An Analysis of the Hammond Assimilation Wetland: System Response, Nutria Herbivory, and Vegetation Recovery

By

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Abstract

The City of Hammond, Louisiana began discharging secondarily-treated municipal effluent into Four Mile Marsh in the northwestern portion of the Joyce wetlands in the fall of 2006. At the time discharge began, the wetlands south of South Slough had been isolated from virtually all freshwater inflow from the surrounding watershed for over a half century, due primarily to the construction of South Slough. Immediately following effluent discharge in 2006, there was robust growth of herbaceous vegetation. By late fall 2007 the emergent wetlands in the immediate vicinity of the effluent discharge began to decline, and within a year nearly the entire marsh south of the discharge pipe along South Slough had converted to open water or mudflat. By 2010, there had been substantial recovery of the marsh. A number of hypotheses have been presented to explain the conversion of the marsh to open water and mudflats, including increased pH, disease, and reduced belowground biomass and increased soil decomposition due to high nutrients. However, there is no reported instance in the scientific literature of a marsh deteriorating in one year due to nutrient inputs. Here we discuss the various hypotheses and present suggestions for future activities to prevent marsh deterioration from occurring at other herbaceous wetlands receiving treated municipal effluent. Intensive field studies provide the most conclusive data that the marsh loss was primarily caused by the introduced rodent nutria (*Myocaster coypus*), and that recovery is occurring as a result of aggressive nutria control, indicating that nutria control was essential to recovery of the herbaceous vegetation. Negative impacts were not observed for mature baldcypress growing in the area of discharge, where growth rates were greater than 5 times those of trees not receiving effluent in the lower Joyce area and Maurepas swamp. Increased flooding due to lack of drainage from the area is hindering marsh recovery. Water control structures have been installed that will allow water level drawdown and this should lead to enhanced marsh recovery.

Introduction

The City of Hammond, Louisiana, began discharging secondarily-treated municipal effluent into the northwestern Joyce wetlands (Figure 1), in the fall of 2006. Immediately following the discharge there was robust vegetation growth during the following growing season (Figure 2). By late fall 2007, however, the emergent wetlands in the immediate vicinity of the effluent discharge began to decline, and within a year nearly the entire marsh south of the discharge had converted to open water or mudflat. Numerous individuals and environmental organizations began to express concern about this conversion. A number of hypotheses have been presented to explain the conversion to open water and mudflat. Here we discuss these hypotheses and present suggestions for future activities to prevent marsh deterioration from occurring at other herbaceous wetlands receiving treated municipal effluent. It should be noted, however, that the Hammond assimilation wetlands are no longer degrading, and that vegetation is reestablishing, especially near the outfall system. Our studies clearly demonstrate that the marsh loss was primarily caused by the introduced rodent nutria (*M. coypus*) and that recovery occurred as a result of aggressive nutria management.

Numerous individuals and environmental organizations began to express concern about this conversion. In 2010, the Lake Pontchartrain Basin Foundation conducted a day-long workshop on the Hammond discharge site. A meeting summary and presentations from the workshop are available online at SaveOurlake.org. The report contributes further to the scientific discussion of the response of the wetlands to the discharge.

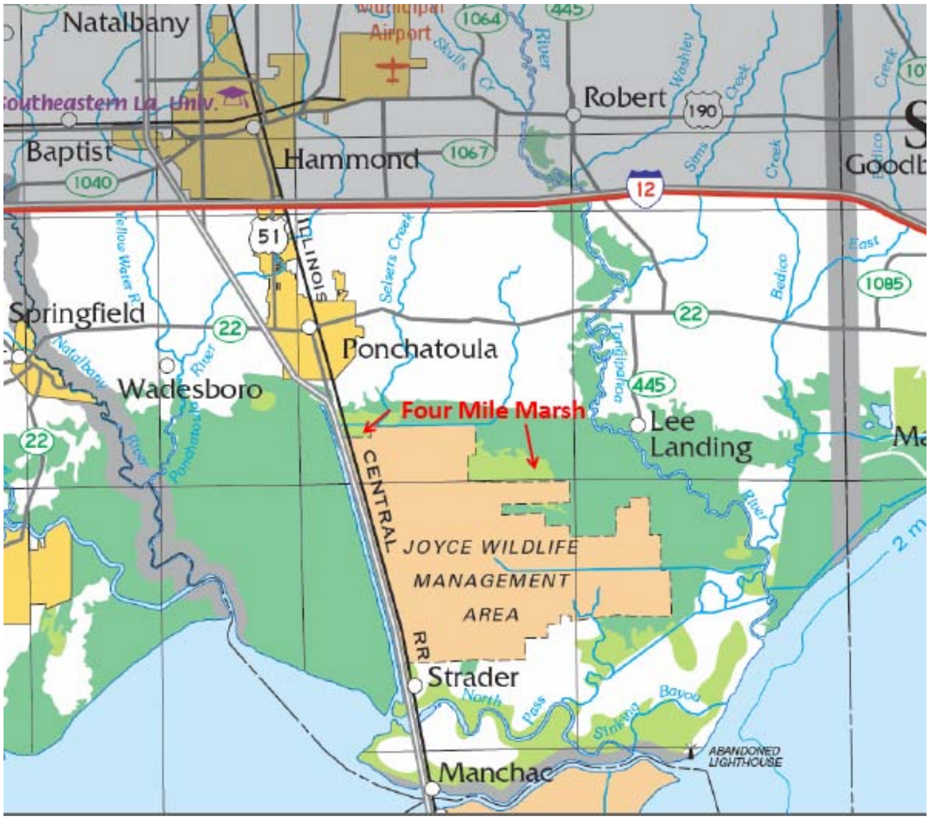


Figure 1. Location of Joyce Wetlands south of Ponchatoula, Louisiana.

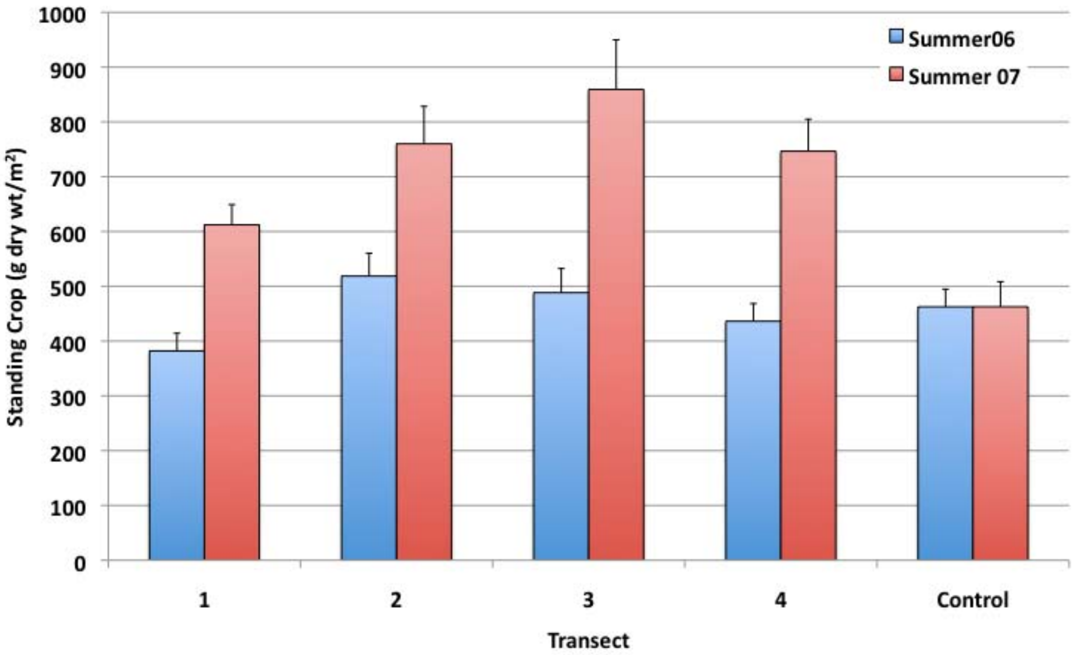


Figure 2. Peak herbaceous aboveground standing crop (g dry weight/m^2) during 2006 (blue) and 2007 (red) growing seasons at subunits 1-4 at the Hammond assimilation wetland and the nearby control site.

The Joyce Wetlands

History of the Joyce wetlands. The Hammond assimilation wetlands are a part of the Joyce wetlands that extend from North Pass, located north of Pass Manchac, to the Pleistocene uplands that begin south of the city of Ponchatoula (Figure 1). They are bordered to the east by the Tangipahoa River, which floods the eastern portion of the wetlands and to the west by the railroad and Interstate 55. The wetland tract originally stretched further west, but the railroad and the canals dug for construction of US-51 and I-55 and accompanying spoil banks hydrologically divided the region.

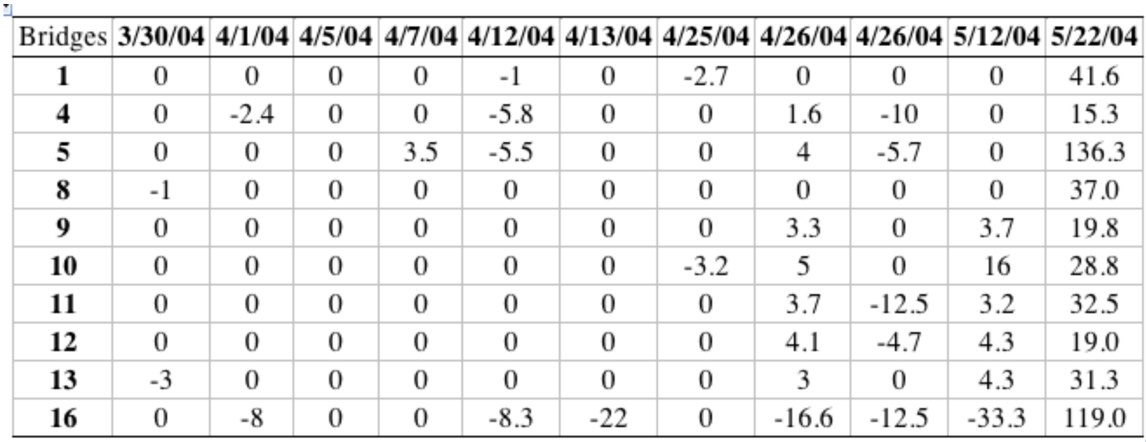
The original hydrology of the Joyce wetlands was characterized by diffuse runoff from uplands directly to the north, and by flow from Selsers and Big Branch creeks, which drain an 86 km^2 watershed that extends north of Hammond. This freshwater input was generally a one-way southerly flow into the wetlands. Water flux into the area from the south was generally tidally driven. A series of well-developed tidal channels along North Pass attest to this tidal influence. Middle Bayou is the largest of these tidal channels, reaching further north than any of the other bayous. Stinking Bayou is located south and east of Middle Bayou. There are no well-defined natural channels from the south in the northern three-fourths of the Joyce wetlands. There are several natural channels, such as Black Bayou, that connect to the Tangipahoa River. These end well east of Seven Mile Marsh and do not directly impact the Hammond assimilation wetlands.

Thus, under pristine conditions, diffuse runoff from uplands to the north and flow from Selsers and Big Branch creeks drained into the upper Joyce wetlands, and slowly moved south. As water flowed closer to North Pass, movement became more bi-directional due to tidal action. The astronomical tide range at North Pass is generally less than 30 cm; however, southeasterly winds can raise water levels by 50 cm or more. On average, there are seven frontal passages per year that lead to such meteorological tides (Day and Shaffer 2009). Hurricanes surge can exceed 1.8 m in the northern portions of the Joyce wetlands. Thus, the generally unidirectional southerly flow from the north changes to a bi-directional flow in the south during frontal setups, but overall net freshwater movement is to the south.

With the exception of Four Mile marsh (herein called the Hammond assimilation wetlands) and Seven Mile marsh, the entire Joyce tract was baldcypress-water tupelo (*Taxodium distichum - Nyssa aquatica*) forest well into the 20th century. There were extensive forested wetlands between North Pass and Pass Manchac, and one of the last remaining old growth forest areas was south of Pass Manchac. Between 1865 and the early 1950s, the cypress forest was harvested, including one tree estimated by timber interests to be 4,000 years old. The cypress trees were first thinned to build a rail line from New Orleans to points north of Lake Pontchartrain (Mancil 1972, 1980). In 1852, the New Orleans Jackson and Great Northern Railroad built a track across the 'Manchac land bridge', the isthmus between Lakes Pontchartrain and Maurepas. When the Civil War broke out, the rail line became a strategic target, with the Manchac land bridge, for a time, being the dividing line between the Confederate and Union armies. Much of the railroad was built on wooden bridges, which were torched by both sides. Cypress harvesting began shortly after the Civil War, when the trees were attractive and convenient for a growing nation looking for building materials. The rebuilt railroad made it easier to move cypress to markets in the Midwest and Northeast. In the 1870s, timber barons began to harvest the gigantic trees, and by the 1890s, steam equipment and pull-boat logging gave the industry more capability to harvest in such difficult terrain (Burns 1980, Keddy et al. 2007). From the late 1800s through the early 1900s, small logging and farming villages thrived on the land bridge (Keddy et al. 2007). The Louisiana Cypress Lumber Co. milled its last cypress log in 1956.

Changes in Hydrology of the Joyce Wetlands. There have been a number of substantial changes over the past century that significantly altered the hydrology of the Joyce wetlands. The earliest impact was the construction of the railroad in the mid 19th century, as described above. The railroad was initially built on a bridge spanning the wetlands, but it was rebuilt on the raised embankment that stands today. Although the embankment has a number of small openings for drainage, it eliminated most east-west water movement. The north-south hydrology described in the previous section remained largely unchanged for nearly a century. U.S. 51 constructed in 1926, parallel to and just west of the railroad, further reduced east-west flow. Generally, culverts were constructed under the road that corresponded to the openings under the railroad embankment, and therefore the hydrology was not altered much from what occurred with the building of the embankment. Measurements show that there is little exchange through these culverts and openings (Table 1) both before and after effluent discharge began. A year-long dye study was conducted where velocity and direction of water flow was measured five times between July 2009 and August 2010. During each dye study, water flow was measured at several points along each board walk south of the discharge pipe, at openings under the railroad, at several points along the Joyce boardwalk, and in the swamp 0.7 km to the east of the discharge pipe. These measurements showed that the primary flow after discharge began was in the southeastern direction. Only after heavy rains or elevated lake levels was there significant flow under the railroad.

Table 1. Water flux through culverts and bridges on the western edge of the JWMA (+= flow into the JWMA, -= flow out of the JWMA (cm/sec)). Bridge #1 is located near North Pass, and #15 is located near South Slough. The Wildlife and Fisheries boardwalk (and the Mid Site) is located between #14 and #15. No data is given for bridges 2, 3, 6, 7, 14 and 15 because flow was measured to be zero.



Bridges	3/30/04	4/1/04	4/5/04	4/7/04	4/12/04	4/13/04	4/25/04	4/26/04	4/26/04	5/12/04	5/22/04
1	0	0	0	0	-1	0	-2.7	0	0	0	41.6
4	0	-2.4	0	0	-5.8	0	0	1.6	-10	0	15.3
5	0	0	0	3.5	-5.5	0	0	4	-5.7	0	136.3
8	-1	0	0	0	0	0	0	0	0	0	37.0
9	0	0	0	0	0	0	0	3.3	0	3.7	19.8
10	0	0	0	0	0	0	-3.2	5	0	16	28.8
11	0	0	0	0	0	0	0	3.7	-12.5	3.2	32.5
12	0	0	0	0	0	0	0	4.1	-4.7	4.3	19.0
13	-3	0	0	0	0	0	0	3	0	4.3	31.3
16	0	-8	0	0	-8.3	-22	0	-16.6	-12.5	-33.3	119.0

Probably the most significant change for the Joyce wetlands was the dredging of South Sough in the early 1950s, which almost completely captured flow from Selsers and Big Branch creeks, as well as diffuse runoff from the north. The spoil generated from dredging of South Sough was placed on the south side of the canal, effectively blocking almost all input of freshwater from the upland watershed. Rainfall on the 86 km^2 watershed generates an average of about 100-million gallons/day (385,000 m^3/day). Rainfall is distributed fairly evenly over the year, but high evapotranspiration during summer (Figure 3) results in most water flowing into the wetlands during the fall (~35%), winter (>50%), and spring (~10%). Heavy rainfall events, such as tropical storms and hurricanes, yield a much higher flows during summer. In addition to these inputs, increasing development in the watershed over the past several decades has resulted in larger volumes of runoff due to increased area of impervious surfaces, such a pavement (Booth and Jackson 1997; Wang et al. 2001; Walsh et al. 2005). Almost all of this freshwater, nutrient, and sediment input to the Joyce wetlands ceased once South Sough was constructed. Because runoff could not flow south into Joyce, wetlands north of South Slough likely flooded more often and with greater depth.

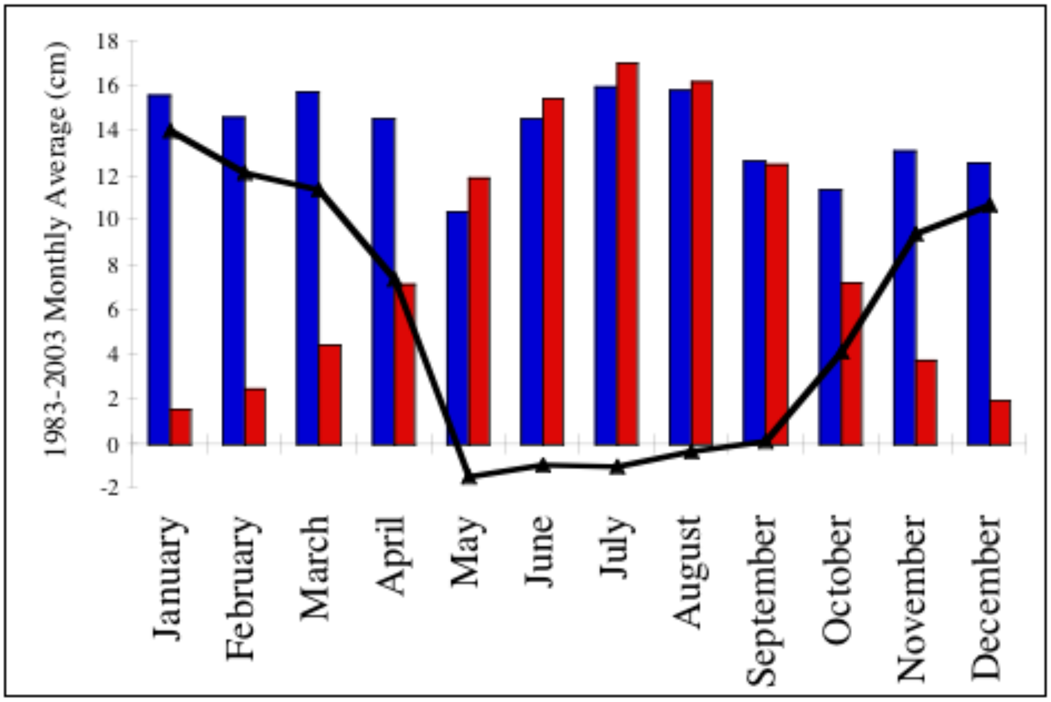


Figure 3. Average rainfall (blue), potential evapotranspiration (red) and net surplus/deficit (line).

Another change was the construction of the Ponchatoula sewage treatment lagoon system north of South Slough in the 1960s. This plant discharges roughly 1 million gallons of secondarily-treated effluent per day (MGD), which flows into a canal that drains into South Slough. Secondary treatment is designed to lower biochemical oxygen demand (BOD) but does not remove nutrients to tertiary levels. Water level fluctuations in South Slough, due to runoff from the watershed and water level fluctuations in Lake Pontchartrain, would have regularly flooded the wetlands north of South Slough. Thus, increased runoff from the watershed, the spoil bank along the south side of South Slough, and discharge from the Ponchatoula treatment plant have likely led to increased flooding and higher nutrient inputs to the wetlands north of South Slough than occurred prior to the construction of South Slough. Our "control" marsh is north of South Slough and therefore receives these higher nutrient inputs and is considerably healthier than most marshes in the Manchac/Maurepas system.

The construction of I-55, and its associated borrow canal, created another major hydrological change for the Joyce wetlands. This 60-m wide and approximately 3-m deep canal now provides a direct, efficient hydrological connection between the northwest part of the Joyce wetlands and Pass Manchac. This has two important effects. First, when lake water levels rise, higher saline water can move rapidly up the canal and enter the northwestern part of the Joyce wetlands. This is especially important during low rainfall periods. Conversely, during high rainfall events, the I-55 canal allows rapid outflow of water from South Slough. Second, rainfall can drain out of the Joyce wetlands into the canal on the western side near the railroad, depriving the wetlands of this freshwater input. It is only during heavy rainfall that there is significant flow under the railroad.

Saltwater intrusion has killed large areas of baldcypress-water tupelo forest in coastal Louisiana. Surface water salinity was 3.5 ppt immediately south of South Slough during summer, 2006, just prior to effluent discharge (Lundberg 2008). Salinity has been a growing problem since the leveeing of the River and opening of the Mississippi River Gulf Outlet in the 1960s (Shaffer et al. 2009a). Practically all baldcypress and water tupelo trees on Jones Island and south of Pass Manchac have been killed primarily due to high salinities (Shaffer et al. 2009b). The lower third of the Joyce wetlands have also experienced a high rate of forest loss due to high salinities (Figure 4). Water Tupelo has been eliminated from the lower two-thirds of the Joyce wetlands due to salinity stress. Tupelo do not occur more than a few hundred meters south of the Joyce boardwalk. This is because tupelo is a strictly freshwater species while baldcypress can tolerate salinities of 3-4 ppt and short-term salinity increases of >5 ppt (Allen et al. 1994, 1996, 1997; Campo 1996; Conner 1994; Conner et al. 1997; McLeod et al. 1996; Shaffer et al. 2009a,b).

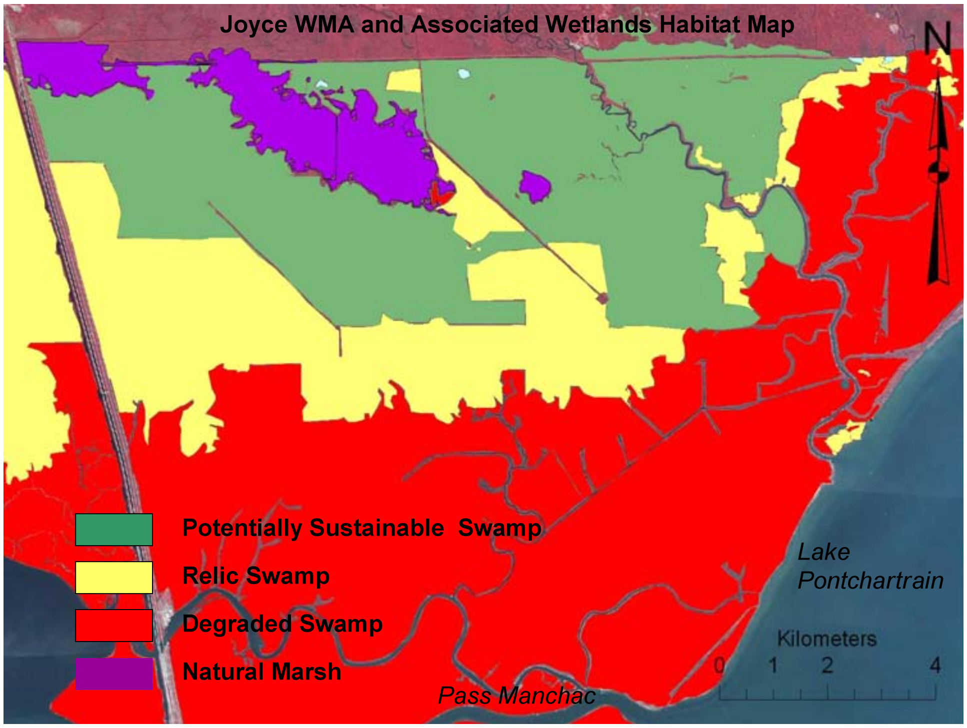


Figure 4. Recent changes in vegetation in the Joyce wetlands. Map was created by overlaying a 1956 vegetation map with 2010 aerial photography. Relic swamp has been impacted by salinity to a level that tupelo were killed.

By the time treated effluent discharge began in November 2006, the wetlands south of South Slough had been isolated from virtually all freshwater inflow from the watershed for over a half century. Other than treated effluent, rainfall is the only important freshwater input into the area. The impact of this isolation from historical watershed inputs, especially on the marshes, is largely unknown. This isolation would have subjected the area of Four Mile marsh to periodically higher salinity and lower nutrient inputs. The impacts of this isolation need to be studied in more detail in order to fully understand the dynamics of the Joyce wetlands.

Timeline of Events to Establish and Monitor the Hammond Assimilation Wetland

- 2004-2005 - Use Attainability Analysis of Four Mile Marsh (hereto referred to as the Hammond Assimilation Wetland) was conducted at the beginning of the 2006 growing season, prior to initiation of the treated effluent outfall system. Measurements were made of vegetation composition and productivity, soil and water chemistry, and hydrology (Day et al. 2004).
- 2006 A baseline study of the Hammond Assimilation Wetland was conducted at the beginning of the 2006 growing season, prior to initiation of the treated effluent outfall system. Four 700 m transects were established along with a nearby control (Figure 5). Data from the baseline study is presented below.
- Fall 2006 - Discharge of treated effluent began.
- 2006-2007 – Began intensive annual monitoring of water and soil chemistry and vegetation species composition and productivity.
- Late fall 2007 - Wetland conversion to open water began. Cut vegetation and dug up roots were observed and attributed to nutria herbivory.
- 2008 - Wetlands directly receiving effluent discharge were nearly completely converted to open water or mudflat.
- 2008 – Ten 16 m^2 nutria exclosures paired with ten controls were planted with nine individuals of southern cattail (*Typha domingensis*).
- 2008 – Cattail in all controls were destroyed within 48 hours of planting while those in cages spread to 100% cover within 2 months.
- 2008-2009 – Greater than 2,000 nutria shot in the Hammond Assimilation Wetlands.
- 2010 – Marsh vegetation reestablishes in greater than 50% of Hammond Assimilation Wetland, especially dense within 200m from outfall area. Dominant vegetation species have shifted since composition was quantified in 2006 (Figure 6).
- 2008-2010 – Growth rates of mature baldcypress near outfall system average 15mm/yr of diameter growth, whereas diameter growth is 2-3mm/yr elsewhere in Manchac/Maurepas system.

- Spring 2010 – Several large (800m^2) exclosures were built to prohibit nutria herbivory but allow waterfowl herbivory. These cages had essentially 100% vegetative cover.
- Fall 2010 – Additional 16 m^2 exclosures were installed at distances from the outfall system ranging from 300 m to 600 m.
- Spring 2011 – Twelve additional 16 m^2 exclosures were installed between 100 m to 250 m. All exclosures and controls were planted with *Schoenoplectus californicus*, *Panicum hemitomon*, and *Pontederia cordata* into Gulf Saver bags, control bags, and directly into the soil.

Figure 5. Map of Hammond assimilation wetland, which receives 4 MGD of secondarily-treated municipal effluent from the Hammond wastewater treatment plant. The area is bordered to the west by the railroad and Interstate-55, to the north South Slough, and to the south by a baldcypress-water tupelo forest. The control site is located to the east, just north of South Slough.

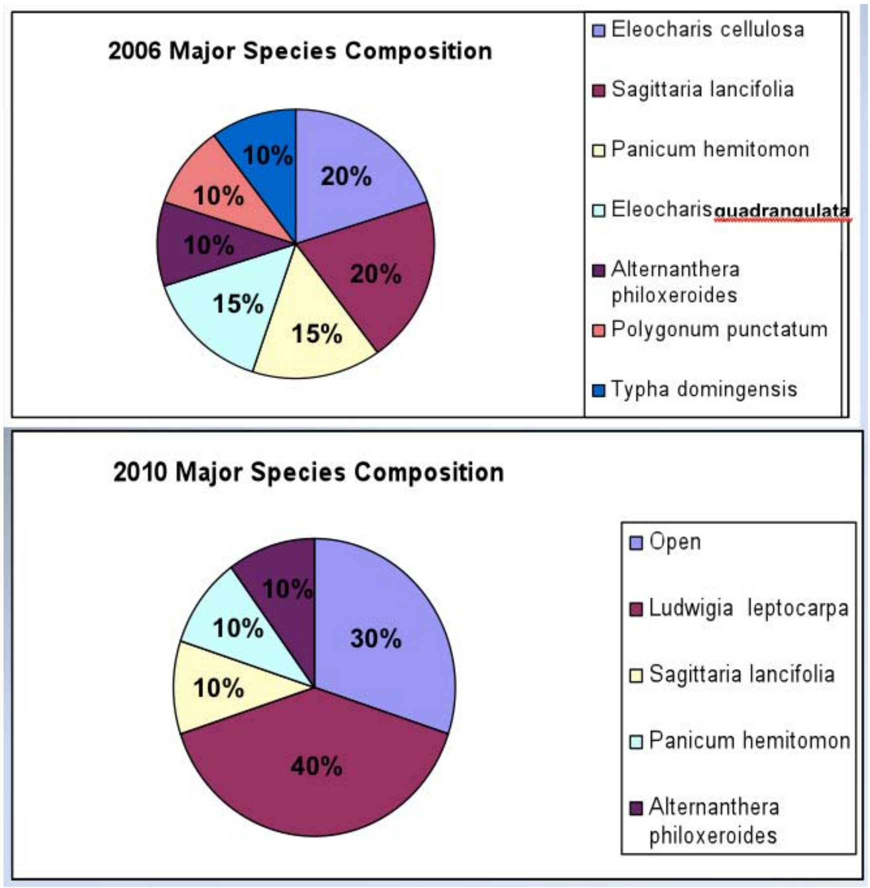


Figure 6. Dominant vegetation species in 2006 and 2010 in the Hammond assimilation wetland.

Issues Raised Following Initiation of the Hammond Assimilation Wetland

Control Sites. Location of the control site in wetlands to the north of South Slough may have introduced an upward bias. As indicated earlier, the wetlands south of South Slough have been largely isolated from watershed inputs for over a half-century. South Slough shunts most water from the watershed west under the railroad to the I-55 canal where it flows rapidly out of the system. The wetlands north of South Slough (see Control in Figure 5) have probably experienced increased water volumes with higher nutrient concentrations due to drainage modifications and increased impervious surfaces in the watershed, as well as inputs from non-point source runoff. As mentioned previously, the wetlands north of South Slough are relatively healthy compared to most in the Manchac/Maurepas system; nevertheless, the biomass production of wetlands receiving municipal effluent was statistically greater than the control wetlands during the first growing season after discharge was initiated (i.e., 2007; Figure 2) and prior to *en masse* invasion of nutria.

Nutrient Concentrations and Loading Rate Calculations. Richardson and Nichols (1985) reported that total nitrogen (TN) of raw sewage ranged from 20-85 mg/l, while TN of secondarily treated effluent ranged from 15-40 mg/l. Total phosphorus (TP) concentrations of raw sewage and treated effluent ranged from 4-15 mg/l and 3-10 mg/l, respectively. The relationship between loading rate and percent uptake of N and P serves as a general design guideline of how much nutrient loading a wetland can assimilate. We have made extensive TN and TP measurements of secondarily treated effluent over the past 20 years. Mean concentration of effluent discharged into assimilation wetlands were about 10 mg/l for TN and 3 mg/l for TP (Hunter et al 2009a).

There are a number of considerations when calculating loading rates for nitrogen and phosphorus. Data presented by Richardson and Nichols (1985) can be used to calculate a range of retention efficiencies from the loading rates of assimilation wetlands. Graphs constructed by Richardson and Nichols (1985) show a non-linear relationship between nutrient loading and percent uptake for a variety of natural and constructed wetlands (Figure 7). In the initial loading rate calculations, the area of the Joyce Wildlife Management Area was used (4047 ha), because that is where most of the discharge ultimately flows. For estimates of assimilation, the total area of wetland that water flows over should be used. In the case of the Hammond assimilation wetlands, there is nearly complete assimilation because of the large area of wetlands (Lundberg 2008, Lundberg et al. 2011). Subsequent hydrology measurements discussed confirm that most of the water flows southerly through the Joyce Wildlife Management Area. To further refine the loading rate analysis, we also calculated loading rates for two additional areas. An area of 400 ha (880 acres) was used to encompass the area between South Slough and the northern part of the Joyce forested wetlands (where the Mid site is located). This is the area where the greatest reduction of nitrogen and phosphorus occurs. Loading rates for TN and TP for this area were 15.7 g N/m^2/yr and 1.6 g P/m^2/yr. We also used an area of 81 ha (200 acres) to approximate the area owned by the City of Hammond. This is the area where the highest loading occurs and where significant reductions in nitrogen and phosphorus concentrations occur. Loading rates for TN and TP for this area were 77.4 g N/m^2/yr and 7.9 g P/m^2/yr.

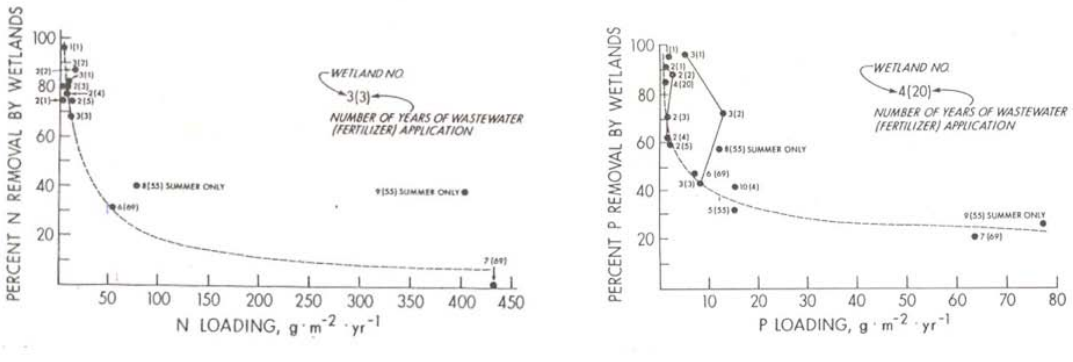


Figure 7. Percent removal of nitrogen (N) and phosphorus (P) as a function of loading rate (Richardson and Nichols 1985).

The 81-ha area where the highest loading rates occur is the area most strongly impacted by the discharge. This is the area where the marsh was degraded (i.e., eaten) first and is also the area that has recovered most strongly. Portions of this wetland have recovered sufficiently during peak biomass at the end of summer to support a person walking on the surface.

Forested Wetlands. Forested wetlands have been used for assimilation for several decades in south Louisiana (Day et al. 2004). The following forested wetland assimilation sites (and years of operation) are currently functioning: Breaux Bridge (60 years), Amelia (37 years), Mandeville (21 years), Thibodaux (18 years), Luling (5 years), and Broussard (4 years). The Luling site received some overflow of treated effluent before the site was permitted. Some of the sites (Breaux Bridge, Amelia, and Mandeville) received treated effluent for considerable times before the discharge was permitted. There has been no measured degradation of any of these sites due to the discharges (Hunter et al. 2009b, Brantley et al. 2008). Where it has been measured, net primary productivity and accretion have been high. Shaffer et al. (2009a) and Conner et al. (2002) documented that 10-15% of trees in coastal forested wetlands are being lost each year in the Maurepas, Barataria, and Terrebonne Basins. In contrast, Izdepski et al (2009) reported that an emergent wetland had become established in what was shallow open water at the Thibodaux assimilation site and mortality of baldcypress proximal to the outfall system at the Hammond assimilation wetland has been zero.

In addition, the site at the Central Wetlands Unit, adjacent to the Gore Pumping Station in St. Bernard Parish, has received input of both pumped stormwater and treated effluent from the Riverbend oxidation pond (since 1962) for over 40 years before effluent discharge was discontinued after Hurricane Katrina (Day et al. 1997). This is the only one of two areas in the Central Wetlands Unit where cypress trees were not killed by saltwater intrusion after construction of the Mississippi River Gulf Outlet (Shaffer et al. 2009b). The other site is located near a stormwater pumping station and receives regular inputs of freshwater.

During winter of 2007, 440 experimental baldcypress seedlings were planted along 700 m transects in Subunits 1-4 and the control site (Figure 2). Eight seedlings were planted and protected from nutria every 50 m from the discharge pipe to 400 m, then every 100 m to the Joyce forest. Diameter stem growth was highest near the outfall and decreased linearly with distance from discharge pipe (Figure 8). The slightly higher elevation near the spoil bank and outfall system may have contributed to higher growth rates there, but higher growth rates also occurred from 50 m to 300 m where there was no higher elevation.

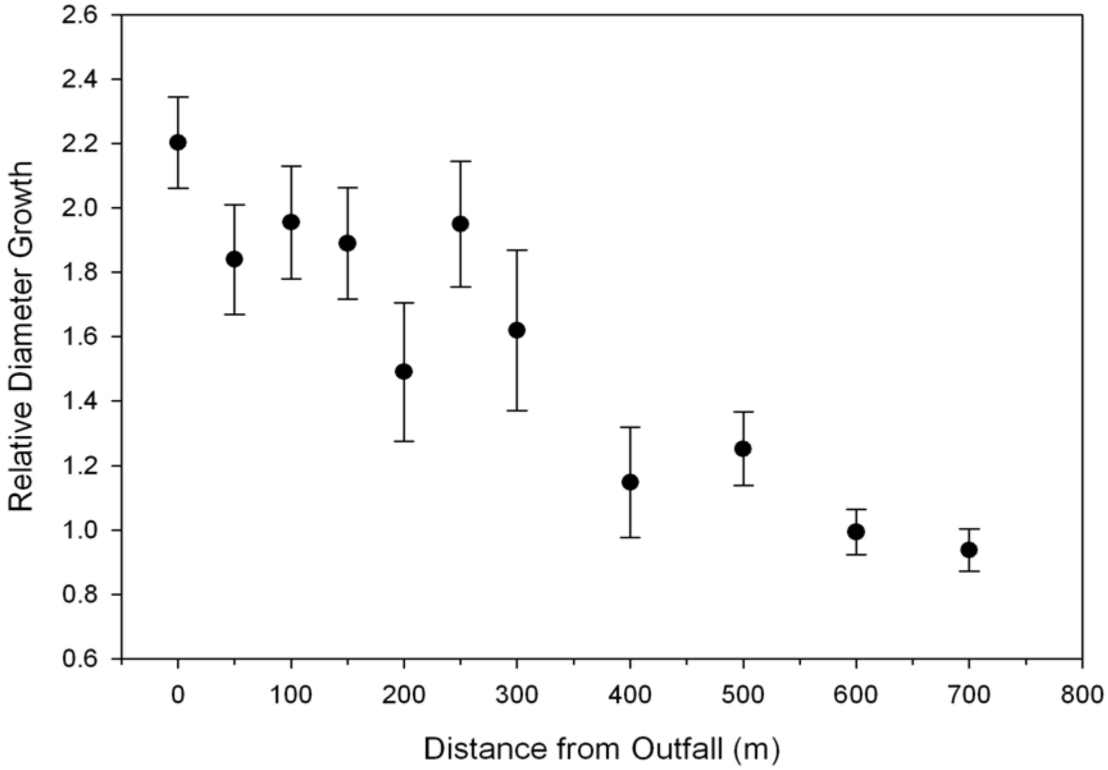


Figure 8. Mean (±1 s.e.) relative basal diameter growth of baldcypress seedlings at various distances from effluent discharge within the experimental subunits 1-4.

Interestingly, inorganic nutrient concentrations followed the same pattern with distance from outfall to the Joyce forest as baldcypress seedling production (Figure 9). Concentrations of ammonia, phosphorus, and silica were nearly undetectable by the time they reached the Joyce forest.

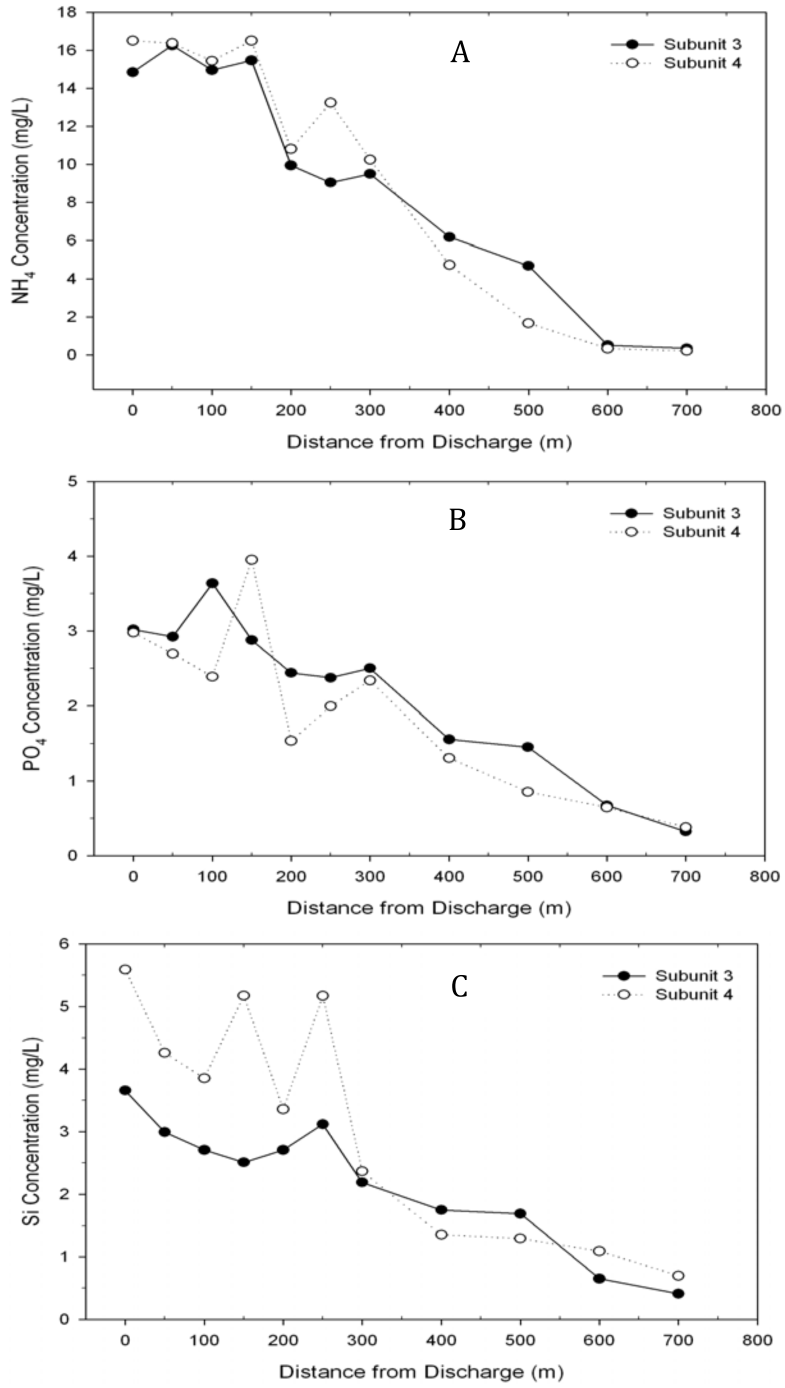


Figure 9. Concentrations of specific inorganic, plant-available compounds (NH_4, PO_4, and Si) at various distances from effluent discharge in the Hammond assimilation wetland.

Several other studies have been conducted in the Manchac/Maurepas system that include the effects of nutrient loading on baldcypress seedlings. Two of these studies (Forder 1995, Boshart 1997) were conducted contiguous with reliable sources of nutrient-rich non-point source runoff and found no difference in diameter increase of control and fertilized seedlings (Table 2). Others in more remote nutrient-poor areas (Myers 1993, Campo 1996, Beville 2002) found statistical increases in diameter growth of baldcypress seedlings under enriched inorganic nutrient levels. None, however, demonstrated increases in growth nearly as great as those of the Hammond assimilation wetland following initiation of discharge (Lundberg 2008).

Table 2. Nutrient enrichment studies of baldcypress seedlings in the Manchac/Maurepas area.

Study	Fertilizer	Seedling Diameter (mm)
Lundberg 2008	Y (effluent)	13.48
	N	6.38
Campo 1996	Y	4.25
	N	2.2
Boshart 1997	Y	4.25
	N	4.5
Forder 1995	Y	2
	N	4
Beville 2002	Y	9
	N	5
Myers 1993	Y	7.5
	N	4.1

Mature Baldcypress. During early 2007, we obtained initial diameters of mature baldcypress located along the discharge system as well as at the northern edge and interior of the Joyce WMA (n = 60). We also followed the stem growth of about 2,225 mature trees in the Maurepas swamp for over the last decade (Shaffer et al. 2009b). Overall, trees at the Maurepas swamp grow an average of 2-3 mm/yr (Figure 10), and those at the Joyce WMA even less. In contrast, mature baldcypress growing near the outfall area of the Hammond assimilation wetland are averaging greater than 5-fold increases in diameter growth. Baldcypress growth in the Bonnet Carré Spillway, which has received floodwaters from the Mississippi about once a decade was more than twice that in the adjacent LaBranche wetlands (Personal communication, Richard Keim, School of Renewable Natural Resources, LSU).

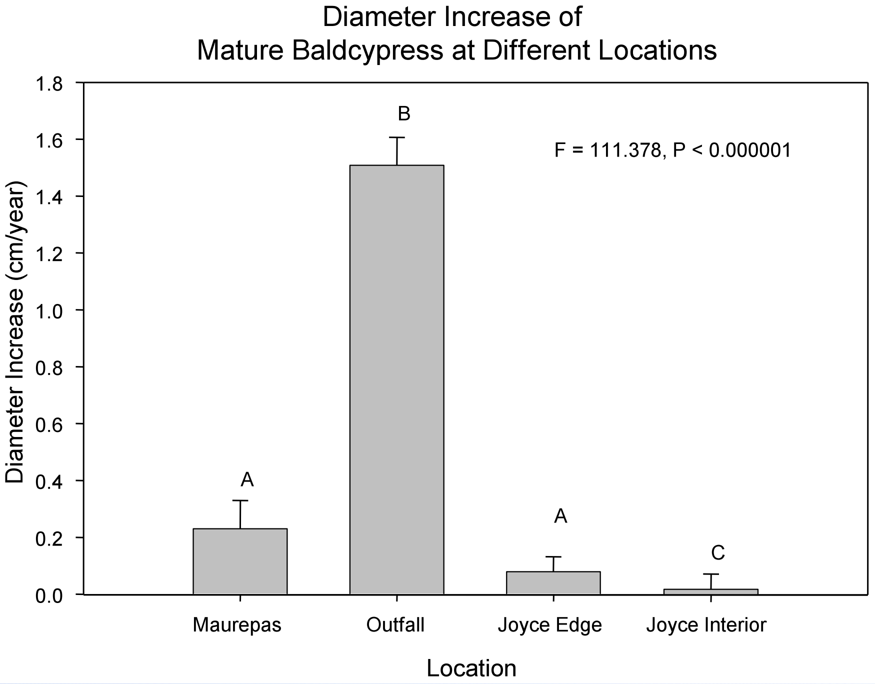


Figure 10. Diameter increase of mature baldcypress growing in the Maurepas swamp, the outfall area of the Hammond assimilation wetland, the northern edge of the Joyce WMA, and the interior Joyce forest.

Level of Treatment Prior to Discharge. Over the past several years, the Hammond Sewage Treatment plant has exceeded its limits for TSS and BOD a number of times. In 2008, the plant experienced problems with BOD and TSS exceeding permitted limits. This occurred when a milk processing facility discharged large volumes of milk that had exceeded the sell-by date into the sewer system. As compared to raw sewage entering a treatment facility that has TSS and BOD generally in the range of 200-300 mg/l, the BOD of milk is greater than 1500 mg/l. In order to address this problem, a pretreatment system was installed at the milk facility and there are no longer problems with excessive TSS and BOD entering the plant. Since that time, the plant has mostly been within permit limits. Although the performance of a treatment plant affects the quality of water being discharged into an assimilation wetland, this problem is one that needs to be addressed at the sewage treatment plant and should not be confused with the performance of the assimilation wetland itself.

Measures of project success. An important issue in the evaluation of wetland assimilation systems is the development of measures of success. There is a need for a mitigation plan if the project fails to meet project goals. Measures of project success are included in the LPDES permit and are dependent upon vegetation, water, and soils monitoring. Because this monitoring occurs both before and after discharge of treated effluent, post-effluent conditions (e.g., species composition, metal concentrations, water nutrient concentrations) can be compared to preeffluent conditions. In addition, conditions in the area receiving treated effluent can be compared to the reference wetland, which is not influenced by the effluent.

The LPDES permit for the Hammond assimilation wetland states
"If wetland monitoring shows that there is:

- More than a 20% decrease in naturally occurring litterfall or stem growth; or
- Significant decrease in the dominance index or stem density of bald cypress;

the permittee shall develop a study and test procedures to determine the origination of the cause within 180 days. A determination shall be made to indicate whether or not the impact to the natural wetland was caused by the effluent. The permittee must demonstrate to the Department of Environmental Quality what has caused the problem within 9 months of the decrease in any of the above required biological criteria and develop a comprehensive plan for the expeditious elimination and prevention of such cause. The plan shall be implemented within 90 days of the determination of the cause. The plan shall provide specific corrective actions to be taken to achieve compliance with the above biological criteria within the shortest period of time." These criteria were developed by LDEQ, in conjunction with wetland ecologists, to protect the health and integrity of the assimilation wetland. At the Hammond site, a series of measurements were made, as described in this study, to determine the causes of wetland deterioration at the site. We also have implemented measures to correct the situation. These include control of nutria herbivory and wetland restoration and are described below.

Hypotheses for the Conversion of the Marsh to Open Water

pH impacting decomposition. *The pH of the influent was higher than the pH of the surface water in the assimilation wetland, and this higher pH increased decomposition rates and impacted the organic matter content and strength of soil.* While it is correct that mean pH in the assimilation wetland was lower prior to discharge (about 5.5-6) than after discharge began (about 7), there were few studies conducted to determine how a small increase in pH would correlate to an increase in decomposition rates. Organic matter decomposition in wetlands is regulated by a number of factors, including dissolved oxygen, temperature, pH, organic matter quality, nutrients, and the availability of other terminal electron acceptors (Craft 2001, Mitsch and Gosselink 2007). In addition, both bacteria and fungi contribute to decomposition, and optimum pH for bacterial decomposition occurs between 6 and 8, while optimum pH for fungal decomposition occurs between 4 and 6. Thus, numerous factors are impacting decomposition. Higher rates of decomposition would be expected after marsh deterioration caused by nutria as there would be high levels of decomposing organic matter. This issue should be investigated in further studies of the area.

Nutria herbivory. *Herbivory, primarily by nutria, was responsible for most of the marsh deterioration at the Hammond assimilation wetland.* Based on data from manipulative experiments as well as observations of nutria activity, it is clear that nutria were a dominant cause of marsh deterioration in the zone near the discharge pipe (Lundberg 2008). After effluent discharge was initiated in November 2006, there was robust vegetative growth at the area receiving discharge, with greatly increased net primary production during the 2007 growing season, due to the increased nutrient and freshwater input (Figure 11).

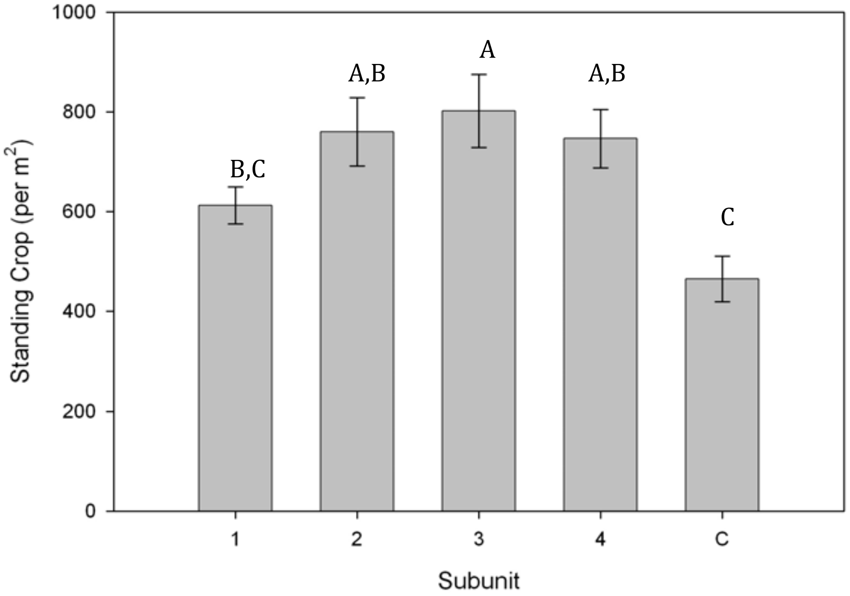


Figure 11. Mean (± 1 s.e.) aboveground herbaceous standing crop of permanent study plots within subunits of the Hammond assimilation wetland and a control marsh during the 2007 growing season. Bars with different letters indicate a Bonferroni-adjusted statistical difference.

Moreover, there was a linear decrease in biomass production from 400-700 m from the outfall system (Figure 12), indicating that nutrients were enhancing biomass in the wetland.

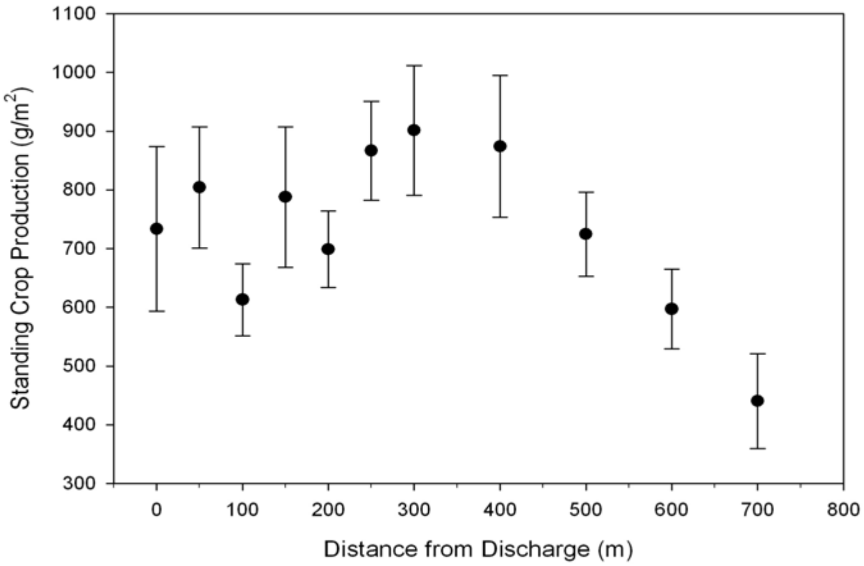


Figure 12. Mean (± 1 s.e.) aboveground herbaceous standing crop of permanent study plots at various distances from wastewater discharge.

By the end of fall of 2007, large numbers of nutria were observed feeding on vegetation in the assimilation wetland. The animals were initially concentrated at Sub-unit 4 (Figure 2), but moved westward as herbivory progressed. Nutria began by devouring vegetation within the first 200 m of the discharge pipe, then moved southerly. By the end of the winter in 2007, the wetland was mostly denuded. At this point, nutria were hunted aggressively, with an estimated 2,000 killed (Chris Carrell, unpublished data), and populations have been maintained at a low density to date through weekly hunting excursions. A similar situation occurred in the Big Branch NWR where about 6000 nutria were killed in one summer by sharp shooters (Chris Carrell, unpublished data). To further demonstrate that the conversion from wetland to open water was caused primarily by nutria, ten 16 m^2 exclosures and 10 controls were constructed and planted with nine individuals of southern cattail (*T. domingensis*) seedlings. Within 48 hours of planting all individuals in the control plots were destroyed, whereas plants in exclosures had 100% survival and spread rapidly. The control plots were replanted several times, and each time suffered 100% mortality due to nutria herbivory.

To quantify belowground plant biomass in the presence and absence of nutria, 50-cm deep, 15-cm diameter, cores were extracted inside and outside of the 16 m^2 exclosures. Belowground biomass was nearly 3-fold higher inside of exclosures ($F = 30.040$, $p = 0.00027$) than in controls (Figure 13).

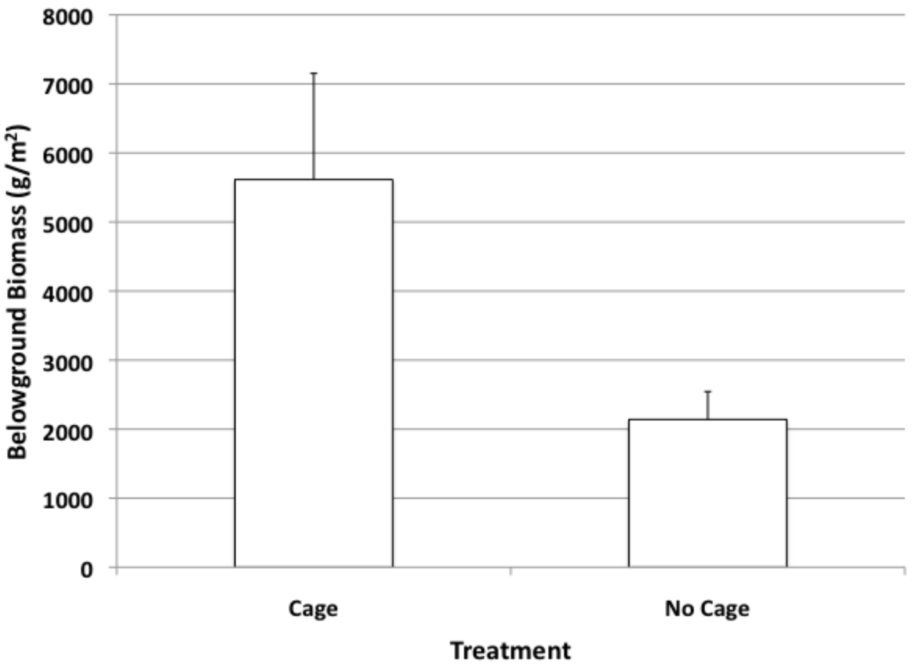


Figure 13. Live belowground biomass (g dry weight/m^2) in the nutria exclosures compared to that of controls.

As a second form of nutria exclusion, 50 cm x 50 cm marsh ‘pillows’ were constructed out of vinyl-coated crab wire, and sprigs of maidencane (*Panicum hemitomon*) were enclosed within, along with foam floatation (Figure 14). Fifty of these pillows were placed along catwalks in Sub-units 1 and 4 (Figure 2). Although both nutria and waterfowl foraged on the vegetation growing through the wire of the pillows, survivorship has been nearly complete and re-growth has occurred from roots inside the pillows.

Figure 14. Photograph of marsh pillow containing maidencane.

Nutria caused species-specific effects as well. Originally, we hypothesized that cattail (*Typha spp.*) would proliferate with increased nutrient loading, eventually replacing all other herbaceous species, especially near the discharge area. This pattern is referred to as 'cattailization' and is a widespread phenomenon (Odum 1988). In contrast, prior to initiation of discharge cattail was homogeneously located throughout the study area (Figure 15, top), whereas one year following initiation of discharge this species became far less abundant near the outfall area (Figure 15, bottom). During the fall 2007 vegetation surveys, clipped cattail was observed in essentially all of the permanent (but uncaged) plots near the discharge area, with the herbivory attributable to nutria.

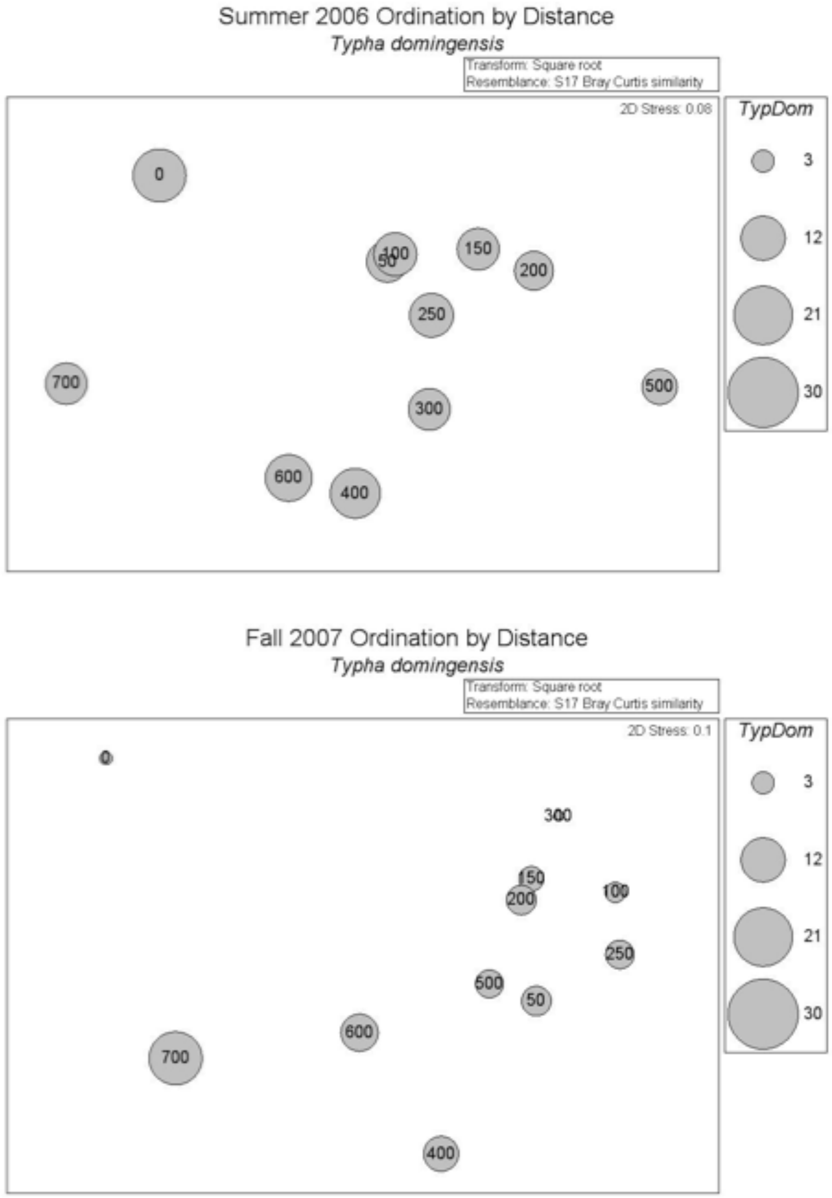


Figure 15. NMDS ordination bubble plot of study sites based on percent cover of southern cattail at the Hammond assimilation wetland. Numbers inside the bubbles indicate distance (m) from the outfall system and the size of each bubble is percent vegetative cover.

Waterfowl herbivory. *Herbivory also was attributable to grazing by waterfowl*. Shortly following initiation of effluent discharge, waterfowl recruited to the area *en masse*. Surveys indicate a 3-fold increase in waterfowl abundance at the Hammond assimilation wetland compared to other areas of the Manchac/Maurepas system (David Brown, Eastern Kentucky University unpublished data). During winter months, approximately 2,000 waterfowl use this area daily (Curtis Hymel, unpublished thesis data). In an attempt to tease apart the herbivory impacts due to nutria and waterfowl, we constructed 20-m x 40-m nutria exclosures during the spring and summer of 2010. While waterfowl cannot land in the 4 by 4 meter exclosures, the larger cages enable waterfowl to forage, yet prevent entry by nutria (Shaffer et al. 1992). Within 4 months these large cages filled with herbaceous vegetation. However, by mid-December 2010, waterfowl had consumed nearly all vegetation in the cages except for cutgrass (*Zizaniopsis miliacea*). Although this study remains in its early stages, we now know that waterfowl, especially during the winter months, are eating as much or more herbaceous vegetation as are nutria. Our 16 m^2 exclosures, which prevent herbivory of both nutria and waterfowl, remain 100% vegetated unless nutria were able to enter through breaks in the wire mesh.

Reduced belowground biomass. *The discharge of treated effluent resulted in low belowground biomass, especially near the discharge system.* Recent reports in the literature have suggested that high nutrient loadings lead to reduced belowground biomass (Turner et al. 2009, Turner 2010). In order to investigate this hypothesis we measured belowground biomass at the Hammond assimilation wetland. Belowground live biomass of bulltongue (*Sagittaria lancifolia*) was measured along the 700 m transects in Sub-units 1 and 4, from the zone of discharge south towards the Joyce forest. We cored through a single species to test the effect of nutrients on belowground biomass at varying distances from the outfall source; *S. lancifolia* is the only vegetative species present from 0 m to 700 m from discharge. Live belowground biomass averaged 2,330 g dry wt/m^2 and ranged between 286 and 5,676 g dry wt/m^2 with no trend of increasing or decreasing biomass along the transect (Figure 16).

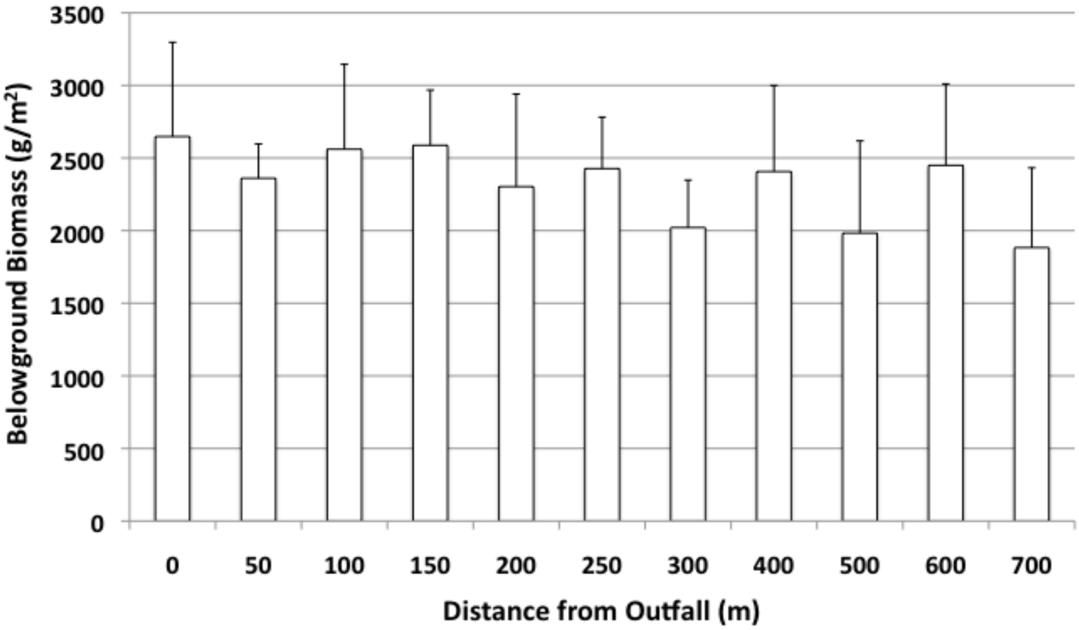


Figure 16. Belowground biomass (g/m^2) of soil cores taken through bulltongue at the Hammond assimilation wetland. No pattern was found from the outfall to 700m away.

Overall, belowground biomass was statistically greater at the eastern end of the outfall system (Sub-unit 4), where the effluent has been discharged throughout much of the study period, compared to the western end (Sub-unit 1) which has received the lowest input (Figure 17).

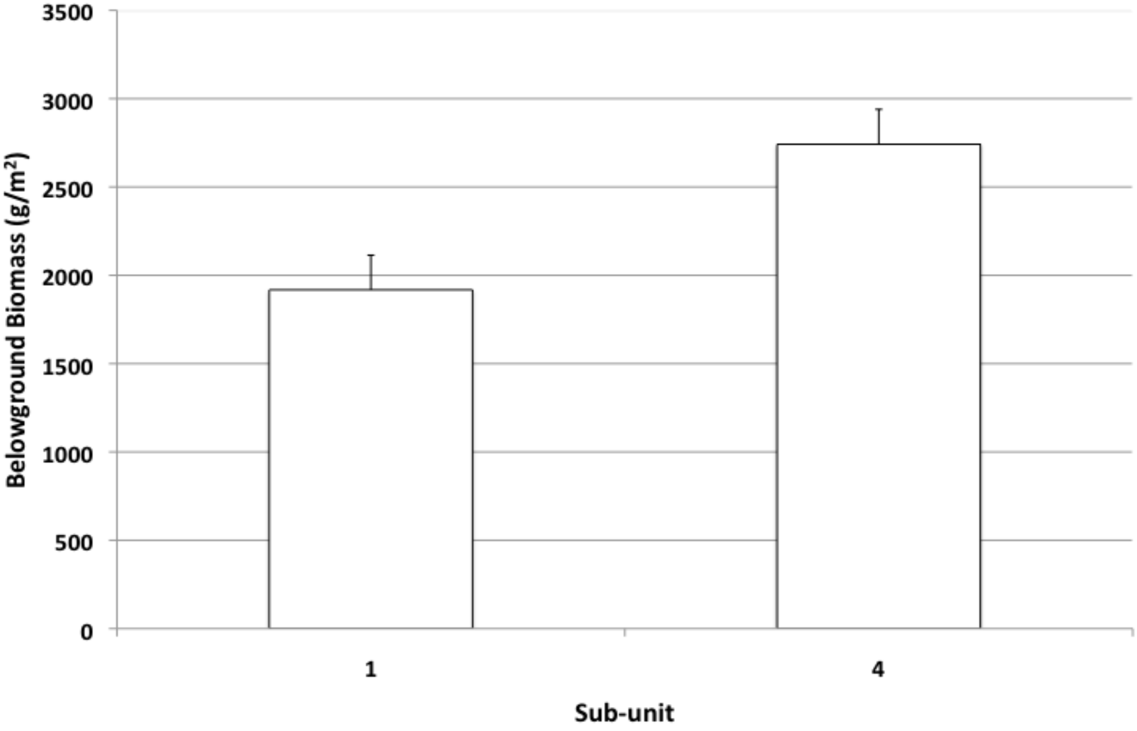


Figure 17. Belowground biomass from cores taken through individual bulltongue at Sub-unit 1 (lowest nutrient loading) and Sub-unit 4 (highest nutrient loading) of the Hammond assimilation wetland.

It is unlikely that excessive nutrient loading led directly to the marsh deterioration. As discussed above, nutrient loading to the marsh area was low to moderate. There is no example that we know of where similar nutrient loading led to marsh deterioration within a year of the initiation of discharge. Coastal marshes have received much higher loading for long periods of time and not converted to open water. For example, nitrogen and phosphorus has been applied to the Great Sippewissett salt marsh in Massachusetts since the early 1970s. Total nitrogen dosage rates ranged from 18 to greater than 157 gN/m^2/yr (Valiela et al. 1975, 1976, Giblin et al. 1980, 1986, Turner et al. 2009, Fox et al. 2011). Turner et al. (2009) reported that there was a reduction of belowground biomass and marsh elevation gain, but the marsh remains intact after more than 35 years of nutrient additions. Fox et al. (2011) suggested that the accretion reduction might be overstated because of solubility of ash. Likewise, shorter term nutrient additions to Louisiana salt marshes that resulted in reduced belowground biomass and soil strength were significantly higher than loading rates at the Hammond site (Darby and Turner 2008a,b,c, Turner 2010). In addition, vegetative cover is highest closest to the outfall area and growth of mature baldcypress near the discharge area is over 3-fold higher than that of the Maurepas swamp and 5-fold higher than that of the nearby Joyce forest.

There are a number of coastal marsh systems in Louisiana where long-term nutrient loading has not caused marsh deterioration. The Atchafalaya and Wax Lake deltas have developed with very high river input and the region affected by Atchafalaya River input has low wetland loss rates compared to most of the coastal zone (Britsch and Dunbar 1993, Barras et al. 2008). Day et al. (2011a) reported that salt marshes near lower Fourleague Bay have been stable for over a half century while salt marshes east of Bayou Terrebonne have high rates of deterioration. Day et al. (2011b) reported similar findings for the northwestern Mediterranean where coastal marshes with riverine input had much higher rates of accretion and elevation gain and were likely to survive accelerated sea-level rise. In contrast, Swarzenski et al. (2008) reported that floating *P. hemitomon* marshes in the Bayou Penchant region affected by inflow of Atchafalaya were more reduced, the organic matter substrate was more decomposed and accumulated more sulfur than *P. hemitomon* floating marshes in the Barataria Basin that do not receive river water. They concluded that the continual input of river water leads to a reducing soil environment, increased sulfide and inorganic nutrients in porewater, and internally generated alkalinity, and that these conditions lead to organic matter decomposition and root decomposition. Swarzenski et al. did not consider is the impact of grazing by nutria. Sasser et al. (2004) conducted a study in the same Bayou Penchant area of the impact of excluding nutria on floating marsh growth. They found that excluding nutria resulted in a dramatic increase in both above ground and belowground growth. The results of Swarzenski et al. and Sasser et al. suggest that there is a complex interaction among fresh water input, nutrients, and nutria grazing. This topic needs to be studied in more detail before definitive conclusions can be drawn.

Izdepski et al. (2009) reported that discharge of treated effluent into a shallow open water area near Thibodaux, LA led to the formation of a freshwater floating marsh. In the Central Wetlands Unit (CWU) in St. Bernard Parish, the only area where cypress survived saltwater intrusion from the Mississippi River Gulf Outlet (MRGO) was in two locations where surface runoff was pumped into the area (Shaffer et al. 2009). At the Gore pumping station, surface runoff combined with treated effluent was pumped into wetlands of the CWU since the early 1960s, a relic cypress forest has survived and soil strength is high. Wetlands impacted by the Caernarvon diversion suffered considerable wetland loss during Hurricanes Katrina and Rita. Howes et al. (2010) reported that soils of low salinity wetlands in the upper Breton Sound estuary had significantly lower shear strength compared to higher salinity wetlands. They concluded that this was due to biomorphological differences between high and low salinity plant species, which may be exacerbated by low mineral sediment and high freshwater and nutrient inputs. Day et al. (2009, 2011c) measured above and below ground biomass and decomposition at streamside marshes in near, intermediate, and far locations from the Caernarvon diversion and at a reference location. Belowgound biomass was high and there were little to no difference in decomposition among sites.

Disease. *Plant diseases induced by high nutrients caused vegetation death at the Hammond assimilation wetland.* Photographs have been presented showing indications of various plant diseases (e.g., *Fusarium* fungi) at the assimilation wetland, but no quantitative data have been presented to support these observations. There was no quantitative information presented on whether these were a common occurrence in the marsh or whether they have been reported to affect marsh growth. There were no data presented on any studies on disease impacts in marshes. Until such information is presented, this should be considered as an untested hypothesis. It is known that fertilized agricultural crops are more susceptible to diseases but very little is known about wetland vegetation. It might be reasonable to think that disease could affect individual species. However, it is extremely unlikely that a pathogen or pathogens could impact an entire marsh of diverse plant species because each species would vary in its susceptibility to the pathogen (Dr. Reymond Schneider and Dr. Laurence Datnoff, LSU Plant Pathology Department, personal communication). This is a topic area that should be investigated in more detail.

Loss of soil strength with increasing nutrient loads. *Soil strength declines in marsh soils with increasing nutrient loads.* Turner (2010) hypothesized that increasing nutrient loads to salt marshes would result in reduced soil strength by increasing soil metabolism and lowering root and rhizome biomass. However, loading rates in the experiments of Turner (2009, 2010) were much higher than loading rates seen in Louisiana assimilation wetlands (<15 g N/m^2 and 4 g P/m^2; Figure 18). The units in Figure 16 are 1000 kg N or P ha^{-1}. 1000 kg ha^{-1} is equivalent to 100 g m^{-2} yr^{-1}. Thus, significant decrease in soil strength did not occur until the dose of N was several hundred to greater than 1000 g N m^{-2} yr^{-1}. Thus, the results may not be applicable to these wetlands. In addition, the method used to estimate soil strength (e.g., use of a hand vane tester) may not appropriate for use in organic soils.

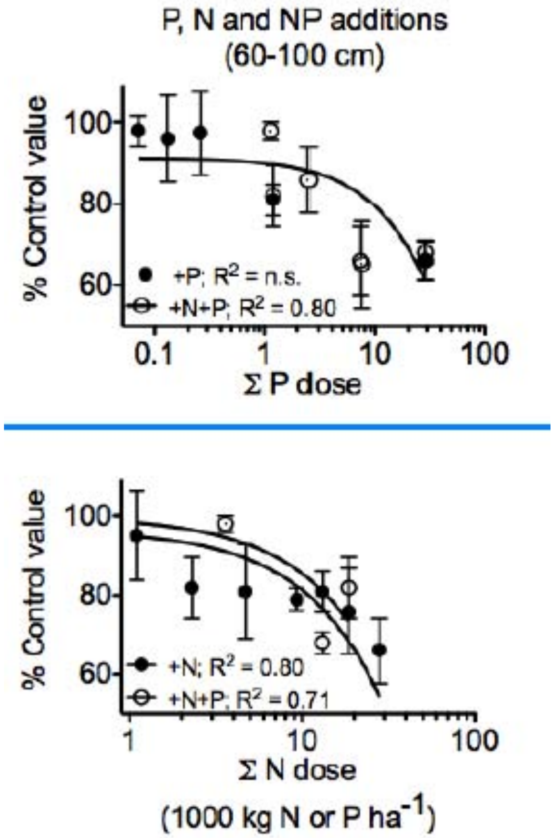


Figure 18. Estimates of soil strength (as a percent of soil strength in a control marsh) for several salt marshes in coastal Louisiana (from Turner 2010). Note that on the horizontal axis, 1 is equivalent to 100 g N/m^2 and 10 is equivalent to 1000 g N/m^2.

Conclusions

The information presented in this paper shows that nutria herbivory was the primary cause of marsh deterioration in the Hammond Assimilation Wetlands. This is supported by observations of high nutria densities and herbivory as the wetland deteriorated, exclosure studies showing that when nutria were excluded, the marsh flourished, local extinction of *Typha domingensis* - a species that generally dominates under eutrophied conditions, and that the marsh recovered – especially within 200 m of the discharge area - following aggressive nutria control. However, the marsh deterioration due to nutria led to a cascade of additional impacts. For example, when the marsh opened up, large numbers of waterfowl were attracted to the area. Grazing of herbaceous vegetation by waterfowl is as important as nutria, especially during winter. As restoration proceeds, an important question is: Is it desirable to maintain some level of waterfowl or try to restore the whole area to marsh?

It is unlikely that excessive nutrient loading led directly to the marsh deterioration. As discussed above, nutrient loading to the marsh area was low to moderate. There is no example that we know of where similar nutrient loading led to marsh deterioration within a year of the initiation of discharge. Coastal marshes have received much higher loading for long periods of time and not converted to open water. For example, nitrogen and phosphorus has been applied to the Great Sippewissett salt marsh in Massachusetts since the early 1970s. Total nitrogen dosage rates ranged from 18 to greater than 157 g N/m^2/yr (Valiela et al. 1975, 1976, Giblin et al. 1980, Turner et al. 2009). There was a reduction of belowground biomass and marsh elevation gain, but the marsh remains intact after more than 35 years of nutrient additions. Likewise, shorter term nutrient additions to Louisiana salt marshes that resulted in reduced belowground biomass and soil strength were significantly higher than loading rates at the Hammond site (Turner and Darby 2008a,b,c, Turner 2010). In addition, vegetative cover is highest closest to the outfall area and growth of mature baldcypress near the discharge area is over 3-fold higher than that of the Maurepas swamp and 5-fold higher than that of the nearby Joyce forest.

There are a number of coastal marsh systems in Louisiana where long-term nutrient loading has not caused marsh deterioration, with the Atchafalaya region being the most notable. The land loss rate in the wetlands affected by the Atchafalaya River is the lowest in coastal Louisiana (Day et al. 2000; Barras 2008). Day et al. (2011a) reported that salt marshes southeast of the Atchafalaya River have been stable for over a half-century, while salt marshes further east not in contact with river water have high rates of deterioration. Day et al. (2011b) reported similar findings for the northwestern Mediterranean where coastal marshes with riverine input had much higher rates of accretion and elevation gain and were likely to survive accelerated sea-level rise.

After aggressive nutria control began at the Hammond Assimilation Wetland, the marsh began to recover, especially near the discharge. There was high below ground biomass and with significant soil strength by the end of the 2010 growing season. There has been a shift from perennial to annual species so that the marsh dies back in winter but some perennial species are present. Planting of nutria resistant species such as *S. californicus* (giant bullwhip) and a hybrid of *Typha* in conjunction with nutria control holds the possibility of restoring a marsh dominated by perennial species. Construction of a second outfall system in a baldcypress – water tupelo forest either to the east or south of the current outfall system to enable pulsing and periods of drawdown would greatly enhance recovery and sustainability of these wetlands. The closure of the drainage pipe next to the railroad at the west end of the discharge pipe caused elevated water levels that likely inhibited revegetation. As of June 2011, water control structures with flap gates have been installed at each end of the discharge pipe. This will allow water levels at one of the pipes to be lowered when discharge is at the other end of the pipe. Alternatively, discharge will be moved to the center of the system and both culverts will be opened to maximize drawdown potential; under this scenario water will still flow over > 500 m of wetland, ensuring adequate bioremediation.

Suggested Further Research

A number of factors could have contributed to wetland deterioration at the Hammond assimilation site. These include nutrient loading, nutria and waterfowl herbivory, increased pH, and increased water levels. Additional studies are needed to determine the interactive impacts of these factors, in concert with nutria herbivory. We suggest that studies be carried out to look at the impacts of these factors, both singly and in combination. Further vegetation surveys and measurements of above- and belowground biomass production need to be made, along with quantification of decomposition rates. One question that should be addressed is the impact of a half-century of hydrological isolation on the response of the Hammond assimilation wetland to municipal effluent inputs. The study should also include the wetland area north of South Slough that has received regular surface runoff from the surrounding watershed and from the Ponchatoula treatment plant, yet has not deteriorated. Studies should be conducted at the same time as restoration so that the effectiveness of restoration can be quantified and accelerated.

Suggested Restoration Approaches. By the end of the growing season in fall 2010, over 50% of the wetland area that turned to open water during the winter of 2007-8 had re-vegetated, with the strongest reestablishment occurring closest to the discharge pipe (Figure 19). We believe that the vegetative recovery would have been more rapid and more complete had the effluent been periodically pulsed to a nearby baldcypress- water tupelo swamp. That is, it may be that the nutria impact was exacerbated by excessive flooding. We believe that this wetland benefits greatly from the nutrients and freshwater supplied by the outfall system, but a pulsing paradigm would be more optimal for maximum ecosystem function, along with continued management of nutria. The least expensive project has been implemented and involves opening culverts to South Slough at the western and eastern ends if the discharge system. A more effective pulsing plan would utilize the spoil bank of South Slough to pipe wastewater approximately 0.7 km eastward and release it in a mature baldcypress-water tupelo swamp. Alternatively, a more expensive plumbing to the interior wetlands of the Joyce WMA would be enormously beneficial for that highly degraded ecosystem.

As mention above, work has begun on different restoration approaches at the Hammond Assimilation Wetland. One suggestion involves construction of a wetland at the Hammond wastewater treatment plant to remove a portion of the nutrients, because one hypothesis is that excessive nutrients are a cause of deterioration. But this should not be done until it has been demonstrated that nutrients are causing any problems. At the wetland site, additional exclosures and controls, planted with several species of herbaceous vegetation at varying distances from the effluent discharge point, has been implemented to better elucidate the effects of nutrient loading in the presence and absence of nutria and/or waterfowl. Marsh pillows should be deployed along varying distances as well to determine if marsh growth can be differentially enhanced (Charles Sasser, LSU, personal communication).

Gulfsaver bags also have been deployed to determine their impact on growth. Finally, nutria-resistant vegetative species such as *Peltandra virginica, S. californicus* and a hybrid *Typha* (that we have isolated) should be widely transplanted to the assimilation wetland. Much of this work has already begun.

Summary and Conclusions

The information presented in this paper shows that nutria herbivory was the primary cause of marsh deterioration in the Hammond Assimilation Wetlands. This is supported by observations of high nutria densities and herbivory as the wetland deteriorated, exclosure studies showing that when nutria were excluded, the marsh flourished, local extinction of *T. domingensis* - a species that generally dominates under eutrophic conditions, and that the marsh recovered – especially within 200 m of the discharge area - following aggressive nutria control. However, the marsh deterioration due to nutria led to a cascade of additional impacts. For example, when the marsh opened up, large numbers of waterfowl were attracted to the area. Grazing of herbaceous vegetation by waterfowl is as important as nutria, especially during winter. As restoration proceeds, an important question is: Is it desirable to maintain some level of waterfowl or try to restore the whole area to marsh?

The Hammond Assimilation Wetlands are a part of the Joyce wetlands that extend from North Pass, located north of Pass Manchac, to the Pleistocene uplands. The natural hydrology of these wetlands was characterized by runoff from uplands directly to the north and by flow from several creeks, which drained an 86 km^2 watershed that extends north of Hammond. This freshwater input was generally a one-way southerly flow into the wetlands. Water flux into the area from the south was generally tidally driven and bidirectional. With the exception of Four Mile Marsh and Seven Mile Marsh, the entire Joyce tract was baldcypress-water tupelo (*Taxodium distichum - Nyssa aquatica*) forest well into the 20th century.

There have been a number of substantial changes over the past century that significantly altered the hydrology of the Joyce wetlands. A railroad built on embankment eliminated most flow with wetlands to the west. Exchange is limited to a number of small openings under the railroad through which water flows only when there is heavy rainfall or large tidal water-level variations in Lake Pontchartrain. The most significant hydrological change for the Joyce wetlands was the dredging of South Slough in the early 1950s, which eliminated almost all flow from the watershed to the north. Runoff now flows under the railroad and south via the canal dredged for I-55. The I-55 canal allows higher saline waters to enter the northwestern part of the Joyce wetlands when lake levels increase. During high rainfall events, the I-55 canal allows rapid outflow of water from South Slough.

Saltwater intrusion has killed large areas of baldcypress-water tupelo forest in the entire Manchac/Maurepas system. Water tupelo has been eliminated from the lower two-thirds of the Joyce wetlands due to salinity stress. Surface water salinity in Four Mile Marsh was as high as 3.5 ppt during summer, 2006. By the time treated effluent discharge began in November 2006, the wetlands south of South Slough had been isolated from virtually all freshwater inflow from the watershed for over a half century. The impacts of this isolation need to be studied in more detail to fully understand the dynamics of the Joyce wetlands.

Based on data from manipulative experiments as well as observations of nutria activity, it is clear that nutria were a dominant cause of marsh deterioration in the zone near the discharge pipe. Before nutria impacts began there was significant increase in growth of vegetation. Diameter stem growth of planted seedlings was highest near the outfall and decreased linearly with distance from discharge pipe. Inorganic nutrient concentrations decreased with distance from outfall to the Joyce forest in a manner similar to baldcypress seedling production. Mature baldcypress growing near the outfall area of the Hammond Assimilation Wetland are averaging greater than 5-fold increases in diameter growth compared to much lower growth rates in the lower Joyce area and in the Maurepas swamp. After effluent discharge was initiated in November 2006, there was robust vegetative growth at the area receiving discharge, with greatly increased net primary production during the 2007growing season, due to the increased nutrient and freshwater input.

Following vegetation impacts by nutria, they were hunted aggressively and about 2000 nutria were killed during the winter of 2007-2008 and nutria continue to be hunted. Nutria exclosures and controls were constructed and planted with southern cattail (*T. domingensis*). Plants in exclosures had 100% survival and spread rapidly while those outside the exclosures were destroyed. Belowground biomass was nearly 3-fold higher inside of exclosures than in controls. Marsh 'pillows' were constructed and planted with sprigs of maidencane (*Panicum hemitomon*). Both nutria and waterfowl foraged on the vegetation growing through the wire of the pillows, but survivorship has been nearly complete and re-growth has occurred from roots inside the pillows. Construction of larger exclosures that excluded nutria but not waterfowl showed that after nutria opened up the marsh, waterfowl grazing became important, especially during the winter. At the end of the growing season in 2010, the marsh within 200 m of the outfall could support a person. By the end of the 2010 growing season, about half of the marsh had recovered with both annual and perennial plants (Figure 19).

A number of other hypotheses have been offered to explain the loss of marsh at the site, including increased pH, reduced belowground biomass and increased decomposition due to high nutrients, and disease. To test the decreased belowground biomass hypothesis we cored live roots of bulltongue (*Sagittaria lancifolia*) from 0 m – 700 m along two transects. Belowground biomass of bulltongue was high and did not vary with distance from the outfall. In addition, there is no reported instance of a marsh deteriorating in one year due to nutrient input.

It is unlikely that excessive nutrient loading led directly to the marsh deterioration. As discussed above, nutrient loading to the marsh area was low to moderate. There is no example that we know of where similar nutrient loading led to marsh deterioration within a year of the initiation of discharge. Coastal marshes have received much higher loading for long periods of time and not converted to open water.

After aggressive nutria control began at the Hammond Assimilation Wetland, the marsh began to recover, especially near the discharge. There was high below ground biomass and with significant soil strength at the end of the 2010 growing season. There has been a shift form perennial to annual species so that the marsh dies back in winter but some perennial species are present. Planting of nutria resistant species such as *S. californicus* (giant bullwhip) and a hybrid of *Typha* in conjunction with nutria control holds the possibility of restoring a marsh dominated by perennial species. Installation of gated culverts at the western and eastern ends of the discharge system has already enabled a lowering of water level by about 40 cm. A more substantial pulsing paradigm could be established through construction of a second outfall system in a baldcypress – water tupelo forest either to the east or south of the current outfall.

Figure 19. Vegetation at the Hammond Assimilation Wetland in Spring 2007 (left) and in September 2010 (right).

Literature Cited

Allen, J.A., J.M. Chambers, and D. McKinney. 1994. Intraspecific variation in the response of *Taxodium distichum* seedlings to salinity. Forest Ecology and Management. 70:203-214

Allen, J.A., S.R. Pezeshki, and J.L. Chambers. 1996. Interaction of flooding and salinity stress on Baldcypress (*Taxodium distichum*) Tree Physiology 16:307-313.

Allen, J.A., J.L. Chambers, and S.R. Pezeshki. 1997. Effects of salinity on Baldcypress seedlings: physiological responses and their relation to salinity tolerance. Wetlands 17(2):310-320.

Barras JA, Bernier JC, Morton RA (2008) Land area changes in coastal Louisiana—A multi decadal perspective (from 1956 to 2006). US Geological Survey Scientific Investigations Map 3019, 14 scale 1#250;000.

Beville, S.L. 2002. The efficacy of a small-scale freshwater diversion for restoration of a swamp in southeastern Louisiana. M.S. thesis, Southeastern Louisiana University. Hammond, 86p.

Booth, D. B., and C. R. Jackson. 1997. Urbanization of aquatic systems - degradation thresholds, storm water detention, and the limits of mitigation. Journal of the American Water Resources Association 22:1-19.

Boshart, W.M. 1997. The conservation and restoration of a Baldcypress swamp: an investigation of macronutrients, competition, and induced vegetation dynamics as related to nutria herbivory. M.S. thesis, Southeastern Louisiana University. Hammond, 67p.

Brantley, C.G., J.W. Day, Jr., R.R. Lane, E. Hyfield, J.N. Day, and J.Y. Ko. 2008. Primary production, nutrient dynamics, and accretion of a coastal freshwater forested wetland assimilation system in Louisiana. Ecological Engineering 34:7-22.

Britsch, L. D. and J. B. Dunbar. 1993. Land loss rates: Louisiana coastal plain. Journal of Coastal Research 9:324-338.

Burns, A. C. 1980. Frank B. Williams, Cypress Lumber King. Journal of Forest History 24:127-133.

Campo, F.M. 1996. Restoring a repressed swamp: the relative effects of a saltwater influx on an immature stand of Baldcypress (*Taxodium distichum* (L.) Richard). M.S. thesis, Southeastern Louisiana University. Hammond, 89p.

Conner, W.H. 1994. The effect of salinity and waterlogging on growth and survival of Baldcypress and Chinese tallow seedlings. Journal of Coastal Research 10 (4):1045-1049.

Conner, W.H., K.W. McLeod, and J.K. McCarron. 1997. Flooding and salinity effects on growth and survival of four common forested wetland species. Wetlands Ecology and Management 5:99-109.

Conner, W.H., I. Mihalia, and J. Wolfe. 2002. Tree community structure and changes from 1987 to 1999 in three Louisiana and three South Carolina forested wetlands. Wetlands. 22:58-70.

Craft, C.B. 2001. Biology of wetland soils. Pages 107-136 In Wetland Soils: Genesis, hydrology, landscapes, and classification. CRC Press, Boca Raton, Florida.

Darby F.A. and R.E. Turner RE. 2008a. Below- and aboveground *Spartina alterniflora* production in a Louisiana salt marsh. Estuaries & Coast 31:223–231.

Darby, F.A., and R.E. Turner. 2008b. Below- and aboveground biomass of *Spartina alterniflora*: Response to nutrient addition in a Louisiana salt marsh. Estuaries & Coasts 31:326-334.

Darby F.A. and R.E. Turner RE. 2008c. Effects of eutrophication on salt marsh root and rhizome accumulation. Marine Ecology Progress Series 363:63-70.

Day, J., J. Rybczyk, R. Pratt, M. Sutula, A. Westphal, T. Blahnik, P. Delgado, P. Kemp, A. J. Englande, C-Y. Hu, G. Jin, and H-W. Jeng. 1997. A Use Attainability analysis for longterm wastewater discharge to the Poydras-Verret wetland in St. Bernard Parish, LA. Report to the Lake Pontchartrain Basin Foundation. LSU Baton Rouge, and Tulane University New Orleans. 60 p. plus appendices.

Day, J.W. Jae-Young Ko, J. Rybczyk, D. Sabins, R. Bean, G. Berthelot, C. Brantley, L. Cardoch, W. Conner, J.N. Day, A.J. Englande, S. Feagley, E. Hyfield, R. Lane, J. Lindsey, J. Mistich, E. Reyes, and R. Twilley. 2004. The use of wetlands in the Mississippi Delta for wastewater assimilation: A review. Journal of Ocean and Coastal Management 47:671-691.

Day, John, Robert Lane, Matthew Moerschbaecher, Ronald DeLaune, Robert Twilley, Irving Mendelssohn, and Joseph Baustian. 2009. The impact of the Caernarvon diversion on above- and belowground marsh biomass and decomposition in the Breton Sound Estuary after Hurricane Katrina. Final Report to the Louisiana Coastal Area (LCA) Science and Technology Program Office. Interagency Agreement 2512-07-01. 87 p.

Day, J.W., G.P. Kemp, D.J. Reed, D.R. Cahoon, R.M. Boumans, J.M. Suhayda and R. Gambrell. 2011a. Vegetation death and rapid loss of surface elevation in two contrasting Mississippi delta salt marshes: The role of sedimentation, autocompaction and sea-level rise Ecological Engineering. DOI:10.1016/j.ecoleng.2010.11.021.

Day, J.W., C. Ibanez, F. Scarton, D. Pont, P. Hensel, J.N. Day, and R. Lane. 2011b. Sustainability of Mediterranean deltaic and lagoon wetlands with sea-level rise: the importance of river input. Estuaries and Coasts. 34:483-493.

Day, John, Robert Lane, Matthew Moerschbaecher, Ronald DeLaune, Robert Twilley, Irving Mendelssohn, and Joseph Baustian. 2011c. Vegetation and soil dynamics of a Louisiana estuary receiving pulsed Mississippi River water following Hurricane Katrina.

Day, J.W., Jr. and G.P. Shaffer. 2009. Ecology of the LaBranche Wetlands: the Role of Human Impacts. Expert Report. LADOTD, Baton Rouge, LA. 58 pp.

Forder, D. R. 1995. Timber harvesting, coppicing and artificial regeneration of a cypress muck swamp. M.S. Thesis, Southeastern Louisiana University, Hammond, LA.

Fox, Liza, Ivan Valiela, Erin L. Kinney. 2011. 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 and Coasts. (Revised manuscript).

Giblin, A., A. Bourg, I. Valiela, AND J. M. Teal. 1980. Uptake and losses of heavy metals in sewage sludge by a New England salt marsh. Am. J. Bot. 67: 1059–1068.

Giblin, A. E., G. W. Luther III, and I. Valiela. 1986. Trace metal solubility in salt marsh sediments containing sewage sludge. Estuarine, Coastal and Shelf Science 23: 477-498.

Howes, N., Duncan M. FitzGerald, Zoe J. Hughesa, Ioannis Y. Georgioub, Mark A. Kulpb, Michael D. Minerb, Jane M. Smithc, and John A. Barras. 2010. Hurricane-induced failure of low salinity wetlands. Proceeding of the National Academy of Sciences. 107: 14014-14019.

Hunter, R.G., J.W. Day, Jr., R.R. Lane, J. Lindsey, J.N. Day, and M.G. Hunter. 2009a. Nutrient removal and loading rate analysis of Louisiana forested wetlands assimilating treated municipal effluent. Environmental Management 44:865-873.

Hunter, R.G., J.W. Day, Jr., R.R. Lane, J. Lindsey, J.N. Day, and M.G. Hunter. 2009b. Impacts of secondarily treated municipal effluent on a freshwater forested wetland after 60 years of discharge. Wetlands 29:363-371.

Izdepski, C., J. Day, C. Sasser, and B. Fry. 2009. Early floating marsh establishment and growth dynamics in a nutrient amended wetland in the lower Mississippi delta. Wetlands. 29:1004-1013.

Keddy, P.A., D. Campbell, T. McFalls, G.P. Shaffer, R. Moreau, C. Dranguet, and R. Heleniak. 2007. The Wetlands of lakes Pontchartrain and Maurepas: origins, processes, and restoration. Environmental Reviews. 15:43-77.

Lundberg, C.J. 2008. Growth rates of Baldcypress (*Taxodium distichum*) seedlings in a treated effluent assimilation marsh. Master's Thesis. Southeastern Louisiana University. 73 pp.

Mancil, E. 1972. A historical geography of industrial cypress lumbering in Louisiana. Ph.D. Dissertation Louisiana State University, Baton Rouge, LA.

Mancil, E. 1980. Pullboat logging. Journal of Forest History 24:135-141.

McLeod, K.W., J.K. McCarron, and W.H. Conner. 1996. Effects of inundation and salinity on photosynthesis and water relations of four southeastern coastal plain forest species. Wetlands Ecology and Management 4(1):31-42.

Mitsch, W.J. and J.G. Gosselink. 2007. Wetlands, 5th ed. Van Nostrand Reinhold, New York, NY.

Myers, R.S. 1993. Wetlands restoration in the Manchac Wildlife Management Area: the relative effects of herbivory, flooding, competition and macronutrients. M.S. thesis, Southeastern Louisiana University. Hammond, 85p.

Odum, W.E. 1988. Predicting ecosystem development following creation and restoration of wetlands. *Proceedings of a Conference: Increasing Our Wetland Resources: Mayflower Hotel, Washington, DC, October 4–7, 1987* (eds J. Zelazny & J.S. Feierabend), pp.67-70. National Wildlife Federation, Washington, DC.

Richardson, C.J., and D.S. Nichols. 1985. Ecological analysis of wastewater management criteria in wetland ecosystems. In: Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. Van Nostrand Reinhold Company, NY, pp. 351–391.

Shaffer, G. P., C. E. Sasser, J. G. Gosselink, and M. Rejmanek. 1992. Vegetation dynamics in the emerging Atchafalaya Delta, Louisiana, USA. J. Ecology 80:677-687.

Shaffer, G.P., J.W. Day, S. Mack, G.P. Kemp, I. van Heerden, M.A. Poirrier, K.A. Westpahl, D. FitzGerlad, A. Milanes, C. Morris, R. Bea, and P.S. Penland. 2009a. The MRGO navigation project: a massive human-induced environmental, economic, and storm disaster. Journal of Coastal Research. 54:206-224.

Shaffer, G.P., Wood, W.B, Hoeppner, S.S, Perkins, T.E, Zoller, J.A, and D. Kandalepas. 2009b. Degradation of Baldcypress – Water Tupelo Swamp to Marsh and Open Water in Southeastern Louisiana, USA: An Irreversible Trajectory? Journal of Coastal Research, Special Issue. 54:152-165.

Turner, R.E., B.L. Howes, J.M. Teal, C.S. Milan, E.M. Swenson, and D. Goehringer-Toner. 2009. Salt marshes and eutrophication: An unsustainable outcome. Limnology and Oceanography 54:1634-1642.

Turner, R.E. 2010. Beneath the salt marsh canopy: Loss of soil strength with increasing nutrient loads. Estuaries and Coasts DOI 10.1007/s12237-010-9341-y. Published online September 2010.

Valiela, I., J. M. Teal, and W. J. Sass. 1975. Production and dynamics of experimentally enriched salt marsh vegetation: Below ground biomass. Limnology and Oceanography 21:245-252.

Valiela, I., J. M. Teal and N. Y. Persson. 1976. Production and dynamics of experimentally enriched salt marsh vegetation: Belowground biomass. Limnology and Oceanography 21:245-252.

Walsh, C. J., T. D. Fletcher and A. R. Ladson. 2005. Stream restoration in urban catchments through redesigning stormwater systems: looking to the catchment to save the stream. North American Benthological Society 24: 690–705.

Wang, L. Z., J. Lyons, P. Kanehl, and R. Bannerman. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. Environmental Management 28: 255-266.