Stormwater diversion as a potential coastal wetland restoration method

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STORMWATER DIVERSION AS A POTENTIAL COASTAL WETLAND RESTORATION METHOD

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ABSTRACT

The Barataria-Terrebonne estuary has been eroding at a rate of up to 103.6 km^2 yr^{-1} for several decades. If the current rate of loss is not reduced, an additional 2,550 km^2 of coastal wetlands will be lost by the year 2050. Currently, stormwater in Terrebonne Parish is pumped into canals, ultimately discharging into the Gulf of Mexico. An opportunity exists to use this stormwater for wetland restoration; however, the ecological impacts of stormwater diversions on wetlands are unknown.

The objectives of this project were to 1) to investigate the seed banks of a degraded marsh to determine if a viable seed source exists, 2) to gather baseline soil chemistry of a degraded marsh prior to receiving stormwater input, and 3) to compare the soil P chemistry, metal concentrations and accumulation rates, and sedimentation rates for different wetlands receiving stormwater input. An existing degraded marsh that is scheduled to receive stormwater input in the fall of 2004 was selected for the baseline study. These results will be compared to conditions after a new stormwater pump becomes operational. Coastal wetlands that have been receiving stormwater for <10 years and 30 years were selected to carry out the sediment chemistry study.

A total of 370 stems germinated from the seed banks from the vegetated areas and a total of 2 stems were counted in the seed banks from the mudflat. These results suggest that replanting will be necessary for establishment of vegetation in the mudflat areas. There was a significant increase ($p<0.05$) in P concentrations in soil that had been receiving stormwater for 30 years when compared to soil that had been receiving for stormwater for <10 years, which may lead to alteration of plant communities as seen in the Everglades. Sediment profiles indicated a significant difference in Fe, Al, Zn, and Pb

concentrations when comparing the age of the pump ($P<0.05$). However, soil metal concentrations were found below EPA toxicity thresholds even in coastal wetlands receiving stormwater input for 30 years. The results from this study will be used as a foundation for future studies of stormwater input to coastal Louisiana wetlands.

CHAPTER 1: INTRODUCTION

Historically, delta cycles of the Mississippi River were responsible for the large-scale sediment distribution that created the coastal wetlands of Louisiana. This occurs when a river changes flow direction and the river then flows into a body of open water (Scruton 1960; Coleman and Gagliano, 1964). This slows the velocity of the river, resulting in sediment deposition. Sediment accumulates over time and eventually appears as new land (Welder 1959; Coleman et al. 1969). This new land then becomes colonized by wetland plant species. As the cycle continues, the river again changes course abandoning the previous delta. The abandoned delta is now in a state of decay. The river switches course repeatedly; therefore, an area is always either being built or in state of decay (Louisiana Coastal Wetlands Conservation and Restoration Task Force, 1998; Reed, 1995). The new land is maintained through sediment deposition from seasonal floods (Louisiana Coastal Wetlands Conservation and Restoration Task Force, 1998).

Due to hydrological restrictions, such as levees and dams, placed on the Mississippi River to prevent flooding and direct water flow, natural sediment deposition has been greatly reduced. Had these restrictions not been placed on the river, suspended sediment in the floodwater would be deposited in adjacent wetlands (Louisiana Coastal Wetlands Conservation and Restoration Task Force, 1998).

Louisiana coastal wetland loss is partially a result of these altered conditions of the Mississippi River. The construction of dams on the Missouri and Arkansas rivers, which are major tributaries of the Mississippi River, has reduced the sediment added into the river, decreasing the amount available to coastal wetlands (Kesel, 1989). The amount of sediment found in the river today is approximately 59 million cubic meters, which is roughly half that found prior to the 1950s (Meade and Parker, 1985). Today, because of

the construction of levees, sediment deposition is hindered. Other factors contributing to coastal land loss are sea-level rise, subsidence and compaction, fluid withdrawal, and canal dredging (Boesch *et al*., 1994)

Coastal wetlands provide numerous environmental and economic benefits. Wetlands serve as habitat to many migratory bird species along with many important fish and mammal species. Coastal wetlands also provide substantial protection from storm surge. Coastal wetlands absorb the impact of storms, thus protecting the inland structures (Mitsch and Gosselink, 1986). In addition, Louisiana coastal wetlands are important for the oil industry along with the fishing and oyster production. Oil and gas production in the Barataria-Terrebonne estuary produced annual revenues of $2.4 billion between 1988 and 1994 (Industrial Economics, 1996). The commercial fisheries value for the thirteen parishes of the estuary from 1989-1992 was $193.6 million (McKenzie et al., 1995). The total value loss is projected to be in excess of $37 billion by 2050 (Louisiana Coastal Wetlands Conservation and Restoration Task Force, 1998).

The Barataria-Terrebonne estuary has been eroding at a rate of up to 103.6 square kilometers of marsh a year for several decades. If the current rate of loss is not slowed by the year 2050, an additional 2,550 km^2 of coastal wetlands will be lost (Louisiana Coastal Wetlands Conservation and Restoration Task Force, 1998). One of the contributing factors to the deteriorated state of this estuary is that the Mississippi River is leveed, thereby preventing the historical spring flooding into the Barataria-Terrebonne estuary (Reed, 1995). Another important factor is the submergence rate. This estuary is sinking at a rate of 1cm yr^{-1} (Reed, 1995).

There have been several efforts in south Louisiana to restore deteriorated wetlands. There is a freshwater diversion project at Caernarvon, Louisiana. This

diversion project, still in operation, began in 1991. The Caernarvon project, the focus of many studies, diverts water from the Mississippi River into the Breton Sound estuary. This diversion has created approximately 1.64 km^2 of new marsh (Lane et al, 1999). Before the diversion, the area was becoming increasingly saline, but now has reported lower levels of salinity and no significant impact to the water quality from the diversion (Lane et al, 1999). DeLaune et al. (2003) concluded the reintroduction of freshwater and mineral sediment by the Caernarvon diversion is necessary for the enhancement of marsh vegetation. DeLaune et al. (2003) found that there was higher accretion at the sites nearest the input. They also found higher levels of P at the sites nearest the input. The Caernarvon diversion is lowering the salinity of the Breton Sound estuary and therefore reducing the sediment required for marsh maintenance (DeLaune, 2003).

Previously, freshwater diversions have been created within the Barataria-Terrebonne estuary. In 1992, the Louisiana Department of Natural Resources (LDNR) began one such diversion at West Pointe a la Hache, Louisiana. After five years of monitoring, salinity decreased and plant species increased from eight species at 21 sites in 1992 to 26 species at 36 sites in 1997 (Haywood and Boshart, 1998).

Similarly, the LDNR began a freshwater diversion project near Naomi, Louisiana in 1998. After five years, there were no signs of spatial difference in the plant communities. These results are encouraging since no freshwater marsh was lost (Boshart, 1998). In addition to the marsh remaining intact, the areas that are still classified as brackish now have plants more indicative of lower saline conditions (Boshart, 1998).

Lane et al. (2001) found that as Mississippi River water was diverted through the Bonne Carre spillway, there was a decrease in TN of 26-30% and a decrease of TP of 50-59%. Total suspended sediments also decreased 82-83 % as the Mississippi River water

passed through the spillway (Lane et al., 2001). The results of this study led to the recommendation by Lane et al. (2001) that all diversions of Mississippi River water first be diverted into wetlands before reaching lakes.

Currently, stormwater in Terrebonne Parish, Louisiana is pumped into canals and ultimately dumped into the Gulf of Mexico. An opportunity exists in the Barataria-Terrebonne estuary to use this stormwater for wetland restoration. There are 256 pumping stations in the Barataria and Terrebonne basins. These stations have the potential to be diverted, either completely or partially, into the wetlands (Richards, 1994).

Stormwater diversion is a relatively new tool for wetland restoration. There have been few studies conducted using the exact methods proposed in this project. However, there have been projects that diverted river, agricultural and wastewater to wetlands with some success. These diversions transport high levels of nutrients and sediment. Because stormwater diversions are expected to yield similar results to other methods of diversions utilized in earlier studies, results from these studies can be viewed in comparison.

Agricultural drainage and wastewater are alternative diversion methods. Both of these approaches were used to restore a dry prairie wetland in Canada. After five years, the impacted area had 194 vascular plant species (White and Bailey, 1999). Wetlands not only benefit from the addition of freshwater and sediment, the plant species present may also be able to absorb (at low concentrations) the heavy metals associated with these sources (Srivastav et al., 1993; 1994).

There are several areas in Louisiana in which municipal wastewater is being added to freshwater wetlands. Day et al., (1999) determined that wastewater additions resulted in improvement of water quality, vertical accretion, and increased productivity. These results indicate that wastewater addition could serve as a tool in wetland

restoration (Day et al., 1999). It was determined that while nutrient removal rates are site specific and dependent on nutrient loading rates, wetlands receiving the wastewater should assimilate all of the NO3, and more than 60% of the P (Day et. al, 2000).

Terrebonne and LaFourche parishes lie within the Barataria-Terrebonne estuary in south Louisiana. There are currently 256 stormwater pumps located within these two parishes. Of these, 60 pumps discharge directly into wetlands. In addition to the pumps that are already in place, there is a new pump planned for installation fall 2004 at the Pointe Au Chien Wildlife Management Area (PAC). There were two main aspects of this project. One was to determine baseline soil conditions and to evaluate the soil seed bank at PAC. This information will be compared to data collected after the pump is installed and operational. The other part of the project involved pumps that are already operational. Four sites were selected, 2 that had received stormwater input for 30 years and 2 that had received stormwater input for <10 years. Sediment phosphorus and metal concentrations were compared at these sites.

CHAPTER 2: BASELINE SOIL CHEMISTRY AND SEED BANK CHARACTERIZATION PRIOR TO STORMWATER DIVERSION IN A DEGRADED LOUISIANA MARSH

INTRODUCTION

Coastal erosion is a major environmental problem in the United States and Louisiana accounts for 80% of the total national loss (Boesch et al., 1994). The Barataria-Terrebonne estuary, located in the southern portion of the state, has some of the highest loss rates (100 km^2 yr^{-1}). If the current rate of loss is not slowed, an additional 2500 km^2 of coastal wetlands will be lost by the year 2050 (Louisiana Coastal Wetlands Conservation and Restoration Task Force, 1998).

Louisiana's coasts were shaped through a series of deltas which the Mississippi River formed and then abandoned, and now the modern delta is currently in a state of deterioration (Coleman, et al., 1998). Flood-control efforts throughout the Mississippi River Basin have limited the connection of the Mississippi River with the adjacent wetlands. Without this connection to a freshwater and sediment source, these coastal wetlands have succumbed to subsidence (Louisiana Coastal Wetlands Conservation and Restoration Task Force, 1998). Currently, there are several efforts in south Louisiana to restore these deteriorated wetlands including freshwater diversions, wastewater additions, and, the diversion of municipal stormwater into adjacent wetlands.

There are several river diversion projects throughout south Louisiana that provide deteriorated wetlands with a freshwater and nutrient source. One diversion project is the Caernarvon diversion located south of New Orleans, which pulses Mississippi River water into the Breton Sound. The Caernarvon structure has the capacity to divert

226 $m^3 s^{-1}$. The diversion project at Caernarvon was originally designed to increase freshwater flow therefore decreasing salinity, however because of the increased input of sediment this diversion has created approximately 1.64 km^2 of new marsh (Lane et al., 1999). Freshwater from the diversion lowered salinity levels with no significant negative impacts to water quality. (Lane et al, 1999) River diversions, once used primarily to lower salinity of open water, have been found to be most successful when first introduced to wetlands. Wetlands act as transformers decreasing nutrient levels that could pose as sources of eutrophication in open water (Lane et. al, 2001).

Additional freshwater diversions have been created within the Barataria-Terrebonne estuary. In 1992, the Louisiana Department of Natural Resources (LDNR) began diverting Mississippi River water at West Pointe a la Hache, Louisiana. This diversion project had three different categories of discharge. These included no flow, partial flow (<1,072 cfs) and full flow (>1,072 cfs) (Haywood and Boshart, 1998). The receiving basin had significantly lower salinity during the period of partial or full flow when compared to salinity levels at times of no flow. After 5 years, salinity decreased and plant species increased from 8 species at 21 sites in 1992 to 26 species at 36 sites in 1997 (Haywood and Boshart, 1998).

Similarly, the LDNR began a freshwater diversion project near Naomi, Louisiana in 1998. There was no change in the aerial extent of communities after 5 years, indicating no loss of freshwater marsh (Boshart, 1998). In addition to the marsh remaining intact, the areas previously classified as brackish now had plants more indicative of freshwater marsh (Boshart, 1998).

Both natural and constructed wetlands can serve as treatment methods for wastewater, and this approach has been used on several restoration projects throughout

North America. White and Bayley (1999) reported an increase in species richness after 5 years of wastewater addition to a Canadian prairie wetland. Coastal wetlands receiving municipal wastewater influx had higher accretion rates and plant productivity (Day et al., 1999).

In light of the positive results of other large-scale restoration projects, stormwater diversions are being explored as another potential method to restore wetlands; however, the ecological impacts of stormwater diversions are poorly understood at this time. Stormwater pumps are currently located throughout south Louisiana, with the majority discharging into open water; however, they have the potential to be diverted into wetlands. There are, in fact, approximately 60 pumps currently discharging directly into wetlands, and an additional pump is scheduled for installation for the fall of 2004. This research site represents a degraded coastal marsh isolated from a freshwater source. Through the introduction of stormwater, we will expose the marsh to freshwater, nutrients, and sediments. The overall objective of this project is to investigate the rehabilitative effect of stormwater additions to coastal wetlands. We report here on the baseline soil chemical characteristics before the introduction of stormwater and evaluate the existing seed bank.

METHODS

Study Area

The study area is located within the Pointe Au Chien Wildlife Management Area (PAC), south of Houma, Louisiana (figure 2.1). The Terrebonne Parish government scheduled the installation of a stormwater pump for the fall of 2004. Once a freshwater

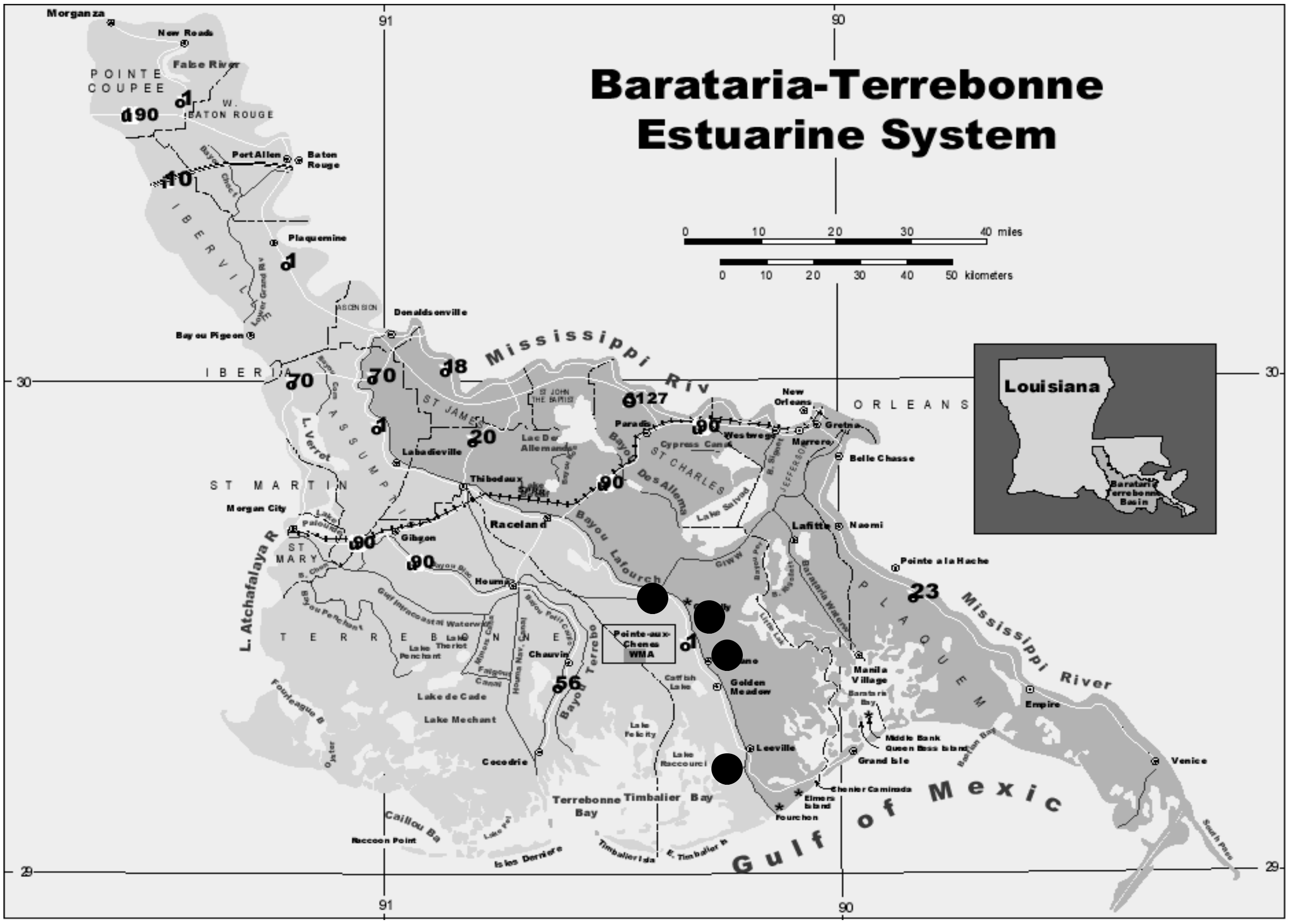


Figure 2.1. Schematic of the Barataria-Terrebonne Estuary. The four black circles are the locations of stormwater pump sites that were selected for comparison in this study.

marsh, PAC now represents a salt/brackish marsh community. Saltwater intrusion has killed off many of the original species and replaced them with more salt tolerant species.

Nine experimental plots were established within the future discharge area. Three experimental plots were randomly located within each of three strata located at different distances from the pump location (figure 2.2). Three control plots were established in an adjacent, hydrologically isolated marsh. This design allowed us to evaluate the impact area of the discharge effluent. We established paired mudflat and vegetated subplots within each experimental plot. Control plots were similar to the experimental plots with vegetated and mudflat subplots present at each plot.

Seed Bank Characterization

In June 2002, we collected cores at a depth of 10 cm at each subplot with a McCauley auger. Two cores were taken from each vegetated subplot to compensate for the volume of root material and provide similar soil volume as the mudflat cores. The cores were transported on ice and refrigerated at $4^{o}C$ for 10 days.

We prepared basins measuring 32 cm x 17.5 cm x 10 cm were prepared by adding 5.5 cm of sterilized sand. Then we added two 8-oz plastic cups, slotted around the bottom sides to each basin to water the soil without disturbing the seeds. Next we labeled each basin for one of the subsamples and randomly placed them on the bench in the greenhouse. We thoroughly mixed each subsample and spread it into its individual basin approximately 1cm deep and added deionized water until the water was even with the soil surface. To ensure that the sand was not supplying any of the seeds, we filled two control basins with sterilized sand only.

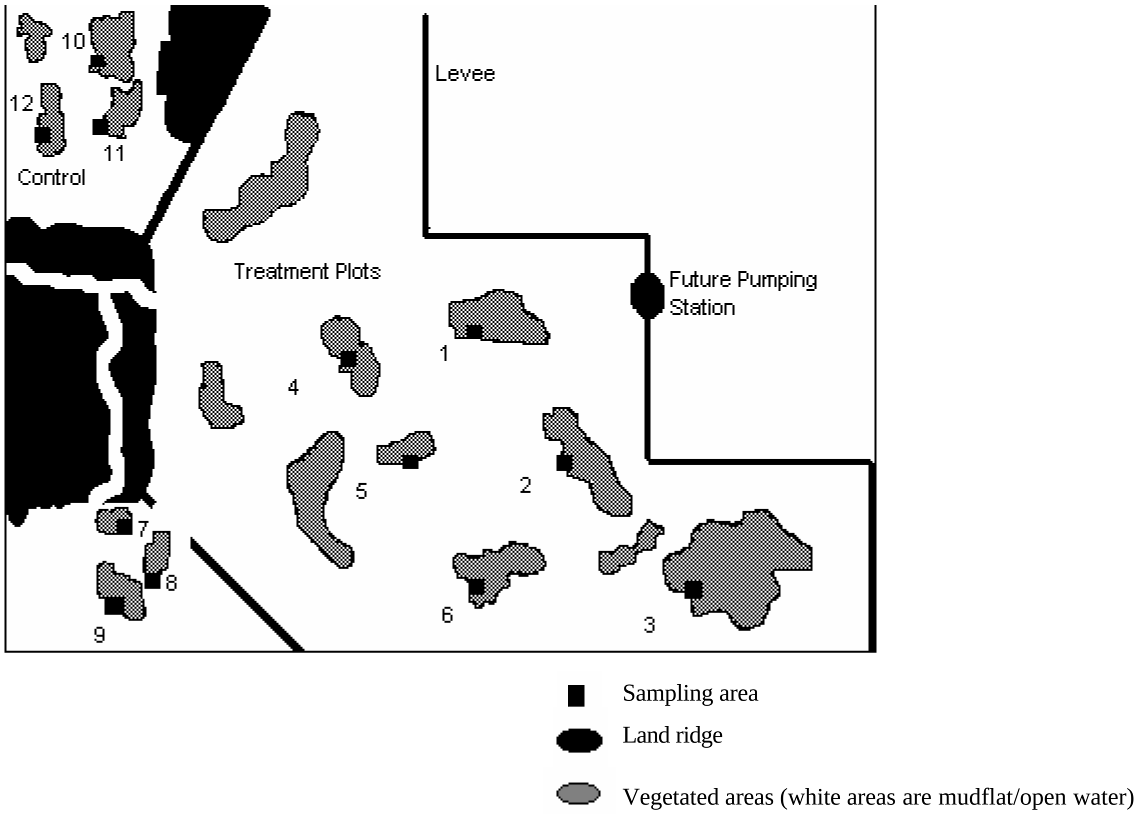


Figure 2.2. Study area at PAC. Plots 1-9 are treatment areas and each are paired vegetated and mudflat area. Plots 10-12 are control plots. The control plots are hydrologically isolated by a land ridge. The control plots also have paired vegetated and mudflat areas.

Every two days we added approximately 250 ml of water to each basin. The water level was allowed to rise to the level of the sand, moistening the soil and after 7 days, the first seedling germinated. We then allowed the seedlings to grow until physical differences could be identified, and we then grouped each seedling accordingly. We transplanted a representative of each group to a larger separate container until species identification to was possible.

All remaining seedlings were identified to genus, counted, and removed after 56 days and then we gently raked the soil to allow for a new germination period. No new germination was observed after 99 days.

Vegetation Sampling

To determine the current species composition at PAC four 1-m^2 plots were randomly established in each of the designated vegetated areas. To determine the species present at PAC, a visual observation of percent coverage was completed within each 1-m^2 plot in July 2003. The results from each plot were averaged. The current vegetation community was assessed through visual estimation of percent coverage using randomly distributed 1-m^2 plots in the vegetated areas.

Baseline Soil Chemistry

Two samples were taken from each treatment area. Each core was a composite of three cores. Soil cores were transported on ice and refrigerated for 7 days, thoroughly mixed and sieved to remove root material and further homogenize the soil. The soil was then dried at 100^{o}C for 24 hours to determine moisture content. For nutrient analysis 10 grams wet soil samples were exposed to 50 mL 2M KCl for 1 hour. The samples were centrifuged and the supernatant was filtered with Whatman 42 filter paper. Metal analysis was completed by placing 2 grams of wet soil into digestion tubes. The samples

were exposed to 1:1 HNO_3 and heated at 95°C for 15 minutes. After the samples were cooled to room temperature, 5 mL of concentrated HNO_3 was added. Samples were heated for 2 hours, cooled to room temperature and then 3 mL of 30% H_2O_2 + 2 mL deionized water was added. The samples were heated for an additional two hours, cooled, and concentrated HCl was added. The samples were heated for an additional 15 minutes. Samples were cooled overnight, filtered and brought to volume in a 100 ml flask. Metals were analyzed using inductively coupled argon plasma spectrometry (ICP) and NO_3^- and NH_4 were analyzed with a Lachat Automated Flow Analyzer (Lachat, 1999).

RESULTS AND DISCUSSION

Seed Bank Characterization and Vegetation Composition

There was growth in 92% of the basins containing soil from the vegetated sites. Within these basins, 410 stems emerged, representing *Amaranthus australis* (Gray) Sauer, *Eleocharis* sp. (R.), *Pluchea odorata* (L.) Cass. , *Cyperus odoratus* (L.), and *Bacopa caroliniana* (Walt.) B. L. Robins. Three stems were counted in the mudflat soil. The species that germinated from the mudflat soil were *Echinochloa crus-galli* (L.) Beauv. and *Leptochloa fascicularis ssp. fascicularis* (Lam.) Gray.

Both *Spartina patens* (Ait.) Muhl. (37%) and *Spartina alterniflora* (Loisel.) (37%) are currently dominant species at PAC. Each species represents 37% of the total percent coverage for a total of 74% *Spartina spp*. of the vegetation at PAC (figure 2.3). The vegetative community at PAC differed from the seed bank, which held as the dominant species *Eleocharis* sp. and *Pluchea odorata* (figure 2.4). There were 236

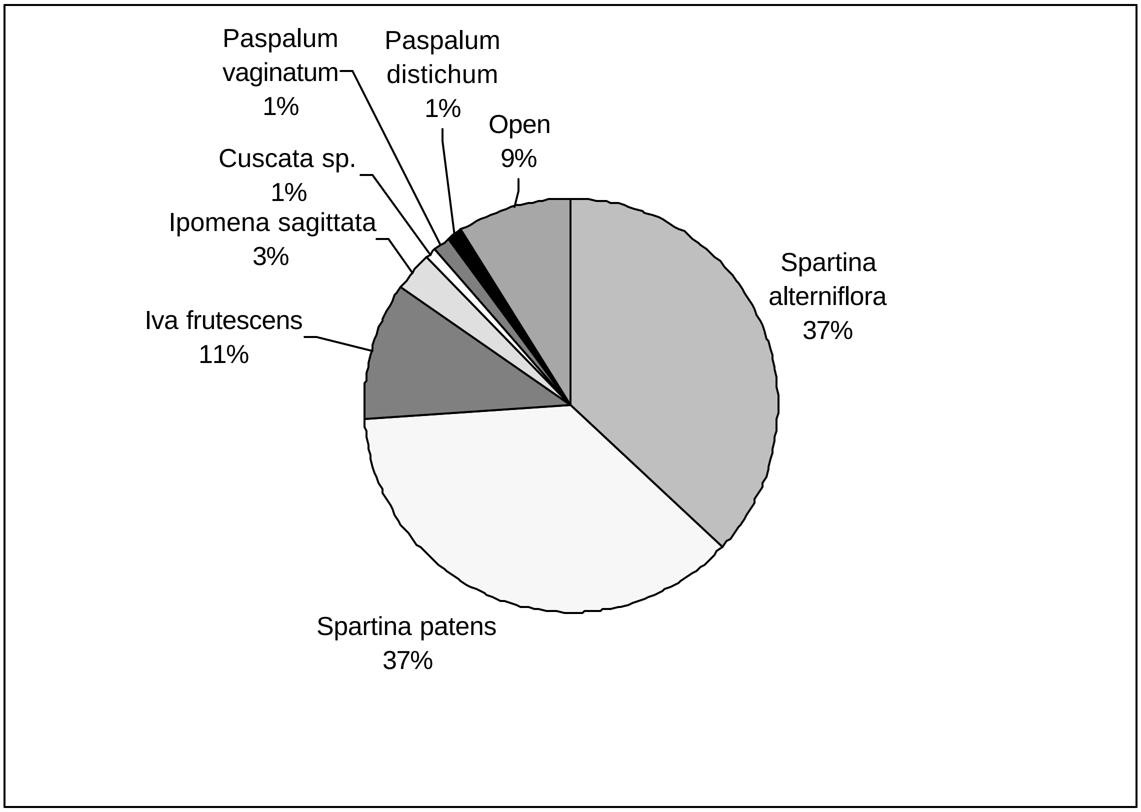


Figure 2.3. Species composition within the vegetated areas at PAC determined through visual observation by randomly placing 1-m^2 plots along a 50-m transect. *Spartina alterniflora* and *Spartina patens* are the dominant species, representing 74% of the total species.

Eleocharis sp. stems and 146 *Pluchea odorata* stems. Twenty-one *Cyperus odoratus* stems, 7 *Amaranthus australis*, and 2 *Bacopa caroliniana*. None of the stems counted were *Spartina* spp.

Eleocharis sp. has a wind-dispersed seed, and *Pluchea odorata* has a long-lived seed. Therefore it is possible that the seed bank represents vegetation communities of adjacent areas or historical communities. Leck et al. (1989) suggested that the differences seen between the vegetation and the seed bank might be caused by both the viability and number of seeds contributed by the dominant species. Leck et al. (1989) also suggested that the conditions present might not allow for germination of the species within the seed bank. The dense canopies of the vegetation currently growing at PAC may explain the absence of the seed bank species in the vegetation community (Baldwin et al. 1996). It has been suggested that increased salinity also may contribute to the lack of seed bank species represented in the vegetative community (Baldwin et al. 1996). In previous studies, Baldwin found that wetlands with a dominant *Spartina* sp. community did not have *Spartina* sp. in the seed bank (Baldwin et al. 1996 and Baldwin and Mendelssohn 1998) but had *Eleocharis parvula* (Romemer and J. A. Schultes) as the dominant species in the seed bank. The results presented here are consistent with his findings.

The absence of viable seeds within the mudflat area may be partially caused by the current flooding regime. The construction of levees has isolated PAC from a freshwater source, and the system's only water supply is the adjacent open water of Little

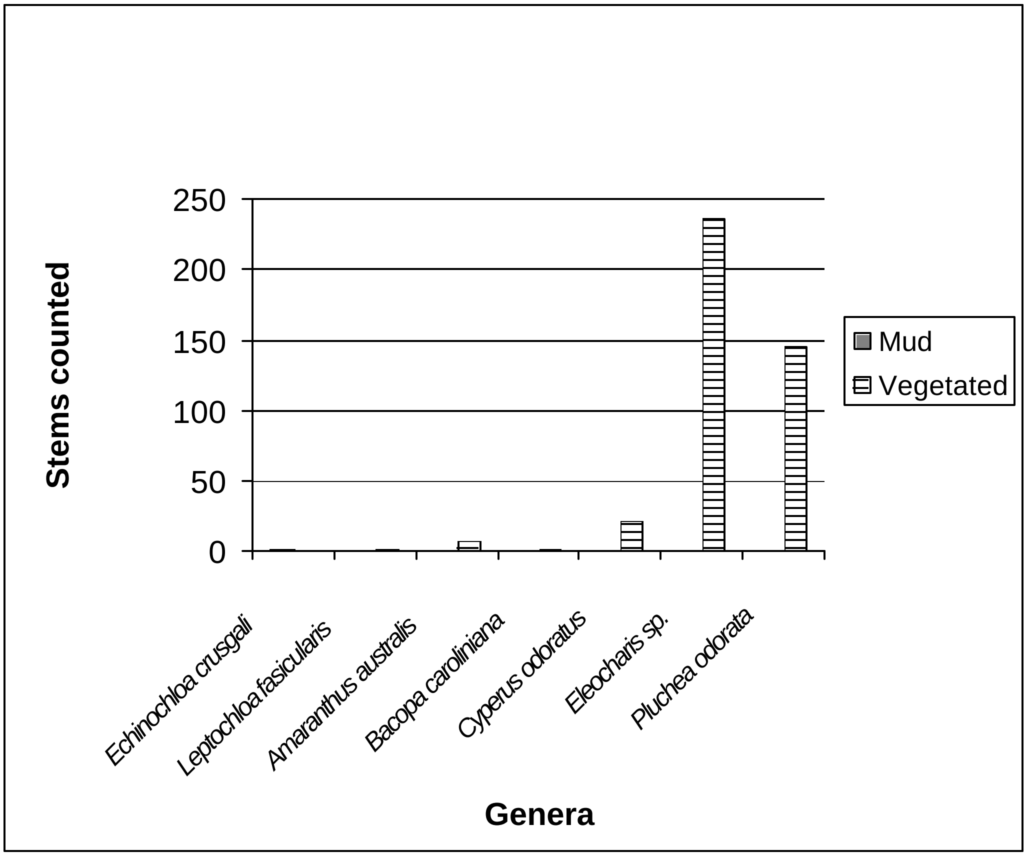


Figure 2.4. Genera observed in the seed banks upon freshwater addition in a greenhouse experiment. *Eleocharis* sp. and *Pluchea odorata* were the dominant species present in the seed bank collected from PAC.

Lake. This brackish water source annually floods the mudflats of PAC April-October. Flooding and increased salinity are proven stressors (Baldwin et al. 1996), and it is probable that the current hydrologic regime of PAC led to the lack of viable seeds found in the mudflat. Seasonal flooding may have also contributed to the seed accumulation on the vegetated areas. Many wetland species seeds can float and will accumulate in the vegetated areas, resulting in a larger seed bank on the vegetated sites than the mudflat areas (Leck et al. 1989). van der Valk and Davis (1978) found that as little as 2 cm of water can inhibit germination in species. Mudflat species may be able to germinate under drawdown conditions; however, they are often unable to donate seeds to the seed bank because they are destroyed upon flooding (van der Valk and Davis 1978).

In addition to inundation and salinity stress, the mudflat soil is not well consolidated. The soil is unconsolidated because the levee at the site has been under construction and has breached in several storms. These conditions have loosened the sediment and possibly mixed the soil, preventing the establishment of seeds. Mixing the soil may bury seeds and lead to increased mortality or, because they were hidden, lack of collection in our sample. It has been previously reported that soils of fine texture have lower rates of seedling success and seed viability (Leck et al. 1989).

Considering the seed bank present in the vegetated areas, we believe that the addition of freshwater from stormwater diversions will enhance the growth and development of freshwater species. The mudflat areas, however, do not currently have the same potential capacity. The results from our study suggest that the mudflat areas will not vegetate naturally under the current flooding patterns. To restore this area of open water/mudflat to a viable wetland, manual replanting may be necessary.

Baseline Soil Chemistry

Because the stormwater that will be introduced to PAC will be a collection of runoff from the surrounding area, metal and nutrient concentrations are a concern and will be monitored. Since wetlands have proven to have the ability to absorb metals from contaminated water (Pardue et al. 1988), therefore baseline concentrations have been determined before the pump is installed. Stormwater pumps are not always operational, therefore, the vegetation present should be able to aid in removal of any excess metals and nutrients with sufficient recovery time between exposure periods.

The heavy metals measured in the soil at PAC include Cu, Zn, Cd, Pb, Cr, and Ni. Throughout the study site Zn was in the highest concentration 18.63 ug g^{-1} – 23.61 ug g^{-1}, while Cd was below detection in all samples. We also measured Fe, Mn, Ca, and Mg along with NO_3^- and NH_4^+. Fe was found in the highest concentration 4658 ug g^{-1}- 6623 ug g^{-1}, while Mn is found in lowest concentration 148.9 ug g^{-1} -572.3 ug g^{-1} .

There have been few studies of background metal concentrations in the soil of south Louisiana. DeLaune et al. (1981), Pardue et al. (1988), and DeLaune and Gambrell (1996) documented their findings of metal concentrations along the coastal zone of Louisiana. These studies included areas with a known source of contamination and areas that had no known contamination. PAC currently has no known source of contamination. The metal concentrations currently present in the soil at PAC are within the range previously reported for the uncontaminated sites. DeLaune et al. (1981) concluded that the sites investigated in that study had low concentrations of heavy metals and therefore represented a pristine marsh. The concentrations reported here for PAC are consistent with the findings by DeLaune et al. (1981). The contaminated areas that were

Table 2.1. Mean soil nutrients and metal concentrations in soil at Pointe Au Chien WMA prior to stormwater input. Concentrations in ug g^{-1} (±SE)

Constituent	Vegetated treatment	Mudflat treatment	Vegetated control	Mudflat control
	------------------------------	ug g^{-1}	-------------------------------	
Cu	4.79 (2.36) a*	1.14 (0.76) b	4.42 (2.34) a	0.78 (0.64) b
Zn	18.83 (1.31)	18.63 (1.48)	21.29 (1.37)	23.61(1.54)
Cd	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
Pb	4.61 (0.99)	4.77 (0.91)	1.01 (0.82)	0.00 (0.00)
Cr	1.78 (1.35)	0.40 (0.26)	0.63 (0.24)	0.00 (0.00)
Ni	2.67 (0.55)	3.10 (0.44)	1.20 (0.62)	0.00 (0.00)
Fe	5144 (487) a	6623 (442) b	4658 (207) a	5886 (265) b
Mn	572.3 (185) a	300.0 (46) b	566.6 (122) a	148.9 (4.36) b
Ca	1286 (41) a	3302 (26) b	1556 (80) a	4321 (360) b
Mg	1871 (54)	1800 (87)	2086 (134)	1987 (62)
P	292 (31)	170 (7)	240 (20)	174 (12)
Na	5747 (238)	5231 (399)	7228 (473)	7725 (204)
NH_4	2.411 (0.001)	2.345(0.001)	1.889(0.001)	1.623(0.001)
NO_3	0.018 (0.001)	0.012(0.001)	0.008(0.001)	0.016(0.001)

*a is significantly different than b when comparing vegetated areas to mudflat areas.

investigated had higher concentrations of metals in the deeper sediment when compared to the surface sediment. This was concluded to indicate that the higher concentrations of metals correlate with pollution that occurred many years ago and the lower surface concentrations indicate that fewer metals have been discharged in recent years (DeLaune and Gambrell, 1996).

Recent studies have focused on wetlands' ability to remove metals present in urban runoff. Scholes et al. (1999) investigated the removal of metals by constructed wetlands. This study found that with high loading rates during storm events, the removal of metals was high and removal was variable during dry weather. Based on a 50-minute retention time, Scholes et al. (1999) found removal efficiencies for Zn 55%, Pb 62%, Cu 85%, Ni 77% and Cr 63%. Walker and Hurl (2002) found that the total concentrations of metals decreased as the stormwater traveled through the wetland. Walker and Hurl (2002) were interested in the importance of sedimentation and chemical and biological processes in the removal of heavy metals. They found that the total amount of metals decreased through the wetland, but the relative concentration varied with distance. Zinc, lead, and copper all decreased with distance from inlet, chromium remained constant, while arsenic concentrations increased with distance. Walker and Hurl (2002) concluded that this indicated that other processes than sedimentation were responsible for metal removal. Based on these studies we expect removal of heavy metals by the sediment at the sites closest to the discharge site for the majority of the metals measured.

The research design at PAC takes into consideration distance from the pump, however, because the pump has not been installed the only treatment reported here is the difference in soil chemistry between the paired vegetated and mudflat areas. A mixed model in SAS was used to analyze the metal data set from PAC (SAS Institute, 1999).

When comparing the concentrations of metals and nutrients in the soil of the mudflat and vegetated areas there was a significant difference in the concentrations of Cu, Fe, Mn, and Ca ($P<0.05$). There was not a significant difference between the mudflat and vegetated areas for NO_3^- or NH_4^+. Ammonium concentrations were higher than NO_3^- at PAC. NH_4^+ has been shown to be present in higher concentrations than nitrate in other studies of wetland soils (e.g. Buresh et al., 1981, Reddy et al., 1976, Smith et al., 1982). Nitrate is present in the aerobic soil or water column in wetlands. Once diffused into the anaerobic layer, NO_3^- is reduced to NH_4^+. Wetland soils are predominately anaerobic therefore producing much more NH_4^+ than NO_3^-. NO_3^- is in low concentration because of the small oxidized layer present in wetland soils. The microbes quickly use the NO_3^- that is present and N_2 gas is formed through denitrification. The samples taken at PAC were at a depth of 10-cm, which would encompass a much larger anaerobic area than aerobic, therefore it would be expected that NH_4^+ would be the dominant form of N. The concentrations of metals and nutrients found at PAC are reported in Table 2.1. These data will serve as an indicator of the wetland's ability to remove metals and nutrients once the pump is fully operational.

CONCLUSIONS

Because of the absence of any seed bank, it will be necessary to replant the mudflat areas in order to establish vegetation. Given the difficulty of working in the unconsolidated sediments, direct seeding may be a better alternative. The metals and nutrient data for PAC are currently within a range previously reported for the coastal zone of Louisiana and will serve as baseline data once the pump is installed. As earlier stated, the majority of the stormwater pumps currently discharge directly into open water. Wetlands can reduce the concentration of potential contaminants before they are

discharged into coastal waters. The addition of freshwater, sediments, and nutrients can rehabilitate degraded coastal wetlands.

It is our recommendation that terraces are established within the mudflat and these areas be replanted or reseeded. We are optimistic that seeds from outlying areas may be successfully introduced to this area if recruitment and germination conditions are suitable. Higher plant species diversity will be expected as the elevation is raised and a freshwater source is supplied to the area at PAC through the addition of stormwater.

CHAPTER 3: SOIL PHOSPHORUS FRACTIONS IN LOUISIANA COASTAL WETLANDS AFTER 10 AND 30 YEARS OF STORMWATER INPUT

INTRODUCTION

Stormwater management is essential in areas of low elevation in south Louisiana. Currently, stormwater is collected through a series of ditches and canals, and ultimately discharged into a drainage canal or directly into a wetland. Terrebonne and LaFourche parishes lie within the Barataria-Terrebonne estuary in south Louisiana. There are currently 256 stormwater pumps located within these two parishes. Of these, 60 pumps discharge directly into wetlands. The primary purpose of these pumps is to relieve the southern parishes of flooding after storms; however their potential for rehabilitation of the coastal wetlands has been a recent focus (Richards 1994). There are several restoration projects throughout south Louisiana that are attempting to reconnect deteriorated marshes with a freshwater and nutrient source.

There have been several river diversion projects in south Louisiana that had a great deal of success. Caernarvon is a major river diversion that has been the focus of much research. The Caernarvon structure has the capacity to divert 226 $m^3 s^{-1}$. The diversion project at Caernarvon was originally designed to increase freshwater flow therefore decreasing salinity; however because of the increased input of sediment this diversion has created approximately 1.64 km^2 of new marsh (Lane et al., 1999). Freshwater from the diversion lowered salinity levels with no significant negative impacts to water quality (Lane et al, 1999). Wetlands act as transformers decreasing nutrient levels that could pose as sources of eutrophication in open water (Lane et. al, 2001). DeLaune et al. (2003) found that the diversion of the Mississippi River water into

the Breton Sound increased the rate of accretion. In light of the positive results of other large-scale restoration projects, stormwater diversions are being explored as another potential method to restore wetlands; however, the ecological impacts of stormwater diversions are poorly understood at this time. Dolan et al. (1981) found that Florida wetlands receiving treated wastewater removed P from the water. Dolan et al. (1981) reported that 70% of the phosphorus that entered the system was stored within the soil. Wetlands not only can act to improve water quality, but the additional P source can stimulate plant growth and productivity of the marsh because phosphorus is an essential plant nutrient.

The Everglades have been the site of much research because it has receiving increased nutrient input over the last few decades from agricultural runoff. Qualls and Richardson (1995) investigated the nutrient concentration along a gradient in the Everglades. They found that the while the P concentration decreased with distance from input, soil in the marsh was impacted several km into the interior of the marsh. Craft and Richardson (1998) used ^{137}Cs dating to determine that the high rate of P accumulation at sites enriched by P in the Everglades is recent accumulation.

There have been several studies that have focused on wetland's ability to uptake P from overflow. Moustafa (1999) examined wetlands' ability to remove P from surface water in the Everglades. This study found that the wetlands investigated were effective in P removal. Mousatafa (1999) reported P removal rates of 64% and 71%. Chiang et al. (2000) found that after four years of nutrient addition to the Everglades there was an increase in P uptake by all plants investigated. There was also an increase in soil P in both the labile and occluded forms, while most was Ca-bound P (Chiang et al., 2000).

Chiang et al. (2000) concluded that the cumulative addition of P was more important than the annual loading rates in determining how the ecosystem will respond.

By examining the soil of areas that have received stormwater over time, we can determine if the addition of this stormwater has an effect on the P chemistry. The objective of this project was to evaluate soil P chemistry in wetlands that have been receiving stormwater for 30 years with wetlands that have been receiving stormwater for <10 years. I hypothesized that the older wetlands would have higher soil P concentrations.

METHODS

Study Site

All the sites selected are located within the Barataria-Terrebonne estuary, located in the southeastern portion of Louisiana (Figure 3.1). All sites chosen for this study represent degraded coastal marsh systems and have been receiving stormwater between 5 and 29 years. The sites selected have had minimal human impact (i.e. dredging). Based on visual observation, the land use of the area is primarily rural and agricultural.

The sites were selected based on the marsh type, maximum discharge of the pump, and age of the pump. All wetlands selected represent a coastal grass marsh. Two of the wetlands have been receiving stormwater for 30 years, and two have been receiving stormwater for <10 years. This allows for an investigation of the soil chemistry of the receiving basins to determine the impact of stormwater input over time. Each site was separated into vegetated plots and adjacent mudflat areas. The mudflat areas are seasonally flooded in the late spring and summer months and are then areas of open water. The pumps discharging into the area have maximum discharge rates between

4020 L s^{-1} – 7305 L s^{-1}. Each pump averages 1-hour day^{-1} at maximum discharge (Blaise LeCompte, personal communication, 2003).

P Fractionation

Soil samples were collected February 2003 at 10-cm depth using an aluminum corer. At each site, soil was collected from vegetated areas closest to the pump discharge and the adjacent mudflat region. There were five replicates taken at both the vegetated and mudflat areas. Each replicate was a composite of three cores. Samples were transported back to Louisiana State University on ice and stored at 4°C until samples were processed. A subsample of each sample collected was dried to determine moisture content of the soil. After moisture analysis was complete, a 0.5 g wet weight soil sample was placed in a 50 mL centrifuge tube. The P fractions of each soil sample were analyzed using a modified version of the Hedley fractionation (Tiessen, and Moir, 1993; Cross and Schlesinger, 1995). In this procedure, the samples are exposed to a sequential series of reagents (figure 3.2). The reagents used vary in alkali and acidic strengths to extract different forms of P. The resin and $NaHCO_3$ fractions represent the inorganic P (Pi) that is most readily available to the plants (labile Pi). The NaOH fraction represents the non-occluded Pi associated with Fe and Al. The sonicated NaOH fraction represents occluded Pi held at the soil surface associated with Fe and Al phosphates. The HCl fraction represents the occluded Ca-bound P. The residual P fraction represents the most stable phosphates (Cross and Schlesinger, 1995; Tiessen and Moir, 1993).

A general linear model procedure of the analysis of variance was used to determine if the age of the pump had a significant effect on the P concentration and to determine whether the soil was collected from the mudflat or vegetated areas (treatments) had a significant effect on the P concentrations. (SAS Institute, 1999).

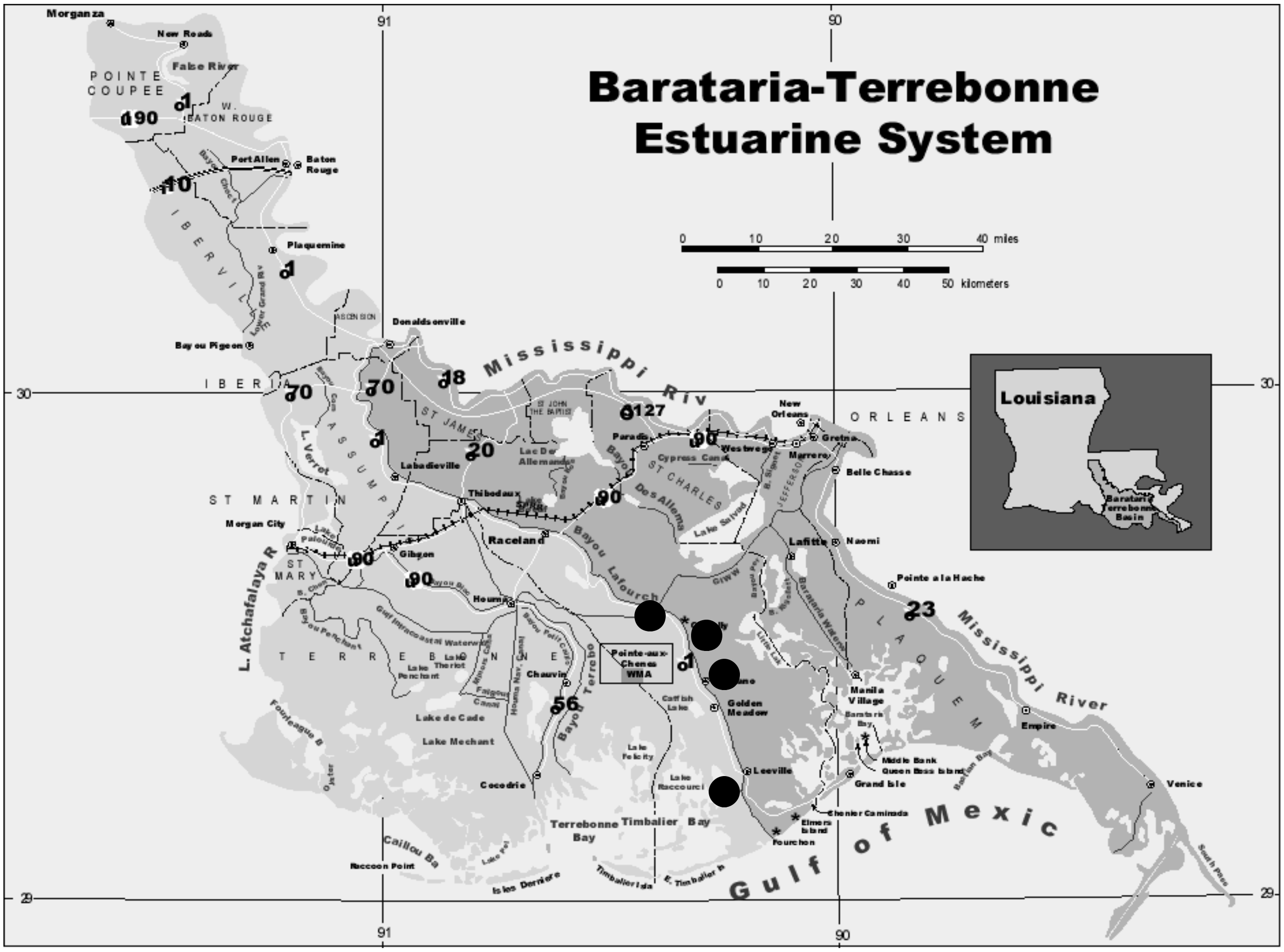


Figure 3.1. Schematic of the Barataria-Terrebonne Estuary. The four black circles are the locations of stormwater pump sites that were selected for comparison in this study.

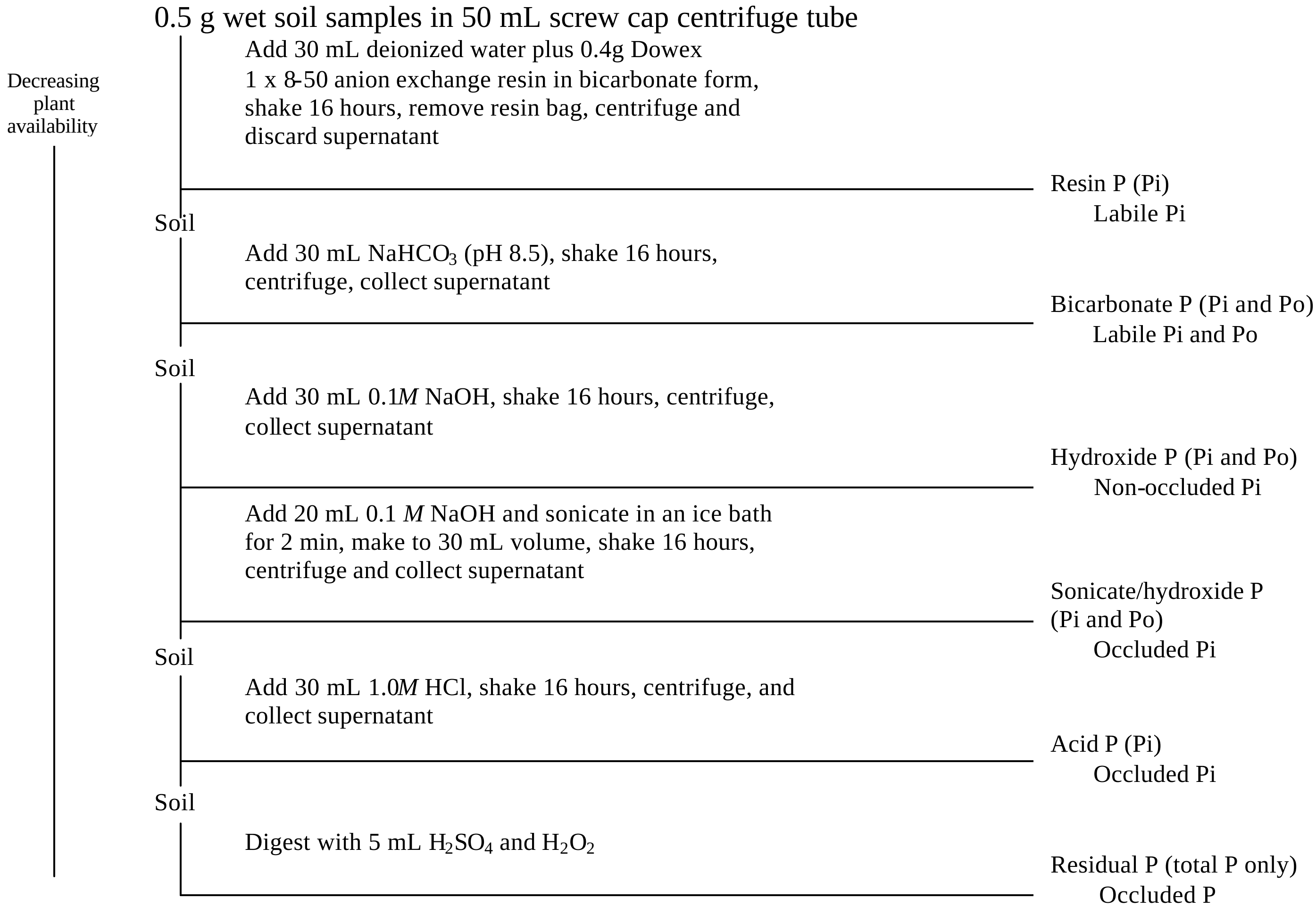


Figure 3.2. Sequential phosphorus fractionation method for soils (Cross and Schlesinger, 1995)

TP and Dissolved P in Water

Water samples were collected in August 2003. Samples were collected in acid washed liter bottles. Six liters were collected from each site, three from each pump and three from each receiving basin. The samples were transported back to Louisiana State University on ice and were stored under refrigeration. 100 ml subsamples were filtered through 0.45 uM syringe filters less than 24 hours after being collected and frozen. These samples were thawed to room temperature immediately prior to P determination. The water was exposed to a Murphy-Riley color solution and the absorbance value was measured on a spectrophotometer to determine the dissolved P concentration. Total P concentrations of unfiltered water samples were determined on a Lachat Automated Flow Analyzer (Lachat, 1999) following persulfate digestion (Qualls, 1989).

To calculate loading rates to the wetland the discharge was adjusted to allow for one hour at maximum discharge day^{-1} year^{-1} for each pump (Blaise, Lecompte, personal communication, 2003). An areal estimate was not done for any of the sites. To determine the loading rate the concentration of the metals were multiplied by the maximum discharge. This rate of flow provided an estimate of metal concentration entering the system. The annual load was multiplied by the time the pump had been operational to determine the total metal concentration that has been exposed to the system.

- Annual loading rate = (Max Q*Total P concentration* 365)
- Total loading rate = annual loading rate * years pump is in operation

RESULTS

P Fractionation

The resin-extractable P fraction was found in the highest concentration in both age groups (Table 3.1). This fraction ranged from 17 ug g^{-1} to 100 ug g^{-1}. The NaHCO3 fraction ranged in concentrations of 2 ug g^{-1} to 19 ug g^{-1}. The NaOH fraction ranged in concentrations of 3 ug g^{-1} to 43 ug g^{-1}. The sonicated/NaOH fraction ranged in concentrations of 0.4 ug g^{-1} to 4 ug g^{-1}. The HCl fraction ranged in concentrations of 0.7 ug g^{-1} to 131 ug g^{-1}. The residual Pi concentrations ranged in concentrations of 1 ug g^{-1} to 5 ug g^{-1}. The total soil Pi ranged in concentrations of 24 ug g^{-1} to 291 ug g^{-1} (Table 3.1).

The labile fraction (resin Pi + bicarb Pi) represented the highest percentage of the total at all sites, followed by the HCl-Pi. The sonicated/NaOH Pi represented the lowest percentage of the total at all age sites (figure 3.3).

The total Pi concentration (the sum of all the Pi fractions) was significantly higher in the soil collected from both the mudflat and vegetated plots of the wetlands that received stormwater for 30 years when compared to wetland soil that received stormwater for <10years (Table 3.1). The $NaHCO_3$, resin, NaOH, and sonicated/NaOH fractions were also all significantly higher in the sites that received stormwater for 30 years when compared to those that received for <10 years ($p<0.05$). Both the NaOH- and sonicated/NaOH- fractions showed a significant difference when comparing the mudflat soil to the vegetated soil ($P<0.05$).

Table 3.1. Mean (± SE) inorganic soil P fractions (ug g^{-1}) of Louisiana coastal wetlands receiving stormwater.

P Fraction	----30 year old pumps--- vegetated	mudflat	----<10 year old pumps--- vegetated	mudflat
[1]Resin Pi	100 (27) *	70 (29) *	17 (2) *	24 (4) *
[1]$NaHCO_3$ Pi	19 (8) *	4 (2) *	2 (1) *	2 (1) *
[2]NaOH Pi	43 (14)*(a)	12 (3)*(b)	5 (1)*	3 (1)*
[3]Sonic/NaOH Pi	4 (1.7)*(a)	0.6 (0.2)*(b)	0.4 (0.2)*	0.6 (0.1)*
[4]HCl Pi	131 (85)	48 (43)	0.7 (0.2)	12 (7)
[5]Residual Pi	4 (1.6)	5 (2)	1 (0.4)	1 (0.3)
[6]Total Pi	291 (105)*	139 (60)*	24 (5)*	44 (11)*

* Indicates means were significantly different (P<0.05) between 30 year sites and sites <10 years old.

a, b indicates means were significantly different (P<0.05) between the mudflat and the vegetated soil within an age class

[1]Resin Pi and $NaHCO_3$ Pi is labile P

[2]NaOH Pi is non-occluded P that is associated with Fe and Al

[3]Sonic/NaOH Pi is occluded P that is held at the soil surface and associated with Fe and Al

[4]HCl Pi is occluded P that is Ca-bound

[5]Residual P is the most stable P

[6]Total Pi is the sum of all Pi fractions

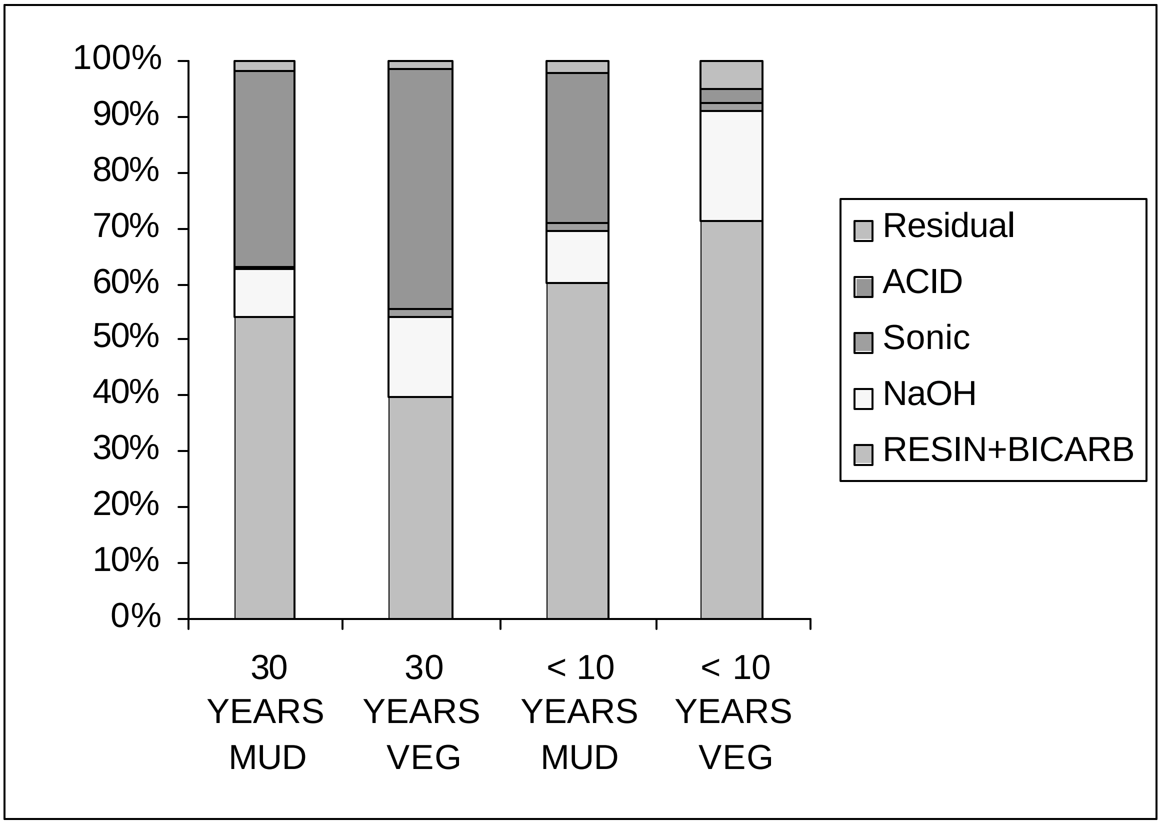


Figure 3.3. Relative percentages of P fractions when compared to the total Pi concentration in each group. Resin and Bicarb fractions are the most available for uptake by the plants. NaOH fraction is the non-occluded Fe and Al-P. The Sonic fraction is occluded and held to the internal surfaces of the soil. Acid fraction is occluded and associated with Ca-bound P. Residual P is the most stable P.

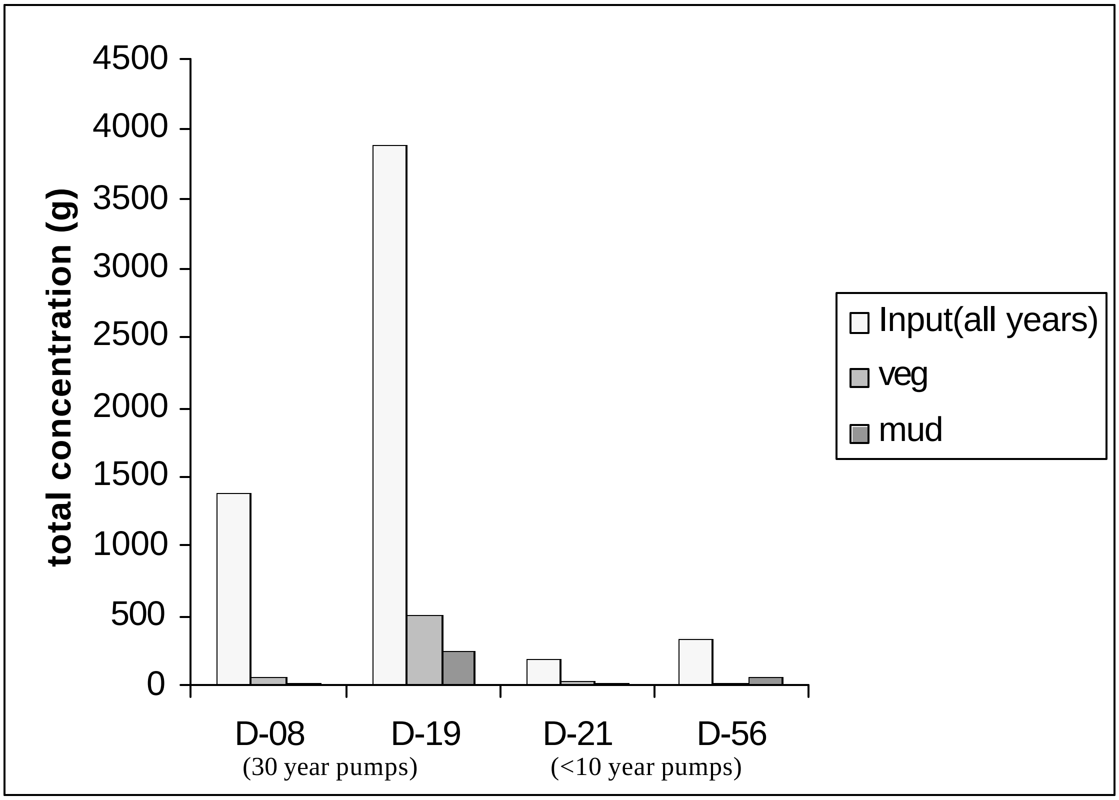


Figure 3.4. Comparison of the dissolved P in the input water and the TPi in the soil of the site. The input is calculated by multiplying the yearly load by the number of years that the pump has been operational.

TP and Dissolved P in Water

The TP concentrations in the water samples collected from the pump of each site ranged from 0.2 mg L^{-1} to 0.7 mg L^{-1} (Table 3.2). The dissolved P concentrations in the water samples collected from the pump of each site were approximately 0.10 mg L^{-1} all sites except one that had a concentration of 0.37 mg L^{-1} (Table 3.3).

While total P loading was greater in the 30 year sites than the 10 year sites, there were some site differences as well. One site exposed to stormwater for 30 years (D-19) had more dissolved P concentrations in the water collected from the pump than the other sites. This site has a much larger drainage area than the other sites. The D-19 site collects runoff from an area of 70.8 ha, while the other sites (D-08, D-21 and D-56) have drainage areas of 10.1 ha, 7.2 ha, and 29.9 ha, respectively. The P concentration in the soil was higher than the other sites (Figure 3.4). This could indicate that the site differences could be a function of loading concentrations. The maximum discharge of each pump and the mean loading rates can be found in tables 3.2 and 3.3.

DISCUSSION

The labile forms (resin-Pi, NaHCO3-Pi and the NaOH-Pi) contribute the most to the total Pi at both age groups. However, the Pi at the vegetated sites of the wetlands receiving stormwater for <10 years is almost completely found in the labile pool, with <10% of the Pi represented in the occluded form (all other fractions). Labile Pi constitutes over 50% of the total Pi in both the wetlands that have been receiving stormwater for 30 years and those receiving for <10 years (figure 3.3). The sites investigated in this study have been receiving an additional nutrient source for several

Table 3.2. Total phosphorus concentrations, annual loading rates and total loading rate

Pump ID	Max Q ($L\ s^{-1}$)	Concentration ($mg\ L^{-1}$)	-------loading rates------- Annual ($Mg\ year^{-1}$)	Total (Mg)
D-08[1]	4020	0.70	0.36	11.1
D-19[1]	6031	0.66	0.34	10.3
D-21[2]	4020	0.42	0.22	1.1
D-56[2]	7305	0.29	0.28	2.5

[1] Operated 30 years
[2] Operated <10 years

Table 3.3. Dissolved phosphorus concentrations, annual loading rates and total loading rate

			-------loading rates-------	
Pump ID	Max Q ($L\ s^{-1}$)	Concentration ($mg\ L^{-1}$)	Annual ($Mg\ year^{-1}$)	Total (Mg)
D-08[1]	4020	0.13	0.68	2.06
D-19[1]	6031	0.37	0.29	8.79
D-21[2]	4020	0.10	0.05	0.26
D-56[2]	7305	0.10	0.09	0.86

[1] Operated for 30 years
[2] Operated for <10 years

years. The Pi increase in the sites that have been receiving stormwater for 30 years when compared to those receiving for <10 years indicates that P has been entering the system and has been stored in the soil. The high amounts of resin-Pi (labile P) may be influenced by the time of year the samples were collected. Qualls and Richardson (1995) found that the labile pool was highly variable depending on the season the soil was sampled. If the soil samples had been collected in the summer months, or during the growing season, the labile Pi in the soil may have been reduced because the plants would have taken it up. Buresh et al. (1980) found that P associated with plants was higher in the June collection than the September collection. The sites receiving stormwater for 30 years have a higher percentage of the total Pi found in occluded forms (sonicated/NaOH Pi, HCl-Pi, and residual P) when compared to sites receiving stormwater for <10 years. Cross and Schlesinger (1995) found that labile Pi was present in the least amount, while the majority of the phosphorus pool was occluded phosphorus. They concluded that this indicated geological processes regulated the phosphorus cycle in the soils investigated in that study. This is not the case for the data presented here. The phosphorus pool is largely made up of labile Pi. This suggests that biological processes regulate the phosphorus cycle at the sites selected for this study.

The areas that have been receiving stormwater for 30 years had a larger input of phosphorus than sites receiving stormwater for <10 years (Table 3.2). There is a significant increase in the Pi concentrations found in the soil of the areas that received stormwater for 30 years. The increase in phosphorus concentration at the sites that had greater P loading is not surprising; however, the distribution of available forms within the soil is interesting (figure 3.2). A larger portion of the phosphorus pool was made up of occluded P in the sites receiving stormwater for 30 years when compared to those

receiving < 10 years. There are several possible explanations for the increase in occluded Pi at the older sites. Hydrological conditions may have influenced the distribution of the Pi forms. The moisture content at the sites that received stormwater for 30 years was lower than the moisture content of the soils collected from the wetlands that have been receiving stormwater for <10 years. When soil is dried, the phosphorus availability is altered. Phosphorus is more available to plants under flooded conditions (Mahapatra and Patrick, 1971; Patrick and Khalid, 1974; Patrick, 1992), therefore if sites are not subject to flooding there may be an increase in occluded Pi.

Another possible explanation for the increase in the occluded P at the sites receiving stormwater for 30 years may be due to weathering. HCl-extractable Pi (Ca-bound Pi) contributed more to the total P pool at the sites receiving stormwater for 30 years. It is thought that the amount of Ca-bound Pi is an indicator of the amount of weathering of the soil (Tiessen et al., 1984). Therefore, because more of the total P pool is made up of Ca-bound Pi at the sites receiving stormwater for 30 years, this may be an indicator that these sites have experienced more weathering. In other words, the P entering the system has been stored over time in occluded forms. Qualls and Richardson (1995) reported that sites nearest the input of P had more total Pi in the Ca-bound fraction. Qualls and Richardson (1995) also found that the Ca-bound P is greatly increased in the areas receiving P enrichment when compared to those sites with no enrichment. Ca-bound Pi was in the greatest concentration at the site with the largest loading rate (Table 3.1). This may indicate that this process is driven by nutrient enrichment.

Cooke et al. (1992) found that the sediment receiving wastewater for a decade had the majority of Pi in the NaOH- Pi fraction. This fraction is associated with oxalate-

extractable Fe. The NaOH- Pi fraction was a small percentage of the TPi for both age classes indicating that the sites investigated here may have low oxalate-extractable Fe concentrations. The NaOH- Pi may also be removed by the previous ($NaHCO_3$-Pi) fraction (Levy and Schlesinger, 1999).

Residual P was found in lower concentrations than previously reported (Cross and Schlesinger, 1995 and Hedley et. al, 1982). Hedley et al. (1982) suggests that because one-quarter of the bacterial cell is non-extractable P, aggregation of residual P can be slowed through the reintroduction of bacteria. The stormwater pumps frequently discharge water into the wetlands, which is a possible new supply to the bacterial population, resulting in the low concentrations of residual P at these sites.

A fractionation was not performed on the P entering the system or the soil in the collection ditches or canals. The possibility that different levels of labile and occluded P entered the systems cannot be ruled out. Since the P fractions in the stormwater being pumped onto the sites were not measured, it is possible that the individual sites received different amounts of labile and occluded P. These forms either react within the system or are buried and stored in the form they entered (Sundby et al., 1992).

CONCLUSIONS

Phosphorus concentrations are higher in the sites receiving stormwater for 30 years when compared to those receiving stormwater for <10 years. Phosphorus is being stored in the soil in labile forms more than the occluded forms. Phosphorus is being stored in both the mudflat and the vegetated areas at all sites. Site differences do exist that may be related to size of drainage basin. The site that had the highest P loading rate had the highest occluded Pi concentration present.

An investigation of the soil in the collection canals and P fraction of the stormwater entering the system should be done in the future. By examining the collection canals, one may understand the transformations of P taking place there and the effect of the P concentration in the receiving wetland.

CHAPTER 4: SOIL METAL CONCENTRATIONS IN LOUISIANA COASTAL WETLANDS AFTER 10 AND 30 YEARS OF STORMWATER INPUT

INTRODUCTION

Coastal Louisiana is experiencing some of the highest rates of coastal loss in the country (Boesch, et al. 1994). There are many restoration projects attempting to slow this loss and to help restore the coast. There are many efforts currently being applied to the land loss issue in Louisiana. These include river diversions, wastewater additions and most recently the investigation of stormwater input as a method of restoration.

There have been several river diversion projects in south Louisiana that had a great deal of success. Caernarvon is a major river diversion that has been the focus of much research. The Caernarvon structure has the capacity to divert 226 m^3s^{-1}. The diversion project at Caernarvon was originally designed to increase freshwater flow therefore decreasing salinity; however because of the increased input of sediment this diversion has created approximately 1.64 km^2 of new marsh (Lane et al., 1999). Freshwater from the diversion lowered salinity levels with no significant negative impacts to water quality (Lane et al, 1999). Wetlands act as transformers decreasing nutrient levels that could pose as sources of eutrophication in open water. River diversions, once used primarily to lower salinity of open water, have been found to be most successful when first introduced to wetlands because wetlands remove nutrients and suspended sediments (Lane et. al, 2001). DeLaune et al. (2003) found that the diversion of the Mississippi River water into the Breton Sound increased the rate of accretion. In light of the positive results of other large-scale restoration projects, stormwater diversions are being explored as another potential method to restore wetlands; however, the ecological impacts of stormwater diversions are poorly understood at this time.

Stormwater is collected through a series of ditches and canals, and ultimately discharged into a drainage canal or directly into a wetland. The primary purpose of these pumps is to relieve the southern parishes of flooding after storms; however, their potential for rehabilitation of the coastal wetlands has been a recent focus.

Through addition of stormwater to a degraded area, a freshwater, nutrient and sediment source should be reconnected to the degraded coastal wetlands therefore, aiding in the rehabilitation of the wetland. However, when adding influent, such as runoff, there is increased concern that there will be an accumulation of toxic metals in the ecosystem.

There have been several studies that focus on wetland's ability to remove metals from water. Scholes et al. (1999) found that constructed wetlands in England and Wales removed between 41% and 85% of metals introduced to the system. The differences in removal rates depended on the metal being removed and the sampling routine (Scholes et al., 1999). Scholes et al. (1999) concluded that during wet weather, wetlands have a large capacity to remove urban pollutants from runoff.

Carleton et al. (2000) focused on constructed wetlands in the Washington D.C. area and found that using constructed wetlands may be a cost-effective method of improving water quality at water detention facilities. Carleton et al. (2000) found that the wetlands were efficient in removal of constituents only when storm volumes were lower than the capacity of the marsh. In this project, Carapeto and Purchase (2000) focused their study on constructed wetland's ability to remove Cd and Pb from urban runoff. The concentrations of Cd and Pb decreased with distance from the runoff source (Carapeto and Purchase, 2000). They found that the wetlands removed higher amounts of Cd during storm events than during dry events.

Walker and Hurl (2002) found that while sedimentation is the primary removal process, biological and chemical processes are also important in the removal of heavy metals. By comparing sites that have been receiving stormwater for different amounts of time, it is possible to determine if long-term addition of stormwater to the wetlands has an affect on the soil metal chemistry. Through depth profiles, a historical record can be formed for the areas that have been receiving stormwater.

The Barataria-Terrebonne estuary lies with the southern portion of Louisiana and is experiencing some of the highest erosion rates of the coast (Louisiana Coastal Wetlands Conservation and Restoration Task Force, 1998). Terrebonne and LaFourche parishes are within the Barataria-Terrbonne estuary. There are currently 256 stormwater pumps located within these two parishes (Richards, 1994) and 80 pumps discharge directly into wetlands. This study focuses on four wetlands that have been receiving stormwater between 5 and 30 years. The sites were selected based on the marsh type, maximum discharge of the pump, and age of the pump. I hypothesis that metal concentrations will be higher in sites that have been receiving stormwater longer.

METHODS

Study Area

All the sites selected are located within the Barataria-Terrebonne Estuary Complex, which is located in the southeastern portion of Louisiana (Figure 4.1). All sites chosen for this study represent degraded coastal marsh systems and have been receiving stormwater between 5 and 30 years. The sites selected have had minimal human impact (i.e. dredging). The land use of the area is primarily rural and agricultural.

Each site was separated into vegetated plots and adjacent mudflat areas. The mudflat areas are seasonally flooded in the late spring and summer months and are then areas of

open water. The pumps discharging into the area have maximum discharge rates between $4020\ L\ s^{-1}$ – $7305\ L\ s^{-1}$. Each pump averages 1-hour day^{-1} at maximum discharge (Blaise LeCompte, personal communication, 2003).

Surface Soils

Soil samples were collected in February 2003 at a 10 cm depth using an aluminum corer. At each site, soil was collected from vegetated areas closest to the pump discharge and the adjacent mudflat region. There were five replicates taken at both the vegetated and mudflat areas each replicate was a composite of three cores. Samples were transported back to Louisiana State University on ice and stored at $4^{o}C$ until processing. A subsample of each sample collected was dried to determine moisture content of the soil. Metal analysis was completed by placing 2 grams wet soil into digestion tubes. The samples were exposed to 1:1 HNO_3 and heated at $95^{o}C$ for 15 minutes. After the samples were cooled to room temperature, 5 mL of concentrated HNO_3 was added. Samples were heated for 2 hours, cooled to room temperature and then 3 mL of 30% H_2O_2 + 2 mL deionized water was added. The samples were heated for an additional two hours, cooled, and concentrated HCl was added. The samples were heated for an additional 15 minutes. Samples were cooled overnight, filtered and brought to volume in a 100 ml flask. Metals were analyzed using inductively coupled argon plasma spectrometry (ICP).

Depth Profiles

Two cores were taken from the vegetated areas of each site. The cores were 15 cm in diameter and sampled to a depth of approximately 45 cm. The cores were transported back to LSU and frozen until analysis was completed. The cores were cut

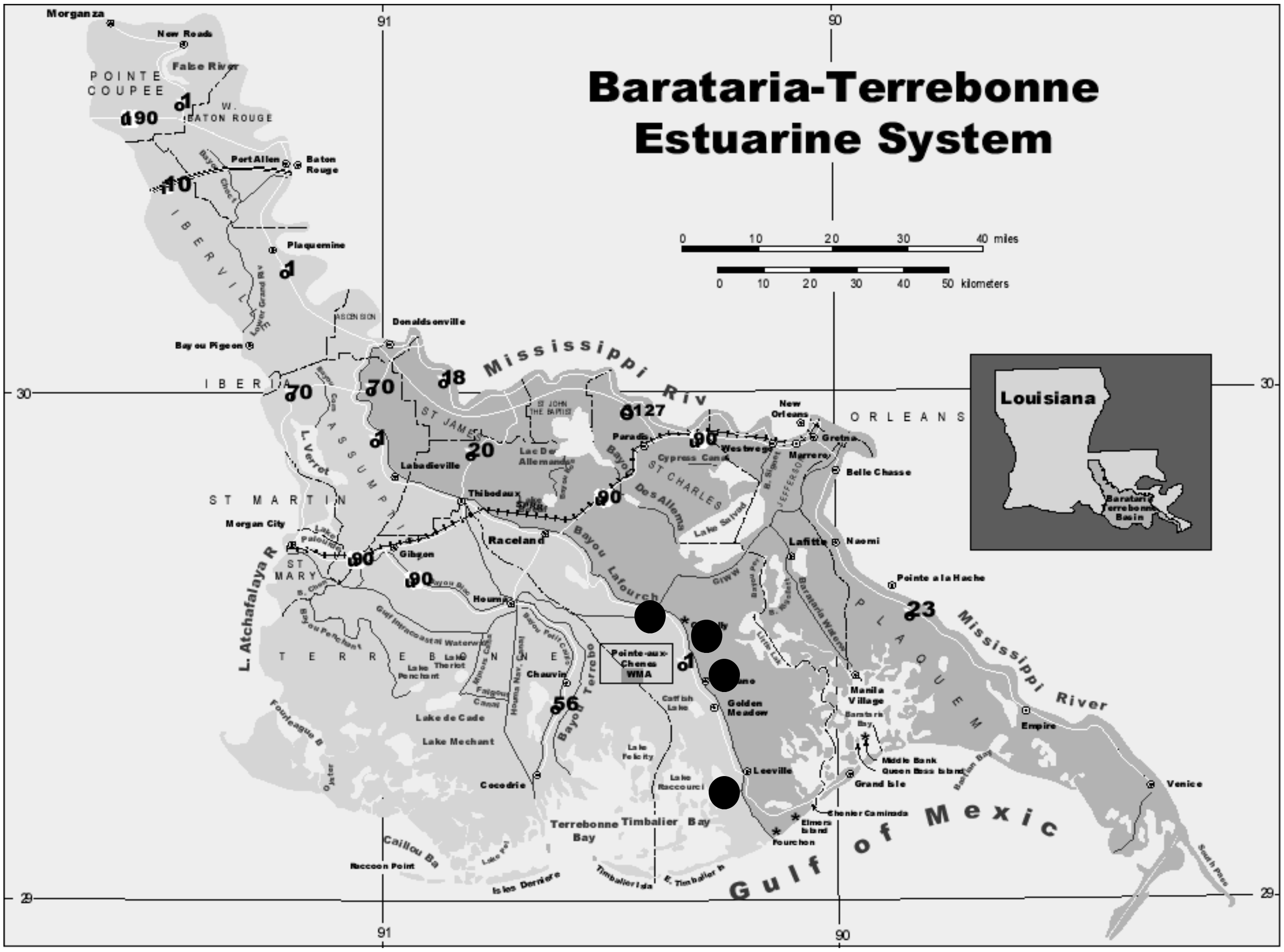


Figure 4.1. Schematic of the Barataria-Terrebonne Estuary. The four black circles are the locations of stormwater pump sites that were selected for comparison in this study.

into 3 cm sections, weighed and dried. After drying, the cores were ground to pass through a 0.5 mm sieve. The core sections were then analyzed for ^{137}Cs content to date the sediment, and 1 gram subsamples of each section were digested using the nitric acid digestion procedure described above.

Statistical Analysis

The general linear model procedure of analysis of variance was used to determine if the age of the pump had a significant effect on the metal concentrations for both the sediment profile and the surface soil samples. It was also used to determine if the depth of the soil had a significant effect on the metal concentrations. A separate general linear model procedure of analysis of variance was used to determine if the mudflat or vegetated samples (treatment) effected the metal concentration and to determine if there was a significant interaction between the age of the pump and the treatment (SAS Institute, 1999).

Metal Loadings

Water samples were collected in August 2003. Samples were collected in acid washed liter bottles. Six liters were collected from each site, three from each pump and three from each receiving basin. The samples were transported back to Louisiana State University on ice and were stored under refrigeration. For metal analysis, 50 mL subsamples were exposed to concentrated nitric acid and heated.

To calculate loading rates to the wetland the discharge was adjusted to allow for one hour at maximum discharge day^{-1} year^{-1} for each pump (Blaise, Lecompte, personal communication, 2003). An areal estimate was not done for any of the sites. To determine the loading rate the concentration of the metals were multiplied by the maximum discharge. This rate of flow provided an estimate of metal concentration

entering the system. The annual load was multiplied by the time the pump had been operational to determine the total metal concentration that has been exposed to the system.

- Annual loading rate = (Max Q*Total metal concentration* 365)
- Total loading rate = annual loading rate * years pump is in operation

RESULTS AND DISCUSSION

Surface Soils

The elements measured in the soil collected at the sites receiving stormwater included Cu, Zn, Cd, Pb, Cr, Ni, Fe, Mn, and Al. Iron was found in the highest concentration in the surface samples in all age groups (12.28 mg g^{-1} to 19.08 mg g^{-1}). While Cd was found in the lowest concentration (0.0 mg g^{-1} - 0.002 mg g^{-1}) (Table 4.1). The wetlands that had been receiving stormwater for <10 years had a significantly higher concentration of Cd than the sites receiving stormwater for 30 years ($p<0.05$).

The loading rates varied greatly depending on the site and metal. D-21, which has received stormwater for <10 years, consistently has high loading rates. Loading rates for each metal can be seen in Tables 4.2 - 4.9. Cadmium was not included because the concentrations found in the water were below detection at all sites. This is an interesting observation because this is the only metal that was significantly different when comparing surface soil of the wetlands that have received stormwater for 30 years with those that have received for <10 years. Because Cd was below the detection in the water collected from the pumps at all sites, it is probable that there is another source of metals influencing the wetlands.

Depth Profiles

The elements measured in the soil collected in the cores included Cu, Zn, Cd, Pb, Cr, Ni, Fe, Mn, and Al. The elements found in the highest concentration throughout the depth profiles are Fe, Mn, and Al (Figure 4.2 (a-c)). Iron had the highest concentration at all depths in both age groups (7.08 mg g^{-1} – 19.57 mg g^{-1}). Callaway et al. (1998) reported that Fe and Mn sediment profiles in flooded areas usually have the highest concentrations in the surface soil and the concentrations decrease with depth. This is not the case in any of the wetlands observed here. Iron is constant with depth in the core from the area receiving stormwater for <10 years, while there are several peaks throughout the sediment profile of the wetlands receiving stormwater for 30 years. The concentration of Fe in the soil is higher throughout the sediment profile in the area receiving stormwater for 30 years when compared to the sediment profile of the area receiving stormwater for <10 years. At approximately 36 cm depth the concentrations at the two sites are similar (Figure 4.2 (c)). Manganese is constant through the sediment profile from the wetlands receiving stormwater for 30 years (Figure 4.2 (a)). Manganese concentration did have an increase in the surface soil of the wetland receiving stormwater for <10 years, but also has a peak in the concentration at a depth of approximately 27 cm. It is not known why these profiles differ from those reported by Callaway et al. (1998).

Zinc, Pb, and Cu are found in moderate amounts throughout the depth profile in both age groups (Figure 4.2 (d-f)). There is an increase in Cu concentration in the sediment profile of the wetlands receiving stormwater for <10 years at approximately the

Table 4.1. Mean surface soil metal concentrations in soil at sites receiving stormwater for 30 years and sites receiving stormwater for <10 years. Concentrations in mg g^{-1} (±SE)

Constituent	-------30 year sites------- Vegetated	Mudflat	---------<10 year sites----- Vegetated	Mudflat
	------------------------------ mg g^{-1}		-------------------------------	
Cu	0 (0)	0 (0)	0.01 (0.01)	0.14 (0.12)
Zn	0.08 (0.08)	0.05 (0.03)	0.16 (0.08)	0.32 (0.21)
Cd	0.00 (0.00)*	0.00 (0.00)*	0.001 (0.01)*	0.002 (0.01)*
Pb	0.01 (0.01)	0.01 (0.01)	0 (0)	0.02 (0.01)
Cr	0.02 (0.02)	0.01 (0)	0.01 (0)	0.01 (0)
Ni	0.03 (0.01)	0.01 (0)	0.01 (0)	0.03 (0.01)
Fe	19 (7)	12 (3)	12 (1)	17 (2)
Mn	0.43 (0.21)	0.32 (0.12)	0.40 (0.11)	0.64 (0.23)
Al	7 (2)	8 (2)	7 (1)	8 (2)

* indicates the age of the pump has a significant effect on the concentration of the metal

Table 4.2. Cu concentrations, annual loading rates and total loading rate.

Pump ID	Max Q ($L\ s^{-1}$)	Concentration ($ug\ L^{-1}$)	-------loading rates------- Annual ($kg\ year^{-1}$)	Total (kg)
D-08[1]	4020	0	0	0
D-19[1]	6031	0	0	0
D-21[2]	4020	0.06	3.16	15.8
D-56[2]	7305	0	0	0

[1] Operated 30 years
[2] Operated <10 years

Table 4.3. Zn concentrations, annual loading rates and total loading rate.

Pump ID	Max Q ($L\ s^{-1}$)	Concentration ($ug\ L^{-1}$)	-------loading rates------- Annual ($kg\ year^{-1}$)	Total (kg)
D-08[1]	4020	0.001	0.052	1.56
D-19[1]	6031	0.038	3.01	90.3
D-21[2]	4020	0.079	4.17	20.8
D-56[2]	7305	0.414	39.73	357.7

[1] Operated 30 years
[2] Operated <10 years

Table 4.4. Pb concentrations, annual loading rates and total loading rate.

Pump ID	Max Q ($L\ s^{-1}$)	Concentration ($ug\ L^{-1}$)	-------loading rates------- Annual ($kg\ year^{-1}$)	Total (kg)
D-08[1]	4020	0	0	0
D-19[1]	6031	0	0	0
D-21[2]	4020	0.235	12.4	62.1
D-56[2]	7305	0	0	0

[1] Operated 30 years
[2] Operated <10 years

Table 4.5. Cr concentrations, annual loading rates and total loading rate.

Pump ID	Max Q ($L\ s^{-1}$)	Concentration ($ug\ L^{-1}$)	-------loading rates------- Annual ($kg\ year^{-1}$)	Total (kg)
D-08[1]	4020	0	0	0
D-19[1]	6031	0.002	0.15	4.75
D-21[2]	4020	0.013	0.68	3.43
D-56[2]	7305	0	0	0

[1] Operated 30 years
[2] Operated <10 years

Table 4.6. Ni concentrations, annual loading rates and total loading rate.

Pump ID	Max Q ($L\ s^{-1}$)	Concentration ($ug\ L^{-1}$)	-------loading rates------- Annual ($kg\ year^{-1}$)	Total (kg)
D-08[1]	4020	0	0	0
D-19[1]	6031	0	0	0
D-21[2]	4020	0.063	3.32	16.6
D-56[2]	7305	0	0	0

[1] Operated 30 years
[2] Operated <10 years

Table 4.7. Fe concentrations, annual loading rates and total loading rate.

Pump ID	Max Q ($L\ s^{-1}$)	Concentration ($ug\ L^{-1}$)	-------loading rates------- Annual ($kg\ year^{-1}$)	Total (kg)
D-08[1]	4020	2.01	106	3185
D-19[1]	6031	0.64	50.7	1521
D-21[2]	4020	2.76	145	728
D-56[2]	7305	0.40	38.3	345

[1] Operated 30 years
[2] Operated <10 years

Table 4.8. Mn concentrations, annual loading rates and total loading rate.

Pump ID	Max Q ($L\ s^{-1}$)	Concentration ($ug\ L^{-1}$)	-------loading rates------- Annual ($kg\ year^{-1}$)	Total (kg)
D-08[1]	4020	2.69	142	4262
D-19[1]	6031	0.45	35.6	1069
D-21[2]	4020	2.92	154	771
D-56[2]	7305	0.51	48.9	440

[1] Operated 30 years
[2] Operated <10 years

Table 4.9. Al concentrations, annual loading rates and total loading rate.

Pump ID	Max Q ($L\ s^{-1}$)	Concentration ($ug\ L^{-1}$)	-------loading rates------- Annual ($kg\ year^{-1}$)	Total (kg)
D-08[1]	4020	0.30	15.84	475
D-19[1]	6031	0.33	26.15	784
D-21[2]	4020	2.19	115.6	578
D-56[2]	7305	0.65	62.3	561

[1] Operated 30 years
[2] Operated <10 years

same time that the pump was installed (Figure 4.2 (e)). However, there is a rapid decrease in concentration and is found at very low concentrations near the surface. None of the other constituents measured seem to show a marked increase in concentration associated with the installation of the stormwater pump. As can be seen in Figure 4.2 (a-i), with the exception of Cu, the metal concentrations do not have any peaks associated with the addition of stormwater to the area. Nickel, Cr, and Cd are found in the lowest concentrations throughout the depth profile in both age groups (Figure 4.2 (g-i)). Cadmium was found in the lowest concentration (0.001 mg g^{-1} – 0.002 mg g^{-1}).

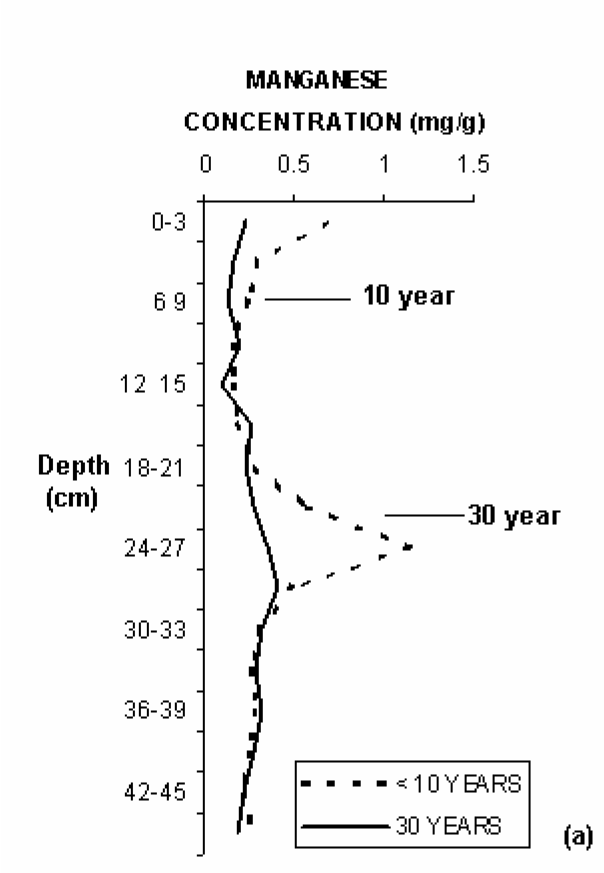
The metals present at the sites are being stored in the soil, however, if there is a change in the hydrology or pH of the area the metals present could be released (Gambrell, 1994). DeLaune et al. (1981) characterized Airplane Lake as a pristine marsh in south Louisiana. Cadmium, Mn and Cu are all found in higher concentrations in the soil in both age groups when compared to the concentrations reported by DeLaune et al. (1981). Iron is the only metal that falls within the range previously reported. Zinc, and Pb have peaks that exceed the findings by DeLaune et al. (1981), but do not consistently exceed the concentrations throughout the profile. The depth profiles of this study are higher than Airplane Lake throughout all depths; therefore it is not clear if the addition of stormwater is what is influencing the metal concentration of our sites.

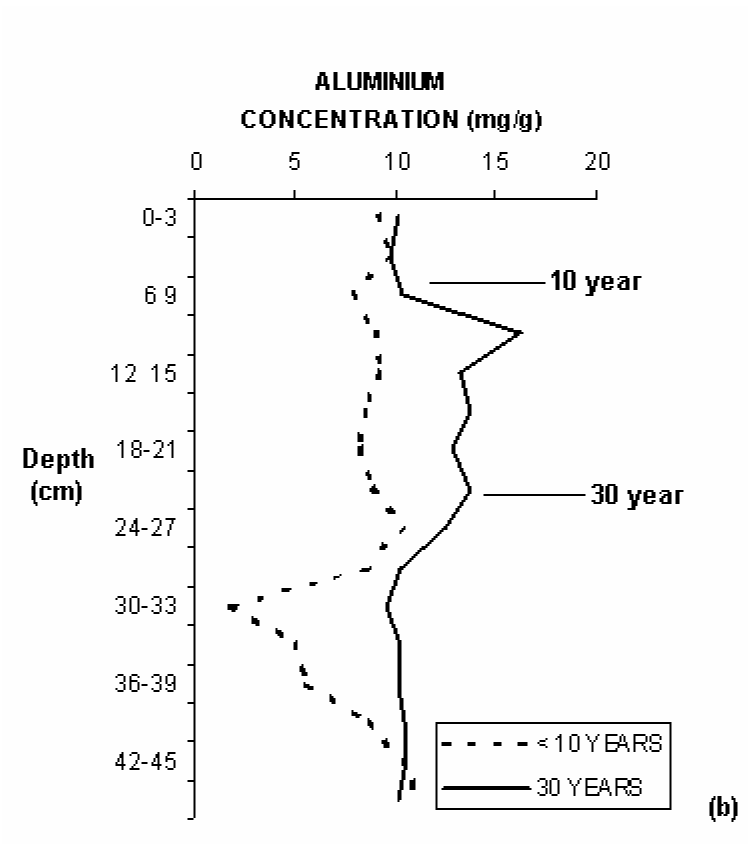
^{137}Cs dating indicated that the sites that have been receiving stormwater for 30 years have had an accretion rate of 0.75 cm yr^{-1} and that the sites receiving stormwater for <10 years have been accreting at a rate of 1.05 cm yr^{-1} (Figures 4.3 and 4.4). The accretion rates established through the ^{137}Cs dating were used to locate the approximate date of pump installation in the sediment profile (Figure 4.2 (a-i)). After estimating the point in the depth profile that the pumps were installed, it was possible to compare

accretion rates before and after pump installation to determine if the pump had a significant impact on the accretion rate of the sites. By conducting a t-test in SAS, it was determined that the pump did not have a significant impact on accretion rates at the sites (SAS Institute, 1999).

Zinc concentrations were significantly different when comparing the cores collected at the sites receiving stormwater for 30 years with those receiving stormwater for <10 years ($p<0.05$). Lead, Fe, and Al are also significantly different when comparing the two age groups ($P<0.005$). The concentrations of these metals are consistently greater in the sites receiving stormwater for 30 years throughout the depth profile. There is not an increase associated with the installation of the pump. The loading rates for Zn, Pb, Al and Fe are all higher in the sites receiving stormwater for 10 years than the sties receiving stormwater for 30 years (Tables 4.3, 4.4, 4.7, and 4.9). Therefore, it is not known at this time if the stormwater is the cause of the significant difference seen between age groups.

Pardue et al. (1988) found that areas that have high accretion rates could mask the increases in metal concentrations. The concentrations in the sediment profile may be diluted by incoming sediment in areas that have high accretion rates (Pardue et. al, 1988). Since the sites examined in this study did not have high enough accretion rates (2.0 cm yr^{-1}), it is unlikely that the metal concentrations were diluted by incoming sediment (Pardue et al. 1988).





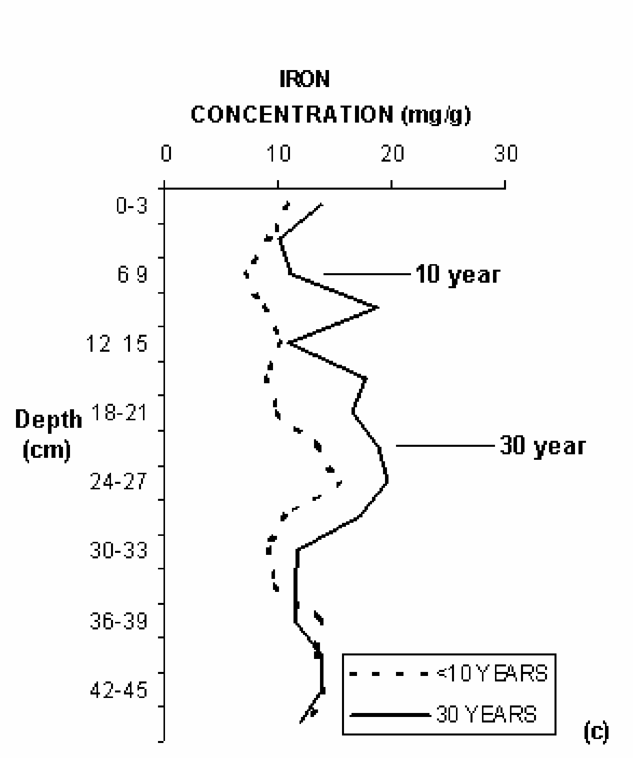


Figure 4.2 (a-c). Depth profiles of Mn, Al and Fe. Dotted line is the soil that received stormwater for 10 years and the solid line received for 30 years. Time is marked in profile.

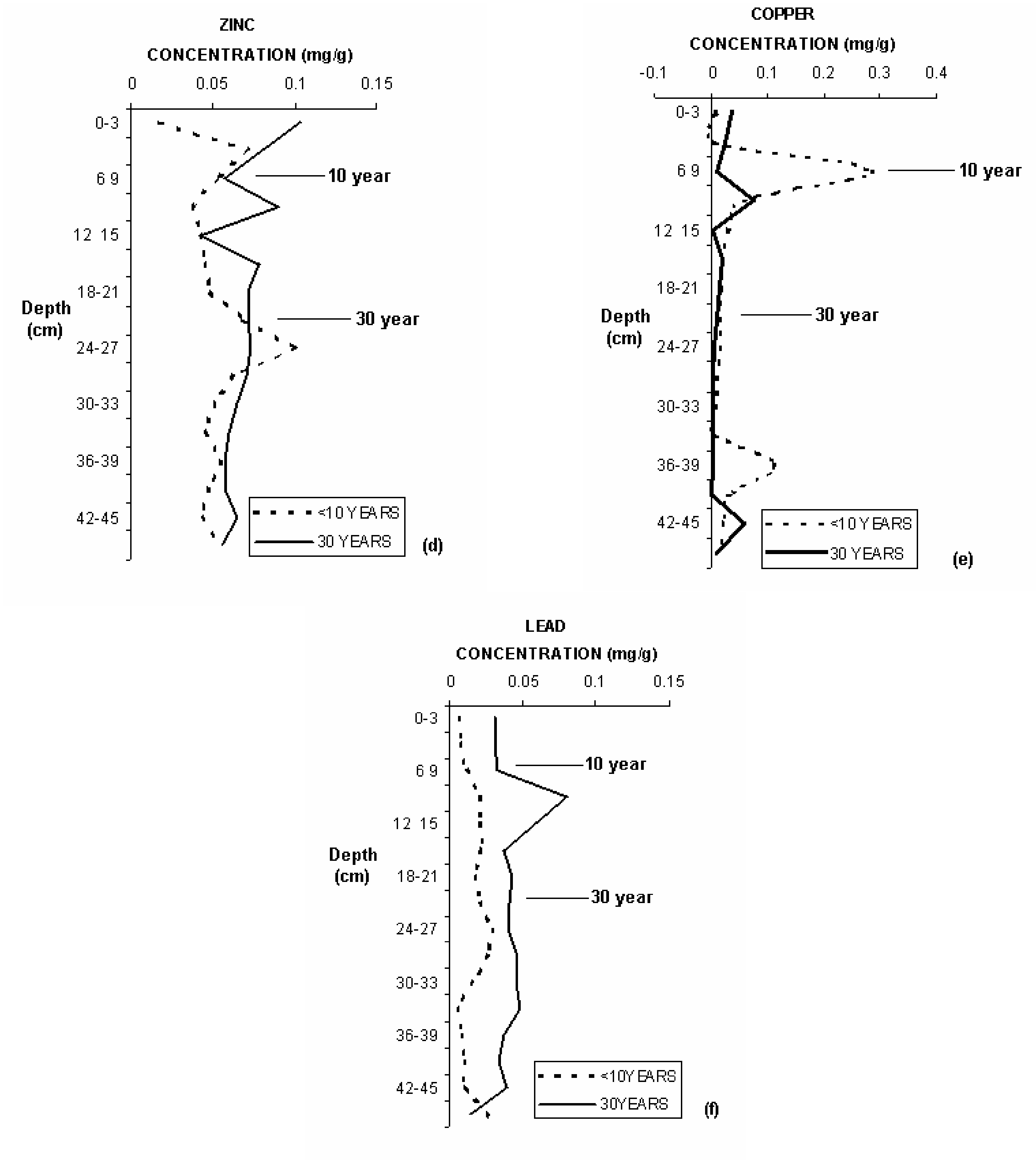


Figure 4.2 (d-f). Depth profiles of Zn, Cu and Pb.

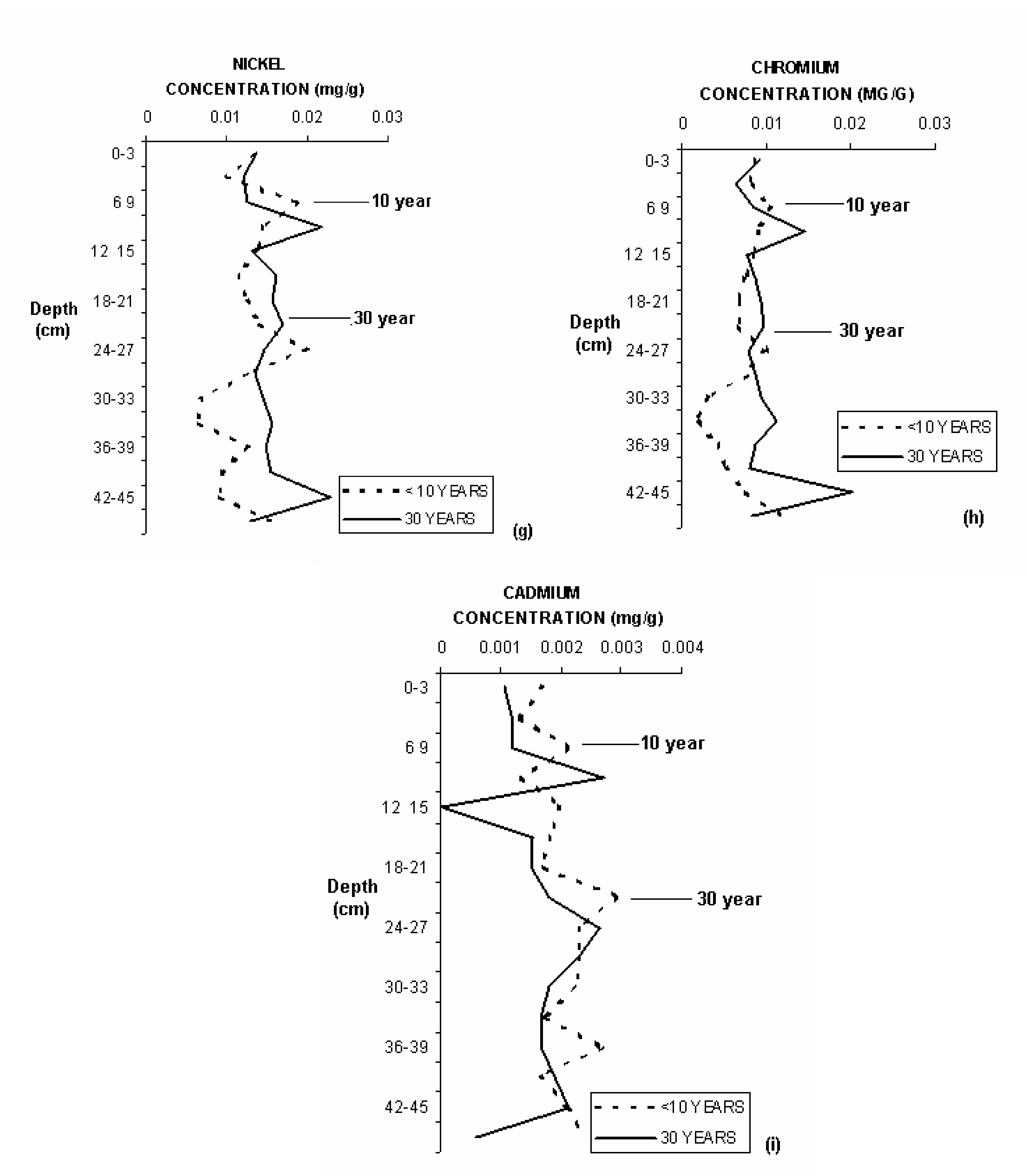


Figure 4.2 (g-i). Depth profiles of Ni, Cr and Cd.

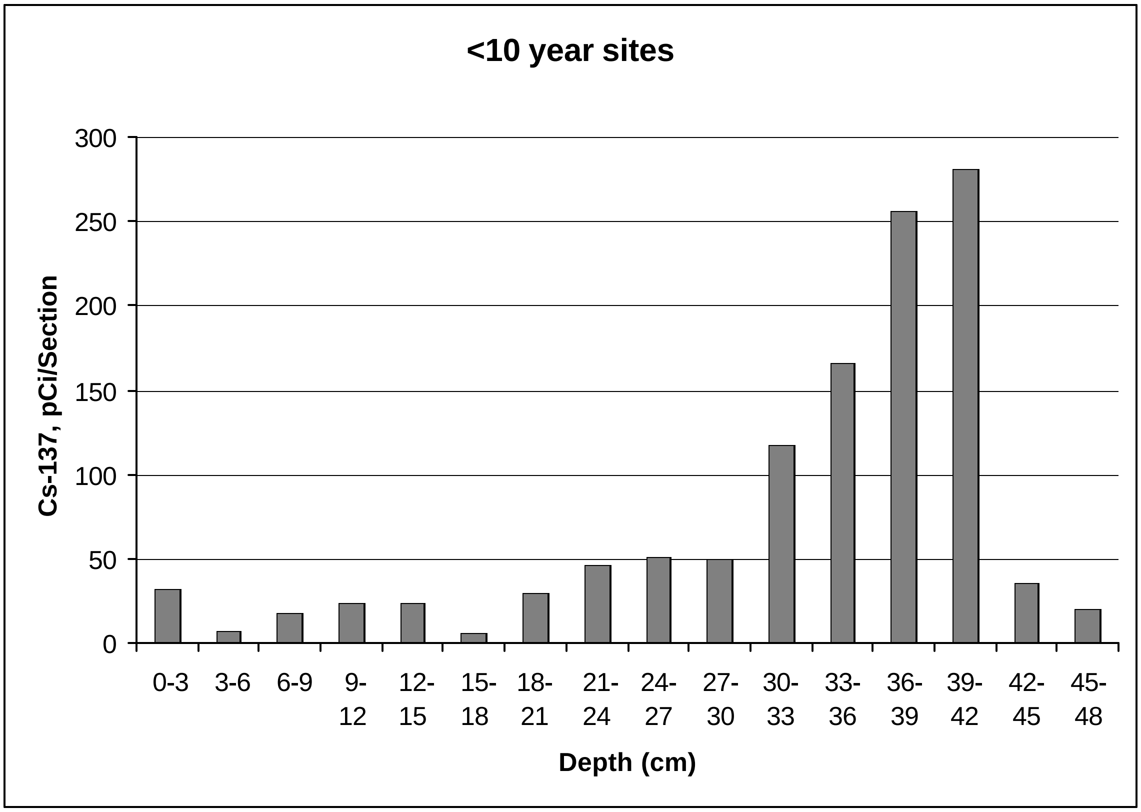


Figure 4.3. ^{137}Cs activity in a depth profile of sediment collected at the sites receiving stormwater for <10 years. The peak at depth 39-42 indicates the year 1963. 1963 is used because this was the end of nuclear testing therefore, the ^{137}Cs activity diminishes after this time. Using the depth of the 1963 peak and the time elapsed since 1963, an approximate accretion rate can be calculated. The accretion rate for the <10 year old sites is 1.05 cm yr^{-1}.

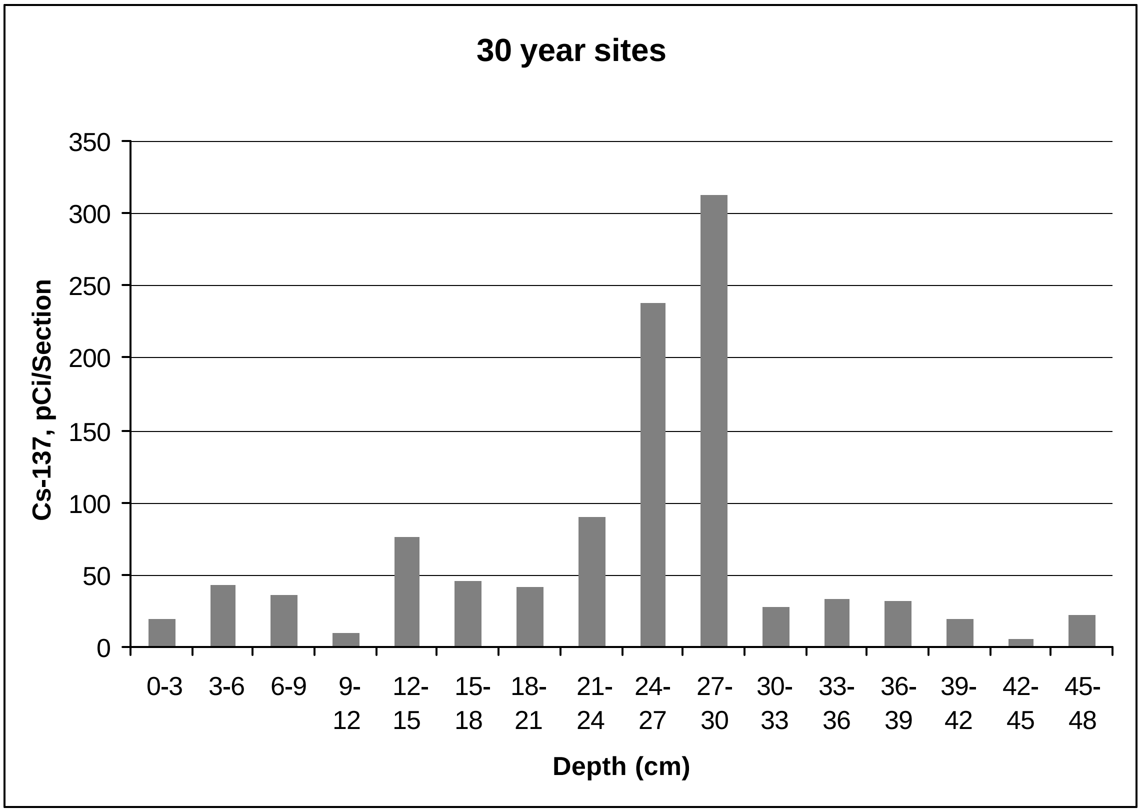


Figure 4.4. ^{137}Cs activity in a depth profile of sediment collected at the sites receiving stormwater for 30 years. The peak at depth 39-42 indicates the year 1963. 1963 is used because this was the end of nuclear testing therefore, the ^{137}Cs activity diminishes after this time. Using the depth of the 1963 peak and the time elapsed since 1963, an approximate accretion rate can be calculated. The accretion rate for the 30 year old sites is 0.75 cm yr^{-1}.

All of the metals measured at the sites that have been receiving stormwater for 30 years have a peak at approximately the same depth (9-12 cm) (Figure 4.2 (a-i)). There is also a peak at the sites that have been receiving stormwater for 30 years at a depth of 42-45 cm. The depth correlates to approximately 1988-1991. The sediment profile indicates that at these sites, there was an event that caused the metal concentration to increase. This peak in metal concentration may be due to a large storm event such as Hurricane Andrew (1992).

When introducing municipal stormwater to an ecosystem, one area of concern is metal concentrations. Metal concentration in sediment has been the focus of several studies to determine the biological effects of introducing metals to an ecosystem. Metal toxicity to meiofauna and plants has been an area of recent research. Meiofauna can uptake metals not only from the water column but also from the concentrations present in the sediment (Millward et al, 2001). Once ingested by the meiofauna the metals may become incorporated into the food chain. Plants can uptake metals and have been the focus of research for use in the treatment of wastewater. While plants can remove metals from sediment, their growth is inhibited at higher concentrations (Srivastav et al, 1994).

Through these studies and others like them, standards have been set for metal concentrations in the soil. After comparing the metal concentrations in the cores collected from the stormwater sites with the standards set by USGS for sediment quality, it was determined that the concentrations for all the heavy metals examined fall well below the standards set. The standards set by the USGS for the EPA reflect probable effect concentrations (PECs). Any concentration above the set PECs is likely to be harmful to the ecosystem (U.S. EPA, 2000). Table 4.10 shows the average metal concentrations found at the stormwater pump sites compared to the PECs.

Table 4.10. Average metal concentrations from stormwater pump sites compared to the probable effect concentrations (PECs) determined by the USGS. Concentrations above PECs are considered harmful to the ecosystem. All concentrations are in mg kg^{-1}.

Metal	10-year sites	30 year sites	PECs
Cd	2.01	1.60	4.98
Cr	7.27	9.76	111
Cu	41.2	18.0	149
Pb	16.0	41.3	128
Ni	12.5	15.4	48.6
Zn	52.5	68.5	459

CONCLUSIONS

There was not a significant difference in surface soil metal concentrations when comparing the age groups. Cadmium was the exception and was significantly different between age groups, however Cd was higher in the sites that have been receiving for <10 years when compared to the sites that have been receiving stormwater for 30 years. There was a significant difference in the concentrations of Zn, Pb, Fe, and Al in the depth profiles when comparing age groups. The concentrations of these metals are higher in throughout the sediment profile and do not decrease with depth. The metals present in the soil of all the sites are well below the established standards for sediment quality set by the EPA. Through examining the sediment profile, it does not appear that there is an increase in metal concentration following the installation of the stormwater pumps. Further studies need to be conducted to determine the impact of metals to the area.

CHAPTER 5: SYNTHESIS

After evaluating areas that have been receiving stormwater for different lengths of time, I found that the areas that have been receiving stormwater for 30 years have a higher concentration of P in the soil than those areas that have been receiving stormwater for <10 years. Further investigation will need to be done to determine if the controlling factor in P accumulation is time exposed to stormwater or the loading rates or both. One of the sites that had received stormwater for 30 years also had a drainage area three times larger than the other sites. It is probable that the increased drainage area directly affected the loading rate and P concentrations measured in the soil.

Chiang et al. (2000) concluded that the cumulative addition of P was more important than the annual loading rates in determining how the ecosystem will respond. Long term loading of P altered plant communities in the Everglades, causing increased productivity and changes in species composition (Chiang et al, 2000). Species composition was not formally analyzed in this study, however, through visual observation, plant communities appear to be similar at the different age groups. Richardson and Qian (1999) determined that the average North American wetland could assimilate P when the loading rate remained below 1g m^{-2} yr^{-1}. Once the loading rate is elevated above this, there is the risk of altering the plant community. The P concentration in the stormwater is comparable to levels found in Mississippi River water; therefore this research can be compared to the results found by Lane et al (1999) at the Caernarvon freshwater diversion site. A total phosphorus loading rate of 0.9 - 2.0 g m^{-2} yr^{-1} was reported for the Caernarvon diversion, however there were not any adverse effects associated with the P addition reported (Lane et al, 1999). The next step in this

research should be to determine the areal extent of the wetland impacted, and determine the assimilating capacity.

^{137}Cs dating the cores made it possible to estimate where in the profile the stormwater pumps were installed. There were no peaks in metal concentrations associated with any of the metals at any of the sites at the time of pump installation. There was a significant difference in the Fe, Zn, Al, and Pb concentrations ($P<0.05$) when comparing the sediment profile from the different age groups, however, the concentrations were higher before the pump was installed, therefore, indicating that the differences between age groups may be site variation. The loading rates of Fe, Zn, Al and Pb were higher in the sites receiving stormwater for <10 years than the sites receiving stormwater for 30 years, which may indicate that there is another source of metals to the sites.

By investigating sites that have historically received stormwater we can begin to understand the differences in long-term effects (30 years) and the short-term effects (<10 years) of stormwater addition to the soil chemistry. Through ^{137}Cs dating and examining the metal concentrations of a sediment profile, it is possible to determine the historical record of that area. There does not seem to be any increase in metal concentration related to the stormwater pump installation date. The baseline data gathered at PAC will be another useful tool in determining the effect stormwater addition has on soil chemistry. To our knowledge there has not been a study of this kind before. By determining the soil chemistry of PAC before the stormwater pump is installed, it can be compared to the soil chemistry after pump installation. This will give a more direct answer as to the effect the stormwater addition has on soil metal and nutrient chemistry. This data along with the study of areas receiving stormwater for 30 years and <10 years will be used in

determining the direction of future studies of stormwater additions for use of wetland restoration.

Pointe Au Chien is designed to investigate the effect distance from the pump has on concentration of nutrients and metals in the soil. Walker and Hurl (2002) found that concentrations of metals decreased with distance from the source. Qualls and Richardson (1995) also found that there was a decrease in nutrient concentration with increasing distance from the enrichment source. Based on these studies, I hypothesis that a similar gradient will be present at PAC. The sites closest to the pump will have higher concentrations of nutrients and metals than sites further away.

While it is expected to see an increase in nutrient concentration once the pump is installed, there is not reason to anticipate a marked changed in accretion. The mudflat areas, which have been classified as unconsolidated and lacking a viable seed source, will have to be replanted or reseeded for these areas to change to vegetated areas. ^{137}Cs dating does not indicate that there is an increase in accretion rates at the sites receiving stormwater for 30 years compared to those receiving stormwater for < 10 years. Vegetated terraces may be necessary near the pump discharge to aid in the collection of sediment.

Many future studies need to be conducted to determine the efficacy of using stormwater as a restoration method. Further investigation needs to be done in the drainage basin itself. Transformations of nutrients and metals may be taking place throughout the collection ditches that lead to the pump. Seasonal trends should be investigated along with the behavior of the wetland following a storm event.

Currently there are two methods of discharging stormwater in the Terrebonne and LaFourche parishes in Louisiana. One is to discharge the stormwater directly into

wetlands and the other is to dig a drainage canal for the stormwater that will ultimately discharge the water into the Gulf of Mexico. Based on the data presented here, I conclude that stormwater discharge at the four wetland sites examined is not negatively impacting the metal concentrations in the soil. These areas are receiving a freshwater and nutrient source. I would recommend that no action be taken to alter the current practice of stormwater addition to these wetlands. However, it is important to note that before stormwater is added to other wetlands an evaluation of the drainage area is necessary. Land use directly relates to the nutrient and contaminant concentrations in the stormwater and wetlands may become overloaded. This approach for wetland restoration/stormwater treatment should be evaluated on a case by case basis.

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VITA

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