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Nutrient interactions, plant productivity, soil accretion, and policy implications of wetland enhancements in coastal Louisiana

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NUTRIENT INTERACTIONS, PLANT PRODUCTIVITY, SOIL ACCRETION, AND POLICY IMPLICATIONS OF WETLAND ENHANCEMENTS IN COASTAL LOUISIANA

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ABSTRACT

Ecosystem response, stakeholder interactions, and the policy implications to a wetland assimilation project are reported here for the City of Mandeville, St. Tammany Parish, Louisiana. Between September 1998 and October 2004, input of secondarily treated wastewater effluent was found to have a net positive effect on the downstream wetland receiving basin. The major hydrologic inputs to the system are the effluent, precipitation, and back water flooding from Lake Pontchartrain. Nutrient levels were generally low except in the immediate vicinity of the outfall and removal efficiencies of N and P ranged from 44% to 87% and 25% to 93%, respectively. On average, TN and TP removal efficiencies were 59% and 69%, respectively, for the study period. Aboveground net primary production of the freshwater forest system was high downstream of the effluent discharge. Also downstream of the outfall, accretion rates were double the rate of relative sea level rise in the area. Re-direction of nutrient-enhanced effluents from open water bodies to wetland ecosystems may maintain plant productivity, sequester carbon, maintain coastal wetland elevations in response to sea-level rise in addition to improving overall surface water quality, reducing energy use, and increasing financial savings. Stakeholder interactions can often be as difficult to resolve as scientific questions. Further progress to improve water quality and regulate point source pollution often requires adjustment in policy strategies to enhance society's capacity to deal with more problematic issues of non-point source pollution. High cost and economic impacts on communities will propel the search for cost-effective water quality management. In addition, cooperation between the public and private sectors can build trust, consensus, and the ability to implement coastal resource projects. In this Mandeville, Louisiana, case, the use

of science-driven solutions in natural resource management was successful in developing cost savings and coastal wetland preservation from the renewable ecological engineering technology of wetland wastewater assimilation. Integration of a national carbon and wetland policy may stimulate investments in energy efficiency and wetland enhancement. The next step must be the development of an annual revenue source to encourage state and private landowners to work together by providing incentive to promote wetland enhancement.

CHAPTER 1

INTRODUCTION

Background

Several important issues are related to the utilization of coastal wetlands for wastewater assimilation, as demonstrated in a case study at Mandeville, Louisiana. The scientific issues with regards to wetland wastewater assimilation have often been addressed in terms of primary productivity, nutrient removal and water quality, soil elevation dynamics, and the effects on aquatic organisms and animal populations. Broader issues with regard to wetland wastewater assimilation, such as stakeholder interactions, energy savings and carbon sequestration have received less attention. The Mandeville, Louisiana, project included a research component to evaluate scientific concerns as well as including significant interactions between public groups and private individuals ultimately to achieve project implementation. Future projects like the demonstration study at Mandeville may assist communities in achieving natural resource sustainability in light of increased energy demand and climate change.

Literature Review

Currently, the Louisiana coastal zone is experiencing tremendous change due to natural processes and anthropogenic activities. For the past 5,000 years, Mississippi River inputs introduced freshwater, sediments, and nutrients to downstream interdistributary wetlands of the deltaic plain (Roberts 1997). This once fertile delta contains about 2.2 million ha of tidal marshes, coastal swamps, and shallow open water areas. However, the deltaic wetlands and estuaries have been in a state of decline since the construction of Mississippi River flood protection levees in the early 1900s (Mossa 1996; Day et al. 2005). The levees

were initially discontinuous and non-uniform (circa 1880s), but with the Flood Control Act of 1928, these levees have been uniformly maintained to a height of 4-6 m along both sides of the river channel from the confluence of the Red, Atchafalaya and Mississippi Rivers, to Head of Passes, a river channel distance of approximately 500 km. Regional impacts such as increasing human populations and global impacts like climate change are expected to intensify the problems of degraded water quality, subsidence, and wetland loss in the coastal zones (Day et al. 2000b, 2004).

Breaux and Day (1994) first proposed a policy to address two of the major environmental problems in coastal Louisiana: high levels of surface water pollution and the high rate of wetland loss. Their work, as well as other studies, indicates that secondarily treated wastewater can stimulate primary productivity and enhance wetland vertical accretion through increased organic matter production and deposition (Odum 1975, 1978; Turner et al. 1976; Mudroch and Copobianco 1979; Bayley et al. 1985; Clarke et al. 1985; EPA 1987; Kuenzler 1987; Knight 1992; Craft and Richardson 1993; EPA 1994; Rybczyk 1997; Hesse et al. 1998; Rybczyk et al. 2002). Studies also indicate that wetland systems, both natural and constructed, are effective at removal of nutrients and pollutants by physical, chemical, and biological processes (Kemp and Day 1981, 1984; Godfrey et al. 1985; Hammer 1989; Conner et al. 1989; Kadlec and Alvord 1989; Faulkner and Richardson 1989; Knight et al. 1994; Richardson 1999; Jackson and Pardue 2000; Pardue and Shin 2000; Shin et al. 2000; Kassenga et al. 2003). In general, the use of coastal wetlands for wastewater assimilation can expect benefits through improved water quality, higher accretion rates, an increase in plant productivity, and the overall conservation of financial and energy resources (Day et al. 2004).

The handling of raw municipal wastewater undergoes several processes (Kadlec and Knight 1996). Primary treatment will remove most of the solids and reduce the amount of total suspended solids (TSS) through a sedimentation process, whereas, secondary treatment generally consists of the additional removal of solids, a reduction in biochemical oxygen demand (BOD), and a reduction in the amount of nutrients (Richardson and Davis 1987; Kadlec and Knight 1996). After disinfection through chlorination or ultraviolet light, the main purpose of tertiary wastewater processing is the further removal of nutrients from the watercourse (Boyt 1976; Richardson and Davis 1987; Kadlec and Knight 1996). Tertiary wetland treatment has begun to be advocated to control excessive nutrient loading to receiving areas and is dependent on the ability of these systems to remove nutrients, primarily nitrogen and phosphorus, by interaction with vegetation, soils, and the atmosphere (Richardson and Davis 1987; Kadlec and Knight 1996).

The nutrient cycling and transformations that occur in wetlands are important because these mechanisms can either temporarily or permanently remove nutrients from the wastewater stream (Patrick 1990; Schlesinger 1978; Mitsch and Gosselink 1993; Boustany et al. 1997). Organic nitrogen forms in wastewater include urea, uric acid, amino acids, and proteins while inorganic forms include ammonium (NH_4^+), ammonia (NH_3), nitrite (NO_2^-), nitrate (NO_3^+), nitrous oxide (N_2O), and nitrogen gas (N_2) (Chen and Patrick 1980; Reddy et al. 1980; Boustany et al. 1997; Kadlec and Knight 1996). Important nitrogen transformations in wetlands include: 1) ammonification (or mineralization), 2) nitrification, and 3) denitrification (Kadlec and Knight 1996). The process of denitrification can be considered a permanent loss pathway in wetlands for nitrogen whereas plant uptake can be considered a

long-term loss pathway if the organic material becomes incorporated into the anaerobic soils (Boustany et al. 1997; Day et al. 2004).

The processes of phosphorus removal in wetlands can take place by adsorption-precipitation by wetland soils, by plant uptake, and by immobilization by microorganisms (Brown et al. 1984; Kadlec and Knight 1996; Yarbro 1979). Again, as with nitrogen, phosphorus uptake in plants can be considered a long-term loss due to storage and incorporation with the soil (Day et al. 2004; Nessel 1978). Only burial is considered a permanent sink for phosphorus (Brown et al. 1984; Richardson and Craft 1993). Nutrient retention or loss in wetlands is related to loading rate, with higher retention at low loading rates (Richardson and Nichols 1985; Faulkner and Richardson 1989; Richardson 1999; Mitsch et al. 2001; Fisher and Acreman 2004; Mitsch and Jorgensen 2004). In coastal Louisiana, where wetland areas are generally large and there is a high burial capacity due to subsidence, high nutrient retention can be expected if loading rates can remain low (Blahnik and Day 2000; Zhang 1995; Zhang et al. 2000; Day et al. 2004). This high rate of burial due to subsidence makes the assimilation of wastewater effluent another possible restoration method in coastal wetlands impacted by high rates of relative sea level rise (RSLR = eustatic sea level rise plus subsidence) (Day et al. 2004).

Research has shown that wetland soil surfaces can persist in the face of RSLR when vertical accretion and elevation gain equals or exceeds the rate of water level rise (Delaune et al. 1983; Cahoon et al. 1995; Nyman and DeLaune 1991a). The addition of wastewater effluent can advance vertical soil accretion by increased organic matter production and subsequent incorporation with soil material, and thus help offset RSLR (Rybczyk et al. 2002; Day et al. 2004). Rybczyk et al. (2002) indicated that coastal sediment accretion dynamics

over time would be able to keep pace with RSLR where decomposition, dewatering, and compaction of sediments and organic matter were buried and reduce the volume and thickness of each yearly cohort. Day et al. (2004) cite an enhanced rate of carbon burial and sequestration as a result of providing wastewater effluent to subsiding coastal wetlands. Because subsiding coastal wetlands can endlessly accrete and bury carbon-rich materials, these habitats have tremendous potential to accumulate or sequester carbon over time (Chmura et al. 2003; Day et al. 2004; Brevik and Homburg 2004). Increasing primary productivity is critical in parts of coastal Louisiana where subsidence can result in a RSLR nearly ten times greater than eustatic sea level rise (Conner and Day 1988; Penland et al. 1988). Increased productivity leads to higher organic soil formation that can enhance the accretion necessary to offset subsidence (Day et al. 2004).

Forested wetlands have productivity responses to additions of wastewater effluent (Mitsch and Ewel 1979; Taylor et al. 1990; Delgado 1995; Rybczyk et al. 1995,1996; Hesse et al. 1998). The addition of wastewater effluent to wetlands can be thought of as providing nutrient fertilization for the plants, but there exists the possibility of negative impacts to a wetland due to wastewater application (Kadlec 1983). Some wetland systems in Florida (Boyt et al. 1977; Ewel and Bayley 1978; Lemlich and Ewel 1984; Nessel and Bayley 1984) and Louisiana (see Day et al. 2004) continue to remove major amounts of wastewater nutrients after 20-45 years. Some studies indicate an increase in the net primary productivity of forests with nutrient addition (Brown 1981; Ewel 1976). Brown (1981) showed that a cypress dome in Florida receiving wastewater had a significant increase in net primary productivity, litterfall, and plant biomass. A tree ring analysis in a Louisiana forested wetland showed a significant and sustained increase in stem growth for nearly 50 years due to the

long-term addition of wastewater (Hesse et al. 1998). However, some studies depict no overall increase in litterfall (Deghi et al. 1980) while other forest responses are dependent on the nature of the effluent. When raw wastewater was discharged to a cypress wetland in Florida, growth initially decreased, but then increased growth when secondarily-treated wastewater was added (Lemlich and Ewel 1984).

Increasingly, wetlands receiving secondarily treated wastewater are being utilized for ancillary benefits in addition to enhancements to water quality, accretion and plant productivity (Hammer 1989; Knight 1992, 1997; Worrall et al. 1997). The presence of water, vegetation, and elevated levels of nutrients can attract fish and wildlife species to these areas (Knight 1992, 1997). In turn, these fish and wildlife populations attract the attention of humans for harvested products (furbearers, alligators), recreational opportunities, and education (Knight 1997).

As wetlands have been shown to be effective wastewater assimilation systems, they are also being investigated as financially sound investments and as being energy efficient (Breaux et al. 1995; Day et al. 2004). Day et al. (2004) lists several economic analyses that depict cost savings by using wetland assimilation over conventional wastewater treatment on the order of millions of dollars for initial construction as well as the cost for operating and maintaining the conventional plant. Wetland wastewater treatment can also produce energy savings over a 20 year period equivalent to over 10,000 barrels of crude oil (Ko et al. 2004).

Study Area

Present day coastal Louisiana was formed over the past 5000 years as sedimentation from the Mississippi River created overlapping deltaic lobes in the shallow nearshore water of the Gulf of Mexico (Scruton 1960; Roberts 1997). Historically, floods from the Mississippi

River provided sources of freshwater, nutrients, sediments and organic materials into the Mississippi River delta plain (Delaune et al. 1983; Hatton et al. 1983; Roberts 1997, Day et al. 2000b), and massive overbank flooding and river crevasses provided water directly to the wetlands adjacent to the river and its distributaries (Hatton et al. 1983; Kesel 1988, 1989; Roberts 1997; Day et al. 2000b). This introduction of mineral sediments contributed directly to vertical accretion and the associated nutrients stimulated organic soil formation (Nyman and Delaune 1991; Mendelssohn and Morris 2000). Recently, due to river channelization and flood protection levees along the lower Mississippi River, riverine input to most of the delta plain has been reduced dramatically, resulting in declining plant productivity and reduced regeneration, low soil accretion rates, prolonged periods of inundation, and coastal wetland breakup and loss (Conner and Day 1988, 1992; Day and Templet 1989; Day et al. 2000b).

The City of Mandeville, Louisiana in southern St. Tammany Parish is situated on the northeastern edge of the Mississippi River alluvial plain on Pleistocene Terrace soils of the Gulf Coastal plain (Figure 1). Bayou Chinchuba and Bayou Castine form the western and eastern boundaries of the City of Mandeville, respectively, and each discharges into Lake Pontchartrain, a 1600 km^2 brackish-water lagoon. Both of these tidal streams are typical of other streams with forested wetlands in the Gulf Coastal plain, containing seasonally flooded cypress-tupelo and bottomland hardwood forests in the lower drainage basins. These forested wetlands are dominated by water tupelo (*Nyssa aquatica*), swamp blackgum (*Nyssa biflora*) and baldcypress (*Taxodium distichum*) (Day et al. 2000a). Swamp blackgum typically dominates the upper floodplain portion of both watersheds, and water tupelo and baldcypress become much more dominant downstream (Rheinhardt et al. 1998; Day et al. 2000a).

The area has a subtropical climate, with a mean annual air temperature of 21.1^{o}C and mean annual rainfall of 151.4 cm yr^{-1} (U.S. Geological Survey (USGS) 1998). Stream flows in Bayous Chinchuba and Castine vary greatly, depending on precipitation events. The lower floodplains of these bayous appear to be hydrologically controlled by stream flow and back flooding from Lake Pontchartrain.

Since 1989, the City of Mandeville has discharged its treated wastewater into the Bayou Chinchuba wetland (7195.5 m^3 day^{-1}). The wastewater facility currently consists of three (61 x 183 m) aerated lagoon cells, a three-celled rock reed filter, and an ultraviolet disinfection system.

Global Objectives and Hypotheses

Excessive nutrient input into the lower Mississippi River basin and the nearshore coastal zone of Louisiana has created a suite of water quality problems (Turner and Rabalais 1991). This nutrient introduction has resulted in changes in ecosystem structure, phytoplankton blooms and large areas of low dissolved oxygen (Turner and Rabalais 1991). In addition, a vast network of canals for navigation and flood-protection levees have isolated many wetlands to hydrologic connections and retained nutrients within these open water bodies (Day et al. 2004).

Recent efforts to restore and enhance wetlands in the subsiding delta region of Louisiana have focused on attempts to decrease vertical accretion deficits by either physically

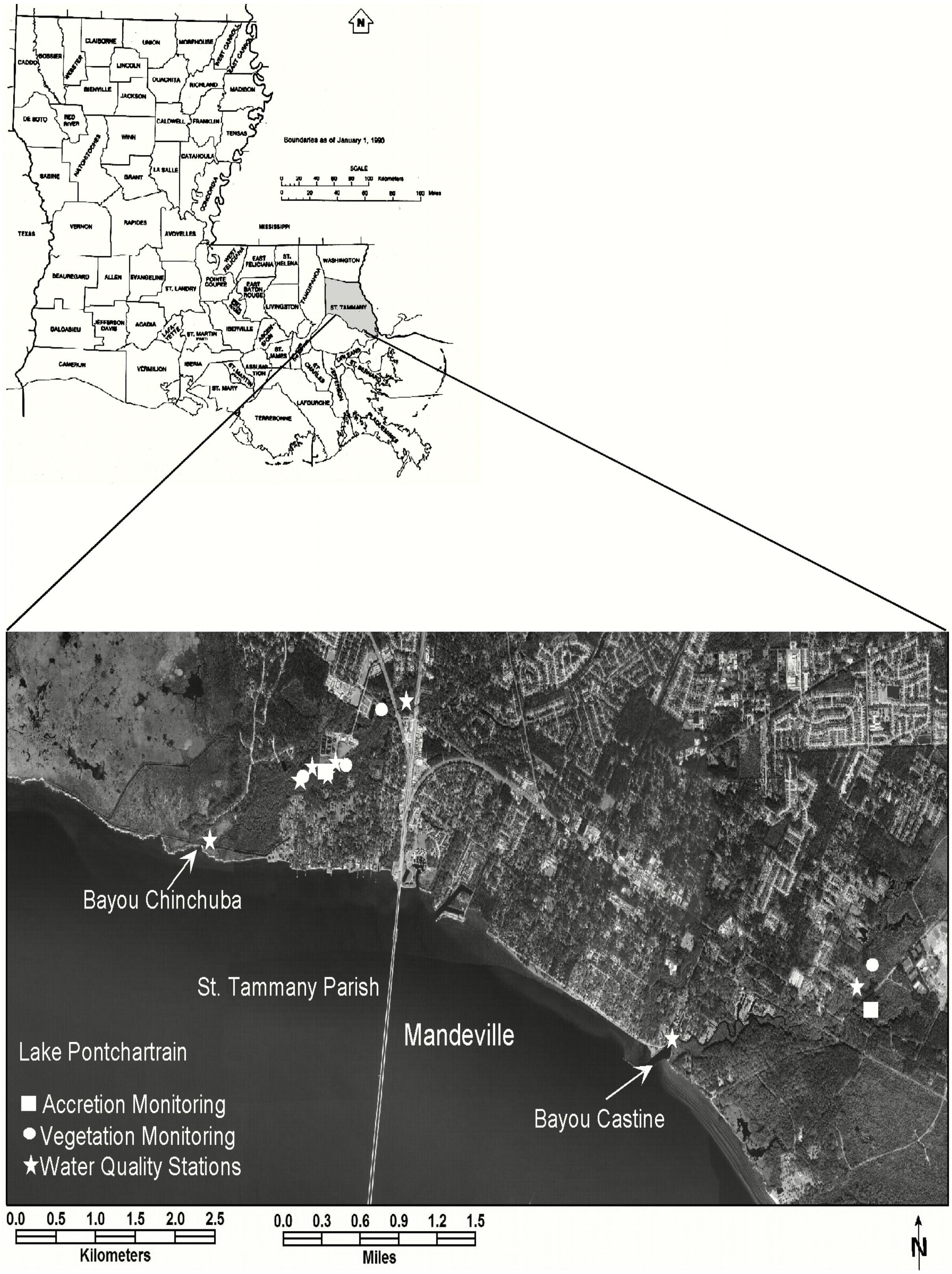

Figure 1. Map of Study Area

adding sediments to wetlands using diversions or dredged material (Day et al. 2004; Delaune et al. 2003; Delaune and Pezeshki 2003), or by installing sediment trapping mechanisms (i.e. sediment fences, Boumans et al. 1997), thus increasing elevation and relieving the physio-chemical flooding stress (Boesch et al. 1994; Mendelssohn and Morris 2000). Breaux (1992) and Day et al. (1992) proposed an alternative strategy to add nutrient rich secondarily treated wastewater to hydrologically isolated and subsiding wetlands, that in turn could promote vertical accretion through increased organic matter production and deposition. Their work, and other studies, has shown that treated wastewater does stimulate productivity and accretion in wetlands and could be a viable wetland sustainability undertaking (Odum 1975; Turner et al. 1976; Mudroch and Copobianco 1979; Bayley et al. 1985; Knight 1992; Craft and Richardson 1993; Rybczyk et al. 2002; Hesse et al. 1998).

Increasingly, other issues will contribute a significant role in wetland enhancement and restoration efforts in coastal Louisiana. Global climate change, largely from anthropogenic activities, has been predicted to change not only temperature and a resulting rise in sea level, but also may lead to an increase in the number and magnitude of coastal storms, decreases in soil moisture and reductions in freshwater runoff (Niklaus and Korner 2004, Day et al. 2005).

The availability of current energy resources and the funding needed to complete restoration projects are other issues that will be important in efforts to restore coastal Louisiana. Prior restoration projects were constructed with relatively inexpensive energy resources. Costs for these projects as well as proposed costs for planned coastal restoration work were valued at 50-70% of the price of current crude oil dollars (LACOAST 2005).

Inexpensive and energy-efficient solutions will be likely alternatives to coastal restoration measures.

The aim of this research was to conduct an investigation of nutrient retention, wetland productivity, and soil accretion in a wetland system north of Lake Pontchartrain, near the City of Mandeville, St. Tammany Parish, Louisiana (Figure 1). In addition, investigations into the use of science-driven solutions in the decision-making and policy-making department of society, and a possible national policy of nutrient resource management were conducted that may guide or encourage future researchers and policy-makers towards a national investment for coastal wetland enhancement.

Specific hypotheses were:

(1) Wastewater input has no effect on overall hydrologic input to these wetlands;

(2) Wastewater nutrient constituents were similar along the stream length;

(3) There was no effect of wastewater on aboveground NPP;

(4) Accretion rates were similar at adjacent wetland systems;

(5) Stakeholder interactions would have no effect on project development.

The alternate hypotheses included yearly effects and wastewater input on overall hydrology, nutrient constituents, aboveground NPP and accretion. In addition, interactions of stakeholders into the project planning process constituted an alternate hypothesis for project implementation.

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CHAPTER 2

FORESTED WETLAND DYNAMICS RECEIVING TREATED MUNICIPAL WASTEWATER: NUTRIENT INTERACTIONS, FOREST PRODUCTIVITY, AND ACCRETION

Introduction

The coastal zone of Louisiana is experiencing tremendous change due to natural and anthropogenic processes. Increasing human populations and global climate change will likely exacerbate problems of degraded water quality, subsidence, and coastal wetland loss in the coastal zone (Day et al. 2000; 2004). Increasingly, natural or constructed wetlands are being used for wastewater assimilation. Although wetlands have been used to treat wastewater for centuries, only in the past several decades has the response to such use been studied in a comprehensive, scientific manner (Richardson and Davis 1987; Kadlec and Knight 1996; Day et al. 2004). The ability of wetlands to perform water purification functions has been well established for natural watersheds (Khalid et al. 1981a; 1981b; Kemp et al. 1985; Nichols 1983; Richardson and Nichols 1985; Knight et al. 1987; Richardson and Davis 1987; U.S. EPA 1987; Conner et al. 1989; Faulkner and Richardson 1989; Kadlec and Alvord 1989; Kadlec and Knight 1996; Day et al. 2004). Studies in the southeastern United States have shown that wetlands chemically, physically, and biologically remove pollutants, sediments and nutrients from water flowing through them (Wharton 1970; Shih and Hallett 1974; Kitchens et al. 1975; Boyt, 1976; Nessel 1978; Yarbro 1979; Tuschall et al. 1981; Yarbro et al. 1982; Nessel and Bayley 1984; Kuenzler 1987; Hesse et al. 1998; Day et al. 1999; 2004; Lane et al. 1999; 2001; Zhang et al. 2000; Rybczyk et al. 2002).

From an ecological perspective, interest in wetlands to assimilate effluent is based on the principle that the free energies of the natural system are both capable of and efficient at driving the cycle of production, use, degradation, and reuse (Odum 1978; Breaux and Day 1994; Mitsch and Jorgensen 2004). The basic idea underlying wetland wastewater assimilation is that the rate of application must balance the rate of decay or immobilization, and the primary mechanisms by which this balance is achieved are physical settling and filtration, chemical precipitation and adsorption, and biological metabolic processes resulting in eventual burial, storage in vegetation, and denitrification (Conner et al. 1989; Kadlec and Alvord 1989; Patrick 1990). Municipal effluent discharge generally introduces nutrients as NO_3, NH_4, PO_4, and organic forms, and these nutrients can be removed in the short-term by plant uptake, and in the long-term by peat and sediment accumulation, and in the case of nitrogen, by the process of denitrification (Hemond and Benoit 1988; Day et al. 2004). Wetlands with relatively long residence times are best suited for Biochemical Oxygen Demand (BOD) reduction, bacteria dieback, and elimination of microorganisms (Reed 1991; Kadlec and Knight 1996).

Wetlands have the ability to remove nutrients from inflowing water, primarily dependent on the volume and nutrient concentration of the input water, and the area of the receiving wetlands (Mitsch et al. 2001). This is expressed as the loading rate which is non-linearly related to nutrient uptake with high nutrient uptake at low loading rates and low uptake at high loadings (Richardson and Nichols 1985; Faulkner and Richardson 1989; Mitsch et al. 2001). Nutrient uptake is also influenced by temperature and the hydrology of the specific wetland site (Blahnik and Day 2000).

In Louisiana, there are sites that have received discharges from 10 to 50 years and continue to have high nutrient removal rates (Zhang et al. 2000; Day et al. 2004). In addition, as long as peat accumulation remains below the water surface, nutrients can continue to be processed in natural wetland systems (Deghi et al. 1980; Rybczyk et al. 2002). In the Louisiana coastal zone, a high rate of relative sea level rise (RSLR) due to geologic subsidence gives rise to a high burial rate as a permanent loss pathway for nutrients (Rybczyk et al. 2002; Day et al. 2004).

Coastal wetlands have been shown to persist in the face of RSLR when vertical accretion equals or exceeds the rate of subsidence (Delaune et al. 1983; Baumann et al. 1984; Stevenson et al. 1986). In the past, seasonal overbank flooding of the Mississippi River deposited large amounts of sediments into the interdistributary wetlands of the delta plain (Roberts 1997). Not only did these floods provide an allochthonous source of mineral sediments, which contributed directly to vertical accretion, but also the nutrients associated with these sediments promoted vertical accretion through increased autochthonous organic matter production and deposition due to organic soil through increased root growth (Delaune et al. 1983). This sediment and nutrient source was largely eliminated by the early 20th century with the completion of levees along the entire course of the lower Mississippi (Mossa 1996), resulting in vertical accretion deficits (RSLR > accretion) throughout the coastal region (Hatton et al. 1983; Day et al. 2000).

Recently, there have been renewed efforts to restore or enhance wetlands in the subsiding delta region of Louisiana. These attempts have focused on eliminating vertical accretion deficits by either physically adding sediments to wetlands using diversions or dredged material (Day et al. 2004; Delaune et al. 2003; Delaune and Pezeshki 2003), or by

installing sediment trapping mechanisms (i.e. sediment fences, Boumans et al. 1997), thus increasing elevation and relieving the physio-chemical flooding stress (Boesch et al. 1994, Mendelssohn and Morris 2000). There have been numerous studies indicating that treated wastewater will stimulate productivity and increase accretion in wetlands (Odum 1975; Turner et al. 1976; Mudroch and Copobianco 1979; Bayley et al. 1985; Day et al. 1992; Knight 1992; Craft and Richardson 1993; Hesse et al. 1998; Rybczyk et al. 2002). Day et al. (1992) proposed using a complementary restoration strategy in coastal Louisiana by adding nutrient rich secondarily treated wastewater to hydrologically isolated and subsiding wetlands, in an effort to promote vertical wetland accretion through increased organic matter production and deposition.

This chapter reports on the investigation of nutrient retention and forest productivity in a wetland system north of Lake Pontchartrain, near the City of Mandeville, St. Tammany Parish, Louisiana (Figure 1). Specific objectives with this research were to: 1) quantify nutrients introduced from a wastewater treatment facility into a forested wetland as well as the loading rate and removal efficiency for N and P; 2) measure the impact of this nutrient-enhanced wastewater on plant productivity and accretion in the area. The null hypotheses were:

(1) wastewater input has no effect on overall hydrologic input to these wetlands;

(2) wastewater nutrient constituents were similar along the stream length;

(3) there was no effect of wastewater on aboveground NPP;

(4) accretion rates were similar at adjacent wetland systems.

The alternate hypotheses included yearly effects and wastewater input on overall hydrology, nutrient constituents, aboveground NPP and accretion.

Study Area

Bayou Chinchuba and Bayou Castine form the western and eastern boundaries of the City of Mandeville, respectively, and each discharges into the northern portion of Lake Pontchartrain. Both of these streams are similar in size and discharge and are typical of other streams with forested wetlands in the Gulf Coastal plain (Ewel and Odum 1984; Felley 1992, USGS 1998). Both the Bayou Chinchuba and Bayou Castine lower drainage basins are seasonally flooded cypress-tupelo and bottomland hardwood forests, with a forest floor approximately 0.6 m above Mean Sea Level (MSL). The streams are tidal near the lake. Soils are classified as an Arat silty clay loam (fine silty, siliceous, non-acid, thermic Typic Hydraquents; Trahan et al. 1990). The floodplains of Bayou Chinchuba and Bayou Castine are hydrologically distinct from adjacent wetland areas due to higher elevation ridges that border each side of the floodplain area.

Bayou Chinchuba and Bayou Castine are broadleaf and needle-leafed deciduous forested wetlands dominated by water tupelo (*Nyssa aquatica*), swamp blackgum (*Nyssa biflora*) and baldcypress (*Taxodium distichum*). Bayou Chinchuba and to a lesser degree Bayou Castine exhibit vegetation zonation from the upper portion of the watershed to the lower portion. Swamp blackgum typically dominates the upper floodplain portion of both watersheds, and water tupelo and baldcypress become much more dominant downstream (Rheinhardt et al. 1998; Day et al. 2000). As the bayou becomes deeper and the floodplain wider, baldcypress becomes the dominant forest tree in the lower floodplain of both tidal streams (Rheinhardt et al. 1998). Water tupelo may also be sensitive to low salinity water in the lower reaches of both streams (Day et al. 2000).

The area has a subtropical climate, with a mean annual air temperature of 21.1^{o}C and mean annual rainfall of 151.4 cm yr^{-1} (U.S. Geological Survey (USGS) 1998). Stream flows in Bayous Chinchuba and Castine vary greatly. USGS measurements during 1998 indicated that during a majority of the year in Bayou Chinchuba, there was no discernible flow downstream of West Causeway Approach (approximately 1500 m upstream of the treatment facility outfall). However, after two rainfall events measuring 25.1 cm on 6-7 March 1998 and 8.5 cm on 14-15 July 1998, flow in Bayou Chinchuba at West Causeway Approach was measured at 1.8 m^{3} s^{-1}on 10 March 1998 and 1.0 m^{3} s^{-1}on 15 July 1998 (USGS 1998). The Bayou Chinchuba and Bayou Castine floodplains are swamps hydrologically controlled by stream flow and back flooding from Lake Pontchartrain. The area immediately downstream of the wastewater treatment facility does not have a well-defined channel area and flow from the plant spreads out through the surrounding forest areas.

Flood tides from Lake Pontchartrain can typically enter through the mouth of Bayou Chinchuba and Bayou Castine and cause backwater flooding. Tides at Manchac Pass to the west typically show a 9 to 12 cm range (Gibson and Gill 1988), however storm tides associated with tropical disturbances in the summer and fall can increase tides to much higher levels.

Little is known about near-surface groundwater interactions in the area, but in general there is little lateral groundwater movement in the fine-grained sediments of south Louisiana (USGS 1998). The low conductivity of clays (10^{-6} mm/sec, Terzaghi and Peck, 1968) coupled with the low topographic gradient indicates that horizontal and vertical groundwater velocities are more likely dominated by surface water pressure (head) and density (salinity) gradients than gravity or soil permeability. Vertical exchange of surface and groundwater is

also likely minimal. During prolonged periods of dryness, some loss of surface water to the ground would be expected when water levels rise and surface soils were not yet saturated. The lack of flow in Bayou Chinchuba during dry periods suggests that ground water input is low but does not eliminate the possibility that the bayou may recharge the water table aquifer.

Since 1989, the City of Mandeville has discharged its treated wastewater into the Bayou Chinchuba wetland (7195.5 m^3 day^{-1}). The wastewater facility currently consists of three (61 x 183 m) aerated lagoon cells, a three-celled rock reed filter, and an ultraviolet disinfection system. In 1998, the City of Mandeville was issued a discharge permit by US EPA with the following criteria: 30 day average BOD of 10 mg l^{-1} with daily maximum of 15 mg l^{-1}; 30 day average TSS of 15 mg l^{-1} with daily maximum of 23 mg l^{-1}; 30 day average NH_4 of 5 mg l^{-1} with daily maximum of 10 mg l^{-1}; 30 day average fecal coliforms of 200 colonies/ 100 ml with daily maximum of 400 colonies/ 100 ml (US EPA 1998).

Methods

<u>Hydrology</u>

Precipitation was monitored daily with a gauge located near the wastewater facility outfall. These data were compared on a monthly basis to 30-year average precipitation records at Covington, Louisiana, approximately 11 km north (NOAA 2003). Evapotranspiration was calculated on a monthly basis using Thorntwaite's equation and utilizing average monthly air temperature at the Covington, Louisiana, weather station. Water balance was determined by subtracting monthly precipitation from the monthly adjusted evapotranspiration rate. Data from a USGS gaging station on Bayou Chinchuba upstream of the wastewater facility outfall and the measured rate of flow from the wastewater facility were used to determine these additional hydrologic inputs.

Water Chemistry

Samples for water chemistry analysis were collected in September and November 1998; January, March, April, May, July, October, and December 1999; and April and October 2000. Water samples were collected in 500 ml acid-washed polyethylene bottles, stored on ice and taken to the laboratory for analysis. Nutrient analyses were performed using standard methods outlined by the Environmental Protection Agency and the Louisiana Department of Environmental Quality, (U. S. Environmental Protection Agency 1979) and included the following: nitrate-nitrite, (NO_x-N); ammonium, (NH_4-N); total N, (TN); phosphate, (PO_4-P); and total phosphorus, (TP). Chlorophyll a samples were analyzed by a modified version of the method suggested by Strickland and Parsons (1972). Chlorophyll was extracted as described by Burnison (1980) and was measured fluorometrically with a Turner Designs model 10-AU fluorometer (Greenburg et al. 1985). Within one week of sample collection, total suspended sediment (TSS) was determined by filtering 100-200 ml of sample water through pre-rinsed, dried and weighed 47 mm 0.45 um Whatman GF/F glass fiber filters. Filters were then dried for 1 hr at 105°C, weighed, dried for another 15 minutes, and reweighed for quality assurance (Greenberg et al. 1985).

For the loading rate analysis at Mandeville, I used concentrations and discharge from the treatment plant and the area of receiving wetlands. Total amounts of nitrogen and phosphorus discharged from the treatment plant were calculated based on TN and TP concentrations and the average discharge of the treatment plant (0.083 $m^3 s^{-1}$). The effective area of wetlands used in the loading rate calculations was based on the floodplain area downstream of the treatment plant outfall on Bayou Chinchuba (approximately 98 ha).

Forest Composition and Productivity

In July 1998, two 20 x 20 m plots were randomly established in four separate swamp areas, 1500 m upstream of the outfall treatment area along Bayou Chinchuba, at the treatment area outfall along Bayou Chinchuba, 200 m downstream of the treatment area along Bayou Chinchuba, and at a reference site along Bayou Castine. The upstream plots were located in an area of Bayou Chinchuba floodplain not affected by water flow from the treatment plant outfall. Within each plot, all trees > 2.5 cm in diameter at breast height (dbh) were tagged. Average diameter, basal area, relative dominance, absolute density, relative density, survivorship, and importance value (IV) were calculated for each tree species (Barbour et al., 1980). The IV of each major tree species in the plots was based on the density (total number) and dominance (basal area) in each of the plots. Mortality rates in each plot were calculated as an exponential decay rate:

Average annual mortality rate = 1-(S/No)1/y;

Where: S = Number of Survivors

No = Number of original stems

Y = Number of years between samples

Data were square-root transformed to normalize data and analysis of variance (ANOVA) used to determine if average annual rates differed.

In forested sites, biomass production is defined as the sum of litterfall and wood production (Newbould 1967). To estimate aboveground productivity within each plot, litterfall was collected from 5-0.25 m^2 litter traps with 1 mm mesh bottoms for a total of 10 litter traps at each monitoring site. The boxes were elevated to a height of 2 m above the forest floor to prevent inundation during high water periods. Litterfall was collected monthly

from August 1998 to March 2002. Litter was separated into leaves, reproductive material, and woody material, dried at 60°C for 48 hours, and weighed. Individual litterfall-trap data were converted to g/m^2 and then log transformed to normalize the data and reduce correlations between means and variance. A 2-way ANOVA was conducted to determine if litterfall varied among the eight plots over the three years, and an a-posteriori Tukey multiple comparisons test was employed to detect any significant differences.

Initial dbh measurements were taken in February 1999 when trees were dormant and trees were re-measured again in December 1999, January 2001, and January 2002 during the dormant season. Algorithms from Megonigal et al. (1997) were used to calculate biomass for each tree species. Change in biomass represents annual wood production and when used with annual leaf litterfall determines the aboveground net primary production in each plot. Data were log transformed and a 2-way ANOVA (4X3 factorial) was employed to determine if aboveground net primary productivity varied. Tukey multiple comparisons was used to detect any significant differences in the four reference areas or by years.

Sediment Accretion

Accretion was monitored at 2 swamp forest locations, one in Bayou Chinchuba and another at Bayou Castine (Cahoon et al. 1995). Feldspar was laid out 1 cm thick in three separate 0.25 m^2 plots in April 1999 and the sediment layer thickness was measured in October 2000 and again in October 2004. The three plots at each site were spaced linearly in the swamp forest along a transect from the bayou edge to the edge of the floodplain adjacent to the elevated ridge bordering the site. At the end of each sampling period, measurements of the depth of material above the marker to the nearest mm were made at 10 random locations within each plot. Each plot was considered an experimental unit and ANOVA was used to test

for total accretion between sites. Data were square-root transformed to normalize the data and an a-posteriori Tukey multiple comparisons test was employed to detect any significant differences.

Results

Hydrology

The area receives water from several primary sources, specifically rainfall, periodic stream flow from Bayou Chinchuba, local runoff, flood tides from Lake Pontchartrain, and effluent from the wastewater facility (Table 1). During the study, average annual precipitation in the Bayou Chinchuba watershed area was 149.5 cm. Comparison with the 30-year rainfall averages at Covington indicates rainfall deficits in the area from October 1998 until November 2000 during an extended region-wide drought (Figure 2). Evapotranspiration (ET) rates in the area ranged from 16.7 cm in July 1998 to 9.3 cm in December 1999. In general, evapotranspiration rates exceeded precipitation for much of the first two years of the study; rainfall began to approximate normal levels again in June 2000 (Figure 3). High rainfall events occurred during November 2000, March 2001, and June 2001, although much of the precipitation that fell in June 2001 was due to the effects from Tropical Storm Allison (Figure 3). Typically in south Louisiana, ET slightly exceeds P from May through August (Keim et al. 1995).

Downstream of the wastewater facility outfall, the floodplain area of Bayou Chinchuba receives approximately 7191.5 $m^3\ day^{-1}$ (0.083 $m^3\ s^{-1}$) of freshwater input from the wastewater plant. This amount of water delivered to the wetlands can be substantial during low precipitation periods, but it can also be overwhelmed during periods of high rainfall (Table 1). During the course of the study, flow from Bayou Chinchuba upstream of the

outfall discharge was low and I observed the effluent stream from the wastewater treatment facility generally bypassed the outfall vegetation plots due to low water levels during this time.

Table 1. Contribution of hydrologic inputs to Bayou Chinchuba, Mandeville, Louisiana. Data are annual average from 1998-2002. Floodplain area downstream of outfall is approximately 100 ha.

Input Source	Potential Hydrologic Contribution
Precipitation[1]	0.43 cm day^{-1}
Bayou Chinchuba (low normal)[2]	0.0 cm day^{-1}
Bayou Chinchuba (high flow)[2]	13.0 cm day^{-1}
Effluent to Bayou Chinchuba[3]	0.7 cm day^{-1}

[1]data from rain gauge at Chinchuba
[3]daily records from city of Mandeville
[2]data from USGS gauge

Water Chemistry

Nutrient levels in the study area are generally low with the exception of the outfall from the facility into Bayou Chinchuba (Figure 4). In general, nutrient concentrations peaked at the plant discharge location, and decreases in nutrient concentrations occurred downstream of the plant. The plant operated within the established permit criteria over the course of this study for the permitted constituents, with one exception. The samples collected in July, October and December 1999, the TSS concentration at the outfall sampling location averaged 78 mg l^{-1}, but may be due to input from upstream as that location averaged 135 mg l^{-1} for those three sample dates. The upstream sample location consistently had the lowest concentration of nitrogen species and total phosphorus along Bayou Chinchuba (Figure 4), in part due to the infrequent flows at this location during the region-wide drought. Samples collected in late 1999 and 2000 were on days associated with precipitation events. Therefore, to analyze any differences in nutrient concentrations as rainfall patterns approached the long-

term average, averages of the first six sample dates were compared with the final five nutrient collections (Figure 4).

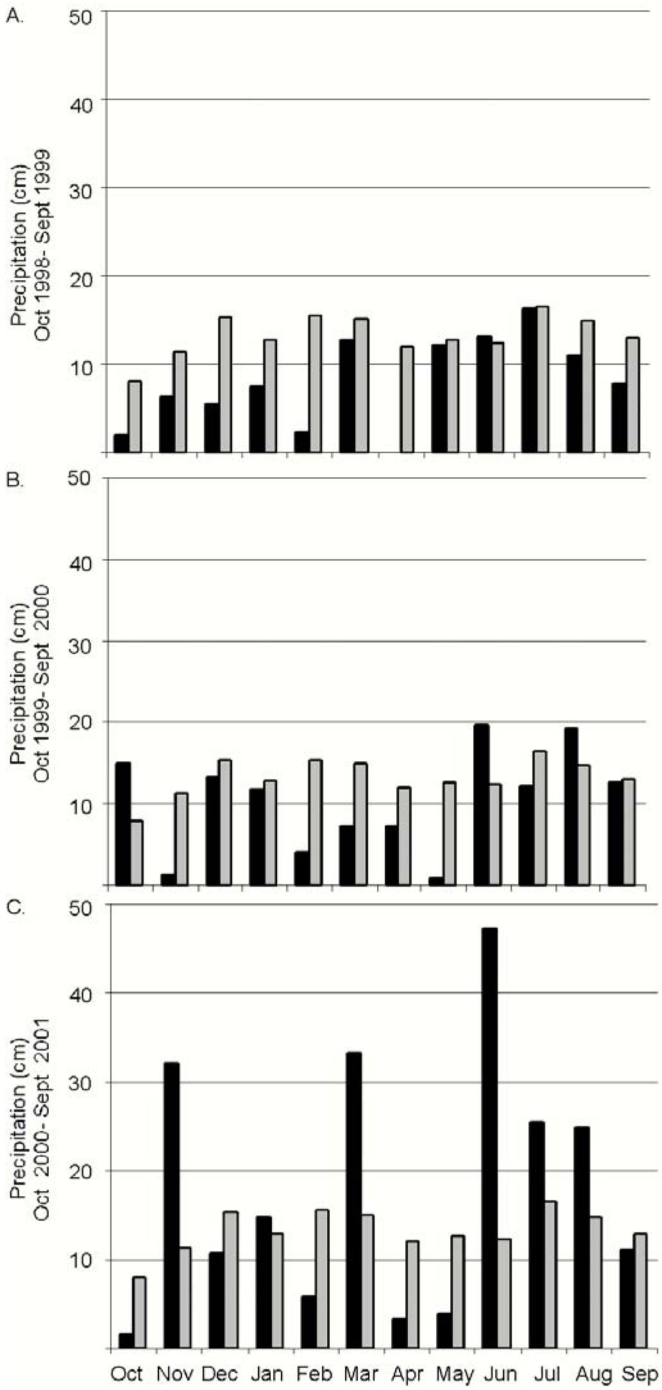

Figure 2. Mandeville precipitation (black bars) vs. 30 year average precipitation at Covington, LA (shaded bars). A) 1999 B) 2000 C) 2001

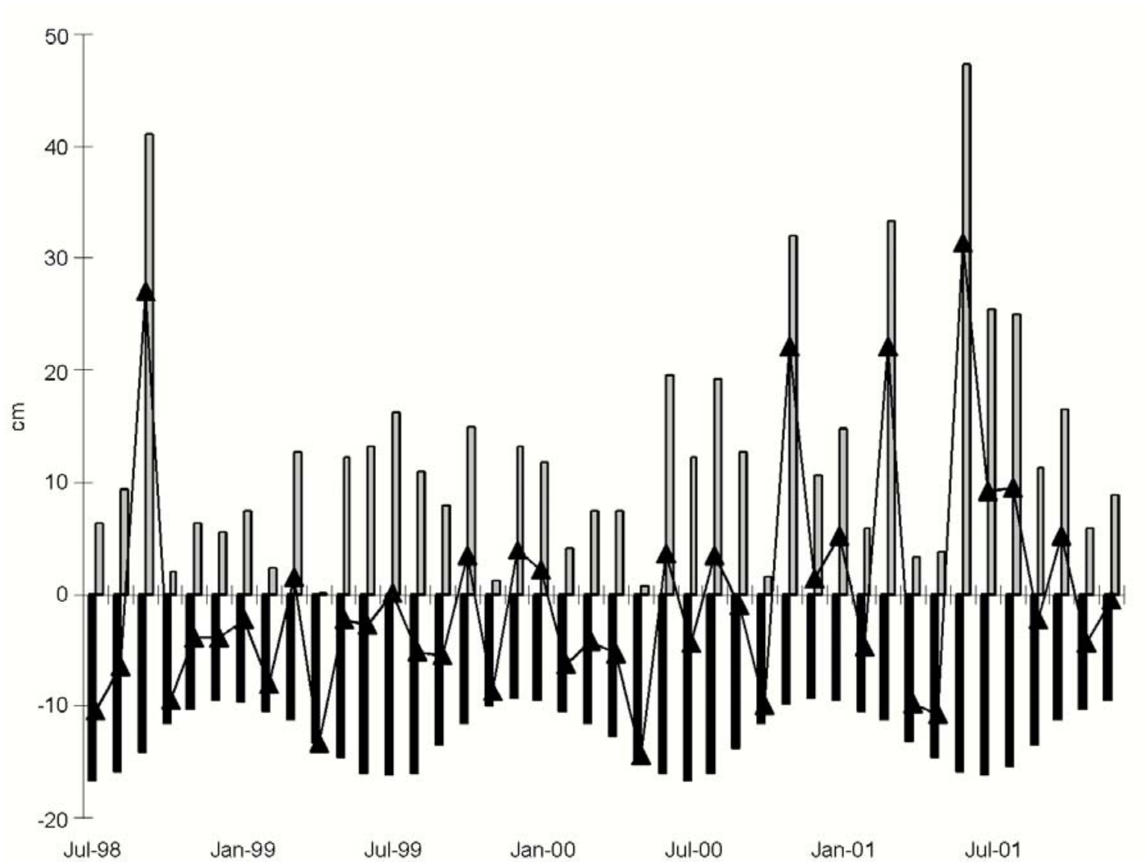

Figure 3. Water budget for study area. Black bars represent adjusted evapotranspiration, shaded bars indicate Mandeville precipitation and the black triangles indicate Mandeville potential evapotranspiration.

Nitrate concentrations averaged 1.8 mg-N l^{-1} at the outfall location, decreased to 0.2 mg-N l^{-1} at the lake, and averaged 0.3 mg-N l^{-1} in the control area (Figure 4A). Though nitrate concentrations were higher on average downstream of the outfall compared to the average concentration of nitrate upstream of the plant discharge, on average, nitrate concentrations declined by 55-93% within 1200 m of the outfall location (Figure 4A). Average decrease in nitrate concentration within 1200 m of the outfall over the entire sampling effort was approximately 82%.

Ammonium concentrations ranged from near detection limits upstream of the outfall to an average high of 2.2 mg-N l^{-1} at the outfall location (Figure 4B). Ammonium

concentration at the control locations on Bayou Castine averaged 0.2 mg-N l^{-1}. Ammonium nitrogen concentrations within 1200 m of the outfall location ranged from an increase of 36% to a decrease of 94% (Figure 4B).

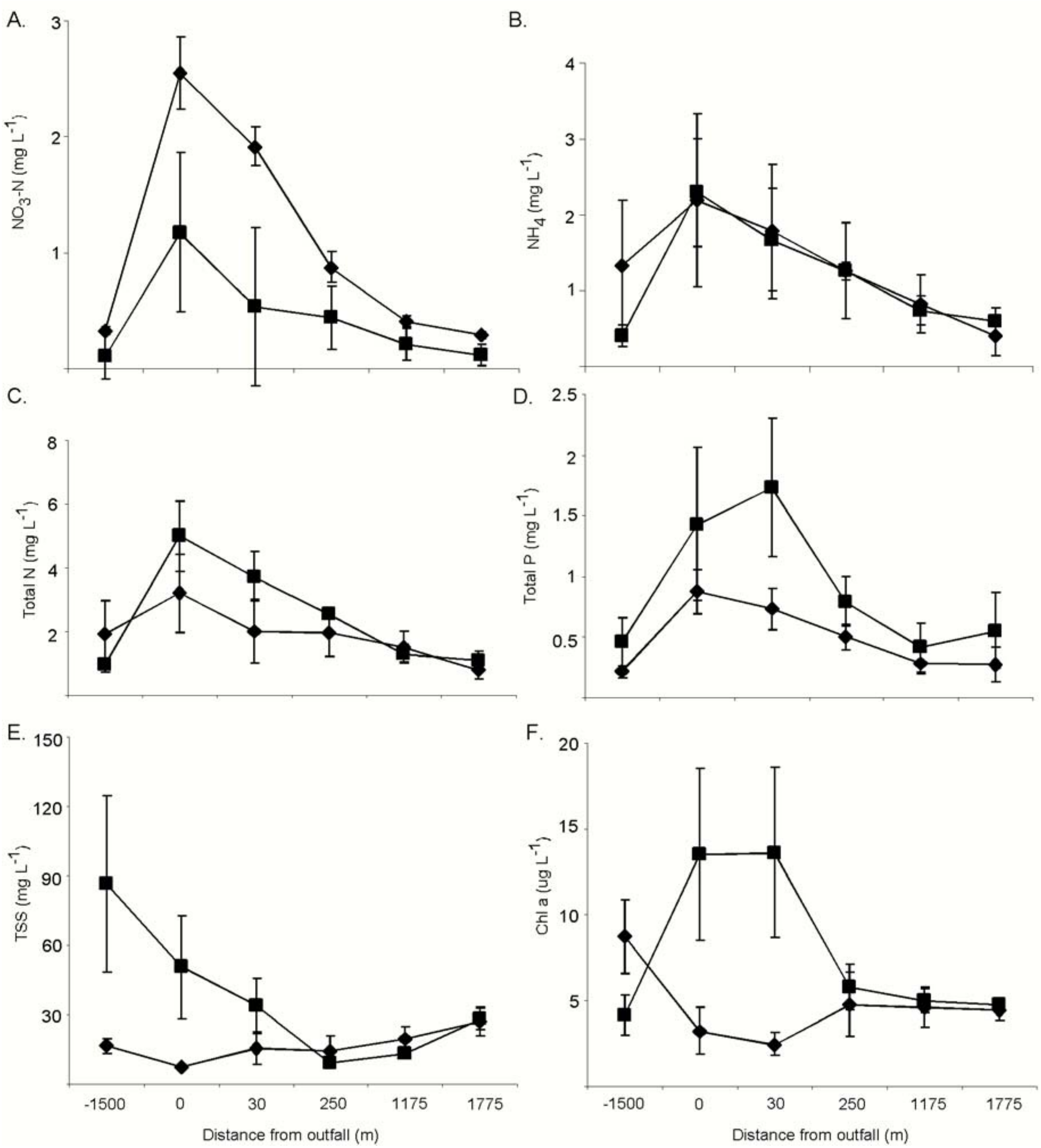

Figure 4. Nutrient concentrations (mean ±SE) in the study area for two periods (Sept 1998-May 1999=■ and July 1999- October 2000=◆) with increasing distance from the outfall. A) NO_3-N, B) NH_4-N, C) Total N, D) Total P, E) TSS, and F) Chl a.

Total nitrogen concentrations averaged 4.3 mg-N l^{-1} at the outfall location while concentrations upstream and downstream were between 1.4 -3.1 mg-N l^{-1} (Figure 4C). Total nitrogen at the control locations on Bayou Castine averaged 0.6 mg-N l^{-1}. On average, TN declined by 59% from the average outfall concentration (Figure 4C), and ranged from an increase of 32% to reductions of 87% downstream of the outfall location.

Total phosphorus concentrations ranged from below 0.5 mg-P l^{-1} upstream and downstream of the outfall to an average of 1.1 mg-P l^{-1} at the outfall location (Figure 4D). Decreases in total phosphorus were noted downstream of the outfall in a range from 25-93%, and averaged 69% reductions over all sample dates (Figure 4D). Total phosphorus levels at the control stations on Bayou Castine were also low, ranging from below detection limits to 0.5 mg-P l^{-1}.

Total suspended sediment concentrations ranged from less than 10 mg l^{-1} to 221 mg l^{-1} over the course of the study (Figure 4E). In general, TSS at the outfall location was low and within the plant permit criteria. TSS at the control sites on Bayou Castine ranged from 10-80 mg l^{-1}.

Chlorophyll a levels on Bayou Chinchuba varied from 1-10 µg l^{-1} (Figure 4F). There appeared to be no noticeable trend in the level of chlorophyll a in Bayou Chinchuba (Figure 4F). The sample sites on Bayou Castine had similar chlorophyll a values from 4-10 µg l^{-1}. The high productivity and dense canopy of the forested wetlands resulted in shaded conditions at the sample sites that probably contributed to the relatively low levels of chlorophyll.

<u>Forest Composition and Productivity</u>

The forest canopy community structure at Bayou Chinchuba and Bayou Castine (Table 2) is largely comprised of baldcypress (*Taxodium distichum*), tupelo gum (*Nyssa*

aquatica), and swamp blackgum (*Nyssa biflora*). Average tree diameter and basal area were similar between areas (Table 3). Midstory was composed of ash and red maple, understory was sparse general ground cover. Litterfall peaked during September in the plots with a swamp blackgum component; otherwise litterfall peak was December or bimodal with a peak in the fall and winter. Litterfall contribution by species reflected the composition and IV for each plot (Table 2). Tree mortality over the first two years was low, however, after the third growing season, mortality rates increased over all areas, but this increase was not significant ($F=5.1$, $P=0.6$) (Figure 5).

Leaf litter from the downstream plots was significantly higher than the other sampled areas for each of the years sampled (Table 3). Annual mean litterfall for Bayou Castine, Upstream Chinchuba, Outfall Chinchuba, and Downstream Chinchuba was 649, 498, 532, and 746 g m^{-2} yr^{-1}, respectively. There were significant litterfall differences between forest plots ($F=35.54$, $P<0.001$) (Table 3) but there was no effect by year ($F=0.41$, $P=0.8$). The interaction term was marginally significant between forest plots and year on leaf litter ($F=1.64$, $P=0.046$), possibly as a result of changing hydrological conditions between forest plots and increases or decreases in annual primary production.

Annual mean stem growth for Bayou Castine, Upstream Chinchuba, Outfall Chinchuba, and Downstream Chinchuba was 217, 462, 386, and 456 g m^{-2} yr^{-1}, respectively, but these differences were not significant. Total aboveground net primary production over the three study years averaged 959.9 g m^{-2} yr^{-1} for the upstream plots at Bayou Chinchuba; 917.1 g m^{-2} yr^{-1} for the Bayou Chinchuba outfall plots; 1202.7 g m^{-2} yr^{-1} for the Bayou Chinchuba downstream plots; and 799.2 g m^{-2} yr^{-1} for the Bayou Castine reference area (Figure 6). There were significant differences between the sampled areas ($F=7.01$, $P<0.05$) (Table 3) and

there was no effect by year ($F=0.97$, $P=0.4$). Comparisons test indicated the aboveground net primary productivity in the downstream area of Bayou Chinchuba was significantly higher than the other three reference areas ($P<0.001$) (Table 3 and Figure 6). There was no significant difference among the outfall, upstream, or Bayou Castine reference areas ($P>0.5$) for aboveground net primary productivity (Table 3 and Figure 6).

Table 2. Forest community structure at Bayou Chinchuba and Bayou Castine, St. Tammany Parish, LA USA.

Plot	Species	DBH (cm)	Stem Density (number ha^{-1})	BA (m^2 ha^{-1})	IV
Castine A					
	Taxodium distichum	39.5	125	18.5	24.2
	Nyssa aquatica	25.8	175	10	17.1
	Nyssa sylvatica	16.7	550	13.6	35.1
	Acer rubrum	11.5	250	3.3	12.9
	Fraxinus caroliniana	9.5	225	2	10.6
	Living Trees		1325	47.6	
	Dead Trees		75	0.4	
Castine B					
	Taxodium distichum	34.5	175	17.9	25.2
	Nyssa aquatica	35	300	30.7	43.1
	Nyssa sylvatica	22.2	250	11	23.8
	Acer rubrum	5.6	100	0.3	6.2
	Fraxinus caroliniana	9.6	25	0.2	1.6
	Living Trees		850	60.3	
	Dead Trees		25	0.2	
Upstream A					
	Taxodium distichum	-	0	0	0
	Nyssa aquatica	33.5	475	48.1	50.4
	Nyssa sylvatica	16.3	625	15.8	32.2
	Acer rubrum	13.8	175	4.1	8.8
	Fraxinus caroliniana	5.7	225	0.7	8
	Living Trees		1500	69.6	
	Dead Trees		25	0.2	

Table 2. cont'd

Plot	Species	DBH (cm)	Stem Density (number ha^{-1})	BA (m^2 ha^{-1})	IV
Upstream B					
	Taxodium distichum	22.3	75	3.7	4.5
	Nyssa aquatica	29.1	400	31.8	32.1
	Nyssa sylvatica	22.5	625	31.4	37.5
	Acer rubrum	9.5	375	3.5	11.9
	Fraxinus caroliniana	6.7	400	1.6	11.3
	Living Trees		1975	72.6	
	Dead Trees		125	0.6	
Outfall A					
	Taxodium distichum	40	125	16.5	22.2
	Nyssa aquatica	27.9	475	34	59.3
	Nyssa sylvatica	22.1	125	4.9	12.1
	Acer rubrum	15.2	75	1.7	6.2
	Fraxinus caroliniana	-	0	0	
	Living Trees		800	57.4	
	Dead Trees		50	0.45	
Outfall B					
	Taxodium distichum	22.7	175	13.3	19.7
	Nyssa aquatica	32.4	400	38.8	51.9
	Nyssa sylvatica	22.1	200	9	17.5
	Acer rubrum	3	125	0.4	6.7
	Fraxinus caroliniana	7.1	75	0.3	4.1
	Living Trees		975	61.9	
	Dead Trees		75	9.8	
Downstream A					
	Taxodium distichum	39.1	100	14.9	14.7
	Nyssa aquatica	25	750	43.9	60.6
	Nyssa sylvatica	19	175	5.6	10.7
	Acer rubrum	10.6	175	3	8.8
	Fraxinus caroliniana	6.3	75	0.2	3
	Living Trees				
	Dead Trees		75	1.2	

Table 2. cont'd

Plot	Species	DBH (cm)	Stem Density (number ha^{-1})	BA (m^2 ha^{-1})	IV
Downstream B					
	Taxodium distichum	30.9	325	27.4	37.6
	Nyssa aquatica	24.8	350	18.6	31.1
	Nyssa sylvatica	16.5	150	3.6	9.6
	Acer rubrum	17.5	225	8.2	16.8
	Fraxinus caroliniana	6.4	100	0.4	4.7
	Living Trees		1150	58.4	
	Dead Trees		250	2	

Table 3. Aboveground net primary production ($g/m^2/yr$) at Bayou Chinchuba and Bayou Castine, St. Tammany Parish, LA USA.

	1999			2000			2002		
	Stem	Leaf	Total	Stem	Leaf	Total	Stem	Leaf	Total
Bayou Castine									
A	297.8	604.9	902.7	163.9	611.1	775.0	191.6	730.0	921.6
B	204.4	548.1	752.5	182.7	627.0	809.7	260.6	773.0	1033.6
Upstream Chinchuba									
A	623.0	446.0	1069.0	466.6	366.1	832.7	494.2	457.9	952.1
B	369.1	592.2	961.3	348.7	501.0	849.7	470.2	624.1	1094.3
Outfall Chinchuba									
A	333.0	577.6	910.6	420.8	538.8	959.6	319.0	647.0	966.0
B	524.6	446.9	971.5	301.6	493.5	795.1	414.4	485.5	899.9
Downstream Chinchuba									
A	724.3	743.8	1468.1	672.1	745.0	1417.1	254.1	827.5	1081.6
B	425.6	670.5	1096.1	370.7	714.2	1084.9	292.4	775.6	1068.0

Accretion

Accretion at the two monitoring sites varied significantly (Table 4). There were significant differences in the plots after 1.5 years ($F = 20.4$, $P<0.0001$) and after 5.5 years ($F = 220.2$, $P< 0.001$). Accretion rates along Bayou Chinchuba from 1999 to 2004 ranged from 9.3 mm yr -1 along the elevated ridge plot location to 11.8 mm yr -1 at the bayou edge plot (Table 4), and this increase was significantly higher than Bayou Castine ($F = 81.5$, $P<0.001$).

The natural levee ridge at Bayou Castine also showed a relatively high accretion rate; however, the remaining accretion plots were near or below 3 mm yr -1 (Table 4).

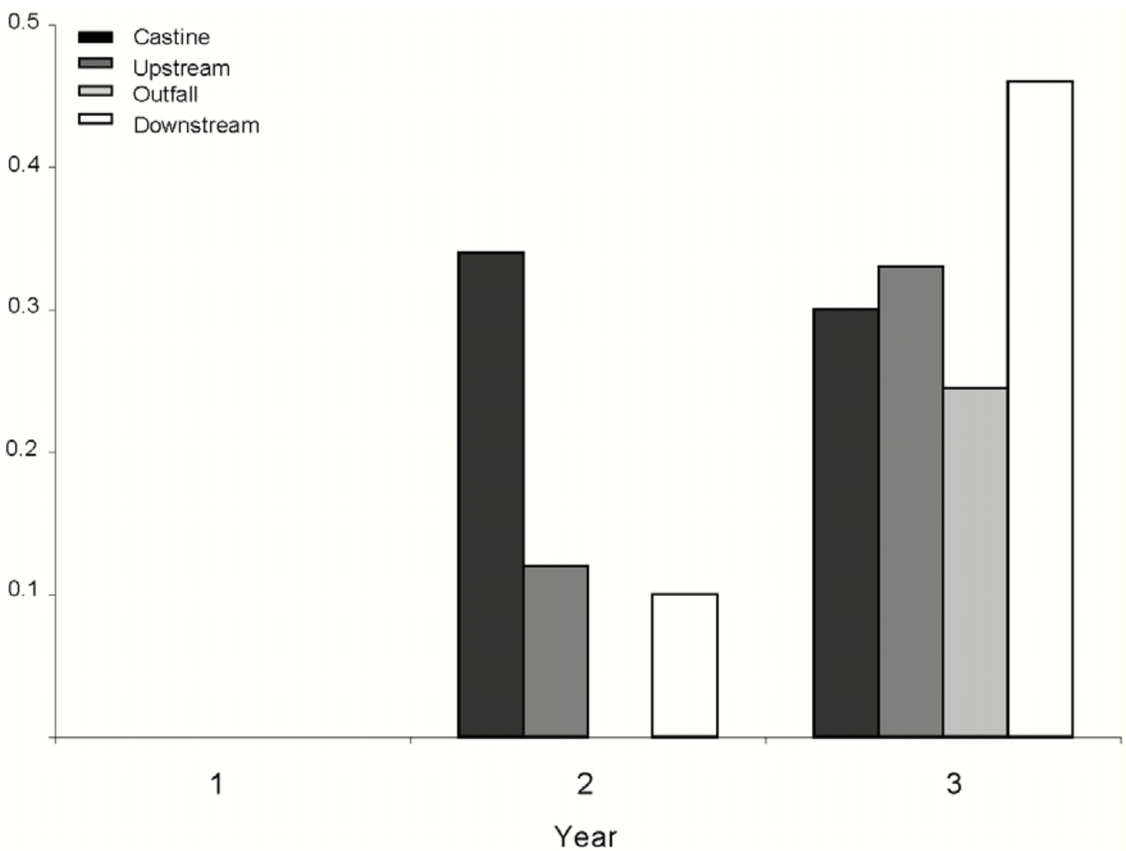

Figure 5. Tree mortality in the study area.

Discussion

During a three year monitoring study at Mandeville, Louisiana, the discharge of secondarily treated effluent into Bayou Chinchuba resulted in nutrient reduction, enhanced wetland forest productivity, and increased accretion downstream of the outfall. The effluent stream appeared to buffer the downstream floodplain from drought effects but was overwhelmed during high bayou discharges, usually associated with precipitation events.

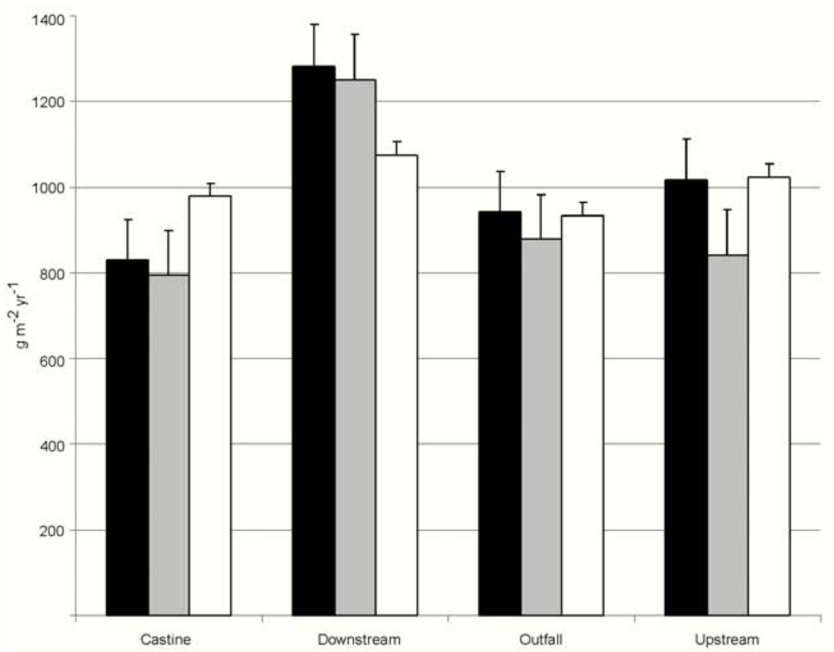

Figure 6. Aboveground net primary productivity at paired forest plots near Mandeville, Louisiana for three annual periods. 1999- black bars, 2000- shaded bars, and 2001- white bars.

Table 4. Total accretion (mm) and accretion rates (mm yr^{-1}) (mean + SE) utilizing feldspar technique for wetland plots near Mandeville, St. Tammany Parish, LA USA.

	Total Accretion (Apr 99-Oct 00)	Accretion Rate (Apr 99-Oct 00)	Total Accretion (Apr 99-Oct 04)	Accretion Rate (Apr 99-Oct 04)
Chinchuba Ridge	8.0±1.7	5.3±1.2	51.3±0.8	9.3±0.1
Chinchuba Mid	9.0±1.5	6.0±1.0	55.5±0.8	10.1±0.2
Chinchuba Bayou	23.0±4.0	15.3±2.7	65.1±1.3	11.8±0.3
Castine Ridge	36.0±2.7	24.0±1.8	43.5±2.0	7.9±0.4
Castine Mid	13.9±1.0	9.3±0.7	14.8±0.7	2.7±0.1
Castine Bayou	7.9±0.7	5.3±0.5	11.6±1.4	2.1±0.3

Hydrology

Hydrology in the study area was dominated for most of the study period by the effluent from the wastewater treatment facility. This constant source of nutrient enhanced freshwater served to buffer the downstream portion of Bayou Chinchuba from a region wide drought (U.S. Geological Survey data). Similar reductions in salinity were noted in St. Bernard Parish where discharge of secondarily treated effluent is maintaining a cypress stand

in a brackish marsh area (Day et al. 1997). Other studies from south Louisiana indicate that freshwater can offset salinity induced stress to coastal plant communities (Day et al. 1997; Martin and Shaffer 2005, Myers et al. 1995, Shaffer et al. 2001, McKee et al. 2004). For coastal wetland forests in Louisiana, fresh water from wastewater effluent and riverine discharges can be important in buffering salinity intrusion, especially in drought years (Day et al. 1997, 2000; Martin and Shaffer 2005; Myers et al. 1995; Shaffer et al. 2001; McKee et al. 2004).

Water Chemistry

There were reductions in nutrient concentrations in Bayou Chinchuba with distance from the plant, indicating nutrient assimilation in the swamp system. This is consistent with findings for other wetlands in Louisiana and elsewhere (Zhang et al. 2000; Blahnik and Day 2000; Day et al. 2004; Mitsch et al. 2001, 2005; Lane et al. 1999, 2001, 2003). The results for nutrient concentrations were similar to values reported for other wetland forests in Louisiana and other areas (Kemp and Day 1984; Zhang et al. 2000; Day et al. 2004; Lane et al. 2003).

I used these results to calculate nutrient loading and uptake for Bayou Chinchuba based on the initial loading amount from the outfall of the wastewater facility. For Bayou Chinchuba, the percent reduction of nitrate-nitrite was approximately 82% between the outfall station and Bayou Chinchuba at Lake Pontchartrain, for most sampling periods. The percent reduction of total nitrogen between these stations was approximately 59% for the study. Total phosphorus reductions ranged from 25% to 93% with an average reduction of 69% over the study. The lowest reduction percentages generally corresponded to times when high rainfall events decreased the residence time of water from Bayou Chinchuba into Lake Pontchartrain.

Nutrient inflow into a wetland is normally expressed as a loading rate, which integrates the nutrient concentration and volume of the inflow and the area of the receiving wetland, usually expressed as g N or P m^{-2} y^{-1}. Most authors have shown that nutrient removal is inversely related to the loading rate (Richardson and Nichols 1985; Kadlec and Knight 1996; Mitsch et al. 2001, 2005; Fisher and Acreman, 2004). At low loading rates nutrient removal efficiency is high. But, as loading rates increase, nutrient removal efficiency decreases rapidly (Richardson and Nichols 1985; Kadlec and Knight 1996). Average total nitrogen loading rate for the 98 ha area below the outfall on Bayou Chinchuba was 20.1 g m^{-2} yr^{-1} and the average loading rate for total phosphorus was 5.4 g m^{-2} yr^{-1} (Table 5). Nutrient loading rate calculations for Bayou Chinchuba indicated excessive loading for these wetlands compared to other treatment systems in Louisiana (Day et al. 2004) and alternate wetland areas were located to decrease these loading rates and ensure high levels of nutrient retention to the wetland areas (Day et al. 2000).

Table 5. Percent nutrient reductions and loading rates for wastewater treatment at a 98 ha area near Mandeville, St. Tammany Parish, Louisiana.

Nutrient	Effluent Concentration (mg L^{-1})	Outlet % Reduction	Loading Rate (g m^{-2} y^{-1})
Total N	7.5	65	20.1
Total P	2	50	5.4

Forest Composition and Productivity

Bayou Chinchuba and Bayou Castine have stem densities and basal areas that are similar to other forested wetlands in the southeastern U.S. (Conner and Day 1982). These forest structure indices are also similar to or higher than at the Amelia wastewater treatment site (Day et. al 1997) and the Breaux Bridge wastewater treatment site (Delgado-Sanchez

1995), and much higher than at the Thibodaux wastewater treatment site (Day et. al. 1993). The forest composition at Bayou Chinchuba and Bayou Castine appears to be correlated with the floodplain width and distance from Lake Pontchartrain (salinity). As noted by Rheinhardt et al. (1998), for other low-order streams in the coastal plain, swamp blackgum usually dominates these forested wetlands in the headwaters, while water tupelo and baldcypress typically dominate mid-reach portions. This is readily apparent in the Bayou Chinchuba plots, as the relative density and relative dominance of swamp blackgum decreases from the upstream to the downstream plots, and both water tupelo and baldcypress dominate the outfall and downstream plots.

The litterfall at the four swamp sites are similar to levels found in other alluvial, flowing water systems in the southeastern U.S. These figures for aboveground production are above averages for similar forest habitats in the southeastern U.S. I noticed differences in the carbon allocation for the amount of stem and leaf growth in the vegetation plots by year which could be affected by the hydrology or the amount of carbon turnover. Clearly, the two downstream plots on Bayou Chinchuba were very productive, likely due to the fertilizing effects of the discharge. This enhanced productivity is similar to other studies involving wetland wastewater treatment, in that, wetland systems can effectuate high nutrient reduction and increase primary productivity (Ewel 1978; Lemlick and Ewel 1984; Conner et al. 1989; Day et al. 1994, 1997, 2004; Rybczyk et al. 1996). Forested wetland systems can provide an additional benefit by storing carbon on a long-term basis.

Tree mortality rates over the first two years were low, however, after the third growing season, rates increased at all areas along Bayou Chinchuba, presumably due to stresses associated with the region wide drought during the study. The mortality rates observed in this

study were similar to rates observed by Conner et al. (2002) in areas where the hydrology of the swamp had not been altered. Although these mortality rate increases were not significant, what is of interest however is that the downstream receiving area, although experiencing greater mortality, also exhibited enhanced productivity, as surviving trees grew into available growing space. This finding can potentially provide another avenue for carbon sequestration, the rapid turnover, burial and compaction of plant material in wetland systems (Comins and McMutrie 1993; Craft et al. 1995; Hamilton et al. 2002; Luo and Reynolds 1999). Increasing plant productivity from elevated nutrient availability and uptake can lead to rapid carbon turnover, but in a wetland system, the release of carbon back into the atmosphere from decomposition is reduced (Comins and McMutrie 1993; Hamilton et al. 2002; Melillo and Aber 1984; Luo and Reynolds 1999).

<u>Accretion</u>

Our study confirmed the findings of Rybczyk et al. (2002) that wastewater effluent can significantly increase accretion rates and thus help offset RSLR. In their study, Rybczyk et al. (2002) presented a conceptual model of coastal sediment accretion dynamics over time where decomposition, dewatering, and compaction of sediments and organic matter reduce the volume and thickness of each yearly cohort. I have modified this conceptual model to include treatment forested wetlands where the yearly cohort is not reduced by decomposition, dewatering and compaction, but instead is subject to high perennial plant productivity (both aboveground and belowground) and nutrient burial over time leading to a linear response as opposed to a curvilinear response in marsh habitats (Figure 7). Rybczck et al. (2002) posed several questions as to how the overall accretion balance would change and how the wetland system would respond to a surface that became subaerial. I agree that decomposition rates

would increase as the soil became oxygenated and rates of mineral deposition would decrease also, as elevation increased (Rybczck et al. 2002). However, if nutrient enhanced water were continuously placed over the surface of a forested wetland, high productivity and high rate of carbon turnover can lead to carbon burial, increasing soil elevations, and forcing water above the soil surface, thereby lowering the rate of decomposition (Rybczyk et al. 2002).

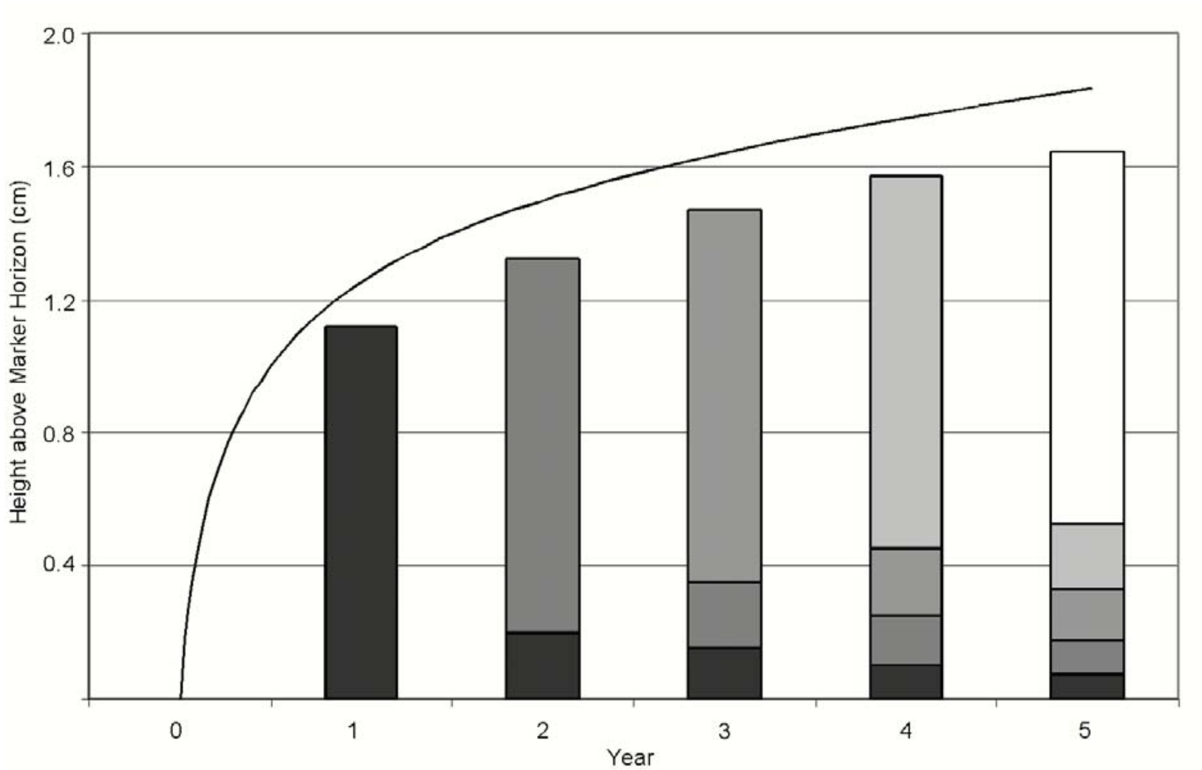

Figure 7. Conceptual marsh accretion model over time (after Rybczyk et al. 2002).

The downstream plots had significantly higher primary production as well as higher tree mortality rates. This served to provide more room for expansion of the remaining trees into available growing areas, and also served to place this stored carbon from stems and limbs into the wetland area for future burial. Leaf litter alone contributed significantly higher rates of accretion over time in these plots, the high accretion rate observed downstream of the outfall may be in part due to the mineral soils present at the study site, as this has been shown

to increase relative elevation rates in coastal areas (Figure 8; Day et al. 1999). RSLR in the Lake Pontchartrain basin has been estimated at 4.5 mm yr^{-1} based on tide gauge analysis (Turner 1991). This area of the Louisiana coastal zone is experiencing lower overall rates of subsidence due to the proximity of Pleistocene sediments near the surface (Penland 1988). RSLR rates are much higher in areas of the former Mississippi River alluvial plain (Turner 1991).

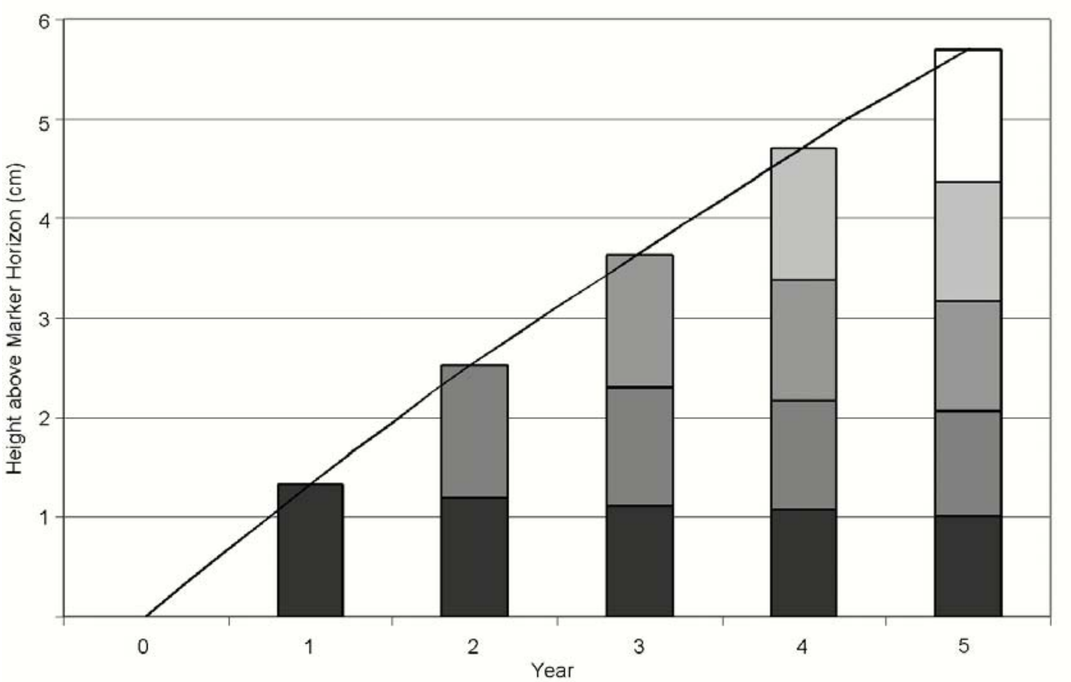

Figure 8. Conceptual swamp accretion model based on data at Mandeville, Louisiana

Summary

In this study conducted on the long-term input of secondarily treated wastewater to a wetland ecosystem near the City of Mandeville, St. Tammany Parish, Louisiana, there were positive responses related to nutrient reduction, increased plant productivity and higher accretion downstream of the discharge location. The effluent into the system was a major source of freshwater to the system during a period of drought and provided elevated levels of

nutrients to the forested wetland. Removal efficiencies of N and P reached 87% and 93%, respectively. On average, the nitrogen loading rate for the 98 ha area below the outfall on Bayou Chinchuba was 20.1 g m^{-2} yr^{-1} and the loading rate for total phosphorus was 5.4 g m^{-2} yr^{-1}. Aboveground net primary production of the freshwater forest system and accretion rates were both high downstream of the effluent discharge. Clearly, the aboveground production on the downstream forest plots at Bayou Chinchuba was significantly higher than the other three reference plots, likely due to the fertilizing effects of the discharge. This enhanced productivity is similar to other studies involving wetland wastewater assimilation. Accretion rates downstream of the outfall were significantly higher than the reference location. Management or re-direction of nutrient-enhanced effluents to wetland ecosystems can maintain plant productivity, sequester carbon, and maintain coastal wetland elevations in response to sea-level rise.

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CHAPTER 3

AN ANALYSIS OF STAKEHOLDER INTERACTIONS IN THE USE OF WETLAND ASSIMILATION: A CASE STUDY FOR THE CITY OF MANDEVILLE, LOUISIANA

Introduction

The use of wetlands for the assimilation of treated municipal effluent has become a viable alternative to conventional treatment from environmental and economic perspectives (Kadlec and Knight 1996; Day et al. 2004; Mitsch and Jorgensen 2004). Scientific issues related to wetland assimilation have received considerable attention. However, the process of planning, permitting, constructing and monitoring a wetland assimilation system is very complicated and involves a wide range of stakeholders. These stakeholder interactions have received relatively little attention. In this paper, I use the City of Mandeville as a case study to illustrate the complexity of stakeholder interactions in developing coastal restoration projects such as these and how this fits into the environmental planning process.

Present day coastal Louisiana was formed over the past 5000 years as sedimentation from the Mississippi River created overlapping deltaic lobes in the northern portion of the Gulf of Mexico (Scruton 1960, Roberts 1997). Historically, floods from the Mississippi River provided sources of freshwater, nutrients, sediments and organic materials into the Mississippi River delta plain (Delaune et al. 1983; Hatton et al. 1983; Roberts 1997, Day et al. 2000a), and massive overbank flooding and river crevasses provided water directly to the wetlands adjacent to the river and its distributaries (Hatton et al. 1983; Kesel 1988, 1989; Roberts 1997; Day et al. 2000a). This introduction of mineral sediments contributed directly to vertical accretion and the associated nutrients stimulated organic soil formation (Nyman and Delaune 1991, Mendelssohn and Morris 2000).

However, due to river channelization and flood protection levees along the lower Mississippi River, riverine input to most of the delta plain has been reduced dramatically, resulting in declining plant productivity and reduced regeneration, low soil accretion rates, prolonged periods of inundation, and coastal wetland breakup and loss (Conner and Day 1982, 1988a, 1988b; Day et al. 2000a). Consequently, the coastal zone of Louisiana is experiencing tremendous change due to natural and anthropogenic processes. Regional effects such as increasing human populations and global impacts like climate change are expected to intensify the problems of degraded water quality, subsidence, and coastal wetland loss in the coastal zones (Day et al., 2000a; 2004).

A recent review paper (Day et al. 2004) on using coastal wetlands for wastewater assimilation found that accrued benefits to the natural and human system include: improved water quality, increased substrate accretion rates, increased plant productivity, and conservation of financial and energy resources. Studies have indicated that wetland systems, both natural and constructed, are effective at removal of nutrients and pollutants by physical, chemical, and biological processes (Godfrey 1985; Hammer 1989; Conner et al. 1989; Kadlec and Alvord 1989; Faulkner and Richardson 1989; Richardson 1999; Khalid 1981a, 1981b; Knight 1987; Patrick 1990; Richardson and Nichols 1985; USEPA 1987; Zhang et al. 2000; Richardson and Davis 1987; Kadlec and Knight 1996).

In addition, wastewater effluent can significantly increase accretion rates and thus help offset the effects of relative sea level rise (RSLR) (Rybczyk et al. 2002, Day et al. 2004). The application of wastewater effluent to coastal wetlands has the capability to increase soil accretion by an increase in organic matter production and integration with soil material (Rybczyk et al. 2002). Subsiding coastal wetland habitats have a nearly limitless capacity and

potential to sequester carbon over time provided the mechanisms are in place to receive nutrient-rich waters and the habitat can effectively assimilate the nutrient load (Chmura et al. 2003, Day et al. 2004, Brevik and Homburg 2004). Subsidence in coastal Louisiana can result in a RSLR nearly ten times greater than eustatic sea level rise and the ability to increase primary productivity is critical to maintain coastal elevations (Conner and Day 1988a; Penland et al. 1988). Increased productivity results in enhanced organic soil formation and belowground plant productivity that in turn, enhance the overall accretion needed to offset subsidence (Day et al 2004). As wetlands have been shown to be effective wastewater assimilation systems, they are also being investigated as financially sound investments and as being energy efficient (Odum 1978; Day et al. 2004).

There has also been an increased interest in wastewater wetlands for other benefits. Nutrient introduction to wetlands can also attract an increase in the number of fish and wildlife species to an area and the presence of water, vegetation, fish and wildlife provide human populations with commercial products (furbearers), recreational possibilities and environmental education opportunities (Hammer, 1989; Knight, 1992; 1997; Worrall et al., 1997).

The issues of water quality, wetland restoration, global climate change, and energy availability are intertwined in the Louisiana coastal zone. Nutrient enrichment combined with excessive nutrient input from the Mississippi River basin has led to a suite of water quality problems and the resulting eutrophication has led to problems such as algal blooms, changes in ecosystem community structure, and low dissolved oxygen (hypoxia or anoxia). A noted example is the large hypoxia zone off the Louisiana coast caused by excessive nutrients in the Mississippi River (Turner and Rabalais, 1991). Mitsch et al. (2001, 2005) proposed that the

restoration or creation of wetlands throughout the Mississippi Basin could be part of the solution to this problem, as the reduction of nutrient inputs to water bodies is a key element in the solution of over-enrichment problems. In coastal Louisiana, a network of navigation canals for water-borne commerce, drainage, petroleum activity and flood-protection levees have left many wetlands hydrologically altered (Day et al. 2000a). This short-circuiting has led indirectly to loss in normal wetland functions, such as water quality improvement (Day et al. 2004). Many of these hydrologically-altered wetlands are also isolated from nutrient sources either from the Mississippi River or from wastewater effluent (Day et al. 2004). These isolated natural wetlands provide a practical economic solution for many small communities that are widely dispersed in the coastal zone (Breaux and Day 1994).

Anthropogenic activities have also led to changes in global climate change and evidence is mounting towards changes in water availability and flow, sea level rise and coastal storms, and effects on soil moisture and drought (Niklaus and Korner, 2004). Elevated carbon levels in the atmosphere have effects not only in the abiotic environment, but also has been found to affect terrestrial systems interacts with nutrient cycling, especially nitrogen (Niklaus and Korner, 2004). The major predicted climate change forcings that will affect the coastal zone are accelerated sea level rise, increased temperature, changes in freshwater runoff, and changes in the frequency and intensity of tropical storms (e.g., see Poff et al. 2001; Day et al. 2005). Increased temperature and reduced fresh water runoff may lead to higher salinities in coastal areas.

Templet and Meyer-Arendt (1988) proposed a regional water management approach in Louisiana utilizing river diversions to combat the long and short-term sedimentation events that were eliminated due to flood-control projects and the management of the Mississippi

River for navigation. Breaux and Day (1994) proposed a strategy utilizing treated wastewater effluents to simultaneously counter two major environmental problems in the Louisiana coastal zone: high levels of surface water pollution and the high rate of wetland loss. Use of wetlands to assimilate effluent improves water quality and stimulates wetland accretion (e.g., Rybczyk et al. 1996; Rybczyk 1997). Other studies have shown that secondarily treated wastewater can both stimulate productivity and promote vertical accretion in wetlands through increased organic matter production and deposition (Odum, 1975; Turner et al., 1976; Mudroch and Copobianco, 1979; Bayley et al., 1985; Knight, 1992b; Craft and Richardson, 1993, 1995; Rybczyk, 1997; Hesse et al., 1998; Rybczyk et al., 2001). Nutrient uptake has also been demonstrated in freshwater diversion projects off the lower Mississippi River (Lane et al., 1999; 2001; Mitsch et al., 2001).

Recent efforts to restore and enhance wetlands in the subsiding delta region of Louisiana have focused on attempts to decrease vertical accretion deficits by either physically adding sediments to wetlands or by installing sediment trapping mechanisms (i.e. sediment fences), thus increasing elevation and relieving the physio-chemical flooding stress (Boesch et al., 1994). While these efforts can provide noticeable results, a policy that would provide high-nutrient level water into the coastal wetland system of Louisiana has the potential to provide greater results over time (Templet and Meyer-Arendt, 1988; Breaux and Day, 1994; Mitsch et al., 2001).

Objectives

The objective of this study is to describe stakeholder interactions that took place in the process of planning and implementation of a wetland assimilation system at Mandeville, St. Tammany Parish, Louisiana.   The idea was to ascertain if development of science-driven

solutions to natural resource management were used as a process in creating successful public-private partnerships. In addition, a determination wanted to be made on the use of these solutions as well as generation of the successful partnerships would assist decision-makers in developing cost effective, coastal wetland preservation projects.

Environmental Setting

Faced with barriers to physical growth in the City of New Orleans (Mississippi River, Lake Pontchartrain, wetland soils), population growth in the metropolitan area has expanded to sites with lower relative population densities, and available open land on upland soils (US Census data 2000). The population of the City of Mandeville, north of Lake Pontchartrain, in southern St. Tammany Parish has increased from 8,645 to 10,489 over the past decade (US Census data 2000). Many in Mandeville commute to New Orleans over the Lake Pontchartrain Causeway, completed in 1958 (see Figure 1). Between 1964 and 1989, Mandeville operated two separate oxidation lagoons, one discharging into a marsh area to the west of Bayou Chinchuba, and the other discharging directly into Bayou Chinchuba upstream of its current discharge location. Since 1989, upon construction of the current facility, the City of Mandeville has discharged all of its treated wastewater into the Bayou Chinchuba wetland and discontinued the use of the oxidation lagoons. The Mandeville Wastewater Facility currently consists of three (61 x 183 m) aerated lagoon cells, a three-celled microbial rock reed filter system, and an ultraviolet disinfection system. The facility was built under the advice of consultants to the Environmental Protection Agency (EPA) and operated under discharge permits issued by Louisiana Department of Environmental Quality (LDEQ) and EPA (Reed 1991). Over time, the rock reed system began discharging excessive amounts of ammonium-nitrogen into Bayou Chinchuba, presumably due to organic matter reducing the

available surface area in the microbial rock-reed filter and allowing the wastewater stream to flow directly on the wetland surface.

Bayou Chinchuba is mostly a broadleaf and needle-leafed deciduous forested wetland dominated by water tupelo (*Nyssa aquatica*), swamp blackgum (*Nyssa biflora*) and baldcypress (*Taxodium distichum*) (Day et al. 2000b). The Bayou Chinchuba swamp has hydrologic inputs from precipitation, drainage from the surrounding uplands, and occasional backwater flooding from Lake Pontchartrain. This area also receives approximately 7191.5 m^3 day^{-1} (0.083 m^3 s^{-1}) from the wastewater facility (see chapter 2).

Extensive wetlands exist in proximity to Bayou Chinchuba and the Mandeville Wastewater Facility. In addition to the lower Bayou Chinchuba drainage basin, a large brackish water marsh hydrologically isolated from riverine input is situated approximately 1 km west of the treatment facility east of the Tchefuncte River (Day et al. 2000b). This marsh has undergone change in vegetation over the past 40 years, presumably from indirect anthropogenic impacts. The City of Mandeville operated under a traditional water quality permit, whereby the criteria limits for wastewater discharge were: 30 day average BOD of 10 mg l^{-1} with daily maximum of 15 mg l^{-1}; 30 day average TSS of 15 mg l^{-1} with daily maximum of 23 mg l^{-1}; 30 day average NH_4 of 5 mg l^{-1} with daily maximum of 10 mg l^{-1}; 30 day average fecal coliforms of 200 colonies/ 100 ml with daily maximum of 400 colonies/ 100 ml (US EPA 1998).

The Regulatory and Policy Environment

There are several important regulatory and policy issues with regards to wetland wastewater assimilation (Day et al. 2004). Probably, one of the most important issues is to be in compliance with the Federal Clean Water Act statutes and the Louisiana Water Pollution

Control Regulations (Day et al. 2004). These laws were passed to protect public water resources and require certain permitted criteria to protect beneficial uses (Day et al. 2004). The use of total maximum daily loads (TMDLs) and nutrient limits on stream and water bodies are other issues that must be considered (Day et al. 2004). TMDL is the calculation of the maximum amount of a pollutant that a water body can receive and still be within water quality standards for that pollutant (Day et al. 2004). The national strategy that has been employed to implement this policy has been to approach it from a region or specific water body as opposed to national criteria, use regional nutrient criteria and target ranges and continue to monitor and evaluate state management programs (EPA 2004).

In the future, there may be problems that arises for small communities in watersheds dominated by other pollutant sources (such as agriculture or a large city), whereby TMDL calculations will force very low discharge limits on effluent from an entire watershed even if a small community contributes only a small fraction of the total loading (Day et al. 2004). However, TMDL calculations can also be used beneficially to determine seasonal or annual loadings into a water body and determine appropriate reduction goals.

There are substantial economic and energy savings for small communities and non-toxic industrial processors (Day et al. 1992; Ko et al. 2004). 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.

Planning System and Public Participation

Beginning in 1991, effluent testing at Mandeville revealed exceedances in ammonium-nitrogen concentrations on a recurring basis over a period of nine years (see Table 6; US EPA Enforcement Case Report, Day et al. 2000b). Due to the repeated nature of these criteria violations, the EPA fined the City of Mandeville $56,500 and promulgated an administrative order requiring the City to take actions to come into compliance (US EPA Enforcement Case Report). The initial proposed solution was a $6.5 million conventional treatment consisting of aerated lagoons and an activated sludge plant. Because Mandeville faced a probable future of twice the number of users into their system and the likelihood of further expensive upgrades to the system, the City turned to regulatory and academic sources for assistance. A citizens-based public advocacy group, the Lake Pontchartrain Basin Foundation (LPBF), approached the City of Mandeville in 1998 and inquired about alternatives to solving the City of Mandeville's wastewater treatment needs.

Table 6. Timeline of wastewater discharge to Bayou Chinchuba, Mandeville, La (1964-present)

1964	City of Mandeville constructs an open air, 20 ha oxidation lagoon adjacent to Bayou Chinchuba to process wastewater from city residents
1988	City expands facility and constructs, with assistance from US EPA, present day wastewater facility, which currently consists of three (61 x 183 m) aerated lagoon cells, a three-celled microbial rock reed filter system, and an ultraviolet disinfection system (Reed 1991).
1991-2000	Wastewater facility at Mandeville exceeds permit requirements for ammonium-nitrogen.
June 1997	City receives notice of non-compliance from EPA and LDEQ
March 1998	Preliminary Feasibility Analysis completed and submitted to City of Mandeville and regulatory authorities recommending development of UAA to process wastewater into adjacent wetlands.
April 1998	Citizens Advisory Group for the Mandeville Wastewater Assimilation project meets for first time. Wetland landowners express support for project.
Sept. 1998	Preparation of UAA begins

Table 6. cont'd

Oct. 1999	City of Mandeville requests construction cost-sharing with US Army Corps of Engineers under ecosystem restoration authority to provide wastewater to nearby degraded marsh.
Feb.2000	UAA submitted to City of Mandeville with recommendations to use the Bayou Chinchuba floodplain and an adjacent brackish-water marsh east of the Tchefuncte River for wastewater assimilation.
April 2000	City of Mandeville begins negotiations with landowners to purchase land for wastewater assimilation.
Dec.2000	US EPA complaint against Mandeville filed with court and final order lodged
Jan.2000	Civil expropriation case filed by Mandeville to acquire wetlands for wastewater assimilation project
June 2001	Case number 06-1999-0142 concluded with US EPA with payment of fines totaling $56,500 for numerous and repeated violations of permit requirements
July 2001	expropriation of wetland selected for wastewater assimilation receives court approval.
Oct. 2003	Federal cost-sharing project becomes delayed due to revenue allocation to Homeland Security and Defense.
Feb. 2005	Mandeville City Council approves resolution to use city revenues for assimilation project
June 2005	City of Mandeville accepts bid for Phase I (of II) for assimilation project construction
Sept. 2005	post construction monitoring for assimilation project to begin

This initiated the formation of a public-private partnership to address this regional problem. A citizen's advisory group was formed to provide public input to the process, of which I became a member. The city on advice from the public-private group, in turn, asked Louisiana State University (LSU) to explore environmental alternatives to a mechanical treatment facility and begin baseline data collection that would be used as part of an Use Attainability Analysis, required by LDEQ and EPA for a change in the permit discharge limits. Scientists with LSU employed an environmental planning process modeled after Templet (1986) (Figure 9). The wastewater treatment problem at Mandeville was identified, alternative solutions were developed and evaluated, (including an expensive conventional

treatment system with necessary upgrades) a preferred alternative was selected (wetland assimilation), and upon reaching consensus with various stakeholder groups, a process of implementation was begun.

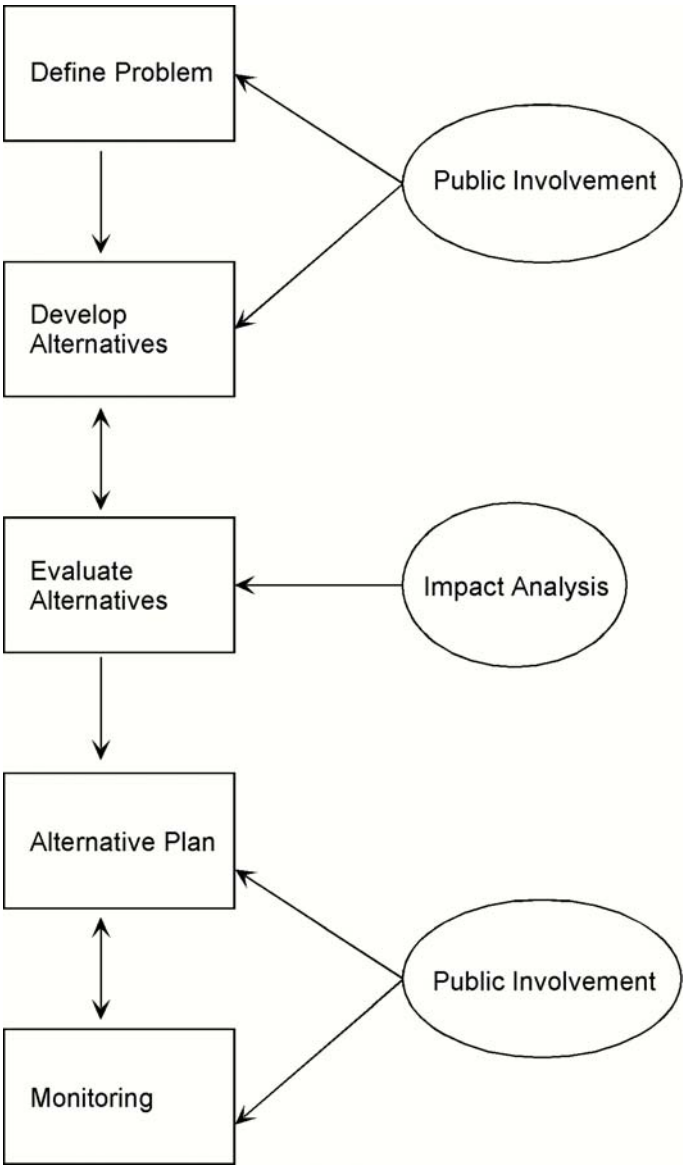

Figure 9. Environmental Planning Process.

Public involvement took place during the problem identification, alternative development, and preferred alternative phases. A Citizens Advisory Committee was established to assist in the planning process. Citizens, including landowners, as well as

representatives of public advocacy groups, Federal resource agencies and state regulatory authorities, as well as members of the City of Mandeville administration participated in planning sessions. The majority of the participants in the sessions were in agreement with the proposed scientific solutions to the problem.

Bardes and Oldendick (2000) found that in general, Americans are supportive of environmental protection and restoration efforts. Their findings indicate that the American public believes not enough funding is being directed to environmental actions; and although a majority would pay higher prices for environmental protection, higher taxes and decreases in a standard of living to protect the environment was not favored by a majority polled (Bardes and Oldendick, 2000). During this project all parties, including some affected wetland landowners, were in general agreement that a course favoring environmental protection and restoration was needed.

Adoption of Planning Documents

In the State of Louisiana, the LDEQ, in consultation with EPA, regulates wetland wastewater assimilation and the discharge of treated effluent. Over the past 15 years, scientists, regulatory personnel and dischargers have developed an approach to ensure wetland treatment systems meet water quality goals (Day et al. 1999; Day et al. 2004) and several projects have been initiated (Delgado 1995; Day et al. 1993, 1995, 1997, 1999, 2000b, 2004; Conner et al. 1989). In most cases, a preliminary feasibility analysis (PFA), generally lasting 2-4 months, is carried out to determine if a particular discharger is a promising candidate for wetland assimilation. The PFA generally includes an analysis of such factors as wetland size and characteristics, preliminary loading rate calculations, distance from the existing plant to the wetlands, landowner issues, and discussions with regulatory personnel. If

the preliminary analysis deems wastewater assimilation is feasible, a year-long Use Attainability Analysis (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 (Day et al. 2004).

The wetland assimilation system is designed so that loading rates are low to ensure a high nutrient retention in the wetland. The UAA analysis found that Mandeville was a likely candidate and recommended the City proceed in implementing a wetland assimilation system (Day et al. 2000b). Upon completion of the UAA (Day et al. 2000b), the City of Mandeville approved the document and proceeded with implementation of the project as well as a settlement with the EPA. Due to excessive nutrient loading into the Bayou Chinchuba floodplain, and then onto Lake Pontchartrain, it was recommended that alternate wetland sites be used for further assimilation (Day et al. 2000b).

The City also expressed interest with the US Army Corps of Engineers to cost-share the ecosystem restoration project under their Section 206 program. The Section 206 program under the Water Resources Development Act of 1990, allows the cost-shared construction of ecosystem restoration projects to take place, whereby the local or non-federal sponsor of the project provides 25% of the initial cost of the project as well as take over operation and maintenance for a 50 year period. The U.S. Army Corps of Engineers prepared planning and environmental compliance documents to facilitate a cost-sharing agreement with the City. The City contracted with a private consultant to submit the revised LPDES permit. Acquisition of the wetlands by the City of Mandeville was included as part of the initial cost-share requirement of the City. The revised permit reiterated the recommendations of the

UAA, a 60% discharge into a nearby brackish-water marsh and 40% discharge to Bayou Chinchuba with biologically-based criteria and less reliance on chemical-based water sampling for compliance. Nutrient limits on the discharge are not needed because the loading rate into the wetland is low enough at both wetland sites to ensure nutrient assimilation. There continues to be a requirement to monitor for toxic materials as well as disinfection for pathogens.

Implementation

Part of the recommendations contained within the UAA for the City of Mandeville project was to divert 60% of their current discharge to a privately owned brackish-water marsh approximately 5 km west of Bayou Chinchuba east of the Tchefuncte River (Day et al. 2000b). The City of Mandeville utilized an appraiser to determine fair market value of the property and promptly negotiated with the landowners based on this value. The landowners of this marsh acreage pushed for future value of the property and asked for a value greater than 5 times the appraised value. As part of the future value estimate of the property, the landowners had partnered with a residential developer to create a water-based housing development. The City in turn filed a civil procedure to expropriate the land needed for public use and to prohibit conflicting future use of this marsh. The case was settled by both parties in July 2001 (Table 6).

This court case signifies the need of all sides of this issue to reach common ground. States and local governments are increasingly turning to private landowners to achieve public use goals. In this case, the private landowner was willing to accept other uses for the property, for the highest economic gain. Researchers are continuing to investigate ways to bring science into the public policy arena and this case is noteworthy because the local

government used the scientific data collected as the justification to expropriate the wetland property.

Construction of the initial phase of the wetland assimilation project began in September 2005. The City of Mandeville expects to complete construction in early 2006. Implementation of a revised water flow plan and environmental monitoring will begin upon completion of construction.

Conclusion

The Mandeville case study illustrates the kinds of challenges most coastal communities face as they attempt to move forward with long-term sustainability. Command and control type environmental policies have accomplished much, as regulation of point source pollution and improved water quality indicate. Further progress requires adjustment in policy strategies to enhance society's capacity to deal with the more intractable problems of non-point source pollution and comprehensive, sustainable solutions to environmental problems. High cost and economic impacts on communities propel the search for cost effective means and for ways to encourage cooperation between the public and private sectors. Such cooperation can build trust and consensus. In this case, the use of a science-driven solution based on an ecologically engineered approach was successful in developing cost savings and coastal wetland preservation from a renewable technology. In addition, public-private partnerships were formed, and they helped to establish effective working relationships among business officials, state and local policy makers, scientists, environmentalists, and community leaders. The legal case illustrates the kinds of unanticipated problems that can arise and shows how these problems can be overcome. The analysis showed that stakeholder interactions are often as complicated and difficult to solve as

scientific issues. The planning model used as part of this study allowed a logical procession or guide during alternative plan development, and the interaction of the public at important steps in the decision-making process. Regional resource management is feasible and in some instances essential for managing coastal resources in both a cost effective and environmentally sound manner.

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CHAPTER 4

CAN EFFECTIVE INTEGRATION OF NATIONAL CARBON POLICY AND NATIONAL WETLAND POLICY TAKE PLACE?

Introduction

A number of criteria are relevant for consideration of environmental policy instruments, including cost-effectiveness, distributional equity, and political feasibility (Bohm and Russell, 1985; Revesz and Stavins, 2004). When compared to traditional command-and-control approaches for air and water pollution, economic-based incentives can 1) provide the least-cost method of control, 2) lowered overall pollution to or below acceptable standards, 3) generate revenue, and 4) promote innovation of new technologies (Smith, 2004). In this paper, I use examples from Louisiana to show how wetland enhancement, through the use of nutrient enhanced effluents and carbon sequestration can be achieved in a complimentary manner.

Breaux and Day (1994) proposed a market-based strategy to counter two major environmental problems in the Louisiana coastal zone: high levels of surface water pollution and the high rate of wetland loss. Their work, and other studies, show that secondarily treated wastewater stimulates productivity and enhances vertical accretion in wetlands through increased organic matter production and deposition (Odum, 1975; Turner et al., 1976; Mudroch and Copobianco, 1979; Bayley et al., 1985; Conner et al. 1989; Day et al. 1992; Knight, 1992; Craft and Richardson, 1993; Rybczyk et al. 1996; Rybczyk, 1997; Hesse et al., 1998; Day et al. 1999; Rybczyk et al., 2002). Nutrient uptake has also been demonstrated in government-sponsored freshwater diversion projects off the lower Mississippi River (Lane et al., 1999; 2001; Mitsch et al., 2001, 2005, Day et al. 2005).

The coastal zone of Louisiana is experiencing tremendous change due to natural and anthropogenic processes. The Mississippi delta, a 2.2 million ha area of tidal marshes, coastal swamps, and shallow open water areas, has been in a state of decline since construction of Mississippi River flood control levees was completed in the early 1900s (Mossa 1996; Day et al., 2005). Pervasive alteration of the hydrology has also contributed to the deterioration of the delta (Day et al., 2000b, 2005). Past inputs from the Mississippi River introduced freshwater, sediments, and nutrients into the wetlands of the delta plain. This input provided an allochthonous source of mineral sediments, contributing directly to vertical accretion, and the nutrients associated with these sediments promoted vertical accretion through increased autochthonous organic matter production and deposition, and the formation of soil through increased plant growth (Delaune et al., 1983). Freshwater input reduces salinity and sulfide stresses (Delaune et al. 2003; Delaune and Pezeshki 2003). The Mississippi River deltaic plain has a high rate of geologic subsidence due to compaction, consolidation, and dewatering of sediments (Roberts 1997). Coastal wetlands can persist in the face of relative sea level rise (RSLR, the combination of eustatic sea level rise and subsidence) when vertical accretion equals or exceeds the rate of subsidence (Delaune et al., 1983; Baumann et al., 1984; Stevenson et al., 1986; Brevik and Homburg, 2004). As long as wetlands are intact, a high rate of RSLR due to geologic subsidence gives rise to a high burial rate as a permanent loss pathway for nutrients (Rybczyk et al., 2002; Day et al., 2004). However, the isolation of the river from the delta plain due to levee construction (Mossa, 1996) has created vertical accretion deficits (RSLR > accretion) throughout the coastal region (Hatton et al., 1983). Impacts due to increasing human population and global climate change will likely exacerbate

problems of subsidence and wetland loss in the Louisiana coastal zones (Boesch et al. 1994; Day et al., 2000a,b; 2004, 2005).

Global Climate Change and Wetlands

The Intergovernmental Panel on Climate Change (IPCC 1996) and Schlamadinger and Marland (2000) estimate that human activities (primarily the burning of fossil fuels) currently add about 7.9 Gigatons of Carbon (C) (Gt = 1 billion metric tons) annually to the current level of 775 Gt C in the atmosphere. The IPCC (2000) has estimated that 600 Gt C is in living plant tissue (mainly trees) and 1600 Gt C is in soil organic matter such as litter, humus and peat deposits. Currently, terrestrial systems are estimated to absorb 2.3 Gt C per year (25% of human emissions) and 1.6 Gt C are released annually to the atmosphere due to deforestation (Malcolm and Pitelka, 2000).

Wetland loss can exacerbate the effects of global climate change directly through the loss of vegetation and indirectly by decomposition of peat deposits that result in a net release of carbon dioxide, CO_2, to the atmosphere (IPCC 2000). Even if plant productivity increases due to global warming and increasing CO_2, wetland systems can result in a net release of CO2 to the atmosphere if humus and peat deposits decrease due to decomposition (Malcolm and Pitelka, 2000). Other research suggests that physical disturbances to ecosystems such as discing or plowing of agricultural fields, or a seasonal wetting and drying cycle in wetlands could release “trapped” microbial exoenzymes with the result being a burst in decomposition activity (Melillo and Aber, 1984). Therefore, carbon-nutrient interactions have important implications, not only for nutrient availability and decomposition (Melillo and Aber, 1984), but also for regional and global climate changes as well as the possible long-term sequestration of carbon.

During the first year of the Clinton Administration, a tax on energy was proposed based on the heat content of a particular fuel (Global Policy 2005). The base rate was proposed at 25.7 cents/million British thermal units (BTU), and in addition 34.2 cents/million BTUs were to be levied on refined fuel products. This is equivalent to about 25 cents/gallon of refined gasoline. The tax sought to decrease the reliance on carbon products by making alternative fuel sources more attractive to consumers as well as contributing tax dollars to reduce the burgeoning Federal budget deficit (Global Policy 2005). The United States Congress eventually passed the Transportation Fuel Tax Bill that President Clinton signed into law on October 1993 that raised gasoline prices an average of 13.814 cents/gallon and deposited the revenues into the U.S. General Fund (Global Policy 2005).

In 1998, under the Kyoto Protocol, most industrialized countries agreed to carbon emission reductions to offset increasing concerns with global climate change. The commitments ranged from reduction in actual emissions to financing forestry projects that sequester carbon. The United States government, concerned about domestic economic productivity and its relation to other growing world economies, not only refused to ratify the agreement, but pursued a domestic agenda of increasing absolute emissions by 14% above 2000 levels (Claussen, 2004).

Therefore, the question addressed in this paper is what type of financial or economic policy mechanism can be used in this country to slow the rate of carbon increases to the atmosphere and slow the rate of wetland loss in the United States? In order to investigate this policy, I decided to investigate the problem from a market-based outlook, which relied upon providing all effected parties with generally positive benefits in return.

A Market Based Proposal for Mitigating Climate Change

The current model of carbon-nutrient interaction as it relates to Louisiana coastal wetlands is shown in Figure 10. Fossil fuels are removed from coastal areas, sold to consumers, and the resulting financial structure of the model is geared back to developing more fossil fuel reserves or improving the infrastructure for additional fossil fuel consumption. These positive feedback loops, if left to continue, would lead to divergent behavior and probably to destruction of the current system. The negative externalities in this model are an increase in greenhouse gases, which lead to global climate change and sea level rise and thus, indirectly to a loss of coastal wetlands and coastal productivity. In the case of Louisiana, fossil fuel production in the coastal zone has also led directly and indirectly to coastal wetland loss (Day et al., 2000b; Morton et al., 2002).

As an alternative to the current model, I propose a competing model (Figure 11), where externalities are internalized because of economic incentives. The positive feedback loops in the current model have been neutralized by negative loops in an effort to maintain the system over time. This new model places an additional Federal carbon tax of $0.013 on every liter ($0.05/gallon) of US motor gasoline consumed. This figure was chosen in part by a comparison with other carbon taxes in developed nations and because it was less than the amount proposed in past carbon tax proposals put forth before the U.S. Congress or the Bush Administration (Burtraw and Portney, 2004; Global Policy 2005). Last year, during the U.S. presidential elections, a proposal was forwarded to both of the major candidates that would impose a $0.06/gallon gasoline tax that would generate an estimated 8 billion in the year 2006 (Burtraw and Portney 2004). Of course, the optimal carbon tax level would be the most

efficient if it was set at a level that would reflect the marginal cost of abatement as well as the marginal damage costs of pollution (Global Policy 2005).

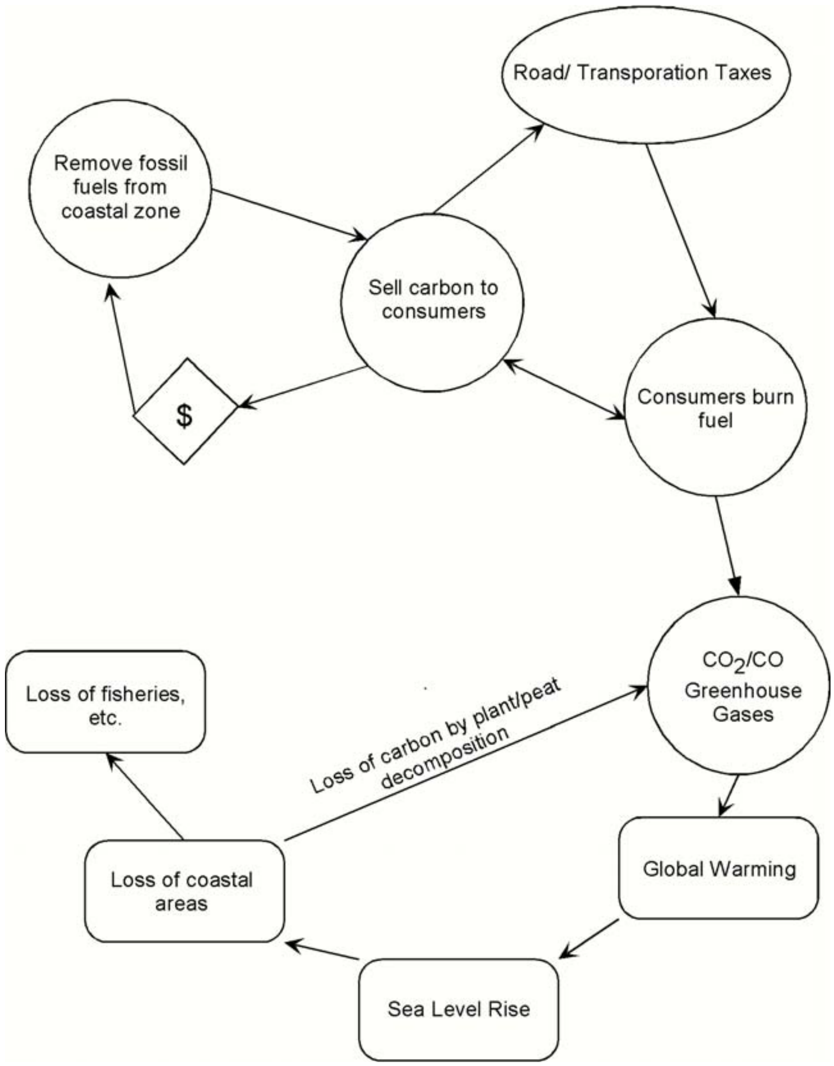

Figure 10. Current model of carbon flow in the Louisiana coastal zone

Based on 2003 domestic energy consumption levels, this would generate approximately $6.8 billion annually. The carbon tax would raise Federal taxes on gasoline to $0.062/liter ($0.234/gallon), an increase of 27% of current tax levels, but only a 2.2% increase in prices paid by consumers at the pump. In addition, instead of placing the revenues from this tax into the General Fund, I propose a national wetland subsidy to public and private

landowners who encourage carbon burial and sequestration on their wetland property by re-directing (allowing) secondarily treated wastewater, non-point source runoff, or nutrient-enhanced river water to flow through the wetland. The proceeds from the Federal carbon tax would be used to direct and fund this subsidy program. On average, if $6.8 billion could be generated through the use of an annual carbon tax, approximately $1940/ha ($790/ac) could be distributed and paid annually to qualified wetland landowners. Higher subsidies would be allocated to landowners of freshwater forested wetlands that receive nutrient enhanced water inputs, and can store C at a higher level than non-forested lands. Smaller subsidies would be paid to other wetlands that receive water inputs, but are not as efficient at storing C (Rybczyk et al. 2002, Delaune et al. 2003, Day et al. 2005).

Day et al. (2005) presented several approaches to restoring wetlands in the Mississippi delta region and argued that restoration efforts would need to be more intensive or focus on ecological engineering techniques to offset the impacts of global climate change and the decreasing amount and increasing cost of energy. Mitsch et al. (2005) recently estimated that 2.2 million ha of wetlands in the Mississippi River basin would be required to remove about 40% of the total nitrogen discharging to the Gulf of Mexico without impact to agricultural production in the basin or a transfer of pollution to another area.

Accretion in coastal areas can be significantly increased by the addition of nutrient-rich water (see Figures 7 and 8). Rybczyk et al. (2002) and Brantley (2005) (Chapter 2) showed that wastewater effluent can contribute significantly to an accretion surplus in the face of RSLR. In their study, Rybczyk et al. (2002) described a conceptual model of coastal sediment accretion dynamics in a marsh over time where decomposition, dewatering, and compaction of sediments and organic matter reduce the volume and thickness of each yearly

cohort (see Figure 7). Brantley (2005) (Chapter 2) modified this conceptual model to include treatment forested wetlands in environments with more mineral soils where the yearly cohort is reduced significantly less by decomposition, dewatering and compaction, but instead is subject to high perennial plant productivity and nutrient burial over time leading to a more linear response as opposed to a curvilinear response in marsh habitats (Figure 8).

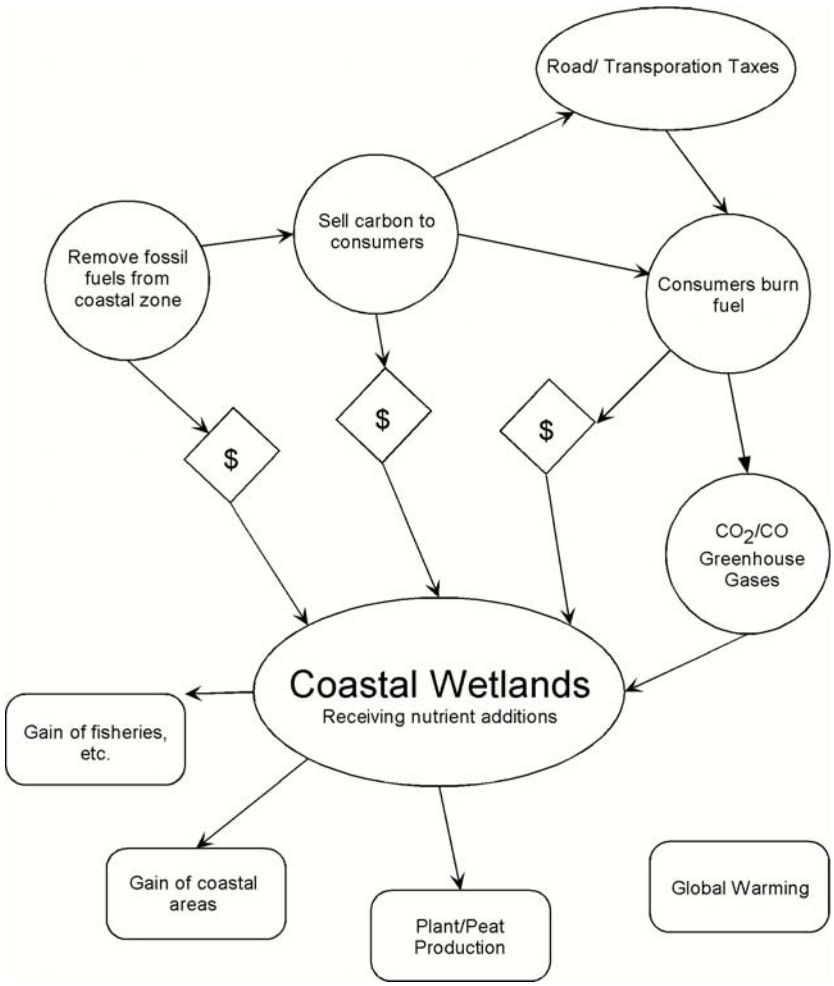

Figure 11. Alternative model of carbon flow under a revised carbon tax policy.

Rybczyk et al. (2002) posed several questions as to how the overall accretion balance would change and how the wetland system would respond to a surface that became subaerial. They suggested that decomposition rates would increase as the soil became oxygenated and

rates of mineral deposition decrease, as elevation increased (Rybczyk et al., 2002). However, they also suggested that if nutrient enhanced water was continuously flowed over the surface of a forested wetland, high productivity and a high rate of carbon turnover would lead to carbon burial, increasing soil elevations, and forcing water above the soil surface, thereby lowering the rate of decomposition (Rybczyk et al., 2002). In addition, since the soil is continuously submerging in coastal Louisiana, there is a continuing high rate of burial.

Predictions for the future are that tree growth and net primary productivity (NPP) will be initially stimulated as CO2 levels become elevated, but will eventually decline over time as tree growth exceeds the rate of N mineralization, therefore resulting in increasing N limitation (Comins and McMurtrie, 1993; Luo and Reynolds, 1999; Hamilton et al., 2002). However, increasing temperatures may increase soil respiration, releasing carbon. Increasing N inputs to forested wetlands that can efficiently sequester C at a relatively fast rate is prudent nutrient resource management.

The economic aims of this strategy are primarily to develop alternative energy strategies and increase energy efficiency through the use of individual entrepreneurship and free markets, and lower the dependence on fossil fuels. A tax on the demand side of the problem would likely push more consumers to seek out alternatives that lessened the amount of income dedicated to energy consumption, as opposed to supply side taxation that would do little to stem consumer demand (Smith 2004). There may also be some indirect benefit to public agencies as the free market approach would lessen some of the energy burden these agencies face. A carbon tax should also lead to lower carbon emissions due to lower consumer demand and the development of efficient alternative fuel sources. In addition, a wetland subsidy would promote additional restoration and creation efforts on the part of state

and local governments, and private landowners, who wish to capture a portion of the subsidy and gain economically from increased productivity on their lands (Hey et al. 2005). This would lead to increased carbon sequestration.

Consumers should expect lower fuel prices over time as research and development into alternative energy sources become available and as the demand for fossil fuel decreases. Fossil fuel companies would see no tax or potential fines for their emissions and could initially redirect funding to research and development of alternative fuel sources as well as promoting wetland preservation and restoration. There must be some consideration given to lower income earners as this proposed tax is regressive, but can be neutralized by indexing the payments to inflation or increase the level of personal deductions households may claim on their Federal income taxes. On a global basis, the world human population can expect a stabilization of greenhouse gas emissions through more focus on the overall energy demand side. In addition, wetland landowners in the United States can expect property enhancement incentives, and domestic taxpayers would see no overall increase from an environmental policy tax on carbon consumers.

Summary

In summary, wetland areas have tremendous capabilities for carbon burial and storage. The addition of nutrient-rich water serves to increase overall production, thereby increasing the wetland capability for carbon processing. Energy policy in the United States has been lacking for several decades, but when acted upon has focused entirely on supply side with little emphasis on stemming consumer and business demand for fossil fuel consumption.

A Federal carbon tax of $0.013 on every liter ($0.05/gallon) of US motor gasoline consumed would generate approximately $6.8 billion annually. In addition to stemming the

growing consumer demand for fossil fuel products in this country, and spur reinvestment into alternative fuel sources, the proceeds from the tax would be used as a subsidy to wetland landowners, thereby creating another financial incentive to manage and restore these ecosystems (Hey et al. 2005).

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CHAPTER 5

SUMMARY AND CONCLUSIONS

This study investigates the response of a coastal, freshwater wetland ecosystem to the long-term input of secondarily treated municipal effluent from the City of Mandeville, St. Tammany Parish, Louisiana. In addition, other issues with regard to wetland wastewater assimilation, such as stakeholder interactions, and the potential for carbon sequestration were addressed. Measurements of hydrology, nutrients, and aboveground net primary productivity were made from September 1998 through March 2002. Substrate accretion measurements were made in October 2000 and October 2004. Beginning in 1998 and continuing through 2005, I also evaluated the various steps that were taken by a public-private partnership to achieve project implementation.

The major hydrologic inputs to the system are the effluent, precipitation, and back water flooding from Lake Pontchartrain. Nutrient levels were generally low except in the immediate vicinity of the outfall, where total N averaged 6.5 mg-N l^{-1} and total P averaged 3.7 mg-P l^{-1} over the 3 years sampled. Removal efficiencies of N and P were as high as 75% and 95%, respectively. Mean net primary production of the freshwater forest system was significantly higher downstream of the effluent discharge (1202 g m^{-2} yr^{-1}) compared to the control site (799 g m^{-2} yr^{-1}), a site at the outfall (917 g m^{-2} yr^{-1}) and a site 1500 m upstream of the outfall (960 g m^{-2} yr^{-1}). Downstream of the outfall, accretion rates were double the rate of relative sea level rise in the area.

The relatively constant flow of secondarily treated wastewater buffered the downstream area from salinity intrusion during a region-wide drought. However, the nutrient loading rate was excessive for the 98 ha floodplain area. The calculated load rate for N into

the system was estimated to be 20.1 g m^{-2} yr^{-1} and the load for phosphorus was estimated to be 5.4 g m^{-2} yr^{-1}. In addition to improving overall surface water quality, the possible re-direction of these nutrient-enhanced effluents to other wetland ecosystems would maintain or increase plant productivity, sequester carbon, and maintain coastal wetland elevations in response to on-going sea-level rise. The local municipality could expect increasing financial savings over time through the use of wetland assimilation instead of an energy-intensive conventional treatment plant.

Thus, the situation at Mandeville, Louisiana illustrates the challenges most coastal communities face as they attempt to move forward with long-term sustainability. Command and control type environmental policies have accomplished much, as regulation of point source pollution and improved water quality indicate. Further progress in this area requires an adjustment in policy strategies to enhance society's capacity to deal with the more difficult problems of non-point source pollution and comprehensive solutions to environmental problems. Increasingly high cost, mainly due to the rising cost for energy, and economic impacts on communities propel the search for cost effective means and for ways to encourage cooperation between the public and private sectors.

Initiation of the ecological analysis also led to the formation of a public-private partnership, and they helped to establish effective working relationships among business officials, state and local policy makers, scientists, environmentalists, and community leaders. To help achieve many of the environmental and financial goals of the partnership, wetland landowners were approached for suitability of possible assimilation of nutrient-enhanced effluents on privately-owned property. The resulting outcome showed that stakeholder interactions are often as complicated and difficult to solve as scientific issues. In this case,

although some wetland owners were acceptable to the idea wetland assimilation on their property, others had conflicting uses and economic issues. Formation of a strong partnership with science as support enabled the municipality to acquire the wetland area needed to successfully implement the project. Therefore, it appears that regional resource management is feasible and in some instances essential for managing coastal resources in both a cost effective and environmentally sound manner. Also, it may be time to reassess policies we have for pollution regulation, climate change, energy and wetland restoration. Currently, air and water pollution regulation in the United States is a command-and-control policy whereby the Federal government generally sets pollution standards, and state governments enforce these standards. National wetland policies require that negative impacts to wetlands are mitigated. By contrast, climate change policies rely little on government intervention, create limited opportunities for investment, and appear these policies are not likely to change in the foreseeable future. Energy policy in the United States has been lacking for several decades, but when acted upon has focused entirely on supply side with little emphasis on stemming consumer and business demand for fossil fuel consumption. A market-based, national policy of nutrient resource management can spur private investment into alternative fuel sources and provide national investment for wetland enhancement that in turn, can mitigate the increasing effects of climate change. In addition, wastewater and river diversions can not only improve overall surface water quality, but will reduce energy use and increase financial savings. The proceeds of a national carbon consumption tax can be used to provide state and private landowners with a monetary incentive to promote wetland enhancement.

A Federal carbon tax of $0.013 on every liter ($0.05/gallon) of US motor gasoline consumed would generate approximately $6.8 billion annually. In addition to stemming the

growing consumer demand for fossil fuel products in this country, and spur reinvestment into alternative fuel sources, the proceeds from the tax would be used as a subsidy to wetland landowners, thereby creating the necessary financial incentive to manage and restore these ecosystems. In summary, wetland areas have tremendous capabilities for nutrient assimilation, burial and storage. The addition of nutrient-rich water to wetlands can increase or maintain plant productivity, sequester carbon, and maintain coastal wetland elevations in response to on-going sea-level rise. Financial incentives for landowners are probably necessary to help achieve wetland restoration efforts.

VITA

Christopher G. Brantley was born in Atlanta, Georgia, on 24 September 1962. He attended public schools in St. Tammany Parish, Louisiana, and graduated from St. Paul's School in Covington, Louisiana, in May 1980. In the fall of 1980 he entered Louisiana State University and received the Bachelor of Science in Forestry degree in May 1985. In January 1986 he entered Southeastern Louisiana University and received the Master of Science degree in May 1989.

In January 1997 he was admitted to the Graduate School at Louisiana State University where he is now a candidate for the Doctor of Philosophy degree in the Department of Oceanography and Coastal Sciences.