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Influence of a Freshwater River Diversion on Sedimentation and Phosphorus Status in a Wetland Receiving Basin

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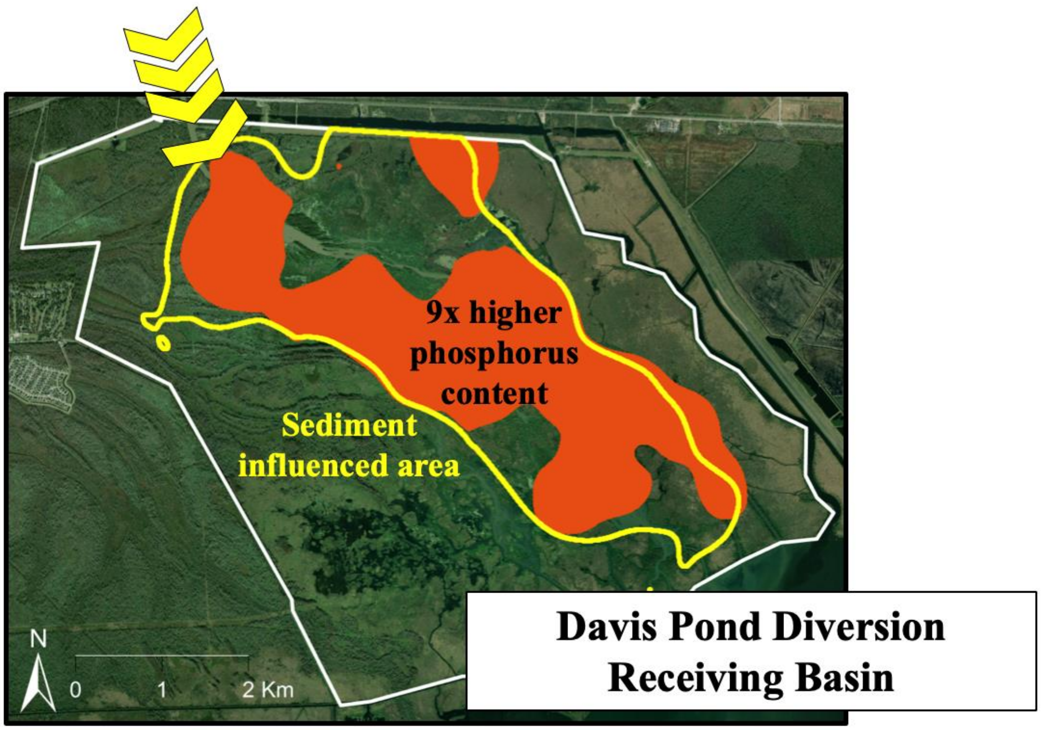
Abstract

Mississippi River Delta wetlands were isolated from river influence due to leveeing in the 1900's. Surface water diversions were primarily designed to manage salinity and maintain marsh vegetation by reintroducing Mississippi River water and nutrients into adjacent wetlands. Phosphorus (P) is a major limiting nutrient that can control productivity, but in excess can contribute to wetland eutrophic conditions and water quality degradation. Most wetland soil characterization assessments consider soil total P, however, this parameter alone cannot describe P bioavailability due to difference in organic and inorganic forms. A soil characterization of the Davis Pond diversion was done in 2007, before full-scale operation began, and in 2018 after 11 years of river loading. The top 10 cm of soil from 140 stations each year were analyzed for physiochemical properties and both organic and inorganic P forms. Mineral content is used to delineate areas of river diversion influence and compare P stocks between hydrologically isolated marsh areas and where effective river diversion reconnection took place. The river diversion resulted in a nearly 100% increase in soil mineral content and 58% increase in bulk density. The dominant source of soil P has changed from organic P to inorganic P in 29% of the wetland area, significantly associated with mineral content of the soil. Inorganic P stocks in diversion influenced areas are 9 times higher than those which remained isolated from riverine materials. The study showed that long-term addition of mineral sediments and inorganic P did not lead to deleterious effects in the wetland. This is the first study in the Mississippi River delta

to spatially track river reconnection driven wetland P dynamics and this study can provide valuable information for predictive models for sediment diversions for coastal restoration.



1. Introduction

Many riparian wetlands serve as a natural buffer situated between anthropogenic-influenced rivers and natural estuarine systems (Gilliam, 1994; Blinn and Kilgore, 2001; Ducey et al., 2015). Wetland areas act as a transformer or sink for nutrients due to various biotic and abiotic processes that remove nitrogen (N) and phosphorus (P) from the water column and improve water quality (Zedler & Kercher, 2005). However, construction of continuous levees down the length of the Lower Mississippi River over the past century has led to sediment and nutrient deprivation in Louisiana’s once connected riparian wetlands (Day et al. 2000, 2018). Wide spread coastal land loss is confounded by increasing salinity gradients and flooding, driven by sea level rise, which also threaten important fisheries (Chesney et al,, 2000; Cowan et al., 2014) and wetland vegetation in Barataria Basin (Delaune & White, 2012). The future of the Mississippi River Delta depends on timely implementation of restoration projects by the state at varying spatial scales (Day et al., 2016). One such project type is the large-scale sediment diversion, still in planning and permitting stages, that will transport significant volumes of water and associated mineral sediment from the Mississippi River into Louisiana coastal wetlands to aid in building land in some locations and slowing land loss in others (CPRA, 2017). Diversion projects from past restoration efforts, like the Davis Pond diversion, provide a small number of real-life analogs for sediment diversions and can give some insight of how best to implement restoration of the Louisiana coast. The Davis Pond diversion diverts an average of 45 m^3s^{-1} water from the Mississippi River through a diversion channel and empties into a freshwater wetland which connects to the large coastal Barataria Basin at the southern end (McAlpin et al. 2008). This river diversion was designed primarily to deliver freshwater and nutrients to the wetland and coastal basin to control salinity. Operation of the diversion did not intend to deliver other needed materials such as inorganic sediment (Barras et al. 2003, Snedden et al. 2007), although others have reported locally significant land building in several freshwater diversion systems (Lopez et al., 2014; Day et al., 2016; Keogh et al., 2019).

Concentrations of nutrients such as N and P have nearly doubled in the Mississippi River since the early 20th century (Goolsby et al., 2000; Evers et al., 1992), and some evidence suggests that river reconnection could encourage wetland eutrophication in response to nutrient enrichment in the receiving basins and reduce wetland resilience (Darby and Turner, 2008; Swarzenski et al., 2008, Turner et al. 2019). In addition to water and dissolved nutrients, freshwater diversions are, in fact, transporting sediment from the river into their respective basins (Keogh et al., 2019), although the rate of sedimentation throughout the entire Davis Pond receiving wetland is unclear (DeLaune et al., 2008). Application of fine-grained inorganic sediments increases the density of wetland soils (Anisfeld and Hill, 2012) and denser soils may be more stable in contrast with highly organic soils that are more susceptible to erosion (Howes et al. 2010, Sapkota and White, 2019). However, riverine sediments also contain dissolved and particulate nutrients that can alter the biogeochemical processes within the system. DeLaune et al. (2016) found that established marshes in an area receiving material from the Atchafalaya River, called the Wax Lake Sub Delta, have enhanced rates of plant production and sediment carbon (C) accumulation. In that case, the wetland serves as a means to remove or store nutrients, however, under highly loaded conditions nutrient enrichment can lead to deleterious effects on wetland soils. Wetland eutrophication is an increasingly common end result of nutrient pollution which can reduce the quality of ecosystem functions provided by wetland habitats (Zedler & Kercher 2005; Bargu et al, 2019).

Phosphorus is commonly a limiting nutrient, especially in wetland systems with low mineral input (Reddy et al, 1995). Abundance of P in relation to C and N is a main determinant in the net primary production of any aquatic system. Total P is generally measured in wetland monitoring studies, and as such, understanding P availability contributes to interpreting overall health and functioning of a wetland system. Deltaic wetlands have P associated with both riverine sediment and highly organic peat soil (Reddy and DeLaune 2008). Organic P (OP) forms by the conversion of dissolved inorganic P immobilized into organic matter (OM) by either microbes or plants. Up to 80% of bioavailable P removal can be performed by plants depending on the productivity of the system (Reddy et al. 1995; Reddy et al., 1998). Inorganic P (IP) in a river system is present either in the form of dissolved inorganic phosphate (PO_4^-) in river water or more substantially as ortho-phosphate minerals associated with iron (Fe) and aluminum (Al) (Zhang et al., 2012).

A critical baseline wetland soil study in the Davis Pond diversion wetland in 2007 by Kral et al. (2012) did not demonstrate a strong spatial component driving the distribution of total P before full river reconnection began, likely due to the low volume operation of the diversion up to that point. Kral et al. (2012) found that total P distribution was weakly correlated to any other organic or inorganic soil parameter and this has been shown by others (Rivero et al., 2007; Tipping et al., 2016; Adams et al., 2018). Analysis of inorganic and organic soil P fractions may help identify drivers of spatial variability in P availability and can be used to delineate areas of river influence in wetlands. In the years following initial sampling, the wetland has accreted 0.59 to 1.03 cm yr^{-1} of new soil each year (DeLaune et al., 2013). A resampling effort of the newly accreted 0-10 cm soil layer was conducted in 2018 to document the ~11-year record of hydrologic restoration on the soils of this coastal, deltaic freshwater wetland. We hypothesize that the relative contribution of Mississippi River sediments and associated metal bound P to the Davis Pond wetland has significantly increased soil P stocks over time and should provide a spatial indication of river influence. These changes may be linked to increased nutrient availability and ecosystem productivity.

2. Materials and Methods

Study Site

The Davis Pond Mississippi River Diversion discharges into a fresh water wetland ponding area located in northern Barataria Basin, Louisiana, USA (Figure 1). The diversion structure is approximately 19 km upstream the Mississippi River and on the opposite bank from the city of New Orleans. At maximum flow, the diversion delivers up to 300 m^3 s^{-1} of water into the 37.6

km^2 ponding area, however operation of the diversion is limited by Mississippi River stage because water is redirected passively by 14 ft box culverts. Since 2007, average diversion discharge was 58 m^3s^{-1} with a median discharge of 36 m^3s^{-1}, and diversion operation was negligible on an average of 44 days each year (USGS, 2018). The ponding area is surrounded by levees on the northern, western and eastern sides directing flow to the southeastern boundary. That boundary is composed of a low rock weir cut with several outflow canals, allowing water to enter into Lake Cataouache and then Lake Salvador eventually entering Barataria Bay further south. The Davis Pond Freshwater diversion was built in 2002, however, water flow issues prevented operation of the diversion at full capacity before 2009, once modifications to the flow path and upgrades to the levees were completed (Chuck Villarubia, per. comm).

2.1. Sampling

The sampling locations were identified using spatial simulated annealing (SSA) (van Groenigen et al., 1999) approach (Kral et al., 2012). At these locations push core samples of the top 10 cm of soil were collected at 140 stations in May through July 2007 and May through June 2018. Dried, ground archived soils samples from 2007 were used for phosphorus fractionation and metals analyses.

2.2. Soil physiochemical properties

Soil moisture and bulk density (BD) were determined by drying field moist sub-samples at 70°C until at constant weight. Total P was separated into total inorganic P (IP) and total organic P (OP). For IP extraction, 0.5 g of dried ground samples were shaken for 3 hours on longitudinal shaker with 25 mL of 1M HCl, followed by 10-minute centrifugation at 4000 g. The supernatant fluid was filtered through a 0.45-µm membrane filter and stored at 4ºC (Reddy 1998; Reddy and DeLaune, 2008). Dried ground sub samples were combusted at 550°C for 4 hours in a muffle furnace for loss-on-ignition method of organic matter (OM) (Sparks, 1996) and ash content determination, from here ash content is referred to as mineral content (MC). After combustions, ashed samples were digested with 6 N HCl, then filtered for total P analysis (Andersen 1976). Total P and total IP were determined with an AQ2 Automated Discrete Analyzer (SEAL Analytical, West Sussex, England), using US EPA ascorbic acid automated colorimetric procedure (Method 353.2; US EPA). Total organic P was determined by the difference (TP-IP). Total metals (Fe, Al, Ca, and Mg) were analyzed on 40 randomly selected 2018 samples from the 6 N HCl digestion with inductively coupled plasma atomic emission spectroscopy (Hitachi High-Technologies America, Schaumburg, IL, USA).

2.3. Statistical analysis workflow

This study used geostatistical means to identify spatial and temporal patterns in the distribution of soil phosphorus fractions in the Davis Pond wetland. The work flow consisted of (1) univariate analysis of between year measured site P data; (2) visualization of patterns in spatial distribution of soil P fractions across the wetland and over time by modeling soil variables through spatial auto-correlation; (3) creating a new dataset from modeled data in order to make univariate and multivariate comparisons between years and between distinct areas of the wetland.

2.4. Univariate Analysis

Paired t-tests were conducted in order to evaluate how soil characteristics changed at each site between sampling years. *P* values < 0.05 were considered significant at an α of 0.05.

2.5. *Geostatistical Analysis*

A geostatistical approach was used to describe the spatial variability in soil characteristics across the Davis Pond wetland from archived 2007 data and analysis from 2018. This analysis assumes that the observed variation in soil properties is a result of a random function that determines the value of a variable of interest, Z, at any given region. A linear model of describing Z at location x is formulated as $Z(x) = \mu + \varepsilon$ ($\mathbf{x}$) where μ is the unknown and invariant mean and $\boldsymbol{\varepsilon}(\mathbf{x})$is the stochastic spatially autocorrelated component and the stochastic spatially uncorrelated variation drawn from a normal distribution with mean zero.

Assuming the assumption of first order stationarity, then models of the spatially dependent variance are the same over the entire sampled area and are represented with semivariance ($\gamma(\mathbf{h})$) (Eq. 1) in variable Z at separating, or lag, vector **h**.

$$\gamma(\mathbf{h}) = \frac{1}{2} E \{[z(x) - Z(x + \mathbf{h})]^2\} \quad \text{(Eq. 1)}$$

Construction of experimental variograms and fitting of variogram models was performed using R (R Core Team, 2014) within RStudio (RStudio Team, 2015). Classical Matheron's method of moments variogram estimator (Kral et al., 2012) was used for least squares fitting, weighted by the number of sites separated by vector **h.** Authorized models such as the exponential and spherical models were used for variogram fitting (Grunwald et al, 2006; Webster and Oliver, 2007). Final model selection was based on minimization of weighted sum of squares.

Variogram models were used for spatial autocorrelation by ordinary kriging in R with global neighborhood (Ribeiro and Diggle, 2001) to produce predictions of physiochemical properties for ~660,000 100 m^2 grid cells across the study area. Finals map layouts were produced in ArcMap 10.6.1 (ESRI, 2018).

2.6. Soil Nutrient Stock Estimations

Soil nutrient stocks in the 0-10 cm soil layers from each sampling year were estimated using the methodology outlined by Veronesi et al. (2014). Soil total P, IP and OP depth-integrated volumetric content (g P m^{-2}), from here referred to as stocks, and mineral density (g cm^{-3}) was calculated using spatially autocorrelated surfaces with the following equations:

$$\text{Mineral Density} = MC_i \bullet BD_i \qquad \text{(Eq.2)}$$

$$\text{Organic Matter Density} = OM_i \bullet BD_i \qquad \text{(Eq.3)}$$

$$\text{P Stock} = TP_i \bullet BD_i \bullet d \qquad \text{(Eq.4)}$$

$$\text{IP Stock} = TIP_i \bullet BD_i \bullet MC_i \bullet d \qquad \text{(Eq.5)}$$

$$\text{OP Stock} = TOP_i \bullet BD_i \bullet OM_i \bullet d \qquad \text{(Eq.6)}$$

Total P (TP_i) concentration and inorganic phosphorous concentration (TIP_i) were normalized with bulk soil density (g cm^{-3}), mineral density (g cm^{-3}), and organic matter density (g cm^{-3})respectively. The depth of the soil layer (d = 10 cm) was used to produce estimates of depth-integrated P stocks (g P m^{-2}).

2.7. Nutrient Stock Variation Estimations

Error propagation in mineral density and the P, OP and IP fractions stock estimations was addressed by assessing uncertainty associated with each variable. Estimation of variance of the P, IP, and OP stock in each predicted grid cell were computed with the following equations, adapted from Goidts et al. (2009) and Schrumpf et al. (2011):

$$\text{Var (P Stock)} = (\text{P Stock})^2 \bullet \left(\frac{\sigma_{TP}^2}{TP^2} + \frac{\sigma_{BD}^2}{BD^2} + 2\,\frac{\sigma_{TP-BD}}{TP \bullet \mathrm{BD}}\right) \qquad \text{(Eq.7)}$$

$$\text{Var (IP Stock)} = (\text{IP Stock})^2 \bullet \left(\frac{\sigma_{TIP}^2}{TIP^2} + \frac{\sigma_{BD}^2}{BD^2} + \frac{\sigma_{MC}^2}{MC^2} + 2\,\frac{\sigma_{TIP-BD}}{TIP \bullet \mathrm{BD}} + 2\,\frac{\sigma_{MC-BD}}{MC \bullet \mathrm{BD}} + 2\,\frac{\sigma_{TIP-MC}}{TIP \bullet \mathrm{MC}}\right) \qquad \text{(Eq.8)}$$

$$\text{Var (OP Stock)} = (\text{OP Stock})^2 \bullet \left(\frac{\sigma_{TOP}^2}{TOP^2} + \frac{\sigma_{BD}^2}{BD^2} + \frac{\sigma_{OM}^2}{OM^2} + 2\,\frac{\sigma_{TOP-BD}}{TOP \bullet \mathrm{BD}} + 2\,\frac{\sigma_{OM-BD}}{OM \bullet \mathrm{BD}} + 2\,\frac{\sigma_{TOP-OM}}{TOP \bullet \mathrm{OM}}\right) \qquad \text{(Eq.9)}$$

Where Var (P Stock) in each grid cell is the error associated with the P stock prediction in g m^{-2}. σ_{TP}, σ_{TIP}, $\sigma_{MC,}$ $\sigma_{OM,}$ and σ_{BD} are the standard deviation of TP, TIP concentration, MC, OM, and BD. Covariance (σ) of TP, TIP, MC, OM, and BD which were calculated from the raw dataset.

2.8. *Significant change determination*

Variance maps from each year for mineral content, IP and OP density were used to find ΔVar ($\Delta\sigma^2$), or the difference in variance value at each site between years, for each grid cell. In order to determine significance of difference in soil parameter between years (ΔIP) the 95% confidence interval around zero for change in each grid cell for IP density was calculated:

$$\pm 1.96 \cdot \sqrt{\Delta \mathrm{Var}_i(\mathrm{IP})} \qquad \text{(Eq.9)}$$

Where $\Delta\mathrm{Var}_i$(IP) is the difference in uncertainty calculations each year at location *i*. A grid cell with ΔIP Density value within this confidence interval around zero would signify no significant change; those outside were considered significant for our analysis.

2.9 *Mixture Model for defining area of diversion influence*

Mixture distributions are commonly used to model a population composed of two or more sub-populations and have been applied in a variety of disciplines (Everitt & Hand, 1981; Lo et al., 2001: Everitt, 2014). An expectation-maximization (EM) algorithm was used to fit a normal mixture model to the mineral content (%) dataset using the mixtools package in R (Benaglia et al. 2009).

3. Results and Discussion

Sedimentation from diversion influence

Mineral density is a measure of the mass of mineral material present in a given volume of wetland soil, in this case, related to fine grained Mississippi River sediment delivered through the river diversion. Although the Davis Pond freshwater diversion was not specifically designed to bring sediment from the river into the ponding areas, spatial distribution of mineral density (g cm^{-3}) demonstrates that fine-grained mineral sediment has settled out from diverted river water and became incorporated into the organic soils as they accrued (Figure 2). Net contribution of mineral materials is more pronounced throughout the wetland with a 58% increase in average BD and 100% increase in mineral density since the 2007 sampling (Table 1). As expected, soils with high mineral density occur closest to the diversion inflow and mineral density decreases with distance from the river (Day et al., 2009). In 2007, highly mineral soils are limited to northern areas of the wetland which experienced direct impact from preliminary operation of the diversion. After approximately 11 years of diversion operation, however, there is increased mineral density throughout the surface wetland soil, especially down the central preferential flow path.

In order to more accurately define the sediment-influence zones, an expectation-maximization (EM) algorithm was used to fit a normal mixture model to the mineral content (%) dataset (Figure 3). The model identified a bimodal distribution with two distinct components, the low mineral content component (mean = 48.4%, SD=12.94) and high mineral content component (mean = 74.5%, SD= 7.99). We estimated the break point between these populations in the dataset as the approximate overlap of the modelled distribution curves. This process indicated that a soil mineral content threshold of 65% by weight could be used to divide the modelled soil property estimations into two groups with significantly different sediment enrichment. In 2007, this sediment-influenced area was 4.8 km^2 (480 ha), in 2018 that area was over 3 times larger at 15.1 km^2 (1510 ha) (Figure 4). This finding is particularly noteworthy as it was hypothesized by the designers that these diversions would not add appreciable sediment to the wetlands.

Keogh et al. (2019) found that the Davis Pond freshwater diversion effectively captures fine-grained sediment, demonstrating that during moderate and low flow conditions in the summer and fall the sediment retention rate is up to 81%. Increased soil mineral content and consolidation of deposited muddy sediment has been shown to reduce vulnerability of Mississippi River Delta freshwater wetlands to impacts from erosive forces, sea level rise and subsidence (Morris et al. 2013; Graham and Mendelssohn, 2013; Slocum et al. 2005, Xu et al., 2019), especially when that sediment is accompanied by increased OM accumulation. Root systems of wetland plant communities are an important structural component for highly organic soils (DeLaune and White, 2012). Soils rich with organic material tend to have higher buoyancy and are less resistant to physical stress such as storms (Turner et al., 2009; Jafari et al., 2019). However, soils which receive fine-grained mineral sediment will have increased bulk density (Poormahdi et al. 2018) and will be less susceptible to erosion from wind waves (Sapkota and White, 2019*)* or storms (Howes et al., 2010). In addition, regular mineral sediment supply can improve growing conditions for wetland vegetation by increasing elevation, relieving flood stress, and introducing new substrate into which vegetation can grow (Graham and Mendelssohn 2016; Mendelssohn and Kuhn, 2003). Finally, riverine sediments commonly have associations with nutrients, like particulate inorganic P, which can become available after deposition. The remainder of this discussion will focus on the impacts of sedimentation on soil P stocks in the Davis Pond wetland system.

P Dynamics

Differences in total P, OP and IP stocks (g P m^2) between years were significant at the 140 measured sites ($p < 0.0001$) as well as for the means from modeled data (Table 1). Total P stock is greater in 2018, driven by significant gains in IP and a decrease in total OP (Table 1). A large portion of the study area (29% of the total wetland area), mainly located in the diversion influenced zone, switched dominant P form from OP in 2007 to IP in 2018. Allochthonous sources of P are mainly inorganic particulate P or ortho-phosphates associated with mineral sediments from the river. The most abundant P associated metals in Mississippi River sediments are Fe and Al (Sutula et al., 2004). In 2018, the Davis Pond wetland soil Fe and Al concentrations are equally positively correlated with log(mineral density) ($R=0.54$, $p=0.0003$). The presence of IP has a positive relationship with mineral density and bulk density of the soil ($R=0.93$ and 0.88, respectively, $p < 0.0001$), demonstrating that IP loading is closely related to deposition of river sediment. Fe and Al form complexes with IP, and can form minerals, providing long-term P storage (Malecki-Brown, et al., 2007). Under flooded or anoxic conditions, however, Fe will become reduced and dissolved IP may be released into the pore water (Zhang et al., 2012; Adhikari et al., 2015). As mineral sediments are the major driver for IP stock in the soils, then areas of the wetland with significant sedimentation should have greater stocks of inorganic P, a portion of which can be released from the sediment under low redox conditions, contributing to organic matter accumulation through plant uptake (White et al, 2006; Zhang et al., 2012).

Spatial P distribution

Sediment influence zones approximated with soil mineral content by percent allowed us to identify how P dynamics differ between areas influenced by the diversion and areas where no effective river reconnection occurs. River influenced regions in 2007 and 2018 have more than two times higher mean total P stock, and IP stocks are almost nine times larger than non-diversion-influenced soils (Table 2, Figure 5). Almost all sites that experience a statistically significant increase in IP content between years are located within the area of significant mineral sedimentation (Figure 6 a). Delivery of sediments brings large stocks of potentially bioavailable P, associated with Fe and Al (Ghaisas et al. 2019).

In contrast, the spatial distribution of soils with significant changes in OP has a less distinct pattern related to the diversion flow path. A majority of the sites with OP loss are located outside the sedimentation zone (Figure 6 b) and total OP stocks are decreasing with continued isolation from river influence over time (Table 2). Mean OP stocks, however, in the sediment influenced zone did not significantly decrease over time. This pattern is especially important because OP can act as a long-term P storage mechanism and reconnecting coastal freshwater wetlands to river influence aids in maintaining organic material and associated OP accretion. Losses to OM accrual in wetland soils can reduce nutrient storage functions, wetland soil stability, and carbon sequestration (Turner et al., 2009).

River diversion projects receive varying degrees of support from the scientific community because of large uncertainties in predicting impacts from river diversion derived sediment and nutrient enrichment (Quirk et al., 2019; White et al., 2019; Jafari et al., 2019). This study provided the first significant spatial data set covering over a decade of time as an opportunity to define river reconnection influence on receiving wetland soils. Coastal managers can use the results of this study to understand potential effects from the currently planned large sediment diversions. Introduction of riverine materials through Mississippi River diversions dramatically alters soil conditions in wetland systems due to introduction of fine-grained sediments and potentially bioavailable forms of P. Overall, we identified positive impacts from nearly continuous diversion influence on freshwater wetland soil structure, nutrient status, and organic matter accumulation.

Significant river influence in this wetland occurred down a central preferential flow path that is likely driven by elevation within the basin. Designs for larger sediment diversions should allow dispersion of diverted materials across larger areas of receiving basins while also considering the importance of small-grained sediments for land building and marsh nourishment. P dynamics in active deltaic systems are difficult to track over time due to interactions between P forms and environmental conditions which drive P cycling (Adams, et al., 2018). Previous to this study, the long-term impacts of lower Mississippi River hydrologic restoration projects had yet to be statistically quantified due to little or no pre-sampling. This is the first study to describe and model changes of soil characteristics in a river diversion restoration wetland over space and time for use in identifying the extent and intensity of river diversion influence. Determining the effectiveness of freshwater diversions will ensure their continued use not only as examples for other diversions, but as effective restoration tools which can contribute to regional scale coastal restoration goals.

Tables

Table 1. Summary of Soil Physiochemical Properties [a]

	Bulk Density (g/cm^3)	Mineral Density (g/cm^3)	Organic Matter Density (g/cm^3)	Phosphorus (g/m^2)	Inorganic P (g/m^2)	Organic P (g/m^2)
Measured 2007	**0.12 ± 0.010**	**0.07 ± 0.004**	0.049 ± 0.02	**10.2 ± 3.2**	**3.28 ± 0.002**	**3.03 ± 0.27**
Measured 2018	**0.21 ± 0.012**	**0.14 ± 0.005**	0.047 ± 0.01	**15.3 ± 3.2**	**7.43 ± 0.04**	**2.30 ± 0.35**
Paired t-test *p*[b]	***p* < 0.0001**	***p* < 0.0001**	*p* = 0.64	***p* < 0.0001**	***p* < 0.0001**	***p* < 0.0001**
Predicted 2007	0.093 ± 0.038	**0.060 ± 0.0002**	0.059 ± 0.003	**10.5 ± 0.12**	**2.2 ± 0.19**	**3.7 ± 0.05**
Predicted 2018	0.147 ± 0.065	**0.120 ± 0.0003**	0.062 ± 0.004	**15.1 ± 0.20**	**5.3 ± 0.35**	**3.0 ± 0.05**

[a] Data from 140 measured sites and predicted dataset for 100 m^2 grid cells, presented with standard error. Bold signifies significant difference between years.
[b] Results from paired t-tests are for data from measured sites only.

Table 2. Mean P stock by year of each soil P fraction inside and outside of the sediment influenced area. [a]

P Fraction	Year	River influenced (g P m^{-2})	Non-River Influenced (g P m^{-2})
Total P	2007	**10.98 ± 0.92** *	**6.21 ± 0.01** *
	2018	**8.67 ± 0.54** *	**7.16 ± 0.02** *
Inorganic P	2007	11.03 ± 0.9 *	**0.91 ± 0.01** *
	2018	10.83 ± 0.6*	**1.64 ± 0.03***
Organic P	2007	2.32 ± 0.8*	**3.95 ± 0.05***
	2018	2.08 ± 0.3*	**3.66 ± 0.10***

[a] Organic P was found by normalizing concentration with organic matter density, inorganic P was found using mineral density. Bold indicates significant difference between years.
[*] Indicates significant difference between wetland areas.

Figures

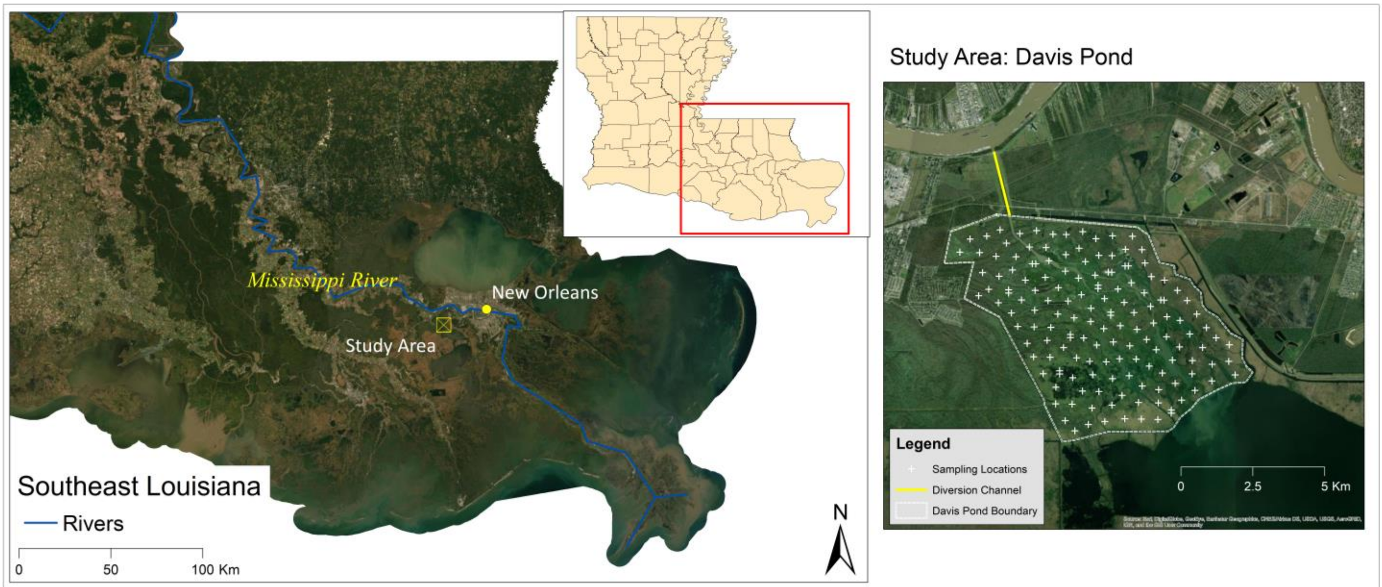


Figure 1. (a) Satellite image of southeastern Louisiana, the study area is located southwest of New Orleans. (b) Satellite image of study wetland area, Davis Pond, with 140 sampled stations. Adapted from GoogleEarth imagery.

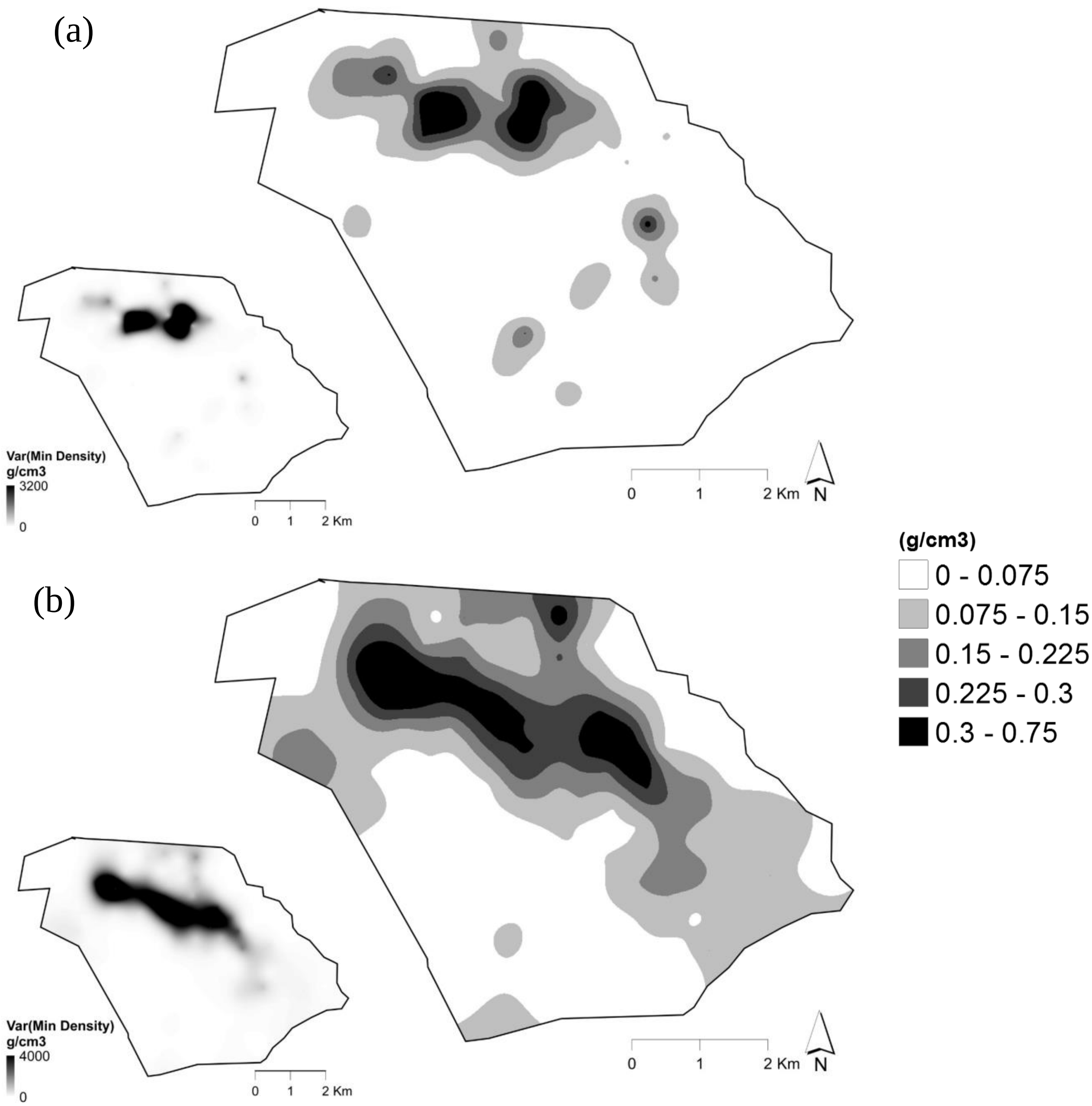


Figure 2. Maps of mineral density (g cm^{-3}) in 0-10 cm soil layer in (a) 2007 and (b) 2018. Small adjacent maps represent associated variance values across the wetland for each prediction.

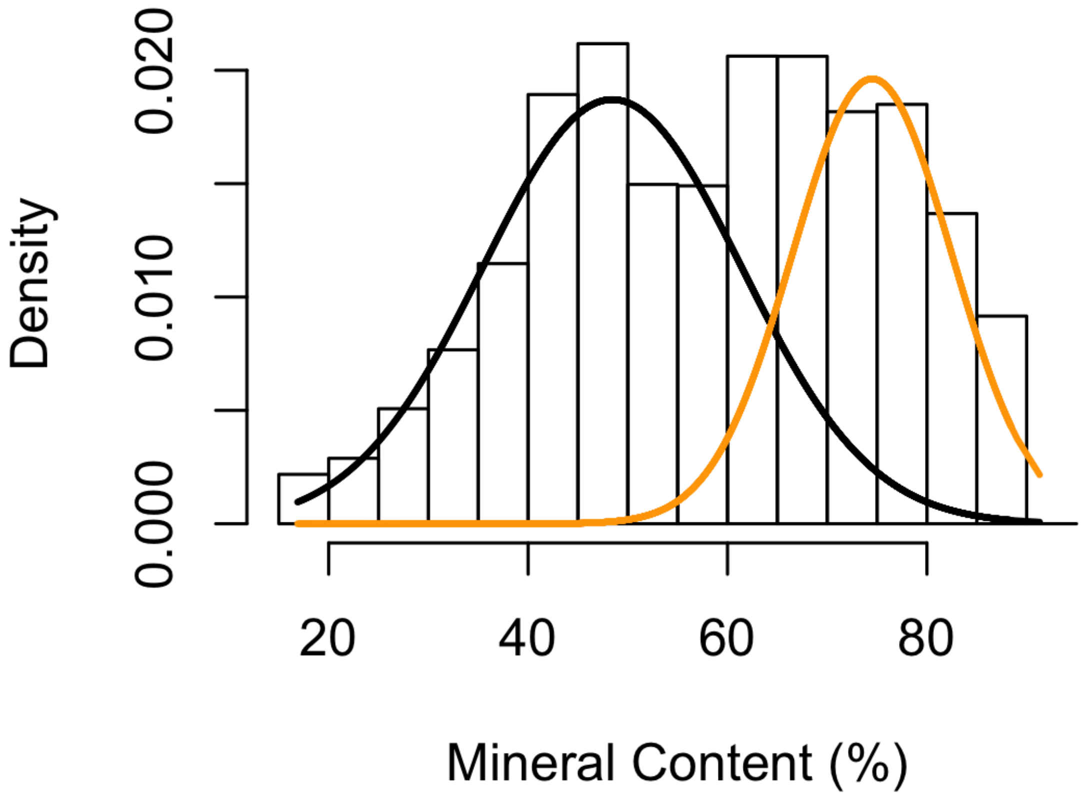


Figure 3. Density plot with modelled component distributions calculated from the EM Algorithm mixture model of the 2018 mineral content (%) data. Low mineral content component in black, high mineral content component is orange.

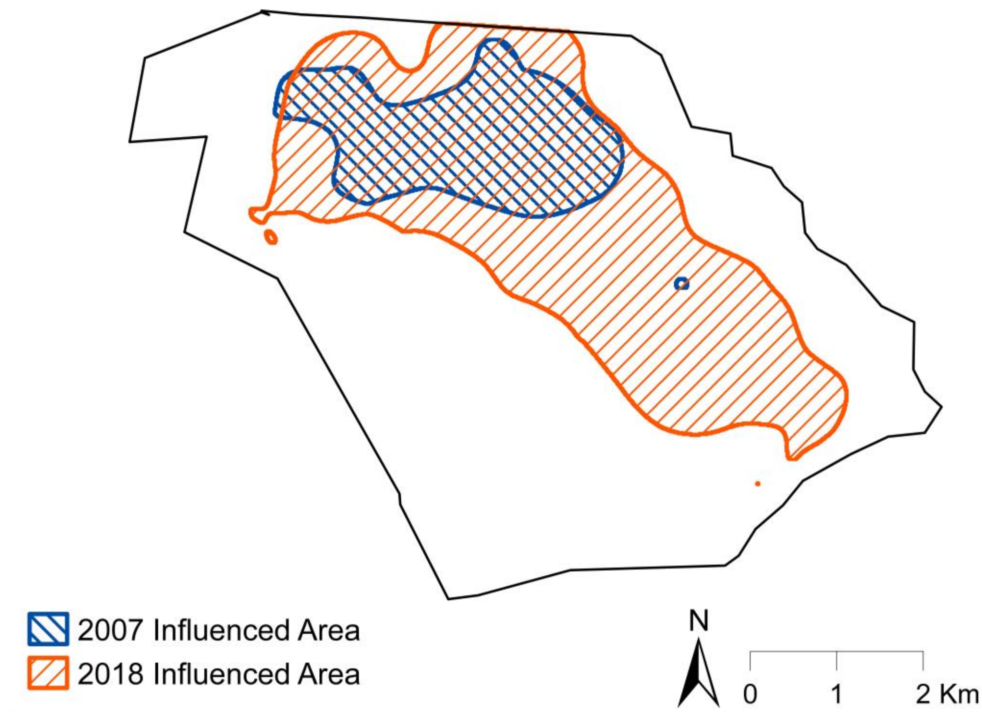


Figure 4. Map of sediment influenced zone from 2007 (blue, back facing shading) and 2018 (orange, forward facing shading) defined by mixture model from mineral content (%) dataset.

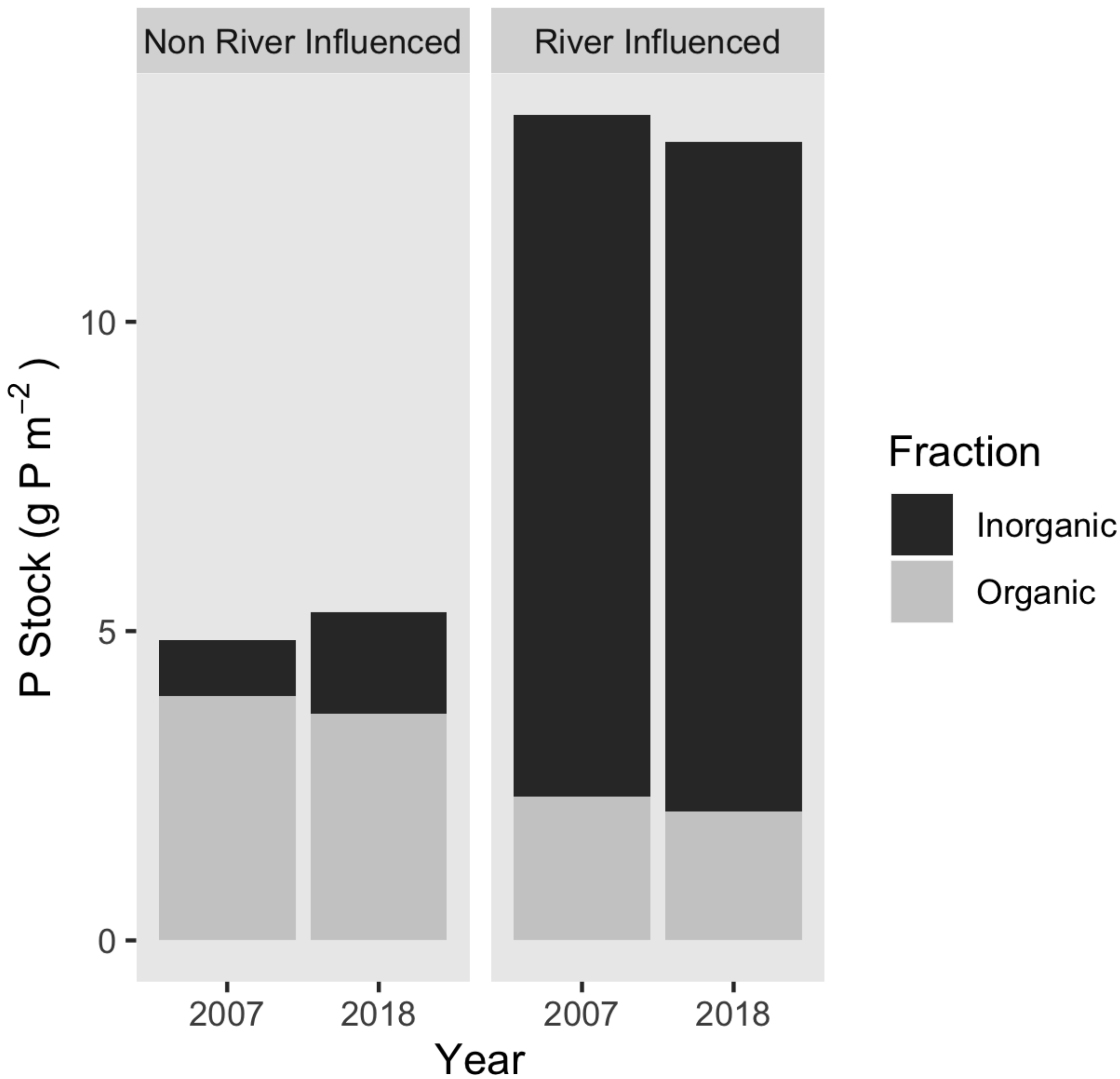


Figure 5. Contribution of P stock for each soil P fraction by year in the diversion sediment influenced zone and the non-influenced zone.

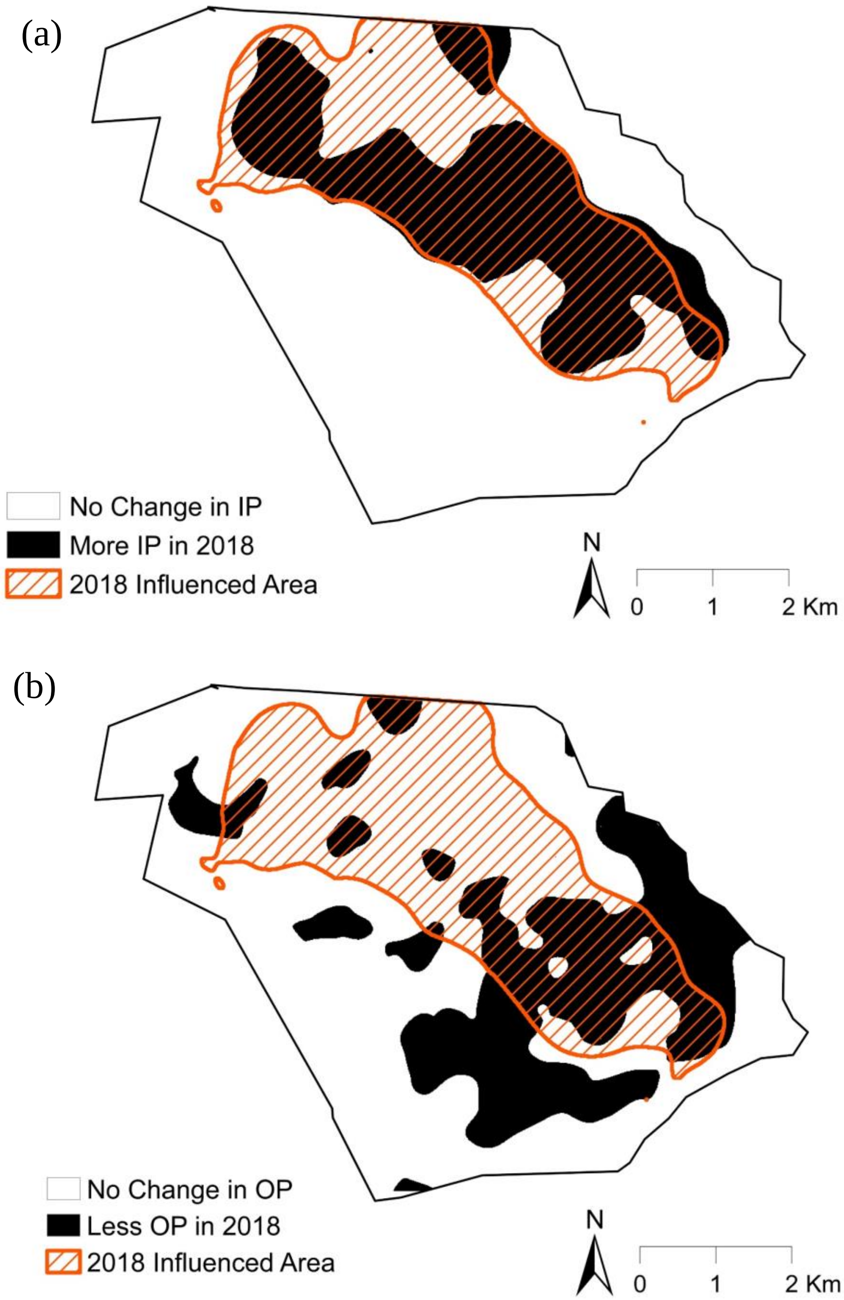


Figure 6. Davis Pond map, black area demarcates where between years new soils had (a) an increase in IP stock or (b) a decrease in OP stock that was outside the 95% confidence interval for change in each variable. Orange shaded area is diversion sediment impact zone in 2018 (mineral content > 65%).

Table Captions

Table 1. Mean bulk density, mineral and organic matter density, and P stock for each fraction at 140 measured sites and from predicted dataset presented with standard error. Results from paired t-tests are for data from measured sites. Bold signifies significant difference between years.

Table 2. Mean content stocks by year of each soil P fraction inside and outside of the sediment influenced area. Organic P stock was found by normalizing concentration with organic matter density, inorganic P stock was found using mineral density. Bold indicates significant difference between years. [a] indicates significant difference between wetland areas.

Figure Captions

Figure 1. (a) Satellite image of southeastern Louisiana, the study area is located southwest of New Orleans. (b) Satellite image of study wetland area, Davis Pond, with 140 sampled stations. Adapted from GoogleEarth imagery.

Figure 2. Maps of mineral density (g cm^{-3}) in 0-10 cm soil layer in (a) 2007 and (b) 2018. Small maps represent associated variance values across the wetland for each prediction.

Figure 3. Density plot with modelled component distributions calculated from the EM Algorithm mixture model of the 2018 mineral content (%) data. Low mineral content component in black, high mineral content component is orange.

Figure 4. Map of sediment influenced zone from 2007 (blue, back facing shading) and 2018 (orange, front facing shading) defined by mixture model from mineral content (%) dataset.

Figure 5. Contribution of stock for each soil P fraction by year in the diversion sediment influenced zone and the non-influenced zone.

Figure 6. Davis Pond map, black area demarcates where between years new soils had (a) an increase in IP stock or (b) a decrease in OP stock that was outside the 95% confidence interval for change in each variable. Orange shaded area is diversion sediment impact zone in 2018 (mineral content > 65%).

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