Chapter 2
Using Natural Wetlands for Municipal Effluent Assimilation: A Half-Century of Experience for the Mississippi River Delta and Surrounding Environs

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Introduction

The ability of wetlands to improve water quality is well established, with hundreds, if not thousands, of scientific studies published in peer-reviewed journals and books (e.g., Godfrey et al. 1985; Moshiri 1993; Lane et al. 1999, 2002, 2004, 2010; Hunter and Faulkner 2001; Mitsch and Jorgensen 2003; Kangas 2004; Kadlec and Wallace 2009; Hunter et al. 2009a, b; Seo et al. 2013; Shaffer et al. 2015). Use of natural ecosystems for assimilation of nutrients and suspended sediments in treated municipal

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effluent is neither new nor strictly non-traditional (Day et al. 2004). There are thousands of wetland treatment systems worldwide with hundreds of years of operational experience (Kadlec and Wallace 2009). Because wetlands naturally occupy lower landscape positions within a watershed, they are ideally located to serve as biological filters, removing nutrients and sediment from water running off the surrounding landscape before it enters an open water body such as a river or lake.

Studies throughout the world have shown that wetlands chemically, physically, and biologically remove pollutants, sediments and nutrients from water flowing through them (Zhang 1995; Day et al. 2004; Alexander and Dunton 2006; Conkle et al. 2008; Meers et al. 2008; Kadlec and Wallace 2009; Vymazal 2010; Shaffer et al. 2015). Some questions remain as to the ability of wetlands to serve as long-term storage nutrient reservoirs, but examples of long-term sustainability are cypress systems in Florida that continue to remove major amounts of nutrients in treated effluent even after 20–45 years (Boyt et al. 1977; Ewel and Bayley 1978; Lemlich and Ewel 1984; Nessel and Bayley 1984), and the Breaux Bridge and Amelia assimilation wetlands that have received treated effluent for 70 and 47 years, respectively (Hesse et al. 1998; Blahnik and Day 2000; Ko et al. 2004; Day et al. 2006; Hunter et al. 2009b).

With regard to water quality, the primary constituents of interest in treated municipal effluent are nitrogen, phosphorus, and suspended solids, which includes both mineral sediments and particulate organic matter. The basic principle underlying wetland assimilation of these constituents 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). Treated effluent typically introduces nutrients as a combination of inorganic (e.g., nitrate + nitrite (NO_x), ammonia (NH_3), and phosphate (PO_4)) and organic forms, both dissolved and particulate. Nitrogen and/or phosphorus from treated effluent can be removed by short-term processes such as plant uptake, long-term processes such as peat and sediment accumulation, and permanently by denitrification (Reddy and DeLaune 2008).

In the Mississippi River Delta, there are ten assimilation wetlands currently receiving discharge of secondarily-treated, disinfected municipal effluent and four others awaiting permits or under review, as of July 2017 (Fig. 2.1). The assimilation systems in the Mississippi River Delta are not constructed wetlands, however, they are also not "natural" wetlands because they have been highly impacted by anthropogenic activity.

The Mississippi River Delta is a profoundly altered regional ecosystem covering over 10,000 km^2. Over 25% of coastal wetlands in the Mississippi River Delta were lost in the twentieth century. One of the primary causes is the almost complete isolation of the delta plain from the Mississippi River by levees that prevent regular riverine input that occurred under natural conditions before human alterations (Day et al. 2007, 2014). The river provided fresh water, mineral sediments, and nutrients during annual floods. This annual flooding maintained a salinity gradient and provided sediments to promote wetland formation and nutrients to enhance productiv-

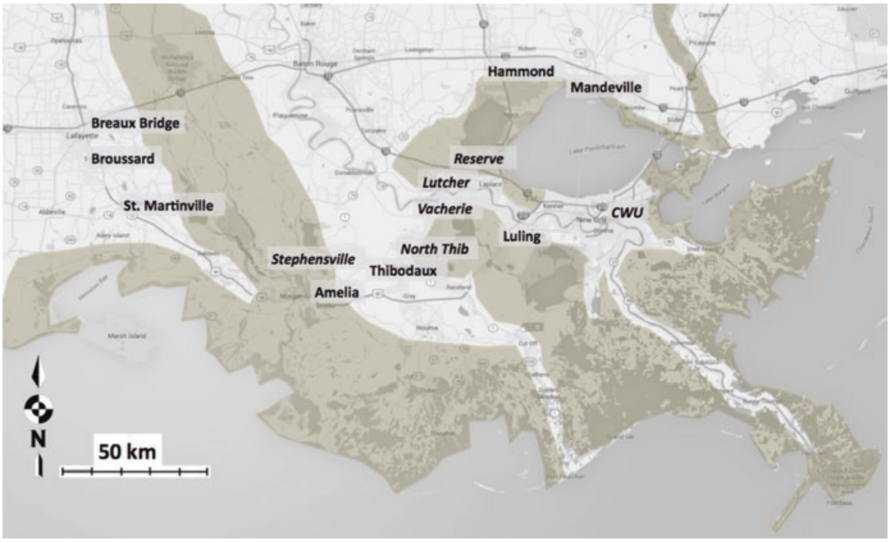

Fig. 2.1 Location of wetland assimilation projects in coastal Louisiana. Municipalities in italics indicate recently completed or ongoing ecological baseline studies. Note that Breaux Bridge, Broussard, and St. Martinville are not impacted by coastal water levels. All the other sites are at or near sea level and are impacted by sea level rise. This inhibits the ability of these sites to drain and have dry periods. Shading indicates areas with a high proportion of wetlands

ity. In addition, there has been a pervasive alteration of hydrology both in the horizontal plane due to spoil banks and canals, as well as vertically caused by enhanced subsidence due to fluid withdrawal (mainly oil and gas), compaction, and drainage. All of the wetland assimilation systems discussed here are in areas where the natural hydrology has been fundamentally altered by human activities.

Wetland assimilation in Louisiana can achieve sustainable low cost tertiary treatment of secondarily-treated municipal effluent while benefiting and restoring wetlands (Day et al. 2004; Hunter et al. 2009a, b). A properly designed wetland assimilation system can be a more economical and sustainable means of tertiary treatment compared to conventional engineering options. The cost of tertiary treatment is a concern as the U.S. Environmental Protection Agency (USEPA) is requiring increasingly stringent limits in discharge permits for wastewater treatment plants. Out of 105 major wastewater treatment facilities in Louisiana, only 12% (13 plants) monitor for nitrogen and phosphorus concentrations, compared with an average of 57% in the 12 states included in the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force. Of the 13 treatment facilities monitoring nutrients in Louisiana, 10 discharge into assimilation wetlands (Hypoxia Task Force 2016).

Freshwater resources, including treated effluent, should be used in a manner that results in the greatest benefits to society. However, municipalities cannot be expected to bear all costs for wetland assimilation projects, so when possible they should be integrated into larger restoration efforts where a variety of funding sources are used.

By doing so, Louisiana benefits from improved water quality with lower cost and energy investments, and from restoration of degraded, yet valuable, wetland ecosystems. The work that is being done in Louisiana is informed by a rich history of scientific and applied experience, taking advice from leading scientists in the field in designing wetland assimilation systems such as Drs. John Day, Jr., William Mitsch, Curtis Richardson, Michael Odgen, Robert Kadlec, Robert Knight, and Scott Wallace.

In this chapter, we discuss the history of wetland assimilation in the Mississippi River Delta. We first provide a background on the environmental setting of the Mississippi River Delta and then discuss steps involved in establishing an assimilation wetland, approaches for ensuring project success, and benefits of these systems. Finally, we review monitoring data from several currently operating systems and discuss recent controversy concerning assimilation wetlands.

The Environmental Setting of the Mississippi River Delta

The functioning and status of assimilation wetlands in the Mississippi River Delta cannot be fully understood without considering the environmental setting of the delta itself—a system formed over the past 6000–7000 years by flooding from the Mississippi River that is on a rapid non-sustainable trajectory of deterioration. Flood control levees and the pervasive alteration of hydrology have isolated wetlands from annual flooding from the river with other deleterious affects including prolonged flooding, saltwater intrusion, and conversion to open water (Kesel 1988, 1989; Mossa 1996; Roberts 1997; Day et al. 2007). Wetland loss in the twentieth century was catastrophic, with approximately 25% of coastal wetlands lost since the middle of the twentieth century (Barras et al. 2008; Couvillion et al. 2011).

A central cause of wetland loss in the delta is subsidence. Subsidence is a natural geologic process due to the compaction, consolidation, and dewatering of sediments. Under natural conditions, sediment deposition from the river and *in situ* organic soil formation balanced subsidence in much of the delta. Now, relative sea level rise, the combination of subsidence plus eustatic sea-level rise, is greater than accretion in much of the delta leading to progressively increased flooding. This has an important impact on several of the assimilation wetlands we review in this chapter. Breaux Bridge, Broussard, and St. Martinville (Fig. 2.1) are located 4–5 m above sea level and are not affected by coastal water levels, which allows them to drain during dry periods. These wetlands also are far enough inland that they are not flooded by hurricane storm surges. All the other assimilation wetlands are affected by coastal water levels, which leads to prolonged and sometimes permanent flooding that prevents them from having dry periods and makes them susceptible to storm surge. Thus forested wetland sites in the coastal zone are generally permanently flooded, which prevents recruitment of baldcypress and water tupelo seedlings that need several months of dry ground to germinate (Allen et al. 1996). These freshwater coastal forested wetlands are also threatened by saltwater intrusion.

Other factors exacerbating wetland loss 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), intentional and unintentional impoundments (Day et al. 1990; Boumans and Day 1994; Cahoon 1994), and herbivory by nutria and other herbivores (Shaffer et al. 1992, 2015; Evers et al. 1998). Almost a third of the delta has been isolated or semi-isolated through the purposeful or accidental construction of various types of impoundments (Day et al. 1990). Throughout this paper we will show how human impacts have negatively impacted areas where wetland assimilation projects are established.

Sustainability in the Mississippi River Delta

Sustainability in the Mississippi River Delta is difficult in the face of increasingly severe climate impacts. Climate change will impact the delta through accelerated eustatic sea-level rise (Meehl et al. 2007; FitzGerald et al. 2008; Pfeffer et al. 2008; Vermeer and Rahmstorf 2009; IPCC 2013; Koop and van Leeuwen 2016; Deconto and Pollard 2016), more severe hurricanes (Emanuel 2005; Webster et al. 2005; Hoyos et al. 2006; Goldenberg et al. 2001; Kaufmann et al. 2011; Mei et al. 2014), drought (IPCC 2007; Shaffer et al. 2015), more erratic and extreme weather (Min et al. 2011; Pall et al. 2011; Royal Society 2014) and increased Mississippi River discharge (Tao et al. 2014). Recent estimates are that sea level will rise between 1 and 2 m by 2100 (Horton et al. 2014; Koop and van Leeuwen 2016; Deconto and Pollard 2016). The combination of accelerated sea-level rise, more intense hurricanes, and drought will lead to increased wetland loss and enhanced saltwater intrusion. Coastal baldcypress—water tupelo swamps are especially susceptible to climate change impacts that increase salinity.

Decreasing energy availability and higher energy prices will limit options for restoration of deltas and complicate human response to climate change (Day et al. 2005, 2007, 2014, 2016a; Tessler et al. 2015). The implication of future energy scarcity is that the cost of energy will increase during the coming decades (Campbell and Laherrere 1998; Deffeyes 2001; Bentley 2002; Hall and Day 2009; Murphy and Hall 2011; Day et al. 2005, 2016a) and the cost of energy-intensive activities will also increase significantly. In a future characterized by scarce and expensive energy, maintaining traditional infrastructure will likely become increasingly unsustainable. Advanced conventional municipal wastewater treatment is very energy intensive, and the use of wetlands offers an energy efficient means to achieve tertiary treatment.

Establishing an Assimilation Wetland

Process Overview

In the State of Louisiana, LDEQ, with oversight from the USEPA, regulates wastewater treatment and the discharge of treated municipal effluent. Over the past 25 years, scientists, regulatory personnel, and dischargers have worked closely to

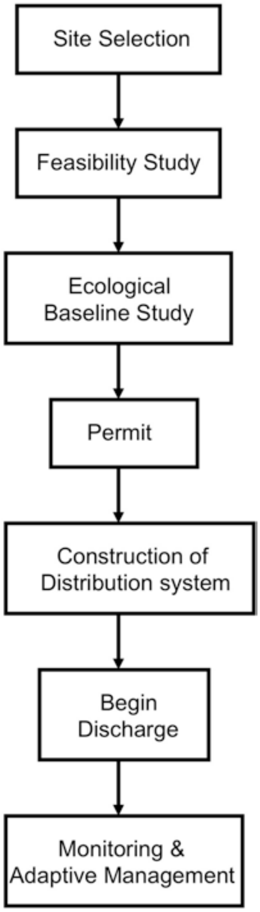

Fig. 2.2 Steps in the establishment of a wetland assimilation project

develop an approach where wetland assimilation systems meet water quality goals while protecting and restoring wetlands (Day et al. 2004; Louisiana Department of Environmental Quality 2010, 2015; Shaffer et al. 2015). Wetlands are carefully selected and monitored prior to discharge of treated effluent and these actions are part of a process to ensure project success (Fig. 2.2).

Site Selection

The process of establishing a wetland assimilation project begins with identification of a suitable candidate wetland (Fig. 2.2). All wetland ecosystems are not created equal and some are clearly unsuited for wetland assimilation. The LDEQ has recognized several wetland types that are not appropriate for assimilation, including seasonally flooded pine flatlands with carnivorous plants and areas heavily used for recreation and oyster production. There are a number of factors taken into consideration for site selection including location, size, hydrology, ecological condition, land ownership, and competing uses. Later in this chapter we address the issue of freshwater herbaceous wetlands.

Feasibility Study

After a candidate wetland is selected, a feasibility study is conducted to determine if the discharge of municipal effluent into the candidate wetland is possible (Fig. 2.2). The feasibility study usually lasts 2–4 months, depending upon the size and complexity of the wetland. During the feasibility study, wetland characteristics (hydrology, soils, vegetation, fauna) are described, along with assessment of surrounding landscape uses, expected nutrient loading rates, and presence of protected flora and fauna and archaeological or historical sites. A preliminary conceptual design of the treated effluent distribution system also is included.

Ecological Baseline Study

If the feasibility study finds the candidate wetland suitable for assimilation, a year-long ecological baseline study (EBS) is conducted (Fig. 2.2). The purpose of the EBS is to describe in detail the baseline ecological conditions of the candidate site, including hydrology, soil and water chemistry, accretion rate, and vegetative species composition and productivity. In addition, a preliminary engineering design and cost analysis are conducted. The EBS then forms part of the permit application, which is the fourth step in the process (Fig. 2.2). The EBS may be carried out at the same time that the permit applications are submitted and under review, however, the EBS data must be completed before final approval of the permits.

Regulatory and Permitting

Several permits may be required for a wetland assimilation project. An LPDES permit is required under the authority of the Federal Clean Water Act and the Louisiana Environmental Quality Act. These two acts require criteria (as set forth in the permit) to protect the beneficial uses (e.g., fish and wildlife propagation) and contain an anti-degradation policy that limits lowering of water quality. The LPDES permit designates biochemical oxygen demand (BOD), total suspended solids (TSS), and fecal coliform effluent limits for discharge to the wetland (Table 2.1) and also outlines monitoring requirements (discussed in next section) and nutrient

Table 2.1 Design characteristics of wastewater treatment plants and LPDES mean monthly and weekly permit limits for BOD, TSS and fecal coliform at Louisiana assimilation wetlands

Municipality	LPDES permit #	Year 1st Issued	Treatment system	Design capacity (MGD[a])	BOD (mg/L)	TSS (mg/L)	Fecal coliform (colonies/100 mL)	Population (2010 census)
Amelia	90999	2007	Oxidation pond with chlorination-dechlorination	0.9	30/45	90/135	200/400	2459
Breaux Bridge	30578	2003	3 oxidation ponds and chlorination-decholorination system	1.27	30/45	90/135	200/400	8139
Broussard	33786	2007	3 oxidation ponds and chlorination-decholorination system	1.0	30/45	90/135	200/400	8197
Guste Island	122552	2008	2-cell facultative lagoon with chlorination-dechlorination system	0.6	30/45	90/135	200/400	Private subdivision—Data not available
Hammond	19578	2010	3 cell oxidation lagoon and chlorination-dechlorination system	8.0	30/45	90/135	200/400	20,019
Luling	43356	2008	Facultative oxidation pond with UV disinfection	3.5	30/45	90/135	200/400	12,119
Mandeville	19420	2003	3 aerated lagoon cells, 3-celled rock reed filter, and UV disinfection	4.0	CBOD 10/15	15/23	200/400	11,560
Riverbend	19244	Pending	Oxidation pond with chlorination-dechlorination system	0.7	30/45	90/135	200/400	5000 homes, Population unknown
St. Martinville	19216	2006	63.7 ha facultative lagoon; UV disinfection; cascade aeration structure	1.5	30/45	90/135	1000/2000	6114
Thibodaux	19012	2004	Aerated lagoon and high-rate trickling filter; UV disinfection	4.0	30/45	30/45	200/400	14,566

[a]Million gallons per day

loading rates. Generally, effluent limits are somewhat less restrictive than for direct discharge to open water bodies because of the ability of wetlands to process and assimilate nutrients and organic matter without deleterious effects (Day et al. 2004). The permit requires disinfection so that pathogens are not discharged to wetlands and toxic materials must be below state and federal limits.

Water Quality Standards (WQS) are provisions of Louisiana State Law and these standards are applied to each assimilation wetland. Water Quality Standards consist of policy statements pertinent to water quality necessary to preserve or achieve the objectives of the standards, designated uses for which public waters of the state are to be protected, and criteria which specify general and numerical limitations for various water quality parameters that are required for designated water uses (Louisiana Department of Environmental Quality 2015). Water Quality Standards for assimilation wetlands in Louisiana serve to protect and preserve the biological and aquatic community integrity.

A CUP may be required for an assimilation wetland project, usually for the discharge pipeline installation. The CUP process is part of the Louisiana Coastal Resources Program (LCRP) that works to preserve, restore, and enhance Louisiana's valuable coastal resources. The purpose of the CUP application process is to make certain that any activity affecting Louisiana's coastal zone, such as a project that involves either dredging or filling, is performed in accordance with guidelines established in the LCRP. The guidelines are designed so that development in the coastal zone can be accomplished with the greatest benefit and the least amount of damage. Section 404(b)(1) of the Clean Water Act serves to regulate the alteration or discharge of dredged and/or fill material into U.S. waters, including wetlands. Many wetland assimilation projects require a 404 permit, particularly for the placement and installation of the effluent discharge pipeline.

Construction and Monitoring

After the necessary permits are issued, the discharger begins construction of the distribution pipeline and, upon completion, discharge and monitoring begin (Fig. 2.2). Monitoring of vegetation, soils, water, and hydrology is required in the LPDES permit for the life of the project and annual monitoring reports are submitted to LDEQ. Continual cooperation among those involved (e.g., municipality personnel and/or dischargers, ecological monitoring team, and regulatory agencies) is essential to ensure proper management over the life of the project.

Approaches to Ensure Success of an Assimilation Wetland

To ensure a healthy and sustainable assimilation wetland and a successful project, treated effluent must be discharged into the assimilation wetland at an appropriate loading rate, which is explained in detail below. The effluent must be disinfected and free from toxins to avoid endangering fauna or flora. Strict policy guidelines must be adhered to as well as long-term monitoring to detect potential problems and to achieve water quality goals, as well as to maintain a healthy wetland.

Appropriate Loading Rate

The basic principal underlying the use of wetlands for municipal effluent assimilation is that the rate of effluent discharge to the wetland must balance the rate of nutrient removal. Therefore, one of the most important factors in designing wetland assimilation systems is the loading rate. In general, loading rate refers to the rate per unit of area at which a material (e.g., a constituent in the effluent) is discharged into a system over a given time period. High nutrient loading rates to wetland systems may not allow for sufficient processing time, resulting in a wetland that is overloaded in nitrogen and/or phosphorus and that has a reduced capacity for assimilation of nutrients in the future. Conversely, at low loading rates, the wetland may have a higher capacity to remove nutrients than at high loading rates.

Specific to wetland systems receiving secondarily-treated municipal effluent, loading rates are normally calculated using nutrient concentrations (i.e., total nitrogen and phosphorus) of the municipal effluent, the volume of the discharge, and the area of the receiving wetland. For wetland assimilation systems in Louisiana, typical loading rates for total nitrogen (TN) range from 2 to 15 g/m^2/year and for total phosphorus (TP) from 0.4 to 4 g/m^2/year (Day et al. 2004). Removal efficiencies for TN and TP at these loading rates average between 65 and 90%, while NO_x removal is between 90 and 100%. Nutrient removal efficiency is the percentage of nutrients removed from the overlying water column and retained within the wetland ecosystem or released into the atmosphere. Richardson and Nichols (1985) reviewed a number of wetlands receiving municipal effluent and found a clear relationship between loading rate and nutrient removal efficiency (Fig. 2.3). The relationship between nutrient removal efficiency and loading rate is not linear, with very effi-

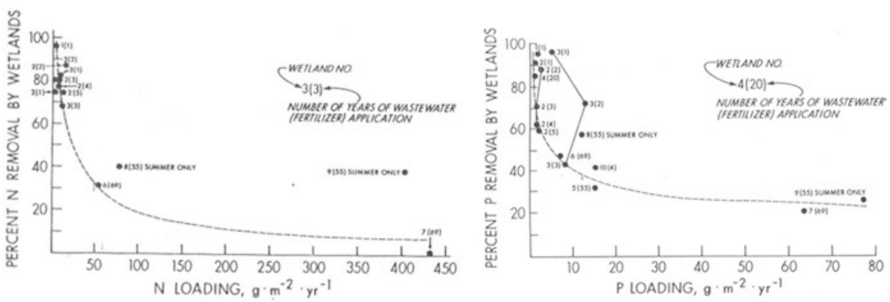

Fig. 2.3 Nitrogen and phosphorus removal efficiency as a function of loading rate in various municipal effluent assimilation wetlands (taken from Richardson and Nichols 1985)

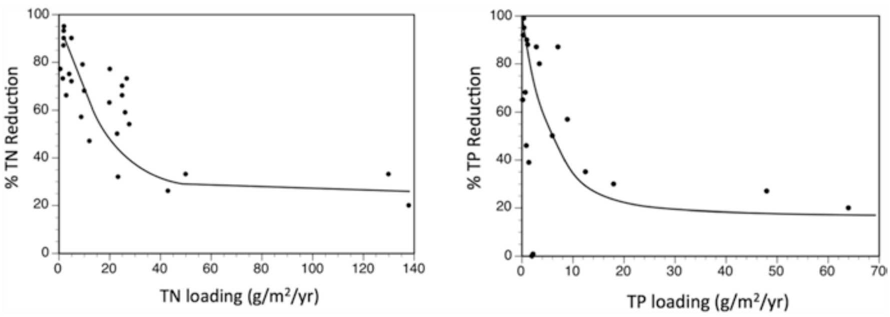

Fig. 2.4 Nitrogen and phosphorus removal efficiency as a function of loading rate in Louisiana wetlands receiving secondarily treated municipal effluent, stormwater, or diverted Mississippi River water

cient nutrient removal at low loading rates, and rapidly decreasing removal efficiency as loading rates rise. Mitsch et al. (2001) found a similar loading-uptake relationship for wetlands in the upper Mississippi basin.

The curves generated by Richardson and Nichols (1985) are derived from data of wetland assimilation systems located in many different parts of the United States. Data from assimilation wetlands, stormwater wetlands, and coastal wetlands receiving diverted Mississippi River water showed that this relationship was generally true for wetlands in Louisiana (Fig. 2.4). Nutrient uptake also has been reported in coastal wetlands receiving Atchafalaya River water (Lane et al. 2002) and Mississippi River water (Lane et al. 1999, 2004).

Effluent Disinfection

Contamination by human pathogens is an important issue that must be considered in wetland assimilation since these pathogens can be transferred to other animal species as well as to humans. For this reason municipal effluent is disinfected prior to release into the wetlands. Nevertheless, studies have shown that pathogens are rapidly degraded in wetlands, much more so than in open water bodies such as lakes, streams or bayous (Kadlec and Wallace 2009). Proper disinfection is a particular concern for all municipal effluent treatment plants, and dischargers are responsible for regularly monitoring the effectiveness of disinfection systems. Commonly used disinfection methods include chlorination followed by dechlorination, UV radiation, and ozone. Chlorination is the most commonly used method of disinfection but, although dechlorination reduces toxic chlorine residuals, it increases operator costs and may introduce hazardous chemicals into the aquatic environment depending on the method used. Ozone and UV radiation are the cleanest disinfection methods, but have operator costs as well (City of New York Department of Environmental Protection and HydroQual, Inc. 1997).

Contaminants

Metals

The LDEQ currently requires cadmium, chromium, copper, iron, lead, magnesium, nickel, selenium, silver, and zinc concentrations to be measured at specific time increments in surface water, soils, and vegetation of wetlands receiving treated municipal effluent. Metal concentrations of surface waters at assimilation wetlands in Louisiana have been very low, with most concentrations below the detectable limit. There also have been no detectable differences in metal concentrations in sediments or vegetation between the assimilation wetlands and reference wetlands (Table 2.2). Similar results have been obtained for all assimilation wetlands in Louisiana. In general, there is little evidence that metal contamination is a problem for assimilation wetlands.

Contaminants of Emerging Concern

The impact of pharmaceuticals and personal care products (PPCPs) and endocrine disrupting compounds (EDCs) in natural systems has become an important issue (Koplin et al. 2002; Boyd et al. 2004). These compounds are excreted into sewage systems and enter the aquatic environment with the discharge of treated municipal wastewater (Li et al. 2014). Impacts from PPCPs are most pronounced in smaller streams where effluent discharge makes up a large proportion of the flow. Conventional wastewater treatment plants have limited ability to remove PPCPs due to short retention times while natural and constructed wetlands can promote removal through a number of mechanisms, including photolysis, plant uptake, microbial

Table 2.2 Mean metal concentrations in soils and vegetation at effluent discharge sites and nearby reference sites

	Soils concentration (mg/kg)		Vegetation concentration (mg/kg)	
Analyte	Discharge	Reference	Discharge	Reference
Cadmium	1.31 ± 0.25	1.08 ± 0.15	0.62 ± 0.06	0.79 ± 0.20
Chromium	14.44 ± 1.64	16.64 ± 2.20	0.93 ± 0.16	0.63 ± 0.08
Copper	19.18 ± 1.88	21.35 ± 3.18	3.29 ± 0.72	5.09 ± 0.96
Iron	14,343 ± 1538	12,067 ± 1567	146 ± 25	125 ± 23
Lead	16.20 ± 1.72	20.20 ± 2.52	1.23 ± 0.29	0.72 ± 0.10
Magnesium	3081 ± 397	2725 ± 326	2195 ± 408	2719 ± 468
Nickel	13.69 ± 1.18	16.02 ± 2.43	1.90 ± 0.34	2.20 ± 0.38
Selenium	2.99 ± 0.23	3.11 ± 0.44	3.65 ± 0.47	3.02 ± 0.31
Silver	0.90 ± 0.16	0.44 ± 0.07	0.67 ± 0.02	0.68 ± 0.13
Zinc	78.36 ± 7.77	82.80 ± 11.66	27.45 ± 3.38	46.74 ± 8.32

Data shown are means (± standard error) for samples collected at Breaux Bridge, Broussard, Hammond, Luling, and Mandeville assimilation wetlands degradation, hydrolysis, and sorption to soil (White et al. 2006; Li et al. 2014; Verlicchi and Zambello 2014). Verlicchi and Zambello (2014) reviewed 47 studies of constructed wetlands used for reducing concentrations of contaminants of emerging concern and concluded that these systems have the potential to remove many contaminants, including naproxen, salicylic acid, ibuprofen and caffeine. Removal efficiency was related to a variety of factors, including type of constructed wetland (e.g., free surface, sub-surface), hydraulic retention time, type of pre-treatment, redox potential, and environmental conditions.

Recent research has shown that most PPCP parent compounds are removed rapidly from the water column through various degradation or removal pathways (Boyd et al. 2003; Batt et al. 2007). For example, a recent study of feedlot wastes found a decrease of 83–93% in estrogenic activity as the wastewater flowed through a constructed wetland system (Shappell et al. 2007). Some municipalities are currently using wetlands to remove PPCPs from effluent and percent reduction is related to residence time in the treatment system. Treatment systems using ponds with long residence times (10–30 day) are more effective in removing PPCPs than highly engineered systems with a short residence time (12–18 h). Low loading rates, long residence times, and diverse microbial habitats in wetland assimilation systems should further promote the breakdown of PPCPs.

One group of personal care products known as nonylphenol ethoxylates (NPEOs) enter the environment due to their use in paints, inks, detergents, pesticides and cleaners. NPEOs can be as much as 10% of the total dissolved organic carbon entering a wastewater treatment plant (Ahel et al. 1994). Vegetated treatment wetlands have demonstrated the ability to remove up to 75% of NPEOs from domestic wastewater (Belmont et al. 2006).

A study at the Mandeville wastewater treatment facility, which consists of both a series of aeration lagoons, a constructed wetland, and natural wetlands, showed that the treatment system decreased the concentrations of nine types of PPCPs by 90% or more, dependent on the compound (Conkle et al. 2008). For most compounds reduction in concentration occurred over a 30-day treatment period in the aerated lagoons. However, the adjacent forested wetland also showed significant (6–52%) removal for several common pharmaceuticals (Conkle et al. 2010).

Little research has focused on sorption in wetland soils though this may be an important removal mechanism since many compounds are likely to bind to soils that have charged binding sites such as clays and organic matter found in wetland soils. Three estrogenic compounds, Bisphenol-A, 17ß-estradiol and 17α-ethinylestradiol (Clara et al. 2004), and three antibiotic compounds, ciprofloxacin, norfloxacin and ofloxacin (Conkle et al. 2010), have been shown to have high sorption coefficients, indicating that sorption is a major pathway for compound removal from the water column. Estrogenic sorption was studied using sewage sludge, while the antibiotics were tested on a wetland soil containing 20% organic matter. Research to date has not addressed the fate of these compounds once bound in soil or their effect on microfauna.

Concentrations of these compounds of concern are highest closest to the point of effluent discharge. The experience with the Mandeville wastewater treatment system suggests that in order to minimize exposure of the biota to higher concentrations of these compounds, discharging into a settling pond is necessary prior to discharge into the receiving wetland. This allows for initial removal as well as providing time-averaged concentrations before discharge into the wetland system.

Policy Considerations

The use of wetlands for municipal effluent assimilation has important implications for total maximum daily loads (TMDLs) and nutrient limits. A TMDL is a calculation of the maximum amount of a pollutant that a water body can receive and still meet EPA and state environmental water quality standards. In the case of water quality problems related to over-enrichment and eutrophication, the pollutants of interest are nutrients and non-toxic organic compounds. One problem that may arise for small municipalities in a watershed dominated by other pollutant sources (such as agriculture or a large city) is that the TMDL allocation for the municipality will be very low, necessitating greater wastewater treatment prior to discharge to receiving water bodies. The use of wetland assimilation provides an economical means for such additional water quality improvement (Ko et al. 2004, 2012).

Land Ownership

Communities in Louisiana have employed a variety of strategies to work with landowners when utilizing wetlands for assimilation of nutrients and sediments in various wastewaters. These strategies range from outright land purchase to a memorandum of understanding (MOU) and flowage easements (Table 2.3).

Ecological Monitoring

Monitoring of vegetation, hydrology, water quality and soils is a vital component of any wetland assimilation project. Requirements for monitoring are outlined in the LPDES permit for discharge of treated effluent into a wetland (Table 2.4). Vegetation data provide information on the health and vigor of the plant community, and whether vegetative species composition or dominance is being altered due to effluent addition. Water gauge data provide information about hydrology and changes in the depth and duration of inundation. Metals and nutrient data of soils and vegetation determine if there is an accumulation of these materials that could become problematic. Surface water quality data provide information of the efficiency of the system in removing nutrients from the water column. Data are collected from the assimilation wetland and from an ecologically similar reference wetland that is not

Table 2.3 Assimilation wetland land use agreements

Community	Type of agreement	Collaborating entity
Mandeville	Purchase	City owns land
Hammond	Memorandum of understanding	LDWF and City owns 230 acres
St. Charles—Luling	Flowage easement	Private landowner
Thibodaux	Flowage easement	Private landowner
Amelia	Flowage easement	Private landowner
Broussard	Purchase	NA
St. Martinville	Purchase	NA
Breaux Bridge	Memorandum of understanding	Nature conservancy

Table 2.4 LPDES monitoring requirements for a typical wetland assimilation project in Louisiana

	Wetland component			
Parameter	Flora	Sediment	Surface water	Effluent
Species classification	P			
Percentage of whole cover (for each species)	P			
Growth studies	A			
Water stage			M	
Metals: Mg, Pb, Cd, Cr, Cu, Zn, Fe, Ni, Ag, Se	P	P	P	S
Metals analysis: Hg, As		P		
Nutrient analysis I: TKN, TP	P	P	S	
Nutrient analysis II: NH_3N, NO_2N, NO_3N, PO_4		P	S	
Others: BOD5, TSS, pH, Dissolved Oxygen			P	
Accretion Rate		P		

P: Periodically—Sampling must be made once during March through May and once during September through November in the fourth year of the permit period for three Assimilation areas and one Reference area
A: Annually—Sample once per year at three Assimilation areas and one Reference area
M: Monthly—Samples should be taken at three Assimilation areas and one Reference area each month.
S: Semi-annually—Sample twice per year. Once during September through February and once during March through August at three Assimilation areas and one Reference area

impacted by the treated effluent. By comparing data between the assimilation and reference wetlands, as well as pre- and post-discharge data at the assimilation wetland, it is possible to determine if the assimilation wetland is being positively or negatively impacted by effluent addition.

According to the LPDES discharge permit, if wetland monitoring indicates that there is: (A) more than a 20% decrease in naturally occurring litterfall or stem growth; or (B) significant decrease in the dominance index or stem density of baldcypress; then, the permittee shall conduct such studies and tests as to determine if the impact to the wetland was caused by the effluent. Thus, monitoring provides a mechanism for evaluating the impacts of treated effluent on an assimilation wetland. It is important to note that wetland monitoring requirements may be modified by LDEQ if data indicate that changes are necessary.

Nutria Management

Nutria (*Myocaster coypus*), an introduced rodent, can severely impede attempts to restore baldcypress swamps (Myers et al. 1995) and herbaceous wetlands (Shaffer et al. 1992; Evers et al. 1998; Shaffer et al. 2015) in coastal Louisiana. This has certainly been the case for the Manchac land bridge, Jones Island, Sawgrass Bayou, and Big Branch National Wildlife Refuge in the mid and upper Lake Pontchartrain Basin, where nutria have killed tens of thousands of baldcypress seedlings and hundreds of hectares of herbaceous marsh (Effler et al. 2007; McFalls et al. 2010). Nutria appear to be able to detect the higher protein content of wetland vegetation with higher nutrient content, whether from fertilizer (Shaffer et al. 2009; Ialeggio and Nyman 2014) or treated municipal effluent (Lundberg 2008; Shaffer et al. 2015). Very few nutria were observed during the pre-discharge data collection phase at the Hammond assimilation wetland. However, within 12 months of discharge initiation, nutria numbers increased dramatically (Shaffer et al. 2015). Nutria are very prolific and can breed any time of year, producing at least 2 litters per year with an average of 4.5 young per litter (http://www.nutria.com/site).

Nutria herbivory can be controlled by several methods, such as nutria exclusion devices for seedlings, which are very effective in preventing herbivory of individual seedlings (Myers et al. 1995). For large areas, however, the only effective protection is population reduction. In an attempt to manage nutria populations in coastal Louisiana, the Coastal Wetlands Planning, Protection, and Restoration Act (CWPPRA) established the Coastwide Nutria Control Program (CNCP). The goal of the CNCP, which is managed by the Louisiana Department of Wildlife and Fisheries, is to encourage the harvest of nutria by paying trappers $5 per tail. In the 2015–2016 trapping season, 349,235 nutria tails, worth $1,746,175 in incentive payments, were collected from 274 participants (http://www.nutria.com/control_program). As part of the adaptive management at the Hammond assimilation wetland, one trapper and several hunters shot over 2000 nutria in one season (Shaffer et al. 2015). This is discussed in more detail below.

Benefits of Wetland Assimilation

Wetland Restoration

The introduction of treated municipal effluent into degraded forested wetlands of Louisiana is a major step towards their ecological restoration. The nutrient component of municipal effluent increases wetland vegetation productivity (Rybczyk et al. 1996; Hesse et al. 1998; Lundberg 2008; Hunter et al. 2009b; Shaffer et al. 2015), which helps offset regional subsidence by increasing organic matter deposition on the wetland surface, thereby decreasing flooding duration and producing a positive feedback loop of increased ecosystem vigor and resiliency (Fig. 2.5). The

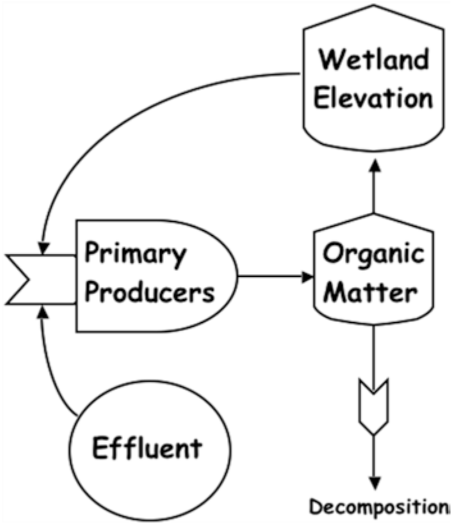

Fig. 2.5 Conceptual model of the effects of effluent application to wetlands

freshwater component of effluent also provides a buffer against saltwater intrusion events, especially during periods of drought, which are predicted to increase in frequency in the future due to global climate change (IPCC 2001). For example, the prolonged drought of 2000–2001 led to widespread death of baldcypress in the Lake Pontchartrain basin when saltwater intruded into these areas (Shaffer et al. 2009; Day et al. 2012).

Recent efforts to restore and enhance wetlands in the subsiding delta region have focused on attempts to decrease vertical accretion deficits by either physically adding sediments to wetlands or by installing sediment trapping mechanisms (e.g., sediment fences), thus increasing elevation and relieving the physio-chemical flooding stress (Day et al. 1992, 1999, 2004; Boesch et al. 1994). Breaux and Day (1994) proposed an alternate restoration strategy by hypothesizing that adding nutrient rich secondarily-treated municipal effluent to hydrologically isolated and subsiding wetlands could promote vertical accretion through increased organic matter production and deposition. Their work, and other studies, has shown that treated municipal effluent does stimulate productivity and accretion in wetlands (Rybczyk 1997; Hesse et al. 1998; Brantley et al. 2008; Hunter et al. 2009b; Shaffer et al. 2015). Rybczyk et al. (2002) reported that effluent discharge into the Thibodaux assimilation wetlands increased accretion rates by a factor of three (Fig. 2.6). DeLaune et al. (2013) reported that the Davis Pond river diversion in southern Louisiana led to accretion rates in receiving wetlands of more than 1 cm/year.

Over the past several decades, many attempts have been made to restore degraded baldcypress-water tupelo swamps in coastal Louisiana. In general, four primary interacting factors have been responsible for the very limited success of these restoration attempts: saltwater intrusion, persistent flooding, nutria, and lack of nutrients.

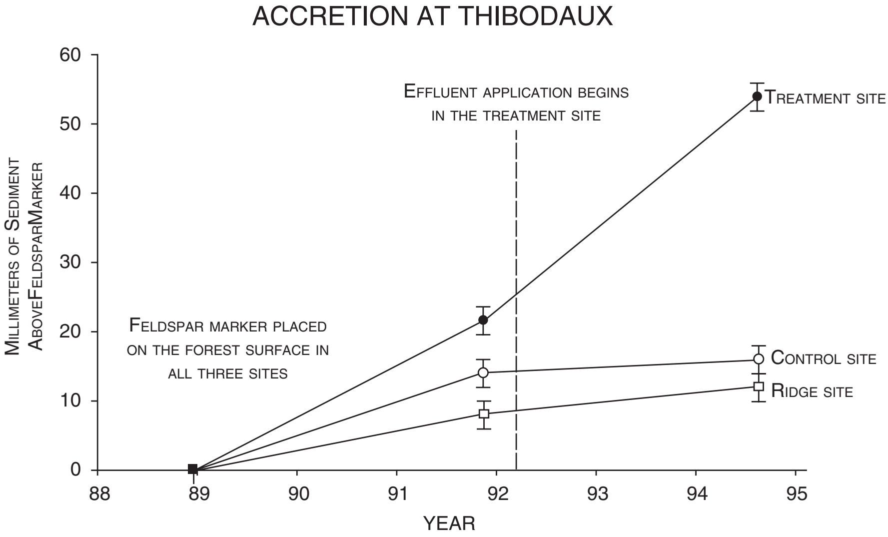

Fig. 2.6 Sediment accretion measured using feldspar horizon markers at the Thibodaux assimilation wetland (treatment site) and reference wetland (control site); From Rybczyk et al. 2002)

Carefully implemented wetland assimilation projects solve all four problems. Municipal effluent is a source of fresh water to otherwise hydrologically isolated wetlands, buffering saltwater intrusion events and providing fresh water during droughts. Hurricane storm surge can cause long-term changes in porewater salinity in coastal swamps and marshes but salt water can be flushed from soils by discharge of municipal effluent, river diversions, or other sources of fresh water (Steyer et al. 2007). Although input of treated effluent does not reduce persistent flooding, consistent input of municipal effluent decreases the residence time of water in a wetland, pushing out toxins (e.g., sulfides) that accumulate under stagnant conditions. Through stringent management nutria populations can be held in check. Finally, nutrient rich municipal effluent addition promotes increased rates of primary production and soil accretion, an important part of any restoration plan for wetlands in coastal Louisiana.

Enhanced Productivity

Secondarily-treated effluent delivers nutrient rich water to wetlands, stimulating vegetative productivity. While this could lead to eutrophication in some aquatic systems, many regions of the Gulf Coast are isolated from historic pulses of nutrients and sediments by dams, dikes, and levees, and, thus, are nutrient limited. Treated effluent can be used to enhance and restore productivity to these areas (Day et al. 2004).

Table 2.5 Mean litterfall, stem growth, and total net primary productivity (NPP) of forested wetlands receiving discharge of treated effluent and reference wetlands in Louisiana

Site	Discharge site			Reference site		
	Litterfall (g/m^2/year)	Stem growth (g/m^2/year)	Total NPP (g/m^2/year)	Litterfall (g/m^2/year)	Stem growth (g/m^2/year)	Total NPP (g/m^2/year)
Breaux Bridge (2002–2013)	56.9 ± 8.7	12.2 ± 2.1	69.2 ± 10.7	34.5 ± 4.1	23.3 ± 3.7	57.8 ± 7.8
Broussard (2007–2013)	39.7 ± 7.5	32.2 ± 6.0	72.0 ± 13.5	14.7 ± 3.2	17.4 ± 1.3	32.1 ± 4.6
Luling (2008–2013)	31.9 ± 2.7	21.7 ± 2.0	53.6 ± 4.7	17.6 ± 0.9	24.5 ± 2.3	42.1 ± 3.2

Data shown are means of post-discharge monitoring data collected by Comite Resources, Inc. (monitoring time period in parenthesis). Data are based on mean productivity per tree

Net primary productivity is generally higher at wetlands receiving discharge compared to corresponding reference wetlands (Table 2.5).

Hesse et al. (1998) conducted a tree ring analysis to document long-term effects of discharge of treated municipal effluent on the growth rate of baldcypress at the Cypriere Perdue assimilation wetland near Breaux Bridge, Louisiana. Treated effluent has been discharged into this wetland since 1954, but long-term monitoring has only been conducted since the city was issued an LPDES permit specific to assimilation wetlands in 2001. Growth chronologies from 1920 to 1992 were developed from cross-dated tree core samples taken from Discharge (Treated) and Reference (Control) sites with similar size and age classes. Significant differences in growth response between sites showed a consistent pattern of growth enhancement in the site receiving treated effluent (Fig. 2.7).

Shaffer et al. (2015) found that growth of baldcypress seedlings at the Hammond assimilation wetland was greatest where treated effluent was discharged and growth followed a linear decrease to 700 m from discharge. The diameter increase of mature baldcypress trees located along the effluent discharge pipe was five times greater than that of the Maurepas swamp and tenfold higher then trees at the Reference site. Baldcypress seedlings planted within 20 m of the effluent outfall system in 2008 averaged over 8 m tall in 2010 and were growing 2.01 cm/year (+0.08 cm/year S.E.) in diameter (Shaffer et al. 2015). There have been numerous studies showing either increased growth or no effect to baldcypress that are exposed to highly nitrified water. For example, Brantley et al. (2008) found significantly higher baldcypress growth downstream of effluent discharged from the Mandeville Bayou Chinchuba wastewater treatment plant. Shaffer et al. (2009) found increased growth rates in the Maurepas basin in areas receiving regular non-point source inputs, as did Effler et al. (2007) for trees given nutrient amendments. At the Amelia assimilation wetland, total NPP was higher at the Discharge site than at the corresponding Reference site (Day et al. 2006).

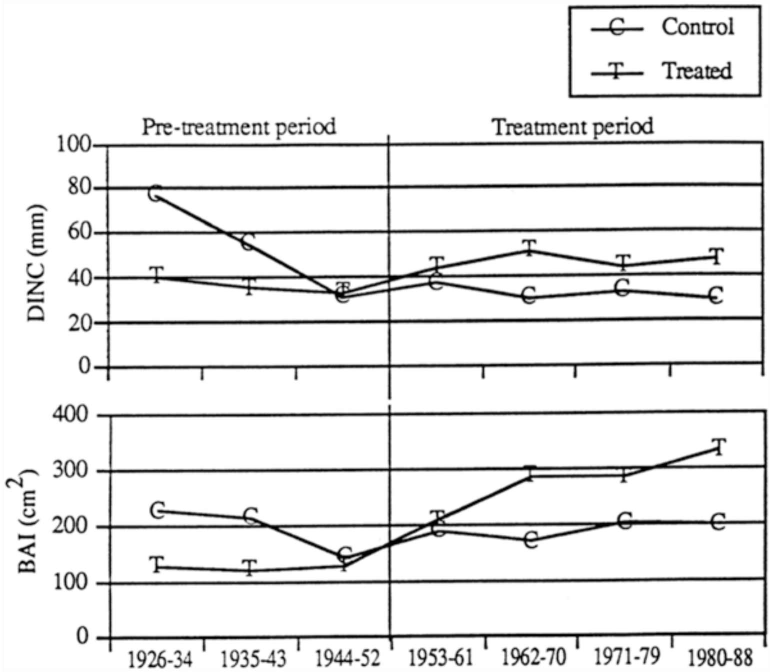

Fig. 2.7 Average periodic diameter increment (DINC) and basal area increment (BAI) growth/tree for each 9-year interval for baldcypress in the Cypriere Perdue Swamp; Taken from Hesse et al. (1998)

Nutrient Reduction

Water quality improvement of municipal effluent has been well documented in the assimilation wetlands at Amelia (Day et al. 2006), Breaux Bridge (Blahnik and Day 2000; Hunter et al. 2009b), Hammond (Shaffer et al. 2015), Luling (Hunter et al. 2009a), Mandeville (Brantley et al. 2008), St. Bernard (Day et al. 1997b), and Thibodaux (Zhang et al. 2000; Izdepski et al. 2009). Reduction of NO_x, NH_3, total Kjeldahl nitrogen (TKN), and total phosphorus typically range between 60 and 100% (Table 2.6). At most sites, concentrations of nitrogen and phosphorus were reduced to background levels by the time surface water left the wetland.

Burial in sediments integrates numerous processes that remove nutrients from treated effluent, including the settling of organic and inorganic sediments from the water column, microbial uptake, and the incorporation of organic matter (e.g., leaf litter or roots) into the sediment. Plant uptake cannot be considered a long-term loss unless the nitrogen and phosphorus are stored in persistent woody tissue and then ultimately harvested or buried in the wetland. Nutrients assimilated by herbaceous plants, however, can remain unavailable for long periods if they are asso-

Table 2.6 Percent nutrient reductions of effluent entering and leaving the assimilation wetlands in coastal Louisiana

Site	Parameter	Discharge concentration (mg/L)	Outlet concentration (mg/L)	% Reduction
Amelia[a]	TKN	2.98	1.00	66
	Total P	0.73	0.06	92
	NO_x	0.80	<0.01	100
Breaux Bridge[b]	PO_4	1.00	0.20	80
	Total P	2.90	0.30	87
Hammond[c]	NH_3	9.85	0.50	95
	NO_x	6.18	0.01	100
	PO_4	3.55	0.01	100
	Total P	4.04	0.04	99
Luling[c]	NO_x	0.52	0.19	63
	PO_4	0.62	0.20	67
Mandeville[c]	NH_3	1.90	0.60	68
	NO_x	5.86	1.09	81
	PO_4	3.31	1.64	57
	Total P	3.82	1.64	57
St. Bernard[d]	TKN	13.60	1.40	90
	Total P	3.29	0.23	95
Thibodaux[e]	NO_x	8.70	<0.10	100
	TKN	2.90	0.90	69
	PO_4	1.90	0.60	68
	Total P	2.46	0.85	66

[a]Day et al. (1997a)
[b]Day et al. (1994)
[c]Comite Resources, Inc. monitoring data
[d]Day et al. (1997b)
[e]Zhang et al. (2000)

ciated with refractory organic matter that becomes incorporated in the soils (Morris et al. 2013).

Increased nutrient inputs at many wetland treatment systems leads to a growth of algae. When light and nutrients are not limiting, algae can contribute significantly to the food web and nutrient cycling (Kadlec and Wallace 2009). However, because most of the assimilation wetlands in Louisiana are forested and have a closed canopy, particularly where effluent discharge occurs, algal blooms do not occur. Two exceptions include the Hammond and Thibodaux assimilation wetlands. At the Hammond assimilation wetland, treated effluent is discharged into an emergent freshwater wetland. Nutria herbivory of the wetland vegetation caused the system to largely degrade to open water which was subsequently colonized by algae. As the system recovered, emergent marsh species colonized the area and shaded out the growth of the algae and floating aquatic species. At the Thibodaux assimilation wetland, the system had very sparse growth of degraded baldcypress when discharge of treated effluent began. Over time, a floating marsh emerged and, again, shaded out the growth of algae. Both of these assimilation wetlands are discussed in more detail later in this chapter.

Carbon Sequestration

The term 'carbon sequestration' describes removal of atmospheric carbon dioxide (CO_2), usually by plants, and permanent storage of the fixed carbon in the ecosystem. Carbon sequestration is mostly viewed in relation to mitigating CO_2 released during the burning of fossil fuels (Williams 1999; Lal 2004; Euliss et al. 2006). Wetlands located in the Louisiana coastal zone have the potential to permanently store carbon due to high regional geological subsidence of 2–10 mm/year (Penland et al. 1988). Rybczyk et al. (2002) found that the Thibodaux assimilation wetlands had significantly higher accretion rates compared to an adjacent reference wetland. Because this accretion was due primarily to an increase in organic matter (OM) rather than mineral sediments, significant carbon burial occurred (OM generally consists of 50% carbon by weight). Estimates of carbon burial pre-(1375 kg C/ha/year) and post-effluent addition (3680 kg C/ha/year) indicate that 2305 kg C/ha/year of additional carbon was sequestered due to the discharge of municipal effluent (Day et al. 2004).

Global warming has become a major worldwide concern that has facilitated significant growth in emissions trading programs collectively referred to as carbon markets. Projects that reduce greenhouse gas emissions generate 'carbon offsets'. A carbon offset (mt CO_2e), also referred to as a carbon credit, is a metric ton reduction in emissions of CO_2 or greenhouse gases made to compensate for, or to offset, an emission made elsewhere (Murray et al. 2011). For a variety of financial, environmental, and political reasons, substantial interest exists for carbon offsets derived from terrestrial landscapes, including wetland ecosystems. The carbon sequestered in coastal and marine ecosystems has been termed 'blue carbon' (Mcleod et al. 2011; Sifleet et al. 2011). Allowing entities to privately invest in wetland restoration projects to offset greenhouse gas emissions elsewhere holds promise as a new carbon offset sector. In the future, the ability to sell carbon credits may provide an important source of revenue for municipalities in Louisiana using wetland assimilation of municipal effluent. Lane et al. (2017) recently documented net carbon sequestration at the Luling wetland assimilation system.

Mitigation of Impacts of Global Climate Change

There are two important global trends that should be considered as part of an analysis of wetland assimilation. These trends are global climate change and the cost and availability of energy, specifically oil. Three climate trends have important implications for wetlands located along coastal Louisiana: accelerated sea level rise; greater frequency of strong hurricanes; and more frequent and longer durations of drought, all of which lead to saltwater intrusion. The introduction of treated municipal effluent to wetlands directly counters the last of these trends (via freshwater addition), and indirectly counters the others through increased vegetative productivity (providing hurricane protection) and accretion (which increases wetland surface elevation).

Energy and Economic Savings

The availability and cost of energy will likely become an important factor affecting society in the near future. Over the past decade, increasing information has appeared in the scientific literature suggesting that world oil production is peaking or will peak within a decade or two, implying that demand will consistently be greater than supply, and that the cost of energy will increase significantly in the coming decades. Conventional sewage treatment is expensive and highly energy intensive compared with wetland assimilation. Economic cost benefit analyses of wastewater treatment operations at the Breaux Bridge and Thibodaux assimilation wetlands (Breaux 1992; Breaux and Day 1994; Breaux et al. 1995; Ko et al. 2004, 2012) conservatively estimated capitalized cost savings using wetland assimilation rather than conventional tertiary treatment (Table 2.7).

A study of the feasibility of using wetlands for assimilation of shrimp processing wastewater also demonstrated significant cost savings (Day et al. 1998; Cardoch 2000). The avoided cost estimate approach was used to compare costs of conventional on-site treatment of the shrimp processing effluent by the dissolved air flotation method with the cost of wetland assimilation. The annualized cost of the conventional treatment calculated to $214,000 per year, as compared to wetland assimilation costs of $63,000 per year, for a potential cost savings of $1,500,000 over 25 years (Day et al. 2004).

In conventional treatment, for every unit of carbon of organic matter oxidized in BOD reduction two to three units of carbon are released to the atmosphere as CO_2 from the burning of fossil fuels. Virtually no fossil fuels are needed for wetland assimilation. Thus, wetland assimilation has a much lower greenhouse gas impact

Table 2.7 Cost comparisons for three wetland assimilation projects (Day et al. 2004)

Site	Cost of conventional treatment	Cost of assimilation wetland	Cost savings
Breaux Bridge[a]	$3,300,000	$664,000	$2,636,000
Thibodaux[b]	$1,650,000	$1,150,000	$500,000
Dulac[c]	$2,200,000	$700,000	$1,500,000

[a]Costs reported in 2000 dollars. Capitalized costs are discounted at 7% for 20 years
[b]Costs reported in 1992 dollars. Capitalized costs are discounted at 9% for 30 years
[c]Costs reported in 1995 dollars. Capitalized costs are discounted at 8% for 25 years

than conventional tertiary treatment systems. Wetland assimilation also offsets CO_2 production through significant carbon sequestration from increased above- and belowground production. Sequestration of carbon, as soil organic matter, is especially significant in subsiding areas like the Mississippi River Delta.

An Overview of Wetland Assimilation Systems in Louisiana

Selected Case Studies: Thibodaux and Amelia, Louisiana

Thibodaux

Although the Thibodaux assimilation wetland is not the longest functioning system, it is one of the most intensively studied wetlands, with 3 years of baseline study (1989–1992) before discharge began. Additional studies and monitoring following the guidelines outlined in the LPDES permit have continued to the present.

The Pointe au Chene wetland, located 10 km southwest of Thibodaux, Louisiana, is a 231-ha subsiding baldcypress-water tupelo swamp on the back slope of Bayou Lafourche, a former distributary of the Mississippi River that was cut off in 1904 (Fig. 2.8). Historically, Bayou Lafourche carried an average of 12% of the

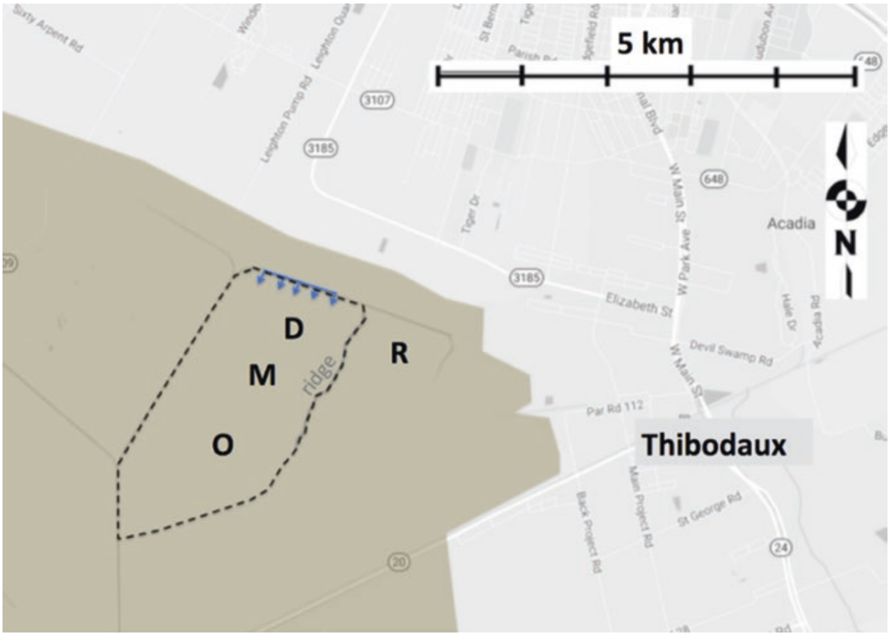

Fig. 2.8 Location of monitoring sites at the Thibodaux assimilation wetland. *R* Reference site, *D* Discharge site, *M* Mid site, *O* Out site. The *blue lines* indicate location of the discharge pipe and outlets, and the *black dashed line* the assimilation wetland

Mississippi River, or about 1100 m^3/s. Prior to the dredging of the Terrebonne-Lafourche drainage canal the wetland received upland runoff from the natural levee of Bayou Lafourche. Now, the spoil bank along the canal prevents any upland runoff from entering the site. The canal is directly connected to coastal waters so that water levels at the site are affected by coastal water levels (Conner and Day 1988). The area was further altered by the construction of a road for access to an oil drilling site on the western side of the assimilation wetland. The area is also bisected by a minor distributary ridge from Bayou Lafourche which separates the assimilation wetland from the reference wetland. The Reference area is about 10 cm higher and the ridge is about 40 cm higher than the assimilation wetland. Because of the hydrological changes, this area only received rainwater prior to effluent discharge. All flow from the area leaves via a 100-m wide shallow wetland channel between the access road and the ridge. There are three monitoring sites in the assimilation wetland that follow the flow of surface water from the discharge of treated effluent to where surface water leaves the wetland (termed Discharge, Mid, and Out sites), along with a reference wetland monitoring site.

The soils are classified as Fausse (very fine, montmorillonitic, nonacid, thermic typic fluvaquents) and effectively restrict groundwater exchange (Zhang et al. 2000). Over the past decades the study area experienced increased flooding due to subsidence and isolation from outside freshwater inputs and transitioned from bottomland hardwood forest to baldcypress-water tupelo swamp. The area immediately adjacent to the effluent input was a shallow, treeless, open water area cleared for the construction of a power line right of way. The dominant woody vegetation is baldcypress (*Taxodium distichum*) and water tupelo (*Nyssa aquatica*), with some black willow (*Salix nigra*) and swamp red maple (*Acer rubrum* var. *drummondii*). Because it is impounded and permanently flooded, there has been no forest regeneration in the Thibodaux assimilation wetland. Forested wetlands in the Verret basin, of which the Thibodaux wetland is part, began flooding in the early 1970s and by the late 1980s, most forested wetlands in the basin were permanently flooded (Conner and Day 1988). Relative sea level rise in the basin now exceeds 1.0 cm/year.

In a recent paper, Conner et al. (2014) analyzed water level changes in the Verret basin from 1986 to 2009 and concluded that the combination of rising water levels, hurricanes, and altered hydrology is fundamentally changing the structure of the forested wetland community. They concluded that the number of baldcypress trees is decreasing over time due to the loss of adult trees and lack of recruitment, and as Chinese tallow (*Triadica sebifera*) and red maple (not the major forest canopy trees one commonly associates with forested wetlands) continue to become common in the canopy through time, the system is losing its “swamp” character. Thus, it is important to understand that at the beginning of effluent discharge, the site was a fundamentally altered system with permanent flooding, stagnant conditions, and a dying bottomland hardwood community. In the 25 year history of the site, relative water level rise has been about 27 cm.

During the EBS study, surface water height was measured monthly at Discharge and Reference sites and mean water levels were about 10 cm deeper at the Discharge site, reflecting the difference in base elevations of the two sites. Measurements were not made for 2 years during the permit application and review process, but were restarted when effluent discharge began. Water levels at both the Discharge and Reference sites increased by 15–20 cm during this time. This was a reflection of hurricane Andrew in August 1992, ongoing relative water level rise, and most importantly, to higher rainfall in the several years after discharge began. A Before-After-Control-Impact (BACI) analysis showed that there was a significant increase in water levels at both sites but no difference between the Discharge and Reference areas, indicating that the increase was not due to the discharge of treated effluent (Rybczyk et al. 2002).

Mean net primary productivity (NPP) was higher at the Reference area compared to the Discharge area during the 2 years prior to effluent discharge. Productivity was likely affected by higher surface water levels and periods of inundation at the Discharge area than at the Reference area, but the most important factor was higher tree density at the Reference area. Although NPP decreased at both Discharge and Reference areas during the 2 years following discharge, the Discharge area still had lower productivity. Rybczyk et al. (1995) showed that decreased productivity at both sites was due to Hurricane Andrew, the eye of which passed within 80 km of the site. Tree mortality was higher and litterfall was lower at both sites due to the hurricane. NPP declined slightly but not significantly at the Reference site, while NPP increased slightly at the Discharge site. Because of the robust monitoring and additional measurements carried out after the hurricane, the impacts of the hurricane and the discharge of treated effluent were able to be separated.

Rybczyk et al. (2002) measured litter decomposition and accretion over feldspar marker horizons before and after discharge began. BACI statistical analysis revealed that neither leaf litter decomposition rates nor initial leaf litter nitrogen and phosphorus concentrations were affected by discharge of treated effluent. A similar analysis revealed that final nitrogen and phosphorus leaf litter concentrations did significantly increase in the Discharge site relative to the Reference site after effluent discharge began. Total pre-effluent accretion, measured 34 months after feldspar horizon markers were laid down, averaged 22.3 ± 3.2 mm and 14.9 ± 4.6 mm at the Discharge and Reference sites, respectively, and were not significantly different. However, total accretion measured 68 months after the markers were installed and 29 months after effluent discharge began averaged 54.6 ± 1.5 mm at the Discharge site and 19.0 ± 3.2 mm at the Reference site and were significantly different. Additionally, after discharge of treated effluent began, the estimated rate of accretion in the Discharge site (11.4 mm year^{-1}) approached the estimated rate of relative sea level rise (12.3 mm year^{-1}). Most of this increased accretion was attributed to organic matter inputs, as organic matter accumulation increased significantly at the Discharge site after effluent application began, while mineral accumulation rates remained constant. These findings indicate that there is a potential for using treated effluent to balance accretion deficits in subsiding wetland systems.

Ten years after the discharge of treated effluent began in 1992, switchgrass (*Panicum virgatum)* became established along the wetland boundary where effluent was discharged and by 2003 began to extend into the wetland. Within 4 years a highly productive emergent wetland developed with floating marsh characteristics.

Izdepski et al. (2009) studied the dynamics of this floating marsh community. They reported that NPP, total belowground biomass, NO_3, and plant-tissue $\delta^{15}N$ ratios varied significantly along a 75-m marsh transect, while mean plant-tissue $\delta^{13}C$ values differed between the dominant species. The area nearest the effluent discharge had the highest NPP (3876 g/m^2/year), total belowground biomass (4079 ± 298.5 g/m^2), and mean NO_3 (5.4 ± 2.9 mg/L). The mean $\delta^{15}N$ of pennywort (*Hydrocotyle umbellate*) floating marsh was less enriched at 0–75 m (9.7 ± 1.9%) compared to 100–200 m (21.0 ± 3.8%). The $\delta^{13}C$ of the belowground peat mat of the floating marsh was similar to switchgrass but not pennywort, indicating that switchgrass was forming the mat. Nutrient availability affected NPP and $\delta^{15}N$. NPP was greater than most reported values for floating marsh from 0 to 45 m then decreased along with NO_3 concentrations and $\delta^{15}N$ further from the effluent source. The herbaceous wetlands still persist. These results suggest that nutrient rich fresh water can promote restoration of some floating marshes.

Rybczyk et al. (1998) developed a wetland elevation/sediment accretion model for the site to determine if addition of treated municipal effluent could stimulate organic matter production and deposition to the point that sediment accretion would balance relative sea level rise. They simulated the effect of predicted increases in eustatic sea level rise (ESLR) on wetland stability and determined the amount of additional mineral sediment that would be required to compensate for relative sea level rise. The model also simulated primary production (roots, leaves, wood, and floating aquatic vegetation) and mineral matter deposition, both of which contribute to changes in elevation. Simulated wetland elevation was more sensitive to estimates of deep subsidence and future ESLR rates than to other processes that affect wetland elevation (e.g., rates of decomposition and primary productivity). The model projected that although the addition of treated effluent would increase long-term accretion rates from 0.35 to 0.46 cm/year, it would not be enough to offset the current rate of relative sea level rise. A series of mineral input simulations revealed that, given no increase in ESLR rates, an additional 3000 g/m^2/year of mineral sediments would be required to maintain a stable elevation.

Keim et al. (2012) used tree-ring analysis to evaluate the combined effects of rising water levels and 13 years of municipal effluent addition on baldcypress growth at the Thibodaux assimilation wetland. Trees at the Discharge, downstream outflow, and adjacent Reference areas all experienced increased growth coinciding with a period of widespread rapid subsidence and water level increases in the late 1960s. Tree growth at the Discharge and outflow sites began to decrease before discharge of treated effluent began in 1992, and afterward was apparently unaffected by effluent discharge. In contrast, trees at the Reference site have not experienced growth declines. Hydrological changes caused by subsidence have apparently overwhelmed any effect of treated effluent on baldcypress growth. Increasing inundation may have increased growth initially by eliminating competition from species less tolerant of inundation; however, after a decade of sustained flooding, growth declined steadily. Release of baldcypress from competition continues at the topographically higher Reference site, but growth will likely subsequently decrease as ongoing subsidence and ESLR cause more prolonged inundation. These data suggest that short-term increases in water level and nutrients stimulated growth of baldcypress, but long-term increased inundation was a net stressor, and was more important than nutrient limitations in controlling growth at the Discharge site.

From 2010 to 2014, researchers at Nicholls State University, LA carried out a series of detailed studies on productivity, biogeochemistry, and benthic population dynamics (Minor 2014). They reported that water levels were higher at the Discharge site compared to the Reference, but were similar at the Out site. The initial studies in the early 1990s showed that the Discharge site had higher water levels because the soil surface was lower. The development of the flotant marsh at the site impedes flow resulting in a higher water level near the inlet (Izdepski et al. 2009). Monitoring indicates that the wetland is still reducing nutrients to background levels after 25 years of operation. Forest productivity measured on an aerial basis was lower at the Discharge site but was similar at Mid, Out, and Reference sites. The lower productivity at the Discharge site was due to low tree density. Minor (2014) noted that by 2014 almost all bottomland hardwood species had died, a process that had begun before effluent discharge. If the flotant marsh is taken into consideration, the productivity of the Discharge site is higher than the Reference and Out sites. Macrofaunal assemblages were different between the Reference and Discharge sites, likely as a result of differences in surface water levels and the resulting vegetation community structure.

Amelia

The Ramos Swamp assimilation wetland is a continuously flooded tidal freshwater forested wetland located south of Lake Palourde, approximately 2 km north of Amelia, Louisiana (Fig. 2.9). Although treated municipal effluent discharge began in 1973, the system was not permitted until 2007. The City uses a 13.4 ha oxidation pond with a chlorination/dechlorination system for sewage treatment. Since this wetland is over 40 km from the coast, daily water level fluctuations are less than a few cm. The swamp area directly affected by the effluent flow is 77 ha within a larger forested wetland area of over 1000 ha. The dominant vegetation is baldcypress, water tupelo, black willow, swamp red maple, and green ash (*Fraxinus pennsylvanica*). Because the forest does not have a completely closed canopy, there is floating aquatic vegetation dominated by mosquito fern (*Salvinia minima*), duckweed (*Lemna minor*), watermeal (*Wolffia* sp.), pennywort (*Hydrocotyle ranunculoides*), and water lettuce (*Pistia stratiotes*). Submerged aquatic vegetation is predominantly hornwort (*Ceratophyllum* sp.). The flooded soils consist of well consolidated riverine clay, overlain by high organic clays and topped by a poorly consolidated 30–60 cm layer of plant detritus (Lytle et al. 1959). There are three monitoring sites in the assimilation wetland along the flow of surface water from the discharge of treated effluent to where surface water leaves the wetland (termed Discharge, Mid, and Out sites) and a nearby Reference wetland that is not affected by treated effluent.

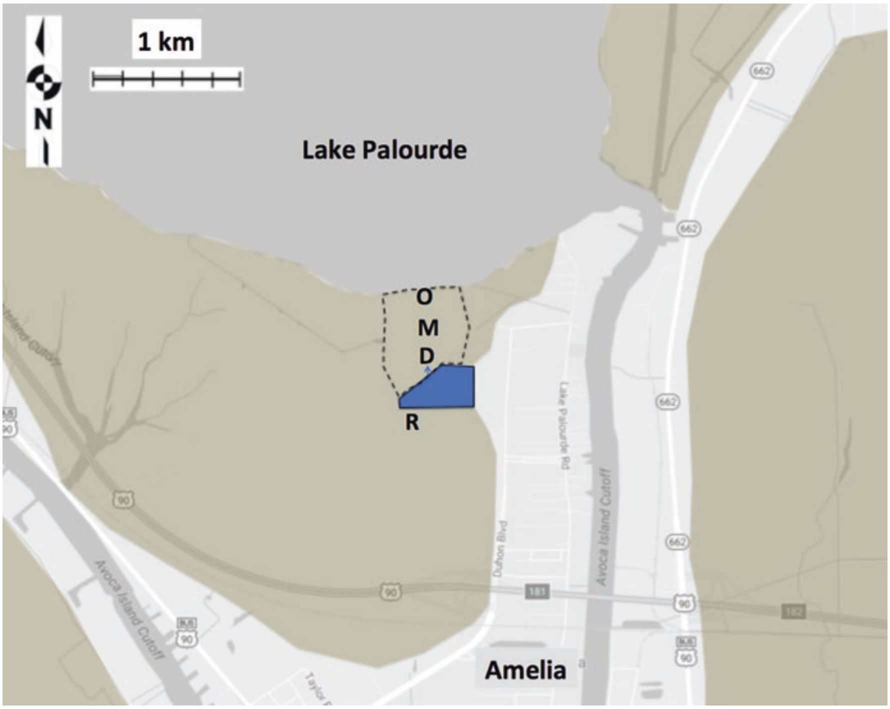

Fig. 2.9 Location of monitoring sites at the Amelia Ramos Swamp assimilation wetland. *R* Reference site, *D* Discharge site, *M* Mid site, *O* Out site. The water body north of the site is Lake Palourde. The *blue* polygon indicates the location of the oxidation pond, and the *black dashed line* the assimilation wetlands

Day et al. (2008) measured surface water nutrient concentrations at Discharge and Reference sites. TKN concentrations (2.0–4.0 mg/L) accounted for almost 75% of TN. NH_4-N was about 25% of TN and ranged from 0.4–1.0 mg/L. NO_x-N was generally less than 1% of TN. Within the wetland, PO_4 concentrations (0.1–0.9 mg/L) were about 50% of TP. TN and TP were reduced by about 79% and 88%, respectively, as water flowed through the wetland. These removal rates are consistent with low loading rates of 9.4 g TN and 1.2 g TP/m^2/year.

Day et al. (2008) also measured vegetation productivity and benthic community structure at assimilation and Reference wetlands. Litterfall was significantly greater at the Discharge site (717 g/m^2/year) compared to the Reference site (412 g/m^2/year). Stem growth ranged from 302 to 776 g/m^2/year and was not statistically different among the Reference, Discharge, and Out sites. Total NPP was highest at the Discharge and Out sites (1467 and 1442 g/m^2/year, respectively) and these values were significantly higher than NPP values at the Reference site (714 g/m^2/year). Total individuals, total species, and species richness of macroinvertebrates was greatest near the effluent outfall and declined away from the discharge. The long-term addition of secondarily treated municipal effluent resulted in a high level of nutrient retention, enhanced forest productivity, and minimal impact on benthic community structure.

Long-Term Monitoring of Assimilation Wetlands in Louisiana

Breaux Bridge Cypriere Perdue

The city of Breaux Bridge has been discharging secondarily-treated municipal effluent into the Cypriere Perdue swamp since the late 1940s. The city treatment system includes three oxidation ponds and a chlorination-dechlorination system with the capacity to treat 1.0 MGD flow. From 2001 to 2013, average monthly discharge into the assimilation wetland was 0.96 MGD and average concentrations of TN and TP were 8.44 and 2.42 mg/L, respectively, with mean TN and TP loading rates during this time of 1.89 and 0.24 g/m^2/year, respectively (Table 2.8).

The Cypriere Perdue swamp is a 1470 ha baldcypress-water tupelo wetland and bottomland hardwood forest located in St. Martin Parish, 3.5 km west of Breaux Bridge, Louisiana. The wetland is dominated by water tupelo, baldcypress, swamp red maple, black willow, and Chinese tallow, as described by Hesse et al. (1998). Under natural conditions, flow from the area was to the south with some flow likely going to the Vermillion River. During high water periods, backwater flooding from the Vermillion can raise water levels at the site by over 2 m. The Ruth Canal now connects the Vermillion River with Bayou Teche, a former distributary of the Mississippi River, and almost no flow from the site goes south of the Canal. The original location of the Reference site was moved because high water levels caused effluent to flow into the area and the Out site was moved because short-circuiting caused effluent to by-pass the area (Fig. 2.10).

Blahnik and Day (2000) studied the hydrology and nutrient loading rates at the Breaux Bridge assimilation wetland. Pond discharge, surface water elevations, and fluorescent dye travel times were recorded to assess surface water hydrology, and water samples were collected for NO_x, NH_3, PO_4, and TSS analyses. Wetted surface area increased with pond discharge rate, and 58–66% of surface water flow was concentrated in shallow channels covering only 10–12% of the total study area. Hydraulic retention time was much longer (0.9–1.1 days) than minimum dye travel times (2–3 h) through the 4 ha study area. Higher pond discharge rates created more treatment surface area, and higher constituent loading rates produced higher removal rates. They concluded that higher nutrient loads could be assimilated without requiring significant increases in wetland area.

Monitoring between 2001 and 2013 showed that mean TN concentrations declined from the Discharge site to the Out site and concentrations at the Out site were actually lower than at the Reference site (Fig. 2.11). The type of nitrogen, or

Table 2.8 Mean effluent discharge, TN and TP concentrations in effluent, and loading rates for assimilation wetlands in Louisiana

Municipality	Years for data summary	Mean discharge (MGD)	Mean TN (mg/L)	Mean TP (mg/L)	Mean TN loading rate ($g/m^2/year$)	Mean TP loading rate ($g/m^2/year$)	% of TN in treated EFFLUENT		
							NH_3	NO_x	Org N[c]
Breaux Bridge	2001–2013	0.96	8.44	2.42	1.89	0.24	49.6	6.1	44.3
Broussard	2007–2013	0.59	24.64	3.45	14.75	2.62	69.3	9.2	21.5
Hammond	2007–2013	3.90	17.91	3.64	2.39	0.48	52.6	32.5	14.9
Luling	2006–2013	1.58	7.06	2.34	2.52	0.84	29.2	9.3	61.5
Mandeville BC[a]	2006–2013	1.19	14.36	3.31	56.50	13.90	43.2	40.2	16.6
Mandeville TM[b]	2009–2013	1.44	15.52	3.02	7.48	1.46			
St. Martinville	2011–2013	0.74	5.40	1.85	8.70	3.00	29.3	20.0	50.7

[a]Mandeville Bayou Chinchuba
[b]Mandeville Tchefuncte Marsh
[c]Organic Nitrogen

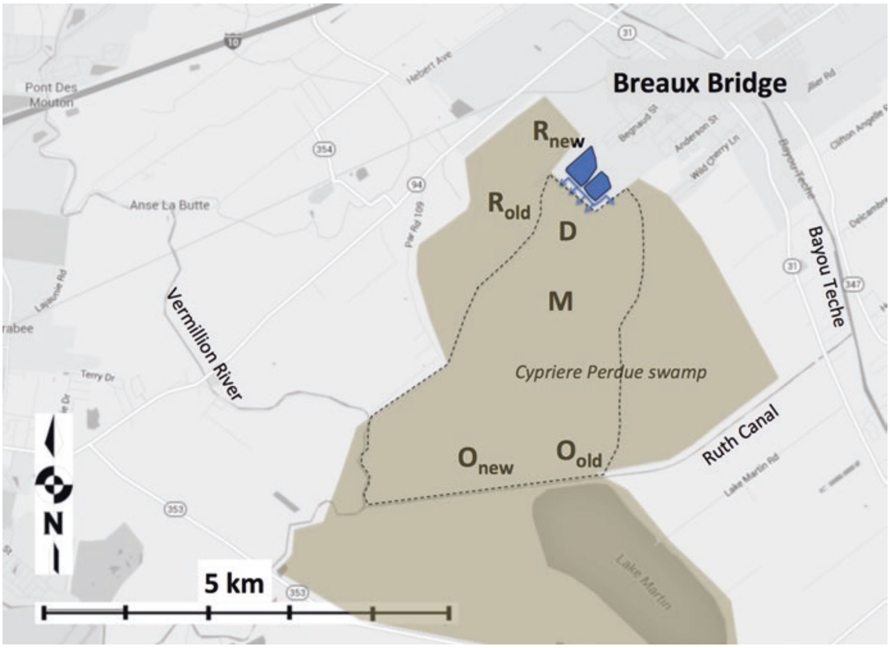

Fig. 2.10 Location of wetland monitoring sites at the Cypriere Perdue swamp, Breaux Bridge, Louisiana. R_{old} Original Reference site, R_{new} New Reference site, *D* Discharge site, *M* Mid site, O_{old} Original Out site, and O_{new} New Out site. The *blue* polygons indicate the location of the oxidation ponds, *blue lines* the discharge pipe and outlets, and the *black dashed line* the assimilation wetlands

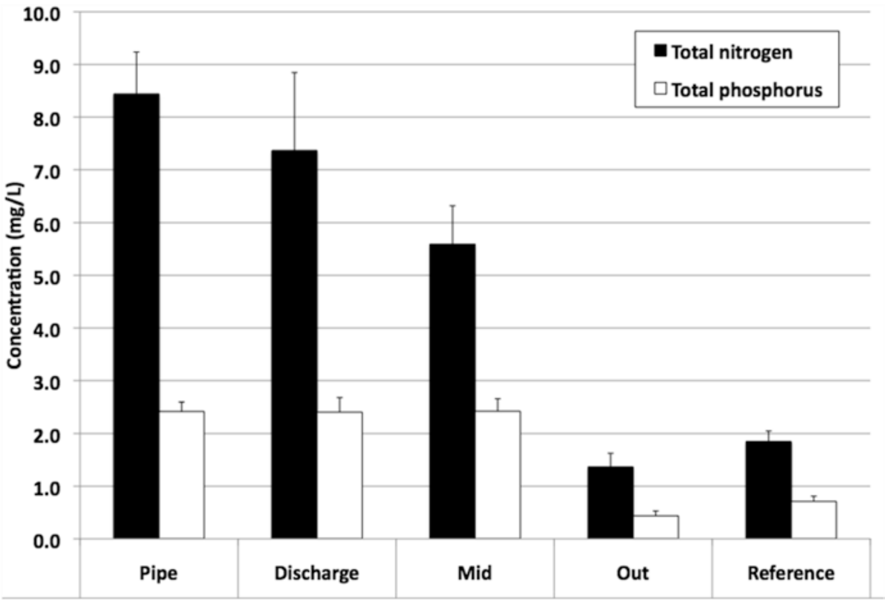

Fig. 2.11 Mean total nitrogen and total phosphorus concentrations at the effluent pipe and in surface water at the Breaux Bridge assimilation wetland between 2001 and 2013. Error bars represent standard error of the mean

nitrogen species, impacts nutrient removal because NO_x rapidly diffuses into anoxic soil layers where it is used as an electron acceptor and reduced to gaseous end products such as nitrous oxide and nitrogen gas (Reddy and DeLaune 2008). Thus, NO_x is removed more rapidly than NH_3, which must first be nitrified before being denitrified. Of the TN concentration, 49.6% is NH_3, 6.1% is NO_x, and 44.3% is organic nitrogen. Since NH_3 is a much larger percentage of TN than NO_x at this assimilation wetland, TN concentrations did not drop as rapidly as seen in other assimilation wetlands where the effluent is highly nitrified (i.e., Hammond, Mandeville Bayou Chinchuba and Tchefuncte Marsh assimilation wetlands). TP concentrations decreased to background conditions at the Out site (Fig. 2.11). Phosphorus is typically removed through vegetation uptake and abiotic retention in soils but it has no permanent removal mechanism such as denitrification for nitrogen (Reddy and DeLaune 2008).

Because of differences in tree density among sites, mean woody or litterfall productivity for each site is divided by the number of trees to determine mean productivity per tree. Mean litterfall and woody productivity are added together to determine mean NPP per tree. At the Breaux Bridge assimilation wetland, mean annual NPP in the Discharge and Reference wetlands ranged between about 30 and 135 g/m^2/year per tree from 2002 to 2015 (Fig. 2.12). In general, litterfall productivity was a higher percentage of NPP than woody productivity.

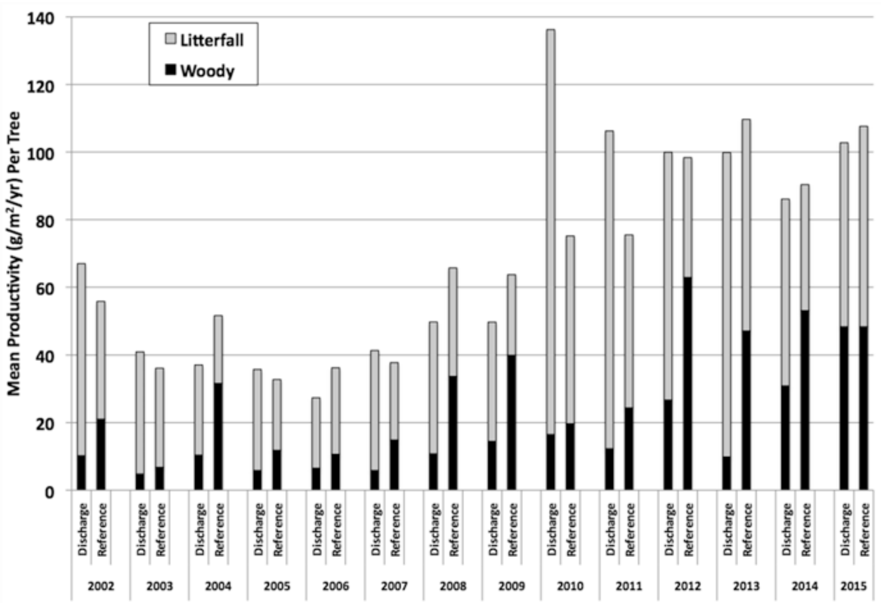

Fig. 2.12 Mean annual litterfall and woody productivity at the Breaux Bridge Cypriere Perdue assimilation wetland

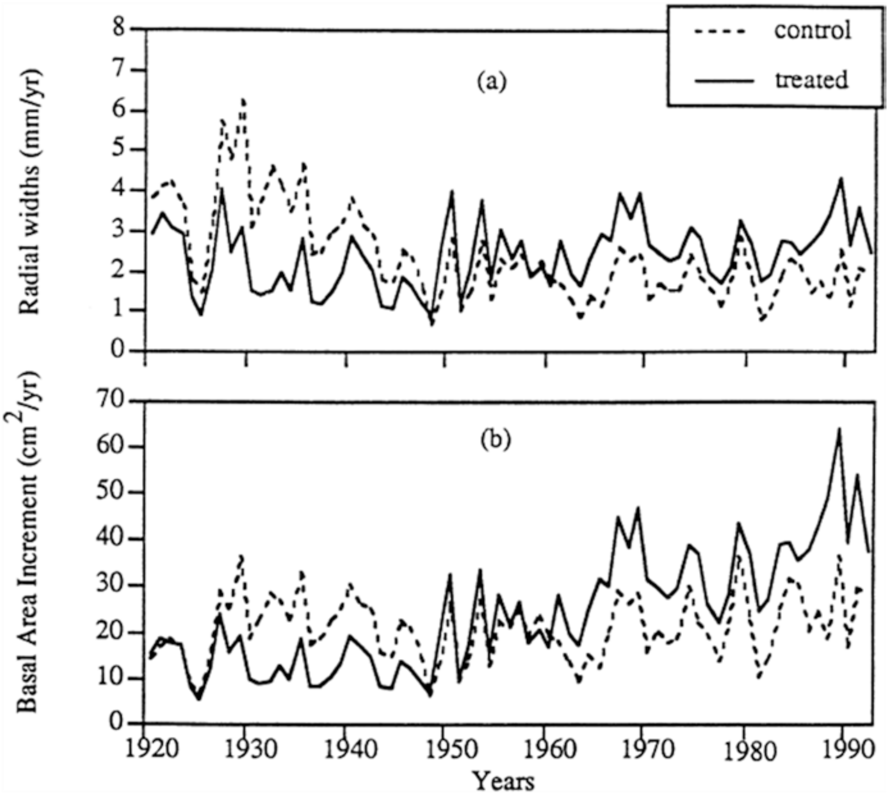

Fig. 2.13 Mean annual ring width chronologies of (**a**) diameter increment (DINC); and (**b**) basal area increment (BAI) for baldcypress in the Cypriere Perdue swamp (Hesse et al. 1998). Discharge of treated effluent began in the late 1940s

Long-term tree ring analysis of baldcypress growth at the site provides a context within which short-term variability can be interpreted (Hesse et al. 1998). Results of tree ring analysis show that after effluent discharge to the swamp began the late 1940's, growth rate was consistently higher in the Discharge area than in the Reference area (Fig. 2.13; Hesse et al. 1998). There was also a high degree of variability in these data, indicating that swamp productivity can vary dramatically. Growth of baldcypress is lower during periods of warm spring weather and drought (Stahle and Cleaveland 1992; Cleaveland 2006; Keim and Amos 2012). Day et al. (2012) reported that growth rates of baldcypress in two sites in the Pontchartrain Basin were most strongly correlated with May Palmer Drought Severity Index (PDSI). This high variability in year-to-year growth rates of baldcypress must be taken into consideration when interpreting forested wetland growth rates at assimilation wetlands because they are so strongly affected by climatic variability.

Broussard Cote Gelee

The City of Broussard discharges secondarily-treated effluent into 300 acres of the Cote Gelee forested wetlands via two outlets that can be operated independently (Fig. 2.14). The city treatment system includes three oxidation ponds and a chlorination-dechlorination system with the capacity to treat 1.0 MGD flow. From 2007 to 2013, average monthly discharge into the assimilation wetland was 0.59 MGD and average concentrations of TN and TP were 24.64 and 3.45 mg/L, respectively, with mean TN and TP loading rates during this time of 14.75 and 2.62 g/m^2/year, respectively (Table 2.8).

The Broussard Cote Gelee assimilation wetland is primarily a baldcypress-water tupelo swamp, but in the slightly more elevated parts of the area there is a mixed forest with bottomland hardwood species, such as pumpkin ash (*Fraxinus profunda*), water hickory, swamp red maple, and water elm (*Planera aquatica*). Under natural conditions, surface water from uplands adjacent to the site flowed

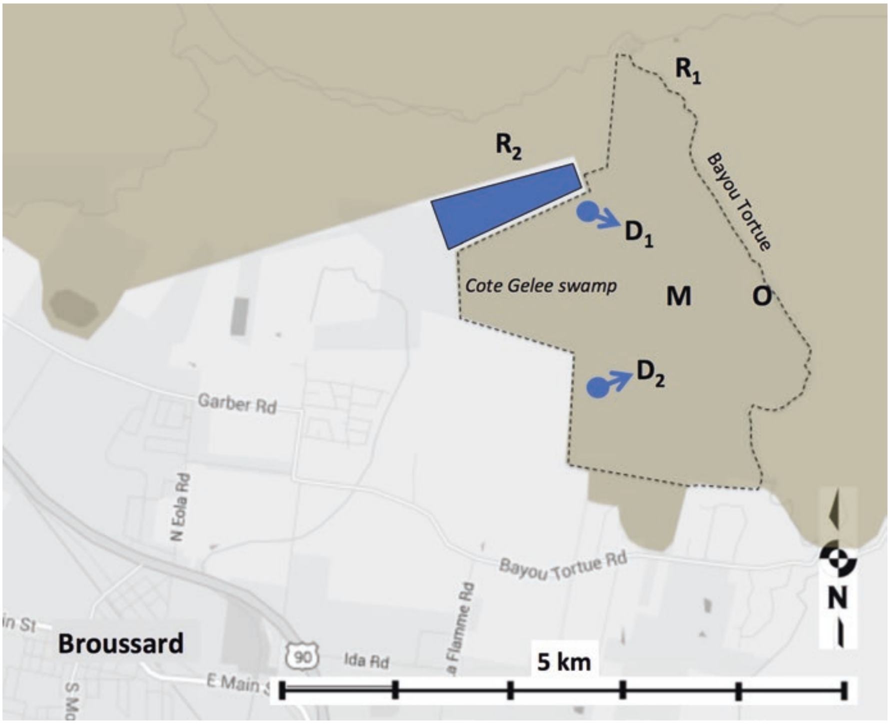

Fig. 2.14 Location of wetland monitoring sites at the Cote Gelee swamp, Broussard, Louisiana. R_1 First Reference site, R_2 Second Reference site, D_1 First Discharge site, D_2 Second Discharge site, *M* Mid site, *O* Out site. The *blue* polygon indicates the location of the oxidation ponds, the *blue arrows* the outlet pipes, and the *black dashed line* the assimilation wetlands

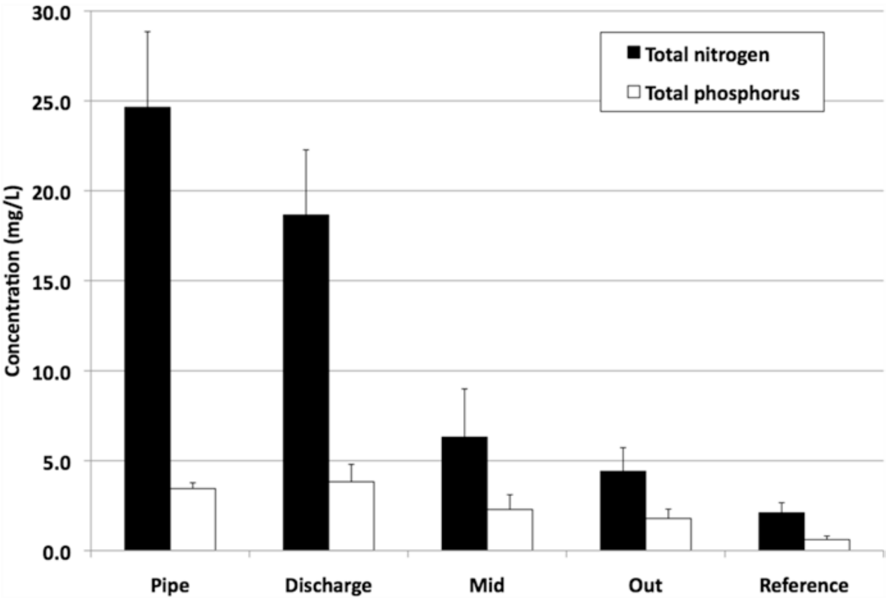

Fig. 2.15 Mean total nitrogen and total phosphorus concentrations measured at the effluent pipe and in surface water at the Broussard assimilation wetland between 2007 and 2013. Error bars represent standard error of the mean

through the forested wetlands and then into Bayou Tortue. A road embankment and a shallow dredged canal have altered water flow through the site and led to the site being over-drained with well-oxidized soils and a high level of subsidence due to soil oxidation. Exposed roots throughout the region suggest the soil surface has subsided one to 2 ft and this condition could lead to a massive blow-down of the forest during a major storm passage. There are six sites where monitoring data were collected (Fig. 2.14). The discharge of treated effluent is switched every 2 months between the Discharge 1 and Discharge 2 sites to prevent prolonged inundation.

Mean TN and TP concentrations of surface water at the study sites declined from the Discharge site to the Out site (Fig. 2.15). Treated effluent entering the Broussard assimilation wetland is 69.3% NH_3, 9.2% NO_x, and 21.5% organic nitrogen. Like the Breaux Bridge assimilation wetland, the high NH_3 concentration may explain why TN concentration was still fairly high in surface water at the Discharge site. By the time surface water reached the Mid site, however, mean TN concentration had dropped by about 75%.

Mean annual NPP was typically much higher at the Discharge site than at the Reference site at the Broussard assimilation wetland (Fig. 2.16). Data shown for the Reference site are the annual average for the two Reference sites that are monitored.

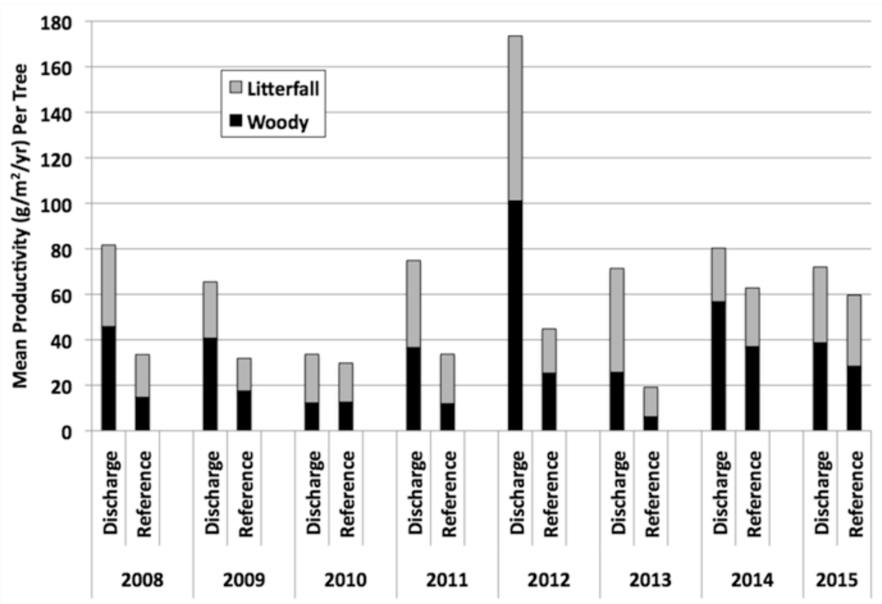

Fig. 2.16 Mean annual litterfall and woody productivity at the Broussard Cote Gelee assimilation wetland

St. Martinville

The City of St. Martinville began discharging secondarily-treated municipal wastewater from the city treatment facility to the Cypress Island Coulee wetlands in 2011. The St. Martinville wastewater treatment facility has a maximum design flow of 1.5 MGD, and consists of a 63.7-ha facultative lagoon with ultraviolet disinfection and a cascade aeration structure. From 2011 to 2013, the system had an annual average flow of 0.74 MGD and average concentrations of TN and TP were 5.40 and 1.85 mg/L, respectively, with mean TN and TP loading rates during this time of 8.70 and 3.00 g/m^2/year, respectively (Table 2.8).

The Cypress Island Coulee wetlands are located adjacent to the treatment facility and consist primarily of baldcypress-water tupelo swamp and red maple. These wetlands have been degraded by urbanization and conversion of surrounding areas to agriculture and are characterized by over-drained soils and subsidence. Because these wetlands were used for rice and crawfish production, they consist of a number of shallow "ponds" separated by low levees (Fig. 2.17). Secondarily-treated effluent is discharged at six different locations around the South basin of the wetlands and surface water drains into the Cypress Island Coulee after flowing through the wetlands (Fig. 2.17). To monitor the effects of this discharge on the vegetation of the

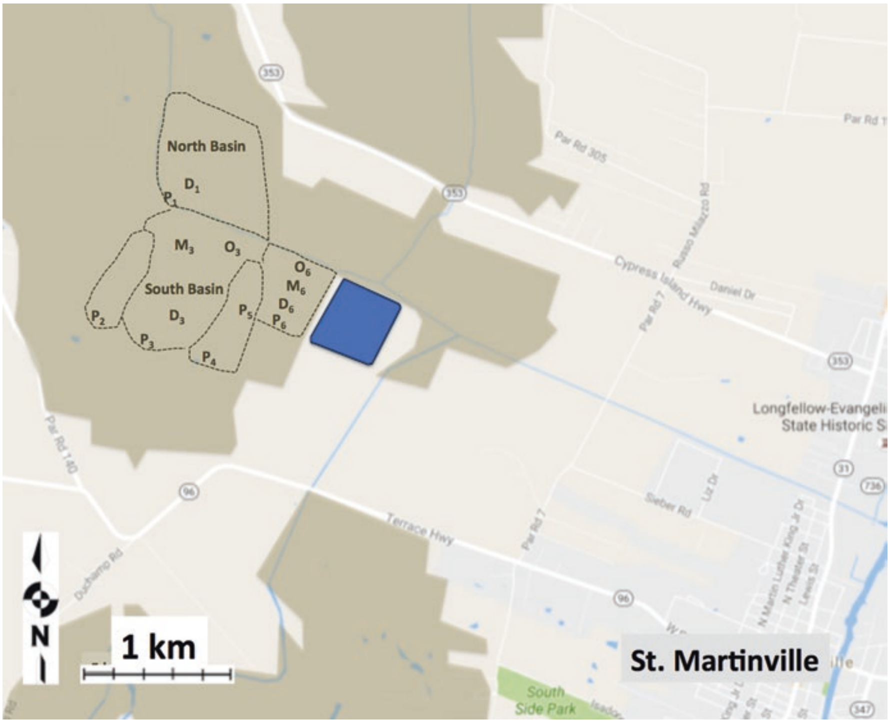

Fig. 2.17 Location of discharge pipes and wetland monitoring sites at the St. Martinville assimilation wetland. *P* pond number, *D* Discharge sites, *M* Mid sites, and *O* Out sites. The *blue* polygon indicates the location of the facultative lagoon, and the *black dashed line* the assimilation wetlands

receiving wetlands, two sets of three study sites (Treatment, Mid, and Out) were established, as well as three Reference sites located in nearby wetlands (One near Breaux Bridge (Fig. 2.10) and two near Broussard, LA (Fig. 2.14)).

Mean TN concentration discharged into the wetland declined from the Discharge site to the Out site (Fig. 2.18) and concentration at the Out site was similar to that measured at the Reference sites. Of the TN treated effluent discharged into the St. Martinville assimilation wetland, 29.3% is NH_3, 20% is NO_x, and 50.7% is organic nitrogen.

Mean annual NPP was typically higher at the Discharge site than at the Reference 2 site and lower than the Reference 1 site at the St. Martinville assimilation wetland (Fig. 2.19). Data shown for the Reference 2 site are the annual average for the two Reference sites monitored at the Broussard assimilation wetland. During most of the years when biomass was monitored, litterfall made up a higher percentage of NPP than woody biomass.

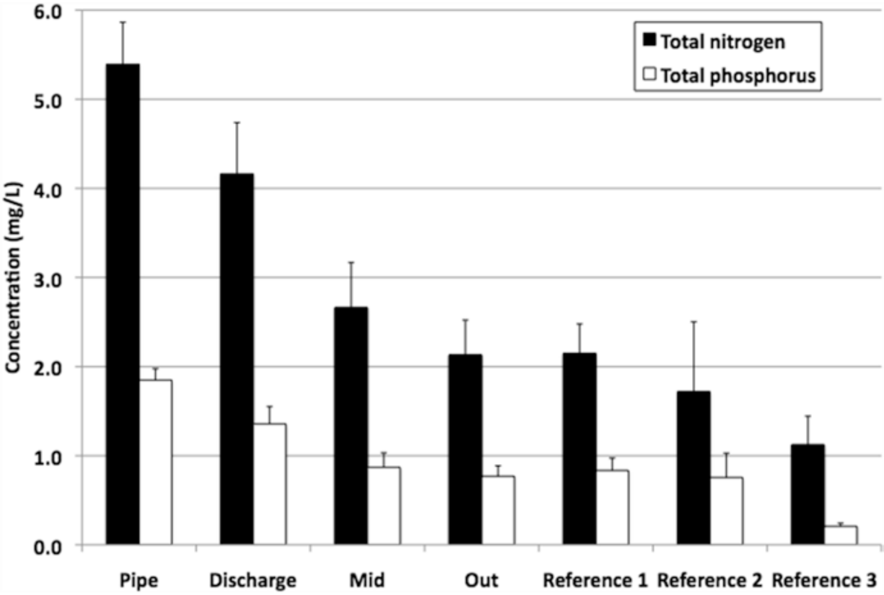

Fig. 2.18 Mean total nitrogen and total phosphorus concentrations at the effluent pipe and in surface water at the St. Martinville assimilation wetland and Reference sites between 2011 and 2015. Error bars represent standard error of the mean

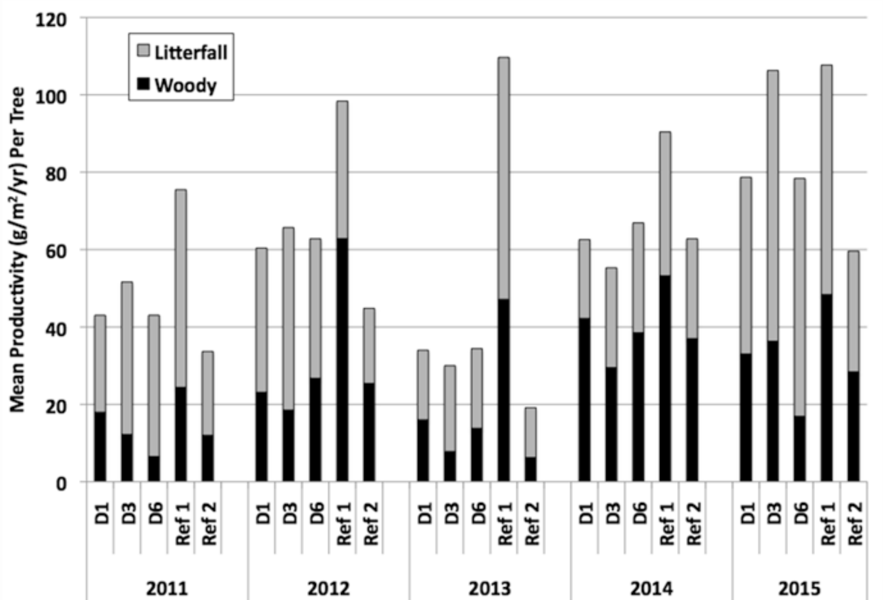

Fig. 2.19 Mean annual litterfall and woody productivity at the Discharge sites (D1, 3, 6) and Reference sites (Ref 1 and 2) at the St. Martinville assimilation wetland

Luling

The city of Luling discharges secondarily-treated effluent into forested wetlands adjacent to the wastewater oxidation pond. The city's treatment system consists of a facultative oxidation pond with ultraviolet disinfection. From 2007 to 2013, average monthly discharge into the assimilation wetland was 1.58 MGD. Effluent had mean concentrations of 7.06 and 2.34 mg/L for TN and TP, respectively, and mean TN and TP loading rates were 2.52 and 0.84 g/m^2/year, respectively (Table 2.8).

The assimilation wetland is located directly to the east of the oxidation pond. The 608-ha wetland is a continuously flooded freshwater forested wetland dominated by water tupelo and baldcypress. The site is located within the Davis Crevasse, a large crevasse splay that was formed in the nineteenth century when the river broke through the flood control levees and deposited a depositional splay of about 150 km^2 (Day et al. 2016b). Under natural conditions, water from the site where the oxidation pond is now located flowed in a southeasterly direction towards Lake Cataouatche through forested wetlands and freshwater marsh. The dredging of Cousins Canal short circuited water flow directly to the lake. The current discharge is a partial return to more normal water flow with the exception of continuous flooding. Three sites were established at the Luling assimilation wetland, including Discharge, Mid, and Out sites, and a Reference site was located nearby. The Discharge, Mid, and Reference sites are forested while the Out site is a freshwater emergent marsh (Fig. 2.20). The marsh Reference site is the same as the Hammond marsh Reference site (described below).

Mean TN and TP concentrations of surface water at the Luling assimilation wetland did not decline between the effluent pipe and the Discharge site but did decrease between the Discharge and Out sites (Fig. 2.21). Like the Breaux Bridge and Broussard assimilation wetlands, nitrogen entering the Luling assimilation wetland is higher in NH_3 than NO_x. The Luling effluent is 29.2% NH_3, 9.3% NO_x, and 61.5% organic nitrogen. Organic nitrogen is removed more slowly than NH_3 or NO_x because it must be decomposed before these constituents are released. Both mean TN and TP concentrations of surface water at the Out site were very similar to those measured at the Forested and Marsh Reference sites.

Mean annual NPP was higher every year for the Discharge site compared to the Reference site at the Luling assimilation wetland (Fig. 2.22). End-of-season-live (EOSL) biomass collection began at the marsh Out site in 2008. Between 2008 and 2015, the Out site typically had higher productivity than the marsh Reference site (Fig. 2.23). During 2013, however, EOSL biomass at the Reference site was greater than at the Out site.

Mandeville Bayou Chinchuba and Tchefuncte Marsh

The City of Mandeville's wastewater treatment system includes three aerated lagoon cells, a three-celled rock reed filter, and an ultraviolet disinfection system. Effluent was discharged into the Bayou Chinchuba assimilation wetland starting in 1998, but

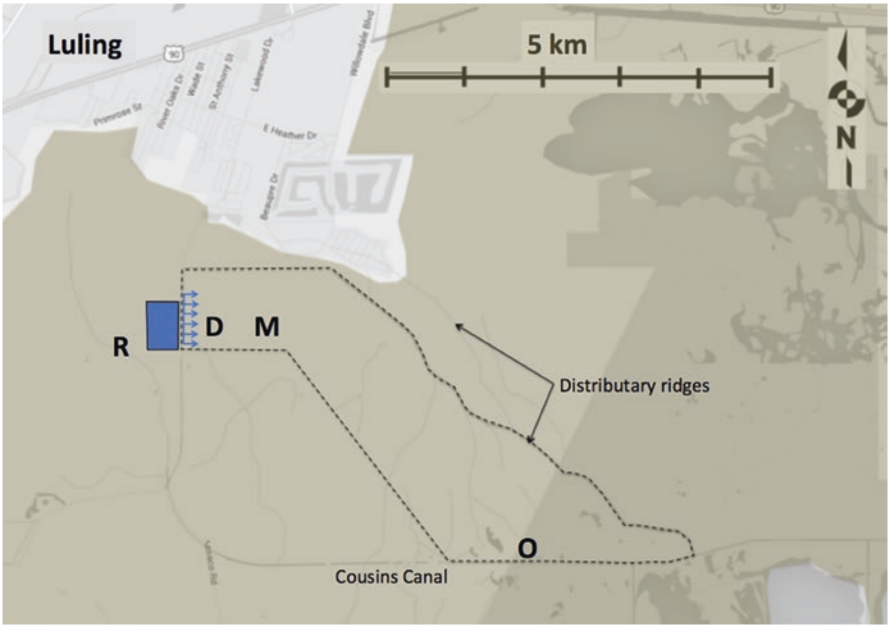

Fig. 2.20 Location of wetland monitoring sites at the Luling assimilation wetland. *R* Reference site, *D* Discharge site, *M* Mid site, *O* Out site. The marsh reference is located at the Hammond assimilation wetland. The distributary ridges were formed in the nineteenth century when the Davis Crevasse deposited river sediments in the area. The *blue* polygon indicates the location of the oxidation pond, *blue lines* the discharge pipe and outlets, and the *black dashed line* the assimilation wetlands

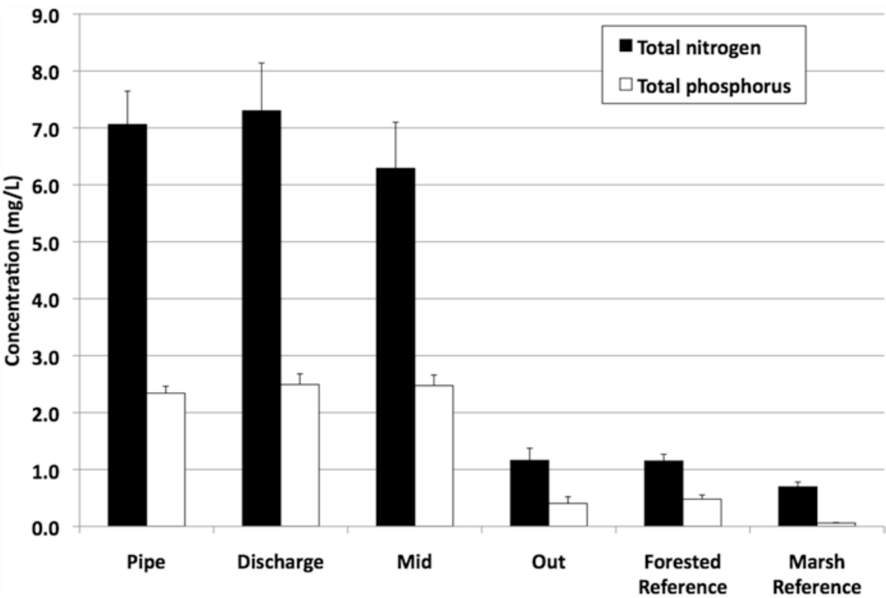

Fig. 2.21 Mean total nitrogen and total phosphorus concentrations at the effluent pipe and in surface water at the Luling assimilation wetland between 2006 and 2013. Error bars represent standard error of the mean

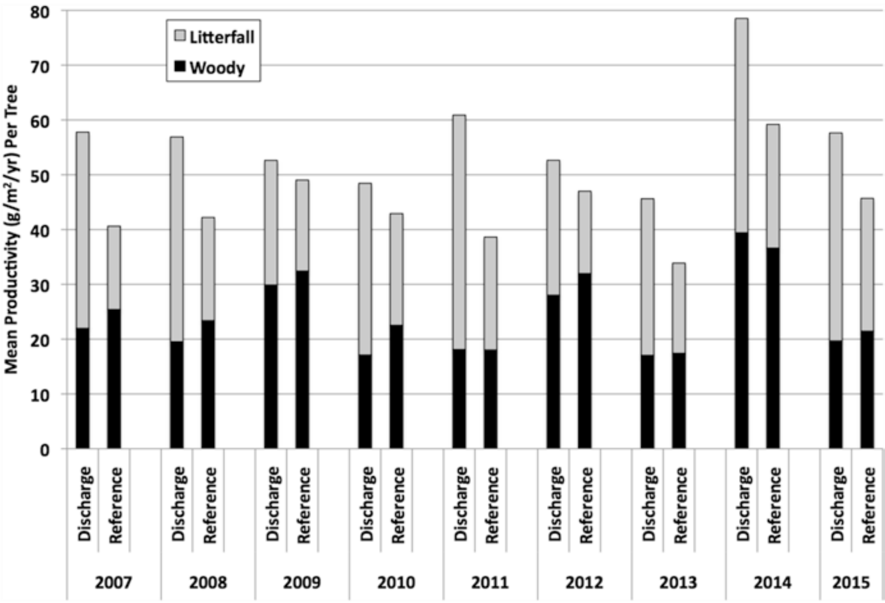

Fig. 2.22 Mean annual litterfall and woody productivity at the Luling assimilation wetland

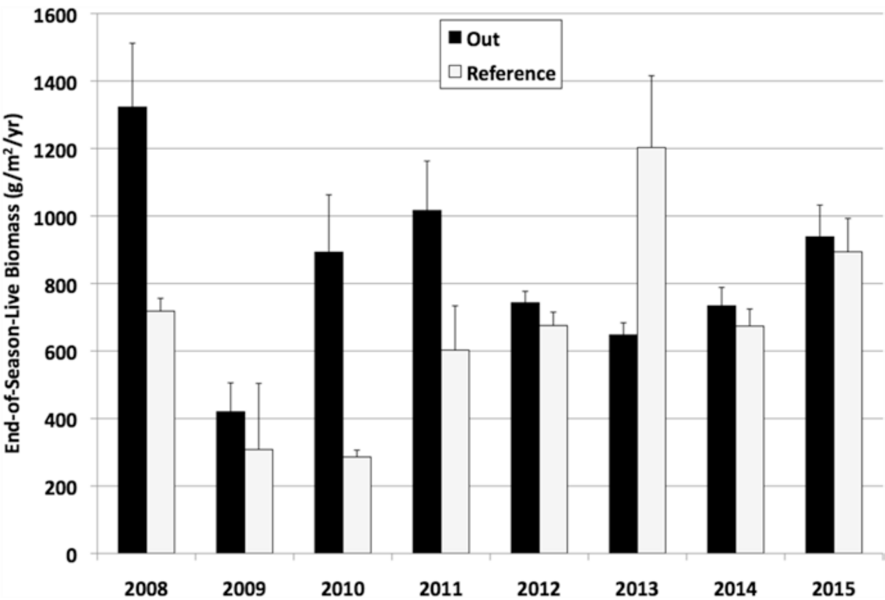

Fig. 2.23 Mean annual end-of-season-live biomass at the Luling assimilation wetland. Error bars represent standard error of the mean

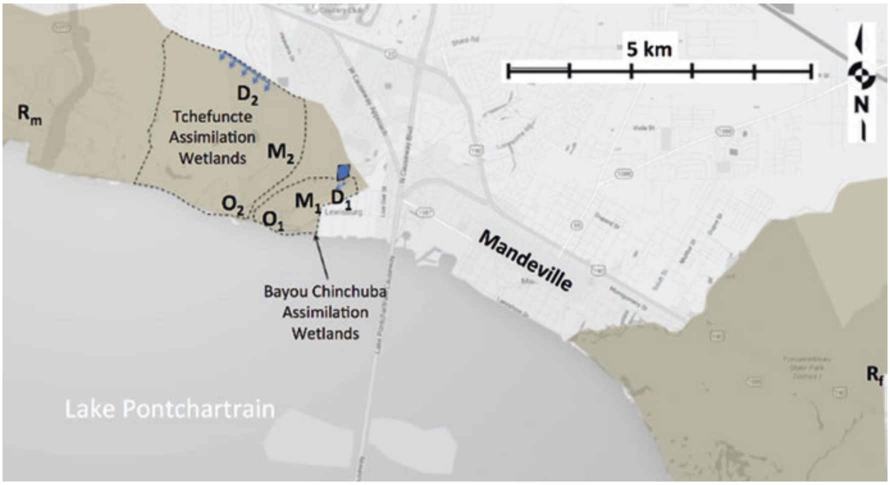

Fig. 2.24 Location of wetland monitoring sites at the Tchefuncte Marsh (TM) and Bayou Chinchuba (BC) assimilation wetlands. R_m Marsh Reference site, R_f Forested Reference site, D_1 BC Discharge site, M_1 BC Mid site, O_1 BC Out site, D_2 TM Discharge site, M_2 TM Mid site, O_2 TM Out site. The *blue* polygon indicates the location of the aerated lagoons, *blue lines* the discharge pipe and outlets, and the *black dashed line* the assimilation wetlands

the LPDES permit was not issued until 2003. Due to high loading rates a second assimilation wetland was established at the Tchefuncte Marsh in 2009 (Fig. 2.24) and effluent was then split between the two assimilation wetlands. Bayou Chinchuba and Bayou Castine (where the forested Reference site is located) are a combination of swamp and bottomland hardwood forest dominated by water tupelo, baldcypress, and swamp blackgum (*Nyssa biflora*). At the Tchefuncte Marsh, the Discharge site is forested, while the Mid, Out, and Reference sites are emergent marsh dominated by creeping panic (*Panicum repens*), dotted smartweed (*Polygonum punctatum*), bulltongue arrowhead (*Sagittaria lancifolia*), and cattail (*Typha latifolia*).

Average monthly effluent discharge into the Bayou Chinchuba assimilation wetland from 2006 to 2013 was 1.19 MGD, with average TN and TP concentrations of 14.36 and 3.31 mg/L, respectively (Table 2.8). Mean loading rates for TN and TP were 56.5 and 13.9 g/m^2/year, respectively. This assimilation wetland has higher loading rates than the 15 g/m^2/year for TN and 4 g/m^2/year for TP maximum limits set by the LPDES permit. It is important to note that even though loading rates are much higher than limits set by the LPDES permit, at the Discharge site mean TN concentration of surface water was 7.58 ± 0.95 mg/L and TP concentration was 1.97 ± 0.20 mg/L between 2006 and 2013. By the time surface water reached the Bayou Chinchuba Mid site, concentrations of total nitrogen and total phosphorus were reduced by at least 50% (2.31 ± 0.38 mg TN/L and 0.98 ± 0.14 mg TP/L) from those measured at the Discharge site and concentrations in the Bayou Chinchuba Out site were similar to the Marsh Reference site (Fig. 2.25). Thus, based on these

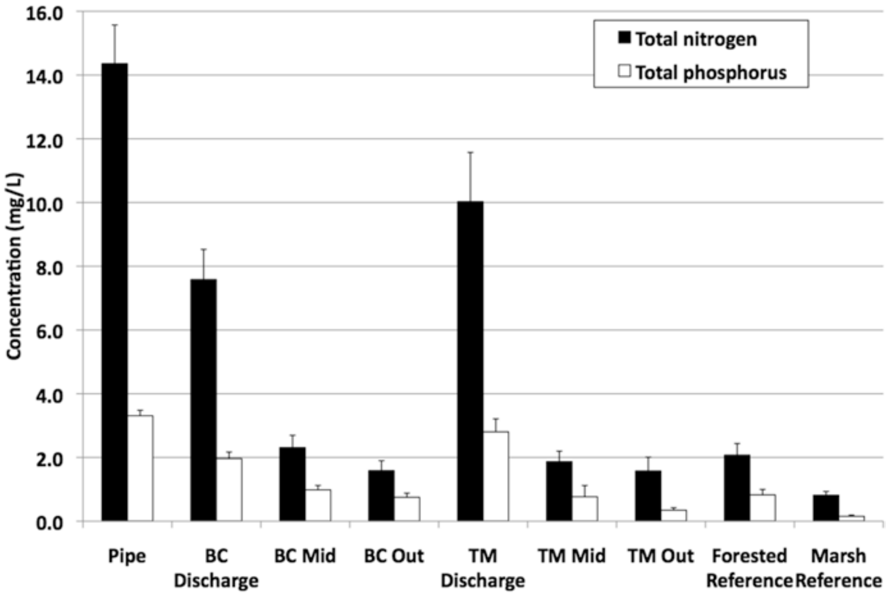

Fig. 2.25 Mean total nitrogen and total phosphorus concentrations at the effluent pipe and in surface water at the Mandeville Bayou Chinchuba (BC) and Tchefuncte Marsh (TM) assimilation wetlands. Data were collected between 2006 and 2013 for Mandeville Bayou Chinchuba and between 2009 and 2013 for Mandeville Tchefuncte Marsh. Error bars are standard error of the mean

data, it appears that the wetland has the capacity to assimilate and reduce nutrients at loading rates much higher than those allowed by the LPDES permit.

Average monthly discharge from 2009 to 2013 into the Tchefuncte Marsh assimilation wetland was 1.44 MGD and average concentrations of TN and TP were 15.52 and 3.02 mg/L, respectively, with mean TN and TP loading rates during this time of 7.48 and 1.46 g/m^2/year, respectively (Table 2.8). Even though the Tchefuncte Marsh and Bayou Chinchuba assimilation wetlands receive effluent from the same source (Mandeville wastewater treatment facility) mean nutrient concentrations are not always the same because they were calculated for different time periods and concentrations of the effluent vary over time.

By the time surface water reached the Mid sites at the Bayou Chinchuba and Tchefuncte Marsh assimilation wetlands, mean total nitrogen concentrations were almost as low as the Reference sites (Fig. 2.25). The Mandeville effluent is 43.2% NH_3, 40.2% NO_x, and 16.6% organic nitrogen. Like the Hammond assimilation wetland, effluent entering these wetlands is high in NO_x and, as stated previously, in anoxic soils NO_x is rapidly removed through denitrification. The high NO_x removal also was seen in the decrease between the point of effluent discharge and the Bayou Chinchuba Discharge and Tchefuncte Marsh Discharge sites. Mean TP concentrations at the Mid, Out, and Reference sites were very similar and generally less than about 1.0 mg/L.

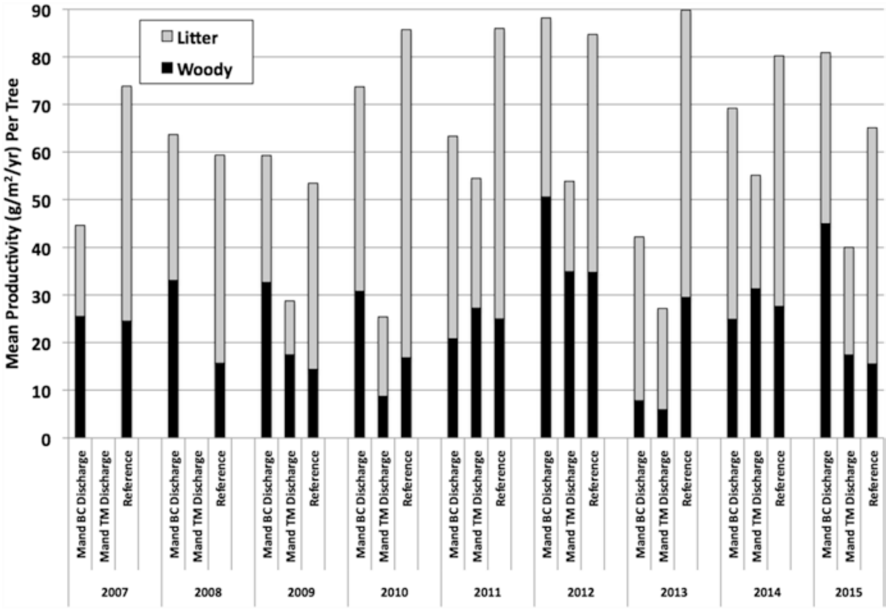

Fig. 2.26 Mean annual litterfall and woody productivity at the Mandeville Bayou Chinchuba (BC) and Tchefuncte Marsh (TM) assimilation wetland

In the Mandeville Bayou Chinchuba assimilation wetland, mean annual NPP was often higher at the Discharge sites than at the Reference site (Fig. 2.26), but results varied from year to year. Monitoring of vegetation biomass did not begin until 2009 at the Mandeville Tchefuncte Marsh wetland. EOSL biomass at the marsh sites (Bayou Chinchuba Out, Tchefuncte Marsh Out, marsh Reference) followed similar trends, but was higher at the Reference site than at the Bayou Chinchuba Out site (Fig. 2.27).

Hammond

During the fall of 2006, the City of Hammond began discharging secondarily-treated effluent into Four Mile Marsh located in the northwest corner of the Joyce wetlands, approximately 11 km southeast of Hammond, Louisiana (Fig. 2.28). The city treatment system has the capacity to treat 8 MGD. Dry weather flow averages about 2.7 MGD but inflow and infiltration can raise discharge as high as 17 MGD (Lane et al. 2015). Influent wastewater is passed through the Hammond wastewater treatment plant headworks and then piped to a three-cell oxidation lagoon. After secondary treatment, effluent is disinfected with chlorine and then de-chlorinated. From 2007 to 2013, average monthly discharge into the assimilation wetland was 3.9 MGD, with average concentrations of TN and TP of 17.91 and 3.64 mg/L,

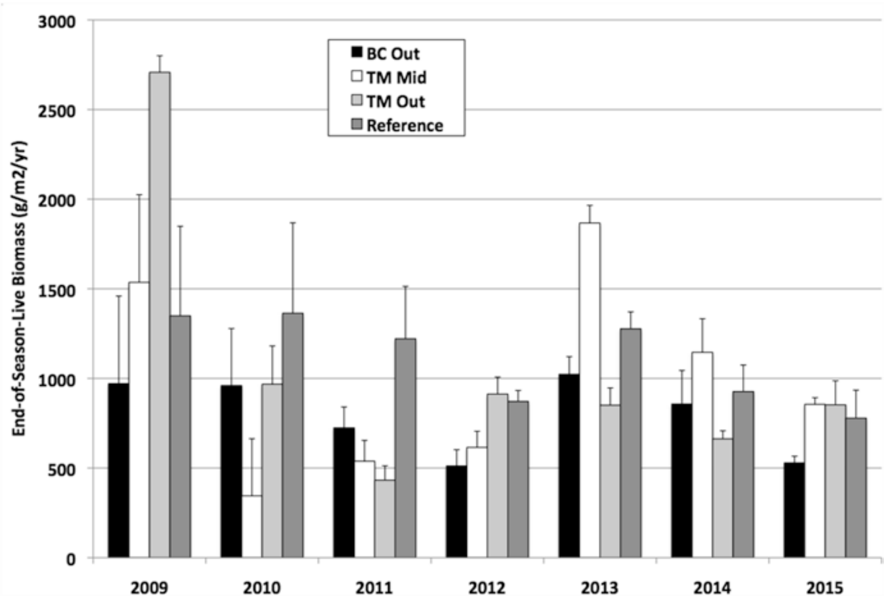

Fig. 2.27 Mean annual end-of-season-live biomass at the Mandeville Bayou Chinchuba (BC) and Tchefuncte Marsh (TM) assimilation wetland sites. Error bars represent standard error of the mean

respectively, and mean TN and TP loading rates of 2.39 and 0.48 g/m^2/year, respectively (Table 2.8).

The Hammond assimilation wetland has generated controversy because of marsh deterioration during the winter of 2007–2008. After initiation of discharge in Fall 2006, there was robust marsh growth during the 2007 growing season. During the 2007–2008 winter, about 150 ha of marsh deteriorated. Since that time, there has been partial recovery of the area. Field observations and exclosure experiments indicated that the deterioration was due to herbivory by nutria (Shaffer et al. 2015). Others proposed that the deterioration was due to excessive inundation and nutrient induced decomposition. We address these issues below.

The east Joyce Wetlands (EJW) bordering northwest Lake Pontchartrain have a long history of human induced changes (Lane et al. 2015), such as leveeing of the Mississippi River that eliminated almost all riverine input to the area and segmentation of the east and west Joyce wetlands by the construction of a railroad, U.S. highway 51, and Interstate 55. Dredged drainage canals and associated spoil banks have channeled watershed input around the wetlands, especially South Slough that channelizes most upland runoff north of the assimilation wetlands directly to the I-55 canal. The deep canal associated with I-55 causes both rapid short-circuiting of freshwater runoff to Lakes Maurepas and Pontchartrain and saltwater intrusion deep into fresh and formerly freshwater wetland areas. Increasing salinity has caused widespread loss of freshwater forested wetlands in the area (Shaffer et al. 2009, 2015).

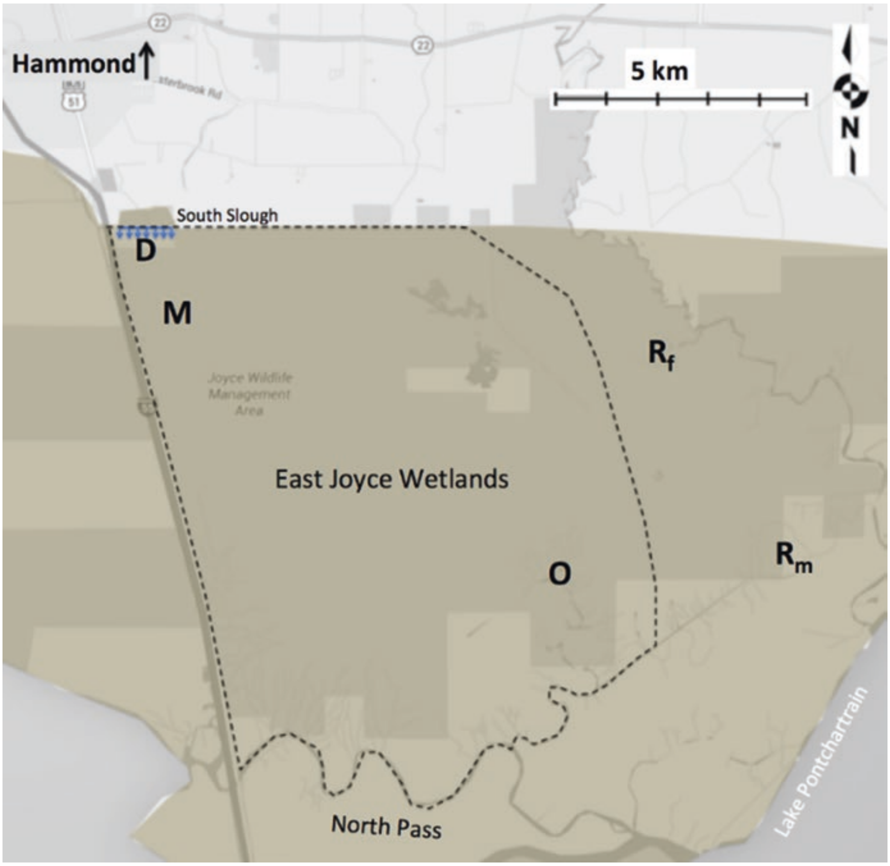

Fig. 2.28 Location of wetland monitoring sites at the Hammond assimilation wetland. R_m Marsh Reference site, R_f Forested Reference site, *D* Discharge site, *M* Mid site, *O* Out site. The *blue lines* show location of the discharge pipe and outlets, and the *black dashed line* the assimilation wetlands

Field measurements and a hydrological model showed that short-circuiting from the wetlands south of South Slough to the I-55 canal was minimal and most flow through the wetlands was to the southeast (Lane et al. 2015). Water levels in the Hammond assimilation wetland were highly variable prior to the beginning of effluent discharge in 2006, with relatively high water levels that did not increase substantially from 2007 through summer 2009 despite the addition of municipal effluent. Post-effluent water levels lacked the variability of the pre-discharge period and were about 20 cm higher from late 2009 until 2014 due to high rainfall in 2009, 2012 and 2013 and high effluent inflow due to significant inflow and infiltration (I&I) into the city collection system. Historical net watershed inputs averaged 2.69 cm/year over the 4 km^2 area immediately south of the effluent distribution system, compared to 0.38 cm/year for the effluent and 0.13 cm/year for direct precipitation. Salinity increased from north to

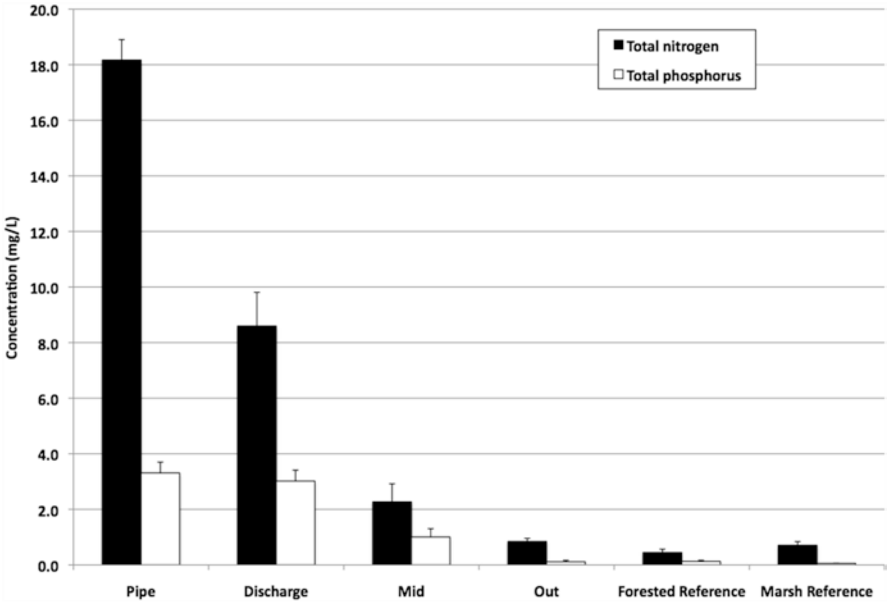

Fig. 2.29 Mean total nitrogen (TN) and total phosphorus (TP) concentrations at the effluent pipe and in surface water at the Hammond assimilation wetland between 2007 and 2013. Error bars represent standard error of the mean

south with strong seasonality, averaging 1.9–2.1 PSU near the lake to 0.4–0.6 PSU in the northwestern EJW. Peak salinities were 4.6–5.1 PSU near the lake and 1.8 PSU in northwestern EJW. There was a significant decrease in salinity beginning in 2010 coinciding with the closure of the Mississippi River Gulf Outlet, high precipitation in the fall and winter of 2009, and in 2012 and 2013, and continuing operation of the assimilation system with high I&I in those years.

The Discharge, Out, and Marsh Reference sites are emergent freshwater wetlands dominated by cutgrass (*Zizaniopsis miliacea*), bulltongue (*Sagittaria lancifolia*), soft rush (*Juncus effusus*) and cattail. The Mid and Forested Reference sites are freshwater forested wetlands dominated by pond cypress (*Taxodium distichum var. nutans*), and further south vegetation transitions to wiregrass (*Spartina* patens) and relict baldcypress forest. The Hammond assimilation wetland is the only assimilation wetland in coastal Louisiana with an herbaceous emergent marsh at the Discharge site; all of the other assimilation wetlands have forested Discharge sites.

Mean TN and TP concentrations in surface water at the Hammond assimilation wetland declined steadily from the Discharge site to the Out site (Fig. 2.29). In particular, mean TN concentration decreased by more than 60% between the Pipe and Discharge site and between the Discharge and Mid sites. Unlike the effluent at the Breaux Bridge, Broussard, and Luling assimilation wetlands, effluent discharged into the Hammond assimilation wetland is highly nitrified. In the Hammond effluent, TN is 52.6% NO_x, 32.5% NH_3, and 14.9% organic nitrogen. The high percentage of NO_x in the effluent leads to rapid nitrogen removal through denitrification, and is the primary reason why TN decreases rapidly as surface water moves through the wetland; NO_x decreased by almost an order of magnitude within 100 m and to <0.1 mg/L within 700 m.

The forested Reference site is highly degraded and had a much lower stem density than the Mid site until the Reference site was re-located in 2012 (Fig. 2.30). When perennial productivity was normalized for stem density, the Mid site was more productive than the Reference site but litterfall was higher in the Reference site. At the Reference site, as the number of trees declined, the amount of leaf litter produced by each tree increased greatly, most likely due to increased light availability.

At the Hammond assimilation wetland, Shaffer et al. (2015) found a linear decrease occurred in the concentrations of NH_3 and PO_4 from the outfall pipe along a 700-m transect. Inorganic nutrients were essentially non-detectable 600 m from the outfall pipe. In addition, baldcypress seedlings planted where effluent is discharged at the Hammond assimilation wetland had aboveground production that followed a remarkably similar pattern as that of inorganic nutrients, with seedling growth greatest at the outfall pipe and decreasing linearly to 700 m from the discharge pipe. The Mid site is almost 1000 m away from the discharge pipe and nutrient concentrations were low and no differences in growth based on nutrients from the effluent should be expected.

Mean EOSL biomass was higher at the Discharge site than at the Reference site in almost every year monitored (Fig. 2.31).

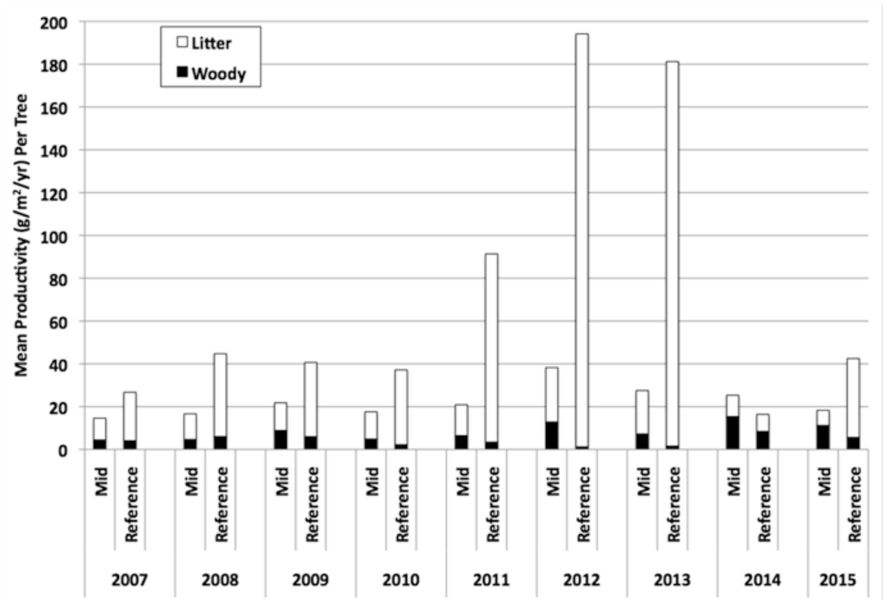

Fig. 2.30 Mean annual litterfall and woody productivity at the Hammond assimilation wetland

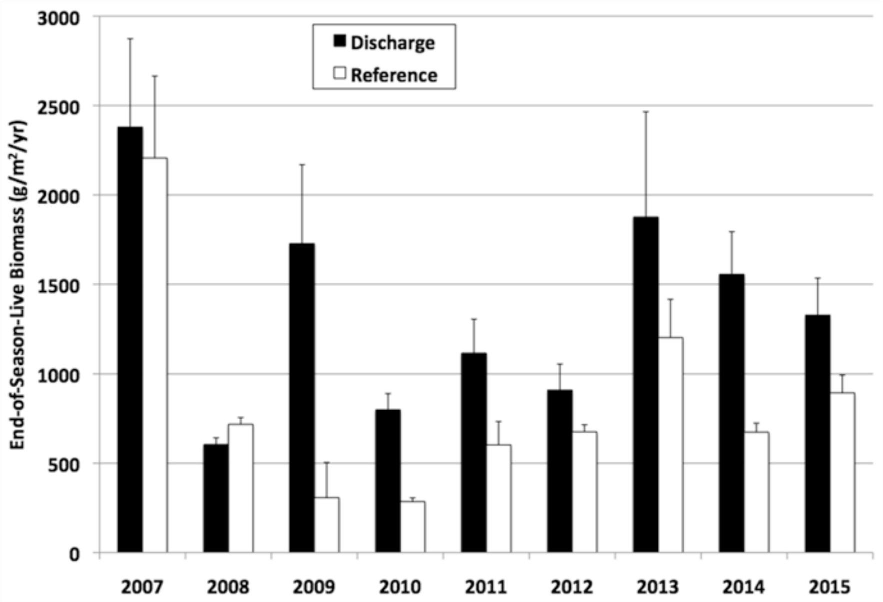

Fig. 2.31 Mean annual end-of-season-live biomass at the Hammond assimilation wetland. Error bars represent standard error of the mean

Adaptive Management at the Hammond Assimilation Wetland

At the Hammond assimilation wetland, immediately following effluent discharge in 2006, there was robust growth of herbaceous vegetation (Shaffer et al. 2015). By late fall 2007, vegetation biomass at the emergent wetland in the immediate vicinity of the effluent discharge began to decline and within months about 150 ha of emergent wetland had converted to open water, degraded marsh, or mudflat. By 2009, however, there had been substantial recovery of the wetland (Shaffer et al. 2015). During spring of 2008, to experimentally determine if the conversion from wetland to open water was caused by nutria herbivory, ten 16-m^2 exclosures were constructed using 2-m wide vinyl-coated crab wire dug approximately 0.4 m into the ground (Shaffer et al. 2015). Ten 16-m^2 paired controls were not enclosed. All of the exclosures and controls were planted with nine individuals of southern cattail (*Typha domingensis*). Cattail displayed nearly 100% cover inside of all ten exclosures within a 3-month period. In contrast, vegetation in all ten controls was completely destroyed within a few days of planting. The control plots were replanted four times, and each time suffered 100% mortality due to nutria herbivory (Shaffer et al. 2015).

The initial enhancement of herbaceous biomass followed by decline and partial recovery has engendered intense controversy and serves as an example of adaptive management to such an event (Shaffer et al. 2015; Lane et al. 2015;

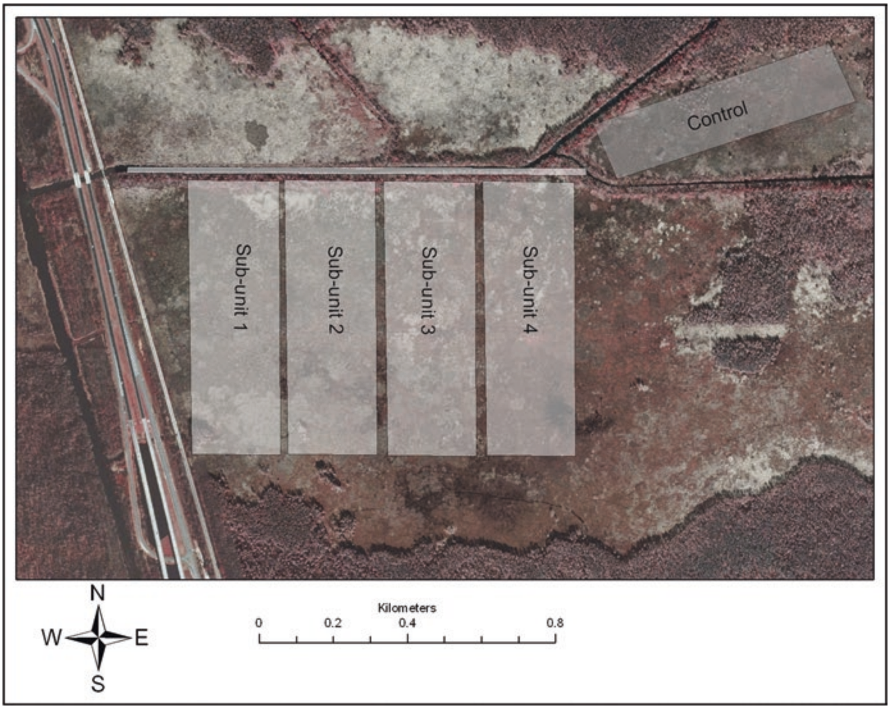

Fig. 2.32 Map of the Hammond assimilation wetland. The *white line* indicates the discharge pipe. Interstate-55 is shown west of the assimilation wetland and Joyce Wildlife Management Area, a baldcypress–water tupelo swamp, is shown south of the marsh

Bodker et al. 2015). The events at Hammond became caught up in a broader controversy of the role of nutrients in coastal wetland health versus the impact of intense herbivory by nutria. In the remainder of this section, we review the results of the monitoring required by the LPDES permit, discuss additional studies carried out to assess causes of wetland deterioration, review actions by the City of Hammond to improve the treatment system, and consider proposals for adaptive management.

To study the impact of effluent addition and marsh deterioration and recovery, a number of studies were conducted at the Hammond assimilation wetland. Four 700-m long subunits were established in the assimilation wetland and a Reference subunit was added at a nearby area isolated from effluent addition (Fig. 2.32). Beginning in winter 2007, and continuing through spring 2008, approximately 6000 baldcypress seedlings grown under different conditions (bare root and potted) were planted in Subunits 1–4 and the Reference subunit (Lundberg et al. 2011). The seedlings were planted from 0 to 700 m from the outfall pipe. Basal diameter growth of the seedlings was monitored over one growing season. Mean basal diameter growth for seedlings in subunits 1–4 ranged from 9.5 ± 0.9 mm to

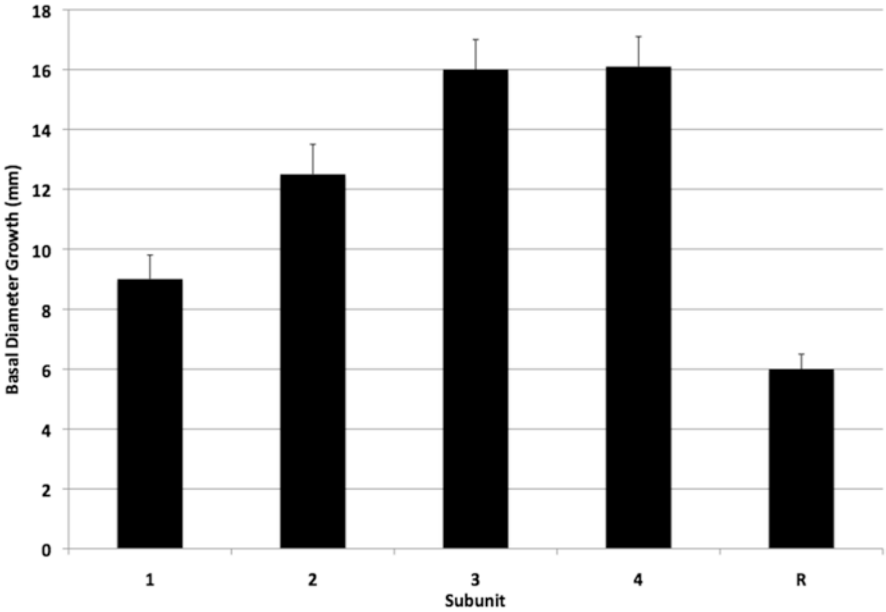

Fig. 2.33 Mean (± SE) relative basal diameter growth of baldcypress seedlings planted within four experimental subunits (1–4) and a reference (R) subunit at the Hammond assimilation wetland

16.1 ± 1.4 mm, with the highest production in Subunits 3 and 4, where most of the effluent was discharged. Mean basal diameter growth for seedlings in the Reference subunit was 6.4 ± 0.9 mm. As mentioned, the planted seedlings were grown under four types of conditions, namely 1-year old 'bare root' seedlings (BR1), 2-year old bare root seedlings (BR2), 5-month old 'Root Production Method' seedlings (RPM5), and 10-month old RPM seedlings grown in 3-gal pots. The RPM seedlings had significantly higher diameter growth than the BR1 seedlings, which had higher growth than the BR2 seedlings (Figs. 2.33 and 2.34; Lundberg et al. 2011).

The Hammond assimilation wetland have been highly effective at improving water quality while providing enormous benefits to wetland health. Once the secondarily-treated effluent reaches 150 m away from the outfall system there is a significant linear decrease in NH_4 and PO_4, declining to undetectable concentrations at 700 m from the outfall system (Fig. 2.35; Lundberg 2008). Growth rates of the baldcypress seedlings serve as surrogates of nutrient assimilation, as they too follow a linear decrease in growth from approximately 150–700 m (Fig. 2.36).

Using the average diameter growth across all distances at the Hammond assimilation wetland and comparing that value with several studies conducted in Manchac/Maurepas ecosystem, indicates a two- to sixfold higher growth rate at the Hammond assimilation wetland (Table 2.9).

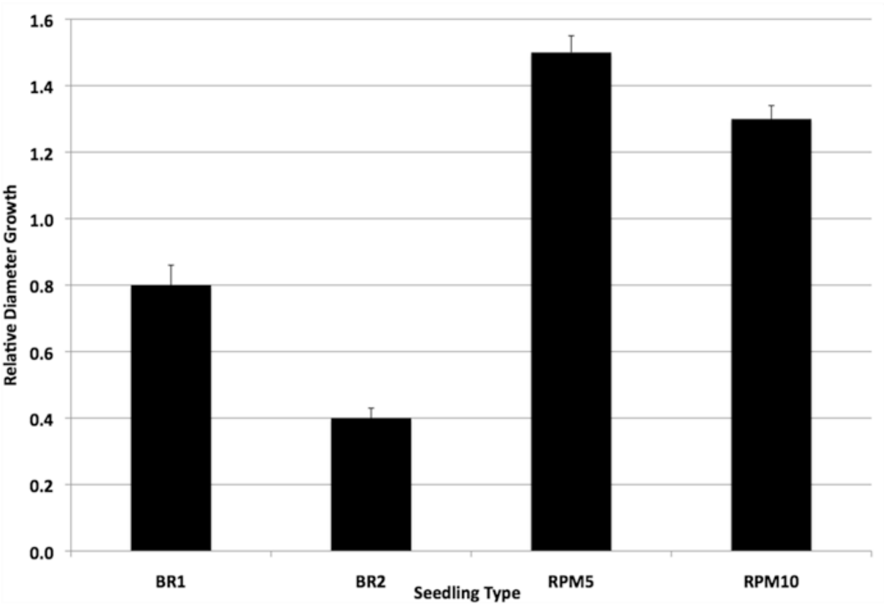

Fig. 2.34 Mean (± SE) relative basal diameter growth of various baldcypress seedling types (*BR1* bare root 1-year olds, *BR2* bare root 2-year olds, *RPM5* root production 5-month olds, *RPM10* root production method 10-month olds)

Summary of the Causes of Wetland Deterioration at the Hammond Assimilation Wetland

In response to the initial wetland deterioration observed at the Hammond assimilation wetland, several studies were conducted to determine the cause of the wetland decline. Shaffer et al. (2015) reviewed a number of hypotheses that were proposed to explain the changes in the assimilation wetland after effluent application began, including herbivory by nutria, excessive nutrients, reductions in above- and belowground biomass, increased soil decomposition due to high nutrient concentrations, prolonged inundation, toxicity, increased pH, and disease. Based on intensive field and mesocosm studies, they concluded that the initial marsh loss was primarily caused by nutria herbivory, and secondarily by waterfowl herbivory, and that significant recovery of the herbaceous vegetation occurred as a result of aggressive nutria control (>2000 eliminated). Marsh destruction due to nutria grazing has been observed frequently in Louisiana (Shaffer et al. 1992; McFalls et al. 2010; Holm et al. 2011) and nutria preferentially graze nutrient-enriched wetland vegetation (Ialeggio and Nyman 2014). After culling of the nutria population, vegetation recovery has been most pronounced near the point of effluent discharge.

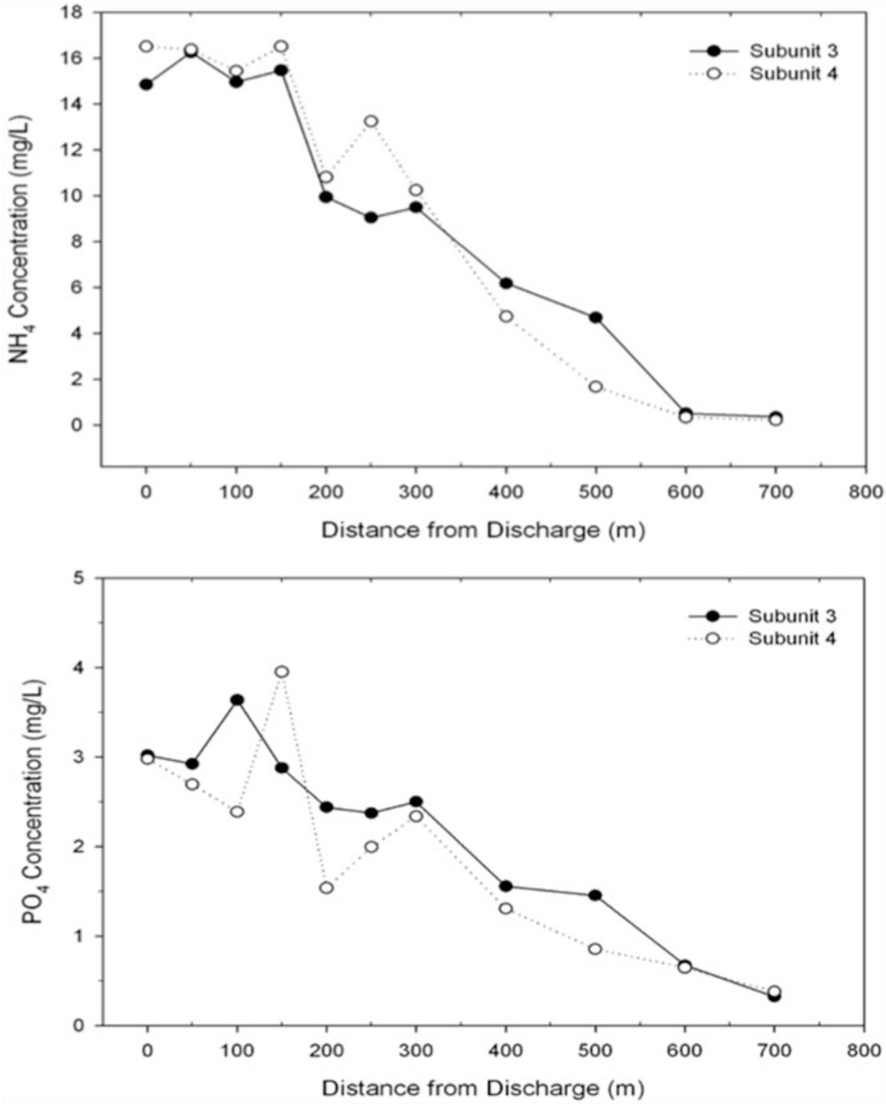

Fig. 2.35 NH$_4$ and PO$_4$ concentrations at various distances from effluent discharge at the Hammond assimilation wetland (Lundberg 2008)

Bodker et al. (2015) measured organic matter loss, gas production, and soil strength in experiments with and without added treated municipal effluent and concluded that added nutrients led to a significant increase in decomposition and a decrease in soil strength that was responsible for the wetland deterioration. To test this hypothesis, we determined the annual loading rate for the Hammond assimila-

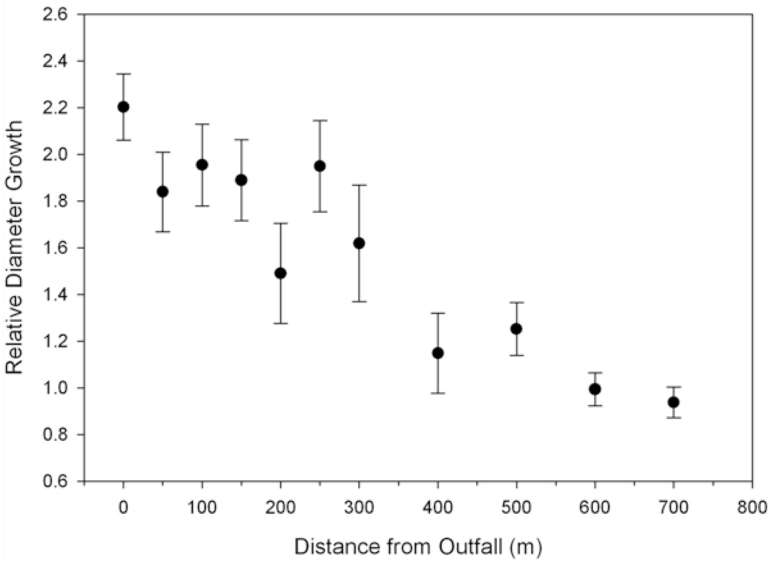

Fig. 2.36 Mean (± 1 s.e.) relative basal diameter growth of RPM baldcypress seedlings at various distances from effluent discharge at the experimental subunits (Lundberg 2008; Shaffer et al. 2015)

Table 2.9 Results of various studies of baldcypress seedling growth within the Maurepas drainage basin

Study	Fertilizer	Diameter (mm)	Height (cm)
Lundberg (2008)	Y	13.48	23.97
	N	6.38	11.32
Beville (2002)	Y	8	–
	N	5	–
Boshart (1997)	Y	4.25	–
	N	4.5	–
Campo (1996)	Y	4.25	–
	N	2.2	–
Forder (1995)	Y	2	–
	N	4	–
Myers et al. (1995)	Y	7.5	9.7
	N	4.1	4.2

Time-release Osmocote 18-6-12 commercial fertilizer was used for all fertilizer treatment studies excluding Lundberg (2008), which used secondarily-treated effluent

tion wetland and calculated how much organic matter could be decomposed if all NO_3 in the effluent stream were denitrified. The equation for denitrification is:

$$C_6H_{12}O_6 + 4NO_3 = 6CO_2 + 2N_2 \quad \text{(Mitsch and Gosselink 2015)}$$

Based on this equation, 1 g of NO_3-N reduced in denitrification results in the oxidation of 1.28 g of carbon, or 2.57 g organic matter assuming a 50% carbon content. If we assume an average NO_3 concentration of 8 mg/L (Shaffer et al. 2015), and discharge of 4.1 MGD (Lane et al. 2015), the total annual nitrogen load would be 44,300 kg-N/year. Using a receiving wetland area of 121 ha, the zone of immediate impact, approximately 94.2 g/m^2 of soil organic matter could be decomposed by denitrification over an annual period if all NO_3 were reduced to N_2 via denitrification and the organic matter substrate was from soil organic matter. However, this amount is more than compensated for by new vegetative production (Shaffer et al. 2015), generally only 60–70% of available NO_3 is denitrified (Mitsch and Gosselink 2015), and most importantly, most soil organic matter is not a suitable substrate for denitrification while effluent is an excellent source of low molecular weight organic compounds for denitrifying bacteria. We thus reject the hypothesis that nutrients were responsible for the wetland deterioration at the Hammond assimilation wetland. Decomposition rates estimated by Bodker et al. (2015) were equivalent to or less than the first month of litterbag decomposition measured over 15 months that found no differences in decomposition with distance or depth (Shaffer et al. 2015). The decomposition rates based on gas production and stoichiometric calculations were less than 5% and 3.5%, respectively, of the soil organic matter substrate used in the experiments for summer temperatures. The Bodker et al. (2015) gas evolution experiments were carried out at temperatures between 22–35 °C. Decomposition would have been significantly less during winter, and thus could not have been responsible for the marsh deterioration that occurred over a 6-month period in winter 2007–2008. There is an extensive scientific literature on wetlands receiving treated effluent that shows few negative impacts on productivity, accretion, or decomposition (Kadlec and Wallace 2009). We know of no study that reports such rapid deterioration of a wetland over a short period of time due to nutrient enrichment. On the other hand, there are numerous studies showing that nutria herbivory has led to rapid destruction of hundreds of hectares of coastal wetlands in less than a year (e.g., Shaffer et al. 1992; McFalls et al. 2010; Holm et al. 2011).

Over the past several years, the BOD limit of 30 mg/L has been exceeded a number of times at the Hammond wastewater treatment facility, with reported concentrations up to 50 mg/L. The City of Hammond has implemented a number of improvements to better treat wastewater to meet LPDES limits, including:

- Installation of a new aeration system to reduce BOD;
- Upgrading an oxidation ditch system to pre-treat high BOD inputs from a milk processing facility and a medical center and installation of pretreatment systems at these two facilities;
- Installation of a nitrification-denitrification system to reduce NH_3 concentrations;
- An evaluation of toxicity reduction in the effluent; and
- Reduction of inflow and infiltration to the collection system up to 50%.

The City is also exploring other options for improving the overall treatment system. One is the construction of a pipeline that would allow the discharge of treated effluent west of I-55 into a freshwater forested wetland. Pulsing the water east and west would allow the current assimilation wetland to have periods of no flow that would allow better drainage of the wetlands. It also would serve as a buffer to saltwater intrusion in wetlands west of I-55 where there has been widespread loss of freshwater forested wetlands.

Current Issues Concerning Assimilation Wetlands in Coastal Louisiana

Pulsing

In our opinion, all assimilation wetlands should have two independent outfall areas to maximize the pulsing paradigm and allow water levels to subside. Pulsing is currently included at most of the assimilation projects in Louisiana and is accomplished by having at least two discharge outlets so that effluent can be introduced to different parts of the wetland. At the St. Martinville project, the effluent can be discharged into different cells. One of the goals of pulsing is to allow the system to "draw down". The idea is that under natural conditions, most freshwater wetlands in coastal Louisiana generally have no standing water during part of the year. Pulsing has been successful at the three systems that are not impacted by coastal water levels (Breaux Bridge, Broussard, and St. Martinville) but the forested wetland sites that are impacted by coastal water levels (Thibodaux, Luling, Mandeville, Amelia, and Hammond) are permanently flooded to the extent that regeneration cannot occur. Thus, even if discharge to these sites was stopped, there would still be permanent or near permanent flooding. The herbaceous wetlands at Mandeville and Riverbend are tidal and experience regular flooding and draining. The Hammond herbaceous wetland is located about 10 km north of Lake Maurepas and is largely isolated from direct hydrologic exchange with either Lake Maurepas or Lake Pontchartrain. Because of this, there is no daily tidal signal in the assimilation wetland. Water levels are controlled by seasonal water level variability due to changes in lake levels, watershed input, local precipitation, and effluent discharge. The effluent discharge has dampened water level variability in the Hammond assimilation wetland (Lane et al. 2015). As mentioned above, it has been suggested that pulsing to forested wetlands west of I-55 would allow a return to a more normal hydrology as well as buffer saltwater intrusion on both sides of I-55.

Fresh Marshes

The experience at Hammond has raised questions about the use of fresh herbaceous marshes as assimilation wetlands. Although we believe that the marsh deterioration at Hammond was due mainly to nutria herbivory, it has been suggested that adding nutrients to wetlands leads to lower belowground productivity, higher decomposition rates, and loss of soil strength. This is based on considerable discussion over the past several years about the effects of nutrient loading to coastal wetlands (Darby and Turner 2008a, b, c; Day et al. 2013; Davis et al. 2017; Deegan et al. 2012; Graham and Mendelssohn 2014; Morris et al. 2013; Nyman 2014; Swarzenski et al. 2008; Turner 2011; van Zomeren et al. 2012; Snedden et al. 2015). Based on these findings, this is an issue that must be carefully analyzed when considering discharge of treated effluent to fresh marshes. In addition, nutria control needs to be incorporated into management plans.

Global Change

Coastal wetlands in Louisiana will be impacted by increasingly severe climate impacts. These impacts can affect wetland assimilation systems as well as be mitigated by them (Day et al. 2005, 2007, 2016a). These include sea-level rise that is expected to increase by 1–2 m by 2100 (FitzGerald et al. 2008; Pfeffer et al. 2008; Vermeer and Rahmstorf 2009; IPCC 2013; Koop and van Leeuwen 2016; Deconto and Pollard 2016), more category 4 and 5 hurricanes (Emanuel 2005; Webster et al. 2005; Hoyos et al. 2006; Goldenberg et al. 2001; Kaufmann et al. 2011; Mei et al. 2014), drought (IPCC 2007; Shaffer et al. 2015), more erratic weather (Min et al. 2011; Pall et al. 2011; Royal Society 2014) and other factors. The combination of accelerated sea-level rise, more frequent intense hurricanes, and drought will lead to more inundation and salinity stress. For example, the 2000–2001 drought raised salinities above 12 psu in western Lake Pontchartrain (Day et al. 2012) and to 9 psu in the Manchac basin (Shaffer et al. 2009). These conditions will especially impact low salinity to fresh wetlands. By providing fresh water and stimulating vertical accretion, discharge of treated effluent can make low salinity and fresh wetlands more sustainable in the face of these climate extremes.

Decreasing energy availability and higher energy prices will make energy intensive wetland restoration and management more expensive, limit options for restoration and complicate human response to climate change (Day et al. 2007, 2014; Tessler et al. 2015). The implication of future energy scarcity is that the cost of energy will be higher in coming decades (Bentley 2002; Day et al. 2005, 2016a; Hall and Day 2009; Murphy and Hall 2011) and the cost of energy-intensive activities also will increase significantly (Tessler et al. 2015). Advanced wastewater treatment systems are energy intensive and expensive to construct and operate. Wetland assimilation is a low energy approach that can offer a sustainable treatment approach in a time of growing climate impacts and increasing energy costs.

Conclusions

There are ten active assimilation wetlands in coastal Louisiana and another four with permit applications pending. Results of annual monitoring show that nutrient concentrations of surface waters decrease with distance, reaching background levels before water leaves the wetland. While nutrient concentrations decrease, vegetative productivity is enhanced. In degraded forested wetlands being used as assimilation wetlands, baldcypress and water tupelo seedlings are often planted, which thrive in the nutrient rich environment. However, nutria are attracted to vegetation with increased nutrient concentrations, and this introduced species must be monitored and controlled. Pulsing of effluent between two or more sites should be incorporated to prevent prolonged flooding and to encourage seedling development and growth.

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