EVALUATING THE IMPACTS OF ENVIRONMENTAL PARAMETERS ON SHORELINE EROSION AND RELATED ASPECTS: ASSESSING THE CURRENT STATUS OF VEGETATION, SEDIMENTS, AND BIOTA

A thesis submitted in partial fulfillment of the requirements for the degree

MASTER OF SCIENCE

in

ENVIRONMENTAL STUDIES

by

LINDSAY GOODWIN
2007

at

THE GRADUATE SCHOOL OF THE COLLEGE OF CHARLESTON

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ABSTRACT

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Erosion and shoreline loss represent an eminent hazard in coastal zones. Erosion threatens both lives and valuable property in coastal communities. Shoreline change research has focused primarily on exposed shorelines. This study has focused on shoreline change of an estuarine river system in Bluffton, South Carolina. Two methods were employed to analyze shoreline change. First, short-term erosion was measured at 47 field stations for a term of 19 months. Second, long-term shoreline change rates were calculated from historical imagery (1965-2006) utilizing ArcGIS and the Digital Shoreline Analysis System (DSAS) extension created by the US Geological Survey (USGS). Mean short-term erosion at the 47 field stations was 0.4 meters of erosion per year. Mean long-term erosion assessed at the 498 transects generated by the DSAS was 0.25 m of erosion per year. In addition to the short term and long term erosion rates other natural factors influencing erosion; such as, stem density, below-ground biomass, flow, and sediment composition, were also assessed. A positive relationship occurred between belowground biomass and short-term erosion rates; however, no other relationships existed between the above factors and short-term rates of erosion.

The study found that mean long-term erosion assessed only at the 47 field stations through DSAS analysis was 0.4 meters of erosion per year. This study supports the understanding that the estuarine system examined is predominantly unaffected by anthropogenic activity. The development boom that is evident throughout South Carolina has not affected this system at the present time as shown by the long-term and short-term erosion rates that are nearly equal.

ACKNOWLEDGEMENTS

I would like to thank the Palmetto Bluff LLC, Dr. Loren Coen, Dr. Linda Walters, and Paul Sacks for their generous support of this project through the SC Sea Grant Consortium, and my committee members Dr. Norm Levine, Dr. Keith Walters, Dr. Virginia Shervette, and Ms. Saundra Upchurch for your encouragement and support. I would also like to thank the many people who assisted with the various aspects of this research including, Claudia Jendron, Andy Hollis, Majbritt Bolton-Warberg, Michael Hodges, Amanda Powers, Ben Dyar, Lee Taylor, Charlie Kaufman, and especially the wonderful people who have made up the SRS section both past and present. Thank you to Dr. Virginia Shervette for your patients and relentless support. A special thanks to Doug Hancock and Patty Richards at Palmetto Bluff for allowing me access and use of the resources at Palmetto Bluff (and Doug thanks for the smoothies!). Lastly, thank you to my loving Mother for your encouragement and financial support and to Bethany Fiem for putting up with me and supporting me.

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INTRODUCTION AND LITERATURE REVIEW

GENERAL OVERVIEW

Erosion and shoreline loss represent an eminent hazard in coastal zones. As coastal areas continue to attract more people and community infrastructures are threatened by shoreline loss, an increased need for accurate information concerning past and present shoreline change exists. Erosion threatens human lives and valuable property in coastal communities residing along both exposed and sheltered coastal areas. Until recently, shoreline change research has focused primarily on exposed shorelines. However, current increases in coastal populations have prompted a rise in development along sheltered shorelines. According to Morton and Miller (2005), sheltered shorelines, such as estuaries and lagoons, comprise the largest expanse of eroding shoreline in the South Atlantic US. Few studies have examined shoreline change of sheltered coasts and fewer have determined causal factors related to shoreline change and the resulting loss of marsh habitat along sheltered shorelines (Morton 2003). In order to address the paucity of research concerning erosion in sheltered shorelines, the current study sought to provide a baseline analysis of shoreline change along the undeveloped sheltered shoreline of Bluffton, South Carolina. In addition, this study sought to better understand some of the processes controlling shoreline movement in the study area.

SHELTERED SHORELINES

Sheltered shorelines are defined as coastal areas protected significantly from wind-driven waves and are often located within estuaries (National Resource Council

2007). As such, they are characterized by low energy conditions that foster habitats and ecological communities including most estuarine habitats. Estuarine environments are one of the most productive ecosystems in the world on both an environmental and economic level (NCCR II 2005). Estuaries intercept and transform polluted waters, buffer shorelines from storm events and flooding, and provide critical habitat for numerous organisms (Boesch and Turner 1984, Mitsch and Gosselink 1986). Estuarine habitats such as marshes, oyster reefs, mud flats, and salt pans function as sites for breeding, feeding, and shelter for economically and ecologically valued fishes and invertebrates (Boesch and Turner 1984, Day et al. 1989). In the southeastern US, the existence of large drainage basins and resultant sediment loads have produced extensive marsh systems within estuaries (Roman, C.T. et. al. 2000). Marshes are a major contributor to overall estuarine productivity. A number of factors, including climate change, sea-level rise, and coastal development, threaten estuarine marsh systems. Although many of the same factors influencing erosion of exposed shorelines are present in sheltered shoreline systems, wind-driven wave energy and tidal currents are moderate in sheltered shorelines, but remain important factors influencing shoreline erosion rates (NRC 2007).

Sheltered shorelines have characteristically shorter reaches and smaller littoral cells than open coast shorelines (NRC 2007). The littoral cell is characterized by three zones; the erosion zone, transport zone, and deposition zone (NRC 2007, Keller 2008). The erosion zone usually occurs at the cutbank or outer bend of the channel. Flow increases at the cutbank during flood tides and slows at the inner bend allowing deposition to occur in the form of point bars (Keller 2008). The characteristically straighter area between erosion and accretions zones is the sediment transport zone (NRC 2007). The channel migrates laterally and forms meanders through these erosion and accretion processes.

NATURAL FACTORS INFLUENCING SHORELINE CHANGE

Shifts in shorelines and changes in habitat result from both anthropogenic and natural causes of erosion. River systems form natural meanders through processes of erosion and accretion (Pye 2000). These natural processes are accelerated and exacerbated through storm events, sea-level rise, wave energy, bioturbation, loss of vegetation, and degradation of land (e.g., Pye 2000, Hughes and Paramor 2004, Morris et al. 2004, Wolters et al. 2005). Land changes occur in the form of channel erosion, beach erosion, bluff erosion and coastal subsidence (Morton 2003). Coastal subsidence occurs in areas where sea-level rise exceeds vertical accretion rates (Hensel et al. 1999).

Local factors play an important role in controlling or contributing to erosion. Phillips (1986) examined shoreline erosion variability in the Delaware Bay over different spatial scales and found that several morphological features of the shoreline varied primarily at the local scale while sediment characteristics varied at larger scales. These features included: cross sectional area of shoreline, slope, and width of peat outcrops. Phillips (1986) also found that erosion rates were highly variable at the local scale and short-term differences were more important than long-term trends. Shoreline morphology and local variations in exposure to wind waves were the primary erosion controls. Phillips (1986) also found that a need existed for further analysis of marsh sediment properties. Currently, soil surveys and geologic surveys lump marsh soils into one classification. Within class variability of marsh soils needs further examination as marsh soil type may play an important role in shoreline change.

Shoreline erosion of estuaries on the southeastern coast of the US has been attributed primarily to waves and currents (e.g. Sanford 1994, Rogers and Skrabal 2001). Wave energy and impact are related directly to water depth, bank slope, shoreline configuration, and nearshore topography (Jackson 1995). Also, areas of low elevation are more susceptible to chronic erosion than erosion caused by storm events (Rogers 2001). As tidal height increases, waves build in height and energy. During severe storm events, tides can reach unprecedented heights allowing larger waves further into the estuary. If the saltmarsh remains exposed during storm events, erosion can be severe; however, if the saltmarsh is submerged, erosion is moderated (Rogers and Skrabal 2001).

Another factor that may play an important role in determining shoreline change is the presence and density of marsh vegetation. Moller and Spencer (2002) reported that wave energy dissipation occurred along a vegetated marsh shoreline in the first few meters of a salt marsh with a 40% reduction of wave energy within the first 10 m. This was consistent for cliffed and non-cliffed sites; however, wave energy attenuation was twice as great at cliffed sites. Non-cliffed, ramped marsh edges gradually dissipated wave energy over the marsh surface. Cliffed edges are susceptible to erosion and cliff retreat caused by wave energy (Moller and Spencer 2002).

Coastal vegetation can enhance stabilization of shorelines and help prevent erosion by retaining sediments in the extensive root systems and buffering wave energy in areas of high stem densities (Gleason et al. 1979). Extensive root systems are able to make a more condensed and cohesive sediment structure by trapping the soil in the root systems (Garafalo 1980, Materne 2000). *Spartina alterniflora* and other grasses such as *Juncus roemerianus* also have been shown to influence channel migration and morphology (e.g., Garafalo 1980). In areas where vegetation has declined, sediments are left exposed to erosive wave and wind energies without the retaining effect of the plant root systems (e.g., Pye et al. 2004).

Root and rhizome densities of salt marsh plants such as *S. alterniflora* and *J. roemerianus* also influence benthic organism abundances (e.g., clams, mussels, or fiddler crabs) (Capehart and Hackney 1989). Bertness (1984) found that mussel byssal threads in *Geukensia* both bind and hold surface sediments in place. Oysters facilitate shoreline stabilization and decrease erosion by creating a barrier against wave energy (Meyer et al. 1997, Chose 1999). Additionally, burrowing animals (e.g., fiddler crabs) may impact the stability of shorelines and affect marsh surface area (Katz 1979). Fiddler crabs are sediment transporters and their burrowing activities influence the nature of sediment structures. The bioturbation resulting from burrowing activities can weaken the sediment binding and, consequently enhance the shoreline susceptibility to wave energy and other erosive forces (Katz 1980, McCraith et al. 2003, Letzsch et al. 1980). However, fiddler crabs also actively remix sediment and increase nutrient availability and root drainage ultimately leading to increased marsh production (e.g. Bertness 1985, Katz 1979, McCraith et al. 2003). The increase in vegetation may counteract sediment instability resulting from fiddler crab burrowing.

The effects of burrowing organisms on sediment erosion remain controversial. For example, in southeast England salt marshes *Nereis diversicolor* has been shown to decrease populations of pioneer plant species (Hughes and Paramor 2004). However, others have rejected this conclusion stating that saltmarsh loss in southeast England should be attributed primarily to sea-level rise, in combination with increased storm frequency and loss of sediment availability (Morris et al. 2004). Research by Pye and van der Wal (2004), also suggest that historic changes in wind and wave energy coincide with periods of rapid erosion. Although storm events can severely alter habitats, in sheltered shorelines storms can be beneficial to marsh systems by importing sediment and nutrients as well as dispersing seeds, thereby promoting marsh growth (Morton 2003).

ANTHROPOGENIC IMPACTS

Coastal development continues to increase at accelerated rates. More than 50% of the total U.S. population resides within coastal areas (National Coastal Conditions Report II 2005). From 1980 to 2003, coastal areas in the southeast U.S experienced a 58% population increase (e.g., Crossett et. al. 2004). Inevitably, this increase in population creates added environmental pressures. Pollution, loss of habitat, decreased water quality, and physical change of the environment are all conditions threatening coastal regions as a result of continued development (Kennish 2001). As development and urbanization increase, land use changes and impervious surfaces threaten environmental stability. Impervious surfaces limit ground water recharge and increase upland runoff thus reducing flood control in the surrounding watershed and increasing pollution (Mallin et. al. 2000, Glasoe and Christy 2004).

With coastal development, recreational activities such as water sports and boating continue to increase at alarming rates. Approximately 77 million beach goers visit the coast yearly and of them, 48 million use recreational motor-boats (National Survey on Recreation and the Environment 2006). Recreation and tourism in coastal states brings in $560 billion dollars a year (National Survey on Recreation and the Environment 2006). Coastal areas have much to offer both in recreation and resources; however, there is a price to pay for this inherent attraction. Several studies have shown that increased boating activities diminish the health of coastal environments (Anderson 1972, 1976, Nanson et al. 1994, Kennish 2001, Bauer et al. 2002). Seagrass scarring, channel erosion, and increased turbidity are some of the direct impacts caused by increased boat traffic (Anderson 2002). Seagrass scarring can displace benthic communities and increased turbidity affects water quality and photosynthesis (Anderson 2002, Bishop 2004). Boat wakes can affect sediment disposition by washing away clays and silts leaving more "coarse" and erodable substrate and also cause undercutting of river banks and shorelines (Bishop 2003). Undercut banks are more prone to slumping, which often results in additional large areas of sediment and associated marsh vegetation breaking off and washing away (Anderson 1972, 1976, Nanson et al. 1994, Bishop 2003, 2004). Construction of Marinas as a result of increased recreational boating also may negatively impact flora and fauna (Bishop 2003). Boat traffic and recreational watersports tend to be concentrated in estuaries due to characteristically low wave energy, warmer water, and public accessibility (Nordstrom 1992 and Bishop 2003).

COMBINED ANTHROPOGENIC AND NATURAL IMPACTS

Distinguishing between natural and anthropogenic causes of erosion can be difficult. Because of this, the need for extensive baseline analysis studies of shoreline change is great. For example, Castillo et al. (2002) quantified vertical and horizontal erosion rates over a 4 year period in the Odiel marshes of SW Spain. The study region was divided into North and South zones. Erosion rates varied from an average of -20 cm/yr in the Northern zone to -25 cm/yr in the Southern zone. The outcome of the study suggested that the southern portion of the marsh, which was subject to significant boat traffic and development, experienced greater erosion than the northern portion, which did not incur the same anthropogenic pressures. Castillo (2002) also asserted that the loss of wetlands as a result of erosion could be explained by sea level rise (Castillo 2002). Without a baseline analysis prior to anthropogenic impacts, natural and human induced changes in shoreline are difficult to distinguish.

Several studies have examined the loss of saltmarsh in southern England. These losses were attributed to: (1) sea level rise; (2) vegetation loss; (3) bioturbation; and (4) a lack of sediment availability caused by constructed seawalls (Pye 2000, Hughes and Paramor 2004, Morris et al. 2004, Wolters et al. 2005). However, salt-marsh loss in southern England has primarily been attributed to coastal squeeze or coastal subsidence resulting from the combination of sea-level rise and decreased sediment availability (Pye 2000, Morris et al. 2004, Wolters et al. 2005). Erosion and accretion are cyclical processes necessary to a healthy marsh system. The removal of large sections of salt-marsh from this system through the construction of seawalls may have detrimental effects on the salt-marsh ecosystem and morphological functions such as sediment transport. Salt marshes that have adequate sediment supply can adapt to sea level rise by moving landward and upward. Sheldon (1968) discussed sedimentation processes of Essex England. Sheldon (1968) found sediment reworking was the most important short-term sedimentation process, while deposition was the most important long-term sedimentation process. Wave action contributes to reworking sediment by bringing sediment into suspension. As waves erode the edge of the salt marsh, the sediment is redeposited during flood tides on the marsh surface. The process moves the salt marsh landward while increasing the elevation (Reed 1988). In the Essex Region, and much of South East England, seawalls prevent the marsh from migrating in this manner and the outcome of the resistance caused by the seawalls is the loss of salt marsh through drowning or "squeezing" outward. French and Burningham (2003) found that sediment supply in the Blyth estuary of Suffolk is sufficient to combat sea-level rise. Changes in salt-marshes of the Blyth estuary are attributed to estuarine morphology rather than sea-level rise as in other southeast England estuaries. Erosional responses in the Blythe estuary result from wind and wave energy, as well as other morphodynamic changes.

Chose (1999) examined two comparable tidal marsh regions in the southeastern US, North Inlet and Murrells Inlet located in South Carolina. Murrells Inlet is subject to significant development and boat traffic while North Inlet is considered a more pristine study site. The purpose of the study was to determine what factors had the greatest influence on erosion, both anthropogenic and non-anthropogenic, in the two areas of study. Chose (1999) concluded that steep slopes with high percent sand content were more erosive than gradual slopes. The study also found that the presence of oyster beds significantly decreased rates of erosion. Chose (1999) also examined bioturbation, but did not find a correlation between fiddler crab burrow density and erosion.

EROSION CONTROL DEVICES

Most erosion control devices were developed for exposed shorelines. Seawalls, revetments, breakwaters, groins, and jetties are examples of hard-stabilization devices used to control erosion. In general, when used in sheltered shoreline areas, these hard-stabilization devices are utilized on an individual property lot basis and do not necessarily protect an intact stretch or reach of sheltered shoreline (NRC 2007, Douglass and Pickel 1999). Hard-stabilization devices have associated negative impacts. For example, seawalls and revetments only protect the structure directly behind them and remove sediment from the immediate beach, while wave energy continues to cause scouring (NRC 2007, Douglass and Pickel 1999). Flanking along margins of a seawall then causes further erosion. In the state of South Carolina, seawalls and revetments were recently outlawed as a form of shoreline stabilization because of their negative impacts on adjacent areas and the loss of beach habitat (1990 S.C. Acts 607; 1990 S.C. S.B. 391; 1990 S.C. R. 748).

The above mentioned erosion control devices do not prevent erosion but, instead displace erosion. There has recently been a shift to soft stabilization devices such as geotextile tubes—which act similarly to groins and jetties but are not permanent structures——nourishment projects, dune plantings, and living shorelines (vegetation plantings and constructed oyster reefs). Seawalls are a commonly seen feature in developed areas of protected shorelines. There has been a recent push toward living shorelines due to the negative impacts of seawalls and the environmentally positive impacts of vegetation plantings and constructed oyster reefs (Davis and Fitzgerald 2004, NRC 2007).

RELATED RESEARCH

The Palmetto Bluff Development located in Beaufort South Carolina consists of 20,000 acres primarily bordered by the May River, the Cooper River, and the New River. The developers of Palmetto Bluff wish to minimize the ecological impacts of associated development. Development plans are in place to construct a new village and marina in the Big House region of the community (Figure 1). An environmental impact statement (EIS) was conducted by Research Planning Inc.(RPI) and other contractors and submitted to the South Carolina Office of Ocean and Coastal Resource Management (OCRM). Historical average erosion rates were estimated to be approximately 0.61 m/yr through shoreline mapping. Seven permanent field stations were established by RPI for direct erosion monitoring. The permanent erosion stations were located in the vicinity of the projected marina and monitored every six months. The erosion monitoring area was limited to the projected marina site and no in depth sampling was conducted to determine erosion variables. RPI estimated maximum erosion rates of 60.96 cm/yr along the marina-impacted area (Research Planning Inc preliminary report, 2004).

Building upon previous efforts, South Carolina Department of Natural Resources (DNR) began research in collaboration with researchers from the University of Central Florida (UCF) in 2005. Part of the collaborative efforts between researchers from DNR and UCF included conducting field tests concerning the impacts of boat wakes on oyster reefs and measuring shoreline erosion along the creeks leading to the proposed marina. Although, the area near the Palmetto Bluff Development has few oyster reefs, oyster recruitment was measured to determine the potential for reef development.

THESIS RESEARCH GOALS

This study builds on the goals of the larger SCDNR / UCF grant funded study. The intent of the thesis research is to expand our understanding of erosional trends, examine potential factors influencing erosion in the region surrounding Palmetto Bluff, and provide a baseline assessment of environmental parameters prior to increased development pressures and marina construction. The study area included the navigational pathway to the marina that would inevitably experience increased boat traffic. Specific goals included: 1) Examining historical erosion trends among depositional, parallel, and erosional river morphology zones within the study area through digital interpretation which was important because knowledge of past erosion and accretion processes will aid in understanding current erosion rates and processes; 2). Measuring short-term erosion rates through direct field measurements which was important because current field measurements will help determine present trends in erosion and accretion; and 3) Examining site-specific characteristics that may correlate with short-term erosion rates, which was important because it provides a baseline analysis of natural factors that may be influencing current erosion rates.

MATERIALS AND METHODS

STUDY AREA

The study area is located in Beaufort, near the South Carolina and Georgia border, approximately 3 km inland of the Atlantic Ocean (Figure 1). The study area consisted of two tidally influenced rivers that border the Palmetto Bluff Development, the New River and Cooper River, and a tributary of the Cooper River, Ramshorn Creek (Figure 2). Ramshorn Creek connects to the Cooper River and the Cooper River connects to the New River.

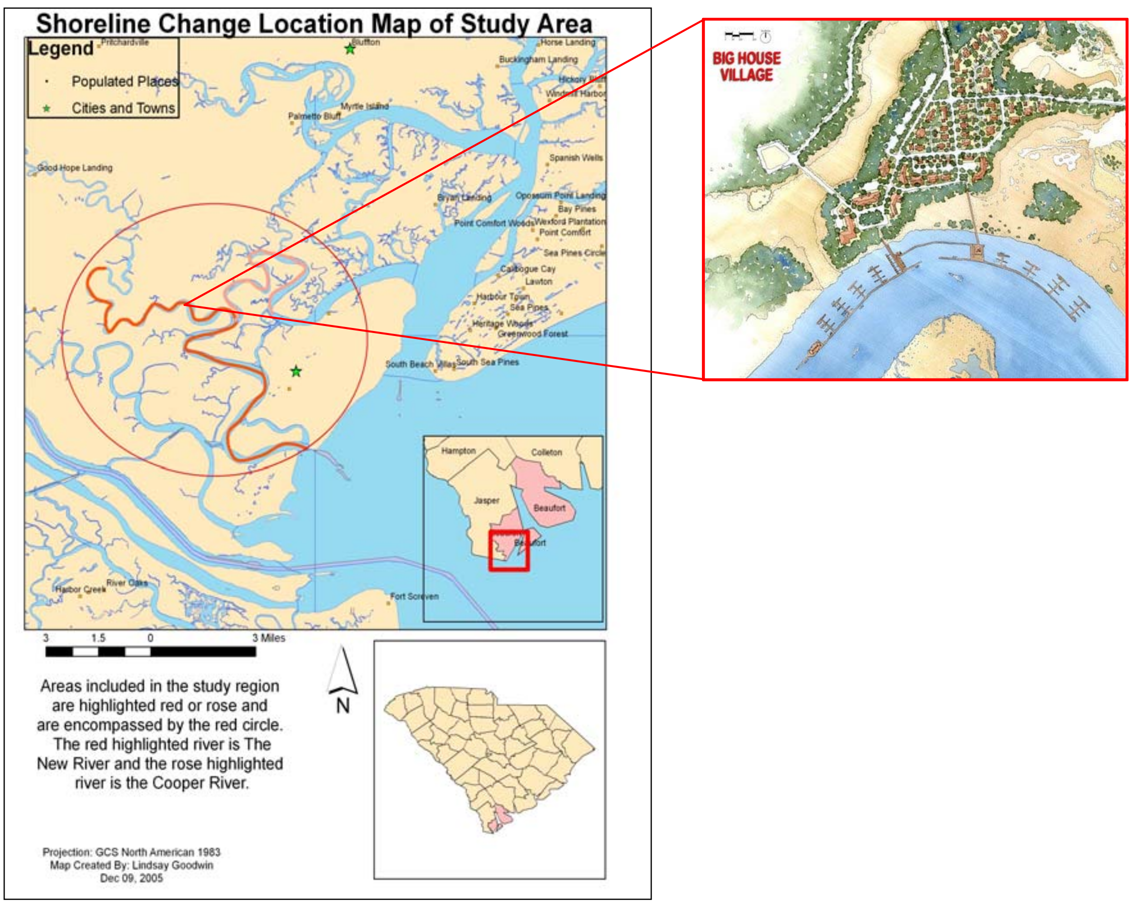


Figure 1: The study area is located in Beaufort, SC, between Hilton Head, SC, and Savannah, Georgia. The Big House Village development location is approximately three miles from the Atlantic Ocean. The expanded view of the Big House Village shows development plans for the community and Marina that will ensue in 2007.

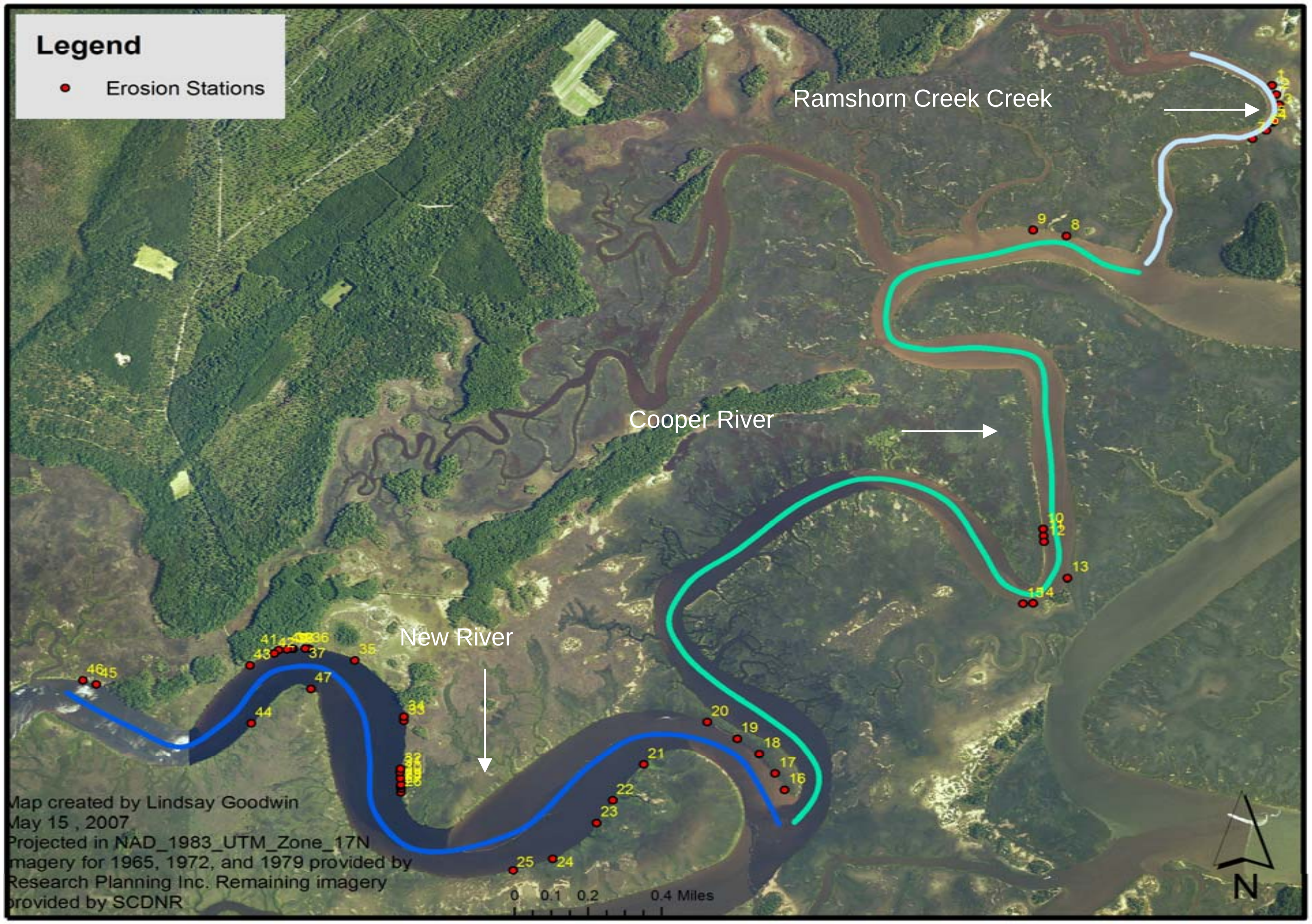


Figure 2. The New River, Cooper River, and Ramshorn Creek are the adjoining waterways of the study area.

The Cooper River is part of the May River/Calibogue Watershed (Figure 3). Recreation and aquatic life are fully supported in the Cooper River (SCDHEC 2003). The channel width of the Cooper River section of the study area ranged from 96–160 m with a mean width of 129 m. The channel width of the Ramshorn Creek section of the study area ranged from 50–61 m across with a mean width of 53 m.

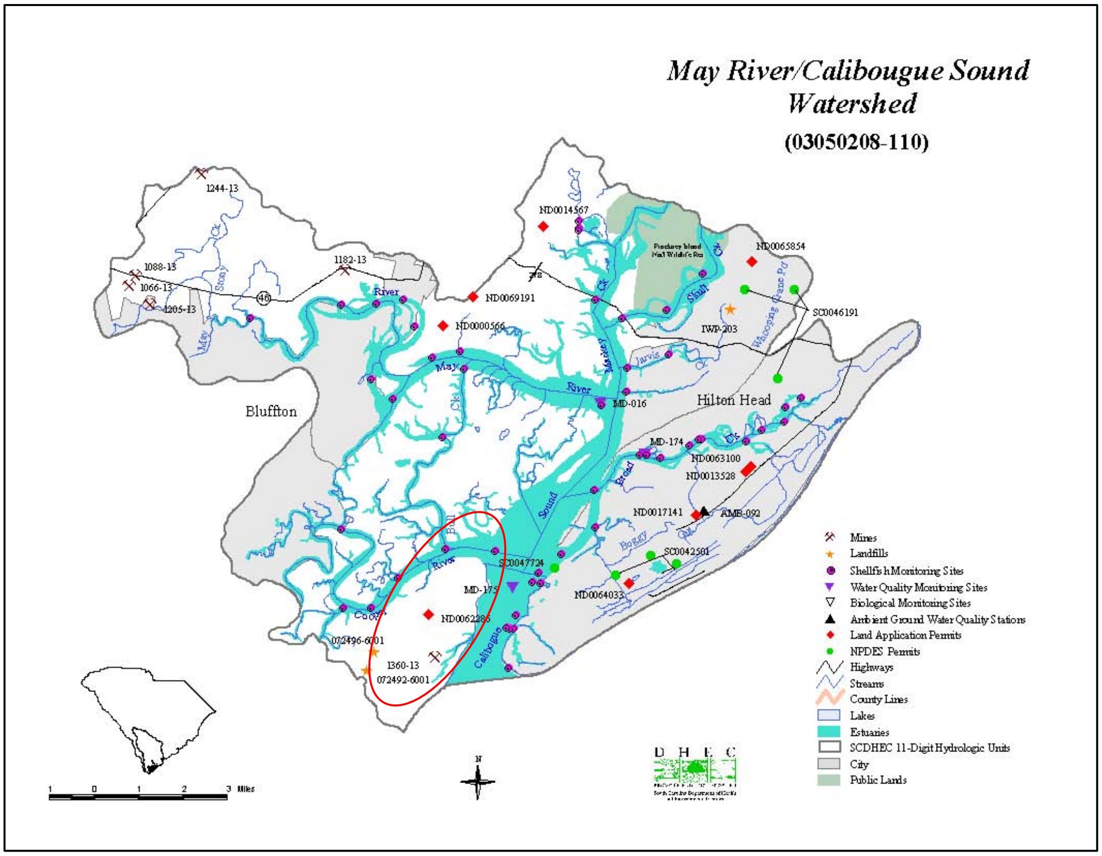


Figure 3. Location map of the Cooper River in relation to the surrounding watershed. This Figure was taken from the SCDHEC Watershed Water Quality Assessment (2003). The Cooper River portion of the study area (the area circled in red) is part of the May River/Calibogue Sound watershed.

The New River is part of the Great Swamp, New River, and Wright River watersheds (Figure 4). The headwaters of the New River begin at the Great Swamp and the base of the New River connects with Ramshorn Creek (SCDHEC 2003). The channel width of the New River section of the study area ranged from 159–420 m with a mean width of 253 m. SCDHEC (2003) has designated the New River as not suitable for recreation due to high fecal coliform bacteria levels and not suitable for aquatic life due to high pH levels.

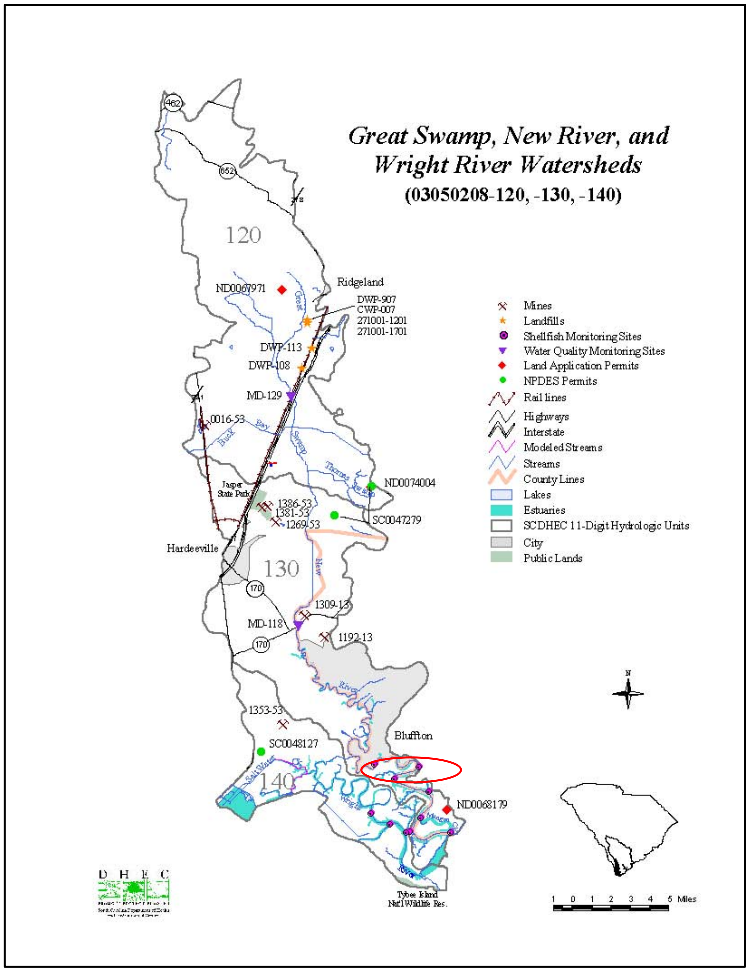


Figure 4: Location map of the New River in relation to the surrounding watershed. This figure was taken from the SCDHEC Watershed Water Quality Assessment (2003). The New River portion of the study area (area circled in red) is part of the Great Swamp, New River and Wright River Watersheds.

A total of 47 intertidal sampling stations were selected in the study area (Figures 2 and 5). The tidal range in the study area was 2–3 m (SCDHEC 2003). The salt marshes are dominated by *Spartina alterniflora* (smooth cordgrass) and *Juncus roemerianus* (black needlerush). The underlying geology is primarily saltmarsh Holocene deposits and Pleistocene barrier island and fringe deposits (Doar 2002).

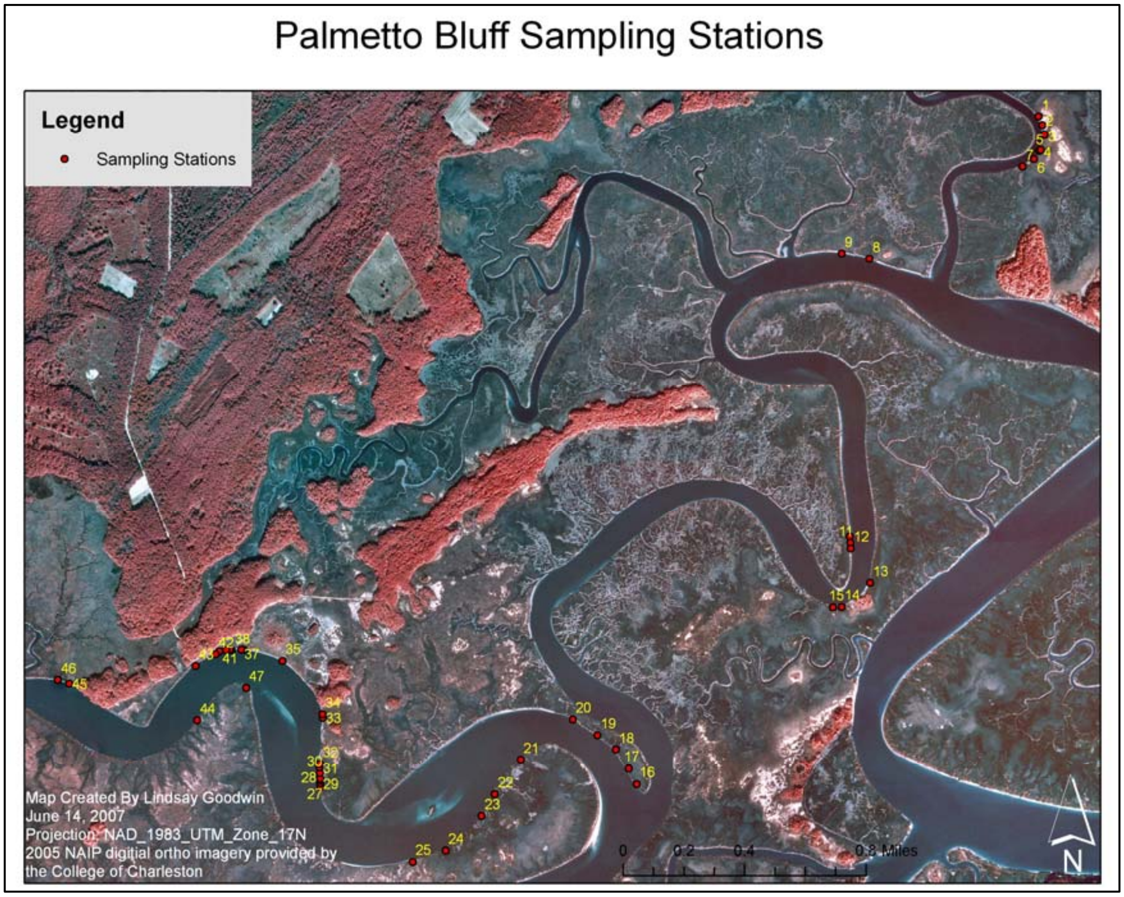


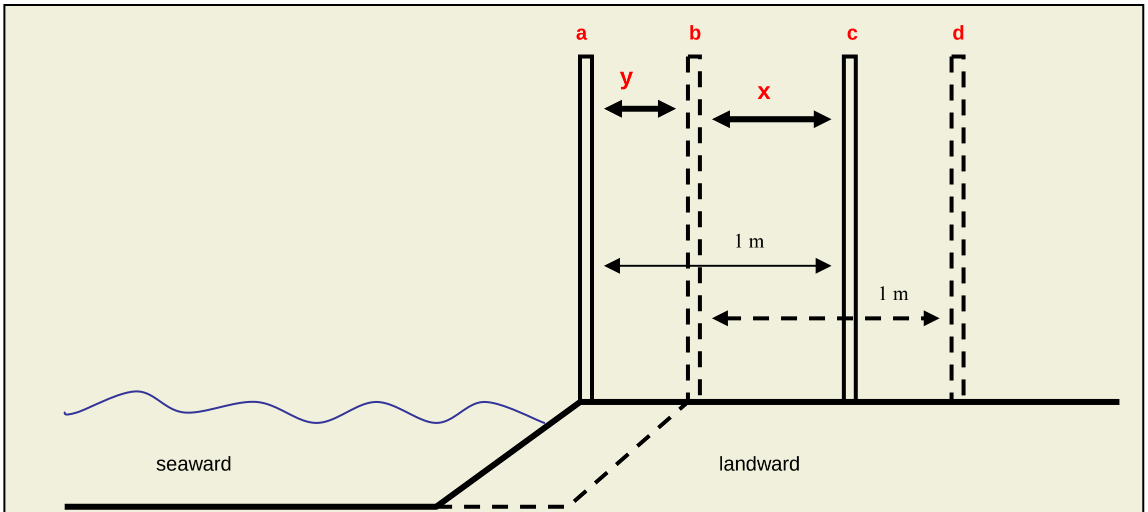
Figure 5: Location of sampling stations (n = 47) within the study area. Sampling stations were deployed as part of a larger ongoing study.

SHORELINE CHANGE ANALYSIS

Short-term Direct Determination of Shoreline Change

Short-term erosion rates were quantified by directly measuring shoreline change over time. Initial and subsequent shoreline positions were assessed for a period of 19 mo at 47 stations. Seven stations were located along Ramshorn Creek, eight stations were located along the Cooper River, and the remaining 31 stations were located along the New River (Figure 2). A total of six measurements were made for each station over the 19 mo period. Initial deployment for short-term shoreline change rate measurements occurred June 2005. Subsequent erosion measurements were collected: July 2005, August 2005, September 2005, January 2006, May 2006, August 2006, and December 2006. Erosion was measured at all stations but one for a total of 19 months. At station 16 erosion poles were lost after month 11 so, the erosion rate was calculated based on a total of 11 months.

Shoreline edges were identified as the farthest landward horizon where differences in elevation occurred (Figure 6). In June 2005, for each of the 47 stations, two poles (3/4" PVC, 2 m long) were used to measure changes in shoreline throughout the study period. These poles are subsequently referenced as the "landward" and the "shoreline" poles. The shoreline pole delineated a shorelines edge and the landward pole was place 1 m inland from the shoreline pole (Figures 6 and 7). In subsequent sampling periods, the distance from between the landward pole and the current shoreline edge was measured. At that time the landward pole was reset and placed directly 1 m inland from the new position of the shoreline pole (Figures 6 and 7).



a., Pole indicating initial shoreline; **b.,** Initial pole set 1 m inland. Dashed line represents where the shoreline was located on subsequent sampling. **c**., Indicates the pole marking the new seaward (shoreline) edge. **d.,** Indicates new pole marking 1 m inland. **y.,** is the distance from the original shoreline to the shoreline edge. x. Indicates the distance measured from pole **b**. to the new seaward edge.

Figure 6: The shoreline pole placement for short term erosion rate assessment.

Figure 7: Field measurements of shoreline change. A measuring stick was used to reset the seaward pole and landward pole to a distance of 1 m apart.

The incremental changes (y_t) in shoreline between sampling periods were calculated with the following equation: $y_t = 1 - x_t$; where x equals the distance between the new shoreline and the old landward pole at time t. Cumulative erosion rates and monthly erosion rates were calculated for each of the 47 stations. A cumulative erosion rate was calculated as the sum of all incremental change values and monthly erosion rates were calculated as the cumulative erosion rate divided by 19 months.

Long-term Characterization in Shoreline Change

Long-term shoreline changes (1965-2006) were assessed using digitized aerial imagery. Research Planning Inc. supplied black & white aerial photographs for 1965, 1972, and 1979. Color infrared aerial photographs for 1999 were obtained from the SC Department of Natural Resources GIS Clearinghouse website. The College of Charleston supplied 2005 color infrared digital ortho imagery acquired by the National Agriculture Imagery Program. The Marine Resources Division of the South Carolina Department of Natural Resources supplied 2006 color infrared digital imagery (Figure 8).

Aerial photography from 1965 – 1999 were scanned at 600 dpi and rectified by RPI according to National Mapping Standards; the resolution and pixel size of imagery from each time period is provided in Table 1. The pixel error is equivalent to the pixel size (E_p). Stations 1–9 have no aerial coverage for the year 1979. Stations 1-24 have no aerial coverage for the year 1965. Metadata was not provided from Research Planning Inc for the rectified aerial photographs. In order to assess error in the rectification process (E_r), imagery from 1965, 1972, 1979, 1999, and 2005 was georeferenced using ArcMap 9.0 to the 2006 imagery provided by SCDNR. A minimum of 4 control points were identified per image and an RMS (root mean squared) error was calculated for each year (Table 1).

The vegetation line was used to delineate the shoreline position. The channel margins on the images were digitized on-screen using ArcMap 9.1 (see table 1 for effective resolutions). The digitized shapefiles were laid on top of the aerial photographs for visual inspection and display purposes (Figure 9). The maximum potential error in the digitizing process (E_d) was determined for each photograph. Several points in each layer were chosen to indicate perceived shoreline positions. The distance between points was measured and the average maximum distance determined the maximum potential digitizing error for the specified year (Figure 10). Buffers were created around each ditized shoreline to illustrate the average maximum digitizing error (Figure 11). The total shoreline position error (E_{sp}) is the square root of the sum of errors: Pixel error (E_p) + rectification error (E_r) + the digitizing error (E_d): $E_{sp} = \sqrt{Ep^2 + Er^2 + Ed^2}$. The annual shoreline position error is the E_{sp}/time (Table 1: Morton et al. 2004).

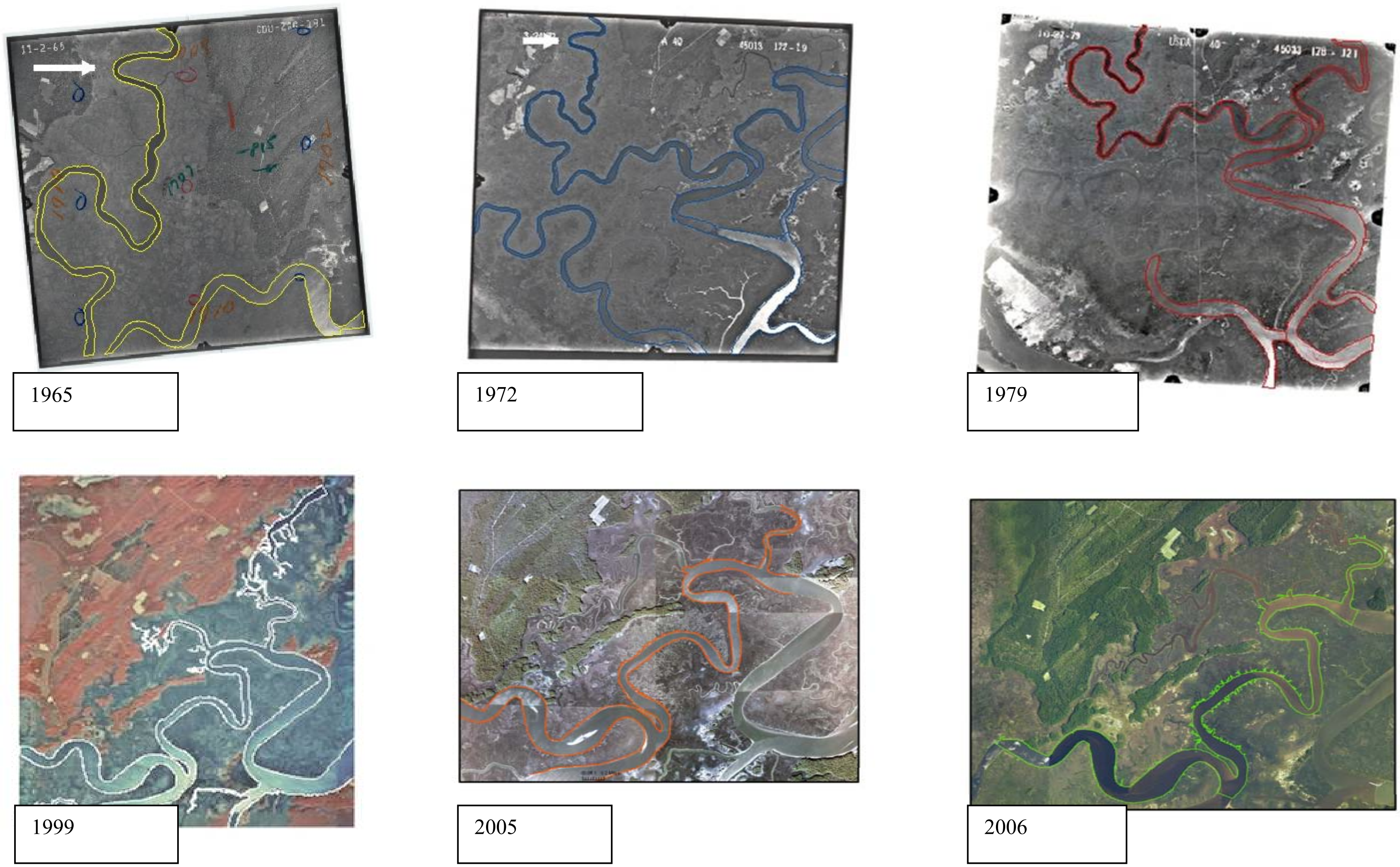


Figure 8: Shorelines were digitized from aerial imagery for the years 1965, 1972, 1979, 1999, 2005, and 2006. Imagery for 1965, 1972, 1979 and 1999 was scanned at 600 dpi.

Table 1: Resolution and error for digitized maps of the study area for each year analyzed. The true resolution was determined by applying the following equation: maps scale $<$ 1:20,000 (Nominal Pixel Size * 39) * 30. The effective resolution is the extent to which the imagery could be magnified and still have easily distinguishable features.

Year	Nominal Pixel Size (m)	True Resolution	Effective Resolution	Positional Accuracy (m)	RMS	Max Digitizing Error	Total Error	Total Annualized error
1965	2 m	1:2000	1:1100	≤ 1.8 m	0.41	3	2.33	0.09
1972	3.5 m	1:4000	1:2000	≤ 4 m	0.10	3	2.57	0.14
1979	3.8 m	1:4500	1:4500	≤ 4 m	0.42	7	3.35	0.30
1999	1 m	1:1100	1:1100	≤ 0.9 m	1.57	3	2.36	0.50
2005	1 m	1:1100	1:1100	≤ 0.9 m	0.18	3	2.04	3.17
2006	0.25 m	1:300	1:300	≤ 0.9 m	0.00	3	1.80	3.01

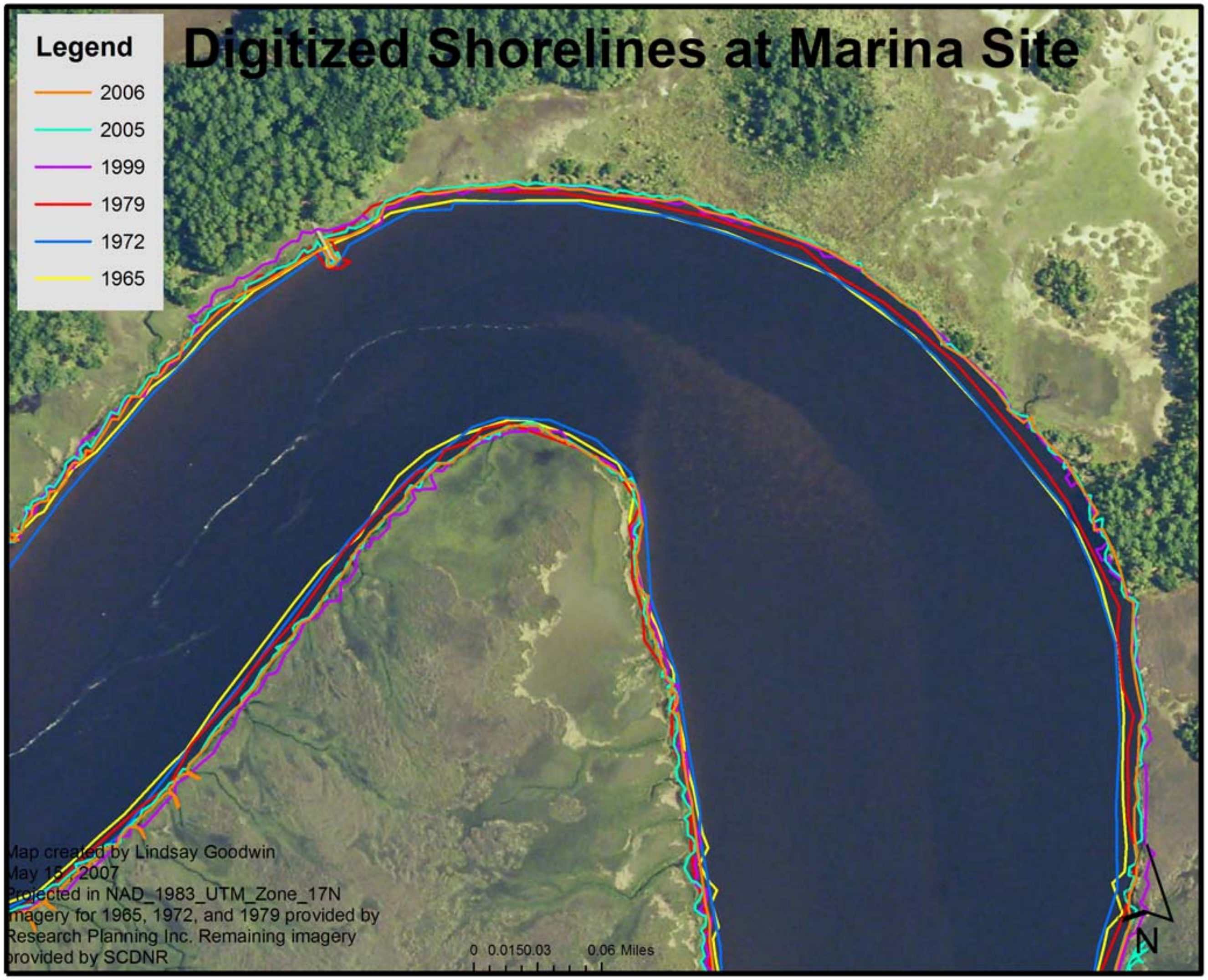
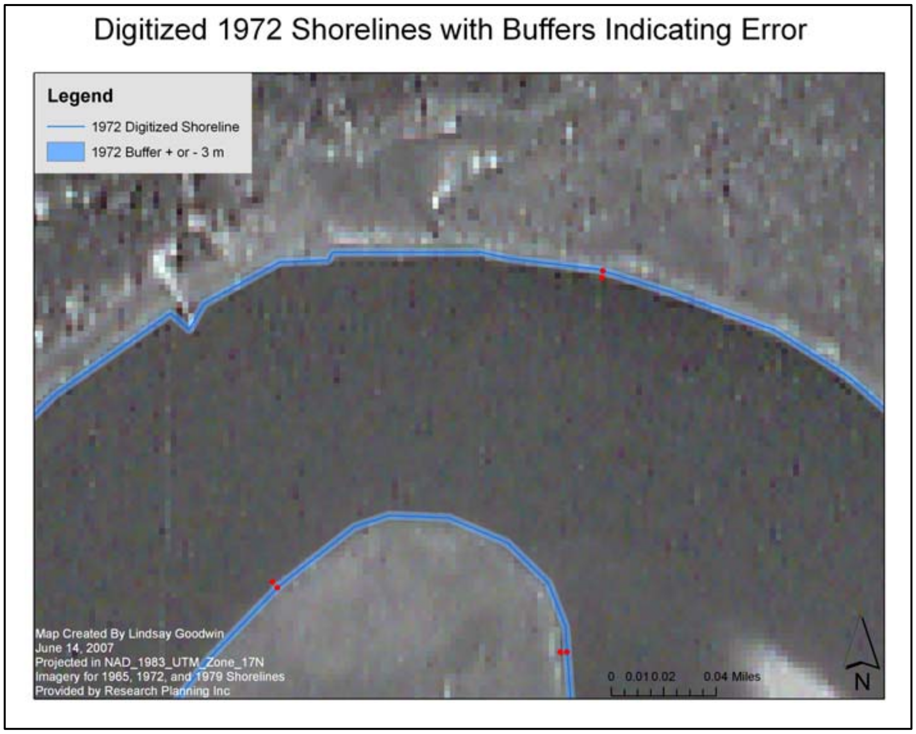


Figure 9. Map showing digitized shorelines overlaid on the 2005 imagery. The outer bank of this region is the projected marina development site.

a)



b)

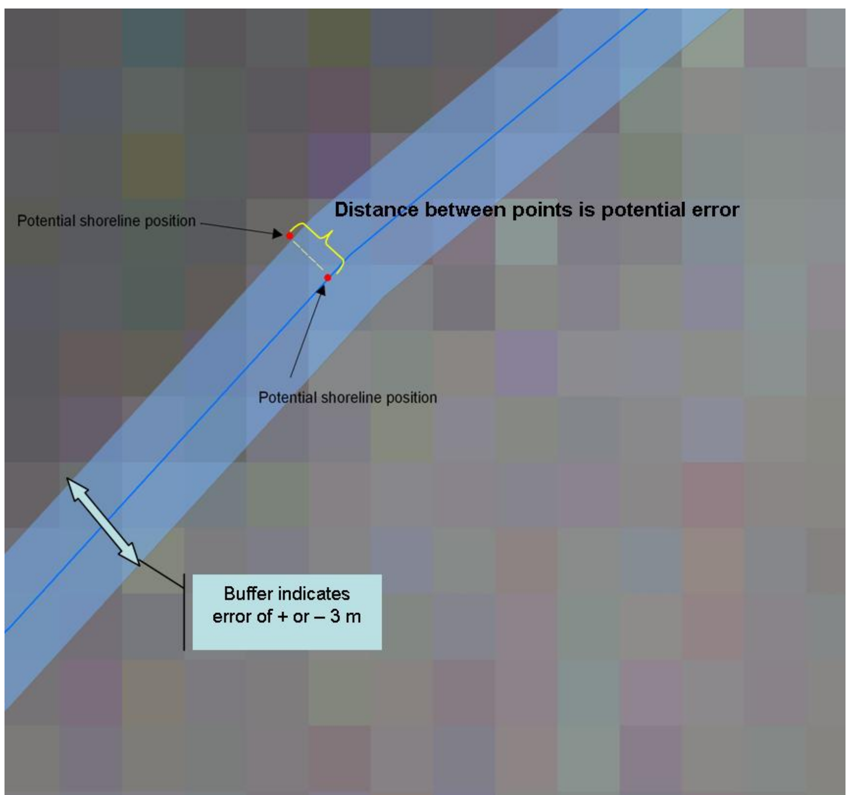


Figure 10: a) Example of points chosen for digitizing error analysis of 1972 shoreline. The red dots indicate potential perceived shoreline position; and b) The distance between the points was measured in order to determine potential error.

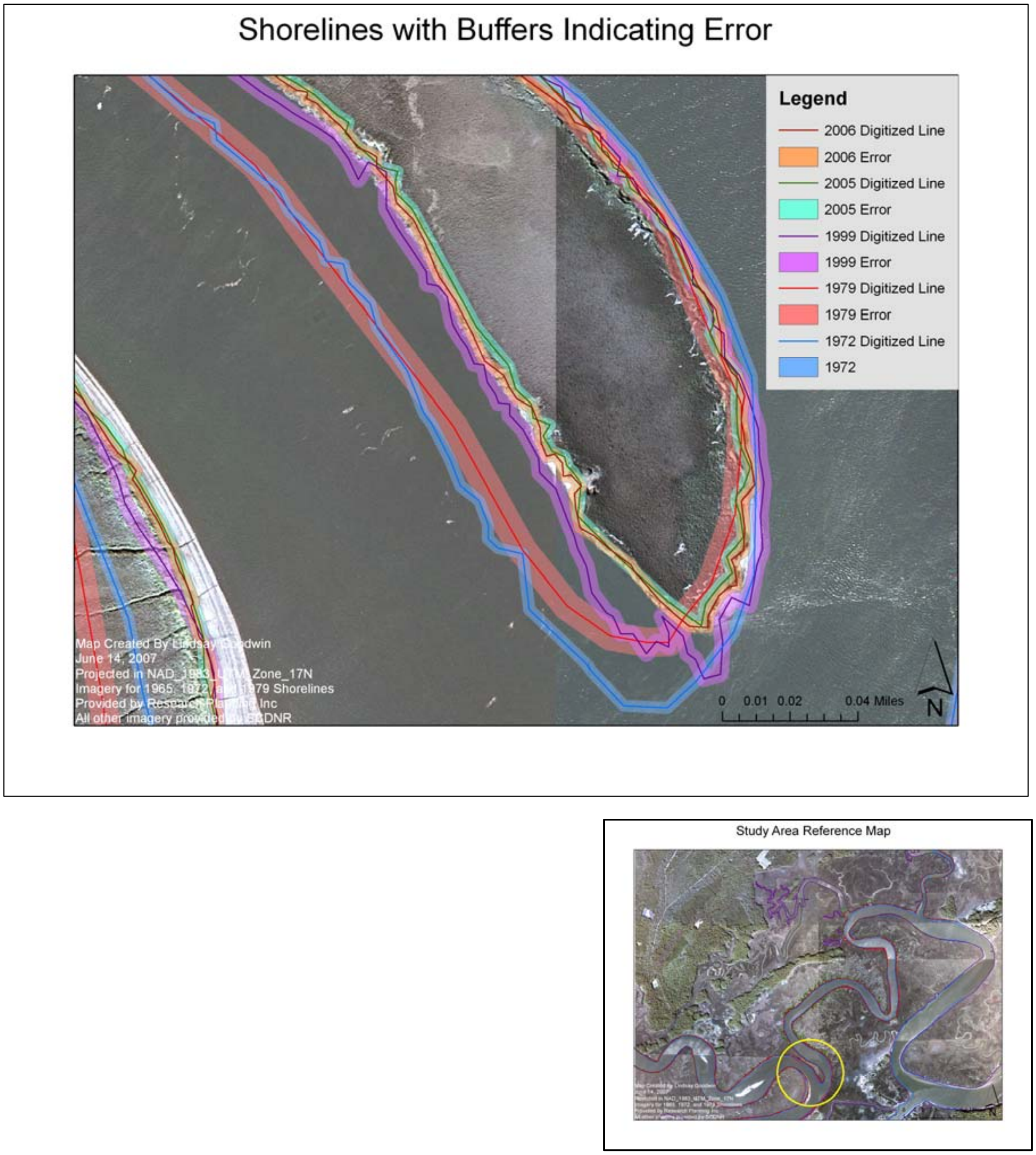


Figure 11. A buffer was applied to the digitized shorelines to indicate potential error in the digitizing process. The far left corner of the enlarged image is a point bar with visible accretion from 1972 to 2006. The area in the middle of the enlarged image is an eroding peninsula. The small image on the bottom right shows the location of the enlarged figure indicated by the yellow circle.

Methods for calculating long-term shoreline changes were taken from the USGS (Theiler et al. 2005). Using vector based historic shorelines, the Digital Shoreline Analysis System (DSAS) originates orthogonal transects from a designated baseline through the shorelines at a specified transect spacing and then calculates associated statistics (Theiler et al. 2005). This study is one of the first studies in South Carolina to use the DSAS system for evaluating erosion rates with in an estuarine system. The approximate center line of the channel was digitized on the 2006 imagery and used as the "baseline" for analysis. The center line was not intended to be an accurate assessment of the mid-channel but, merely a reference line from which the orthogonal transects were projected. For data management purposes, the channel was separated into two features classes, one containing all features of the left bank and the other containing all features of the right bank. The feature classes were further separated into left bank a, b, c and right bank a, b, c. Each segment represented a separate feature class for a total of six feature classes (Figure 12). Using the DSAS 3.2 ArcGIS extension, transects were drawn in 10 m and 50 m increments from the baseline, extending past the farthest shoreline delineation (Figures 13 and 14). This process was repeated for all six feature classes.

Left Bank and Right Bank a,b, and c

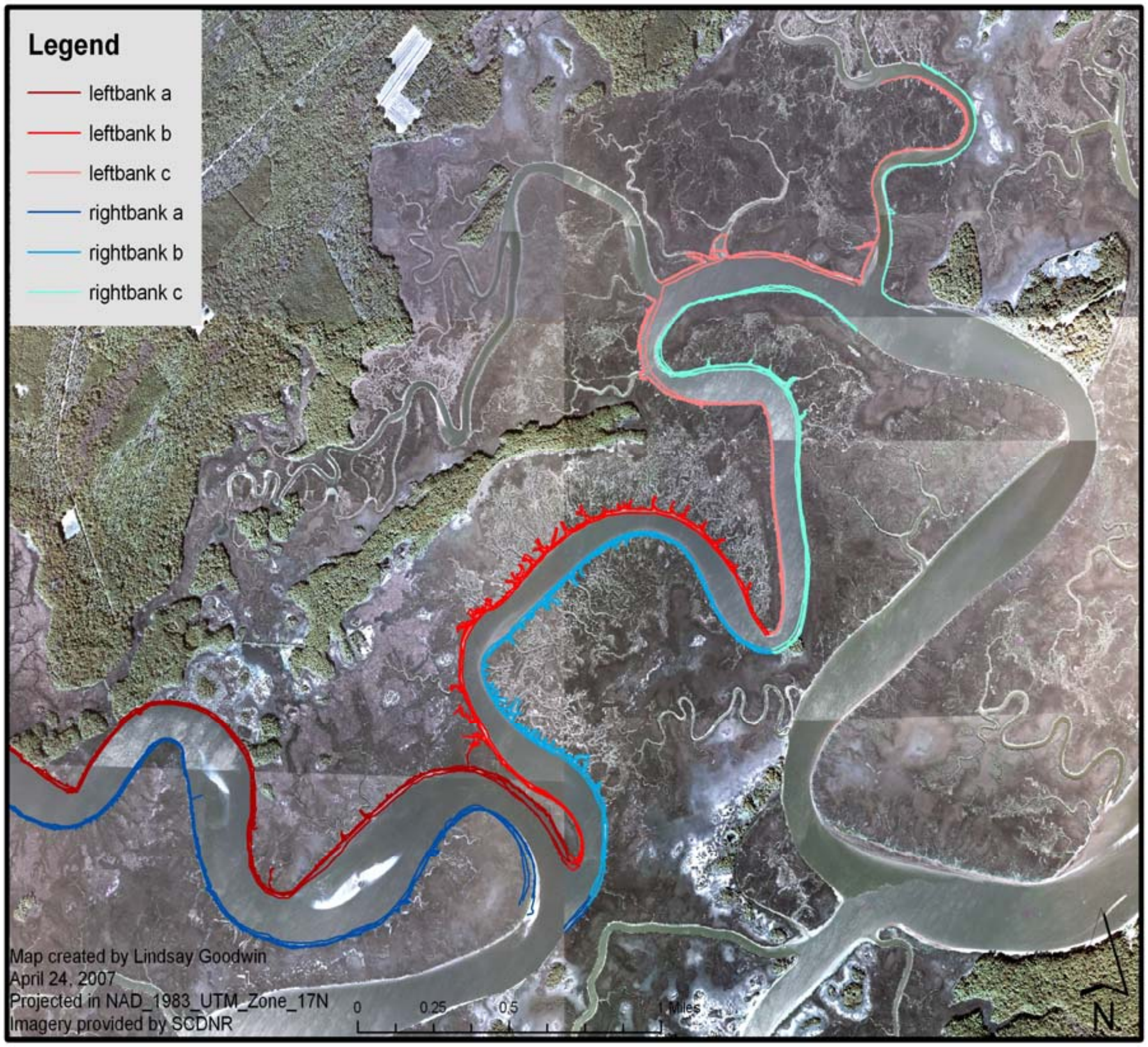


Figure 12. The study area was divided into six segments (left bank a, b, c and right bank a, b, c). Each segment represents a separate feature class for use in the DSAS analysis of long-term shoreline change.

Transect Intersect Points

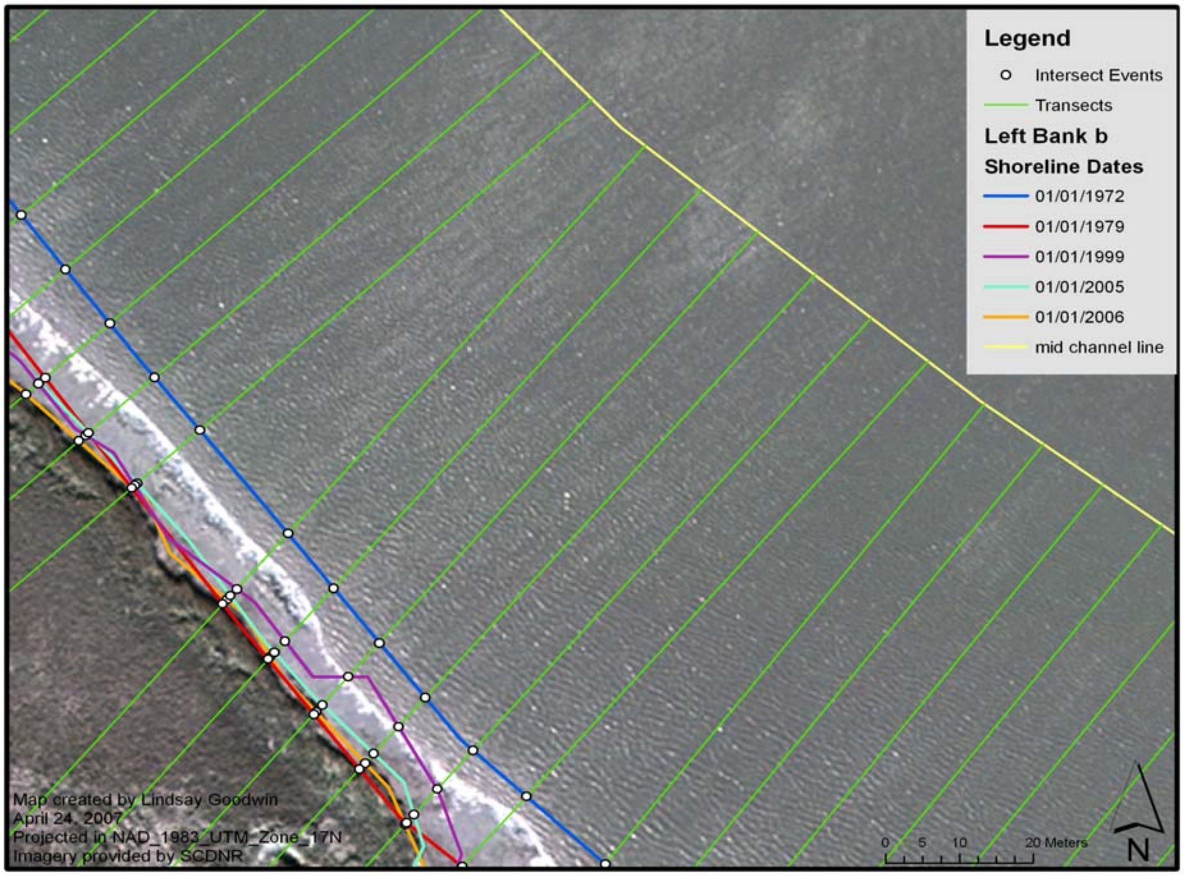
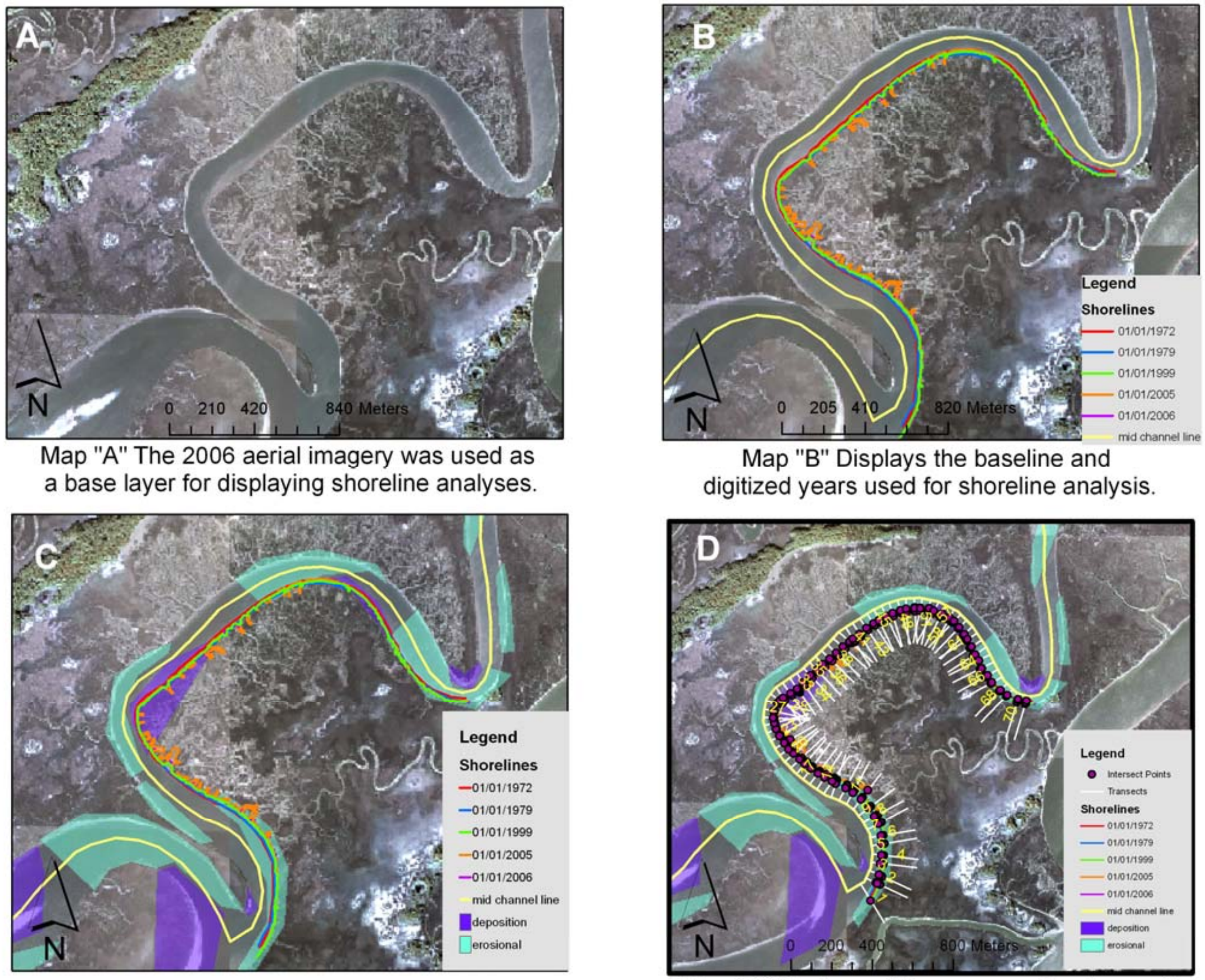


Figure 13. The Digital Shoreline Analysis System (DSAS) projects transects from the baseline (in this case the mid channel line) through the digitized shorelines.



Map "A" The 2006 aerial imagery was used as a base layer for displaying shoreline analyses.

Map "B" Displays the baseline and digitized years used for shoreline analysis.

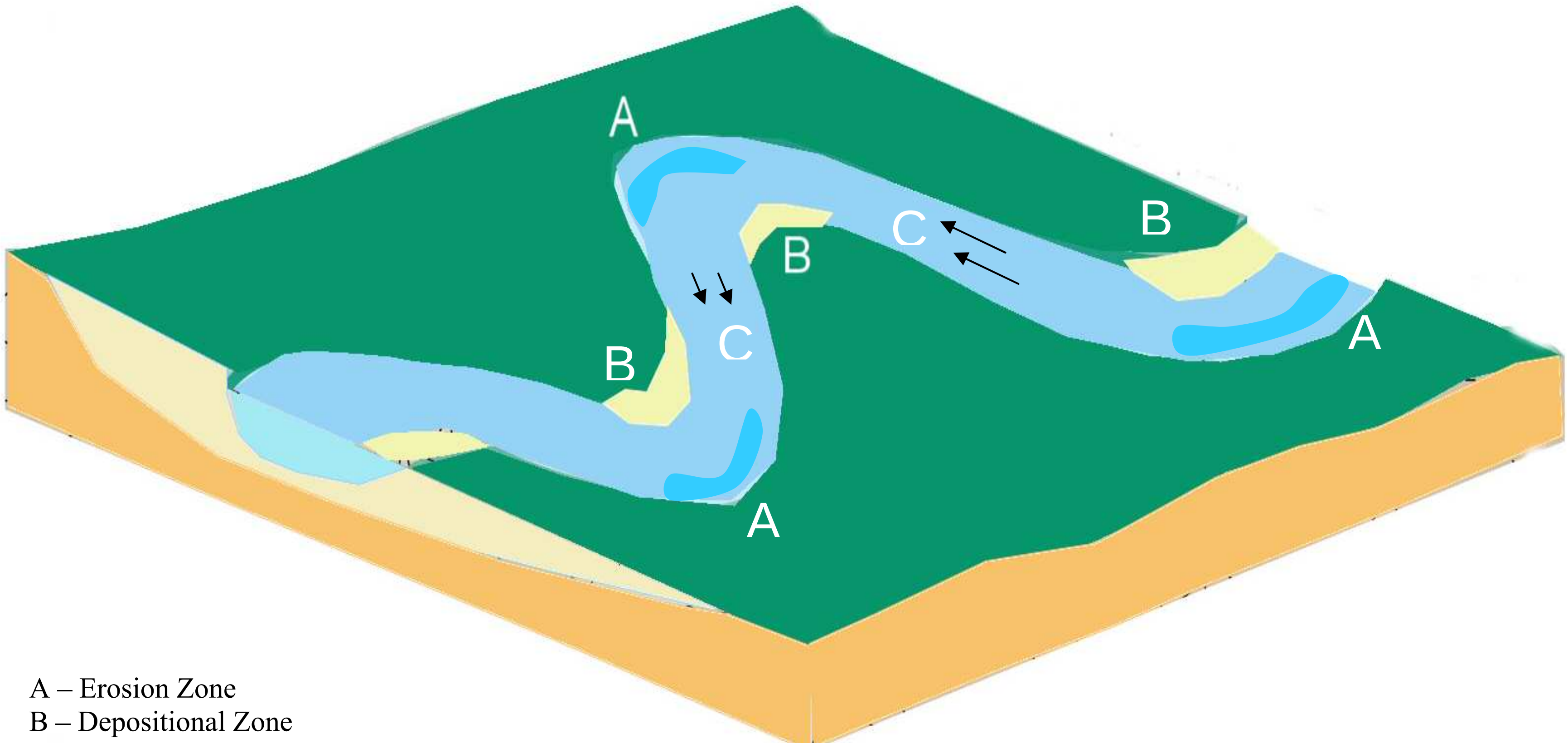
Map "C" Displays the baseline, digitized shorelines, and river morphology classifications.

Map "D" Displays all layers as well as transects and intersect points produced by the DSAS.

Figure 14: Steps in preparing data and analyzing data in right bank “b” for the digital shoreline analysis.

After all transects were drawn, an additional field was created labeled "River Morphology" and added to each of the shoreline attribute tables. The river morphology zones were classified as either an erosional, parallel flow, or depositional zone. A zone was classified erosional if visible signs of erosion occurred on or near the cutbank of the channel (Figures 15 and 16). A zone was classified as parallel flow if there was neither visible erosion nor accretion and there was neither a cutbank nor pointbar (Figures 15 and 16). A zone was classified as depositional if there was visible deposition and usually occurred opposite of a cutbank either on or near the point bar (Figures 15 and 16). The classifications were made through visual inspection of the 2006 imagery. The 2006 imagery was used because it was the most recent, had the greatest accuracy, and was taken at low tide making it easier to delineate areas of erosion and deposition (Figure 16). The 50 m transects were used to compare the overall long-term rate of shoreline change among the three river morphologies for the study area. The transects were classified according to the river morphology zone the transect intersected.

A Weighted Linear Regression rate of change was generated by the DSAS. This rate was used for comparative and statistical analysis. According to Gent et al. (2007) weighted linear regression methods are less susceptible to outliers and are the best erosion rate method when uncertainties in the data, such as accuracy and error, are understood. The DSAS program assigns a weight to specific years digitized based on the shoreline position accuracy assigned by the user.



A – Erosion Zone
B – Depositional Zone
C- Transport or Parallel Flow Zone

Figure 15: Diagram showing zones of erosion and deposition. Depositional areas and point bars are usually opposite of erosion zones and pools. Pools are deep portions of the river formed by the current. Point bars are formed through deposition as are sand bars (not shown here). Figure taken from….

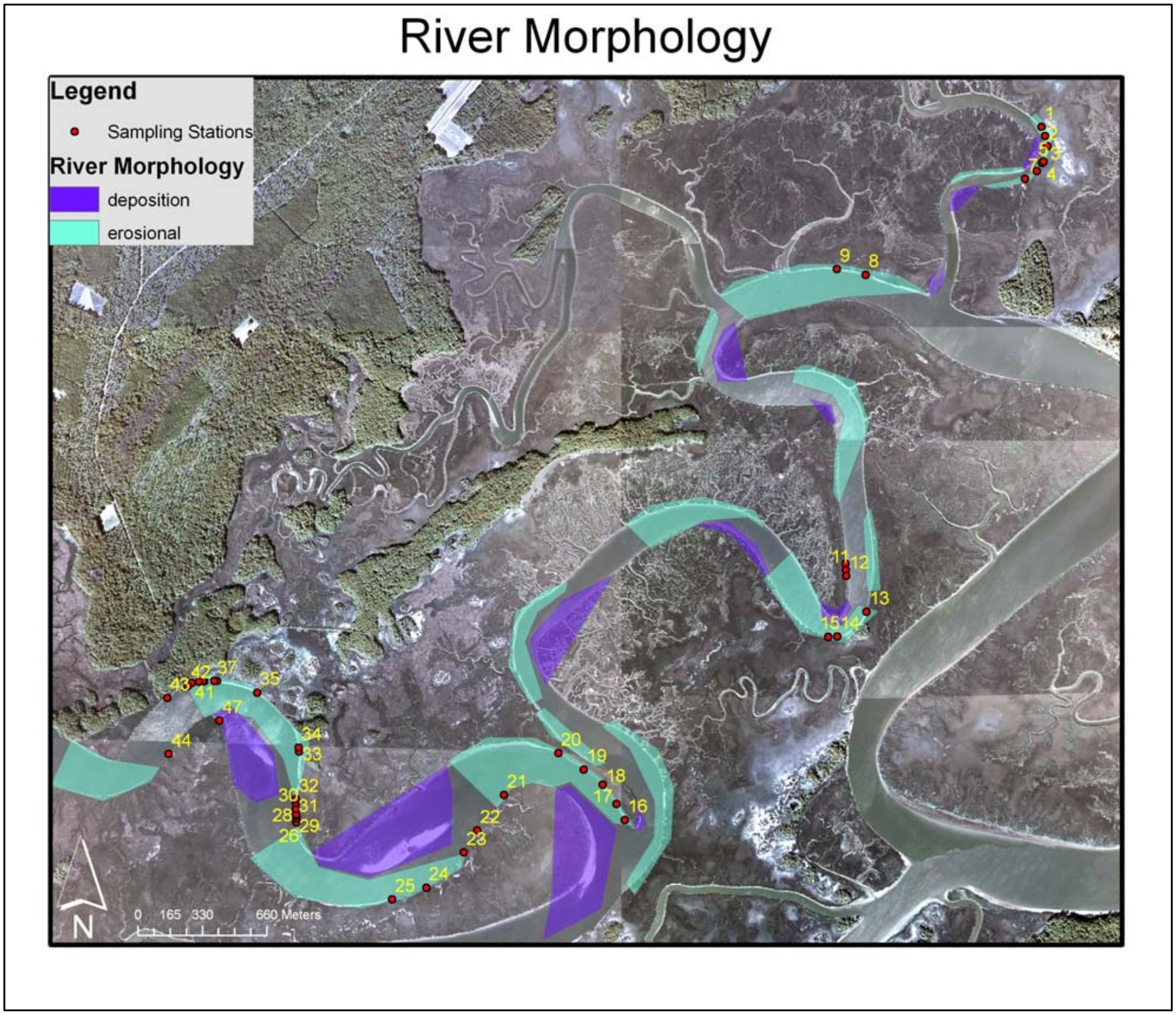


Figure 16: The Study area with river morphology classifications. All areas in the study region that are not colored are classified as parallel flow. This classification was made through visual inspection of the imagery. Erosional zones were generally opposite of depositional zones.

Long-term vs Short-term

For comparative analysis, a long-term shoreline change rate was calculated at each of 47 short-term field sampling stations using the 10 m transects generated by the DSAS. A total of 34 stations were located in an erosional zone. A total of 12 stations were located in a parallel flow zone and only one station was located in a depositional zone. The WLR rate of change was used to quantify long-term shoreline change at the 47 stations.

BIOLOGICAL, GEOLOGICAL, AND HYDROLOGICAL VARIABLES

Stem Density

Live and dead stem counts were taken at a total of 20 of the 47 stations. Specifically, stem densities were taken at stations 1, 3, 6, 7, 9, 11, 13, 15, 19, 21, 22, 25, 27, 31, 32, 33, 36, 38, 41, 44, and 47. Three 0.25 m^2 quadrats were placed behind the landward erosion pole with quadrats placed one meter directly behind the landward pole, approximately one meter to the left and right of the landward pole (Figure 17). All intact live and dead stems were counted within each quadrat.

Below-ground Root Biomass

Below-ground biomass samples were collected at the same 20 stations as stem density in the same quadrats (Figure 17). Specifically, five cores (4 cm diameter) were collected at each of the 20 stations to a depth of 20 cm. In areas of soft sediment a plastic tube was used for coring and in areas of coarse, compact sediments a PVC tube with a 4 cm inside diameter was used. Two cores were extracted from the right and left quadrats and one from the center quadrat. The coring tube was haphazardly placed inside the quadrat and pushed into the sediment by hand or using a hammer when needed. Cores were removed from the tube with a plunger, sealed in labeled plastic bags, and taken back to the lab and frozen until analyzed.

In the lab, cores were cut into 10 cm sections and each section was rinsed over a 1 mm mesh sieve to separate below-ground biomass (live and dead roots and rhizomes) from the sediment. Roots were then dried at 80°C for a minimum of 24 h to a constant dry weight, allowed to cool in a desiccator filled with fresh desiccant, and weighed to the nearest 0.0001g.

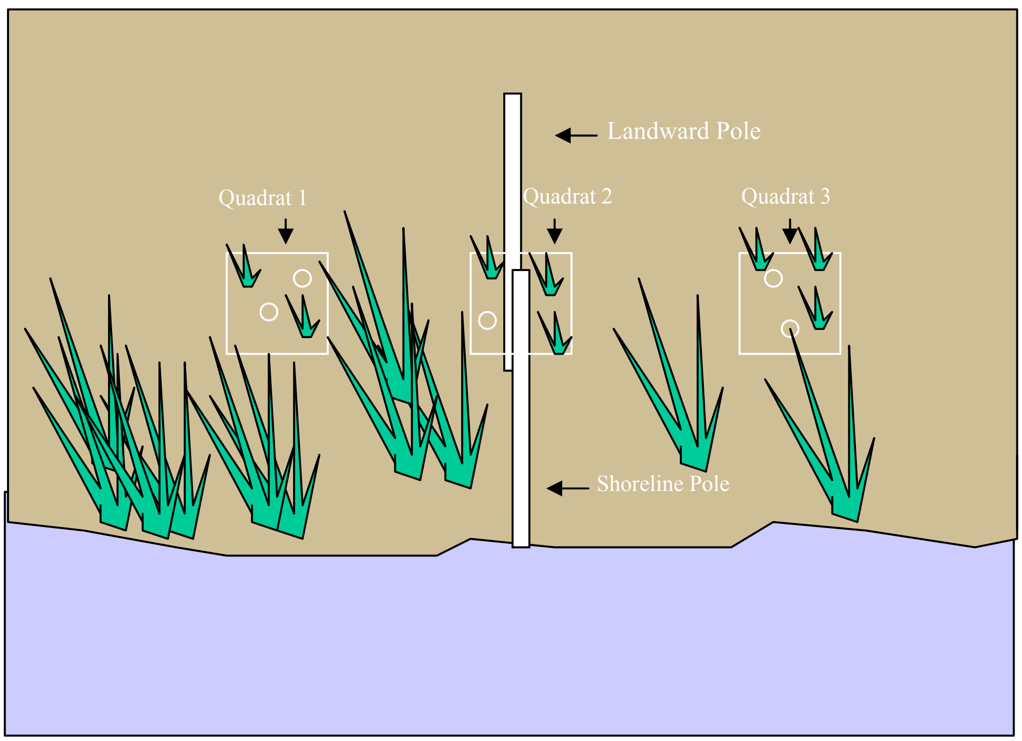


Figure 17. Stem density and core sampling design. Live and dead stems were counted from each quadrat. Two cores were extracted from quadrat 1 and quadrat 3. One core was extracted from quadrat 2 for a total of five cores per sampling station.

Sediment Grain Size Analysis

Sediment data were collected from the same cores processed to collect below-ground biomass data. Cores were rinsed over 1 mm and 63 µm stacked mesh sieves to separate roots and sand from the cores. Sand was collected in the 63 µm mesh sieve. The silt/clay mixture was collected in a bucket or beaker and the volume of each beaker was recorded. After all sediments were washed through the sieves, the remaining water and silt/clay mixture was agitated using a mechanical magnetic stirrer at a medium speed and a 50 ml aliquot was removed from, placed in a pre-weighted aluminum boat, and dried for 48 h at 80°C to a constant dry weight. (Figure 18). The sand portion remaining on the 63 µm sieve was dried for 24 h to a constant weight at 80°C, and weighed.

The silt/clay dry weight (x) was multiplied by the beaker volume (V_b), then divided by the aliquot volume (V_a) to yield the adjusted sediment weight (W_{cs}): $(x^* V_b)/V_a$. Percent sand was calculated with the following equation: W_s/W_t; where W_t = total dry weight and W_s = sand dry weight. Percent silt/clay was calculated with the following equation: W_{cs}/W_t; where W_{cs} = silt/clay dry weight.

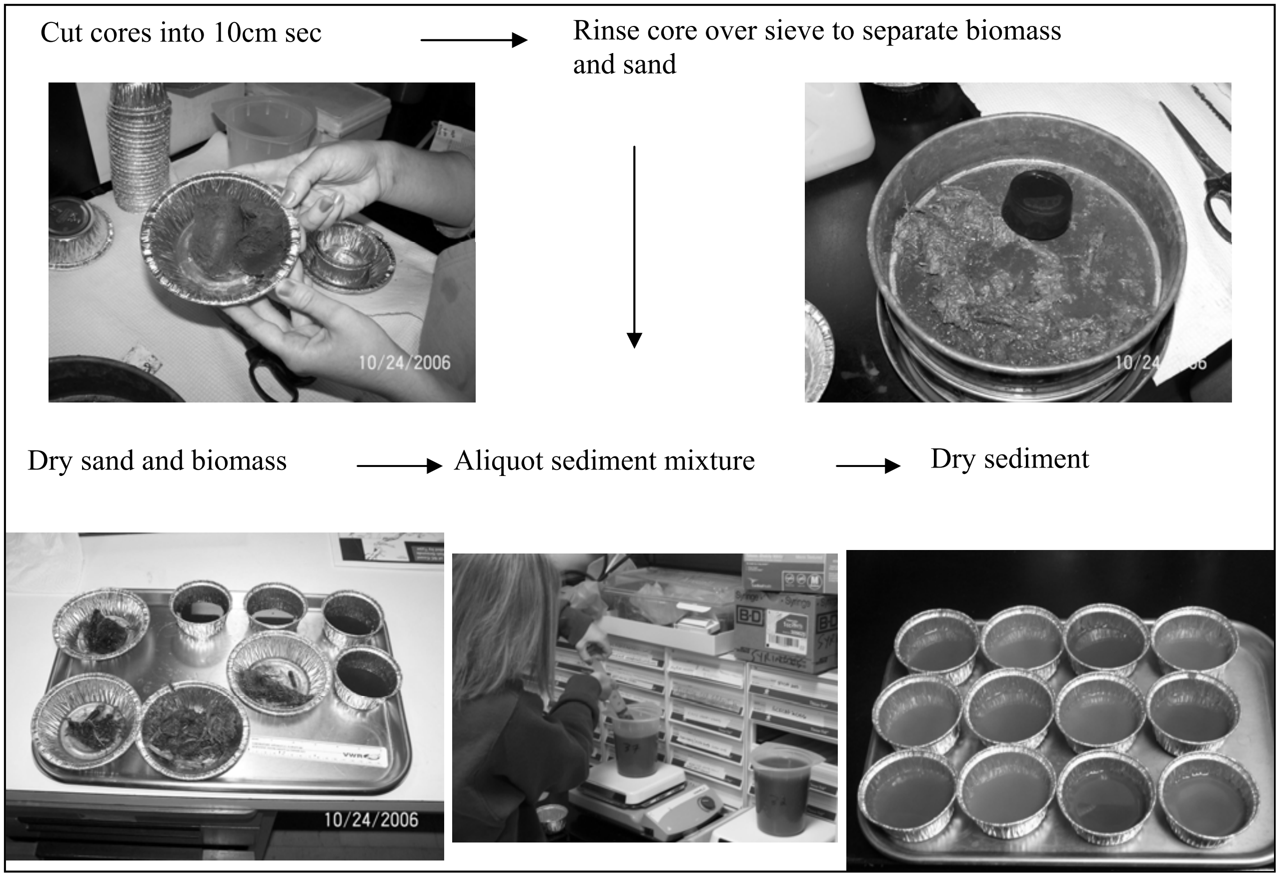


Figure 18. Flow chart indicating the process of analyzing cores for belowground biomass and percent sand. The first step was to cut each core in 10 cm sections. Each section was then rinsed over two stacked sieves (340 μm mesh and 63 μm) in order to separate to biomass from the sand and all water and sand were saved and put aside for later analysis.

Long-term Flow Analysis

Long-term flow was measured using plaster cylinder dissolution (Figure 19). Plaster cylinders were constructed according to the methods modified from Judge et al (1997). Plaster and water was mixed (4 parts plaster: 1 part water) and poured into 16 x 100 mm glass culture tubes. Each cylinder was dried for 24 h and then removed from the glass tubes by carefully breaking the tubes and removing the glass pieces. Once removed, the cylinders were dipped in polyurethane varnish (water soluble satin or gloss), air dried, and cut into 3.5 cm long sections. The end of each cylinder was then sanded flat so that it was level and free of polyurethane to allow dissolution to occur from the top of the cylinder down and not from the polyurethane coated sides or bottom. Prior to deployment, cylinders were labeled on the coated bottoms, dried for 12 h at 70°C, allowed to acclimate to room temperature for 12 h, and weighed to the nearest 0.0001 g. Paired cylinders were mounted on precut polypropylene perforated plates (5 cm x 10 cm) using silicone caulking.

In June 2006 a pilot experiment was conducted deploying paired cylinders at nine sampling stations for nine days (1 plate per station, each plate containing 2 cylinders). Based on results, the final deployment time was shortened to three days; many of the plaster cylinders were eroded too severely to analyze. Representative sampling stations were chosen based on observed erosion rates (encompassing high and low rates), and differences in vegetation, slope, and sediment. Plaster cylinders were deployed at the following stations: 1, 3, 6, 7, 9, 11, 13, 15, 16, 19, 21, 22, 25, 27, 31, 32, 33, 36, 38, 41, 44, and 47. Paired replicates were deployed at stations 1, 16, 36, 27, and 41 for a total of

27 plates. Plates were placed near the bank edge, approximately 1.5 m seaward, and at a height above low-tide but fully submerged during high tide.

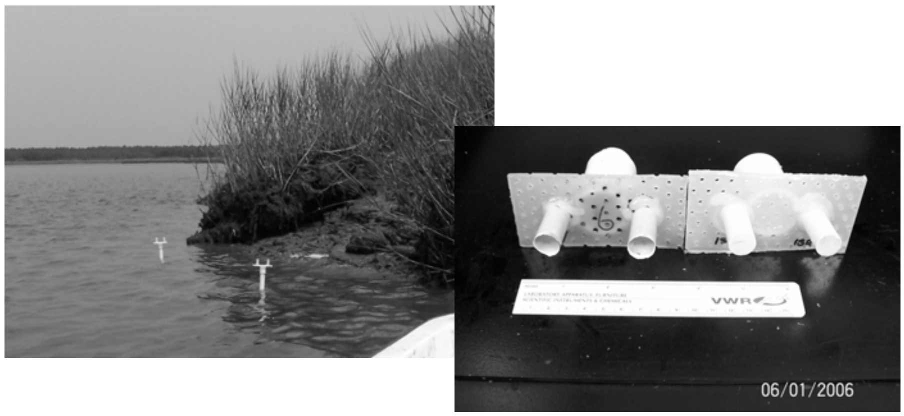


Figure 19**.** Long-term flow was measured through plaster dissolution techniques. Plaster cylinders were deployed at low tide (left photo) and retrieved after three days. Plaster cylinders were weighed pre and post deployment. (Right photo) Post deployment cylinders are on the left and pre deployment cylinders are on the right. Plaster dissolution is used as a surrogate for flow.

Short-term Analysis of Flow

Short-term flow was measured at stations 1, 3, 11, 13, 15, 16, 21, 27, 32, 36, 44 and 47 using a Marsh-McBirney Flo-Mate (Model 2000) current meter on the following dates: July 10 – July 13 2006 and August 14 2006. Measurements were taken at the shoreline adjacent to the station and in the river channel. Vertical profiles were made sampling at 50 cm intervals starting at the surface to the bottom or a maximum of 8 m depth. Care was taken, especially near shore, not to let the boat or any other obstructions disturb the current meter or the flow of water. All measurements were taken during ebbing spring tide.

STATISTICAL METHODOLOGIES

Short-term shoreline change

A T-test was used in order to evaluate the hypothesis that mean short-term shoreline change rates did not vary between erosional zone stations and parallel flow zone stations. The one depositional zone station was not included in this analysis. The dependent factor was field erosion rate and the independent factor was river morphology zone. Short-term erosion rate values did not violate the assumptions of normality or homogeneity of variance and were left untransformed.

Long-term shoreline change

As previously stated, the 50 m transects were used to compare the overall long-term rate of shoreline change among the three river morphologies for the study area using WLR shoreline change rates for each transect. The variable violated the assumption of normality so the Kruskal-Wallis test, the non-parametric equivalent of analysis of variance (ANOVA), was used in order to test for significant differences in the long-term rate of shoreline change among erosional, parallel, and depositional zones. The Kruskal-Wallis test is less powerful than a standard ANOVA because the requirements for normality are relaxed but, conclusions drawn from it are valid for the data collected in this study. The Mann-Whitney test is a non-parametric t-test and was applied in order to determine if significant differences in rate of shoreline change occurred in pair-wise comparisons between the river morphology categories. In combining these two measures, a robust understanding of the data can be developed.

Long-term shoreline change versus short-term erosion rate

In order to determine if long-term shoreline change values and short-term erosion rates were consistent at each of the 47 stations, a paired T-test was utilized. The long-term shoreline change value used for this test was taken from the closest 10 m transect. The null hypothesis tested was: no significant difference existed between the long-term and short-term rates across the 47 stations.

Biological, geological, and hydrological variables

. The Kolmogorov-Smirnov test was used to determine normality for all variables being investigated. .Much of the data were non-normally distributed so the Spearman's Rank Order Correlation, a nonparametric correlation analysis, for pair-wise comparisons, was applied. Correlation analyses were employed to determine if relationships existed between the biological (stem density, belowground biomass), geological (percent sand, and short-term erosion rate), and flow (long-term) measurements at the 20 stations.

A Nested ANOVA is a technique that explains the linked variability between independent, dependent, and nested variables. Nested ANOVAs were used to evaluate the hypothesis that the above listed variables did not vary by station between the two river morphology zones with a sufficient number of observations (erosional and parallel flow zones). Therefore in this analysis the biological, geological and hydrological variables were the dependent variables and the river morphology was the independent variable while the station was considered the nested variable in all analysis.

In order to evaluate the hypothesis that flow did not vary between erosional and parallel flow zones a Student's T-test was used testing for significant difference of the mean plaster dissolution rate per station.

RESULTS

SHORELINE CHANGE ANALYSIS

Short-term Direct Determination of Shoreline Change

Overall, short-term shoreline change rates ranged from 0.27 cm/mo of erosion at station 47 to 10.27 cm/mo erosion per month at station 5 (Figure 20, Table 2). In erosional zone stations, erosion ranged from a low of 0.32 cm/mo at station 32 to a high of 10.27 cm/mo of erosion at station 5 (Figure 20, Table 2). In parallel flow zone stations, erosion ranged from a low of 0.54 cm/mo at station 23 to a high of 6.94 cm/mo at station 22 (Figure 20, Table 2). The only station located in a depositional zone was station 47, which also was the station with the lowest overall erosion rate at 0.27 cm of erosion per month (Figure 20, Table 2). Mean short-term shoreline change for all 47 stations was 3.36 cm per/mo (+/- 0.380 SE).

Long-term Characterization in Shoreline Change

The weighted linear regression (WLR) for shoreline change in depositional zones of all banks, ranged from 0.72 m/yr of erosion to 1.9 m/yr of accretion (n = 128) with a mean rate of 0.01 m/yr (+/- 0.030 SE) of erosion (Table 3). In erosional zones of all banks, shoreline change ranged from 3.01 m/yr of erosion to 0.78 m/yr of accretion (n = 242) with a mean rate of 0.38 m/yr (+/- 0.012 SE) of erosion (Table 3). In parallel flow zones of all banks, shoreline change ranged from 1.05 m/yr of erosion to 1.05 m/yr of accretion (n = 148) with a mean rate of 0.22 m/yr (+/- 0.006 SE) of erosion (Table 3). The greatest amount of erosion occurred in the erosional zone of left bank c with 3.01 m/yr of erosion. The greatest amount of accretion occurred in the parallel flow zone of left bank c with 1.05 m/yr of accretion (Table 3; Appendix A, Figure 30). The mean long-term shoreline change rate for all transects was 0.25m/yr (+/- 0.02 SE) of erosion (n

= 490). Mean long-term erosion for transects intersecting field erosion stations was 0.41m/yr (+/- 0.05 SE) of erosion (n = 47). Appendix A, figures 28-33 illustrate the WLR for all river bank sections and river morphology categories.

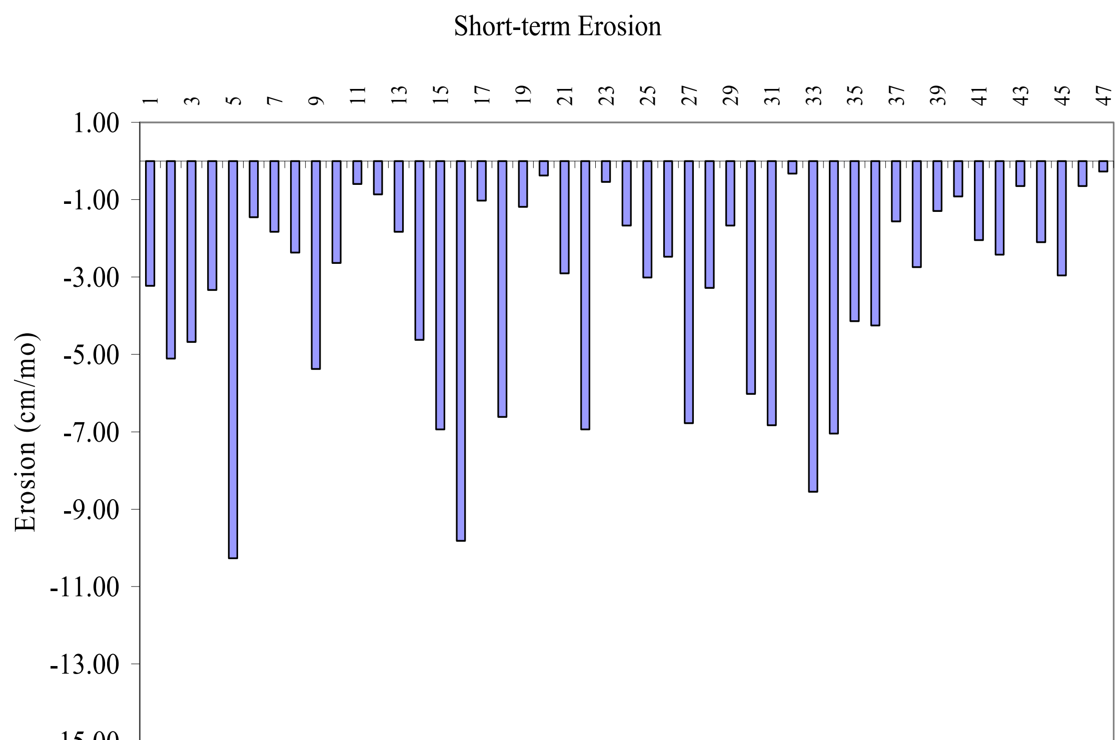


Figure 20: The erosion rate per month was determined by dividing the cumulative erosion rate by the total number of months deployed. Overall 33 stations were located in an erosional zone, 12 stations were located in a depositional zone and 1 station was located in a depositional zone.

Table 2: Short-term erosion rates per river morphology zone. In the mean column means are given for all but the depositional zone where n = 1.

Short-term Erosion	Mean	min	max	n
Parallel Flow	-2.8763441	-6.935483871	-0.537634409	12
Depositional	-0.2688172	-0.268817204	-0.268817204	1
Erosional	-3.6268472	-10.2688172	-0.322580645	33
All	-3.363782	-10.2688172	-0.268817204	47

Table 3: Long-term shoreline change descriptive statistics for the weighted linear regression (WLR) generated by the DSAS analyses.

Bank Designation	n	WLR m/yr Mean WLR	Min WLR	Max WLR
Right bank a				
Parallel Flow	27	-0.35	-0.73	0.49
Depositonal	21	0.55	-0.16	1.9
Erosional	42	-0.34	-0.75	0.29
Right bank b				
Parallel Flow	12	-0.17	-0.46	-0.01
Depositional	26	-0.27	-0.71	0.34
Erosional	29	-0.23	-0.69	0.16
Right bank c				
Parallel Flow	36	-0.19	-1.05	0.71
Depositional	17	0.13	-0.37	0.78
Erosional	39	-0.12	-1.02	0.78
Left bank a				
Parallel Flow	17	-0.19	-0.45	0.02
Depositional	20	-0.31	-0.72	0.11
Erosional	49	-0.49	-1.42	0.08
Left bank b				
Parallel Flow	17	-0.30	-0.58	-0.06
Depositional	7	-0.12	-0.23	0.01
Erosional	43	-0.45	-0.98	-0.17
Left bank c				
Parallel Flow	39	-0.13	-0.9	1.05
Depositional	16	0.05	-0.45	0.33
Erosional	41	-0.65	-3.01	0.22
All Banks				
Parallel Flow	142	-0.22	-1.05	1.05
Depositional	128	-0.01	-0.72	1.9
Erosional	228	-0.38	-3.01	0.78

BIOLOGICAL, GEOLOGICAL, AND HYDROLOGICAL RESULTS

Stem Density

Stem densities were assessed at a total of 20 stations. Mainly two species of marsh plants contributed to the stem densities: *Spartina alterniflora* and *Juncus roemanus*. Overall, *Spartina* stems were counted at 19 out of 21 stations. *Spartina* stem density ranged from 2 to 51/quadrat. *Juncus* stems were collected at 4 out of 21 stations. *Juncus* stem density ranged from 2 to 126/quadrat at stations where it was present. Overall combined stem density ranged from 2/quadrat at station 38 to 126/quadrat at station 36. In erosional zones, stem density ranged from 2/quadrat to 126/quadrat with a mean of 23.35/quadrat (+/- 4.424 SE; n = 42). In parallel flow zones, stem density ranged from 3/quadrat to 19/quadrat with a mean of 10.26/quadrat (+/- 1.255 SE; n = 15). In depositional zones, stem density ranged from 7/quadrat to 27/quadrat with a mean of 16/quadrat (+/- 5.859 SE; n = 3; Table 4).

Table 4: Mean (number of samples) and standard error (SE) for below-ground biomass, sand content, and stem density for each of the river morphology categories. Depositional values were not incorporated into the statistical analyses because only one station (with multiple samples at that station) represented a depositional zone.

	Erosional		Parallel		Depositional	
Parameter	Mean	SE	Mean	SE	Mean	SE
Biomass (g)	7.21 (n = 66)	0.568	2.48 (n = 29)	0.452	1.32 (n = 5)	0.650
% Sand	0.32 (n = 44)	0.040	0.16 (n = 15)	0.062	0.04 (n = 3)	0.012
Stems (no./m)	23 (n = 42)	4.4	10 (n = 15)	1.3	16 (n = 3)	5.9

Below-ground Biomass

Mean Below-ground biomass per station ranged from 0.33 g (+/- 0.069 SE) at station 9 to 12.63 g (+/- 1.073 SE) at station 32 (Table 5, Figure 22). In erosional zones, total below-ground biomass per core ranged from 0.18 g to 17.91 g with a mean of 7.32 g (+/- 0.566 SE; n = 65). In parallel flow zones, belowground biomass per core ranged from 0.28 g to 9.73 g with a mean of 2.40 g (+/- 0.442 SE; n = 30). In depositional zones, below-ground biomass per core ranged from .029 g to 2.09 g with a mean of 1.32 g (+/- 0.294; n = 5). Figure 32 – 36 of Appendix A illustrate below-ground biomass per station and core by depth.

Table 5**:** Mean, minimum, and maximum values of below-ground biomass for the three river morphology categories.

River Morphology	**n**	**Mean (g)**	**Min (g)**	**Max (g)**
Parallel	30	2.41	0.27	9.74
Depositional	5	1.32	0.29	2.10
Erosional	65	7.32	0.18	17.92
All	100	5.55	0.18	17.92

Sediment

Overall, mean percent sand per station ranged from 2% (+/- 0.1 SE) at station 11 to 76% (+/- 0.2 SE) at station 33 (Figure 23). In erosional zones, sand content per core ranged from 2% sand to 79% sand with a mean 32% sand (+/- 0.040 SE; n = 44). In parallel flow zones, sand content per core ranged from 1% sand to 89% sand with a mean of 16% sand (+/- 0.062 SE; n = 15). In depositional zones, sand content per core ranged from 2% sand to 6% sand with a mean of 3.5% sand (+/- 0.010 SE; n = 3). See Appendix A Figures 41 - 45 for percent sand per station and core by depth.

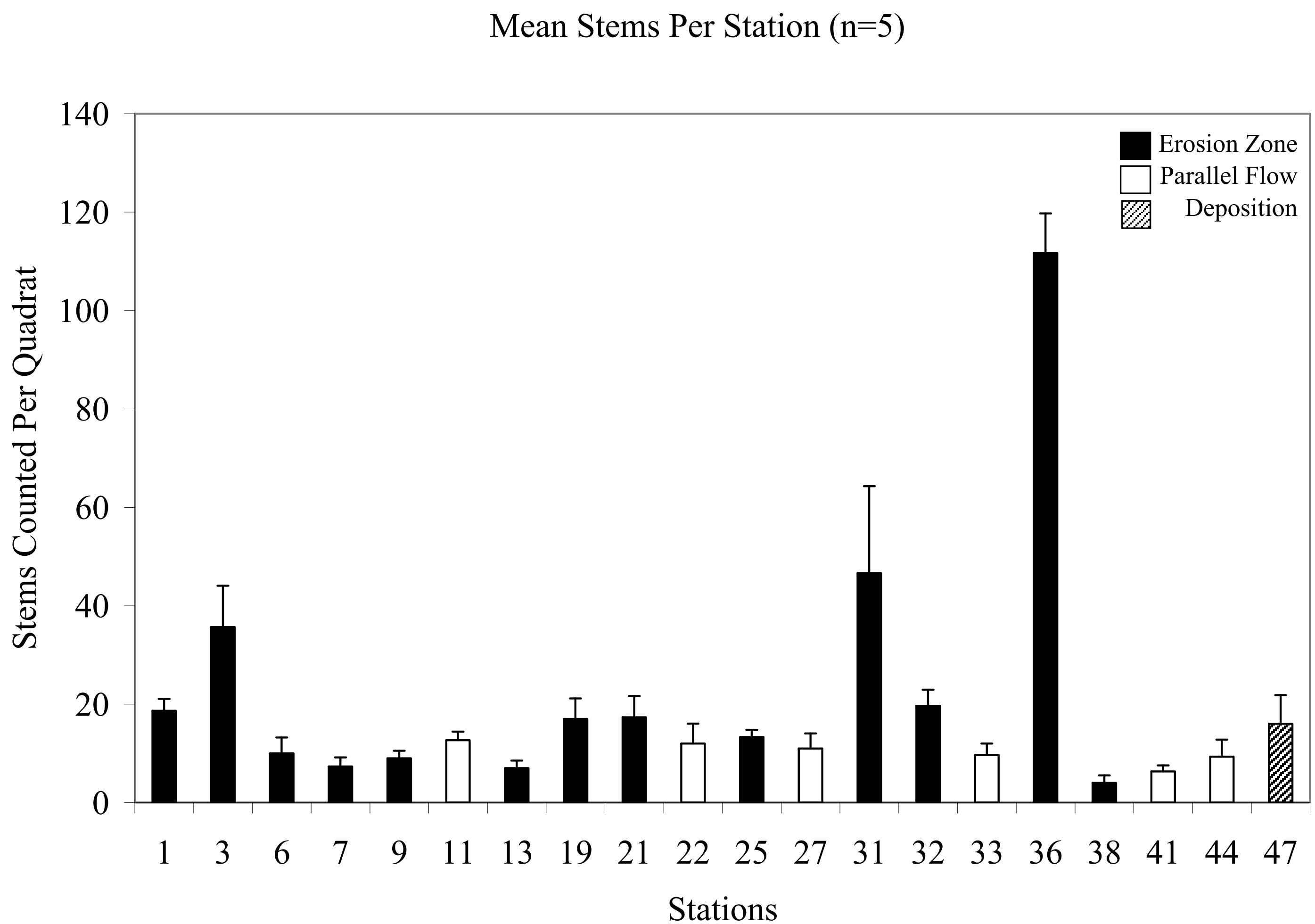


Figure 21: Mean stem density per staion.

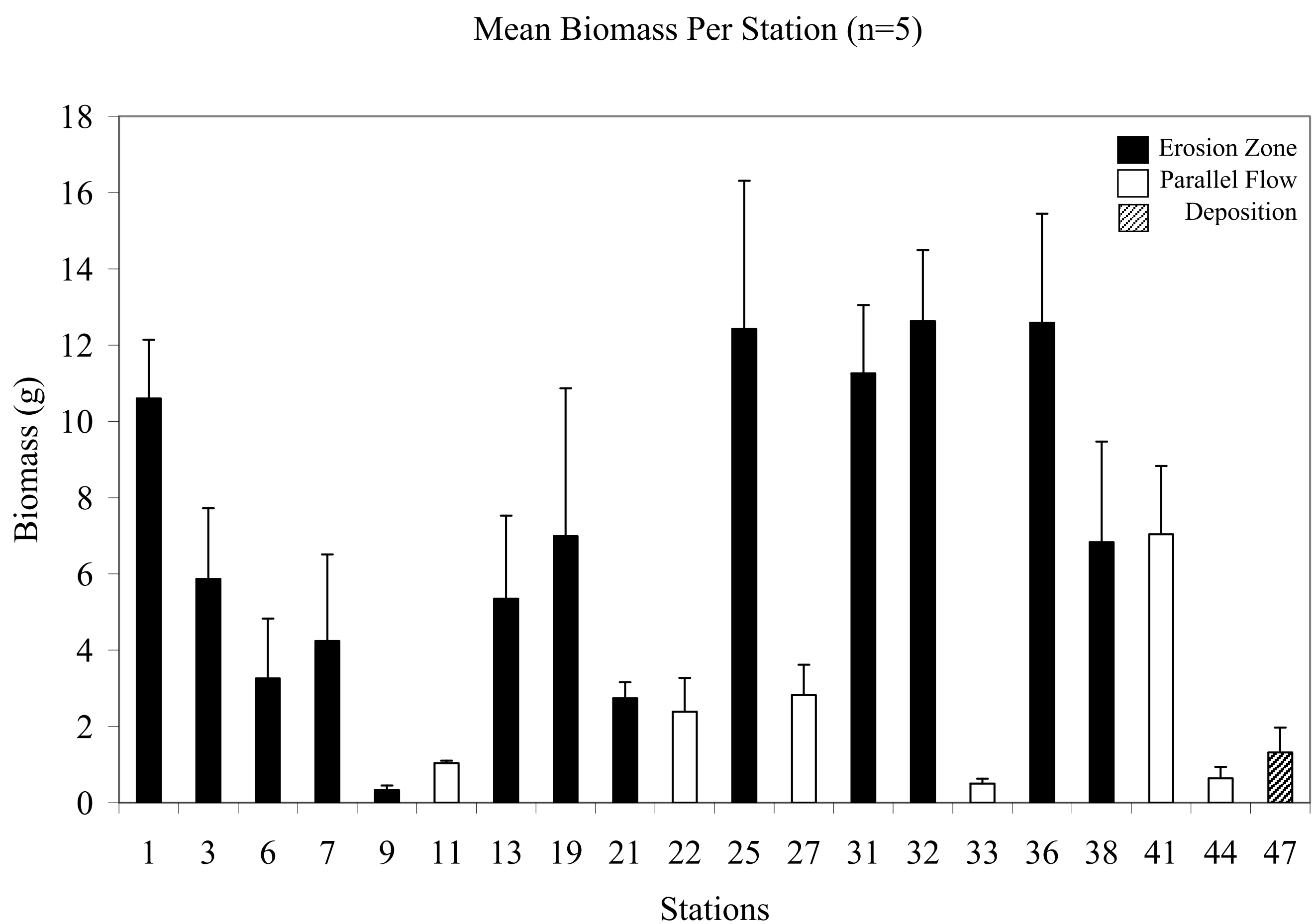


Figure 22: Mean below-ground biomass per station.

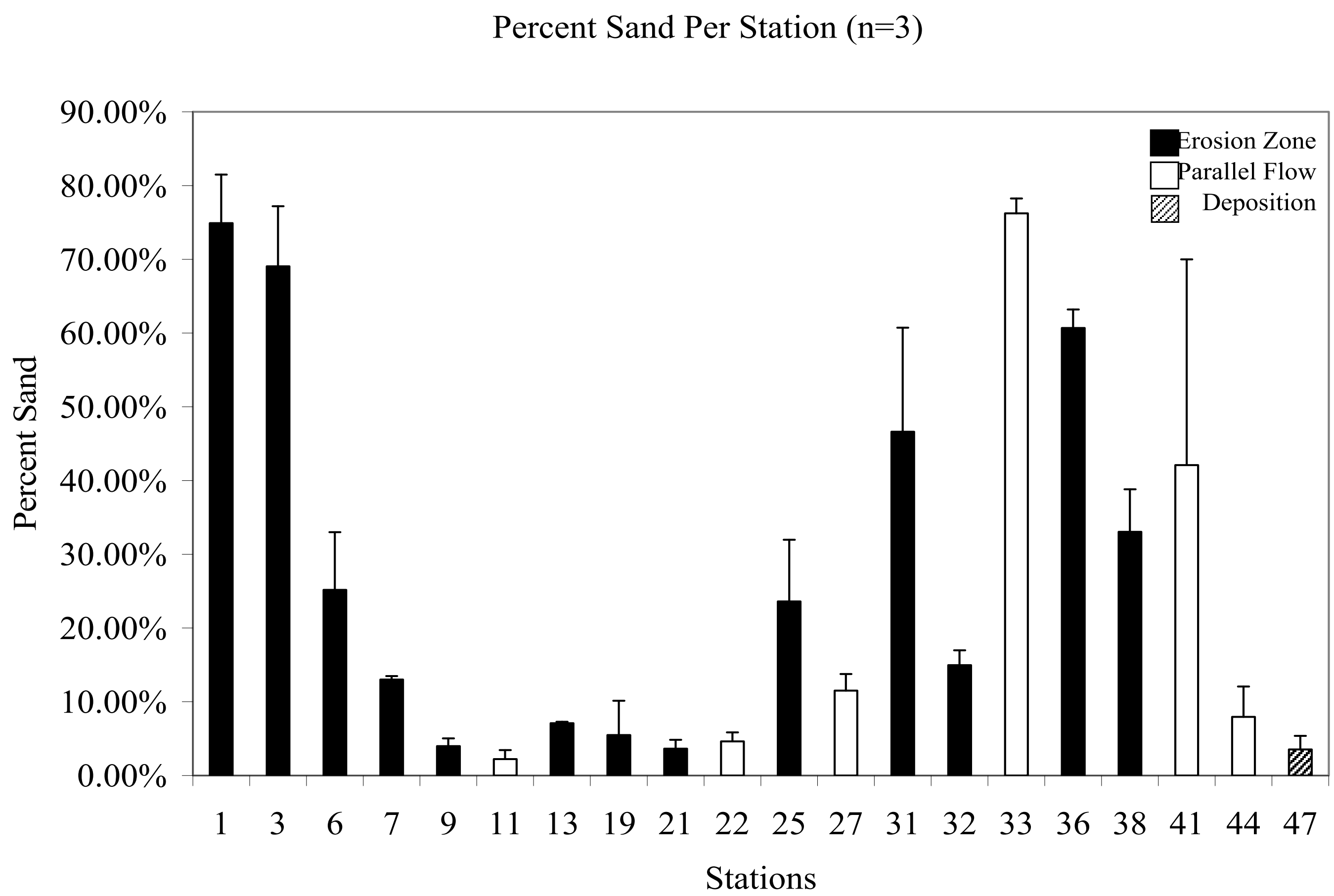


Figure 23: Mean percent sand per station. Of the stations analyzed for percent sand content, three cores were analyzed per station. Overall, 39 cores were analyzed for percent sand in erosional zones, 18 cores in parallel flow zones and 3 cores in depositional zones.

Long-term and Short-term Flow Analysis

Mean proportion of dissolution, which was our measure of relative flow, for stations in depositional zones was 0.15 (+/- 0.012 SE). This indicates that on average 15% of the original plasters dissolved by the time of collection. Mean proportion of dissolution for stations in parallel flow zones was 0.12 (+/- 0.014 SE). This means that on average 12% of the original plaster dissolved by the time of collection. The one station located in a depositional zone had a plaster proportion of dissolution of 0.11 indicating that 11% of the plaster dissolved by the time of collection.

Velocity was measured at 17 stations. Maximum velocity ranged from 0.06 m/sec at station 11 to 0.96 m/sec at station 27. Minimum velocity ranged from 0.00 m/sec at station 21 to 0.22 m/sec at station 44. Mean velocity ranged from .15 m/sec at station 21 to .69 m/sec at station 16. See Appendix A Figures 46 - 57 for velocity measured per station by depth.

RESULTS OF THE STATISTICAL ANALYSES

Analyses of long-term and short-term shoreline change rates

The long-term shoreline change rates varied significantly across the three categories of river morphology (Kruskal Wallis Test: $\chi^2 = 95.47$, $p < 0.001$; Table 6, Figure 25) and exhibited the following relationship of shoreline loss: depositional < parallel < erosional. Mean short-term shoreline change rates did not differ significantly across erosional and parallel flow zone stations (T-test: T = -0.19; p = 0.850). In addition, short-term and long-term shoreline change rates did not vary significantly from each other across the 47 stations (Paired T-test: T = 1.313; p = 0.268).

When using plaster dissolution to measure flow, no significant difference occurred between the two river morphology zones with sufficient numbers for comparison. This means that mean plaster dissolution did not vary significantly between erosional and parallel zones (Student's T-test: $t = 1.17$, d.f. $= 17$, $p = 0.257$).

Correlation Analyses of Variables

A significant correlation was detected between percent sand content and below-ground biomass ($p = 0.02$). No significant correlation was detected between percent sand and the following variables: stem density ($p = 0.45$); flow ($p = 0.65$); erosion ($p = 0.07$). No significant correlation was detected between below-ground biomass and the following variables: stem density ($p = 0.439$); flow ($p = 0.319$); and erosion ($p = 0.638$). No significant correlation was detected between stem density and flow ($p = 0.62$) or stem density and erosion ($p = 0.054$). No significant correlation was detected between flow and erosion ($p = 0.431$). Of the ten Spearman's Rank Order Correlations conducted among the five variables (stem density, below-ground biomass, percent sand, flow, and short-term erosion) only one significant relationship emerged (Table 7). Percent sand was significantly correlated with below-ground biomass (Spearman's rho = 0.514; $p = 0.02$). No other combination of the variables were significantly correlated.

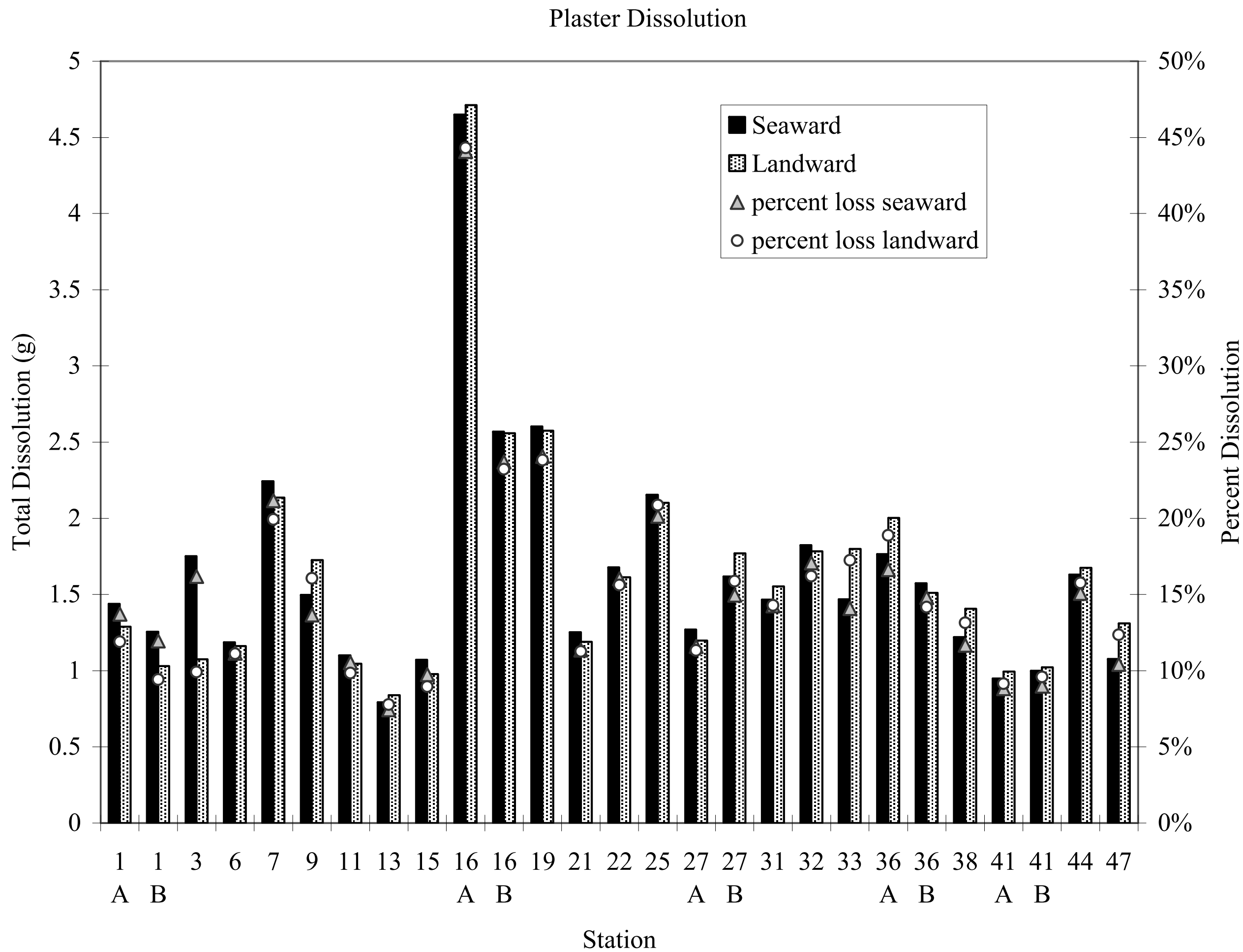


Figure 24: This graph illustrates total plaster dissolution over the three day deployment period for both the landward and seaward cylinder. Each cylinder was slightly different in weight pre-deployment. Percent of plaster dissolution is also illustrated as a reference.

Table 6. Statistical tests for comparing long-term erosion rates among and between the three categories of river morphology. Kruskal-Wallis was used to test for a significant difference in erosion among the three river categories then the Mann-Whitney Test was used in order to determine the pair-wise significant differences between the categories.

Comparision	Test Statistic	Significance
Kruskal-Wallis Test		
Among three categories: Erosional, Parallel, and Depositional	$\chi^2 = 95.47$	< 0.001
Mann-Whitney Test		
Erosional x Parallel	U = 12376.5	< 0.001
Erosional x Depositional	U = 5060.0	< 0.001
Depositional x Parallel	U = 4683.5	< 0.001

Table 7: Spearman's Rank Order Correlations of the sedimentological, biological, and flow variables. Correlations were significant at the 0.05 level (2-tailed).

		Variables				
Variables		**Percent sand**	**Below Ground Biomass**	**Stem Density**	**Flow**	**Erosion (cm/mo)**
Percent Sand	Correlation Coefficient	1.000	0.514	0.176	0.108	0.411
	Sig. (2-tailed)	-	0.020	0.458	0.650	0.072
	n	20	20	20	20	20
Below Ground Biomass (g)	Correlation Coefficient	0.514	1.000	0.183	0.235	-0.112
	Sig. (2-tailed)	0.020	-	0.439	0.319	0.638
	n	20	20	20	20	20
Stem Density	Correlation Coefficient	0.176	0.183	1.000	0.116	0.436
	Sig. (2-tailed)	0.458	0.439	-	0.627	0.054
	n	20	20	20	20	20
Flow	Correlation Coefficient	0.108	0.235	0.116	1.000	0.187
	Sig. (2-tailed)	.650	0.319	0.627	-	0.431
	n	20	20	20	20	20
Erosion (cm/mo)	Correlation Coefficient	0.411	-0.112	0.436	0.187	1.000
	Sig. (2-tailed)	0.072	0.638	0.054	0.431	-
	n	20	20	20	20	20

ANOVA Results

Critical ANOVA assumptions were examined for each data set, appropriate transformations were applied when indicated, and variances were assessed when homogeneity assumptions could not be satisfied (Table 8). Below-ground biomass varied significantly between river morphology zones ($F = 6.993$; $p = 0.017$) and between stations ($F = 17.882$; $p < 0.001$). Below-ground biomass was significantly higher in erosional zones than parallel flow zones ($p = 0.017$; Figure 25). Sand content was not significantly different between erosional and parallel flow zones ($F = 1.623$; $p = 0.220$; Figure 26), but was significantly different between stations ($F = 32.157$; $p < 0.001$). Stem density was not significantly different between erosional and parallel flow zones ($F = 1.052$; $p = 0.319$; Figure 27), but was significantly different among stations ($F = 19.982$; $p < 0.001$).

Table 8 : Individual nested ANOVA results testing for significant differences in mean below-ground biomass among River Morphologies, in mean sand content among River Morphologies, and in mean stem density among each river morphology category.

Source	df	Mean Square	F	Sig.
Biomass				
River Morphology	1	495.249	6.993	0.017
River Morphology (Station)	17	70.825	17.882	0.000
Percent Sand				
River Morphology	1	0.477	1.623	0.220
River Morphology (Station)	17	0.302	32.157	0.000
Stem Density				
River Morphology	1	1893.985	1.052	0.319
River Morphology (Station)	17	1800.504	19.982	0.000

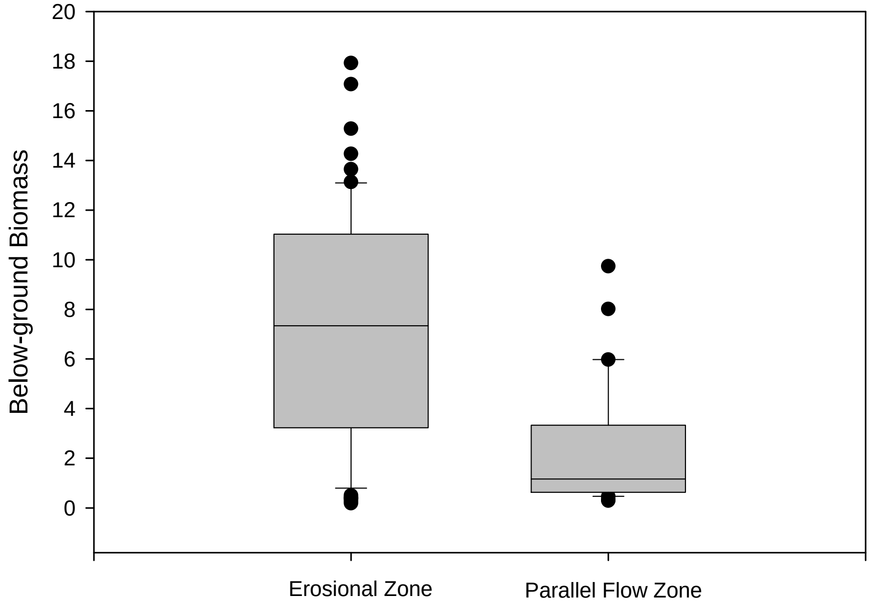


Figure 25*:* Box plot of below-ground biomass in erosional and parallel flow zones. Rectangles represent the middle half of the data range with an end at each quartile. The horizontal line inside the box represents the median. Vertical lines with horizontal line attached at extremes represent the range of data excluding outliers. Circles represent outliers and asterisks represent extreme outliers. Below-ground biomass was greater in erosional zones

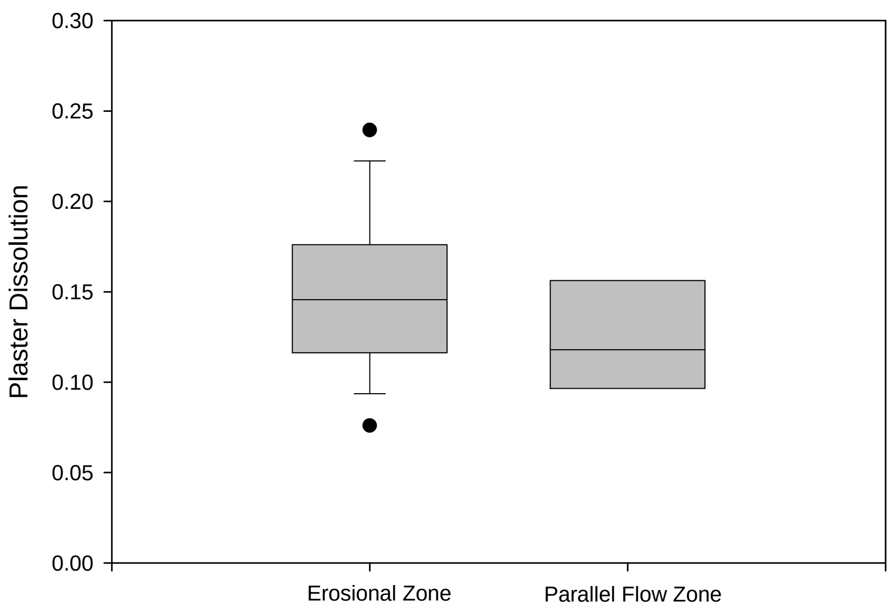


Figure 26: Box plot of below-ground biomass in erosional and parallel flow zones. Rectangles represent the middle half of the data range with an end at each quartile. The horizontal line inside the box represents the median. Vertical lines with horizontal line attached at extremes represent the range of data excluding outliers. Circles represent outliers and asterisks represent extreme outliers. Below-ground biomass was greater in erosional zones

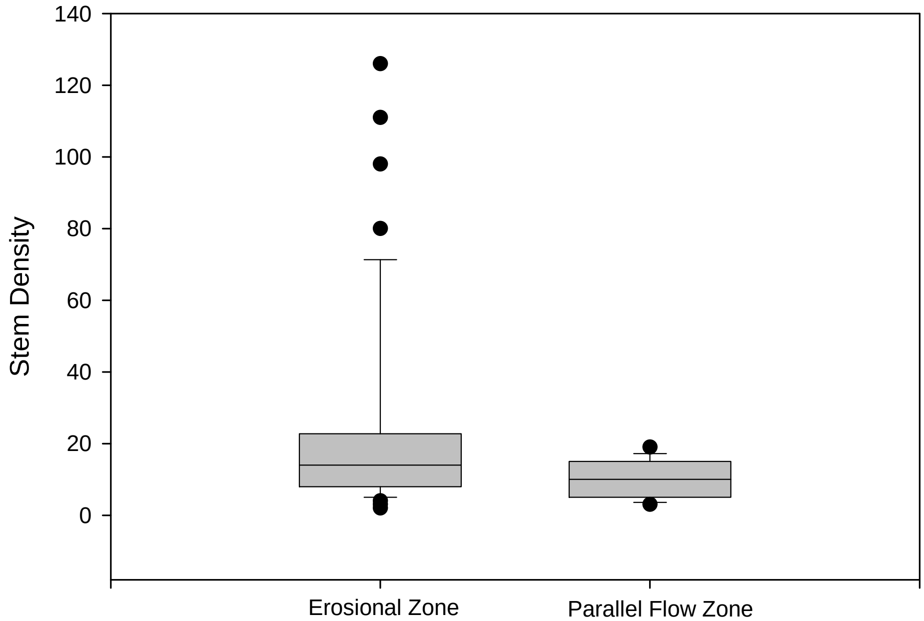


Figure 27: Boxplot of stem density from all grass species in erosional and parallel flow zones. Rectangles represent the middle half of the data range with an end at each quartile. The horizontal line inside the box represents the median. Vertical lines with horizontal line attached at extremes represent the range of data excluding outliers. Circles represent outliers and asterisks represent extreme outliers. There was no significant difference found between zones.

DISCUSSION AND RECOMENDATIONS

Long-term erosion was greatest in erosional zones, then parallel flow zones, followed by depositional zones. The findings are indicative of natural river meandering. Sediment accumulates at pointbars, whereas erosion usually occurs along the cutbanks (Keller 2008). Mean long-term erosion of the entire study area was 0.25 m/yr of erosion . Price (2005) examined erosion in the Guana Tolomato Matanzas National Estuarine Research Reserve in Florida and documented a mean erosion rate of 0.44 m/yr from 1970 – 2002. That study attributed erosion to boat wake impacts. Erosion rates may be less in the current study because currently there is minimal boat traffic,. Price (2005) also did not find evidence of meandering. Estuarine shorelines are dynamic in nature and differ on a site-by-site basis and comparisons of separate study areas concerning influencing factors should recognize such differences.

Inherent uncertainty and error exists in estimating shoreline change through digital interpretations. When examining the long-term shoreline change rates, the total annual shoreline position error did not exceed the recorded change in shoreline position. If the shoreline position error had exceeded the shoreline change rate, then more accurate data would have been necessary for the analysis to be valid. Shoreline change of river channels is generally less than the observed change that occurs in exposed shorelines. High resolution data is needed to best minimize error in assessing shoreline position and detecting shoreline change. Table 1 provided estimates of accuracy based on FGDC standards (Foote et al. 1995). These are maximum estimates and may overestimate the error of positional accuracy.

The Digital Shoreline Analysis System used in this study was an effective and efficient tool for analyzing long-term shoreline change and should be utilized in future studies. This study is one of the first studies in South Carolina to use the DSAS system for evaluating erosion rates with in an estuarine system. The correspondence between short term rates and the long term rates from the aerial photography show the power in this methodology. DSAS has been successfully used in other estuarine environments (Burman et. al. 2007) and with increased accuracy in future data sets will become a powerful tool for managing the estuarine environment. The Weighted Linear Regression function used in this analysis is valuable because it allows the user to apply greater emphasis to imagery with greater accuracy and resolution.

In the current study, mean short-term shoreline change based on the 47 field sampling stations was 0.4 m/yr of erosion. When long-term shoreline change was analyzed using the WLR derived from DSAS measurements at 498 transects the mean erosion rate was 0.25 m/yr of erosion. The mean short-term erosion rate was higher than mean long-term erosion which may signify one of two possible scenarios. The first is that the region is experiencing an erosional period. Accretion was not present at any of the 47 erosion stations. A second possible scenario is that there was a sampling bias that skewed the data. This second scenario is the more probable scenario. There was only one field station in a designated depositional zone as compared to over 128 transects in depositional zones used in the DSAS analysis. When long-term shoreline change was analyzed using the WLR at only the transects that intersected the 47 field sampling stations with only one transect intersecting a depositional zone, the mean WLR was 0.41 m/yr of erosion which appears to be equivalent to the mean short-term erosion rate derived from direct sampling methods. This suggests that the short-term direct rates may have overestimated the overall erosion in the study area because of the lack of site representation in depositional zones.

Further direct measurements of depositional zones and point bars may show the presence of accretion. Sand bar and point bar formations and expansion are present upon visual inspection of the available imagery signifying that deposition and accretion are occurring in some areas. There are more areas of erosion than accretion present in both the short-term and long-term analyses. The study area shows signs of an overall widening of the channels. This suggests that the estuarine environment is currently in an erosional phase where overall erosion in the channels is greater than deposition and the formation of more marshland.

Below-ground biomass was greater in erosional zones than in the parallel flow zones indicating that vegetation may not be a dominant controlling factor for erosion in the study area. Much of the observed erosion processes involved large sections of marsh, erosion cusps, (Ginsberg and Perillo 1990), slumping off in sheet-like formation (Figure 58). The existing marsh rests on Pleistocene materials such as beach sands (old sand dunes), and clayee sands (Doar 2003; Figure 59). The slope of the marsh edge in conjunction with soft sediment resting on hard sediment may have more control in this type of erosion process. Ginsberg and Perillo (1990) studied erosional processes in an estuarine channel in Argentina. They determined that erosion cusps are caused by underwater erosion by tidal currents during flood tides and the associated slumping of these features is caused by gravitational pull. Tension cracks are usually associated with erosion cusp formations and indicate slope instability (Ginsberg and Perillo 1990). This process of erosion cusp formation appeared to be present in much of the *spartina* dominated marsh suggesting that the geology and biologic activity may act in congress to destabilize the area in the flow regime of the erosional zone. As the tide moves out and the water level drops, the plants dewater from the inner marsh to the outer bank. As the inner marsh dries, the marsh at the outer bank remains saturated with water and may cause tension cracks resulting from the gravitational pull.

In the area of the projected marina site, vertical banks have formed where the shoreline has been undercut by wave energy (Figures 59 and 60). When the undercutting reaches a threshhold point, the overhanging marsh is no longer supported and falls off in clumps or large masses. In this case, vegetation density and root mass does not appear to mitigate wave energy. Moller and Spencer (2002) found that vertical banks experience a greater amount of wave energy than gradual slopes. In such areas where undercutting occurs, the binding power of the roots does not appear to counter the forceful wave energy. Although many studies have found that vegetation does help mitigate erosion, the erosional mechanisms (undercut banks, and erosion cusp formations) in this area may overpower the binding capabilities of roots and the buffering capacity of stems.

This study did not find a relationship between sand content and erosion, although other studies have found sediment variation to significantly influence erosion. Chose (1999) documented that sand content was greater in areas of higher erosion. In the current study, sediment analysis of deeper cores (1 m +) may have revealed correlations between sediment measurements and erosion rates. The layer of sand found beneath the marsh was usually visible in areas with steep banks where the marsh had eroded back and the underlying sand layer was left exposed forming a platform in front of the marsh. Possibly, the underlying geology may have a greater influence on erosion processes than the sediment composition of the upper marsh layer where the current study focused its sediment collection efforts.

In the current study, no significant relationship between erosion rates and flow was documented even though flow plays a major role in shaping a river channel (Ahnert 1960). Flow is usually greatest just before a river bend forming deep pools near the cutbank. Flow generally starts to slow around the bend allowing deposition to occur usually in the form of sandbars and point bars (Keller 2008). The maximum velocity recorded in this study was approximately 1 m per second. This is consistent with RPI findings in their 2004 studies where the average maximum velocity near the projected marina site was from 0.75 to 0.9 m per sec. The fact that the current study did not find a relationship between flow and short-term shoreline change rate may indicate that the methods used to assess flow were not refined enough to detect a correlation. A relationship may have emerged if a different method of measuring the flow rate was used.

Techniques for armoring shorelines include soft-stabilization and hard-stabilization (NRC 2007). Hard stabilization techniques such as embankments and revetments have been shown to negatively impact adjacent shorelines and displace available sediment. Natural, soft stabilization techniques, such as planting vegetation or the ecologically more sensitive armoring by creating oyster reefs, have become a more common and less intrusive method used in low energy environments (NRC 2007, Piazza 2005)..

Oyster reefs in sheltered shorelines have been shown to effectively mitigate erosion (Meyer et al. 1997, Chose 1999, Piazza et al. 2005). Constructing oyster reefs for erosion control purposes not only utilizes natural and native materials but, also creates new habitat. Productive reefs continue to strengthen and grow after construction and are more efficient in soft sediment than limestone or other heavy materials used in hard stabilization techniques (Piazza 2005). Vegetation planting is also a suitable erosion control technique for fetch limited shorelines (Rogers and Skrabal 2001). Plantings work best between mid and high water lines and in areas of limited boat traffic and wave energy (Rogers and Skrabal 2001). Plantings are not suitable in high energy environments or areas with steep slopes (Rogers and Skrabal 2001).

The area surrounding Palmetto Bluff is well vegetated but does not have many oyster reefs. As part of an on-going study, recruitment of oyster larvae was monitored to determine if oyster reefs could potentially be created to help prevent erosion. Recruitment trays were deployed and the study found that oysters were abundant where substrate was available. Using oyster reefs as an erosion control method in this area could potentially aid in mitigating shoreline loss and could also create essential habitat for numerous species. However, reefs could not be established successfully in areas of erosion where steep slopes occur and undercut banks are prevalent as in the area of the projected marina site.

Future boat traffic is a major concern that could cause devastating erosion problems within the current study area. Several studies have shown that distance from shore and speed of boats are the two main factors influencing boat wake impact (Chose 1999, Macfarlane and Renilson 1999, Anderson 2002). Boats that travel within 50 m or less of the shore at a medium speed have a greater impact than boats either on a plane or idling at the same distance. Boats traveling further than 50 m from the shore have less impact, but still have the least impact when moving under 5 mph or on a plane. Keeping boat traffic at a greater distance from the shore may help prevent accelerated erosion caused by increased boating. This could be accomplished by implementing channel markers and no wake zones.

There is a navigational pathway of particular concern leading from the projected marina site at Big House Landing to Mooreland Landing or the Village Marina. This pathway is Ramshorn Creek and is the most narrow channel in the study area (Stations 1 – 7 encompass a portion of Ramshorn Creek). Because boaters are inevitably close to the shore in Ramshorn Creek, channel markers will not be an effective method for limiting boat impacts. A no-wake zone could be implemented, but the length of the zone would likely need to extend the entire length of the channel, which is more than a 5 mile stretch due to the sinuous nature of the channel. Currently, the primary users of this pathway are crabbers, employees of Palmetto Bluff, jet skiers, and kayakers. The most effective way to protect Ramshorn Creek may be to designate it as off-the-map area and limit its use to kayakers, crabbers, and employees. Ramshorn Creek may not be a suitable pathway since most boats cannot get through the area of concern at low or mid tide. A map displaying alternate routes from Big House Landing to the Village Marina and Mooreland landing could provide boaters with preferable alternative routes.

CONCLUSIONS

The estuarine environment in the study area is currently in a healthy natural state. The system exhibits all of the characteristics that would be expected in a low gradient estuarine environment. The stream morphology currently controls the erosional regions of the system. The USGS's Digital Shoreline Analysis System proved to be a viable and valuable tool for assessing erosional rates in the study area. Using aerial photography seems to be a time effective technique for doing a study of this type and should be used for planning the field design of similar studies. The correspondence between the measured short-term rates with the long-term rates derived from the aerial photography show that the region currently has a stable erosional profile. The study is not able to make conclusive statements on the degree to which geology or biology currently control the erosion in the region. More than likely the controls are a combination of geology and biology acting together in a site specific format. Statistical results did indicate significant variability at the site level.

This study supports the understanding that the estuarine system examined is predominantly unaffected by anthropogenic activity. The development boom that is evident throughout South Carolina has not affected this system at the present time since the long-term and short-term erosion rates are nearly equal. This is important because the study can be used as a baseline to investigate the effects of the permitted marina as it opens and begins to be used by boaters. There are several likely outcomes of placing the marina in the study area. Based on the work of Price (2006) and Chose (1999), the wave action of the boats will increase the wave energy in the channels leading to more erosion and a widening of the channels over time. The nature of the estuarine system is to move to an equilibrium point suggesting that when the channel widens sufficiently the processes will stabilize and return to acting like a more natural system (with a higher energy level).

One of the most ideal conservation strategies for reducing environmental degradation of the study area would be for the Palmetto Bluff Development to provide educational brochures and conservation overviews for resident boaters and getting the community involved in all protection initiatives. Residents may be more likely to follow community guidelines and suggestions if they know the potential implications of their actions and they have an investment in preserving the area in which they live.

Recommendations for future research

Shoreline change research in the current study area should be on-going. Continued monitoring of shoreline change through direct ground measurements throughout the creation of the marina will help in the overall understanding of anthropogenic impacts on protected shorelines. In addition, examination of the bathymetry and how it relates to erosion would help in understanding where the eroding sediments are accumulating. A study in Monterey Bay, California, has been monitoring tidal scour and channel erosion in relation to inlet stability through the use of bathymetry data and they found severe erosion and scour along curves of the tidal channel that had once been areas of deposition (Israel and Watt 2006). Bathymetry data would allow the researcher to identify areas of deposition and have an accurate assessment of sandbar migration. As previously discussed, large sections of marsh have been observed slumping off into the channel. Detailed bathymetry data could help in understanding what happens to the marsh slumps once they fall into the channel.

Vertical erosion and accretion measurements would also aid in understanding marsh processes. Castillo et al. (2002) found a positive linear correlation between horizontal bank erosion and vertical intertidal plain erosion and accretion signifying that eroded sediments were not being deposited in the intertidal region adjacent to the eroding bank. French and Burningham (2003) measured vertical accretion of the marsh surface in southeast England and found that marsh accretion was occurring at levels that far exceeded sea-level rise. If accretion had been less than the relative sea-level rise rate, it would have indicated marsh subsidence. In the current study, both short-term and long-term shoreline change measurements indicated an overall erosional trend. Vertical accretion measurements would help to determine if eroded sediment is being redeposited on the marsh surface during flood tides.

SCDNR and OCRM have been monitoring water quality in areas surrounding Palmetto Bluff. Future monitoring efforts should incorporate the impacts of run-off on erosion and water quality as impervious surfaces increase. Monitoring and mapping marsh vegetation trends will also be useful in assessing the role of marsh vegetation in mitigating development impacts; such as, run-off and soil erosion.

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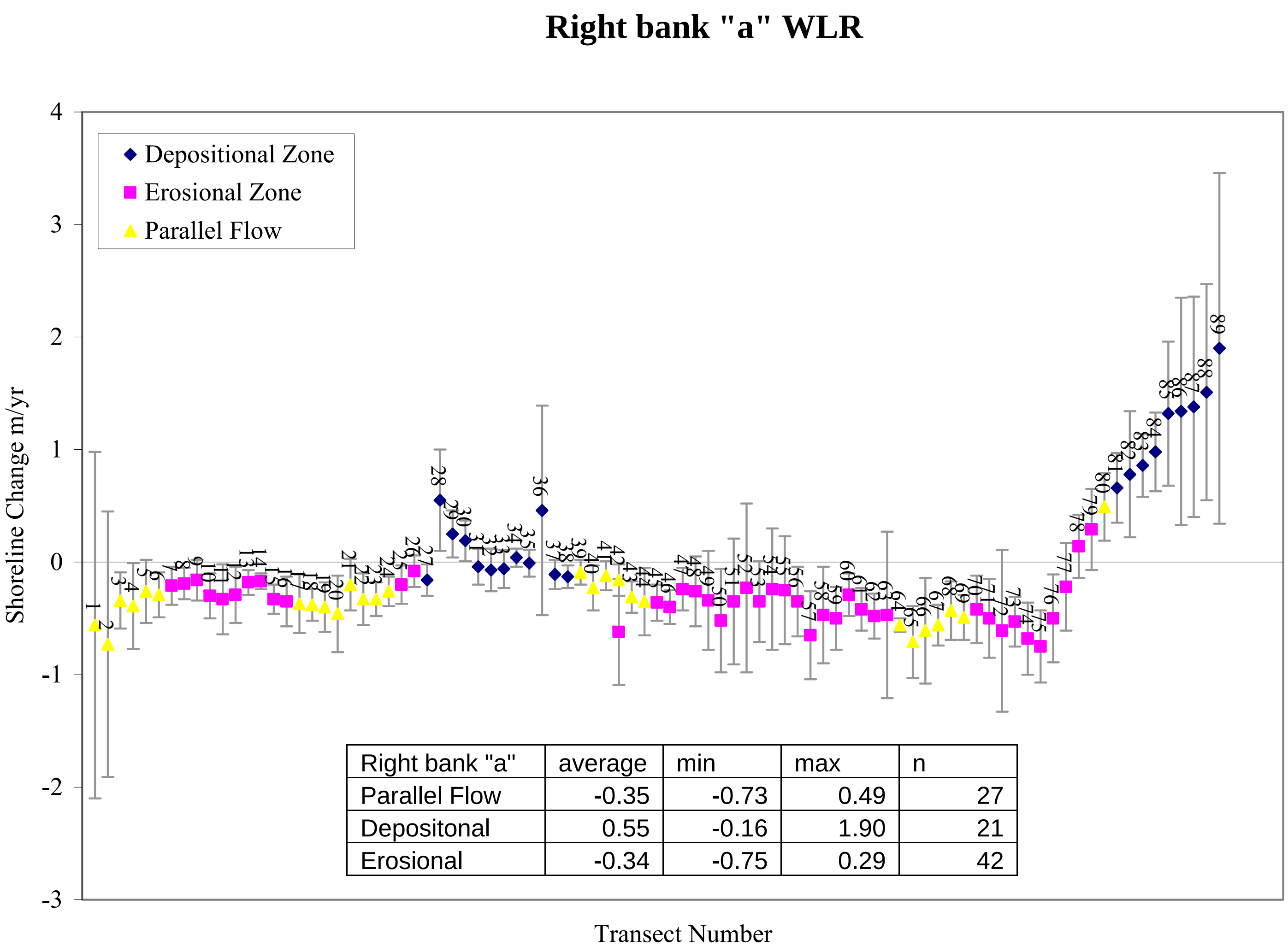
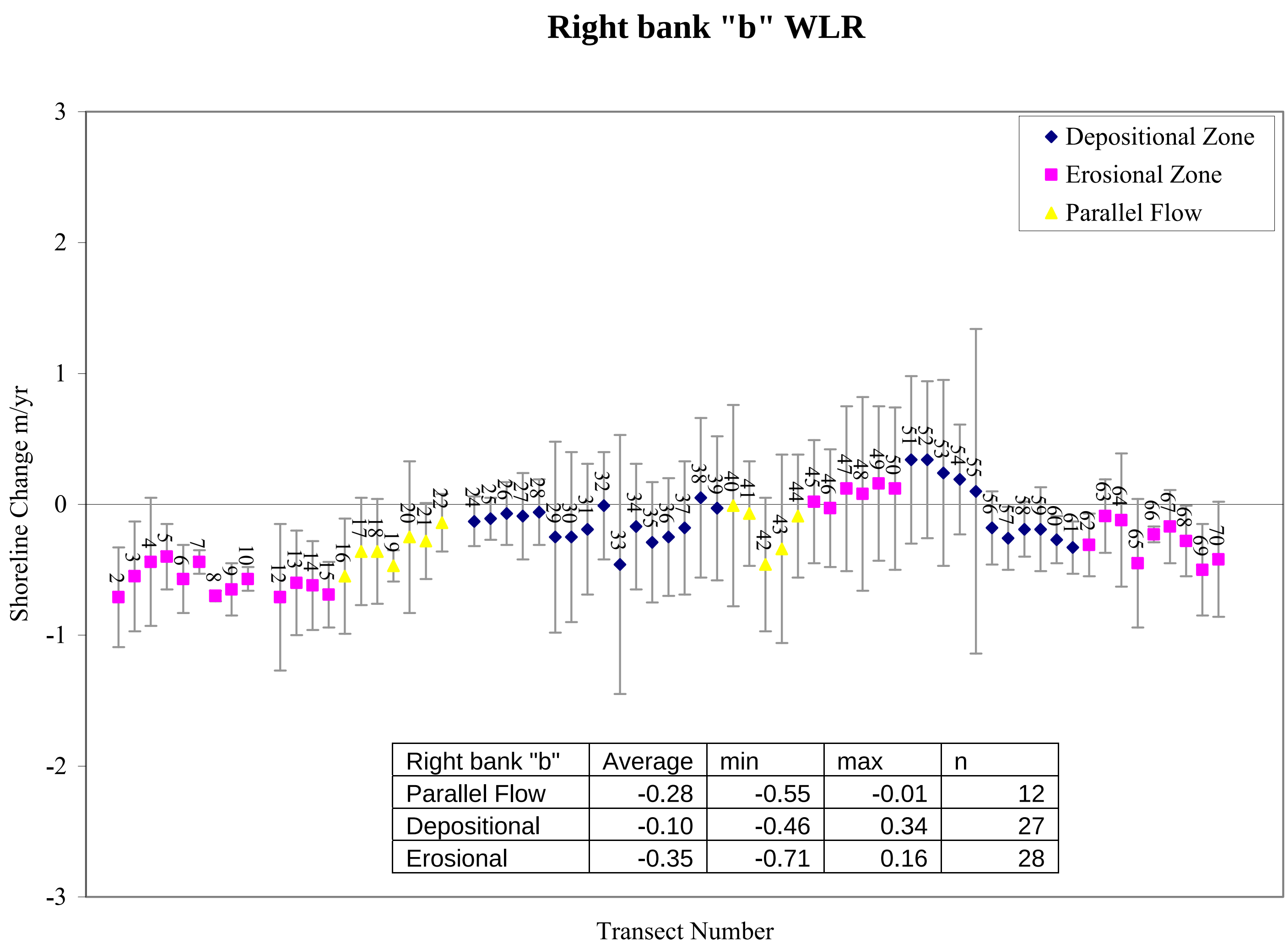
APPENDIX A



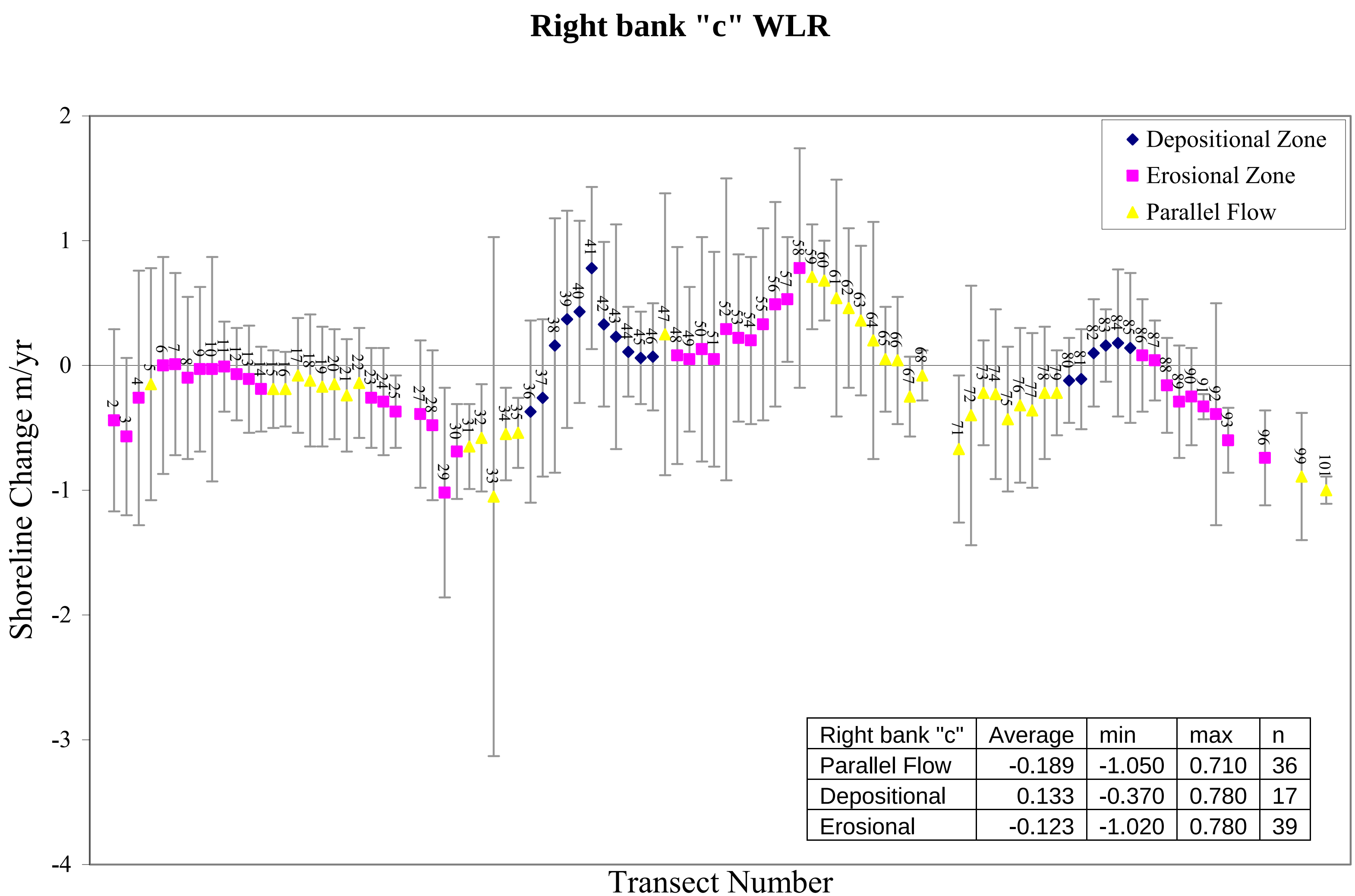
Right bank "a"	average	min	max	n
Parallel Flow	-0.35	-0.73	0.49	27
Depositonal	0.55	-0.16	1.90	21
Erosional	-0.34	-0.75	0.29	42

Figure 28 The weighted linear regression (WLR) was created by the DSAS. A weight is assigned to each year based on the accuracy of imagery defined by the user. A linear regression rate is then calculated for shoreline change. Depositional zones of right bank “a” incurred the least erosion. Erosional and parallel flow zone incurred similar average erosion rates.



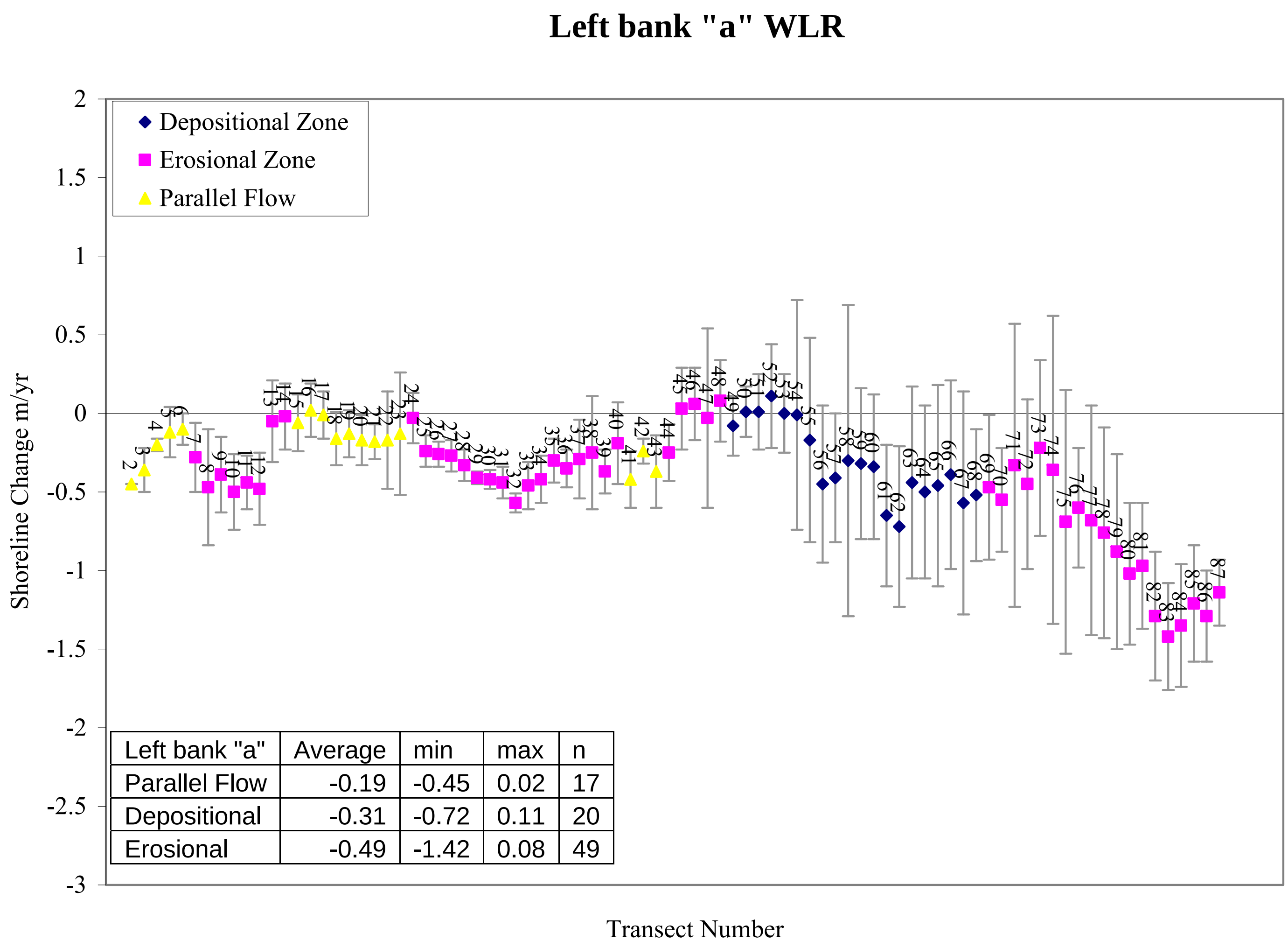
Right bank "b"	Average	min	max	n
Parallel Flow	-0.28	-0.55	-0.01	12
Depositional	-0.10	-0.46	0.34	27
Erosional	-0.35	-0.71	0.16	28

Figure 29: The weighted linear regression (WLR) was created by the DSAS. A weight is assigned to each year based on the accuracy of imagery defined by the user. A linear regression rate is then calculated for shoreline change. Depositional zones of right bank “b” incurred the least erosion followed by parallel flow zones and erosional zones.



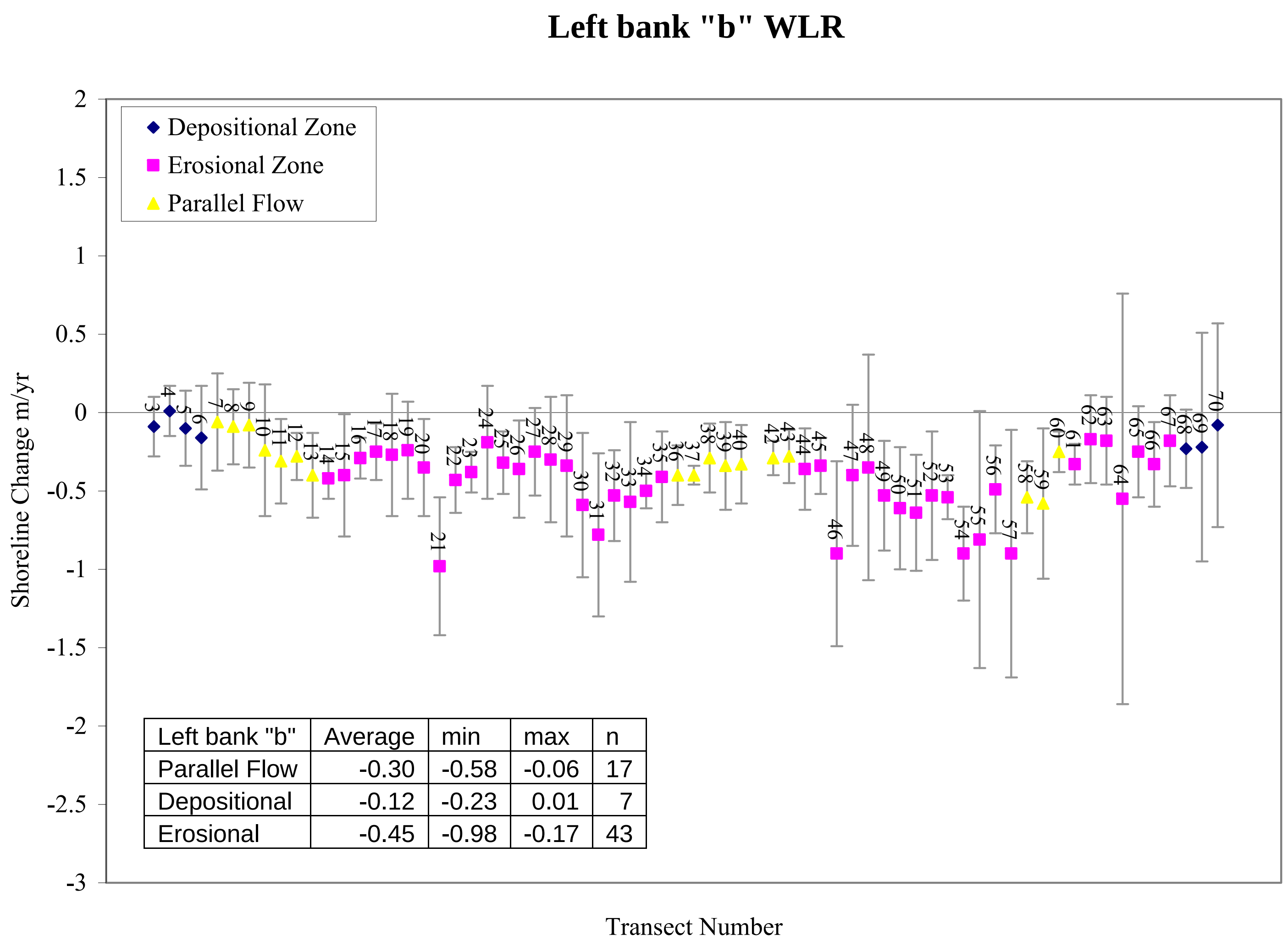
Right bank "c"	Average	min	max	n
Parallel Flow	-0.189	-1.050	0.710	36
Depositional	0.133	-0.370	0.780	17
Erosional	-0.123	-1.020	0.780	39

Figure 30: The weighted linear regression (WLR) was created by the DSAS. A weight is assigned to each year based on the accuracy of imagery defined by the user. A linear regression rate is then calculated for shoreline change. Depositional zones of right bank “c” incurred the least erosion followed by erosional zones and parallel flow zones.



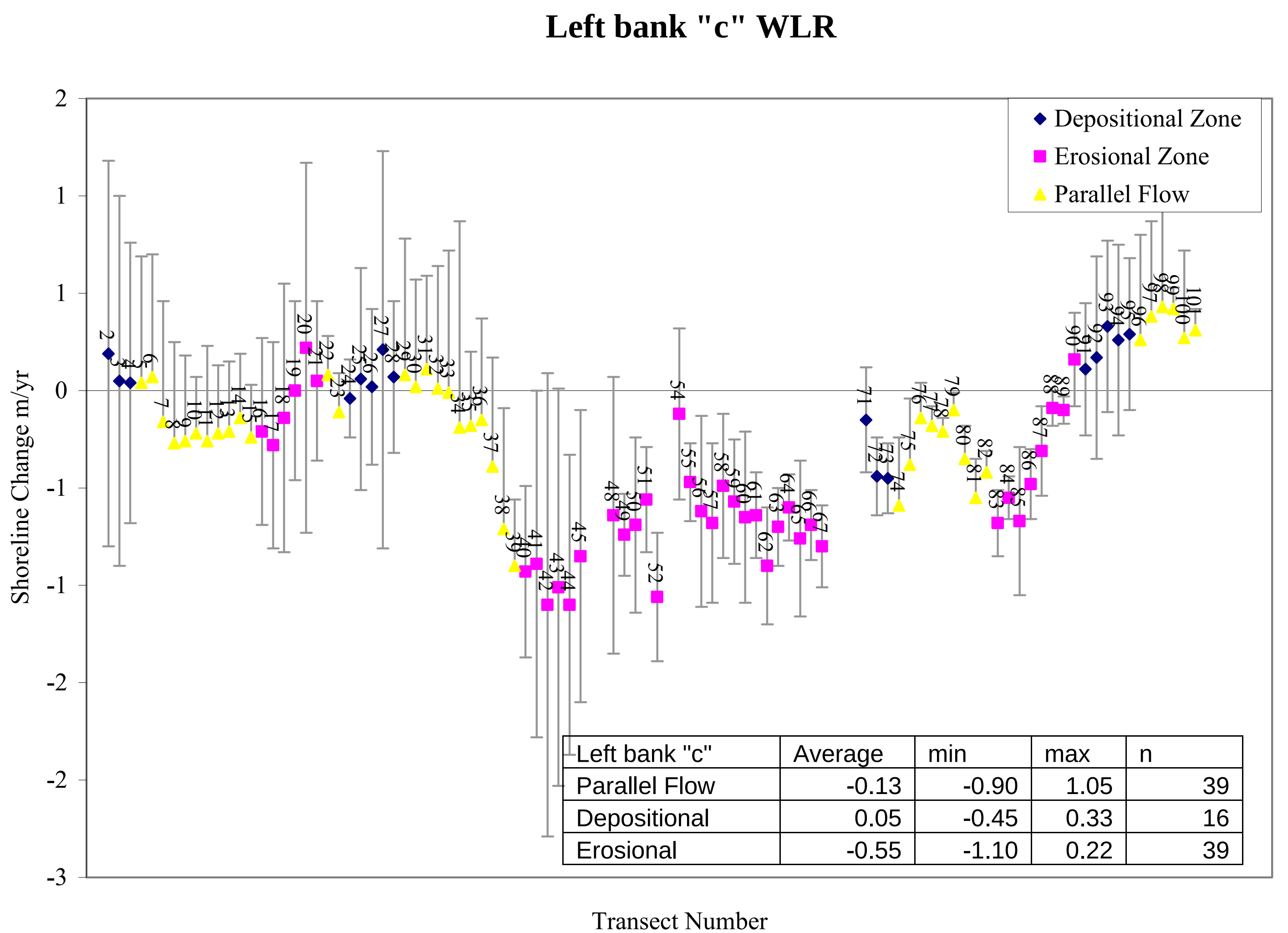
Left bank "a"	Average	min	max	n
Parallel Flow	-0.19	-0.45	0.02	17
Depositional	-0.31	-0.72	0.11	20
Erosional	-0.49	-1.42	0.08	49

Figure 31: The weighted linear regression (WLR) was created by the DSAS. A weight is assigned to each year based on the accuracy of imagery defined by the user. A linear regression rate is then calculated for shoreline change. parallel flow zones of left bank “a” incurred the least erosion followed by Depositional zones and erosional zones.



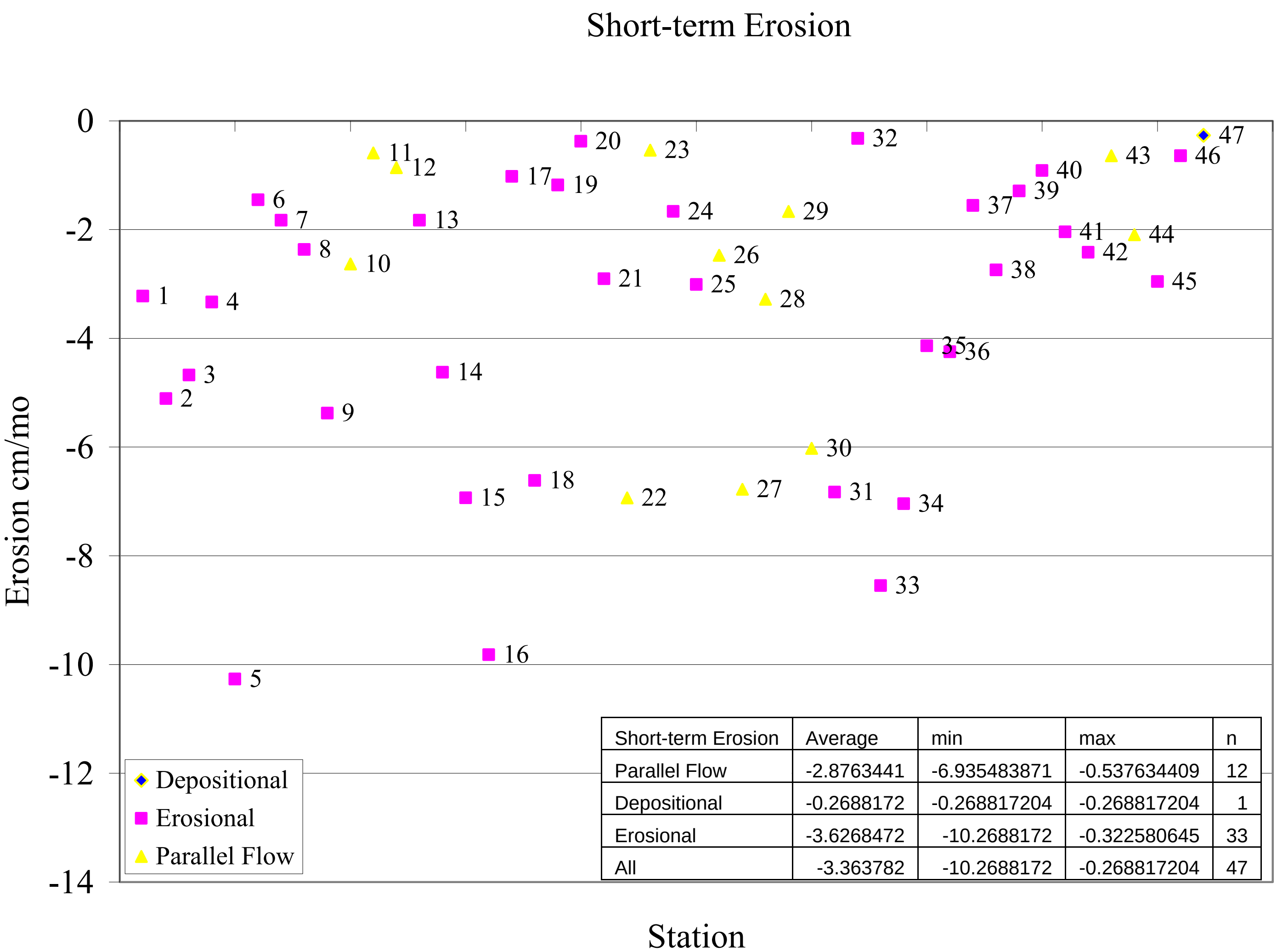
Left bank "b"	Average	min	max	n
Parallel Flow	-0.30	-0.58	-0.06	17
Depositional	-0.12	-0.23	0.01	7
Erosional	-0.45	-0.98	-0.17	43

Figure 32: The weighted linear regression (WLR) was created by the DSAS. A weight is assigned to each year based on the accuracy of imagery defined by the user. A linear regression rate is then calculated for shoreline change. Depositional zones of left bank “b” incurred the least erosion followed by parallel flow zones and erosional zones.



Left bank "c"	Average	min	max	n
Parallel Flow	-0.13	-0.90	1.05	39
Depositional	0.05	-0.45	0.33	16
Erosional	-0.55	-1.10	0.22	39

Figure 33: The weighted linear regression (WLR) was created by the DSAS. A weight is assigned to each year based on the accuracy of imagery defined by the user. A linear regression rate is then calculated for shoreline change. Depositional zones of right bank "c" incurred the least erosion followed by parallel flow zones and erosional zones.



Short-term Erosion	Average	min	max	n
Parallel Flow	-2.8763441	-6.935483871	-0.537634409	12
Depositional	-0.2688172	-0.268817204	-0.268817204	1
Erosional	-3.6268472	-10.2688172	-0.322580645	33
All	-3.363782	-10.2688172	-0.268817204	47

Figure 34: The erosion rate per month was determined by dividing the cumulative erosion rate by the total number of months deployed. Erosion rates for stations 1 – 25 were underestimated due to lost data in December of 2005. Overall 33 stations were located in an erosional zone, 12 stations were located in a depositional zone and 1 station was located in a depositional zone.

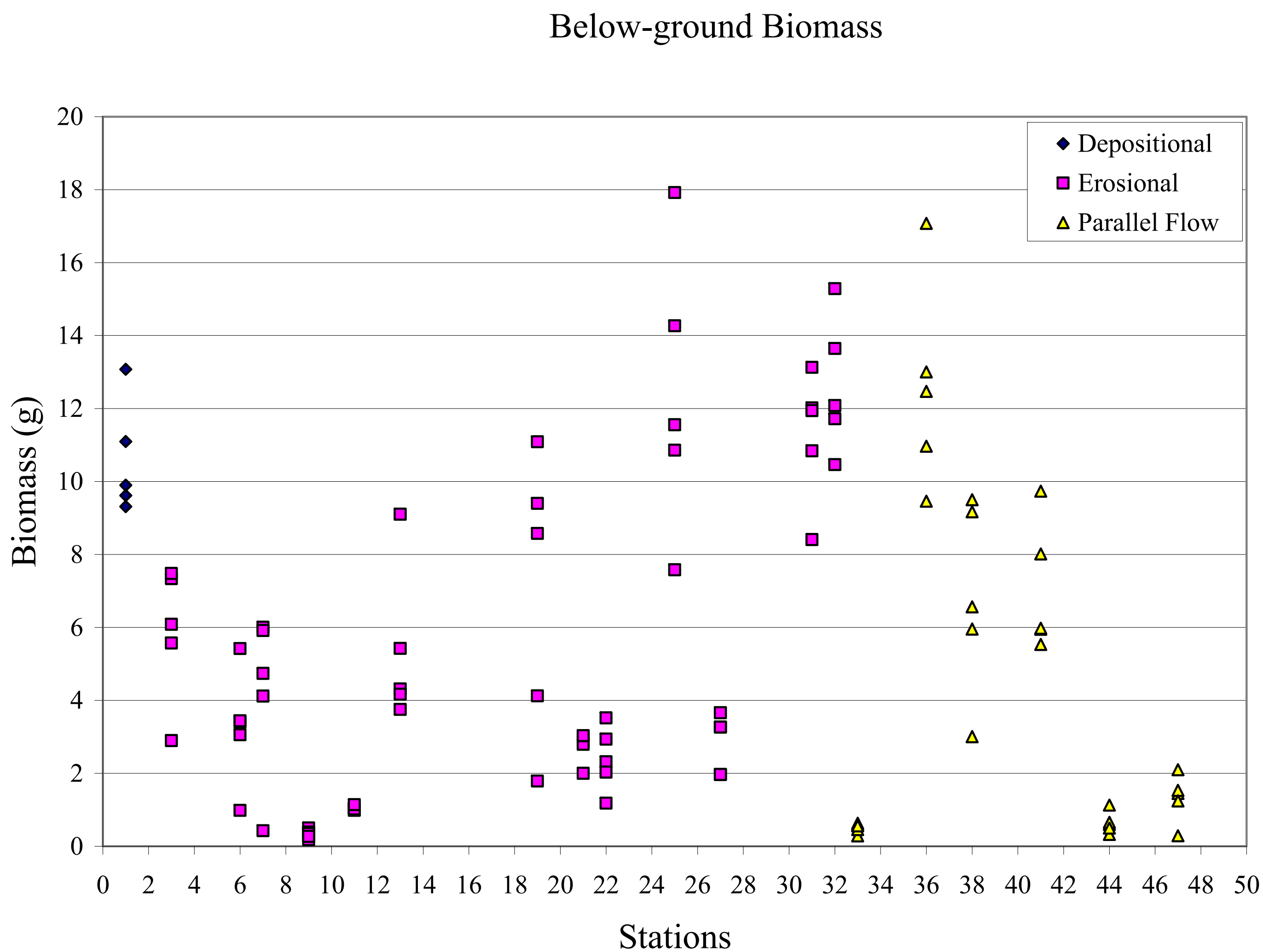


Figure 35: Total below-ground biomass per station and as depicted by zone. Overall, 65 cores were located in erosional zones, 30 were located in parallel flow zones, and five were located in depositional zones.

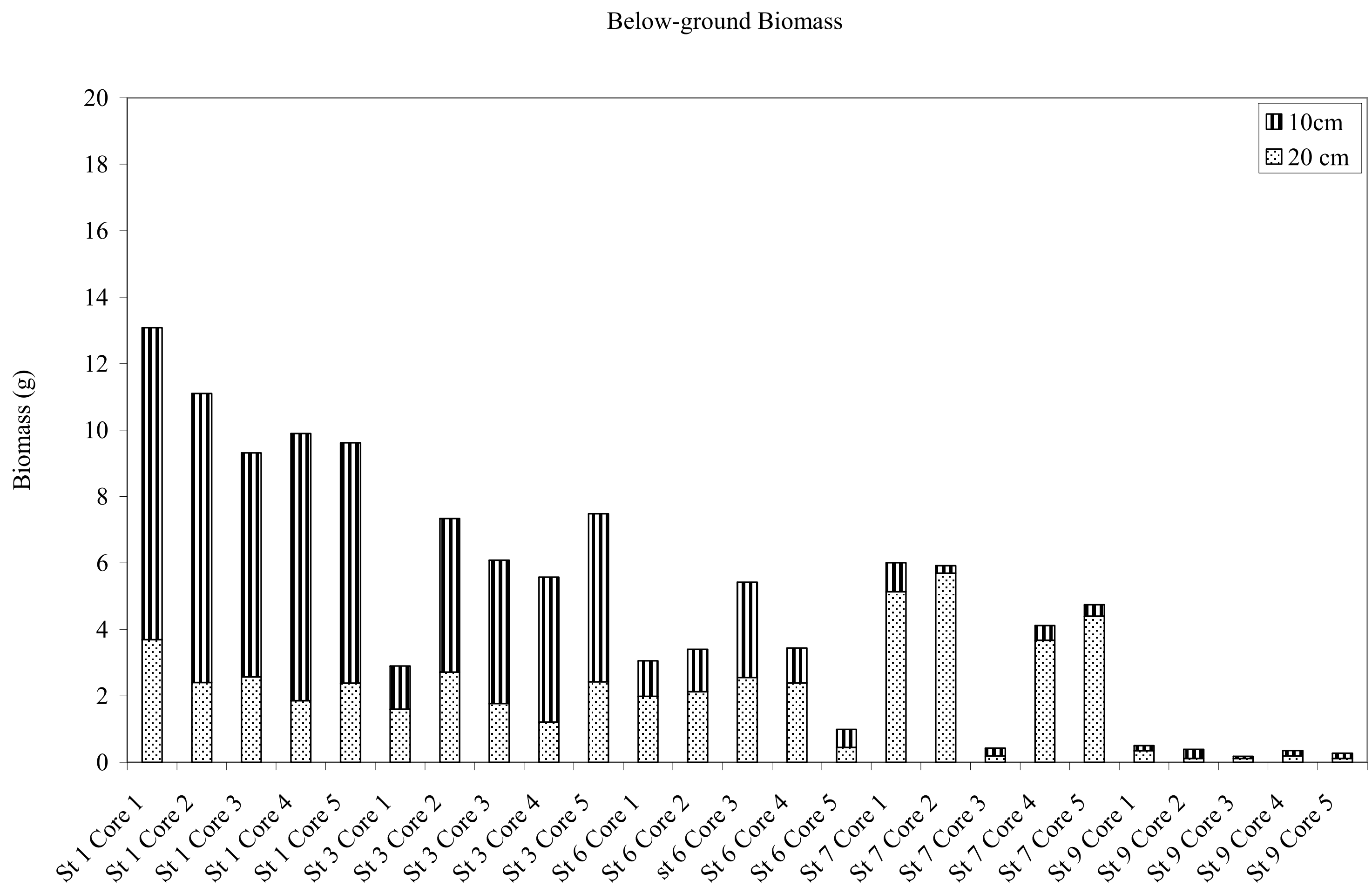


Figure 36: Cores were extracted at 20 cm depth. In the lab, cores were cut into sections and analyzed to determine if belowground biomass was greater from 0 - 10 cm depth or 10 – 20 cm in depth.

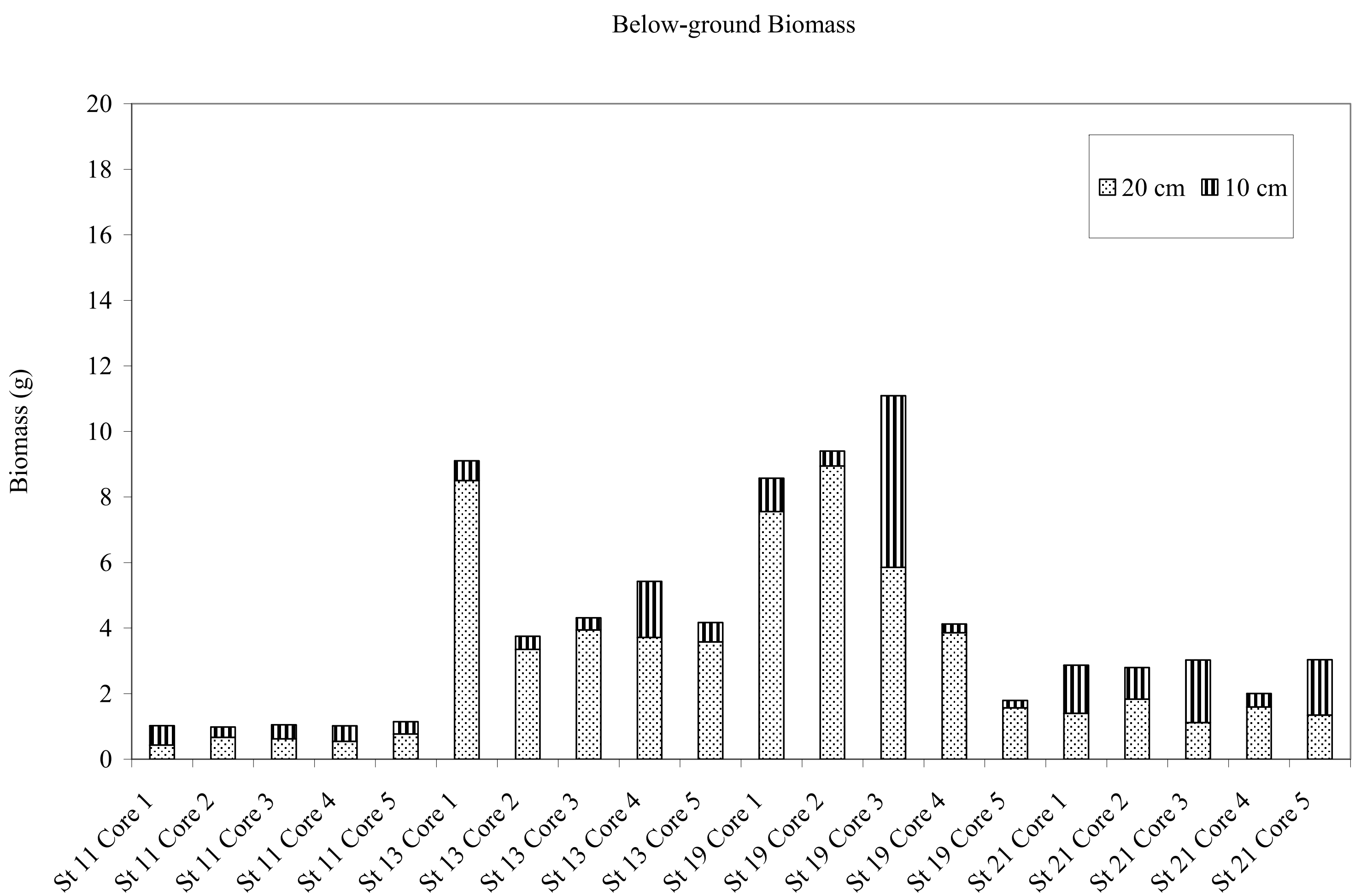


Figure 37: Cores were extracted at 20 cm depth. In the lab, cores were cut into sections and analyzed to determine if belowground biomass was greater from 0 - 10 cm depth or 10 – 20 cm in depth.

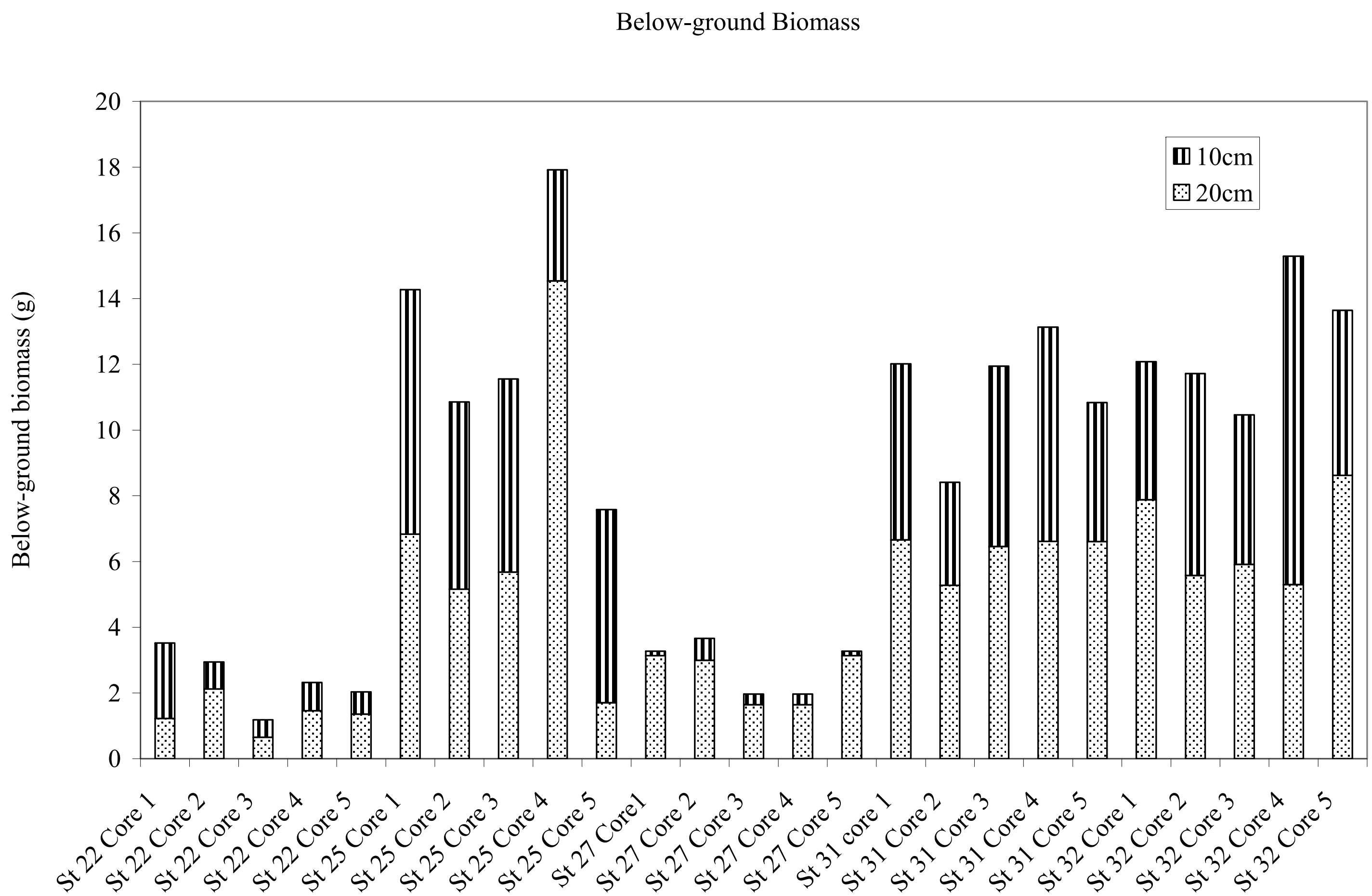


Figure 38: Cores were extracted at 20 cm depth. In the lab, cores were cut into sections and analyzed to determine if belowground biomass was greater from 0 - 10 cm depth or 10 – 20 cm in depth.

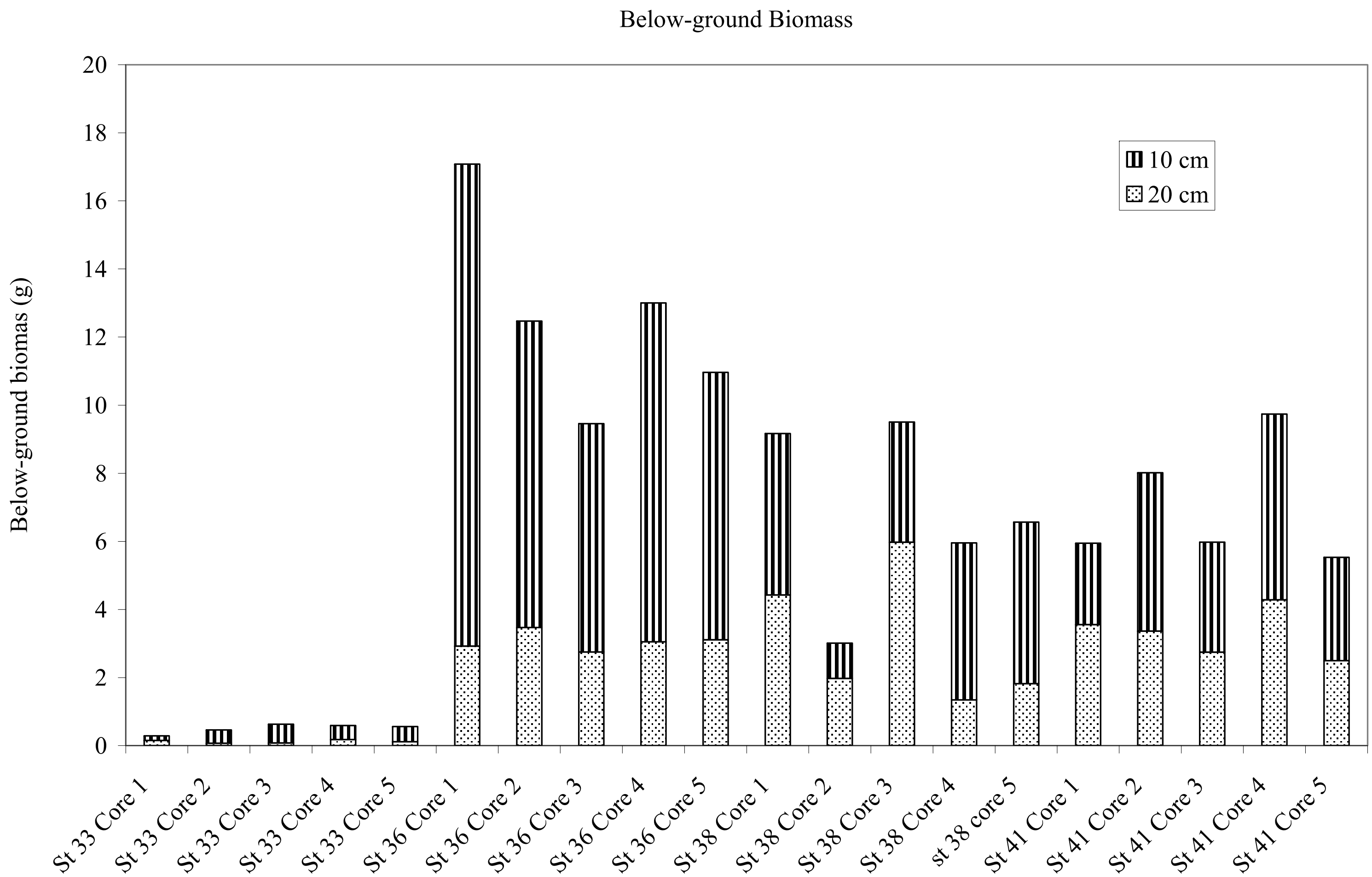


Figure 39: Cores were extracted at 20 cm depth. In the lab, cores were cut into sections and analyzed to determine if belowground biomass was greater from 0 - 10 cm depth or 10 – 20 cm in depth.

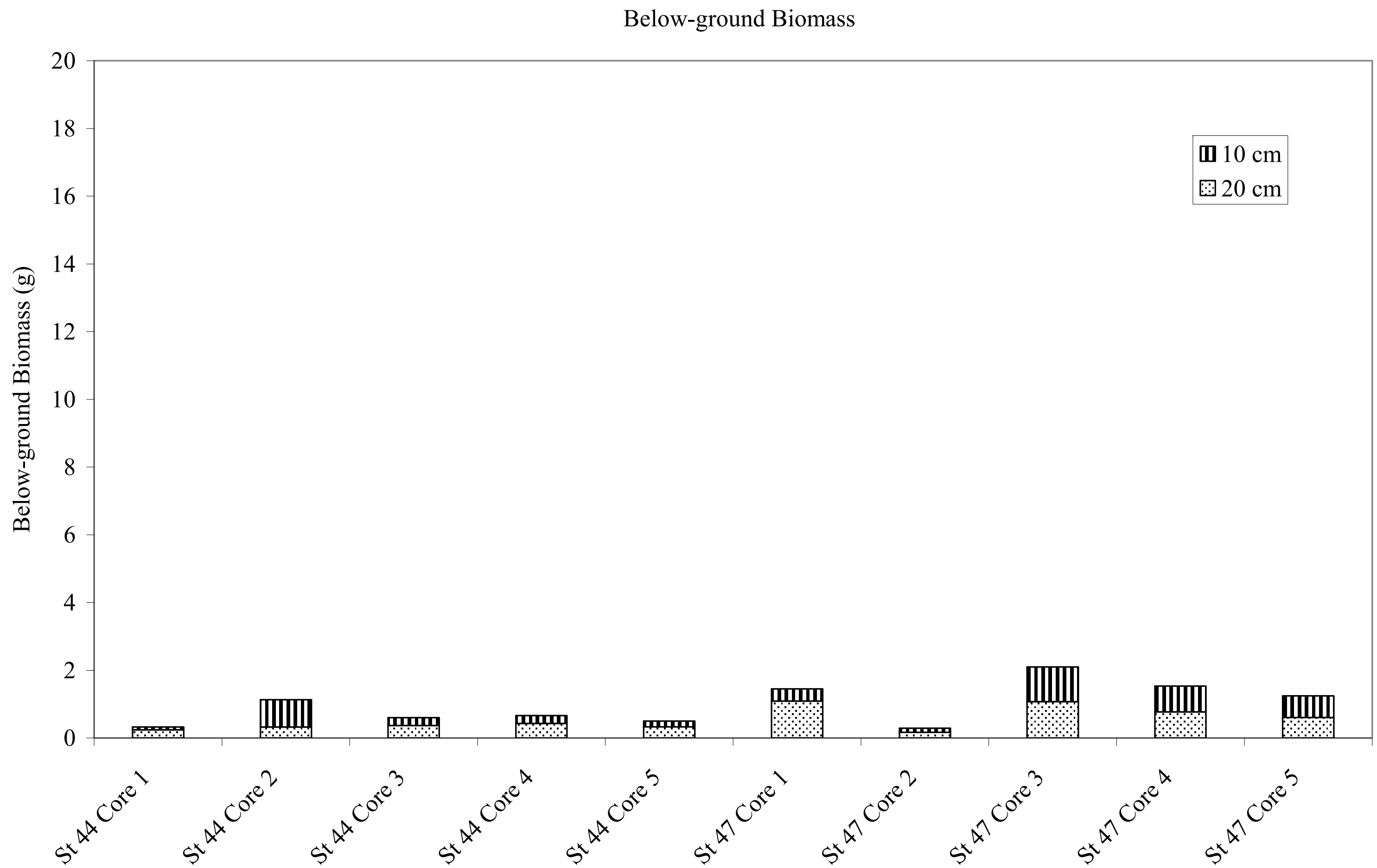


Figure 40: Cores were extracted at 20 cm depth. In the lab, cores were cut into sections and analyzed to determine if belowground biomass was greater from 0 - 10 cm depth or 10 – 20 cm in depth.

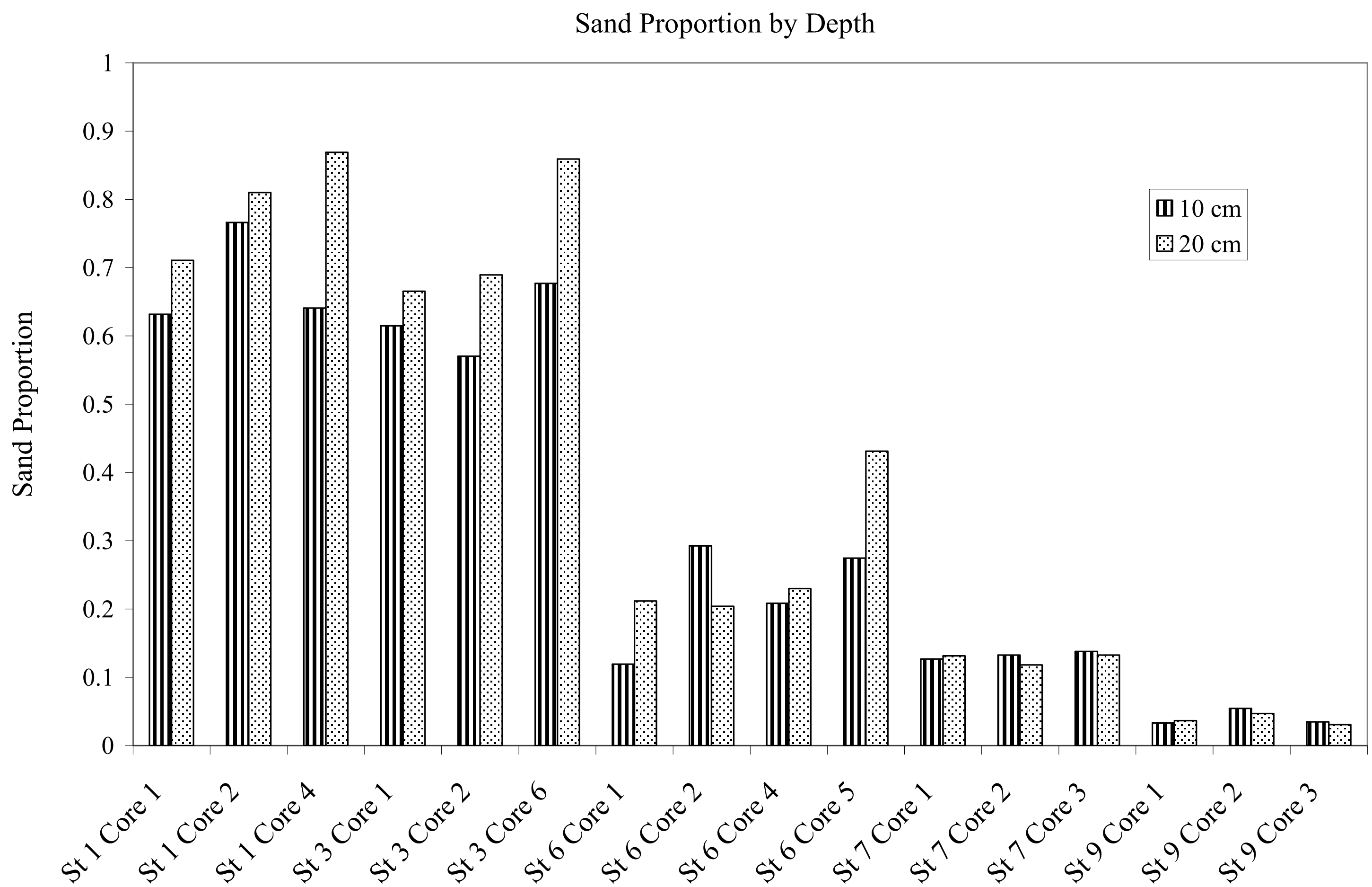


Figure 41: Cores were extracted at 20 cm depth. In the lab, cores were cut into sections and analyzed to determine if sand content was greater from 0 - 10 cm depth or 10 – 20 cm in depth.

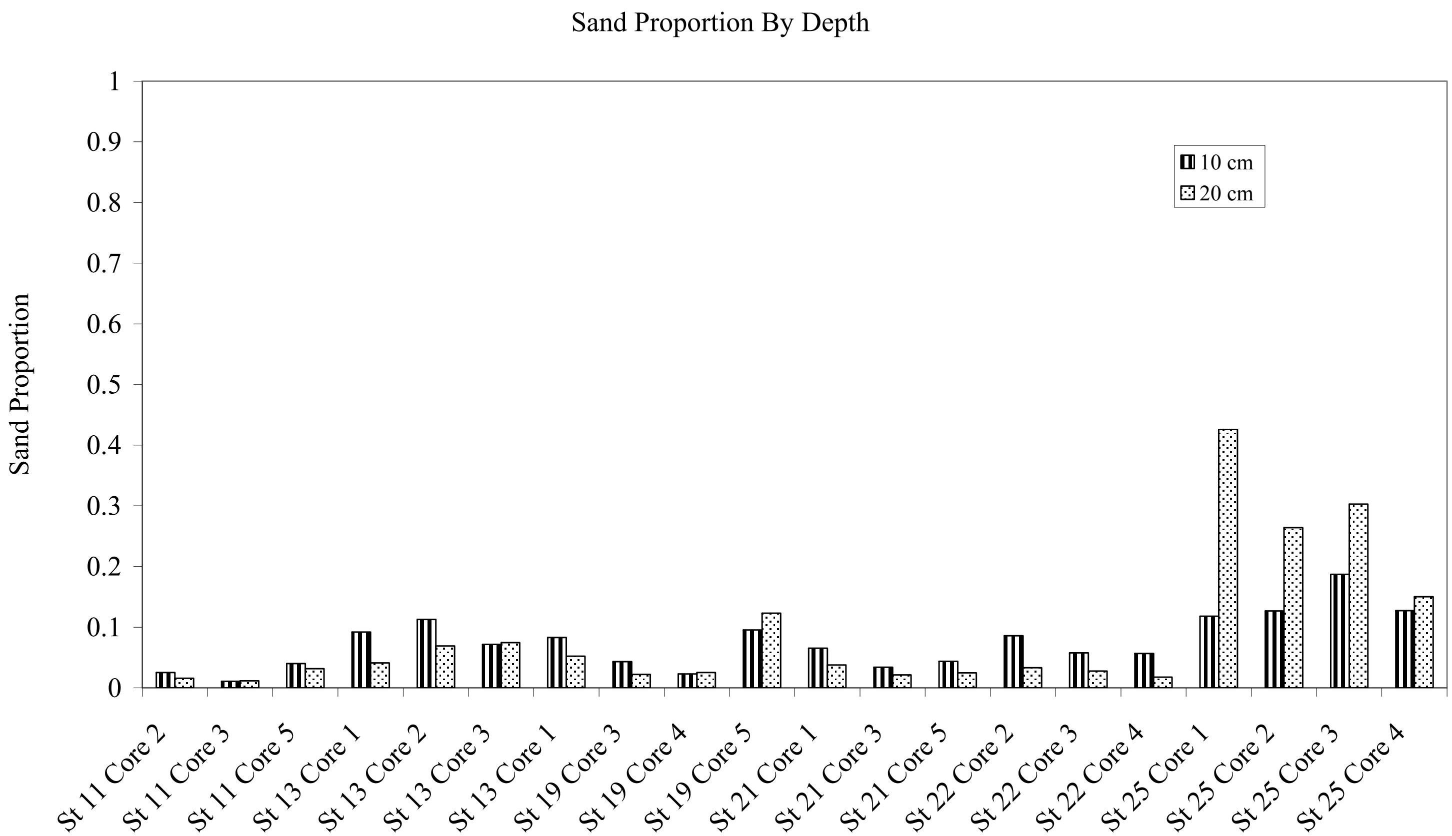


Figure 42: Cores were extracted at 20 cm depth. In the lab, cores were cut into sections and analyzed to determine if sand content was greater from 0 - 10 cm depth or 10 – 20 cm in depth

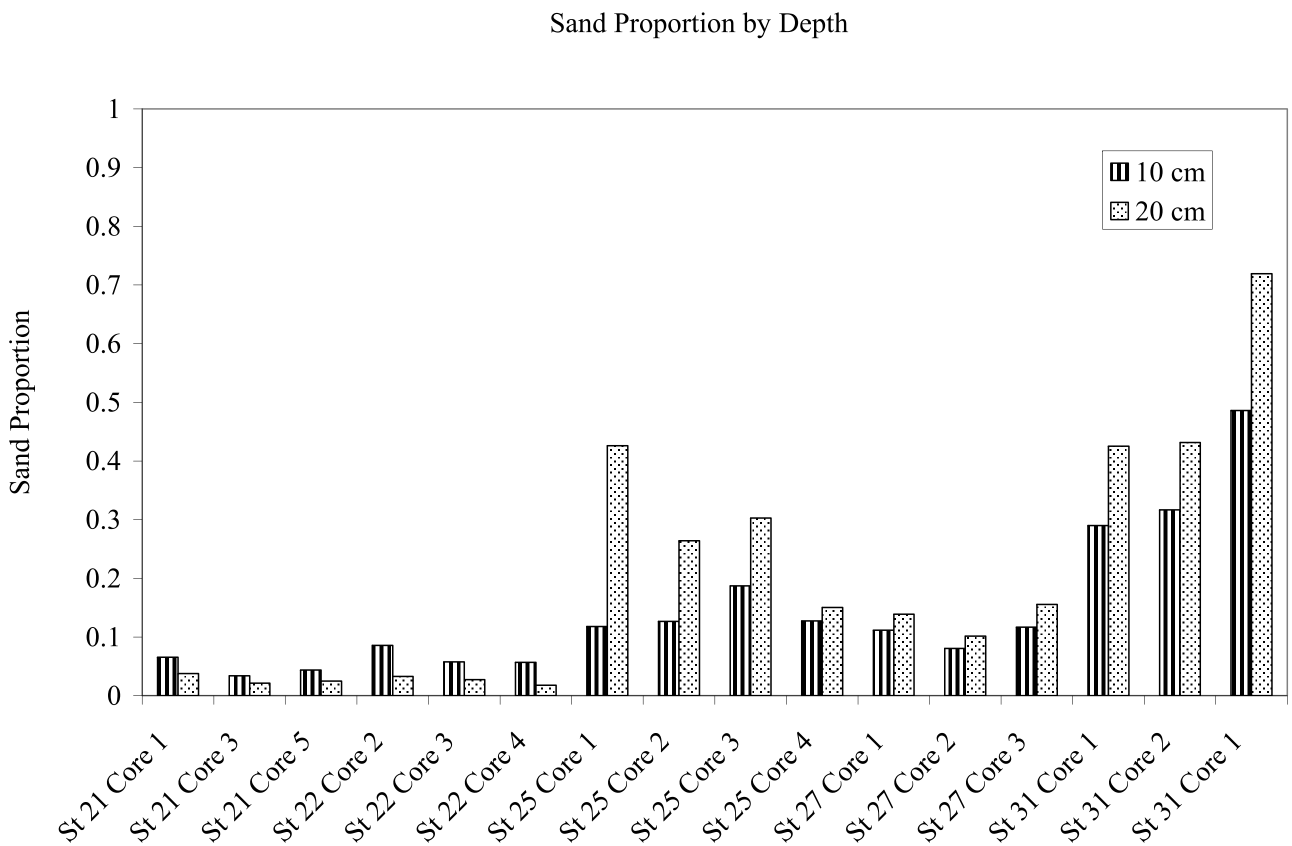


Figure 43: Cores were extracted at 20 cm depth. In the lab, cores were cut into sections and analyzed to determine if sand content was greater from 0 - 10 cm depth or 10 – 20 cm in depth.

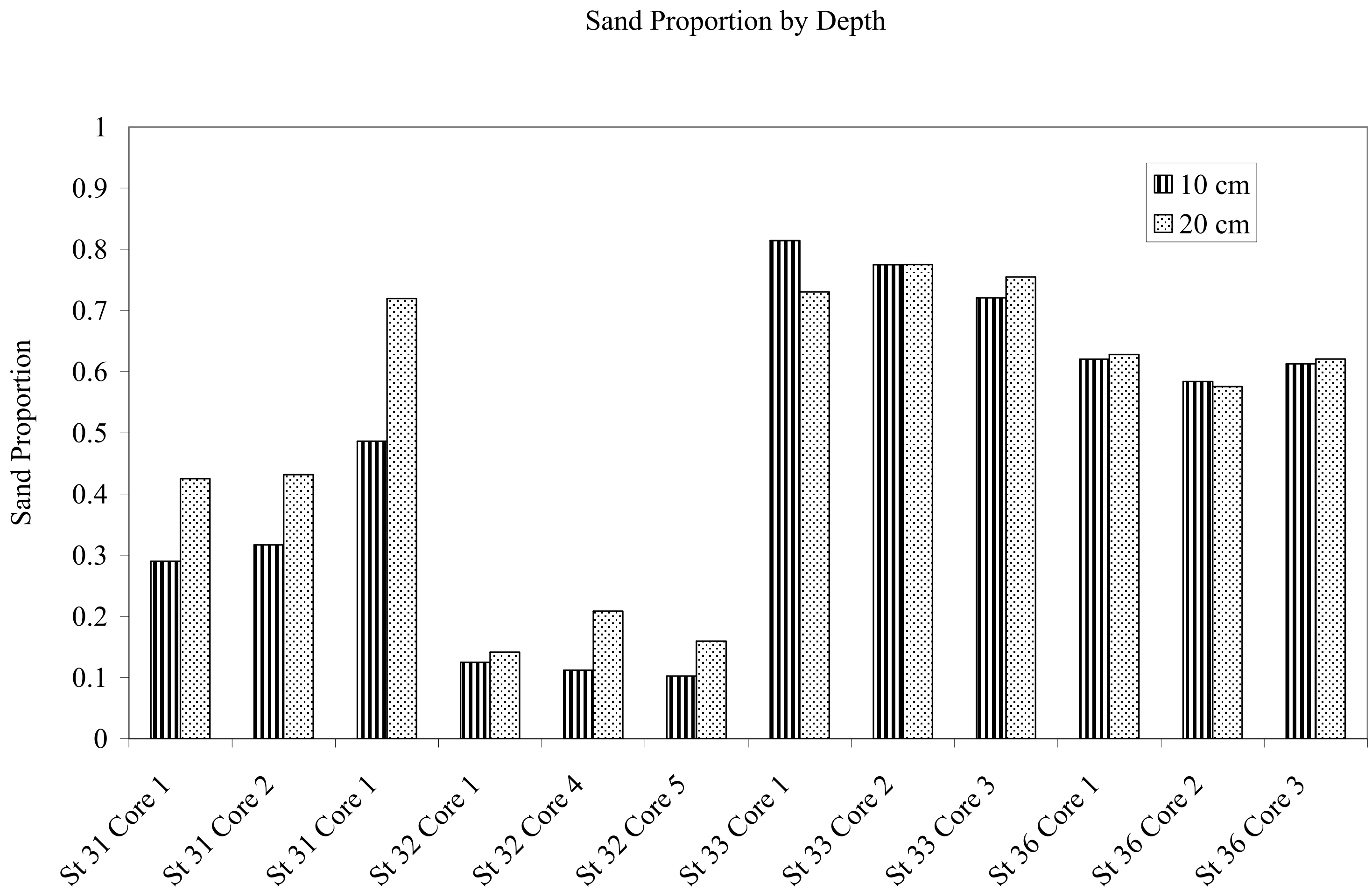


Figure 44: Cores were extracted at 20 cm depth. In the lab, cores were cut into sections and analyzed to determine if sand content was greater from 0 - 10 cm depth or 10 – 20 cm in depth.

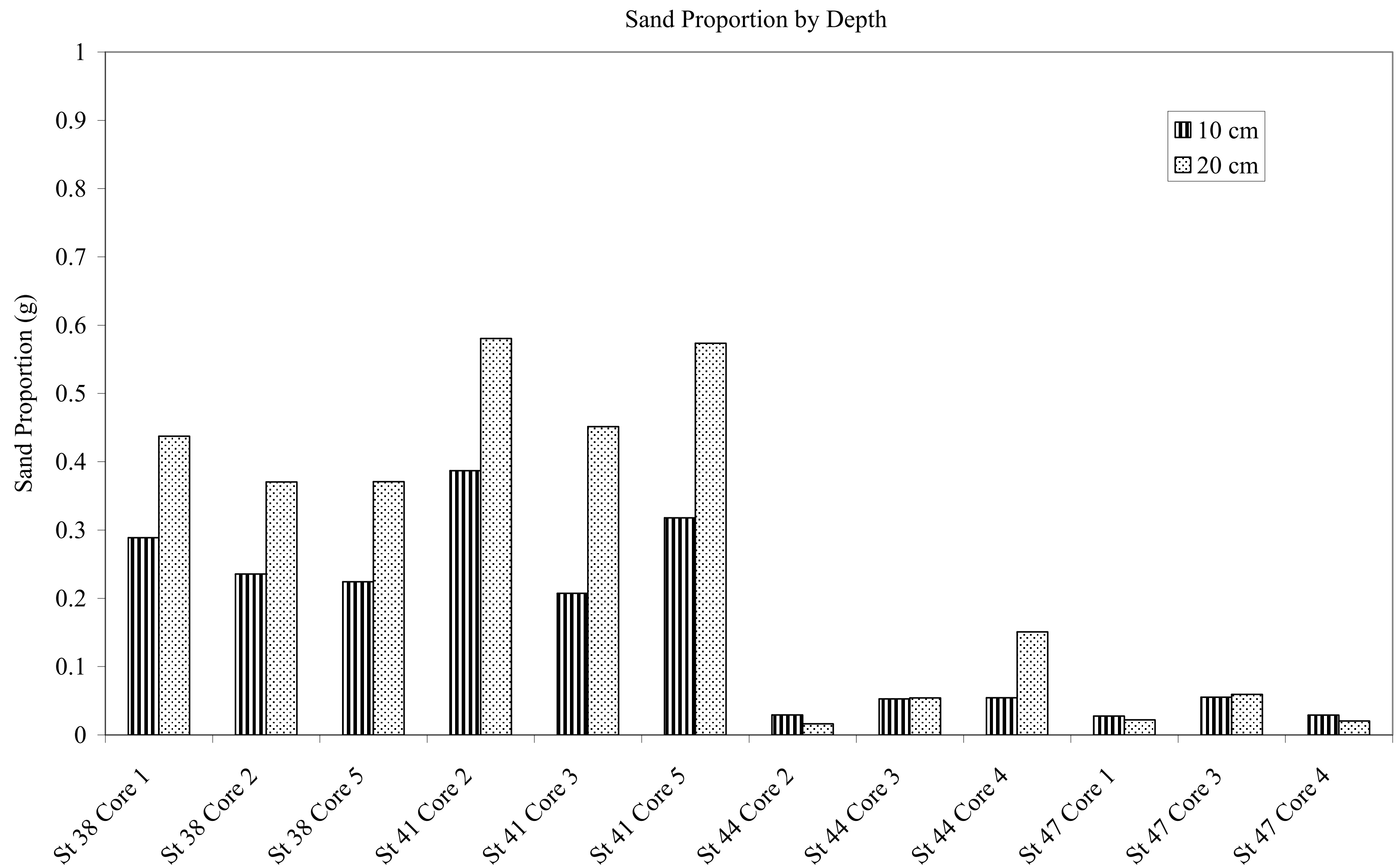


Figure 45: Cores were extracted at 20 cm depth. In the lab, cores were cut into sections and analyzed to determine if sand content was greater from 0 - 10 cm depth or 10 – 20 cm in depth.

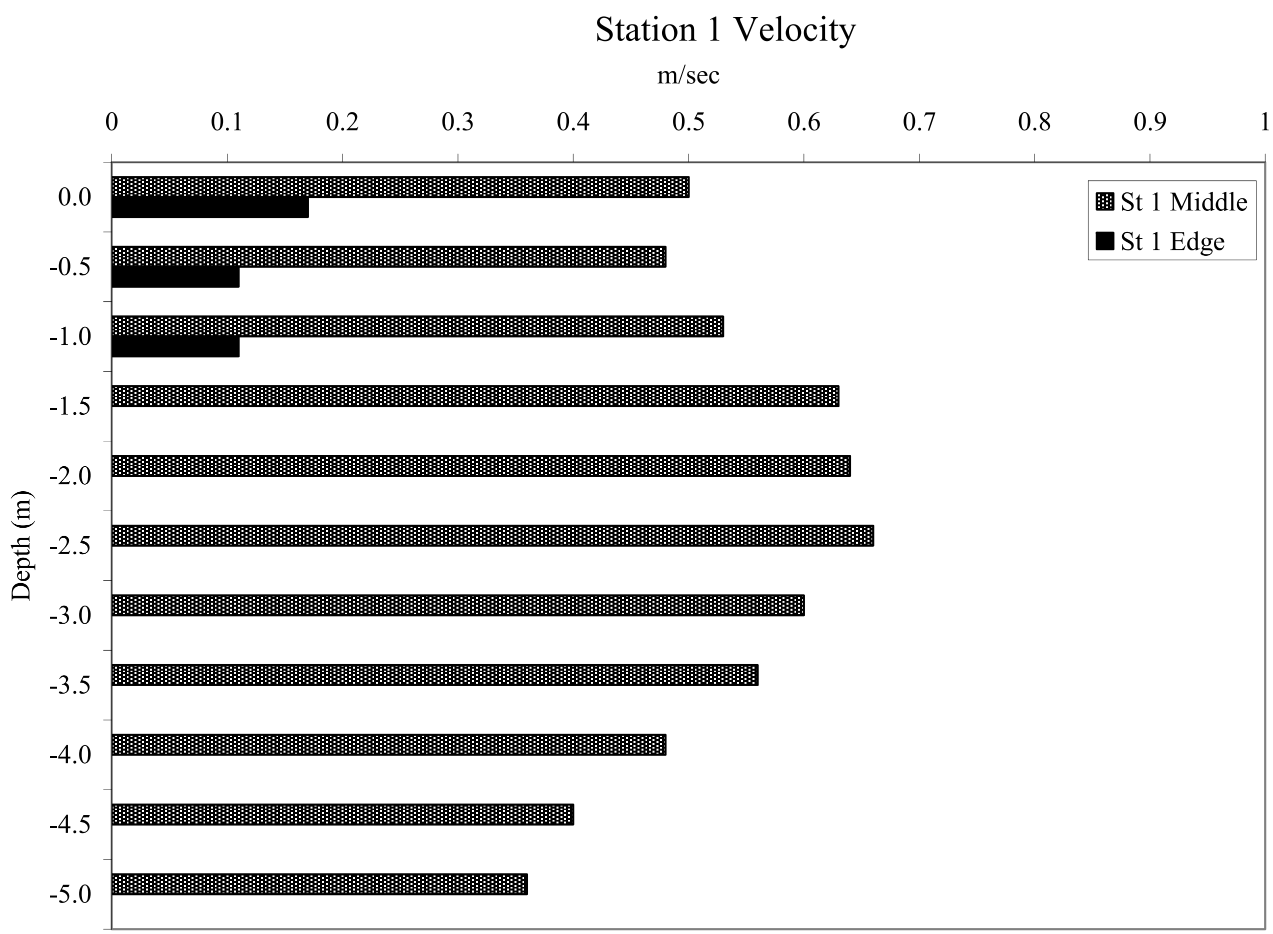


Figure 46: Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

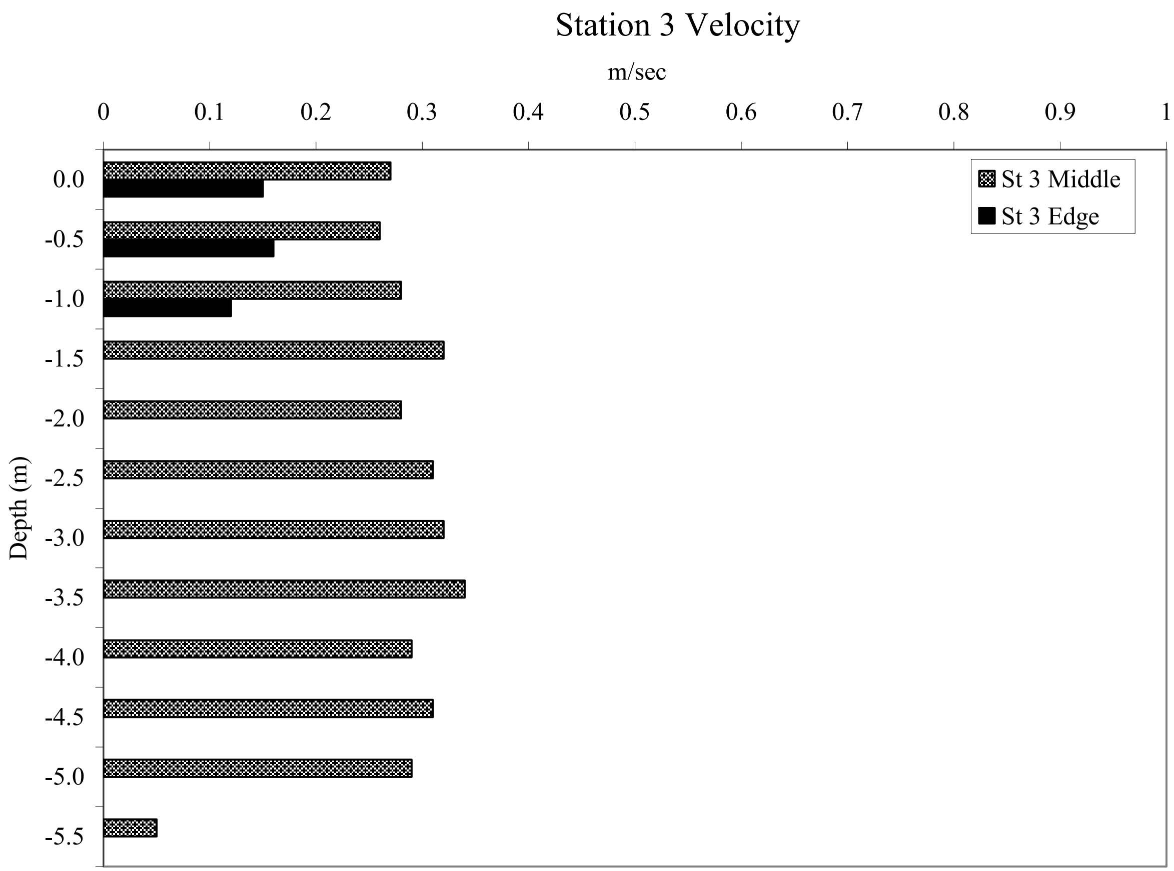


Figure 47 : Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

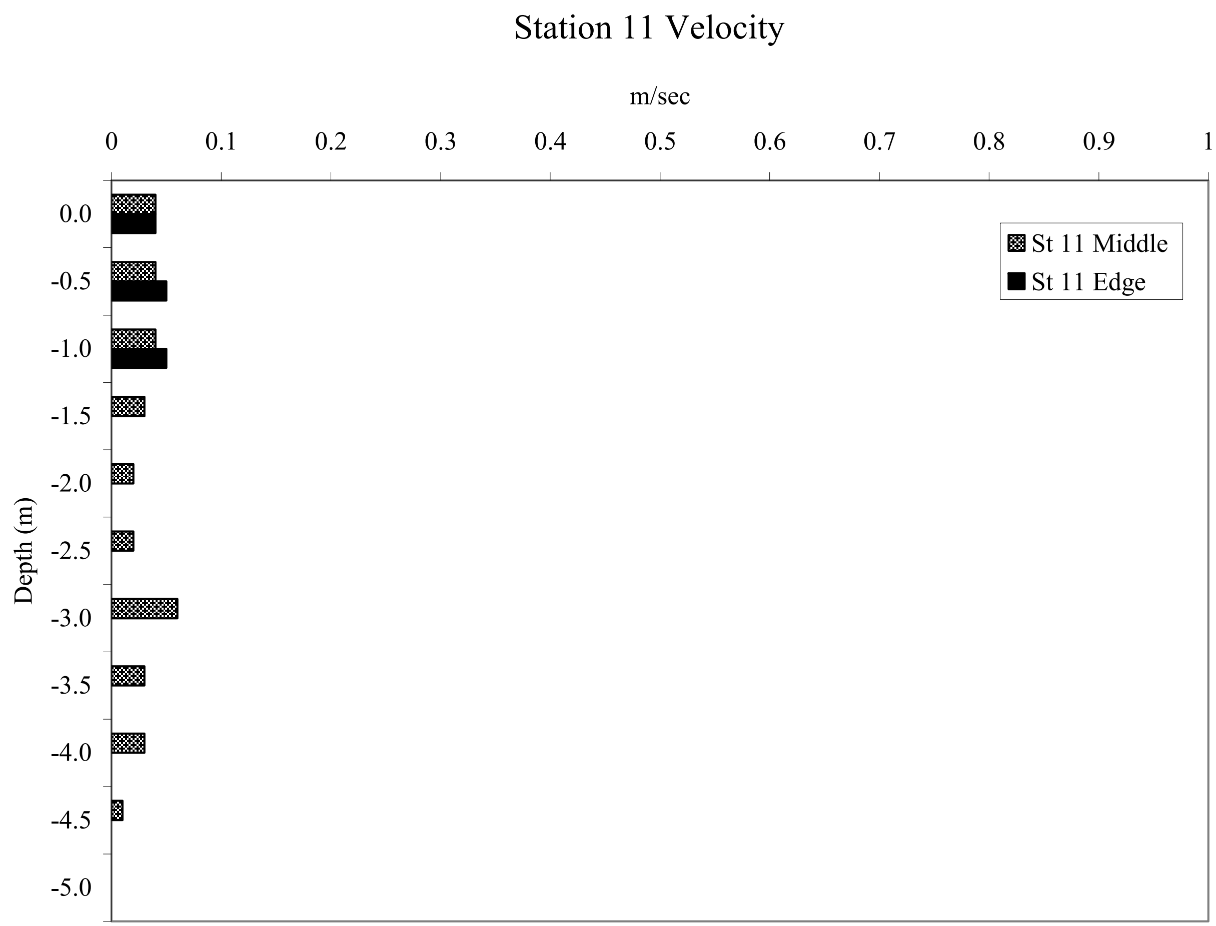


Figure 48 Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

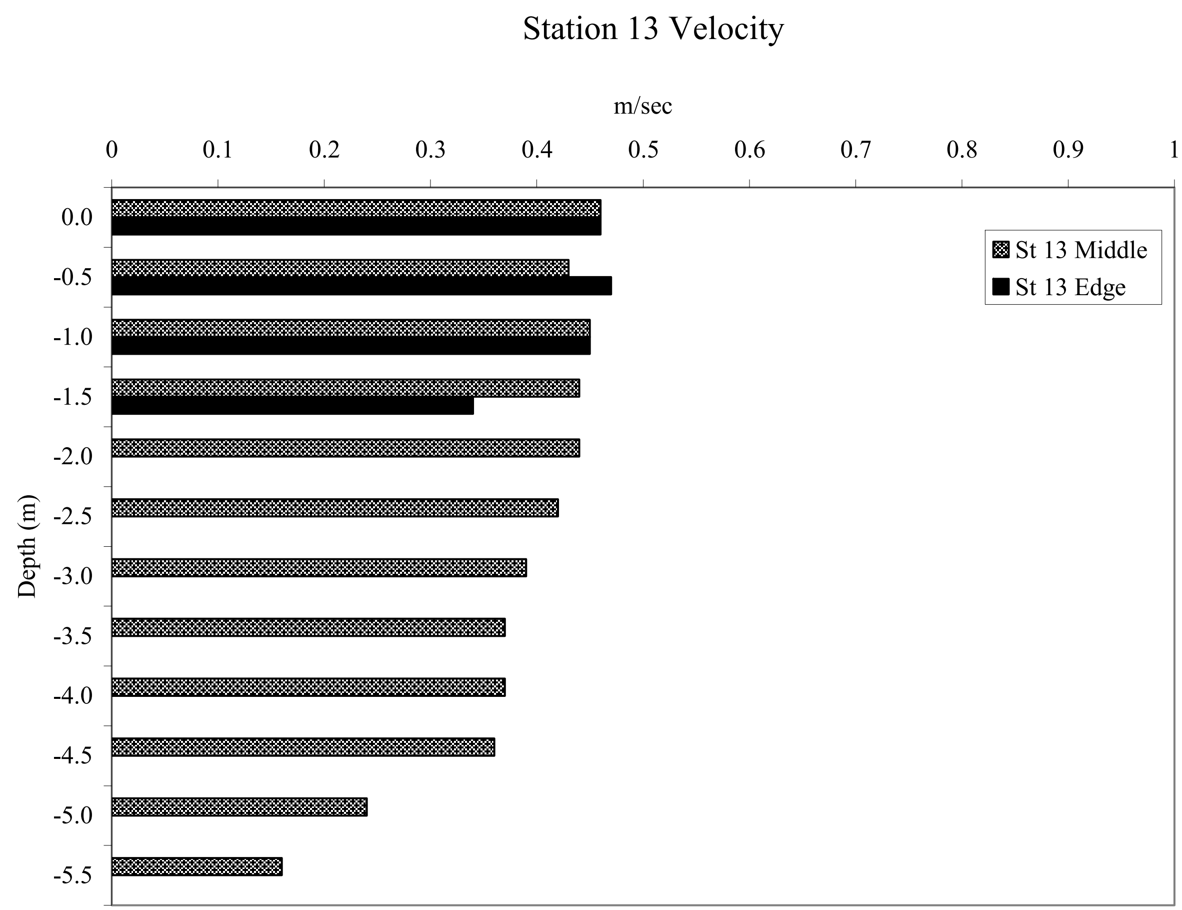


Figure 49: Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

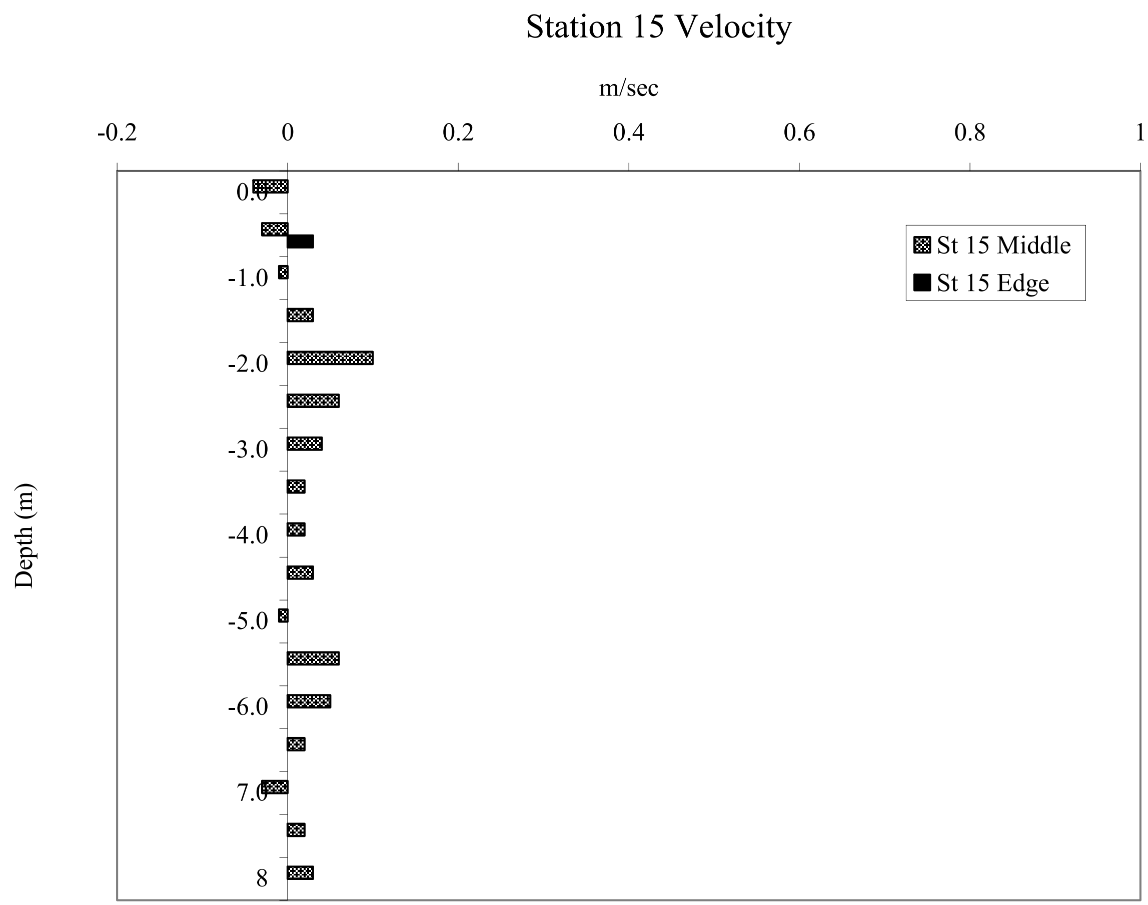


Figure 50: Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

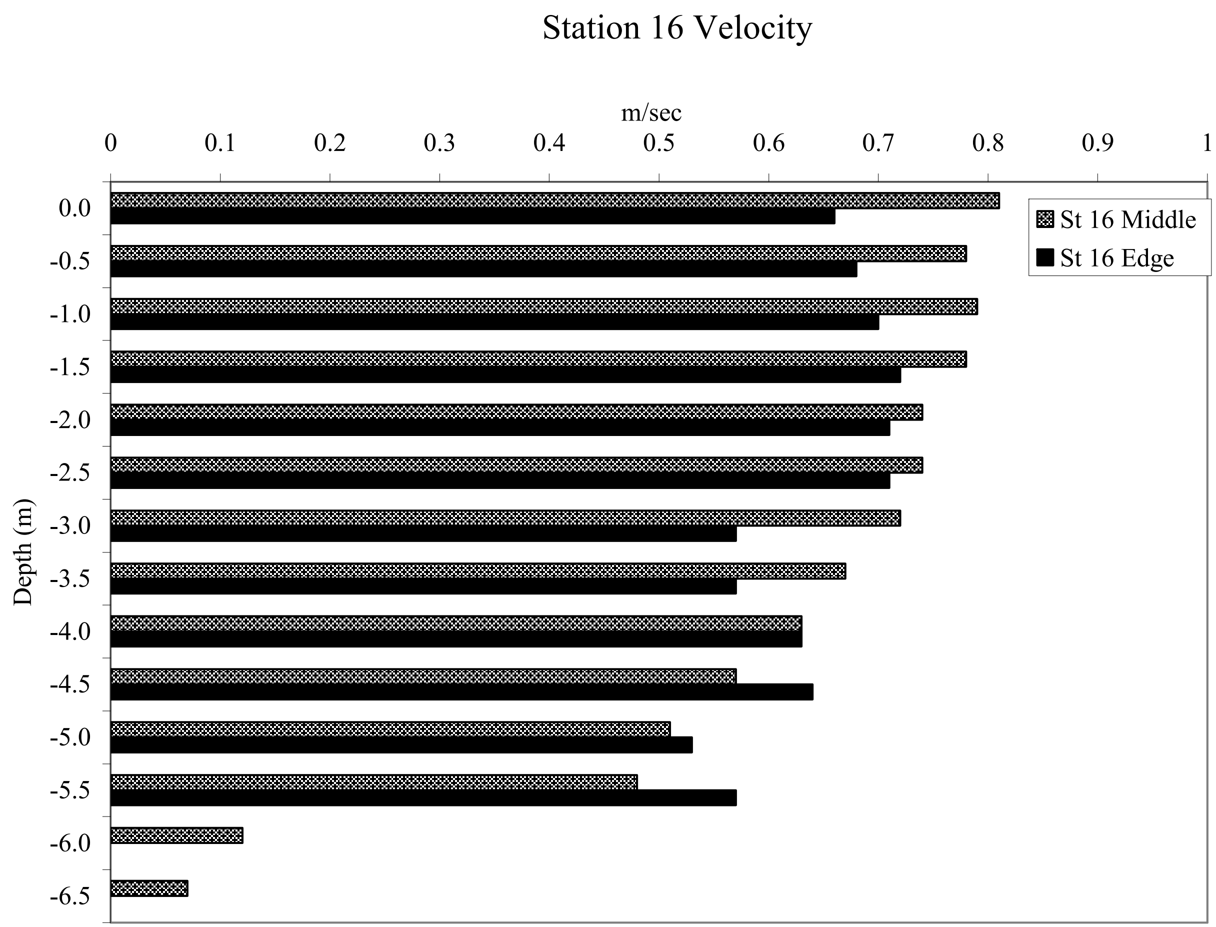


Figure 51: Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

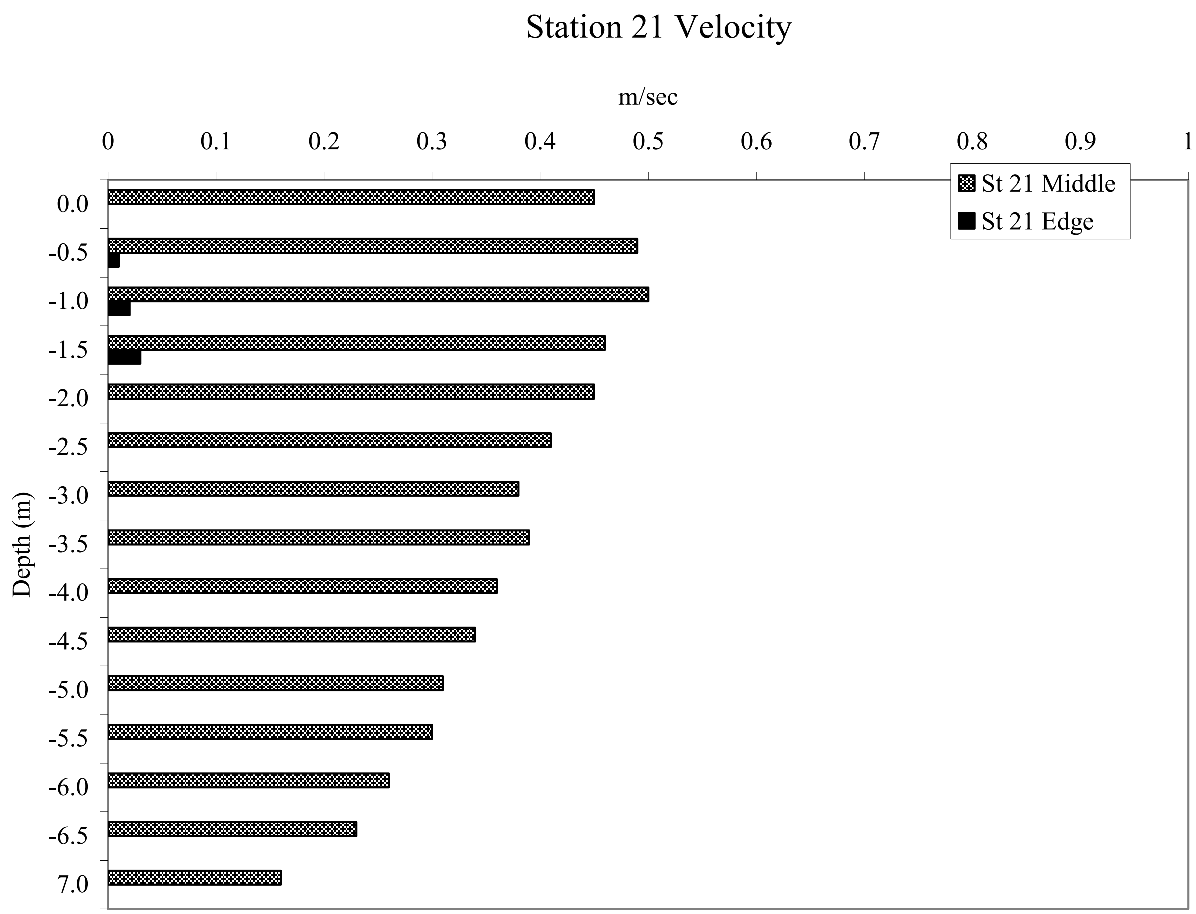


Figure 52: Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

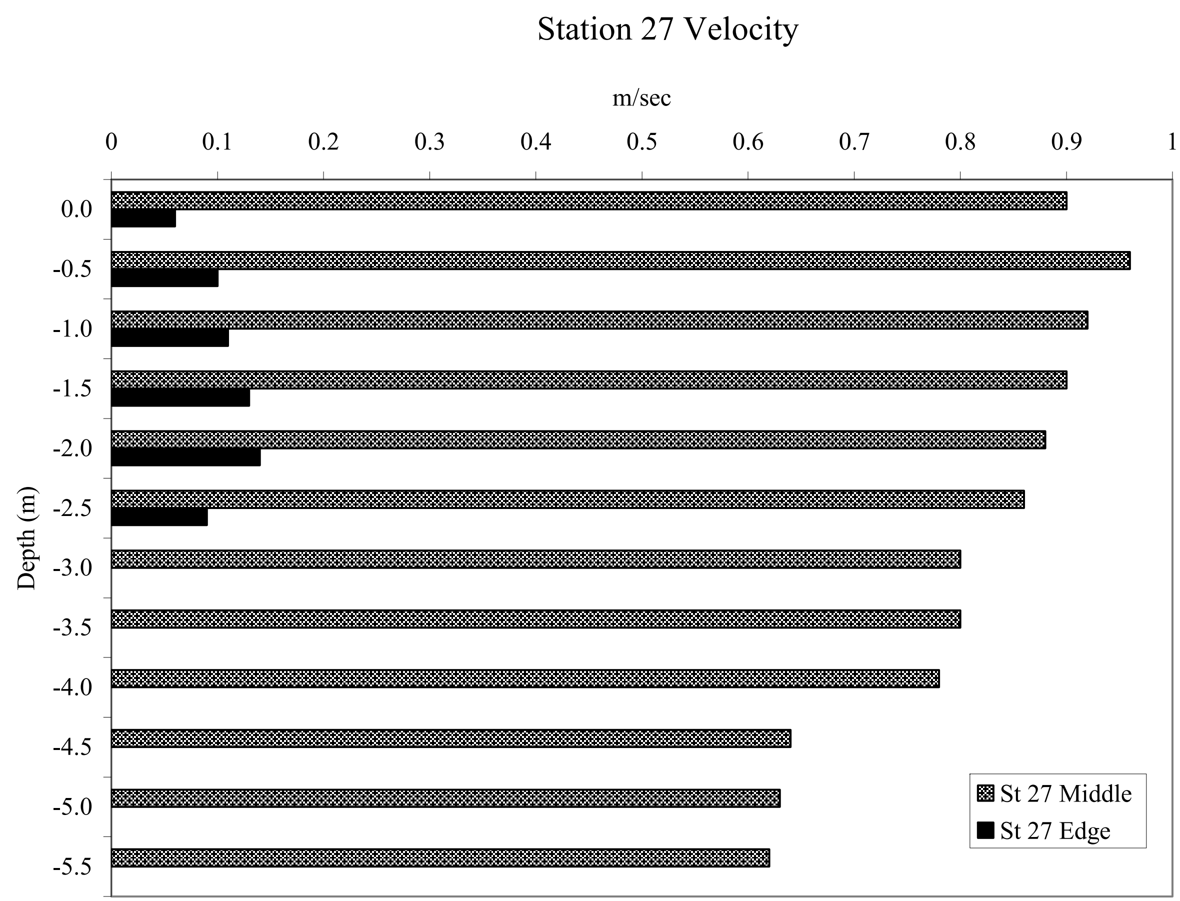


Figure 53: Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

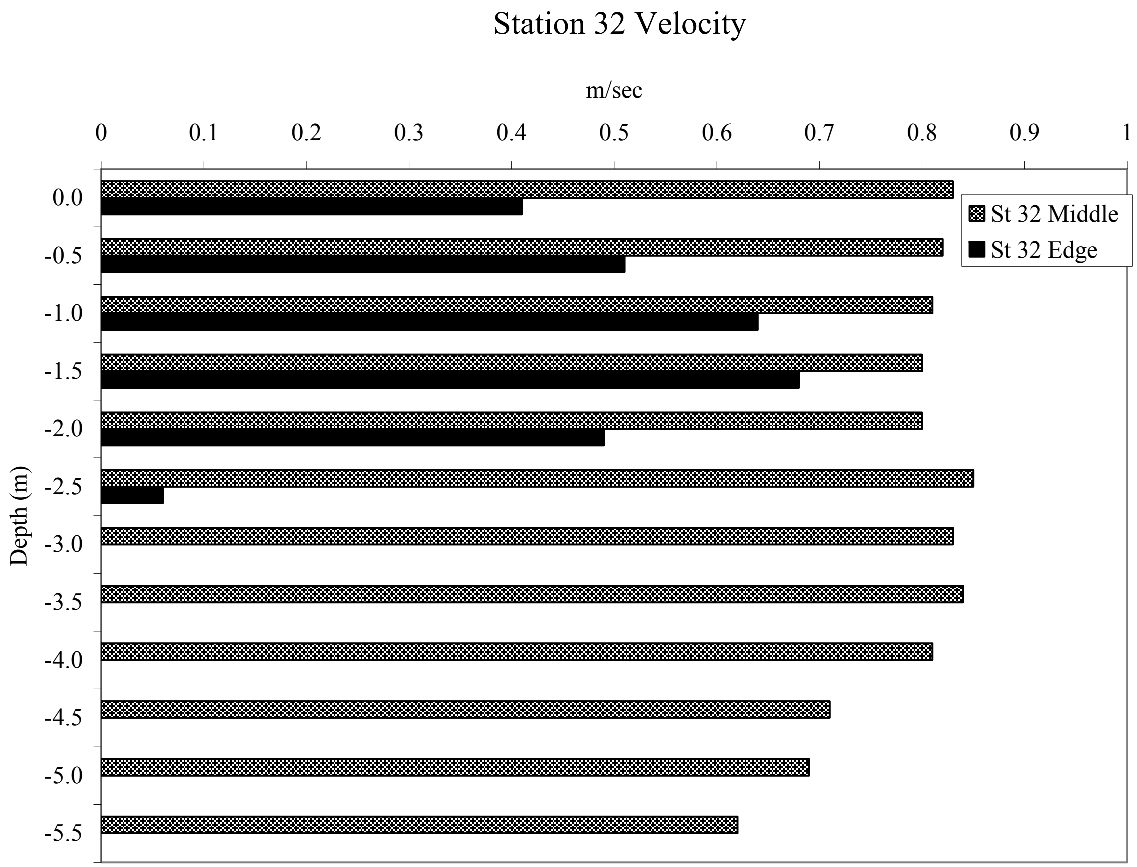


Figure 54: Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

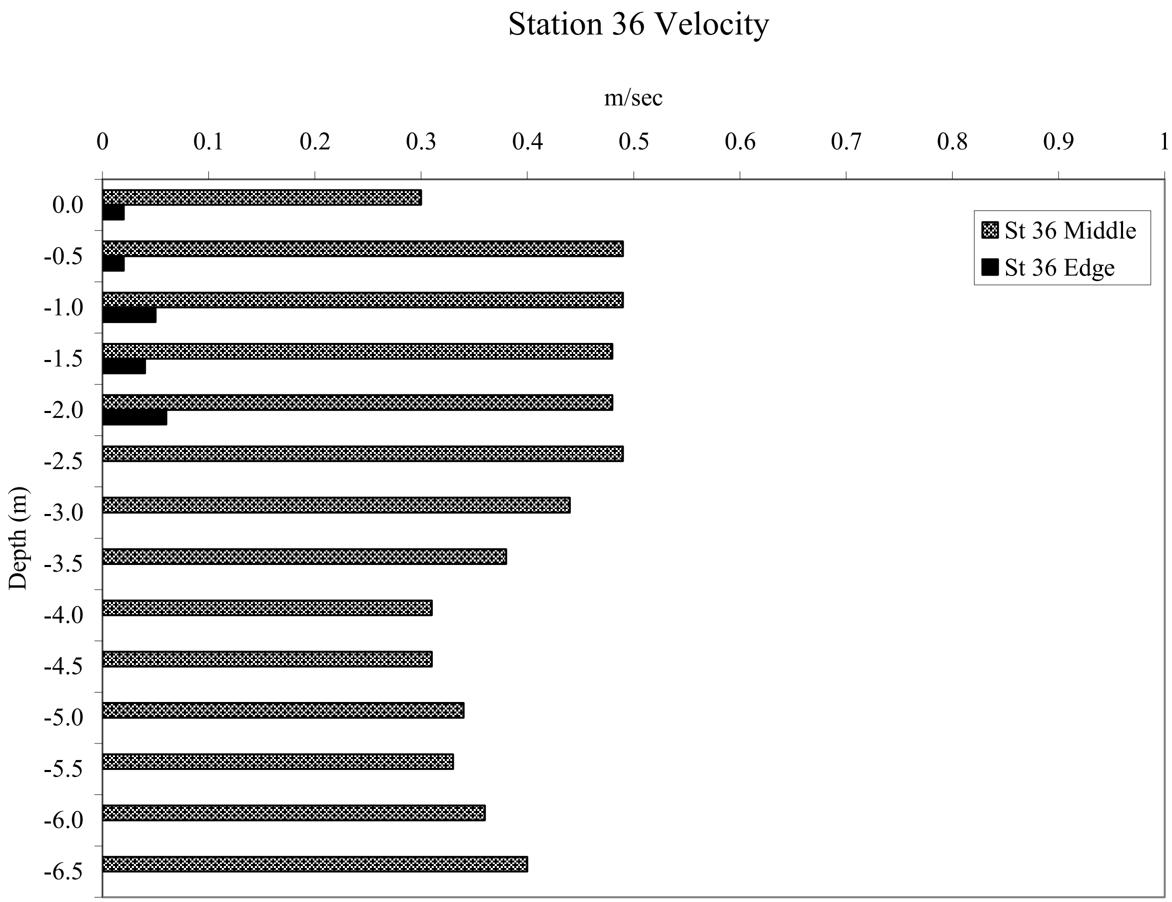


Figure 55: Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

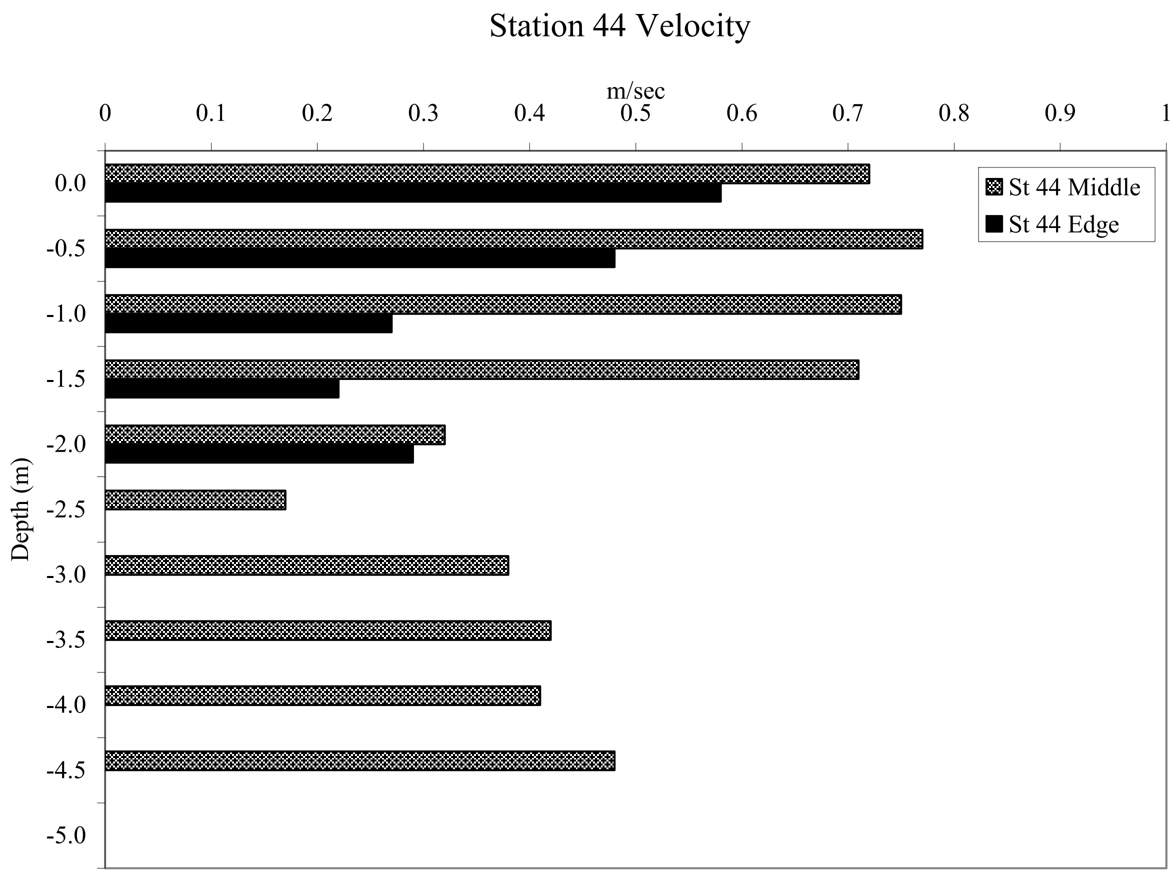


Figure 56: Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide.

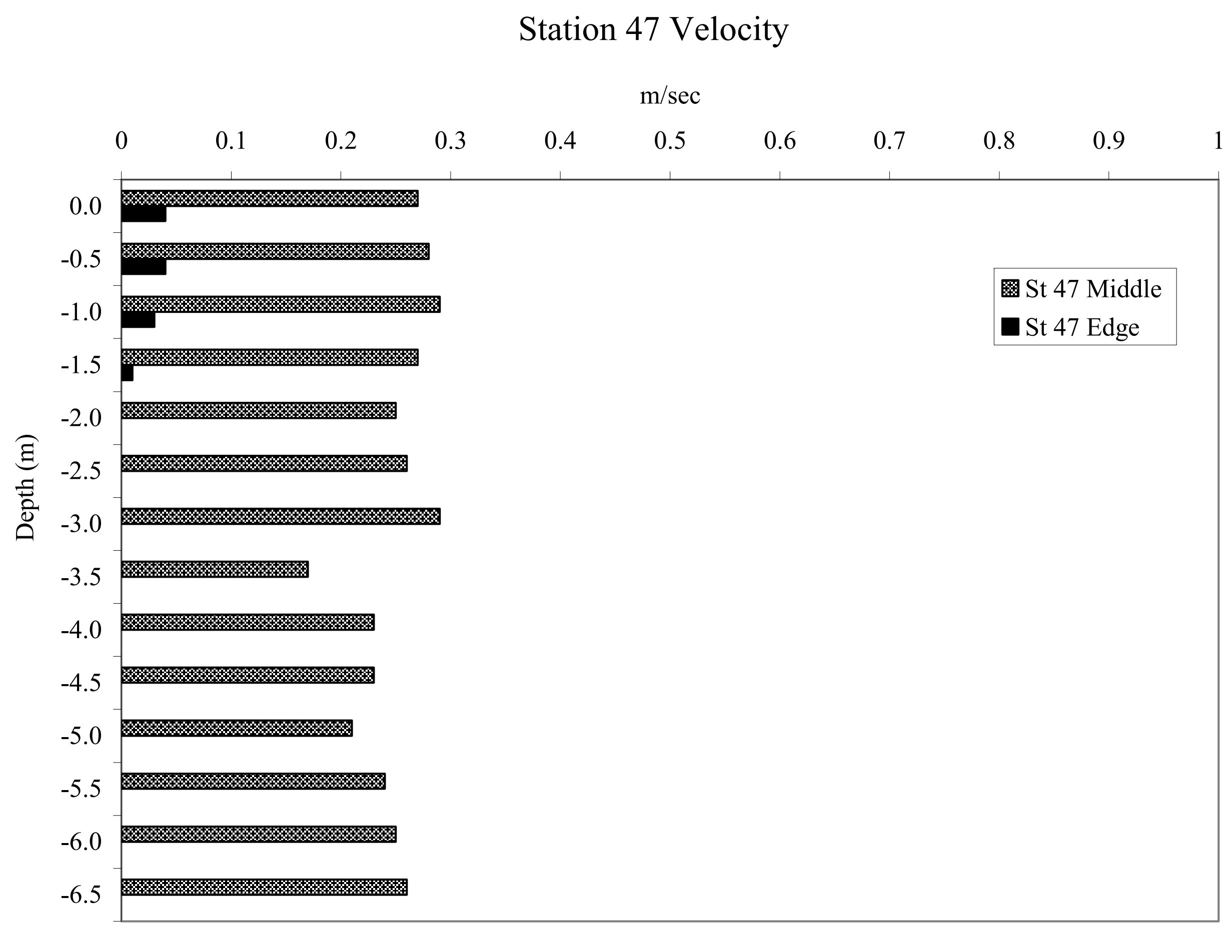


Figure 57: Vertical profiles were made by sampling at 50 m intervals starting from the water surface to the bottom or to a maximum of 8 m depth. Measurements were taken both at the approximate center of the channel and at the channel's edge in the vicinity of specified erosion stations. All measurements were taken during an ebbing spring tide

Figure 58: Photo of large masses of marsh slumping. This photo was taken in the area between stations 16 and station 18.

Figure 59: Photo of *Juncus* dominated marsh on exposed clay/sand shelf near stations 27 - 30.

Figure 60 : Photo of undercut bank at low tide near the projected marina site.