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The use of wetlands in the Mississippi Delta for wastewater assimilation: a review

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Abstract

The use of wetlands for treatment of wastewaters has a number of important ecological and economic benefits. Adding nutrient rich treated wastewater effluent to selected coastal wetlands results in the following benefits: (1) improved effluent water quality; (2) increased accretion rates to help offset subsidence; (3) increased productivity of vegetation; and (4)

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financial and energy savings of capital not invested in conventional tertiary treatment systems. We present as case studies results from several wetlands that are receiving secondarily treated wastewater in coastal Louisiana. At one site where sedimentation accumulation was measured, rates of accretion increased significantly after wastewater application began in the treatment site (from 7.8 to 11.4 mm yr^{-1}) and approached the estimated rate of regional relative sea level rise (RSLR) (12.0 mm yr^{-1}). No corresponding increase was observed in an adjacent control site. This suggests that the application of nutrient-rich wastewater can help coastal wetlands survive sea level rise. In the same site, surface water nutrient reduction, from the effluent inflow to outflow (1600 m), ranged from 100% for nitrate-nitrogen (NO_3-N) to 66% for total phosphorus (P). At a second site, a forested wetland that has been receiving wastewater effluent for 50 years, N and P were both reduced by more than 90%. Nutrient reduction is due to three main pathways: burial, denitrification and plant uptake. Dendrochronological analysis at the second site revealed that stem growth increased significantly in the treatment site after wastewater applications began, and was significantly greater than an adjacent control site. Similar increases in productivity have been measured in a number of wetland treatment sites. Economic analyses comparing conventional and wetland systems indicate savings range from \$500,000 to \$2.6 million. In addition there are substantial energy savings.

1. Introduction

Numerous studies have shown that wetlands can be effective tertiary processors of wastewater effluent [1]. Previous studies indicate that both natural and constructed wetlands have been successfully used to purify effluent [1–4]. Wetlands are efficient at removing excess nutrients and pollutants by physical settling and filtration, chemical precipitation and adsorption, and biological metabolic processes that result in burial, storage in vegetation, and denitrification [3,5,6]. These wetland functions can be especially critical for the coastal regions in Louisiana affected by degraded water quality caused, in part, by inadequate sewage treatment [7].

Wastewater effluent may also serve as a restoration tool in coastal wetlands impacted by high rates of relative sea level rise (RSLR). Wetlands have been shown to persist in the face of RSLR when vertical accretion and elevation gain equals or exceeds the rate of water level rise [8,9]. Historically, seasonal overbank flooding of the Mississippi River deposited sediments and nutrients into the wetlands of the delta plain [10–12]. Not only did these floods provide an allocthonous source of material or mineral sediments, which contributed directly to vertical accretion, but the nutrients associated with these sediments promoted vertical accretion through organic matter production as well as deposition [13,14]. This sediment and nutrient source to most coastal forested wetlands and marshes in the Mississippi delta has been eliminated since the 1930s with the completion of levees along the entire course of the lower Mississippi River, resulting in vertical accretion deficits (accretion < RSLR), prolonged periods of inundation, lowered productivity, marsh loss, and a lack of regeneration in forested wetlands [15,16]. Primarily because of these impacts, there has been a massive loss of coastal wetlands [16].

In these stressed wetland systems in the Mississippi delta, there are 4 primary benefits derived from wetlands wastewater treatment: (1) improved water quality; (2) increased accretion rates; (3) increased productivity of vegetation; and (4) the financial and energy savings of capital not invested in conventional tertiary treatment systems (see Fig. 1 [17,18]). The high rate of burial due to subsidence and higher than national average rates of denitrification due to warm temperatures are additional reasons for the use of wetland treatment in Louisiana. Increasing vegetative productivity is especially crucial in many parts of Louisiana where coastal subsidence in the Mississippi delta results in a RSLR nearly 10 times greater than eustatic sea level rise [15,19]. Increasing productivity results in greater root production leading to organic soil formation that can enhance the accretion necessary to offset the subsidence that is contributing to wetland loss.

Since 1988, a group of scientists in Louisiana has been working with EPA, the Louisiana Department of Environmental Quality (DEQ), and several dischargers to assess the impact of forested and marsh wetland wastewater assimilation projects in coastal Louisiana (for a general policy review see [18]). The dischargers include several municipalities and industries. All of the potential and actual receiving wetlands are hydrologically altered by some combination of levees, spoil banks, highways, oil and gas access roads, or railroad lines. In addition, prior to wetland treatment, all effluent was discharged directly into open water bodies. Wetland discharge provides additional treatment by removing nutrients from the effluent before entering open water bodies.

To examine the effect of wetland treatment in the Mississippi delta on effluent water quality, sediment accretion, productivity, and economic savings, we review results of a number of wetland treatment studies. This information is available in several different formats. Municipalities and industries contemplating the use of wetland treatment are required to conduct a use attainability analysis (UAA) that is submitted to the Louisiana Department of Environmental Quality as part of the

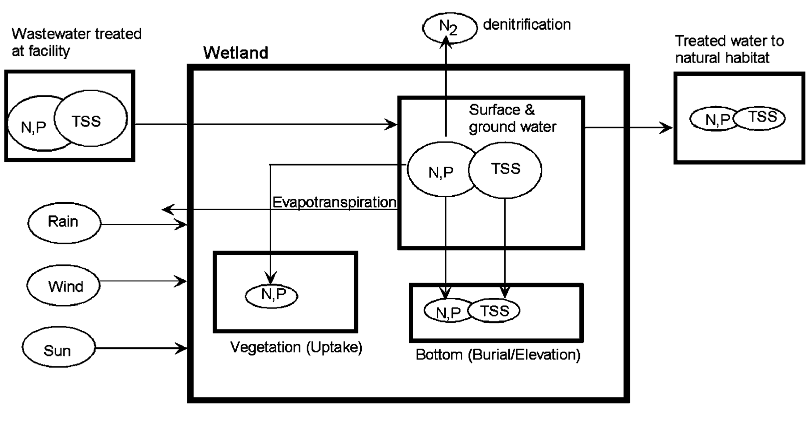

Fig. 1. A conceptual model of wastewater assimilation by wetlands showing the three main pathways of permanent nutrient uptake; vegetative uptake, denitrification, and burial (from [29]).

permit process. A UAA describes background ecological conditions of the candidate site (hydrology, soil and water chemistry, vegetation, animal populations), analyzes the feasibility of wetland treatment, and provides preliminary engineering design and cost analyses. A number of UAA studies have been carried out (e.g. [20–22]). We have also published various aspects of these studies in the scientific literature [11,18,23–33]. Additional project information can be found in a number of theses and dissertations [17,34–42]. Throughout this review, when we use the term significant, it implies a statistical significance that is documented in the references. Sampling of benthic invertebrate and nekton communities did not indicate any clear effects due to wastewater discharge and results are not presented here (e.g. [40]). This paper is adapted from a technical report published in the proceedings of a symposium held in Louisiana [43].

2. Improved effluent water quality

We hypothesized that effluent water quality will be improved through efficient nutrient uptake and removal pathways within forested wetlands. Loading rates and percent nutrient reductions for municipal wastewater treatment wetlands are listed in Table 1. Data from the Point au Chene treatment wetland for the City of Thibodaux offers an example for the impact of effluent on water quality.

The Thibodaux site consists of two almost permanently flooded, subsiding, forested wetlands, separated by a slightly elevated bottomland hardwood ridge. Since 1992, the 231 ha treatment wetland has received secondarily treated municipal wastewater at the average rate of 15,140 $m^3 d^{-1}$. The wetland on the eastern side of the ridge, which is not impacted by the effluent, serves as a control site. Baseline monitoring of vegetation, soils, surface water, hydrology, and fauna, at both sites, began in 1988. Extended inundation was documented during the baseline studies [44]. A comprehensive site description is provided by Breaux and Day, Zhang et al. and Rybczyk et al. [18,32,33].

Measurements taken at Thibodaux by Zhang et al. [32] indicate that water quality was improved as nutrients were significantly reduced and assimilated. The effluent stream was highly nitrified, with nitrate-nitrogen (NO_3-N) being the dominant form of N and soluble phosphate-phosphorous (PO_4-P) accounting for about 77% of the total P in the effluent. After passage through the treatment swamp, the concentrations of many water quality parameters at the output station were significantly reduced compared with the influent concentrations. From 1992 through 1996, the mean annual reduction (from inflow to outflow) of NO_3-N, the dominant form of nitrogen in the effluent, ranged from 96% to 99% (Fig. 2). At the output station, the NO_3-N concentration was below the detection limit (<0.1 mg l^{-1}) during most sampling periods, indicating that the swamp system removes NO_3-N completely. Fig. 3 illustrates reductions of NO_3-N concentrations as a function of distance traveled in the swamp. Within 800 m, concentrations are comparable to those found in the control site. NO_3-N was taken up by growing plants, immobilized to organic N, or removed by denitrification [24].

Table 1
Loading rates and percent nutrient reductions in at wastewater treatment forested wetlands in coastal Louisiana

Site	Treatment basin (ha)	Nitrogen loading ($g\,m^{-2}\,yr^{-1}$)	Phosphorus loading ($g\,m^{-2}\,yr^{-1}$)	Nutrient	Effluent concentration discharge	Outlet	% Reduction
Amelia[a]	1012	1.96–3.92	0.22–0.42	TKN	2.98	1	66
				Total P	0.73	0.06	92
Breaux Bridge[b]	1475	1.87	0.94	NO3-N	0.8	<0.1	100
				PO4-P	1	0.2	80
				Total P	2.9	0.3	87
St. Bernard[c]	1536	2	0.42	TKN	13.6	1.4	89.7
				Total P	3.29	0.23	95
Thibodaux[d]	231	3.1	0.6	NO3-N	8.7	<0.1	100
				TKN	2.9	0.9	69
				PO4-P	1.9	0.6	68
				Total P	2.46	0.85	66

All concentrations are reported as $mg\,l^{-1}$.
[a] [21].
[b] [20].
[c] [22].
[d] [36].

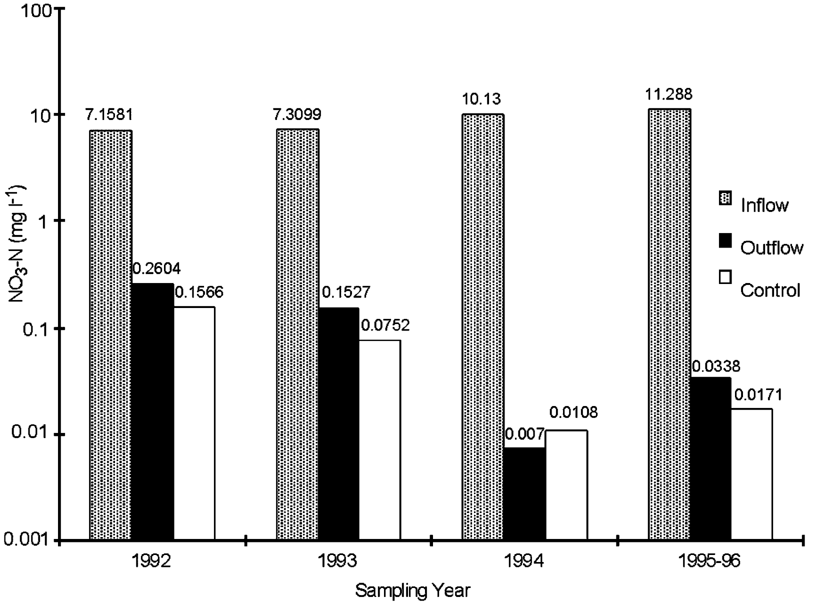

Fig. 2. Mean annual concentrations of NO_3 at the Thibodaux site for the effluent inflow pipe, the treatment outflow, and at the control site. Inflow concentrations are reduces by more than 95%. Note logarithmic scale (modified from [33]).

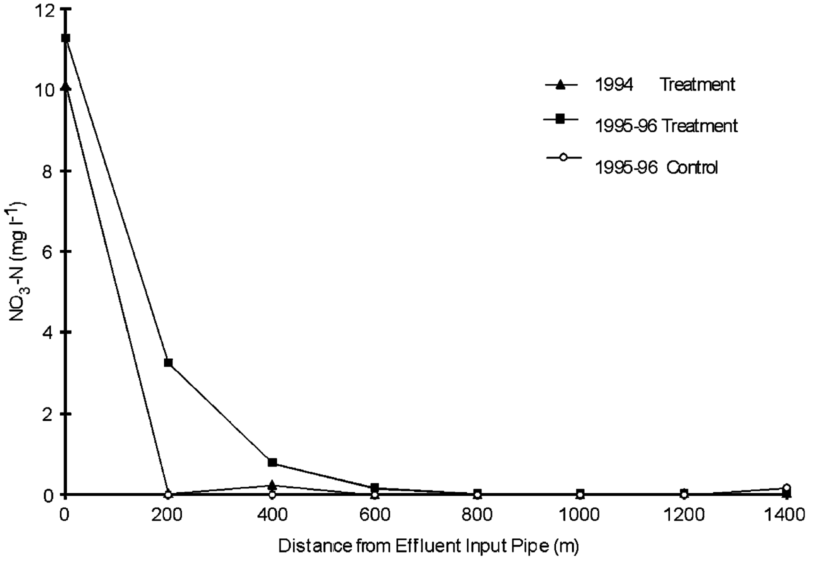

Fig. 3. Mean concentrations of NO_3 along sample transects at the Thibodaux site at treatment and control sites (modified from [33]).

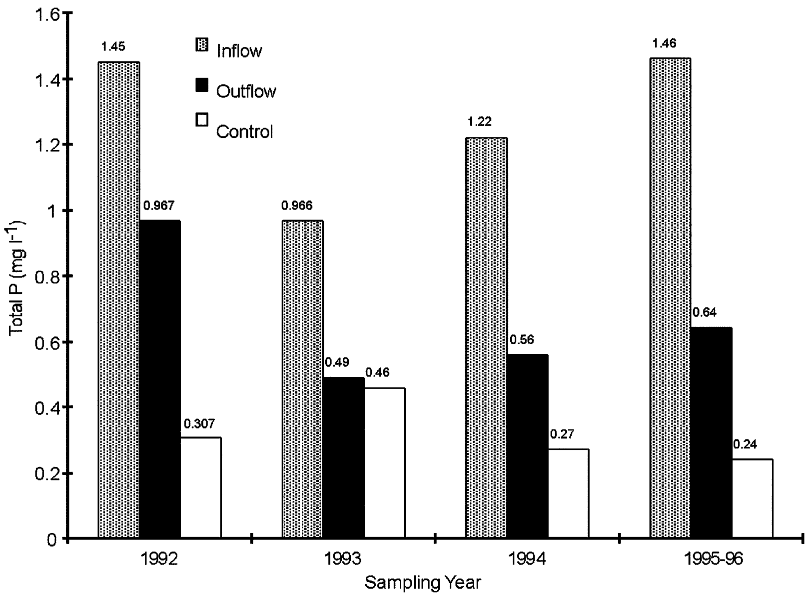

Fig. 4. Mean annual concentrations of total phosphorus at the Thibodaux site for the effluent inflow pipe, the treatment outflow, and at the control site. Inflow concentrations are reduces by an average of 66% (modified from [33]).

Zhang et al. [32] described the effects of wastewater effluent on water quality, sediment nutrient concentrations, and the chemical composition of floating aquatic vegetation at the Pointe au Chene site. This study assessed the long-term ability of the swamp to treat secondarily treated wastewater effluent from the city of Thibodaux. In general Zhang et al. found that, within the immediate 231 ha treatment zone, N and P concentrations in the water were reduced 100% and 66% (Figs. 3 and 4, Table 1), respectively, from effluent inflow to outflow. In a related review, Rybczyk et al. [30] concluded that the effective tertiary processing of effluent at this site could be attributed to the following: (1) The dominant species of N in the effluent is the oxidized NO_3 form and not the reduced species, NH_4. These naturally dystrophic wetlands readily denitrify NO_3, resulting in a net loss of N to the system as N_2 or N_2O gas (see [24,27]); (2) Loading rates are low compared to other wetlands treatment sites. For example, the State of Florida has adopted regulations for wetland wastewater management that established maximum P loading rates of $9\,g\,m^{-2}\,yr^{-1}$ for hydrologically altered wetlands [45], an order of magnitude higher than at most of our sites, and; (3) High rates of accretion and burial of sediments in these subsiding systems provide a permanent sink for phosphorus.

Similar water quality improvements have been documented for the treatment wetlands at Amelia, Breaux Bridge, and St. Bernard (Table 1). These high reduction rates of N and P further indicate that the wetlands there act as a net nutrient sink and that the sites are effective providers of tertiary treatment. For comparison, in

Florida, the tertiary advanced waste treatment standards for total N and total P are 3 and $1\,mg\,l^{-1}$, respectively. For many of these sites, nutrient concentrations have been well below that limit, indicating that tertiary treatment was achieved.

3. Removal pathways for N and P in coastal wetlands

Effluent N and P entering a wetland can either be buried in the sediments or assimilated and stored by plants (Fig. 1). Nitrogen has an additional atmospheric loss pathway via the microbial transformation of NO_3 to N_2 gas (denitrification). Burial in the sediments, or soil formation, integrates numerous processes that remove nutrients from wastewater, including; the settling of organic and inorganic sediments in the water column (and associated adsorbed or assimilated nutrients), microbial uptake, and the incorporation of allogenic organic matter (leaf litter and roots for example) into the sediment matrix. Plant uptake cannot be considered a long-term loss pathway unless the N and P are stored in persistent woody tissue and then ultimately harvested. Nutrients assimilated by herbaceous plants can also remain unavailable for long periods if they are associated with refractory organic matter that becomes incorporated in the soils (we consider this eventual loss pathway as a sediment burial pathway however). Plant uptake can be considered a permanent loss pathway, of course, if emergent or floating aquatic plants are harvested. Denitrification, an anaerobic process, can be a significant loss pathway in wetlands, especially if the dominant species of N in the effluent is the oxidized NO_3.

At the Pointe au Chene treatment site described previously, we have measured effluent loading rates [33], rates of sediment accretion [32], primary production [39,46], rates of denitrification [24,27], sediment nutrient concentrations [33], and the physical characteristics of the soil (i.e. bulk density) [32] since effluent additions began in 1992. This has allowed us to quantify loss pathways of N and P at the 231 ha treatment site (Table 2).

4. Increased sediment accretion

As indicated in the introduction, current evidence indicates that rising water levels are leading to wetland loss, coastal erosion, and salt water intrusion in a number of coastal areas [47,48]. If coastal wetlands do not accrete vertically at a rate equal to the rate of RSLR (RSLR = eustatic sea level rise plus subsidence) they can become stressed, due either to waterlogging or salt, and ultimately disappear [14,18,49,50]. In coastal regions, especially deltas, naturally high rates of subsidence can exceed rates of eustatic sea level rise by an order of magnitude [19,51]. For example, while the current rate of eustatic rise is between 1 and $2\,mm\,yr^{-1}$ [52], RSLR in the Mississippi delta is in excess of $10\,mm\,yr^{-1}$, thus eustatic sea level increase accounts for only 10–15% of total RSLR. However, predicted increases in the rates of eustatic sea level rise associated with global warming [53] have led to concerns over coastal wetland loss worldwide [47,48,54–56]. Furthermore accretion deficits (sediment

Table 2
Estimated fate of effluent N and P entering the Thiboduax treatment site

		Total N ($g\,m^{-2}\,yr^{-1}$)	Total P ($g\,m^{-2}\,yr^{-1}$)
A.	Storage in sediments (burial) Calculated as the mean rate of accretion in the immediate impact zone ($1.14\,cm\,yr^{-1}$) × mean conc. of total N ($4.95\,mg\,g^{-1}$) or P ($1.25\,mg\,g^{-1}$) in the upper 4 cm of soil × mean bulk density ($0.13\,g\,cm^{-3}$) of soil in the upper 4 cm.	7.3	1.8
B.	Storage in woody vegetation Calculated as mean annual increase in bole wood ($285\,g\,m^{-2}\,yr^{-1}$) × mean conc. of N (0.39%) and P (0.11%) in wood[a].	1.1	0.03
C.	Potential denitrification rates	36	—
D.	Total	44.6	1.83
E.	Loading rate Calculated as the mean hydraulic loading rate of $6.3 \times 10^6\,l^{-1}$ day × mean N and P effluent concentrations of 12.6 and $2.46\,mg\,l^{-1}$ respectively × basin area (231 ha).	12.5	2.4

All values used to calculate removal and loading rates were derived from data collected at the Thibodaux sites except for estimates of woody tissue N and P.

[a]Concentrations of N and P in woody tissue were not measured at the Thibodaux site. Concentrations used here are means from bottomland hardwood swamps as reported by Johnston [63].

accretion < RSLR) in many coastal systems are not only the result of high rates of RSLR, but are also the consequence of hydrologic alterations such as dams, dikes, and levees that restrict that natural movement of nutrients and suspended sediments into wetlands [11,16]. In systems affected by high rates of RSLR, hydrologic alterations, or both, coastal wetlands not only treat wastewater, but the effluent can serve as a wetland restoration or enhancement tool. Specifically, the discharge of secondarily treated effluent into wetlands can stimulate biomass production and enhance sediment accretion rates [32,57].

Recently, Rybczyk et al. [32] reported on the effects of adding nutrient-rich, secondarily treated wastewater to a subsiding, forested wetlands at Thibodaux. They found the addition of the effluent promoted vertical accretion through increased organic matter production and subsequent deposition and allowed accretion to keep pace with rates of RSLR that approached $1.23\,cm\,yr^{-1}$. Background sediment accretion rates at the site, as determined by ^{137}Cs activity in the sediments, averaged only $0.44 \pm 0.04\,cm\,yr^{-1}$ resulting in an accretion deficit of $0.79\,cm\,yr^{-1}$ [39] and continuously flooded conditions.

To determine whether wastewater applications stimulated accretion, a feldspar horizon marker technique [58] was utilized to estimate accretion rates in the site receiving effluent and in an adjacent control site, both before (1988–1991) and after (1992–1994) wastewater applications began in the treatment site. Pre-effluent accretion rates averaged $0.78\,cm\,yr^{-1}$ in the treatment site and $0.52\,cm\,yr^{-1}$ in the control site and were not significantly different (Fig. 5). It should be noted that

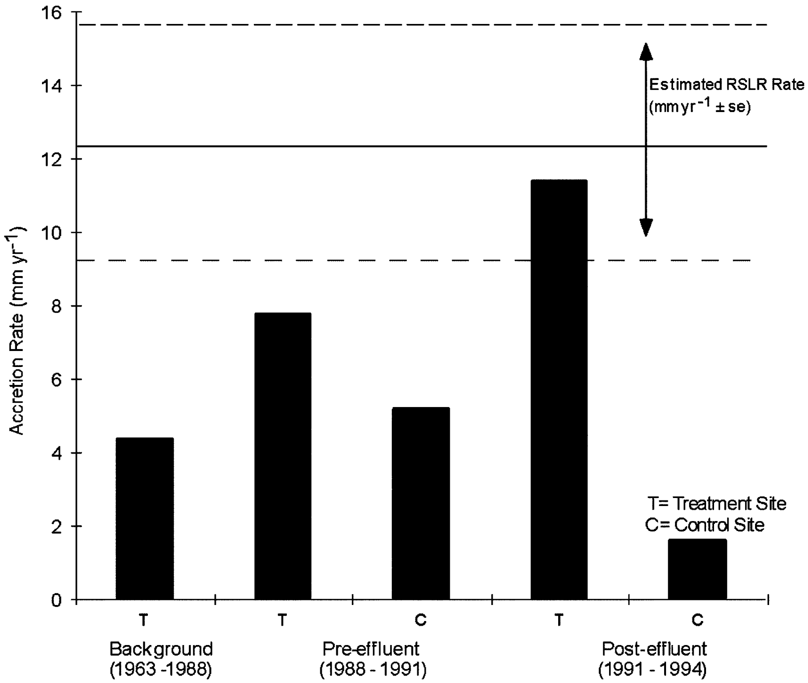

Fig. 5. Background (from ^{137}Cs measurements), pre-effluent, and post-effluent accretion rates, and RSLR for treatment and control at the Thiboduax site. T and C were not significantly different during the pre-treatment period, but T was significantly greater during the post-effluent period (from [32]).

short-term measurements of accretion rates derived from feldspar markers are typically higher than long term measurements derived from ^{137}Cs analysis, since short-term measurements do not integrate the compaction and decomposition that occurs over long time periods. Thus this is responsible for the difference between background rates of accretion, as measured by ^{137}Cs, and pre-effluent accretion rates as measured with feldspar markers.

After wastewater application began, accretion rates in the treatment site (1.1 cm yr^{-1}) were significantly higher than accretion rates measured at the control site (0.14 cm yr^{-1}) (Fig. 5). Additionally, estimated accretion rates in the treatment site fell within one standard deviation of the estimated rate of RSLR in the region [32]. An analysis of sediment accumulation rates (accretion rate x % organic or %mineral matter) indicated that only the rates of organic matter accumulation increased significantly after effluent additions began in the treatment site (Table 3). The authors attributed this to effluent-stimulated organic matter accretion. Productivity measurements collected at this site [59], showed that the production of floating aquatic vegetation (predominantly *Lemna* sp. and *Hydrocotyl* sp) increased in the treatment site after the introduction of wastewater effluent and increased in relationship to the control site.

Table 3
Average mineral and organic matter accumulation rates in the Pointe au Chene Swamp before effluent applications began (1963–1988) and from the period 1988–1993, a period that included 2 years of effluent additions in the treatment site

Time period	Accumulation rates (g dry weight m^{-2} yr^{-1} $\pm$ se)	
	Mineral matter	Organic matter
1963–1988	2302.0 ± 29.4	275.9 ± 3.3
1988–1993	2004.6 ± 67.0	736.7 ± 58.3

Only organic matter accumulation rates were significantly different between time periods ($P < 0.05$).

It could also be hypothesized that nutrient enrichment would stimulate the decomposition of organic matter, thus negating any increase in accretion due to increased organic matter accumulation. To test these hypotheses, Rybczyk et al. [32], in the same study, measured decomposition rates and litter nutrient dynamics in the wetland receiving wastewater effluent and in the adjacent control site, both before and after effluent applications began. A before-after-control-impact (BACI) statistical analysis revealed that neither leaf-litter decomposition rates nor initial leaf-litter N and P concentration were affected by wastewater effluent. A similar analysis revealed that final N and P leaf-litter concentrations did significantly increase in the treatment site relative to the control after effluent was applied. A wetland elevation/sediment dynamics model developed for this system revealed that changes in wetland elevation were much more responsive, or sensitive, to changes in primary production than to changes in rates of decomposition [31]. This would suggest that increased organic matter production and accretion would offset any increase in rates of decomposition. The model also indicated that nutrient addition alone was not sufficient to lead to long term restoration of the forested wetland and that some mineral sediment input was necessary.

5. Carbon sequestration

The data on accretion and burial discussed above indicate that addition of nutrient-rich effluent to subsiding wetland can substantially enhance the rate of carbon burial and sequestration. Sediment carbon burial before the application began was 1375 k C ha^{-1} yr (assuming C equals 50% of organic matter). After the application of wastewater effluent to the Thibodaux swamp began, accretion rates increased and calculated carbon burial rates increased to 3680 k C ha^{-1} yr, an increase of almost a factor of three. Thus, these results indicate that an additional benefit of wetland treatment is carbon sequestration. It is possible that this burial could be used as a carbon credits and partially offset treatment costs.

6. Increased productivity

Secondarily treated effluent delivers nutrient-rich water to wetlands, stimulating vegetative productivity. While this could lead to eutrophication in some aquatic systems, many wetlands are naturally dystrophic [60]. In regions of the coast that are isolated from historic pulses of nutrients and sediments by dams, dikes and levees (see [11,12]), wastewater could be used to enhance and restore productivity.

Long-term effects of wastewater effluent discharge to coastal systems can be assessed by evaluating data from a forested wetland in Breaux Bridge, Louisiana, that has been receiving wastewater effluent for over 50 years. The town of 6000 people discharges its effluent from an oxidation pond ($3785\,m^3\,d^{-1}$) to a 1475-ha cypress-tupelo wetland [18,23]. Monitoring of the effluent impact site and an adjacent reference site began in 1992. A comprehensive site description is provided by Day et al. [20].

A dendroecological ecological analysis was conducted [28,34] to determine the long term impacts of wastewater effluent on aboveground productivity (Fig. 6). Stem wood growth from 1920 to 1992 was measured at the treatment site and an adjacent control site. An annual diameter increment ratio was calculated by comparing stem wood growth from the treatment site versus the stem wood growth at the control site. Records indicate that the city began discharging into the forested wetland between 1948 and 1953. Before wastewater application began, Hesse et al. [28] found statistically significant higher growth in the control site than at the treatment site.

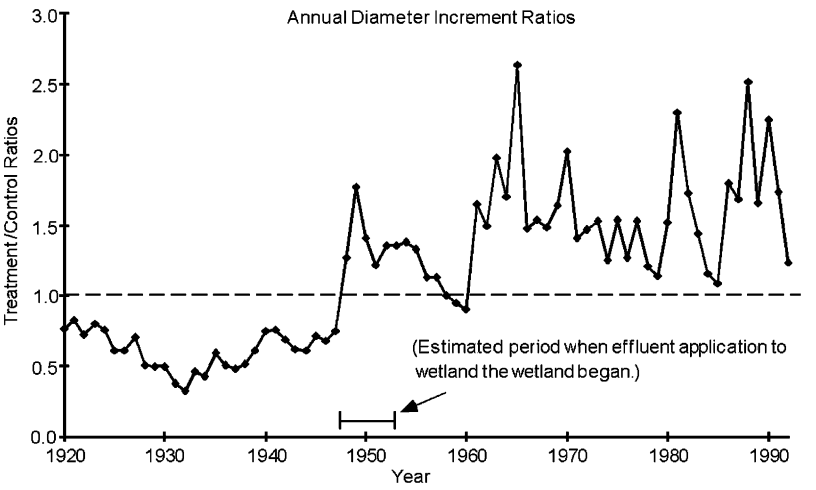

Fig. 6. The ratio (treatment/control) of annual wood stem growth (measured as growth in diameter) for bald cypress trees at Breaux Bridge. Before wastewater application to the treatment site began in the late 1940s or early 1950s, growth in the control site was significantly higher than growth in the treatment site ($P > 0.05$). After wastewater application began, stem growth was significantly higher in the treatment site. Both sites were similar in size and structure (from [28]).

However, after onset of treatment, there was increased growth in the treatment site, resulting in statistically significant higher annual diameter increment ratios [28]. A spike in the annual diameter increment ratios coincides with the onset of treatment.

Short term records at this site also confirm these findings. In January 1994, the effluent discharge was switched from the historic wetland (old treatment site), to a new site that had not previously received effluent (new treatment site). In 1992, permanent plots were established at both sites to measure annual litterfall and stem growth (Table 4). There was no statistically significant difference in the total aboveground production between the old treatment site and the new treatment site during 1993 [35]. However, during 1994 and 1995, when effluent discharge was switched to the new treatment site, total production was significantly higher in the new treatment site compared to the old treatment site [35]. Most of this difference was due to increases in stem wood biomass in the new treatment site and not leaf production.

Similar results have been reported for the other treatment wetlands. For example, a study conducted at Amelia also indicates an increase of primary productivity. The City of Amelia discharges secondarily treated effluent into a forested wetland as part of its treatment system to polish secondarily treated sewage effluent [21,42]. A year-long study on primary productivity indicates enhanced litterfall in the treatment sites (Table 5 [21,42]).

Table 4
Aboveground production ($g\,m^{-2}\,yr^{-1} \pm$ s.e.) measured at Breaux Bridge treatment site [35]

Year	Site	Stem wood production	Leaf production	Total aboveground production
1993	Old treatment[a]	780 ± 358.5	420	1200.9
1993	New treatment	677.9 ± 69.21	514	1191.9
1994	Old treatment	593.2 ± 46.8	547.3 ± 9.2	1140.5
1994	New treatment[a]	1383.4 ± 186.4	745.8 ± 8.2	2129.2
1995	Old treatment	574.8 ± 187.4	705.2 ± 81.1	1280
1995	New treatment[a]	847.7 ± 200.1	763.6 ± 45.5	1611.3

[a]Indicates site receiving wastewater effluent.

Table 5
Total mean litterfall ($g\,m^{-2}$) collect at the Amelia treatment wetland from September 1995–September 1996 [42]

Site	Mean litterfall ($g\,m^{-2}$) $\pm$ s.e.
Control 1	581.09 ± 35.68
Control 2	42.45 ± 38.24
Treatment	716.65 ± 38.08
Lake 1 Site 1	546.06 ± 47.24
Lake 1 Site 2	666.35 ± 49.52

Table 6
Mean % vegetative cover in permanent 1 m × 1 m plots measured seasonally from 1989 through 1995

Year/season	Control	Treatment
1989 Spring	51% (2)	19% (2)
1989 Summer	17% (6)	23% (2)
1989 Fall	30% (9)	37% (3)
1990 Fall	63% (7)	89% (7)
1991 Spring	3% (3)	21% (5)
1991 Summer	1% (1)	8% (3)
1992 Summer	11% (10)	99% (6)
1992 Winter	6% (6)	100% (4)
1992 Fall	11% (3)	98% (3)
1994 Summer	23% (4)	100% (3)
1994 Winter	77% (5)	98% (1)
1995 Summer	77% (3)	100% (2)

The treatment site started receiving effluent in the spring of 1992. Parenthesis indicates the number of species present.

Our studies have shown that the production of herbaceous vegetation in coastal wetlands, both emergent and floating, is also stimulated by wastewater effluent, and may contribute to sediment accretion to a greater extent than does woody vegetation [39]. At the Thibodaux forested wetland site, the percent cover of herbaceous vegetation was monitored seasonally from 1989 to 1995 in a series of permanent 1 m × 1 m plots established at the treatment site and in an adjacent control site, both before and after wastewater effluent was directed into the treatment site (Table 6). Before wastewater effluent was applied to the treatment sites in 1992, percent cover ranged from ranged from 1% to 89% in both sites and was commonly less than 25%. Highest percent cover occurred during the fall, and was probably associated with warm temperatures and the opening of the largely deciduous upper canopy. After effluent additions began in the spring of 1992 however, the treatment site was permanently and completely covered with a thick mat of floating aquatic vegetation (primarily *Lemna*, *Salvinia*, and *Hydrocotyl*).

7. Substantial economic and energy savings

Conventional wastewater treatment is often very expensive for the loads generated from many of the small communities in southern Louisiana. Wetland assimilation can provide an affordable and effective waste treatment option. A series of papers [17,18,25,29], conducted economic cost benefit analyses of the wastewater treatment operation at Breaux Bridge and Thibodaux (Table 7). They conservatively estimated a capitalized cost savings, using natural wetland wastewater treatment rather than conventional tertiary treatment. At Breaux Bridge, the estimated costs savings was approximately $2.6 million, over a 20-year period. At Thibodaux, there is a potential

Table 7
Cost comparisons for three wetlands treatment projects

Site	Conventional treatment	Wetland treatment	Cost savings
Breaux Bridge[a]	\$3,300,000	\$664,000	\$2,636,000
Thibodaux[b]	\$1,6500,000	\$1,150,000	\$500,000
Dulac[c]	\$2,200,000	\$700,000	\$1,500,000

[a]Cost reported in 2000 dollars. Capitalized costs are discounted at 7% for 20 years [29].
[b]Costs reported in 1992 dollars [18,25]. Capitalized costs are discounted at 9% for 30 years.
[c]Costs reported in 1995 dollars [26]. Capitalized costs are discounted at 8% for 25 years.

savings of approximately \$500,000. However, capitalized savings could be as high as \$1,300,000 over a 30-year period, depending upon the disinfection system employed prior to wetland discharge.

Non-toxic, industrial processors, such as shrimp processors, can benefit from using wetlands for their highly seasonal loads. A study was recently conducted to determine the feasibility of using wetlands for treatment of shrimp processing wastewater in Dulac, Louisiana [41,59]. The avoided cost estimate approach was used to compare costs of conventional on-site treatment of the shrimp processing effluent (the dissolved air flotation method) with the cost of wetland treatment, which is in this case. The annualized cost of the conventional treatment would be \$214,000 per year, as compared to wetland treatment costs of \$63,000 per year. This is a potential cost savings of \$1.5 million dollars over 25 years (Table 7).

Ko et al. [29] conducted a comparative embodied energy analysis, which calculates energy used directly and indirectly, for wetland treatment versus a conventional sand filtration system for Breaux Bridge. The energy capital cost of building a sand filtration method was estimated as 40 Tera joules (TJ), and annual operation and maintenance (O&M) cost was 2.1 TJ. In total, 82.6 TJ would have been used to treat wastewater over 20 years for capital costs and accumulated O&M costs for sand filtration. The energy capital cost for wetland treatment was estimated as 1.5 Giga joules (GJ), and annual O&M cost was 447 GJ, resulting in 11.1 TJ of energy costs for wetland treatment over 20 years. Thus, wetland treatment used 7.4 times less energy than sand filtration. The embodied energy savings of 71.5 TJ over 20 years is equivalent to 11,354 barrels of crude oil. After considering additional benefits of wetland treatment, such as wetland maintenance and increased net primary productivity, the benefit-cost ratio of the wetlands method is about 14.3 times higher than the sand filtration method.

Most wetland treatment has focused on constructed wetlands primarily to provide a high degree of hydrologic control. In Louisiana, a dense network of canals and levees has left many wetlands hydrologically isolated and this confers a similar degree of hydrologic control as for constructed wetlands. With natural wetlands plentiful, it is generally unnecessary to construct artificial wetlands in Louisiana. These isolated natural wetlands provide a practical economic solution for many small communities that are widely dispersed in the coastal zone.

8. Regulatory and policy considerations

In the State of Louisiana, the La. DEQ, in consultation with EPA, regulates wastewater treatment and the discharge of treated effluents. Over the past 15 years, scientists, regulatory personnel and dischargers have worked closely to develop an approach where wetland treatment systems meet water quality goals. In most cases, a preliminary feasibility analysis, generally lasting 2–4 months, is carried out to determine if a particular discharger is a promising candidate for wetland treatment. This analysis includes a determination of such factors as wetland size and characteristics, preliminary loading rate calculations, distance from the existing plant to the wetlands, and discussions with regulatory personnel. If it is decided to continue with the process, a year-long UAA is carried out that describes background ecological conditions of the candidate site (hydrology, soil and water chemistry, vegetation, animal populations, and toxic materials), analyzes the feasibility of wetland treatment, and provides preliminary engineering design and cost analyses. The wetland treatment system is designed so that loading rates are low to ensure a high nutrient retention in the wetland (see Table 1 for examples of loading rates). The UAA then forms part of the permit application. The permit designates effluents limits for discharge to the wetland and outlines monitoring requirements for the permit. Generally, effluent limits are considerably less than for direct discharge to a water body because of the ability of the wetland to process and assimilate nutrients and organic matter in the effluent. For example, a municipality that would receive limits of 10,15, 5 mg l^{-1} (BOD_5, TSS, NH_3) for discharge directly to an open water body might receive limits of 30, 30 mg l^{-1} (BOD_5, TSS). Permits do not set forth nutrient limits because the design loading rate is low enough to ensure high nutrient assimilation. There is a requirement for disinfection so that pathogens are not discharged to wetlands and toxic materials must be below state and federal limits. Permits also have a "dystrophic exception" because of low naturally occurring dissolved oxygen levels in wetlands. This means that there is no limit set for dissolved oxygen. After the permit is issued, the discharger constructs the project, starts discharge and initiates monitoring. Monitoring is required for the life of the project with annual monitoring reports.

One of the most important objectives of wetland assimilation is to ensure compliance with the Louisiana Water Pollution Control Regulations and the Federal Clean Water Act. The purpose of these laws is to protect or enhance public water including wetlands, including beneficial uses such as fish and wildlife propagation. The laws require criteria (as set forth in the permit) to protect the beneficial uses and contain an anti-degradation policy that limits lowering of water quality. As demonstrated by the information on the different wetland projects, these systems enhance wetlands and lead to substantial water quality improvement.

The use of wetlands for wastewater assimilation has important implications for two regulatory and policy issues relative to water quality. These are 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 water quality standards and an allocation of that amount to the pollutants sources. In the case of water quality problems related to over-enrichment and eutrophication, the pollutants of interest are nutrients and non-toxic organic compounds (i.e., sewage). The criteria by which successful attainment of water quality goals is normally dissolved oxygen (DO), generally $5\,\mathrm{mg\,l^{-1}}$. There is a broader issue with regard to TMDLs if a DO of $5\,\mathrm{mg\,l^{-1}}$ is appropriate for sluggish, lowland streams of South Louisiana, often bordered by wetlands, where natural DO levels are frequently below this value. But there is no doubt that there has been significant water quality deterioration in watersheds where nutrients and organic compounds come from a variety of sources including agriculture, sewage, and urban non-point sources. A problem that arises for a small municipality in a watershed dominated by other pollutant sources (such as agriculture or a large city) is that TMDL calculations will force very low discharge limits on its effluent. The use of wetland assimilation allows higher discharge limits to the wetland because in a well designed treatment system, there is little if any measurable discharge to the water body on the other side of the receiving wetland.

Nutrient enrichment has led to a suite of water quality problems in water bodies world wide. Eutrophication leads to problems such as algal blooms—sometimes with the development of harmful algal species, changes in community structure, and low dissolved oxygen (hypoxia or anoxia). One well-known example is the large hypoxia zone off the Louisiana coast caused by excessive nutrients in the Mississippi River. Turner and Rabalais [61] and Mitsch et al. [62] proposed that the restoration or creation of wetlands throughout the Mississippi basin could be part of the solution to this problem. The reduction of nutrient inputs to water bodies is a key element in the solution of eutrophication problems. The setting of nutrient limits involves the development of over-enrichment assessment tools at the regional watershed and water body levels. There is a major focus on the development of water body-type technical guidance and region specific nutrient criteria. States must adopt numerical nutrient criteria into their water quality standards. Wetland assimilation can be an important way to meeting nutrient reduction standards because, as demonstrated above, a well designed system can result in nutrient reduction in excess of 90%.

9. Conclusions

Results from several ongoing and completed studies of wetland treatment systems in Louisiana indicate that they are achieving the ecological goals of enhancing water quality, stimulating vertical accretion, and increasing productivity. At low loading rates, nutrient reductions are high, often greater than 80%, due to plant uptake, denitrification, and burial. There are substantial economic and energy savings for small communities and non-toxic industrial processors. The regulatory review and permit process ensures that projects comply with State and Federal clean water laws. As water quality regulations become more stringent, and federal grants become less available, it will be increasingly difficult for small coastal communities to meet water quality standards using conventional treatment methods. Wetland wastewater treatment can provide an economically viable, effective and sustainable alternative to expensive conventional tertiary treatment. Additionally, it serves as a means for wetland restoration in the subsiding coastal zone.

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