TIDAL MARSH MITIGATION IN THE OGEECHEE RIVER ESTUARY, GA: SHORT AND LONG TERM CHANGES

by

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(Under the Direction of Merryl Alber)

ABSTRACT

Short and long-term changes were examined in Georgia tidal marshes to evaluate mitigation progress. Water conditions, sedimentation, and vegetation were monitored in a former rice impoundment before and after removal of hydrologic restrictions. Water conditions (water level, salinity, pH, dissolved oxygen) improved immediately, but after 2 years vegetation cover remained low and sedimentation rates were extremely high. A GIS was used to compare channel density, order, sinuousity, and edge habitat in aerial photographs of a natural marsh with those in a formerly impounded marsh 50 and 100 years after restoration. Drainage patterns in the restored marsh were highly altered and did not exhibit substantial changes over time. Channel density and edge habitat were highest in the restored marsh, but edge may not equal access to the intertidal area. Natural drainage patterns are not expected to develop in the mitigation site over the long term (100 years).

INDEX WORDS: mitigation, restoration, drainage patterns, impoundment, edge, geomorphology, diked wetlands, sinuosity, channel order, GIS, aerial photographs, estuary, Georgia, rice plantation

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CHAPTER 1

BACKGROUND

Coastal wetlands in the southeastern United States, particularly salt marshes, have been well studied with regard to their importance in providing both fish and wildlife habitat and critical environmental and water quality functions. Although most studies have focused on salt marsh habitats, lessons learned from these environments may be generally applied to brackish marshes as well. It is estimated that 75 % of the commercially important fish and shellfish in the United States have some life stage associated with salt marsh habitats (DeVoe and Baughman 1986). The subtidal creeks and intertidal vegetated community found in salt marshes provide a rich food resource and diverse refuge for aquatic and terrestrial species (Boesch and Turner 1984; Rozas and Reed 1993; Rozas 1995). Although most species found in the marsh are not permanent residents, there are many organisms that can be found in these habitats on particular tidal stages, such as nekton that forage on the marsh surface on the flooding tide, or bird species that are looking for prey on exposed mudflats as the tide ebbs (Kneib 1984). There are also many organisms that use these habitats for specific life stages. Numerous fish species, such as spot (*Leostomus xanthurus*) and silver perch (*Bairdella chrysoura*), are abundant in marsh habitats during their juvenile or larval stages (DeVoe and Baughman 1986).

Historically, the functions of coastal wetlands were not recognized, and as a result a significant portion of marshes in the United States have been destroyed or degraded. Within the last century, Baca and Kana (1988) estimate that over half of the southeast's tidal wetlands have been destroyed due to a combination of development and natural causes (such as sea level rise) with loss rates as high as 185,000 ha/yr. It is only in the last few decades that research has provided evidence for the value of wetland functions (Mitsch and Gosselink 2000) and that laws have been passed to protect marsh habitat from further eradication. The Clean Water Act was enacted in 1977; section 404 of the Act restricts the destruction of a marsh habitat unless another functionally equivalent marsh is created or restored to mitigate for its loss. This is a complicated objective because of the difficulty in determining whether a functionally equivalent marsh has been created or restored when the complexities of marsh processes are not always well understood.

The general goals of restoration, creation, and enhancement are to reestablish natural processes, which in theory should lead to the restoration of marsh function (Simenstad and Thom 1996). Many studies have found that allowing natural processes to dictate the speed of restoration is preferable to complex engineering and construction (Mitsch and Wilson 1996; Zedler 2001). Additionally, focusing on residual areas of previously functioning marsh is generally more successful than constructing new marsh habitat because it takes more time, effort, and knowledge to create viable new habitat. In a wetland restoration project in New York, allowing the natural colonization of marsh vegetation through the natural dispersal of propagules was more successful than attempting to mimic colonization through various planting methods (Brown and Bedford 1997). However, a more active role may be necessary if no propagules are nearby or if the natural processes are degraded (Simenstad and Thom 1996).

Some problems in the field of restoration ecology are the determination of success or failure, what indicators should be used to make these determinations, how often these goals are met, and how long it takes to reach them. What defines a success story when it is often not possible to return a wetland habitat to its previous status or function? Usually the most appropriate alternative is to remove modifications, reestablish natural processes and work to achieve defined goals of success identified early in the process. Overall, the success rate of mitigation projects is low and in some cases approved mitigation projects are not even attempted or very little monitoring takes place so success or failure can not be determined (Mitsch and Wilson 1996; National Research Council 2001). Success can be a subjective term and has been defined as occurring when a system sustains viable populations and natural processes or when it is functionally comparable to nearby reference wetlands (Mitsch and Wilson 1996). Some studies have used specific criteria such as a 75% vegetative cover as a measure of success, but have ignored other important factors such as the determination of vegetation species (Mitsch and Wilson 1996). Other studies have compared certain parameters, such as water levels and fish use, to nearby reference wetlands that act as standards of function for restored marshes (Burdick *et al.* 1997; Haltiner *et al.* 1997). Critiques of wetland restoration studies stress that an understanding of wetland functioning, permitting sufficient time for natural functions to develop, and allowing for the self-designing capacity of nature should all be considered (Mitsch and Wilson 1996).

Even more difficult than evaluating success is the determination of which organisms or processes should be used to indicate functionality since processes are so interwoven. Most studies examine a suite of processes that are relevant to the particular system to address targeted questions and goals. There are certain indicators that are inherent in marsh habitats that can be used to indicate wetland functionality, including appropriate vegetative community and surface elevations, water quality, development of a drainage channel system, and a proper hydrologic regime (Simenstad and Thom 1996; Burdick *et al.* 1997; Weinstein *et al.* 1997).

Vegetation change is probably the most well-studied parameter in wetland restoration (Mitsch and Wilson 1996) because the response time is often rapid, changes are easily tracked, and small modifications in vegetative structure can alter faunal assemblages and influence stream migration (Garofalo 1980). It is important to track vegetation because plants provide stability to the marsh surface, they are the basis for the detrital food chain, and they serve as a habitat and food source for numerous species. Vegetation is sensitive to small changes in water levels and salinities and visibly responds to these changes quickly (Burdick *et at.*1997; Niering and Warren 1980). Numerous studies have shown that after tidal restrictions were removed, the vegetative community responded within a couple of years, although more time was often needed for full development of the targeted vegetative assemblage (Simenstad and Thom 1996; Burdick *et al.* 1997; Williams and Zedler 1999). The speed at which wetland vegetation develops into the targeted assemblage depends on local conditions, whether the surface elevation is at the correct level to support vegetation, and whether there is a source of propagules nearby.

Sediment input and marsh surface elevation are important to the understanding of wetland restoration because when an area has been hydrologically restricted, the sediment input decreases and the marsh begins to subside. When a marsh is tidally restricted and drained of water, soil salinities can either increase or decrease (Rozsa and Orson 1993; Haltiner *et al.*1997; Zedler 2001) water tables lower, and the rate of decomposition can increase, which in turn increases compaction and ultimately also decreases marsh elevation (Dent *et al.* 1976; Roman *et al.* 1984; Rozsa and Orson 1993). Subsidence is a common problem in marshes that have had hydrologic restrictions and consequently elevation changes should be monitored when tracking restoration.

Small changes in elevation can result in significant marsh community change or prevent vegetation establishment (Rozsa and Orson 1988; Zedler 2001). In some cases, it may be better to allow the reintroduction of tidal water to occur slowly so that the system can adjust sedimentologically to the new tidal prism. If restrictions are removed all at once, an open water habitat may be created, vegetation will be submerged, and the peat will be waterlogged (Rozsa and Orson 1988).

The utilization of newly restored marsh habitats by birds, nekton, and macroinvertebrates is often indicative of a functioning wetland habitat. This may occur within days of simply reducing tidal restrictions (Simestad and Thom 1996; Burdick *et al.* 1997; Raposa 2002). Studies have shown that after hydrologic restrictions are removed, densities and species richness of epibenthic species and fishes can reach levels similar to reference marshes within 2-5 years, depending on the characteristics of the particular system (Simenstad and Thom 1996). Again, the amount of time needed to see nekton using these restored habitats depends on whether similar habitats nearby support these species. Visual observations of nekton and bird presence/usage can be important, but it may be more informative to examine foraging success and growth rates of specific, targeted species to get information on the productivity and functional value of the restored habitat (West and Zedler 2000).

The presence and health of biological communities is dependent on the water quality of coastal wetlands. Hydrologically restricted wetlands often exhibit variable dissolved oxygen levels and higher nutrient concentrations that create additional stresses on species (DeVoe and Baughman 1986; Brockmeyer *et al.* 1997). Stagnant water resulting from decreased circulation may lead to stratified water conditions in marshes, creating a bottom later of anoxic water with high sulfide levels (Brockmeyer *et al.* 1997). Monitoring of salinity levels is an additional important water quality parameter because the distribution of vegetation and organisms is influenced by small changes in salinity.

The formation of a network of drainage channels is critical in the development of a marsh because they serve as conduits of exchange between the marsh surface and the estuarine environment (Weinstein *et al.* 1997). Horton (1945) demonstrated theoretically that stream order is related to the number of stream segments, the lengths of these individual stream segments, and drainage basin area through simple geometric relationships. When these relationships are plotted on a semilogarithmic scale, they plot as a straight line and appear to have similar ranges of values. The stream channel calculations were adapted to salt marshes by Strahler (1952), who found similar relationships in tidal channels. Zeff (1999) compared natural marshes from various scientific studies and provided evidence for similar drainage characteristics (bifurcation ratios, sinuosities, and channel numbers) among natural marshes. Conclusions from these studies indicate that drainage composition may be used as an indicator of overall marsh function. There are some restoration examples where the tidal range was only partially restored, resulting in inadequate flooding regimes and an undeveloped drainage system (Burdick *et al.* 1997; Sinicrope *et al.* 1990). A successful restoration of the Gog-Le-Hi-Te wetland in Washington saw a change in the drainage channels from a first order system to a fourth order system in 3 years (Simenstad and Thom 1996). These changes are important to study in the early stages of restoration efforts because hydrologic development is usually seen within months (Burdick *et al.*1997) although it may take 20 years or more to reach a fully developed drainage system (Williams and Orr 2002). Some studies have shown that reestablishment of tidal hydrology can result in restoration success with little or no human manipulation (Brockmeyer *et al.* 1997).

The time needed to restore a habitat to a successfully functioning system can be highly variable and depends on the degree of modification and the specific functions of interest. More rapid success is seen in habitats with less severe damage and those that have sources of natural propagules nearby (Zedler and Callaway 1999). In general, changes in hydrologic processes and fish utilization occur within days whereas plant community changes and development occur within months to years (Burdick *et al.* 1997). Elevation changes and peat development occur on times scales of years to decades (Burdick *et al.* 1997). For many restoration studies, major changes are usually seen after 10 years. Most recent evaluations of restoration projects indicate that long term monitoring is critical for documenting the restoration of highly degraded habitats and in marsh creation studies.

The most successful examples of marsh restoration often proceed by restoring hydrologic conditions that restrict or prevent tidal flow. These restrictions come from diking of wetlands to prevent flooding in developed areas or from the construction of roads or dams that prevent or restrict tidal exchange between adjacent wetland habitats. Marshes have also been impounded for agricultural activities such as rice cultivation and waterfowl hunting. Successful restoration of these degraded habitats occurs by the removal of hydrologic restrictions and allowing natural processes to return. These habitats have therefore become promising restoration sites since restoration is more straightforward and less disturbing than marsh creation methods.

The research presented in this thesis first examines the short term changes that occurred when hydrologic restrictions were removed from an impounded marsh in the Ogeechee River Estuary in Georgia. Monitoring began before restrictions were removed and continued for two years after flapgates installed for historic rice farming operations were removed. Data were also collected in a nearby formerly impounded marsh to determine realistic reference levels of marsh functions. The parameters that were measured include vegetation percent cover, sediment deposition and grain size, and water quality. After three years, conditions appeared to be improving once hydrologic restrictions were removed, as evidenced by reduced variability in water quality and increased sediment deposition. However, vegetation cover is still much lower in the impoundment site than in the reference marsh.

Because it often takes many years for a restored wetland to reestablish natural functions, this study also focused on long-term indicators of marsh restoration success by examining the formation of a channel system in a formerly impounded marsh that was restored more than 50 years ago and compared this information to the existing channel network in a natural marsh system. Drainage parameters such as drainage density, sinuosity, channel distribution, and amount of marsh edge were calculated for the two marshes. There were significant differences between the natural and restored marsh and after a 50 year period, only minor changes had occurred in the restored marsh. Although drainage density was much higher in the restored marsh than in the natural marsh sinuousity was lower in the lower order channels of the restored marsh. Interestingly, the greater amount of marsh edge found in the restored marsh suggests that not edge is created equal.

The observations made in this project are useful for characterizing short term changes that occurred in the impoundment after restoration as well as for predicting the type of changes that may occur over longer time periods. Although there is some evidence of short term recovery that may meet restoration goals (*i.e.* water quality), the tidal creek network analysis suggests that even after 50 years the area will not be fully equivalent to and undisturbed system.

CHAPTER 2

TUCKER MITIGATION SITE SHORT-TERM MONITORING

INTRODUCTION

As part of a mitigation effort by the Georgia Department of Transportation (GDOT), plans were made to restore a hydrologically restricted wetland in the Ogeechee River to functionally replace wetland habitat destroyed in two highway-widening projects by purchasing the Tucker Mitigation Site. Mitigation plans consisted of removing two partly functional flapgates that remained from a historic rice plantation and served to restrict water exchange between the impounded wetland and surrounding areas. Because many mitigation projects are criticized for their brevity and minimal level of involvement, this site was purchased in 1996 with plans to monitor ecological change over a 10-year period. Skidaway Institute of Oceanography was designated to conduct the monitoring and analyses at this site, which consisted of monitoring changes in water conditions, sedimentology, and benthic characteristics combined with the analysis of aerial photographs to examine changes in drainage pattern formation. The data described here represent observations of water conditions, vegetation, and sedimentation, all of which I was responsible for collecting.

STUDY AREA AND METHODS

Study Site

The Tucker mitigation site is located in a brackish marsh in the Ogeechee River Estuary near Savannah, Georgia. Tidal fluctuations of approximately 1.5–2 meters and salinity ranges of 0-10 psu characterize marshes in this area, although elevated salinity levels have been recorded under drought conditions. In the 1800's, this area was used extensively for rice cultivation (Baca and Kana 1988; Sullivan 1999). Dikes were built around the wetland habitats, the trees were removed, and flapgates were installed to provide controlled conditions appropriate for rice cultivation (Sullivan 1999). Once rice culture was abandoned, many dikes fell into disrepair and tidal conditions were eventually restored. However, some impounded areas were maintained for wildlife preserves or hunting grounds, particularly for migrating waterfowl (DeVoe and Baughman 1986; Sullivan 1999). At this particular site channels approximately three meters wide, were excavated around the perimeter of the wetland and the material was used to build the outer dike that impounds the wetland. The rice plantation was subdivided into approximately 20 smaller plots with each plot having dikes on the outer edges and an internal canal system with channels approximately one meter in width. The exterior dikes were recently (in the last 20 years) repaired and maintained, although the internal canal system was allowed to degrade. The area was drained in the summer and flooded in the winter for the migrating waterfowl (GDOT Report 1996). Differences in water levels between the two seasons ranged from summertime lows when water was retained only in the perimeter ditches to wintertime maximum depths of 1.22 meters over the entire area (GDOT Report 1996). Dikes repairs were necessary every 1-3 years because of severe erosional problems exacerbated by flood conditions, particularly for the riverside and southernmost areas of the impoundment.

At the time GDOT purchased the wetland in 1996, two flapgates remained that restricted tidal exchange between the river and the impoundment (Fig. 1). The flapgates are essentially culverts that have covers to prevent water movement in and/or out of the impounded wetland. By this time, the marsh surface had subsided approximately 30 cm, presumably due to reduced tidal exchange (GADOT Report 1996). When monitoring began in 1998 the cover over the culvert was permanently in the open position, the tidal fluctuation averaged 60-70 cm directly at the river flapgate. Sometime in 1996, a breach (henceforth called the large natural breach) occurred near the location of the Valambrosia Canal flapgate on the southeastern dike and by 1999 it was approximately 50 feet wide (Fig. 1). A second breach on the same dike, but closer to the river (henceforth called small natural breach), was discovered in 1999 and is approximately five feet wide. In December 1999, the two remaining flapgates were removed to further reduce restrictions to tidal flow in the impounded area (Fig. 1). There are no current plans to remove other portions of the dike because of the documented historical value of the dike system, although they will not be maintained and will be allowed to naturally deteriorate. Some places on the riverside dike already show signs of overwash.

Impoundment sites monitored by SkIO are being compared to a nearby wetland that is being monitored as a reference site. The reference site is slightly downstream and separated from the impoundment area by Valambrosia Canal (Fig. 1). The reference marsh was also used for rice cultivation in the past, and sparse remnants of the dikes and canals remain. However, it appears to be a functional marsh with tidal ranges characteristic of the area, a developed drainage system, and native vegetation. The data presented here represents observations made between August 1998 and September 2001, although sampling is ongoing.

Water Conditions

Beginning in August 1998, data loggers were deployed to monitor general water conditions at the study site. Instruments were deployed in a small channel near the river flapgate on the inside of the impoundment (Outer Impoundment site) and in a similar area in the reference marsh (Reference site) (Fig. 1). Minisondes (Model 4) from the Hydrolab Corporation monitored conductivity, water temperature, dissolved oxygen, pH, depth, and redox at 15-minute intervals and were deployed for a three-week sampling period every three months (Table 1). Instruments were mounted on steel poles that were vibracored into the sediment in small channels of the impounded area and reference marsh. In October 1999, an additional Minisonde (Model 4a) was purchased to monitor water conditions in the inner impoundment (Inner Impoundment site) (Fig. 1). The Hydrolab at the Inner Impoundment site did not measure pH. This instrument was placed in Peachtree Canal, which runs through the center of the impoundment. In February 2000, the Outer Impoundment Site was moved approximately 50 m further into the canal due to extremely high current velocities flowing through the new opening in the dike created by the removal of the flapgate.

Instruments were calibrated before and after deployment according to Hydrolab manuals. Data retrieved from the instruments were edited if necessary when calibrations were not in a specified range. Some additional editing was necessary when the instrument came out of the water at low tide, which is recorded by one of the sensors.

Vegetation

To study changes in the vegetation as tidal conditions were restored in the area, I established two sets of paired transects in the reference and impounded marshes (in areas that should receive similar tidal input and be of similar elevations) so that changes can be assessed and compared over time (Fig. 2). Reference and Impoundment site B represent the pair of transects located closer to the breach and Reference and Impoundment site R represent the pair of transects located closer to the river (Fig. 2). The transects were 200 meters in length and were sampled at 20 meter intervals for species composition and percent cover in a 0.8 m^2 circular quadrat. Transects were established in October 1999 and were sampled again in July 2000 and August 2001 to track changes in vegetation.

Sedimentology

Grain size

Samples for grain size analysis were collected quarterly from August 1998 – September 2001 (Table 1). Samples were collected with a petite ponar grab sampler at five stations [in creeks at the outer impoundment site, the reference site, the canal that runs in between the reference and impoundment sites (Valambrosia Canal), directly outside the flapgate near the main channel of the Ogeechee River (Ogeechee Embayment site), and the main channel of the Ogeechee River]. Grain size distributions, which show the relative proportions of sand, silt, and clay, are reported at 0.25 phi (ϕ) intervals. The phi scale is a log-based scale used primarily by geologists to aid in mathematical computations of grain sizes (i.e., size (mm) = 2ϕ). Sieves were used to separate the sand fractions (-2ϕ to 4ϕ) and a Sedigraph 5100 was used to analyze the silt and clay fractions (4ϕ - 12ϕ). When the silt and clay fractions constituted less than 10% of the total, samples were not run on the Sedigraph.

Deposition Rates

Sedimentation rates in the impounded and reference marshes were determined by using sediment deposition plates. These studies took place at three locations – inside the reference marsh near SkIO Vegetation Transect B, inside the impounded marsh near SkIO Vegetation Transect B, and in the impounded marsh near the Hydrolab deployment site in the inner impoundment (Fig. 2). At all sites, three large unfinished clay plates (15 cm x 15 cm) were permanently placed at the marsh surface and the thickness of sediment deposition was measured over time. These systematic deposition studies began in January 2001. However, in October 1999, three plates were placed initially in the impounded area near SkIO Vegetation Transect B in order to evaluate the sampling methodology and determine approximate deposition rates inside the impoundment. Information from these plates continued to be collected when the other plates were initiated in January 2001.

RESULTS

Water Conditions

Water depth

The presence of the dike system and culverts resulted in a dampened tidal range in the impounded site. Before the flagates were removed, the tidal ranges at the impoundment site averaged 0.7 m, compared to the reference site, where the tidal range averaged 2.0 m (Fig. 3).

Moreover, canals in the impoundment remained flooded at low tide with approximately one meter of water. Additionally, much of the marsh surface, particularly near the Outer Impoundment Site and away from the large breach had standing water at low tide. Although it was not expected that the canals would drain completely at every low tide, these results indicate that the entire impoundment remained flooded, which has implications for vegetation characteristics (see next section).

The removal of the flapgates in December 1999 resulted in a substantial increase in tidal range at both impoundment sites (Figs. 3, 4). At the Outer Impoundment Site, there was an immediate increase to an average tidal range of 1.3 m. As a consequence, low tide depths at this site increased after March 2000 due to a change in the instrument location. However, high current velocities and associated channel scouring removed the instrument pole and forced a change in study location away from the opening in the dike created when the flapgate was removed. Channel depths are relatively deep in this area due to channel scouring, resulting in greater water retention at low tide compared to the rest of the impoundment.

The response to flapgate removal was smaller and proceeded more slowly at the inner impoundment site, most likely because it is located further from the new opening in the dike. Water depths at the inner impoundment site indicate that it is now draining completely of water at low tides and this is likely representative of most of the impoundment (Fig. 3).

Salinity

The salinity observations were obtained from all three study sites both before and after flapgate removal. Salinity patterns in the impoundment and reference sites were similar, although slightly higher salinity values were observed in the impoundment as compared to the reference site before flapgate removal (Fig. 5). Mean salinities throughout all deployments before flapgate removal averaged 3.0 ± 2.0 psu (mean ± standard deviation) at the outer impoundment site as compared to a mean salinity at the reference site of 2.0 ± 1.9 psu (Table 2). Salinities at the inner impoundment site were much higher than those at either of the other two sites. However, data from this instrument were only available for the two deployments before flapgates were removed. When just these two deployments are considered, the inner impoundment site averaged 4.5 ± 1.9, as compared to salinities of 2.2 ± 1.5 psu at the outer impoundment site and 3.0 ± 1.3 psu at the reference site.

After flapgate removal, the mean salinities at all of the sites increased, including the salinity at the reference site. Mean salinities at the outer impoundment site, the inner impoundment site, and the reference marsh averaged 5.9 ± 4.6 psu, 7.1 ± 5.8 psu, and 5.1 ± 4.6 psu, respectively. The salinities at the outer impoundment site and at the reference site were fairly comparable, which suggests that the increase is due to interannual variation, rather than to flapgate removal. Increased salinities are likely due to the drought conditions that persisted during this period (Fig. 6), as the unusually high salinities in the August 2000-December 2000 deployments coincided with decreased precipitation (total precipitation totaled 44 cm whereas in the previous year it was over 70 cm during the same time period).

After flapgate removal, higher salinity levels were again seen in the inner impoundment site as compared to the other two sites. This may be because the inner impoundment study site is located some distance away from a river culvert and therefore experiences reduced tidal exchange and less dilution of concentrated salts. In contrast, the outer impoundment site received constant flushing from river water moving in and out of the culvert at the flapgate and directly from the river after flapgate removal. An alternative explanation for elevated salinities in the inner impoundment may be due to the influence of the Little Ogeechee River, which tends to have higher salinities than the main channel of the Ogeechee River (J. Blanton, Skidaway Institute of Oceanography, pers. comm.). Aerial photographs indicate strong water movements from the large breach opening, which is directly across from a tidal creek connected to the Little Ogeechee River, towards the location of the inner impoundment site. We have addressed this hypothesis by taking synoptic water samples in August 2000 at various stations in waters believed to be influenced by both river systems, the impoundment, and the reference site (Fig. 7). Salinities in these samples support the idea that more saline water is coming from those canals that lead to the Little Ogeechee River.

pH

Similar pH levels were observed at the reference site and the impoundment site, both before (6.7 ± 0.4 and 6.8 ± 0.3, respectively) and after (7.0 ± 0.3 and 7.1 ± 0.6) flapgate removal (Fig. 8). These values are also in keeping with those measured in other studies of the Ogeechee River (Meyer 1997). Although the Ogeechee River is a black water river and might be expected to be acidic, pH values in the lower Ogeechee River are almost neutral (pH=6.5) due to carbonate inputs (Meyer 1997). In July 1999, the pH dropped to 5.4 in the reference site and 5.8 in the impoundment site. This is most likely due to the large amount of precipitation that occurred in June and July of that year (Fig. 9). Studies have shown that precipitation is more acidic than surface waters, which can cause sudden decreases in pH values following large rainfall events (Ringwood and Keppler 2002). Several spikes in pH were observed in the impoundments before flapgate removal. The causes of this are uncertain, but spikes in pH can be associated with eutrophication and supersaturated oxygen levels (Ringwood and Keppler 2002). Once the flapgates were removed, these spikes were eliminated.

Water temperature

Temperature patterns were similar among the impounded and reference sites with obvious seasonal patterns of lower temperatures in winter and higher temperatures in the summer (Fig. 10). Mean temperature values were similar among all sites before and after flapgate removal (Table 3), although the overall temperature ranges were five degrees greater in the impoundment sites than in the reference site. Before the flapgates were removed, the difference between minimum and maximum temperature ranges was 29.9°C inside the impoundment and 24.2°C at the reference site. This difference can most likely be attributed to the restriction in water exchange, as shallow, stagnant water in the impoundment was likely influenced more by changes in air temperatures with less river water exchange to buffer these changes. After flapgate removal, the temperature ranges inside the impoundment were more similar to the reference site, with maximum difference of 31.2°C and 29.8°C, respectively (Table 3).

Dissolved Oxygen

Dissolved oxygen observations for the study period are shown in Figure 11. Only the first week of each deployment is represented in this data set, as the dissolved oxygen sensors tended to foul easily and compromise their calibrations after approximately seven days in the field. The information gathered from the outer impoundment site before flapgate removal indicates that dissolved oxygen was highly variable in the impoundment site as compared with more stable values in the reference site, although mean values were similar (all range from 5.0 ± 1.2 to 5.6 ±

1.8 mg/L). Dissolved oxygen values in the impoundment before flapgate removal ranged from almost 0 to over 13 mg/l, indicating supersaturation. This wide range of values would provide a harsh environment for many aquatic organisms. Once the flapgates were removed, the variability in dissolved oxygen decreased inside the impoundment and approached reference levels. The standard deviation surrounding the mean value has decreased from 2.6 mg/L before flapgate removal to 1.6 mg/L after flapgate removal. In contrast, the standard deviation surrounding the mean in the reference site averaged 0.7 mg/L throughout all deployments.

Vegetation

The October 1999 vegetation survey was completed several months before flapgate removal. At that time, there was a substantial difference in species composition and percent cover between the reference and impounded marshes (Fig. 12). The two transects in the reference marsh averaged approximately 85 % cover and supported an assemblage of brackish marsh vegetation. The vegetation along the reference transect closest to the river was dominated in both 1999 and 2001 by *Juncus effusus* and *Typha* spp., whereas the vegetation at the reference transect closest to the river was predominantly *Spartina* spp. In contrast, the two transects in the impoundment site averaged 8.5 % cover and when vegetation was present, only *Zizaniopsis milacea* was observed.

Transect data collected in October 2001 reflected an increase in vegetation to approximately 20 % cover inside the impoundment (Fig. 13). There was some recruitment of *S. alterniflora*, *S. cynosuroides,* and *Amamrythus cannabinus* inside of the impoundment. Impoundment Site B, which is located closer to the breach, had greater vegetation cover and likely receives better tidal flushing. Recruitment of *S. alterniflora* was observed at the reference site as well. This is likely due to the high salinity levels from drought conditions.

Sedimentology

Grain size

Grain size distributions were markedly different between the impoundment and the reference site (Fig. 14). At the reference site, the silt and clay fraction composed 9-97 % of the total (Table 4). Bottom sediments at this site consisted of muds interbedded with sand layers of varying thickness, so the variable proportion of sand likely depends on the thickness of the sand lens where the grab sampler hits the bottom. Larger peaks in the sand fraction equate to thicker sand layers. The sediment comprising the silt and clay fraction at this site was well distributed across the range of sediment sizes. Other studies have suggested that sediment in estuaries moves dominantly as flocs (large aggregates of particles of varying sizes) (Milligan 2001). If this is the case, a wide distribution of sediment sizes would be expected in the bed and the sediment would be poorly sorted (Milligan 2001). In the main channels in the impoundment, the sediment was composed mainly of sands, with the silt and clay fraction constituting less than 10% of the grain size distribution. The fine fraction in many cases consisted of degraded organic material with little or no silt and clay. The impoundment study site was in close proximity to one of the flapgates where rapid water exchange would tend to break up flocs or keep them in suspension, either of which would not allow them to settle.

Differential particle transport was also reflected in the main grain sizes and sorting values at the two sites (Table 5). At the reference site, the mean and sorting values were variable depending on the sand component of the sample. Sorting values were higher at the reference site (indicating poor sorting) because particles across the entire distribution range are represented, in contrast to the impoundment where well-sorted distributions (indicated by low sorting values) represent unimodal fine sand.

Suspended sediment

The ranges of suspended sediment concentrations observed in this study (20 – 80 mg/L; Table 6) are in keeping with concentrations observed in Georgia estuaries (Suk 1999). The concentrations observed here were somewhat variable because instrument deployments, and therefore times when water samples were taken, did not always occur at the same stage of the tide. In all deployments before flapgate removal (with the exception of the March 2, 1999 deployment), the suspended sediment concentrations at the reference site were greater than those observed at the impoundment site (Table 6), and averaged 26.9 mg/L in the reference site as compared to 13.1 mg/L in the impoundment site. After flapgate removal, the mean suspended sediment in the reference site increased 36.5 mg/L whereas in the impoundment site it was 28.2 mg/L, and in the last deployment (6/01) concentrations at the two sites were comparable (reference site: 24.7 mg/L, impoundment site: 27.2 mg/L). The increase in suspended sediment in the impoundment site after flapgate removal indicates more particulate material is being delivered to the area.

Deposition from Sediment Plate Deployments

Deposition plates placed in the inner impoundment, at Impoundment Vegetation Transect B, and at the reference site indicate sediment accumulation at all sites, but with more rapid sedimentation occurring inside the impoundment as compared to the reference site (Fig. 15).

The sediment plates placed inside the impoundment in October 1999 suggest sediment accumulation rates of approximately 2 cm/year (Fig. 15). This is encouraging, since elevation levels inside the impoundment are lower than in nearby unrestricted marshes due to subsidence.

DISCUSSION

Before the flapgates were opened to restore tidal flow to the Tucker mitigation site, elevation measurements reported by the Georgia Department of Transportation inside the impoundment indicated that the surface had subsided approximately 0.3 m relative to adjacent marshes. Wetlands that have been impounded for a long period of time often succumb to problems with subsidence when lack of water exchange and therefore sediment input lead to compaction of the marsh surface (Williams and Orr 2002). The resulting increase in water depth likely prevented the establishment of vegetation as the proper elevation may not have been present. Before the breach occurred in 1996, some emergent vegetation was present in the impoundment, but it was predominantly found in areas where elevations were higher, such as where internal dikes were present (GDOT report 1996). Once the breach opened in 1996 and there was only one large opening in the dike, tidal flushing was not adequate to remove the water added to the system over time and the area gradually retained more water, a phenomenon known as tidal choking (Rydberg and Wickbom 1996).

Baseline conditions at the Tucker Mitigation site when monitoring was initiated by SkIO in 1998 indicate that hydrologic restrictions resulted in a highly modified environment compared to the reference site. Despite one opening in the perimeter dike system, tidal ranges at the impoundment site were significantly reduced and the impoundment was inundated during most stages of the tide. Although average water conditions were similar in the impounded and reference sites, the degree of variability around the means was much higher in the impoundment sites, with wide fluctuations in dissolved oxygen, temperature, and pH. In contrast, only the salinity data showed little difference between the reference and the outer impoundment site before flapgates were removed. However, elevated salinities were measured at the inner impoundment site. It is likely that higher salinity levels were found here because this site was located away from any tidal exchange and experienced less dilution of concentrated salts. An alternative explanation, or one that was exacerbated by the difference in flushing rates, is that higher salinity waters from the Little Ogeechee River entered the impoundment near the study site.

Grain size and suspended sediment data showed differences in the sedimentary environment between the impoundment and reference site before the flapgates were opened. At the impoundment, sediment was composed predominantly of sands with little silt and clay, whereas the reference site had greater, although variable, amounts of silt and clay. This was confirmed by the suspended sediment data, which indicated that less silt and clay was suspended in the water column at the impoundment than at the reference site. The lack of particulates in the grain size data and the reduced suspended sediment concentrations before flapgate removal likely contributed to the marsh subsidence in the impoundment.

The documented subsidence of 0.3 m that occurred inside the impoundment resulted in increased inundation that likely prevented vegetation establishment, as described above. The vegetation transect measured by GDOT (See Fig. 2 for transect locations) show a shift between 1996 and 1999 from an emergent vegetation community to one dominated by submerged and aquatic vegetation species. Because the impoundment was inundated at most tidal stages, the emergent vegetation was stressed and in this three year period emergent vegetation was replaced by floating and aquatic species (Fig. 16). Transects 1 and 2 have the highest occurrence of submerged vegetation, which is in accordance with GDOT observations that these transects were completely under water even at extreme low tides. Because these two transects are farthest away from tidal exchange and therefore sediment input, this area may be lower in elevation, which would result in greater water depths. The other three transects do not show as much of an increase in submerged/aquatic vegetation but still show decreased emergent vegetation over the three year period.

The poor water quality and excessive tidal inundation observed in the impounded area likely resulted in harsh water conditions for faunal communities, a situation commonly found in impoundments (Baughman and DeVoe 1986). Shallow water and reduced tidal exchange caused water temperatures in the impoundment to fluctuate, which would be stressful for organisms. Limited water exchange likely reduced the ability of nekton to enter the impoundment and if they were successful at accessing the marsh through the breach or through one of the culverts, they likely became trapped inside the impoundment. Moreover, the fact that little vegetation was present meant that few refuges exist for the nekton that did make it to the impounded area.

There were immediate signs of improved conditions in the mitigation site once the flapgates were opened in 1999. The tidal range immediately increased within days by approximately a meter at the Outer Impoundment site, but took approximately a year to reach reference levels. At the Inner Impoundment site, tidal range gradually increased over a period of a year and a half and was approximately 0.5 m lower than the tidal range found at the Outer Impoundment site and reference site. This is presumably due to the distance of this study site from any openings in the perimeter dikes. As the dike system begins to deteriorate, the range may increase due to better tidal circulation within the impoundment.

Wide fluctuations and spikes in such things as temperature, dissolved oxygen, and pH declined after flapgate removal, although in general, less favorable conditions were observed at the inner impoundment site than at the outer impoundment site due to the close proximity of the latter to the newly created opening in the dike. Spikes in the data before flapgate removal had been eliminated for the most part after tidal circulation improved and had moved closer to reference levels by the end of the study period.

Whereas the tidal range and some of the water quality parameters suggest that conditions are quickly improving in the impoundment site, the sedimentation and vegetation data show that it may take some time for these parameters to reach reference levels. There have been extremely high levels of sedimentation inside the impoundment since flapgate opening and elevation is still increasing. Sedimentation rates continued to be much higher than the reference site throughout the study period. High sedimentation has been noted in other newly restored sites in comparison to their reference sites (Williams and Orr 2002). However, Williams and Orr (2002) noted that sediment deposition rates were typically correlated to stem density (Gleason *et al.* 1979). Since in their study little vegetation was present, they hypothesized that surface sediments were being resuspended due to strong tidal flow on the unvegetated marsh surface. Long term sediment deposition data at the Tucker site indicate that marsh elevation is increasing despite low vegetative cover, but since long term comparative data to the reference marsh is not yet available, it will be important to continue monitoring deposition to ensure that elevation continues to increase.

Although there has been some recruitment of vegetation inside the impoundment since the flapgates opened, a majority of the transect points sampled in this study had no vegetation present, particularly at the site near the river where tidal flushing is not as strong as it is near the large breach. Healthy vegetation that is comparable in composition to reference standards is essential in wetland rehabilitation and restoration. Marsh plant zonation is greatly influenced by minute changes in elevation and problems with vegetation establishment may occur if elevation levels are not sufficient to allow colonization of marsh plants. Examinations of recent aerial photographs and vegetation transect data suggest that the impoundment site is sparsely vegetated, with most vegetation occurring in areas of higher elevations where former internal dikes existed and in areas with significant tidal flushing (and likely higher sedimentation rates). Low marsh elevation can slow vegetation colonization as well as channel development (Cornu and Sandro 2002).

Tidal restoration has been noted to be most successful when hydrologic restrictions are removed to allow the wetland to passively reach reference levels of wetland function (Mitsch and Wilson 1996; Williams and Orr 2002; Zedler 2002). In a highly altered and degraded habitat, such as The Tucker site with its extensive ditching and diking, it may take longer to reach reference levels than what is found in most restoration studies (Mitsch and Wilson 1996). Increased utilization of the marsh by nekton often reaches reference levels in most systems within 2-5 years (Rozas 1988; Simenstad and Thom 1996). However, a longer period of time may be necessary at the Tucker site because of only four possible points of exchange with adjacent marshes. As breaches and flapgate openings increase in size and the perimeter dike begins to deteriorate, the potential for colonization and use of the impoundment should increase. It has been suggested that it takes 15-20 years to determine restoration success in freshwater marshes and that coastal marshes require additional time, possibly up to 50 years (Frenkel and Morlan 1991; Mitsch and Wilson 1996). Most monitoring periods do not even attempt to measure success over this time frame, much less incorporate enough time to ensure that natural disturbances, such as drought, do not upset the development of ecological function. As the Tucker mitigation site has only been monitored now for four years since flapgate removal, it will be important to continue to monitor the progress of this system in comparison to reference areas. Finally, as the Tucker mitigation site was compared to realistic reference levels of a nearby formerly impounded marsh, it will also be important to evaluate if, and how, the marsh functions at the Tucker Mitigation site develop to be similar to those found in natural marsh systems.

Table 1. Water Quality Sampling Schedule. Hydrolabs were deployed at three sites approximately every four months to measure water depth, conductivity, pH, and dissolved oxygen every 15 minutes during a three week deployment. An X indicates periods for which data were collected. The Inner Impoundment Site was not established until October 1999. Gaps in data in 2001 were due to instrument malfunction.

Sampling Dates	Outer Impoundment	Inner Impoundment	Reference Site
August 1998	X		X
November 1998	X		X
March 1999	X		X
July 1999	X		X
October 1999	X	X	X
December 1999	X	X	X
February 2000	X	X	X
March 2000	X	X	X
June 2000	X	X	X
August 2000	X	X	X
October 2000	X	X	X
January 2001		X	
April 2001		X	X
June 2001	X	X	X
September 2001	X	X	X

Table 2. Comparison of Mean Salinities. Mean salinity and standard deviation before and after flapgate removal is reported for the Outer Impoundment and Reference sites. Because the Inner Impoundment site was only established for two deployments before flapgate removal, these data have been excluded from the comparison.

Salinity (psu)	Inner Impoundment	Outer Impoundment	Reference Site
Before flapgate removal			
Mean		3.0	2.0
Standard deviation		2.1	1.8
After flapgate removal			
Mean	7.1	5.9	5.6
Standard deviation	5.8	4.6	4.6

Table 3. Temperature Differences Before and After Flapgate Removal.

Temperature Data (°C)	Outer Impoundment	Inner Impoundment	Reference Site
Before Flapgate Removal			
Mean ± standard deviation	23.4 ± 6.5		21.5 ± 6.3
Maximum	37.4		32.7
Minimum	7.4		8.5
Maximum Variation	29.9		24.2
After Flapgate Removal			
Mean ± standard deviation	22.0 ± 6.4	20.6 ± 6.6	21.5 ± 6.0
Maximum	36.0	36.3	35.2
Minimum	4.9	4.4	5.4
Maximum Variation	31.2	31.9	29.8

Table 4. Comparison of Percent Sand, Silt, and Clay for the Reference and Impoundment Sites.

	Reference Site		Impoundment Site	
Deployment	**Percent Sand**	**Percent Silt and Clay**	**Percent Sand**	**Percent Silt and Clay**
21 Aug 98			99.9	0.1
27 Oct 98	90.5	9.5		
20 Nov 98	59.2	40.8	95.0	5.0
09 Dec 98	25.0	75.1	97.3	2.7
02 Mar 99	82.7	17.3	93.9	6.1
22 Mar 99	33.8	66.2	98.2	1.8
28 Jul 99	43.0	57.0		
04 Jan 00			99.4	0.7
07 Sep 01	2.8	97.2	89.3	10.7

Table 5. Grain Size for the Reference and Impoundment Sites. Data from the Impoundment and Reference sites broken down into the mean and sorting values for each deployment.

Grain Size (phi)	Reference Site		Impoundment Site	
Deployment	**Mean**	**Sorting**	**Mean**	**Sorting**
21 Oct 1998			2.6	0.6
27 Oct 1998	1.4	2.6		
20 Nov 1998	4.4	4.3	2.5	0.6
09 Dec 1998	7.1	3.7	2.5	0.5
02 Mar 1999	2.4	3.3	2.5	0.6
22 Mar 1999	6.84	4.3	2.4	0.5
28 Jul 1999	5.74	4.4		
04 Sep 00			1.8	0.7
07 Sep 01	10.3	2.5	2.7	2.7

Table 6. Comparison of Mean Suspended Sediment Measurements for the Reference and Impoundment Sites. Insufficient replication of samples prohibited the report of standard deviations with the means.

Deployment	Reference Site (mg/L)	Outer Impoundment Site (mg/L)
7/9/98	53.8	14.9
8/21/98	29.0	23.9
9/10/98	28.8	12.5
12/9/98	23.4	17.3
3/2/99	13.0	13.9
3/22/99	14.0	9.0
7/28/99	26.3	6.1
8/8/00	48.3	29.2
6/1/01	24.7	27.2

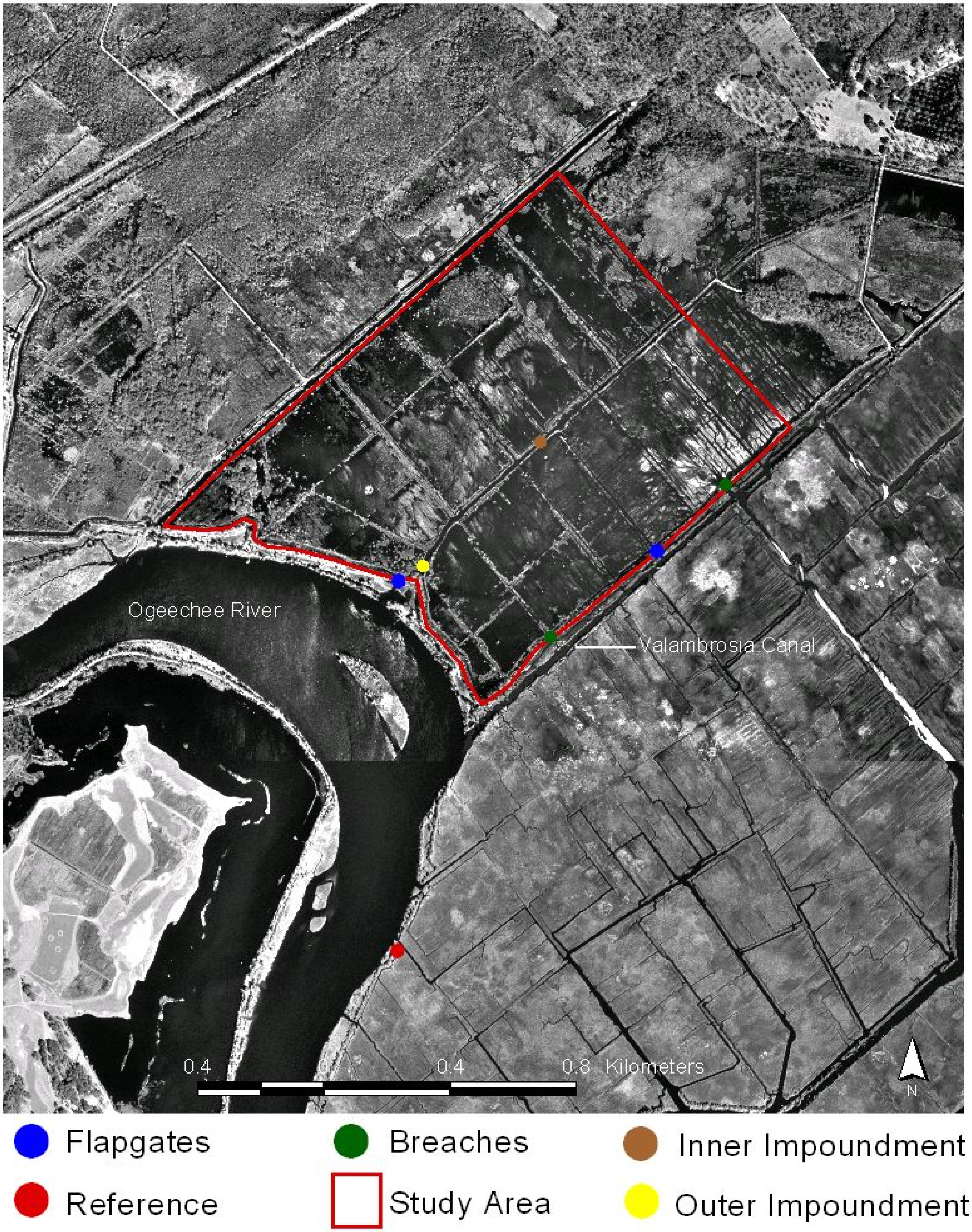

Figure 1. Study Sites at the Tucker Mitigation Site. This figure depicts the Tucker Mitigation site and the three study sites (Reference Site, Outer Impoundment Site, and the Inner Impoundment Site) where monitoring measurements were taken.

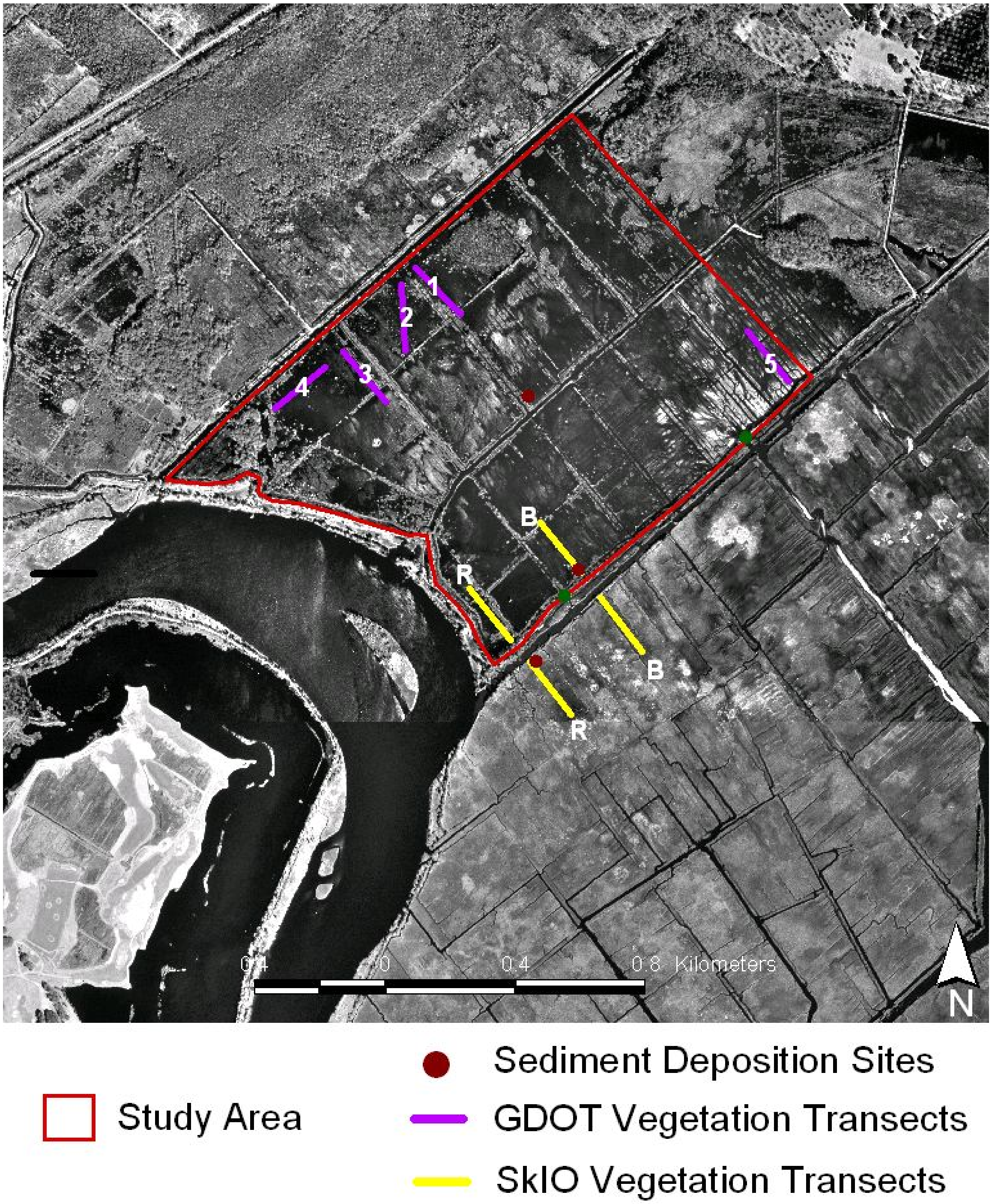

Figure 2. Sampling Sites at the Tucker Mitigation Site. Four vegetation transects were established to compare vegetation cover and species in the impoundment and in the reference site before and after flapgates were opened. GDOT established five transects inside the impoundment in 1996 to study vegetation change inside the impoundment over time. Sediment deposition sites were established to monitor sediment accumulation inside the impoundment and compared to measurements taken at the reference site.

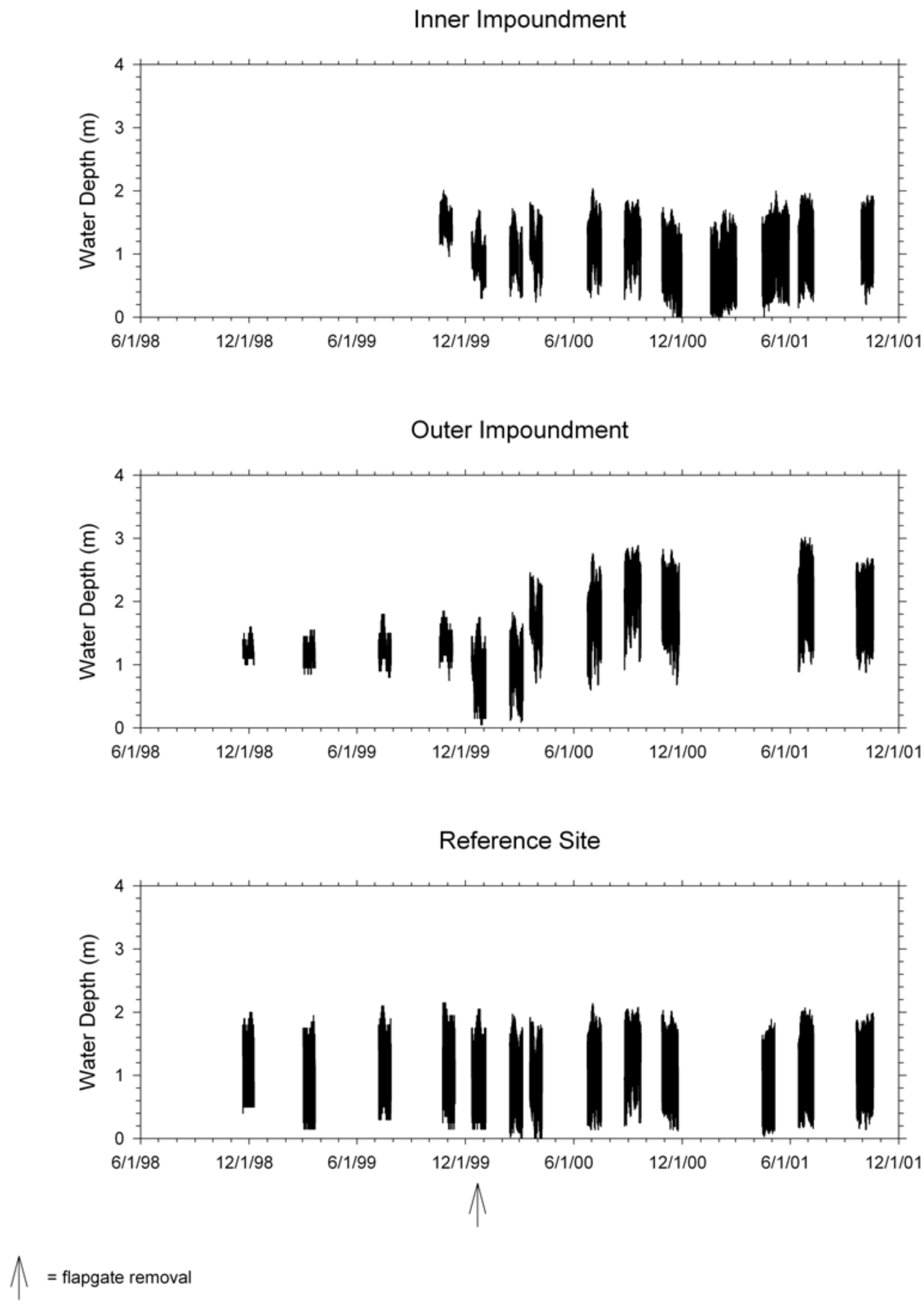

Figure 3. Comparison of Water Depths at the Outer Impoundment, Inner Impoundment, and Reference sites. Data was taken with Hydrolab Minisondes, which recorded depth measurements every 15 minutes. The arrow depicts when flapgates were removed. Note that during some of the early deployments at the reference site, the full tidal range was not captured because the instrument came out of the water during low water.

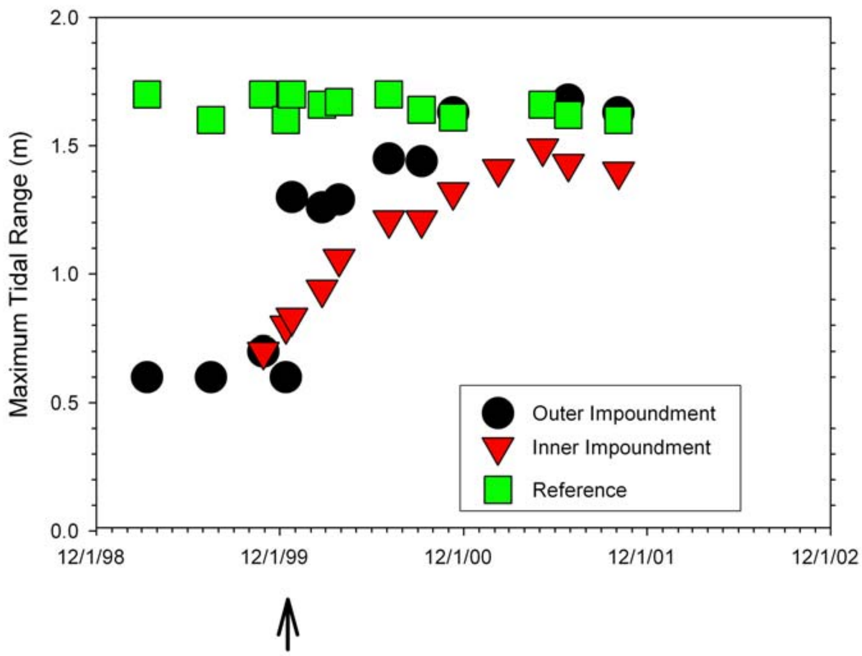

Figure 4. Comparison of Tidal Ranges at the Outer Impoundment, Inner Impoundment, and Reference sites.

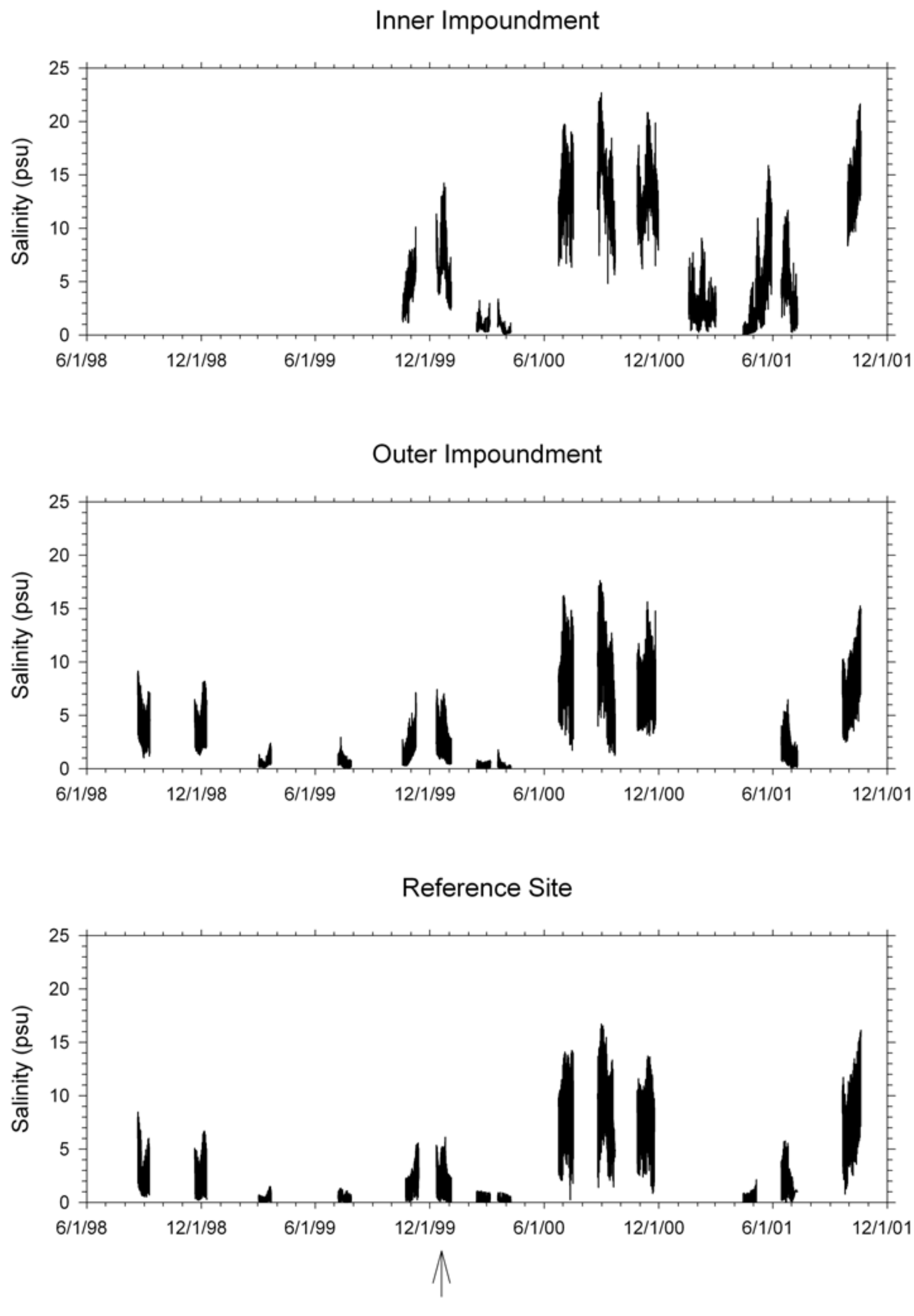

Figure 5. Comparison of Salinity at the Outer Impoundment, Inner Impoundment, and Reference sites.

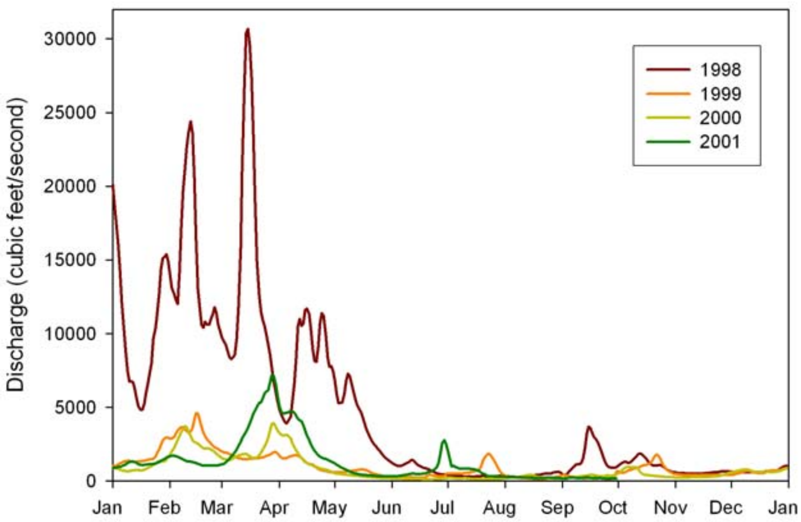

Figure 6. Daily Streamflow Averages for the Ogeechee River. Discharge data for 1998-2001 at the USGS station at Eden, Georgia.

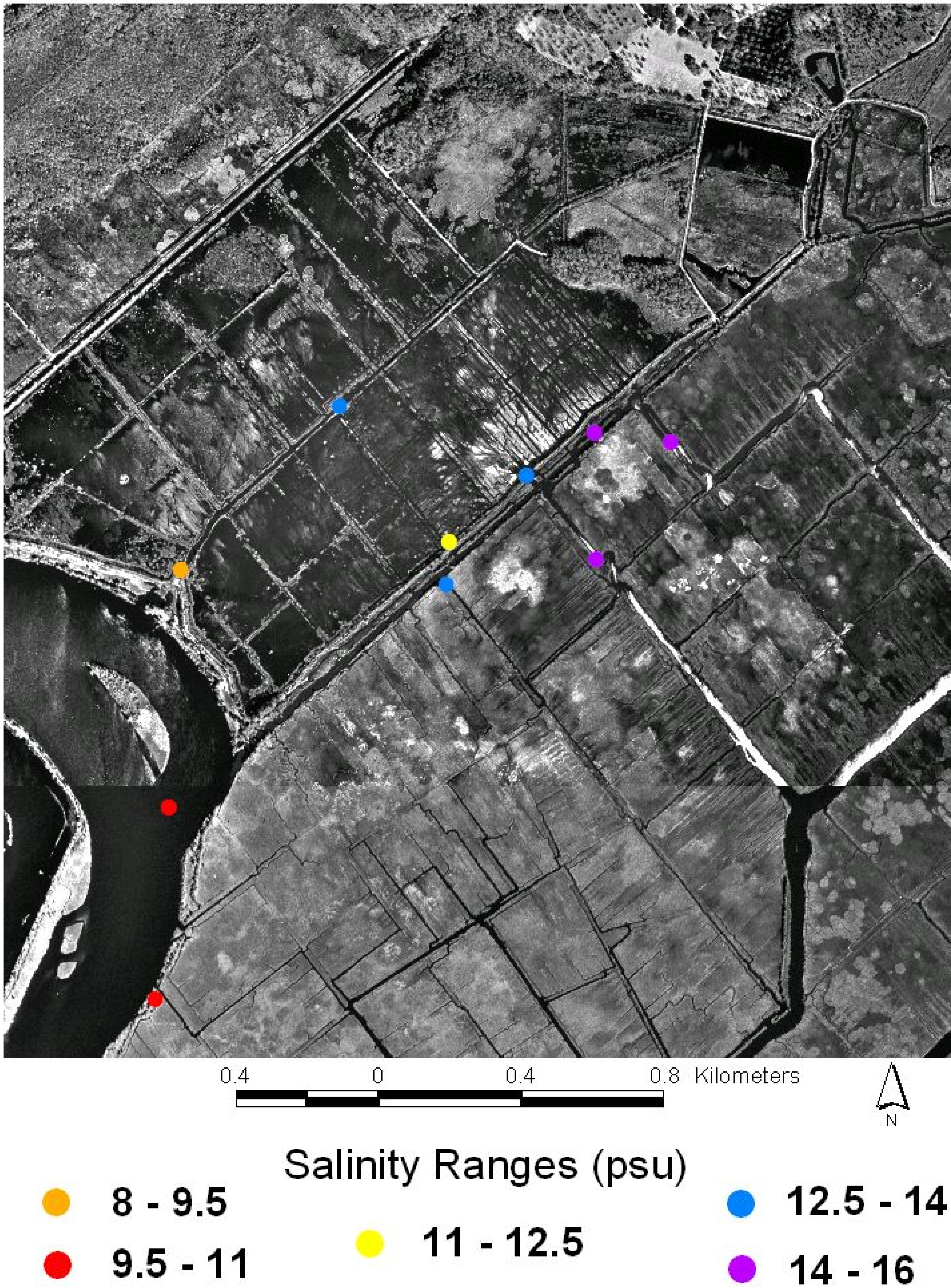

Figure 7. Salinity Survey of Tidal Creeks Surrounding the Tucker Mitigation site. Salinity measurements were taken at stations in the Ogeechee River, inside the impoundment, and in the surrounding canals to determine salinity influences inside the impoundment from the Ogeechee and Little Ogeechee Rivers.

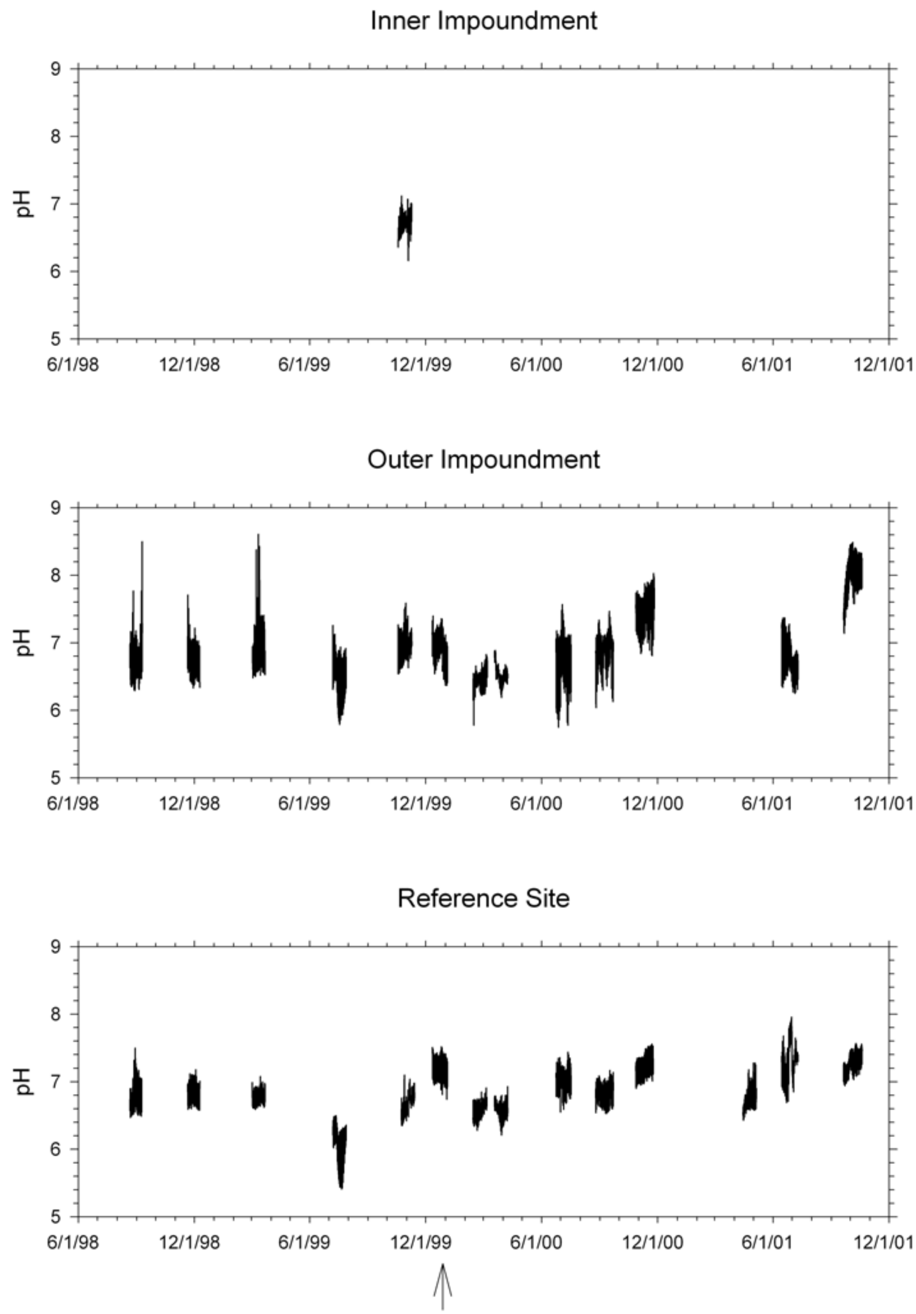

Figure 8. Comparison of pH at the Outer Impoundment, Inner Impoundment, and Reference sites. Data was taken with Hydrolab Minisondes, which recorded pH measurements every 15 minutes. The lack of data at the Inner Impoundment site is because the instrument did not have a pH sensor. The one deployment where pH was measured was because a replacement instrument from Hydrolab was used while the Inner Impoundment Minisonde was being repaired.

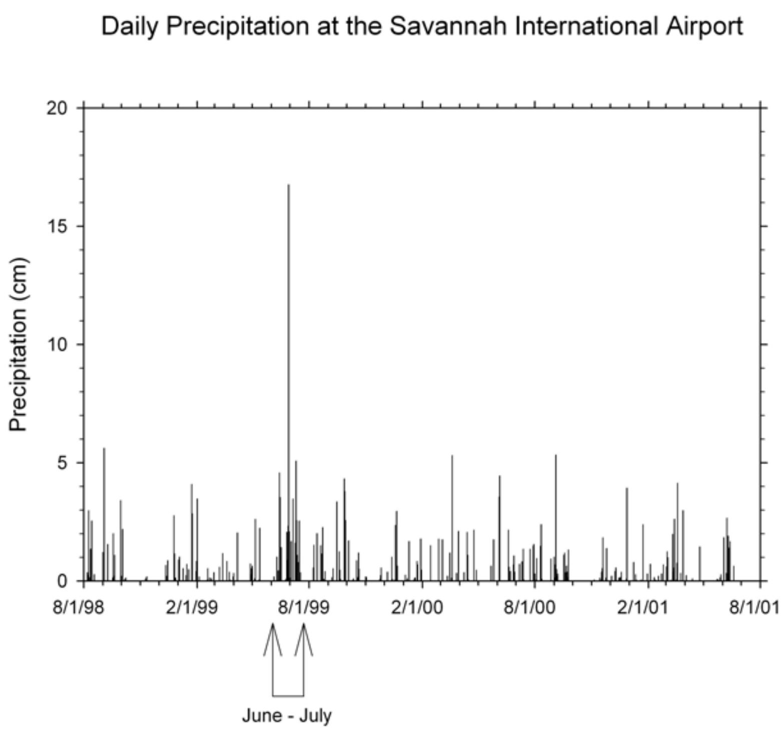

Figure 9. Daily Precipitation Data from the Savannah International Airport. Note the high rainfall that occurred in June and July of 1999, which resulted in low salinities and pH.

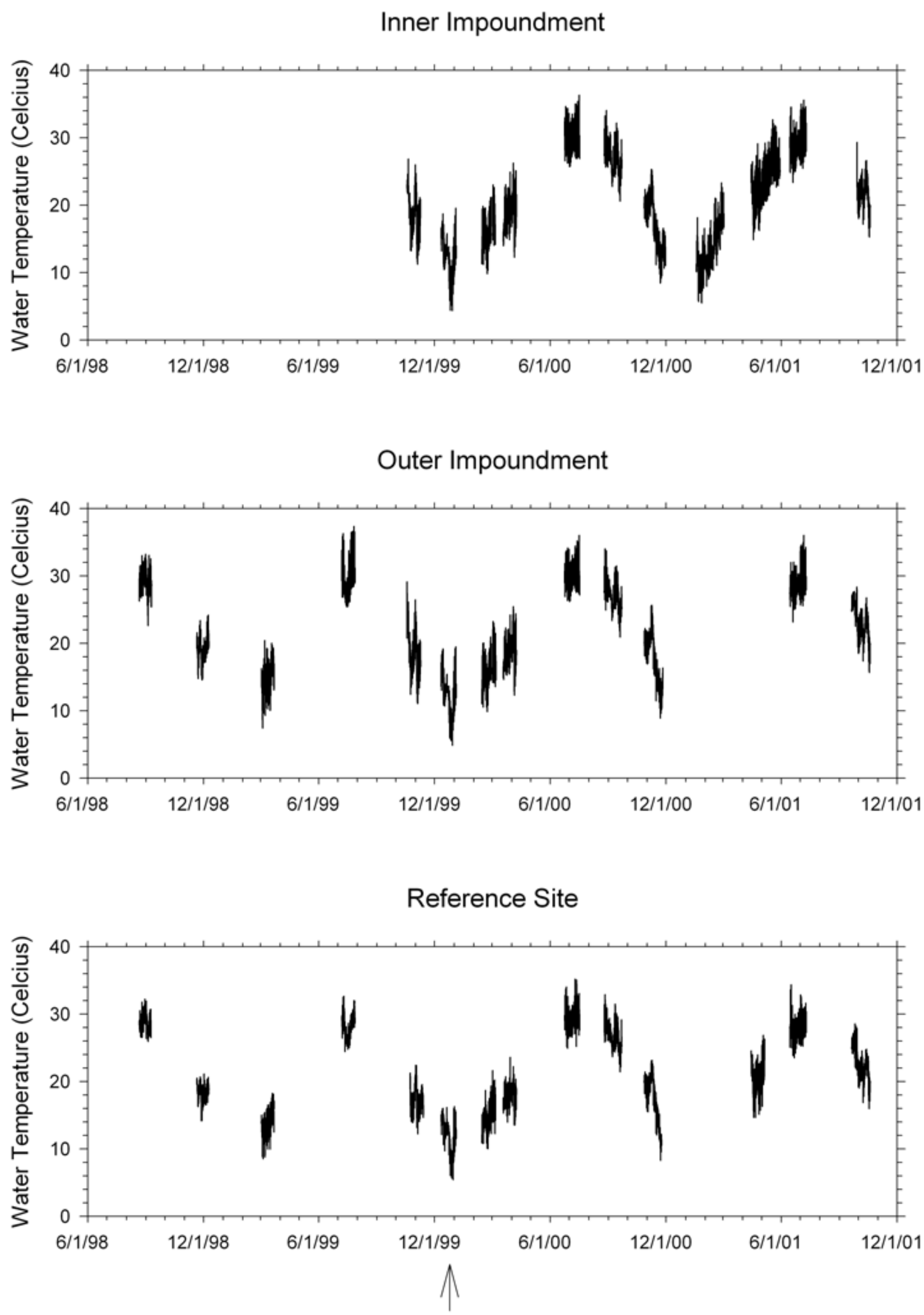

Figure 10. Comparison of Water Temperature at the Outer Impoundment, Inner Impoundment, and Reference sites.

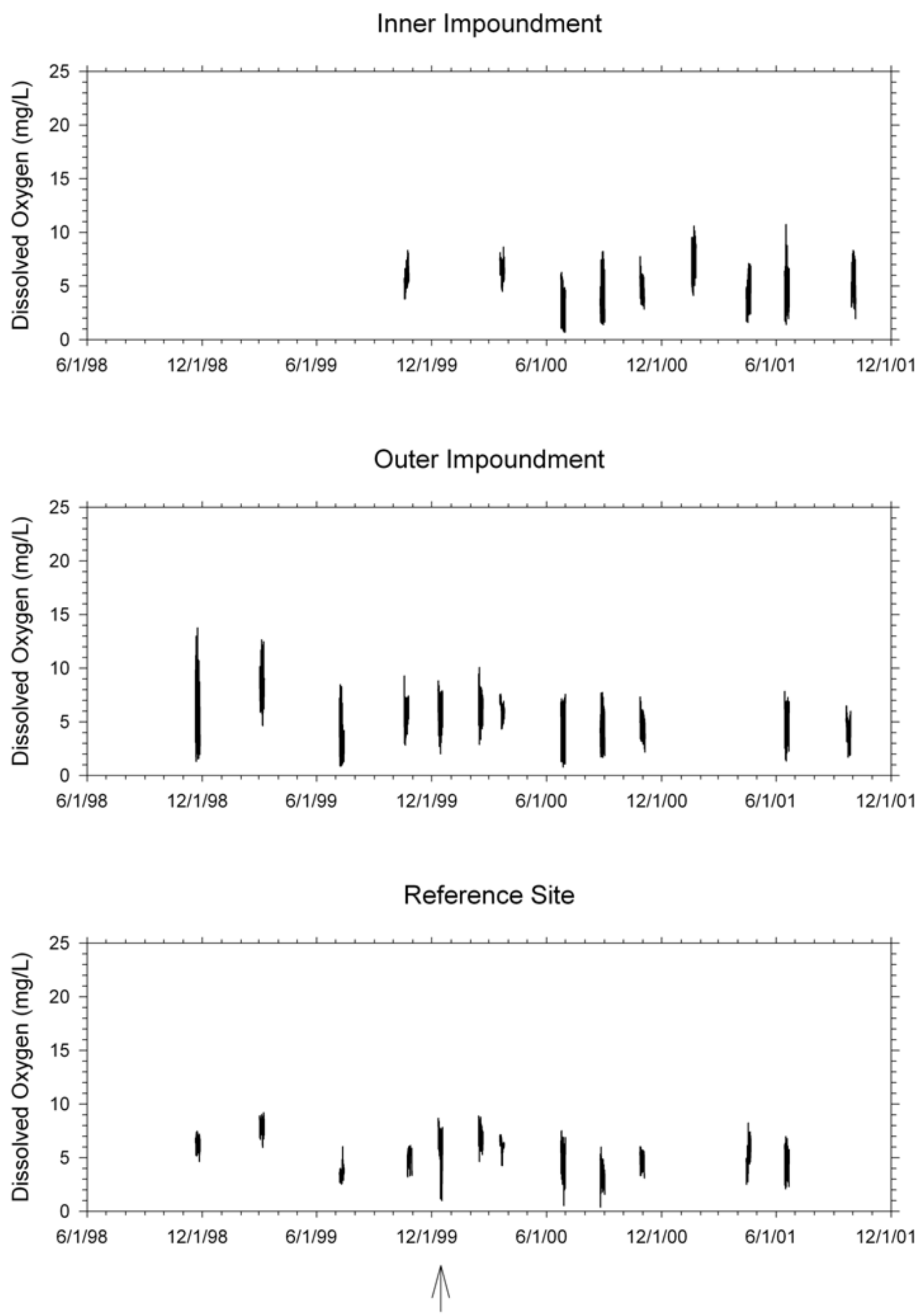

Figure 11. Comparison of Dissolved Oxygen at the Outer Impoundment, Inner Impoundment, and Reference sites. Only the first week of each three week deployment is shown here, as the sensors would tend to foul after a week and produce unreliable data.

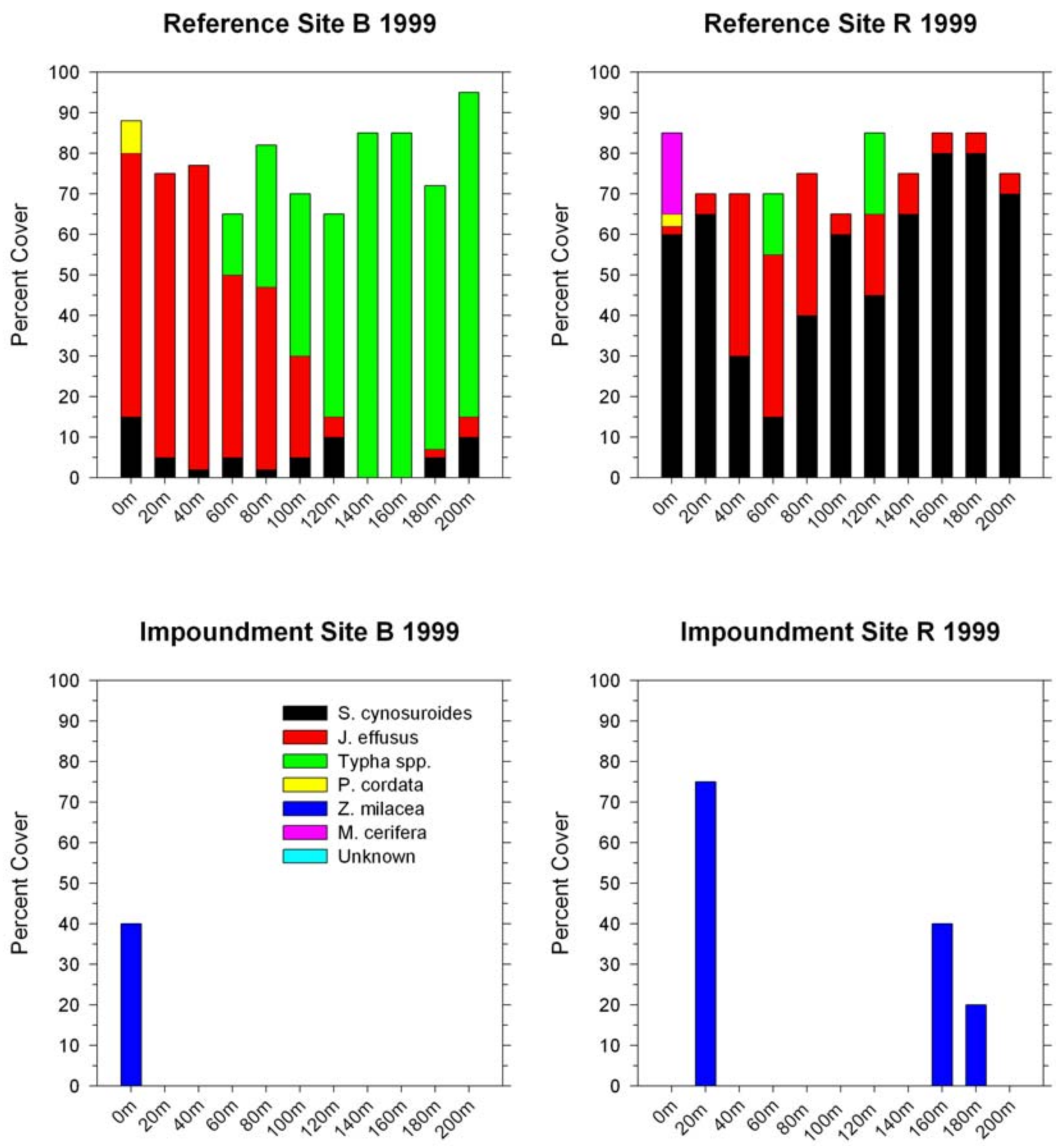

Figure 12. 1999 Vegetation Transects. Measurements of percent cover were taken at 20 meter intervals in impoundment and reference transects in August 1999. Impoundment and Reference Transect B are the pair of transects that are close to the large natural breach and Impoundment and Reference Transect R pair are the pair of transects located closer to the Ogeechee River.

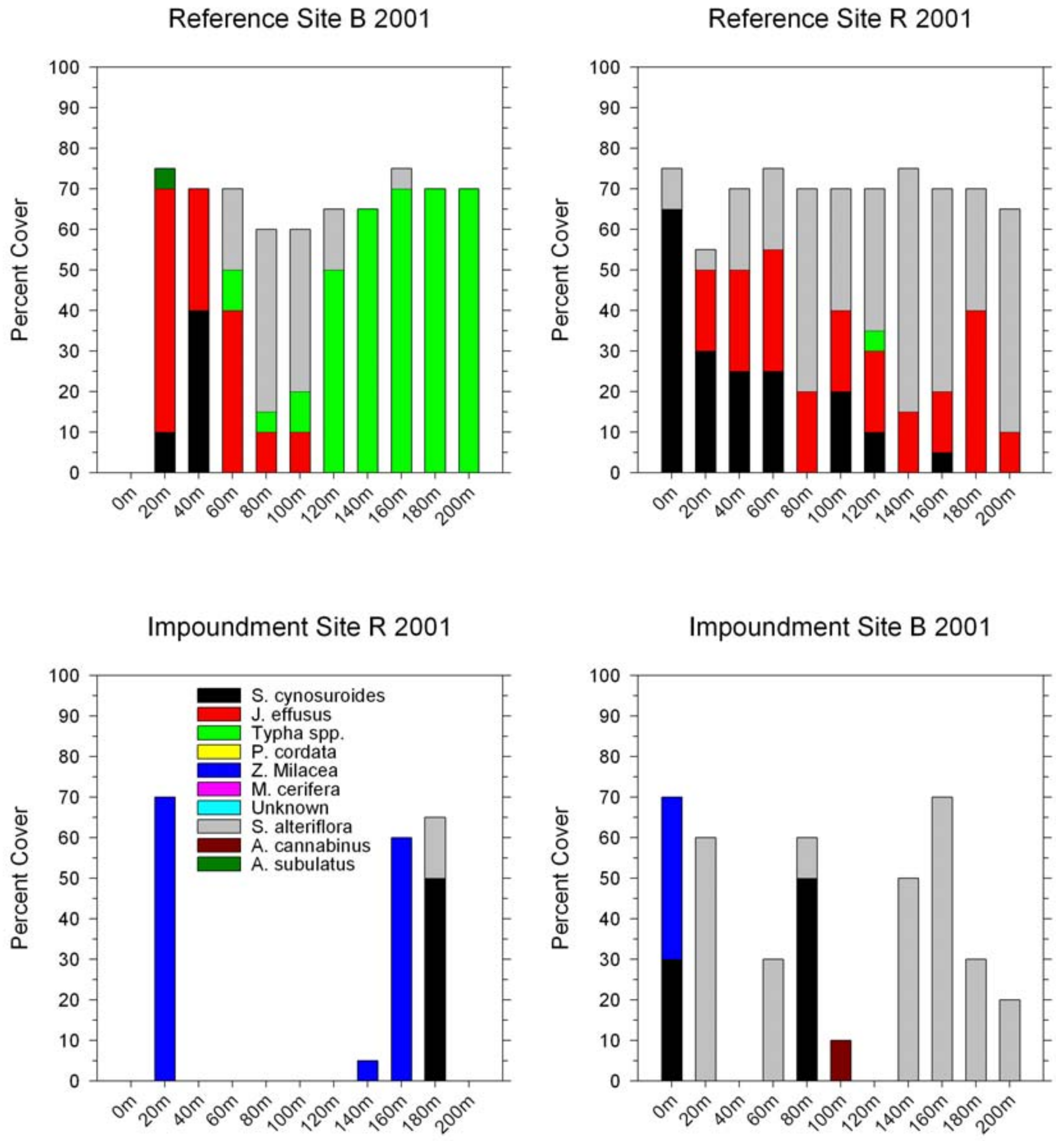

Figure 13. 2001 Vegetation Transects. Measurements of percent cover were taken at 20 meter intervals in impoundment and reference transects in October 2001. Impoundment and Reference Transect B are the pair of transects that are close to the large natural breach and Impoundment and Reference Transect R pair are the pair of transects located closer to the Ogeechee River.

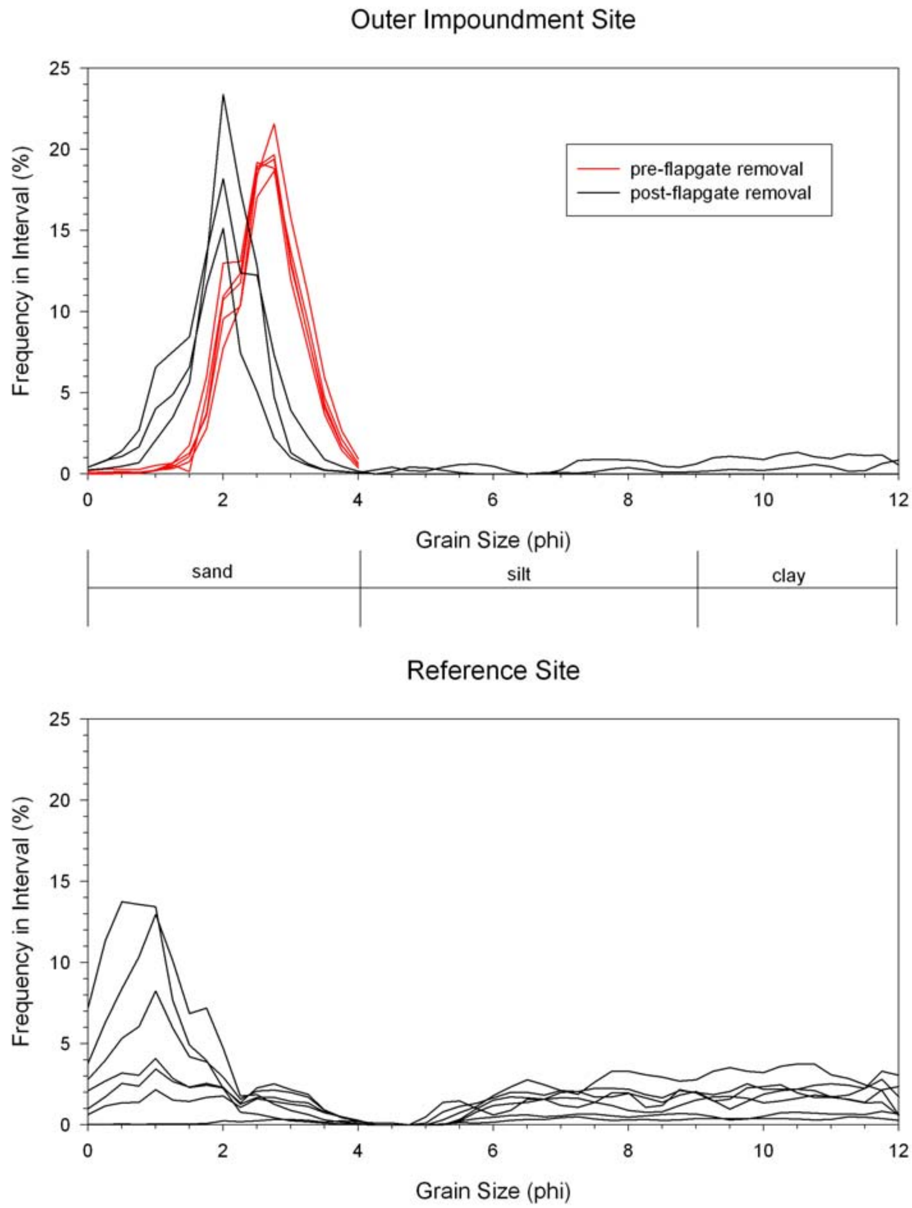

Figure 14. Comparison of Grain Size Data from the Outer Impoundment, Inner Impoundment, and Reference sites. Grain size data taken from the Outer Impoundment and the Reference Site taken before and after flapgate removal. The black lines for the reference site represent samples from deployments before and after flapgate removal. The black lines for the impoundment site represent samples from deployments after flapgate removal, while red lines represent deployment before flapgate removal.

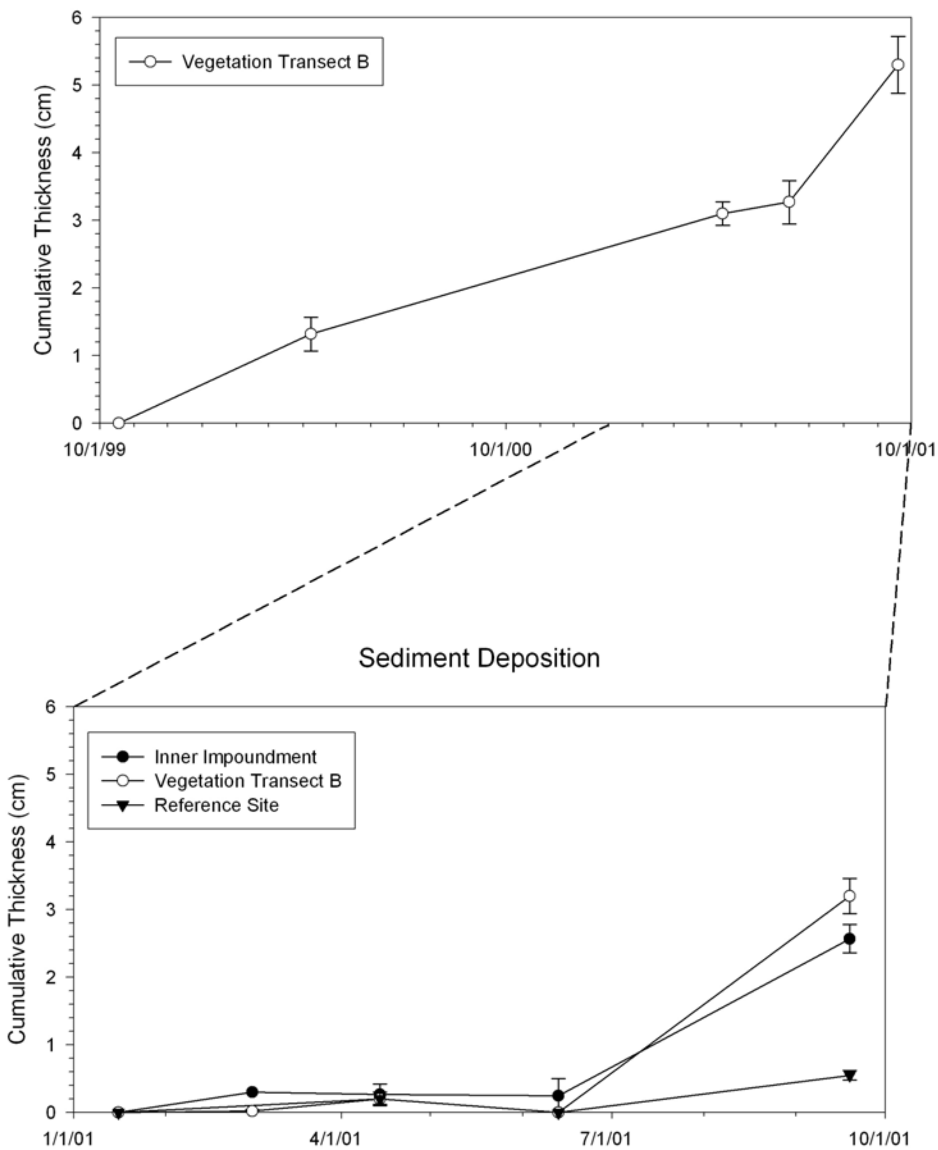

Figure 15. Comparison of Sediment Cumulative Thickness at the Outer Impoundment, Inner Impoundment, and Reference sites. Sedimentation rates were quantified by placing three unglazed ceramic plates (15 cm x 15 cm) at the marsh surface. Plates were initially placed at Vegetation Transect B in October 1999 to gain long-term information on impoundment sedimentation. In January 2001, plates were placed at Vegetation Transect B, Inner Impoundment, and the Reference Site for comparative studies.

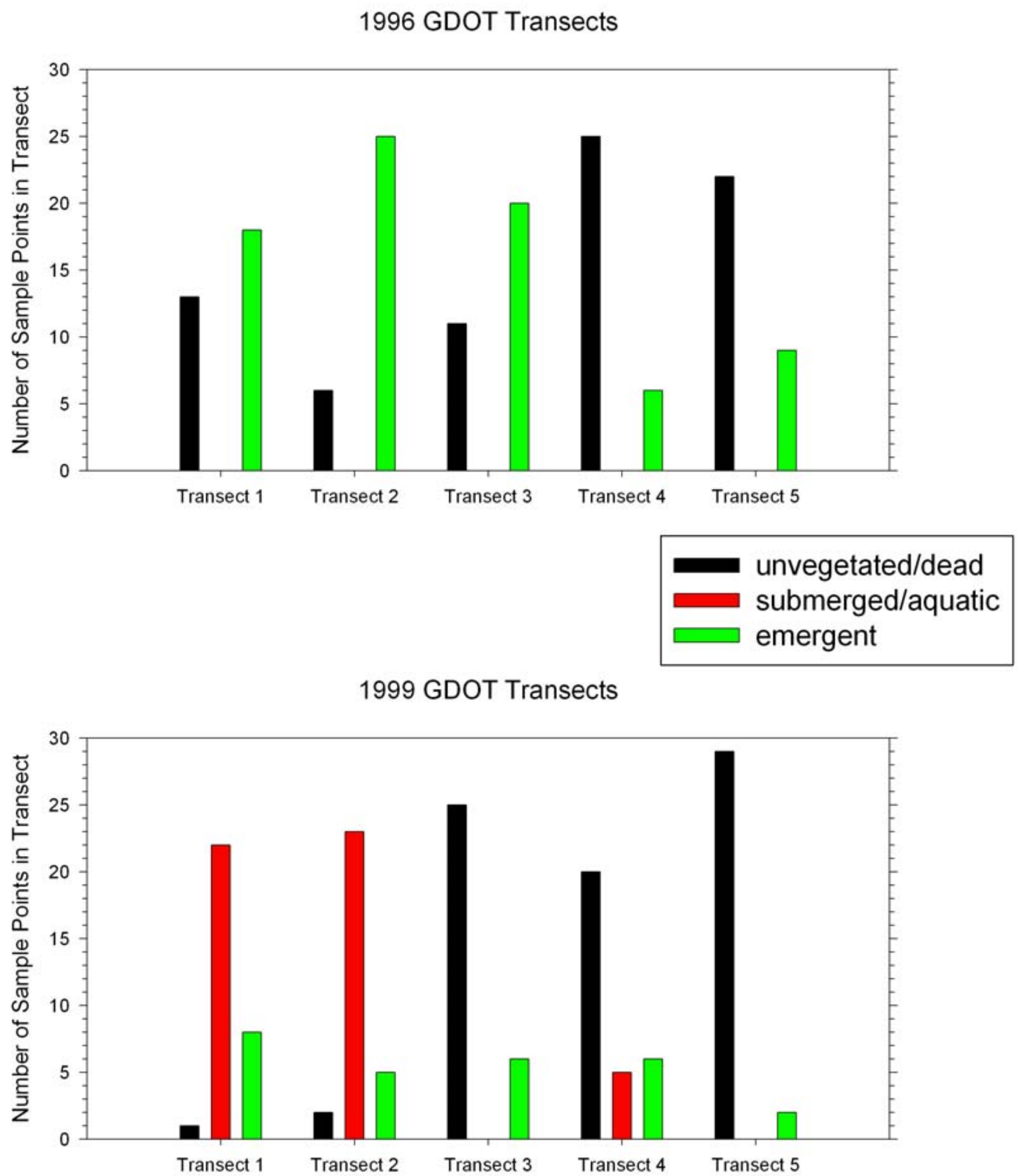

Figure 16. GDOT Vegetation Transects. Vegetation transects completed inside the impoundment by Georgia Department of Transportation in August of 1996 and 1999. See Figure 2 for transect locations.

CHAPTER 3

GEOMORPHIC COMPARISONS OF A RESTORED AND NATURAL MARSH USING A GIS

INTRODUCTION

Tidal marshes are dissected by a complex network of creeks and channels that provide a route for the delivery of sediment, oxygenated waters, and nutrients to the marsh as well as a means of removing material. The tidal marsh channel network has received increased attention from scientists and managers who recognize the importance of a well-developed drainage configuration in marsh restoration, creation, and enhancement projects (Zedler 2001). Creeks and channels give free-swimming nekton a route to the intertidal area via the creek edge and they also provide pathways into the marsh interior (Rozas and Zimmerman 2000; Zedler 2001). Ensuring the proper amount of edge has been demonstrated to be important for many fishery species, as many organisms move back and forth from the tidal creek to the marsh surface with the water (Minello *et al.* 1994; Rozas and Zimmerman 2000). Proximity to the water's edge and plant community composition were shown to be the most important factors in influencing the distribution of nekton in a marsh in the Gulf of Mexico (Rozas and Zimmerman 2000). In

Louisiana, Peterson and Turner (1994) found that many species on the marsh surface, particularly non-marsh residents, were concentrated within a few meters of the tidal channel.

Although edge habitat is understood to be important in terms of accessibility of the marsh surface to nekton, the importance of creek size is not often considered, particularly in restoration projects. Several studies have found that juvenile fish utilization of tidal channels depends on channel size, with increased densities in smaller channels (Rozas and Odum 1987; Rozas 1992; Desmond *et al.* 2000). Many organisms prefer to use small creeks and rivulets to access the marsh, rather than cross over the creekbank in a larger channel (Rozas *et al.*1988), and fish species have been shown to increase their food intake and grow faster when they have access to the marsh surface through small channels (Madon *et al.* 2001). However, the number of fish that actually use smaller channels for marsh access depends on the relative density of these channels in the marsh drainage system (Rozas *et al.* 1988).

A properly functioning drainage network corresponds to Horton's law of stream channel composition, or stream order, which was first studied in fluvial systems (Horton 1945). Horton found that stream order is related to the number of channel segments, the mean sinuous channel length, and drainage basin area, through geomorphic relationships. These geomorphic relationships were applied to tidal systems by Pestrong (1965), and numerous studies of marsh systems have utilized geomorphic characteristics to evaluate drainage systems. Some of the characteristics that are often considered include channel order, channel distribution, sinuosity, marsh edge, channel length, bifurcation ratios, and drainage density (Pestrong 1965; Garofalo 1980; Coats *et al.* 1995; Weinstein *et al.* 1997; Zeff 1999; Desmond *et al.* 2000; Sanderson *et al.* 2000). A comparison of drainage systems in natural marshes on the West coast, East coast, and the Gulf demonstrated that all had similar characteristics in terms of bifurcation ratios, number of channels per order, mean sinuous lengths per order, and sinuosity ratios (Zeff 1999).

Geomorphic parameters in drainage systems have been examined in restoration projects as well, although most of these studies have taken place in the Pacific coast region (Pestrong 1965; Coats *et al.* 1995; Williams and Zedler 1999; Desmond *et al.* 2000; Sanderson *et al.* 2000; Sanderson *et al.* 2001). Some studies of drainage patterns have also been conducted in the Gulf of Mexico, but the majority of these focus on maximizing nekton production rather than restoration success (Zimmerman *et al.* 2000; Minello 2002). Given that creek patterns are likely influenced by both tidal range and type of vegetation (Garofalo 1980; Coats *et al.* 1995), there is a recognized need for comparative information to be collected in other types in marshes in different environments (Coats *et al.* 1995).

Marsh vegetation and tidal regimes differ greatly between the Pacific, Atlantic, and Gulf of Mexico coasts: the Pacific coast in characterized by mixed tides, the Gulf coast has diurnal tides and the Atlantic coast has semidiurnal tides. Tidal ranges in southern California are typically three meters and less than one meter in the Gulf. In the northern United States, tidal ranges average more than two meters and in the southeastern U.S. they range from one to three meters, with the largest tidal range found along coastal Georgia. The dominant marsh vegetation species on the West coast are *Salicornia virginia* and *Spartina foliosa*, in contrast to the Gulf marshes and Southeastern marshes where *Spartina alterniflora* dominates. In more northern areas of the Atlantic, *Spartina patens* is the dominant marsh vegetation species.

This is the first study completed on the East Coast that calculated current and historic geomorphic characteristics in a restored marsh and compared these results to a natural marsh system. In this study, the geomorphic variables channel order, channel distribution, drainage density, and drainage area were examined in a passively restored brackish marsh in coastal Georgia from historic aerial photographs. These parameters were then assessed in photographs 50 years older to assess long-term changes in drainage patterns. A natural marsh was also analyzed to determine whether the drainage network in a restored habitat was comparable to that of a natural system.

STUDY AREA AND METHODS

Study Site

Drainage channel analysis was performed on aerial photographs of a formerly impounded marsh on the Ogeechee River in coastal Georgia and compared to photographs of a nearby natural marsh (Fig. 17). The formerly impounded marsh (called the restored marsh) has a remnant dike system from historical rice cultivation in the 1800s (Sullivan 1999), which began to naturally deteriorate in the early 1900s, allowing the initiation of passive restoration. This formerly impounded marsh was used as the reference marsh in the evaluation of short term changes of a mitigation project (see Chapter 2). In this study, I characterized the drainage patterns in this marsh approximately 50 years after restoration, in 1949, and 50 years after that in 1999. The natural marsh is located nearby in the Little Ogeechee River, a small branch of the Ogeechee River, which does not have the direct freshwater inputs required for rice cultivation. Drainage patterns in this marsh were characterized based on photographs taken in 1999. Salinity ranges are slightly higher in the natural marsh habitat than at the restored former rice impoundment, because all habitats in the area that had a similar salinity range as the restored marsh were also used for rice cultivation. However, water level predictions at the natural marsh average approximately two meters and are comparable to those at the restored site (Nobeltec Tides and Currents Program version 3.0). Plant communities in the restored marsh are currently a brackish mix, consisting of *Spartina alterniflora*, *Spartina cynosuroides*, *Typha* spp., *Juncus effusus*, and *Juncus roemerianus*. In the natural marsh, the plants assemblage is dominated by *S. alterniflora* and *J. roemerianus*.

Remote Sensing

A Geographic Information System (GIS) was used to characterize drainage patterns in the restored marsh in both 1949 and 1999 and in the natural marsh in 1999. I obtained aerial photographs of the study areas representing 1949 (black and white at 1:20,000) and 1999 (color infrared at 1:12,000). The 1949 aerial photographs came from the US Department of Agriculture, Agricultural Stabilization and Conservation Service (now called the Farm Service Agency), which has photographed agricultural land since the 1930s. The 1999 photographs came from the National Aerial Photography Program associated with the United States Geological Survey. These photographs can be purchased as digital orthophoto quadrangles that are already georeferenced with a ground resolution of 1 m. Approximately four photographs were used in a mosaic of each study site.

The series of photographs from 1949 were scanned in and georeferenced with ArcInfo (ESRI, version 8.1) using the 1999 photographs as the base map from which to establish the coordinate system on the older photographs. Great care was taken to choose ground control points (GCPs) that have remained stable over a period of 50 years. Typical GCPs that were chosen for rectification included individual trees, stable vegetation patches, road intersections, and, when necessary, intersections of creeks. Using creek intersections for rectification was not optimal since the objective of the analysis was to assess the extent of channel changes, but in cases where using these intersections for GCPs was unavoidable they were always used in conjunction with other types of GCPS. Approximately 10-15 GCPs were used to rectify each photograph. The REGISTER command (all ArcInfo commands are written in capital letters) was used to establish the ground control points for each photograph and the RECTIFY command was used to perform the georeferencing process.

The ESRI recommendation for Root Mean Square (RMS) error suggests that the total error for each georeferenced photograph should be between 0.004 – 0.008 inches. This equates to an error that should range from 2.0 m – 4.1 m for the 1949 photographs. Although every effort was made to keep the error within this range (i.e. removing and adding GCPs to see if that reduced the error term), it was not possible to get the RMS errors this low on most photographs. Final RMS errors on the rectified photographs ranged between 1.0 m and 7.1 m. The National Map Accuracy Standards (NMAS) states that on all maps at scales of 1:20,000 or more, not more than 10% of the points tested should be in error by more than 1/20th of an inch. This error would equate to a 16.9 m error on the 1949 aerial photographs, which is much larger than the RMS errors in this study.

Drainage Patterns

The outer lines of tidal channels were digitized on-screen from both 1949 and 1999 photographs using ArcView (version 3.2) to make line shapefiles for drainage density and area calculations. The images were viewed on-screen at a scale of 1:750 to 1:2000, depending on the size of the channel. The outer line themes were then converted to ArcInfo coverages using the SHAPEARC command. Polygon topology was established using the CLEAN function. The fuzzy tolerance used for the CLEAN command was 0.001 m. Any incomplete polygons or vertices were corrected in ArcEdit using ARCSNAP and ADD to connect two line segments. The coverages were then CLEANed again and then converted to shapefiles for use in ArcView. All polygons in the attribute tables were viewed and labeled appropriately as marsh or channel. The shapefile was then DISSOLVEd using the Geoprocessing Wizard so that polygons were grouped according to habitat category (marsh or water). The final channel themes from the restored marsh areas were CLIPped to a digitized study area theme to ensure that they were the same area. Area of each habitat category could then be calculated.

Drainage density and drainage area were calculated using polygon themes. The amount of error associated with digitizing polygons for geomorphic analyses was also calculated. Three different areas ranging from 3571.97 m^2 to 23669.9 m^2 were each digitized in triplicate in both the 1949 restored marsh and the 1999 natural marsh photographs. Mean drainage area and standard deviation were computed for each area. Errors averaged 8% for the 1949 photographs and 4% for the 1999 photographs.

Buffer Analysis

To determine the amount of marsh edge, calculations were performed to determine how much of the marsh in each study area was within three meters of drainage channels. Buffers of three meters were assigned to the channel shapefiles and then SUBTRACTed from the marsh theme in ArcView to result in a shapefile of the marsh habitat that was more than three meters from a channel for the restored and natural marsh in 1999 and the restored marsh in 1949.

Tidal Marsh Geomorphology

The centerlines of all channels in both the restored and natural marshes in 1999 were digitized to perform additional analyses of channel order, channel order distribution, bifurcation ratios, lengths, and sinuosity ratios. The channel order was determined and entered into the attribute table for the entire drainage network in the study area using Horton's (1945) method for assigning stream order, where small, first order streams originate in the marsh system. When two first order streams meet, a second order stream is formed and this process proceeds until all channels are labeled and ordered. The total number of channels, their distribution, and lengths were ascertained using GIS once the orders were assigned. All final channel and centerline themes were CLIPped to a digitized study area theme for the restored marsh to ensure that themes for the 1949 and 1999 were the same area, similar to the process completed for the polygon drainage themes. The attribute tables were DISSOLVEd based on channel order so that total sinuous length and total number of channels could be determined for each channel class.

Assignment of stream order was not performed for the 1949 restored marsh. The 1949 marsh system is artificial and channels were so interconnected that there did not appear to be a logical ordering system where smaller order streams joined to form higher order streams, and so any channel order assignments would have been arbitrary.

Sinuosity ratios, bifurcation ratios, and drainage density were calculated based on channel centerlines, following Pestrong (1965):

1) Bifurcation Ratio (R_b): $R_b = N_u / N_{(u+1)}$

where U is the channel order and N_u is the number of channels of a particular order. This ratio provides an estimate of the number of channel segments of each particular channel order in relation to the number of channel segments of the next highest order.

2) Sinuosity Ratio (R_s): $R_s = L_{sin} / L_{str}$

where L_{sin} is the total sinuous length of a channel segment and L_{str} is the total straight length of a channel segment. This ratio provides an estimate of the level of deviation from a straight path for each particular channel order. The sinuosity ratio tends to increase with increasing channel order.

3) Drainage Density (DD): $DD = L_t / A_t$

where L_t is the total length of all channels in the study area and A_t is the total area. This ratio is used as a measure of the density of tidal channels in a drainage area.

RESULTS

Natural and restored marshes have very different drainage systems due to differences in the amount of open water, channel meandering, and marsh habitat in the two systems (Figs. 18 – 21). In the natural marsh, channels are highly branched and sinuous, with large expanses of vegetated marsh habitat (Figs. 18 – 19). Tidal creeks appear to have obvious upstream and downstream directional flow, with smaller channels splitting off larger channels towards the headwaters. In the 1949 restored marsh there is a high density of drainage channels that are typically straight, with little meandering. Channels patterns have no obvious upstream and downstream flow direction. In this system, numerous areas exist where double, parallel channels enclose islands of marsh. Fifty years later, in1999, the restored marsh looks similar in appearance to the 1949 marsh, but with fewer drainage channels and consequently more marsh habitat (Fig. 21). Parallel channels have merged together to form wider channels. Some channels exhibit signs of meandering, but in general straight channels still dominate the study area. At all of the study sites, marsh parcels not near large dominant channels were less dissected as compared to these areas adjacent to large channels.

Drainage Patterns

The drainage patterns in the natural marsh were very different than those in either observation of the restored marsh. Both the drainage area (square meters of drainage per total marsh area) and the drainage density (total length of channels per total marsh area) of the natural marsh were lower than what was observed in the restored marsh (Table 7). As compared to the reference marsh, the natural marsh had approximately half the channel density and area and more marsh habitat. Open water in the natural marsh only accounted for 5% of the total study area, in contrast to the restored marsh, where 9-12% of the area was open water (Table 7). Because channel density was so high in the restored marsh, there was greater marsh habitat within three meters of a channel (70-80%) as compared to the natural marsh (40%).

There were also differences in drainage patterns between the 1949 and 1999 restored marsh. Drainage channels in the restored marsh totaled approximately 12% of the study area in 1949 and decreased to 9.5% in 1999 (Table 7). Indications as to where channel loss has occurred can be seen in a comparison of the drainage network in the 1949 and 1999 restored marshes and in an overlay of the two coverages (Figs. 20-22). Dominant channel pathways increased in width from

1949 to 1999 with less efficient channel routes deteriorating over this period of time (Figs. 22, 23). Some channels that had previously penetrated into marsh parcels had disappeared by 1999, particularly in areas not directly connected to major channel pathways. In several cases, narrow channels located close to one another joined together to form large channels, particularly in dominant channel pathways. Although the drainage patterns in the restored marsh in 1999 were more similar to those found in the natural marsh than those observed in 1949, they were still very different than the natural marsh. In 1999, after approximately 100 years of passive restoration, the drainage area, drainage density, area of open water, and the area of marsh within three meters of a channel all remained approximately 50% higher than what was observed in the natural system.

Tidal Marsh Geomorphology

Geomorphic calculations for the marsh study sites generally followed Horton's laws of drainage for fluvial systems. The number of lower order channels exceeded the number of higher order channels in both the restored and natural marshes; sinuous length of the lower order channels was longer than that of the higher order channels; and both the number of channels and the mean sinuous length were inversely proportional to increasing channel order (Table 8). Bifurcation ratios, which give information on the distribution of the number of channel segments in each order in relation to the number of channel segments in the next highest order, were found to be approximately 3.9 and 3.7 in the restored and natural marsh, respectively, which is comparable to those observed in other marsh systems (Strahler 1952; Coats *et al.* 1995).

Although both the natural and restored sites generally followed Horton's laws, there were clear differences between them in the numbers of channel segments per order, sinuosity,

sinuosity ratios, and bifurcation ratios (Table 8). The proportions of the number of channels per order were similar between study sites, but the total number of channels per square kilometer was lower in the natural marsh. This was particularly true for first order channels (198/km^2 in the restored marsh vs. 142/km^2 in the natural marsh; Table 8). As channel order increased, the number of channel segments became more similar.

. In general, mean sinuous length increased with channel order in both the restored and natural marshes (Fig. 24). In the restored marsh, the lower order channels were the ones that were straightened for rice cultivation, yet these lower order channels are naturally less meandering so the sinuosities are still comparable to those in the natural marsh. However, the mean sinuous length was much lower in the restored marsh fifth order channel than in the natural marsh despite the fact that it was less likely to have been as highly altered for agriculture.

Sinuosity ratios (an estimate of the degree of meandering) were fairly similar between the two marshes, averaging 1.2 in the restored marsh and 1.4 in the natural marsh (Table 8). However, the relationship between sinuosity ratio and channel order differed at the two sites (Fig. 26). Whereas in the natural marsh sinuosity ratios increased with channel order, in the restored marsh the ratios did not vary between the first through fourth order channels (although it increased in the fifth order). This means that sinuosity was greater in the third and fourth order channels in the natural marsh as compared to the restored marsh.

Overall, sinuosity appeared to increase from 1949 to 1999 in the restored marsh. Due to the difficulties in determining channel order for the 1949 marsh, only general observations of sinuosity change could be made. A simple overlay of the two channel themes from the 1949 and 1999 restored marshes reveals that while some increase in sinuosity occurred between 1949 and 1999, the drainage network still retained the characteristics of a man-made system (Figs. 22, 23, 25). The lower order channels, particularly the first and second order channels have increased in sinuosity over fifty years (Fig. 23). However, little to no sinuosity increase was found in the third and fourth order channels.

The bifurcation ratios found in both marshes (3.9 in the restored marsh and 3.7 in the natural marsh) are comparable to those found in other natural systems. Data collected from numerous natural marshes in California all had an average bifurcation ratio of 3.5 (Strahler 1952; Coats *et al.* 1995). The range of individual bifurcation ratios found for channel order were found to fall within the range of values determined for marshes in California (2-10; Coats *et al.* 1995). The ratio was slightly higher in the restored marsh due to the increased bifurcation in the higher order channels. In contrast, the number of second order channels in relation to the number of third order channels was much lower in the restored marsh than in the natural marsh. Bifurcation ratios have been demonstrated to be useful in marsh enhancement projects where stream channels are added to the drainage network, as the bifurcation ratios help determine the numbers of channels per channel order to assess proper channel distribution.

DISCUSSION

Drainage Patterns

The geomorphic comparisons of creek networks in the natural marsh and restored rice impoundments suggest that restoration is a very slow, long-term process. After approximately 50 years of passive restoration, drainage channels in the 1949 restored marsh were predominantly straight, particularly in the mid-order channels. The total length of channels was approximately double that found in the natural marsh and the drainage patterns were different.

Moreover, large expanses of marsh habitat in the natural marsh could be found that were dissected by sinuous creeks and channels, whereas marsh parcels were highly fragmented with straight channels in the restored marsh. Channels in the natural marsh followed logical patterns associated with bi-directional flow whereas channels in the restored marsh were grid-like without obvious circulation patterns.

Although there is some evidence that 50 years later, in 1999, there was a shift in channel geomorphology in the restored marsh toward the characteristics found in the natural marsh, the two systems were still very different. Despite a decrease in channel density and a small increase in sinuosity, the 1999 restored marsh retained the characteristics of a man-made system. Drainage density and drainage area both decreased approximately 30% in 50 years, but remained at least 50% higher than what was found in the natural marsh. Channel loss occurred in areas away from major channels and in areas where multiple channels with center islands have joined together to form larger channels. It is likely that the few remaining multiple channels and islands of marsh that exist will be replaced by wider and/or larger channels over time, as has been observed in California (Haltiner *et al.* 1997). Due to the complexity of the artificial channel system in the 1949 restored marsh it was not possible to quantify sinuousity and the actual change over time. However, in a qualitative examination of the centerline overlay of the two marshes, there only appears to be minor changes in the meandering in these channels between 1949 and 1999.

Edge Considerations

This study indicated that edge habitat is highest in the 1949 restored marsh, lower in the 1999 restored marsh, and lowest in the natural marsh. Approximately 70 – 80% of the marsh area in the 1949 restored marsh was within a few meters of a drainage channel, which would have provided a large amount of edge habitat. Numerous studies have documented the high abundances of nekton and infauna at the marsh-channel interface or within the first few meters of the channel edge (Rozas *et al.* 1988; Minello 2002). For example, white shrimp, brown shrimp, and blue crabs in the Gulf of Mexico are mostly found within a few meters of drainage channels on the marsh surface (Minello 2002). Nekton may be found at the marsh edge because of the increased numbers of prey species that are also found here (Minello *et al.* 1994; Whaley and Minello 2002). A high density of tidal creeks would maximize marsh edge and therefore the use of marsh edge (Coats *et al.* 1995).

These results imply that the restored marsh would provide better habitat value due to the increased edge habitat associated with high drainage densities. However, the value of edge habitat depends on the species studied, the physical characteristics of the marsh-channel interface, and the length of inundation/elevation of the marsh surface. One study noted that although transient species (shrimp and blue crabs, Minello 2002) were most frequent at the marsh edge, resident marsh species were found in greater abundances in the interior marsh (Peterson and Turner 1994). There are also differences in species composition depending on distance from creek edges (Thomas *et al.* 1990; Rozas 1992 Baltz *et al.* 1993; Peterson and Turner 1994; Coats *et al.* 1995). Interior marsh habitat has been demonstrated to be extremely important for smaller resident marsh species, such as *Fundulus heteroclitus* which, like other small resident fish, finds refuge in the densely vegetated marsh surface (McIvor and Odum 1988; Rozas *et al.* 1988; Kneib and Wagner 1994; Kneib and Craig 2001). Therefore, although higher marsh edge habitat may be beneficial for some species, a diversity of habitats may promote species diversity.

Another consideration for evaluating edge habitat is its accessibility. Although it may appear that the restored marsh provides more edge for nekton, steep edge slopes and high marsh elevation may compromise accessibility to the marsh surface. McIvor and Odum (1988) studied nekton usage of the marsh surface in channel systems with different geomorphologies and found that fish and free-swimming invertebrate abundances were lower in marsh surfaces that were adjacent to steep erosional slopes. Steep channel edges have been found in ditched marsh systems, possibly due to the higher current velocities associated with straight channels (Poizat and Crivelli 1997; Zimmerman *et al.* 2000). Rozas (1992) found that in some cases, steep creek sides only allow access by nekton in extremely high tides. Additionally, in rice impoundments, the material dug from the channels was used to form internal dikes around parcels of marsh inside the impoundment, creating high side levees (Sullivan 1999). Steep channel sides and possibly high channel side levees may have interfered with nekton access to the marsh (Rozas 1998; Rozas and Zimmerman 2000). Cornu and Sadro (2002) found that the density of fish distribution in a restored marsh was related to the shape of the creek bank next to the marsh and the number and characteristics of rivulets providing access to the marsh surface. Many studies of the importance of marsh edge examine the abundance and distribution of species relative to the distance from the channel edge in the natural marsh (Minello *et al.* 1994; Rozas and Zimmerman 2000). However, the findings from these studies may not be directly transferable to restored marshes if access to the marsh surface is different.

Even in cases where edge slope allows for adequate access to the marsh, one must also consider the importance marsh surface elevation in determining the benefits of high edge habitat. Elevation influences the hydroperiod, which is the length of time the marsh surface is under water and therefore available to nekton. In a comparison of a series of natural and restored marshes in Texas, Delaney *et al.* (2000) found that the restored marshes generally had high elevations, both at the marsh edge and in the marsh interior. Elevation data were not taken in this study, but general observations indicate that the restored marsh may be more characteristic of a high marsh environment, which is only intermittently flooded and undergoes fewer immersions than low marsh. This suggestion is supported by my observations of the plant species in the field, as well as by indications from aerial photographs. In addition, the presence of hogs indicates that there is likely reduced flooding in the area. A reduction in the hydroperiod would lessen the importance of marsh edge for nekton and could result in a less dense drainage network over time (Wadsworth 1980).

Two explanations for the lower amount of edge observed in the natural marsh is that due to the scale of the photographs, some of the smaller channels may not have been correctly digitized and/or the smallest rivulets may have been omitted. It was easier to see smaller channels in the restored marsh where channels are artificial and straight, as compared to the natural marsh. Although it was not possible to verify the 1999 photointerpretations with field data, this explanation is unlikely because the photographs were high quality infra-red images with a resolution of one meter and most first order channels (smallest channels measured in this study) were wider than one meter. In addition, overlapping photographs were available for each study area to provide multiple opportunities to assess channel boundaries. However, the smaller rivulets that occur at the most upper reaches of the first order channels were not delineated in the present study. Rivulets breach the boundary between the channel bed and the marsh surface and are important in that they provide easier access to the marsh surface for small resident nekton than crossing over channel levees. None of the photographs had a high enough resolution to capture these rivulets, which have been estimated to be approximately 0.3 – 0.5 cm wide (West and Zedler 2000). Vegetation often grows in the rivulet boundaries and would make them difficult to see even if larger scale photographs were used for analyses. Rivulets are extremely dense in natural marshes; however, it is not likely that these features are present in ditched environments. If these had been included in the channel network I might have drawn a different conclusion with regard to the amount of edge habitat in the natural system. The inclusion of rivulets in the estimates of drainage density would have likely resulted in much greater drainage density estimates and greater edge habitat in the natural marsh. Not only would the addition of rivulets increase drainage density, but they may also provide an important access point to the intertidal marsh surface by nekton (Cornu and Sadro 2002).

Tidal Marsh Geomorphology

Channel morphology is influenced by complicated interactions among tidal range and energy, vegetation, and fauna and this study provides much needed information about the geomorphology of the marshes in the South Atlantic Bight. Pacific coast marshes experience similar tidal ranges as Georgia marshes, but have much less extensive spans of intertidal marsh habitat and different dominant vegetation. Gulf coast marshes have similar vegetation, but they experience much smaller tides than Georgia, are inundated more than Atlantic marshes, and have more fragmentation resulting in higher amounts of edge habitat (Zimmerman *et al.* 2000). Most previous studies of tidal marsh geomorphic parameters come from the Pacific coast, focused predominantly on California (Williams and Orr 2002). Studies conducted on the Gulf Coast have focused on how nekton production and abundance have been influenced by drainage characteristics, but the geomorphology has been less well characterized.

Densities of first order channels in both the restored and natural marsh were comparable to what has been found in geomorphic studies in California. Coats *et al.* (1995) reported that the number of first order streams ranged from 76-86% of the total number of streams, compared to 87% found in this study's natural marsh. No channel distributions were reported for fifth order system restored marshes, but two restored marsh fourth order systems in California had 71% and 78% first order streams. In this study of a fifth order restored marsh, approximately 77% of the channels were found to be first order creeks.

Geomorphic parameters also generally fell within the range of values observed in marsh systems in other areas. The number of channel segments per channel order and the mean sinuous length per channel order were comparable to those reported in other studies (Figs. 26, 27). Differences between channel numbers and mean sinuous lengths per order among sites referenced in these figures are not clear and are in need of further study. It has been suggested that differences between the number of channels per order at the sites represented in these figures are likely due to differences in study area and the size of the drainage system (Novakowski *et al.* 2004). The two marsh systems with the lowest overall number of channel segments were the two studies that measured channel lengths in fourth order systems (versus the fifth order systems studied elsewhere). Differences in the slopes of mean sinuous length are likely due to differences in salinity and vegetation. Sinuosity is higher in salt marshes, where vegetation exerts control of patterns of meandering whereas in freshwater systems, hydrodynamics influences sinuosity more than vegetation (Garofalo 1980; Zeff 1999). Although salinities and vegetation were not determined in the studies used in the Hortonian analyses, it can be noted that the regression line from California in 1963, whose slope is the most different from the other studies, was taken from a freshwater marsh while the others were from saline marshes. Mean sinuosity was lower in the freshwater marsh in the smaller channels, but quickly increased in the larger channels.

Drainage densities were somewhat lower than what has been observed in California, where various investigators have reported densities between 0.03 – 0.06 m/m^2 in natural systems (Coats *et al.* 1995; Sanderson *et al.* 2000). I found a drainage density of 0.01 m/m^2 in the natural marsh and higher values in the restored marsh (0.03 m/m^2 and 0.02 m/m^2 in 1949 and 1999, respectively). A study in South Carolina reported an average drainage density of 0.013 m/m^2 (Novakowski *et al.* 2004), which is comparable to the value observed here. Average sinuosities for two natural marshes in California were 1.2 and 1.5 (Coats *et al.* 1995), compared to sinuosities of 1.4 for the natural marsh in Georgia. Restored marshes in California had average sinuosities of 1.1, which is similar to the sinuosity ratio of 1.2 found in the restored marsh in this study.

Due to the major differences in tidal range and vegetation species between the Gulf, Pacific, and Atlantic coasts and the influence these parameters have on shaping geomorphic processes, it has been pointed out that gaining geomorphic information from different regions is important (Coats *et al.* 1995). Most of the studies that have used geomorphic parameters to evaluate differences between restored and natural marshes have been conducted on the California coast. The results reported here suggest that there are in fact many geomorphic similarities between Georgia and Pacific marshes in terms of channel order distribution (Fig. 27) and drainage density.

Using Horton's laws of drainage analysis may be useful as an overall guide to channel distribution in marshes, but it does not get at the observed differences in drainage patterns between the restored marsh and other natural systems. Drainage densities and the quantity of edge habitat more closely reflect drainage pattern differences between the restored and natural marsh. The type of conditions optimum for nekton is not something that is generally known and is in need of further study (Kneib 2000; Weinstein *et al.* 2000). This study highlights the need for further research to not only quantify edge habitat in these created and restored marshes, but also to determine the quality of edge habitat available and accessible to nekton.

The data from restored marshes give some indication of the condition of restored former rice plantation marshes in comparison to natural systems and the degree of success that can be expected when using these marshes, or any other marsh that has been ditched or highly altered, to mitigate wetland loss. A century has passed since hydrologic restrictions were removed from the restored marsh and yet little changes in the drainage system have taken place. Despite higher edge habitat, the unnatural, straight channels present in the restored marsh may inhibit access to the marsh surface by nekton and the results here indicate that between 1949 and 1999 only slight changes in channel density and sinuosity occurred. Whereas numerous other studies have indicated success in restoring formerly impounded marshes by removing hydrologic restrictions (Weinstein *et al.* 1997, it may be that due to the extensive ditching and diking that occurred in these former rice plantations, these comparisons may not apply. A study in Georgia compared drainage maps over a 200 year period and found little change in the natural drainage patterns of larger channels (Wadsworth 1980). Once artificial drainage patterns have been established in former rice plantation marshes, it may take centuries for natural drainage patterns to develop, if they develop at all.

Table 7. Comparison of Drainage Patterns in Natural and Restored Marshes. Area and drainage densities were calculated from drainage channel polygons created from channel digitizations.

Site	Drainage Density (total length of channels m/km^2)	Drainage Area (total area of channels m/km^2)	Area of Open Water	Marsh Area within 3 m from channel
1949 Restored Marsh	28251	124194	12%	82%
1999 Restored Marsh	21324	95908	9.5%	72%
1999 Natural Marsh	14028	50346	5%	42%

Table 8. Geomorphic Calculations for the 1999 Restored and Natural Marsh. The calculations are calculated based on centerline digitizations for the restored and natural marsh in 1999. These calculations are arranged by channel order. Because the total study areas in the restored and natural marshes were not exactly equivalent, the number and lengths of channels for the restored marsh have been normalized for comparison. Data is shown as the total number of channels or the length of channels in a 1 km^2 area. Bifurcation ratios are calculated by determining the number of channel segments of a particular order with the number of segments for the next highest order. Theses marshes are fifth order systems; therefore, there would be no bifurcation ratio for the fifth order channels.

Site	Order	Number (per km^2)	Number (% of total)	Total Sinuous Length (m/km^2)	Mean Sinuous Length(m/ km^2)	Straight Length (m/km^2)	Sinuosity Ratio	Bifurcation Ratio
RM 1999	1	198.4	74.8	11834.5	59.6	10522.7	1.1	4.2
	2	46.8	17.6	5444.4	116.3	4655.4	1.2	3.4
	3	13.9	5.2	2489.5	179.1	2096.7	1.2	4.6
	4	3.0	1.1	1151.4	383.7	977.7	1.2	11
	5	0.3	0.1	325.7	10856.7	190.0	1.7	-
	mean						1.28	3.9
NM 1999	1	142.0	77.1	7620.4	53.7	6646.8	1.2	4.3
	2	33.3	18.1	3735.3	112.2	3082.0	1.2	5.9
	3	5.6	3.0	1249.2	223.1	910.2	1.4	3.5
	4	1.6	0.9	696.7	435.4	429.7	1.6	6
	5	0.3	0.1	711.4	23713.3	408.9	1.8	-
	mean						1.44	3.7

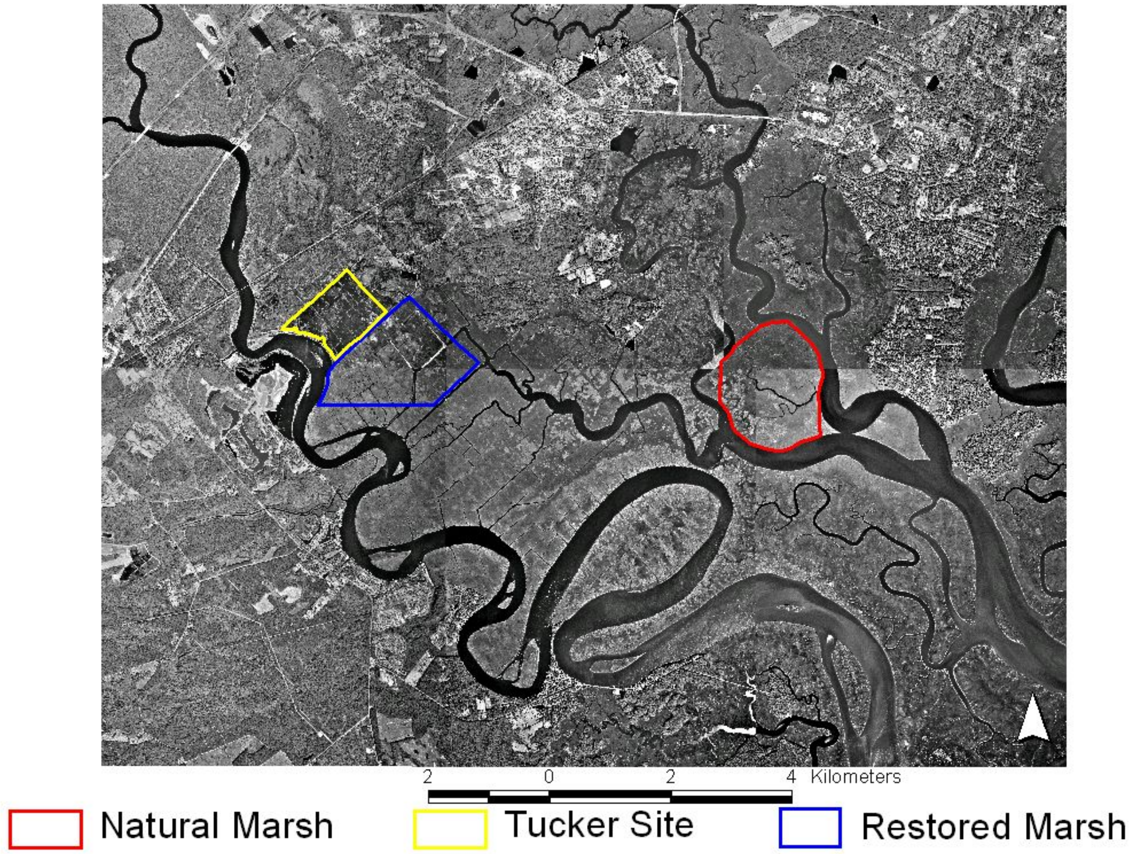

Figure 17. Study Sites in the Ogeechee River, Georgia. A mosaic of 1999 digital orthophotos is shown with the locations of the natural marsh (outlined in red), restored marsh (outlined in blue), and Tucker site (outlined in yellow) examined in this study.

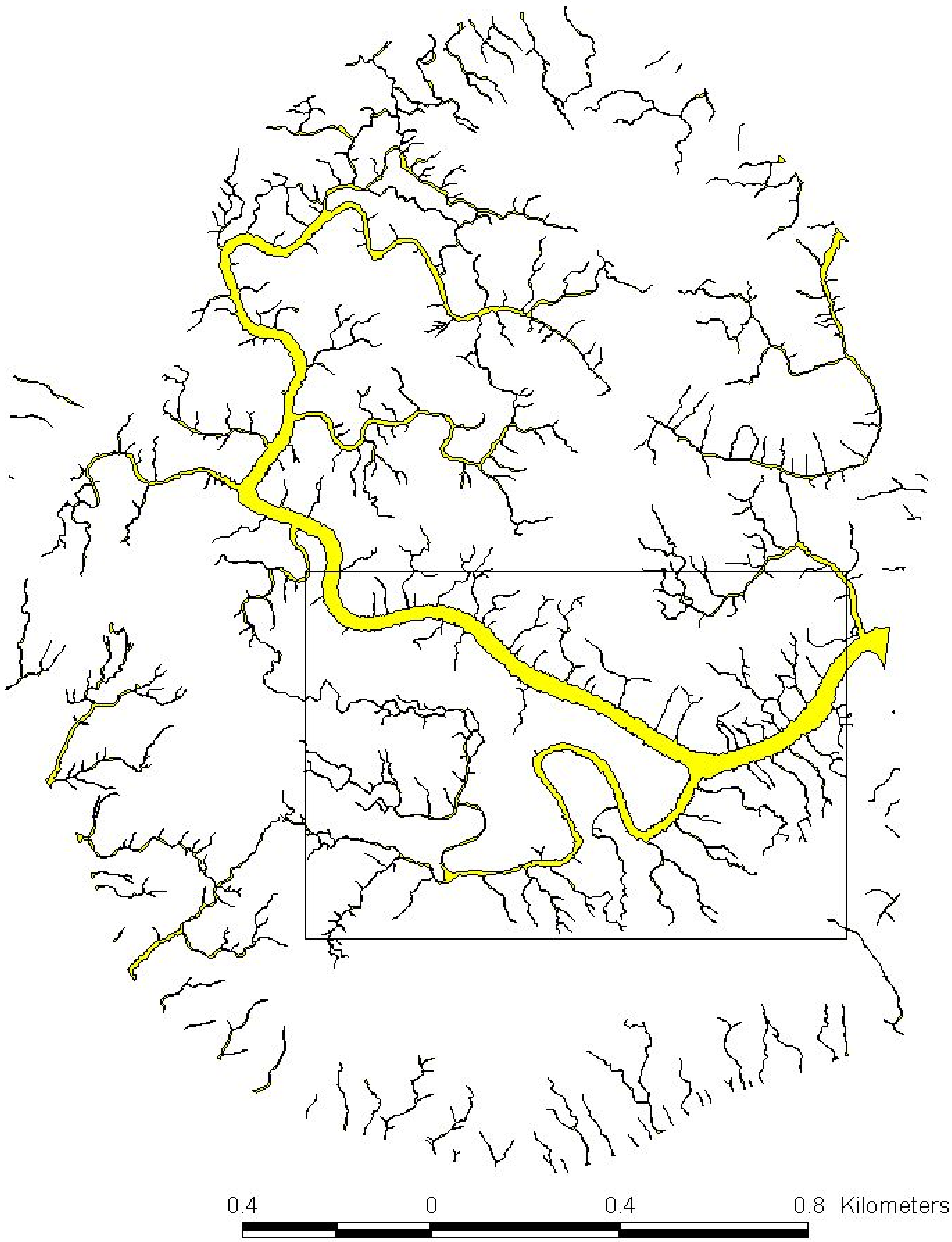

Figure 18. Drainage Patterns of the Natural Marsh in 1999. The boundaries of channel beds were digitized to form drainage area polygons. The box depicts the area enlarged in Fig. 19.

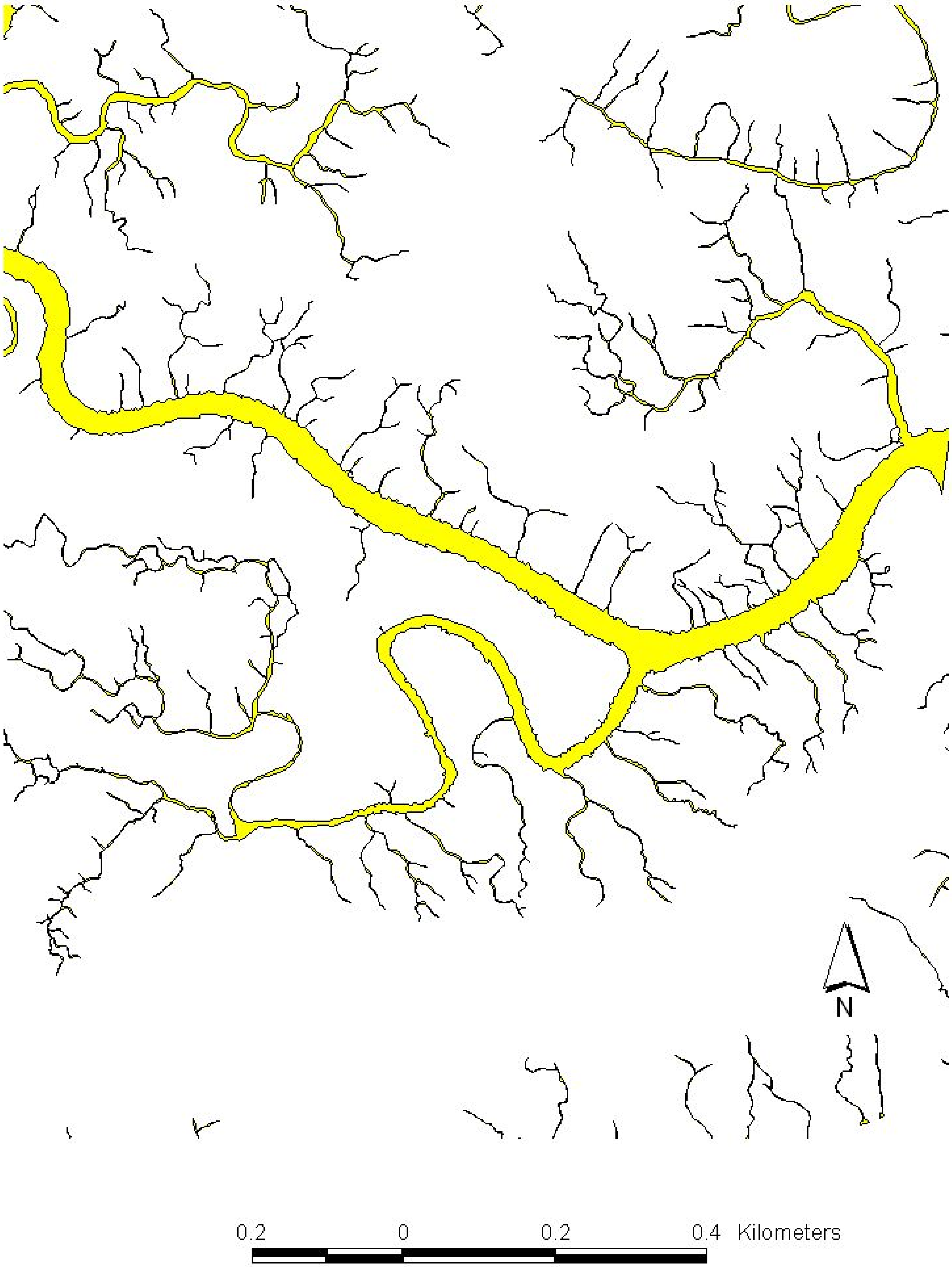

Figure 19. Enlargement of the Drainage Patterns in the 1999 Natural Marsh. A section has been enlarged to more easily depict some of the characteristics of the drainage system. Note the highly branched, sinuous channels. The bottom of the figure depicts a parcel of marsh with few drainage channels that is not directly next to a large channel.

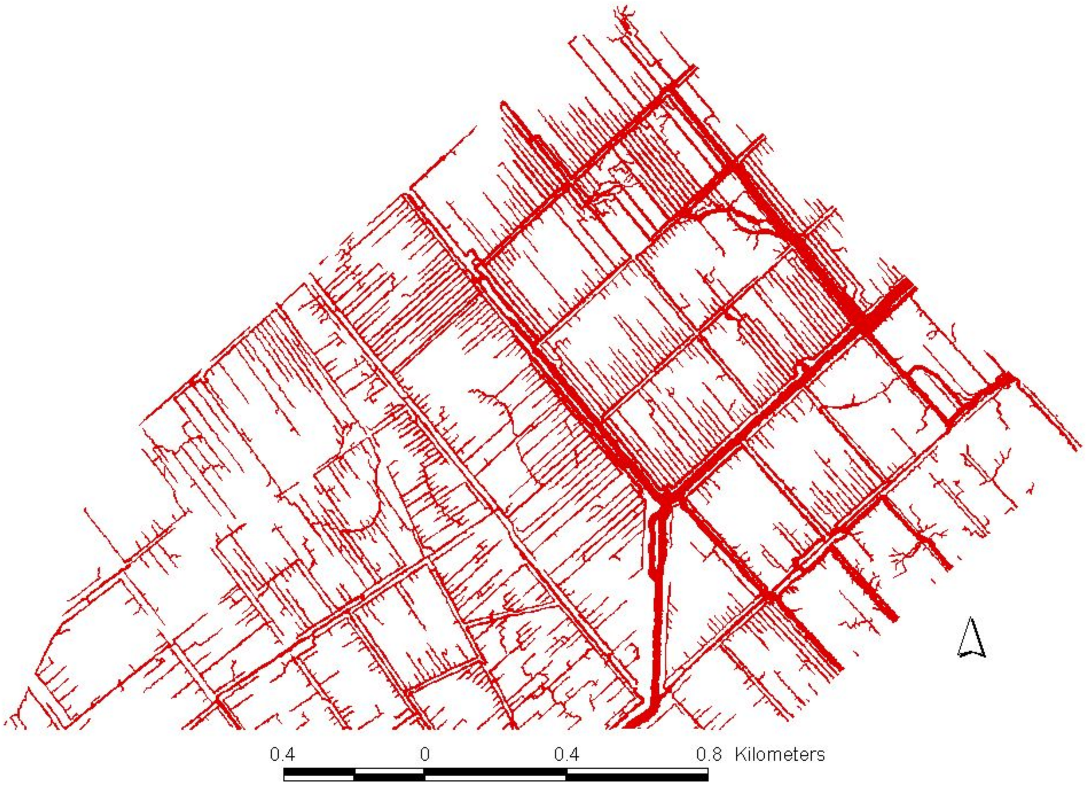

Figure 20. Drainage Patterns of the Restored Marsh in 1949. The boundaries of channel beds were digitized to form drainage area polygons.

Figure 21. Drainage Patterns of the Restored Marsh in 1999. The boundaries of channel beds were digitized to form drainage area polygons.

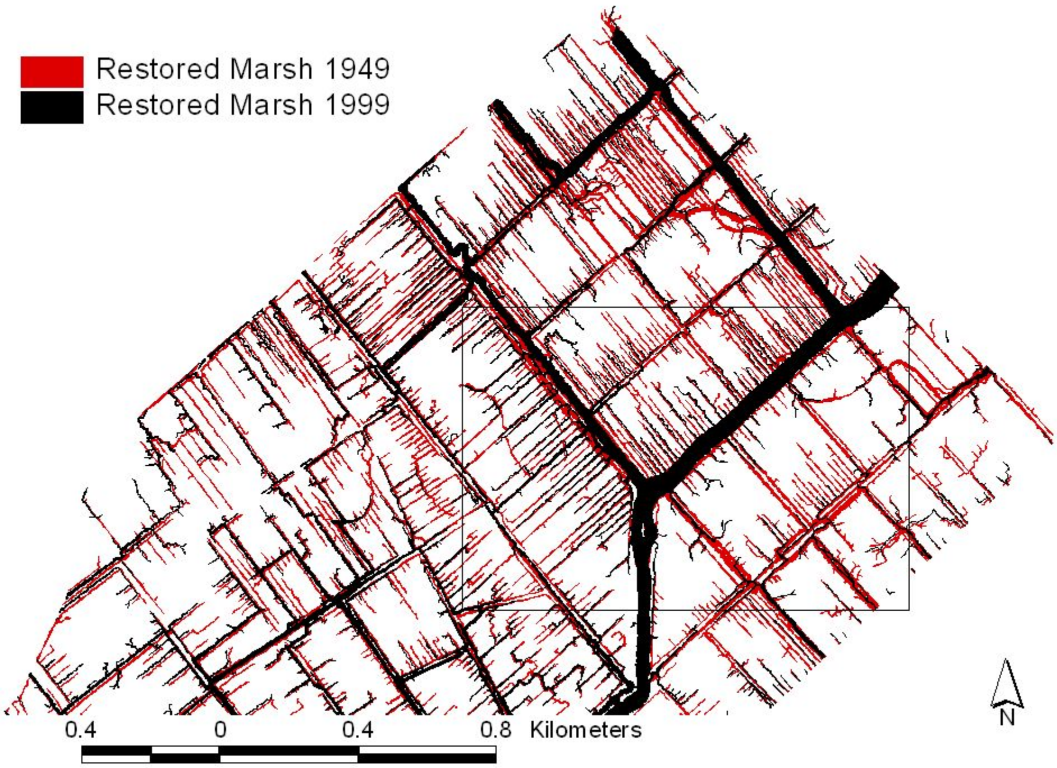

Figure 22. Overlay of the 1949 and 1999 Restored Marsh. An overlay of the two marshes provides an opportunity to examine changes that have occurred over fifty years, particularly channel loss and changes in channel widths. The box depicts the area enlarged in Fig. 21.

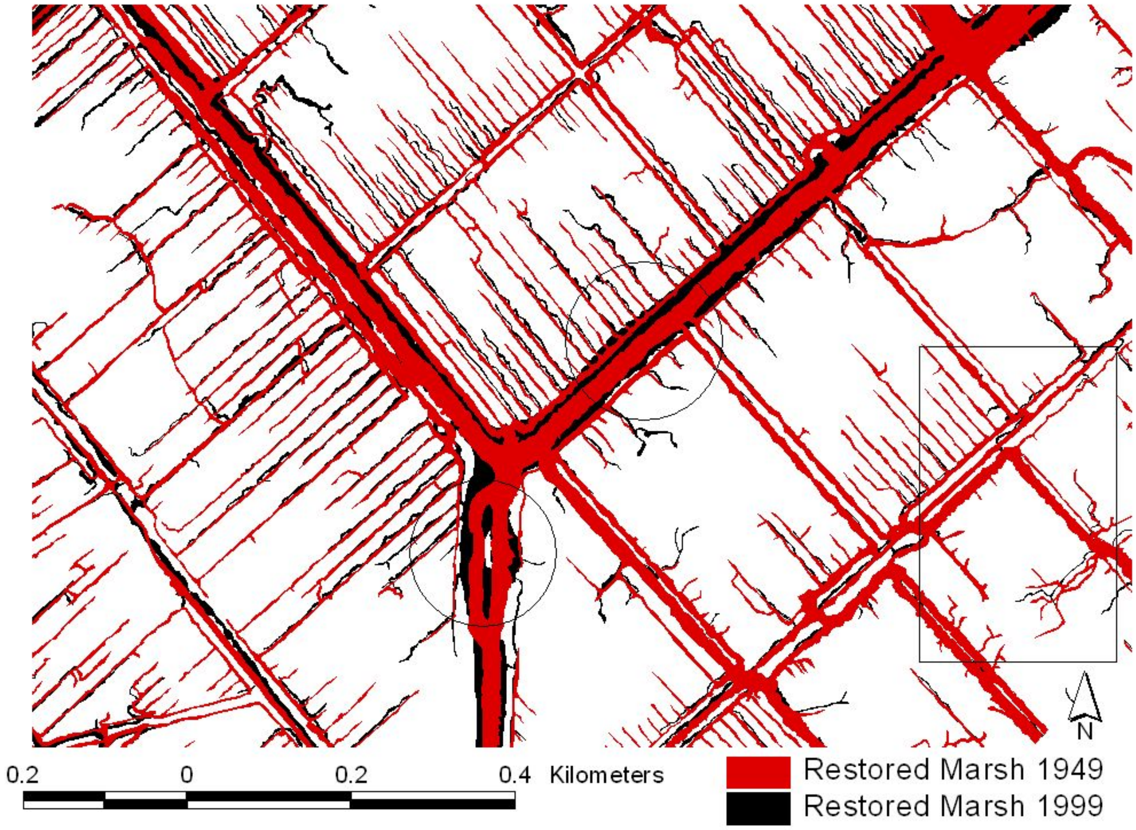

Figure 23. Enlargement of 1949 and 1999 Restored Marsh Overlay. A section of the overlay of the restored marsh in 1949 and in 1999 has been enlarged to more easily see the changes that have occurred over a fifty year time period. Near the center of the figure is an example of doubled narrow channels joining to form one wide channel. Bottom center of the figure depicts examples of the reduction of marsh islands in the center of channels. In the bottom right, there has been an extreme reduction in the density of channels that are not directly connected to the dominant channel pathways. In the center of the enlargement, an increase in sinuosity of first order channels can be seen.

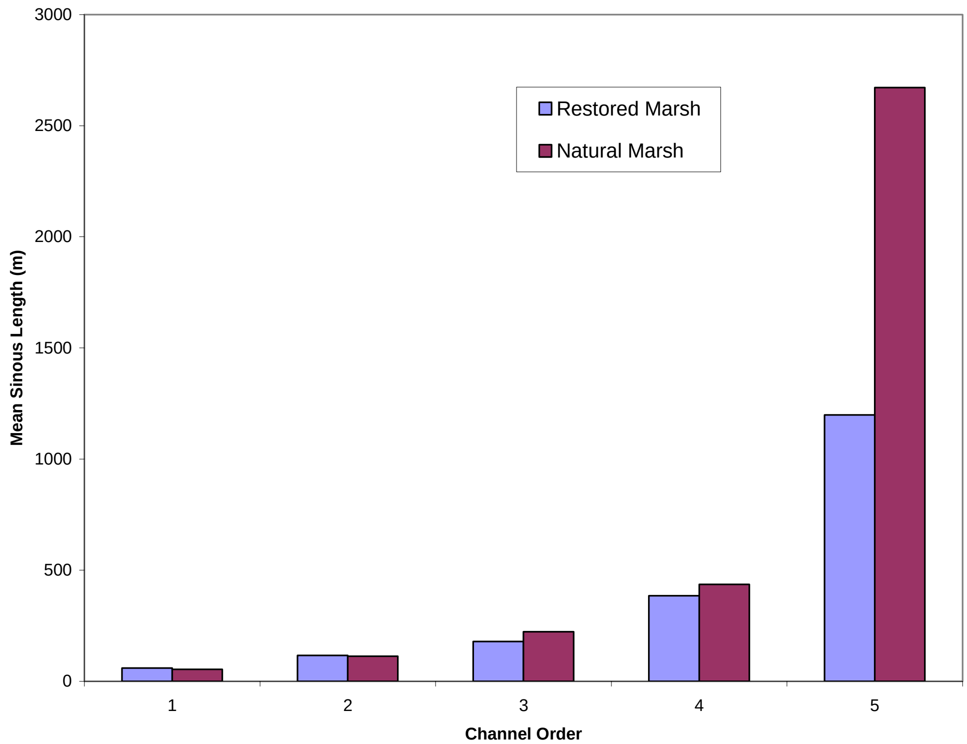

Figure 24. Mean Sinuous Length Plotted by Channel Order for the Restored and Natural Marsh. The mean sinuous length is calculated by dividing the total sinuous channel length for each order by the number of channels for that order.

Figure 25. Overlay of the Centerline Digitizations of the 1949 and 1999 Restored Marsh.

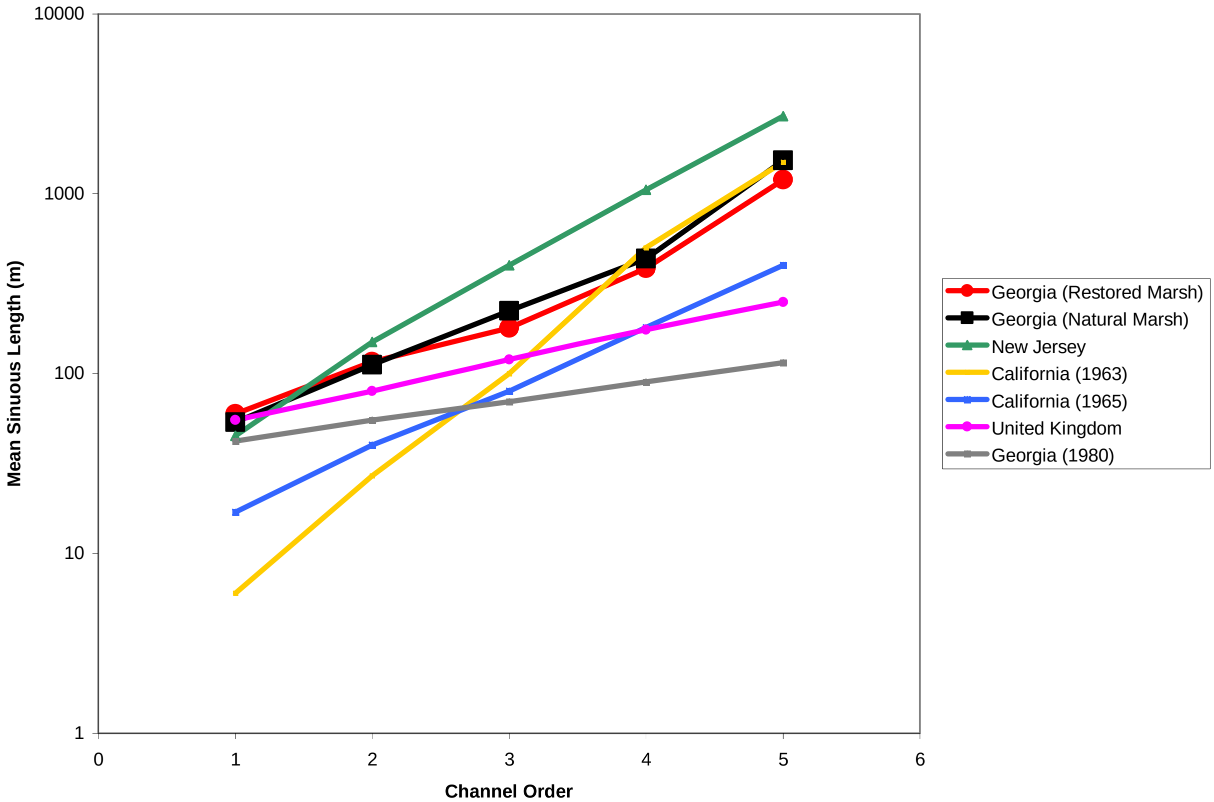

Figure 26. Mean Sinuous Length by Channel Order. This figure depicts the mean sinuous length of channels by order for the restored and natural marsh in 1999. Studies of natural systems in Georgia, California, England, New Jersey, and California are also shown for comparison. Data for Wadsworth (1980) and Pestrong (1965) were taken from their respective papers, but data for Myrick and Leopold (1963), Pethick (1980), and Zeff (1999) were estimated from figures in Zeff (1999).

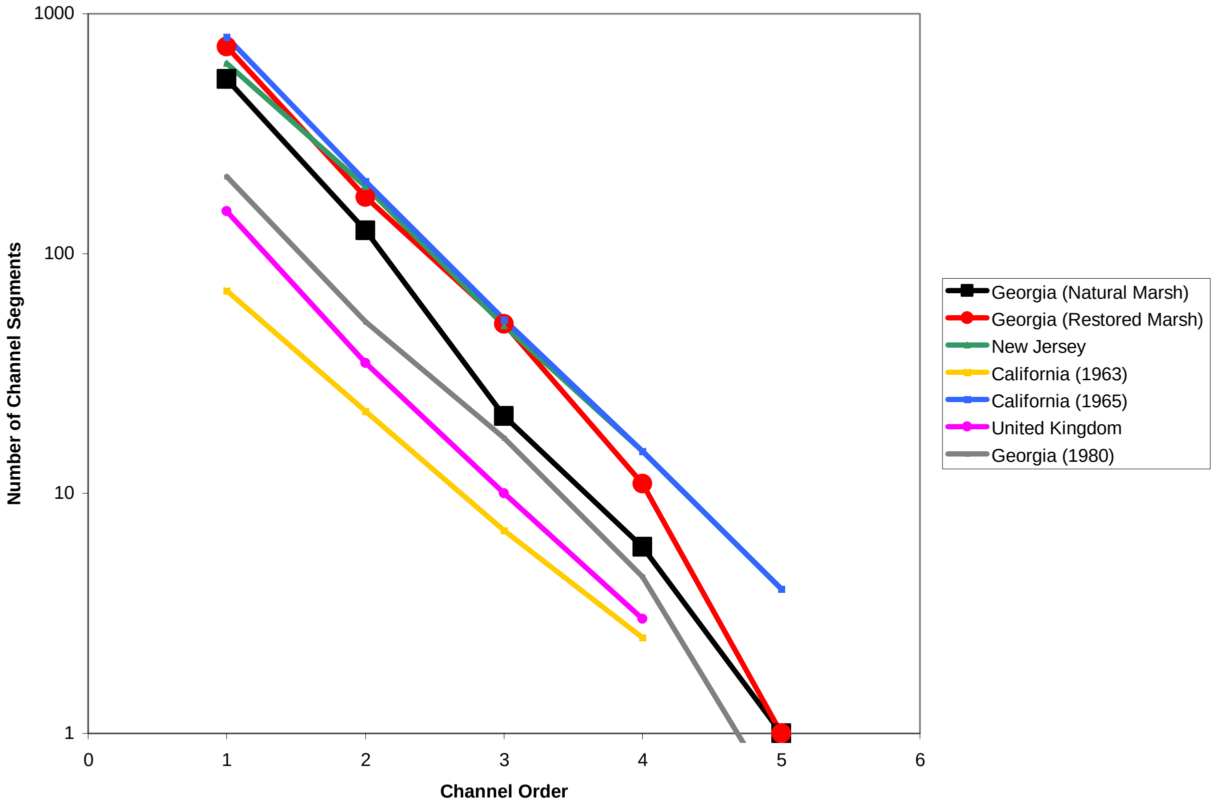

Figure 27. Number of Channel Segments per Channel Order. This figure depicts the number of channel segments for each order channel for the restored marsh and natural marsh. Studies of natural systems in Georgia, California, England, New Jersey, and California are also shown for comparison. Data for Wadsworth and Pestrong were taken from their respective papers, but data for Myrick and Leopold, Pethick, and Zeff were estimated from figures in Zeff 1999.

CHAPTER 4

CONCLUSIONS

This study evaluated the success of tidal marsh mitigation in the brackish portion of the Ogeechee River in coastal Georgia in an area that was extensively altered for rice cultivation. I examined the short-term changes that occurred in a recently restored impoundment site (the Tucker Mitigation site) as well as the long-term changes that occurred in a formerly impounded site that was restored almost 100 years ago. Short-term changes were quantified by measuring hydrologic conditions, vegetation, sediment characteristics, and water quality both before and immediately after flow was returned to the impoundment. These observations were compared to conditions in a nearby reference marsh, which was a formerly impounded marsh also used in rice cultivation. The Tucker Mitigation Site showed dramatic signs of improvement over the five years since the perimeter dike was breached. High levels of sediment deposition occurred, which allowed marsh vegetation to colonize in some areas. Total percent vegetation cover is still fairly low, but there was an increase in vegetation after two years and subsequent observations have seen that trend continue (personal observations). Water conditions improved as tidal range increased to reference levels and water quality has approached conditions similar to those in the reference marsh.

Because of the criticisms of evaluating mitigation success after only a few years of monitoring, a formerly impounded marsh (the same area that was used as the reference site in the short-term study) that was restored in the early 1900s was studied to examine how conditions change over time. Using this information, predictions could then be made for the Tucker site. I chose to focus on drainage characteristics because of the extensive ditching and diking that has occurred in coastal Georgia and South Carolina because of past rice cultivation and mosquito ditching practices. In aerial photographs, it becomes easily apparent which areas were historically used for these practices due to the straight channel patterns that persist in these marshes, despite the cessation of rice cultivation in the early 1900s. Since many studies cite the importance of a proper drainage network for nekton production, I thought it would be important to examine how drainage channels change over time in these formerly impounded marsh sites, which are frequent along the coast of Georgia and South Carolina.

Drainage characteristics were examined in the restored marsh over time and these changes were compared to drainage in a natural marsh. The geomorphic calculations suggest that the restored marsh provides valuable habitat for nekton due to the high drainage densities and the elevated amount of edge habitat in comparison to the natural marsh. In natural marsh systems, edge habitat is created by highly branched, sinuous channels that dissect marsh parcels, creating edge habitat. Despite low sinuosity in the restored marsh, the extremely dense network of straight channels provided more edge habitat than in the natural system. The importance of edge habitat has been demonstrated for numerous fisheries and it appears that the restored marsh habitat provides close to twice the amount of edge habitat than the natural marsh.

Whereas the geomorphic calculations in this study suggest that the restored marsh may be providing more valuable habitat than the natural marsh, there is some evidence that suggest that additional research should be completed to adequately address not only the amount of available edge habitat, but also the quality of habitat the marsh can provide. There is evidence that inside the marsh edge there are higher elevations than found in the restored marsh, possibly from reduced tidal flushing due to high channel velocities and steep channel edges, which would reduce access to the marsh by nekton. Additionally, rivulets in the natural marsh were likely to have been overlooked due to the scale of photographs in this study. The addition of rivulets to the natural marsh would have resulted in higher estimates of drainage density and edge habitat. It is likely that the amount of edge habitat is equal in both the restored and natural marsh, but the actual access to the marsh surface is reduced in the restored marsh due to steep creek sides and absence of rivulets.

Results from the study were intended to not only evaluate the success of a wetland mitigation project, but also to make recommendations for the establishment of a proper drainage network at the Tucker Mitigation Site. Restoration has been achieved in many former rice impoundments through the natural deterioration of dikes, resuming hydrologic connections. While it is likely that the Tucker Mitigation Site will reach reference levels of similarly restored former impoundments in terms of vegetation cover, sediment characteristics, water quality, hydrology, and drainage patterns, the likeness of these formerly impounded marshes to natural systems has yet to be determined. A century after hydrologic restrictions were removed in the restored marsh site, natural drainage patterns have yet to develop and no significant changes were observed after 50 years of study using aerial photography. Therefore, one can expect that the Tucker Mitigation site will not likely develop natural drainage networks over the next 100 years.

Future studies should address nekton edge usage in man-made and ditched marsh habitats to determine habitat quality in altered environments. In addition, it will also be important to determine how steep channel edges and high velocity straight channels affect the length of tidal inundation on the marsh and the implications that this has for nekton and other organisms. This information will be useful in the restoration of formerly impounded marshes, as well as in the design of channels in created marshes.

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