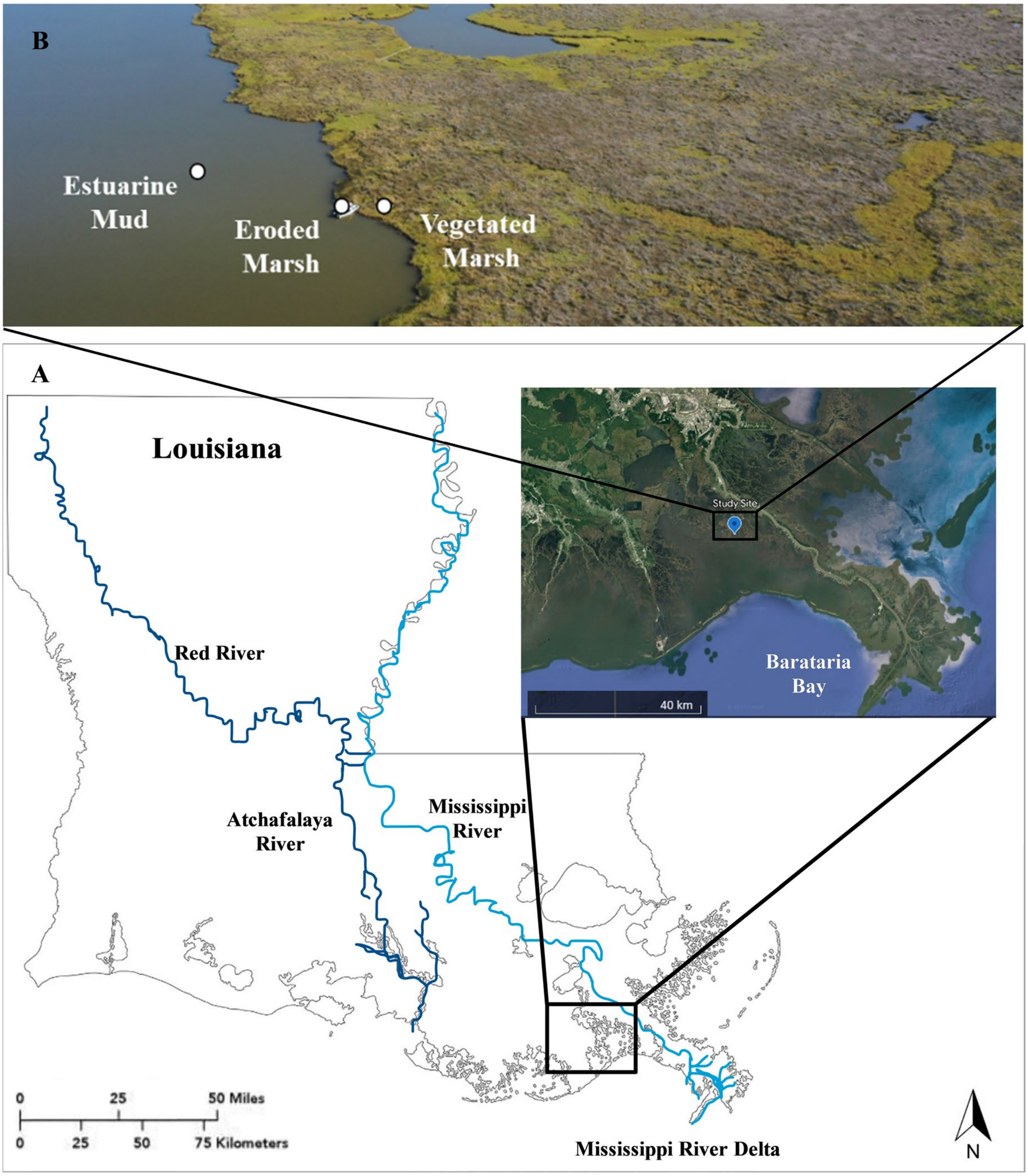


Fig. 1 **a** The State of Louisiana with the path of the Mississippi River, and the inset shows a Google Earth satellite images of Barataria Basin, Louisiana in the Mississippi River Delta with the sampling site; **b** is a low angle oblique drone photograph depicting the sampling transect

is brackish marsh with some saline marsh present toward the bay and open water.

Field sampling

In June 2022, three soil and sediment substrates were sampled at our wetland site: vegetated organic marsh, submerged eroded marsh, and submerged estuarine sediment (Fig. 1B). The vegetated organic marsh soil was located 3.0 m into the intertidal marsh platform, and the aboveground vegetation was clipped at the time of core sampling. The eroded organic marsh soil was located 0.5 m from the marsh edge into the open water and is the product of collapsed marsh due to wave action that is submerged 100% of the time. The submerged estuarine sediment was located 125 m from the marsh edge and is the bay-bottom sediment typically found in open water areas. Eight, 15 cm long, 7 cm diameter, intact, field-replicate cores were taken at each substrate zone. In addition, eight extra cores in the vegetated organic marsh were taken to receive river sediment collected in Mardi Gras Pass (29°31′32.6"N 89°41′06.3"W), a Mississippi River crevasse. Marsh cores were collected using a sharpened acrylic tube, gently pushed into the soil to avoid compaction (Upreti 2019). Bay-bottom estuarine sediment cores were collected using a piston core sampling device. All cores from each substrate type were collected within a 2 m^2 quadrat. Previous research in Barataria Bay has shown similar soil characteristics across three different wetland islands (Vaccare et al 2019). Also, quadruplicate cores were collected, extruded, and sectioned into 0–5 cm and 5–10 cm sections at each substrate zone for initial soil characterization. The soil sections were placed in Ziploc bags, and the 32 intact cores were sealed with rubber stoppers on the bottom and covered. Additionally, four mineral sediment samples (0–5 cm) were collected at Mardi Grass Pass for soil characterization. All samples were transported back to the Wetland and Aquatic Biogeochemistry Laboratory (WABL) at Louisiana State University (LSU). In the lab, the soil section samples were weighed, transferred to polyethylene sediment containers, homogenized, and stored at 4 °C until analysis.

Intact-core incubation: nitrate reduction potential under an aerobic water column

Upon return to the laboratory, the overlying site water was removed by siphon from the intact cores. Then, 2 cm of mineral mud from the Mardi Gras Pass was placed over eight randomly selected vegetated marsh cores to mimic future conditions after Mid-Barataria Sediment Diversion operation. All cores were reflooded with deionized water to mimic the river freshwater condition to a 15 cm water column and placed into a ~ 21 °C water bath and left to equilibrate overnight while the water column was bubbled with room air using aquarium pumps. Sixteen cores (4 per substrate) were spiked to bring water column concentration to 2.0 mg NO_3-N L^{-1}, using a liquid KNO_3 standard, representing the Mississippi River's high nitrate concentration during a spring flood event (Mitsch et al. 2005; Pellerin et al. 2014; Tuffilaro et al. 2024). The remaining 16 cores (4 per substrate) were spiked to bring the water column concentration to 0.5 mg NO_3-N L^{-1} to mimic decreased concentrations further from the river diversion input. The 2.0 mg NO_3-N L^{-1} cores were incubated and sampled daily for up to ~ 11 days until nitrate concentration dropped to below ~ 0.5 mg NO_3-N L^{-1}. The 0.5 mg NO_3-N L^{-1} cores were incubated and sampled daily for up to ~ 7 days until water column nitrate concentration dropped to below ~ 0.05 mg NO_3-N L^{-1}. The water column was slowly bubbled with room air to maintain a well-mixed aerobic water column to match the water column characteristics of the field (Steinmuller et al. 2018). The cores were incubated in the dark to prevent algal uptake of N, which could affect areal nitrate reduction rates. Water samples (7 mL) were collected once daily, filtered using 0.45 µm syringe filters, acidified with diluted H_2SO_4 to a pH $<$ 2, and stored at 4 °C until analysis. Surface water sampled from cores was replaced with 7 mL of DI water to maintain a constant core water column. Water samples were analyzed colorimetrically for NO_3^- (U.S.EPA, Method 353.1) concentrations on a SEAL AQ300 Automated Discrete Analyzer with a method detection limit of 0.006 mg N L^{-1} (U.S.EPA 1993).

Soil physicochemical properties

The following soil physicochemical properties were analyzed on top 0–5 cm and 5–10 cm soil samples ($n = 28$): moisture content, bulk density (BD), percent organic matter (%OM), total carbon (TC), total nitrogen (TN), and total phosphorus (TP). Gravimetric moisture content was determined by weighing the homogenized soil subsamples before and after drying at 70 °C until constant weight. Bulk density was calculated for the soil intervals on a dry weight basis. Dried sediment subsamples were ground using a ball mill grinder. Total C and N values were measured on homogenized dried, ground subsamples of soil using a Costech1040 CHNOS Elemental Combustion System with method detection limits of 0.07 g C kg^{-1} and 0.005 g N kg^{-1}, respectively (Costech Analytical Technologies, Inc. Valencia, California). Total P was measured following the Andersen (1976) ashing-digestion method. A ~ 0.3 g of dried and ground subsamples were placed into 50 mL beakers and placed into a muffle furnace at 550 °C for 4 h. Loss on ignition, equivalent to wt % organic matter, was determined by dividing the ashed sample weight by the pre-burn dry sample weight (Sparks 1996). Combusted samples were moistened with DI water and dissolved in 20 mL of 6.0 M HCl. The beakers were transferred to a hot plate at 100–120 °C until dry, and then the temperature was raised to high (~ 370 °C) for an additional hour. Samples were then moistened with DI water and 2.25 mL of 6.0 M HCl was added and placed back on the hot plate until near boiling. After cooling, samples were filtered through a Whatman #41 filter paper into 50 mL volumetric flasks and diluted to volume with DI water. The TP of the digests was determined colorimetrically using a SEAL AQ300 Automated Discrete Analyzer (SEAL Analytical Inc., Mequon, Wisconsin) using U.S.EPA method 365.1 with a detection level 0.002 mg P L^{-1} (U.S. Epa 1993).

Extractable ammonium (NH_4^+), nitrate (NO_3^-), soluble reactive phosphorus (SRP), and dissolved organic carbon (DOC) were measured by placing ~ 5 g of thoroughly homogenized field moist soil subsamples into 40 mL centrifuge tubes. The samples for extractable NH_4^+, NO_3^-, and SRP were extracted with 20 mL of 2 M KCl. Samples were shaken on a longitudinal shaker for an hour and then centrifuged in a Sorvall RC, 5C Plus (Weaverville, NC) centrifuge at 4000 g for 10 min at 10 °C. Samples were vacuum filtered through a 0.45 µm membrane filter, acidified with concentrated H_2SO_4 for preservation, and refrigerated at 4 °C until analysis. Samples were analyzed on a SEAL AQ300 Automated Discrete Analyzer with detection limits of 0.006 mg NO_3-N L^{-1}, 0.007 mg NH_4-N L^{-1}, and 0.002 mg P L^{-1}, using EPA methods 126-A Rev. 5, 103-A Rev. 4, 118-A Rev. 5, respectively. The samples for extractable DOC were extracted with 25 mL of 0.5 M K_2SO_4. The centrifuge tubes shook for an hour, centrifuged for 10 min at 4000 g, and the supernatant was subsequently filtered through a 0.45 µm membrane filters. Samples were acidified to a pH $<$ 2 with 12 M HCl and stored at 4 °C until analysis. The DOC concentrations were determined on a Shimadzu TOC-V CNS Analyzer (Kyoto, Japan).

Microbial biomass nitrogen (MBN) was determined using the chloroform fumigation method after Brookes et al. (1985) with modifications by White and Reddy (2000). Fumigate and non-fumigate samples were prepared by adding duplicate 5 g of wet, homogenized soil subsamples into centrifuge tubes. Twenty-five ml of 0.5 M K_2SO_4 of extract were added to non-fumigate samples and treated as previously described for extractable DOC. The fumigate samples were fumigated with chloroform, placed in a glass vacuum desiccator, vacuum sealed, and incubated for 24 h. Then, the samples were extracted with 25 mL of 0.5 M K_2SO_4, shaken on a longitudinal shaker for 1 h, and centrifuged at 4000 g for 10 min at 10 °C. Samples were then vacuum filtered through a 0.45 µm membrane filters into scintillation vials, acidified with concentrated HCl to a pH $<$ 2 for preservation, and refrigerated at 4 °C until analysis. Microbial biomass nitrogen was analyzed on a Shimadzu TOC-V CNS Analyzer. The difference in total dissolved N between the non-fumigate and fumigate paired samples represents the size of the microbial pool (Brookes et al. 1985; Vance et al. 1987).

Statistical analysis

The nitrate concentration was converted to an areal basis (mg m^{-2}) and plotted vs. incubation time. The slope of the linear regression line provided the nitrate reduction rate in units of mg m^{-2} d^{-1} (Roy & White 2012). Nitrate reduction rates for the 2.0 mg NO_3-N L^{-1} treatment were calculated over 7 days, except for the vegetated marsh soil whose concentration decreased below 0.5 mg NO_3-N L^{-1} by day 3. Nitrate reduction rates for the 0.5 mg NO_3-N L^{-1} treatment were calculated over 5 days, except for the vegetated marsh soil whose concentration decreased below 0.05 mg NO_3-N L^{-1} by day 2. All values were reported as a mean of all samples (N = 4 cores) ± standard deviation for each substrate. Statistical analyses were performed using the "aov" and "lm" functions in R version 4.0.3 (R Foundation for Statistical Computing, Vienna, Austria; R Studio Inc, Boston, MA, USA). One way ANOVAs were run for soil characteristics and nitrate reduction rates. ANOVAs were first tested for normality using the Shapiro–Wilk test ($\alpha < 0.05$) and visually with normal Q-Q plots. The assumption of homogeneity of variances was tested using Levene's test ($\alpha < 0.05$). A generalized linear model (GLM) was used when the data did not meet the assumptions of an ANOVA even after performing transformations. The model with the lowest AICc score was chosen, and if the models had identical AICc scores, model selection was based on the residual plots. Then, ANOVA or GLM was performed using Tukey's honestly significant difference (Tukey's HSD) post hoc test for pairwise comparisons between substrates.

To determine if NO_3^- reduction rates between substrates significantly differ from each other and between nitrate treatments, a GLM was used using a Gamma distribution with a log link (NO_3^- reduction rate ~ substrate * treatment). One-way, single-factor ANOVAs ($\alpha < 0.05$) were used to examine differences in physicochemical and microbial soil properties between each soil substrate type for each soil section.

Results and Discussion

Intact-core incubation: nitrate reduction under an aerobic water column

The average water column NO_3^- reduction rate for the vegetated marsh, post-diversion marsh, eroded marsh, and estuarine sediment were 23.5 ± 2.7, 11.8 ± 0.43, 3.42 ± 0.66, and 8.61 ± 0.19 mg NO_3-N m^{-2} d^{-1}, respectively, for the 0.5 mg NO_3-N L^{-1} spike; and 71.1 ± 2.7, 27.8 ± 4.5, 19.7 ± 1.2, and 13.0 ± 0.75 mg NO_3-N m^{-2} d^{-1}, respectively, for the 2.0 mg NO_3^- -N L^{-1} spike (Table 1). Across all four soil/sediments and for both initial NO_3^- concentrations, the average areal NO_3^- reduction rates were significantly different from one another, with the highest rate in the vegetated marsh soil (Fig. 2).

The average post-diversion marsh NO_3^- reduction rate was ~ 50% of the average vegetated marsh NO_3^- reduction rate for the cores spiked with 0.5 mg NO_3-N L^{-1}; while the mean post-diversion substrate NO_3^- reduction rate was ~ 39% of the mean vegetated marsh NO_3^- reduction rate for the cores spiked with 2.0 mg NO_3-N L^{-1}. These results demonstrate a ~ 49.8% [42.9,56.7] decrease in NO_3^- reduction rate

Table 1 Average areal nitrate reduction rates (mean ± standard deviation; n = 4) from aerobic incubations over 5 days for the 0.5 mg NO_3^- -N L^{-1} treatment, and 7 days for the 2.0 mg NO_3^- -N L^{-1} treatment

Water column condition	NO_3^- treatment	N form	Vegetated marsh	Post-diversion marsh	Eroded marsh	Estuarine sediment
	mg NO_3-N L^{-1}		mg m^{-2} d^{-1}	mg m^{-2} d^{-1}	mg m^{-2} d^{-1}	mg m^{-2} d^{-1}
Aerobic	2.0	NO_3^-	71.1 ± 2.7^a	27.8 ± 4.5^b	19.7 ± 1.2^c	13.0 ± 0.75^d
	0.5	NO_3^-	23.5 ± 2.7^a	11.8 ± 0.43^b	3.42 ± 0.66^d	8.61 ± 0.19^c

Different letters denote significant differences within each nitrate treatment. The letter 'a' denotes a higher significant difference between substrates with decreasing value for subsequent letters

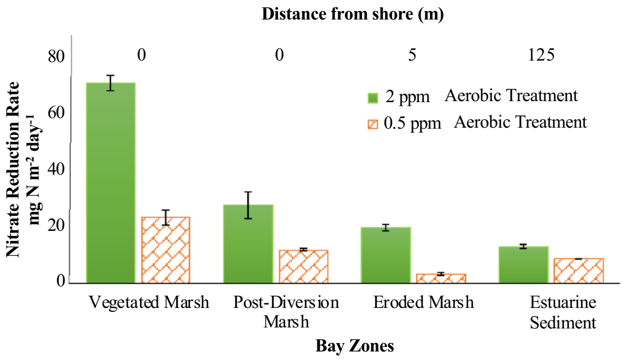


Fig. 2 Average ± standard error (n = 4 cores) of areal nitrate reduction rates for the intact core experiment under aerobic water column conditions for each bay zone and nitrate treatment (0.5 mg N L^{-1} and 2.0 mg N L^{-1})

once the riverine mineral sediment is placed onto the vegetated organic marsh under lower nitrate concentration range; and a 60.9% [54.4,67.4] decrease in NO_3^- reduction rate under the higher range of nitrate concentration (Fig. 2).

The decrease in NO_3^- reduction rate may be explained by the soil physicochemical characteristics of both the vegetated organic marsh and mineral post-diversion river sediment. The concentration of labile organic carbon can increase by the presence of vegetation in the marsh, increasing microbial communities, and enhancing denitrification rates by 55% (HinShaw et al. 2017; Jiang et al. 2017). In addition, the organic marsh soil has lower bulk density values and higher soil moisture contact essentially equivalent to higher soil porosity (Table 2). These aforementioned conditions can lead to a more direct pathway for diffusion from the water column into the anaerobic layers of the soil, which can increase the water column NO_3^- reduction rate. A meta-analysis study

Table 2 Average sediment/soil physicochemical properties (mean ± standard deviation; n = 4) determined for homogenized samples for both sampling depths at three substrate types in Barataria Basin

	0–5 cm soil interval				5–10 cm soil interval		
	Vegetated marsh	Post-diversion marsh*	Eroded marsh	Estuarine sediment	Vegetated marsh	Eroded marsh	Estuarine sediment
Moisture content (%)	78.1 ± 1.78^{a}	33.2 ± 0.25^{d}	82.5 ± 1.44^{b}	47.1 ± 3.4^{c}	76 ± 2.35^{b}	84.4 ± 0.66^{a}	42.6 ± 5.04^{c}
BD (g cm^{-3})	0.216 ± 0.02^{c}	0.927 ± 0.08^{a}	0.189 ± 0.02^{c}	0.758 ± 0.09^{b}	0.248 ± 0.05^{b}	0.169 ± 0.01^{c}	0.854 ± 0.13^{a}
OM (%)	25.4 ± 1.94^{a}	3.4 ± 0.25^{d}	32.7 ± 3.83^{b}	6.8 ± 0.48^{c}	25.2 ± 3.36^{b}	39.7 ± 3.95^{a}	5.6 ± 0.48^{c}
TP (mg kg^{-1})	551 ± 31.5^{b}	599 ± 7.52^{a}	518 ± 24.2^{b}	501 ± 47.0^{b}	502 ± 50.6^{a}	531 ± 15.6^{a}	510 ± 12.1^{a}
TC (g kg^{-1})	107 ± 11.1^{b}	12.2 ± 0.53^{d}	154 ± 20.2^{a}	28.9 ± 2.04^{c}	109 ± 12.7^{b}	200 ± 30.0^{a}	22.5 ± 2.89^{c}
TN (g kg^{-1})	6.83 ± 0.28^{b}	0.8 ± 0.02^{d}	8.16 ± 0.9^{a}	1.55 ± 0.08^{c}	5.98 ± 0.71^{b}	9.54 ± 0.75^{a}	1.37 ± 0.18^{c}
C:N	15.6 ± 1.31^{b}	15.2 ± 0.31^{b}	18.8 ± 0.45^{a}	18.6 ± 0.41^{a}	18.3 ± 0.69^{b}	20.8 ± 1.66^{a}	16.4 ± 0.96^{c}
Ext. NH_4^+ (mg N kg^{-1})	11.9 ± 2.23^{b}	30.2 ± 1.24^{a}	13.1 ± 2.07^{b}	17.8 ± 3.21^{b}	13.3 ± 1.38^{b}	14.6 ± 0.93^{b}	33.8 ± 5.27^{a}
Ext. PO_4^- (mg P kg^{-1})	$0.33 \pm 0.1^{a,b}$	$0.16 \pm 0.14^{b,c}$	0.61 ± 0.23^{a}	0.15 ± 0.06^{c}	0.69 ± 0.31^{a}	0.91 ± 0.52^{a}	0.14 ± 0.06^{b}
Ext. DOC (mg C kg^{-1})	284 ± 24.4^{a}	54.3 ± 5^{b}	265 ± 24.5^{a}	57.6 ± 8^{b}	257.9 ± 18.2^{b}	363 ± 45.5^{a}	39.9 ± 6.5^{c}
MBN (mg kg^{-1})	61.4 ± 9.4^{a}	9.1 ± 2^{c}	26.8 ± 3.2^{b}	2.2 ± 0.4^{d}	55.6 ± 18.2^{a}	5.5 ± 0.8^{c}	13.2 ± 1.9^{b}

Different letters indicate significant differences within each soil property and depth section ($p < 0.05$) based on a one-way ANOVA. The letter 'a' denotes a significantly higher value with decreasing value for subsequent letters

BD Bulk Density, *OM* Organic Matter, *MBN* Microbial Biomass Nitrogen, *Ext*. Extractable

*The post-diversion marsh represents riverine mineral sediment from Mardi Gras Pass, a Mississippi River crevasse splay of 55 publications with 419 denitrification rates from different vegetated wetlands habitats reported that the presence of plants overall increased denitrification rates by ~ 55% (Alldred & Baines 2016). Conversely, the 2 cm of fine-grained mineral sediments on top of the marsh substrate (i.e., post-diversion substrate) can limit the diffusion rate of nitrate into the soil, therefore slowing the overall NO_3^- reduction rate. The vegetated marsh has statistically higher values than the post-diversion marsh for moisture content, organic matter, TC, TN, extractable DOC, and MBN, enhancing NO_3^- microbial respiration (Table 2). The trend of greater N reduction in marsh soils vs. benthic sediments has been previously observed in coastal Louisiana. One study found there is 2–3 times decrease in denitrification potential between vegetated marsh and benthic sediments (Rivera-MonRoy et al. 2013). Also, benthic N reduction rates were 2.7 times smaller in the subtidal sediments compared with the vegetated marsh in a more saline region of Barataria Basin (Vaccare et al. 2019) (Table 3).

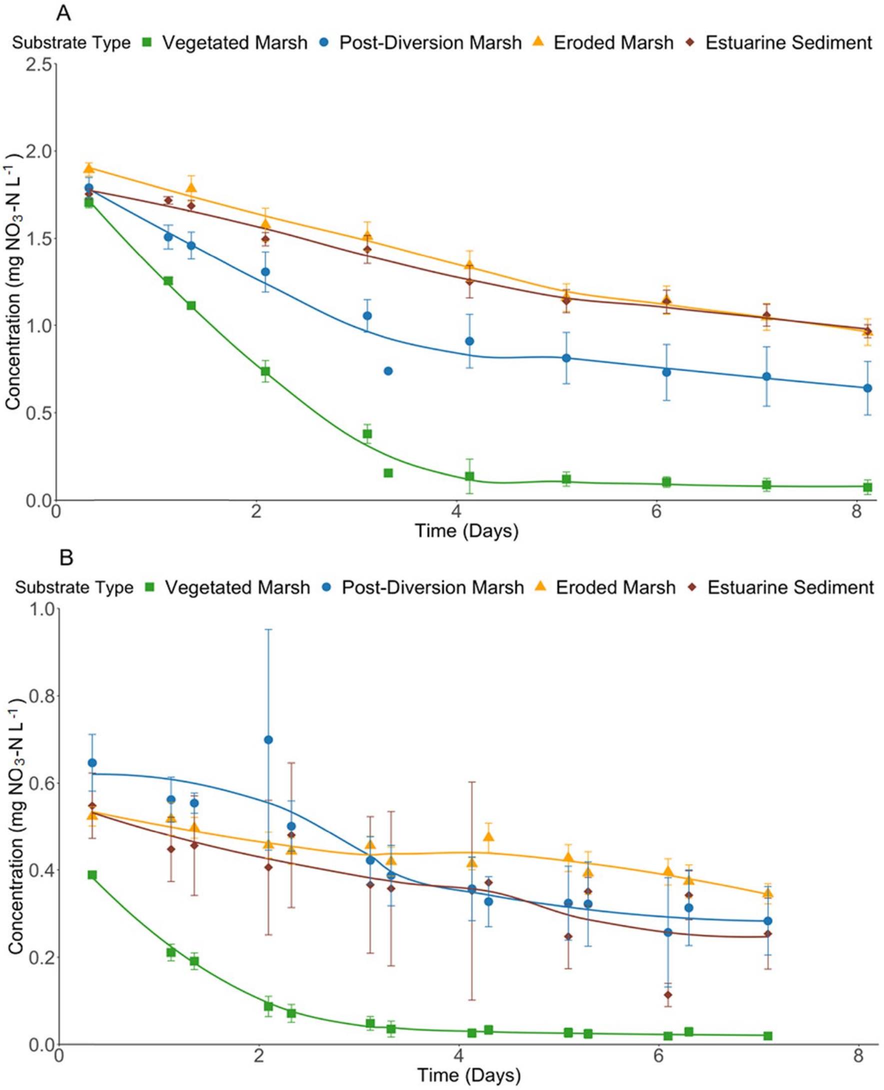
Barataria Basin is undergoing continuous erosion at the edge of the vegetated marsh platform. The interwoven root mat of the vegetated marsh is eroded by wave action, causing the marsh to subside into the bay. The remaining underlying layers of older marsh are then submerged (Vaccare et al. 2019) and this facie is what is referred to as the eroded marsh substrate. DOC values at 0–5 cm soil depth in both substrates were not significantly different (Table 2). However, there was a ~ 85.4% [81.7,89.2] decrease in NO_3^- reduction rate in the eroded submerged marsh compared to the vegetated organic marsh under low nitrate concentration range; and a 72.3% [70.3,74.3] decrease in NO_3^- reduction rate under the high range of nitrate concentration (Fig. 2). The soil physiochemical properties for each substrate zone indicate that the vegetated marsh and the eroded marsh zones are statistically similar for the top 0–5 cm for mean bulk density, TP, extractable NH_4^+, PO_4^- and DOC, representing their similar origin (Table 2). However, the vegetated marsh has significantly higher values for MBN (61.4 ± 9.4 mg kg^{-1}) compared to the eroded marsh (26.8 ± 3.2 mg kg^{-1}), which likely contributed to lower NO_3^- reduction in the eroded marsh substrate (Table 2). In addition, the mineral sediment which has settled on top of the eroded marsh organic soil may restrict and reduce the diffusion rate of NO_3^-, decreasing the rate of delivery of NO_3^- from the water column to the anaerobic soil layer.

In cores spiked with 0.5 mg NO_3-N L^{-1}, the average aerobic NO_3^- reduction rate for the vegetated marsh, post-diversion marsh, eroded marsh, and estuarine sediment decreased by 66.9% [63.0,70.9], 57.6% [50.6,64.5], 82.7% [79.3,86.2] and 33.8% [29.8,37.9], respectively, compared to cores spiked with 2.0 mg NO_3^- -N L^{-1}. Across all four substrates spiked with 2.0 mg NO_3-N L^{-1}, there was a relatively linear NO_3^- reduction rate from 300 mg NO_3-N m^{-2} (2.0 mg L^{-1}) down to 150 mg NO_3-N m^{-2} (1 mg L^{-1}) at which the slope (rate) decreases and becomes asymptotic (Fig. 3A). On the other hand, the linear NO_3^- reduction rate was observed from 75 mg NO_3-N m^{-2} (0.5 mg L^{-1}) down to 37.5 mg NO_3-N

Table 3 Average aerobic denitrification rates reported in previous studies along the Louisiana coast

Study area	Incubation temp	Salinity (PSU)	Nitrate reduction rate (mg N m^{-2} day^{-1})		References
			Marsh soil	Benthic sediment	
Barataria Basin, LA	20 °C	4.7	24.5 ± 7.06		Cheng & White 2022
Barataria Basin, LA	20 °C	0.1–0.4	47.5 ± 0.60	29.6 ± 2.08	Upreti et al. 2021
Barataria Basin, LA	20 °C	4.4	50.6 ± 3.89	19.7 ± 3.00	Bowes 2018
Barataria Basin, LA	21 °C	8.9	71.1 ± 2.7	13 ± 0.75	This study
Barataria Basin, LA	21 °C	11	29.3 ± 3.28	10.8 ± 0.62	Vaccare 2019
Barataria Basin, LA	25 °C	14	19.1 ± 1.80		Levine et al. 2017
Wax Lake Delta, LA	20 °C	0.1–0.4	117 ± 9.37	32.4 ± 0.30	Upreti et al. 2021
Wax Lake Delta, LA	22 °C	0.1–0.2	85.11	68.54	Li et al. 2020
Wax Lake Delta, LA	20 °C	0.2	27.1 ± 7.08		Hurst 2016

Fig. 3 Water column nitrate concentration over time (mean ± standard error; n = 4) for starting concentrations of **A** 2.00 mg NO_3-N L^{-1}, and **B** 0.5 mg NO_3-N L^{-1}. Incubations were run under an aerobic water column for vegetated marsh, future scenario post-diversion marsh, the submerged eroded marsh and the mineral estuarine mud sediment in Barataria Bay, LA



m^{-2} (0.25 mg L^{-1}) in the cores spiked with 0.5 mg NO_3-N L^{-1} (Fig. 3B). According to Fickian diffusion law, the speed of the solute diffusion is proportional to the concentration gradient, with the solute moving from regions of high to low concentration (Hussein 2007). Thus, NO_3^- reduction rates were faster when the water column NO_3^- concentrations were greater vs. lower, primarily due to the primary driver of Fick's first law, that being the concentration gradient.

Based on the diffusive transport equation (Eq. 1), diffusion for the 2.0 mg NO_3-N L^{-1} to 1 mg NO_3-N L^{-1} should be hypothetically 4 times faster than going from 0.50 mg NO_3-N L^{-1} down to 0.25 mg NO_3-N L^{-1}. In the transport equation (Eq. 1), J_D is the diffusion flux, φ is porosity, D_s the diffusion coefficient, C is the concentration, and x a distance term. Experimentally, our results demonstrated, on average, 3.17 ± 1.9 times faster nitrate flux rate in the higher vs. low concentration treatment, which suggests other factors playing a role in NO_3^- removal and diffusive equations should be used with caution in predictive models.

$$\left(J_D = -\varphi D_s \frac{dC}{dx}\right) \tag{1}$$

Allochthonous nitrate removal in louisiana coastal systems: management implications

Wetlands can remove NO_3^- from the water column through microbial respiratory denitrification or plant and algal biomass assimilation (Jiang et al. 2017). While there is conservatory biogeochemical pathways including assimilative NO_3^- reduction and dissimilatory NO_3^- reduction to ammonia, recent studies have found that these pathways are minimal in these coastal wetlands with the vast majority of NO_3^- reduce through the denitrification pathway (Upreti et al. 2021; Vaccare et al 2019). High soil % OM is positively associated with high denitrification rates in wetlands (Henry & Twilley 2014). Therefore, it has also been generally accepted that to remove the allochthonous NO_3^-, a high soil % OM is necessary to maintain wetland denitrification rates. However, a recent study found that younger, emerging natural marshes (<43 years) with low % OM content and higher BD, in Wax Lake Delta were capable of higher denitrification rates when compared to older degrading marshes (~100 s of years old) with higher % OM content in upper Barataria Basin. Thus, substantive denitrification can be sustained in wetlands even with low %OM at environmentally relevant NO_3^- concentrations (Upreti et al. 2021). Another study in northern Barataria Basin also reported higher denitrification rates in a dredged-created mineral marsh (~9 years old) compared to a more organic-rich natural marsh (Cheng & White 2022). These findings are related to the fact that in most cases, nitrate is the limiting factor in denitrification in the environment. Stoichiometrically, denitrification requires 5 mol of carbon to reduce 4 mol of nitrate–N (VanZomeren et al 2012; White et al 2019). Considering that nitrate is generally measured in a few ppm in the environment and carbon in wetland soils is measured in percent, it is far more likely that NO_3^- is the limiting factor for denitrification. Additionally, a study in the actively growing Wax Lake delta in Louisiana found that recently deposited mineral sediments with low TC and TN content could have high denitrification rate under high bedload shear stress as advection dominated over diffusion (Hurst et al. 2019). Another important factor affecting denitrification is the soil/sediment time of exposure to NO_3^-. Gardner and White (2010) demonstrated that the denitrifying enzyme activity in coastal wetland soils was well correlated with the surface water NO_3^- concentrations and length of exposure. The timing of when the river water is released can impact N removal because of temperature. Earlier in the spring, the Mississippi River water is colder and as such, lower rates have been reported with a drop in water temperature (Bowes et al. 2022). Therefore, other factors can play a dominant role in regulating denitrification, beyond the simple correlation with organic matter content.

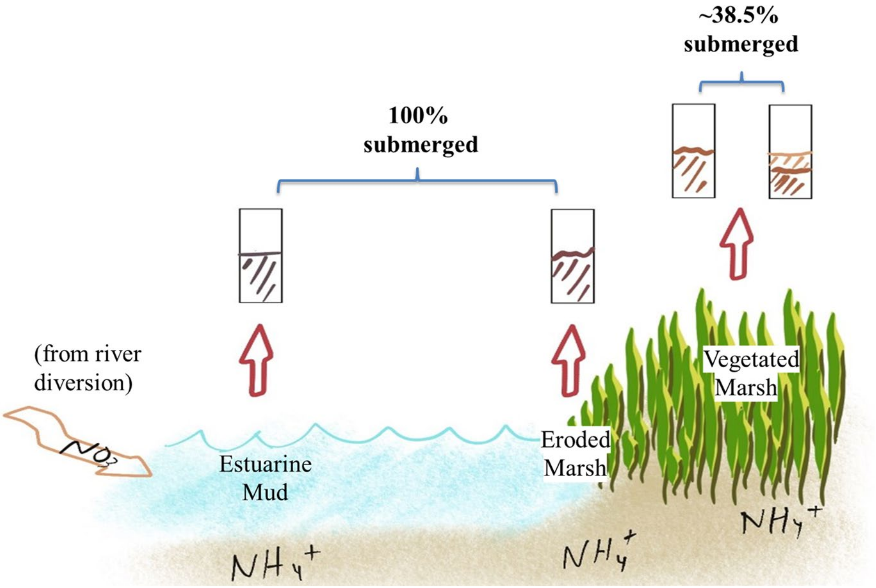
The post-diversion, future scenario treatment produced a potential NO_3^- reduction rate (i.e., 27.8 ± 4.5 mg NO_3-N m^{-2} day^{-1}) that is not significantly different from previous denitrification rates found in Barataria Basin for natural, organic marsh soils, such as 24.5 ± 7.1 (Cheng & White 2022), and 29.3 ± 3.3 mg NO_3-N m^{-2} d^{-1} (Vaccare et al. 2019) using the same methodology (Table 3). Also, this rate is not significantly different from denitrification rates found within the mudflats of Wax Lake Delta, at 27.1 ± 7.08 N m^{-2} d^{-1}. Most studies addressing N removal efficiency in coastal Louisiana have found that wastewater treatment wetlands, despite the high NO_3^- loading, maintain their nutrient sink characteristics even after 26–70 years of operation. Given these studies on treatment wetlands, this suggests a river diversion operated for about 1 month per year will not lead to reductions in denitrification potential over time (Day et al. 2018, 2019). Also, changing salinity has been found to impact denitrification rates (Marks et al 2016). However, it has been shown that while salinity pulses penetrate the coastal wetland soil relatively quickly (McKee et al 2016), a recent study has shown the reverse (surface freshening over saline porewater) is not as easily reversed (Feder and White 2024).

In order to better understand the impact of large-scale, river reconnection and restoration strategies on the wetland biogeochemical processing of nitrate, it is not enough to know the NO_3^- reduction rates across the coastal basin. The Water Institute of the Gulf and Louisiana CPRA have created an Integrated Biophysical Model to perform long-term simulations of essential ecosystem components (hydrodynamics, morphodynamics, vegetation, and nutrient dynamics) in coastal/deltaic systems to predict response to global change scenarios and restoration techniques (Baustian et al. 2018). Using the model, simulations were run at CRMS 0224 (proximal to our study site) to predict water level change in model years 2020, 2030, 2040, and 2050, which was simulated to be one, ten, twenty and thirty years after river diversion operation, respectively. The water level output predicts that there will not be a significant change in water level once the sediment diversion is operated at most sites beyond a few km distance from the inflow (Meselhe et al. 2016).

These model results suggest that while Barataria Basin will receive nutrients and sediment from the diversion, there is little to no water level change and much of the river water will not be driven up on to the vegetated marsh platform during the period of operation (Meselhe et al. 2016). Consequently, the spring-neap tidal cycle and the winds will ultimately continue to control the hydroperiod in the microtidal lower basin (Li et al. 2011). One study found that vegetated marsh platforms in Barataria Bay are flooded only 31–46% of the time (Valentine & Mariotti 2019). This finding indicates that although the vegetated marsh platform has the highest NO_3^- reduction rate, it will also have limited contact with the high nitrate water distributed from the river. The eroded marsh and estuarine sediment substrates, both of which are submerged 100% of the time and will be in continuous contact with the surface water, provides the greatest opportunity to reduce NO_3^- in the water column (Fig. 4).

Given these constraints, the relative contribution of each soil/sediment for potential nitrate removal was simply calculated by multiplying the associated NO_3^- reduction rate by the % submergence time. Although the eroded marsh NO_3^- reduction rate is ~ 70.9% of the post-diversion marsh, and the estuarine sediment NO_3^- reduction rate is ~ 46.8% of the post-diversion marsh, they could potentially provide up to 1.84 × and 1.21 × more NO_3^- reduction than the post-diversion marsh due to increased contact time (rate x contact time), on an area equivalent basis, respectively. This hydrologic-link result occurs because both (i.e., eroded marsh and estuarine sediment) are flooded 100% of the time, while the post-diversion marsh will be submerged ~ 38.5% of the time. However, these data would need to be scaled to the areal coverage within the diversion area also considering hydraulic loading rate and substrate limitation. It is clear the coastal marsh ecosystem, on an equivalent area basis, can still provide the ecosystem service of water quality improvement through substantial N microbial respiration in subtidal substrates, ameliorating water quality and protecting Louisiana's coastal fisheries. This example serves to demonstrate that it is necessary to not only address NO_3^- concentration level, NO_3^- reduction rates, soil types and organic matter content, but is also necessary to link inundation period to more accurately predict the overall spatial and temporal NO_3^- reduction rate. This linkage is transferable globally, as recent restoration trends have focused on restoring freshwater flows to

Fig. 4 The organic-rich vegetated marsh substrate is flooded ~ 38.5% of the time, while the eroded marsh substrate and the estuarine sediment substrate are 100% of the time submerged



several other coastal ecosystems (Herbert et al. 2011; Zedler 2017; Chua et al. 2024).

Conclusion

This study examined differences in nitrate removal rates between intertidal vegetated organic-rich marsh soil, subtidal eroded organic-rich marsh soil, and subtidal estuarine sediment in mid-Barataria Basin as well as a future scenario where the marsh soil is capped by mineral river sediment after river reconnection. Soil from vegetated wetlands expressed the highest NO_3^- reduction rates (i.e., 71.1 ± 2.7 mg NO_3-N m^{-2} d^{-1}) under higher inorganic N concentrations (NO_3^- ~ 2.0–0.5 mg N L^{-1}) and decreased ~ 66.9% [63.0,70.9] under lower N concentration (NO_3^- ~ 0.5–0.05 mg N L^{-1}) driven by difference in concentration gradient. In contrast, overall lower NO_3^- reduction rates (i.e., < 19.7 ± 1.2 mg NO_3-N m^{-2} d^{-1}) were observed in the subtidal sediments. The research found that there was a ~ 55.3 ± 7.9% decrease of nitrate removal rate in the organic-marsh soil once mineral riverine sediments cap the organic soil. These findings can assist in better predicting nutrient responses to river input in support of the basin-wide integrated biophysical model concerning operating the sediment diversions, which is a critical restoration type in the Louisiana Coastal Master Plan. Therefore, modeling efforts should incorporate spatial and temporal variability when seeking to predict nutrient loading and fate. Further, these findings can inform effective nutrient management strategies in river deltas around the world. All of the world's great rivers contain elevated nitrate concentrations, which subsequently flow into deltaic coastal areas and this research clearly demonstrates that interception by wetlands and estuarine sediments can significantly reduce N loading to coastal waters.

Acknowledgements The authors would like to acknowledge Eddie Weeks, Jacob Cheng and James Anderson for field assistance. This study was supported by the U.S. Department of the Treasury through the Louisiana Coastal Protection and Restoration Authority's Center of Excellence Research Grants Program under the Resources and Ecosystems Sustainability, Tourist Opportunities, and Revived Economies of the Gulf Coast States Act of 2012 (RESTORE Act) (Award No. 1 RCEGR260007-01-00). The statements, findings, conclusions, and recommendations are those of the authors and do not necessarily reflect the views of the Department of the Treasury.

Author contributions Both Authors contributed to the study conception and design. Data collection and analyses were performed by MMP. The first draft of the manuscript was written by MMP and JRW commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Funding This study was supported by the U.S. Department of the Treasury through the Louisiana Coastal Protection and Restoration Authority's Center of Excellence Research Grants Program under the Resources and Ecosystems Sustainability, Tourist Opportunities, and Revived Economies of the Gulf Coast States Act of 2012 (RESTORE Act) (Award No. 1 RCEGR260007-01-00).

Data availability The datasets generated during the current study are not publicly available due to formatting issues but are available from the corresponding author on reasonable request. All data will be provided upon request to the Corresponding Author by email.

Declarations

Conflict of interest Mercedes M. Pinzon and John R. White declare they have no relevant financial interests to disclose.

References

Alldred M, Baines SB (2016) Effects of wetland plants on denitrification rates: a meta analysis. Ecol Appl 26(3):676–685

Andersen JM (1976) An ignition method for determination of total phosphorus in lake sediments. Water Res 10:329–331

Baustian MM, Meselhe E, Jung H, Sadid K, Duke-Sylvester SM, Visser JM, Allison MA, Moss LC, Ramatchandirane C, Sebastiaan van Maren D, Jeuken M, Bargu S (2018) Development of an Integrated Biophysical Model to represent morphological and ecological processes in a changing deltaic and coastal ecosystem. Environ Model Softw 109:402–419

Bevington AE, Twilley RR, Sasser CE, Holm GO (2017) Contribution of river floods, hurricanes, and cold fronts to elevation change in a deltaic floodplain, northern Gulf of Mexico, USA. Estuar Coast Shelf Sci 191:188–200

Blum MD, Roberts HH (2009) Drowning of the Mississippi Delta due to insufficient sediment supply and global sea-level rise. Nat Geosci 2(7):488–491

Bowes K (2018) From the River to the Gulf: An Investigation of Biogeochemical Cycling in Wetland Soils and Coastal Shelf Sediments (master's thesis). Louisiana State University, Louisiana, USA

Bowes K, White JR, Maiti K, Meselhe E (2022) Surface water temperature impacts on denitrification: implications for river reconnection. Sci Total Environ 828:154397

Brookes PC, Landman A, Pruden G, Jenkinson D (1985) Chloroform fumigation and the release of soil nitrogen: a rapid direct extraction method to measure microbial biomass nitrogen in soil. Soil Biol Biochem 17(6):837–842

Burgin AJ, Hamilton SK (2007) Have we overemphasized the role of denitrification in aquatic ecosystems? A review of nitrate removal pathways. Front Ecol Environ 5(2):89–96

Byrnes MR, Britsch LD, Berlinghoff JL, Johnson R, Khalil S (2019) Recent subsidence rates for Barataria Basin, Louisiana. Geo-Mar Lett 39(4):265

Cheng JZ, White JR (2022) Dredge-material created coastal marshes are more effective at improving water quality than natural marshes in early-stage development. Ecol Eng. https://doi.org/10.1016/j.ecoleng.2022.106814

Chua X, Yang Y, Kondolf G, Oeurng C, Sok T, Zhang S, Xixi L (2024) Can restoring water and sediment fluxes across a mega-dam cascade alleviate a sinking river delta? Sci Adv 10:eadn9731

Church, J., Clark, P., Cazenave, A., Gregory, J., Jevrejeva, S., Levermann, A., et al., (2013) Intergovernmental Panel on Climate Change. (2014). Climate Change 2013 - The Physical Science Basis: Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

Coastal Protection and Restoration Authority of Louisiana (CPRA). (2017). Louisiana's comprehensive master plan for a sustainable coast. Retrieved from. http://coastal.la.gov/ wp-content/uploads/2017/04/2017-Coastal-Master-Plan_Web-Book_CFinal-with-Effective-Date-06092017.pdf.

Coastal Protection and Restoration Authority (CPRA). (n.d.). Coastwide Reference Monitoring System. Retrieved May 14, 2024, from https://www.lacoast.gov/crms_viewer/Map/ CRMSViewer

Couvillion, B., Beck, H., Schoolmaster, D., Fischer, M., (2017). Land area change in coastal Louisiana (1932 to 2016). U.S. Department of the Interior, U.S. Geological Survey. https://pubs.usgs.gov/sim/3381/sim3381_pamphlet.pdf

Das A, Justic D, Inoue M, Hoda A, Huang H, Park D (2012) Impact of Mississippi River diversions on salinity gradients in a deltaic Louisiana estuary: ecological and management implications. Estuarine Coast Shelf Sci 111:17–26

Day JW, DeLaune RD, White JR, Lane RR, Hunter RG, Shaffer GP (2018) Can denitrification explain coastal wetland loss: a review of case studies in the Mississippi Delta and New England. Estuar Coast Shelf Sci 213:294–304

Day JW, Hunter RG, Lane RR, Shaffer GP, Day JN (2019) Long-term assimilation wetlands in coastal Louisiana: review of monitoring data and management. Ecol Eng 137:7–20

Day JW, Conner WH, DeLaune RD, Hopkinson CS, Hunter RG, Shaffer GP, Kandalepas D, Keim RF, Kemp GP, Lane RR, Rivera-Monroy VH, Sasser CE, White JR, Vargas-Lopez I (2021) A review of 50 years of study of hydrology, wetland dynamics, aquatic metabolism, water quality and trophic status, and nutrient biogeochemistry in the Barataria Basin, Mississippi Delta – system functioning, human impacts and restoration approaches. Water 13:642

DeLaune RD, White JR (2012) Will coastal wetlands continue to sequester carbon in response to an increase in global sea level? A case study of the rapidly subsiding Mississippi River Deltaic Plain. Clim Chang 110(1–2):297–314

Feder RF, White JR (2024) Impact of freshwater river reconnection on porewater salinity and ammonium availability in coastal brackish marsh soils. Sci Total Environ 926:172131

Georgiou, I. Y., FitzGerald, D. M., Stone, G. W. (2005). The Impact of Physical Processes along the Louisiana Coast. Journal of Coastal Research, 72–89.

Henry KM, Twilley RR (2014) Nutrient biogeochemistry during the early stages of delta development in the Mississippi River Deltaic Plain. Ecosystems 17(2):327–343

Herbert DA, Perry WB, Cosby BJ, Fourqurean JW (2011) Projected reorganization of Florida Bay seagrass communities in response to the increased freshwater inflow of Everglades restoration. Estuaries Coasts 34:973–992

Hinshaw SE, Tatariw C, Flournoy N, Kleinhuizen A, Taylor C, Sobecky PA, Mortazavi B (2017) Vegetation loss decreases salt marsh denitrification capacity: implications for marsh erosion. Environ Sci Technol 51(15):8245–8253

Hurst N, White JR, Baustian J (2016) Nitrate reduction in a hydrologically restored bottomland hardwood forest in the Mississippi River watershed, Northern Louisiana. Soil Sci Soc Am J 80:1698–1705

Hurst NR, White JR, Xu K, Ren M (2019) Nitrate reduction rates in sediments experiencing turbulent flow conditions. Ecol Eng 128:33–38

Hussein E (2007) Chapter four: transport. In: Radiation mechanics: principles and applications in nuclear engineering. Elsevier, pp 247–310. https://doi.org/10.1016/B978-008045053-7/50005-7

Jankowski KL, Törnqvist TE, Fernandes AM (2017) Vulnerability of Louisiana's coastal wetlands to present-day rates of relative sealevel rise. Nat Commun 8:14792

Jiang Y, Li Y, Zhang Y, Zhang X (2017) Effects of HRT on the efficiency of denitrification and carbon source release in constructed wetland filled with bark. Water Sci Technol 75(12):2908–2915

Jung H, Nuttle W, Baustian MM, Carruthers T (2023) Influence of increased freshwater inflow on nitrogen and phosphorus budgets in a dynamic subtropical estuary, Barataria Basin, Louisiana. Water 15(11):1974

Levine BM, White JR, DeLaune RD (2017) Impacts of the long-term presence of buried crude oil on salt marsh soil denitrification in Barataria Bay, Louisiana. Ecol Eng 99:454–461. https://doi.org/10.1016/j.ecoleng.2016.11.017

Li, C., White, J. R., Chen Changsheng, Lin, H., Weeks, E., Galvan, K., Bargu, S. (2011). Summertime tidal flushing of Barataria Bay; transport of water and suspended sediments. J Geophys Res 116(C4)

Li S, Christensen A, Twilley RR (2020) Benthic fluxes of dissolved oxygen and nutrients across hydrogeomorphic zones in a coastal deltaic floodplain within the Mississippi River delta plain. Biogeochemistry 149(2):115–140. https://doi.org/10.1007/s10533-020-00665-8

Louisiana Department of Natural Resources Coastal Engineering Division (LDNR), (2008). Coastwide Reference Monitoring System (CRMS) Survey Report: Site Number CRMS 0224. Retrieved from: https://www.lacoast.gov/crms_viewer/crms_public_data/survey_reports/CRMS0224SurveyReport.pdf.

Marks BW, Chambers LG, White JR (2016) Effects of fluctuating salinity on potential denitrification in coastal wetland soils and sediments. Soil Sci Soc Am J 80:516–526

McKee M, White JR, Putnam Duhon LA (2016) Simulated Storm Surge Effects on Freshwater Coastal Wetland Soil Porewater Salinity and Extractable Ammonium Levels: Implications for Marsh Recovery after Storm Surge. Estuar Coast Shelf Sci 181:338–344

Mendelssohn, I., Andersen, G., Baltz, D., Caffey, R., Carman, K., Fleeger, J. et al. (2012). Oil Impacts on Coastal Wetlands: Implications for the Mississippi River Delta Ecosystem after the Deepwater Horizon Oil Spill. (2012). BioScience, 62(6), 562–574.

Meselhe, E. A., Baustian, M. M., Sadid, K. M., Xing, F., Costanza, K., Allison, M. A., Jarrell, E., Richards, C. P., Pahl, J. (2016). Morphologic and ecologic analysis of a proposed network of Mississippi River sediment diversions. Ocean Sciences Meeting, 2016, @Abstract EC41A-04.

Mitsch W, Day J, Zhang L, Lane R (2005) Nitrate-nitrogen retention in wetlands in the Mississippi River basin. Ecol Eng 24(4):267–278

Morton RA, Bernier JC, Barras JA, Ferina NF (2005) Historical subsidence and wetland loss in the Mississippi delta plain. Transactions - Gulf Coast Association of Geological Societies 55:555–571

Pellerin BA, Bergamaschi BA, Gilliom RJ, Crawford CG, Saraceno JF, Frederick CP, Downing BD, Murphy JC (2014) Mississippi River nitrate loads from high frequency sensor measurements and regression-based load estimation. Am Chem Soc 48(21):12612–12619

Peyronnin NS, Caffey RH, Cowan JH, Justic D, Kolker AS, Laska SB, McCorquodale A, Melancon E, Nyman JA, Twilley RR, Visser JM, White JR, Wilkins JG (2017) Optimizing sediment diversion operations: working group recommendations for integrating complex ecological and social landscape interactions. Water 9(6):368

Rabalais NN, Turner RE, Scavia D (2002) Beyond science into policy: Gulf of Mexico hypoxia and the Mississippi River. Bioscience 52:129–142

Reddy KR, DeLaune RD (2008) Biogeochemistry of Wetlands: Science and Applications, 1st edn. CRC Press

Reinhardt M, Muller B, Gachter R, Wehrli B (2006) Nitrogen removal in a small constructed wetland: an isotope mass balance approach. Environ Sci Technol 40:3313–3319

Rivera-Monroy VH, Branoff B, Meselhe E, McCorquodale A, Dortch M, Steyer GD, Visser J, Wang H (2013) Landscape-Level Estimation of Nitrogen Removal in Coastal Louisiana Wetlands: Potential Sinks under Different Restoration Scenarios. Journal of Coastal Research, Special Issue 67:75–87

Roy E, White JR (2012) Nitrate flux into the sediments of a shallow oligohaline estuary during large flood pulses of Mississippi River water. J Environ Qual 41(5):1549

Sapkota Y, White JR (2021) Long-term fate of rapidly eroding carbon stock soil profiles in coastal wetlands. Sci Total Environ 753:141913

Shaw JB, Estep JD, Whaling AR, Sanks KM, Edmonds DA (2018) Measuring subaqueous progradation of the wax Lake Delta with a model of flow direction divergence. Earth Surf Dyn 6:1155–1168

Smith M, Tiedje JM (1979) Phases of denitrification following oxygen depletion in soil. Soil Biol Biochem 11(3):261–267

Sparks, D. (ed.). (1996). Methods of soil analysis. Part 3. Chemical Methods. Book Set 5. Madison: SSSA

Steinmuller, H., Dittmer, K., White, J., & Chambers, L. (2018). Understanding the fate of soil organic matter in submerging coastal wetlands soils: A microcosm approach. Geoderma. 337(4).

Tufillaro N, Piazza BP, Reddy S, Baustian J, Sousa D, Grötsch P, Lalović I, De Moitié S, Zurita O (2024) Linking optical data and nitrates in the Lower Mississippi River to enable satellite-based monitoring of nutrient reduction goals. Ecohydrol. https://doi.org/10.1002/eco.2631

U.S. EPA, (1993). Methods for Determination of Inorganic Substances in Environmental Samples. EPA/600/R-93/100 (Washington D.C).

Upreti, K., (2019). Evaluating Seasonal Nutrient Fluxes in Emerging and Eroding Wetlands of the Louisiana Delta Plain. Doctoral Dissertation. Louisiana State University, Louisiana, USA.

Upreti K, Rivera-Monroy VH, Maiti K, Giblin A, Geaghan JP (2021) Emerging wetlands from river diversions can sustain high denitrification rates in a coastal delta. J Geophys Res Biogeosciences. https://doi.org/10.1029/2020JG006217

Upreti K, Rivera-Monroy VH, Maiti K, Giblen A, Castaneda-Maya E (2022) Dissimilatory nitrate reduction to ammonium (DNRA) is marginal relative to denitrification in emerging-eroding wetlands in a subtropical oligohaline and eutrophic coastal delta. Sci Total Environ 815:152942

Vaccare J, Meselhe E, White JR (2019) The denitrification potential of eroding wetlands in Barataria Bay, LA, USA: implications for river reconnection. Sci Total Environ 686:529–537

Valentine K, Mariotti G (2019) Wind-driven water level fluctuations drive marsh edge erosion variability in microtidal coastal bays. Cont Shelf Res 176:76–89

Vance E, Brookes P, Jenkinson D (1987) An extraction method for measuring soil microbial biomass C. Soil Biol Biochem 19(6):703–707

VanZomeren CM, White JR, DeLaune RD (2012) Fate of nitrate in vegetated brackish coastal marsh. Soil Sci Soc Am J 76:1919–1927

Wang H, Steyer GD, Couvillion BR, Beck HJ, Rybczyk JM, Rivera-Monroy VH et al (2017) Predicting landscape effects of Mississippi River diversions on soil organic carbon sequestration. Ecosphere. https://doi.org/10.1002/ecs2.1984

White JR, DeLaune RD, Justic D, Day JW, Pahl J, Lane RR, Boynton WR, Twilley RR (2019) Consequences of Mississippi River diversions on nutrient dynamics of coastal wetland soils and estuarine sediments: a review. Estuar Coast Shelf Sci 224:209–216

White JR, Reddy K (2000) Influence of phosphorus loading on organic nitrogen mineralization of everglades soils. Soil Sci Soc Am J 64(4):1525

Wood SE, White JR, Armbruster CK (2017) Microbial processes linked to soil organic matter in a restored and natural coastal wetland in Barataria Bay, Louisiana. Ecol Eng 106(Part A):507–514

Zedler JB (2017) What's new in adaptive management and restoration of coasts and estuaries? Estuaries Coasts 40:1–21