TRANSGRESSION OF DELAWARE'S FRINGING TIDAL SALT MARSHES: SURFICIAL MORPHOLOGY, SUBSURFACE STRATIGRAPHY, VERTICAL ACCRETION RATES, AND GEOMETRY OF ADJACENT AND ANTECEDENT SURFACES

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Wendy L. Carey

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by

Wendy L. Carey

Approved: Carolyn A. Thoroughgood
Carolyn A. Thoroughgood, Ph.D.
Dean of the College of Marine Studies

Approved:
John C. Cavanaugh, Ph.D.
Interim Associate Provost for Graduate Studies

I certify that I have read this dissertation and that in my opinion it meets the academic and professional standard required by the University as a dissertation for the degree of Doctor of Philosophy.

Signed: John C. Kraft

John C. Kraft, Ph.D.
Professor in charge of dissertation

I certify that I have read this dissertation and that in my opinion it meets the academic and professional standard required by the University as a dissertation for the degree of Doctor of Philosophy.

Signed:

James E. Pizzuto, Ph.D.
Professor in charge of dissertation

I certify that I have read this dissertation and that in my opinion it meets the academic and professional standard required by the University as a dissertation for the degree of Doctor of Philosophy.

Signed: David

David E. Krantz, Ph.D.
Member of dissertation committee

I certify that I have read this dissertation and that in my opinion it meets the academic and professional standard required by the University as a dissertation for the degree of Doctor of Philosophy.

Signed:

John L. Gallagher, Ph.D.
Member of dissertation committee

I certify that I have read this dissertation and that in my opinion it meets the academic and professional standard required by the University as a dissertation for the degree of Doctor of Philosophy.

Signed: Stephen P. Leatherman

Stephen P. Leatherman, Ph.D.
Member of dissertation committee

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ABSTRACT

Fringing tidal salt marshes in coastal Delaware provide ideal sites to evaluate the relationship between geometry of adjacent and antecedent surfaces and wetland development and evolution. Vegetative zonation patterns, surface elevation, subsurface stratigraphy, and depth to the pre-transgressive interface were studied along 69 survey transects extending from adjacent uplands, through transitional shrub (*Iva frutescens/Baccharis halimifolia*) and high marsh (*Spartina patens/Distichlis spicata*) environments, into low marsh (*Spartina alterniflora*) areas. Over 400 cores illustrate how Delaware's salt marshes have transgressed over coastal plain deposits during the past several hundred years. Five microfacies are distinguished in fringing salt marsh deposits based on sedimentologic characteristics (grain size, texture, color) and plant species identified in the subsurface based on rhizomatal characteristics of vegetative remains: pre-transgressive sand facies, basal soil facies, high-marsh transition facies, high-marsh facies, and low-marsh facies. Patterns of changing environments at the wetlands/uplands boundary over the past 100-200 years were documented; stratigraphic relationships between these facies are overwhelmingly transgressive, with wetlands developing over surfaces which were previously upland, and low marsh environments transgressing over high marsh environments.

Survey and core data indicate that surface vegetative zonation patterns in Delaware's fringing tidal salt marshes are related to antecedent geometry and depth to the pre-transgressive sand surface. High marsh environments, characterized by *S. patens* and *D. spicata*, are restricted to areas where both antecedent topography and adjacent upland surface have a gentle slope, and where depth to pre-transgressive surface is less than 40 cm. In contrast, low marsh *S. alterniflora* environments are generally found where antecedent and adjacent slopes are steep, and where thickness of marsh sequence exceeds 40 cm.

Vertical accretion rates calculated in fringing *S. alterniflora* marshes using ^{210}Pb, ^{137}Cs, and oak:ragweed pollen ratios range from 0.12 to 0.52 cm/yr. Vertical accretion rates determined by the three geochronologic methods demonstrate that different rates may characterize any site depending on the time interval considered. Short-term (30 year) ^{137}Cs rates are higher than the long-term (250 year) pollen rates, yet are consistently lower than the ^{210}Pb (100 year) rates. Although ^{210}Pb vertical accretion rates suggest that Delaware's fringing salt marshes are keeping pace with the rate of sea-level rise (0.31 cm/yr), ^{137}Cs rates indicate that fringing wetlands may be in the process of becoming submerged; i.e., they are not keeping pace with the rate of sea-level rise. Anthropogenic impacts such as ditching, dredging and inlet stabilization resulting in water-level alterations may exert significant influence on salt marsh vegetative environmental transitions and associated vertical accretion rates. Subsurface changes in wetland subenvironments and associated vertical accretion rates should be used only as indicators or evidence of variation in local relative water level changes.

Although vertical accretion and landward migration of low marsh *S. alterniflora* vegetative zones are confirmed by the observed transgressive overlap relationships of low marsh over high marsh zones, new high marshes are not forming in many areas. Continued development and formation of fringing high marshes in Delaware is affected by steepness of the adjacent upland slope; high marshes may be progressively replaced by low marsh environments in areas where adjacent upland slopes are too steep to provide adequate accommodation space relative to tide range and sea-level rise.

The future areal extent of fringing tidal salt marshes is related to rates of transgression of wetland environments onto adjacent upland surfaces as well as erosion of marsh shorelines. Wetland transgression rates in Indian River and Rehoboth Bays determined from metric map data can be attributed to specific and varying combinations of antecedent geometry and human impacts on local relative tidal range. Average wetland transgression rates measured from metric maps range from 0.25 m/yr (steep slope) to 1.85 m/yr (shallow slope), while average erosion rates range from 0.2 to 5.0 m/yr. Wetland transgression rates calculated from cross-section data, known slopes, and estimates of sea-level rise are much lower than measured rates of transgression, ranging from 0.08 m/yr (steep slope) to 0.44 m/yr (shallow slope). This suggests that human impacts on local relative sea-level are significant, and that the overall areal extent of many Delaware fringing salt marshes is decreasing.

Chapter 1

INTRODUCTION

Purpose and Scope

Much attention has been focused recently on the potential impacts of rising sea level on the fate of coastal and estuarine wetland environments. Direct impacts of changing sea level, increased tidal inundation, and storm surge effects on coastal salt marshes have the potential for significant consequences to the longevity of these wetland environments. Previous studies have demonstrated that, over the short term, the areal extent of Delaware's coastal wetlands is presently being reduced due to both natural processes, such as shoreline erosion (Maurmeyer, 1978; Swisher, 1982) and human intervention (Hardisky and Klemas, 1983). Longer-term impacts of rising sea level on wetlands environments are often combined with analyses of vertical accretion rates in marshes, and the long-term scenario of high rates of relative sea level rise outpacing the short-term accretionary limits of the marshes (Stevenson *et al.*, 1986; Kearney and Stevenson, 1991; Kraft *et al.*, 1992). In many cases, it has been assumed that landward growth of marsh environments will continue as long as rates of vertical accretion (sediment accumulation) on wetlands are greater than or equal to that of sea level rise. In fact, rates of landward migration, or lateral expansion, of marsh environments over adjacent uplands are relatively unknown. Although many studies have examined marsh response to rising sea level (Redfield and Rubin, 1962; DeLaune *et al.*, 1983; Orson *et al.*, 1985), most have concentrated on marsh surface accretion rates (DeLaune *et al.*, 1983; Stevenson *et al.*, 1986), the pervasive high rates of coastal erosion, and consequent rapid destruction of the coastal marsh lithosome (Phillips, 1986). The mechanisms and rates by which coastal wetland environments transgress adjacent uplands has not been described and characterized.

It is the intent of this study to develop methods which can be used to predict the future extent of fringing tidal wetlands in Delaware through a detailed examination of the wetlands/uplands boundary and the morphology of the upland surface over which the marsh is encroaching. Research objectives include: 1) description and characterization of morphology, slope, and surface and subsurface environments at the wetlands/uplands boundary; 2) chronologic evaluation of surface and subsurface wetland vegetative distribution and environmental changes; 3) utilization of dating techniques (^{210}Pb, ^{137}Cs, pollen analyses) to establish timelines as a predictive tool for determining rates of upland surface inundation over the last 50-100 years; and 4) evaluation of the likelihood of continued landward migration of coastal wetlands over the adjacent upland surface, linking antecedent topography (geometry and elevation) and environments to the process of transgression.

General Discussion

The Holocene marine transgression, which has resulted in landward and upward migration of many coastal environments along the entire Delaware Bay and Atlantic coastal zone, has had a profound effect on coastal wetlands, resulting in formation of new wetlands as well as deterioration and destruction of established wetlands. These processes have occurred throughout the past 14,000 years, and are also perceptible on human time scales. Extant tidal marshes of coastal Delaware began developing approximately 3,000-3,500 years ago (Kraft and John, 1976; Kraft and Chrzastowski, 1985; Kraft and Belknap, 1986), and it has generally been assumed that the continued existence of coastal salt marshes in the stratigraphic record demonstrates that sediment accumulation within the Delaware Bay and Inland Bays estuaries had kept pace with local relative sea level rise. However, the question of "continued existence" of coastal salt marshes in Delaware and other areas of the east coast of the United States has recently been re-evaluated. For example, Fletcher *et al.* (1993) describe a transgressive event in Delaware's tidal marshes that was probably caused by an acceleration in the rate of regional sea-level rise in Delaware Bay at 1,800 years ago. Examination of cores obtained in a Delaware Bay salt marsh reveal a stratigraphic sequence where marsh facies are absent, and estuarine/marine sediments are found to be conformably overlying terrestrial units (Fletcher *et al.*, 1993). Evidence of transgressive drowning of salt marshes has similarly been suggested in New Jersey (Meyerson, 1972) and Connecticut (van de Plassche, 1991; Varekamp *et al.*, 1992).

These previous studies demonstrate that sea-level movement has continuously exerted a dominant control on the development and longevity of salt marshes (Fletcher *et al.*, 1992, 1993), and that intermittent fluctuations in the rate of sea-level rise may have had a significant impact on tidal wetlands. However, data acquired through subsurface coring (stratigraphic evidence), and observation of surface environments as well as the areal extent of coastal marshes along Delaware shorelines have proven that, for the most part, tidal wetlands have been able to maintain vertical accretion rates comparable to relative rates of rising sea level (Kraft and John, 1976; Kraft and Belknap, 1986; Fletcher *et al.*, 1990).

Previous Work: Geology of Delaware Tidal Wetlands

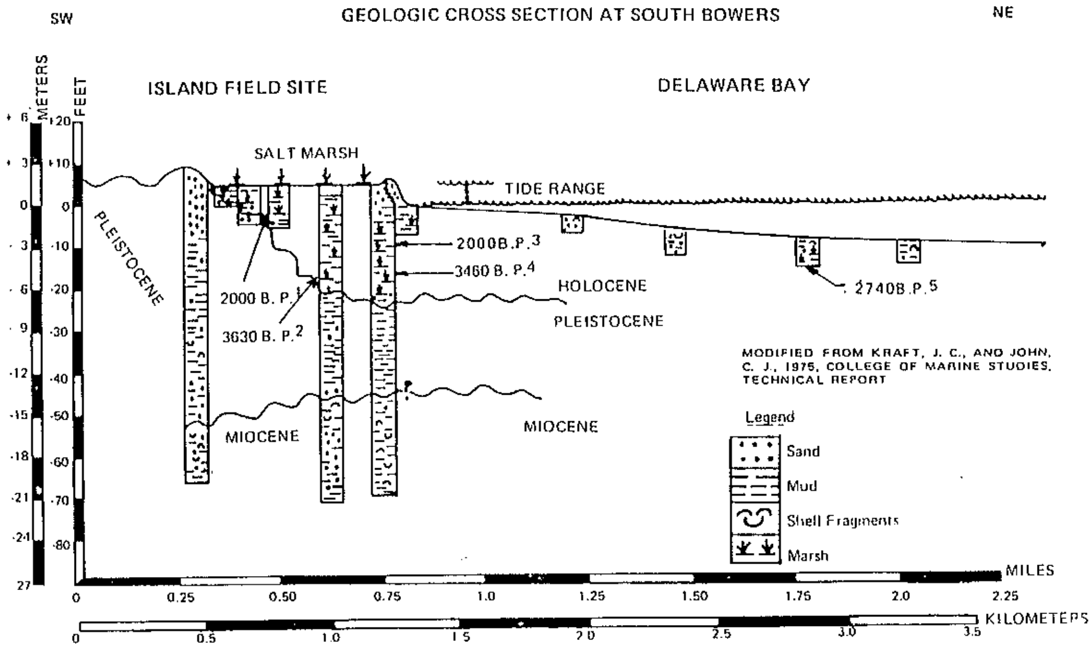
Previous geologic research on Delaware's coastal tidal marshes has provided data on general geomorphology and geology of Holocene marsh sediments and stratigraphy (Kraft, 1971a, 1971b; Elliot, 1972, 1973; Allen, 1974, 1977; Richter, 1974; Weil, 1976; Belknap and Kraft, 1977, 1985; Khalequzzaman, 1989; Whallon, 1989; Yi and Kraft, 1989; Fletcher *et al.*, 1990, 1991; Stedman, 1990; Yi *et al.*, 1991; Yi, 1992; Pizzuto and Rogers, 1992; John and Pizzuto, 1995). Detailed investigations of Holocene stratigraphic sequences and paleo-environments, especially marsh sub-environments, have been conducted and reported by Allen (1974, 1977, 1978) and Yi (1992). The present study expands on previous work and provides a detailed characterization and interpretation of surficial morphology and subsurface stratigraphic relationships over the last 150 to 200 years at the wetlands/uplands boundary area of Delaware's fringing tidal salt marshes.

Much of the past research demonstrates that wetlands and sea level have been in approximate equilibrium over the past 3,000-5,000 years, with the exception of possible intermittent rapid fluctuations of sea-level rise and consequent transgression of tidal wetlands environments in Delaware Bay at 1,800 years before present (Fletcher *et al.*, 1993). However, when considering more recent time scales, the continued existence of tidal wetlands in Delaware is affected by local factors that are not necessarily related to sea level. These include, but are not limited to, variations in sediment supply, tidal range, tidal channel processes, compaction, supply of freshwater from upland sources, geometry of adjacent upland surfaces, and, ultimately, extent of tidal inundation (hydroperiod) and sediment accumulation rates. Sedimentation rates and vertical accretion rates in salt marshes in Delaware (Khalequzzaman, 1989; Whallon, 1989; Kraft *et al.*, 1989, 1992; Fletcher *et al.*, 1993) are highly variable, with rates of marsh accretion ranging from 0.20 cm/yr to 0.78 cm/yr. Conceptual models have been developed concerning cycles of marsh development, vertical accretion rates, anthropogenic effects, and the ability of coastal wetlands to either: 1) keep pace with relative sea-level rise (accretion rate = rate of sea-level rise); 2) outpace relative sea-level rise (accretion > rate of sea-level rise); or 3) become inundated (accretion rate < rate of sea-level rise) (Orson *et al.*, 1985). Such models demonstrate that while studies of vertical accretion rates are usually concentrated in the regularly flooded intertidal, or low marsh portions of the salt marsh, attention needs to be turned to lateral accretion;

i.e., to the wetland/upland boundary area, where the leading edge of the marine transgression, and the ultimate fate of the coastal salt marsh lies. Because many of Delaware's coastal wetlands are being eroded at the bayward interface, transgression onto the upland area must take place at comparable rates in order to maintain the areal extent of tidal wetlands.

Although the characteristics and rates of landward migration of present and future marsh systems are generally unknown, historical data and the geologic record provide information on the nature of wetlands transgression over the last several thousand years. Existing stratigraphic data (Kraft and John, 1976, 1979; Kraft *et al.*, 1987, 1989; Chrzastowski, 1986) clearly demonstrate that marshes have migrated in a landward direction during the Holocene period. It is usually assumed that marsh environments have migrated landward as a geometric unit with concurrent retreat of adjacent coastal environments and lithosomes. Previous geologic studies and associated cross-sectional interpretations commonly depict Holocene marshes, lagoons, and beaches overlying pre-Holocene or Pleistocene sediments, as seen in Figure 1.1 (Kraft and John, 1976, 1979). Interpretations of core data (Figure 1.2) also clearly demonstrate that Delaware Bay and Inland Bay marsh environments were over 3 times wider than present circa 3,000 years ago (Kraft and John, 1976; 1979; Chrzastowski, 1986; Kraft and Belknap, 1986; Kraft *et al.*, 1992).

Many interpretations of the pre-Holocene surface at the leading edge of the Holocene transgression are generally schematic; i.e., the wetlands/uplands boundary area is often depicted as a dramatically steepened adjacent Pleistocene (pre-Holocene)



MODIFIED FROM KRAFT, J. C., AND JOHN, C. J., 1975, COLLEGE OF MARINE STUDIES, TECHNICAL REPORT

Radiocarbon dates are shown in Calendar Time Masca Correction: Radiocarbon years, 5568½ life = [1]1952 ± 45, [2]3314 ± 63, [3]1950 ± 55, [4]2999 ± 59, [5]2550 ± 100.

Figure 1.1 Cross-sectional interpretation of Holocene sediments and their general relationship to pre-Holocene or Pleistocene sediments (after Kraft and John, 1975).

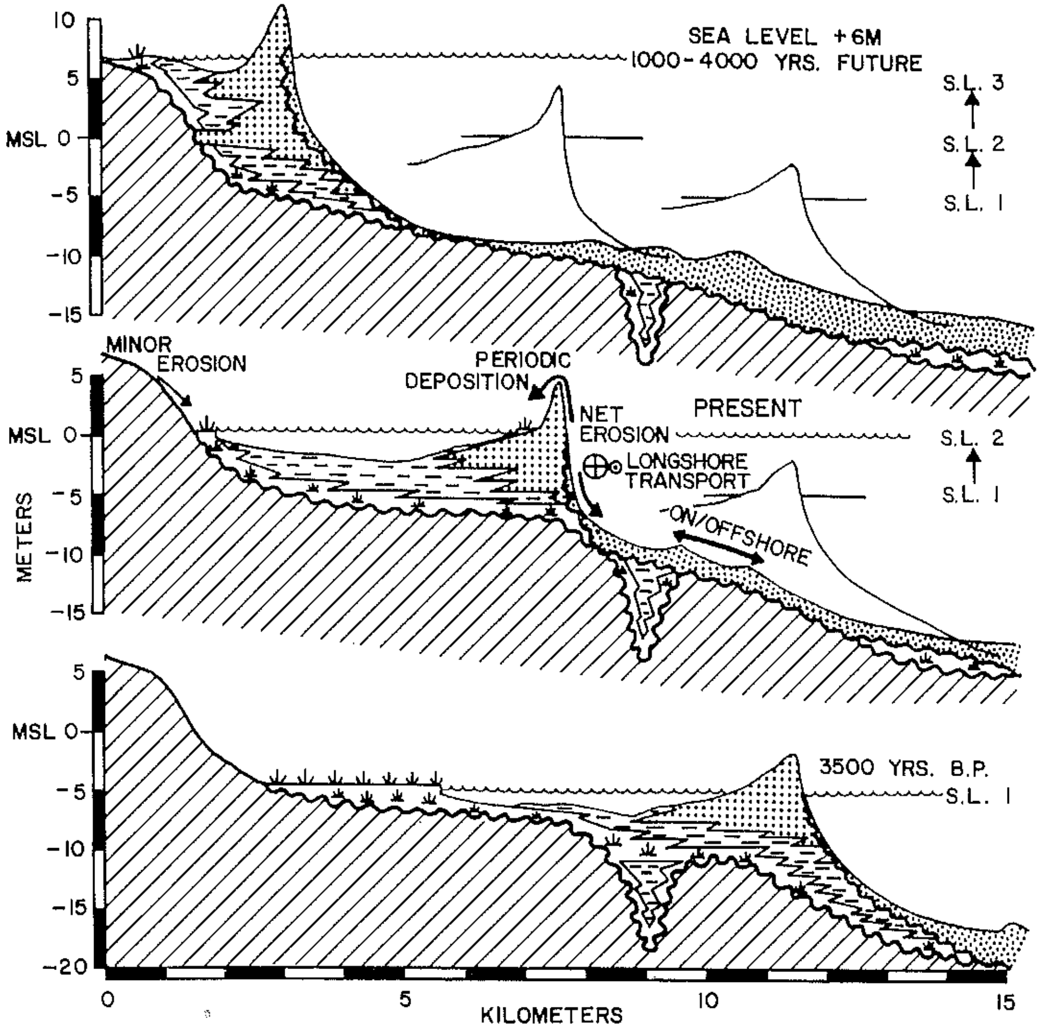


Figure 1.2 Schematic diagram of geologic profiles across Rehoboth Bay and adjacent barrier. The middle profile depicts the present configuration of pre-Holocene surface and various paralic lithosomes, including marsh, lagoon, and barrier. The bottom-most diagram shows configuration of Delaware Atlantic coastal area approximately 3,500 years before present; note wider lateral extent of fringing salt marsh wetlands. Upper diagram depicts a much reduced Rehoboth Bay and fringing marsh area after a projected sea level rise of 5 to 6 meters (from Kraft *et al.*, 1992).

surface. A preliminary assessment of the geometrical complexities of wetlands transgression and preservation vs. erosion and sea-level rise has been made by Phillips (1986). This study employs a more detailed examination of this underlying, laterally adjacent antecedent topography to determine if there is a relationship between a possible narrowing of the coastal marsh zone and the geometry of adjacent uplands. It is then possible to demonstrate if the areal extent of tidal wetlands and associated migration rates may decline dramatically due to this underlying topographic control.

An initial postulate is that the ability of a coastal salt marsh to continue transgression in a landward direction varies between two extremes: 1) a steep pre-Holocene slope over which wetlands cannot continue to transgress, or where the rate of transgression is extremely slow; and 2) a gentle, shallow pre-Holocene slope over which wetlands can continuously migrate landward and upward at a rate equivalent to or close to the rate of wetland disappearance (erosion) at the wetland/water interface. Between these two extremes, continued transgression of Delaware's coastal salt marshes may be influenced by other critical factors such as vertical accretion rates, relative sea level rise rates, local subsidence, sediment compaction, and location of the coastal wetland relative to other wetland environments. For example, tidal wetlands may be able to transgress over palustrine and riverine wetlands (Orson *et al.*, 1990; Whallon, 1989; Pizzuto and Rogers, 1992; John, in progress), but not over steeper slopes of the pre-Holocene material.

The ability or likelihood of a coastal wetland to continue landward migration is of fundamental importance because the wetland/upland boundary conditions may ultimately control and govern the evolution and continued survival of coastal wetlands. While a gently sloping pre-Holocene surface will likely prolong the lifetime of the marsh as sea level rises, a steeply sloping surface may shorten it significantly. This study investigates the hypothesis that the relationship between the rate of sea-level rise and the slope of the upland being transgressed may be a significant environmental parameter controlling wetland surface vegetative zonation as well as longevity of fringing tidal wetlands in coastal Delaware. Other environmental parameters such as flooding regime/hydroperiod (extent and duration of tidal inundation), sediment supply (marsh accretion rate), salinity, and sediment chemistry are additional primary controlling factors. Even if sediment accumulation in the frequently flooded areas of a wetland keep pace with relative sea level rise, it may be the nature of the wetland/upland boundary that ultimately controls the fate of wetlands during a marine transgression.

Objectives

The purpose of this investigation is to determine the relationship between subsurface antecedent geometry, wetland surface and subsurface vegetative zones or lithosomes, and the rate and processes of landward migration of coastal wetland environments. The principle objectives of this study are: 1) to describe and characterize the nature of the wetlands/uplands boundary at the leading edge of the Holocene transgression in Delaware; 2) to determine relationships between coastal wetlands and the surfaces they are transgressing; 3) to document the evolution and recent history of fringing coastal wetlands; and 4) to assess the potential physical impacts of relative sea-level rise on the areal extent of existing coastal wetlands in Delaware.

This study is an investigation and characterization of surficial morphology and subsurface stratigraphy of sub-environments in Holocene marsh deposits at the wetlands/uplands boundary along fringing tidal salt marshes of Delaware's Inland Bays and along the western shoreline of Delaware Bay. This study provides additional data on the fate of tidal wetlands along the Delaware coast, and utilizes methods which can be used to predict the future extent of fringing tidal wetlands in Delaware. Temporal and spatial changes in surficial morphology and zonation patterns of halophytic vegetation reflect the environmental and physical processes and factors affecting fringing salt marshes. Short-term changes (50-200 years) in these vegetated salt marsh sub-environments are reflected in subsurface sediments. Data on the stratigraphic relationships of marsh sub-environments reveal these changes and can provide insight into temporal and spatial distribution of extant fringing marshes in Delaware's coastal areas. The basis of this investigation is detailed analysis of sedimentary and stratigraphic data obtained from 400 closely spaced Eijelkamp cores collected from 69 survey transects along the Inland Bays and Delaware Bay shoreline. Additionally, it is essential to characterize the vegetative distribution and zonation patterns in surficial marsh environments and to recognize the environmental parameters affecting their distribution. Thus, an understanding of plant morphology (species composition, structure, and biology) of the halophytic plant community in Delaware's fringing salt marshes is also required. Major species investigated in this study include: *Iva* *frutescens* L. (marsh elder), *Baccharis halimifolia* L. (groundsel bush), *Phragmites australis* (Cav.) Trin. ex Steud. (giant reed), *Spartina patens* (Ait.) Muhl. (salt meadow hay), *Distichlis spicata* (L.) Greene (spike grass), and *Spartina alterniflora* Loisel. (smooth cordgrass). These prerequisites permit identification of marsh sub-environments in the subsurface based on macroscopic vegetative remains, thus establishing the basis for determining temporal and spatial patterns and variations in vertical and horizontal (landward) movement of fringing coastal salt marshes in Delaware.

Various geomorphic scenarios at the landward edge of coastal salt marshes along the Delaware Bay and Atlantic coast are reviewed, and the varied geomorphic and stratigraphic characteristics of the study sites are examined to determine if and how they relate to the status of the marine transgression. General goals are outlined as follows:

1) Determine the general sedimentary, vegetative, and stratigraphic characteristics of the landward edges of transgressing tidal marsh systems. A series of core transects from an upland (pre-Holocene) surface across the upland, transitional-, high-, and low-marsh environments characterizes the modern wetlands/uplands boundary.

2) Determine the relative surface elevation of wetland vegetative zonation, subsurface wetland environments, and adjacent and underlying topographic slopes. These data are used to determine if both surface and subsurface vegetative zonation patterns as well as rates of inundation of upland surfaces are related to the slope of the upland being transgressed.

3) A detailed examination and description of surface and subsurface wetlands environments, when considered with Walther's Law, are used to evaluate migration of wetland environments through space and time. A micro-stratigraphic model is constructed to document the characteristics and evolution of the wetlands/uplands boundary, seen as the leading edge of the Holocene transgression; areas are characterized according to their "likelihood of continued transgression." Stratigraphic relationships of wetlands environments, their ages, and relationships to pre-Holocene surfaces are established to determine the principal factors controlling the areal extent of existing tidal wetlands in Delaware.

4) Establish chronostratigraphic relationships and utilize various dating methods to establish time lines as a tool for estimating rates of inundation over the last 50-100 years. Chronologic data are acquired to determine rates of wetland transgression over adjacent upland areas.

5) Characterize/describe landward migration of Holocene wetlands environments as controlled by antecedent topography.

6) Results of the characterization of the wetlands/uplands boundary and associated underlying slopes make it possible to predict the future extent of tidal wetlands in Delaware. Rates of transgression and landward migration of wetlands can be compared to and combined with rates of concurrent shoreline erosion of wetlands environments. The results permit calculation and prediction of the trends in the areal extent of tidal wetlands loss in Delaware.

The overall objectives are to: describe characteristics of the wetlands/uplands boundary along Delaware's Inland Bays and western Delaware Bay shorelines, investigate the relationship between antecedent and laterally adjacent topography (pre-Holocene geometry) and the nature of the transgressive boundary over time, and develop models of these relationships to permit prediction and evaluation of the future areal and volumetric extent of Delaware's coastal wetland environments.

Summary

At present, the distribution of wetlands is dependent upon and, in some cases, at equilibrium with, existing tidal conditions, wave energy, tidal flooding frequency and duration, sedimentation rates, local climate, and sea level fluctuations. Traditional views of marshes and their development hold that developing and emerging salt marshes are sites of sediment accumulation where the rate of accretion exceeds or keeps pace with the cumulative effects of submergence and sea level rise. The coastal salt marsh is considered to be an evolving landform that goes through a cycle of emergence, continued accretion, and, depending on availability of sediment and relative sea level rise, either continued transgression, or inundation. Many previous studies have focused on the rates of sediment accretion in marshes, and the question of whether wetlands can "keep pace" vertically with relative sea level rise and possible increases in the rate of sea level rise. However, the question of continued transgression and lateral, or horizontal migration of the marsh, needs to be addressed.

The future extent of Delaware's tidal marshes is not only controlled by shoreline erosion and vertical accretion rates, but also by rates of landward migration of the wetlands system. This study has resulted in compilation of data required to determine the detailed subsurface stratigraphy and surface vegetative distribution of wetlands/uplands boundary zones. Similarly, detailed mapping of antecedent geometry and other possible factors controlling rates of lateral shifts in wetlands environments at the leading edge of the Holocene transgression has been determined. Mapping surface elevations and determining slopes of both laterally adjacent and underlying topographic surfaces permits projection of the effects of rising sea level on the future extent of wetlands in Delaware. With projected decreases in net area of Delaware's coastal wetlands over the next century, the detailed analyses conducted in this study provide information that may have wide application to projections, potentials, and limitations of new marsh development.

Chapter 2

REGIONAL SETTING AND STUDY AREA

Regional Geologic Setting

Introduction

The mid-Atlantic region of the east coast of North America is characterized by a broad continental shelf and coastal plain (Figure 2.1). Delaware Bay is a major embayment in the Atlantic Coastal Plain, located on the partially emerged northwest flank of the subsiding Baltimore Canyon trough of the geosyncline (Sheridan *et al.*, 1974). In this area, Atlantic continental shelf sediments range in thickness from approximately 12,000 m in the Baltimore Canyon trough geosyncline (Belknap and Kraft, 1977) to 2,000 meters in Delaware (Minard *et al.*, 1974). Delaware's Inland Bays (Rehoboth Bay, Indian River Bay, and Little Assawoman Bay) are located along the Atlantic coast of the Delmarva Peninsula (Figure 2.2), an emerged part of the Atlantic coastal plain.

The emergent portion of the coastal plain in Delaware is characterized by Cretaceous, Tertiary, and Quaternary sediments; surficial sediments (up to 45 meters thick) are generally characterized by a veneer of Pleistocene gravel, sand, silt, and clay of fluvial, marginal marine and marine depositional origin (Jordan, 1962, 1964, 1974;

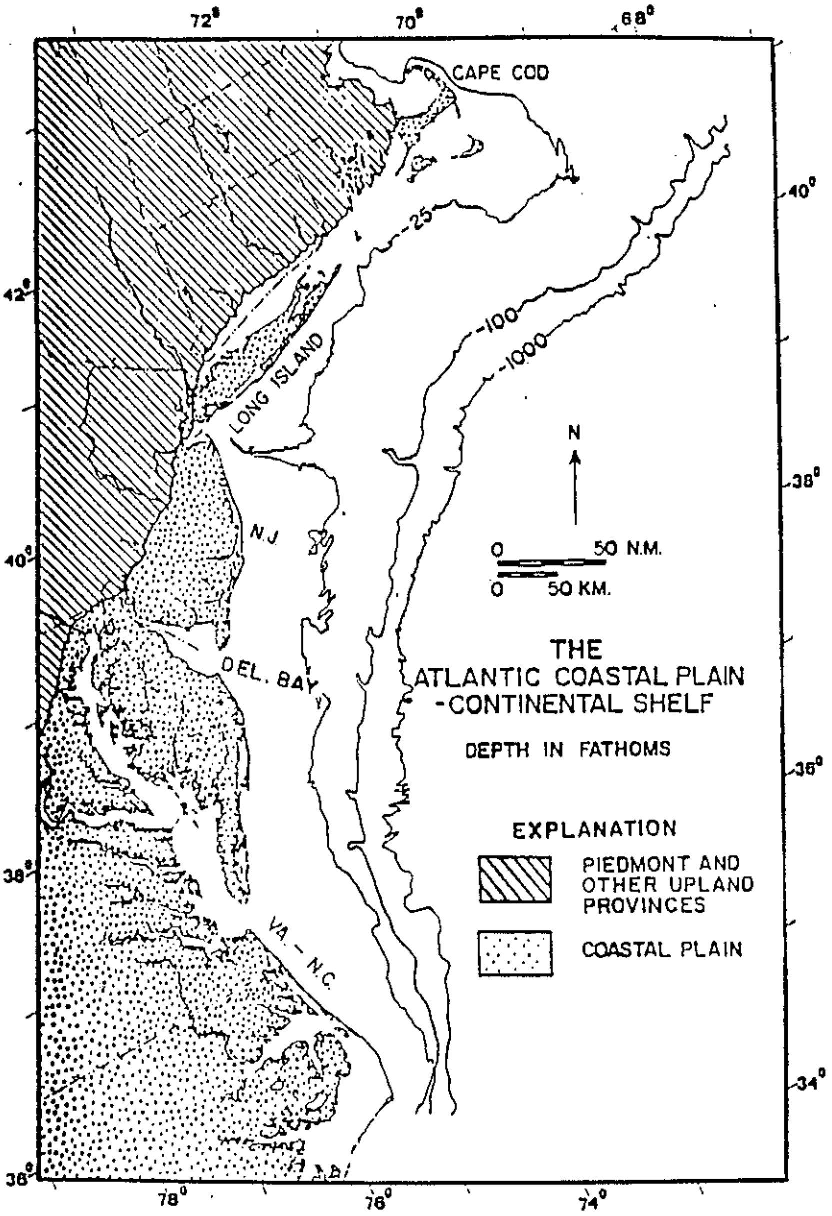


Figure 2.1 Atlantic coastal plain and continental shelf (from Allen, 1978).

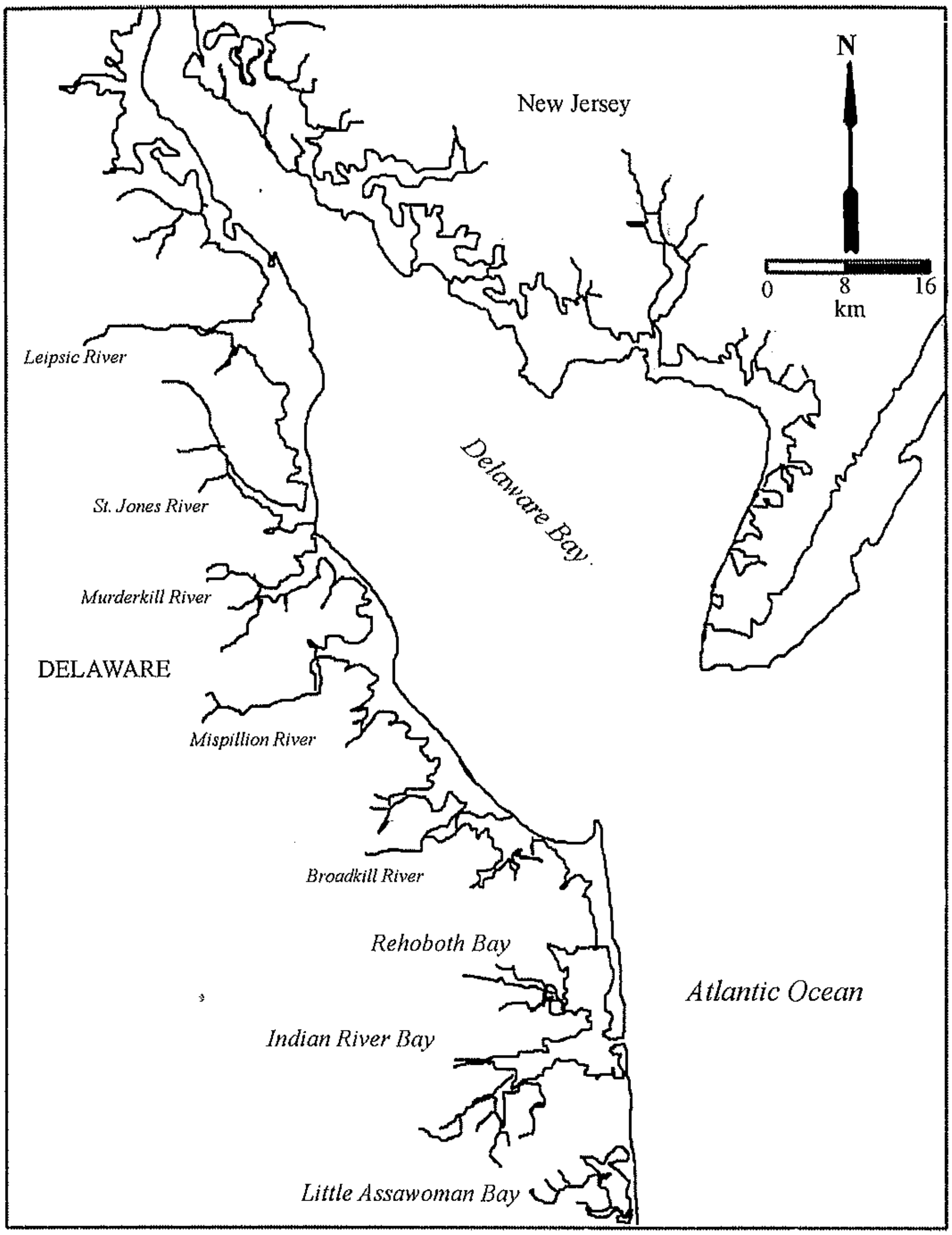


Figure 2.2 The Delaware Bay estuary and Atlantic coast (from Yi, 1992).

Maurmeyer, 1978; Chrzastowski, 1986). This coastal plain surface presently has little relief and elevations rarely exceed 30 m. Quaternary glacio-eustatic sea-level fluctuations have primarily influenced the evolution, stratigraphy, and geomorphology of the region, and have ultimately controlled the distribution of Holocene depositional environments (Chrzastowski, 1986).

Pre-transgressive Surface

Pleistocene units form a surface over which the Holocene sedimentary environments migrate and provide a major source of sediment to the Holocene during transgression (Kraft *et al.*, 1973). During the Wisconsinan glaciation, the study area was approximately 230 km south of the maximum extent of the Laurentide ice sheet (McIntyre *et al.*, 1976; Chrzastowski, 1986). During peak and waning glaciation, the ice margin extended into the headwater-drainage area of the ancestral Delaware River, and during this time the ancestral Delaware was an arterial meltwater stream (White, 1978; Chrzastowski, 1986). Lowered base level, increased stream discharge, and the underlying unconsolidated coastal plain and continental-shelf sediments allowed the ancestral Delaware River to incise a deep and narrow valley 40-60 m below present mean sea level (MSL) along its course beneath the present Delaware Bay inner shelf (Sheridan *et al.*, 1974; Weil, 1976; Belknap and Kraft, 1985; Chrzastowski, 1986). Other major Pleistocene features are the deeply incised tributary valleys to the ancestral Delaware River and Bay (Kraft *et al.*, 1976); these valleys are likely the result of multiple cycles of incision and sediment infill (Chrzastowski, 1986). The result of valley incision during Wisconsinan time is that along both the Delaware estuarine and ocean coasts, the Holocene pre-transgression surface is a series of coast-perpendicular antecedent valleys and associated paleo-interfluves (Kraft *et al.*, 1979; Chrzastowski, 1986).

Holocene fill of these antecedent valleys and burial of associated paleo-interfluves is dominated by mud (Richter, 1974; Chrzastowski, 1986). Chrzastowski (1986) explained that the surficial expression of these antecedent valleys and transgressed paleo-interfluves corresponds to the broad salt marshes along the Delaware Bay coast and lagoons along the ocean coast. Both the estuarine coast salt marshes and ocean coast lagoons have a digitate, coast-perpendicular morphology which reflects the relief and configuration of the pre-transgressive surface (Chrzastowski, 1986). Intermediate to these low-lying areas of transgression are digitate uplands or "necks" of exposed Pleistocene sediments representing interfluves yet to be transgressed (Chrzastowski, 1986). It is the relationship between this antecedent pre-transgressive surface and the Holocene marsh sediments that is the subject of this investigation.

Holocene Coastal Environments in Delaware

Holocene coastal environments in Delaware include nearshore shallow marine sands; the Atlantic Coast barrier system and Cape Henlopen spit complex (beaches, dunes, washover fans, marshes, lagoons, inlets, and tidal deltas); and the Delaware Bay estuary (washover barriers and broad marshes) (Kraft, 1971a, b). These coastal units have been migrating across the continental shelf and coastal plain in response to relative sea-level rise (Allen, 1978), as documented by Kraft *et al.* (1979), Kraft and Allen (1975), Swift *et al.* (1972), and Belknap *et al.* (1976). Allen (1978) reports that the oldest post-glacial coastal-marsh sediments in Delaware are dated at 11,000 years before present and are buried at 30 m below present sea level in the channel of the ancestral Indian River along the Atlantic Delaware coast (Kraft *et al.*, 1976). Marsh deposits at a depth of 27 m at South Bowers are dated at nearly 10,000 years before present, indicating that coastal sedimentation was occurring in the vicinity of at least part of the present bay shoreline by this time (Belknap and Kraft, 1977; Allen, 1978).

Holocene Local Relative Sea-Level

General

The Delaware coastal zone area is undergoing a relatively rapid marine transgression due to the combined effects of eustatic sea-level rise and tectonic subsidence (Maurmeyer, 1978). Eustatic rise of water level is due primarily to melting of ice caps since the end of Pleistocene glaciation, and subsidence is occurring in response to sedimentary and tectonic effects of the Baltimore Canyon trough geosyncline (Belknap and Kraft, 1977) and water-loading of the continental shelf (Bloom, 1967). Coastal fringing tidal salt marshes which border the western shore of the Delaware Bay and Delaware Inland Bays have formed in response to this rise in Holocene relative sea level.

Local Sea-Level Variations

Several versions of a Holocene sea-level rise curve have been developed for the Delaware coastal area from radiocarbon-dated coastal-marsh sediments (Belknap, 1975; Kraft, 1976; Belknap and Kraft, 1977). The local relative sea-level rise curve of Belknap and Kraft (1977) is used for purposes of this study (Figure 2.3). Sea level has been rising relative to the Delaware Coast since the end of the last glacial period approximately 15,000 years ago (Kraft *et al.*, 1987). The Holocene local relative sea level curve depicts three segments with different rates of sea-level rise in the past 12,000 years. The smooth curve consists of three segments: the first indicates sea level rising at rates of 32.4 cm/century from 12 ka to 4.8 ka; the second indicates sea level rising at about 19.8 cm/century from about 4.8 ka to 1.8 ka; and the third indicates sea level rising at 8.5 cm/century from about 1.8 ka to present.

However, tide gauge data collected over the past 50-100 years in coastal Delaware and nearby Atlantic City, New Jersey, indicate rates of 0.33 to 0.39 cm/year over the period from 1920 to 1980 (Hicks and Crosby, 1974). Figure 2.4 depicts tide gauge data from Breakwater Harbor, near Cape Henlopen, Delaware (Demarest, 1978). The rate of local sea-level rise in Breakwater Harbor since 1920 is approximately 0.33 cm/yr, as compared to the slower rate of 0.20 cm/yr for the last several millennia (Kraft *et al.*, 1987). This indicates that coastal Delaware may be undergoing a rapid acceleration in the rate of sea-level rise, mainly due to subsidence and sediment compaction within the Atlantic Coastal Plain. Such short-term changes in relative sea

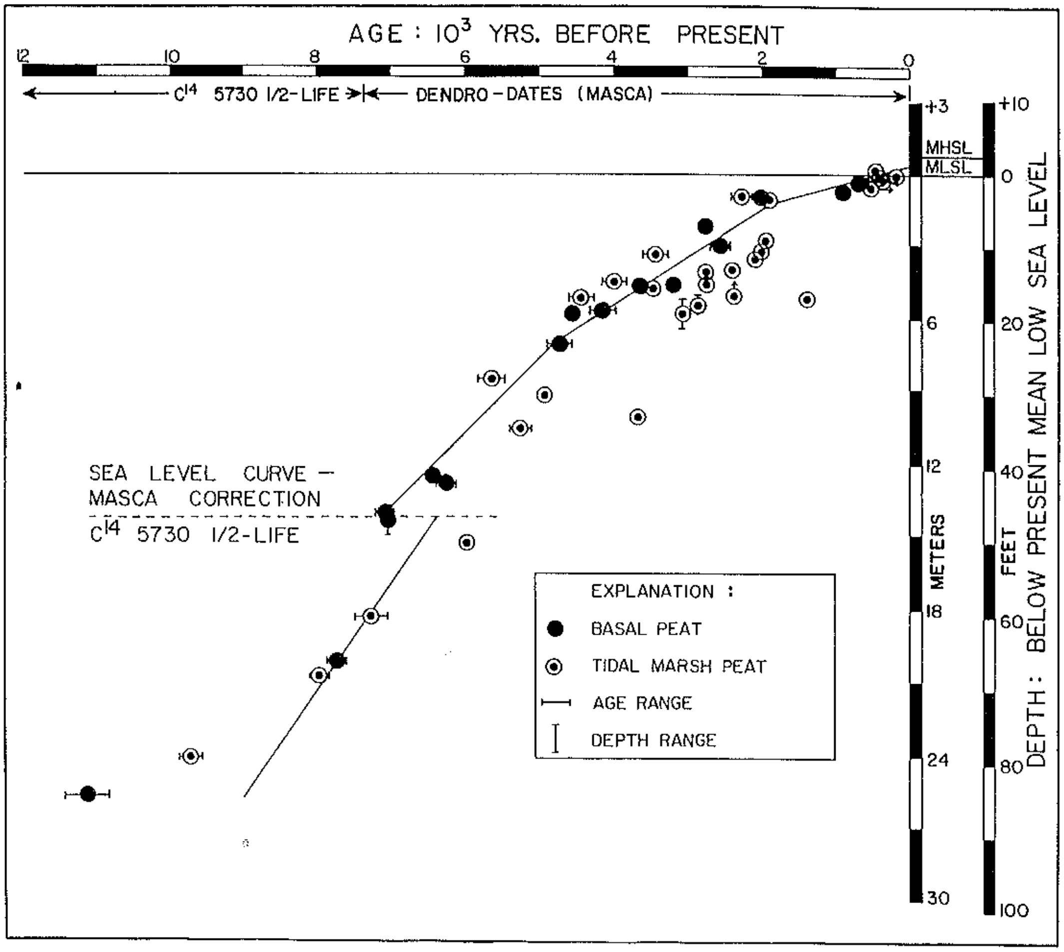


Figure 2.3 Local relative sea-level curve for the Delaware coast (from Belknap and Kraft, 1977).

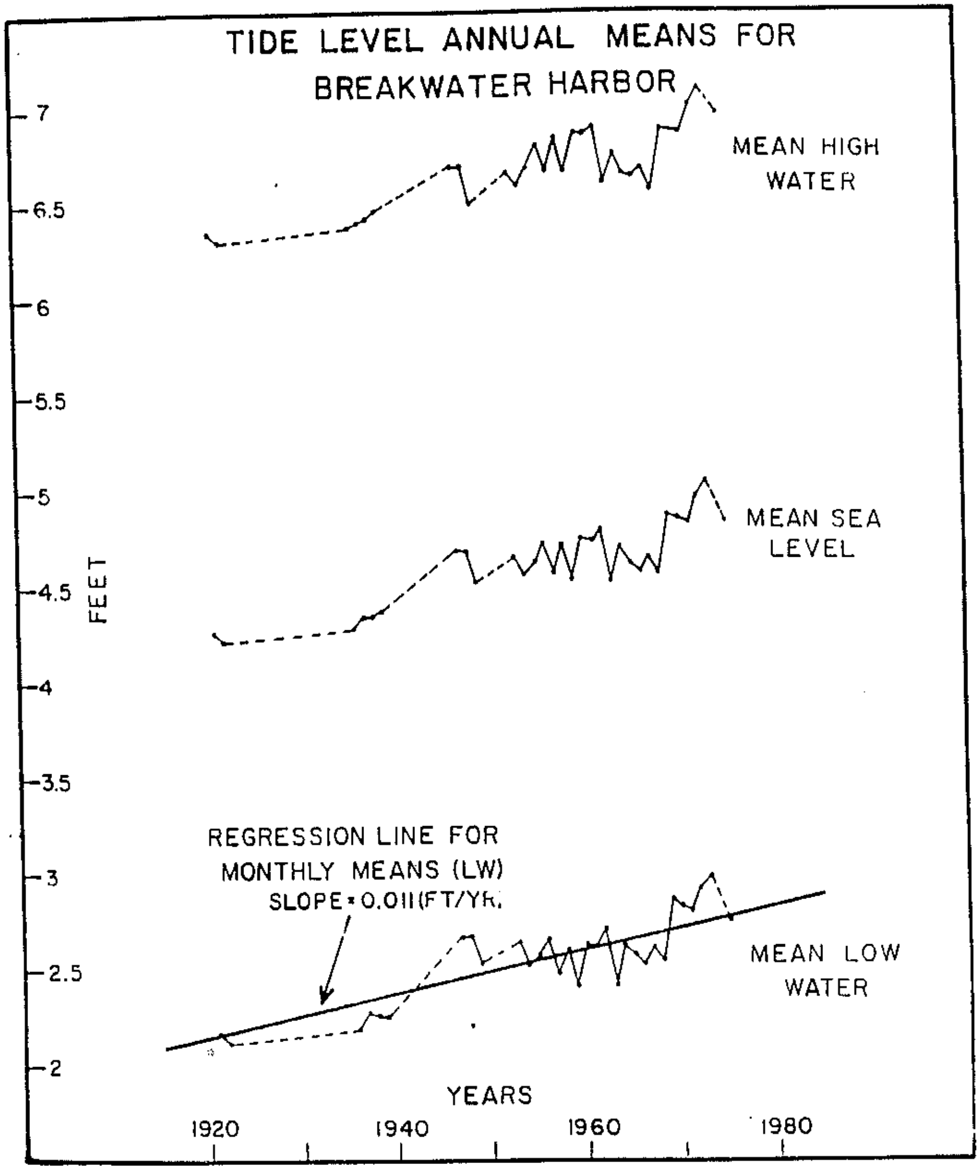


Figure 2.4 A regression line of tide gauge data, showing a rate of 0.33 cm/yr of sea-level rise. Tide gauge data were recorded by NOAA, National Ocean Service, from Breakwater Harbor, near Cape Henlopen, Delaware (from Demarest, 1978).

level could greatly modify sedimentary processes and deposits in coastal areas (Allen, 1977; Maurmeyer, 1978).

In recent decades, human activities have contributed to global warming, causing an acceleration in the rate of sea-level rise. Many scientists agree that continued global warming will result in an acceleration in the rate of sea-level rise; however, predictions for future rates of sea-level rise vary, based on methodologies and assumptions. Table 2.1 summarizes the results of some of these studies and provides a means of comparing predictions for global sea-level rise to those for local sea-level rise in Lewes, Delaware, and Ocean City, Maryland (Washburn, 1991). All of these scenarios reflect the fact that relative sea level along the Atlantic Coast is rising at a faster rate than worldwide.

Global Sea-Level Rise

Many studies (Jelgersma, 1988; Pirazolli, 1986, 1989) have demonstrated that there have been fluctuations in sea level through geologic time, and that sea level has been relatively stable over the past five to six thousand years. The term 'sea level' usually refers to mean sea level, approximated by the long-term average of high- and low-tide levels, or mean tide level. Research based on tide-gauge data from around the world indicates that global sea level has risen 18 cm during the past century (1.8 mm/yr) (Douglas, 1991). However, it should be noted that tide-gauge records from around the world show substantial variations in mean sea level trends (Pirazolli, 1986; Bird, 1993). A generalized worldwide sea level curve is depicted in Figure 2.5.

Table 2.1 Projected scenarios of sea-level rise; values expressed as cm above present MSL (after Washburn, 1991).

(Year)	2000	2050	2080	2100
Global				
Current Trends	2.54	10.16	15.24	17.78
Hoffman et al. 1983				
Low	5.08	22.86	-	55.88
Mid-Low	7.62	53.34	-	144.78
Mid-High	12.70	78.74	-	215.90
High	17.78	116.84	-	345.44
Hoffman et al. 1986				
Low	2.54	20.32	-	55.88
High	5.08	55.88	-	368.30
NAS Estimate (Revelle, 1983)	-	-	71.12	-
Lewes, Delaware				
Current Trends	12.70	30.48	43.18	48.26
Hull and Titus 1986				
Low	15.24	45.72	-	91.44
Mid-Low	20.32	76.20	-	180.34
Mid-High	22.86	101.60	-	251.46
High	27.94	139.70	-	381.00
NAS Estimate (Revelle, 1983)	-	-	101.60	-
Ocean City, Maryland				
Current Trends	7.62	25.40	-	-
Titus et al. 1985				
Mid-Low	12.70	66.04	-	-
Mid-High	17.78	91.44	-	-

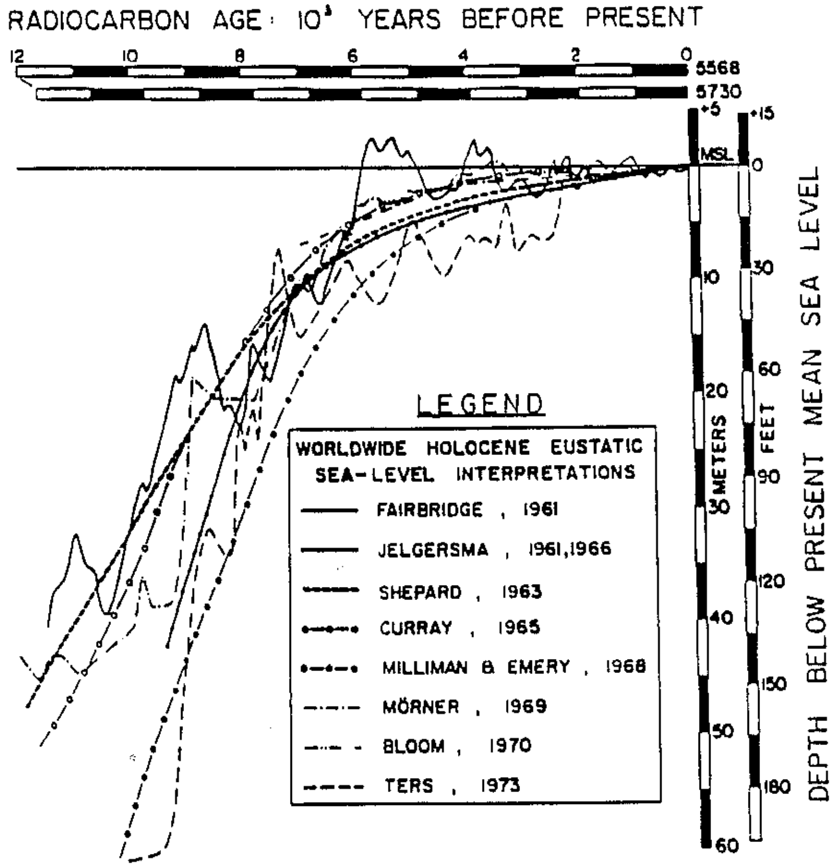


Figure 2.5 Worldwide sea level curves (from Belknap and Kraft, 1977).

It has been predicted that the greenhouse gases (notably carbon dioxide, methane, and nitrous oxide produced by industrial and agricultural activities) will double during the coming century, resulting in an increase of 1.5 to 4.5°C in the mean temperature of the lower atmosphere (Bird, 1993; Douglas, 1995; IPCC, 1995). This increase in temperature may cause an expansion of the volume of near-surface ocean water (the steric effect), and partial melting of the world's snowfields, ice sheets, and glaciers, releasing water into the oceans (Bird, 1993; Douglas, 1995; IPCC, 1995). However, although much of the past and predicted rise in sea level may be explained by climatic warming, local conditions such as subsidence of land masses, river flow, and weather also contribute to this effect.

Estimates of the magnitude of predicted sea-level rise vary considerably. The most recent predictions are that global mean sea level will rise 15 cm by the year 2050, and about 34 cm by 2100, with a 1% chance that global warming will raise sea level 1 meter in the next 100 years and 4 meters in the next 200 years (Douglas, 1995; IPCC, 1995; Titus and Narayanan, 1995). Although estimates may change as more data are collected, it is realistic to consider the implications and possible effects of a global sea level rise of 0.50 to 1 meter over the next 100 to 200 years (Bird, 1993). Titus and Narayanan (1995) have developed normalized sea level projections for the next 200 years; estimated sea level rise predictions for Lewes, Delaware are shown in Table 2.2.

However, as described later in this dissertation, factors other than anthropogenic climate change will cause an increase in relative sea level. These factors

Table 2.2 Projected sea level rise (cm) at Lewes, Delaware, compared with 1990 levels. Based on Titus and Narayanan (1995) normalized projections and historic rate of sea-level rise of 0.31 cm/yr (after Titus and Narayanan, 1995).

Projected Sea Level Rise (cm) at Lewes, Delaware
Based on Historic Rate of Sea-Level Rise of 0.31 cm/yr and
Normalized to Titus and Narayanan (1995) Projections;
Compared with 1990 Levels (cm)
Sea Level Projection by Year:

Cumulative Probability (%)	Year				
	2025	**2050**	**2100**	**2150**	**2200**
10	12	20	35	53	70
20	12	22	44	66	88
30	14	25	50	76	102
40	15	27	54	85	116
50	16	29	59	93	129
60	17	32	64	103	143
70	19	34	70	115	163
80	20	37	78	130	190
90	23	42	89	156	239
95	25	46	100	184	296
97.5	28	50	112	217	361
99	30	57	126	260	467

include compaction and subsidence of the land, groundwater depletion, and anthropogenic effects on tidal range, tidal prism, and sediment input to estuarine environments.

Area of Study

The present investigation involves an analysis of the wetlands/uplands boundary along the Delaware Bay between Bowers Beach and Cape Henlopen, Delaware (sites DB-1 to DB- 15); along the Lewes and Rehoboth Canal (sites L&R-1 to L&R-5); and along the fringing marsh shorelines of Rehoboth Bay (sites RB-1 to RB-24), Indian River Bay (sites IRB-1 to IRB-19), and Little Assawoman Bay (sites LAB-1 to LAB-7). Figures 2.6 to 2.9 are index maps showing the areas of study; site numbers and location names of study areas are listed in Table 2.3 through 2.6.

Delaware Bay and Estuary

General

The Delaware Estuary extends 217 kilometers from its mouth at the Atlantic Ocean to the head of tide at Trenton, New Jersey. Delaware Bay is the lower part of the estuary, and extends from Cape May, New Jersey and Cape Henlopen, Delaware in the south to a line between Arnold Point, New Jersey, and Smyrna River, Delaware in the north (Maurmeyer, 1978). The estuary formed as the valley of the Delaware River was drowned during the Holocene marine transgression; Delaware Bay is a sub-triangular shaped body of water, approximately 75 kilometers long and up to 44 kilometers wide.

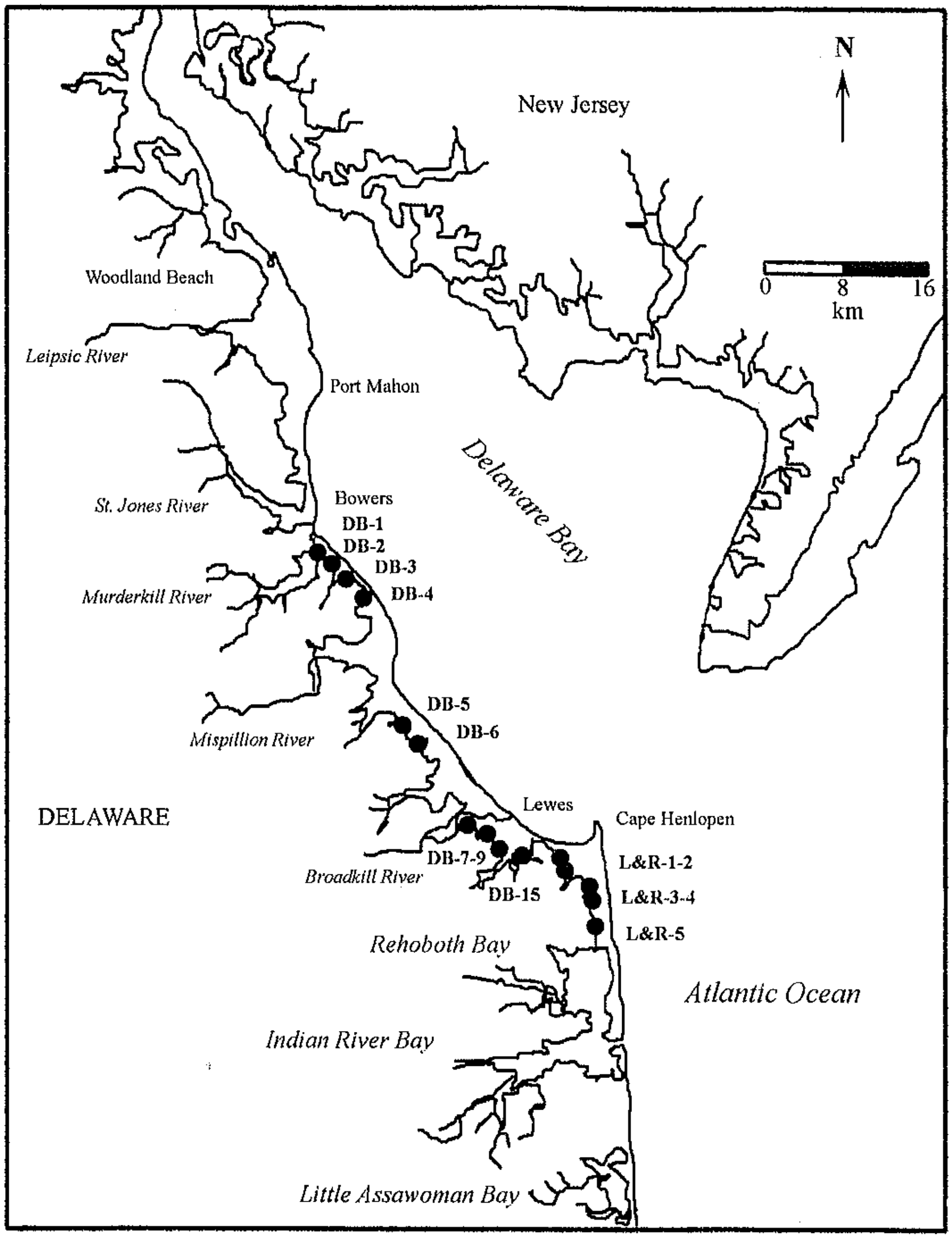


Figure 2.6 Study areas and sample locations along the wetlands/uplands boundary of western Delaware Bay and the Lewes and Rehoboth Canal. See Table 2.3 for location name corresponding to site number.

Table 2.3 Site number and location name of study areas along Delaware Bay and Lewes and Rehoboth Canal wetlands/uplands boundary. See Figure 2.6 for site locations.

Site Number	Location Name
DB-1	Island Field-1
DB-2	Island Field-2
DB-3	McKim-1
DB-4	McKim-2
DB-5	Prime Hook Radio Station
DB-6	Wall Island-1
DB-7	Oyster Neck Road-1
DB-8	Oyster Neck Road-2
DB-9	Ritter-1
DB-10	Ritter-2
DB-11	Ritter-3
DB-12	Ritter-4
DB-13	Hercules-2
DB-14	Canary Creek-1
DB-15	Pagan Creek-1
L&R-1	Wolfe Runne-1
L&R-2	Wolfe Runne-2
L&R-3	Glade-1
L&R-4	Glade-2
L&R-5	Anderson

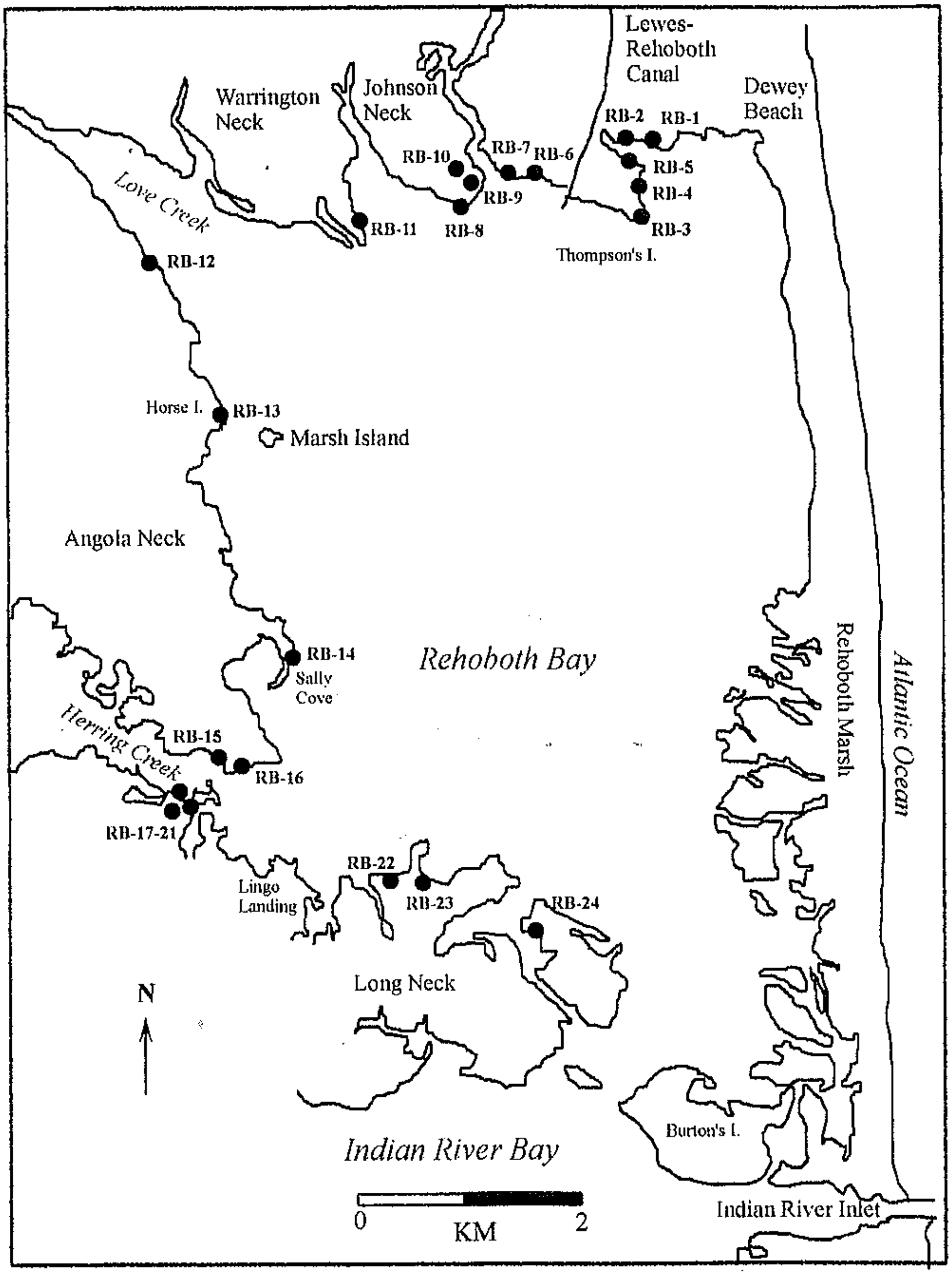


Figure 2.7 Study areas and sample locations along the wetlands/uplands boundary of Rehoboth Bay. See Table 2.4 for location name corresponding to site number.

Table 2.4 Site number and location name of study areas along Rehoboth Bay wetlands/uplands boundary. See Figure 2.7 for site locations.

Site Number	Location Name
RB-1	Dodd-1
RB-2	Dodd-2
RB-3	Thompsons Island-2
RB-4	Thompsons Island-3
RB-5	Thompsons Island-4
RB-6	Bay Vista-1
RB-8	Draper-1
RB-9	Draper-2
RB-10	Draper-3
RB-11	Sugar Shack
RB-12	Mills-1
RB-13	Horse Island
RB-14	Camp Arrowhead-2
RB-15	Delaware Wildlands-1
RB-16	Delaware Wildlands-2
RB-17	Pot Nets North-1
RB-18	Pot Nets North-2
RB-19	Pot Nets North-3
RB-20	Pot Nets North-4
RB-21	Pot Nets North-5
RB-22	Nats Marsh-1
RB-23	Nats Marsh-2
RB-24	Pullover-1

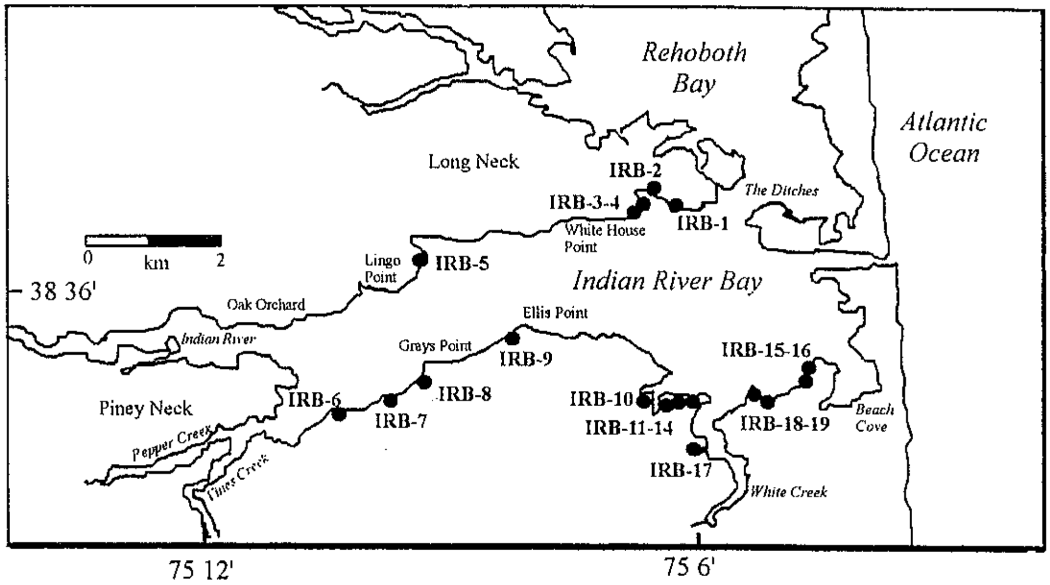


Figure 2.8 Study areas and sample locations along the wetlands/uplands boundary of Indian River Bay. See Table 2.5 for location name corresponding to site number.

Table 2.5 Site number and location name of study areas along Indian River Bay wetlands/uplands boundary. See Figure 2.8 for site locations.

Site Number	Location Name
IRB-1	Indian Landing
IRB-2	Boat House Cove
IRB-3	Burton-1
IRB-4	Burton-2
IRB-5	Lingo Point
IRB-6	Bay Colony-1
IRB-7	Murray-1
IRB-8	Murray-2
IRB-9	Blackwater Point
IRB-10	Bethany Forest
IRB-11	Bethany Bay-4
IRB-12	Bethany Bay-3
IRB-13	Bethany Bay-2
IRB-14	Bethany Bay-1
IRB-15	Pasture Point Cove-1
IRB-16	Pasture Point Cove-2
IRB-17	Seawinds
IRB-18	Wm.Derrickson-1
IRB-19	Wm. Derrickson-2

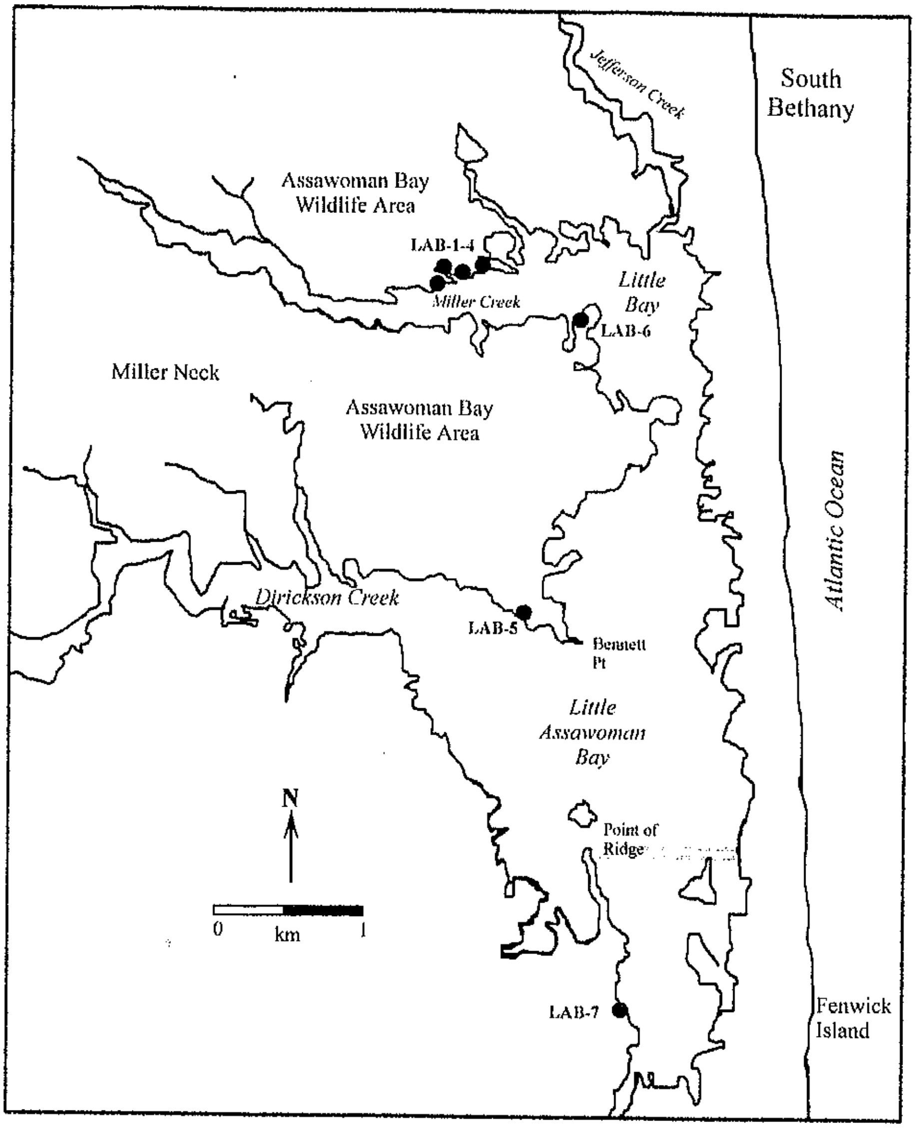


Figure 2.9 Study areas and sample locations along the wetlands/uplands boundary of Little Assawoman Bay. See Table 2.6 for location name corresponding to site number.

Table 2.6 Site number and location name of study areas along Little Assawoman Bay wetlands/uplands boundary. See Figure 2.9 for site locations.

Site Number	Location Name
LAB-1	Muddy Neck-1
LAB-2	Muddy Neck-2
LAB-3	Muddy Neck-3
LAB-4	Muddy Neck-4
LAB-5	Mulberry Landing
LAB-6	Strawberry Landing
LAB-7	Little Assawoman Bay

A deeply incised channel up to 46 meters in depth occurs in the southwestern part of the bay off Cape Henlopen, and represents the axis of the ancestral Delaware River (Maurmeyer, 1978). Approximately 80% of the bay consists of shallow subtidal flats less than ten meters deep (Schuster, 1959; Polis and Kupferman, 1973; Weil, 1976; Maurmeyer, 1978).

The evolution of coastal environments is greatly influenced by the physical forces operating in the coastal zone (Polis and Kupferman, 1973; Maurmeyer, 1978). The following sections describe the processes active in the Delaware Bay estuarine environment.

Winds

As summarized by Maurmeyer (1978), the direction, speed, duration, and fetch of winds blowing over Delaware Bay determine the nature of waves generated within most of the bay. Maurmeyer (1978) reports that on a yearly basis, prevailing winds blow from the northwest, and dominant winds from the northeast (U.S. Army Corps of Engineers, 1977). Monthly data compiled by Maurmeyer (1978) show that prevailing winds in winter months blow from the northwestern quadrant and in summer months from the southwest; during major storms such as northeasters and hurricanes, winds generally blow from the northeast.

Waves

The large fetch of Delaware Bay allows wind to generate large waves within the estuary. The distribution of waves in Delaware Bay generally correlates with wind speed and direction trends (Maurmeyer, 1978). A comparison of shoreline erosion rates and a wave-frequency rose diagram for Delaware Bay suggests that most shore erosion is caused by local wind-generated waves (Weil, 1977; Allen 1977). Waves in Delaware Bay are less than 0.6 m 80% of the time (Maurer and Wang, 1973; Allen, 1977; Maurmeyer, 1978).

Tides

Tides in Delaware Bay are semi-diurnal; at the mouth of Delaware Bay, mean and spring tide ranges are approximately 1.2 m and 1.5 m, respectively. Mean tidal range generally increases from approximately 1.2 m in the lower part of the bay to 2.2 m in the upper portion of the bay at the head of the tide at Trenton (Allen, 1977; Maurmeyer, 1978). The tidal range in the tributaries of Delaware Bay is gradually reduced by friction from the mouth of the rivers to the headward limit of tidal intrusion. Also, the duration of tidal inundation is drastically reduced along the upper limit of marsh vegetation where the marsh is inundated by only the highest tides (Allen, 1977).

Salinity

Salinity varies geographically and seasonally in Delaware Bay, ranging from 33 ppt near the mouth of the bay to 6 ppt in the upper sections of the bay (Mather *et al.*, 1973). Significantly higher and lower salinities occur within the coastal marshes and will be discussed later in greater detail.

Marsh/Shoreline Characteristics

Shoreline sediments along the Delaware Bay are derived from many sources. Eroding subcrops and outcrops of pre-Holocene sands and pebbles provide an irregularly spaced but locally very important source of sediment to the shoreline. The Delaware River, its tributaries, and tidal rivers transport loads of predominantly fine-grained (silt and clay) sediments into the estuary (Mansue and Commings, 1974). Fine sands and muds of subtidal flats may be resuspended and transported during storms (Weil, 1977). The shelf and Atlantic coast is also a source of sediment transported into the mouth and along the margins of Delaware Bay (Allen, 1977), as sediment is redistributed along the shoreline by erosion and transport. Erosion of relict-marsh tidal flats, and wind- and water-borne sand from beaches contribute a significant volume of sediment to back-barrier marshes (Allen, 1977).

Weil (1977) notes that shorelines formed of marsh sediments erode at greater rates than sandy shorelines or the even more-resistant shorelines formed of pre-Holocene sediments. Some marsh shorelines are located over deep pre-Holocene incised valleys and the higher rates of erosion may be due in part to subsidence by compaction of underlying fine-grained sediments (Allen, 1977).

Salt marshes along the Delaware Bay shoreline were studied by Elliott (1972), Allen (1974, 1977) and Richter (1974). Elliott (1972) studied the Great Marsh north of Lewes, defined sedimentary facies and environments of deposition, and presented a model for the development and maintenance of coastal marshes (Maurmeyer, 1978). Allen (1974) analyzed core data to reconstruct the paleogeography of marshes at Lewes Creek and South Bowers. The petrology and stratigraphy of coastal marshes along Delaware Bay were studied by Allen (1977); seventeen facies and subfacies were identified based on lithologic and floral components (Maurmeyer, 1978). Richter (1974) determined the three dimensional configuration of Delaware Bay coastal marshes from core data, and calculated that 0.2 x 106 m^3 of coarse-grained material (sand and gravel) and 2.3 x 106 m^3 of fine-grained sediments (silt and clay) of Holocene age underlie present Delaware Bay salt marshes (Maurmeyer, 1978).

Inland Bays

General

Delaware's Inland Bays consist of three interconnected water bodies: Rehoboth Bay, Indian River Bay, and Little Assawoman Bay. These bays are shallow (< 2.3 m maximum depth), microtidal (< 2 m mean tidal range) lagoons, with a combined water area of 86.2 km^2, and approximately 32.7 km^2 of adjacent intertidal salt marshes. Rehoboth Bay is the largest bay (33.8 km^2), with a mean tidal range of 0.10 m and spring range of 0.20 m. Its total length of shoreline measures 40.2 km, with 12.1 km^2 area of adjacent marsh (Chrzastowski, 1986). Indian River Bay (29.3 km^2) has a coast-perpendicular configuration, and a mean tidal range from 0.80 m at the mouth to 0.20 m at the head, and spring tidal range of 0.9 to 0.30 m. Total length of shoreline in

Indian River Bay is 38.4 km, with approximately 6.9 km^2 of adjacent marsh area (Chrzastowski, 1986). Little Assawoman Bay (23.1 km^2) is the smallest of the Inland Bays, and scant data have been acquired regarding its general characteristics. Mean- and spring-tide ranges are approximately 0.10 and 0.20 m, respectively. The low-energy, transgressive shoreline of Little Assawoman Bay can be divided into three major geomorphic types, including irregular marsh shoreline (79%), sandy pocket beaches (4%), and artificial stabilization structures such as rubble and bulkheads (17%). As is typical of the other two Inland Bays, the approximately 13.5 km^2 of adjacent marsh area at Little Assawoman Bay consists of broad emergent wetlands which have developed on topographically low areas along the western shore and on relict flood tidal deltas/overwash fans in eastern portions.

The action of natural physical forces, such as wind, waves, tides, and currents have affected and controlled the development and evolution of fringing marsh environments in the Inland Bays. Discussions of physical processes in the bays are presented in Polis and Kupferman (1973); Delaware Coastal Management Program (1977); Lanan and Dalrymple (1977); Dennis and Dalrymple (1978); Carey (1979); Swisher (1982); Chrzastowski (1986); Wong (1987); Crouse (1989); Raney *et al.* (1990) and Weston *et al.* (1993). The following summary is compiled from these sources.

Wind

Analysis of wind data from Indian River Inlet indicates that prevailing winds in the Inland Bays area are from the southwest in spring and summer, and from the northwest and northeast in the winter months. Winds above 48 kph are most frequently from the northwest, and winds of the highest velocities are most frequently from the northeast (Polis and Kupferman, 1973). The velocity, duration, and fetch (distance of open water over which the wind blows) determine the characteristics of waves within the bays. As these wind parameters increase, wave parameters (height, period, and energy) also increase. The fetch in the Inland Bays is limited, due to the small size of the bays. Swisher (1982) considered 5795 m (approximately 5.8 km) to represent the average fetch for Rehoboth Bay. Crouse (1989) calculated fetches for 40 sites along Rehoboth Bay, and documented a maximum fetch of approximately 10.5 km for winds from the northeast. The Delaware Coastal Management Program (1977) states that the fetch across Indian River Bay is 9.7 km for easterly winds, and considerably less for all other directions. No published information concerning fetch for Little Assawoman Bay is available. However, rough measurements indicate that the maximum fetch is less than 4.8 km for the northerly and northwesterly winds.

Waves

The Inland Bays are generally low-energy wave environments. The shallowness of the water and limited fetch preclude development of extreme wave heights in the Inland Bays. Yet, the presence of wave-generated ripple marks in sections of the bottom of Rehoboth Bay (Swisher, 1982) and Indian River Bay (Carey, 1979) indicates that waves are capable of bottom sediment transport in the bays. According to calculations by Swisher (1982) utilizing wave forecasting curves developed by the U.S.

Army Corps of Engineers (1977), wind speeds of approximately 32.2 kph would generate 0.3 m high waves within Rehoboth Bay. The Delaware Coastal Management Program (1977) calculated that maximum wave heights of 0.76 m to 1.07 m can be expected for wind velocities of 80 kph in Indian River Bay. Due to the short fetch and shallower water depths, wind-generated waves in Little Assawoman Bay can be expected to be smaller than in the other bays for most wind conditions.

Tides

Tides in the Inland Bays are semidiurnal, with a periodicity of 12.42 hours. Tides enter Indian River Bay directly through Indian River Inlet; as tides propagate into Rehoboth Bay through the "Ditches", they are constricted and attenuated, resulting in much lower tidal energy in Rehoboth Bay than in Indian River Bay (Wong, 1987). Lanan and Dalrymple (1977) report a phase lag of 1.75 hours at the Ditches for both ebb and flood phases; Karpas (1978) reports phase lags of 3 hours at Massey's Ditch and 5 hours at the entrance to the Lewes and Rehoboth Canal in northern Rehoboth Bay. Raney *et al.* (1990) present field data and results of a numerical model to predict tide ranges at various locations in Rehoboth Bay and Indian River Bay; comparison with historical data sets suggest that the tide range in the Inland Bays has increased, due to the increase in the cross-sectional area and tidal prism on Indian River Bay (Raney *et al.*, 1990).

Currents

Currents within the Inland Bays result from wind-generated waves and the effects of tides, and are significant with regard to water and sediment transport within the bays. Field measurements of current velocities, as well as numerical models predicting current velocities, have been reported for Rehoboth Bay and Indian River Bay by numerous investigators (Lanan and Dalrymple (1977); Carey (1979); Swisher (1982); Raney *et al.* (1990)). Swisher (1982) reports that wave-induced longshore currents in Rehoboth Bay most likely result in net southerly transport along the western shoreline in the winter months, and northerly transport in the summer months, due to seasonal variations in prevailing wind directions. Analysis of current ripple patterns on the bay bottom suggests that bottom-current velocities approach 30-35 cm/sec in the nearshore zone. The tidal currents in Indian River Inlet are important in water exchange between the ocean and the bays, and sediment transport into and out of the bays.

Salinity

Salinities in the Inland Bays are characterized by a wide range, varying from oligohaline to euryhaline both within and between basins. Rehoboth Bay is typically polyhaline, with salinities ranging from 18 to 25 ppt throughout the year. Rehoboth Bay receives fresh water from two major tributaries, Love and Herring Creeks, with salt water exchange occurring between Rehoboth Bay and Lewes and Rehoboth Canal (to the north) and Masseys Ditch (to the south). Indian River Bay is characterized by a wide range of salinities, from euryhaline (25-30 ppt) at Indian River Inlet to oligohaline

0.5-5 ppt) at Millsboro, the head of tide. Primary sources of freshwater input from tributaries include: Indian River, Stockley Branch, Pepper Creek, Vines Creek, Blackwater Creek, and White Creek. Little Assawoman Bay is a polyhaline (18-25 ppt) water body, with freshwater flow contributed from Dirickson and Miller Creeks. Tidal salt water connections include the Assawoman Canal (to the north) and Assawoman Bay through the Ditch (to the south).

Marsh/Shorelines Characteristics

Fringing marshes along the northern, western, and southern shores of the Inland Bays are low-lying tidal wetlands. These fringing marshes are irregular to crenulate in planform, and generally exhibit an erosional undercut scarp at the water's edge. The scarps, generally 0.15-0.30 meters above mean sea level, consist of a dense mat of sediment, peat, and roots. Examples of fringing marshes include wetlands along the tidal tributaries of the Inland Bays, such as Love Creek and Herring Creek in Rehoboth Bay; Indian River, Pepper Creek, and White Creek in Indian River Bay; and Miller Creek and Dirickson Creek in Little Assawoman Bay. Dominant vegetation in the marshes includes salt marsh cordgrass (*Spartina alterniflora* Loisel.), salt meadow hay (*Spartina patens* (Ait.) Muhl.), spike grass (*Distichlis spicata* (L.) Greene), marsh elder (*Iva frutescens* L.), groundsel bush (*Baccharis halimifolia* L.), and giant reed (*Phragmites australis* (Cav.) Trin. ex Steud.).

In contrast to the fringing marshes, backbarrier marshes, developed over relict flood tidal delta and overwash sand deposits associated with the Atlantic Coast barrier beach system, have formed along the eastern shores of the Inland Bays. Examples of backbarrier marsh areas, which are characterized by numerous marsh islands dissected by channels and ditches, include Rehoboth Marsh in Rehoboth Bay; Mare Marsh and Salt Marsh in Indian River Bay; and Daisy Marsh in Little Assawoman Bay. These backbarrier marshes developed and evolved through different processes than the fringing marshes, and have not been included in this study.

Chapter 3

BACKGROUND INFORMATION AND PREVIOUS WORK

Introduction

The base of existing information for this study consists of present knowledge of: 1) wetlands formation, evolution, and development; 2) wetlands zonation patterns; 3) sediment accumulation (vertical accretion) in wetlands; and 4) wetlands adjustments to sea level rise inferred over a broad area over the last several thousand years. Stratigraphic sequences based on cores taken in modern coastal environments, and associated environmental interpretations have provided evidence for past sea levels and locations of ancient marshes. These data have been used as the foundation for constructing sea level rise curves, reconstructing locations of past shorelines, and characterizing the areal extent of past wetlands in coastal Delaware.

Marshes: Definition and Classification

Definition

Coastal salt marshes have been defined as "environments high in the intertidal zone where a generally muddy substrate supports varied and normally dense stands of halophytic plants" (Allen and Pye, 1992a). They form on open coasts, in tidal embayments, behind barrier islands and in deltaic or nondeltaic estuaries. Salt marshes usually develop between approximately mean sea level and higher high water (Frey and Basan, 1985; Allen and Pye, 1992a), and their development is primarily controlled by three physical factors which affect the elevation of the marsh surface: sediment supply, tidal range, and movement of relative sea level. In general, plant species richness increases with elevation, but at high elevations, species distributions tend to be governed by competition whereas low elevation limits are governed mainly by the plants' physiological tolerances (Pielou and Routeledge, 1976). Although a strong relationship has been demonstrated between frequency and duration of tidal inundation and plant species distribution (Moody, 1978; Hutchinson, 1982; Dawe and White, 1986), the vertical distribution and ecological roles of certain species may also be influenced by the local geographic setting.

Tidal salt marshes in Delaware are closely tied to adjacent estuarine systems, and are found along the margins of Delaware Bay and the Inland Bays (Figure 2.2). While tidal marshes may include freshwater marshes, the tidal marshes examined in this study are restricted to those in brackish or saline areas, and may be defined as areas of halophytic grassland and dwarf shrubs adjacent to tidal saline water bodies. The configuration of these tidal salt marsh areas is largely determined by: 1) the vertical range of the tide, which delineates tidal flooding depths and the vertical extent of vegetation; 2) the form of the tidal cycle (e.g. diurnal, semi-diurnal) which controls the frequency and duration of submergence and emergence; 3) salinity range; 4) the frequency and intensity of mechanical disturbance (e.g., waves, winds, sedimentation);

and 5) sediment accumulation or erosion. Natural or human-induced changes in rate or intensity of any of the physical, chemical, or biological processes controlling salt marsh development contributes to the natural variability of salt marshes. Daiber (1986) described the tidal salt marsh as a domain of gradients, due to differential deposition of sediments, and possible ranges of tidal excursions and salinity fluctuations.

Classification

Many different classification systems have been developed for wetlands areas; typical tidal wetland community types in Delaware have been summarized for the National Wetlands Inventory by Tiner (1985), using the Cowardin system (Cowardin *et al.*, 1979). Table 3.1 outlines examples of typical estuarine wetland communities in Delaware, as described by Tiner (1985). Note that salinity, vegetation, and tidal water regime are primary controlling factors in this classification scheme.

Wetland environments investigated in this study are restricted to coastal tidal salt marshes, and in this report the term "wetland" is used interchangeably with "tidal salt marsh" or polyhaline/mesohaline "estuarine emergent" wetland (classification of Cowardin *et al.*, 1979). Estuarine emergent wetlands are the low-lying grassy areas around the perimeter of Delaware Bay and Inland Bays that are flooded by tidal waters for varying periods depending on elevation. These wetlands are vegetated by a variety of macrophytic plants depending on frequency and duration of tidal flooding, substrate, and salinity.

Table 3.1 Examples of estuarine wetland communities in Delaware (from Tiner, 1985).

Wetland Type (Halinity)	Dominance Type	Associated Plants	Water Regime (flooding)
Emergent (polyhaline)	Smooth Cordgrass (tall form)	(none)	regularly flooded
Emgergent (polyhaline)	Spike Grass and Black Grass	High-tide Bush, Big Cordgrass, Smooth Cordgrass, Salt Meadow Hay, Seaside Goldenrod	irregularly flooded
Emergent (polyhaline)	Spike Grass	Salt Meadow Hay, Common Threesquare, High-tide Bush, Seaside Goldenrod, Big Cordgrass, Rose Mallow, Sea Myrtle, Wax Myrtle	irregularly flooded
Scrub-Shrub (polyhaline)	High-tide Bush	Sea Lavender, Spike Grass, Salt Meadow Hay, Common Reed, Wax Myrtle, Poison Ivy	irregularly flooded
Emergent (mesohaline)	Switchgrass	Wax Myrtle, Sea Myrtle, Big Cordgrass, Red Cedar	irregularly flooded
Emergent (mesohaline)	Cattail, Rose Mallow, and Salt Meadow Hay	Wax Myrtle, Salt Marsh Bulrush, Switchgrass, Common Reed, Poison Ivy, Red Cedar	irregularly flooded
Emergent (mesohaline)	Smooth Cordgrass (short form) and Salt Meadow Hay	High-tide Bush, Big Cordgrass, Water Hemp, Germander	irregularly flooded
Emergent (oligohaline)	Cattail and Rose Mallow	Common Reed, Smooth Cordgrass	irregularly flooded

Emergent wetlands are typically divided into two general zones based on elevation and general physiographic conditions. These lateral zonations are commonly referred to as: low marsh, or regularly flooded wetlands; and high marsh, or irregularly flooded wetlands (Tiner, 1985; Figure 3.1). The low marsh areas extend from about mean sea level to the mean high tide level, and are alternately flooded by the tides and exposed subaerially at least once (usually twice) daily. In high and moderate salinities, the low marsh is dominated by the tall form of smooth cordgrass (*Spartina alterniflora* Loisel.).

The high marsh ranges from above the mean high water mark to the extreme spring high tide level. This zone is generally exposed to air, and is less frequently flooded by the tides, usually during the semi-monthly spring tides and during storms. Vegetation composition will vary in high marsh zones depending on elevation relative to the tide, drainage characteristics of the substrate, and salinity. A greater diversity of plant life is found in the high marsh zone than in the low marsh, including: the short form of smooth cordgrass (*S. alterniflora*), salt hay grass (*Spartina patens* (Ait.) Muhl.), spike grass (*Distichlis spicata* (L.) Greene) , black needlerush (*Juncus roemerianus* Scheele) , glassworts (*Salicornia bigelovii* Torr., *Salicornia europaea* L., and/or *Salicornia virginica* L.), sea lavender (*Limonium carolinianum* (Walt.) Britt.), orach (*Atriplex patula* L.), marsh elder (*Iva frutescens* L.), groundsel bush (*Baccharis halimifolia* L.), and giant (or common) reed (*Phragmites australis* (Cav.) Trin. ex Steud.). Giant cordgrass (*Spartina cynosuroides* L.), narrow leaved cattail (*Typha angustifolia* L.), three-square (*Scirpus* L. spp.), rose mallow (*Hibiscus mosheutos* L.),

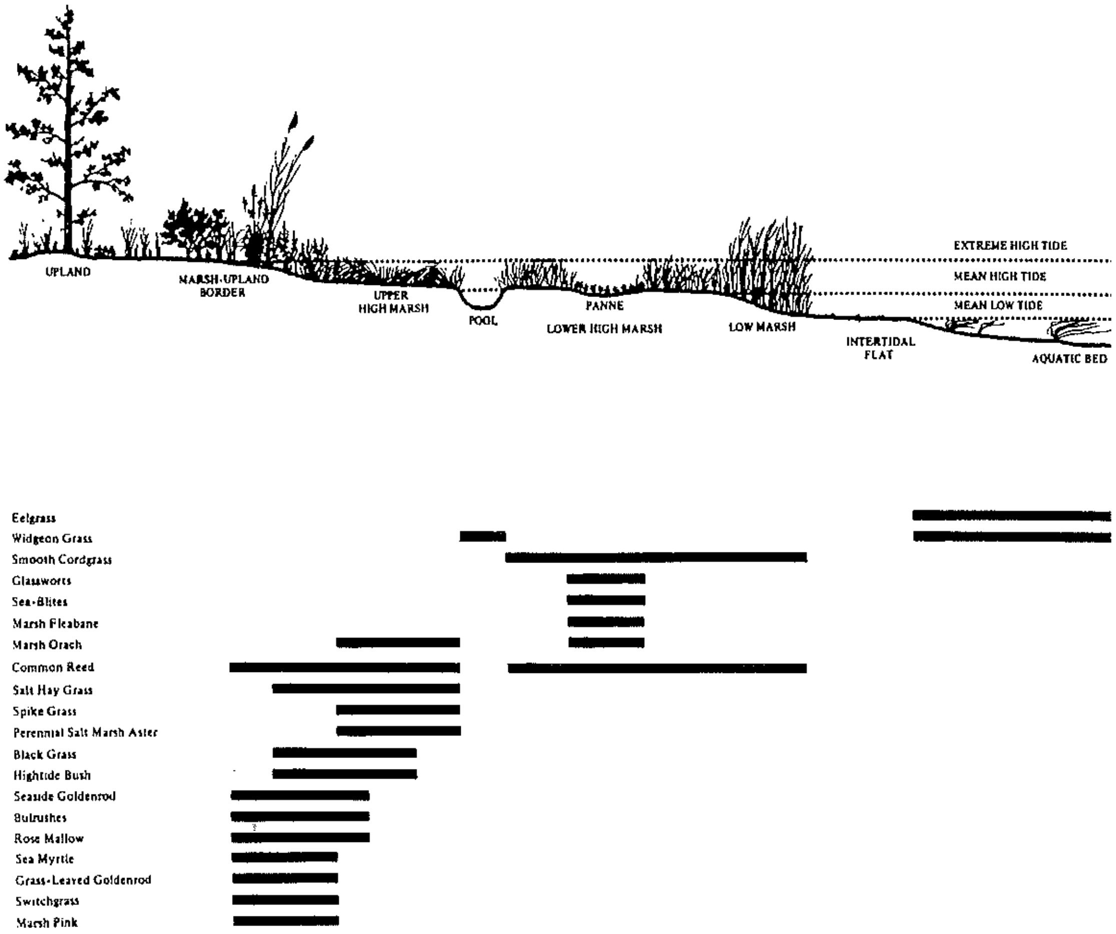


Figure 3.1 Generalized distribution of vegetation in Delaware's tidal salt marshes (after Tiner, 1985; Titus, 1988).

and water hemp (*Amaranthus canabinus* L.) are common in moderate to low salinity marshes.

In keeping with the concept that tidal wetlands possess distinctive vegetation types forming a pattern of gradients across the marsh surface, a classification system has been established by Daiber *et al.* (1976) for the tidal marshes of Delaware. This classification system is based on the associations of marsh vegetative zones, as salt marsh vegetation is an accurate and specific reflection of the many complex factors acting in marsh areas. Vegetative zones established and mapped in surface and subsurface environments for this study closely follow the floral zones influenced by saline tidal waters, as established by Daiber *et al.* (1976). Beginning at the low tide level, Zone I is dominated by smooth cordgrass, *S. alterniflora*. Moving upward along the gradient, Zone II is dominated by salt meadow hay, *S. patens*, and spike grass, *D. spicata*, which together comprise the high marsh "salt hay" association. Moving landward, Zone III is dominated by marsh elder (high-tide bush), *Iva frutescens*, and groundsel bush, *Baccharis halimifolia*, as well as by salt hay. Zone IV is dominated by giant reed, *Phragmites australis*.

Evolution and Development of Coastal Tidal Salt Marshes

Marsh Formation and Development

As previously described, tidal salt marshes are dynamic coastal features shaped by the interaction of water, sediments, and vegetation. Salt marsh development is due to the interaction of several variables: 1) the rise of sea level, which is a long term process which ultimately shapes and controls the evolution of the coast and ultimately the salt marsh; 2) tides, which provide not only energy for introduction of sediments into the system, but also a mechanism for dispersal of sediment over the marsh surface; 3) accumulation of organic and inorganic sediments, which ultimately controls the existence of the salt marsh; and 4) the growth of halophytic vegetation.

Many investigations, theories, and studies on marsh development and evolution have been described in the literature. For a comprehensive review on literature related to wetlands and sea-level rise, the reader is referred to Gehrels and Leatherman (1989) which reviews the history of marsh studies in light of the changing views and approaches of coastal scientists regarding the role of sea-level rise in marsh development. A brief description of coastal marsh studies relating to marsh evolution and development follows.

Historically, three major processes of marsh development have been described: 1) accumulation of marine sediments within protected bays and lagoons (Redfield and Rubin, 1962); 2) submergence of upland areas in direct response to the long-term rise of sea level (Redfield, 1972); and 3) accumulation of fluvial sediments associated with the formation of a river delta system (Gosselink, 1980).

There has been much discussion on processes of genesis and physical evolution of wetlands under the influence of rising sea levels and subsiding land surfaces, with the concept that wetlands development is a product of sea-level rise evolving over a period of hundreds of years. The early theory that wetlands were drowned uplands resulting from land subsidence evolved in the mid 1800s; Cook (1856) and Mudge (1858) characterized land subsidence as the primary factor controlling both development and erosion of coastal wetlands. These studies provided a background for Shaler's (1885) theory of mud deposition and vegetative colonization over tidal flats which, for 75 years, was the widely accepted theory of wetlands development and evolution. Through the first half of the twentieth century, the scientific community accepted either land subsidence or mud deposition followed by wetlands vegetative colonization as the principle causes for wetlands development.

Although Davis (1910) explained the process of marsh accretion as being driven by post-glacial sea-level rise, a conceptual model of marsh function in an environment of rising sea level was not developed until 1972 (Redfield, 1972). As a result of an extensive analysis of Barnstable marsh, Redfield (1972) demonstrated that marshes accreted both vertically and laterally in response to sea-level rise, incorporating the detailed relationships between vegetation, tide, sedimentation, and rising (relative) sea level (Figure 3.2).

Wetlands and Sea-Level Rise

It is generally accepted that long-term accumulation of tidal salt marsh sediments is dependent upon the relationship of the marsh surface to local relative sea level; in order for a marsh to exist and accumulate sediments, it must be spatially and temporally tied to sea level (Bloom, 1964; Chapman, 1974; Belknap and Kraft, 1977). Holocene marsh deposits in Delaware have been documented to be several meters thick, lying stratigraphically between backbarrier lagoonal deposits and underlying pre-

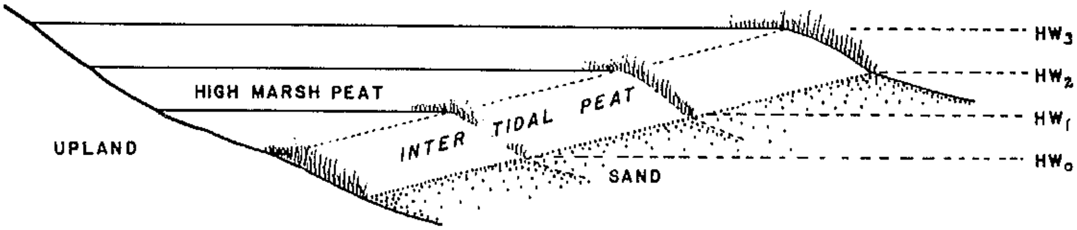


Figure 3.2 Theoretical structure of salt marsh which is spreading over accumulating sediment on a sand flat and over the upland in the course of rising sea level (from Redfield, 1972).

Holocene surfaces (Belknap and Kraft, 1977). The initiation, growth and development of these Holocene marshes are attributed to the slowing of Holocene sea-level rise over the past 7,000 years (Kraft *et al.*, 1987).

Many investigations into the evolution and development of salt marshes have demonstrated that they have the ability to accrete vertically in response to moderate rates of sea level rise (1-10 mm/yr) (Redfield, 1972; McCaffrey and Thomson, 1980; Allen and Rae, 1988; French, 1994). In addition, other studies (Pethick, 1980; Gehrels and Leatherman, 1989; Streif, 1989; French, 1994) have suggested that sea-level rise is a critical and important factor in marsh initiation, development, and maintenance. There are, however, widely varying figures regarding the present rates of vertical accretion in salt marshes and the associated local rates of relative sea-level rise, as described by DeLaune *et al.* (1983, 1992), Orson *et al.* (1985), Stevenson *et al.* (1986, 1988), Walker *et al.* (1987) and French (1993, 1994). Attention has been focused on "accretionary deficits" which often result in submergence and erosion of marsh areas (Gagliano *et al.*, 1981; Baumann *et al.*, 1984; Stevenson *et al.*, 1986, 1988; Penland *et al.*, 1990; French, 1994), along with estimates of sediment accumulation rates that are (or would be) required for maintenance of marsh elevations relative to projected future sea-level rise (National Research Council, 1987; Warrick and Oerlemans, 1990; French, 1994).

Complex relationships between vertical accretion and sea-level rise ultimately affect and control marsh surface level adjustments to the possibility of accelerated sea-level rise. The potential influence of a change in mean sea level on flooding frequency and soil salinity in the salt marsh would likely result in changes in vegetative growth characteristics and possible changes in vegetative zonation patterns. Because the salinity gradient in marsh soil is a function of inundation frequency and duration, a change in mean sea level will ultimately affect not only waterlogging conditions of the sediment, but also soil salinity and associated vegetation type.

The relative change in mean sea level is the local rise or fall of sea level due to both eustatic sea level change and local subsidence or uplift of the coast line. The gradual rise in relative sea level along the coast of Delaware is resulting in landward migration of intertidal wetlands, whereas a rapid rise, as is happening along the coast of Louisiana, may cause significant loss of intertidal wetlands (DeLaune *et al.*, 1987; Morris *et al.*, 1990).

Wetlands Formation and Sea-Level Rise in Coastal Delaware

Tidal marshes of Delaware are relatively recent landforms. During the worldwide Holocene marine transgression, valley mouths were inundated to form estuaries (Delaware Bay) into which rivers flowed, and coastal lowlands were submerged to form broader embayments (Bird, 1993). Spits and barriers of sand have built up across these embayments to form coastal lagoons such as Delaware's Inland Bays. These coastal lagoons are generally shallow, and are linked directly or indirectly to the open sea by one or more tidal entrances (e.g., Indian River Inlet; Roosevelt Inlet; Ocean City Inlet). They are typically estuarine, with a salinity gradient increasing from the mouths of in-flowing rivers and streams to seawater at tidal entrances. These inland bays have generally been reduced in depth and area by accumulation of sediment washed in from rivers and from the sea, organic deposits such as peat, and possible swamp encroachment over the past 6000 years.

Geologic research (Belknap and Kraft, 1985; Stedman, 1990; Fletcher *et al.*, 1990) on Delaware's marshes shows that while the coastal marsh lithosome in the paralic region developed as early as 10-11,000 years ago (Kraft and Belknap, 1986), most modern coastal marsh environments initially developed 3,000 to 4,000 years ago (Belknap and Kraft, 1977, 1985; Kraft and Belknap, 1986). This period of expansive marsh development along Delaware's coasts was coincident with the mid-Holocene reduction in the rate of sea level rise, when the rate of relative sea level rise in Delaware was approximately 0.21 cm/yr. As described in the model of marsh formation developed by Redfield and Rubin (1962), emerging and developing salt marshes undergo periods of expansion in response to rising sea level, ultimately resulting in accumulation and vegetative colonization of marine sediments, and submergence of adjacent upland environments. As sea level continues to rise, the marsh continues to expand in a landward direction. It is commonly assumed that the growth of the salt marsh will continue as long as rates of accretion (sediment accumulation) are equal to or greater than the rate of relative sea level rise (Orson *et al.*, 1985), and erosion of wetland fringes does not exceed landward expansion of wetlands (Phillips, 1986). An objective of this investigation is to determine if there are other controls on the ability of a wetland to migrate in a landward direction, especially antecedent topography and its control on hydroperiod.

Over the past 2,000 years, the rate of local relative sea level rise has averaged 0.12 cm/yr (Belknap and Kraft, 1977; Kraft and Belknap, 1986). During this period, the areal extent of Delaware's tidal wetlands was significantly reduced as the landward migration of other coastal environments continued. More recently, tide-gauge records along Delaware's coast indicate a relative sea level rise of 0.33 cm/year (Hicks and Hickman, 1988). This accelerated sea level rise over the last 100 years has apparently resulted in increased shoreline erosion of Delaware's tidal wetlands. In addition, greenhouse warming scenarios predict even more rapid increases in sea level (Hoffman *et al.*, 1983; Titus, 1988), with associated dramatic wetlands losses. The longevity of Delaware's coastal marshes depends upon the marshes' ability to keep pace with rising sea levels, not only in a vertical direction, but also horizontally. The lateral migration of wetlands environments and concurrent survival of coastal wetlands depends upon continued submergence of adjacent upland environments, and/or continued gradual conversion of brackish/freshwater marshes to salt marshes as sea level rises. If landward migration of wetlands environments is not occurring, continued existence of upper transitional zones and high marsh areas may be in jeopardy, while vertical accretion in the low marsh (*S. alterniflora*) zone may keep pace with continued sea level rise.

Tidal Wetland Zonation Patterns

Wetlands Zonation

The distribution of wetlands is dependent upon and at equilibrium with tidal-range, flooding frequency and duration, wave energy, sedimentation rates, local climate, and sea level fluctuations. The elevation of wetlands relative to mean sea level determines zonation and areal extent of tidal wetland habitat. Numerous researchers have surveyed the distribution of plants and species diversity within intertidal salt marshes throughout the United States (Teal 1958; Wilson 1962; Good 1965; Daiber, 1969; Reimold *et al.* 1975; Nixon 1982). These studies show that tidal salt marshes have specific vegetative zonation and distribution patterns, controlled by physical factors, including, but not limited to: 1) salinity; 2) frequency and duration of tidal flooding; 3) topography, or elevation of the marsh surface; and 4) substrate characteristics, both sedimentary and chemical.

Tidal salt marshes in Delaware can be divided into several zones, based on elevation and general physiographic conditions. These lateral zonations are commonly referred to as: 1) low marsh, or regularly flooded wetlands; and 2) high marsh, or irregularly flooded wetlands; and 3) transitional, or rarely flooded wetlands. The low marsh areas extend from about mean sea level to the mean high tide level, and are alternately flooded by the tides and exposed subaerially at least once (usually twice) daily. In high and moderate salinities, the low marsh is dominated by the tall form of smooth cordgrass (*S. alterniflora*). The high marsh ranges from above the mean high water mark to the extreme high tide level. This zone is less frequently flooded by the tides, usually during the semi-monthly spring tides or during storm conditions, and is generally exposed to air. Composition of the flora will vary in high marsh zones depending on elevation relative to the tide, drainage characteristics of the substrate, and salinity; typical vegetation includes: the short form of smooth cordgrass (*S. alterniflora*), salt meadow hay (*S. patens*), and spike grass (*D. spicata*). Above the high marsh zone is a transitional zone between upland and wetland environments. This transitional zone is only inundated by extreme storm tides, and species common to this area include: switchgrass (*Panicum virgatum*), marsh elder (*Iva frutescens*), groundsel bush (*Baccharis halimifolia*), black needlerush (*Juncus gerardii*), and giant reed (*Phragmites australis*).

It is not the intent of this project to provide detailed species inventories of marsh zonation, but rather to utilize the existing lateral zonation and vegetative distributions to determine recent (last 100 years) changes in wetlands environments at the landward edge of the ongoing transgression. That is, to determine if there is a relationship between the existing (and expected) lateral distribution of surface wetlands "zones" and the underlying stratigraphic lithosomes directly adjacent to the antecedent "upland" surface that is being transgressed.

General Environmental Factors Affecting Zonation Patterns

Tidal salt marshes possess distinct ecological conditions which control vegetative community structure, distribution and zonation. These ecological conditions, which have been described in the literature (Cottam *et al.*, 1938; Chapman, 1960; Adams, 1963; Queen, 1973; Daiber, 1974), include interacting physical, chemical, and biological factors which control the vegetative distribution in a salt marsh: 1) likelihood of submergence; 2) type of substrate; 3) salinity of soil and water; 4) adaptation to mechanical action of wind, water, and sun; 5) soil chemistry, pH, and aeration; 6) depth of water table; and 7) soil nutrients. However, it is generally considered that tidal inundation and salinity are the primary factors in determining marsh plant distributions (Daiber, 1974).

The most important environmental factors affecting the wetlands/uplands boundary of Delaware's fringing salt marshes include: 1) natural factors such as tides, salinity, water table, surface level changes (vertical accretion, sediment input), and storms; and 2) human-induced changes such as mosquito ditching, mowing, grazing, agricultural impacts (land clearing activities), and development of adjacent uplands.

Natural Processes and Factors

Tides. Tidal influences are probably the most significant environmental factor responsible for vegetative zonation patterns in Delaware's fringing tidal wetlands. The wetland area dominated by *S. alterniflora* is subjected to regular flooding by saline waters twice a day. The extent of the marsh dominated by *S. patens*, *D. spicata*, *I. frutescens*, and *B. halimifolia* vegetation is flooded less often, and may be flooded completely only during the highest spring tides. The high upland margin of *P. virgatum* and terrestrial vegetation is flooded only during extreme storm events.

Salinity and Water Table. The shoot systems of tidal salt marsh plants are submerged in saline water at every flooding, while the root systems are in saline solutions of varying concentrations, often less than those of the flooded shoots. Subsoil seepage of fresh water from the uplands may create a brackish environment in what would otherwise be an area of high salinity due to surface tidal flows. Many marsh species grow more luxuriantly in the less-saline situation, and this implies that their abundance in the marshes is an example of an "emergence into dominance" where relieved of the competition of other less salt-tolerant species.

Extreme salinities, up to 110 ppt, may be found on certain occasions in the pannes. The water table below the unflooded marsh may fluctuate only slightly. Its salinity fluctuates scarcely at all, and its level varies from 5 to 40 cm below the surface. These extremes are possibly important in altering the vegetation, with effects that persist for many years afterwards.

Surface Level Changes. Past studies and information obtained in this study imply that surface-level changes are an active process in the salt marsh. These changes are likely related to four processes: 1) normal up-building of the peat level, possibly in correlation with the rise in relative sea level; 2) soil erosion, assumed to result locally from either excessive mowing or grazing; 3) deposition, as known to occur on natural levees along ditches and tidal creeks; and 4) compaction of the marsh peat, a phenomenon known to occur in wetland areas (Kaye and Barghoorn, 1964; Pizzuto and Schwendt, in review). However, in the limited area of the wetlands/uplands boundary, where thickness of marsh sediments do not exceed 2 m, compaction is not likely a significant factor.

Storms. Hurricanes and northeasters may have important impacts on tidal marshes, especially at the wetlands/uplands border, although the direct impact of such storms on the herbaceous marshes themselves is not readily apparent. At the wetlands/uplands border, destructive effects of storms include physical breakage of terrestrial vegetation (saplings, trees), killing by salt spray and salt water inundation and/or intrusion into the groundwater, and smothering by storm debris or wrack.

Human Impacts

Removal of Surface Vegetation: Mowing, Grazing, and Burning.

Mowing on some of Delaware's fringing tidal marshes has apparently occurred continuously since colonial times, and has likely affected the nature of the vegetation. Although mowing has not generally been considered a significant vegetational factor in most of the previous literature of Atlantic Coast marshes, there is a strong probability that prolonged mowing of the high marsh zones may reduce the vitality of predominant high marsh species (*S. patens/D. spicata*), eventually resulting in bare soil and subsequent erosion. Similarly, grazing and burning effects are likely important to the ecology of the wetlands/uplands boundary area. Stachecki (personal communication) reports that many *S. patens/D. spicata* marshes were used as grazing areas for cattle and sheep in Delaware Bay and Inland Bay marshes; relict fence posts in high marshes provide evidence of previous grazing activities in many areas surveyed for this report (see photograph, Figure 3.3). Florschutz (1959) reports that some natural and ditched salt marshes have been used as cattle pasture or hay cropping, and that others have been burned late in the fall to induce sprout growth as a winter food for wild geese. However, historical data is typically anecdotal and not readily available for documentation of these impacts.

Mosquito Ditching. As part of a state-wide effort to control mosquito populations, extensive areas of salt marsh were ditched in the early part of this century. As a result of these extensive modifications, essentially all salt marshes in Delaware have been affected in some way by mosquito ditching. Direct effects of mosquito ditching include: conspicuous margin effects that are related to deposition of spoil removed from ditches, the bare soil on sides of the ditches, and natural levees that develop on each side of the ditch. These natural levees likely alter the surface drainage conditions of the marsh, and may produce associated vegetative changes in ditched areas. In some areas, spoil deposition of sediment along the sides of the ditches is so uniform and extreme that small but affective dikes are formed which effect drainage in the area. Indirect effects include: draining pools, increasing tidal flooding into higher sections of the marsh, altering range and distribution of organisms that may impact wetland vegetation, and altering suspended sediment input into wetlands.

Biological Adaptations

Tidal salt marsh plants exhibit various biochemical, morphological, and physiological adaptations which permit them to grow in periodically flooded saline soils

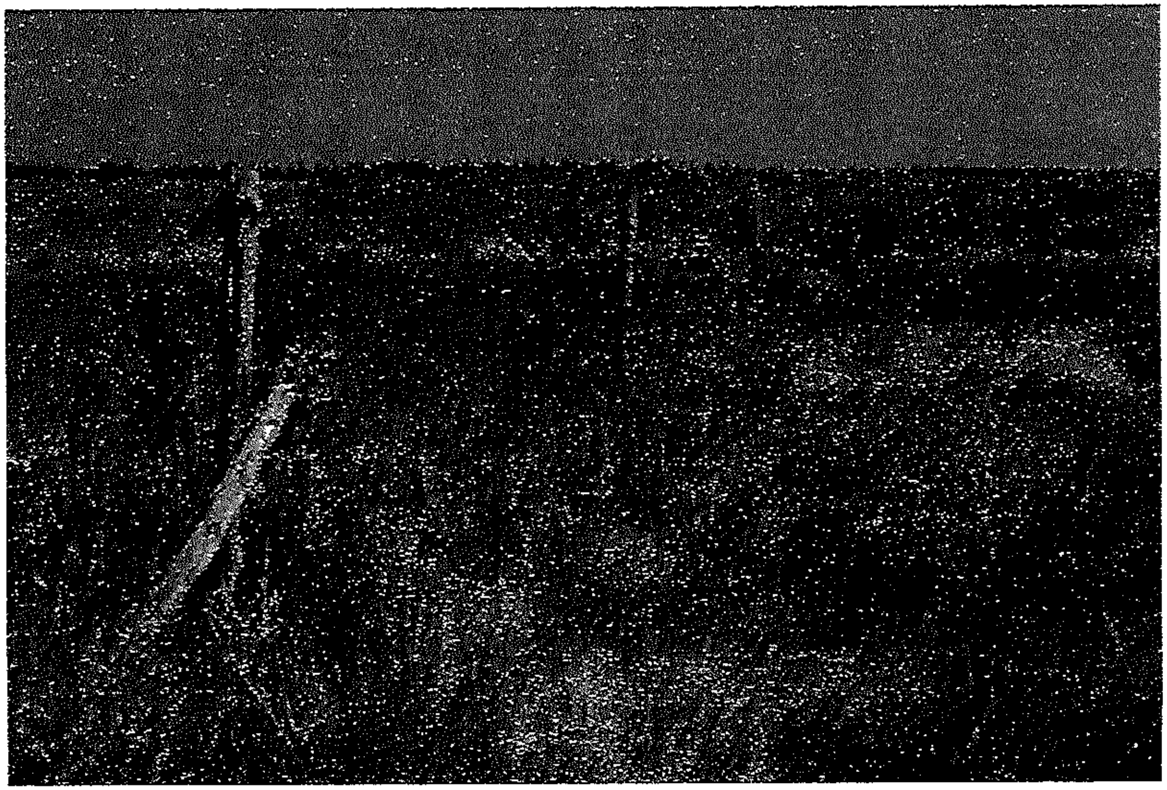


Figure 3.3 Photograph of old fence posts in *S. patens* marsh, possibly indicative of previous grazing by domesticated animals.

(Armstrong, 1979; Hook, 1984; Naidoo *et al.*, 1992). The roots of wetland plants, growing in waterlogged soils, must be able to survive anaerobic and chemically reduced conditions resulting from the rapid depletion of oxygen by microbial oxidation of organic matter as well as root respiration processes (Turner and Patrick, 1968; Gambrell and Patrick, 1978). Wetland plants are able to oxygenate the rhizosphere (Figure 3.4), allowing them to survive the low Eh (reducing conditions) of salt marsh soils (Armstrong, 1979). Thus, survival of wetlands plants in these stressful conditions is dependent upon the efficiency of oxygen transport from aerial parts of the plants to the roots via aerenchyma tissue (Armstrong, 1979; Burdick and Mendelssohn, 1990; Naidoo *et al.*, 1992). Formation of aerenchyma tissue in wetland species not only reduces the amount of respiring tissue relative to root volume, but also facilitates oxygen diffusion to the living tissues and rhizosphere where reduced soil components are detoxified (Williams and Barber, 1961; Armstrong, 1967; Mendelssohn and Postek, 1982; Burdick, 1989). Productivity of wetland species has been positively correlated with substrate redox potential (Eh) (Linthurst, 1979; DeLaune *et al.*, 1984; Bandyopadhyay *et al.*, 1993), as the plants switch from direct transport of oxygen from shoot to root for maintenance of aerobic conditions (Armstrong, 1979) to anaerobic pathways (DeLaune *et al.*, 1984).

In addition to waterlogging stresses, coastal salt marsh plants are subjected to salinity stress. Growth of coastal marsh plants is also affected by the salinity of the rhizosphere (Nestler, 1977; Valiela *et al.*, 1978) which has been found to adversely affect net photosynthesis (Pezeshki *et al.*, 1987), exert toxic effects on the plants, and

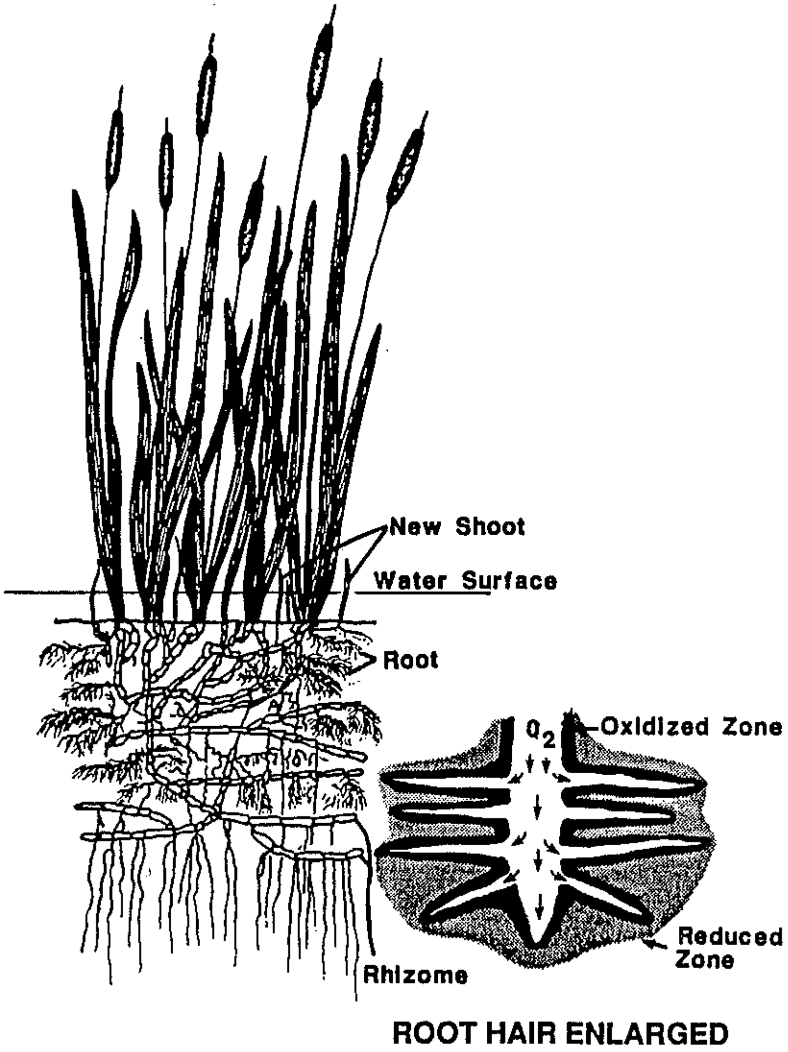


Figure 3.4 Wetland plants have the unique ability to transport oxygen to support their roots growing in anaerobic substrates (from Mitsch and Gosselink, 1994).

interfere with water and nutrient uptake processes (Linthurst, 1979; DeLaune *et al.*, 1983).

The combined effects of waterlogging and salinity on plant growth and metabolism are poorly understood (van der Moezel *et al.*, 1989), but it is generally acknowledged that aeration and salinity are two of the major factors influencing growth and distribution of *Spartina* species in salt marshes (Nestler, 1977; Smart and Barko, 1980; Valiela *et al.*, 1978). Anatomical and metabolic responses to waterlogging and salinity relate to the surface vegetative distribution controlled by environmental and physical factors at the wetlands/uplands boundary of tidal salt marshes.

Naidoo *et al.* (1992) compared the metabolic and anatomical response of *S. alterniflora* and *S. patens*, which differ in salinity and flooding tolerance, to different levels of waterlogging and salinity stressors. While both species are adapted to waterlogging and salinity, *S. alterniflora* appears to be more tolerant of reducing soil conditions and less responsive to higher salinity than *S. patens* (Naidoo *et al.*, 1992). The effects of waterlogging on *S. patens* included negligible production of new, positively gravitropic roots, while older *S. patens* roots were grayish to blackish in color, and generally in poor condition (Naidoo *et al.*, 1992). Alternatively, under the same conditions, the flooded *S. alterniflora* plants produced a profuse mass of new roots, whitish to brownish in color, viable, and all positively gravitropic (Naidoo *et al.*, 1992).

Burdick (1989) examined root aerenchyma development in *S. patens* in response to flooding, and his results suggest that "the ability of *S. patens* to develop aerenchyma in response to flooding may be important in maintaining flood tolerance and competitive ability in habitats at various positions on a soil waterlogging gradient" (Burdick, 1989, p. 780).

Waterlogging of soils has been shown to cause severe mortality of *S. alterniflora* in stagnant and unaerated soil/water conditions (Linthurst, 1979). Those plants which did survive extreme waterlogging conditions were significantly shorter, less dense, and had significantly less biomass than those grown in aerated systems. However, *S. alterniflora* has developed adaptational mechanisms, both morphologic and metabolic, to allow growth under hypoxic conditions. Aerenchyma tissue of the stem can provide oxygen from the atmosphere and increased alcohol dehydrogenase (ADH) activity can support greater fermentation rates in submerged *S. alterniflora* roots (Armstrong, 1979; Mendelssohn *et al.*, 1981).

Hellings (1990) determined that under salinity conditions of 0 ppt and 30 ppt, increased flooding resulted in higher recoverable underground reserves in *S. alterniflora* plants. Flooding of *S. alterniflora* resulted in stimulated growth and subsequent regrowth of photosynthate translocated to overwintering rhizomes. Although this holds true at both 0 and 30 ppt, the plant grows optimally in freshwater conditions. Thus, raising water levels in less-saline habitats for promotion of *S. alterniflora* growth should result in a healthy, dense, productive marsh; the best conditions for *S. alterniflora* growth involved flooding of substrate at surface levels, especially in marshes where salinity of retained water is 30 ppt or less. In more saline areas, increased flooding conditions resulted in less evident advantage. The metabolic and anatomical response of *S. patens* and *S. alterniflora* plants to waterlogging and salinity stress may provide an explanation for zonation of these species in the salt marsh (Naidoo *et al.*, 1992).

Vertical Accretion in Salt Marshes

Sediment Accretion in Wetlands

Accumulation of sediment is crucial to initiating development of and maintaining the marsh environment. Sediment supplied to marshes maintains the surface elevation of wetlands relative to surrounding waters, and provides essential nutrients to wetland vegetation. Primary sources of sediment in wetlands include: 1) sands, silts, clays, and organic material transported from adjacent water bodies onto the marsh surface by tides, waves, and storm surges; 2) sediment and organics transported downslope into the wetlands from erosion of the adjacent upland; and 3) in situ deposition of organic material from grasses and organisms growing on the wetland. Vertical accretion rate of sediments in salt marshes have been used to provide estimates of long-term rates of coastal submergence and/or the ability of a wetland to keep pace with relative sea level rise. Studies have focused on short-term fluxes in marsh accretion using marker horizons and similar techniques (Stumpf, 1983; Khalequzzaman, 1989; Kraft *et al.*, 1989; 1992). However, results from these short-term flux studies may be influenced by storm events and other factors, including continuous compaction which lowers the marsh surface relative to sea level (Kearney and Ward, 1986). The scientific literature describes variable, yet ubiquitous and continuous, rates of vertical accretion in Holocene tidal salt marshes in the eastern United States, with many of these rates tied closely to the continual rise in sea level over the last 7000 years.

Vertical Accretion, Sea-Level Rise, and Salt Marshes

Because estuarine coastal marshes occupy the transition zone between terrestrial and marine ecosystems, they are extremely susceptible to shoreline erosion and submergence by rising sea level. Continued growth, development, and maintenance of these estuarine fringing wetlands will depend on both landward migration and/or vertical growth in response to sea-level rise.

Vertical accretion in salt marshes occurs through accumulation of inorganic sediments, generally sands, silts and clays, and organic sediments consisting primarily of plant roots and rhizomes. The rate of vertical accretion is critical in maintaining marsh elevation relative to sea-level rise. In areas where sediment accumulation rates are low, marshes may be inundated and die back. The wetlands of the Mississippi River Delta are examples of marsh loss due to insufficient sediment supply relative to local subsidence and sea-level rise (Nyman *et al.*, 1990). In contrast, many estuarine marshes of the Atlantic coast do not receive large inputs of sediment and therefore are dependent on accumulation of organic matter to support their growth (Stevenson *et al.*, 1986; Bricker-Urso *et al.*, 1989; Craft *et al.*, 1993).

In order for salt marshes to maintain themselves, accretion rates must at least equal the rate of sea-level rise (Thom, 1992). The relationship between sea-level rise and salt marsh accretion has been studied in many marshes of the East and Gulf coasts of the United States. Various techniques have been utilized, including sediment horizon markers such as sand (Stoddart *et al.*, 1989), radioactive isotopes including ^{210}Pb (Armentano and Woodwell, 1975) and ^{137}Cs (DeLaune *et al.*, 1978). Most studies have determined that salt marshes have the capacity to keep pace with a rate of sea-level rise on the order of that recorded over the past century (Orson *et al.*, 1985; Kearney and Ward, 1986; Sharma *et al.*, 1987; Bricker-Urso *et al.*, 1989). However, other studies have demonstrated that marshes would disappear under greater rates of relative sea-level rise (Hatton *et al.*, 1983; Stevenson *et al.*, 1986; Leatherman *et al.*, 1995).

Accretion in coastal salt marshes is largely controlled by the amount of sediment deposition and organic matter accumulation, which is ultimately controlled by the frequency, depth, and duration of tidal flooding (Gosselink and Turner, 1978; Stevenson *et al.*, 1986). Craft *et al.* (1993) report that several studies have observed that accumulation of organic matter in marsh soils is greater in marshes that are irregularly flooded by spring and storm tides as compared to regularly flooded marshes (Gosselink and Turner, 1978; Craft *et al.*, 1988). They also note that sediment deposition is generally lower in wetlands that are irregularly flooded (Gosselink and Turner, 1978; Craft *et al.*, 1993).

Salt marsh vegetation has played a significant role in the shaping of intertidal landforms along the coastlines of embayments, estuaries, and lagoons (Bird, 1993). The growth of these plant communities has been facilitated where a large tidal range produces a broad intertidal zone, and where there has been an abundant supply of sediment, especially silt and clay, along with peaty residues derived from the plants themselves. In temperate regions, salt marsh vegetation has trapped sediment and marshes continue to grow where sediment is being supplied at a sufficient rate for their continued upward, seaward, and landward development. In a landward direction, a transition occurs from salt to brackish to freshwater vegetation such as reeds and rushes, swamp scrub and forests, in a zonation resulting from plant succession as accreting sediment builds the substrate up to and above high-tide level. A rising adjacent upland is marked by the transition to terrigenous vegetation (Bird, 1993).

This pattern of vegetational succession is the result of vertical accretion during the Holocene marine transgression. There is often stratigraphic evidence of changes in vegetation related to the Holocene history of submergence followed by a period of relative still stand. On coasts that are still subsiding such as the southern New England coastline, salt marshes are found where they are aggrading to keep pace with the relative rise of sea level (Redfield, 1972; Bird, 1993). Salt marshes are being drowned and eroded in areas where rates of vertical accretion are less than the rate of relative sea-level rise, and they are developing and spreading in areas where rates of vertical accretion exceed the rate of sea-level rise (Orson *et al.*, 1985).

In general, a global sea level rise will impede salt marsh progradation and ultimately result in erosion along the seaward or bayward margins of the marshes (Phillips, 1986). As marine submergence proceeds, the landward margins of the salt marsh can be expected to transgress over the adjacent upland surface at a rate related to the transverse gradient; salt marsh vegetation will move landward to displace freshwater or terrigenous vegetation communities on low lying upland surfaces. Around the submerging shores of Delaware Bay and Delaware's Inland Bays, salt marsh plants are invading the adjacent low-lying uplands, and there are sites where the upland forest is dying as salt marshes encroach. Similar effects have been documented in Maryland (Darmody and Foss, 1979; Stevenson *et al.*, 1986).

The landward migration of salt marsh communities will be impeded where the adjacent upland surface rises steeply. In these situations, the upper intertidal zone narrows significantly or disappears as sea level rises. The same effect is observed where embankments (rip-rap, bulkheads, dikes, berms) have been established at the upper margins of a salt marsh in order to reclaim land for agricultural, recreational, residential, or commercial purposes. Where sedimentation is sufficient to maintain vertical accretion, the existing salt marshes will persist as sea level rises (Pethick, 1981); however, maintenance of the areal extent of the marsh surface will only be possible where adjacent upland slopes are sufficiently low and/or undeveloped.

Although the rate of vertical accretion in wetlands is crucial to maintenance of wetland surface elevation with rising sea level, horizontal growth or expansion of the marsh may be equally significant to maintenance of distinct wetland vegetative zones (e.g., maintenance of high marsh zones as well as low marsh environments). The longevity of salt marsh environments in coastal Delaware may ultimately depend upon bidirectional expansion, with vertical accretion accompanied by marsh encroachment of adjacent uplands (Redfield and Rubin, 1962; Orson *et al.*, 1985; Phillips, 1986). If the slope of the adjacent upland and antecedent topography is too steep, vertical accretion of salt marsh sediments will not necessarily result in concurrent lateral expansion or landward migration of salt marsh environments. Stratigraphic relationships and geometry of the Holocene/pre-Holocene surface must also be factored into predictions of survivability of wetlands, and interpretations of both long-term and short-term impacts of rising sea level.

The Salt Marsh Lithosome

Identification of Holocene Marsh Sediments: Previous Work

Allen (1974) discusses the many physical, chemical, and biological methods that have been used to describe peat deposits in Holocene sediments. These methods include: color, texture, organic content, water content, sediment content, salinity content, and identification of plant fragments. Generally, indirect methods of determining peat characteristics have been largely based on general textural characteristics such as sediment content and color, while direct methods of plant fragment identification are only possible when preservation is excellent.

Davis (1910) noted that fresh or brackish water peats are dark, almost black in color when exposed to air; salt marsh peats are dark to medium brown in color (Allen, 1974). Bloom (1964; Bloom and Stuiver, 1963) distinguished fresh and brackish water peats from salt marsh peats by color, texture, and organic content. Bloom (1964) observed that salt marsh peats consisted of a mass of fibrous roots with inorganic sediment forming a coherent brown layer. The texture of fresh to brackish water peats was much like "well-rotted sawdust" and contained more organic material (Bloom, 1964; Allen, 1974).

Redfield (1965b) characterized high and low marsh peats on the basis of water content. Low marsh peats were found to contain less water than high marsh peats. McCormick (1969) identified high marsh and low marsh peats in Holocene material by textural means. A fine textured, or low marsh peat contained a low amount of sediment. A coarse textured or high marsh peat contained a high amount of sediment, whose source was attributed to upland runoff. Elliott (1972) observed in a Delaware marsh system, that sedimentation by tidal processes contributed more sediment to the marsh than sedimentation by upland runoff. It seems apparent that a criterion such as sedimentological texture can be used to describe or identify peats only if the role and dominance of all the sediment sources can be established (Allen, 1974). Meyerson (1972) defined "peat fabric" as the percent organic material and used this as one criterion to distinguish peat deposits. This is based on the same criterion as McCormick (1969) and Elliott (1972); "peat fabric" simply measures the vegetation component instead of the mineralogic component (Allen, 1974).

Recently, more detailed investigations of Holocene freshwater peat deposits (Whallon, 1989; Pizzuto and Rogers, 1992) have demonstrated that the organic content of wetland sediments can be used for identification of depositional subenvironments. Riverine, estuarine, and palustrine wetland systems were distinguished from each other by loss-on-ignition analyses (Whallon, 1989; Pizzuto and Rogers, 1992).

Allen (1974) reported that many investigators have used plant fragments in peats as a method of plant species identification. Others have cited the shape of organic remains as criteria for identification (McCormick, 1969; Newman and Munsart, 1968;

Elliott, 1972; and Meyerson, 1972). Caution must be used in utilizing plant fragments to identify plant species in the subsurface. First, the plant fragments must be preserved well enough that distinctive and characteristic details are easily recognizable. Second, a thorough understanding of and familiarization with species anatomy and morphology is required. For example, the distinction between root, rhizome, stem, and leaf must be made; flat epidermal sheets from rhizomes and stems might easily be misidentified as fragments of leaves if the anatomy of the fragments is not studied (Allen, 1974). Hubbard and Stebbings (1968) identified plant fragments in peats during a study of the stratigraphy of the Keysworth marsh, Poole Harbor, England. They differentiated rhizome, root, stem, leaf, seed, and fruit of at least twenty different plant species and divided the sediment layers into zones according to the plant distribution. Similarly, Niering *et al.* (1977) identified subsurface plant fragments (especially rhizomes) to species level in salt marsh peats.

General Sedimentary Characteristics: The Holocene Marsh Lithosome

Salt marshes along the Delaware Bay coast have been studied by Kraft (1971), Elliott (1972), Allen (1974, 1977), Richter (1974), Khalequzzaman (1989), Whallon (1989), Stedman (1990), Pizzuto and Rogers (1992), John and Pizzuto (1995), and Yi (1992). Elliott (1972) and Stedman (1990) studied the Great Marsh north of Lewes, Delaware, defined sedimentary facies and environments of deposition, and presented a model for the development and maintenance of coastal marshes. Allen (1974) analyzed core data to describe the petrology and stratigraphy of coastal marshes, and to form paleogeographic reconstructions of the marshes at Lewes Creek and South Bowers. Richter (1974) determined a three-dimensional configuration of coastal marshes from core data along the Delaware Bay coast, while Stedman (1990) and Yi (1992) described detailed facies and paleoenvironments in subsurface sediments along the Delaware Bay coast. Both stratigraphy and paleogeographic reconstruction of marshes located in tidal streams and river valleys have been investigated by Whallon (1989), Pizzuto and Rogers (1992), and John (in progress).

The Holocene marsh mud lithosome in Delaware has been described in detail and characterized by Chrzastowski (1986) and Yi (1992). Chrzastowski (1986) described the marsh mud lithosome as "an organic-rich mud in which the organic material is primarily in-place remains of *Spartina* and *Distichlis* roots, stems, and blades, with additional possible remains of *Phragmites*, *Iva* and *Baccharis*" (page 71). Additionally, he determined that fine-grained inorganic sediments with high water content dominate the Holocene sequence of the lagoon-barrier systems in Rehoboth and Indian River Bays. Plant remains were found to be a dominant component of the sediment either as detrital (as in tidal stream and lagoonal mud) or in-place (as in marsh mud) fragments (Chrzastowski, 1986). Typically, abundance of organic material within the lithosome is variable; Elliott (1972, 1973) estimated marsh mud organic content for the Great Marsh, Lewes, Delaware, to range from 1% to 90%, and Chrzastowski (1986) reports similar variations in organic content. Only in rare cases (less than 10% of all marsh muds examined) did Chrzastowski (1986) identify beds with a density of plant remains to warrant classification as true peats (i.e., organic content 60% or greater).

Yi (1992) identified salt marsh microfacies in subsurface sediments based on: grain size (mud); color (variable); sedimentary structure (bioturbation by plant rootlets, rhizomes, and stems); stratigraphic position (primarily at top section of sedimentary sequences); foraminifera (primarily *Trochammina macrescens*, *T. inflata*, *T. squamata*, and *Miliammina fusca*); other microfossils (diatoms, mites); and macrofossils (crabs, snails). Other characteristics of Delaware Holocene salt marsh microfacies, as described by Yi (1992) include: mostly lower organic content than freshwater microfacies and brackish marsh microfacies; 30-50% organic content; water content variable; mostly dense mat of plant rootlets and rhizomes at top of the cores; and sometimes organic content decreasing upward. In addition, sub-microfacies or nannofacies of salt marsh microfacies were identified based on dominant plant species (e.g., *S. alterniflora*, *S. patens*, and *D. spicata*) (Yi, 1992).

Although these previous studies provide data on general stratigraphic characteristics and relationships in Delaware's tidal wetlands, they have not specifically addressed the potential current and future impacts of sea-level rise on longevity of Delaware's estuarine marshes. This study is complementary to other studies, and also unique in that it focusses on marsh evolution and development as related to antecedent topography and local relative changes in water levels.

The subsurface stratigraphy of wetland environments directly adjacent to uplands should demonstrate the principle of Walther's Law, i.e., sedimentary sequences that occur laterally should be seen vertically in the same or similar conformable stratigraphic sequences. The expansion of low marsh over high marsh, over transitional wetlands, over upland surfaces should be clearly evident in cores. Thus, a record of process and rate of the ongoing transgression can be detailed through examination of subsurface stratigraphy. This can provide information regarding the leading edge of the transgression, the longevity and vegetative characteristics of remaining tidal wetlands, and a possible indication of the influence of hypothesized changes in rates of sea-level rise.

Chapter 4

SURFICIAL MORPHOLOGY OF THE WETLANDS/UPLANDS BOUNDARY, DELAWARE'S FRINGING TIDAL SALT MARSHES

Introduction

The purpose of this chapter is to document relative elevation and vegetative characteristics of the wetlands/uplands border of Delaware's tidal salt marshes, including the surface vegetative zonation and distribution, and inferred correlations with environmental factors. In addition, data are presented that contribute to the basic knowledge of Delaware's tidal salt marshes, particularly in regard to the segregation of wetland plant communities with respect to thickness of marsh sediments (or depth to the pre-transgressive surface), relative surface elevation, and adjacent upland slopes.

The effects of adjacent and antecedent topography are not typically considered as factors controlling vegetative zonation patterns in coastal salt marshes. For example, Redfield (1972) emphasized biotic control over vegetative composition, and modeled salt marsh development with specific sequences of vegetative zonation over time. As noted by Clark (1986), many previous studies of coastal vegetation have emphasized the inherent stability of salt marshes and the importance of autogenic succession as a driving mechanism for successional change. Although this study does not focus on ecological succession and plant community interrelationships, it does document that pre-existing topography and other physical factors such as substrate characteristics exert control on coastal vegetation patterns in Delaware's fringing tidal salt marshes. Successional patterns in the salt marsh vegetative community at the wetlands/uplands boundary may be initiated by pre-existing physiographic and topographic relationships and directed by fluctuations in the physical environment to which the vegetation responds.

The tidal salt marshes of the Delaware estuarine coast are distinctive and easily recognizable landforms. They are generally flat meadows at or below the level of the highest tides, bounded on the landward side by scrubby and forested uplands and by freshwater swamps and marshes, and bounded on the seaward/bayward side by an escarpment (0.1-0.5 m) or a bay-mouth sandbar. The wetlands substratum is generally an organic accumulation of fibrous peat, mixed with silt or sand.

General knowledge of local tidal-marsh vegetation along Delaware's coast is extensive, and has been summarized in papers by Daiber *et al.* (1976) and Tiner (1985). Similarly, general knowledge of the geologic history of tidal wetlands in Delaware has been documented by many research projects, including but not limited to: Elliott, (1972); Fletcher *et al.*, (1990, 1993); John and Pizzuto, (1995); Khalequzzaman, (1989); Kraft, (1971); Kraft *et al.*, (1987); Pizzuto and Rogers, (1992); Richter, (1974); Stedman, (1990); Stumpf, (1983); Whallon, (1989); Yi, (1992), and many works in progress.

Distinct ecological conditions control salt marsh vegetative community structure, distribution and zonation patterns. As previously discussed, these conditions include many interacting factors which exert control on the vegetation in a salt marsh (Cottam *et al.*, 1938; Chapman, 1960; Adams, 1963; Queen, 1973; Daiber, 1974).

These interacting physical, chemical, and biological factors influence initiation, evolution, and continuing development of Delaware's fringing tidal salt marshes; some are long-term, with constant, continuously acting impacts such as relative sea-level rise, normal sediment supply and tidal changes. These long-term processes are punctuated by more extreme and rapid changes in factors such as tidal range (possibly human-induced), or other more extreme fluctuations, such as dramatic changes in sediment input, or salinity changes due to evaporation of saline waters on the marsh surface. Data on the distribution of surface vegetative distribution patterns, relative elevations, and relationships to adjacent and antecedent topography contributes to our understanding of the evolution and development of salt marsh environments.

Methods

Sixty-nine wetland transects were surveyed to determine relative elevations of specific sections of the marsh at the wetlands/uplands boundary, to document characteristic vegetation at various elevations, and to obtain representative cross-sections of fringing marshes along the Delaware coast. In addition, five detailed elevation surveys were conducted at selected representative sites. From the elevation data, frequency of flooding was inferred based on tide range and site characteristics. Using this information it is also possible to infer and estimate the landward shift of wetland communities and possible net loss of wetland area associated with various scenarios of sea-level rise.

To determine how an increase in sea level will impact wetlands, an understanding of surficial morphology and plant distribution at the wetlands/uplands boundary zone is required. Four discrete zones were delineated based on relative elevation and vegetation type: uplands; transitional shrub wetlands; high marsh; and low marsh.

Surveying: Establishment of Transects

Transects were surveyed to establish: a) the topography or relative elevation of the marsh surface and related vegetative distribution; b) the slope of adjacent uplands; and c) the slope of underlying antecedent topography. An automatic level and rod were used to determine relative elevations for all transects. Each transect began at a benchmark established in an upland area (terrestrial vegetation dominant) adjacent to the wetland boundary, and continued bayward, perpendicular to the wetlands/uplands boundary. Transects extended 10 to 30 m into the *S. alterniflora* dominated marsh area. For each transect, elevation and distance from the upland benchmark were measured at 3- m intervals into the *S. alterniflora* dominated wetland environment. Elevations were also surveyed at breaks in slope or changes in plant species composition. A Total Station was used for detailed surveys of five transect sites.

Over 400 hand-driven Eijelkamp cores were obtained along the 69 survey transects. Depth to the pre-transgressive sand was measured as an indicator of marsh thickness at the wetlands/uplands boundary. These subsurface data were used to map the geometry of antecedent topography at representative points along the transects.

Wetlands Zonation Patterns

Survey data were used to map vegetational zonation patterns of wetlands surface environments. Lateral distribution of vegetational zonation and relative surface elevation of each vegetative zone (upland; transitional zone; high marsh; low marsh) were determined at each transect site.

Wetland zonation patterns were established based on plant identification and a qualitative estimate of species dominance; plant species composition was determined by estimating percentage cover at 3- m intervals along the elevation transect. Dominant plant species in Delaware tidal salt marshes and their associated wetland zone (as used in this study) include: 1) *Spartina alterniflora* (smooth cordgrass): low marsh zone; 2) *Spartina patens* (salt meadow hay)/*Distichlis spicata* (spike grass): high marsh zone; and 3) *Iva frutescens* (marsh elder)/*Baccharis halimifolia* (groundsel bush): shrub zone. Plant species were identified by examining characteristics of living plants, as described in the following paragraphs.

***Spartina alterniflora* (Smooth Cordgrass)**. *Spartina alterniflora* Loisel, commonly called smooth cordgrass, is the dominant plant species of salt marshes on the Atlantic coast, comprising approximately 90% of these marshes (Silverhorn, 1982). *S. alterniflora* grows from southern Canada to Florida and the Gulf states, and generally occurs in extensive stands in saline areas or mixed with *Juncus roemerianus* and other species in brackish areas (Eleuterius, 1980). There are two growth forms of this species: tall *S. alterniflora*, 1.0 to 2.1 m in height, and short *S. alterniflora*, 0.30 to 0.61 m. The former grows along the margins of tidal creeks, guts, natural channels and other areas subject to daily tidal flooding and the latter occupies higher levels of the marsh near the upper limit of daily tidal influence. The tall form produces a lone, narrow inflorescence in August and the short form seldom produces inflorescence. The primary means of propagation is via rhizomes and roots (Silverhorn, 1982).

***Spartina patens* (Salt Meadow Hay)**. *Spartina patens* (Ait.) Muhl. is a perennial rhizomatous halophytic grass of low to medium height (0.3 to 0.9 m), which is commonly found in high marsh areas in cowlicked mats from Quebec to Florida and Texas, and inland in New York and Michigan. *S. patens* leaves are narrow and linear (less than 0.5 cm wide and up to 0.5 m long), with margins rolled inwardly. Its stems are slender, stiff, and hollow. Open terminal inflorescence (panicle up to 20 cm long) is usually composed of 3-6 spikes alternately arranged and diverging from the main axis at 45-60 degree angles. Flowering period is late June into October. *S. patens* is commonly found in high marsh zones, or in areas of elevation that are generally flooded only by tides that exceed the mean high level (infrequently flooded salt and brackish marshes) (Daiber *et al.*, 1976). However, it has been reported to occur at times in frequently flooded marsh zones. *S. patens* is also found on wet beaches, sand dunes, and upper borders of salt marshes, and can be found in inland and saline areas. Daiber *et al.* (1976) report that *S. patens* is typically found in marsh areas where the marsh surface is formed of peat with less mud than in the smooth cordgrass zone, which has been confirmed in this study. Optimum soil type reported for *S. patens* is loose, sandy soil, although it can tolerate a wide range of soils, except very fine-grained soils such as clay, mud, and muck (Barnett and Crewz, 1990).

In cultivated plots, as reported by Somers (1979), *S. patens* grew well in well-drained soils watered at least three times per week with water of 25-30 ppt salinity. *S. patens* grass is propagated vegetatively and by seeds. The plant seeds profusely, and invades low moist sites readily (Woodhouse and Knutson, 1982).

***Distichlis spicata* (Spike Grass)**. *Distichlis spicata* (L.) Greene is a rhizomatous perennial, dioecious, C4 grass with a tolerance for high salinity, waterlogging and long periods of drought. The grass is erect and low growing (0.20 to 0.60 m tall), with stiff, hollow and round stems; creeping rhizomes form dense mats. Numerous linear leaves (2-3 mm wide) with shoot margins usually rolled inwardly and sheaths overlapping, are distinctly two-ranked. The terminal inflorescence bears one or two types of crowded spikelets on separate plants (dioecious); male spikelets have 8-12 flowers, female spikelets usually have 5 flowers. Flowering period is from August to October. *Distichlis* is typically found in salt marshes at elevations that are generally flooded only by tides that exceed the mean high level. It is often found intermixed with *S. patens*, or in pure stands. *Distichlis* grows slowly, and appears to prefer soils which do not remain waterlogged (Somers *et al.*, 1978). It is found from New Brunswick and Prince Edward Island, south to Florida and Texas and Mexico, and is found locally inland to Missouri. Globally, *Distichlis* is found from Canada to Argentina, Australia, and Sudan (Yensen *et al.*, 1985).

Optimum growth salinity ranges are reported to be 5-30 ppt. *D. spicata* can tolerate very high salinity (Somers, 1979), with *D. spicata* seeds planted into dredge-spoil plots germinating in presence of water of 30 ppt salinity (Somers *et al.*, 1978). Germination and growth proceeded satisfactorily on sandy soil when flooded 3 times weekly with water of 25-32 ppt (Somers, 1979). Wrona and Epstein (1980) collected *D. spicata* and grew plants in solution cultures with salinity increasing at a rate of 10% seawater per week until a level twice that of seawater was reached. Inland ecotypes died, but coastal ecotypes appeared healthy.

Rumer (1982) and Gallagher (1985) report on response of *Distichlis* to fluctuating salinities. Growth response at constant high salinity and in 2-day fluctuating salinity was approximately one-third that at constant low salinity. Yield of *Distichlis* of one-week fluctuating salinity treatment was two-thirds that of the yield in low salinity treatment. Thus, it appears that *Distichlis* may lack the ability to make rapid biochemical adjustments to short-term salinity fluctuations, resulting in poor growth.

Distichlis spreads vegetatively and by seeds; Woodhouse and Knutson (1982) report that transplanting success using sprigs has been poor, and initial growth slow. However, others (Barnett and Crewz, 1990) characterize *D. spicata* as having a rapid growth rate, with dense cover found in the second year at some sites.

***Iva frutescens* (Marsh elder)**. *Iva frutescens* L., a dicotyledon, is a bushy-branched woody shrub which grows to approximately 0.6 to 3.5 m in height. Leaves (4.5 to 9.0 cm long and 8 to 40 mm wide) are finely toothed, and location on the branches are opposite; however, those on flowering branches are sometimes alternate.

Blades are typically lanceolate, minutely gladular, with three main ascending veins. Flowers in heads are surrounded by bracts; the functional stamens and pistils are found in separate flowers but in the same heads. *I. frutescens* generally has more finely toothed leaves (with blades tapering towards both ends) than *Baccharis halimifolia*. *I. frutescens* is common in brackish or salt-water habitats, and is typically located in marsh/upland margins, on small hummocks in salt or brackish water marshes, and/or along ditches.

***Baccharis halimifolia* (Groundsel bush)**. *Baccharis halimifolia* L., also a dicotyledon woody plant, is a multi-branched shrub or tree, growing to 1 to 2 m in height. Although *B. halimifolia* is deciduous, its bare stems remain somewhat greenish in color throughout the winter. In contrast, the bare branches of *I. frutescens* have a whitish-grey appearance. Its elliptical to broadly obovate shaped leaves (6 cm long and 4 cm wide) are alternate, grayish green and glandular-punctate, with long blades characterized by few to several conspicuous leaf teeth. Compared to *I. frutescens*, leaves are usually coarsely toothed, mostly toward the apex. White flowers which bloom and ripen in late fall are found in heads surrounded by bracts; the species is dioecious, or unisexual, with sexes found on different plants. Fruits are cylindrical. Leaves are reported to be poisonous, and cattle have died after eating them. *B. halimifolia* is primarily a coastal species, ranging from Massachusetts to Florida, and areas of the Gulf Coast. *B. halimifolia* is commonly found along the edges of salt marshes, and in freshwater marshes, swales, along pond shores, fencerows, and old fields.

***Phragmites australis* (Giant Reed)**. *Phragmites australis* (Cav.) Trin. ex Steud. is a coarse rhizomatous perennial which typically grows to 4 m tall and forms extensive dense colonies. Broad, acutely tapering leaf blades are typically 50 cm in length and 5 cm wide. Characteristic inflorescence is a dense tawny to purplish panicle approximately 50 cm long and 20 cm wide. *P. australis*, a very aggressive plant, has long, creeping rhizomes which enable it to propagate vegetatively into areas very quickly. *P. australis* is common along the upper edges of tidal and non-tidal marshes, along pond margins, ditches, and disturbed habitats.

Surficial Morphology: Results and Discussion

Species Distribution, Abundance, and General Zonation Patterns

Introduction. Five vegetative zones were mapped according to species dominance: 1) terrestrial (upland) vegetation zone; 2) *I. frutescens/B. halimifolia* (shrub) zone; 3) *S. patens/D. spicata* (high marsh) zone; 4) *S. patens/D. spicata/S. alterniflora* mixed zone; and 5) *S. alterniflora* (low marsh) zone. The *I. frutescens/B. halimifolia* zone occurs in either a narrow or broad zone (depending upon adjacent upland slope) parallel to and bordering the adjacent upland. Similarly, the *S. patens/D. spicata* zone occurs in either a narrow or wide band (depending upon adjacent slope) in an area between the *S. alterniflora* and *I. frutescens/B. halimifolia* zones. *S. alterniflora* vegetation dominates the salt marsh on the bayward side of the *I. frutescens/B. halimifolia* and/or *S. patens/D. spicata* zone.

The most seaward community of vascular plants is composed primarily of *S. alterniflora*. On its landward edge, the *S. alterniflora* vegetative zone is either abruptly or gradationally succeeded by a band or zone dominated by *S. patens* and/or *D. spicata*. Gradational transition between vegetation types consists of a mixed zone where both *S. alterniflora* and *S. patens* appear to be equally distributed; a sharp between-zone contact is characteristic when little or no overlap between species is observed. Along the transect, the *S. patens/D. spicata* zone is replaced in the landward direction by the *I. frutescens/B. halimifolia* shrub zone, which is the most landward portion of the salt marsh community.

Terrestrial Upland Zone. Characteristic vegetation of upland forested areas adjacent to fringing tidal wetlands depends on elevation, slope, and soil type. In upland areas with sufficient elevation and well-drained soils (e.g., Evesboro, Sassafras, or Rumford soils), upland forests are primarily oak, with a mixture of hickory, holly, sassafras, and loblolly pine. In adjacent upland areas with lower elevations and poorly drained hydric soils (Pocomoke, Fallsington, or Johnston soils), hydrophytic vegetation dominates. This assemblage is dominated by sweet gum and maple, and includes facultative vegetation such as loblolly pine, American holly, greenbriar, poison ivy, arrow-wood, blueberry, myrtle, bayberry, and sweet pepper bush. Switchgrass (*Panicum virgatum)* and broomsedge (*Andropogon spp.)* are common herbaceous plants adjacent to fringing tidal salt marshes.

***I. frutescens/B. halimifolia* Shrub Zone**. The *I. frutescens/B. halimifolia* shrub community forms a distinct belt between the tidal marsh grasses and the upland zone. This zone is typically 2-3 m wide, although its width may vary from 1-15 m. The vegetation is usually restricted to the shrubs *I. frutescens* and *B. halimifolia*, although *S. patens* is commonly found growing beneath the shrubs where enough light penetrates to the soil surface. The substratum of the *I. frutescens/B. halimifolia* shrub zone is typically sand intermixed with or overlain by an organic "soupy" material 5-10 cm deep. Due to its higher topographic position relative to other marsh areas, the shrub zone is inundated by salt water only during extreme events such as major storms.

***S. patens/D. spicata* High Marsh Zone**. The *S. patens/D. spicata* high marsh zone typically occurs landward of the *S. alterniflora* low marsh zone, yet bayward/seaward of the upland and shrub zones. Its areal extent and general shape vary depending on local conditions and factors, but it usually develops in a band parallel to the wetlands/uplands interface. The *S. patens/D. spicata* community is readily separable on the seaward side from the *S. alterniflora* community, while the transition to the upland is more gradational; *S. patens* is often observed in the understory of terrestrial upland vegetation. The *S. patens/D. spicata* zone is essentially a pure grassland, with few other species interspersed. *I. frutescens* and *B. halimifolia* are occasionally observed in the high marsh zone, but they generally only occur as seedlings; these shrubs rarely mature into the taller shrub form except where *S. patens* is thinned out by shading.

S. patens and *D. spicata* form a closed, stable community, with a dense cover of grass which mats down heavily into cowlick formations. The substrate is generally flat and firm, and very stable for foot traffic. However, in many *S. patens/D.*

spicata zones, the flatness of the high marsh substrate may abruptly change to an extremely hummocky surface. The relief in these hummocky areas is approximately 10 to 20 cm; in most cases, the hummocky topography occurs at the bayward edge of the high marsh zone, and coincides with the transition to *S. alterniflora*. The tops of the hummocks are vegetated by *S. patens/D. spicata*, while the lower elevations between hummocks are either barren, or colonized by *S. alterniflora*. A possible explanation for these hummocky features is that high tide levels may be increasing, resulting in erosion of the substrate, or because of inundation and bioturbation and differential lowering of the substrate surface.

S. alterniflora Low Marsh Zone. The *S. alterniflora* community occurs on the bayward edge of the wetlands/uplands boundary. The land surface of the low marsh zone is typically lower than other wetland zones, although parts are not flooded at every high tide. This vegetative community is readily distinguished from the bare mud flats on the bayward/seaward side and the fine grassy and/or shrubby vegetation on the landward side. The *S. alterniflora* low marsh zone is a monospecific stand of coarse halophytic grasses; however, other species, including *Salicornia* and *Limonium* are commonly interspersed.

Species Distribution: Relative Elevation

General. Relative elevation measurements were made at 3- m intervals along all transects from the terrestrial upland, through the shrub and high marsh zones, and into the low marsh *S. alterniflora* zone. Because this investigation focuses on the wetlands/uplands boundary, elevation surveys did not extend to the seaward/bayward edge of the *S. alterniflora* zone. Elevation data were recorded 20-30 m past the point at which *S. alterniflora* vegetation became dominant.

Plant species composition changed with increasing relative elevation, although the range of relative elevation was small and at times there was overlap of species distribution (e.g., *S. patens* typically overlapped with the *I. frutescens/B. halimifolia* zone; *S. patens/D. spicata* occasionally overlapped with the *S. alterniflora* zone).

General Transect Site Results. Results of surveys of 69 individual transects are included in Appendix A as cross-sectional elevation profiles. Data presented in each cross-section include relative elevation of the wetland surface, depth to a sandy substratum (antecedent topography), slope of the adjacent upland, and width of the *S. patens/D. spicata* (high marsh) zone. The demarcation between upland terrestrial zones plant species and salt-tolerant species is often abrupt because of changes in surface slope. However, in many cases there is an overlap of upland and salt marsh species, as *S. patens* typically was found growing beneath upland tree canopies as it encroaches into the uplands. In cases where species overlap occurred, dominant species were used to demarcate each zone; these are areas where slopes are almost uniform between the upland and wetland surfaces. In all cases, the wetland transect consisted of a series of low-relief steps between areas of generally uniform elevation; i.e., the leading edge of the transgression, including *P. australis* and *I. frutescens/B. halimifolia* (shrub zone) was generally found at elevations above the *S. patens/D.*

spicata (high marsh zone), which was always at elevations greater than the *S. alterniflora* dominated (low marsh zone) wetland areas.

Selected Transect Site Results. Selected representative cross-sections demonstrate the characteristics of the wetlands/uplands boundary and its surficial morphology in Delaware's salt marshes.

Bay Vista-1. A relative elevation profile for the Bay Vista-1 site, northern portion of Rehoboth Bay, is depicted in Figure 4.1. Steeply sloping upland areas are characterized by terrestrial trees and shrubs such as loblolly pine, American holly, sassafras, and white oak. At the base of the upland slope the wetlands/uplands boundary area is dominated by poison ivy, *P. australis* vegetation and remains of dead or dying *I. frutescens* plants. Bayward, there is a sharp transition to a *S. alterniflora* low marsh community. The boundary between *S. alterniflora* and *P. australis* is marked by a 4 cm decrease in relative surface elevation. The *S. patens/D. spicata* high marsh zone is absent at this site.

Glade-1 . A relative elevation profile for the Glade-1 site, located along a river axis tributary to the Lewes and Rehoboth Canal, is depicted in Figure 4.2. The upper sections of the steeply sloping upland are dominated by loblolly pine, white and red oak, cherry, and sumac; the lower elevations are characterized by red maple, sweet bay magnolia, and tupelo. *P. australis* dominates the area where the upland intersects the marsh. Channelward of the *P. australis* zone, the elevation drops by 3-6 cm and *S. alterniflora* is the dominant vegetation. The *S. patens/D. spicata* high marsh zone is again absent in this wetland plant community.

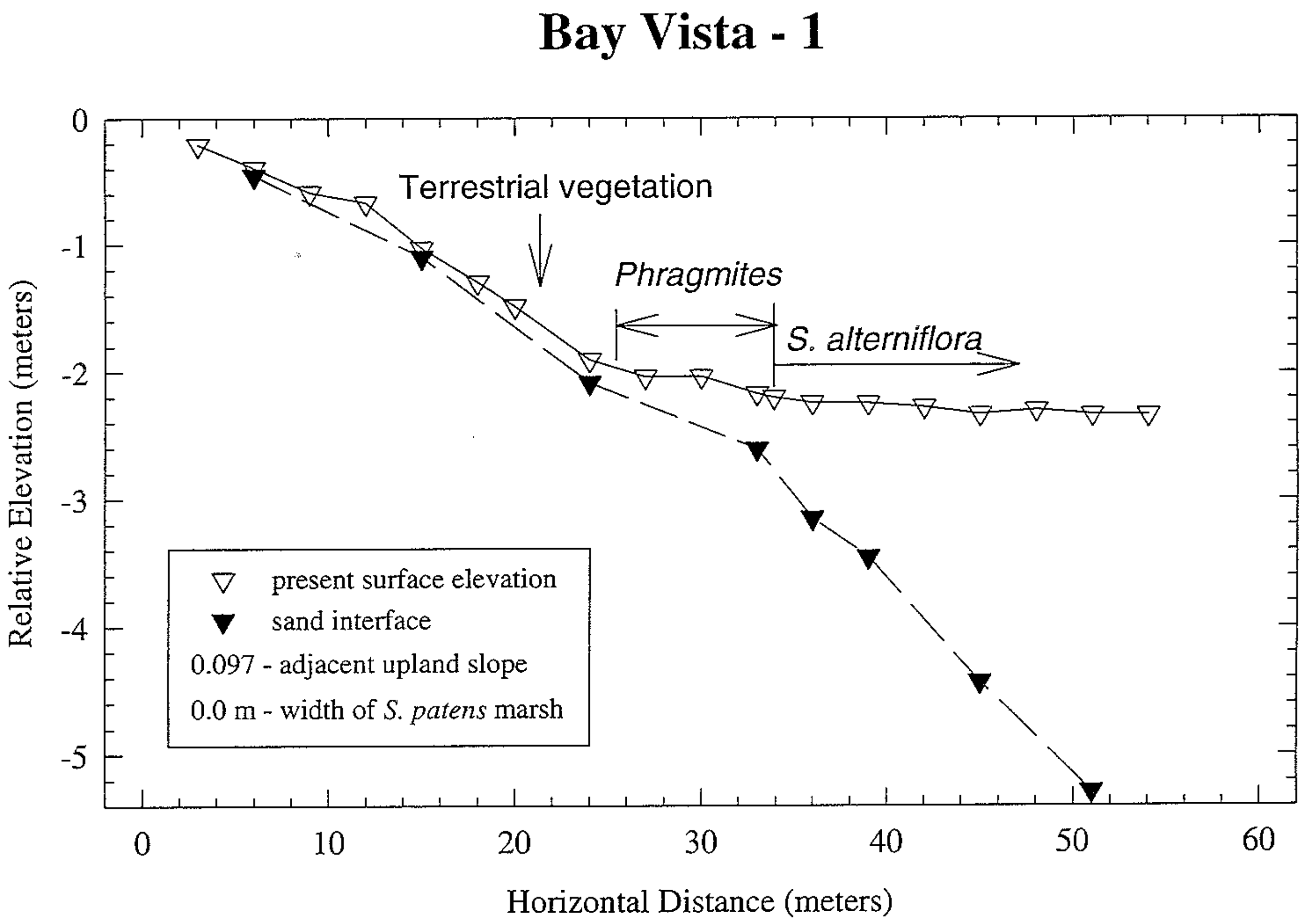


Figure 4.1 Relative elevation profile of Bay Vista-1 site, northern Rehoboth Bay.

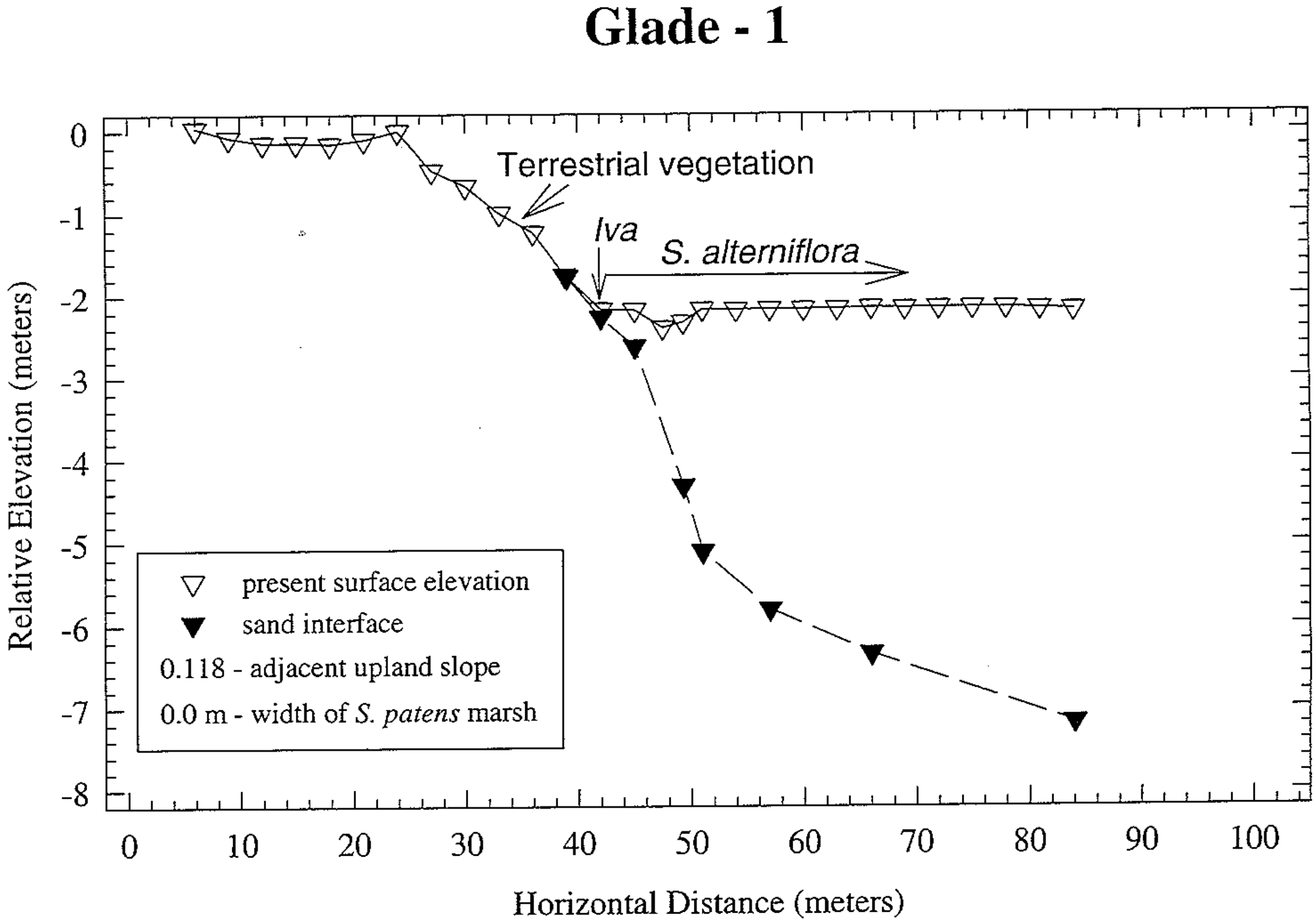


Figure 4.2 Relative elevation profile of Glade-1 site, Lewes and Rehoboth Canal.

Camp Arrowhead-2. Figure 4.3 depicts the relative elevation cross-section for the Camp Arrowhead-2 site, located on the western shore of Rehoboth Bay. In contrast to the two study sites described above, this area is characterized by a gently sloping wooded upland which is vegetated by loblolly pine, cherry, American holly, scrub oak, and bayberry. *I. frutescens, B. halimifolia* and *S. patens* are colonizing areas that were previously dominated by upland trees, as is evidenced by dead tree stumps. At slightly lower elevations (mean elevation difference of approximately 6 cm), *S. patens/D. spicata* (high marsh) becomes the dominant vegetative community. The transition between high and low marsh is gradational with a mixed zone of *S. patens/D. spicata* and *S. alterniflora*. *S. alterniflora* (low marsh) vegetation characterizes the most channelward portion of the marsh, at elevations approximately 2-3 cm lower than the high marsh zone.

Ritter-4. A relative elevation cross-sectional profile of the Ritter-4 site, located along the southwestern shore of Delaware Bay, is shown in Figure 4.4. This site is characterized by a gently sloping wooded upland (greenbriar, bayberry, loblolly pine, holly, and poison ivy), with a gradational transition to *I. frutescens/B. halimifolia* and *S. patens* salt marsh vegetation. The high marsh *S. patens/D. spicata* zone is approximately 12 m wide at this site, with a sharp transition to *S. alterniflora*-dominated low marsh in a channelward direction. There is an approximate 4 cm difference in relative elevation between the high marsh and low marsh plant communities at the wetlands/uplands boundary at this site.

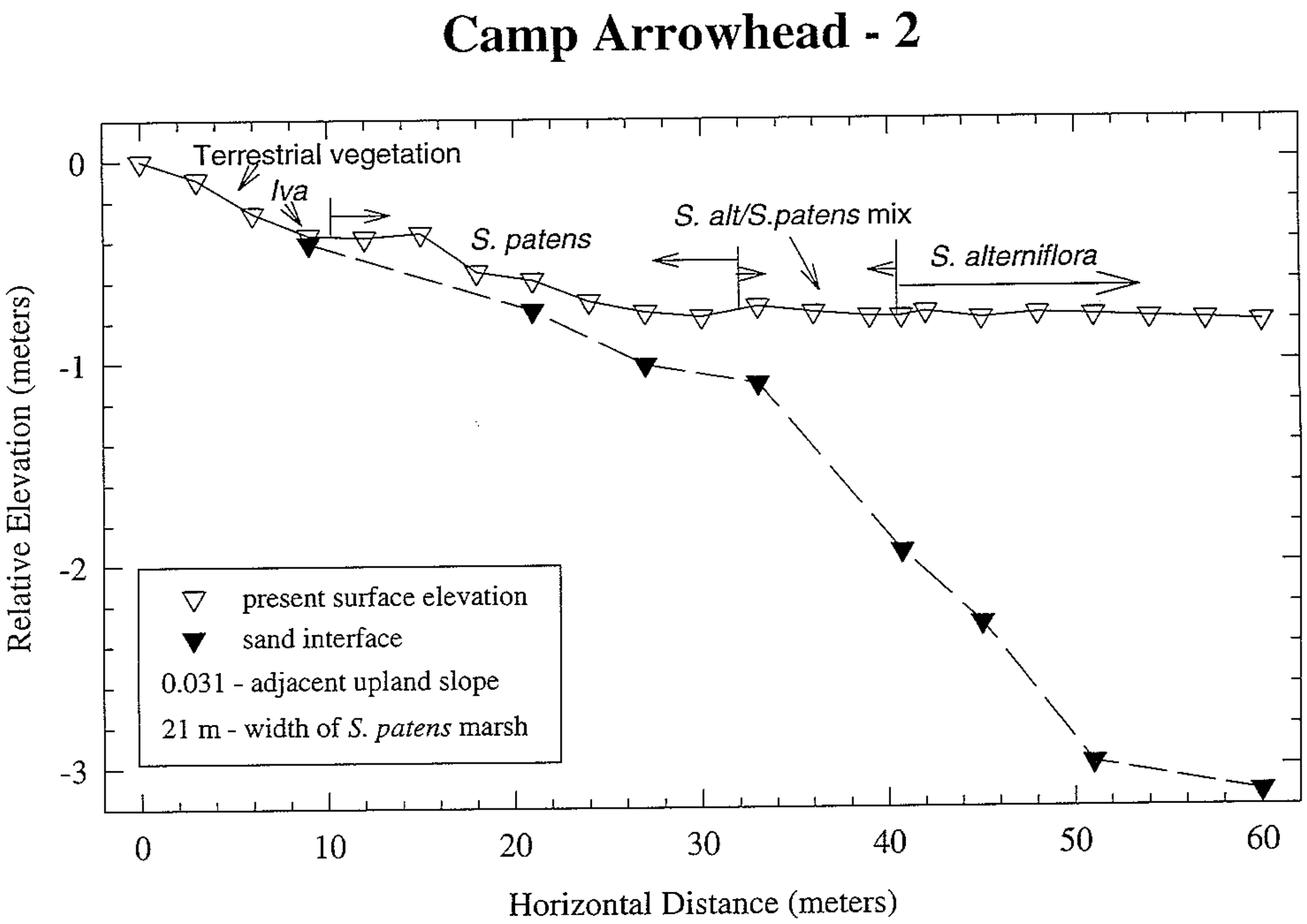


Figure 4.3 Relative elevation profile of Camp Arrowhead-2 site, western Rehoboth Bay.

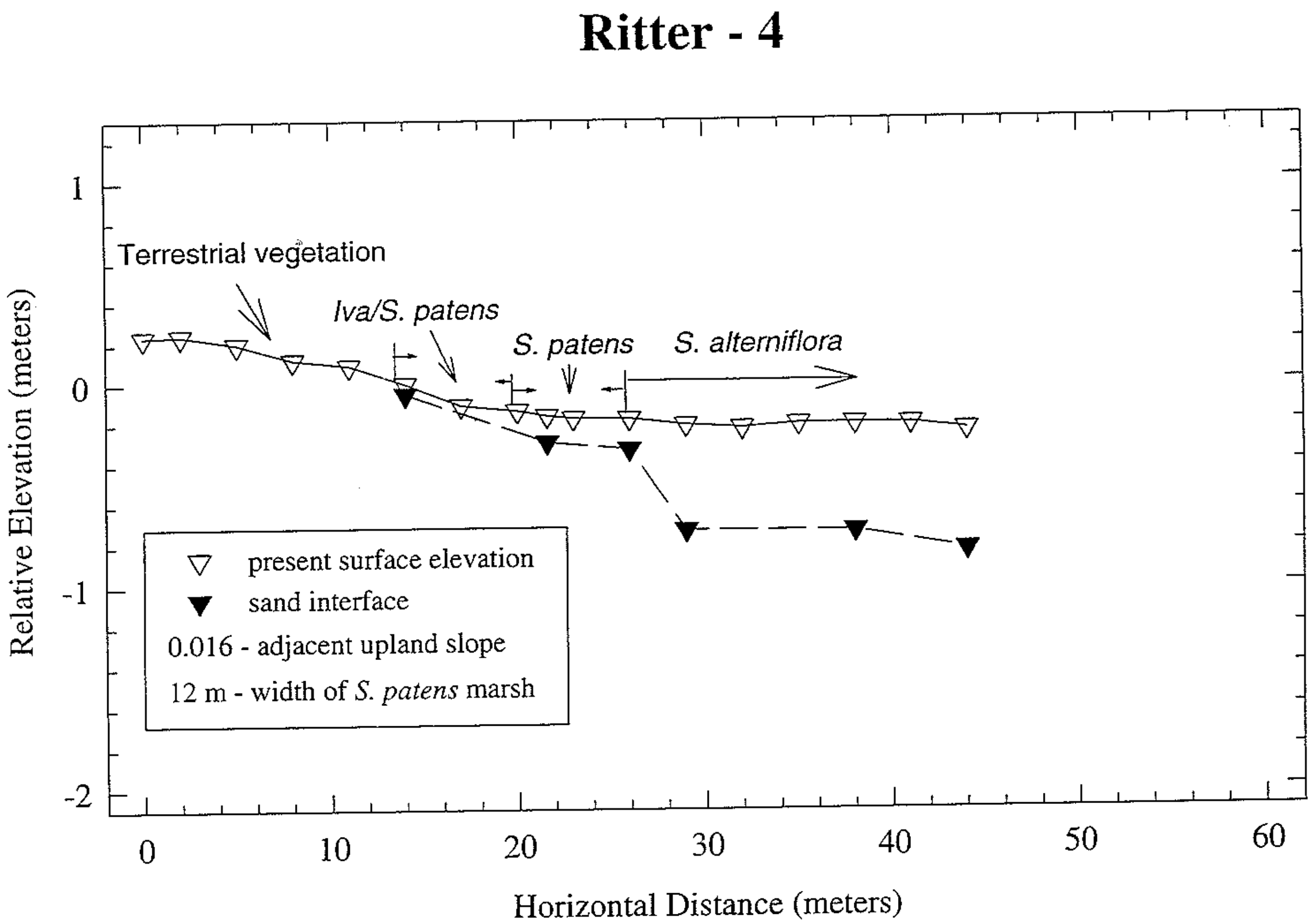


Figure 4.4 Relative elevation profile of Ritter-4 site, southwestern Delaware Bay.

Mean Relative Elevation of *S. alterniflora* and *S. patens* Vegetative Zones

Differences in mean relative elevation of the *S. patens/D. spicata* high marsh and *S. alterniflora* low marsh zones are presented in Appendix B. The transition between these plant communities typically occurs over a mean elevation difference of 17 ± 1.4 cm (range = 2 to 46 cm; standard deviation = 11 cm). The differences in mean relative elevation between *S. patens* and *S. alterniflora* for all sites are summarized in Table 4.1, and presented as a frequency distribution in Figure 4.5.

Table 4.1 Summary of differences in mean relative elevation between *S. patens/D. spicata* (high marsh zones) and *S. alterniflora* (low marsh zones) at individual transect locations.

Difference in mean relative surface elevation (m): *S. alterniflora* < *S. patens*	
Mean	0.17
Standard Deviation	0.11
Standard Error	0.01
Sample Size	53
Maximum	0.46
Minimum	0.02

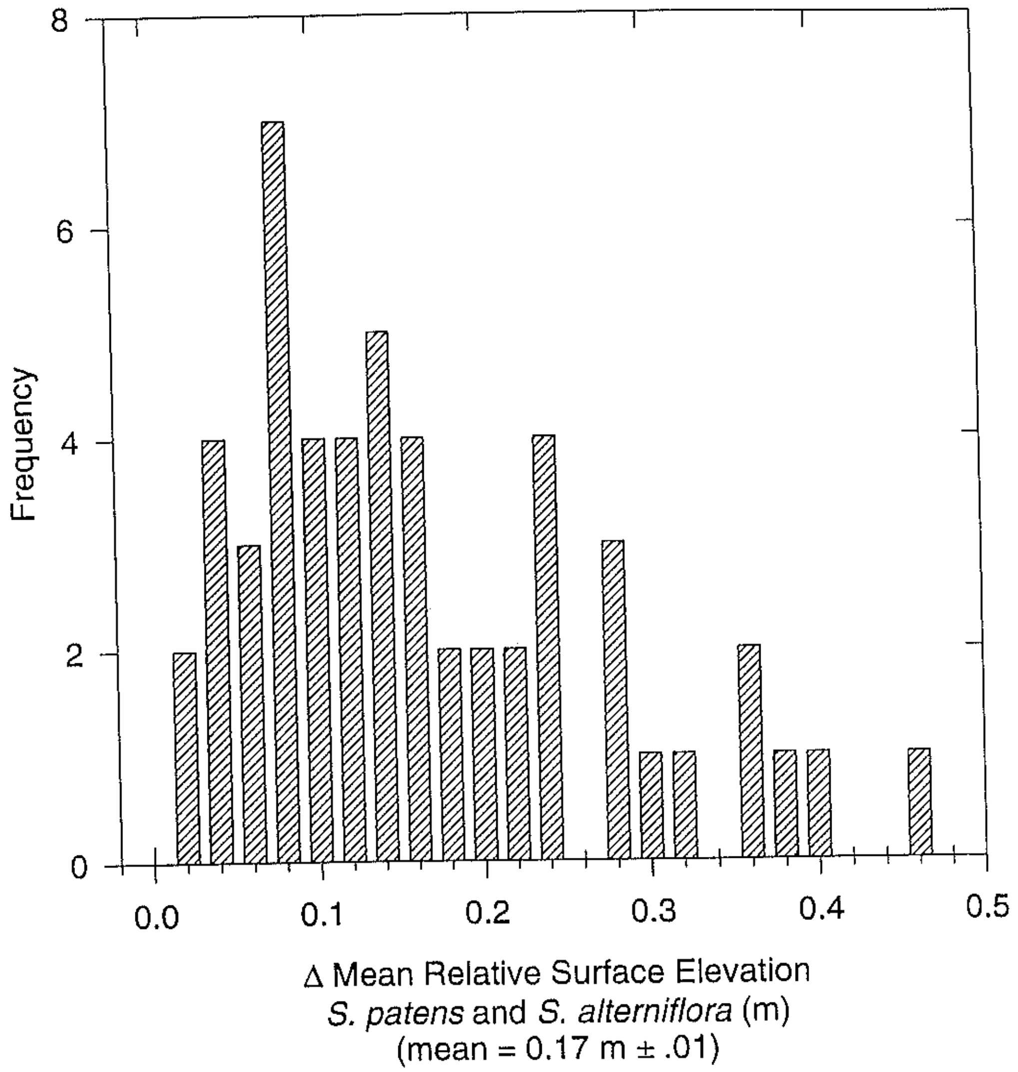


Figure 4.5 Histogram of mean relative elevation differences between *S. patens* and *S. alterniflora* vegetative zones at individual transect locations.

Detailed Survey Sites: Relative Elevation

Laser theodolite surveys provide more precise relative elevation data for five study sites: Boat House Cove (Indian River Bay), Delaware Wildlands and Pot Nets North (Rehoboth Bay), and Oyster Neck Road and Prime Hook Radio Station (Delaware Bay).

At the Boat House Cove (BHC) site (Table 4.2), the difference in mean elevation for the *S. alterniflora* surfaces vs. the *I. frutescens* surface is 51 cm (Figure 4.6). Note the absence of the *S. patens/D. spicata* (high marsh) zone at this location.

Table 4.2 Boat House Cove (BHC) site, northern Indian River Bay. Difference in mean relative elevation: *S. alterniflora*, *I. frutescens*.

BHC Site Elevation Data	Relative Elevation (m)	Relative Elevation (m)	Relative Elevation (m)
	S. alterniflora	*S. patens*	*I. frutescens*
Mean (± s.e.)	-2.18 ± 0.009	n/a	-1.67 ± 0.03
Std. Deviation	0.04	n/a	0.15
Maximum	-2.09	n/a	-1.33
Minimum	-2.26	n/a	-1.90
Sample Size	21	n/a	20

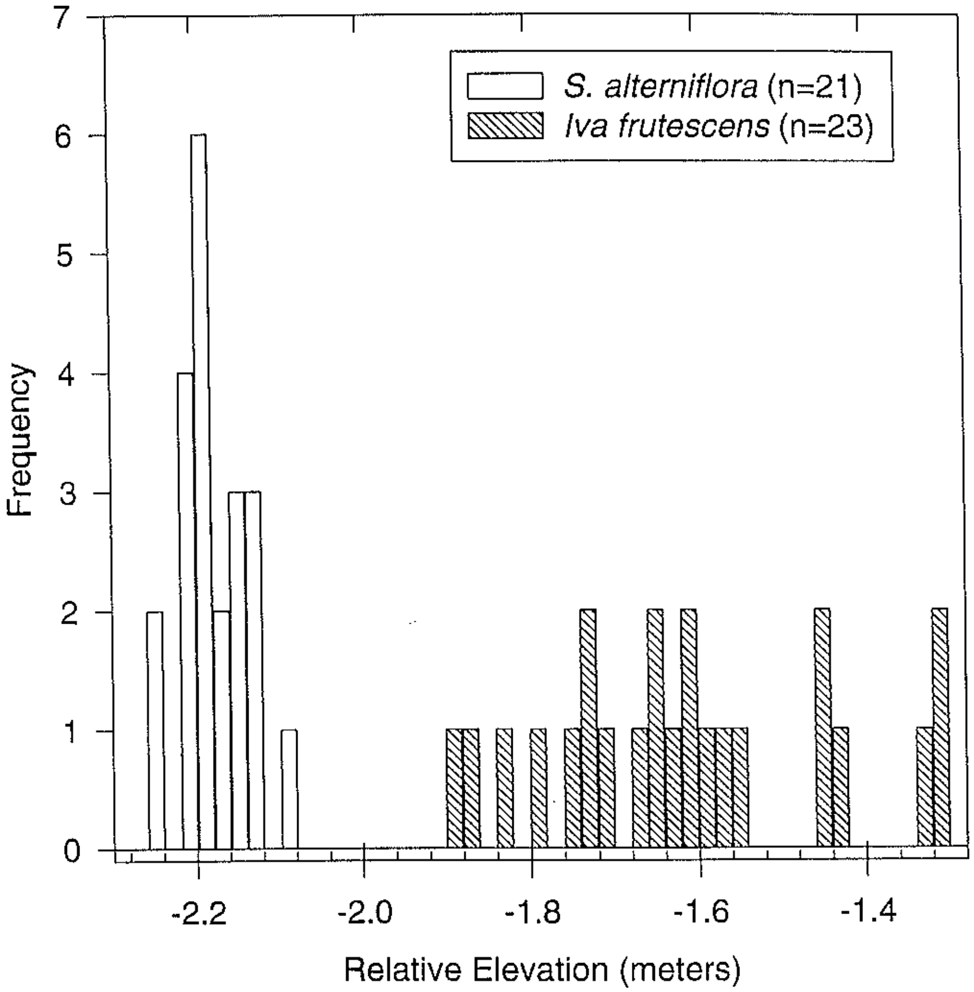


Figure 4.6 Histogram of mean relative elevation differences between vegetative zones, Boat House Cove (BHC) site.

At the Delaware Wildland (DEWL) site (Table 4.3), the difference in mean elevation between the *S. alterniflora* and the *S. patens* surface is 13 cm (Figure 4.7).

Table 4.3 Delaware Wildlands (DEWL) site, western Rehoboth Bay. Difference in mean relative elevation: *S. alterniflora*, *S. patens*.

DEWL Site Elevation Data	Relative Elevation (m)	Relative Elevation (m)
	S. alterniflora	*S. patens*
Mean (± s.e.)	-0.46 ± 0.006	-0.33 ± 0.03
Std. Deviation	0.03	0.17
Maximum	-0.36	0.26
Minimum	-0.49	-0.45
Sample Size	21	26

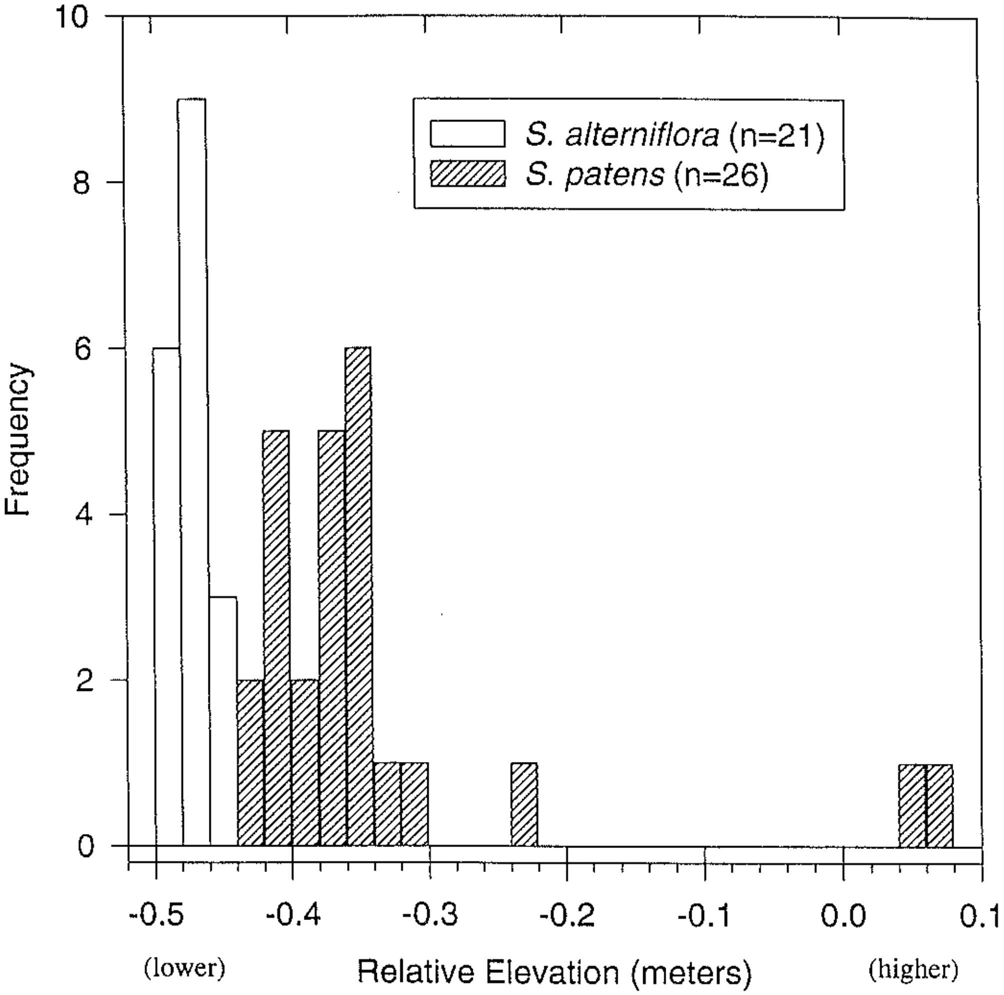


Figure 4.7 Histogram of mean relative elevation differences between vegetative zones, Delaware Wildlands (DEWL) site.

At the Pot Nets North (PNN) site (Table 4.4), the difference in mean elevation between *S. alterniflora* and *S. patens* vegetated sites is 7.3 cm, while the difference between mean elevations of *S. alterniflora* and *S. patens/I. frutescens* sites combined is 17 cm (Figure 4.8).

Table 4.4 Pot Nets North (PNN) site, southwestern Rehoboth Bay. Difference in mean relative elevation: *S. alterniflora*, *S. patens*.

PNN Site Elevation Data	Relative Elevation (m)	Relative Elevation (m)	Relative Elevation (m)
	S. alterniflora	*S. patens*	*I. frutescens*
Mean (± s.e.)	-0.201 ± 0.004	-0.128 ± 0.004	0.088 ± 0.02
Std. Deviation	0.025	0.021	0.095
Maximum	-0.154	-0.080	0.262
Minimum	-0.246	-0.174	-0.055
Sample Size	31	26	17

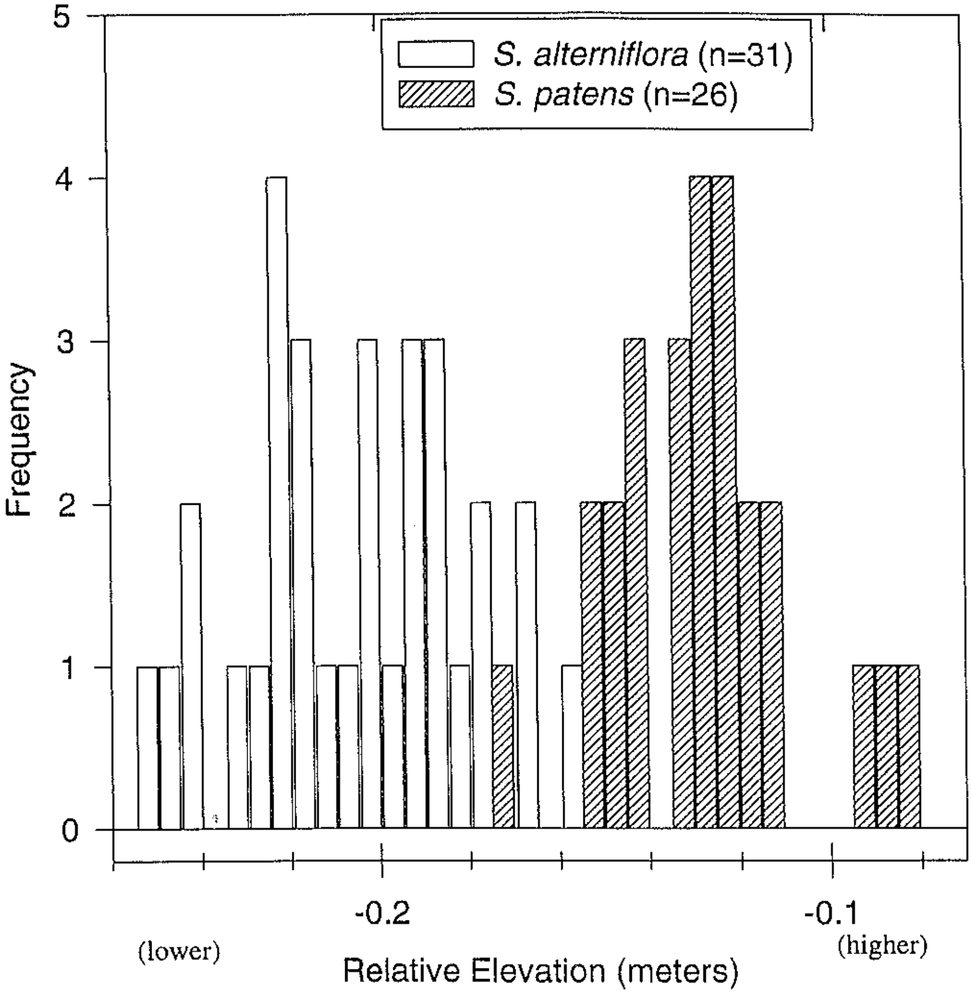


Figure 4.8 Histogram of mean relative elevation differences between vegetative zones, Pot Nets North (PNN) site.

At the Oyster Neck Road (ONR) site (Table 4.5), the difference in mean elevation between *S. alterniflora* and *S. patens* surfaces is 4.9 cm, while the difference between *S. alterniflora* and *S. patens/I. frutescens* combined mean elevations is 6.9 cm (Figure 4.9).

Table 4.5 Oyster Neck Road (ONR) site, western Delaware Bay. Difference in mean relative elevation: *S. alterniflora*, *S. patens*.

ONR Site Elevation Data	Relative Elevation (m)	Relative Elevation (m)
	S. alterniflora	*S. patens*
Mean (± s.e.)	-0.064 ± 0.003	-0.015 ± 0.03
Std. Deviation	0.015	0.138
Maximum	-0.041	0.326
Minimum	-0.096	-0.109
Sample Size	28	29

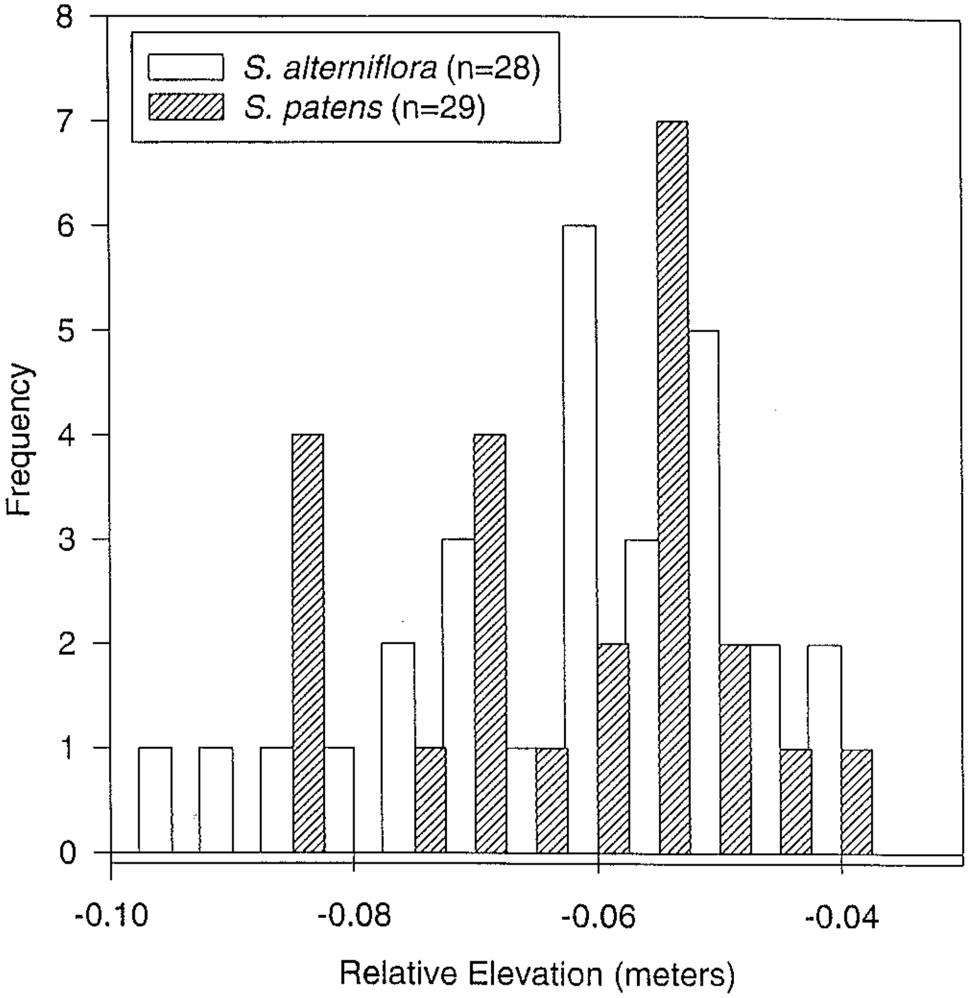


Figure 4.9 Histogram of mean relative elevation differences between vegetative zones, Oyster Neck Road (ONR) site.

At the Prime Hook Radio Station (PHRS) site (Table 4.6), the difference in mean elevation between *S. alterniflora* and *S. patens* is 8 cm; while the difference between *S. alterniflora* and *S. patens/Iva* surfaces combined is also 8 cm (Figure 4.10).

Table 4.6 Prime Hook Radio Station (PHRS) site, western Delaware Bay. Difference in mean relative elevation: *S. alterniflora*, *S. patens*.

PHRS Site Elevation Data	Relative Elevation (m)	Relative Elevation (m)	Relative Elevation (m)
	S. alterniflora	*S. patens*	*I. frutescens*
Mean (± s.e.)	-0.014 ± 0.004	0.066 ± 0.03	0.067 ± 0.02
Std. Deviation	0.016	0.155	0.067
Maximum	0.024	0.600	0.173
Minimum	-0.048	-0.050	-0.024
Sample Size	19	27	17

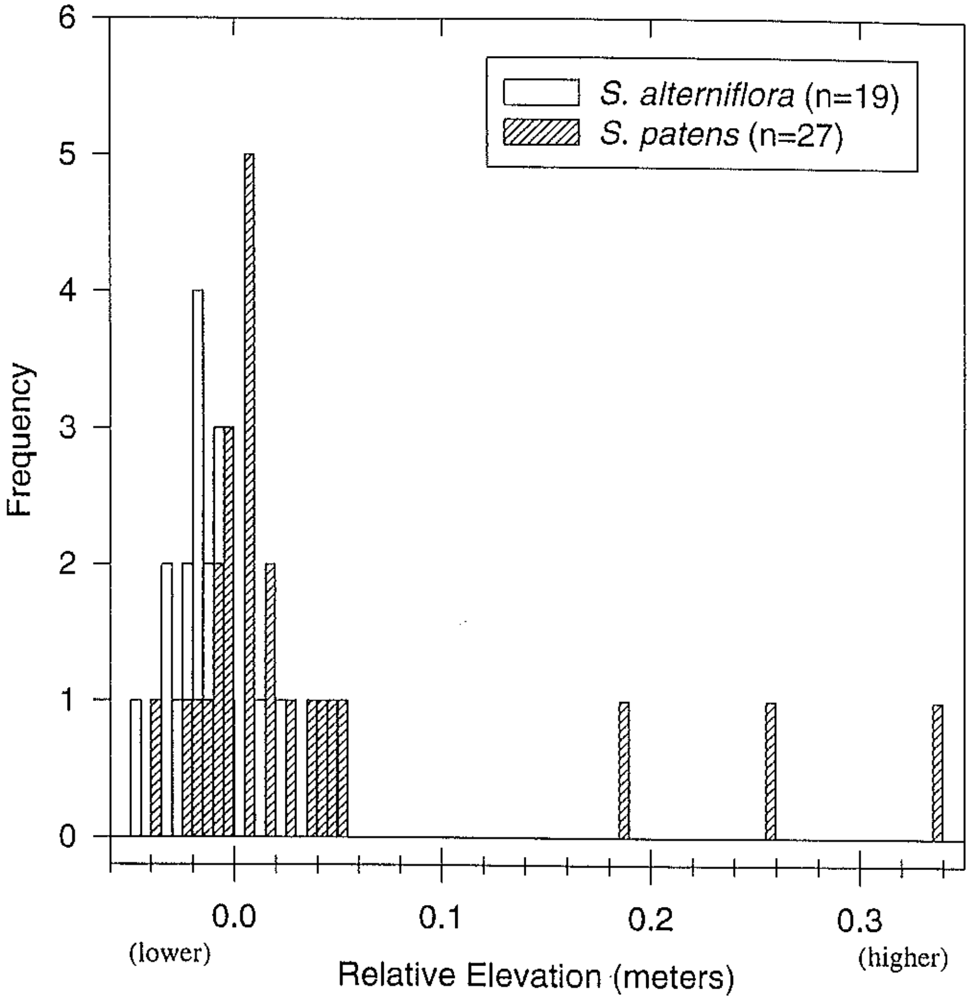


Figure 4.10 Histogram of mean relative elevation differences between vegetative zones, Prime Hook Radio Station (PHRS) site.

Species Distribution: Depth to Pre-Transgressive Sand (Marsh Thickness)

General. Previous studies have noted a possible relationship between substrate and marsh vegetative zonation (Teal and Kanwisher, 1961; Edwards and Frey, 1977; Frey and Basan, 1985), but not as related to antecedent geometry. In this study, striking differences in wetland vegetation were noted to be strongly associated with depth to pre-transgressive sand surface in Delaware fringing tidal salt marshes. Differences in marsh thickness were correlated with surface vegetative zonation for the 69 individual transects and for the five detailed survey sites.

Individual Transect Sites. Depth to the pre-transgressive sand for each surface vegetation zone is presented in Appendix C; data are summarized for all transects in Table 4.7 and presented as a histogram in Figure 4.11. A non-parametric Mann-Whitney statistical test shows a significant difference ($P<0.005$) between *S. alterniflora* and *S. patens* vegetative zones.

Table 4.7 All individual transect sites: average marsh thickness (depth to pre-transgressive sand) data.

Marsh Thickness: Depth to Pre-Transgressive Sand (cm)			
All Transects	*S. alterniflora*	*S. patens*	*I. frutescens*
Mean	**87.3 ± 4.54**	**22.4 ± 1.54**	**16.9 ± 1.23**
Std. Dev.	60.09	18.88	11.76
Max.	267.0	120.0	52.0
Min.	11.0	0.0	0.0
Sample Size	175	151	92

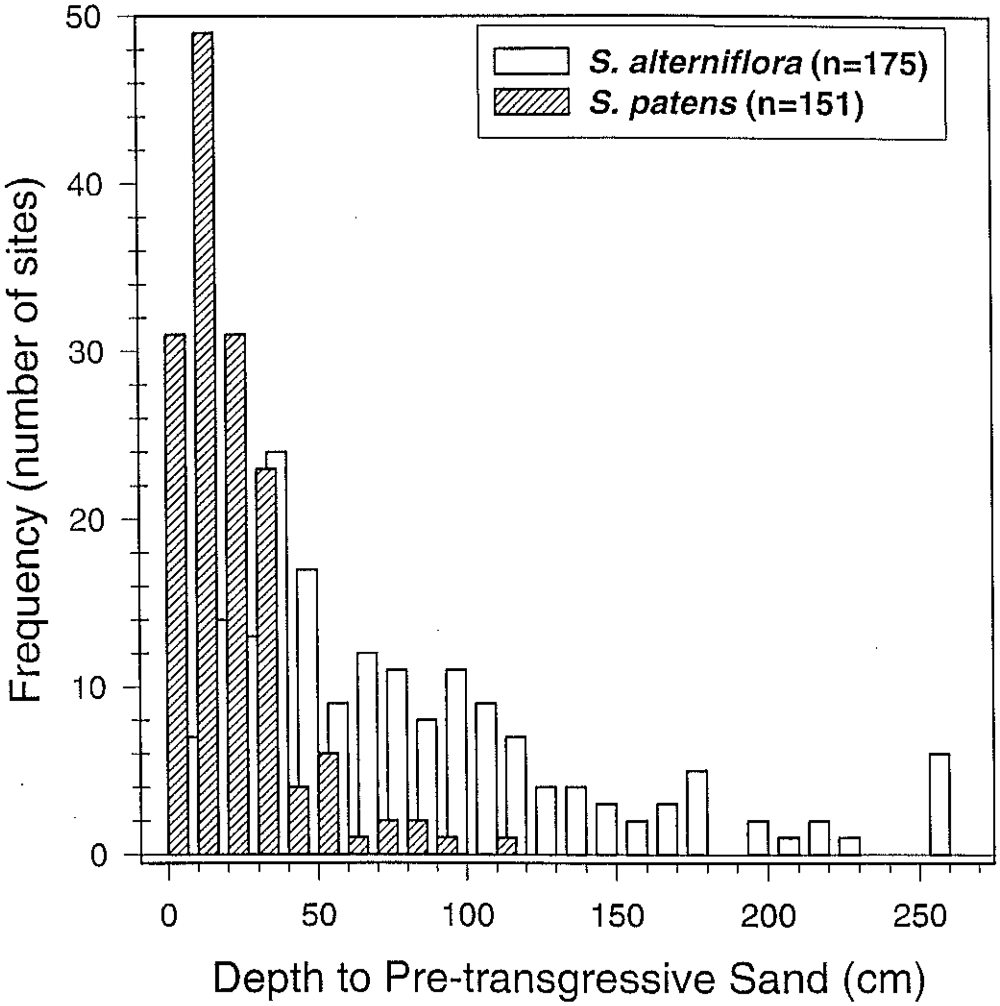


Figure 4.11 Histogram of marsh thickness (depth to pre-transgressive sand) for all individual transect sites.

Detailed Survey Sites. Depth to the pre-transgressive sand was also measured at the five detailed survey locations; these data are presented in the following paragraphs and tables.

1) Boat House Cove (BHC) Site: Detailed information on marsh thickness was obtained at 49 points in the Boat House Cove marsh at the wetlands/uplands boundary area. The data on surface vegetation and depth to pre-transgressive sand surface are presented in Table 4.8 and Figure 4.12. Note the absence of *S. patens* in the surface vegetative community at the Boat House Cove site. *I. frutescens* vegetation is generally restricted to areas where depth to pre-transgressive sand is less than 12 cm. It should be noted that marsh thickness in the *S. alterniflora* zones only include sites within 20 meters of the wetlands/uplands boundary. Marsh thickness of the *S. alterniflora* zones would likely increase bayward along this transect.

Table 4.8 Boat House Cove site: data on surface vegetation and depth to pre-transgressive sand.

Marsh Thickness: Depth to Pre-Transgressive Sand (cm)			
BHC Site	*S. alterniflora*	*S. alt/Iva border*	*I. frutescens*
Mean (± s.e.)	**87.5 ± 7.78**	**30.0 ± 6.02**	**4.9 ± 0.56**
Std. Dev.	35.63	15.92	2.56
Max.	137.0	58.0	12.0
Min.	24.0	12.0	2.0
Sample Size	21	7	21

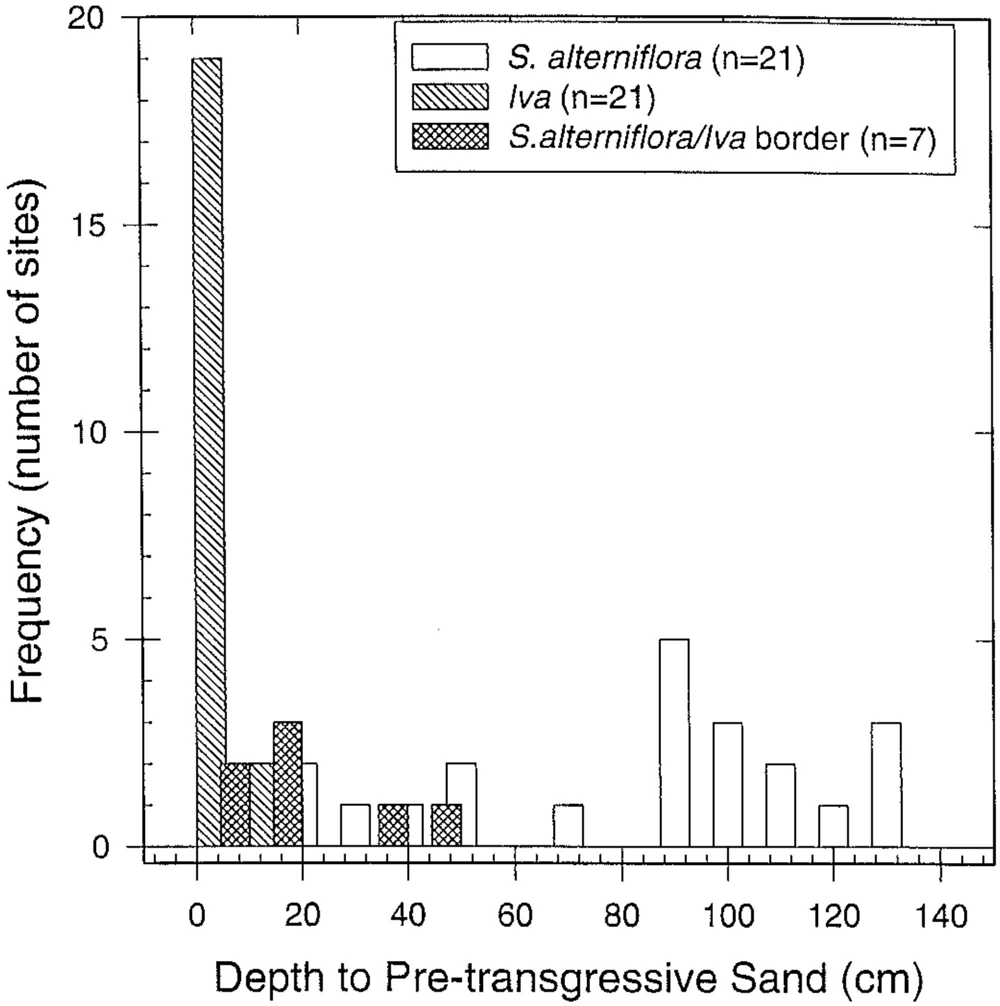


Figure 4.12 Boat House Cove site: histogram of marsh thickness (depth to pre-transgressive sand).

2) Delaware Wildlands (DEWL) Site: Detailed information on marsh thickness was obtained at 70 points at the Delaware Wildlands site. Depth to pre-transgressive sand surface by surface vegetation zone are shown in Table 4.9 and Figure 4.13.

Table 4.9 Delaware Wildlands site: data on surface vegetation and depth to pre-transgressive sand.

Marsh Thickness: Depth to Pre-Transgressive Sand (cm)				
DEWL Site	*S. alterniflora*	*S.alt/S.patens border*	*S. patens*	*I. frutescens*
Mean (± s.e.)	**96.4 ± 8.95**	**39.7 ± 4.22**	**26.2 ± 2.34**	**13.4 ± 1.99**
Std. Dev.	40.03	12.66	12.36	7.73
Max.	162.0	71.0	45.0	29.0
Min.	34.0	29.0	4.0	4.0
Sample Size	20	9	28	15

S. patens vegetation is generally found in areas where the depth to pre-transgressive sand is less than 30 cm; mean marsh thickness of *S. alterniflora* and *S. patens* marsh areas is significantly different (P<0.005), based on a Mann-Whitney non-parametric statistical test.

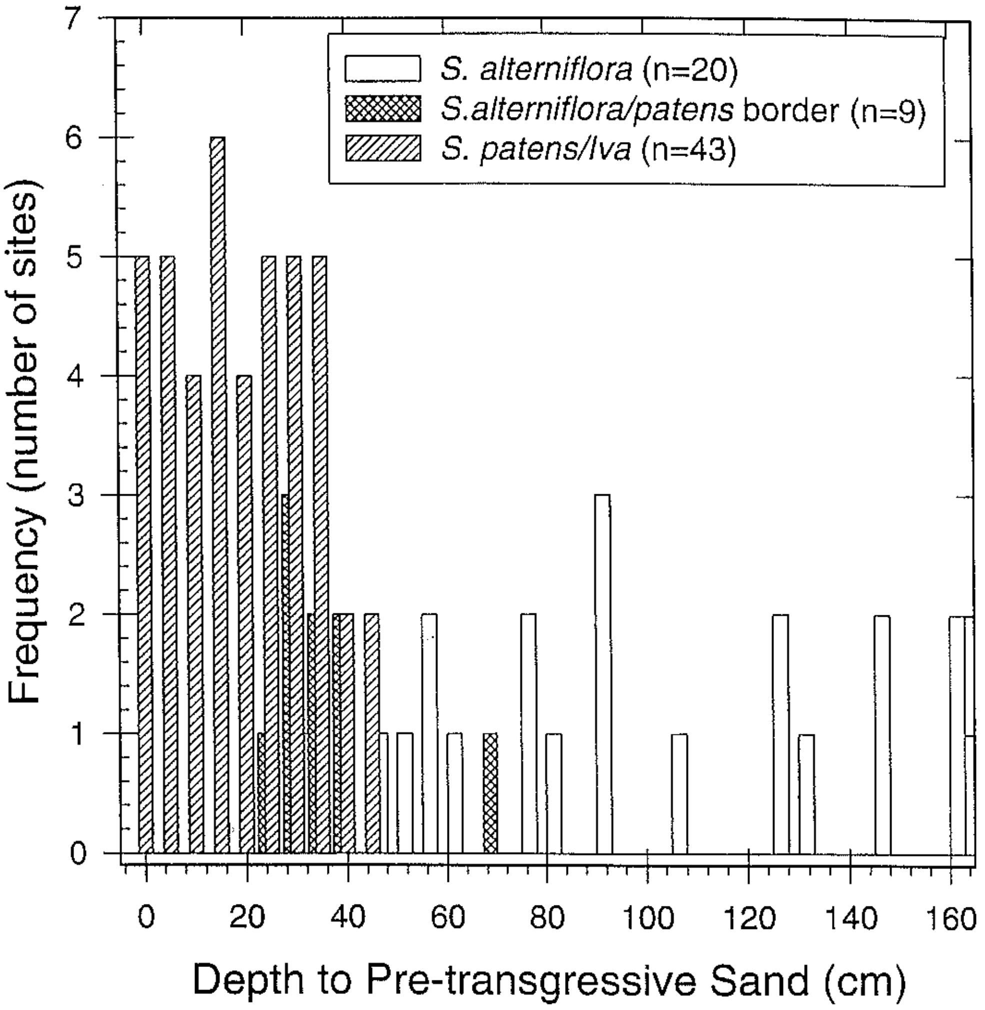


Figure 4.13 Delaware Wildlands site: histogram of marsh thickness (depth to pre-transgressive sand).

5) Pot Nets North (PNN) Site: Detailed information on marsh thickness was obtained at 104 points in the Pot Nets North site at the wetlands/uplands boundary area. The data on surface vegetation and depth to pre-transgressive sand surface are shown in Table 4.10 and depicted in Figure 4.14; there is a significant difference between mean marsh thickness of *S. alterniflora* vegetation and *S. patens* marsh surface areas (P<0.005).

Table 4.10 Pot Nets North site: data on surface vegetation and depth to pre-transgressive sand.

Marsh Thickness: Depth to Pre-Transgressive Sand (cm)				
PNN Site	*S.alterniflora*	*S.alt/patens*	*S. patens*	*I. frutescens*
Mean (± s.e.)	**76.5 ± 11.57**	**18.5 ± 1.49**	**14.5 ± 0.43**	**9.0 ± 0.99**
Std. Dev.	64.43	8.04	2.18	4.20
Max.	252.0	38.0	19.0	15.0
Min.	23.0	4.0	11.0	4.0
Sample Size	31	29	26	18

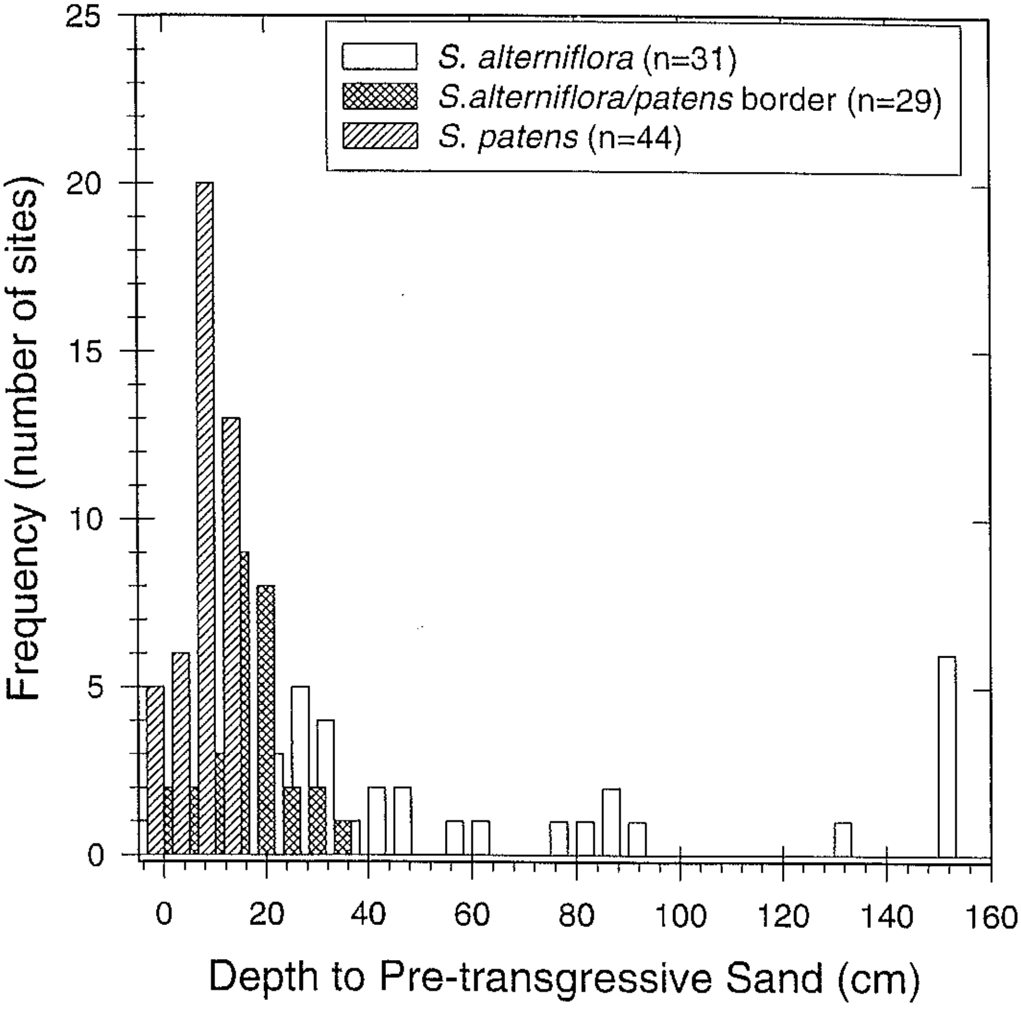


Figure 4.14 Pot Nets North site: histogram of marsh thickness (depth to pre-transgressive sand).

3) Oyster Neck Road (ONR) Site: Detailed information on marsh thickness were obtained at 118 points in the Oyster Neck Road study site at the wetlands/uplands boundary area. The data on surface vegetation and depth to pre-transgressive sand surface are shown in Table 4.11 and depicted in Figure 4.15; mean depths to pre-transgressive sands are significantly different (P<0.005) based on a Mann-Whitney one-way analysis of variance.

Table 4.11 Oyster Neck Road site: data on surface vegetation and depth to pre-transgressive sand.

Marsh Thickness: Depth to Pre-Transgressive Sand (cm)			
ONR Site	*S. alterniflora*	*alt/patens border*	*S. patens*
Mean (± s.e.)	**85.6 ± 5.02**	**43.1 ± 2.85**	**25.3 ± 2.46**
Std. Dev.	26.07	13.95	13.01
Max.	150.0	73.0	57.0
Min.	42.0	23.0	5.0
Sample Size	27	24	28

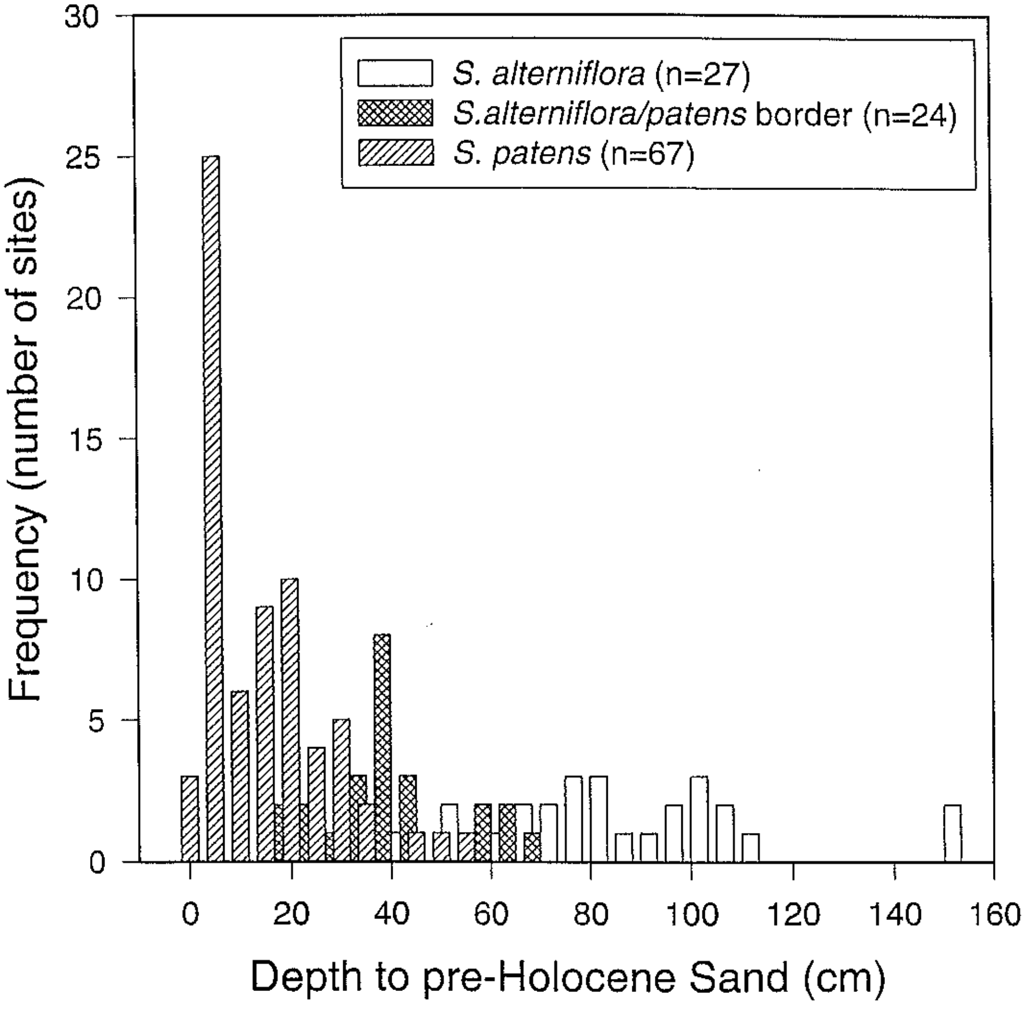


Figure 4.15 Oyster Neck Road site: histogram of marsh thickness (depth to pre-transgressive sand).

4) Prime Hook Radio Station (PHRS) Site: Detailed information on marsh thickness was obtained at 72 points in the Prime Hook Radio Station study site at the wetlands/uplands boundary area. The data on surface vegetation and depth to pre-transgressive sand surface are shown in Table 4.12 and depicted in Figure 4.16; a one way analysis of variance demonstrates that mean marsh thickness of *S. alterniflora* and *S. patens* marsh areas is significantly different (Mann-Whitney non-parametric statistical test, P<0.005).

Table 4.12 Prime Hook Radio Station site: data on surface vegetation and depth to pre-transgressive sand.

Marsh Thickness: Depth to Pre-Transgressive Sand (cm)			
PHRS Site	*S. alterniflora*	*S. alt/patens border*	*S. patens*
Mean (± s.e.)	**96.8 ± 8.00**	**60.7 ± 10.88**	**19.1 ± 2.04**
Std. Dev.	35.78	36.09	13.04
Max.	170.0	154.0	55.0
Min.	48.0	34.0	2.0
Sample Size	20	11	41

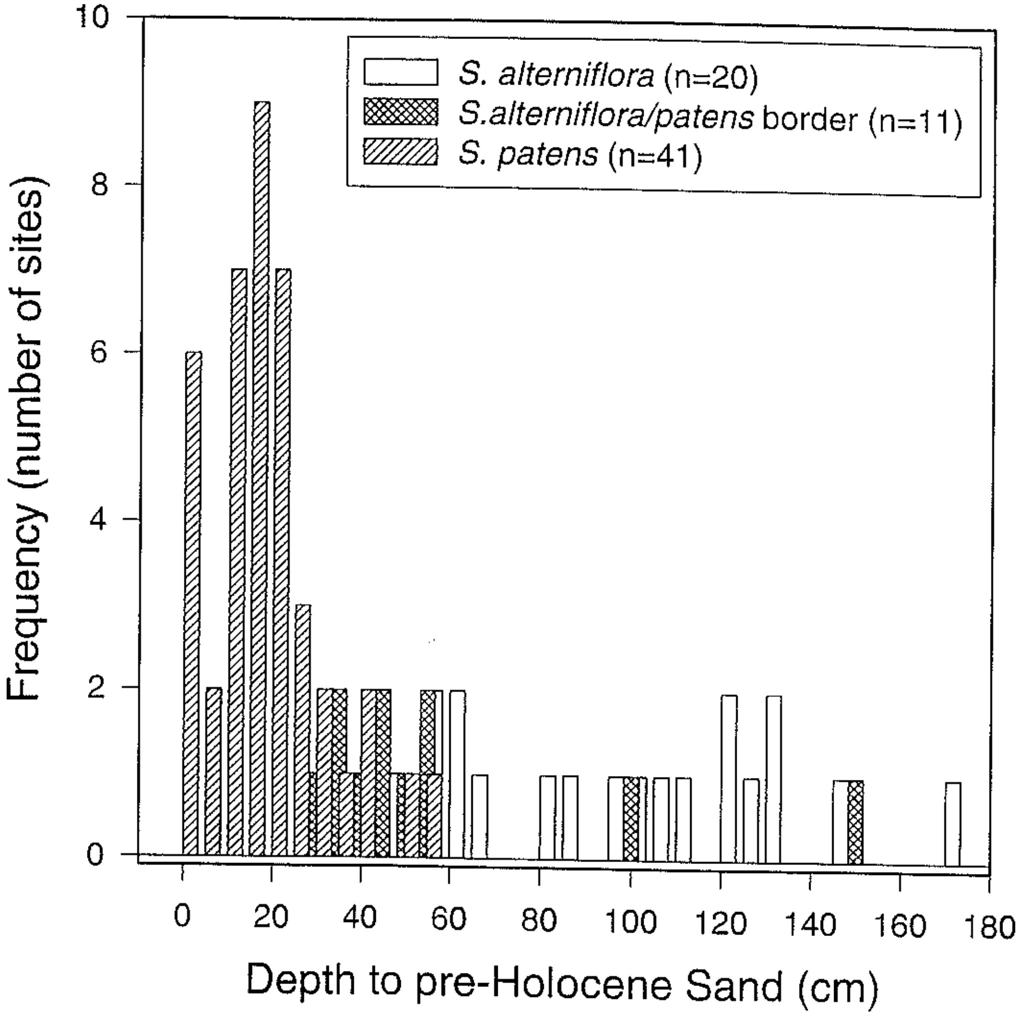


Figure 4.16 Prime Hook Radio Station site: histogram of marsh thickness (depth to pre-transgressive sand).

Discussion

The correlation between depth to pre-transgressive sand and vegetative zones suggests that abiotic factors influence the distribution of vegetation in Delaware's fringing salt marshes. Within constraints of this study (i.e., transects extend only 50-100 m from the wetlands/uplands boundary), vegetative zones can be distinguished on the basis of depth to pre-transgressive sand, or muddy vs. sandy substrates. The wetlands/uplands boundary area is typically underlain by pre-Holocene barrier sands, with variable subsurface slopes and antecedent topography. Typically, *S. patens/D. spicata* (high marsh) zones are restricted to areas where depth to pre-transgressive sand is less than 40 cm, while *S. alterniflora* (low marsh) zones are in areas where depth to pre-transgressive sand exceed 40 cm. Similar trends have been observed in salt marshes of Georgia, in both the Sapelo Island (Edwards and Frey, 1977) and Duplin River marshes (Teal and Kanwisher, 1961).

In the course of this study, a definitive relationship has been noted between the thickness of marsh sediment (depth to pre-transgressive sand) and surface vegetative zonation patterns. This relationship is not surprising when one considers the biotic factors and physiological adaptations of salt marsh vegetation. Silty/muddy low marsh sediments are highly reduced, whereas high marsh sediments are more oxic because of increased sand content and permeability of the substrate. Additionally, high pH values (e.g., >6.5 to 8.5) and low Eh values (-155 mv to +100 mv) have been measured in cores from low marshes in Georgia (Edwards and Frey, 1977); these highly reducing conditions likely exert control on species distribution.

Although it is perhaps intuitively obvious that *I. frutescens* and *S. patens* would be found in areas where the depth to pre-transgressive sands are shallower, it is interesting to note how consistent the "depth to sand" divisions are, especially for the *S. patens* vegetation. It is often assumed that distribution of *S. patens* is controlled by elevation and, ultimately, frequency and duration of tidal inundation. *S. patens* is often considered to be restricted to areas of the marsh above the mean high water level, i.e., that are only flooded during spring tides and storm events.

These data clearly demonstrate that depth to antecedent sandy substrates is greatest for *S. alterniflora* vegetation, with depths to pre-transgressive sandy surfaces decreasing for *S. patens* and *I. frutescens/B. halimifolia* vegetative zones. This provides additional data which show the importance of antecedent topography or slope on the development and evolution of fringing wetlands in Delaware.

Species Distribution: Slope and Width of *S. patens/D. spicata* zone

The slope of the upland is directly related to width (or presence) of *S. patens/D. spicata* high marsh zones and may ultimately control the fate of tidal wetlands. Slopes of adjacent upland areas were calculated from the survey data, and are expressed as tangent of slope, i.e., rise/run. Slopes ranged from shallow (0.0067) to steep (0.232), with corresponding widths of high marsh (*S. patens/D. spicata* zone) ranging from 67 m to 0.0 m, respectively. These data are presented in Appendix D.

The relationship between width of the *S. patens/D. spicata* zone and slope of adjacent upland surface is shown in Figure 4.17. A relationship between adjacent slope and width of high marsh environment can be further defined when considered in conjunction with the 'threshold depth' of marsh thickness (or depth to pre-transgressive sand). As depicted in Figure 4.17, the width of high marsh *S. patens* environments can be defined as the threshold depth (e.g., 0.5 m) / slope of adjacent upland surface. The plot of the calculated width line (threshold depth/slope) falls within the data envelope of measured widths and slopes.

Study sites with shallow upland slopes (<0.08) are characterized by broad *S. patens* zones, with *S. patens* and *I. frutescens* colonizing upland surfaces. Evidence of this transgression includes dead tree stumps in *S. patens* marshes (Figure 4.18) and many dead and dying trees at the wetlands/uplands boundary. In contrast, steep-slope areas (>0.09) are generally characterized by narrow *S. patens* zones (<2 m wide) or by *S. alterniflora* vegetation directly at the base of the upland slope (Figure 4.19). In most cases where a steep terrestrial upland intersects the plane of the tidal marsh, there is a narrow transitional fringe of *I. frutescens* and/or *P. australis*, with little or no *S. patens*. Similarly, there is typically little evidence of dead or dying upland vegetation on steep slopes, because the wetlands have not migrated laterally over the upland surface.

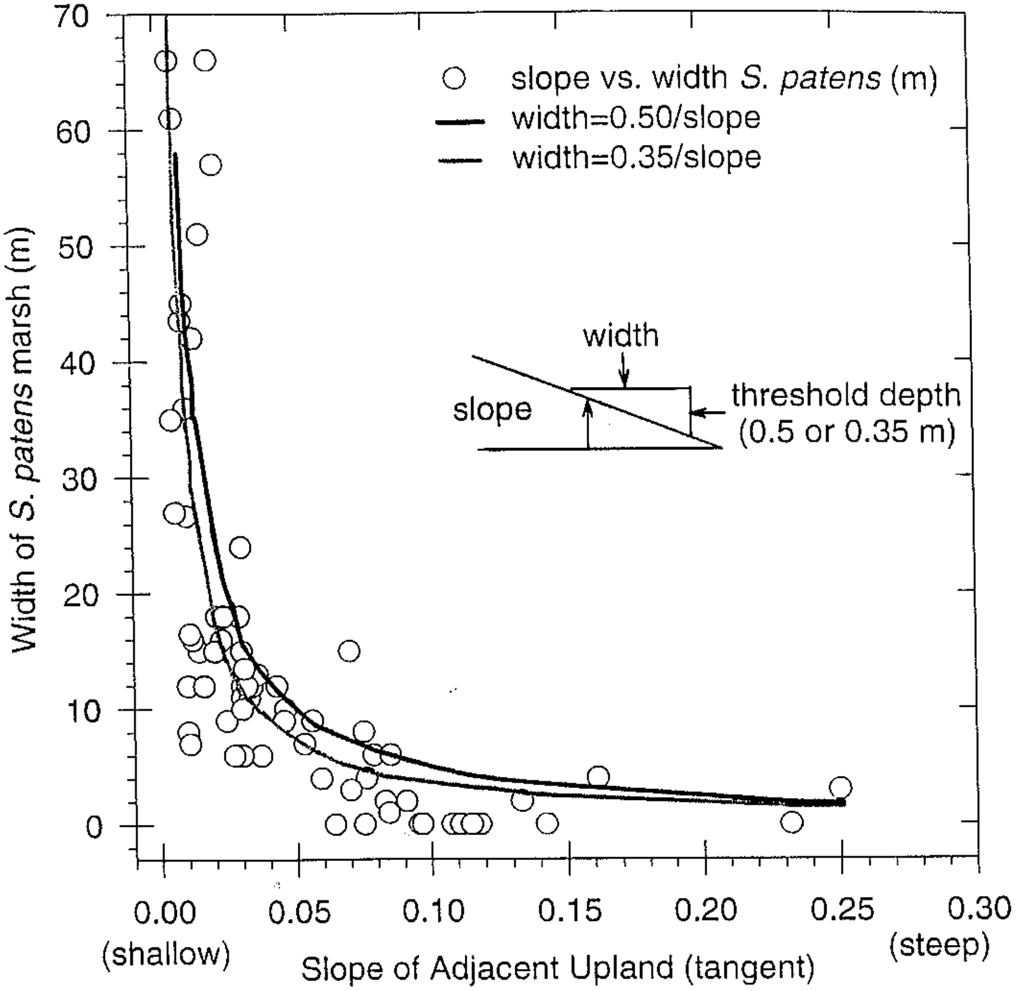


Figure 4.17 Relationship between width of *S. patens*/*D. spicata* zone, slope of adjacent upland surface, and marsh thickness. See Appendix D for data included in graph.

Figure 4.18 Photograph of dead and dying trees and *S. patens/D. spicata* vegetation at wetlands/uplands boundary zone, low slope adjacent upland.

Figure 4.19 Photograph of *S. alterniflora* vegetation at base of steeply sloping adjacent upland. Note absence of *S. patens/D. spicata* vegetative zone.

Generally, steep adjacent upland slopes characterize fringing wetlands along tidal creeks and rivers, where antecedent stream/river valley incisions have resulted in the present-day steep topography and geomorphology.

Data presented in Figure 4.17 suggest that evolution, longevity, and/or development of *S. patens* high marshes is restricted in steeply sloping areas. Except for three sites where the *S. patens* zone is < 2 m wide, the data for slopes above 0.10 indicate that high marshes do not form on steep slopes. Although one of these outlier points can be explained by human impact (sediment runoff onto wetlands), the other two are related to variable antecedent topography, where a shallow sandy platform/shelf is at the base of the steep adjacent upland slope. Although high marshes may exist at any slope, they tend to approach a minimum width, depending on geometry of antecedent surfaces.

The data show a definitive geometric relationship between the slope of upland surface and the distribution of vegetation in adjacent tidal salt marshes. In areas with a low slope adjacent to wetlands, the high marsh has a broader zone for development, with associated vegetative diversity. In contrast, a steep slope provides little or no 'accommodation space' for either establishment or landward movement of high marsh environments; this ultimately results in absence of high marsh environments as low marsh environments continue to move landward and upward through time. This relationship between adjacent upland slope and width of *S. patens* zone further emphasizes the control exerted by antecedent and adjacent topography on Delaware's fringing tidal salt marshes.

Surficial Zonation Patterns: Discussion

Introduction

Zonation is a conspicuous and well documented feature of salt marshes throughout the world (Chapman, 1974). Generally, zonation patterns are ascribed to tidal influences (duration and frequency) due to the fact that many salt marsh species occupy a characteristic vertical range relative to sea level. More detailed examinations of the relationships between salt marsh vegetation and its environment have demonstrated that sediment characteristics, which are also related to tidal regime, may be causative agents of vegetative zonation in salt marshes (Niering and Warren, 1980; Vince and Snow, 1984). Plant distributions have been correlated with soil properties (Vince and Snow, 1984) such as soil salinity (Mahall and Park, 1976), soil type (Gray and Bunce, 1972), waterlogging (Gray and Scott, 1977) and nutrient availability (Pigott, 1969), but competitive interaction is rarely considered (Vince and Snow, 1984; Snow and Vince, 1984).

Community Structure and General Ecology

This investigation documents that high marsh vegetation species (*S. patens/D. spicata*) are initial colonizers of shallow-sloping adjacent upland surfaces at the landward edge of fringing tidal salt marshes. At the same time, the channelward or bayward edge of the high marsh zone is being transgressed by the *S. alterniflora* low marsh zone. At most study sites, higher-elevation vegetative communities are being transgressed by lower-elevation communities, as demonstrated by *S. patens* and *I. frutescens/B. halimifolia* vegetation in areas of dead tree stumps (Figure 4.18); *S. patens/D. spicata* vegetation in areas of dead and dying *I. frutescens/B. halimifolia*; and *S. alterniflora* invading areas of dead *I. frutescens/B. halimifolia* and eroding hummocks of *S. patens/D. spicata* (Figure 4.19).

Therefore, low-marsh assemblages may be considered a "climax community" in Delaware's fringing salt marshes. This has also been documented in Massachusetts salt marshes by Clark (1986) and Clark and Patterson (1985) who observed that this scenario is in direct contrast to Chapman's (1974) description of high-marsh assemblages as "climax communities."

Regional patterns of vegetative zonation parallel to the wetlands/uplands boundary support an interpretation that abiotic variables control zonation patterns; and the influence of adjacent upland slope and antecedent topography must be considered as an integral part of the physical environment in fringing tidal salt marshes. These results do not alter the fact that salt marsh vegetation responds to environmental and physiological stress (Valiela and Teal, 1974; Mendelssohn *et al.*, 1981; Burdick, 1989), or that tidal influence may ultimately regulate productivity (as related to salinity, dissolved oxygen, sulfides, iron, nitrogen, etc.). Antecedent topography and upland slope clearly influence vegetative zonation, and may ultimately control the future areal extent of tidal salt marshes in Delaware.

Zonation of tidal salt marshes is usually related to elevation relative to tide levels (Chapman, 1974), but questions remain about species composition along an elevation gradient as a result of floristic competition. Data from this study relate relative elevation and adjacent/antecedent topography to surface vegetative zonation. Caution must be used in applying these data to questions of community structure, competition and general plant ecology. Additional detailed studies on general ecology, ecological principles, and complex environmental variables (tidal flooding, siltation, and soil characteristics) are required before 'ecological' interpretations can be made.

In this study, species composition changed either gradationally or sharply with increasing relative elevation, depending on the slope of adjacent and antecedent topography. Past research suggests that salt marsh zonation occurs in response to environmental factors that change along the elevation gradient such as frequency and duration of tidal inundation (Vince and Snow, 1984). More data on variations in flooding regimes at the wetlands/uplands boundary are required before flooding frequency can be related to vegetative zonation. However, limited tide gauge data and observations made over three field seasons in the study areas suggest that tidal range is significantly dampened across the areal extent of the wetland. It is unlikely that tidal inundation alone determines plant distribution; more likely, interactions between vegetation, tides, and sediment characteristics (especially salinity and waterlogging) control vegetative zonation in the wetlands/uplands boundary. Additional data must be collected and analyzed before specific correlations can be made regarding effects of edaphic factors on wetland vegetative zonation patterns. However, existing vegetation patterns and relationships to both mean relative elevation and depth to pre-transgressive sandy substrate are consistent with previous research conducted on vegetation patterns and soil chemistry (Niering and Warren, 1980; Nixon, 1982; Teal, 1986; Warren and Niering, 1993). The better-drained sandy substrate underlying the *S. patens/D. spicata* vegetation should have a lower redox potential and lower sulfide concentrations than thicker muddy substrates, and therefore would be a less stressful environment for marsh vegetation.

Investigations by Clark and Patterson (1985) and results of this study suggest that salt marsh vegetation responds to frequent and severe fluctuations in the physical environment. Additionally, the role of between-species competition and its influence on zonation must be considered. Snow and Vince (1984) suggest that there is a complementary relationship between the roles of tolerance of physical factors and between-species competition in various ecosystems; i.e., "species occurring in zones along a physical gradient are often limited by physiological tolerance toward one end of the gradient, and by competitive ability towards the other, where many species could potentially survive and reproduce" (p. 682).

Model: Development and Succession of Fringing Salt Marshes

Many previous studies have surveyed the distribution of plants and species diversity within intertidal salt marshes throughout the United States (Teal, 1958; Wilson, 1962; Good, 1965; Stroud and Cooper, 1968; Reimold *et al.*, 1975; Turner, 1976; Nixon, 1982). It is not the intent of this study to provide detailed species inventories as related to elevation and zonation, but to examine the temporal and spatial relationships between uplands, high marsh and low marsh environments at the leading edge of the transgressive wetland system in coastal Delaware.

The vegetative zonation observed in Delaware's fringing marshes is consistent with other East Coast marshes. The dominant plants of the three primary zones differentiated in this study include: *I. frutescens/B. halimifolia* in the transitional marsh zone; *S. patens/D. spicata* dominate in the high marsh; and *S. alterniflora* dominates in the low marsh. Although the author recognizes that the short form *S. alterniflora* can be a dominant species in the high marsh with medium and tall *S. alterniflora* dominant in the low marsh, no distinction was made in this study between tall, medium, and short forms of *S. alterniflora* because relative elevations were measured. That is, *S. patens* and *D. spicata* were always found at slightly higher elevations than either short, medium or tall form *S. alterniflora.*

In Delaware, the tidal salt marsh is a geologic landform developing in conjunction with post-glacial progressive submergence of the coastal zone. Many models of tidal salt marsh development proposed by biologists and ecologists, especially those developed for emerging coasts, suggest that plant succession in salt marshes progresses in the other direction. That is, many plant ecologists hypothesize that plant succession in coastal tidal wetlands culminates in a high and dry *S. patens* or *Juncus* marsh, and ultimately an upland area. Delaware's fringing tidal wetlands appear to have developed the "opposite" way; i.e., plant succession moves from higher, drier uplands to high marsh *S. patens*, and ultimately to low marsh *S. alterniflora* wetlands. The stages associated with "progressive submergence" have been described in the literature (Niering *et al.*, 1977; Orson *et al.*, 1987; Nydick *et al.*, 1995).

Characteristics of the progressive submergence, development, and evolution of Delaware's fringing tidal salt marshes are outlined below:

I. As sea level rises tidal waters flood previous upland surfaces and extend up stream valleys to create lagoons, tidal creeks, and freshwater tidal swamps and marshes.

II. As submergence progresses, salinity increases and adjacent bordering upland and/or freshwater swamp is inundated. Lagoons fill with silt to a certain level then the shallow margins are colonized by *S. alterniflora* and *S. patens* (e.g., the Shaler marsh development theory).

III. Submergence continues, the estuarine and wetland environments migrate landward, the upland and/or freshwater swamps are overrun by encroaching tidal marsh. Thickness of the *S. patens* horizon remains unchanged; thickness of the *S. alterniflora* peat horizon increases with continued sea-level rise. Depending on the adjacent upland slope, the areal extent of the *S. patens* zone either increases or decreases over time. Spatial and temporal relationships of vegetative community zonation respond to many interacting forcing functions, including rate of relative sea-level rise, slope of adjacent upland, and rate of vertical accretion of various vegetative zones.

IV. Submergence continues and more of the upland and freshwater swamp are inundated and destroyed. Adjacent upland slope and antecedent geometry control not only rate of landward migration of the wetland, but also distribution of surface vegetative zonation. Broad expanses of *S. patens* marsh will continue to migrate spatially and temporally only if relatively low slopes (<0.10) are available; based on data obtained in this study, widest *S. patens* marshes are likely to be found where adjacent upland slopes are < 0.05. Otherwise, *S. patens* environments will be "pinched out" or minimized between advancing *S. alterniflora* wetlands from the bayward side and terrestrial vegetation on steep adjacent slopes. In addition, rates of erosion on bayward/seaward side will control areal extent of fringing tidal salt marshes; if rate of erosion exceeds rate of landward migration, fringing wetlands will not maintain areal extent in space and time.

Summary and Conclusions

1) Zonation patterns of wetland surface environments were documented at the wetlands/uplands boundary zone. Five vegetative zones were mapped according to species dominance: a) upland (terrestrial vegetation) zone; *b) I. frutescens/B. halimifolia* (shrub) zone; c) *S. patens/D. spicata* (high marsh) zone; d) mixed *S. patens/D. spicata/S. alterniflora* zone; and e) *S. alterniflora* (low marsh) zone.

2) Plant species composition in tidal marshes changes with elevation, although the range of elevation change is small and species distributions often overlap. The transition between *S. patens/D. spicata* (high marsh) zone and *S. alterniflora* (low marsh) zone occurs over a mean elevation difference of 17 ± 1.4 cm, with a range of 2 to 46 cm.

3) Surficial wetland zonation patterns at the wetlands/uplands boundary are strongly associated with antecedent topography and depth to the pre-transgressive sand surface. The *S. patens/D. spicata* (high marsh) zone and *S. alterniflora* (low marsh) zone may be distinguished by depth to pre-transgressive sand. At the wetlands/uplands boundary, the *S. patens/D. spicata* zone is restricted to areas where depth to pre-transgressive sand is less than 40 cm, while the *S. alterniflora* zone is found in areas where depth to pre-transgressive sand exceeds 40 cm.

4) The width of the *S. patens/D. spicata* (high marsh) zone is directly related to slope of the adjacent upland surface. This suggests that antecedent topography controls the rates of landward migration of tidal wetlands, and vegetative zonation at the wetlands/uplands boundary. Sites with shallow adjacent upland slopes (<0.05) were characterized by broad *S. patens/D. spicata* zones, while areas with steep slopes (>0.10) are generally characterized by *S. alterniflora* vegetation directly at the base of the upland slope, with either no high marsh zone or a very narrow (2-3 m) band of *S. patens* between the upland and the *S. alterniflora* marsh. Steeply sloping adjacent uplands provide little or no accommodation space for either establishment or landward movement of high marsh environments. This ultimately results in the reduction or disappearance of the high marsh as low marsh environments continue to move landward and upward in space and time.

Chapter 5

SUBSURFACE STRATIGRAPHY OF WETLANDS/UPLANDS BOUNDARY AREA

Introduction

The shallow stratigraphy at the wetlands/uplands boundary of Delaware's fringing tidal salt marshes provides information on recent (100-200 year) development of marsh paleoenvironments. Changes in paleoenvironment or vegetative species with depth in subsurface cores indicate changes in associated environmental and physical forces which drive wetland development. Stratigraphic patterns are related to antecedent geometry and subtle variations in configuration of the pre-transgressive surface. As observed in mapping of surficial vegetative zones, the slope of the pre-transgressive surface controls the development and successional patterns of the salt marsh.

Most previous investigators studying the Holocene of Delaware described the marsh lithosome as one general unit (e.g., Kraft *et al.*, 1989; Chrzastowski, 1986; Stedman, 1990). However, in several more detailed studies, the marsh lithosome was subdivided into microfacies (Allen, 1974, 1977; Yi, 1992). None of these previous studies investigated the narrow boundary zone between wetlands and uplands in Delaware's fringing tidal salt marshes.

A primary objective of this investigation is to characterize surface and subsurface vegetative distribution patterns at the wetlands/uplands boundary of fringing salt marshes. This portion of the study focuses on assemblages of plant remains in subsurface units for comparison with the established surficial vegetative zones. Substrates in the wetlands/uplands boundary zone may be characterized by sediment texture (sand vs. mud), vegetative remains, and changes in vegetation with depth (e.g., succession from high marsh to low marsh). Changes in vegetative remains in the subsurface are indicative of changes in depositional environment; subsurface transition from *Spartina alterniflora* to *Spartina patens/Distichlis spicata* rhizomes and stems indicates submergence of the former high marsh environment and transition to a low marsh wetland environment.

Methods

Field Data Collection

Eijelkamp Cores. Subsurface data were obtained in 1993, 1994, and 1995 from over 400 hand-driven Eijelkamp gouge auger cores at 69 transect sites in fringing tidal marshes along the margins of Delaware Bay, Rehoboth Bay, Indian River Bay, and Little Assawoman Bay (Figures 2.6-2.9). Transects were perpendicular to the wetlands/uplands boundary, with cores taken at representative intervals along each transect. Surficial vegetative characteristics and distance from wetlands/uplands boundary were recorded. Cores were described in the field, and lithologic units were visually identified, based primarily on texture (sand/mud), color (Munsell), vegetative remains, and estimated percent organic content. Vegetation in cores was identified from rhizome, shoot, and stem characteristics as described by Niering *et al.* (1977) and Orson and Howes (1992). Cores were driven through the total thickness of marsh deposits, into the underlying pre-transgressive sandy sediments, and ranged from <1 m in length to >2 m in length, depending on marsh thickness and antecedent geometry. Generally, no compression of peat was detected, but if compaction was noted the area was resampled. Changes in stratigraphic sequences were mapped in detail, along with the depth to pre-transgressive sandy sediments. These data were used to determine the topography of the surface over which the marsh is transgressing. Analysis and interpretation of core data permit reconstruction of the evolution of the wetlands/uplands boundary during transgression.

Vibracores. Twelve cores for geochronological analyses and detailed examination of macrophyte remains were obtained at four study sites by vibracoring with a piston attachment (Foster *et al.*, 1974; Lanesky *et al.*, 1979; Chrzastowski, 1986). The piston is held stationary at the marsh surface by a cable and minimizes rodding and compaction by sealing the pipe at the top of the sediment as the pipe is driven down. Aluminum-alloy irrigation pipe (7.3 cm id; pipe thickness 0.12 cm) was used to core through the entire Holocene marsh sequence (usually < 2.5 m). A 7-horsepower cement vibrator was used to produce approximately 1300 vibrations per minute at the vibrator head, thus vibrating the core pipe into the subsurface sediment. The root mat of the upper 25 cm of the marsh was cut with a knife to allow free penetration by the core pipe and minimize rodding and compaction. In addition, the aluminum pipe was sharpened to facilitate movement through subsurface organic remains. Prior to core extraction, the amount of rodding or compaction was recorded as the difference in elevation between the sediment surface inside the pipe and the marsh surface outside the pipe (Chrzastowski, 1986); rodding and compaction varied from 1 cm to 2.5 cm, and was not considered significant.

Vibracores were transported to the laboratory, cut in half with an electric saw, and photographed. One half of each vibracore was sent to Dr. Grace Brush, Johns Hopkins University, for pollen analyses (Chapter 6). The remaining core halves were used to identify plant remains and characterize sediment types. Observations on color (Munsell), grain size, organic fragments, and general lithology were made in the laboratory.

Short cores. Sixty short cores were collected during the summer and late fall of 1993 at four sites for detailed examination of above- and below-ground morphology and anatomy of *Spartina alterniflora* Loisel., *Spartina patens* (Ait.) Muhl., and *Distichlis spicata* (L.) Greene which are primary contributors to salt marsh peat. Plastic pipe cores (16 cm diameter, 25 cm length) were driven into the marsh surface and removed with a shovel. The cores were extruded from the plastic pipe, cut into 10-cm sections, and washed through a screen box (1-mm mesh). The morphology of root, rhizomatal, and stem material left on the screen mesh was examined macroscopically; large plant fragments were identified to species. Before coring, above-ground biomass was clipped at ground level, bagged, and later identified in the laboratory.

As described in Chapters 3 and 4, dominant plant species in Delaware tidal salt marshes are *S. alterniflora* (smooth cordgrass), *S. patens* (salt meadow hay), and *D. spicata* (spike grass). These three plants were used as reference samples to identify marsh plant stems, leaves, roots and rhizomes in the subsurface cores. Root, rhizome, and basal stem systems are emphasized because they are potentially the most important contributors to salt marsh peat. Identification of plant species was first made by examining root morphology, and leaf, stem and rhizome characteristics of living plants; this knowledge was then applied to macroscopic vegetative remains found in the subsurface. General morphologic characteristics of the more common marsh plants in Delaware salt marshes are described in a previous chapter, but details are also included in this section. A general description of habitat has also been given previously for each species.

Macroscopic Identification of Vegetative Remains

The vegetative organs used to identify plant fragments in subsurface cores were the stem, rhizome, leaf, and roots. Rhizomes encountered in core sections were identified with a published key by Niering *et al.* (1977) (Figure 5.1), and by comparison with reference material obtained from the short cores. Field identification was based on gross morphology, and was limited to the larger, sheathed rhizomes of *S. alterniflora*, the smaller, finer rhizomes of *S. patens* and *D. spicata*, and woody fragments of *Baccharis halimifolia* L. and *Iva frutescens* L. shrubs. The plant fragments recovered

KEY TO SPECIES IN SALT MARSH PEAT

1. Rhizomes greater than 10 mm in diameter; roots numerous, flaccid, 3-4 mm in diameter extending 20-30 cm or more below the rhizomes; pith hollow ..... *Phragmites communis*

1. Rhizomes less than 10 mm in diameter 2

  2. Rhizomes 1.5 mm or more in diameter present 3

    3. Rhizomes shiny, tannish brown, 2-4 mm in diameter, with conspicuous encircling scars or scales; pith solid with scattered vascular bundles; cortex with thin radial septa delimiting air chambers, when not collapsed (A) ...... *Distichlis spicata*

    3. Rhizomes not as above 4

      4. Rhizomes over 2.5 mm in diameter, tannish-brown, usually collapsed; ensheathing, scale-like leaves shredded; few small roots (B) *Spartina alterniflora*

      4. Rhizomes 1.5 to 2.5 mm in diameter 5

        5. Rhizomes with ensheathing scale-scars present, tannish-brown, pith hollow; cortex with a ring of air chambers; peat dense and fibrous with many fine roots (C) .... *Spartina patens*

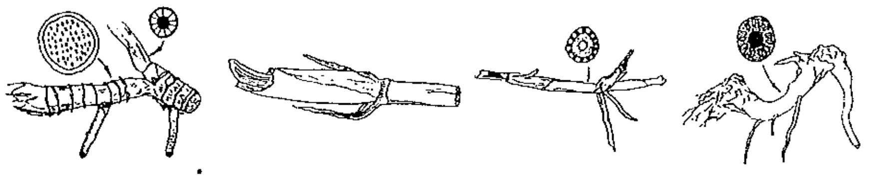
        5. Rhizomes lacking ensheathing, scale-like leaves, reddish-brown; pith solid, dark when wet, delineated by a narrow, reddish-brown band; cortex light with septa delineating air chambers; roots distinctly wavy and wiry (D) .... *Juncus gerardi*

  2. Rhizomes absent; peat usually black with numerous rootlets, often white, 0.25-0.5 mm in diameter .... Forb peat
  Root stalks up to 10 mm or more occasionally present 6

          6. Bulb-like root stalks covered with bristle-like fibers; central cylinder very hard, blood red and barbed in longitudinal section, with many fibrous roots up to 1 mm in diameter .... *Triglochin maritima*

          6. Root stalks woody, 5 to 10 mm or more in diameter, with smaller laterals; all parts black, wrinkled, with striated texture; pith and cortex dark and poorly differentiated .... *Limonium carolinianum*

Note: *Panicum virgatum* (Switchgrass) rhizomes are found in upland soil or at the marsh-upland interface. Rhizomes are 1 to 4 mm in diameter, usually collapsed and flaccid with very long, pointed, ensheathing, scale-like leaves and few roots or rootlets.



A *Distichlis spicata* *B* *Spartina alterniflora* *C* *Spartina patens* *D* *Juncus gerardi*

Figure 5.1 Key to species in salt marsh peat from Niering *et al.* (1977).

and identified in the cores were used to reconstruct the low marsh, high marsh, and transitional/upland subenvironments.

In subsurface marsh sediments, thin roots form fibrous networks and are the most abundant plant material, but roots are not easily identifiable to species by macroscopic inspection. Instead, rhizomes, stems, stem bases, and leaf fragments must be used for macroscopic identification of halophytes. Stem bases (sheathing bases or ligules) are usually found in-place and mark a paleomarsh surface more exactly than rhizomes (Allen, 1974) (Figure 5.2). Rhizomes may extend 10-15 centimeters below the sediment surface, while sheathing bases occur partially buried surface (Allen, 1974).

Monocot roots are usually adventitious, fibrous, branched, and occur at basal nodes of rhizomes and stolons (Allen, 1974). Stems and rhizomes are usually hollow cylinders with solid nodes. Leaves extend from the sheathing base which is present at the lower part of the stem and is a perennial portion of the plant. The ligule is a small membranous flap of tissue located where the leaf blade joins the sheathing base. Dicots possess a tap root system with lateral or branched roots which penetrate the sediment to greater depths than the adventitious root system of monocots (Allen, 1974).

Thus, plant fragments from cores were used to identify changes in plant species which indicated changes in marsh environment, i.e., low marsh environments (*S. alterniflora*), high marsh environments (*S. patens*, *D. spicata*), and high-marsh transition shrub environments (*I. frutescens*, *B. halimifolia*).

Identifying characteristics reported here are based on field and laboratory observations and on the literature cited, primarily Allen (1974):

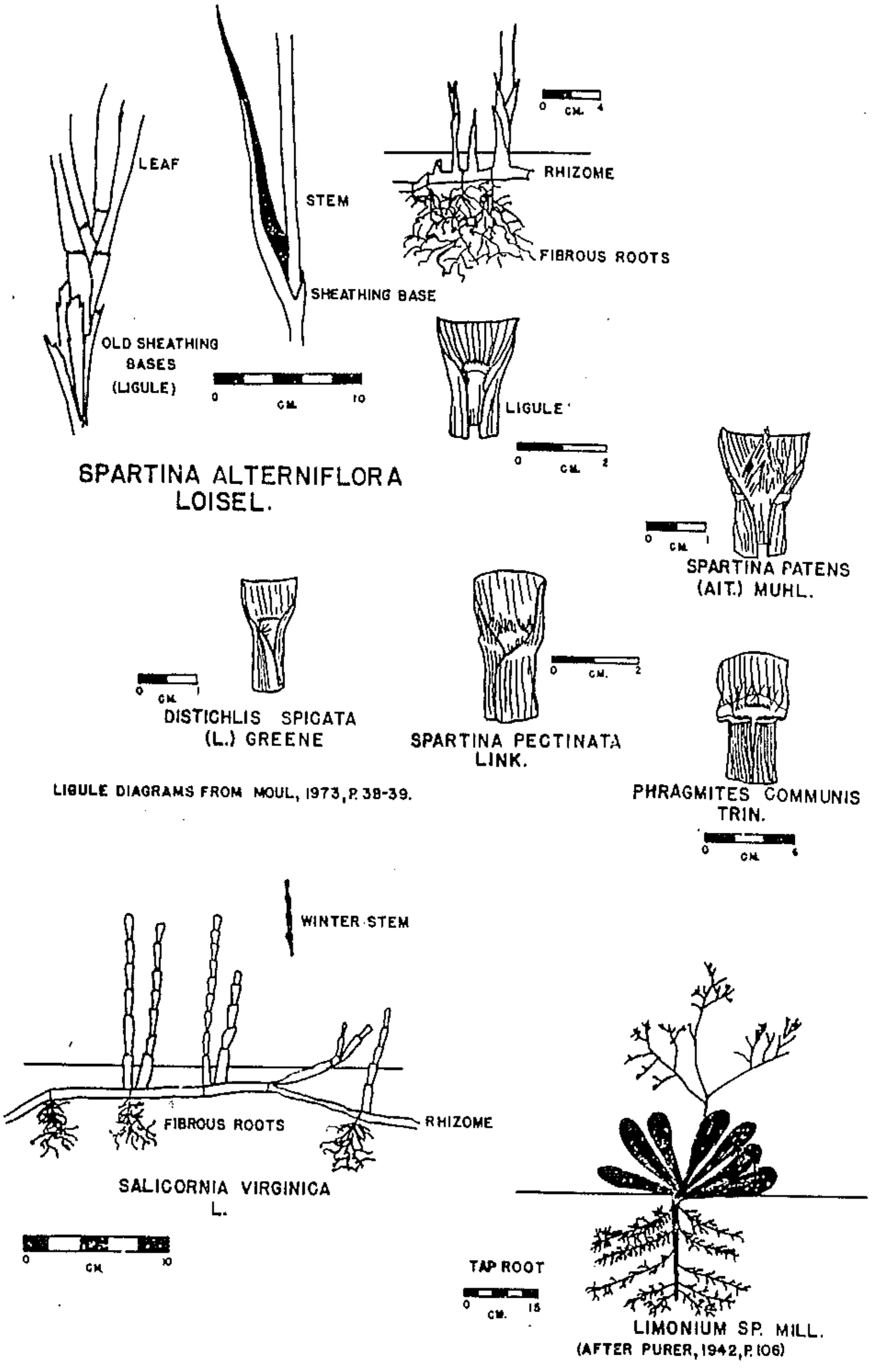


Figure 5.2 Line drawing depicting stem base and rhizomes and relationship to paleomarsh surface (from Allen, 1974).

1) *Spartina alterniflora* Loisel., a monocotyledon perennial grass, is typically 0.5 to 2.5 m tall, with broad (1-1.5 cm) leaves and a large, hollow-jointed stem (Figure 5.3). *S. alterniflora* is characterized by large, "woody" rhizomes covered with papery scales, and has a dense, fibrous root network which extends 15-30 cm below the surface. Rhizomes are typically greater than 2.5 mm but less than 10 mm in diameter, tannish-brown in color, and collapsed. Relatively long (1-2 cm) ensheathing scale-like leaves are present, with few small roots.

2) *Spartina patens* (Ait.) Muhl., also a monocotyledon perennial grass, has slender (1-3 mm) leaves approximately 0.2-0.4 m long (Figure 5.4). *S. patens* rhizomes are slender and tough; roots are fibrous and branching. *S. patens* rhizomes, typically tannish brown in color with ensheathing scale-scars, are smaller than those of *S. alterniflora*, ranging from 1.5 to 2.5 mm in diameter. *S. patens* peats typically consist of dense and fibrous organic matter with many fine roots.

3) *Distichlis spicata* (L.) Greene is a monocotyledon, perennial grass with leaf widths 2-3 mm, and length 0.5-1.5 m (Figure 5.4). *D. spicata* leaf sheaths overlap, and plant height is 0.2-0.6 m. Its slender, creeping tannish brown rhizomes, typically 2-4 mm in diameter, are characterized by conspicuous encircling scars or scales.

4) *Iva frutescens* L. and *Baccharis halimifolia* L. are dicotyledon woody shrubs (Figure 5.5). Leaves are either thick, and elliptical to broadly obovate (*Baccharis*) or distinctly toothed (*Iva*). These shrubs are characterized by a primary tap root system, easily distinguishable from the rhizomes and fibrous root network of the grasses.

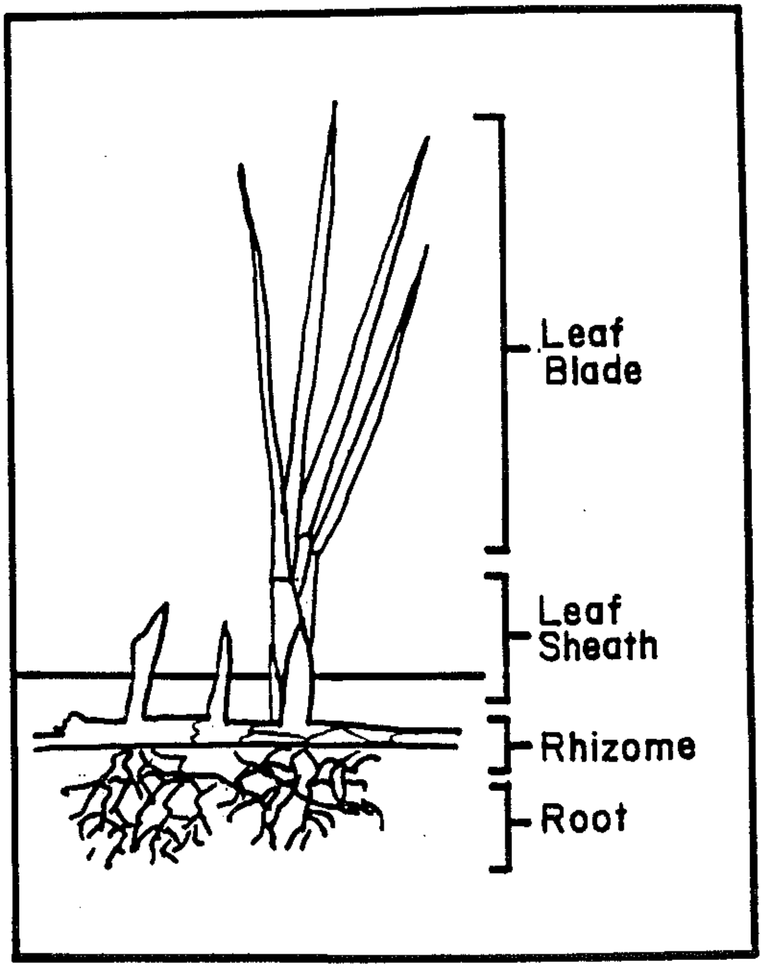


Figure 5.3 Line drawing of the morphology of *Spartina alterniflora* (15 cm to 2 m tall) (from Allen, 1977).

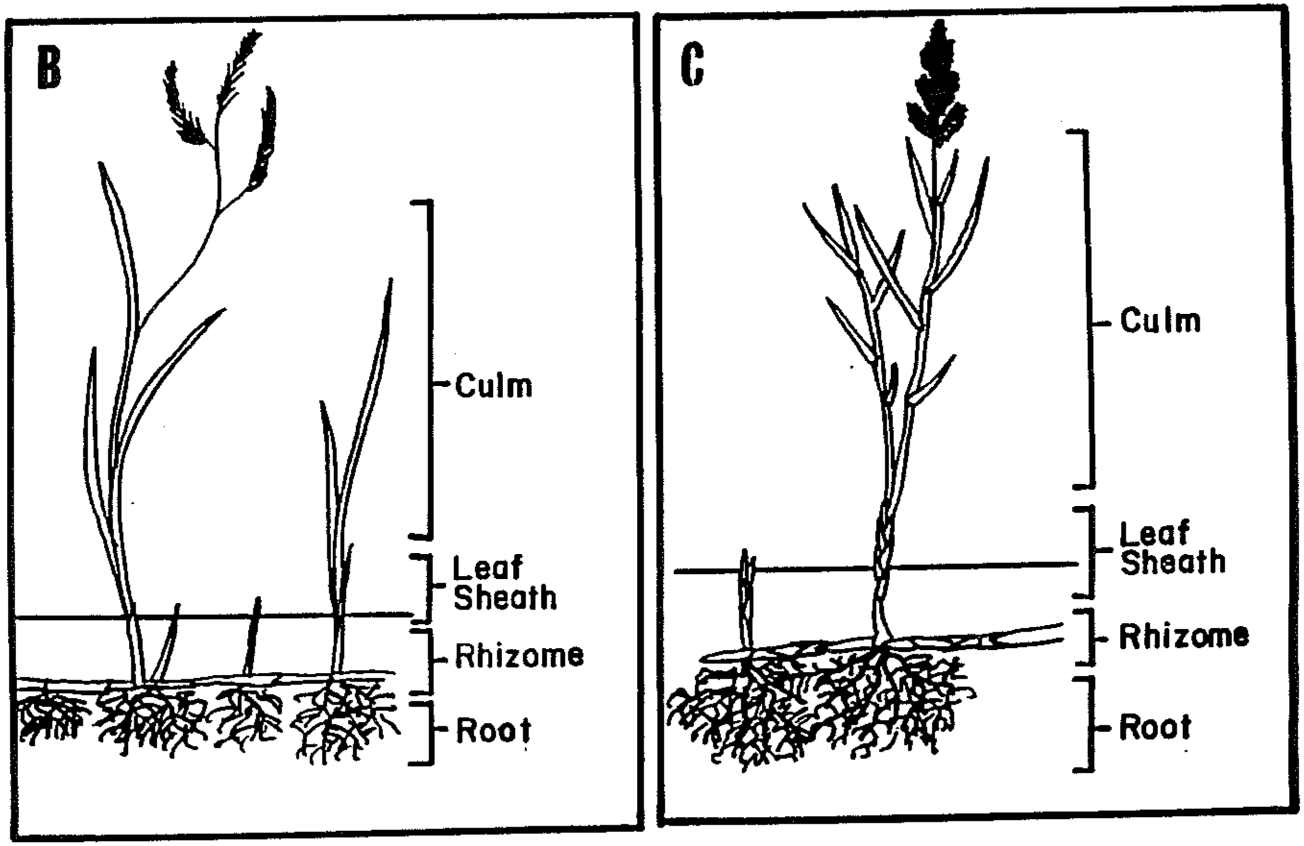


Figure 5.4 Line drawing of the morphology of (B) *Spartina patens* (15 cm to 35 cm tall) and (C) *Distichlis spicata* (15 cm to 50 cm tall) (from Allen, 1977).

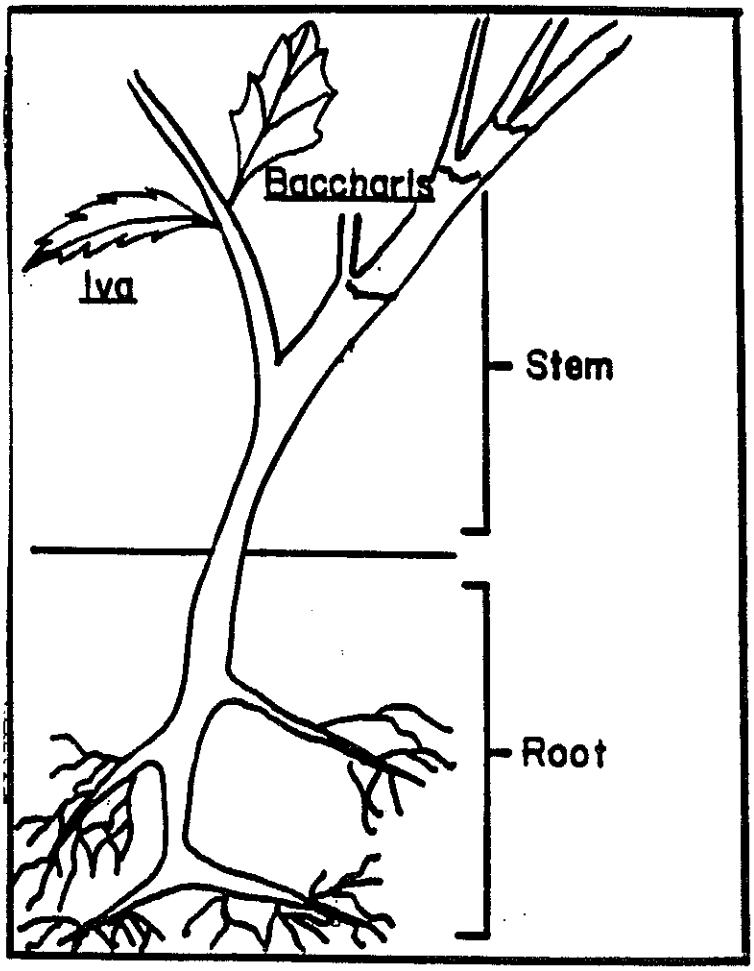


Figure 5.5 Line drawing of the morphology of *Iva frutescens* and *Baccharis halimifolia* (1 m to 3 m tall) (from Allen, 1977).

5) *Phragmites australis* (Cav.) Trin. ex Steud. rhizomes are recognized primarily by their size, typically greater than 10 mm in diameter (Figure 5.6).

Macroscopic identification of plant materials from surface and living subsurface samples were thus identified and recognizable, and remains of non-living subsurface salt marsh vegetation were then examined macroscopically to identify salt marsh environments in the subsurface and relate them to their modern environment.

Comments on Preservation of Vegetative Remains

Plant remains are well preserved in cores of the upper 1.5 m of salt marsh sediments, allowing identification by macroscopic examination. Changes in species composition with depth in the subsurface can be used to reconstruct the recent (100-300 year) evolution of the marsh by comparison with the surface zonation and application of Walther's Law.

The accumulation and potential preservation and of plant fragments is primarily related to the surface conditions and secondarily to depth and length of time buried (Allen, 1974). Allen (1974) reports that peat composition is controlled by processes which occur after the plants die and before they are incorporated into the peat. Chemical, biological, and physical processes contribute to the breakdown of vegetation on the marsh surface; the degree of degradation depends on the environmental conditions of a specific marsh zone. Allen (1974) reports that the subsurface portion of the plant (roots and rhizomes) is always more likely to be preserved than the aerial portion of the plant (stems and leaves). The aerial portion may be transported away by

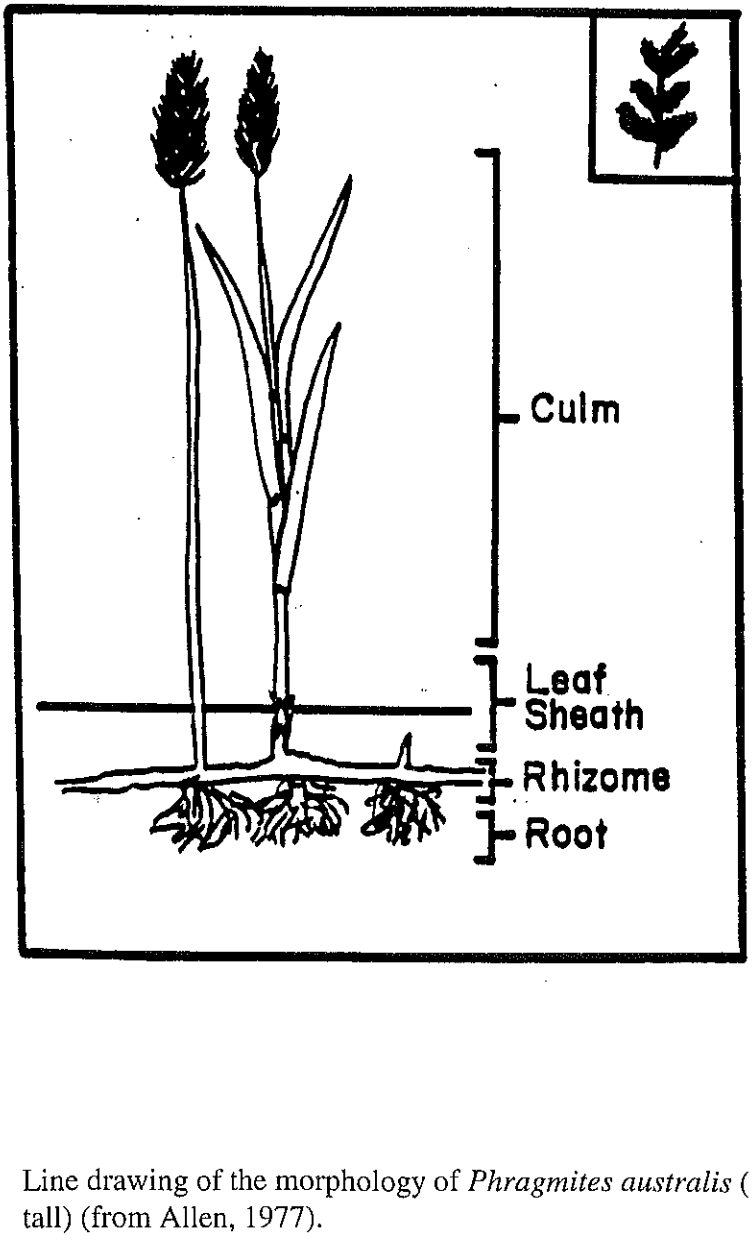


Figure 5.6 Line drawing of the morphology of *Phragmites australis* (15 cm to 2 m tall) (from Allen, 1977).

tidal waters, or may decompose before burial. Preservation potential is enhanced by rapid sedimentation, or reduced by erosion (normal tidal processes and storms) and decomposition (high heat, humidity, light, macro- and micro-organism activity) (Allen, 1974).

Plant material that is exposed on or above the marsh surface is less likely to be preserved or is preserved in a highly decayed state, while material that is intruded into the sediment such as roots and rhizomes has a greater chance of preservation in recognizable form (Allen, 1977). Many factors affect the preservation potential of plant remains, including: characteristics of the original material (e.g., fundamental chemical differences between tissues [Barghoorn, 1949a; 1949b]); water level; abundance and types of macro- and micro-organisms on surface and in subsurface; and chemical-physical-biological characteristics of sediments. In this study, vegetative remains were extremely well preserved in the upper meter of all cores, with rhizomes, stems, and even leaves easily identifiable. Between 1 and 1.5 m depths, rhizomatal fragments were still recognizable, but in smaller pieces; generally leaves and stems were not identifiable below 1 m. At approximately 2 m (usually the greatest depth penetrated by coring), fewer pieces of rhizomatal fragments were available for identification, while woody fragments were still intact. With increasing depth, preservation potential decreases to a point where positive identification of vegetative remains requires microscopic examination (as described by Allen [1977]).

In summary, the composition and degree of preservation of plant fragments in subsurface sediments depend upon biotic and abiotic factors such as sedimentation rate, tidal inundation patterns, storm events, climatic factors, and bacterial decomposition. Because this investigation involved only those sediments deposited most recently (300 years) in the marsh, macroscopic identification of plant remains was possible.

Characterization of Facies and Depositional Environments

Introduction

Environmental and sedimentary parameters were used to interpret the distribution of marsh facies in vertical and lateral sequences, and to infer the recent evolution of the wetlands/uplands boundary. Vegetative remains and sediment type (lithology) distinguish tidal wetland environments from terrestrial uplands, freshwater wetlands, and lagoonal environments. Although relative sea-level rise is a primary control, other external influences such as land clearing, agriculture, mosquito ditching, and dredging may be equally influential in controlling wetland development and evolution.

Based on differences in sedimentary characteristics, vegetative content, and spatial relationships, the recently deposited (last 300 years; upper 1.5 meters) Holocene marsh sequence found at the wetlands/uplands boundary is divided into 3 principal facies and 4 micro-facies:

1. Pre-transgressive sand facies
2. Soil transition-zone facies
3. Salt marsh facies

a. *Scirpus* spp./*Phragmites australis*; freshwater/brackish marsh facies

b. *Iva frutescens/Baccharis halimifolia*; transitional zone facies

c. *Spartina patens/Distichlis spicata*; high marsh facies

d. *Spartina alterniflora*; low marsh facies

This chapter describes the sedimentary characteristics of these facies and micro-facies based on macroscopic examination of vibracores and Eijelkamp cores obtained in this study. Additional descriptive characteristics of these sub-environments were obtained by examining modern surface wetlands environments, and from detailed analyses from previous studies (John, 1977; Carey, 1979; Chrzastowski, 1986; Yi, 1992). All grain size descriptions are based on the Wentworth (1922) classification; colors are wet colors referenced to the Munsell Soil Color Chart.

The primary criteria used for identification of sedimentary micro-facies are: 1) sedimentary texture (grain size, estimate of organic content, degree of plant decomposition); 2) color; 3) vegetative remains; and 4) lateral and vertical relationships. As reported by Yi (1992), color of sediments reflects a degree of organic material decomposition and amount of organic content in the study area. Therefore, dark gray, brown, and black colors usually indicate organic-rich sediments which are interpreted to be marsh and other wetland sediments. In contrast, gray to olive gray colors generally indicate organic-poor or inorganic sediments which are interpreted to be non-marsh sediments, such as lagoonal or estuarine sediments. The dominant plants used in identification of salt marsh sub-environments in the cores are listed in Table 5.1.

Table 5.1 Dominant plants used in identification of salt marsh environments.

Family	Species	Common Name
Compositae	*Baccharis halimifolia* L. *Iva frutescens* L.	Groundsel bush Marsh elder
Cyperaceae (sedges)	*Scirpus americanus* Pers. *Scirpus robustus* Pursh	Three square rush Bulrush
Gramineae (grasses)	*Distichlis spicata* (L.) Greene *Panicum virgatum* L. *Phragmites australis* *Spartina alterniflora* Loisel. *Spartina cynosuroides* (L.) Roth *Spartina patens* (Ait). Muhl.	Spike grass Switch grass Common or Giant Reed Smooth cordgrass Big cordgrass Salt Meadow Hay
Juncaceae	*Juncus gerardi* Loisel.	Black grass

Walther's Law (the correlation of lateral and vertical sedimentary sequences) was also used to interpret sub-environments.

Parameters Used in Identification of Facies and Sub-Environments

Several environmental and sedimentary parameters were utilized to characterize and identify sediments, depositional environments, and facies and micro-facies. Qualitative observations were made of the modern marsh environments and adjacent uplands, as well as macroscopic characteristics of sediments in core samples. Descriptive terms, such as high, intermediate and low, and ranges of values are relative and based on core- to- core comparisons. Many of the parameters in this study have been used successfully by other investigators (Allen, 1977; Chrzastowski, 1986; and Yi, 1992).

Substrate Parameters. Grain size was estimated macroscopically. Most marsh sediments are composed of silt and clay (Allen, 1974; Elliott, 1972), but sand is an important component near washover barriers, dune systems, upland surfaces, and along the banks of tidal creeks (Edwards, 1973; Basan, 1975; Allen, 1977). In this study, marsh sediments were generally comprised of silt and clay sized sediment, while adjacent upland surfaces were composed primarily of clean sand. Distinct sand layers were not found in cores. These discrete mineralogic layers typically represent periods of major deposition during storm events; these storm deposits are not recorded in the upper two meters of Delaware's fringing tidal salt marshes.

Mud facies were not subdivided into micro-facies in this study; fluvial mud micro-facies were not distinguished from tidal flat/tidal stream mud micro-facies, lagoonal mud micro-facies, or estuarine mud micro-facies. As Yi (1992) reported, the distinction between fluvial, tidal flat/tidal stream, lagoonal, and estuarine mud micro-facies is extremely difficult because each of these micro-facies is characterized by homogeneous inorganic gray-to-dark-gray muds.

Vegetation Parameters. As discussed previously, plant fragments were identified through macroscopic examination of organs or organ fragments, including roots, stems (rhizomes and culms), leaf sheaths, and leaves. The botanical term "stem" includes both the aerial stem, or culm, and the subsurface stem, or rhizome. Rhizomes have roots extending from nodes and thus can usually be distinguished from the culm macroscopically (Allen, 1977). Because the subsurface plant organs have higher preservation potential than the aerial organs, most stems found in the subsurface are rhizomes. The leaf sheath surrounds the stem, and joins the leaf at the abscission zone (Allen, 1977).

Salt marsh plants can be identified confidently from the microscopic morphology and anatomy of stems, as described by Allen (1977) and Niering *et al.* (1977). Subsequently, identification of changes in vertical sections can be used to document general conditions or sub-environments at the core site. A summary of distinguishing plant characteristics for various species is shown in Table 5.2; the fine rhizomes with distinct leaf scales of high-marsh species such as *S. patens* and *D. spicata* and can be readily distinguished from the larger, sheathed rhizomes of *S. alterniflora*.

Table 5.2 Summary of distinguishing plant characteristics used for identification of subsurface vegetative remains (after Allen, 1974).

Species	Rhizome diameter (mm)	Distinguishing characteristics
Spartina alterniflora	2.5-10.0	1. Center of rhizome is hollow, often collapsed. 2. Leaf sheaths are common in the sediment. 3. Orange mottling common around small roots. 4. Fragments light tan, but may be black.
Spartina patens	1.0-3.0	1. Rhizome is slender, straight; hollow center. 2. Dense interlocking network of roots and rhizomes or 3. Loosely bound sediment with laminations containing few roots and abundant stems.
Distichlis spicata	1.5-4.0	1. Rhizome is knobby with distinct leaf scales. 2. Rhizome has solid center. 3. Dense interlocking network of roots and rhizomes or 4. Loosely bound sediment with laminations containing few roots and abundant stems.
Phragmites australis	8.0-20.0	1. Stem has hollow center and is rigid. 2. Leaf sheaths are common in sediment.
Scirpus spp.	4.0-8.0	1. Rhizome has solid center and fragments are frequently black or dark orange. 2. Culm is triangular in some species. 3. *S. robustus* rhizome possesses a tuber.

This difference has often been used as a tool for interpreting fluctuations between high- and low-marsh environments in core sections (Mudge, 1858; Davis, 1910; Johnson, 1925; Knight, 1934; Clark and Patterson, 1985).

Characterization of Salt Marsh Environments of Deposition: Results and Discussion

Characteristics of Identified Sub-Environments and Facies

Pre-Transgressive Surface and Facies. The pre-transgressive surface represents those sediments deposited at a site before the sequence of marsh muds (freshwater marsh and tidal salt marsh) were deposited and is the surface over which the wetland is migrating. In most cases the pre-transgressive surface is the same as the pre-Holocene surface; the term pre-Holocene surface is not used because the subaerial surface underlying the marsh sediments may indeed represent part of Holocene time.

Pre-transgressive deposits were characterized by grain size, organic content, and stratigraphic position. Each of the five vibracores and most of the 400 Eijelkamp cores penetrated pre-transgressive sediments, which were then used to map antecedent topography. Depth to the pre-transgressive surface generally varied from 0.01 m to no more than 2.0 m, depending on proximity to the wetlands/uplands boundary and antecedent geometry. At sites near rivers and streams, and the associated incised valleys, cores greater than 2 m in length often did not penetrate the pre-transgressive surface due to the steep slope of the antecedent surface.

The three-dimensional morphology of the pre-transgressive surface is important for understanding the evolution, development, and lateral migration rates of fringing tidal wetlands. The pre-transgressive surface may be recognized as a soil zone or a mixture of marsh mud with pre-Holocene sand (Kraft, 1971a, 1971b), and/or on the basis of radiocarbon dates (Yi, 1992). Qualitative criteria for identifying the pre-transgressive surface (Kraft, 1971a, 1971b; Marx, 1981; Chrzastowski, 1986; Yi, 1992) include: 1) changes in sediment characteristics (mottling, oxidation); 2) high degree of compaction or cementation; 3) little or no organic material, or a significant decrease in the amount of organic material; 4) presence of clean, well-sorted sand; 5) oxidation resulting in orange coloration; 6) development of a soil horizon; 7) position relative to identified facies of geomorphic features; and 8) presence of basal peat.

All of the above criteria were utilized in this study to identify the pre-transgressive surface. Greater emphasis was placed on the presence of clean, well-sorted sand, decrease in amount of organic material, and distinctive changes in texture (sudden change of color, grain size, and organic content). The boundary between the pre-transgressive sediments and the overlying marsh sediments was typically clear and unmistakable. The author acknowledges that well-sorted sand below organic marsh sediments are likely to be pre-Holocene, but might be fluvial sand, headland beach sand, barrier sand, tidal delta sand, and tidal channel sand deposited during the Holocene, as discussed by Yi (1992).

The pre-transgressive sediments examined macroscopically in this investigation vary in grain size from coarse to very fine sand, silty sand, sandy silt, and sandy clay. These deposits are generally yellow, yellowish gray, orange, tan, and brown. Pebbles are also characteristic of pre-transgressive sediments, but are not found in every sequence. Modern analogs of these subsurface pre-transgressive sediments include soils/sediments of contiguous subaerial uplands. Upland soils at most sites are Evesboro, Sassafras, or Rumford sandy loams or loamy sands. Soil textures range from fine to coarse sand throughout the soil profile; the upper soil horizons are strongly acidic because of decaying oak and coniferous leaf litter.

Soil Transition-Zone Facies. The soil transition-zone facies is typically a highly reduced blue-black, thin (2-4 cm) sedimentary layer positioned stratigraphically between the underlying pre-transgressive sands and overlying marsh sediments. This layer is characterized by a distinctive dark blue-black color (Munsell Chart 2.5YN2/1) and trace amounts of organic material, although small vertical rootlets may penetrate this layer. Typical grain size and texture varies from muddy sand to sandy mud.

This unit has been described as a soil transition-zone because its stratigraphic position is analogous to that of the humus layer found in adjacent upland areas. However, it may not represent a true soil horizon. The distinctive black color of this facies may be due to very low redox potentials and reduced organic end products, including hydrogen sulfide, methane, and ferrous compounds.

Fringing Salt Marsh Facies. The following classification of salt marsh facies is based primarily on macroscopic identification of vegetative remains, but also on other properties of the surficial and subsurface deposits (Allen, 1977). Each of the four micro- or sub-facies corresponds to a modern depositional environment distinguished by a community of plants. Although these environments are generally distinct, they locally grade into one another and the facies may not always be separable; this characteristic was similarly observed by Allen (1977). Since only macroscopic identification of sediments and subsurface environments was utilized in this study, further detailed subdivisions of micro-facies was not possible.

Fringing salt marsh facies can be subdivided into four microfacies:

1) low marsh facies, *Spartina alterniflora*; 2) high marsh facies, *Spartina patens/Distichlis spicata*; 3) upland transitional facies, *Iva frutescens/Baccharis halimifolia*; and 4) freshwater/brackish marsh facies, *Scirpus* spp./*Phragmites australis*. A description of macroscopic characteristics and depositional environment for each subfacies follows.

Low Marsh Sub-facies, *Spartina alterniflora*.

Low Marsh Depositional Environment: The low marsh is exposed to regular wave attack and tidal currents. Plant fragments transported by tidal flow contribute a large volume of detritus to nearby tidal creeks and estuaries. The low marsh is usually densely populated by fiddler crabs (*Uca pugnax* and *U. pugilator*) and larger mud crabs such as *Sesmarma* spp. Active burrowing by crabs, especially adjacent to creek banks, disturbs lower peat layers (20-25 cm to 1 m depths), mixes subsurface layers, and contributes to erosion on wave-cut marsh banks (Allen, 1974). Additionally, invertebrate populations contribute to the breakdown of plants on the marsh surface. *S. alterniflora* is the dominant plant of the low marsh; its roots and rhizomes form a dense mat which is resistant to erosion. The large size and sturdy nature of this plant increases its chance of preservation (Allen, 1974). Although most leaves and stems from each year's growth are consumed by herbivores and detritivores or transported away by tides, the roots, rhizomes and basal stem of the plant remain in place unless severe erosion occurs.

Characteristics of Low Marsh Facies: The low marsh subenvironment is characterized by muddy silt and clay, with a very minor quantity of sand. Plant roots and rhizomes are extensive, binding and lending coherence to the substrate. This substrate is poorly drained, with tidal waters remaining ponded in some low-lying areas. *Spartina* vegetation is less dense, or may be absent (substrate may be barren of vegetation) in these ponded areas. Physical sedimentary structures were not observed in cores taken in this subenvironment due to the homogeneous nature of sediment, root mat, and vegetation type. Evidence of bioturbation is virtually non-existent in the *S. alterniflora* environments at the wetlands/uplands boundary area, although shells of *Geukensia demissa* (ribbed mussel) and burrows of *Uca pugnax* (fiddler crab) are found adjacent to ditches. Low marsh environments include low marsh *S. alterniflora* zones which cover extensive areas of intertidal marsh zones. *S. alterniflora* low marsh sediments comprised of mud (silts and clays), are generally extremely fibrous and high in organic content; sediments are brown to dark brown in color, and may exhibit bright orange mottles around roots. A dense network of roots and rhizomes is characteristic of these deposits. Rhizomes are identifiable by size, node characteristics, and the presence of a hollow interior in cross-section. Roots and rhizomes are tan, yellow, dark brown, or transparent. Broad sheet-like fragments of *S. alterniflora* leaf sheaths or leaves, and well-preserved basal culms are common in upper sections of marsh sediments.

S. alterniflora low-marsh sediments show little degradation of organic material because the sediment consists primarily of subsurface components (roots and rhizomes) and inorganic particles (Allen, 1977). *S. alterniflora* leaves are not a major portion of subsurface sediments because much of the above-ground detrital material is removed by tides (Allen, 1977).

High Marsh Sub-facies, *Spartina patens/Distichlis spicata*.

High Marsh Environment: Plants and plant material in the high marsh are less likely to be washed away by the tides than those in the low marsh, but they are subjected to high humidity and heat at the surface and aerobic bacteria which accelerate decomposition (Allen, 1974). However, cowlicks of grass formed by *S. patens* near the end of the growing season are high temperature, high humidity environments where decomposition by aerobic bacteria and fungi is accelerated. Portions of the previous year's growth are still present the next growing season, indicating that the leaf and stem material escape total and rapid decay. The preservation potential of *D. spicata* is similar to that of *S. patens* (Allen, 1974).

Characteristics of High Marsh Facies: Sediments of the high marsh are generally uniform, consisting primarily of slightly muddy to 'clean' sand overlain by approximately 20-30 cm of a muddier dense mat of *S. patens/D. spicata* roots and rhizomes. Substrates in the high marsh subenvironment are firm, and although one may infer that the sandy substrates are permeable and aerated, few field data of the degree of waterlogging and subsurface saturation in this area exist. Qualitative field observations suggest that these sediments are frequently saturated by high water tables. Although frequency and duration of tidal inundation on the marsh surface may be limited, waterlogging may impact this subenvironment. Physical sedimentary structures were not observed, and bioturbation is assumed to be minimal except for occasional burrows of *Uca* spp.

High marsh environments sampled in this study include *S. patens* zones, *D. spicata* zones, and mixed *S. patens/D. spicata* zones; there was no attempt made to separate *S. patens* and *D. spicata* plant remains. These sediments are finely fibrous, with a very dense network of roots, rhizomes, and leaf sheaths. Roots and rhizomes are generally well preserved, yet highly decayed vegetation fragments can be found with increasing depth. Culms and culm fragments may also be distinguished, and are recognizable as slender stems without root nodes. Sediments are brown to dark brown, with light brown matrices indicating minor decay. Vegetative remains are typically very well preserved at the surface, while only rhizomatal fragments may be found in sections buried 1-2 meters below the surface. Sediment texture is sandy mud, generally coarser than that found in *S. alterniflora* zones, although muddy sediments may also be characteristic.

High-marsh environments are flooded only by storm tides and spring high tides; therefore they are exposed to aerobic oxidizing conditions most of the time, and are less likely to receive sediment from tidal action than low-marsh environments (Allen, 1977). Vegetative preservation potential thus varies with tidal sedimentation and surficial conditions such as temperature and degree of subaerial exposure. For example, high marsh sediments would decay more quickly during periods of low tidal sedimentation and dry conditions (aerobic decay would be high); slower rates of decay would occur when rate of accumulation of organic or mineral sediments is high enough so that surface litter is buried before decomposition (Allen, 1977). Essentially, the oxygenation of the sediment controls the preservation of organic material in this sub-facies.

The transition between *S. alterniflora* (low marsh) and *S. patens/D. spicata* (high marsh) in surface and subsurface environments may be either sharp or gradational. Although *S. alterniflora* typically develops a monospecific community and *S. patens/D. spicata* associations are usually pure, the plant types do coexist (Allen, 1977). This association is most common at the transitional zone between low marsh and high marsh, and is perhaps more often recorded in transgressive subsurface sediments than in surface environments. In the subsurface, this subfacies includes characteristics of both low- and high-marsh environments; i.e., rhizomatal fragments of *S. alterniflora* mixed with *S. patens* and/or *D. spicata*.

Upland Transitional Sub-facies, *Iva frutescens/Baccharis halimifolia*.

Upland Transitional Environment: Plants in the upland transitional environment are only affected by the extremes of spring tides and storm events. This environment is more directly impacted by adjacent upland characteristics such as groundwater and surface water runoff, sediment supply, and slope. If the adjacent upland is adequately stabilized with vegetation, little or no mineral sediment is likely to be transported directly into this zone; accumulation of fine mineral sediment may occur during episodic storm conditions. Similarly, the organic component of sedimentation in this zone is minimal due to characteristics of the woody, shrubby plants themselves. Although *S. patens/D. spicata* grasses are occasionally associated with the shrubby transition zone, shading by *B. halimifolia*, *I. frutescens*, and upland trees and shrubs limits light penetration to the ground surface.

Characteristics of Upland Transitional Facies: *I. frutescens* and *B. halimifolia* sediments are identified by very low organic content except for large woody fragments of branches, stems, and/or tap roots. Tap root systems and woody branches are easily identifiable in the subsurface. The majority of *I. frutescens* and *B. halimifolia* that fall to the marsh surface decompose quickly unless rapidly buried (Allen, 1974). Sedimentary texture is typically granular and sandy, but poorly sorted, as high amounts of organic sediment and fine-grained mineral sediment may accumulate in the narrow zoned occupied by these species. Sediments are typically described as 'mucky' and 'soupy', with few roots, rootlets, and rhizomes to hold it together. Sediments range from dark gray to black to dark red brown, suggesting high rates of decay; the high organic content in the coarse mineral sediment is likely related to formation and decomposition of surface litter. Surface litter is rarely removed by tides and sediment accumulation rates are low.

Freshwater/Brackish Marsh Sub-facies, *Phragmites australis/Scirpus* spp..

Freshwater/Brackish Marsh Depositional Environment: This sub-facies occupies a similar environment to that of the shrubby upland transitional facies in that it is only impacted by extreme tides, and is perhaps more directly influenced by adjacent upland conditions (groundwater, surface water runoff, sediment supply, slope, etc.). Shading and competition by both *Phragmites australis* and *Scirpus* limit growth of high marsh grasses *S. patens* and *D. spicata.*

Freshwater/Brackish Marsh Facies Characteristics: *P. australis* may be abundant in the upland transitional sub-facies zone. *P. australis* sediments are typically sandy and granular, but like the *I. frutescens* and *B. halimifolia* zones, are also loose and appear to be composed of highly decayed organic 'soup'. Roots, rhizomes, and culm fragments are abundant and contribute to the cohesiveness of the sediment. Rhizomes are rigid, hollow (except at nodes), large (up to 20 mm in diameter), and easily identifiable. Leaves are not commonly observed in subsurface sediments, but roots and rhizomes are typically well preserved. Vegetative remains are tan to brownish, while sediment color is similar to that of *I. frutescens* and *B. halimifolia* zones, ranging from dark grey to black to reddish brown (indicative of high rates of decomposition and decay). In addition, oxidizing conditions may exist in the more porous sandy material found at the wetlands/uplands boundary.

Scirpus spp., typical of brackish marsh systems, is sometimes found intermixed in the upper edges of *S. patens/D. spicata* zones in the transitional area between salt marsh and upland, usually where there is greater influence of freshwater. *Scirpus americanus* and/or *Scirpus robustus* plant fragments are easily recognizable due to their sharply triangular culm, pronounced sheathing base, and rhizomes which commonly forms tubers. *Scirpus* fragments are usually reddish brown or black in color.

General Stratigraphic Patterns

Stratigraphic cross-sections for the 69 sites are included in Appendix E. Several representative sites have been selected to illustrate stratigraphic patterns and are included in the following discussion.

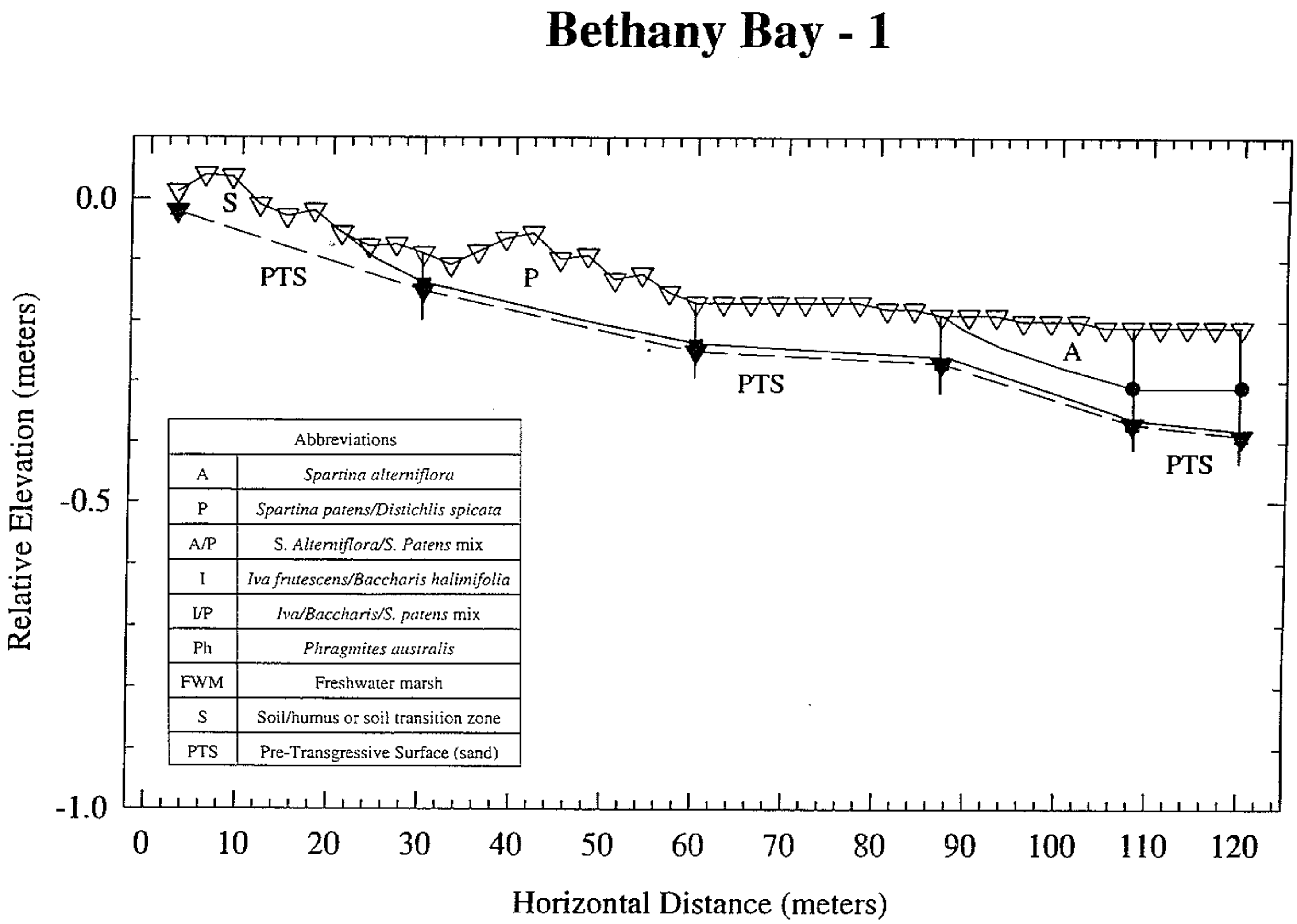
In the study area, restricted to the wetlands/uplands boundary zone, simple stratigraphic relationships can be defined: recently developed salt marshes directly overlie pre-transgressive sands, or previously terrestrial upland surfaces. At all sample sites, stratigraphy was 'transgressive' in nature with no evidence of seaward progradation; i.e., pre-transgressive sediments were directly overlain by high marsh deposits, which were overlain by low marsh deposits. At the wetlands/uplands border, the narrow thickness of salt marsh sediments is likely due to the fact that salt water influence has occurred recently (200-300 years). Before encroachment of salt water, the study area had been subaerially exposed, colonized by terrestrial vegetation, as is evidenced by numerous sites with remnant tree stumps, some identified by bark fragments as loblolly pine (see photograph, Figure 5.7). The slope of adjacent and antecedent topography has a direct effect on the nature and extent of salt marsh encroachment, as is demonstrated by the following transect descriptions.

Figure 5.7 Photograph of remnant tree stumps in marsh area presently colonized by *S. patens* vegetation.

Bethany Bay - 1. The cross-section for the Bethany Bay-1 site (Figure 5.8) shows a simple stratigraphic sequence of salt marsh encroachment over a shallow-sloping upland. The *S. patens* unit directly overlies pre-transgressive sands, and is overlain by *S. alterniflora*-dominated sediments which thicken bayward. High marsh facies have developed over previously subaerially exposed terrestrial uplands, and low marsh facies have developed over high marsh facies. This is a good example of the broad expanse of *S. patens* marsh typically found over wide, shallow sloping antecedent pre-transgressive surfaces.

Bay Vista - 1. In contrast, the Bay Vista-1 cross-section (Figure 5.9) depicts a more steeply sloping adjacent upland and antecedent topography. In this transect, the *S. patens* zone is absent at the surface and in subsurface facies. A narrow band of *I. frutescens* is present at the base of the steeply sloping terrestrial upland, and *S. alterniflora* is presently encroaching into the shrub environment. Stratigraphically, only two salt marsh facies are recognized; in the subsurface, *S. alterniflora* remains are dominant, with no evidence of *S. patens/D. spicata* (high marsh). *S. alterniflora* (low marsh) sediments directly overlie sediments with woody remains of *I. frutescens* (shrub/transition zone).

Wolfe Runne - 2. This cross-sectional transect (Figure 5.10) is similarly characterized by steeply sloping adjacent and antecedent topography. However, because the site location is farther from salt water influence, the wetland is brackish and dominated by *P. australis*. It should be noted, however, that there is no high marsh



Abbreviations	
A	*Spartina alterniflora*
P	*Spartina patens/Distichlis spicata*
A/P	S. *Alterniflora/S. Patens* mix
I	*Iva frutescens/Baccharis halimifolia*
I/P	*Iva/Baccharis/S. patens* mix
Ph	*Phragmites australis*
FWM	Freshwater marsh
S	Soil/humus or soil transition zone
PTS	Pre-Transgressive Surface (sand)

Figure 5.8 Subsurface stratigraphy: Bethany Bay cross-section. See Figure 2.8 for study site location.

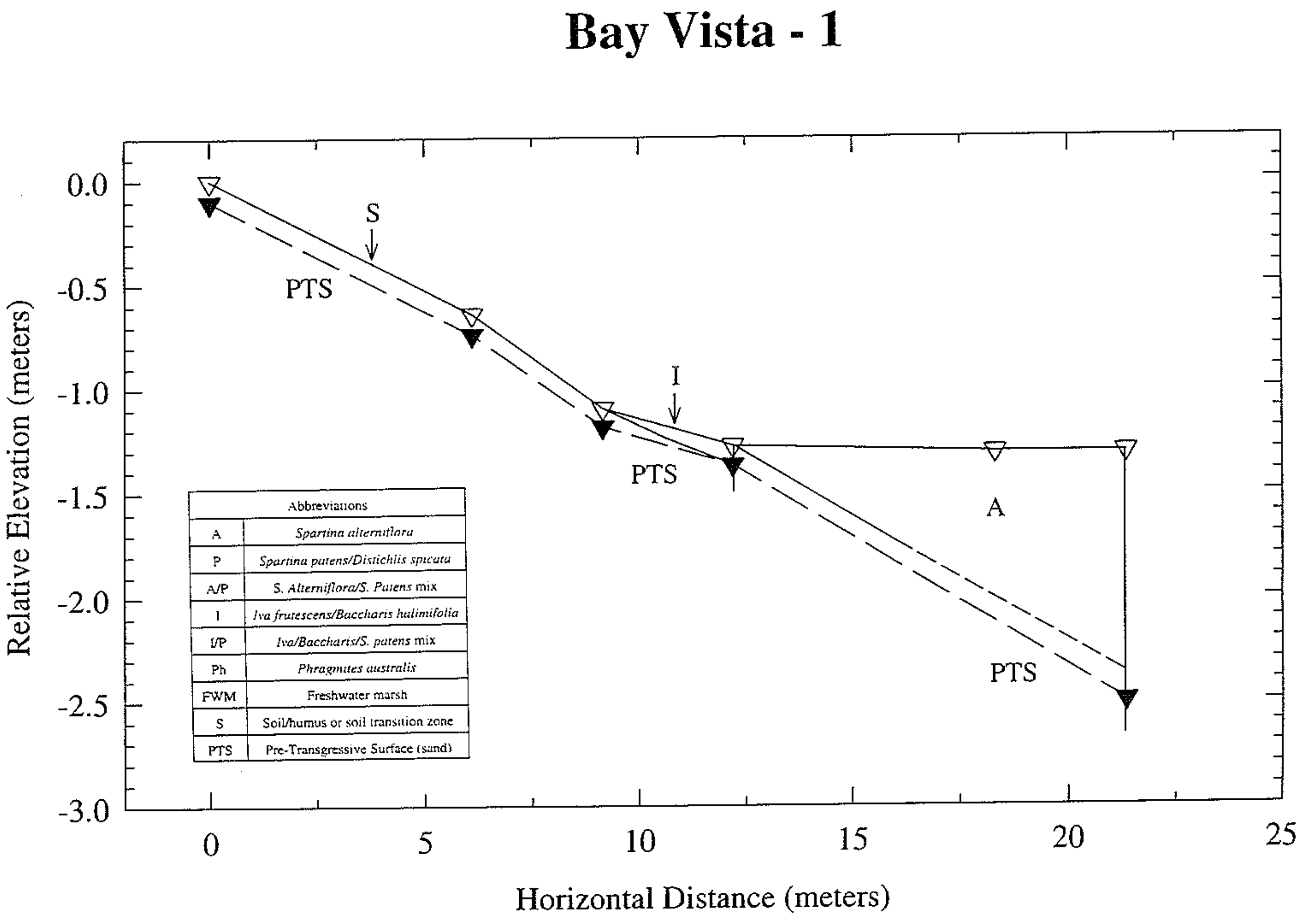


Figure 5.9 Subsurface stratigraphy: Bay Vista-1 cross-section. See Figure 2.7 for study site location.

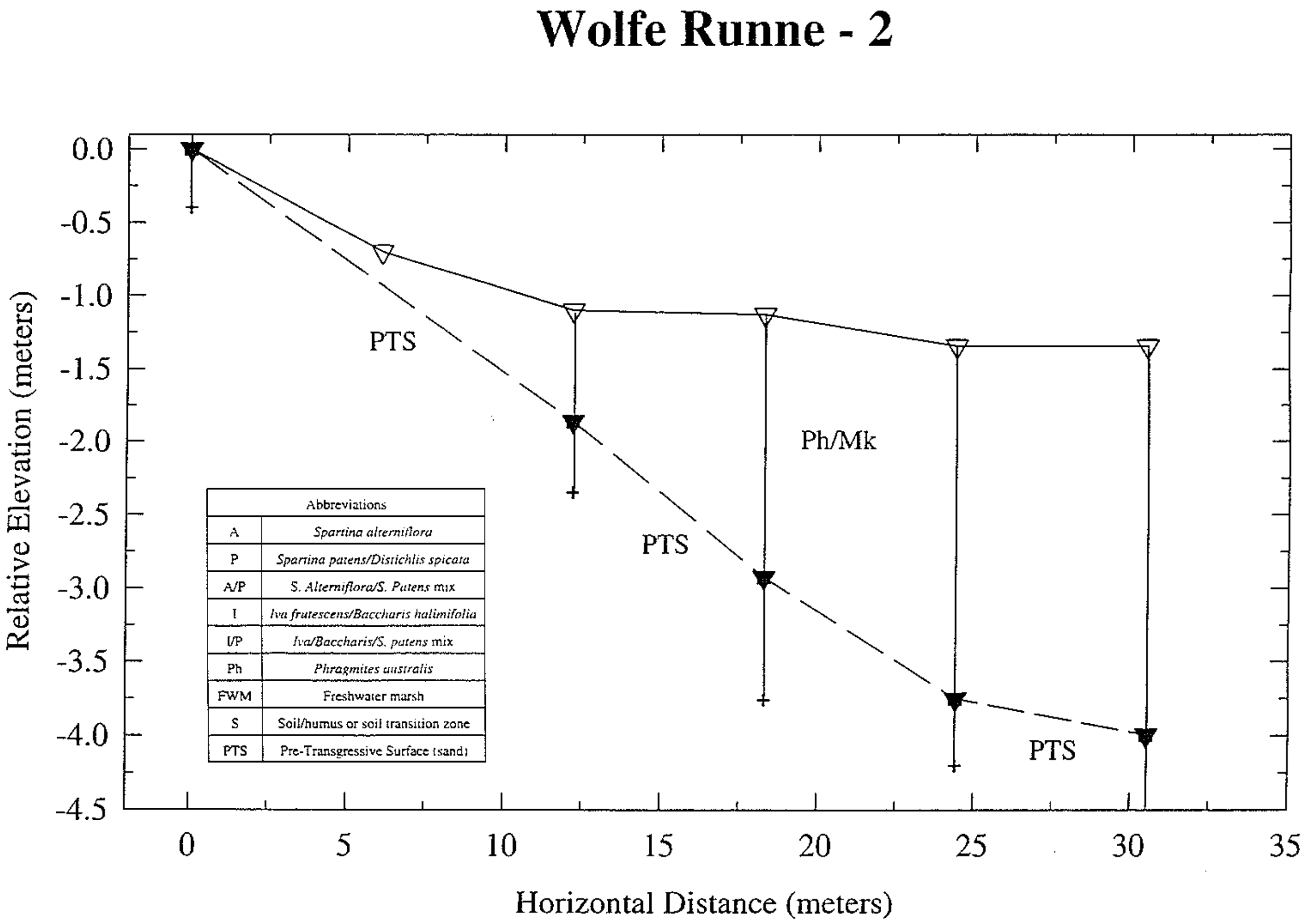


Figure 5.10 Subsurface stratigraphy: Wolfe Runne-2 cross-section. See Figure 2.6 for study site location.

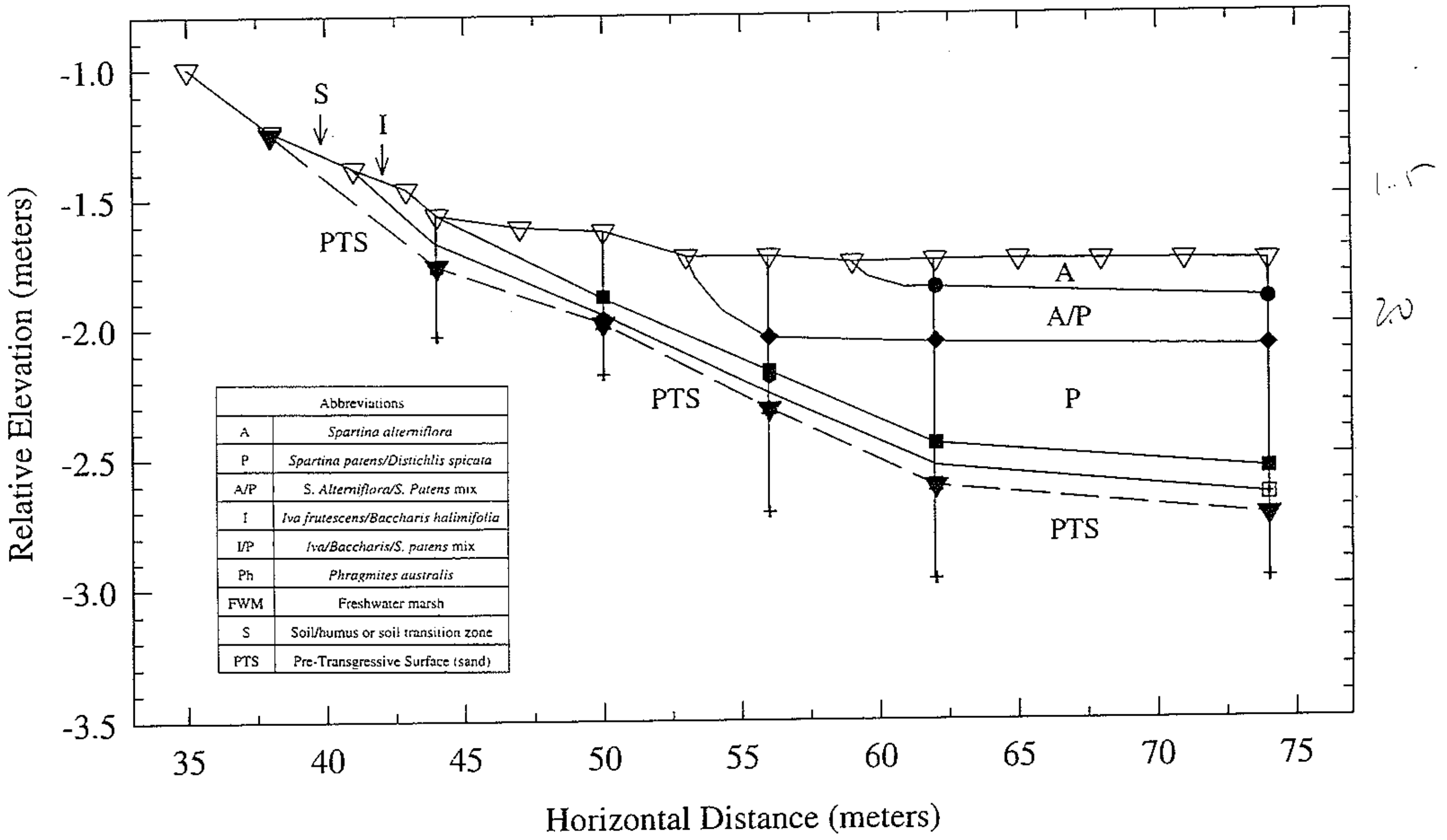
environment or shrub/transition zone present. The *P. australis* brackish marsh facies directly overlies the pre-transgressive surface.

Delaware Wildlands - 2. All major salt marsh environments are represented in this cross-section and transect (Figure 5.11), located along the western shore of Rehoboth Bay, and it provides an illustration of Walther's Law. On the surface, this transect depicts a gradual transition from uplands to shrub/transition zone, through a *S. patens/D. spicata* high marsh, a mixed *S. patens/S. alterniflora* zone, and ultimately, a *S. alterniflora* dominated low marsh environment. All salt marsh sub-environments are represented in the subsurface stratigraphy. Low marsh environments overlie high marsh environments, which overlie the pre-transgressive sandy surface; this sequence suggests that saline water is encroaching in this area.

Ritter - 2. This cross-sectional transect (Figure 5.12) similarly depicts all major salt marsh surface and subsurface zonation patterns. A narrow *I. frutescens/B. halimifolia* shrub/transition zone is located adjacent to the terrestrial upland. Adjacent to and above the shrub zone, *S. patens/D. spicata* high marsh environments are found, which are in turn overlain by a narrow *S. patens/S. alterniflora* mixed zone, and finally a *S. alterniflora*-dominated low marsh zone. Again, this sequence is indicative of a wetter, more saline environment.

Pot Nets North - 5. This cross-section (Figure 5.13) represents another example of the transgressive nature of the wetlands/uplands boundary. A much wider expanse of *S. patens/D. spicata* high marsh forms on the broad, shallowly sloping adjacent upland.

Delaware Wildlands - 2



Abbreviations	
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A/P	S. *Alterniflora/S. Patens* mix
I	*Iva frutescens/Baccharis halimifolia*
I/P	*Iva/Baccharis/S. patens* mix
Ph	*Phragmites australis*
FWM	Freshwater marsh
S	Soil/humus or soil transition zone
PTS	Pre-Transgressive Surface (sand)

Figure 5.11 Subsurface stratigraphy: Delaware Wildlands-2 cross-section. Figure 2.7 for study site location.

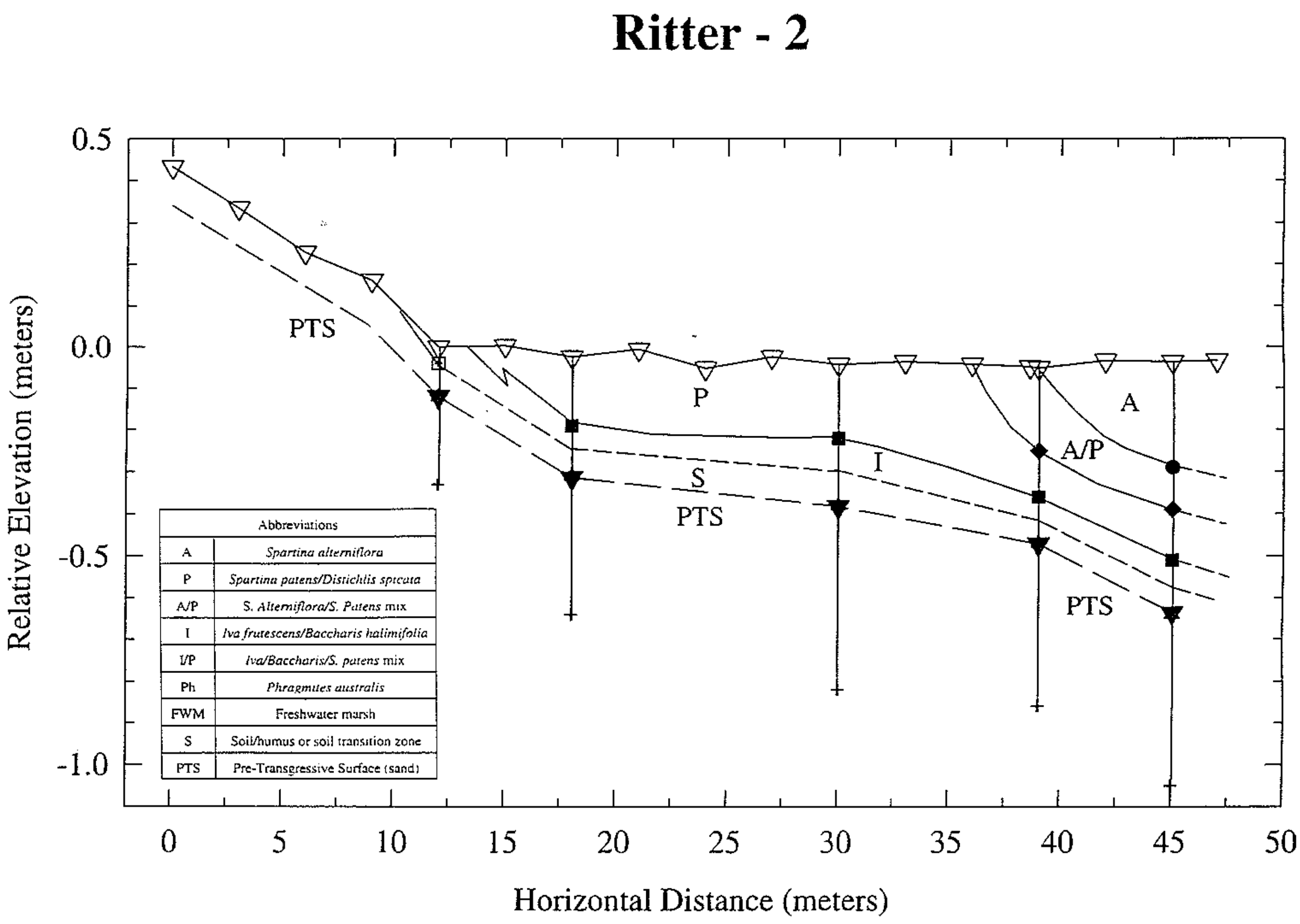
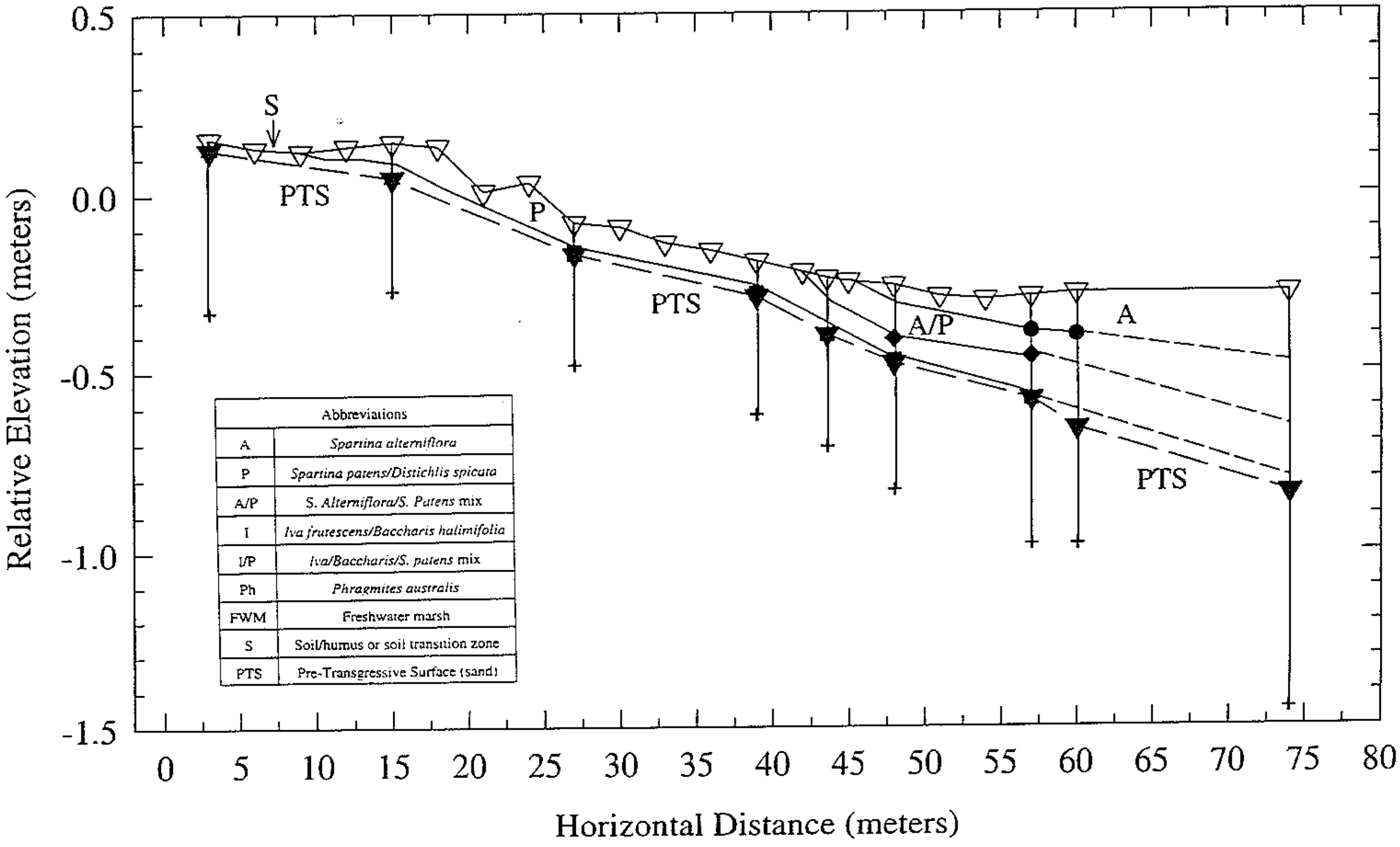


Figure 5.12 Subsurface stratigraphy: Ritter-2 cross-section. See Figure 2.6 for study site location.

Pot Nets North - 5



Abbreviations	
A	*Spartina alterniflora*
P	*Spartina patens/Distichlis spicata*
A/P	*S. Alterniflora/S. Patens* mix
I	*Iva frutescens/Baccharis halimifolia*
I/P	*Iva/Baccharis/S. patens* mix
Ph	*Phragmites australis*
FWM	Freshwater marsh
S	Soil/humus or soil transition zone
PTS	Pre-Transgressive Surface (sand)

Figure 5.13 Subsurface stratigraphy: Pot Nets North-5 cross-section. See Figure 2.7 for study site location.

Boat House Cove - 2. In contrast, the transect and cross-section depicted in Figure 5.14 illustrates how a steeply sloping upland surface does not provide conditions necessary for development of a broad *S. patens/D. spicata* high marsh environment. Although *S. patens/D. spicata* remains have been identified in the subsurface, high marsh vegetation remains are found overlying shallow-sloping antecedent topography. That is, the high marsh sub-facies is identified in the subsurface where the pre-transgressive slope is shallow but not where the slope is steep. *S. alterniflora* (low marsh) characterizes surface sedimentary units, indicating increasing inundation by salt water over time.

Thickness of Subsurface *S. patens/D. spicata* Unit

Comparison of thickness of surface and subsurface *S. patens/D. spicata* units might yield interesting information regarding present and past environmental conditions affecting high marsh vegetative zonation patterns. That is, in the absence of compaction, erosion, and bioturbation (none of which should have a significant effect on the high marsh environment), the distribution of buried thicknesses of *S. patens* units should be similar to present unburied thicknesses (J. Pizzuto, personal communication). The thickness of subsurface *S. patens* units are tabulated in Appendix F. In determining thickness of subsurface *S. patens* units, it was assumed that *S. alterniflora/S. patens* mixed zones in the subsurface were once *S. patens* zones (mixing occurred as *S. alterniflora* transgressed landward and upward over previously existing *S. patens* environments); thus, thickness of mixed zones is included in the *S. patens* zone

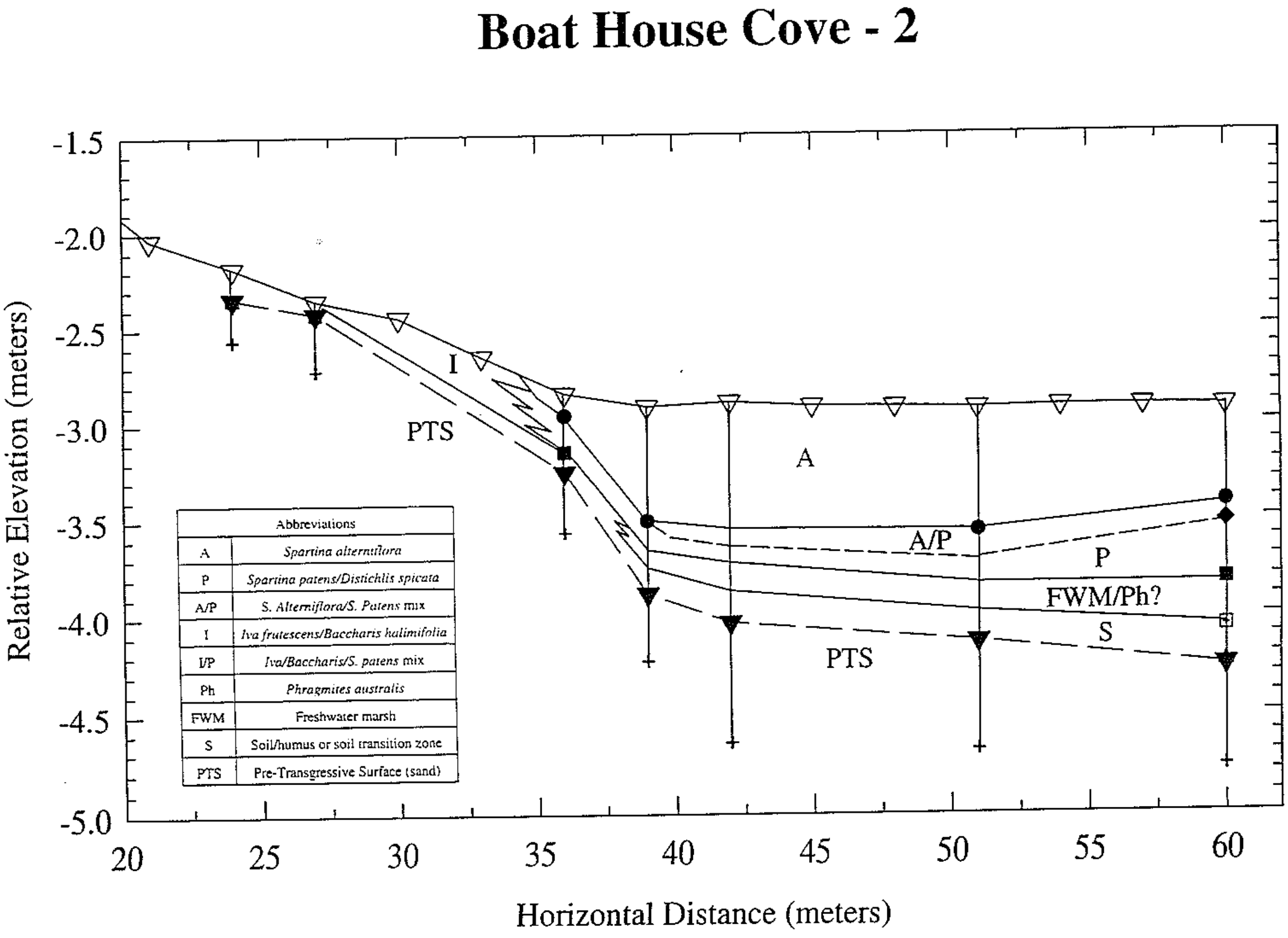


Figure 5.14 Subsurface stratigraphy: Boat House Cove-2 cross-section. See Figure 2.8 for study site location.

average. For the 160 subsurface *S. patens* units measured in cores, mean unit thickness is 24.72 ± 1.50 cm, which is comparable to mean surface-unit thickness (n=151) of 22.44 ± 1.54 cm. Table 5.3 shows general statistics for subsurface thickness of *S. patens* units; a Kruskal-Wallis one-way analysis of variance for the 311 cases shows that the thickness of subsurface *S. patens* units are <u>not</u> significantly different from the present-day surface *S. patens* environments.

Table 5.3 General statistics for subsurface thickness of *S. patens* units.

All Transects/Cores	Thickness *S. patens* unit (cm)
Sample Size	160
Mean ± standard error	**24.72 ± 1.50**
Standard Deviation	19.02
Maximum	110.00
Minimum	2.00

Figure 5.15 is a histogram depicting frequency of occurrence of various subsurface *S. patens* unit thicknesses.

The similarity in mean thickness of *S. patens* in surface and subsurface sediments suggests that at least until the present time, *S. patens* environments have been evolving in a similar fashion over the last 200 years or so. Thus, because the distribution of present day *S. patens* environments is also associated with the nature of the antecedent topography (depth to pre-transgressive sands as well as slope of

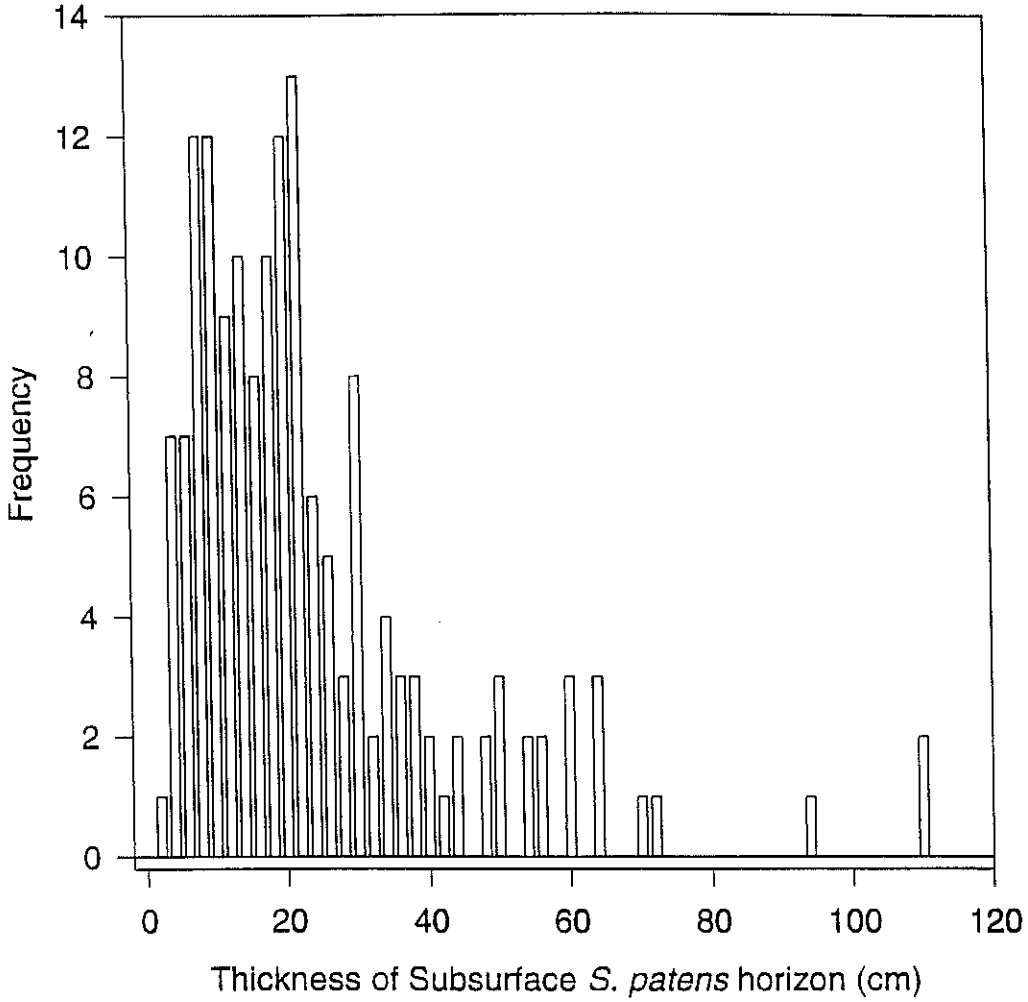


Figure 5.15 Histogram of frequency of occurrence of various thicknesses of subsurface *S. patens* units.

antecedent and adjacent surfaces), it may be possible to model the distribution of historic *S. patens* environments. Existence of *S. patens* marsh environments appears to be restricted to areas where shallow sloping sandy upland surfaces are adjacent to areas influenced by salt water; in modern environments where steeply sloping adjacent or antecedent geometries are found, the areal extent of *S. patens* environments is severely limited. This holds true for past high marsh environments as well, as supported by the stratigraphy, and will be modeled as part of the synthesis in Chapter 8.

Stratigraphic Patterns: Orientation of Contacts Between Subsurface Units

It has been noted that in transect cross-sections, some of the contacts between low and high marsh, and occasionally other units, curve upwards as they approach the ground surface (J. Pizzuto, personal communication). If the form of the shallow contacts is actually preserved, then a recent change in the orientation of this contact would imply a change in the rate of transgression; in this case, to slower transgression (J. Pizzuto, personal communication). It is felt that due to potential preservation effects and underground biomass dynamics of wetlands vegetation, detailed interpretations and conclusions regarding rates of transgression cannot be inferred from the form of the contacts as depicted in the cross sections.

In this study, detailed inferences regarding rates of transgression or identification of episodes of transgression and regression are not made from generalized stratigraphic profiles depicting vegetation changes with depth at the wetlands/uplands boundary for the following reasons:

Macroscopic Evaluation of Vegetative Plant Remains. This project involved characterization of trends observed in surface and subsurface salt marsh environments. The primary question addressed in the study: Is there a change in salt marsh sub-environments (as determined by plant species distribution) with depth in the subsurface at the wetlands/uplands boundary? Plant identifications are made through macroscopic evaluation of roots, rhizomes and stems, but not by microscopic examination (Allen, 1974, 1977). Additionally, although it is sometimes evident which rhizomes are alive and which are dead, specific methods and techniques for separation of living and non-living plant materials were not applied. Thus, the distribution of living and non-living rhizomatal remains was not established in subsurface cores; this would also affect the thickness of subsurface units mapped.

Preservation Effects.

1) Possible erosion of *S. patens* surface: In many transect locations, the transition from the *S. patens/D. spicata* zone to the *S. alterniflora* zone is characterized by hummocky topography. Typically, *S. patens/D. spicata* vegetation occurs on small hummocks or tussocks of higher elevation than surrounding *S. alterniflora*. There may be an erosive mechanism which acts to lower the elevation of the wetland surface as transgression continues, thereby creating an environment more suitable for *S. alterniflora* colonization. Although the physical process of surface elevation reduction was not examined in this study, possible mechanisms could include: surface runnels caused by upland runoff, removal of sediment during storm conditions when tides and waves may impinge on the high marsh/low marsh border, and biological impacts such as

burrowing and channelization by mummichogs and other fish as the surface of the marsh is flooded due to water level increases over time.

Erosion (or reduction in elevation) of high marsh surfaces would result in stratigraphic relationships depicted in many profiles from this study; the 'apparent' spatial and temporal distribution of high marsh and low marsh environments could thus be misinterpreted.

2) Possible dominance of *S. alterniflora* rhizomatal remains over thinner *S. patens* remains: As low marsh environments transgress over high marsh environments, remains of *S. alterniflora* rhizomes and stems may overwhelm vegetative remains of high marsh species. Rhizomes and stems of *S. alterniflora* are much larger and thicker than those of *S. patens* and *D. spicata*, and may dominate subsurface environments over time. Thus, there is the possibility that by mapping subsurface environments with the Eijelkamp coring device, one sub environment may have 'masked' the presence of the other, resulting in irregularities in reconstructed stratigraphy of the area.

Although care must be taken in making detailed inferences from the methodologies applied and the data obtained in this study, the author is confident that the general conclusions concerning the temporal and spatial relationships of high marsh and low marsh subenvironments are correct.

Underground Biomass Dynamics. The relationship between surface vegetative growth and subsurface rhizome distribution (penetration to depths of 20 and 25 cm) precludes traditional interpretation of stratigraphic profiles. Interaction and intermixing of low marsh vegetative remains over previously existing high marsh deposits occurs as transgression continues; i.e., even if *S. alterniflora* is present in surface vegetative distribution for only one year, it is represented in the subsurface to depths of 20 and 25 cm. The presence of *S. alterniflora* rhizomes in the subsurface at a particular depth, therefore, does not necessarily imply that the corresponding surface environment (e.g., as drawn with time line) was colonized by *S. alterniflora.*

Underground Biomass Profiles: *S. alterniflora* and *S. patens*. Research has been conducted on distribution of underground biomass and productivity in Atlantic coastal salt marshes (Valiela *et al.*, 1976; Gallagher *et al.*, 1977; Gallagher and Plumley, 1979; Gallagher and Kibby, 1981; Gallagher and Howarth, 1987). Gallagher and Howarth (1987) examined recoverable underground reserves from stands of three growth forms of *S. alterniflora* and one stand of *S. patens* in Massachusetts, and found that most recoverable reserves of living plants were found within 20 cm of the marsh surface. Similarly, in Oregon, Gallagher and Kibby (1981) reported rhizome growth for backmarsh *S. alterniflora* to be no deeper than 20 cm. Valiela *et al.* (1976) sampled salt marsh sediments to depths of 15-20 cm for their below-ground primary production research in a Massachusetts tidal marsh. Generally, it has been established that although living roots and rhizomes may extend to a subsurface depth of 35 cm, the vast majority of roots and rhizomes are concentrated in the upper 15-25 cm (Valiela and Teal, 1974; Gallagher *et al.*, 1977; Gallagher *et al.*, 1984) (Figures 5.16 and 5.17).

Thus, while rhizomatal remains in the subsurface may be useful in identification of salt marsh plant species, caution must be used in making inferences of temporal and spatial relationships in the subsurface and in assigning a specific marsh

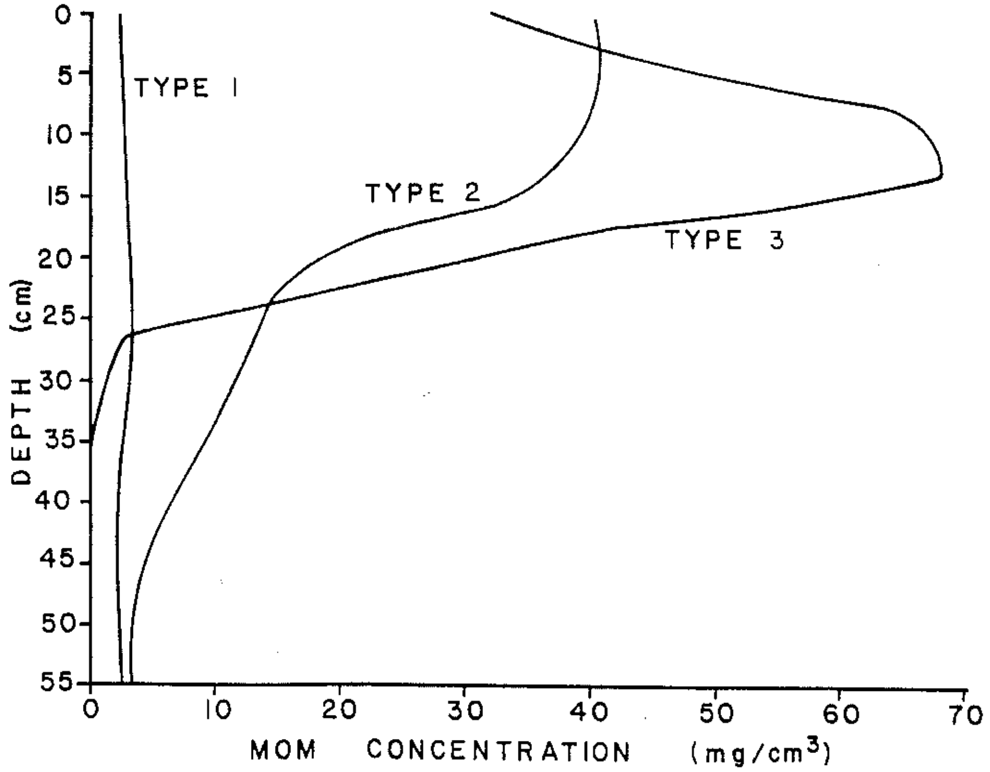


Figure 5.16 Diagrams depicting three types of macro-organic matter profiles (from Gallagher *et al.*, 1977).

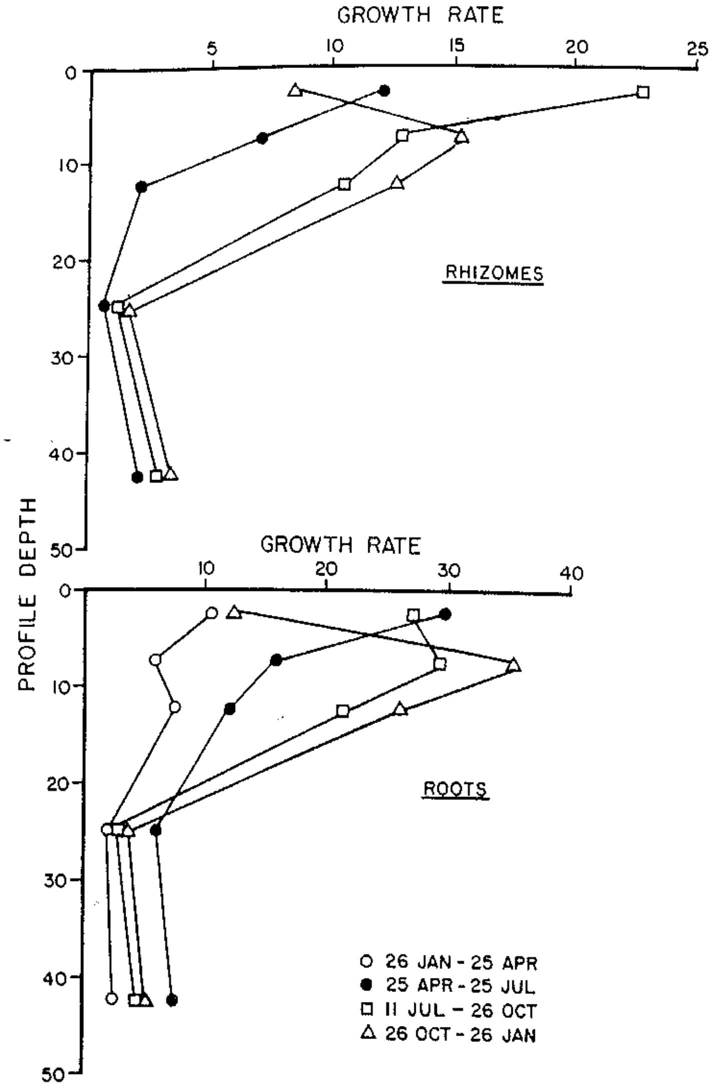


Figure 5.17 Diagram depicting rhizome and root growth as a function of profile depth (cm) in a Georgia *Spartina alterniflora* salt marsh (from Gallagher *et al.*, 1984).

surface environment to a time-depth horizon. The presence of *S. alterniflora* rhizomes at a 20 cm depth in the subsurface may simply represent the present surface wetland vegetative characteristics. It is not necessarily appropriate to assign a 'before present' time/depth relationship to that subsurface horizon containing *S. alterniflora* rhizomes, based on the assumption that in the past, *S. alterniflora* vegetation was growing on the surface at the same time it is represented in the subsurface.

Summary. Data obtained in this study, through identification of macroscopic plant remains, demonstrate that there is a change in salt marsh vegetative species with depth in Delaware's fringing tidal salt marshes. Inferences regarding temporal and spatial relationships of subsurface marsh environments must be made with caution due to the narrow thickness of vegetative zones mapped in the subsurface and possible overlap of rhizome penetration with depth. Vegetative characteristics of fringing tidal salt marshes change with depth in subsurface deposits and are overwhelmingly transgressive, with low marsh environments overlying higher marsh environments, and high marsh environments transgressing over previously upland surfaces. The subsurface profiles imply a landward movement of *S. patens* and *S. alterniflora* environments over the past 200 years.

Stratigraphic Patterns: Models

Stratigraphic patterns observed in the 69 transects can be presented in three models, depending on the nature of both antecedent and adjacent topography, as illustrated schematically in Figures 5.18 through 5.20.

Shallow:Shallow. Fringing tidal salt marshes with gently sloping adjacent uplands and gently sloping antecedent topography are dominated by high marsh *S. patens/D. spicata* vegetation (5.18). As transgression continues, *S. patens/D. spicata* vegetation will migrate landward over the adjacent upland surfaces and *S. alterniflora* (low marsh) will transgress over the high marsh. Surface and subsurface expression of high marsh *S. patens/D. spicata* environments is likely.

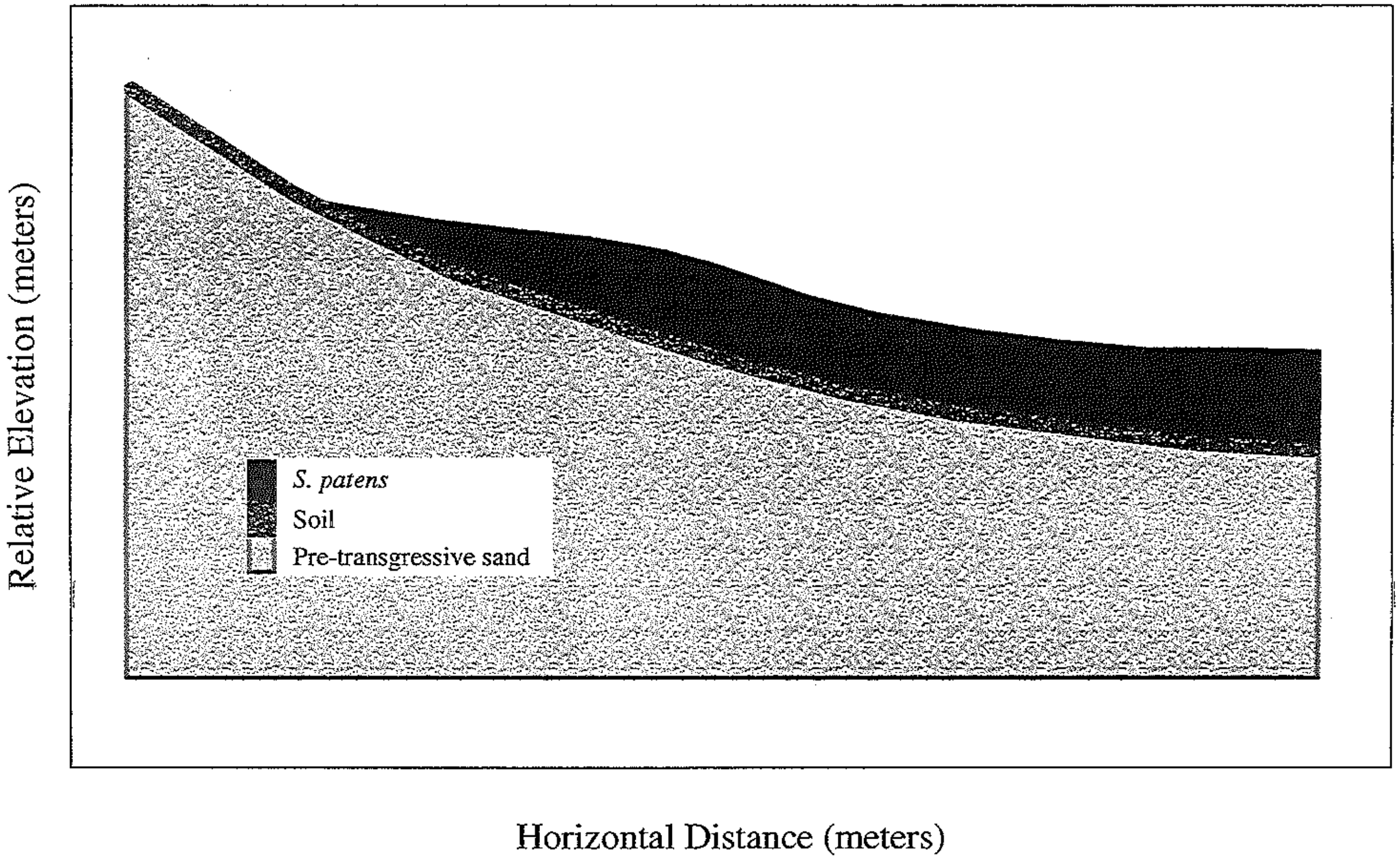
Steep: Steep. Fringing salt marshes with steeply sloping adjacent uplands and steeply sloping subsurface geometry are dominated by *S. alterniflora* low marsh environments, with notable absence of a high marsh zone (Figure 5.19). *S. alterniflora* vegetation extends landward to the wetlands/uplands interface, which is colonized by a narrow band of *I. frutescens/B. halimifolia* and/or *P. australis* vegetation. Steeply sloping antecedent geometry is indicative of past environmental conditions which preclude development of a *S. patens/D. spicata* high marsh zone.

Mixed: Steep:Shallow, or Shallow: Steep. Fringing salt marshes with variable antecedent and adjacent slopes are typically characterized by 'classic' salt marsh vegetative zonation patterns. This includes an upper fringe of *I. frutescens/B. halimifolia* vegetation, a band of high marsh *S. patens/D. spicata* zone (width depending on adjacent and antecedent slopes), a mixed zone of *S. patens/S. alterniflora* (representing landward movement of low marsh over high marsh), and a *S. alterniflora* low marsh zone (Figure 5.20).

General characteristics of these composite transects are outlined in Table 5.4.

Figure 5.18 Schematic diagram depicting stratigraphic patterns observed in fringing tidal salt marshes characterized by shallow adjacent slopes and shallow antecedent geometry.

Schematic Cross-Section
Shallow Antecedent and Adjacent Slope



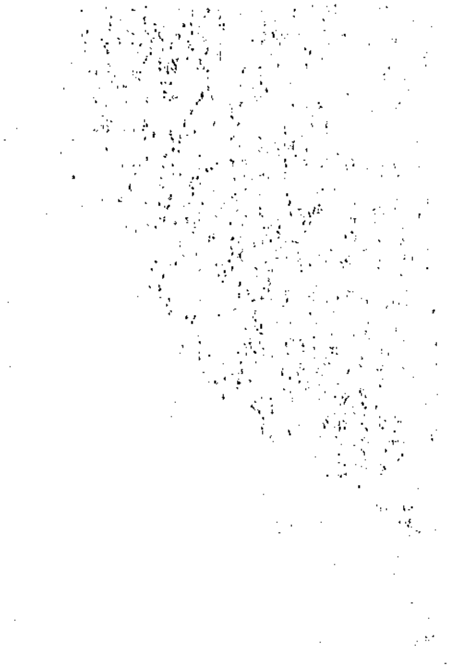


Figure 5.19 Schematic diagram depicting stratigraphic patterns observed in fringing tidal salt marshes characterized by steep adjacent slopes and steeply sloping antecedent geometry.

Schematic Cross-Section
Steep Antecedent and Adjacent Slope

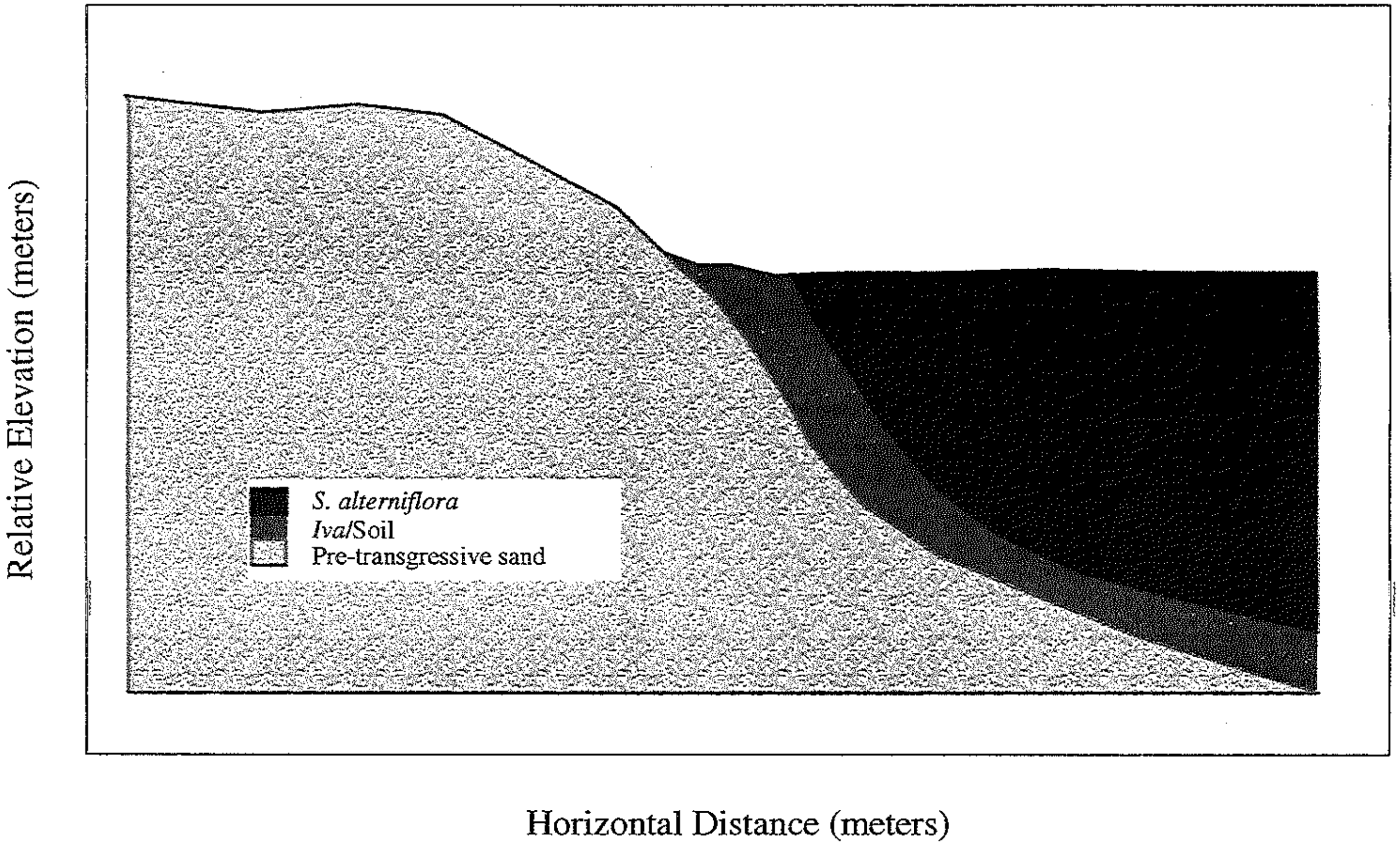


Figure 5.20 Schematic diagram depicting stratigraphic patterns observed in fringing tidal salt marshes characterized by variable adjacent slopes and antecedent geometries.

Schematic Cross-Section
Variable Antecedent Slope

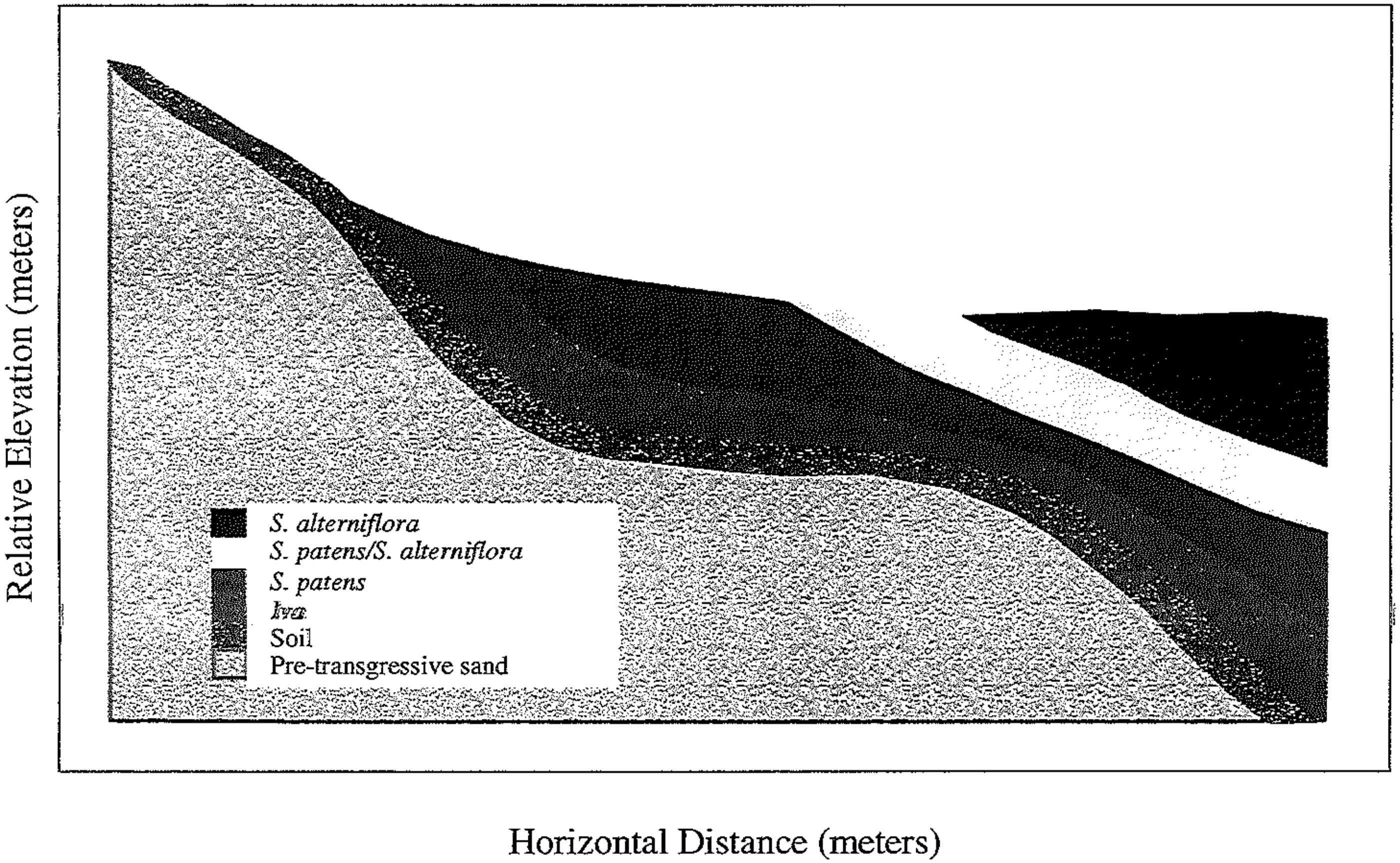


Table 5.4 Adjacent/Antecedent Slope and Characteristic Surface and Subsurface Environments (✚ indicates presence of environment; ◘ indicates absence of environment, or high probability of absence of environment).

Topography		Surface/Subsurface Environments			
Adjacent	Antecedent	*P. australis*	*I. frutescens*	*S. patens*	*S. altern.*
shallow	shallow	◘	✚	✚	✚
steep	steep	✚	✚/◘	(◘)	✚
steep	shallow	◘/(✚)	✚	◘/(✚)	✚
shallow	steep	◘	✚	◘	✚

Summary: Subsurface Vegetative Changes and Characterization of Wetlands/Uplands Boundary

Delaware salt marshes exhibit numerous subenvironments within the overall low and high marsh areas. The distinctiveness and 'preservation potential' of individual features within these subenvironments is highly variable and may be gradational. They ultimately depend upon interacting factors including depth, time, and environment of burial. Although sub-environmental changes mapped and delineated in this study are 'micro' and 'minor' and are not likely to be differentiated at greater depths (> 2 meters), their documentation is important in demonstrating the transgressive nature of Delaware's fringing tidal marshes. Change in vegetative species in the subsurface at the wetlands/uplands boundary is not only indicative of submergence in this environment, but also suggests that vegetative changes in salt marshes are complex and variable and may be attributed to short term changes such as sediment supply, tidal range, and soil characteristics, and the long-term effects of relative sea-level rise.

Subsurface Marsh Patterns: Discussion

Introduction

Historic changes in marsh environments have been recorded by many investigators, notably Yapp (1922), Richards (1934), Bourn and Cottam (1950) and Redfield (1972), and more recently by Warren and Niering (1993). The response of vegetation to changing physical conditions is well known, and spatial variations in vegetation patterns can occur over time; subsurface sediments document these vegetation patterns and provide a record of temporal and spatial environmental changes. However, many previous studies have noted the difficulty of correlating strata and environment of deposition between core locations due to the variability of conditions within the marsh (Meyerson, 1972; Heusser *et al.*, 1975; Niering *et al.*, 1977; Clark and Patterson, 1985). Clark and Patterson (1985) note that for any core site, a given fluctuation in external conditions (e.g., sea level, climate, sediment supply) is recorded, but is affected by: 1) distance from the site to adjacent ecotones at a point in time; 2) the responsiveness of those ecotones to environmental change; and 3) the magnitude of environmental change. In this investigation, however, the closely spaced surveying and coring conducted in the study area allow detailed interpretation of strata with respect to depth and time. In addition, detailed information of slope, vegetation, and sedimentology for adjacent upland environments permits inferences of the response of the marsh to external forcing functions, i.e., cause and effect relationships. The conclusions drawn in this study apply only to the wetlands/uplands boundary area of the fringing marsh, and not to the entire marsh. The seaward or bayward marsh edge is characterized by different community structure, physical characteristics, and external and internal physical forcing mechanisms.

The level of detail obtained in the stratigraphic interpretations of wetlands cores in this study permits accurate reconstruction of marsh development at the wetlands/uplands boundary. Surficial wetland zonation patterns, as described in the previous chapter, suggest that vegetation is restricted to relatively discrete communities. Through reconstruction of tidal marsh vegetative zonation, it is possible to determine marsh response to changing environmental conditions, although it is difficult to say which external physical factors are actually causing changes in environmental conditions (degree of tidal influence, sediment type, and depth to antecedent topography are likely to be primary factors).

Factors to Consider. Local relative sea-level change is the single most important long-term physical factor affecting marsh systems along the Delaware coast. This factor controls to some degree many other factors such as erosion rates and vertical accretion rates. Erosion of the wetland fringe and/or changes in barrier beach shoreline configuration affect marsh systems by changing inlet locations and modifying drainage patterns, which in turn may impact sediment sources and volumes, as well as tidal fluctuations. Subsidence due to compaction may be a significant factor where thick sequences of marsh sediments or other compressible sediments underlie living marshes; some marshes may have high sedimentation rates but show very little change in elevation or vegetation distribution because of subsidence due to compaction of underlying sediments. Extreme events such as fires and storm erosion or deposition can cause major changes in vegetation patterns. This summary of factors stresses the importance of local conditions in controlling the distribution of marsh environments.

Marsh Inception. Marsh inception, as used here, is modified from the definition of Pethick (1980): the time at which marsh vegetation appears on previously vegetated or unvegetated upland surfaces. This colonization takes place at or above the high water mark of spring and storm tides, where terrestrial vegetation gives way to halophytic wetland plants. *I. frutescens/B. halimifolia* are typical shrubs in this area, while *S. patens* is the normal initial herbaceous colonizer of uplands in Delaware; other authors note that *Panicum virgatum* is often found in the wetland/upland transition zone (Niering and Warren, 1980).

Throughout this study of the wetlands/uplands boundary, one distinct pattern of marsh facies distribution was identified in the core sequences, with several important and notable variations. In all cases, stratigraphic relationships were transgressive, with low marsh *S. alterniflora* overlying high marsh *S. patens/D. spicata* zones. Variations in this transgressive stratigraphy are related to slope of adjacent upland and antecedent slope (slope of the pre-transgressive surface).

The overwhelmingly transgressive nature of the stratigraphic sequences sampled at the wetlands/uplands boundary reflects the general response of vegetation to sea-level rise. It is represented in cores from fringing marshes along the western shore of Delaware Bay, and along the wetlands/uplands boundary of Delaware's Inland Bays. The sequence is preserved in short cores located at the leading edge of the transgression, and variations in this transgressive sequence are observed where vegetation changes occur perhaps more rapidly or more slowly, depending on adjacent and antecedent slopes.

Variations in the transgressive sequence may also be caused by human-induced factors, such as mosquito ditching and inlet stabilization. Mosquito ditching can have variable impacts on marsh vegetation; ditching can lower the water table and increase drainage, thereby effectively raising the marsh surface and providing an environment more suitable to high marsh fringing vegetation (Allen, 1977).

Increased tidal flooding can also result in greater amounts of sediment transported into high marsh areas. Yet, at the same time, mosquito ditching can increase the amount of tidal flooding in a marsh area, and, when mosquito ditches are not maintained, reduced drainage can occur, resulting in effective lowering of the marsh surface.

Reconstruction of Marsh Development at Wetlands/Uplands Boundary or Leading Edge of Marine Transgression: Discussion

Antecedent Topography and/or Morphometry of Pre-transgressive Surface. Exposed forest floor layers are found within the present high- and low-marsh zones, identified by the presence of tree stumps, wood and bark fragments, and leaf litter. Pre-transgressive sand was found to underlie marsh deposits at a depth of 0.02 m to >3 m within a 40 m horizontal distance perpendicular to the wetlands/uplands boundary. High-marsh peat with *S. patens*/*D. spicata* rhizomes was typically found above the pre-transgressive sand, with a transition to low-marsh peat with *S. alterniflora* rhizomes and stems in the uppermost section of the cores.

Slope of the pre-transgressive surface is quite variable, and controls not only the rate of landward migration of tidal salt marsh environments, but also surface vegetative zonation of fringing wetlands. Three general types of antecedent topographies have been identified in Delaware's fringing tidal salt marshes: 1) shallow antecedent and adjacent topography; 2) steep antecedent and adjacent topography; and 3) intermediate, or 'stepped' antecedent and adjacent topography. General stratigraphic characteristics can be described for various antecedent topographies using lithology and macroscopic vegetative remains.

History of the Marsh. Results of stratigraphic analyses and reconstruction of recent (150-200 year) wetland histories from core data indicate a landward migration of tidal salt marsh environments in Delaware. Stratigraphic data demonstrate that pre-transgressive sandy sediments are overlain by high-marsh peat, and in areas where low-marsh environments are present on the marsh surface, low marsh peats are typically found above high-marsh peat remains. In some cases, rhizomes and macroscopic vegetative remains provide evidence that freshwater wetlands have been transformed to salt marsh, with remains of *Scirpus* and *P. australis* overlain by *S. patens/D. spicata* and *S. alterniflora* rhizomes and stems.

General changes in lithology and macroscopic plant remains seem to have occurred uniformly throughout the study area, although thickness of deposits and rates of transgression are likely to be quite variable. Although the study area has been impacted by a regional relative sea-level rise, more localized conditions including dredging, sediment load, upland runoff, mosquito ditching, inlet openings and closures, changes in tidal range, differences in adjacent upland slope and antecedent topographies etc. are causative factors that result in both temporal and spatial variations in stratigraphic histories.

The pattern of transition from high-marsh deposits to low-marsh deposits indicates that a change in mean high-water levels which could have occurred due to either an overall decrease in sediment supply and/or an increase in water level (tidal range, sea level); i.e., the rate of local relative sea-level rise has exceeded the rate of sedimentation. The transgressive nature of the stratigraphy in the area suggests that physical factors such as changes in sediment supply and tidal range are controlling the landward movement of Delaware's tidal marshes.

Historical clearing of forests and heavy agricultural use in the watersheds of the study area supplied much of the sediment to tidal salt marshes. However, over the past 20-30 years, erosion and sediment control measures have reduced the amount of sediment supplied to the waterways. Overall change in species distribution with depth in the cores therefore reflects change in environmental characteristics, especially tidal range and sediment supply, with a change from high-marsh to low-marsh indicative of an increase in tidal influence. Such a transition can be explained by a general sustained increase in water level relative to the marsh surface caused by increases in tidal range or decrease in sediment supply, or an overall increase in the rate of relative sea-level rise. Other possible impacts include early human occupation of coastal areas, with clearing of adjacent upland forested areas, and use of *S. patens* wetlands areas for pasturage (Chet Stachecki, personal communication; Chuck Fithian, personal communication).

Application to Sea Level Reconstruction. In order to relate stratigraphic changes to sea-level change, a relationship must be defined between vegetational changes in the subsurface and environmental factors. Changes in coastal marsh stratigraphy is inevitably tied to interpretation of tidal water levels, but detailed interactions and associations between tidal marsh vegetation and associated environmental conditions cannot be overlooked. Correlation of changes in tidal salt marsh vegetation in the subsurface at various locations along the Delaware coast must be inexorably tied to modern analogues and definitions of vegetation-environmental relationships. Consideration of salinity, waterlogging, soil chemistry, sediment type, and water table changes must be made along with the frequency and duration of tidal inundation factor. Environmental changes observed in the subsurface may reflect either changes in sea-level or more isolated and site specific periodic fluctuations in tidal influence. Shifts and changes in marsh sub-environments can clearly be caused by many interacting physical, chemical, and biological factors; vegetation response to environmental change must be given specific and detailed consideration.

Suggestions for research which would permit more detailed interpretations of relationship between vegetational changes and environmental conditions: 1) detailed pollen analyses; 2) macrofossil analyses; 3) precise dating of cores; 4) detailed local tide-gauge records; and 5) detailed knowledge of historic human activities and potential impacts in adjacent upland areas. These data would permit interpretation of environmental changes in the subsurface and identification of these changes as sea level or disturbance responses, and therefore provide evidence for direct interpretation of potential physical, chemical, and biological factors contributing to overall marsh development.

Fringing Tidal Salt Marsh Development. Past studies have demonstrated that vegetation plays a dominant and essential role in development of salt marshes, trapping and anchoring sediment, contributing to both mineralogic and organic components of vertical accretion. Many past models of salt marsh development conceived for emerging coasts (e.g., Redfield, 1972) explain that as vertical accretion continues in the marsh, *S. patens* (which simultaneously expands in the upland direction) replaces *S. alterniflora*, and future accretion is closely tied to sea-level rise. Thus, core sections typically include tidal flat sediments overlain by *S. alterniflora* deposits, followed by high marsh deposits. In the landward direction, high marsh sediments are typically found overlying upland soils. This model of wetland development and evolution requires that sediment supplies are sufficient to keep up with the rate of sea-level rise, which is often assumed to be a gradual and uninterrupted process. However, the transgression of *S. alterniflora* (low-marsh) deposits over *S. patens/D. spicata* (high-marsh deposits) is not typically reported in previous studies. This model requires not only a steady increase of sea level rise, resulting in the migration of high marsh environments over adjacent uplands, but also the possibility of episodic events or disturbances superimposed on the gradual and steady influence of sea-level rise. The most likely possibilities for disturbances are changes in local environments, such as storms, human impacts, inlet dynamics, sediment supply, and tidal range. However, the many interacting biotic and abiotic factors which influence zonation patterns in a salt marsh are perhaps too complex and dynamic to permit simple interpretation of individual events. Zonation and successional changes in salt marsh environments are extremely complex, and it may be unwise to infer ecological principles from subsurface vegetational changes. As pointed out by Clark and Patterson (1985), modeling of marsh development may not be possible over the short term, as assumptions of constant sediment supply and sea-level rise in the absence of environmental or human disturbance may be unrealistic. Past models developed by Redfield (1972) and Chapman (1974) consider longer time frames. Models must include the response of vegetation to important external disturbances resulting from human impacts such as variation in sediment supply, and changes in tide range (barrier island and inlet dynamics) and water table levels in adjacent uplands.

Reduced sediment input would slow vertical growth of the marsh relative to rising sea level, and would contribute to elevational differences between high and low marsh environments. Increased tidal flooding of sites with lower accretion rates would produce edaphic changes which in turn would result in changes in plant assemblages. Anthropogenic factors are likely contributing to changes in surface and subsurface vegetation patterns in Delaware's fringing salt marshes, especially those factors which affect waterlogging and saturation of high marsh soils. Thus, along with the long-term impacts of continuing sea level rise, multiple short-term factors may be interacting to produce the changes in both surface and subsurface vegetation patterns observed at the wetlands/uplands boundary of Delaware's fringing tidal salt marshes.

Based on changes observed in a narrow band of salt marsh along the wetlands/uplands border, one can speculate about future development of *S. patens* high marshes or increased areal extent of low-marsh vegetation at the expense of high marsh environments. While rates of sea level rise and other factors (tide range, sediment supply) control the distribution of low-marsh *S. alterniflora* environments, it is ultimately the slope of adjacent uplands which controls the future areal distribution of *S. patens* environments. This has been addressed in Chapter 4 and will be discussed again in Chapters 7 and 8. As noted by Warren and Niering (1983), these changes may occur prior to any direct wetlands loss resulting from coastal submergence. This may be a first stage of submergence or an indicator that submergence is about to occur.

Summary and Conclusions

Changes in subsurface environments in fringing marsh systems along the Delaware Bay and Inland Bays are related to biotic and abiotic, natural and anthropogenic factors. Closely spaced cores and macroscopic analyses of vegetative remains provides records of salt marsh vegetation histories. Several major patterns have been identified in stratigraphic relationships of salt marsh subenvironments.

1) The stratigraphic pattern observed in 69 transects is a transgressive sequence of subaerially exposed terrestrial pre-transgressive sands overlain by salt marsh sediments.

2) Thickness of marsh sedimentary units, or depth to the pre-transgressive surface, provides evidence for the spatial and temporal inception of fringing tidal salt marsh development over pre-existing upland surfaces. Wetlands have developed over previously upland surfaces in areas where adjacent slopes are shallow; recently developing high marshes can be found in areas of shallowly sloping adjacent uplands, but they are typically absent from sites with steeply sloping upland and antecedent surfaces.

3) Schematic models with different antecedent geometries are developed to illustrate the stratigraphy at the wetlands/uplands boundary. These sections show some

of the subtle variations influencing marsh environment distribution and zonation, such as depth to pre-transgressive sands or slope of adjacent and underlying surfaces.

4) The evolution from one marsh environment to another with a slightly lower elevation (e.g., transition from *S. patens* to *S. alterniflora* environments) often but not always, occurs in areas with shallowly sloping antecedent and adjacent surfaces. Temporal and spatial transitions from high marsh environments to low marsh environments indicate the increasing inundation by salt water. However, evolution to 'lower' salt marsh environment may also be indicative of increased submergence due to increased water levels caused by short-term anthropogenic influences such as changes in tidal range and decreased sediment supply.

5) Stratigraphic relationships found in tidal wetlands can be linked to changes in factors which affect tidal marsh evolution, such as tide range and water level fluctuations because marsh vegetation zonation/histories are so closely linked to external physical factors. The overall long-term factor controlling marsh deposition is sea-level rise. However, short-term anthropogenic influences on tidal range and sediment supply may exert significant control on evolution and development of fringing salt marshes at the wetlands/uplands boundary area.

6) Empirical evidence (dead trees, fencing, etc.) suggests that, within the confines of the restricted study area (wetlands/uplands boundary zone), temporal inception of wetland development is likely to range from 200 years before present to the present. Over this short term, tidal marsh development is more likely related to changes in sediment supply and tidal range than to fluctuations in sea level. Minor fluctuations

which have occurred in this restricted geographic area are reflected in the development of the salt marsh. Over the short-term, it is difficult to suggest or imply that rise in sea level is the primary driving force behind development of fringing salt marshes. Perhaps impacts of greater significance are related to local fluctuations in sediment supply and tidal range, especially when these local factors are superimposed on sea-level rise.

7) Characteristics of adjacent uplands, especially slope, are critical in determining spatial and temporal changes in surface and subsurface vegetation and vegetative zonation (plant succession) at the wetlands uplands boundary, as well as the future areal extent of fringing tidal wetlands in Delaware.

8) At the wetlands/uplands boundary, marsh development is proceeding in a landward direction, normal to the shoreline; i.e., fringing tidal marshes are developing only in a landward (transgressive) direction, not a seaward (regressive) direction, as previous models suggested.

Chapter 6

GEOCHRONOGY AND VERTICAL ACCRETION

Introduction

Cores from six study sites were selected for determination of vertical accretion rates based on analyses using excess activity of ^{210}Pb, maximum activity of ^{137}Cs, and oak:ragweed pollen ratios (data are presented in Appendix G). Sites selected for detailed sedimentation rate analyses include: Prime Hook Radar Station (PHRS) (pollen only, no ^{210}Pb or ^{137}Cs) and Oyster Neck Road (ONR) sites along the Delaware Bay shoreline; Wolfe Runne (WR) site along the Lewes/Rehoboth Canal (no pollen analysis); Delaware Wildlands (DEWL) and Pot Nets North (PNN) along the southwestern shore of Rehoboth Bay; and Boat House Cove (BHC) on the northern fringe of Indian River Bay. Visual observation of both the coring site conditions and internal structure of all cores suggested that little or no bioturbation (aside from root growth and penetration) had taken place in the core; i.e., there was no evidence of burrowing or tunneling by infauna that would have mixed the marsh sediments.

Additional data on sediment accretion rates in Delaware salt marshes were compiled from the literature (Church *et al.*, 1981; Yi, 1992; Kraft *et al.*, 1992). Previous studies have shown that rates of aggradation of coastal salt marsh surfaces in

Delaware vary greatly (Figure 6.1). All data were used to interpret the spatial and temporal variation in vertical accretion rates in Delaware salt marshes over the last 100-250 years.

These rate analyses permit comparison of vertical accretion in similar wetland environments that are impacted by variable external parameters such as sediment supply, antecedent slope, and local tidal range. This results in detailed knowledge of the chronologic sequence of wetland development, evolution, and transgression at various wetland locations in coastal Delaware.

Methods

Field Methods

^{210}Pb and ^{137}Cs Cores. A total of five wetland sites were sampled for ^{210}Pb and ^{137}Cs analyses, including two sites in fringing marshes along the Delaware Bay coast (Oyster Neck Road and Wolfe Runne) and three sites in fringing tidal marshes along Delaware's Inland Bay shoreline (Boat House Cove, Delaware Wildlands, and Pot Nets North). See Figure 6.2 for site locations. To limit the influence of different environmental factors such as vegetation type on sedimentary/accretionary processes, the sites selected were spatially similar ("upper low marsh zone") relative to the wetlands/uplands boundary, and were dominated by short form *Spartina alterniflora* Loisel vegetation. Cores for ^{210}Pb and ^{137}Cs analyses were obtained at four of the sites (Oyster Neck Road, Wolfe Runne, Delaware Wildlands, Pot Nets North) by hammering a 16-cm diameter plastic pipe into the marsh sediments to a depth of 50 cm. A

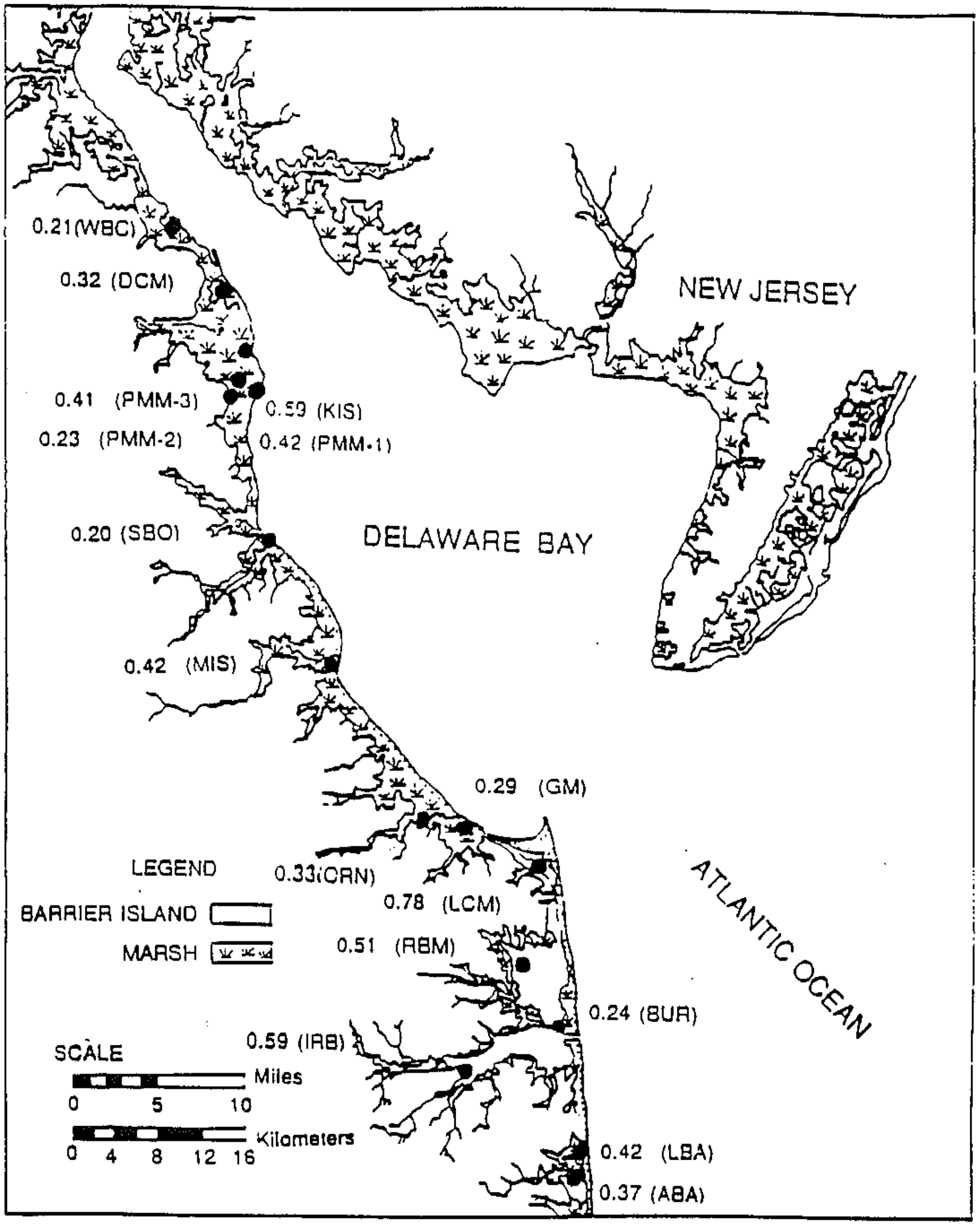


Figure 6.1 Previous studies show that rates of sedimentation or aggradation of coastal salt marshes surfaces in Delaware vary greatly (from Kraft et al., 1989; courtesy of H. -I. Yi).

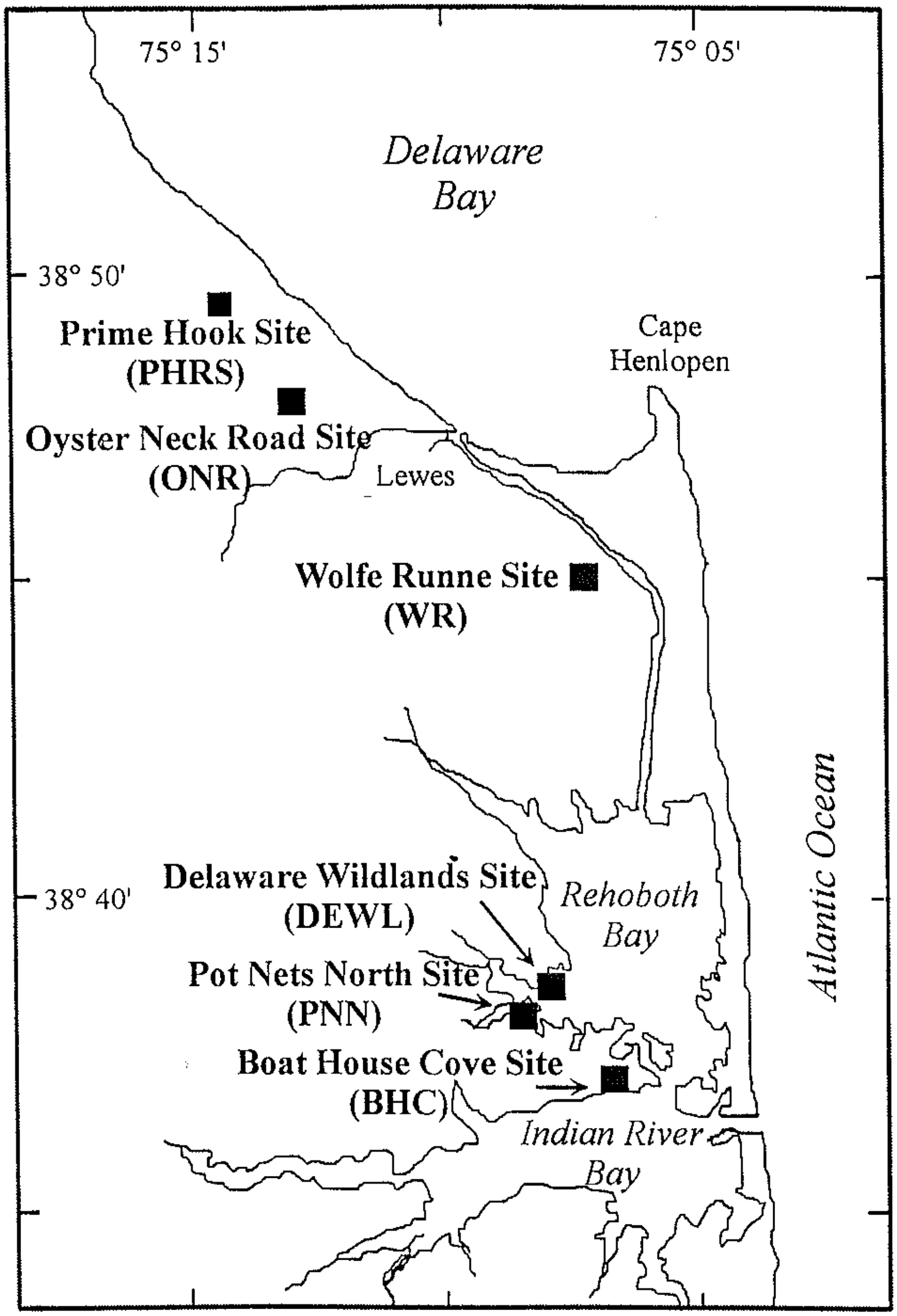


Figure 6.2 Location map of sampling sites for radionuclide and pollen analyses.

vibracore was obtained for radionuclide analyses at the Boat House Cove site. Compaction of sediment during coring was minimized by trimming surface vegetation and cutting subsurface roots and rhizomes with a serrated knife; compaction was estimated to be within 1% to 3%. Immediately after extraction, cores were sealed with plastic and tape in the field and taken to the laboratory for sampling. Samples were obtained by slicing the core in 2.5-cm sections from the bottom to the top, to avoid sectioning contamination.

Pollen Cores. Vibracores were obtained for pollen analyses at five study sites: Prime Hook Radio Station, Oyster Neck Road, Delaware Wildlands, Pot Nets North, and Boat House Cove. Vibracores were sealed, labeled, and transported to the laboratory of Dr. Grace Brush, Johns Hopkins University, for analyses.

Laboratory Methods

Moisture Content, Organic Content, and Dry Bulk Density. Moisture content (% water), organic content (% weight loss on ignition), and dry bulk density of sediments for the five cores were determined at 2.5 cm increments. Water content was measured by oven-drying the pre-weighed wet samples at 65° C for five days, or until there was no change in dry weight over time. The weight percent of water content of each sample was defined by the equation: [(wet weight - dry weight)/wet weight] x 100.

Percent organic content was determined using the method of weight loss on ignition (LOI) (Nozaki *et al.*, 1978). Homogenized subsamples (1-3 grams) of the oven-dried samples were ashed in a muffle furnace at 425° C overnight (Lord, 1980), and then re-weighed. The weight percentage of ash is considered to be the weight percent of organic content; the weight percent of organic content of each sediment sample is defined by the following equation: [(dry weight - ash weight)/dry weight] x 100.

Dry bulk density was determined for each 2.5-cm section in each core, calculated from the weight and volume of each slice according to the formula: (dry weight of sediment/volume core section), with units g/cm^3.

Pollen Analyses. Pollen analyses were conducted by Dr. Grace Brush, Johns Hopkins University, using oak:ragweed ratios as follows: A measured volume of sediment is washed in hydrochloric acid, hydrofluoric acid, and acetylzed with a mixture of nitric acid and acetic anhydride to remove carbonates, silicates, and organic material. The residue is then washed in glacial acetic acid, distilled water and alcohol, then stored in tertiary butyl alcohol. Aliquots of known volume (usually 0.2 ml) are mounted in silicone oil on microscope slides, and all pollen in the aliquot is identified and counted. Identifications are made using various pollen identification manuals and also, when necessary, using the Smithsonian (Washington, D.C.) pollen reference collections (Grace Brush, personal communication).

Geochronologic determinations were developed through detection of decline in pollen of oak and concurrent sharp rise in pollen of agricultural weeds such as ragweed. It is assumed that a decline in oak pollen and concurrent increase in ragweed pollen resulted from large-scale European land clearance in Delaware from approximately 1750 through the early 19th century. Changes in pollen profiles reflect historically dated changes in vegetation and land use, thereby providing a method for calculating average sedimentation rates. Sedimentation rates for each interval of a core were calculated by adjusting the average sedimentation rate to the pollen concentration for each interval using the equation:

$$R_{0\text{-}1} = (n/n_{0\text{-}1})R$$

where

$R_{0\text{-}1}$ is the sedimentation rate for the interval 0-1

n is the average number of pollen grains per unit of area in the interval

$n_{0\text{-}1}$ the number of pollen grains per unit area in the interval 0-1

R is (d/t), the average sedimentation rate between two dated horizons (depth of dated horizon in the core/time in years) (Brush, 1989).

Vertical accumulation of pollen is used as a surrogate of time; i.e., the concentration of the pollen in any interval of sediment reflects the rate of accumulation of the other particles that make up that sediment. If the rate of sediment accumulation increases, the concentration of pollen will be correspondingly less, and if the rate of sediment accumulation decreases, pollen concentrations will be correspondingly greater (Brush, personal communication). The validity of dating sediments using this method has been demonstrated by identifying historically dated events (Brush, 1989).

^{210}Pb and ^{137}Cs Dating Technique

Five study sites were selected to measure sediment accretion rates of marsh environments; ^{210}Pb and ^{137}Cs analyses were performed by Dr. Najid Hussain (Dr.

Thomas M. Church's Laboratory, College of Marine Studies, University of Delaware, Newark, Delaware). Cores were sectioned into 2.5 cm interval samples, which were then dried at 60° C for 5 days, or until no changes in weight were determined. Subsamples of 1-3 g from each interval were ashed at 420° C overnight for determination of organic content (LOI). The remaining dried sample was homogenized, placed into 100 cm^3 aluminum cans, hermetically sealed with canning equipment, and used for direct gamma counting. The aluminum cans used are 6 cm diameter by 3.5 cm height, with a bottom surface area of approximately 28.3 cm^2.

Gamma Spectrometry. Radionuclide analyses were conducted by Dr. Najid Hussain with a non-destructive technique utilizing a germanium detector coupled with a multi-channel analyzer. All radioisotopes used in this study (^{210}Pb, ^{226}Ra, and ^{137}Cs) were measured by a hyperpure low-background Ge gamma detector (intrinsic germanium coaxial detector, PGT Model No. NIGC 2020). This germanium detector has a 21.65% relative efficiency, with a range of 10 KeV to 1480 KeV, and an energy resolution of 0.371 keV. ^{210}Pb and ^{137}Cs emit gammas at 47 KeV and 662 KeV, respectively (Krishnaswami *et al.*, 1971, 1980). Because of the high resolution of this detector, a large number of clearly defined photopeaks, with energies indicative of specific radioactive nuclides, could be identified. The pulses from this detector were analyzed with a multichannel buffer analyzer having 4096 channels, coupled to an MCA/simulation program disk-based microcomputer. Background contribution of the HPGe detector was reduced by surrounding the detector by a 1 cm thickness steel and 5 cm thickness lead castle (N. Hussain, personal communication).

Low background gamma counting for direct measurement of the gamma ray of ^{210}Pb with acceptable analytical precision has previously been used as a method for establishing geochronology of core sediments based on ^{210}Pb activities (Cutshall *et al.*, 1983; Appleby *et al.*, 1986; French *et al.*, 1994; Hussain *et al.*, in press). The principal difficulty of gamma-ray counting of ^{210}Pb is correction for self-absorption of the radiation by the sample because the attenuation coefficient for the 46.5 keV gamma-rays is highly dependent on sample composition (Cutshall *et al.*, 1983; Hussain, personal communication). Calibration of detector efficiency, effects of self-absorption, and sample geometry was accomplished through use of standard materials (^{210}Pb point source, US EPA pitchblende, ^{137}Cs standard, and ^{207}Bi standard) (Hussain, pers. comm). According to Hussain *et al.* (in press) in order to obtain the self absorption factors of ^{210}Pb for each sample, six standard samples which have different amounts of quartz were made, and counted with a ^{210}Pb point source placed on top of the can. The self absorption factors for each standard sample are provided by an equation:

$$SF = 0.902 + 0.0064\ W$$

where SF is self-absorption factor and W is sample weight. It is assumed that the effect of self-absorption is negligible for the other high-energy nuclides (Hussain *et al.*, in press).

Vertical Accretion Rates: Results and Discussion

Sediment Characteristics

Dry bulk density, moisture content, and percent loss-on-ignition were calculated for six cores: Prime Hook Radio Station, Oyster Neck Road, Wolfe Runne, Delaware Wildlands, Pot Nets North, and Boat House Cove.

Dry Bulk Density. Dry bulk densities for the marsh cores range from 0.08 to 0.52 g/cm^3, with greatest values generally occurring in either the top 2.5 cm of the core and/or within the 40-45 cm section of each core, as shown in Figures 6.3 to 6.8. Dry bulk density is highly variable throughout the cores. However, dry bulk density is generally lowest at the surface of the core, varies with depth, and then decreases sharply at the base (50 cm) of each core; many variations are observable within and between cores and sample sites.

Dry bulk density was higher in the low organic matter sediments between 25- 45 cm depth, as compared to the higher organic sediments in the top 15-20 cm of the cores. In most cores, bulk density was lowest at the surface and increased with depth. In summary, dry bulk density of individual slices varies with depth over the length of the core; the curves demonstrate that mineralogenic sediment accumulation has varied in a complex way during the period of accretion.

McCaffrey and Thomson (1980) determined that variation in dry-matter accumulation is due to a varying inorganic-matter contribution; the organic material accumulates in a highly linear manner over depth. This suggests that inorganic and

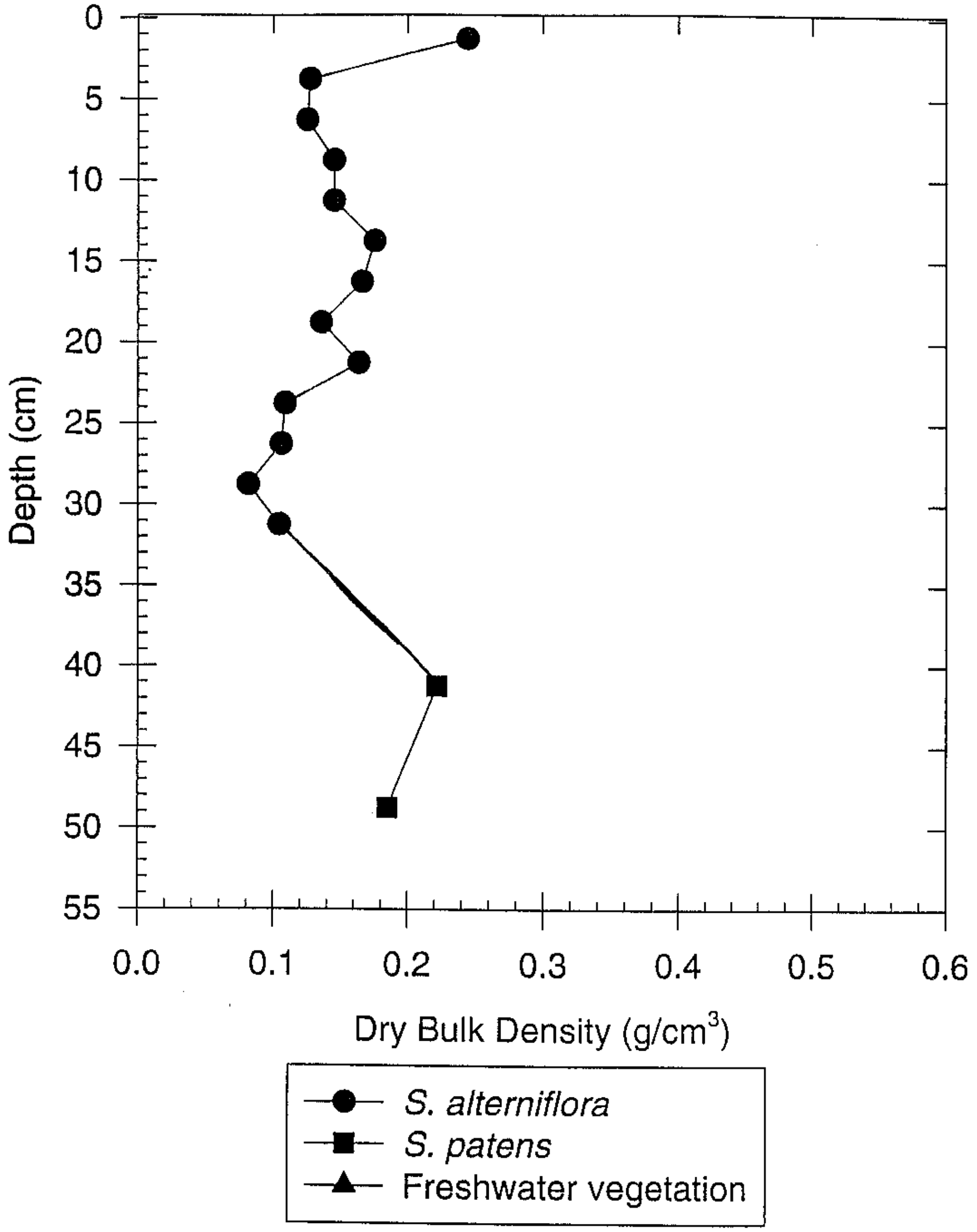


Figure 6.3 Dry bulk density (g/cm^3) vs. depth plot for Prime Hook Radio Station core.

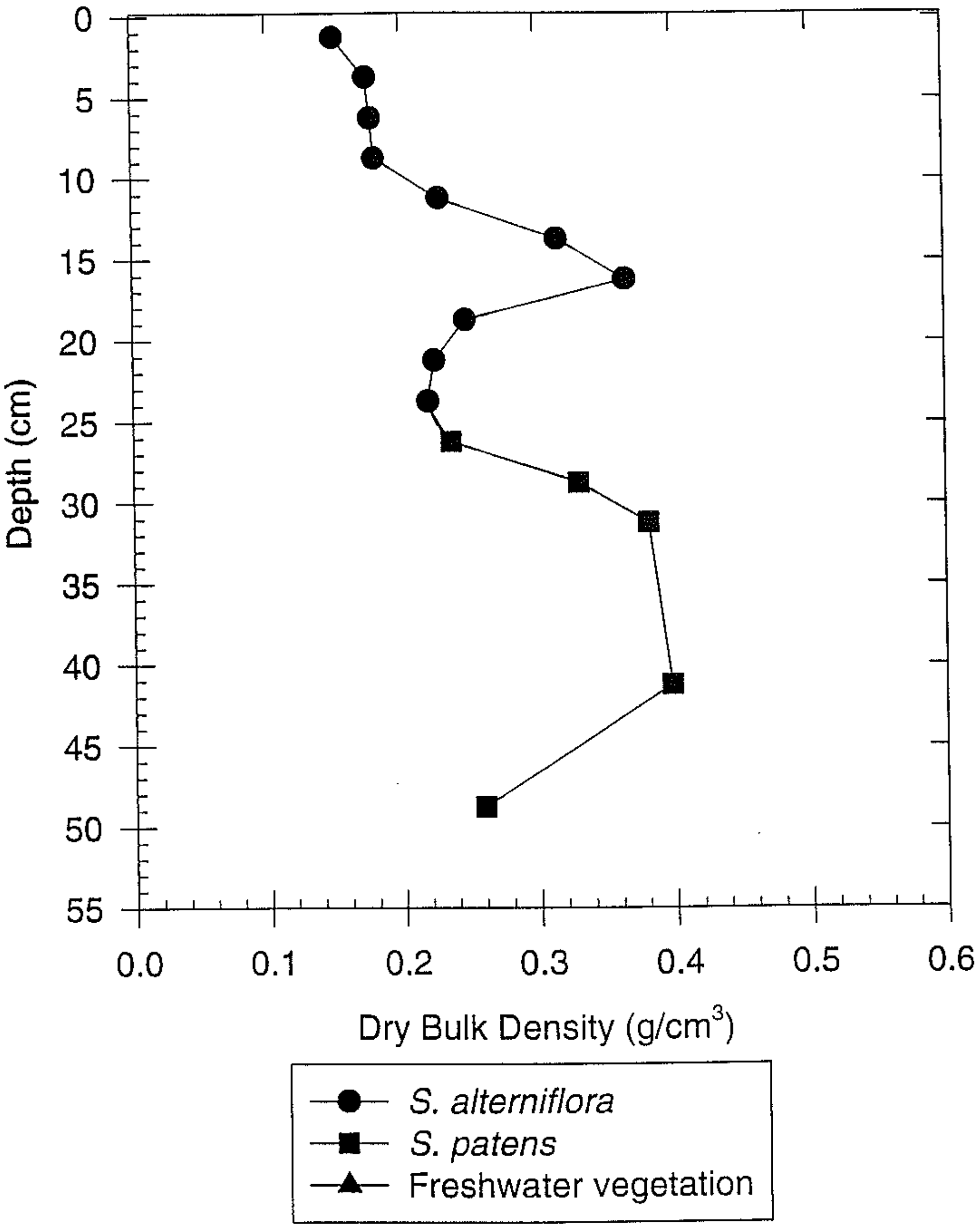


Figure 6.4 Dry bulk density (g/cm^3) vs. depth plot for Oyster Neck Road core.

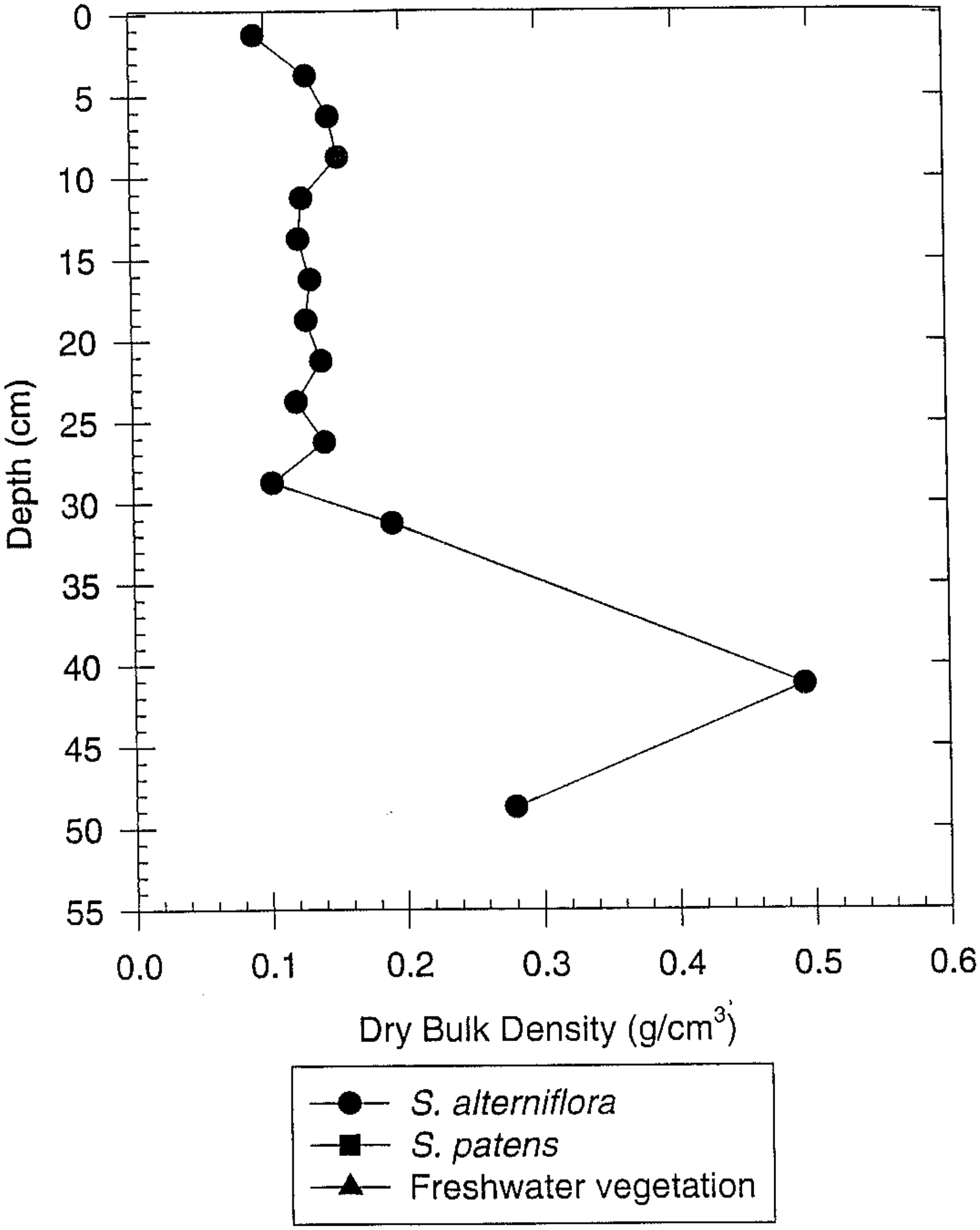


Figure 6.5 Dry bulk density (g/cm^3) vs. depth plot for Wolfe Runne core.

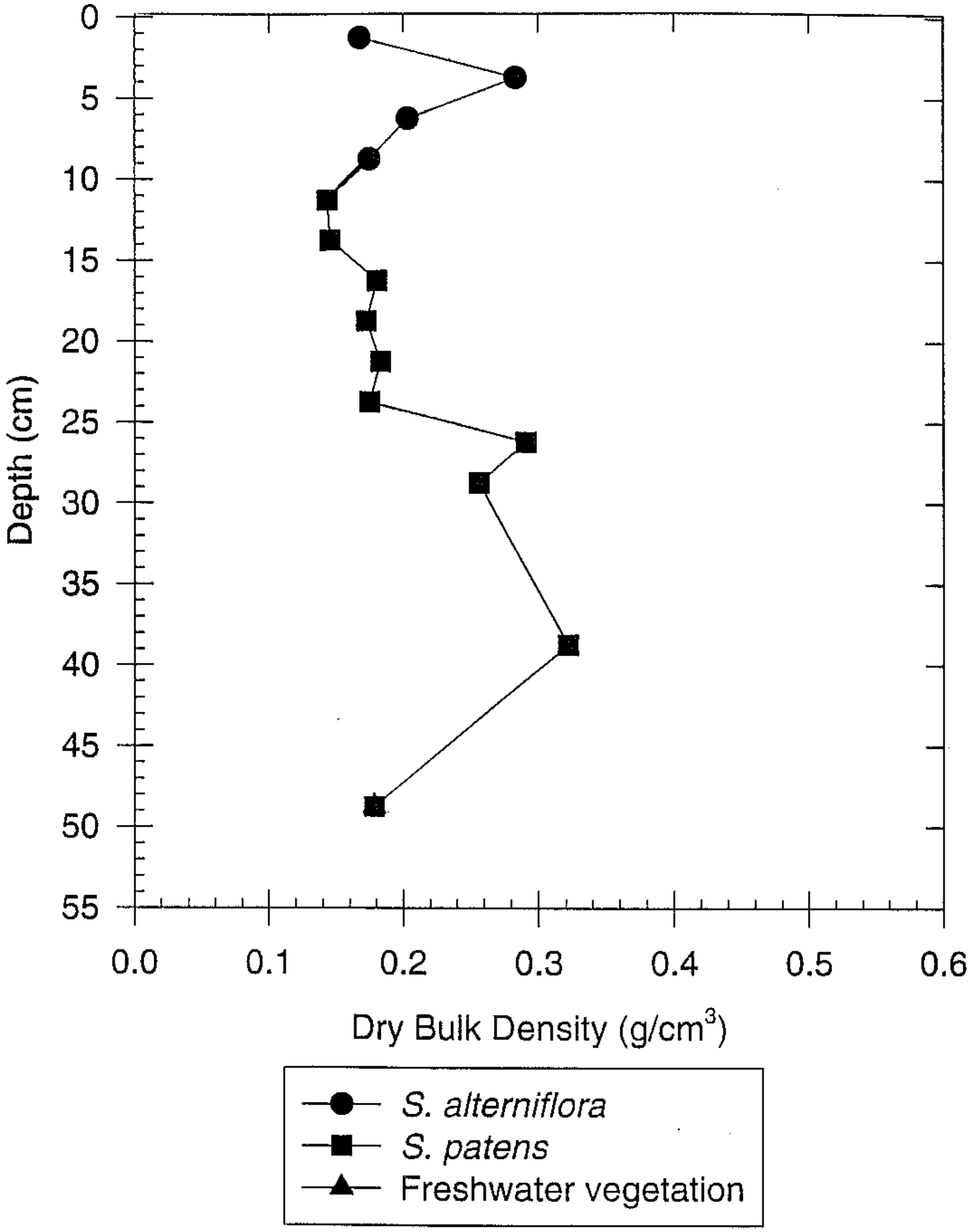


Figure 6.6 Dry bulk density (g/cm^3) vs. depth plot for Delaware Wildlands core.

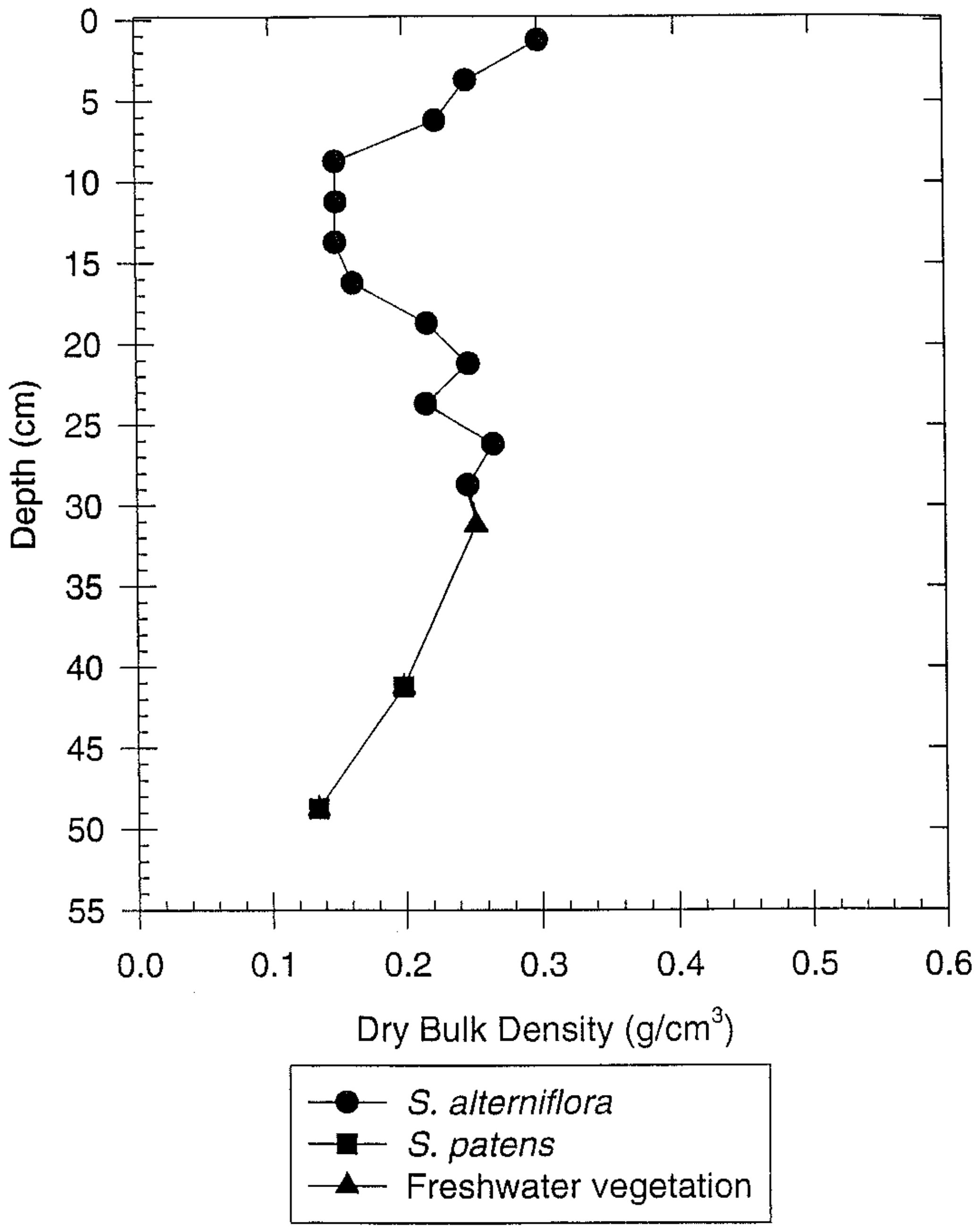


Figure 6.7 Dry bulk density (g/cm^3) vs. depth plot for Pot Nets North core.

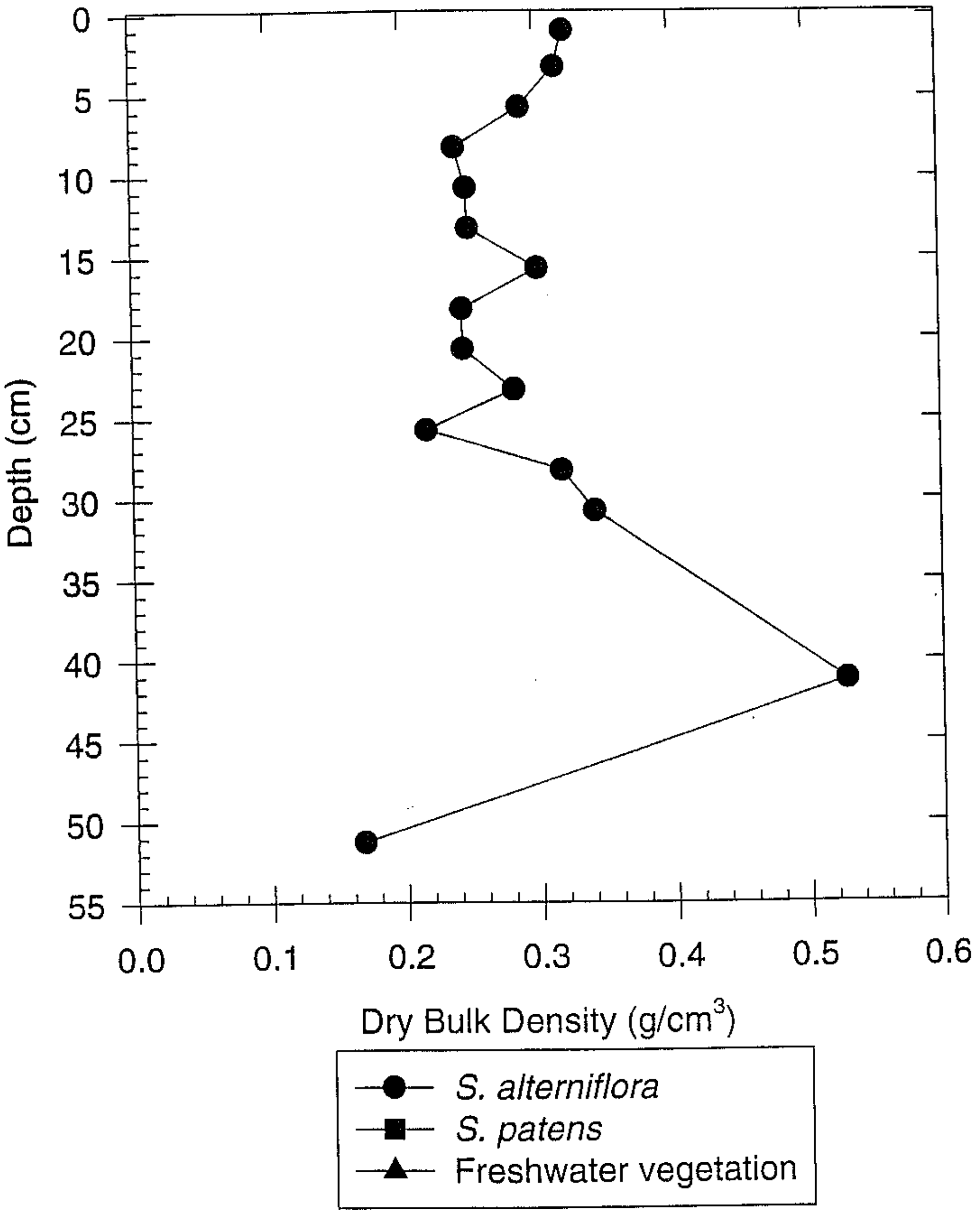


Figure 6.8 Dry bulk density (g/cm^3) vs. depth plot for Boat House Cove core.

organic matter accumulate independently (McCaffrey and Thomson, 1980). Armentano and Woodwell (1975) determined that there was no trend with depth in either organic content or density, and concluded that neither compaction nor decomposition occurs in the recently deposited materials.

Changes in bulk density reflect environmental change, as a transition from mineralogenic to organogenic sedimentation. For example, reduction in bulk density may reflect the stage at which marsh development first began at each site. At that time, finer-grained, more-organic materials began to accumulate, reflected by the higher LOI content of the core. These changes may also reflect either natural or human impacts on the wetland system such as storms, extreme flooding, or dam construction and erosion/sediment control implementation that would impact sediment supplied to the marsh system.

Scatterplot and correlation analyses (Pearson correlation coefficient, Bonferroni-adjusted probability, and Spearman correlation coefficient) of dry bulk density and organic matter content were conduced. Pearson correlation coefficients and Bonferroni probabilities ($P < 0.05$) suggest a linear relationship between dry bulk density and organic content in cores from Wolfe Runne, Oyster Neck Road, Prime Hook Radio Station, Pot Nets North, and Boat House Cove, but not in the Delaware Wildlands core.

Previous studies have found that soil bulk density varies inversely with organic matter content (McCaffrey and Thomson, 1980; Bricker-Urso *et al.*, 1989). Zones of relatively high inorganic matter are usually indicated by a pronounced increase in the dry bulk density plots. This is observed at the base of the marsh sequences in the

Wolfe Runne and Boat House Cove cores. In these cores, increases in dry bulk densities with depth indicate transitions between sub-environments or vegetative communities, or transition from a wetland environment and upland soils. Kearney *et al.* (1994) and Bricker-Urso *et al.* (1989) also found sharp increases in dry bulk density at depth in marsh cores, suggesting initiation of marsh development over pre-transgressive soil surfaces.

Moisture Content. Moisture content for the six cores ranges from 65.7% to 87.1% (Figures 6.9 to 6.14). Water clearly dominates the mass in these salt marsh sediments. Water content is relatively constant in the upper 25 cm of all cores, and varies by less than 20% over the entire 50 cm length of the cores. Statistical tests (Pearson coefficient correlation, Bonferroni probability, Spearman coefficient correlation, and regression analysis) show no significant variation of the moisture content with depth.

Loss on Ignition. Loss on ignition (LOI) content for the six cores ranges from 12.2% to 78.6% (Figures 6.9 to 6.14). Organic content is generally greatest in surface layers (top 5-20 cm), which are dominated by *S. alterniflora* roots and rhizomes. LOI values generally decrease with depth in core sections dominated by *Spartina patens* (Ait.) Muhl./*Distichlis spicata* (L.) Greene vegetation; however, LOI values increase with depth when freshwater vegetative remains are found at the base of the core (e.g., Pot Nets North core). A sharp increase in LOI% is also seen at 50 cm in the Boat House Cove and Wolfe Runne cores, where *S. alterniflora* vegetative remains characterize the entire 50 cm core length; i.e., there is no change in wetland

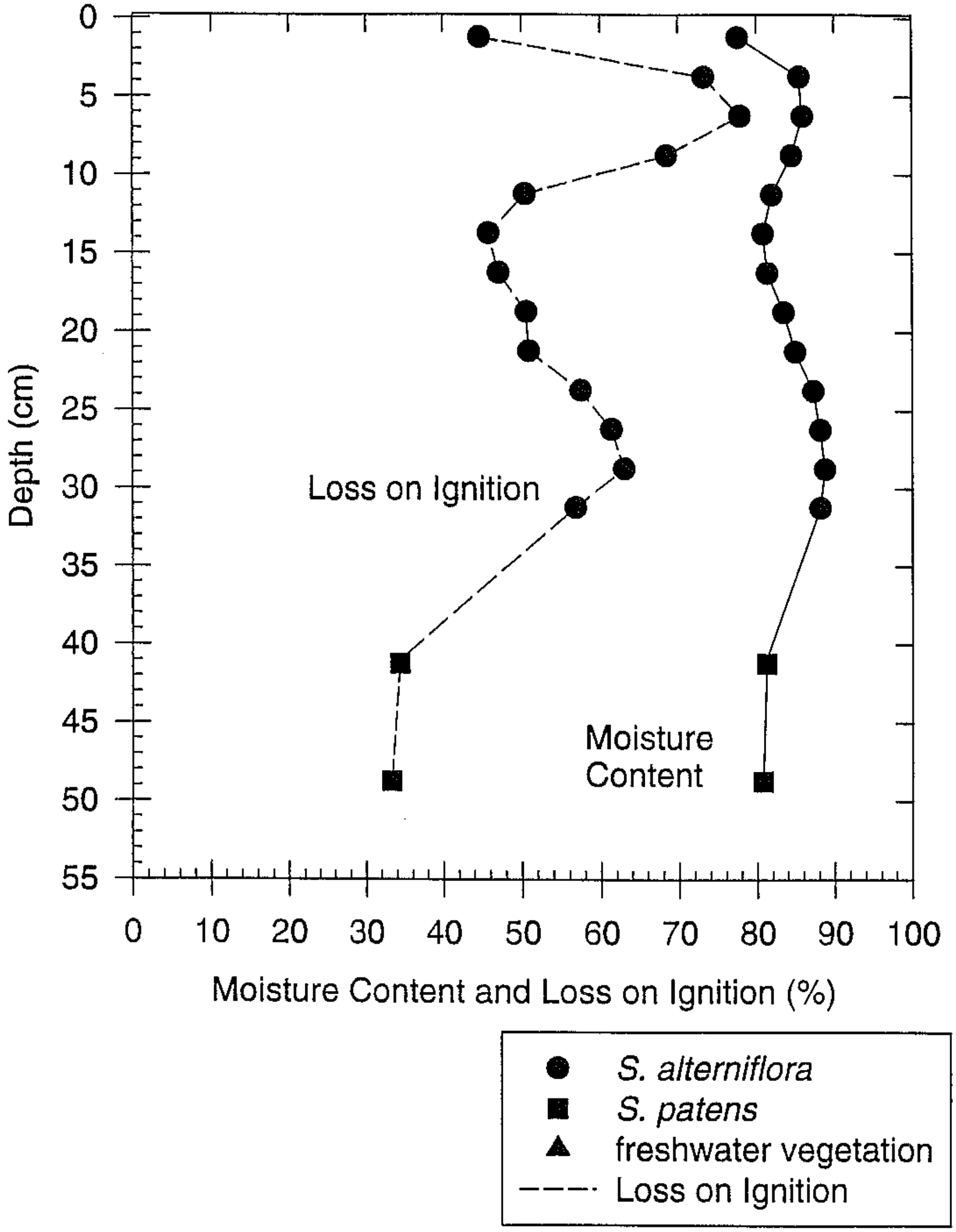


Figure 6.9 Moisture content (%) and Loss on ignition (%) vs. depth plot for Prime Hook Radio Station core.

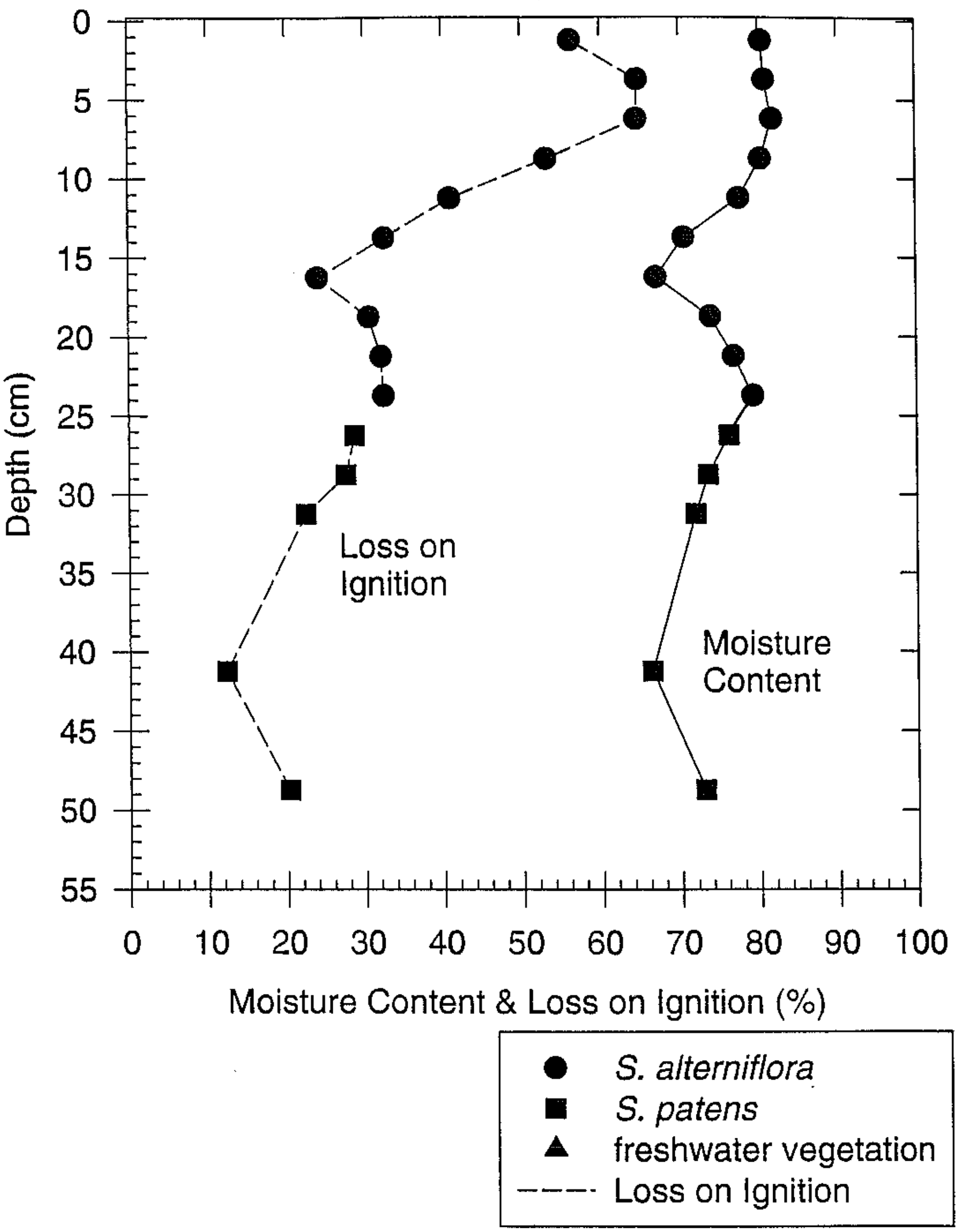


Figure 6.10 Moisture content (%) and Loss on ignition (%) vs. depth plot for Oyster Neck Road core.

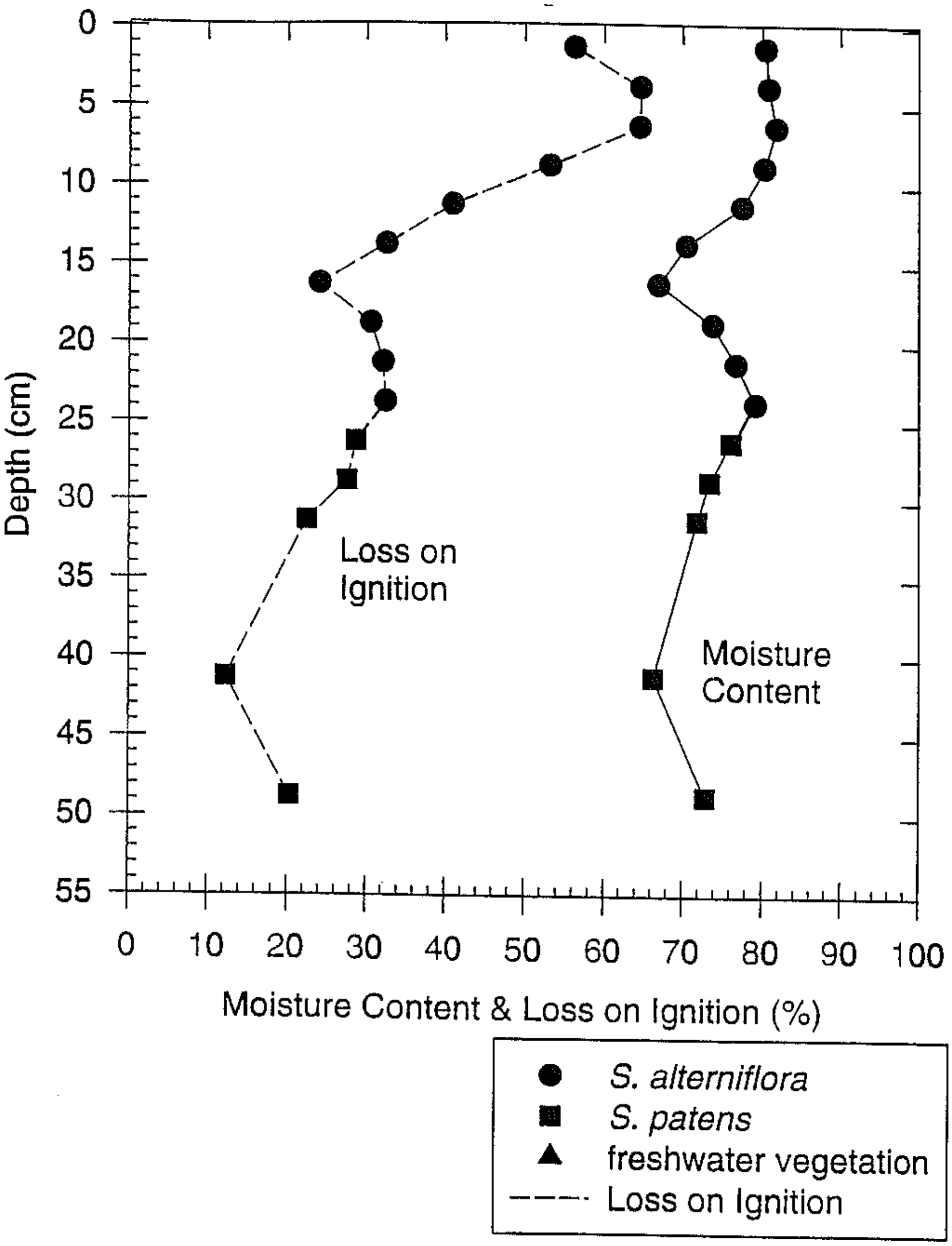


Figure 6.11 Moisture content (%) and Loss on ignition (%) vs. depth plot for Wolfe Runne core.

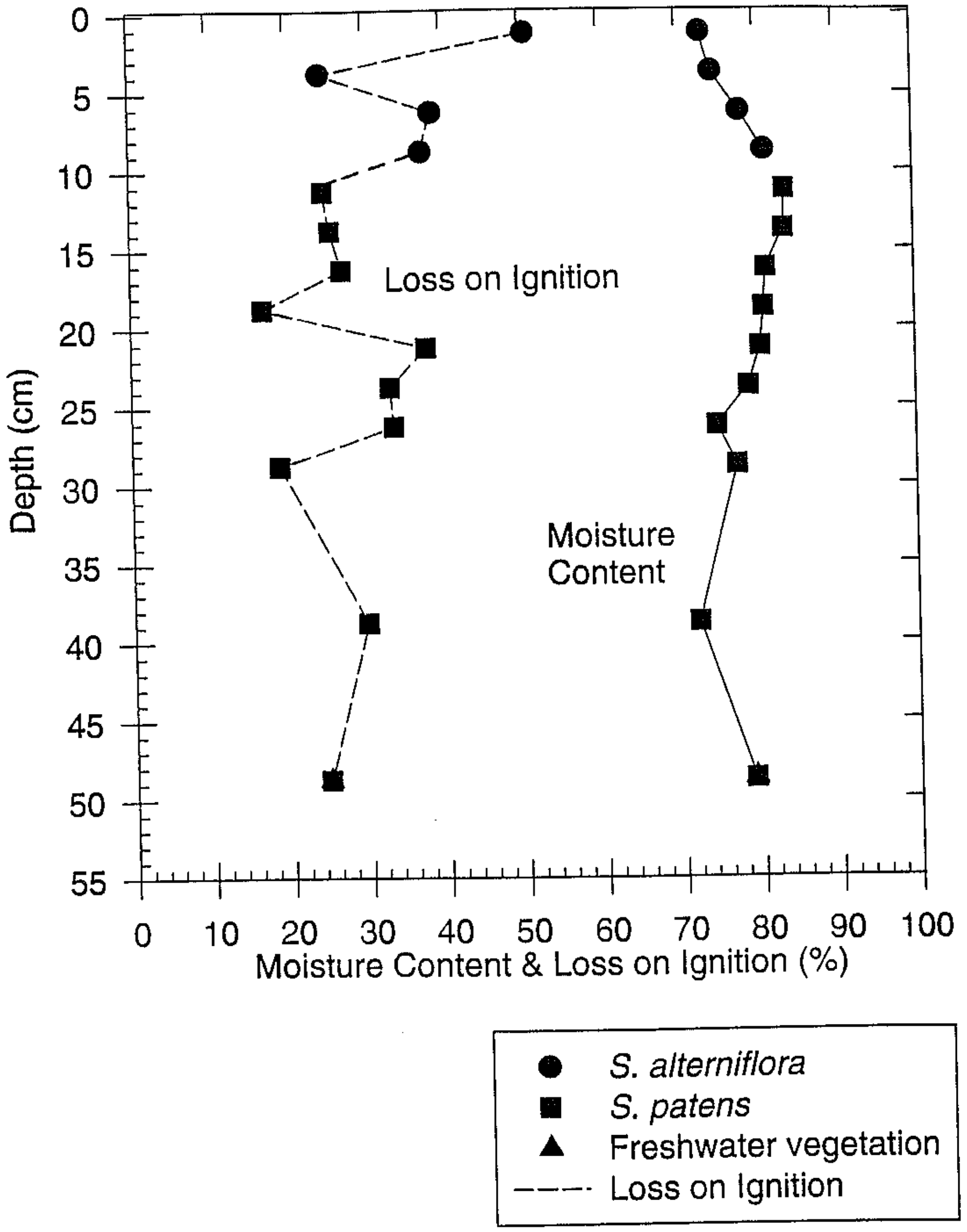


Figure 6.12 Moisture content (%) and Loss on ignition (%) vs. depth plot for Delaware Wildlands core.

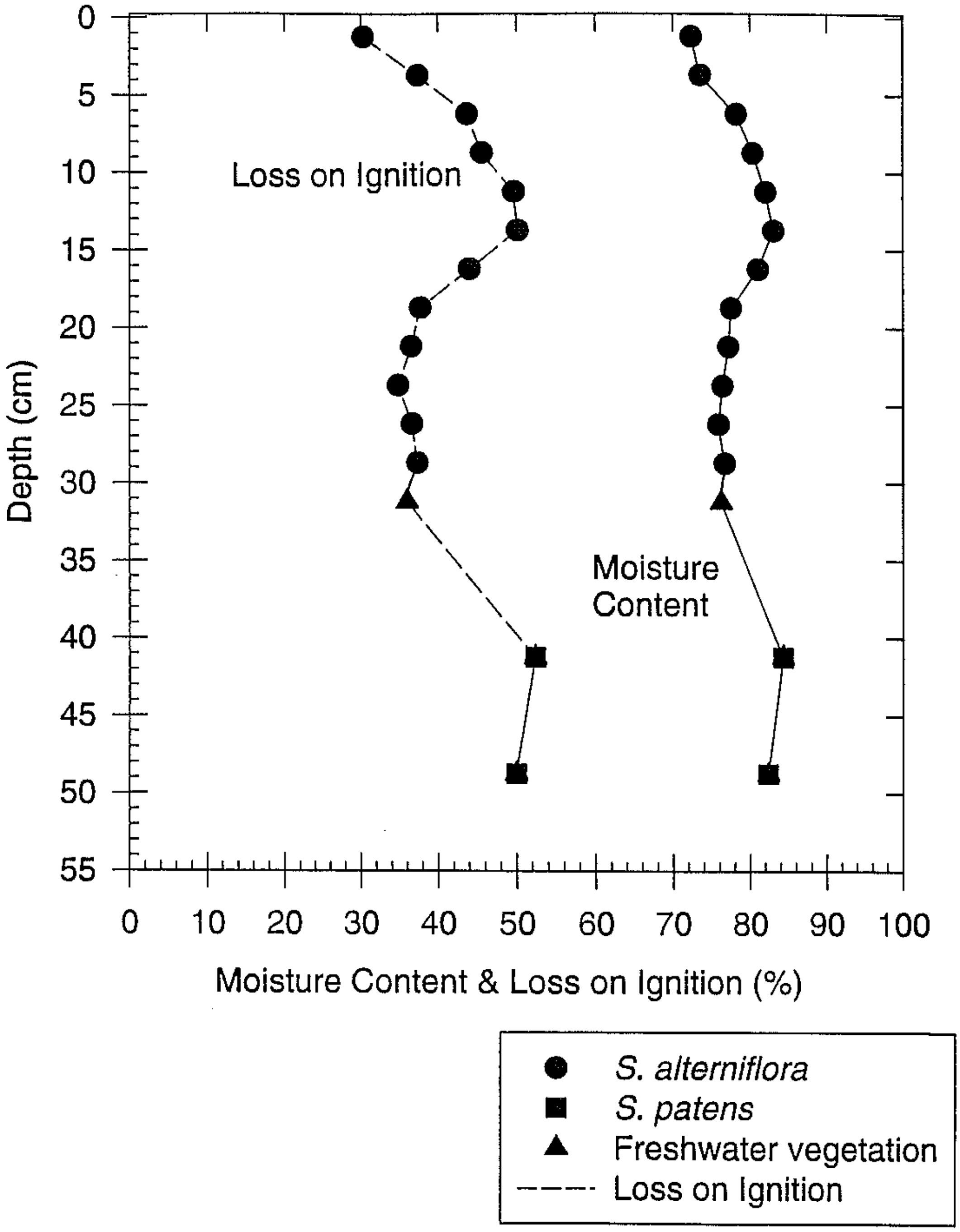


Figure 6.13 Moisture content (%) and Loss on ignition (%) vs. depth plot for Pot Nets North core.

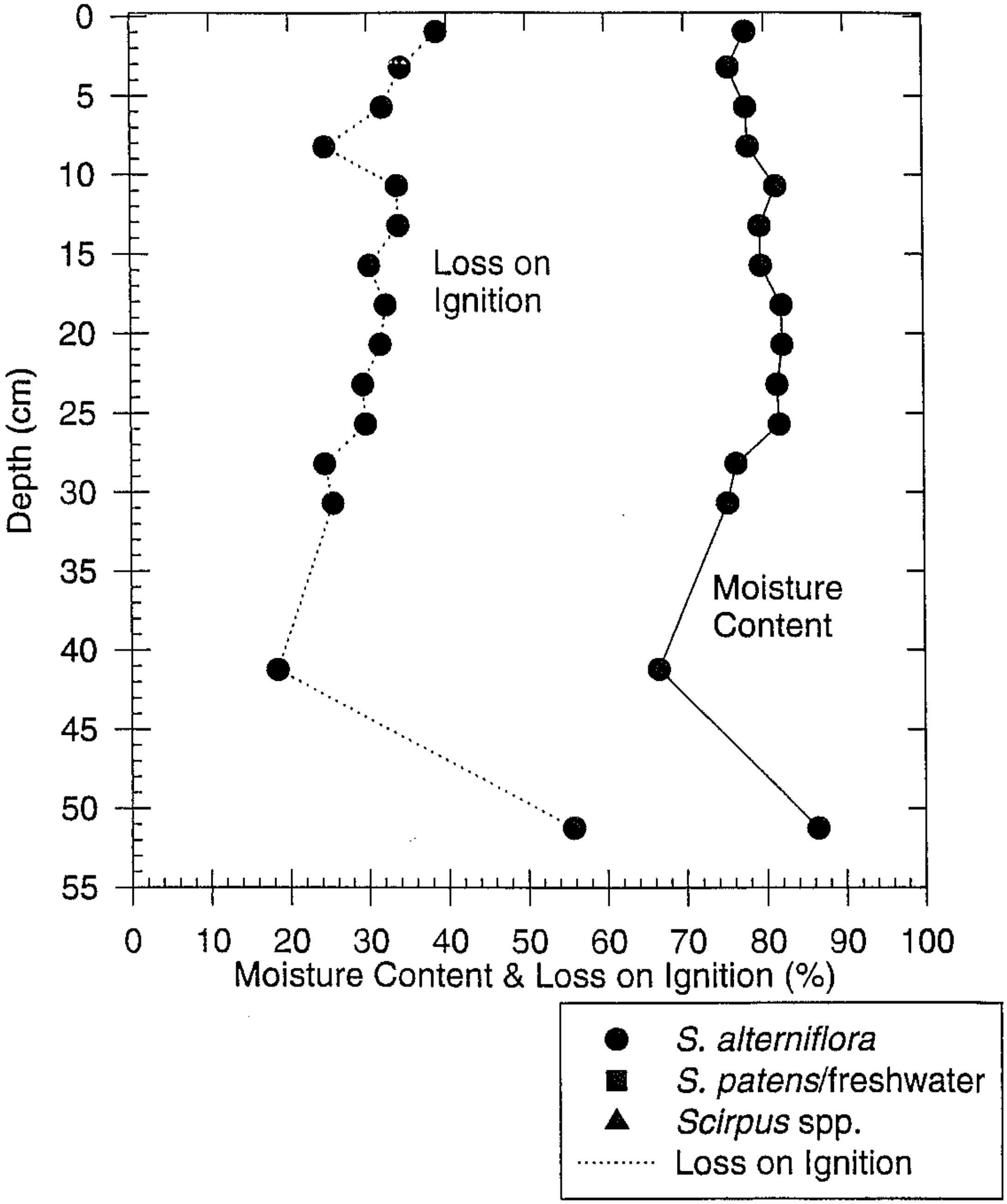


Figure 6.14 Moisture content (%) and Loss on ignition (%) vs. depth plot for Boat House Cove core.

subenvironment with depth, but the increase in LOI% is likely indicative of a change in condition of the *S. alterniflora* marsh.

The results demonstrate that these "upper low-marsh" samples are highly organic, with loss on ignition values as high as 78%. However, LOI values are reduced below 15 cm depth in most cores, reflecting either a change in marsh sub-environment from low to high marsh, or a biological feedback system whereby organic below-ground productivity is affected by local environmental conditions in the marsh (e.g., tidal inundation or waterlogging).

The range of LOI content determined in this study (12.7 - 78.6%) generally exceeds those reported for Rhode Island salt marshes (30-60%, Bricker-Urso et al, 1989), Delaware marshes (14-33%, Church *et al.*, 1981; 20-40%, O'Shea, 1986), and South Carolina marshes (3-20%, Sharma *et al.*, 1987) (Table 6.1). It is likely that the high organic content in the "upper low marshes" sampled in this study is due to position relative to tidal inundation and frequency, and a biological feedback mechanism which possibly stimulates organic productivity in the absence of mineralogic sediment supply. Sampling sites were located at the upper edge of the *S. alterniflora* zone, away from streamside influences and near the *S. patens/D. spicata* border. Rates of organic matter accumulation may be higher in these irregularly flooded marshes because mineral sediment input is generally lower. Higher organic content in the upper *S. alterniflora* salt marsh may be caused by several factors, including: 1) reduced rates of decomposition caused by waterlogging, and 2) reduced tidal flooding and export of

Table 6.1 Range of concentrations of organic matter in upper layers (0-30 cm) of salt marshes of eastern U.S.A. (after Kim *et al.*, in press).

Location	Surface Vegetation	Organic Matter (%)	Reference
Indian River Bay, DE	*S. alterniflora*	25 - 56	This study
Rehoboth Bay, DE	*S. alterniflora*	18 - 52	This study
Rehoboth Bay, DE	*S. alterniflora*	32 - 54	This study
Delaware Bay, DE	*S. alterniflora*	23 - 62	This study
Wolfe Glade, DE	*S. alterniflora*	44 - 79	This study
Wolfe Glade, DE	*S. alterniflora*	44 - 79	Kim *et al.*, in press
Great Marsh, DE	*S. alterniflora*	14 - 33	Church *et al.*, 1981
Nanticoke River, MD	*S. cynosuroides*	20 - 43	O'Shea, 1986
Rhode Island	*S. alterniflora*	30 - 60	Bricker-Urso *et al.*, 1989
North Inlet, SC	*S. patens*	3 - 20	Sharma *et al.*, 1987

detritus (Hackney and de la Cruz, 1980; Stevenson *et al.*, 1986; Craft *et al.*, 1988, 1993).

This relatively high LOI content may be indicative of the importance of vegetation and organic material in vertical accretion and maintenance of surface elevation at the wetlands/uplands boundary, as opposed to mineral or inorganic components of sedimentation. Stevenson *et al.* (1986) reported that waterlogging of the soils slows decomposition and enhances organic matter accumulation as compared to coastal wetlands that are flooded by tides. Similarly, Craft *et al.* (1993) demonstrated the contribution of organic matter to vertical accretion by enhancing soil volume.

Autocompaction

Previous studies on shallow cores of marsh sediments as well as data from this study on both moisture content and organic content of core samples obtained for radionuclide analyses suggest that compaction effects will not affect radiometric dates. However, autocompaction is commonly cited as a process which can influence interpretations of both geochronologic data and the evolution of coastal wetlands. Coastal wetland deposits may consolidate extensively under their own weight as they accumulate (Bloom, 1964; Kaye and Barghoorn, 1964; Pizzuto and Schwendt, in press). Autocompaction is the consolidation of sedimentary deposits under their own weight, due to the slow (but steady) expulsion of pore fluids from the sediment over time, by overburden or loading. Muds and peats compact due to the progressive decrease in pore space over time resulting from overburden loading. The amount of sediment compaction (or void reduction) is controlled by: sediment type (lithology, including organics), initial water content, the rate of loading and thickness of the overburden, the initial thickness of the sedimentary unit, and the underlying substrate.

Autocompaction is a significant factor in the evolution of coastal wetland deposits in areas of thick, highly organic sediments, with the greatest compaction in freshwater marsh peats, rather than salt marsh peats (J. Pizzuto, personal communication). Autocompaction must be considered when reconstructing sea level histories from dated marsh deposits. Dated stratigraphic horizons may not be at the elevation of original deposition. Valid sea level histories can only be developed when dated horizons are restored to the original elevation of deposition; i.e. the amount of compaction must be evaluated, especially when dealing with deep valley fills of muddy organic sediments. However, the effects of compaction are not considered in this study; the shallow depth of cores obtained for radionuclide dating (50 cm in length) permits determination of relatively short-term accretion rates without accounting for decomposition, dewatering, compression, and compaction that marsh sediments would be subject to upon deeper burial.

Radionuclide Analyses

General. ^{210}Pb and ^{137}Cs analyses were used to examine historical rates of sediment accumulation over the past 30-150 years. ^{210}Pb and ^{137}Cs analyses data and gamma counting results are included in Appendix G. There are few techniques available for measurement of sedimentation rates over shorter periods of time. Marker horizons such as storm debris (Siccama and Porter, 1972; Stumpf, 1983) and specifically applied material (Bloom, 1967), or stakes (Ranwell, 1964) have been used as markers in wetlands against which deposition or erosion of sediments has been measured over a several-year period. Such techniques are appropriate for determination of extremely short term rates and are subject to errors such as erosion of marker horizons and atypical deposition at the site due to the marker itself (Armentano and Woodwell, 1975). Armentano and Woodwell (1975) document that the naturally occurring radioisotope ^{210}Pb is a useful dating technique that is free of such errors, and other investigations have shown that in undisturbed sediment, ^{210}Pb activity along a vertical profile of sediment provides an accurate estimate of the mean rate of deposition (Koide *et al.*, 1972; Krishnaswami *et al.*, 1971; Armentano and Woodwell, 1975).

The total amount of ^{210}Pb in sediments includes both 'supported' (derived from sediment ^{226}Ra radioactive decay) and 'unsupported' (derived from atmospheric fallout) components. 'Supported' ^{210}Pb is produced from radioactive decay of ^{226}Ra, a daughter product in the ^{238}U series, and a precursor of ^{210}Pb. The amount of ^{226}Ra-supported ^{210}Pb activity is calculated indirectly by determining the amounts of ^{214}Bi and ^{214}Pb in the sediments. 'Unsupported' or excess ^{210}Pb is derived from ^{222}Rn gas, which escapes from surface rocks into the troposphere, and decays through short-lived daughter elements to ^{210}Pb. Excess ^{210}Pb is added to the stratosphere through atmospheric mixing, or returned to the earth's surface by precipitation where it is absorbed by fine-grained inorganic sediments. After deposition into sediments, the atmospherically derived ^{210}Pb decays exponentially as a function of residence time. In analyses of sediment cores, excess (unsupported) atmospherically derived ^{210}Pb is estimated by subtracting the ^{226}Ra-supported ^{210}Pb from the total amount of ^{210}Pb in the sediments. Analysis of ^{210}Pb compares the activity of 'unsupported' or 'excess' ^{210}Pb with the background activity of 'supported' ^{210}Pb. An average sediment accumulation rate over the last 100-150 years is determined by plotting the log of the unsupported ^{210}Pb activities per cumulative sediment dry weight versus depth, and calculating an accumulation rate over intervals of constant slope (Koide *et al.*, 1972; 1973; Hakanson and Jansson, 1983).

The ^{137}Cs dating technique is based on the known chronology of this man-made radioisotope (half life of 30 years) produced in nuclear explosions. Atmospheric fallout began in 1954-1955 and reached a peak around 1964 (Hakanson and Jansson, 1983). A horizon can be dated by first appearance (1954-1955) or peak activity (1964) of ^{137}Cs, and the depth and date of this horizon can be used to calculate a sediment accumulation rate.

Determination of Vertical Accretion Rates: ^{210}Pb. In order to examine the historical patterns of deposition of the radionuclide, the sediment cores were dated using two models. Accretion rates were calculated from the results of ^{210}Pb analyses using both the constant initial activity model, and the constant flux model (McCaffrey and Thomson, 1980; Oldfield and Appleby, 1984; Bricker-Urso *et al.* 1989). The maximum dating range for these methods is 100-150 years, depending on the amount of excess ^{210}Pb compared to background levels. In addition, the nature of the sediment (e.g., grain size, organic matter content and distribution, salt content) and possible bioturbation and other post-depositional processes may also impact ^{210}Pb distributions in the sediment column (Church *et al.*, 1981; Sharma *et al.*, 1987; Bricker-Urso *et al.*, 1989; French *et al.*, 1994).

Sediment accretion rates determined through these two ^{210}Pb dating methods must be evaluated under the assumptions of the constant flux and the constant initial concentration methods. The constant flux method assumes that no ^{210}Pb is supplied with the sediment, and that all the ^{210}Pb is supplied from the atmosphere, at a known and constant rate (atmospheric flux rate). The ^{210}Pb, once deposited from the atmosphere, is rapidly taken up by sediment particles (Appleby and Oldfield, 1978). Atmospheric flux rate for Lewes, Delaware, based on rainwater collections is 0.8 dpm/cm^2/yr (Hartman, 1987; Hussain, unpublished data). Additionally, based on calculated values of ^{210}Pb inventories in three cores from Wolfe Glade, Delaware, an average atmospheric flux of ^{210}Pb (obtained by dividing the inventory with average life of ^{210}Pb ($1/\lambda$)), is 0.7 dpm/cm^2/yr (Kim *et al.*, 1996).

The constant initial concentration method assumes that all the ^{210}Pb is supplied attached to the sediment, that none of the ^{210}Pb is supplied from the atmosphere, and that the activity of the ^{210}Pb on the sediment is constant. Additionally, it is assumed that no migration, addition, or subtraction of ^{210}Pb from the sediment column has occurred other than by radioactive decay (Koide *et al.*, 1972). If these assumptions are satisfied for the constant initial activity method, the excess ^{210}Pb activity will decrease exponentially with depth in the sediment.

In theory, the concentrations of ^{210}Pb decrease exponentially with increasing depth. The distribution of specific excess ^{210}Pb activity follows a simple exponential decay curve. This simple exponential decay curve indicates a relatively constant rate of sediment accretion, while variations from the curve may indicate many possible factors (e.g., that accretion may not have been constant through time) (Bricker-Urso *et al.*, 1989), as is evident in some of the curves obtained in this study. Assuming a constant sedimentation rate and a constant flux of ^{210}Pb to the sediments, the mean sedimentation rate can be calculated from the slope of the regression equation.

Constant Initial Activity Model. In the constant initial activity method, ^{210}Pb is assumed to be deposited with the sediment, that the sediment has a constant initial activity, and that the sedimentation rate is constant. The vertical accretion rate is calculated from an equation, and the age of a sediment slice is determined (Koide *et al.*, 1972; 1973; Oldfield and Appleby, 1984).

The sedimentation rate is calculated from the relationship (Koide *et al.*, 1972; Oldfield and Appleby, 1984):

$$A_z = A_o e^{(-\lambda z/S)}$$

where: λ = the decay constant of ^{210}Pb (0.03114/year)

z = depth (cm) in core

A_o = activity (dpm/g) of excess ^{210}Pb at z=0 (surface)

(measured as ^{210}Pb - ^{226}Ra)

A_z = activity (dpm/g) of excess ^{210}Pb at depth z (cm) in core

S = sediment accretion rate (cm/yr)

The exponential function equation can be transformed into a linear equation by taking Ln:

$$\mathrm{Ln}A_z = \mathrm{Ln}\, A_o - (\lambda/S)z$$

A (specific excess activity ^{210}Pb (dpm/g)) vs. z (depth) should plot as a straight line on semi-log scale. The sedimentation rate is obtained from the slope of the line ($-\lambda/S$). Therefore, the sedimentation rate, S, is proportional to the slope of the log excess ^{210}Pb vs. depth, from which A_o is calculated.

Figures 6.15 through 6.19 depict the semi-log plots of specific excess ^{210}Pb (dpm/g) vs. depth (cm) for the five sample sites. Sedimentation rates derived from the slopes of these regression lines range from 0.31 ± 0.03 cm/yr to 0.52 ± 0.09 cm/yr, with a mean rate of 0.39 ± 0.04 cm/yr (standard deviation: 0.09; 95% CI = 0.11).

Rates for cores analyzed in this study are presented in Table 6.2. These rates lie within the range of 0.26 to 1.1 cm/yr determined for other salt marshes in the United States by the ^{210}Pb method (Armentano and Woodwell, 1975; McCaffrey, 1977; Goldberg *et al.*, 1979).

Table 6.2 Vertical accretion rates determined by ^{210}Pb analyses.

Site	^{210}Pb accretion rate (cm/yr) (constant initial activity)	^{210}Pb accretion rate (cm/yr) (constant flux)
Boat House Cove	0.42 ± .03	0.41 ± .02
Delaware Wildlands	0.52 ± .09	0.26 ± .01
Pot Nets North	0.32 ± .03	0.35 ± .01
Oyster Neck Road	0.40 ± .05	0.34 ± .02
Wolfe Runne	0.31 ± .03	0.32 ± .02

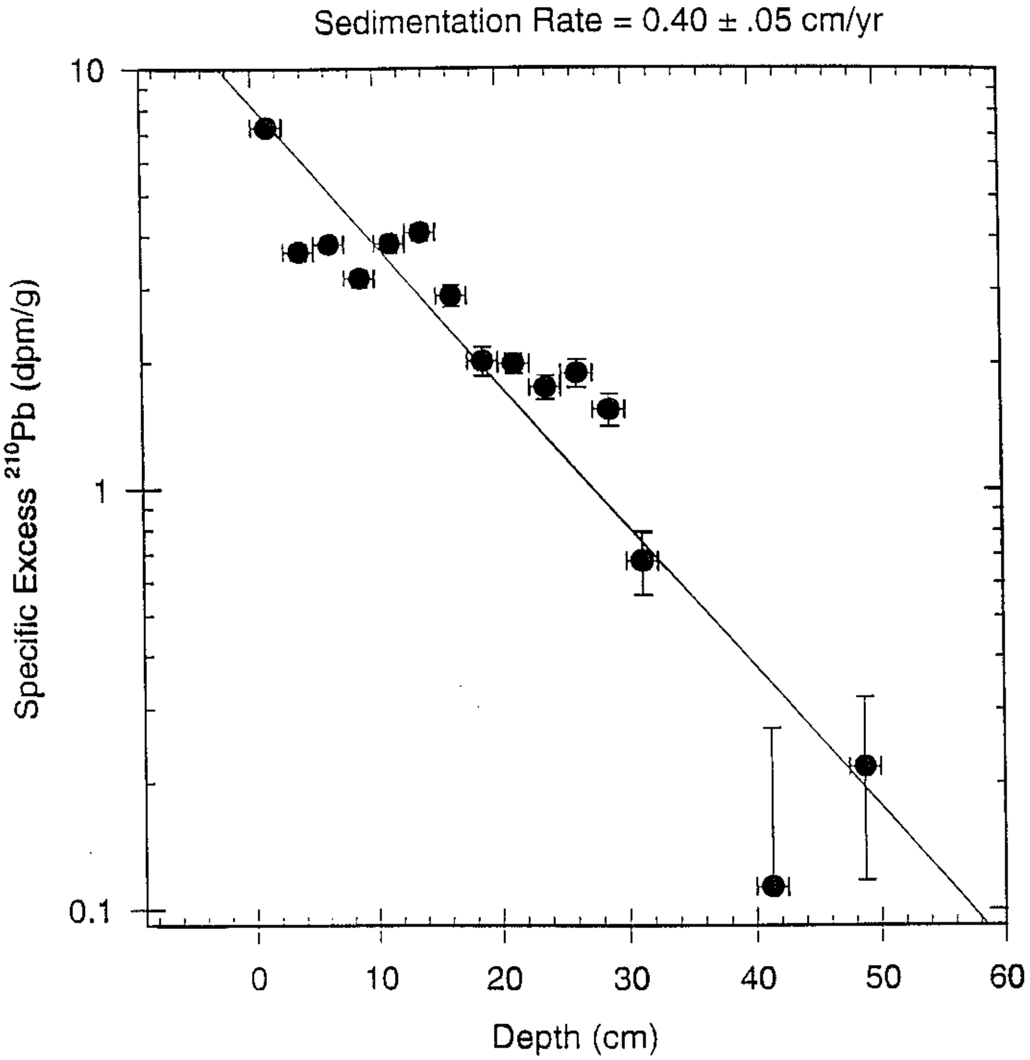


Figure 6.15 Depth profile of excess ^{210}Pb in Oyster Neck Road salt marsh (constant initial activity method). The absence of a vertical error bar indicates that counting areas are smaller than the symbol size.

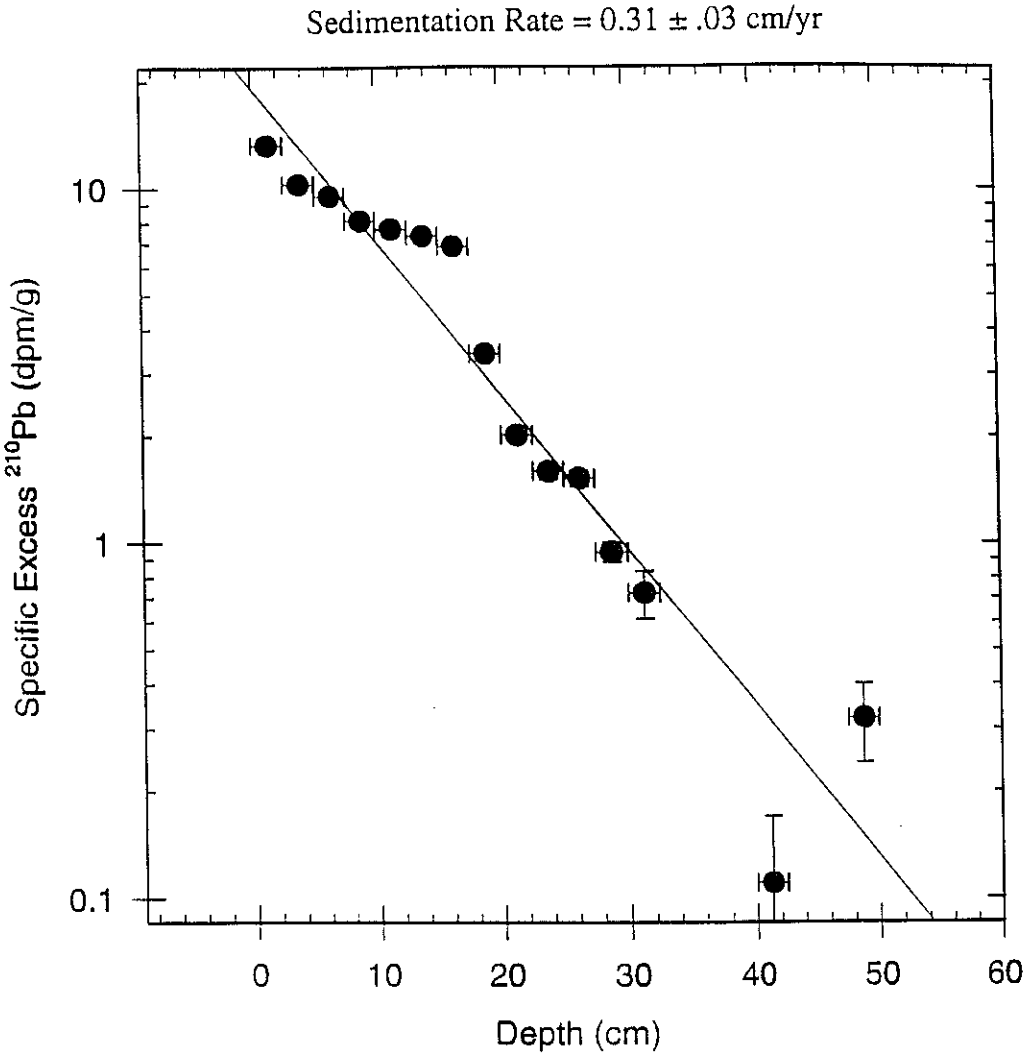


Figure 6.16 Depth profile of excess ^{210}Pb in Wolfe Runne salt marsh (constant initial activity method). The absence of a vertical error bar indicates that counting areas are smaller than the symbol size.

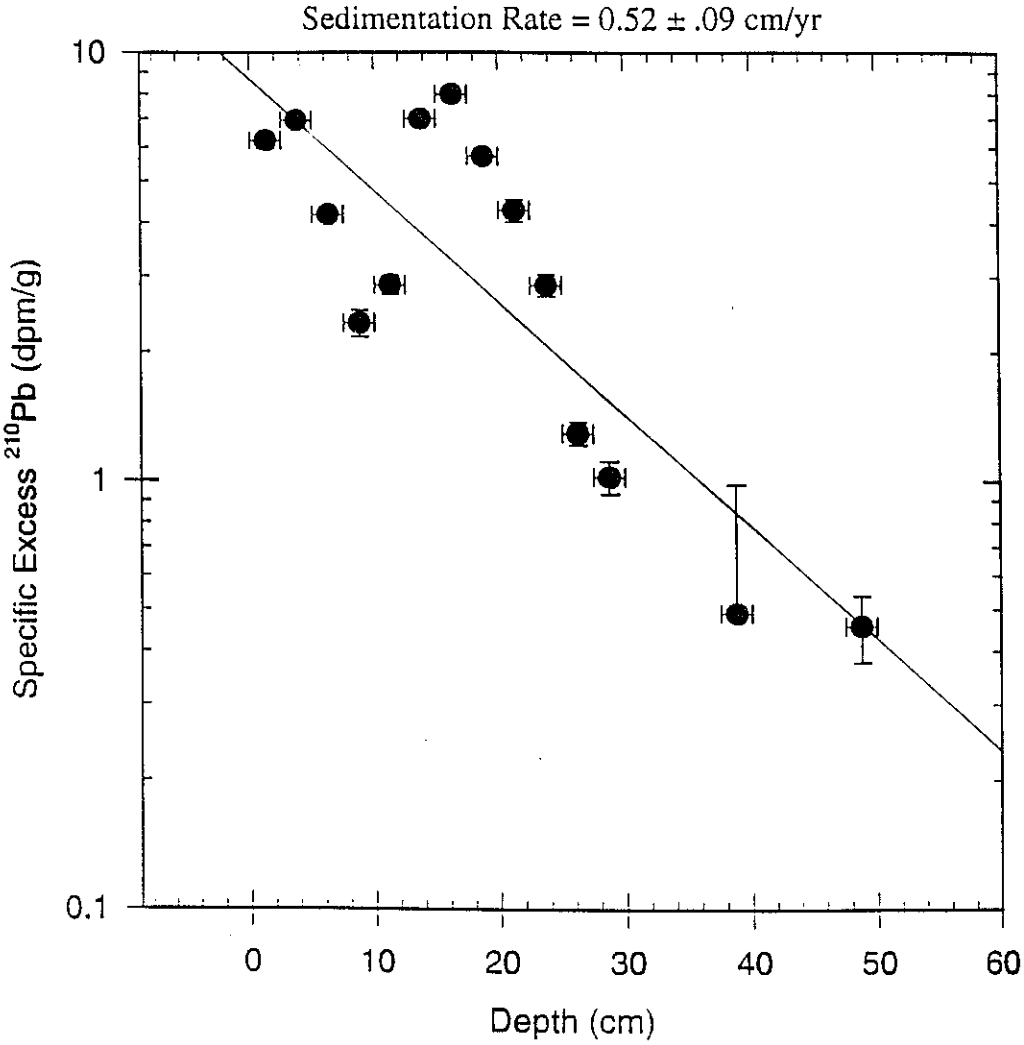


Figure 6.17 Depth profile of excess ^{210}Pb in Delaware Wildlands salt marsh (constant initial activity method). The absence of a vertical error bar indicates that counting areas are smaller than the symbol size.

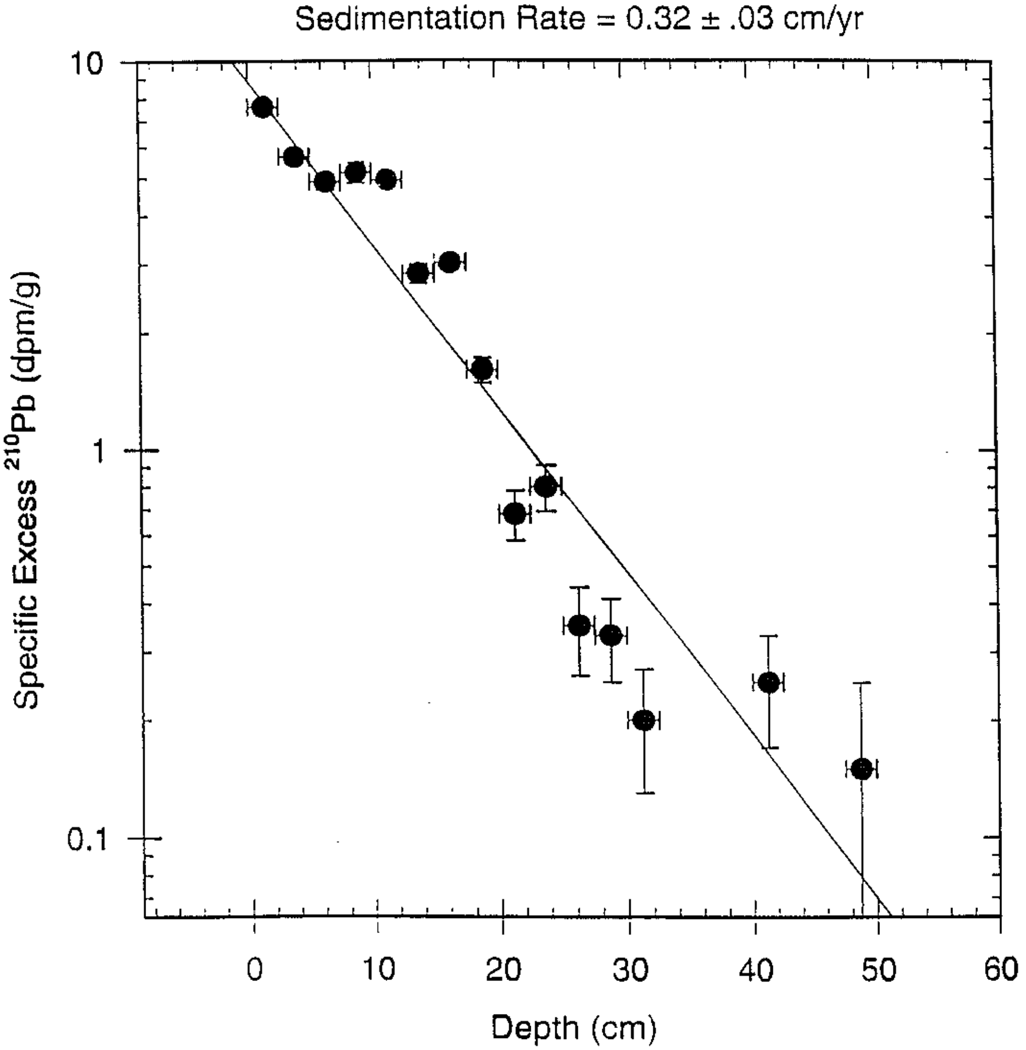


Figure 6.18 Depth profile of excess ^{210}Pb in Pot Nets North salt marsh (constant initial activity method). The absence of a vertical error bar indicates that counting areas are smaller than the symbol size.

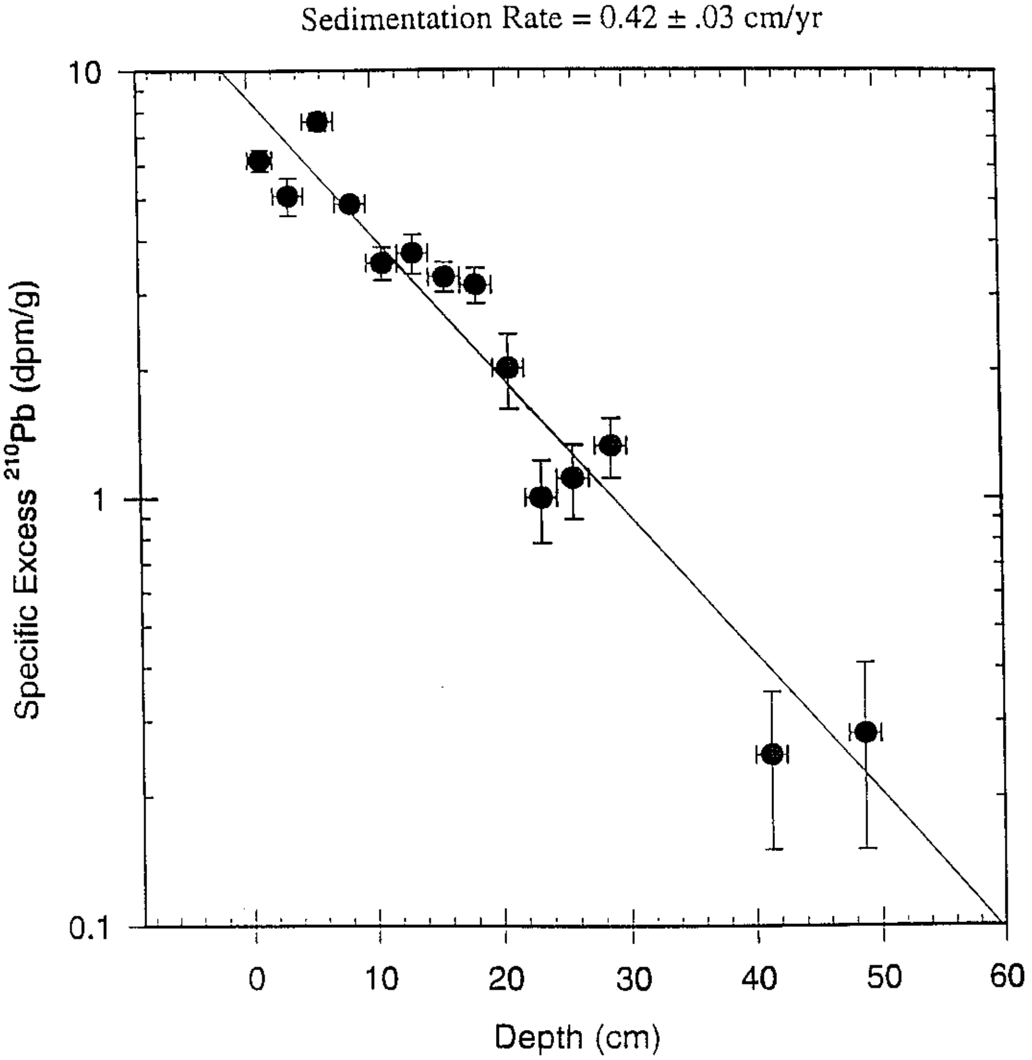


Figure 6.19 Depth profile of excess ^{210}Pb in Boat House Cove salt marsh (constant initial activity method). The absence of a vertical error bar indicates that counting areas are smaller than the symbol size.

Constant Flux Model. As seen in the semi-log plots of specific excess ^{210}Pb, the distribution of excess ^{210}Pb activity is not always a simple exponential decay curve. Variations in the curve suggest that the sedimentation rate may not have been constant through time. This may be a result of compaction of the marsh sediment (unlikely in this study), variation in sediment supply, changing tidal flooding conditions, or effects of organic content on ^{210}Pb calculations. Therefore, in sediments which show departures from an exponential decrease with depth, the constant flux model may be used to determine accretion rate, with the primary assumption that the sites have been exposed for most of the time to the atmosphere.

The constant flux model interprets irregularities as changes in vertical accretion rate through time (Goldberg, 1963; Appleby and Oldfield, 1978; McCaffrey and Thomson, 1980; Bricker-Urso *et al.*, 1989), taking into account possible effects of compaction or other external factors. The age of a core section at any given depth is calculated by the relationship:

$$A_z = A_o e^{-\lambda t}$$

where

t = age in years

A_o = total amount of excess ^{210}Pb in sediments

A_z = total amount of excess ^{210}Pb below depth z

λ = decay constant of ^{210}Pb (0.03114/yr)

The age of a core section in this model is provided by converting this equation to the form:

$t = -1/\lambda \ \mathrm{Ln}\ (1- \sum A_z/\sum A)$

where: $\sum A_z$ = integrated unsupported ^{210} Pb activity from surface to depth z (cm)

$\sum A$ = total unsupported ^{210}Pb activity in the core

λ = decay constant of ^{210}Pb (0.03114 /yr)

The calculated ages of sections of the core are shown in Figures 6.20 to 6.24, and vertical accretion rates are shown in Table 6.2. Vertical accretion rates determined from the constant flux method range from 0.26 ± 0.01 cm/yr to 0.41 ± 0.02 cm/yr, with a mean vertical accretion rate of 0.34 ± 0.02 cm/yr (standard deviation: 0.05; 95% CI = 0.07). The straight line in many of the diagrams indicate a constant rate of sediment accumulation; additionally, the straight line is an indirect test which suggests that the constant flux method can well describe the geochronology of these cores.

Another check of the assumptions made of the constant flux method, i.e., the assumption that all ^{210}Pb is supplied from the atmosphere at constant rate and/or that discontinuities in accumulation have not occurred, can be accomplished by summing the cumulative ^{210}Pb activity in each core. This sum (cumulative inventory) of ^{210}Pb activity (dpm/cm^2) in the core should approach a steady state which equals the deposition rate of ^{210}Pb in the area (C_l) divided by the decay constant (K). ^{210}Pb deposition rates calculated for the Lewes, Delaware, area by Hartman (1987), Hussain (unpublished data), and Kim *et al.* (1996) are 0.8 dpm/cm^2/yr and 0.7 dpm/cm^2/yr, respectively; the decay constant (K) is 0.03114. Table 6.3 presents measured ^{210}Pb

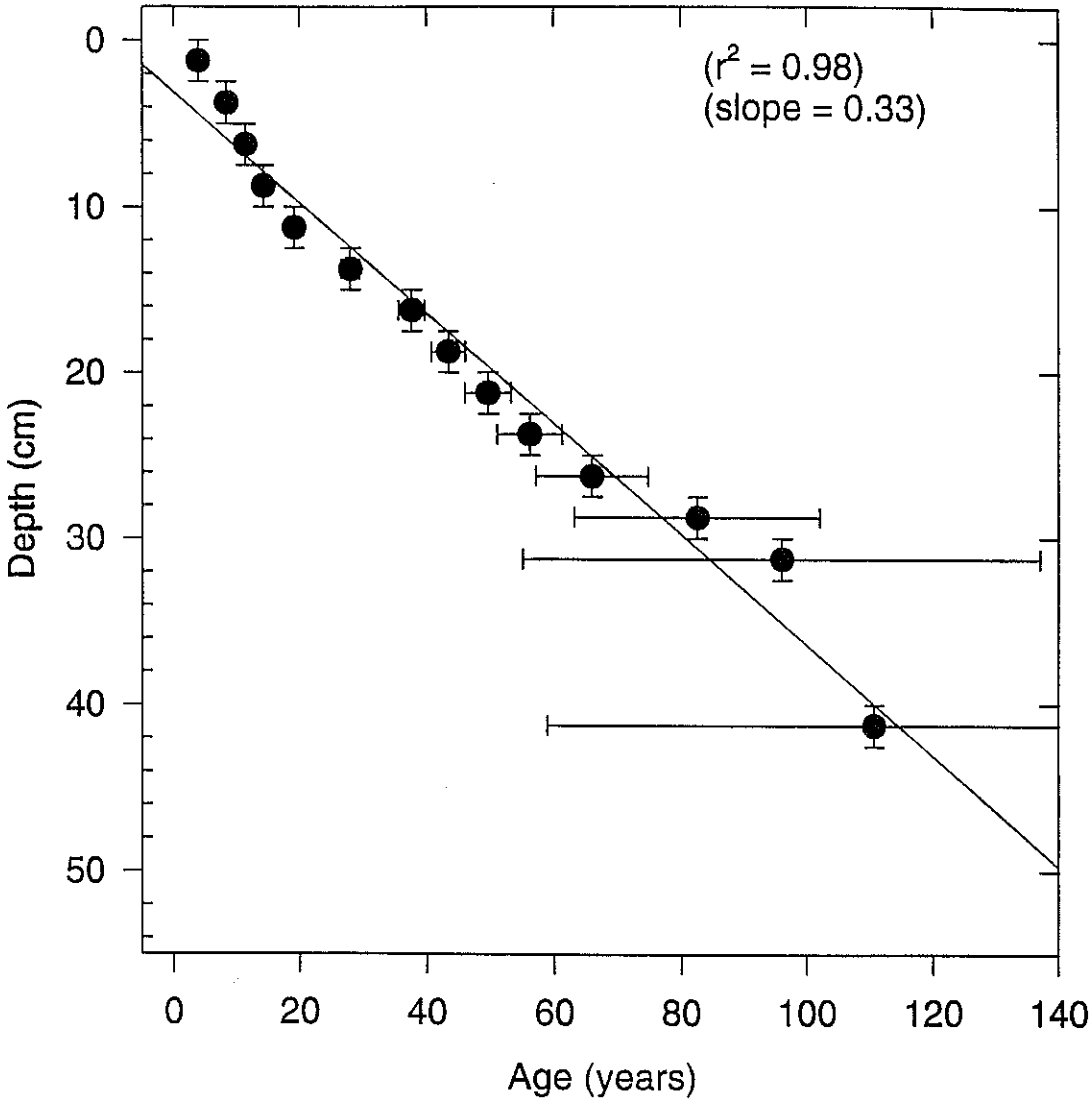


Figure 6.20 ^{210}Pb dates for the Oyster Neck Road salt marsh core (constant flux method). The absence of a horizontal bar indicates that age errors are smaller than the symbol size.

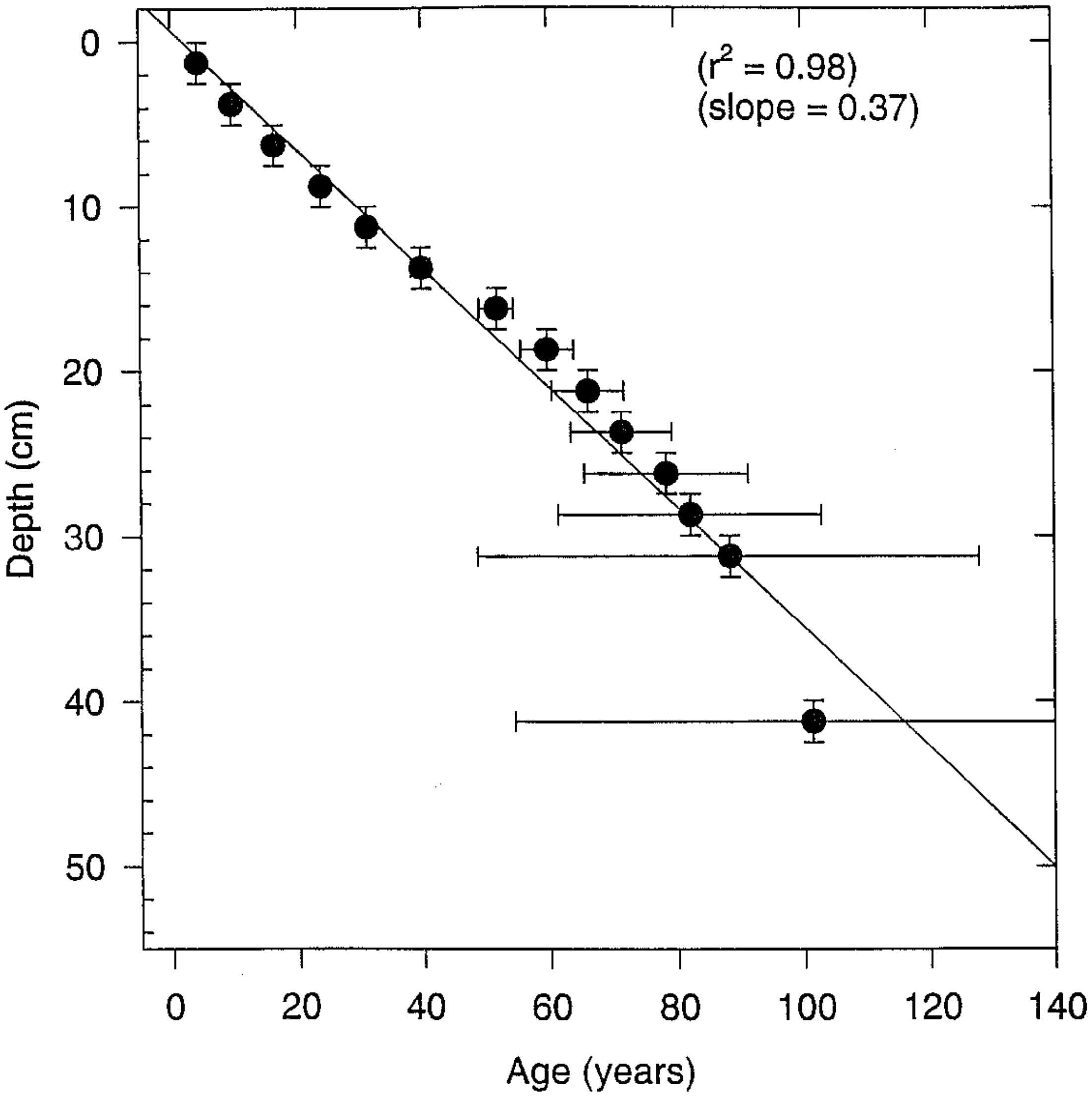


Figure 6.21 ^{210}Pb dates for the Wolfe Runne salt marsh core (constant flux method). The absence of a horizontal bar indicates that age errors are smaller than the symbol size.

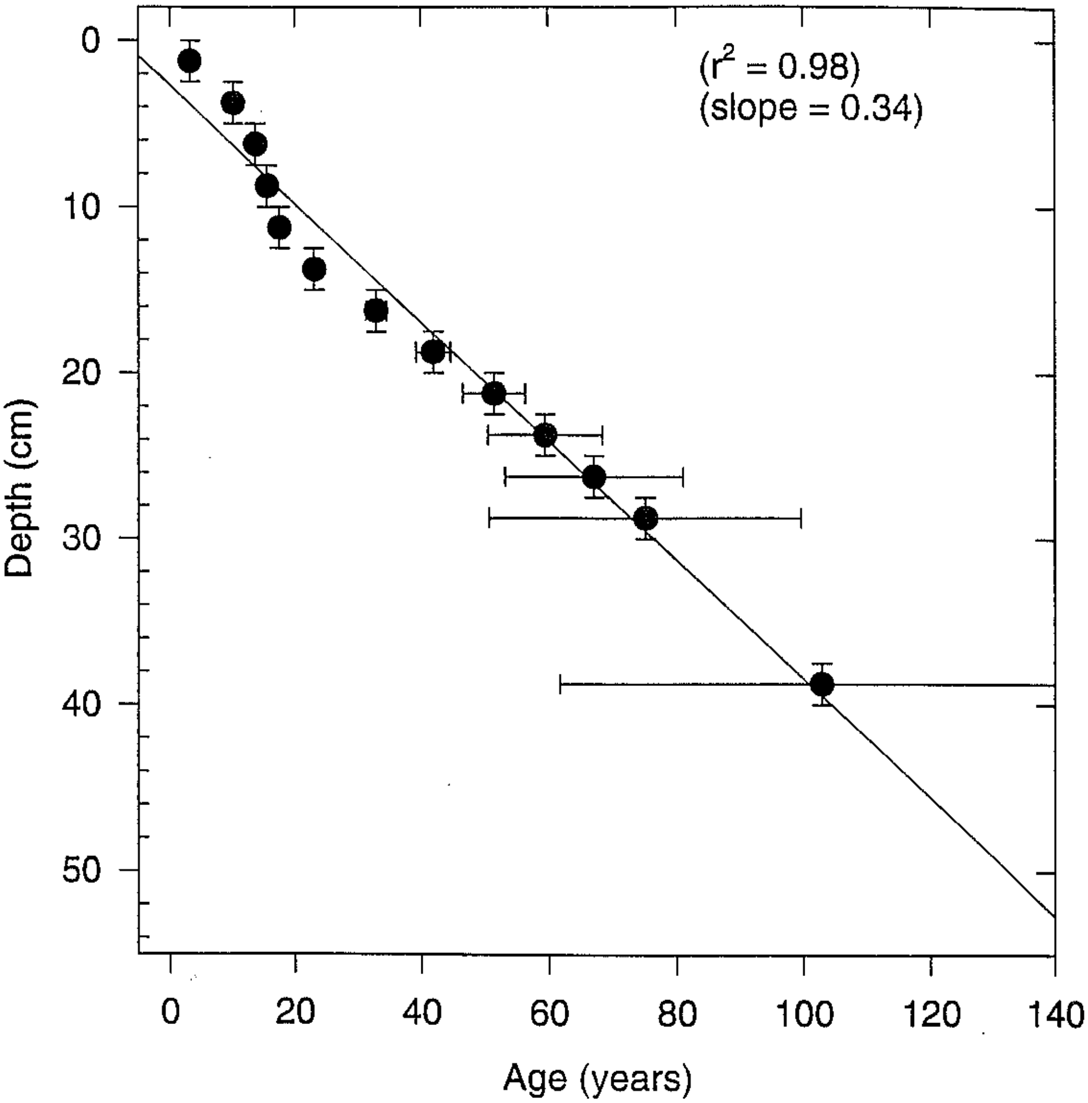


Figure 6.22 ^{210}Pb dates for the Delaware Wildlands salt marsh core (constant flux method). The absence of a horizontal bar indicates that age errors are smaller than the symbol size.

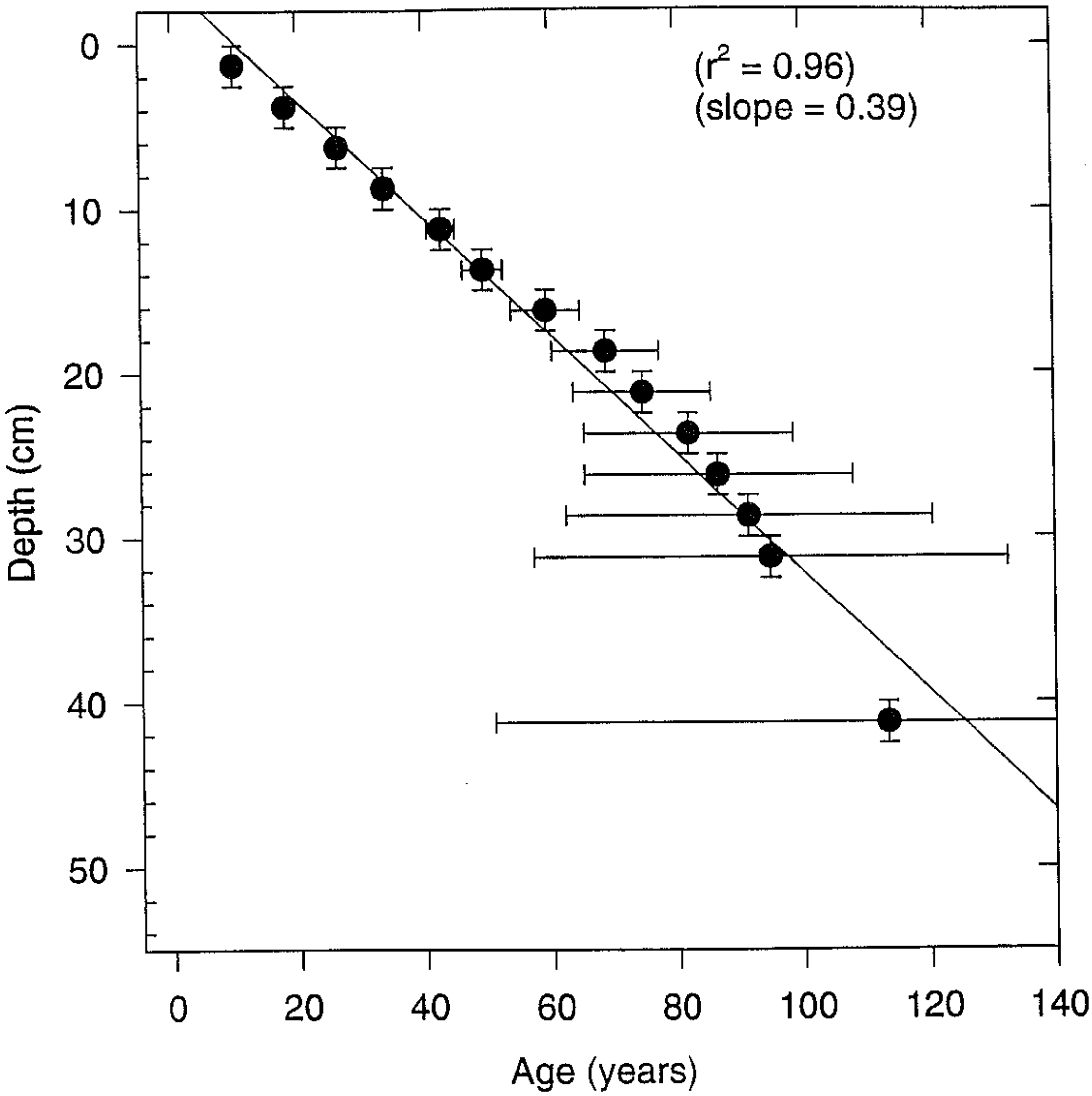


Figure 6.23 ^{210}Pb dates for the Pot Nets North salt marsh core (constant flux method). The absence of a horizontal bar indicates that age errors are smaller than the symbol size.

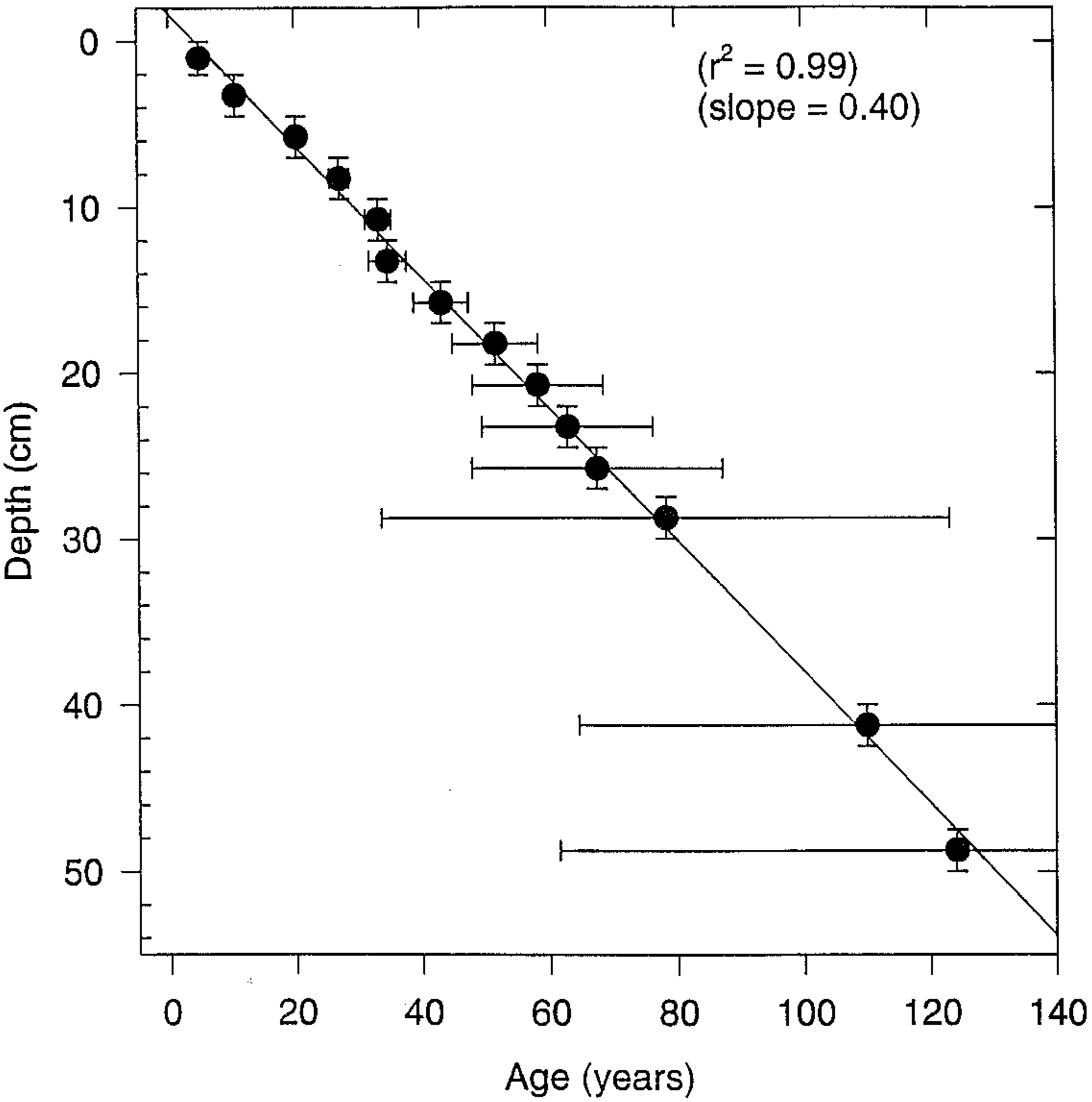


Figure 6.24 ^{210}Pb dates for the Boat House Cove salt marsh core (constant flux method). The absence of a horizontal bar indicates that age errors are smaller than the symbol size.

inventories (dpm/cm^2) for salt marshes in this study, those calculated from known deposition rates, and those measured for U.S. lowland soils. Close agreement of ^{210}Pb inventories indicates that the assumptions of the constant flux method have been satisfied.

Table 6.3 Inventories of ^{210}Pb in salt marsh soils (this study) and lowland soils of the United States (dpm/cm^2) (after Kim *et al.*, 1996).

Site	^{210}Pb Inventory (dpm/cm^2)
Boat House Cove	27.68
Wolfe Runne	25.06
Delaware Wildlands	26.00
Pot Nets North	21.26
Oyster Neck Road	22.22
calculated: C_I/K (0.8/0.03114)	25.69
calculated: C_I/K (0.7/0.03114)	22.48
Delaware Salt Marsh*	25.60
U.S. Lowland (mean)*	27.70

* Data from Graustein and Turekian (1986) cited by Kim *et al.* (1996)

^{137}Cs Activity

General. In the northern hemisphere, the atmospheric fallout of ^{137}Cs from nuclear testing started in 1954 (Perkins and Thomas, 1980), peaked in 1963

(Hakanson and Jansson, 1983), and then declined substantially due to curtailment of above-ground nuclear testing. The peak of ^{137}Cs concentration in sediment is assumed to be the result of the peak in atmospheric ^{137}Cs fallout, and can be used as a geochronological tracer layer (Sharma et al, 1987; Zwolsman *et al.*, 1993; French *et al.*, 1994). Identification of the 'peak concentration' ^{137}Cs horizon therefore permits calculation of a time-averaged sediment accumulation rate for the overlying sediment. As described by Kim *et al.* (in press), ^{137}Cs has been often used as a geochronologic tracer in sediments of lakes (Krishnaswami *et al.*, 1971; Miller and Heit, 1986), salt marshes (DeLaune *et al.*, 1978; Sharma *et al.*, 1987; Chmura and Kosters, 1994), and estuaries and bays (Hutchinson and Prandle, 1994; Sugai *et al.*, 1994).

The utility of ^{137}Cs in assessing salt marsh sedimentation rates depends on the following assumptions (Milan *et al.*, 1995): 1) the marsh is rapidly accreting; 2) there is no import or export of ^{137}Cs into the system other than from atmospheric fallout, and 3) the ^{137}Cs is fixed in the sediment column and does not migrate vertically in either direction.

Results. As depicted in Figures 6.25 to 6.29, a ^{137}Cs peak was detected in each of the five cores analyzed, with peak ^{137}Cs activity occurring at approximately the same level (8.25 to 11.25 cm) in each core. Maximum activity of ^{137}Cs (1963 horizon) permits calculation of vertical accretion rates by dividing amount of sediment accumulated by the specific time interval. The accretion rates determined through use of ^{137}Cs as a geochronologic tracer are 'short term' rates, measuring vertical accretion in the past 30 years.

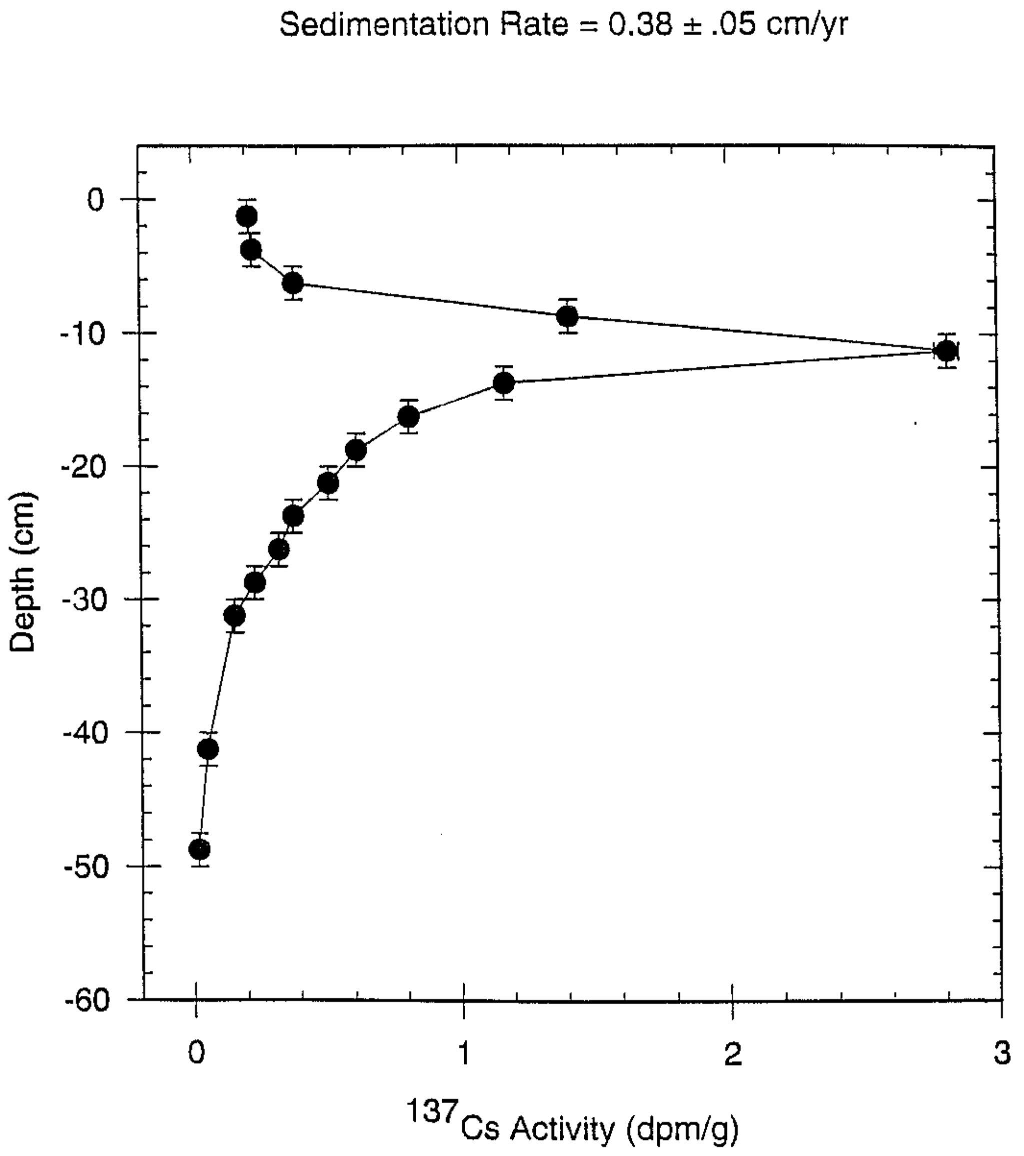


Figure 6.25 Depth profile of ^{137}Cs in Oyster Neck Road salt marsh core. Horizontal bars represent counting errors; absence of those indicates that counting errors are smaller than symbol size.

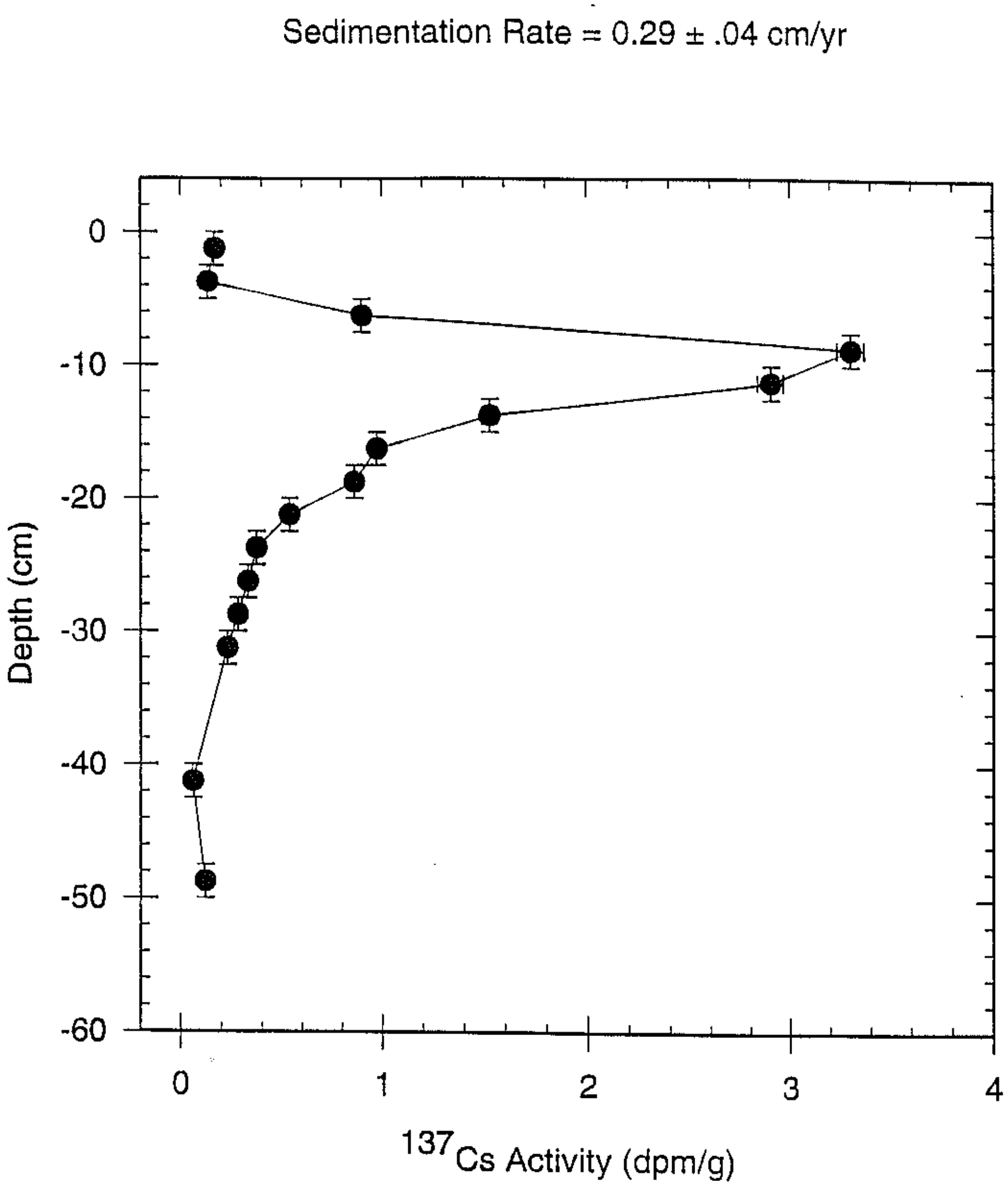


Figure 6.26 Depth profile of ^{137}Cs in Wolfe Runne salt marsh core. Horizontal bars represent counting errors; absence of those indicates that counting errors are smaller than symbol size.

Sedimentation Rate = 0.29 ± .04 cm/yr

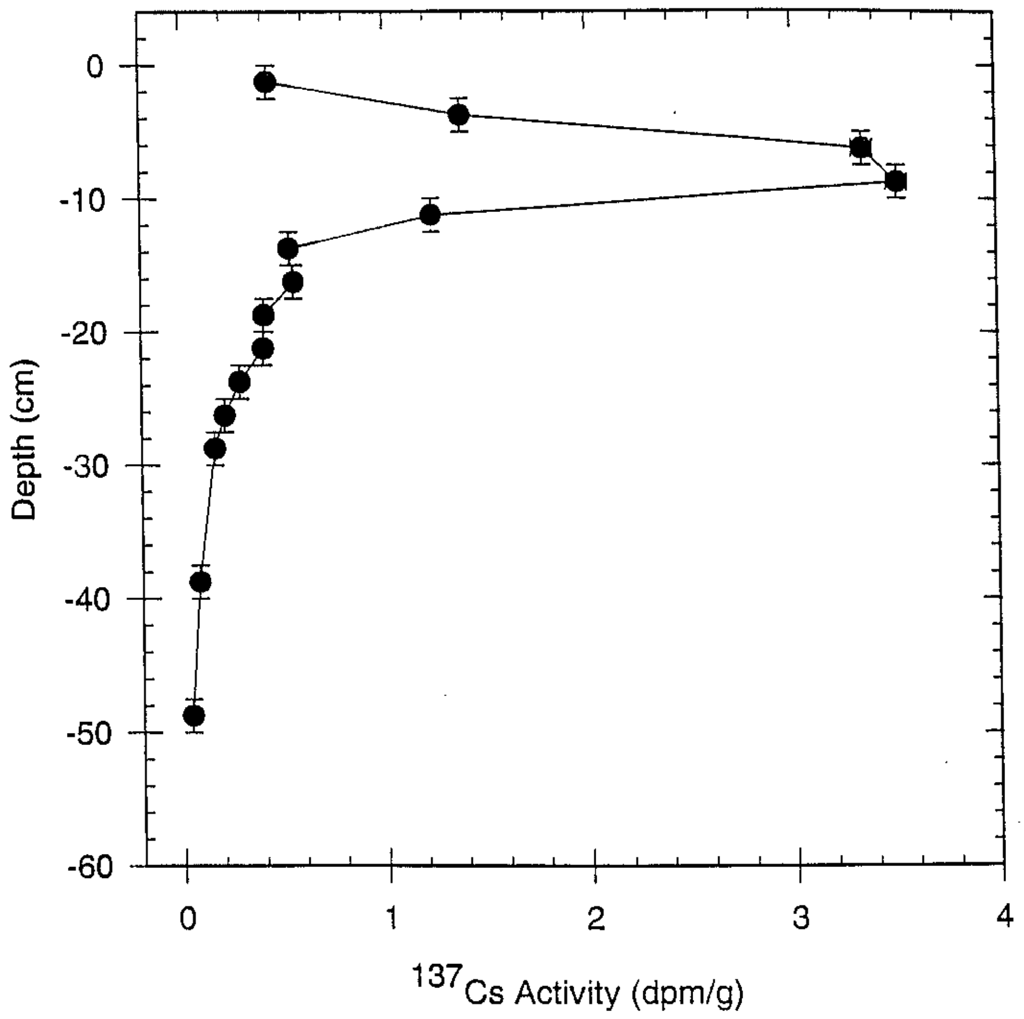


Figure 6.27 Depth profile of ^{137}Cs in Delaware Wildlands salt marsh core. Horizontal bars represent counting errors; absence of those indicates that counting errors are smaller than symbol size.

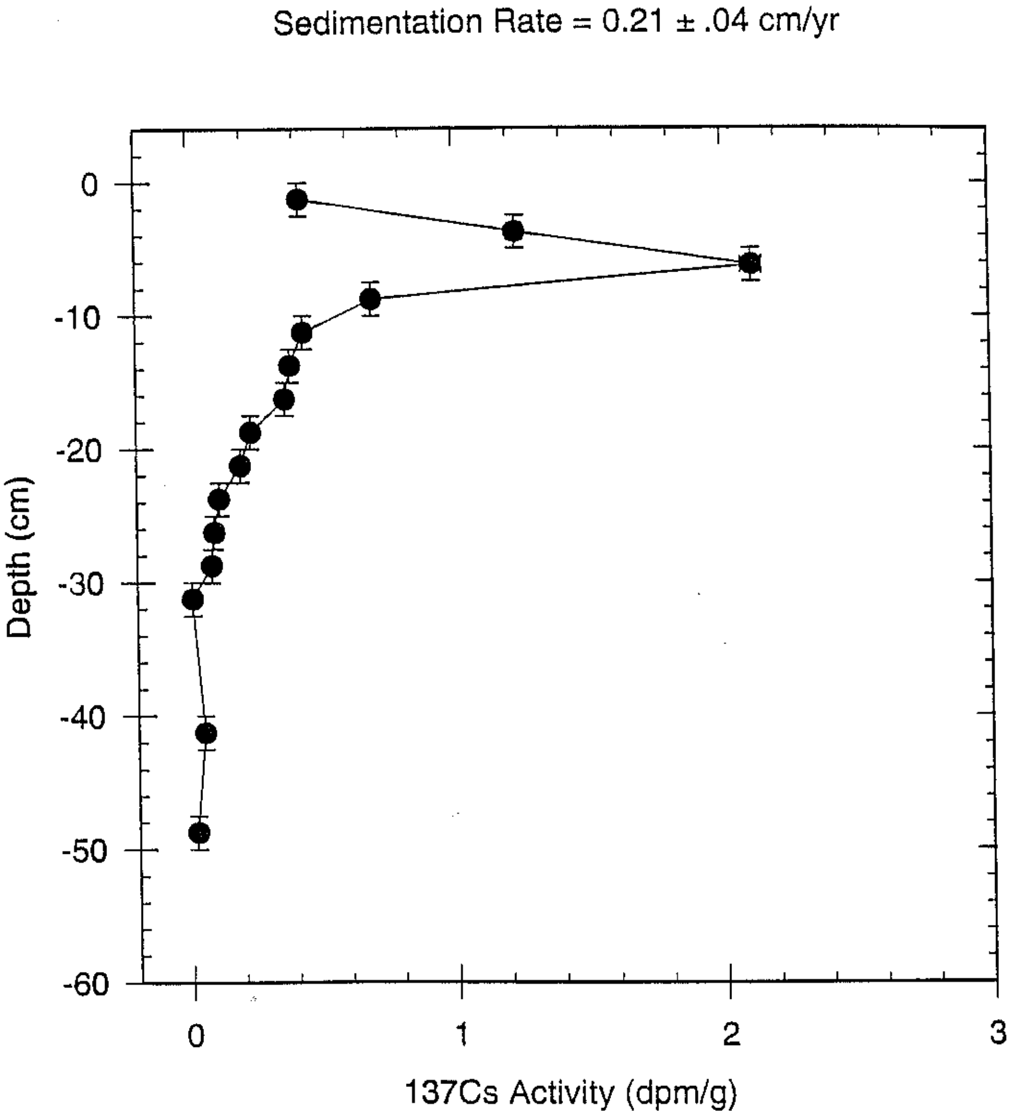


Figure 6.28 Depth profile of ^{137}Cs in Pot Nets North salt marsh core. Horizontal bars represent counting errors; absence of those indicates that counting errors are smaller than symbol size.

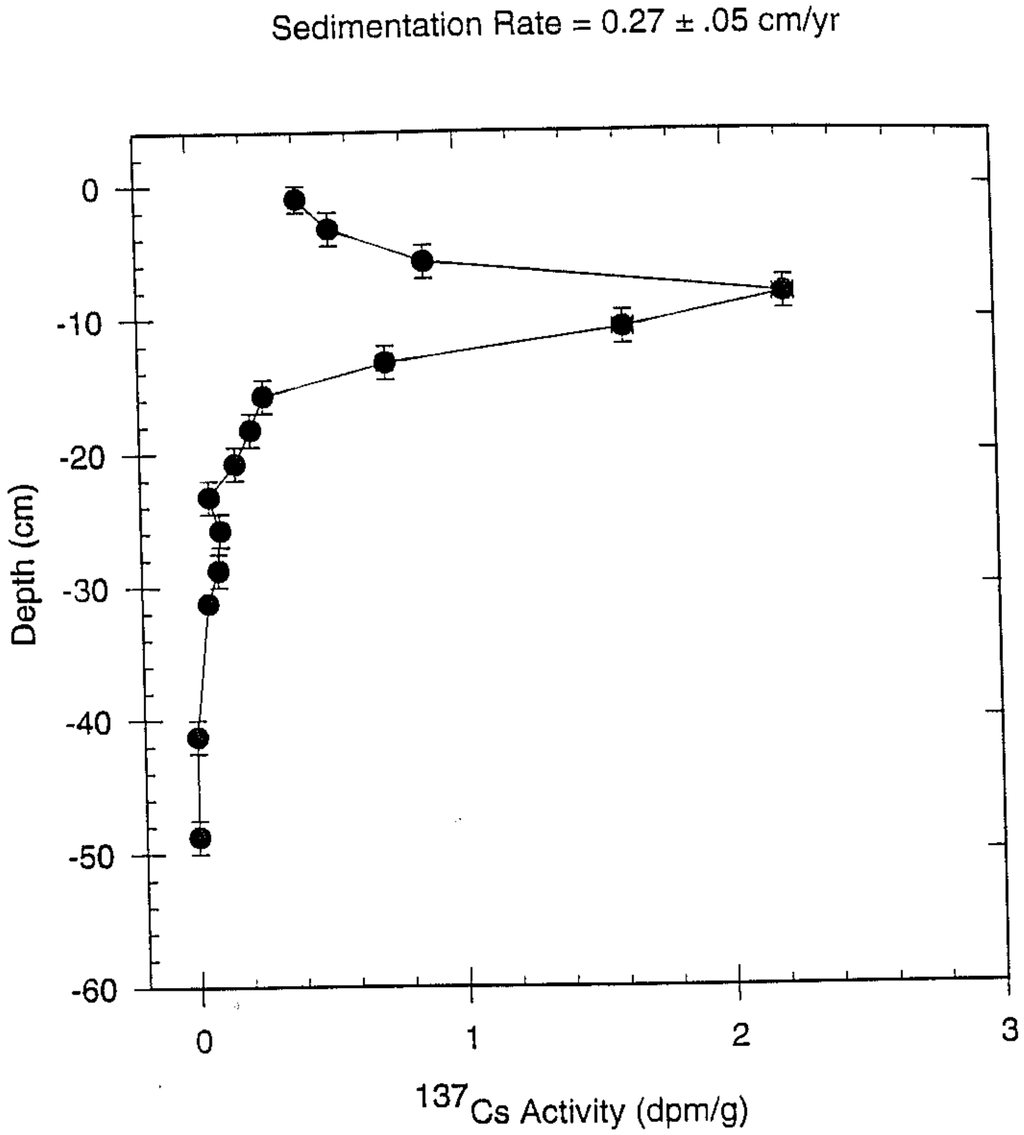


Figure 6.29 Depth profile of ^{137}Cs in Boat House Cove salt marsh core. Horizontal bars represent counting errors; absence of those indicates that counting errors are smaller than symbol size.

Vertical accumulation rates of sediments in the five cores ranged from 0.21 ± .04 cm/yr to 0.38 ± .05 cm/yr (Table 6.4), with an overall mean rate of 0.29 ± 0.03 cm/yr (standard deviation: 0.06; 95% CI = 0.08).

Table 6.4 Vertical accretion rate of sediments based on ^{137}Cs analyses.

Site	^{137}Cs accretion rate (cm/yr)
Boat House Cove	0.27 ± .05
Delaware Wildlands	0.29 ± .04
Pot Nets North	0.21 ± .04
Oyster Neck Road	0.38 ± .05
Wolfe Runne	0.29 ± .04

Discussion. The ^{137}Cs accretion rates suggest that for the last 30 years vertical accretion in salt marshes in Delaware has been lower than the present rate of sea-level rise (0.31 cm/yr). This indicates that at the *S. alterniflora*/*S. patens* boundary, the salt marsh may be in a transitional 'accretionary deficit stage' (phrase coined by Stevenson *et al.*, 1987), in which both organic and mineral sediment input are not sufficient to 'keep pace' with rising sea level. As local relative sea-level rises or tidal range increases, the marsh may either become inundated (i.e., drown), or there may be a natural feedback mechanism by which organic and/or mineral sedimentation increases to compensate for the sea level rise. The transgression of low marsh environments over high marsh environments indicates that while high marsh environments cannot 'keep pace' with the present rate of rising water level, the low marsh is able to maintain itself. Perhaps there is a time lag between inundation of the wetland surface, waterlogging of wetland sediments, and the ability of the salt marsh to maintain or adjust vertical accretion rates. However, it should be noted that the predicted mean rate of sea-level rise for the region of 0.35-0.82 cm/yr (Titus and Narayanan, 1995) may be so much greater than natural accretion rates that it may not be possible for the marsh to maintain its elevation.

The maximum ^{137}Cs accretion rate measured (0.38 cm/yr at the Oyster Neck Road site, Delaware Bay) was greater than the mean rate of sea-level rise, which suggests that this salt marsh can accrete at rates at least as great as predicted sea-level rise rates. It is difficult to ascertain why one salt marsh would have a higher 30-year rate of vertical accretion than others.

Results and Discussion: Pollen Analyses

Results of pollen analyses provided by Dr. Grace Brush are shown in Figures 6.30 to 6.34. The agricultural horizon (based on oak:ragweed ratio) was identified in all five cores; average sedimentation rates calculated from the agricultural horizon range from 0.12 to 0.30 cm/yr (Table 6.5). Based on pollen analyses, the mean vertical accretion rate is 0.20 ± 0.03 cm/yr (standard deviation: 0.07; 95% CI = 0.09).

PIZZUTO-CAREY DELAWARE CORES
CORE ONR

Depth	Oak	Ragweed	Other	Total	Oak:rag
8	70	65	145	280	1.2
20	30	110	60	200	0.3
34	45	60	155	260	0.7
46	60	30	135	225	2
60	120	30	225	375	4
72	50	5	215	270	10
86	110	30	290	430	3.7
99	70	10	115	195	7
112	35	35	290	360	1
125	125	25	242	392	5
138	0	0	0	0	
154	0	0	0	0	

Total post-European pollen
965
Average pollen per level
241
Agricultural horizon 1750
Ave. post-Eur sed. rate 0.24 cm yr-1

Sed rates (cm yr-1)
0.28
0.2
0.26
0.22

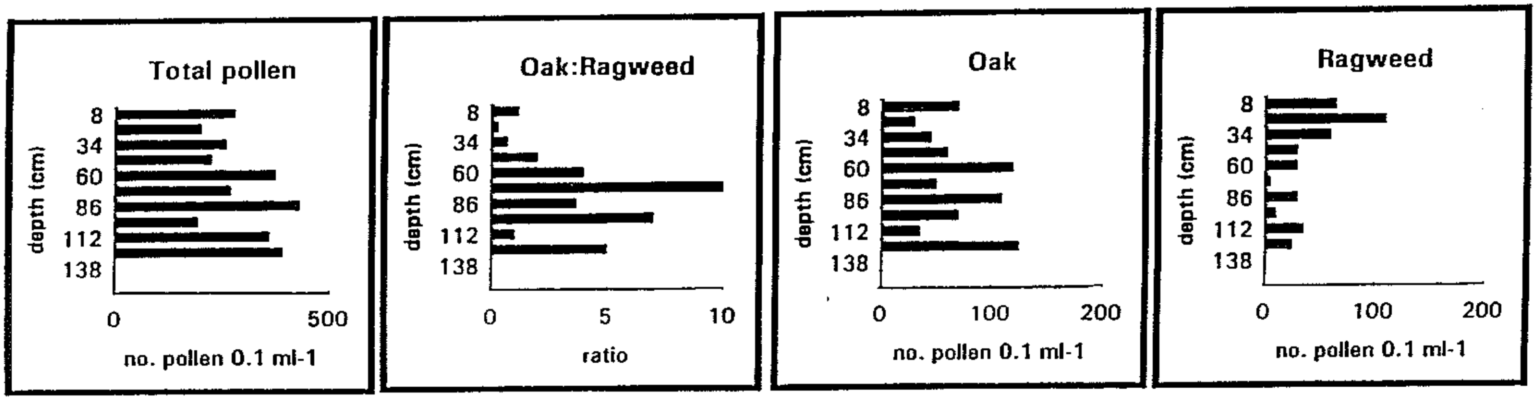

Total pollen
Oak:Ragweed
Oak
Ragweed

Figure 6.30 Oak:ragweed pollen data: Oyster Neck Road salt marsh core (data analyzed and provided by G. Brush).

PIZZUTO-CAREY DELAWARE CORES
CORE PHRS

Depth	Oak	Ragweed	Pine	Other	Total	oak:rag
5	5	47	38	48	138	0.1
20	11	72	71	105	259	0.1
35	21	60	66	33	180	0.3
50	26	44	32	84	186	0.6
64	15	5	28	66	114	3
80	31	2	28	101	162	15
95	9	17	41	105	172	0.6
105	24	14	13	89	140	1.7
120	5	3	2	36	46	1.7
135	1	0	0	2	3	
150	0	0	0	0	0	
165	0	0	0	0	0	

Total post-European pollen
763
Average pollen at each level
191
Agricultural horizon : 1750
Average sedimentation rate: 0.2 cm yr-1.

Sed rates (cm yr-1)
0.14
0.27
0.19
0.19

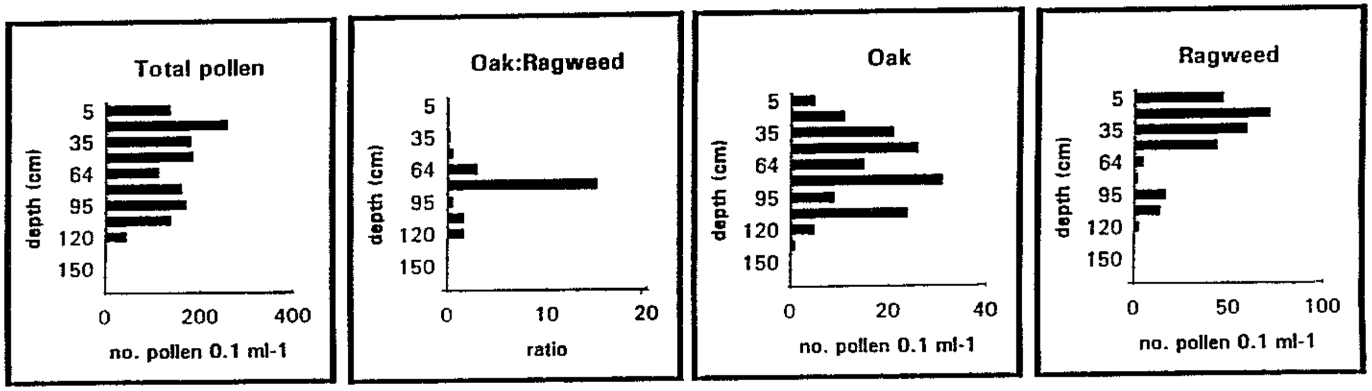


Figure 6.31 Oak:ragweed pollen data: Prime Hook Radio Station salt marsh core (data analyzed and provided by G. Brush).

PIZZUTO-CAREY DELAWARE CORES
CORE WL

Depth	Oak	Ragweed	Pine	Other	Total	Oak:Rag	Sed rates (cm yr-1)
5	12	23	75	39	149	0.5	0.26
15	19	86	62	122	289	0.2	0.5
25	7	16	62	52	137	0.4	0.24
35	21	11	35	76	143	1.9	0.25
45	28	26	65	123	242	1.1	0.42
25	11	47	42	121	221	0.2	0.39
60	10	13	34	64	121	0.8	0.21
73	8	13	3	40	64	0.6	0.11
80	0	0	0	0	0		
85	2	1	4	11	18	2	
100	0	0	0	0	0		
115	0	0	0	0	0		
130	19	21	20	117	177	0.9	

Total pre-European pollen
1366

Average pollen per level
171

Agricultural horizon 1750

Average sedimentation rate: 0.3 cm yr-1

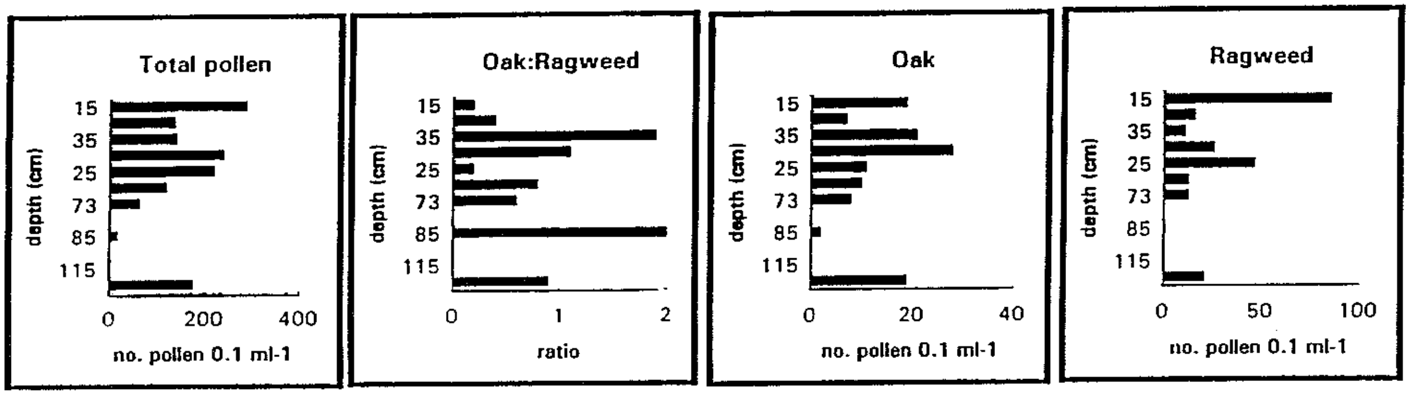


Figure 6.32 Oak:ragweed pollen data: Delaware Wildlands salt marsh core (data analyzed and provided by G. Brush).

PIZZUTO-CAREY DELAWARE CORES
CORE PNN

Depth	Oak	Ragweed	Pine	Other	Total	oak:rag	Total post-European pollen	Sed rates (cm yr-1)
3	14	14	41	63	132	1	619	0.14
14	31	26	54	53	164	1.2	Average pollen at each level	0.17
26	26	26	54	81	187	1	155	0.19
39	31	10	29	66	136	3.1	Agricultural horizon 1750	0.14
51	29	4	24	47	104	7.2	Average sedimentation rate: 0.16 cm yr-1	
63	53	4	47	100	204	13.2		
74	6	2	3	45	56	3		
85	25	2	23	184	234	12.5	alot of Osmunda and some Dryopteris	
96	20	3	15	168	206	6.7	alot of Osmunda and charcoal ?	
106	23	4	21	151	199	5.7	alot of Osmunda	
117	33	0	23	111	167			
128	17	0	17	72	106			

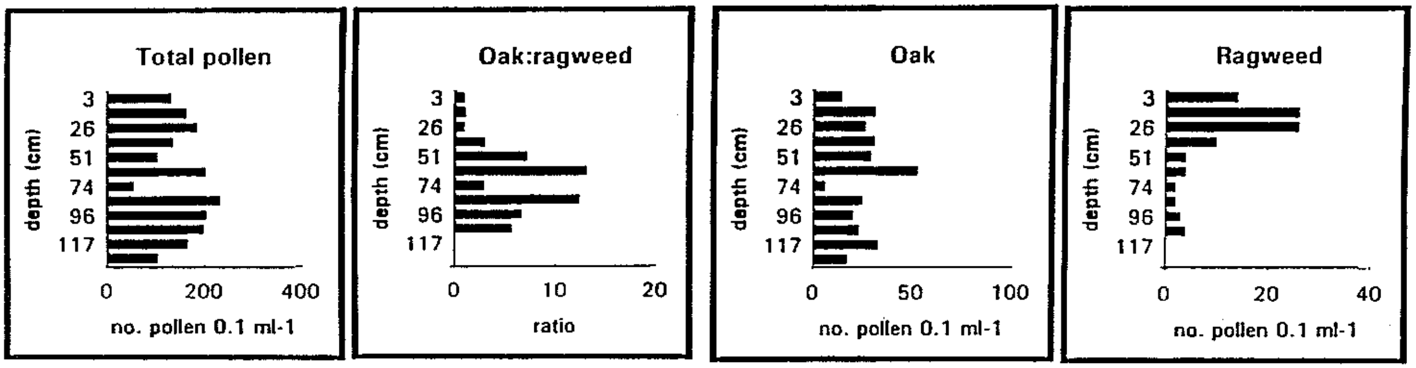


Figure 6.33 Oak:ragweed pollen data: Pot Nets North salt marsh core (data analyzed and provided by G. Brush).

PIZZUTO-CAREY DELAWARE CORES
CORE BHC

Depth	Oak	Ragweed	Pine	Other	Total	Oak;rag	Total post-European pollen	Sed rates (cm yr-1)
4	15	17	36	43	111	0.9	477	0.08
17	7	42	49	46	144	0.2	Average pollen per level: 159	0.11
30	13	36	53	120	222	0.4	Agricultural horizon 1750	0.17
43	18	5	21	148	192	4	Average sedimentation rate 0.12 cm yr-1	
56	0	0	16	29	45			
69	23	2	8	88	121	11		
82	9	0	19	38	66			
108	3	11	14	116	144	0.3	alot of Dryopteris	
120	3	12	5	106	126	0.2	alot of Chenopodium	
134	0	0	0	0	0			
140	0	1	0	1	2			

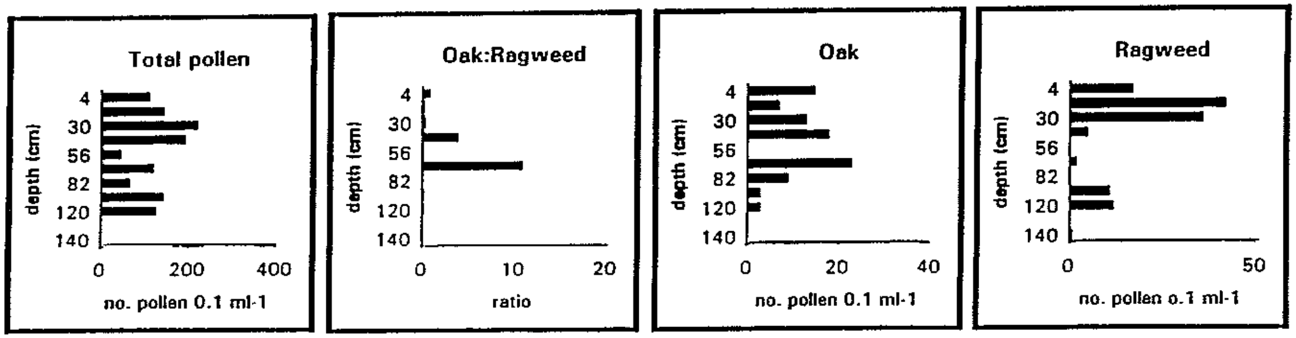


Figure 6.34 Oak:ragweed pollen data: Boat House Cove salt marsh core (data analyzed and provided by G. Brush).

Table 6.5 Vertical accretion rate of sediments based on pollen analyses.

Site	Pollen-based accretion rate (cm/yr)
Boat House Cove	0.12 ±0.05
Delaware Wildlands	0.30 ±0.11
Pot Nets North	0.16 ±0.03
Oyster Neck Road	0.24 ±0.04
Prime Hook Radio Station	0.20 ±0.07

The date 1750 has been chosen as the time that agricultural activity was initiated in the area (G. Brush, personal communication; C. Fithian, personal communication). Initial clearing of the land for agriculture is reflected in the stratigraphic record by changes in the percent of ragweed pollen and in the ratio of oak to ragweed pollen (Brush *et al.*, 1982). The vertical pollen profiles provide a clear record of forest clearance associated with agriculture, as demonstrated by the increase in ragweed pollen concentrations at approximately 25-30 cm in each core. Conversely, abandonment of farms or other changes in land use are recognized by the decrease in ragweed pollen towards the top 10-15 cm in most cores.

It should be noted that vertical accretion rates determined through pollen geochronology are dependent upon the date selected for initiation of agricultural activity in the area. If an earlier or later date is chosen, accretion rates could be either much lower or much higher, respectively. Thus, there are limitations in utilization of this methodology for vertical accretion rate determinations; i.e., the nature of evidence provided by the oak:ragweed agricultural horizon is somewhat arbitrary.

Comparison of Vertical Accretion Rates by Methodology

Estimates of vertical accretion rates at any site vary considerably with time scales (Table 6.6). The mean vertical accretion rates were compared by site and methodology with a two-way ANOVA without replication. While accretion rates vary only slightly between sites, there was a significant difference in accretion rate ($P < 0.05$) between methodologies. Long-term (250-year) vertical accretion rates as determined from pollen analyses are consistently lower than calculated 100-year (^{210}Pb) or short-term 30-year rates (^{137}Cs) (Figure 6.35). If long-term pollen rates are used to assess vertical accretion at the five sample sites, the fringing tidal salt marshes in Delaware appear to be below the threshold for maintaining elevation versus sea-level rise. The changes in subsurface vegetative characteristics with low marsh environments transgressing over high marsh environments suggests that inundation or transgression of high marsh environments is occurring; i.e., sedimentation rates in the high marsh are not adequate to maintain an elevation above rising water levels. Because Delaware's fringing wetlands have not become totally inundated and submerged over the past 250 years, it is plausible that sedimentation rates in *S. alterniflora* subenvironments have 'kept pace' with the rate of relative local sea-level rise. However, although 100-year vertical accretion rates have been adequate to maintain *S. alterniflora* environments,

Table 6.6 Comparison of vertical accretion rate by time scale and methodology.

Site	^{210}Pb Vertical Accretion Rate (cm/yr) (Constant Initial Activity)	^{210}Pb Vertical Accretion Rate (cm/yr) (Constant Flux Method)	^{137}Cs Vertical Accretion Rate (cm/yr)	Pollen Vertical Accretion Rates (cm/yr)
(Time Frame)	(100 year)	(100 year)	(30 year)	(250 year)
Boat House Cove	0.42 ±0.03	0.40 ±0.01	0.27 ±0.05	0.12 ±0.05
Delaware Wildlands	0.52 ±0.09	0.34 ±0.01	0.29 ±0.04	0.30 ±0.11
Oyster Neck Road	0.40 ±0.05	0.30 ±0.01	0.38 ± .05	0.24 ±0.04
Pot Nets North	0.32 ±0.03	0.39 ±0.02	0.21 ±0.04	0.16 ±0.03
Wolfe Runne	0.31 ±0.03	0.37 ±0.01	0.29 ±0.04	NA
Prime Hook	NA	NA	NA	0.20 ±0.07
Mean	**0.39**	**0.34**	**0.29**	**0.20**
Std. Dev.	0.085	0.054	0.061	0.070
Std. Err.	0.038	0.024	0.027	0.031
95% [CI]	0.106	0.067	0.076	0.087

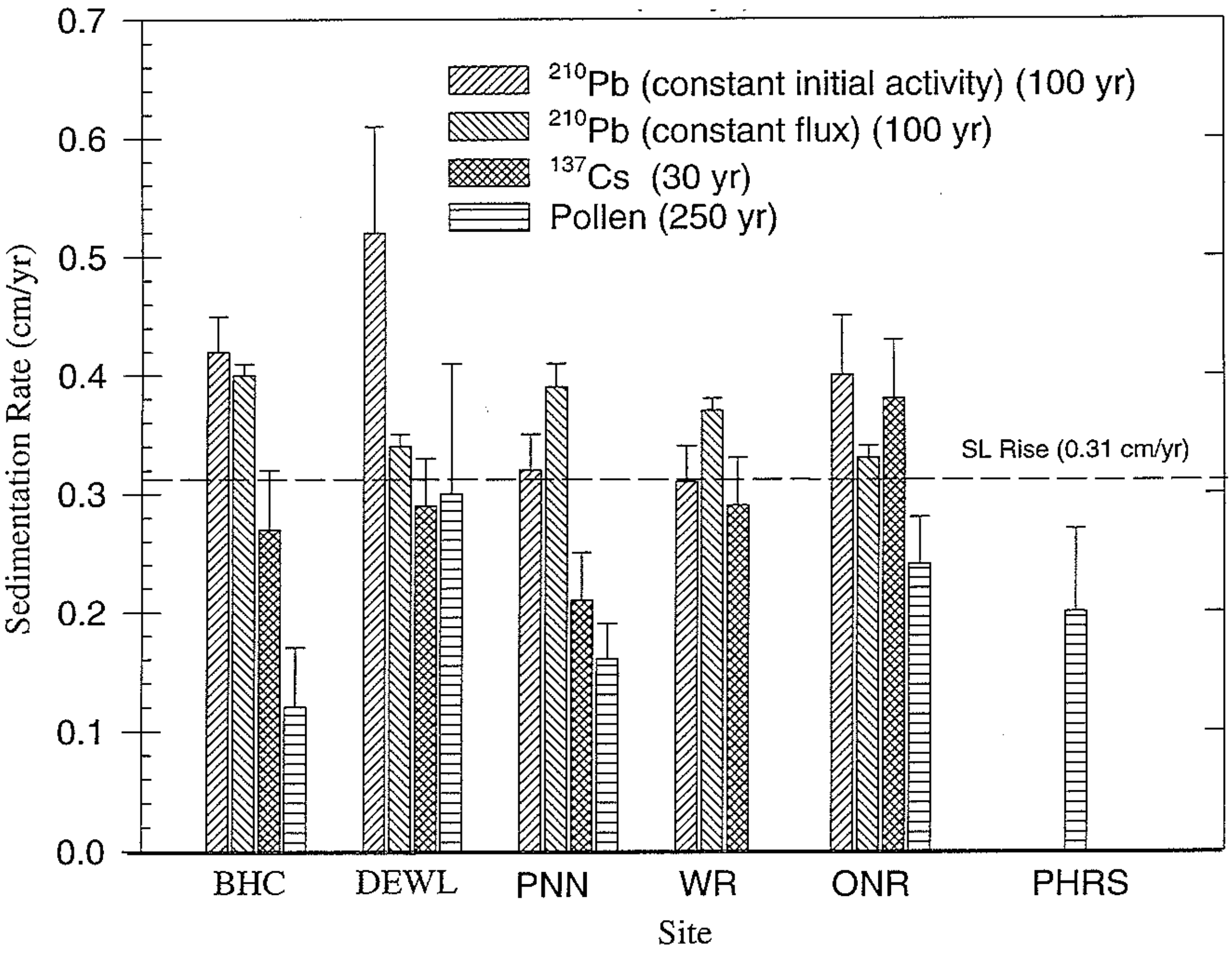


Figure 6.35 Variation in vertical accretion rate by site and methodology.

30-year ^{137}Cs data suggest that present (and possibly future) rates of vertical accretion may not be adequate to maintain even low marsh surface elevations.

The most obvious explanation for this trend is that there has been a higher average sedimentation rate during the past 100 years than during the past 250 years, or that the rate of vertical accretion is adequate to maintain low marsh environments, but not high marsh environments. This suggests that the relationship between sediment supply, tidal range, and local relative sea-level rise must have changed over the last 250 years. If accompanied by adequate increases in sediment supply, natural factors such as increases in rates of relative sea-level rise would result in greater vertical accretion rates in salt marshes. Human induced changes to local relative sea-level rise, i.e., changes in tidal range caused by dredging, ditching, and inlet stabilization, could similarly have significant impacts on vertical accretion rates in salt marshes. This topic was considered previously, and will be examined further in a later chapter.

Research in the Chesapeake Bay has demonstrated similar trends in the range of vertical accretion rates over various time scales (Kearney and Ward, 1986; Kearney *et al.*, 1994). Kearney *et al.* (1994) infer that these differences reflect an adjustment in marsh vertical accretion rates to the acceleration in relative sea-level rise after 1920, as documented in Atlantic Coast tide gauge records. Additionally, DeLaune *et al.* (1983) suggest that the derivation of lower rates for longer time intervals is not surprising as carbon losses and compaction may shrink older marsh sediments. However, as discussed previously, compaction and carbon loss are not considered to be factors in the 50-cm cores obtained for ^{210}Pb and ^{137}Cs analyses, and are not likely contributing factors to vertical accretion rates determined through pollen analysis, where the agricultural horizon was found at a maximum depths ranging from 30 to 73 cm.

Variation in Vertical Accretion Rates

Introduction

The rate of vertical sediment accretion, measurable on a range of time scales, is an important index of the dynamic level of a sedimentary environment (Allen and Rae, 1988). Previous studies have reported that historically marsh systems are often sites of major sediment accumulation, because vertical accretion rates are equal to or greater than sea level rise (Kearney and Ward, 1986; Stevenson *et al.*, 1986). However, data from this study (Table 6.6) indicate that long-term (250-year) and short-term (30- year) vertical accretion rates in Delaware fringing salt marshes are generally lower than the current estimate for the rate of sea-level rise (0.31 cm/yr), while 100-year vertical accretion rates, calculated from ^{210}Pb profiles, generally approximate or exceed the 0.31 cm/yr rate of local relative sea-level rise.

Although there is little direct evidence of the actual mechanism of salt-marsh accretion, two processes generally considered to be of primary importance are silt deposition and organic productivity (McCaffrey and Thomson, 1980). The concept that salt marsh accretion is a function of silt deposition from the water column is likely correct for early stages of marsh development. Historically, authors have considered silt and sand to be crucial in the first stages of marsh build up and establishment (Shaler, 1885; Chapman, 1960). In areas of abundant sediment supply (such as the Mid-Atlantic coastal plain), many salt marsh deposits consist primarily of inorganic particulate detritus with minor amounts of plant remains (Bouma, 1963; Redfield, 1972; McCaffrey and Thomson, 1980). Sediment deposition may also enhance plant growth by adding nutrients to the marsh (DeLaune *et al.*, 1981; Bricker-Urso *et al.*, 1989; Craft *et al.*, 1993). This addition of particulate nutrients to the marsh may act as a positive feedback by stimulating plant growth and organic matter accumulation (Bricker-Urso *et al.*, 1989; Craft *et al.*, 1993). On the other hand, salt marshes are found in areas lacking an abundant silt supply, such as the New England marshes described by McCaffrey and Thomson (1980).

In Delaware, salt marsh sediments at the leading edge of the salt marsh, within 30 meters of the wetlands/uplands boundary, are composed of plant remains (field observations; approximately 40% by weight), along with silt- and clay-sized inorganic particles. The organic matter consists of an extensive and interwoven system of fibrous roots and rhizomes (the underground portion of the plant); the above-ground parts of the plant (i.e., stem and leaves), are rarely preserved in subsurface sediment which emphasizes the critical importance of below-ground biomass and productivity.

Many studies have documented the importance of soil organic matter in vertical accretion, both by enhancing soil volume (DeLaune *et al.*, 1983; Bricker-Urso *et al.*, 1989; Craft *et al.*, 1993) and by providing structural support and pore space (DeLaune & Patrick, 1980; Hatton *et al.*, 1983; Bricker-Urso *et al.*, 1989). Although marshes are able to grow vertically in response to sea-level rise through accumulation of organic matter from emergent vegetation, and via contribution of inorganic sediment to the surface of the marsh; Craft *et al.* (1993) concluded that in irregularly flooded marshes, vertical accretion occurs via in situ production and accumulation of organic matter. In regularly flooded marshes, allochthonous inputs of mineral sediment were found to be responsible for vertical growth of the marsh surface (Craft *et al.*, 1993). They also concluded that ultimately vertical growth of the marshes in response to rising sea-level will be dependent on a concomitant increase in allochthonous mineral sediment inputs (Craft *et al.*, 1993).

Spatial Variation in Salt Marsh Accretion Rates. There does not appear to be a trend in vertical accretion rates with respect to geographic location in this study; i.e., there were no significant differences found in vertical accretion rates among the five study sites. It is interesting that the accretion rate of Delaware's Inland Bays lagoonal marshes appears to have been as great or greater than those of the Delaware Bay marshes, despite a limited sediment supply and a much smaller tidal range. In this study, the spatial similarities in vertical accretion rates are likely due to the similarities in location of the sample sites relative to the wetlands/uplands boundary or the *S. alterniflora*/*S. patens* vegetative boundary. Each sample was obtained at the upper fringe of the marsh; perhaps environmental conditions such as frequency of tidal inundation are similar between sample stations. That is, even along the Delaware Bay coast, with a tide range of 1.2 m, the upper fringe of tidal wetlands is only subjected to tidal inundation and frequency at time periods equal to those found in Inland Bay fringing marshes, where tide range is 0.5 meters or less. Without detailed knowledge of factors such as tidal range, external sediment input, in-place organic matter accumulation, below-ground productivity, and/or hydrography, regional trends in accretion rates are difficult to assess and explain.

In this study, all sites may have been exposed to variable changes in water level which affect the type of vegetation colonizing the wetland, and the vigor or general health of the vegetation. This in turn may impact both above- and below-ground biomass, the major component of organic sedimentation at the study sites. Due to the fact that none of the cores contained discrete layers of mineral sediments, episodic deposition (e.g., storm events, upland runoff) of inorganic sediments has not been a factor in vertical accretion at the wetlands/uplands boundary. Thus, changes in conditions and processes affecting organic sediment supply may ultimately explain differences in average accretion rates.

Temporal Variation in Salt Marsh Accretion Rates. The data clearly suggest that salt marsh accretion rates vary through time; causes of these variations in accretion rates are likely due to short-term changes in local tidal range, sediment supply, storms, vegetation changes, human impacts, and other aperiodic events. Additionally, data obtained in this study on vegetative changes in the subsurface suggest that the nature and type of wetland sub-environments (i.e., high marsh to low marsh) have also changed over the same time period. Thus, it is possible that the observed changes in sedimentation rate over time actually indicate changes in wetland sub-environments over time due to transgression and/or inundation.

Temporal differences in vertical accretion rate relative to the rate of sea-level rise may relate directly to the spatial transition of wetland vegetative sub-environments (e.g., transgression of low marsh over high marsh). The following section includes a brief summary of site-specific changes in vertical accretion rate over time, possible changes in inundation frequency and duration, and associated changes in salt marsh environment (identified from vegetative remains in subsurface) over the same period of time. These interacting factors and relationships suggest that temporal changes in vertical accretion rate may actually be related more directly to local changes in environmental conditions rather than sea-level rise.

Inland Bay Sites

Delaware Wildlands Site. At seen in the ^{210}Pb depth vs. age plot (Figure 6.22), there is a break in slope at 11.75 cm, which corresponds with a change in vegetation from *S. patens/D. spicata* to *S. alterniflora*. The change in wetland sub-environment could have many causes, including change in tidal range, resulting from the effects of the stabilization of Indian River Inlet after 1938. Figure 6.36 is a plot of LOI and change in vegetative species annotated with local environmental changes for the Delaware Wildlands core; approximate dates on the graph are based on radionuclide analyses. ^{137}Cs data are used to approximate the 1964 date, while the ^{210}Pb vertical accretion rates (both constant initial activity and constant flux methods) are used to approximate the date of inlet stabilization (1938). Note that changes in organic content may be associated with changes in local environmental condition. For example,

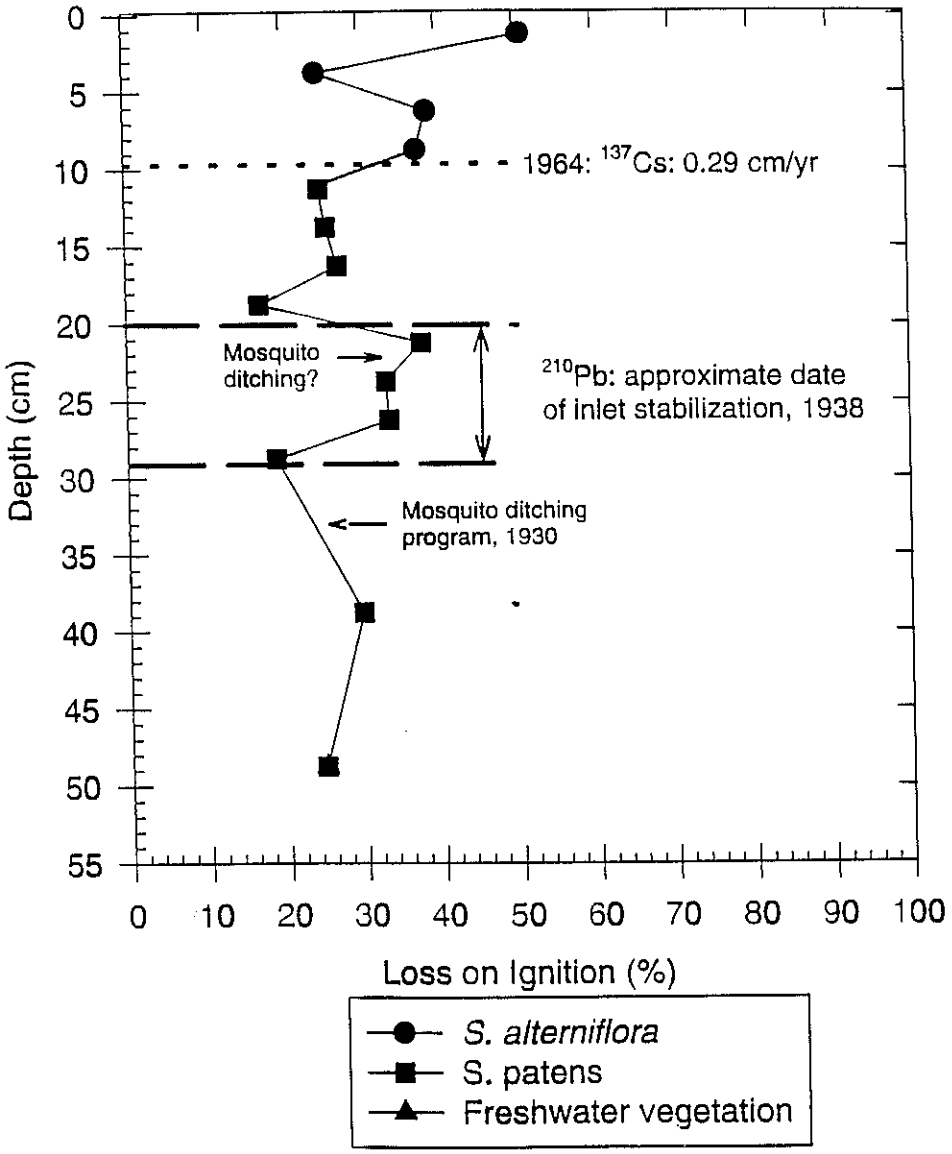


Figure 6.36 Delaware Wildlands site: LOI and vegetation type vs. depth plot, annotated with local environmental changes.

the decrease in organic content for *S. patens* vegetation corresponds to the stabilization of Indian River Inlet; the decrease in LOI values may reflect the deterioration of *S. patens* vegetation resulting from increased flooding and waterlogging of marsh sediments, with the eventual transition from *S. patens* to *S. alterniflora* subenvironment.

Pot Nets North Site. The LOI/vegetative species vs. depth plot for Pot Nets North is shown in Figure 6.37. Approximate location of the 1938 inlet stabilization horizon is based on ^{210}Pb vertical accretion rates. Freshwater vegetation characterized this site prior to stabilization of Indian River Inlet, and *S. patens* remains are found at the base of this core. The transition from *S. patens* vegetation to *Scirpus* (more tolerant of waterlogging and flooding) may indicate increased flooding and/or tidal range at this site. The transition from freshwater marsh to salt marsh resulted from the increasing effects of salt water and tidal range as the ephemeral Indian River Inlet was stabilized in 1938. The increase in LOI content of the *S. alterniflora* in this core may be related to the many attempts to stabilize and maintain Indian River Inlet prior to the final inlet stabilization. Similarly, effects of the mosquito ditching program in 1930, (at approximately 25 cm depth) are noted, as increased flooding and sediment supply are reflected as increases in LOI content. However, as accretion rates decreased over the last 30 years, as suggested by the ^{137}Cs data, organic content of subsurface *S. alterniflora* also began to decrease, perhaps as a result of increased waterlogging stress on the vegetation.

Boat House Cove Site. The Boat House Cove core (Figure 6.38) is characterized solely by *S. alterniflora* vegetation, with only slight variations in LOI

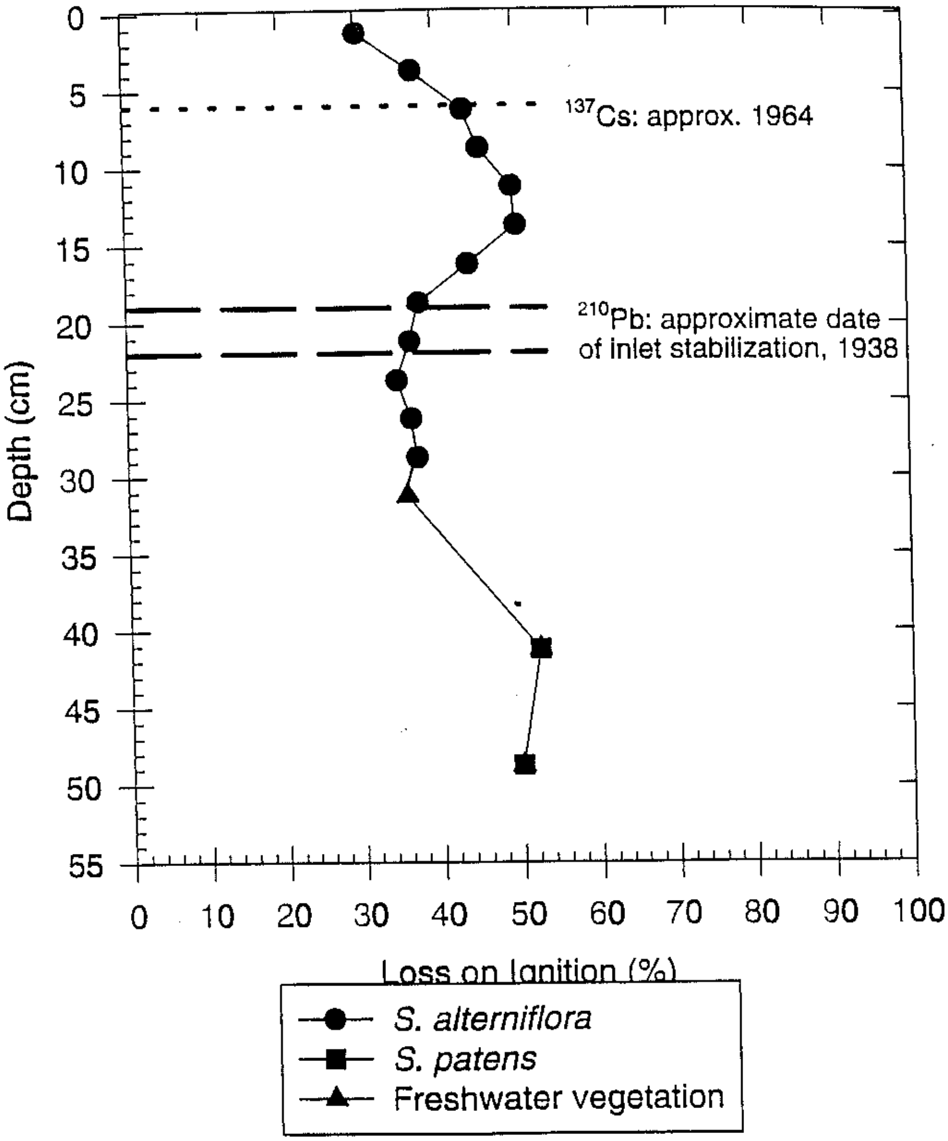


Figure 6.37 Pot Nets North site: LOI and vegetation type vs. depth plot, annotated with local environmental changes.

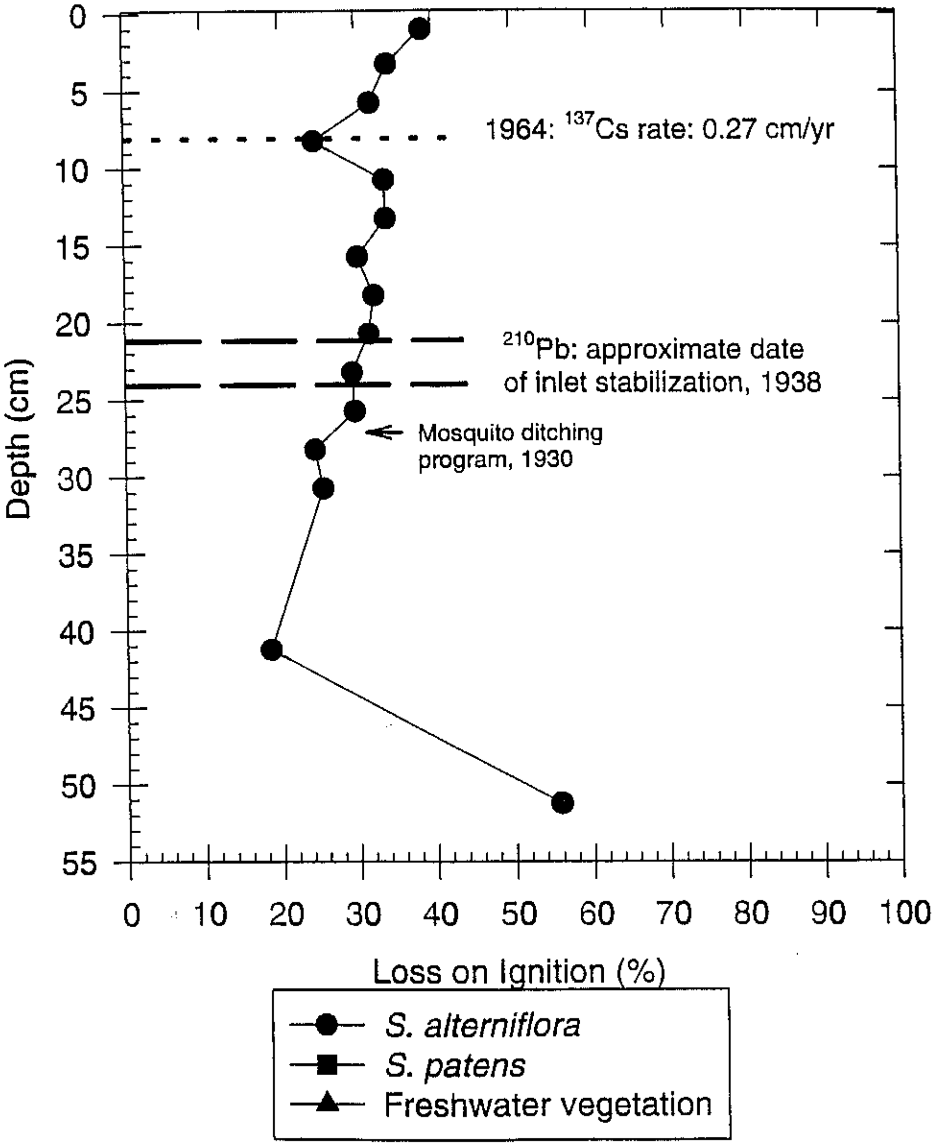


Figure 6.38 Boat House Cove site: LOI and vegetation type vs. depth plot, annotated with local environmental changes.

content. Increases in flooding and sediment supply resulting from inlet stabilization and mosquito ditching may be reflected in the gradual increase in LOI values from 26 cm to 11 cm depth. However, it is interesting to note that at 8-11 cm, when ^{137}Cs data suggests a decrease in vertical accretion rate from 0.40 cm/yr (100 year ^{210}Pb rate) to 0.27 cm/yr, the marsh responded first by a decrease then a gradual increase in LOI value. This may reflect the biological feedback mechanism or response of the marsh to increase and maintain surface elevation through increased below-ground productivity (Naidoo *et al.*, 1993; Hellings, 1990).

Delaware Bay, and Lewes and Rehoboth Canal Sites

Wolfe Runne Site. The Wolfe Runne core is characterized solely by *S. alterniflora* vegetation, with a general trend of increasing LOI content over time (Figure 6.39). Changes in vertical accretion rate and LOI content of subsurface sediments may be associated with changes in tidal prism/tide range and sediment supply caused by construction of the Lewes and Rehoboth Canal and the mosquito ditching program.

Oyster Neck Road Site. A similar pattern can also be observed in the Oyster Neck Road core (Figure 6.40). A change in vegetative sub-environment, from *S. patens* to *S. alterniflora* (high marsh to low marsh) corresponds to the closure of the Broadkill Inlet, the stabilization of Roosevelt Inlet, and the initiation of the mosquito ditching program. The increases in tidal flooding and sediment input resulted in a change in wetland vegetation and an increase in organic content of sediments.

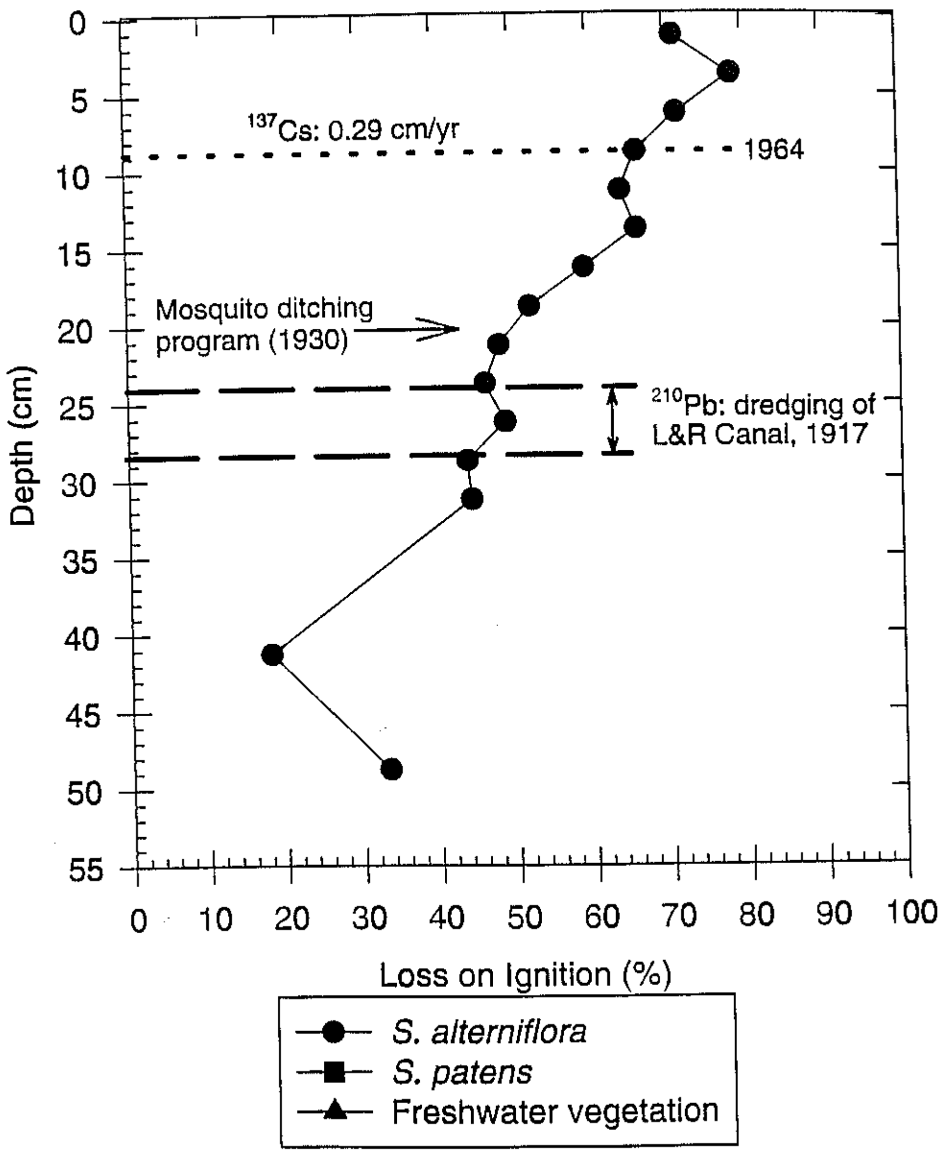


Figure 6.39 Wolfe Runne site: LOI and vegetation type vs. depth plot, annotated with local environmental changes.

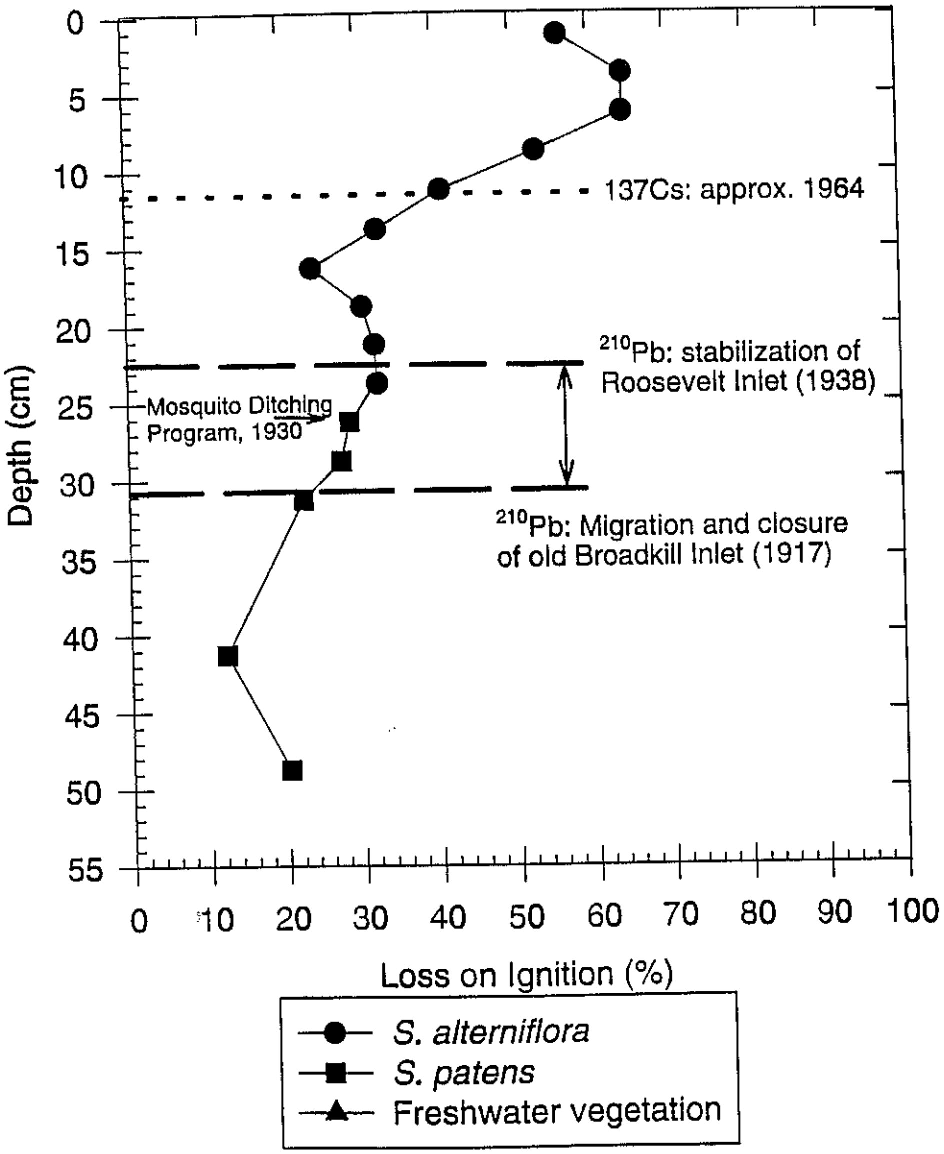


Figure 6.40 Oyster Neck Road site: LOI and vegetation type vs. depth plot, annotated with local environmental changes.

Vertical Accretion and Sea-Level Fluctuations

Variation in vertical accretion rate in a salt marsh can be attributed to many factors such as changes in local sediment supply, tidal range, marsh elevation, vegetation type, and general 'health' of the vegetation, as well as to changes in rates of sea-level rise. Over the longterm (thousands of years), stratigraphic data clearly show that substantial deposits of estuarine silts and rooted peats (including present day salt marshes) have accumulated along the margins of the estuary in response to the rise of sea level during the post-glacial period. The long-term effects of sea-level rise on coastal tidal wetlands are clearly evident in the stratigraphic record. The upward, glacio-eustatic movement of global sea level which has slowed considerably over the last few thousand years (Fairbanks, 1989), resulted in the evolution and development of Delaware's tidal wetland environments.

A direct relationship between vertical accretion rates and rates of sea-level rise may not necessarily be established due to the many possible variations in sediment supplied to the marshes. It is tempting to interpret changes and trends in wetlands accretion rates as indicators of trends in sea level rise; although it is likely that long-term sedimentation rates have been largely determined by the rate of sea-level rise, many human-induced factors have impacted vertical accretion in salt marshes over the last 100 years. Fluctuations over time can represent local changes in vegetation or sediment supply resulting in changing environmental conditions in the marsh. Human-induced changes may be reflected in the decreased sedimentation rate observed in the most recent core section, possibly indicative of reduced supply of sediment from rivers, streams and overland runoff in the last 10 to 20 years.

Vertical accretion rates of 0.52-0.30 cm/yr calculated from 100-year ^{210}Pb data are generally greater than the rate of local sea-level rise (0.31 cm/yr). These higher '100-year' rates may be the result of changing tidal ranges, deepening of channels, dredging operations, inlet stabilization, and/or the mosquito ditching program. According to historical records, approximately 44,000 acres of Delaware's tidal marshes were ditched in the 1930s by the Civilian Conservation Corps for mosquito control (Florschutz, 1959; Swisher, 1982; Stachecki, personal communication). At that time, ditching was considered to be a beneficial practice, intended to improve wetlands for human use, but it may be that ditching enhanced vertical accretion in the marsh through increased flooding and sediment supply.

Summary and Conclusions

Data on salt marsh accretion rates estimated from ^{210}Pb, ^{137}Cs, and pollen data are used to determine the trends of salt marsh accretion in Delaware's fringing coastal marshes over the past 30 to 250 years. Vertical accretion rate estimates provided through these three methodologies are dependent upon the nature of evidence, i.e., the limitations of the methodologies. Accretion rates determined through pollen methodologies are dependent upon the somewhat arbitrary date selected for the agricultural horizon. Similarly, the radionuclide dating methods could be affected by possible erosion, bioturbation, and/or migration of the radionuclide through the sediment column. Although all of the vertical accretion rate data are consistent with sea-level rise, limitations of the methodologies utilized, or the nature of evidence, must be considered when making inferences from these data.

While there do not appear to be differences in vertical accretion rate between study sites, there are differences in the rates of vertical accretion through time. Although the long-term effects of sea-level rise on salt marsh development cannot be ignored, local physical and biological factors may have controlled these trends. Individual short-term (30-year) accretion rates from this study (after the 1960s) are below the rate of local relative sea-level rise (estimated at 0.31 cm/yr).

Accretionary balance, defined as the rate of vertical accretion minus rate of relative sea-level rise, has been positively correlated with mean tide range (Stevenson *et al.*, 1986; Craft *et al.*, 1993). Coastal marshes in microtidal environments typically exhibit an accretionary deficit, and are therefore susceptible to submergence under current rates of sea-level rise. For the marsh sites examined in this study, measured accretionary balances over the past 30 years range from -0.10 to +0.07 cm/yr (mean = -0.02 cm/yr), suggesting that these microtidal marshes would not be capable of maintaining their elevation if the rate of sea-level rise increased. However, accretionary balances over the past 100 years range from -0.01 to +0.41 cm/yr (mean=0.09 cm/yr), suggesting that most of these marshes would maintain elevation with increased rates of sea-level rise. Generally, these results suggest that the fringing coastal salt marshes in Delaware may be susceptible to submergence by an increased rate of sea level rise unless accompanied by an increase in mineral sediment deposition.

The 100-year estimates of vertical accretion rate, higher than the relative sea-level rise rate, suggest that vertical accretion rates may have been enhanced by local factors other than sea level change. Thus, using changes in accretion rates of marshes as evidence of recent variation in sea level should be done cautiously, especially over time periods of 100-200 years. This note of caution may be applicable in many situations, as the age of a marsh, local conditions, the "equilibrium state" of the marsh, and many internal and external forcing factors may impact the rate of salt marsh accretion, especially over short time periods.

Many studies have documented a natural "feedback" system, characteristic of salt marshes that appear to maintain themselves with respect to relative sea-level rise. In this study, at many sample sites, there is a distinct transition from high marsh *S. patens* environments to low marsh *S. alterniflora* environments, suggesting that increases in local relative sea-level rise have indeed resulted in a natural transition to wetland environments with greater organic production and subsequent increases in the organic component of sedimentation. Similarly, Morris *et al.* (1990) document higher productivities in areas that have experienced increased wetland flooding; their data show that areas with the lowest accretion rate had the greatest standing stock and volatile solids level. They found a strong correlation between net annual aboveground production of *S. alterniflora* and mean sea level, attributing higher productivities to increased wetland flooding.

Future research should examine the relationship between organic content of soils and accretion rates; it would be useful to model the effects of sea-level rise on the structure of the marsh, which may be indicative of changes in important processes such as carbon cycling and nutrient fluxes occurring in marsh soils.

Summary of Vertical Accretion Rate by Methodology

^{210}Pb.

1) All of the profiles of radionuclide analyses show a marked excess of ^{210}Pb activity in the upper section of the cores. The amount of excess ^{210}Pb activity decreases with depth. Sedimentation rates were calculated using the depth profiles of excess ^{210}Pb activities.

2) Calculated sedimentation rates based on ^{210}Pb analyses for the 5 sites are:

Site	^{210}Pb accretion rate (cm/yr) (constant initial activity)	^{210}Pb accretion rate (cm/yr) (constant flux)
Boat House Cove	0.42 ± .03	0.41 ± .02
Delaware Wildlands	0.52 ± .09	0.26 ± .01
Pot Nets North	0.32 ± .03	0.35 ± .01
Oyster Neck Road	0.40 ± .05	0.34 ± .02
Wolfe Runne	0.31 ± .03	0.32 ± .02

3) ^{210}Pb rates indicate that over the past 100 years the vertical accretion in Delaware's fringing tidal marshes has kept pace with the rate of submergence.

^{137}Cs.

1) ^{137}Cs plots demonstrate that an horizon of peak activity is present in all five sample sites.

2) Calculated sedimentation rates based on ^{137}Cs analyses for the five sites are:

Site	^{137}Cs accretion rate (cm/yr)
Boat House Cove	0.27 ± .05
Delaware Wildlands	0.29 ± .04
Pot Nets North	0.21 ± .04
Oyster Neck Road	0.38 ± .05
Wolfe Runne	0.29 ± .04

3) Based on ^{137}Cs rates alone, vertical accretion rates at most sites have not kept pace with the rate of submergence.

Pollen.

1) Long term (250 year) sedimentation rates based on pollen (agricultural horizon) analyses at the five study sites range from 0.12 to 0.30 cm/yr.

2) If only long-term pollen rates are considered, most sites appear to be below the threshold of maintaining elevation versus sea-level rise.

Vertical Accretion.

1) Calculated vertical accretion rates are limited by the nature of evidence and methodologies utilized. Rates based on pollen analyses are dependent upon date selected for agricultural settlement in Delaware, and are therefore somewhat arbitrary. Similarly, radionuclide-dependent rates of vertical accretion are limited by quality of radionuclide dates and condition of salt marsh sediment core itself.

2) Vertical accretion rates determined by the three geochronologic methods for all sites demonstrate that different rates may characterize any site depending on the length of time interval considered.

3) Short-term (30-year) ^{137}Cs rates are higher than the more long-term pollen (250-year) rates, yet are consistently lower than the ^{210}Pb (100-year) rates, based on a two-way ANOVA without replication which indicates significant differences between mean accretion rates determined by each method.

4) Short-term (30-year) accretion rates suggest that Delaware's fringing tidal wetlands may be in the process of becoming submerged; i.e. they are not keeping pace with the rate of sea-level rise.

5) Anthropogenic impacts such as ditching, dredging and inlet stabilization may exert significant influence on salt marsh vegetative environmental transitions and associated vertical accretion rates. Thus, using changes in wetland subenvironment (vegetative characteristics) and accretion rates as evidence of variations in sea level should be done cautiously.

TRANSGRESSION OF DELAWARE'S FRINGING TIDAL SALT MARSHES: SURFICIAL MORPHOLOGY, SUBSURFACE STRATIGRAPHY, VERTICAL ACCRETION RATES, AND GEOMETRY OF ADJACENT AND ANTECEDENT SURFACES

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Wendy L. Carey

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Chapter 7

LANDWARD MIGRATION OF FRINGING TIDAL WETLANDS

Introduction

The areal extent of fringing tidal wetlands depends on transgression of the wetland onto adjacent upland surfaces and erosion of the marsh shore. Many previous studies have documented historic shoreline changes along the Inland Bay and Delaware Bay coasts; average annual erosion rates have been calculated for both marsh shorelines and sandy beaches, along with analyses of physical forces which impact these rates. However, the rate of landward migration of fringing tidal salt marshes has not been previously documented. Past patterns and short-term rates of landward transgression of Delaware's fringing tidal wetlands can be determined from historical charts and aerial photographs. This study presents qualitative analyses of historical aerial photographs to evaluate temporal and spatial changes occurring at the wetlands/uplands boundary of fringing salt marshes. In addition, metric maps are used to calculate quantitatively lateral transgression rates at several study sites.

Methods

Trends in historical shoreline change and wetlands/uplands boundary change in the Inland Bays and along the southwestern shore of Delaware Bay were determined from historical maps, aerial photos, and metric maps, along with data previously analyzed by others (Swisher, 1982; French, 1990; Leatherman and Harris, undated manuscript). When combined with ground surveys and observations collected for this study (as described in previous chapters), these methods provide the qualitative and quantitative information required to make inferences regarding changes in wetlands/uplands boundary location, historic distribution of wetland zonation patterns, and past shoreline positions.

Aerial Photographs

Historical aerial photographs were used to obtain qualitative information regarding changes in the areal extent of fringing tidal wetlands at three study site locations: Boat House Cove, Pot Nets North, and Oyster Neck Road. These sites were selected as case study areas to investigate changes salt marsh vegetative zonation patterns.

Aerial photographs have often been used as accurate data sources to monitor changes in coastal tidal salt marshes and shoreline position over selected intervals of time, and provide information on trends in areal extent of tidal wetlands. The usefulness of aerial photos depends on scale, vertical angle, time of day the photo was taken, cloud coverage, and ability of the individual to interpret the photos (Tanner, 1978; Hayden *et al.*, 1979; Swisher, 1982).

Table 7.1 lists the set of low altitude aerial photographs used in this study. The 1968 aerial photograph was used as a base map, from which fixed reference points such as mosquito ditches, houses, and road intersections were used to make qualitative observations of changes in shoreline position and vegetation at the wetlands/uplands boundary. Only qualitative observations were made, due to potential problems with scale variations, transference of shorelines from one year to another, and the crenulate planform of both the marsh shoreline and the wetlands/uplands boundary.

Interpretations of vegetative changes at the wetlands/uplands boundary were based on textural, tone, and shading patterns in the aerial photographs. The major wetland plant species were distinguished by their appearance (grey tone and texture); the wetland/upland boundary formed the transition from predominately salt-tolerant (halophytic) plant species to upland (glycophytic) plant species (Anderson and Wobber, 1973; Hardisky and Klemas, 1983). With assistance from Dr. Michael A. Hardisky, general vegetative zones were established and outlined: terrestrial upland (easily observed as trees); shrub zone (*Iva frutescens* L. and *Baccharis halimifolia* L. shrubs); high marsh zone (*Spartina patens* (Ait.) Muhl./*Distichlis spicata* (L.) Greene); and low marsh zone (*Spartina alterniflora* Loisel.). Although in many cases it was difficult to resolve changes between tidal marsh sub-environments, it was generally obvious where the wetlands/uplands boundary was located.

Table 7.1 Aerial photographs used in this study. Source of photos: U.S. Department of Agriculture, as provided by Delaware Department of Natural Resources and Environmental Control, Division of Soil and Water Conservation and Division of Fish and Wildlife.

Site Location	Year of Photography	Scale
Boat House Cove	1926	1:20,000
	1938	1:20,000
	1954	1:20,000
	1968	1:20,000
	1981	1:12,000
Pot Nets North	1926	1:20,000
	1938	1:20,000
	1954	1:20,000
	1968	1:20,000
	1981	1:12,000
Oyster Neck Road	1926	1:20,000
	1938	1:20,000
	1954	1:20,000
	1968	1:20,000
	1981	1:12,000

Field verification is an integral part of any photo-interpretation study. Photo interpretation is considered to be efficient and accurate for inventorying wetlands (Tiner, 1985), but may have limitations. Wetland/upland and high marsh/low marsh relationships delineated on the aerial photographs, ground-truthing field work was conducted for verification of subtle photo signatures. These ground truthing surveys were conducted to collect information on existing plant communities of various wetland areas and to gain confidence in detecting and classifying changes in the wetland/upland boundary from aerial photography.

Data obtained by Swisher (1982) from historic aerial photographs of Rehoboth Bay were used for shoreline change analyses. Rates of shoreline change for each shoreline cell or segment were calculated, with minimum average error of ±0.1 meters/year (Swisher, 1982).

Metric Maps

The Coastal Mapping Group, Coastal Research Laboratory, University of Maryland, has produced shoreline change maps of Indian River and Rehoboth Bays using the Metric Mapping technique (Leatherman, 1983; Crowell *et al.*, 1991). The historical shoreline change maps were produced at a scale of 1:5,000, and show shoreline and marsh changes between 1944 and 1989. Data sets used in production of the maps include National Ocean Survey (NOS) "T" sheets and three sets of aerial photographs. Two sets of "T" sheets used for this project include 1944 and 1962; for Indian River Bay, the 1845 "T" sheet was not used because of the apparent inaccuracies in the mapping of the marsh/upland boundaries (Leatherman and Harris, undated manuscript). Stable points on the NOS "T" sheets are accurate to within 0.3 mm of their actual location at the scale of the map. The aerial photographs used were corrected for radial and tilt distortions and scale differences.

Metric Mapping Process. Metric mapping is a computer-assisted cartographic technique that is highly accurate and emulates the most precise photogrammetric techniques (Leatherman, 1983; Leatherman and Clow, 1983; Crowell *et al.*, 1991, 1993). Original maps are transformed to State Plane coordinates, data are entered through X-Y coordinate digitization of shoreline data, photographs are corrected for distortion introduced by flight-path irregularities, and accurate maps are plotted. The entire metric mapping technique can be divided into four distinct tasks: 1) data annotation and selection; 2) digitization and data transformation; 3) data adjustment; and 4) map plotting.

Control points and annotation of the mean high water line and marsh/upland interface were selected before the data digitization process. The mean high water line was demarked on aerial photographs using both the tide tables and the tonal differences in beach sand. The wetland/upland boundary was delineated by the contrast between upland forest/field and wetlands grasses *(Phragmites australis* (Cav.) Trin. ex Steud. and *S. patens*).

Control points were annotated based on known geographic coordinates. Maps dated 1929 and earlier had coordinates and triangulation stations updated to the

1927 North American Datum (NAD) to eliminate errors and distortions. This was accomplished using the Least Squares Fitting, Transformation and Interpolation (LEFTI) program, which removes latitude and longitude distortions inherent in earlier cartographic products.

Digitization was accomplished through use of an x-y digitizer. Shorelines were transformed into numerical data and broken down into small segments or arcs which were assembled into continuous streams of shoreline data for map file assembly. Digitized data were transformed from digitizer coordinates to state plane coordinates using the transformation subroutine in the program (Leatherman, 1983; Leatherman and Clow, 1983; Crowell *et al.*, 1991, 1993).

Aerial photographs were transformed after normal corrections for scale and location errors as well as flight-induced errors such as tilt and radial distortion. After all shorelines were transformed to state plane coordinates and assembled into year group files, errors in line segments (e.g., mismatches) were corrected with the TIE program. The TIE program was used to identify line segments which were out of alignment by more than 1 foot per each 1000 feet of scale. Those line segments out of alignment greater than this limit were redigitized with additional control features to eliminate rotational problems. In the final phase, map sheets were generated and assembled. The final step was printing accurate, legible maps with a HP plotter.

Accuracy Assessment. The metric mapping technique is subject to a limited number of error sources introduced by source materials or human factors (Leatherman, 1983). Each map and photograph was subject to resection and transformation programs to assess the accuracy of scale, degree of rotation, and translation; aerial photographs were also checked and corrected for tilt, radial and scale distortions induced by aircraft flight irregularities. The resection and transformation programs assess the magnitude of error of each of the digitized control points on the media; normal maximum acceptable error is 1 foot of error for every 1,000 feet of scale. For example, 10 feet of error would be the maximum acceptable error for a 1:10,000 scale map. For this metric mapping project, the majority of detectable error was less than 0.5 feet per thousand feet of scale (5 feet for a 1:10,0000 scale map) (Leatherman and Harris, undated manuscript).

In addition to media errors, human factors are the other primary source of error in the metric mapping technique, especially map and photograph interpretation and digitizing errors. Photographic interpretation errors were minimized by using magnifying devices to determine the mean high water line, the marsh/upland interface, and to identify control features.

Erosion Rates: Results and Discussion

Aerial Photographs: Historic Shoreline Changes

Shoreline data analyzed and interpreted by Swisher (1982) for Rehoboth Bay were used to obtain shoreline change rates from 1938 to 1981. Historical shoreline changes in Rehoboth Bay have been extremely variable, ranging from -9.5 meters/year due to storms, to +13.8 meters/year due to filling (Swisher, 1982). Mean erosion rates in most areas have been approximately 0.6 meter/year or less (Swisher, 1982).

A summary of the historic shoreline changes for Rehoboth Bay compiled from the aerial photographs, as determined by Swisher (1982) is shown in Table 7.2 and Figure 7.1. These data demonstrate that changes in shore erosion in time and space have been extremely variable. The long term geologic processes that have formed Rehoboth Bay and the short-term shoreline processes and human alteration that have modified it are all evident in the recent historical changes around the Bay (Swisher, 1982). Natural and human-induced causes of shoreline change as summarized by Swisher (1982) include: long-term sea-level rise, storms (waves, winds, tides), rapid fluctuations in water level, inlet stabilization, dredging, filling, and placement of shoreline structures.

Metric Maps: Erosion Rates

Delaware Bay shore erosion rates based on metric map data have been calculated and summarized by French (1990), and are presented in Figures 7.2 and 7.3 and in Table 7.3. These shoreline-change maps demonstrate an overall transgressive coastline with few areas of significant accretion. Average total net change for the study area between the years 1882-1977 was -128 meters, or approximately -1.3 m/yr (±0.03 m/yr), which is considerable higher than the Atlantic coast average of -0.79 m/yr (National Research Council, 1987; French, 1990). Erosion rates for Port Mahon and Bombay Hook in northern Delaware Bay were consistently high during the 135-year time period (French, 1990). Rates of -9 m/yr are common, with rates up to -15 m/yr occurring at some locations; long-term (1882-1977) average erosion is -6.25 m/yr (+/-

Table 7.2 Statistical values (m/y) for historical shoreline changes. Note that in most cases the standard deviation is larger than the mean indicating a high variation in the data (from Swisher, 1982).

Shoreline Orientation	Time Interval		
	1938-54	1954-68	1969-81
Northern			
Mean	-0.3	+1.1	-0.2
Std. Dev.	(0.9)	(3.2)	(0.5)
Western			
Mean	-0.4	-0.6	-0.5
Std. Dev.	(0.2)	(0.4)	(2.3)
Southern			
Mean	-0.2	-0.2	-0.1
Std. Dev.	(0.6)	(1.1)	(1.5)

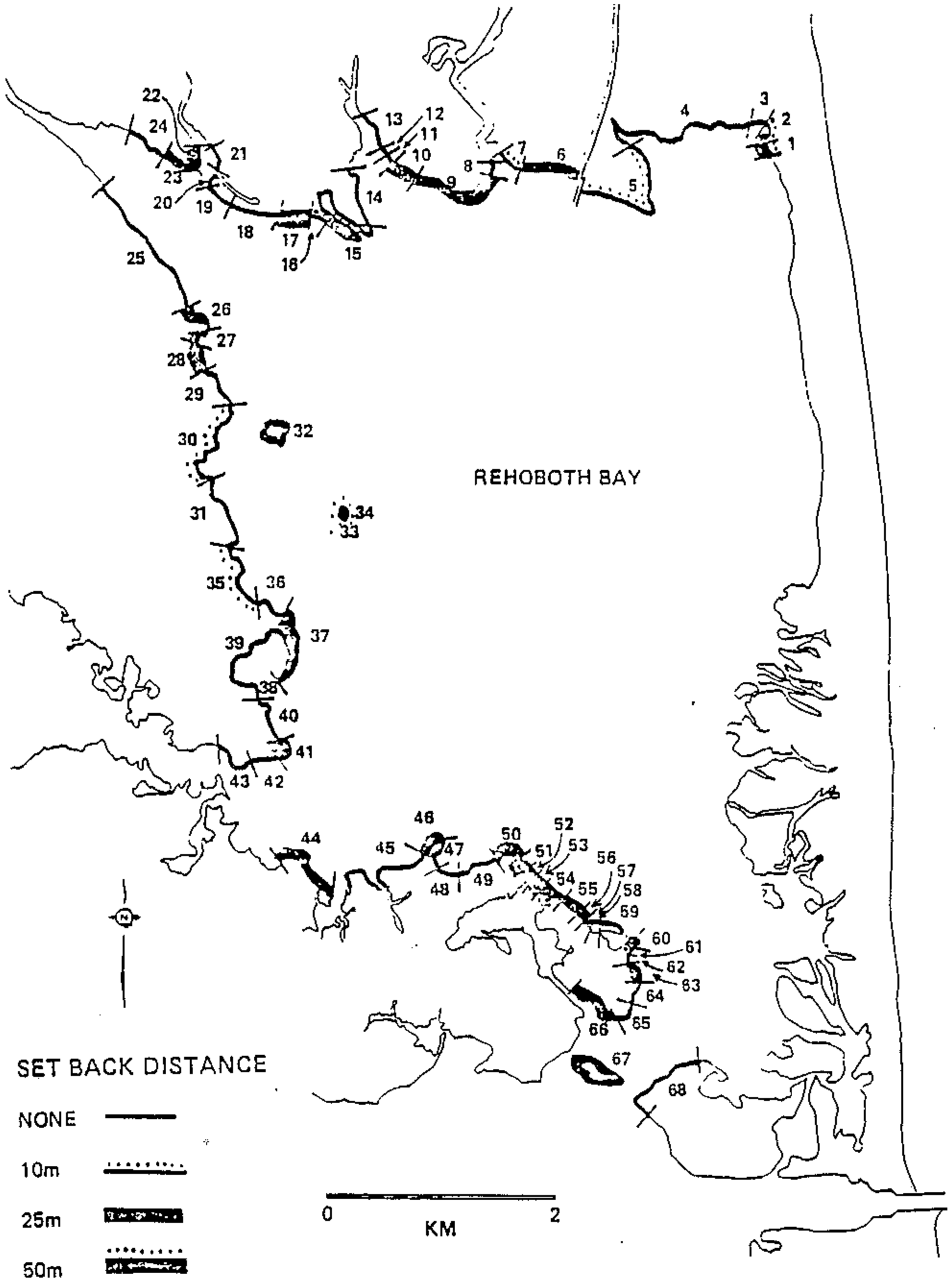


Figure 7.1 Erosion prone areas of Rehoboth Bay. Cells selected to qualify for a certain set back distance if the historical erosion rate for that cell is greater than or equal to the critical erosion rate at least half the time. The critical erosion rate is less than 0.2 m/yr for no set back distance; 0.2 m/yr for a 10 meter set back distance; 0.5 m/yr for a 25 meter set back distance; and 1 m/yr for a 50 meter set back distance (from Swisher, 1982).

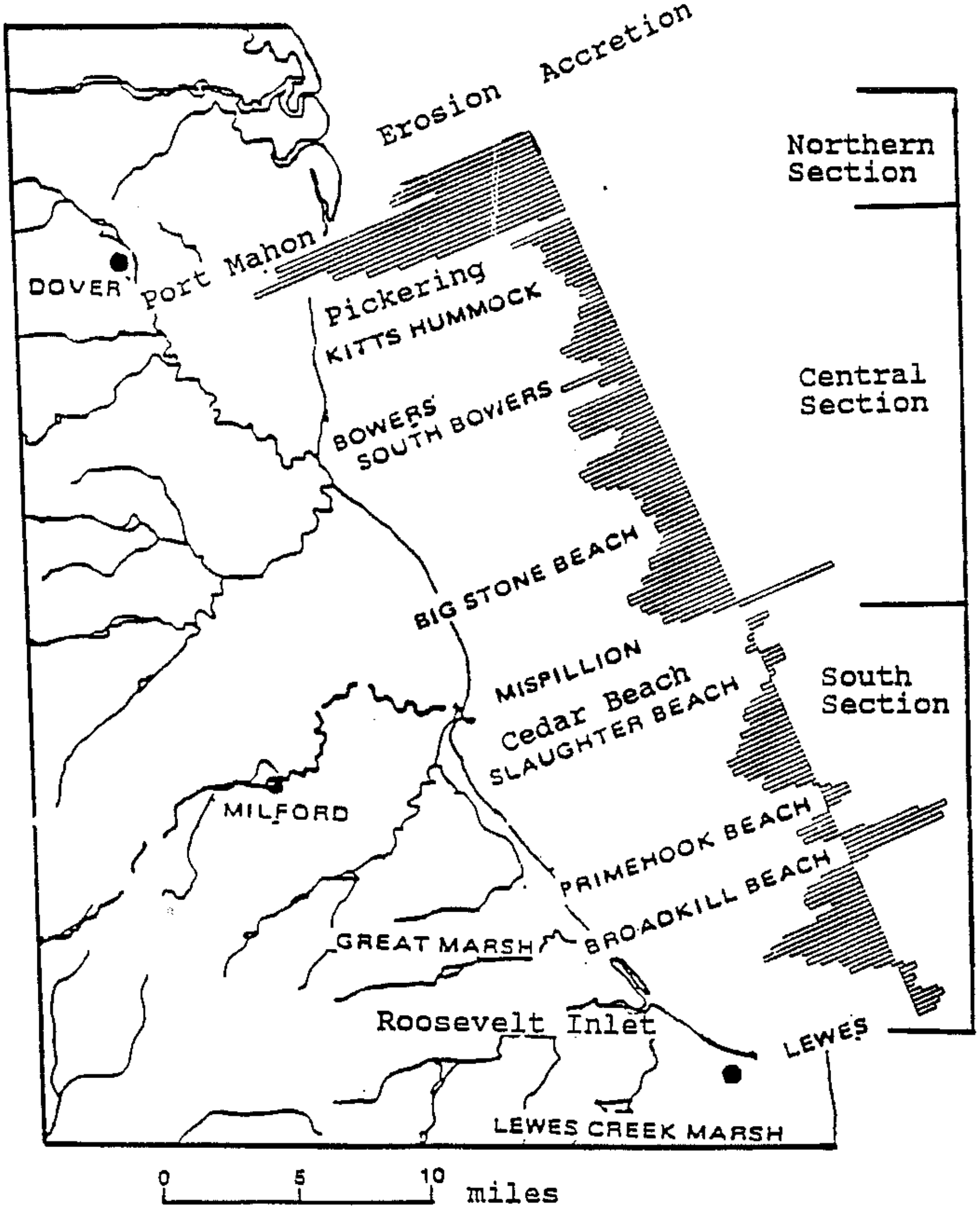


Figure 7.2 General pattern of shoreline changes along the western Delaware Bay coast: 1882-1977 (from French, 1990).

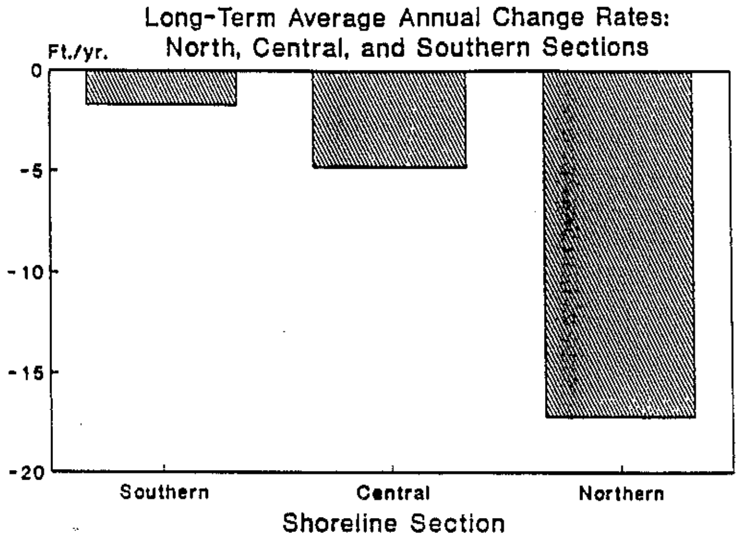


Figure 7.3 Spatial variations in shoreline change rates along Delaware Bay for each of the three sections of the study area. Graph depicts differences in long-term (1882-1977) change rates between the three sections of the study area (from French, 1990).

Table 7.3 Average rates of change for each of the three sections of Delaware Bay for each of the data time spans (from French, 1990). * N=North; C=Central; S=South. N/A=Data not available.

	Sequential Data Sets						Total Spans	
*	1842-1882g	1882g-1910	1910-1943g	1943g-1954	1954-1969g	1969g-1977	1842-1977	1882g-1977
N	-24.2	-19.3	-11.8	-12.3	-18.9	-25.8	-18.4	-17.2
C	-4.4	-6.1	-4.6	+2.6	-7.2	-2.4	-4.7	-4.8
S	N/A	N/A	N/A	+1.6	-4.7	+4.4	N/A	-1.7
± Range of Error:								
	±0.5	±0.8	±0.6	±1.7	±1.2	±2.3	±0.1	±0.2

0.03 m/yr) in the area of Port Mahon. In the southern section of Delaware Bay, where sites for this study are located, periods of pronounced erosion alternate with stability/accretion, suggesting that changes of the shoreline are largely storm driven (French, 1990).

Historical Shoreline Changes. For this study, the erosion rates calculated by Swisher (1982) for Rehoboth Bay are used as generalized shore erosion rates for the Inland Bays fringing marsh systems. These results show that over the past 45 years (1938-1981), a pattern of overall erosion characterizes most of the wetland shores along Rehoboth Bay, with mean erosion rates averaging -0.6 m/yr or less.

As reported by French (1990), along the Delaware Bay shoreline during the time period 1954-1969, overall erosion rate for the southern section is -1.4 m/yr (±0.4 m), while the long-term (1882-1977) overall erosion rate for the southern section is -0.52 m/yr (±0.06 m) (French, 1990). Over the same time span, erosion rates are much higher in northern and central sections of Delaware Bay, averaging -5.24 m/yr (± 0.06 m) in the northern section and -1.46 m/yr (±0.06 m) in the central section.

Processes of Marsh Shore Erosion. Swisher (1982) concluded that marsh shores erode differently than sandy beaches. Marsh erosion is a combined process of undercutting and slumping caused by reflected storm waves and variations in water level. Under both normal and storm conditions, marsh shorelines had scarped profiles, and erosion processes were concentrated in a narrow zone around the scarp (Swisher, 1982). Marsh shorelines do not have a short-term repair mechanism for erosion, as do sandy beaches. Swisher (1982) concluded that results from both a long-term aerial photo study and a short-term monitoring study indicate that marsh shores erode faster than beach shores.

Deposition and erosion in a marsh are related to tidal hydraulics, storms, sediment supply, plant colonization, biological trapping, and marsh growth (Letzsch and Frey, 1980; Swisher, 1982; Stumpf, 1983; Frey and Basan, 1985). Erosion in a marsh is primarily controlled by storms, shear stress, rotational slides, and animal burrowing (Pferd, 1970; Frey and Basan, 1971; Hardaway, 1980; Letzsch and Frey, 1980; Swisher, 1982). An erosional marsh shoreline is characterized by a scarped profile (Hardaway, 1980) and slump blocks are common (Frey and Basan, 1971). The dense sediment-root matrix along the marsh shoreline is extremely resistant to wave attack, but may be frequently undercut, facilitating slumping (Kraft, 1971; Hardaway, 1980). However, marsh vegetation simultaneously dampens energy of incoming waves and reduces current velocity and bottom shear stress, thereby mitigating erosion and causing deposition (Dean, 1978; Swisher, 1982).

Marsh shore erosion is consistent over time; erosion of marsh shores is not characterized by extreme events such as storms, but rather retreat at a relatively regular rate. This is likely due to the composition and structure of the marsh's cohesive substrate and extensive rhizomatal network. Another reason that extreme erosion rates are not found to correlate with storm events (as is often the case along sandy shorelines) is that if the storm surge is sufficiently high, the marsh may be completely inundated. If this occurs, the wave energy will no longer be expended on the marsh edge, but will instead be dissipated among the culms of the marsh grass.

Sea-level rise may be the underlying cause for much of the marsh shore erosion (French, 1990). Present rates of sea-level rise exceed 0.30 meters per century (Lyles *et al.*, 1988), double the rate of the past 2,000 years (0.12 m/century) (Belknap, 1975), with the most obvious result manifested as increased erosion at the water's edge (National Research Council, 1987). Other contributing factors may include low rates of sediment supplied to the marsh edge, along with the activity of clams and other bivalves, as well as crabs and other crustaceans which loosen the normally cohesive substrate allowing normal wave activity to erode the edge.

Once a marsh edge is "scarped" and the marsh/water interface is a vertical wall, it is unlikely that there will ever be accretion on the bayward or seaward side of the marsh. This is especially the case when there is little or no sediment transport or sediment supplied to the littoral system. The marsh edge actually becomes a vertical wall like a bulkhead, and once the vertical wall is established, there is increased erosion, and accretion at this steep edge is virtually impossible.

Therefore, except in sandy areas or areas where there is a drop in water level, marsh accretion on the bayward/seaward side would not occur. Along Delaware's Inland Bays and along the southern portion of Delaware Bay, marsh progradation in the seaward direction is <u>not</u> occurring at any of the study sites. Perhaps marsh progradation may be found up tidal creeks or on point bar/meanders where sediment is supplied from steep, eroding upland banks, or perhaps along sandy pocket beaches of the inland bays. However, only erosion and undercutting occurs in areas of established marsh with a scarp configuration at the wetlands/water interface.

Thus, with characteristic marsh shore erosion occurring along both the Delaware Bay and Inland Bay shores, the future areal extent of Delaware's fringing tidal wetlands will depend on the balance between rate of erosion, present width of marsh, and rate of landward migration of the marsh onto adjacent upland surfaces. If erosion on the bayward/seaward edge exceeds encroachment over adjacent uplands, the areal extent of fringing tidal wetlands will diminish over time.

French (1990) calculated that current average erosion rates of -6.25 m/yr are common in the Port Mahon area of Delaware Bay. At this rate, the entire marsh (approximately 3000 m wide) behind Port Mahon will be gone in less than 500 years. If sea-level rise rates triple over the next 100 years (National Research Council, 1987), then the local rate of erosion could reach as much as 30 m/yr, eliminating the marsh in only 100 years (French, 1990). Along the Inland Bays shorelines, where fringing marshes are narrower (tens of meters wide), much marsh area could be eliminated in 100-200 years under present erosion rates of 0.6 m/yr or less. These estimates reflect only marsh-edge erosion.

Marsh-edge erosion is not the only process which results in deterioration, degradation, and loss of wetland areas. Submergence of wetland areas may occur due to interior ponding and inundation (Stevenson *et al.*, 1986; Leatherman *et al.*, 1995). Extensive marsh loss due to inundation, internal ponding, and submergence (Orson *et al.*, 1985; Kearney *et al.*, 1988) may rapidly convert wetlands to shallow open-water areas where erosion rates may accelerate.

Although erosion of the marsh shore is a long term geologic process that has occurred over the past several thousand years, the historic response of the salt marsh has been to migrate landward and upward in space and time. As water levels have risen, low-lying adjacent uplands have been converted to wetlands, as is demonstrated in the stratigraphic record. However, due to anthropogenic factors, this conversion may no longer be a viable process for maintaining the areal extent of fringing tidal wetlands. Human development has typically constrained landward advancement of wetlands through extensive stabilization of shorelines and conversion of coastal woodlands and farms to urban and residential development (Leatherman *et al.*, 1995). Development of waterfront areas has removed buffer zones which would have become wetlands in response to sea-level rise. Only in areas where rates of landward migration of the salt marsh onto adjacent upland surfaces equal or exceed the rate of marsh submergence and erosion will fringing tidal wetlands exist in the future. The process and rate of wetland encroachment onto adjacent uplands has not previously been determined, and is examined in the following section.

Transgression Rates: Results and Discussion

Aerial Photographs: Historic Changes at the Wetlands/Uplands Boundary

The following section discusses trends in recent (70 years) historic changes in vegetation at the wetlands/uplands boundary, as revealed by aerial photographs of several study sites.

Oyster Neck Road. The Oyster Neck Road (ONR) site is located along the western shore of Delaware Bay, at the northeastern end of Oyster Rocks Neck. Figures 7.4 through 7.6 are reproductions of the 1938, 1954, and 1968 aerial photographs of the study area, with the wetlands/uplands boundary outlined in red. The greatest change in the character of the wetlands/uplands boundary area occurred between 1954 and 1968, (Figures 7.5 and 7.6). The 1938 photograph (Figure 7.4) shows the ONR site characterized by upland agricultural fields and a fringe of terrestrial trees and shrubs; the salt marsh is dominated by *S. alterniflora* with a fringe of *S. patens/D. spicata* adjacent to the upland. The position of the adjacent creek (outlined in blue) does not change significantly over the 70-year time period. The 1938 photo shows that mosquito ditches have been excavated in the marsh. By 1954 (Figure 7.5), the wetland/upland boundary area has moved landward, with a more extensive network of mosquito ditches in the surrounding salt marsh. The 1968 photograph (Figure 7.6) illustrates that the salt marsh has continued to migrate over adjacent uplands, with extensive changes in the configuration of the wetlands/uplands boundary. Halophytic vegetation has migrated laterally over previously upland surfaces; this is evident in the isolation of an upland island in the newly established salt marsh area. Ground-truthing surveys conducted in 1995 confirm this trend; an isolated upland island is presently surrounded by *S. patens* high marsh and *S. alterniflora* low marsh. Remnant tree stumps are observed in the *S. patens* wetland (see 1995 ground-view photograph, Figure 7.7).

Pot Nets North. The Pot Nets North (PNN) site is located at the end of Long Neck, southwestern shore of Rehoboth Bay, at the mouth of Guinea Creek. The

Figure 7.4 Oyster Neck Road site: reproduction of 1938 aerial photograph depicting location of wetland/upland boundary (red line) and tidal creek (blue line) for reference point.

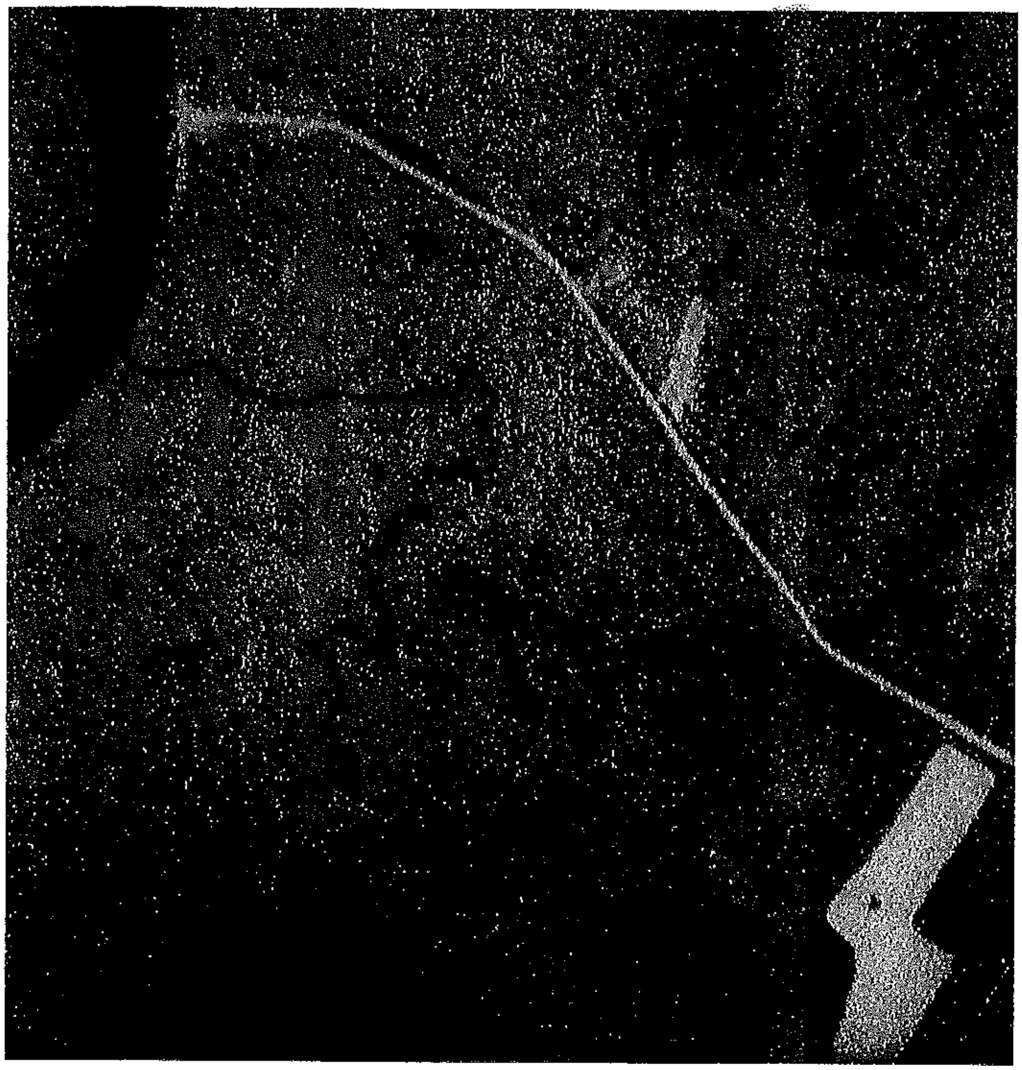
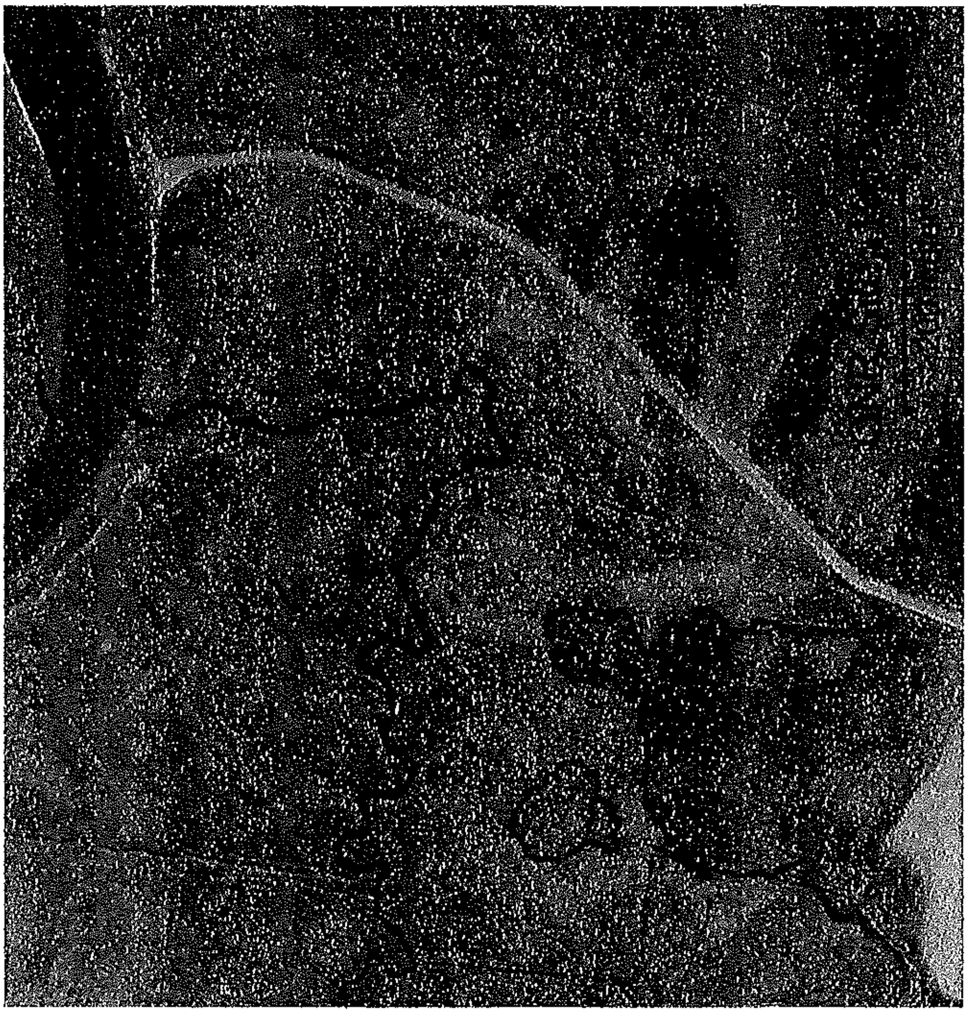


Figure 7.5 Oyster Neck Road site: reproduction of 1954 aerial photograph depicting location of wetland/upland boundary (red line) and tidal creek (blue line) for reference point.

Figure 7.6 Oyster Neck Road site: reproduction of 1968 aerial photograph depicting location of wetland/upland boundary (red line) and tidal creek (blue line) for reference point.



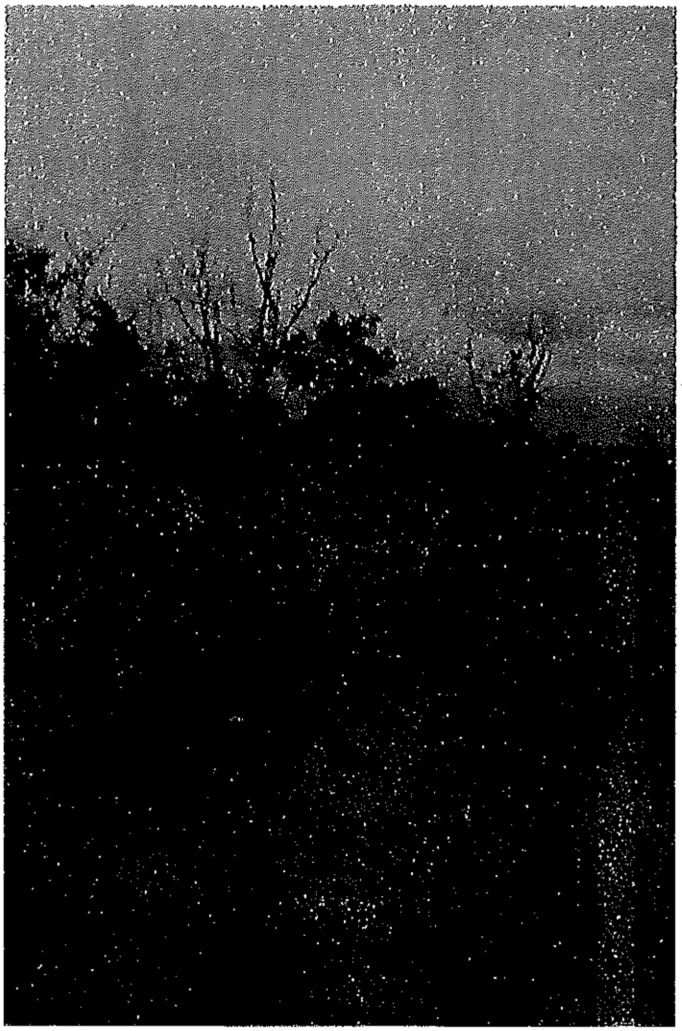


Figure 7.7 1995 ground view photograph of Oyster Neck Road site showing remnant tree stumps in *S. patens* dominated salt marsh.

area is presently characterized by low-lying agricultural fields and loblolly pine forests on a shallowly sloping upland, as well as trailer park home sites. Historical aerial photographs of this location show a pattern of landward migration of fringing tidal wetlands over adjacent wooded uplands. Figure 7.8 depicts the site in 1926; note the nature and extent of the wetlands/uplands boundary (outlined in red) along the agricultural field and along the margins of the natural creek. By 1968 (Figure 7.9) tidal wetlands had migrated landward over adjacent shallowly sloping forested areas, leaving isolated hummocks of trees surrounded by tidal marsh. However, there is little or no landward movement of tidal wetlands along the steeper sloping sides of the tidal creek valley. Few mosquito ditches are observed in the 1926 aerial photograph, but a much more extensive network of mosquito ditches appears in the 1968 photograph. Figure 7.10 is a conceptual depiction of the Pot Nets North site in 2025 based on extrapolation of generalized historical trends into the future, with continued landward movement of wetlands over adjacent wooded upland areas.

Boat House Cove. Located on the northern shore of Indian River Bay, the Boat House Cove (BHC) site provides an excellent illustration of landward movement of tidal wetlands adjacent to three different upland surfaces: 1) agricultural farmland; 2) steeply sloping upland; and 3) shallowly sloping upland. The 1938 aerial photograph (Figure 7.11) depicts the forested upland area as a 'boot shaped' peninsula in the salt marsh. The base of the 'boot', closest to the shore of Indian River Bay, is characterized by a steeply sloping vegetated upland (BHC transect, this study), while the backside of the 'boot' is characterized by a shallowly sloping upland area vegetated by loblolly pine,

Figure 7.8 Pot Nets North site: reproduction of 1926 aerial photograph depicting location of wetland/upland boundary (red line) and tidal creek (blue line) for reference point.

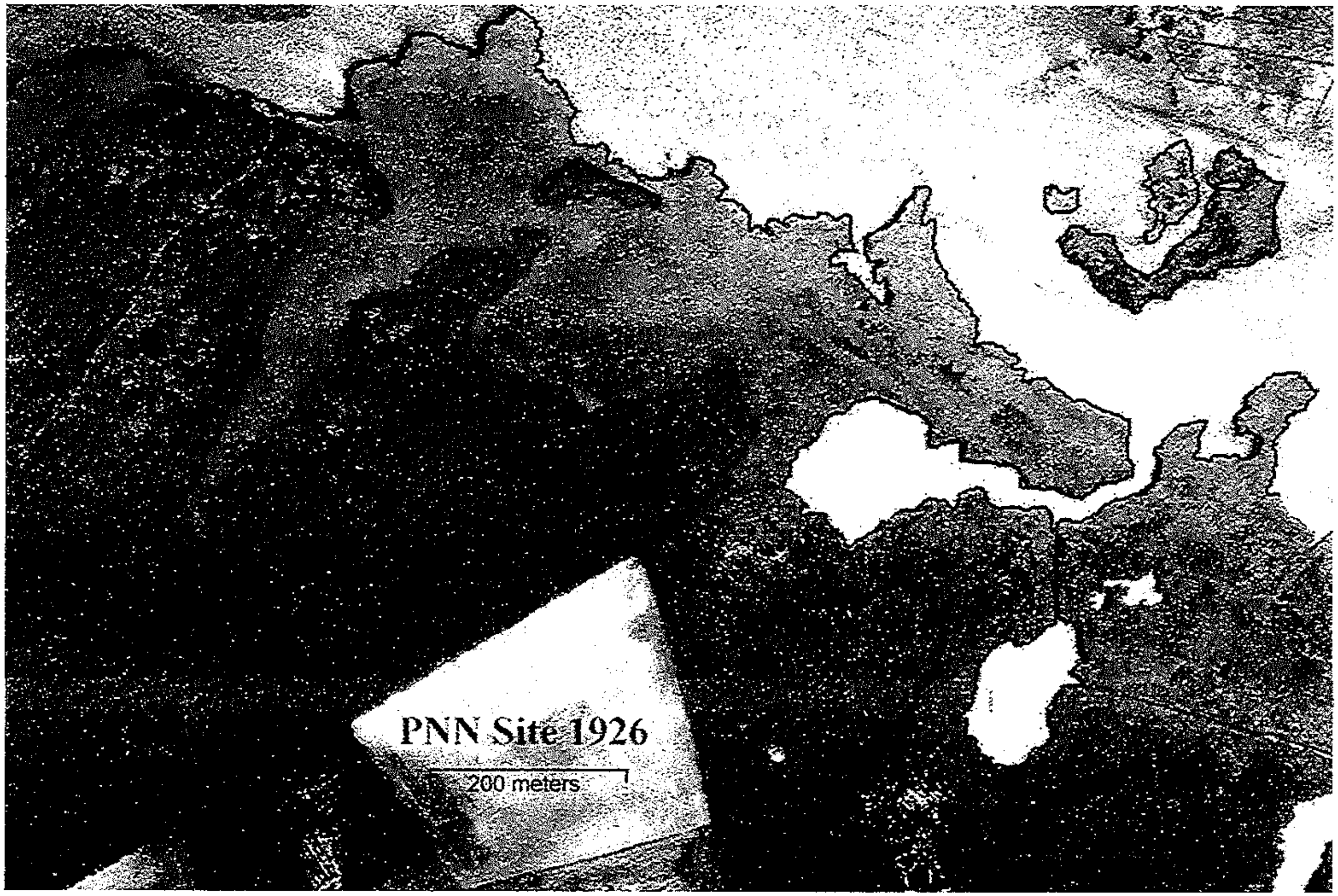
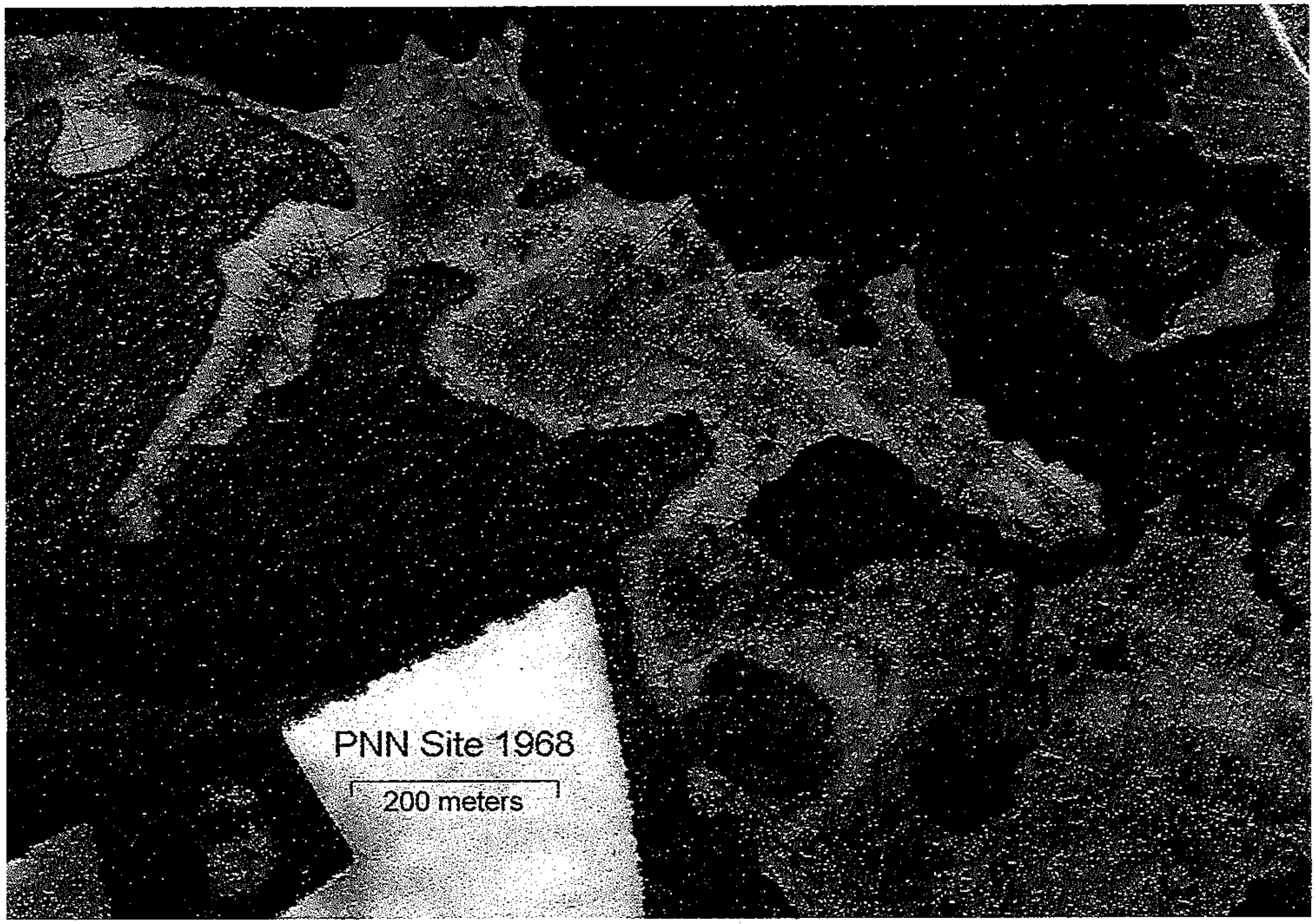

PNN Site 1926
200 meters

Figure 7.9 Pot Nets North site: reproduction of 1968 aerial photograph depicting location of wetland/upland boundary (red line) and tidal creek (blue line) for reference point.


PNN Site 1968
200 meters

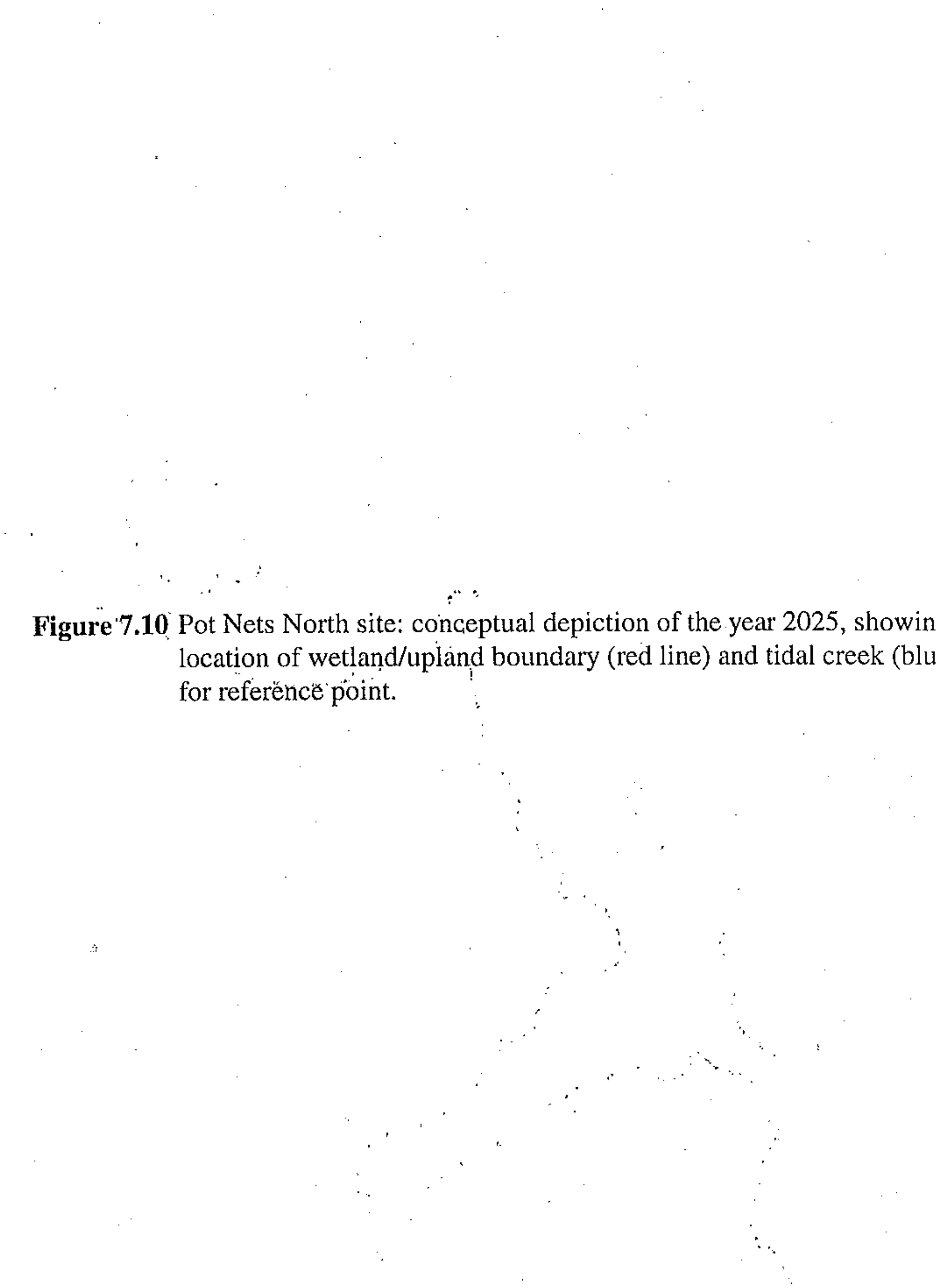


Figure 7.10 Pot Nets North site: conceptual depiction of the year 2025, showing location of wetland/upland boundary (red line) and tidal creek (blue line) for reference point.

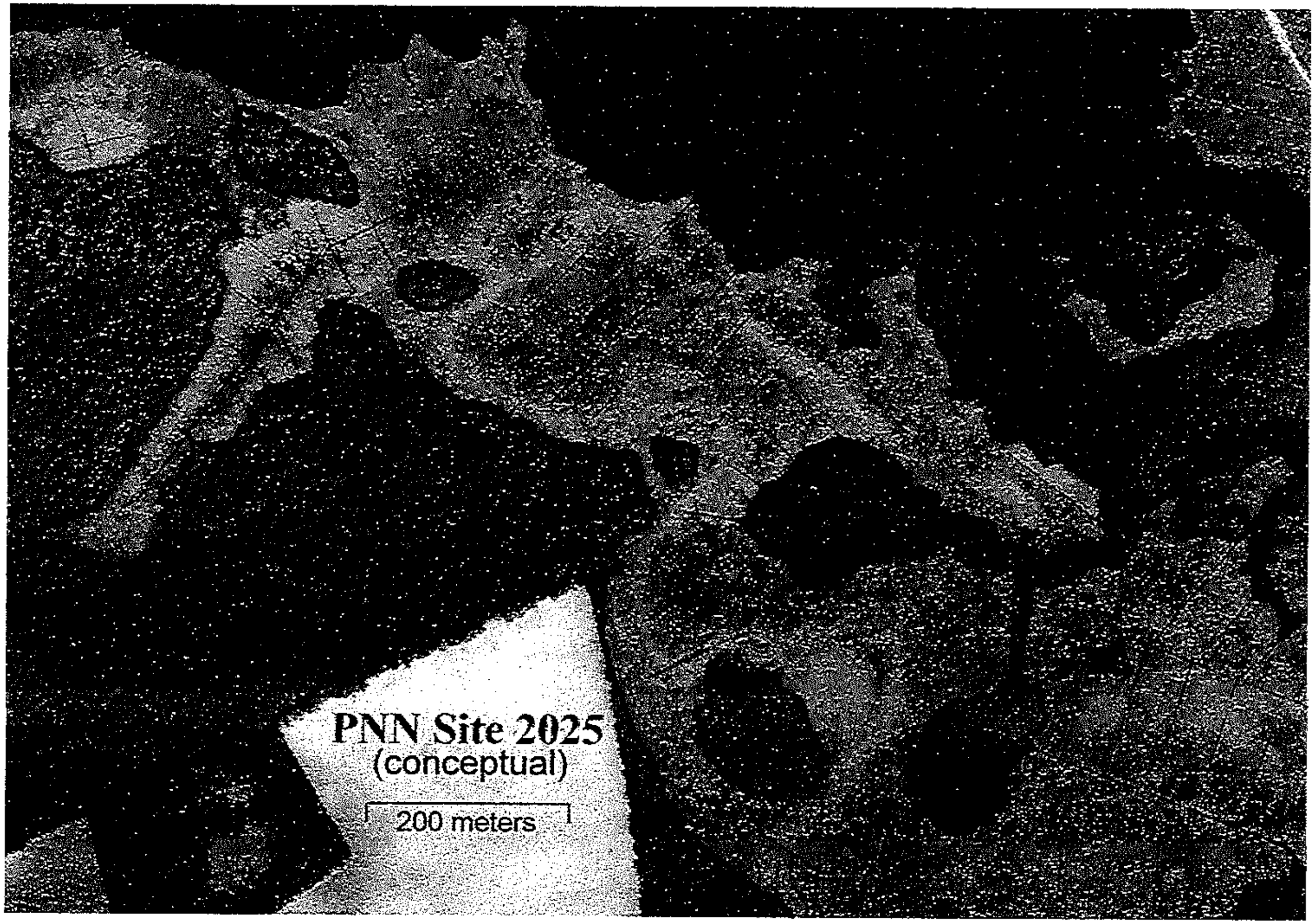

PNN Site 2025
(conceptual)
200 meters

Figure 7.11 Boat House Cove site: reproduction of 1938 aerial photograph depicting location of wetland/upland boundary (red line) and tidal creek (blue l for reference point.

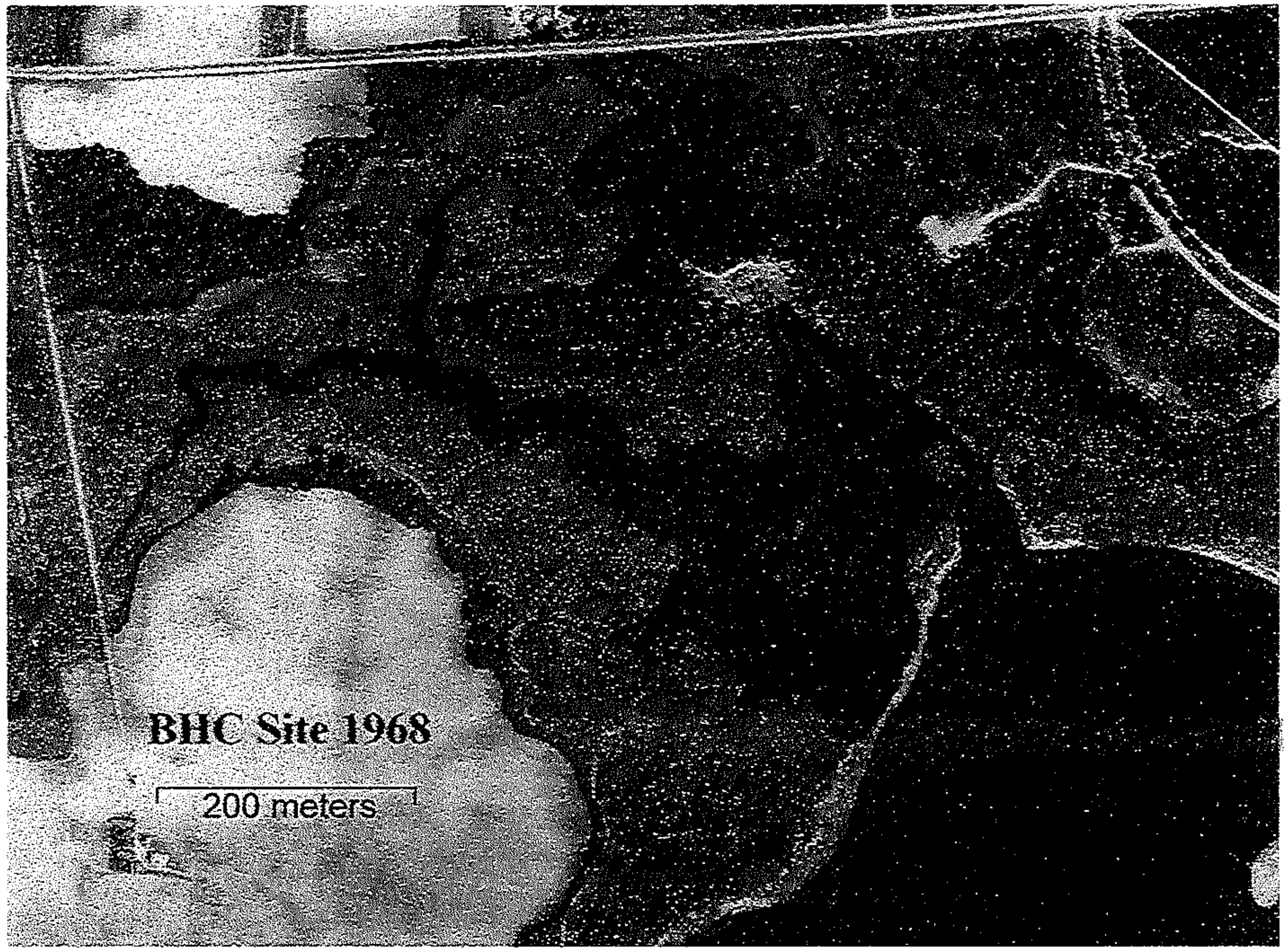

200 nm

poison ivy, and greenbriar. By 1968 (Figure 7.12), changes in the location of the wetlands/uplands boundary are easily observable along the shallowly sloping backside of the upland island, and in the area of the adjacent tidal creek (east side of photograph). Additionally, an extensive network of mosquito ditches are evident. This 1968 photograph clearly shows the dramatic changes in vegetative characteristics at the wetlands/uplands boundary in the shallowly sloping areas. Zones that had been previously wooded are now dominated by *S. patens/D. spicata* high marsh vegetation. However, note that in the area characterized by steep adjacent uplands, there has been little or no change in the characteristics of the vegetation. These changes have also been documented by ground-truth surveying in 1995; as illustrated in the ground view photograph (Figures 7.13 and 7.14), there are many tree stumps in areas now colonized by *S. patens* and *S. alterniflora*. Changes have taken place very rapidly at this site; Figure 7.15 is a conceptual depiction of Boat House Cove study area in the year 2025. Note that tidal wetlands have transgressed over the shallowly sloping wooded uplands, but not over the steeply sloping areas.

Metric Maps: Transgression Rates

Introduction. Historical changes in both the shoreline configuration and the location of the wetlands/uplands boundary were quantified for study sites in Indian River and Rehoboth Bays (see location map, Figure 7.16). Changes of the shoreline and wetlands/uplands boundary were measured from Metric Maps provided by Leatherman's

Figure 7.12 Boat House Cove site: reproduction of 1968 aerial photograph depicting location of wetland/upland boundary (red line) and tidal creek (blue line) for reference point.


BHC Site 1968
200 meters

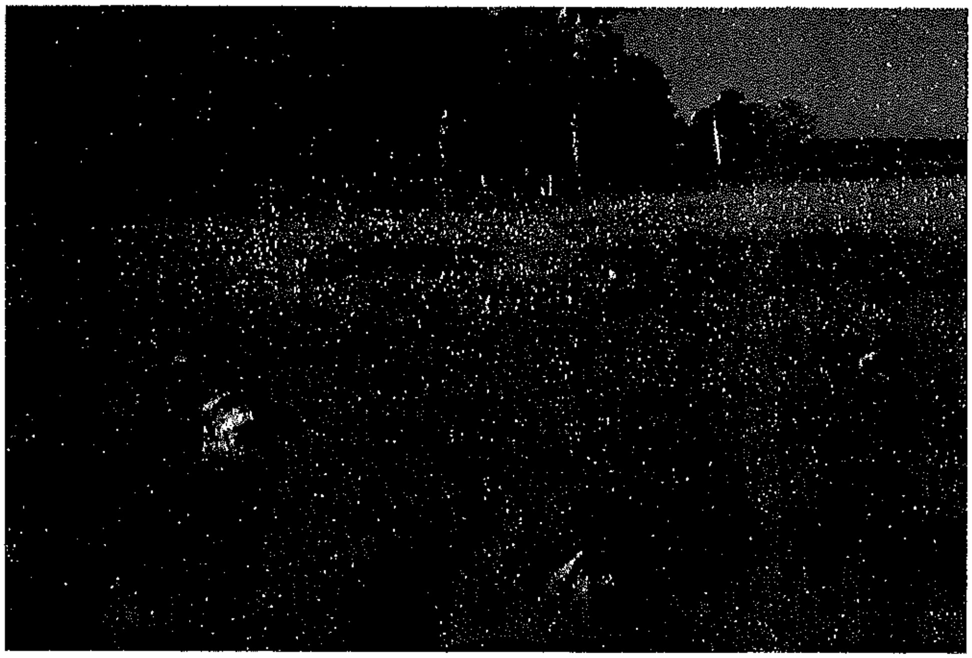
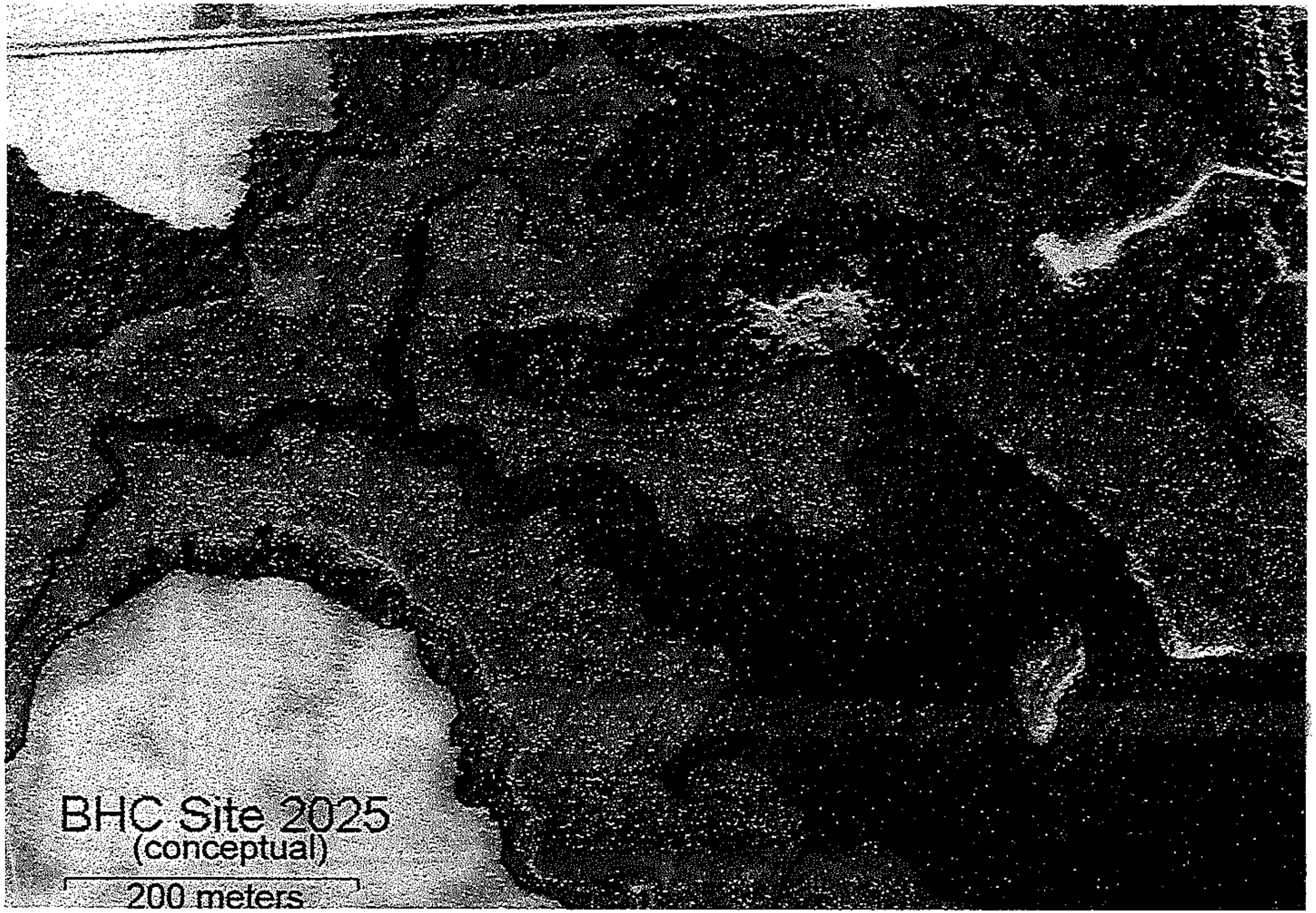


Figure 7.13 1995 ground view photograph of Boat House Cove site showing remnant tree stumps in *S. patens* dominated salt marsh.

Figure 7.14 1995 ground view photograph of Boat House Cove site showing remnant tree stumps in *S. patens* dominated salt marsh.

Figure 7.15 Boat House Cove site: conceptual depiction of the year 2025, showing location of wetland/upland boundary (red line) and tidal creek (blue line) for reference point.


BHC Site 2025
(conceptual)
200 meters

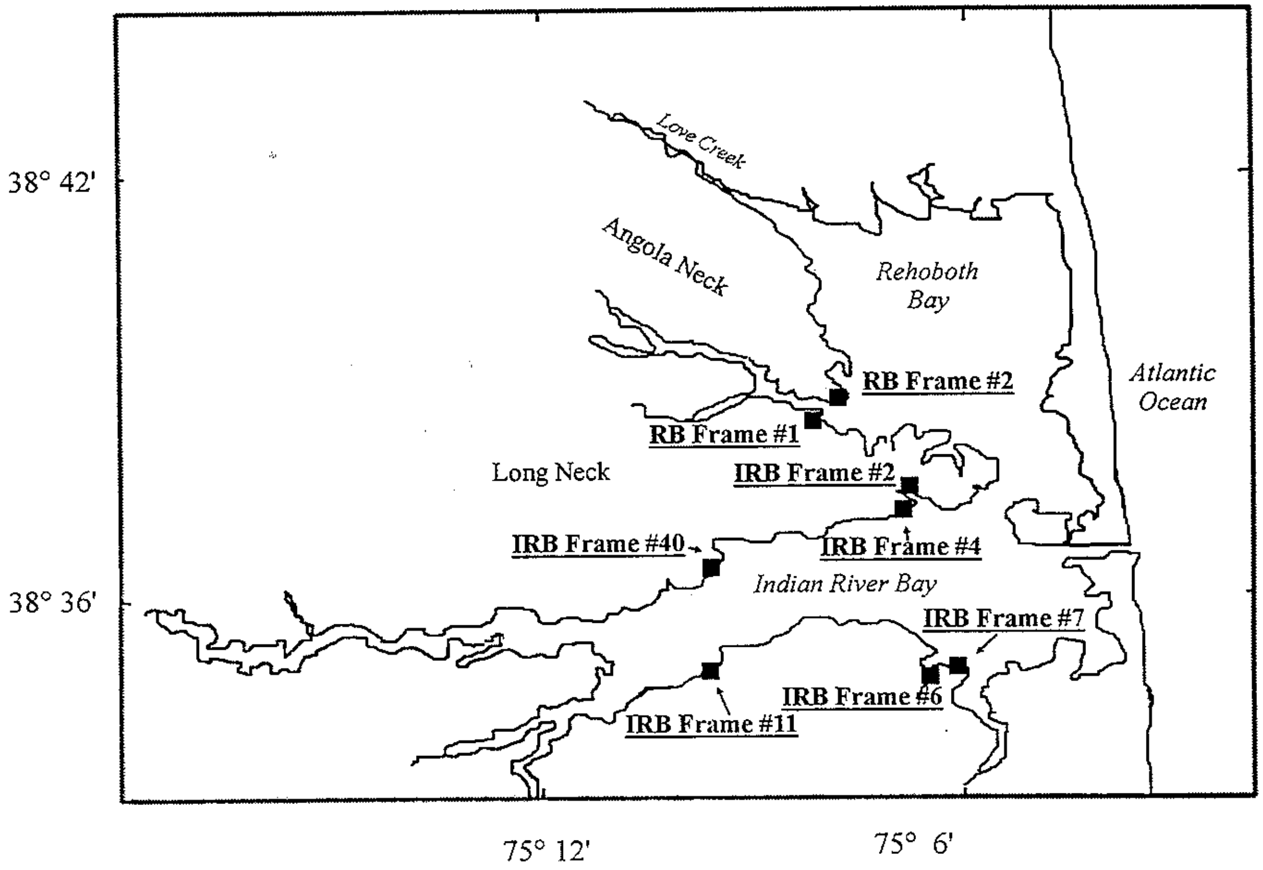


Figure 7.16 Map depicting location of metric map study sites, Indian River and Rehoboth Bays.

Coastal Mapping Group. Measured changes are correlated with various environmental conditions and with possible human influences.

Transgressive movement, or landward migration of wetland environments, is observed at the wetlands/uplands boundary in many areas, with noticeable spatial and temporal variation in rates of transgression as calculated from the metric maps. Rates of landward transgression of Delaware's salt marshes may also be characterized based on slope of adjacent upland, present marsh characteristics, vertical accretion rates, and projected rates of sea-level rise.

Indian River Bay: Frame #40. The metric map for Indian River Bay Frame #40 is depicted in Figure 7.17. Table 7.4 shows the landward migration rates of fringing tidal wetlands for four transects along the northwest side of Indian River Bay in the vicinity of the Lingo Point transect.

Table 7.4 Indian River Bay, Frame #40, in the vicinity of Lingo Point. Average annual wetland landward migration rates (m/yr); (+) indicates landward migration of wetland environments or encroachment of wetlands over uplands; (-) indicates wetlands loss or encroachment of uplands over wetland.

Transect	1944-1954	1954-1968	1968-1989	1944-1989	Notes:
A	+ 10.6	+0.6	+ 11.6	+8.0	shallow slope
B	+ 2.4	+ 1.3	+ 3.6	+ 2.6	shallow slope
C	+ 2.1	+2.1	+0.3	+2.0	shallow slope
D	+ 2.1	+ 1.1	+0.1	+0.8	shallow slope

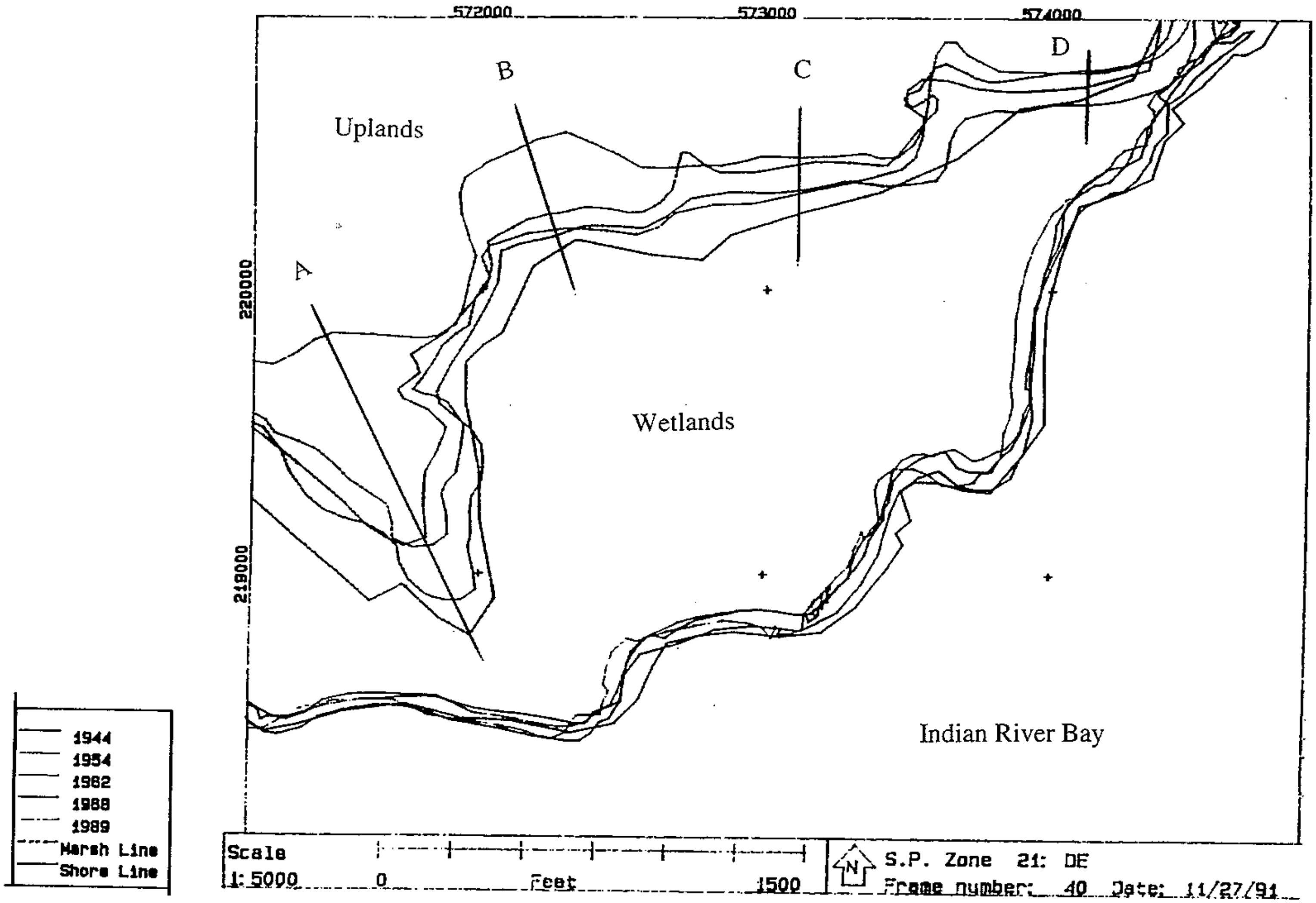


Figure 7.17 Metric map for Indian River Bay, Frame #40.

Indian River Bay: Frame #2. The metric map for Indian River Bay Frame #2 is depicted in Figure 7.18. Table 7.5 shows the landward migration rates of fringing tidal wetlands for six transects along the north side of Indian River Bay in the vicinity of the Boat House Cove and Burton profile site transect locations.

Table 7.5 Indian River Bay, Frame #2, Burton's Farm Field. Average annual wetland landward migration rates (m/yr); (+) indicates landward migration of wetland environments or encroachment of wetlands over uplands; (-) indicates wetlands loss or encroachment of uplands over wetland.

Transect	1944-1954	1954-1962	1962-1968	1944-1968	Notes:
A	-1.2	0.0	+4.6	+0.6	steep profile
B	-0.6	+1.5	-2.5	+0.4	steep profile
C	+4.0	-0.8	-3.1	-0.6	steep profile
D	+0.9	-1.1	-3.6	-0.9	steep profile
E	0.0	-0.8	-0.5	-0.4	steep profile
F	+2.1	+1.9	-6.6	+0.1	steep profile

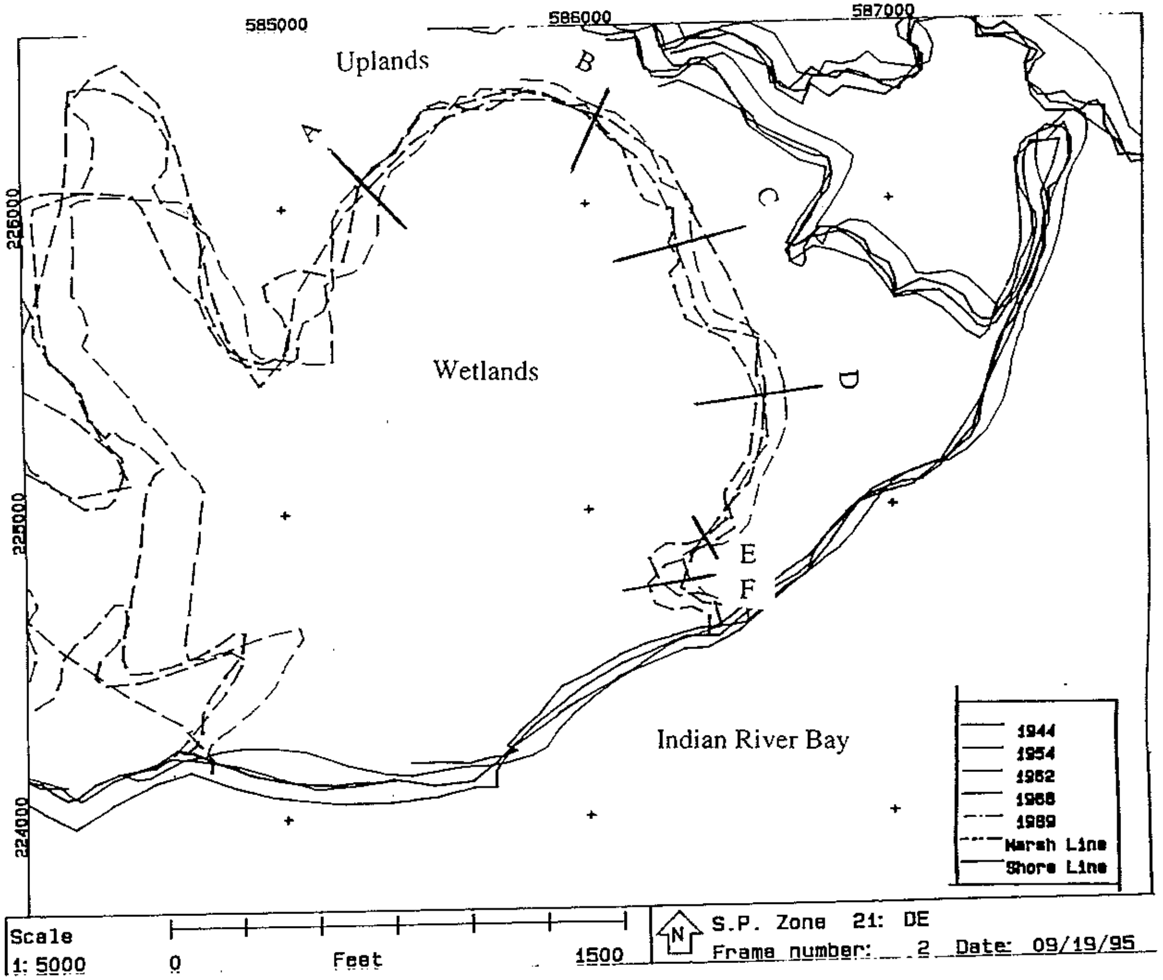


Figure 7.18 Metric map for Indian River Bay, Frame #2.

Indian River Bay: Frame #4. The metric map for Indian River Bay Frame #4 is depicted in Figure 7.19. Table 7.6 shows the landward migration rates of fringing tidal wetlands for six transects along the northern side of Indian River Bay in the vicinity of the Boat House Cove transect location.

Table 7.6 Indian River Bay, Frame #4, Boat House Cove Site. Average annual wetland landward migration rates (m/yr); (+) indicates landward migration of wetland environments or encroachment of wetlands over uplands; (-) indicates wetlands loss or encroachment of uplands over wetland.

Transect	1944-1954	1954-1962	1962- 1968	1944-1968	Notes:
A	+ 2.2	- 1.1	+ 2.2	+1.1	shallow slope
B	+ 3.5	- 3.8	+ 3.3	+ 1.0	shallow slope
C	+ 3.1	+ 1.9	+ 5.8	+3.4	shallow slope
D	+ 4.1	+0.8	+ 7.3	+ 3.8	shallow slope
E	- 5.0	+ 2.7	+ 2.6	- 0.6	steep slope
F	- 6.5	- 1.1	+ 1.5	- 0.2	steep slope

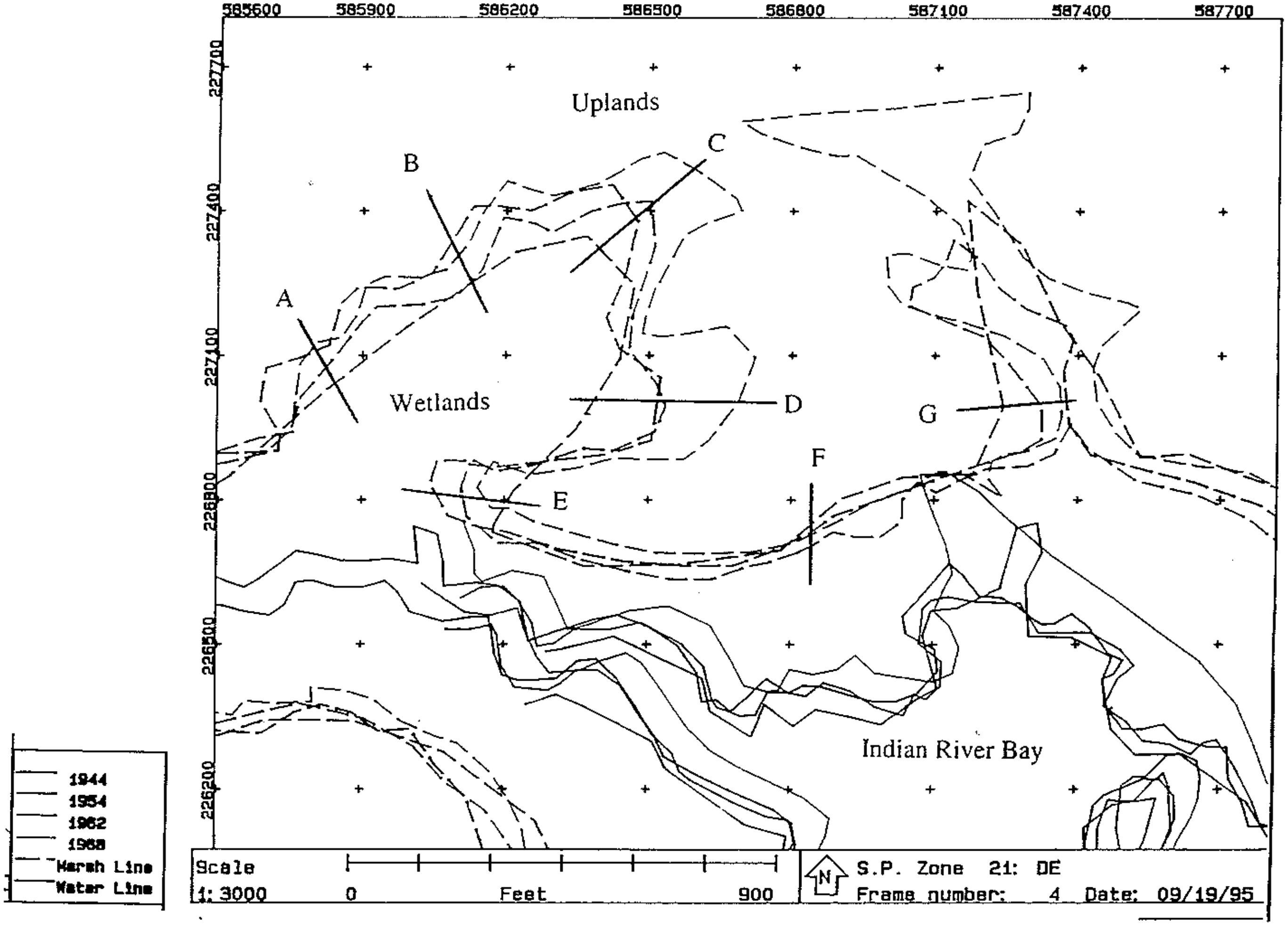


Figure 7.19 Metric map for Indian River Bay, Frame #4.

Indian River Bay: Frame #6. The metric map for Indian River Bay Frame #6 is depicted in Figure 7.20. Table 7.7 shows the landward migration rates of fringing tidal wetlands for five transects along the southern side of Indian River Bay in the vicinity of the Bethany Bay transect locations.

Table 7.7 Indian River Bay, Frame #6, Big Marsh Point. Average annual wetland landward migration rates (m/yr); (+) indicates landward migration of wetland environments or encroachment of wetlands over uplands; (-) indicates wetlands loss or encroachment of uplands over wetland.

Transect	1944-1954	1954-1968	1968-1989	1944-1989	Notes:
A	- 2.1	+2.0	+0.4	+0.3	steep slope
B	+1.2	+2.2	+2.3	+2.0	shallow slope
C	+0.9	+0.9	+1.3	+1.1	shallow slope
C1	+0.9	+2.2	+1.9	+1.8	shallow slope
C2	- 0.6	+2.6	+0.6	+1.0	shallow slope

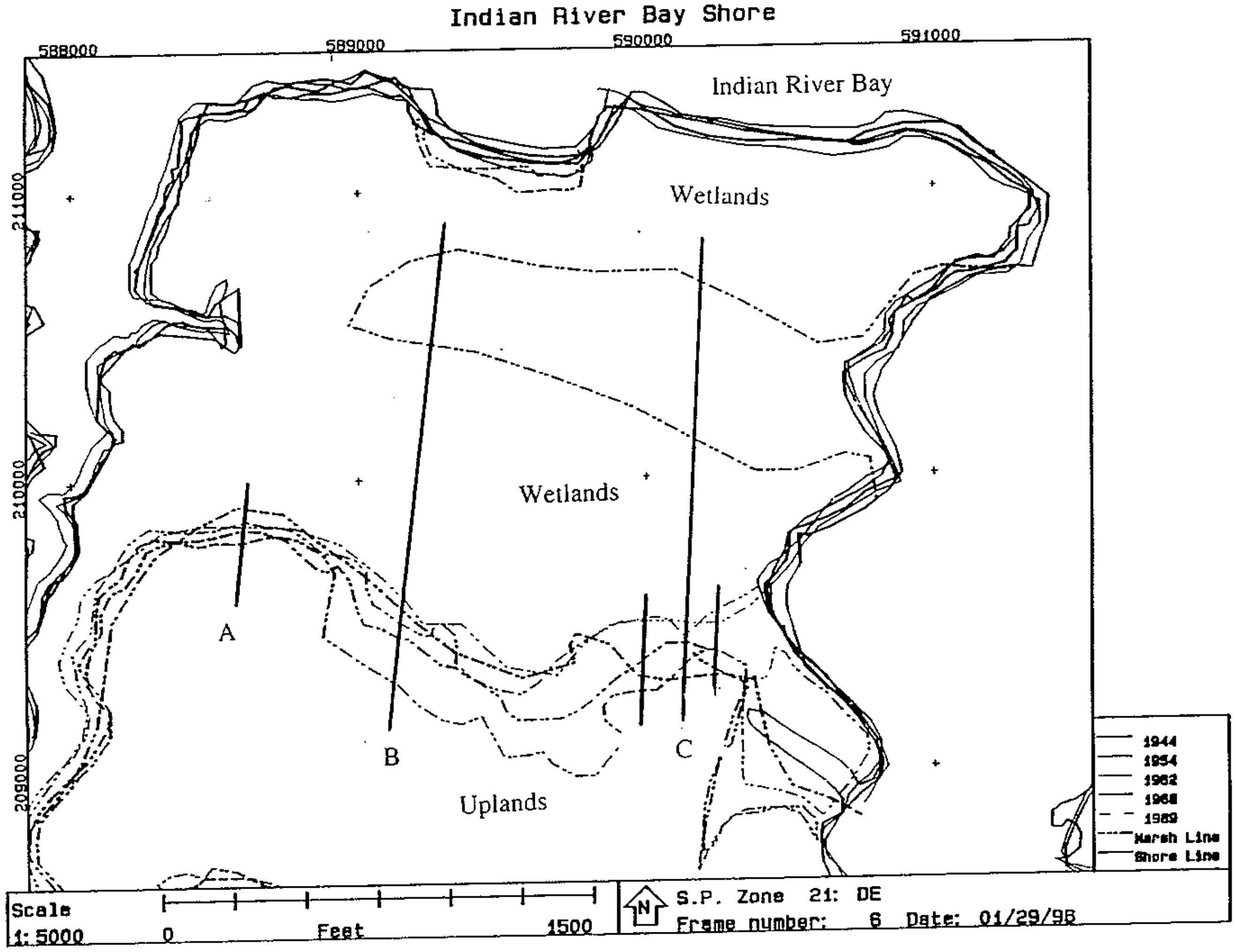


Figure 7.20 Metric map for Indian River Bay, Frame #6.

Indian River Bay: Frame #7. The metric map for Indian River Bay Frame #7 is depicted in Figure 7.21. Table 7.8 shows the landward migration rates of fringing tidal wetlands for three transects along the southern side of Indian River Bay.

Table 7.8 Indian River Bay, Frame #7. Average annual wetland landward migration rates (m/yr); (+) indicates landward migration of wetland environments or encroachment of wetlands over uplands; (-) indicates wetlands loss or encroachment of uplands over wetland.

Transect	1944-1954	1954-1968	1968-1989	1944-1989	Notes:
A	+0.6	+ 2.4	+ 5.7	+ 5.2	river valley axis
B	+17.1	+2.6	+ 2.0	+ 5.6	river valley axis
C	+ 1.2	+ 2.2	+0.6	+ 1.2	side slope of river valley

Indian River Bay: Frame #11 The metric map for Indian River Bay Frame #11 is depicted in Figure 7.22. Table 7.9 shows the landward migration rates of fringing tidal wetlands for one transect along the southwestern side of Indian River Bay.

Table 7.9 Indian River Bay, Frame #11, Ellis Point. Average annual wetland landward migration rates (m/yr); (+) indicates landward migration of wetland environments; (-) indicates wetlands loss or encroachment of uplands over wetlands.

Transect	1944-1954	1954-1968	1968-1989	1944-1989	Notes:
A	+0.9	+0.9	+0.2	+0.5	steep slope

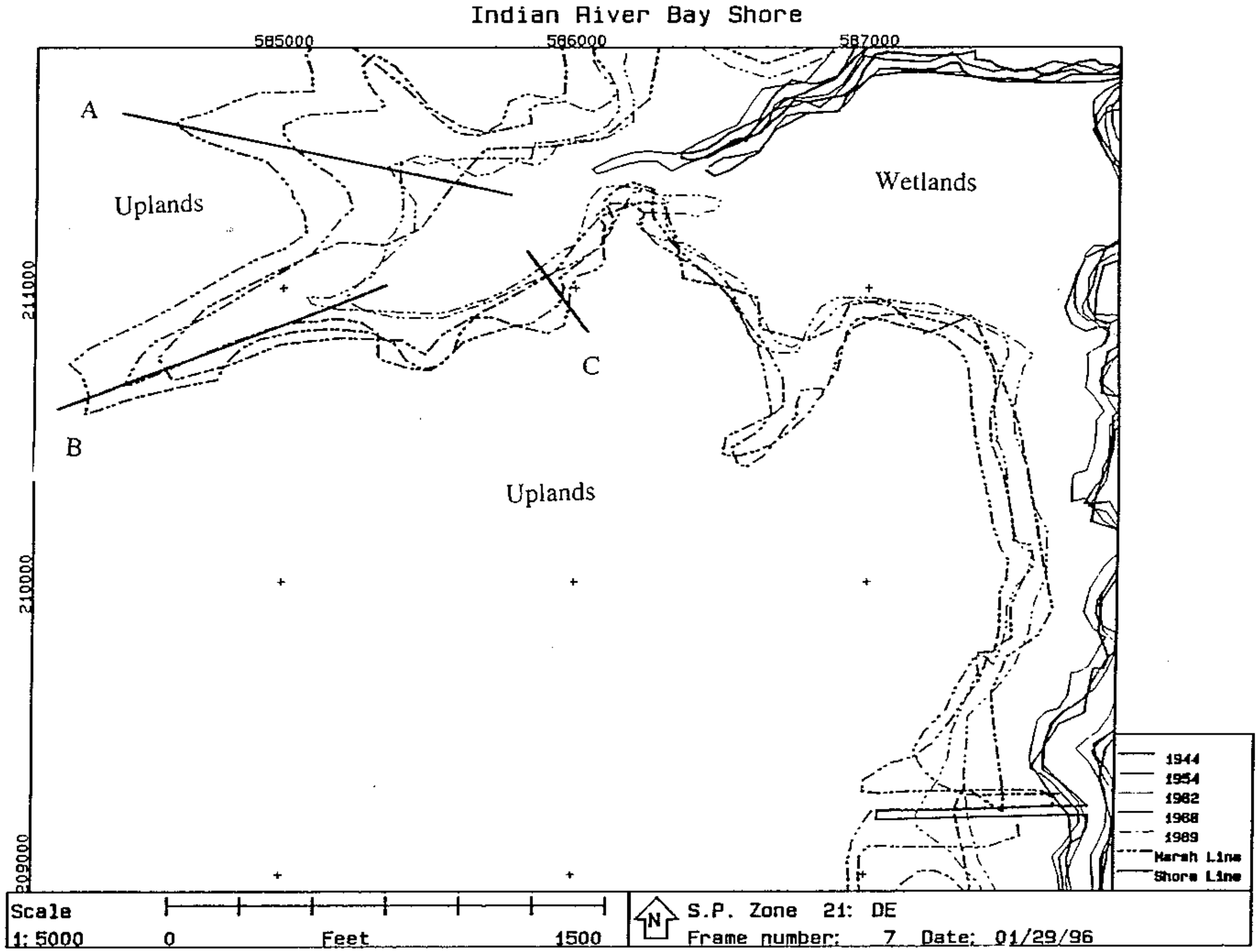


Figure 7.21 Metric map for Indian River Bay, Frame #7.

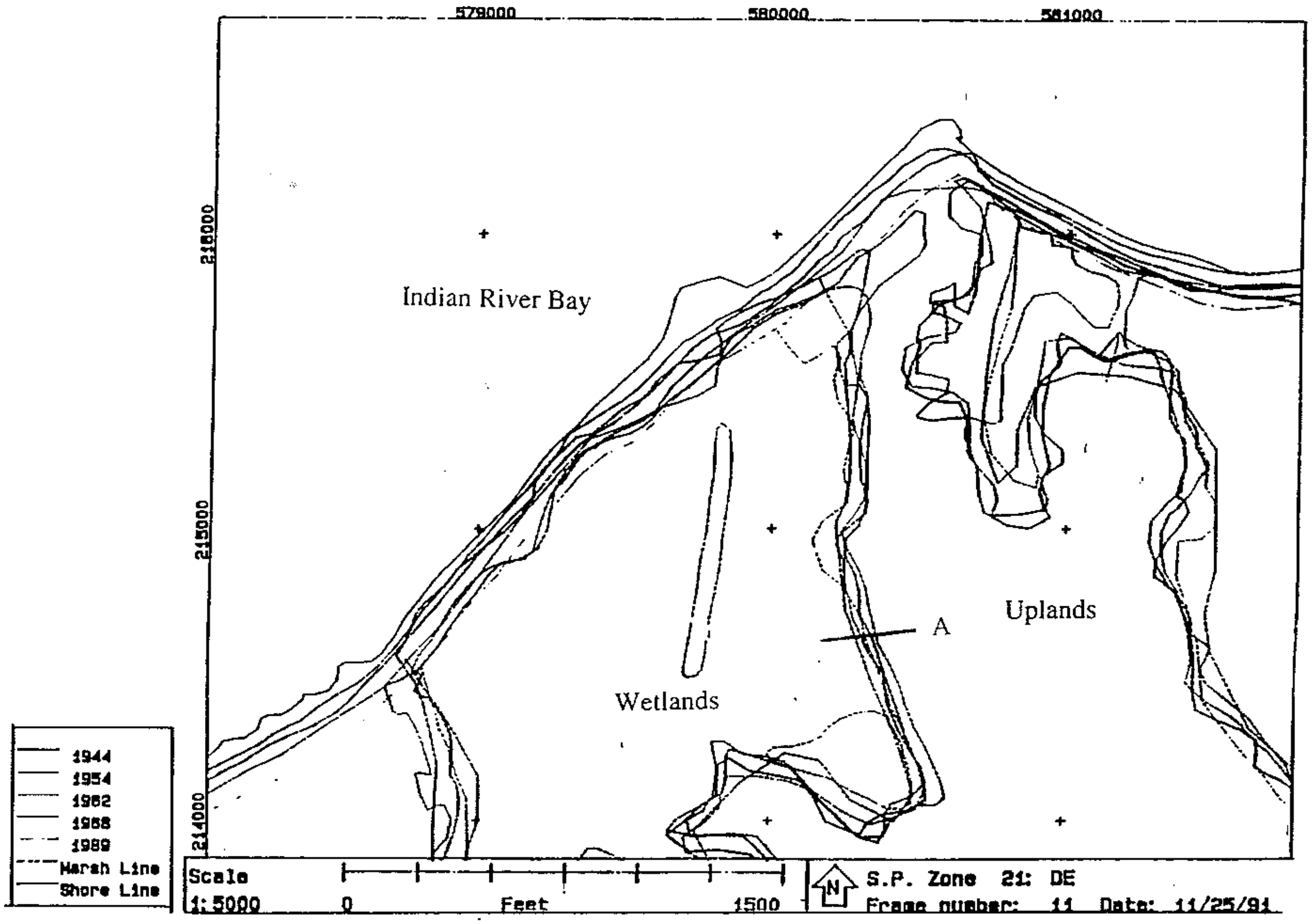


Figure 7.22 Metric map for Indian River Bay, Frame #11.

Rehoboth Bay

A location map for Rehoboth Bay metric maps is shown in Figure 7.16.

Rehoboth Bay: Frame #1. The metric map for Rehoboth Bay Frame #1 is depicted in Figure 7.23. Table 7.10 shows the landward migration rates of fringing tidal wetlands for six transect along the southwestern Rehoboth Bay in the vicinity of the Pot Nets North transect locations.

Table 7.10 Rehoboth Bay, Frame #1, Pot Nets North Site. Average annual wetland landward migration rates (m/yr); (+) indicates landward migration of wetland environments or encroachment of wetlands over uplands; (-) indicates wetlands loss or encroachment of uplands over wetland.

Transect	1944-1954	1954-1968	1968-1989	1944-1989	Notes:
A	n/a	+ 5.2	+ 2.3	+1.8	stream valley axis
B	+ 1.8	+ 1.1	+0.4	+1.0	stream valley axis
C	+ 1.8	+0.7	+ 6.0	+ 3.4	shallow slope
D	0.0	+ 1.1	-0.1	+0.1	steep slope
E	- 1.2	+ 1.3	-0.4	+0.1	steep slope
F	+ 4.9	+ 1.7	- 1.5	+0.9	shallow slope

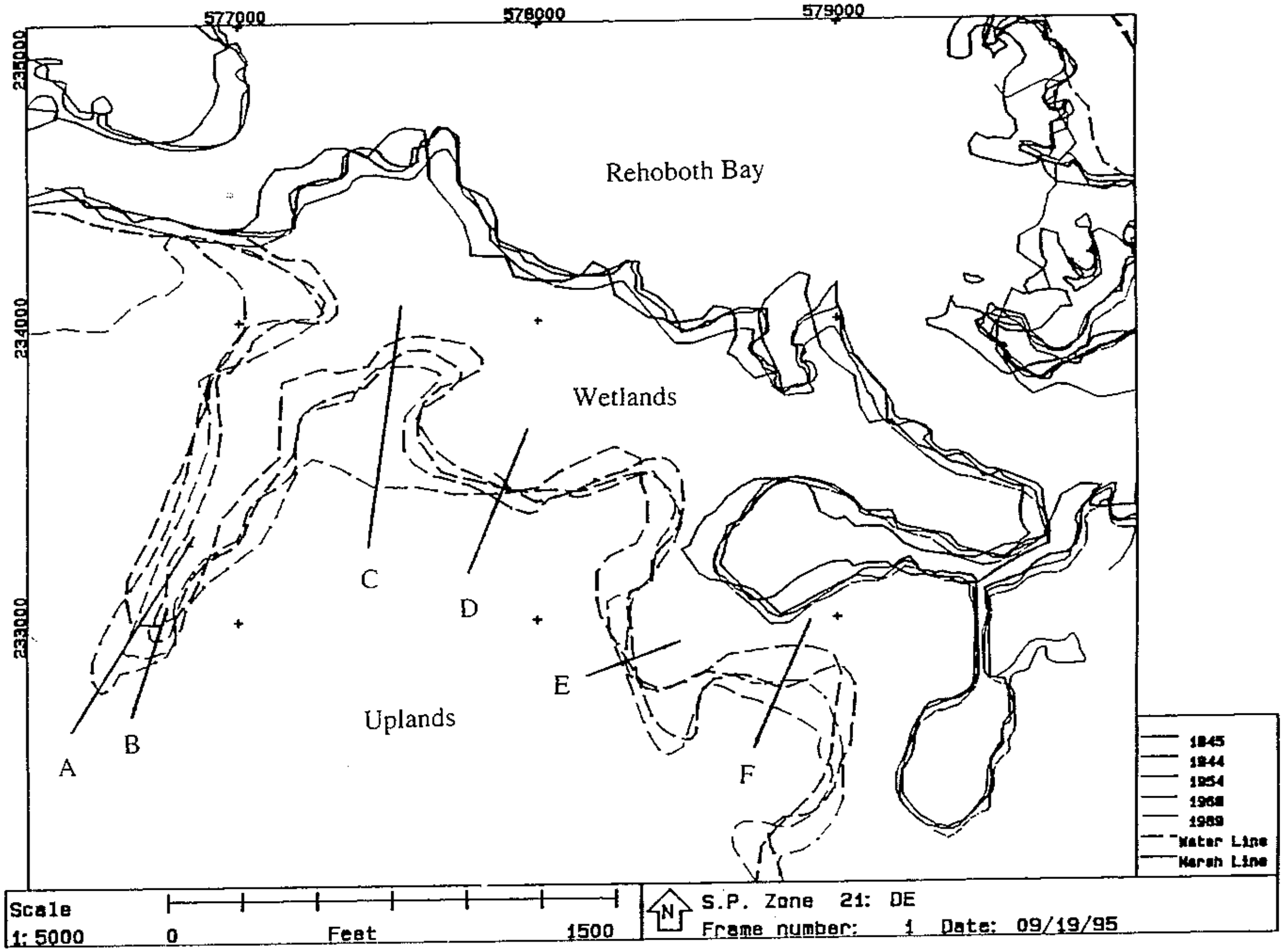


Figure 7.23 Metric map for Rehoboth Bay, Frame #1.

Rehoboth Bay: Frame #2. The metric map for Rehoboth Bay Frame #2 is depicted in Figure 7.24. Table 7.11 shows the landward migration rates and shore erosions rates of fringing tidal wetlands for three transect along the western side of Rehoboth Bay in the vicinity of the Delaware Wildlands transect locations.

Table 7.11 Rehoboth Bay, Frame #2, Delaware Wildlands site. Average annual wetland landward migration rates (m/yr); (+) indicates landward migration of wetland environments or encroachment of wetlands over uplands; (-) indicates wetlands loss or encroachment of uplands over wetland.

Transect	1845-1954	1954-1968	1968-1989	1845-1989	Notes:
A	+0.5	+0.9	+0.3	+0.4	steep slope
B	+0.5	+0.4	-0.4	+0.4	steep slope
C	+ 1.6	+ 0.0	+0.1	+ 1.2	shallow slope

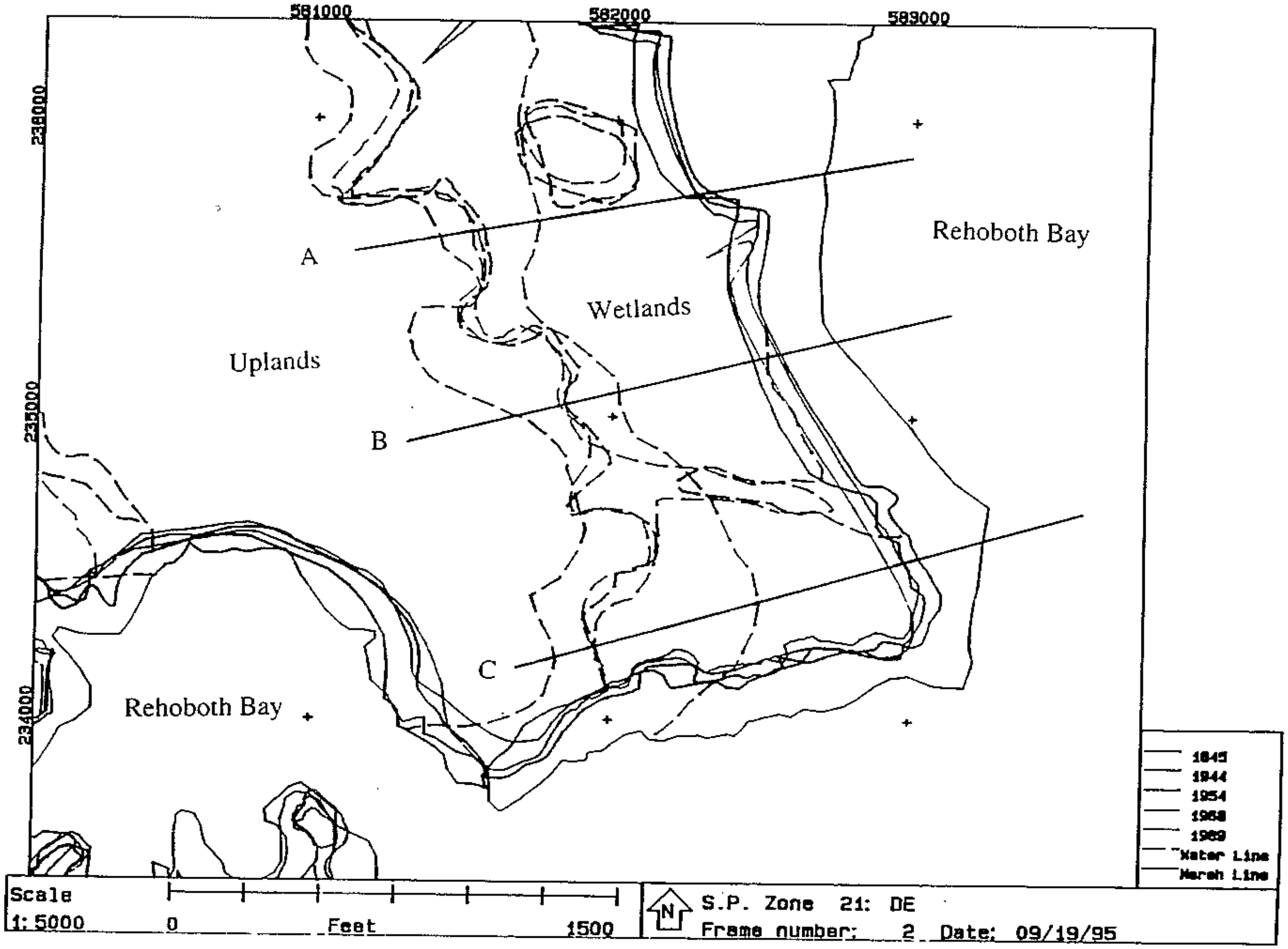


Figure 7.24 Metric map for Rehoboth Bay, Frame #2.

General Characterization: Landward Migration Rates of Fringing Tidal Wetlands

The following general characterizations of landward migration rates of fringing tidal wetlands are indicative of past trends that may be applied qualitatively to future scenarios. Analyses of position changes in shorelines and wetlands/uplands boundary along Indian River and Rehoboth Bays are dependent upon interpretations of historical-change maps and measurements made along transects perpendicular to the shoreline/boundary area. The overall averaged annual rates of lateral movement of the wetlands/uplands boundary for each time span in Indian River and Rehoboth Bays are shown in Tables 7.12 and 7.13, respectively.

Table 7.12 Indian River Bay: Mean rates of landward migration of fringing tidal wetlands (m/yr); (+) indicates landward migration of wetland environments or encroachment of wetlands over uplands; (-) indicates wetlands loss or encroachment of uplands over wetland.

	1944-1954	1954-1962	1954-1968	1962-1968	1968-1989	1944-1968	1944-1989
Overall	+2.87	n/a	+1.71	n/a	+2.24	n/a	+2.35
Steep Slope	n=10 -0.75 sd 3.16	n=8 +0.30 sd 1.51	n=2 +1.41 sd 0.77	n=8 -0.94 sd 3.69	n=2 +0.30 sd 0.21	n=8 -0.20 sd 0.53	n=2 +0.44 sd 0.14
Shallow Slope	n=11 +1.99 sd 1.34	n=4 +1.36 sd 0.77	n=7 +1.77 sd 0.66	n=4 +4.65 sd 2.32	n=7 +1.44 sd 1.25	n=4 +2.31 sd 1.48	n=7 +1.60 sd 0.66
Valley Axis	n=4 +7.38 sd 7.93	n/a	n=4 +1.96 sd 0.90	n/a	n=4 +4.97 sd 4.91	n/a	n=4 +5.00 sd 2.82

Table 7.13 Rehoboth Bay: Mean rates of landward migration of fringing tidal wetlands (m/yr); (+) indicates landward migration of wetland environments or encroachment of wetlands over uplands; (-) indicates wetlands loss or encroachment of uplands over wetland.

	1845-1954	1944-1954	1954-1968	1968-1989	1944-1989
Overall	n/a	+1.52	+1.62	+0.91	+1.16
Steep Slope	n=2 +0.53 sd 0.00	n=2 -0.60 sd 0.85	n=4 +0.93 sd 0.37	n=4 -0.17 sd 0.34	n=4 0.25 sd 0.20
Shallow Slope	n=1 +1.60 sd 0.00	n=2 +3.35 sd 2.19	n=3 +0.78 sd 0.86	n=3 +1.54 sd 3.94	n=3 +1.85 sd 1.35
Valley Axis	n/a	n=2 +1.80 sd 0.00	n=2 +3.15 sd 2.90	n=2 +1.37 sd 1.32	n=2 +1.39 sd 0.62

In general, the Metric Maps show a landward migration of fringing tidal wetlands, which is typical in a transgressive coastal area. Average total net change at the wetlands/uplands boundary for all areas studied between the years 1944 and 1989, including the stream valley axis areas, is +1.76 meters per year. If the stream valley axis rates are removed from the long-term average, the total net change at the wetlands/uplands boundary is +1.03 m/yr from 1944-1989, and +1.06 m/yr from 1944-1968. However, these rates are averages from measurements at selected study sites. Rates of landward migration are likely to be highly variable throughout the study area, and many external parameters may affect individual rates at specific locations. These rates should be representative of general trends of lateral migration of Delaware's fringing tidal wetlands.

Indian River Bay. In general, the highest rates of lateral migration occurred during the 1944 to 1954 time span, with an average annual rate of landward migration of +2.87 meters/year. This is the period of time after the inlet was opened and stabilized by the U.S. Army Corps of Engineers in 1938, with a consequent increase in tidal range and tidal flooding in Indian River and Rehoboth Bays. The lowest average annual rate of lateral migration of +1.71 m/yr occurred between 1954 and 1968. The period between 1968 and 1989 is again characterized by a more rapid rate of landward migration (+2.24 m/yr). This may reflect the deepening of the inlet channel and associated increases in tidal range, which would have a significant effect on transgression rates.

Rehoboth Bay. The landward migration rates of fringing tidal wetlands in Rehoboth Bay are more variable than those in Indian River Bay. The period with the greatest overall net change is 1954-1968, with an average annual transgression rate of +1.62 m/yr, while the 1944-1954 period had an overall transgression rate of +1.52 m/yr. The time period with the lowest rate of landward transgression (+0.91 meters per year) is 1968-1989, when the bay may have reached an equilibrium condition with regard to tidal range and sediment supply. This decrease in the rate of landward migration may also be related to decreases in sediment supply resulting from anthropogenic impacts such as mosquito ditching and inlet stabilization.

Landward Migration Rate and Adjacent Upland Slope. A correlation was found between rate of landward migration of the wetland surface and antecedent topography. The study sites selected for historical rate analyses are representative of three distinct antecedent topographies, or adjacent upland geomorphologies: 1) a low-lying, shallowly sloping adjacent upland area previously colonized by upland vegetation such as loblolly pine; 2) a steeply sloping adjacent upland; and 3) upstream portions of tidal streams tributary to the bays. Where adjacent upland topography is steep, average lateral migration rates are lower, ranging from -0.94 to +1.41 meters per year, with a mean of +0.35 meters per year. Where the adjacent upland slope is shallow, average lateral migration rates are greatest, ranging from +0.78 to +4.65 m/yr with a long-term mean of +1.73 m/yr. The most rapid rates of landward migration of the wetlands uplands boundary area are found in headwater areas of tidal streams tributary to the bays. In these areas, average lateral migration rates range from +1.60 to +9.44 meters per year, with an average of +3.20 m/yr. Figure 7.25 is a graphical representation of average annual landward transgression rate (m/yr) for steep slope, shallow slope, and stream valley axis areas. A more detailed description of the three geomorphic zones follows.

Shallow Sloping Upland. Examples of shallow sloping adjacent upland areas include: the northwestern portion of the Boat House Cove upland site, the Pot-Nets North study area, and the Big Marsh Point location in southern Indian River Bay. These areas experienced the greatest changes in landward movement of the wetlands/uplands boundary between 1945 and 1968. These areas are typically

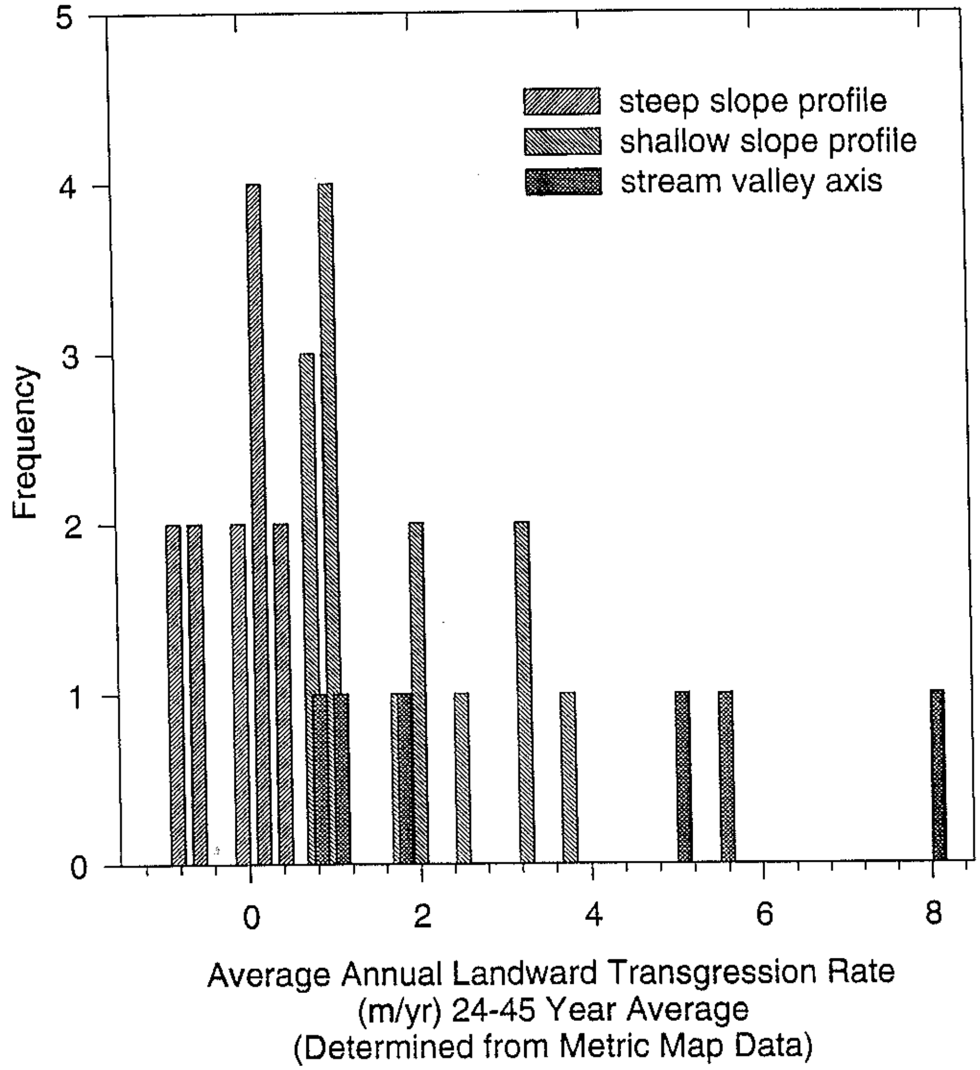


Figure 7.25 Histogram of average annual landward transgression rate (m/yr) for steep slope, shallow slope, and stream valley axis areas (24-45 year average, as determined from metric map data).

characterized as the extreme distal portion of upland necks, comprised of gently sloping (0.005 tangent) sandy uplands colonized by upland species such as panic grass, poison ivy, and loblolly pine. Ground-truth photographs of the sites demonstrate that the areas are presently colonized by salt marsh vegetation (*S. alterniflora*, *S. patens*, *D. spicata*); remains of the upland forest surrounded by marsh grasses (Figures 7.26 and 7.27) demonstrate dramatic landward migration of the tidal wetlands.

Steep Adjacent Uplands. Fringing wetland areas laterally adjacent to steeply sloping uplands are characterized by low rates of landward migration of wetland environments, or by loss of wetland areas due to run-off or erosion of upland sediments onto the fringing wetlands. These steeply sloping uplands can be characterized by well-established upland vegetation (oak, holly) directly adjacent to the fringing marsh, or by agricultural land with a plow berm.

Landward lateral migration rates for these steep sites are generally low (Figure 7.25). Some areas experienced loss of wetlands areas (conversion of wetlands to uplands) at the wetland/upland boundary, likely caused by upland sediment runoff onto wetlands due to agricultural activities or by intentional filling.

Examples of this geomorphic type include the fringing wetlands surrounding the Burton agricultural land at the Steeles Cove location; the uplands and wetlands adjacent to the Delaware Wildlands site, Angola Neck; many sections of upland on the western and northern fringes of Rehoboth Bay; and many areas along the southern section of Indian River Bay. These more steeply sloping adjacent uplands are

Figure 7.26 Photograph (1995) of shallowly sloping wetlands/uplands boundary area and evidence of landward movement of wetlands over uplands, western shore of Rehoboth Bay.

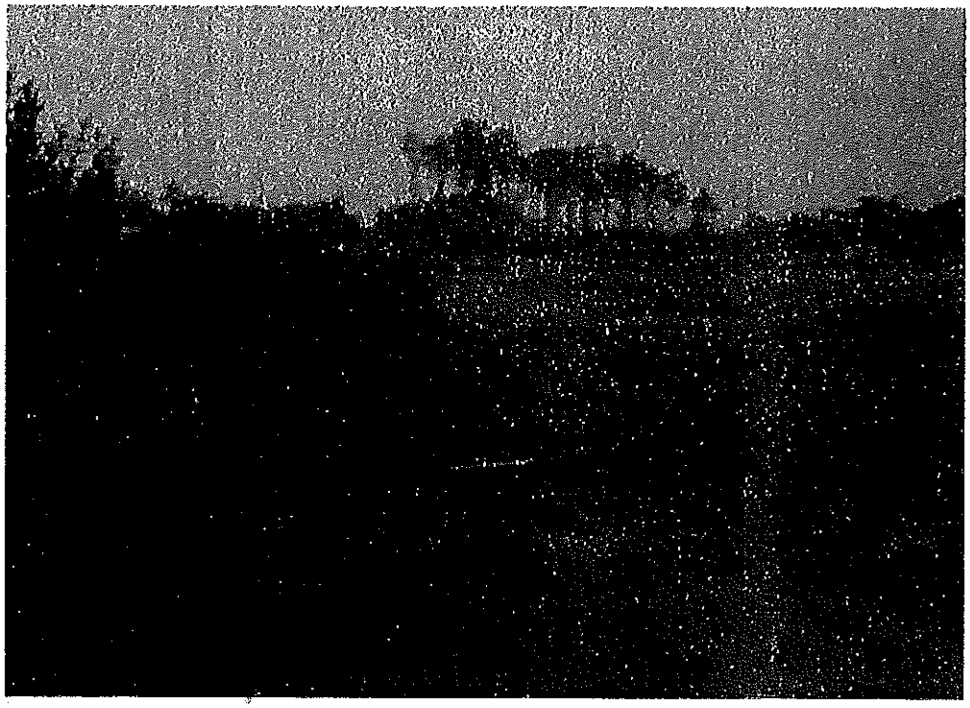


Figure 7.27 Photograph (1995) of shallowly sloping wetlands/uplands boundary area and evidence of landward movement of wetlands over uplands, southern shore of Rehoboth Bay.

the "sides" or lateral portions of pre-Holocene necks (as opposed to the distal portion or "tips" of the pre-Holocene necks which are more shallowly sloping).

Stream Valley Encroachment. In many areas where tidal streams are tributary to Indian River and Rehoboth Bays, landward migration of fringing tidal wetlands has occurred in a headward direction. An example of this is the Pot Nets North study area.

The overall long-term average of lateral wetland migration up stream valley axes is 3.20 m/yr, with average rates ranging from 1.37 to 7.38 m/yr. Although migration rates are relatively high in these tidal streams, the areal extent of new wetlands created is small due to the V-shaped geometry of the stream valley. Instead of new wetlands becoming established over broad low-lying flat areas, wetlands are established within the narrow confines of the pre-existing valley. In addition, these wetlands are likely to be replacing freshwater wetlands (marshes or swamps), not pre-existing upland areas.

Field studies and ground-truthing of the fringing tidal wetlands in stream valleys tributary to the Inland Bays (Figure 7.28) illustrate that *S. alterniflora* or *P. australis* colonize these new areas, rather than *S. patens* which commonly colonizes gently sloping upland sites.

Figure 7.28 Photograph (1995) of fringing tidal salt marsh adjacent to uplands in stream valley axis, Canary Creek marsh.

Landward Migration Rates and Historical Erosion Rates

Landward Migration Rates

Measured Rates of Transgression vs. Calculated Rates of Transgression. Rates of landward migration of fringing tidal wetlands as determined from Metric Map Data are on the order of meters per year, as shown in the composite histogram, Figure 7.25. However, when rates of marsh transgression are calculated based on sea-level rise and measured slopes (rate of sea-level rise/slope), landward migration of wetlands is much lower, on the order of centimeters per year (Figures 7.29 and 7.30).

The difference in the measured and calculated rates is surprising, and may be explained by local relative sea-level rise in the Inland Bays. The calculated rates of landward migration are based on rates of relative sea-level rise of 0.35 and 0.40 cm/yr, which are reasonable predicted averages for the Delaware coast (Lyles *et al.*, 1988). However, because of stabilization of Indian River Inlet, the wetlands in the Inland Bays may have been affected by local relative rates of sea-level rise much greater than 0.35 cm/yr. Tidal range in Indian River Bay has increased 77 cm from 1941 to 1988 (Raney *et al.*, 1990); similarly, tidal range in Rehoboth Bay has increased 38 cm over the same time period. Thus, the actual rate of local relative sea-level rise in Indian River Bay from 1941 to 1988 has been 0.82 cm/yr; in Rehoboth Bay the local relative sea-level rise has been 0.40 cm/yr. When landward migration rates are calculated with measured

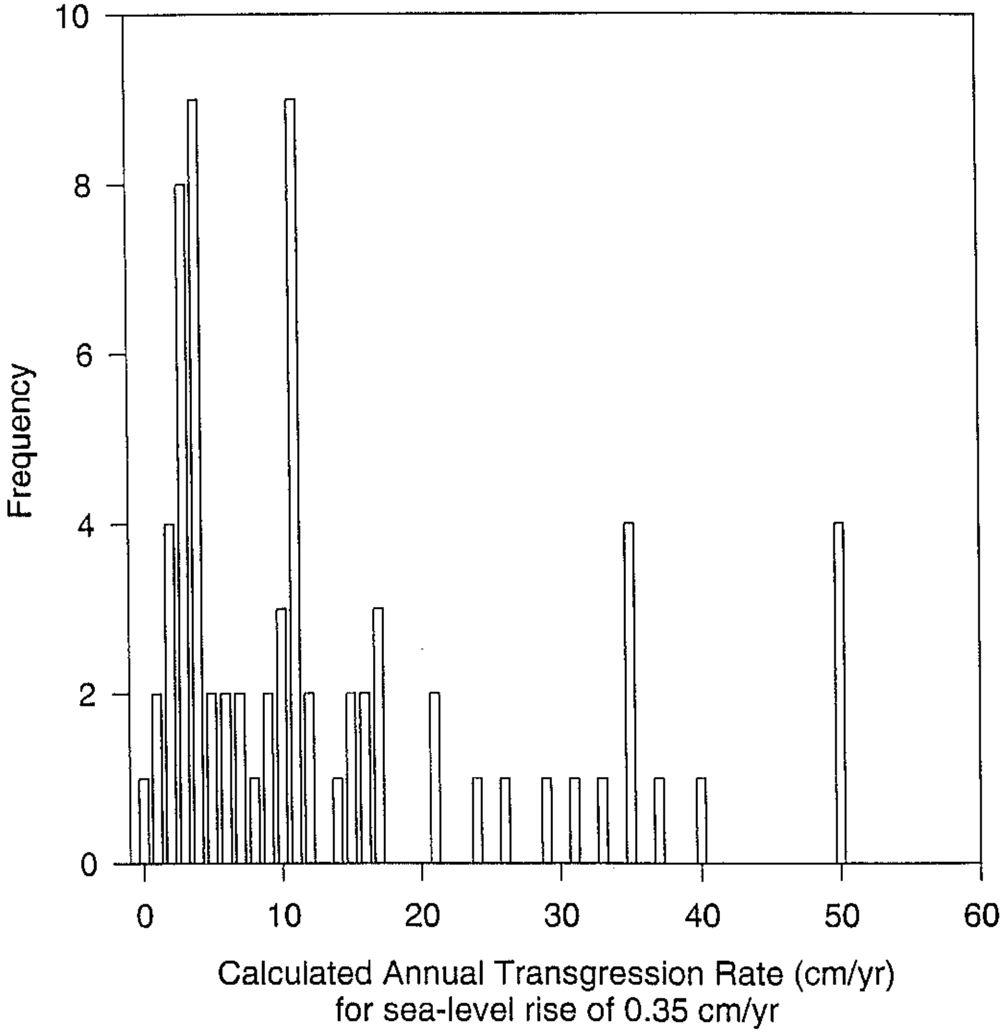


Figure 7.29 Histogram of calculated rates of landward transgression (cm/yr) based on transect slope calculations with a projected sea-level rise of 0.35 cm/yr.

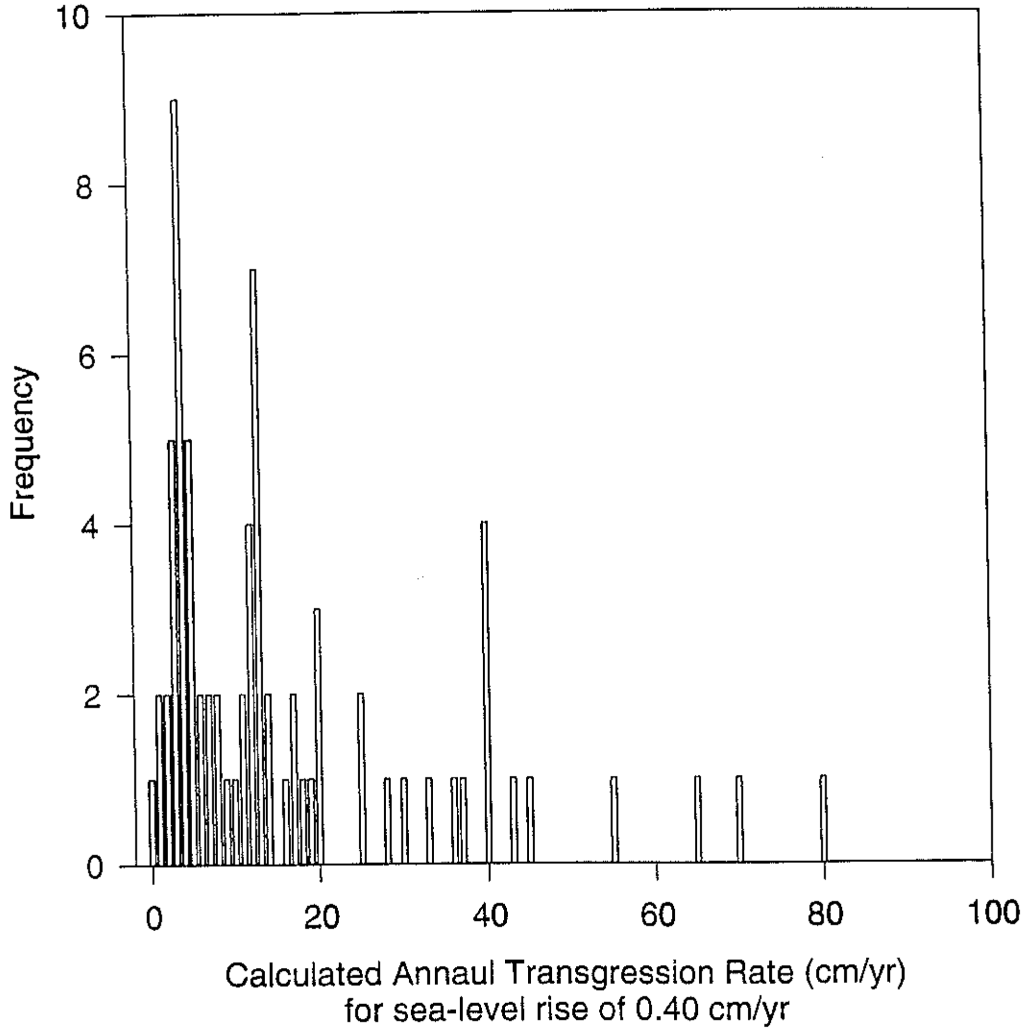


Figure 7.30 Histogram of calculated rates of landward transgression (cm/yr) based on transect slope calculations with a projected sea-level rise of 0.40 cm/yr.

slopes and a rate of sea-level rise of 0.82 cm/yr, the rates of transgression are more similar to the measured rates (Figure 7.31).

Because Metric Maps are only available for the Indian River and Rehoboth Bays, measured rates of landward migration of fringing salt marsh environments may be considered extreme rates which are higher than would be measured in other marshes in Delaware. Rates calculated from measured slopes and projected scenarios of sea-level rise may actually provide more realistic estimates of transgression rates for fringing tidal salt marshes.

Rate of Transgression vs. Rate of Erosion.

Erosion rates have been determined for Delaware Bay and Rehoboth Bay shores (Maurmeyer, 1978; Swisher, 1982; French, 1990), as discussed earlier in this chapter. Similarly, predictions have been made regarding location of Inland Bay shorelines over the next 100-200 years (Figure 7.32).

Figure 7.33 is a histogram of the relationship between the nominal rate of marsh transgression (calculated for a 0.31 cm/yr rise in sea level) and the rate of marsh shore erosion. The diagram clearly shows that erosion rates exceed the rate of transgression, suggesting that over time, areal extent of fringing wetland environments will decrease. The time frame within which Delaware's fringing tidal wetlands will disappear depends on the slope of adjacent upland surfaces, the rate of sea-level rise, and the rate of erosion. This time frame will be examined in the integration chapter.

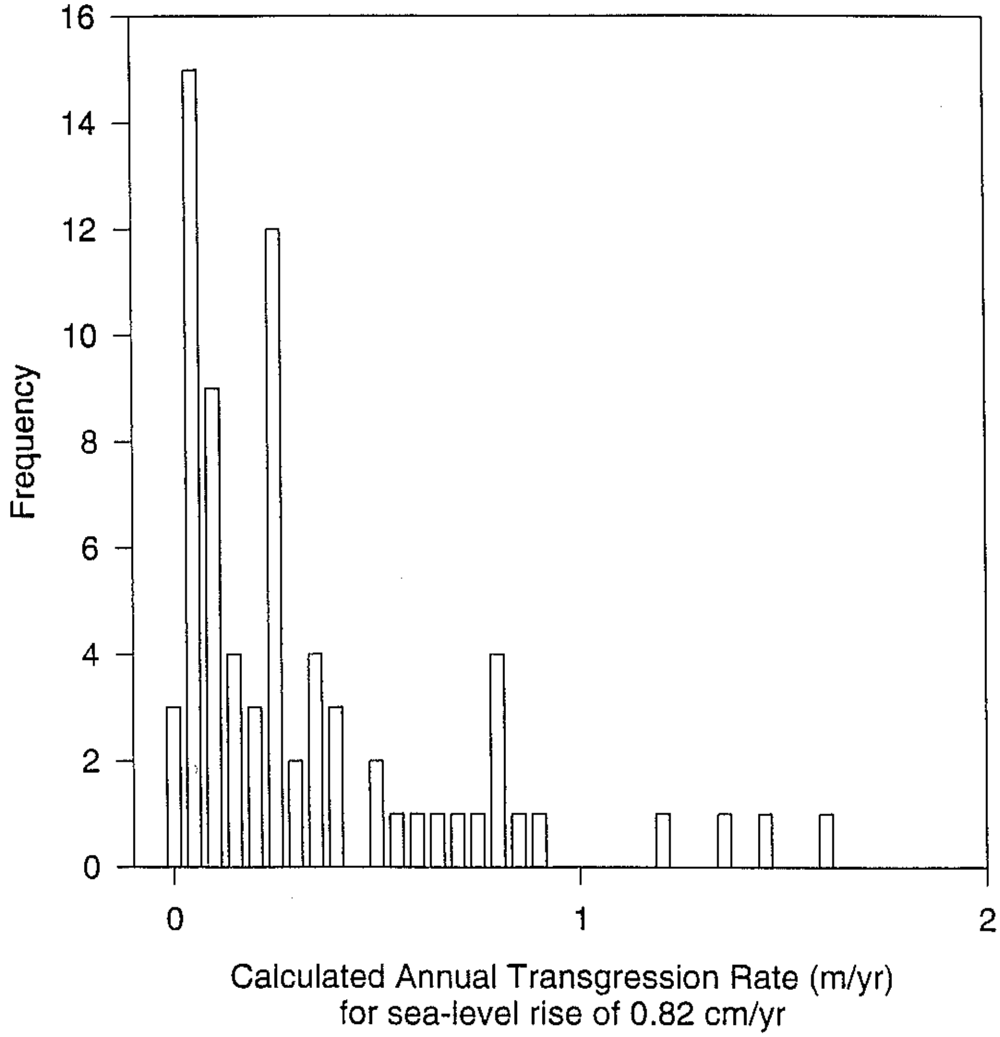


Figure 7.31 Histogram of calculated rates of landward transgression (cm/yr) based on transect slope calculations with a projected sea-level rise of 0.82 cm/yr.

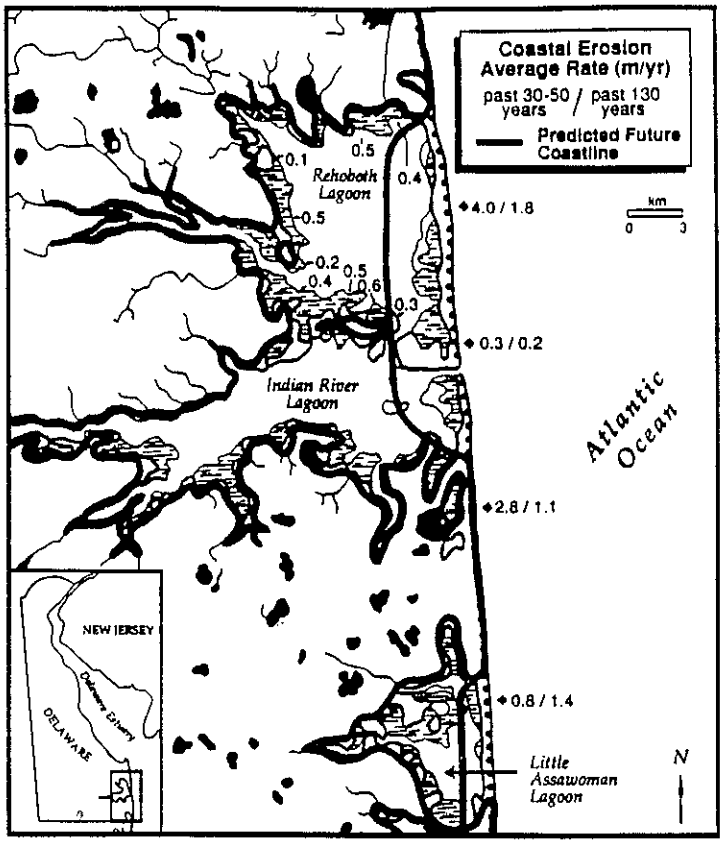


Figure 7.32 Salt marshes along the Inland Bays of Delaware showing rates of shoreline erosion over the past 100 years (based on Swisher, 1982). Heavy lines indicate projected shorelines 200 years into the future (from Kraft *et al.*, 1992).

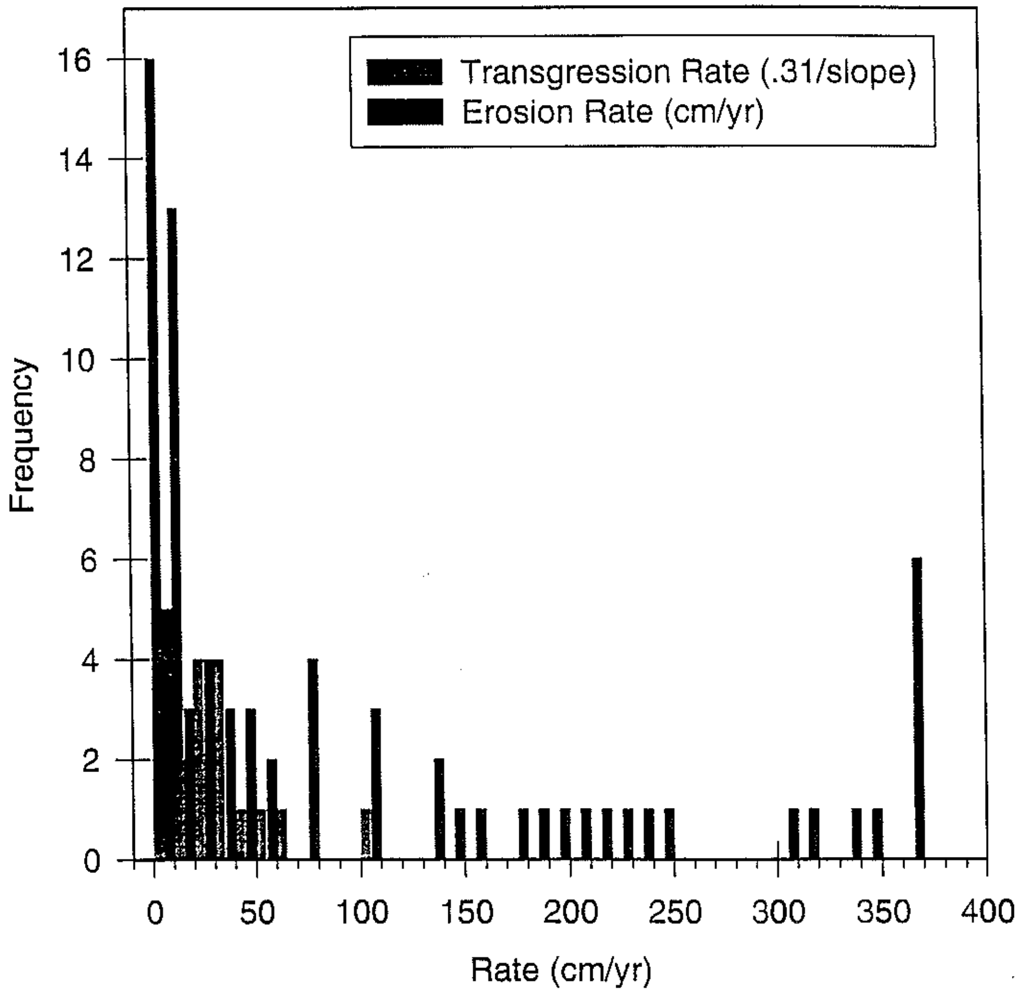


Figure 7.33 Histogram of calculated nominal rate of marsh transgression and rate of marsh shoreline erosion.

Summary and Conclusions

Erosion of Wetland Shores

1) From 1882-1977, wetland shorelines of Delaware Bay have a history of erosion, averaging -5.24 m/yr in the northern section, -1.46 m/yr in the central section, and -0.52 m/yr in the southern section, while during the time period 1954-1969, overall erosion rate for the southern section was -1.4 m/yr.

2) From 1938-1981, shore erosion rates for Rehoboth Bay averaged -0.6 m/yr or less, with measured erosion rates ranging from -0.2 to -1.6 m/yr over varying time periods. However, a high degree of spatial and temporal variability is characteristic of these erosion rate measurements.

3) Accelerating rates of local relative sea-level rise appear to be the driving mechanism causing marsh erosion (shoreline recession). Rate of sea level rise over the past century (0.31 cm/yr) is more than double the rate of the past 2,000 years (0.12 cm/yr) (Belknap, 1975), the period during which these marshes formed. Anthropogenic impacts (such as inlet stabilization, mosquito ditching, and dredging) have contributed to erosion processes by changing water levels in these bays over the past 70-100 years; the response to increased water levels is erosion at the bayward edge of the marsh. A more immediate reason for these high rates of erosion is the general deficiency of sediments in the area combined with the nature of the eroding shoreline, i.e. the vertical

scarp which makes accretion unlikely except in the presence of significant amounts of sediment.

3) The process of wetland shore erosion, characterized by undercutting and slumping (as described by Maurmeyer, 1978; Swisher, 1982; French, 1990) results in development of a vertical scarp at the wetland/water interface; this precludes the process of accretion in the absence of high sediment input, and minimizes the likelihood of a short-term repair mechanism for erosion.

4) Erosion of wetland shores is related to shoreline type and local energy conditions. As French (1990), Swisher (1982), and Maurmeyer (1978) concluded, marsh shores erode faster than beach shores; additionally, marsh shorelines appear to erode in a relatively continuous pattern through time.

Landward Migration of Fringing Tidal Wetlands

1) Average rates of transgression of wetlands onto upland surfaces measured from Metric Maps range from +0.25 to +1.85 m/yr.

2) Patterns of transgression of wetlands over adjacent uplands clearly indicate that lateral migration rates are primarily controlled by slope of adjacent uplands. Three distinct model categories have been identified based on geomorphic response and transgression rates:

(a) Low slope areas: These sites have experienced the greatest changes in landward migration of the wetlands/uplands boundary. From 1944-1989, mean landward migration rates of shallowly sloping areas in Indian River and Rehoboth Bays were +1.60 m/yr and +1.85 m/yr, respectively.

(b) Steep slope areas: These sites are characterized by either low rates of landward migration of wetland environments or by loss of wetland area due to encroachment of uplands over wetlands. From 1944-1989, mean landward migration rates of steeply sloping areas in Indian River and Rehoboth Bays were +0.44 m/yr and +0.25 m/yr, respectively.

(c) Stream valley axes: These V-shaped valley areas are characterized by rapid movement of fringing salt marshes in a headward direction, with average rates ranging from +1.39 m/yr to +5.00 m/yr. However, pre-existing geometry confines newly established wetlands to the narrow stream valley.

3) Long-term transgression rates observed at each of the sites can be attributed to specific and varying combinations of antecedent geometry and tidal range, amplified by human impacts (inlet stabilization).

Future Areal Extent of Fringing Wetlands: Rates of Erosion vs. Rates of Transgression

1) Future areal extent of fringing tidal wetlands is based on accretion or aggradation via landward transgression combined with loss via bayside (seaward) erosion.

2) Results of this study show that while measured rates of landward accretion in the Inland Bays range from +0.25 to +1.85 m/yr, calculated rates based on measured slopes and various scenarios of sea-level rise are much lower, on the order of

centimeters per year, ranging from +0.20 m/yr to +0.60 m/yr. Measured erosion rates for the Inland Bays average -0.6 m/yr (ranging from -1.6 to -0.2 m/yr), while much greater rates (-5.24 to -0.52 m/yr) are characteristic of Delaware Bay marsh shorelines.

3) It is predicted that the net long-term result will be a net decrease in wetland area, as rates of erosion outpace rates of landward migration of fringing tidal salt marshes.

Chapter 8

INTEGRATION: DELAWARE'S FRINGING TIDAL SALT MARSHES, LOCAL RELATIVE SEA-LEVEL RISE, VERTICAL ACCRETION, SHORE EROSION, AND ANTECEDENT TOPOGRAPHY

Introduction

The continued existence of fringing tidal marshes in Delaware ultimately depends upon several interacting factors, including rate of local relative sea-level rise, vertical accretion rates, accommodation space afforded by antecedent topography (adjacent upland slopes), rates of shore erosion, and human impacts, not only on physical processes affecting tidal wetlands, but also coastal management policies. Although previous research has suggested that rapid rises in sea level may result in salt marshes being unable to keep pace with sea-level rise and concurrent large-scale loss of salt marshes, the impacts of increasing inundation on landward movement of wetland vegetation zones must also be considered and quantified. Landward migration of wetland environments is most likely to occur in low-lying freshwater tidal (or non-tidal) marsh areas and/or along low-lying and gently sloping adjacent upland surfaces. A number of coastal watersheds in Delaware currently have significant areas of freshwater wetlands mapped at elevations just above the elevations of tidal salt marshes; these areas consist primarily of freshwater tidal marshes along the axes of rivers or streams, such as the Murderkill, Mispillion, or Broadkill Rivers, or along Cedar or Prime Hook Creeks. It is likely that tidal salt marshes will transgress landward within these watersheds at the expense of coastal freshwater wetland areas. However, in many locations, factors of coastal slopes and scarps, as well as human induced changes in tidal range, sediment supply, and shoreline characteristics may reduce these landward processes and rates.

This chapter addresses these general factors in conceptual model scenarios which consider the future areal extent of Delaware's fringing tidal wetlands, as well as vegetative zonation patterns of future wetlands. In addition, the 'micro-scale' stratigraphy, survey data, and conceptual models obtained in this study are incorporated into previously described large scale stratigraphic patterns and history of coastal marshes, especially as they relate to antecedent geometry. Finally, this chapter addresses how human induced changes to physical processes may result in significant changes in wetland vegetative distribution, zonation, erosion, and landward migration rates.

General Stratigraphic Relationships

Coastal Paralic Environments and Fringing Wetland Stratigraphy: The Past 10,000 Years

The record of geologic evolution and development of Delaware's coastal tidal salt marshes relative to other coastal paralic environments has been preserved in subsurface sediments and has been described in stratigraphic cross-sections. Previous investigations of Delaware's coastal paralic environments and subsurface stratigraphy documents existence of past tidal marshes and indicate that a relationship exists between antecedent topography and morphology of paralic subenvironments (Kraft *et al.*, 1976; Belknap and Kraft, 1977; John, 1977; Belknap and Kraft, 1985; Oertel and Kraft, 1994). The geologic record demonstrates that tidal salt marshes have existed in Delaware's coastal areas for the past 2,000 years, and their three-dimensional geometry is controlled by antecedent topography. When combined with detailed information regarding the nature and geometry of the present wetlands/uplands boundary obtained in this study, predictions can be made regarding future temporal and spatial distribution of fringing tidal wetlands in Delaware.

In the geologic record, fringing tidal marsh deposits are typically restricted to areas along margins of numerous tidal channels and estuaries (e.g., antecedent stream valleys), along the mainland shoreline (wetlands/uplands boundary), and/or over broad flat areas (e.g. former interfluve/tidal flat areas) (Oertel and Kraft, 1994). Surficial environments are similarly affected; the relationship between mainland topography (interfluves/headlands) and drainage systems on adjacent coastal environments is depicted in Figure 8.1, a diagram of Delaware's coastal barrier system (e.g., Indian River and Rehoboth Bays) and adjacent Atlantic coastal plain (Pleistocene marine and paralic sediments). The diagram clearly indicates the irregular nature of the mainland margins of Indian River and Rehoboth Bay lagoon complex, due to submerged stream valleys. Previous work (Kraft,1971a; Chrzastowski, 1986; others) has shown that the paleovalleys of the ancestral Love Creek, Herring Creek, and Indian River extend under

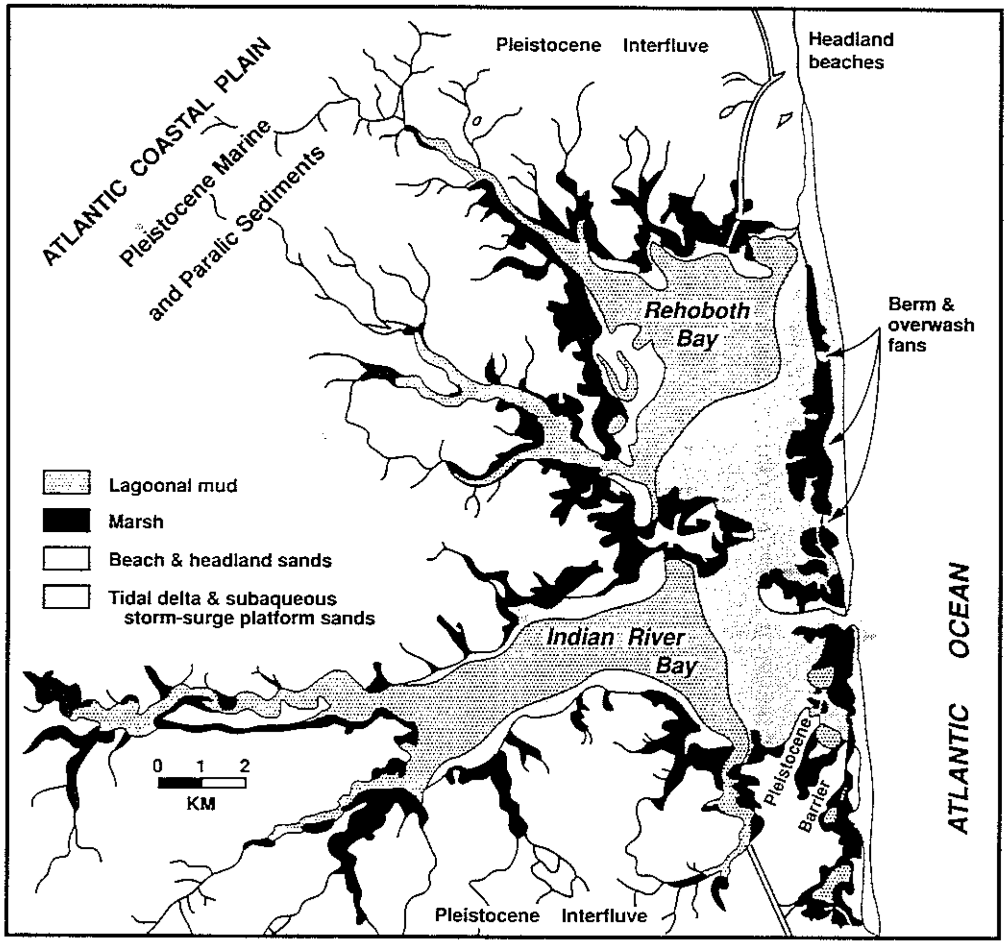


Figure 8.1 Map of the northern Delaware coast showing the interfluve/headland-beach relationship, and the drainage system/barrier-beach relationship (from Oertel and Kraft, 1994).

the barrier spits and eventually connect to the main thalweg of the ancestral Delaware River system under the Delaware shoreface. As depicted in Figure 8.2, ancestral stream valleys connect to the ancestral Delaware River along the entire length of the northern Delmarva peninsula (Oertel and Kraft, 1994). The submergence of these stream valleys during the Late Holocene transgression has produced an irregular shoreline along the margins of associated lagoons; these stream valleys and irregular shorelines presently exert significant control on the distribution and areal extent of fringing tidal marshes.

There has been considerable stratigraphic research conducted in barrier/lagoon systems along the Delmarva peninsula which provides data on spatial and temporal evolution and development of coastal salt marshes. Many of these studies (Kraft 1971a; Belknap and Kraft, 1985) emphasize the importance of topographic characteristics which interact with marine processes to produce morpho-stratigraphic patterns of coastal barrier systems (Oertel and Kraft, 1994), and illustrate the importance of interaction between depositional environment and antecedent topography. During periods of relatively constant rates of sea-level rise, variations in inundation of upland surfaces may be affected by slope, elevation, drainage density, drainage type, and drainage orientation. For example, within Delaware's coastal barrier lagoon systems, antecedent topography and associated drainage system (ancestral stream and river valleys) are submerged by coastal inundation; these relationships have been mapped based on stratigraphic interpretation of cores and seismic records (Figure 8.3). In making stratigraphic interpretations, basal peat deposits found in the buried shoreface are used as markers for tracing the position of the mainland fringe marsh during

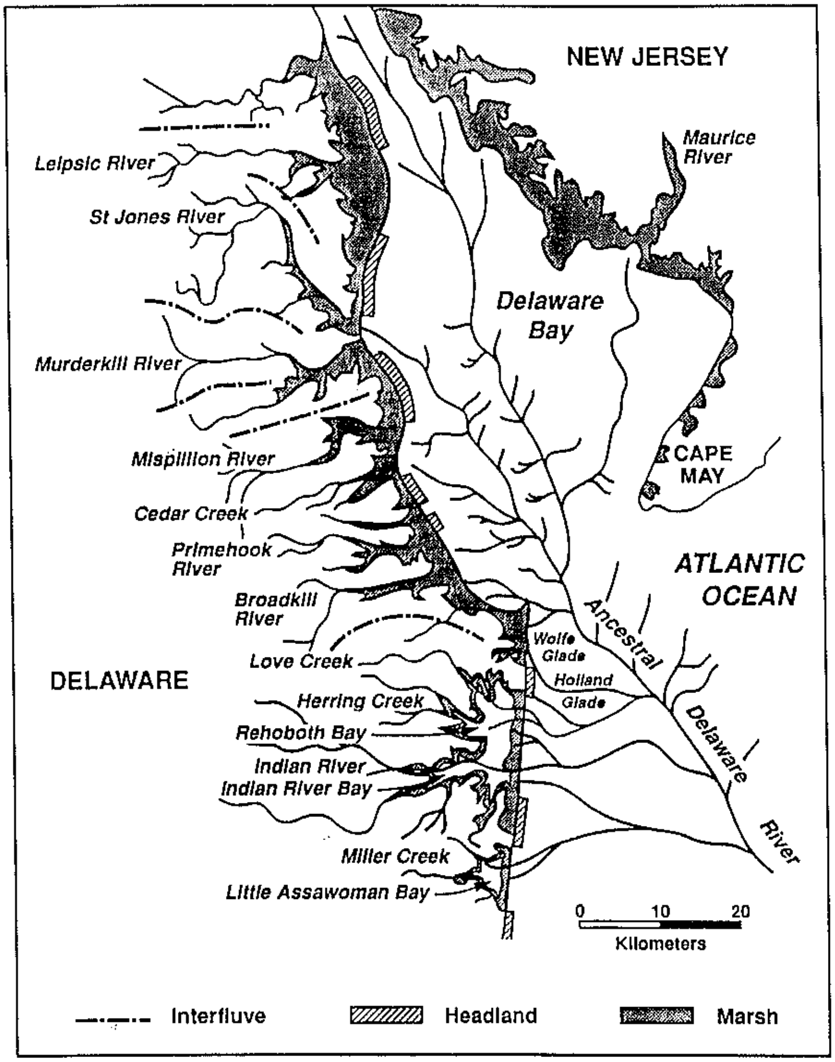


Figure 8.2 Map of the northern Delmarva peninsula showing the ancestral Delaware River and the trace of low-order streams that cross the present shoreline and connect with channel fills of the transgressed Delaware drainage system (from Oertel and Kraft, 1994; based on Kraft, 1971a; Belknap and Kraft, 1977, 1981; and Fletcher *et al.*, 1990).

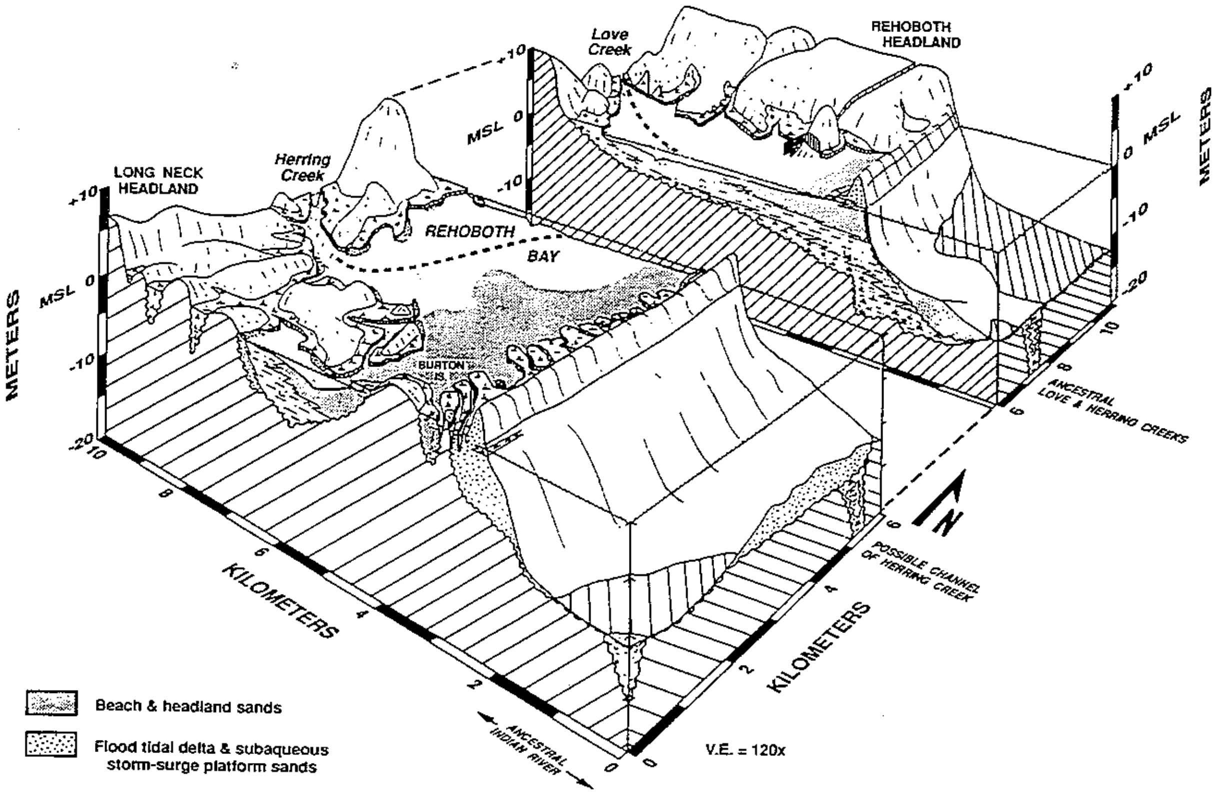


Figure 8.3 Diagrammatic block diagram of the northern Delmarva Peninsula showing the drainage pattern of the buried, late Holocene pre-transgressed land surface (from Oertel and Kraft, 1994; after Belknap and Kraft, 1985).

transgression (Oertel and Kraft, 1994). These fringing marsh deposits may be relatively continuous or spatially discontinuous, occurring in fragmented segments along margins of numerous tidal channels and estuaries, around lagoonal islands, and along the mainland shoreline, as depicted in Figures 8.4 and 8.5 (Oertel and Kraft, 1994).

The control exerted by antecedent topography on distribution of fringing wetland environments is clearly documented. In Figure 8.5, for example, the dated core logs extending 12 to 15 meters below mean low sea level clearly illustrate how the areal extent of fringing tidal wetlands is controlled by antecedent topography. As fringing marsh muds and peats developed from 7075-5640 years before present, landward movement of fringing wetland environments was truncated by steep Pleistocene antecedent topography. Only after relative local sea-level had reached an elevation above the pre-existing vertical Pleistocene scarp (at 5640 years before present) did fringing wetlands migrate in a landward direction.

Fringing Wetland Stratigraphy: The Past 100-200 Years

Similar patterns are observed in stratigraphic relationships mapped at the wetlands/uplands boundary in this study at a much more detailed scale. The transgressive nature of fringing salt marsh stratigraphy at the wetlands/uplands boundary has been described in previously (Chapter 5). When time lines are drawn on surveyed cross-sections, correlations between antecedent topography and geometry and areal extent of encroaching salt marsh environments are clearly illustrated.

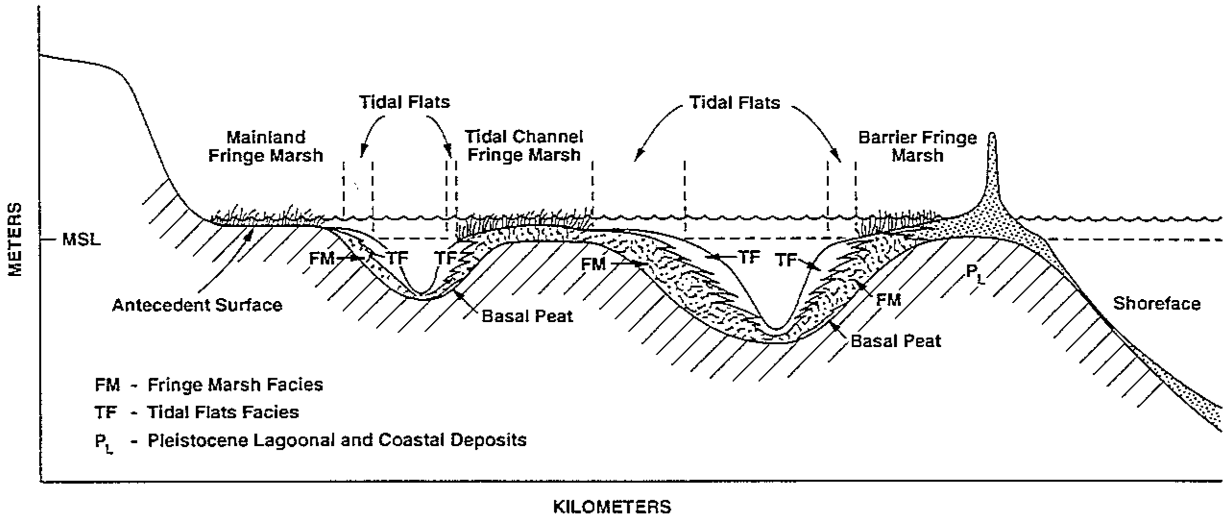


Figure 8.4 Cross-section of an idealized coastal lagoon showing how the thick sequences of fine-grained sediment (tidal flat facies and fringe-marsh facies) are related to depressions in the antecedent topography. Note that although marsh facies may occur at similar depths (at adjacent valleys), the facies are not laterally continuous because of the intervening interfluvial high (from Oertel and Kraft, 1994).

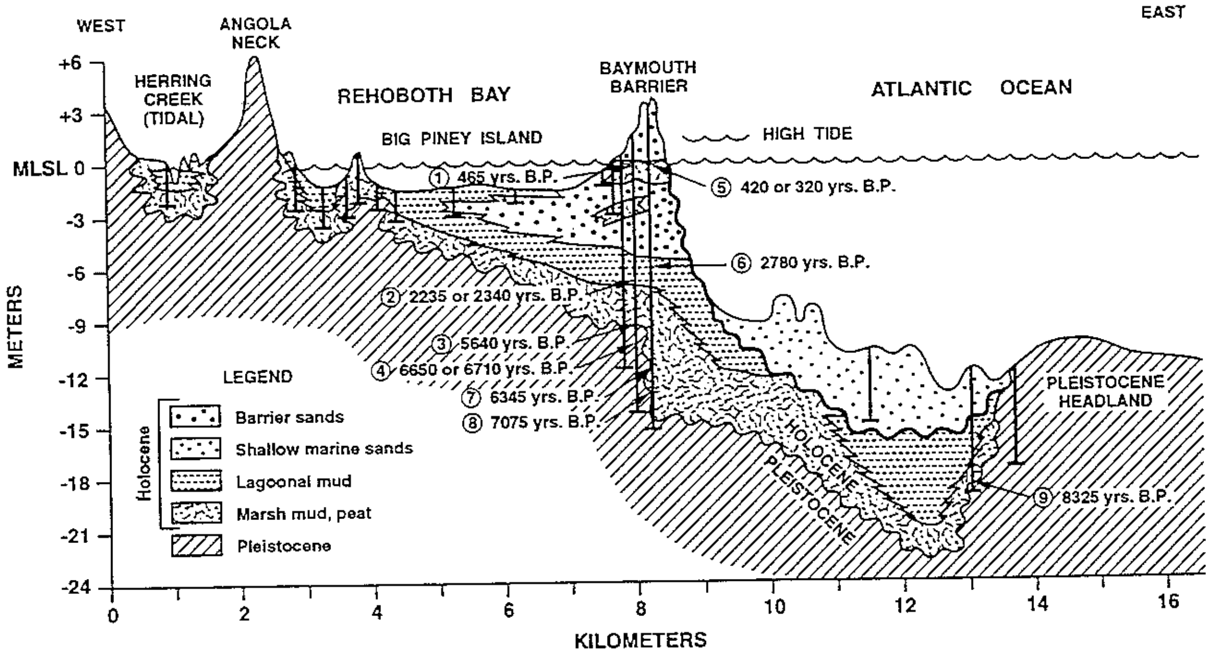


Figure 8.5 Cross-section of the Rehoboth Bay lagoon showing an interpretation of the stratigraphy and lithosomes (from Oertel and Kraft, 1994; after Kraft *et al.*, 1973).

Figure 8.6 is a stratigraphic cross-section of the Boat House Cove, a steeply sloping site, with time lines drawn according to vertical accretion rates as determined by radionuclide analyses (Chapter 6). Time lines are shown at -80 cm (approximately 190 years before present) and -60 cm (approximately 143 years before present). At 190 years before present, the *Spartina patens* wetland area was greater than 23 m wide; however, by 143 years before present, the *S. patens* wetland was less than 2 meters wide. This clearly illustrates the relationship between antecedent topography and the distribution of tidal wetland zonation patterns: a broad *S. patens* high marsh environment existed where the pre-transgressive surface was gently sloping and where the depth to pre-transgressive surface was approximately 0.5 m or less. Note that as the migrating wetlands encountered the more steeply sloping adjacent upland (at 143 years before present), the areal extent of the *S. patens* marsh was reduced in width.

This cross-section also illustrates the effect of a steeply sloping pre-transgressive surface on wetland migration rates. From 190 years before present to 143 years before present (47 year time frame), the wetlands migrated only 5 meters over a steep pre-transgressive slope, at an average annual rate of 0.06 m/yr. Similarly, from 143 years before present to present, the wetland landward migration rate approximated 0.07 m/yr.

Figure 8.7 depicts time lines on the stratigraphic cross-section of the intermediate sloping Delaware Wildlands site. A ^{210}Pb determined vertical accretion rate of 0.34 cm/yr results in time lines drawn at -63 cm (185 ybp), -38 (112 ybp), and -18 cm (53 ybp), while a vertical accretion rate of 0.52 cm/yr results in: -63 cm (121

Figure 8.6 Cross-section with time lines: Boat House Cove study site, northern River Bay.

Boat House Cove - 2

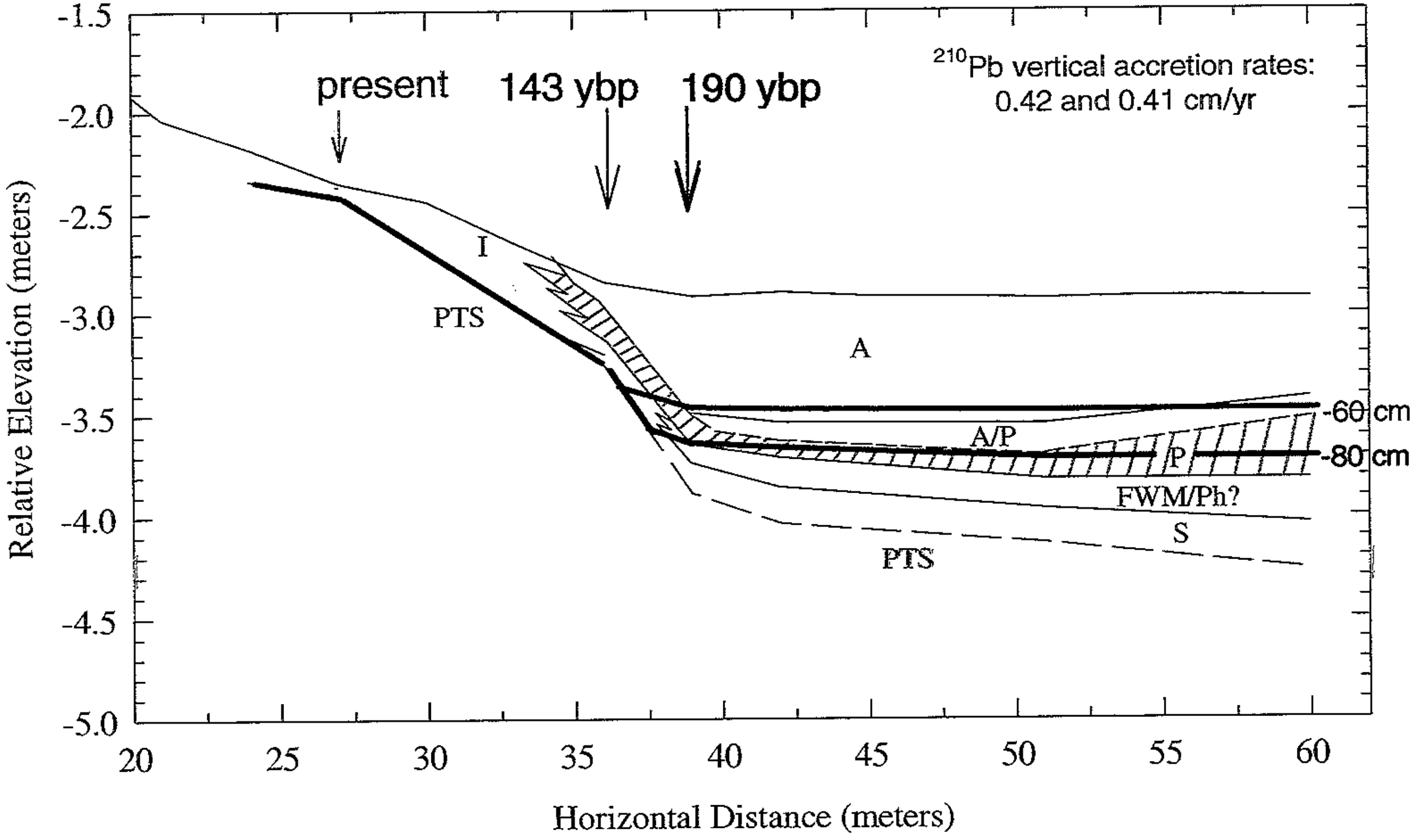
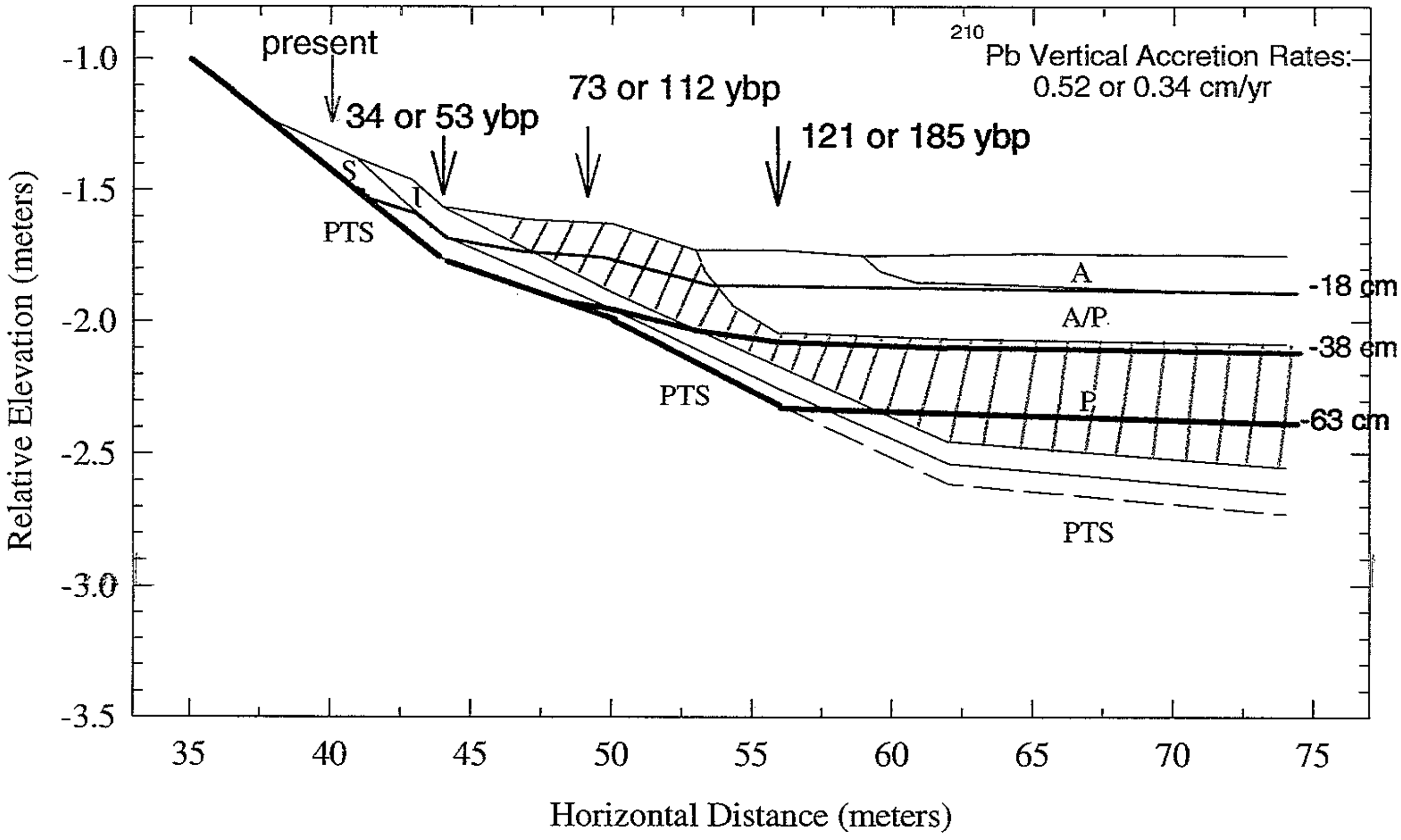


Figure 8.7 Cross-section with time lines: Delaware Wildlands study site, western Rehoboth Bay.

Delaware Wildlands - 2



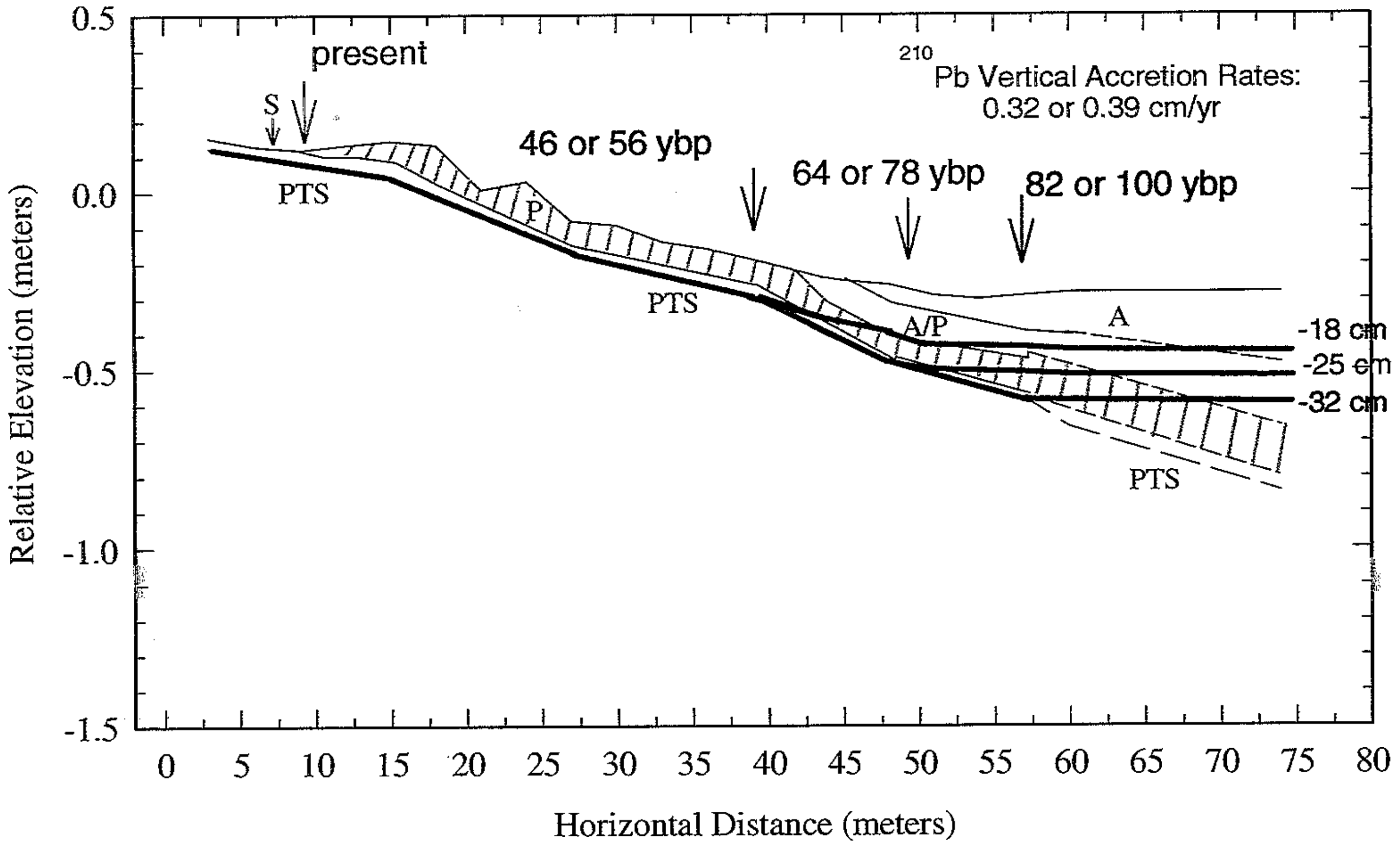
ybp), -38 (73 ybp), and -18 cm (34 ybp). Calculated landward migration rates at this site from 185 years before present to present are 0.08 m/yr which are similar to those calculated from the Boat House Cove cross-section and to the nominal transgression rates calculated for known slopes and sea-level rise scenarios. Rates of 0.08 m/yr are much lower than the transgression rates measured from metric maps for Indian River and Rehoboth Bays, suggesting that anthropogenic effects may have impacted transgression rates in the Inland Bay fringing wetlands.

Figure 8.8 depicts the cross-section for a low slope site, Pot Nets North-5. A ^{210}Pb determined vertical accretion rate of 0.32 cm/yr results in time lines drawn at -18 cm (100 ybp), -25 cm (78 ybp) and -32 cm (56 ybp), while an estimated vertical accretion rate of 0.39 cm/yr results in the following time lines: -18 cm (82 ybp), -25 cm (64 ybp) and -32 cm (46 ybp). Calculated landward migration rates of fringing tidal wetlands at this site for an estimated vertical accretion rate of 0.39 cm/yr are as follows: 82-64 years before present: rate=0.36 m/yr; 64-46 years before present: rate=0.44 m/yr. If a vertical accretion rate of 0.32 cm/yr is used as an estimate, landward migration rates of fringing tidal wetlands at this gently sloping site are as follows: 100-78 years before present: rate=0.30 m/yr; and 78-56 years before present: 0.36 m/yr. These rates of transgression are four to five times greater than those calculated for more steeply sloping sites.

These data clearly demonstrate that primary consideration must be given to local and regional geomorphic control, specifically substrate and subsurface geometry,

Figure 8.8 Cross-section with time lines: Pot Nets North study site, south western Rehoboth Bay.

Pot Nets North - 5



when developing models for prediction and quantification of future areal extent of salt marshes.

Future Areal Extent of Delaware's Fringing Tidal Wetlands

Landward Migration of Fringing Tidal Wetlands and Slope of Adjacent Uplands

Although previous studies have demonstrated that a geometric relationship exists between vertical accretion in wetlands and marsh encroachment onto adjacent uplands (Redfield, 1972; Orson *et al.*, 1985; Phillips, 1986), details concerning adjacent upland slopes and rates of landward migration of salt marshes have not been established. Phillips (1986) developed a simple model which describes the relationship between shore erosion, slope of the upland surface, and accretion rates necessary to maintain a constant marsh area. Similarly, for any given increase in the rate of local relative sea-level rise, predictions can be made regarding rates and areal extent of landward migration of fringing tidal wetlands. A simple diagram (Figure 8.9) illustrates the basic geometric relationship between upland slope, water level increases, and landward migration. Landward movement of wetland environments will occur only if marsh vertical accretion can keep pace with increases in water level. Submergence will occur if rates of vertical accretion do not maintain marsh surface elevations relative to increasing water levels.

Application of these general geometric relationships can be readily illustrated by using data from two transect elevation surveys made for this study. The sites chosen for illustrative purposes include Bay Vista-1 site, with a steep adjacent

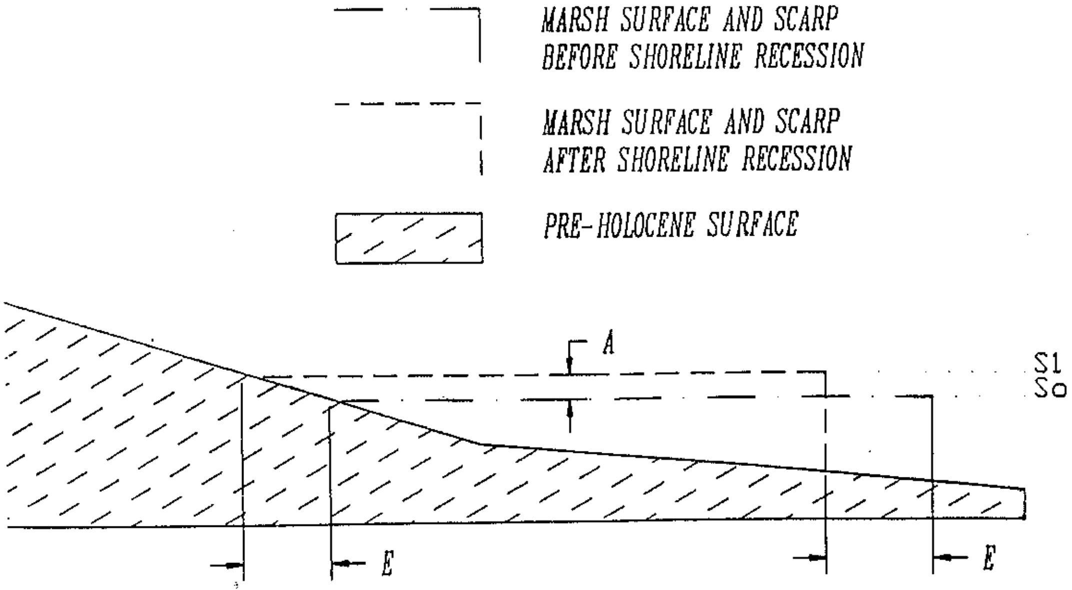


Figure 8.9 Conceptual model showing surface accretion necessary to maintain marsh area for a given rate of shoreline erosion. If the distance from the upland intercept of the marsh to the land/water interface is to remain constant during a shoreline recession of magnitude E, then surface accretion of A is necessary (from Phillips, 1986).

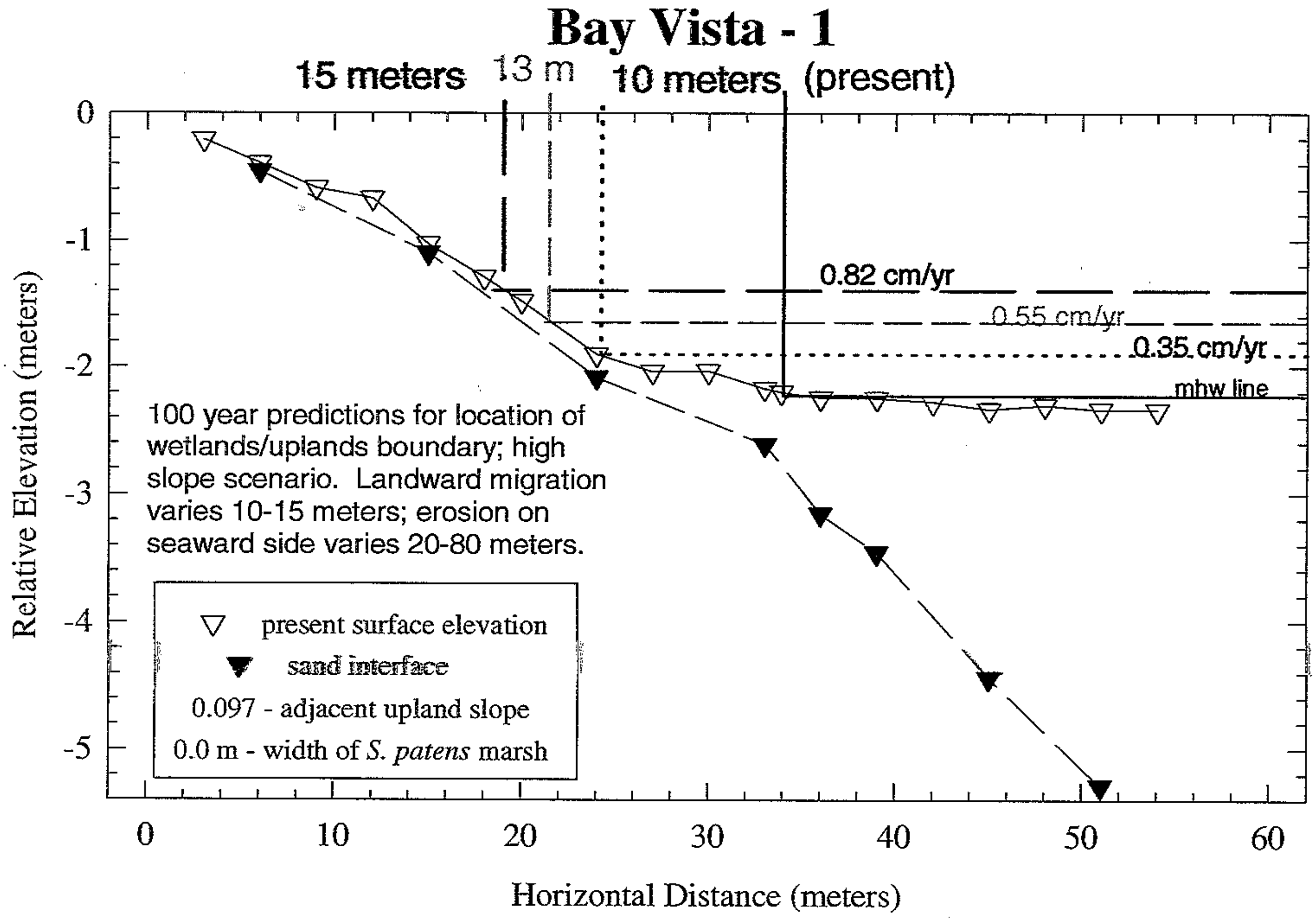
upland slope (0.097), and the Pot Nets North-5 site, with a gently sloping adjacent upland (0.005). For purposes of illustration, it is assumed that the rate of vertical accretion can maintain marsh surface elevation relative to sea-level increases. Based on U.S. E.P.A. (Titus and Narayanan, 1995) estimates, projected water level increases are made for the next 100 years, including low, medium, and high rate projections (0.35 cm/yr, 0.55 cm/yr, and 0.82 cm/yr, respectively).

Surface and subsurface antecedent topography and profile data for Bay Vista-1, a steeply sloping upland site, are depicted in Figure 8.10. Over the next 100 years, total landward migration of wetland environments at this site will range from 10 to 15 meters. If concurrent erosion of the marsh at the bayward edge occurs at rates varying from 0.2 to 0.8 m/yr (20-80 m total over the next 100 years), the areal extent (width) of the fringing salt marsh will diminish over time.

In contrast, Figure 8.11 depicts a similar scenario for a gently sloping site, Pot Nets North-5, with 100 year projections of water level increases of 35, 55, and 82 cm. Concurrent landward migration of fringing wetlands over adjacent low sloping upland ranges from 25 to 57 to 82 meters, respectively. If the similar rates of marsh fringe erosion are applied (0.2 to 0.8 m/yr, or 20-80 m total over 100 years), the areal extent of the wetland adjacent to this low sloping upland will generally be maintained over time.

Vegetative Zonation Patterns and Geometry of Antecedent and Adjacent Upland Surfaces. Surficial zonation patterns can also be predicted based on the slope of adjacent upland surfaces. Results of this study show a relationship between

Figure 8.10 100 year predictions for location of wetlands/uplands boundary, Bay Vista study site: high slope scenario. Landward migration varies 10-15 meters; erosion on seaward side varies 20-80 meters.

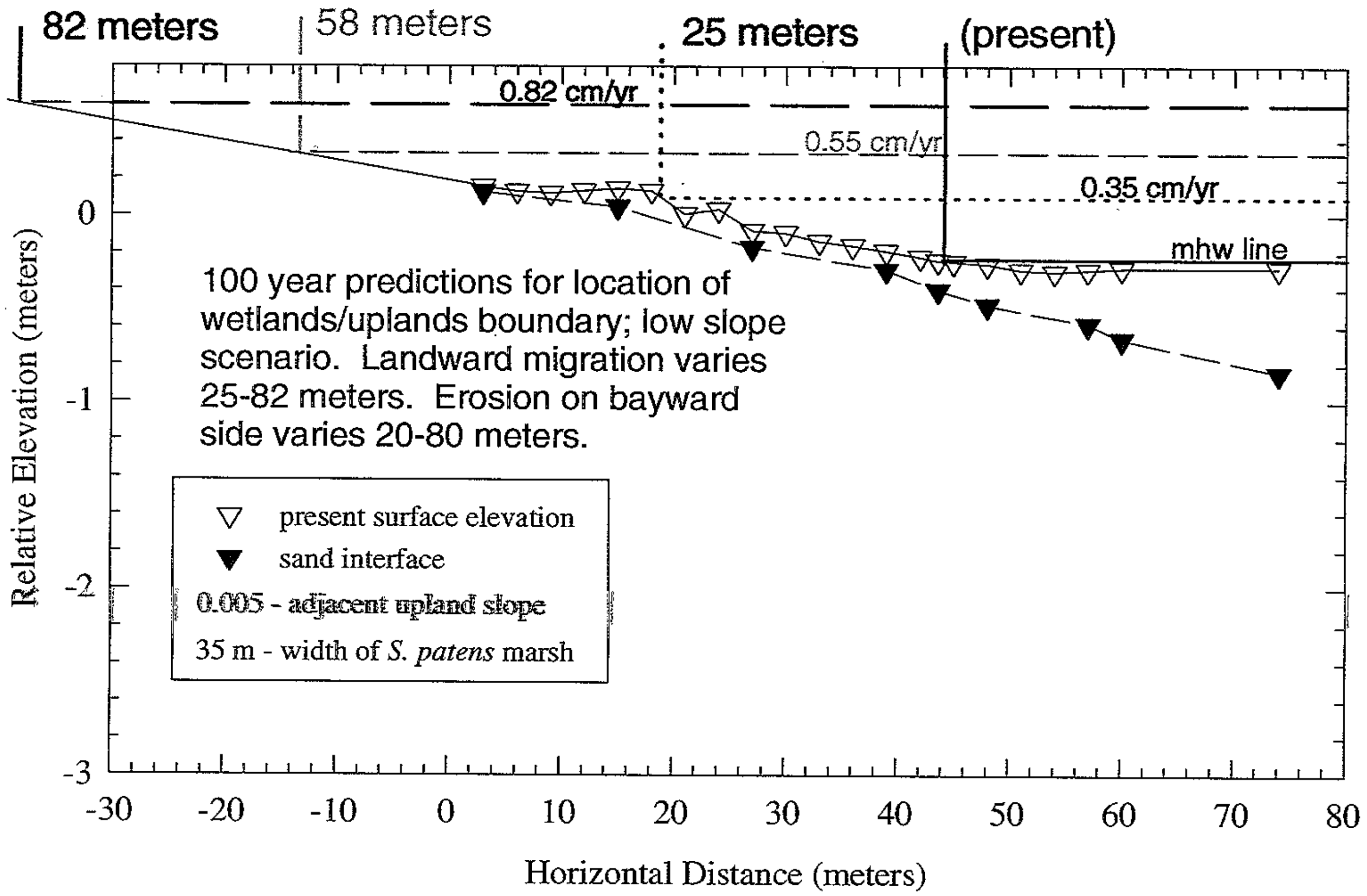


Bay Vista - 1

100 year predictions for location of wetlands/uplands boundary; high slope scenario. Landward migration varies 10-15 meters; erosion on seaward side varies 20-80 meters.

Figure 8.11 100 year predictions for location of wetlands/uplands boundary, Pot Nets North-5 study site: low slope scenario. Landward migration varies 25-82 meters; erosion on seaward side varies 20-80 meters.

Pot Nets North - 5



depth to pre-transgressive sandy sediments and distribution of surface vegetation in fringing salt marshes. *S. patens* is characteristically restricted to areas where depth to pre-transgressive sands is less than 40 cm, while *Spartina alterniflora* vegetation characterizes wetland/upland boundary areas where antecedent and adjacent slopes are steep, and a shallow sandy sediment 'platform' is not found at the base of the upland (depth to pre-transgressive surface is generally > 40 cm). Thus, as water levels increase and landward migration of salt marshes occurs, *S. patens* high marsh environments are likely to be found only on gently sloping upland surfaces. *S. patens* vegetation is not likely to be found in future wetland areas established over and adjacent to steeply sloping upland surfaces.

In contrast, as adjacent upland surfaces are flooded, *S. alterniflora* dominated wetlands are likely to be found at the base of steeply sloping uplands as these surfaces are flooded. For example, along margins of tidal streams and rivers, where Metric Map data indicated greatest rates of landward migration, steep antecedent geometry precludes development of *S. patens* marshes at these locations. A present example of this scenario is shown in Figure 8.12, a photograph of a fringing tidal salt marsh characterized by *S. alterniflora* vegetation within and adjacent to a pre-existing stream valley.

Model Scenarios

The future areal extent of coastal tidal wetlands in Delaware is dependent on several factors, including slope of adjacent upland, rate of marsh shore erosion, and

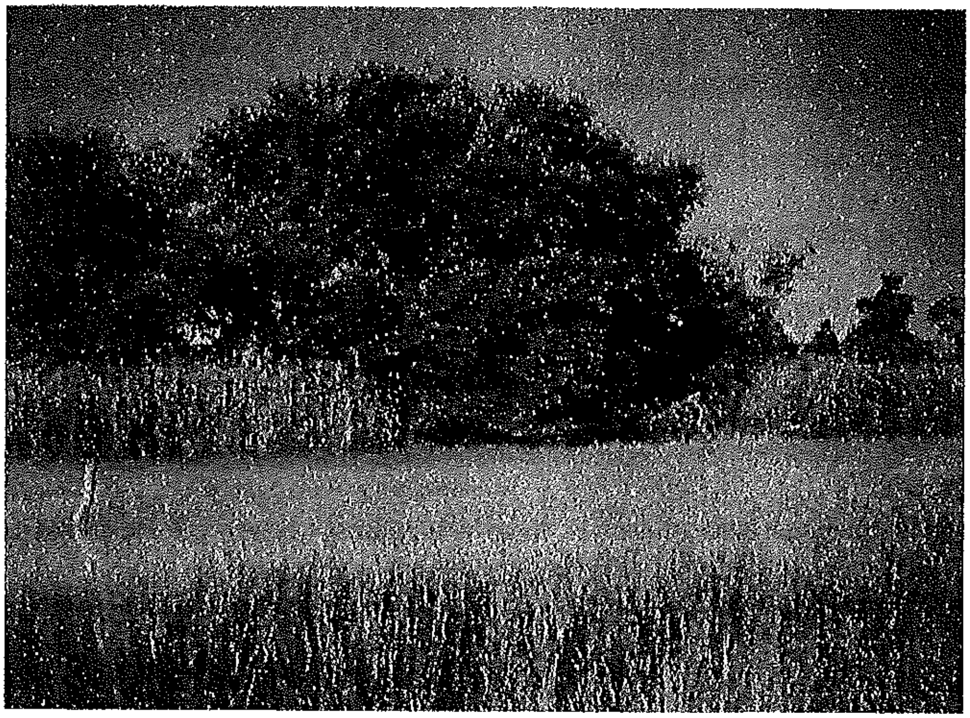


Figure 8.12 Photograph depicting wetland zonation pattern adjacent to steeply sloping adjacent upland in stream valley axis area; note *S. alterniflora* vegetation at base of upland slope, and absence of *S. patens* zone.

rates of relative sea-level rise. Many previous studies examining rates of tidal wetland loss have dealt with quantitative analyses of marsh shore erosion rates and rates of vertical accretion. However, one of the purposes of this investigation is to examine the factors influencing landward wetland migration and to quantify the rate of marsh transgression over adjacent upland surfaces. After establishing basic relationships between surficial morphology, wetland vegetative zonation, depth to pre-transgressive sand, and adjacent upland slopes, a conceptual model for rates of landward transgression and changes in wetland zonation patterns at the wetlands/uplands boundary can be developed. Factors contributing to future areal extent and landward transgression of Delaware's coastal tidal wetlands include parameters such as rate of relative local sea-level rise, sediment supply (vertical accretion rate), erosion rate, and slope of adjacent upland surface.

Major assumptions used in any scenario modeling concern the rate of sea-level rise and the average sedimentation rate in fringing tidal wetlands. Based on recent studies (Titus and Narayanan, 1995), three scenarios of future sea-level rise over the next 100-200 years are selected: baseline (0.55 cm/yr), low (0.35 cm/yr), and high (0.83 cm/yr). A conceptual model for future wetland transgression and zonation must also consider marsh sedimentation rates and peat formation which act to increase the elevation of the marsh surface as relative sea level rises. Generally, sedimentation and peat formation have kept pace with rising sea level during the past century over most of the East Coast (Boesch *et al.*, 1982; Orson *et al.*, 1985; Stevenson *et al.*, 1986). However, if rates of marsh vertical accretion do not keep pace with sea level rise, wetlands zones will be submerged or will migrate landward, depending on adjacent upland slopes (Phillips, 1986). Similarly, even if vertical accretion rates keep pace with the rate of sea level rise, unless the wetlands migrate landward, fringing wetlands may not survive due to erosion on the bayward or seaward edge.

Vertical accretion rates in Delaware's fringing tidal salt marshes range from 0.27 to 0.52 cm/yr, as determined in this study. These rates are consistent with rates reported in the literature for the mid-Atlantic region (Meyerson, 1972; Lord, 1980; Church *et al.*, 1981; Kim *et al.*, in press). For the purposes of this conceptual model, vertical accretion rates of 0.27 cm/yr, 0.35 cm/yr, and 0.52 cm/yr are assumed (the actual rate will vary across any wetland transect).

Erosion rates and processes also ultimately affect the future areal extent of fringing tidal wetlands. As reported in the previous chapter, erosion rates for fringing tidal wetlands are highly variable, from 0.2 m/yr to 1.5 m/yr. For the purpose of this conceptual model, low (0.1 m/yr), medium (0.5 m/yr) and high (1.2 m/yr) rates of marsh shore erosion will be used in the model.

The final controlling factor on transgression of wetlands over adjacent upland surfaces is the slope of the surface being transgressed. As measured in survey transects in this study, adjacent upland slopes are highly variable, ranging from low (0.003), to medium (0.02), to high (0.11). Table 8.1 depicts the relationships between various factors controlling future areal extent of fringing tidal wetlands in Delaware, including slope of adjacent upland, rate of relative sea level rise, rates of transgression, and rates of marsh shore erosion.

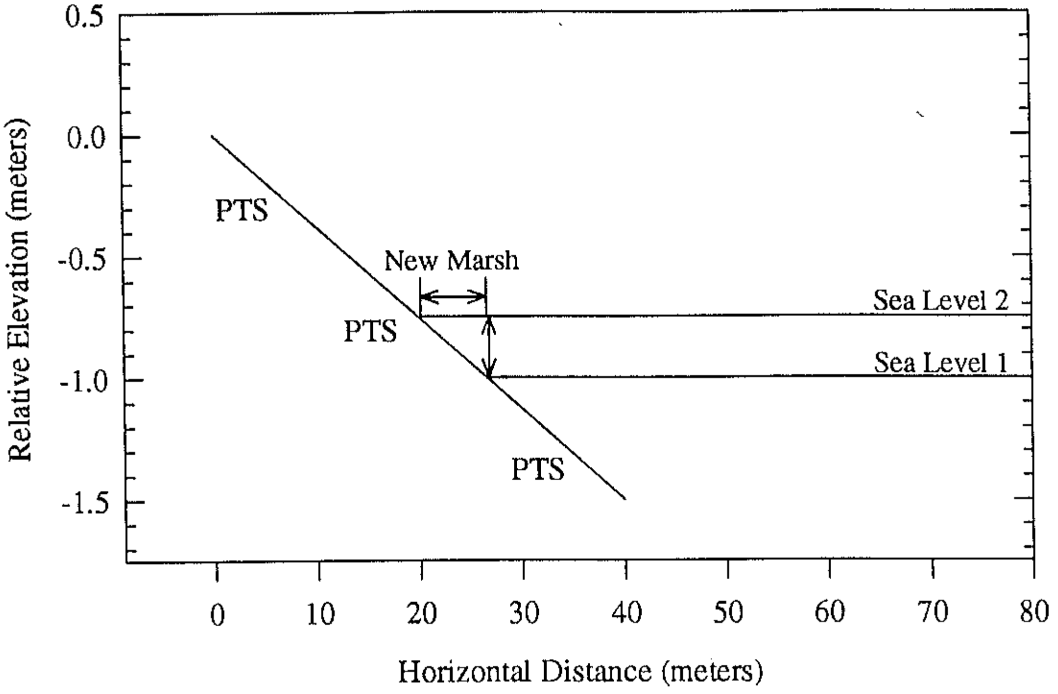
Table 8.1 Summary of factors controlling future areal extent of fringing tidal wetlands in Delaware.

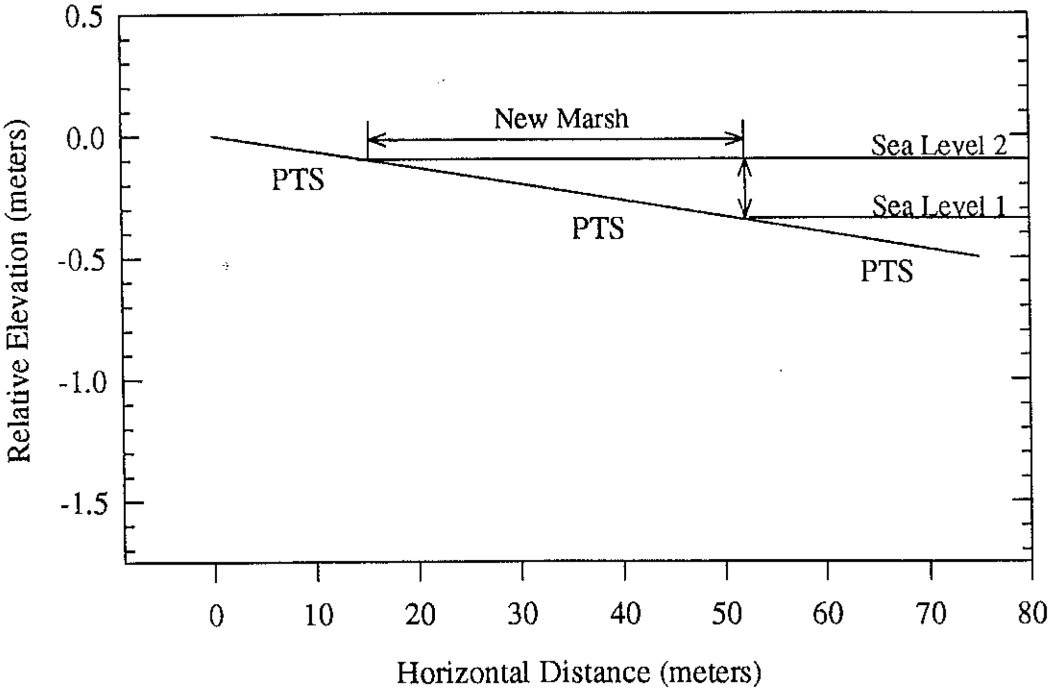
	Slope	100 yr SLR (0.35 cm/yr)	100 yr transgression	100 yr erosion (0.5 m/yr)	Net change in marsh width
(Low) Gentle	0.003	35 cm	116 m	50 m	+ 66 m
	0.007	**35 cm**	**50 m**	**50 m**	**no net change**
Steep (High)	0.02	35 cm	17.5 m	50 m	- 32.5 m
	0.1	35 cm	3.5 m	50 m	- 46.5 m

<u>Slope, Sea-Level Rise and Transgression Rate</u>. If the rate of vertical accretion is equal to the rate of sea-level rise, a direct relationship can be established between slope of adjacent upland, rate of sea level rise, and rate of landward transgression of wetland environments (Figure 8.13). This conceptual relationship can be observed at several of the transect sites surveyed for this study. For example, the profile of the Bay Vista study site (Figure 8.10) is relatively steep, with a 0.096 slope; there is no *S. patens* marsh developed between *S. alterniflora* wetlands and the adjacent upland. An increase in sea-level rise results in formation of only a small area (10-15 m) of new wetland in a 100-year period (Figure 8.10), as this is related directly to the adjacent upland slope.

Area of transgression for similar rates of sea-level rise at Pot Nets North-5 (Figure 8.11), with a lower adjacent upland slope of 0.005, shows a wide area of shallow subsurface antecedent topography adjacent to the upland where *S. patens/Distichlis spicata* are dominant species. Accordingly, for the same increase in

Figure 8.13 Conceptual diagram showing relationship between slope of adjacent upland surface and areal extent of landward migration of fringing tidal wetlands.


Effects of 0.25 Meter Sea Level Rise on a Steep Slope


Effects of a 0.25 Meter Sea Level Rise on a Gentle Slope

the rate of sea-level rise, a greater areal extent (25-82 m) of new marsh is formed over the same 100-year time period.

Thus, for identical conditions of rates of vertical accretion and sea-level rise, the areal extent of landward migration of wetlands at the Pot Nets North gently sloping site is approximately four times greater than that at the steeply sloping Bay Vista site, demonstrating the importance of pre-existing antecedent topography on geometry of future salt marshes.

The following observations may then be derived: 1) rates of nominal transgression increase as slopes decrease; and 2) for any given wetland, there may be a critical value for development of *S. patens* marshes, which depends on the slope of adjacent upland (≤ 0.09), geometry of the pre-transgressive sub-surface (depth to pre-transgressive sand < 40 cm), and sediment supply. As slopes increase, continued development/establishment of *S. patens/D. spicata* marshes is more critically dependent on antecedent geometry and sufficient sediment supply. Additionally, loss of fringing tidal wetlands is controlled by loss at the wetlands/water interface due to erosion.

Future Areal Extent of Fringing Tidal Wetlands. Ultimately, the future areal extent of fringing tidal wetlands depends not only on the slope of adjacent upland and rate of sea-level rise, but also on the extent of erosion on the bayward edge of the wetland. As has been previously demonstrated by Phillips (1986), the areal extent of wetlands will decrease over time if the rate of landward migration does not keep pace with the rate of erosion. Based on information obtained in this investigation, it is possible to develop conceptual models for future areal extent of Delaware's fringing tidal wetlands, assuming that the ideal situation is to maximize the presence of tidal salt marshes.

Best Case Scenario. Adjacent slope: low; erosion rate: low; rate of vertical accretion is equal to rate of sea-level rise. Result: continued transgression of wetland environments over time, maintenance of spatial relationships of vegetative zonation patterns. Examples: Pot Nets North, Draper-1, Bethany Bay-1.

Worst Case Scenario. Adjacent slope: high; erosion rate: high; rate of vertical accretion is less than rate of sea-level rise. Result: little or no landward movement of tidal wetlands; high marsh environments disappear, replaced by *S. alterniflora* low marsh areas. Eventually, with high rates of erosion along the marsh shoreline at the marsh/water interface, wetland area will disappear as erosion continues to base of adjacent upland. Examples: Thompson's Island, Rehoboth Bay, northwest side of Indian River Bay, and portions of Little Assawoman Bay shoreline (e.g. at mouth of Miller Creek, there is no fringing wetland because of low bluff adjacent to agricultural field).

Anthropogenic Impacts. When human impacts are considered in these scenarios, the future areal extent of fringing tidal wetlands may be altered. Development, shoreline stabilization, and 'hardening' of the shoreline will preclude any landward movement of fringing tidal wetland environments. Although installation of bulkheads and rip-rap at marsh shorelines may limit or prevent short-term erosion of the marsh, wetlands will eventually become inundated and drowned as sea level rises. Retaining walls installed at the wetlands/uplands boundary preclude landward migration of wetlands over adjacent uplands and will reduce potential development of new fringing wetlands. In contrast, vegetative stabilization and marsh creation in shoreline areas will increase wetland areas, and mitigation projects such as conversion of uplands to wetlands through grading and planting will result in net increases in marsh areas.

Future of Fringing Tidal Wetlands. It is evident from the previous discussion that the ultimate controlling factor in marsh transgression in terms of maintaining areal extent is geometry of adjacent upland surfaces. For a given rate of sea-level rise, low slope uplands will be transgressed more rapidly than uplands with steeply sloping faces. Although transect surveys in this study measured directly adjacent uplands (within 10s of meters of the wetland/upland boundary), the long-term future extent of wetlands as determined by sea-level rise is affected by adjacent slopes of broader areas. For example, even in areas characterized by low-lying agricultural fields, natural or artificial topographic features such as remnant dunes or the typical 0.5-1.0 meter plow berm that is established between the flat field surface and adjacent wetland often create short term vertical barriers to wetland transgression. However, once a threshold elevation of sea level is reached, the rate of transgression will accelerate over and above the berm, and wetland colonization of flat agricultural surfaces will proceed rapidly. Examples of this type of adjacent topography include areas adjacent to Little Assawoman Bay and southern Delaware Bay.

In addition, subsurface antecedent slope and depth to sand (marsh thickness) are important in determining the type of wetland that will develop as transgression occurs. Cross-sectional diagrams depicting various composite variations in slope/antecedent geometry conditions (Figures 8.6-8.8) illustrate that there is a great variation not only in the rate of marsh transgression, but also in the type of marsh (high marsh vs. low marsh) that develops as transgression continues. A major impact of geometry of adjacent uplands is the control exerted on the extent of future high marsh and transitional marsh zones. A scenario with an adjacent low slope would result in relatively minor and gradual changes, with the number of acres of transition marsh, high marsh, and low marsh maintaining spatial relationships and balance over time. Higher mean water levels over gentle slopes would gradually displace upland areas, killing plant species that cannot tolerate frequent tidal inundation, but promoting growth of more salt-tolerant species, such as *Iva frutescens*, *Baccharis halimifolia*, *S. patens*, and *D. spicata*. Low/intermediate adjacent slope scenario may result in the distribution of each sub-environment undergoing major changes, and thereby profoundly affect marsh ecology as high marsh environments are replaced by low marsh environments. In addition, existing low marsh areas may be eliminated by shore erosion, while transitional *I. frutescens*/*B. halimifolia* shrub zones may also be eliminated. Even with a low-lying adjacent upland slope, *S. patens* would move landward, but might disappear due to increased competition and shading effects of existing upland vegetation.

A high slope adjacent upland scenario would result in a diminished area of *S. patens* (and other high marsh species) as this zone become compressed. *S. patens* wetlands may not migrate significantly in a landward direction as sea level rises over a steep slope, and *S. alterniflora* will transgress over the bayward side of *S. patens* wetlands as inundation takes place.

In terms of process variables that contribute to wetland transgression, antecedent slope may be of greater significance than the rate of sea-level rise, as the slope ultimately determines future areal extent of the marsh surface and the distribution of vegetative zonation patterns and sub-environments within the marsh.

Predictive model scenarios may be used in conjunction with observed slopes, erosion rate estimates, and predicted rates of sea-level rise to determine the likelihood of suitable or necessary conditions for wetland transgression to occur. For example, with the current rate of sea-level rise and assuming vertical accretion rates are sufficient for the marshes to 'keep pace', higher rates of landward migration of tidal wetlands will occur in low slope, undeveloped areas (e.g., Little Assawoman Bay and Delaware Bay as opposed to the western shoreline of Rehoboth Bay).

Using model examples, the Inland Bays are divided into general geographic segments: northern, western, and southern. Swisher's (1982) characterization of shore erosion rates are combined with generalized characterization of topography and slope of surrounding uplands within two kilometers of wetlands/uplands boundary to determine short- and long-term future of fringing marshes in these areas (Table 8.2).

Summary

Based on this research, it is possible to predict future areal extent of fringing tidal wetlands in Delaware and the general distribution of high marsh/low marsh zones based on adjacent upland topography. In areas where tidal wetlands transgress over previously existing tidal freshwater wetlands in stream and river valley

Table 8.2 Predicted short- and long-term future of fringing marshes along the Inland Bays and Delaware Bay.

Shoreline Segment	Erosion Rate (m/yr)	Relative Slope Adjacent Upland	Short term (decades) future marsh	Long term (centuries) future marsh
Little Assawoman Bay: generally low slopes surrounding bay, but the limiting factor may be the 0.5 m 'berm' surrounding parts of agricultural fields to the west				
Northern	moderate(-0.3)	low; forested	mod/high	high
Western	high (-0.5)	low;plow berm	low	high
Southern	low (-0.2)	low;developed	high	low
Rehoboth Bay: upland necks on one side = steep; other side = shallow				
Northern	moderate(-0.3)	mixed	mod/high	high
Western	high (-0.5)	high	low	low
Southern	low (-0.2)	low	high	high
Indian River Bay: upland necks on one side = steep; other side = shallow				
Northern	high/mod(-0.4)	mixed	moderate	moderate
North-Western	high (-0.5)	high	low	low
Southern	high/mod(-0.4)	low	high	high
South-Western	mod (-0.3)	low	moderate	moderate
Delaware Bay: low sloping farmland after initial scarp is overtopped				
West-Southwest	high	low; low scarp	low	high

axes with steeply sloping edges, salt marsh vegetation will be dominated by *S. alterniflora* vegetation, with no high marsh *S. patens* zone. Alternatively, in areas where flat, low-lying agricultural fields become inundated by saline water, broad expanses of *S. patens* high marsh zones will be prevalent.

Based on existing topographies of land areas adjacent to fringing tidal wetlands, it is possible that over the short term, fringing wetlands may be "pinched out" between upland slopes and eroding shores. However, over the long term, especially in the vicinity of Delaware Bay and Little Assawoman Bay, broad expanses of fringing tidal wetlands will likely exist in the future. The nature of upland topography adjacent to Indian River and Rehoboth Bays, with the high elevations of Pleistocene necks, may preclude future development of broad tidal wetlands in these areas.

Natural and Anthropogenic Impacts on Local Relative Sea Level Changes and Areal Extent of Fringing Tidal Salt Marshes

Introduction

Development and evolution of tidal wetlands is determined by many interacting factors: sea-level rise, tidal inundation regime, source of sediment, and the growth of plant species tolerant to salt water inundation and tidal fluctuations. While previous chapters have evaluated natural and human impacts on zonation patterns in salt marshes, this chapter's focus is their impacts on erosion and aggradation of salt marshes. Depending on the interaction of these natural factors, tidal marsh areas may be enlarged or reduced; human activities within the watershed of tidal wetlands exert additional impacts on the areal extent of tidal wetlands in Delaware.

It is difficult to separate or prioritize the overall effects of the many interacting natural and anthropogenic factors which have impacted fringing tidal wetlands in Delaware. However, consideration of possible impacts and time frames in which they operate demonstrate that over the short term (decades), human induced alterations during the last 200 years may have had more significant effects than natural forces on transgression of salt marshes over adjacent upland surfaces. Natural forces have impacted tidal wetlands through wave-induced shore erosion, emergent marsh inundation, and landward marsh transgression caused by relative sea-level rise. Human induced impacts on tidal wetlands include alterations and manipulations resulting from: filling for development, ditching and draining for mosquito control, diking for flood prevention, inlet stabilization, dredging for navigation, hydrological modifications for highway construction, aquatic habitat restoration, and vegetation control for wildlife habitat enhancement. The most significant human impacts on future areal extent and landward transgression of fringing tidal wetlands include alterations to sediment supply and relative tidal water levels in the marsh. Dams along rivers and streams, flood protection levees, and erosion and sediment control measures prevent water and sediment from reaching coastal wetland systems, while dredging, ditching, and inlet stabilization projects may dramatically alter extent of tidal inundation in coastal marsh systems.

The extensive low elevation of tidal salt marshes, and their dependence upon a balance between vertical accretion (sediment supply, both mineral and organic), and duration and frequency of tidal inundation, make them extremely vulnerable to even minor fluctuations in any of these controlling variables. Potential impacts of natural forces and human alterations on coastal tidal wetlands are considered in more detail in the following sections.

Sedimentation and Relative Water Levels

Sediment supply. Over the last 250 years, tidal wetlands of Delaware have been affected by human use of shorelines and adjacent upland areas. The earliest European settlers began damming the mouths of tidal creeks and creating impoundments for cultivation and pasture (Orson *et al.*, 1992; Stutz, 1992; C. Fithian, personal communication). The conversion of coastal watersheds to farmland and subsequent cultivation increased the flow of terrestrial sediment to tidal waters, influencing marsh sedimentation rates.

Sea-level rise. The equilibrium model of the relationship between sea level and marsh surface level is based on the connection between sediment accumulation rates and rates of sea-level rise. Models have been developed to illustrate the three basic scenarios relating sea-level rise and salt marshes: 1) if sea level rises rapidly, the marsh will drown, as it cannot 'keep pace' with the rate of sea level rise; 2) if sea level rises slowly, the marsh can maintain itself, balancing the increase in flooding frequency with accumulation of sediments; 3) if the rate of sea-level rise decreases, and available sediment increases, the marsh will build in elevation, continuing to 'outpace' sea-level rise (ultimately resulting in marsh loss due to invasion of upland species) until flooding frequency and sediment supply decrease.

Past studies have demonstrated that wetland response to a modest increase in the rate of sea-level rise consists of concurrent increases in sedimentation and vegetative growth which can ultimately maintain the surface area of most tidal wetlands through a natural process of landward and upward migration, as has happened in the past.

Tidal range. Although past research has clearly indicated that increases in tidal range increases sedimentation rate (Ranwell, 1964; Harrison and Bloom, 1977), this factor is generally overlooked when vertical accretion rates are discussed. Anthropogenic effects on tidal range in fringing tidal salt marshes may actually be more significant than the long-term effects of sea-level rise.

The tidal marshes of Delaware are highly sensitive to local tidal elevation, and must maintain an elevation that is intertidal. Increasing the frequency and duration of tidal flooding can result in changes in wetlands community structure, especially in microtidal areas. The vulnerability of coastal wetlands to inundation and submergence depends largely on tidal range. Surface elevations of coastal wetlands are generally less than one tidal range above mean sea level; thus, if a dramatic increase in sea level occurred, equal to the height of one tidal range, all existing wetlands would drown.

Floodwater Hydrography. Although detailed examinations of tidal floodwater hydrography were not made in this study, general observations and previous work permit speculation and suggestions for future research. Stumpf (1981) determined that the period of flooding on the Holland Glade marsh surface is distinctly asymmetric and that the sediment load transported by 'normal' tides cannot supply the entire marsh with sediment required to maintain it against sea-level rise. The frequency and magnitude of tidal flooding and associated sediment supply can be dramatically affected by seasonal variables, especially at the wetlands/uplands boundary area of tidal salt marshes. Tide gauge data obtained in a ditch at a Delaware Bay site (Prime Hook Radio Station) demonstrated that the study area (*S. alterniflora* and *S. patens/D. spicata* wetlands/uplands boundary area) was not influenced by semi-diurnal tides. This upper section of marsh was flooded only during periods of storm tide, or was flooded by upland runoff during thunderstorms.

At the Delaware Wildlands site along the western shore of Rehoboth Bay, tide gauge data obtained from a ditch located in *S. alterniflora*-dominated wetlands within 15 m of the *S. patens*/D. *spicata* zone, demonstrates that the wetlands are influenced by a typical semi-diurnal flooding pattern during late summer and fall. However, in winter months when northwest winds dominate, high tide levels do not flood the marsh surface on a regular basis; rather, days and weeks can pass between tidal inundation of the *S. alterniflora*-dominated marsh surface.

Seasonal effects on tide range and associated sediment supply have also been noted by Stumpf (1983) who documented reductions in mean water levels in Breakwater Harbor from November to April, causing reduced frequency and magnitude of tidal flooding and associated sediment supply to the marsh surface. Stumpf (1983) also noted decreases in creek suspended sediment supply in winter months compared to summer and fall months, and reached a conclusion that during storm-free tides, sedimentation is no more likely to occur in the winter than the summer.

Hydrologic Alterations, Diversions, and Changes in Tidal Range

Delaware Bay. Wetland hydrologic regimes and distribution patterns of salt marsh plant communities are impacted by natural and manmade channels which are created and maintained for navigational purposes. Historically, many creeks and rivers have been dredged, affecting the circulation and flooding patterns, as well as flow and tidal elevation, of adjacent embayments and tidal marshes. Large scale alterations of the hydrology of tidal marsh systems are evident in channel straightening and diversion projects throughout the Delaware Estuary (Wetlands Research Associates, 1994). Tidal creek channel straightening projects, conducted in the late 1800s and early 1900s are evident in the Mispillion, Leipsic, Smyrna, and St. Jones rivers (Wetlands Research Associates, 1994).

Historically, dredging has been conducted to establish or maintain navigation channels for commercial and recreational purposes. During the 1800s and early 1900s, dredging was also conducted to acquire fill material for wetland areas adjacent to settlements. Over the last 50 years, most dredging projects have been related to navigation, and occur throughout the Delaware Bay and Inland Bays areas. For example, operation and maintenance dredging requirements for the Delaware River are high, especially for maintenance of a 35' to 40' shipping channel. Although related changes in tidal range have not been calculated, it is likely that increased dredging and deepening of the Delaware River channel has resulted in an increased tidal prism in the estuary and associated changes in tidal range in adjacent salt marsh areas.

Roosevelt Inlet/Broadkill River. For the study sites located at the wetlands/uplands boundary along the southwestern portion of Delaware Bay, migration patterns and history of Broadkill River and Roosevelt Inlet openings and closures have likely had impacts on tidal range and wetland development (E. Maurmeyer, personal communication). The progradation of Cape Henlopen during the early part of the nineteenth century greatly affected littoral transport along the Broadkill Beach system. Wave refraction and flood tidal currents deflected the mouth of Broadkill River northward approximately 1650 meters between 1842 and 1882 (Maurmeyer, 1974; Kraft *et al.*, 1975). However, by 1917, Broadkill spit growth and new inlet formation resulted in major geomorphic changes to the Broadkill River area; Broadkill River became the present tidal slough between the marsh and barrier island presently called Broadkill Beach (Kraft *et al.*, 1975). By 1938, Roosevelt Inlet was dredged and stabilized to replace Old Broadkill Inlet, farther to the north, which had serious silting problems (Maurmeyer, 1974). Aerial photographs of the area show that Old Broadkill Inlet was closed by 1954, creating Broadkill Sound to the north (Maurmeyer, 1974). Thus, over the past 150 years tidal salt marshes adjacent to Broadkill River have been subject to changes in hydraulic regime; tidal range has likely increased since stabilization of Roosevelt Inlet, and tidal flooding frequency and duration have likely been altered.

Lewes and Rehoboth Canal. Dredging of the Lewes and Rehoboth Canal between 1912 and 1917 dramatically changed tidal wetlands adjacent to the waterway. Before canal dredging, Snow (1902, 1913) indicated that the marsh in the vicinity had a small tidal influence and was dominated by *S. patens* and *Juncus scirpoides* vegetation. Lewes and Rehoboth Canal construction resulted in an increase in the local tide range from 0 cm ('negligible' according to U.S. Congress, 1906; Stumpf, 1983) to 80 cm (Stumpf, 1983), thereby increasing tidal flooding in salt marsh areas adjacent to the waterway (Stumpf, 1983). The dredging of the canal also increased the tidal range in Rehoboth and Indian River Bays.

Mosquito Ditching. Daiber (1986, 1987) provides a detailed summary of impacts of mosquito ditching on Delaware tidal salt marshes. The Civilian Conservation Corps (CCC) created an extensive network of parallel ditches throughout Delaware's salt marshes during the period 1932 to 1938. Studies such as those conducted by Bourn and Cottam (1939) documented that the mosquito ditching program resulted in changes in marsh hydrology, plant community distribution, and wildlife. As a result, the ditching approach to mosquito control ceased in the early 1940s, although ditch cleaning programs continued into the 1960s (Florschutz, 1959; Smith, 1968).

Changes in wetland hydrology and vegetative community structure as a result of mosquito ditching are 'bidirectional', including both 1) drainage of surface water of the marsh, thereby enhancing high marsh environments, and 2) increased flooding of the marsh by creating ingress channels for tidal flow and circulation in marsh community areas where daily influence of tides had not previously existed. It has been reported (Florschutz, 1959) that ditched marshes in Delaware flood and drain twice in a normal day, resulting in "more benefit to surrounding estuaries and bays than from a natural marsh which probably only floods completely during extremely high tides." Possible increases in sedimentation rates in salt marshes due to the mosquito ditching program may be inferred from the frequent 'filling in' of ditches which required maintenance ditching every five to ten years (Florschutz, 1959).

Although many investigations of mosquito ditching and effects on salt marshes report that substantial lowering of the water table in the area results in a marked change in the vegetation of the ditched marsh, Florschutz (1959) reports that, when ditched, many of the heaviest mosquito producing marshes in Delaware, typically dominated by *S. patens*, *S. alterniflora*, and *D. spicata* vegetation, do not have drastic vegetative changes except along the ditches on the spoil banks where *Hibiscus*, *I. frutescens*, and *B. halimifolia* invade.

Indian River Inlet. Indian River Inlet has a long history of shoaling, closures, openings, and migration. Migration and changes in configuration of the inlet have been recorded since 1800, with historical changes in inlet location and configuration documented in charts and maps. The inlet was stabilized in 1938 with stone jetties by the U. S. Army Corps of Engineers; prior to 1938, the inlet had migrated and closed several times, as described by local residents and in transcripts from Congressional Hearings. Between the years 1800 and 1925, the ephemeral inlet migrated, opening and closing as the channel was dredged (to approximate depths of 1.2 m) and subsequently shoaled to its original, 'natural' depth of 0.6 meters.

Closure of the inlet resulted in decreased salinity in Indian River and Rehoboth Bays, with concurrent decreases in shellfish populations. In addition, water levels rose in the bay due to land drainage accumulation; in 1929, local farmers reported that fast land adjacent to the bay was being flooded. In 1938, the federal government authorized an inlet stabilization project for the area; channel dredging and jetty construction were initiated in 1938 and completed in 1940.

Changes in tidal flow in Indian River and Rehoboth Bays have been documented by Karpas (1978) and Lanan and Dalrymple (1978) who demonstrate that tides are a dominating factor in the hydrodynamics of the bays. More recently, a numerical model investigation was conducted for Indian River Inlet and Indian River and Rehoboth Bays (Raney *et al.*, 1990) in which a model of the tidal prism in the bay system was developed, and the evolution of the tidal prism and its influence on system hydrodynamics was investigated. Raney *et al.* (1990) demonstrate that tremendous changes have occurred in bay system hydrodynamics over the last 50 years; tidal prism at Indian River Inlet has increased by a factor of approximately 4.5, and the tidal prism entering Rehoboth Bay has increased by a factor of approximately 3.8, with corresponding increases in surface water elevation and velocities (Figure 8.14). These changes in tidal prism, along with channel dredging within the bays, have likely increased the flushing of the bays and tidal flooding of the bays and fringing wetlands.

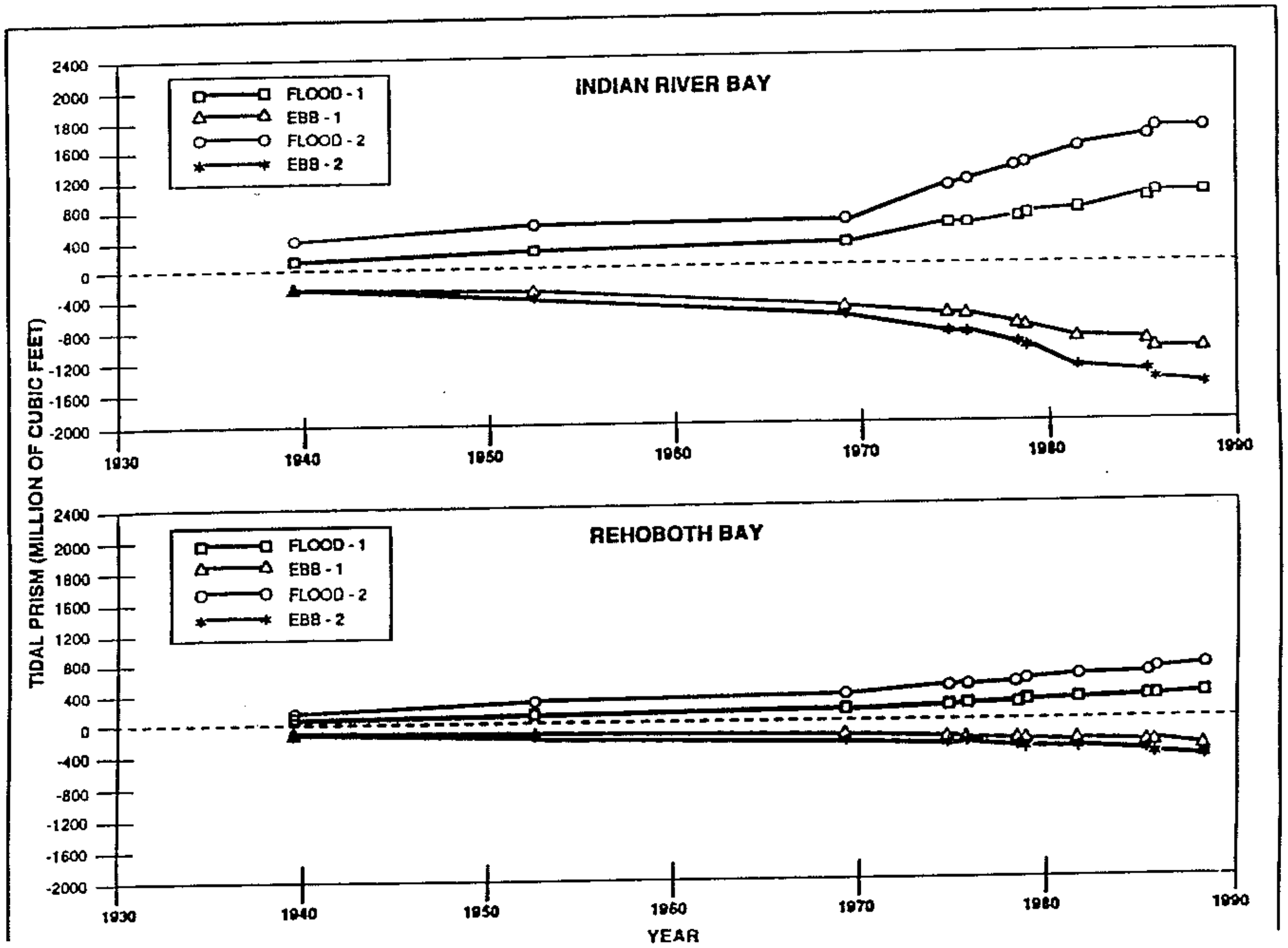


Figure 8.14 Variation of tidal prism in Indian River and Rehoboth Bays, 1941 to 1988 (from Raney *et al.*, 1990).

Tidal amplitude in the bays has also been modified as the cross-sectional area of the inlet has increased over time. Raney *et al.* (1990) developed and verified a model of change in surface elevation in the bays caused by enlarging the cross-section of the inlet. Spring tide elevations in both Indian River and Rehoboth Bay areas are characterized by higher high tides and lower low tides over the time period from 1941 to 1988 (Figure 8.15). The spring tide range at the mouth of Vines Creek (southwestern Indian River Bay) was 15 cm in 1941, and increased to 92 cm by 1988. For Dewey Beach (northeastern Rehoboth Bay), tidal range for the same time period increased from 8 to 46 cm. Such increases in tidal range have likely had an impact on the spatial and temporal distribution of fringing wetland environments in Indian River and Rehoboth Bays. It would be difficult to separate these tidal ranges effects on water level increases in the Bays from the overall effect of sea-level rise.

Management Implications

An obvious management response to maintain areal extent of Delaware's fringing tidal wetlands is to attempt to avoid, minimize, or delay all possible adverse impacts caused by increases in water level relative to marsh surface elevation. These adverse impacts include not only erosion of the bayward edge of the marsh, but also submergence and/or inundation of the marsh surface. Additionally, maintenance of high marsh and low marsh surface vegetative distribution will require detailed attention to surface elevation and marsh thickness or subsurface substrate characteristics, as well as adjacent upland slopes. Structural or mechanical management responses include: jet-

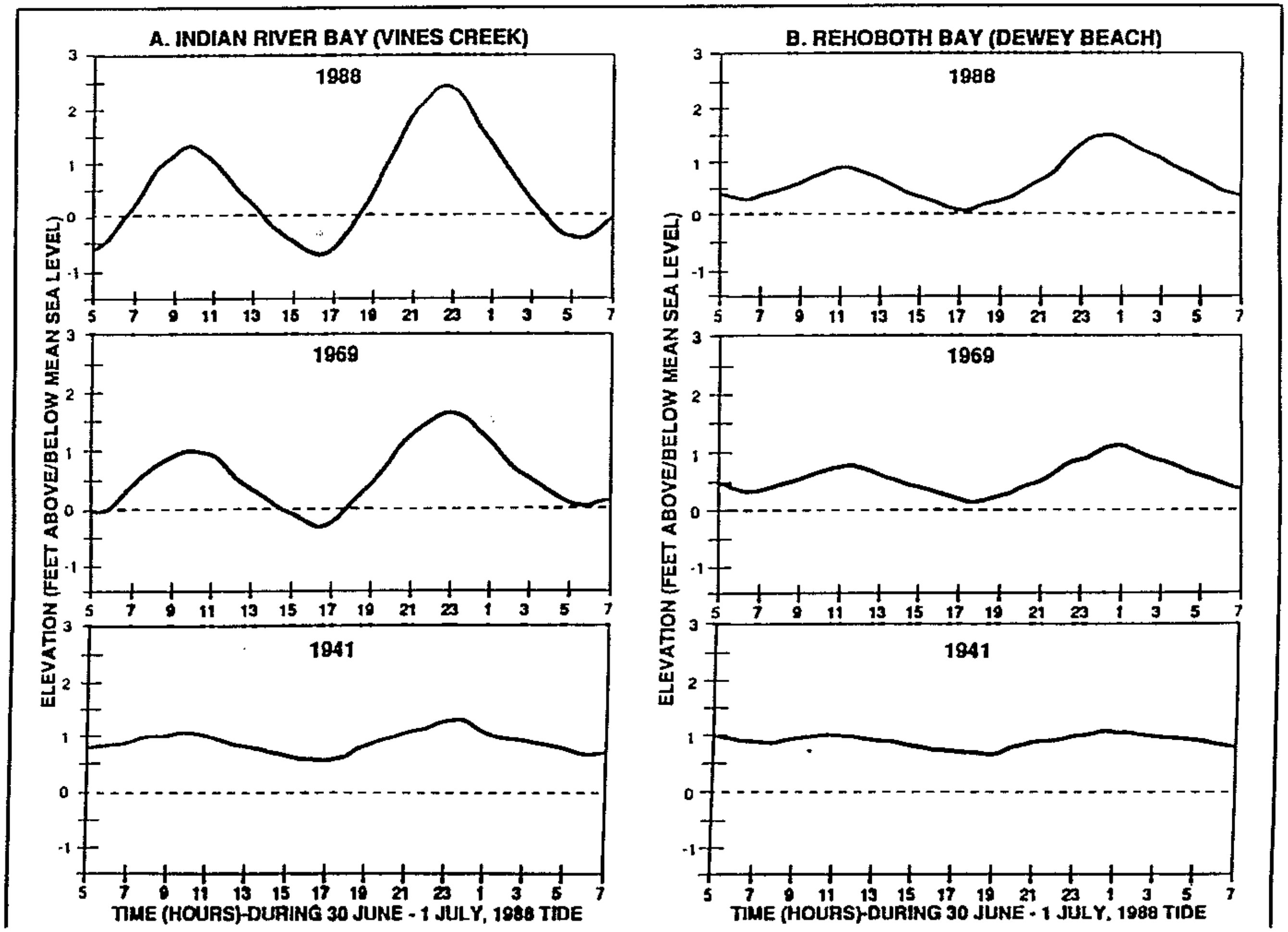


Figure 8.15 Computed changes in historic tide heights for Indian River and Rehoboth Bays, 1941 to 1988 (from Raney *et al.*, 1990).

spray spoil deposition over subaqueous bottoms or emergent wetlands to create new wetlands or to elevate existing wetlands; use of flood gates and other structures to manage frequency, duration, and height of tidal inundations; and excavation of adjacent uplands to suitable elevations for expansion or creation of new tide marsh areas.

Alternatively, management responses could include development and implementation of a new statutory authority that would create upland buffer zones landward of the upland marsh edges, in which no construction or significant disturbance could occur. This would permit natural transgression of wetlands environments over adjacent upland surfaces, which is especially important in low-sloping areas where high marsh and low marsh environments would be spatially and temporally maintained. Depending on spatial salt marsh vegetative characteristic requirements, possible set-back and building restriction lines could be based on elevation and slope of adjacent upland areas rather than a linear distance from mean high water. Set back lines could be established based on projected rates of landward transgression of wetlands vs. slope of adjacent uplands (Figures 8.16 and 8.17).

There are many applications of this research to design and implementation of wetland mitigation projects, especially with regard to both elevation, substrate characteristics, and the importance of subsurface geometry to wetland vegetative zonation patterns. It is hoped that this project has resulted in further consideration and understanding of the potential impacts of physical characteristics of substrate and adjacent and antecedent geometries on surface vegetative distribution in salt marshes.

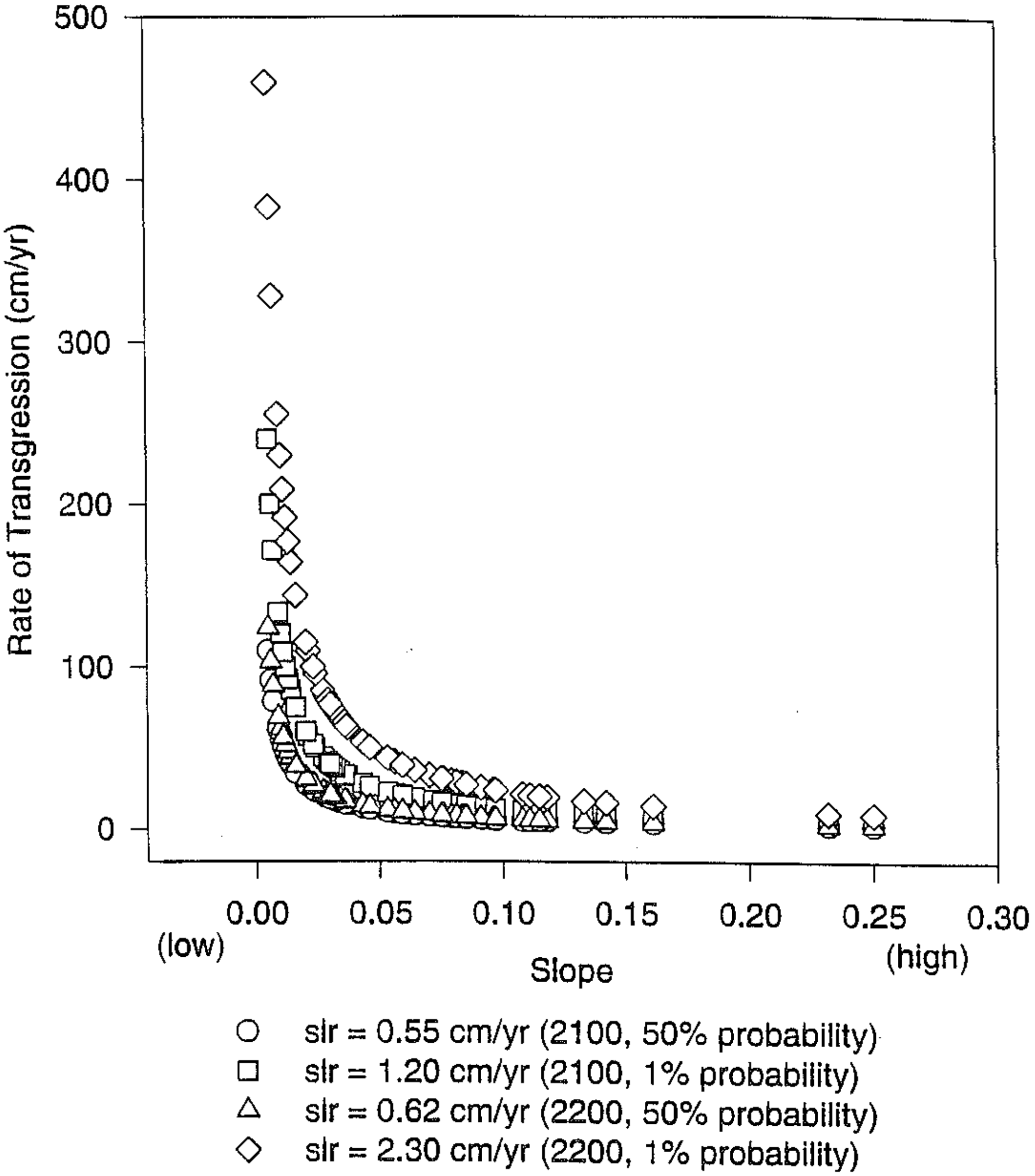


Figure 8.16 Project rates of landward transgression of wetlands vs. slope of adjacent uplands for EPA estimates of sea-level rise.

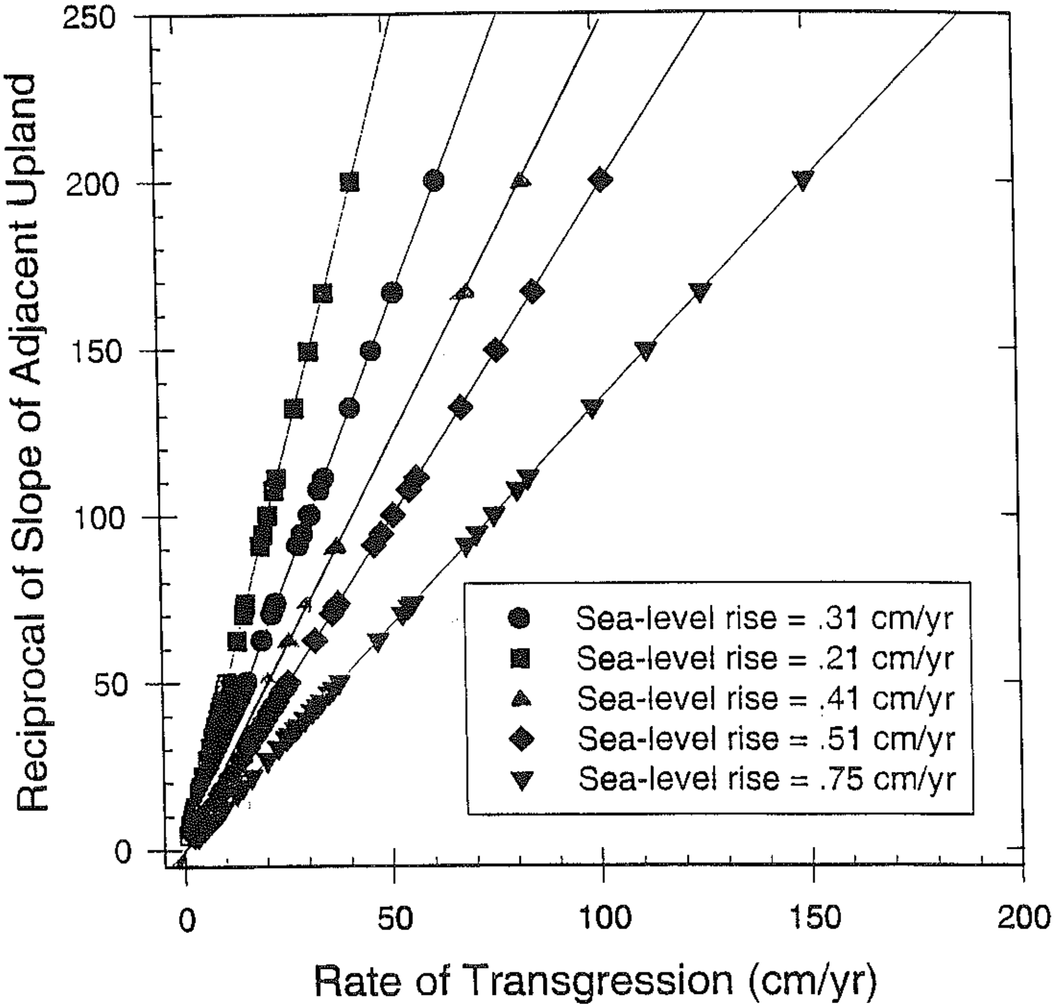


Figure 8.17 Nominal rate of transgression (cm/yr) vs slope of adjacent uplands for various scenarios of sea level rise.

Summary and Conclusions

1) At the wetlands/uplands boundary zone in Delaware's fringing tidal salt marshes, the general stratigraphic pattern is one of low marsh transgressing over high marsh deposits, and high marsh overlying pre-transgressive sandy sediments.

2) Slope of the adjacent upland surface ultimately controls spatial distribution and geometry of fringing wetland sedimentary deposits and the areal extent of future fringing tidal salt marshes in Delaware.

3) Geometry of the pre-transgressive surface controls the distribution of past and present high marsh *S. patens*/D. *spicata* zonation patterns. *S. patens/D. spicata* high marsh zones will be restricted to areas where depth to pre-transgressive sands is less than 40 cm and/or where adjacent upland slope is shallow/gentle.

4) Fringing tidal salt marshes along margins of stream and river valleys will be dominated by *S. alterniflora* vegetation, even to the base of steeply sloping adjacent uplands. A narrow band of *I. frutescens/B. halimifolia* and/or *Phragmites australis* may be present at wetland/upland boundary. However, *S. patens* environments will be noticeably absent.

5) Rates of landward migration of fringing tidal wetlands over the past 200 years, as determined from geologic cross-sections, range from 0.08 m/yr to 0.44 m/yr. These rates are similar to the nominal transgression rates calculated from known slopes and estimated rates of sea-level rise; however, the rates are much lower than those measured for the Inland Bays areas from metric map data. This suggests that rates

measured for Indian River and Rehoboth Bays from metric maps may be extreme rates of transgression due to anthropogenic impacts on tidal range and water levels in the bays.

6) Increases in tidal range in the study area have been significant over the last 100 years due to anthropogenic impacts related to dredging, inlet stabilization, and mosquito ditching. For example, the Lewes and Rehoboth Canal was built between 1912 and 1917, and changed the tide range from 'negligible' (U.S. Congress, 1906) to 80 cm (Stumpf, 1981). Similarly, from 1941 to 1988, tidal range increased in Indian River and Rehoboth Bays from 15 to 92 cm and from 8 to 46 cm, respectively.

7) As local relative sea-level continues to rise, temporal and spatial relationships of fringing salt marsh subenvironments are likely to change. However, detailed evaluation of alterations in sediment supply, tidal range and overall water level must be made before variation/changes in salt marsh environment are attributed solely or specifically to rates of sea-level rise.

8) Relationships between sea-level rise, tidal inundation, inorganic sediment supply, in situ organic component of sedimentation, and antecedent topographies must be closely examined before predictions regarding areal extent of Delaware's fringing tidal wetlands can be established.

Chapter 9

SUMMARY AND CONCLUSIONS

Data obtained on surficial morphology and subsurface stratigraphy in this study provide a chronology for coastal salt marsh evolution and vegetative development in Delaware's fringing tidal wetlands. Changes in water level, salinity, and sediment supply, affected in the long-term by sea-level rise, and in the short-term by anthropogenic effects on tidal hydrology, are the most critical and most common disturbance variables. However, geometry of the antecedent surface, or pre-Holocene topography, exerts a previously unconsidered environmental control on the distribution of vegetative zones within the salt marsh. The significance of subsurface stratigraphy and both antecedent and adjacent topography should be considered when examining the physical factors which ultimately determine distribution of vegetative zones and areal distribution of salt marshes.

Time-transgressive patterns, including vegetative zonation shifts in response to sea-level and other changes in hydrology and tidal range, were documented using surface and subsurface indicators of spatial and temporal variability. Detailed mapping of surface vegetative distribution patterns related to marsh thickness and adjacent upland slope along with examination of macroscopic vegetative remains and detailed stratigraphic interpretations of wetland vegetative transitions, results in detailed characterization of the nature of the wetlands/uplands boundary of Delaware's fringing tidal marshes. In addition, geochronologic data from ^{210}Pb, ^{137}Cs, and pollen analyses permit general correlation of observed stratigraphic changes with historic documentation of environmental and physical changes in the marsh.

Sequential historical maps of the study area illustrate the dynamic nature of changes in Delaware's coastal wetland environments, as well as the transgressive nature of spatial and temporal vegetative relationships mapped in fringing wetlands in coastal Delaware. The activity and impact of anthropogenic alterations to inlets and tidal range, along with longer-term impacts of sea-level rise, are illustrated in the landward migration of vegetative zones. Similarly, subsurface stratigraphy is indicative of the overwhelmingly transgressive nature of the wetlands/uplands boundary area. Although the composition of salt marsh vegetative assemblages is primarily determined by tidal range, salinity, and water-table fluctuations, these factors are ultimately affected by geometry and slopes of the surface of inundation. Thus, vegetative succession in the upper reaches of a salt marsh may be primarily controlled by physical site conditions.

Data obtained in this study indicate that salt marsh vegetation changes which have occurred over a time period of 30-200 years are characterized by transgression or submergence. Greatest changes in marsh vegetation patterns have occurred in low-sloping areas where high marsh environments have migrated over adjacent upland surfaces, and low marsh areas have concurrently transgressed pre-existing high marsh areas. However, in areas characterized by steeply sloping adjacent and antecedent geometry, floristic patterns have not demonstrated the same extensive areal changes. Rather, in marshes adjacent to steeply sloping uplands, *Spartina alterniflora* vegetation has remained dominant over the past 50-100 years, as high marsh zones either never existed, or were 'pinched-out' between transgressing low marsh and adjacent upland.

Short term vegetative changes documented in this study result not only from a rate of vertical marsh growth lower than local relative sea-level rise, but also from the nature of antecedent topography. Natural and anthropogenic factors resulting in increased tidal inundation have produced changes in the salt marsh environment that ultimately result in floristic change. In areas where the salt marsh has kept pace with the rate of relative local sea-level rise, the areal distribution of resulting vegetative zones is ultimately dependent upon simple geometric relationships of adjacent slope and antecedent depth to pre-transgressive sand. In areas where vertical growth of the marsh surface has been less than that of local relative sea-level rise, the effects of pre-existing geometry become even more significant as wetland vegetative zones become pinched out between steep uplands and the effects of inundation and encroaching seawater.

Rapid shifts in both subsurface stratigraphy and surface vegetative zonation patterns mapped at the wetlands/uplands boundary are likely due to increases in tidal range caused by anthropogenic effects (inlet stabilization, ditching, dredging, etc.) superimposed on long-term sea-level rise. Although geochronologic data suggest that over the past 100 years, wetlands are 'keeping pace' with sea-level rise, shorter term (30-year) rates of vertical accretion in the Inland Bays suggest that wetlands are not "keeping pace" with sea-level rise. Transgression of both upland and wetland sites, upland slopes ranged from shallow (0.0067) to steep (0.232), with corresponding widths of high marsh ranging from 67 meters to 0.0 meters, respectively. These data suggest that, ultimately, adjacent upland slopes control not only the fate of coastal tidal wetlands, but also vegetative zonation patterns at the wetlands/uplands boundary.

4) Five microfacies are distinguished in fringing salt marsh deposits based on sedimentologic characteristics (grain size, texture, color) and plant species identified in the subsurface based on rhizomatal characteristics of vegetative remains in over 400 cores. These five microfacies include: pre-transgressive sand facies, basal soil facies, high marsh transition facies, high marsh facies, and low marsh facies.

5) Stratigraphic relationships between these facies are overwhelmingly transgressive, with wetlands developing over previously upland surfaces, and low marsh environments transgressing over high marsh environments.

6) Vertical accretion rates calculated in fringing *S. alterniflora* marshes using ^{210}Pb, ^{137}Cs and oak:ragweed pollen ratios range from 0.12 to 0.52 cm/yr. Vertical accretion rates determined by the three geochronologic methods suggest that rates may vary depending on length of time interval considered. Long-term (250-year) pollen rates and short-term (30-year) ^{137}Cs rates are generally lower than the current estimate for the rate of sea-level rise (0.31cm/yr), while 100-year ^{210}Pb rates generally approximate or exceed the 0.31 cm/yr rate of local relative sea-level rise.

7) Short-term (30-year) accretion rates suggest that Delaware's fringing tidal salt marshes may be in the process of becoming submerged; i.e., they are not keeping pace with the rate of sea-level rise. Over the past 100 years, anthropogenic

impacts such as ditching, dredging, and inlet stabilization may have exerted significant influence on salt marsh vegetative transitions and associated vertical accretion rates due to changes in water level and sediment supply.

8) Between 1941-1988, increases in tidal range in Indian River and Rehoboth Bays are documented to be 77 cm and 38 cm, respectively, primarily due to stabilization of Indian River Inlet. Similarly, after dredging of the Lewes and Rehoboth Canal in 1917, tidal range in adjacent salt marshes increased by 80 cm. These rapid changes in tidal range have exerted significant influence on salt marsh vegetative transitions and associated vertical accretion rates. Thus, use of changing patterns in wetland subenvironments and accretion rates as evidence of variations in sea level should be done cautiously.

9) Areal extent of fringing tidal wetlands is dependent upon transgression of wetland environments onto adjacent upland surfaces as well as erosion of marsh shores. Previously established erosion rates for wetlands along the Rehoboth Bay shoreline range from -0.2 to -1.6 m/yr, with an overall average rate of -1.0 m/yr. Along the southern Delaware Bay shoreline, erosion rates range from -0.2 to -1.4 m/yr, with the average erosion rate estimated to be -0.5 m/yr. Higher erosion rates (up to -5.2 m/yr) characterize central and northern sections of the Delaware Bay shoreline.

10) Historical changes in location of the wetlands/uplands boundary were quantified for study sites in Indian River and Rehoboth Bays. Landward migration of wetland environments is observed in many areas, with spatial and temporal variation in rates of transgression measured from metric maps. Rates of landward transgression of

Delaware's fringing salt marshes may be characterized based on slope of adjacent uplands. Where adjacent upland topography is steep, average lateral migration rate is +0.35 m/yr. However, where the adjacent upland slope is shallow, average landward migration rate is approximately +1.7 m/yr. The most rapid rates of landward migration of the wetlands/uplands boundary are found in headwater areas of tidal streams, with an average rate of +3.2 m/yr.

11) Calculated lateral migration rates based on adjacent upland slope, rates of vertical accretion, and predicted rates of relative sea-level rise, range from 0.2 to 0.5 cm/year, depending on the slope of the upland surface being transgressed. However, Delaware's marsh shorelines are currently being eroded at rates of 0.2 to 5.0 m/yr, suggesting that the areal extent of many fringing tidal salt marshes is decreasing.

12) As illustrated by an aerial photograph of Angola Neck along the western shore of Rehoboth Bay (Figure 9.1), Delaware's fringing tidal wetlands are ephemeral features. Not only do they encroach adjacent upland surfaces at variable rates, but they exhibit a high degree of variability in zonation pattern and areal extent due to variations in adjacent slope and geometry of pre-existing surface.

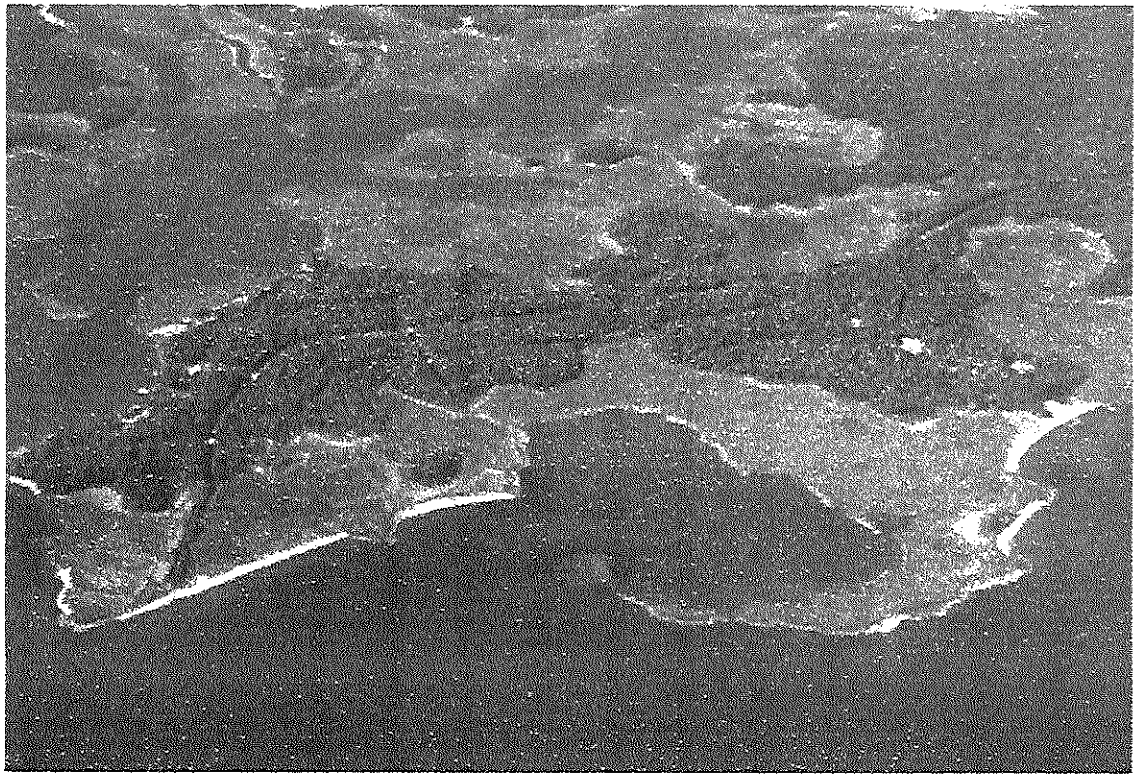


Figure 9.1 Aerial photograph, western shore of Rehoboth Bay, illustrating ephemeral nature of fringing tidal wetlands (photograph by E.M. Maurmeyer).

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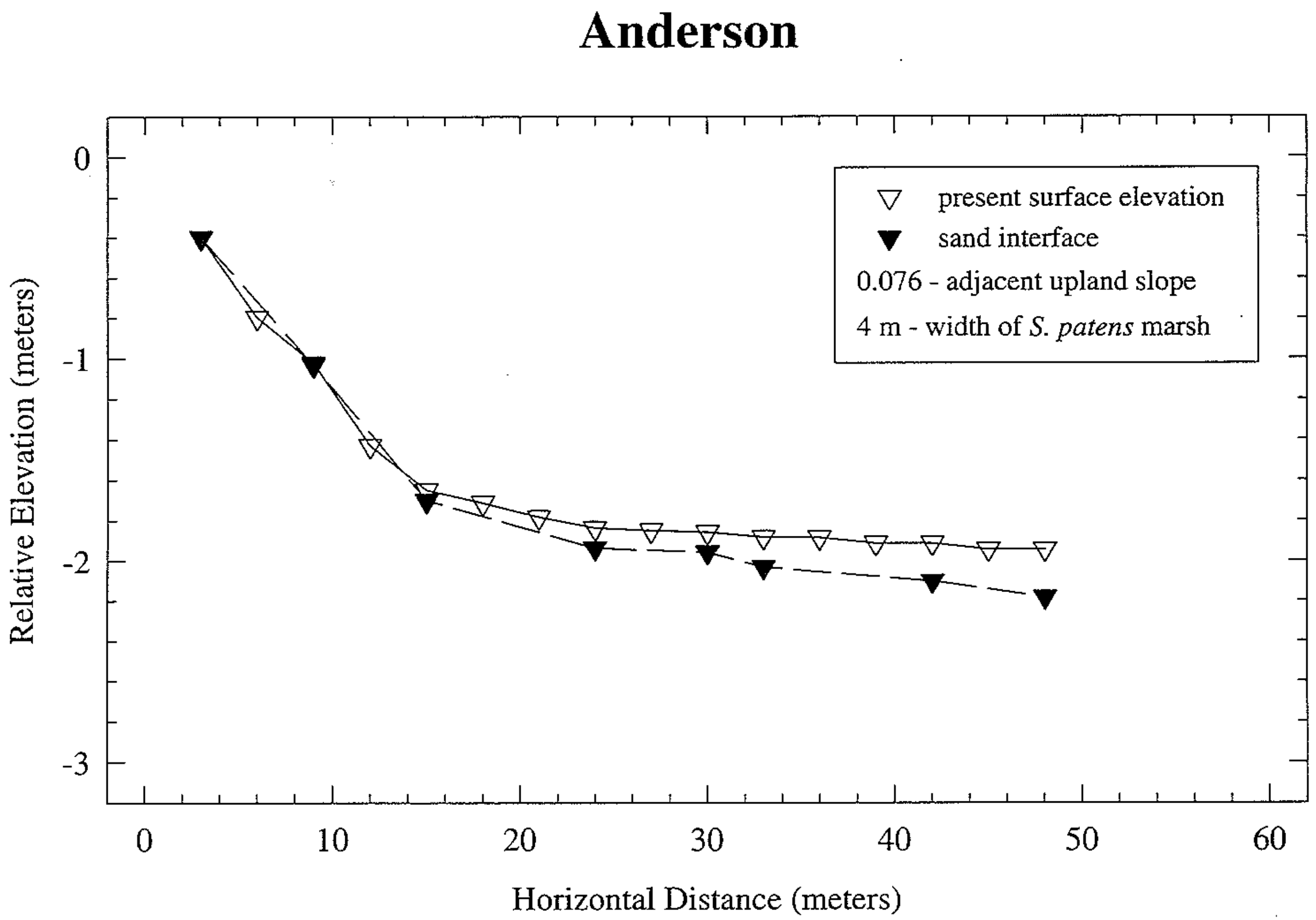
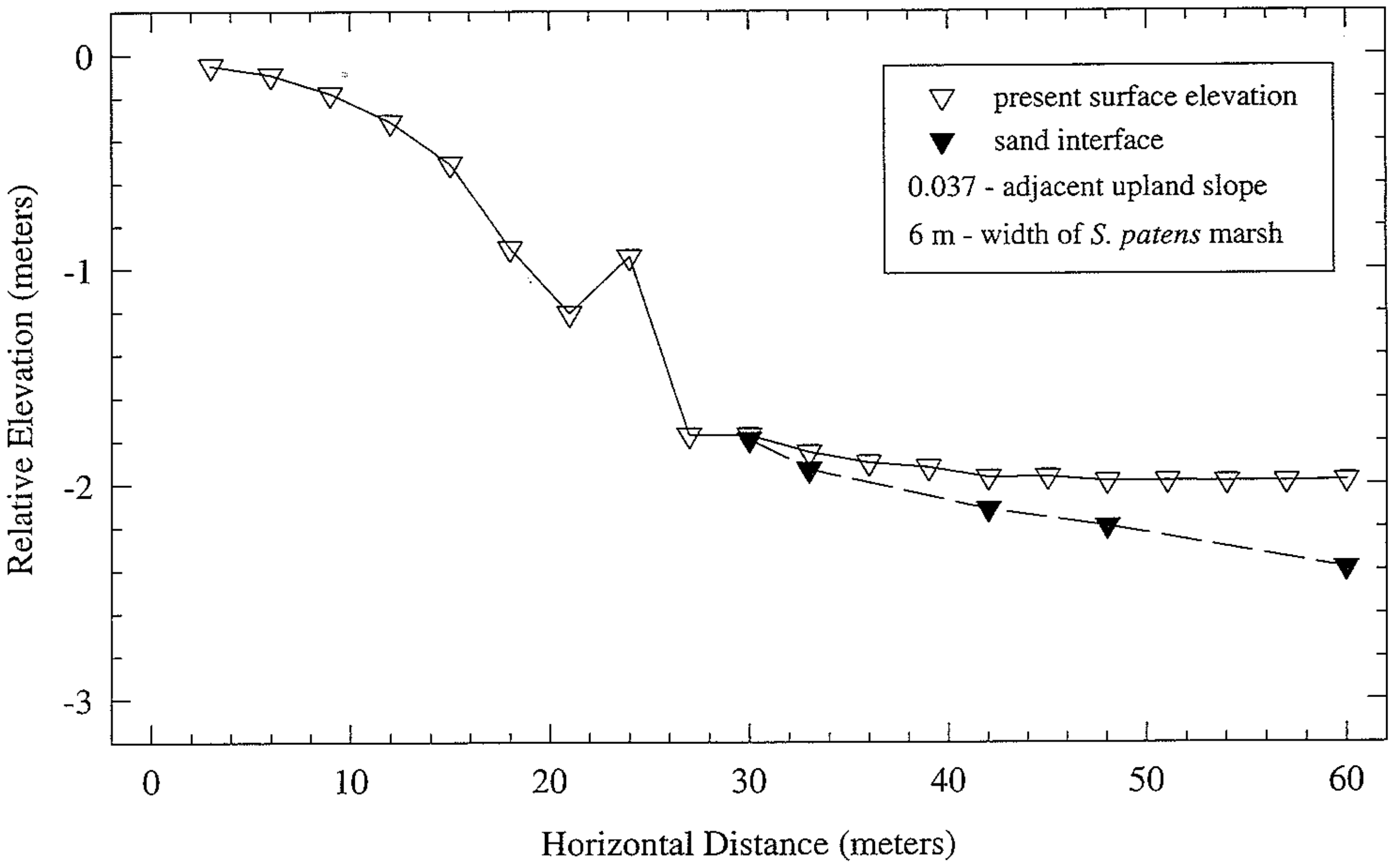
APPENDIX A

Cross-Sectional Elevation Profiles:
Marsh Surface Elevations, Adjacent and Antecedent Topography

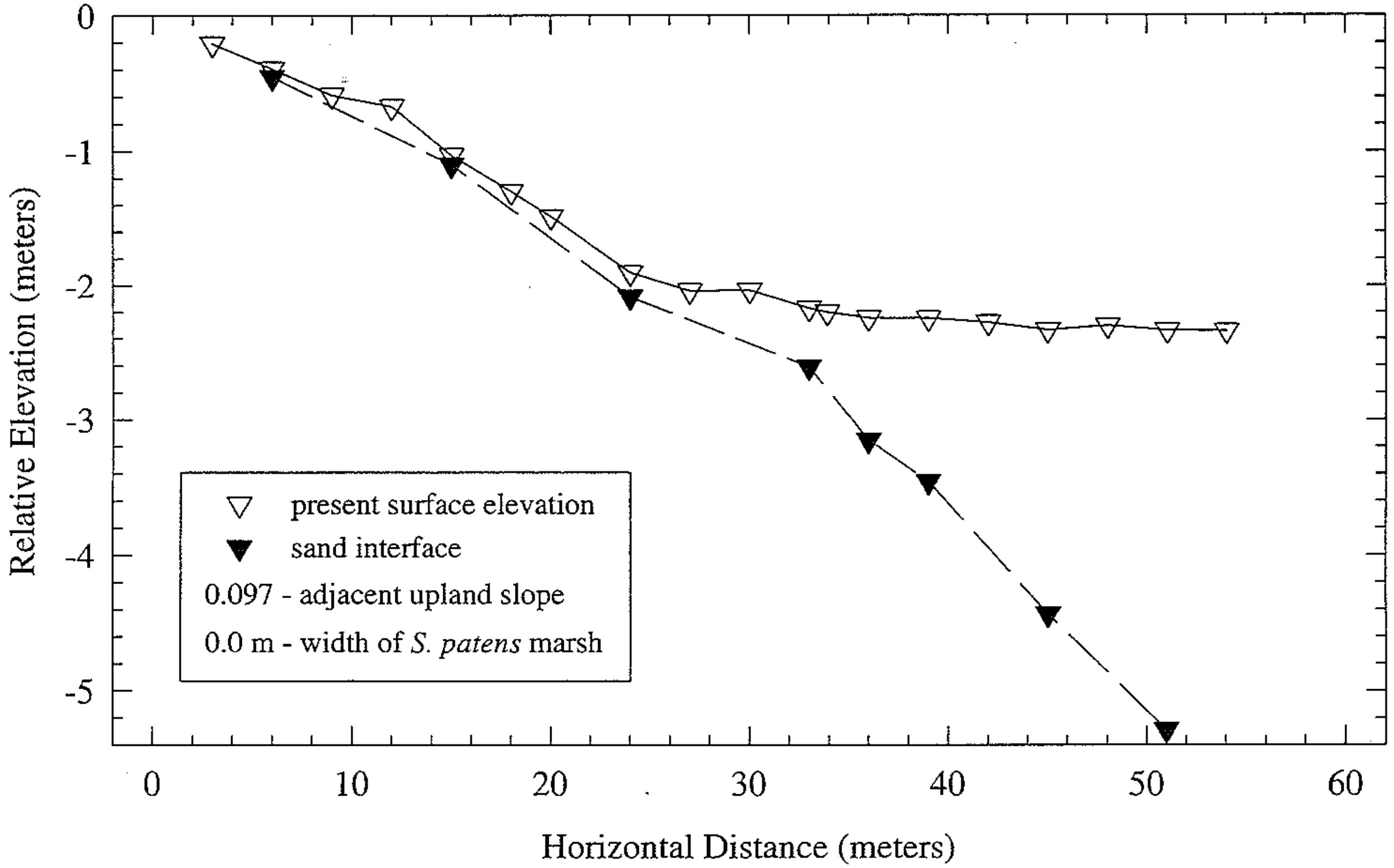
69 Individual Transect Sites


Anderson
▽ present surface elevation
▼ sand interface
0.076 - adjacent upland slope
4 m - width of *S. patens* marsh
Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60

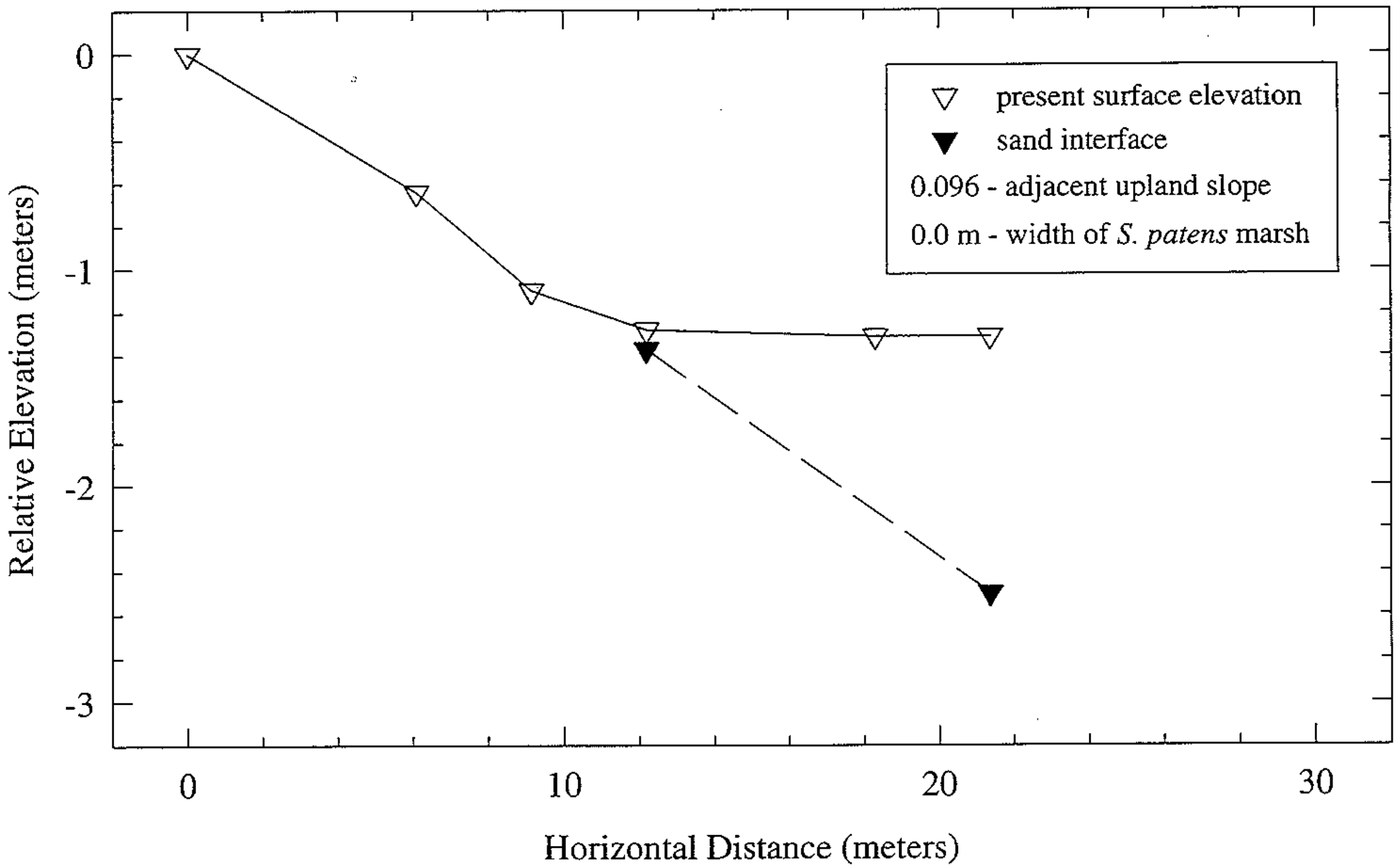
Bay Colony - 1


present surface elevation
sand interface
0.037 - adjacent upland slope
6 m - width of *S. patens* marsh
0
-1
-2
-3
Relative Elevation (meters)
0
10
20
30
40
50
60
Horizontal Distance (meters)

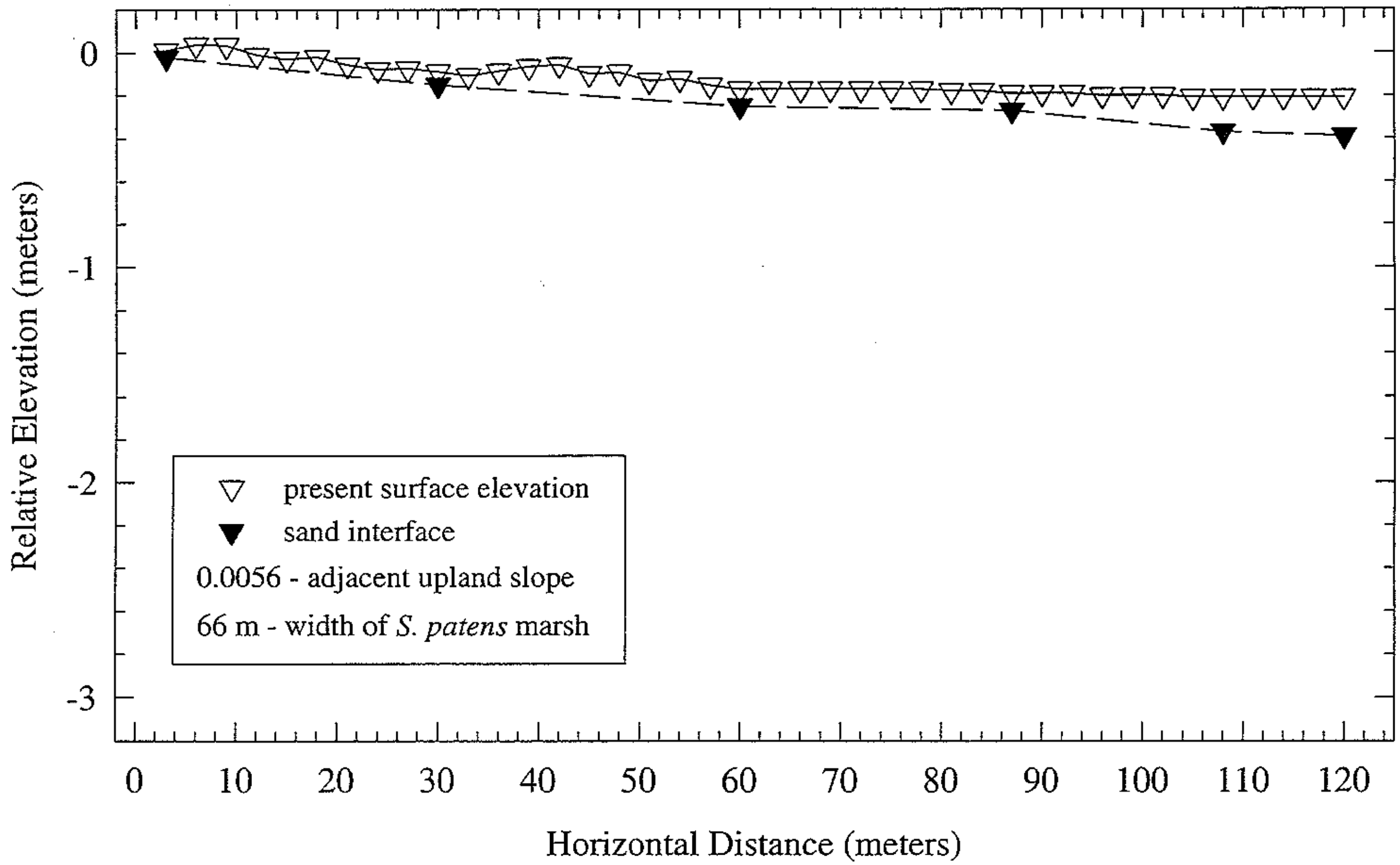
Bay Vista - 1


0
-1
-2
-3
-4
-5
Relative Elevation (meters)
0
10
20
30
40
50
60
Horizontal Distance (meters)
▽ present surface elevation
▼ sand interface
0.097 - adjacent upland slope
0.0 m - width of *S. patens* marsh

Bay Vista - 2


▽ present surface elevation
▼ sand interface
0.096 - adjacent upland slope
0.0 m - width of *S. patens* marsh
Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30

Bethany Bay - 1



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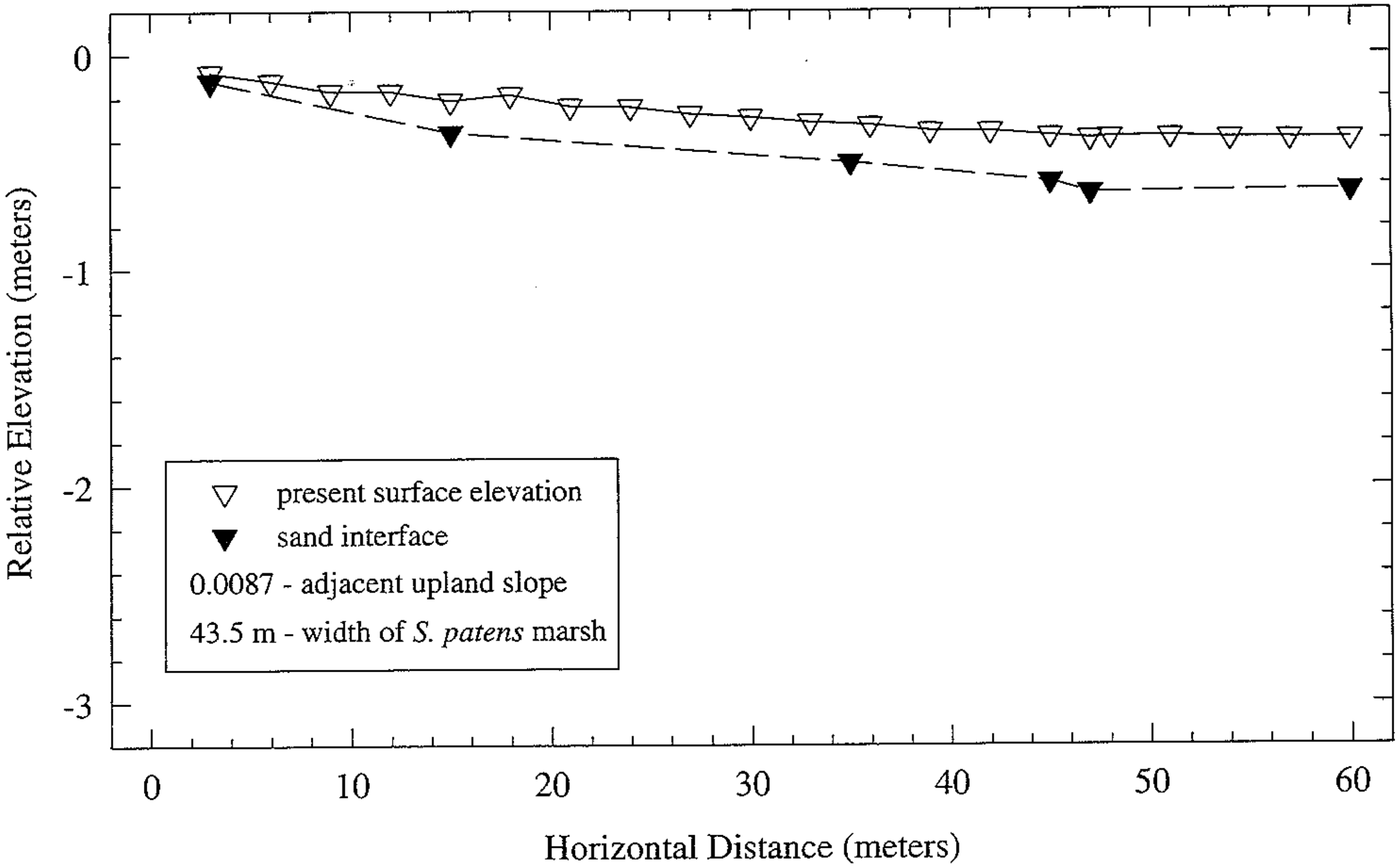
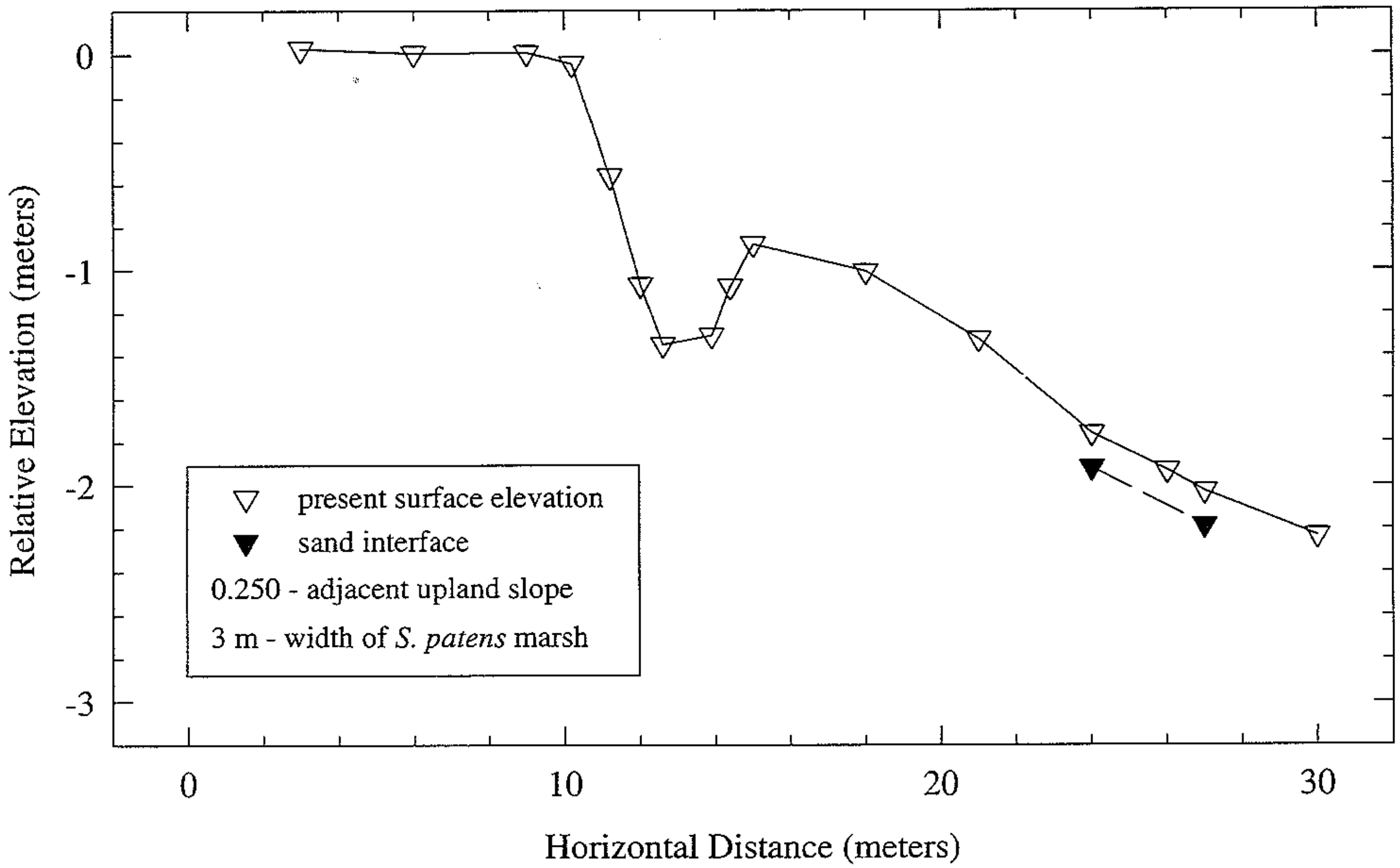
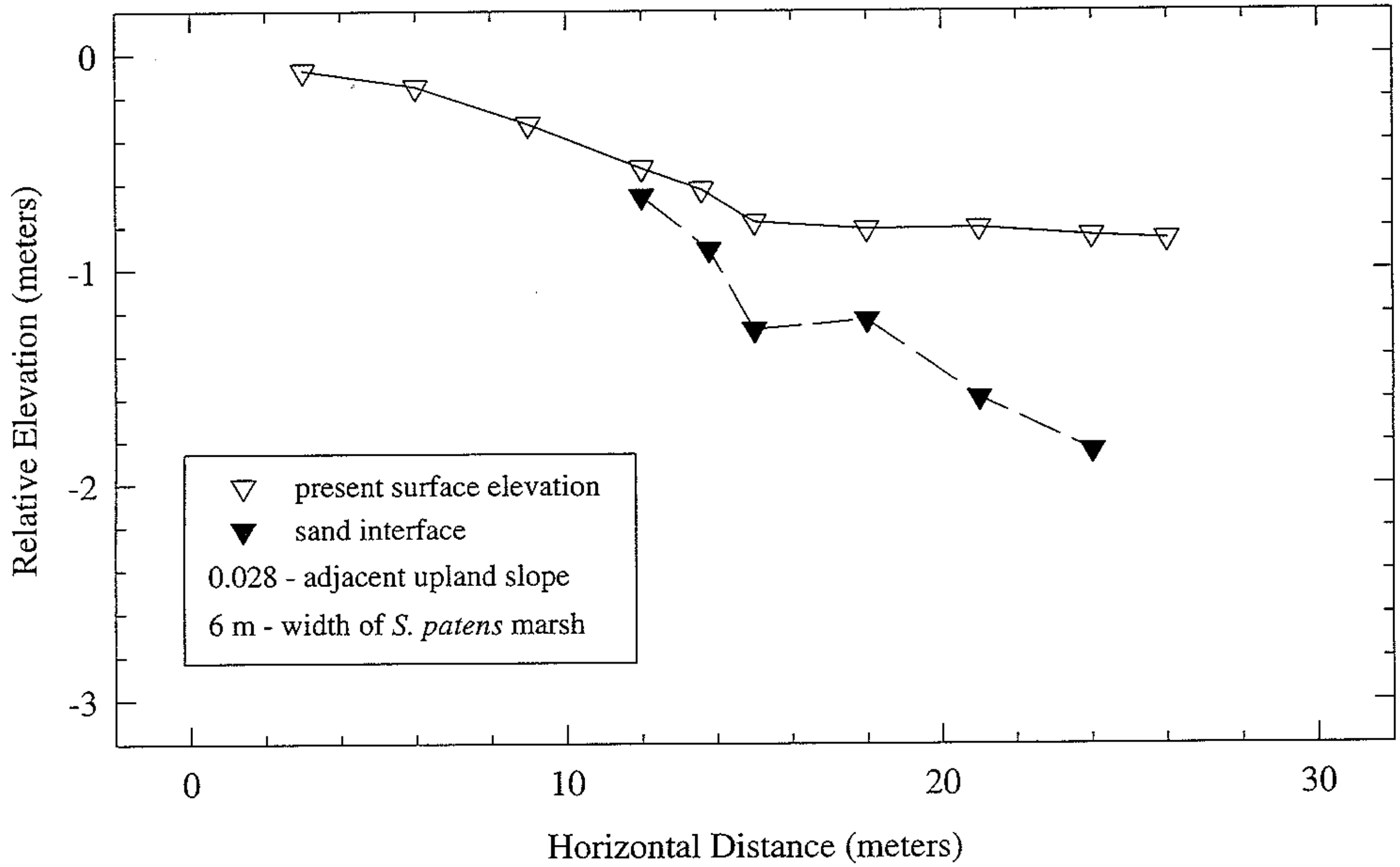


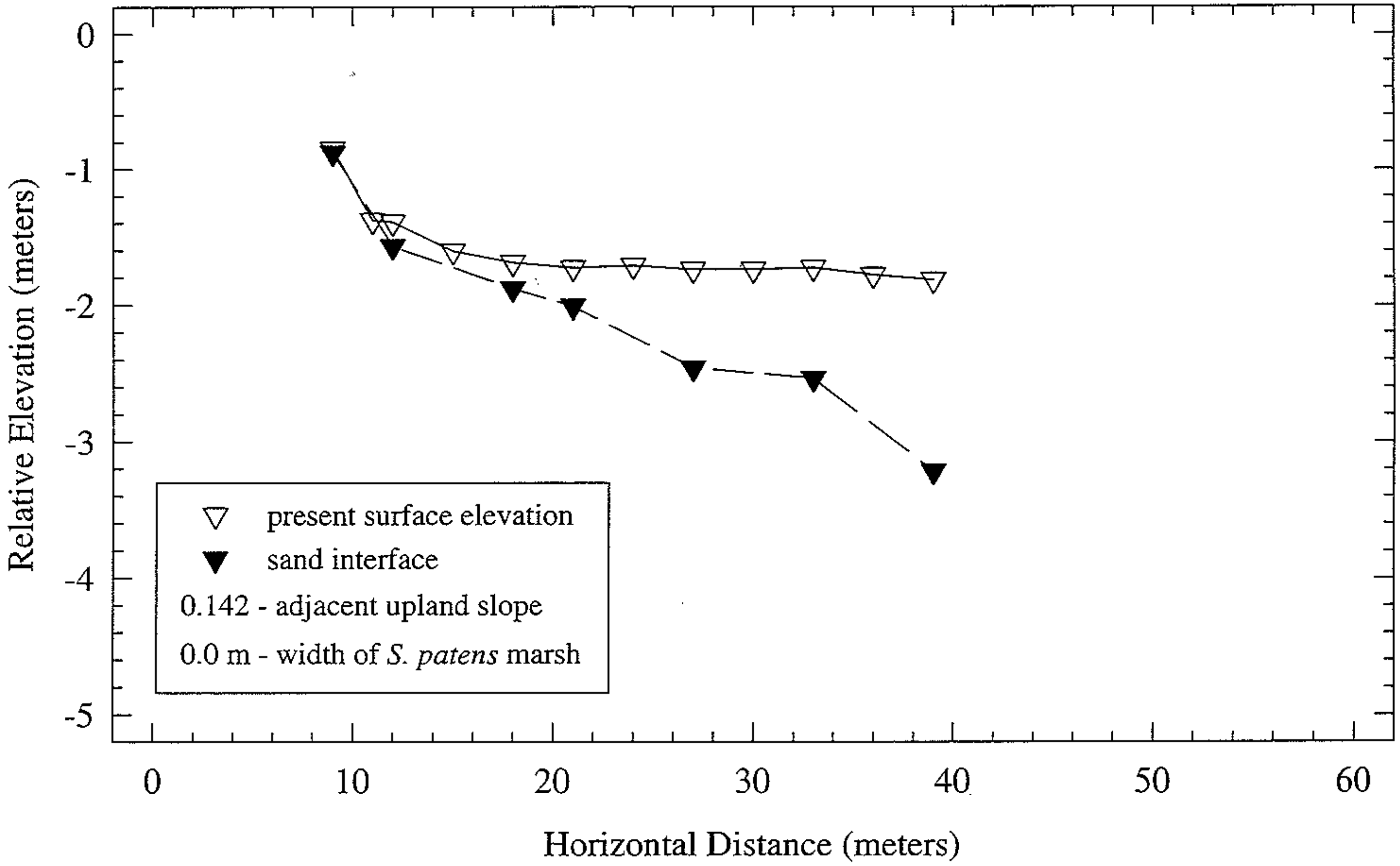
Bethany Bay - 3



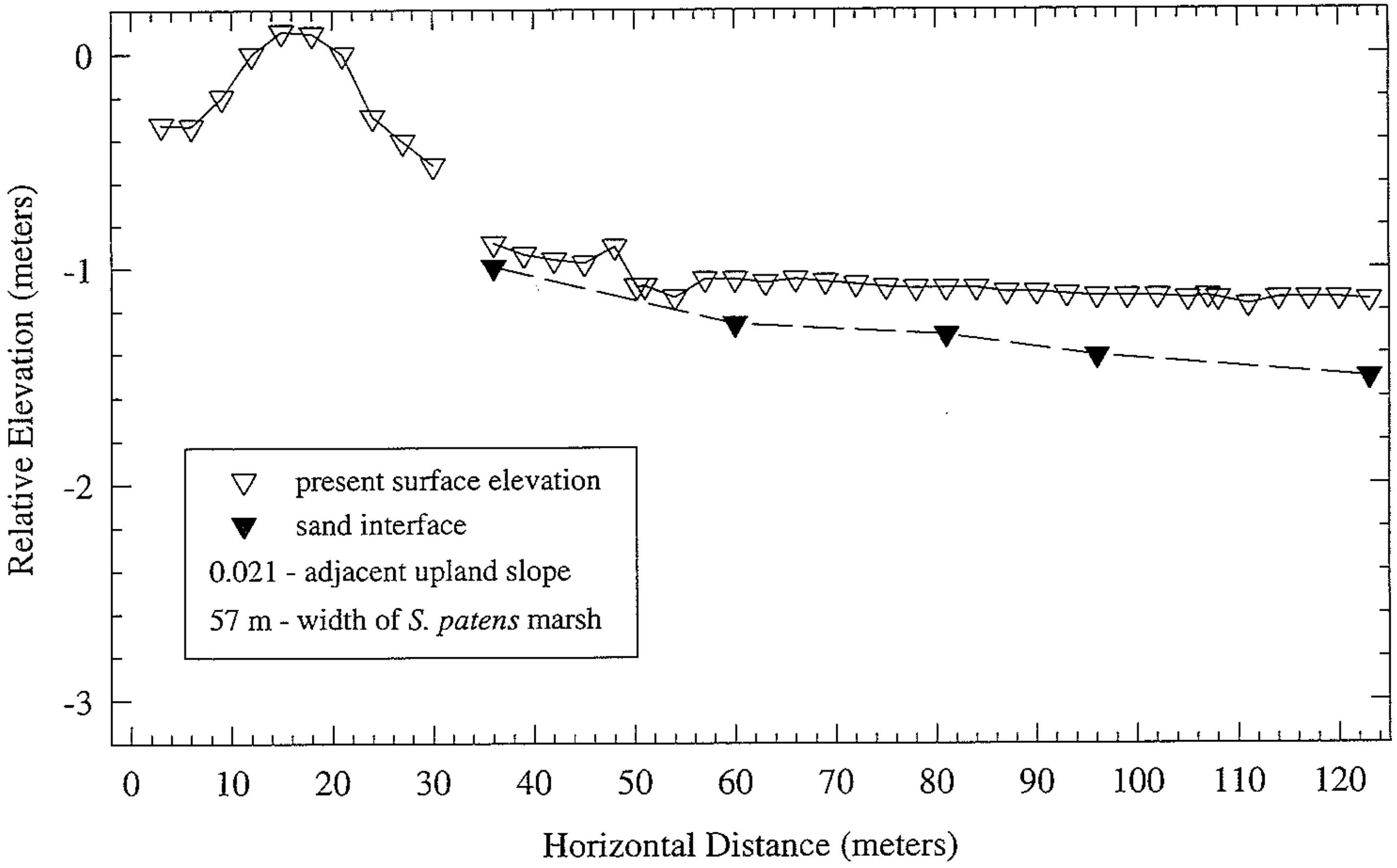
Bethany Bay - 4



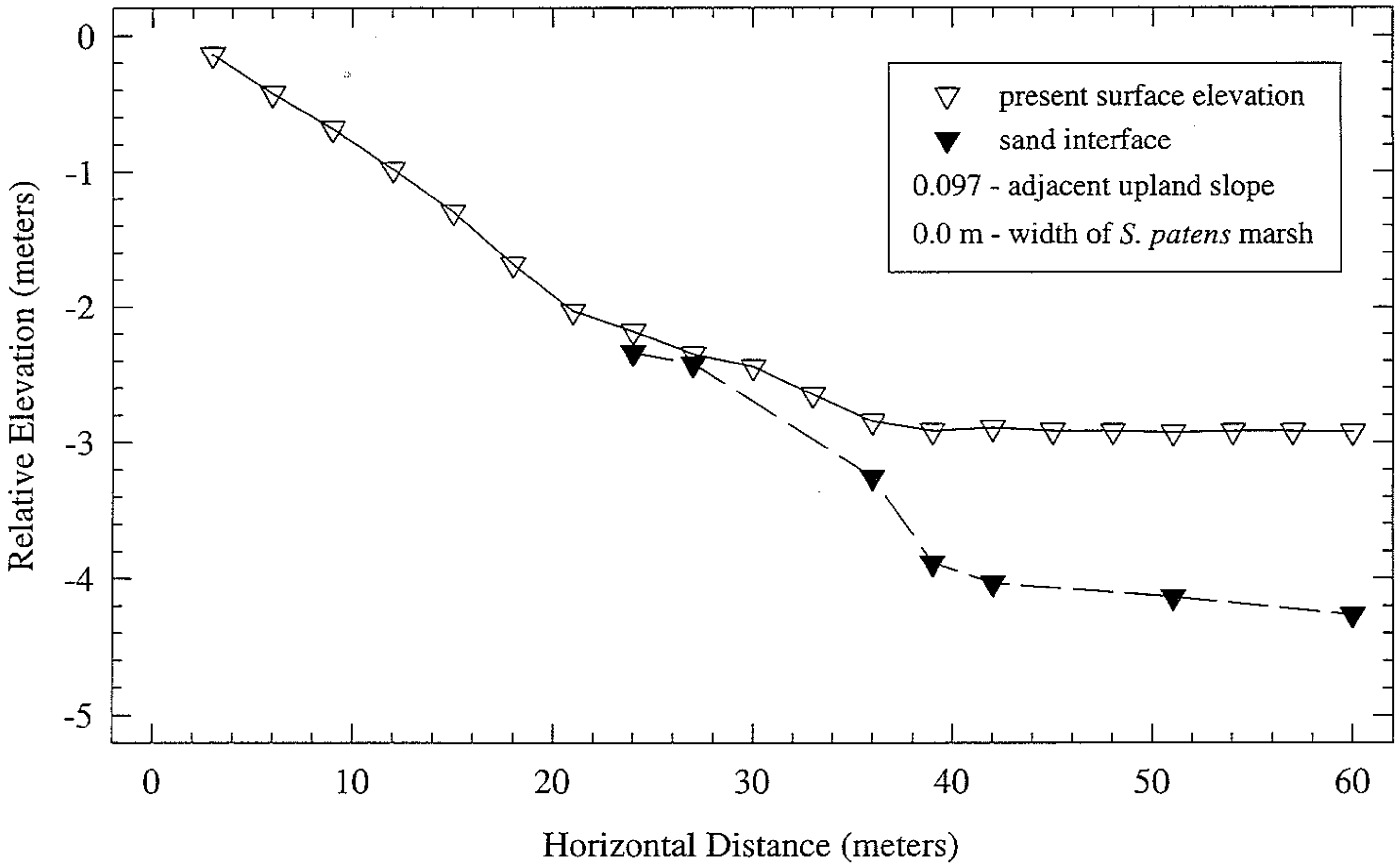
Bethany Forest - 1



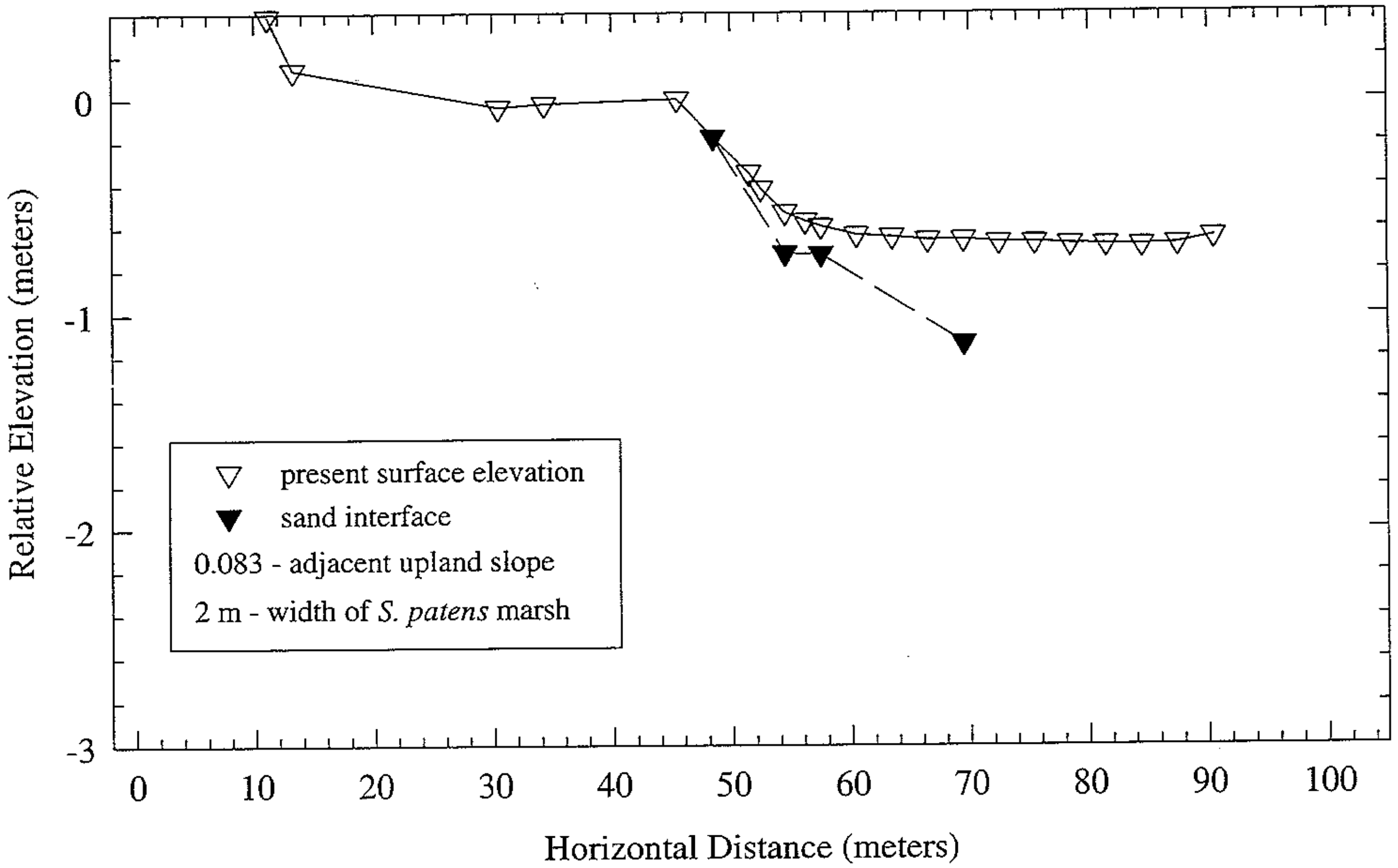
Blackwater Point



Boat House Cove - 2

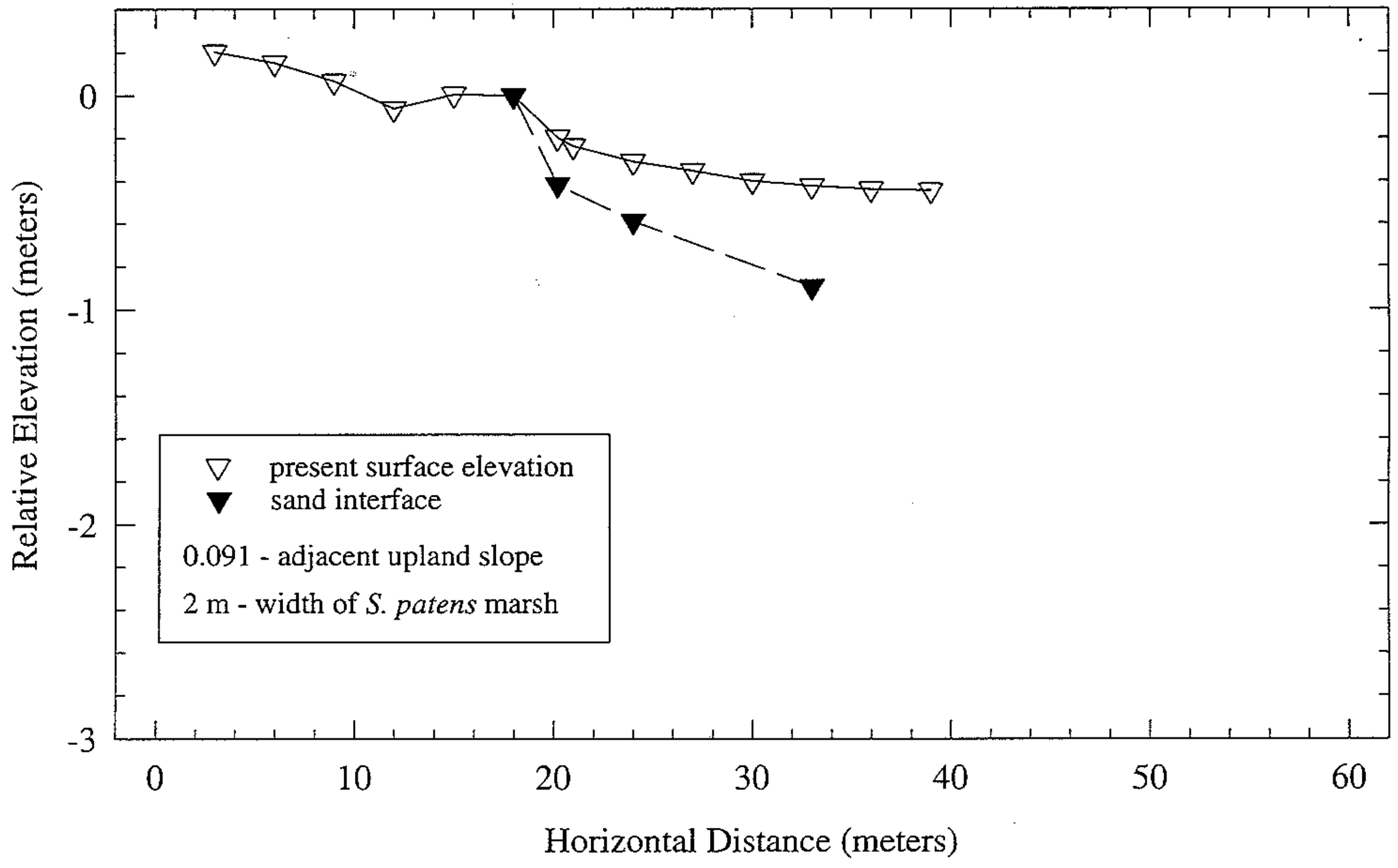


Burton - 1



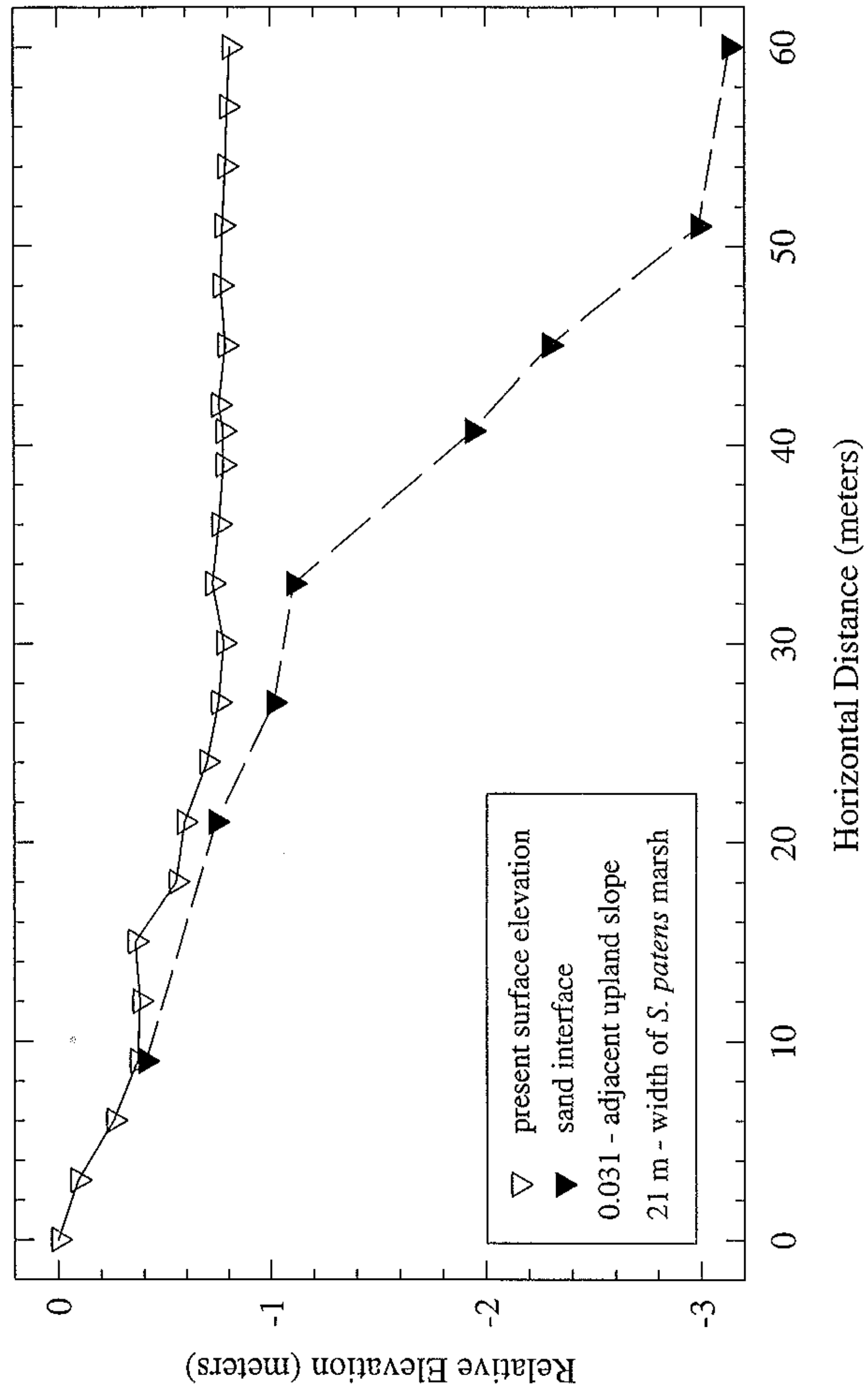
Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60
70
80
90
100
▽ present surface elevation
▼ sand interface
0.083 - adjacent upland slope
2 m - width of *S. patens* marsh

Burton - 2

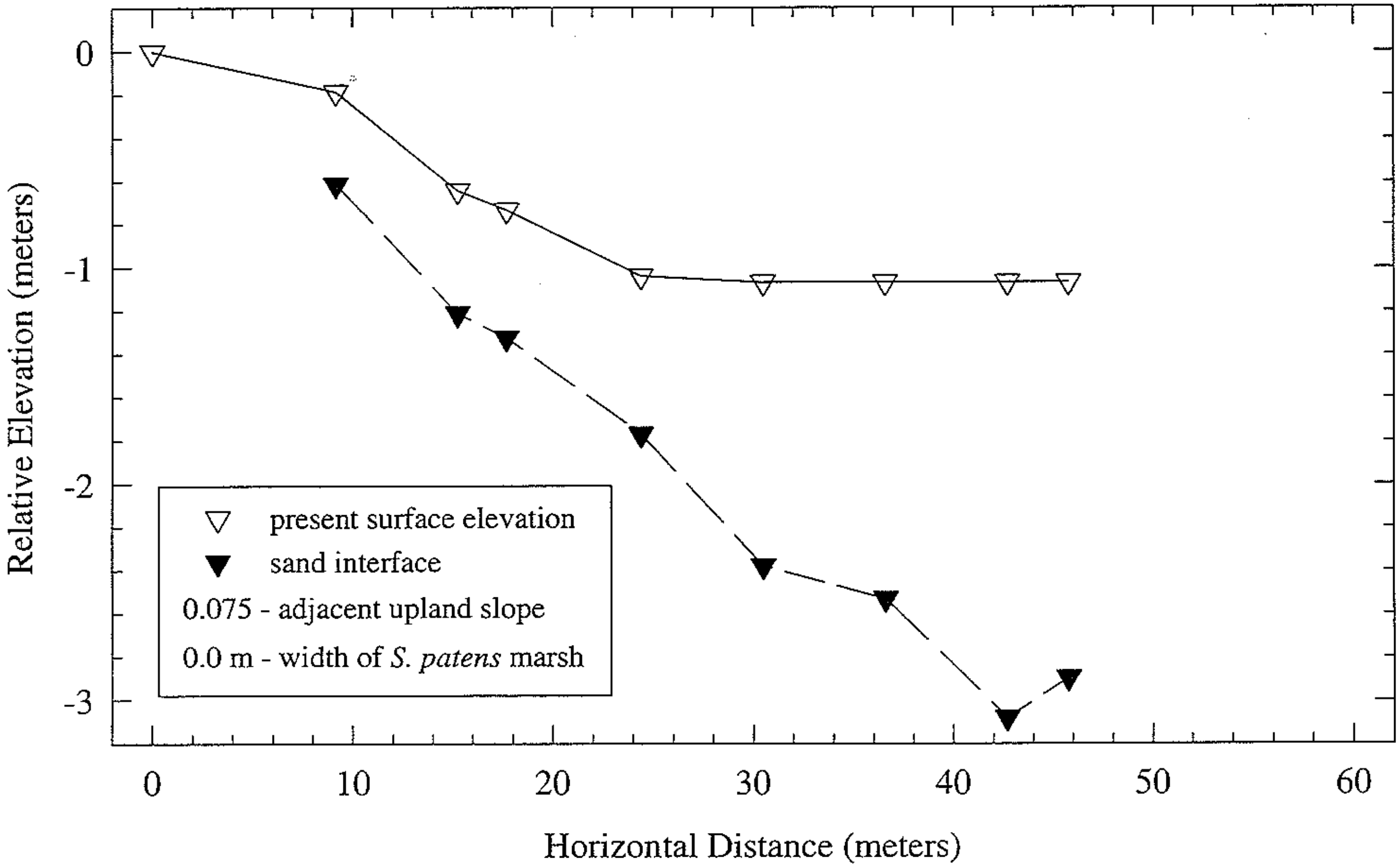


Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60
▽ present surface elevation
▼ sand interface
0.091 - adjacent upland slope
2 m - width of *S. patens* marsh

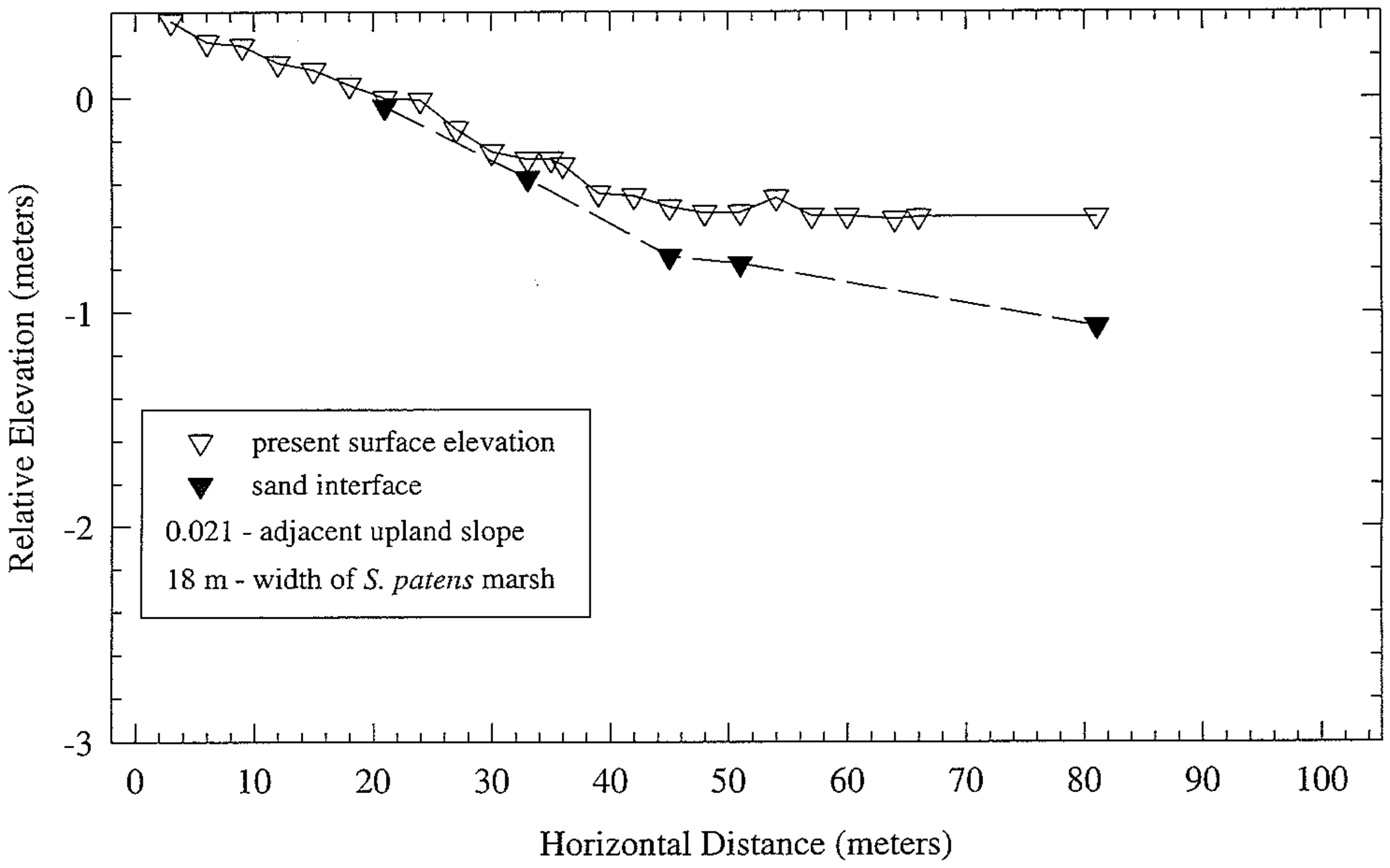
Camp Arrowhead - 2



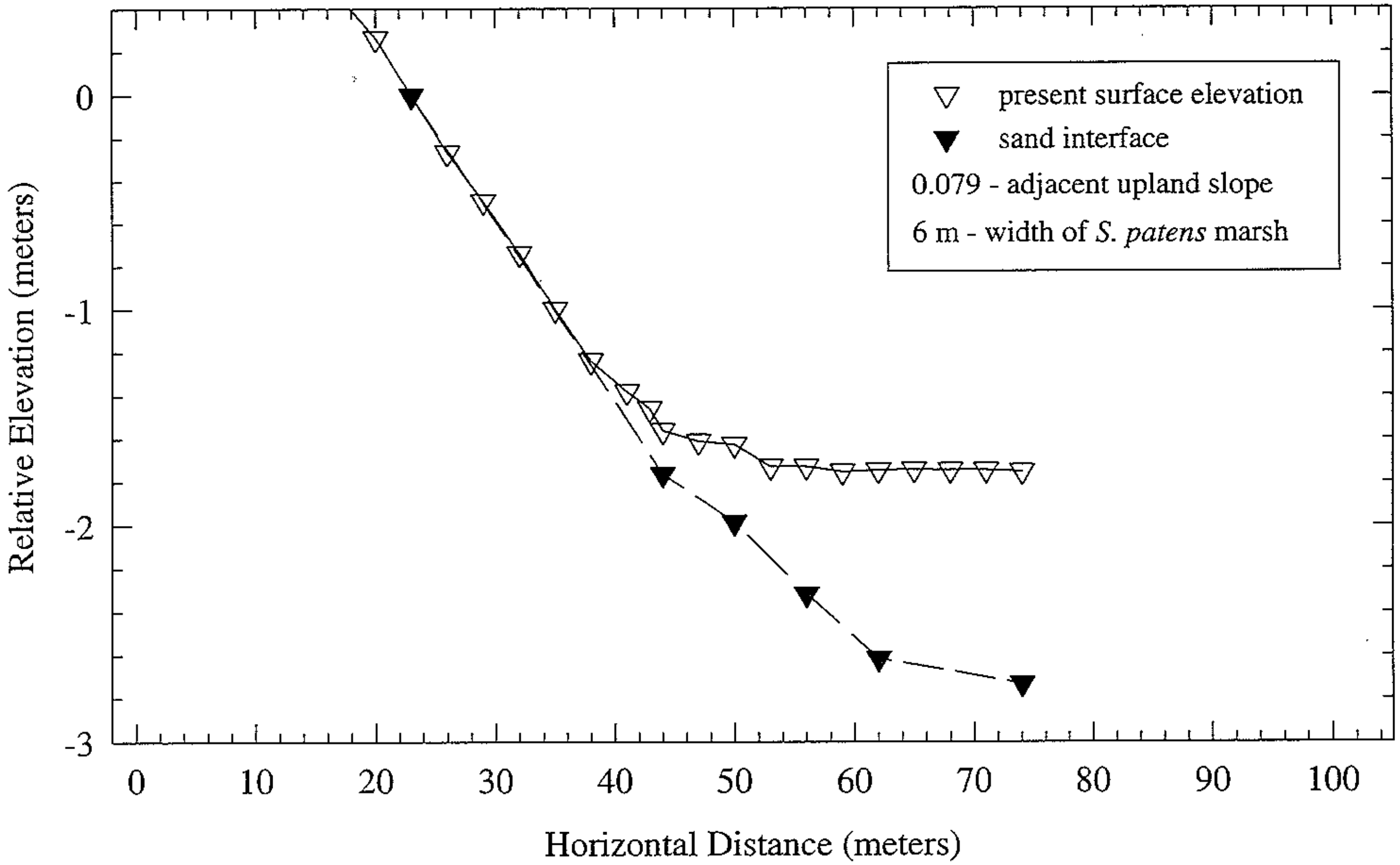
Canary Creek - 1



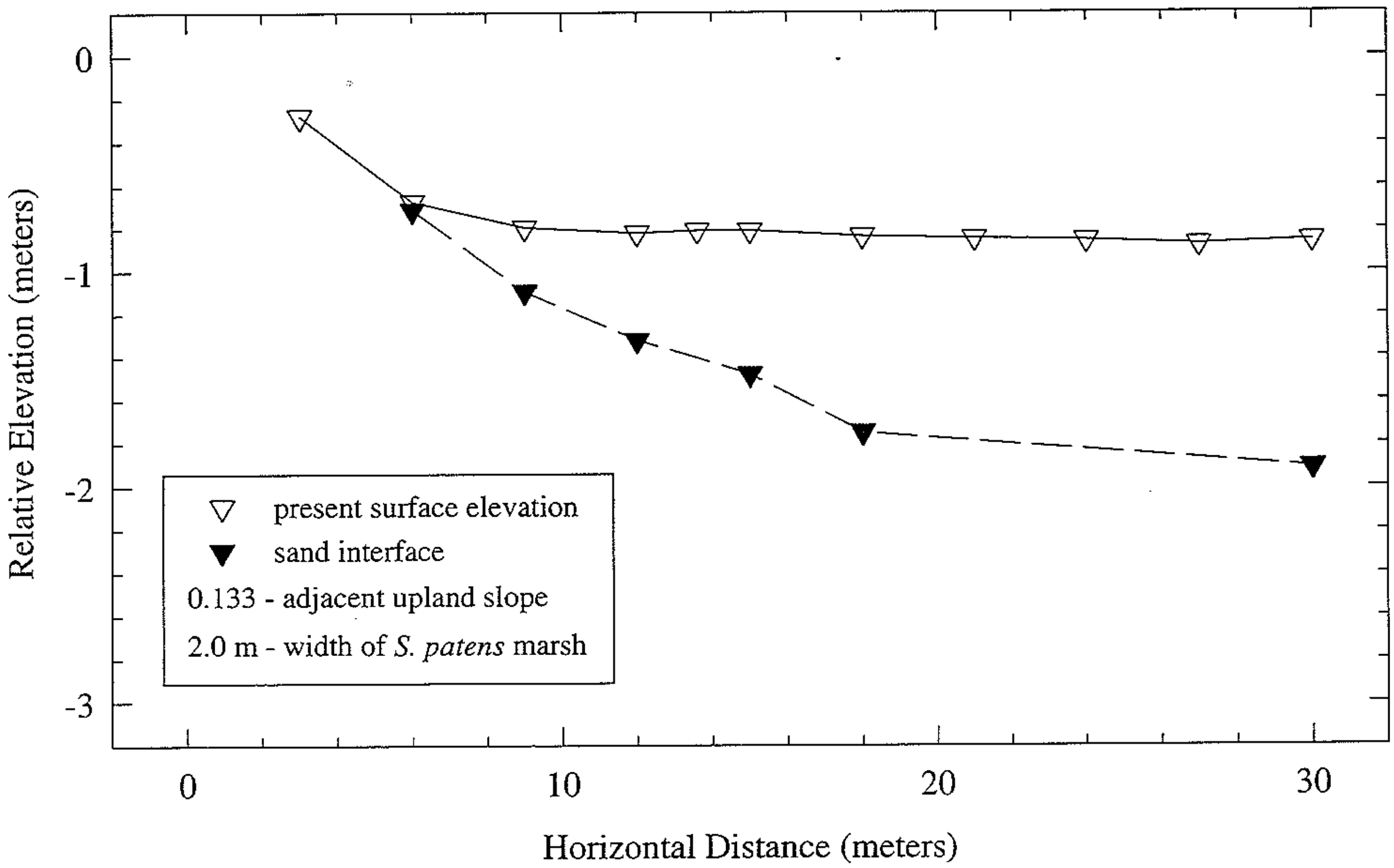
Delaware Wildlands - 1



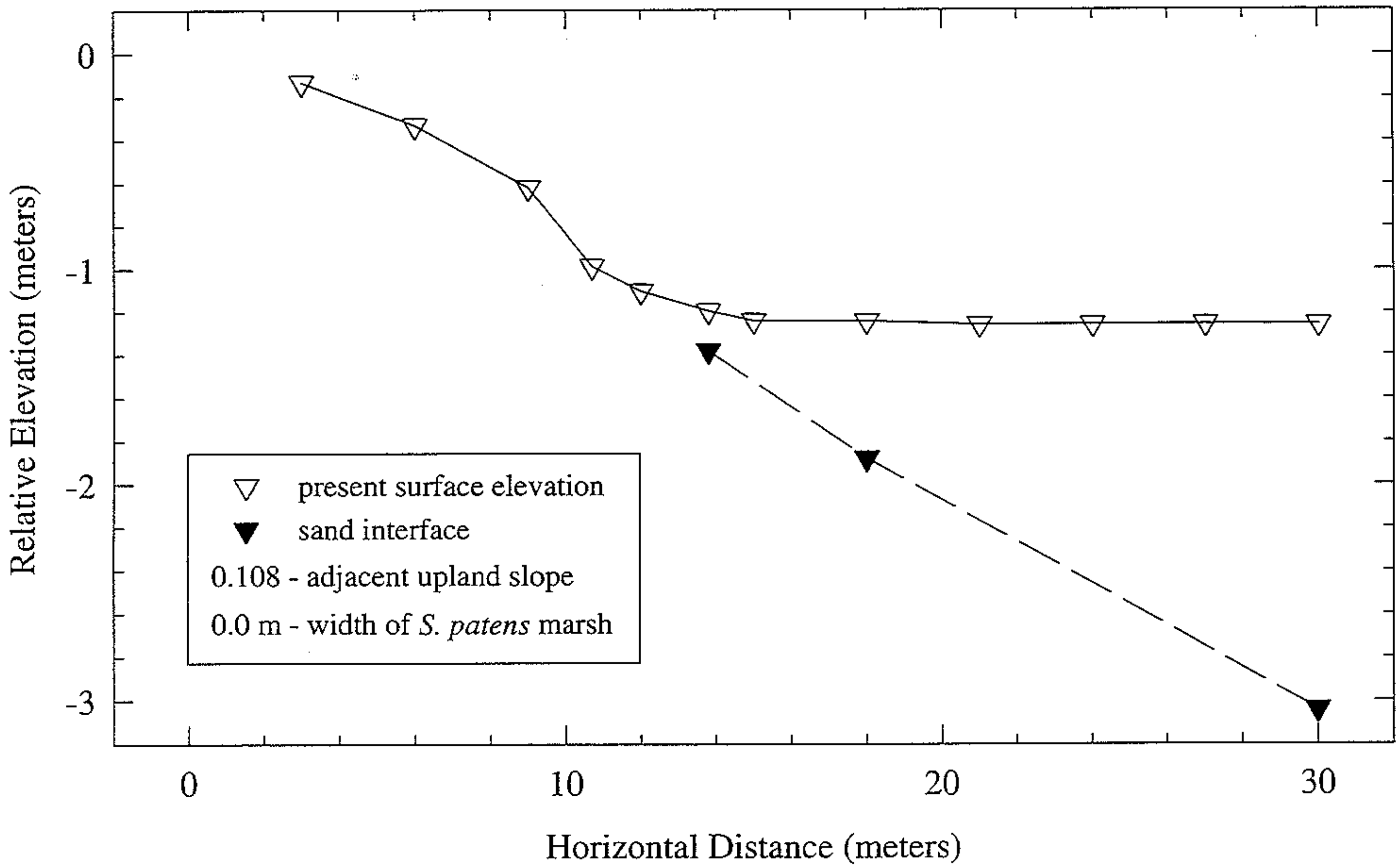
Delaware Wildlands - 2

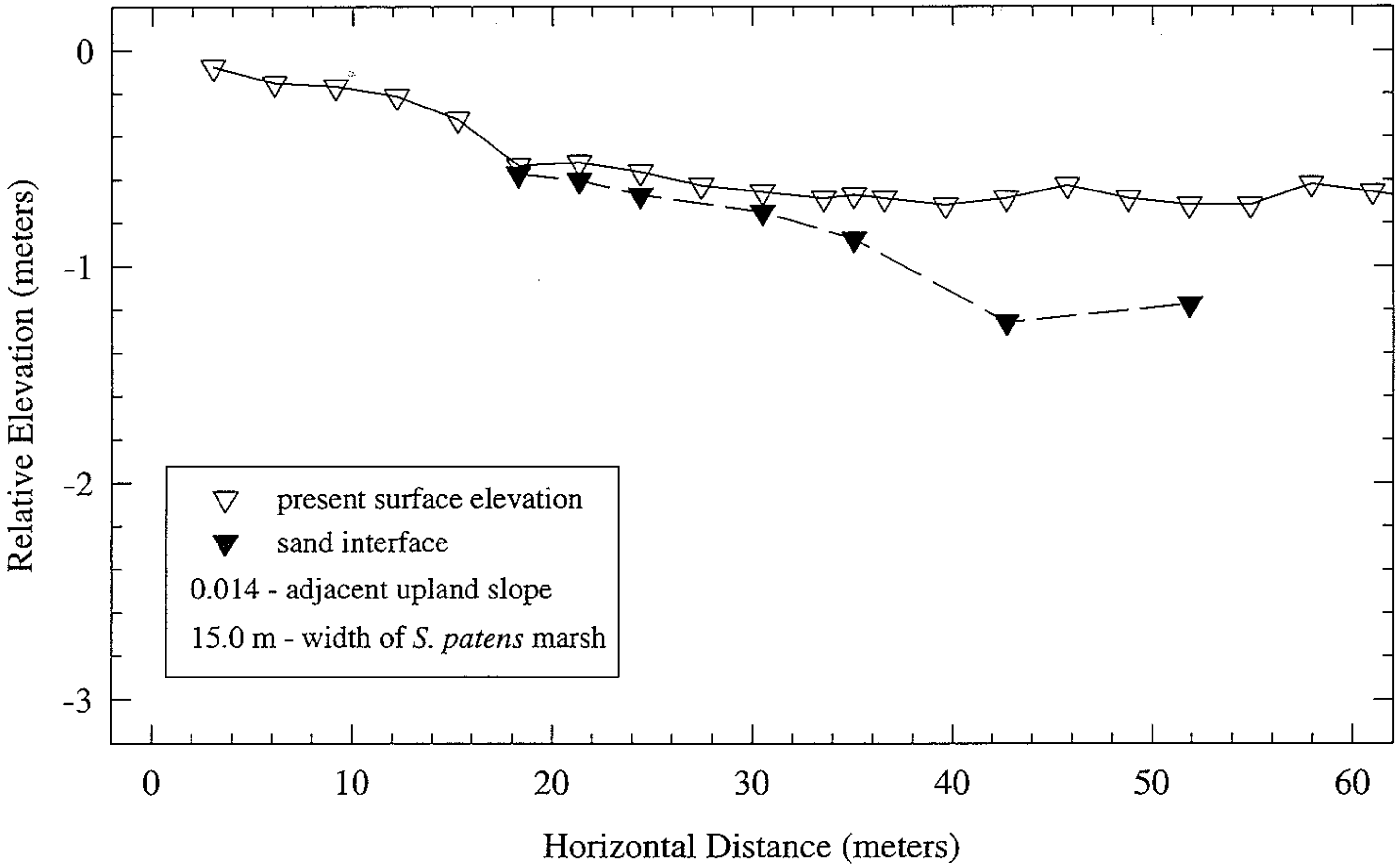


Dodd - 1


Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
▽ present surface elevation
▼ sand interface
0.133 - adjacent upland slope
2.0 m - width of *S. patens* marsh

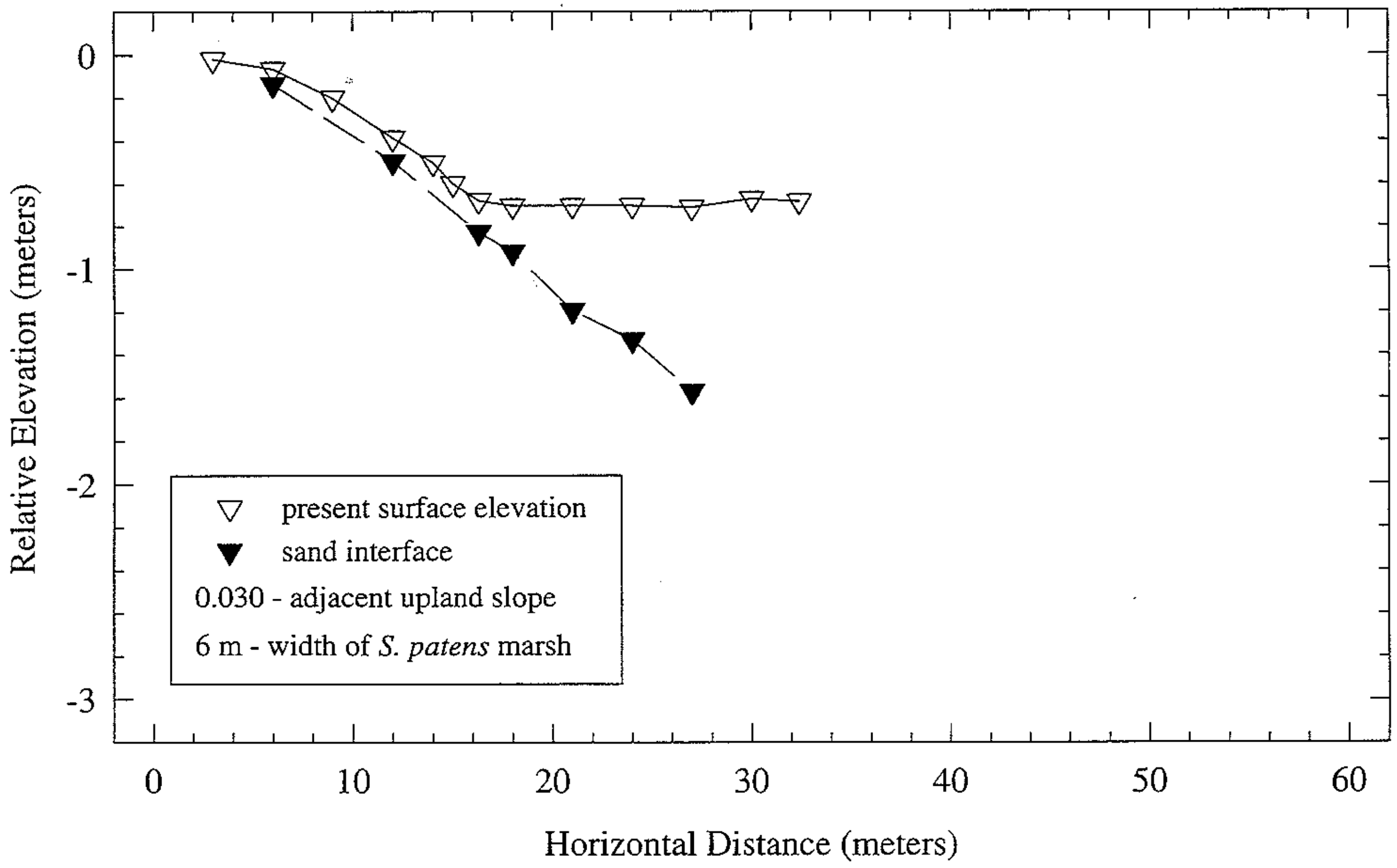
Dodd - 2


0
-1
-2
-3
Relative Elevation (meters)
0
10
20
30
Horizontal Distance (meters)
▽ present surface elevation
▼ sand interface
0.108 - adjacent upland slope
0.0 m - width of *S. patens* marsh



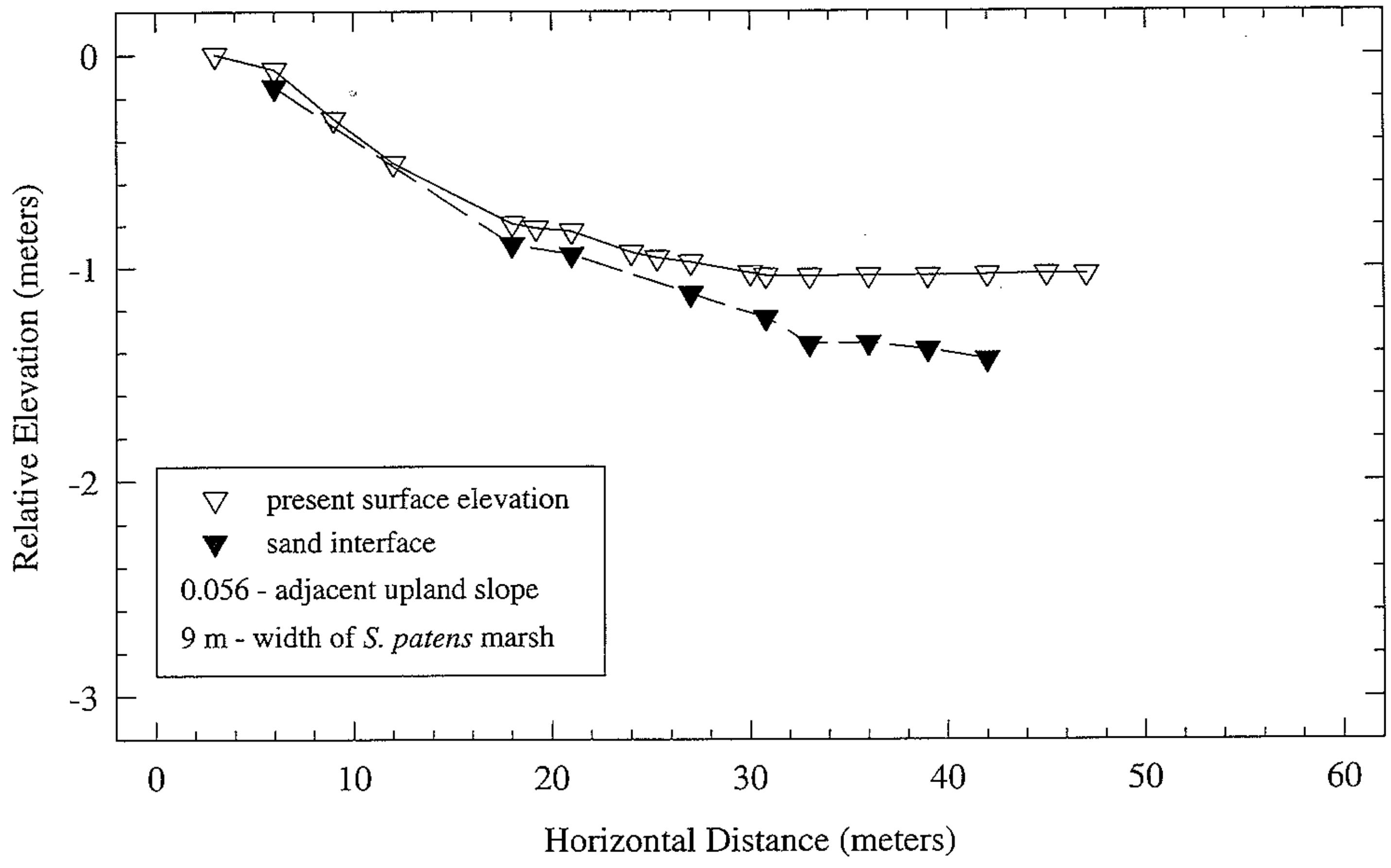
Draper - 1

Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60
▽ present surface elevation
▼ sand interface
0.014 - adjacent upland slope
15.0 m - width of *S. patens* marsh


Draper - 2

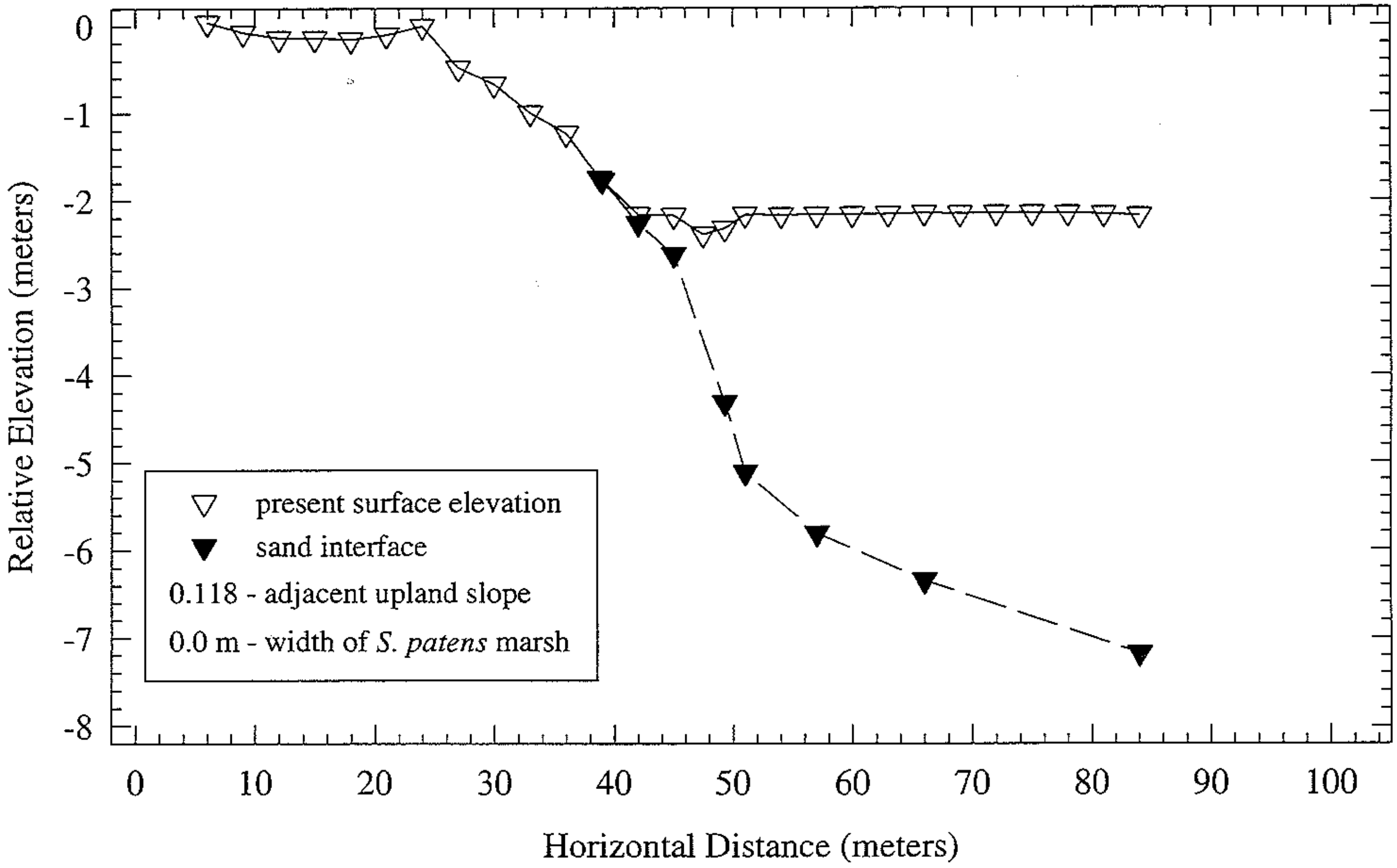
present surface elevation
sand interface
0.030 - adjacent upland slope
6 m - width of S. patens marsh
Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60

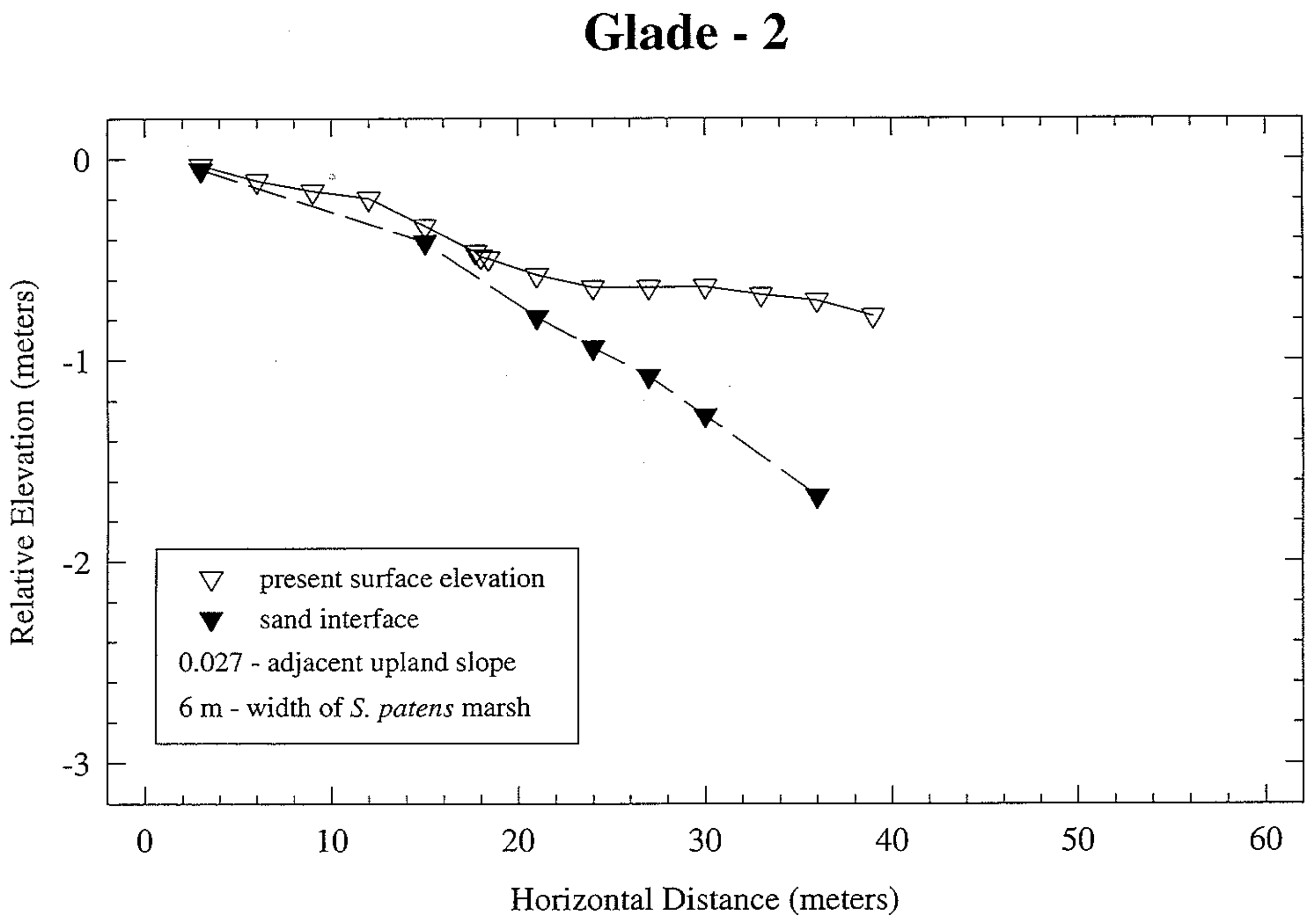
Draper - 3



0
-1
-2
-3
Relative Elevation (meters)
0
10
20
30
40
50
60
Horizontal Distance (meters)
▽ present surface elevation
▼ sand interface
0.056 - adjacent upland slope
9 m - width of *S. patens* marsh

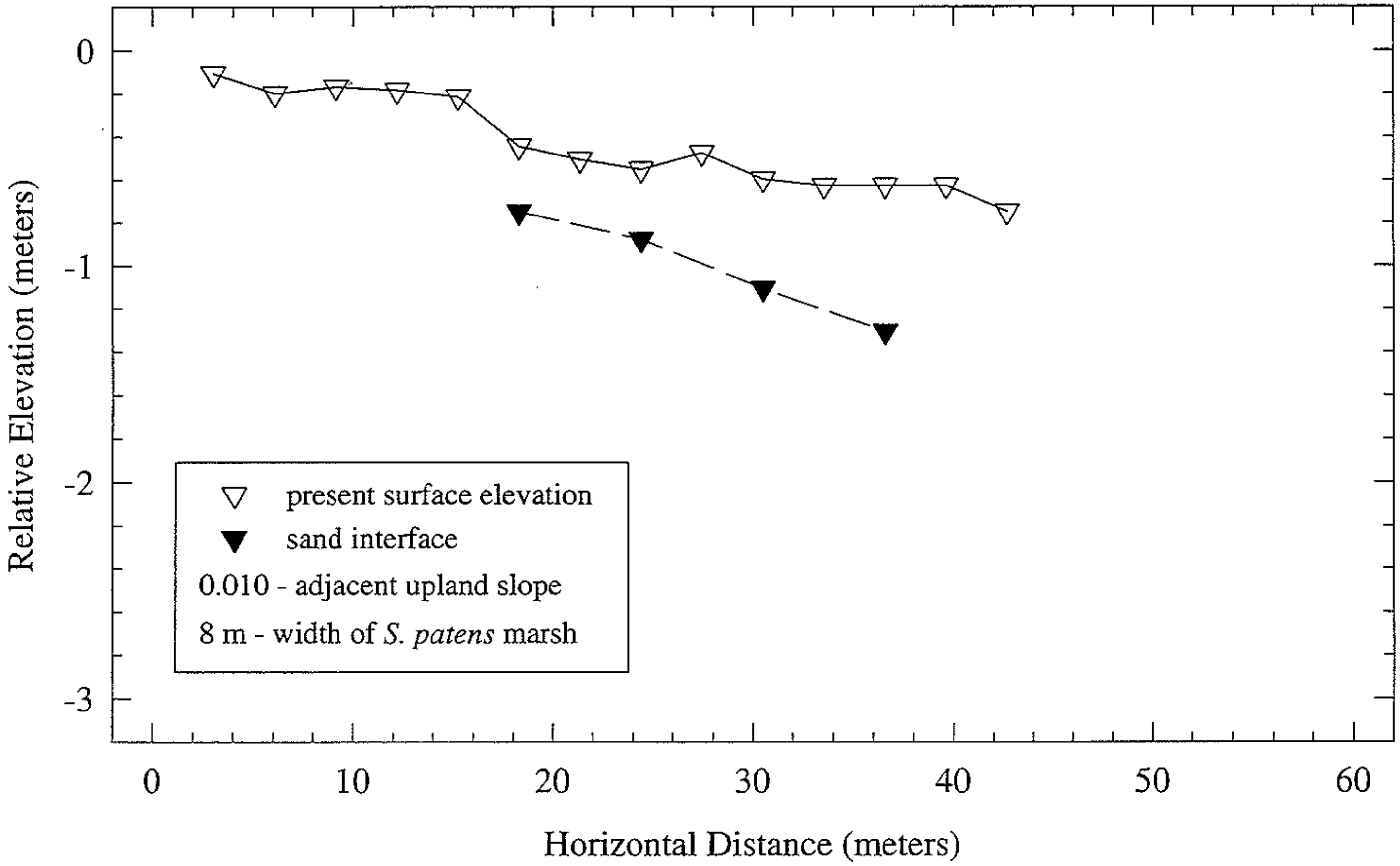
Glade - 1


0
-1
-2
-3
-4
-5
-6
-7
-8
Relative Elevation (meters)
0
10
20
30
40
50
60
70
80
90
100
Horizontal Distance (meters)
▽ present surface elevation
▼ sand interface
0.118 - adjacent upland slope
0.0 m - width of *S. patens* marsh

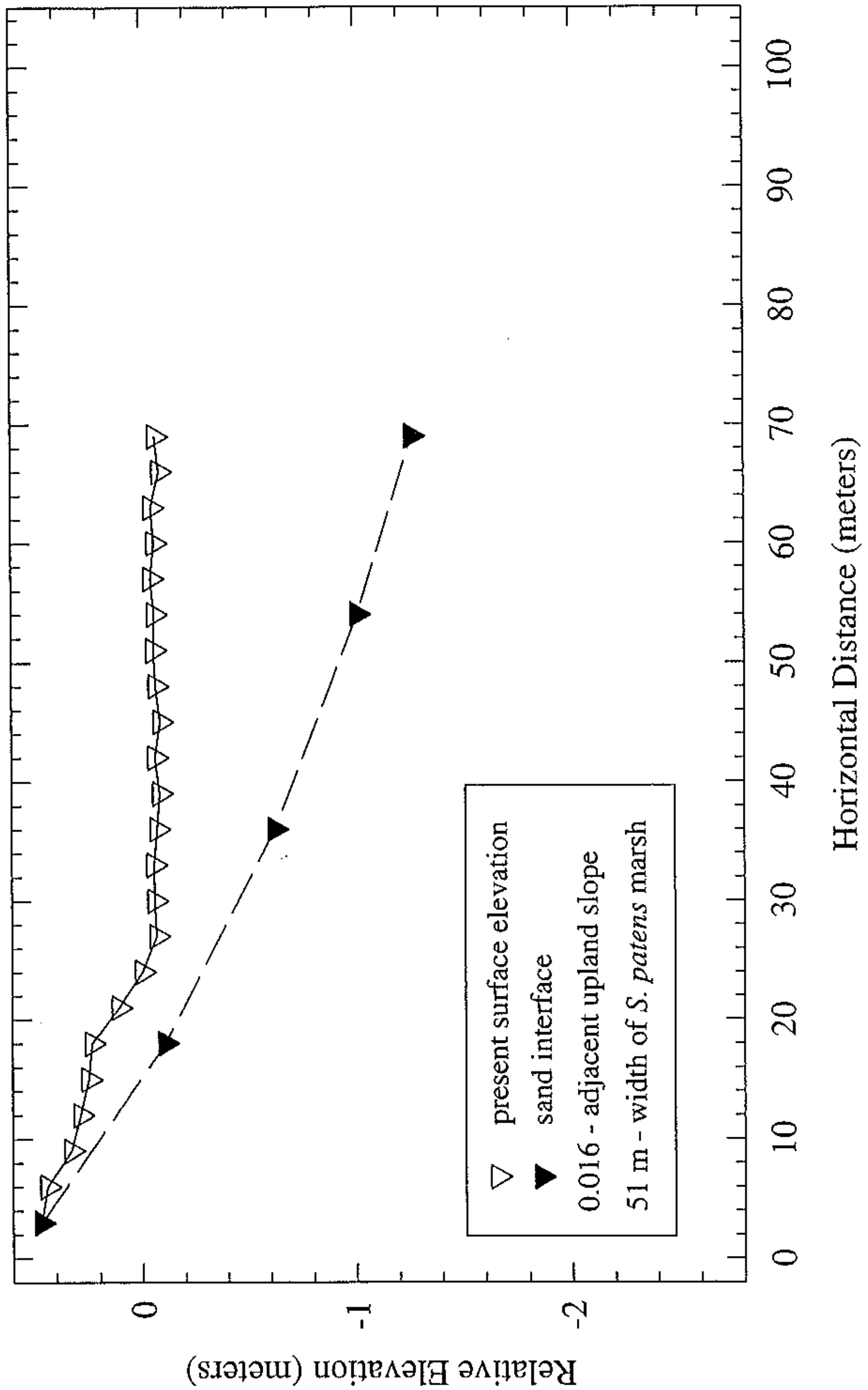

Glade - 2

Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60
▽ present surface elevation
▼ sand interface
0.027 - adjacent upland slope
6 m - width of *S. patens* marsh

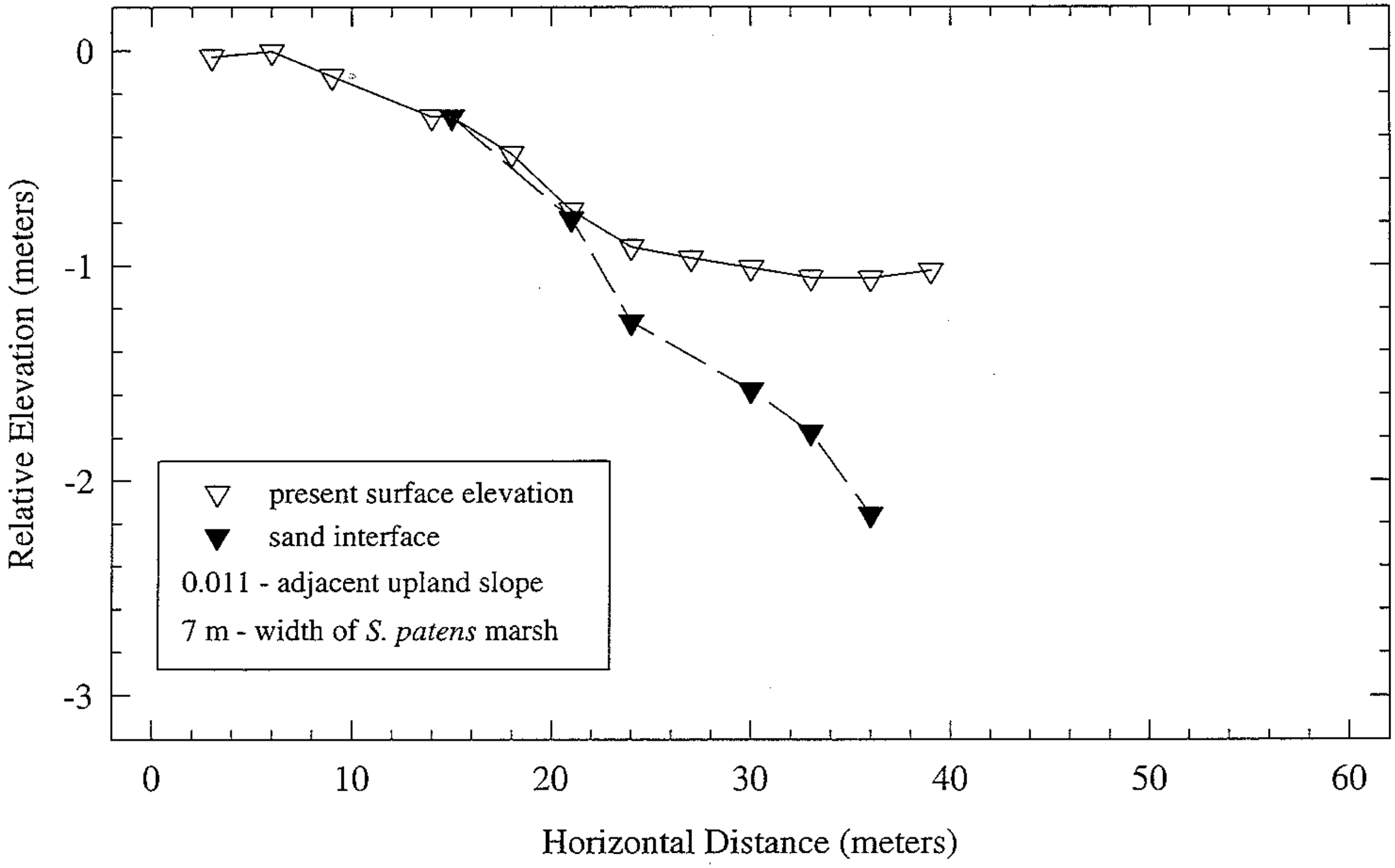
Hercules - 2


0
-1
-2
-3
Relative Elevation (meters)
0
10
20
30
40
50
60
Horizontal Distance (meters)
▽ present surface elevation
▼ sand interface
0.010 - adjacent upland slope
8 m - width of *S. patens* marsh

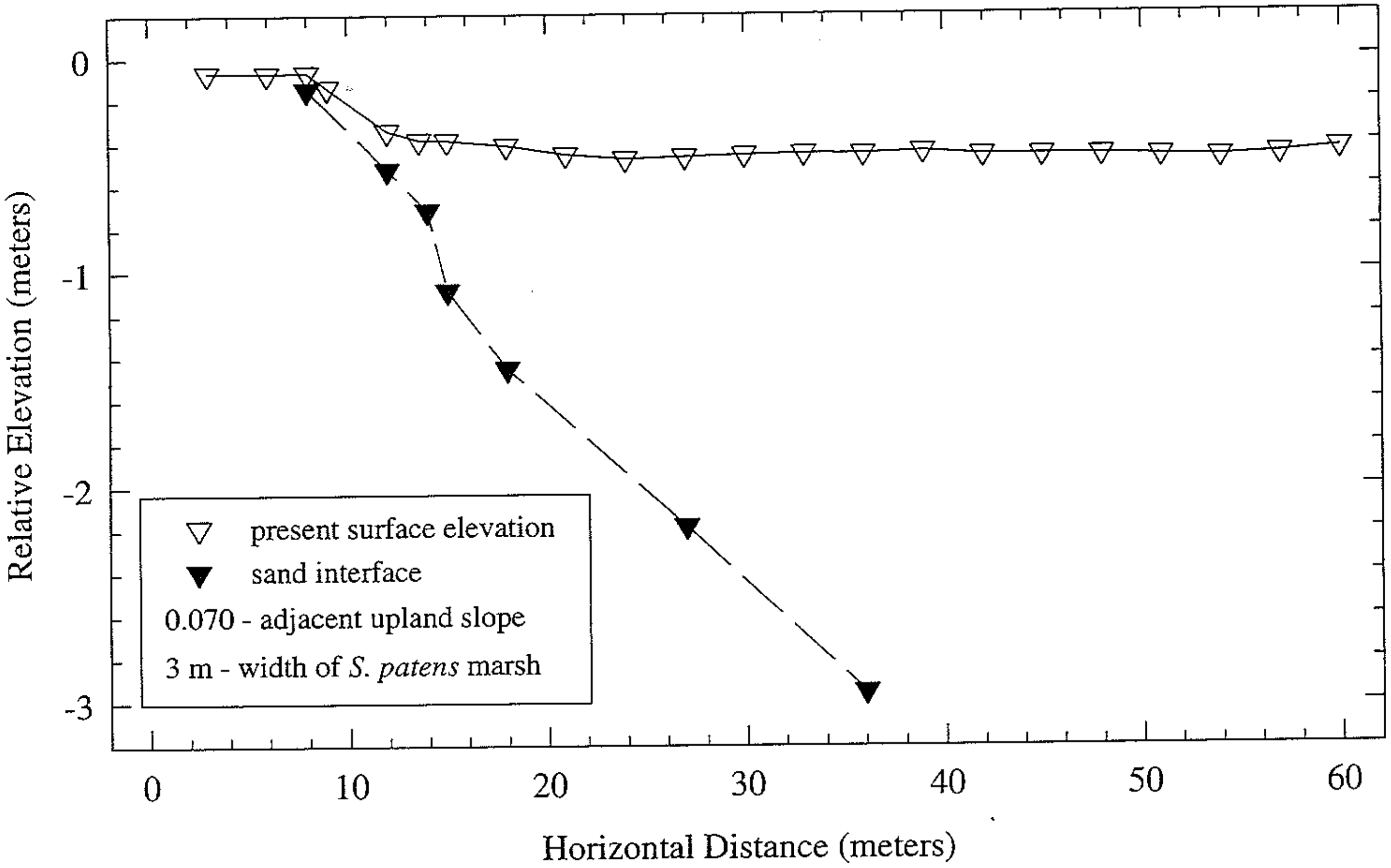
Horse Island



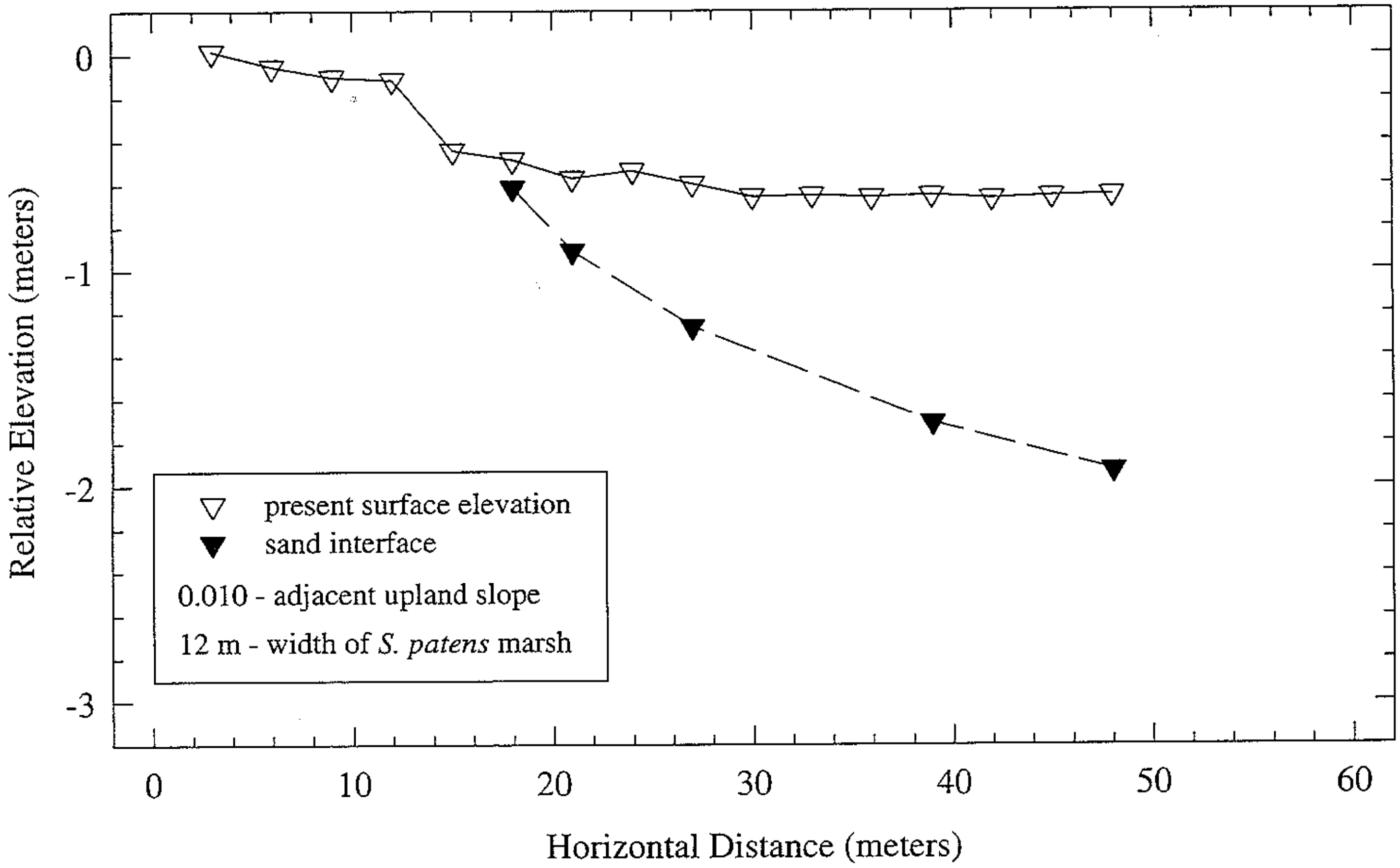
Indian Landing



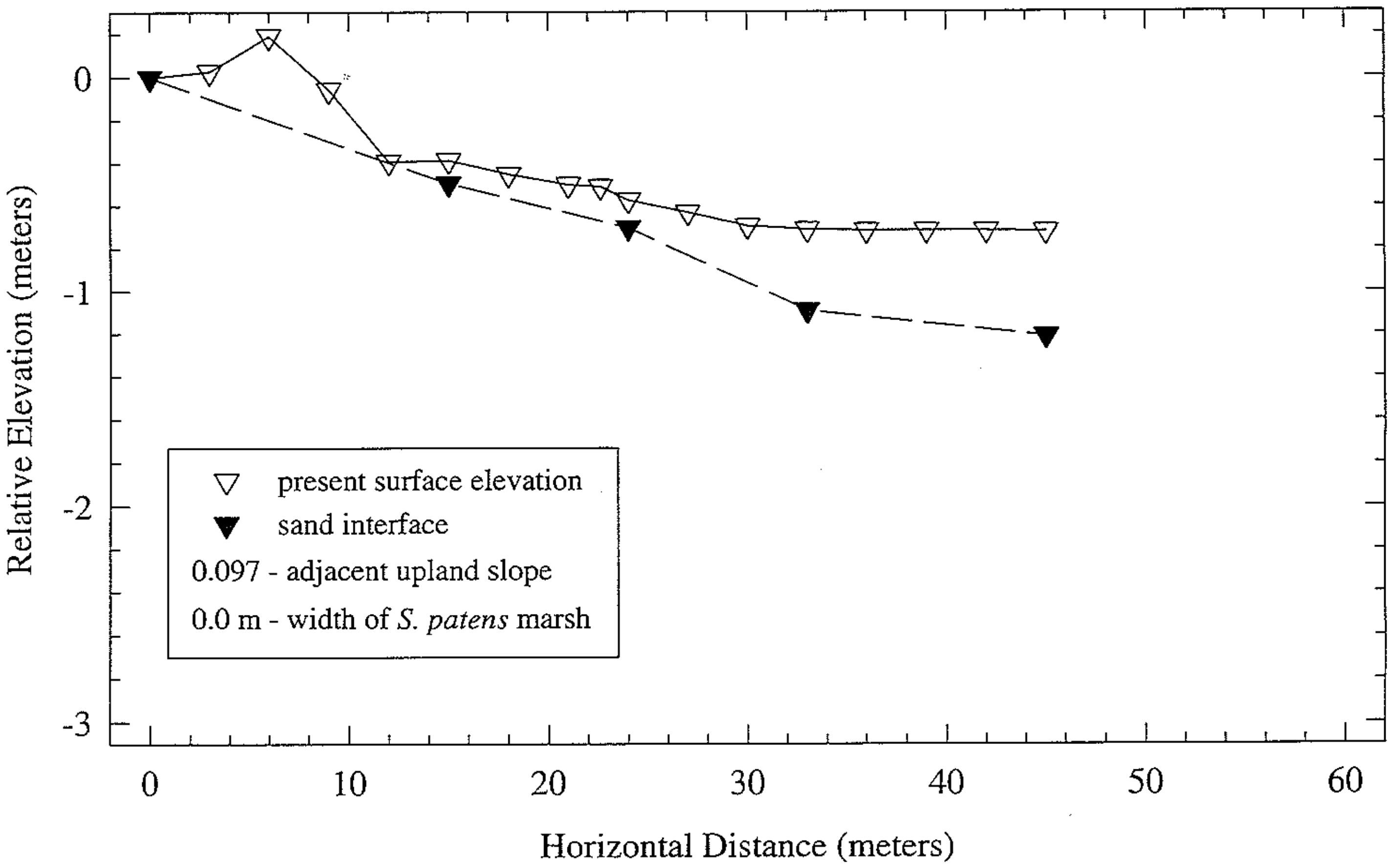
Island Field - 1



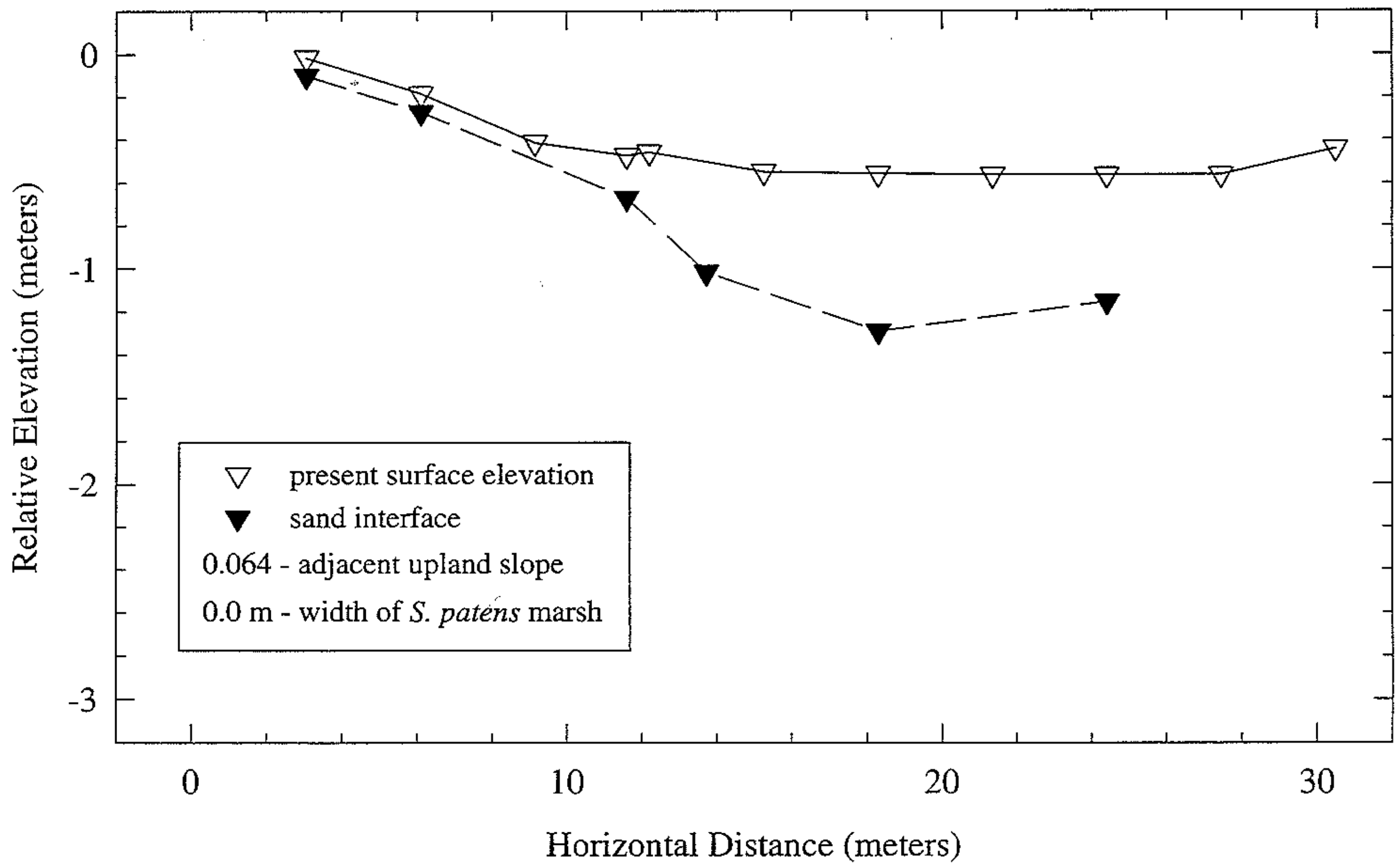
Island Field - 2



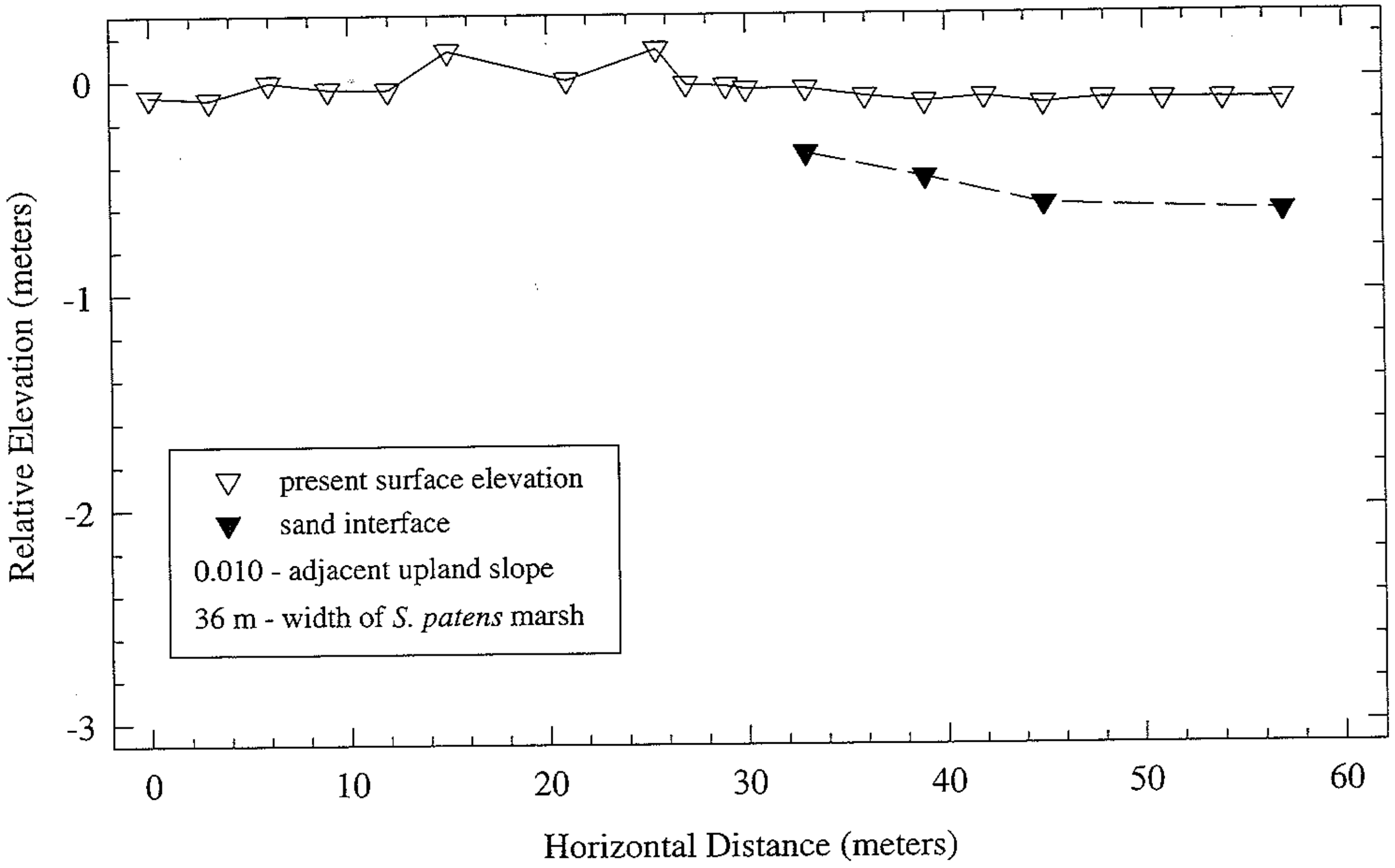
Lingo Point

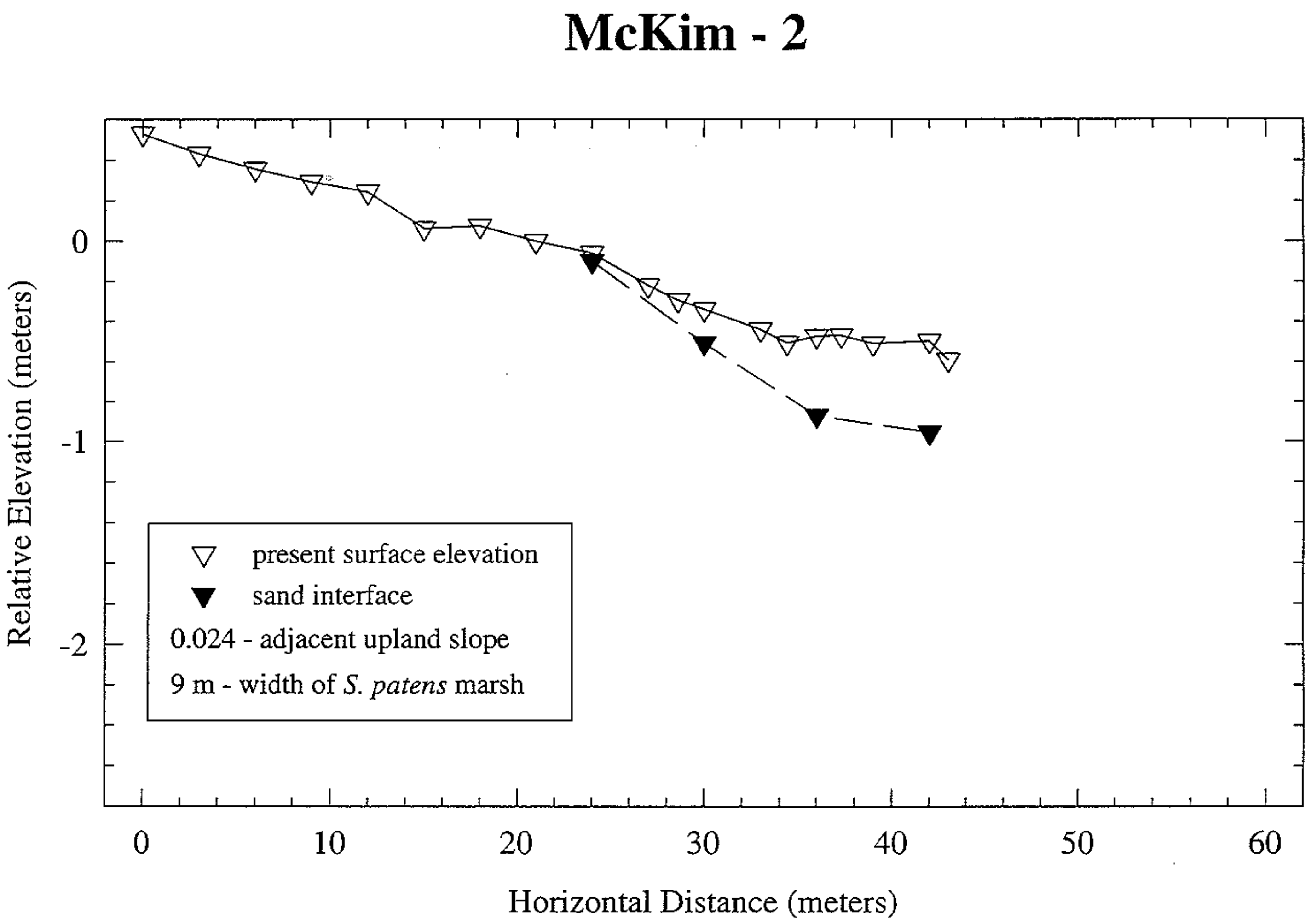

▽ present surface elevation
▼ sand interface
0.097 - adjacent upland slope
0.0 m - width of *S. patens* marsh
Relative Elevation (meters)
0
-1
-2
-3
0
10
20
30
40
50
60
Horizontal Distance (meters)

Little Assawoman Bay - 1

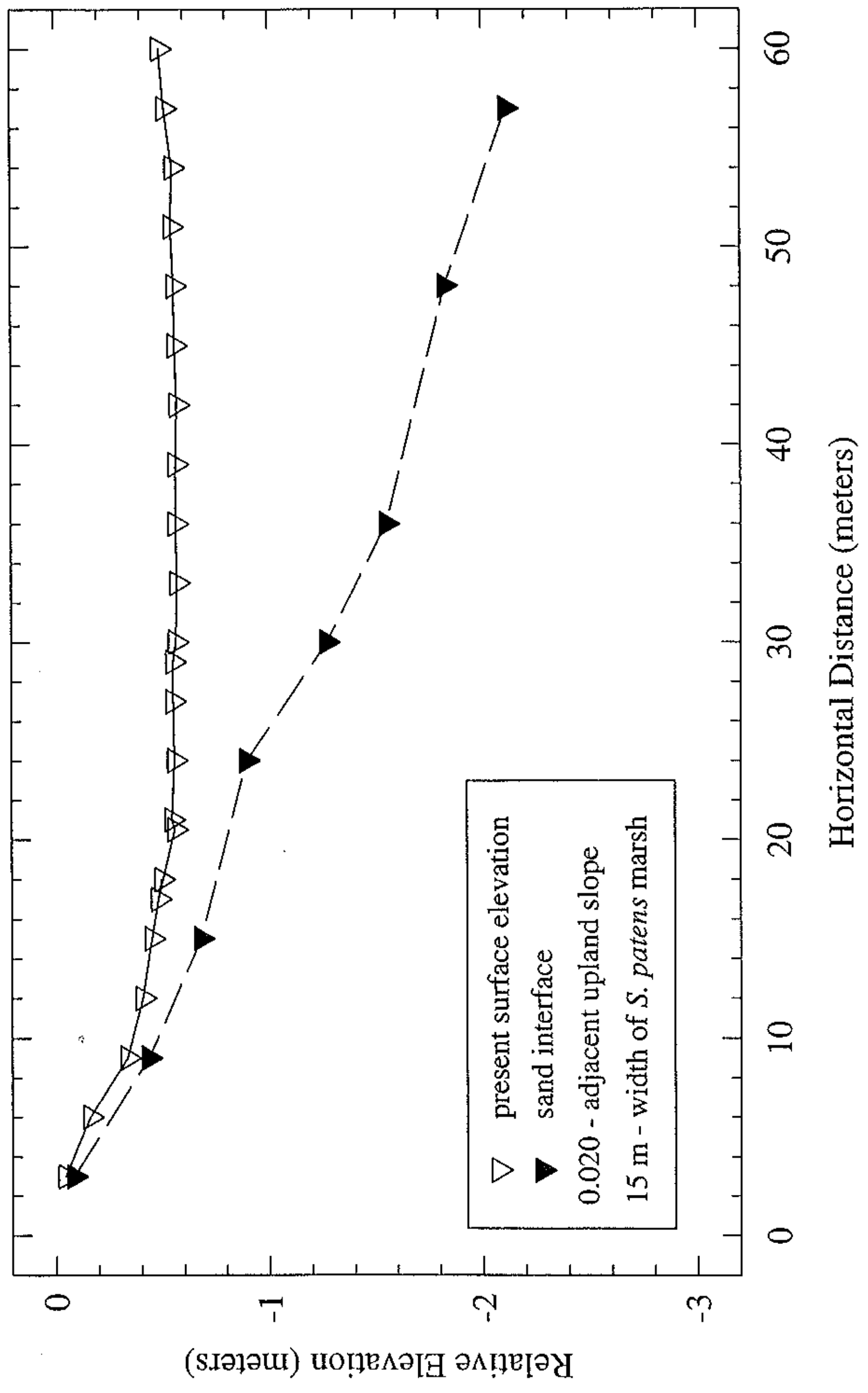


McKim - 1

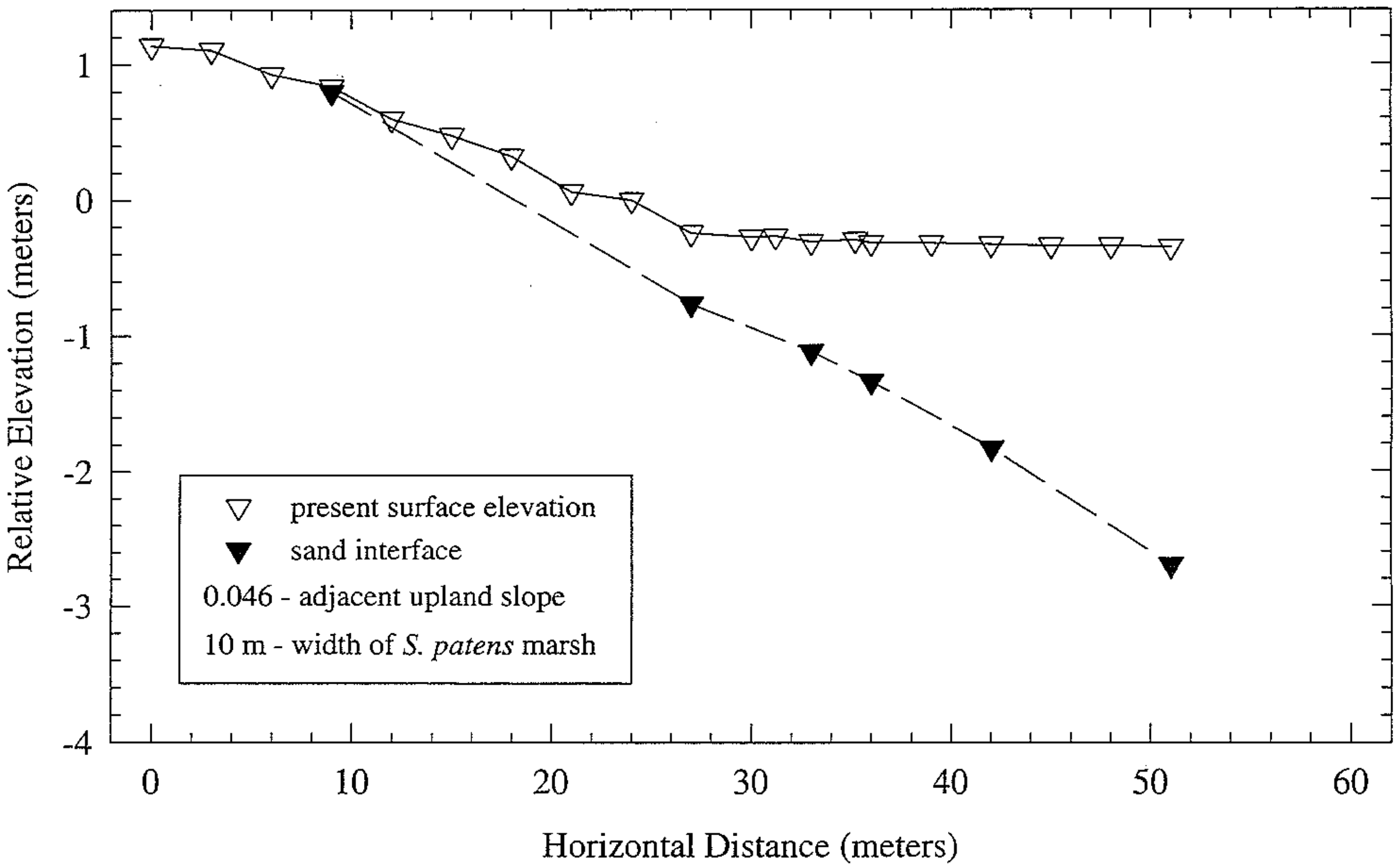

0
-1
-2
-3
Relative Elevation (meters)
0
10
20
30
40
50
60
Horizontal Distance (meters)
▽ present surface elevation
▼ sand interface
0.010 - adjacent upland slope
36 m - width of *S. patens* marsh


McKim - 2
present surface elevation
sand interface
0.024 - adjacent upland slope
9 m - width of *S. patens* marsh
Relative Elevation (meters)
0
-1
-2
Horizontal Distance (meters)
0
10
20
30
40
50
60

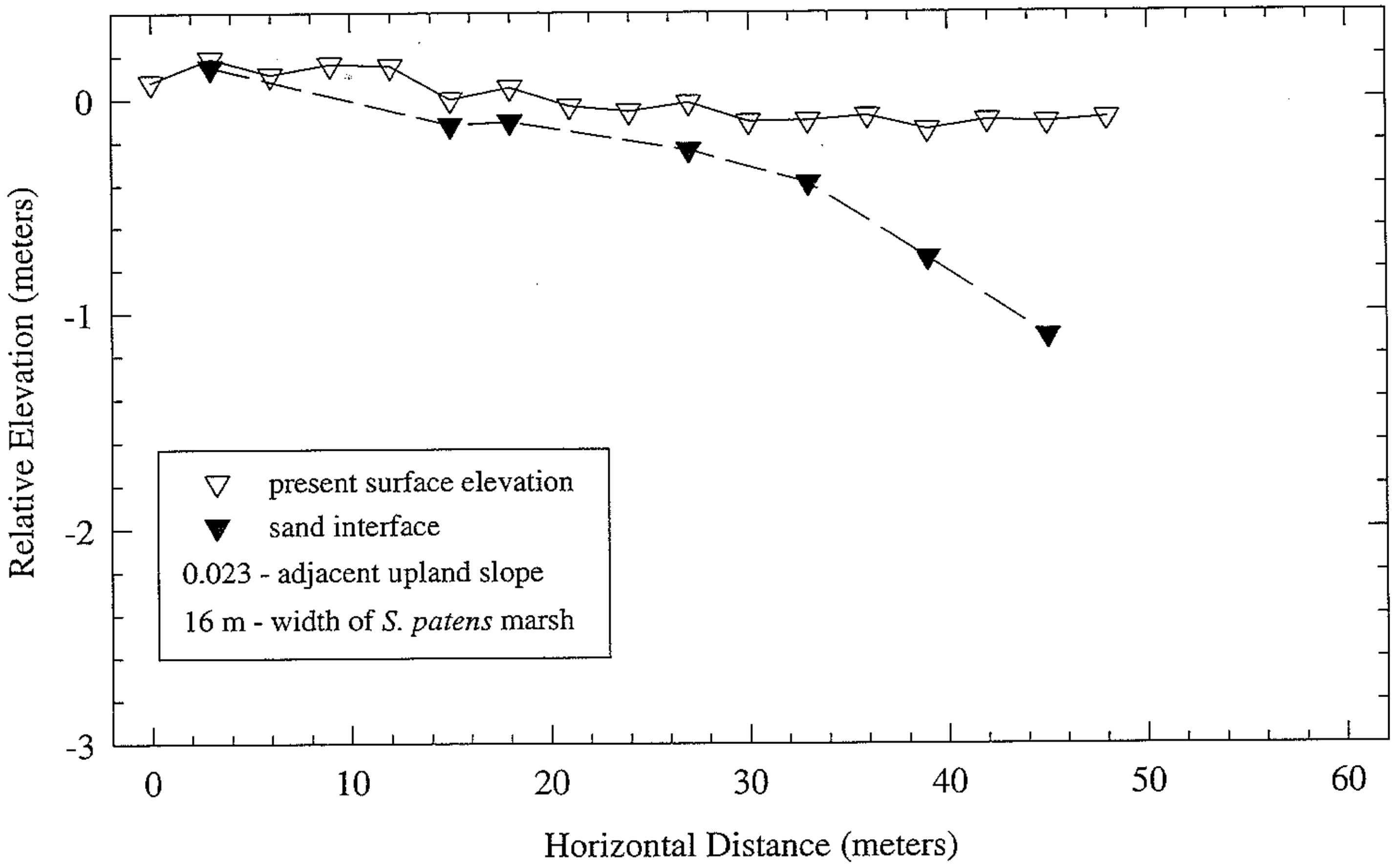
Mills - 1


present surface elevation
sand interface
0.020 - adjacent upland slope
15 m - width of *S. patens* marsh
Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60

Muddy Neck - 1

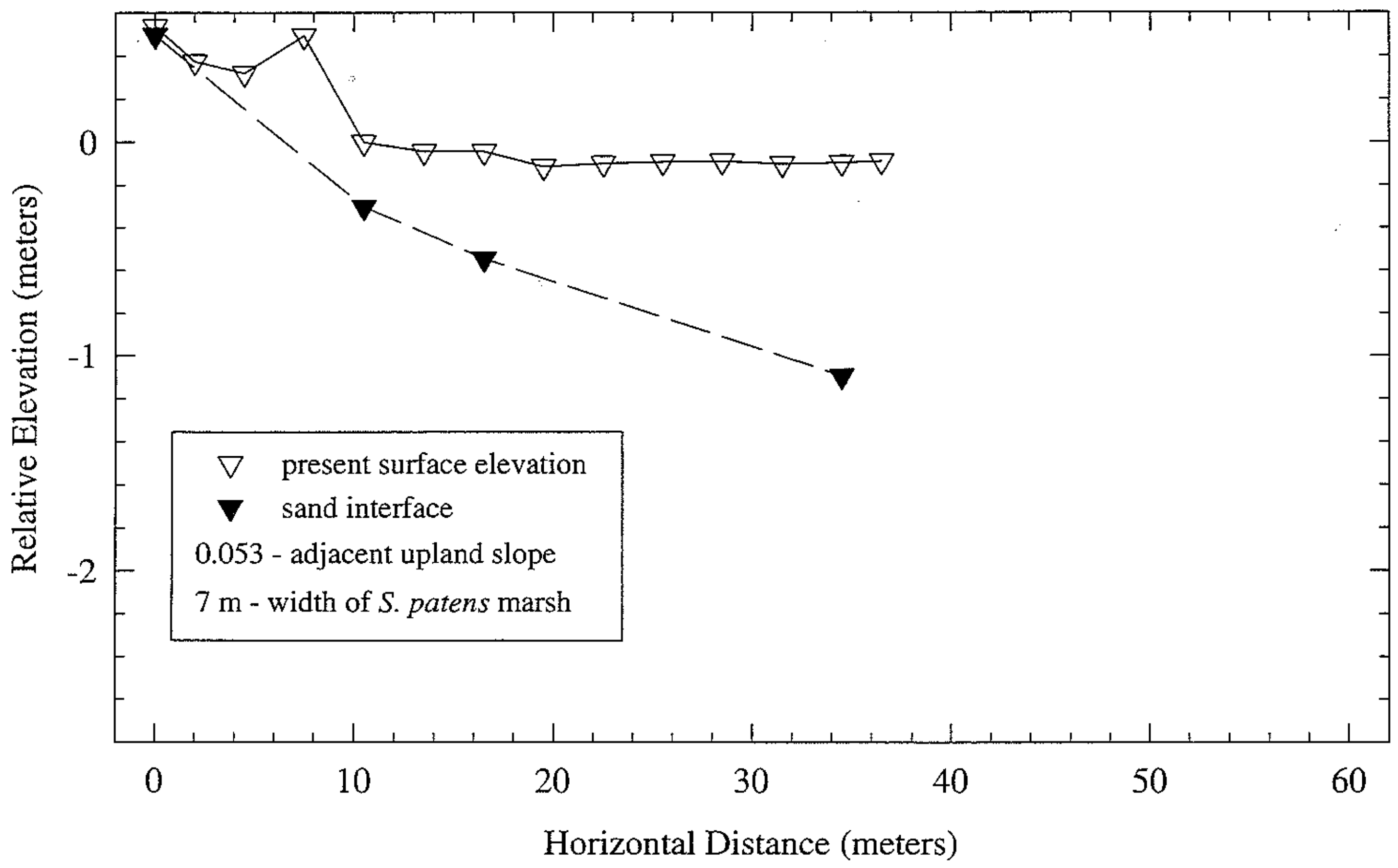

present surface elevation
sand interface
0.046 - adjacent upland slope
10 m - width of *S. patens* marsh
Relative Elevation (meters)
1
0
-1
-2
-3
-4
0
10
20
30
40
50
60
Horizontal Distance (meters)

Muddy Neck - 2



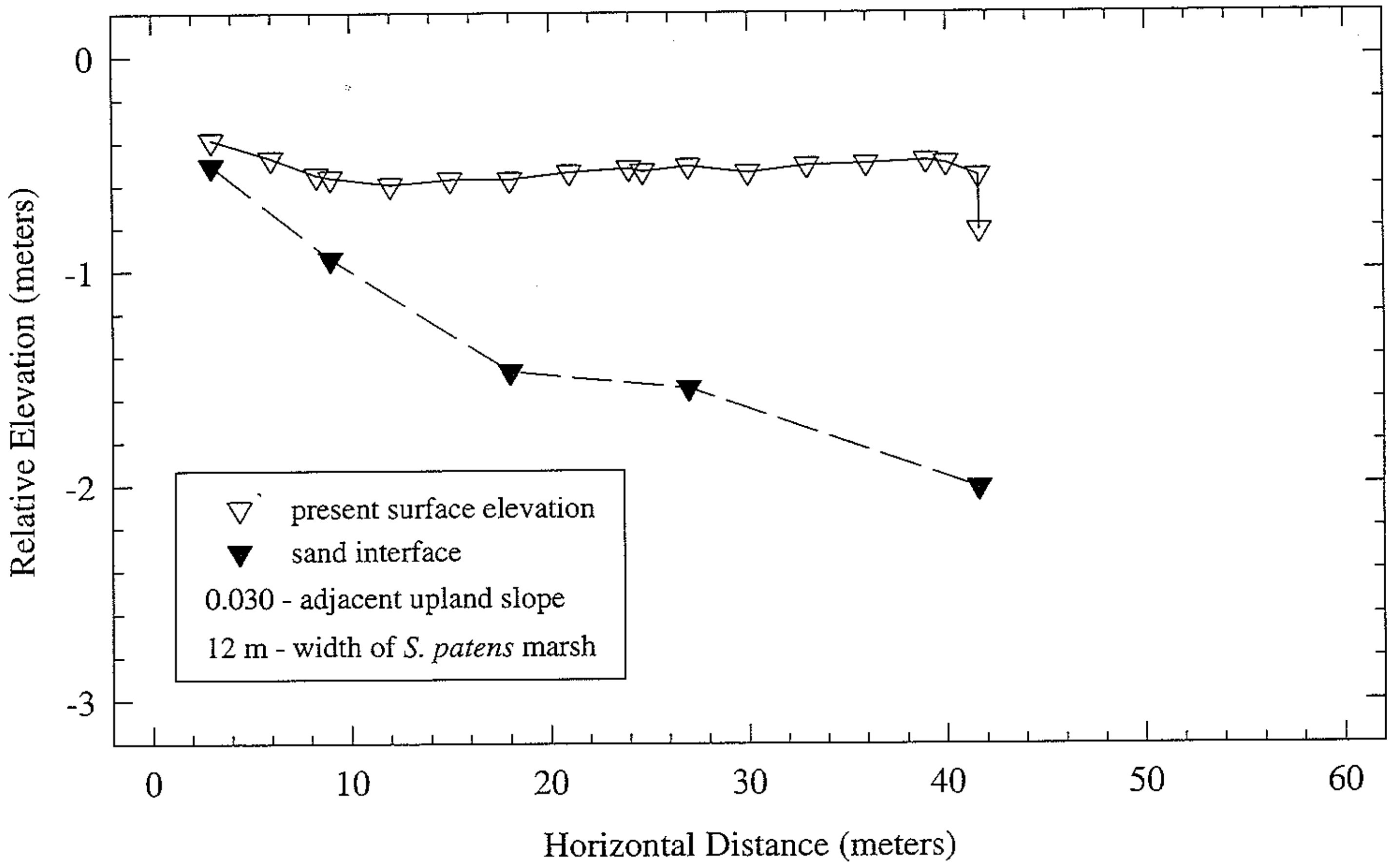
0
-1
-2
-3
Relative Elevation (meters)
0
10
20
30
40
50
60
Horizontal Distance (meters)
▽ present surface elevation
▼ sand interface
0.023 - adjacent upland slope
16 m - width of *S. patens* marsh

Muddy Neck - 3

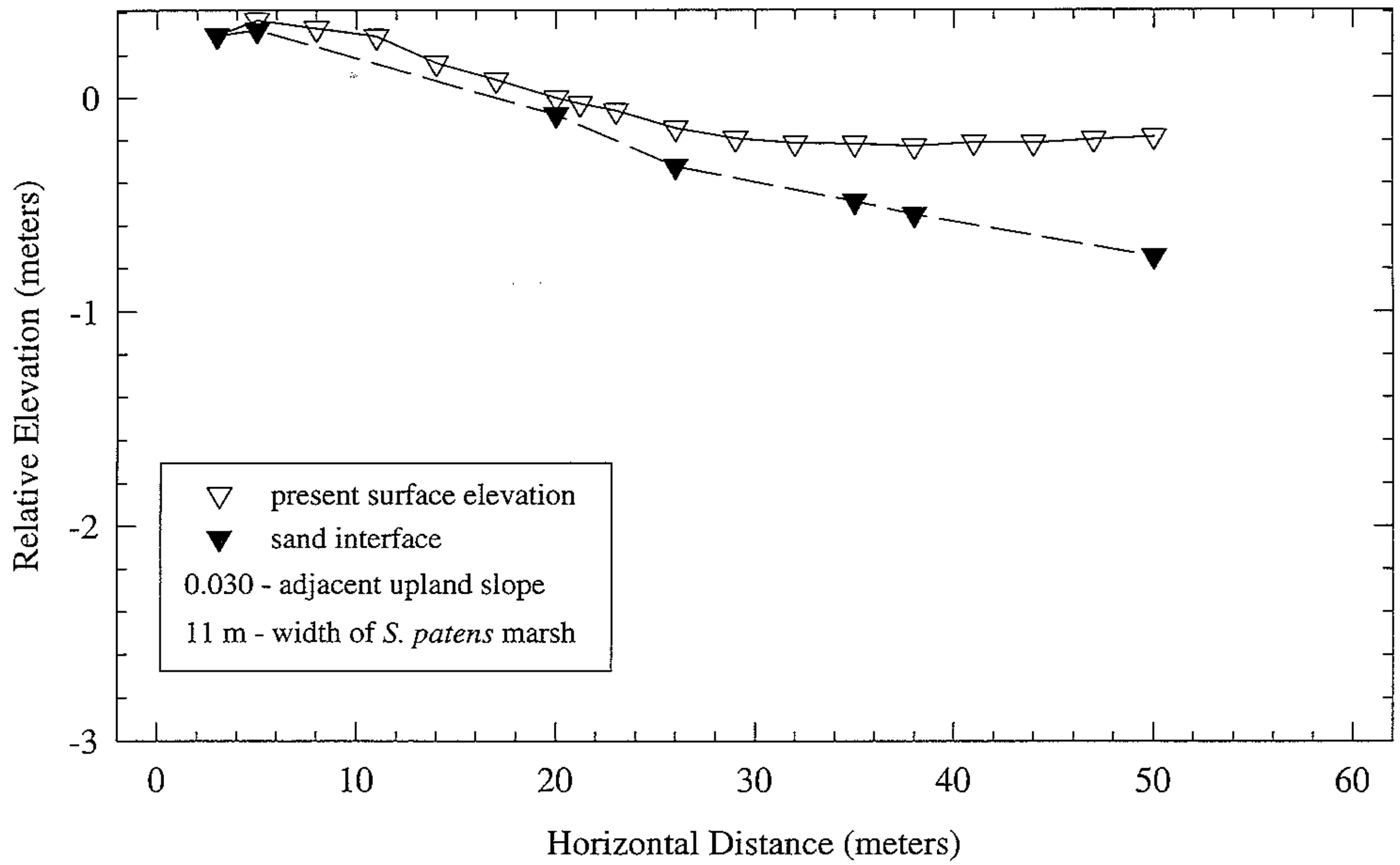


0
-1
-2
Relative Elevation (meters)
0
10
20
30
40
50
60
Horizontal Distance (meters)
▽ present surface elevation
▼ sand interface
0.053 - adjacent upland slope
7 m - width of *S. patens* marsh

Mulberry Landing

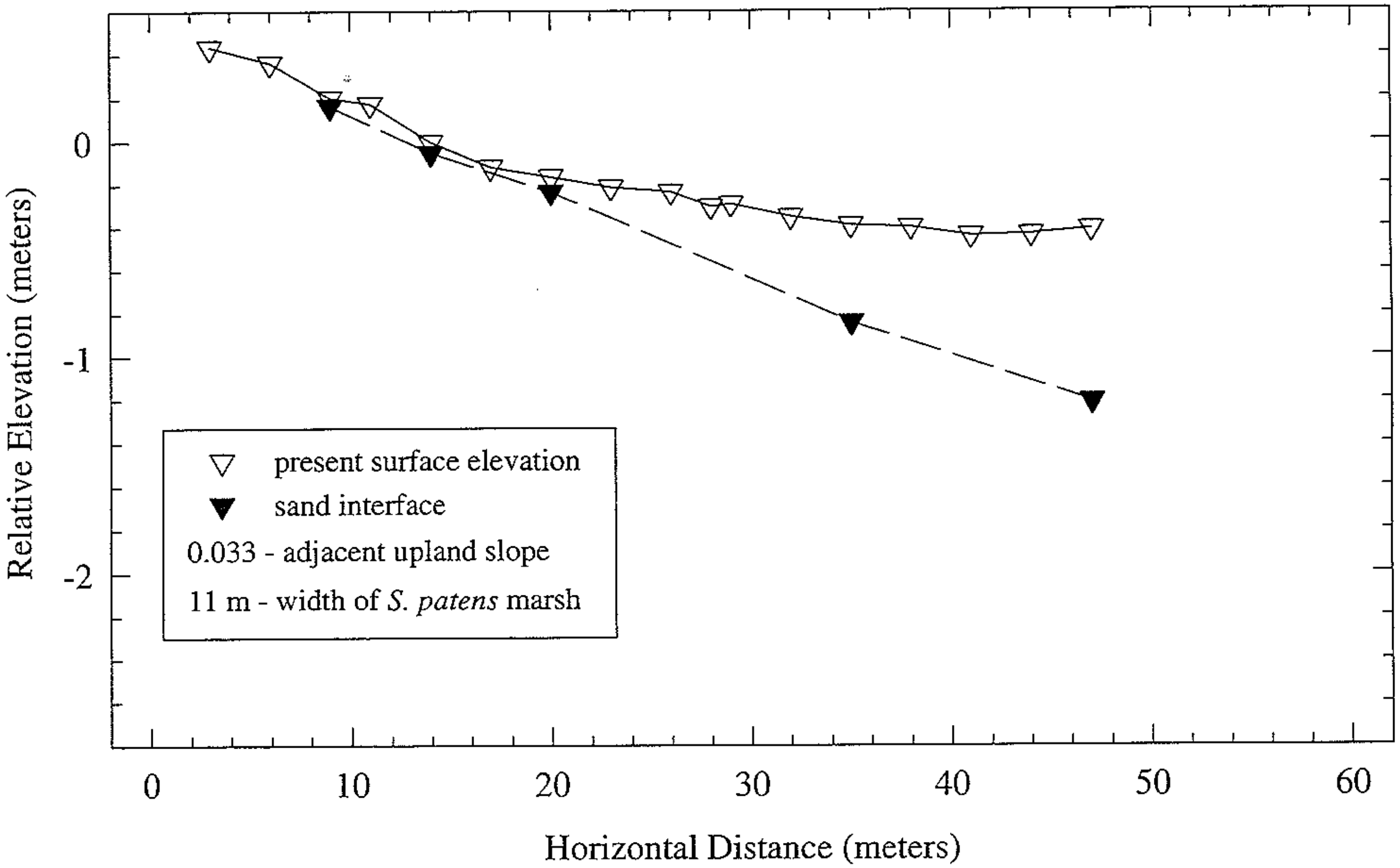


Murray - 1



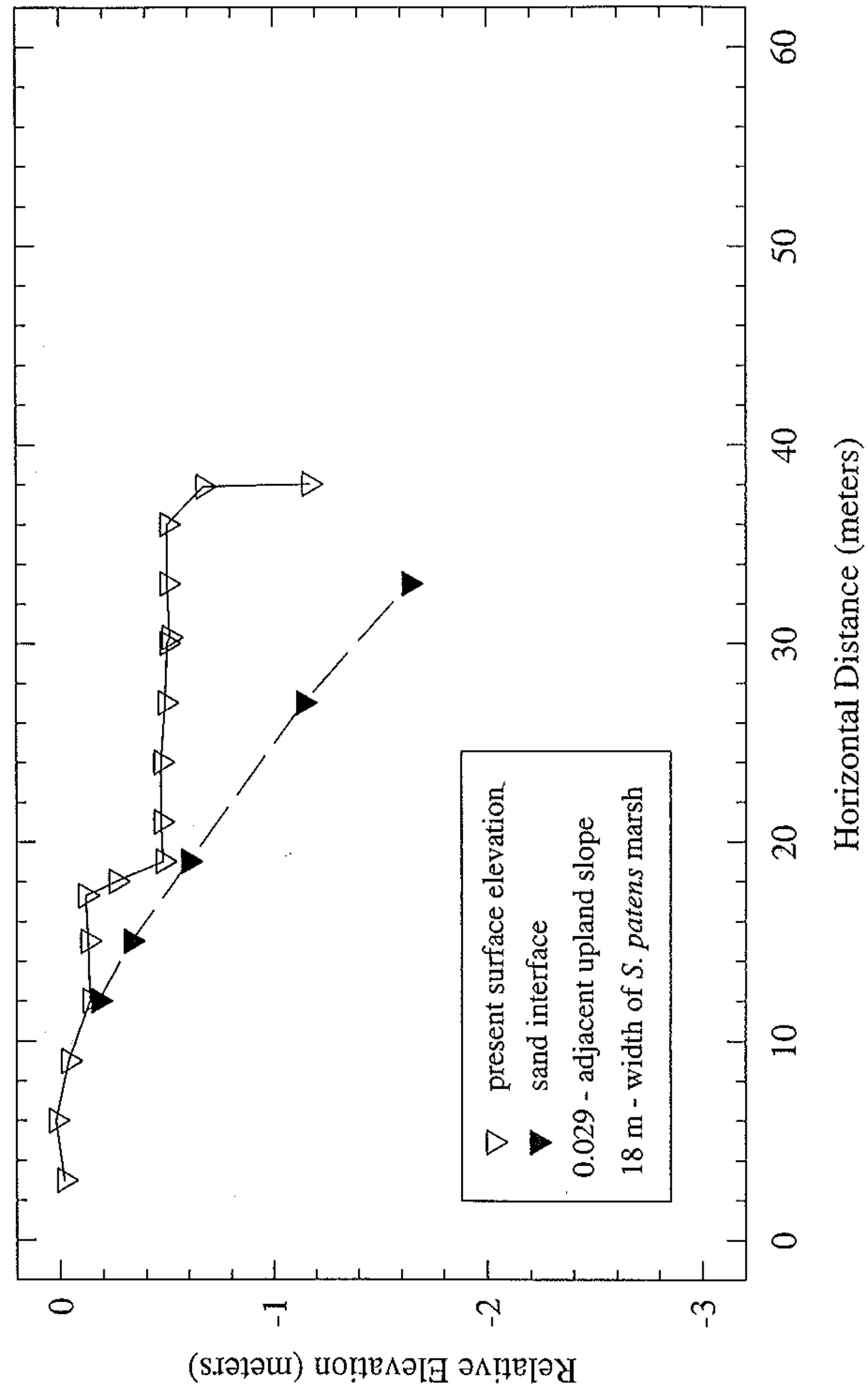
Relative Elevation (meters)
0
-1
-2
-3
0
10
20
30
40
50
60
Horizontal Distance (meters)
▽ present surface elevation
▼ sand interface
0.030 - adjacent upland slope
11 m - width of *S. patens* marsh

Murray - 2

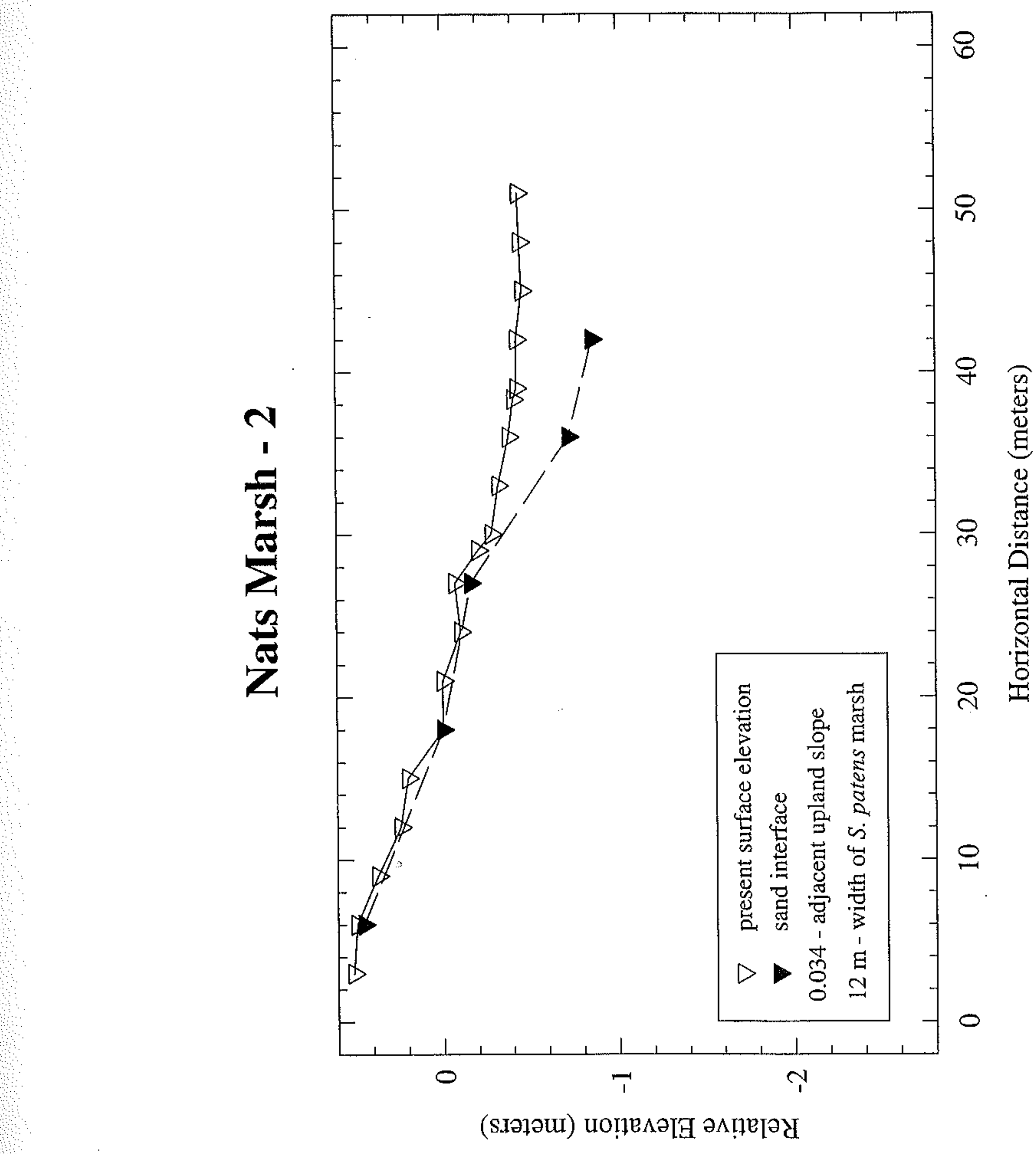


▽ present surface elevation
▼ sand interface
0.033 - adjacent upland slope
11 m - width of *S. patens* marsh
Relative Elevation (meters)
0
-1
-2
Horizontal Distance (meters)
0
10
20
30
40
50
60

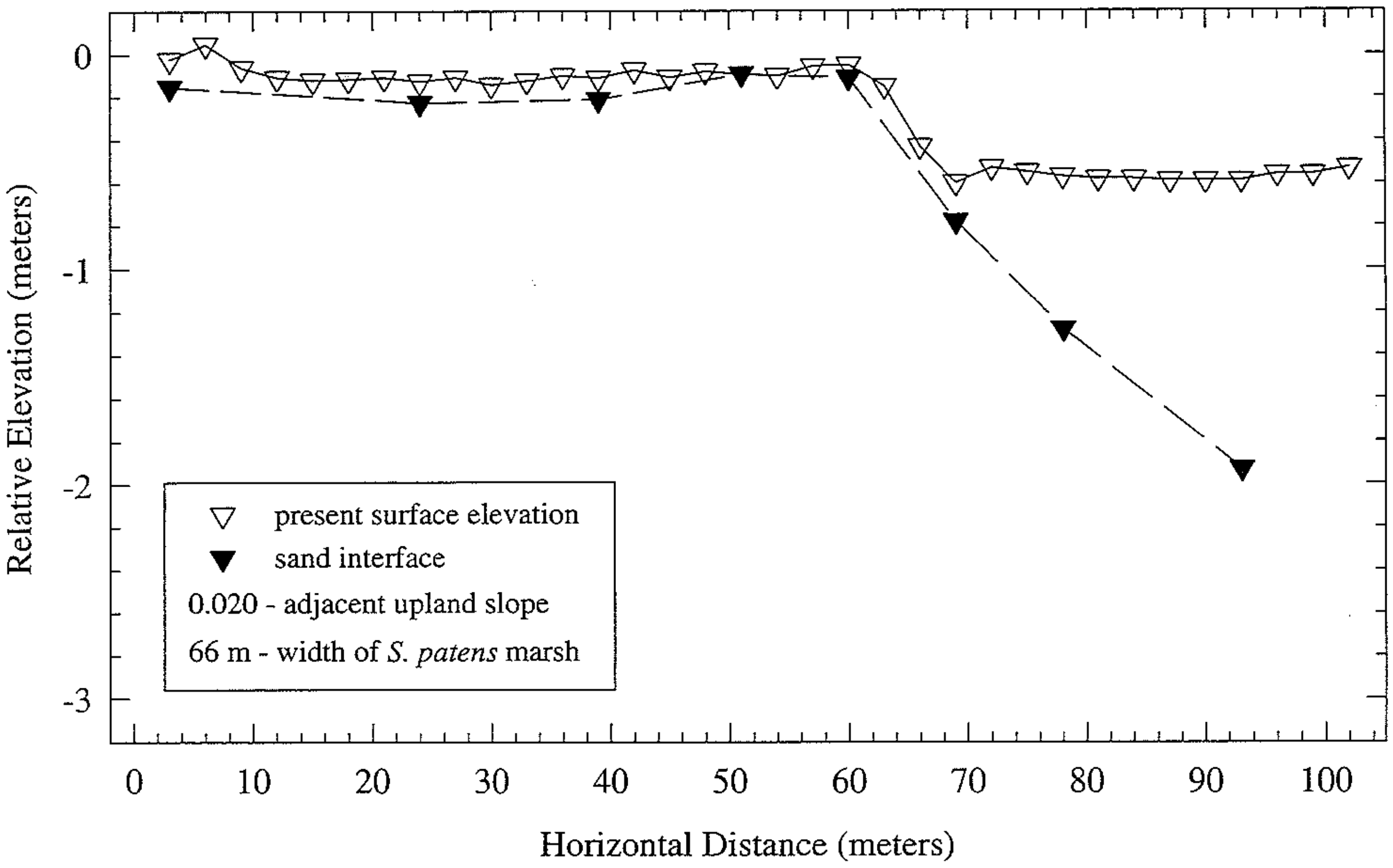
Nats Marsh - 1



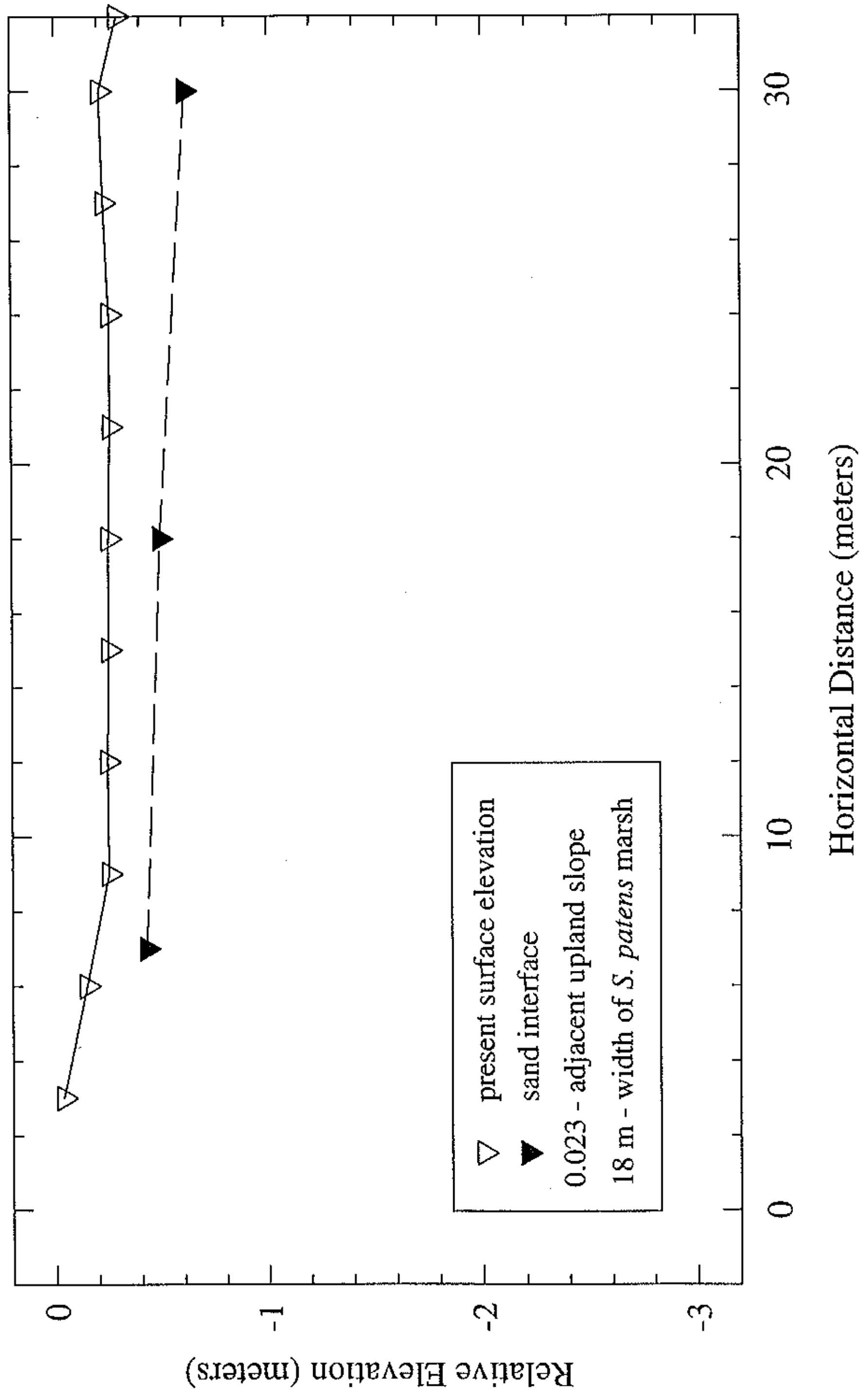
▽ present surface elevation
▼ sand interface
0.029 - adjacent upland slope
18 m - width of *S. patens* marsh
Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60


Nats Marsh - 2
▽ present surface elevation
▼ sand interface
0.034 - adjacent upland slope
12 m - width of *S. patens* marsh
Relative Elevation (meters)
0
-1
-2
Horizontal Distance (meters)
0
10
20
30
40
50
60

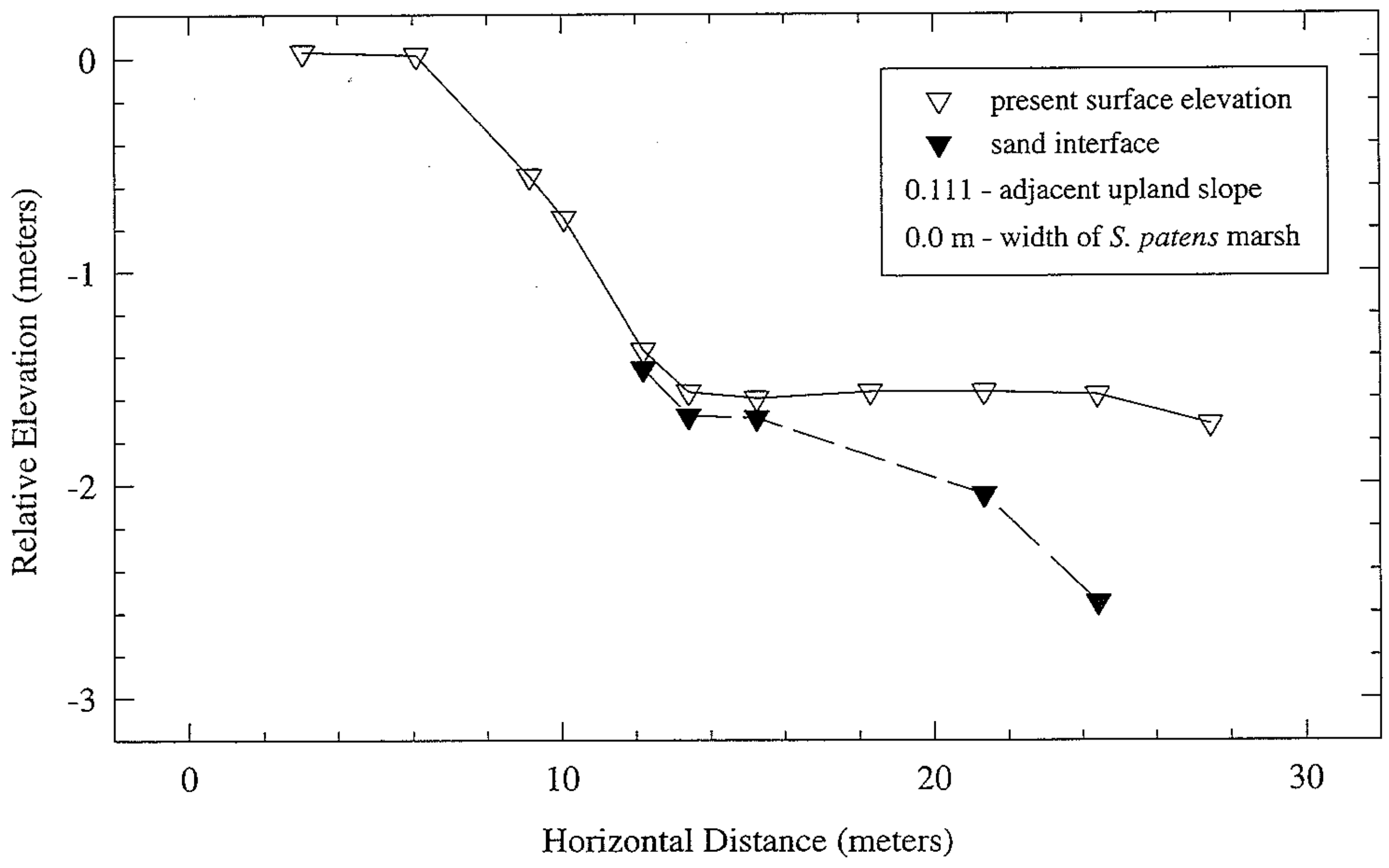
Oyster Neck Road - 1



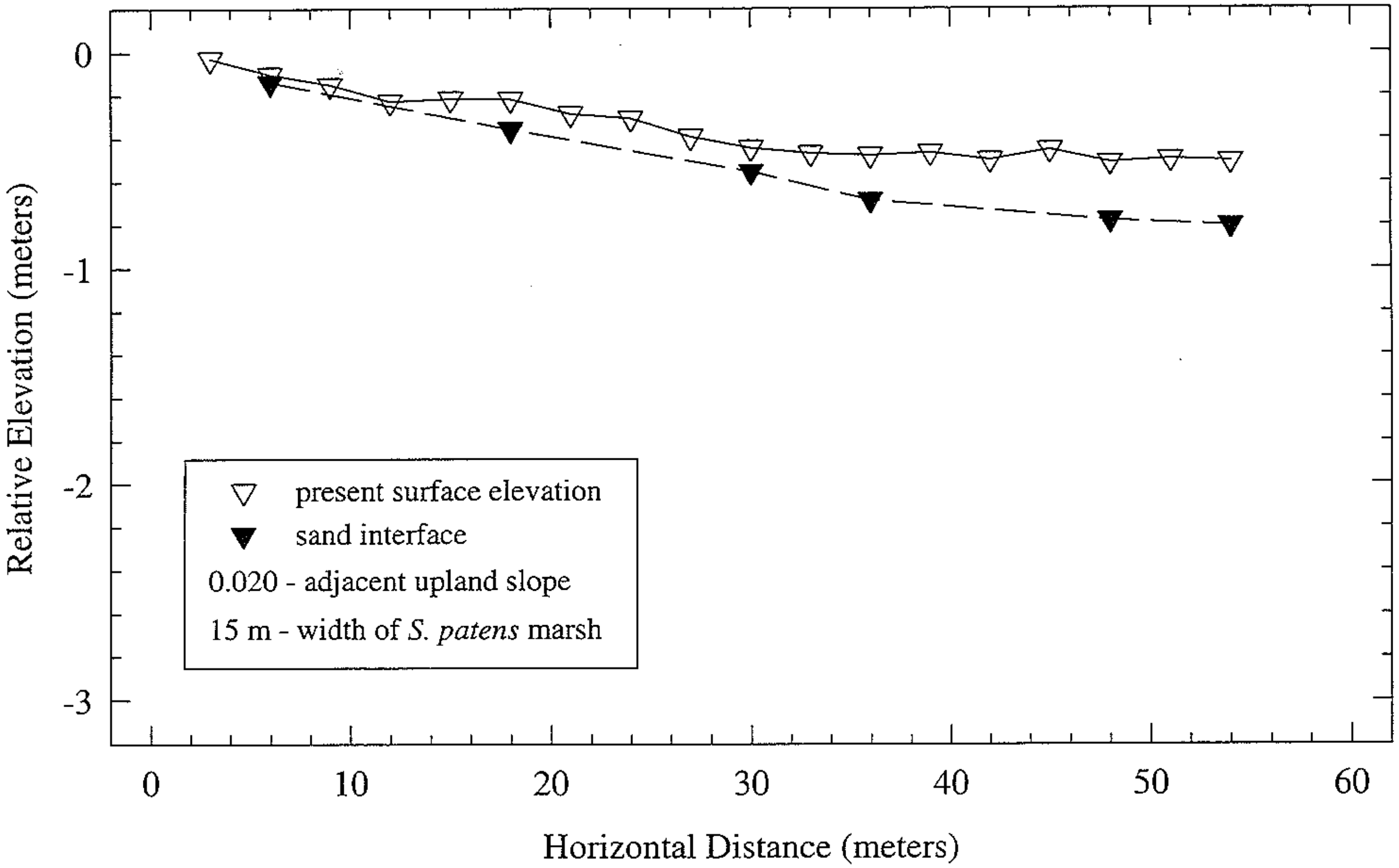
Oyster Neck Road - 2



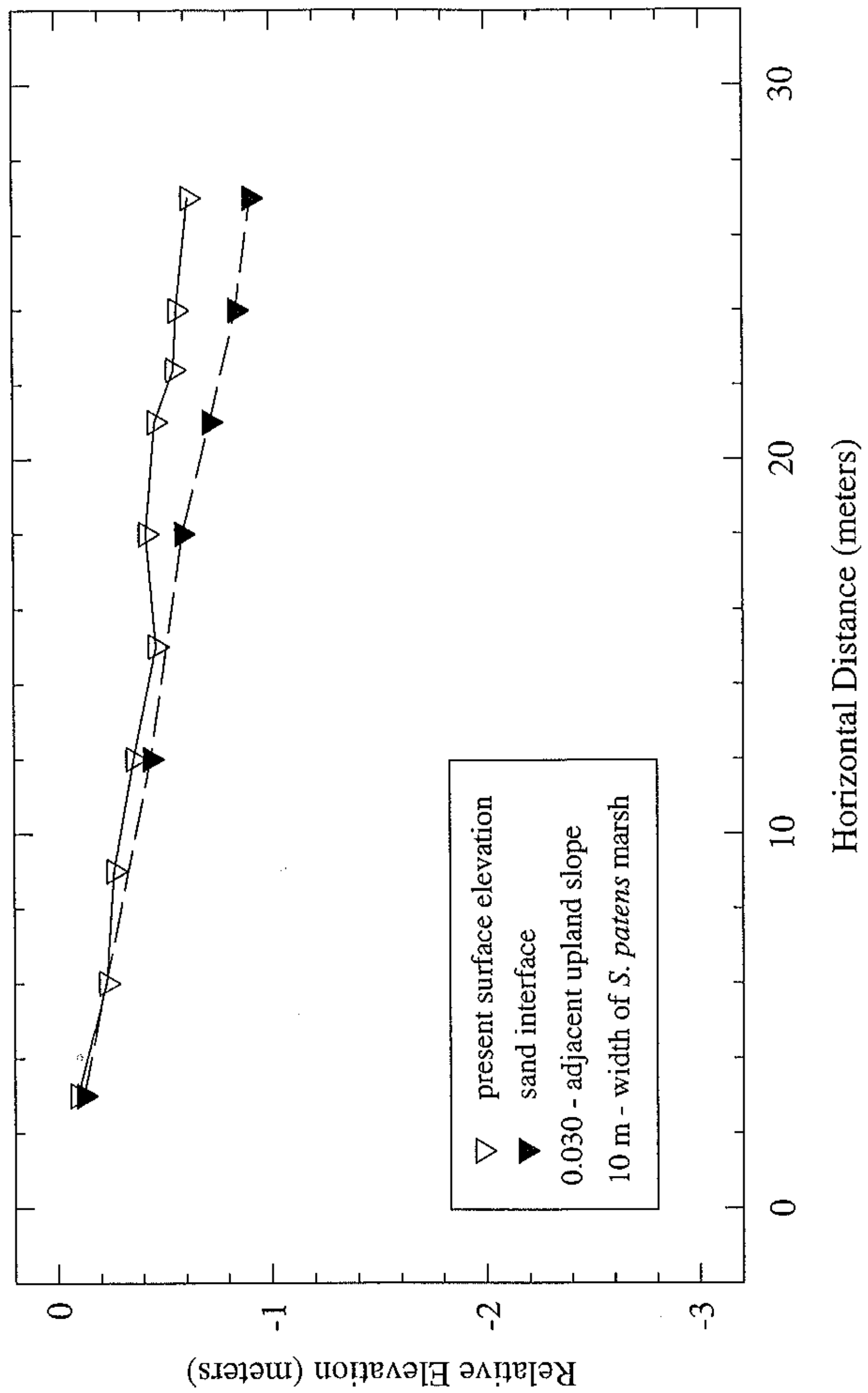
Pagan Creek Village

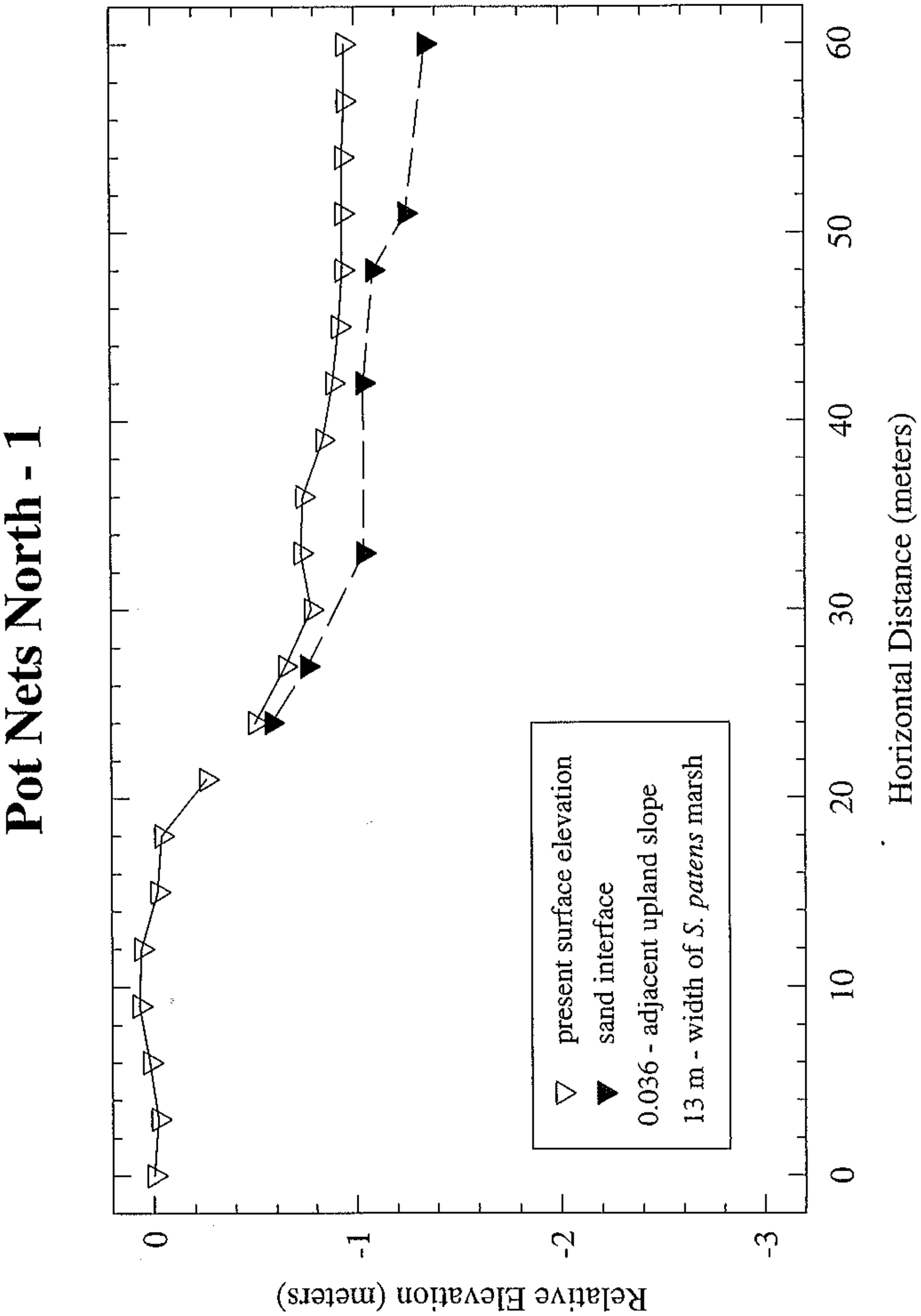


Pasture Point Cove - 1

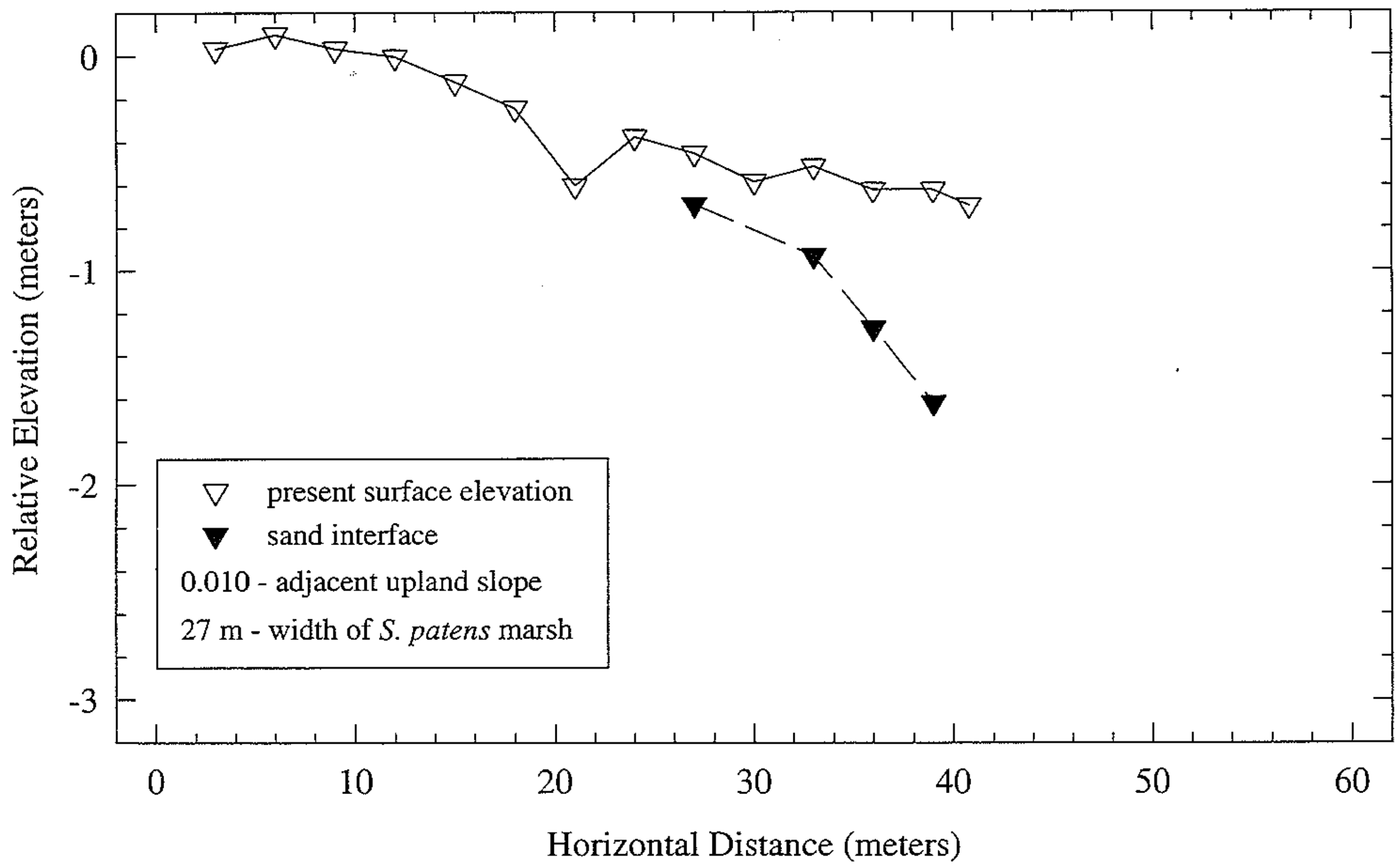


Pasture Point Cove - 2

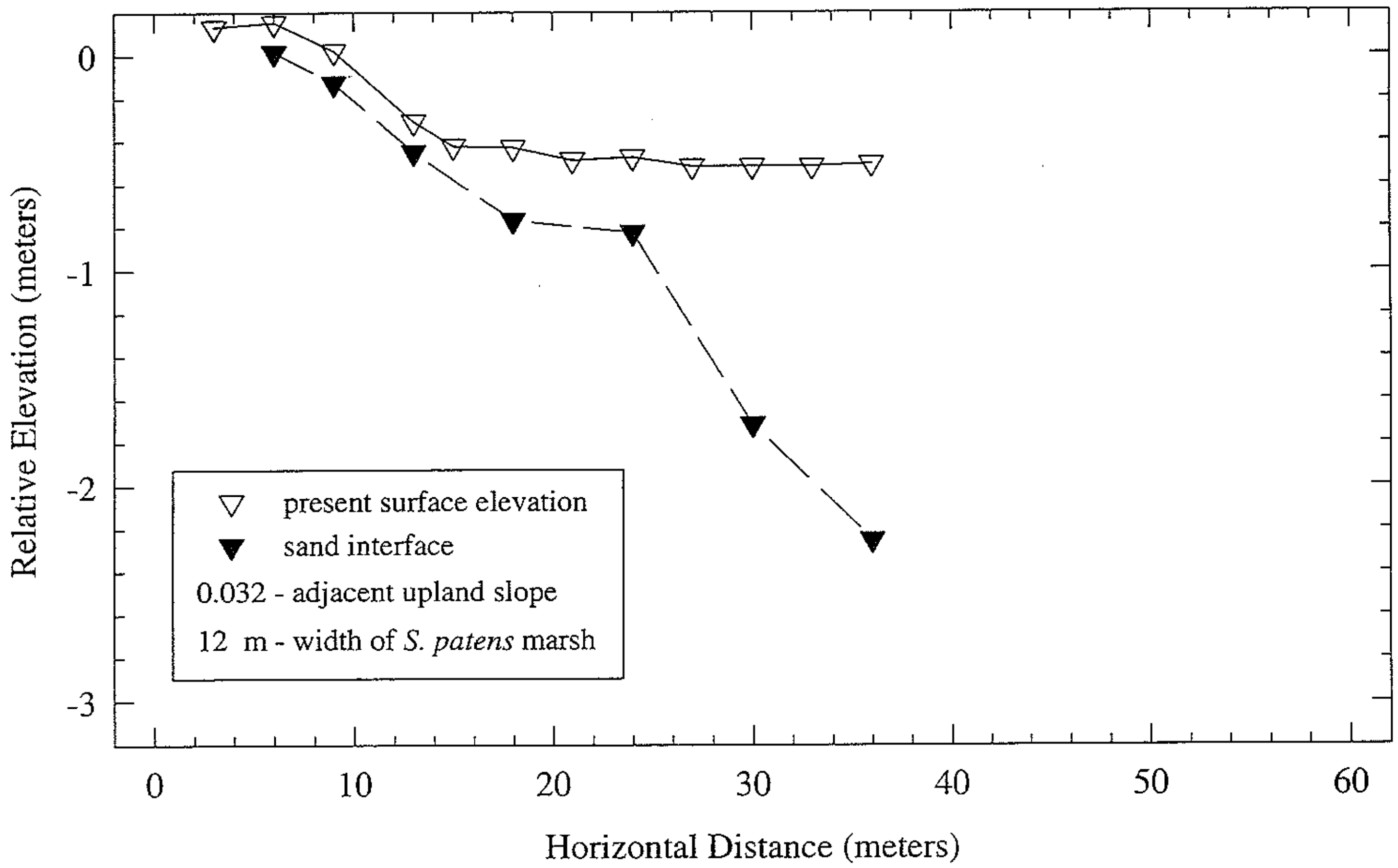



Pot Nets North - 1
Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60
▽ present surface elevation
▼ sand interface
0.036 - adjacent upland slope
13 m - width of *S. patens* marsh

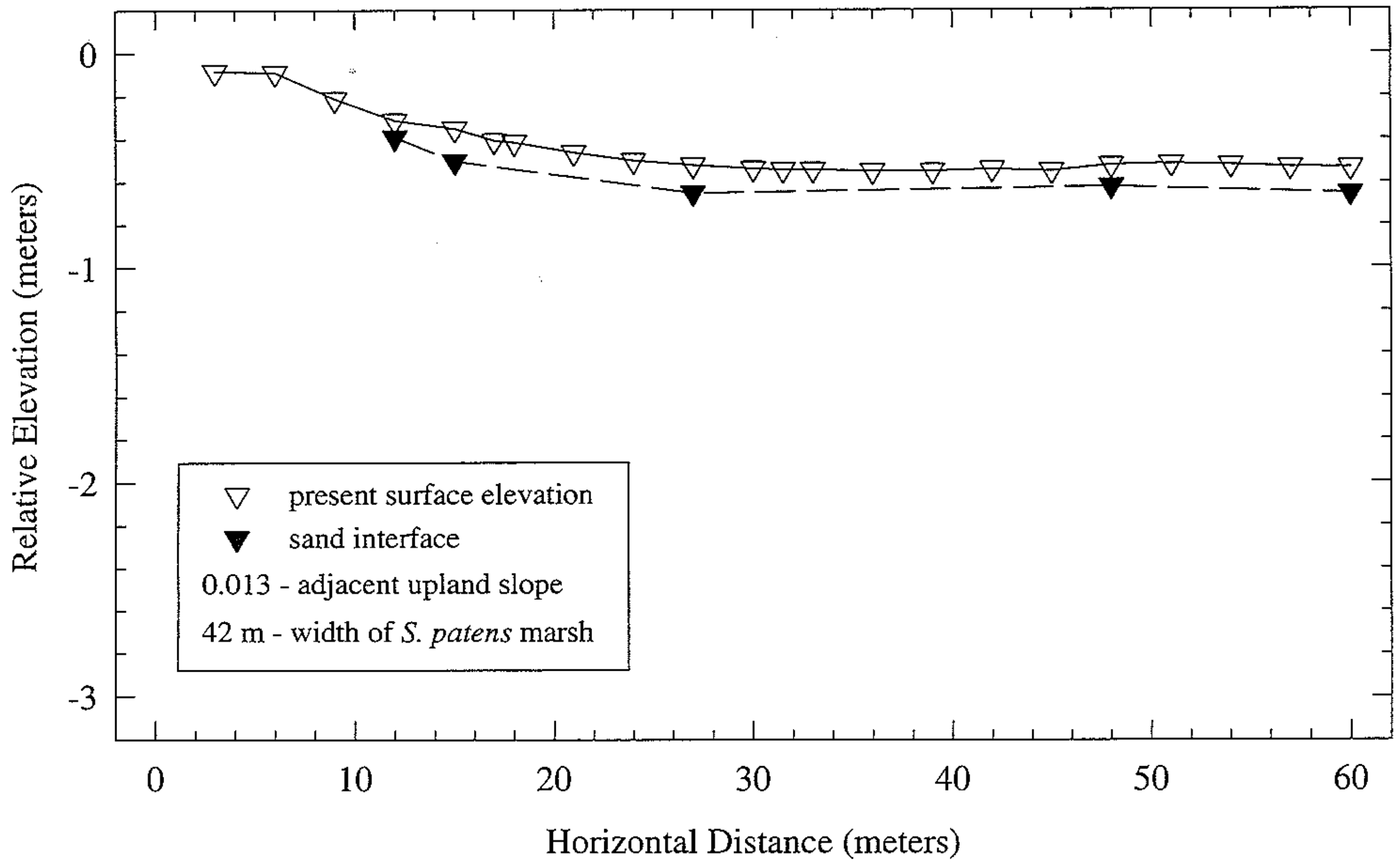
Pot Nets North - 2



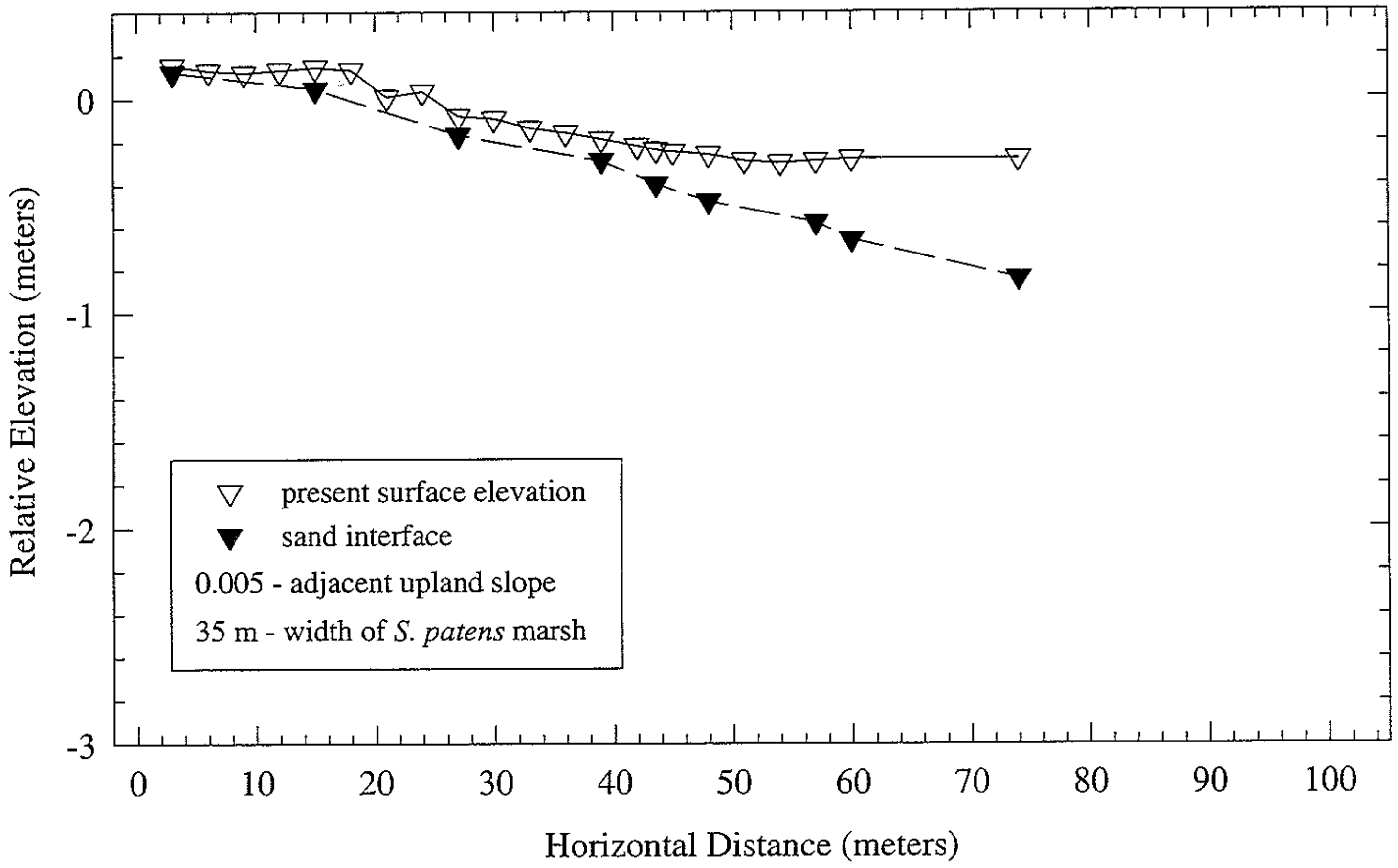
Pot Nets North - 3



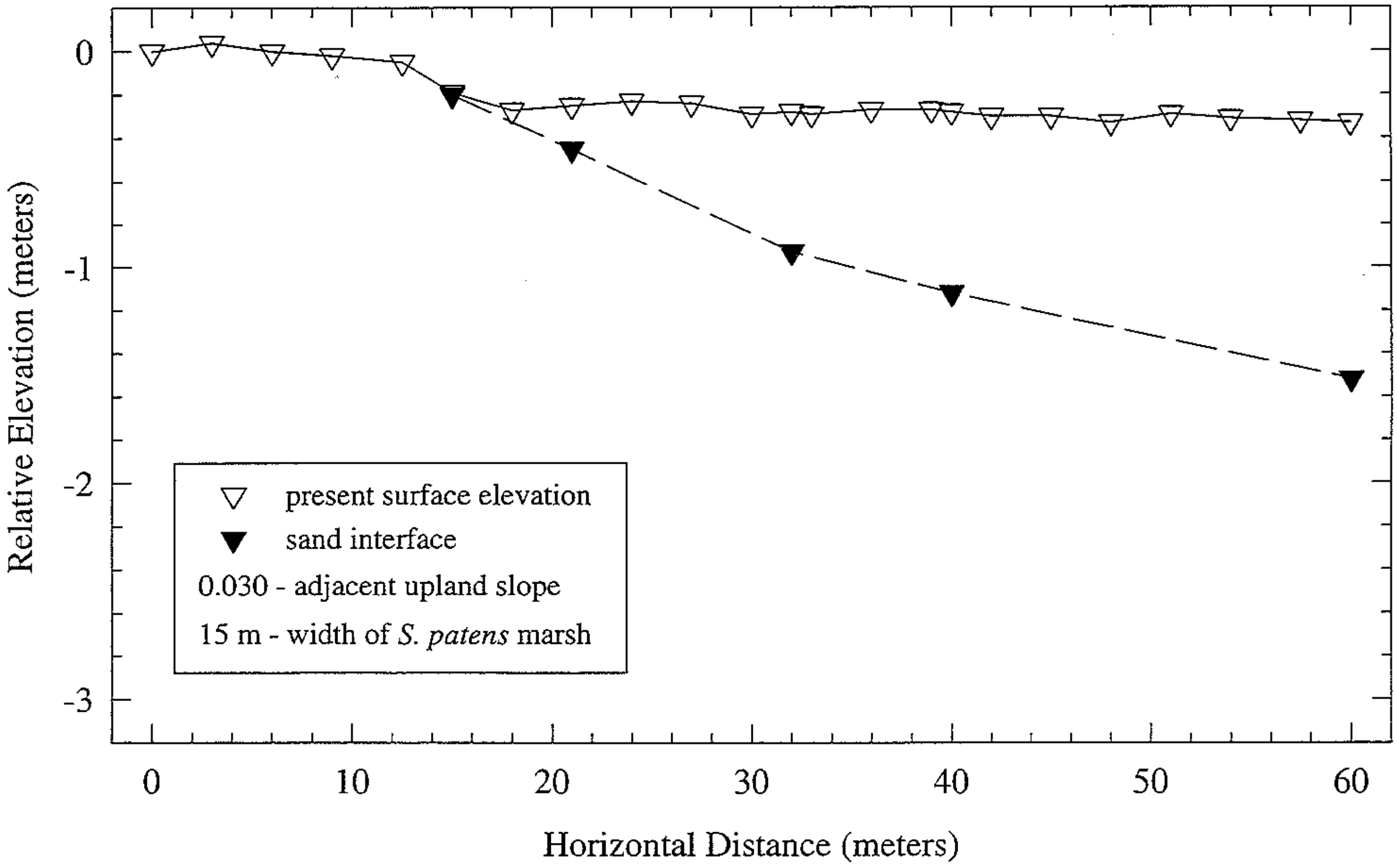
Pot Nets North - 4



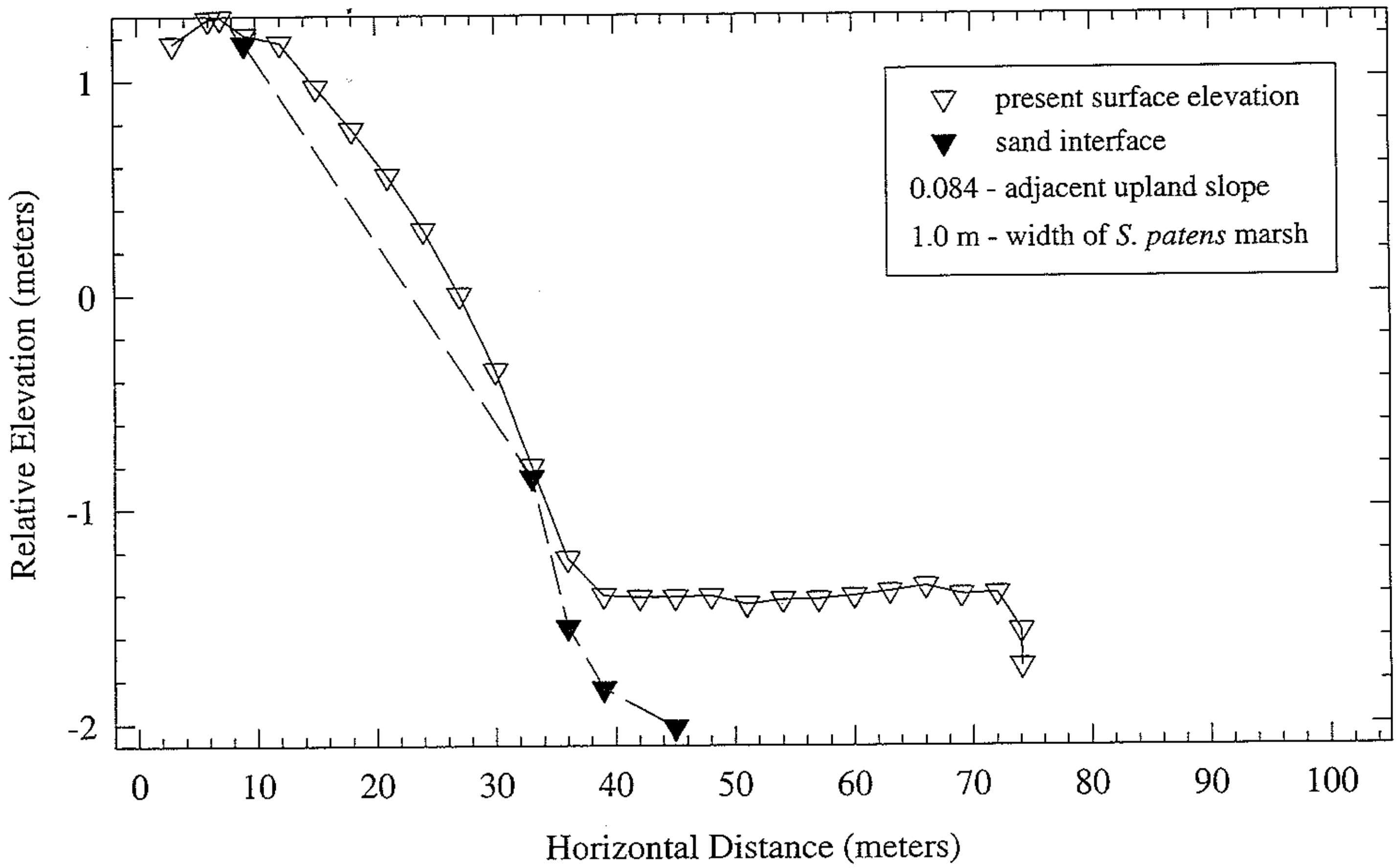
Pot Nets North - 5



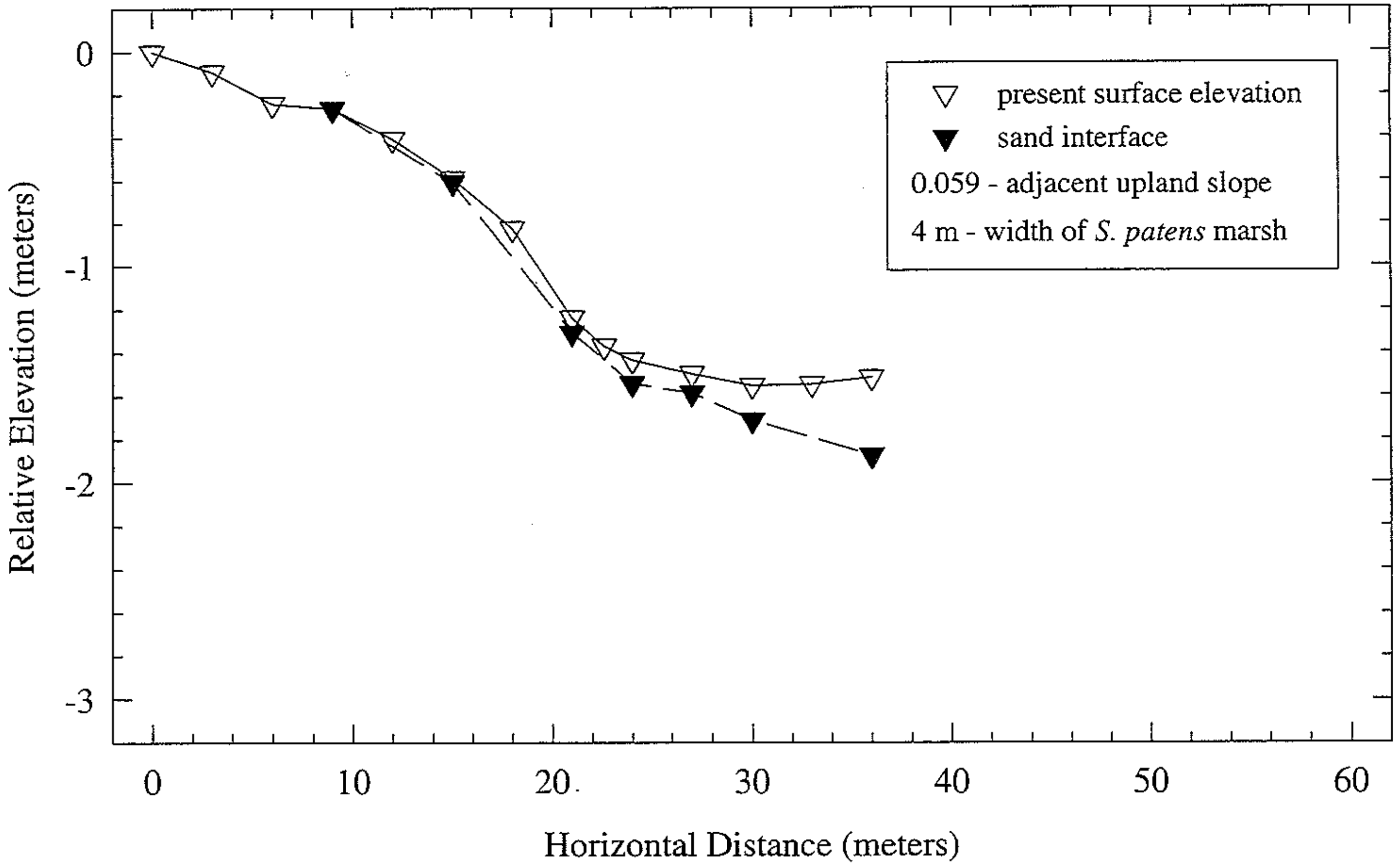
Prime Hook Radio Station



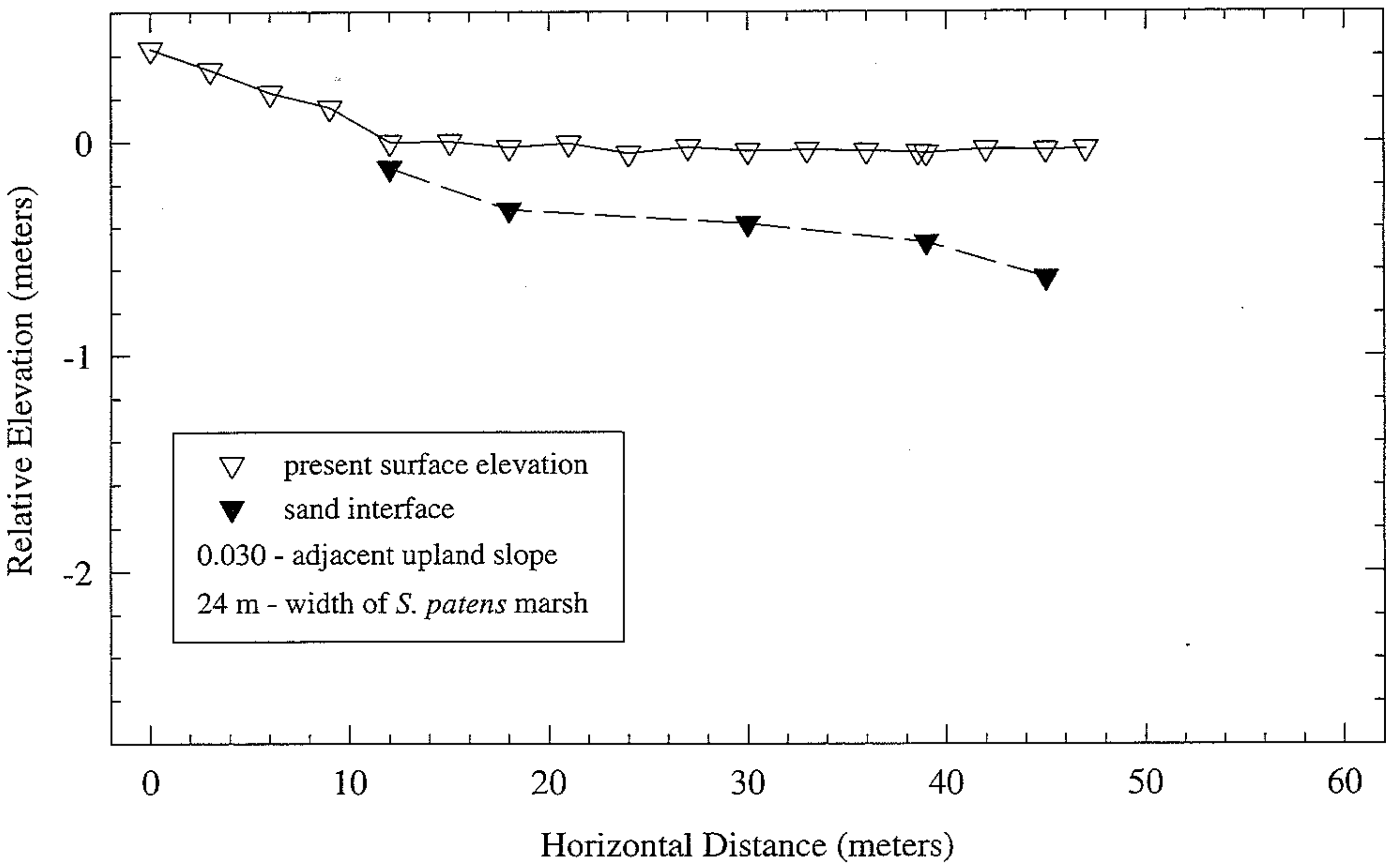
Pullover - 1


present surface elevation
sand interface
0.084 - adjacent upland slope
1.0 m - width of *S. patens* marsh
Relative Elevation (meters)
1
0
-1
-2
Horizontal Distance (meters)
0
10
20
30
40
50
60
70
80
90
100

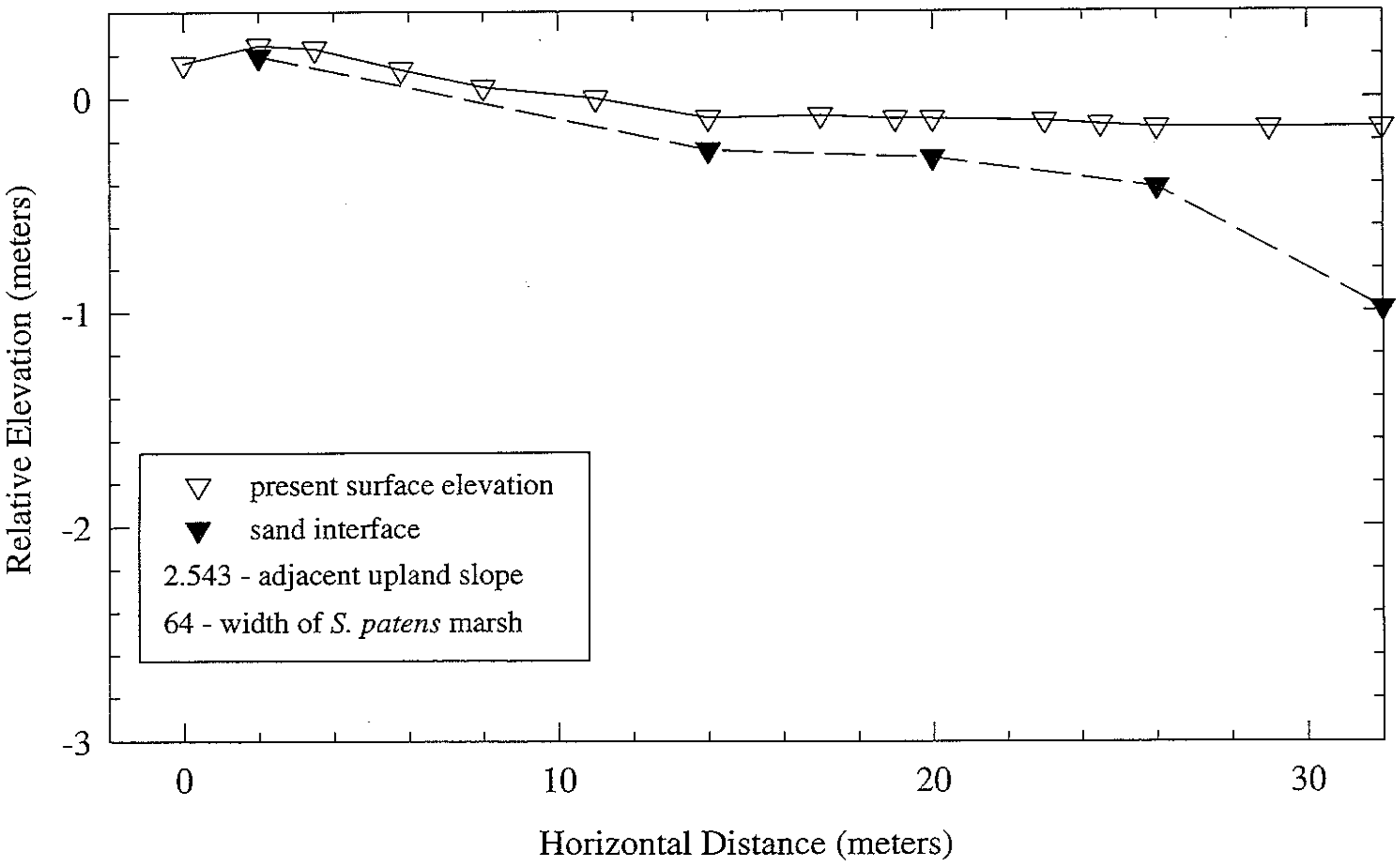
Ritter - 1


present surface elevation
sand interface
0.059 - adjacent upland slope
4 m - width of *S. patens* marsh
Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60

Ritter - 2

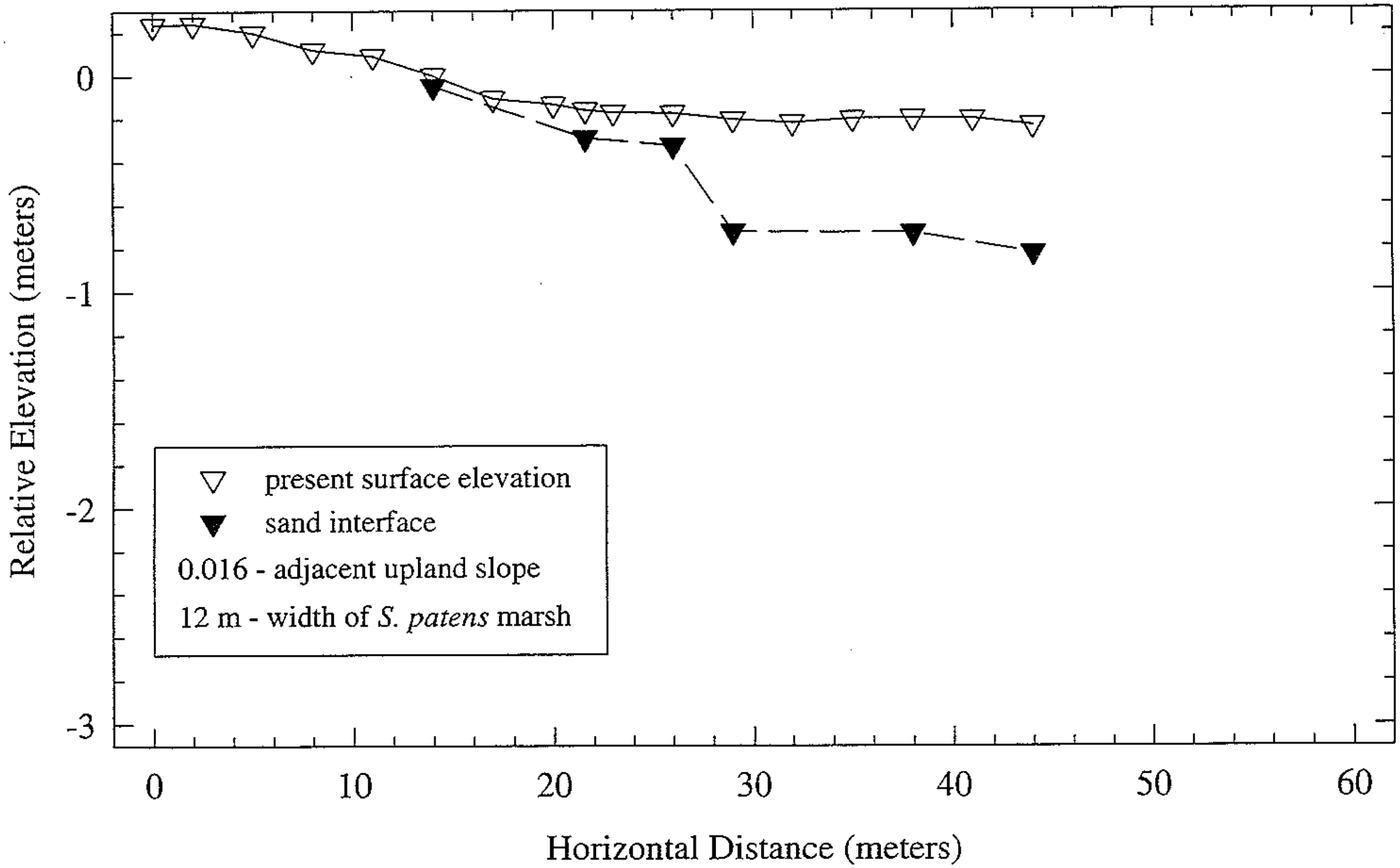

present surface elevation
sand interface
0.030 - adjacent upland slope
24 m - width of *S. patens* marsh
Relative Elevation (meters)
0
-1
-2
Horizontal Distance (meters)
0
10
20
30
40
50
60

Ritter - 3



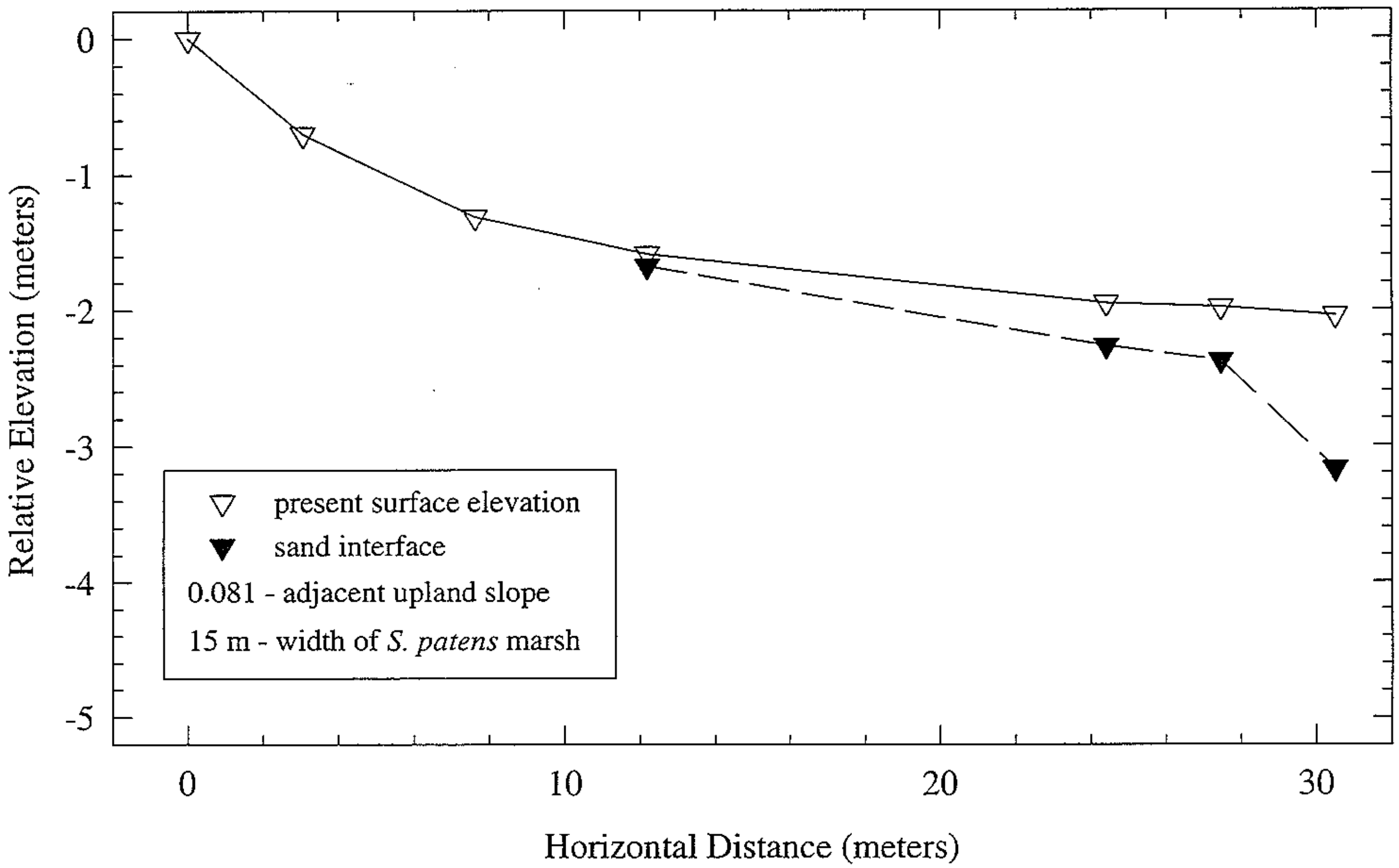
Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
▽ present surface elevation
▼ sand interface
2.543 - adjacent upland slope
64 - width of *S. patens* marsh

Ritter - 4



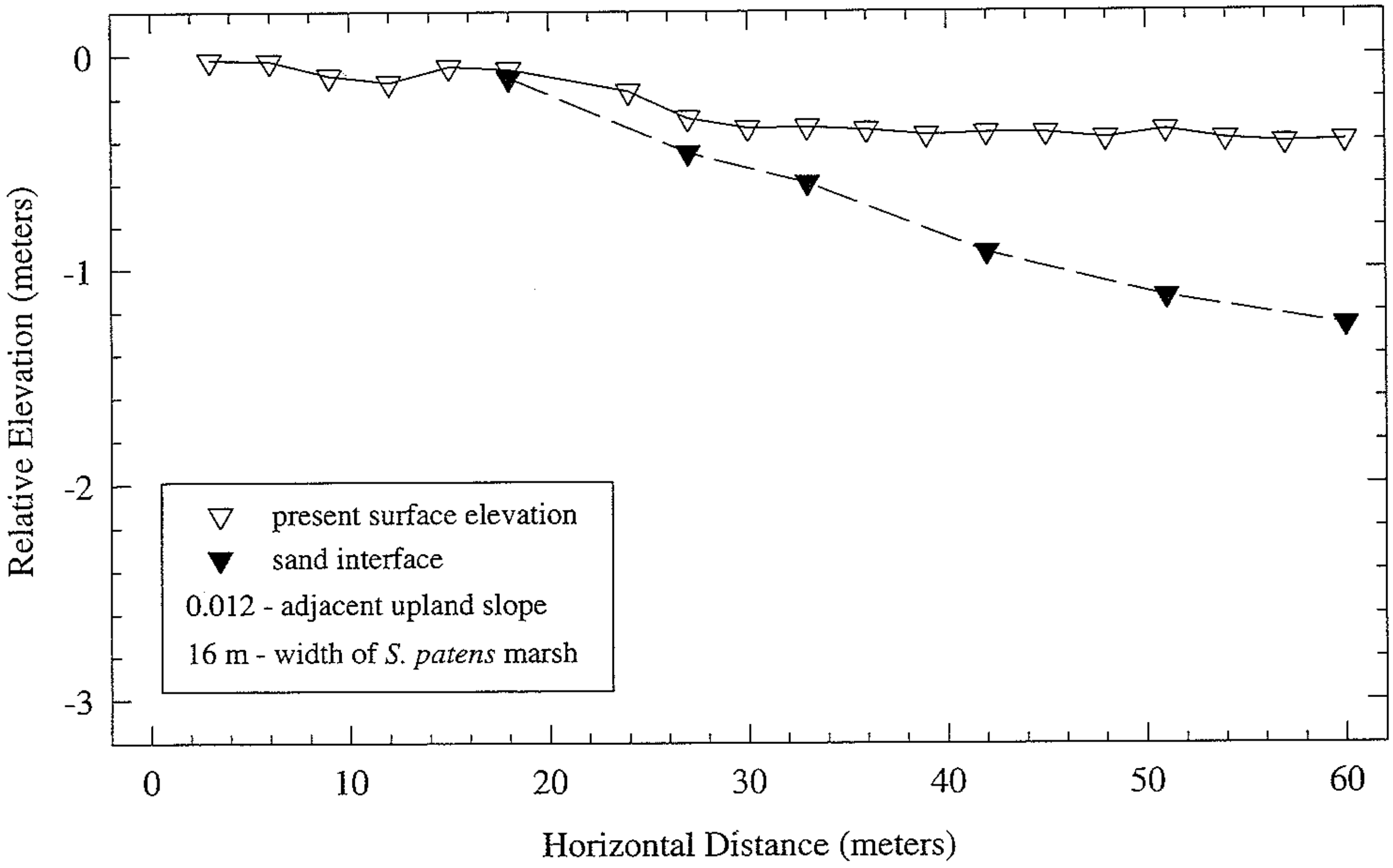
Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60
▽ present surface elevation
▼ sand interface
0.016 - adjacent upland slope
12 m - width of *S. patens* marsh

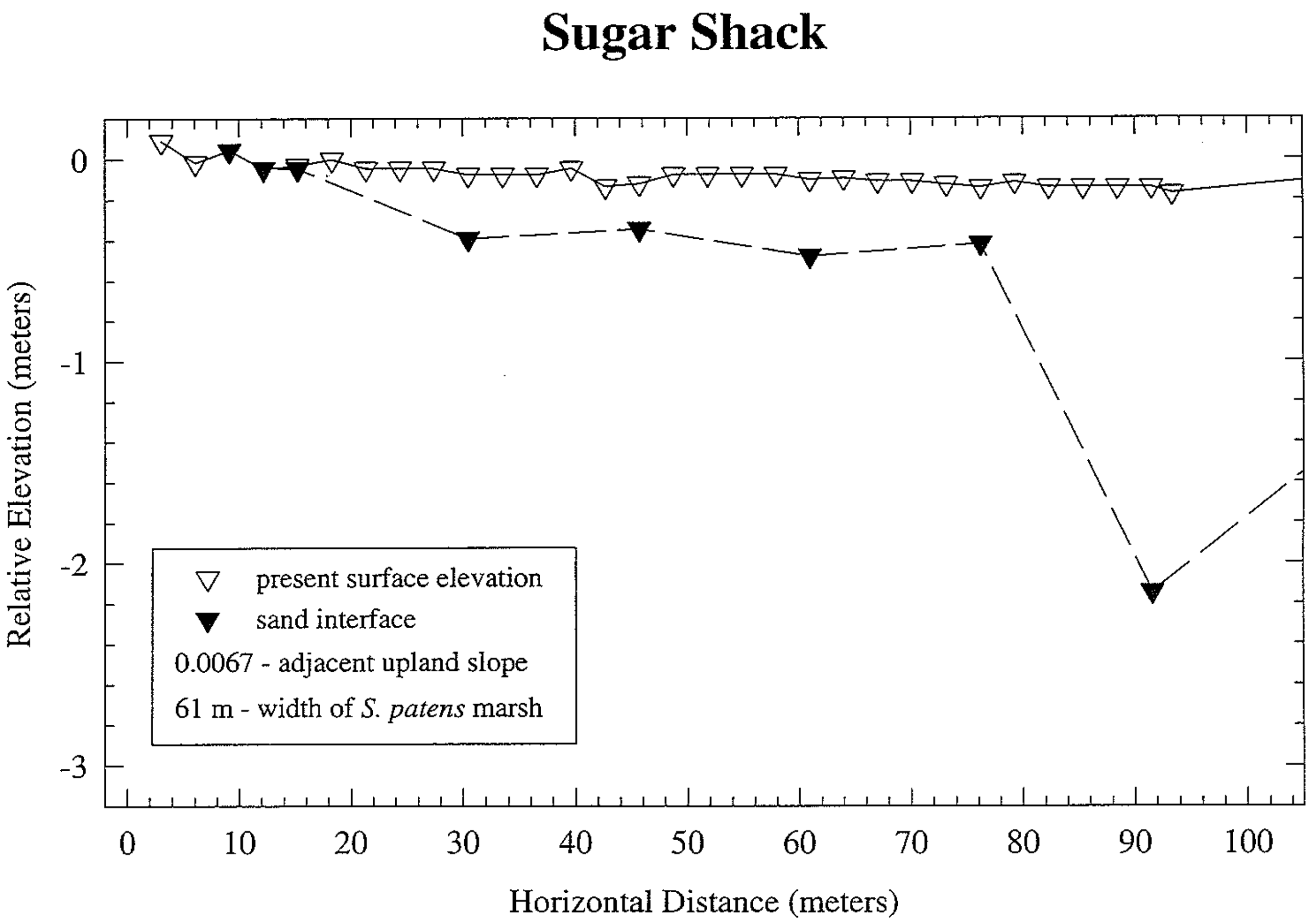
Seawinds



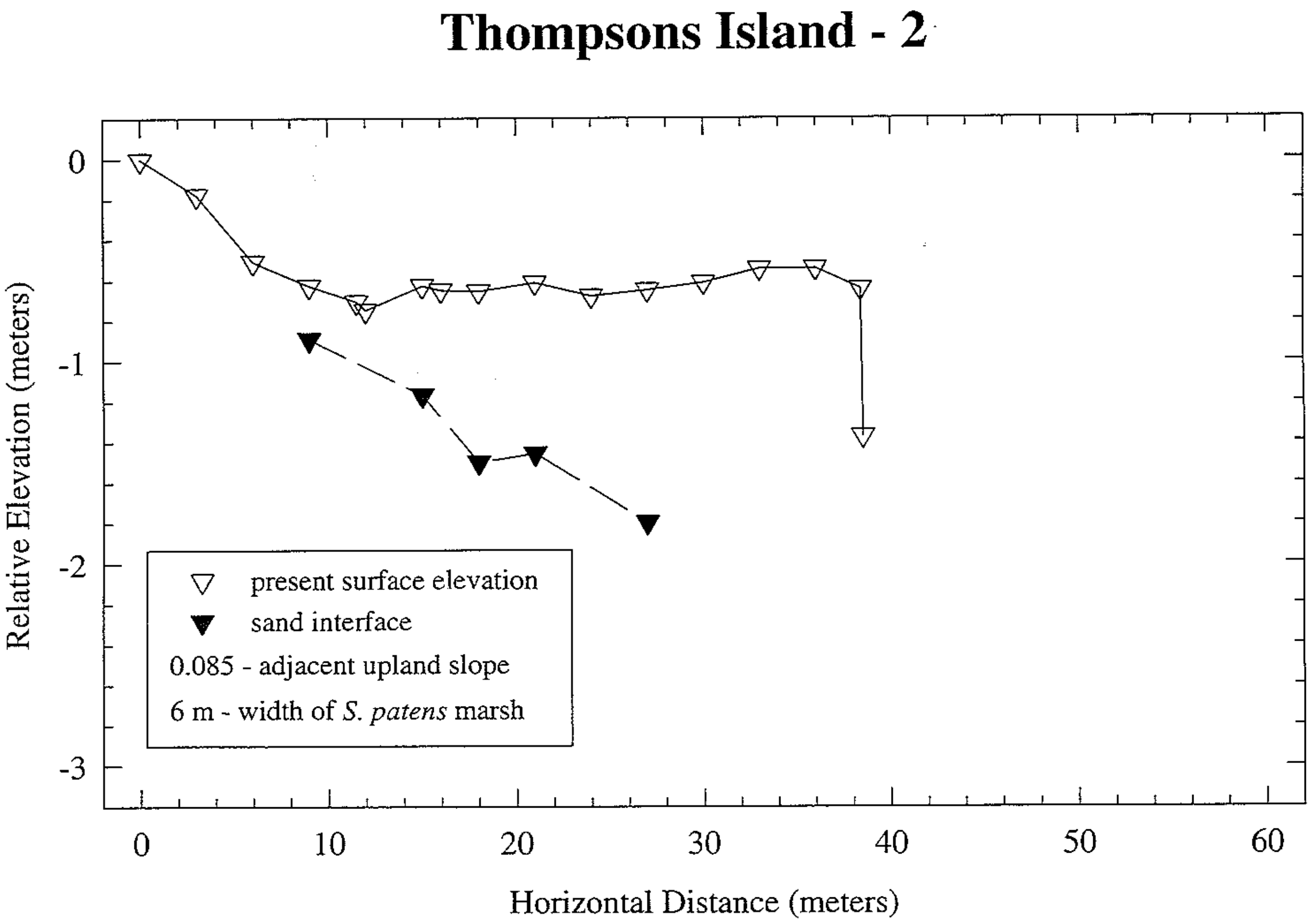
Relative Elevation (meters)
0
-1
-2
-3
-4
-5
Horizontal Distance (meters)
0
10
20
30
▽ present surface elevation
▼ sand interface
0.081 - adjacent upland slope
15 m - width of *S. patens* marsh

Strawberry Landing



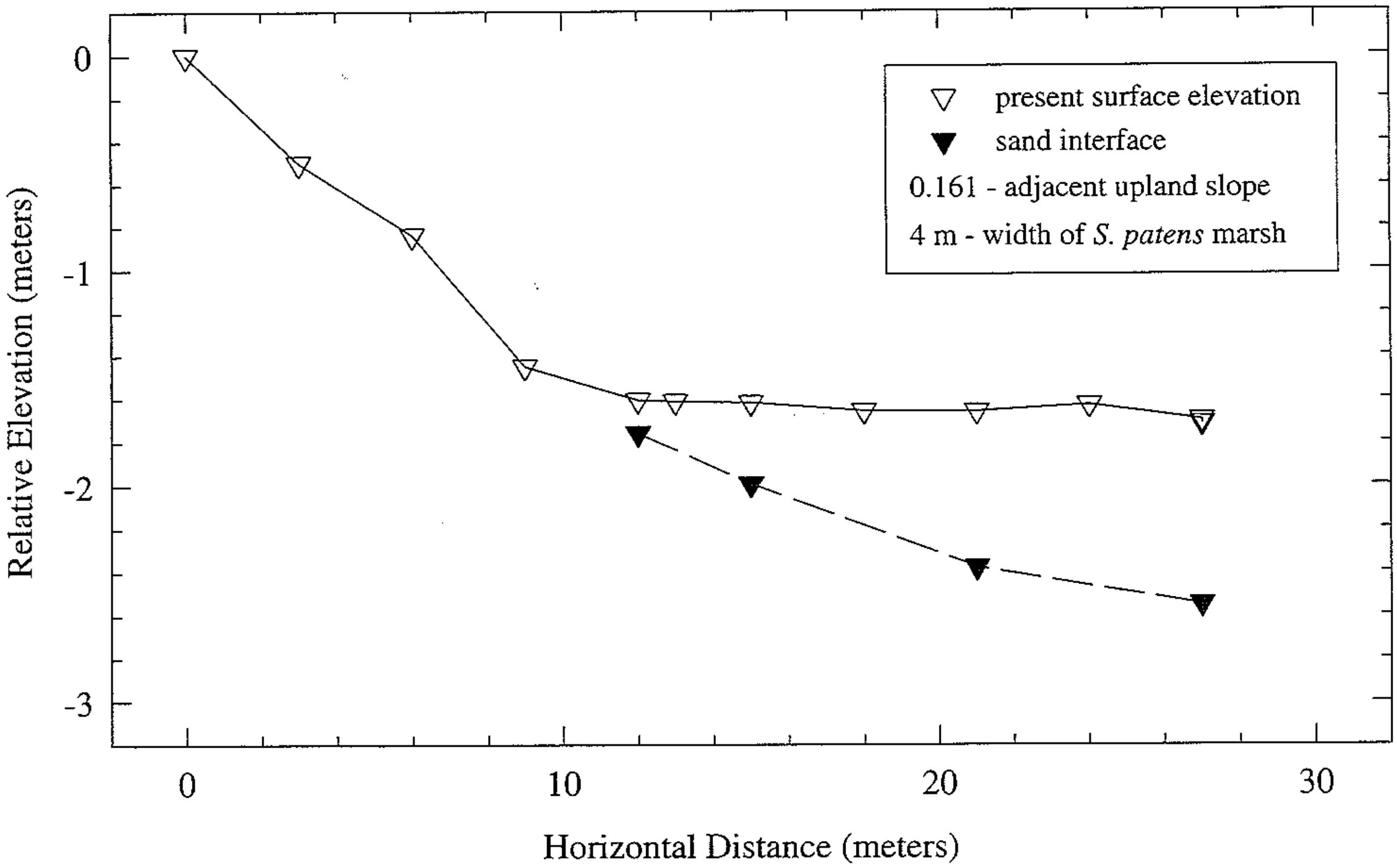

Sugar Shack

Relative Elevation (meters)
0
-1
-2
-3
Horizontal Distance (meters)
0
10
20
30
40
50
60
70
80
90
100
▽ present surface elevation
▼ sand interface
0.0067 - adjacent upland slope
61 m - width of *S. patens* marsh

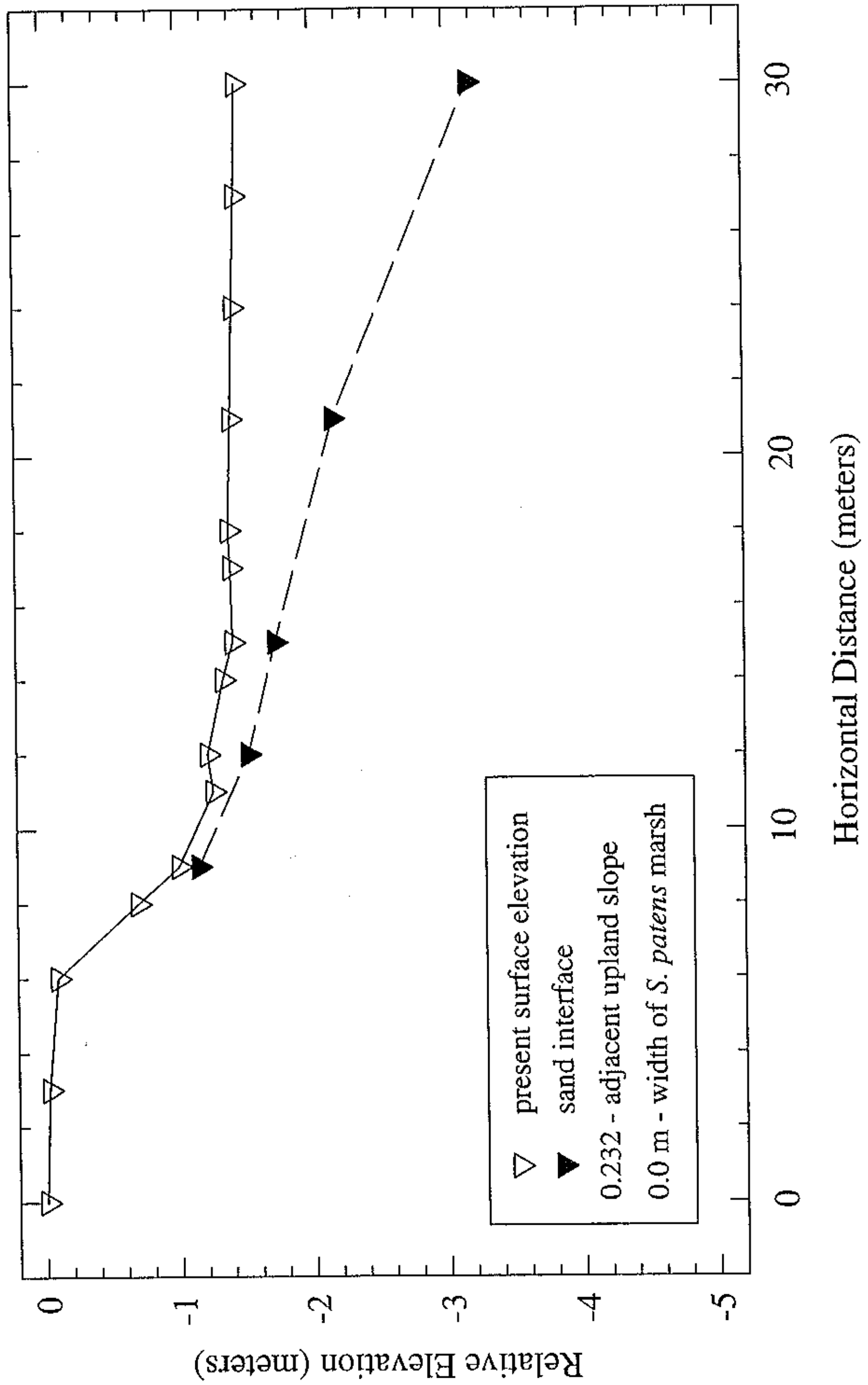


Thompsons Island - 2

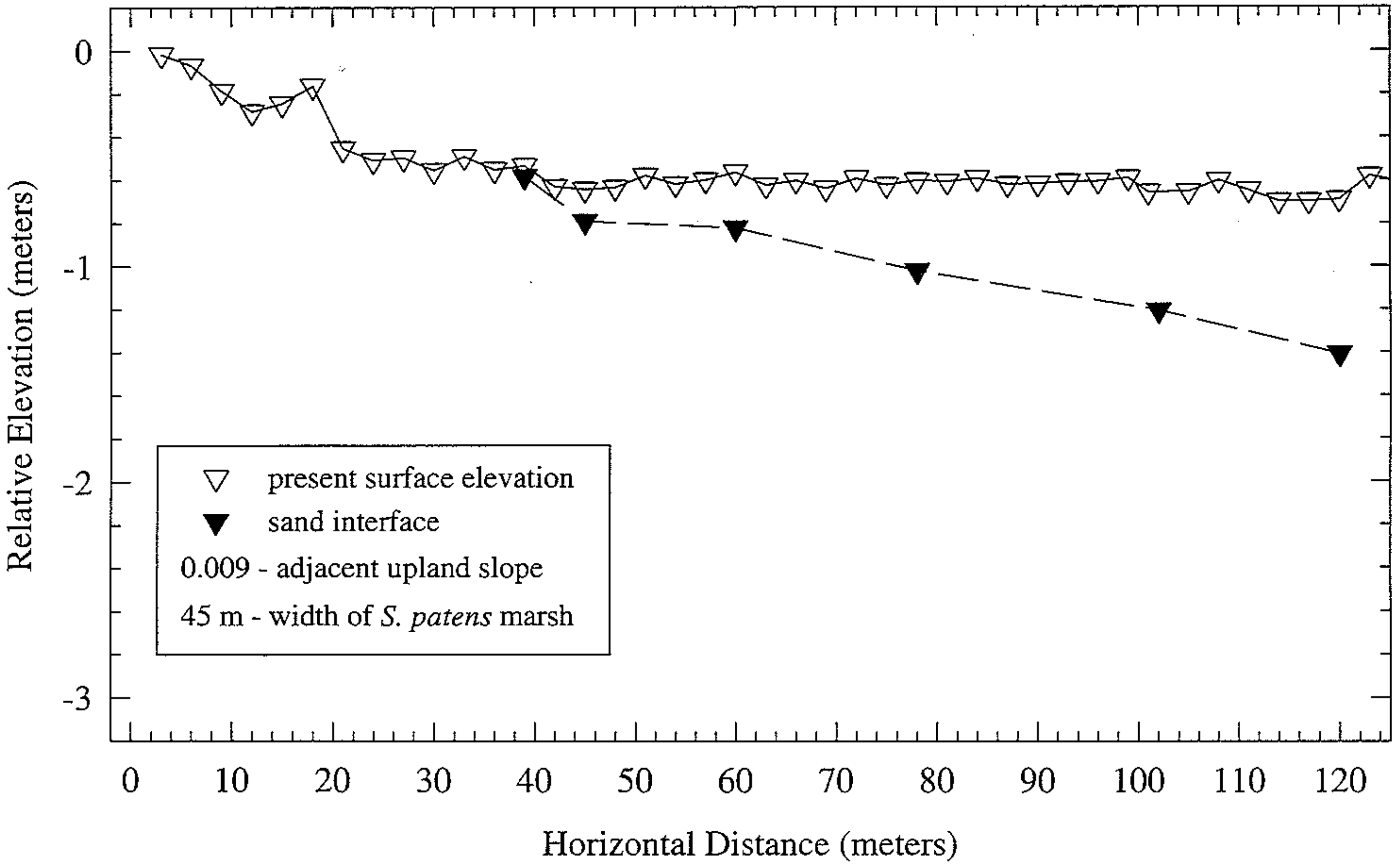
Thompsons Island - 3



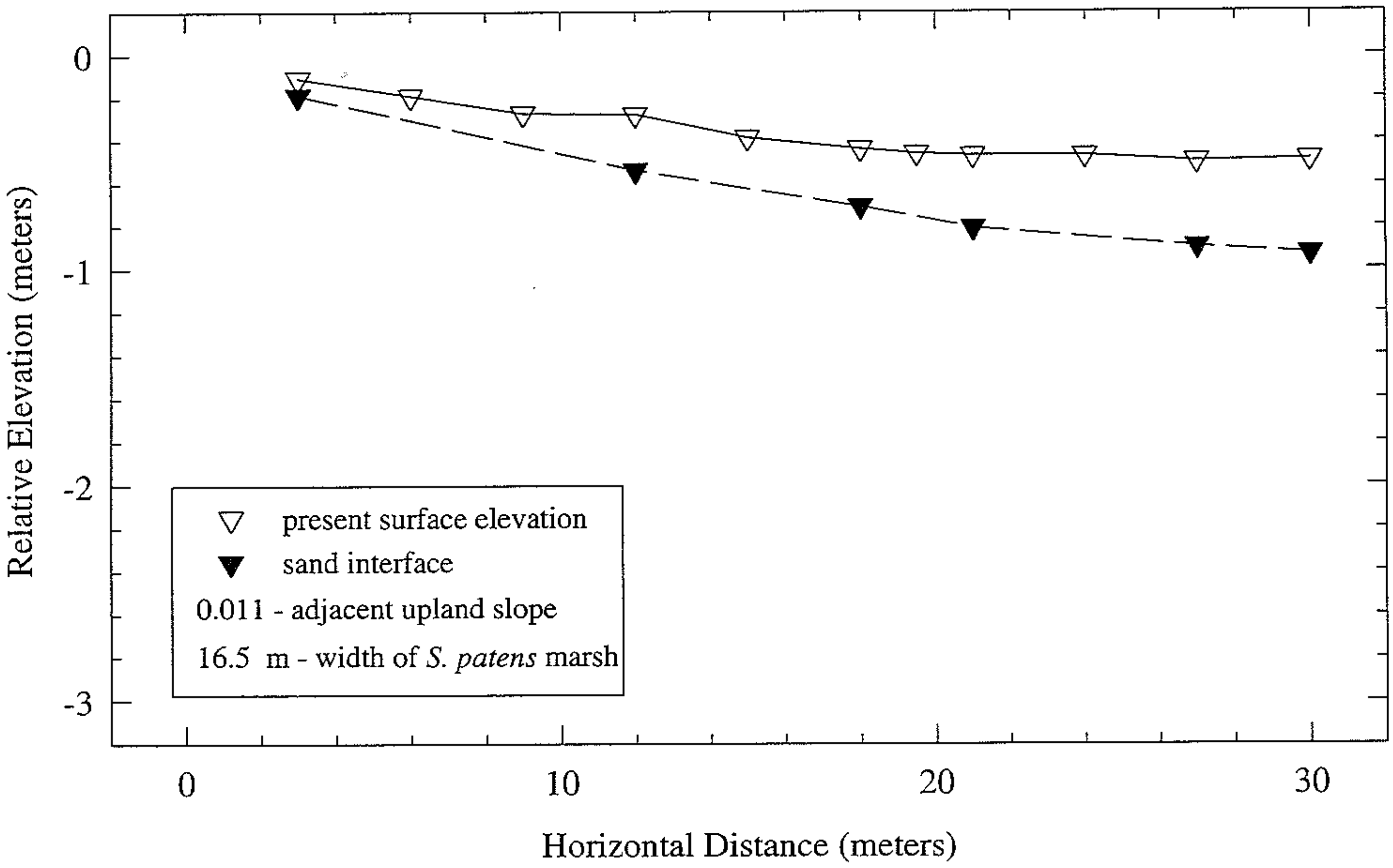
Thompsons Island - 4



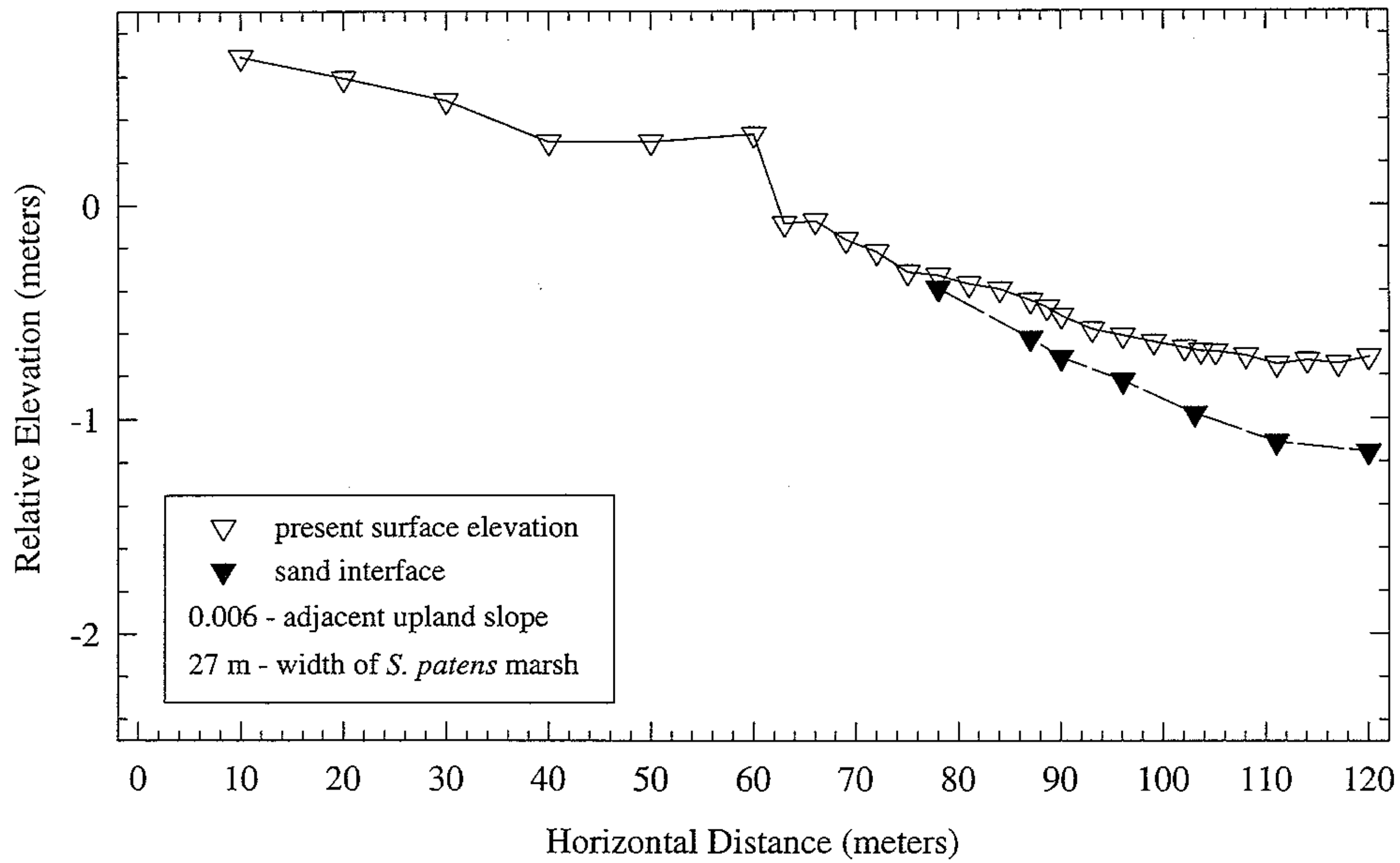
Wall Island


0
-1
-2
-3
Relative Elevation (meters)
0
10
20
30
40
50
60
70
80
90
100
110
120
Horizontal Distance (meters)
▽ present surface elevation
▼ sand interface
0.009 - adjacent upland slope
45 m - width of S. patens marsh

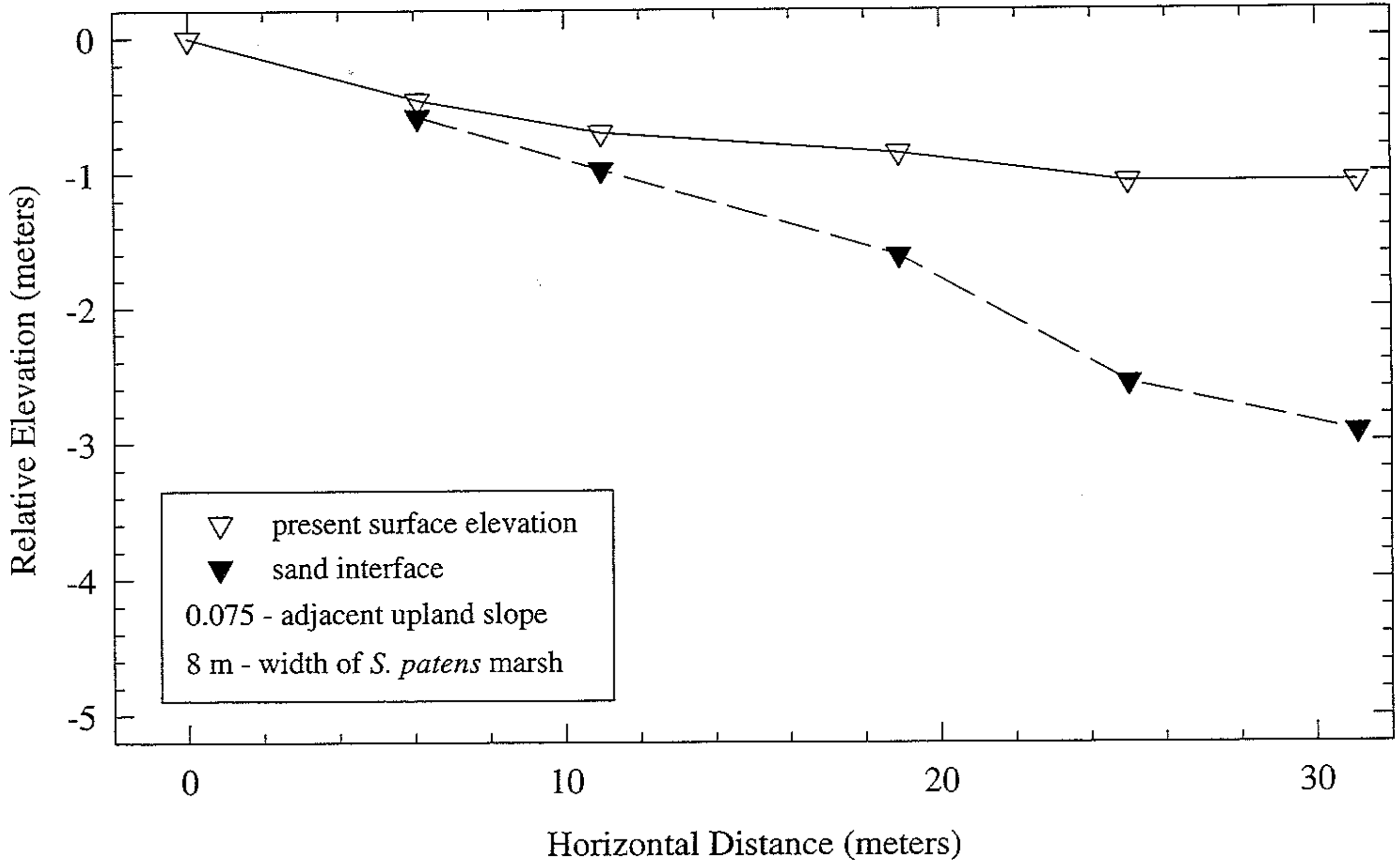
Wm. Derrickson - 1



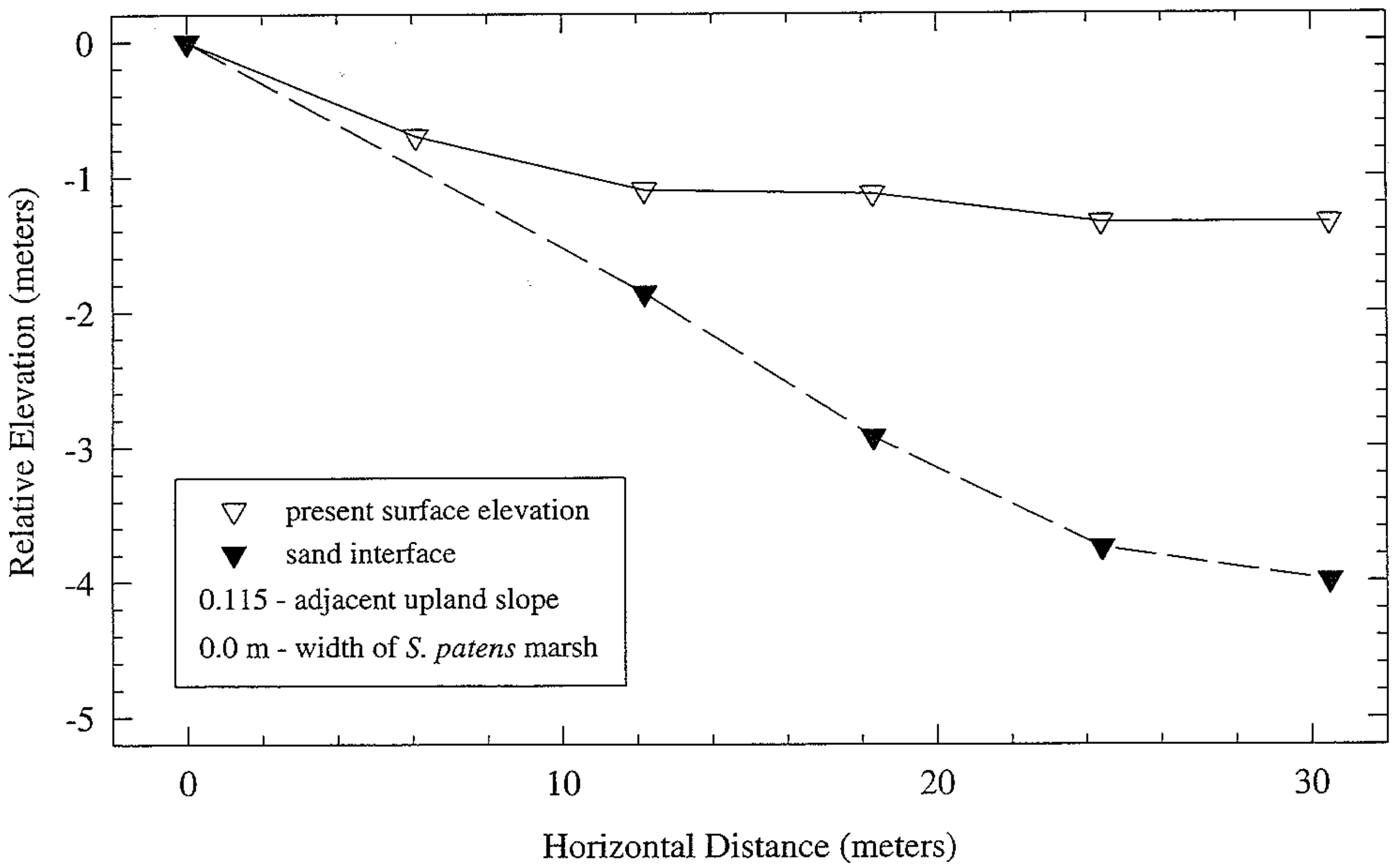
Wm. Derrickson - 2



Wolfe Runne - 1



Wolfe Runne - 2



APPENDIX B

Mean Relative Elevations of High Marsh (*S. patens/D. spicata*) zones and Low Marsh (*S. alterniflora*) zones for all sites

Site #	Location	Mean Relative Elevation (m)		Mean Relative Elevation (m)		Δ Mean Relative Elevation (m)
		S. patens	std. dev.	*S. alterniflora*	std. dev.	*alt < pat*
L&R-5	Anderson	-1.83	0.03	-1.91	0.03	-0.08
IRB-6	Bay Colony 1	-1.89	0.04	-1.98	0.01	-0.09
IRB-11	Bethany Bay 1	-0.06	0.05	-0.19	0.02	-0.13
IRB-12	Bethany Bay 2	-0.28	0.04	-0.38	0.01	-0.10
IRB-13	Bethany Bay 3	-1.76	0.00	-2.06	0.15	-0.30
IRB-14	Bethany Bay 4	-0.74	0.10	-0.83	0.02	-0.09
IRB-9	Blackwater Point	-1.04	0.06	-1.13	0.02	-0.09
IRB-3	Burton 1	-0.26	0.12	-0.63	0.07	-0.36
IRB-4	Burton 2	-0.22	0.03	-0.39	0.05	-0.18
RB-13	Camp Arrowhead	-0.60	0.17	-0.78	0.02	-0.18
RB-14	DE Wildlands 1	-0.28	0.03	-0.52	0.04	-0.24
RB-15	DE Wildlands 2	-1.31	0.10	-1.68	0.10	-0.37
RB-1	Dodd 1	-0.81	0.01	-0.85	0.01	-0.04
RB-7	Draper 1	-0.57	0.12	-0.67	0.04	-0.10
RB-8	Draper 2	-0.49	0.11	-0.69	0.02	-0.20
RB-9	Draper 3	-0.88	0.08	-1.03	0.01	-0.15
L&R-4	Glade 2	-0.50	0.05	-0.68	0.06	-0.17
DB-11	Hercules 2	-0.49	0.05	-0.64	0.06	-0.15
DB-1	Island Field 1	-0.33	0.11	-0.46	0.01	-0.13
DB-2	Island Field 2	-0.52	0.06	-0.65	0.02	-0.12
DB-3	McKim 1	-0.01	0.07	-0.10	0.01	-0.09
DB-4	McKim 2	-0.10	0.15	-0.48	0.07	-0.38
RB-11	Mills 1	-0.52	0.04	-0.55	0.02	-0.03
LAB-1	Muddy Neck 1	-0.19	0.15	-0.33	0.01	-0.14
LAB-2	Muddy Neck 2	-0.01	0.05	-0.11	0.02	-0.09
LAB-3	Muddy Neck 3	-0.03	0.02	-0.10	0.01	-0.07
LAB-5	Mulberry Landin	-0.53	0.03	-0.81	0.00	-0.28
IRB-7	Murray 1	-0.08	0.08	-0.21	0.02	-0.12
IRB-8	Murray 2	-0.18	0.05	-0.38	0.06	-0.20
RB-21	Nats Marsh 1	-0.32	0.17	-0.64	0.27	-0.32

RB-22	Nats Marsh 2	-0.19	0.14	-0.43	0.02	-0.24
DB-7	Oyster Neck Rd.	-0.11	0.08	-0.57	0.02	-0.46
DB-8	Oyster Neck Rd.	-0.11	0.01	-0.34	0.24	-0.22
IRB-15	Past. Point Cov	-0.24	0.09	-0.48	0.02	-0.24
IRB-16	Past. Point Cov	-0.42	0.05	-0.58	0.04	-0.15
RB-16	Pot Nets North	-0.75	0.03	-0.92	0.04	-0.17
RB-17	Pot Nets North	-0.36	0.22	-0.65	0.04	-0.29
RB-18	Pot Nets North	-0.24	0.23	-0.49	0.03	-0.25
RB-19	Pot Nets North	-0.18	0.04	-0.27	0.02	-0.10
DB-5	Prime Hook	-0.20	0.10	-0.30	0.02	-0.10
DB-7	Ritter 1	-1.39	0.11	-1.54	0.02	-0.15
DB-8	Ritter 2	-0.03	0.02	-0.04	0.01	-0.02
DB-9	Ritter 3	-0.10	0.01	-0.14	0.00	-0.04
DB-10	Ritter 4	-0.12	0.07	-0.21	0.02	-0.09
IRB-17	Seawinds	-1.62	0.32	-2.01	0.04	-0.40
LAB-6	Strawberry Land	-0.35	0.03	-0.39	0.02	-0.04
RB-10	Sugar Shack	-0.06	0.04	-0.22	0.51	-0.16
RB-3	Thompsons Isl.	-0.64	0.09	-0.69	0.24	-0.05
RB-4	Thompsons Isl.	-1.61	0.01	-1.67	0.03	-0.06
DB-6	Wall Island	-0.60	0.03	-0.67	0.09	-0.07
IRB-18	Wm. Derrickson	-0.31	0.10	-0.48	0.02	-0.16
IRB-19	Wm. Derrickson	-0.41	0.08	-0.69	0.05	-0.28
L&R-1	Wolfe Runne 1	-0.78	0.11	-1.00	0.12	-0.22

Mean Difference in Elevation: -0.17 m (Std. Dev. =0.11 m; n=53; max = -0.02, min = -0.46)

APPENDIX C

Depth to Pre-Transgressive Surface (Marsh Thickness)
for *S. alterniflora* and *S. patens* Marsh Surface Vegetation

Depth to pre-transgressive sandy surface (cm)	
Surface Vegetation: *S. patens*	Surface Vegetation: *S. alterniflora*
30	110
0	183
0	146
0	75
7	264
3	116
30	115
32	124
38	130
4	72
8	125
11	143
10	198
20	179
36	20
2	16
20	58
0	72
4	109
8	47
11	95
31	67
11	56
15	45
15	50
26	53
53	132
15	170
37	65
30	84
51	119
57	46
3	57

Depth to pre-transgressive sandy surface (cm)	
Surface Vegetation: *S. patens*	Surface Vegetation: *S. alterniflora*
8	110
8	31
16	40
12	83
15	87
21	101
8	72
0	84
15	75
26	183
42	67
13	91
10	106
10	64
0	178
0	96
18	18
28	25
24	24
15	72
35	55
10	71
13	134
12	40
4	34
24	50
42	120
28	150
8	65
15	100
21	50
18	42
8	79

Depth to pre-transgressive sandy surface (cm)	
Surface Vegetation: *S. patens*	Surface Vegetation: *S. alterniflora*
27	16
10	21
9	41
10	29
11	36
14	44
11	34
10	40
23	44
34	72
8	81
17	140
26	29
16	38
26	56
38	41
90	97
10	114
10	124
11	134
15	21
22	27
18	30
33	71
70	99
13	127
34	28
5	29
10	56
15	78
21	82
55	104
82	146

Depth to pre-transgressive sandy surface (cm)	
Surface Vegetation: *S. patens*	Surface Vegetation: *S. alterniflora*
20	44
48	91
65	121
10	210
9	267
34	20
5	32
32	32
0	35
13	40
0	49
22	63
28	86
26	104
38	182
23	227
24	70
36	106
11	128
21	19
21	24
28	30
7	64
18	44
52	97
81	200
50	267
20	267
22	267
16	267
75	181
9	114
12	43

Depth to pre-transgressive sandy surface (cm)	
Surface Vegetation: *S. patens*	Surface Vegetation: *S. alterniflora*
34	60
15	13
18	48
28	42
13	38
15	49
30	151
35	221
17	232
40	116
15	51
31	59
22	87
35	98
35	36
	45
	80
	27
	56
	32
	27
	150
	102
	165
	100
	100
	16
	11
	60
	85
	59
	53
	52
	52
	40
	46
	205
	111

APPENDIX D

Slope of Adjacent Upland Surface and
Associated Width of *S. patens* (high marsh) surface

Site Number	Location	Slope of Adjacent Upland Surface	Width of *S. patens* (surface environ.)
		(tangent)	(m)
L&R-5	Anderson	0.076	4.00
IRB-6	Bay Colony-1	0.037	6.00
RB-6	Bay Vista-2	0.096	0.00
IRB-11	Bethany Bay-1	0.006	66.00
IRB-12	Bethany Bay-2	0.009	43.50
IRB-13	Bethany Bay-3	0.250	3.00
IRB-14	Bethany Bay-4	0.028	6.00
IRB-10	Bethany Forest-1	0.142	0.00
IRB-9	Blackwater Point	0.021	57.00
IRB-2	Boat House Cove	0.097	0.00
IRB-3	Burton 1	0.083	2.00
IRB-4	Burton 2	0.091	2.00
RB-13	Camp Arrowhead2	0.043	12.00
DB-12	Canary Creek 1	0.075	0.00
RB-14	DE Wildlands-1	0.021	18.00
RB-15	DE Wildlands-2	0.079	6.00
RB-1	Dodd-1	0.133	2.00
RB-2	Dodd - 2	0.108	0.00
RB-7	Draper-1	0.014	15.00
RB-8	Draper-2	0.030	6.00
RB-9	Draper-3	0.056	9.00
L&R-3	Glade-1	0.118	0.00
L&R-4	Glade-2	0.027	6.00
DB-11	Hercules -2	0.010	8.00
RB-12	Horse Island	0.016	51.00
IRB-1	Indian Landing	0.011	7.00
DB-1	Island Field-1	0.070	3.00
DB-2	Island Field-2	0.010	12.00
IRB-5	Lingo Point	0.097	0.00
LAB-7	Little Assawoman	0.064	0.00
DB-3	McKim-1	0.010	36.00
DB-4	McKim-2	0.024	9.00
RB-11	Mills 1	0.020	15.00
LAB-2	Muddy Neck-2	0.023	16.00

Site Number	Location	Slope of Adjacent Upland Surface	Width of *S. patens* (surface environ.)
		(tangent)	(m)
LAB-3	Muddy Neck-3	0.053	7.00
LAB-4	Muddy Neck-1	0.046	9.00
LAB-5	Mulberry Landing	0.030	12.00
IRB-7	Murray-1	0.030	11.00
IRB-8	Murray-2	0.033	11.00
RB-21	Nats Marsh-2	0.029	18.00
RB-22	Nats Marsh-2	0.034	12.00
DB-7	Oyster Neck Road 1	0.020	66.00
DB-8	Oyster Neck Road 2	0.023	18.00
DB-13	Pagan Creek	0.111	0.00
IRB-15	Past.Pt.Cove-1	0.020	15.00
IRB-16	Past.Pt.Cove-2	0.030	10.00
RB-16	Pot Nets North 1	0.036	13.00
RB-17	Pot Nets North 2	0.010	26.70
RB-18	Pot Nets North 3	0.032	12.00
RB-19	Pot Nets North 4	0.013	42.00
RB-20	Pot Nets North 5	0.005	35.00
DB-5	Prime Hook RS	0.030	15.00
RB-23	Pullover-1	0.084	1.00
DB-7	Ritter-1	0.059	4.00
DB-8	Ritter-2	0.030	24.00
DB-9	Ritter-3	0.031	13.50
DB-10	Ritter-4	0.016	12.00
IRB-17	Seawinds	0.081	15.00
LAB-6	Strawberry Landing	0.012	16.00
RB-10	Sugar Shack	0.007	61.00
RB-3	Thompsons Is-2	0.085	6.00
RB-4	Thompsons Is-3	0.161	4.00
RB-5	Thompsons Is-4	0.232	0.00
DB-6	Wall Island-1	0.009	45.00
IRB-18	Wm.Derrickson-	0.011	16.50
IRB-19	Wm.Derrickson-	0.006	27.00
L&R-1	Wolfe Runne-1	0.075	8.00
L&R-2	Wolfe Runne-2	0.115	0.00

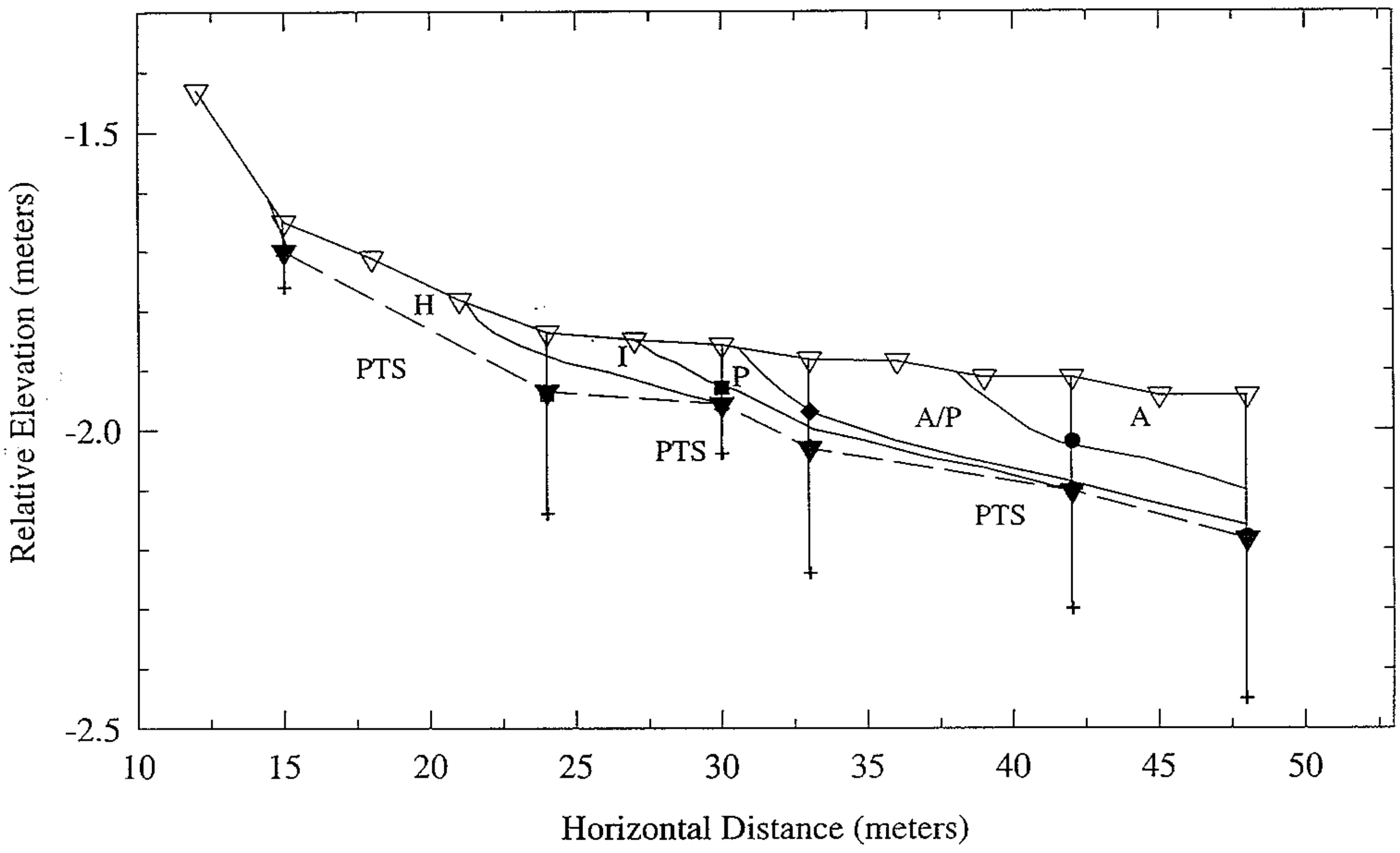
APPENDIX E

Stratigraphic Cross-Sections of 69 Individual Transect Sites

(For relative elevation profiles of cross-sectional profile sites, see Appendix A)

Abbreviations	
A	*Spartina alterniflora*
P	*Spartina patens/Distichlis spicata*
A/P	S. *Alterniflora/S. Patens* mix
I	*Iva frutescens/Baccharis halimifolia*
I/P	*Iva/Baccharis/S. patens* mix
Ph	*Phragmites australis*
FWM	Freshwater marsh
S	Soil/humus or soil transition zone
PTS	Pre-Transgressive Surface (sand)

Anderson



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Relative Elevation (meters)
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45
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Horizontal Distance (meters)
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A/P
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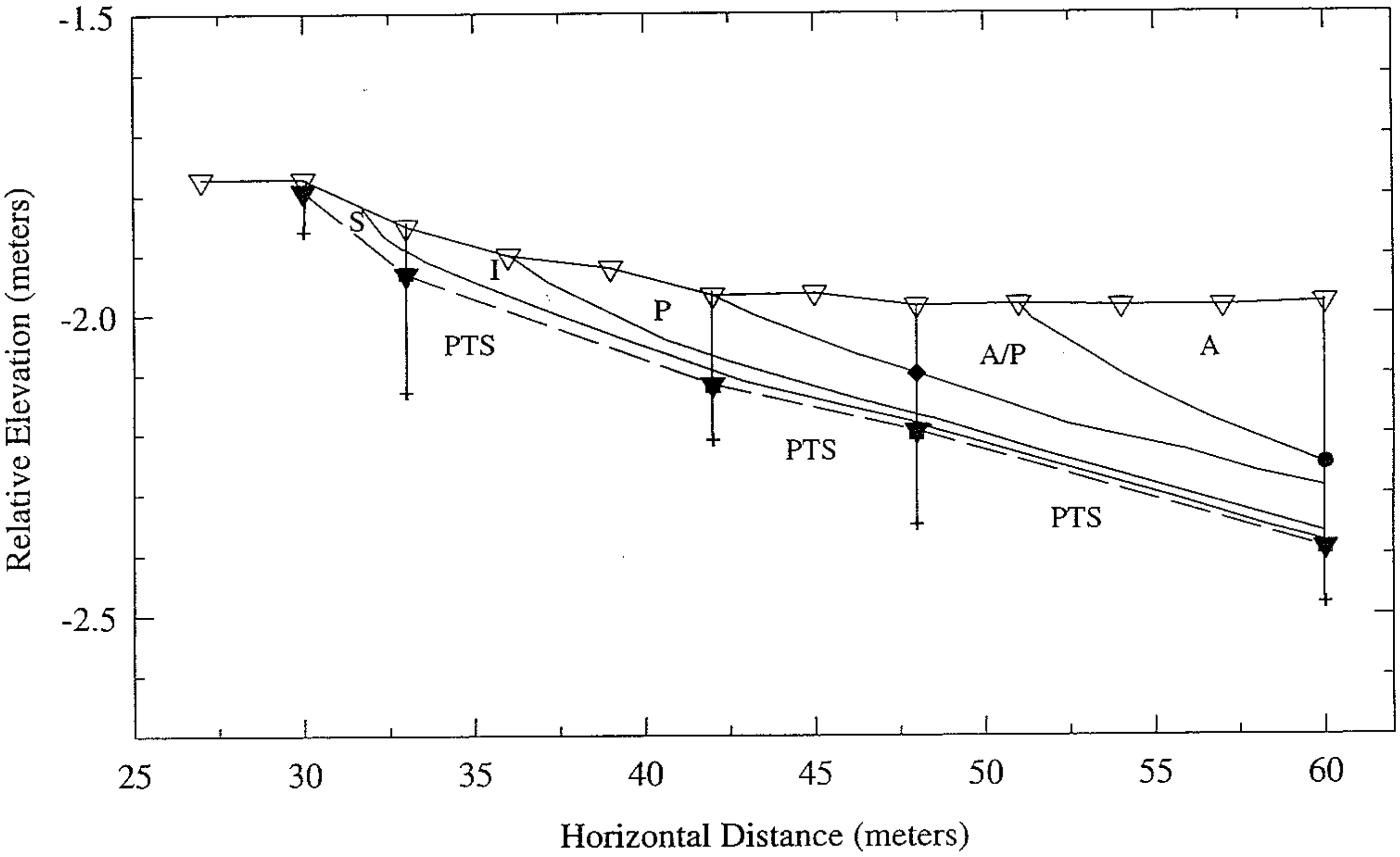
PTS

Bay Colony - 1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Relative Elevation (meters)
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55
60
Horizontal Distance (meters)
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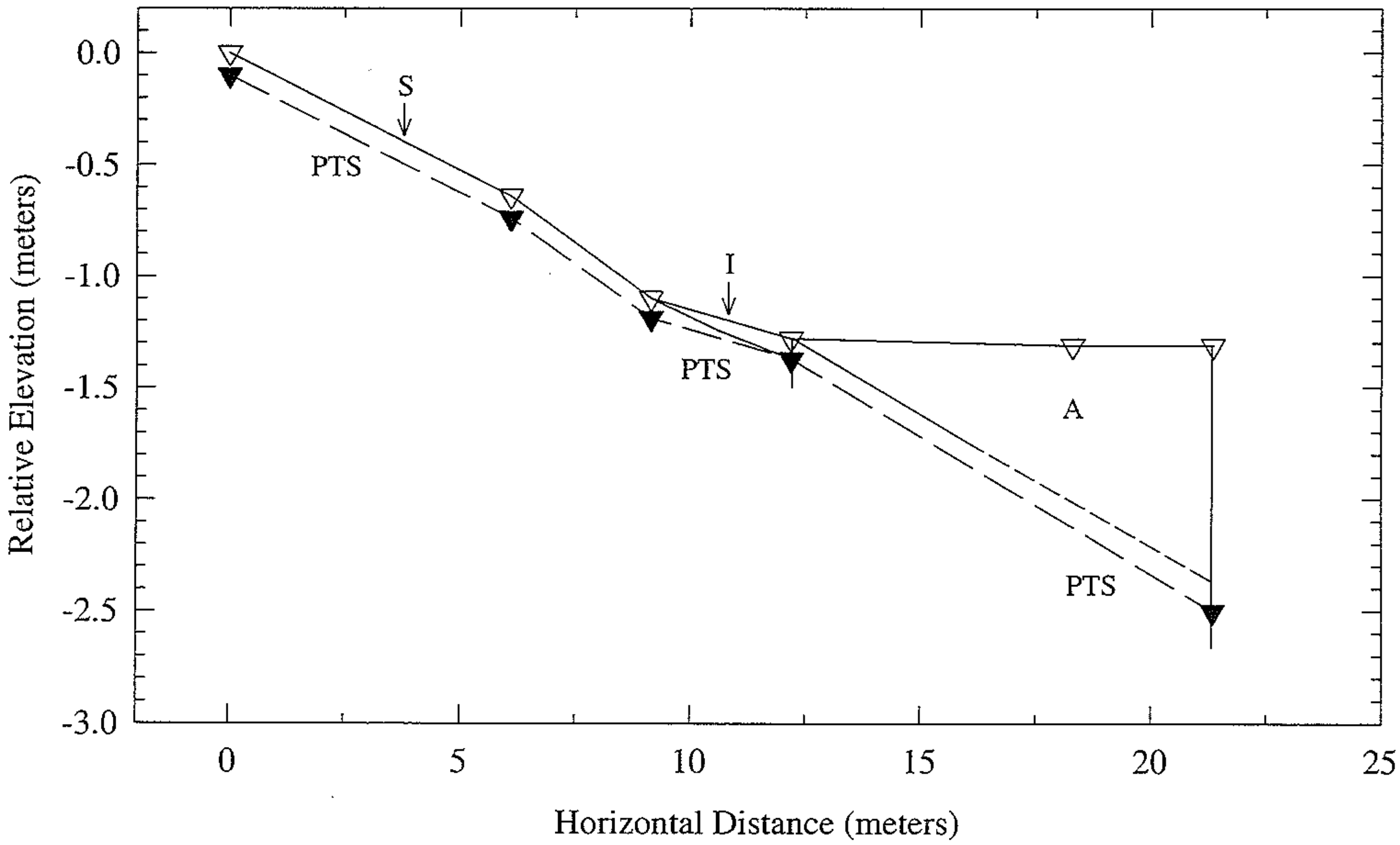
PTS

Bay Vista - 1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Relative Elevation (meters)
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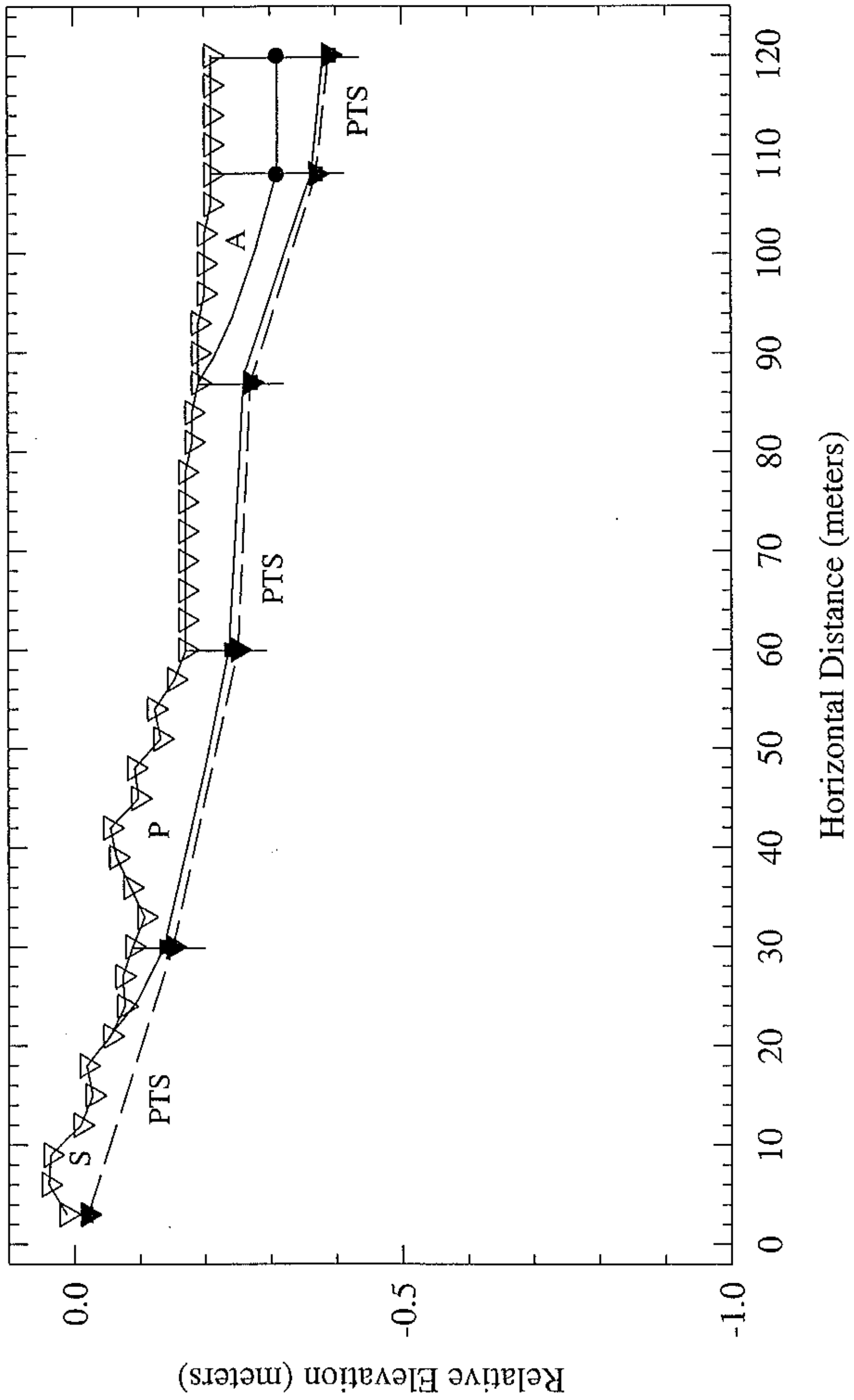
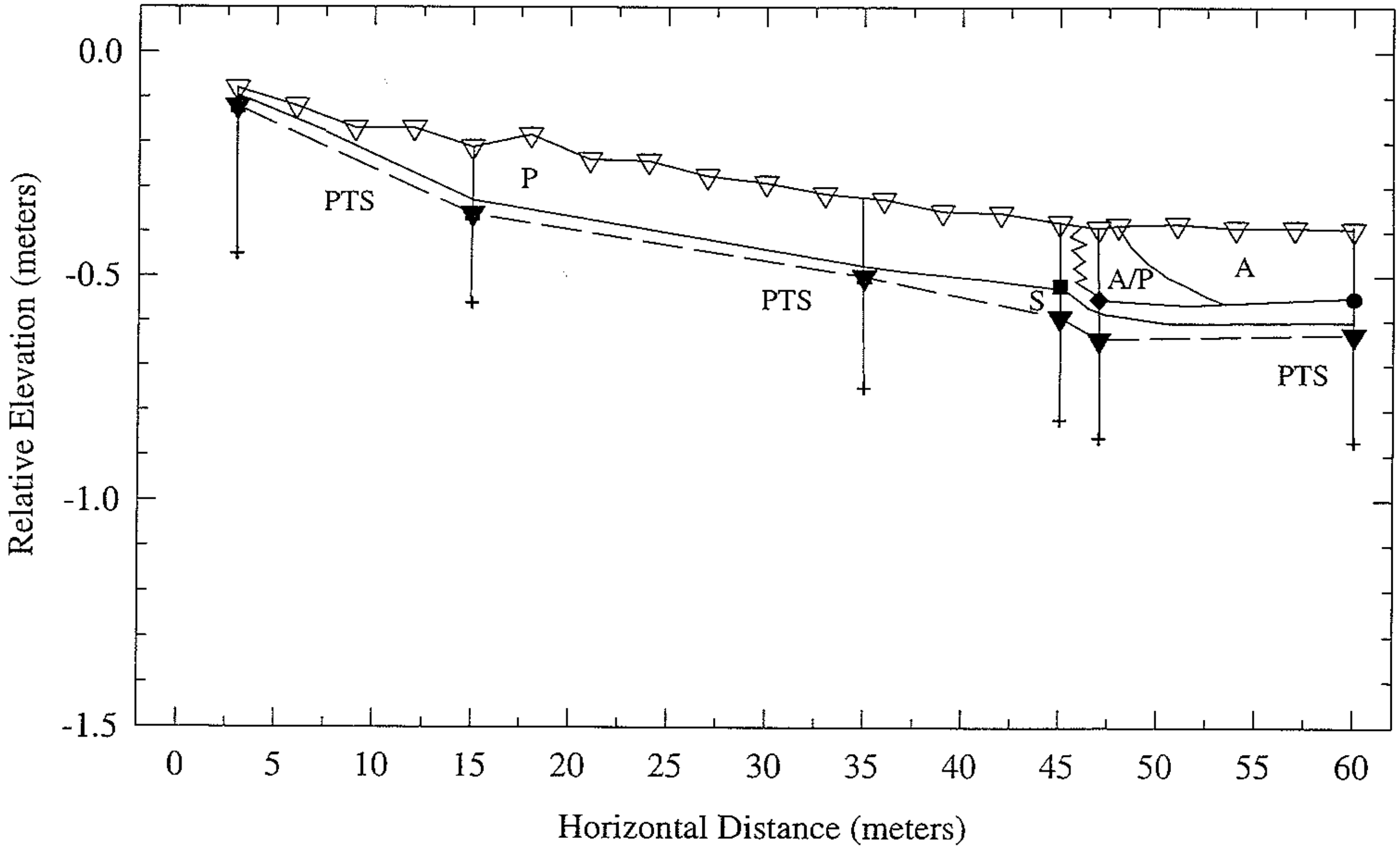
Horizontal Distance (meters)

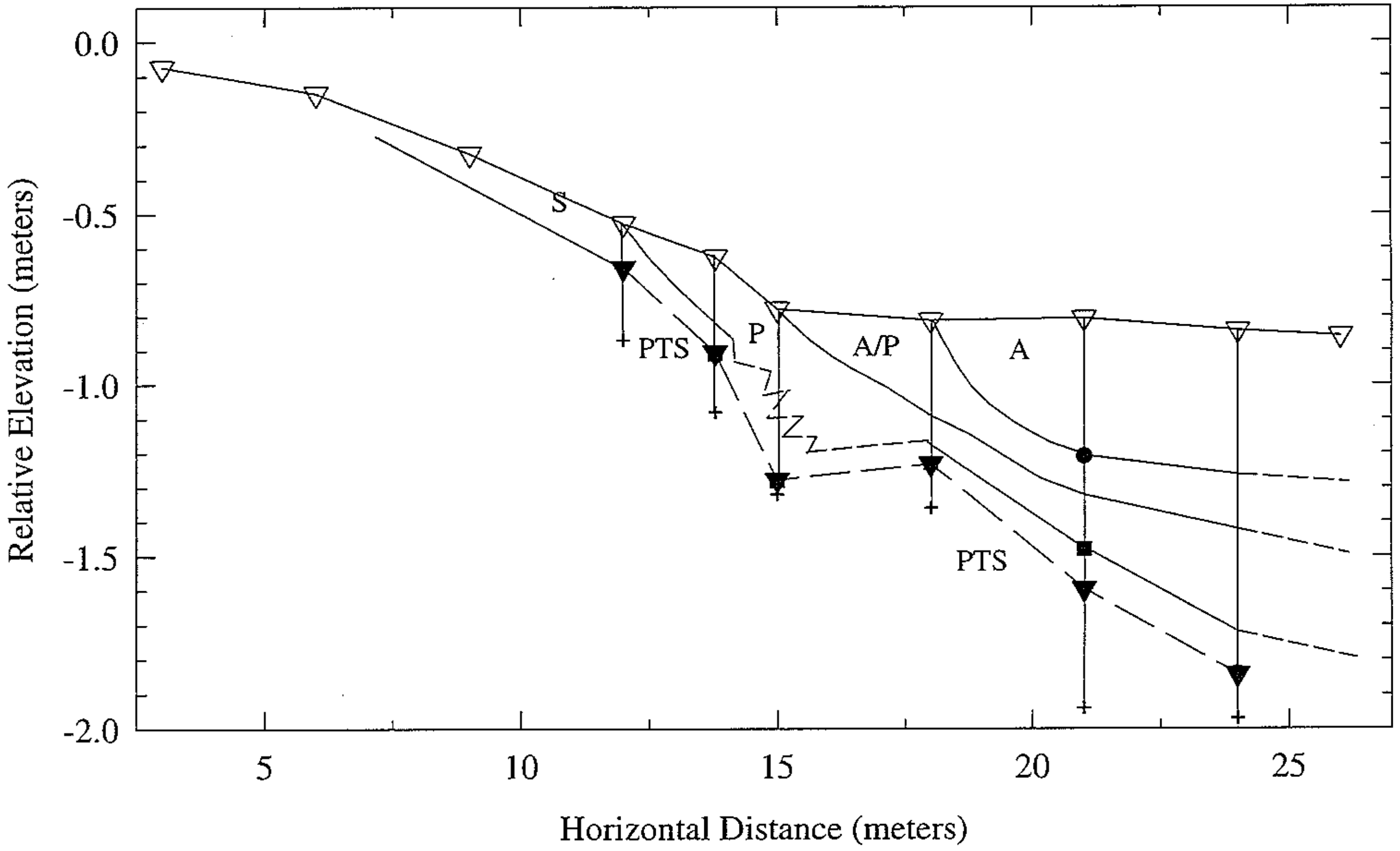
Bethany Bay - 1



Bethany Bay - 2



Bethany Bay - 4



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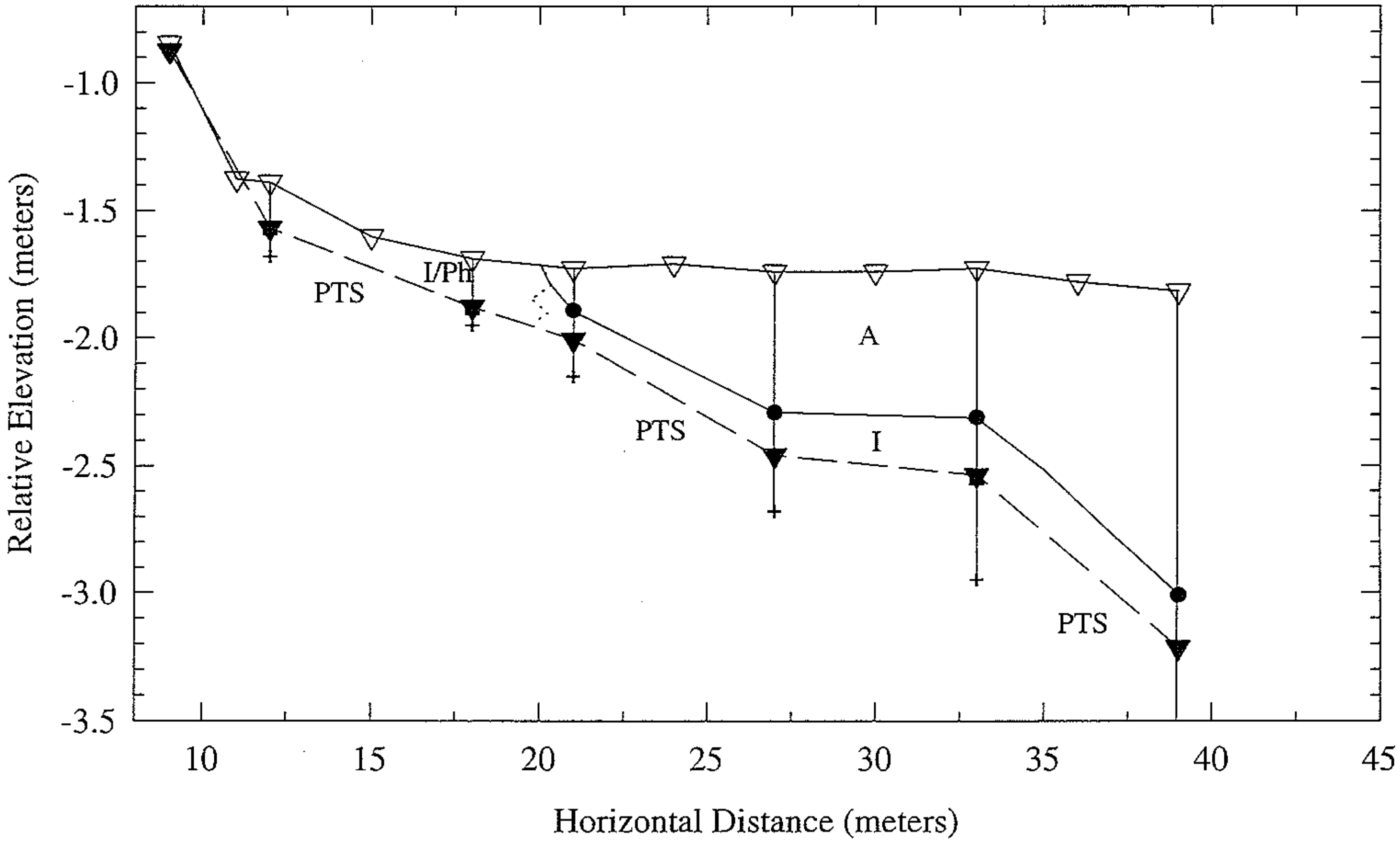
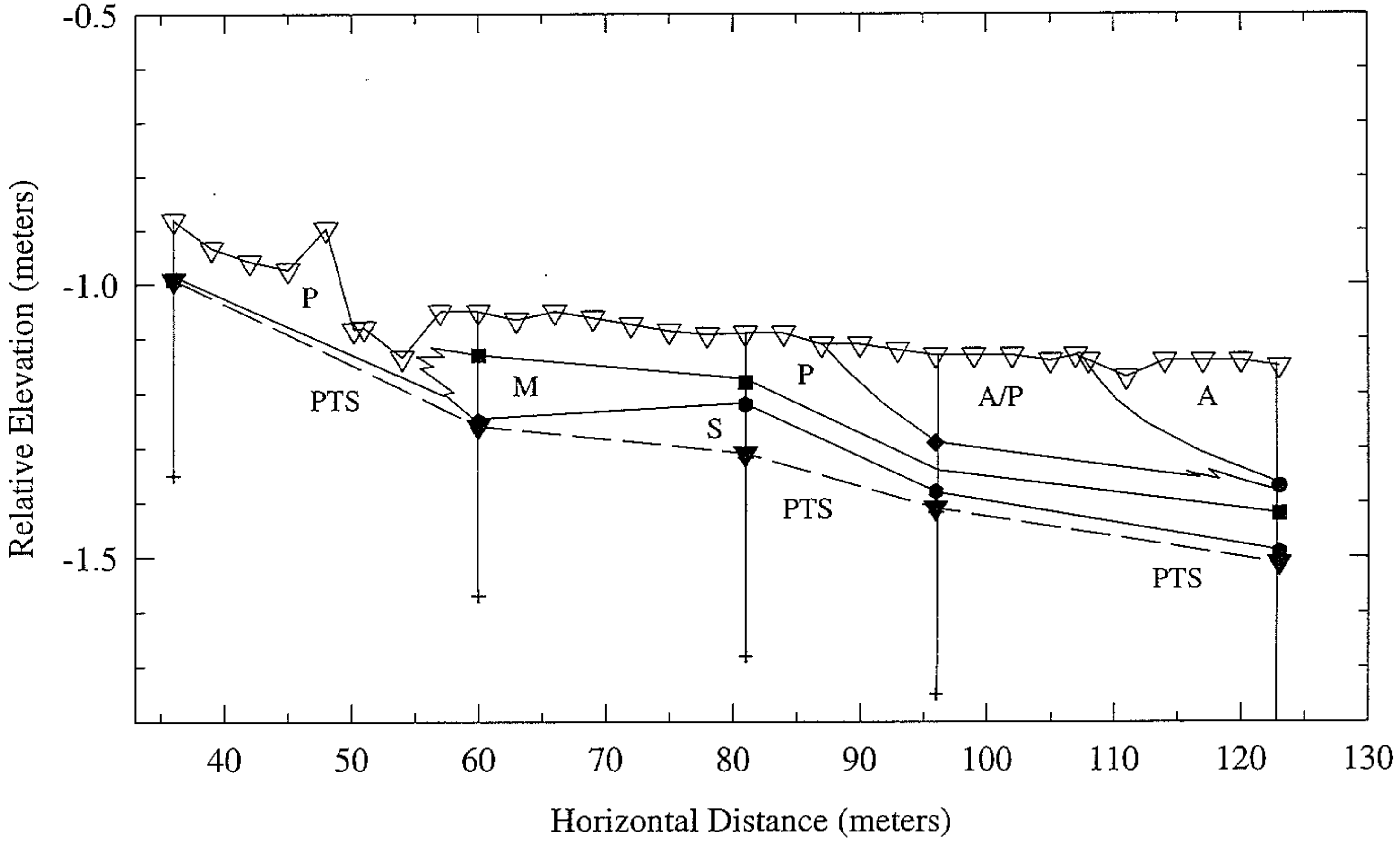
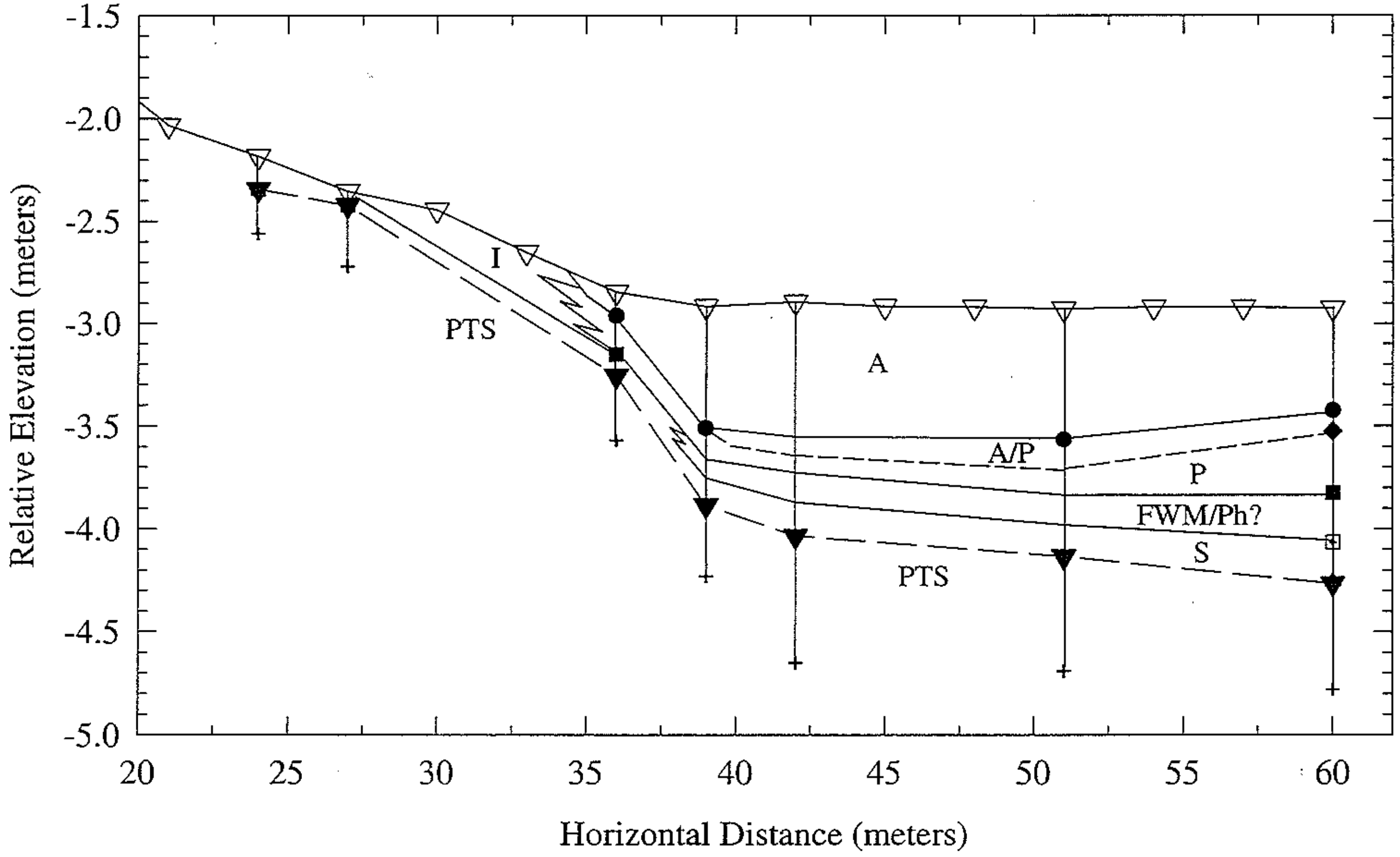


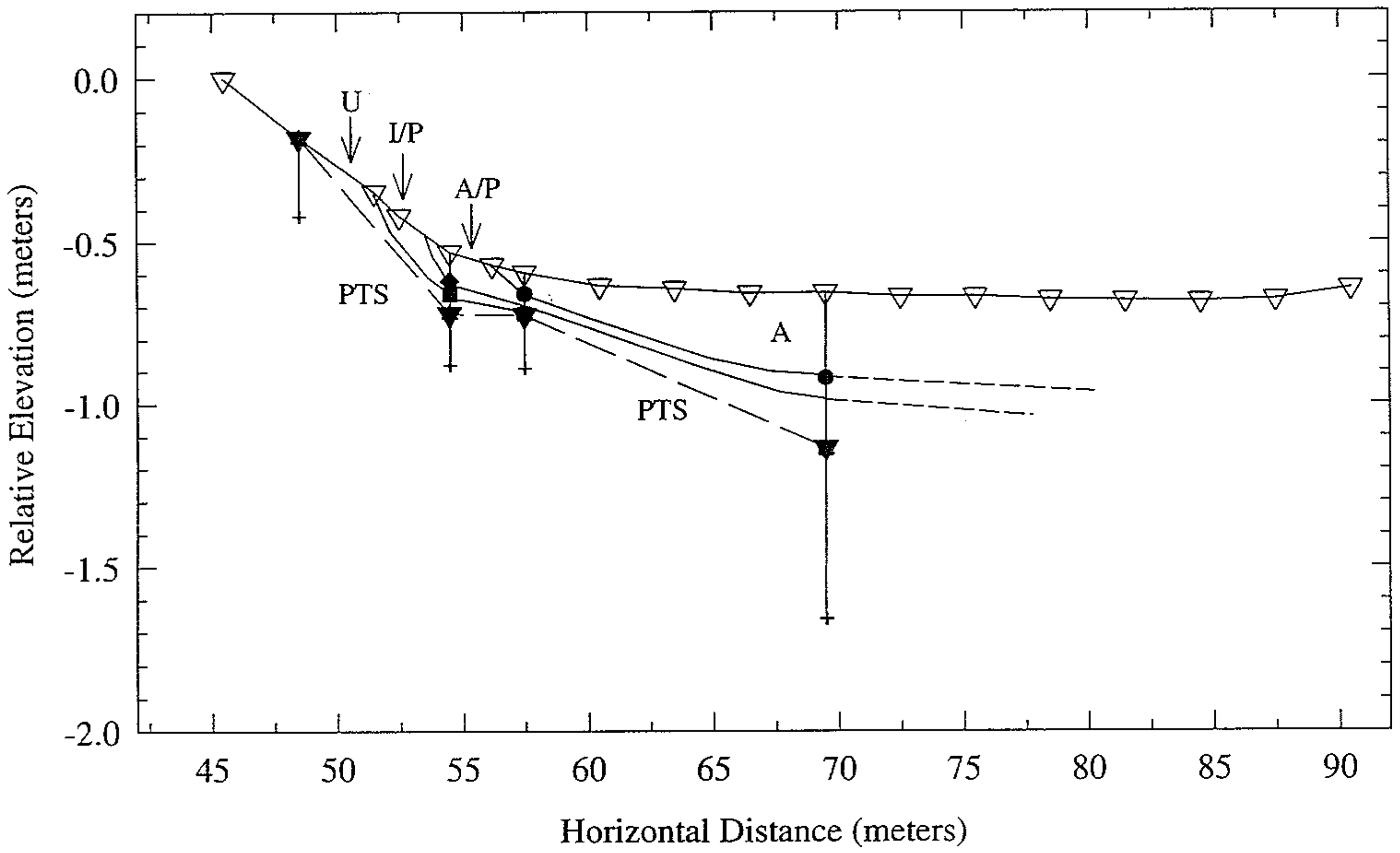
Blackwater Point



Boat House Cove - 2



Burton - 1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-2.0
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50
55
60
65
70
75
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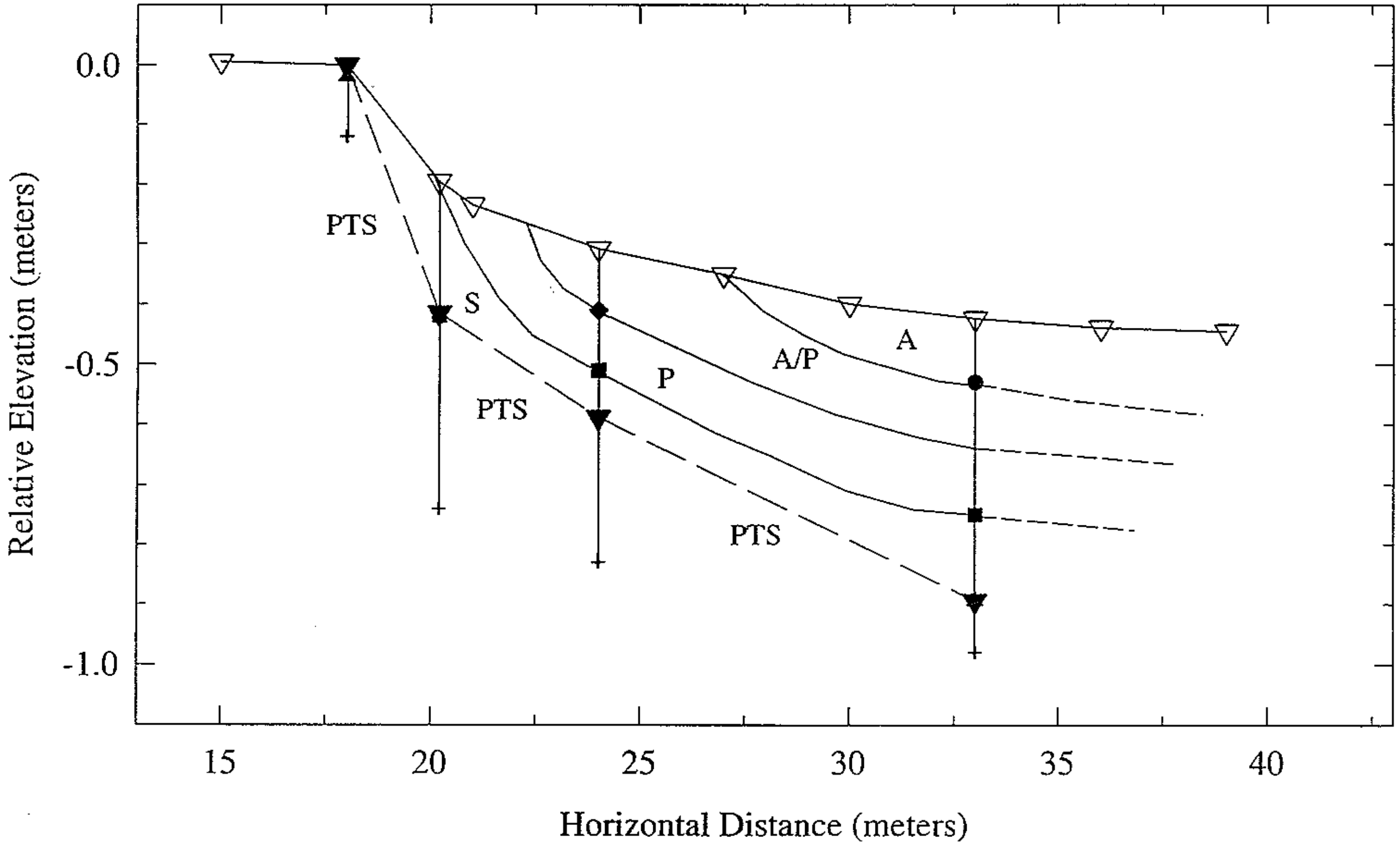
85
90
Horizontal Distance (meters)

Burton - 2


0.0
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Relative Elevation (meters)
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Horizontal Distance (meters)
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PTS
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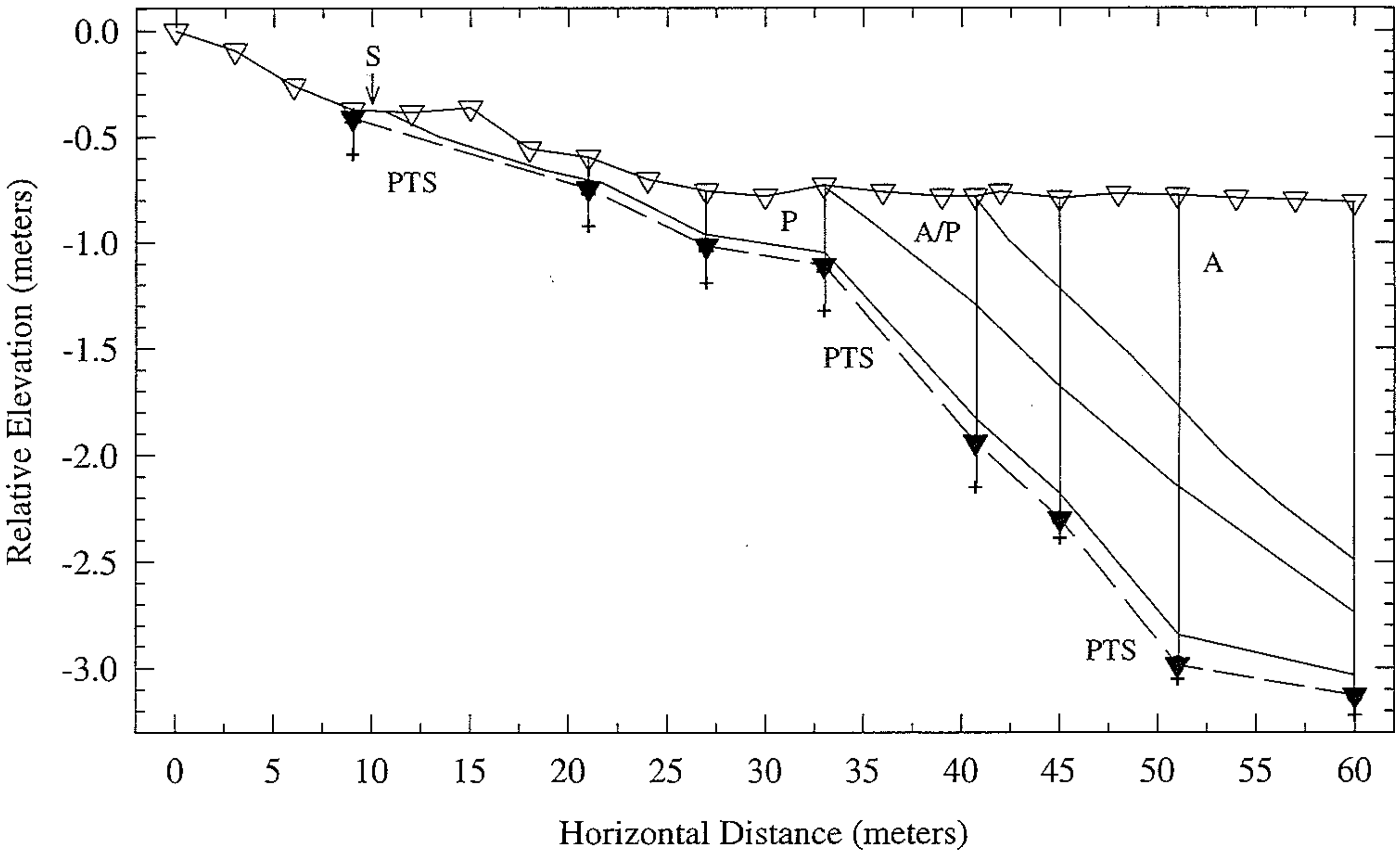
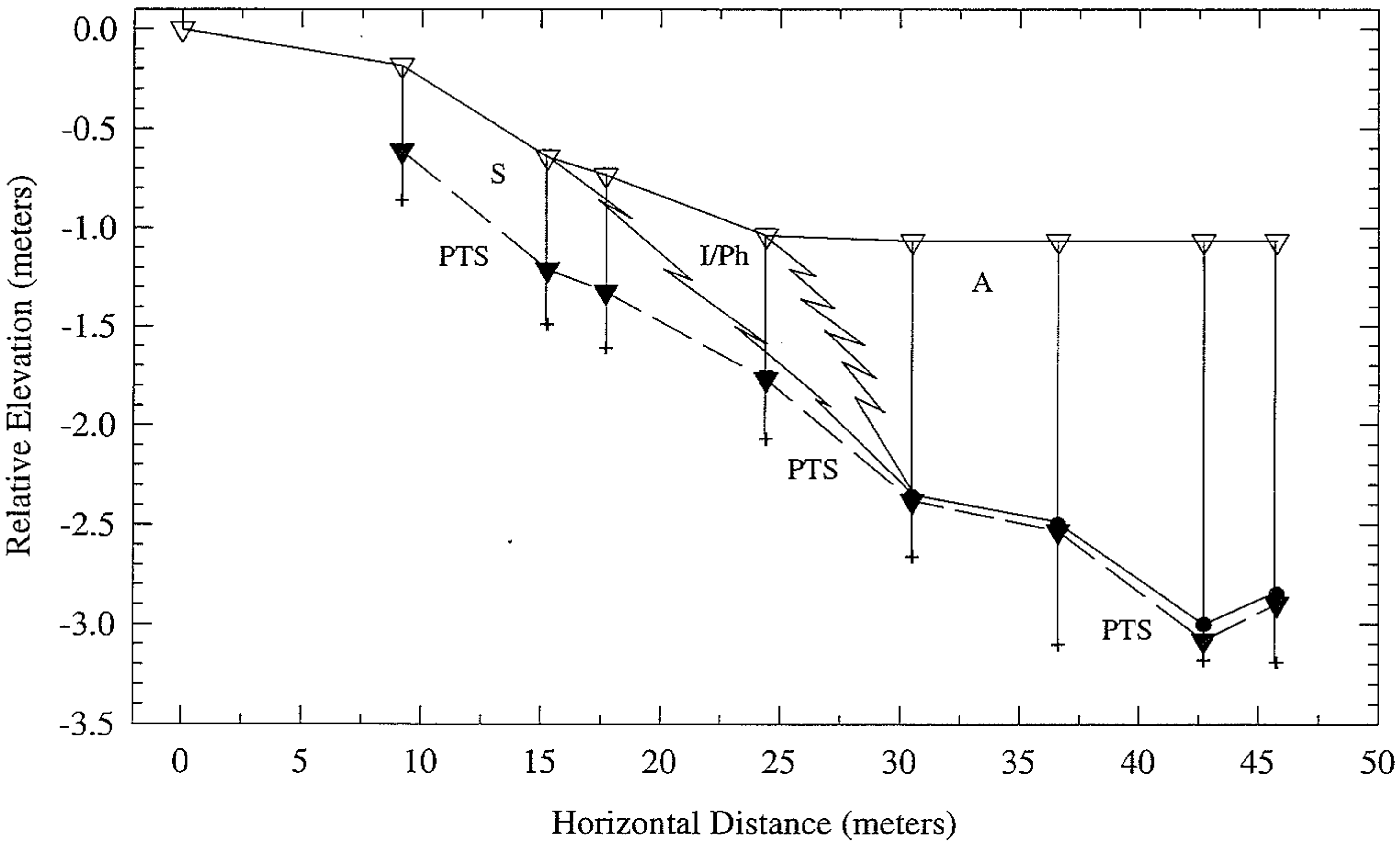
PTS

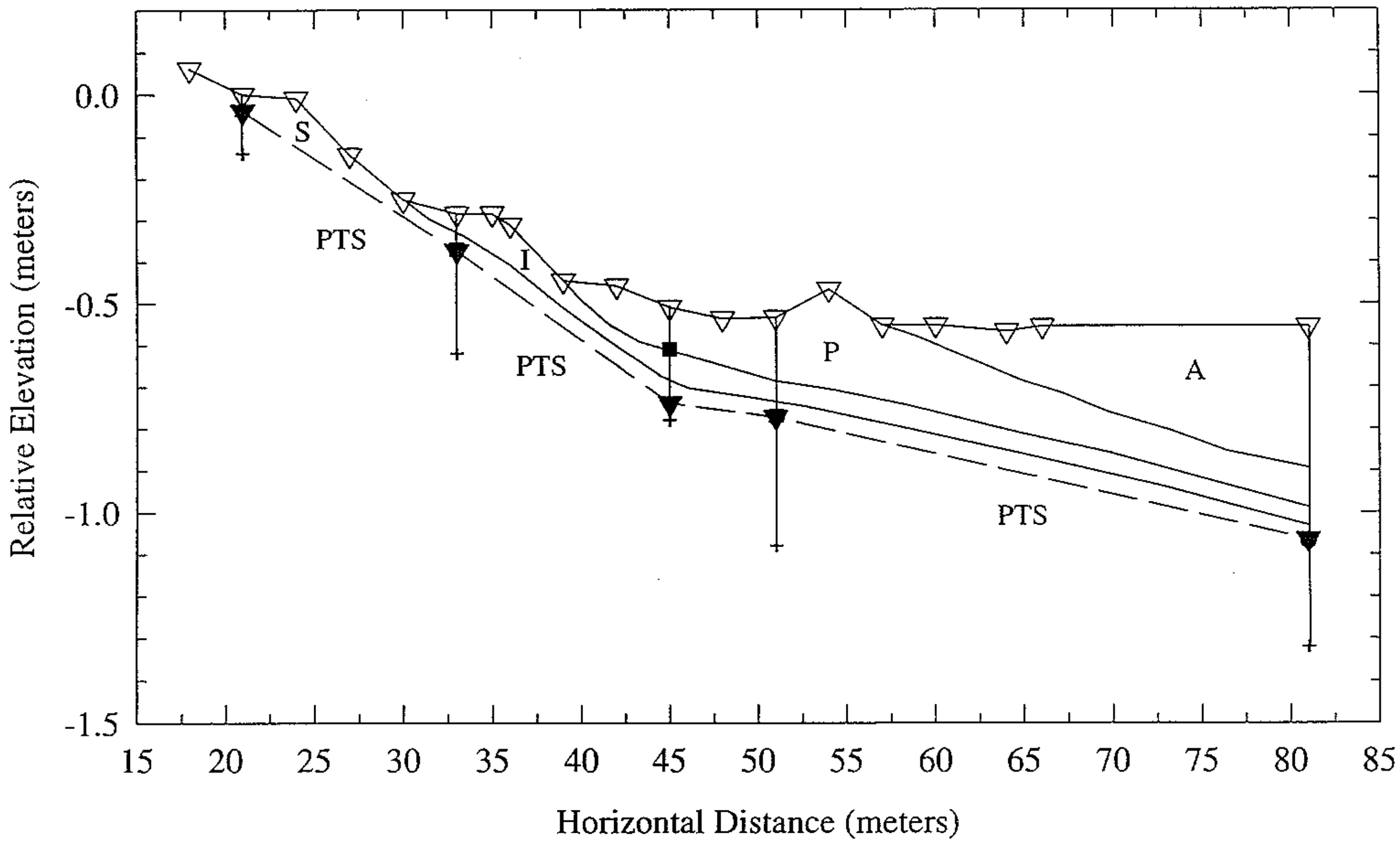
Camp Arrowhead - 2



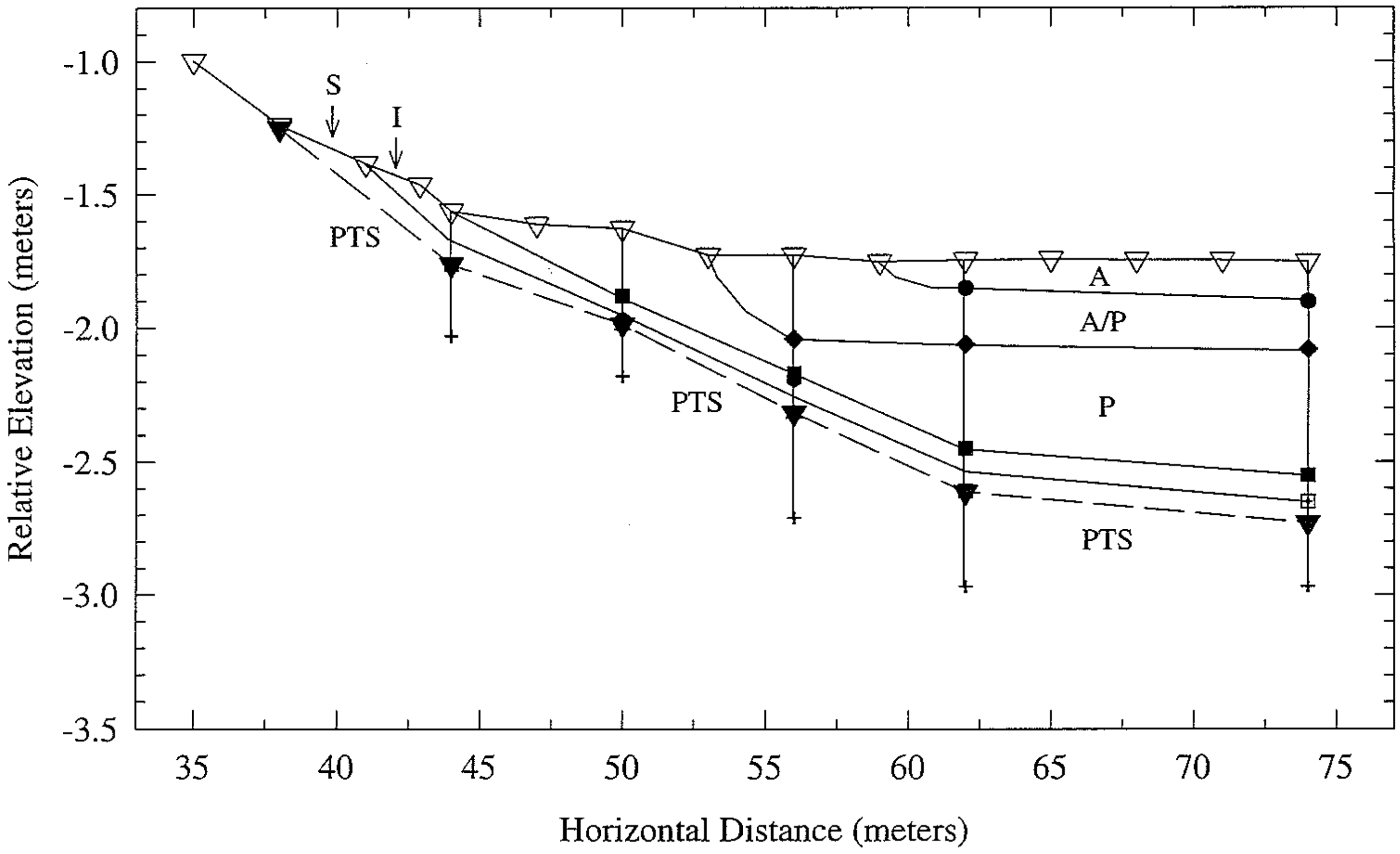
Canary Creek - 1



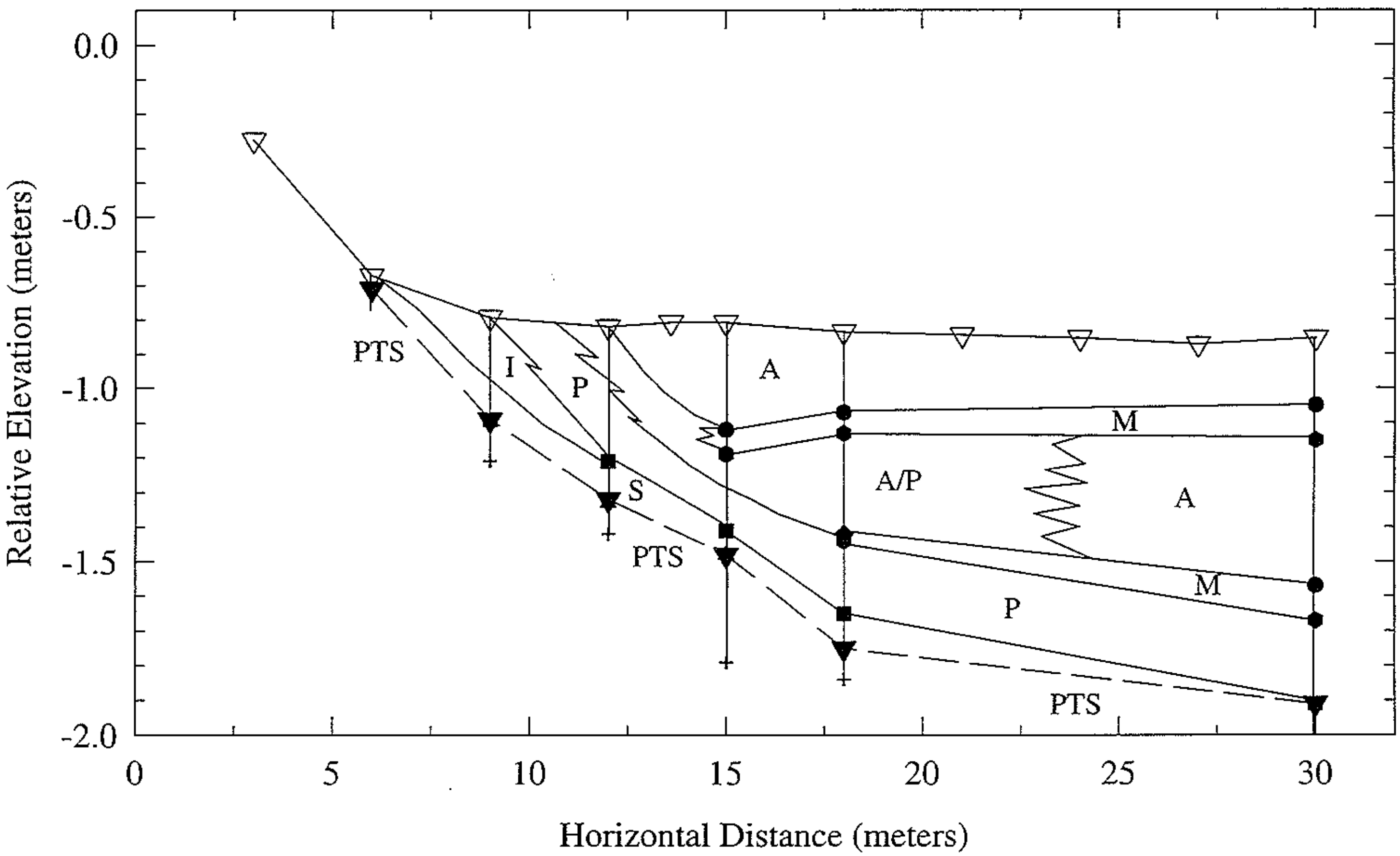
Delaware Wildlands - 1



Delaware Wildlands - 2



Dodd - 1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Horizontal Distance (meters)
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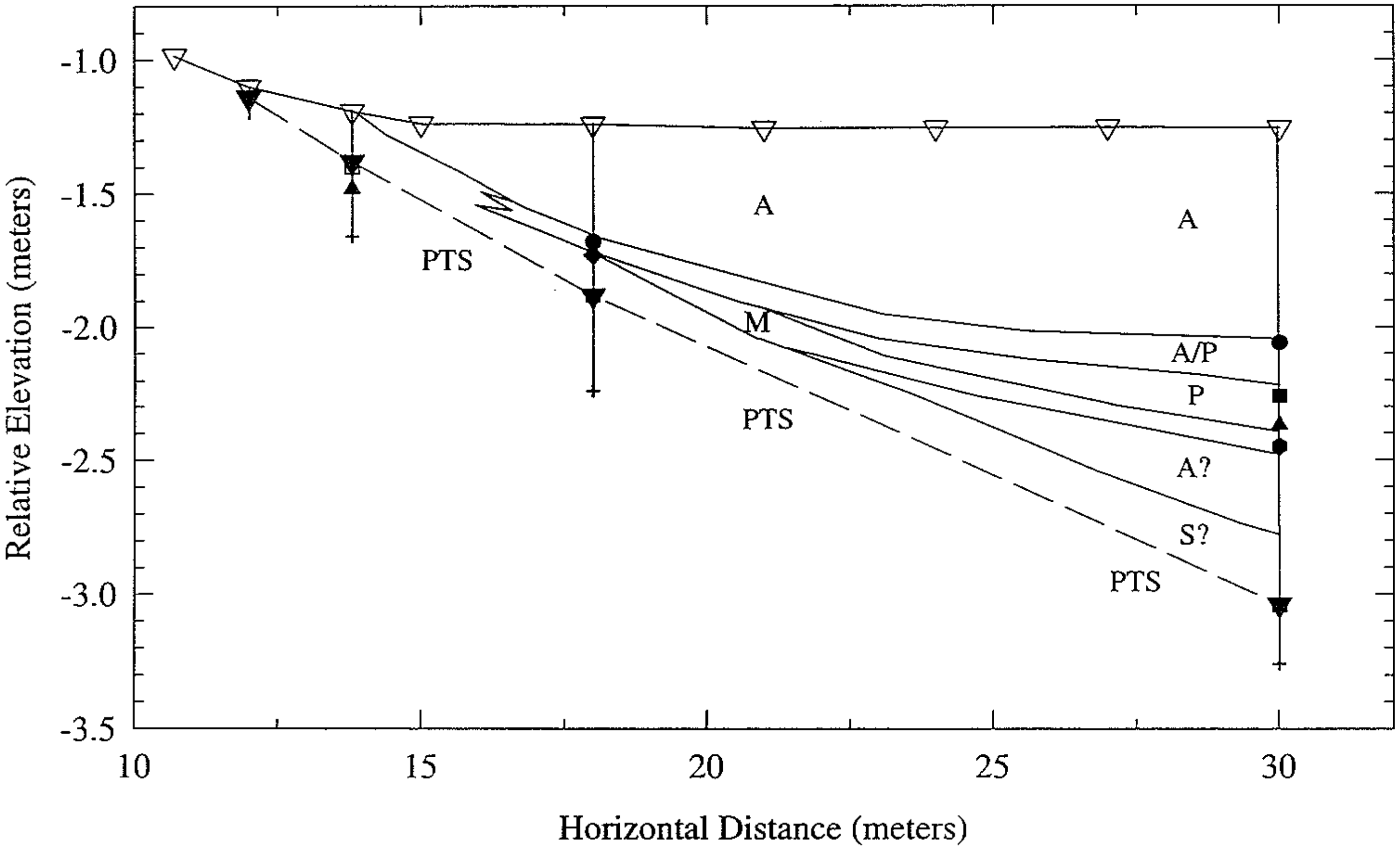
Dodd Marsh - 2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Horizontal Distance (meters)
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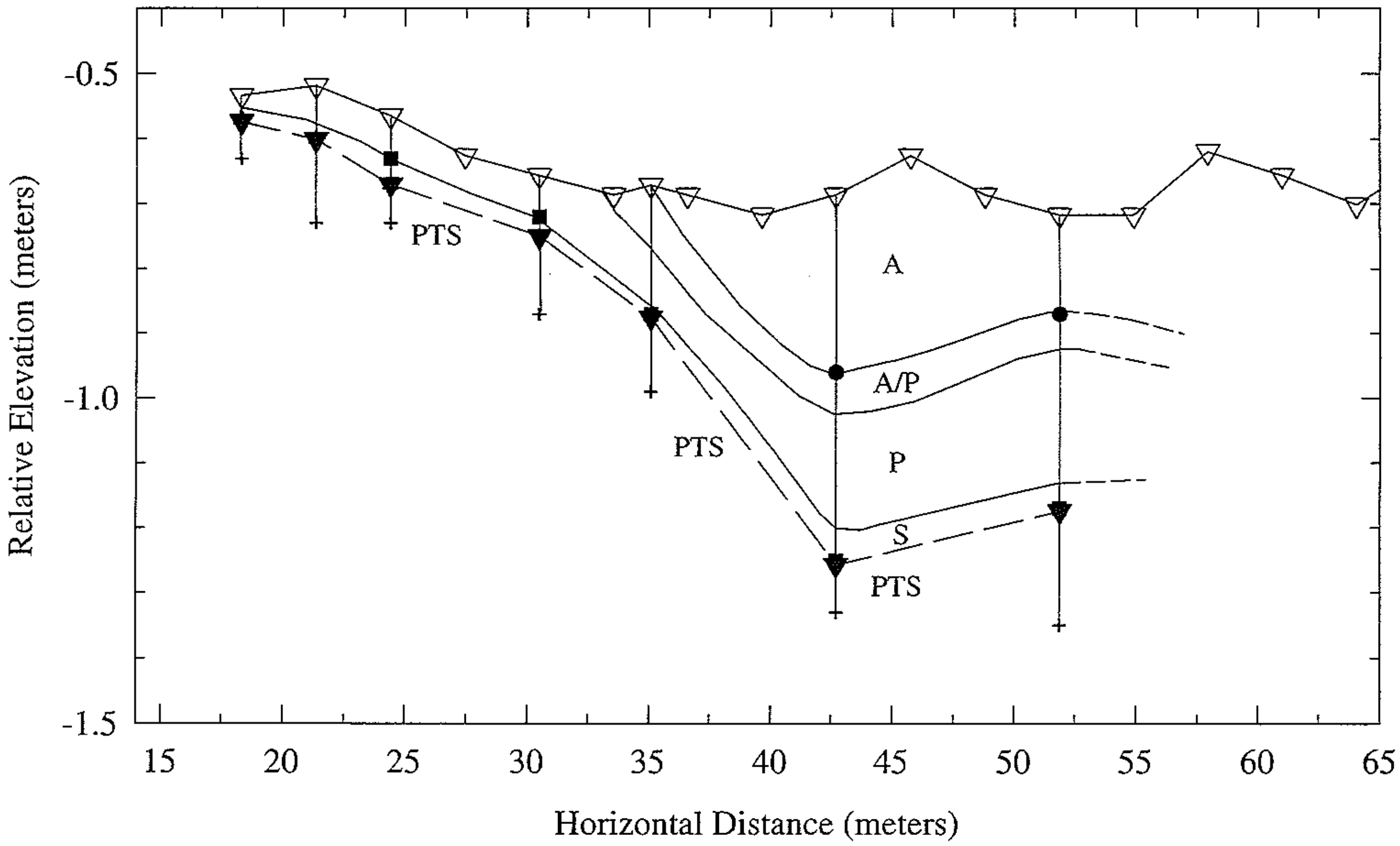
Draper - 1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Horizontal Distance (meters)
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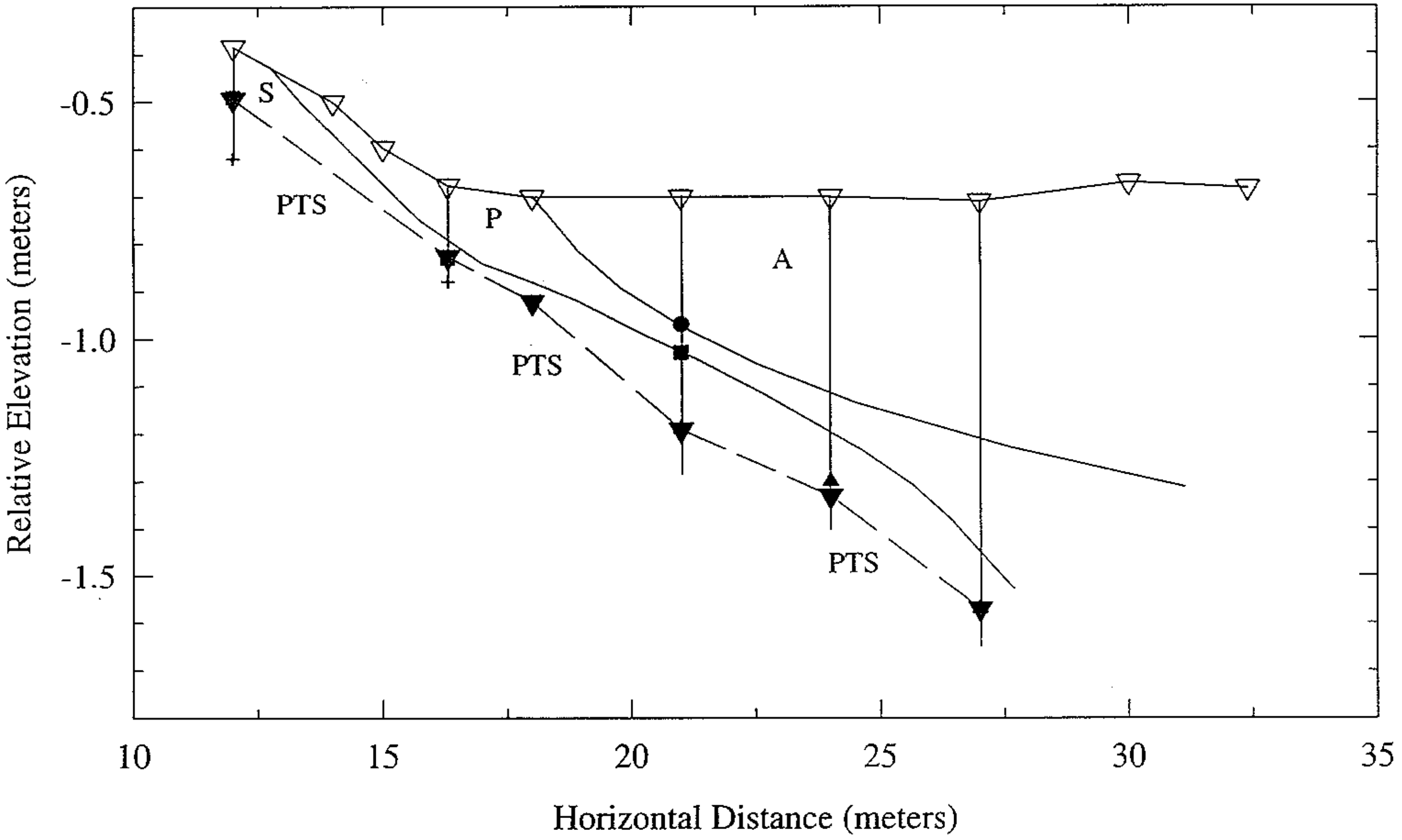
Draper - 2



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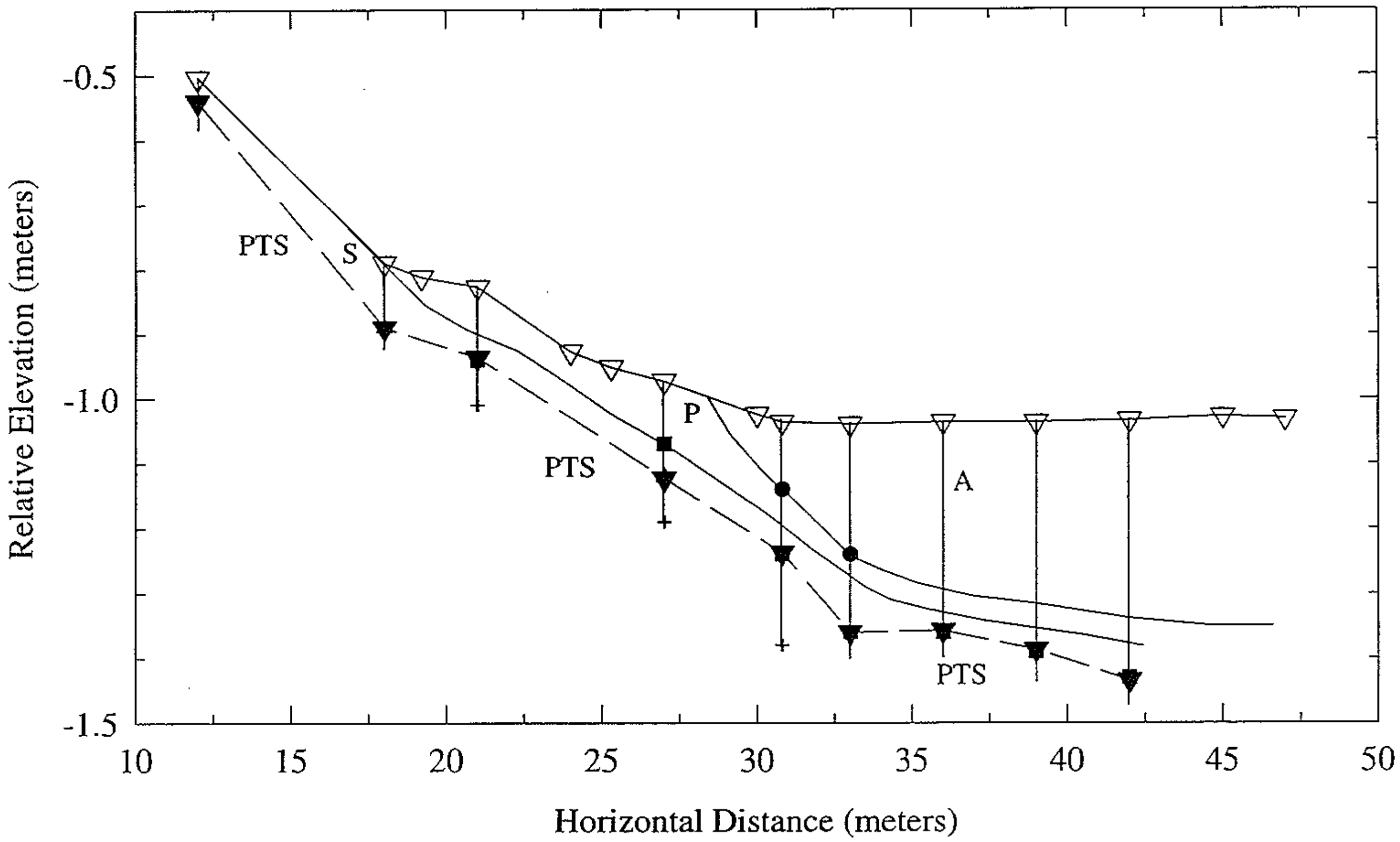
Draper - 3



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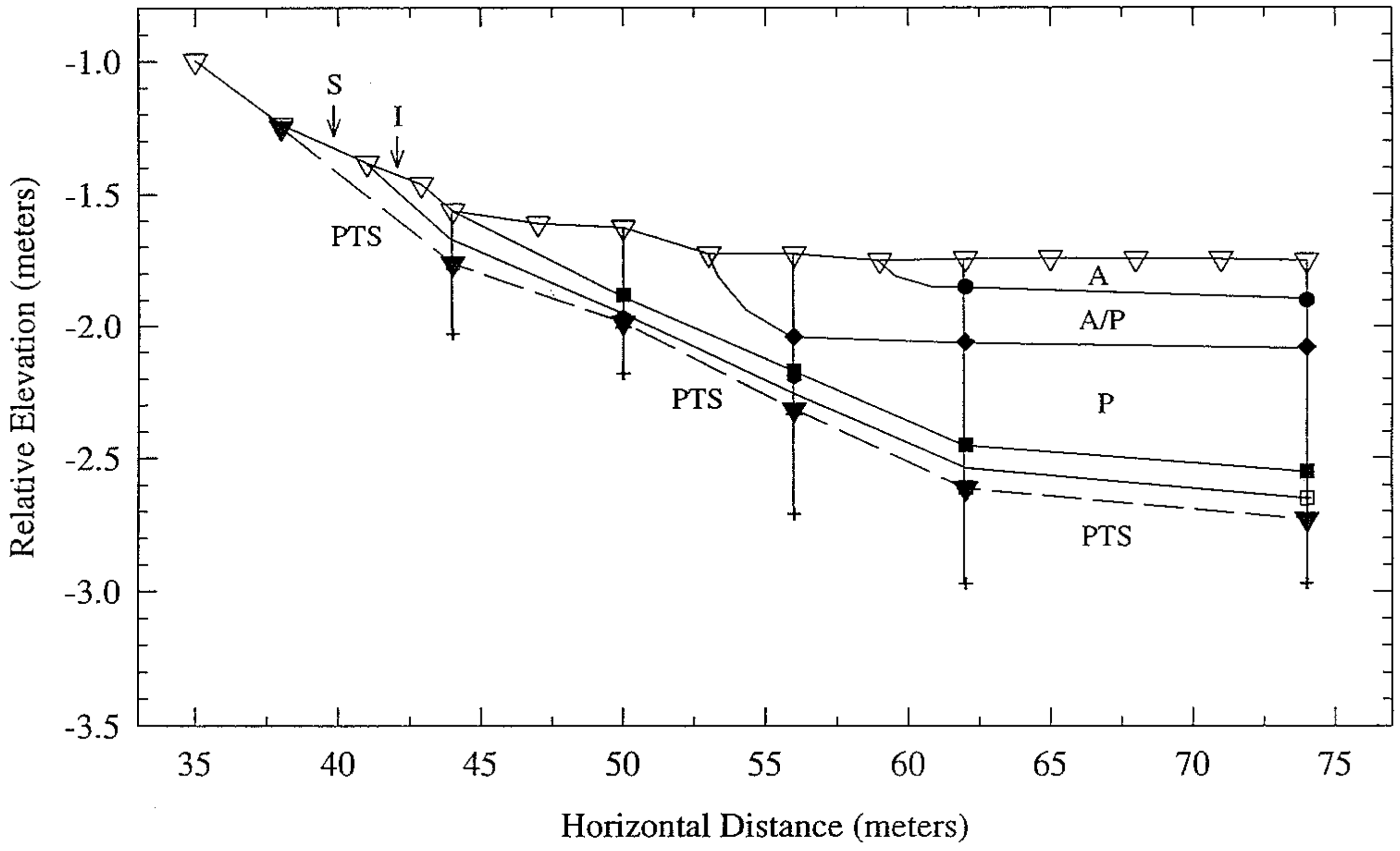
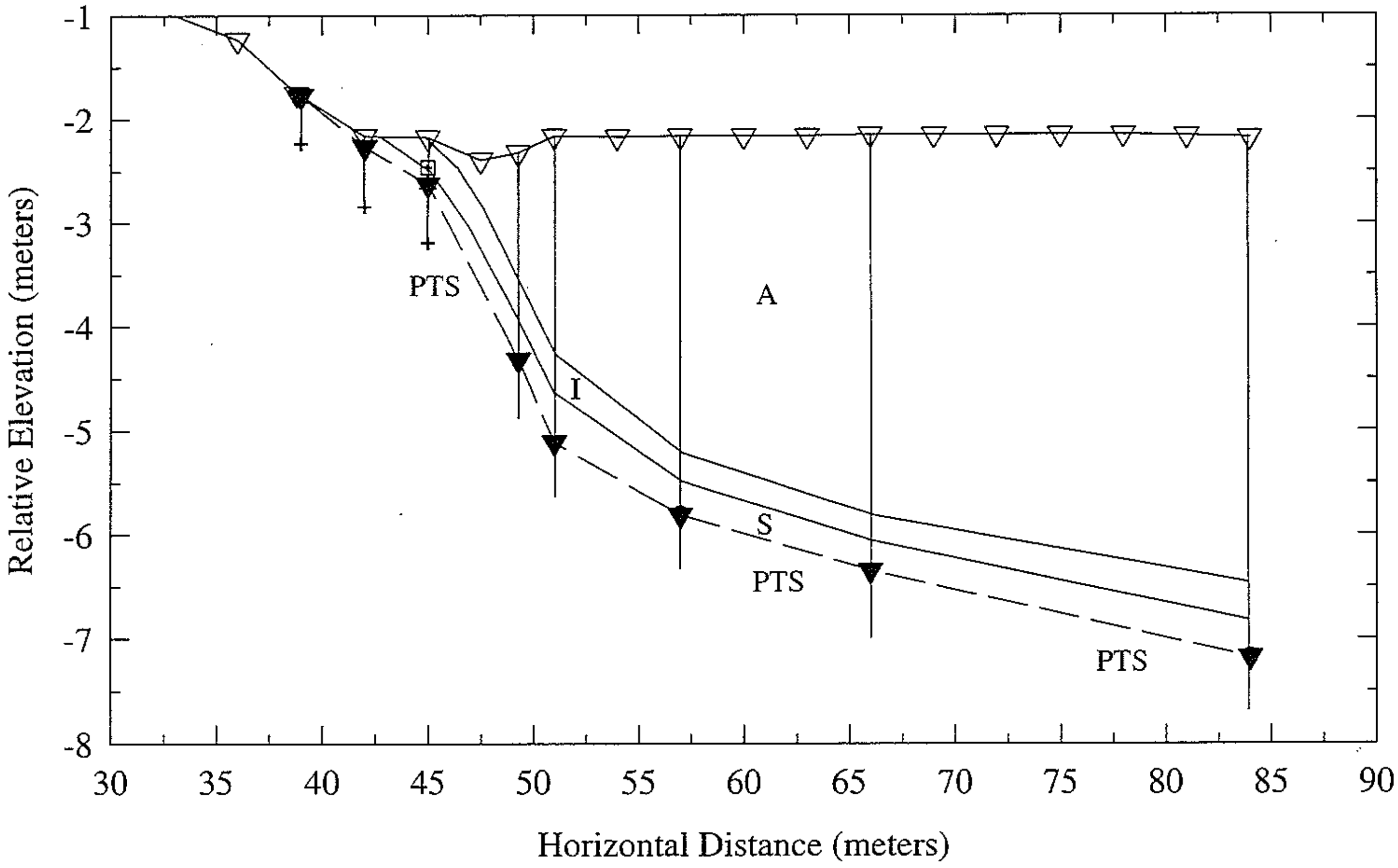
Delaware Wildlands - 2



Glade - 1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Relative Elevation (meters)
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Horizontal Distance (meters)
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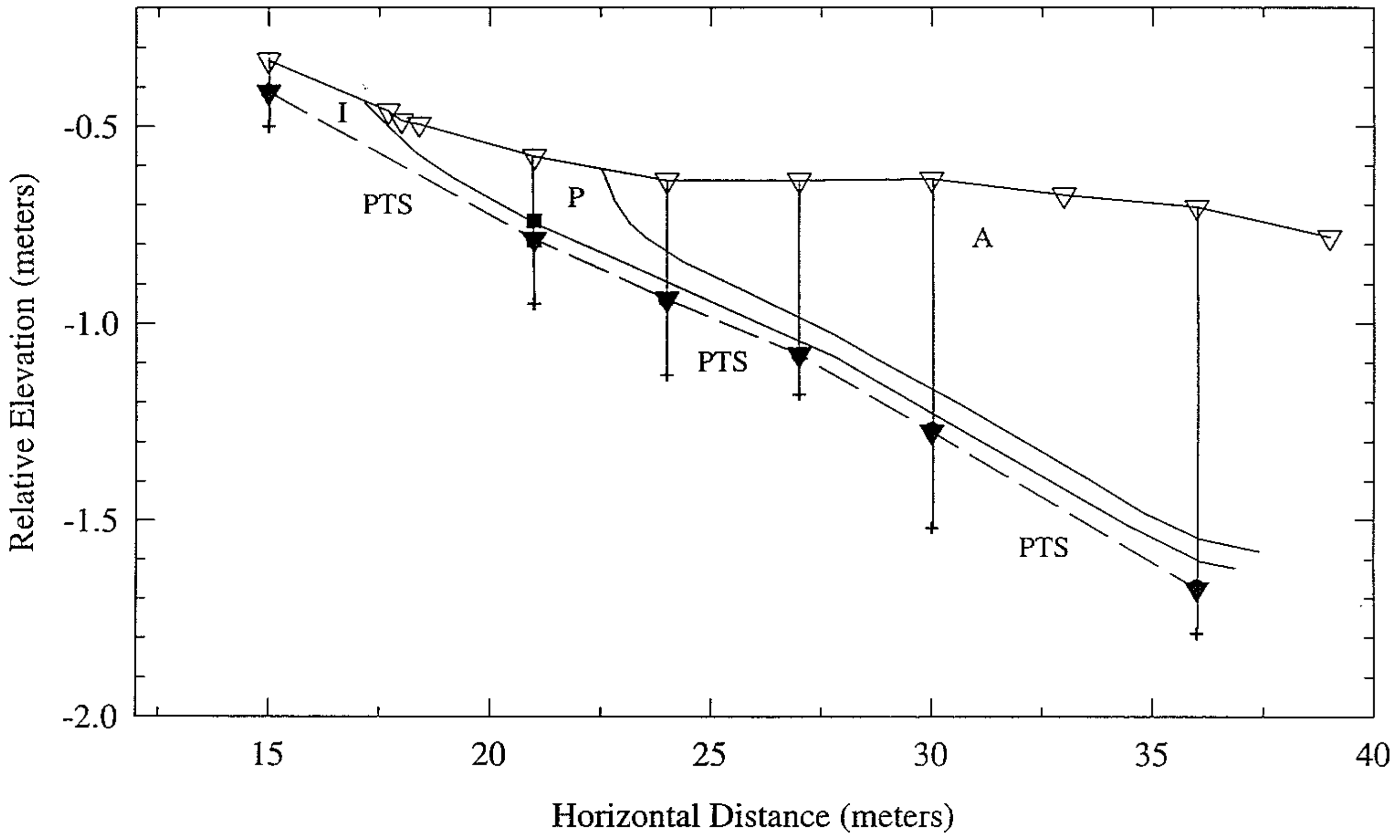
Glade - 2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Relative Elevation (meters)
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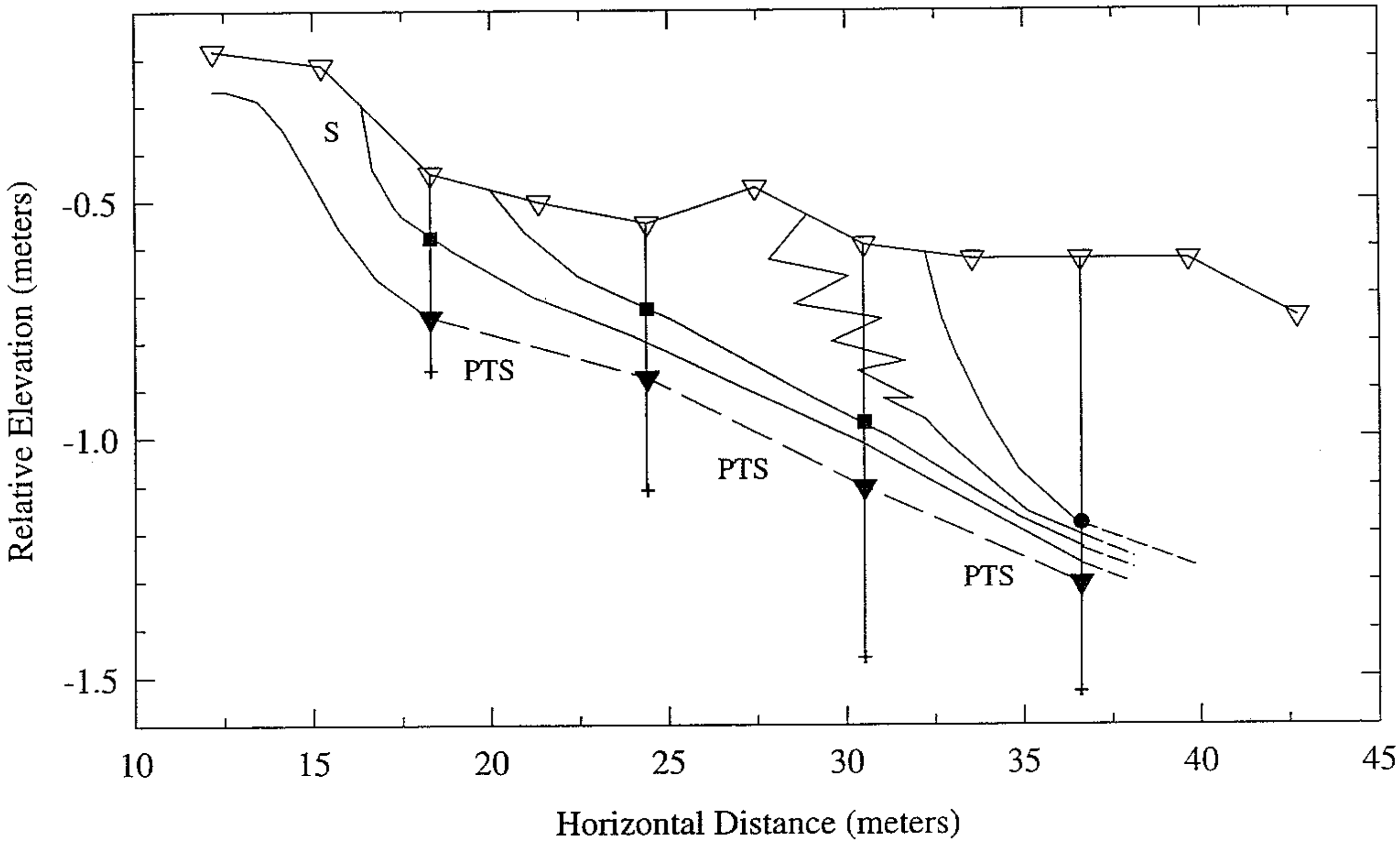
Horizontal Distance (meters)

Hercules - 2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Horizontal Distance (meters)
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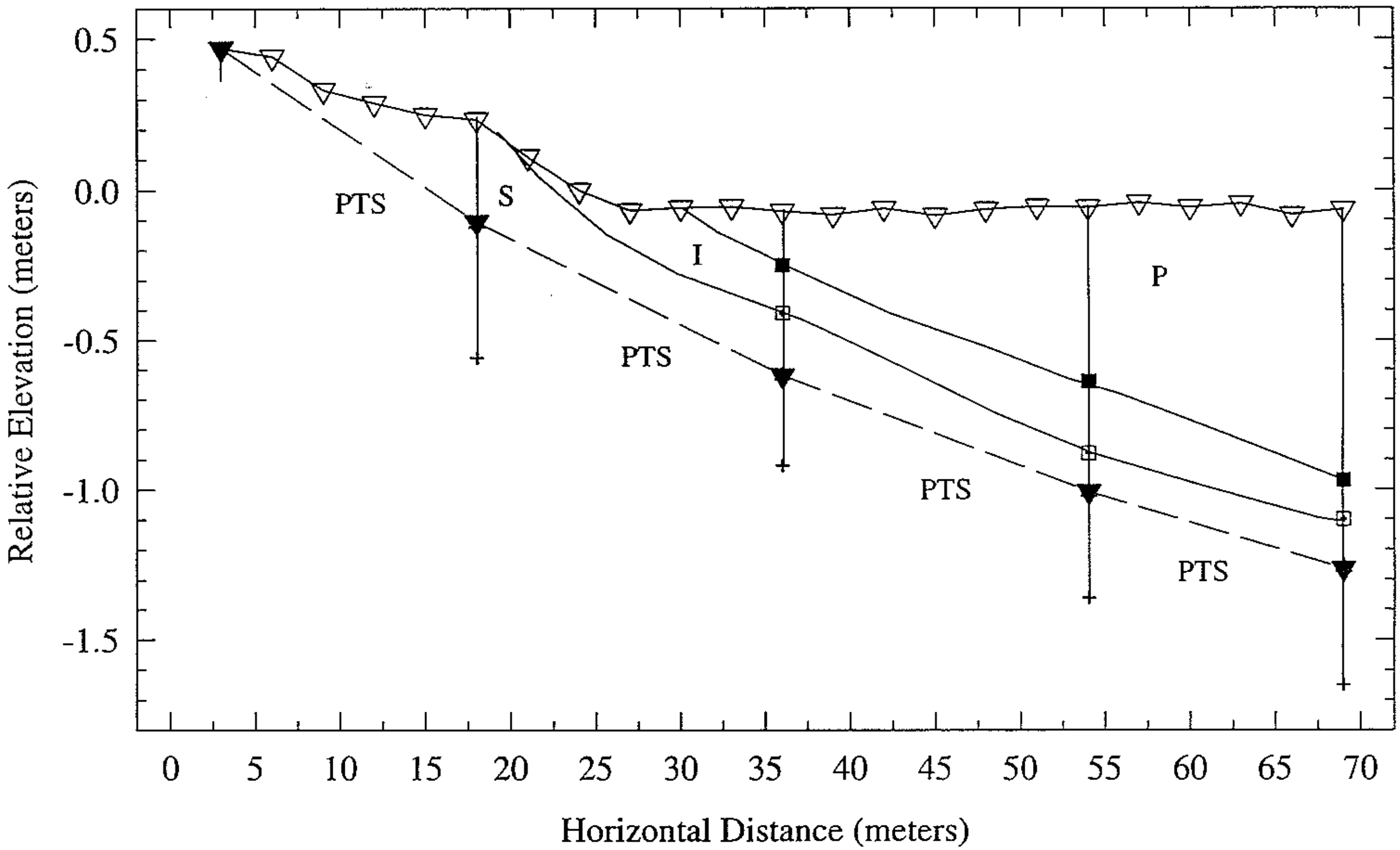
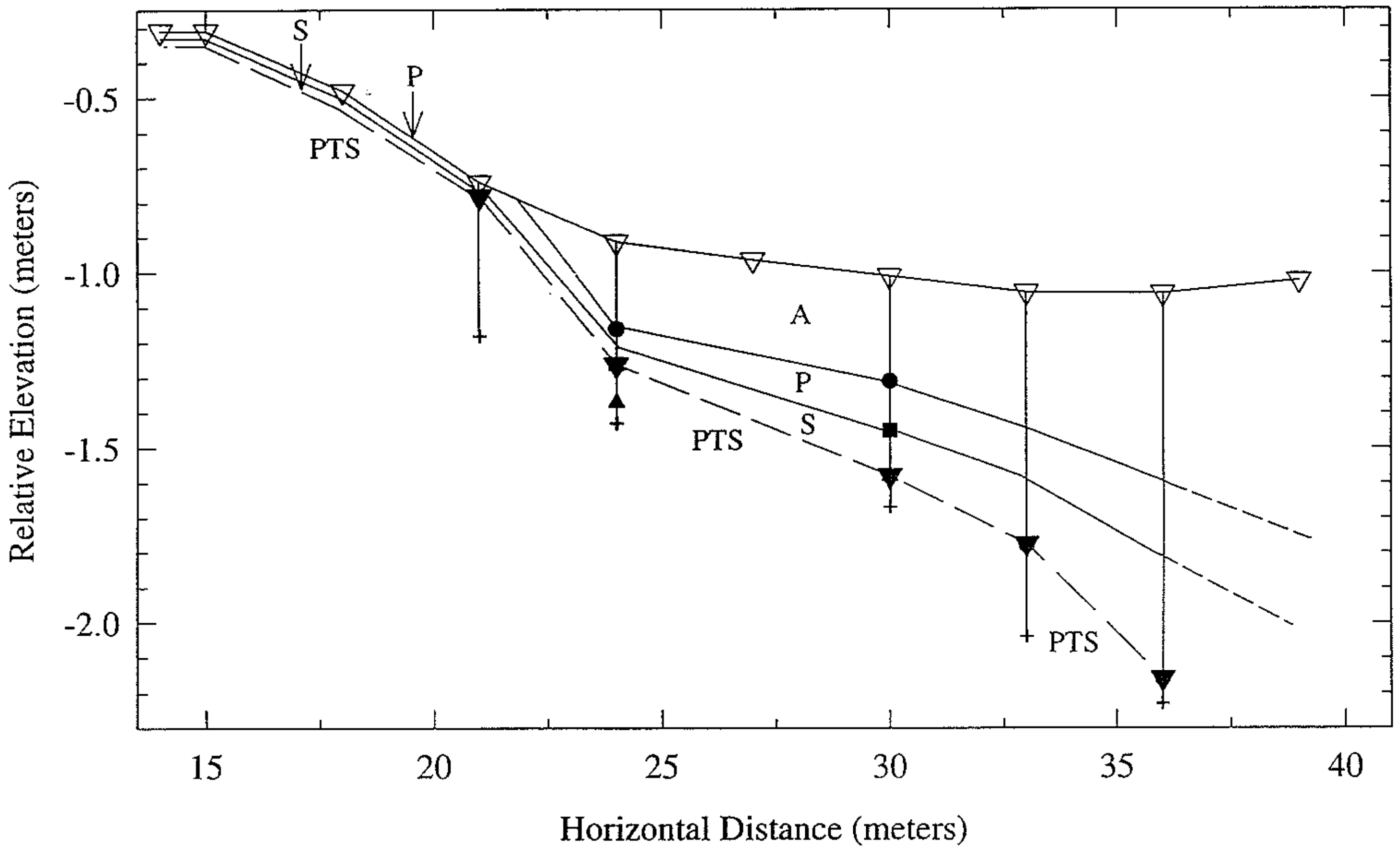
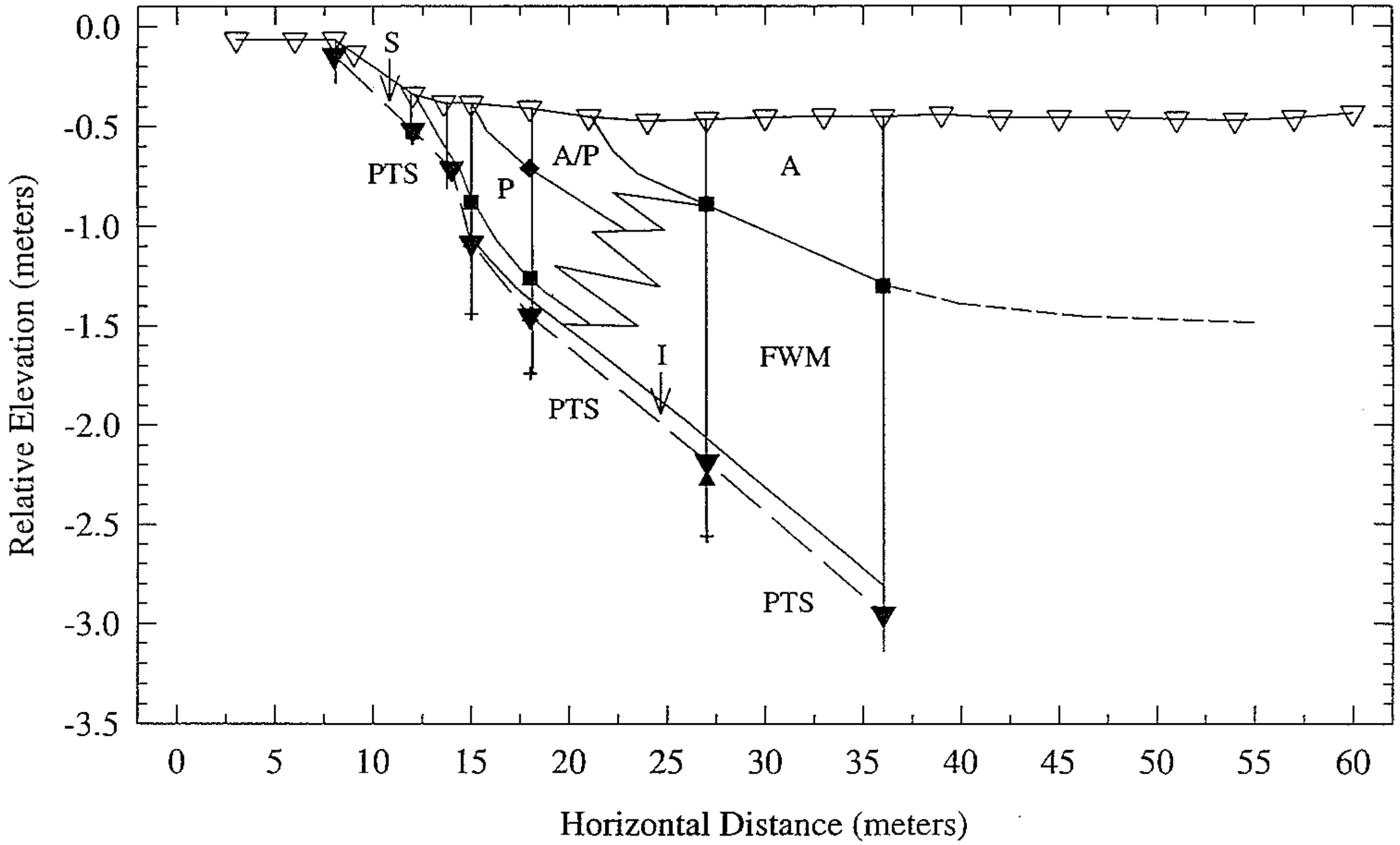
Horse Island



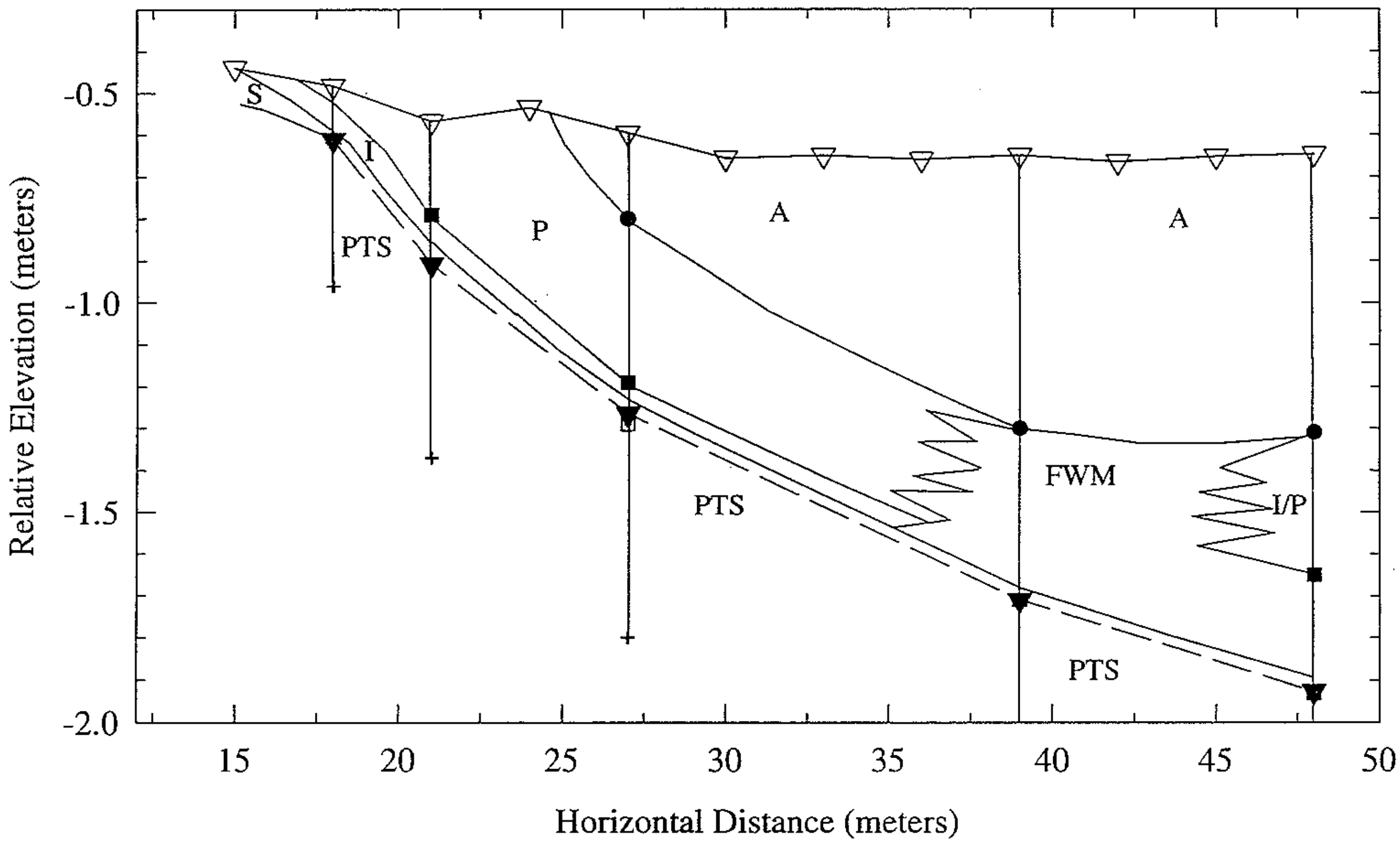
Indian Landing - 1



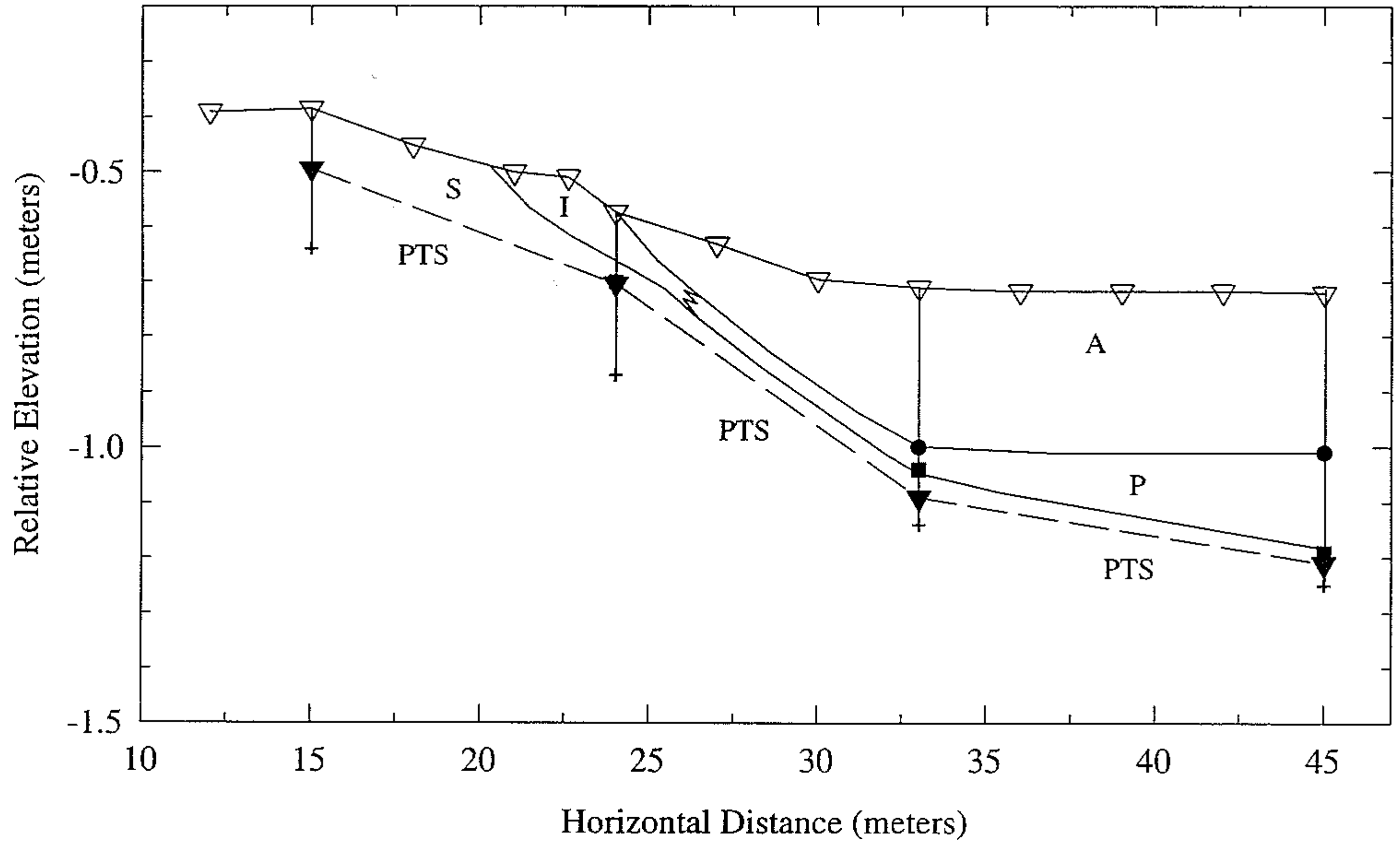
Island Field - 1



Island Field - 2



Lingo Point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Relative Elevation (meters)
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Horizontal Distance (meters)
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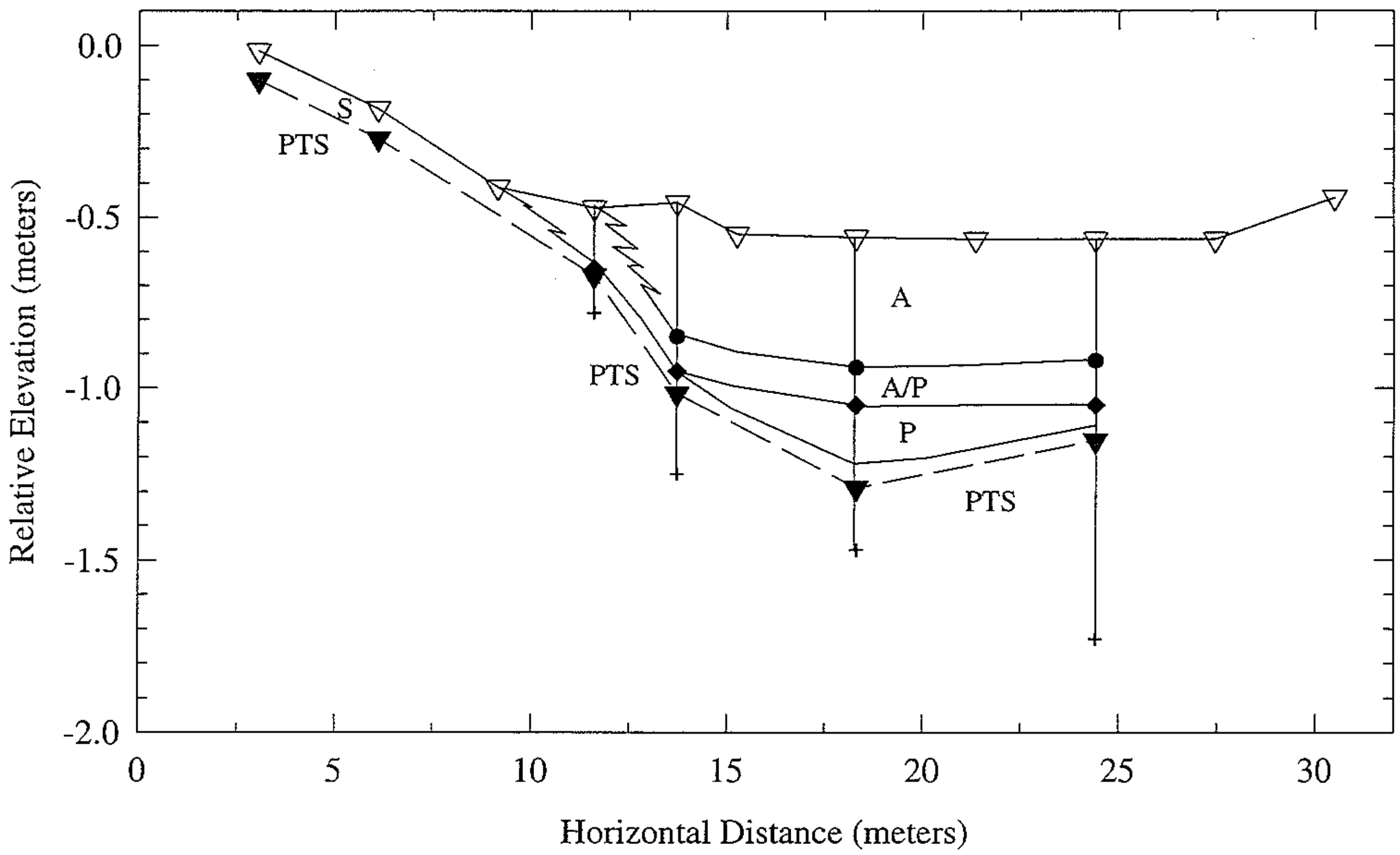
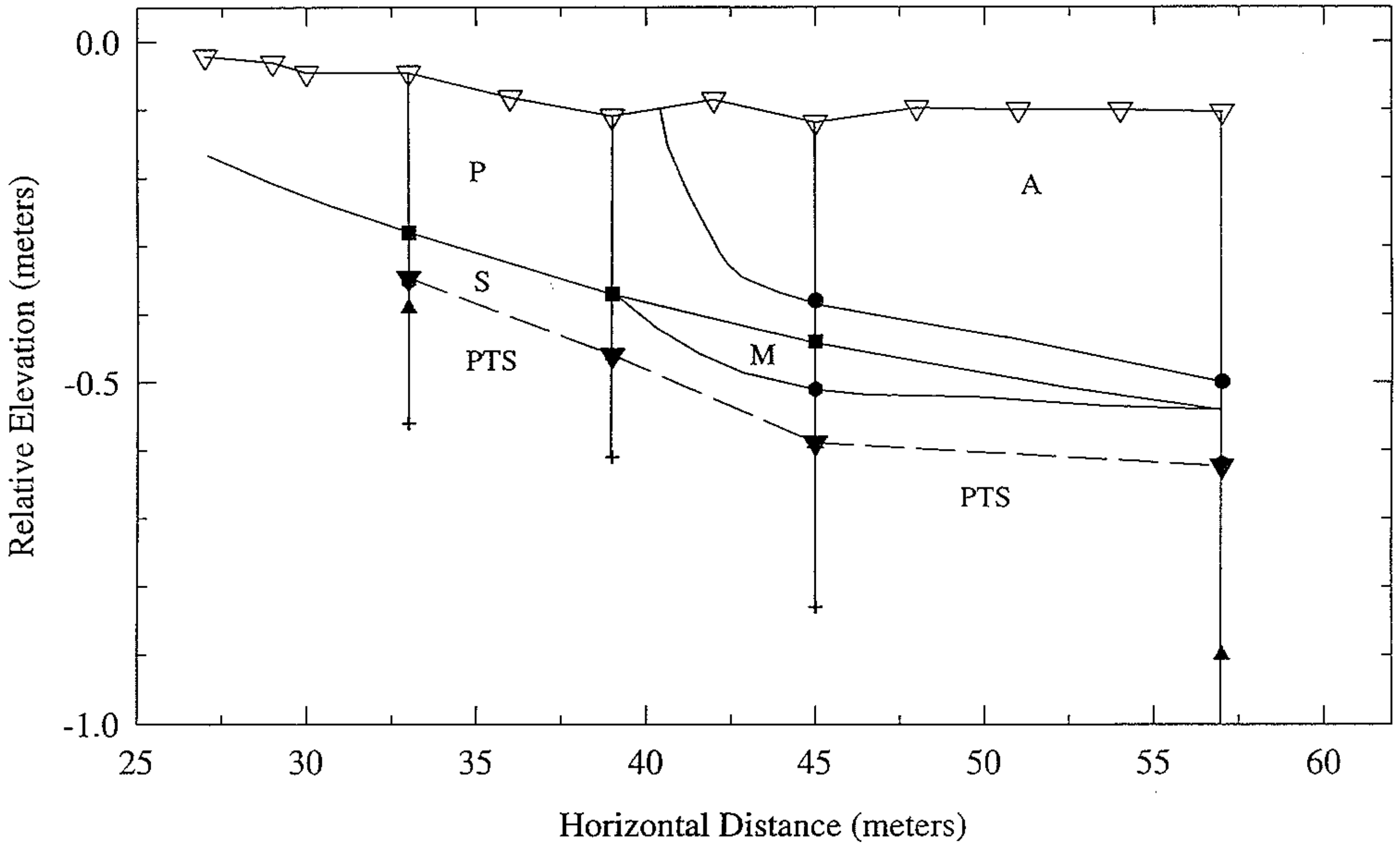
Little Assawoman Bay - 1



McKim - 1



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Relative Elevation (meters)
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Horizontal Distance (meters)
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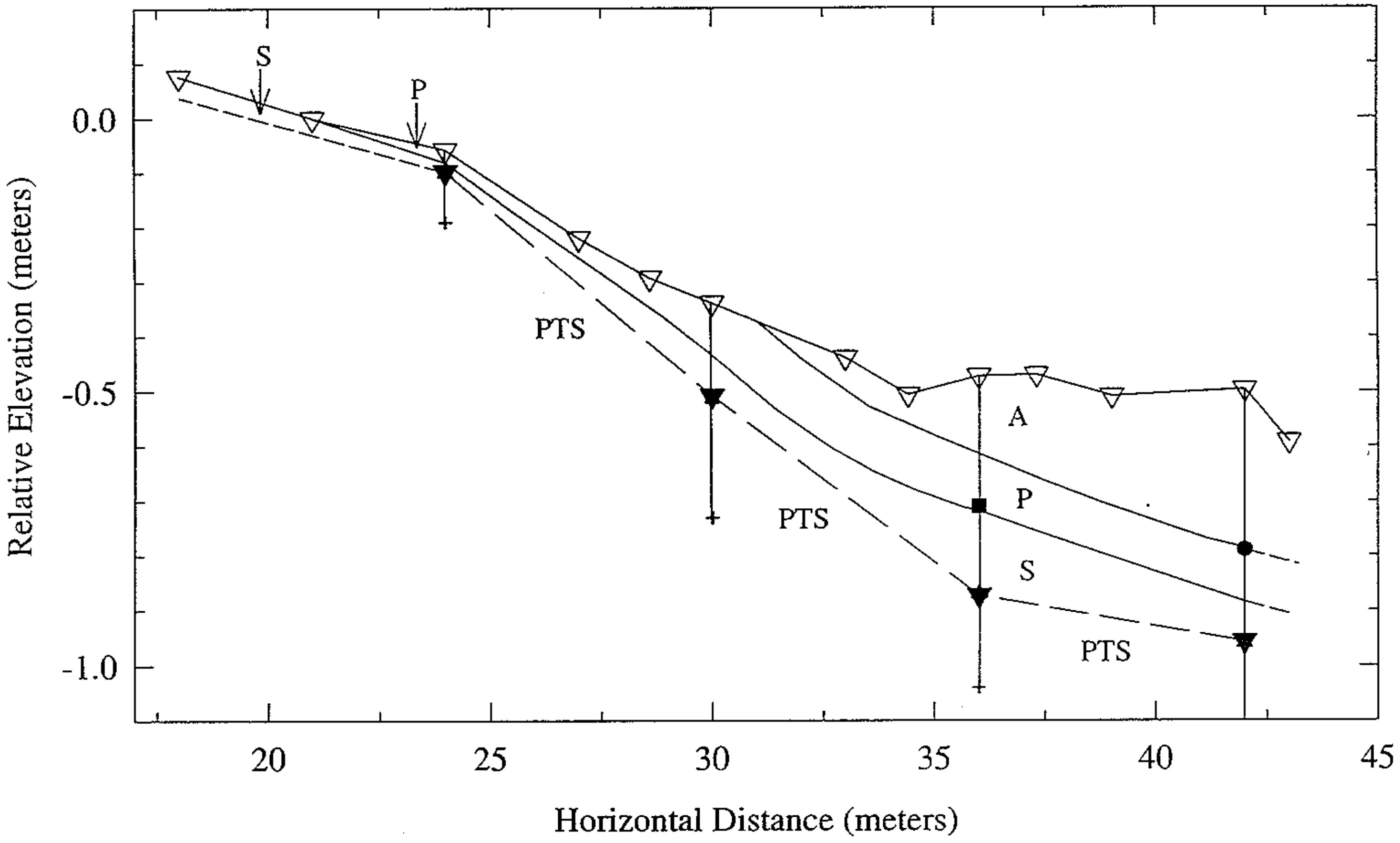
McKim - 2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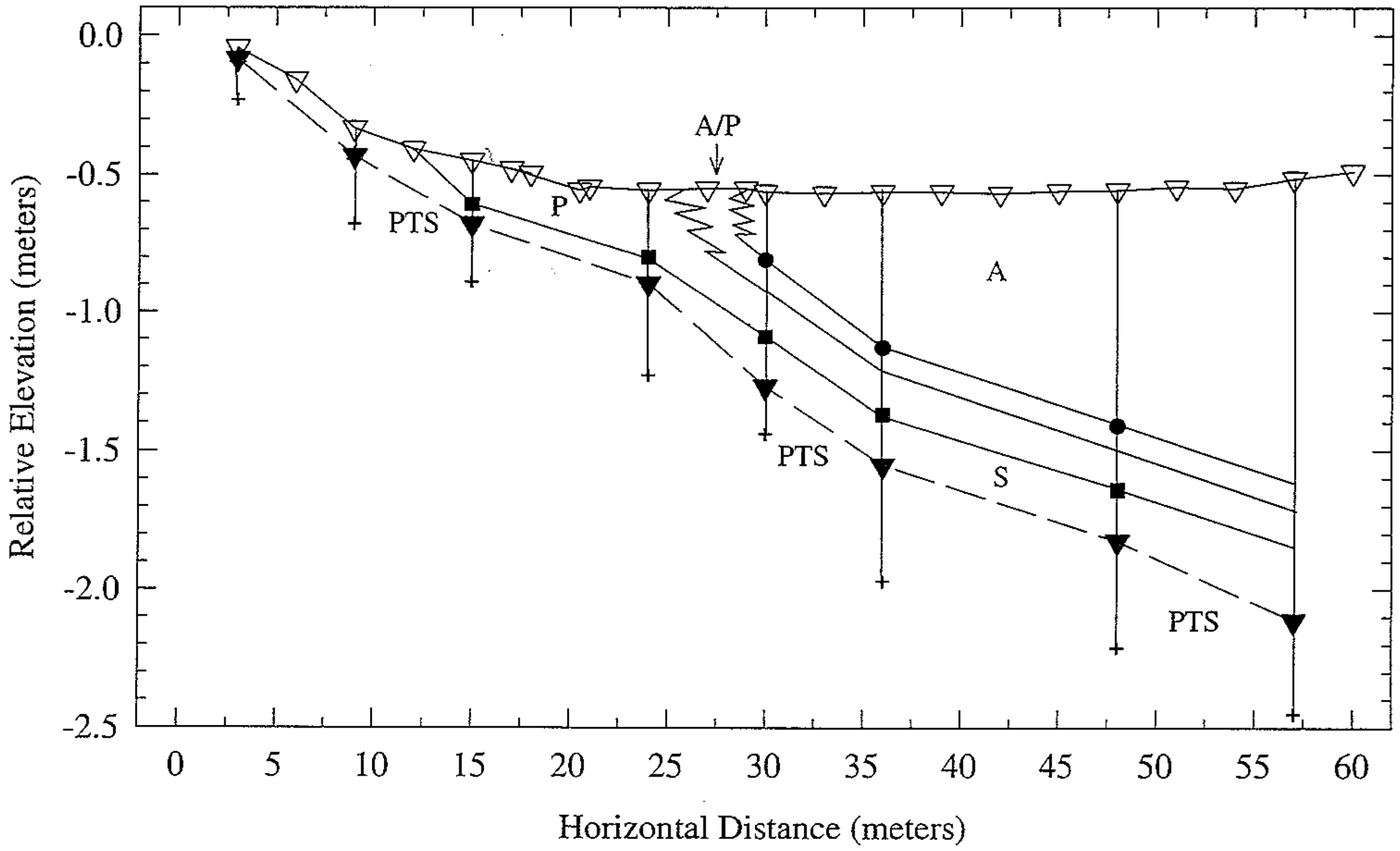
Relative Elevation (meters)
Horizontal Distance (meters)

Mills - 1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Horizontal Distance (meters)
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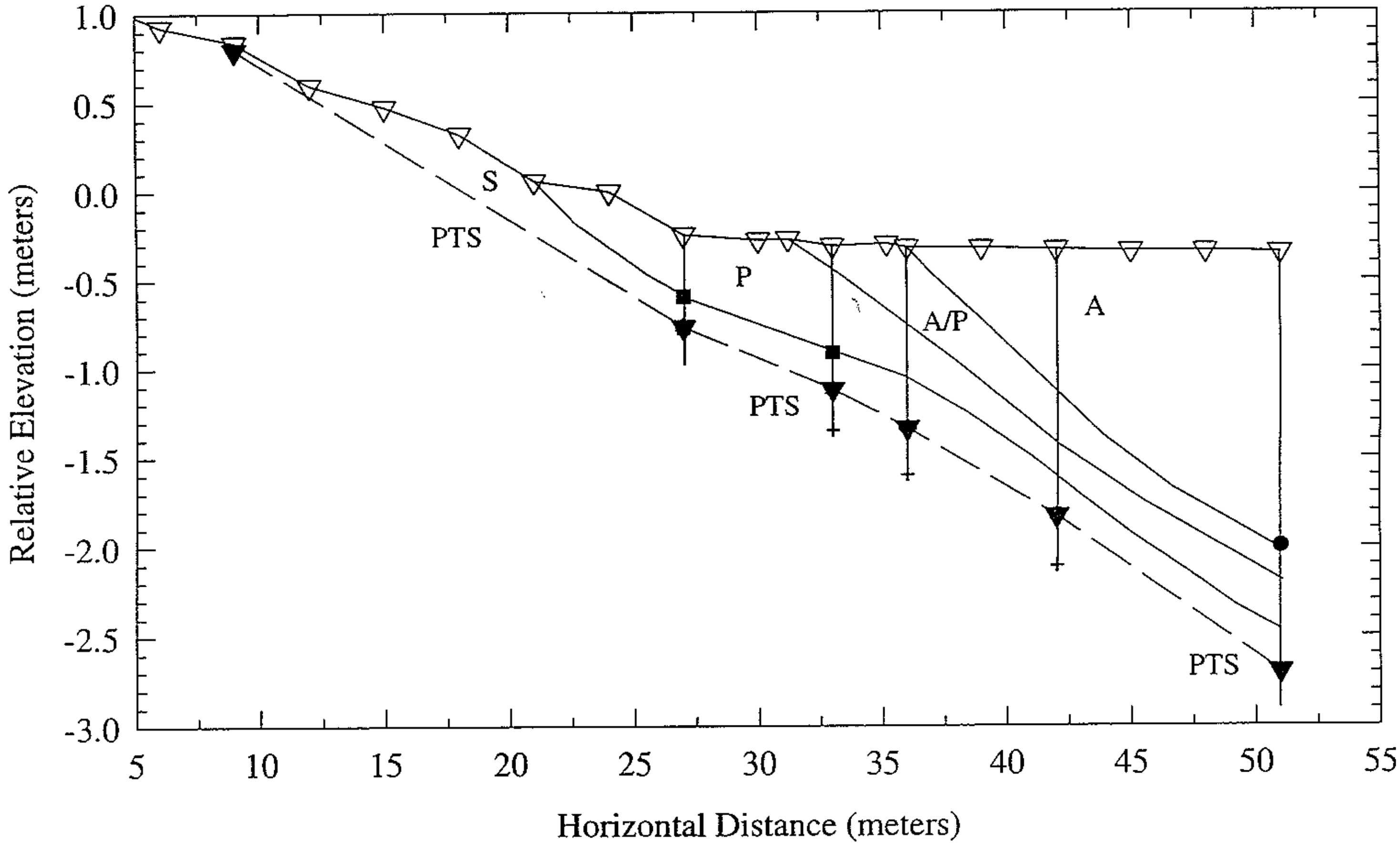
Muddy Neck - 1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Horizontal Distance (meters)
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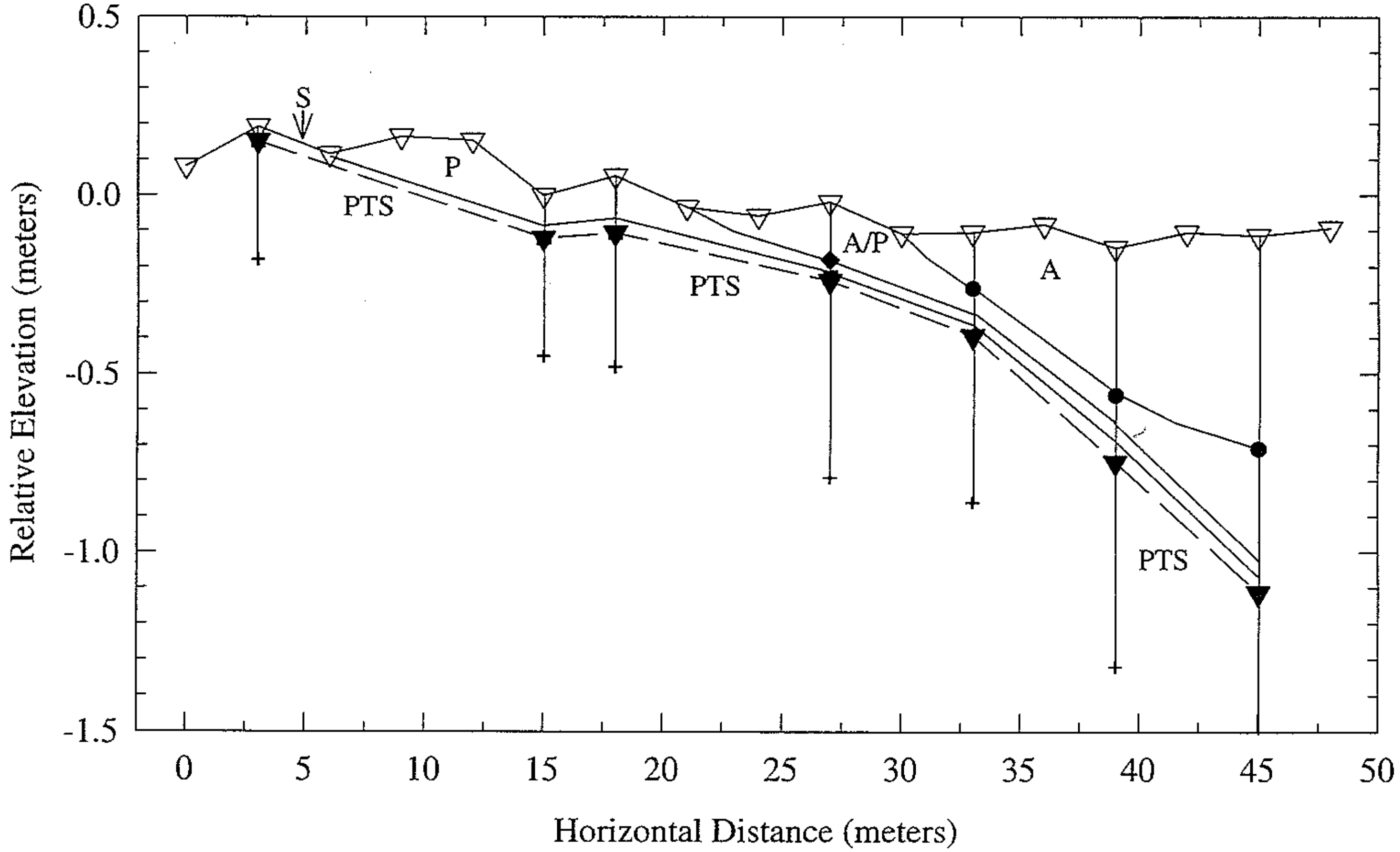
A/P
A
PTS
PTS

Muddy Neck - 2


S
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-1.5
Relative Elevation (meters)
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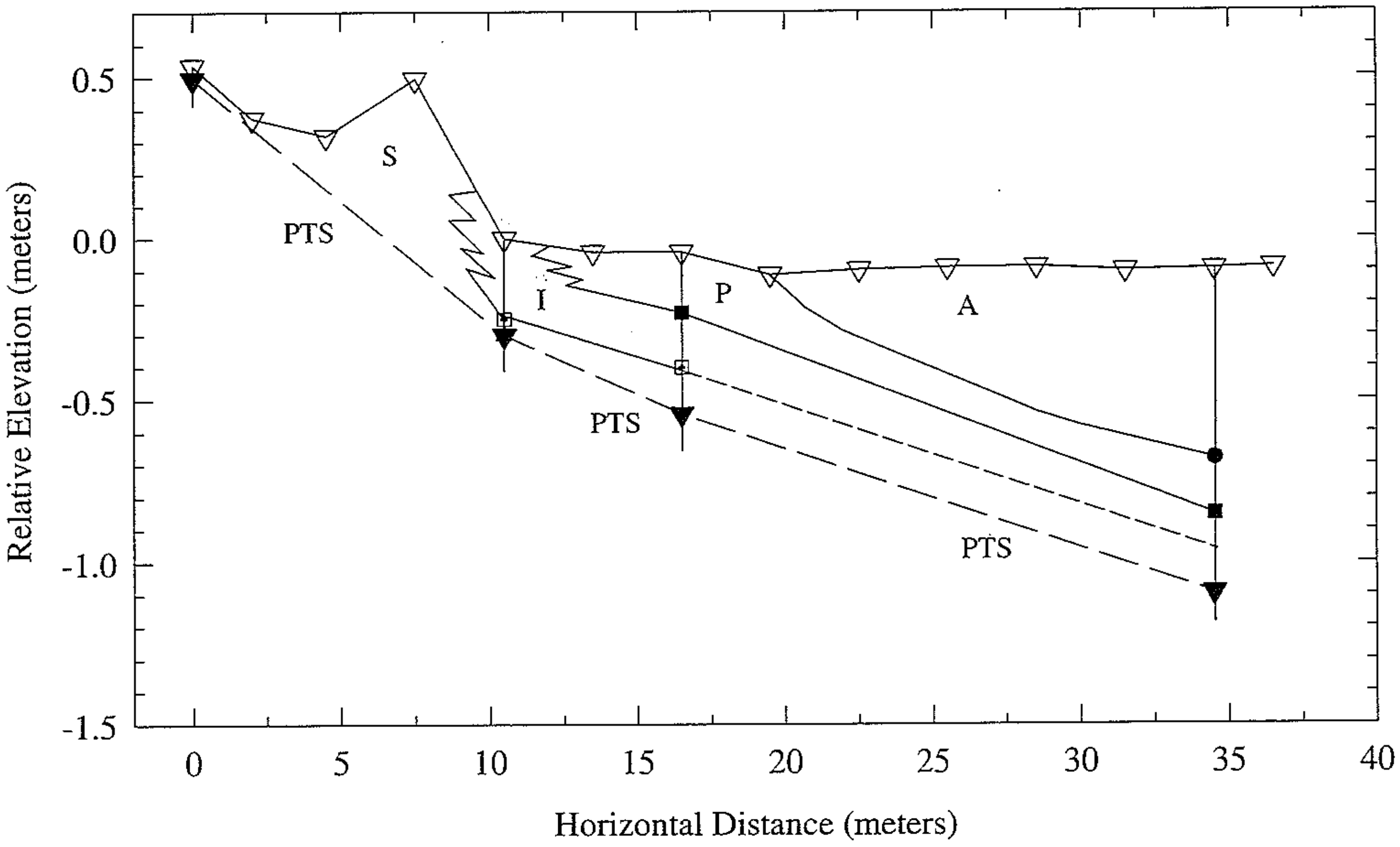
Horizontal Distance (meters)

Muddy Neck - 3



0.5
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-1.5
Relative Elevation (meters)
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Horizontal Distance (meters)
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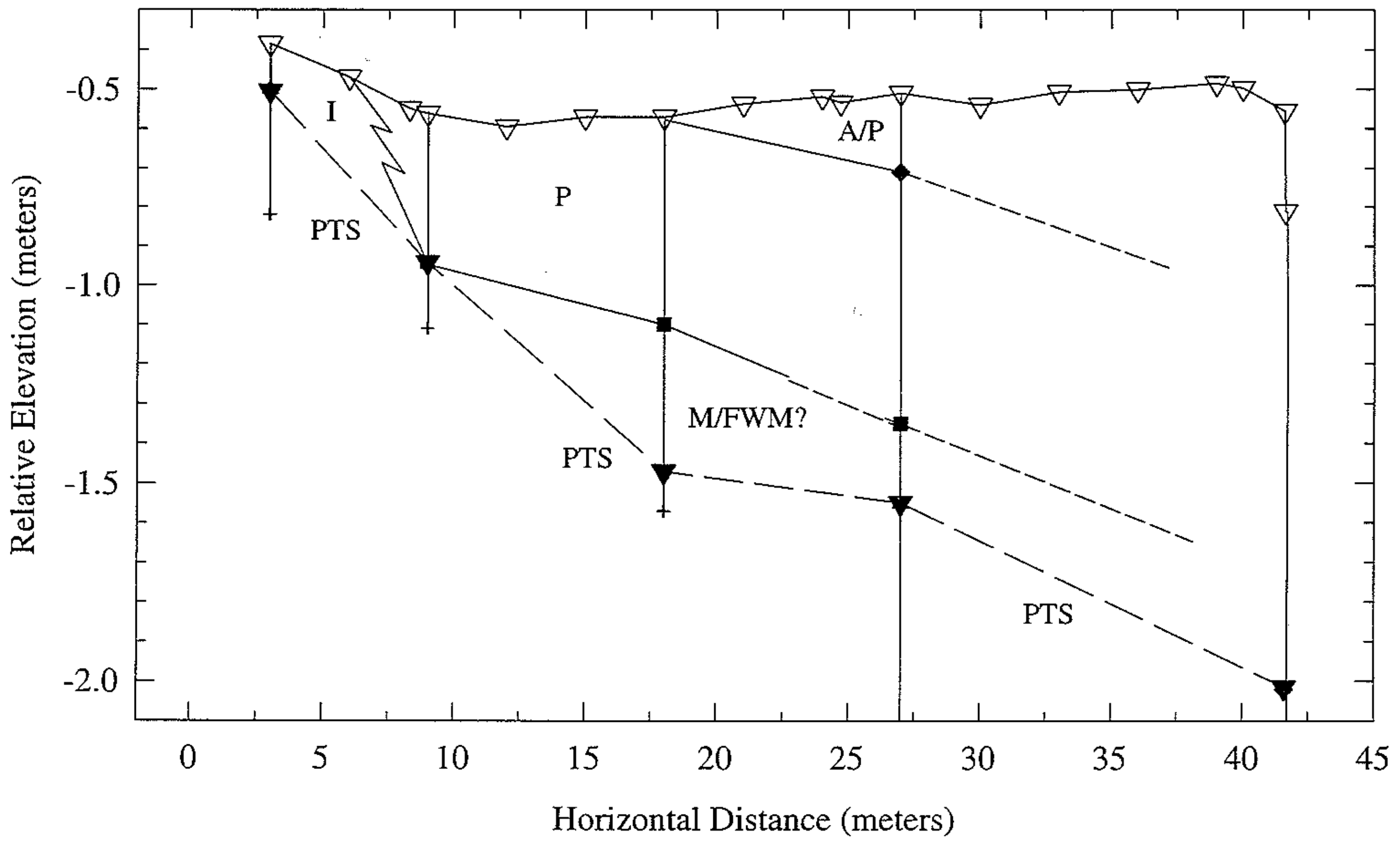
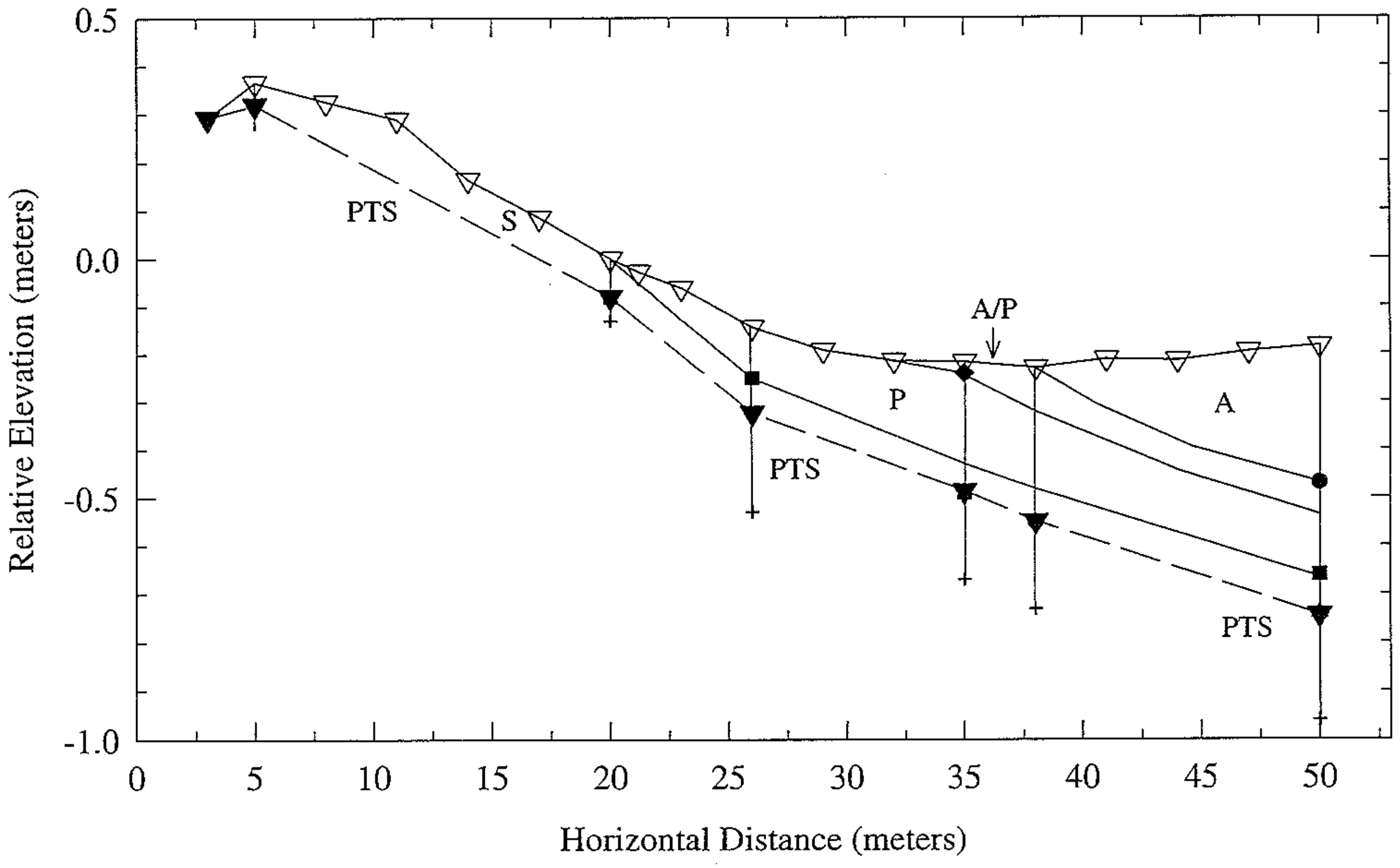
Mulberry Landing



Murray - 1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Relative Elevation (meters)
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Horizontal Distance (meters)
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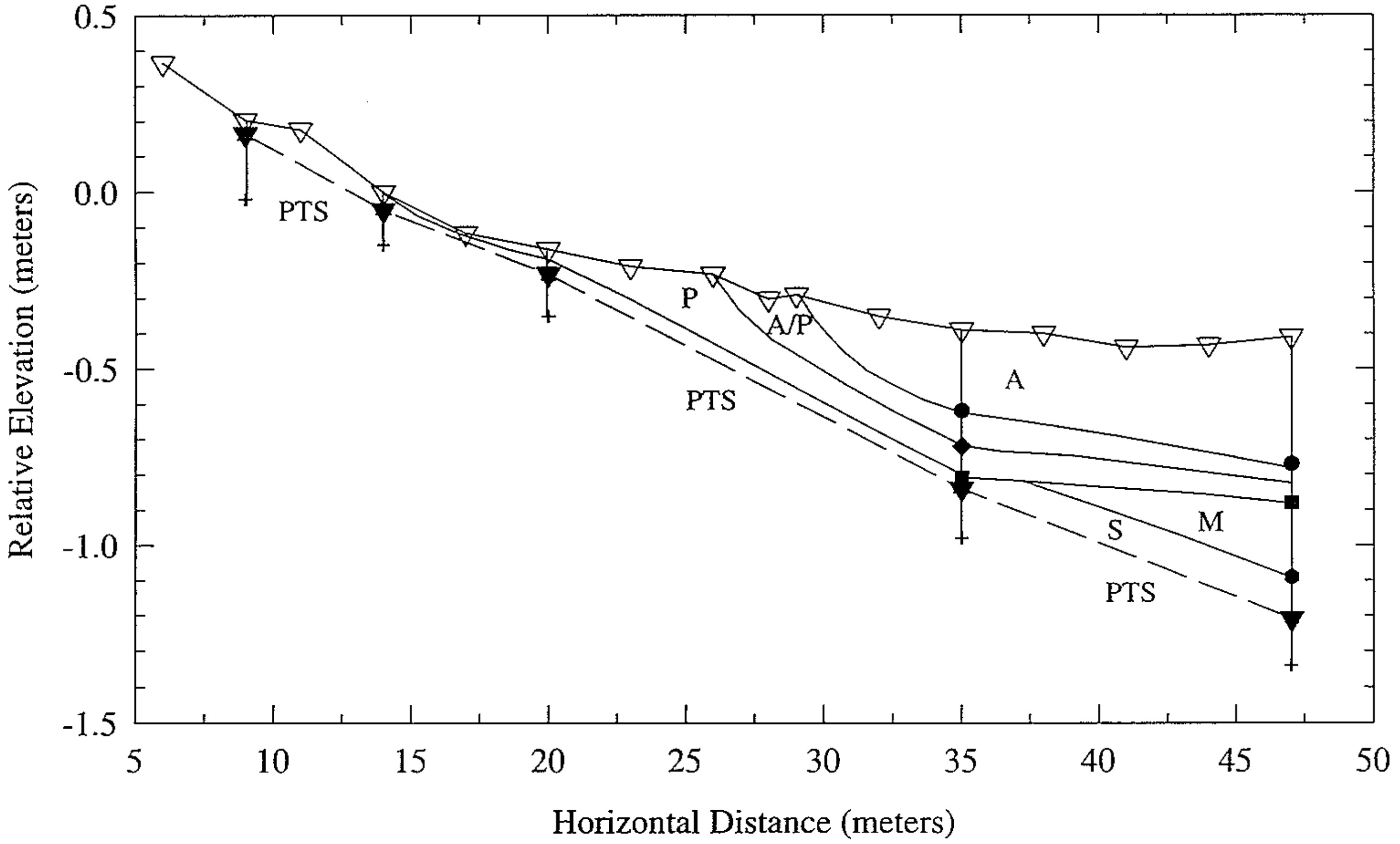
PTS
PTS

Murray - 2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Relative Elevation (meters)
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Horizontal Distance (meters)
PTS
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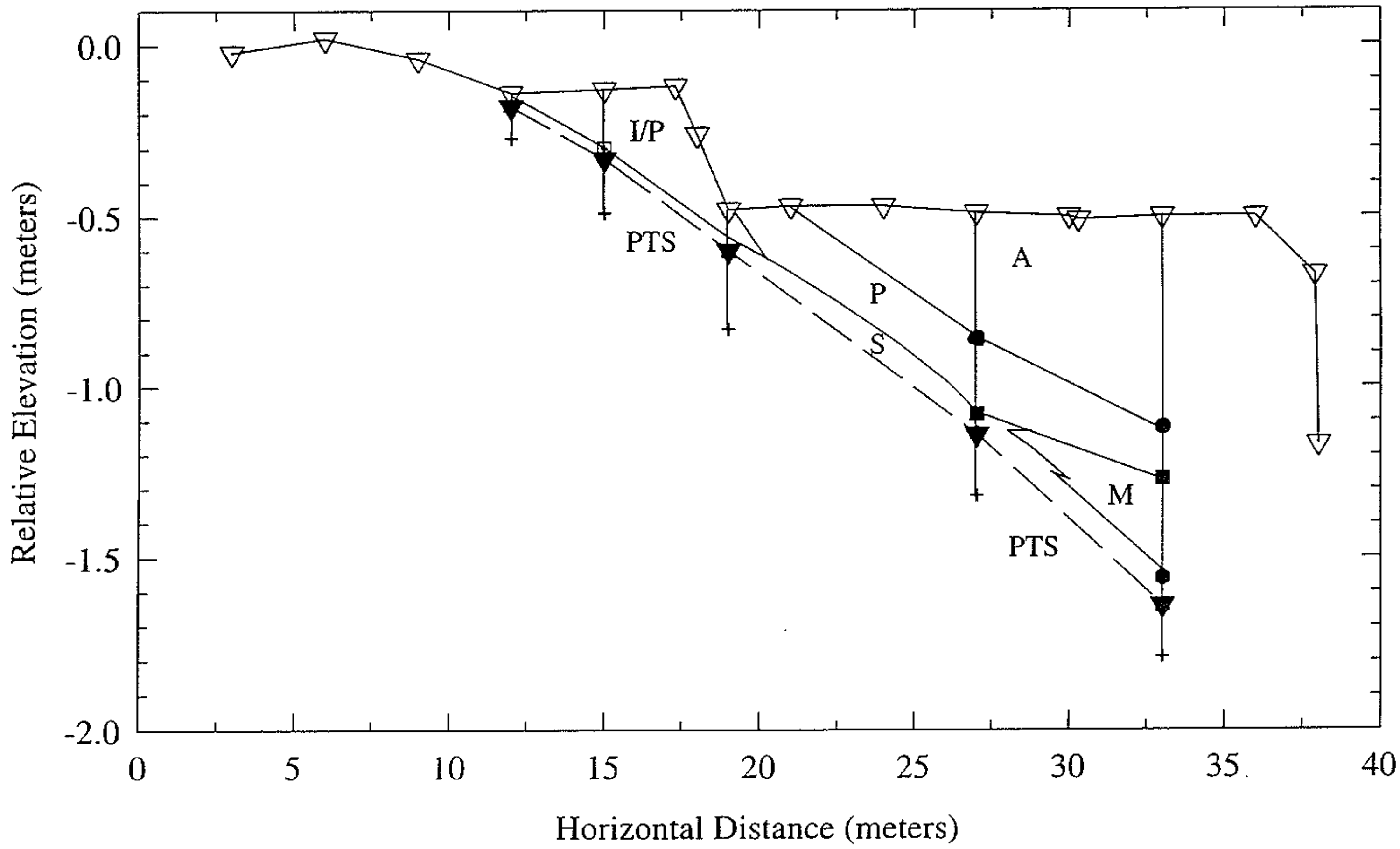
PTS

Nats Marsh - 1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-1.5
-2.0
Relative Elevation (meters)
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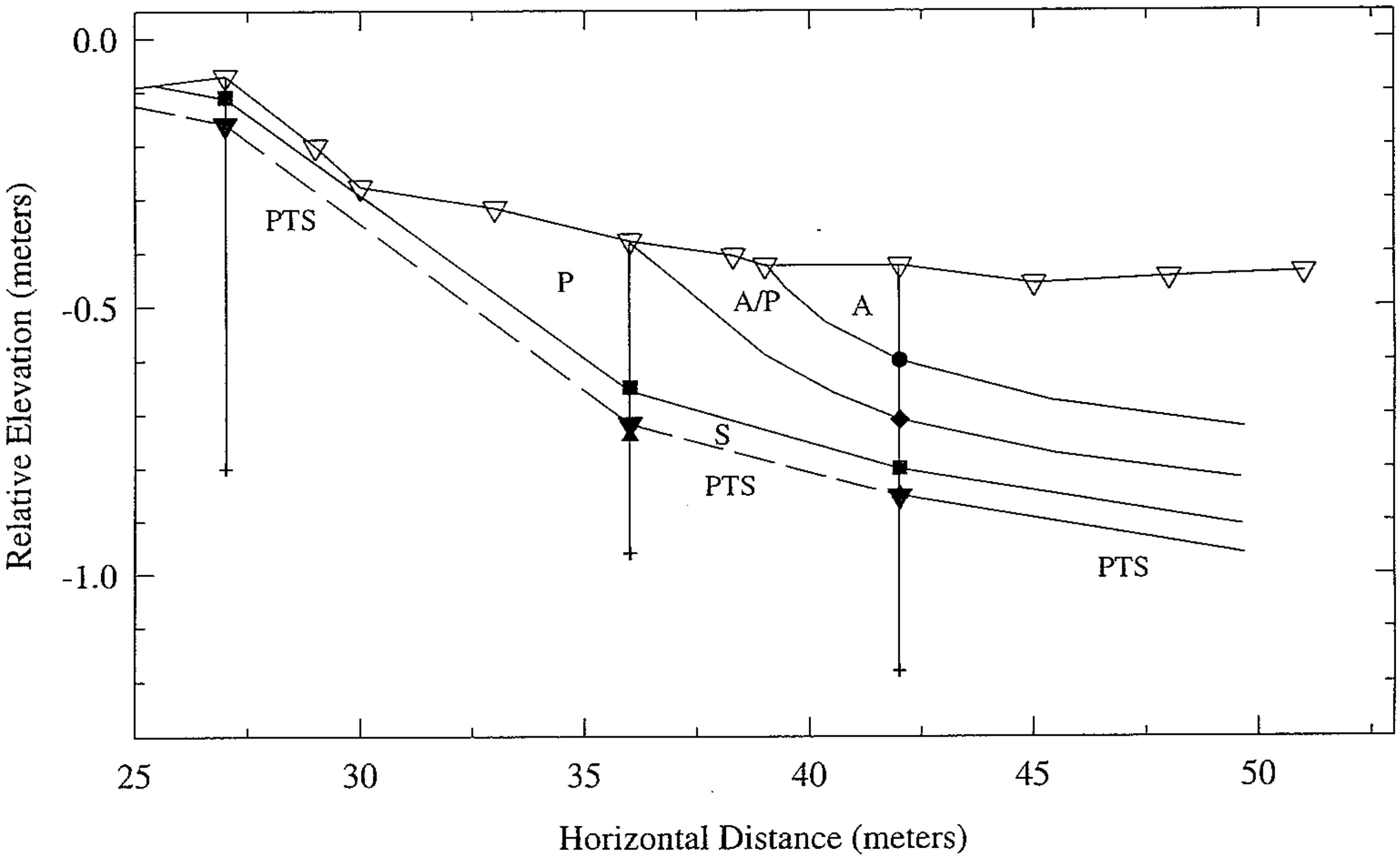
40
Horizontal Distance (meters)
I/P
PTS
A
P
S
M
PTS

Nats Marsh - 2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Relative Elevation (meters)
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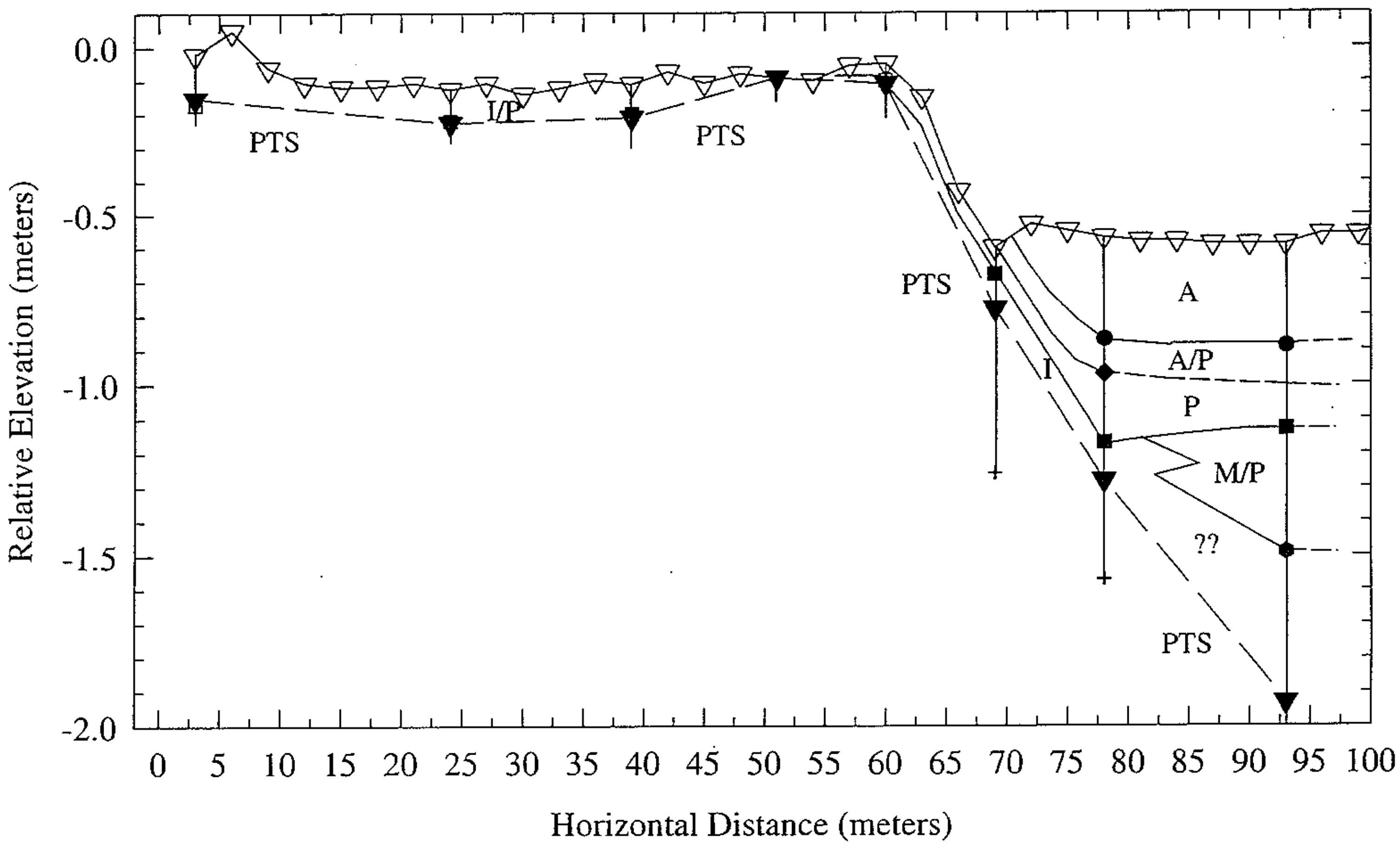
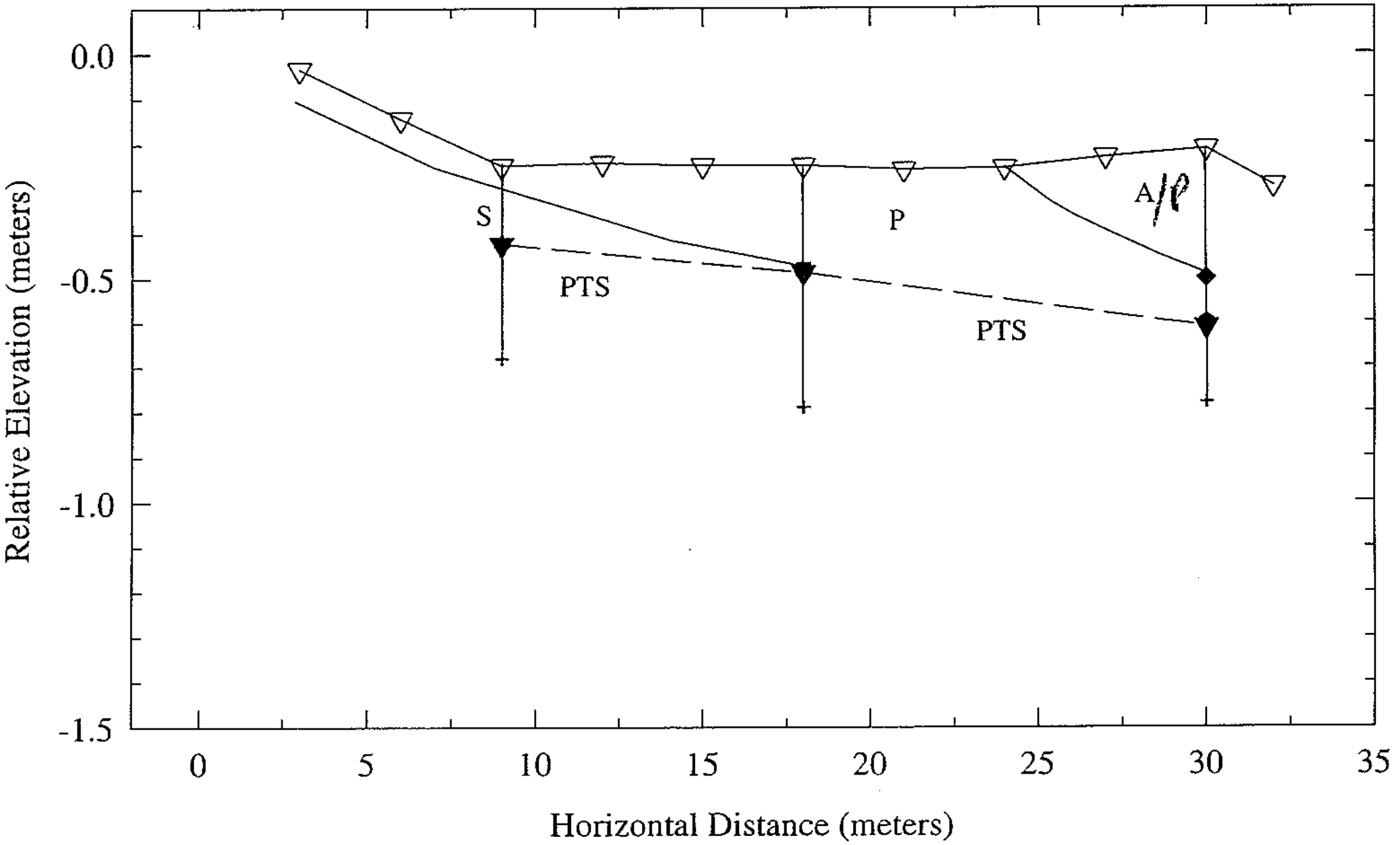
30
35
40
45
50
Horizontal Distance (meters)
PTS
P
A/P
A
S
PTS
PTS

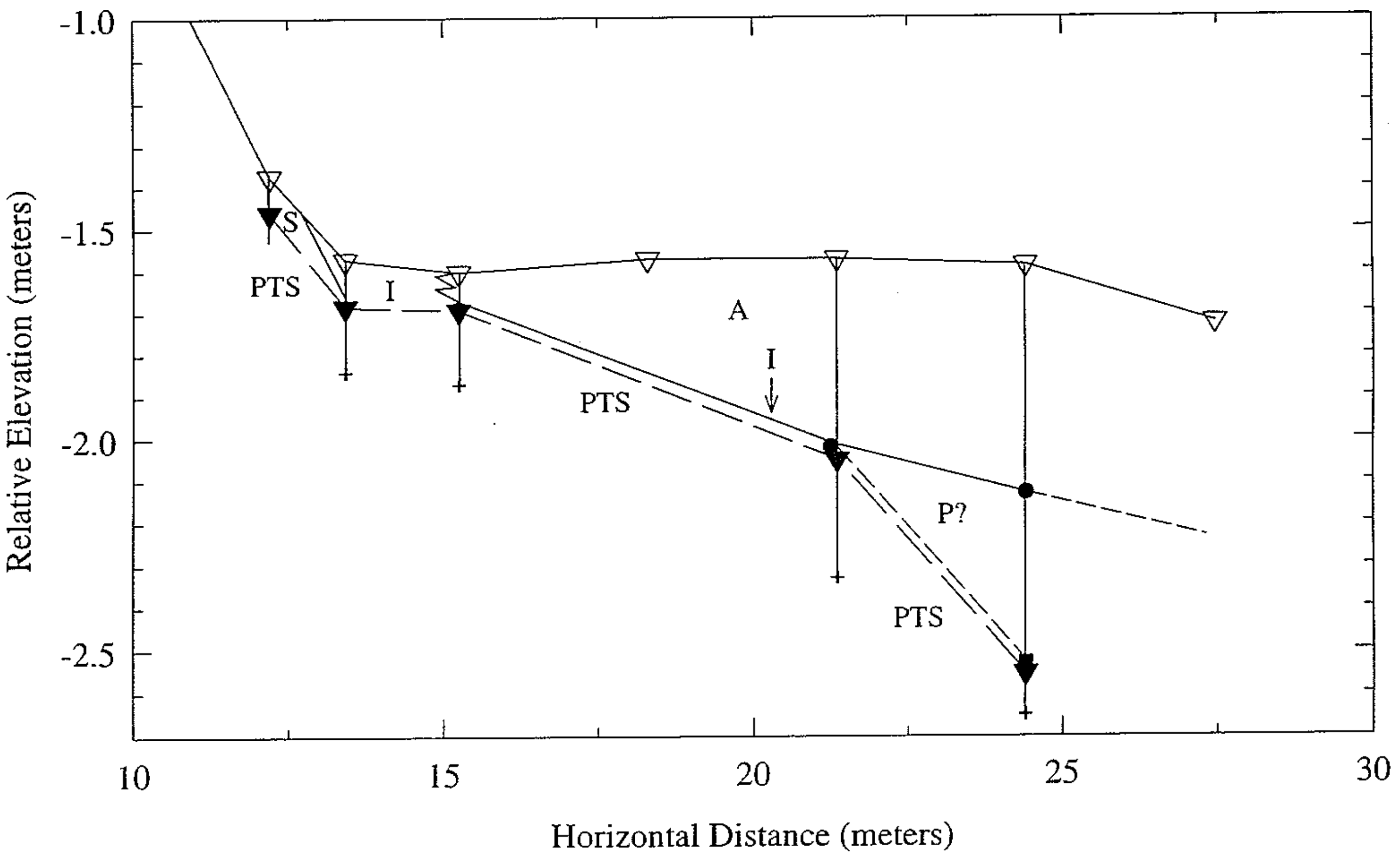
Oyster Neck Road - 1



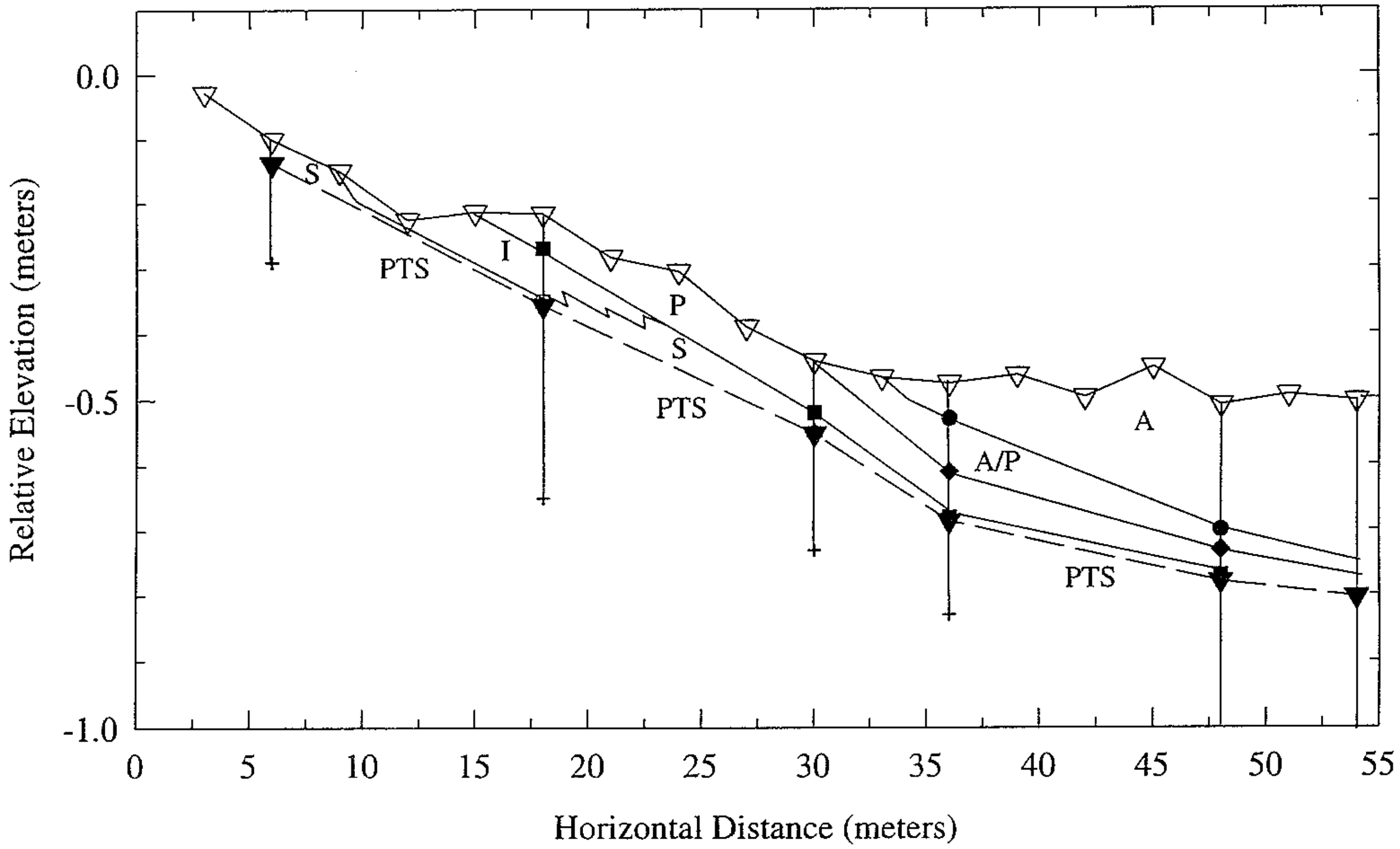
Oyster Neck Road - 2



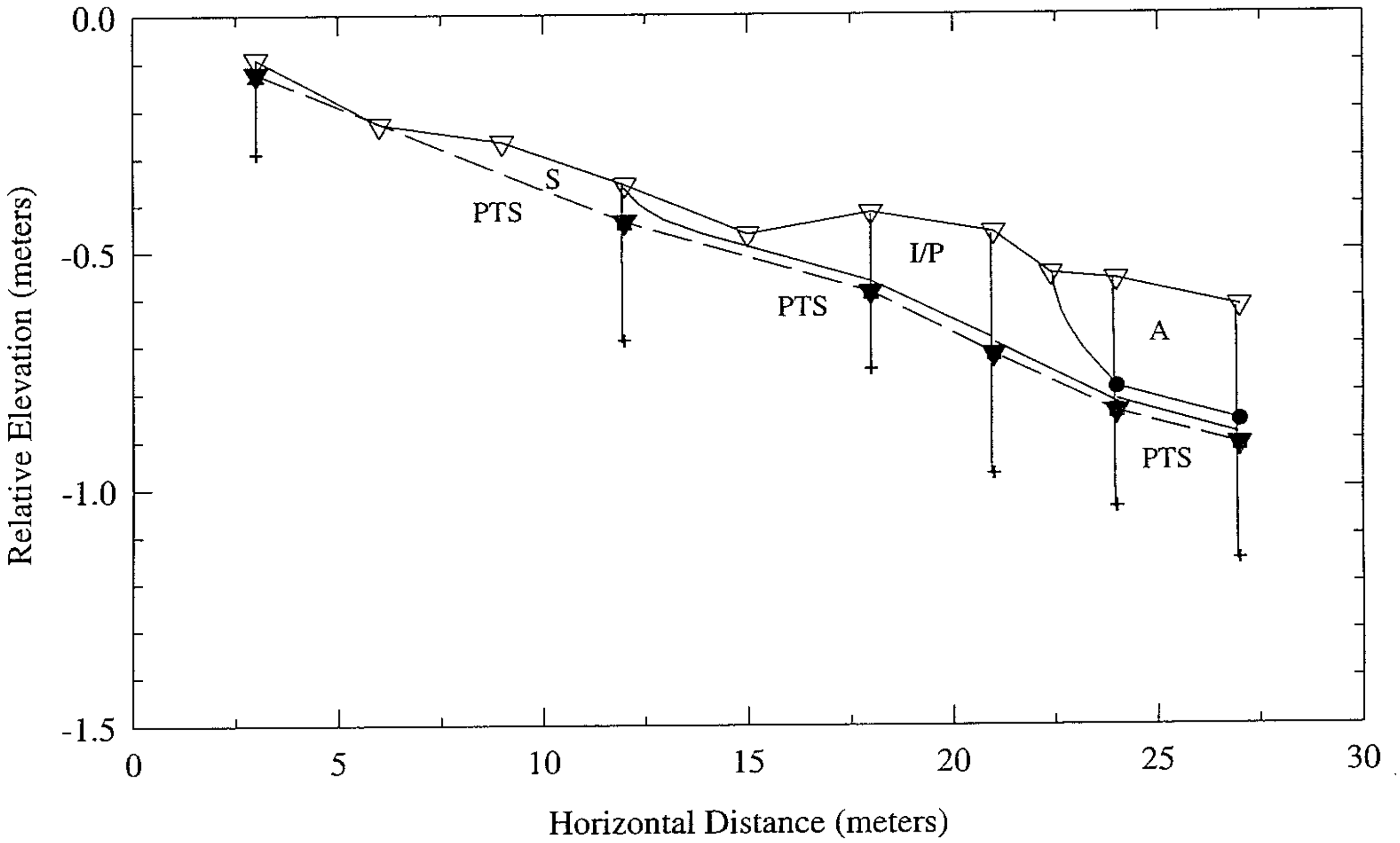
Pagan Creek Village



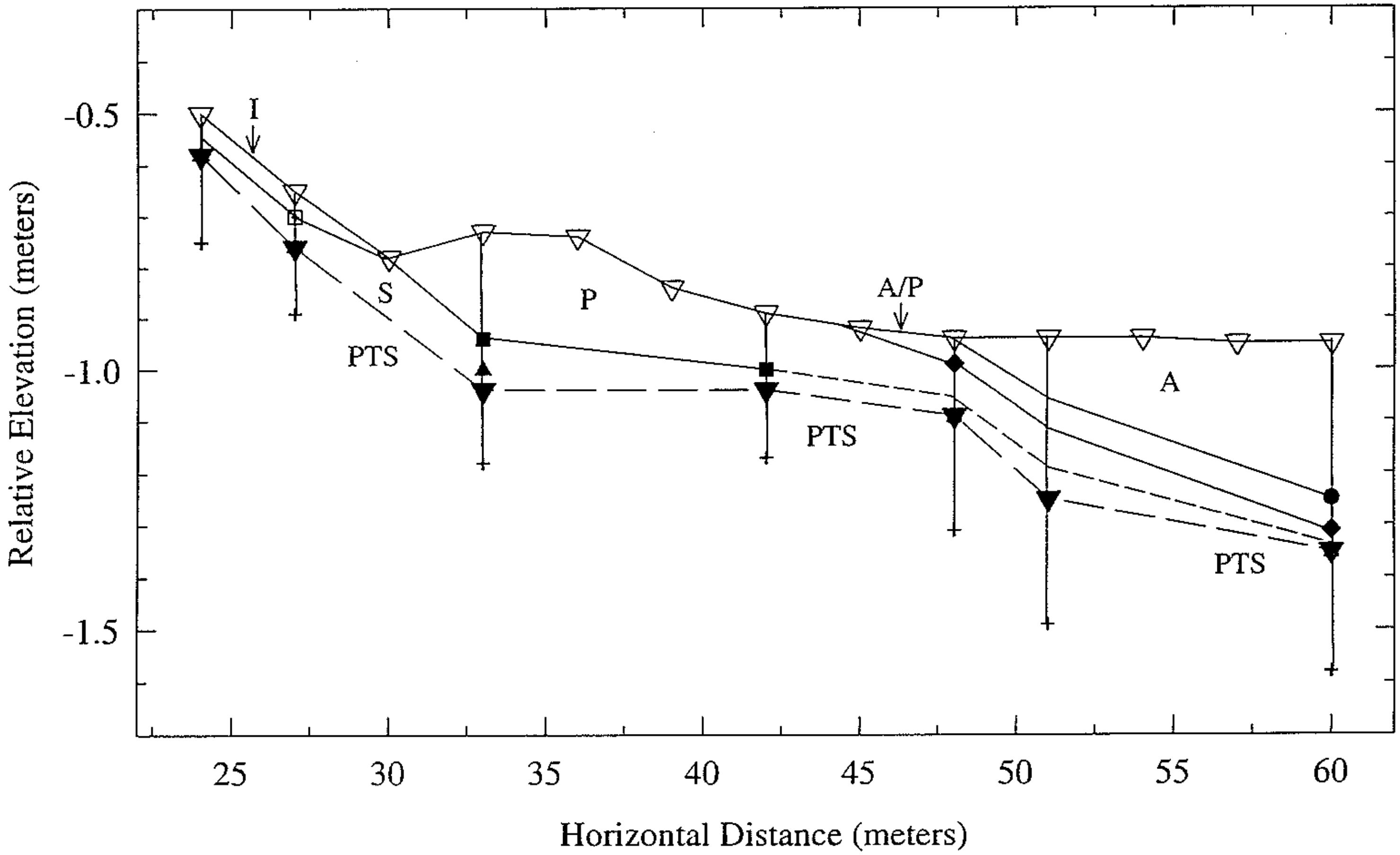
Pasture Point Cove - 1



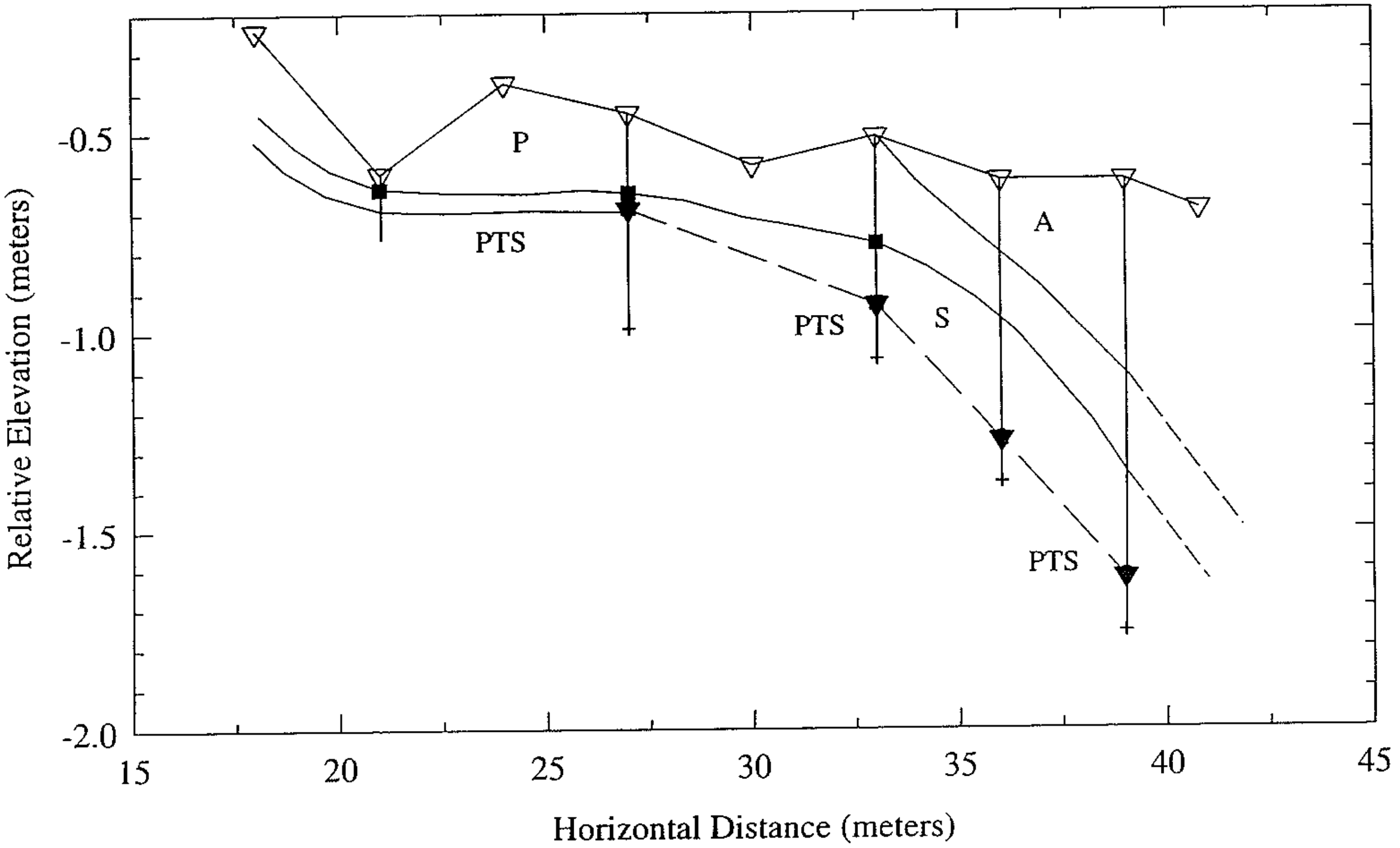
Pasture Point Cove - 2



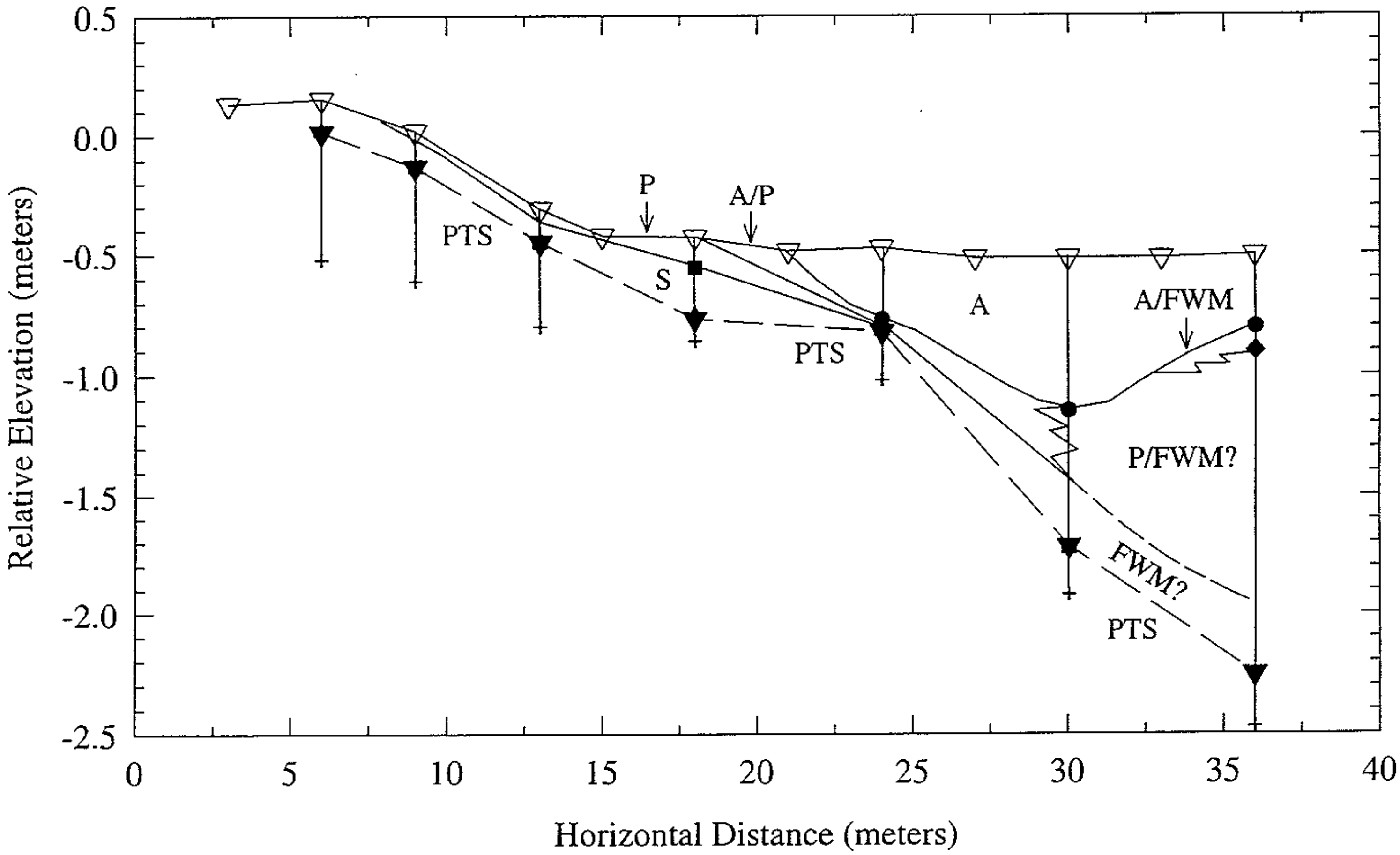
Pot Nets North - 1



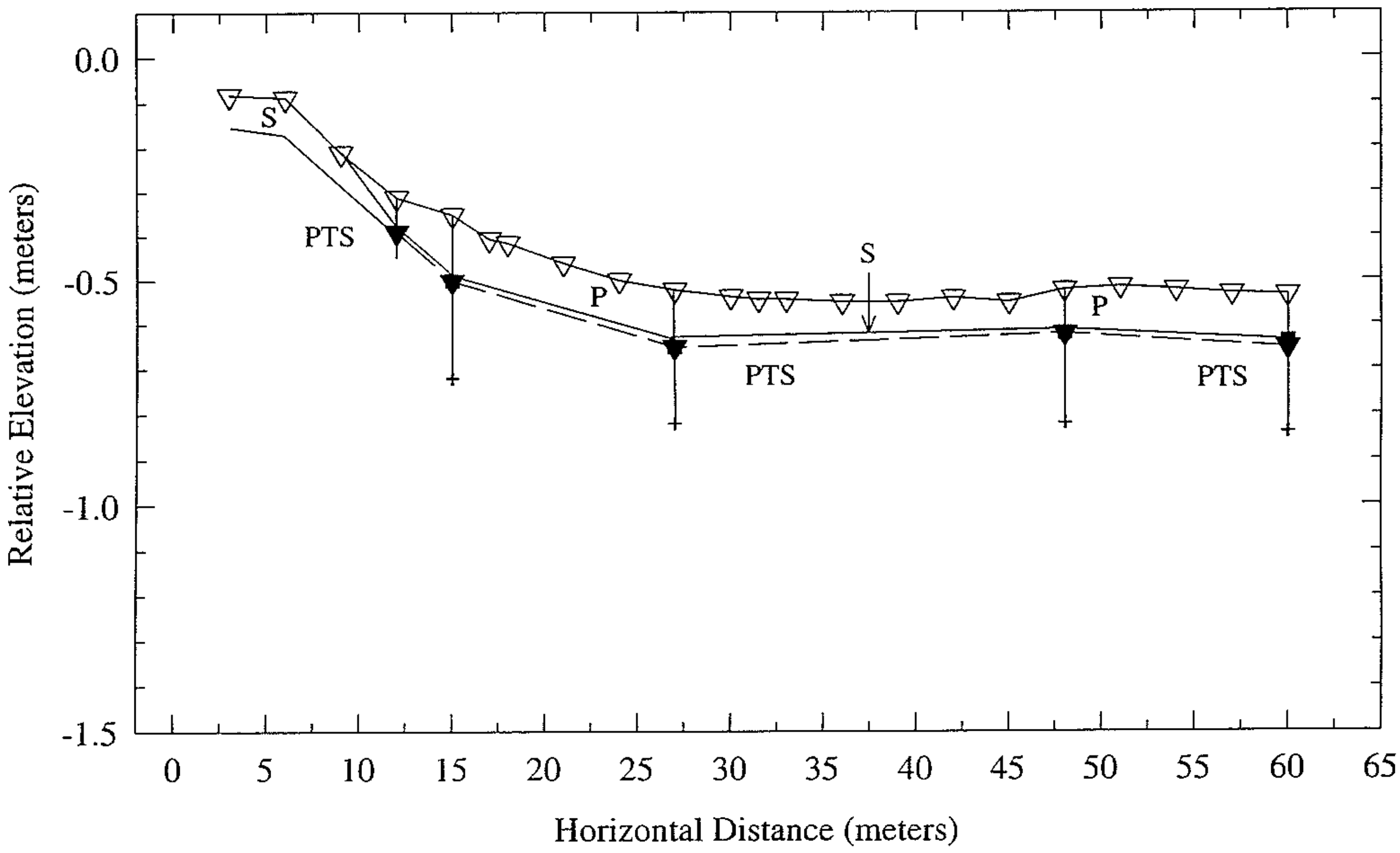
Pot Nets North - 2



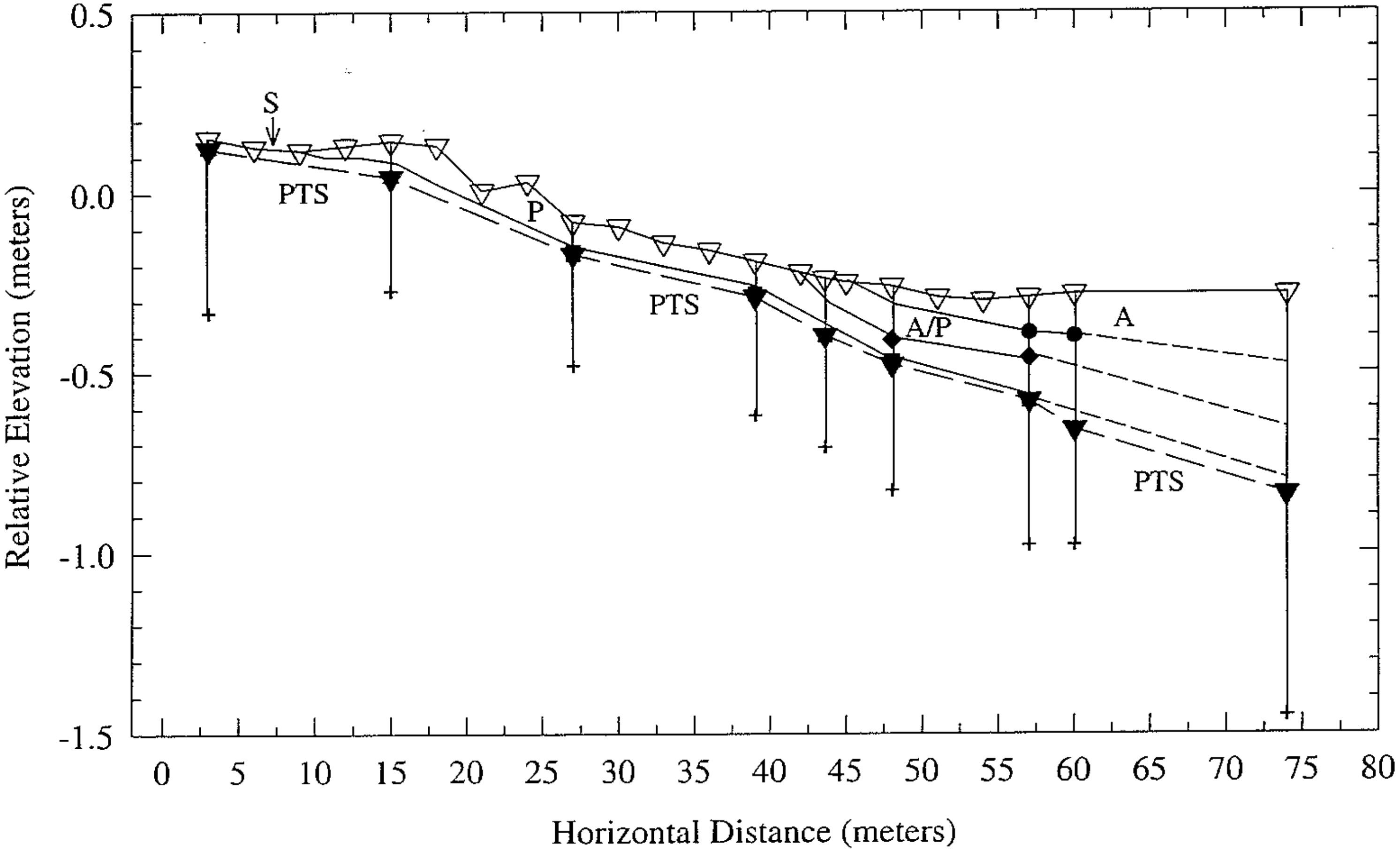
Pot Nets North - 3



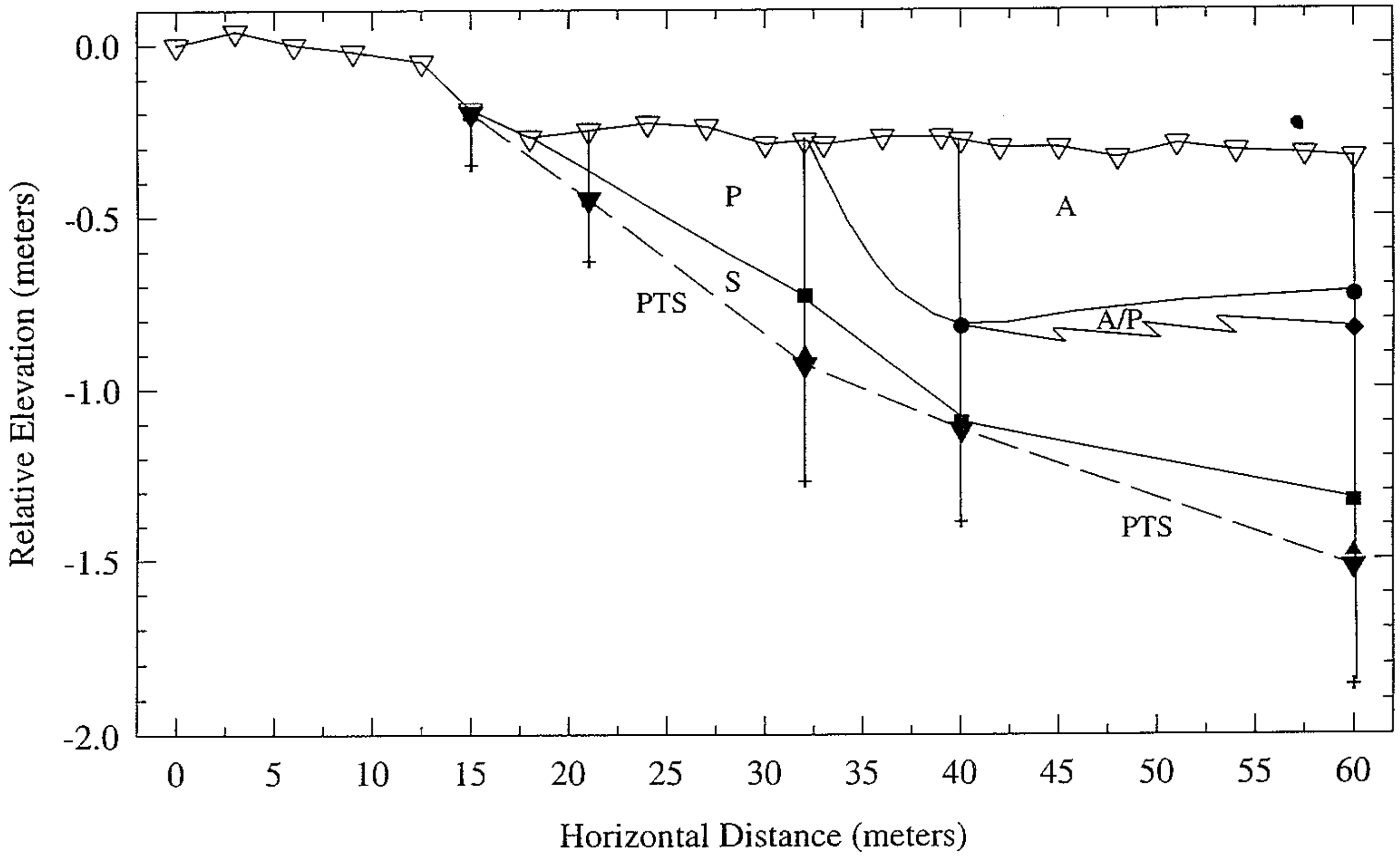
Pot Nets North - 4



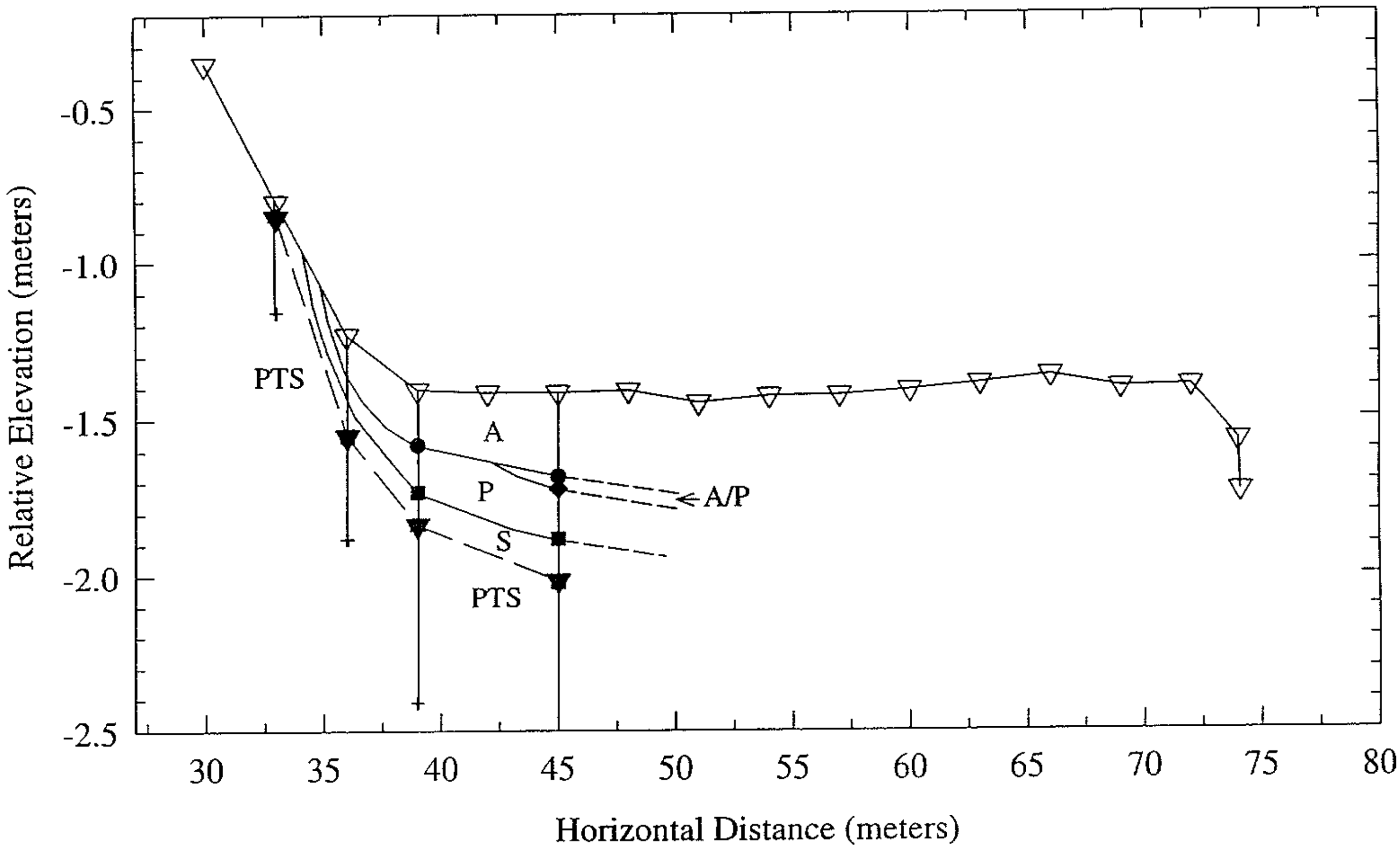
Pot Nets North - 5



Prime Hook Radio Station



Pullover - 1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Horizontal Distance (meters)
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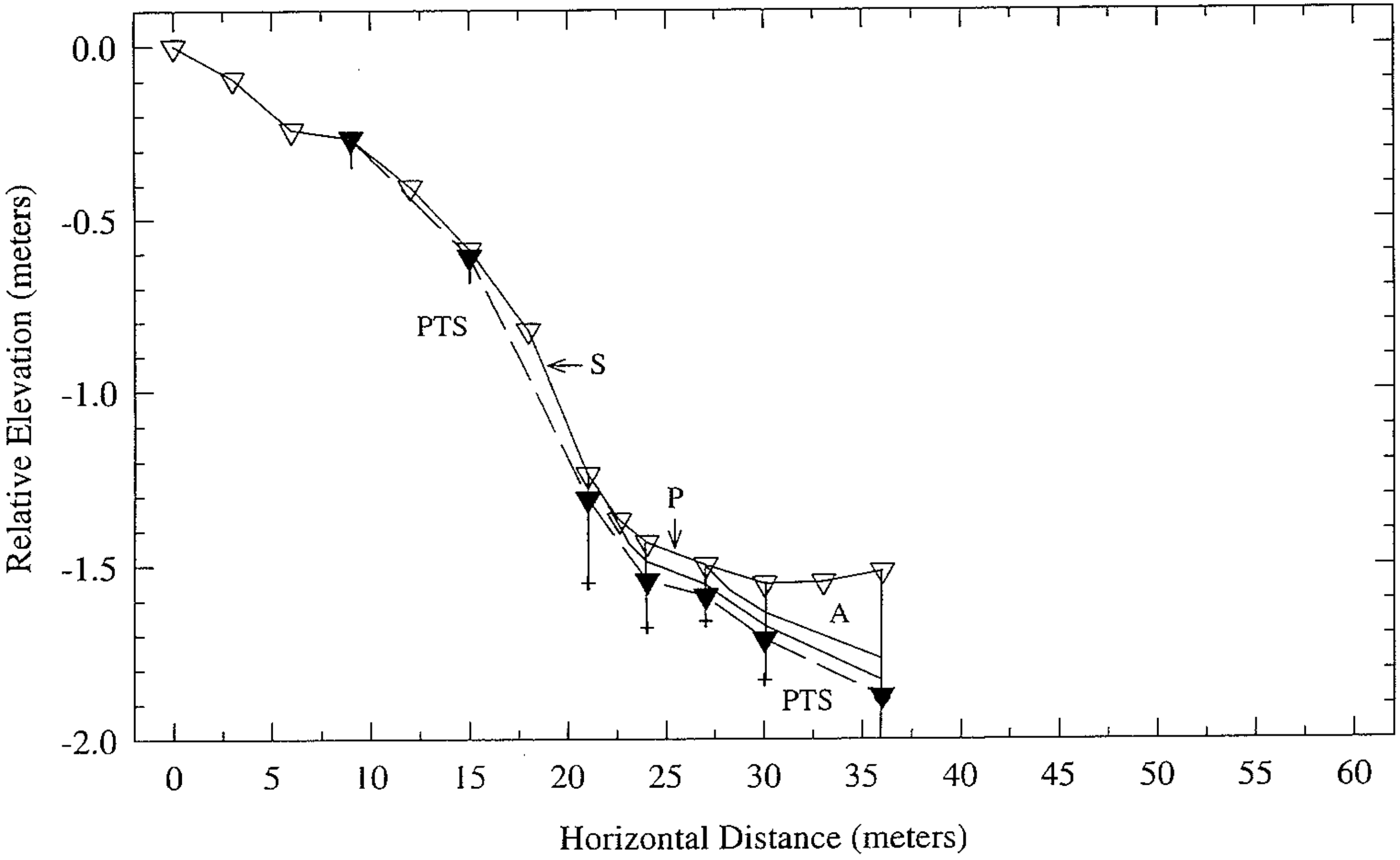
50
55
60
65
70
75
80
PTS
A
P
S
PTS
A/P

Ritter - 1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Horizontal Distance (meters)
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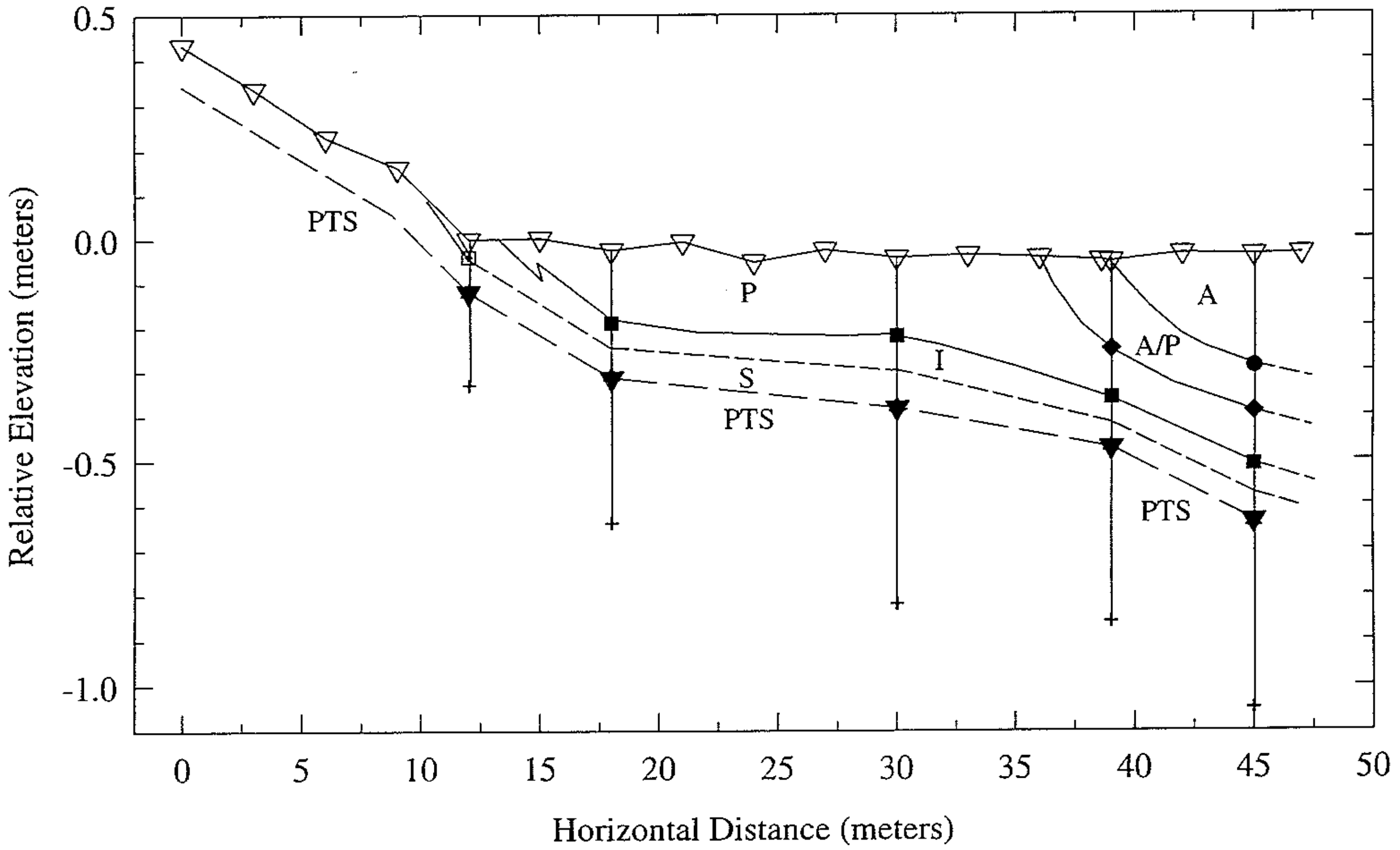
10
15
20
25
30
35
40
45
50
55
60
PTS
S
P
A
PTS

Ritter - 2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Relative Elevation (meters)
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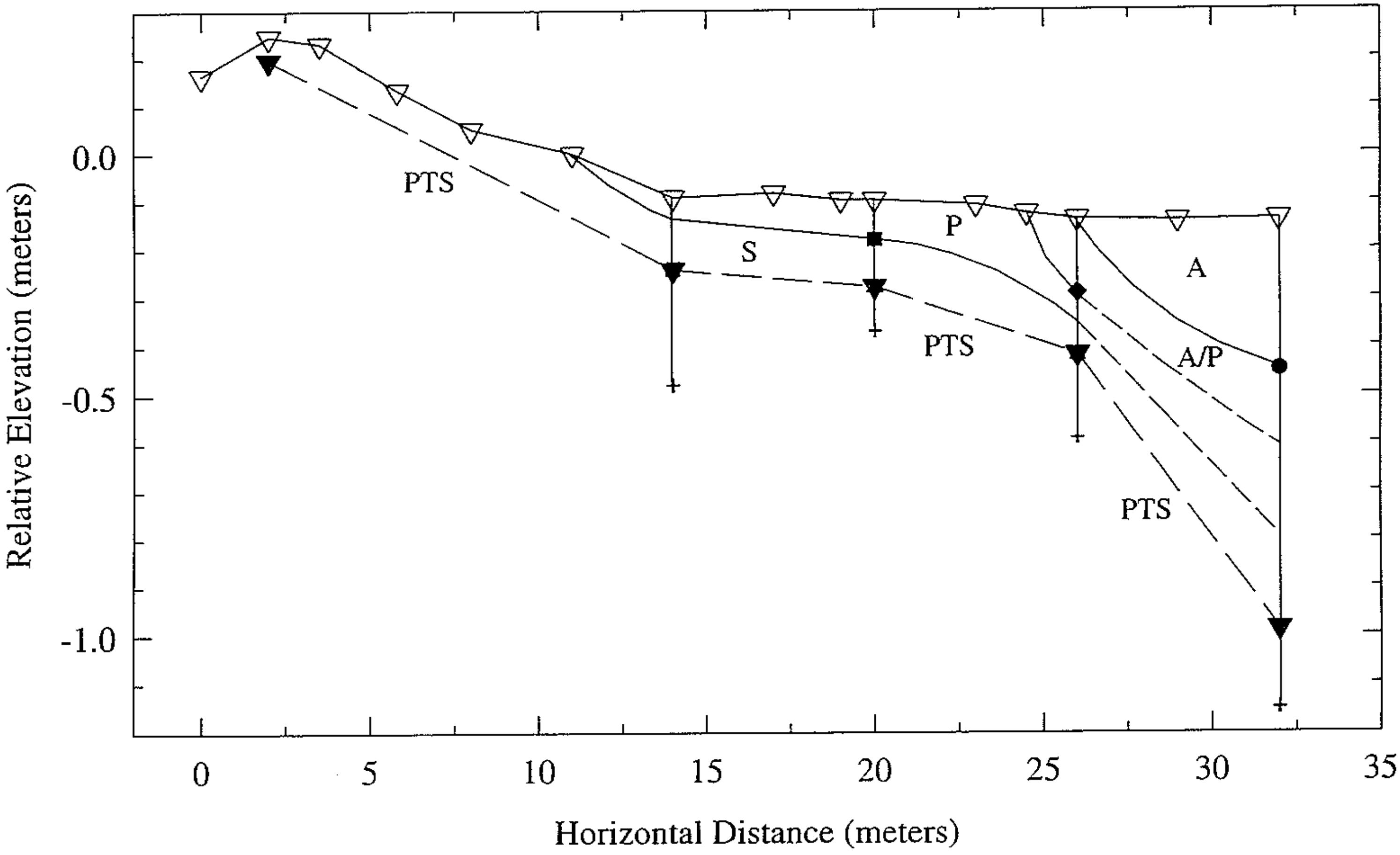
5
10
15
20
25
30
35
40
45
50
Horizontal Distance (meters)
PTS
P
A
A/P
I
S
PTS
PTS

Ritter - 3



PTS
S
P
A
A/P
PTS
PTS
0.0
-0.5
-1.0
Relative Elevation (meters)
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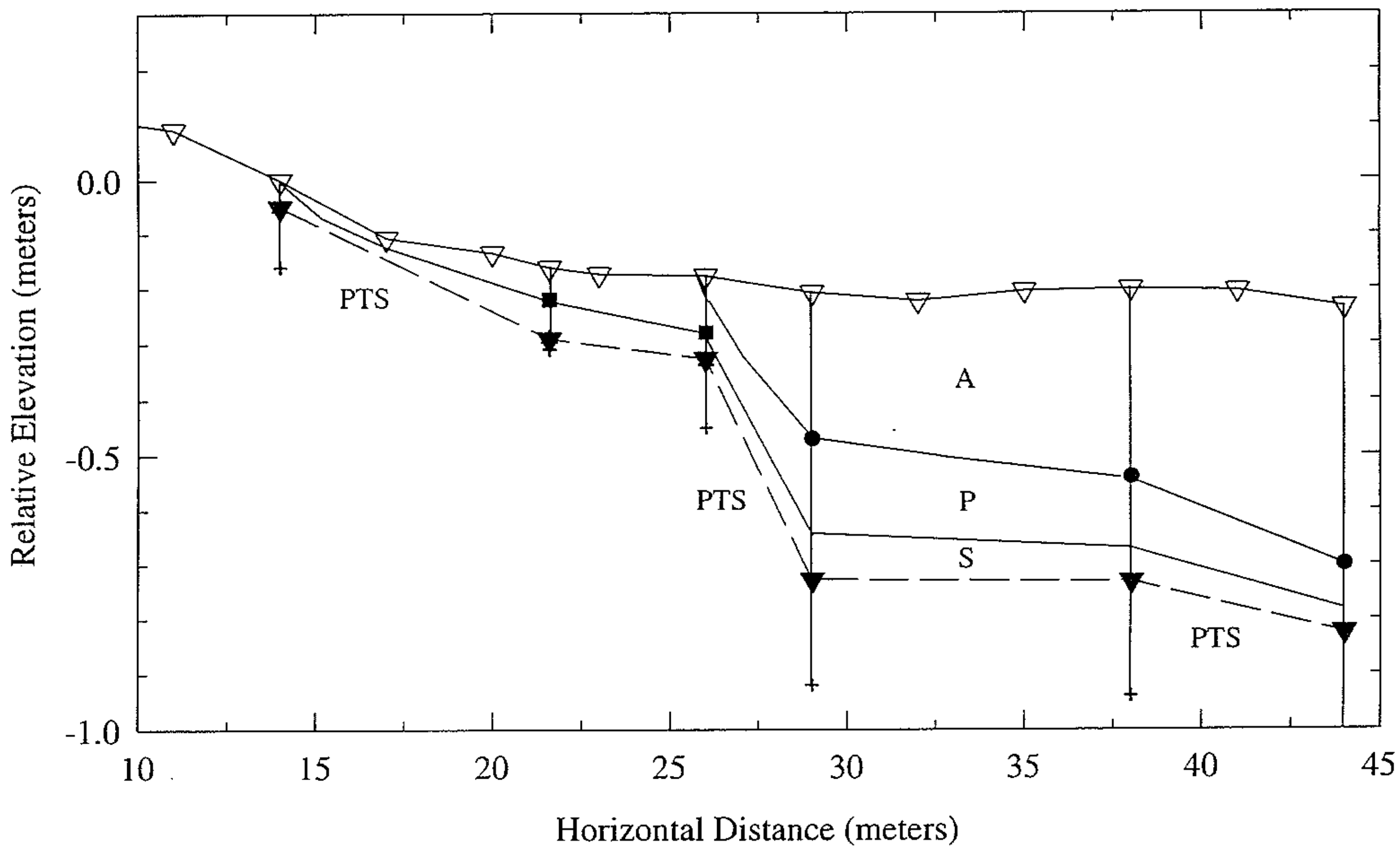
5
10
15
20
25
30
35
Horizontal Distance (meters)

Ritter - 4


PTS
PTS
A
P
S
PTS
Relative Elevation (meters)
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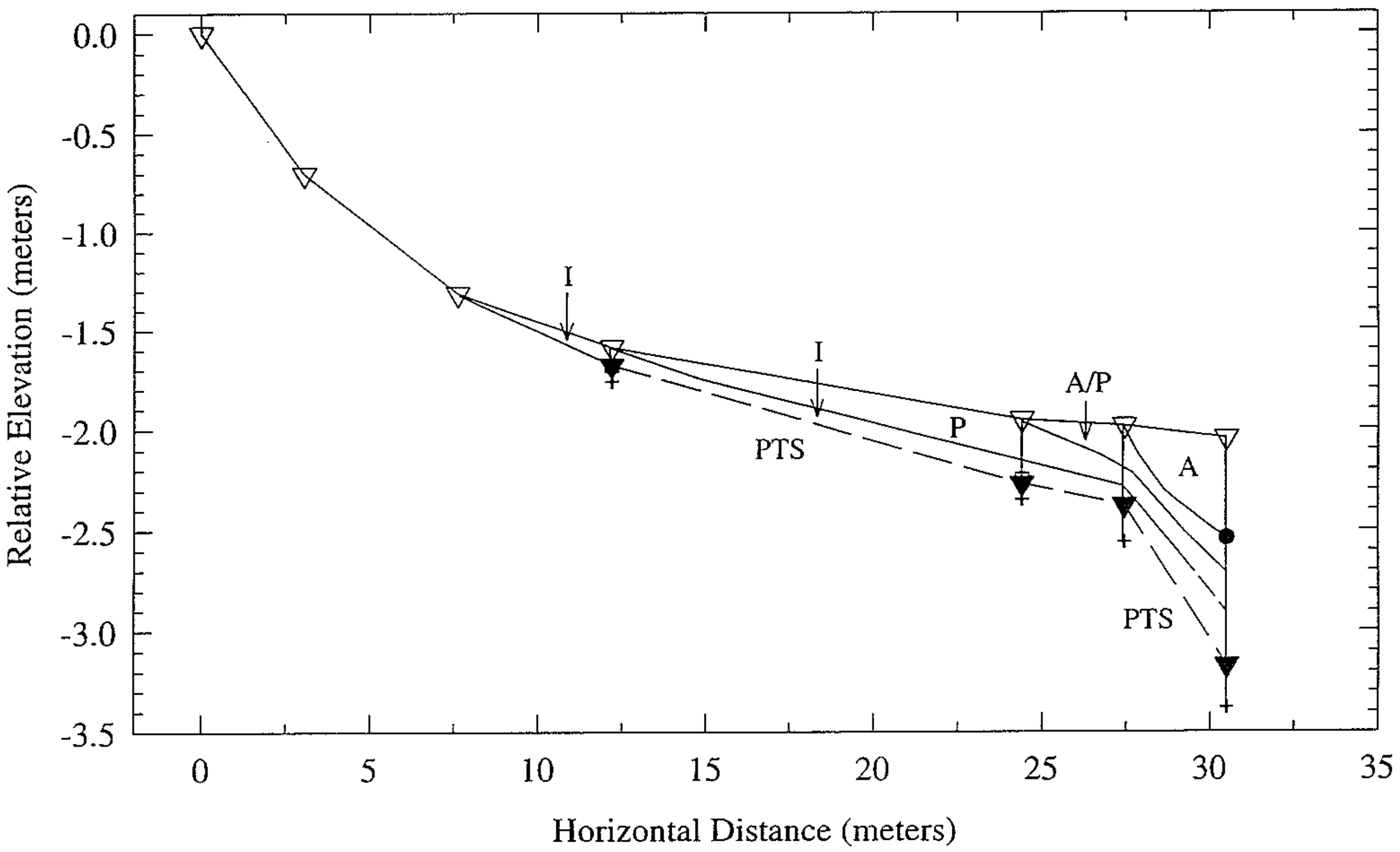
-0.5
-1.0
10
15
20
25
30
35
40
45
Horizontal Distance (meters)

Seawinds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-1.0
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-2.0
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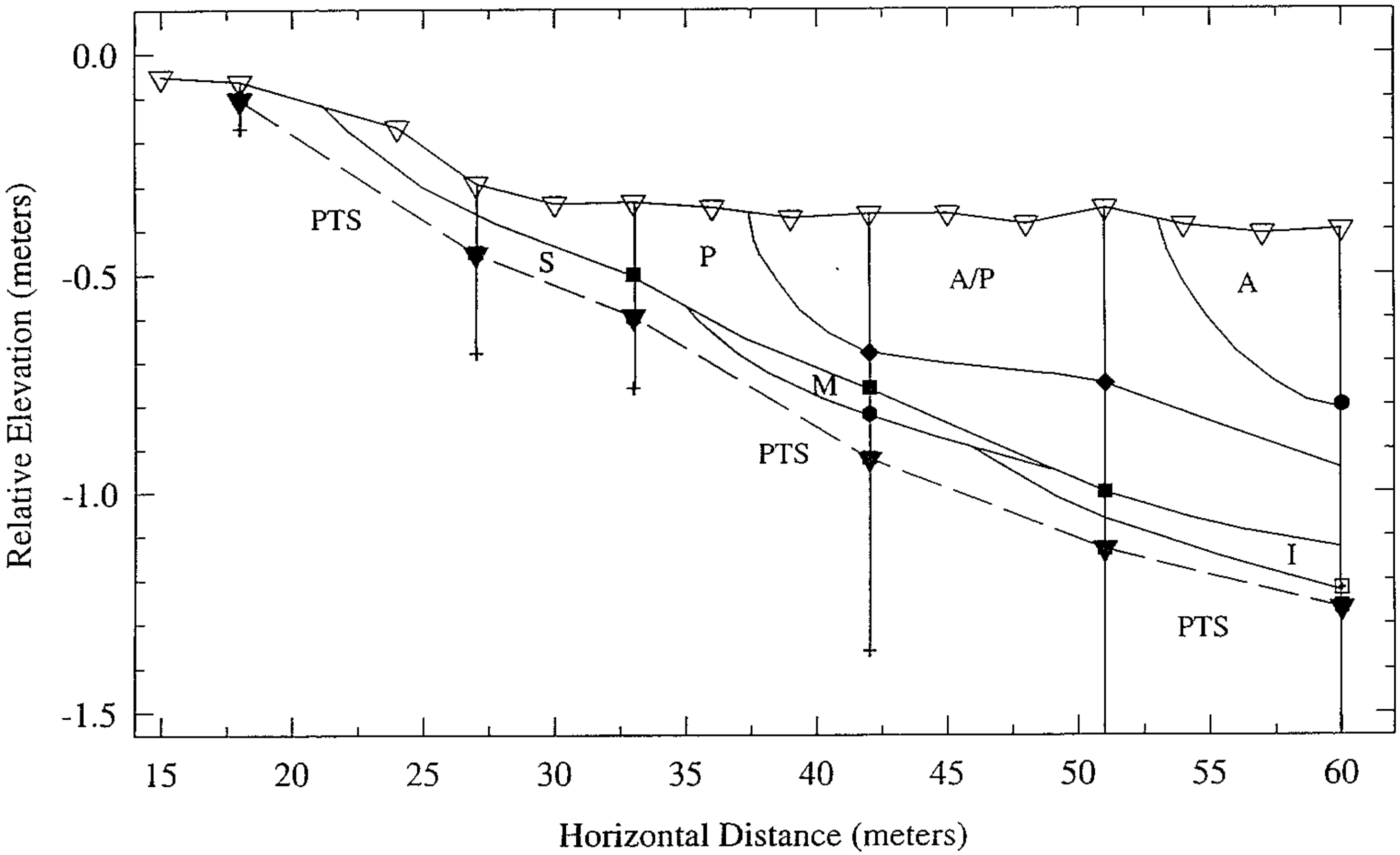
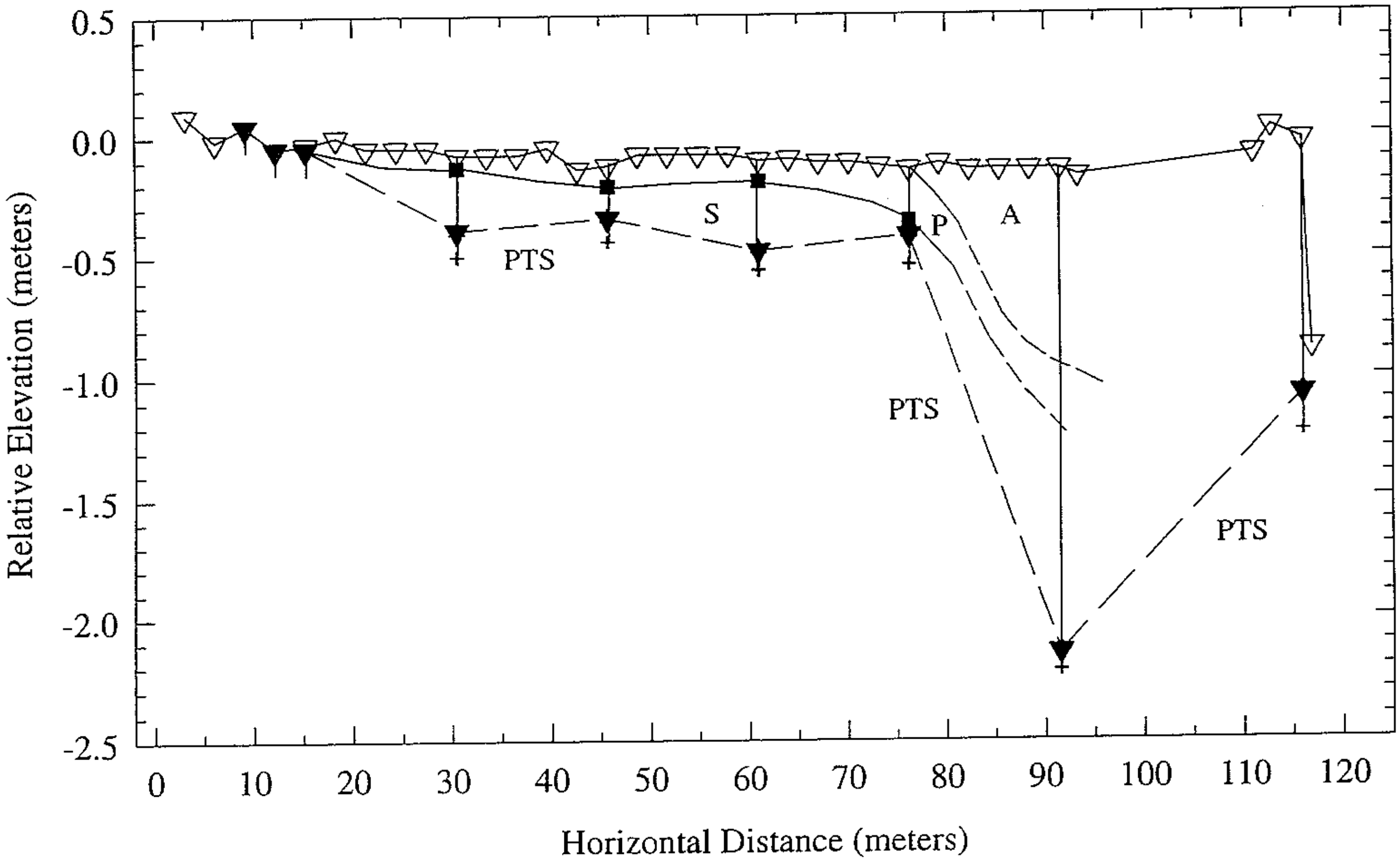
-3.0
-3.5
Relative Elevation (meters)
0
5
10
15
20
25
30
35
Horizontal Distance (meters)
I
I
A/P
P
PTS
A
PTS

Strawberry Landing



Sugar Shack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-1.0
-1.5
-2.0
-2.5
Relative Elevation (meters)
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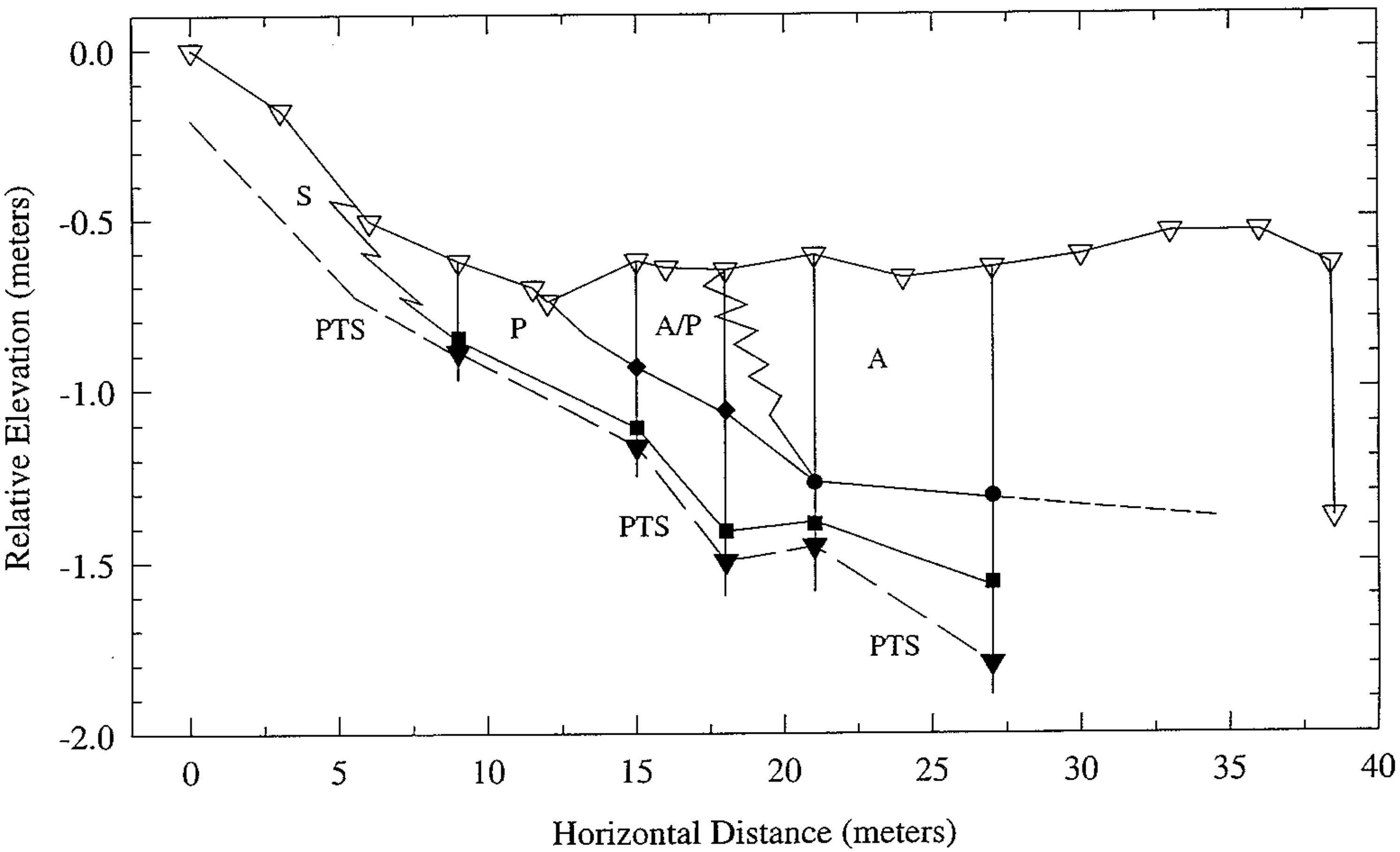
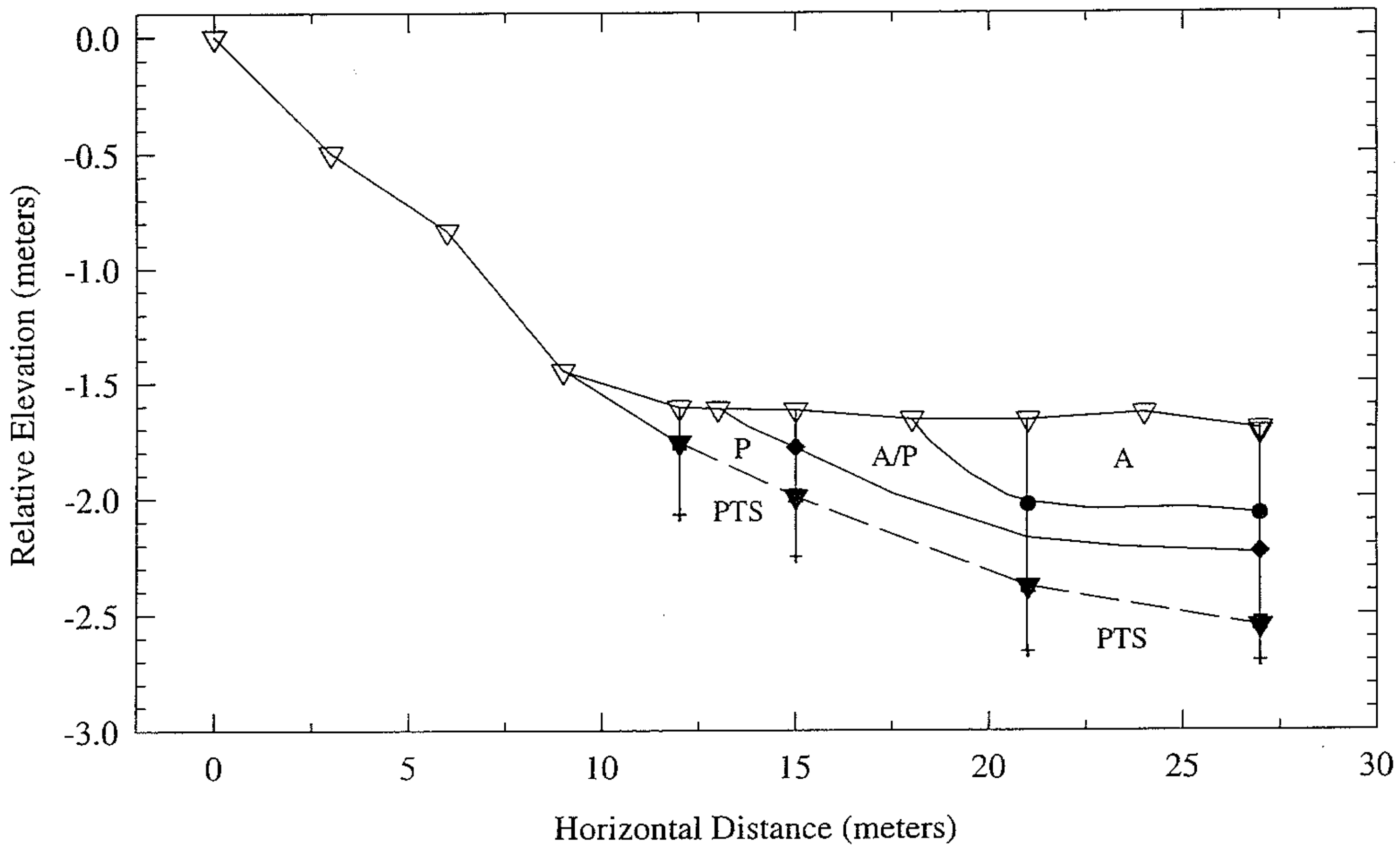
10
20
30
40
50
60
70
80
90
100
110
120
Horizontal Distance (meters)
PTS
S
P
A
PTS
PTS

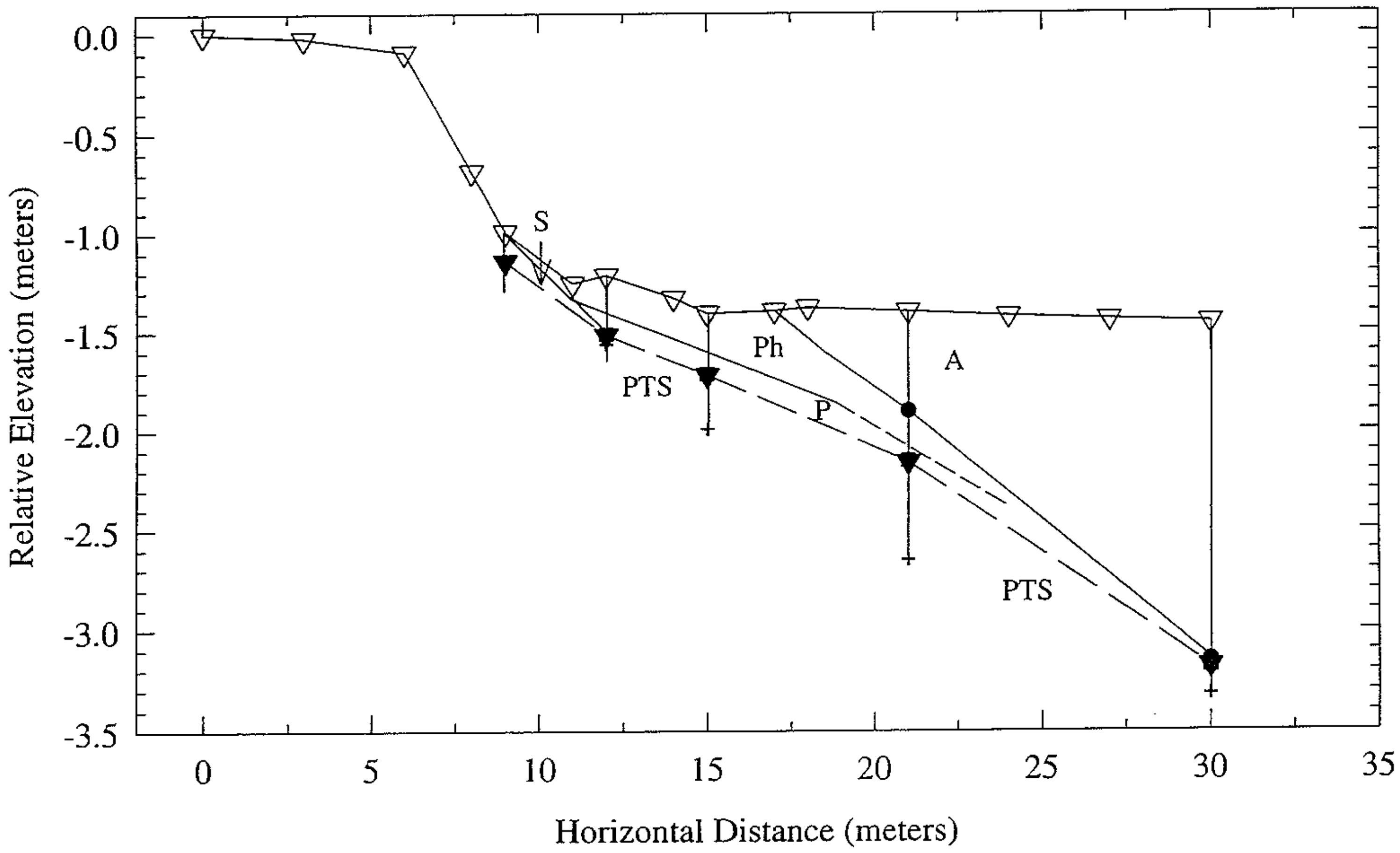
Thompsons Island - 2



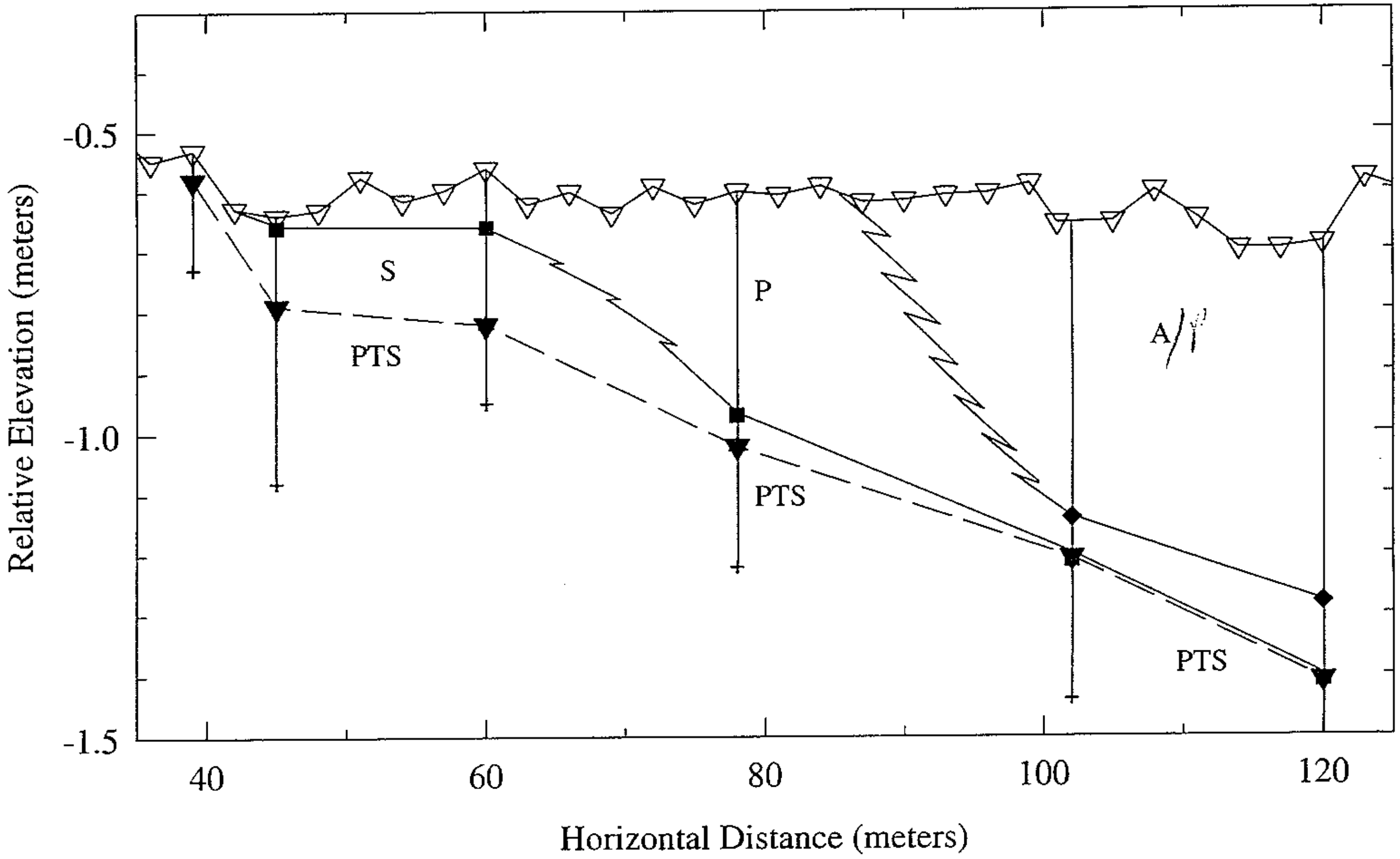
Thompson Island - 3



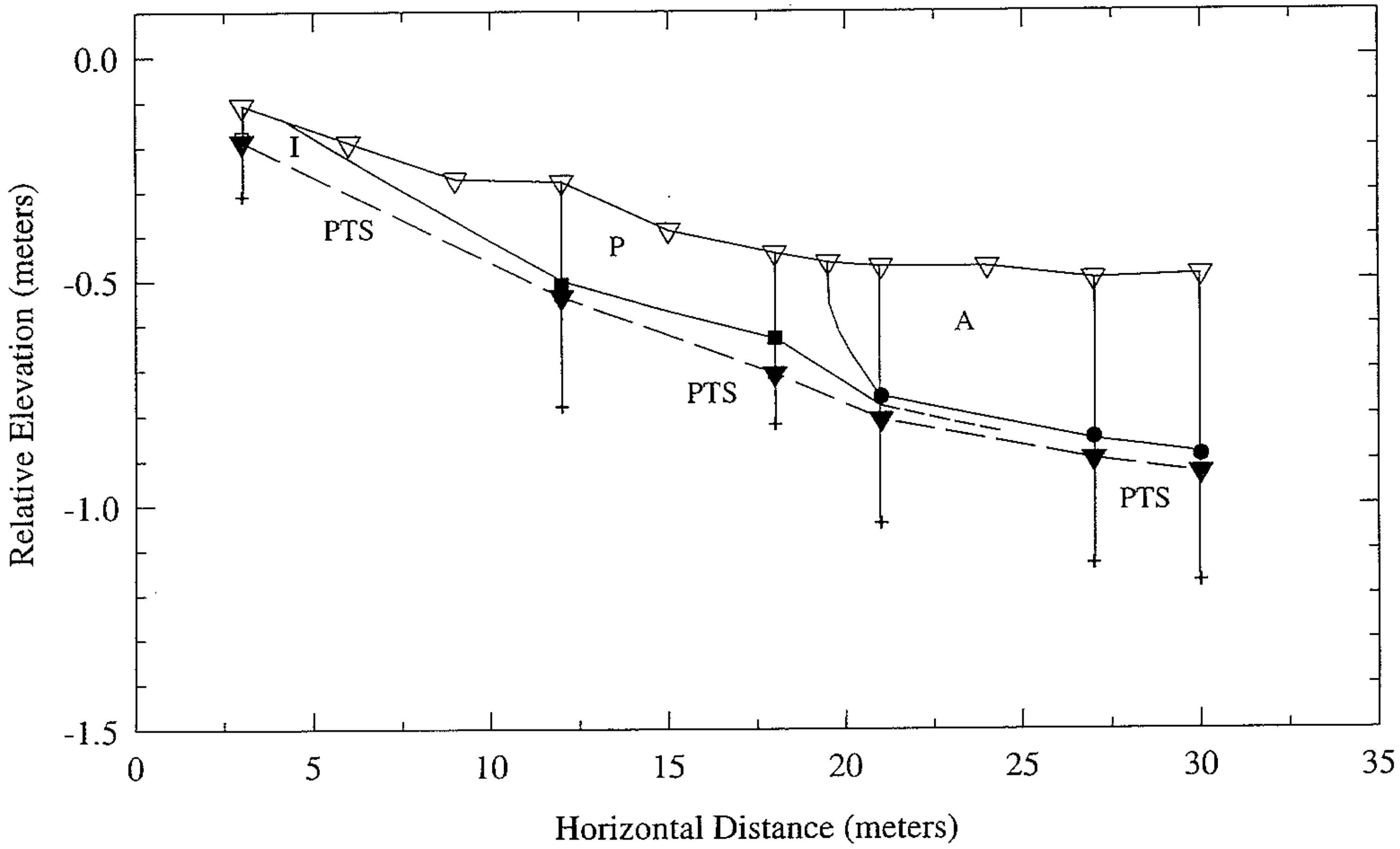
Thompson Island - 4



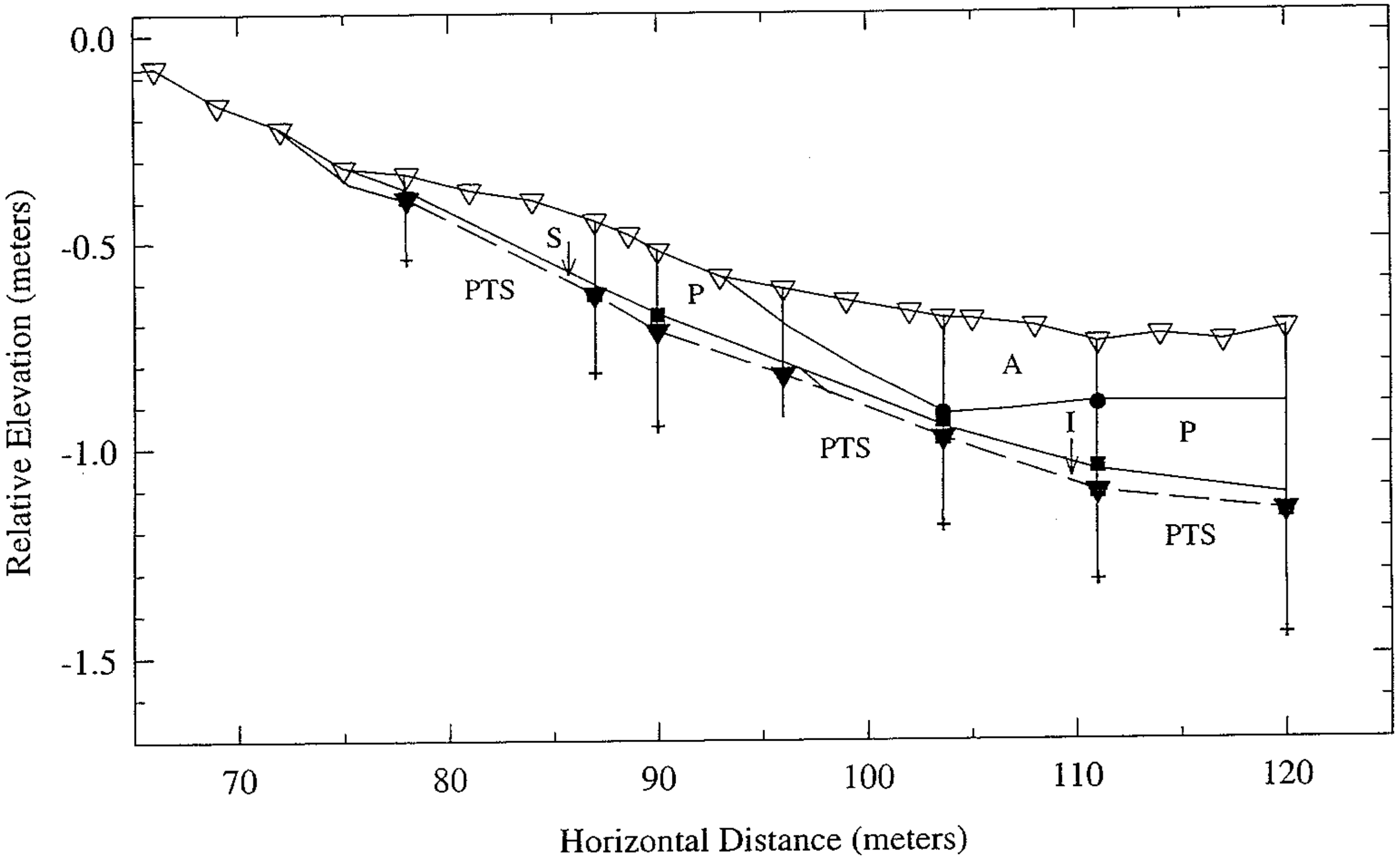
Wall Island - 2



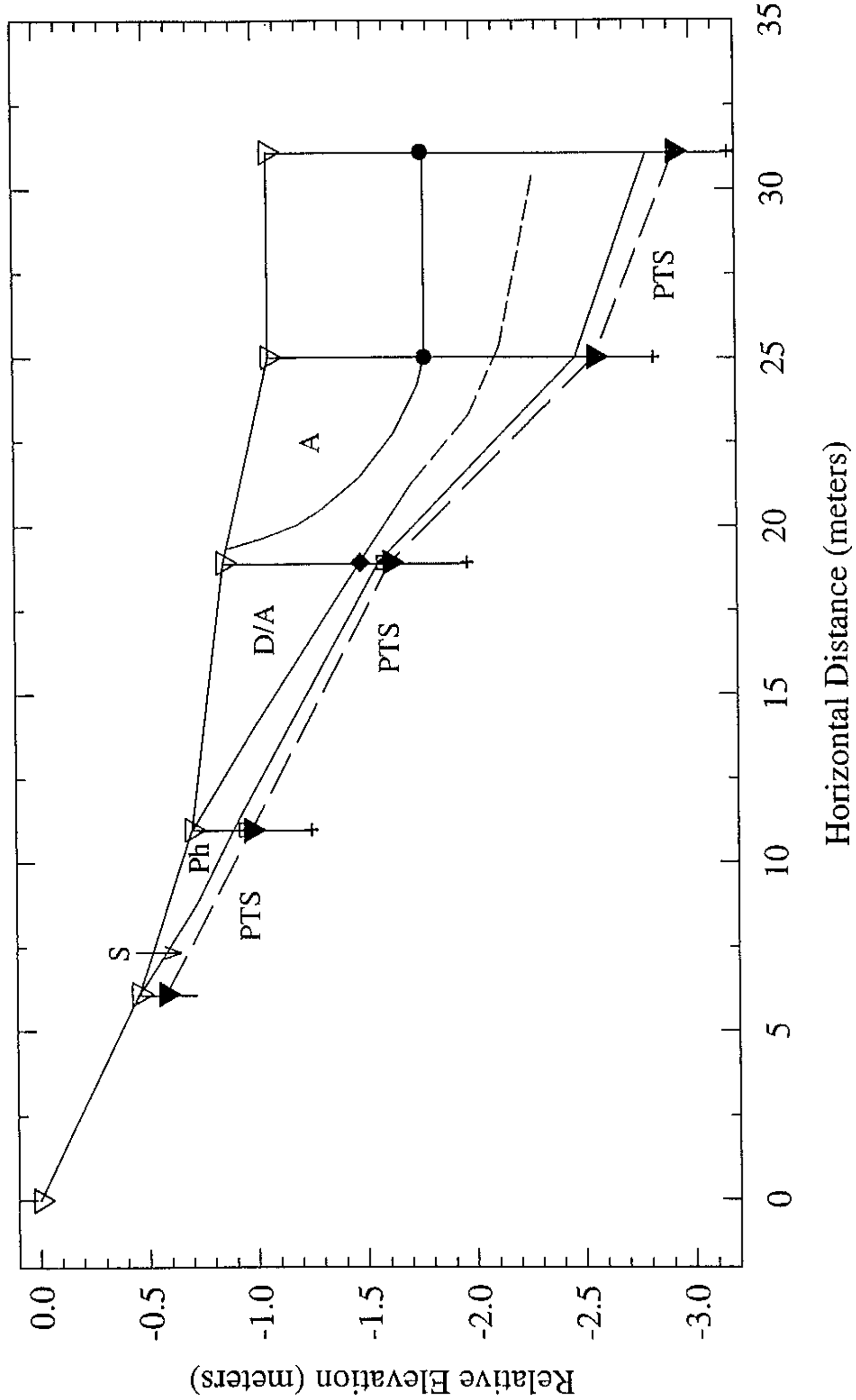
Wm. Derrickson - 1



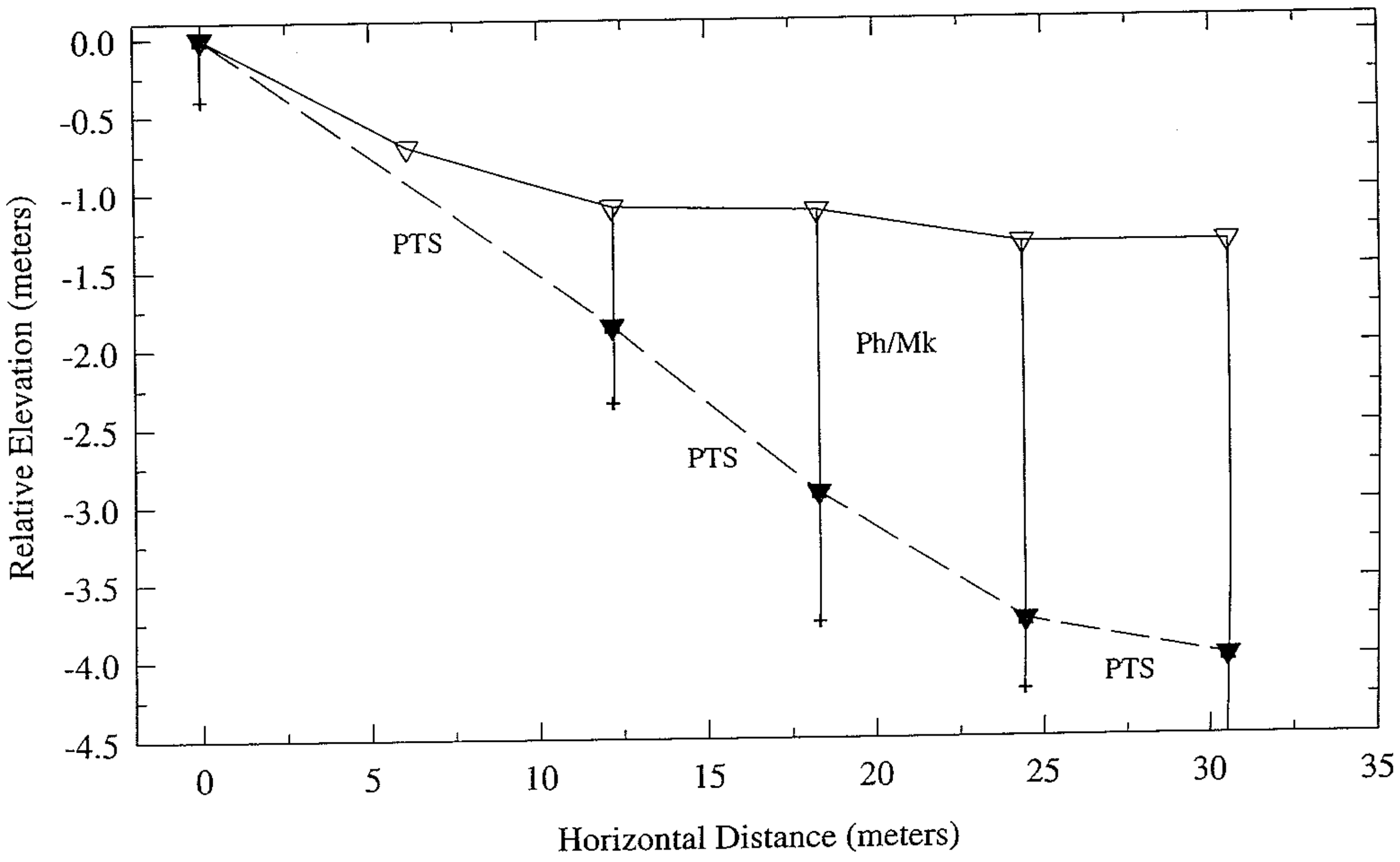
Wm. Derrickson - 2



Wolfe Runne - 1



Wolfe Runne - 2



APPENDIX F

Thickness of Subsurface *S. patens* Units

Site #	Site Location	*S. patens* units: SubsurfaceThickness (cm)
L&R-5	Anderson	9
IRB-6	Bay Colony	21
		14
RB-6	Bay Vista	17
IRB-11	Bethany Bay 1	7
		8
		6
		8
IRB-12	Bethany Bay 2	16
		8
IRB-14	Bethany Bay 4	27
		37
		48
IRB-10	Bethany Forest	12
	(Iva)	17
		23
		20
IRB-9	Blackwater Pt.	22
		8
IRB-2	Boat House Cove	19
		40
		17
		20
		30
IRB-3	Burton 1	21
		6
		16
IRB-4	Burton 2	20
		22
RB-13	Camp Arrow 2	110
		95
		110
		56
RB-14	DE WL 1	12

Site #	Site Location	*S. patens* units: SubsurfaceThickness (cm)
RB-22	Nats Marsh 2	20
DB-7	ONR 1	8
		30
		24
DB-8	ONR 2	11
IRB-15	Past Point Cove	15
		7
IRB-16	Past. Point 2	8
		5
		5
RB-16	PNN 1	10
		11
		12
RB-17	PNN 2	18
		23
RB-18	PNN 3	4
RB-20	PNN 5	19
		15
		18
		22
DB-5	PHRS	28
		60
RB-24	Pullover	15
		20
DB-7	Ritter 1	6
		8
DB-8	Ritter 2	32
		22
DB-9	Ritter 3	22
		36
DB-10	Ritter 4	10, 18
		15
		10

Site #	Site Location	*S. patens* units: SubsurfaceThickness (cm)
RB-15	DE WL 2	44
		28
		10
		60
		65
		45
RB-1	Dodd 1	39
		22
		21
		24
RB-2	Dodd 2	20
		30
RB-7	Draper 1	20
		26
		27
RB-8	Draper 2	11
		9
		22
RB-9	Draper 3	7
		5
		5
		10
L&R-3	Glade 1	30
	(Iva)	30
		30
		30
L&R-4	Glade 2	8
		8
		19
DB-11	Hercules 2	43
		5
RB-13	Horse Island	16
	(Iva)	24
		13

Site #	Site Location	*S. patens* units: SubsurfaceThickness (cm)
IRB-1	Indian Ldg.	14
		15
		20
DB-1	Island Field 1	55
		50
DB-2	Island Field 2	39
		34
IRB-5	Lingo Point	4
		18
LAB-7	LABay-1	10
		24
		18
DB-3	McKim 1	6
DB-4	McKim 2	11
		12
RB-11	Mills 1	28
		24
		23
		22
LAB-1	Muddy Neck 1	60
		70
		50
		50
LAB-2	Muddy Neck 2	9
		15
		36
LAB-3	Muddy Neck 3	17
LAB-5	Mulberry Landing	64
IRB-7	Murray 1	27
		19
		26
IRB-8	Murray 2	19
		11
RB-21	Nats Marsh 1	22

Site #	Site Location	*S. patens* units: SubsurfaceThickness (cm)
IRB-17	Seawinds	32
		38
LAB-6	Strawberry L.	40
		65
		31
RB-2	Thompsons 2	17
		35
		12
		25
RB-3	Thompsons 3	21
		35
		48
RB-4	Thompsons 4	12
		12
		12
DB-6	Wall Island	55
		72
IRB-19	Wm. Derrick. - 2	2
		15
		10
		22
L&R-1	Wolfe Runne 1	35
		57

APPENDIX G

Geochronology (^{210}Pb and ^{137}Cs) Data

Boat House Cove Core

Depth	Concentrations				Activities		
	%	%		dpm/g		dpm/g	
(cm)	WC*	LOI†	DBD^	^{210}Pb	^{210}Pb±	^{137}Cs	^{137}Cs±
0-2.0	77.63	38.73	0.32	6.17	0.36	0.41	0.02
2.5-5.0	75.53	34.21	0.32	5.09	0.51	0.53	0.02
5.0-7.5	77.67	31.91	0.29	7.60	0.38	0.88	0.02
7.5-10.0	77.96	24.52	0.24	4.87	0.16	2.22	0.04
10.0-12.5	81.38	33.73	0.25	3.54	0.31	1.62	0.04
12.5-15.0	79.37	33.87	0.25	3.73	0.40	0.73	0.03
15.0-17.5	79.49	30.17	0.30	3.29	0.26	0.27	0.02
17.5-20.0	82.05	32.24	0.24	3.14	0.30	0.22	0.02
20.0-22.5	82.13	31.53	0.25	2.01	0.40	0.16	0.02
22.5-25.0	81.47	29.32	0.28	1.00	0.22	0.06	0.01
25.0-27.5	81.74	29.64	0.22	1.11	0.22	0.10	0.01
27.5-30.0	76.28	24.4	0.32	1.32	0.21	0.09	0.01
30.0-32.5	75.19	25.44	0.34	nd	nd	0.05	0.01
40.0-42.5	66.53	18.41	0.53	0.25	0.10	0.00	0.00
50.0-52.5	86.41	55.74	0.17	0.28	0.13	0.00	0.00

Errors quoted are 1 sigma counting statistics
* water content
† loss on ignition
^ dry bulk density
nd not determined

Delaware Wildlands Sediment Core

Depth	Concentrations				Activities		
	%	%		dpm/g		dpm/g	
(cm)	WC*	LOI†	DBD^	^{210}Pb	^{210}Pb±	^{137}Cs	^{137}Cs±
0-2.5	73.15	50.73	0.17	6.22	0.26	0.43	0.02
2.5-5.0	74.54	24.49	0.28	6.93	0.24	1.38	0.03
5.0-7.5	77.99	38.74	0.20	4.17	0.13	3.34	0.05
7.5-10.0	81.03	37.41	0.17	2.33	0.17	3.51	0.05
10.0-12.5	83.56	24.70	0.14	2.86	0.15	1.23	0.03
12.5-15.0	83.44	25.54	0.15	6.98	0.27	0.53	0.02
15.0-17.5	81.05	27.01	0.18	7.97	0.28	0.55	0.02
17.5-20.0	80.71	16.83	0.17	5.71	0.18	0.40	0.02
20.0-22.5	80.25	37.72	0.18	4.27	0.26	0.40	0.01
22.5-25.0	78.71	33.02	0.17	2.86	0.17	0.28	0.01
25.0-27.5	74.62	33.42	0.29	1.29	0.08	0.21	0.01
27.5-30.0	77.12	18.77	0.26	1.02	0.09	0.16	0.01
37.5-40.0	72.06	29.76	0.32	0.49	0.49	0.08	0.01
47.5-50.0	78.91	24.64	0.18	0.46	0.08	0.04	0.01

Errors quoted are 1 sigma counting statistics

* water content

† loss on ignition

^ dry bulk density

Oyster Neck Road Sediment Core

Depth	Concentrations				Activities		
	%	%		dpm/g		dpm/g	
(cm)	WC*	LOI†	DBD^	^{210}Pb	^{210}Pb±	^{137}Cs	^{137}Cs±
0-2.5	80.41	56.24	0.15	7.25	0.27	0.21	0.02
2.5-5.0	80.79	64.63	0.17	3.66	0.17	0.23	0.01
5.0-7.5	81.71	64.54	0.18	3.82	0.14	0.38	0.01
7.5-10.0	80.24	53.16	0.18	3.17	0.15	1.41	0.03
10.0-12.5	77.48	40.93	0.23	3.83	0.19	2.81	0.05
12.5-15.0	70.45	32.56	0.31	4.07	0.19	1.17	0.03
15.0-17.5	66.92	24.00	0.36	2.88	0.17	0.81	0.02
17.5-20.0	73.79	30.55	0.25	2.01	0.16	0.61	0.02
20.0-22.5	76.71	32.11	0.22	1.98	0.11	0.51	0.01
22.5-25.0	79.14	32.39	0.22	1.74	0.12	0.38	0.01
25.0-27.5	76.11	28.68	0.24	1.88	0.14	0.32	0.01
27.5-30.0	73.39	27.54	0.33	1.54	0.14	0.23	0.01
30.0-32.5	71.82	22.45	0.38	0.67	0.12	0.15	0.01
40.0-42.5	66.28	12.25	0.40	0.11	0.16	0.05	0.01
47.5-50.0	72.91	20.21	0.26	0.22	0.10	0.02	0.01

Erros quoted are 1 sigma counting statistics.

* water content

† loss on ignition

^ dry bulk density

Pot Nets North Sediment Core

Depth	Concentrations			Activities			
	%	%		dpm/g		dpm/g	
(cm)	WC*	LOI†	DBD^	^{210}Pb	^{210}Pb±	^{137}Cs	^{137}Cs±
0-2.5	72.34	30.26	0.30	7.62	0.25	0.42	0.02
2.5-5.0	73.51	37.34	0.25	5.67	0.24	1.23	0.03
5.0-7.5	78.19	43.70	0.22	4.89	0.22	2.11	0.04
7.5-10.0	80.37	45.59	0.15	5.17	0.32	0.69	0.02
10.0-12.5	82.03	49.63	0.15	4.93	0.16	0.43	0.02
12.5-15.0	83.07	50.13	0.15	2.83	0.16	0.38	0.01
15.0-17.5	81.02	44.01	0.16	3.02	0.12	0.36	0.01
17.5-20.0	77.51	37.68	0.22	1.60	0.12	0.23	0.01
20.0-22.5	77.13	36.50	0.25	0.68	0.10	0.19	0.01
22.5-25.0	76.35	34.81	0.22	0.80	0.11	0.11	0.01
25.0-27.5	75.82	36.57	0.27	0.35	0.09	0.09	0.01
27.5-30.0	76.64	37.28	0.25	0.33	0.08	0.08	0.01
30.0-32.5	76.20	35.94	0.25	0.20	0.07	0.01	0.01
40.0-42.5	84.30	52.32	0.20	0.25	0.08	0.05	0.01
47.5-50.0	82.39	49.92	0.13	0.15	0.10	0.02	0.00

Errors quoted are 1 sigma counting statistics.
* water content
† loss on ignition
^ dry bulk density

Wolfe Runne Sediment Core

Depth	Concentrations				Activities		
	%	%		dpm/g		dpm/g	
(cm)	WC*	LOI†	DBD^	^{210}Pb	^{210}Pb±	^{137}Cs	^{137}Cs±
0-2.5	87.17	71.23	0.09	13.23	0.31	0.17	0.01
2.5-5.0	86.80	78.65	0.13	10.28	0.31	0.13	0.02
5.0-7.5	85.62	71.59	0.15	9.50	0.24	0.90	0.03
7.5-10.0	83.93	66.21	0.15	8.10	0.27	3.30	0.07
10.0-12.5	84.14	64.15	0.13	7.67	0.27	2.90	0.07
12.5-15.0	85.45	66.17	0.12	7.33	0.23	1.53	0.04
15.0-17.5	85.26	59.30	0.13	6.84	0.23	0.97	0.03
17.5-20.0	84.64	52.20	0.13	3.41	0.11	0.86	0.02
20.0-22.5	84.66	48.18	0.14	2.01	0.08	0.54	0.01
22.5-25.0	84.85	46.35	0.12	1.59	0.08	0.37	0.01
25.0-27.5	84.98	48.85	0.14	1.52	0.08	0.33	0.01
27.5-30.0	84.74	43.88	0.10	0.94	0.06	0.28	0.01
30.0-32.5	84.95	44.43	0.19	0.72	0.11	0.23	0.01
40.0-42.5	65.72	18.09	0.49	0.11	0.06	0.06	0.01
47.5-50.0	77.72	33.29	0.28	0.32	0.08	0.12	0.01

Errors quoted are 1 sigma counting statistics.

* water content

† loss on ignition

^ dry bulk density

Wolfe Runne Sediment Core

Depth	Concentrations			Activities			
	%	%		dpm/g		dpm/g	
(cm)	WC*	LOI†	DBD^	^{210}Pb	^{210}Pb±	^{137}Cs	^{137}Cs±
0-2.5	87.17	71.23	0.09	13.23	0.31	0.17	0.01
2.5-5.0	86.80	78.65	0.13	10.28	0.31	0.13	0.02
5.0-7.5	85.62	71.59	0.15	9.50	0.24	0.90	0.03
7.5-10.0	83.93	66.21	0.15	8.10	0.27	3.30	0.07
10.0-12.5	84.14	64.15	0.13	7.67	0.27	2.90	0.07
12.5-15.0	85.45	66.17	0.12	7.33	0.23	1.53	0.04
15.0-17.5	85.26	59.30	0.13	6.84	0.23	0.97	0.03
17.5-20.0	84.64	52.20	0.13	3.41	0.11	0.86	0.02
20.0-22.5	84.66	48.18	0.14	2.01	0.08	0.54	0.01
22.5-25.0	84.85	46.35	0.12	1.59	0.08	0.37	0.01
25.0-27.5	84.98	48.85	0.14	1.52	0.08	0.33	0.01
27.5-30.0	84.74	43.88	0.10	0.94	0.06	0.28	0.01
30.0-32.5	84.95	44.43	0.19	0.72	0.11	0.23	0.01
40.0-42.5	65.72	18.09	0.49	0.11	0.06	0.06	0.01
47.5-50.0	77.72	33.29	0.28	0.32	0.08	0.12	0.01

Errors quoted are 1 sigma counting statistics.

* water content

† loss on ignition

^ dry bulk density