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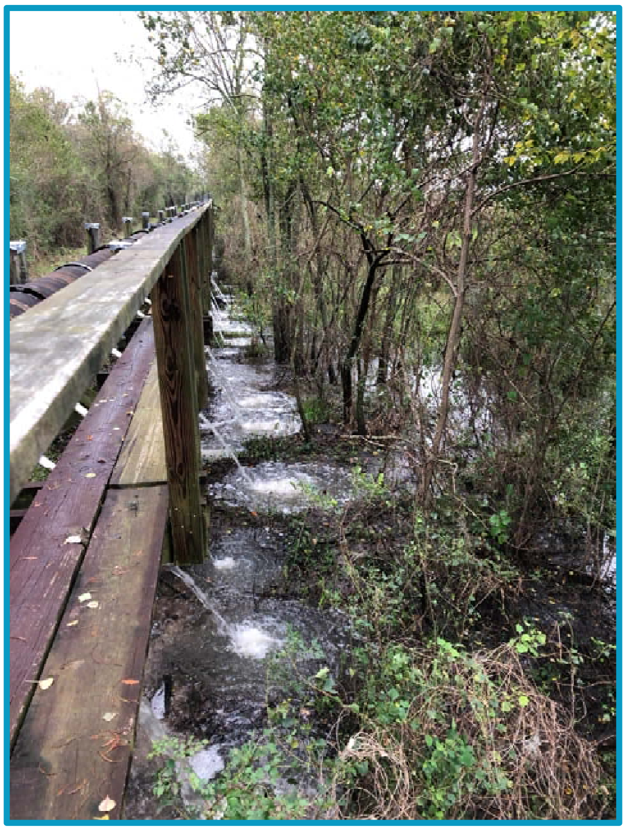
Wetland Evaluation Report

South Slough Wetland Study
Hammond, Louisiana

Prepared for: Louisiana Department of Environmental Quality



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REPORT TO: Louisiana Department of Environmental Quality

PREPARED BY: Naturally Wallace Consulting, LLC

DATE: May 22, 2019

NOTE:

This report represents an independent evaluation of the City of Hammond South Slough Wetland assimilation system. The conclusions and recommendations represented herein reflect the professional opinion of the author, and do not necessarily represent the views or opinions of the Louisiana Department of Environmental Quality or its employees and officials.

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Executive Summary

The study was an independent evaluation of the City of Hammond, Louisiana wetland wastewater assimilation project, defined as the “South Slough Wetland” by LDEQ and permitted by the Department under Louisiana Pollution Discharge Elimination System (LPDES) Permit LA0032328. The South Slough Wetland includes a section of freshwater marsh (locally known as Four Mile Marsh) immediately downstream of the effluent distribution pipeline. Beyond that, effluent can spread to the south and east over extensive cypress-tupelo swamps as water moves towards Lake Pontchartrain. Most of these swamps are degrading due to increased salinity and rising sea levels. The project represents an attempt to partially reverse these negative trends while providing the City a cost-effective means of effluent discharge.

Discharge of secondarily-treated municipal wastewater effluent began November 2006. After about one year of operation, the fresh water marsh converted to open water and mudflats during 2008-2009. This area has largely revegetated with a mixed plant community (including annuals) that is different than the original marsh community, in which *Panicum hemitomon* (maidencane) was a keystone species. The reason for this marsh conversion is an ongoing dispute in the scientific literature, with different proponents advocating that either nutrients or nutria were the dominant cause of the vegetation change.

Despite the marsh conversion, the system has consistently and successfully met expectations for nutrient assimilation, salinity reduction, and enhanced plant productivity. Over the period of record (2006 - 2017), the system has produced the water quality benefits expected from wetland assimilation projects. Comparing data averages from the NEAR and OUT monitoring locations:

- Total Kjeldahl Nitrogen (TKN) was reduced from 10.0 to 0.9 mg/L.
- Ammonia Nitrogen (NH_4-N) was reduced from 6.2 to 0.2 mg/L.
- Nitrate Nitrogen (NO_3-N) was reduced from 1.4 to 0.1 mg/L.
- Total Phosphorus (TP) was reduced from 3.2 to 0.2 mg/L.
- Salinity was 1.66 PPT at the OUT location but was only 0.29 PPT at the NEAR location.

Concentrations of TKN, NO_3-N, NH_4-N and TP were at ecosystem background levels at the OUT location, indicating these nutrients had been completely assimilated by the wetland. The addition of treated wastewater effluent, a low-salinity water supply, was effective in lowering salinity levels at the NEAR and MID locations, keeping salinity well below levels that cause stress to cypress and tupelo trees.

The “fertilizer effect” of available nutrients resulted in increased plant productivity. Measurements of plant biomass production over the growing seasons when nutrients were available (2007 – 2017) all indicated enhanced plant growth:

- For the marsh vegetation, End of Season Live Biomass (EOSL) was 2.2X greater at the NEAR location compared to the Marsh Control.
- For the forest vegetation, Litterfall was 2.8X greater at the MID location compared to the Forest Control.
- Similarly, Stem Growth was 2.8X greater at the MID location compared to the Forest Control.
- For cypress trees studied at the MID location, the average growth rate was 1.87X faster after effluent application began compared to the 20 years prior to the project.

This study used a mathematical modeling approach to estimate the "active assimilation zone", which is the area actively involved in nitrogen (N) and phosphorus (P) assimilation. This has not been done for previous assimilation wetlands in Louisiana, which have historically used a "loading chart" approach. These modeling calculations indicate that the "active assimilation zone" is far smaller than the overall South Slough Wetland (5-16% of the total project area). Most of this nutrient assimilation zone is in the freshwater marsh that underwent large vegetative changes after effluent discharge began.

Modeling calculations were completed to determine possible causes of the marsh conversion. Grazing by nutria was found to be the most likely cause, but to occur in the two-year period described, there are three likely underlying factors:

- The pre-existing marsh was a relatively fragile, "relict" plant community that developed prior to the dredging of the South Slough Canal and the construction of I-55. Grazed vegetation died and did not grow back.
- Nutria are "wasteful feeders" and destroy about 10X more vegetation than they actually consume.
- Nutria were attracted to the area by the fertilized vegetation that grew once treated wastewater effluent was introduced.

The marsh changes are likely irreversible as the area has converted to floating mats of vegetation. However, this phenomenon is widespread in coastal Louisiana marshes, and has occurred in many areas not involving wastewater assimilation. Effluent application is protecting the area from salinity intrusions and is enhancing plant growth and biomass productivity.

Recommendations

For the City of Hammond:

1. Continuance of the City of Hammond effluent discharge is strongly recommended for the following reasons:
    a. The system is clearly successful in meeting the objectives of nutrient assimilation, salinity reduction, and productivity enhancement.
    b. Effluent assimilation is clearly enhancing the growth of cypress trees at the MID location.
    c. Discontinuing the discharge is highly unlikely to return the marsh to the pre-project state, due to the structural changes (development of open water and floating mats) which have occurred. Enhanced biomass production through effluent assimilation is the best means to promote "in-filling" of the floating mats back to a fixed marsh community.
2. Changes in the marsh community should continue to be monitored in a more comprehensive manner, including plant survey, the status of open water and floating mats, and changes over time. To date, most of the studies done on the marsh area has been outside the existing LDPES permit. Adding one or more permanent monitoring locations should be considered.
3. Nutria control should be ongoing and documented more thoroughly on an annual basis.

4. Since the system uses multiple application zones on the distribution pipeline, the date(s) and volume(s) of water discharged into each application zone should be documented on an annual basis.
5. Hydraulic control structures that allow discharge north into South Slough should be replaced with water-tight structures that allow positive operator control over this discharge path.

For Future Wetland Assimilation Projects:

1. The assumption that a "do nothing" option (no effluent addition) represents "no change" (maintenance of current wetland ecosystems) is questionable at best in coastal Louisiana. There is a considerable body of evidence that indicate that both freshwater marshes and forested swamps in coastal Louisiana will continue to decline and disappear without human interventions to re-introduce sources of fresh water, nutrients and sediments. Evaluation of future assimilation projects should therefore consider both outcomes; what changes to the marsh/swamp would happen with effluent addition, and what likely changes will occur without effluent addition.
2. Many wetlands in coastal Louisiana are in a relict state, where the current plant communities developed prior to human-induced changes. Evaluation of future assimilation projects should consider that stability of the pre-project vegetative community, and what likely changes will occur both with the absence of effluent application and with effluent application.
3. Advances in wetland science now make it possible to calculate the estimated size of the "active assimilation zone". Historically, wetland assimilation projects in Louisiana have relied on a "loading chart" approach which assumes that the entire project area is involved in nutrient assimilation. Because the "loading chart" approach cannot estimate the size of the active assimilation zone, it provides no insight into optimal placement of monitoring locations. It is recommended that future projects utilize modern methods to locate monitoring locations within the context of the active assimilation zone.
4. The location of compliance monitoring points should be established relative to the anticipated extent of the active assimilation zone. Having a single MID monitoring location between the NEAR and OUT locations only provides a single data point on what is happening inside the active assimilation zone. This is not adequate to accurately monitor the system. Also, the active assimilation zone will take multiple years to fully develop. The zone will be about 20% developed after one year, 50% developed after four years, and 90% developed after 10 years. Having at least three monitoring points located in the area where the active assimilation zone will develop would provide much more information about the active assimilation zone and the rate of formation.

1.0 Introduction

The State of Louisiana supports the use of natural wetland systems to assimilate treated wastewater and enhance wetland ecosystems. The City of Hammond operates one such wetland assimilation system, the South Slough Wetland, which is regulated by the Louisiana Department of Environmental Quality (LDEQ) under permit LA0032328.

Treated effluent represents a steady supply of fresh water, helping to protect freshwater wetlands from the effects of salinity and seawater intrusion. Nutrients present in treated effluent can promote enhanced growth of wetland vegetation (Day *et al.* 2004), resulting in greater accretion of wetland soils, which also protects against subsidence and associated seawater intrusion (Hunter *et al*. 2018). These effects are especially beneficial in the vulnerable forested swamp wetlands in the Lake Pontchartrain basin (Shaffer *et al*. 2016).

The immediate discharge area is owned by the City of Hammond and flows are distributed along a piping system approximately 1,200 meters long located on the northern edge of the wetland assimilation area. Water flows primarily to the south, (Figures 1.1 and 1.2) entering the Joyce Wildlife Management Area (JWMA), which contains over 4,500 hectares (10,000 acres) of wetland habitat east of Interstate 55; flows eventually entering the northern reaches of Lake Pontchartrain (Lane *et al*. 2016).

The South Slough Wetland site was selected based on ecological studies done to assess baseline conditions and the anticipated effects of wastewater assimilation, as summarized in the City of Hammond Use Attainability Analysis (UAA) report (UAA 2005). Wetland habitat was primarily described as forested cypress-tupelo swamp in the JWMA with areas of emergent marsh habitat in the area proposed for wastewater discharge. Other assimilation projects in Louisiana are almost exclusively forested swamps (Day *et al*. 2018a) and the presence of a marsh wetland at Hammond was not addressed in detail while the project was in the planning stages (UAA 2005).

However, the immediate area of wastewater assimilation was an emergent marsh in which *Panicum hemitomon* (maidencane) was a keystone species. This area is locally known as Four Mile Marsh (Lundberg *et al*. 2011, Bodker *et al*. 2015, Shaffer *et al*. 2015). Four Mile Marsh appears to have never been a forested swamp (USDA 1905) for reasons that likely date back to the geologic formation of the Lake Pontchartrain basin (Saucier 1963, Flocks *et al*. 2009) further discussed in Section 3.1.

Within about one year after effluent addition began, there was a dramatic shift in the marsh plant community. The *Panicum* community largely died and converted to open water and “mudflats” (Bodker *et al*. 2015, Shaffer *et al*. 2015). These “mudflats” are floating mats of relict *Panicum* biomass (Turner *et al*. 2018), which have been subsequently been colonized by a diverse group of annual and perennial emergent marsh plants (Weller & Bossart 2017).

The shift in the marsh plant community has been controversial engendering local media coverage and calls for regulatory policy changes. This has been fueled by debate in the scientific community, which is largely divided into two camps: 1) nutrient addition caused destabilization of the marsh soils (Bodker *et al*. 2015, Turner *et al*. 2018) and 2) intense grazing by nutria resulted in changes to the plant community (Shaffer *et al*. 2015, Day *et al*. 2019).

Figure 1.1 – City of Hammond Assimilation Wetland (from Shaffer *et al.* 2015). The project area estimated in the Use Attainability Analysis (UAA, 2005) is approximately 4,500 hectares. The active assimilation zone (estimated by modeling in this study) was approximately 204 ha in 2006, increasing to 660 ha in 2018.

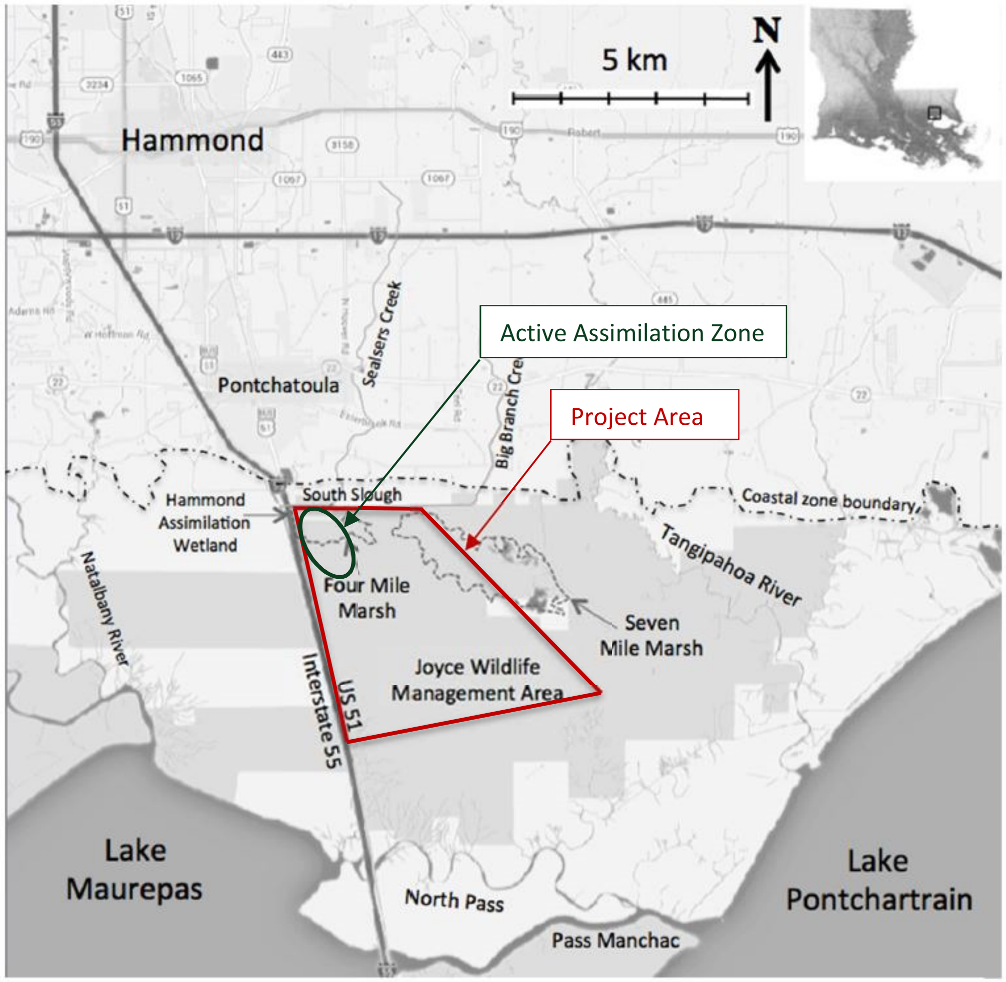
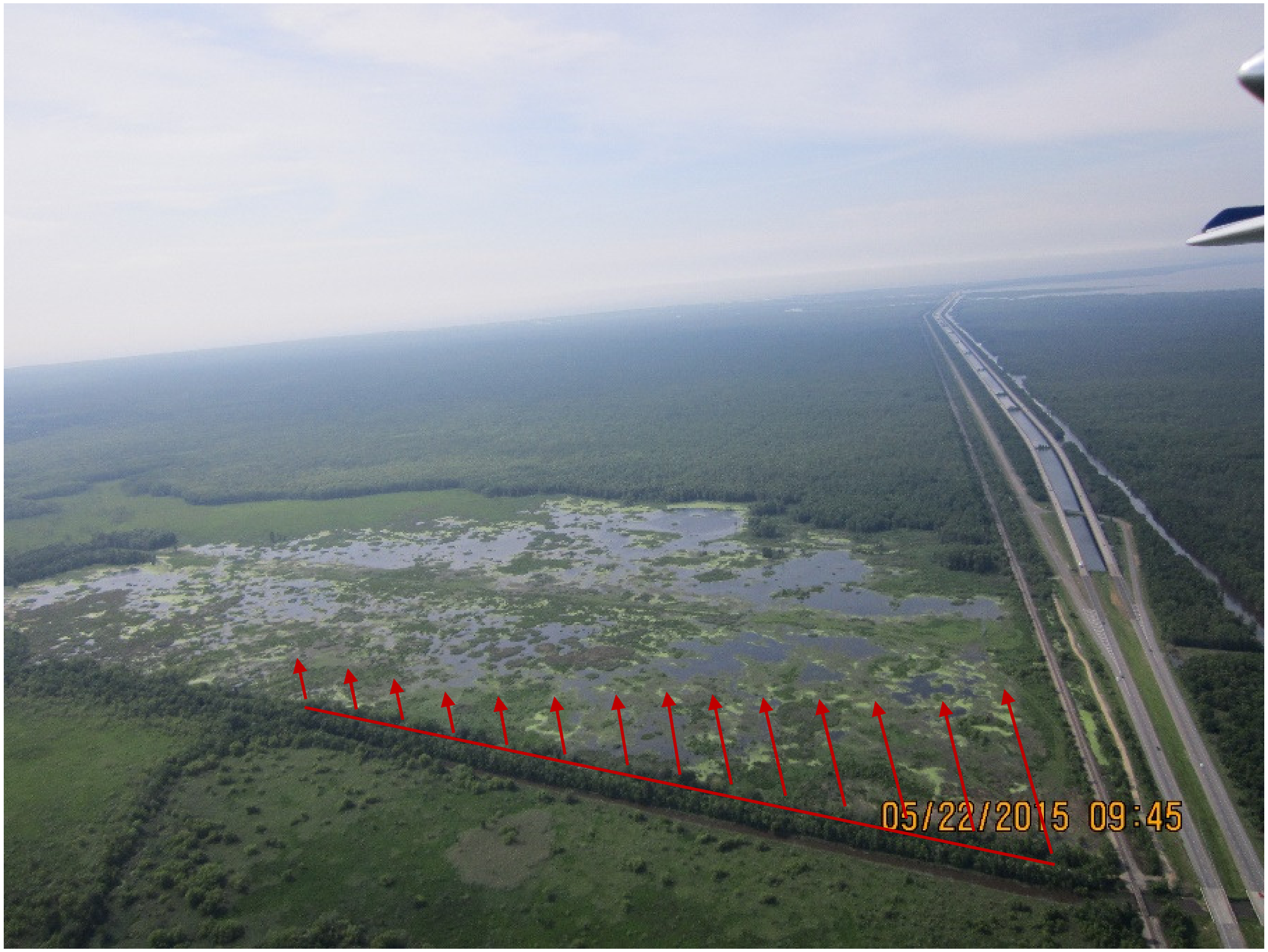


Figure 1.2 – Aerial View of the South Slough Wetland (including the wetland assimilation area), looking southeast (photo taken by LDEQ May 22, 2015). Red arrows indicate the primary direction of effluent movement as indicated by Lane *et al*. (2016).



The debate over the merits of wastewater assimilation wetlands is further complicated by the fact that wetlands (both forested swamps and emergent marshes) are declining throughout the Lake Pontchartrain basin as a result of drainage, channelization, relative sea level rise and increasing salinity (Saucier, 1963, Visser *et al*. 1999, Shaffer *et al*. 2016) As a result the "do nothing" alternative is not a return to a static status quo, and represents acceptance that widespread loss of freshwater wetlands in the Lake Pontchartrain basin is inevitable.

The goal of this evaluation was to conduct an independent review of current and historical data associated with the Hammond Assimilation Wetland project, conduct a site visit of the wetland assimilation area (Appendix A), make conclusions based on this information regarding environmental and ecological conditions at the South Slough Wetland, and present these findings in this report to LDEQ.

2.0 Regulatory Background

The City of Hammond, Louisiana discharges treated wastewater effluent into the South Slough Wetland (thence into the Joyce Wildlife Management Area) under LPDES permit LA0032328. This is a wastewater wetland assimilation discharge as authorized under Louisiana Administrative Code (LAC) LAC 33:IX.1109.J. LAC 33:IX.1109.J defines wetland types, the water quality use designation for wetlands used for wastewater assimilation, and references procedures for implementation of wetland assimilation projects.

2.1 Types of Wetlands

LAC 33.IX.1109J.2 recognizes two types of wetlands, forested and non-forested (marsh) as further defined under LAC 33.IX.1105.

Forested Wetlands: LAC 33.IX.1105 defines forested wetlands as including "bottomland hardwood swamps" and "cypress-tupelo swamps", which would apply to the forested regions of the Joyce WMA receiving wastewater effluent from the City of Hammond.

Freshwater Marshes: LAC 33.IX.1105 defines these as wetlands that support typical vegetation including cattail (*Typha angustfolia*), bulltongue (*Sagittaria* spp.), maidencane (*Panicum hemitomon*), water hyacinth (*Eichornia crassipes*), pickerelweed (*Pontideria cordata*), alligatorweed (*Altemanthera philozeroides*), and *Hydrocotyl spp*. (This listing of "typical" vegetation includes both emergent and floating wetland plant species). Freshwater marshes are further defined as having a salinity normally less than 2 parts per thousand (ppt). The definition also recognizes two subtypes of freshwater emergent wetlands: floating and attached:

- Floating wetlands are defined as areas where the wetland surface substrate is detached and is floating above the underlying deltaic plain.
- Attached wetlands are defined as areas where the vegetation is attached to the wetland surface and is contiguous with the underlying wetland substrate and can be either submerged or emergent.

The Hammond wastewater assimilation project has both types of wetlands; Four Mile Marsh (the immediate assimilation area) is a freshwater marsh under the definition of LAC 33.IX.1105, and the surrounding swamp areas in the Joyce WMA are forested wetlands under the definition of LAC 33.IX.1105.

2.2 Water Quality Use Designation; Level of Treatment Required

LAC 33.IX.1109.J.3 states that wetlands approved for wastewater assimilation projects are assigned designated uses of:

- Secondary contact recreation, which is defined under LAC 33.IX.1111 as water contact activity where prolonged or regular full-body contact is either incidental or accidental, and the probably of ingesting appreciable amounts of water is minimal.
- Fish and wildlife propagation, which is defined under LAC 33.IX.1111 as the use of water for aquatic habitat, food, resting, reproduction, cover and/or travel corridors. This use also includes maintenance of water quality to a level that prevents damage to indigenous wildlife and/or aquatic life species associated with the aquatic environment and the contamination of aquatic life biota consumed by humans.

The level of treatment required under the LPDES permit is determined by the water quality use designation of the receiving water body, consistent with the Clean Water Act (USC 33 1251). LAC 33.IX.1109.J.4 lists additional treatment criteria, including

- A numerical dissolved oxygen criterion is not necessary to protect the beneficial use of fish and wildlife propagation.
- General criteria under LAC 33.IX.1113.B apply for aesthetics, color, taste and odor, toxic substances, oil and grease, foam, nutrients, flow and radioactive materials.
- Criteria for assessment of the biological integrity of wetlands (LAC 33.IX.1113.B.12.b) apply. This is defined that the discharge area shall have no more than a 20 percent reduction in the rate of total above-ground wetland productivity over a five-year period as compared to a reference area. Methods of measuring above-ground productivity are found in the current *Water Quality Management Plan, Volume 3, Section 10, Permitting Guidance Document for Implementing Louisiana Surface Water Quality Standards* ("Section 10").

2.3 Rationale for Permitting Wetland Assimilation Projects

LDEQ recognizes that many of Louisiana's wetlands are deteriorating due to changes in hydrology and the resultant lack of nutrients, suspended solids, and a high natural subsidence rate ("Section 10"). Therefore, the department may allow the discharge of the equivalent of secondarily treated effluent into wetlands for the purposes of nourishing and enhancing those wetlands ("Section 10"). The underlying ecological model is that addition of secondarily-treated nutrient-rich municipal wastewater to south Louisiana wetlands will promote vertical accretion through increased organic matter production and deposition, counteracting the effects of hydrological isolation and subsidence. ("Section 10").

The stated rationale of wastewater assimilation projects is increased accretion in wetlands to offset relative sea level rise. Increased accretion in wetlands is accomplished by increased biomass production as a result of nutrients present in the treated wastewater effluent. This closely follows the benefits outlined in Day *et al*. 2004, and the wetland assimilation wetland web page maintained by LDEQ (https://deq.louisiana.gov/page/wetland-assimilation). These summaries of stated benefits are based on the policy outline proposed in Breaux & Day, 1994.

Adding nutrient rich treated wastewater effluent to selected coastal wetlands is claimed to result in the following benefits (Day *et al*. 2004):

- Improved water quality
- Increased accretion rates to help offset subsidence
- Increased productivity of vegetation
- Financial and energy savings of capital not invested in conventional tertiary treatment systems

LDEQ's web site on wetland assimilation projects (https://deq.louisiana.gov/page/wetland-assimilation) contains a similar list of stated benefits:

Stated benefits of wetland assimilation projects for the environment include:

- Removes direct discharges of treated wastewater into state waterbodies.
- Can help prevent saltwater intrusion into the wetland.
- Add an abundance of needed nutrients into the wetland to stimulate plant growth.
- Carbon sequestration.

Stated benefits of wetland assimilation projects for a permittee include:

- Less operation and maintenance costs
- A "green" approach to wastewater treatment

The stated benefits of financial and energy savings are based on economic studies by Ko *et al*. (2004, 2012), which support the premise that wetland assimilation is a more cost-effective alternative for small communities than other treatment/discharge alternatives.

Breaux & Day, 1994 presented a set of "tentative standards" for the Thibodaux wastewater treatment site, which were considered (at that time) to be protective of assimilation wetlands from any adverse effects due to wastewater application:

1. No more than 20% decrease in naturally occurring litterfall or stem growth.
2. No significant decrease in the dominance index or stem density of bald cypress.
3. No significant decrease in faunal species diversity and no more than a 20% decrease in biomass.

In the rationale presented in Breaux & Day, 1994, accretion is seen as the primary overriding benefit, whereas preservation of plant communities is a secondary concern, due to an assessment that the status quo represents an ongoing loss of Louisiana's coastal wetlands. Breaux & Day, 1994 state:

> "Wastewater application to wetlands does not usually lead to biological communities identical to those either preceding application or surrounding the receiving site (*emphasis added*). For Louisiana, the object is both to treat and to maintain wetlands. In a state with a relative sea level rise four times the average of any other state… the first problem to be addressed should be to keep the land above water. Only after succeeding in that attempt will we have the option of determining exactly what type of vegetation is optimal."

2.3.1 Carry-Over into City of Hammond Permit

The concept of no more than a 20% decrease in productivity (1) and (3) from Breaux & Day 1994 was later incorporated into the language of LAC 33:IX.1113.B.12. All three criteria are included almost verbatim in the City of Hammond Permit LA0032328, Part II, Section D.2.

It is important to note that the only species specifically mentioned in LAC 33:IX1113.B.12 in regards to the dominance index is bald cypress (*Taxodium distichum*). Since bald cypress one of the dominant tree species in forested swamps in Louisiana, a decrease in the dominance index or stem density would reflect a major shift in the vegetative community within the forested swamp. It is important to note that no such "protection" exists for freshwater marsh plant species, despite the fact that both forested swamp and freshwater marshes fall under the scope of wastewater assimilation discharges in LAC 33:IX.1109.J. Thus, it is entirely possible to comply with the permit requirements (no significant decrease in diversity, no more than a 20% decrease in litterfall, stem growth and biomass) for a freshwater marsh even though the plant community within the marsh undergoes significant changes as a result of wastewater assimilation.

2.4 Permitting of Wetland Assimilation Projects ("Section 10")

The Louisiana Administrative Code (LAC 33.IX.1113.B.12.b) reference a stand-alone guidance document for permitting wetland assimilation projects. This is the *Water Quality Management Plan, Volume 3, Section 10, Permitting Guidance Document for Implementing Louisiana Surface Water Quality Standards* ("Section 10"). "Section 10" defines the information required for a permit application (feasibility assessment and baseline study) and the requirements for ongoing monitoring under the permit once issued.

Permit Application Process

Approval of wetland assimilation permits requires that the applicant complete the following:

A Feasibility Assessment that includes:

- Delineation of the available wetlands
- A list of landowners and the availability of ownership and/or easement agreements
- A description and the suitability of the type (classification) of the wetlands available
- The number of acres of wetlands required for assimilation
- Uses that currently exist within the wetland
- Long-term average loading rates (and basis for calculations) to the wetland (not to exceed 15 g TN/square meter/year and 4 g TP/square meter/year
- A proposed reference area for evaluation purposes
- A hydrology and hydrograph of the proposed assimilation area and possible distribution system layout

A Baseline Study that includes:

- Classification of the flora present
- Vegetative productivity
- Sediment analysis for metals and nutrients
- Water level measurements/analysis, including salinity, dissolved oxygen, conductivity, nitrogen series and total phosphorus
- Water quality measurements
- Accretion measurements

Ongoing Permit Monitoring

Upon permit issuance, the permittee is required to conduct ongoing biological measurements to ensure the biological integrity of the wetland. The quantity and frequency of the measurements will be dependent upon the flow of the discharge and the loading rate to the wetland, but may include, but is not limited to, sampling in the discharge site for variations in:

- Floral species diversity
- Above-ground productivity
- Water stages
- Metals and nutrient analysis from plant tissue samples
- Metals and nutrient analysis from sediment samples
- Water quality analysis of metals, nutrients and other components
- Accretion measurements

As found in LAC 33:IX.1113.B.12, the following biological criteria shall apply to a wetland receiving a discharge:

- Due to effluent addition, the discharge area shall have no more than a 20% reduction in the rate of total above-ground wetland productivity over a 5-year period as compared to a reference area

2.5 City of Hammond Permit

The discharge into the South Slough Wetland (and thence to the Joyce Wildlife Management Area) by the City of Hammond is regulated under LPDES Permit No. LA0032328, last issued July 7, 2010. This permit follows the requirements of LAC 33:IX.1109.J and the *Water Quality Management Plan, Volume 3, Section 10, Permitting Guidance Document for Implementing Louisiana Surface Water Quality Standards* ("Section 10").

The permit requires the City to monitor the wastewater discharge at Outfall 001 (defined as the point of discharge from the last treatment unit prior to distribution to the wetlands and mixing with other waters), and to monitor the assimilation wetland at three different locations within the wetland (NEAR, MID and OUT), as well as two reference monitoring locations (designed at the Forest Control and Marsh Control). These monitoring locations were identified prior to the project in the *Hammond Wetland Wastewater Assimilation Use Attainability Analysis* (UAA, 2005).

The Forest Control location was moved in 2012 to a more representative location (Hunter *et al*. 2018).

Monitoring requirements for Outfall 001 are summarized in Table 2.1.

Table 2.1 – City of Hammond Outfall 001 Monitoring Requirements

Parameter	Discharge Limitations		Measurement Frequency	Comments
	Weekly Average	Monthly Average		
Flow			Continuous	Stated design flow is 8 MGD
BOD_5	30 mg/L	45 mg/L	5/week	
TSS	90 mg/L	135 mg/L	5/week	Not subject to 85% removal requirement
pH	6 - 9		5/week	
Total Residual Chlorine (TRC)	No Measurable (NM)		5/week	< 0.1 mg/L = NM
Fecal Coliforms	200 CFU/100 mL	400 CFU/100 mL	5/week	
Total Nitrogen (TN)	Report		Quarterly	< 15 g/m^2-yr in assimilation area
Total Phosphorus (TP)	Report		Quarterly	< 4 g/m^2-yr in assimilation area
Metals: Mg, Pb, Cd, Cr, Fe, Ni, Ag, Se	Report		1/6 months	
Total Cu	2.02 lb/day	0.85 lb/day	1/month	< 10 ug/L = 0
Total Hg	0.004 lb/d	0.002 lb/d	1/month	< 0.2 ug/L = 0
Total Zn	15.71 lb/d	6.62 lb/d	1/month	<20 ug/L = 0

Monitoring of the wetland assimilation area is also required. Monitoring requirements are summarized in Table 2.2.

Table 2.2 – City of Hammond Wetland Assimilation Monitoring Requirements

Parameter	Wetland Component [(1)]		
	Flora	Sediment	Surface Water
Water Stage			Monthly
Nutrient Analysis I: TKN, TP	4th Year of the Permitting Cycle	4th Year of the Permitting Cycle	Quarterly
Nutrient Analysis II: NH_3N, NO_2N, NO_3N, PO_4		Every 4th Year	Quarterly
Growth Studies	Annually		
Accretion Rate		4th Year of the Permitting Cycle	
Species Classification	4th Year of the Permitting Cycle		
Percentage of Whole Cover (for each species)	4th Year of the Permitting Cycle		
Metals: Mg, Pb, Cd, Cr, Cu, Zn, Fe, Ni, Ag, Se	4th Year of the Permitting Cycle	4th Year of the Permitting Cycle	4th Year of the Permitting Cycle
Others: BOD, TSS, pH, dissolved oxygen			4th Year of the Permitting Cycle

Note:

1. Sampling required at the three wastewater assimilation area sites (NEAR, MID, OUT) and the two control sites (Forested and Marsh).

3.0 Regional History

Understanding the situational context of the South Slough Wetland requires an understanding of the geologic, climate and human influences on the site that have contributed the current situation.

3.1 Geologic History

Interpreting what is occurring at the Hammond Assimilation Wetland site requires a background understanding of how and why the landscape formed, as this has impacts even to the present day.

3.1.1 Pleistocene Landform Development

The modern (Holocene) landscape of the Mississippi River basin is the product of climate change, sea level rise, glacial melting and erosion that has occurred since the period of the maximum cold during the last Ice Age (Pleistocene) about 22,000 to 18,000 years ago (Flocks *et al*. 2009). During this time period, sea level was about 120 meters (390 feet) below current levels.

Sections of the Gulf of Mexico that are now underwater were an outwash plain of the glacial Mississippi River. The glacial Mississippi was fueled by melting glaciers and carried a much higher volume of flow and sediment load than the modern (Holocene) river. With the lower sea level during the Pleistocene, drainage systems had more erosion potential and incised into the upland sediments. This resulted in entrenched drainage valleys (Figure 3.1). In the Hammond area, this is seen most clearly in the upland valleys associated with the Tangipahoa River and other drainageways such as Big Branch Slough and the Ponchatoula River (USDA, 1905) that are cut into the upland (Prairie terrace) landscape (Flocks *et al*. 2009).

There is a network of geologic faults in southern Louisiana (Fisk 1944, Saucier 1963) (Figure 3.2). The Baton Rouge – Denham Springs (BR-DS) fault system is an active fault zone extending northwest to southeast along the north shore of Lake Pontchartrain (Flocks *et al*. 2009). These slip faults caused an abrupt lowering of the land level, forming the northern edge of the Lake Pontchartrain basin. This fault system is observed in the abrupt topographic changed observed at the transition between the upland Prairie terrace deposits and the northern edge of the Pontchartrain basin (USGS 2015, USDA 1905). Locally, this occurs south of the City of Ponchatoula, with a slip differential between 5 – 15 feet (Saucier 1963). This drop in land level is readily observable today as one drives south from Ponchatoula towards the Hammond Assimilation Wetland site.

Figure 3.1 – Soil Map of Project Area (from USDA, 1905)

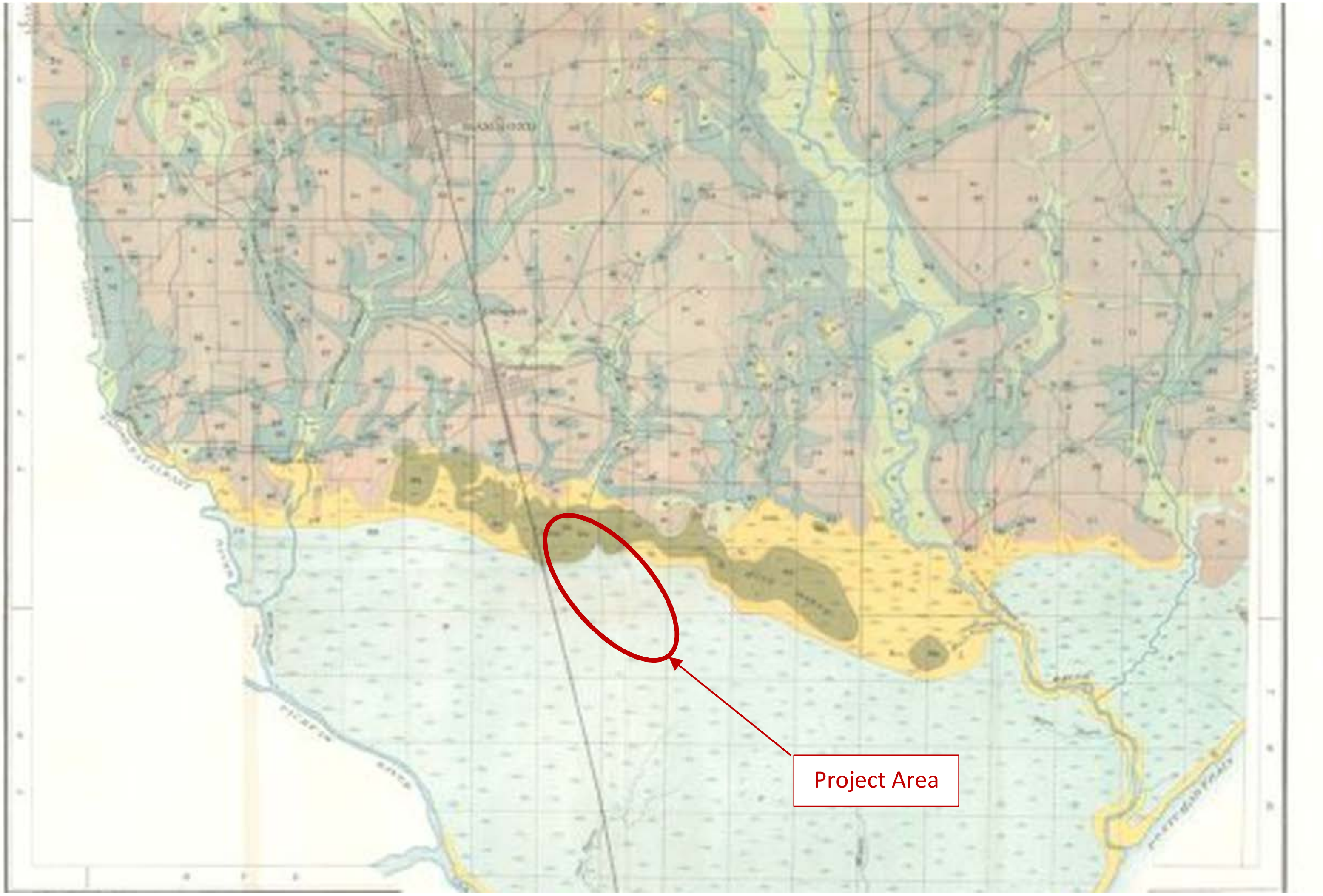
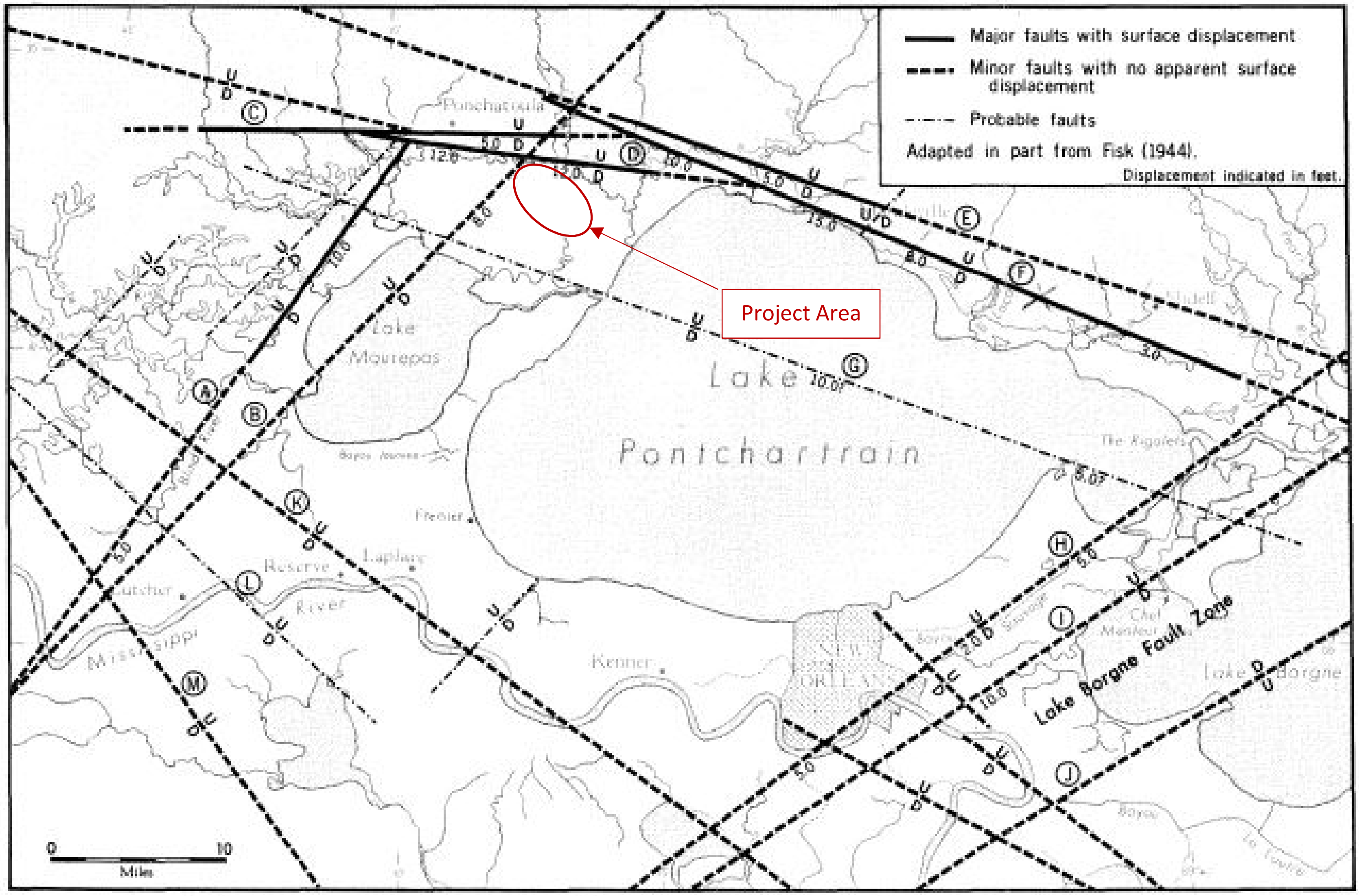


Figure 3.2 – Geologic faults in the Lake Pontchartrain basin (from Saucier, 1963)



3.1.2 Enclosure and Formation of Lake Pontchartrain

As the Ice Age was winding down, glacial melting from approximately 18,000 to 4,000 years ago raised the sea level to about the current (1990's) elevation (Saucier, 1994). Rising sea levels flooded the Pontchartrain basin, forming an embayment that was bounded on the north side by the BR-DS fault system. South of the fault system, the incipient Lakes Maurepas, Pontchartrain and Borgne developed in response to subsidence along multiple fault lines where the Prairie terrace now dropped below rising sea levels (Lopez, 1991). This early Pontchartrain embayment was a shallow open saltwater bay that supported tidal-flat species (Darnell, 1962). Within these strata, oyster shells are present (Saucier, 1963) beneath brackish fauna, indicating that the salinity gradually decreased over time. The gradual shift to lower salinities was brought about by the formation of barrier island systems along the northern Gulf Coast, stretching from Mobile Bay to south of Lake Pontchartrain. These barrier islands gradually increased the flux of fresh water through the area, and decreased salt water intrusion.

By about 4,000 years ago, the Pontchartrain embayment was progressively becoming a fresh water body because of the closure of the southern outlet by barrier island systems migrating westward and the Mississippi River Delta lobes migrating eastward (Frazier, 1967). By approximately 2,900 years ago, the modern forms of Lake Pontchartrain and Lake Maurepas were in place (Otvos, 1978) (Figure 3.3).

The modern Lake Pontchartrain Basin is one of the largest freshwater wetlands along the Gulf of Mexico (approximately 150,000 ha) (Keddy *et al*. 2007). The natural vegetation of the region remains fresh or brackish marshes, mixed with swamps dominated by bald cypress (*Taxodium distichum*) and tupelo (*Nyssa aquatica*). Yearly flooding by the Mississippi River was once a major factor controlling vegetation patterns, but has been greatly impaired by the construction of artificial levees for flood control.

3.1.3 Milton's Island Beach Trend and Formation of Four Mile Marsh

The mainland shoreline of the final-stage Pontchartrain embayment has been partially traced from the vicinity of Slidell westwards towards Ponchatoula. The Milton's Island Beach Trend (Saucier, 1963, Figure 3.4) consists of a sand ridge which now lies buried under marsh and swamp deposits or is located beneath Lake Pontchartrain. Deeper water areas on both the north (landward) and southern (toward Lake Pontchartrain) were gradually in-filled with sediment and organic deposits that support the marsh and swamp habitats that delineate the modern-day shoreline of Lake Pontchartrain. Saucier (1963) describes this as:

> Throughout a total distance of about 35 miles, the south or Gulf side of the ridge is relative steep and fairly uniform in plan. When the ridge was an active coastal feature, the irregular northern or sound side probably consisted of a series of ridges, swales and flats and sandy shoals in shallow water. (Page 51).

Figure 3.3 – Modern Lake Pontchartrain basin (from Saucier, 1963)

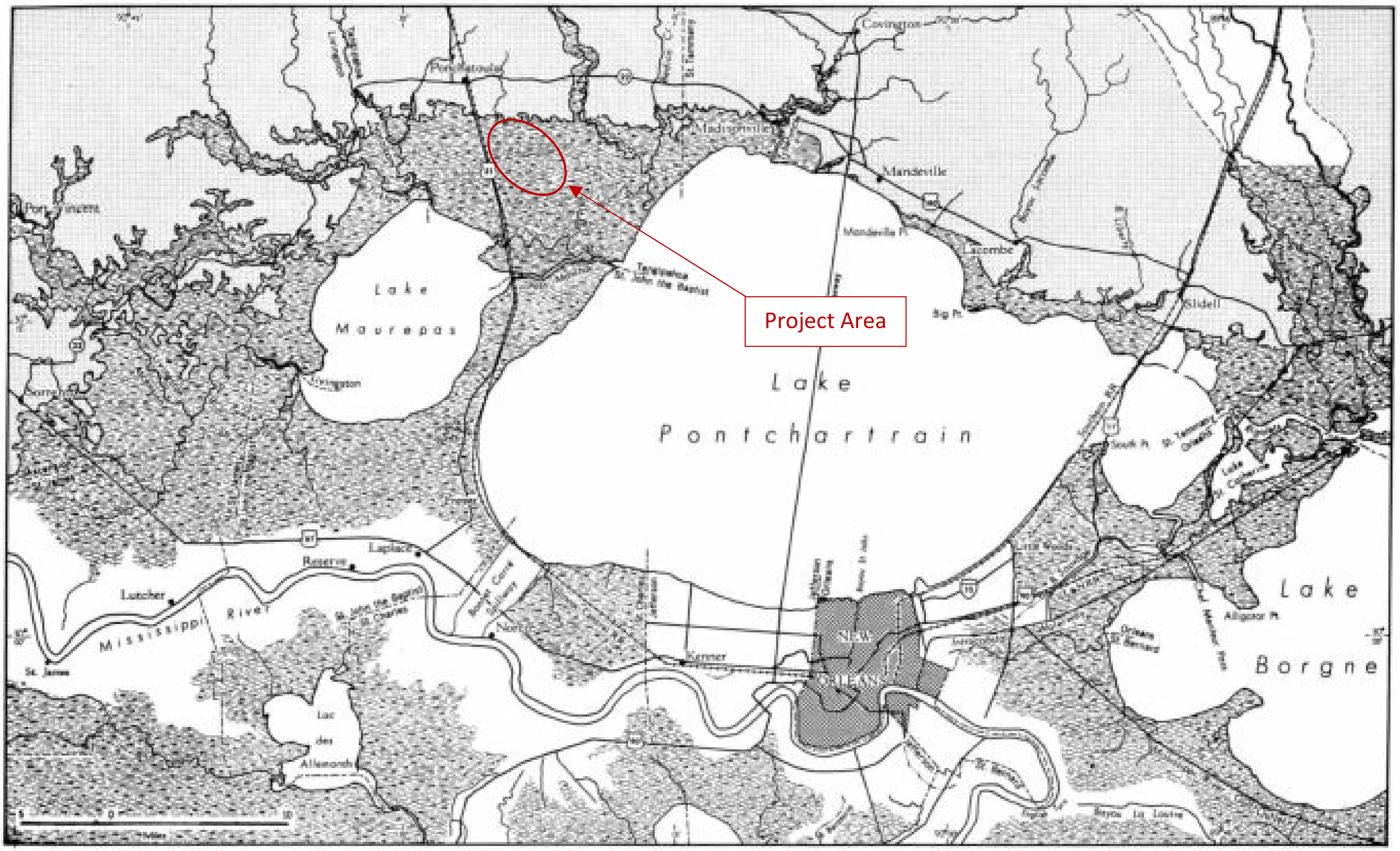
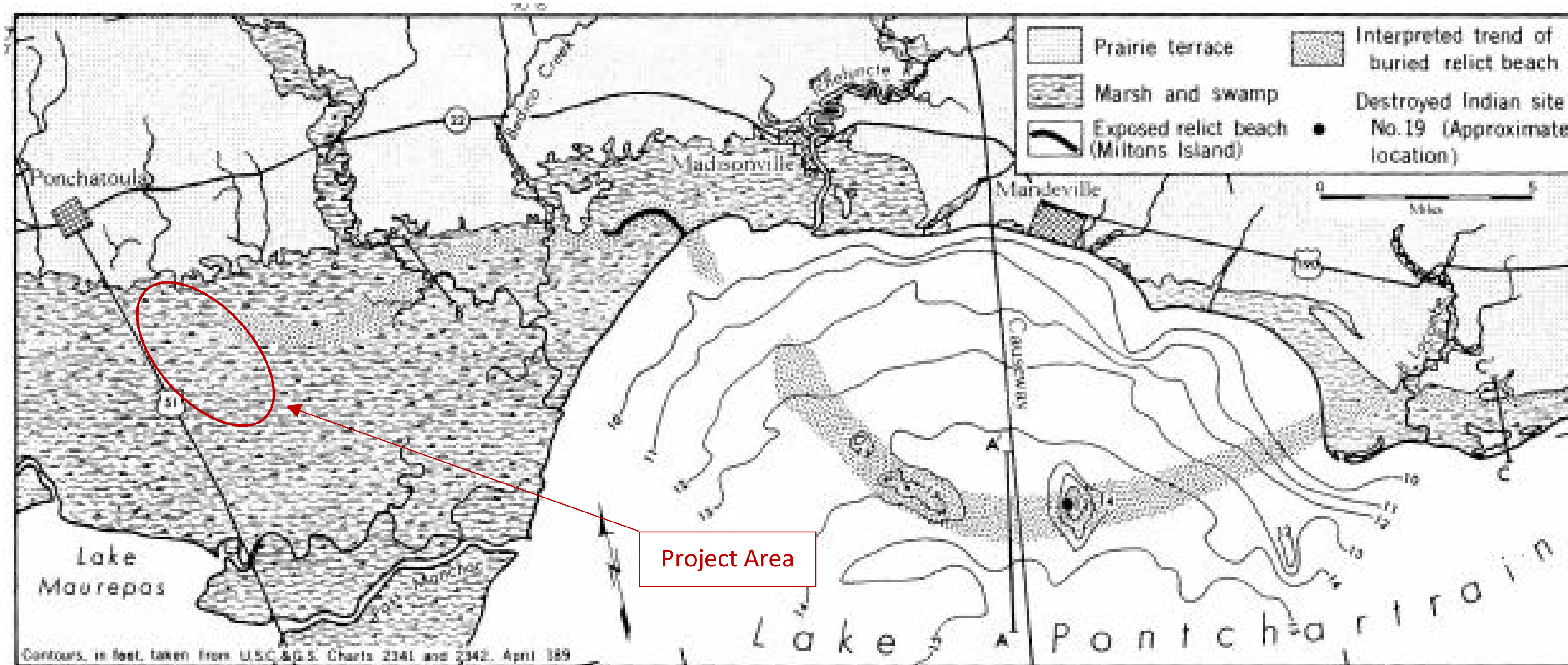


Figure 3.4 – Milton's Island Beach Trend (from Saucier, 1963)



The Milton's Island Beach Trend runs through the area, extending west of Ponchatoula (Saucier 1963). This places the sand ridge directly in the region of Four Mile Marsh and the Hammond Assimilation Wetland. It is likely that this relict sand ridge influenced the flow of water and deposition of sediments during the gradual in-filling process forming the current shoreline of Lakes Pontchartrain and Maurepas, resulting in the modern-day Four Mile Marsh and Seven Mile Marsh (Figure 1.1) and the forested cypress-tupelo swamps to the south that make up the majority of the Joyce WMA.

An origin of Four Mile Marsh based on coastal geomorphology thus appears likely. There is no available evidence that the Marsh was created by human activities such as logging, dredging, fire, drainage, etc. Available evidence points to this being a stable marsh community over the modern period of record, including the 1905 Soil Survey Map of Tangipahoa Parish (USDA 1905, Figure 3.1), which pre-dates the dredging of the South Slough canal and construction of U.S. 51 and I-55 (but does not predate the railroad), the 1943 logging map of Winters *et al*. (1943) and historical aerial photographs from 1965, 1972, 1998, 2005, 2007, 2009, 2010, 2013 and 2018 (USGS, 1965, USGS 1972, USGS 1998, USGS 2005, USDA 2007, USDA 2009, USDA 2010, USDA 2013, Google Maps 2018) (see Appendix B). A non-anthropogenic origin of Four Mile Marsh also dovetails with estimates that the lower organic sediments in the marsh are approximately 1,200 years old (Turner, unpublished data, cited in Bodker *et al*. 2015).

The implication for the City of Hammond project is that it appears Four Mile Marsh does not support, and never did support, hydrology suitable for the development of a forested swamp. (If the area could have been colonized by cypress trees, that would have already happened over the last 1,000+ years). Any measurement of project outcomes should be based on the understanding that the portion of Four Mile Marsh involved in wastewater assimilation will not transition to forested swamp, and any future system-states for that area (with or without effluent addition) will involve herbaceous vegetation (emergent, floating, submerged, salt marsh) or open water.

3.2 Modern History

As discussed in Section 3.1, the origin of Four Mile Marsh (the fresh water marsh in the immediate area of effluent assimilation) is likely due to coastal geomorphology and is not a result of human activities such as logging. However, human impacts have dramatically changed the drainage, hydrology and vegetation of the region.

Habitation of the area by Native Americans was based on the evolving landscape of Lake Pontchartrain, and settlements were likely relocated over time in response to rising sea levels and the gradual shift in ecosystems that occurred in response to the transition from saltwater to fresh water in the Lake Pontchartrain embayment (Saucier, 1963). These settlements did not alter the hydrology and vegetation patterns of the region to any appreciable extent. The early stages of European settlement likewise followed occupation of natural levees and ridges: however later developments evoked significant changes. Lopez (2003) summarized major periods of human impacts on the region as follows:

- 1718-1844: Natural Levee and Ridge Utilization
- 1812-1895: Severing the Mississippi River from Lake Pontchartrain (1812-1895)
- 1890-1938: Commercial deforestation (1932-1990).
- 1950-1989: Water pollution

3.2.1 Mississippi River Flood Control Efforts

The forested swamps and fresh water marshes of the Lake Pontchartrain basin developed over the last 4,000 – 2,900 years (Darnell, 1962; Otvos, 1976) in response to the changing balance of fresh and saltwater. This was primarily driven by the annual flooding of the Mississippi River, which was the major source of fresh water, nutrients and sediments (Day *et al.* 2000) in the region. The sustained effort to channelize the Mississippi River starting in 1812 (Lopez, 2003) led to a gradual reduction in these flood events.

The land level in the region is subsiding (Penland & Ramsey, 1990) due to geologic factors at the same time that sea level is rising (Thomson, 2000) in response to climate change. Historically (in the last 2,900 years), forested swamps in the region have been able to trap and produce enough sediment (accretion) to offset this increase in relative sea level rise (RSLR) (Shaffer *et al*. 2009a). This allowed forested swamps to dry out for a portion of the year, which is critical to seed germination and regeneration of the forest (DeBell & Naylor 1972; Conner *et al*. 1986; Keim *et al*. 2006). Without sufficient accretion, the forested swamps stay permanently flooded. New seedlings cannot be produced and the relict forests will eventually disappear as trees die of old age, disease, salinity increases and blowdowns from storm events (Shaffer *et al.* 2009a; Shaffer *et al*. 2016).

This Mississippi River was the major source of both mineral sediments and nutrients (to stimulate biomass production) leading to vertical accretion and survival of the swamps. Cutting off the Mississippi floodwaters denies forested swamps of these critical inputs as well as a reliable supply of fresh water (Day *et al*. 2000, Day *et al*. 2007, Shaffer *et al.* 2009a, Day *et al*. 2012). The resulting lack of fresh water has resulted in increasing salinity within Lakes Pontchartrain and Maurepas, which is exacerbated by droughts (Day *et al*. 2012, Shaffer *et al*. 2016), and the presence of the Mississippi River Gulf Outlet (MRGO) canal which operated from 1968-2009 (Shaffer *et al*. 2009b; USCOE, 2012). The associated spikes in salinity has resulted in widespread forest death in the swamps (Day *et al*. 2012; Shaffer *et al*. 2016).

3.2.2 Railroads, Highways, Dredging and Logging

In addition to regional impacts associated with Mississippi River flood control efforts, a number of man-made changes have impacted the project area over the years. The New Orleans, Jackson and Great Northern Railway opened in 1854 (Perrin, 2000). This railroad route takes advantage of the Manchac land bridge as a north-south corridor to New Orleans. The railroad has become the defining landscape feature of the area and now forms the western boundary of the Hammond Assimilation Wetland (Figure 3.1).

The original railroad was on wooden trestles and was destroyed during the Civil War. When rebuilt, the railroad was constructed on an earthen embankment that stands today (Keddy *et al*. 2007). The embankment was built from dredged spoils and thus created a north-south canal that facilitates movement of water. At the same time, there are only a limited number of drainage culverts running east-west under the railroad embankment, greatly restricting water flow in that direction.

The railroad alignment served as the route for highway construction (Figure 1.1). U.S. Highway 51 was constructed in 1926 and Interstate 55 (I-55) in the 1960's. I-55 left a wide (>60 m) and deep (>5 m) canal that now serves as a major drainage channel in the region (Lane *et al*. 2016). The I-55 canal routes upland drainage southward directly into Lake Maurepas and also serves as a conduit for salt water to move northward into formerly freshwater wetlands during droughts and storm surges (Keddy *et al*. 2007). The canal is thus considered to be a major contributor to the death of cypress-tupelo forests in the southern half of the eastern Joyce wetlands (Keddy *et al*. 2007).

The Timber Act of 1876 resulted in swampland being declared unsuitable for cultivation and unavailable to private individuals; consequently, large tracts were sold to lumber companies. Leonard Strader acquired over 2,833 ha (7,000 acres) of land northeast of Pass Manchac between 1885 and 1892 as the Strader Lumber Company (later the Owl Bayou Lumber Company) (Keddy *et al*. 2007). This likely would have included the forested swamps south of Four Mile Marsh in the JWMA. The town of Strader (on the railroad north of North Pass, now the location of the Port Manchac intermodal facility) and Owl Bayou (now part of the I-55 drainage canal) are both shown on the Tangipahoa Parish Soil Survey Map of 1905 (USDA 1905, Figure 3.1).

Old-growth cypress trees were massive by modern standards; some being over 3.6 m (12 feet) in diameter and estimated to be more than 1,000 years old (Keddy *et al*. 2007). With the advent of swamp railroads and logging pullboats circa 1890, cypress logging began in earnest, with most swamps cut over by 1925 (Mancil, 1972). Lumber extraction reached a maximum in the early 1900's; by 1934, Louisiana had over 647,497 ha of cutover swamp and only 8,903 ha in remaining cypress forest (Norgress, 1947). This is consistent with the estimated age of cypress trees in the Hammond Assimilation Wetland (Section 8.0). At the Hammond MID monitoring location, all the cypress trees appear to be second-growth (after the logging era) and only one tree sampled could have potentially germinated before 1890. Relict open-water channels from pullboat runs are still readily apparent on aerial photos today throughout the JWMA.

The Hammond Assimilation Wetland was cut off from upland drainage from the north as a result of the dredging of the South Slough canal in the 1960's (Lane *et al*. 2016). The South Slough canal intercepts the drainages of Big Branch Creek and Sealser's Creek as well as overland flow coming off the upland terrace to the north that delineates the edge of the Lake Pontchartrain Basin (Figure 1.1). This canal also intercepts effluent discharges from the City of Ponchatoula wastewater treatment plant. Discharge is directed into the I-55 drainage canal and thence flows south towards Lake Maurepas.

Spoil from the dredging of the South Slough canal forms a spoil bank on the south side of the canal. Under normal water levels, this spoil bank effectively serves as a raised berm and prevents any significant flow of water between the Hammond Assimilation Wetland and the South Slough canal. The effluent distribution pipeline is constructed on this spoil bank (Figure 4.1).

3.2.3 Nutria

Nutria (*Myocastor coypus*) are a non-native wetland herbivore introduced to Louisiana marshes in the 1930's (Holm *et al*. 2011). Nutria are opportunistic feeders that exploit a variety of floating aquatic, submerged and woody species (>60 plant species in Louisiana). Nutria are considered "wasteful feeders", often destroying 10X more plant material than actually consumed (Holm *et al*. 2011). Although the animals are not very mobile, often only moving a few km from their home area, they can reach population densities exceeding 43 animals per hectare (Kinler *et al*. 1987).

There is some speculation that nutria displaced the native muskrat (*Ondatra zibethicus*), which was the foundation of an economically successful fur trapping industry in the 1930's and 1940's (O'Neill, 1949; Boscareno, 2009). However, this is complicated by the fact that muskrat populations collapsed after the 'great muskrat eat out' during 1945-1947 (O'Neill, 1949; Holm *et al*. 2011) prior to widescale competition from nutria.

In addition, alligators, which are a top-down predator of both muskrats and nutria, declined precipitously between 1850 and 1960. As reported by McIlheny (1935), alligators (*Alligator mississippiensis*) were common until 1900, and all but exterminated from Louisiana by 1935. Trapping was suspended in 1962 and alligator harvesting resumed in 1972. As a top-down predator, robust predation by alligators would regulate herbivore populations (including nutria), which would thus affect wetland ecosystems in coastal Louisiana (Keddy *et al*. 2007). These presumed effects would include:

- Increased biomass of plants in marshes.
- Shifts in species composition towards species more favored by grazers.
- Increased land accretion from increased organic matter accumulation.
- Increased rates of regeneration of trees, especially bald cypress.

However, there is considerable evidence that the opposite is occurring. Annual aerial surveys beginning in 1998 indicated that up to 321 – 415 km^2 (80,000 – 100,000 acres) of Louisiana's 14,164 km^2 (3.5 million acres) coastal wetlands were already damaged by nutria, prompting a $68 million USD appropriation under the Coastal Wetlands Planning, Protection and Restoration Act (CWPPRA, 2003) to initiate a coastal Louisiana nutria control program. Nutria control is ongoing, and 170,471 nutria tails were harvested in the 2017-2018 season (LDWF, 2018).

Attempts to regenerate cypress swamps without nutria control have generally been failures, with 100% seedling mortality (Myers *et al*. 1995; Geho *et al*. 2007) or only a very low survival percentage (Conner, 1995).

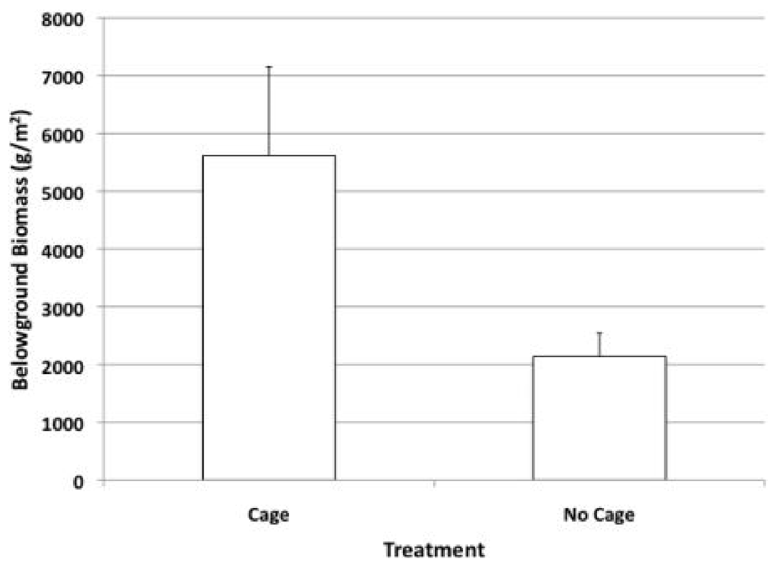
Grazing pressure by nutria also has a strong influence over the development of marsh plant communities as summarized by Holm *et al*. (2011). In an attempt to restore thick-mat *Panicum* (maidencane) floating wetlands, Sasser *et al*. (2005) studied the transplantation of *Panicum* back into *Eleocharis* (spikerush) thin floating mats in an attempt to restore them back into thick-mat *Panicum* systems. The impact of nutria grazing was decisive; in plots available to nutria, essentially 100% of the transplants were consumed within three months, and end-of-season biomass for plots where transplants were protected from grazing was roughly 3X higher (aboveground) and 2X higher (belowground) than the pre-existing condition (with grazing) or other treatments. The differentials in biomass reflect the differences between the thick-mat *Panicum* marsh vs. the thin-mat *Eleocharis* vegetative communities.

The visual effects of nutria grazing on marsh biomass is immediate and obvious (Figures 3.5 and 3.6). At the Hammond Assimilation Wetland, *Typha domengensis* (cattail) displayed nearly 100% cover when protected from nutria. However, cattails in all ten controls where nutria were not excluded were completely destroyed within 48 hours (Day *et al*. 2011; Shaffer *et al*. 2015). The control plots were replanted four times and all four times the cattails were destroyed by nutria (Figure 3.6). In the same study, when plants were protected from nutria, belowground biomass was about 3X greater (Figure 3.7)

Figure 3.5 – *Panicum hemitomon* (maidencane) protected from nutria grazing in Terrebonne Parish (from Holm *et al*. 2011).

Figure 3.6 – *Typha domingensis* (cattail) protected from nutria grazing at the Hammond Assimilation Wetland in 2008, after intense grazing over the fall and winter of 2007-2008 (from Day *et al*. 2019).

Figure 3.7 – Belowground biomass in the freshwater marsh region of the Hammond Assimilation Wetland with nutria exclusion (cages around plant enclosures) and without nutria exclusion (no cages). From Shaffer *et al.* 2015.



While nutrients increase the production of plant biomass (Anisfeld & Hill, 2011; Hillmann, 2011) nutria grazing can offset this increase (McFalls, 2004). Nutrients appear to do more than just grow more biomass; fertilized vegetation is apparently highly preferred by nutria. In a study comparing fertilized and non-fertilized plants, nutria (*Myocastor coypus*) showed a significant preference for fertilized plants. Across three plant species, (*Panicum hemitomon*, *Sagittaria lancifolia*, *Spartina patens*), fertilized vegetation mass loss was 79.4% vs. 9.3% for non-fertilized vegetation when nutria were given a choice about which plants to eat (Ialeggio & Nyman, 2014).

One explanation of the vegetation changes that occurred in the fresh water marsh at Hammond is that it was the result of nutria herbivory (Day *et al*. 2011, Shaffer *et al*. 2015, Day *et al*. 2019). This is a combination of the known effects of nutria grazing (Figures 3.5, 3.6) with the known preference of nutria for fertilized vegetation (Ialeggio & Nyman, 2014). The role of nutria in the Hammond system has been disputed by Bodker *et al*. 2015 and Turner *et al*. 2018, who contend that soil decomposition as the result of nutrient additions is the leading cause of the marsh conversion.

Shaffer *et al*. 2015 reported that over 2,000 nutria were killed after the conversion of the marsh to mudflats and open water, which occurred over an impact area of approximately 122 ha (300 acres) (Turner *et al*. 2018). This would put the nutria density at approximately 16 animals per hectare; almost in the middle of the range summarized by Holm *et al*. 2011 and well below the maximum of 43 animals per hectare reported by Kinler *et al*. 1987.

Plant biomass cycling and nutria herbivory within the Hammond Assimilation Wetland is discussed in more detail in Section 6.3.

3.2.4 Wetland Losses in the Modern Period

The dramatic decline of both fresh water marshes and cypress-tupelo swamps are a regional phenomena unrelated to wastewater assimilation projects. To identify a true “baseline” condition from the region, one would have to go back in time prior to 1812 when channelization of the Mississippi River began

(Lopez, 2003). Since that time, the old-growth cypress trees were logged starting in the 1890's (Mancil, 1972), freshwater marshes were burned to provide muskrat habitat in the 1930's (O'Neill, 1949), nutria were introduced in the late 1930's (Holm *et al*. 2011), alligators were nearly extirpated (McIlheny, 1935), drainage/channelization projects were ongoing (Saucier, 1963), and relative sea level rise is increasing (Penland & Ramsey, 1990; Thompson, 2000) resulting in a doubling of flooding in the Manchac area compared to 50 years ago.

The ongoing drivers of increased flood duration means that cypress-tupelo swamps cannot regenerate (DeBell & Naylor 1972; Conner *et al*. 1986; Keim *et al*. 2006) and once the relict trees are killed by increasing salinity, they cannot come back (Day *et al*. 2012; Shaffer *et al*. 2016). Efforts to regrow swamp forests are stymied by salinity pulses and nutria predation (Conner, 1995; Myers *et al*. 1995; Geho *et al*. 2007; Day *et al*. 2012; Shaffer *et al*. 2016). Without the ability to regrow trees, these areas would become open water (in the short term) followed by a conversion to freshwater or brackish-water marsh, depending on the salinity.

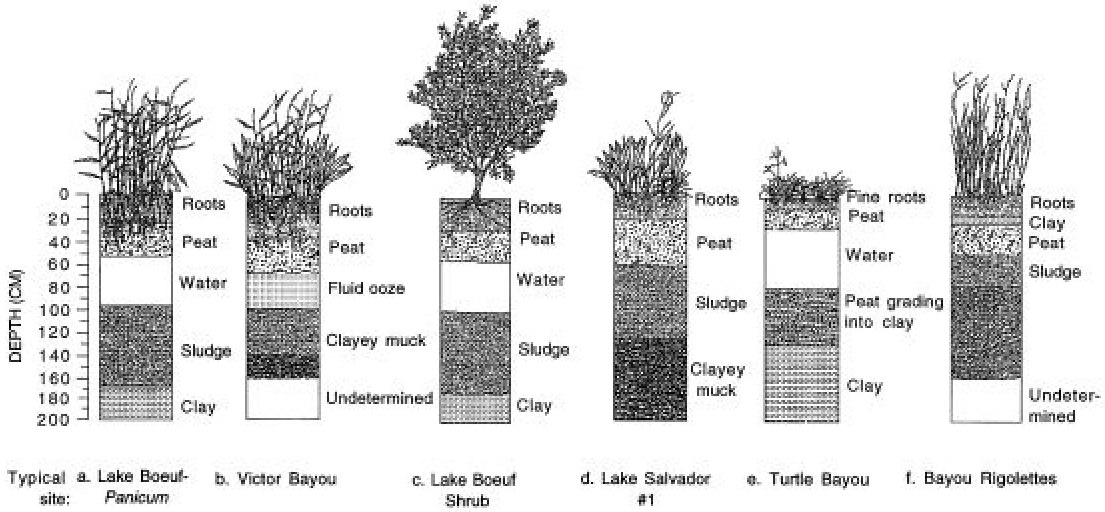
In the 1940's extensive freshwater and brackish water marshes (flotant) were reported and mapped in coastal Louisiana wetlands. O'Neill (1949) described two type of floating marshes in Louisiana, covering over 100,000 ha of the Mississippi Delta. In freshwater areas, O'Neill (1949) described extensive *Panicum hemitomon* (maidencane) marshes that floated freely, easily supporting the weight of a person. The substrate was an organic root-bound mass. Sub-dominant vegetation species included *Typha latifolia*, *Zizaniopsis miliacea* and *Scirpus validus*; also *Sagittaria latifolia* in disturbed areas such as grazed marsh.

This vegetation species list is similar to that compiled by Saucier (1963), and presumably included large areas of marsh that were not yet impacted by nutria, which were introduced in the late 1930's (Boscareno 2009). During this timeframe, muskrat harvests declined steadily from 1945 to 1965, whereas nutria harvests did not begin to any appreciable degree until about 1955 (Holm *et al*. 2011).

Sasser *et al*. (1996) describes five types of floating marsh. He describes thick-mat *Panicum* marshes, dominated by *Panicum hemitomon* in association with *Leerzia oryziodes* and *Sagittaria lancifolia*. This is described as the upper 20-30 cm being a mass of live and dead intertwined roots holding together a decomposing root mass. Below this active root zone, between about 30 to 50 cm, the active roots were fewer and the root mass more decomposed and finer in structure (peat zone). This mat was floating on a distinct layer of water that was usually clear. This type of marsh matches the description of O'Neill (1949) who suggested that these originally developed as attached-growth marshes, but the increasing amounts of organic substrate, and the resulting increase in buoyancy coupled with subsidence, eventually tears the root-bound mat free from the underlying substrate. (Mechanisms of floating-mat development are further described in Section 7.2.1).

Sasser *et al.* (1996) also describes types of floating marshes that appeared to evolve from *Panicum* marshes; in that the new plant communities had colonized rafts of root-bound organic substrate that was deposited by *Panicum* (maidencane) (Figure 3.8). In some cases, these rafts were seasonally buoyant, and could sink during the winter when bacterial activity and gas lift was lowest, and then re-float during warmer seasons when biological activity accelerates.

Figure 3.8 – Different types of floating marshes in coastal Louisiana derived from *Panicum* mats (from Sasser *et al*. 1996).



As seen in Figure 3.8, the Lake Boeuf marsh (a) is *Panicum*-dominated (maidencane) marsh; the Victor Bayou marsh (b) is a *Panicum*-dominated marsh that is apparently in-filling with organic matter; the Lake Boef shrub (c) marsh is a *Myrica*-dominated (wax myrtle) shrub marsh that apparently evolved from a *Panicum* marsh; the Lake Salvador marsh (d) is a *Sagittaria*-dominated (bulltongue) marsh that was formerly mapped by O'Nell (1949) as a *Panicum* marsh, the Turtle Bayou marsh (e) is an *Eleocharis*-dominated marsh that was formerly a *Panicum* marsh (but may have converted to open water prior to being colonized by *Eleocharis*).

Decline of thick-mat *Panicum* floating marshes has been documented in the northwestern Terrebonne Basin of coastal Louisiana (Visser *et al*. 1999), with *Panicum* marshes declining from 67% to 19% of fresh and oligohaline marshes between 1968 and 1992. Conversion of floating marsh areas from thick-mat *Panicum* marshes to thin-mat *Eleocharis* floating marshes increased 3% to 53% over the same timeframe (Visser *et al*. 1999), and in some cases, the thin-mat marshes were adjacent to open-water areas that were formerly marsh.

The thin mat floating marshes are generally thought to be a degraded form of *Panicum* marsh and are dominated by *Eleocharis baldwinii* early in the growing season. Later in the year, other plants such as *Ludwigia leptocarpa*, *Phylla nodiflora* and *Bidens laevis* overtop the *Eleocharis* and dominate the late summer flora (Sasser *et al.* 1996). The root zone of the *Eleocharis* mats is much weaker and easily disrupted. Because the mat is thinner than *Panicum* mats, it is not as buoyant, may seasonally float or sink, and will not support the weight of a man.

In a separate study, Sasser *et al*. (2005) attempted to convert thin-mat *Eleocharis* marshes back to thick-mat *Panicum* marshes by transplanting *Panicum* seedlings. These efforts were unsuccessful unless the plots were enclosed and protected from nutria grazing (Figure 3.6).

It is important to note that the sites studied by Sasser *et al.* (1996) and Visser *et al*. (1999) were not wastewater assimilation wetlands, therefore effluent addition did not play a role in the shifts of the plant communities. Both Sasser *et al*. (1996) and Visser *et al.* (1999) speculated that a variety of factors, including subsidence, salinity, grazing (nutria) and eutrophication could shift a *Panicum*-dominated marsh to another system-state involving different herbaceous species, shrubs, or open water.

In reality, the factors of nutrient availability (eutrophication) and grazing pressure (nutria) are highly inter-related, as Ialeggio & Nyman (2014) demonstrated that nutria strongly prefer fertilized vegetation (compared to non-fertilized) and Holm *et al.* (2011) demonstrated that attempts to restore thick-mat *Panicum* marshes were essentially futile in the face of grazing pressure from nutria.

3.2.5 Possible Future Wetland Ecosystems in Coastal Louisiana

As a result of these known ecosystem drivers (salinity increases, relative sea level rise, nutria, lack of fresh water, nutrient, and sediments), neither freshwater marshes or forested cypress-tupelo swamps are stable ecosystems in coastal Louisiana. Extrapolating from the known changes already occurring, it appears that there are eight possible future system-states of coastal wetlands in the region (adapted and modified from Keddy *et al*. 2007):

1. Preservation of existing natural freshwater marshes. Contingent upon an appropriate fresh water supply (river water diversion or treated wastewater effluent)
2. Conversion to brackish marshes. Rising sea levels and little climate change result in increasing salinity while temperatures remain largely the same. Resultant vegetation shift is to *Spartina patens* and *Juncus roemerianus*.
3. Conversion to open water. Further land subsidence or rising sea level leads to conversion to open water as brackish marsh areas are progressively inundated in the future.
4. Exotic vegetation. Spread of non-native plants suited to warmer climates, plus ongoing climate change could lead to wetlands dominated by herbaceous species such as water hyacinth (*Eichornia crassipes*), alligator grass (*Alternanthera philoxeroides*) and *Colocasia esculenta* (Elephant ear), and tree species such as Chinese Tallow (*Triadica sebifera*).
5. Preservation of existing cypress-tupelo swamps. This requires effectively managing multiple stressors, including salinity, nutrient limitation, herbivory, (Myers *et al*. 2005) and semi-permanent flooding (Shaffer *et al*. 2003).
6. Conversion to bottomland hardwood forests. With large amounts of fresh water plus increased levels of sedimentation, high rates of accretion plus protection from salinity makes it possible that water tupelo (*Nyssa aquatica*) and swamp tupelo (*Nyssa biflora*) could dominate future forested swamps, with associated bottomland hardwood species such as ash and maple.
7. Conversion to scrub-shrub wetlands. With frequent droughts associated with ongoing climate change, areas could become dominated by sea myrtle (*Bacchis halimifolia),* Jesuit's bark (*Iva frutescens)*, and wax myrtle (*Morella cerifera*).
8. Conversion to mangrove swamps. Rising sea levels plus a warmer climate in southern Louisiana could lead to a gradual conversion to mangrove swamps. Black mangrove (*Avicenna germinans*) currently occurs only a short distance south of the project site, and occasional winter frosts appear to be what limits northern expansion.

In the author's opinion, the ongoing effects of rising sea levels and warmer climates result in the most likely future system-states (2, 3, 4, 7 and 8). Active management by human intervention is necessary to preserve freshwater marshes and cypress-tupelo swamps (1 and 5). Conversion to bottomland hardwood forests (6) would require an unprecedented amount of human intervention and represents the most unlikely future outcome.

4.0 Project Description

The City of Hammond operates a wetland assimilation system, termed the South Slough Wetland, which is regulated by the Louisiana Department of Environmental Quality (LDEQ) under permit LA0032328. Requirements imposed by LDEQ under the permit are summarized in Section 2 and are consistent with other wetland wastewater assimilation systems in the State.

Wastewater from the City of Hammond is treated in a 3-stage aerated lagoon to secondary treatment standards. After biological treatment, secondary effluent is disinfected (chlorination/dichlorination) prior to being discharged to the assimilation wetland. At the assimilation wetland, flows are distributed along a piping system approximately 1,200 meters long located on the northern edge of the wetland assimilation area (Figure 4.1). Water flows primarily to the south, (Figures 1.1 and 1.2) entering the Joyce Wildlife Management Area (JWMA), which contains over 4,000 hectares (10,000 acres) of wetland habitat east of Interstate 55, with flows eventually entering the northern reaches of Lake Pontchartrain (Lane *et al*. 2016).

Figure 4.1 – South Slough distribution pipe and boardwalk

Throughout the various studies and reports to date, different areas and terminologies have been used to describe the wetland system. For the sake of clarity, this report uses the following definitions:

- South Slough Wetland: The total area of wetlands considered for effluent utilization under LPDES Permit LA0032328, including approximately 4,050 hectares (10,000 acres) of the East Joyce Wetlands (EJW) and the effluent distribution area owned by the City of Hammond (UAA, 2005).
- Effluent distribution area: Land owned by the City of Hammond and utilized for purposes of effluent distribution. This area is approximately 230 acres (Hunter *et al*. 2018).
- Four Mile Marsh: The area of naturally-occurring emergent wetland vegetation (including the effluent distribution area) that existed prior to the effluent application project. The boundaries of Four Mile Marsh appear unchanged from historical aerial photographs dating back as far as 1965 (USGS 1965) and the 1905 soil survey map (USDA 1905). This area is approximately 750 acres (Lundberg *et al*, 2011)
- Hammond Assimilation Wetland area (HAW): Area of the Four Mile Marsh and adjacent forested swamps where changes in the vegetative community have occurred after effluent application began in 2006. While this area has not been field delineated, it is estimated at 122-130 hectares (300-320 acres) (Shaffer *et al*. 2015; Turner *et al*. 2018). Calculations to support a detailed estimate of the extent of the HAW are summarized in Appendix C.
- Joyce Wildlife Management Area (JWMA): The state wildlife area managed by the Louisiana Department of Wildlife & Fisheries including the effluent distribution area. This is approximately 34,600 acres (Lane *et al*, 2016) and includes all of Four Mile Marsh not owned by the City of Hammond. The area east of I-55 is sometimes termed the East Joyce Wetlands (EJW). Other than two emergent marsh areas (Four Mile Marsh and Seven Mile Marsh), the majority of the EJW is forested swamp.

The wastewater assimilation project is based on a Use Attainability Analysis completed by Comite Resources Inc. (UAA 2005). At the time the UAA was completed, the method available to system designers was the “loading chart” approach of Nichols (1983) and Richardson & Nichols (1985). The “loading chart” approach makes the assumption that the entire 10,000 acres (4,050 hectares) of the South Slough Wetland would be involved in effluent assimilation. Because the available project area is very large, the projected “loading rates” (mass load divided by entire wetland area) were low, and based on the charts presented in Nichols (1983) and Richardson & Nichols (1985), percentage removals of nitrogen (N) and phosphorus (P) were projected to be high. The limitations and drawbacks of using the “loading charts” of Nichols (1983) and Richardson & Nichols (1985) as a predictive design tool are discussed further in Appendix C.

The UAA (2005) characterizes the South Slough Wetland as:

> *“Flora communities in the South Slough wetland are mostly cypress-tupelo-willow forested wetlands to the north of South Slough, transitioning south of the slough into cattail-Sagittaria dominated marsh. After being processed by the South Slough wetland effluent will flow into the Joyce Wildlife Management Area. This area is characterized by freshwater forested wetlands and fresh to brackish marshes dominated by spartina sp., with minor species consisting of bulltongue, maidencane, alligatorweed, cattail, common rush, pickerelweed, swamp milkweed and swamp knotweed.*
>
> *It is noteworthy that the wetlands immediately south of the spoil bank bordering South Slough are dominated by cattail and willow. This likely reflects perodic inflow of high nutrient waters from South Slough. It is expected that the cattails and willow will expand in the freshwater marsh of the South Slough wetland, but there should be no composition changes in freshwater marshes further south and east in the JWMA wetlands.”*

With the benefit of hindsight, this seems an inadequate description of the project site, given that Four Mile Marsh is the major effluent assimilation area. The UAA describes areas north of South Slough (which are not part of the project) and describes areas of the Joyce WMA that are well downstream of the active assimilation zone. The UAA does note that there is an area of "cattail and willow" immediately downstream of the effluent distribution pipe, and notes that the pre-existing cattails and willows may expand within the freshwater marsh, but there would be no composition changes in freshwater marshes further south and east in the Joyce WMA wetlands. This description of the assimilation area would make it easy for the reader to assume that no major vegetation changes would happen as a result of the project.

Again, with the benefit of hindsight, this turned out to be inaccurate due to several factors.

- The origin of Four Mile Marsh appears to be poorly understood in the UAA. That Four Mile Marsh likely had its origins from coastal geomorphology (Section 3.1); that drainage changes the region meant the pre-existing vegetation developed from hydrology that no longer existed (Section 3.2); and that the pre-existing *Panicum* marsh community may have not been all that stable (Section 3.2.4) were not addressed.
- By describing the area as cattails and willows that may expand, this creates the impression that vegetation changes would be "more of the same" instead of the large-scale shifts that actually occurred in Four Mile Marsh.
- By stating that "no composition changes" would occur at some point further south and east of the distribution pipeline, no quantitative description of the expected area of the active assimilation zone was put forth. This statement is sufficiently vague enough that it could be interpreted many different ways.

The actual trajectory of the vegetative community turned out to be very different than that described in the UAA. Within approximately one year after effluent discharge commenced, the marsh vegetative community began to deteriorate, and within months, nearly the entire marsh south of the discharge had converted to open water and mudflats (Shaffer *et al*. 2015). After conversion to open water and mudflats, two alternative narratives have been presented in the scientific literature:

1. Vegetation changes were brought about by intense nutria grazing (Day *et al*. 2011; Lundberg, 2011; Shaffer *et al.* 2015). Adding nutrients to the area could have made the existing vegetation more attractive to nutria (Iallegio & Nyman, 2014), and regrowth was heavily impacted by nutria (Day *et al*. 2019).
2. Vegetation changes were the result of nutrients. Nutrient addition lead to reduced belowground biomass and structural weakening of the marsh soil (Bodker *et al*. 2015, Turner *et al*. 2018).

The nature of the pre-existing vegetation and the marsh "recovery" is also in dispute. Bodker *et al*. 2015 and Turner *et al*. 2018 describe the pre-existing marsh attached *Panicum*-dominated mats. Day *et al.* (2011), presents data that the pre-existing marsh vegetation was relatively diverse (Table 4.1) and that recovery was underway by 2010. Shaffer *et al.* 2015 states that marsh recovery was underway by 2010, but also dependent on nutria activity and the cool-season extent of *Hydrocotyle* (water pennywort), a floating species

Table 4.1 – Dominant vegetation in the marsh area within the Hammond Assimilation Wetland (from Day *et al*. 2011, Day *et al*. 2019)

Species	2006	2010	2019 [(1)]
Open Water	---	30%	---
Panicum hemitomon (maidencane)	15%	10%	---
Sagittaria lancifolia (bulltongue)	20%	10%	X
Alternanthera philoxeroides (alligator weed)	10%	10%	X
Ludwegia leptocarpa (willow primrose)	---	40%	X
Eleocharis cellulosa (spikerush)	20%	---	---
Eleocharis quadrangulata (spikerush)	15%	---	---
Polygonum punctatum (smartweed)	10%	---	X
Typha domengensis (cattail)	10%	---	X
Zizaniopsis miliacea (rice cutgrass)	---	---	X
Hydrocotyle spp. (pennywort)	---	---	X

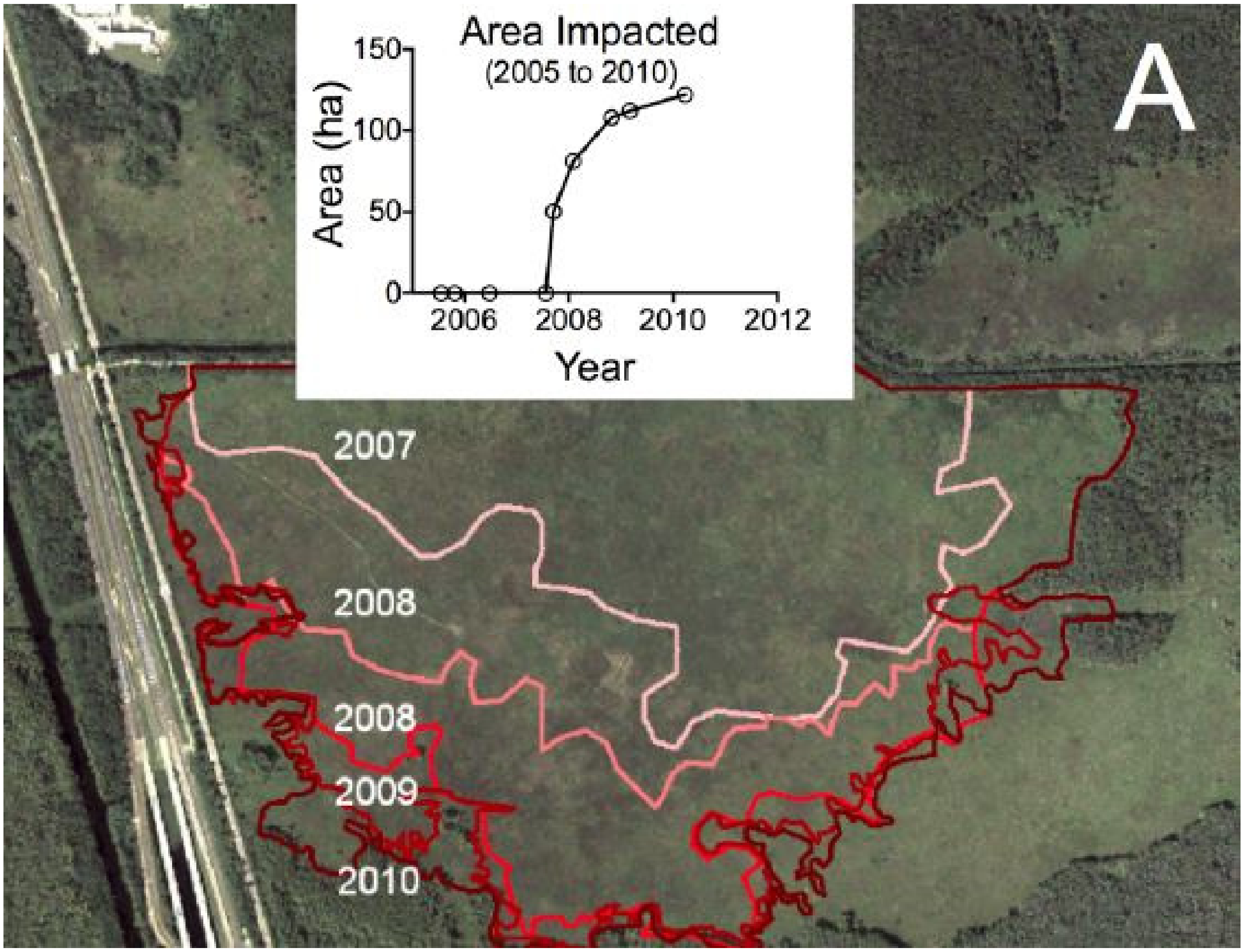
Note:

1. Data from Day *et al* (2019) and presumed to come from previous growing seasons; species listed as present are indicated with an "X", percentages were not provided. 2006 and 2010 data from Day *et al*. (2011)

The 2006 species survey in Table 4.1 depicts a plant community that likely developed from a relict *Panicum* mat (Figure 3.8) as described by Sasser *et al*. (1996). Additional evidence on marsh recovery is presented in Weller & Bossart (2017) who studied insect diversity at the site in 2007 (original marsh vegetation), 2008-2009 (declining marsh) and 2012 (recovering marsh). Their findings indicate that insect diversity closely matched vegetative diversity, and the recovery of insect diversity by 2012 was indicative of a more diverse marsh vegetative community.

Bodker *et al*. 2015 and Turner *et al*. 2018 contend that the original marsh was anchored (not floating) and dominated by *Panicum hemitomon*, and that nutrient addition resulted in the collapse of the *Panicum*-dominated marsh community and the conversion of the system to floating mats and open water. The conversion to open water as described by Bodker *et al.* 2015 and Turner *et al*. 2018 reported occurred between 2007 and 2010 (Turner, 2019), as shown in Figure 4.1.

Figure 4.2 – Estimated conversion to open water/mudflats (from Turner, 2019)



As a practical matter, the area of impact is quite large, 300+ acres and difficult to survey in a comprehensive manner. Access to most of the area is by airboat or drone only; surveying a small subset of the overall assimilation area could lead to very different conclusions about the vegetative community and structure. Most interpretations to date have been based on aerial photos (Shaffer *et al.* 2015; Turner 2017, 2019) which can be further complicated by mats that are seasonally buoyant (Sasser *et al*. 1996), the presence of seasonal floating vegetation (Shaffer *et al*. 2015) and fluctuating water levels. Based on satellite imagery from 1956-2015, there have been periods of open water within Four Mile Marsh prior to the effluent assimilation project (Allen, 2016), so it is by no means clear that the pre-existing marsh community was stable.

The fact that there is not a clear baseline of the Four Mile Marsh area (comprehensive species inventory, extent of anchored vs. floating mats) is a major limitation of the UAA study (UAA, 2005) and a major factor in the ongoing debate.

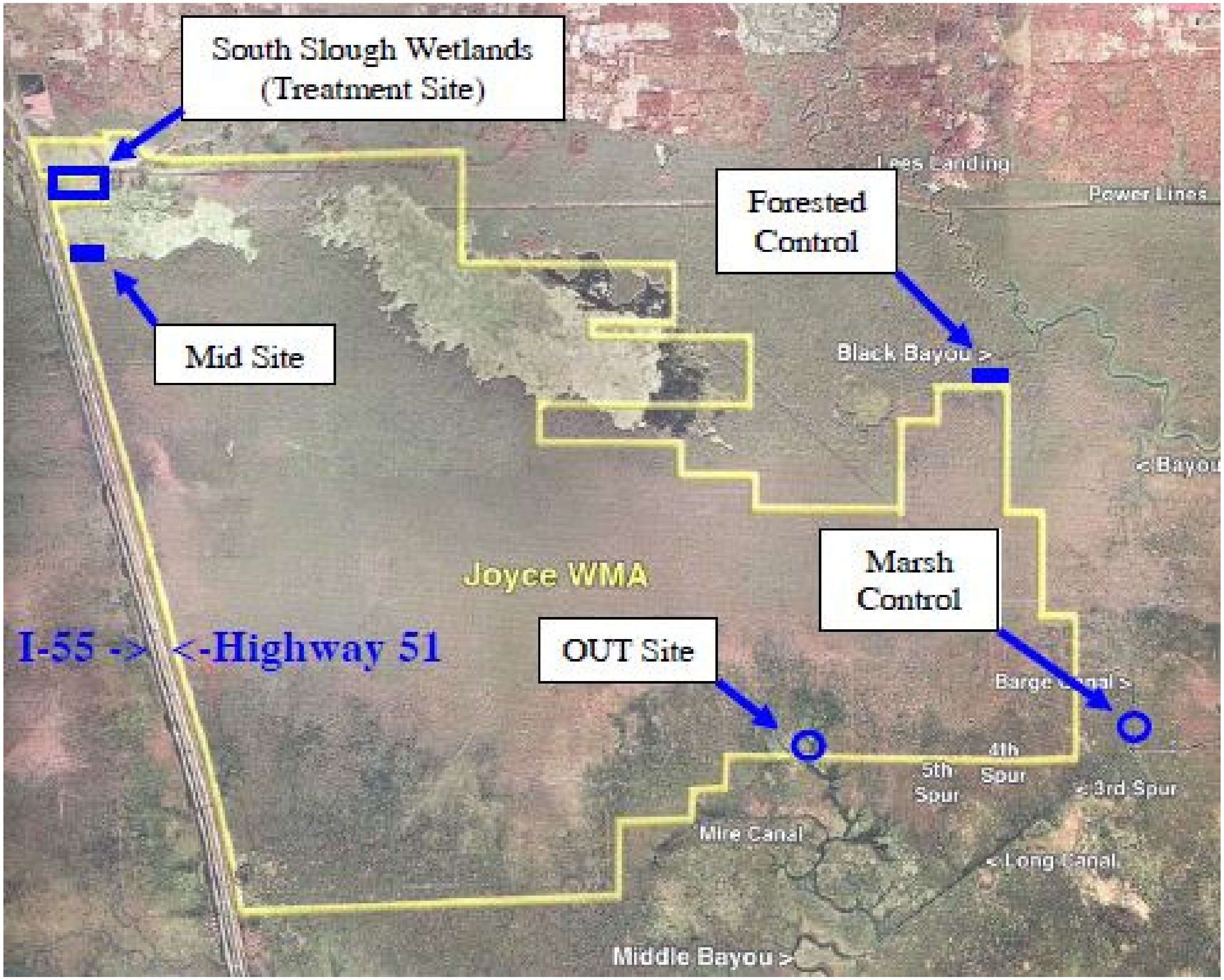
5.0 Permit-Related System Compliance

Performance criteria for the City of Hammond wetland assimilation system are defined in LPDES Permit LA0032328 (Section 2). Monthly discharge monitoring reports (DMRs) and annual wetland system monitoring reports (Comite Resources 2007, 2008, 2009, 2010, 2011, 2012, 2013, 2014, 2015, 2016, 2017) were obtained from LDEQ.

As discussed in Section 4, effluent is spread over a 1,200-meter elevated distribution pipeline (Figure 4.1). The site is hydrologically isolated on the north and west sides to a large extent and effluent spreads though the South Slough wetlands generally in a south and east direction towards Lake Pontchartrain (Section 3). The South Slough wetland area is somewhat arbitrarily considered to be roughly 4,500 hectares (10,000 acres) within the much larger East Joyce Wetlands, which are roughly 14,000 hectares (34,600) acres in extent). The zone of active nutrient assimilation is much smaller, and has generally been considered the section of Four Mile Marsh immediately downstream of the effluent distribution pipe (Figures 1.2, 4.1). The zone of active nutrient assimilation is further discussed in Section 6.

Locations of the various monitoring points (UAA, 2005) are shown in Figure 5.1. The Forested Control monitoring location was moved in 2012 to a new location (northeast of the effluent distribution pipeline) believed to be more representative of baseline forested swamp conditions in the area (Hunter *et al*. 2018). As discussed further in Section 6, all monitoring locations except for NEAR and MID are well outside the active assimilation zone of the wetland.

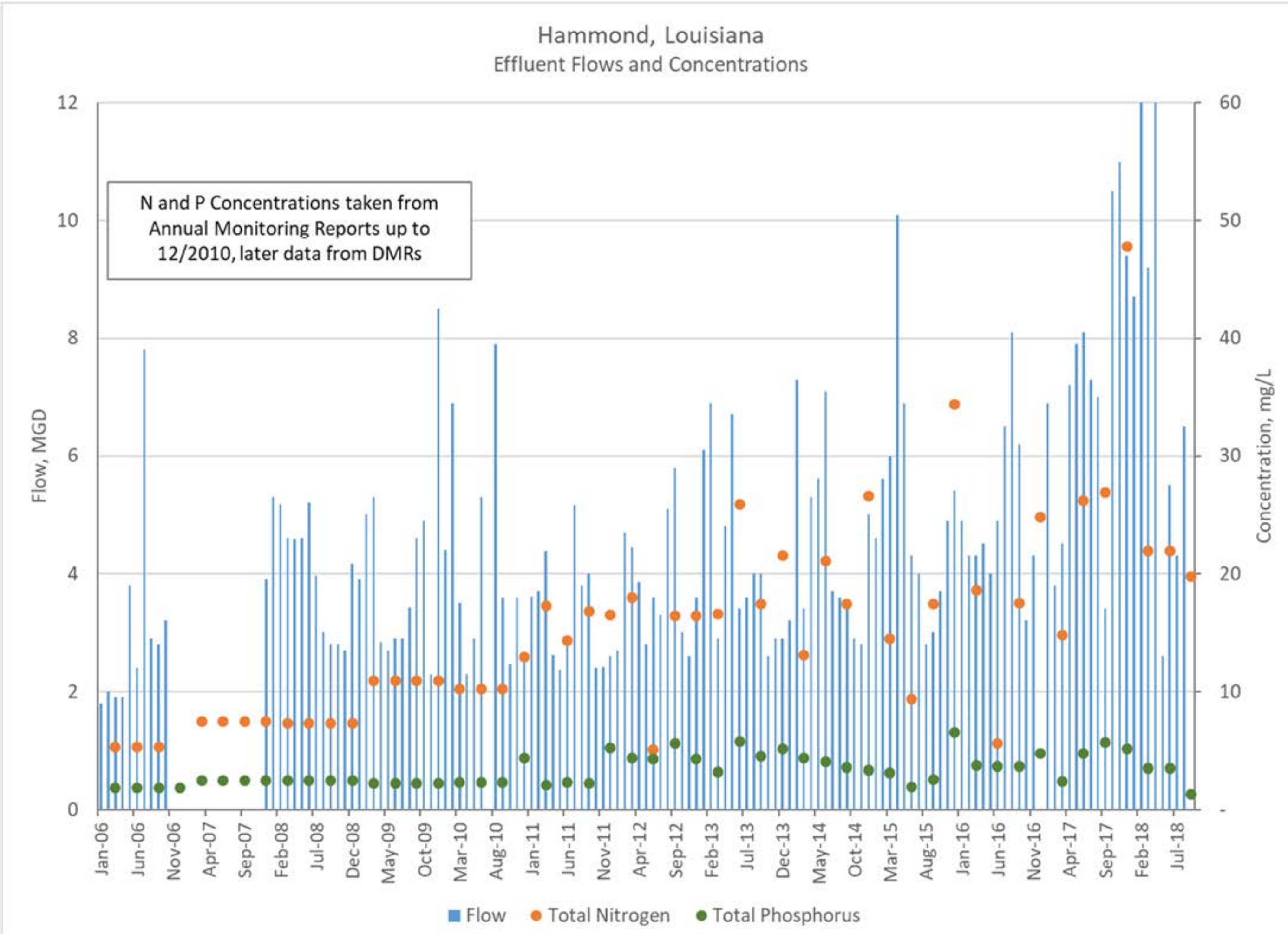
Figure 5.1 – LA0032328 Monitoring Locations (from UAA, 2005)



5.1 Effluent Application and Extent of Effluent Spreading

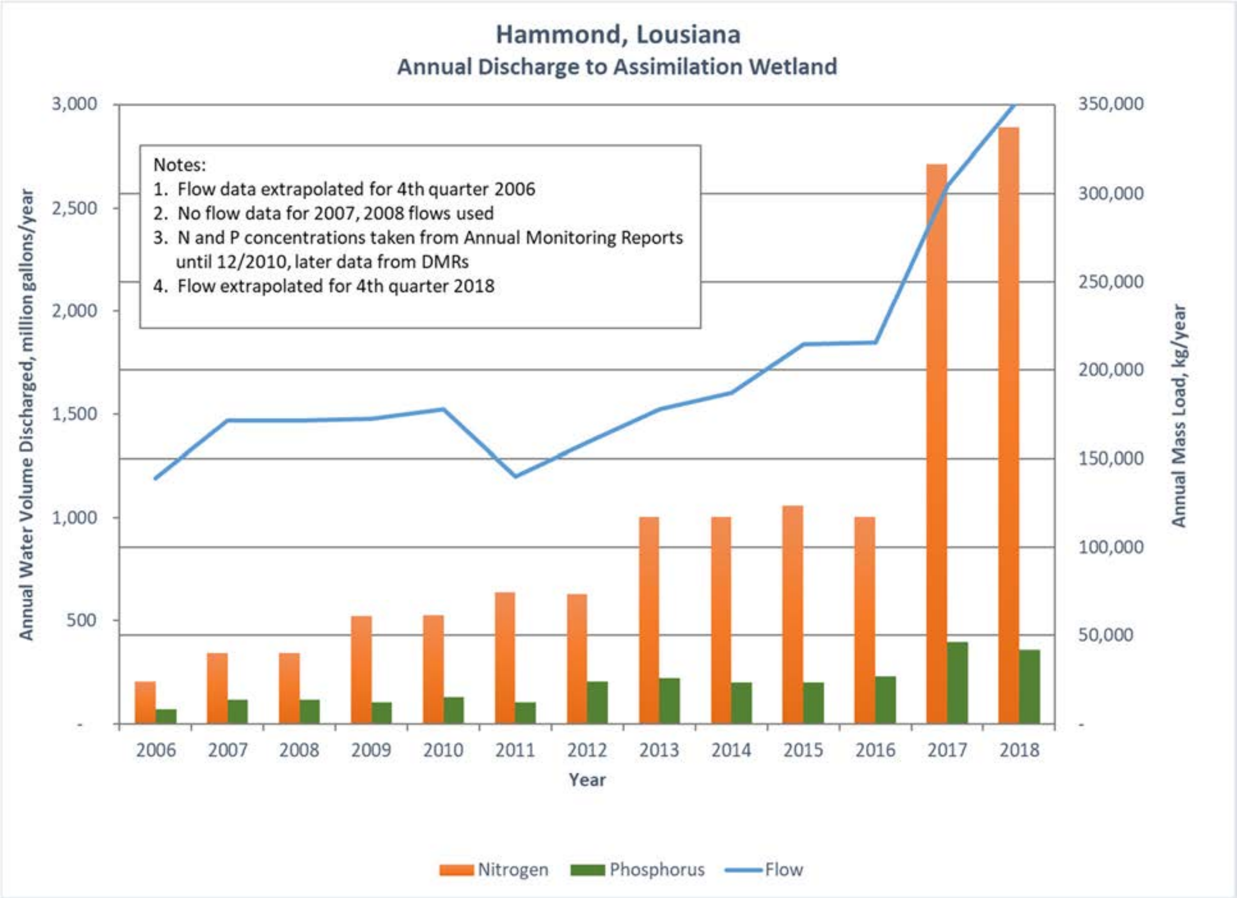
Effluent application (monthly flows and concentrations of total nitrogen and total phosphorus) are summarized in Figure 5.2:

Figure 5.2 – Effluent flows and concentrations applied to the South Slough Wetland



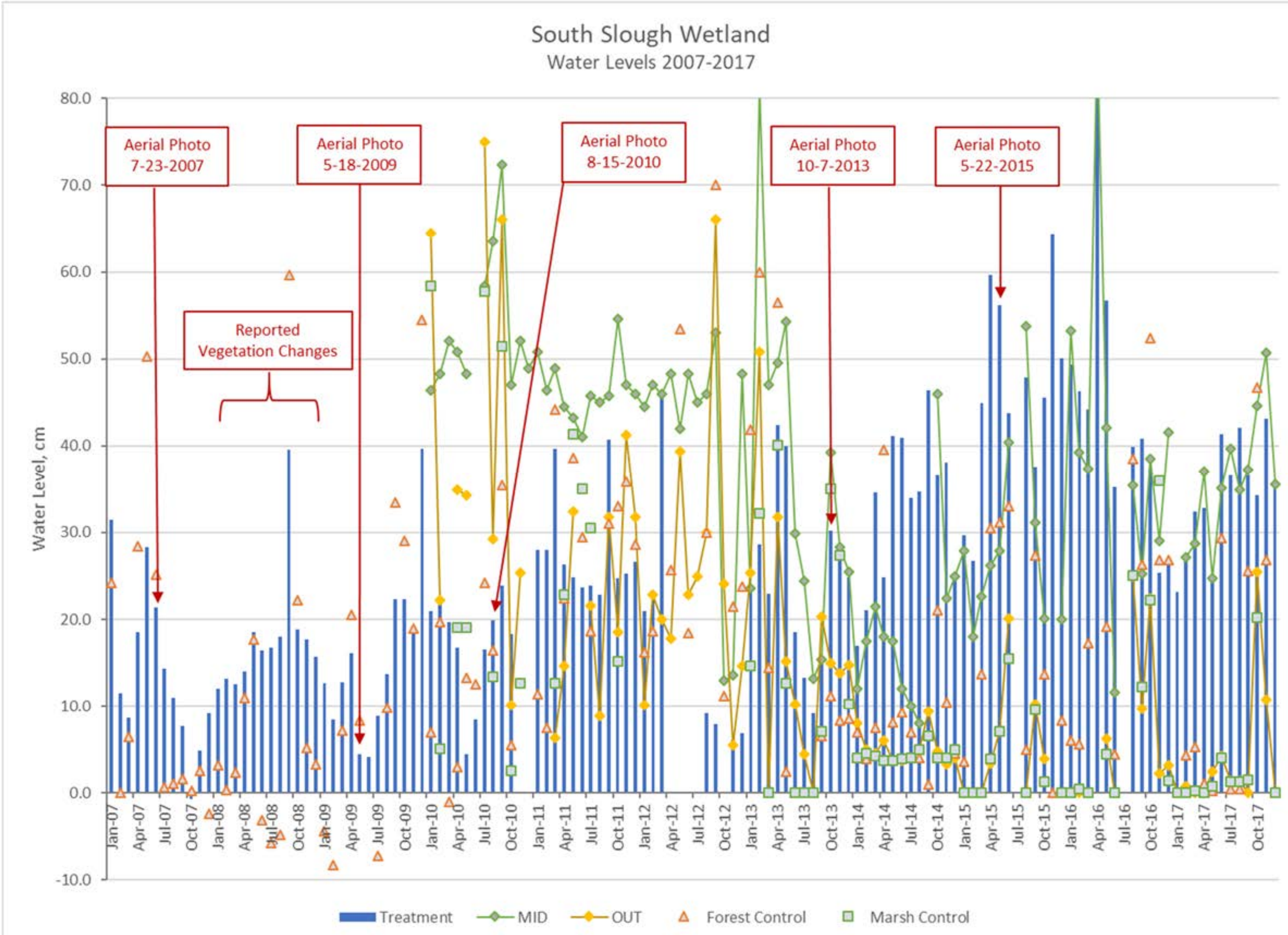
As seen in Figure 5.2, there has been an increase in both flow and concentration over time. Combining the flow and concentration data into annual discharge loads results in Figure 5.3.

Figure 5.3 – Annual discharge loads of nitrogen and phosphorus applied to the South Slough Wetland



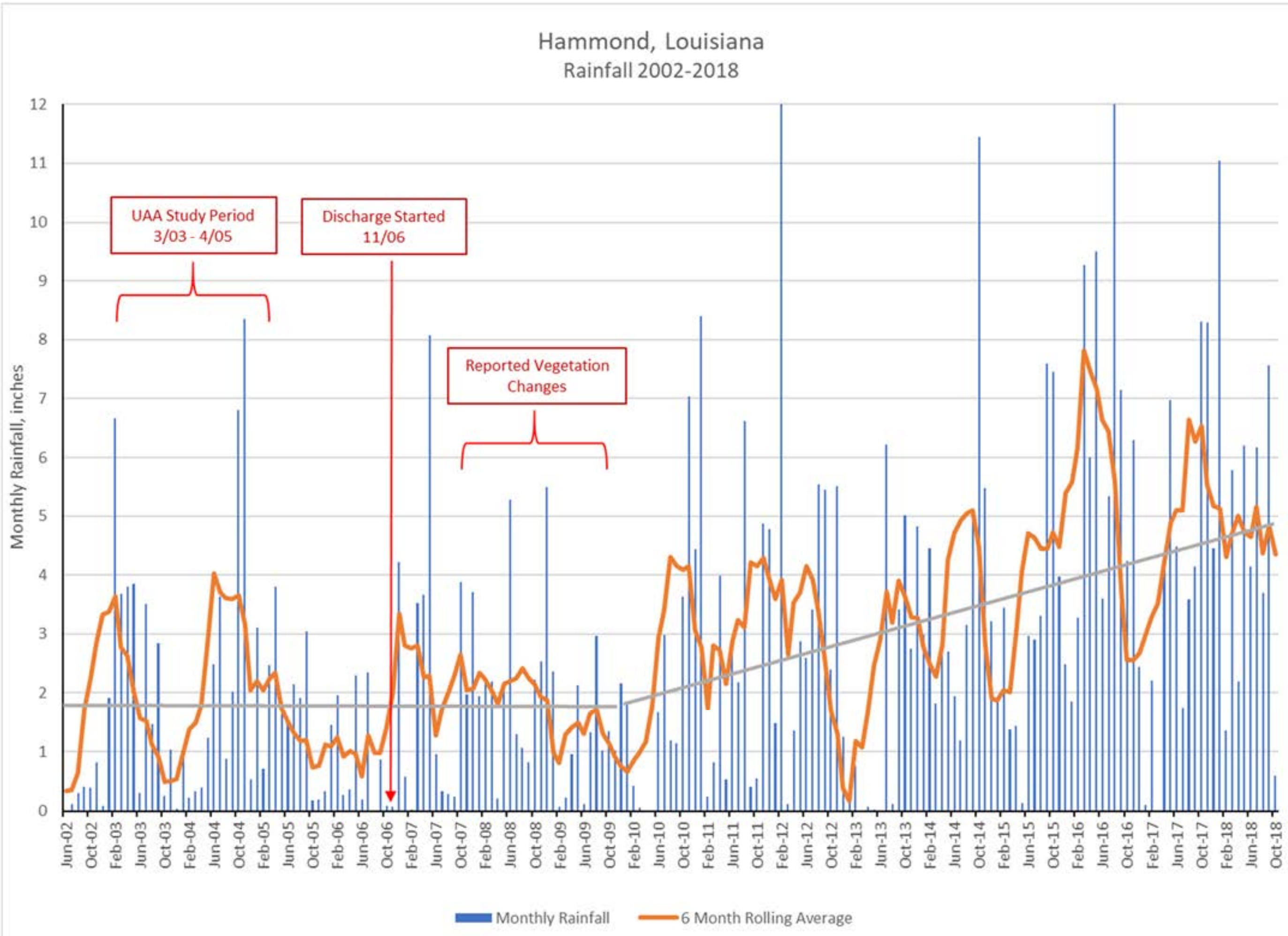
Effluent application represents a stable supply of fresh water to the receiving wetlands which has resulted in more stable water levels within the assimilation area compared to background monitoring locations. Figure 5.4 summarizes monthly water levels recorded at the site. Because water levels have varied over the years, these are correlated to the historical aerial photos summarized in Appendix B.

Figure 5.4 – Monthly water levels recorded at the South Slough Wetland



An important contributor to the site hydrology is precipitation, because channelization projects to the north and west (Section 3) have largely cut off other water inputs. Monthly rainfall data for the City of Hammond (June 2012 – October 2018) was summarized; results are presented in Figure 5.5.

Figure 5.5 – Monthly rainfall totals for the City of Hammond, Louisiana



As seen in Figure 5.5, rainfall amounts in the area have been increasing since 2010. The combination of this increasing rainfall and the increasing effluent flows (Figure 5.2) is the likely cause of the increase in water levels observed at the NEAR monitoring location (Figure 5.4).

During the marsh conversion reported by Day *et al*. 2011, Lundberg *et al*. 2011, Bodker *et al*. 2015, Shaffer *et al*. 2015, Turner *et al.* 2018, Day *et al*. 2019, neither effluent flows (Figure 5.2), water levels (Figure 5.4) and rainfall (Figure 5.5) were particularly high or low. This seems to rule out a hydrologic cause (flooding or drought) for the observed vegetation changes in Four Mile Marsh (Figures 1.2, 4.1).

The extent to which effluent can theoretically spread is a function of the difference between precipitation (P) and evapotranspiration (ET). When ET exceeds, P, there is a net water loss from the wetland and effluent is consumed to make up this water deficit. When P exceeds ET, there is a surplus of water and effluent can theoretically spread to the outlet of Lake Pontchartrain. Data from the IWMI World Water and Climate Atlas (IWMI, 2009) was downloaded for New Orleans, Louisiana to assess the net monthly differential between P and ET. Results are shown in Figure 5.6.

As seen in Figure 5.6, there is a net deficit of water (ET exceeds P) during the Spring and Fall. Late summer is close to a water balance, and winters tend to be wet (P exceeds ET). The results of Figure 5.6 and the monthly flow data of Figure 5.2 were used to calculate the extent of effluent spreading. These results are summarized as a percentile distribution in Figure 5.7.

Figure 5.6 – Average precipitation and evapotranspiration for New Orleans, Louisiana

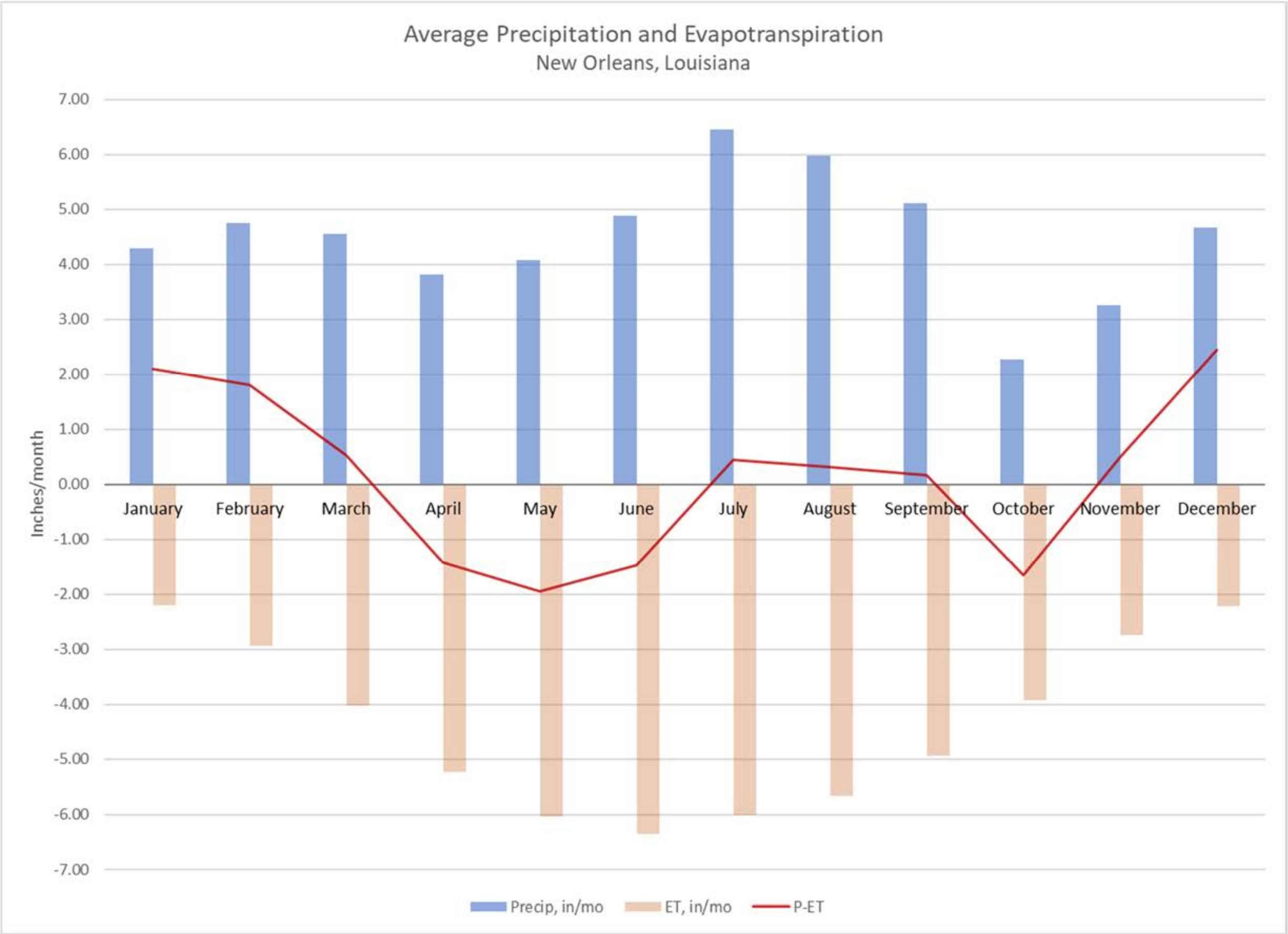
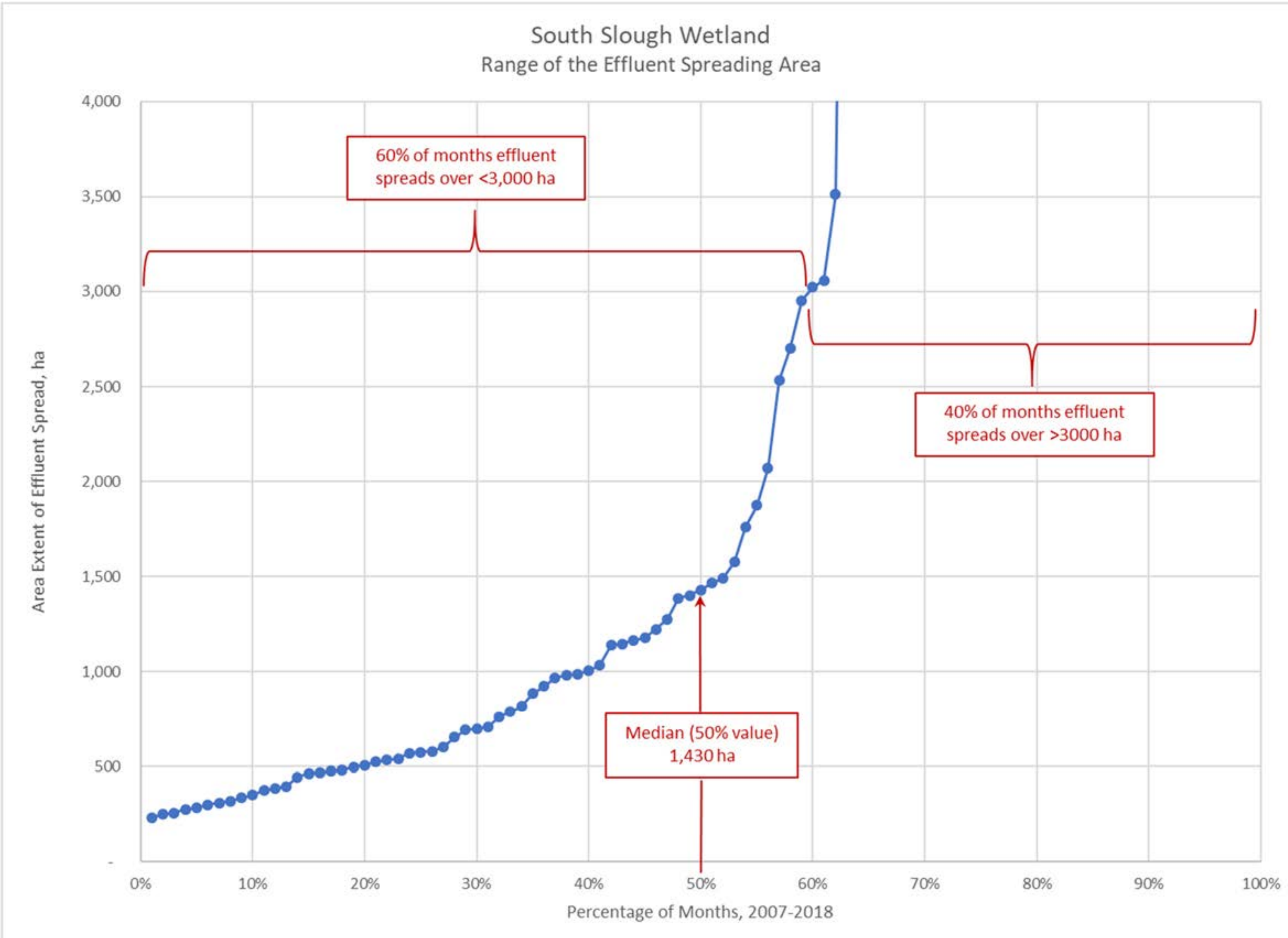


Figure 5.7 – Range of effluent spreading area within the South Slough Wetland



As seen in Figure 5.7, about 40% of the time, effluent can theoretically spread over the entire South Slough Wetland area. This assumes uniform sheet flow, which is likely not the case (Blahnik & Day 2000), but uniform sheet flow is more likely during periods of water abundance. This is when P exceeds ET. During other months of operation, there was a net water deficit (ET exceeded P) and it was assumed the volume of effluent applied that month was consumed to offset the water deficit. This results in the percentile distribution seen in Figure 5.7. For about 60% of the time, water spread to less than 3,000 ha (7,400 acres), and 50% of the time, that extent of effluent spread was less than 1,430 ha (3,500 acres). It is important to note from Figure 5.7 that the 122 ha (300 acres) of the marsh conversion area was always within the zone of effluent spread. There was never a hydrologic limitation on effluent being able to spread across the marsh region and into the forested swamps south and east of the MID monitoring location.

5.2 Permit-Monitored Nutrient Assimilation and Biomass Productivity

Results for water quality parameters (ammonia-nitrogen, total kjeldahl nitrogen (TKN), total phosphorus) and plant biomass productivity (end of season live biomass (EOSL), litterfall production and stem growth) are summarized in Figures 5.8 – 5.15.

As seen in Figures 5.8, 5.9. 5.10 and 5.11, ammonia nitrogen, TKN, nitrate nitrogen and phosphorus are removed (assimilated) between the NEAR and MID monitoring locations. The other monitoring locations (OUT, Forest Control and Marsh Control) are outside the zone of active assimilation and represent background concentrations. Since the MID concentrations are higher than the background concentrations, one can conclude that the MID location is still within the zone of active nutrient assimilation (further discussed in Section 6). One of the stated benefits of wetland wastewater assimilation projects is that they remove nutrients through assimilative processes (discussed further in Section 2). Based on the available data, the South Slough Wetland project is clearly successful in removing nutrients. Water quality improvement from wetland assimilation is summarized in Table 5.1.

Figure 5.8 – Ammonia Nitrogen (NH_4-N) assimilation in the South Slough Wetland

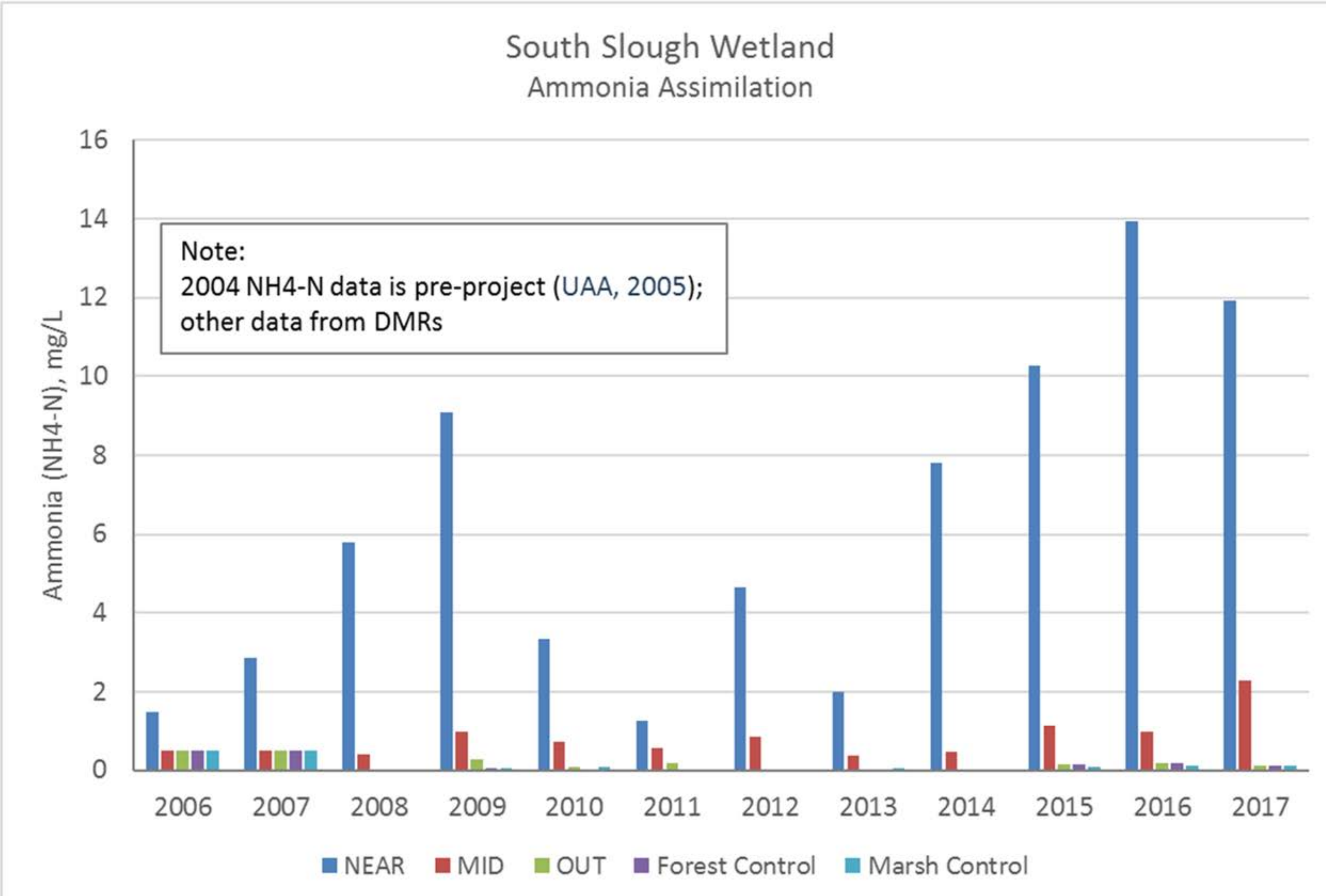


Figure 5.9 – Total Kjeldahl Nitrogen (TKN) assimilation in the South Slough Wetland

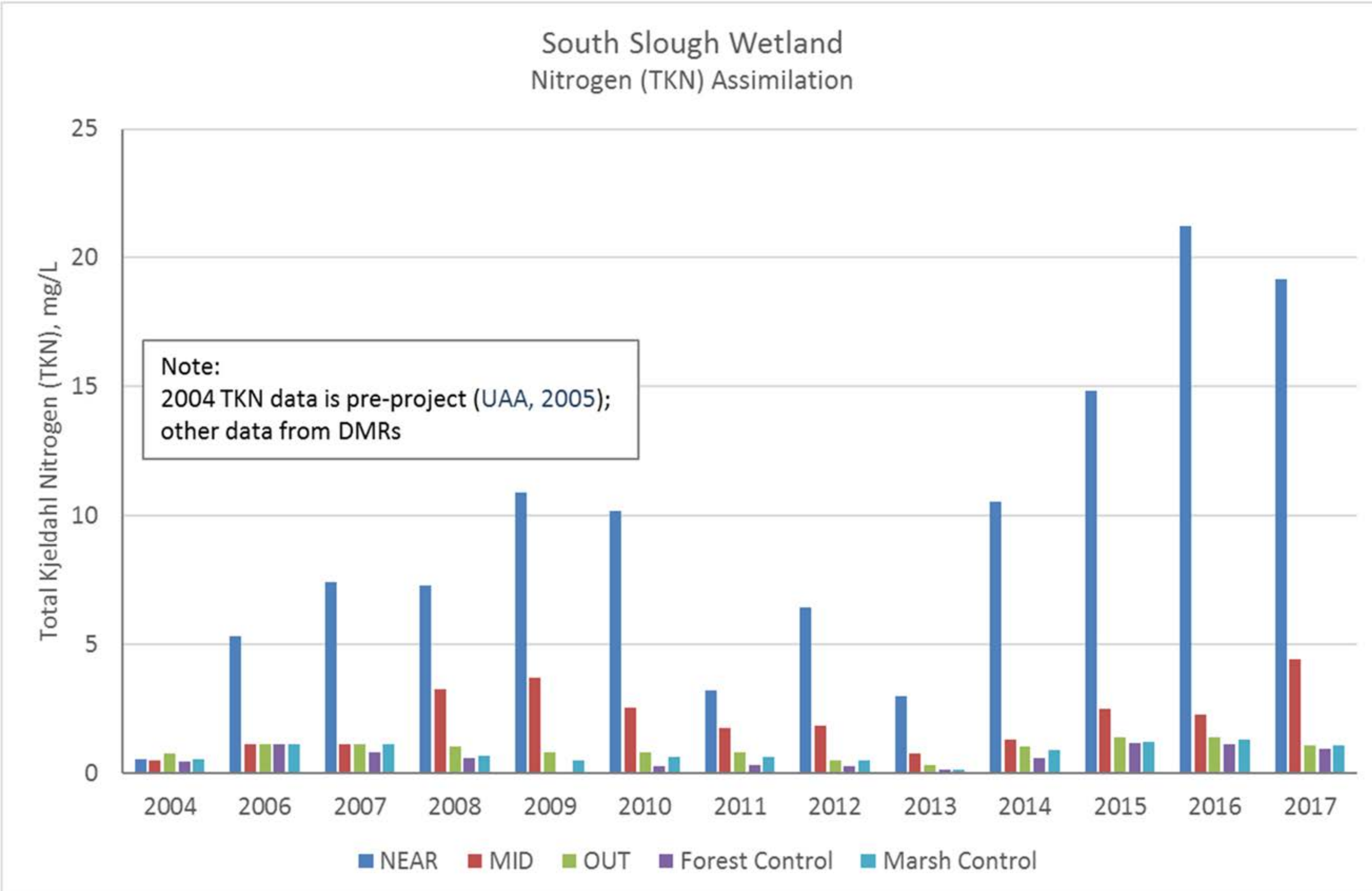


Figure 5.10 – Nitrate Nitrogen (NO_3-N) assimilation in the South Slough Wetland

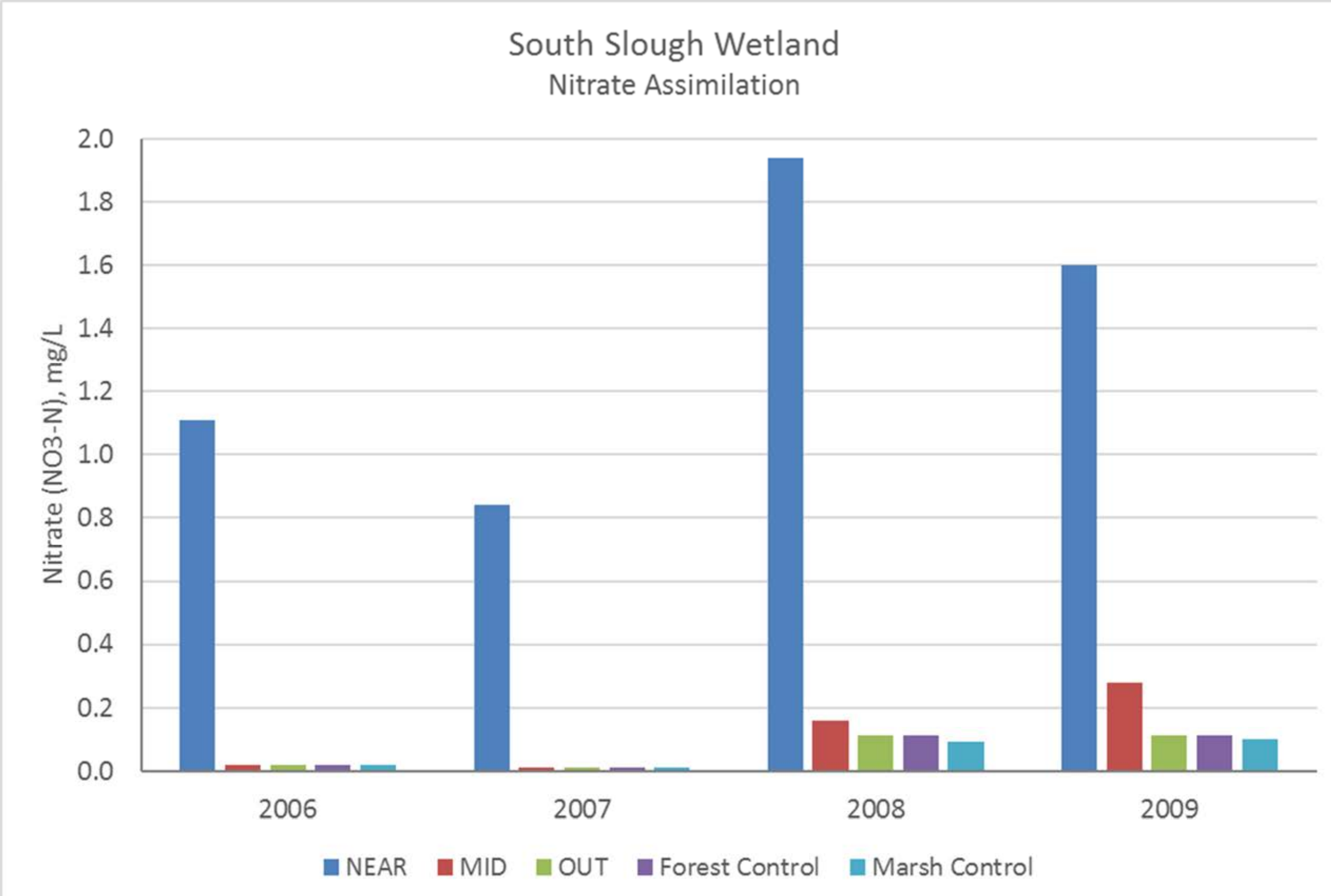


Figure 5.11 – Total Phosphorus (TP) assimilation in the South Slough Wetland

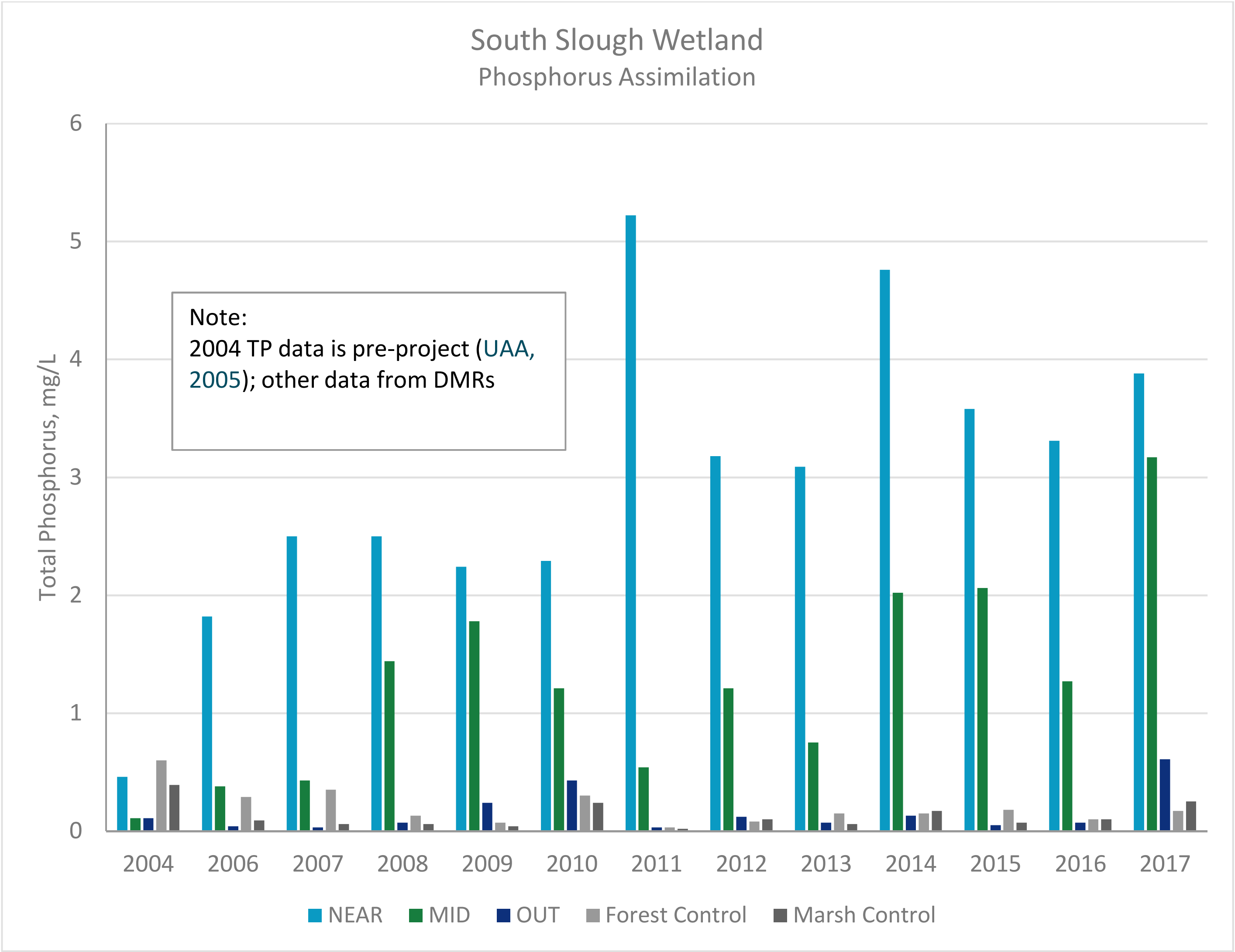


Figure 5.12 – Salinity concentrations in the South Slough Wetland

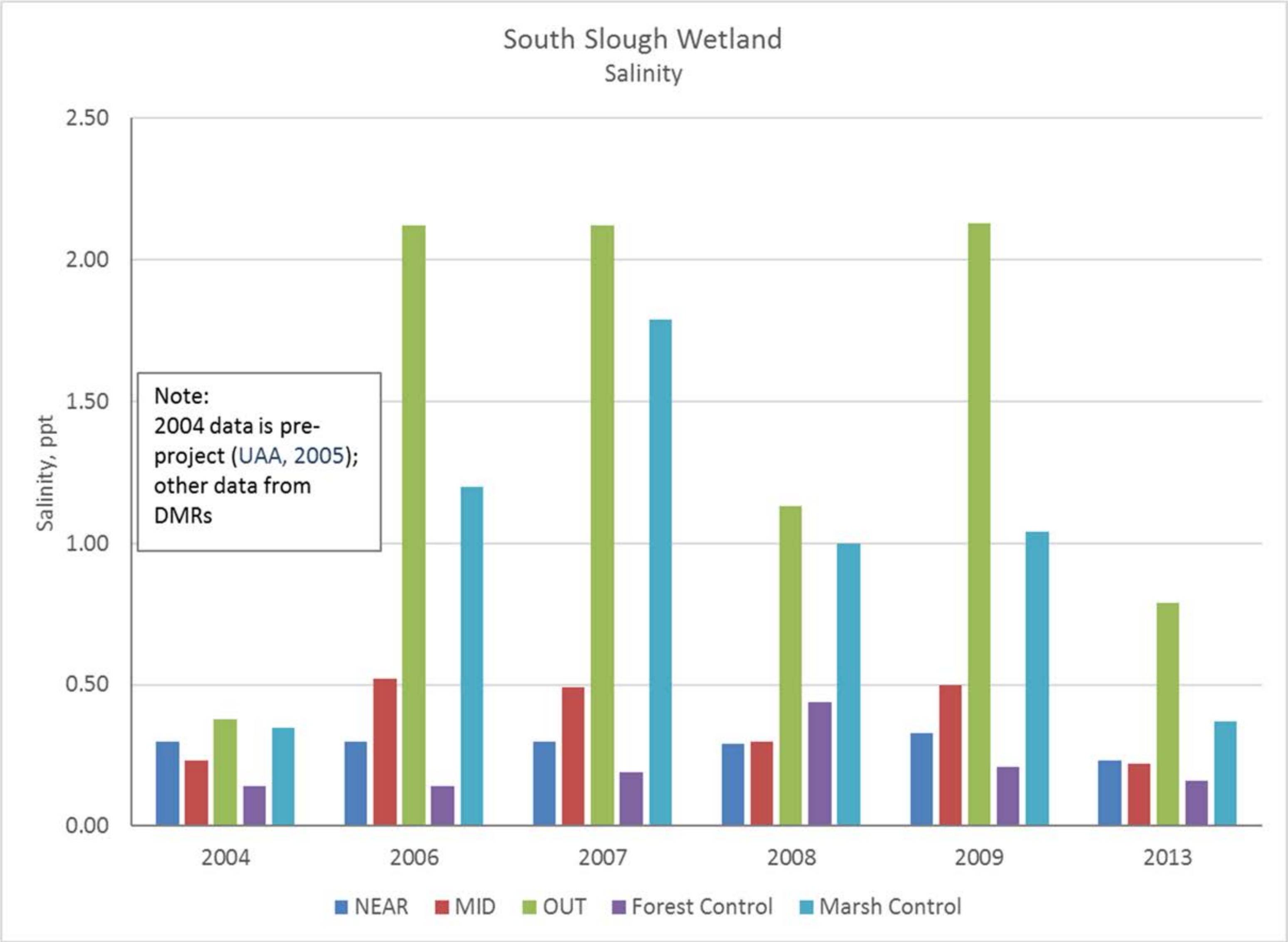


Figure 5.13 – End of Season Live Biomass (EOSL) production in the South Slough Wetland

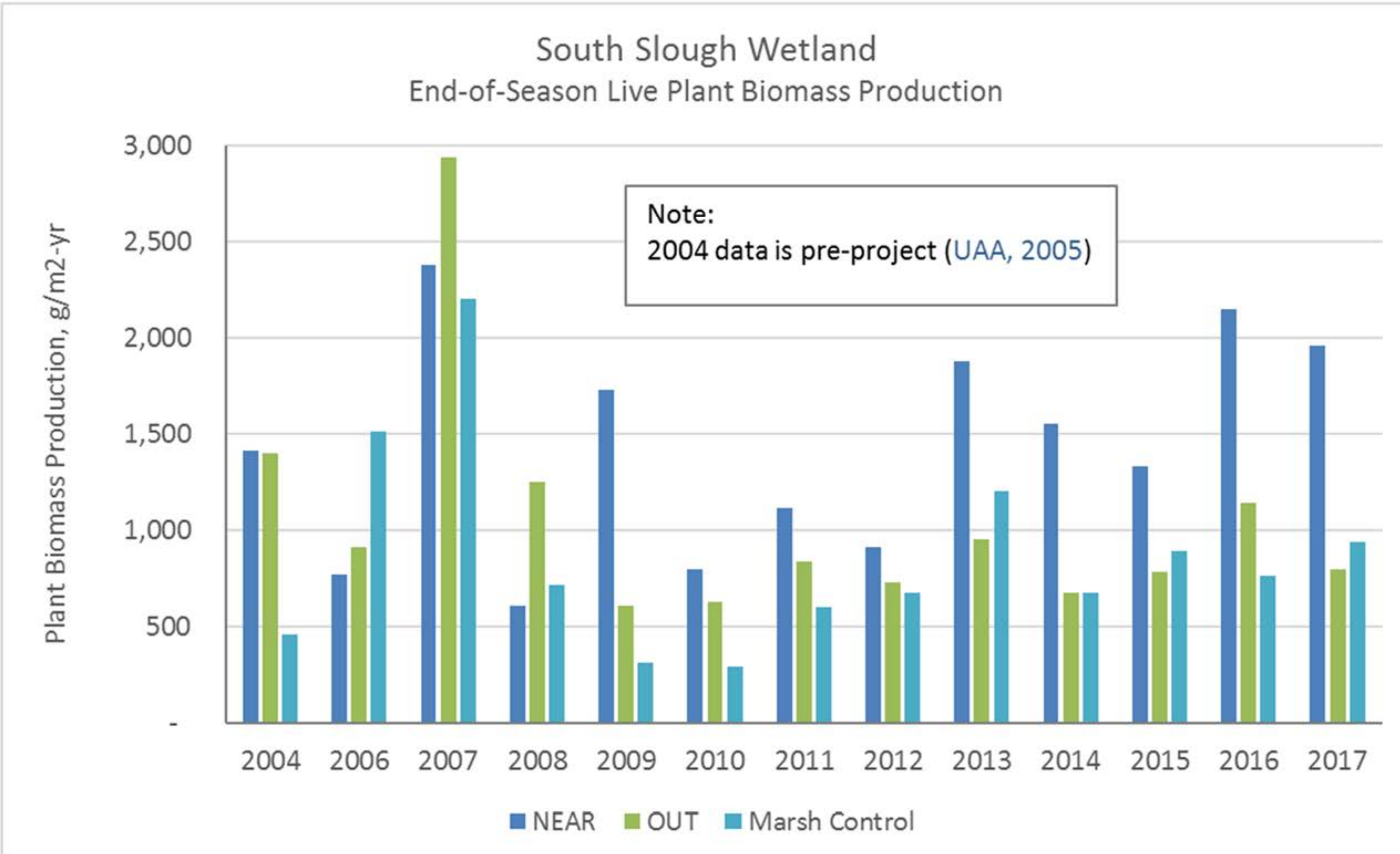


Figure 5.14 –**Litterfall plant biomass production in the South Slough Wetland**

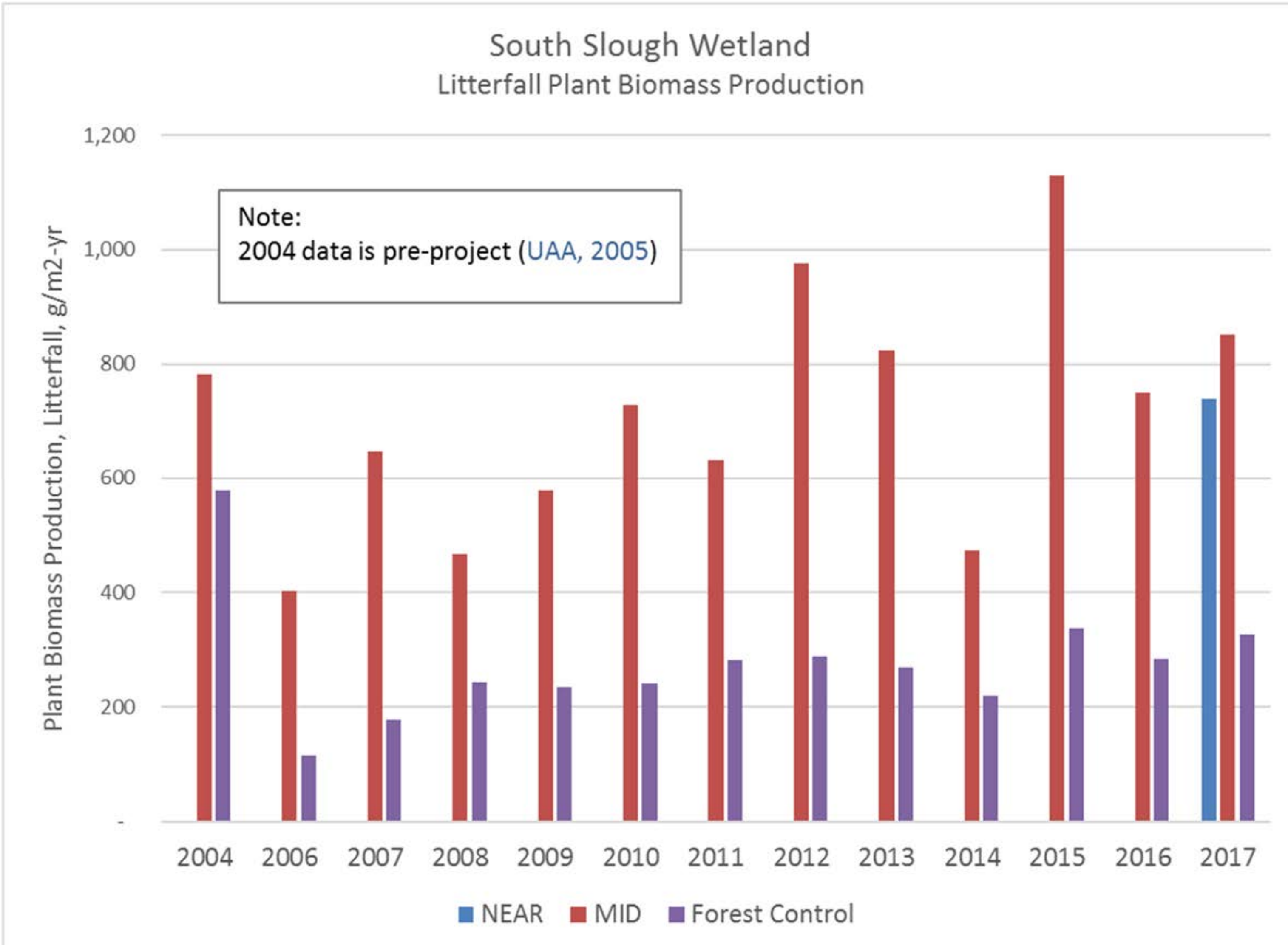


Figure 5.15 –Stem growth plant biomass production in the South Slough Wetland

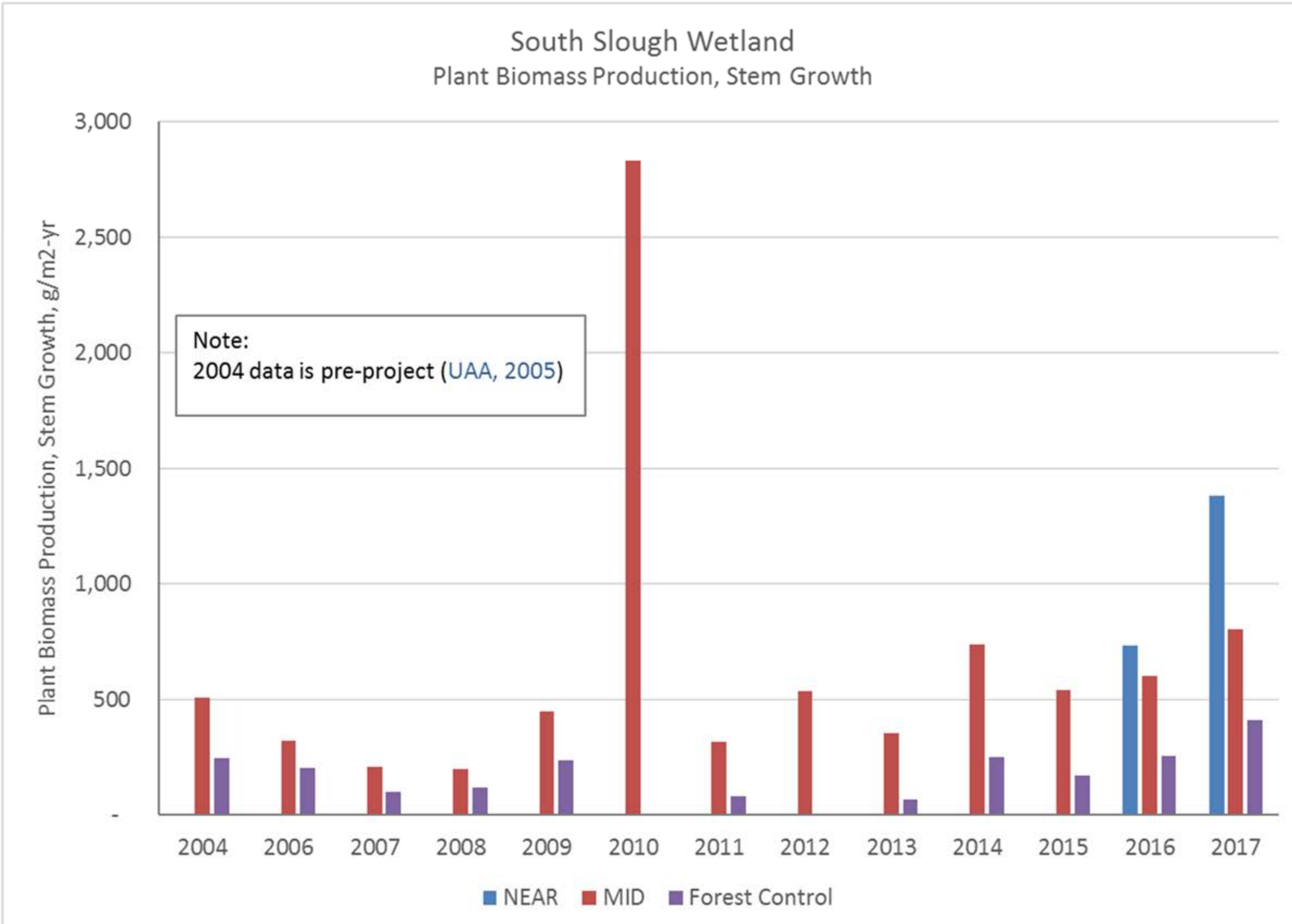


Table 5.1 – Summary of water quality parameters in the Hammond Assimilation Wetland, 2006 - 2017

	Average Concentration from 2006 - 2017				
Parameter	NEAR	MID	OUT	Forest Control	Marsh Control
Total Kjeldahl Nitrogen (TKN), mg/L	10.0	2.2	0.9	0.6	0.8
Ammonia Nitrogen (NH4-N), mg/L	6.2	0.8	0.2	0.1	0.1
Nitrate Nitrogen (NO3-N), mg/L (1)	1.4	0.1	0.1	0.1	0.1
Total Phosphorus (TP), mg/L	3.2	1.4	0.2	0.2	0.1
Salinity, PPT (2)	0.29	0.41	1.66	0.23	1.08

Note:

1. Nitrate nitrogen data from 2006, 2007, 2008 and 2008.
2. Salinity data from 2006, 2007, 2008, and 2013.

As seen in Figure 5.12, salinity concentrations are generally lowest at the NEAR and MID locations, presumably because the low-salinity wastewater effluent serves as a barrier against the intrusion of more saline waters from Lake Pontchartrain. This is consistent with the effluent spreading calculation summarized in Figure 5.7.

Protection of freshwater marshes and swamps by preventing saline intrusion is one of the stated goals of wetland wastewater assimilation projects (discussed further in Section 2). Based on the available data, the South Slough Wetland project is clearly successful in preventing elevated salinity levels within the effluent spreading zone.

End of Season Live Biomass (EOSL) is used to measure biomass production in herbaceous (marsh) wetlands. As seen in Figure 5.13, EOSL is generally higher in the NEAR monitoring location (9 out of 11 years) than the two background locations (OUT and Marsh Control). One of the stated regulatory goals of wetland wastewater assimilation projects is that biomass production will be enhanced (discussed further in Section 2). Enhanced biomass production is presumably linked to greater rates of soil accretion (Day *et al*. 2004). The South Slough Wetland project is clearly successful in enhancing EOSL.

Both litterfall (Figure 5.14) and stem growth (Figure 5.15) are used to measure biomass production in woody (forested) swamps. Originally, the only monitoring locations for woody biomass were the MID and OUT concentrations. Due to successful transplantation of cypress seedlings (Lundberg *et al.* 2011, Shaffer *et al*. 2015), monitoring at the NEAR site for stem growth started in 2016, and for litterfall in 2017. Based on the concentrations of ammonia, TKN and phosphorus (Figures 5.8, 5.9, 5.11) at the MID location, one would conclude that this site is still in the zone of active assimilation. This is consistent with the enhanced production of litterfall and stem growth biomass at the MID location relative to the background Forest Control locations (Forest Control location was moved in 2012).

One of the stated goals of wetland wastewater assimilation projects is that biomass production will be enhanced (discussed further in Section 2.0). Enhanced biomass production is presumably linked to greater rates of soil accretion (Day *et al*. 2004). The South Slough Wetland project is clearly successful in enhancing litterfall and stem growth biomass as a result of nutrient assimilation.

One measurement of the "fertilizer effect" of wastewater assimilation is the growth ratio of fertilized vs. unfertilized plants (further discussed in Section 8.0). Growth ratios for the different monitoring locations in the Hammond Assimilation Wetland are summarized in Table 5.2.

Table 5.2 – Growth ratio between monitoring locations in the Hammond Assimilation Wetland, 2007 - 2017

Parameter	Ratio	Value
EOSL	NEAR – OUT – Marsh Control	2.2 - 1.3 - 1.0
Litterfall (1)	NEAR – MID – Forest Control	2.3 - 2.8 - 1.0
Stem Growth (2)	NEAR – MID – Forest Control	3.1 - 2.8 - 1.0

Note:

1. Litterfall data for the NEAR site is only for 2017.
2. Stem growth data for the NEAR site is only for 2016 and 2017.

Results summarized in Table 5.2 are from 2007 (the first full growing season after effluent application began) through the available period of record in 2017. These growth ratios are very similar to those presented in Table 8.2, indicating that:

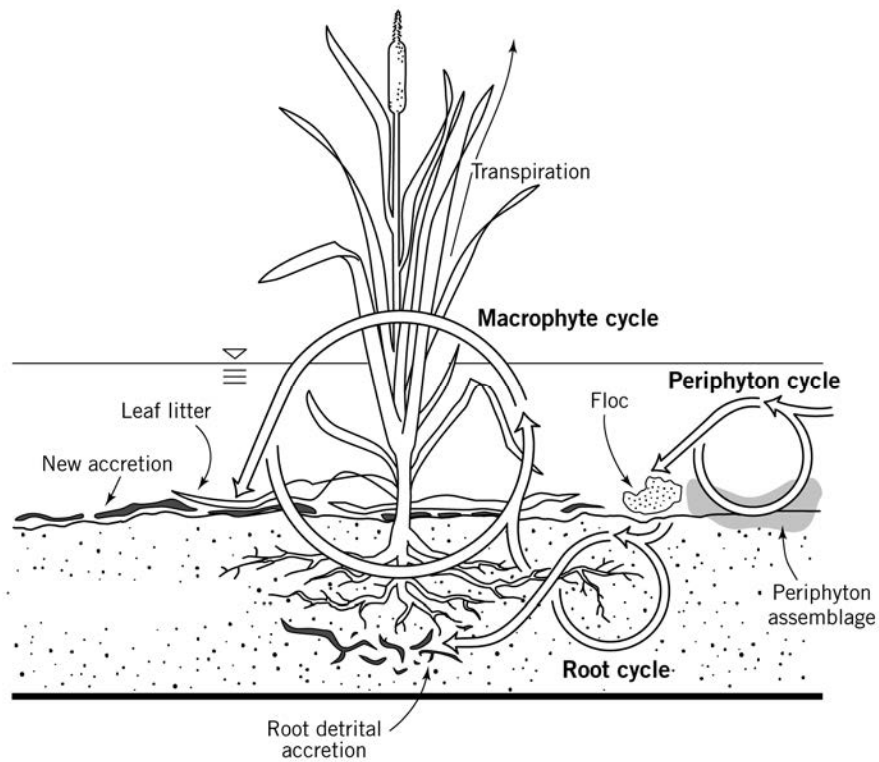
1. The increased rate of tree growth estimated from tree ring analysis at the MID site (Section 8.0) resulted in a growth ratio of 1.87. The further corroborates that the MID location experiences a fertilizer effect, which the litterfall and stem growth ratios of Table 5.2 also support.

2. Growth ratios in Table 5.2 are similar to other studies presented in Table 8.2. This suggests the fertilizer effect seen in the Hammond Assimilation Wetland is similar to other wetland assimilation projects studied in the scientific literature.

6.0 The Active Nutrient Assimilation Zone

While forms of oxidized nitrogen can rapidly be lost to the atmosphere via denitrification, the remaining nitrogen (N) and phosphorus (P) interact with the wetland ecosystem. Wetlands process and store nutrients through a variety of biogeochemical cycles (Reddy & DeLaune, 2008; Kadlec & Wallace, 2009), the most significant of which is the plant biomass cycle (Figure 6.1).

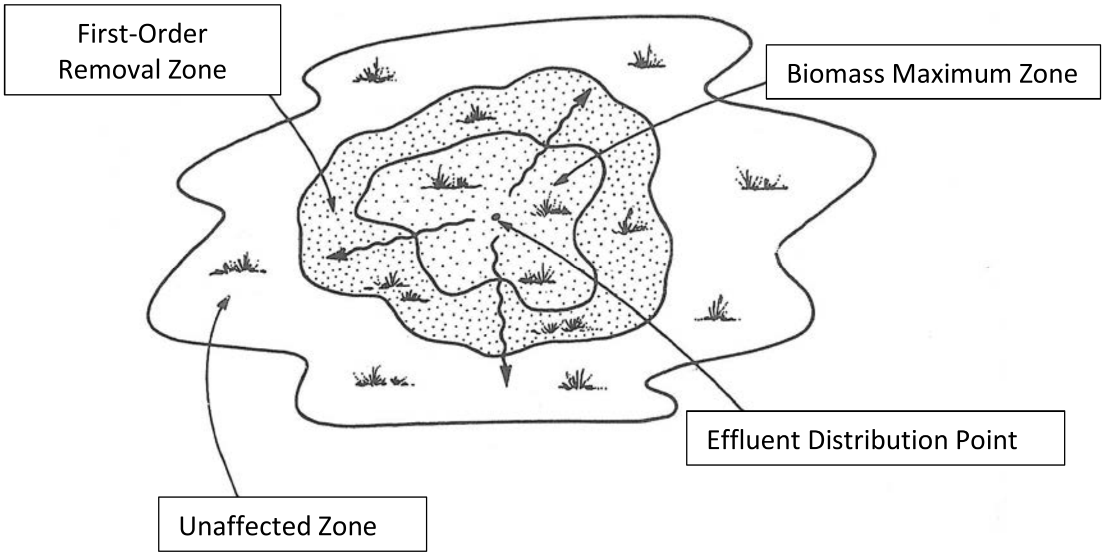
Figure 6.1 – Plant biomass cycling in wetlands (from Kadlec & Wallace, 2009)



There is a "fertilizer effect" in assimilation wetlands where the added nutrients (N and P) in the effluent stimulates both a greater production in plant biomass and a greater nutrient content in plant tissues (Kadlec 1985, Kadlec 1997, Kadlec 2009a, Kadlec & Bevis 2009, Lundberg *et al*. 2011, Ialeggio & Nyman 2014, Shaffer *et al*. 2015). However, there is an upper limit of how much biomass can be produced annually, even if there is an unlimited supply of N and P.

While N can be oxidized, denitrified (subject to oxygen availability) and lost to the atmosphere, phosphorus remains bound in plant biomass. Since roughly 20% of the plant biomass is not degradable (Kadlec & Wallace 2009) N and P in this biomass is stored in the wetland through the accretion of new organic sediments. Since there is an upper limit on the amount of biomass that can be produced, and hence an upper limit on N and P which can be stored in the system (even if there is an unlimited supply of nutrients), assimilation wetlands can be divided into three zones (Kadlec, 1985) as shown in Figure 6.2.

Figure 6.2 – Schematic of nutrient assimilation zones in wetlands (from Kadlec, 1985)



According to Kadlec (1985), these zones include:

1. Biomass Maximum Zone: In this "saturated zone", loadings of nutrients (typically phosphorus) exceed the maximum possible uptake/storage of the plant biomass cycle, even at maximum biomass production and maximum nutrient content in plant tissues. Removal of nutrients in this zone follows a zero-order removal rate (g/m^2-yr) where the mass of nutrient removed is a function of the maximum biomass cycle.

2. First-Order Removal Zone: In this "zone of rapid removal", available nutrients (typically phosphorus) are within the bounds of the plant biomass cycle. Plant growth demonstrates a fertilizer effect, as plants with access to more nutrients grow at a greater rate and contain more nutrients in their tissues. As a result, there is a gradient in plant growth reflecting the decreasing availability of nutrients as flows move away from the inlet distribution point. Removal of nutrients in this zone follows a first-order process (m/year), with decreasing removals at lower concentrations.

3. Unaffected Zone: In this zone, nutrients have been reduced to background concentrations, and no longer play a role in stimulating plant growth. Removals in this region are generally low and reflect the natural biomass cycle of the wetland without any additional nutrients.

Zones 1 and 2 make up the "active assimilation zone" where applied nutrients are utilized and stored within the wetland.

6.1 Long-Term Experience at Houghton Lake, Michigan

One of the most extensively studied assimilation wetlands in the United States was at Houghton Lake, Michigan, which operated for 30 years (1978 to 2008) under an in-depth and long-term monitoring program (Kadlec 2009a). While Houghton Lake was a northern assimilation project, it has important similarities to the Hammond Assimilation Wetland area, including:

1) Effluent discharge was to a fresh water marsh (sedge-willow) community.
2) The marsh had large internal storages of carbon (as peat) that had accumulated over time prior to commencement of the project.
3) An active assimilation zone developed over time, which occupied roughly 83 ha out of the total 700 ha wetland (Kadlec & Bevis, 2009), consistent with Figure 6.2.
4) In the active assimilation zone (Figure 6.3):
    a. There was a shift from the pre-existing marsh community (sedge-willow) to a *Typha* (cattail) dominated vegetative community.
    b. There was a fertilizer response, characterized by a much larger standing biomass (about 3X the non-fertilized areas of the marsh). Increased biomass was also accompanied by increases in plant tissue nitrogen (2X) and phosphorus (3X) content.
    c. In the areas closest to effluent distribution, the pre-existing fixed marsh substrate developed into a floating mat.
5) Effluent application was only during the growing season (May to October in Michigan). No effluent was applied during the dormant season. With effluent is applied year-round to the Hammond Assimilation Wetland, the project location in coastal Louisiana also has a year-round growing season.

Houghton Lake was a very successful nutrient assimilation project. The wetland stored 94% of the incoming phosphorus (P) and removed 95% incoming inorganic nitrogen over the 30-year period of record. Phosphorus was stored in new biomass (about 17% of stored P), increased soil sorption (about 3% of stored P) and accretion of new soils and sediments (about 80% of stored P) Kadlec (2009a).

There is considerable evidence that the Hammond Assimilation Wetland is undergoing structural changes within Four Mile Marsh. For instance, the data of Lundberg (2008) (Figure 6.4) for herbaceous marsh biomass production at Hammond appears to follow the same pattern as Figures 6.2 and 6.3

As seen in Figure 6.4, herbaceous biomass appears to operate near a maximum plateau level from the effluent distribution pipe out to a distance of 400 m. This is the region within the assimilation wetland where nutrient availability exceeds the biomass assimilation capacity of the wetland. As effluent flows further away, nutrients start to become a limiting resource, and plant biomass decreases as nutrients are less available.

Figure 6.3 – Assimilation zone occupying 83 ha within a 700-ha assimilation wetland at Houghton Lake, Michigan (from Kadlec, 2009b). The assimilation zone took approximately 9 years to develop, and then was stable for the remainder of the 30-year period of operation.

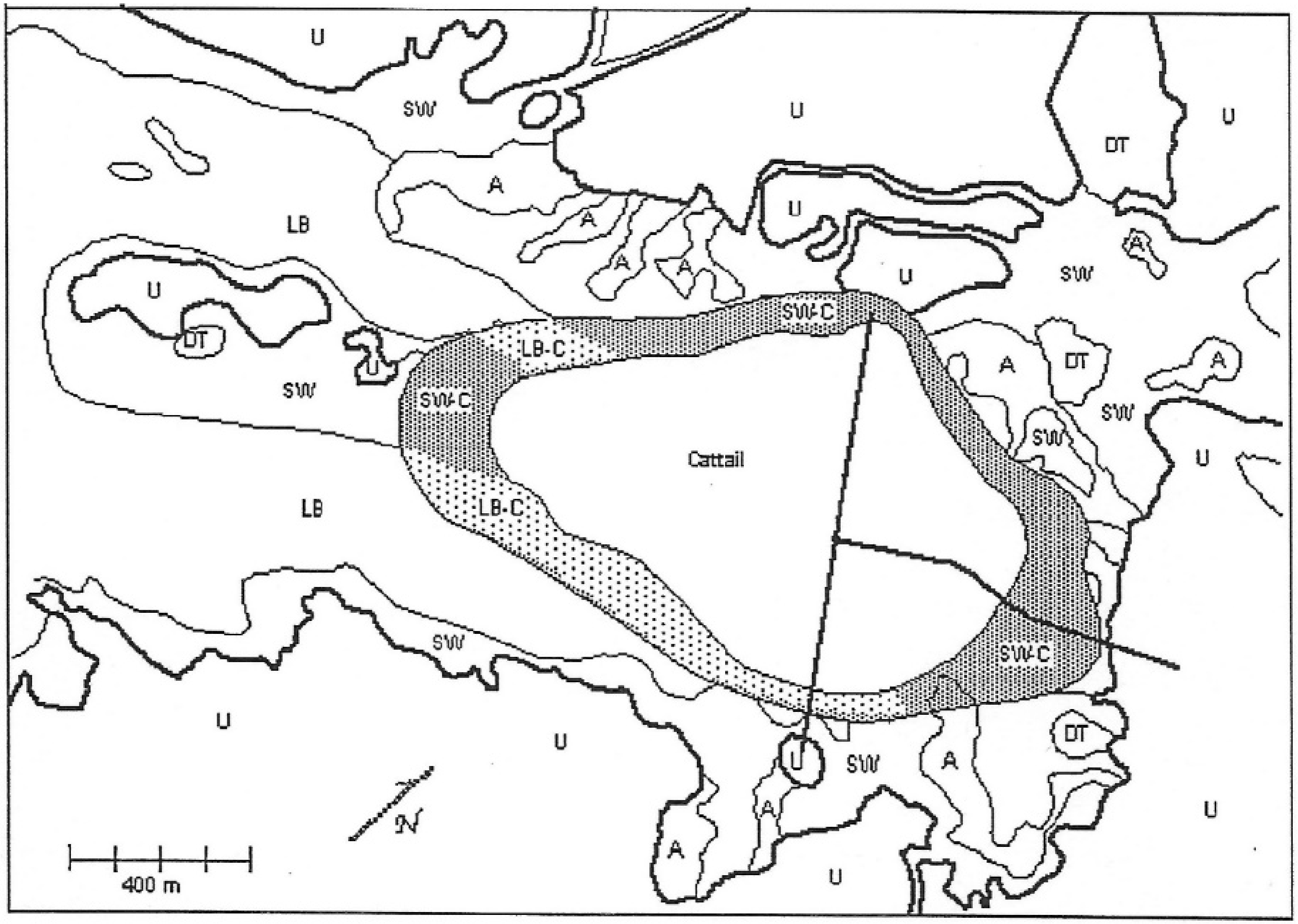
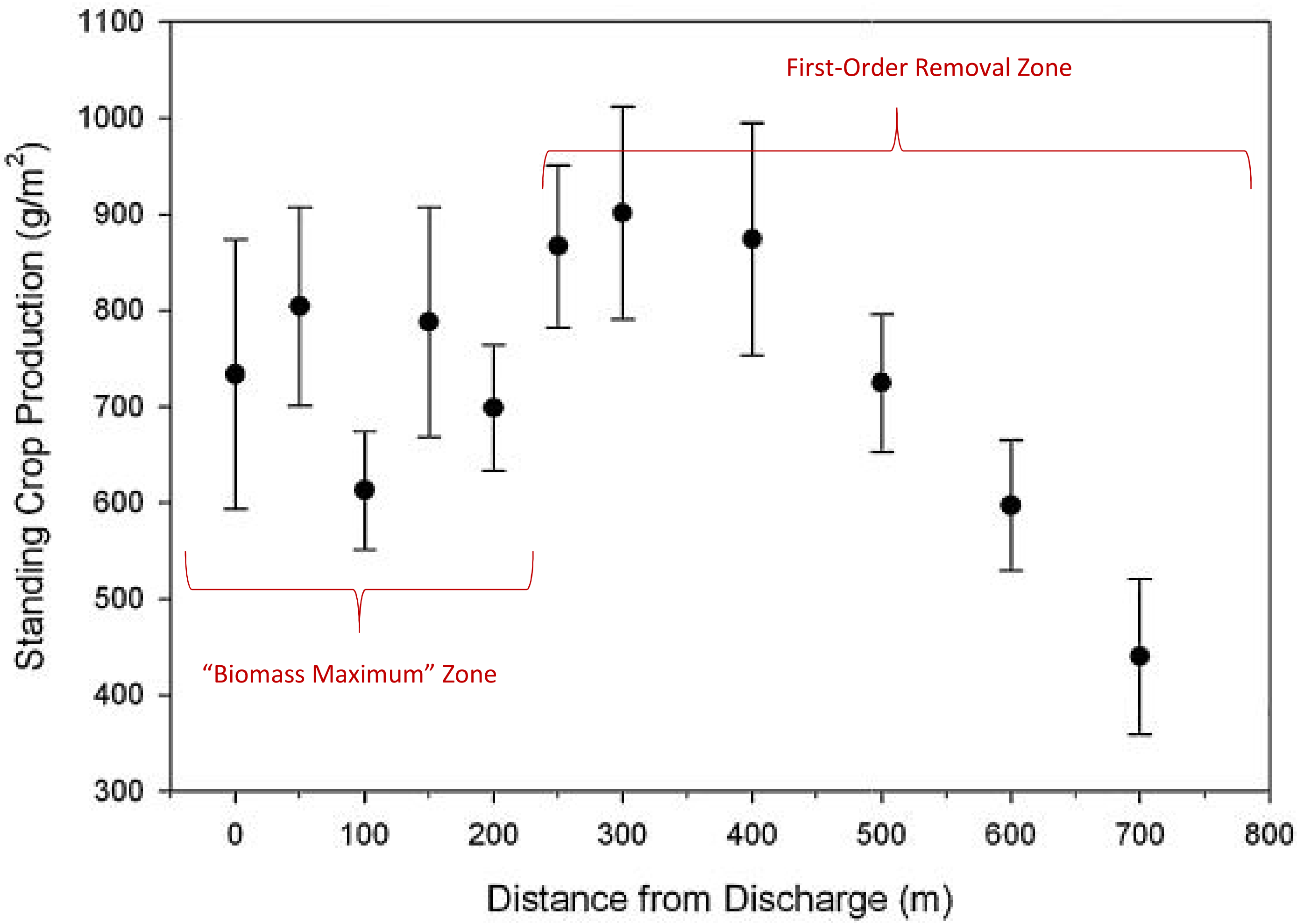


Figure 6.4 – Mean aboveground herbaceous biomass production, Hammond Assimilation Wetland in 2007 (prior to nutria "eat out") (Lundberg 2008; Shaffer *et al*. 2015).



6.1.1 Speed of Assimilation Zone Development at Houghton Lake, Michigan

Because the Houghton Lake project had the benefit of many years of comprehensive monitoring, transect data from multiple years was used to parameterize the removal of phosphorus, ammonia (NH_4-N) and nitrate. Phosphorus was the rate-limiting parameter and took the largest area to assimilate.

Ammonia (NH_4-N) was not loaded above biomass maximum uptake and all removals were first-order (Zone 2 in Figure 6.2). Nitrate (NO_3-N) is a highly sought after electron acceptor in reduced wetland environments (Reddy & DeLaune, 2008) and was almost completely removed within 100 m of the inlet distribution pipe (consistent with the Hammond Assimilation Wetland).

At the inlet, phosphorus loadings from the distribution pipe were greater than the ability of the plant biomass cycle to uptake, bury and store (accrete) the phosphorus. As a result, phosphorus removal followed a zero-order pattern of approximately 14.4 g/m^2-yr. As loadings were reduced further away from the inlet loading pipe, available phosphorus could be uptaken within the bounds of the plant biomass cycle, with a gradient in biomass productivity reflecting lower phosphorus availability (first-order removal) similar to that seen in Figure 6.4 for the Hammond Assimilation Wetland. At the edge of the assimilation zone, phosphorus was at background concentrations and no further vegetative changes were noted. Key parameters for the Houghton Lake system are summarized in Table 6.1.

Table 6.1 – Rate coefficients for the Houghton Lake, Michigan wetland assimilation system (Kadlec, 2009a)

Parameter	Rate Coefficient
Phosphorus (inlet region, zero-order)	14.4 ± 4.9 gP/m^2-year
Phosphorus (overall, first order)	19.2 ± 6 m/year
Ammonia (NH_4-N)	29.8 ± 8.7 m/year
Nitrate (NO_3-N)	Removed within 100 m of inlet pipe

The fertilizer response of wetland plant biomass to the added phosphorus was modeled using the "biomachine" method of Kadlec (1997) as:

$$\frac{N}{N_{max}} = \frac{(C - C')}{(C - C') + s}$$

Where: N = local biomass density (g/m^2)

N_{max} = maximum biomass density that can exist (g/m^2)

C = local P concentration (gP/m^3)

C' = lowest P concentration that supports growth (gP/m^3)

s = half-saturation P concentration for biomass (gP/m^3)

For the Houghton Lake system, Kadlec and Bevis (2009) selected C' = 0.02 mg/L and s = 0.2 mg/L as this provided the best results for annual data fitting between 1982 and 2008, when the wetland assimilation system was essentially running at steady-state. This implies that at very low phosphorus concentrations, the actual plant biomass (N) is only about 5% of the maximum biomass possible (N_{max}).

6.1.2 Rate of Plant Community Development at Houghton Lake, Michigan

Development of the new vegetative community in the Houghton Lake assimilation zone took approximately 9 years (with effluent application 6 months per year) to spread across 63% of the final assimilation area. The grow-in time for development of a new, larger biomass crop also took approximately 9 years. Once developed, the assimilation zone area was stable over the remaining 30-year period of record. The rate of expansion of the assimilation area was described mathematically as (Kadlec & Bevis, 2009):

$$A = A_{\max}\left(1 - e^{-t/b}\right)$$

Where: A_{max} = Area of maximum assimilation zone expansion (83 ha for Houghton Lake)

t = time since discharge began (years)

b = time constant (9.0 years for Houghton Lake)

The Houghton Lake site in Michigan experiences a growing season that is approximately 6 months long. Effluent application was also for 6 months of the year (corresponding to the growing season). Considering the year-round growing season in southern Louisiana and the year-round effluent application by the City of Hammond, it is expected that the development time constant of the assimilation zone would be roughly half the time (4.5 years) compared to Michigan.

6.2 Estimation of the Active Assimilation Zone in the Hammond Assimilation Wetland

Assuming that the rate coefficients developed at Houghton Lake (Table 6.1) can be applied (a reasonable assumption given that Kadlec & Wallace, 2009 found little difference in phosphorus rate coefficients between warm-climate and cold-climate surface flow wetlands), and the time constant is halved to reflect the year-round growing season in southern Louisiana (4.5 years vs. 9 years), this same modeling approach can be applied to the Hammond Assimilation Wetland.

Annual operating data from the City of Hammond (DMRs from 2006-2018) were utilized to estimate the area required for nitrogen (N) and phosphorus (P) assimilation based on the parameters summarized in Table 6.2.

Table 6.2 – Assimilation area modeling parameters for the Hammond Assimilation Wetland

Parameter	Nitrogen (1)	Phosphorus
Rate coefficient, m/yr	29.8	19.2 (4)
Flow, m^3/yr	From DMRs	From DMRs
C_{in} mg/L	From DMRs	From DMRs
C_{out}, mg/L (2)	1.0	0.02
C*, mg/L (3)	0.6	0.002

Notes:

1. Includes total nitrogen, although NO_3-N likely disappeared within 100 m of the effluent distribution pipe. (Rate coefficient for NO_3-N estimated at 138 m/yr; Kadlec, 2009a).
2. Outlet concentrations selected based on estimates of best possible advanced tertiary treatment technology capabilities.
3. Ecosystem background concentrations (C*) based on background site monitoring data from the City of Hammond (Comite Resources 2007-2017).
4. This is in the 73rd percentile of phosphorus rate coefficients summarized in Kadlec & Wallace, 2009.

The estimated area needed for nutrient assimilation was calculated for each year between 2006 (year that effluent assimilation started) to 2018 (last year data was available). 2006 calculations assume that effluent was applied to the wetland for the entire year, although effluent assimilation did not begin until late Fall. Modeling results are presented in Figure 6.5.

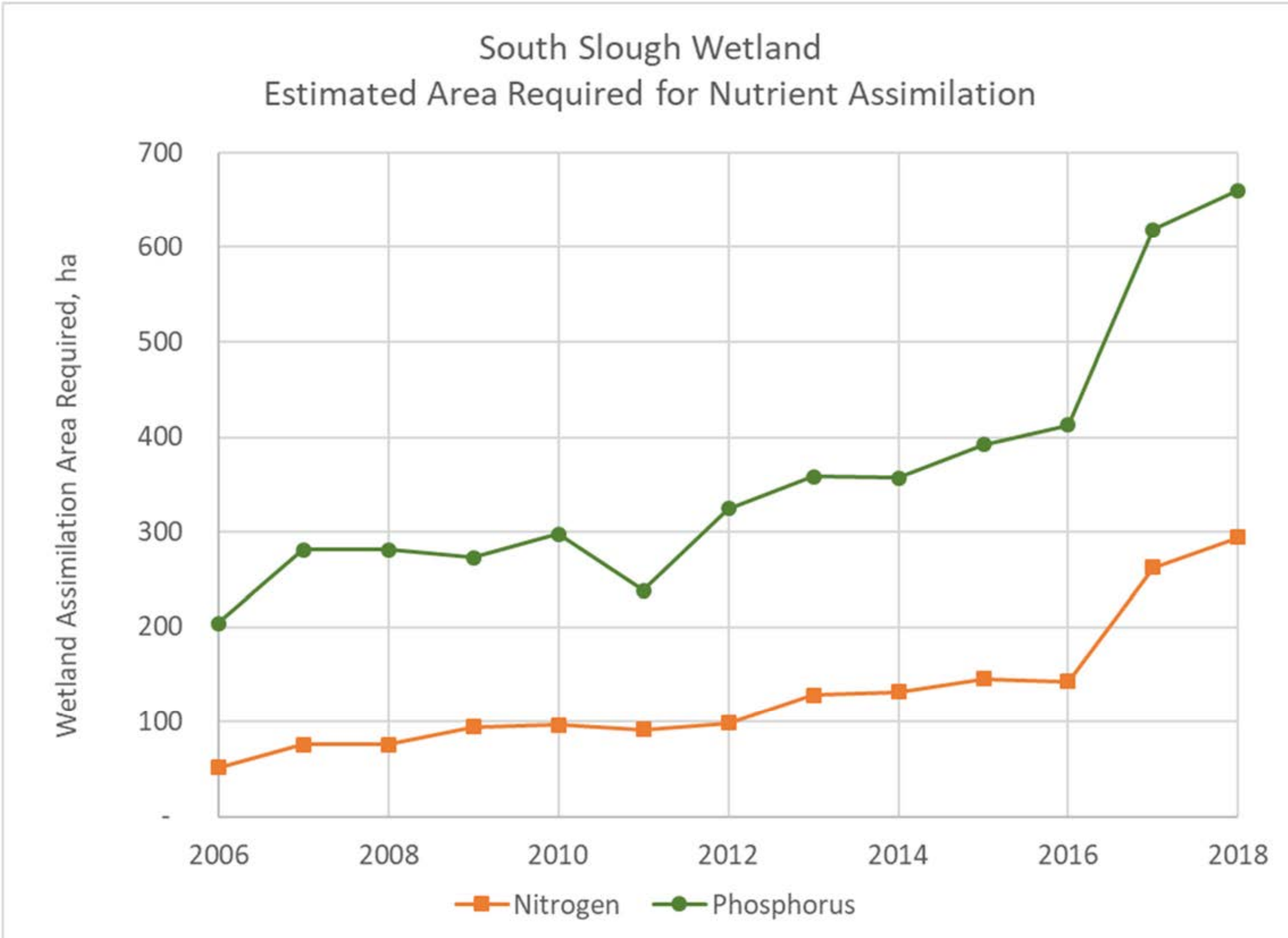
As seen in Figure 6.5, during the early years of the project (2007-2010), approximately 300 ha of wetland area would have been involved in the phosphorus assimilation zone. This is the region of the receiving wetland that would have displayed a "fertilizer effect" of increased plant biomass reflecting Zone 1 and Zone 2 behavior in the active assimilation zone (Figure 6.2).

Modeling the assimilation area in this way indicates that nutrient assimilation in the active assimilation zone (Zones 1 and 2 of Figure 6.2) is much greater than the "loading chart" concepts of Nichols (1983) and Richardson & Nichols (1985) would indicate. The drawbacks and limitations of the "loading chart" approach are described in more detail in Appendix C.

Dividing the annual mass loads by the area of active assimilation indicates that nitrogen loadings were 46-121 gN/m^2-yr. This is within the range of nitrogen loadings expected to be contained within the plant biomass cycle, which is generally less than 120 gN/m^2-yr (Kadlec & Wallace, 2009). Based on nitrogen, the plant biomass cycle would have been operating at 81-97% of maximum in the active assimilation zone.

The same approach applied to phosphorus indicates that phosphorus loadings were 4.0-7.4 gP/m^2-yr in the active assimilation zone. Based on phosphorus, the plant biomass cycle was running at roughly 99% of maximum in the active assimilation zone.

Figure 6.5 – Estimated active assimilation zones within the South Slough Wetland



6.3 Speed of Marsh Conversion in the Assimilation Zone at the Hammond Assimilation Wetland

The 300-ha area encompasses the area of marsh conversion described by Day *et al*. 2011, Bodker *et al*. 2015 and Shaffer *et al*. 2015 that occurred in 2008-2009. One theory of the marsh conversion is that it was due to the added nutrients. If this is the sole cause, then the marsh conversion is related to biomass cycling. The estimates of the impacted areas described by Turner (2017, 2019), Figure 4.1 were modeled to determine the necessary turnover rates in the plant biomass that would have had to occur to engender the conversion to open water and mudflats in the time period described.

The initial response of the herbaceous marsh vegetation within the Hammond Assimilation Wetland was increased biomass growth (Lundberg *et al*. 2011). This rate of increased growth is consistent with the "fertilizer effect" within assimilation wetlands (Kadlec 1997) as described in Figure 6.4. Beginning in late fall of 2007 (after about one year of effluent assimilation), there was a shift (within a few months) to open water and mudflats (Shaffer *et al*. 2015, Bodker *et al*. 2015) representing a collapse of the pre-existing plant community, followed by subsequent evolution of a replacement plant community (Weller & Bossart 2017). Several theories have been advanced regarding why the ecosystem shift occurred, but the major ones are that:

1. There was a major nutria "eat out" of the newly-fertilized marsh vegetation (Shaffer *et al*. 2015, Day *et al*. 2019), since the fertilized vegetation was very attractive to nutria (Ialeggio & Nyman, 2014) The pre-existing plant community (which may have had *Panicum* as a keystone) did not grow back. The initial transition period was open water and mudflats, which has been replaced over time by a more diverse community of annual and perennial marsh plants colonizing the relict *Panicum* mats (Weller & Bossart 2017).
2. Increases in nutrient loading from effluent application lead to increased rates of degradation of organic matter within the wetland soils (Bodker *et al.* 2015) coupled with decreasing belowground biomass (Darby & Turner, 2008a, 2008b; Turner, 2011). This resulted in instability of the wetland soils leading to collapse of the pre-existing marsh plant community (Turner *et al.* 2018).

The speed of the marsh conversion compared to other wetland assimilation projects is interesting. In the case of two Michigan wetland assimilation projects (Houghton Lake and Kinross), Kadlec & Bevis (2009) and Kadlec & Bevis (1990) were able to observe changes in the plant community within the assimilation zone. While both wetlands were actively grazed by muskrats (*Ondatra zibethicus*) this did not lead to "eat out" collapses of the pre-existing plant community and the observed shifts in vegetation gradually occurred over time in response to nutrient assimilation.

6.3.1 Biomass Cycling

The speed of open water development at Hammond was estimated by Turner (2017, 2019) as transitioning from 0 → 122 ha between 2006 and 2010. These data were modeled to determine what speed development response rate and phosphorus assimilation rates would be required to match these observations. The best-fit results were a phosphorus assimilation rate of ≈46 m/yr (as opposed to 19.6 m/yr), and a speed development response rate of ≈0.7 years (as opposed to 4.5 years). The implications of this are discussed below:

- A phosphorus assimilation rate of 46 m/yr falls in about the 86th percentile of *k*-rate coefficients determined by Kadlec & Wallace (2009) for 282 surface-flow wetlands. While not out of the realm of possibility, this is certainly in the upper range observed for treatment wetlands.
- However, a speed development rate of 0.7 years is extremely rapid. Kadlec (1997) describes the speed development rate as the inverse of the burial fraction of plant biomass (amount of annual plant biomass contributing to accretion). A speed development rate of 4.5 years results in a burial fraction of 22%, and a biomass cycling rate of ≈3 times per year (4 months) which falls within the observed range of annual biomass production and cycling in existing treatment wetlands (Kadlec & Wallace 2009) and the data of Anisfeld & Hill (2011) for salt marshes. A speed development rate of 0.7 years implies 143% burial, which would require the entire plant biomass of the marsh to cycle (turn over) approximately 20 times per year (0.6 months). This is extremely rapid. In a study of soil CO_2 respiration rates and soil carbon balance during the peak of the summer growing season (July and August), Wigand *et al*. 2009 reported mean turnover rates of 2.6 months for *Spartina patens* and 6.8 months for *Spartina altenifolia*. However, two of the twelve marshes studied by Wigand *et al.* 2009 had cycling rates close to 0.6 months. This implies that the active assimilation area would have to cycle at some of the fastest rates ever recorded in the literature and do so year-round (instead of just during the peak summer months) if nutrients were the sole cause of the vegetation changes.

For nutrients to be the sole cause of the community shift, the Hammond Assimilation Wetland would have to be able to respond to nutrient loadings in an extremely rapid manner unique amongst wetland ecosystems. This lends credence to the argument that other external factors were a contributing factor in the ecosystem change, as it appears the adaptation of the plant community to increased nutrient loads could simply not happen quickly enough to engender the changes described in Figure 4.2 and by Turner, 2019.

6.3.2 Nutria Herbivory

Holm *et al*. (2011) developed a model to describe the impact of nutria herbivory on marsh biomass, based on one average-sized nutria being able to consume 72.4 kg of dry organic matter per year (of which 64% would be returned to the marsh as excreta). Holm *et al*. (2011) state this is equivalent to the annual biomass produced by 24 m^2 of marsh. This works out to a biomass productivity of 3,000 g/m^2-yr, which is in line with productivity estimates of ≈2,400 g/m^2-yr at both Houghton Lake and Hammond (Comite Resources, 2007; Kadlec & Bevis, 2009).

The worst-case scenario is that none of the consumed vegetation grows back (it dies and is eventually replaced by other species) using the lower biomass estimate of the Hammond Assimilation Wetland This set of assumptions indicate an average-sized nutria could consume approximately 30 m^2 of marsh vegetation in a single growing season. However, if nutria are the "wasteful feeders" described in Holm *et al*. (2011) and destroy approximately 10 times more vegetation than they consume, the area of vegetation that could be wiped out by a nutria in a single growing season is approximately 300 m^2.

As discussed in Section 3, nutria in the marsh area was reportedly about 2,000 animals killed (Shaffer *et al*. 2015) in an impact area of approximately 122 ha (300 acres) (Turner *et al*. 2018). Assuming that 100% of the nutria were killed, the pre-cull density was approximately 16.4 animals per hectare, which is in the mid-range of nutria summarized by Holm *et al*. (2011). 16.4 animals per hectare is one nutria per 610 m^2 of marsh area. Using the information from Holm *et al*. (2011) and the assumptions listed above, each nutria could have impacted 300 m^2 (49%) of this 610 m^2 in a single growing season.

If the grazed vegetation did not grow back, this moving "eat out" of 60 ha per year would have spread, such that the entire 122 ha would have been consumed in approximately two years. This matches the data of Turner (2019), who estimated 2007 losses at 0 → 50 ha, 2008 losses at 50 → 108 ha, 2009 losses at 108 → 112 ha, and 2010 losses at 112 → 122 ha.

These calculations indicate that nutria herbivory provides a plausible explanation for the marsh conversion, but it is predicated on several important underlying assumptions:

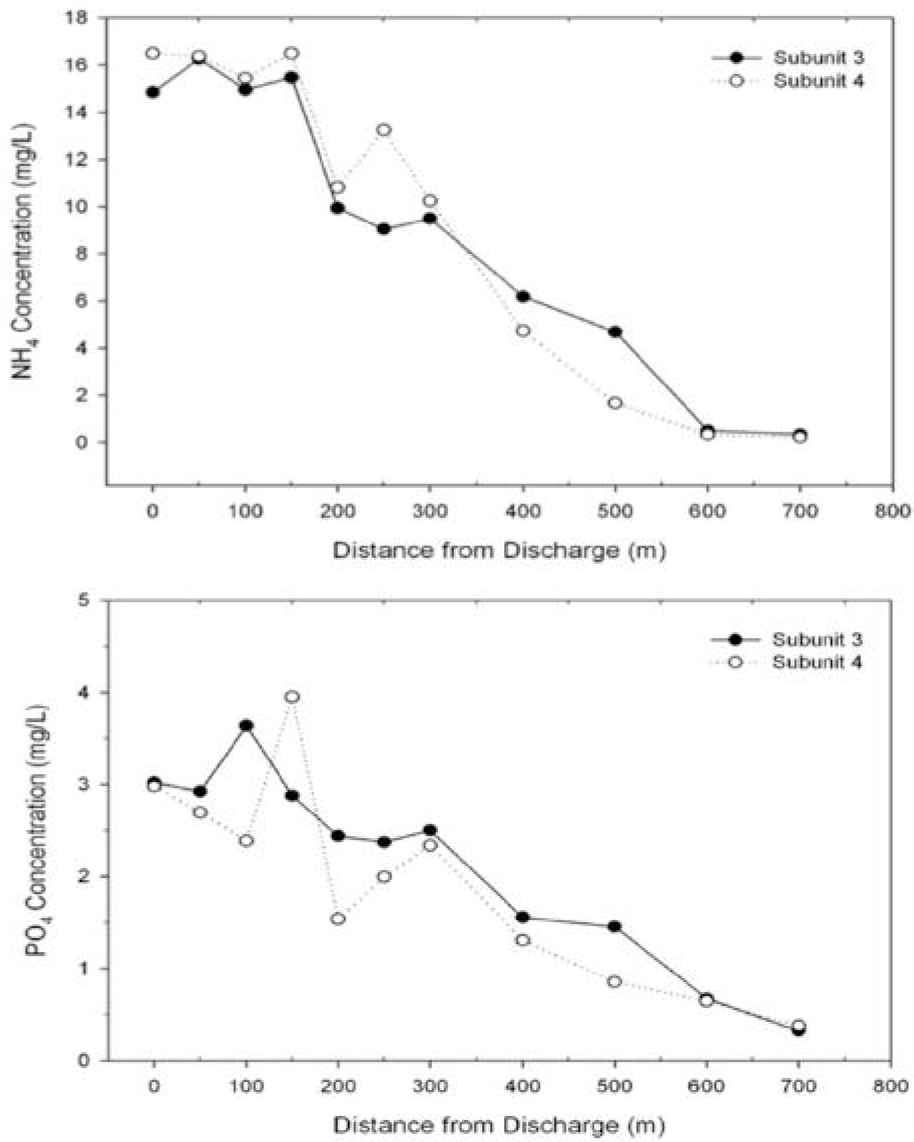
1. Nutria are "wasteful feeders" as described by Holm *et al*. (2011) and destroy 10 times more vegetation than they actually consume.
2. The marsh plant community was a relict system that developed under hydrologic conditions that no longer existed, even before the wastewater assimilation project (Section 3). The relict system was not all that stable and thus subject to switching to a different system-state as described by Sasser *et al*. (1996) in the face of a new set of ecosystem drivers. Consequently, vegetation grazed by nutria died and did not grow back.
3. The nutria had to come from somewhere. The role of nutrient-fertilized vegetation as a nutria attractant (Ialeggio & Nyman, 2014) had to occur in order to generate the nutria densities reported.

7.0 Biomass Production, Carbon Storages and Wetland Soils

Within the zone of active nutrient assimilation described in Section 6, a number of internal biogeochemical processes occur (Kadlec, 1997, Reddy & DeLaune, 2008, Kadlec & Wallace, 2009). To understand the actual process of nutrient assimilation, it is necessary to examine what occurs within the active assimilation zone.

Most wastewater assimilation wetlands in Louisiana have a total area much larger than the active assimilation zone (Hammond included) (see Appendix C for more details). Transect data near the effluent distribution pipeline is extremely valuable in this regard (Brantley *et al*. 2008, Lundberg, 2008) because it shows what occurs inside the zone of active assimilation (Figure 7.1).

Figure 7.1 – Removal of NH_4-N and PO_4-P in the Hammond Assimilation Wetland (Lundberg, 2008)



7.1 Denitrification and Organic Carbon

Oxidized nitrogen is rapidly removed in natural wetlands, and the role of organic carbon in this process requires attention as it plays a key role in net soil stability and soil production (accretion). Nitrate nitrogen (NO_3-N) is a highly sought-after electron acceptor in reduced environments such as natural wetlands (Mitsch & Gosselink 2007). The accumulation of organic-rich sediments (peat) in wetlands provides a built-in supply of organic carbon that can be used by bacteria to reduce NO_3-N to gaseous forms (N_2, N_2O) that are released to the atmosphere. The stoichiometric requirement for denitrification is approximately 1.25 mol of C per mol of N (Reddy & DeLaune 2008):

$$5(C_6H_{12}O_6) + 24(NO_3^-) + 24\ H^+ \rightarrow 30(CO_2) + 12(N_2) + 42(H_2O)$$

Wetland organic matter that is stored as peat is approximately 44-48% carbon (Sterner & Elser 2002). Thus, wetlands that have accumulated organic sediments over geologic timescales can have an extremely large supply of stored carbon relative to the applied loading of oxidized nitrogen from influent wastewaters. This is consistent with observations that forested wetlands in the Mississippi River delta have the capacity to uptake very large quantities of oxidized nitrogen (NO_3-N) (Lane *et al.* 2003), and the rapid and complete removal of wastewater effluent NO_3-N in the Thibodaux assimilation wetlands (Zhang *et al*. 2000).

7.1.1 Mining of Stored Carbon from Wetland Sediments

Short-term (6 week) gas-release studies have been performed by Bodker *et al*. 2015 to estimate the degradation rate of organic matter from different plant materials from the Hammond Assimilation Wetland site. As pointed out by Day *et al*. 2019, these rates of degradation were consistent with long-term (18 month) studies conducted by Shaffer *et al*. 2015. Stoichiometric calculations have been carried out on those results by Day *et al*. 2018b and Day *et al*. 2019 to demonstrate that the soil organic matter loss would be between 1.5% to 4.7% per year.

The range in these estimates reflect the variability in how readily biodegradable the plant materials studied in Bodker *et al*. 2015 were. The *Panicum* marsh mat was the least degradable, and the *Taxodium* needles were the most degradable. This is consistent with observations that the labile fraction of primary production (leaves, needles and non-woody litterfall) are the most biodegradable, contribute the most to denitrification, and thus do not contribute significant to sediment accumulation (Morris *et al*. 2014)

A range of denitrification potentials can be derived from the study of denitrification bioreactors that use wood chips and similar materials. Assuming that wetland peat material has a low bulk density (≈0.1 g/cm^3) and is 95% organic matter (Rezanezhad *et al*. 2016), and that organic matter is ≈48% carbon (Sterner & Elser 2002), each 1 m^3 of wetland sediment contains ≈45,600 g of stored carbon (not factoring in the annual carbon input from the plant biomass cycle).

The rate of denitrification is controlled by the degradation rate of the peat which releases the bioavailable carbon (rather than the rate of the denitrification reaction). The release rate of bioavailable organic carbon is dependent on the water temperature and the type of organic material being decomposed. Since it is difficult to measure the release rate of carbon before it is consumed in denitrification, the removal rate of NO_3-N is usually used as a direct measure. For a peat-based constructed wetland, Kleimeier *et al*. 2018 measured this at approximately 2.4 g/d of NO_3-N per m^3 of peat material at water temperatures consistent with the Hammond project. (This nitrate removal rate is somewhat lower than most wood chip bioreactors (Addy *et al*. 2016), indicating that organic material that accumulates in wetlands as peat is relatively refractory compared to wood chips.

This set of assumptions indicate that the wetland sediments would lose ≈2.6 g/d of C per m^3 of peat material until nitrate (NO_3-N) is exhausted. Assuming that the most biologically active zone of the wetland soil/sediment is in the top 1 m, this equates to ≈2.6 g/m^2-d of carbon loss. In a 12-year study of a South Carolina *Spartina alternifolia* marsh, Morris & Bradley (1998) measured the additional soil carbon loss from nutrient addition at ≈795 g/m2-yr, or ≈2.2 g/m^2-d. This close correlation to ≈2.6 g/m^2-d indicates that the assumptions presented herein for carbon mining appear reasonable.

This rate of carbon loss is roughly 0.0056% per day (≈2% per year) of the stored carbon. This is in rough agreement with the estimates of Day *et al*. 2018b (1.5% to 4.7% per year) based on the most refractory materials studied (the *Panicum* marsh mat) studied by Bodker *et al.* 2015.

In other words, if all the stored peat could be broken down into bioavailable carbon (an unlikely event), no new carbon was added from the annual plant biomass cycle, and no carbon from the effluent BOD was utilized, it would take approximately ≈50 years for all of the peat in the top 1 m to be consumed via denitrification, even with an unlimited nitrate supply. This makes it unlikely that the rapid and widespread vegetation changes reported in the marsh community (further discussed in Section 6) were the result of soil decomposition as hypothesized by Bodker *et al*. 2015 and Turner *et al.* 2018.

7.1.2 Closing the Carbon Balance: Carbon Sourced from the Annual Plant Biomass Cycle

There is a "fertilizer effect" in assimilation wetlands where the added nutrients (N and P) in the effluent stimulates both a greater production in plant biomass and a greater nutrient content in plant tissues (Kadlec 1985, Kadlec 1997, Kadlec 2009a, Kadlec & Bevis 2009, Lundberg *et al*. 2011, Ialeggio & Nyman 2014, Shaffer *et al*. 2015). However, there is an upper limit of how much biomass can be produced annually, even if there is an unlimited supply of N and P. Therefore, there is an upper bound to how much organic carbon can be generated annually to fuel denitrification.

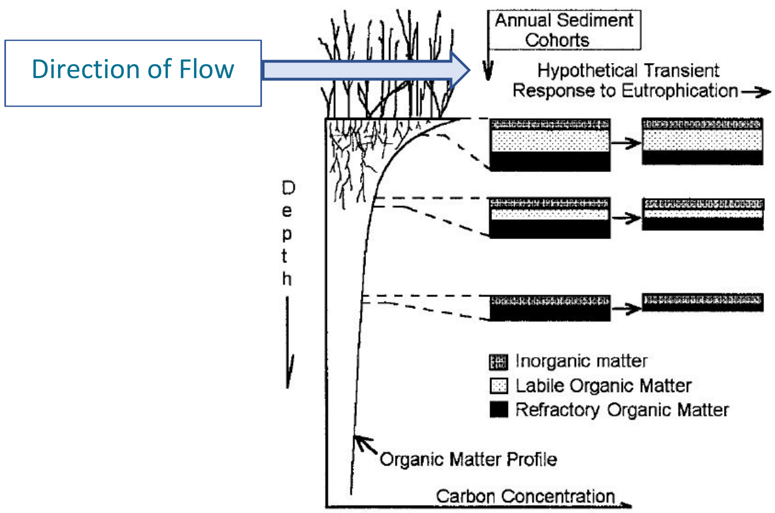
In the case of Hammond, herbaceous aboveground plant productivity was approximately 2,379 g/m^2-yr (dry weight) within the zone of maximum biomass production during 2007 prior to the "eat out" event by nutria (Comite Resources, 2007). This is a reasonable estimate of maximum productivity for a marsh system and agrees closely with the Houghton Lake, Michigan results of Kadlec & Bevis 2009 (≈2,400 g/m^2-yr). Assuming this plant biomass material was ≈48% carbon (Sterner & Elser, 2002), this would represent fixation of roughly 1,140 g/m^2-yr (3.1 g/m^2-d) of C.

This addition of new carbon every year from the plant biomass cycle (3.1 g/m^2-d) is enough to approximately balance out the amount of carbon "mined" from the stored peat by denitrification (≈2.6 g/m^2-d). This indicates that a heavily-fertilized marsh running under biomass maximum conditions can offset carbon losses to denitrification induced by effluent assimilation and maintain a stable carbon balance. In a 12-year study of a South Carolina *Spartina alternifolia* marsh, Morris & Bradley (1998) measured the additional soil carbon loss from nutrient addition at ≈795 g/m^2-yr, or ≈2.2 g/m^2-d. However, the net carbon loss estimated by Morris & Bradley (1998) was only ≈40 g/m^2-yr (≈0.11 g/m^2-d). The almost-net closure of the carbon balance was attributed by the authors to the role of the plant biomass cycle in the overall carbon cycle.

7.2 Spatial Distribution of Organic Matter within the Wetland

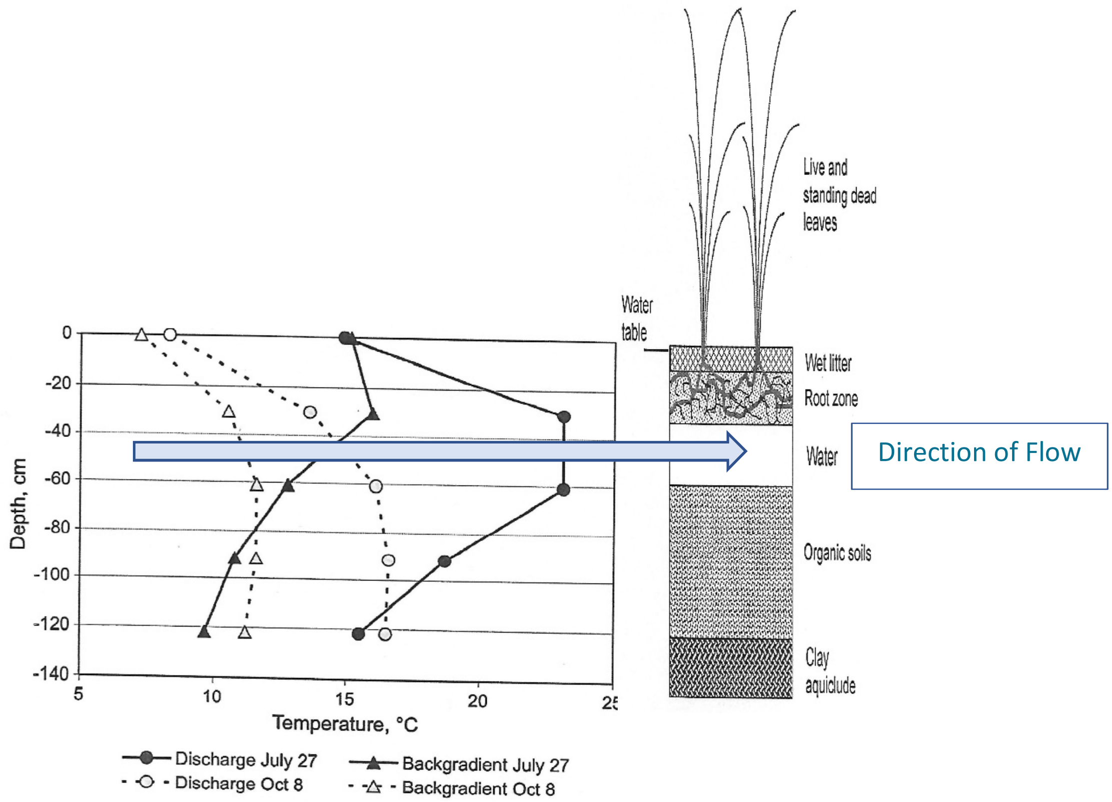
The spatial distribution of organic carbon in marsh systems is usually thought of as a "layer cake" phenomena where each annual cycle of plant biomass adds a new layer of material (mineral sediments, readily-degradable organic matter, and refractory organic matter) to the top of a vertical stack (Kadlec & Wallace, 2009). These annual deposits undergo changes as organic matter decomposes and are buried under subsequent annual layers. Morris & Bowden (1986) extended this concept to include the effect of nutrient availability on the storage of organic carbon (Figure 7.2).

Figure 7.2 – "Layer cake" model of Morris & Bowden 1986, where nutrient addition results in more biomass production, but potentially a reduction in refractory carbon storage (Morris & Bradley 1998)



In marsh systems that transition to floating mats, this is complicated by the fact that the preferential flow path is under the floating mat, not over it. Kadlec (2009b) demonstrated preferential flow underneath the floating mat at Houghton Lake due to the temperature differential between the effluent and the wetland water temperatures (Figure 7.3).

Figure 7.3 – Water temperature profiles during 2001 at Houghton Lake, Michigan (Kadlec, 2009b). Backgradient readings are wetland water temperatures upstream of the assimilation area. Effluent is warmer than the wetland waters. Floating mat profile is from Kadlec & Bevis (2009), stretched to match the vertical scale of the water temperature readings.



The system response of Morris & Bowden (1986), where the wetland both produces and consumes more biomass in response to nutrient loading agrees with the "biomachine concept" of Kadlec (1985, 1997). However, with under-mat flows as documented at Houghton Lake (Kadlec 2009b), it is possible that while "new" carbon is added via increased aboveground biomass litterfall, "old" carbon from the under-mat region is being utilized for nutrient assimilation. This would be a reinforcing mechanism for floating mat development from formerly attached peat wetlands. This has been documented for wastewater assimilation wetlands at Houghton Lake and Kinross, Michigan (Kadlec & Bevis 1990, 2009). This is also consistent with the observations of Turner et al. 2018 for the freshwater marsh areas within the Hammond assimilation wetland.

However, the studies of Sasser *et al*. 1996 and Visser *et al*. 1999 document the widespread conversion of non-wastewater *Panicum* marshes in Louisiana to floating-mat systems, so it appears that organic carbon conversions due to effluent-derived nutrients cannot be the sole cause of marsh conversion. The development of floating mats in assimilation wetlands thus bears more examination.

7.2.1 Development of Floating Mats

Floating mats must be almost entirely organic to be buoyant enough to float. Their buoyancy is a result of gas spaces in plant rhizomes and also gases generated by decomposition products (Hogg & Wein, 1988, Krusi & Wein, 1988). There are several natural mat formation mechanisms (Kadlec & Bevis, 2009):

1. The delamination and floating of unvegetated organic substrates from deeper sediment; germination of plants occurs after emergence. This is a peat "float-up" process. Since the Hammond system was fully vegetated prior to the assimilation project, it was likely not operative within the assimilation zone.
2. The rhizomes of aquatic plants colonize the water surface from a nucleus of aquatic vegetation that is either unattached or expanding from the shore. This is a "grow-over" process and appears to be the means by which the floating mat was established at Thibodaux, Louisiana (Izdepski *et al*. 2009). This is a possible means of floating mat expansion at Hammond.
3. Units of rooted vegetation and substrate split simultaneously from the bed and float to the surface (Saucier, 1963). This is a "mat floating" process and can be caused by major storm events. Photos presented by Turner *et al*. 2018 demonstrate "mat floating" at Hammond in the wake of Hurricane Isaac in 2012.
4. Upward root retreat, with accompanying detachment from underlying soils. This mechanism has been advocated by Turner *et al*. 2018 based on shear vane measurements at Hammond. The causes of upward root retreat are unclear. One scientific position is that this is caused by reductions in belowground biomass as a result of increased nutrient availability (Darby & Turner, 2008a, 2008b, Turner 2011) but there are numerous studies that show that nutrients result in increased belowground biomass (Valiela *et al*. 1976; Anisfeld & Hill, 2011; Hillmann, 2011).
    a. This is complicated by the fact that nutrient additions often cause major shifts in the plant community, where deeper-rooted plant species can be replaced by more prolific (but shallower-rooted) plants. Kadlec (2009b) noted that the sedge community at Houghton Lake, Michigan had a rooting depth of approximately 50 cm, but this was supplanted by *Typha*, which only had a rooting depth of ≈25 cm. Portions of the *Typha*-colonized zone later went on to become a floating mat in the areas closest to the inlet distribution pipe. The wetland assimilation system at Kinross, Michigan underwent a similar transformation from a pre-existing sedge peatland to a floating *Typha* mat (Kadlec & Bevis 1990).
5. Dissolution or destabilization of the rooting soils by chemical processes. (Peat soils are acidic and can lose structure upon exposure to alkaline waters).

Because the effluent addition can both enhance plant biomass production and consume soil organic carbon simultaneously, this has the potential to affect soil strength.

7.2.2 Impact of Nutrient Availability on Wetland Soil Strength

Organic-rich marsh soils generally lose strength (as determined by shear vane measurements) as a function of depth (Hollis & Turner, 2018). Part of this trend is due to the presence of live roots and rhizomes, which predominantly grow in the upper portion of the soil profile (Valiela *et al*. 1976). Live root tissues are stronger than dead ones, and in the "growth zone" of the upper soil profile, root structures are replenished and replaced.

In the absence of ongoing inputs of mineral sediments, the soil matrix below the "growth zone" is mainly comprised of dead plant materials, both fibrous (peat) and particulate (muck). This accumulation of organic matter is generally "refractory", meaning that the easily-decomposable (labile) organic materials have been removed over time. This decomposition process was historically in equilibrium with the redox conditions, nutrient availability, and matter inputs (plant biomass and mineral sediments) occurring over the lifetime of the marsh development (Reddy & DeLaune, 2008). Sudden shifts in nutrient availability, biomass cycling, redox potential and other factors can alter this equilibrium, affecting rates of production and decomposition of organic materials in the soil profile.

In a 5-year study using *Spartina alternifolia*, a salt marsh species, Turner, 2011 observed a loss of soil strength at depths > 80 cm in the soil profile as a result of nutrient addition (N, P and N+P). Changes in soil strength in the upper soil profile (including the upper 30 cm zone of active root/rhizome formation) was not affected by nutrient addition.

In a study using *Spartina patens*, a brackish marsh species, Hollis & Turner (2019) noted a reduction in the tensile strength of small live roots (0.5-1.0 mm) after 212 days but not after 60 days as a result of nutrient addition (N, P and N+P).

Cotton strings and canvas strips have been used as substitutes for dead plant matter. Turner, 2011 observed a loss of tensile canvas strips as a result of N+P addition. In a study based on the Hammond assimilation site, Bodker *et al*. 2015, observed the highest loss of tensile strength in cotton string was observed closest to the effluent distribution pipe, which was presumed to have the highest nutrient availability.

Turner *et al*. 2018 have claimed that there is a reduction of soil strength at Hammond in the effluent assimilation zone associated with the development of open water and floating mats. Soil strength is dependent on multiple factors and the relationship to nutrient availability is not clear-cut in the scientific literature.

Loss of soil strength is believed to be due to the combined effects of:

1. Changes in the production/morphology of belowground biomass (less roots and potentially more rhizomes) as a result of increased nutrients.
2. Degradation of newly-produced litterfall and existing soil organic matter as a result of increased nutrients (N, P) and when these materials are used as a carbon source for denitrification of oxidized forms of nitrogen (NO_3-N).
3. Gas uplift as a result of increased belowground gas formation resulting from enhanced denitrification and organic matter degradation caused by nutrient addition.
4. Increasing aboveground biomass much more than belowground biomass, this making the system "top-heavy" and susceptible to storm damage, delaminating the belowground substrate.

Changes in Belowground Biomass Production/Morphology

Darby & Turner (2008a) conducted an experiment on the biomass production of a Louisiana salt marsh grass, Spartina alternifolia in response to N and P additions. Where there was increased aboveground biomass production due to N and N+P additions, they reported that belowground biomass was reduced with P addition, mainly due to less rhizome production in the top 10 cm of the wetland sediment. Similar conclusions of reduced belowground biomass as a function of nitrogen inputs were reported by Darby & Turner (2008b) and Wigand *et al*. 2009.

However, other studies have not demonstrated a reduction in belowground biomass as a result of nutrient addition. Valiela *et al*. (1976) studied nutrient additions (N + P) to *Spartina alternifolia* and *Spartina patens* in a Massachusetts salt marsh (Table 7.1). While root production decreased in the study depth (0-20 cm), rhizome production did not. In a similar study of *Spartina alternifolia*, Anisfeld & Hill (2011) found no change in belowground biomass as a result of nutrient additions (N, P and N+P).

Table 7.1 – Estimated annual live biomass production for a fertilized salt marsh, adapted from Valiela *et al*. (1976)

	Control	Low Fertilization	High Fertilization (1)
N Loading, g/m^2-yr	---	43.7	131.0
P Loading, g/m^2-yr	---	11.4	34.1
Spartina alternifolia			
Aboveground	424	956	1,321
Belowground	3,498	5,637	3,315
Root:Shoot ratio (R:S)	**8.3**	**5.9**	**2.5**
Spartina patens			
Aboveground	632	1,379	1,256
Belowground	2,520	3,612	3,540
Root:Shoot ratio (R:S)	**4.0**	**2.6**	**2.8**

Note:

1. Nitrogen loading rates < 120 g/m^2-yr are generally taken up in the plant biomass cycle (Kadlec & Wallace 2009) due to the “fertilizer effect” of nutrient application and increased biomass production (Kadlec 1985, 1997).

A multi-year study done on a *Spartina alternifolia* salt marsh in Connecticut also did not demonstrate a reduction in belowground biomass as result of nutrient addition (Anisfeld & Hill 2011), as summarized in Table 7.2.

Table 7.2 – Effect of nutrient application to *Spartina alternifolia* biomass. Adapted from Anisfeld & Hill (2011).

	Control	N	N + P	P
2007				
N Loading Rate, g/m^2-yr		210	210	
P Loading Rate, g/m^2-yr			93	93
Net Aboveground Primary Productivity, gC/m^2-yr	222	344	397	239
Net Belowground Primary Productivity, gC/m^2-yr	670	537	660	540
Root:Shoot ratio (R:S)	**3.0**	**1.6**	**1.7**	**2.3**
2009				
N Loading Rate, g/m^2-yr		105	105	
P Loading Rate, g/m^2-yr			46.5	46.5
Net Aboveground Primary Productivity, gC/m^2-yr	153	355	452	136
Net Belowground Primary Productivity, gC/m^2-yr	384	463	394	383
Root:Shoot ratio (R:S)	**2.5**	**1.3**	**0.9**	**2.8**

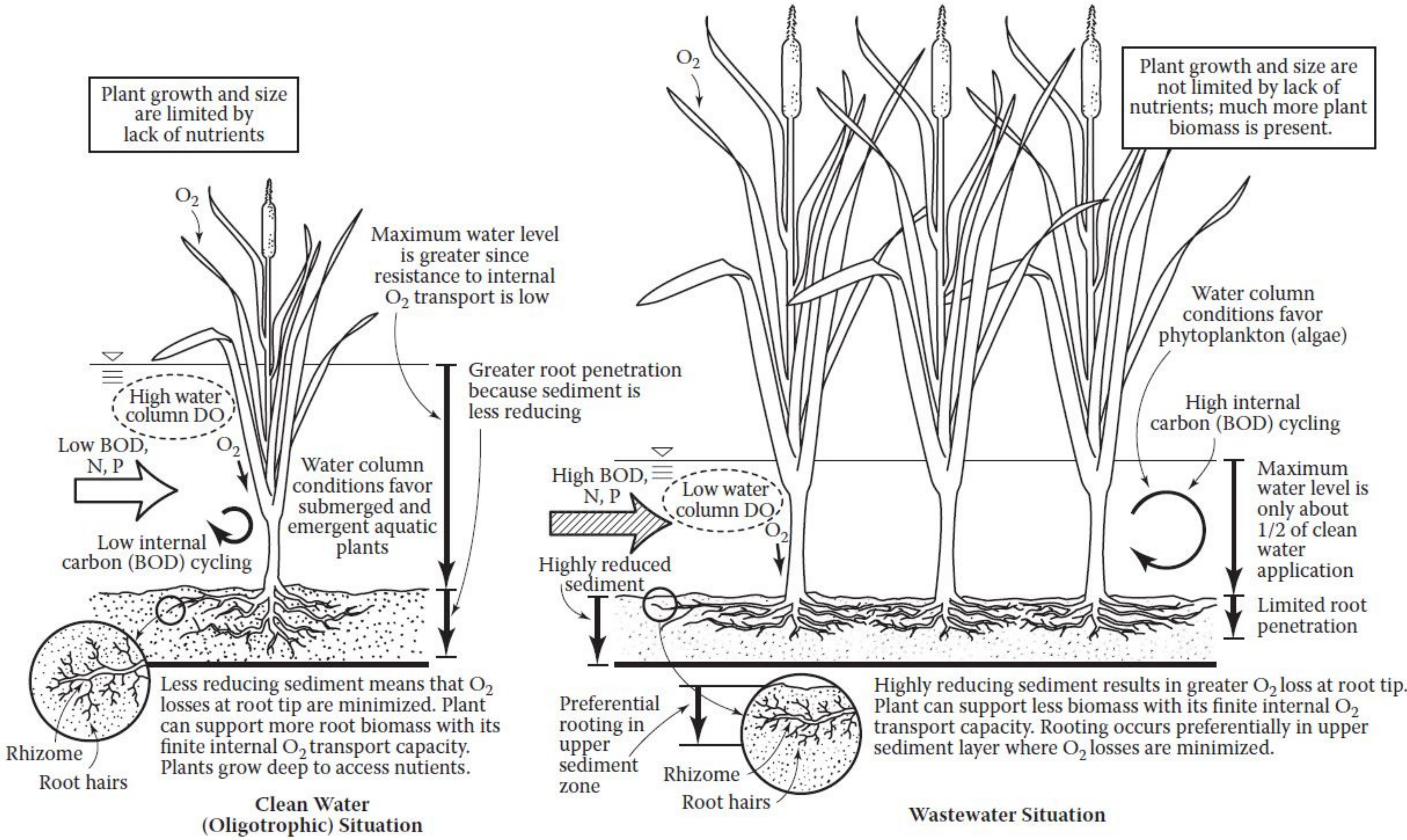
Hillmann (2011) conducted a 5-year study on the effects of fertilization using a total of eleven plant species. These included *Cephalanthus occidentalis* (buttonbush), *Peltrandra virginica* (arrow arum), *Panicum hemitomon* (maidencane), *Pontederia cordata* (pickerelweed), *Taxodium distichum* (bald cypress), *Nyssa aquatica* (water tupelo), *Sagittaria lancifolia* (arrowhead), *Spartina patens* (wiregrass), *Spartina alterniflora* (smooth cordgrass), *Cladium jamaicense* (sawgrass) and *Typha domengensis* (cattail). Hillmann (2011) determined that fertilization increased the total belowground biomass (combined data set of all species) while reducing the root:shoot ratio at the same time (Table 7.3).

Table 7.3 – Above- and belowground biomass as function of salinity and fertilization. Adapted from Hillmann (2011).

Treatment	0F	0	3 ppt	6 ppt
N Loading, g/m^2-yr	90	---	---	---
P Loading, g/m^2-yr	13	---	---	---
Salinity, ppt	0	0	3	6
Aboveground biomass, g/m^2-yr	10,571	2,001	2,145	1,895
Belowground biomass, g/m^2-yr (<30 cm depth)	15,482	7,745	8,292	5,987
Belowground biomass, g/m^2-yr (>30 cm depth)	2,376	1,392	686	818
Total belowground biomass, g/m^2-yr	19,542	9,137	8,978	6,805
Root:Shoot (R:S)	**1.8**	**4.6**	**4.2**	**3.6**
Percentage <30 cm depth	79%	85%	92%	94%

The data of Valiela *et al*. 1976, Anisfeld & Hill (2011) and Hillmann (2011) demonstrates an important point observed in many treatment wetlands; while there is a "fertilizer effect" related to N and P availability, aboveground biomass increases much more relative to belowground biomass, and there is a net reduction in the root:shoot ratio (R:S). If nutrients are not a limiting resource, plants can expend more of their photosynthetic energy on the production of aboveground biomass (Levin *et al*. 1989, Marschner, 2012) as shown in Figure 7.4.

Figure 7.4 – Relative effect of nutrient availability in aboveground and belowground wetland plant biomass (Wallace & Knight, 2006; Kadlec & Wallace, 2009)



Degradation of Existing Soil Organic Material

In a 12-year study of a South Carolina *Spartina alternifolia* marsh, Morris & Bradley (1998) measured the additional soil carbon loss (via CO_2 respiration) from nutrient addition at ≈795 g/m^2-yr, or ≈2.2 g/m^2-d. In a study of Connecticut *Spartina alternifolia* marsh, Anisfeld & Hill (2011) found a much lower carbon loss attributable to N addition; ≈70 g/m^2-yr. Both of these studies found values much lower than that reported by Wigand *et al*. (2009), who reported values ≈1,500 g/m^2-yr of additional carbon loss; however, the values of Wigand *et al*. (2009) probably represent close to maximum values since respiration was only measured during peak summer growing periods.

While observed rates of soil carbon loss were variable, all three studies reported a net increase in CO_2 respiration (and soil carbon loss) associated with nutrient addition. This increase in CO_2 respiration may not necessarily correlate with a loss of wetland sediments. While Morris & Bradley (1998) measured additional soil carbon loss (via CO_2 respiration) from nutrient addition at ≈795 g/m^2-yr, the net carbon loss was only ≈40 g/m^2-yr, implying that 95% of the carbon lost via CO_2 respiration was replenished. The almost-net closure of the carbon balance was attributed by the authors to the role of the plant biomass cycle in the overall carbon cycle, however there was a net loss of soil organic carbon over time, almost all in the upper 10 cm of the soil organic matter, which was the zone of nutrient application.

Litterfall Degradation

Several studies have been carried out that study the decomposition rate of leaf and stem litter material. In a study of a *Spartina alternifolia* salt marsh, Anisfeld & Hill (2011) found no difference in litter degradation rates as a result of nutrient addition, with loss rates of ≈8% in 100 days and ≈20% after one year.

In a study of the Hammond marsh assimilation area, Bodker *et al*. 2015 conducted degradation experiments, estimating loss rates of ≈14.7% for effluent waters compared to ≈9.6% for reference water after roughly 70 days. Bodker *et al*. 2015 attributed the difference to nutrients present in the treated effluent used in the study. A preceding study done by Shaffer *et al*. 2015 looked at decomposition rates at different locations within the Hammond assimilation wetland; locations further away from the effluent distribution pipe were presumed to have lower nutrient inputs. Shaffer *et al.* 2015 found no difference in decomposition rates at different locations, implying nutrients did not play a role in increasing decomposition. The data of Shaffer *et al*. 2015 indicated loss rates of ≈15% in one month and ≈40% after one year. Comparing the two studies done at Hammond, Day *et al*. 2019 noted that results essentially overlap and thus the correlation to nutrient availability was questionable.

In a 4-year study done at the Thibodaux, Louisiana wetland assimilation site, Rybcyzk *et al*. 2002 found no difference in litterfall decomposition rates with and without wastewater effluent.

Gas Uplift

There is evidence that introduction of oxidized forms of nitrogen (nitrate) results in increased gas production due to denitrification in wetland soils. In a salt marsh experiment, Turner & Bodker (2016) demonstrated increased gas production when NO_3-N was added to *Spartina alterniflora* soils, but addition of P had no effect on gas production.

While denitrification appears logical in a wetland substrate, studies using non-oxidized forms of nitrogen have also demonstrated increased levels of CO_2 evolution (Morris & Bradley 1998, Anisfeld & Hill 2011, Wigand *et al*. 2009, Bodker *et al.* 2015).

Thus, it appears that gas formation will increase as a result of nutrient availability. The question then becomes whether or not the gas is produced in sufficient quantities to exert significant buoyancy forces on the wetland substrate. Gas formation is often associated with methane (CH_4) formation in freshwater wetlands, but in coastal situations, methane formation is often precluded by the abundance of sulfate (SO_4). Redox measurements have not been carried out in the Hammond Assimilation Wetlands, therefore opinions regarding gas uplift as a significant mechanism are speculative.

Soil Accumulation (Accretion)

In an experiment in a *Spartina alternifolia* salt marsh, Anisfeld & Hill (2011) found an increase in net accretion rates (accretion – subsidence) as a result of nutrient additions (N, P, N+P). While there was some soil carbon loss attributable to N addition (≈70 g/m^2-yr), this was offset by increased biomass production, and greater sediment trapping efficiencies. A similar conclusion was drawn by Morris & Bradley (1998), who observed an increase in net accretion despite apparent soil carbon loss (≈40 g/m^2-yr).

In a study done at the Thibodaux, Louisiana wetland assimilation site, Rybcyzk *et al*. 2002 calculated accretion rates of 11.4 mm/yr for the assimilation area vs. 1.4 mm/yr in a nearby control site, vs. a relative sea level rise (RSLR) at the site of 12.3 mm/yr. Since the site received treated wastewater effluent and no significant sources of mineral sediments, the increased rate of accretion was attributed to biomass production, decomposition and storage.

Conner & Day (1988, 1991) found that vertical sediment accretion averaged 8.8 mm/yr for forested swamps in the Lake Verret basin of south-central Louisiana, vs. a RSLR of 13.7 mm/yr.

A model to evaluate the role of nutrient addition to accretion was developed by Rybcyzk *et al*. 1998, who concluded that increased biomass production due to effluent application would increase accretion by 0.35-0.46 cm/yr; however, this would be insufficient to offset RSLR of 0.69-1.74 cm/yr (Penland & Ramsey 1990).

These projects all demonstrate that the net effect of nutrient availability (when all mechanisms are considered) is to increase soil accumulation (accretion).

Conclusions

The addition of nutrients will increase the production of plant biomass. While different researchers have drawn different contributions about whether or not belowground biomass increases or decreases, scientific studies concur that the root:shoot ratio decreases with nutrient availability. As a result, much more of the additional plant biomass is produced aboveground than belowground.

This increase in aboveground biomass can make the vegetative community “top-heavy” and susceptible to wind throw and storm damage, delaminating the underlying wetland substrate (Figure 7.5). Once an “under mat” flow path develops, this is the path of least resistance, carrying water and nutrients below the floating mat (Kadlec, 2009a, Kadlec & Bevis, 1989, 2009).

Figure 7.5 – Wind throw damage at the Hammond Assimilation Wetland (from Turner *et al*. 2018)

In this situation, the carbon mined for nutrient assimilation is belowground, while the new carbon stored is aboveground. In the short term, this appears to be a self-reinforcing mechanism to promote floating mats in marsh wetlands. Over long time periods, it is entirely possible for the aboveground biomass to accumulate to the extent that it would push the floating mat down, resulting in an anchored marsh substrate.

While this in-filling process is a naturally-occurring phenomena in wetlands (Mitsch & Gosselink, 2007), no wastewater assimilation wetlands have operated long enough to measure this in detail. One could argue that the floating mat development in the former open-water section of the Thibodaux wetlands (Izdepski *et al*. 2009) is an example of this process in its early stages, and that given enough time (many decades of operation) long-term infilling of assimilation marshes could be documented with verifiable rate coefficients.

8.0 Tree Growth

The vast majority of this study was based on documents from the historical record. As a means of independently verifying whether or not effluent assimilation was having any effect (negative or positive), on the receiving wetlands, a small subset of cypress trees were sampled as part of this study.

8.1 Results from the Hammond MID Location

Core samples were taken from cypress (*Taxodium distichum*) trees at the MID monitoring location on November 14, 2018 (Figure 8.1). Dendochronology (tree ring) analysis was selected because it allows analysis over an extended time period before and after the start of effluent assimilation.

Figure 8.1 – Use of increment borer at the Hammond MID location, November 14, 2018

A small selection of bald cypress (*Taxodium distichum*) trees were sampled with an increment borer on November 14, 2018 as part of this study. Ten trees were sampled (with nine recoverable core samples) in the immediate location of the MID monitoring point of the Hammond Assimilation Wetland, located north of the JWMA boardwalk.

Samples were prepared into wooden mounts and tree ring increments were measures with a digital microsope (Figure 8.2). Results are summarized in Table 8.2.

Average ring increment growth was 2.25 mm/yr (5.5 mm/yr diameter increase) for the period from 2007-2018 (after effluent addition). Going back earlier in time, the average ring increment growth was 1.25 mm/yr (2.5 mm/yr diameter increase) for the period 1986-2006 (the 20 years prior to effluent addition). The growth ratio was **1.87**, indicating the same trees grew 1.87 times faster after effluent addition started began in the Fall of 2006.

Figure 8.2 – Tree ring measurement using a digital microscope

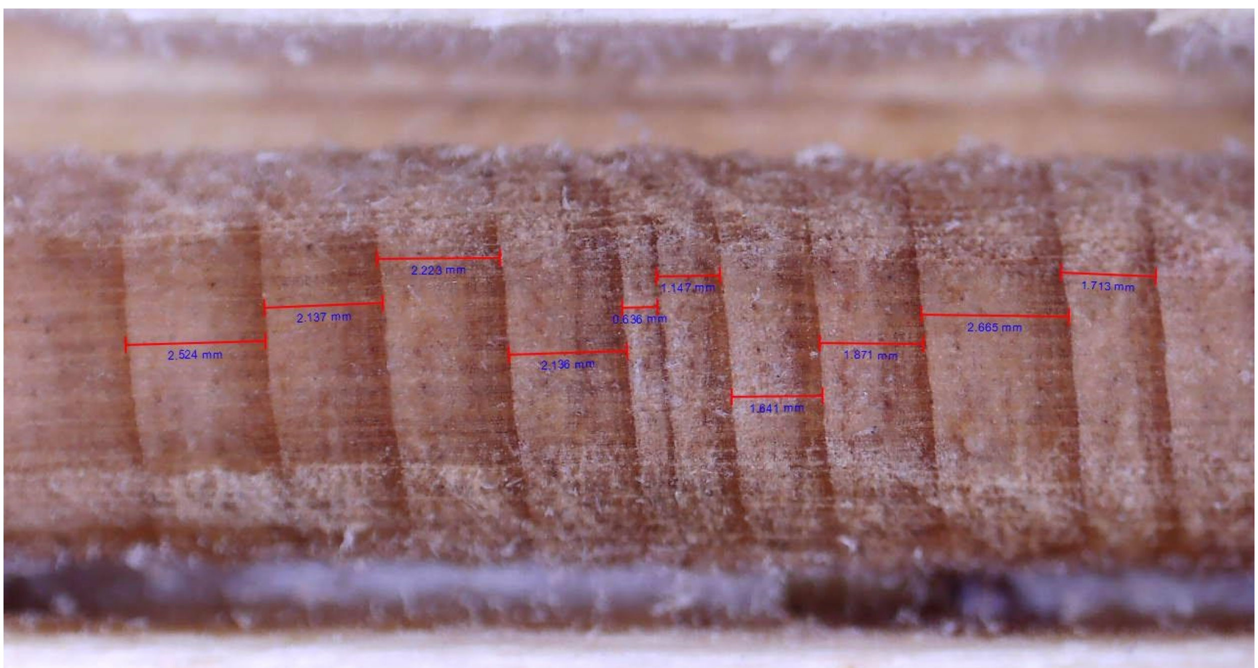


Table 8.1 – Tree growth data from Hammond MID site, 1986-2006 (pre-project) vs. 2007-2018 (with effluent application)

Tree Sample	Radial Tree Growth, mm/yr		Ratio
	1986-2006	2007-2018	
1A	1.052	1.337	1.27
3A	1.345	2.356	1.75
4A	0.790	1.230	1.56
5A	2.183	2.786	1.28
6A	1.263	1.818	1.44
7A	0.940	2.764	2.94
8A	1.064	2.588	2.43
9A	1.093	2.539	2.32
10A	1.532	2.805	1.83
Mean Value	*1.251*	*2.247*	***1.87***

The results of Table 8.1 are part of a larger data set collected from the trees, summarized in Figure 8.3.

The dates shown in Figure 8.3 become progressively less accurate as one proceeds back in time. This is due to the fact that errors from false tree rings are possible (Ewel & Parendes, 1984, Young *et al*. 1993, Copenheaver *et al*. 2017), and cross-correlation to known tree cores (Stokes & Smiley, 1968) was not done. This type of cumulative error was deemed acceptable because the period of interest was the most recent past, 1986-2018.

The story presented in Figure 8.3 is consistent with the history of the area (Section 3). Only one of the trees could have germinated before the advent of commercial logging circa 1890 (Mancil, 1972), and all of the trees had germinated before the end of the logging era around 1938 (Norgress, 1947; Lopez, 2003). Tree growth declined after the construction of I-55 and the South Slough Canal in the 1960's (Keddy *et al*. 2007, Lane *et al*. 2015), reflecting the less favorable hydrology that resulted from those projects. This history and variability in cypress growth over time is consistent with that observed by Hesse & Day (1998) for the Breaux Bridge wetlands.

Typically, the rate of tree ring growth decreases with age (Fritts, 1976). At the Hammond MID location, the opposite as happened. The most recent period of record, 1992-2017 is shown in Figure 8.4. While the tree growth at Hammond MID was not compared to a control group, nutrients likely played a major role in enhanced tree growth. The MID location is well within the assimilation zone modeled in this study (Section 5), and the annual monitoring reports prepared for the City of Hammond (Comite Resources, 2007-2017) show that N and P concentrations at this location are still above background concentrations.

Figure 8.3 – Annual growth rates of cypress trees at the Hammond MID location, entire period of record (different symbols refer to individual trees)

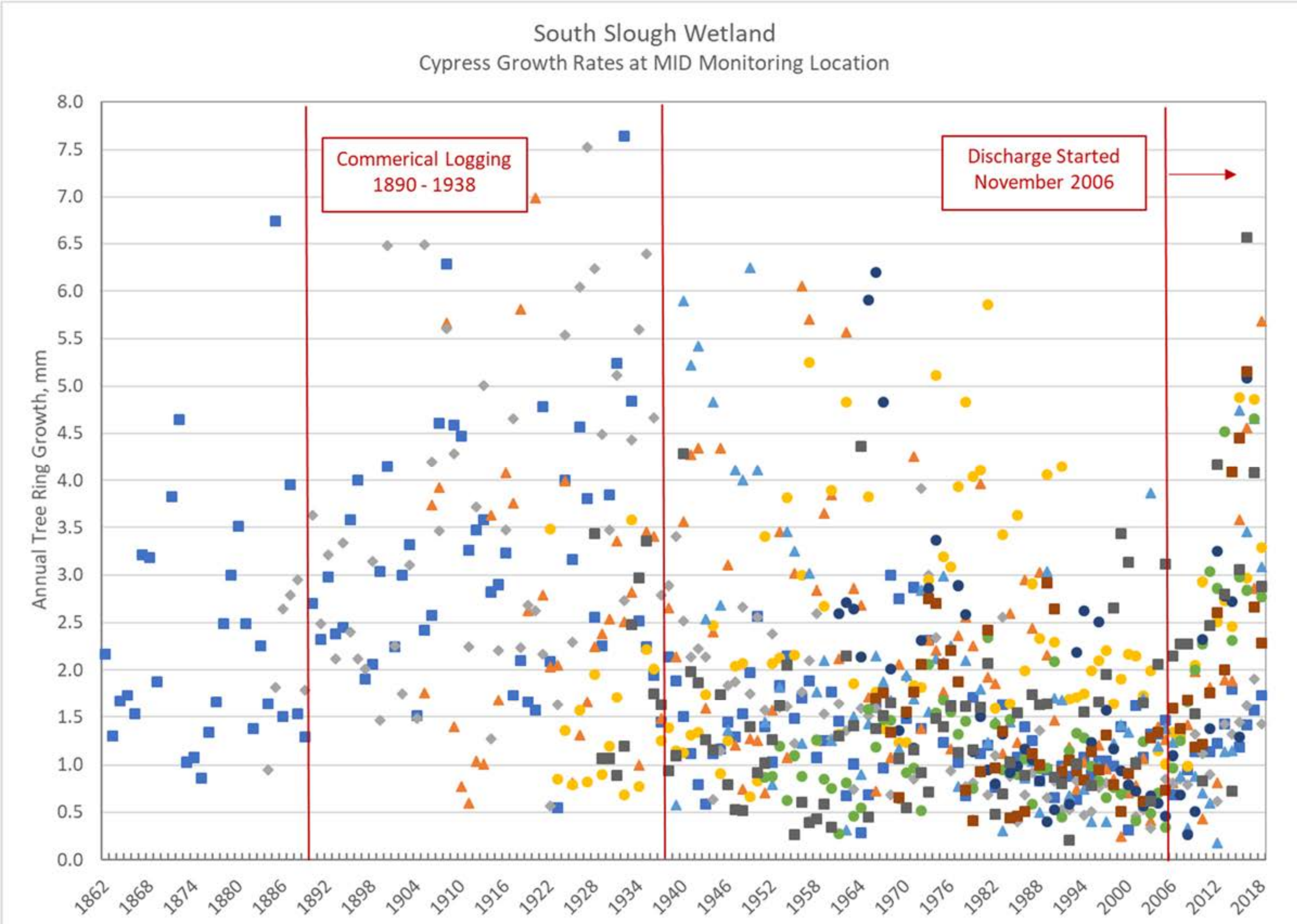
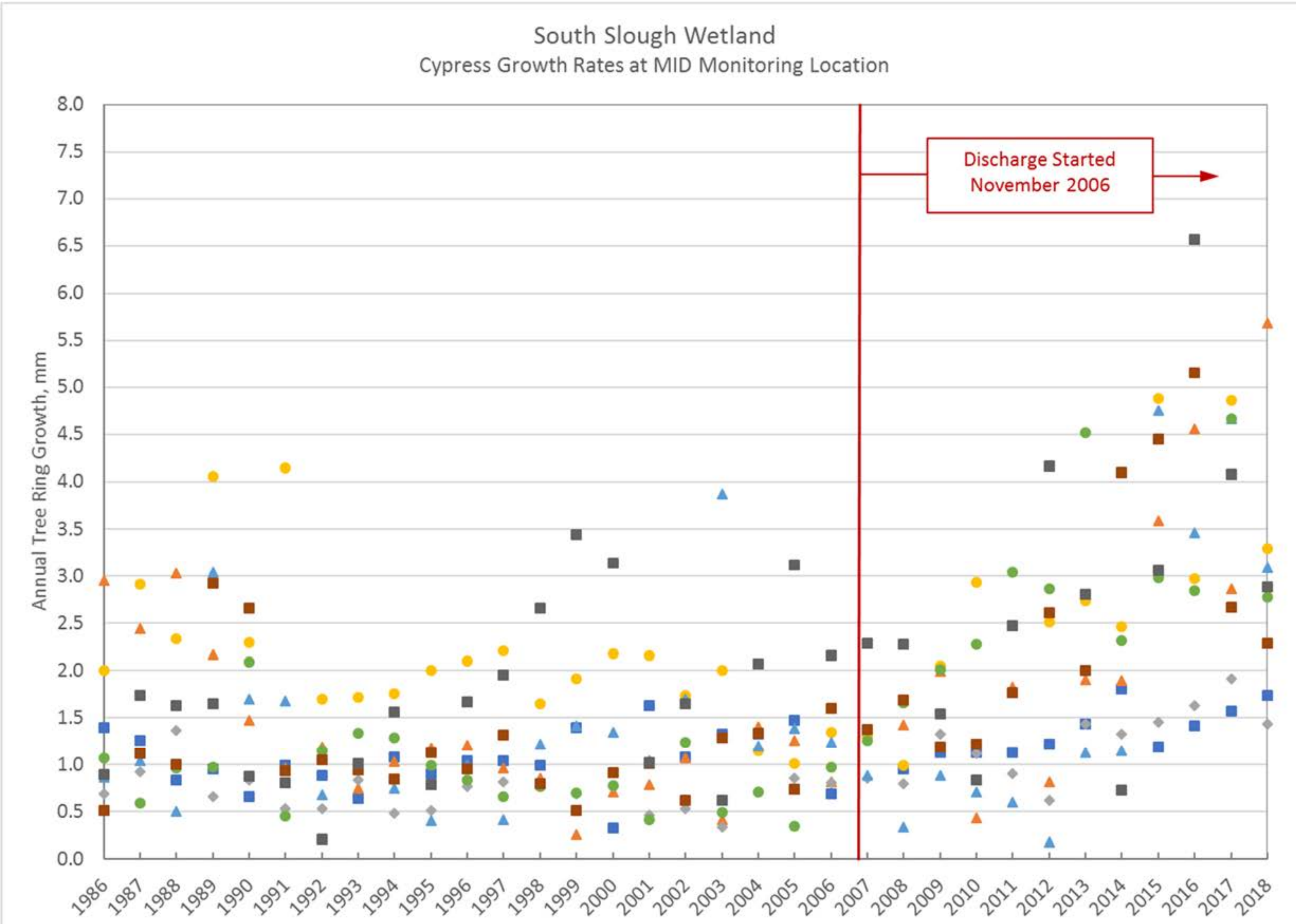


Figure 8.4 – **Annual growth rates of cypress trees at the Hammond MID location, 1986-2018 (different symbols refer to individual trees)**



8.2 Comparisons to Similar Wastewater Assimilation Projects

In a study done on the Breaux Bridge, Louisiana assimilation wetland, Hesse & Day 1998 studied the effect of municipal wastewater additions on the growth of *Taxodium distichum*. During the time period studied after effluent addition began (1948-1992), the growth ratio was **1.52** (0.91-2.64), compared to 1.87 at the Hammond MID location. This indicates that trees in the effluent assimilation area at Breaux Bridge grew 1.52 times faster than those not receiving effluent.

Hillmann *et al*. 2018 studied the fertilizer response of bald cypress (*Taxodium distichum*) and water tupelo (*Nyssa aquatica*). At a maximum fertilization rate of 400 gN/m^2-yr and 58 gP/m^2-yr, the resulting growth ratio was ≈**2.14** (combined *Taxodium* and *Nyssa* data set).

Effler & Goyer (2006) studied the effects of fertilization on *Taxodium* and *Nyssa*. Fertilization increased total biomass, with a growth ratio ≈**2.6** while decreasing the root:shoot ratio. These results are similar to those reported by Hillmann (2011) (see Table 7.3).

Hunter *et al*. 2018 summarized performance of wastewater assimilation wetlands in Louisiana. Comparing discharge monitoring sites with reference sites results in the following growth ratios. Breaux Bridge (2002-2013) **1.2**, Broussard (2007-2013) **2.2**, Luling (2008-2013) **1.3**.

During the period of effluent assimilation, the annual tree ring increment growth was between 2-4 mm/yr, at Breaux Bridge (1948-1992) compared to 2.25 mm/yr (2007-2018) observed at the Hammond MID location.

Nessel *et al*. 1982 studied a Florida pondcypress (*Taxodium distichum* var. *nutans*) wetland that had been receiving septic tank effluent for 41 years. After the introduction of the septic tank effluent (estimated P loading of 4.2 gP/m^2-yr), the growth rate doubled (≈**2**). Cypress trees in a nearby wetland that did not receive septic tank effluent showed no comparable increase in growth rate. In a similar study of Florida pondcypress wetlands, Brown (1981) estimated sewage addition increased *Taxodium* net primary productivity, with a growth ratio of ≈**2.1**.

Keim *et al*. 2012 studied the assimilation wetland at Thibodaux, Louisiana. The site receives an estimated 3.1 g/m^2-yr of N and 0.6 g/m^2-yr of P (loading chart method). The expected increase in tree growth did not occur. This was attributed to increasing inundation of the assimilation area, which has a negative effect on growth rates, and the combination of the two (nutrient addition = greater growth; increased inundation = lower growth) cancelled each other out. This study, relative to other studies, is summarized in Table 8.2.

Lundberg (2008) studied the growth of approximately 6,000 *Taxodium* seedlings planted at the Hammond Assimilation Wetland. Lundberg (2008) noted that cypress seedlings in the vicinity of the discharge pipe grew approximately **2.2** times faster than those 700 m away. However even at a distance of 700 m from the discharge pipe, seedlings would have been well within the active assimilation zone of the Hammond wetlands, so the "baseline" used by Lundberg (2008) still represented a condition of enhanced growth. Considering the pre-project growth rate at the Hammond MID location as a baseline, trees grew 1.87 times faster after effluent addition. If tree growth at the MID location was approximately the same as trees growing 700 m from the discharge pipe, the growth rate at the discharge pipe would be 2.2 X 1.87 = **4.1** times greater. Since trees at the MID location were probably growing more slowly than those 700 m away from the discharge pipe, 4.1X is likely an under-estimate. This is in rough agreement with Shaffer *et al*. 2015 who observed that cypress seedlings at the discharge pipe grew five times faster than those in the Maurepas swamp and ten-fold faster than those in the non-project regions of the JWMA.

Table 8.2 – Growth Response Ratio of *Taxodium* and *Nyssa* to nutrient addition in forested wastewater assimilation wetlands

Study	Location	Period of Record	Growth Response Ratio
This Study	Hammond MID	1986-2018	1.87
Lundberg, 2008	Hammond NEAR	2006-2008	2.2 (1)
Keim *et al*. 2012	Thibodaux	1992-2005	≈1
Hesse & Day, 1998	Breaux Bridge	1811-1993	1.5
Hunter *et al*. 2018	Breaux Bridge	2002-2013	1.2
Hunter *et al*. 2018	Broussard	2007-2013	2.2
Hunter *et al*. 2018	Luling	2008-2013	1.3
Hillmann *et al*., 2018	Experiment	2009-2010	≈2.1
Effler & Goyer, 2006	Experiment	2001-2003	≈2.6
Nessel *et al*. 1982	Florida cypress dome	1934-1975	≈2
Brown, 1981	Florida cypress dome	1972-1976	≈2.1

Note:

1. Baseline of Lundberg (2008) was 700 m away from the effluent distribution pipeline, comparing this to growth at the MID location (this study) indicates the growth response ratio was 4-5X that of the background regions of the Joyce WMA.

9.0 Summary and Conclusions

The study was an independent evaluation of the City of Hammond, Louisiana wetland wastewater assimilation project, defined as the "South Slough Wetland" by LDEQ and permitted by the Department under Louisiana Pollutant Discharge Elimination System (LPDES) Permit LA0032328. The "South Slough Wetland" includes a section of freshwater marsh (locally known as Four Mile Marsh) immediately downstream of the effluent distribution pipeline. This area of marsh is about 122 ha (300 acres) in size. Beyond that, effluent can spread to the south and east over extensive cypress-tupelo swamps greater than 4,000 ha (>10,000 acres) in extent as water moves towards Lake Pontchartrain

The project is based on a Use Attainability Analysis completed by Comite Resources in April 2005 (UAA, 2005). Discharge of secondarily-treated municipal wastewater effluent began November 2006. After about one year of operation, the section of fresh water marsh immediately downstream of the distribution pipeline converted to open water and mudflats during 2008-2009. This area has largely revegetated with a mixed plant community (including annuals) that is different than the original marsh community, in which maidencane (*Panicum hemitomon*) was a keystone species. The reason for this marsh conversion is an ongoing dispute in the scientific literature, with different proponents advocating that either nutrients or nutria were the dominant cause of the vegetation change.

Despite the marsh conversion, the system has consistently and successfully met requirements for nutrient assimilation, salinity reduction, and enhanced plant productivity. Over the period of record (2006 - 2017), the system has produced the water quality benefits expected from wetland assimilation projects. Concentrations of TKN, NO_3-N, NH_4-N and TP were at ecosystem background levels at the OUT location, indicating these nutrients had been completely assimilated by the wetland. The addition of treated wastewater effluent, a low-salinity water supply, was effective in lowering salinity levels at the NEAR and MID locations, keeping salinity well below levels that cause stress to cypress and tupelo trees.

This study used a mathematical modeling approach to estimate the area actively involved in nitrogen (N) and phosphorus (P) assimilation (Kadlec, 1997). This has not been done for previous assimilation wetlands in Louisiana, which have historically used a "loading chart" approach to predict system performance (Nichols, 1983; Richardson & Nichols, 1985). Model parameters were based on 30 years of operation for a similar assimilation project in Michigan (Kadlec & Bevis, 2009) and modified to reflect the year-round operation the Hammond system.

These modeling calculations indicate that the "active assimilation zone" is far smaller than the overall South Slough Wetlands (5% in 2006, now up to approximately 16% of the total area as of 2018), and that N and P cycling in this assimilation zone is far more intense than that predicted by the "loading chart" approach. Most of this active assimilation zone is in the freshwater marsh region that underwent large vegetative changes in 2008-2009 after effluent discharge began.

This study also collected tree ring samples from nine individual cypress (*Taxodium distichum*) trees at the MID monitoring location. Analysis of tree growth indicated that the same trees grew by a factor of **1.87** faster (almost double) after effluent application began. This increase in growth is consistent with other wastewater assimilation wetlands recorded in the scientific literature.

9.1 Conclusions Related to Permit Compliance

1. The system consistently meets objectives (as outlined under LPDES Permit LA0032328) for assimilation of nutrients, reduction of salinity, and enhancement of plant productivity. Over the period of record (2006 - 2017), the system has produced the water quality benefits expected from

wetland assimilation projects (see Section 5.0 for details). Comparing data averages from the NEAR and OUT monitoring locations:

- Total Kjeldahl Nitrogen (TKN) was reduced from 10.0 to 0.9 mg/L.
- Ammonia Nitrogen (NH_4-N) was reduced from 6.2 to 0.2 mg/L.
- Nitrate Nitrogen (NO_3-N) was reduced from 1.4 to 0.1 mg/L.
- Total Phosphorus (TP) was reduced from 3.2 to 0.2 mg/L.
- Salinity was 1.66 PPT at the OUT location but was only 0.29 PPT at the NEAR location.

Concentrations of TKN, NO_3-N, NH_4-N and TP were at ecosystem background levels at the OUT location, indicating these nutrients had been completely assimilated by the wetland. The addition of treated wastewater effluent, a low-salinity water supply, was effective in lowering salinity levels at the NEAR and MID locations, keeping salinity well below levels that cause stress to cypress and tupelo trees.

2. The "fertilizer effect" of available nutrients resulted in increased plant productivity. Measurements of plant biomass production over the growing seasons when nutrients were available (2007 – 2017) all indicated enhanced plant growth:
   - For the marsh vegetation, End of Season Live Biomass (EOSL) was 2.2X greater at the NEAR location compared to the Marsh Control.
   - For the forest vegetation, Litterfall was 2.8X greater at the MID location compared to the Forest Control.
   - Similarly, Stem Growth was 2.8X greater at the MID location compared to the Forest Control.
   - For cypress trees studied at the MID location, the average growth rate was 1.87X faster after effluent application began compared to the 20 years prior to the project (see Section 8.0 for details).

9.2 Conclusions Related to Hydrologic Limitations

1. There are almost no hydrologic limitations to the effective spreading of effluent. Even under the maximum assimilation area estimated (660 ha in Table C.2), effluent spreads out to a larger area than this ≈75% of the time (Figure 5.7). For about 40% of the time, precipitation exceeds evapotranspiration and effluent can theoretically spread to the outlet of Lake Pontchartrain. This makes it very unlikely that there is a constraint on nutrient assimilation due to excessive water loss.

9.3 Conclusions Related to Marsh Conversion

1. During the period of marsh conversion, the effluent flows (Figure 5.2), rainfall (Figure 5.5), and water levels (Figure 5.4) were low and relatively stable. This seems to rule out a hydrologic cause (drought or flooding) as a reason for the observed vegetation changes in the affected sections of Four Mile Marsh.

2. Modeling was completed to determine the rate of biomass cycling (Section 6) that would be required if nutrient addition were the sole cause of the vegetation changes observed in Four Mile Marsh. This would require the system to 1) assimilate phosphorus at a fast rate (86% percentile of observed rates in Kadlec & Wallace, 2009, and 2) cycle (turn over) the biomass approximately 20 times per year (0.6 months). This is extremely fast and is at the highest ranges of the fastest wetlands ever recorded (Wigand *et al*. 2009). In addition, the system would have cycle at this rate year-round, and not just during the peak summer months studied by Wigand *et al*. (2009). This combination has never been recorded in the scientific literature and is considered very unlikely.

3. Modeling was completed to estimate the amount of vegetation loss due nutria herbivory (Section 6), based on the estimated density of animals recorded at Hammond (16.4 nutria per hectare) and the methodology outlined by Holm *et al*. (2011). These calculations indicate that the nutria could have caused a moving "eat out" of 60 ha per year, which would have caused the marsh conversion in the approximately 2-year timeframe described by observers (Shaffer *et al*. 2015, Day *et al*. 2019). These calculations indicate that nutria herbivory provides a plausible explanation for the marsh conversion, but is predicated on several important assumptions:

    a. Nutria are "wasteful feeders" as described by Holm *et al*. (2011) and destroy 10 times more vegetation than they actually consume.

    b. The marsh plant community was a relict system that developed under hydrologic conditions that no longer existed, even before the wastewater assimilation project (Section 3). The relict system was not all that stable and thus subject to switching to a different system-state as described by Sasser *et al*. (1996) in the face of a new set of ecosystem drivers. Consequently, vegetation grazed by nutria died and did not grow back.

    c. The nutria had to come from somewhere. The role of nutrient-fertilized vegetation as a nutria attractant (Ialeggio & Nyman, 2014) had to occur in order to generate the nutria densities reported.

9.4 Conclusions Related to Soil Strength, Biomass Production and Development of Floating Mats

The loss of soil strength, the influence of nutrients on plant growth, and the development of floating mats as a result of effluent application (Section 7) was investigated:

1. While denitrification is an obvious means of soil carbon loss, this would be a factor in close proximity to the effluent discharge pipeline since nitrate is rapidly removed in assimilation wetlands (Zhang *et al*. 2000, Kadlec 2009a), including the Hammond Assimilation Wetland (see Figure 5.10). Even without oxidized forms present, nitrogen appears to stimulate greater CO_2 respiration from wetland soils (Morris & Bradley, 1998; Wigand *et al*. 2009; Anisfeld & Hill, 2011). It is possible to apply environmental stoichiometry to wetland peat degradation rates (Kleimeier *et al*. 2018) and compare those to the litter degradation experiments of Bodker *et al*. (2015), and Shaffer *et al*. (2015) to estimate this carbon loss at approximately 2.6 gC/m^2-d, which represents about ≈2% per year of soil carbon loss. New carbon is added to the wetland soils each growing

season through the plant biomass cycle, which essentially closes the net soil carbon balance to zero, even in zones of maximum nitrate assimilation.

2. While some studies claim that nutrients reduce belowground biomass (Darby & Turner 2008a, 2008b, Turner, 2011), other studies demonstrate the opposite (Valiela *et al*. 1976; Anisfeld & Hill, 2011, Hillmann, 2011). However, the consensus among all studies is that plants produce much more aboveground biomass than belowground biomass (the root:shoot ratio decreases). This places much more of the "new carbon" aboveground than belowground. The center of gravity of the plant biomass moves upward, rendering the system more susceptible to wind throw and storm damage (Figure 7.5; Turner *et al*. 2018) in peat soils. This can further delaminate and separate the underlying organic substrate layers.
3. If under-mat flow develops, this is the flow path of least resistance (Kadlec, 2009a, 2009b; Kadlec & Bevis, 2009). When this occurs, the carbon utilized for nutrient assimilation is "old carbon" from the under-mat region, and the nutrients uptaken by plants produce "new carbon" which is primarily aboveground.
4. The combination of 2 and 3 above can apparently generate a self-reinforcing trend of floating mat development in formerly fixed peat substrates (Kadlec & Bevis, 1989, 2009). While the long-term effects of increased biomass production may eventually lead to in-filling of floating marshes, no wastewater assimilation wetlands have operated long enough to observe this change back to a fixed substrate. The available information indicates that floating mats are a likely outcome of future marsh assimilation projects when the pre-existing marsh is based on accumulated peat, instead of being firmly anchored in a mineral soil.

9.5 Conclusions Related to the Permit Planning Process (Use Attainability Analysis)

The assimilation zone modeling carried out in this study indicates that the active assimilation zone is far smaller than the total project area, and the most intense region (the "biomass maximum" zone) is in the region of Four Mile Marsh where the controversial vegetation changes have occurred. With the benefit of hindsight, this brings into question the scope and suitability of the original Use Attainability Study (UAA, 2005).

While there is no reason to conclude the UAA was done incorrectly (based on the status of wetland science known at the time), in retrospect the UAA is not a very useful document in addressing the ongoing controversy associated with the marsh conversion. This brings to light several key points:

1. The tool available to wetland designers at the time was the "loading chart" approach of Nichols (1983) and Richardson & Nichols (1985). This approach assumes that the entire project area is involved in nutrient assimilation. As a result, the area of the "active assimilation zone" was not estimated. The insight that Four Mile Marsh would form the majority of the active assimilation zone was overlooked, and the freshwater marsh was thus not studied in detail as part of the UAA.
2. Monitoring locations for the project were based on the assumption that the entire 10,000 acre (4,047 ha) South Slough Wetlands was involved in nutrient assimilation. As a result, these monitoring locations (except for the MID location) are located far outside the active assimilation zone. Consequently, permit-related compliance monitoring yields very little information about

how and where the wetlands actually assimilate nutrients. Had the location of the monitoring locations been located in relation to the expected size of the active assimilation zone, data derived from permit-related monitoring would be far more relevant to addressing the ongoing issues related to marsh conversion.

3. Baseline studies were based on the same assumption that the entire 10,000-acre South Slough Wetlands was involved in nutrient limitation. Since the majority of this area is forested swamp, this was a focus of the UAA. However, the majority of the active assimilation zone is actually within Four Mile Marsh. Detailed baseline studies of the marsh were therefore not conducted. This is relevant even to the current day, as there is ongoing debate over the pre-existing nature of the marsh plant community (*Panicum*-dominated or mixed species in a relict *Panicum* mat), the pre-existing nature of the marsh structure (floating mat vs. fixed substrate), the pre-existing role of nutria in the system, the origin and stability of the Four Mile Marsh plant community, etc. The fact that there is no clear map of what the marsh was prior to the project has turned out to be a major problem. The fact that permit-related monitoring does not provide updates on what is happening within the marsh is an ongoing problem.

10.0 Recommendations

Due to the wide-ranging scope of this study, recommendations are broken down into two categories; those specific to the City of Hammond project and those related to LDEQ's approach to permitting wastewater assimilation wetland projects in the State of Louisiana.

10.1 Recommendations Specific to the Hammond Assimilation Wetland

1. Continuance of the effluent discharge is strongly recommended for the following reasons:
    a. The system is clearly successful in meeting the objectives of nutrient assimilation, salinity reduction, and productivity enhancement.
    b. Effluent assimilation is clearly enhancing the growth of cypress trees at the MID location.
    c. Changes in the marsh community should continue to be monitored. Discontinuing the discharge is highly unlikely to return the marsh to the pre-project state, due to the structural changes (development of open water and floating mats) which have occurred.
2. There is only a single permit-related compliance point (MID) in the active assimilation zone. Consideration of multiple monitoring locations within the active assimilation zone should be considered (see Future Permitting Recommendation #4 below).
    a. For the City of Hammond, the NEAR and MID locations are within the active assimilation zone. However, the NEAR location is essentially at the discharge pipe, leaving the MID location as the only monitoring point in the active assimilation zone. It is recommended that future projects have at least three monitoring locations since it will take 4-10 years for the active assimilation zone to develop. The active assimilation zone in the Hammond Assimilation Wetland is also expanding due to an increase in effluent flows and loads that have occurred since the project began (see Table C.2 for details). Due to these factors, having two monitoring stations in Four Mile Marsh (between the NEAR and MID locations), and one additional monitoring location downstream of the MID location near the anticipated edge of the active assimilation zone is recommended.
3. A comprehensive survey of the impacted region of Four Mile Marsh is recommended, including plant species, soil measurements, and water quality measurements. This would be analogous to a mid-project UAA.
    a. Establishment of fixed monitoring locations where annual plant surveys can be conducted is recommended.
4. The City has multiple dosing zones along the distribution pipeline, but how this affects flow in the wetland is poorly understood. For instance, during the November 2018 site visit, some flow was observed leaving the site to the west and to the north (via South Slough). Field studies to better understand this are recommended. Replacement of the two water control structures that connect to South Slough is recommended. Replacement structures should be water-tight and allow positive operator control.

10.2 Recommendations for Future Permitting of Wastewater Assimilation Wetland Projects

1. The assumption that the "do nothing" option (no effluent addition) represents "no change" (maintenance of the current wetland ecosystem) is questionable at best in coastal Louisiana. There is a considerable body of evidence (Sasser *et al*. 1996, Visser *et al*.1999, Shaffer *et al*. 2016, among many others) that indicate that both freshwater marshes and forested swamps in coastal Louisiana will continue to decline and disappear without human interventions to re-introduce sources of fresh water, nutrients and sediments. Evaluation of future assimilation projects should therefore consider both outcomes; what changes to the marsh/swamp would happen with effluent addition, and what changes will occur without effluent addition.

2. Many wetlands in coastal Louisiana are in a relict state, where the current plant communities developed prior to human-induced hydrologic (channelization, drainage) and vegetative (logging, nutria) changes. As a result, marsh communities may not be all that stable (Visser *et al*. 1999) and susceptible to shifting to a different community system-state (Sasser *et al*. 1996), and swamps may be unable to regenerate, subject to salinity-related die-off (Shaffer *et al*. 2016), and also be susceptible to shifting to a different system-state (Keddy *et al*. 2007). Evaluation of future assimilation projects should therefore consider that stability of the pre-project vegetative community, and what likely changes will occur both the absence of effluent application and with effluent application.

3. The use of the "loading chart" approach (Nichols, 1983; Richardson & Nichols, 1985) is based on the assumption that the entire project area is involved in nutrient assimilation. This is inadequate to predict the performance of future assimilation projects because the active assimilation zone is not determined by this method. The use of more current design tools (Kadlec, 1985, 1997, 2009a; Kadlec & Bevis, 2009) that allow estimation of the size of the active assimilation zone is recommended (see Appendix C for more details).

4. The location of compliance monitoring points should be established relative to the anticipated extent of the active assimilation zone. Having a single MID monitoring location between the NEAR and OUT locations only provides a single data point on what is happening inside the active assimilation zone. This is problematic for several reasons:

    a. Relying on a single data point assumes that the extent of the active assimilation zone was established with certainty (which has not been the case for previous assimilation projects in Louisiana).

    b. Relying on a single data point also assumes that the size of the active assimilation zone is static (which is not the case when flows and loads to the system are increasing over time) such that the center of the assimilation zone does not change.

c. Relying on a single data point further assumes that the flow path through the system is always the same, regardless of water level, which is not the case (Blahnik & Day, 2000). In reality, flow through the system will vary at different water levels, and also when effluent is applied to different distribution zones (Lane *et al*. 2015).

The active assimilation zone will take multiple years to fully develop. The zone will be about 20% developed after one year, 50% developed after four years, and 90% developed after 10 years. The area of the active assimilation zone can also increase over time if the effluent discharge is increasing. Having at least three monitoring points located in the area where the active assimilation zone will develop would provide much more information about the active assimilation zone and the rate of formation.

5. Ongoing vegetation surveys should list major species and estimated percent cover for both swamp and marsh areas. This should be done at least every 4th year of the permitting cycle (similar to the reporting schedule in Table 2.2).

11.0 References

For references, visit: https://deq.louisiana.gov/assets/docs/Water/SouthSloughWetlandEvaluationReport_11.0_References.pdf