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Long-term assimilation wetlands in coastal Louisiana: Review of monitoring data and management

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ABSTRACT

The term 'assimilation wetland' has been applied to natural wetlands in Louisiana into which disinfected, secondarily treated municipal effluent is discharged with the dual purpose of improving regional water quality and enhancing vegetation productivity and soil accretion. Some municipalities began discharging treated effluent into wetlands prior to state regulations, which began in 1992. Here we review data and observations from five assimilation wetlands in the Mississippi River Delta receiving discharge of treated effluent for 26–70 years. In addition, we examine two adjacent forested wetlands, one that receives periodic Mississippi River input and one that does not. Information from these sites provides insight into how long-term nutrient input impacts coastal wetlands. Analysis of tree-ring, leaf litter, accretion, and water quality data shows that input of freshwater containing nutrients and sediments leads to enhanced wetland productivity and soil accretion via increased organic matter burial. In addition, long-term data indicate that assimilation wetlands continue to be nutrient sinks even after decades of effluent discharge, with both nitrogen and phosphorus reduced to background levels. Collectively, these data demonstrate that wetlands benefit from long-term discharge of treated municipal effluent. Properly managed wetland assimilation systems can function for long periods and lead to enhancement of degrading wetland communities in coastal Louisiana.

1. Introduction

Assimilation wetlands are natural (i.e., non-constructed) wetlands into which disinfected, secondarily treated municipal effluent is discharged with the dual purpose of improving regional water quality and enhancing vegetation productivity and soil accretion to help offset subsidence and relative sea level rise (RSLR; Hesse et al., 1998; Day et al., 1999; Zhang et al., 2000; Rybczyk et al., 2002; Brantley et al., 2008; Hunter et al., 2018). This practice benefits dischargers by reducing capital and operations and maintenance costs compared to conventional wastewater treatment (Day et al., 2004; Ko et al., 2012; Hunter et al., 2018a,b).

To understand the functioning of assimilation wetlands in south Louisiana, it is necessary to put them into the context of the Mississippi River Delta (MRD) where they are located. The MRD is one of the largest coastal ecosystems in the world, covering about 25,000 km^2 of shallow open waters, wetlands and low relief uplands. The MRD was formed by a series of overlapping delta lobes as the river occupied different distributary channels that delivered water and sediments over the broad deltaic plain (Roberts, 1997; Blum and Roberts, 2012). These distributaries were elevated ridges that separated sub-basins of the deltaic plain. The MRD is characterized by a series of vegetation zones along a salinity gradient, with saline marshes at the coast that grade to brackish and freshwater marshes and swamps in the interior parts of the delta. The distributary ridges and barrier islands formed a skeletal framework that protected fresh interior parts of the delta from direct marine influences. Prior to hydrologic modifications, river water entered the coastal zone via the main river channels, as well as by overbank flow during high water (Day et al., 2007).

Beginning in the 18th century, and greatly accelerating throughout the 20th century, the delta was impacted by a variety of human activities such as levee construction and closure of distributaries, massive hydrological alterations, impoundments, and barrier island losses. Flood control levees built during the last two centuries separated the Mississippi River from most of the deltaic plain, preventing seasonal flooding and inputs of freshwater, nutrients, and sediments to the

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Table 1
Characteristics of wastewater treatment plants and assimilation and other wetlands discussed in this paper.

Location in Louisiana	Date discharge into wetland began	Date received wetland assimilation LPDES Permit	WWTP[1] design capacity (MGD)[2]	Type of wastewater treatment	Freshwater wetland type
Breaux Bridge	Late 1940s	1997	1.27	3 oxidation ponds with chlorination-dechlorination	Forested
Amelia	1973	2007	0.90	Oxidation pond with chlorination-dechlorination	Forested
Mandeville	1989	1998	4.00	3 aerated lagoon cells, 3-celled rock reed filter, UV[3] disinfection	Forested
Thibodaux	1992	1992	4.00	Aerated lagoon with high-rate trickling filter, UV disinfection	Forested
Riverbend	1978	Pending	1.00	Oxidation lagoon, UV disinfection	Forested and fresh to low salinity emergent
Bonnet Carré[3]	1933	Not applicable	Not applicable	Not applicable	Forested
LaBranche[4]	none	Not applicable	Not applicable	Not applicable	Forested and fresh to brackish emergent

1 WWTP = Wastewater treatment plant.
2 MGD = million gallons per day; liters per day = MGD*3.785.
3 Not an assimilation wetland. See discussion titled "Bonnet Carré Spillway and LaBranche Wetlands" for information.
4 Ultraviolet.

surrounding wetlands (Kesel, 1988, 1989; Mossa, 1996; Roberts, 1997; Day et al., 1990, 2000, 2007; Twilley and Rivera-Monroy, 2009). Other factors impacting the delta include altered hydrology due to the proliferation of dredged canals and deep-well fluid withdrawal associated with the oil and gas industry (Turner et al., 1994; Day et al., 2000; Morton et al., 2002; Chan and Zoback, 2007). These alterations have led to salt water intrusion, pervasive alteration of hydrology, and massive loss of about 25% of coastal wetlands. A contributing factor to wetland loss is the regional geologic subsidence (about 10 mm yr^{-1}) caused by compaction and consolidation of sediments that leads to increasing water levels. Prior to ecosystem modifications, soil accretion from riverine inputs of sediments and nutrients were able to compensate for subsidence (Day et al., 2007).

These anthropogenic changes in the delta have strongly impacted coastal freshwater wetlands (Day et al., 2004; Hunter et al., 2018). Because of the high relative water level rise, almost all baldcypress forests in the coastal zone are permanently flooded, which prevents seed germination (Shaffer et al., 2009, 2016). Most of the assimilation wetlands discussed here are threatened by saltwater intrusion and rising water levels, and all have been hydrologically altered due to canals, spoil banks, embankments due to roads or other features that isolate these areas and block upland runoff from entering the sites (Shaffer et al., 2018). In this paper, these conditions are considered when interpreting results.

Scientists in Louisiana began working with the U.S. Environmental Protection Agency (USEPA), Louisiana Department of Environmental Quality (LDEQ), and municipalities in 1988 to develop a framework for design, permitting, and monitoring of assimilation wetlands (Ko et al., 2012; Hunter et al., 2018). Municipal effluent is treated to secondary standards and disinfected prior to discharge into wetlands. Because nutrient concentrations are reduced as effluent moves through a wetland, this process could be considered tertiary treatment, although currently most wastewater treatment facilities in Louisiana are not required to meet tertiary standards. The first city to receive a Louisiana Pollutant Discharge Elimination System (LPDES) permit to discharge treated effluent into a wetland was Thibodaux in 1992. Since then, ten assimilation wetlands have been permitted and another four are under permit review (Hunter et al., 2018). It is important to note that some of the assimilation wetlands that received LPDES permits were discharging treated effluent into the wetlands for many years prior to being permitted. All of the wetlands receiving treated effluent have been hydrologically altered by some combination of levees, spoil banks, highways, oil and gas access roads, or railroad lines. In addition, prior to the permitting of assimilation wetlands, treated effluent was discharged directly into open water bodies, negatively affecting regional water quality (Day et al., 2004).

When considering discharge of treated effluent into wetlands in coastal Louisiana, it is important to determine if this practice benefits these wetlands in the long-term (i.e., over multiple decades). This study reviews data and observations from five wetlands that have received treated effluent from 26 to 70 years. These sites have been studied extensively and are discussed in more detail in published manuscripts (Day et al., 2012, 2004; Hunter et al., 2009a,b, 2016; Lane et al., 2015; Shaffer et al., 2015; and references cited in these papers), and in a recent review (Hunter et al., 2018a). This study also investigates two adjacent forested wetlands, one that receives periodic inflow from the Mississippi River and one that does not. Monitoring data and observations from these sites provide insight into how long-term nutrient loading impacts coastal wetlands. The objectives of this study were to describe these wetlands, to review available data after decades of operation, and to discuss how ecosystem monitoring has provided information for adaptive management and an in-depth understanding of these systems.

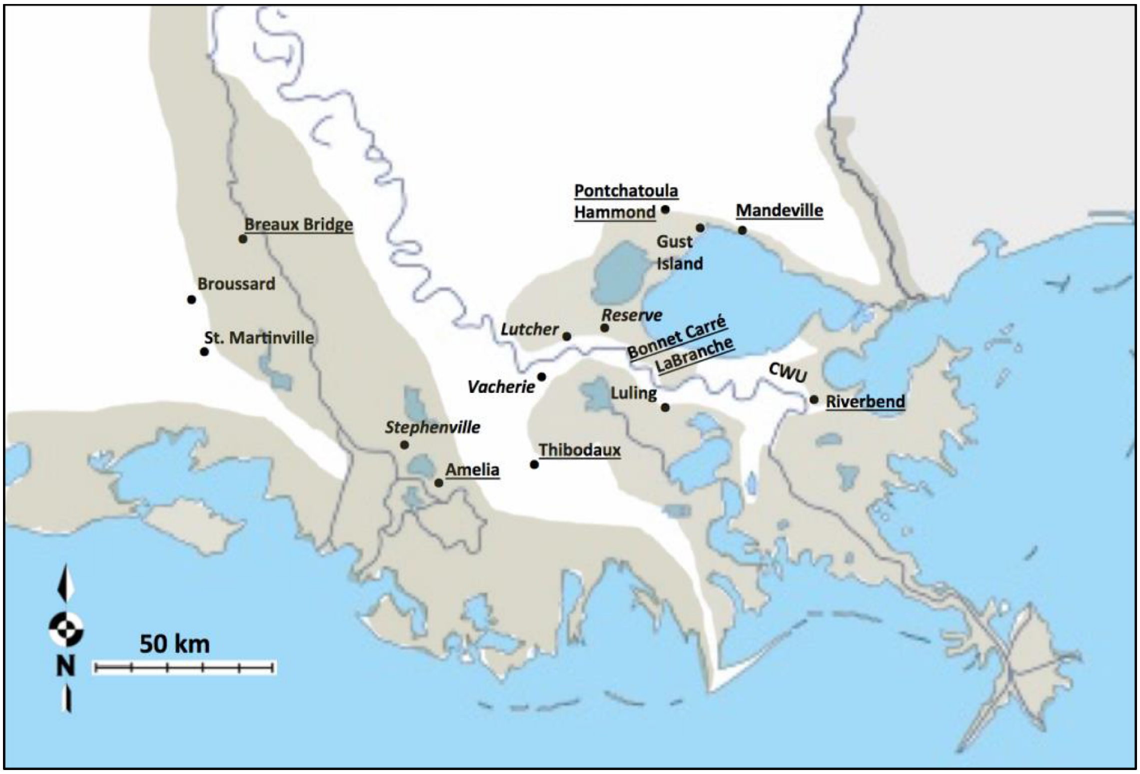


Fig. 1. Location of permitted and potential wetland assimilation projects in coastal Louisiana. Those underlined are discussed in this paper, those in italics indicate recently completed or ongoing ecological baseline studies, and the remainder are active assimilation wetlands. Shading indicates areas with a high proportion of wetlands (modified from Hunter et al., 2018a). Note: Bonnet Carré and LaBranche wetlands are not assimilation wetlands but are included in this discussion to illustrate impacts of periodic long-term nutrient inputs on wetlands.

2. Description of long-term assimilation wetlands

Four municipalities in coastal Louisiana discharged treated effluent into adjacent wetlands for many years prior to LPDES permitting for assimilation wetlands; these include Breaux Bridge, Amelia, Mandeville, and Riverbend (Table 1; Fig. 1). The assimilation wetland for the city of Thibodaux was the first permitted assimilation wetland and it did not discharge to a wetland prior to being permitted. Other municipalities in coastal Louisiana, however, discharge directly or indirectly into wetlands but are not recognized or permitted as assimilation wetlands. One example is the city of Pontchatoula, LA wastewater treatment plant that has discharged into adjacent wetlands for over 50 years. We also discuss the Bonnet Carré spillway wetlands that receive large intermittent flow from the Mississippi River and the LaBranche wetlands located across the eastern spillway levee that receive little fresh water inputs other than precipitation (Fig. 1). Investigation of these wetlands provides better understanding of the impacts of nutrient-rich water flowing into wetlands.

2.1. Breaux Bridge

During the late 1940's, when a city-wide sewage collection system and wastewater treatment plant were first constructed, the city of Breaux Bridge began discharging secondarily treated municipal effluent into a 1470 ha swamp dominated by water tupelo (*Nyssa aquatica*), baldcypress (*Taxodium distichum*), and swamp red maple (*Acer rubrum* var. *drummondii*). LDEQ first issued the city a wetland assimilation LPDES permit in 1997. The permit mandates that the City monitor vegetation species composition and productivity, surface water, soil and vegetation nutrients and metals concentrations, sediment accretion, and hydrology of the assimilation wetlands (Day et al., 2004; Hunter et al., 2018a). The city uses a 3-stage oxidation pond and chlorination-dechlorination disinfection system with the capacity to treat 3785 m^3/day (1.0 million gallons per day (MGD)) prior to discharge to the wetlands. Monitoring of assimilation wetlands is typically conducted near the discharge pipe (termed the discharge site), where the surface water exits the assimilation wetland (termed the out site), and a site between the discharge and out sites (termed the mid site; Hunter et al., 2018a). A nearby reference wetland that is not impacted by the effluent is also monitored for comparison (Fig. 2).

2.2. Amelia

The city of Amelia began discharging treated municipal effluent into a continuously flooded swamp located south of Lake Palourde in 1973. However, the assimilation wetland did not receive an LPDES permit until 2007. Amelia uses a 3-stage aerated lagoon and chlorination-dechlorination disinfection system and an average of 3000 m^3/day (0.8 MGD) is discharged to the wetlands. Water depths in the swamp located between the oxidation pond and Lake Palourde average about 40 cm and the wetland never completely drains. The region is impacted by coastal water level fluctuations, but because it is over 40 km from the coast, daily tidal water level variations are at most only a few cm (Day et al., 2006). Approximately 77 ha of forested wetlands are in the direct path of effluent, with more impacted to east and west as water fans out as it flows to Lake Palourde. The ~77 ha area directly affected by the effluent flow is within a larger forested wetland of over 1000 ha. The dominant vegetation is baldcypress, water tupelo, black willow (*Salix nigra*), swamp red maple, and green ash (*Fraxinus pennsylvanica*). Because the forest does not have a completely closed canopy and sunlight can penetrate, there is floating aquatic vegetation present, which is dominated by mosquito fern (*Salvinia minima*), duckweed (*Lemna minor*), watermeal (*Wolffia* sp.), pennywort (*Hydrocotyl ranunculoides*), and water lettuce (*Pistia stratiotes*). Submerged aquatic vegetation is

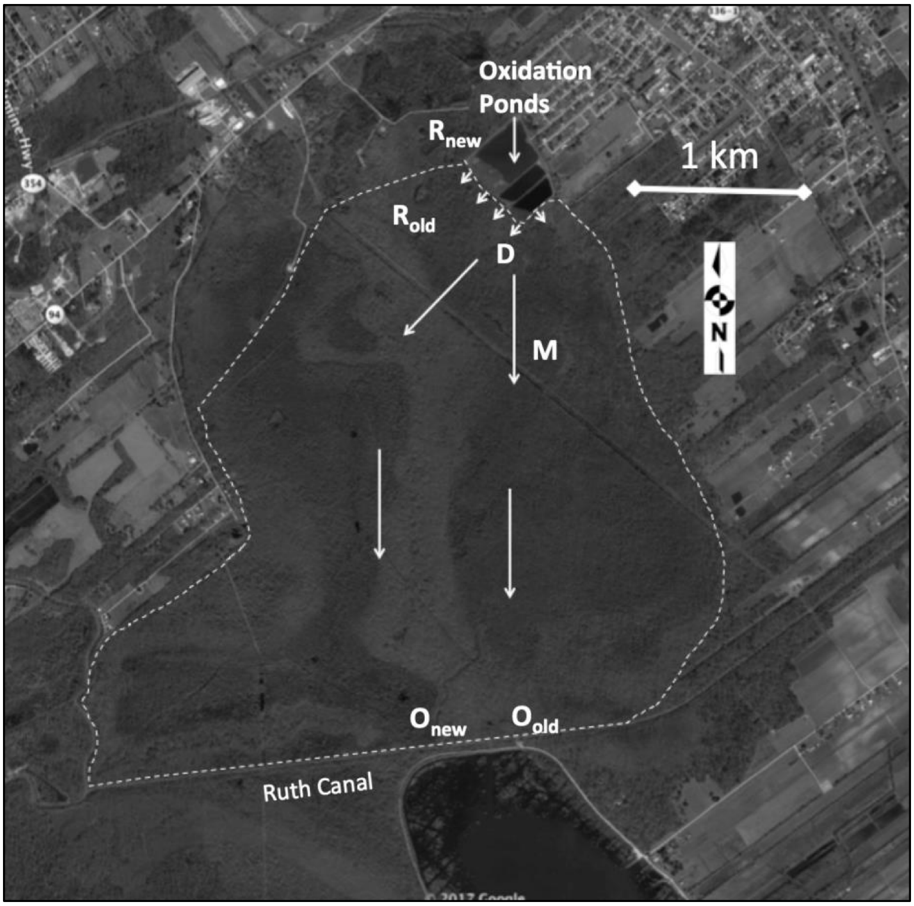


Fig. 2. Location of wetland monitoring sites at the Breaux Bridge assimilation wetland. R = Reference site, D = Discharge site, M = Mid site, and O = Out site. The small arrows indicate discharge locations, the large arrows indicate water flow, and the dashed line delineates the boundary of the assimilation wetland. The Reference site was moved after the permit was issued because the number of discharge points increased and effluent began to enter the old Reference site.

predominantly hornwort (*Ceratophyllum* sp.).

Three monitoring sites are aligned along the flow path of surface water discharged into the wetland to where the water enters Lake Palourde. A reference monitoring site is located nearby in the same swamp but outside the influence of the effluent (Day et al., 2006; Fig. 3). Prior to permitting, only one outlet pipe discharged treated water into the wetland but after the project was permitted, five outlets were constructed along the northern and western border of the large oxidation pond to increase the area of wetlands over which the treated water flowed.

2.3. Riverbend

The Riverbend oxidation pond began discharging treated municipal effluent in 1978 into the western part of the Central Wetland Unit (CWU; Fig. 4). Effluent was discharged into a drainage canal where, along with runoff pumped by the Gore pumping station, it was pumped into a relic baldcypress swamp and a marsh dominated by *Spartina patens* and *Spartina alterniflora*. An LPDES permit application was submitted in 2015, but has not yet been issued as of the publication date of this paper. The oxidation pond currently discharges about 1900 cubic meters per day (0.5 MGD) of secondarily treated municipal effluent directly to the wetlands. The treatment system is a one-cell aerated lagoon with a UV disinfection system.

It is important to put the effluent discharge at Riverbend into the context of the history of the CWU (Hunter et al., 2016). The CWU consists of about 12,000 ha of wetlands and open water east of New Orleans. Prior to 1965, the CWU contained nearly 6000 ha of freshwater forested wetlands and extensive fresh and low salinity emergent wetlands. Salinities in the area ranged from fresh to about 3 practical salinity units (psu). These wetlands provided an important buffer protecting the New Orleans area from storm surge. The construction of the Mississippi River Gulf Outlet (MRGO) shipping channel in the mid 1960s severed the Bayou La Loutre ridge, an old distributary of the Mississippi River that served as a natural barrier to saltwater intrusion from the Gulf of Mexico into the CWU and Lake Borgne. This breach led to saltwater intrusion that killed thousands of hectares of freshwater forested and emergent wetlands in the CWU (Shaffer et al., 2009). The MRGO was closed by a rock dam in 2009 (Hunter et al., 2016), and the area is now primarily brackish marsh and open water that has declined in salinity since the closure.

One of the few remaining stands of baldcypress that survived the opening of the MRGO is located in the area directly impacted by discharge from the Gore pumping station and Riverbend oxidation pond. This regular freshwater discharge prevented the excessive soil salinities that killed baldcypress in most other areas of the CWU and is the primary reason that baldcypress are still alive in this area (Hunter et al., 2016; Shaffer et al., 2018).

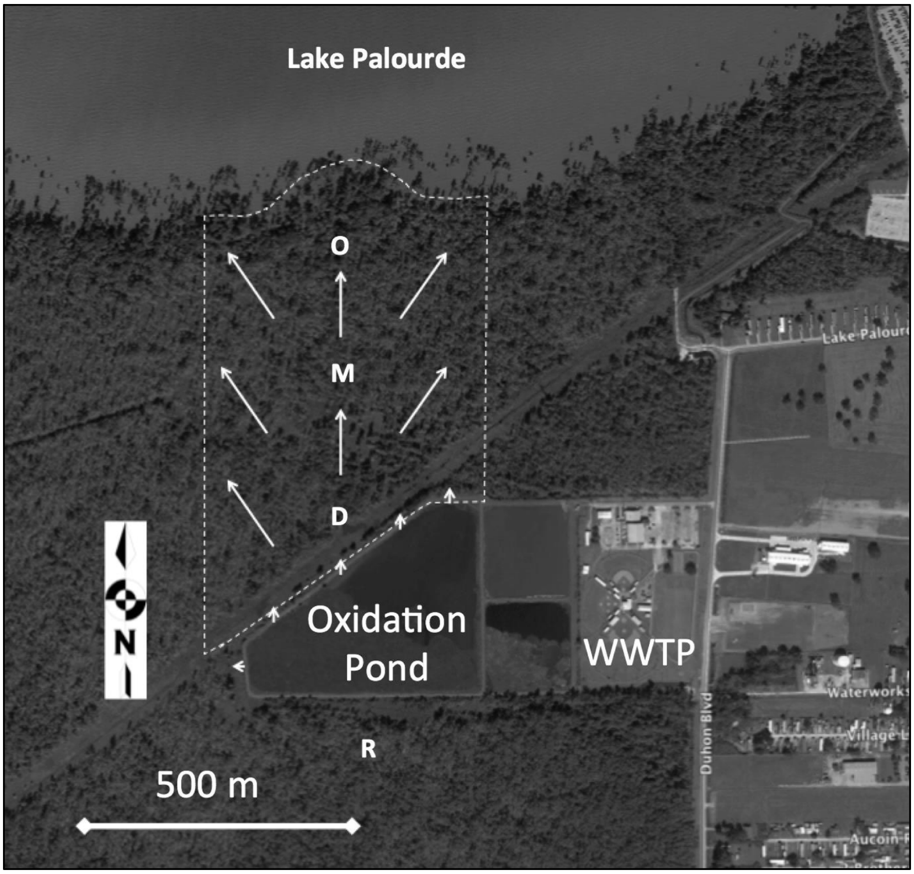


Fig. 3. Monitoring sites at the Amelia assimilation wetland. R = Reference, D = Discharge, M = Mid, O = Out. The small arrows indicate the general location of discharge pipes, the large arrows indicate water flow, and the dashed line delineates the area directly impacted by the discharge, but treated effluent flows over a larger area to the east and west.

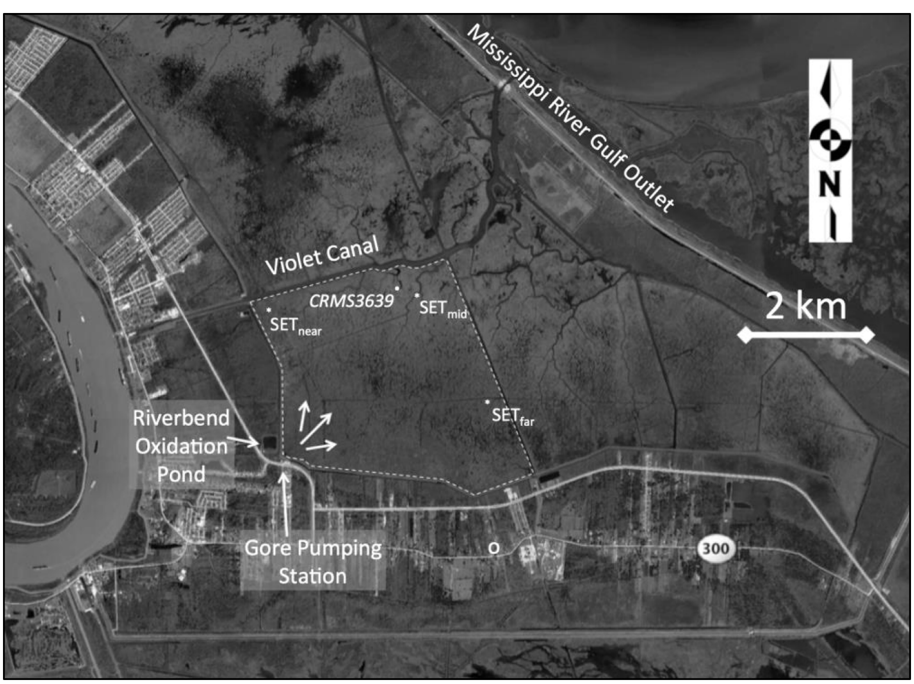


Fig. 4. Location of the Riverbend Oxidation Pond and Gore Pumping station in the southwestern Central Wetlands Unit (CWU). Sites identified with SET are where wetland surface elevation change and accretion were measured. The white arrows indicates where effluent was pumped into the wetland.

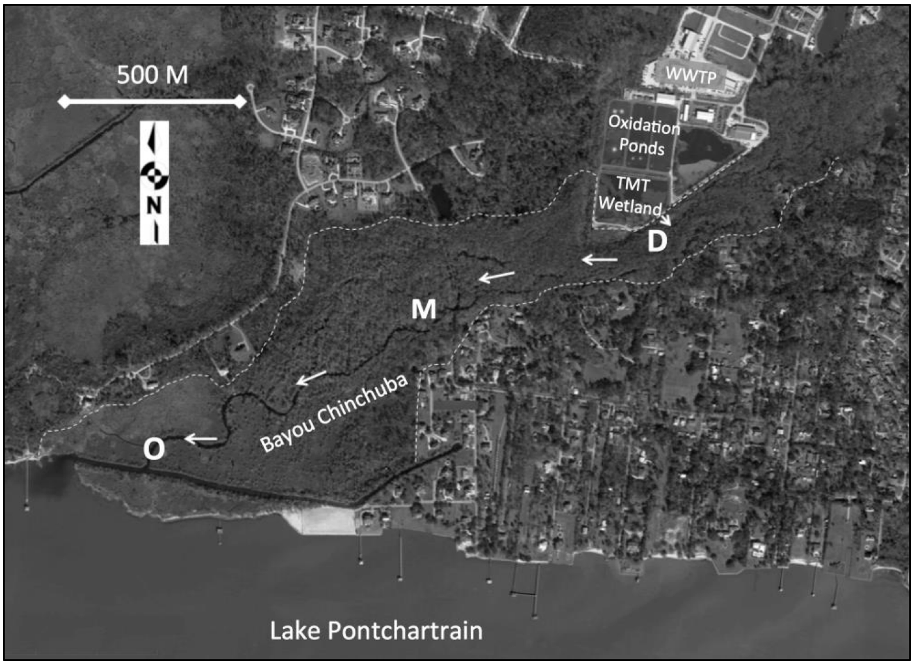


Fig. 5. Location of monitoring sites at the Bayou Chinchuba (BC) assimilation wetlands. D = Discharge site, M = Mid site, O = Out site. The small arrow indicates the discharge location, the large arrows indicate water flow, and the dashed line delineates the boundary of the assimilation wetland. The TMT Wetland is a constructed wetland designed to reduce ammonia levels. The reference site, Bayou Castine, is a similar waterway about 3 km to the east (Brantley et al., 2008).

In an ecological baseline study carried out in 2011, surface water salinities in the vicinity of the Riverbend and Gore discharges were between 0 and 6 psu (Hillmann et al., 2015). Interstitial soil salinity ranged between about 4 and 8 psu in 2011 in the CWU, but Hillmann et al. (2015) recorded soil salinities < 2 psu in areas surrounding each of the pumping stations in the CWU. Other data show that soil salinity in some areas has dropped below 2 psu since 2013 (G. Shaffer, unpublished data). After soil salinities were consistently measured below 2 psu, 3000 baldcypress seedlings were planted in 2014 near the Gore pumping station in an area affected by the discharge and a recent survey found the trees to be alive and thriving.

Vegetative species richness is low in the CWU and throughout the area an unstable marsh platform has developed over a matrix of about 2 million dead baldcypress and water tupelo trunks located just below the soil surface. Soil bulk density ($0.49\,g\,cm^{-3}$), soil strength ($> 4\,kg\,cm^{-2}$), and belowground biomass (> 4000 g dry weight m^{-2}) were greater at the area receiving discharge from Riverbend and Gore than at any of the other study sites ($\sim 0.25\,g\,cm^{-3}$, $< 2.5\,kg\,cm^{-2}$, and $< 2800\,g\,m^{-2}$; Hunter et al., 2016). The lowest bulk density and soil strength occurred at sites with low vegetation biomass that were degrading to open water. Aboveground herbaceous biomass ranged from 1500 to 2000 g dry weight m^{-2} (Hunter et al., 2016) and is limited compared to other coastal marshes in Louisiana (Hopkinson et al., 1978; Day et al., 2013). Belowground live biomass ranged from about 1000 to 4000 g dry weight m^{-2} (Hunter et al., 2016), whereas healthy herbaceous marsh generally has 8000 to 10,000 g dry weight m^{-2} of belowground biomass (Day et al., 2012, 2013; Shaffer et al., 2015).

2.4. Mandeville

The city of Mandeville received an LPDES permit to discharge treated effluent to the Bayou Chinchuba swamp in 1989 (Brantley et al., 2008). The system was expanded in 2009 to discharge to the Tchefuncte marsh to provide more assimilative capacity. Bayou Chinchuba and Bayou Castine (the Reference site; Brantley et al., 2008) are freshwater forested wetlands that are dominated by water tupelo, baldcypress, and swamp blackgum (*Nyssa biflora*; Fig. 5). The hydrology of the Bayou Chinchuba and Bayou Castine swamps is strongly influenced by rainfall and flooding from Lake Pontchartrain (i.e., during southeasterly winds that raise water levels in the lake). The area immediately downstream of the wastewater treatment facility does not have a well-defined channel, and flow spreads over the surface of the surrounding forested wetlands (Brantley et al., 2008). Mandeville uses a 3-stage aerated oxidation pond, a constructed wetland to lower ammonia levels, and a UV disinfection system and discharges an average of $7196\,m^3$/day (1.9 MGD) to the assimilation wetlands.

2.5. Thibodaux

Although not the oldest in operation, the Thibodaux assimilation wetland was the first to be permitted in Louisiana in 1992. The city uses a trickling filter and UV disinfection and discharges an average of $11{,}546\,m^3$/day (3.0 MGD). The Thibodaux assimilation wetland is a 231-ha subsiding baldcypress-water tupelo swamp on the southern slope of Bayou Lafourche, a former distributary of the Mississippi River that was cut off from the river in 1904 (Fig. 6). The wetland is part of a much larger degraded swamp where the hydrology has been substantially altered since the 19th century due to creation of drainage canals and associated spoil banks and road construction (Conner et al., 2014). Prior to the construction of flood control levees, the area was a swamp that received frequent overflow from Bayou Lafourche (Blum and Roberts, 2012). Today, flood control levees prevent riverine input, and water levels are controlled by rainfall and stormwater runoff from the surrounding uplands as well as coastal water levels. When effluent discharge began in 1992, the Thibodaux assimilation wetland was a fundamentally altered system that was isolated and semi-impounded with permanent standing water, stagnant conditions, and a dying bottomland hardwood system.

Water levels at the assimilation wetland are impacted by coastal water levels, with RSLR in the delta now exceeding $1.0\,cm\,yr^{-1}$ (Conner and Day, 1988a; Conner et al., 2014). In the past 25 years, water levels have risen about 27 cm and the bottomland hardwood community that existed on the site died due to permanent flooding. The number of baldcypress trees has decreased over time due to the loss of adult trees, mainly during hurricanes (Rybczyk et al., 1995), and lack of seed germination (Conner et al., 2014). The area immediately adjacent to the effluent discharge pipe was a shallow, treeless, open water area cleared for the construction of a power line right of way and, over time, transitioned into a floating marsh community approximately 10 years after discharge of treated effluent began (Izdepski et al., 2009). Because

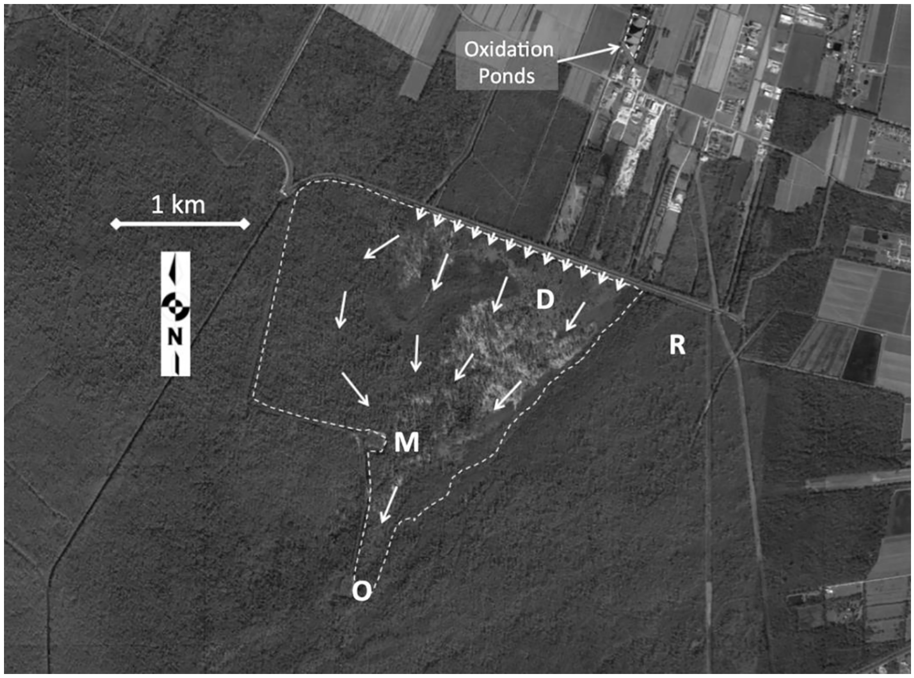


Fig. 6. Location of monitoring sites at the Thibodaux assimilation wetland. R = Reference site, D = Discharge site, M = Mid site, O = Out site. The small arrows indicate discharge locations, the large arrows indicate water flow, and the dashed line delineates the boundary of the assimilation wetland.

this site was the first to be permitted, it has been one of the most intensively studied of the assimilation wetlands.

2.6. Bonnet Carré spillway and LaBranche wetlands

The Bonnet Carré spillway and the LaBranche wetland are examples of two adjacent yet contrasting wetland sites (Fig. 7). The Bonnet Carré spillway has received episodic inputs of large volumes of Mississippi River water while the LaBranche wetland is isolated from significant freshwater or nutrient input. The Bonnet Carré Spillway was completed in 1933 in response to the great flood of 1927 (Day et al., 2012) to decrease Mississippi River stage to minimize flooding threat in New Orleans. The 3.4 km wide spillway is confined by two 8.6 km long levees running from the Mississippi River to Lake Pontchartrain. A water flow regulation structure consisting of 350 floodgates is located at the Mississippi River inlet. Approximately half of the total spillway is forested wetlands (1300 ha; Lane et al., 2001). From 1931 to 2018, the spillway has been opened twelve times during high water events of the Mississippi River, with flows ranging from 3100 to 9000 $m^3 s^{-1}$ (Day et al., 2012). When open, water flows through the spillway into Lake

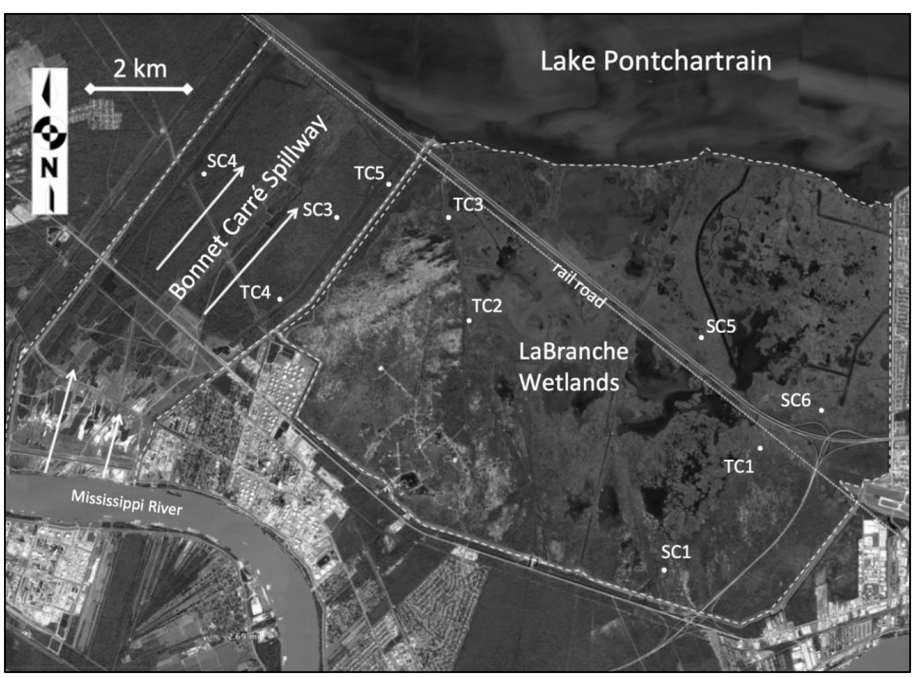


Fig. 7. Location of the Bonnet Carré spillway and LaBranche wetland, along with location of sites where soil cores (SC) and tree cores (TC) were collected. The arrows indicate water flow and the dashed lines delineate the boundary of the wetlands (see Day et al., 2012 for more detail).

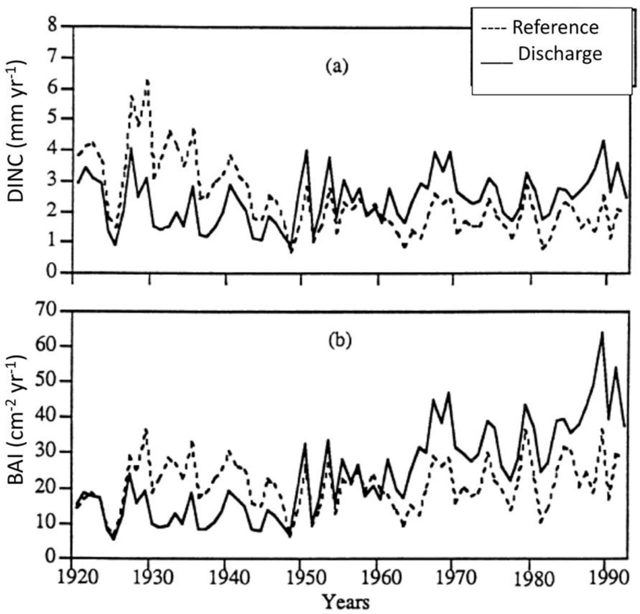


Fig. 8. Mean annual tree ring width chronologies of diameter increment (DINC; top); and basal area increment (BAI; bottom) for baldcypress at the Breaux Bridge assimilation swamp at the Control (Reference) and Treated (Discharge) sites. Discharge of treated effluent began in the late 1940's (from Hesse et al., 1998).

Pontchartrain and then to the Gulf of Mexico. With each opening of the Bonnet Carré spillway, the river deposits an average of about 5 million m^3 of sediment in the spillway, consisting mostly of silts and sands (Lane et al., 2001).

The 8100 ha LaBranche wetland, adjacent to the Bonnet Carré spillway, is composed primarily of non-regenerating baldcypress-water tupelo swamp and fresh herbaceous wetlands near the river, transitioning to brackish marsh and shallow open water ponds closer to Lake Pontchartrain. The wetland borders Lake Pontchartrain to the north while the rest of the area is confined by flood control levees (Cramer et al., 1981; Boumans et al., 1997). The major factors contributing to the deterioration of the LaBranche wetlands are isolation from riverine input by the Mississippi River levees, construction of a railroad levee and several canals that semi-impound the area, erosion, saltwater intrusion during hurricanes and droughts, nutria herbivory, subsidence, and sea-level rise (Pierce et al., 1985; Day et al., 2012). In its natural state, there was regular riverine input and flow of fresh water through the LaBranche wetland. Salinities in the LaBranche wetland can be greater than 10 psu during drought, as occurred during the 2000–2001 regional drought (Shaffer et al., 2009; Day et al., 2012). As a result of these changes, much of the interior wetland converted to open water and many baldcypress and water tupelo trees died during the last half-century (Pierce et al., 1985; Day et al., 2012).

3. Ecological functioning of long-term sites

3.1. Long-term wetland productivity

Analysis of annual tree growth rings can provide information about how environmental conditions affect tree growth. At the Breaux Bridge assimilation wetland, tree rings were analyzed at two sites (one where effluent was discharged and one reference area) to determine impacts of treated effluent on growth patterns. After effluent discharge to the swamp began in the late 1940's, growth rates were consistently higher at the discharge area compared to the reference area (Fig. 8; Hesse et al., 1998). However, the large degree of variability in the data shows that wetland productivity varies considerably with environmental conditions. For example, growth of baldcypress is generally less during periods of warm spring weather and drought (Stahle and Cleaveland, 1992; Cleaveland, 2000; Keim and Amos, 2012). Day et al. (2012) reported that the climate variables that most strongly impacted growth rates of baldcypress in the Lake Pontchartrain basin were precipitation and temperature during April and May.

Keim et al. (2012) evaluated the combined effects of rising water levels and municipal effluent discharge on baldcypress growth at the Thibodaux assimilation wetland using tree ring analysis. Baldcypress trees at the discharge, outflow, and adjacent reference areas (shown in Fig. 6) had increased growth that coincided with widespread rapid subsidence and water level increases in the late 1960s because increasing inundation eliminated bottomland hardwood species and decreased competition for light and nutrients (Fig. 9; Conner and Day, 1988a; Conner et al., 2014). Tree growth at the discharge and outflow sites began to decrease before effluent discharge began in 1992, and then afterward was apparently not significantly affected by the effluent. In contrast, trees at the reference site, which is about 10 cm higher than the Treatment site, did not show similar declines in growth. But Keim et al. (2012) concluded that increasing regional water levels will lead to reduced growth as the reference site experiences increasing levels of inundation. At the treatment site, changes in hydrology caused by subsidence have reduced any positive effect of the treated effluent on long-term woody growth. The growth rates of baldcypress at the Thibodaux assimilation wetland, however, are higher than those recorded in the Maurepas swamp and the LaBranche wetland, which are also permanently flooded but have very sluggish water movement and low nutrient input (Day et al., 2012; Shaffer et al., 2016).

Tree ring analysis was also used to compare long-term growth of baldcypress trees in the Bonnet Carré and LaBranche wetlands (Day et al., 2012). Baldcypress in the Bonnet Carré spillway grew almost twice as fast as those in the LaBranche wetlands after the spillway began operating (Fig. 10). Rain and flowing water increase oxygen to the root zone when compared to stagnant water (Davidson et al., 2006) and typically increase nutrient loading from non-point source runoff (Shaffer et al., 2009). The openings of the Bonnet Carré spillway in 1937, 1945, 1950, 1973, 1975, 1979, 1983, 1997, and 2008 coincided with years of enhanced productivity for both the Bonnet Carré and LaBranche wetlands, but the enhanced growth was less for the LaBranche wetlands. The openings occurred during times of wet weather. Fresh water from the spillway decreases saltwater intrusion and increases sediment deposition in the Bonnet Carré wetlands compared to the LaBranche wetlands. These tree ring studies show a common response of baldcypress both to permanent flooding and to introduction of freshwater with nutrients.

Tree productivity at the Breaux Bridge assimilation wetland has been estimated annually since 2002 from measurements of diameter-at-breast-height (dbh) increase and litterfall as part of the LPDES permit monitoring requirements. Because of differences in tree densities at the monitoring sites, productivity was calculated on a per tree basis. Mean annual net primary productivity (NPP; woody growth and litterfall combined) at the discharge and reference wetlands ranged from 30 to 135 $g\,m^{-2}\,yr^{-1}$ per tree from 2002 to 2015 (Fig. 10). Litterfall is important because it increases soil building and helps counteract regional subsidence. In general, litterfall productivity was a larger percentage of NPP than woody productivity and was greater at the discharge site than the reference site during most of the years monitored (Fig. 11).

At the Amelia assimilation wetland, tree productivity has been measured annually at the discharge and reference sites since 2007. Like the Breaux Bridge wetland, litterfall was typically a greater percentage of NPP than woody growth and significantly greater at the discharge site (717 $g\,m^{-2}\,yr^{-1}$) compared to one of the reference sites (412 $g\,m^{-2}\,yr^{-1}$). Woody growth ranged from 302 to 776 $g\,m^{-2}\,yr^{-1}$ and was not statistically different between the reference and discharge

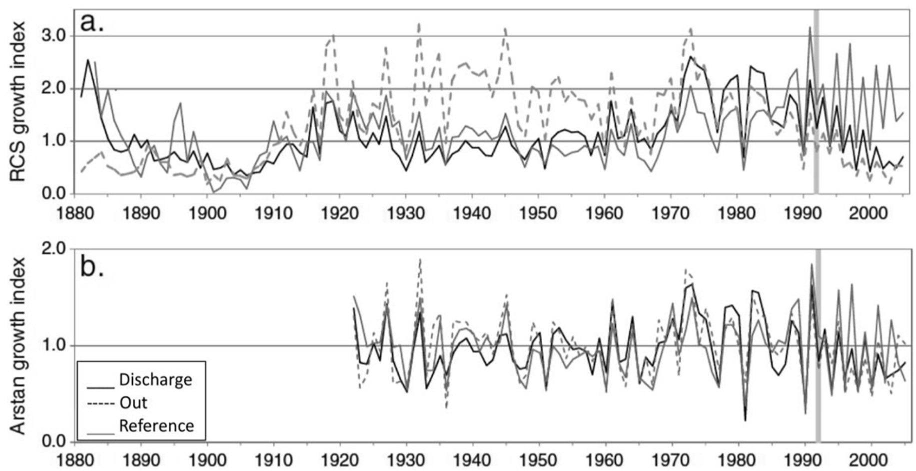


Fig. 9. Tree ring chronologies from the Thibodaux assimilation wetland. The vertical line indicates the onset of effluent addition to the treatment site in 1992 (from Keim et al., 2012). RCS (regional curve standardization) and Arstan growth index describe programs used to detrend tree ring width data to generate growth indices.

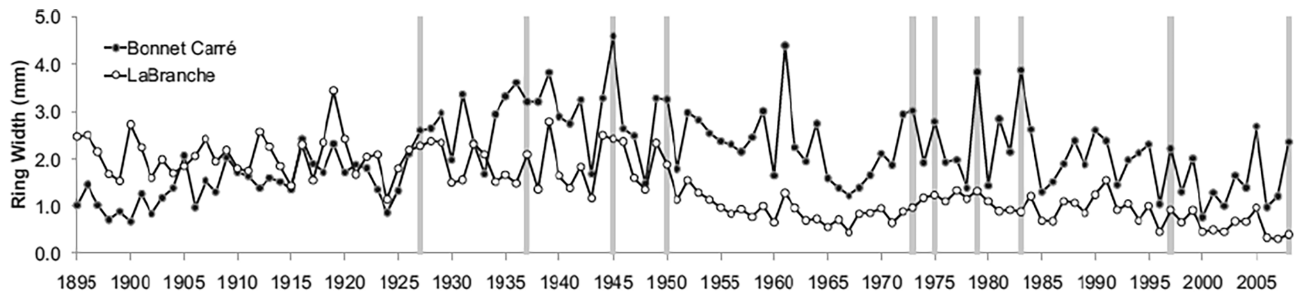


Fig. 10. Tree ring widths at the Bonnet Carré and LaBranche wetlands. Each value is the mean across sampling locations within each site. Vertical gray bars indicate years when the Bonnet Carré spillway was opened. The first grey bar denotes the exceptional 1927 flood that led to the construction of the spillway (From Day et al., 2012).

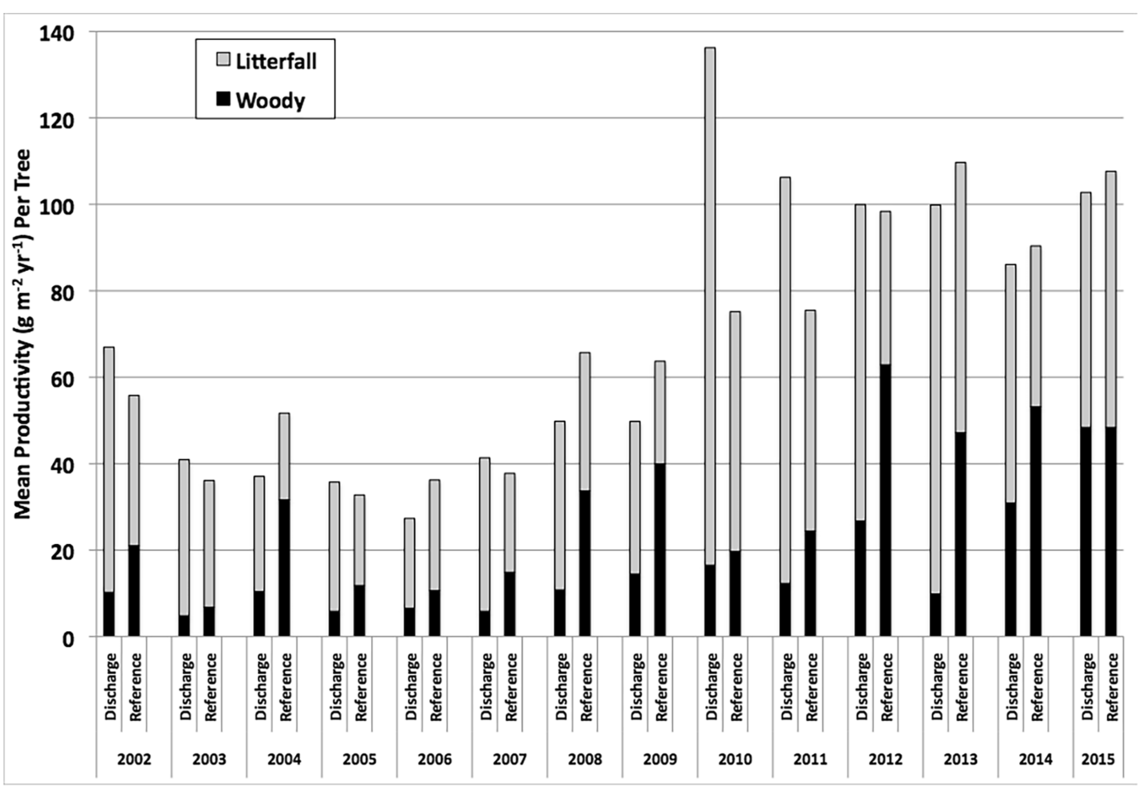


Fig. 11. Mean annual litterfall and woody production at the Breaux Bridge assimilation wetland discharge site and nearby reference site (From Hunter et al., 2018a).

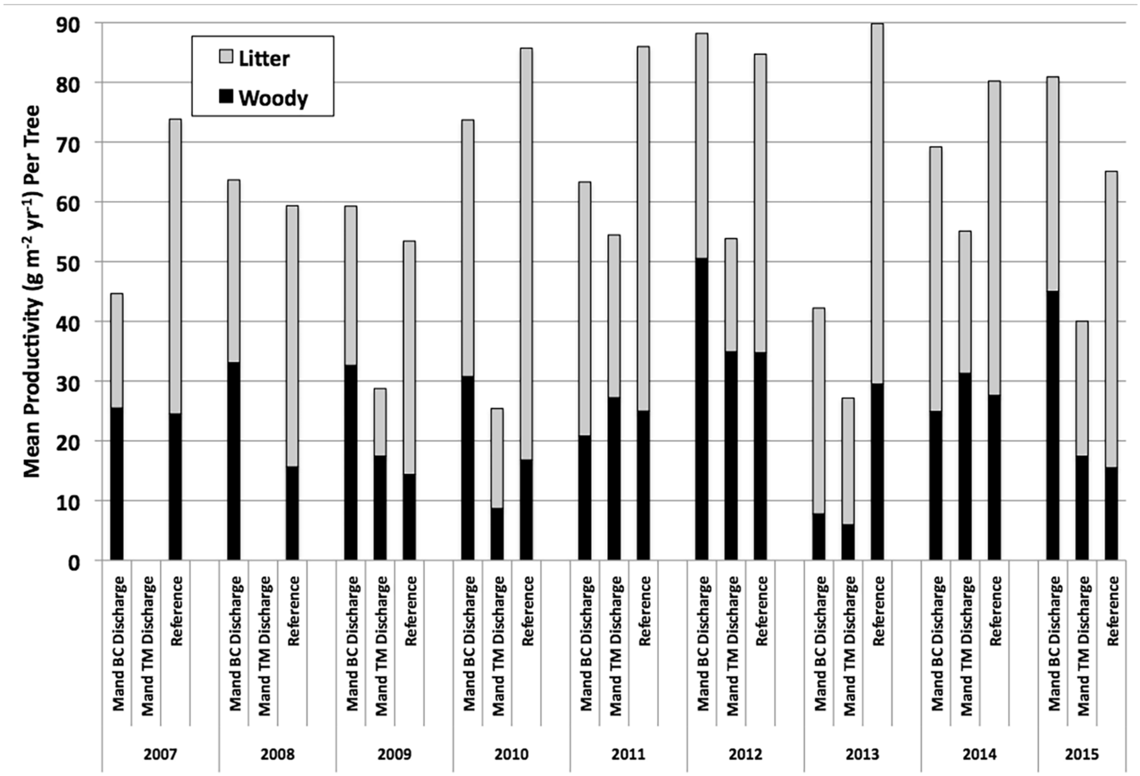


Fig. 12. Mean annual litterfall and woody production at the Mandeville Bayou Chinchuba (BC) and Tchefuncte Marsh (TM) assimilation wetlands (From Hunter et al., 2018).

sites. Total NPP at the discharge site (1467 g m^{-2} yr^{-1}) was significantly higher than at the reference site (714 g m^{-2} yr^{-1}).

Litterfall was also a greater proportion of NPP than woody growth at the Mandeville Bayou Chinchuba wetland (Fig. 12). Although NPP varied from year to year at this assimilation wetland, it was generally greater at the treatment site than at the reference site (Brantley et al., 2008).

At the Thibodaux assimilation wetland, switchgrass (*Panicum virgatum*) became established at the shallow open water area adjacent to the discharge pipe, and by 2003 began to extend into the wetland. Within four years of establishment at the site, a highly productive floating wetland had developed (Izdepski et al., 2009). The area nearest the effluent discharge had the greatest NPP (3876 g m^{-2} yr^{-1}) and total root biomass production (4079 ± 298.5 g m^{-2} yr^{-1}) compared to sites further away from the discharge along a 75-m transect.

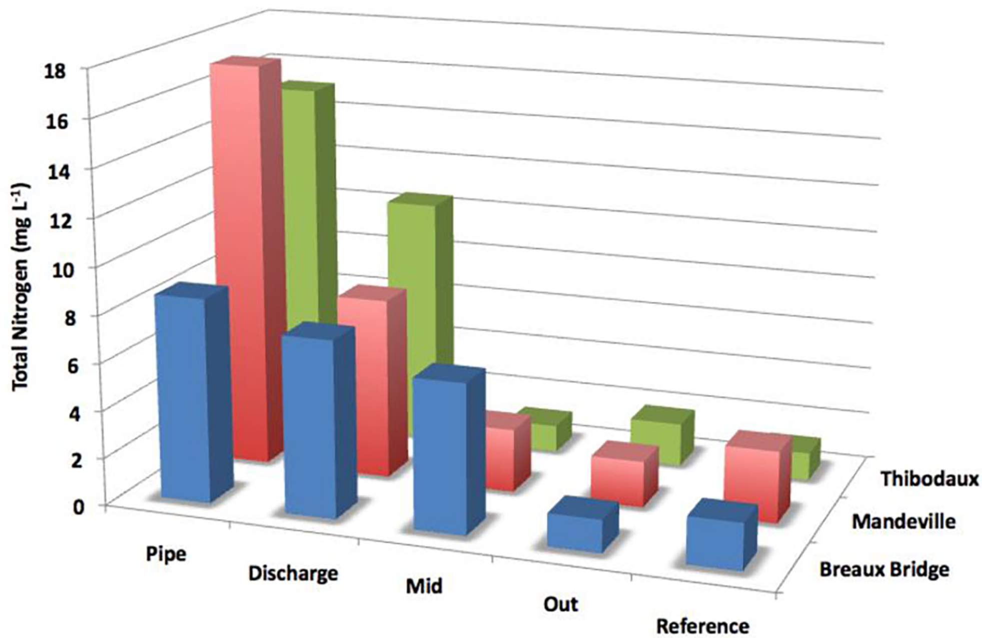
Forest productivity has also been measured at the Thibodaux assimilation wetland since 1992 and was less at the discharge site compared to the mid, out, and reference sites. The lesser productivity at the discharge site was due to sparse tree density because by 2014 almost all bottomland hardwood species had died (Minor, 2014), a process that had started before effluent was discharged into the wetland. If the floating marsh is taken into consideration, the productivity of the discharge site is significantly higher than at the other monitoring sites and the reference site.

3.2. Long-term nutrient removal

Wetlands improve water quality by chemically, physically, and biologically removing pollutants, sediments and nutrients from overlying surface waters (Day et al., 2004; Hunter et al., 2009a,b; Kadlec and Wallace, 2009; Vymazal, 2010; Shaffer et al., 2015). The basic principle underlying wetland assimilation is that the rate of effluent application must balance the rate of removal. The primary mechanisms by which this balance is achieved are physical settling and filtration, chemical precipitation and adsorption, and biological processes that result in burial, storage in vegetation, and denitrification (Reddy and DeLaune, 2008). Nitrogen in treated effluent can be removed permanently by denitrification, and both nitrogen and phosphorus can be removed by woody plant uptake, and burial (Kadlec and Wallace, 2009).

For wetland assimilation systems in Louisiana, typical loading rates for total nitrogen (TN) range from 2 to 20 g m^{-2} yr^{-1} and for total phosphorus (TP) from 0.4 to 3 g m^{-2} yr^{-1} (Day et al., 2004). Removal efficiencies for TN and TP at these loading rates average between 65 and 90%, while nitrate (NO_3) removal efficiency is between 90 and 100%. Long-term data show that assimilation wetlands continue to be nutrient sinks even after decades of effluent discharge, with both TN and TP reduced to background levels (Fig. 13). This is true for all assimilation wetlands in Louisiana (Hunter et al., 2018) indicating that nutrients are reduced to background levels even after decades of discharge. This is likely due to both organic matter accumulation in the subsiding coastal zone as well as long-term accumulation of woody material. Such accumulation has been demonstrated in the Everglades as well as in the Mississippi delta (Reddy et al., 1983; DeLaune et al., 1978; Hatton et al., 1983).

The form of nitrogen in treated effluent discharged into wetlands will impact nitrogen removal rates. In anoxic wetland soils, NO_3 is rapidly reduced to nitrous oxide (N_2O) and nitrogen gas (N_2) via denitrification, while ammonia (NH_3) must be nitrified to NO_3 before denitrification can occur (Reddy and DeLaune, 2008). As can be seen in Fig. 13, total nitrogen has a greater decrease between the pipe, discharge, and mid sites at the Mandeville and Thibodaux wetlands than at the Breaux Bridge wetland. Total nitrogen in treated effluent from the Breaux Bridge wastewater treatment facility averages 6% NO_3, 50% NH_3, and 44% organic nitrogen (ON). Total nitrogen in the treated effluent at the Mandeville facility averages 40% NO_3, 43% NH_3, and 17% ON while at the Thibodaux facility effluent total nitrogen averages 34% NO_3, 27% NH_3, and 39% ON (Hunter et al., 2018). The Mandeville



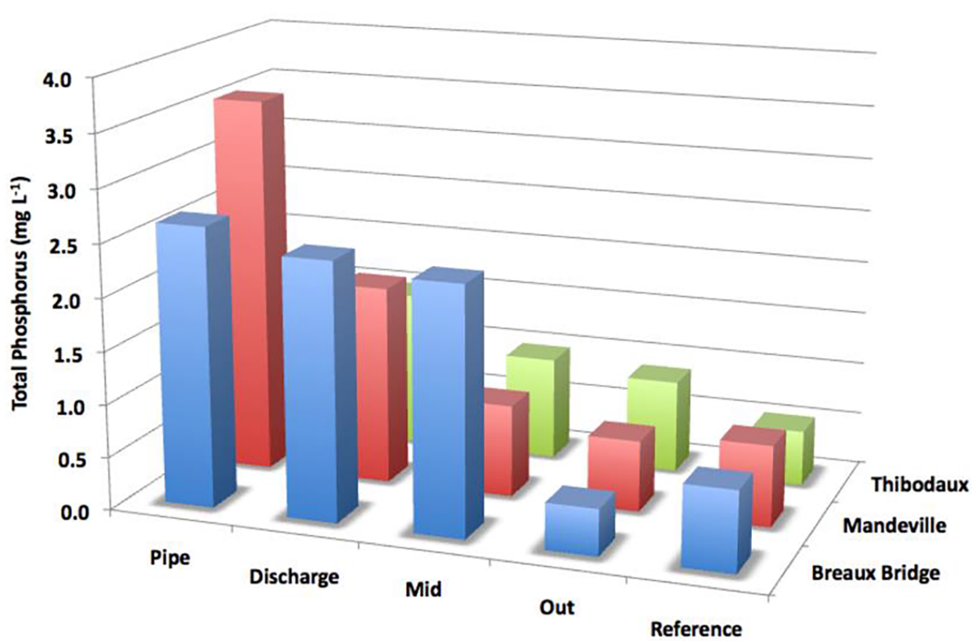


Fig. 13. Total nitrogen (top) and phosphorus (bottom) concentrations of surface waters at three assimilation wetlands. Values are means of 8, 10, and 14 years for Thibodaux, Mandeville, and Breaux Bridge, respectively.

and Thibodaux facilities aerate effluent to increase NO_3 concentrations of the effluent, and thus have higher removal rates of total nitrogen.

3.3. Long-term accretion

Coastal wetland elevation is directly influenced by a complex relationship between relative water level rise (lowering of elevation due to subsidence combined with eustatic sea-level rise) and accretion (vertical accumulation of material). Subsidence in some areas of the MRD is in excess of 10 mm yr^{-1} (Penland and Ramsey, 1990). Mean accretion rates ranged from 12.0 to 32.4 mm yr^{-1} at the assimilation wetlands discussed here and from 0.5 to 7.8 mm yr^{-1} at the reference wetlands (Table 2) demonstrating that the wetlands receiving the effluent are able to accrete at a rate higher than mean subsidence in the MRD plain. Similarly, the Bonnet Carré wetlands had higher mean accretion rate (26.5 mm yr^{-1}) than the LaBranche wetlands (4.3 and 11.4 mm yr^{-1} north and south of the railroad, respectively; Day et al., 2012). Accretion rates vary depending upon nutrient and sediment loading. Increased nutrient inputs to isolated wetlands increase vegetation productivity, which promotes sediment deposition and increases both aboveground and belowground organic matter accumulation (Baumann et al., 1984; DeLaune and Pezeshki, 2003; Lane et al., 2006; Brantley et al., 2008; Jarvis, 2010; Teal et al., 2012). Many studies have shown that organic matter accumulation drives sediment accretion in Louisiana marshes (DeLaune and Pezeshki, 2003; Turner et al., 2006; Craft, 2007; Neubauer, 2008) making it an important component of

Table 2
Mean accretion at wetland sites measured by feldspar marker horizons or Cs137.

Location	Site	No. of Samples	Mean accretion ($mm\,yr^{-1}$)	Standard error	Year installed	Year measured
Breaux Bridge	Discharge	10	12.0	0.2	2006	2017
Breaux Bridge	Reference	10	7.8	0.2	2008	2017
Mandeville Bayou Chinchuba	Discharge	10	12.1	0.2	2008	2017
Mandeville Bayou Castine	Reference	10	0.5	0.0	2006	2017
Thibodaux	Discharge	5	11.4		1988[A]	1994
Thibodaux	Reference	5	1.4		1988[A]	1994
Bonnet Carré		2	26.5		Cs^{137}	2009
LaBranche	South	2	4.3		Cs^{137}	2009
LaBranche	North	2	11.4		Cs^{137}	2009

[1]Although feldspar markers were installed prior to discharge, accretion reported in this table only includes measures made after discharge began in 1992. Breaux Bridge is from (Comite Resources Inc, 2017), Mandeville from Brantley et al. (2008), Thibodaux from Rybczyk et al. (2002) and Labranche from Lane et al. (2006).

restoration in the coastal zone (Hunter et al., 2018).

4. Faunal response

For the early permitted assimilation wetlands, monitoring for fauna (i.e., benthos and nekton) was required. These studies showed no significant differences between treatment and reference wetlands (Conner et al., 1989; Day et al., 1997; Day et al. 2000). Because of this, the requirement for faunal monitoring was dropped but subsequent studies show no detrimental impacts due to effluent discharge.

At the Amelia assimilation wetland, total individuals, total species, and species richness of macroinvertebrates were greatest near the outfall and declined away from the discharge (Day et al., 2006). There was increased abundance of benthic organisms at the station nearest the effluent discharge compared to sites further away. This is consistent with reports of enhanced abundance of benthic organisms in other effluent-impacted forested wetlands (Brightman, 1984; Kadlec and Alvord, 1989). The chironomid and oligochaete-dominated benthos at the near site at Amelia is typical of unimpacted benthos in Louisiana forested wetlands (Beck, 1977; Loden, 1978; Ziser, 1978; Sklar, 1983).

Minor (2014) studied differences in wetland macrofaunal assemblages between discharge and reference sites at the Thibodaux assimilation in 2013–2014 and found significant differences between the two sites. Taxa that occurred at both study sites included western mosquitofish (*Gambusia affinis*), least killifish (*Heterandria formosa*), red swamp crayfish (*Procambarus clarkii*), juvenile crayfish of family Cambaridae, and members of Noteridae family. Macrofaunal species composition and abundance differed between the discharge and reference wetlands primarily because of differences in surface water levels, dissolved oxygen concentrations, and vegetation community structure as well as because the reference site is hydrologically isolated and receives only rainfall input. Compared to results from Conner and Day (1988b), obtained from an ecological baseline study prior to discharge, aquatic macrofaunal assemblages observed by Minor (2014) at the discharge and reference sites indicate similarity to pre-effluent species composition.

Weller and Bossart (2017) reported on changes in benthic insect diversity at the Hammond assimilation wetland over multiple years of sampling, during which time an initially healthy marsh degraded to open water/mud flat due to grazing by the introduced rodent, nutria (*Myocaster coypus*) and subsequently partially revegetated after nutria were controlled (Shaffer et al., 2015). Insects are commonly used to monitor the health of streams and rivers but have rarely been used to study wetlands. During the three-year study, 3984 individuals were collected, representing 33 families and 86 species. Insect diversity tracked the overall condition of the marsh over time. Simpson's diversity was highest before degradation occurred, lowest at the height of degradation, and intermediate during the period of partial recovery. Species richness, however, was highest in the partially revegetated marsh community. Although this community included species characteristic of both the intact and degraded communities, it shared greatest affinity with the intact marsh. The dominant taxa present in these communities shifted from various beetles to chironomid flies and then back to beetles (Weller and Bossart, 2017).

5. Enhancement of degrading wetlands

The Federal Clean Water Act includes an antidegradation clause that prevents any activity that leads to the degradation of water quality. Some believe the antidegradation requirement is a reason to prohibit the use of natural wetlands for assimilation of treated municipal effluent. The assimilation wetlands in the Louisiana coastal zone have been highly degraded due to isolation from Mississippi River input, a subsiding environment, pervasive alteration of hydrology, and salt water intrusion. As shown by the studies discussed in this paper, discharge of treated municipal effluent has enhanced wetland productivity, reduced nutrients to background levels, and enhanced accretion. Thus, rather than degrading wetlands, discharge of treated effluent enhances wetlands.

6. Summary and conclusions

Secondarily treated and disinfected municipal effluent has been discharged to five coastal Louisiana wetlands for decades. The effluent generally has enhanced vegetation growth and sediment accretion and, at all sites, TN and TP concentrations have been reduced to background levels before surface water leaves the wetland. At Breaux Bridge, tree ring analysis documented enhanced woody growth for trees receiving effluent, but with a high degree of variability. The permanently flooded wetland at Amelia continues to have sustained productivity of both trees and floating and submerged aquatic vegetation. At the Central Wetlands Unit, discharge of treated effluent contributed to maintaining a baldcypress wetland despite increased regional salinity due to the opening of the MRGO that killed most baldcypress trees in the area. At Mandeville, discharge into a forested wetland produced greater productivity and accretion and significantly reduced nutrients of overlying waters. The Thibodaux assimilation wetland was the only long-term site that was permitted before discharge began. All bottomland hardwood species were dead or dying when the project began due to pre-existing permanent flooding due to coastal water level rise. After a decade of discharge, a floating marsh became established. This floating wetland remains a highly productive community with net primary production greater than 4000 g dry wt $m^{-2}\,yr^{-1}$. These data indicate that properly managed wetland assimilation systems can function for long periods and lead to enhancement of degrading wetland communities. There are many other wetlands in the coastal zone that receive inputs of freshwater, nutrients, and sediments, including areas with river diversions. Proper management of these wetlands can enhance vegetation productivity and soil accretion in the Louisiana coastal zone.

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