SEDIMENTATION AND FALLOUT CESIUM-137 CYCLING IN A VIRGINIA SALT MARSH

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ABSTRACT

The sediment characteristics and ^{137}Cs profiles were examined at eleven field sites in a salt marsh area near Nassawadox, Virginia. Above- and belowground biomass were sampled at selected locations. The results indicate that fallout ^{137}Cs is preferentially retained by the sediments based on the percent organic matter by weight and the frequency of tidal inundations. The cesium inventories at all locations were less than that predicted from atmospheric fallout. The variability in the cesium peaks precluded their use as a chronostratigraphic marker in the sediments at several locations. A scheme for ^{137}Cs cycling in salt marsh sediments is presented which includes the proposition that export of cesium with organic detritus is a major mechanism of reducing the inventories at the study site. The high organic content, increase in organic density with depth, and the observed root mat in the high marsh of Phillips Creek Marsh indicate that it is accreting organically rather than inorganically. The high aboveground

standing crop in the high marsh indicates that organic matter may be accumulating there due to the high productivity of the vegetation. The high productivity of the vegetation, low export of organic detritus, and infrequent tidal inundations favor cesium retention.

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INTRODUCTION

Rates of sedimentation in salt marshes have been investigated using various methods from measuring the vertical accumulation over a natural or artificial marker horizon (including radioisotopic methods) to using palynological, radiocarbon or stratigraphic methods. The latter give only historical averages involving many meters of sediment. Radioisotopes such as ^{137}Cs and ^{210}Pb have been used to provide higher resolution or to document rates of accumulation within the upper few centimeters of sediment. The rates of accretion, methods used, and time intervals involved for Atlantic and Gulf coasts marshes are presented in Table I.

The accretion rates determined by the radioisotopic methods, ^{137}Cs and ^{210}Pb, in Table I show some consistency with other methods such as particulate or stratigraphic. All three methods (^{210}Pb-being the radioisotopic method) were used in Delaware with very similar results, suggesting that ^{210}Pb is a reliable indicator in salt marshes. Furthermore, McCaffrey (1977) found that ^{210}Pb

Table I. Summary of Salt Marsh Accretion (after Hatton, 1983)

STATE	ENVIRONMENT	ACCRETION RATE (MM/YR)	MEAN SEA LEVEL RISE (MM/YR)	METHOD*	TIME INTERVAL	SOURCE
Mass.	*S. alterniflora*					
	colonizing	51.8	3.4	S	6 yr	Redfield, 1972
	below MHW	2.7			625 yr	
	high marsh	18.3			100 yr	
	high marsh	1.5-2.1			950-1250 yr	
Conn.	*S. alterniflora*	8-10	2.6	P		Bloom, 1967 in Richard, 1978
	S. patens	1-4				
Conn.	S. patens	2-6.6		P	10 yr	Harrison & Bloom, 1977
Conn.	S. patens	1-2		^{210}Pb		McCaffrey, 1977
N.Y.	Mudflats	20.5-45.5	2.9	P	16 months	Richard, 1978
	Mudflats	3.4		H	173 yr	
	S. alterniflora colonizing	9.5-37.0		P	16 months	
	S. alterniflora established	2.0-4.25		P	16 months	
N.Y.	*S. alterniflora* established	2.5		H	171 yr	Flessa et al., 1977
N.Y.	*S. alterniflora* established	4.7-6.3		^{210}Pb	100 yr	Armentano & Woodwell, 1975
N.Y.	*S. alterniflora* established	4.0		^{210}Pb	88	Muzyka, 1976
Del.	*S. alterniflora*	5.1-6.3	3.8	P		Stearns & McCreary, 1957 in Hatton, 1983
Del.	*S. alterniflora* inland	5.0		^{210}Pb		Lord, 1980 in Hatton, 1983
Del.		5.0	3.0	S	-65 yr	Stumpf, 1983
Md.		1.4-6.7	4.0	pollen		Ward et al., 1987
		> above		^{137}Cs		
		1.4-6.7		^{210}Pb		
N.C.	*Juncus roe.* (200m inland)	~2		^{210}Pb		Benninger et al., 1988
		~2		^{137}Cs		
S.C.	*S. alterniflora* (10-15 m inland)	1.3	-3.0	^{137}Cs	24 yr	Sharma et al., 1987
		2.5		^{137}Cs		
		1.6		^{210}Pb		
		2.4		^{210}Pb		
Ga.	*S. alterniflora*	3-5		^{137}Cs		Hopkinson in Hatton, 1983
Ga.	*S. alterniflora* streamside	9		P	2 yr	Letzsch & Frey, 1980
	levee	6				
	S. patens	2				
	ponded water	2.8				
	high marsh	0.5				
La.	*S. alterniflora* Deltaic Plain streamside	13.5	9.2	^{137}Cs	15 yr	Delaune et al., 1978
	inland	5.9-9.4				
La.	*S. alterniflora* Deltaic Plain streamside	15.2				
	inland	9.1		P	4 yr	Baumann, 1980
La.	*S. alterniflora* Backbarrier	5.5		^{137}Cs		Delaune et al., 1986
		7.8		^{137}Cs		
La.	*S. alterniflora* Chenier Plain	4.4-8.2		^{137}Cs		Delaune unpubl in Hatton, 1983

* P-visual particulate, H-historic record, S-stratigraphy

inventories were within 20% of each other and with direct measurements in total precipitation in nearby New Haven, Connecticut. In Maryland, sedimentation rates based on ^{210}Pb were found to correlate with pollen chronostratigraphic markers.

There is less evidence on which to evaluate ^{137}Cs as a marker in the sediments. The short time interval of four years for the particulate method used in Louisiana by Baumann (1980) may under- or overestimate the sedimentation rate, and therefore the rate determined using ^{137}Cs may be similarly misproportioned. Sharma et al. (1987) found that ^{137}Cs correlated with ^{210}Pb at some specific locations in South Carolina. Ward et al. (1987) found that cesium overestimated the rate of sedimentation in Chesapeake Bay, Maryland. Benninger et al. (1988) found that although ^{210}Pb showed the inventory expected from atmospheric fallout, ^{137}Cs was at a deficit in the sediments. Casey et al., 1986 found that the measured inventories of ^{137}Cs were roughly half that expected from records of atmospheric fallout at the Palus Crisium site in Chincoteague Bay, Virginia. They suggest that ^{137}Cs should not be used without considering mobility in pore fluids. If cesium is sufficiently mobile to change the position of the peak, then the accretion rates predicted by cesium and presented in Table I may be incorrect. Cesium mobility and inventories are considered in this study because this method has been used more frequently in recent years to determine accretion rates yet has not been rigorously tested as an appropriate method for salt marsh sediments.

This study evaluates the ^{137}Cs method of determining accretion rates in salt marsh sediments by

1) reviewing the transport of ^{137}Cs in lake and salt marsh sediments, along with the mechanisms behind its sorption and desorption.

2) evaluating the cesium inventories and depth profiles along two transects encompassing eleven locations in the salt marsh sediments of Virginia.

3) comparing the cesium inventories in this study with those from other salt marshes.

The study site is located on the mainland of Delmarva Peninsula and behind Hog Island Bay (Figure 1). The eleven locations are shown in Figure 2: locations 1 through 7 traverse Phillips Creek Marsh beginning with location 1 at the creek bank and ending with location 7 in the high marsh. A second transect established in No Egrets Marsh also runs perpendicular to the creek bank and encompasses locations 8 through 11. The problem of potential cesium mobility in the

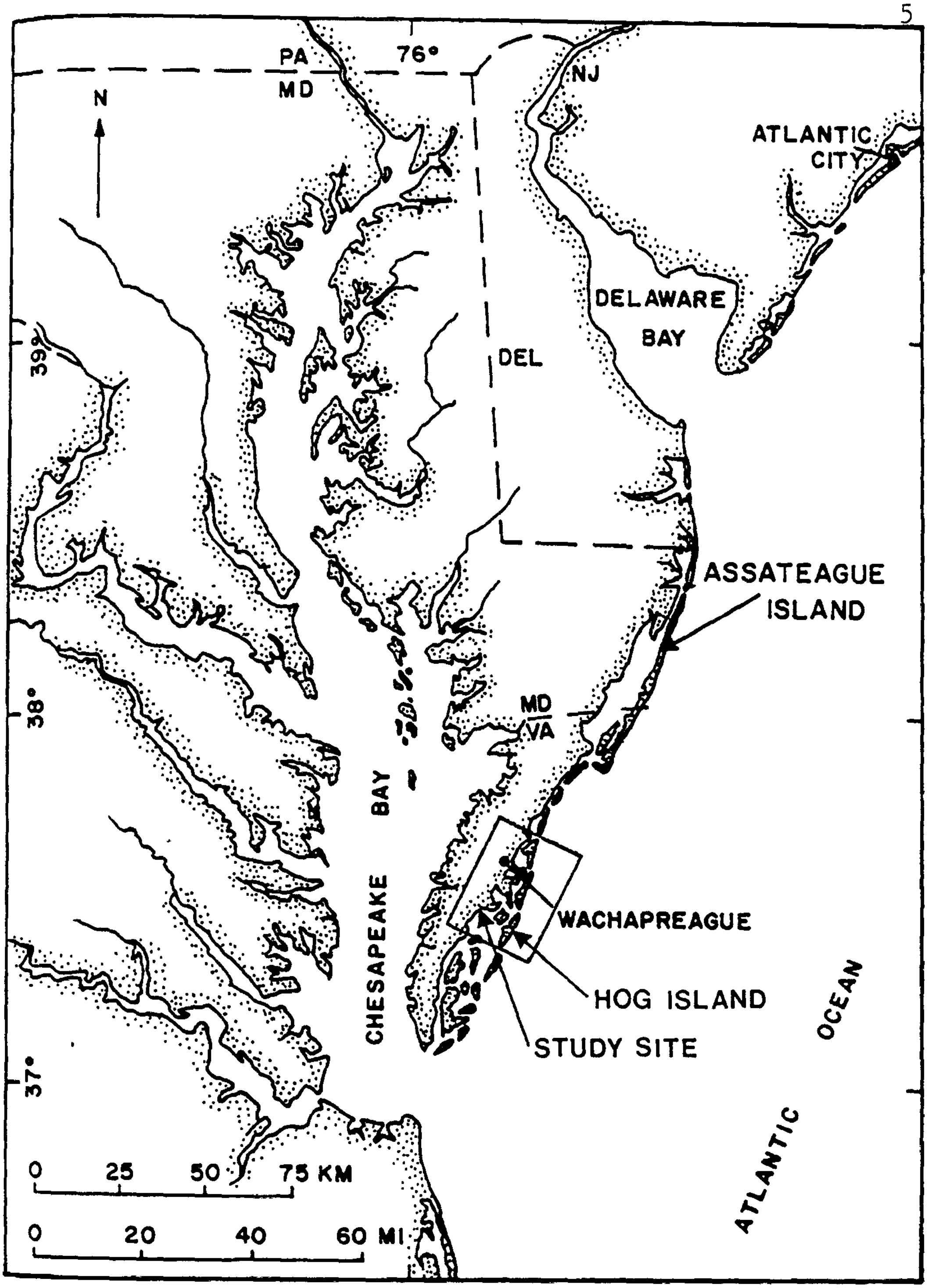
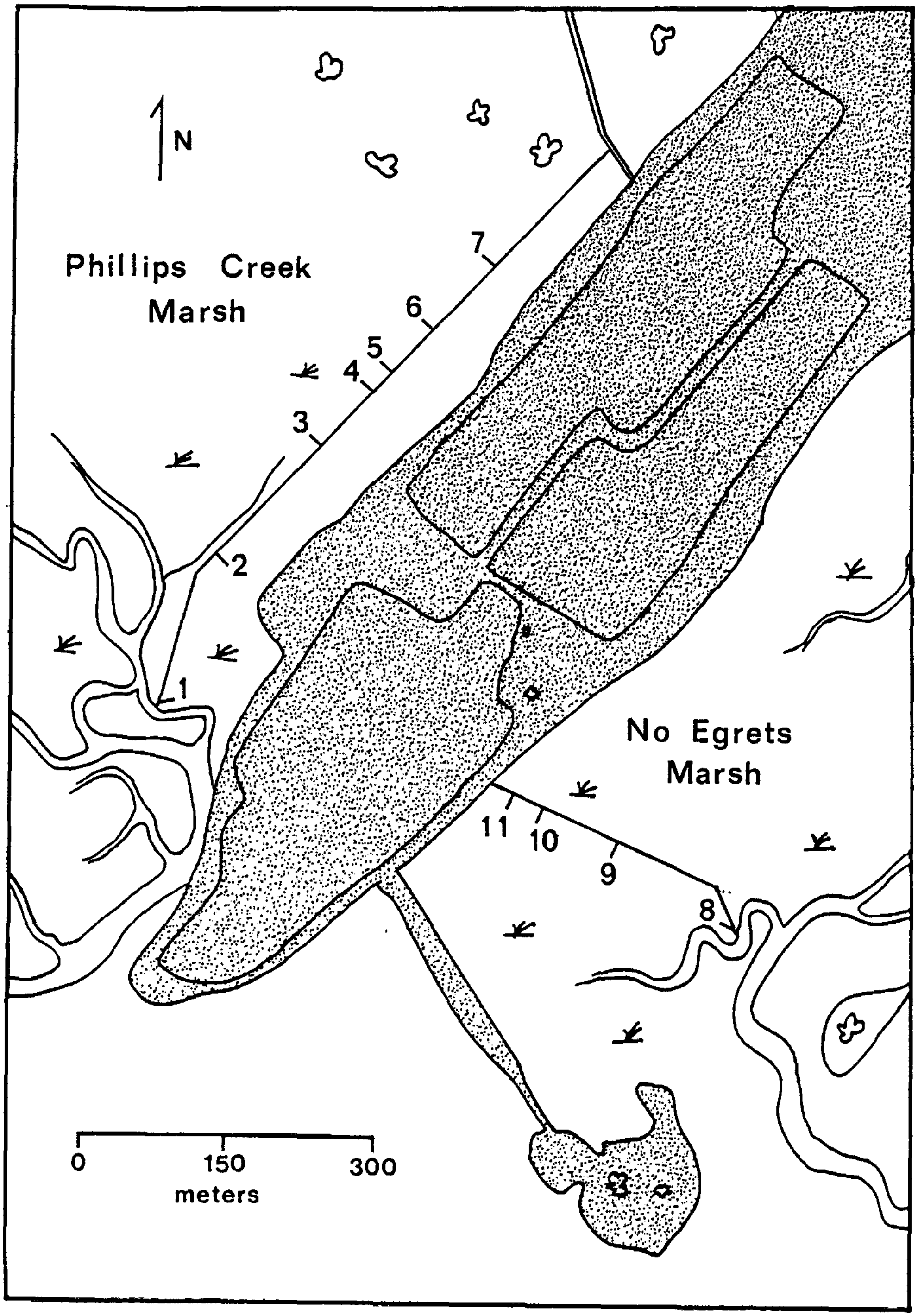


Figure 1 General location map showing the study area behind Hog Island.

427038E, 4146487N, 18N.



426076E, 4145096N, 18N.

Figure 2 Specific location map of the study area which shows the transect in Phillips Creek Marsh with sites 1 through 7 and the transect in No Egrets Marsh with sites 8 through 11.

sediments was anticipated from the onset of the study. Therefore, two sets of objectives were developed. The first and original set of objectives was based on the assumption of relative immobility of the cesium and the determination of accretion rates at each site. The second set of objectives was based on the mobility of cesium. Based on the evidence of cesium mobility presented in the Results section of this thesis and the cation inputs favoring cesium exchange in salt marsh sediments, a second set of objectives was developed and found to be more appropriate:

Objectives Pertaining to Cesium

1) provide evidence for the mobility of cesium and explain the spatial distributions in each marsh.

2) relate cesium mobility and retention to the sediment properties and physical processes occurring in the marshes.

3) examine the role of vegetation in cesium transport.

Rather than determining rates of accretion using ^{137}Cs, the process of marsh accretion was investigated by examining soil properties (percent water, inorganic and organic content by weight and bulk density) and standing crop biomass along each transect. The remaining objectives are listed below followed by further discussion of marsh accretion.

Objectives Pertaining to Marsh Accretion

1) determine trends in soil properties and vegetation and relate these trends to the accretion process.

2) determine if the inorganic component of sedimentation by weight increases toward the creek banks while the organic component by weight decreases.

3) compare spatial changes in vegetation (above- and belowground biomass) with sediment characteristics.

Marsh Accretion

The surface of a marsh is maintained with respect to sea level through sediment accumulation, which involves both inorganic and organic components of sedimentation. The rates of vertical accretion vary spatially within a marsh depending primarily on factors related to the tidal gradient (frequency, depth, and duration of flooding) and the position of major species of vegetation. The tidal gradient affects the rate of inorganic sedimentation because as tidal submergence time decreases from low to high marsh, the overlying water column becomes shallower, and the sediment supply decreases (Richard, 1978). The result is an inorganic depositional pattern that increases toward creek bank areas (Delaune et al., 1983; Hatton et al., 1983; Letzsch and Frey, 1980).

The organic rate of sedimentation also varies because substrate characteristics that influence production and decomposition vary along the gradient. Factors influencing both production and decomposition are reviewed later in this study.

The data in Table I support the trend that deposition in low-marsh habitats exceeds that in high-marsh habitats. In a short-term study (16 months), Richard, 1978, found that colonizing *Spartina alterniflora* areas accrete at a much higher rate than areas of established *S. alterniflora*. He also concluded that mud-flat sedimentation rates vary greatly from year to year based on his long-term rate of 3.4 mm/yr versus the short-term rate of 20.5-45.5 mm/yr. Several authors (Delaune et al., 1978; Baumann, 1980; Letzsch and Frey, 1980) report that streamside rates are many times higher than inland rates: 2-18 times higher depending partially on the time interval involved. Bloom (1967) recorded a range of 8-10 mm/yr in *S. alterniflora* and 1-4 mm/yr in the *Spartina patens* zone which lay above mean high water. According to Redfield, 1972, accretion will be rapid along the margins of the marsh where levees are formed; however, as the margins increase in elevation the accretion rate will diminish providing the inland areas an opportunity to accrete at a faster rate. On a fairly long time interval, the rates of accretion will be comparable throughout the marsh as the nearly horizontal level of the surface is maintained. The data from New York (Table I) provide evidence that this may be the case as the mudflat accretion rate is 3.4 mm/yr (173 yr) which is close to the rate of 2.5 for established _Spartina_ (171 yr). The depositional trend of higher accretion rates initially in the low marsh habitats coincides with the pattern of inorganic deposition which also increases toward open water. This suggests that organic sedimentation plays a primary role in maintaining the elevation of high marsh areas.

A trend which is not apparent from Table I is the relationship between relative sea level changes and sedimentation. Given a sufficient sediment supply, accretion will increase along with the rise in mean sea level. The marsh system will migrate upward and landward due to an increase in the frequency, depth, and duration of tidal flooding. If sea level rises slowly enough for sedimentation and peat formation to keep pace, the marsh can maintain its position at the seaward edge and expand inland (Titus et al., 1984). McCaffrey (1977) found the age/depth relation determined by ^{210}Pb to be in general agreement with an independent measure of sea level rise recorded by the New York City tide gauge. The sedimentation rate of 5 mm/yr since 1917 recorded by peat stratigraphy is slightly in excess of the 3 mm/yr rate determined by NOAA tidal records for Lewes, Delaware. Redfield (1972) found that vertical accretion of the high marsh in Massachusetts was somewhat greater than the rise in sea level, exceeding it by as much as 152.4 mm in 1000 years (0.15 mm/yr). The elevated sedimentation rates found in the deltaic plain of Louisiana are due to rapid relative sea level changes induced by land subsidence; however, the sediment supply is not adequate to keep pace and wetlands are being lost (Delaune et al., 1986). The literature on salt marsh accretion suggests that it will depend primarily on the elevation of the marsh with respect to sea level, the availability of sediment, and the distance from open water. Furthermore, the rates will be variable particularly at the margins. This paper investigates sedimentation and ^{137}Cs cycling in two Virginia salt marshes.

THE TRANSPORT OF ^{137}Cs

A common technique used to date lake and salt marsh cores involves use of the peak in the ^{137}Cs concentration as a marker of the 1963 peak fallout deposition from tests of nuclear weapons (Miller and Heit, 1986). Sediments deposited during fallout deposition are tagged by their high counts of gamma rays which are products of the radioactive decay of ^{137}Cs (half-life=30 years) to ^{137}Ba. The usual assumptions made in using this technique are that: 1) the sedimentation of the radionuclide at the coring site has been in constant proportion to its loading by fallout; 2) the radionuclide is not delayed for a significant period prior to its sedimentation; 3) it is not actively concentrated by organic-rich or other chemically distinctive layers in the sediment; and 4) its mobility by mechanical mixing including bioturbation or molecular diffusion is negligible [Davis et al., 1984]. Recent literature (Robbins et al., 1979; Davis et al., 1984; Miller and Heit, 1986; Sharma et al., 1987) suggests that post-depositional nuclide migration should be considered prior to using this technique for dating. This section first considers ^{137}Cs profiles for lakes and salt marshes and then addresses the processes that affect cesium distribution in the sediments.

^{137}Cs in Lake Sediments

Sedimentation rates for many lakes have been determined using ^{137}Cs in conjunction with the naturally occurring radionuclide ^{210}Pb (half-life=22 years). Although there are studies in which the chronologies based on ^{137}Cs and ^{210}Pb were essentially in agreement (Robbins and Edgington, 1975; Pennington et al., 1973, 1976; Durham and Joshi, 1980), there are other studies which suggest post-depositional migration of ^{137}Cs (Alberts et al., 1979; Dominik et al., 1981; Johnston et al., 1982; Davis et al., 1984). In the first three studies indicating migration of ^{137}Cs, the 1963 peak was too close to the top of the sediment. Johnston et al. found that although ^{210}Pb chronologies agreed with erosional chronostratigraphic markers, the ^{137}Cs date was approximately seven years too recent (Davis et al., 1984). They postulated that the discrepancy was due to delayed transport of detritus to the site of deposition. Dominik et al. concluded that if the ^{210}Pb studies were correct, then the distribution of ^{137}Cs did not reflect the history of atmospheric fallout in the central lake area of Lake Constance, Central Europe. ^{137}Cs was found limited to the upper 2 cm of sediments in this area. The ^{137}Cs distribution could not be explained by mixing (physical or biological), since sediment lamination was well preserved, or by simple molecular diffusion since no evidence for advection of groundwater into the lake could be found. ^{137}Cesium distribution with depth in the eastern part of Lake Constance was found to be in better agreement with the fallout history; However, they interpreted the activity as extending below the 1954-marker which could indicate mixing or slow molecular diffusion.

Davis et al. (1984) studied soft water lakes in New England and Scandinavia, some of which are in granitic terrain and lack clay minerals. The cesium was thought to be entering primarily by organic detritus. The sediment profiles for ^{137}Cs disagreed with expectations based on ^{137}Cs fallout history and with the chronology based on ^{210}Pb and pollen markers. In 30 of the 51 lakes, ^{137}Cs was found at depths dating several decades to a century or more prior to the nuclear weapons testing according to ^{210}Pb dating. Pollen chronostratigraphic markers in the New England lakes also pointed to an anomalously deep distribution of cesium in the sediment. The cesium maximum was however at or near the surface. Davis et al. attributed the ^{137}Cs at pre-fallout depths as due to downward molecular diffusion and adsorption and the surface concentration to net upward diffusion and adsorption, recycling in the water column, and delayed input from the watershed. No effort was made to more clearly define the process(es) of transport. Robbins (1982) concluded that in general, cesium profiles in Great Lakes sediments do not reflect the history of atmospheric deposition.

The studies in which ^{137}Cs was successfully applied to geochronologies show some mobility as well. Robbins and Edgington (1975) observed low surficial activities of ^{210}Pb at several locations and the appearance of ^{137}Cs at greater depths in the core than would be predicted from the inferred sedimentation rate alone. They accounted for this by post-depositional redistribution by physical mixing or biological processes. Simple molecular diffusion was not considered a possibility as the diffusion coefficient was confirmed to be too small. Pennington et al. (1973) studied five lakes in the English Lake District and noted an elongated "tail" of the ^{137}Cs in the two shallow lakes, Esthwaite Water and Blelham Tarn. Diffusion or bottom fauna were thought to be possible explanations.

^{137}Cs in Marsh Sediments

^{137}Cs has also been used to evaluate sedimentation rates in marsh sediments (Delaune et al., 1978; Hatton et al., 1983; Casey et al., 1986; Sharma et al., 1987). Delaune et al. cored a Spartina alterniflora salt marsh taking two cores 7 meters from a stream and another two cores inland 45 meters parallel to the same stream in the Barataria Basin, Louisiana. Two cores were also taken in a shallow adjoining lake. All cores showed a distinct 1963 peak except one core at the lake site to which sediment mixing was attributed. In all four cores the ^{137}Cs activity extended below the 1954 marker (1st year of atmospheric fallout). This "tail" of activity was linked to the rooting depth of S. alterniflora and was speculated as being the result of plant absorption. Hatton et al., 1983 examined the cesium profiles from cores collected from freshwater, intermediate, brackish, and salt marshes. They found a well defined ^{137}Cs maximum in most every profile examined and a close agreement between salt marsh accretion rates using cesium and independent estimates based on a visual particulate method. They also concluded that cesium is found deeper in profiles near the water's edge due to the more rapid aggradation of sediments. The profiles attained a relatively uniform level in the cesium peak at distances exceeding approximately 40 meters. In contrast, Casey (1985) suggests that the downward flow of pore fluid near the edge of the marsh results in a deeper cesium peak. Sharma et al. (1987) cored a salt marsh in South Carolina and found that, except for creek bank sites where bioturbation is intense, the accumulation rates determined by ^{137}Cs were in agreement with those determined by ^{210}Pb. This amounted however to interpreting only two cores for sedimentation rates, although five locations were sampled.

Cesium Sorption and Desorption

The Effect of Mineralogy and Co-ions

It is known from the literature that different clay minerals sorb cesium differently (Sawhney, 1965; Tamura et al., 1960). Illite has a particularly high affinity for cesium and contains a high amount of potassium in its chemical make-up (Tamura et al., 1960). The potassium ions are accommodated between the aluminosilicate sheets. The Conasauga shale, which is very effective in removing cesium from radioactive waste solutions, is 40% illite, 25% probably vermiculite but possibly chlorite or montmorillonite, and 35% quartz.

Three types of sites are important in the binding of ^{137}Cs and help to explain the sorption/desorption of cesium by different clay minerals: 1) easily exchangeable sites on the clay surface where cesium binding is largely electrostatic and relatively non-selective. Sites on the unlayered clay mineral, kaolinite, and montmorillonite approximate this behavior. 2) fixed sites at the edges of the clay mineral lattice where ^{137}Cs is selectively fixed and is displaced only by cations of similar size and charge [K^+, NH_4^+, Rb^+, and perhaps H^+]. Sites on weathered micas such as illite exhibit this behavior. 3) non-exchangeable interlayer sites (not readily exchangeable with any cation) where cesium is entrapped during lattice collapse [Jacobs and Tamura, 1960; Sawhney, 1965, 1972]. The binding of cesium based on mineralogy is considered further in the following section on grain size changes.

In addition to the clay mineral composition of the soil, the cationic environment also affects the sorption and desorption of microquantities of cesium by soils (Schulz et al., 1960; Sawhney, 1965; Kawabata, 1967). Sawhney (1965) studied the sorption of cesium by a synthetic cation-exchange resin over a range of low cesium concentrations in the presence of different counter-ions. The exchange isotherms for the sorption of cesium by montmorillonite in the presence of H^+, K^+, Ca^{2+}, and Al^{3+} showed that cesium is sorbed in preference to H^+ and K^+. The exchange isotherms for the sorption of cesium by illite in the presence of H^+, K^+ and Ca^+ suggested that two exchange sites existed. A portion of the cesium sorbed by illite replaces interlayer potassium ions and is thus fixed. Illite preferred cesium to the three other ions. The isotherms also showed that more cesium is sorbed by illite in the presence of calcium than in the presence of potassium at cesium concentrations below 10^{-3}N. Thus the presence of potassium seems to hinder sorption but not necessarily initiate desorption. Sawhney further noted that cesium is more readily replaced by the univalent cations K^+ and NH_4^+ than by the bivalent cation Ca^{2+}.

Schulz et al. (1960) showed that microquantites of ^{137}Cs were displaced by the clay mineral kaolinite in the following order: $Na^+ \geq Ca^{2+} = H^+ = Mg^{2+} > NH_4^+ > K^+ >> Cs^+$. Illite showed a displacement pattern as follows: $NH_4^+ > K^+ = Cs^+ > H^+ > Na^+ > Ca^{2+} > Mg^{2+}$. Kawabata (1967) found that the higher the concentration of a competing cation, the more inhibitive to the uptake of the radionuclides by river sediments. The rate of inhibition depended on the cationic species and the concentration of cations. Evans et al. (1983) showed that Pond B sediments on the Savannah River Plant, South Carolina, contained 89% kaolinite, 5% illite and 6% mixed chlorite/vermiculite. Salt solutions of NH_4^+ and K^+ were the most effective in displacing ^{137}Cs releasing 15.6 and 11.7 percent respectively. They suggested that the order of displacement was influenced by several ion exchange sites of differing selectivity or that sites of intermediate character were involved.

The Effect of Grain Size Changes

Studies in various environments have shown that cesium sorbes preferentially on fine particles (Hetherington and Jefferies, 1974; Simpson et al., 1976; Wong and Moy, 1984; Carr and Blackley, 1986; Terada et al., 1986; Albertazzi et al., 1987). Most of these studies were conducted in shallow offshore waters or estuaries and the results were consistent: the finer the grain size of the sediment, the higher the concentration of ^{137}Cs per unit mass. These studies indicate that mineralogy and grain size are important in determining whether surface adsorption or incorporation into the lattice occurred.

Hetherington and Jefferies analyzed the surface sediment layers from estuarine mud banks near the Windscale nuclear fuel reprocessing plant in Cumberland, England. They normalized the activity in each fraction (<2μm-clay, 2-63μm-silt, 63-125μm-very fine sand) to that in their coarsest fraction (125-250μm-fine sand) and found that the specific activity of each isotope, including ^{137}Cs, increases as particle size decreases. The clay fraction, although constituting only 15% of the weight of the sample, contained 40% of the radionuclide. In the sand and silt fractions (4-90μm), the concentration of ^{137}Cs (pCi/g dry wt) decreased linerally when plotted with mean particle diameter. The authors interpreted this result as suggestive that uptake is a surface adsorption process for the sand and silt fractions.

Carr and Blackley, 1986, found that ^{137}Cs is strongly correlated with the proportion of fines at each level in the core. They took the natural log of the activity and the percentage of sediment <2μm, <8μm, and <63μm and found a correlation coefficient for each line of 0.95. The correlation coefficient decreased when sediment of size >63μm was included. They concluded that the cesium was preferentially absorbed into the illite lattice since the best correlation was found with the log base e and the slope of the line or isotherm was approximately 1 (implying a relationship with volume rather than surface area). The sediment was approximately 69% illite, 10% chlorite, 13% kaolinite, and 8% illite-montmorillonite.

Terada et al., 1986, discovered that a unit amount of the finest component (<1.9μm, and 12-18% by wt) contained about three times as much ^{137}Cs as that of the coarsest one (16µm, and 42-67% by wt), in spite of lower sodium and potassium contents. The lower sodium and potassium contents indicated that the predominant clay minerals were montmorillonite or kaolinte instead of illite. Preferential sorption by the finer fraction was still observed, showing that a large number of exchange sites for cesium existed. They also performed a sorption experiment in sea water in which only 20-30% of the alkali metals were replaced by cesium for the coarsest component while in the case of the finest component almost all of the alkali metals were exchanged with cesium. The results showed that the distribution coefficients for sorption increased with a decrease in the grain size of the sediments.

Albertazzi and co-workers studied the distribution of ^{137}Cs in nearshore sediments of the northern Adriatic off the lagoon of Venice and at the estuary of the Adige River. ^{137}Cs was shown to accumulate in the finest particles (0.24-16µm range) and to be taken up by surface adsorption processes. The fine pelitic sediments were predominantly carbonates. They determined the specific surface area for coarse and fine sediments to be 2-3 m^2/g and 15-20

m^2/g, respectively.

A study of ^{137}Cs in Lake Michigan sediments showed that the isotope was constant with particle mass but not a function of the particle sizes studied (<2µm to 45µm) [Alberts and Muller,1979]. According to the latter authors, if the distribution of a nuclide is dependent on surface area, then the concentration per gram of sediment should increase dramatically with decreasing particle size since the surface area per unit mass increases with decreasing particle size.

Wong and Moy (1984) concluded that the concentrations of ^{137}Cs, metals, and organic carbon were considerably higher in the fine-grained sediments of the James River Estuary, Virginia.

Many of the papers cited above suggest that ^{137}Cs sorbes onto sediments mainly as a function of the physical properties of the sediments, particularly with respect to high surface area and finer grain size. Cesium is incorporated into the lattice of the layered clay minerals given sufficient time for diffusion. The literature also suggests that if surface adsorption predominates, the cesium concentration will be associated with sediment particle size whereas if ion-exchange into interlayer sites predominates, the cesium concentration will be associated with sediment volume. Since this isotope is preferentially attached to the clay and silt fraction, the peak activity in a cesium profile could coincide with clay mineral accumulation as noted by Brickman, 1978. A concentration of clay minerals would tend to concentrate ^{137}Cs and a deficiency would lessen the adsorbance. Several authors (Brickman, 1978; Sharma et al., 1987; and Terada et al.,1986) have used ^{40}K to interpret grain size changes and changing mineralogy. An increase in ^{40}K could be the result of a total clay mineral increase or an illite abundance increase or a combination of the two. The concentration of potassium in distinct layers will lead to higher cesium retention and the potential for misinterpretaion of the cesium peak.

The Effect of Salinity Variations

Salinity (co-ion concentrations such as Na^+, K^+, Mg^{++}) has been found to hinder sorption of ^{137}Cs and to significantly affect its desorption. Klechkovsky et al. (1959) showed that when the concentration of salt in the medium is increased, the sorption of cesium by soils decreased (Sawhney, 1964). This conclusion was also reached by Aston and Duursma (1973). Many other authors have suggested that major cations successfully compete with ^{137}Cs for sorption sites in highly saline waters (Cheng and Hamaguchi, 1968; Patel et al., 1978; Schell et al., 1980; Stanners and Aston, 1981; Casey et al., 1986).

Stanners and Aston (1981) have observed that ^{137}Cs is adsorbed by eastern Irish Sea sediments by an ion exchange reaction which exhibits a competition with other alkali metal ions especially K^+. The increased solubility of cesium in more saline environments may be due to the increased concentration of K^+ and increased competition for sites for ^{137}Cs adsorption (Larry Benninger, UNC, personal communication).

Patel et al. (1978) showed that the desorption of cesium nuclides, when expressed per unit weight, was dependent on salinity. They used sediments from Bombay Harbor which contained 32-34% illite and shook them in varying salinities for ten days. The maximum desorption (40%) occurred at normal harbor salinity (38 ppt) and the minimum (25%) at the lowest salinity (9.5 ppt). They also calculated the sedimentary half-life of cesium to be approximately two years which is much shorter than its physical radioactive half-life of 30 years. The sedimentary half-life was derived by plotting the annual average cesium content (pCi/g dry sediment) over a six-year period, 1971-1976. The slope was found to be 1.86 years. Patel et al. concluded that the cesium was being depleted at a rate greatly exceeding its natural decay.

Schell et al. (1980) measured distribution coefficients for ^{137}Cs [K_d= concentration of bound radionuclide in dry sediment (dpm/g)/ concentration of radionuclide in water (dpm/ml)] using a constant shaking technique. They found the lowest sorption distribution coefficients in marine environments and the highest values in freshwater systems. The Hudson River sediment-water system, salinity less than 2 ppt, had a K_d of approximately 1700 ml/g after reaching equilibrium. In comparison, the equilibrium K_d for the Hudson Estuary sediment-water system, salinity 15-25 ppt, was approximately 650 ml/g. The lowest K_d value for ^{137}Cs, 61 $\pm$ 24 ml/g, was found for the sediment water system of Saanich Inlet, an anoxic marine basin (Schell et al., 1980). The results show that cesium is more soluble in brackish and saline waters than in freshwater.

Casey et al. (1986), using a simple model for solute transport, concluded that the hypersalinity and low pH of the marsh sediments cause some nuclides to desorb and become transported as solutes. They measured the distribution coefficients for ^{134}Cs between a wide range of sea water solutions and marsh soil. The coefficients decreased dramatically over the salinity range 14 ppt to 270 ppt, the seasonal variation of salinity at the Palus Crisium site, from approximately 175 ml/g to 25 ml/g, respectively. They attribute the desorption to competition with saltwater cations via a reaction such as presented below:

$$Cs^{+} + K\cdot solid \rightleftharpoons Cs\cdot solid + K^{+}$$

The Effect of Decomposition/Redox Changes

The ammonium ion (NH_4^{+}) produced during the decomposition of organic matter is found in relatively high concentrations in marsh sediments. It may contribute to desorption in addition to the cations associated with salinity.

Terada et al. (1986) found that ^{137}Cs can be extracted from sediments with an ammonium nitrate solution. They suggested that although the nuclide is strongly adsorbed in the crystal lattice of some clay minerals, it can be is desorbed from the sediment phase by ammonium ions in high concentrations. Evans et al. (1983) also used ammonium nitrate as an extracting reagent and found an exponential release of cesium from sediments of primarily kaolinitic composition. At a 1 mM concentration of ammonium (commonly found in the upper 20 cm of salt marsh sediments [Casey, 1985]) approximately 4-5% of the cesium was released whereas at 100 mM, it was feasible to release 16-21%. Schulz (1965), recording the uptake of cesium by romaine lettuce plants, found that ammonium sulfate fertilization resulted in a six-fold increase in ^{137}Cs uptake. Brisbin et al. (1974) showed that 55.3% of the radiocesium was desorbed from Steel Creek soil, Barnwell County, South Carolina, (predominantly kaolinite) by 6 N HNO_3 and 24.9% by 1 N HNO_3. Lomenick and Gardiner (1965) were unable to desorb more than 1% of the radiocesium from illite clay except with 6 N HNO_3 which removed 19%.

According to Schulz (1965) potassium, rubidium and cesium, along with NH_4^+, will tend to behave in a similar fashion in soils. Cesium is bound with the highest energy of adsorption followed by rubidium. NH_4^+ and K^+ are held by the soil with about equal tenacity. Schulz, after considering the complementary ion effect, concluded that NH_4^+ or K^+ should liberate a considerable amount (20-80%) of fixed ^{137}Cs into a mobile form. The cesium can then become available to plants or be transported elsewhere in the environment. Both NH_4^+ and K^+ in solution will complete with ^{137}Cs for plant uptake. Schulz contends that NH_4^+ will be removed from competition as microbiological oxidation to NO_3^- takes place. In salt marsh sediments this occurs in the upper 10 cm or so of sediment where the ammonium is essentially consumed by chemical and biological processes (Lord and Church, 1980). However, deeper in the soil horizon NH_4^+ will substitute for ^{137}Cs and may prevent it from reverting to a fixed state.

Alberts et al. (1979) discovered that ^{137}Cs was partially remobilized from bottom sediments into the water column on an annual cycle coincident with summer thermal stratification and hypolimnetic anoxia. This occurred on the Savannah River Plant, South Carolina, from sediments of primarily kaolinitic composition. They speculated that release of cesium coincided with dissolution of ferric oxides during stratification in oxygen poor bottom water. It returned to the sediments when vertical mixing of the lake supplied enough oxygen to reprecipitate ferric oxides (Dominik et al., 1981). Such a redox reaction could occur in the salt marsh as well particularly around plant roots where oxygen diffuses into the surrounding soil and in areas of pore water drainage.

Plant/Soil Relationships

Natural processes which contribute either to decreased ^{137}Cs adsorption to soils and sediments or to its subsequent desorption can increase cesium mobility and biological availability. Radioactive elements that are readily absorbed in the soil by plants are soluble in the soil or are isotopic with elements that have metabolic functions in the plant (Menzel, 1965). The uptake of ^{137}Cs by plants is usually quantified by the use of vegetation/soil concentration ratios or transfer coefficients [both are represented by pCi/g dry tissue/ pCi/g oven dry soil]. In terrestial and freshwater ecosystems the relative concentration ratio for cesium in many soils is 0.01-1, meaning that cesium is not readily available for plant uptake due to its strong fixation on clay minerals (Menzel, 1965; Dahlman and Van Voris, 1976; Cline, 1981). However in saltwater environments, the soil fixing capacity for ^{137}Cs is greatly reduced by the elevated co-ion concentrations and, as will be discussed in this section, is affected by the presence of organic matter. This section emphasizes the role of plants in the geochemical cycling of cesium.

The Uptake of Cesium by Plants

Comparisons of the concentration ratios in various ecosystems show that plant uptake is determined by the soil fixing capacity. The concentration ratios for Oak Ridge soils, floodplain sediments highly contaminated by production reactor effluents in Tennessee, are 0.001 to 0.23 for overstory species and 0.001 to 0.53 for ground cover species (Dahlman and Van Voris, 1976). The low concentration ratios at this site are due to the illite clays which in this terrestrial ecosystem tightly bind the cesium. The highly contaminated sediments of the Savannah River Plant, South Carolina, show a bimodal distribution of cesium activity due to separation of the sediments in stream flow. The low-level soils are coarse-particle sediments with a mean radiocesium concentration of 18.1 pCi/g dry weight. The high-level soils are clay, predominantly kaolinite, and silt with a mean radiocesium concentration of 170.5 pCi/g (Sharitz et al., 1975). The concentration ratios on low-level soils were between 10.8 and 20.1 while the ratio on high-level soils was 0.6. These ratios show an increase in plant uptake on coarse-particle sediments which have fewer binding sites for the cesium. In this situation the soil fixing capacity (meq./100g) is lowered by an increase in particle size. The lower soil fixing capacity allowed greater uptake by plants. Overall, the higher proportion of kaolinite contributed to higher plant to soil ratios of ^{137}Cs over that found for Oak Ridge soils.

Cummings et al. (1969) found a significant correlation coefficient of 0.74 between the oat uptake of ^{137}Cs and the soil fixing capacity of 12 soils. Oat uptake decreased as the soil fixing capacity increased. The correlation coefficient for five Florida soils was 0.90. They explained the higher than average ^{137}Cs concentrations in the milk of the these areas as being a result of the low soil fixing capacity of the highly weathered soils.

Kuhn et al. (1984) compared forest clearings and grasslands contaminated by low concentrations of cesium from radioactive fallout. They concluded that the decreasing clay content of forest clearings over grasslands corresponded to increasing activity in the vegetation. The grassland had concentration ratios between 0.01 and 0.09 whereas the ratios for the forest clearings ranged from 0.05 to 6.2. In both ecosystems, the concentration ratios in plants increased with a decrease in the exchangeable potassium content of the soil. The highest specific activity in the vegetation of the forest clearing was 6937 ± 0.5 pCi/kg dry weight while the same value for the grassland was 238 ± 5.8 pCi/kg.

The studies reviewed here show that when the soil fixing capacity is reduced, cesium uptake by the vegetation is increased. The fixing capacity can be used to predict the extent to which cesium will be taken up by plants.

The Influence of Organic Matter on Cesium Retention

The work of Barber (1964) demonstrated a high correlation (r= 0.94) between ^{137}Cs uptake by ryegrass from pasture soils and percent soil organic matter. A significant correlation coefficient (r= 0.95) was also found for ^{137}Cs uptake and cation exchange capacity on organic matter. Barber suggested that when organic matter is responsible for a large fraction of the total exchange capacity of the soil, the adsorption and subsequent fixation of cesium onto clay minerals is greatly reduced. The cesium held in organic matter has particular significance in that it remains available for plant uptake (Menzel, 1965).

Other studies have shown that organic matter contributes to ^{137}Cs retention in soils (Brisbin et al., 1974; Ritchie et al., 1978; Mattsson, 1974). Brisbin et al. found a correlation coefficient of 0.76 for the linear relationship between radiocesium concentration and percent organic matter and 0.78 for potassium (mg/g dry wt) and radiocesium (pCi/g dry wt) at the Savannah River Plant. Ritchie et al. analyzed data from fourteen North Central states' watersheds, cultivated and non-cultivated. They concluded that some cesium was moving in the watershed and that soil organic matter was the key factor in the retention of cesium in the cultivated watersheds.

Lichens (fungi and algae) effectively accumulate and retain most of the deposited fallout products, including ^{137}Cs (Mattson, 1974). They are responsible for a high concentration of radionuclides to man via the food chain of atmospheric deposition- lichen- reindeer- man in the arctic and subarctic regions. Mattsson studied the lichen _Cladonia alpestris_ and summarized the principal ways for elimination of the cesium from the undisturbed carpets as being the physical decay of the radionuclide and the continuous growth of the lichen plants, the latter combined with a continuous decomposition of older material and grazing by reindeer.

Sanchez et al. (1988) showed that the distribution coefficient (both sorption and desorption) was significantly higher for ^{137}Cs in peat material containing less organic carbon under acidic conditions. They concluded that although other radionuclides such as Co-60 and Sr-85 may be retained by organic complexes, ^{137}Cs is more strongly associated with the inorganic component of peat in terrestrial ecosystems.

On the basis of dialysis experiments, Schell et al. (1981) concluded that organic detritus accumulates ^{137}Cs in freshwater systems but not in marine or estuarine systems.

The Effect of Organic Compounds on Cesium

Results of the studies on the effects of organic compounds show that certain organic compounds can definitely affect sediment- water K_d values (Clayton et al., 1981). Clayton et al. measured the distribution coefficients using a constant shaking technique and found that EDTA and humic acids can solubilize both ^{241}Am and ^{57}Co and that the effect was more pronounced in freshwater than in marine sediment water systems. Other organic compounds, 1-nitroso-2-naphthol, and 1,10 phenanthroline increased their affinity for sediments. However, none of the above organic compounds including acetic and salicylic acids, were found to alter the K_d's for ^{137}Cs. The acetic and salicylic acids were tested in freshwater systems only. There is little if any tendency for cesium to form complexes in natural environments. Soluble organic complexing of cesium probably has little effect on cesium mobility (Onishi et al., 1981).

Diffusion of Cesium

The geochemical behavior of cesium in sediments as reviewed in the previous sections suggests that desorption via ion exchange (particularly with K^+, NH_4^+, and Na^+) is likely to occur in salt marsh sediments. Casey (1985) addressed the potential for migration of ^{137}Cs through pore fluid in salt marsh sediments. He concluded that the peak activity could be displaced by several centimeters or removed completely from the sediment depending on the direction and velocity of flow, the burial rate of the sediment, the adsorption coefficient and the frequency with which the marsh surface is flushed by tidal waters.

Hetherington and Jefferies (1974) calculated an apparent vertical diffusion coefficient of approximately 10^{-6} cm^2/sec for ^{137}Cs in mud of the Ravenglass Estuary, Whitehaven Harbour, in the area of Eskmeals. They reported the clay as 69% illite. Duursma and Bosch (1970) derived a value of 10^{-8} cm^2/sec for sediment in the Mediterranean. Robbins et al. (1979) demonstrated that for experimental lake sediment columns, ^{137}Cs was released from labelled illite and diffused slowly with an effective molecular diffusion coefficient of $6*10^{-10}$ cm^2/yr for freshwater cores.

Seasonal oscillations in the salinity of marsh sediments may have an effect on the ^{137}Cs distribution with depth as well as its inventory at a particular location. Both dilution by rainfall and periodic desaturation of the soil will change salinity profiles (Casey and Lasaga, 1987). In times of high rainfall, the salinity of surface waters decreases and cesium at depth would be expected to diffuse to and concentrate in the surface pore waters. Conversely, when rainfall is minimal cesium would be expected to diffuse to and concentrate in the deeper pore waters (Brickman, 1978). The result over many years would be a broad curve of ^{137}Cs activity with depth and a profile that could not be used to determine a rate of sedimentation (Casey, 1985). Furthermore, salinity gradients in the marsh could affect the inventories by retaining cesium in regions that are less saline on average.

Physical Reworking-Bioturbation

Sediment mixing by benthic organisms can substantially alter the vertical profiles of sediments. The method of redistribution is dependent on the biological species (Robbins et al., 1979; Robbins, 1982; Christensen, 1982). Robbins et al. (1979) studied the effect of amphipods (Pontoporeia hoyi) on the distribution of ^{137}Cs; the amphipods exchanged adjacent parcels of sediment randomly at all depths within a 2 cm mixing zone. Their activity conformed closely to a diffusion analog. The authors' experiments on the effect of tubificid worms (who ingest sediment at depth and excrete it at the surface) resulted in peak broadening of ^{137}Cs and the temporary reappearance of radioactive material in surface sediment.

Fiddler crabs are the most likely agent of bioturbation at the site of the present study. Their effect on the distribution of ^{137}Cs has not been studied. ^{210}Pb has been modelled with respect to bioturbation by fiddler crabs but the results are not applicable to ^{137}Cs (Gardner et al., 1987). Bertness and Miller (1984), who studied crab burrowing, found that their frequency on creek bank sites was an order of magnitude greater than those in the high marsh. Their survey also suggested that *U. pugnax* burrowing in *Spartina patens* was restricted due to a heavy root mat and that burrowing in soft substratum requires support structures for burrow maintenance.

Summary

This section has defined the processes and key factors which affect ^{137}Cs mobility by examining field studies of lakes, estuaries, salt marshes, and terrestrial soils. Evidence from lake studies which have employed ^{137}Cs as a dating technique indicates that one or more assumptions are often incorrect. Most lake studies have not considered particle size distribution and transport within the watershed which have been emphasized and correlated in estuarine studies. The lake studies are important because they suggest transport of cesium by biota as well as by slow molecular diffusion. The assumption of negligible mobility by mechanical mixing including bioturbation or molecular diffusion is violated in many cases. Further difficulty has arisen in using the peak cesium concentration in lakes due to delayed transport of sediment and detritus. The method assumes that delayed transport is also negligible.

Many of the factors affecting the sorption/desorption of ^{137}Cs have been defined in the field studies of estuaries, salt marshes, and terrestrial ecosystems:

The sorption of ^{137}Cs:

1) decreases on a weight basis with an increase in grain size of the sediment component.

2) decreases with increasing salinity or elevated concentrations of co-ions.

3) varies with the mineralogy and the soil fixing capacity.

4) does not change in the presence of many organic ligands but tends to be increased in the presence of organic matter in freshwater and terrestrial ecosystems.

5) will be strongly influenced by the amount and species of complementary ions particularly, NH_4^+, K^+, and Na^+, which can desorb the cesium.

6) onto the soil and its subsequent desorption will have a strong influence on the redistribution of cesium in the soil profile and will affect plant uptake.

Combining the possible mechanisms of ^{137}Cs transport suggested from these studies, the following mechanisms need to be considered for marshes:

1) upward or downward diffusion in interstitial waters including potential desorption by ammonium ions and by conditions of hypersalinity

2) adsorption by plants and export of the cesium with organic detritus

3) bioturbation of sediments by benthic infauna and mixing of sediments by physical reworking (slumping)

Diffusion in interstitial waters is a serious contention because it suggests that ^{137}Cs could be transported significant distances via pore water fluxes during evapotranspiration and infiltration during inundations by tides. Desorption of cesium from sediments suggests potential readsorption by plants and potential export from the marsh through the removal of organic detritus or chemical precipitation such as that proposed by Alberts et al. (1979). Using the cesium peak concentration in the sediment may provide valid results if the affect of remobilizing processes are negligible. However, many factors contributing to cesium mobility in salt marshes are present. Therefore this method should be used with scrutiny and the affect of each process evaluated at each site.

SEDIMENT TRANSPORT AND DEPOSITION IN MARSH SYSTEMS

Sediment transport in a marsh drainage system occurs primarily through the action of tidally-induced open channel flow. Clay-size grains compose a large proportion of tidal sediments such that transport processes involve primarily cohesive sediments (clay) rather than cohesiveless (sand and silts) sediments and transport in suspension rather than as bedload. In this section, sediment dynamics and studies on the distribution and aggregation of sediments is presented first, followed by sediment transport results from tidal flux studies.

Sediment Dynamics

As the water level in the channel reaches bank full stage and spills over onto the marsh surface the cross-section of flow is greatly increased and consequently the velocities drop dramatically. The dissipation of energy (particularly kinetic energy since it is a function of velocity squared) due to decreased velocities favors deposition. Stumpf (1983) found that of the sediment suspended in the water flooding the marsh surface, more than 80% settles within 12 meters of the creek. Deposition is further enhanced by the vegetation which increases sedimentation by reducing flow and turbulence. Deposition takes place during slack tide (high and low tide) as shear velocities drop to zero and the rate of deposition is dependent on the suspended sediment concentration and the settling velocities. However, deposition also occurs at velocities above zero as sediments are trapped within a thin layer of viscous flow close to the bed (Settlemyre and Gardner, 1977; Allen, 1985). Since velocities on the marsh are an order of magnitude less than in the channel, the resistance of deposited mud to erosion is even higher; Erosion on the marsh surface therefore usually occurs as mass failure or slumping.

Sediment Size Distribution and Aggregation

The channels of tidal flat complexes have a consistent pattern of grain size distribution: The sediment decreases in grain size from sand-sized in the largest through-flowing channels to silt- and clay-sized in the smallest dead end channels (Zeff, 1988; Harrison, 1972). Following this sediment trend is an increase in the total organic matter content toward the lower order channels. This is a result of the organic supply from the marsh surface. The sediment trend for the marsh levee and back levee continues from the channels and is progressively finer as well (Zeff, 1988). This means that the back levee and marsh interior are the most removed from allochthonous sediment sources and should have the finer size fraction of sediment deposited from suspension. According to Zeff (1988), sand reaches the high marsh only during storms events.

Information on the sediment size distributions from marsh field studies is scant. A trend toward progressively finer sediment is indicated in the data of Stumpf (1983): The levee deposits had an 86% inorganic content by weight and consisted of 32% coarse silt (>16um) and 42% clay (<4um) whereas the backmarsh had an 80% inorganic content and consisted of 25% coarse silt and 47% clay. The percentages of fine-silt were not noted. In the backmarsh, the material larger than clay size (>4um) had a higher organic content associated with it (53% compared to 37% for the clay sized material). A decrease in grain-size toward the backmarsh was not the trend noted by Edwards and Frey (1977) for salt marsh sediments in Sapelo Island, Georgia. As Figure 3 shows, the substrate was composed of approximately equal parts of silt and clay-size sediment at the creek banks with a low sand content whereas the high marsh sediments were predominantly clean to slightly muddy sands. The lack of silt or clay in the high marsh sands suggests that they were pre-existing sediment and that material deposited from tidal suspensate pertained only to the transitional to creek bank areas. The higher sand content in the transitional marsh was most likely due to erosion of the high marsh sands and deposition on the ebb tide.

Sediment aggradation is the principal mechanism by which clay-size grains are transported to the marsh surface under fair weather conditions (Carson et al., 1988) and supports to some extent the accumulation of finer material toward the backmarsh. Most of the clay particles coagulate to form flocs during interparticle collision and cohesion; The flocs have higher settling velocities than individual particles and accelerate the deposition of fine-grained sediments. Organic flocculation (ingestion of clay grains by organisms who utilize the adhering organic material and excrete the clay as fecal pellets) takes place (Pethick, 1984; Hayter and Mehta, 1986) as well as the formation of organic-mineral aggregates (Carson et

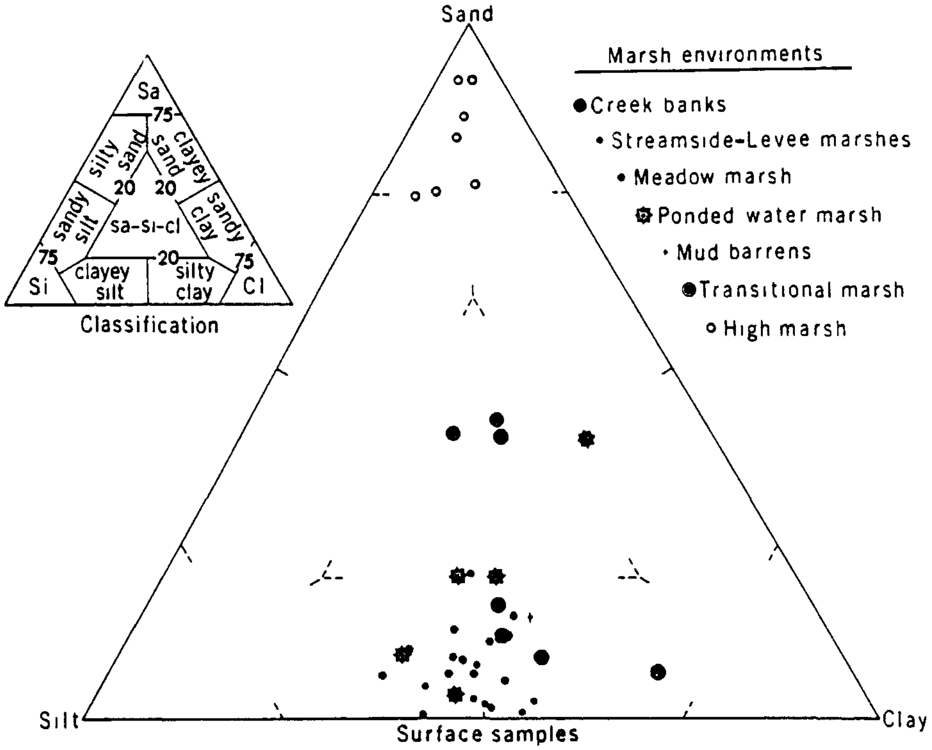


Figure 3 The proportions of sand, silt and clay in salt marsh sediments, Sapelo Island, Georgia (From Frey and Basan, 1978).

al., 1988; Harrison, 1972).

Sediment aggradation has been studied in detail in two areas: the tidal flat complex of Delmarva Peninsula, Virginia, and in Great Sound, southern New Jersey (Harrison, 1972; Carson et al., 1988). Both studies show that fecal pellets (bioaggregates) are found in maximum abundance in the channels and bays and to a more limited extent in the tidal flats. The inorganic content of the fecal pellets in the above study areas consists of quartz, feldspar, and heavy minerals in a wide variety of particle sizes. Harrison (1972) noted the presence of fecal pellets on the blades of Spartina grasses as well as an even greater abundance of organic-mineral aggregates. The organic-mineral aggregates differ from the fecal pellets in that they are composed almost exclusively of grains smaller than 6 μm and have lower settling velocities and densities: the settling velocity of the organic-mineral aggregates are less than 0.047 cm/s compared to the higher settling velocities of the fecal pellets, 0.047-0.500 cm/s (Carson et al., 1988). The organic-mineral aggregates are very abundant on the marsh surface in the Great Sound and comprise 60-80% of the suspensate under fair weather conditions. The fecal pellets which are hydrodynamically equivalent to coarse silt and fine sands (Carson et al., 1988), will be deposited quickly on the marsh surface as the velocities decrease on overbank flow. In the Great Sound, the fecal pellets are dominate in the suspensate only under storm conditions or around maximum tidal flow with velocities exceeding 50 cm/s and require bottom shear stresses in excess of 2 dyn/sq. cm for suspension. Single mineral grains range from clay to fine sand but are dominant in suspension only in the size fractions greater than 88 um and therefore also require maximum velocities for transport.

Storm deposition affects the size distribution on the marsh surface but has not been studied directly. Stumpf (1983) concluded that mechanisms other than normal flooding were required to produce the pattern of sedimentation within the Holland Glade Marsh, Lewes, Delaware. Sedimentary deposits in the backmarsh (coarse-silt) were not formed during normal tidal flooding as this size class was not found in suspension. Storms provide the energy for erosion or accretion because wind affects the velocity of the flow and influences turbulence as well as the concentration of sediment in suspension. An increase in the flux of sediment through cross-sections in the channel coupled with advection to greater distances on the marsh surface would affect the sedimentation pattern, however there are at present no studies during storms that are sufficient to examine sediment transport during such events.

Tidal Flux Studies - Middle Atlantic Coast

The sediment flux in coastal marshes has been examined by measuring suspended sediment concentrations in a tidal channel over a sampling of tidal cycles (Ward 1981; Settlemyre and Gardner,1977; Boon, 1973). Boon studied a typical salt marsh creek, Little Fool Creek, near Wachapreague, Virginia. The sedimentary budget balanced over normal (storm-free) tidal cycles (1972-1973) showed a net removal of sediment having a 10-15% combustible organic content. He found an annual cycle in steric sea level which produces significant variations in the frequency of occurrence of the higher discharge volumes entering and leaving the marsh (the frequency being higher in September and October but significantly lower during January and February; Higher levels of transport were associated with strong ebb residuals). Seasonally, he also found that cooler temperatures enhanced the resistance of cohesive bottom sediments to erosion. His observations showed that as the depth shallowed in the creek, the bottom shear stress and sediment transport capacity increased.

Settlemyre and Gardner (1977) determined the sediment flux during 25 randomly selected tidal cycles for Charleston Harbor tidal creek, South Carolina. Their data, like Boon's, showed a seasonal effect with major export in the summer and minor accumulation in the winter, except when preceded by wind-induced wave-scour of the outlying tidal flats. The cycles that showed major export were characterized by higher ebb turbidity whereas those that showed major import had turbidity peaks during flood flow. In contrast to Boon's data, they recorded an import of inorganic matter. On a seasonally adjusted basis the yearly budgets were -393 g m^{-2} yr^{-1} for total suspended sediment, +280 g m^{-2} yr^{-1} for inorganic suspended sediment, and -673 g m^{-2} yr^{-1} for organic suspended sediment.

Ward (1981) also found a net export of total suspended solids for Bass Creek, Kiawah Island, North Carolina. The maximum suspended sediment concentration in summer was attributed to an increase in both biologic activity and thundershower frequency.

There was a net export of inorganic suspensate during the summer of March 1977 and a balance during the winter. This implies that, under these conditions, the bay or estuary will receive the sediments and that the sedimentation will be primarily subtidal.

The above ebb-dominated marshes all showed net export of sediment. In addition, sediment transport varied seasonally being greater in summer due to higher temperatures (lower resistance of bottom sediments to erosion), increased tidal prisms, and increased suspended sediment concentrations.

Summary

The size distribution of sediments on the marsh surface is complicated by the transportation of pre-existing sediment as well as that brought in by the tidal creeks. Furthermore, storm deposition may significantly alter the pattern of particle sizes normally deposited during fair weather conditions. The formation of aggregates suggests that clay and quartz particles can be deposited simultaneously. The particle size distribution will tend to be progressively finer from the creek bank to the high marsh until influenced by pre-existing sediment. Coarse silts and sands and large clay agglomerates should settle closest to open water and finer sediment should settle out in the backmarsh.

The greater sediment transport during the summer and the potential for export of the organic matter contribute to the sediment dynamics of the marsh.

ORGANIC SEDIMENTATION

The rate of organic matter accumulation in the marsh contributes to the total accretion rate and is generally higher in elevated high marsh areas (Good et al., 1982; Armentano and Woodwell, 1975; Gleason, 1976). The input of organic matter into salt marsh sediments differs from that of other marine systems. As in most marine systems, there are detrital deposits at the sediment surface, however the major organic input in marshes occurs below the sediment surface as the production of roots and rhizomes (Howes et al., 1985; Schubauer and Hopkinson, 1984; Good et al., 1982; Valiela et al., 1976). The long-term accumulation of carbon in marshes is estimated to be at most 100-200 g C m^{-2} yr^{-1} (Howarth and Hobbie, 1982). This rate depends on the balance between the processes of production and decomposition. These processes are related to locations in the marsh, properties of the sediment, and the standing crop biomass.

Factors Influencing Production

Spartina alterniflora is the dominant salt marsh plant between mean low water and mean high water along the Atlantic coast. This species exhibits a productivity gradient such that the tallest grass is found along the creek banks with height and productivity diminishing toward the marsh interior (King and Klug, 1982; Delaune et al., 1983). The different height forms are genetically the same indicating that it is the environmental conditions which affect growth. Both above- and belowground production of Spartina decrease inland so that the gradient has relevance to the belowground production as well. Furthermore, depth profiles show decreasing penetration of Spartina roots and rhizomes into the marsh substrate with increasing distance from the creek bank (Ellison et al., 1986).

The differences in the aboveground productivity of Spartina alterniflora are related to a number of key variables: oxygenation of the root zone, nitrogen availability and uptake, sulfide concentration and salinity (Chalmers, 1982). The most current hypotheses suggest that plant growth is limited by the amount of nitrogen available in relation to factors mediating uptake (Howes et al., 1981, 1986; Delaune et al., 1983). Several authors suggest that sulfide concentration is a major factor limiting uptake of nitrogen; however, the exact mechanism is unclear (Howes et al., 1986; Delaune et al., 1983; Wiegert et al., 1982). The sulfide concentration in pore water is dependent on the oxidation status of the substrate which is determined largely by pore water movement, drainage, and air entry. Bioturbation also enhances aeration and can be a major factor in the redox potential of some salt marsh soils. The interrelationships among the key variables can be complex as salinities usually increase along with evapotranspiration.

The key variables and their influence on productivity have been reviewed in the recent literature (Nestler, 1977; Valiela et al., 1976; Mendelssohn, 1980; Delaune et al., 1983; Weigert et al., 1983; Agosta, 1985; Howes et al., 1986) such that only a short summary is presented here. The literature cited above suggests that stagnation of interstitial water causes a decrease in productivity due to:

1) increased sulfide concentrations- which may limit growth by preventing nitrogen uptake and root development by direct toxic effects.

2) increased anoxic stress- which lowers Eh and increases sulfide concentrations in the pore water.

3) decreased iron (nutrient) availability due to the precipitation of trace metals with sulfide.

4) potential increases in osmotic stress due to the concentration of salts by evapotranspiration.

The increased productivity at the creek bank is likewise associated with pore water movement and its affects on pore water chemistry.

Factors Influencing Decomposition

The primary factors which affect above- and belowground decomposition are the nature of the plant material and decomposing organisms, oxygen supply (Eh and pH) and water supply. Temperature is known to influence these factors and therefore also alters rates of decomposition. Much of the belowground decomposition is mediated by anaerobic fermenting and sulfate-reducing microbes (Clymo, 1983; Nakas, 1986).

Anaerobic Conditions

The rate of decomposition in a marsh is affected by the position of the plant material relative to zones of aerobic and anaerobic conditions in the sediment (Schubauer and Hopkinson, 1984; Reddy and Patrick, 1975). Although decomposition varies greatly in salt marshes, the rate of belowground decomposition is always less than the rate of aboveground decomposition (Hackney, 1987; Hackney and de la Cruz, 1980). Reddy and Patrick reported a 56% reduction in cumulative rice straw decomposition after 128 days of submergence compared to an aerobic decomposition. Hackney, 1987, measured the average annual rate of decomposition on the surface of a North Carolina salt marsh and found it to be 39.7%. The surface decomposition exceeded the rate of belowground decomposition which was 26% (5-15 cm below the surface). He also found that CO_2 evolution, which is the ultimate product of decomposition, decreased approximately 50% under continuously anaerobic conditions compared to all other treatments which included an aerobic period.

The differences in aboveground and belowground rates is probably attributable to differences in the decomposing organisms and their populations, although very little research has been carried out in this regard. Anaerobic bacteria are known to be less efficient than aerobic bacteria in obtaining energy, therefore, the decrease could be due to the lower metabolic efficiency of anaerobiosis (Hackney, 1987). The decay of large amounts of plant biomass belowground results in a high oxygen demand and high rates of sulfate reduction. Sulfate reduction and associated fermentations are known to dominate the decomposition process in the salt marsh (Howarth and Giblin, 1983). Other forms of anoxic respiration such as denitrification and methanogenesis are insignificant in comparison (Howarth and Teal, 1979). However this does not preclude interactions in these processes as Connell and Patrick, 1969, showed that in water-logged soils, sulfate reduction is inhibited by nitrate. Sulfate, an anaerobic electron acceptor, is reduced to H_2S by obligate anaerobic bacteria within an Eh range of +115 to -450 mV (Nakas, 1986). Valiela et al., 1982, found that the larger populations of sulfate-reducing bacteria occurred at depths in peat where the redox potential indicated the presence of a very low oxygen tension. Interpretation of results using sulfate reduction to measure decomposition has proven to be difficult (King, 1988).

Inundation by Tides

Decomposition not only varies with aerobic and anaerobic zones in the sediment, but with the frequency and duration of tidal inundations (Hackney, 1987; Reddy and Patrick, 1975). The position of the aboveground litter in the tidal range affects the rate at which detritus disappears: Aboveground litter on the high marsh is not affected as much by tidal removal as is low marsh detritus. Furthermore, the decomposition of organic matter is stimulated by alternately wetting and drying the soil. Valiela et al. (1982) concluded that decomposition is faster in detritus that is subject to longer tidal submergence, apparently due to a greater degree of leaching and therefore a faster removal of more labile compounds. They placed litter bags at different elevations in Great Sippewissett marsh, Massachusetts, in tall and short Spartina alterniflora and Spartina patens. The percent weight remaining decreased toward the creek bank or tall Spartina alterniflora. The nature of the decomposing material was not discussed. However, the authors did attribute some of the differences to an increase in detritus feeders toward the creekbank.

In both Georgia and New Jersey marshes, belowground biomass of Spartina alterniflora appears to turn over some 2-3 times faster in stands of the tall form (which is more frequently inundated than short Spartina) than in stands of the short form (Howarth and Hobbie, 1982). The rate of decomposition as determined by CO_2 production of ground rice straw was faster in treatments with a greater number of alternate aerobic and anaerobic periods (Reddy and Patrick, 1975).

Nature of the Plant Material

Although spatial variability in decomposition in salt marshes may result from differences in the frequency of tidal inundations, the nature of the plant material is also important. White et al.,1978, concluded that Spartina alterniflora aerial production decomposes much faster than the high marsh species Juncus roemerianus, Distichlis spicata, and Spartina patens. The oven dried shoot material was placed in litter bags staked to the ground. It decomposed to 50% in one month for Spartina alterniflora, three months for the Juncus and five months for Spartina patens. To test the effect of site location on decomposition rates, they placed bags containing D. spicata alongside some of those containing S. alterniflora. A difference in their rates of decomposition (S. alterniflora-50% loss in 2.5 months, D. spicata-50% loss in 5.5 months) suggested that the plant materials rather than the environmental conditions were responsible.

Recently, further investigation into the recalcitrant nature of the plant material has been made (Benner et al., 1986, 1984). The recalcitrant nature of lignocellulosic plant matter in anoxic sediments is frequently cited as a major factor in the accumulation of peat. Spartina alterniflora lignocellulose is composed of 80% polysaccharide and 20% lignin while J. roemerianus lignocellulose consists of 75% polysaccharide and 25% lignin (Benner et al., 1984). Benner et al. found a large difference in the degradation of the lignin component of S. alterniflora and J. roemerianus lignocellulose: after 294 days, 16.9% of the lignin in the Spartina was degraded to $^{14}CO_2$ and $^{14}CH_4$ whereas only 3.5% of the lignin in Juncus was degraded to these same products. Combining this result with the fact that S. alterniflora decomposes aerobically faster than D. spicata, it appears that the nature of the material is as important as the frequency of inundation in explaining decomposition. S. alterniflora may decompose faster both aerobically and anaerobically due to the structural nature of the plant material. The increased recalcitrance of the high marsh vegetation may help to explain the increased organic matter content of some high marsh soils. Since bacteria are the predominant degrader of lignocellulose (Benner et al., 1986), the importance of the decomposing organisms is again stressed.

Another important factor contributing to organic matter degradation is pH. Benner et al., 1985, reported that the mineralization of the polysaccharide components of S. alterniflora and C. walterianna increased several fold with increasing pH (range 3.6 to neutral). Rates of lignin degradation were only minimally affected within this range. Reed, 1981, also demonstrated that decomposition was dramatically decreased at pH 4 relative to pH 6 or 8 (Howarth and Hobbie, 1982). Although salt marshes generally have a neutral pH, microzones probably exist particularly in the rhizosphere due to higher levels of organic acids.

The literature on rates of decomposition showed that it generally decreased inland due primarily to

1) the reduced frequency of tidal inundations- which export aboveground detritus and leach soluble compounds from the soil.

2) the increased recalcitrance of plant material.

Other contributing factors such as pH and the quantity of decomposing organisms have not been studied thoroughly enough to define their roles in the decomposition of organic matter in salt marsh soils.

Summary

The information presented in this section suggests that rates of production and decomposition decrease inland in frequently inundated areas of a marsh. The tidal inundations are important because they remove organic detritus from the marsh surface. The organic matter content of creek bank areas therefore is low due to high export of organic detritus and the fast aerial decomposition of *Spartina alterniflora*. In the backmarshes where organic matter is accumulating, the rate of production is higher than the rate of decomposition in spite of the limitations placed on plant productivity.

MATERIALS AND METHODS

The methods presented in this section were used to achieve the goal of examining the sediment properties, vegetation, and cesium profiles of locations along two major transects at the Virginia study sites. The above factors were examined so that interrelationships among them could be determined. The specific objectives, listed in the introduction, are to evaluate the mobility or immobility of ^{137}Cs at each site, relate cesium mobility and retention to the sediment properties and physical processes occurring in each marsh, and examine the role of vegetation in the transport of cesium. Furthermore, sedimentation will be examined by determining sediment properties and above- and belowground biomass at each site.

Study Area

The salt marsh area in this study, on Virginia's Eastern Shore, is owned by the Nature Conservancy and is part of a Long Term Ecological Research Site. It is located on the oceanside of Delmarva Peninsula and behind Hog Island Bay (Figure 1). The area is within the Nassawadox, Virginia 7.5 minute quadrangle at a latitude of approximately 37° 27' 30'' and a longitude of approximately 75° 50'. The two marsh sites studied are separated by a relict sand ridge: to the east of the sandy peninsula lies "No Egrets" Marsh and to the west lies Phillips Creek Marsh (Figure 2). Although these marshes fringe the mainland, other marshes along the shore are found on the backs of the barrier islands and in the lagoonal system between the islands and the Pleistocene mainland. The mainland was deposited when sea level was higher than present during an interglacial period approximately 35,000 years ago. The barrier islands have formed during the Holocene and are believed to be approximately 3000 years old (Dolan et al., 1980). It is likely that the marshes formed after barrier development and are therefore fairly recent sedimentary environments.

According to Harrison (1972) clay along the Delmarva Peninsula consists predominantly of chlorite and illite. Powder x-ray diffraction of marsh samples from the surface of both creek bank locations in this study and of an underlying clay layer in the high marsh showed the predominant clay mineral to be illite, followed by kaolinite and chlorite (personal communication, R. Mitchell).

Freshwater from the mainland drains into Phillips Creek whereas the tidal creeks on the eastern side are flooded predominantly by water from Hog Island Bay. By observation alone, the eastern marsh is flooded in its entirety on a daily basis and has a gentle slope whereas the western marsh rarely floods in the high marsh areas. Although the marsh is probably underlain by sandy sediments, the predominant sediment is silt and clay (Harrison, 1972).

The marsh vegetation at both sites is predominantly Spartina alterniflora (Table II). It occurs in tall form on the margins of the channels, up to approximately 60 cm, and decreases inland to about 20 cm. On these marshes the tidal limits are gradational and the Spartina gives way to other grasses near the upper limit of tidal flooding. As Table II shows, in the Phillips Creek Marsh, Juncus roemerianus, Spartina patens and Distichlis spicata are present in the high marsh whereas in the "No Egrets" Marsh, a panne area and Salicornia virginica occupy some area at the highest elevations. The low density of the vegetation near the creek banks suggests a fairly recently vegetated mudflat. Peat has formed on the high marsh of Phillips Creek to a depth of approximately 15 cm which is very shallow considering that peat near Sandy Neck, Massachusetts, attains a maximum thickness of about 610 cm (Redfield, 1972). Sea level rise is evident from a tributary of Phillips Creek which is making headward erosion into the upland area and the presence of wood stumps at depth in the high marsh overlain by marine sediments.

Table II. FIELD SITES

Location as in Figure 2	Distance from major creek bank (m)	Primary Vegetation	Date Sampled
Phillips Creek Marsh			
1	3	Sp. alterniflora	3/88
2	197	Sp. alterniflora	8/87
3	353	Sp. alterniflora	8/87
4	435	Juncus roemerianus	8/87
4A	435	Juncus roemerianus	4/88
4B	435	Juncus roemerianus	4/88
5	460	Spartina patens	8/87
6	547	Spartina patens	4/88
7	635	Distichlis spicata	8/87
"No Egrets" Marsh			
8	10	Sp. alterniflora	8/87
9	195	Sp. alterniflora	8/87
10	260	Unvegetated panne	8/87
11	293	Salicornia virginica	8/87

Field Methods

A transect was established in each marsh which ran essentially perpendicular to each creek bank (Figure 2). In August of 1987, sites were chosen along each transect through the transition in vegetation. The predominant type of vegetation and approximate distance along the transect are presented in Table II. After selection of the sites, a square meter plot of the vegetation was clipped (8/87 sites only with the exception of location 9) followed by coring of the sediments. At each location a 19 cm diameter core and a 14 cm diameter core were taken in plexiglass tubing. The cores sampled the upper 40 cm of the sediment. The 19 cm core was immediately sectioned into 2 cm intervals, wrapped in aluminum foil and sealed in plastic bags. Each section of the large diameter core was analyzed for ^{137}Cs distribution, bulk density, percent water, and inorganic/organic matter. The 14 cm core was extracted from the corer into a plastic bag so that macroorganic content could later be quantified. In May of 1988, elevations along the transect were determined with a long-range surveying transit.

Laboratory Methods

Macroorganic Matter

To quantify the aboveground standing crop, the square meter of vegetation was cut close to ground level and air dried. Due to the adherence of sediment on the *Spartina*, this species was then washed and redried so that the weight due to the sediments could be subtracted. To quantify the belowground standing crop, a 14 cm core was placed unsectioned on a very fine mesh netting (0.03 sq. cm's per square). The core was washed and rewashed with water only until no visible trace of the inorganic mud remained. The macroorganic matter was then dried in a Mechanical Convection Oven at approximately 100°C. A portion of the sample was then ground and ashed in a muffle furnace at 500°C to quantify the residual inorganic fraction.

Sediment Analysis

The percent water, total organic and inorganic matter, and bulk density were estimated for each 2 cm section of the 19 cm diameter cores. A wet weight was taken on each section prior to drying to a constant weight in an oven at 100°C so that the water content could be determined. The water content was taken as an estimate of porosity as each core was taken after tidal inundation to ensure saturation of the pores. Organic matter was determined as the weight loss on ignition (LOI%). A ground and homogenized subsample was then ashed in a muffle furnace at 550°C for approximately 24 hours. The samples were ground in a Wiley grinding mill. The weight remaining after drying and ashing is attributable to the fraction of inorganic matter. The bulk density of the sediment was determined as the oven-dried mass of the sample divided by its field volume (g/cc).

Isotope Analysis- ^{137}Cs and ^{40}K

Each 2 cm section of the 19 cm diameter core was dried and ground by the methods outlined in the previous section. The powdered sample was placed in a marinelli container with a maximum placed on the volume of material in order to maintain a constant geometry. The ^{137}Cs content of each section was determined by a multichannel analyzer which recorded the number of counts received at approximately 661.6 KeV, the energy of ^{137}Cs. Another isotope, ^{40}K was counted simultaneously and recorded at an energy of approximately 1460 KeV. The ^{40}K values were not corrected for background, which was high but assumed constant, and represent only relative values. Two detectors were used for counting: a high purity germanium detector (HPGe-2600 V position, labelled Detector A) and a lithium-germanium detector (Detector E, unlabelled). The counts per second above background were calculated by programs which integrated the counts under the ^{137}Cs peak and estimated a counting error. A ^{137}Cs standard was prepared by spiking a powdered sample containing a minimal detectable amount of cesium and placing it in a marinelli container. The activity was compared to a reference source (Amersham Gamma Ray) so that the conversion from counts per second to pCi could be made. The efficiency of Detector A was approximately twice that of Detector E such that the conversion factor was 2601.6 and 4627.3 pCi ct^{-1} sec^{-1}, respectively. All core sections except the following were counted on Detector A- location 1 and all three cores taken at location 4 in Juncus roemerianus.

Most samples were counted for approximately 24 hours.

The Drainage Experiment

The drainage experiment was performed to determine if the cesium peak or the inventory could be changed by draining salt water through the core and inducing anoxia by keeping the core submerged. The objective was to introduce a downward flow of pore fluid that contained cations that should exchange with ^{137}Cs. A cored section of Juncus was left in plexiglass tubing and transported to the laboratory. It was surrounded by saltwater and kept in a bucket until drainage was induced by removing it and placing the core on a metal screen that allowed the water to pass through and into a bucket. This procedure was followed intermittently for approximately two months during which about 136 liters of saltwater were drained through the core.

RESULTS AND DISCUSSION

Soil Properties

A major objective of this study was to relate the sediment properties and vegetation to how the marsh accretes vertically. The following soil properties were measured: bulk density, porosity, inorganic content, and organic content. The above- and belowground standing crops were also measured at selected locations along the transect. This section describes the trends that were found in both soil properties and vegetation and relates these trends to the accretion process. Original values for the soil properties are listed in Appendix 1.

General Trends and Values

The relationship of soil bulk density to mineral and organic density for the Virginia marshes in this study is similar to that found for Louisiana soils by Gosselink and Hatton, 1984. As shown in Figure 4, the bulk density is directly related to mineral density but is independent of organic density in both marshes. Thus, the inorganic density and bulk density profiles with depth correlate closely at each site. Examples

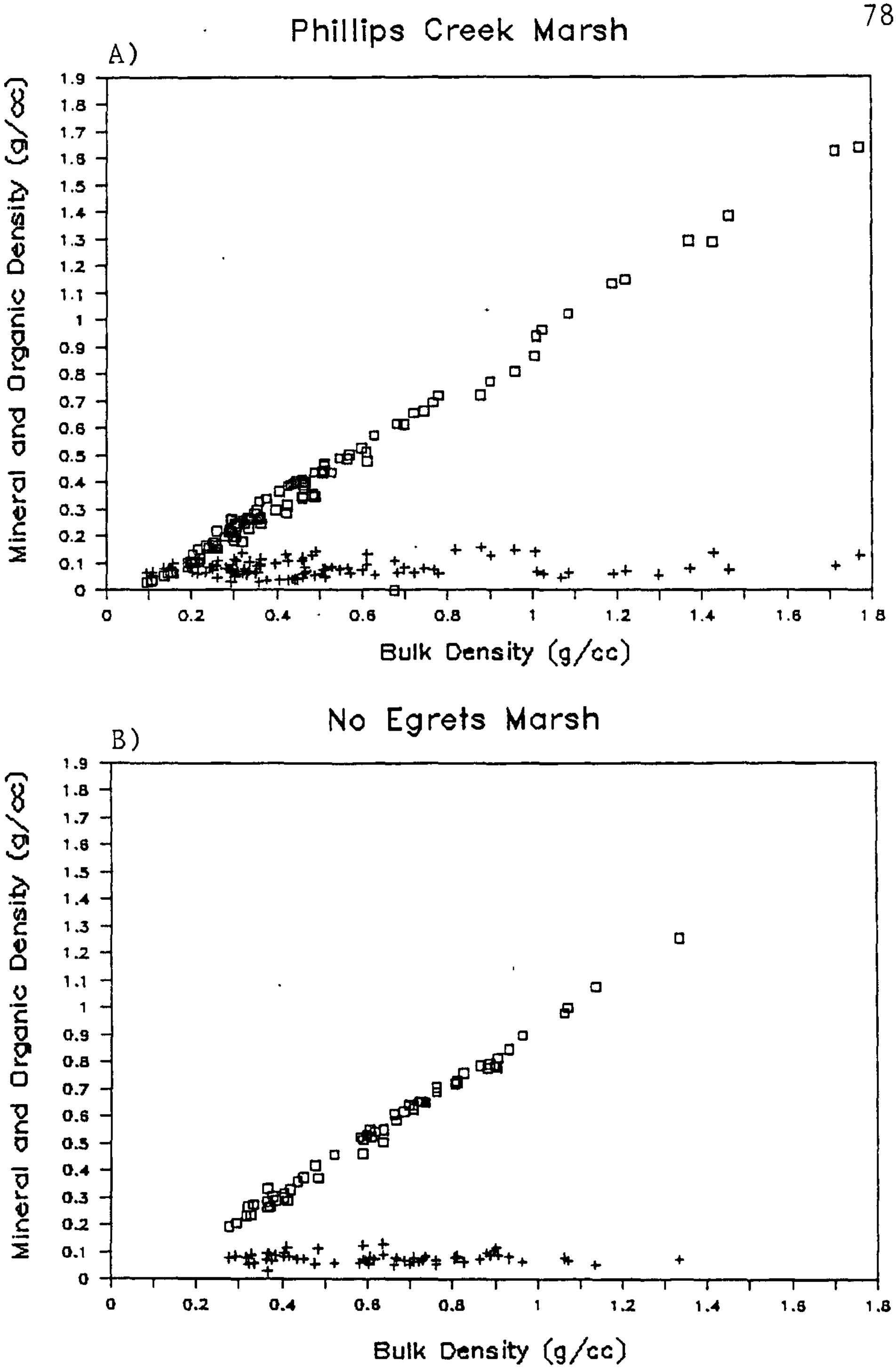


Figure 4 The mineral and organic densities as a function of bulk density for A) Phillips Creek Marsh and B) No Egrets Marsh. The squares represent mineral density values while the pluses represent organic density values.

of this relationship are shown in Figures 5 and 6A. The inorganic density comprises the majority of the bulk density at most locations in each marsh. The accretion process is dominated by inorganic sedimentation except at locations 6 and 7 where the organic density comprises the majority of the bulk density in the upper 10 cm of surface sediment (Figures 5B and 6A). Locations 6 and 7 are the only sites where the organic density increased with depth (Figure 6B) in spite of a dramatic increase in inorganic density as well. These data show that the high marsh in Phillips Creek is accreting organically rather than inorganically.

Although the bulk density is independent of the organic density, a curvilinear relationship exists between bulk density and organic content (% dry wt) as shown in Figure 7. At organic matter contents exceeding 50% in Phillips Creek Marsh, the bulk density is very low, less than 0.25 g/cc. The low bulk density associated with high organic content is a result of the sediment infilling of the organic matrix. This is supported by the high macro-organic concentration found at locations 5 and 7 (Table IV and Figure 15) and the visible root mat occupying the

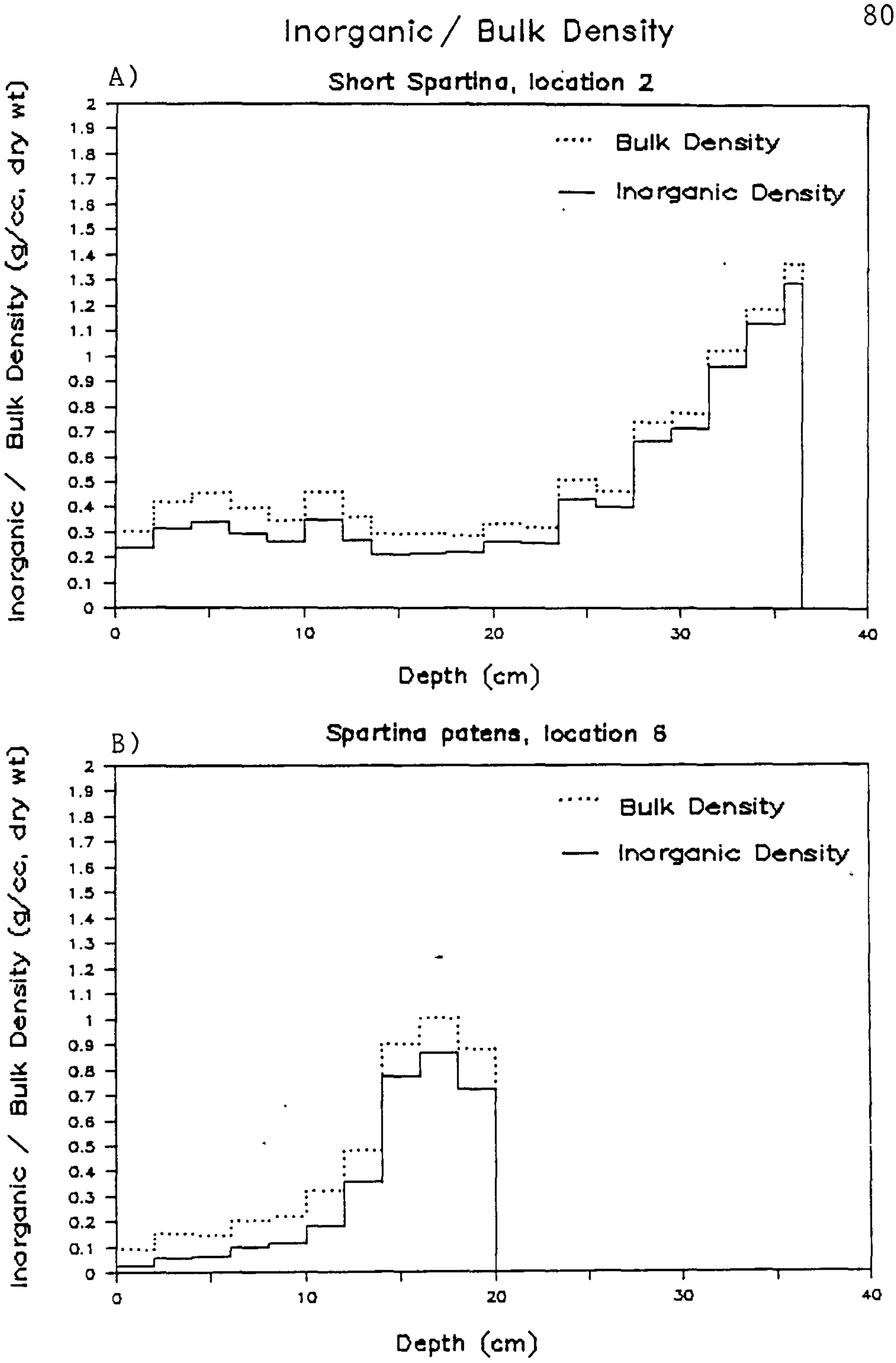


Figure 5 A) The inorganic and bulk densities with depth at location 2.
B) The inorganic and bulk densities with depth at location 6.

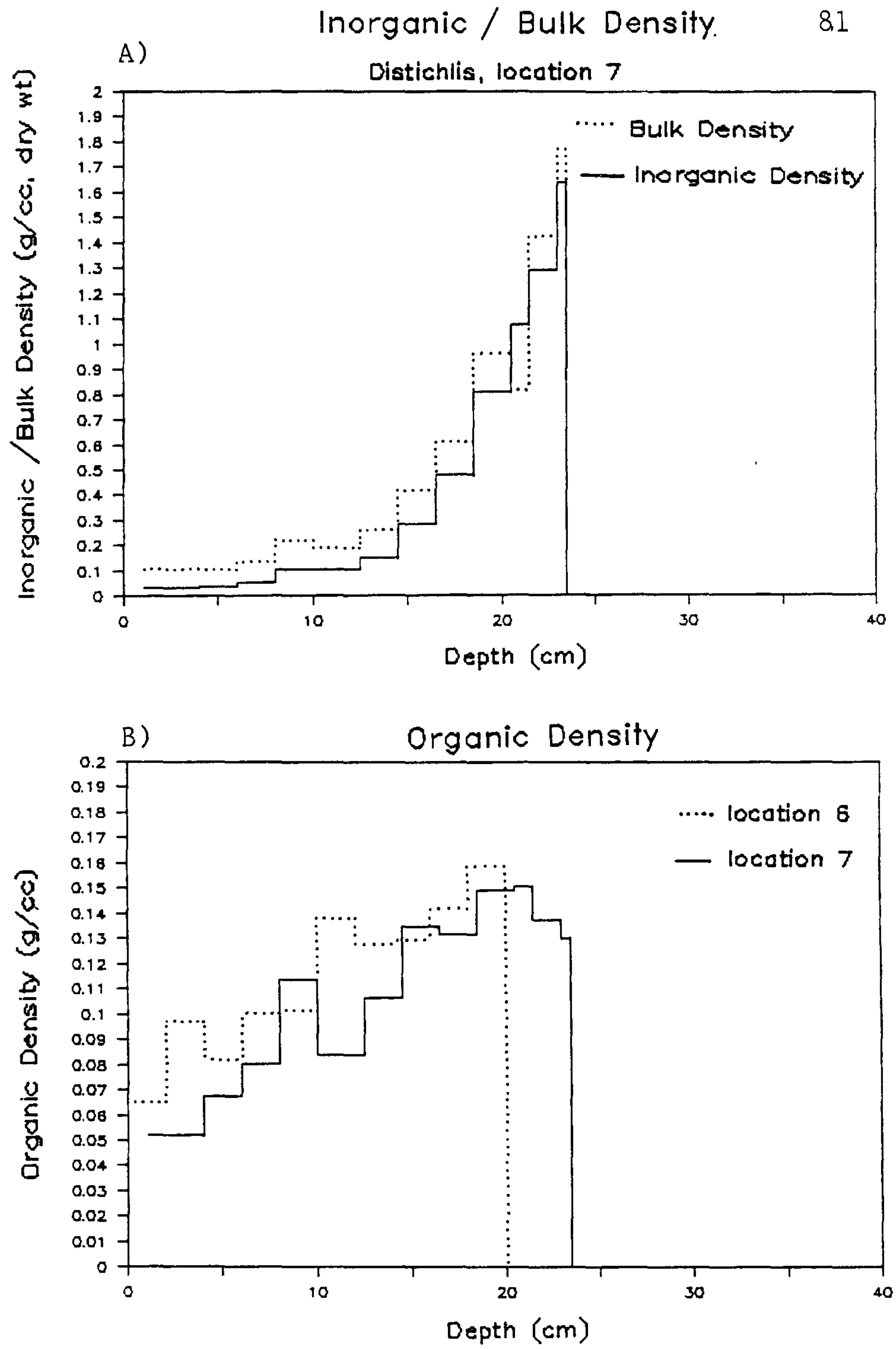


Figure 6 A) The inorganic and bulk density profiles with depth at location 7.

B) The organic density profiles with depth at locations 6 and 7.

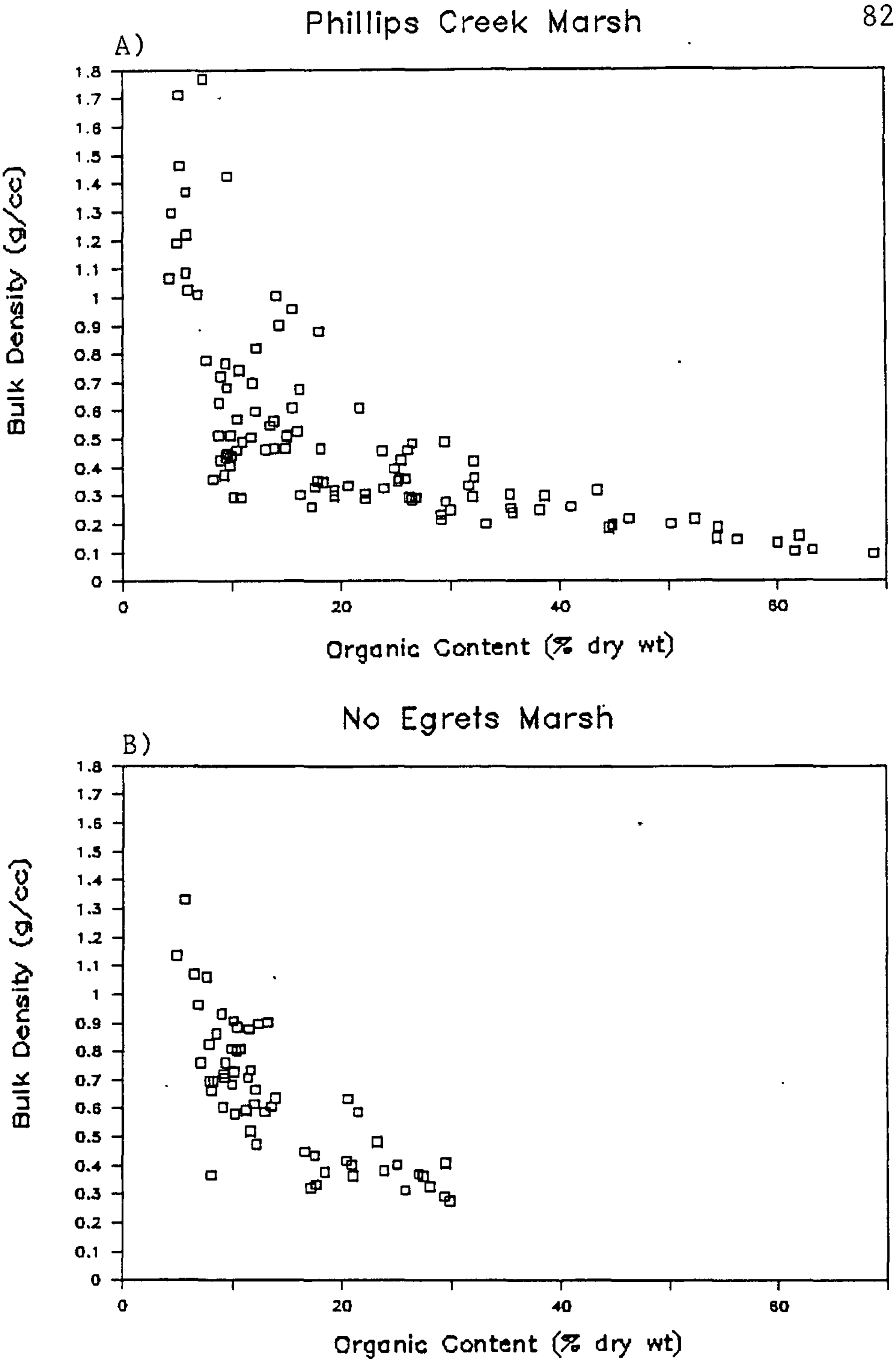


Figure 7 The curvilinear relationship between organic content and bulk density in A) Phillips Creek Marsh and B) No Egrets Marsh.

upper 16 cm or so of sediment from locations 4-7.

Another major trend in these Virginia marshes is an increase in inorganic density (g/cc) with decreasing porosity as shown in Figure 8. The porosity is high when low volumes of inorganic sediments are present. The lowest surface porosities are found at locations 1 and 8 nearest the creek banks in both marshes. The average porosity to a depth of 32 cm in No Egrets Marsh for locations 8, 9, and 10 is 46.1%, 75.1%, and 62.3% respectively. The average porosity values and other soil properties with depth in the Phillips Creek Marsh are influenced by a clay layer which was intersected at depth in locations 4-7. The range of values for each soil property as shown in Table III is greater in Phillips Creek primarily due to this subsurface clay along with the root mat present at locations 5-7.

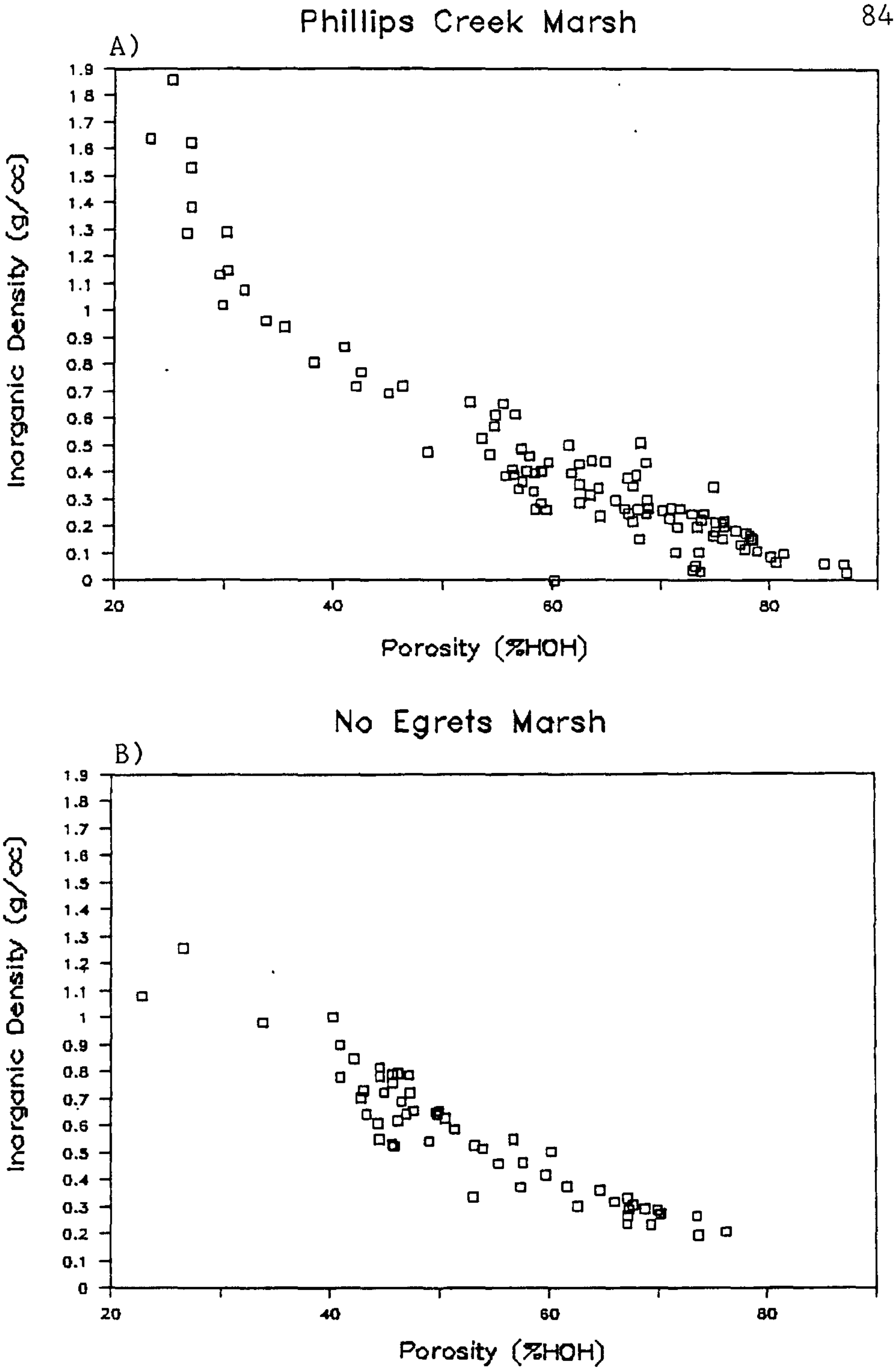


Figure 8 The relationship between inorganic density and porosity in A) Phillips Creek Marsh and B) No Egrets Marsh.

Table III
Soil Properties- Minimums and Maximums in Each Marsh

Soil Property	No Egrets Marsh	Phillips Creek Marsh
Bulk Density (g/cc)	0.28-1.33	0.06-1.77
Inorganic Density (g/cc)	0.19-1.26	0.03-1.86
Organic Density (g/cc)	0.029-0.130	0.03-0.151
% Water by weight	22.8-76.3	23.3-87.2
% Inorganic by weight	70.2-95.1	31.1-95.7
% Organic by weight	4.9-29.8	4.3-68.9

The general trends of soil parameters with depth are an increase in bulk density and inorganic density, a decrease in porosity, and variable organic density trends. Porosity and organic density profiles with depth are shown in Figure 9. The bulk density and porosity values deviate from the above trends at locations 1, 3, and 8. The porosity is relatively constant at these locations to the total depth of the cores (Figure 9B). The constant porosity may be due to mixing of the sediments by bioturbation. In addition, the fluctuations in inorganic density, organic density, and bulk density profiles at each location correlate closely (Figures 10-11). The fairly uniform density curves show that of the small percentage of organic sediments present, an increase

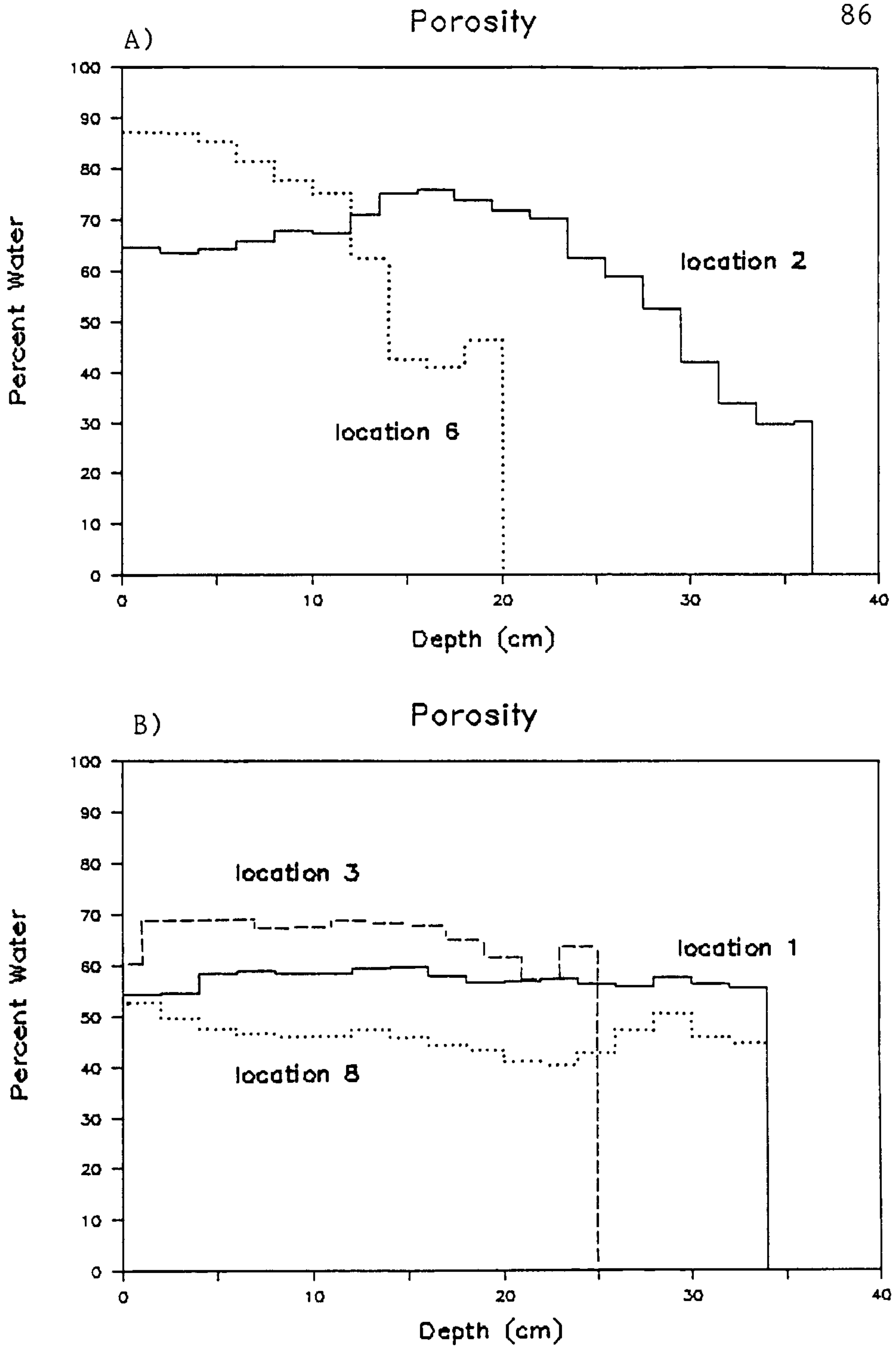
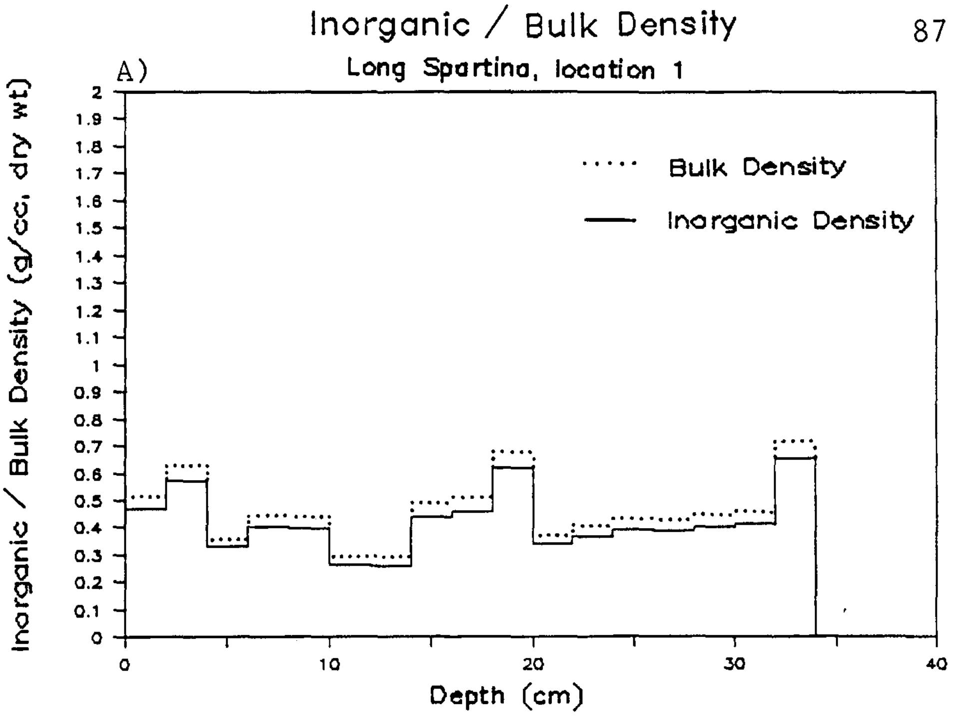


Figure 9 A) The porosity profiles with depth at locations 2 and 6.
B) The fairly uniform porosity profiles with depth at locations 1,3, and 8.



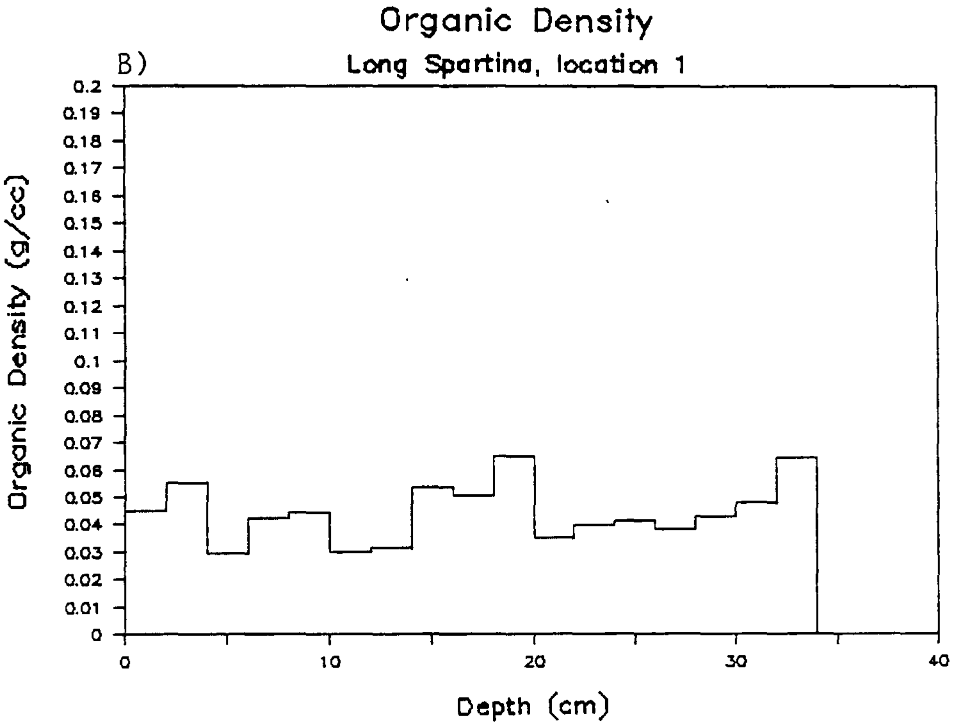


Figure 10 A) The inorganic and bulk densities with depth at location 1.

B) The organic density profile at location 1 which mimics the flucuations in the profiles in A.

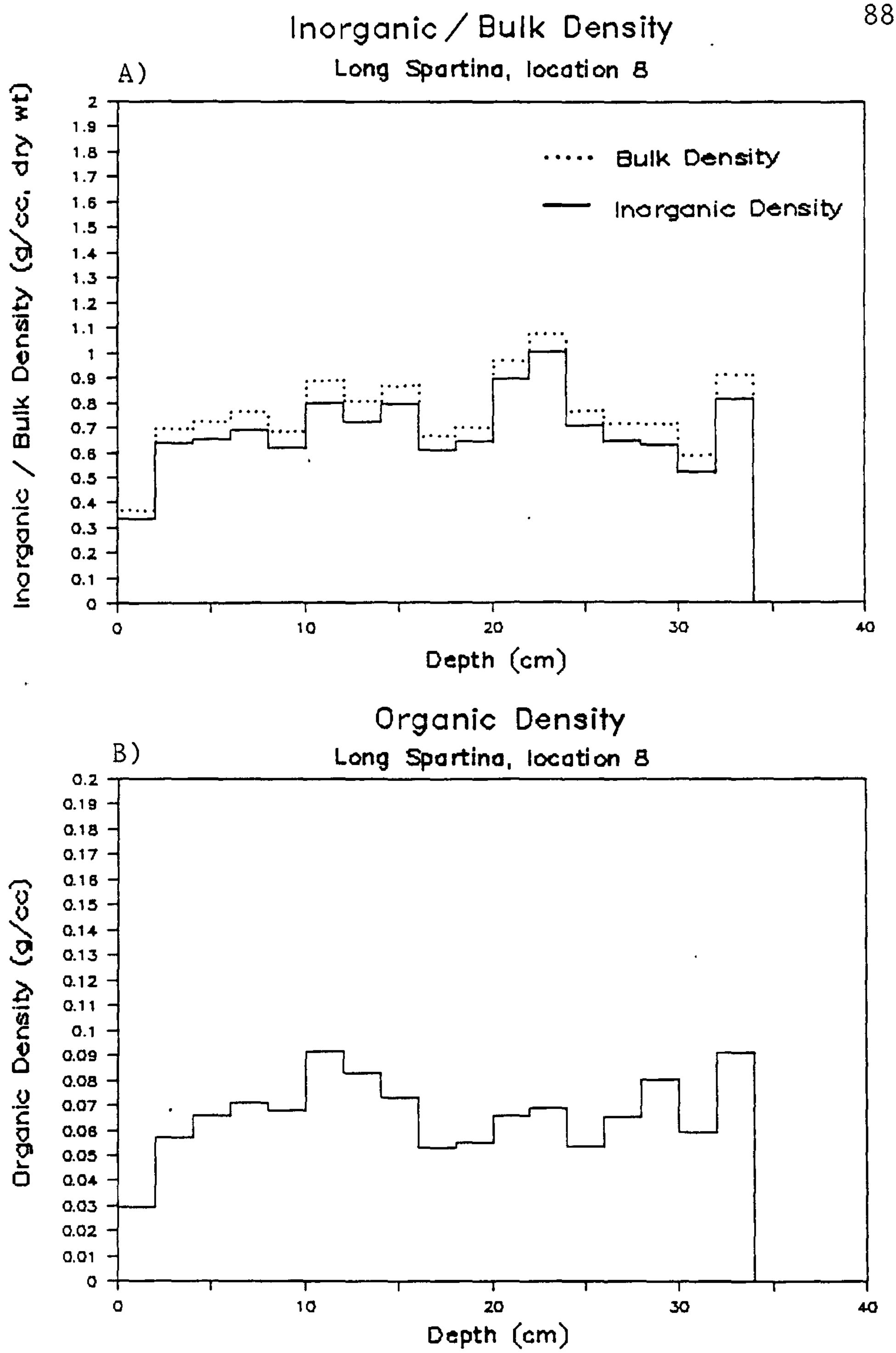


Figure 11 A) The inorganic and bulk densities with depth at location 8.
B) The organic density with depth at location 8.

in their content is usually associated with an increase in the inorganic content as well. A major deviation from the above trends is a dramatic increase in organic density with depth at locations 6 and 7 which has previously been discussed.

The three cores from location 4 provide sites for comparison of the variability in soil parameters. The variability in porosity, inorganic density and organic density is shown in Figures 12 and 13. Organic density is the most variable soil characteristic of the three properties measured. Although all porosity profiles with depth show a general decrease and the inorganic density profiles show an increase with depth, two of the three organic density profiles show a decrease while location 4B shows an increase at 16-18 cm. Porosity and inorganic density are most similar in the upper 10 cm of sediment.

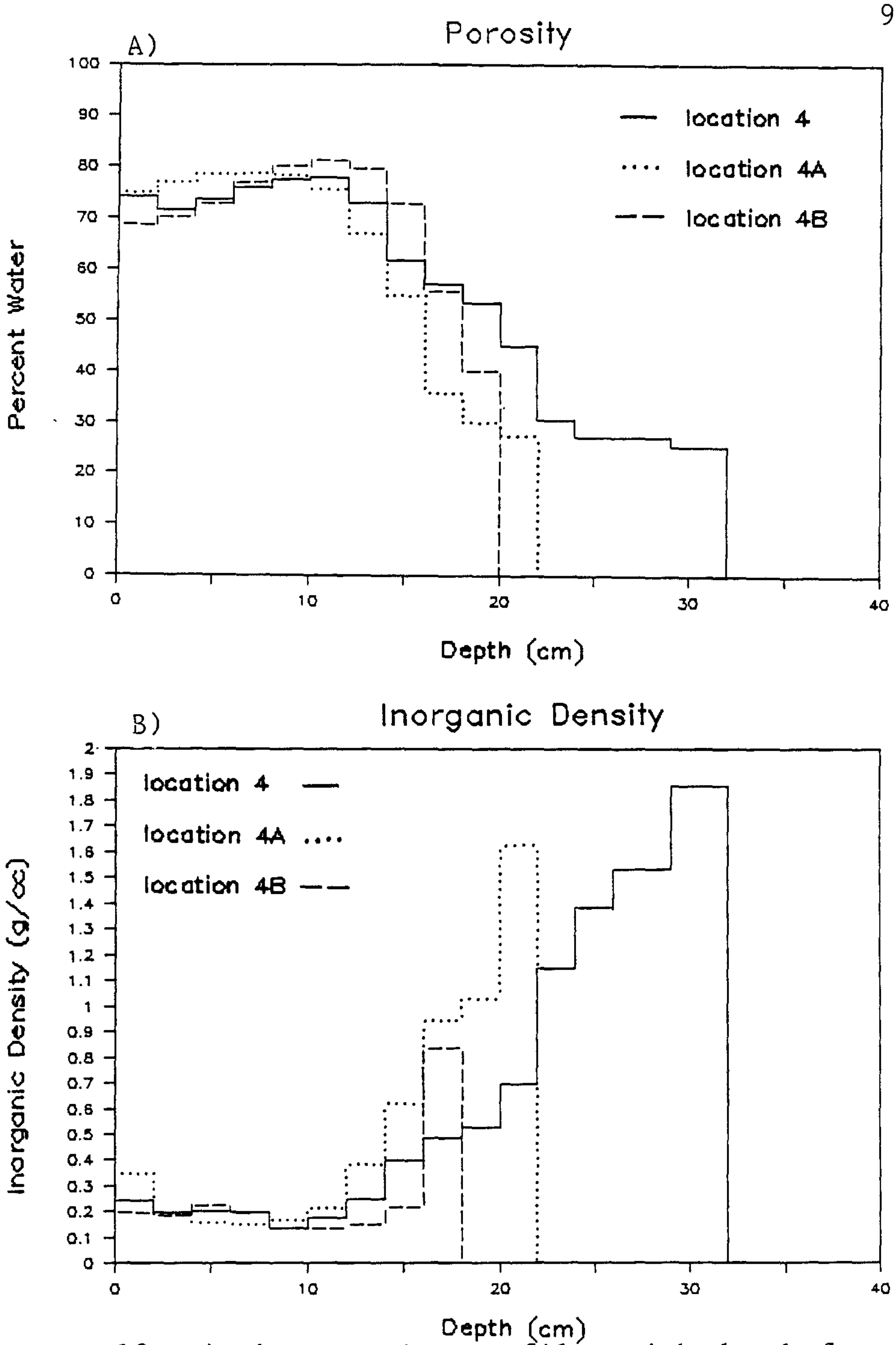


Figure 12 A) The porosity profiles with depth for all three cores, location 4.
b) The inorganic density profiles with depth for all three cores, location 4.

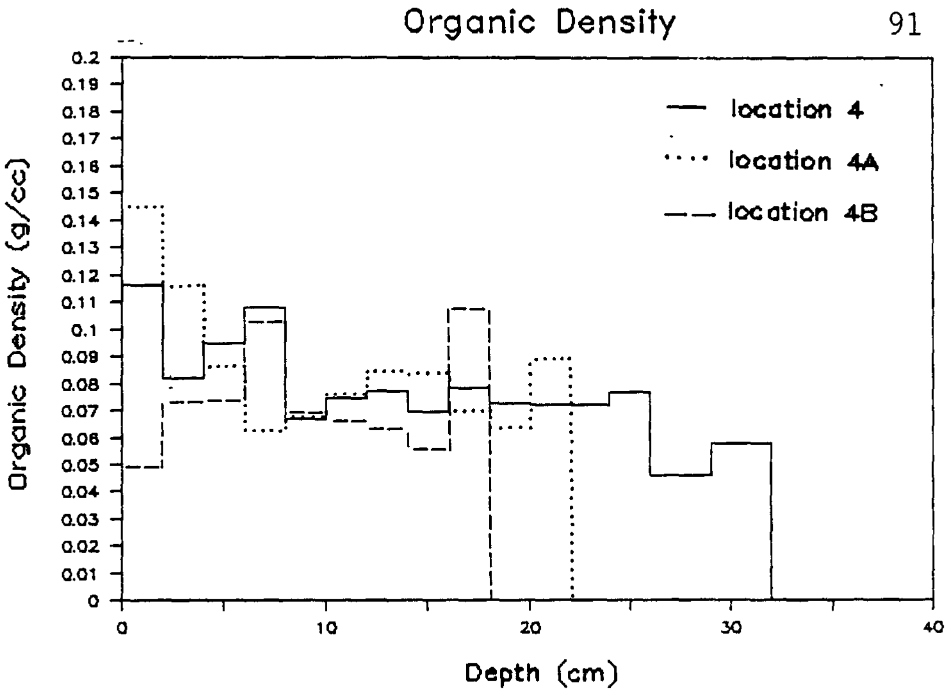


Figure 13 The organic density with depth at locations 4, 4A, 4B.

Organic Matter and Standing Crop Biomass

The literature from previous studies suggests that the inorganic component of sedimentation increases toward creek banks while the organic component by weight decreases (Delaune et al., 1978; Letzsch and Frey, 1980; Hatton et al., 1983). Data from this study show that this trend in general exists, however some variability is present. The average organic matter content (% dry wt) in the surface 16 cm is shown for these soils in Figure 14. The percent organic matter is lowest in each marsh at the location closest to the creek bank. The variability in No Egrets Marsh is the result of sampling in an unvegetated panne area (location 10) where the concentration of organic matter is dependent on detritus washed into the site. At location 11, the percent organic matter (10.7) is close to the concentration found at the creek bank (9.2). The low organic matter content at location 11, in *Salicornia* is the result of minimal above- and belowground standing crops (Table IV) which suggest that productivity of this species is low. In Phillips

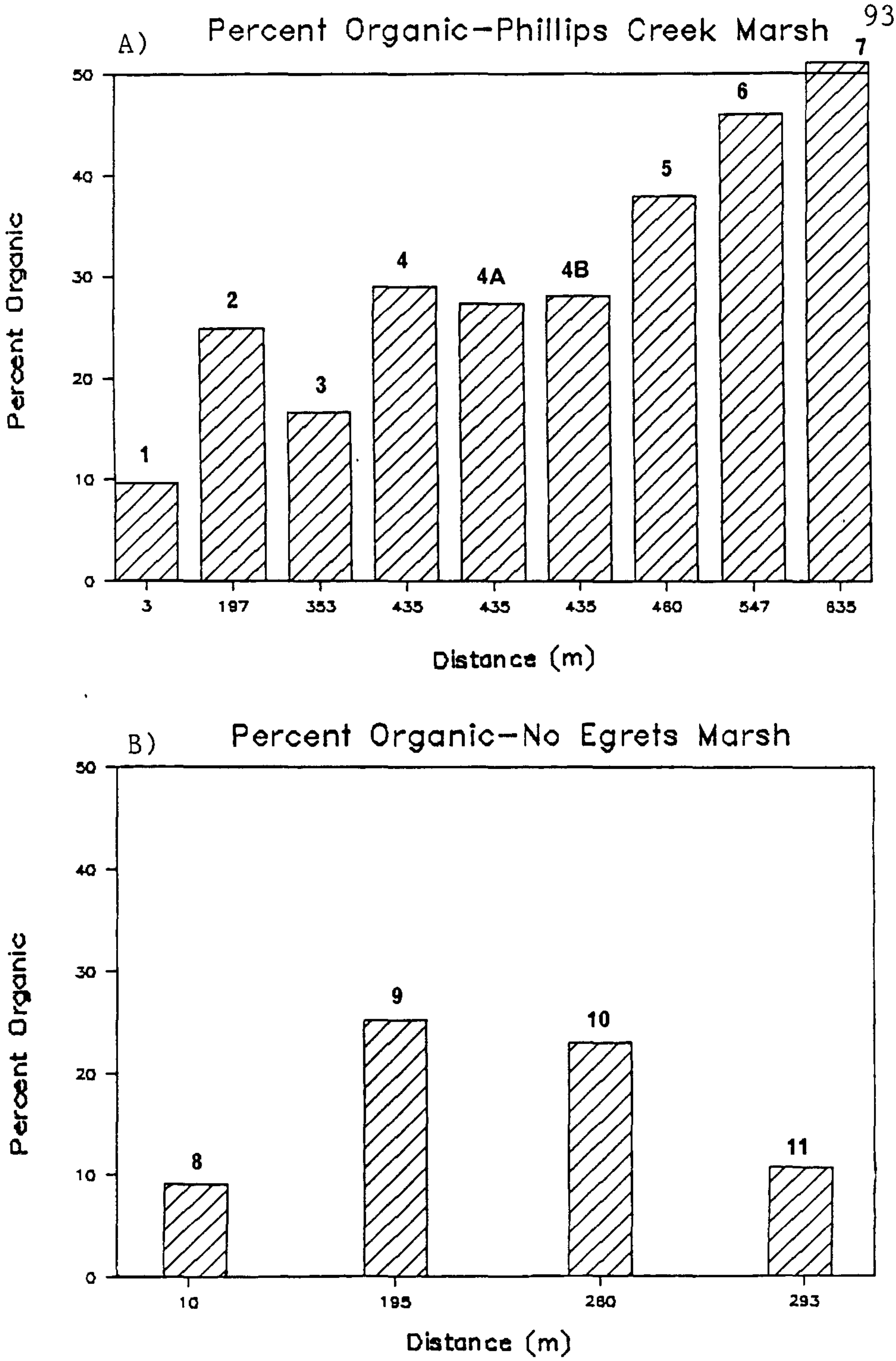


Figure 14 The average organic content by weight in the surface 16 cm of A) Phillips Creek Marsh and B) No Egrets Marsh.

Creek Marsh, the organic content in the top 16 cm of location 3 is low in comparison to locations 2 and 4. Although the aboveground standing crop is significant (893.7 g/m^2) at location 3, the belowground standing crop (220 g/m^2) is the lowest recorded in this marsh (Figure 15). The growth of the tall form of *Spartina* at location 3 is due to the lower elevation or swale which promotes greater drainage of the pore water. This may also influence decomposition and reduce the organic matter concentration by increasing the Eh of the sediments and lowering the sulfide concentrations. In addition, the aboveground standing crop values for *Spartina alterniflora* show a deviation from the productivity gradient found in most marshes.

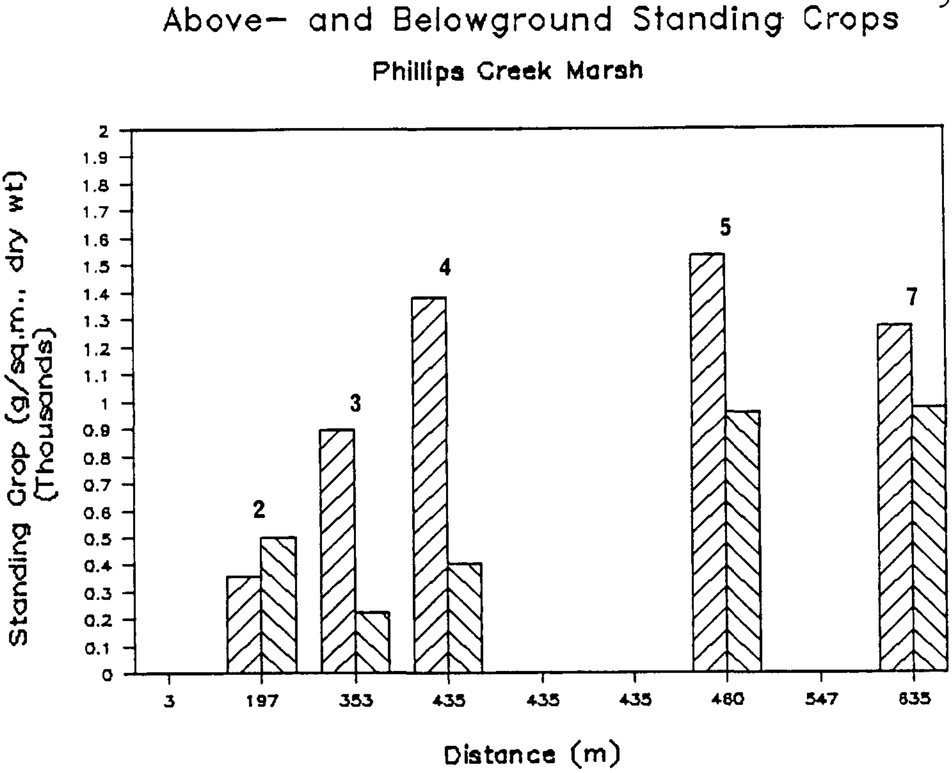


Figure 15 The above- and belowground standing crops in Phillips Creek Marsh. The first bar represents the aboveground standing crop while the second bar represents the belowground standing crop.

TABLE IV
Above- and Belowground Standing Crops

Location	Aboveground * Standing Crop (g/m^2)	Macro-organic** Matter predicted in 2 cm thick slice (g/m^2)
2, short *Spartina*	452	500
3, long *Spartina*	894	220
4, *Juncus*	1379	402
5, *Spartina patens*	1535	960
7, *Distichlis*	1272	972
8, long *Spartina*	300	334
10, panne	---	384
11, *Salicornia*	419	238

* The *Spartina alterniflora* values represent the weight after washing.

** Calculations and ash-free dry weights are presented in Appendix III.

The average organic matter content in the surface 16 cm steadily increases from 28.9% at location 4 to 50.7% at location 7. At depths exceeding 16 cm the average organic matter content for locations 4-7 is affected by a dense clay layer. The high average organic content in locations 4-7 also corresponds to the highest aboveground standing crops. The high standing crop values for locations 5-7 may be a result of the high inorganic nutrients present in the underlying clay layer. The high above- and belowground standing crops are factors contributing to the organic matter accumulation. Values for the percent organic are as high as 70% (dry wt) on the high marsh plateau of Phillips Creek Marsh whereas in No Egrets Marsh the percent organic does not exceed 30%. In addition, the variability in above- and belowground standing crops is higher in Phillips Creek Marsh. The differences in organic matter content, decomposition, and production between the two marshes appears to be influenced by the frequency of inundation.

Summary

The increase in organic density only at locations 6 and 7 in Phillips Creek Marsh, along with the observed root mat, suggests that the high marsh is presently accreting organically rather than inorganically. The percentages of inorganic and organic sediment across the marsh show that in general the inorganic component of sedimentation is increasing toward the creek bank while the organic component by weight decreases. This trend is not rigorously adhered to in No Egrets Marsh. The sediment characteristics show that when the porosity is high, the inorganic sediment content is low. The standing crop values in the back marsh of Phillips Creek Marsh are higher than low marsh values and indicate that organic matter may be accumulating there due to the high productivity of the vegetation and decreased rates of decomposition.

Evidence for the Mobility of Cesium

In terrestrial and freshwater ecosystems, the cesium is partitioned primarily to the inorganic phase with little accumulation in plants or mobility in pore water (Menzel, 1965; Dahlman and Van Voris, 1976; Cline, 1981). The studies of cesium in lakes show evidence for mobility based on slow molecular diffusion, mixing of sediments by benthic organisms, and redistribtuion of sediments by physical mixing (Pennington et al., 1973; Robbins and Edgington, 1975; Dominik et al., 1981; Davis et al., 1984). Other studies show mobility based on the salinity of flood waters which supplies cations (K^+, Na^+, Ca^{2+}, Mg^{2+}) that compete with cesium for binding sites on clay minerals and within plant tissues (Klechkovsky et al., 1959; Sawhney, 1964; Kawabata, 1967; Cheng and Hamaguhi, 1968; Schell et al., 1980, 1981; Stanners and Aston, 1981; Evans et al., 1983). The production of ammonium through decomposition of organic detritus also contributes to competition for sites and reduced retention of the cesium in all environments reviewed. The large and continous supply of cations that can compete with cesium for sorption sites in the salt marsh soil and vegetation suggests that this environment has the conditions necessary for a high degree of mobility and therefore biological accumulation.

Cesium Inventories at the Study Site

The amount of cesium deposited in precipitation can be calculated from radiostrontium deposition. The radiostrontium fallout for Sterling, Virginia, has been recorded and parallels the fallout measurements for New York City (USEPA, Radiation data and reports). A total inventory of cesium was calculated from New York data (Appendix II) because the record of deposition was more complete. The total inventory deposited from the atmosphere from 1954-1987 is 9.88 pCi/cm^2. This figure was confirmed as accurate of deposition found in undisturbed upland cores for New York City by the Department of Energy, Environmental Measurements Laboratory (Kevin Miller, personal communication) and is approximately that used for Virginia soils, 9.76 pCi/cm^2, by Casey (1985).

The cesium inventory at each coring site by area, weight and volume is presented in Figures 16-18. The inventories by area represent the sum of the activity

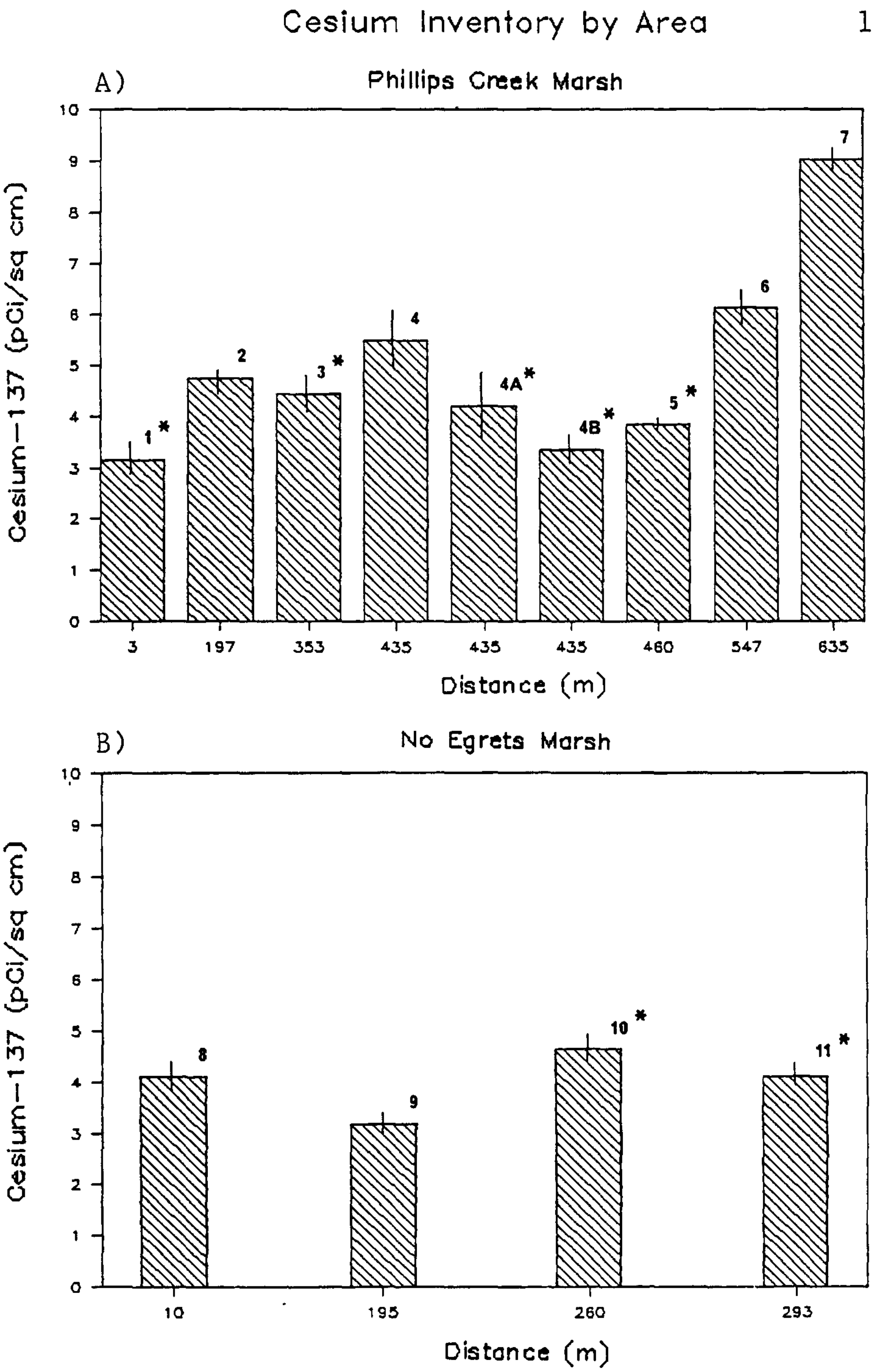


Figure 16 The cesium inventory summed to the total depth of each core divided by the surface area in each location in A) Phillips Creek Marsh and B) No Egrets Marsh. The asterisk means an incomplete inventory.

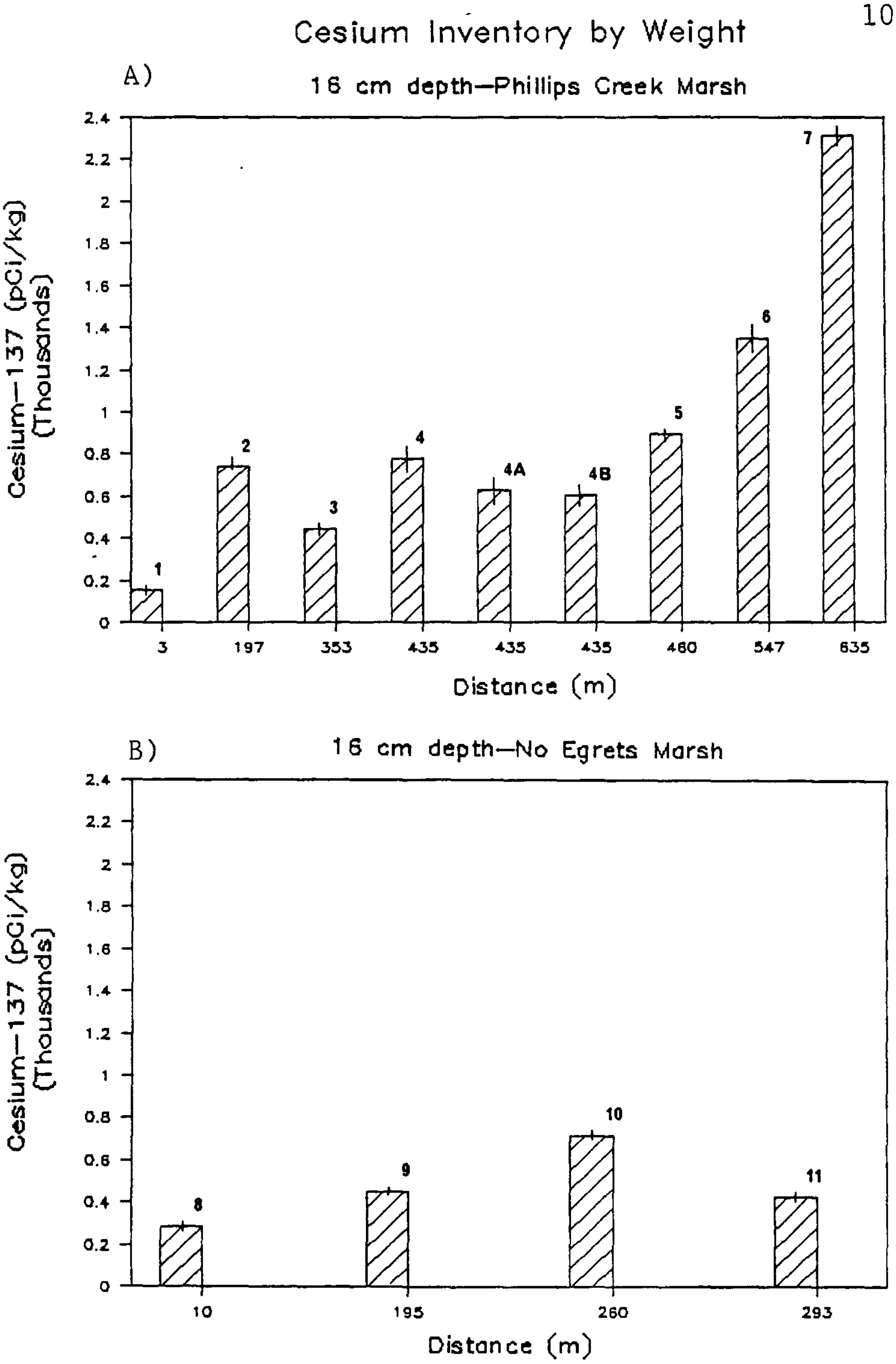


Figure 17 The total cesium inventory by weight in the surface 16 cm of marsh substrate in A) Phillips Creek Marsh and B) No Egrets Marsh.

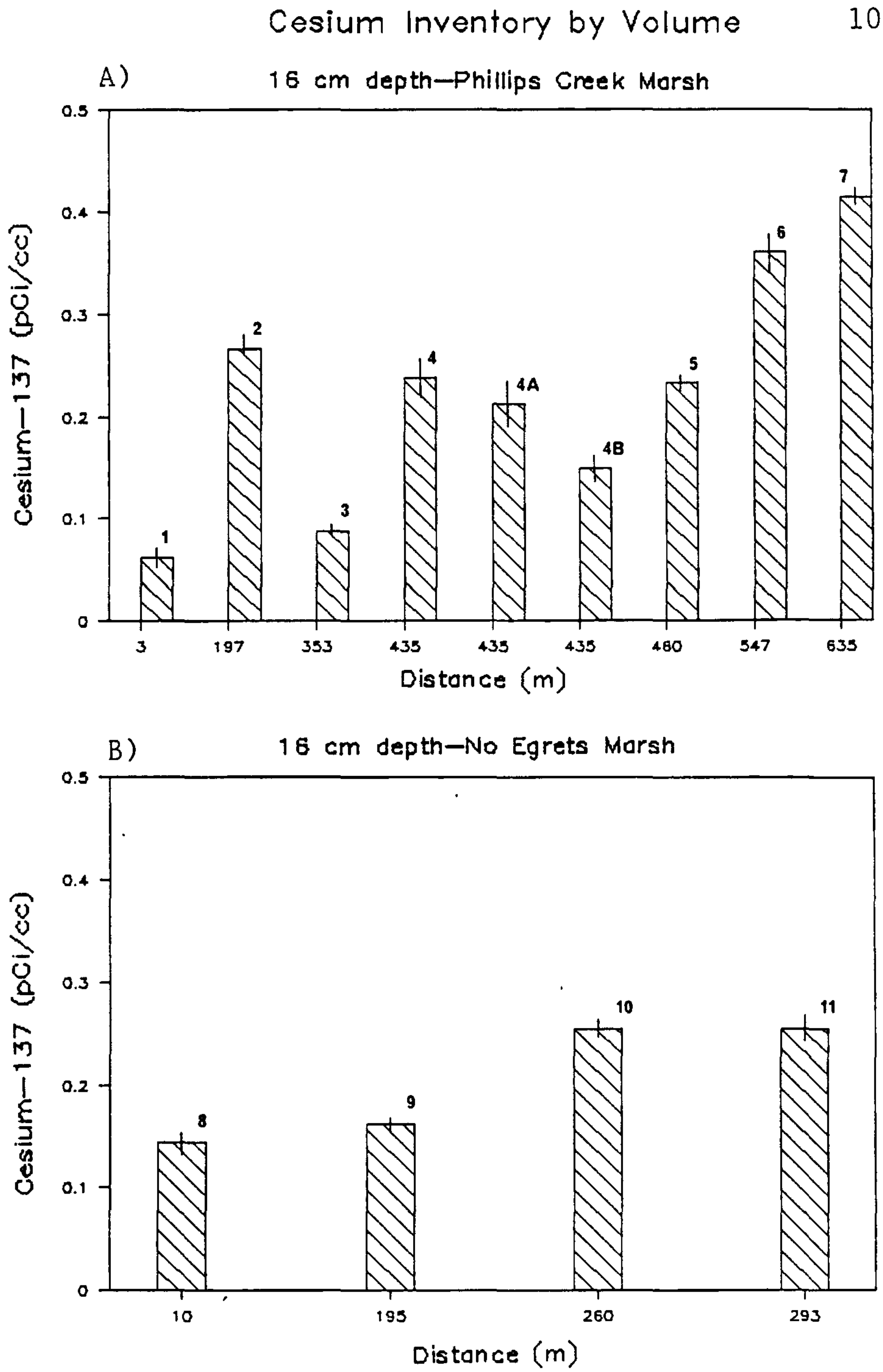


Figure 18 The total cesium inventory by volume in the surface 16 cm of marsh substrate of each location in A) Phillips Creek Marsh and B) No Egrets Marsh.

to the total depth of the core divided by the surface area. The inventories by weight and volume represent the upper 16 cm of marsh substrate. In Phillips Creek Marsh, locations 2, 4, 6, and 7 contain complete inventories (the cesium activity declines to nearly undetectable- < 10 pCi/Kg or < 0.01 pCi/cc- within the sampling interval). Other locations in this marsh, 1, 3, and 5, had activities with depth that remained detectable (incomplete inventories are indicated with an asterisk in Figure 16). The complete inventories indicate an increase in cesium (pCi/cm^2) with increasing distance from the creek bank: the cesium range in pCi/cm^2 at each location respectively is 4.4-4.88, 4.89-6.07, 5.76-6.45, and 8.78-9.26. Thus location 6 may not be statistically different from location 4. Location 7 shows the highest increase in cesium inventory. Comparing the complete inventories by area to the expected inventory, 9.88 pCi/cm^2, location 7 contains most of the cesium deposited from the atmosphere (89-94%) and locations 6, 4, and 2 contain 58-65%, 50-61%, and 45-49%, respectively.

The cesium inventories by weight and volume to a depth of 16 cm are presented in Figures 17 and 18. They show, like the inventory based on area for

Phillips Creek Marsh, an increase in cesium content with distance from the creek bank. This trend holds at 32 cm as well in all cores penetrating to that depth (locations 1,2,4,8,9,10). At a constant depth of 16 cm the expected trend based on the rate of burial (higher at the creek bank and diminishing toward the high marsh) would result in lower cesium inventories at the creek bank. However, the rate of burial does not explain the lower total inventories. Furthermore, the cesium retained by weight and volume tends to be less at the creek bank where the sediment is primarily inorganic.

The three cores taken at location 4 in Juncus roemerianus show slight variations in inventory. The inventory of the 1988 April core (location 4A) is lower than the August 1987 core (location 4) by area and weight (Figures 16-17). The inventory by volume however indicates that they are not significantly different (Figure 18). The difference in the inventories could be due to local variability as well as to the loss of cesium, however the cores were taken within 1m of each other. The core at location 4B was subjected to saltwater drainage and exhibits a lower inventory which is statistically different from 4A only by volume (Based on the counting error, the range of cesium values do not overlap). The interpretation of the trends in cesium inventory will be discussed later in this section.

Cesium Inventories and Data- Other Studies

Delaune et al., 1983, reported the mean ^{137}Cs activity from a levee and backmarsh sample in freshwater, intermediate, brackish, and salt marsh soils of the Barataria Basin, Louisiana: 4.7, 3.5, 1.3 and 0.9 pCi/g respectively. The decrease in cesium concentration from fresh to saltwater shows the lower cesium retention with increasing salinity. The cesium inventories calculated for the intermediate marsh soils (from Figure 2, Hatton et al., 1983) are 11.87, 10.67 and 8.98 pCi/cm^2 for streamside, 20m and 100m respectively. The cesium inventories for the salt marsh soils were considerably lower. Of the two cores from the creek bank, one contained 5.33 pCi/cm^2 and the other contained 5.84 pCi/cm^2. The inland salt marsh cores showed yet even lower inventories, 4.16 and 4.59 pCi/cm^2. In both the intermediate and salt marshes of Louisiana, the inventories decreased with distance from the creek bank. This is contrary to the inventories for Virginia which were found to increase with distance from the creekbank. The expected inventory for Louisiana soils is approximately 4.88 pCi/cm^2 (based on the fallout record for the years 1960-1976, and a correction factor based on the New York record- Appendix II). This shows that the intermediate soils are acting as a sink for the cesium as the actual values are much higher than those predicted from atmospheric precipitation. It is difficult to evaluate depletion or enhancement of the salt marsh sediments because no error bars were given and the inventories are approximately those of atmospheric deposition.

The results of three Virginia cores show an inventory of roughly 50-60% of that expected from records of atmospheric fallout at the Palus Crisium site in Chincoteague Bay (Casey et al., 1986). Casey et al. cited collaboration of their data with information from other marsh sites along the Delmarva Peninsula which also contained less cesium than predicted from atmospheric fallout (C.R. Olsen, Oak Ridge National Laboratories, unpublished data). These results support the data at the study site because of the twelve cores analyzed, none had inventories in excess of that predicted from atmospheric fallout.

The data of Sharma et al. (1987) for salt marshes in South Carolina also show lower inventories than predicted from atmospheric precipitation. The expected inventory for these soils is approximately 5.58 pCi/cm^2 (based on the fallout records for the years 1960-1976, and a correction factor based on the New York record- Appendix 2). The inventories for several South Carolina salt marsh locations were derived from the data, Table 3, of Sharma et al. (1987). Both locations, Town Creek 4 and Bread and Butter 4, had very low inventories, ranging from 1.37-1.78 pCi/cm^2 and 1.62-2.00 pCi/cm^2 respectively. Cesium activity went below detection in these cores and so is interpreted as being the total inventory. These cores were located 10-15 meters from their creek banks and contained at most 32-36% of the activity predicted by atmospheric precipitation. The core closest to Bread and Butter Creek, BB2, was two meters from the creek and had an cesium inventory ranging from 4.53-5.16 pCi/cm^2 (81-92% of the predicted inventory). The data of Sharma et al. clearly shows a higher inventory at the creek bank than inland. The spatial trend is therefore similar to that found by Delaune et al. (1983) in Louisiana but is opposite to that found in the Virginia marshes in this study.

Interpretation of Cesium Inventories

The cesium inventories for salt marshes in Virginia and South Carolina are lower than predictions based on atmospheric precipitation. This suggests that fallout radiocesium was not retained by the sediments after deposition. The high competing cation concentration due to the salinity may not be as effective in Louisiana due to the more rapid accumulation of sediments. This section explores the most feasible explanations for the increasing retention of cesium toward the high marsh found in this study and possible explanations for the reversal of this trend in South Carolina and Louisiana. Three factors will be considered: the influence of the tidal regime on the sorption/desorption of cesium, the retention of cesium by organic matter, and retention based on particle size distribution.

The Tidal Influence

The cesium inventories in this study were found to decrease toward the creek banks which are lower in elevation and are more freqently inundated. The trend in cesium inventories could be explained by

1) a salinity gradient in the marsh
2) the frequency of inundation

both of which would affect the sorption/desorption of cesium.

Normal sea water salinities are known to hinder sorption and initiate desorption as discussed previously. The results of Patel et al., 1978, clearly show that cesium deposited in saline water will be depleted. The absolute levels of cesium in the contaminated sediments from the Trombay clam bed, Bombay Harbor, were found to drop significantly with time. The levels were at a peak in 1971 when the maximum and minimum concentrations were 500 and 165 pCi/g respectively but were considerably lower in 1976 with a maximum of 58 and a minimum of 29 pCi/g. A similar drop was observed over the entire harbor bed (Figure 19) over a period of three years. An average

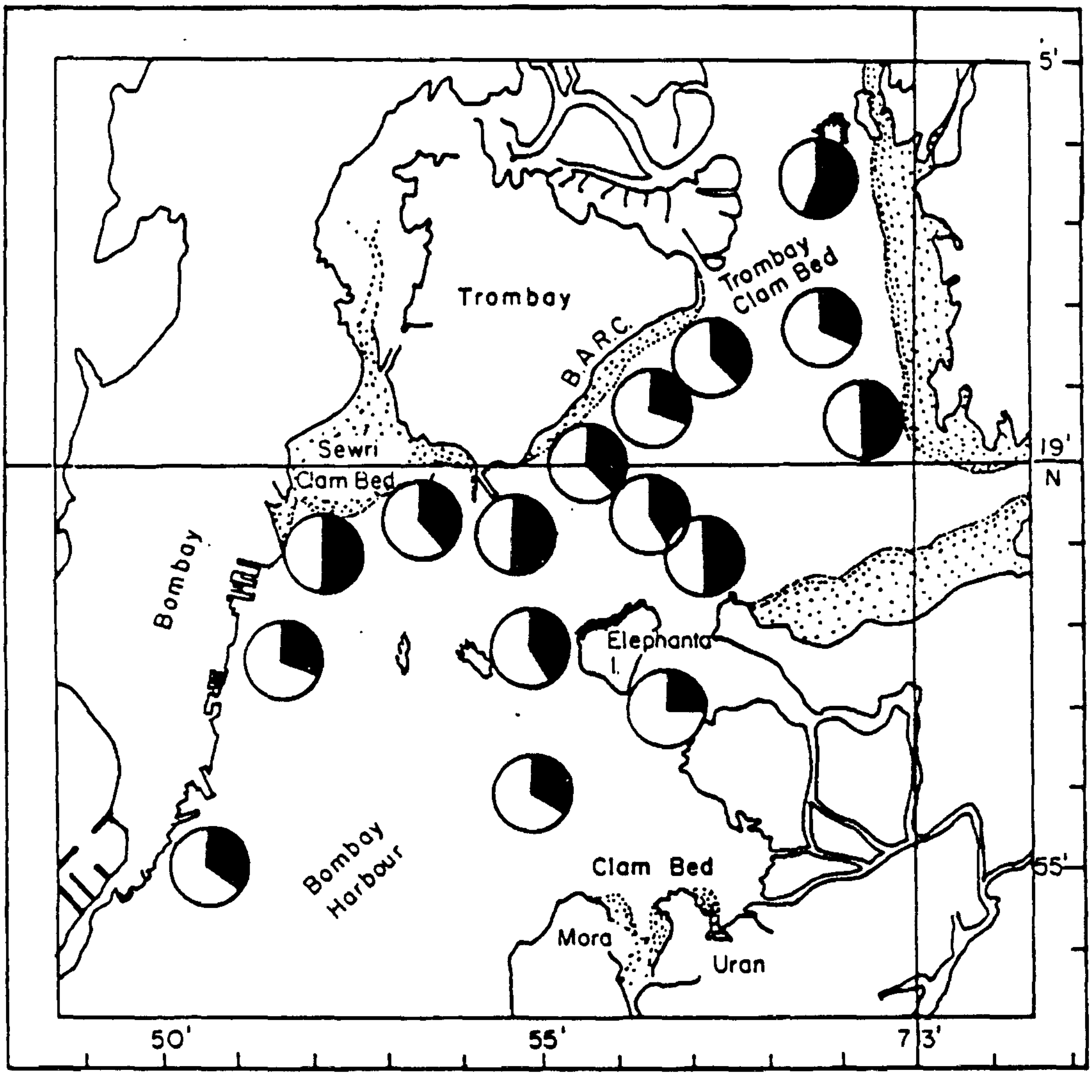


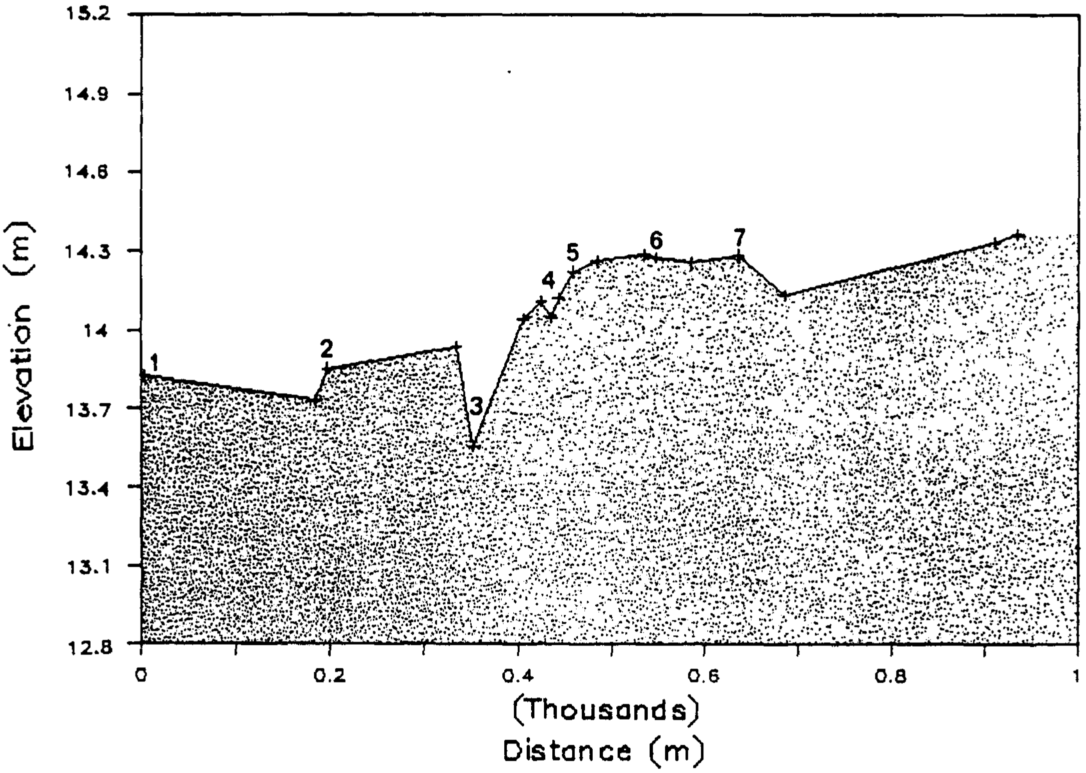
Figure 19 Bombay Harbour clam bed showing the fraction of cesium retained (shaded area) after three years since the initial contamination.

of 53% (range 47%-75%) of the cesium activity was desorbed from sediments up to 30 cm in depth over this time period.

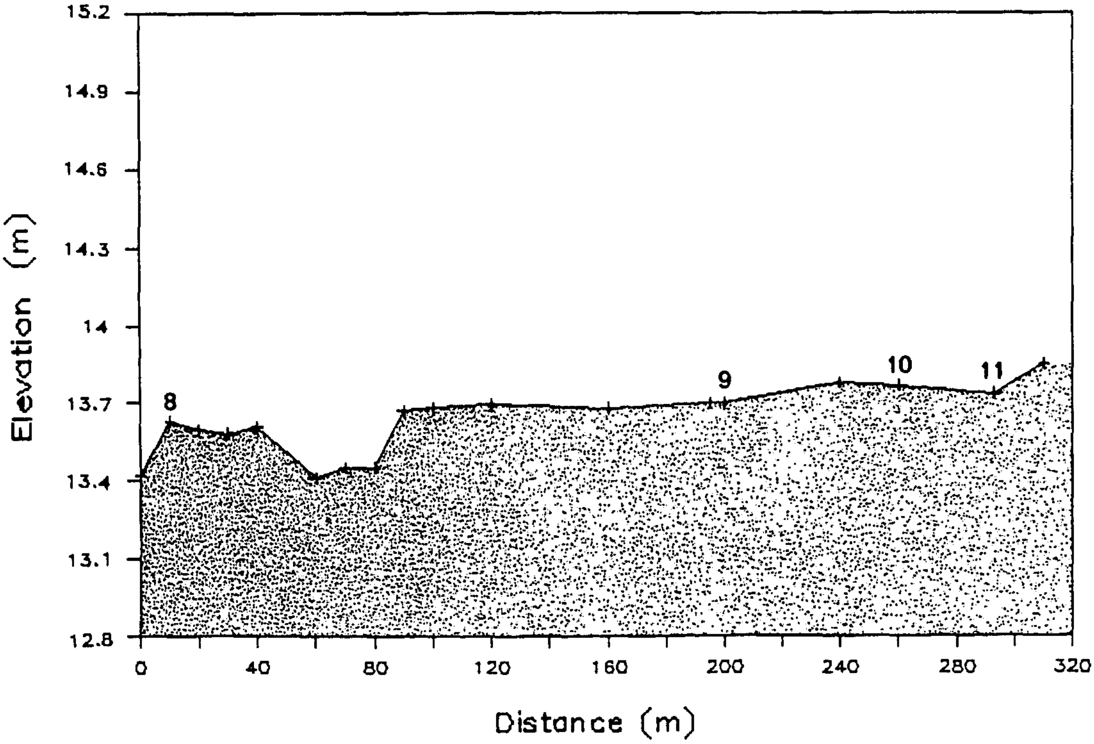
A salinity gradient of saline water at the creek bank moving toward less saline water in the high marsh would mean decreasing sorption and enhanced desorption toward the creek bank. It is feasible that such a gradient exists in these marshes as initial testing of the salinity in Phillips Creek Marsh indicated decreasing salinity toward the back marsh. The organic detritus of the less saline sites would have a greater capacity to retain the cesium based on the results of Schell et al. (1981) which showed that organic detritus accumulated ^{137}Cs in freshwater systems but not in marine systems.

The frequency of inundation and emersion at a site are strongly dependent on its elevation relative to mean sea level. The relative elevations with distance in each marsh are shown in Figure 20 and the original values are reported in Appendix 1. A relationship between emersion and elevation was determined for Bridgeport, Connecticut, where exposure to direct atmospheric input increased rapidly from slightly more than 0.5 for a surface positioned at

Elevation - Phillips Creek Marsh



Elevation - No Egrets Marsh

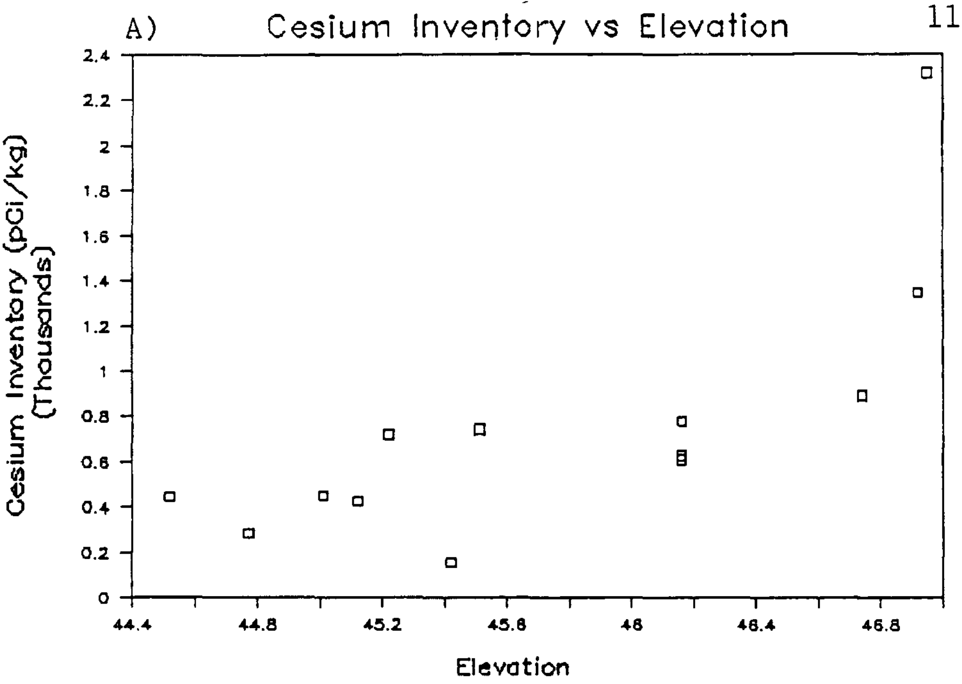


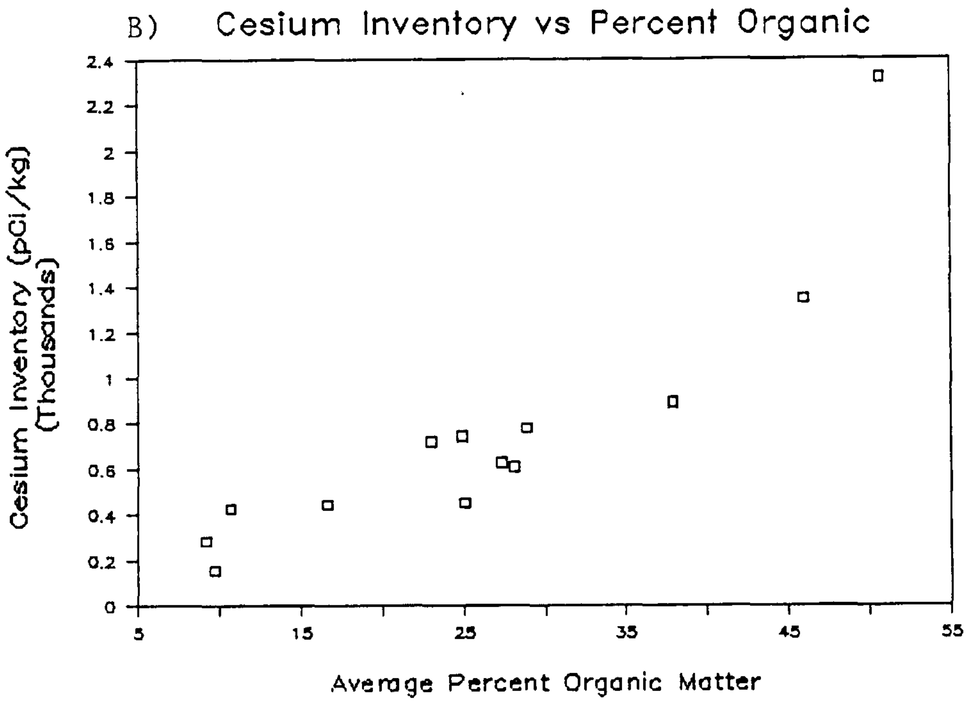
mean sea level to more than 0.9 of the time for surfaces located at MHW (McCaffrey, 1977). By observation, location 7 was inundated only by the highest spring tides and was emersed a large percentage of the time. The juncus-patens area, locations 4 and 5, by the nature of the vegetation and its elevation is approximately at MHW and is probably emersed approximately 50% of the time. The inventory at location 4 was 50-61% of that predicted from atmospheric precipitation. It appears that frequency of inundation or the time a position is exposed to the atmosphere is a primary factor controlling the sortpion/desorption of the cesium. The inventories by area for the Phillips Creek Marsh support this theory as well because it is inundated regularly to the furthest site sampled and has inventories (locations 8-10) that are less than 50% of the expected inventory (Figure 16B).

Emersion is important because of the direct atmospheric input that could occur to the marsh soil and vegetation. The high productivity of the vegetation suggests that a fair amount of cesium will be intercepted by the vegetation and absorbed by the aboveground plant parts. The cesium sorbed during emersion however would be subject to desorption during tidal inundations. At the creek bank the increased frequency of inundation would provide greater competition for binding sites. Since the sorption of cesium into interlayer sites of illite is a slow process requiring months to years (Zucker et al., 1984), the cesium may never become associated with the sediments before being flushed from the system by tides. The highly saline tides would also inhibit fixation of cesium into non-exchangeable interlayer sites and therefore the particles would be more suseptible to cation exchange and desorption. The cesium would be readily available for plant uptake from the soil or absorption from the tidal water. Myttenaere et al. (1969) found that flooded rice absorbed 30 to 100 times more cesium from the water than from the soil. It appears that the percentage of time a location was inundated affected the sorption/desorption of the cesium. The latter proposition is supported by the literature on cesium in a saline environment as previously addressed. According to this theory, the site furthest from the creek bank would be exposed to direct atmospheric input the longest and would be inundated less frequently therefore retaining more of the cesium.

Figure 21A shows the relationship between elevation and cesium inventories in the marshes of this study. In general, the inventories gradually increase but significant retention occurs at the highest elevations in Phillips Creek Marsh. Casey, 1985 predicted decreased inventories where there is relatively rapid flow of pore fluid. At location 3, in the swale of Phillips Creek Marsh, a decreased inventory, as found in this study (Figures 17 and 18), would be expected due to the ground water movement induced by the change in elevation. This is based on the assumption that cesium will be mobile in the saline pore water. Significant downward and lateral flow of pore fluids would be expected at location 1 due to its proximity of 3m from the creek bank. The inventories (Figures 17-18) are the lowest there. This may be partly attributable to the sampling interval though because the cesium curve with depth was just beginning to climax (Figure 33).

The sorption/desorption of cesium based on the frequency of tidal inundations alone however can not explain the reversal of the cesium trends in South Carolina and Louisiana, therefore other factors



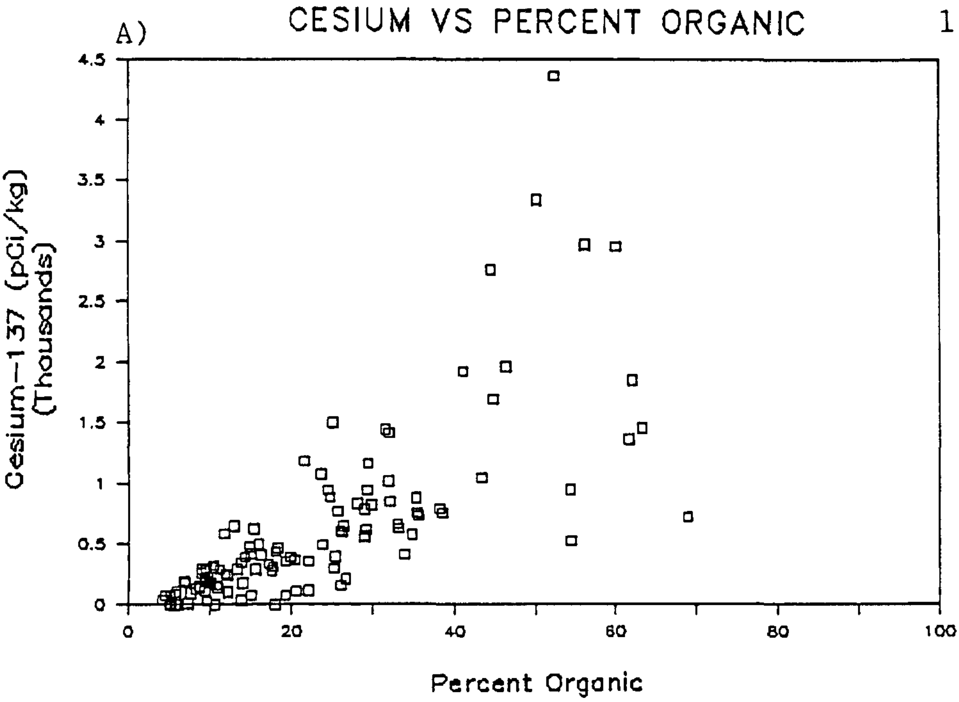


<u>Figure 21</u> The average cesium content by weight in the upper 16 cm of both marshes plotted against A) elevation and B) average percent organic matter.

influencing the spatial distribution of cesium will be considered. As shown in Figure 21B, cesium retention in the marshes of this study is also influenced by the organic matter content and will be considered next.

The Influence of Organic Matter

The data presented thus far shows that cesium retention is suppressed at the creek banks in this study and tends to reach higher concentrations toward the back marshes. The organic matter content of the sediment also increases toward the back marshes. Figure 22A shows an increase in cesium content with an increase in organic matter content by weight. In Figure 23A, the natural log of cesium is plotted against percent organic matter (and indiates a relationship with volume over surface area). At high inorganic contents such as are found near the creek banks, the cesium retention is low. It increases as lower inorganic/higher organic contents are approached. Since an increase in porosity is associated with the organic matter, the cesium content also increases with an increase in porosity as shown in Figure 22B. The natural log of the cesium content



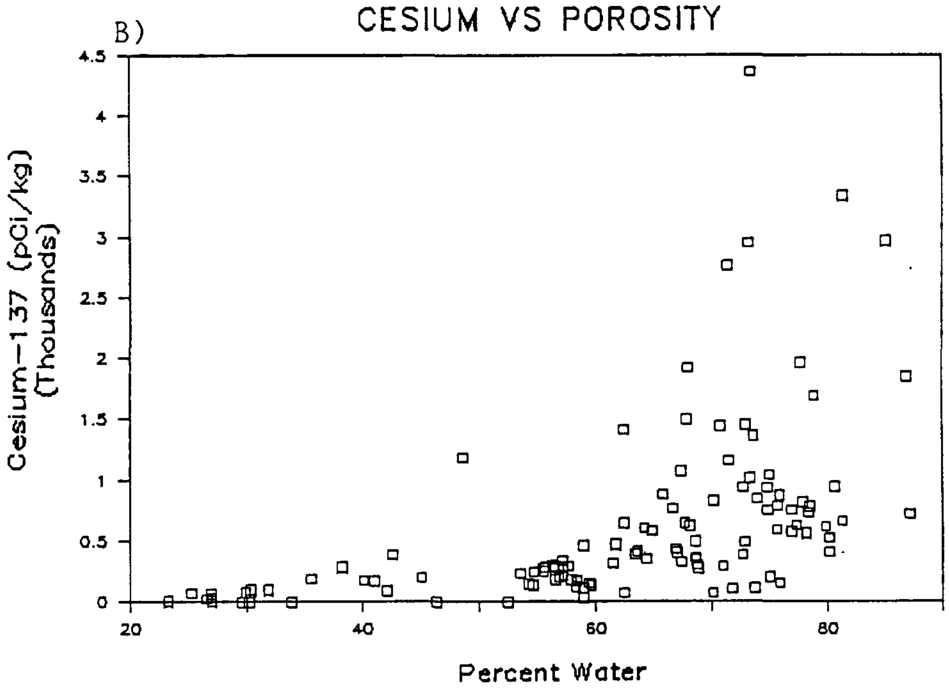


Figure 22 A) The cesium values by weight plotted with percent organic matter by weight in both marshes.

B) The relationship of cesium content by weight plotted against percent water in both marshes.

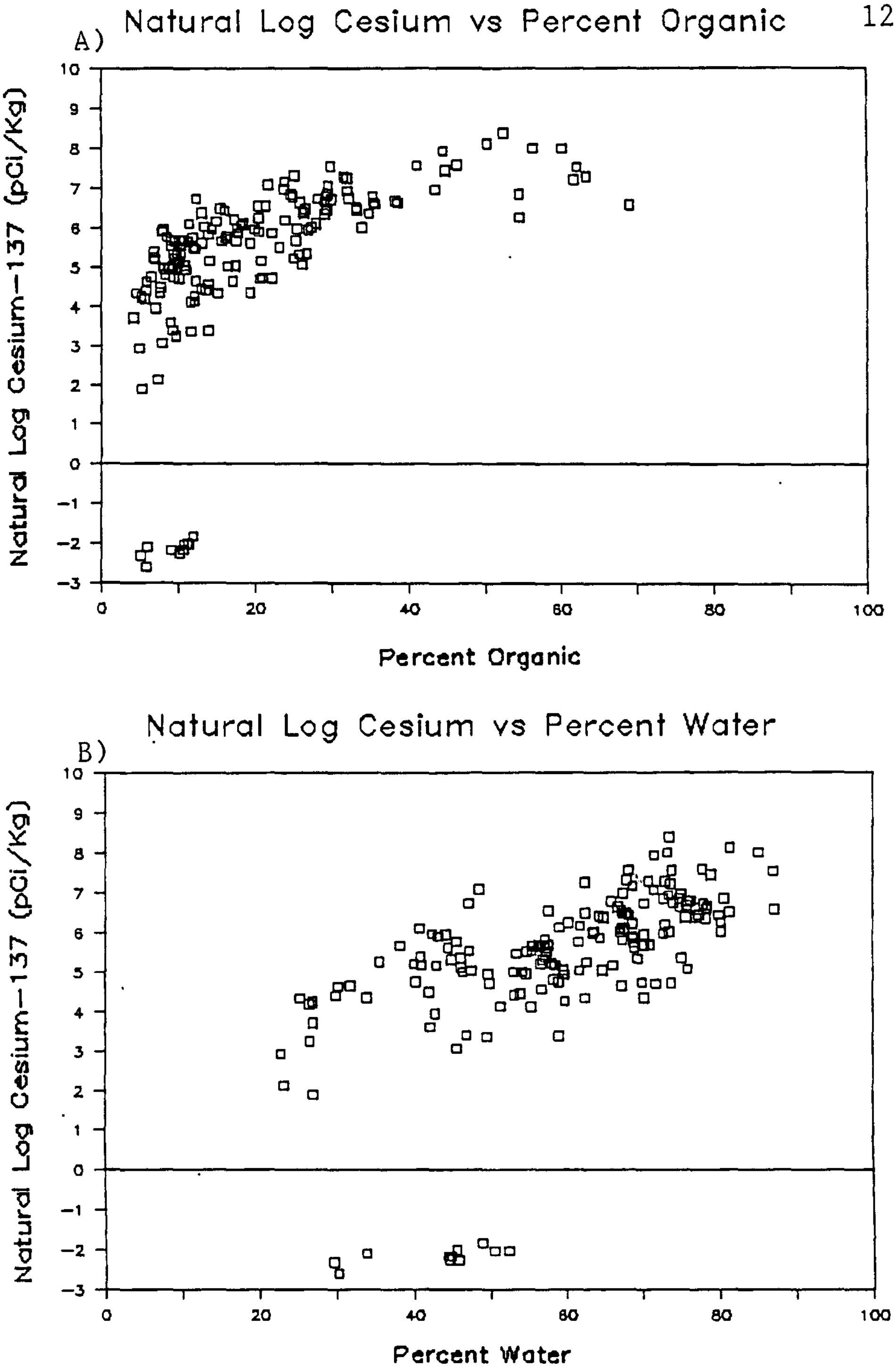
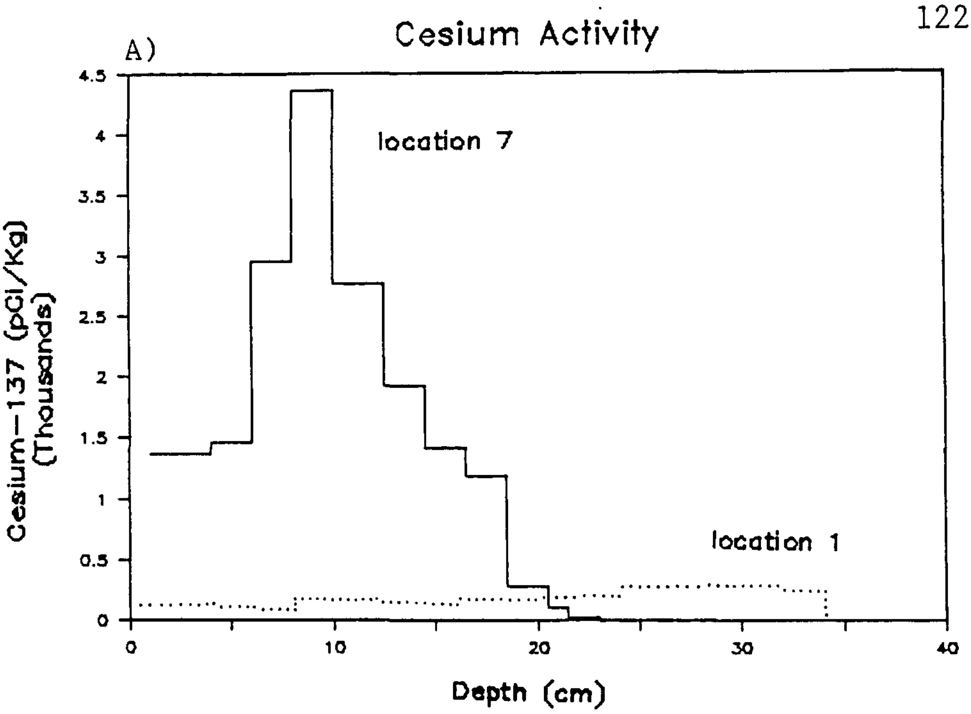


Figure 23 The same values as in Figure 22 but with the natural log of the cesium values.

is plotted against the water content in Figure 23B. The average organic matter content in the surface 16 cm's of Phillips Creek Marsh increases with distance from the creek bank (Figure 14). The fluctuations mimic the cesium inventories by weight with distance in each marsh (Figure 17). The anomaly in cesium retention at location 2 on a volume basis (Figure 18) corresponds to an anomaly in organic matter content.

The magnitude of the difference in cesium retention between the creek bank and high marsh in Phillips Creek is shown in Figure 24A. These depth profiles of cesium by weight show that cesium is consistently low, ranging from 115-297 pCi/kg near the creek bank to a high of 2950 in the root mat of the high marsh, location 7, with a low value of 8.41 pCi/kg in the underlying clay layer. The root mat appears to be retaining the cesium. Although plant material enhances cesium mobility through the production of ammonium, the cesium can be sorbed to peat containing high concentrations of sodium and potassium. The K^{40} data indicates that the potassium retained by the root mat on a weight basis is higher than the potassium retained by the underlying clay layer (Figure 24B). The sections were eqivalent on a



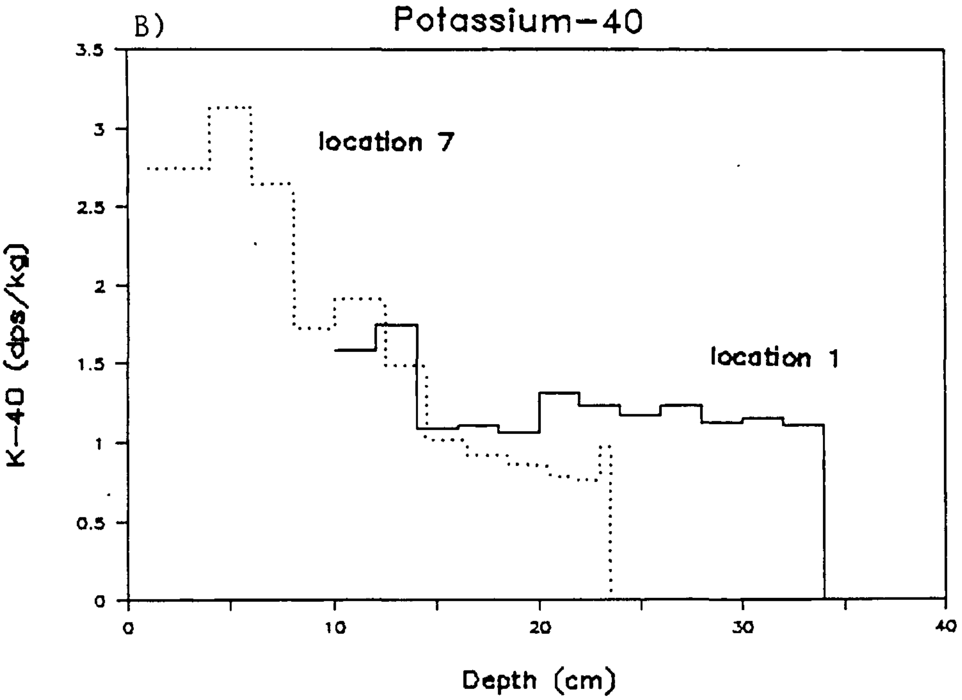
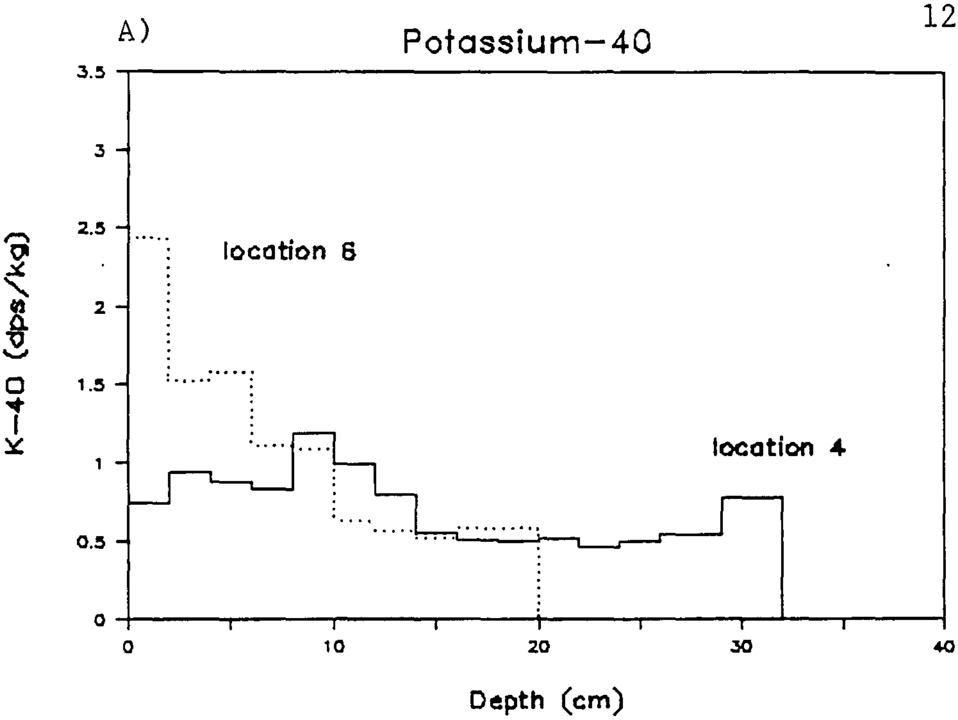


Figure 24 A) The cesium content with depth at location 7 compared to the distribution of activity at location 1.

B) The potassium-40 content by weight with depth at locations 1 and 7.

volume basis. Thus the initial sorption of cesium through cation exchange with potassium in the root mat was probably high. The high retention of potassium by root mat is supported at other locations as well. Location 6 has a higher potassium content by weight in the top 10 cm than location 4 (Figure 25A) and the potassium content of location 5 is greater than at location 2 (Figure 25B). No intercomparisons between graphs can be made because different detectors were used and correction factors were not applied. Both potassium and cesium held in the root mat would be subject to uptake by the vegetation.

The cesium could be retained _in situ_ by cycling through the above- and belowground production. The translocation of cesium to aboveground parts would be expected because cesium is metabolically similar to potassium which is translocated from belowground roots and rhizomes to aboveground plant tissue. The high belowground productivity in the marsh would result in substantial uptake of ^{137}Cs by plants, particularly since the cesium is usually found in the upper 30 cm of sediment which also contains the root zone. Cycling of the cesium through the vegetation and the retention of cesium by organic matter is feasible. A



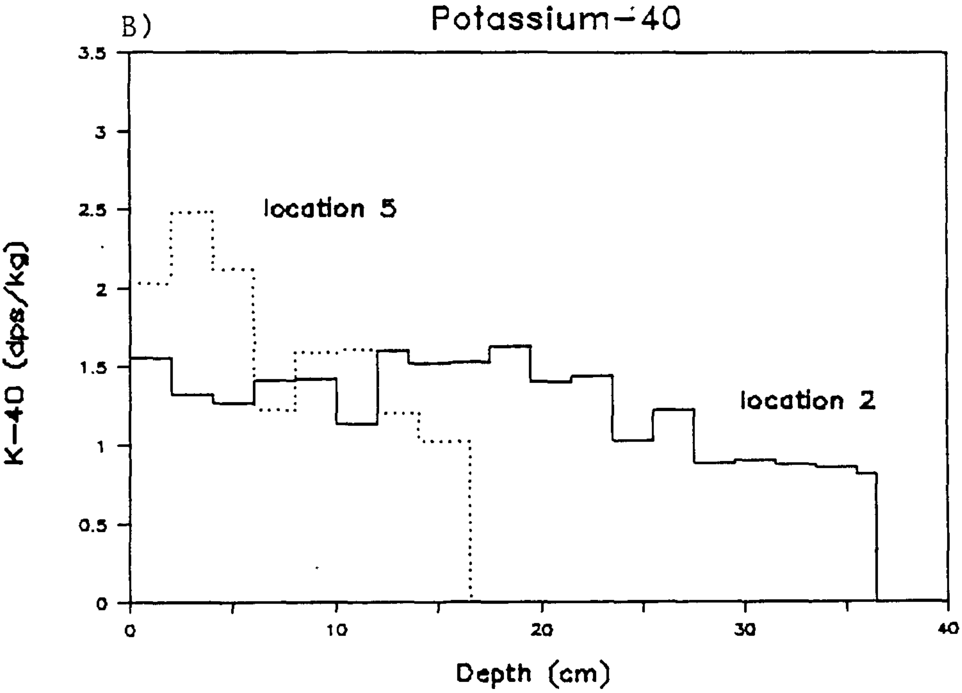


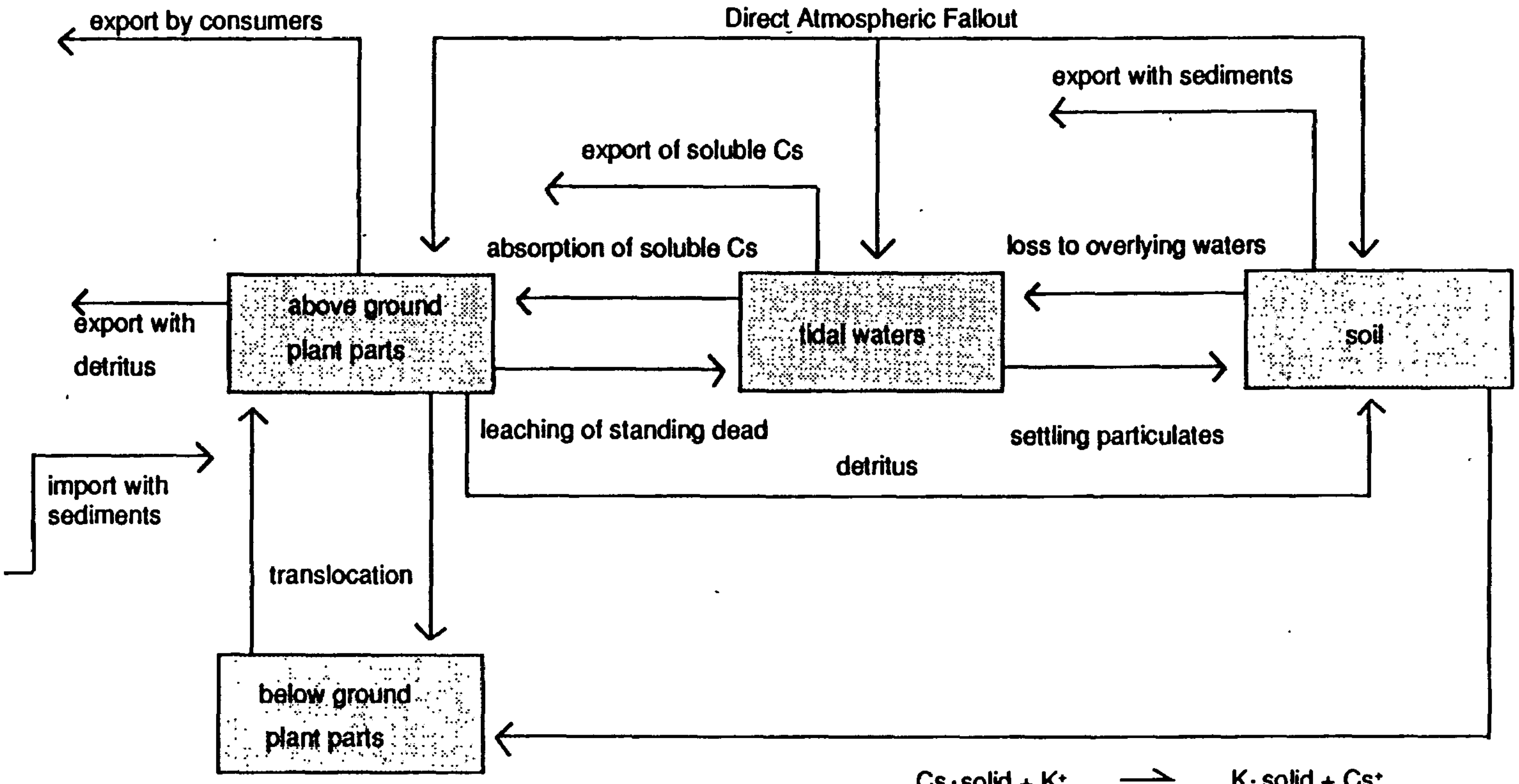
Figure 25 The potassium-40 content by weight with depth in A) locations 6 and 4 and B) locations 5 and 2.

summary of fallout ^{137}Cs cycling in salt marsh sediments is presented in Figure 26. Organic detritus removed from the site would mean an export of cesium and a decrease in the inventory as would export by consumers, export of soluble cesium in the tidal water and export of sediment containing cesium.

The uptake of cesium by plants explains the lower inventories toward the creek bank through the greater export of detritus and the higher rate of decomposition of organic matter. As reviewed previously, many studies support the retention of cesium by organic matter (Barber, 1964; Brisbin et al., 1974; Ritchie et al., 1978; Mattson, 1975). In the saline environment, the cesium held in organic matter will be desorbed but can be retained by the vegetation.

According to Gillham et al (1980), the distribution coefficient for a trace constituent is directly proportional to the cation exchange capacity (CEC) and is inversely proportional to the concentration of the competing cation. This relationship is based on the mass-action equation which describes the exchange of two monovalent cations, one in trace concentrations. Although this

Fallout ^{137}Cs Cycling In Salt Marsh Sediments



$$\mathrm{Cs\cdot solid} + \mathrm{K^+} \rightleftharpoons \mathrm{K\cdot solid} + \mathrm{Cs^+}$$

$$\mathrm{Cs\cdot solid} + \mathrm{NH_4^+} \rightleftharpoons \mathrm{NH_4\cdot solid} + \mathrm{Cs^+}$$

Figure 26 The cycling of cesium-137 in a salt marsh environment.

relationship is too simplistic to express the complex interaction of cations in the salt marsh environment, it does indicate the importance of the CEC in the sorption of cesium. Eagle (1963) showed that organic matter contributes to the CEC of soils by determining a correlation coefficient of 0.733 between exchangeable potassium content and organic matter. Eagle also found that the addition of peat increased the CEC of terrestrial silt-loam soils. He contended that on soils of similar texture and geological origin, the CEC will be closely correlated with the organic matter content. Brady (1984, p.179) cites average figures for the CEC of silicate clay, 0.5 cmol/kg for each 1% by weight, and for well-humified organic matter, 2.0 cmol/kg for each 1%. These averages indicate that in the marshes of this study the CEC will tend to increase with an increase in organic matter content. The cesium would be retained to a greater extent at sites having high cation exchange capacities and therefore, larger distribution coefficients for the sorption of cesium. A correlation of ^{137}Cs with organic matter and potassium was found by Brisbin et al. (1974) in the soil at the Savannah River Plant, South Carolina. Duurma and coworkers found that the distribution coefficients correlated well with the exchangeable potassium content of marine sediments (Onishi et al., 1981). If the sediment had high concentrations of exchangeable potassium, the distribution coefficient was large.

Although two major mechanisms for explaining the distribution of cesium inventories have been proposed, the number of variables affecting cesium distribution makes determining the distribution in other marshes, particularly a reversal in the trend difficult. However, both the Louisianna and South Carolina marshes had well defined levees and dense vegetation at the creek bank compared to inland sites (Delaune et al., 1979; Sharma et al., 1987). In contrast, the creek bank Spartina in the Virginia marshes studied was sparse, with the densest vegetation farther back in the marsh. The reversals could be a results of initial differences in the sorption of cesium by the aboveground plant parts. The transects in this study covered longer distances and a wider range of vegetation.

Differential cesium retention based on particle size is most likely not significant in the marshes of this study due to the homogeneity of the sediment.

The marsh sediment cored at the study sites consisted primarily of silts and clays. Although K^{40} has been used to indicate illite clay content in other studies, the high potassium content of the organic matter in this study indicates that it may delineate sites of higher organic matter content rather than of finer particle size. For example, the K^{40}/kg to a depth of 16 cm in No Egrets Marsh is 1.07, 1.12, and 0.84 for locations 8, 10 and 11 respectively. Thus the panne area in the No Egrets Marsh shows a slightly higher K^{40} content per kilogram than the other sites in this marsh. The panne area would be expected to have a high proportion of fine clay minerals because of the increased time for settling provided by the ponded water. However, this site also has the highest average organic matter content of the three sites as shown in Figure 14B. Furthermore, the incorporation of cesium into the matrix of illite suggests that a relationship with volume rather than surface area is important for these sediments.

The cesium inventories in the marshes of this study can be explained primarily by the frequency of tidal inundations and the organic matter content. The primary role of plants in cesium cycling can occur only if the cesium is relatively mobile. Further evidence of mobility is seen in the cesium profiles with depth which are analyzed next.

Cesium Profiles and Peaks

This section presents the cesium profiles and cites evidence for the mobility of cesium in addition to that indicated by the inventories. The cesium distribution with depth is shown for each location in figures 27-33. A standard curve of cesium deposition has been predicted for each location based on the fallout data and a sampling resolution of 2 cm. The predicted cesium distribution is based on the assumption that the 1963 peak may decrease in magnitude but does not shift position. Both the actual and predicted curves are normalized (each quantity is divided by the total sum) such that the area under each profile is one. Many of the predicted peaks terminate above 0.4, the upper limit of the graphs. The predicted distribution was derived by assuming that the peak in the actual profile is 1963, obtaining an accretion rate, and summing the number of years represented by that rate based on the sampling interval. Calculations are presented in Appendix II. The actual cesium distribution shows in every case, except location 1, a tail of activity below that predicted and in every core, except location 7, a much higher cesium activity than predicted in the intervals above each peak. This apparent peak broadening could be caused by diffusion in both directions, sediment mixing and cesium deposited from the water column rather than directly from the atmosphere.

The higher cesium content of the intervals above each peak is significant because the cesium inventories suggest that only a fraction of the predicted cesium should have been deposited. The upward mobility of the cesium can be characterized by considering the peak-to-surface ratio of each profile (Miller and Heit, 1986). The ratio of the peak inventory to the surface inventory can be used to test for the degree of mixing or delayed wash-in. Miller and Heit defined the surface inventory as the baseline data from 1975-1980 and the peak inventory as being the 1963 maximum. The peak-to-surface ratios are higher in the predicted curves than in the actual curves suggesting that the cesium has either been mobilized from depth or is a delayed response due to sediment deposited from the water column. If the peak-to-surface ratios are considered for the No Egrets Marsh, the predicted curves would have ratios of 7.7, 11.9, 6.6, and 14.1 for locations 8-11. The surface cesium was determined by considering the interval 4 cm from the peak for locations 8 and 9, and 2 cm from the peak for locations 10 and 11. A span of years for the surface ratio could not be used because of the sampling resolution and cesium peaks near the surface. Using this same method, the values for the actual ratios are 1.5, 2.8, 2.0, and 2.3, respectively. The peak-to-surface ratios emphasize that remobilization of the cesium has occurred.

Every cesium profile reviewed by Miller and Heit (1986) for lake sediment cores showed excess ^{137}Cs in the layers corresponding to the post peak fallout period. The surface enhancement of the cesium content was interpreted as due to delayed wash-in of sediment from the watershed. The fact that every cesium profile but one (location 7) in this study also showed this enhancement suggests that sediment deposited from the water column or wash-in is also important to the cesium content of the soils in salt marshes or that cesium was retained near the surface by recycling through the above- and belowground production. If mixing alone were responsible for the enhancement, locations 5 and 6 would most likely not show enhancement due to the thick root mat and lack of bioturbation at these sites.

The downward transport of cesium can be characterized by comparing the sedimentation rates based on the beginning of cesium deposition, 1950, with the rate based on the 1963 peak (Miller and Heit, 1986). The rates using this approach have been summarized in Table V. A value close to 1 indicates little or no peak broadening.

TABLE V
SEDIMENTATION RATES BASED ON CESIUM DISTRIBUTION

Location	Sedimentation rate (mm/yr)		Ratio 1950/1963
	1963	1950	
1	13.2	---	---
2	3.8	8.2	2.17
3	6.7	---	---
4	1.3	6.8	5.20
5	2.9	---	---
6	2.9	4.6	1.58
7	3.8	>6.2	>1.64
8	6.3	7.3	1.16
9	2.9	7.3	2.52
10	2.1	>8.4	>3.99
11	1.3	>4.1	>3.15

The locations exhibiting the lowest values for downward diffusion based on the 1950/1963 ratio are locations 6 and 7 in Phillips Creek Marsh and location 8 in No Egrets Marsh. The high ratio for location 4 suggests significant mobilization - either the 1963 peak has moved up and/or the cesium has diffused to greater depths. The variability in the 1963 peaks between locations 4, 4A, and 4B (2-4 cm, 0-2 cm, 4-6 cm, Table VI), also suggests post-depositional mobility at this location. Table VI lists the depth to each peak and the accretion rate by weight and volume assuming that the peak represents maximum fallout in 1963. In addition to the variability of accretion rates between cores at location 4, the lack of resemblance between the predicted and actual cesium profiles suggests that the actual distribution no longer reflects atmospheric fallout and that an accretion rate should not be predicted using ^{137}Cs at this location.

TABLE VI
CESIUM PEAKS AND ACCRETION RATES

Location	Depth to peak (cm)		Accretion Rate*(mm/yr)	
	by wt	by vol	by wt	by vol
1	30-32	32-34	12.4	13.2
2	10-12	10-12	3.8	3.8
3	13-15	13-15	6.7	6.7
4	2-4	2-4	1.3	1.3
4A	0-2	0-2	0.4	0.4
4B	4-6	4-6	--	--
5	4-6	6-8	2.1	2.9
6	6-8	6-8	2.9	2.9
7	8-10	8-10	3.8	3.8
8	16-18	14-16	7.1	6.3
9	6-8	6-8	2.9	2.9
10	4-6	4-6	2.1	2.1
11	2-4	2-4	1.3	1.3

* assuming the cesium peak in the sediments to represent the peak fallout in 1963.

Although location 7 has 89-94% of the cesium deposited from the atmosphere, the magnitude of the cesium peak is lower and the cesium is distributed to lower depths. The actual cesium distribution shows a double peak whereas the predicted cesium distribution indicates a single peak (Figure 27A). Observation of this core prior to processing showed a dense clayey section beginning at 16.5 cm. Thus, the preferred retention is most likely due to the high inorganic content of the sediment and the subsequent decrease in porosity. The high inorganic content is also

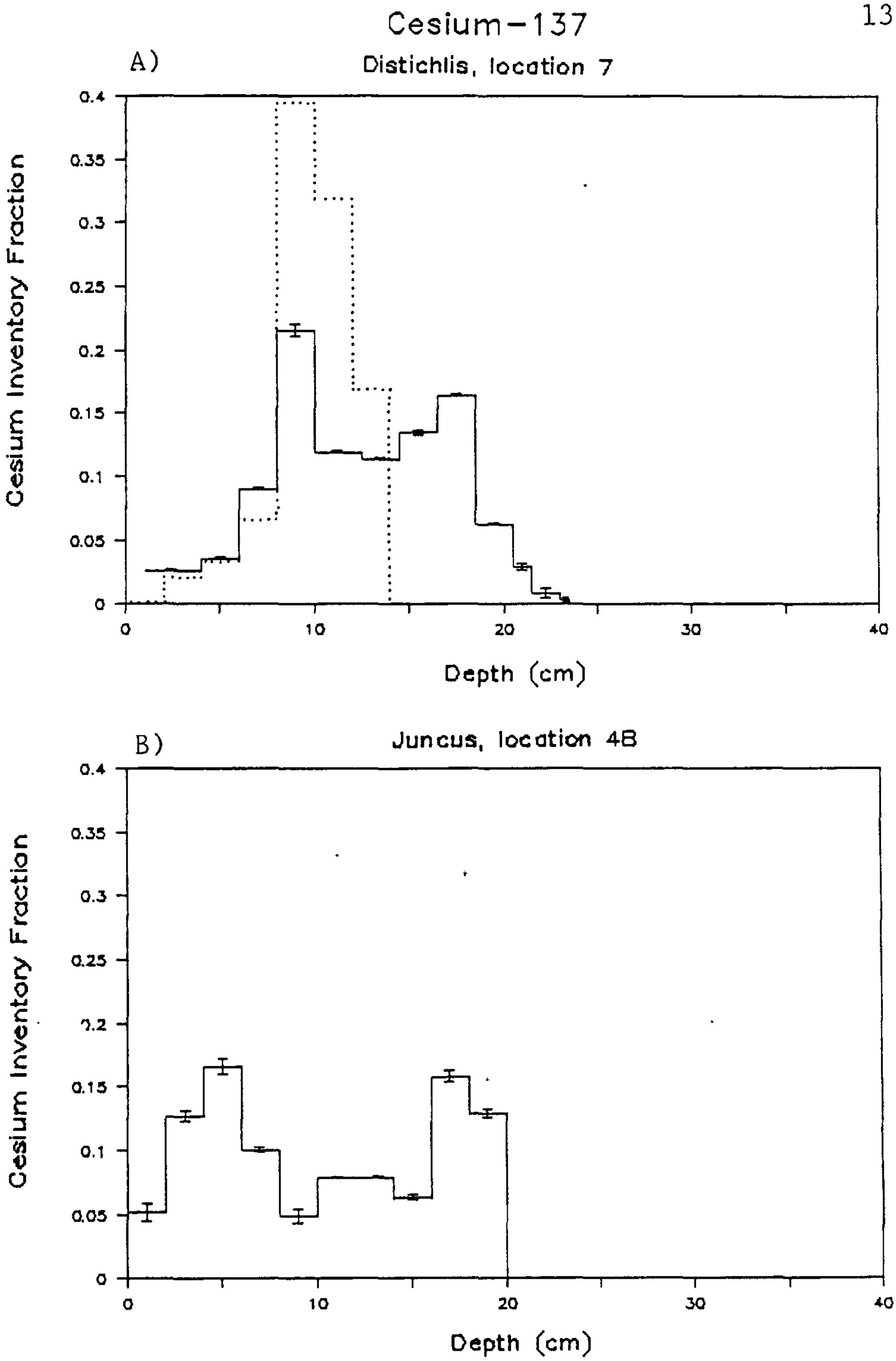


Figure 27 The cesium inventory fraction by volume with depth at A) location 7 and B) location 4B. The solid line is the actual profile while the dotted line is a predicted curve.

suggested by an increase in bulk density at this depth. Since the second peak is not represented by atmospheric fallout and bioturbation of the thick root mat is unlikely, chemical mobility is the probable avenue of relocation. The cesium distribution in the artificially drained core (Figure 27B) also shows a double peak with preferential retention at a depth of 16-18 cm, the approximate depth of the dense clayey substrate. The other cores from location 4 showed different cesium distributions (Figure 28). The reason for the additional peak at the depth interval of 12-14 cm in location 5 is uncertain as the observed clay layer was deeper (Figure 29A). A small second peak is noticeable at a depth of 14-16 cm in location 6 (Figure 29B).

Since locations 5, 6, and 7 are on the high marsh "plateau", their rates of accretion should be similar and are close as shown by the data in Table VI- 2.9, 2.9, and 3.8 mm/yr respectively and in Figures 27 and 29. The average of the three locations gives an accretion rate of 3.2 mm/yr which is very close to the rate of sea level rise for Virginia, 3.4 mm/yr (Boon, 1978). According to the cesium peaks locations 7 and 2 have the same accretion rate. This is contrary to

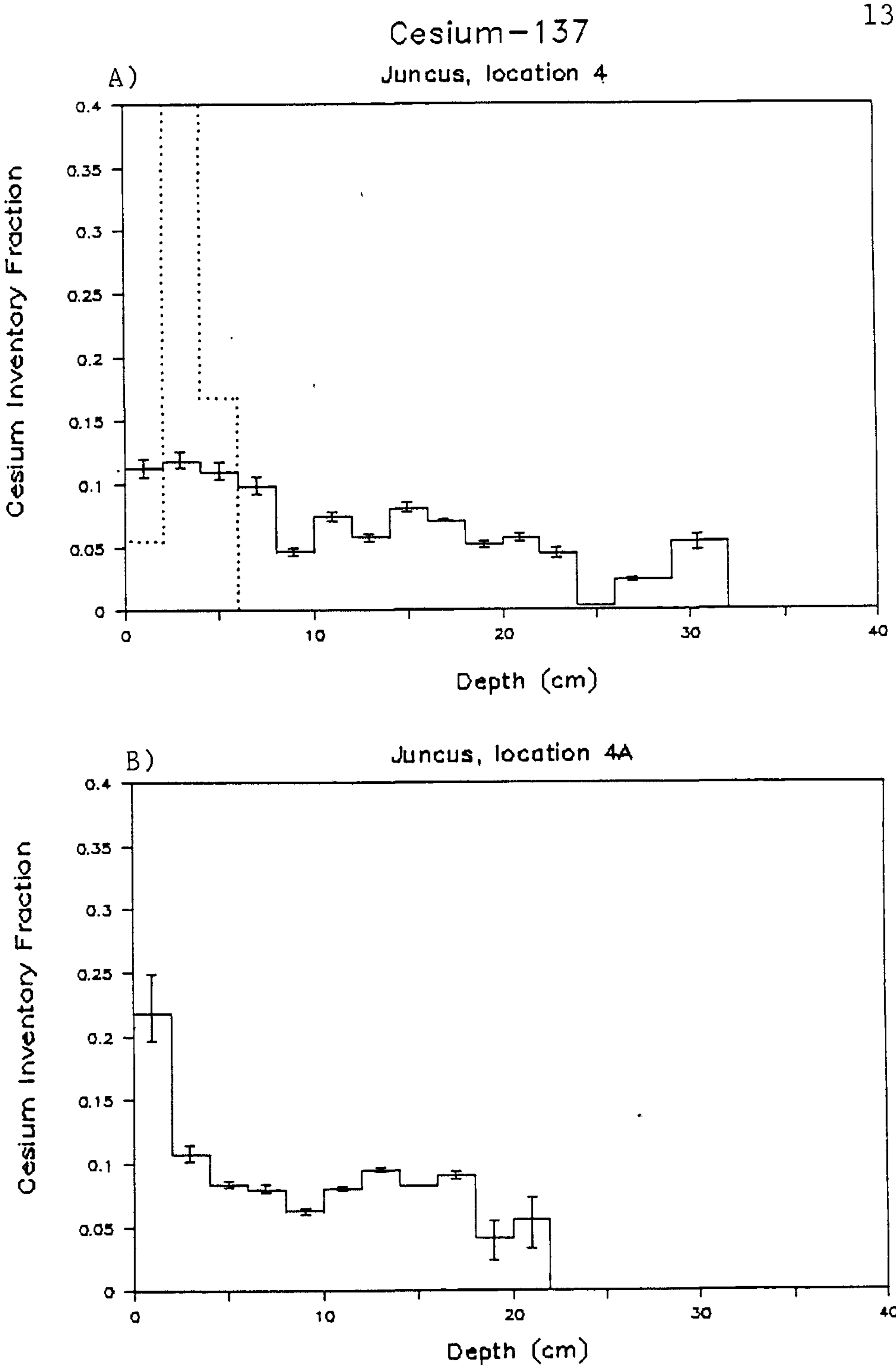


Figure 28 The cesium inventory fraction by volume with depth at A) location 4 and B) location 4A. The solid line is the actual profile while the dotted line is a predicted curve.

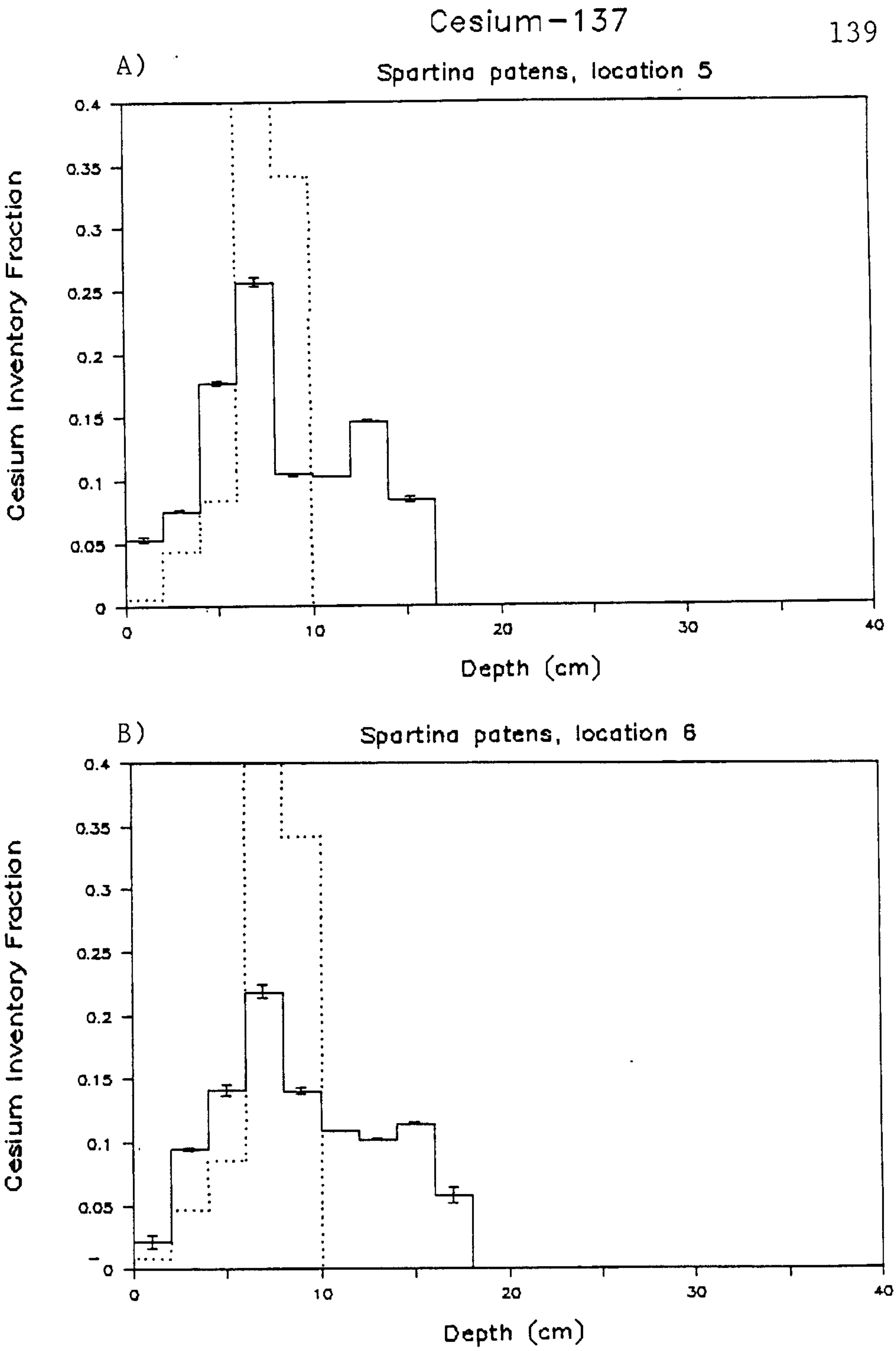


Figure 29 The cesium inventory fraction by volume with depth at A) location 5 and B) location 6. The solid line is the actual profile while the dotted line is a predicted curve.

the observed pattern of deposition in salt marshes (Table I) which shows a general decrease in sedimentation away from the creek bank. If the accretion rate for location 7 is approximately correct, then the accretion rate at location 2 is under-predicted (Figures 27A and 30A). If location 2 is under-predicted and location 4 is uninterpretable showing essentially a monotonic decrease then the accretion rate for location 3 of 6.7 mm/yr may be appropriate and downward mobility of the cesium peak need not be invoked (Figure 30B).

The broad cesium distribution at location 8 along with a constant porosity with depth suggests that bioturbation has been a key factor in cesium remobilization (Figure 31A). The 1950-1963 ratio however indicates that the accretion rate of 6.3 mm/yr may be correct. The long tail documented at location 10 coupled with a high 1950-1963 ratio suggests significant downward diffusion of cesium (Figure 31B). Downward diffusion is also significant at locations 9 and 11 (Figures 32-A and B).

Based on atmospheric fallout and a fast accretion rate (13.2 mm/yr) at the creek bank, there should not be detectable activity on the surface sediments of

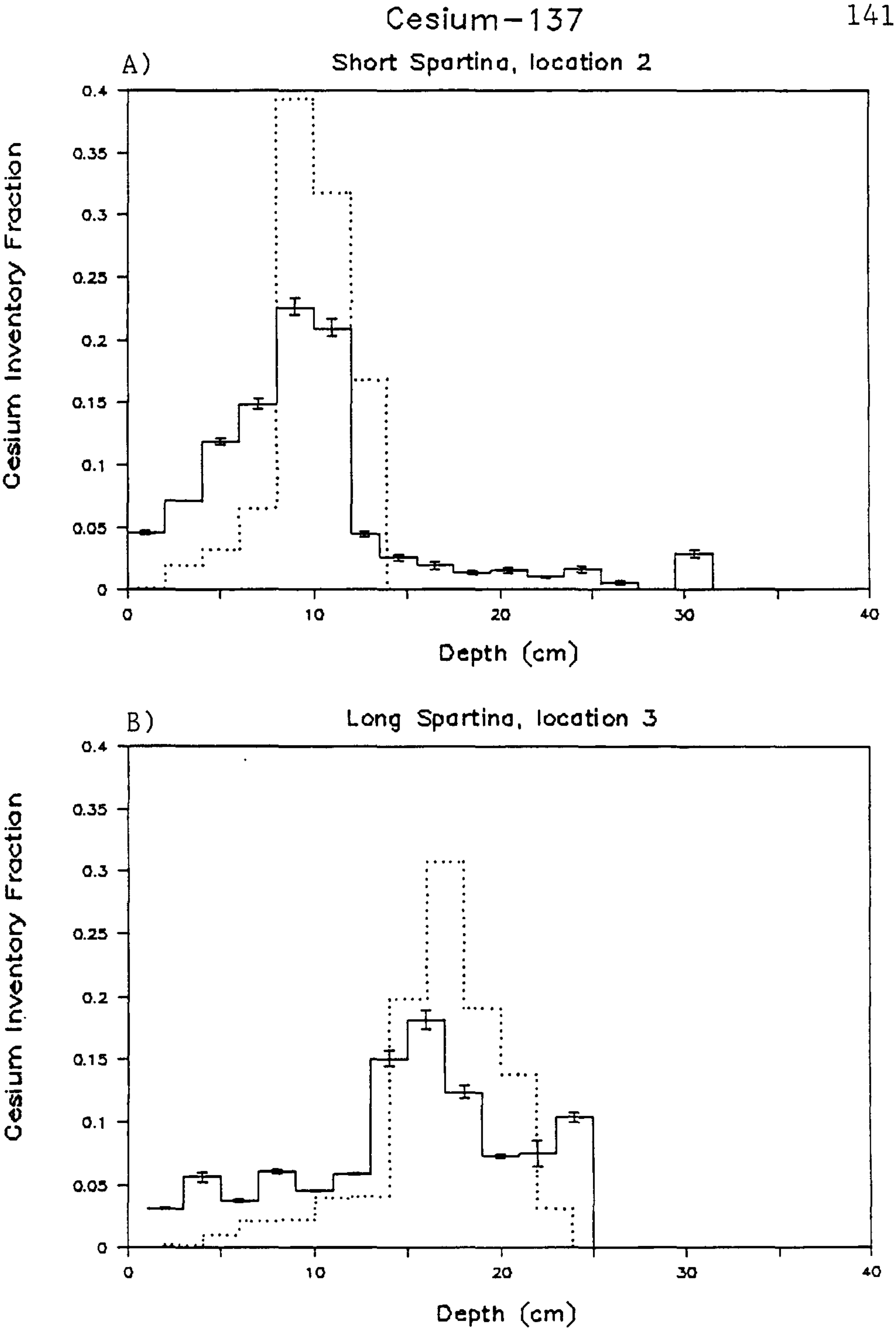


Figure 30 The cesium inventory fraction by volume with depth at A) location 2 and B) location 3. The solid line is the actual profile while the dotted line is a predicted curve.

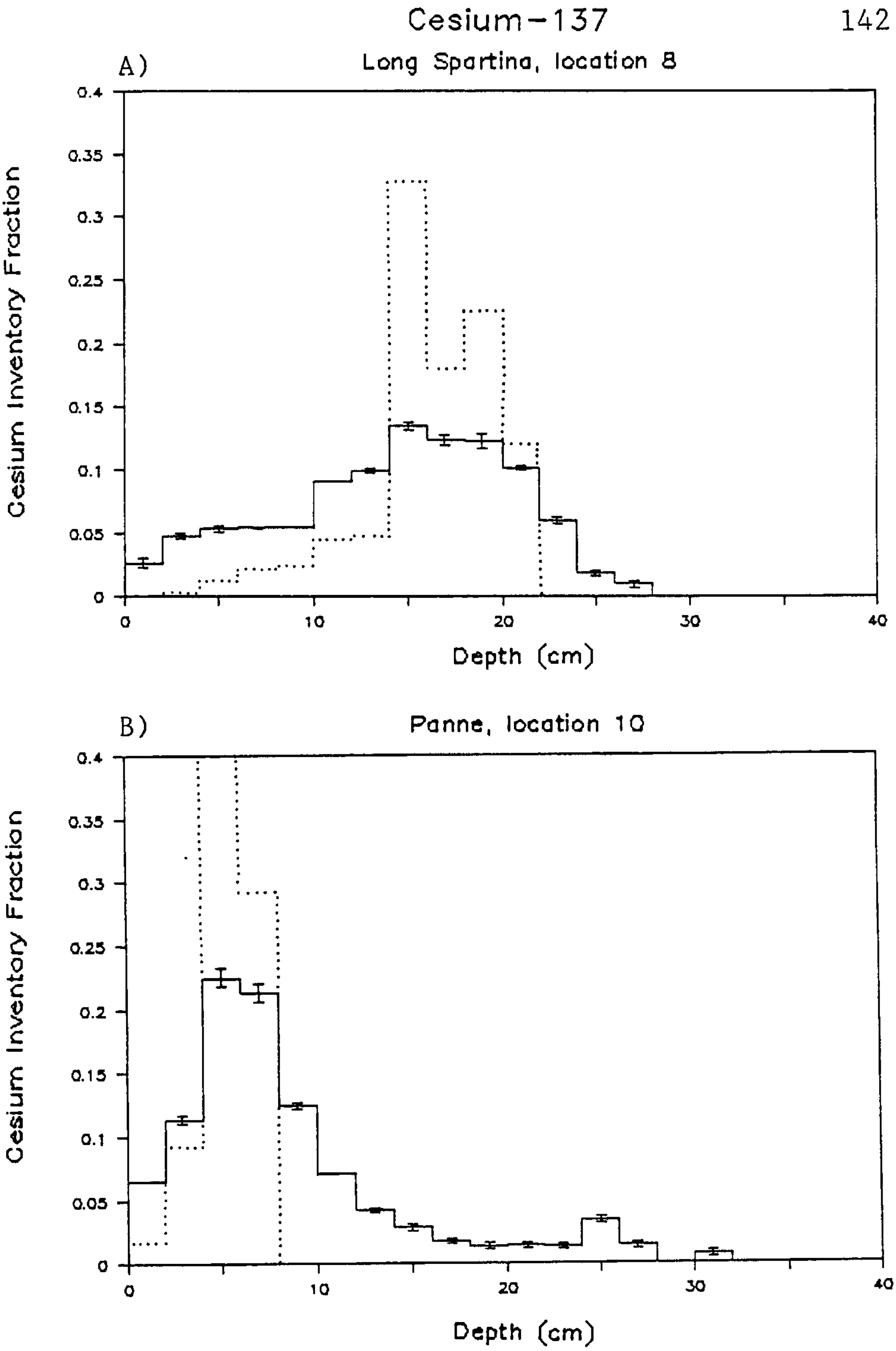


Figure 31 The cesium inventory fraction by volume with depth at A) location 8 and B) location 10. The solid line is the actual profile while the dotted line is a predicted curve.

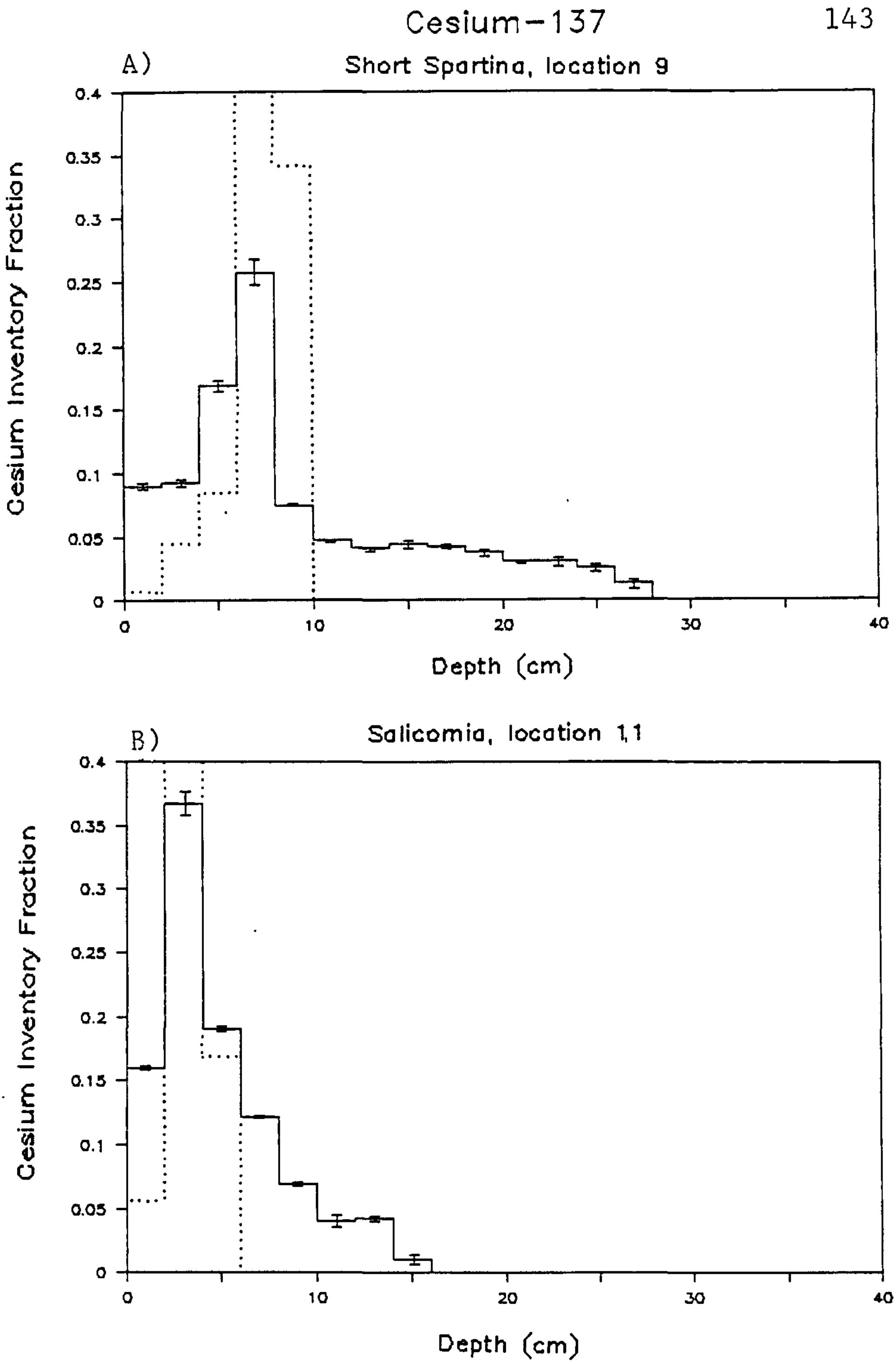


Figure 32 The cesium inventory fraction by volume with depth at A) location 9 and B) location 11. The solid line is the actual profile while the dotted line is a predicted curve.

location 1, yet the activity there is quite significant (Figure 33). Since the groundwater flow is downward, it seems that cesium in the upper sediments should have been depleted not enhanced. Significant redistribution of the cesium from depths exceeding 30 cm through bioturbation seems unlikely. This implies that the accretion rate is an over prediction as proposed previously or that cesium remains closer to the surface due to plant uptake and wash-in.

A deeper cesium peak is found at both locations 1 and 3 than at other locations in this marsh (Table VI) suggesting either a higher rate of burial or movement through the pore water. Based on peak depth a minimum accretion rate of 12.4-13.2 mm/yr would be predicted for location 1. This is an extremely high rate compared to other values using radioisotopic methods and rivals the rates predicted for the rapidly subsiding Deltaic Plain of Louisiana. Location 3, most likely a man-made drainage feature, may have a higher rate of sedimentation due to in-filling of the ditch. However, the cesium peak may also be deeper at location 3 due to the downward flow of groundwater. It is feasible that the cesium peaks at locations 2

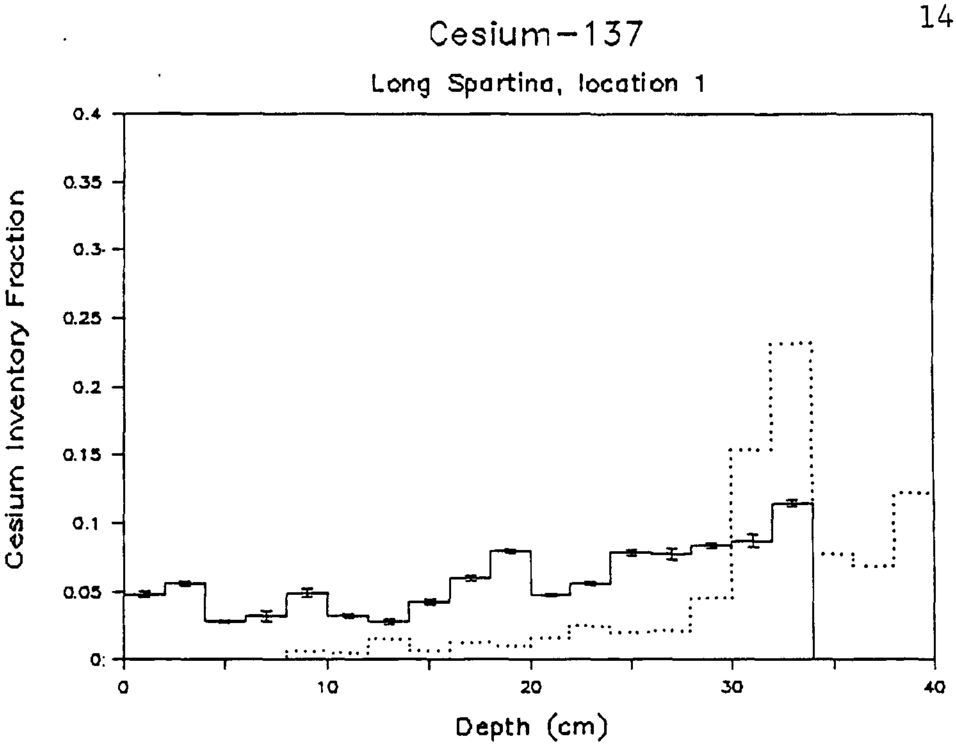


Figure 33 The cesium inventory fraction by volume with depth at location 1. The solid line represents the actual profile while the dotted line is a predicted curve.

and 4 are shallower due to this mobility as well.

Summary

The cesium-137 data from the salt marshes in this study support the literature on the chemical mobility of cesium in a saline environment. The mobility is suggested by the following:

1. All inventories are less than those predicted from atmospheric fallout.
2. The cesium distributions with depth are different from the predicted distributions. The upper sediments contain more cesium than predicted by atmospheric fallout and cesium is found at greater depths than predicted.
3. The cesium-137 peaks which are used to predict accretion rates are too variable, often being too deep or too shallow based on a decreasing sedimentation rate toward inland areas. However, there are sites in each marsh where the predictions support this trend.
    - A) The cesium peaks indicate that the accretion rate for location 2, 197 meters from the creek bank, is the same as for location 7, 635 meters from the creek bank. This is contrary to rates determined in other marshes which show that rates decrease inland. The variability of cesium peaks at location 4 and their lack of resemblance to the predicted profiles indicate that the ^{137}Cs peak is not indicative of the sedimentation rate.
    - B) The consistency of the values at locations 5-7 in Phillips Creek Marsh and their

approximate correlation with the rate of sea level rise suggests that the accretion rate derived from the cesium peak may be a good approximation of the rate of sedimentation. Furthermore, the consistent depth to the clay layer at these locations (16-24 cm) indicates similar accretion rates.

C) The rates of sedimentation predicted in No Egrets Marsh are consistent with a decreasing rate of sedimentation from the creek bank.

4. The retention of cesium toward the creekbank is increasingly suppressed due to the higher frequency of tidal inundations, the lower organic matter content and the greater export of organic detritus.

5. The increasing inventories toward the high marsh can be explained by the decreasing frequency of inundation, the finer particle size of the sediments, and retention of the organic matter. The cesium cycles through the above- and belowground production.

In conclusion, the cesium profiles are in general not historical records of atmospheric fallout but are the result of deposition modified by mixing, chemical remobilization and other diagenetic processes. Use of the cesium peak concentration as a marker in the sediments is therefore not the most appropriate method for determining accretion rates in salt marshes and other methods should be sought.

CONCLUSIONS

Specific conclusions have been presented at the end of most chapters. This section attempts a synthesis of these conclusions. This study shows that the major assumptions behind using the ^{137}Cs technique for dating are not valid for the salt marshes studied. The major assumption that is violated is that the cesium is not actively concentrated by organic-rich layers in the sediment. The organic content of the sediments is the primary factor favoring retention of the cesium. The organic matter is important because it increases the cation exchange capacity of the soil and therefore increases the distribution coefficients for sorption of the cesium. The low cesium inventories in comparison to that predicted by atmospheric fallout at all sites and variability in the cesium peaks provide evidence of the mobility of cesium in this environment. Although illite fixes cesium strongly in the non-marine environment, the evidence from this study suggest that the soil fixing capacity is greatly reduced. This implies that cesium deposited in a saline environment will move rapidly through the food chain.

A unique relationship between the soil, plants, and fallout ^{137}Cs was proposed. The cesium deposited from atmospheric fallout may be sorbed directly by the soil or plants, or deposited in the tidal waters. The high productivity of salt marsh vegetation suggests that a fair amount of cesium will be intercepted by the aboveground plant parts. Translocation of the ceisum through the vegetation is feasible. Some loss of cesium from the system is expected from consumers such as insects, water fowl, and fiddler crabs. However, the cesium exiting in a soluble form or with sediments can be redeposited on the marsh surface so that the loss by these mechanisms is negligible. The major mechanism for loss of cesium from the system is most likely its removal with organic detritus. The low soil fixing capacity for cesium suggests that uptake by the vegetation will be high. If a substantial portion of the net primary productivity is exported then this provides a mechanism for depletion of the cesium. The cesium inventory at a site may be an indication of the amount of net organic detritus removed from the location since fallout began in 1954.

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Appendix 1

A) Values of soil properties and cesium-137 content (IF or if = Inventory fraction).

B) Values for the relative changes in elevation with distance in each marsh.

Appendix 1A

The soil property values for locations 1, 2, 3, and 4.

Depth	Name & No.	Dry Wt g	% Org.	% Inorg	Organic Density g/cc	Inorganic Density g/cc	%HOH	B. Den. g/cc
0-2	pc237	291.91	8.7	91.3	0.045	0.47	54.24	0.51
2-4	pc236	357.91	8.8	91.2	0.055	0.57	54.62	0.63
4-6	pc235	203.67	8.3	91.7	0.030	0.33	58.32	0.36
6-8	pc234	254.43	9.5	90.5	0.042	0.40	58.94	0.45
8-10	pc233	251.05	10.0	90.0	0.044	0.40	58.37	0.44
10-12	pc232	167.45	10.1	89.9	0.030	0.26	58.45	0.29
12-14	pc231	166.30	10.8	89.2	0.032	0.26	59.50	0.29
14-16	pc230	278.86	11.0	89.0	0.054	0.44	59.65	0.49
16-18	pc229	291.52	9.8	90.2	0.050	0.46	57.93	0.51
18-20	pc228	388.63	9.5	90.5	0.065	0.62	56.52	0.68
20-22	pc227	213.51	9.3	90.7	0.035	0.34	56.89	0.37
22-24	pc226	230.97	9.8	90.2	0.040	0.37	57.24	0.41
24-26	pc225	247.62	9.5	90.5	0.041	0.39	56.47	0.43
26-28	pc224	242.51	9.0	91.0	0.038	0.39	55.70	0.43
28-30	pc223	254.87	9.6	90.4	0.043	0.40	57.69	0.45
30-32	pc222	262.43	10.5	89.5	0.040	0.41	56.29	0.46
32-34	pc221	410.23	9.0	91.0	0.065	0.66	55.47	0.72
0-2	SSP44	174.63	22.2	77.8	0.068	0.24	64.42	0.31
2-4	SSP45	242.21	25.5	74.5	0.108	0.32	63.44	0.42
4-6	SSP46	262.51	26.1	73.9	0.120	0.34	64.23	0.46
6-8	SSP47	225.36	24.8	75.2	0.098	0.30	65.91	0.40
8-10	SSP48	200.51	25.1	74.9	0.088	0.26	67.98	0.35
10-12	SSP49	261.18	23.7	76.3	0.109	0.35	67.46	0.46
12-13.5	SSP50	154.05	25.3	74.7	0.091	0.27	71.01	0.36
13.5-15.5	SSP51	166.70	26.8	73.2	0.078	0.21	75.01	0.29
15.5-17.5	SSP52	168.46	26.1	73.9	0.077	0.22	75.84	0.30
17.5-19.5	SSP53	164.43	22.2	77.8	0.064	0.22	73.77	0.29
19.5-21.5	SSP54	190.35	20.6	79.4	0.069	0.27	71.78	0.33
21.5-23.5	SSP55	182.44	19.3	80.7	0.062	0.26	70.16	0.32
23.5-25.5	SSP56	289.48	15.1	84.9	0.077	0.43	62.47	0.51
25.5-27.5	SSP57	265.53	13.9	86.1	0.065	0.40	59.01	0.47
27.5-29.5	SSP58	423.89	10.7	89.3	0.080	0.66	52.46	0.74
29.5-31.5	SSP59	443.48	7.7	92.3	0.060	0.72	42.02	0.78
31.5-33.5	SSP60	583.94	6.0	94.0	0.061	0.96	33.84	1.02
33.5-35.5	SSP61	680.45	5.0	95.0	0.060	1.13	29.58	1.19
35.5-36.5	SSP62	391.80	5.8	94.2	0.080	1.29	30.18	1.37
0-1	LSC31	103.82			0.000	0.00	60.17	0.67
1-3	LSC32	108.50	16.2	83.8	0.057	0.30	68.78	0.35
3-5	LSC33	92.53	17.8	82.2	0.054	0.25	68.72	0.30
5-7	LSC34	101.76	19.3	80.7	0.064	0.27	68.93	0.33
7-9	LSC35	93.11	17.6	82.4	0.053	0.25	67.07	0.30
9-11	LSC36	79.95	16.3	83.7	0.042	0.22	67.40	0.26
11-13	LSC37	162.28	17.3	82.7	0.091	0.44	68.67	0.53
13-15	LSC38	187.48	16.0	84.0	0.098	0.51	68.18	0.61
15-17	LSC39	142.22	15.5	84.5	0.072	0.39	67.75	0.46
17-19	LSC40	155.21	13.1	86.9	0.066	0.44	64.90	0.50
19-21	LSC41	175.70	11.8	88.2	0.067	0.50	61.54	0.57
21-23	LSC42	168.42	10.5	89.5	0.057	0.49	57.16	0.55
23-25	LSC43	157.64	13.4	86.6	0.069	0.44	63.65	0.51
0-2	JUN30	206.10	32.2	67.8	0.116	0.25	73.97	0.36
2-4	JUN29	158.62	29.5	70.5	0.082	0.20	71.52	0.28
4-6	JUN28	168.20	32.0	68.0	0.094	0.20	73.42	0.30
6-8	JUN27	173.77	35.4	64.6	0.108	0.20	75.87	0.30
8-10	JUN26	114.81	33.2	66.8	0.067	0.13	77.33	0.20
10-12	JUN25	142.10	29.9	70.1	0.075	0.17	77.89	0.25
12-14	JUN24	184.83	23.9	76.1	0.077	0.25	72.94	0.32
14-16	JUN23	266.33	14.9	85.1	0.070	0.40	61.79	0.47
16-18	JUN22	321.88	13.9	86.1	0.078	0.49	57.18	0.56
18-20	JUN21	341.19	12.1	87.9	0.073	0.53	53.51	0.60
20-22	JUN20	436.78	9.4	90.6	0.072	0.69	45.01	0.77
22-24	JUN19	695.35	5.9	94.1	0.072	1.15	30.30	1.22
24-26	JUN18	833.55	5.2	94.8	0.077	1.39	26.99	1.46
26-29	JUN17	913.04	4.3	95.7	0.046	1.53	26.95	1.07
29-32	JUN16	1108.54	4.5	95.5	0.058	1.86	25.28	1.30

The cesium and potassium values for locations 1, 2, 3, 4.

Depth	Name & No.	Cs IF pCi/cc	range if	range if	pCi/Kg	pCi/cc	k-40 dps/kg
0-2	pc237	0.048	0.050	0.046	147.20	0.0754	
2-4	pc236	0.056	0.057	0.054	139.87	0.0878	
4-6	pc235	0.028	0.029	0.027	123.69	0.0442	
6-8	pc234	0.032	0.036	0.028	114.50	0.0511	
8-10	pc233	0.049	0.051	0.046	175.21	0.0772	
10-12	pc232	0.032	0.034	0.031	173.67	0.0510	1.58
12-14	pc231	0.028	0.030	0.026	153.34	0.0447	1.75
14-16	pc230	0.043	0.044	0.041	138.15	0.0676	1.09
16-18	pc229	0.060	0.058	0.062	183.25	0.0937	1.11
18-20	pc228	0.079	0.078	0.081	182.70	0.1246	1.07
20-22	pc227	0.048	0.049	0.047	202.40	0.0758	1.31
22-24	pc226	0.056	0.057	0.055	217.41	0.0881	1.23
24-26	pc225	0.078	0.077	0.080	283.07	0.1230	1.18
26-28	pc224	0.077	0.074	0.081	284.92	0.1212	1.23
28-30	pc223	0.083	0.082	0.086	293.85	0.1314	1.13
30-32	pc222	0.087	0.083	0.092	297.27	0.1369	1.15
32-34	pc221	0.114	0.113	0.117	250.06	0.1800	1.11
0-2	SSP44	0.046	0.047	0.045	352.42	0.1080	1.56
2-4	SSP45	0.071	0.071	0.071	392.14	0.1666	1.32
4-6	SSP46	0.119	0.116	0.121	604.08	0.2782	1.27
6-8	SSP47	0.149	0.145	0.153	881.50	0.3485	1.42
8-10	SSP48	0.225	0.219	0.233	1502.95	0.5287	1.43
10-12	SSP49	0.209	0.203	0.216	1071.17	0.4908	1.14
12-13.5	SSP50	0.045	0.047	0.042	289.84	0.1044	1.61
13.5-15.5	SSP51	0.026	0.028	0.023	207.55	0.0607	1.52
15.5-17.5	SSP52	0.020	0.022	0.017	156.73	0.0463	1.53
17.5-19.5	SSP53	0.014	0.015	0.012	110.79	0.0320	1.63
19.5-21.5	SSP54	0.016	0.018	0.013	109.51	0.0366	1.41
21.5-23.5	SSP55	0.010	0.011	0.009	75.99	0.0243	1.44
23.5-25.5	SSP56	0.016	0.019	0.014	75.83	0.0385	1.03
25.5-27.5	SSP57	0.006	0.007	0.004	29.10	0.0136	1.23
27.5-29.5	SSP58	0.000	0.000	0.000	0.13	0.0001	0.88
29.5-31.5	SSP59	0.029	0.032	0.026	87.30	0.0679	0.90
31.5-33.5	SSP60	0.000	0.000	0.000	0.12	0.0001	0.87
33.5-35.5	SSP61	0.000	0.000	0.000	0.10	0.0001	0.86
35.5-36.5	SSP62	0.000	0.000	0.000	0.08	0.0001	0.81
0-1	LSC31						
1-3	LSC32	0.032	0.032	0.031	305.87	0.069	1.65
3-5	LSC33	0.057	0.060	0.052	350.23	0.123	2.59
5-7	LSC34	0.038	0.039	0.037	272.57	0.082	2.41
7-9	LSC35	0.061	0.063	0.059	401.76	0.133	2.60
9-11	LSC36	0.045	0.046	0.045	326.22	0.099	2.99
11-13	LSC37	0.059	0.056	0.060	496.60	0.129	1.66
13-15	LSC38	0.150	0.145	0.157	620.58	0.327	1.53
15-17	LSC39	0.181	0.174	0.189	646.19	0.393	1.83
17-19	LSC40	0.124	0.119	0.130	583.91	0.270	
19-21	LSC41	0.073	0.072	0.075	315.50	0.159	
21-23	LSC42	0.076	0.085	0.065	288.97	0.165	
23-25	LSC43	0.104	0.107	0.100	412.52	0.226	
0-2	JUN30	0.112	0.106	0.120	848.26	0.3067	0.74
2-4	JUN29	0.118	0.112	0.124	1159.17	0.3226	0.94
4-6	JUN28	0.109	0.104	0.116	1015.59	0.2997	0.88
6-8	JUN27	0.098	0.092	0.105	880.18	0.2683	0.83
8-10	JUN26	0.046	0.049	0.043	629.21	0.1267	1.19
10-12	JUN25	0.074	0.078	0.070	816.45	0.2035	1.00
12-14	JUN24	0.058	0.060	0.054	487.54	0.1581	0.79
14-16	JUN23	0.081	0.077	0.084	472.45	0.2207	0.55
~~16-18~~	~~JUN22~~	0.070	0.071	0.069	340.66	0.1924	0.51
18-20	JUN21	0.052	0.054	0.049	236.55	0.1416	0.50
20-22	JUN20	0.057	0.059	0.054	203.26	0.1558	0.52
22-24	JUN19	0.045	0.048	0.041	100.96	0.1232	0.46
24-26	JUN18	0.004	0.004	0.003	6.64	0.0097	0.50
26-29	JUN17	0.024	0.025	0.022	40.77	0.0653	0.55
29-32	JUN16	0.053	0.059	0.046	75.26	0.1464	0.78

The soil property values for locations 4A, 4B, 5, 6, and 7.

Deoth	Name & No.	Dry Wt g	% Org.	% Inorg	Organic Density g/cc	Inorganic Density g/cc	%HOH	B. Den. g/cc
0-2	juna250	279.23	29.4	70.6	0.144	0.35	74.78	0.49
2-4	juna249	170.84	38.6	61.4	0.116	0.18	76.86	0.30
4-6	juna248	137.13	35.6	64.4	0.086	0.15	78.38	0.24
6-8	juna247	122.10	29.1	70.9	0.062	0.15	78.44	0.21
8-10	juna246	132.60	29.1	70.9	0.068	0.16	78.15	0.23
10-12	juna245	163.65	26.4	73.6	0.076	0.21	75.63	0.29
~~12-14~~	~~juna244~~	~~264.87~~	~~18.1~~	~~81.9~~	0.084	0.38	67.00	0.46
14-16	juna243	397.26	11.9	88.1	0.083	0.61	54.80	0.70
16-18	juna242	575.66	6.9	93.1	0.069	0.94	35.57	1.01
18-20	juna241	618.45	5.8	94.2	0.063	1.02	29.91	1.09
20-22	juna240	976.46	5.1	94.9	0.088	1.62	26.98	1.71
0-2	junb300	135.79	20.4	79.6	0.049	0.19	68.68	0.24
2-4	junb301	145.41	28.3	71.7	0.072	0.18	70.21	0.26
4-6	junb302	168.30	24.7	75.3	0.073	0.22	72.75	0.30
6-8	junb303	165.84	34.9	65.1	0.101	0.19	76.92	0.29
8-10	junb304	114.75	33.9	66.1	0.068	0.13	80.11	0.20
10-12	junb305	112.80	33.2	66.8	0.066	0.13	81.29	0.20
12-14	junb306	121.67	29.3	70.7	0.063	0.15	79.84	0.21
14-16	junb307	155.78	19.9	80.1	0.054	0.22	72.73	0.27
16-18	junb308	534.99	11.2	88.8	0.105	0.83	55.51	0.94
18-20	junb309	683.76	6.9	93.1			40.07	1.20
0-2	pat63	107.75	54.5	45.5	0.103	0.09	80.12	0.19
2-4	pat64	85.93	54.4	45.6	0.082	0.07	80.61	0.15
4-6	pat65	112.20	44.8	55.2	0.088	0.11	78.88	0.20
6-8	pat66	190.64	31.6	68.4	0.106	0.23	70.83	0.33
8-10	pat67	141.82	38.2	61.8	0.095	0.15	75.70	0.25
10-12	pat68	146.38	35.5	64.5	0.091	0.17	74.82	0.26
12-14	pat69	203.94	25.8	74.2	0.092	0.27	66.75	0.36
14-16.5	pat70	247.27	18.3	81.7	0.064	0.28	58.96	0.35
0-2	rm220	53.74	68.9	31.1	0.065	0.03	87.21	0.09
2-4	rm219	89.02	62.0	38.0	0.097	0.06	86.92	0.16
4-6	rm218	82.72	56.3	43.7	0.082	0.06	85.07	0.15
6-8	rm217	114.01	50.3	49.7	0.101	0.10	81.36	0.20
8-10	rm216	124.36	46.4	53.6	0.101	0.12	77.75	0.22
10-12	rm215	181.30	43.4	56.6	0.138	0.18	75.01	0.32
12-14	rm214	275.04	26.5	73.5	0.128	0.35	62.47	0.48
14-16	rm213	513.66	14.3	85.7	0.129	0.77	42.51	0.90
16-18	rm212	573.18	14.1	85.9	0.141	0.86	40.99	1.01
18-20	rm211	501.35	18.0	82.0	0.158	0.72	46.33	0.88
0-1	dis14						64.68	0.06
1-2.5	dis13						71.00	0.07
1-4	dis12	72.25	61.6	38.4	0.052	0.03	73.69	0.10
4-6	dis11	61.01	63.2	36.8	0.068	0.04	73.00	0.11
6-8	dis10	76.58	60.0	40.0	0.081	0.05	73.23	0.13
8-10	dis9	123.86	52.4	47.6	0.114	0.10	73.49	0.22
10-12.5	dis8	134.52	44.6	55.4	0.084	0.10	71.48	0.19
12.5-14.5	dis7	147.92	41.1	58.9	0.107	0.15	68.06	0.26
14.5-16.5	dis6	239.55	32.1	67.9	0.135	0.29	62.46	0.42
16.5-18.5	dis5	347.99	21.6	78.4	0.132	0.48	48.57	0.61
19.5-20.5	dis4	546.27	15.6	84.4	0.149	0.81	38.21	0.96
20.5-21.5	dis3	350.14	12.3	87.7	0.151	1.08	31.85	0.82
21.5-23	dis2	609.57	9.6	90.4	0.137	1.29	26.59	1.43
23-23.5	dis1	252.23	7.4	92.6	0.130	1.64	23.26	1.77

Appendix 1A

The cesium and potassium values for locations 4A, 4B, 5, 6, and 7.

Depth	Name & No.	Cs IF pCi/cc	range if	range if	pCi/Kg	pCi/cc	k-40 dps/kg
0-2	juna250	0.218	0.196	0.248	937.54	0.4593	0.65
2-4	juna249	0.108	0.102	0.115	754.53	0.2261	0.87
4-6	juna248	0.084	0.082	0.087	733.76	0.1765	1.05
6-8	juna247	0.080	0.077	0.083	781.63	0.1674	1.19
8-10	juna246	0.063	0.065	0.060	566.70	0.1318	0.98
10-12	juna245	0.081	0.079	0.083	591.10	0.1697	0.83
12-14	juna244	0.095	0.094	0.097	430.35	0.2000	0.58
14-16	juna243	0.083	0.082	0.083	248.93	0.1735	0.43
16-18	juna242	0.091	0.094	0.087	189.91	0.1918	0.52
18-20	juna241	0.042	0.055	0.024	80.86	0.0877	0.48
20-22	juna240	0.056	0.073	0.034	69.07	0.1183	0.47
0-2	junb300	0.0517	0.0577	0.0446	363.39	0.0866	0.08
2-4	junb301	0.1267	0.1230	0.1311	831.03	0.2120	0.14
4-6	junb302	0.1654	0.1597	0.1722	937.18	0.2767	0.13
6-8	junb303	0.1004	0.0986	0.1025	577.16	0.1679	0.13
8-10	junb304	0.0488	0.0538	0.0429	405.94	0.0817	0.13
10-12	junb305	0.0788	0.0783	0.0793	665.81	0.1318	0.13
12-14	junb306	0.0783	0.0791	0.0773	613.45	0.1309	0.14
14-16	junb307	0.0633	0.0649	0.0613	387.44	0.1059	0.14
16-18	junb308	0.1577	0.1533	0.1630	281.12	0.2638	0.16
18-20	junb309	0.1289	0.1316	0.1258	179.81	0.2157	
0-2	pat63	0.053	0.054	0.051	584.86	0.0992	2.04
2-4	pat64	0.076	0.076	0.076	946.96	0.1428	2.49
4-6	pat65	0.177	0.175	0.178	1691.27	0.3329	2.13
6-8	pat66	0.257	0.253	0.261	1444.20	0.4830	1.23
8-10	pat67	0.105	0.106	0.104	792.37	0.1971	1.60
10-12	pat68	0.103	0.104	0.102	752.46	0.1932	1.61
12-14	pat69	0.146	0.145	0.147	768.52	0.2750	1.21
14-16.5	pat70	0.085	0.087	0.083	460.21	0.1597	1.03
0-2	rm220	0.022	0.027	0.017	722.15	0.0681	2.43
2-4	rm219	0.095	0.096	0.094	1849.98	0.2889	1.53
4-6	rm218	0.141	0.137	0.145	2964.88	0.4303	1.58
6-8	rm217	0.219	0.214	0.224	3337.99	0.6677	1.12
8-10	rm216	0.140	0.137	0.144	1964.78	0.4287	1.10
10-12	rm215	0.109	0.108	0.109	1044.77	0.3323	0.63
12-14	rm214	0.102	0.101	0.103	646.76	0.3121	0.57
14-16	rm213	0.114	0.115	0.113	387.32	0.3490	0.52
16-18	rm212	0.058	0.064	0.051	175.50	0.1765	0.59
18-20	rm211	0.000	0.000	0.000	0.00	0.0000	0.58
0-1	dis14						
1-2.5	dis13						
1-4	dis12	0.026	0.027	0.026	1365.04	0.1153	2.74
4-6	dis11	0.035	0.036	0.035	1456.71	0.1559	3.12
6-8	dis10	0.090	0.089	0.091	2950.45	0.3964	2.64
8-10	dis9	0.215	0.211	0.220	4363.09	0.9481	1.72
10-12.5	dis8	0.118	0.117	0.119	2762.96	0.5216	1.91
12.5-14.5	dis7	0.113	0.113	0.114	1922.82	0.4990	1.49
14.5-16.5	dis6	0.135	0.133	0.136	1413.29	0.5940	1.02
16.5-18.5	dis5	0.164	0.163	0.165	1184.05	0.7229	0.92
18.5-20.5	dis4	0.062	0.063	0.061	285.22	0.2733	0.85
20.5-21.5	dis3	0.029	0.031	0.027	103.38	0.1270	0.78
21.5-23	dis2	0.008	0.012	0.005	25.65	0.0366	0.76
23-23.5	dis1	0.003	0.005	0.002	8.41	0.0149	0.98

Appendix 1A

The soil property values for locations 8, 9, 10, and 11.

Depth	Name & No.	Dry Wt g	% Org.	% Inorg	Organic Density g/cc	Inorganic Density g/cc	%HOH	B. Den. g/cc
0-2	LSP91	208.54	8.0	92.0	0.029	0.34	53.03	0.37
2-4	LSP92	397.88	8.2	91.8	0.057	0.64	49.79	0.70
4-6	LSP93	411.68	9.2	90.8	0.066	0.66	47.59	0.72
6-8	LSP94	434.21	9.3	90.7	0.071	0.69	46.46	0.76
8-10	LSP95	390.97	9.9	90.1	0.068	0.62	46.16	0.69
10-12	LSP96	505.74	10.3	89.7	0.092	0.80	46.20	0.89
12-14	LSP97	459.05	10.3	89.7	0.083	0.72	47.30	0.81
14-16	LSP98	492.42	8.5	91.5	0.073	0.79	45.70	0.86
16-18	LSP99	377.46	8.0	92.0	0.053	0.61	44.35	0.66
18-20	LSP100	397.77	7.9	92.1	0.055	0.64	43.32	0.70
20-22	LSP101	549.50	6.8	93.2	0.066	0.90	40.93	0.96
22-24	LSP102	610.69	6.4	93.6	0.069	1.00	40.26	1.07
24-26	LSP103	434.05	7.0	93.0	0.054	0.71	42.81	0.76
26-28	LSP104	404.10	9.3	90.7	0.066	0.64	46.94	0.71
28-30	LSP105	404.21	11.4	88.6	0.081	0.63	50.50	0.71
30-32	LSP106	331.69	10.2	89.8	0.059	0.52	45.82	0.58
32-34	LSP107	517.00	10.0	90.0	0.091	0.82	44.53	0.91
0-2	Bee1	211.31	27.0	73.0	0.100	0.27	70.16	0.37
2-4	Bee2	187.02	28.0	72.0	0.092	0.24	67.18	0.33
4-6	Bee3	234.52	29.4	70.6	0.121	0.29	67.36	0.41
6-8	Bee4	335.63	21.4	78.6	0.126	0.46	57.59	0.59
8-10	Bee5	275.99	23.1	76.9	0.112	0.37	57.46	0.48
10-12	Bee6	230.13	25.0	75.0	0.101	0.30	62.63	0.40
12-14	Bee7	180.66	25.8	74.2	0.082	0.24	69.34	0.32
14-16	Bee8	230.04	20.8	79.2	0.084	0.32	66.05	0.40
16-18	Bee9	256.38	16.5	83.5	0.074	0.38	61.64	0.45
18-20	Bee10	363.59	13.9	86.1	0.089	0.55	56.75	0.64
20-22	Bee11	335.79	12.9	87.1	0.076	0.51	53.91	0.59
22-24	Bee12	346.04	13.5	86.5	0.082	0.53	53.18	0.61
24-26	Bee13	380.43	12.1	87.9	0.080	0.59	51.39	0.67
26-28	Bee14	419.71	11.6	88.4	0.085	0.65	49.68	0.74
28-30	Bee15	351.49	11.9	88.1	0.074	0.54	49.01	0.62
30-32	Bee16	340.19	11.2	88.8	0.067	0.53	45.62	0.60
32-34	Bee17	461.56	10.6	89.4	0.086	0.72	44.90	0.81
0-2	Pan108	208.34	27.4	72.6	0.100	0.27	73.55	0.37
2-4	Pan109	167.08	29.3	70.7	0.086	0.21	76.27	0.29
4-6	Pan110	157.94	29.8	70.2	0.083	0.19	73.76	0.28
6-8	Pan111	218.88	23.8	76.2	0.092	0.29	68.75	0.38
8-10	Pan112	238.02	20.4	79.6	0.085	0.33	67.21	0.42
10-12	Pan113	214.77	18.4	81.6	0.069	0.31	67.64	0.38
12-14	Pan114	190.79	17.6	82.4	0.059	0.28	70.22	0.33
14-16	Pan115	248.22	17.4	82.6	0.076	0.36	64.70	0.44
16-18	Pan116	207.69	21.0	79.0	0.076	0.29	69.92	0.36
18-20	Pan117	183.79	17.1	82.9	0.055	0.27	67.22	0.32
20-22	Pan118	271.71	12.1	87.9	0.058	0.42	59.71	0.48
22-24	Pan119	297.37	11.5	88.5	0.060	0.46	55.36	0.52
24-26	Pan120	415.75	10.1	89.9	0.073	0.66	50.00	0.73
26-28	Pan88	531.40	9.0	91.0	0.083	0.85	42.20	0.93
28-30	Pan89	344.68	9.1	90.9	0.055	0.55	44.46	0.60
30-32	Pan90	470.86	7.8	92.2	0.064	0.76	45.72	0.83
0-2	SAL72	361.99	20.4	79.6	0.130	0.51	60.20	0.64
2-4	SAL73	512.20	12.3	87.7	0.110	0.79	47.18	0.90
4-6	SAL74	502.07	11.4	88.6	0.100	0.78	40.89	0.88
6-8	SAL75	514.69	13.1	86.9	0.118	0.78	44.55	0.90
8-10	SAL76	461.85	9.9	90.1	0.080	0.73	43.01	0.81
10-12	SAL77	606.29	7.6	92.4	0.081	0.98	33.87	1.06
12-14	SAL78	760.05	5.6	94.4	0.075	1.26	26.55	1.33
14-16	SAL79	647.50	4.9	35.1	0.055	1.08	22.79	1.14

Appendix 1A

The cesium and potassium values for locations 8, 9, 10, and 11.

Deotn	Name & No.	Cs IF pCi/cc	ranoe if	range if	pCi/Kg	pCi/cc	k-40 dps/kg
0-2	LSP91	0.026	0.030	0.023	147.75	0.0541	1.46
2-4	LSP92	0.047	0.049	0.045	139.10	0.0971	1.08
4-6	LSP93	0.054	0.056	0.051	152.23	0.1099	1.03
6-8	LSP94	0.055	0.056	0.054	147.93	0.1127	1.03
8-10	LSP95	0.055	0.055	0.055	163.83	0.1124	1.01
10-12	LSP96	0.091	0.091	0.091	209.61	0.1860	1.02
12-14	LSP97	0.099	0.098	0.101	252.00	0.2029	1.05
14-16	LSP98	0.135	0.132	0.138	319.72	0.2762	1.03
16-18	LSP99	0.124	0.120	0.128	381.95	0.2529	1.02
18-20	LSP100	0.123	0.118	0.129	360.39	0.2515	1.00
20-22	LSP101	0.101	0.100	0.103	214.86	0.2071	0.99
22-24	LSP102	0.060	0.063	0.057	115.52	0.1238	0.97
24-26	LSP103	0.019	0.021	0.016	51.09	0.0389	0.92
26-28	LSP104	0.010	0.012	0.008	29.68	0.0210	0.94
28-30	LSP105	0.000	0.000	0.000	0.13	0.0001	0.93
30-32	LSP106	0.000	0.000	0.000	0.11	0.0001	0.89
32-34	LSP107	0.000	0.000	0.000	0.11	0.0001	0.90
0-2	Bee1	0.090	0.087	0.092	383.84	0.1423	
2-4	Bee2	0.093	0.090	0.096	449.82	0.1476	
4-6	Bee3	0.169	0.164	0.173	650.30	0.2676	1.41
6-8	Bee4	0.257	0.248	0.268	693.41	0.4083	1.07
8-10	Bee5	0.075	0.074	0.076	245.52	0.1189	1.18
10-12	Bee6	0.047	0.048	0.046	186.01	0.0751	1.36
12-14	Bee7	0.041	0.043	0.039	205.79	0.0652	1.50
14-16	Bee8	0.045	0.047	0.041	175.18	0.0707	1.37
16-18	Bee9	0.043	0.045	0.041	151.56	0.0682	1.25
18-20	Bee10	0.038	0.041	0.036	95.67	0.0610	1.18
20-22	Bee11	0.032	0.033	0.030	85.01	0.0501	1.13
22-24	Bee12	0.031	0.035	0.027	81.41	0.0494	1.14
24-26	Bee13	0.026	0.029	0.023	61.85	0.0413	
26-28	Bee14	0.013	0.016	0.010	28.62	0.0211	1.10
28-30	Bee15	0.000	0.000	0.000	0.16	0.0001	1.07
30-32	Bee16	0.000	0.000	0.000	0.13	0.0001	1.09
32-34	Bee17	0.000	0.000	0.000	0.12	0.0001	1.00
0-2	Pan108	0.065	0.065	0.065	409.51	0.1497	1.22
2-4	Pan109	0.113	0.111	0.116	894.06	0.2621	1.43
4-6	Pan110	0.225	0.218	0.233	1878.04	0.5204	1.54
6-8	Pan111	0.213	0.206	0.221	1283.47	0.4929	1.21
8-10	Pan112	0.124	0.122	0.127	690.03	0.2881	1.16
10-12	Pan113	0.071	0.071	0.071	436.86	0.1646	1.27
12-14	Pan114	0.042	0.044	0.041	293.36	0.0982	1.35
14-16	Pan115	0.029	0.032	0.026	155.08	0.0675	1.12
16-18	Pan116	0.018	0.020	0.016	112.94	0.0412	1.24
18-20	Pan117	0.014	0.017	0.012	102.43	0.0330	1.37
20-22	Pan118	0.015	0.017	0.012	70.88	0.0338	1.05
22-24	Pan119	0.014	0.015	0.012	60.30	0.0315	0.97
24-26	Pan120	0.035	0.037	0.032	110.11	0.0803	0.92
26-28	Pan88	0.015	0.017	0.012	36.49	0.0340	0.91
28-30	Pan89	0.000	0.000	0.000	0.11	0.0001	0.94
30-32	Pan90	0.008	0.010	0.005	21.24	0.0175	0.93
0-2	SAL72	0.160	0.159	0.161	513.22	0.3259	0.95
2-4	SAL73	0.367	0.358	0.377	834.05	0.7495	0.89
4-6	SAL74	0.191	0.189	0.192	442.08	0.3894	0.93
6-8	SAL75	0.121	0.121	0.121	274.03	0.2474	0.91
8-10	SAL76	0.069	0.070	0.068	173.54	0.1406	0.82
10-12	SAL77	0.040	0.044	0.035	76.62	0.0815	0.78
12-14	SAL78	0.043	0.045	0.040	65.43	0.0872	0.74
14-16	SAL79	0.010	0.014	0.007	18.61	0.0211	0.73

Appendix 1B

The relative changes in elevation with distance in each marsh.

Phillips Creek Marsh

distance (m)	elevation (ft)
3	45.42
185	45.11
197	45.51
335	45.79
353	44.52
405	46.14
425	46.36
435	46.16
443	46.4
460	46.74
485	46.87
535	46.96
547	46.92
585	46.85
635	46.95
685	46.45
910	47.1
935	47.21

No Egrets Marsh

distance (m)	elevation (ft)
0	44.07
10	44.77
20	44.66
30	44.6
40	44.69
60	44.04
70	44.16
80	44.17
90	44.9
100	44.94
120	44.98
160	44.93
195	45.01
200	45.01
240	45.25
260	45.22
293	45.12
310	45.5

Appendix 2

A) Calculations for the predicted inventories of fallout cesium in Virginia, South Carolina and Louisiana.

B) The values for the predicted cesium profiles in this study presented in Figures 27-33.

C) Calculations of the actual cesium inventories at several sites in South Carolina based on the results of Sharma et al., 1987.

Year	NY Sr90 mCi/Km²	Cs137/ Sr90	Orig. Cs b*c (mCi/Km²)	Correct. 86 d*.576 (mCi/Km²)	Correct pCi/cm² e/10	SC Sr90 mCi/Km²	Orig. Cs g*c mCi/Km²	Correct. 86 h*.576mCi/Km²	La Sr90	Orig. Cs j*c	Correct. 86 k*.576
54	2.76	1.9	5.244	3.021	0.302						
55	3.57	1.9	6.783	3.907	0.391						
56	4.43	1.9	8.417	4.848	0.485						
57	4.44	1.9	8.436	4.859	0.486						
58	6.16	2	12.32	7.096	0.710						
59	8.68	2	17.36	9.999	1.000	2.17	4.34	2.49984			
60	1.58	1.9	3.002	1.729	0.173	1.77	3.363	1.937088	0.25	0.475	0.2736
61	2.43	1.9	4.617	2.659	0.266	1.62	3.078	1.772928	1.75	3.325	1.9152
62	12.33	1.4	17.262	9.943	0.994	7.53	10.542	6.072192	5.46	7.644	4.402944
63	23.79	1.3	30.927	17.814	1.781	12.71	16.523	9.517248	9.19	11.947	6.881472
64	15.85	1.4	22.19	12.781	1.278	9.76	13.664	7.870464	9.62	13.468	7.757568
65	5.53	1.5	8.295	4.778	0.478	3.7	5.55	3.1968	2.83	4.245	2.44512
66	2.43	1.5	3.645	2.100	0.210	1.57	2.355	1.35648	1.82	2.73	1.57248
67	1.64	1.5	2.46	1.417	0.142	0.96	1.44	0.82944	0.53	0.795	0.45792
68	1.32	1.6	2.112	1.217	0.122	0.9	1.44	0.82944	0.41	0.656	0.377856
69	1.43	1.6	2.288	1.318	0.132	0.67	1.072	0.617472	0.73	1.168	0.672768
70	1.48	1.9	2.812	1.620	0.162	1.05	1.995	1.14912	0.84	1.596	0.919296
71	1.41	1.9	2.679	1.543	0.154	0.93	1.767	1.017792	0.89	1.691	0.974016
72	0.75	1.8	1.35	0.778	0.078	0.5	0.9	0.5184	0.52	0.936	0.539136
73	0.42	1.8	0.756	0.435	0.044	0.3	0.54	0.31104	0.3	0.54	0.31104
74	0.93	1.8	1.674	0.964	0.096	0.6	1.08	0.62208	0.7	1.26	0.72576
75	0.69	1.9	1.311	0.755	0.076	0.49	0.931	0.536256	0.35	0.665	0.38304
76	0.26	1.9	0.494	0.285	0.028	0.08	0.152	0.087552	0.06	0.114	0.065664
77	0.81	1.7	1.377	0.793	0.079						
78	0.86	2.1	1.806	1.040	0.104			40.74163			30.67408
79	0.29	2.1	0.609	0.351	0.035						
80	0.14	2.1	0.294	0.169	0.017						
81	0.36	2.1	0.756	0.435	0.044						
82	0.04	2.1	0.084	0.048	0.005						
83	0.04	2.1	0.084	0.048	0.005						
84	0.04	2.1	0.084	0.048	0.005						
85	0.01	2.1	0.021	0.012	0.001						
86	0.02	2.1	0.042	0.024	0.002						
87				0	0						
				98.84	9.88 - sum 1954-86						
				72.13478	7.21 - sum 1959-76						
					6.21 - sum 1960-76						

Appendix 2A The cesium inventories are based on the cesium-137/strontium-90 fallout ratios and a correction factor based on the radioactive decay of cesium.

Appendix 2A

Explanation

The cesium inventory for Virginia over the time period 1954-1987 is 9.88 pCi/sq. cm and from 1959-1976 is 7.21 pCi/sq cm. To obtain the present value of 9.88 pCi/sq cm, a correction factor of 1.37 is needed. When this correction factor is applied to the inventory for South Carolina, the value changes from 4.07 to 5.58 pCi/sq cm. The Louisiana record covers 1960-1976, The correction factor based on the Virginia record is 1.59 and the inventory for Louisiana changes from 3.07 to 4.88 pCi/sq cm.

The predicted cesium profiles in Figures 27-33 of the text are based on the accretion rate derived from the peak of the actual profile for that site. The number of years represented by each 2 cm section determines the cesium content using the actual fallout data.

3yr sum	Norm	pt	5yr sum	Norm	7yr sum	Norm
AR 0.63	a/9.88		AR 0.4	d/9.885	AR 0.3	f/9.885
0.003	0.000	1	0.013	0.001	0.062	0.006
0.015	0.001	3	0.205	0.021	0.435	0.044
0.096	0.010	5	0.323	0.033	0.834	0.084
0.211	0.021	7	0.648	0.066	5.180	0.524
0.216	0.022	9	3.889	0.395	3.374	0.341
0.394	0.040	11	3.143	0.319	9.885	1.000
0.396	0.040	13	1.664	0.169		
1.966	0.199	15	9.885	1.003		
3.041	0.308	17				
1.883	0.190	19				
1.362	0.138	21				
0.302	0.031	23				
9.884	1.000					

9.5yr sum	Norm	15yr sum	Norm	1.5yr sum	Norm
AR 0.2	h/9.885	AR .13	j/9.885	AR 1.2	l/9.855
0.166	0.017	0.541	0.055	0.001	0.000
0.901	0.091	7.680	0.777	0.002	0.000
5.944	0.603	1.664	0.168	0.0075	0.001
2.874	0.292	9.885	1.000	0.0075	0.001
9.885	1.003			0.0525	0.005
				0.0435	0.004
				0.1435	0.015
				0.0675	0.007
				0.124	0.013
				0.092	0.009
				0.155	0.016
				0.239	0.024
				0.193	0.020
				0.203	0.021
				0.449	0.046
				1.517	0.154
				2.278	0.231
				0.763	0.077
				0.673	0.068
				1.21	0.123
				0.7285	0.074
				0.6365	0.064
				0.302	0.031
				9.885	1.003

AR = accretion rate

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q
	Vclay + Vorg/ Vtotal	1-LOI	2.5(1-LOI)	LOI	C +D	A*E Bulk D. g/cc	dpm/kg	dpm/kg low	dpm/kg high	G/2220 pCi/g	H/2220	I/2220	vol.	F*M wt sample (g)	J*N pCi	K*N	L*N
TC4																	
	0.47	0.888	2.22	0.112	2.332	1.09604	369	335	403	0.166216	0.150900	0.181531	117.8	129.1135	21.46075	19.48334	23.43817
	0.35	0.856	2.14	0.144	2.284	0.7994	393	360	426	0.177027	0.162162	0.191891	117.8	94.16932	16.67051	15.27070	18.07032
	0.454	0.873	2.1825	0.127	2.3095	1.048513	406	365	447	0.182882	0.164414	0.201351	117.8	123.5148	22.58874	20.30761	24.86987
	0.446	0.89	2.225	0.11	2.335	1.04141	442	398	486	0.199099	0.179279	0.218918	117.8	122.6780	24.42509	21.99364	26.85655
	0.439	0.82	2.05	0.18	2.23	0.97897	258	214	302	0.116216	0.096396	0.136036	117.8	115.3226	13.40236	11.11668	15.68803
	0.471	0.794	1.985	0.206	2.191	1.031961	180	158	202	0.081081	0.071171	0.090990	117.8	121.5650	9.856622	8.651923	11.06132
	0.468	0.854	2.135	0.146	2.281	1.067508	125	85	165	0.056306	0.038288	0.074324	117.8	125.7524	7.080655	4.814845	9.346465
	0.474	0.86	2.15	0.14	2.29	1.08546	41	29	53	0.018468	0.013063	0.023873	117.8	127.8671	2.361511	1.670337	3.052685
	0.426	0.853	2.1325	0.147	2.2795	0.971067	50	35	65	0.022522	0.015765	0.029279	117.8	114.3916	2.576389	1.803472	3.349306
	0.484	0.863	2.1575	0.137	2.2945	1.110538	58	44	72	0.026126	0.019819	0.032432	117.8	130.8213	3.417855	2.592856	4.242855
															123.8405	107.7054	139.9756
														pCi/cm^2 =	1.577586	1.372043	1.783128
BB2																	
	0.402	0.84	2.1	0.16	2.26	0.90852	562	520	604	0.253153	0.234234	0.272072	117.8	107.0236	27.09337	25.06860	29.11814
	0.404	0.837	2.0925	0.163	2.2555	0.911222	751	689	813	0.338288	0.310360	0.366216	117.8	107.3419	36.31252	33.31468	39.31036
	0.384	0.822	2.055	0.178	2.233	0.857472	769	715	823	0.346396	0.322072	0.370720	117.8	101.0102	34.98956	32.53256	37.44657
	0.378	0.816	2.04	0.184	2.224	0.840672	640	584	696	0.288288	0.263063	0.313513	117.8	99.03116	28.54952	26.05144	31.04760
	0.368	0.82	2.05	0.18	2.23	0.82064	679	650	708	0.305855	0.292792	0.318918	117.8	96.67139	29.56751	28.30468	30.83033
	0.37	0.817	2.0425	0.183	2.2255	0.823435	630	577	683	0.283783	0.259909	0.307657	117.8	97.00064	27.52720	25.21142	29.84299
	0.382	0.805	2.0125	0.195	2.2075	0.843265	661	615	707	0.297747	0.277027	0.318468	117.8	99.33661	29.57725	27.51892	31.63558
	0.352	0.815	2.0375	0.185	2.2225	0.78232	687	642	732	0.309459	0.289189	0.329729	117.8	92.15729	28.51894	26.65089	30.38700
	0.344	0.808	2.02	0.192	2.212	0.760928	684	652	716	0.308108	0.293693	0.322522	117.8	89.63731	27.61798	26.32591	28.91005
	0.357	0.811	2.0275	0.189	2.2165	0.791290	653	620	686	0.294144	0.279279	0.309009	117.8	93.21402	27.41835	26.03274	28.80397
	0.346	0.809	2.0225	0.191	2.2135	0.765871	642	618	666	0.289189	0.278378	0.3	117.8	90.21960	26.09053	25.11518	27.06588
	0.348	0.804	2.01	0.196	2.206	0.767688	520	505	535	0.234234	0.227477	0.240990	117.8	90.43364	21.18265	20.57161	21.79369
	0.365	0.792	1.98	0.208	2.188	0.79862	637	587	687	0.286936	0.264414	0.309459	157	125.3833	35.97711	33.15316	38.80106
															380.4225	355.8518	404.9932
														pCi/cm^2 =	4.846147	4.533144	5.159149
BB4																	
	0.33	0.8	2	0.2	2.2	0.726	554	507	601	0.249549	0.228378	0.270720	78.5	56.991	14.22207	13.01551	15.42864
	0.32	0.8	2	0.2	2.2	0.704	645	595	695	0.290540	0.268018	0.313063	78.5	55.264	16.05643	14.81174	17.30111
	0.33	0.79	1.975	0.21	2.185	0.72105	652	606	698	0.293693	0.272972	0.314414	78.5	56.60242	16.62377	15.45093	17.79661
	0.313	0.79	1.975	0.21	2.185	0.683905	602	556	648	0.271171	0.250450	0.291891	78.5	53.68654	14.55824	13.44581	15.67066
	0.339	0.793	1.9825	0.207	2.1895	0.742240	332	311	353	0.149549	0.140090	0.159009	78.5	58.26587	8.713635	8.162472	9.264799
	0.327	0.793	1.9825	0.207	2.1895	0.715966	424	395	453	0.190990	0.177927	0.204054	78.5	56.20337	10.73433	10.00014	11.46852
	0.325	0.8	2	0.2	2.2	0.715	306	285	327	0.137837	0.128378	0.147297	78.5	56.1275	7.736493	7.205557	8.267429
	0.324	0.8	2	0.2	2.2	0.7128	322	294	350	0.145045	0.132432	0.157657	157	111.9096	16.23193	14.82046	17.64340
	0.344	0.808	2.02	0.192	2.212	0.760928	238	214	262	0.107207	0.096396	0.118018	78.5	59.73284	6.403791	5.758031	7.049552
	0.345	0.808	2.02	0.192	2.212	0.76314	249	213	285	0.112162	0.095945	0.128378	78.5	59.90649	6.719241	5.747784	7.690698
	0.34	0.781	1.9525	0.219	2.1715	0.73831	167	143	191	0.075225	0.064414	0.086036	78.5	57.95733	4.359853	3.733287	4.986419
	0.381	0.822	2.055	0.178	2.233	0.850773	206	179	233	0.092792	0.080630	0.104954	78.5	66.78568	6.197229	5.384971	7.009488
	0.367	0.847	2.1175	0.153	2.2705	0.833273	71	54	88	0.031981	0.024324	0.039639	78.5	65.41196	2.092004	1.591101	2.592906
	0.4	0.841	2.1025	0.159	2.2615	0.9046	69	45	93	0.031081	0.020270	0.041891	314	284.0444	8.828407	5.757656	11.89915
	0.41	0.843	2.1075	0.157	2.2645	0.928445	20	16	24	0.009009	0.007207	0.010810	314	291.5317	2.626411	2.101129	3.151694
															142.1038	126.9866	157.2211
														pCi/cm^2 =	1.810240	1.617663	2.002816

Appendix 2C The cesium inventories for sites in South Carolina based on the data of Sharma et al., 1987.

Appendix 2C

Explanation

The data of Sharma et al., 1987 (Table 3) was used to predict the cesium inventories in South Carolina near North Inlet at three sites: Town Creek 4, Bread and Butter 2, and Bread and Butter 4. The calculations were necessary in order to estimate the weight of each sample so that the activity (dpm/kg) could be changed to the activity per section (dpm). The activity on an area basis could then be calculated as the radius of each core was 5 cm. The formula's presented below were used in the calculations. The density of the clay was assumed to be 2.5 g/cc while the density of the organic matter was assigned a value of 1 g/cc.

$$\rho\text{solid} = \rho\text{clay}\ (1\text{-LOI}) + \rho\text{org.}(\text{LOI})$$

$$\text{Bulk density} = \rho\text{solid} * \left[\frac{V_{clay} + V_{org}}{V_{total}}\right],\ V = \text{volume}$$

The inventory ranges for the three sites were 1.37-1.78, 4.53-5.16, and 1.62-2.00 pCi/sq cm, respectively.

Appendix 3

The values and calculations for the above- and below-ground standing crops.

ABOVEGROUND STANDING CROP

Location	dry wt per m*m (g/m*m)	Corrected values for Spartina
2	451.6	355.4
3	1125.6	893.7
4	1378.5	
5	1535.3	
7	1271.7	
8	434.8	300.4
11	418.9	

BELOWGROUND STANDING CROP

Location	dry wt (g)	Depth of core (cm)	Radius of core	Area of core	Bulk Den. of core (g/cc)	Predicted O.M. in 2cm slice (g/m*m)
2	130.6	34.0	7.0	5233.9	0.0250	499.1
3	78.4	25.0	9.5	7088.2	0.0111	221.2
4	89.6	29.0	7.0	4464.2	0.0201	401.4
5	132.9	18.0	7.0	2770.9	0.0480	959.3
7	97.2	13.0	7.0	2001.2	0.0486	971.4
8	89.8	35.0	7.0	5387.8	0.0167	333.3
10	103.2	35.0	7.0	5387.8	0.0192	383.1
11	29.2	16.0	7.0	2463.0	0.0119	237.1

BELOWGROUND STANDING CROP- ASHING

Location	Orig. wt (g)	Ashed wt (g)	% inorg. residue
2	13.53	4.17	30.8
3	17.28	4.80	27.8
4	16.76	5.88	35.1
5	14.36	4.71	32.8
7	16.23	4.64	28.6
8	19.71	6.20	31.5
10	14.48	4.01	27.7
11	17.75	5.82	32.8