

Mississippi river sediment diversions and coastal wetland sustainability: Synthesis of responses to freshwater, sediment, and nutrient inputs



T. Elsey-Quirk^{a,*}, S.A. Graham^b, I.A. Mendelssohn^a, G. Snedden^c, J.W. Day^a, R.R. Twilley^a, G. Shaffer^d, L.A. Sharp^e, J. Pahl^f, R.R. Lane^g

^a Department of Oceanography and Coastal Sciences, Louisiana State University, Baton Rouge, LA, 70803, USA

^b Department of Biological Sciences, Nicholls State University, Thibodaux, LA, 70301, USA

^c U. S. Geological Survey, Wetland and Aquatic Research Center, Baton Rouge, LA, 70803, USA

^d Department of Biological Sciences, Southeastern Louisiana University, Hammond, LA, 70402, USA

^e Louisiana Coastal Protection and Restoration Authority, Lafayette, LA, 70596, USA

^f Louisiana Coastal Protection and Restoration Authority, Baton Rouge, LA, 70802, USA

^g Comite Resources Inc., P.O. Box 66596, Baton Rouge, LA, 70896, USA

ARTICLE INFO

Keywords:

Restoration
Decomposition
Productivity
Eutrophication
Accretion
Mississippi delta

ABSTRACT

Management and restoration of coastal wetlands require insight into how inundation, salinity, and the availability of mineral sediment and nutrients interact to influence ecosystem functions that control sustainability. The Mississippi River Delta, which ranks among the world's largest and most productive coastal wetland complexes, has experienced extensive deterioration over the last century due, in large part, to enhanced vulnerability to relative sea-level rise and lateral erosion caused by a combination of natural processes and anthropogenic modifications of hydrology. This land loss crisis has prompted the State of Louisiana to develop a comprehensive restoration plan that includes constructing and implementing a series of large-scale sediment diversions that will reconnect sediment- and nutrient-rich Mississippi River water to adjacent bays, estuaries, and wetlands. Sediment loading through diversions is predicted to enhance the long-term sustainability of coastal wetlands; however, the additive effects of increased inundation, abrupt changes in the salinity regime, and high nutrient loads on wetland plant growth and organic matter (SOM) decomposition rates, which help regulate accretion and elevation change, is uncertain. Therefore, this review attempts to synthesize existing information to inform predictions of the interactive effects of diversions on these drivers of coastal wetland sustainability. The data suggest that sediment deposition within an optimal elevation range will increase the overall productivity of existing wetlands where prolonged flooding does not counter this effect by limiting plant growth. A reduction in salinity may increase plant productivity and cause vegetation shifts to less salt tolerant species, but seasonal swings in salinity may have unforeseen consequences. Nutrient-loading is predicted to lead to greater above-ground productivity, which, in turn, can facilitate additional sediment trapping; however, belowground productivity may decline, particularly in areas where sediment deposition is limited. In areas experiencing net deposition, nutrient-enrichment is predicted to enhance belowground growth into new sediment and contribute to positive effects on soil organic matter accumulation, accretion, and elevation change. Thus, we contend that sediment input is essential for limiting the negative effects of flooding and nutrient-enrichment on wetland processes. These conclusions are generally supported by the biophysical feedbacks occurring in existing prograding deltas of the Mississippi River Delta complex.

1. Introduction

Coastal wetlands are among the most productive ecosystems on the planet, performing ecological functions that contribute to services of high economic value such as habitat for fish and wildlife, nutrient

transformation and removal, burial and long-term sequestration of carbon, and storm surge protection for coastal communities (e.g., Boesch and Turner, 1984; Chmura et al., 2003; Costanza et al., 2008; Sousa et al., 2010; Batker et al., 2014). Yet despite their high value, coastal wetlands around the world are experiencing a convergence of

* Corresponding author.

E-mail address: tquirk@lsu.edu (T. Elsey-Quirk).

<https://doi.org/10.1016/j.ecss.2019.03.002>

Received 6 September 2018; Received in revised form 25 February 2019; Accepted 4 March 2019

Available online 09 March 2019

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human disturbance and sea-level rise that threatens their long-term sustainability. The world's river deltas are particularly vulnerable to rapid environmental change due to the combined effects of river management and natural processes on flooding regime, sediment supply, salinity, and nutrient loading, which regulate complex feedbacks among plant community composition and productivity, organic matter decomposition, and soil accretion. Coastal wetland management, therefore, requires an understanding of how these hydrogeomorphic factors interact to influence wetland stability and resilience to sea-level rise and other stressors (Kirwan and Megonigal, 2013).

In the Mississippi River delta, as in deltas around the world, wetlands form as vegetation colonizes subaerial sediments deposited in shallow coastal environments. Historically, the Mississippi River and its distributaries freely overflowed their banks and penetrated natural levees as crevasse splays during periods of high flow creating a delta-complex made up of several hydrologic basins separated by active or abandoned distributary ridges (Roberts, 1998; Day et al., 2000; Blum and Roberts, 2012). Eight major delta lobes have been identified including the Atchafalaya, which is in the early stages of development. Each delta lobe, having a lifespan of approximately 1500 years and covering 100s–1000s of km², formed as river channels elongated and bifurcated. Delta lobes are ultimately abandoned through a process of upstream avulsion in favor of shorter, more efficient routes to sea level. Even after abandonment, most distributaries continued to flow with river water during major floods, maintaining a framework of interconnected distributary channels and natural levee ridges (Penland et al., 1988), and extending the life of wetlands in abandoned delta lobes until their ultimate degradation. Numerous crevasses also functioned at a smaller scale during high water for a few years to as long as a century to build sediment splays adjacent to the natural levee with areas of 10s–100s of km² (Saucier, 1963; Weller, 1959; Davis, 2000; Roberts, 1998; Shen et al., 2015). The discharge of mineral sediment during delta formation or through crevasse splays created elevations appropriate for wetland vegetation to colonize and facilitated positive feedbacks among primary productivity, organic matter accumulation, sediment-trapping, and accretion following plant establishment. Freshwater wetlands are common throughout active river deltas where hydrology is dominated by seasonal river flooding. In abandoned deltas, seawater encroachment at the leading edge allows the formation salinity gradients, where salt marshes exist nearest the coast and grade to brackish, intermediate, oligohaline, and freshwater marsh near the river channels and distributaries. Ultimately, these wetlands subside and convert to open water over time without continued sediment input.

Today, modification of river flow by flood control structures and a reduction of sediment supply by upstream dams and artificial levees along much of the lower Mississippi River have greatly reduced the amount of fresh water and sediment supplied to deltaic wetlands (Boesch et al., 1994; Williams et al., 1994). Currently, natural overbank flooding is limited to the extreme lower end of the River, at the Balize Delta and along the lower Atchafalaya River (Allison et al., 2012). Canals constructed for navigation, oil and gas exploration and extraction, drainage, and logging along with spoil banks deposited on wetland surfaces adjacent to canals have also changed the natural flooding and salinity regimes and sediment transport patterns (Swenson and Turner, 1987; Day et al., 2007; Keddy et al., 2007). These human modifications coupled with geologic subsidence and sea-level rise have resulted in a net loss of wetland area since European settlement equal to approximately one-third of the historical extent of the Mississippi River Delta wetlands (Craig et al., 1979; Gagliano et al., 1981; Day et al., 2000), with 4754 to 5640 km² lost between 1932 and 2016 (Couvillion et al., 2017). Moreover, without restoration, an additional 5800 to over 10,000 km² of land area is predicted to convert to open water by the end of the century (Blum and Roberts, 2009). Along with the loss of wetland area, is a corresponding loss of habitat value, ecological functions and services, and cultural and economic resources (Day et al., 2007; Batker et al., 2014). Therefore, reducing land loss and creating

new land through restoration is of both ecological and economic importance.

In an attempt to offset widespread and rapid coastal land loss, the Coastal Protection and Restoration Authority (CPRA) of Louisiana has developed, revised, and expanded three Coastal Master Plans (CMPs; CPRA, 2007, 2012 and 2017). The most recent CMP outlines plans for 79 restoration projects designed to build or maintain approximately 2000 km² of land through a combination of wetland creation, sediment diversions, and barrier island restoration (CPRA, 2017). More specifically, the 2017 CMP proposes seven large-scale sediment diversion projects to be implemented over the next 50 years (Pahl et al., 2019 this volume). River sediment diversions are engineered outlets through existing levees designed for periodic and/or sustained delivery of freshwater, nutrients, and sediment to create new wetlands and combat further degradation (Day et al., 2007; CPRA, 2017; Peyronnin et al., 2017). Once diversion structures are operational, water flows from the river by gravity and is distributed to the receiving basin largely by wind, waves, tides and currents. In essence, diversions are designed to mimic natural crevasse splays and promote sub-delta development (Davis, 2000). In total, coastal restoration and protection efforts in Louisiana represent the largest ever attempt to restore a natural system, with an estimated cost of \$50 billion over 50 years.

The high cost and scale of the proposed coastal restoration in the Mississippi River Delta necessitates improved predictions of how sediment diversions will change important environmental drivers that interact to influence key wetland processes. In this context, our goal was to synthesize existing information on the effects of flooding, freshwater input, nutrient, and mineral sediment availability on processes that influence coastal wetland sustainability, including plant productivity, decomposition, and rates of accretion and elevation change, as well as discuss likely interactions. Finally, we summarize our findings, with support from case studies where rivers flow freely into receiving basins, to help better understand the fate of coastal wetlands receiving diverted river water and sediment. While this review is broadly applicable, our focus is on potential changes to wetland plant productivity, organic matter decomposition, accretion and elevation change that are likely to occur if the Mississippi River is reconnected to adjacent receiving bays and wetlands through sediment diversions.

2. Potential effects of reintroducing river water and sediment to deltaic wetlands

2.1. Sediment deposition

The primary objective of river sediment diversions is to expand wetland area and increase the sustainability of existing wetlands by introducing mineral sediments to subsiding and eroding wetlands. Currently, much of the sediment supplied to wetlands of the eastern portion of the Mississippi River deltaic plain is from submerged sediments resuspended and deposited on the wetland surface during energetic events like frontal passages and hurricanes (e.g., Perez et al., 2000; McKee and Cherry, 2009; Turner et al., 2006). Strategies that enhance continued sediment delivery in the region have the potential to build new land in areas of open water (Rouse et al., 1978; Roberts et al., 2003; Kim et al., 2009; Bevington et al., 2017), and nourish degrading wetlands (Slocum et al., 2005; Baustian and Mendelssohn, 2015). Generally, river input increases sediment supply to adjacent wetlands following an approximate exponential decay function, where higher loads of coarser grained particles settle nearest the sediment source and smaller loads of fine sediments are dispersed and deposited farther afield (Fig. 1; Wheelock, 2003; Day et al., 2009). Plant productivity can be stimulated, which in turn, aids in trapping sediment on the soil surface further (e.g., Stumpf, 1983), representing a positive feedback between sediment deposition and plant growth.

By virtue of their location along protected parts of estuaries and river deltas, coastal wetlands typically have fine-textured soils,

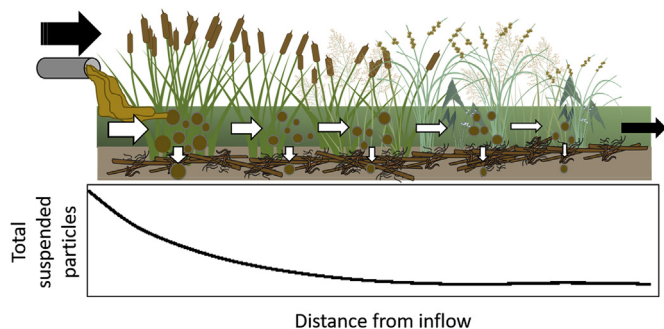


Fig. 1. Schematic illustrating particle settling as a function of distance from the inflow to a wetland (modified from Reddy and DeLaune, 2008).

comprised primarily of silts and clays. Fine-grained sediment particles play a key role in facilitating biogeochemical processes by adsorbing water, nutrient cations, and organic matter. Marsh restoration with coarser-grained sand has been shown to be problematic due to high porosity, rapid organic matter decomposition, low water holding capacity, and low nutrient retention caused by leaching leading to low plant productivity (Langis et al., 1991; Gibson et al., 1994; Boyer and Zedler, 1996). In the context of Mississippi River diversions, the majority (> 80%) of the sediment load is fine grained, with a smaller percentage of sand (< 20%; Allison et al., 2012). The sand fractions will settle rapidly, while finer sediments will be dispersed more broadly. Silt represents the largest fraction of sediment in both the River and bottoms of bays and estuaries (Xu et al., 2016), indicating that river diversions will deliver mostly fine sediments to deteriorating wetlands.

Positive effects of sediment inputs on marsh vegetation, particularly in degrading wetlands, generally correspond with higher soil surface elevations, reduced flooding, improved soil aeration in the rooting zone, low or undetectable levels of the phytotoxin hydrogen sulfide, greater soil bulk density, and higher concentrations of mineral and adsorbed nutrients (e.g., N, P, K, Fe, Mn, and others; Nyman et al., 1990; Wilsey et al., 1992; Mendelssohn and Kuhn, 2003; Slocum et al., 2005; Turner et al., 2006; Schrifft et al., 2008; Slocum and Mendelssohn, 2008; Stagg and Mendelssohn, 2010, 2011; Day et al., 2011; Graham and Mendelssohn, 2013; Baustian and Mendelssohn, 2015). In salt marshes of Louisiana, where the tidal range averages 30 cm, the beneficial effects of sediment addition have been shown to occur when the soil surface elevation is within the mid-to-high intertidal zone (2–11 cm above MSL; Stagg and Mendelssohn, 2011). Here, degrading *Spartina alterniflora* marshes tend to be at least 10–18 cm lower (16–21 cm below MSL) in elevation than healthy marshes (6 cm below - 15 cm above MSL) (Fig. 2). For example, following the accidental discharge of dredged sediment into a *S. alterniflora*-dominated marsh in the Mississippi River Balize Delta (near Venice, LA), total plant biomass was significantly greater where sediment addition exceeded 15 cm compared to a deteriorating reference marsh that received no added sediment (Mendelssohn and Kuhn, 2003). The positive plant growth response associated with increased elevation was still evident seven years after sediment application, especially at moderate sediment additions (initially 15–30 cm) that resulted in relative elevations of 5–12 cm above the reference marsh (Slocum et al., 2005). Schrifft et al. (2008) found comparable results for a *S. alterniflora* salt marsh adjacent to Bayou Lafourche (near Port Fourchon, LA) that received hydraulically dredged sediments after vegetation dieback from a brown marsh event in 2000. In their study, sediment additions that raised the marsh elevation 14–20 cm above reference conditions had rapid vegetation recruitment, and plant cover and species richness that was statistically indistinguishable from healthy reference marshes after two years. Three years later, sediment-amended marsh elevations of 3–10 cm above ambient marsh elevation corresponded with above- and belowground

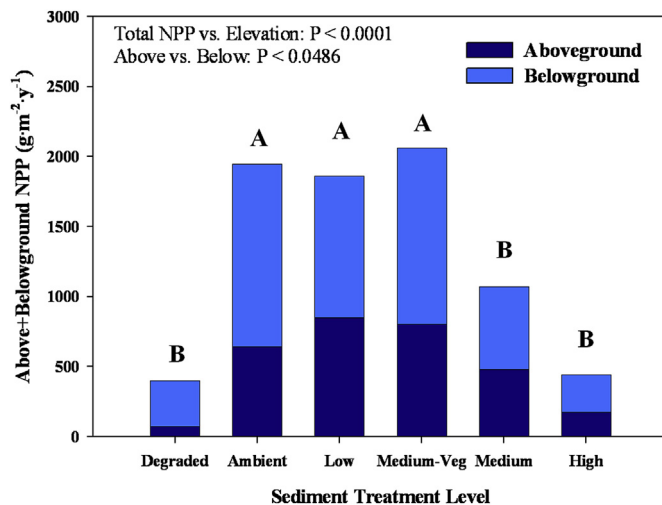


Fig. 2. Total *Spartina alterniflora* above + belowground net primary productivity (NPP) in response to sediment addition (marsh elevations relative to MSL seven years after sediment application in 2002: Degraded = -16 to -21 cm; Ambient = -3 to -6 cm; Low = +3 to +6 cm; Medium-vegetated = +4 to +10 cm; Medium = +9 to +12 cm; High = +10 to +15 cm; Stagg and Mendelssohn, 2011). Different letters represent significantly different means, with respect to the effect of sediment addition on total production ($p < 0.0001$) (Fisher's Protected LSD, $p < 0.05$). (modified from Stagg and Mendelssohn, 2010).

production that was equivalent to natural marsh production, and significantly greater than both the extreme low (degraded reference) and extreme high (high sediment) treatments (Fig. 2; Stagg and Mendelssohn, 2010). Moreover, DeLaune et al. (1990) found that smaller amounts of sediment (e.g., 8–10 cm) added to a degrading *S. alterniflora* salt marsh doubled live aboveground plant biomass by the end of the second growing season, while as little as 4–6 cm had a positive, but non-significant, effect. Natural sedimentation events resulting from hurricane impact resulted in a similar increase in primary productivity, though the intensity of the effect depended on plant species composition (Fig. 3). Regardless, hurricane-induced sedimentation accounted for 75–77% of the variation in primary productivity (Baustian and Mendelssohn, 2015).

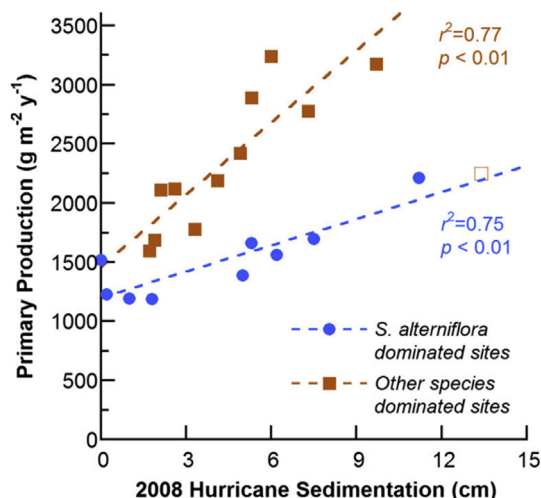


Fig. 3. Aboveground primary production in response to 2008 hurricane (Gustav, Ike) sedimentation. The open square was identified as an outlier and omitted from the regression (Reprinted by permission from Springer Nature, Wetlands, Hurricane-Induced Sedimentation Improved Marsh Resilience and Vegetation Vigor under High Rates of Relative Sea Level Rise, Baustian and Mendelssohn, Copyright, 2015).

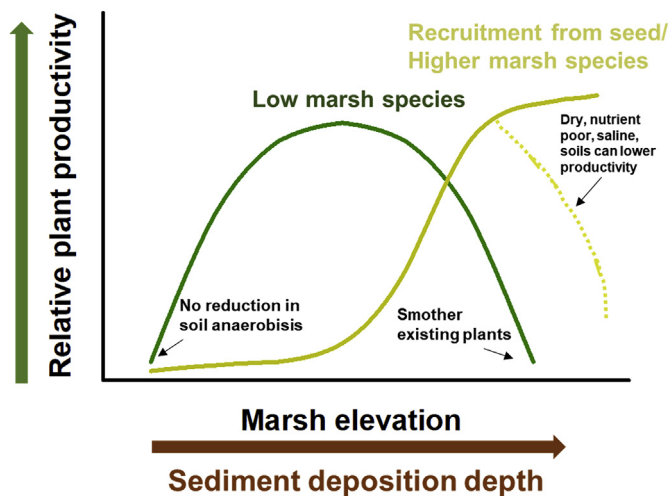


Fig. 4. Conceptual model of wetland plant productivity in response to elevation increases due to sediment deposition.

Although less studied, fresh and brackish marsh species also show positive responses to sedimentation. In a mesocosm study, a suite of 11 freshwater marsh and swamp species had positive belowground growth responses with 1.5 cm of sediment applied per year over three growing seasons (Hillmann, 2011). The mesocosms treated with sediment had an elevation increase of approximately 4.5 cm compared to untreated mesocosms with an elevation decrease of about 1.3 cm, a difference of almost 6 cm. Similarly, following thin-layer sediment enhancement of six brackish marshes representing a chronosequence of restoration sites, La Peyre et al. (2009) found that when interior marsh ponds were filled with sediment, both above- and belowground biomass increased over time, approaching that of the reference marshes within 7 years.

Too little or too much sediment can result in species changes and/or altered ecosystem function (Fig. 4). Sediment depth influences whether existing plants will be smothered or grow through the deposition layer. In a Georgia salt marsh, sediment additions at depths of 8, 15, and 23 cm enhanced biomass of *S. alterniflora*, while 30 cm of sediment reduced biomass, and 61 and 90 cm of sediment caused plant mortality (Reimold et al., 1978). Thus, a sediment depth threshold appears to occur at approximately 30 cm where growth and/or survival of existing *S. alterniflora* plants are impaired. A negative aboveground biomass response to sediment addition may be indicative of above-optimal elevations (Fig. 4). Nonetheless, sediment depths that smother existing vegetation can still foster vegetation colonization through seedling establishment (Reimold et al., 1978; Mendelssohn and Kuhn, 2003). Reimold et al. (1978) found seedling recruitment at the 61 and 90 cm burial treatments resulted in considerable new biomass production and ultimately the greatest live biomass of all the sediment burial treatments after two growing seasons. However, if sediment deposition raises the marsh soil surface to an elevation that prevents regular tidal inundation, vegetation change can occur, or under extreme conditions, conversion of marsh to another habitat type. Slocum et al. (2005) described a shift in vegetation with a high sediment depth, where a former *Spartina alterniflora* marsh became dominated by *Bolboschoenus robustus*, *Distichlis spicata*, and *Spartina patens*. Large sediment applications can result in soils becoming relatively dry, nutrient limited, and more saline, resulting in vegetation stress, sparse plant cover, and low productivity (Slocum et al., 2005; Stagg and Mendelssohn, 2010). At elevations that exceed the range of most tidal flooding, precipitation may leach salt from soils and allow terrestrial vegetation (trees and shrubs) to colonize (Montz, 1972). In contrast, too little sediment does not provide sufficient positive effects on elevation and soil physicochemistry to counter the negative effects of relative sea level rise and excessive flooding, particularly in the rapidly subsiding wetlands of

coastal Louisiana. Marshes with compressible organic soil are especially susceptible to a reduction in surface elevation if the amount of sediment added is too little to overcome compaction resulting from the weight of added sediment (Graham and Mendelssohn, 2013), potentially increasing vulnerability to excessive flooding. Yet, even with relatively small increases in marsh elevation, plant productivity can be stimulated. For instance, experimental applications of hydraulically conveyed fine-grain sediment to thicknesses ranging from 2 to 20 cm provided *Spartina patens*-dominated plots with an immediate increase in soil surface elevation, though realized elevation gains were minimal (< 3 cm) after 2.5 years due to compression of the underlying native marsh soil (Graham and Mendelssohn, 2013). Nevertheless, plots that initially received > 15 cm of sediment maintained small elevation gains, which corresponded with enhanced above- and belowground plant production compared to control plots. Thus, a minimum threshold of sediment nourishment may be necessary to increase plant productivity and have lasting positive effects on marsh surface elevation, despite having only small direct effects on elevation.

The overall net effect of sediment deposition on marsh elevation depends on the balance between elevation increase and compaction, as well as redox- and sediment-mediated effects on decomposition. In isolation of other factors, sediment deposition may increase organic matter decomposition at the surface through enhanced aeration of the rhizosphere, but this effect on total organic matter pools and accretion will likely be low due to the indirect effect of stimulating plant productivity and aboveground trapping of mineral sediment (Mudd et al., 2010; Graham and Mendelssohn, 2013). Furthermore, mineral sediments play a significant role in organic matter preservation, both through adsorption onto mineral surfaces or incorporation into organo-mineral aggregates (Rovira and Vallejo, 2003). Reactive iron plays a particularly important role in preserving organic carbon and this mechanism has shown to account for approximately 15% of organic carbon in deltaic wetlands (Shields et al., 2016). In salt marshes, mineral sediment accumulation has been shown to be positively correlated with labile and total organic carbon accumulation, indicating that an ecologically important relationship exists between mineral sediment availability and organic carbon preservation (Unger et al., 2016).

Deltaic wetlands that have been hydrologically isolated from riverine sediment sources tend to have lower soil bulk densities and a higher fractions of organic matter with increasing distances from coastal and marine sediment sources (Hatton et al., 1983; Howes et al., 2010). Freshwater and oligohaline marsh soils, therefore, have higher organic matter content than salt marshes (e.g., Craft, 2007), which may be due to relatively low sediment input, higher input of organic matter, and/or slower decomposition. River diversions may level or reverse this trend by increasing the soil bulk density and lowering organic matter content in freshwater wetlands nearest the diversion outfall. However, this change in soil composition would coincide with an increase in rates of mineral sediment and organic matter accumulation and accretion (DeLaune et al., 2003).

2.2. Inundation effects

Waterlogging and prolonged inundation are primary factors causing the widespread loss of wetlands in the Mississippi River Delta (DeLaune et al., 1983; Nyman et al., 1993). While river diversions will increase flooding in wetland systems that may already be flood-stressed, corresponding sediment deposition is predicted to raise soil surface elevations such that hydroperiods are reduced over a greater area compared to current conditions. Tidal wetlands naturally experience varying degrees of waterlogging and soil anaerobiosis that are governed primarily by the interplay between water level variability and wetland elevation. Most wetland plant species possess morphological and physiological adaptations to mitigate stresses such as anoxia and low redox potential that can be brought about by inundation (Mendelssohn and Morris, 2000). These adaptations can include aerenchyma, adventitious roots,

and pressurized gas flow from the atmosphere to the rhizosphere (Shaffer and Kandalepas, 2017). However, even the most flood-adapted emergent species must invest additional energy to maintain these adaptations, which can reduce the energy available for nutrient uptake, growth, and reproduction (e.g., Mendelssohn et al., 1981; Bradley and Morris, 1990; Mendelssohn and Morris, 2000 and references therein). Sulfide toxicity imposes additional stress on plants exposed to sea water brought about by inundation (DeLaune et al., 1983; Pezeshki and DeLaune, 2002). For example, *Spartina alterniflora* productivity is negatively affected by soluble sulfide levels greater than 1 mM (or 32 ppm) (Bradley and Dunn, 1989; Koch and Mendelssohn, 1989; Koch et al., 1990), while freshwater marsh species, like *Panicum hemitomon*, are impacted at much lower concentrations (Koch and Mendelssohn, 1989; Koch et al., 1990).

Numerous studies have demonstrated the ability of common wetland plant species to survive under persistently flooded conditions. For instance, Lessmann et al. (1997) observed that both *S. patens* and *S. alterniflora* could survive nearly 60 days of inundation at depths approaching 40 cm. Spalding and Hester (2007) similarly documented *S. patens* survival after one year of persistent inundation at 20 cm at all salinity levels tested (up to 6). *Sagittaria lancifolia*, *Sagittaria latifolia*, and *Sagittaria platyphylla* all tolerated 27 months of inundation, with water levels 30 cm above the soil surface (Martin and Shaffer, 2005). However, continuous flooding comes at an energetic cost, resulting in a decline in productivity. In the field, marsh organ studies that simulate varying marsh elevations (see Morris et al., 2002) have documented decreased productivity due to increased inundation for a number of common Louisiana species including *S. alterniflora* (Voss et al., 2013; Snedden et al., 2015), *Juncus roemerianus* (Voss et al., 2013), *S. patens* (Kirwan and Guntenspergen, 2015; Snedden et al., 2015), *Schoenoplectus americanus* (Schile et al., 2017), and *S. lancifolia* (using a similar approach in a greenhouse; Visser and Sandy, 2009; Visser and Peterson, 2015). Exponential declines in aboveground and belowground production of both *S. patens* and *S. alterniflora* were documented in a marsh organ study in brackish and salt marsh locations in Breton Sound, LA with six step increases in time inundated from 3 to 88% (Fig. 5;

Snedden et al., 2015). With the exception of *S. alterniflora* belowground biomass, higher salinity tended to shift the decay curve down, indicating that inundation impacts may be exacerbated at elevated salinities. *Sagittaria lancifolia*, however, was not as impacted by flooding increases illustrating species-specific responses to inundation (Snedden et al., 2015).

Under flooded conditions, soil organic matter decomposition slows as the concentration of oxygen declines (Gambrell and Patrick, 1978; Day and Magonigal, 1993; Hackney and de la Cruz, 1980; Hemminga et al., 1988). However, field and macrocosm experiments examining the relationship between flooding and organic matter decomposition in wetland soils have not led to a clear model (Hackney and de la Cruz, 1980; Hackney, 1987; Blum 1993; Kirwan et al., 2013; Stagg et al., 2018; Janousek et al., 2017), supporting the conclusion of Kirwan et al. (2013) that organic matter decomposition in already semi-saturated wetland soil is not strongly regulated by increased hydroperiods. Drainage of wetland soil, however, can have a substantial effect on organic matter oxidation and compaction (e.g., Portnoy and Giblin, 1997), but diversions are more likely to increase the hydroperiod and magnitude of flooding in receiving basins.

Accretion rates are predicted to increase with increasing inundation up to a threshold above which plant productivity declines due to flooding stress (e.g., Kirwan et al., 2010; Day et al., 2011). Increased hydroperiod allows greater time for sediments to deposit on the marsh surface, which stimulates positive feedbacks between plant productivity and sediment trapping. At very low elevations, however, soil can remain fluid and deposited sediments are not retained on the marsh surface (Day et al., 2011). Furthermore, when sediment supply is limited, increased inundation can lead to a reduction in plant productivity, and therefore, sediment trapping capacity. Thus, the accretionary response to inundation largely depends on sediment availability.

2.3. Changes in the salinity regime

Depending on the location of the diversion and salinity of the receiving bays, emergent brackish and saline marshes are expected to experience a pronounced reduction in salinity in response to freshwater inflows. Salt water is a constraint for growth and productivity for most halophytes, which utilize energetically taxing physiological mechanisms for excluding, storing or exuding excess salts (Flowers et al., 1977). Many salt marsh plants including *S. alterniflora* have higher productivity under freshwater compared to salt water conditions (Nestler, 1977), possibly due to increased nutrient uptake (Morris, 1984). Therefore, a reduction in mean salinity following a diversion may cause an initial increase in productivity (DeLaune et al., 2005). However, salt tolerant plant species are less competitive under freshwater conditions (Greiner La Peyre et al., 2001), suggesting that fresh and oligohaline marsh plants may replace mesohaline communities under the influence of river diversions. However, extreme salinity fluctuations rather than average salinity conditions tend to be more important in controlling species zonation (Chapin et al., 1993; Bokhorst et al., 2007), survival (Van Peer et al., 2004), and productivity (Knapp et al., 2002).

Planning for a large Mississippi River sediment diversion into the brackish marshes of mid-Barataria basin is currently underway (Pahl et al., 2019 this volume). Barataria Bay marshes currently experience salinities that approach and sometimes exceed 15 during autumn due to saltwater intrusion brought about by elevated sea levels that are typical during that time of year. Mitigating these autumn salinity spikes with freshwater influxes through diversions may be challenging because a positive head differential (i.e., river water surface elevation exceeds estuarine water surface elevation) is required to discharge river water through diversion structures into estuarine receiving basins. During autumn, this head differential typically approaches zero for lower reaches of the river, or in some instances, becomes negative as a result of seasonally low river stages, which are coincident with elevated estuarine water levels (Fig. 6). Several studies have shown decreased

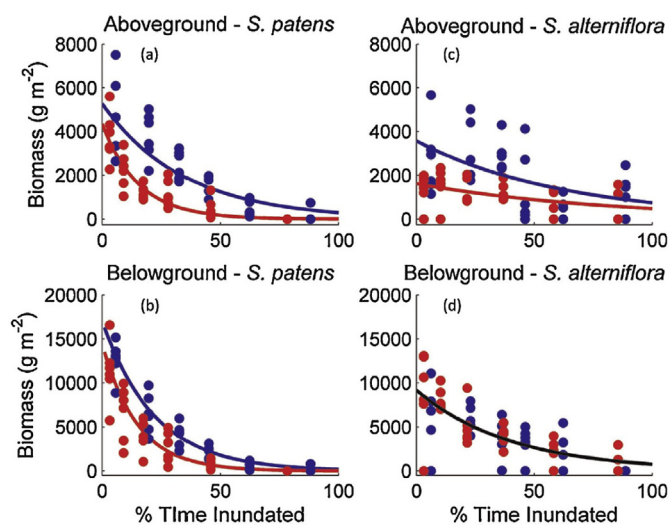


Fig. 5. Aboveground (top) and belowground (bottom) biomass for *Spartina patens* (left) and *Spartina alterniflora* (right) for brackish (blue) and saline (red) sites. Data values are indicated by circles; exponential regressions are represented by lines. Only one line exists for *S. alterniflora* belowground biomass because there were no differences for brackish and saline responses so they were pooled together. (Ecological Engineering, Vol 81, Snedden, G.A., Cretini, K., Patton, B., "Inundation and salinity impacts to above- and belowground productivity in *Spartina patens* and *Spartina alterniflora* in the Mississippi River deltaic plain: Implications for using river diversions as restoration tools" Pages 133–139, Copyright, 2015, with permission from Elsevier).

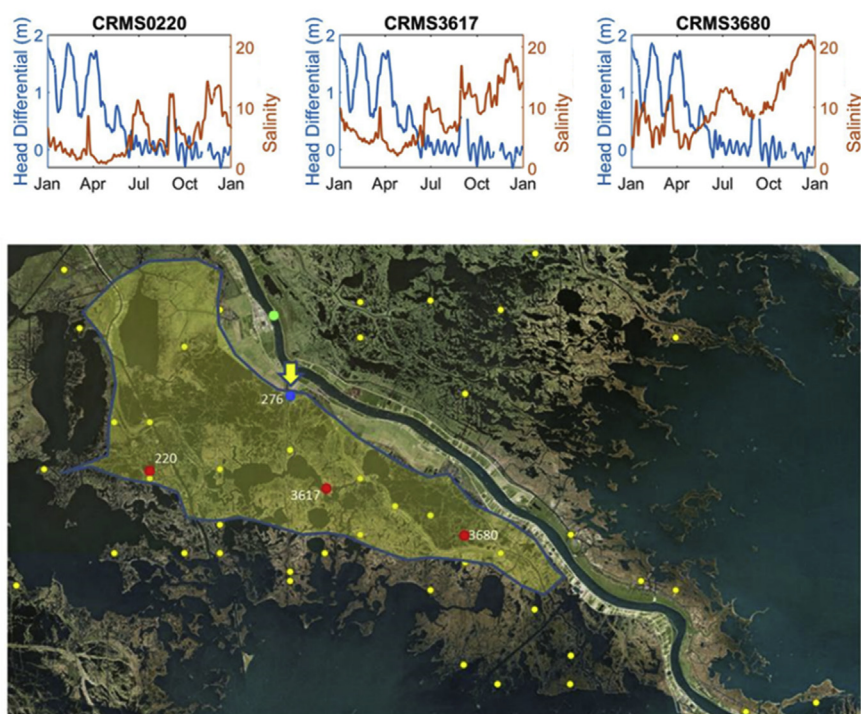


Fig. 6. Top panel: head differential (blue) and salinity (orange) during 2012 for three Coastwide Reference Monitoring System stations. Head differential was calculated as the difference in water surface elevations between the Mississippi River at Alliance and CRMS0276. Lower panel: locations of Mississippi River at Alliance (green circle), CRMS0276 (blue circle), the three CRMS stations where salinity data were obtained (red circles), and the proposed mid-Barataria sediment diversion (yellow arrow). The yellow shaded polygon indicates the region expected to experience > 0.15 m of diversion-induced flooding (https://thewaterinstitute.org/assets/docs/reports/Meselhe_Delft-Hydro_Morph_Production_Runs.pdf).

performance (Baldwin and Mendelsohn, 1998; Greiner La Peyre et al., 2001), tissue damage and chlorosis (Holm and Sasser, 2001), or outright lethality (Pezeshki et al., 1987; McKee and Mendelsohn, 1989; Shaffer et al., 2009) of fresh to intermediate wetland plant species at typical autumn salinity levels found in brackish marshes of the Mississippi River Delta. Therefore, plant community transitions to oligohaline assemblages may not occur as a result of diverting river water into brackish settings. In general, we have a limited understanding of the effects of extreme salinity fluctuations on plant productivity and community dynamics. Thus, predicting the effects of freshwater input through diversions on wetland plant community composition and productivity may be a challenge due to seasonal salinity fluctuations that tend to increase in the bays and estuaries in the fall when river water is lowest and diversions may not be open.

The effects of freshwater introductions through river diversions on organic matter decomposition in brackish and salt water wetlands have not been well-studied. Conversely, the effects of increased salinity on decomposition in brackish and freshwater wetlands have been given much more attention due to the potential negative impacts of salt water intrusion and sea-level rise. In general, changes in salinity can influence soil organic matter decomposition by affecting the plant community type and associated litter quality (Stagg et al., 2018), biogeochemistry (Weston et al., 2006), and microbial function (Weston et al., 2011; Neubauer et al., 2013). Suggestions that decomposition rates are higher in salt as compared to freshwater wetlands based on comparative studies along salinity gradients are challenged by inherent confounding factors, such as species differences, hydrology, sedimentation rates, and soil types that may also influence decay rates (e.g., Craft, 2007; Stagg et al., 2018). On the other hand, experimental incubations have shown that salt water added to tidal freshwater marsh soil can stimulate carbon dioxide and methane fluxes, organic matter mineralization through sulfate reduction and methanogenesis, and ultimately reduce soil organic carbon within one year (Weston et al., 2011). Longer-term (3.5 yr) additions of salt water to tidal freshwater marsh field plots resulted in higher carbon gas flux rates compared to control plots, though no corresponding changes in soil organic matter content were observed (Neubauer et al., 2013). In a comprehensive review of the effects of salinization of freshwater wetlands, Herbert et al. (2015)

found no clear consensus as to how increased salinity affects soil carbon; some studies found enhanced mineralization while others did not. Variation in responses likely depends on a number of factors including the quality and quantity of the organic matter (Rybczyk et al., 1996; Stagg et al., 2018), ambient edaphic characteristics (Hemminga et al., 1991; Mendelsohn et al., 1999; Morrissey et al., 2014), inundation regime (Chambers et al., 2014), initial soil salinity (Kiehn et al., 2013), wetland type (Chambers et al., 2014; Herbert et al., 2015), methodology (e.g., carbon mineralization to CO₂ and CH₄, extracellular carbon-degrading enzymes, cellulose strips, plant litter bags; see Herbert et al., 2015 and other references cited therein), salinity level (Chambers et al., 2011), and duration of salinization (Neubauer et al., 2013). In sum, the effects of salinity on organic matter decomposition are variable and complicated by multiple factors; therefore, direct influences and subsequent effects will likely depend more strongly on local biotic and abiotic processes, and are thus difficult to predict.

2.4. Increased nutrient-loading

Wetlands have long been recognized for their importance in sequestering and transforming land-based nutrients prior to reaching coastal waters (Nixon, 1980; Reddy et al., 1999). However, coastal wetlands can also be sensitive to excess nutrient loading, especially nitrogen (N), because plant growth is generally N-limited (Sullivan and Daiber, 1974; Valiela and Teal, 1974; Gallagher, 1975). Over the last several decades, coastal wetlands have experienced increased exposure to high nutrient loading in many coastal watersheds. Currently, approximately two-thirds of U.S. estuaries show moderate to severe symptoms of eutrophication, with a high prevalence of eutrophic conditions along the Gulf of Mexico and mid-Atlantic coasts (Howarth et al., 2002; Bricker et al., 2007). In the Mississippi River, for example, while ammonium-N has remained fairly low (typically < 0.1 mg/L, Antweiler et al., 1995), nitrate-N increased ten-fold (0.14–1.45 mg/L) from 1900 to 1996 (Goolsby, 2000), and is now the principal form of N in the River along with organic N. Thus, sediment diversions will alter the availability of key nutrients to primary producers and microbes, and recent findings suggest the possibility of negative impacts on coastal wetland processes (Darby and Turner, 2008a, b; Deegan et al., 2012;

Swarzenski et al., 2008; Wigand et al., 2009; Bodker et al., 2015). Although, because processes, such as plant nutrient uptake, are tightly regulated by soil redox conditions and other factors (Bandyopadhyay et al., 1993), plant responses to nutrient enrichment will also be influenced by inundation, salinity, and sediment availability.

Nitrate-N in floodwaters of river diversions has several potential fates before being incorporated into wetland soils, including denitrification and assimilation by phytoplankton and benthic algae or export from the system. Once in wetland soils, nitrate-N is further subject to denitrification, dissimilatory reduction to ammonium, plant and algal uptake, and subsequent ammonification, deposition, burial, or export. Van Zomeren et al. (2012) studied the relative rate of nitrate removal by plants and microbes in a greenhouse study conducted on marsh soils from Breton Sound, LA, and determined that 36% of the added soluble ^{15}N -labeled $\text{NO}_3\text{-N}$ was present in the soil and plant compartments, while the majority (64%) was removed through gaseous loss via denitrification. In wetlands, ammonium is the dominant pool of plant-available N, whereas nitrate tends to be readily denitrified or converted to ammonium through dissimilatory and assimilatory reduction pathways.

Positive relationships between N addition and aboveground plant productivity have been illustrated by numerous field and greenhouse studies across a range of coastal wetlands including tidal freshwater, intermediate, brackish, and salt marsh types (e.g., Valiela and Teal, 1974; Valiela et al., 1976; Mendelssohn, 1979; Morris and Bradley, 1999; Wigand et al., 2004; DeLaune et al., 2005; Frost et al., 2009; Graham and Mendelssohn, 2010; Shaffer et al., 2015). However, the magnitude of aboveground plant responses to nutrient fertilization is inversely related to ambient productivity; greater biomass stimulation occurs when biomass is low with less of a response when ambient biomass is high (Morris, 1991; Morris et al., 2013).

Compared to aboveground productivity, which tends to increase with nutrient-enrichment, belowground biomass responses are variable (Fig. 7). Fertilization of field plots has been shown to increase (Valiela et al., 1976; Tyler et al., 2007; Darby and Turner, 2008a, b; Hunter et al., 2008; Nelson and Zavaleta, 2012), decrease (Valiela et al., 1976; Darby and Turner, 2008a, b; Hines et al., 2006; Davey et al., 2011; Ket et al., 2011; Deegan et al., 2012; Graham and Mendelssohn, 2014, 2016), or have no effect (Gallagher, 1975; Haines and Dunn, 1976; Buresh et al., 1980; Wigand et al., 2004; Darby and Turner, 2008a, b; Hunter et al., 2008; Anisfeld and Hill, 2012) on belowground biomass. Graham and Mendelssohn (2016) assembled the findings of 18 studies that measured the effects of fertilization on standing stock belowground biomass at 34 coastal wetland sites around the U.S. and found that 18 sites (53%) showed a reduction in belowground biomass, while seven (21%) showed an increase, and 10 (29%) showed no response. Similar to aboveground, the belowground response to fertilization (whether negative or positive) tends to be larger when ambient belowground biomass is lower (e.g., $< 2000 \text{ g m}^{-2}$; Fig. 7). Both N and phosphorus (P) have been implicated in eliciting a negative belowground biomass

response (Ket et al., 2011, Darby and Turner, 2008a,b), but we found no clear trend in belowground biomass response with increasing nutrient dosage (Fig. 7). Variable responses are likely due to differences in the biomass pool measured, study duration, quantity and type of nutrient applied, plant species life histories, and environmental conditions (e.g., salinity, flooding, soil characteristics, rate of sedimentation), which interact to affect nutrient uptake and biomass allocation.

In contrast to standing stock belowground biomass, belowground ingrowth, measured as new biomass accumulation in mesh bags filled with root-free substrate, tends to be stimulated by fertilization (reviewed in Graham and Mendelssohn, 2016). Although only a handful of root ingrowth studies have been conducted in fertilized versus non-fertilized wetland field plots (Valiela et al., 1976; McKee et al., 2007; Langley et al., 2009; Anisfeld and Hill, 2012; Graham and Mendelssohn, 2016), responses were either enhanced biomass accumulation or no difference; lower belowground growth has not been observed in response to fertilization. These findings are supported by a number of mesocosm and greenhouse (pot) studies showing the stimulatory effects of nutrients on belowground growth into unoccupied sediment (Haines and Dunn, 1976; Svengsouk and Mitsch, 2001; Ravit et al., 2007; Merino et al., 2010; Hillmann, 2011; Langley et al., 2013; Watson et al., 2015; Matzke and Elsey-Quirk, 2018).

Differential species strategies for resource capture and utilization can result in plant community shifts when nutrients become available in greater quantities, with potentially important implications for ecosystem function and biodiversity. In general, plant species that invest resources in stress-tolerance tend to expand their distributional range when nutrient resources are available through a release from competition by less stress-tolerant, more competitive species (Bertness et al., 2002). For example, in salt marshes in New England, *S. alterniflora*, a stress-tolerant low marsh species, expanded its distributional range into the *S. patens*-dominated high marsh plant community in response to fertilization, thereby reversing the outcome of competitive interactions under ambient nutrient conditions (Levine et al., 1998; Emery et al., 2001). Negative correlations between *S. patens* abundance and nutrient availability across plot and landscape scales further support this relationship (Levine et al., 1998; Emery et al., 2001; Bertness et al., 2002; Wigand et al., 2004; Wigand, 2008). Furthermore, Fox et al. (2012) found long-term fertilization led to expanded occurrence of *D. spicata* both landward and seaward into the native low-marsh *S. alterniflora* monoculture. Similar changes in competitive relationships were observed in a brackish marsh, where N fertilization stimulated *S. patens* and *D. spicata* biomass relative to *Schoenoplectus americanus*, resulting in a shift in plant species composition (Langley and Megonigal, 2010). In species-diverse freshwater wetlands, nutrient enrichment has also been shown to lead to an overall reduction in species richness with rare species particularly sensitive, and a corresponding increase in exotic species (Barendregt et al., 1995; Keddy, 2000; Houlahan et al., 2006), similar to findings in an oligohaline marsh in coastal Louisiana (Slocum and Mendelssohn, 2008). Others have reported no significant effect of

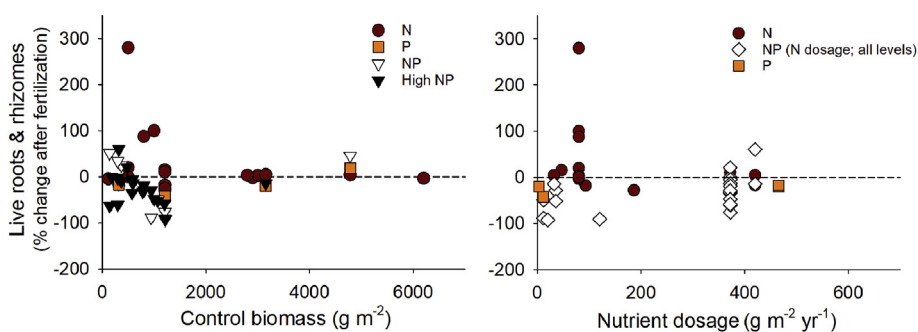


Fig. 7. Percent change in live rhizome and large root biomass in response to fertilization as a function of control or ambient biomass (left) or nutrient dosage (right). Data where live rhizomes and live large roots were not sorted from fine roots and dead biomass were excluded. Studies where both N and P were added were categorized as NP and high NP, where high NP was classified as $\text{N} \geq 50 \text{ g N m}^{-2} \text{ yr}^{-1}$ and $\text{P} \geq 10 \text{ g P m}^{-2} \text{ yr}^{-1}$. NP treatment effects on the % change after fertilization are plotted with respect to the N dosage rather than the P dosage, which was $< 63 \text{ g P m}^{-2} \text{ yr}^{-1}$ in all studies. Data were compiled from Valiela et al., (1976) (*Spartina alterniflora* and *Spartina patens*); Tyler et al., (2007)

(*Spartina alterniflora*); Darby and Turner (2008a, b) (*Spartina alterniflora*); Wigand et al., (2004) (*Spartina patens*); Ket et al., (2011) (*Zizaniopsis miliacea*); Graham and Mendelssohn (2014) (*Sagittaria lancifolia*); Davey et al., (2011) (*Spartina alterniflora*) (n = 39).

nutrient enrichment on species richness, though changes in species dominance patterns were apparent (Graham and Mendelssohn, 2010; Poormahdi et al., 2018).

Nutrient-enrichment of coastal wetlands exposed to sediment diversions from the Mississippi River may also affect secondary consumption of vegetation through herbivory. In the Atchafalaya delta, which receives nutrient-rich water from the Mississippi and Atchafalaya rivers, a near complete eatout of wetland vegetation by the introduced rodent nutria (*Myocaster coypus*) occurred in the late 1970s and early 1980s (Shaffer et al., 1992). Similarly, vegetation cover was rapidly reduced by nutria over a four-month period in the Hammond Assimilation Wetland that receives high nutrient sewage effluent (Hunter et al., 2008; Shaffer et al., 2015). In both of these cases, the vegetation recovered, but under experimental conditions nutria preferentially graze on nutrient-enriched wetland plants (Nyman, 2014) indicating the need for nutria management, particularly in wetlands receiving high nutrient loads through river water input.

High nutrient loading may also affect the rate at which organic matter decomposes. Nutrient enrichment can lead to greater plant uptake and higher tissue nutrient concentrations, such that organic matter quality (e.g., lignin:N), and thereby, decomposability are improved (Valiela et al., 1985; Rybczyk et al., 1996). Rates of leaf litter decay have been directly correlated to initial tissue N content of the litter (Melillo et al., 1984). In addition, higher external availability of nutrients in soil water may increase the efficiency with which microbes use organic matter as an electron donor. A review of 24 wetland decomposition studies by Rybczyk et al. (1996) indicated more consistent and positive decay rate responses when litter quality was improved through internal enrichment compared to when fertilizer was applied to litter externally. However, only the initial stages of decay were sensitive to nutrients, which suggests that the latter stages may be limited more by the availability of labile organic carbon. Thus, the ultimate effect of nutrient enrichment on soil organic matter accumulation may depend on the quantity and physiochemical stability of organic carbon, which can vary among marshes (Unger et al., 2016). Moreover, suggestions that soil organic matter loss through denitrification can lead to wetland deterioration (Kearney et al., 2011; Bodker et al., 2015) are not supported by stoichiometric analysis (Day et al., 2018 this volume). For example, denitrification supported by nitrate in diverted Mississippi River water to Breton Sound marshes could explain less than 1% of marsh decomposition in the area (Van Zomeren et al., 2012). In addition, most soil organic matter is not a suitable substrate for denitrification (Reddy and DeLaune, 2008; Van Zomeren et al., 2012; Day et al., 2018, this volume).

Although many studies have reported an increase in soil respiration rates in response to nutrient addition (e.g., Anisfeld and Hill, 2012; Wigand et al., 2009; Deegan et al., 2012; Wigand et al., 2015), few have reported a co-occurring loss of soil organic matter, accretion or elevation (except see Wigand et al., 2009). Rather, soil respiration rates have been shown to be greater in plots fertilized with N and NP than control plots with no associated loss of soil carbon or elevation (Anisfeld and Hill, 2012). Furthermore, Wigand et al. (2015) found that soil respiration rates were higher in fertilized plots, though CO₂ emission rates increased with increasing rhizome diameter, and declined with declining fine root biomass, suggesting that changes in CO₂ efflux rates were associated with plant respiration and/or greater decomposition of fine root biomass.

Field studies have generally shown positive or null effects of fertilization on marsh accretion and elevation change (Lane et al., 2006; Langley et al., 2009; Anisfeld and Hill, 2012; Morris et al., 2013; Graham and Mendelssohn, 2014). A number of studies have observed that greater aboveground biomass production in response to nutrient enrichment can directly or indirectly affect marsh accretion through increased organic matter input to the soil as litter (Anisfeld and Hill, 2012; Fox et al., 2012; Graham and Mendelssohn, 2014) and a greater capacity to trap sediments due to greater biomass density and height

that reduces flow velocity (Morris et al., 2002, 2013; DeLaune et al., 2005; Mudd et al., 2010; Fagherazzi et al., 2012). In a *Spartina alterniflora*-dominated marsh along Hoadley Creek in Connecticut, combined additions of N and P (but not N or P alone) corresponded with an accretion rate that was 1.5 cm yr⁻¹ greater than control plots (Anisfeld and Hill, 2012). Turner et al. (2009) found no significant change in surface soil accretion, inorganic and organic matter accumulation, or bulk density in a Massachusetts *Spartina alterniflora* salt marsh subjected to 36 years of fertilization. However, the highest fertilization treatment in that study (22.7 g N m² yr⁻¹ and 5.94 g P m² yr⁻¹) had higher accretion and organic and inorganic accumulation rates than paired control plots (Turner et al., 2009). Similarly, fertilized freshwater marsh mesocosms containing 11 species had significantly greater elevation gain compared to unfertilized mesocosms (Hillmann, 2011). Fertilization of field plots in both low marsh *Spartina alterniflora* and high marsh *Spartina patens* in Plum Island, MA, did not have a significant effect on elevation change over a > 10-year period (Morris et al., 2013). Similarly, elevation change rates did not differ from control plots following 13 years of NPK fertilization of an oligohaline marsh in LA because increased shallow subsidence was counterbalanced by a higher rate of surface accretion in fertilized plots (Graham and Mendelssohn, 2014).

While most fertilization studies have focused on plot-level effects in the marsh interior, an ecosystem-level experiment in Plum Island Estuary illustrates negative effects of nutrient-enrichment on creek-banks. Here, dissolved nutrients were added to tidal floodwaters over a nine-year period, producing eutrophic conditions with water nutrient concentrations 10 times ambient concentrations (0.1–1.4 mg/L NO₃-N and 0.16–0.22 mg/L PO₄-P; Deegan et al., 2012). The flooding dynamics in Plum Island Estuary along with continuous fertilization of the tidal water throughout the growing season created extremely high nitrate loading to the creek banks (500–1000 g NO₃-N m⁻² yr⁻¹) with much lower nutrient-loading in the marsh interior (Deegan et al., 2012; Johnson et al., 2016; Day et al., 2018). As a result, nutrient-enrichment led to cascading effects on the biophysical stability of tidal creek banks through a reduction in belowground root and rhizome biomass and an increase in decomposition rates due to lower lignin:N ratios, which led to consolidation and creek bank erosion (Deegan et al., 2012). Although long-term plot-level fertilization studies in New England marsh interiors do not show similar destabilization or conversion to open water at lower N loading rates ranging from 18 to greater than 157 g N m⁻² yr⁻¹ (Valiela et al., 1985; 1976; Giblin et al., 1980, 1986; Fox et al., 2012), creekbanks may be more vulnerable, particularly to extremely high nutrient loading rates. Nutrient loading from Mississippi River diversions will likely be much lower than the experimental loading at Plum Island and many fertilization studies. For example, the nitrate loading rates into Breton Sound, LA, from the opening of a freshwater diversion from the Mississippi River (Caernarvon) were estimated to be 0.9–5.3 g NO₃⁻-N m⁻² yr⁻¹ (Lane et al., 1999; Hyfield et al., 2008).

3. Insights from active deltas and diversions

Shallow coastal bays receiving water through sediment diversions are predicted to evolve over time as sediment accumulates to form islands, which grade in elevation and are intersected by deeper channels. The processes that control the morphological development of these elevation gradients include hydrodynamics, sediment transport, as well as biomass production and sediment trapping. These processes are exemplified in two active deltas associated with the Atchafalaya River, which receives controlled input of Mississippi River water over 350 km north of the mouth of the Mississippi River. The US Army Corps of Engineers ensures that 30% of the Mississippi River flow is diverted to the Atchafalaya River, which is further subdivided by a dredged channel, the Wax Lake Outlet. The mouths of both the Atchafalaya River and the Wax Lake Outlet are forming deltas in the Gulf of Mexico and represent two of the few areas of land gain along the Louisiana coast. Research in

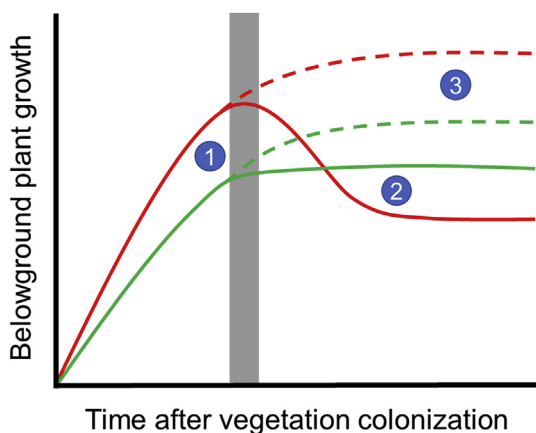


Fig. 8. Hypothesized model of belowground growth under nutrient-limited (green) versus nutrient-enriched conditions (red). Solid lines show (1) plant establishment and initial belowground growth into unexploited soil are stimulated by enrichment with the growth-limiting nutrient, and (2) when the plant community becomes established and maximum aboveground growth is achieved through nutrient enrichment (vertical gray bar), plants equilibrate to nutrient excess by reducing nutrient foraging efforts compared to nutrient-limited growing conditions, while dashed lines show (3) nutrient-enhanced belowground growth as plants exploit open resource space created by sedimentation, annual plant senescence, disturbance, or as plant invasions occur, among other possible conditions promoted by sediment diversions (after Graham and Mendelsohn, 2016).

Wax Lake Delta has shown that the delta development includes the formation of high elevation islands composed of woody plants intertidal and shallow subtidal environments dominated by emergent, floating leaved, and submerged vegetation. The vegetation is dominated by freshwater adapted species, where zonation and composition are controlled to a large extent by elevation as well as rates of sediment deposition and erosion (Johnson et al., 1985; Shaffer et al., 1992; Cahoon et al., 2011; Bevington et al., 2017). Beyond the actively building delta, wetland loss was very low in a broad arc in the central Louisiana coast surrounding the prograding Atchafalaya and Wax Lake deltas (Twilley et al., 2016, 2019, this volume). This implies that in addition to building land, river diversions can limit the loss of adjacent wetlands.

Several freshwater diversions currently connect the Mississippi River to adjacent bays, although most were engineered to regulate estuarine salinity rather than transport sediment. The largest of these freshwater diversions at Caernarvon, LA, became operational in 1991 with a maximum capacity of about 200 m³/sec. Water flows first into Big Mar, a water body that resulted from a failed agricultural impoundment, before entering the upper Breton Sound estuary (Lane et al., 1999, 2006). As a result of river water and sediments discharged through the diversion over a period of 15 years, a small new sub-delta of several hundred ha has formed in Big Mar (Lopez et al., 2014). Similarly, a small sub-delta has formed in the immediate receiving area of the Davis Pond freshwater diversion (Day et al., 2014). Relationships between discharge and the timing and size of delta formation indicate that the lag time for subaerial delta formation and overall area of wetland created may be proportional to discharge.

4. Predictions for river sediment diversions

Proposed large-scale diversions of the Mississippi River will introduce large amounts of fresh water, nutrients, and sediment into coastal wetlands and shallow bays of receiving basins. The response of wetland vegetation to diversions will be dependant on sediment discharge volume in relation to the magnitude of stress regulators (e.g., flooding, salinity) and to resource availability (e.g., nutrients). Increased inundation, particularly during the growing season, and, in

areas where sediment deposition rates are low, may be the greatest concern regarding diversion effects on existing wetlands. Prolonged flooding from diversion openings during the growing season may reduce plant productivity and even cause vegetation death and wetland deterioration in areas receiving low or no additional sediment. This effect may be temporary where sediments accumulate to form deltaic islands that subsequently support wetland vegetation through sedimentation and elevation gain (Cahoon et al., 2011).

The first large-scale sediment diversion project is being planned for implementation in mid-Barataria basin (CPRA, 2017). Recommendations for diversion operations by an expert panel included a gradual increase in diversion operation to promote sediment deposition and limit flooding and erosion of existing wetlands, and seasonal diversion openings to minimize flood risks during the growing season (Peyronnin et al., 2017). Day et al. (2014) suggested that diversions should be large (> 5000 m³/s), but infrequent (active < once a year) to mimic historical functioning of crevasses and sub-delta lobe formation (see also Rutherford et al., 2018). This may help to build land more rapidly, while minimizing negative consequences of inundation and nutrient loading (Day et al., 2014). These recommendations could be better informed by targeted research studies in the future. For example, the development of stable vegetation communities may hinge heavily on the feasibility of mitigating high estuarine salinities that regularly occur during autumn, a time period that typically coincides with reductions or reversals in head differential required to pass water through sediment diversion structures. The potential for seasonal swings in salinity in Barataria Bay exacerbated by a spring fresh water influx through diversions illustrates that diversions may not lead to a first-order freshening of receiving basins, but rather a new salinity regime characterized by greater seasonal variability. The response of wetland plants and soil microbes, therefore, will depend on the timing and magnitude of salinity fluctuations.

The aggregate effects of increased sediment, fresh water, and nutrient delivery via river diversions may increase aboveground productivity of wetland plants, which can promote positive feedbacks that facilitate an increase in sediment deposition and surface accretion. The effect of nutrient loading, in particular, is predicted to depend upon the availability of new substrates resulting from sediment deposition. In areas where diversion-induced sedimentation is limited or absent, nutrient enrichment may result in a decline in standing stock belowground biomass, while belowground growth may be stimulated by nutrient-enrichment in areas where sediments are deposited (Fig. 8; Graham and Mendelsohn, 2016). Field studies in mineralogenic salt marshes of Goat Island in North Inlet, South Carolina, support this hypothesis and showed that sediment input may enhance the belowground biomass response to nutrients (Wigand et al., 2015). Thus, in areas receiving both sediments and nutrients from diversions, root and rhizome biomass (and volume) may increase, thereby contributing to greater rates of plant colonization and overall wetland accretion relative to areas where nutrient and sediment input is limited.

Based on our review of the literature and case studies of river water flowing into bays and estuaries in the Mississippi River Delta, we provide a qualitative assessment of possible ecosystem responses to low and high diversion discharge compared to ambient conditions, i.e., no diversion, in terms of wetland plant productivity, decomposition, and accretion (Fig. 9). Under ambient conditions, wetland plant productivity in Louisiana is relatively high (Hopkinson et al., 1980). Nonetheless, we conclude that, overall, both low and high discharge through diversions will increase the supply of sediment to coastal wetlands and enhance plant productivity, though this response may be diminished to some degree by a simultaneous increase in inundation. Plants receiving nutrient-laden water and a reduction in salinity at a low and high discharge rate may also experience slight increases in total productivity. The anticipated effects of each environmental driver on decomposition are more uncertain because of either variable responses in controlled studies or limited data (Fig. 9, illustrated by polygons with

Degrading deltaic wetlands influenced by river diversions

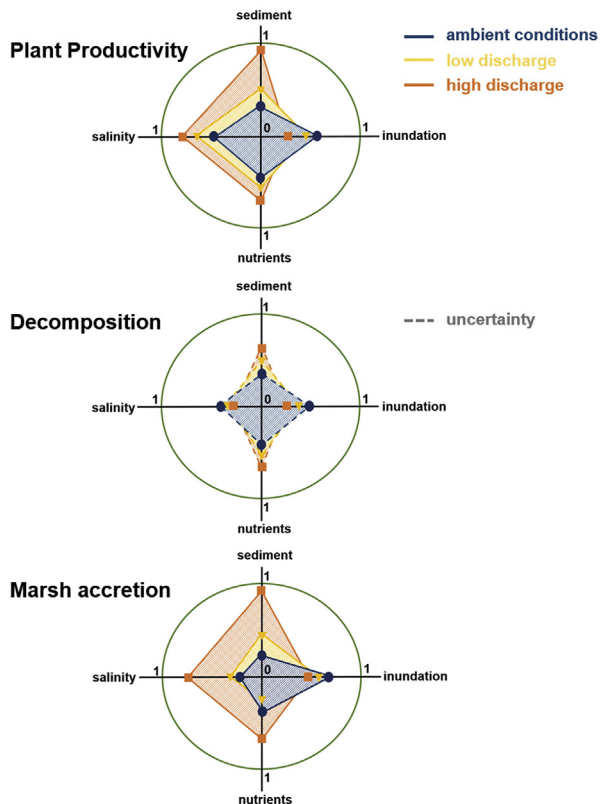


Fig. 9. Radar plots showing the relative magnitude of influence of four key environmental drivers (sediment, inundation, nutrients, and salinity) on wetland plant productivity, soil organic matter decomposition, and accretion (mineral and organic) under ambient, low, and high river discharge conditions. The estimated effect of each driver ranges from high (1) to low (0). The number 1 on each axis represents high productivity, rapid decomposition, and high rates of accretion, while 0 represents low productivity, slow decomposition, and low rates of accretion on respective axes. Dashed outlines of polygons represent areas of greater uncertainty.

the dashed exterior lines). Even so, we predict that the rate of decomposition will increase with increasing sediment deposition with a low and high discharge rate relative to ambient conditions due to increased soil oxygenation. In contrast, decomposition is anticipated to slow slightly compared to ambient conditions with greater inundation. Nutrient supply may increase the rate of decomposition as plant tissues become internally enriched (Deegan et al., 2012), while a reduction in salinity may increase the rate of decomposition by influencing quality and quantity of organic matter and microbial function (Stagg et al., 2018). The rate of soil accretion is anticipated to be high under conditions of elevated sediment supply, reduced salinities, and increased nutrient concentrations that are expected to accompany high discharge events through river diversions. A low discharge river diversion is predicted to increase accretion primarily due to elevated sediment supply, with minimal contributions to accretion occurring from reductions in salinity; however, inundation and nutrient-enrichment may cause a slight reduction in accretion rate relative to ambient conditions. Overall, a high discharge river diversion is predicted to have the greatest positive impact on marsh accretion (Couvillion et al., 2013; Wang et al., 2014), primarily due to high sediment loading and the stimulation of plant productivity resulting from high rates of sedimentation, higher nutrient concentrations, and low salinities. However, the balance of positive influences on production brought about by sediment and nutrient delivery and salinity reduction and the potential negative influence brought about by inundation need to be better

resolved. Predicting the effect of diversions on wetland elevation change is more complicated due to uncertainties in the effects of sediment inputs and spatial variability on subsurface compaction and subsidence.

5. Conclusions

A primary goal of wetland management in areas experiencing wetland loss due to relative sea-level rise and lateral erosion is to maximize vertical accretion and lateral progradation. Sediment input is necessary to supply substrate, reduce soil anaerobiosis, and stimulate wetland plant productivity. Large sedimentation events, which created high elevations, were once an integral part of natural deltaic wetland formation and may again be necessary to counter relative sea-level rise in degraded and subsiding wetlands of the Mississippi River Delta. However, loading rates that supply > 30 cm of sediment at one time may smother existing plants or produce dry saline conditions that reduce overall productivity, though recruitment by seed can foster colonization and growth of high marsh species and, over time, consolidation and subsidence will lower elevations. River diversions may increase inundation in some areas. Areas that receive more water but little sediment may degrade further as a result of river diversions. Freshwater input is likely to have positive overall effects on wetland plant productivity by reducing salinity, and may result in community shifts from more saline and brackish species to species found in more oligohaline and freshwater wetlands. Seasonal fluctuations in salinity, however, may have unknown consequences on plant community composition and productivity. Nutrient loading to wetlands and bays will increase as a result of diverting river water. Nutrient loading tends to stimulate aboveground production, particularly when ambient productivity is low, which in turn, may stimulate sediment-trapping and surface accretion. Belowground productivity may also increase in areas that receive sediment, but decline in areas where sediment input is limited. A number of uncertainties exist regarding the effects of these key environmental factors on decomposition and soil organic matter stocks.

Sediment diversions have the potential to enhance wetland area and sustainability relative to ambient conditions given appropriate controls on diversion operations to maximize sediment delivery (Allison and Meselhe, 2010). We cannot overstate the importance of ensuring high sediment loading to the diversion-affected basins. Otherwise, potential negative effects from prolonged high water and nutrient excess may dominate. While the planned diversions have a real potential to contribute, overall, to a net improvement of wetland sustainability, a number of interacting factors occurring at local and seasonal scales must be considered and closely monitored during implementation. Many of these interactions cannot be fully assessed through small-scale experimentation and thus, diversions will also serve as an important model through which to further test hypotheses and inform future management of Louisiana's coastal wetlands as well as deltaic wetlands throughout the world.

Acknowledgements

Support was provided by the Louisiana Coastal Protection and Restoration Authority Interagency Agreement No. 4400008905 – Task Order 2. We would like to thank C. Swarzenski, D. Day, G. Groseclose, M. Kelsall, A. McClellan, and two anonymous reviewers for their review of the manuscript.

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