

# Hammond Wetland Assimilation Monitoring Report

*February 2019*



**By:** Robert R. Lane, PhD  
Rachael G. Hunter, PhD  
John W. Day, PhD  
Comite Resources  
21245 Smith Rd.  
Covington, LA 70435

**For:** Pete Panepinto, Mayor  
Guy Palermo, Superintendent  
Water & Sewer  
City of Hammond  
P.O. Box 2788  
Hammond, LA 70404-2788

February 25, 2019

[WWW.COMITERES.COM](http://WWW.COMITERES.COM)



Comite Resources, Inc.  
21245 Smith Road  
Covington, LA 70435

John W. Day, Ph.D.  
jwday@comiteres.com

Robert R. Lane, Ph.D.  
rlane@comiteres.com

Rachael G. Hunter, Ph.D.  
rhunter@comiteres.com

225-247-3917

225-439-3931

# Hammond Wetland Assimilation Monitoring Report

## *Summary of Activities: February 2019*

### Boardwalks Need Repair

The boardwalks at the discharge site need to be repaired. Comite Resources personnel are willing to make the repairs, free of charge, if the City provides the lumber. Please call or email Robert Lane (225-247-3917; [rlane@comiteres.com](mailto:rlane@comiteres.com)) to coordinate efforts.

### Turner Rebuttal

The rebuttal to the Turner et al. (2017) paper, which also rebuts the Bodker et al. (2015) paper, has been published in the journal *Wetlands Ecology & Management*. The rebuttal shows that the hypotheses presented by Turner et al. (2017) and Bodker et al. (2015) are flawed and biased, with the underlying premise without supporting evidence. The publication has been appended to this report.

### Site visits

**February 7, 2019:** Comite Resources biologist Jason Day and Joel Mancuso traveled to the Hammond assimilation wetland to conduct monthly monitoring. Discrete water quality parameters (dissolved oxygen, conductivity, temperature, salinity and pH) were measured at the Treatment (BW-1 & BW-4), Mid, Out and Discharge Pipe, as well as the Reference sites M-Con and S-Con (see data below). Leaf litter biomass was collected from each forested site (Treatment, Mid & S-Con). Water level measurements were taken at all sites. Water samples for nutrient analysis and a single BOD sample were collected at each site and the discharge Pipe. Samples were delivered to Pace Environmental in Baton Rouge for analysis. The marsh organ had a staff gauge had a reading of 40.5 cm. Nutria were viewed in the Treatment area.

*Discrete water quality data from the Hammond assimilation wetlands on February 7, 2019.*

Site	Date	DO (mg/l)	Cond (mS)	Temp. (°C)	Sal (PSU)	pH	Water Level (cm)
Pipe	2/7/19	5.4	573.5	19.9	0.3	7.5	.
BW1	2/7/19	1.5	597.7	20.7	0.3	7.2	75.1
BW4	2/7/19	2.8	608.2	24.3	0.3	7.3	33.7
Mid	2/7/19	5.5	456.8	22.8	0.3	7.3	.
Out	2/7/19	3.3	436.4	19.5	0.2	6.9	0.9
M-Con	2/7/19	3.0	262.3	19.6	0.1	7.0	1.0
S-Con	2/7/19	3.8	132.7	21.7	0.1	7.9	22.9

Dissolved oxygen was 5.4 mg/L coming out of the Pipe, and decreased to 1.5 and 2.8 mg/L at BW1 and BW4, respectively, then increased to 5.5 mg/L at the Mid site. DO was between 3-4 mg/L at the rest of the sites. Conductivity ranged from below ~150 mS at the S-Con site to ~610 mS at Boardwalk#4. Water temperature ranged from 19.5 °C to 24.3 °C. Salinity ranged from 0.1 PSU at the reference sites to 0.3 PSU at several of the other sites. Water level was highest at BW1 (75.1 cm), followed by BW4 (33.7 cm) and S-Con (22.9 cm). All these parameters are within expected normal ranges and there are no issues of concern.



*Boardwalk#1 February 7, 2019.*

# Response to: Turner, R.E., J.E. Bodker, and C. Schulz. 2017. The belowground intersection of nutrients and buoyancy in a freshwater marsh. *Wetlands Ecology & Management*: 1–9

John W. Day · Robert R. Lane · Rachael G. Hunter · Gary P. Shaffer

Received: 15 November 2017 / Accepted: 9 November 2018  
© Springer Nature B.V. 2019

**Keywords** Treatment wetlands · Soil strength · Nutria · Wetland restoration

Turner et al. (2017) report on wetland degradation following introduction of secondarily-treated municipal effluent into a freshwater emergent and forested wetland in southeastern Louisiana, referred to as the Hammond assimilation wetland (HAW). They assign the cause of the wetland loss to a combination of increased decomposition and decreased soil strength due to the presence of nutrients from the effluent that led to buoyancy in the marsh soil. They do not, however, discuss or even cite two other papers that have examined the same wetland and have come to different

conclusions (Shaffer et al. 2015; Lane et al. 2015), specifically that nutria herbivory was the main cause of the wetland deterioration (Fig. 1), or a workshop in October 2016 where these issues were discussed in detail. Most importantly, the authors fail to mention or consider that the wetland vegetation began to recover as soon as nutria control was implemented (Fig. 2), though with a different species assemblage most likely due to the combined impacts of herbivory (Shaffer et al. 2015) and perhaps increased water levels (Lane et al. 2015). In general, Turner et al. (2017) selectively cite the literature to support their conclusions. There have been recent concerns that because denitrification, defined as the microbially-mediated reduction of nitrogenous oxides to nitrogen gas, is coupled to the oxidation of organic matter, there is the potential for marsh soil weakening or destabilization as a result of this nitrate addition (Bodker et al. 2015; Turner 2010; Kearney et al. 2011). The observations have primarily been anecdotal or based on simple correlations of nitrate loading and soil strength measurements or measurements of belowground biomass (Darby and Turner 2008a, b, c; Deegan et al. 2012) that do not clearly indicate causation. The two central issues in this paper are the role of nutria in the marsh deterioration and the role of nutrients in causing marsh deterioration. We show below that there is strong evidence that nutria were the primary cause of marsh deterioration at the Hammond assimilation wetland and that based on stoichiometry, the amount of nitrate in the effluent could not explain the observed wetland loss.

---

J. W. Day · R. R. Lane (✉) · R. G. Hunter  
Comite Resources, Inc. (CRI),  
PO Box 66596, Baton Rouge, LA 70896, USA  
e-mail: johnday@lsu.edu

R. R. Lane  
e-mail: rlane@comiteres.com

J. W. Day  
Department of Oceanography & Coastal Sciences,  
College of the Coast & Environment, Louisiana State  
University (LSU), Baton Rouge, LA 70803, USA

G. P. Shaffer  
Department of Biological Sciences, Southeastern  
Louisiana University (SELU), Hammond, LA 70402,  
USA



**Fig. 1** Photo of a fenced 2 × 2-m enclosure that prevented nutria entry established in the wetland in 2008 after intense grazing over the fall and winter of 2007–2008 (left), and another enclosure (center of photo surrounded by wetland vegetation) during the Summer of 2011 (right) after substantial recovery of

the vegetation at the assimilation wetland (note different species assemblage inside the enclosure compared to outside. *Typha domingensis* was the dominant plant in the enclosures). The discharge pipe is located near the trees in the background of both photos



**Fig. 2** Panoramic photograph of the assimilation wetland looking south approximately 100 m from the discharge pipe on September 27, 2017. The area between the boardwalk and the

power lines in the background was open water in 2010 but recovered after nutria removal began

Turner et al. (2017) state that ‘An oligotrophic coastal freshwater marsh converted to open water within months after receiving partially-treated sewage water in fall 2006.’ The assimilation wetland does not receive ‘partially-treated sewage’, but rather secondarily treated and disinfected municipal wastewater. In essence, all municipal effluent is ‘partially-treated sewage’ in the sense that effluent is not treated to the level of drinking water that is the primary water source of the effluent. The term ‘partially-treated’ is not used in the wastewater treatment literature to refer to required levels of treatment. Rather, standards are set for different levels of treatment that are governed by

the type of treatment and the status of the receiving water body.

Turner et al. also state that ‘Consideration should be given to the possibility that the partially-treated sewage can vary in toxicity strength and that toxic effluent spikes could be fatal or produce a profound stress to the trees and various species of marsh vegetation.’ This is speculation and no evidence is presented to show that this has ever happened at the site or at any assimilation wetland in Louisiana or elsewhere. The secondarily-treated and disinfected effluent that is discharged into an assimilation wetland is subject to the same toxicity testing as effluent discharged into an open water body, as per the

requirements of the Louisiana Pollutant Discharge Elimination System permit. Whole effluent toxicity testing with *Daphnia pulex* and *Pimephales promelas* is conducted twice per year, as well as monitoring for magnesium, lead, cadmium, chromium, iron, nickel, silver and selenium. Periodically, all treatment systems are required to carry out a priority-pollutant-scan that includes over 50 organic and inorganic potential pollutants (referred to as PPS). The LDEQ currently requires cadmium, chromium, copper, iron, lead, magnesium, nickel, selenium, silver, and zinc concentrations to be measured at specific time increments for surface water, soils, and vegetation of wetlands receiving treated municipal effluent. Metal concentrations of surface waters at the assimilation wetlands of Breaux Bridge, Thibodaux, St. Bernard, and Hammond have been very low, with most concentrations below the detectable limit at both assimilation and reference wetlands (Hunter et al. 2018). There also have been no detectable differences in metal concentrations for sediments or vegetation between the assimilation wetlands and reference wetlands. Similar results have been obtained for all assimilation wetlands in Louisiana. In general, there is little evidence that toxicity or metal contamination is an issue of concern for assimilation wetlands. Turner et al. (2017) also state that ‘There are other issues, including that the un-managed distribution of pathogens into a nutrient- and organic-rich system is a fertile growth medium for organisms.’ No data are provided to support this statement. The secondarily treated effluent is disinfected using chlorination-dechlorination, so there is no un-managed distribution of pathogens. That is one of the primary goals of the wastewater treatment process.

It is also important to consider in detail the time line of events at the Hammond assimilation wetland when discussing the cause of the wetland deterioration. The wetland did not convert to open water within months of the initiation of effluent discharge. Effluent discharge began in November 2006. During the 2007 growing season, there was robust growth of marsh vegetation with a near doubling of biomass compared to controls, and a pronounced increase in the height of emergent vegetation, which more than doubled (see Fig. 9 from Shaffer et al. 2015). There was no evidence of a negative effect on the vegetation until nutria began to impact the area late Fall of 2007 (Shaffer et al. 2015).

Turner et al. (2017) state ‘The majority of cypress trees planted with protective collars (5000) within the marsh receiving partially-treated sewage either died, floated out of their anchorage, lodged over or manifested signs of abnormal growth (hypertrophy and stunted height). Some cypress trees planted in the firm soil of the pipeline embankment grew well, but other species on this spoil embankment died after the project began’. This is simply inaccurate. As noted above, there was vigorous growth of marsh vegetation in 2007 with much of the emergent vegetation reaching heights of 1.5–2.0 m. Shaffer et al. (2015) concluded that the seedlings died mainly as a result of shading from surrounding vegetation, which grew taller than the seedlings. Hillmann et al. (2018) fertilized baldcypress and water tupelo seedlings in a 16-month mesocosm study at loading rates of 0–400 g N m<sup>-2</sup> year<sup>-1</sup>. Aboveground biomass production increased to 400 g m<sup>-2</sup> year<sup>-1</sup> for both species, whereas belowground biomass production increased to 100 g m<sup>-2</sup> year<sup>-1</sup> then decreased slightly at higher loading rates. Diameter increase for seedlings planted within 100 m of effluent discharge at five assimilation wetlands averaged from 1.1 to 2.5 cm year<sup>-1</sup> over 3–10 years and was about 5–10 times higher than that of nearby natural swamps in the Joyce wetlands. The trees that died were on the low part of the spoil bank and were almost all Chinese tallow (*Triadica sepifera*), an invasive species that has replaced native vegetation in upland areas and is difficult to control. These trees died due to higher water levels and are being replaced by planted baldcypress and water tupelo. We also maintain a nursery of thousands of healthy baldcypress and water tupelo seedlings located adjacent to the discharge pipe and inundated with the effluent. In addition, wetlands have received nutrients from treated municipal effluent for decades in Louisiana and elsewhere without deterioration (Hesse et al. 1998; Brantley et al. 2008; Day et al. 2006, 2018b; Hunter et al. 2009a, b, 2016, 2018).

Based on results from manipulative enclosure experiments, observations of nutria activity, and vegetation recovery after nutria control, it is clear that nutria were the dominant cause of marsh deterioration. Effluent discharge began in November 2006, vegetation grew dramatically through the spring and summer of 2007, then nutria heavily grazed the site within six months (Fall 2007–Spring 2008). Intensive nutria removal began in the spring of 2008 and continued

through the winter of 2008–2009. Approximately 2000 nutria were killed by shooting. Vegetation recovery began during the Spring of 2009 and was most pronounced and consistent nearest the discharge pipe. By 2015, there was considerable recovery of wetland vegetation (Allen 2016). There is now a diverse vegetation community that includes floating aquatics as well as shallow and deep-rooted emergent marsh (Table 1).

The recovery of the site followed a similar trajectory as documented by Izdepski et al. (2009) for the Thibodaux assimilation wetland where floating aquatics first became established followed by rooted emergent floating marsh. When discharge of secondarily treated effluent began in 1992 at the Thibodaux site, there was considerable open water area adjacent to the discharge pipe. In less than a decade, *Alternanthera* and *Hydrocotyle* had grown over much of the open areas and by 2002 *Panicum* had begun to form a thick mat that spread over the area. The marsh has persisted until present. Stable isotope analysis showed that *Panicum* took up nitrate from the effluent and formed the mat (Izdepski et al. 2009).

Turner et al. (2017) state that ‘There was a lack of herbivory damage in April 2009 where the outer boundary of the soil profile was weakened at 50–60 cm depth, and eventually converted to open water.’ Herbivore damage was occurring, as documented by Shaffer et al. (2015) from fall of 2007 through the spring of 2008, and nutria populations were greatly reduced by April 2009.

Turner et al. (2017) question whether the exclosures indicate nutria grazing—‘Wetland-to-open water

conversion through the buoyant uplift and the subsequent movement of floating mats has a consequence to interpreting results from experiments using small exclosures used to experimentally test for herbivore grazing effects. Exclosures keep out the herbivore grazers, but also trap and maintain floating organic matter, *perhaps* (emphasis added) to re-connect to the bottom layer. If the mat rises when disconnecting during flooding water, but cannot float away because of restraint by the exclosure wall, then the continuing presence of emergent vegetation could be interpreted as evidence for herbivore grazing outside the plot, whereas none happened. This problem of misinterpretation is one arising from omitting a disturbed control as part of an experiment. A disturbed control allows access for herbivores, yet maintains the support offered by the wall structure, thereby testing for a ‘cage effect’ on emergent vegetation stability. We have seen exclosure cages at our study area that have one cage wall collapsed, but whose vegetation inside was intact; the area around it was devoid of emergent vegetation. In this case we concluded that herbivore grazing was insignificant.’

Turner et al. (2017) provide this interpretation of the exclosure experiments, however, they do not discuss the experimental design or timeline, and do not have any data to support their claims. The exclosures were constructed on bare mud and the fencing prevented floating organic matter larger than 2–3 cm from entering. Plants in the exclosures were immediately planted and established quickly and were almost completely deeply rooted marsh vegetation, mainly planted *Typha domingensis*, which have roots that

**Table 1** List of species currently growing in the area that was impacted by nutria grazing

Most common species	Other vegetation at the site includes
Giant cutgrass ( <i>Zizaniopsis miliacea</i> ) <sup>a</sup>	Deer pea ( <i>Vigna luteola</i> )
Pennywort ( <i>Hydrocotyle</i> sp.)	Marsh morning glory ( <i>Ipomoea sagittata</i> )
Bulltongue ( <i>Sagittaria lancifolia</i> ) <sup>a</sup>	Walter’s millet ( <i>Echinochloa walteri</i> )
Alligatorweed ( <i>Alternanthera philoxeroides</i> )	Water primrose ( <i>Ludwigia peploides</i> )
Scurgeweed ( <i>Ludwigia leptocarpa</i> )	Climbing hempweed ( <i>Mikanea scandens</i> )
Swamp smartweed ( <i>Polygonum punctatum</i> )	Sheetflow grass ( <i>Panicum gymnocarpon</i> )
Thin-leaved cattail ( <i>Typha domingensis</i> ) <sup>a</sup>	arrow arum ( <i>Peltandra virginica</i> )
	Pickerelweed ( <i>Pontederia cordata</i> )
	American cupscale ( <i>Sacciolepis striata</i> )
	Duck potato ( <i>Sagittaria platyphylla</i> )
	Giant bullrush ( <i>Schoeplectus californicus</i> )
	Soft rush ( <i>Juncus effusus</i> )

<sup>a</sup>Zizaniopsis, Sagittaria, and Typha are deeply rooted

penetrate a meter or more. If the rooted vegetation detached from the soil and floated with rising water, it would tend to fall over and be readily obvious that floating had occurred. This was not the case. In addition, *T. domingensis* repeatedly planted in uncaged paired controls was eaten within 48 h. The authors may have seen vegetation in exclosures where walls were not intact, but this depends on how long the walls were down, and the exclosures were not maintained after the experiment concluded. We have observed instances where a wall was breached and nutria consumed the vegetation within days or weeks. In addition, when larger cages were built (20 m × 20 m), waterfowl landed inside the exclosures and consumed vegetation. The authors made all of their assumptions regarding the exclosures without knowledge of the experimental design or timeline, and without trying to consult with the scientists conducting the experiment.

Widespread damage due to nutria herbivory has been extensively reported for coastal Louisiana (Ford and Grace 1998a, b; Evers et al. 1998; Keeland et al. 2011; McFalls et al. 2010; Sasser et al. 2018; Shaffer et al. 1992). In the Big Branch Marsh National Wildlife Refuge (NWR) located about 50 km east of the Hammond assimilation wetland, a large area of marsh deterioration occurred due to nutria during the same period as the damage that occurred at Hammond (Shaffer et al. 1992; Sasser et al. 2018). Over 7000 nutria were culled by shooting and trapping during 2007 and 2008 at Big Branch Marsh NWR (Table 1 in Sasser et al. 2018). One of the individuals involved in the culling at Big Branch Marsh (Christopher Carrell) also was the primary shooter at Hammond. In total, eight scientists observed and killed nutria at the Hammond assimilation wetland.<sup>1</sup> After the nutria population was reduced, vegetation recovery occurred, with the most robust recovery nearest the discharge pipe. Exclosures that excluded nutria had robust growth, but when nutria were allowed to enter

the exclosures, the vegetation was greatly reduced (Fig. 5 from Shaffer et al. 2015). Similar exclosure experiments have demonstrated the impacts of nutria in the Atchafalaya delta, Bayou Penchant wetlands, and other areas in Louisiana (Ford and Grace 1998a, b; Geho et al. 2007; Gough and Grace 1998a, b; Sasser et al. 2004, 2018; Shaffer et al. 1992). Assimilation wetlands may be more susceptible to nutria damage because nutria preferentially graze on nutrient-enriched wetland vegetation (Ialeggio and Nyman 2014).

Weller and Bossart (2017) reported that insect community diversity tracked the overall condition of the Hammond assimilation wetland over time. Simpson's diversity was highest before degradation occurred, lowest at the height of degradation, and intermediate during the period of partial recovery. Species richness, however, was highest in the partially revegetated marsh community. The community included species characteristic of both the intact and degraded communities, but it shared greatest affinity with the intact marsh. The dominant taxa present in these communities shifted from various beetles to chironomid flies and then back to beetles.

### **Did nutrients increase wetland soil organic matter decomposition at the Hammond site?**

Turner et al. (2017) claim that the damage to the Hammond assimilation wetland was the result of nutrient impacts on vegetation. They draw on Bodker et al. (2015) who reported that nutrients in treated municipal effluent led to the marsh deterioration at Hammond. These studies were developed based on other recent studies suggesting that nutrient additions have deleteriously impacted wetlands (Turner 2010; Kearney et al. 2011). In particular, it has been claimed that denitrification, which is the microbially-mediated reduction of nitrogenous oxides to nitrogen gas, is coupled to the oxidation of organic matter, and this leads to marsh soil weakening or destabilization as a result of nitrate addition to wetlands (Bodker et al. 2015; Turner et al. 2017).

There are many reports of nutrient additions having no or positive impacts on above- and belowground production and decomposition (Haines and Dunn 1976; Valiela et al. 1976; Buresh et al. 1980; Day et al. 2004, 2006; Ravit et al. 2007; Hunter et al. 2009a, b; Carrell 2009; Shaffer et al. 2009; Hillmann

<sup>1</sup> Researchers who observed high nutria populations, some of whom participated in shooting nutria populations included Chris Carrell (SELU), Jason Day (CRI), Eva Hillman (SELU), Montgomery Hunter (CRI), Chris Lundberg (SELU), Joel Mancuso (SELU), Gary Shaffer (SELU), Bernard Wood (SELU). Mr. Carrell also participated in the culling of about 7000 nutria at the Big Branch Marsh National Wildlife Refuge during the same period. Nick Stevens (SELU) is currently involved in monitoring and shooting nutria at the site.

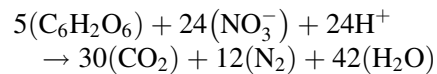


et al. 2015; Anisfeld and Hill 2012; Fox et al. 2012; Zhang et al. 2013; Morris et al. 2013; Graham and Mendelssohn 2014; Steinmuller et al. 2016), while some show negative impacts (Darby and Turner 2008a, b, c; Deegan et al. 2012; Morris and Bradley 1999; Swarzenski et al. 2008; Wigand et al. 2009). The reason for varied responses to nutrient addition is likely associated with variation in localized environmental conditions (e.g., salinity, flooding, soil composition, mineral sediments) that interact to affect nutrient uptake and biomass, and with differences in nutrient loading rates.

Bodker et al. (2015) reported that high nutrients caused increased decomposition that led to the dramatic loss of wetland vegetation at the Hammond assimilation wetland. While there are numerous studies that show both positive and negative impacts of nutrient additions on wetlands, there is no study that shows that nutrient addition led to such a dramatic and rapid loss of wetland vegetation in such a short period of time. Bodker et al. (2015) carried out several short-term experiments (4–6 weeks) that they contend to show that excess nutrients led to the wetland loss. In one experiment, they measured loss of organic matter from marsh soils with added reference water and effluent. The results of their six-week study are consistent with the 18-month litterbag decomposition study by Shaffer et al. (2015) that found no impact of the discharge on decomposition. Typically, litterbag studies find that initial decomposition is rapid followed by a log decrease over time. In addition, the decomposition experiment by Bodker et al. (2015) was carried out at 35 °C ‘to maximize the decomposition rates anticipated during the summer temperature.’ However, 35 °C is much higher than water temperatures measured at the site, which had mean of  $21.9 \pm 0.6$  °C over an 8-year period and never reached 35 °C, and from October to March, 2007–2008 when marsh deterioration occurred, mean water temperature was about 16–17 °C.

In additional experiments, Bodker et al. (2015) incubated several marsh substrates with different nutrient amendments (distilled water, water from a reference wetland, surface water from the wetland receiving treated effluent) and measured gas production over 26 days. Gas production in chambers with effluent was 12% (*Panicum* mat) to 83% (cypress needles) and averaged 34% compared to experiments with water from a reference site. They did not identify

the gas produced or relate gas production to organic matter decomposition. To do this, we carried out stoichiometric calculations that showed that the differences in gas production could explain only a small fraction of observed organic matter decomposition in the experiments. Below is the equation for denitrification (Reddy and DeLaune 2008):



Based on this equation, 30 mol of carbon are utilized for every 24 mol of N. Thus, one mole of  $\text{NO}_3\text{-N}$  (14 g) reduced in denitrification results in the oxidation of 1.25 mol of C ( $12 \times 1.25 = 15$  g) or 30 g organic matter, assuming a 50% carbon content. For the gas production experiments, from 0.7% to 4.9% of the marsh substrate was decomposed with the lowest value being *Panicum* marsh mat. We assumed that all  $\text{NO}_3$  introduced was denitrified therefore our calculations over-estimate organic matter decomposition due to denitrification. The calculations are the same whether  $\text{CO}_2$  or  $\text{CH}_4$  is produced. Day et al. (2018a) calculated the amount of organic carbon needed to support denitrification of all introduced nitrate at the Hammond assimilation wetland. Nitrate concentration drops to background levels ( $< 0.1$  mg/L) within one km of discharge and the smaller area where marsh deterioration occurred (Shaffer et al. 2015). The amount of soil organic matter that could be decomposed annually by denitrification if all  $\text{NO}_3$  were reduced to  $\text{N}_2$  via denitrification would range from 1.5 to 4.7% of the marsh soil organic matter if the marsh soil was the organic substrate used in denitrification. Note that the same results are obtained if the equation is only for the decomposition of organic matter without consideration of  $\text{NO}_3$ . These calculations show that any heterotrophic nutrient interaction in wetlands such as denitrification that involves oxidation of organic matter cannot lead to significant soil organic matter decomposition because ppm concentrations of nutrients demand ppm organic matter while soil organic matter concentration is parts per hundred (Day et al. 2018a).

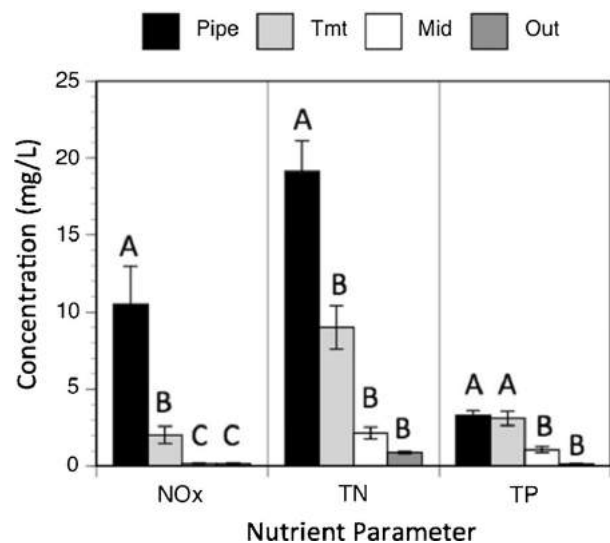
It is unlikely that denitrification could lead to significant marsh soil organic matter decomposition because direct denitrification uses low molecular weight compounds as an organic substrate, and the great majority of soil organic carbon components

cannot serve as an organic substrate for denitrification (Reddy and DeLaune 2008). Labile and low-molecular weight organic matter, such as that found in the treated effluent, can serve as a carbon source for denitrifiers thus organic compounds in the effluent added to the experiments likely served as an organic substrate that yielded gas production. Thus, there are two potential sources of labile organic matter that could support denitrification, organic matter in the wetland soils and that in the municipal effluent. For example, the mean 5-day biochemical oxygen demand (BOD<sub>5</sub>) concentration of municipal effluent is about 30–40 mg/L, which is more than sufficient for the denitrification of the available NO<sub>3</sub>, and is most likely preferable since most organic matter is already in dissolved form and is readily available, not requiring a microbial enzymatic hydrolysis step.

Aboveground productivity in the area receiving the municipal effluent has been measured annually since discharge began and has always been above 600 g m<sup>-2</sup> year<sup>-1</sup> (Shaffer et al. 2015) and often above 1500 g m<sup>-2</sup> year<sup>-1</sup> depending on year and location (LPDES reports for the Hammond assimilation wetland). The contention that tight coupling of denitrification and oxidation of soil carbon stores decreases soil strength and, hence, marsh stability, is not supported by these data or the published literature.

Bodker et al. (2015) also measured relative decomposition at the Hammond assimilation wetland using loss of tensile strength of cotton strings inserted vertically into the soil. Their results ranged from ~ 0.1 to 1.0% day<sup>-1</sup> with an average difference between treatment and reference of ~ 0.3% day<sup>-1</sup>. Cotton-tensile-strength-loss (CTSL) is a measure of cellulose degradation, and most plant roots and leaves are composed of much more than just cellulose. It is a measure of relative decomposition rather than a direct measure of cellulolytic activity (Harrison et al. 1988). By contrast, litter bag decomposition (used by Shaffer et al. 2015) is a direct measure of plant organic matter from a site. The values for CTSL from Bodker et al. (2015) were about an order of magnitude less than values reported for other wetlands in Louisiana and elsewhere that ranged from about 1 to 8% day<sup>-1</sup> (Mendelssohn et al. 1999; Mendelssohn and Slocum 2004; Verhoeven et al. 2001; Slocum et al. 2009; Day et al. 2013), suggesting a possible error in their calculations.

Turner et al. (2017) state that ‘Hunter et al. (2009a) calculated the loading rates at Joyce WMA using the area of 4047 ha at 2.10 g N m year<sup>-1</sup>, which was equivalent to 21 kg N ha year<sup>-1</sup>. But the impact area where land turned to water is much smaller (122 ha) which equates to an annual total N loading rate of about 697 kg N ha year<sup>-1</sup> over the impacted area. The loading rate would be even higher closer to the first exposure to effluent additions. A 10 ha exposure zone at the beginning of waste delivery, for example, would be 8499 kg N ha year<sup>-1</sup>.’ With this logic a loading rate could be extremely large provided smaller and smaller areas are used to calculate it. This is clearly misguided. Only a limited amount of exchange occurs between sediments and the overlying water column, with nitrate disappearing within a few cm (Smith et al. 1982; Reddy and DeLaune 2008). The ‘effective loading rate’ is the loading rate at which nutrient concentrations in the entire water column are processed. It can be determined by the distance needed for surface water nutrient concentrations to drop to background levels, and using the area between that distance and the discharge point as the ‘exposure zone’ to calculate loading rate as Turner et al. (2017) did above. Figure 3 below, for example, shows that the exposure zone for NO<sub>x</sub> is much less than TN since



**Fig. 3** Long-term (2007–2013) water quality data from quarterly sampling from the Hammond Assimilation Wetland ( $\pm 1$  s.e.). NO<sub>x</sub>: nitrate + nitrite; TN: total nitrogen; TP: total phosphorus. Bars with different letters differ according to Tukey–Kramer multiple comparison (from Shaffer et al. 2015). Tmt is 100 m from the discharge, Mid is 1.7 km from the discharge, and Out is ~ 10 km

$\text{NO}_x$  concentrations drop to near background levels by the Mid site ( $\sim 1$  km) compared to the Out site ( $\sim 9.5$  km) for TN. The area of wetland where nitrate is present is larger than the 122 ha area where marsh degradation took place. TN is still present at the Out site located  $\sim 10$  km distant. The area potentially affecting nutrient concentrations is several thousand ha so that the loading rate for TN reported by Hunter et al. (2009a) is correct.

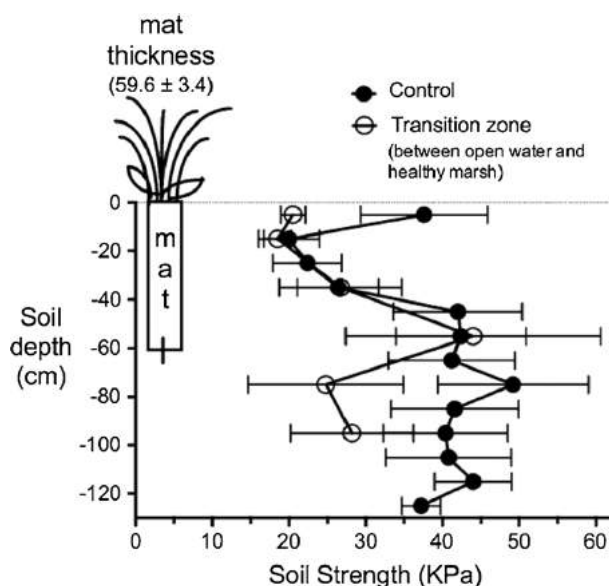
### Soil strength change at the Hammond assimilation wetland

The only quantitative data collected by Turner et al. (2017) were soil strength measurements made with a shear vane (Fig. 4). There were no significant differences in soil strength of the living marsh mat between control and transition zone marshes. Below the living marsh mat, the only depth where standard error bars did not overlap was at 80–90 cm. Thus, there was no impact on soil strength in living marsh soils. These measurements were taken 2 years after marsh deterioration occurred and after significant recovery had taken place. The nutrient data in Fig. 3 show that nutrients were reduced significantly by the time water reached the site measured, and soil strength is different at only one depth (approximately 80–90 cm) and there is high

variability in the measurements. An alternative explanation of marsh buoyancy was that the marsh that floated had been killed by nutria and that decomposition produced gas bubbles that caused the marsh to float during the high water period. Morton and Barras (2011) reported on hurricane impacts on coastal wetlands in Louisiana over a half-century period and created a classification of impact types including floating and redistribution of marsh. Those marshes often recovered to varying degrees after hurricanes, as was the case at Hammond.

Organic matter accumulation and the volume of soil that it generates are primarily functions of the production of refractory organic matter such as lignin, which is not used by denitrifiers as a substrate. The labile fraction of primary production, which is most readily used by denitrifiers, does not significantly contribute to sediment volume (Morris et al. 2014). Based on stoichiometric calculations at the Caernarvon river diversion and the Hammond assimilation wetland, a very small portion of soil organic matter would be decomposed during denitrification if all effluent  $\text{NO}_3$  were reduced (Day et al. 2018a). Thus, it is not likely that nutrient additions are increasing decomposition rates and causing marsh instability.

We know of no example of such rapid dramatic wetland deterioration occurring due to nutrient addition as observed at the Hammond assimilation wetland. There are numerous studies of the impact of nutrients on wetlands, some of which lasted for decades. These include experimental nutrient addition studies, river diversions, and wetlands receiving treated municipal effluent either as planned assimilation wetlands or situations where there was opportunistic discharge that flowed over wetlands. There have been reports of effects of added nutrients, both positive and negative, on above- and belowground biomass, decomposition, soil strength, and accretion, but no study has reported rapid, dramatic deterioration of a wetland such as occurred at Hammond. By contrast, nutria eat outs have occurred often with destruction of hundreds of ha of herbaceous marsh and destruction of bald cypress seedlings (Sasser et al. 2018). The study of Turner et al. (2017) and that of Bodker et al. (2015) show small and subtle direct impacts of nutrients and are consistent with the decomposition study of Shaffer et al. (2015) who found no significant impact of the effluent discharge on above- or belowground decomposition.



**Fig. 4** Soil strength at control and transition zone at the edge of the open water in the Hammond assimilation wetland (from Turner et al. 2017)

## Summary

In summary, Turner et al. (2017) propose that nutrient loading led to marsh deterioration due to elevated levels of belowground organic matter decomposition, which then led to a more buoyant marsh that floated and was misplaced. In drawing their conclusions they selectively cite the literature and make a number of misleading and erroneous statements about the area that are not supported by any data. Turner et al. (2017) rely on an unpublished opinion to dismiss the importance of nutria at the Hammond site. They contend that a hurricane that affected the area in 2012 was a factor, but it occurred 4–5 years after marsh deterioration and during the time wetland vegetation was recovering. The most robust recovery was in the area adjacent to the discharge where nutrient loading was highest. Stoichiometric calculations show that measured nutrient inputs could not have supported the organic matter deterioration that occurred. Though the issues are probably more complex than an argument about whether nutria herbivory or changes in decomposition caused the dieback, the results presented by Turner et al. (2017) are consistent with marsh degradation due to nutria grazing and the subsequent recovery of the marsh.

Although it seems clear that nutria herbivory was the main cause of the marsh deterioration at Hammond and that research carried out at the site do not demonstrate an impact on belowground production or decomposition, nutrient additions may also likely cause a variety of changes to the marsh ecosystem that could take years to unfold (see reviews by Morris et al. 2014; Day et al. 2018a, b). Changes in plant species composition, changes in growth of above- versus belowground plant biomass, and changes in decomposition rates could alter the structure, function, and persistence of the marsh ecosystem. Long-term impacts should be studied to assess the desirability and sustainability of discharging secondarily treated sewage to coastal wetlands, especially marshes to reduce nutrient inputs to receiving waters. Insights into the long-term impact of discharge of secondarily treated municipal effluent on coastal wetlands comes from five assimilation wetlands that have functioned from 27 to 70 years (Hunter et al. 2016; Day et al. 2018b). There are both freshwater forested and emergent wetlands. These studies demonstrate that after decades of discharge, productivity is enhanced,

accretion is increased and nutrient levels are reduced to background levels.

**Acknowledgements** This study was funded by Comite Resources, Inc., which received funding from the City of Hammond. John W. Day, Robert R. Lane, and Rachael G. Hunter acknowledge in the manuscript that they carried out both ecological baseline studies and routine monitoring as employees of Comite Resources, Inc.

## Compliance with ethical standards

**Conflict of interest** JWD, RRL, and RGH acknowledge that they carried out both ecological baseline studies and routine monitoring as employees of Comite Resources (comiteres.com), which received funding from the City of Hammond, however no funds from the city were used for this work.

## References

- Allen Y (2016) Remote sensing of the North Joyce Wetlands from 1952 to 2015. Wetland Assimilation Workshop. Oct 25–26, 2015. Hammond Louisiana
- Anisfeld SC, Hill TD (2012) Fertilization effects on elevation change and belowground carbon balance in a Long Island Sound tidal marsh. *Estuaries Coasts* 35:201–211
- Bodker JE, Turner RE, Tweel A, Schulz C, Swarzenski C (2015) Nutrient-enhanced decomposition of plant biomass in a freshwater wetland. *Aquat Bot* 127:44–52
- Brantley CG, Day JW, Lane RR, Hyfield E, Day JN, Ko J-Y (2008) Primary production, nutrient dynamics, and accretion of a coastal freshwater forested wetland assimilation system in Louisiana. *Ecol Eng* 34:7–22
- Buresh RJ, DeLaune RD, Patrick W (1980) Nitrogen and phosphorus distribution and utilization by *Spartina alterniflora* in a Louisiana Gulf Coast marsh. *Estuaries* 3:111–121
- Carrell CE (2009) Assembly rules and hurricane induced wetland habitat-state change. Master's Thesis. Southeastern Louisiana University, p 84
- Darby FA, Turner RE (2008a) Below- and aboveground *Spartina alterniflora* production in a Louisiana salt marsh. *Estuaries Coast* 31:223–231
- Darby FA, Turner RE (2008b) Below- and aboveground biomass of *Spartina alterniflora*: response to nutrient addition in a Louisiana salt marsh. *Estuaries Coasts* 31:326–334
- Darby FA, Turner RE (2008c) Effects of eutrophication on salt marsh root and rhizome accumulation. *Mar Ecol Prog Ser* 363:63–70
- Day J, Ko J-Y, Rybczyk J, Sabins D, Bean R, Berthelot G, Brantley C, Breaux A, Cardoch L, Conner W, Courville C, Day J, Englande A, Feagley S, Hyfield E, Lane R, Lindsey J, Mistich J, Mitsch W, Reyes E, Twilley R, Yáñez-Arancibia A, Zhang X (2004) The use of wetlands in the Mississippi Delta for wastewater assimilation: a review. *Ocean Coast Manag* 47:671–691
- Day JW, Westphal A, Pratt R, Hyfield E, Rybczyk J, Kemp GP, Day JN, Marx B (2006) Effects of long-term municipal

- effluent discharge on the nutrient dynamics, productivity, and benthic community structure of a tidal freshwater forested wetland in Louisiana. *Ecol Eng* 27:242–257
- Day J, Lane R, Moerschbaecher M, DeLaune R, Mendelsohn I, Baustian J, Twilley R (2013) Vegetation and soil dynamics of a Louisiana estuary receiving pulsed Mississippi River water following Hurricane Katrina. *Estuaries Coasts* 36:1–18
- Day JW, DeLaune RD, White JR, Lane RR, Hunter RG, Shaffer GP (2018a) Can denitrification explain coastal wetland loss: a review of case studies in the Mississippi Delta and New England. *Estuar Coast Shelf Sci* 213:294–304
- Day JW, Hunter RG, Lane RR, Shaffer GP, Day JN (2018b) Long-term assimilation wetlands in coastal Louisiana: review of monitoring data and management. *Ecol Eng* 2:3. <https://doi.org/10.1016/j.ecoleng.2018.09.019>
- Deegan LA, Johnson DS, Warren RS, Peterson BJ, Fleeger JW, Fagherazzi S, Wollheim WM (2012) Coastal eutrophication as a driver of salt marsh loss. *Nature* 490:388–392
- Evers DE, Sasser CE, Gosselink JG, Fuller DA, Visser JM (1998) The impact of vertebrate herbivores on wetland vegetation in Atchafalaya Bay, Louisiana. *Estuaries* 21:1–13
- Ford MA, Grace JB (1998a) The interactive effects of fire and herbivory on a coastal marsh in Louisiana. *Wetlands* 18:1–8
- Ford MA, Grace JB (1998b) Effects of vertebrate herbivores on soil processes, plant biomass, litter accumulation and soil elevation changes in a coastal marsh. *J Ecol* 86:974–982
- Fox L, Valiela I, Kinney EL (2012) Vegetation cover and elevation in long-term experimental nutrient-enrichment plots in Great Sippewissett Salt Marsh, Cape Cod, Massachusetts: implications for eutrophication and sea level rise. *Estuaries Coasts* 35:445–458
- Geho EM, Campbell D, Keddy PA (2007) Quantifying ecological filters: the relative impact of herbivory, neighbors, and sediment on an oligohaline marsh. *Oikos* 116:1006–1016
- Gough L, Grace JB (1998a) Herbivore effects on plant species density at varying productivity levels. *Ecology* 79:1586–1594
- Gough L, Grace JB (1998b) Effects of flooding, salinity and herbivory on coastal plant communities, Louisiana, United States. *Oecologia* 117:527–535
- Graham SA, Mendelsohn IA (2014) Coastal wetland stability maintained through counterbalancing accretionary responses to chronic nutrient enrichment. *Ecology* 95:3271–3283
- Haines BL, Dunn EL (1976) Growth and resource allocation responses of *Spartina alterniflora* Loisel to three levels of NH<sub>4</sub>-N, Fe and NaCl in solution culture. *Bot Gaz* 137:224–230
- Harrison A, Latter P, Walton D (1988) Cotton strip assay: An index of decomposition in soils. ITE Symposium no. 24. Institute of Terrestrial Ecology, Grange-Over-Sands, Great Britain. 176 pp
- Hesse ID, Day J, Doyle T (1998) Long-term growth enhancement of baldcypress (*Taxodium Distichum*) from municipal wastewater application. *Environ Manage* 22:119–127
- Hillmann E, Henkel T, Lopez J, Baker D (2015) Recommendations for restoration: Central Wetlands Unit, Louisiana. Lake Pontchartrain Basin Foundation, New Orleans, p 69
- Hillmann ER, Shaffer GP, Wood WB, Day JW, Day J, Mancuso J, Lane RR, Hunter RG (2018) Above-and belowground response of baldcypress and water tupelo seedlings to variable rates of nitrogen loading: mesocosm and field studies. *Ecol Eng*. <https://doi.org/10.1016/j.ecoleng.2018.08.019>
- Hunter RG, Day JW, Lane RR, Lindsey J, Day JN, Hunter MG (2009a) Impacts of secondarily treated municipal effluent on a freshwater forested wetland after 60 years of discharge. *Wetlands* 29:363–371
- Hunter RG, Lane RR, Day JW, Lindsey J, Day JN, Hunter MG (2009b) Nutrient removal and loading rate analysis of Louisiana forested wetlands assimilating treated municipal effluent. *Environ Manage* 44:865–873
- Hunter R, Day JW, Shaffer G, Lane R, Englande A, Reimers R, Kandalepas D, Wood W, Day JN, Hillmann E (2016) Restoration and management of a degraded swamp and freshwater marsh in coastal Louisiana. *Water* 8(3):71
- Hunter RG, Day JW, Lane RR, Shaffer GP, Day JN, Conner WH, Rybczyk JM, Mistich JA, Ko J-Y (2018) Using natural wetlands for municipal effluent assimilation: a half-century of experience for the Mississippi River Delta and surrounding environs. In: Nagabhatla N, Metcalfe CD (eds) Multifunctional wetlands. Springer, Cham, pp 15–81
- Ialeggio JS, Nyman JA (2014) Nutria grazing preference as a function of fertilization. *Wetlands* 34:1039–1045
- Izdepski CW, Day JW, Sasser CE, Fry B (2009) Early floating marsh establishment and growth dynamics in a nutrient amended wetland in the lower Mississippi delta. *Wetlands* 29:1004–1013
- Kearney MS, Riter JCA, Turner RE (2011) Freshwater river diversions for marsh restoration in Louisiana: twenty-six years of changing vegetative cover and marsh area. *Geophys Res Lett* 38:L16405
- Keeland BD, Draugelis-Dale RO, Darville R, McCoy JW (2011) Effects of herbivory and flooding on reforestation of baldcypress (*Taxodium distichum* [L.] saplings planted in Caddo Lake, Texas. *Tex J Sci* 63:47–68
- Lane RR, Day JW, Shaffer GP, Hunter RG, Day JN, Wood WB, Settoon P (2015) Hydrology and water budget analysis of the East Joyce wetlands: past history and prospects for the future. *Ecol Eng* 87:34–44
- McFalls TB, Keddy PA, Campbell D, Shaffer G (2010) Hurricanes, floods, levees, and nutria: vegetation responses to interacting disturbance and fertility regimes with implications for coastal wetland restoration. *J Coast Res* 26:901–911
- Mendelsohn I, Slocum M (2004) Relationship between soil cellulose decomposition and oil contamination after an oil spill at Swanson Creek, Maryland. *Mar Pollut Bull* 48:359–370
- Mendelsohn I, Sorrell B, Brix H, Schierup H, Lorenzen B, Maltby E (1999) Controls on soil cellulose decomposition along a salinity gradient in a *Phragmites australis* wetland in Denmark. *Aquat Bot* 64:381–398
- Morris JT, Bradley PM (1999) Effects of nutrient loading on the carbon balance of coastal wetland sediments. *Limnol Oceanogr* 44:699–702

- Morris JT, Shaffer GP, Nyman JA (2013) Brinson review: perspectives on the influence of nutrients on the sustainability of coastal wetlands. *Wetlands* 33:975–988
- Morris JT, Nyman JA, Shaffer GP (2014) The influence of nutrients on the coastal wetlands of the Mississippi delta. In: Day JW, Kemp GP, Freeman AM, Muth DP (eds) *Perspectives on the restoration of the Mississippi Delta*. Springer, Dordrecht, pp 111–123
- Morton R, Barras J (2011) Hurricane impacts on coastal wetlands: a half-century record of storm-generated features from southern Louisiana. *J Coast Res* 26:27–43
- Ravit B, Ehrenfeld J, Häggblom M, Bartels M (2007) The effects of drainage and nitrogen enrichment on *Phragmites australis*, *Spartina alterniflora*, and their root-associated microbial communities. *Wetlands* 27:915–927
- Reddy KR, DeLaune RD (2008) *Biogeochemistry of wetlands: science and applications*. CRC Press, Boca Raton
- Sasser CE, Holm GO, Visser JM, Swenson EM (2004) Thin-mat floating marsh enhancement demonstration project, TE-36. Final Report to Louisiana Department of Natural Resources by Coastal Ecology Institute, School of the Coast & Environment. Louisiana State University, Baton Rouge
- Sasser CE, Holm GO, Evers-Hebert E, Shaffer GP (2018) The nutria in Louisiana: a current and historical perspective. In: Day J, Erdman J (eds) *Mississippi Delta restoration*. Springer, Dordrecht, pp 39–60
- Shaffer GP, Sasser CE, Gosselink JG, Rejmanek M (1992) Vegetation dynamics in the emerging Atchafalaya Delta, Louisiana, USA. *J Ecol* 80:677–687
- Shaffer GP, Wood WB, Hoepfner SS, Perkins TE, Zoller J, Kandalepas D (2009) Degradation of Baldcypress-Water Tupelo Swamp to marsh and open water in southeastern Louisiana, USA: an irreversible trajectory? *J Coast Res* 54:152–165
- Shaffer GP, Day JW, Hunter RG, Lane RR, Lundberg CJ, Wood WB, Hillman ER, Day JN, Strickland E, Kandalepas D (2015) System response, nutria herbivory, and vegetation recovery of a wetland receiving secondarily-treated effluent in coastal Louisiana. *Ecol Eng* 79:120–131
- Slocum M, Roberts J, Mendelssohn I (2009) Artist canvas as a new standard for the cotton-strip assay. *J Plant Nutr Soil Sci* 172:71–74
- Smith C, DeLaune R, Patrick W (1982) Nitrate reduction in *Spartina alterniflora* marsh soil. *Soil Sci Soc Am J* 46:748–750
- Steinmuller HE, Graham SA, White JR, McKee M, Mendelssohn IA (2016) A decadal-scale nutrient loading study in a coastal wetland: impacts on soil microbial processes. *Ecol Eng* 97:58–63
- Swarzenski CM, Doyle TW, Fry B, Hargis TG (2008) Biogeochemical response of organic-rich freshwater marshes in the Louisiana delta plain to chronic river water influx. *Biogeochemistry* 90:49–63
- Turner R (2010) Beneath the salt marsh canopy: loss of soil strength with increasing nutrient loads. *Estuaries Coasts* 34:1084–1093
- Turner RE, Bodker JE, Schulz C (2017) The belowground intersection of nutrients and buoyancy in a freshwater marsh. *Wetl Ecol Manag*: 1–9
- Valiela I, Teal JM, Persson NY (1976) Production and dynamics of experimentally enriched salt marsh vegetation: belowground biomass. *Limnol Oceanogr* 21:245–252
- Verhoeven J, Whigham D, van Logtestijn R, O'Neill J (2001) A comparative study of nitrogen and phosphorus cycling in tidal and non-tidal riverine wetlands. *Wetlands* 21:210–222
- Weller MO, Bossart JL (2017) Insect community diversity tracks degradation and recovery of a wastewater assimilation marsh in southeast Louisiana. *Wetlands* 37:661–673
- Wigand CP, Brennan M, Stolt M, Holt, Ryba S (2009) Soil respiration rates in coastal marshes subject to increasing watershed nitrogen loads in southern New England, USA. *Wetlands* 29:952–963
- Zhang Y, Wang L, Xie X, Huang L, Wu Y (2013) Effects of invasion of *Spartina alterniflora* and exogenous N deposition on N<sub>2</sub>O emissions in a coastal salt marsh. *Ecol Eng* 58:77–83