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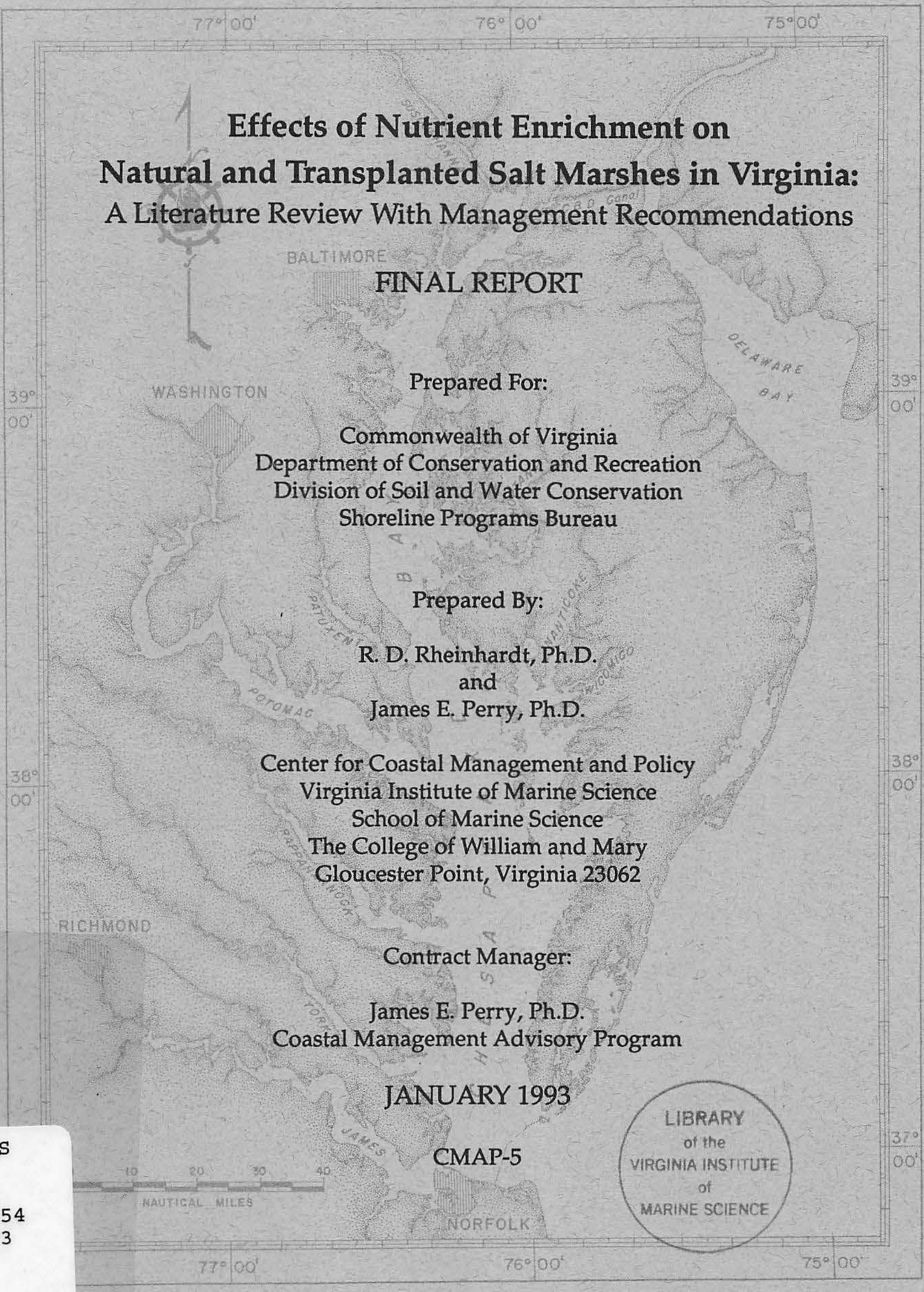

Effects of Nutrient Enrichment on Natural and Transplanted Salt Marshes in Virginia:

A Literature Review With Management Recommendations

FINAL REPORT

Prepared For:

Commonwealth of Virginia
Department of Conservation and Recreation
Division of Soil and Water Conservation
Shoreline Programs Bureau

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INTRODUCTION

At the climax of the Wisconsin glacial period about 18,000 years ago, global sea level was approximately 85 m lower than now and the western Atlantic coastline was located near the edge of the continental shelf (Kennet 1982). As continental glaciers melted during the Holocene, sea-level rose rapidly along the Atlantic coast. In the process, the sea drowned coastal river valleys forming estuaries. The flooding of the lower (coastal plain) valley of the Susquehanna River eventually formed the largest estuary in the world, the Chesapeake Bay.

Coastal salt marshes, which evolved to exploit the ecotone between land and estuary, have kept pace with sea level rise by eroding and reforming (accreting) at the edges of advancing shorelines. The continual erosion and accretion of tidal coastal wetlands has therefore been a natural consequence of sea level rise and estuarine evolution. As long as sea level continues to rise, the dynamic interactions between land and estuary will continue.

The rate at which coastal wetlands erode and accrete is likely linked to the rate at which sea level rises. Should the rate of sea level rise accelerate in response to the predicted warming of the planet's biosphere, the rate of shoreline erosion along Atlantic coast estuaries (including that of Chesapeake Bay) will also likely accelerate.

Coastal erosion is a major concern of shoreline property owners along the tidal portions of Chesapeake Bay. It has been estimated that approximately 85 ha of Chesapeake Bay shoreline have been damaged by erosion each year (Hardaway et al. 1984). Much of this erosion may be a natural consequence of storms, but man-made alterations of the coastline are known to accelerate the process in some instances. Although over the long term, tidal marshes have had no impact on the landward advance of marine transgressions, they do function as a physical buffer between land and sea in the short term and, by so doing, help reduce shoreline erosion (Dean 1978).

Marshes are successful at reducing shoreline erosion because the stems of marsh grasses create a physical impediment to the erosional potential of currents (Benner et al. 1982). By slowing current speed and buffeting waves, marsh grasses cause sediment and other particulate matter to settle out of the water column and onto the marsh (Boto and Patrick 1978, Knutson et al. 1982). By trapping sediment and organic particulates, marshes not only maintain their positions in relation to sea level, but often expand laterally as well (Redfield 1972).

Because marshes have been shown to be effective at damping wave energy and slowing shoreline erosion, the Commonwealth of Virginia has made it a priority to protect tidal wetlands from destruction. To facilitate the latter goal, the Shoreline Erosion Advisory Service (SEAS) of the Virginia Department of Conservation and Recreation encourages tidal shoreline property owners to establish salt marshes along eroding shorelines where such establishment appears feasible (primarily in areas of erosion along low gradient and low energy shorelines). The SEAS program also provides technical assistance for property owners to plant and successfully establish marshes.

In some instances, the planting or transplanting of marshes to stem shoreline erosion has been found to be less costly to property owners and more effective than constructing groins or bulkheads (Knutson 1977). In addition, marshes perform valuable functions which man-made structures are unable to provide; they provide habitat for wildlife (Day et al. 1989), contribute to the maintenance of water quality (Kadlec and Kadlec 1978), and provide dissolved nutrients and carbon to the biota of the adjacent estuary (Valiela 1984). However, marshes are not all equal in their ability to perform the above functions.

It has been assumed that marshes with more biomass per unit area or those with higher stem density are more effective at stemming erosion and stabilizing shorelines, particularly during storms (Hill, pers. commun.). Marsh nutrient enrichment studies conducted in the 1970s found that transplanted marshes produced more biomass and attained higher survival rates when nutrients (fertilizers) were added during their establishment. Because of these findings, SEAS and other resource agencies recommend the subsurface application of fertilizers (as slow release fertilizer pellets) when planting or transplanting marshes, and the broadcasting (surface application) of fertilizers the following season if the planted marshes are thought to be developing poorly.

In addition, SEAS has been advising shoreline property owners to fertilize natural marshes in order to increase marsh plant biomass (Hill, pers. commun.). These recommendations were developed on the basis of visual (qualitative) evaluations which suggested that marshes with higher biomass per unit area might be more effective in reducing erosion. However, no quantitative evidence has been cited to support these recommendations. Moreover, because nutrients added to tidal marshes can potentially be transported from the marsh to the adjacent estuary, such a recommendation could potentially contribute to the eutrophication of the estuary.

This study was initiated in response to a request by SEAS to review their technical recommendations concerning the fertilization of tidal salt marshes and determine whether those recommendations are sound in light of current scientific knowledge. The intention of this analysis was to determine whether benefits are derived from applying nutrients to marshes, and if so, whether those benefits outweigh the costs associated with the potential pollution of Chesapeake Bay's tidal waters.

To provide insight into the effects of adding fertilizers to salt marshes, the scientific literature was examined from four perspectives:

1) the dynamics of nutrient cycling in natural marshes,

2) the effects of nutrient loading (fertilization) on natural marshes,

3) the effects of nutrient enrichment on planted (man-made) marshes, and

4) the effects of marsh fertilization on adjacent estuaries.

LITERATURE REVIEW

Nutrient Cycling in Natural Salt Marshes

Salt marshes of eastern North America are dominated by the salt marsh cordgrass, *Spartina alterniflora* Loisel (hereafter referred to as simply, *Spartina*). Salt mashes are prevalent along tidal, low energy coasts where relative sea level is rising and where there is a sufficient supply of sediments for marshes to trap (trapping sediment enables marshes to maintain their elevation with respect to sea level). In eastern North America, these conditions occur behind barrier island complexes and within coastal bays and estuaries (Mitch and Gosselink 1986).

Spartina occurs in two distinct growth forms: a tall form and a short form. The two forms are generally considered to be stress-mediated ecophenes (Shea et al. 1975), but there is recent evidence suggesting that the two forms may be distinct species (Gallagher et al. 1988). The banks of tidal creeks and the fringes of tidal rivers are dominated by almost monospecific stands of tall-form *Spartina*, which commonly grow 1 to 2 m in height. Thus, tall-form *Spartina* inhabits the intertidal zone and so is flooded daily by tides. The habitat in which tall-form *Spartina* grows is commonly referred to as low marsh or streamside marsh (when extensive areas of marsh occur behind) and fringe marsh (when there is no extensive marsh area behind).

The interior parts of large marsh complexes are dominated by the short-form of *Spartina*, which seldom exceeds 0.5 m in height. These inner or high marsh communities inhabit the higher elevations (high intertidal or supratidal zone) inland of streamside marshes. Although inner marshes are only inundated by higher than normal tides, the soils of inner marshes are continuously saturated and anoxia is more severe in these areas than at streamside locations (Howes et al. 1981, Mendelssohn and McKee 1988). The low redox potential of inner marsh soils has been implicated as one possible factor causing the stunted growth of *Spartina* in inner marshes.

Salt marshes are one of the most productive terrestrial ecosystems in temperate latitudes, annually producing 330 to 3,700 g C/m^2 of aboveground biomass (Mitsch and Gosselink 1986). Belowground production in salt marshes is as high or higher than aboveground production (Valiela et al. 1976, Gallagher and Plumley 1979, Smith et al. 1979, Livingston and Patriquin 1981) with 460 to 4,780 g C/m^2 produced annually belowground (Day et al. 1989). In general, lower latitude marshes are more productive than marshes of higher latitudes and creekbank marshes are more productive than inner marshes.

Prior to the 1980s, it was believed that salt marshes were responsible for the high productivity of the coastal zone because they were found to export high levels of nutrients and carbon to adjacent coastal waters (see Nixon 1980 for a review of the earlier literature). Since 1980, salt marshes have been shown to be highly variable in nutrient output, both spatially (Wolaver and Zieman 1984, DeLuane et al. 1980) and seasonally (Wolaver and Zieman 1984, Scudlark and Church 1989). The current consensus is that salt marshes are generally a sink for nitrogen and phosphorus on an annual basis (DeLuane and Patrick 1980, DeLuane and Patrick 1983a, Wolaver and Zieman 1983) and a transformer of carbon (Wolaver and Spurrier 1988). The exchange characteristics of a marsh appears to depend upon its specific geomorphology (Odum et al. 1979) and its proximity to other wetland systems (Axelrad 1974, Wolaver and Spurrier 1988). Further, major storms, particularly rain storms at low tide, may be an important export mechanism that has rarely been factored into the annual budgets of marshes (Axelrad 1974, Chalmers et al. 1985, Scudlark and Church 1989, Whiting et al. 1989, Morris and Haskin 1990).

Phosphorus Enrichment

Most studies of nutrient cycling in salt marshes conclude that phosphorus is primarily controlled by geochemical processes (Nixon 1980) and that most phosphorus reaches marshes via sediment deposition. Although Reimold (1972) believed that his tracer studies showed that phosphorus is actively removed by *Spartina* (removed from the sediment by roots, pumped to the leaves, and exuded), other workers have refuted this concept (McGovern et al. 1980, Nixon 1980).

It is now generally believed that phosphorus (in the form of orthophosphate) in the sediment is geochemically retained under oxic conditions and released under anoxic conditions (Patrick and Khalid 1974), possibly due to the ability of oxidized iron in the sediment to absorb phosphorus and the reduced forms of hydrated iron oxide to release phosphate (Patrick and DeLuane 1977, Scudlark and Church 1989). On an annual basis, most salt marsh soils appear to function as a reservoir for phosphate and particulate phosphorus (Wolaver and Zieman 1984, Craft et al. 1989), and so probably retain more phosphorus than is required for plant production.

The experimental addition of orthophosphate to natural marshes has been shown to be ineffective in increasing *Spartina* production or average plant height (Sullivan and Daiber 1974, Valiela et al. 1975, Patrick and DeLuane 1976, Buresh et al. 1980), even though phosphorus levels in the plants have been shown to increase (Patrick and DeLuane 1976, Buresh et al. 1980). These results substantiate the prevailing view that sediment supplied to most natural marshes is sufficiently high in phosphorus to supply the nutritional needs of *Spartina* (i.e., phosphorus is not a limiting nutrient). Also, *Spartina* is capable of luxuriant uptake (DeLuane and Patrick 1980). On the other hand, the addition of phosphorus (as orthophosphate) to coarse, mineral soils (such as those derived from dredge spoils) has been shown to enhance *Spartina* production (Broome et al. 1975), indicating that such soils are probably initially low in phosphorus (i.e., phosphorus is a limiting nutrient under such conditions).

Nitrogen Enrichment to Natural Marshes

Nitrogen occurs in seven oxidation-reduction states in salt marsh soils and its transformations among forms is microbially mediated (in contrast to phosphorus which occurs in only one oxidation-reduction state and is geologically transformed). Thus, the cycling and transformations of nitrogen in salt marshes are much more complex than those of phosphorus. It is believed that in most salt marshes, particulate nitrogen and low levels of nitrate (NO_3) and ammonium (NH_4) are tidally transported to the marsh from the estuary, while dissolved organic nitrogen (DON) is exported from the marsh (Axelrad 1974, DeLuane et al. 1980, Whiting et al. 1989, Wolaver and Zieman 1983, Craft et al. 1989). Because NO_3 is quickly denitrified under anoxic conditions (Keeney 1973, Patrick and DeLuane 1976), very little is found in marsh soils, particularly during the growing season. Instead, almost all of the inorganic nitrogen found in salt marshes is in the ammonium form (Mendelssohn 1979).

Ammonium is generated *in situ* by the remineralization of organic matter, particularly via sulfate-mediated respiration (Howarth and Teal 1979, Mendelssohn 1979), and for the most part within the upper 10 cm of the root zone (Lord and Church 1983). *Spartina* takes up ammonium and incorporates it into tissue during the growing season and exports particulate ammonium during the winter (Wolaver and Zieman 1984). Before *Spartina* seneces at the end of the growing season, tissue nitrogen is mobilized and translocated to the rhizome (Mendelssohn 1979). In this way, nitrogen is conserved within salt marshes (Wolaver and Zieman 1983).

Most nitrogen enrichment studies have examined the effects of one of the following three nitrogen forms on *Spartina* production: 1) ammonium (in the form of ammonium nitrate or ammonium sulfate), 2) powdered sewage sludge, and 3) commercial fertilizers (mixture of nitrogen, phosphorus, and potassium). Urea and nitrate (NO_3) nitrogen have also been utilized to a lesser extent. Studies that examined plant tissue nitrogen levels after enrichment found that *Spartina* assimilated the added nitrogen, regardless of the form applied (Broome et al. 1975, Gallagher 1975, Patrick and DeLuane 1976, Valiela and Teal 1976, Chalmers 1979, Buresh et al. 1980).

Because inorganic ammonium (NH_4) is the major form of nitrogen found in salt marsh soils, it is the preferred form to use in manipulative experiments on the effects of nitrogen enhancement (Patrick and DeLuane 1976). The effects of sewage sludge enrichment have been studied because there was some hope that marshes would prove to be a viable and effective tertiary treatment option for domestic sewage (Valiela et al. 1976, Chalmers 1979). The effects of enrichment by commercial fertilizers on planted marshes have been examined because this source of nitrogen is the most readily obtainable by people involved in marsh restoration.

Sewage sludge and commercial fertilizers are not composed solely of nitrogen, but contain other chemical constituents. Thus, should *Spartina* biomass increase following enrichment by such a mixture, it would be impossible to unequivocally determine which of the chemical constituents is responsible for the increase. Investigators have generally assumed, however, that any increase in biomass following the application of sewage sludge or commercial fertilizer is attributable to the nitrogenous component of the mixture.

Patrick and DeLuane (1976) found that belowground fertilization (200 kg/ha of NH_4) of a streamside marsh led to a 15% increase in *Spartina* biomass. Similarly, Valiela et al. (1975) found that a biweekly broadcasting of sewage sludge (at 8.4 and 25.2 g/m^2/wk, 10% N) at streamside sites led to a doubling of tall-form *Spartina* biomass. In these two studies, nitrogen fertilizer was either buried in the substrate or periodically reapplied, thus increasing the likelihood that some of the nutrient would be assimilated before being removed by tides. However, neither study subjected their control plots to the same physical treatment as their experimental plots (i.e., compacting the root substrate of the planted plot via trampling). Therefore, the effects of the experimental treatment in this study (nutrient addition) cannot be statistically separated from the effects of the physical manipulation (trampling) (Hurlbert 1984).

Sullivan and Daiber (1974) found that the application of ammonium sulfate (at 20 g/m^2/month) to a Delaware marsh led to an almost tripling of short-form *Spartina* aboveground standing crop. On Cape Cod, Valiela et al. (1975) found that the biweekly application of sewage sludge to inner marshes at both low (8.4 $g/m^2/wk$) and high ($25.2\ g/m^2/wk$) rates doubled short-form *Spartina* biomass. Buresh et al. (1980) found that after adding radioactively labelled ammonium sulfate to an inner (short-form *Spartina*) marsh, biomass increased 28% over control plots. All of the above studies show that short-form *Spartina* significantly increases in biomass following the addition of nitrogen fertilizer.

Gallagher (1975), working on Sapelo Island, Georgia, found that the addition of 200 kg/ha of ammonium nitrate (placed belowground) significantly increased the live biomass of short-form *Spartina* 1.7 times over that of the controls, but failed to significantly increase the biomass of tall-form (streamside) *Spartina* plots. This experiment was one of the few that applied adequate procedural controls: holes were dug in the control plots that were identical to those dug in the experimental plots (dug to place the fertilizer underground) and trampling appears to have been similar in both control and experimental plots.

The fertilized short-form *Spartina* marsh in Gallagher's study reached a biomass and maximum height intermediate between that of unfertilized short-form *Spartina* plots and unfertilized tall-form *Spartina* plots. Other marsh fertilization studies supported the finding that nitrogen enrichment of short-form *Spartina* increases both its biomass and height (Valiela et al.

1978, Buresh et al. 1980, Haines 1980). However, the plants never attain the biomass or height of natural, unfertilized stands of streamside (tall-form) *Spartina*. The increase in the biomass and height of short-form *Spartina* following nitrogen enrichment suggests that nitrogen is a limiting nutrient in inner marshes. However, the failure of fertilized inner *Spartina* marshes to attain the vitality of tall-form *Spartina* marshes suggests that factors besides low nitrogen may stress inner marshes.

Paradoxically, Mendelssohn (1979) found that interstitial concentrations of NH_4 were about six times higher in an inner marsh than in a streamside marsh. Laboratory experiments of NH_4 uptake by *Spartina* (Morris 1980) suggested that the concentrations of inorganic nitrogen found in inner marshes should not be limiting to *Spartina* growth. It is not known why ammonium enrichment of natural marshes leads to an increase in short-form *Spartina* production or why tall-form *Spartina* often fails to respond to such enrichment. This is a major unresolved puzzle among those studying *in situ* salt marsh nutrition.

The inability of inner marsh *Spartina* to reach the same height as the taller streamside plants following fertilization and the apparent excess of nitrogen in inner marsh sites suggests that the inner marsh (short-form *Spartina* zone) is either secondarily limited by some other resource or resources (Valiela et al. 1978) or that short-form *Spartina* differs genetically from the tall-form (Stalter and Batson 1969, Gallagher et al. 1988).

This has led researchers to explore other environmental factors that might be preventing the assimilation of nitrogen or otherwise limiting short-form *Spartina* production in the inner marsh areas.

Salinity has been found to be stressful to *Spartina*, as substantiated by greenhouse experiments (Linthurst and Blum 1981, Linthust and Seneca 1981). However, salinity has been found to inhibit *Spartina* production only when relatively high, particularly when above 35 ppt (Smart and Barko 1980). In natural situations, inner marshes are usually less frequently flooded by tidal water and have higher evapotranspiration rates. These conditions usually lead to higher interstitial salinity levels in inner marshes (Nestler 1977). However, not all inner marshes are more saline than adjacent streamside marshes. In the deltaic marshes of Louisiana, salinity has been found to be lower in inner marshes, due to high inputs of surficial freshwater (Buresh et al. 1980, Howes et al. 1981).

Although most inner marshes are rarely flooded by tides, they remain saturated and anoxic most of the time. Soil redox values have been found to be much lower in inner marshes than in streamside marshes (Howes et al. 1981, DeLuane et al. 1983b, Mendelssohn et al. 1981, Mendelssohn and McKee 1988). This may be because streamside marshes (tall-form *Spartina* areas) are regularly flooded by tides, which deposit sediment and nutrients, replenish oxygenated water and sulfate (SO_4), and remove toxins (such as sulfides). Thus, tidal flushing may be important to

Spartina productivity.

An *in situ* experiment by King et al. (1982) seems to substantiate the importance of tidal flushing to salt marsh productivity. They found that by irrigating inner marsh sediments, the height and aboveground production of short-form *Spartina* are increased. The importance of aeration was further substantiated in a greenhouse experiment by Linthurst and Seneca (1981) which found that the production of *Spartina* is enhanced when its roots are oxygenated. Thus, differences in oxygen status may be the proximal cause for the height differences in the two *Spartina* forms.

Reduced compounds, particularly sulfides and ammonium, accumulate in inner marshes at higher levels than in streamside locations (Patrick and DeLuane 1977, King et al. 1982, DeLuane et al. 1983b, Lord and Church 1983, King 1988, Mendelssohn and McKee 1988). Some of the sulfide in inner marshes binds with iron monosulfide (FeS) to form pyrite (FeS_2), which is relatively inert (Howarth 1978), and some sulfate is regenerated in the oxic areas around the roots (Lord and Church 1983). Excess sulfide is toxic to *Spartina* if it accumulates to high enough concentrations (Howarth and Teal 1979, Howes et al. 1981, Mendelssohn and McKee 1988). Sulfide also appears to inhibit nitrogen uptake in the field (King et al. 1982) and in culture (Morris 1980, Morris and Dacey 1984, Bradley and Morris 1990). Thus, high concentrations of sulfide may ultimately be responsible for the reduced vigor of *Spartina* in inner marsh areas.

In a laboratory culture experiment, Bradley and Dacey (1990) found that the ammonium uptake rate of *Spartina* was greatest under oxic conditions. When nitrogen was added to short-form *Spartina* in the field, an increase in the rate of aboveground production led to increases in both the evapotranspiration rate, and the amount of oxygen drawn into the sediment (Dacey and Howes 1984, Howes et al. 1981, Howes et al. 1986). The increase in soil oxygen appeared to have lowered soil sulfide concentrations and further reduced stress. Howes et al. (1986) suggested that this feedback mechanism was responsible for the increase in aboveground production following ammonium enrichment to inner marshes. The major problem with this hypothesis is that if inner marshes already posses excess ammonium, then a further addition of ammonium should not be expected to change uptake rates.

Recently, there have been attempts to quantify the effect nutrient enrichment has on marsh infauna. Preliminary data suggests that nutrient additions to natural marshes cause a profound change in infuanal community structure, with a significant loss of polychaetes from the community (Currin, pers. comm.). A change in infaunal composition could change nutrient cycling dynamics in marshes and thereby change their functional attributes.

Nutrient Enrichment to Planted Marshes

A few experimental studies on the effects of nitrogen addition to salt marshes have been conducted in planted (man-made) marshes. Usually, the substrate of planted marshes is initially low in nutrients. This is because the soils of such areas are often sandy (along beaches or eroding shorelines) or are composed of dredge spoil.

On the protected (lagoon) side of a North Carolina barrier island, Broome et al. (1975) found that nitrogen enrichment of planted *Spartina* seedlings produced significantly more biomass than those not receiving nitrogen. The study area was composed primarily of coarse-grained (sandy) beach material that was low in nutrients and was subjected to high ambient salinity (32 ppt). Broome et al. (1975) also found that belowground biomass (roots and rhizomes) produced significantly more biomass than controls following nitrogen enrichment (168-672 kg/ha) on the protected barrier island beach. Their results differ from those Valiela et al. (1976) obtained in a natural marsh (on fine-textured soil) to which fertilizer was added (8.4-25.2 $g/m^2/wk$, 10% N). In the natural marsh, nitrogen enrichment not only failed to increase belowground production, but fertilization actually led to a decrease in root biomass. The above two studies may have differed in their results because of the different characteristics of the marsh substrates (sand vs. silt) and age (created vs. established marsh).

In another study, Broome et al. (1983) determined that nitrogen (224 kg/ha) had to be applied belowground in a slow release-form (Osmocote) in order to significantly increase *Spartina* biomass in a transplanted marsh along an eroding estuarine shoreline (5 to 22 ppt salinity, void of organic matter). Surface application of fertilizer was ineffective in increasing production over controls. It was believed that nitrogen applied to the marsh surface was lost to the estuary before it could be incorporated into plant tissue. Thus, the manner in which nitrogen is applied, broadcast aboveground or buried belowground in a slow release form, is likely to have an effect on the efficiency of plant utilization and the likelihood of excess nutrients entering the adjacent estuary.

Nutrient Losses to the Estuary

Valiela et al. (1973) estimated that although as much as 20% of the nitrogen and 10% of the added phosphorus they applied (by broadcasting sewage sludge fertilizer, 25 g/m^2, 10% N and 6% P) to a inner marsh was lost, most of the loss occurred within three days of application. Working in a Georgia salt marsh, Gallagher (1975) estimated that 53% of the nitrogen he buried in his inner marsh plots (at 200 kg/ha) was bound in the aerial biomass after 10 wk and that 36% was still present 1 yr later (these plants were cohorts and not successive generations). In Louisiana, Patrick and DeLuane (1976) estimated that 29% of the radioactively labelled (^{15}N) ammonium and 1% of the phosphorus they buried in their streamside marshes in May (both applied at a rate of 200 kg/ha) could be accounted for in *Spartina* biomass near the end of the growing season (September). Chalmers (1979) found that sewage sludge broadcast over an inner marsh (100

$g/m^2/wk$, 2% N) retained about half of the added nitrogen after one year. After applying labelled nitrogen ($^{15}NH_4$) belowground (at the rate of 200 kg/ha) to an inland location, Buresh et al. (1980) found that 28% of the labelled nitrogen was recovered from the aboveground portion and 29% from the belowground portion after 4 months.

DeLuane et al. (1983a) found that the application of radioactively labelled ammonium in four belowground installments (10-15 cm belowground at 72 kg N/ha) was almost completely (93%) retained by a vigorous marsh. The experimental location, streamside or inner marsh, was not provided. Further, each of their experimental plots were surrounded by a metal casing inserted 15-20 cm below the surface, which could have affected lateral losses through interstitial spaces.

Although the above studies suggest that much of the fertilizer added to experimental marshes will probably be retained there, an appreciable percentage (20-50%) of the added nitrogen will likely be lost to the adjacent creeks and estuaries (little can be predicted about the fate of the added phosphorus).

Valiela et al. (1973) found a positive correlation between tide height and the amount of nutrients lost from their fertilized inner marsh, indicating that tidal action was primarily responsible for nutrient losses from the marsh. Because streamside marshes are more frequently flooded by tides than inner marshes, loss rates of added nutrients are probably higher at streamside locations than inner marsh locations, particularly in marshes where fertilizers are broadcast over the marsh surface rather than being buried underground. Nutrients are also removed from the marsh by runoff from rainstorms at low tide. Excess nutrients removed by tides or rain runoff flow from salt marshes to marsh creeks and then to adjacent estuaries where they can be assimilated by estuarine phytoplankton.

Estuarine phytoplankton populations are nitrogen limited (Ryther and Dunstan 1971) and nitrogen enrichment of estuarine waters leads to phytoplankton blooms (Barlow et. al 1963, Ryther and Dunstan 1971, Pomeroy et al. 1972, Darnell and Soniat 1981). Phytoplankton blooms lead to anoxic conditions in poorly mixed benthic environments (Jaworski 1981, McErlean and Reed 1981, Webb 1981) and can detrimentally effect estuarine biota by disrupting natural benthic-pelagic interactions (Rowe et al. 1975, Verity 1987). Thus, although there have been no studies examining the fate of applied nutrients, nutrient losses from fertilized salt marshes have the potential to adversely affect estuarine processes.

Salt Marshes as Erosion Buffers

Almost all of the marsh fertilization studies reviewed in this paper examined the influence of fertilization on *Spartina* biomass. No studies were found relating plant biomass to the ability of a marsh to reduce erosion. A laboratory flume experiment conducted by Gleason et al. (1979), found that higher densities of *Spartina* dissipated more wave energy and led to higher accumulations of sediment than lower stem densities. Unfortunately, of all the studies of natural marshes reviewed in this paper, none of them examined the effects fertilization might have on stem rigidity and only one (Valiela et al. 1978) measured its effect on *Spartina* density. In that study, streamside marshes were found to be naturally less dense than inner marshes and fertilization was found to further decrease stem density. Further, fertilized *Spartina* tended to possess a less developed root structure and a higher shoot-to-root ratio (Valiela et al. 1976, Smart and Barko 1980) than unfertilized plants.

The above studies suggest that nitrogen enrichment of *Spartina* may tend to concentrate more biomass in fewer and taller stems than unfertilized plants. Also, fertilized plants were shown to have relatively less root biomass to support taller aboveground portions than shorter unfertilized plants. The possibility exists, therefore, that even though fertilized *Spartina* might appear to be more robust, fertilized plants may tend to be more prone to breakage and uprooting when subjected to high wave energy than unfertilized plants. Thus, if current erosion abatement and sedimentation rates are indeed positively related to stem density as the Gleason et al. (1979) study found and fertilization leads to a reduction in stem density as the Valiela et al. (1978) study found, then one would hypothesize that fertilizing natural marshes could be counterproductive in inhibiting shoreline erosion. Future research should be directed toward field work to further investigate this hypothesis.

CONCLUSIONS

Phosphorus enrichment has been shown to be ineffective in increasing *Spartina* production in salt marshes. Except for salt marshes created upon coarse grained, nutrient-poor substrate, most natural marshes contain sufficient phosphorus for *Spartina* growth and reproduction. Thus, most salt marsh nutrient studies over the past 20 years have focused on the effects that nitrogen addition has on plant biomass and height.

The addition of nitrogen has been shown to significantly increase the aboveground biomass and height of the short-form *Spartina* that dominate inner marshes. The effects of nitrogen enrichment on tall-form *Spartina*, however, is not as clear: tall-form *Spartina* sometimes fails to increase in biomass following nitrogen addition.

The different responses of inner and streamside *Spartina* to nitrogen fertilization may reflect differences in their tidal regimes. Tides may be more likely to remove added nutrients from streamside marshes (where tidal exchange is prevalent) than from inner marshes (where tidal exchange is minimal). An alternative explanation is that tall and short-form *Spartina* are different species and thus have different physiological requirements.

Plant biomass, however, may not be the most important factor in determining the potential of a salt marsh to protect a shoreline from erosion. Paradoxically, one study found that the addition of fertilizer to a marsh not only increased *Spartina* biomass, but reduced *Spartina* stem density as well. This could prove important for coastal erosion control in light of a flume study that found a significant positive relationship between *Spartina* stem density and the dissipation of wave energy and sediment accumulation. This suggests that, by reducing stem density, fertilization may be counterproductive in inhibiting shoreline erosion. However, because so few studies have attempted to directly relate the effects of marsh fertilization with erosion abatement potential, more work must be done in these areas before a more definitive conclusion can be drawn.

One study, however, have shown that fertilizing natural *Spartina* marshes reduces root biomass and increases plant height (particularly in inner marshes). The combined effect of a reduction in belowground biomass and an increase in aboveground biomass and plant height following fertilization could make *Spartina* stems more prone to breakage and the roots less able to stabilize substrate when subjected to high wave energy and currents. Thus, fertilizing marshes could have an adverse effect on the ability of *Spartina* to reduce erosion. Again, more research must be done to examine the relationship between fertilization, stem rigidity, and root adhesive properties (particularly in shoreline marshes) before more definitive conclusions can be drawn.

Experimental evidence concerning the loss of fertilizers from marshes, however, is more conclusive. It has been shown that from 20-50% of the fertilizers applied to marshes is soon lost to adjacent creeks and estuaries. The highest loss rates appear to be associated with streamside locations (tall-form *Spartina* areas). An even higher percentage of fertilizer would be lost if applied during spring tides or prior to heavy rains. Because Chesapeake Bay is already overburdened with nutrients, any fertilizers lost to the estuary would likely contribute to the further eutrophication of the Bay.

Important functions of salt marshes could also be altered by fertilization. The fertilization of a marsh may lead to saturation with nutrients. Once saturated, marshes would lose one of their important functions: their capacity to filter excess organic nutrients from the estuary. Also, if fertilization changes the structure of infaunal communities in marshes, nutrient cycling dynamics could be radically altered as well.

The erosion and accretion of salt marshes are a natural phenomenon of estuarine evolution. Salt marshes tend to be at equilibrium with the coastal processes of erosion and accretion and so can only function as a physical buffer of erosional processes in the short term. Salt marshes cannot prevent sea level rise; their fates are ultimately controlled by the physical attributes of their adjacent coastal regimes.

Marshes are most successful at reducing erosion along low energy coastlines where sediment supply is sufficient for their maintenance. Under such conditions, the reestablishment of eroded marshes following catastrophic events may prove beneficial in the short term. In order to successfully establish marshes in such eroded areas (and upon dredge spoils), the belowground application of a slow release fertilizer is often necessary. This is because coarse-grained sediments (at sandy sites) and dredge spoils are usually low in nutrients.

RECOMMENDATIONS

Any benefits derived from the addition of nutrients to salt marshes must be weighed against the potential deleterious effects nutrient enrichment might have on the marsh and adjacent estuaries. Most North American estuaries, including Chesapeake Bay and its tributaries, are already overburdened with nutrients from urban and agricultural fertilizers and industrial effluent. Adding excessive nutrients to marshes may exacerbate pollution problems, particularly if fertilizers are broadcast at streamside sites.

Because shoreline erosion primarily occurs at streamside sites (along creeks and rivers) and not in inner marshes, marsh fertilization by shoreline property owners may lead to a loss of fringe marshes adjacent to creeks and the estuary. In addition, there is some evidence that an increase in plant biomass following marsh fertilization might lead to a reduction in *Spartina* stem density, and this could lower the capacity of a marsh to prevent shoreline erosion. Therefore, prior to advising shoreline property owners to fertilize established (natural) marshes in order to enhance erosion abatement, it is important to determined whether or not the desired effect of additional fertilizers (i.e., erosion abatement) will actually be achieved.

Presently, marsh fertilization should be reserved for the establishment of new marshes, and in such instances, the belowground placement of slow release fertilizer is recommended. Because *Spartina* translocates nutrients to its rhizomes prior to senescence at the end of the growing season (i.e., it is not expending much of its energy in the uptake of nutrients from the soil), fertilizers should not be administered in the fall. Instead, fertilizer should be applied to planted marshes in the early part of the growing season or during planting. Studies have shown that 200 kg/ha (225 lb/acre) of nitrogen is sufficient to establish a healthy stand of *Spartina*. In many cases, the benefits of establishing marshes or replacing damaged marshes in order to prevent shoreline erosion may outweigh the potential negative consequences of adding nutrients to adjacent creeks and estuaries.

Reducing shoreline erosion using the natural buffering capabilities of salt marshes should continue to be encouraged wherever possible. Salt marshes not only function as erosion buffers, but also convert inorganic nutrients to organic matter and provide shelter and habitat to other estuarine organisms. The protection and planting of salt marshes as a natural erosion buffer should be continued to be encouraged where appropriate, but the public should also be reminded that man-made alterations to shorelines to prevent erosion are only short-term solutions.

For the long-term survival of salt marshes and the benefits they provide to man and the estuarine biota, salt marshes must be allowed to advance shoreward as sea level rises. The SEAS program can provide a important public service by educating the public about the importance of providing buffer zones behind marshes to enable them to advance with the rising sea and continue to provide their important functions into the future.

Further Research Needs

Most scientific studies have examined the effects of fertilization on marsh biomass. However, other parameters, such as stem density and stem rigidity, could be as important as biomass to erosion abatement considerations. It would be beneficial to obtain quantitative evidence on the effect of fertilization on marsh stem density and rigidity and the effects these two parameters have on erosion abatement potential. Quantitative data collected under controlled experimental conditions is essential to elucidate these relationships. Data will have to be obtained from a sufficient number of replicate plots collected over a wide geographic range and physical conditions, combined with meaningful controls (see Hurlbert 1984 for experimental designs).

It is possible that the density of any given marsh may partially reflect the physical parameters (i.e., fetch, tidal regime, nutrient input, etc.) to which it is subjected. The erosion potential of a marsh may also be controlled by the geology of the site. Information on the effects of the above factors, coupled with experimental manipulation of stem density in marshes, might tell us whether adding plants to eroding natural marshes would be expected to have any beneficial, short-term effects on slowing erosional processes. If, however, the biological and physical characteristics of a marsh are in approximate equilibrium with the physical parameters impinging upon the site, a manipulated natural marsh site might quickly return to its pre-manipulated condition once planting and fertilization are discontinued.

Finally, because the Chesapeake Bay is already overburdened with nutrients from agricultural, urban, and suburban runoff, future studies conducted on the effects of marsh fertilization should consider the loss rates and consequences of adding nutrients to adjacent creeks and estuaries.

Literature Cited

Axelrad, D.M. 1974. Nutrient flux through the salt marsh ecosystem. Ph.D. Dissertation, School of Marine Science, College of William and Mary, Gloucester Point, Virginia, USA.

Barlow, J.P., C.J. Lorenzen, and R.T. Myren. 1963. Eutrophication of a tidal estuary. Limnol. and Oceanogr. 8:251-262.

Benner, C.S., P.L. Knutson, R.A. Brochu, and A.K. Hurme. 1982. Vegetative erosion control in an oligohaline environment, Currituck Sound, North Carolina. Wetlands 2:105-117.

Boto, K.G. and W.H. Patrick, Jr. 1978. Role of wetlands in the removal of suspended sediments. *In* Greeson, P.E., J.R. Clark, and J.E. Clark (eds.), Wetlands Functions and Values: the State of Our Understanding. American Water Resources Association, pp. 479-489.

Bradley, P.M. and J.T. Morris. 1990. Influence of oxygen and sulfide concentration on nitrogen uptake kinetics in *Spartina alterniflora*. Ecology 71:282-287.

Broome, S.W., W.W. Woodhouse, Jr, and E.D. Seneca. 1975. The relationship of mineral nutrients to growth of *Spartina alterniflora* in North Carolina: II. The effects of N, P, and Fe fertilizers. Soil Science Soc. of America Proc. 39:301-307.

Broome, S.W., E.D. Seneca, and W.W. Woodhouse, Jr. 1983. The effects of source, rate, and placement of nitrogen and phosphorous fertilizers on growth of *Spartina alterniflora* transplants in North Carolina. Estuaries 6:212-226.

Buresh, R.J., R.D. DeLaune, and W.H. Patrick, Jr. 1980. Nitrogen and phosphorous distribution and utilization by *Spartina alterniflora* in a Louisiana gulf coast marsh. Estuaries 3:11-121.

Chalmers, A.G. 1979. The effects of fertilization on nitrogen distribution in a *Spartina alterniflora* salt marsh. Estuarine Coastal Marine Sci. 8:327-337.

Chalmers, A.G., R.G. Wiegart, and P.L. Wolf. 1985. Carbon balance in a salt marsh: Interactions of diffusive export, tidal deposition and rainfall-caused erosion. Estuarine, Coast, and Shelf Sci. 21:757-771.

Craft, C.B., Broome, S.W., and E.D. Seneca. 1989. Exchange of nitrogen, phosphorous, and organic carbon between transplanted marshes and estuarine waters. J. Environ. Qual. 18:206-211.

Currin, C. and S. Broome. 1992. Ecosystem function in restored and constructed wetlands. Symposium paper at the NOAA COP-EHP Workshop, Nov. 18-20, Gloucester Point, VA. (unpublished)

Dacey, J.W.H. and B.L. Howes. 1984. Water uptake by roots controls water table movement and sediment oxidation in short *Spartina* marsh. Science 224:487-489.

Darnell, R.M. and T.M. Soniat. 1981. Nutrient enrichment and estuarine health. *In* B.J. Neilson and L.E. Cronin (eds), Estuaries and Nutrients. International Symposium on the Effects of Nutrient Enrichment in Estuaries, Humana Press, Inc., Clifton, New Jersey, USA.

Day, J.W., C.A.S. Hall, W.M. Kemp, and A. Yanez-Arancibia (eds). 1989. Estuarine Ecology. John Wiley and Sons, Inc., New York, NY, USA.

Dean, R.G. 1978. Effects of vegetation on shoreline erosional processes. *In* Greeson, P.E., J.R. Clark, and J.E. Clark (eds.), Wetlands Functions and Values: the State of Our Understanding. American Water Resources Association, pp. 415-426.

DeLaune, R.D. and W.H. Patrick, Jr. 1980. Nitrogen and phosphorous cycling in a gulf coast salt marsh. *In* Kennedy, V.S. (ed.), Estuarine Perspectives, 5th Biennial International Estuarine Research Conference. Academic Press: New York, New York.

DeLaune, R.D., T.C. Feijtel, and W.H. Patrick, Jr. 1980. Nitrogen flows in a Louisiana gulf coast salt marsh: spatial considerations. Biogeochemistry 8:25-37.

DeLuane, R.D., C.J. Smith, and W.H. Patrick, Jr. 1983a. Nitrogen losses from a Louisiana gulf coast salt marsh. Estuarine, Coastal and Shelf Sci. 17: 133-141.

DeLuane, R.D., C.J. Smith, and W.H. Patrick, Jr. 1983b. Relationships of marsh elevation, redox potential, and sulfide to *Spartina alterniflora* productivity. J. Soil Sci. Soc. Am. 47:930-935.

DeLuane, R.D., C.J. Smith, and M.D. Tolley. 1984. The effect of sediment redox potential on nitrogen uptake, anaerobic root respiration and growth of *Spartina alterniflora* Loisel. Aquatic Botany 18:223-230.

Gallagher, J.L. 1975. Effect of an ammonium nitrate pulse on the growth and elemental composition of natural stands of *Spartina alterniflora* and *Juncus roemerianus*. Amer. J. Bot 62:644-648.

Gallagher, J.L. and F. Plumley. 1979. Underground biomass profiles and productivity on Atlantic coastal marshes. Amer. J. Bot. 66:156-161.

Gallagher, J.L., G.F. Somers, D.M. Grant, and D.M. Seliskar. 1988. Persistent differences in two form of *Spartina alterniflora*: a common garden experiment. Ecology 69:1005-1008.

Gleason, M.L., D.A. Elmer, N.C. Pien, and J.S. Fisher. 1979. Effects of stem density upon sediment retention by salt marsh cord grass, *Spartina alterniflora* Loisel Estuaries 2:271-273.

Haines, E. 1980. Salt marsh creation: Impact of sewage. *In* Rehabilitation and Creation of Selected Coastal Habitats: Proceedings of a Workshop, Sapelo Island, GA. U.S. Fish and Wildlife Service, Biol. Services Progr., Washington, Dc, Report FWS/OBS-80.

Hardaway, C.S., G.R. Thomas, A.W. Zacherle, and B.K. Fowler. 1984. Vegetative erosion control project: Final report 1984. Virginia Institute of Marine Science, College of William and Mary, Gloucester Point, VA, USA.

Howarth, R.W. 1978. Pyrite: Its rapid formation in a salt marsh and its importance in ecosystem metabolism. Science 203:49-51.

Howarth, R.W. and J.M. Teal. 1979. Sulfate reduction in a New England salt marsh. Limnol. Oceanogr 24:999-1013.

Howes, B.L., R.W. Howarth, J.M. Teal, and I. Valiela. 1981. Oxidation-reduction potentials in a salt marsh: Spatial patterns and interactions with primary production. Limnol. Oceanogr 26:350-360.

Howes, B.L., J.W.H. Dacey, and D.D. Goehringer. 1986. Factors controlling the growth form of *Spartina alterniflora*: Feedbacks between above-ground production, sediment oxidation, nitrogen and salinity. J. Ecol. 74: 881-898.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Monogr. 54:187-211.

Jaworski, N.A. 1981. Sources of nutrients and the scale of eutrophication problems in estuaries. *In* B.J. Neilson and L.E. Cronin (eds), Estuaries and Nutrients. International Symposium on the Effects of Nutrient Enrichment in Estuaries, Humana Press, Inc., Clifton, New Jersey, USA.

Kadlec, R.H. and J.A. Kadlec. 1978. Wetlands and water quality. *In* Greeson, P.E., J.R. Clark, and J.E. Clark (eds.), Wetlands Functions and Values: the State of Our Understanding. American Water Resources Association, pp. 436-456.

Keeney, D.R.. 1973. The nitrogen cycle in sediment-water systems. J. Envir. Quality 2:15-29.

Kennet, J.P. (ed). 1982. Marine Geology. Prentis-Hall, Inc., Englewood Cliffs, New Jersey, USA.

King, G.M., M.J. Klug, R.W. Wiegert, and A.G. Chalmers. 1982. Relation of soil water movement and sulfide concentration to *Spartina alterniflora* production in a Georgia salt marsh. Science 218:61-63.

King, G.M. 1988. Patterns of sulfate reduction and the sulfur cycle in a South Carolina salt marsh. Limnol. Oceanogr. 33:376-390.

Knutson, P.L., R.A. Brochu, W.N. Seelig, and M. Inskeep. 1982. Wave damping in *Spartina alterniflora* marshes. Wetlands 2:87-104.

Knutson, P.L. 1977. Planting guidelines for marsh development and bank stabilization. Coastal Engineering Technical Aid #77-3, U.S. Army Corps of Engineering Research Center, Fort Belvoir, Virginia.

Linthurst, R.A. and E.D. Seneca. 1981. Aeration, nitrogen and salinity as determinants of *Spartina alterniflora* Loisel growth response. Estuaries 4:53-63.

Livingston, D. and D. Patriquin. 1981. Below growth of *Spartina alterniflora* Loisel: Habitat, functional biomass, and nonstructural carbohydrates. Estuarine Coastal Shelf Sci. 12:579-588.

Lord, C.J., III and T.M. Church. 1983. The geochemistry of salt marshes: sedimentary ion diffusion, sulfate reduction, and pyritization. Geochimica et Cosmochimica Acta 47:1381-1391.

Mendelssohn, I.A. 1979. Nitrogen metabolism in the height forms of *Spartina alterniflora* in North Carolina. Ecology 60:574-584.

Mendelssohn, I.A., K.L. McKee, and W.H. Patrick, Jr. 1981. Oxygen deficiency in *Spartina alterniflora* roots: metabolic adaption to anoxia. Science 214:439-441.

Mendelssohn, I.A. and K.L. McKee. 1988. *Spartina alterniflora* die back in Louisiana: time-course investigation of soil waterlogging effects. J. of Ecology 76:509-521.

McErlean, A.J. and G. Reed. 1981. Indicators and indices of estuarine enrichment. *In* B.J. Neilson and L.E. Cronin (eds), Estuaries and Nutrients. International Symposium on the Effects of Nutrient Enrichment in Estuaries, Humana Press, Inc., Clifton, New Jersey, USA.

McGovern, T.A., L.J. Laber, and B.C. Gram. 1980. Characteristics of the salt excreted by *Spartina alterniflora* Loisel and their relation to estuarine production. Estuarine and Coastal. Mar. Sci. 9:351-356.

Mitch, W. J. and J. G. Gosselink (eds). 1986. Wetlands. Van Nostrand Reinhold Company, New York, NY, USA.

Morris, J.T. 1980. The nitrogen uptake kinetics of *Spartina alterniflora* in culture. Ecology 61:1114-1121.

Morris, J.T. and J.W.H. Dacey. 1984. Effects of O_2 on ammonium uptake and root respiration by *Spartina alterniflora*. Am. J. Bot. 71:979-985.

Morris, J.T. and B. Haskin. 1990. A 5-yr record of aerial primary production and stand characteristics of *Spartina alterniflora*. Ecology 71:2209-2217.

Nestler, J. 1977. Interstitial salinity as a cause of ecophenic variation in *Spartina alterniflora*. Estuarine and Coastal Mar. Sci. 5:707-714.

Nixon, S.W. 1980. Between coastal marshes and coastal waters - a review of twenty years of speculation and research on the role of salt marshes in estuarine productivity and water chemistry. *In* Hamilton, P. and MacDonald, K.B. (eds), Estuarine and Wetland Processes, Plenum Press.

Odum, W.E., J.S. Fisher, and J.C. Pickral. 1979. Factors controlling the flux of particulate organic carbon from estuarine wetlands. *In* R.J. Livingston (ed), Ecological Processes in Coastal and Marine Systems. Plenum Press, New York, NY, USA.

Patrick, W.H., Jr. and R.A. Khalid. 1974. Phosphate release and sorption by soils and sediments; effect of aerobic and anaerobic conditions. Science 186:53-55.

Patrick, W.H., Jr. and R.D. DeLaune. 1976. Nitrogen and phosphorous utilization by *Spartina alterniflora* in a salt marsh in Barataria Bay, Louisiana. Est. Coast. Mar. Sci. 4:59-64.

Patrick, W.H., Jr. and R.D. DeLaune. 1977. Chemical and biological redox systems affecting nutrient availability in the coastal wetlands. Geoscience and Man 28:131-137.

Pomeroy, L.R., L.R. Shenton, R.D.H. Jones, and R.J. Reimold. 1972. Nutrient flux in Estuaries. *In* G.E. Likens, (ed.), Nutrients and Eutrophication: The Limiting-nutrient Controversy. Limnol. and Oceanogr., Special Symposium. v. 1.

Redfield, A.C. 1972. Development of a New England salt marsh. Ecol. Monogr. 42:201-237.

Reimold, R.J. 1972. The movement of phosphorous through the salt marsh cord grass, *Spartina alterniflora* Loisel. Limnol Oceanogr 17:606-611.

Rowe, G.T., C.H. Clifford, and K.L. Smith, Jr. 1975. Benthic nutrient regeneration and its coupling to primary productivity in coastal waters. Nature 255:215-217.

Ryther, J.H. and W.M. Dunstan. 1971. Nitrogen, phosphorus, and eutrophication in the coastal marine environment. Science 171:1008-1013.

Scudlark, J.R. and T.M. Church. 1989. The sedimentary flux of nutrients at a Delaware salt marsh site: a geochemical perspective. Biogeochem. 7:55-75.

Shea, M.L., R.S. Warren, and W.A. Niering. 1975. Biochemical and transplantation studies of the growth form of *Spartina alterniflora* on Connecticut salt marshes. Ecology 56:461-466.

Smart, R.M. and J.W. Barko. 1980. Nitrogen nutrition and salinity tolerance of *Distichlis spicata* and *Spartina alterniflora.* Ecology 61:630-638.

Smith, K,, R. Good, and N. Good. 1979. Production dynamics for above- and below-ground components of a New Jersey tidal marsh. Estuarine Coastal Mar. Sci. 9:189-201.

Stalter, R. and W.T. Batson. 1969. Transplantation of salt marsh vegetation, Georgetown, South Carolina. Ecology 50:1087-1089.

Sullivan, M.J. and F.C. Daiber. 1974. Response in production of cord grass, *Spartina alterniflora*, to inorganic nitrogen and phosphorous fertilizer. Chesapeake Science 15:121-252.

Valiela, I., J.M. Teal, and W. Sass. 1973. Nutrient retention in salt marsh plots experimentally fertilized with sewage sludge. Est. Coast. Shelf Sci. 1:261-269.

Valiela, I., J.M. Teal, and W. Sass. 1975. Production and dynamics of experimentally enriched salt marsh vegetation and the effects of experimental treatment with sewage sludge. I. Biomass production and species composition. J. Appl. Ecol. 12:973-981.

Valiela, I., J.M. Teal, and N.Y. Persson. 1976. Production and dynamics of experimentally enriched salt marsh vegetation: belowground biomass. Limnol. Oceanogr. 21:245-252.

Valiela, I., J.M. Teal, and W.G. Deuser. 1978. The nature of the growth forms of the salt marsh grass *Spartina alterniflora* American Naturalist 112: 461-470.

Valiela, I. 1984. Marine ecological processes. Springer-Verlag, New York, New York, USA.

Verity, P.G. 1987. Factors driving changes in the pelagic trophic structure of estuaries, with implications for the Chesapeake Bay. *In* M.P. Lynch and E.C. Krome (eds), Perspectives on the Chesapeake Bay: Advances in Estuarine Science. Chesapeake Research Consortium, Publication #127, U.S. Environmental Protection Agency Document CBP/TRS 16/87.

Webb, K.L. 1981. Conceptual models and Processes of nutrient cycling in estuaries.

Whiting, G.J., H.N. McKellar, Jr., J.D. Spurrier, and T.G. Wolaver. 1989. Nitrogen exchange between a portion of vegetated salt marsh and the adjoining creek. Limnol. Oceanogr 34:463-473.

Wolaver, T.G. and J.D. Spurrier. 1988. Carbon transport between a euhaline vegetated marsh in South Carolina and the adjacent tidal creek: contributions via tidal inundation, runoff and seepage. Mar. Ecol. Prog. Ser. 42:53-62.

Wolaver, T.G. and J. Zieman. 1983. Effect of water column, sediment, and time over the tidal cycle on the chemical composition of tidal water in a mesohaline marsh. Mar. Ecol. Prog. Ser. 12: 123-130.

Wolaver, T.G. and J. Zieman. 1984. The role of tall and medium *Spartina alterniflora* zones in the processing of nutrients in tidal water. Est. Coast. Shelf Sci. 19:1-13.