

*Suffolk County Vector Control &  
Wetlands Management Long Term Plan  
& Environmental Impact Statement*



*Task 3 Literature Review  
Book 9 Part I: Salt Marsh Health*

*Prepared for*

**Suffolk County Department of Public Works  
Suffolk County Department of Health Services  
Suffolk County, New York**

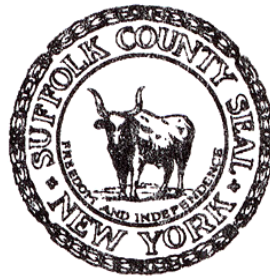
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**SUFFOLK COUNTY VECTOR CONTROL AND WETLANDS MANAGEMENT  
LONG - TERM PLAN AND ENVIRONMENTAL IMPACT STATEMENT**

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### **List of Abbreviations and Acronyms**

|        |   |
|--------|---|
| ICI    | Invertebrate Community Index                  |
| LUI    | Land Use Index                                |
| MA-CZM | Massachusetts Coastal Zone Management         |
| MHHW   | Mean High Higher Water                        |
| MSL    | Mean Sea Level                                |
| NASA   | National Aeronautics and Space Administration |
| OMWM   | Open Marsh Water Management                   |
| PCI    | Plant Community Index                         |
| SET    | Sedimentation-Erosion Tables                  |
| USEPA  | US Environmental Protection Agency            |

## **Executive Summary**

Salt marshes are inherently ephemeral coastal ecosystems. They continually evolve in response to changing environmental conditions, and thus their health is at best subjectively defined and likely to be biased by personal experience. As such, the condition of a natural, pre-Colonial marsh system is only as relevant today as it is possible to accurately reconstruct it. Ultimately, there is no single way to define salt marsh health, and instead the definition must be constructed from a variety of parameters.

A salt marsh that is largely natural and healthy can present a broad array of physical habitats, species diversity, abundances, and temporal variability. This range of expected conditions is due in part because salt-marsh flora and fauna are necessarily adapted to great variability in their physical environment and, therefore, possess considerable tolerance to stress. Consequently, it has not proven straightforward to develop general health assessments for salt-marsh systems.

Any approach that is broad enough to address the wide range of phenomena that comprise salt marshes tends to fail because it does not address important specifics integral to understanding particular marshes and their settings. One US Environmental Protection Agency-sponsored project has certain well-designed aspects to it, for example, but ultimately only calls for the derivation of area or even marsh specific criteria for assessment purposes. Other approaches that key on specific aspects of marsh development or maintenance may only address issues that are not general problems. Two examples are illustrative. One assessment effort focuses on marsh loss, for example, which is pertinent for Louisiana and Jamaica Bay, New York, but of much less utility for areas such as Long Island's Peconic Bay system. Another effort, from Massachusetts, created metrics that attempt to link land use to marsh integrity measures. This may be useful in understanding changes in marsh ecosystems, but probably would not be useful for understanding processes that cause the large losses in Louisiana or Jamaica Bay.

Because of these difficulties this report does not seek one overall assessment means. Instead, it attempts to define salt marsh health by considering:

- physical habitat stability through sediment erosion and deposition
- connectivity of the marsh with adjacent ecosystems

- within-marsh ecological communities
- overall water quality.

Because marshes sit at the high-energy interface of land and sea, a stable or benevolent physical setting is key for marsh development and stability. This stability, by definition, requires the efficient exchange of materials between the marsh and its adjacent upland and marine habitats, which can be defined as connectivity. In conjunction with physical processes, a persistent plant community is also required as a major positive feedback for habitat stability. Marsh plants also comprise the ecosystem's base in terms of primary production and habitat structure. Thus, they play dual roles in physical and ecological health. Finally, water quality encompasses several key components of marsh health including nutrient regime, water and sediment exchange, and marsh-surface hydrodynamics. Taken together, these multiple characteristics reflect key aspects of salt-marsh condition and might be used through monitoring or investigation to assess overall health.

Regardless of the approach chosen, the scope, cost, and success of any wetland assessment will ultimately hinge on the chosen definition of a healthy marsh. This requires that long-term goals such as habitat stability, biodiversity, and water quality, be explicitly defined, and placed in the context of any anticipated manipulations, such as restoration, Open Marsh Water Management, and land use controls.

This will not be an easy task. Caveats are numerous. First, it is apparent from past health assessments that a fully comprehensive approach or “do-it-all” project is not feasible in terms of logistics and cost. This requires agreement on a definition of health that is compatible and achievable within the framework of expected work efforts. Within these logistical constraints, a do-it-all approach cannot be accommodated. As well, too broad a definition of health would likely introduce competing management goals, such as those that focus on biodiversity, vector control, or habitat stability. Conversely, there is a danger of defining goals relating to salt marsh health that are too specific, and that ignore the reality that marshes are dynamic systems which yield benefits under numerous states of health and development, including growing or drowning, being old or young, and whether they are polluted or pristine.

To date, no specific measure has been identified that serves as a sufficiently reliable indicator of marsh health. In large part, this is because it cannot be assumed that pristine or healthy salt marshes are similar in key physical and biological characteristics. A way around this problem



may be to think of indicators of health in the negative, in that monitoring would seek to measure differences from acceptable conditions. It may also be that the presence of clear and negative trends, such as rapid marsh loss and/or dominance of non-native species, would be another means of determining a marsh is not healthy.

Trend analysis is likely to be the more useful assessment means, because in most cases salt marshes are already altered or impacted, with limited possibility for restoration to pristine conditions. Thus, data trends can reveal whether an otherwise impacted marsh system is relatively healthy or is losing its health. In systems as complex and variable as salt marshes, just identifying such trends would be important and useful accomplishments.

Despite the complexity of marsh systems, and many and varied ways they have been changed and manipulated, there are several characteristics that might be expected of all systems in reasonable health. These include a relatively stable vegetated area, limited extent of invasive plants, and presence of obligate marsh species. Table 1 suggests how these might be measured and evaluated to create first-order assessments of marsh health.

Table 1 – Proposed first-order indices for marsh health on Long Island, to be more explicitly defined based on discussion of user groups.

| <b>Health Indicator</b>   | <b>Good Condition</b>   | <b>Alert Status</b>  |
|---|---|--|
| Marsh stability   | Net loss of vegetated wetland <1% per year  | Net loss of vegetated wetland >3% per year   |
| Plant health (for <i>S. alterniflora</i> only – health of the high marsh presumably threatened by <i>Phragmites</i> invasion rather than vegetation loss as in the low marsh) | <5% of vegetated marsh with stem densities below 100/m <sup>2</sup><br>or<br>total below-ground biomass from 0-20 cm >3000 g/m <sup>2</sup> | >10% of vegetated marsh with stem densities below 100/m <sup>2</sup><br>or<br>total below-ground biomass from 0-20 cm <1500 g/m <sup>2</sup> |
| Invasive species  | <30% <i>Phragmites</i> sp.  | >50% <i>Phragmites</i> sp.   |
| Resident finfish  | Killifish group represented in most or all suitable habitats  | Killifish group absent from >30% of suitable habitats  |
| Species of Interest (e.g., marsh sparrows, terrapins, forb plants, others)  | Stable population or consistent use of marsh by species of special State or Federal status  | No species of concern present or viable  |
| Temporal trends   | Selected indicator does not trend negatively in 3 or more consecutive years   | Selected indicator trends negatively in 3 or more consecutive years  |
| Note: marsh characteristics between Good and Alert condition should be considered to be Of Concern and monitored closely  |   |  |

## 1. Marsh Health Assessment Techniques

The literature survey has shown that marsh health may be defined in many ways that are often disjoint. These definitions develop from the background and interests of involved parties.

The broadest division of health categories is between those oriented toward biological resources and those focused on habitat stability of plants and physical attributes. In areas where marsh loss has not traditionally been a focus, such as New England and Georgia, health assessments are often geared toward ecological attributes. Examples include the Massachusetts Coastal Zone Management (MA-CZM) projects (Carlisle et al., 2004) and the Alberts et al. (1999) US Environmental Protection Agency (USEPA) study. In contrast, areas experiencing significant marsh loss such as Maryland and Louisiana more often develop assessment strategies to evaluate health in terms of physical habitat degradation and conversion to open water. A prime example is the Kearney et al. (1999) National Aeronautics and Space Administration (NASA) study; there have been many more efforts in Louisiana.

Presently, the status of New York salt marshes spans both of these categories, with a general eastward gradient of degraded to more pristine systems. Habitat stability is perhaps the more relevant issue in the South Shore Estuary, particularly in the western salt marshes, whereas the less impacted systems of the Peconic Estuary might be more appropriately assessed for their ecological uniqueness and integrity.

### 1.1 Complex Assessments

A clear demonstration of the lack of consensus on the definition of marsh health is illustrated by numerous requests-for-proposals from USEPA that target health metrics for coastal ecosystems. Only one such academic research study has been funded for work specifically on salt marshes (Alberts et al., 1999). This large, multidisciplinary study investigated numerous potential health indicators within macrophyte, fungal, and microcrustacean communities. The project report concludes that only

the relative abundance of tanaids and amphipods at a marsh site together with the inverse of the incidence of *P. pandalicola* in estuarine populations of grass shrimp may provide a promising measure of environmental stress.

Plant gas exchange was also noted as a seemingly good health indicator, but

was found to vary with tidal range, marsh elevation, and pore-water salinity. [Therefore] future attempts to use this indicator as a measure of plant health must standardize for these environmental factors

which in itself introduces many new variables that will limit the sensitivity of such an indicator. A project related publication (Pennings et al., 2002) further concluded that

“overall, these measures [of lower trophic level responses] showed little promise as rapid indicators of salt marsh health.”

The ambivalent results of this federally funded research reflect the inherent difficulty of characterizing an ecosystem that is influenced by many interrelated factors, factors that support a broad range of normal conditions.

USEPA recently produced a relatively comprehensive manual, *Methods for Evaluating Wetland Condition* (USEPA, 2002a, b). It includes protocols for both freshwater and tidal marshes. Its general approach is to collect detailed, transect-based measures of vegetation, birds, invertebrates, and water quality. The primary goals were to develop methods for evaluating:

- 1) the overall ecological condition of wetlands using biological assessments; and
- 2) nutrient enrichment of wetlands, which is one of the primary stressors damaging wetlands in many parts of the country

(USEPA, 2002b)

The USEPA approach is strongly weighted toward the ecological integrity and biodiversity of marsh systems and, aside from water quality measures, does not evaluate physical conditions such as sedimentation and hydrology. The primary targets appear to be species of special concern within amphibian, invertebrate, bird, and forb plant groups.

Based in part on the USEPA approach, a series of case studies has been conducted. One was initiated by MA-CZM in 1996. This series of projects, called the “Massachusetts Wetland Bioassessment Pilot,” is the only one of these projects to have considered Atlantic salt marshes. It was conducted at sites on Cape Cod and North Shore, Massachusetts. Each of three completed studies has generated results that indicate decreasing biological integrity in association with increasing land-use stressors. The land use stressors included modeled nitrogen load and area of impervious upland. An analysis of results showed that the strongest correlations among stressors

and indicators were achieved when individual indicators were combined into single indices (Carlisle et al., 2004), as

different metrics, both for plants and invertebrates, displayed different signals and the combination of these various indicators into an overall index serves to integrate these characteristics or variables.

MA-CZM found that the multi-indicator Plant Community Index (PCI) and Invertebrate Community Index (ICI) “responded inversely and rather predictably to two of the three disturbance indicators,” being Land Use Index (LUI) and modeled nitrogen. However, correlations were not especially strong, with  $R^2$  values of 0.45 ( $p < 0.01$ ) for PCI and 0.51 ( $p < 0.01$ ) for ICI against Land Use Index and 0.35 ( $p < 0.05$ ) and 0.42 ( $p < 0.05$ ) against modeled nitrogen. Nonetheless, the findings of the MA-CZM study are encouraging and suggest that appropriate metrics can be developed for assessing ecological communities in salt marshes.

However, the conclusions of the report note that

this work is very resource intensive [and] that field effort and resource needs quickly add up....more dialogue needs to occur at policy and funding levels as Federal and State governments decide whether understanding the condition of wetland resources is an important investment

(Carlisle et al., 2004)

This is a striking statement that emphasizes the high level of commitment required for such assessments. It also stresses the importance of precisely defining marsh health at the start of a project and targeting an assessment approach that is compatible with the end-user’s final objectives, presumably some form of marsh restoration or management.

## 1.2 Rapid Assessments

Turner et al. (2004) compared above- and below-ground biomass in 12 healthy and impaired *S. alterniflora* marshes of Louisiana, where the health of the marsh was determined *a priori*. They found that both the accretion rate and above-ground biomass was indistinguishable between the different kinds of sites, which is consistent with findings from Long Island marshes (Kolker et al., 2004). Turner et al. reported that below-ground biomass varied significantly and consistently with marsh health. On average, degraded sites had approximately 50 percent less total below-ground root biomass, and over 80 percent less live root biomass, compared to healthy sites. Because accretion rates strongly correlate with biomass in Louisiana, as is the case in mid-

Atlantic areas as well (Gross et al., 1991), Turner et al. suggest that the organic biomass is the primary control on marsh-surface elevation. This makes it a relevant indicator of marsh health. They concluded that, in order to sustain a marsh, or to conduct a reasonable restoration project, it is important to understand the below-ground live root matrix and microenvironment. Unfortunately, no universal value for below-ground biomass is appropriate, as even a comparison among healthy *S. alterniflora* marshes in Louisiana and North Carolina shows a four-fold difference (Gross et al., 1991; Turner et al., 2004). Nevertheless, it may be that constructing below-ground biomass values for Long Island salt marshes, bearing in mind differences between physical settings, such as those on the north shore, in the Peconic Estuary, and those in the South Shore Estuary, will help to categorize the health of the marshes.

Another rapid assessment of salt-marsh health was created Kearney et al., premised on the observation that marshes erode in a quantifiable pattern from increasing areas of standing water within a marsh system (Kearney et al., 1988; Kearney et al., 1999; Kearney and Stevenson, 1991). A simple model for estimating marsh loss as a function of the amount of water in a given area was developed. Table 2 shows the resulting the classification scheme. Actual assessments of marsh health for the US mid-Atlantic coast were then based on the area of open water, determined from 30 m<sup>2</sup>-pixel TM Landsat data, gridded to four hectares per cell in the final output product. Tidal stage was not considered, which introduced a potentially significant source of error. The coverage included New York State, including all Suffolk County marshes. Results published on the project website capture the general westward trend of increasing marsh deterioration along Long Island's South Shore. However, several large areas identified as severely to completely deteriorated (Classes 3 and 4) included Nissequogue River and Hempstead Bay; other assessments, such as Muschacke (2001) do not agree with such findings, and suggests that the technique may not always be successful. Furthermore, the results present only a single snapshot of the marsh systems, and do not reflect temporal trends that may otherwise show stability or limited rates of degradation. Another limitation, noted in the reports, is that the actual controls of observed marsh-surface and open-water patterns cannot be deduced. For these, further groundtruthing and field research are required to understand the system. Nonetheless, these data

can give a manager some idea of how much time elapses between different stages of deterioration, making possible timely intervention

which is an important issue, illustrated by the rapid Jamaica Bay marsh losses.

Table 2. Marsh health classifications, based on the percent of open water within a marsh boundary (Kearney et al., 1999)

| <b>Class</b> | <b>Open Water</b> | <b>Status</b>          | <b>Responses</b>   |
|--------------|-------------------|------------------------|--|
| 1            | 0-20%             | Healthy                | healthy stable substrate   |
| 2            | 21-30%            | Moderate Deterioration | tidal streams widen  |
| 3            | 31-50%            | Severe Deterioration   | ponds enlarge and coalesce, streams widen further  |
| 4            | >50%              | Complete Deterioration | large ponds remain all year, streams cut off islands and enlarge ponds, allowing greater wind fetch, creating a feedback that leads to more rapid deterioration. |

Despite limitations, aerial photo or satellite surveys are perhaps the best way, in terms of producing relevant results cost effectively, to rapidly and widely assess marsh stability. Caveats remain, as groundtruthing continues to be an essential tool for remote sensing and results do not reveal the original cause of degradation, nor the impact of the degradation on the larger biological community. For example, work by Turner et al. (2004) on below-ground biomass shows this to truly be an early indicator of plant stress as it decreases greatly before observable changes in aboveground biomass or stem density, which are the indicators mapped by remote sensing. Therefore, remote sensing of marsh loss may track indicators that occur too late in the degradation process to be of great help for management, particularly of small systems that can disappear relatively quickly. Nevertheless, benefits of remote sensing are that it yields integrated measures of marsh losses or gains, thereby accounting for the natural heterogeneity of marsh systems, not just localized losses or gains. Even an otherwise healthy marsh can experience local erosion without significant degradation, and so changes in marsh area extent can reveal losses not recognized by ground monitoring alone.

Remote sensing has proven to be a rapid and relatively inexpensive technique for large-scale assessments. Indeed, remote sensing in combination with GIS analysis has proven essential for regional and global studies of wetlands, particularly from remote or unstable areas. At a more local scale, however, the limitations of remote sensing begin to compete with its advantages. Despite major advances in image resolution and interpretation, significant groundtruthing continues to be required because users typically want fine-scale assessments (less than 10 m<sup>2</sup>) when working at a local scale. Error remains large at boundaries, where spectra must be

averaged to pixel size, which for satellites collecting relevant spectral bands remains at best greater than or equal to 10 m. This boundary error is especially relevant to marshes, which typically have long perimeters that are poorly defined because of changing tidal levels. Meter-scale resolution is presently available in the visible-light spectrum from platforms such as Quickbird and Ikonos, but infrared remains the most suitable wavelength for marsh assessments because of its superb land-water contrast. One very consequential limitation of remote sensing is that it does not generally reveal the cause of observed changes, necessitating additional ground-based investigation to identify causative mechanisms and develop a management strategy. As a consequence, remote sensing is perhaps best suited as a monitoring tool rather than a stand-alone assessment technique (Kearney et al., 1999).

Ground-based field efforts comprise the other major approach for assessing marsh health. Some benefits of a ground-based approach include:

- the opportunity for continuous or repeat observations
- definition of specific targets, such as species of concern
- direct and high resolution, mm to tens of m, measures of processes and conditions.

Negatives often associated with ground-based research are that it is labor intensive and consequently expensive. Otherwise, it is often limited in its spatial and temporal coverage. In recognition of these limitations, but also of the importance of the resulting data, USEPA has fostered a state-level “Wetland Volunteer Monitoring Program” (U.S. EPA, 2002a). This is intended to involve the public in wetland stewardship, and to perhaps significantly enhance the database from which key management strategies will be developed.



## **2. Characteristics of a Healthy Marsh**

Another approach to develop a definition of a healthy marsh is to describe attributes that belong to healthy marshes, as opposed to attempting the holistic approach discussed above.

### **2.1 Physical habitat stability**

#### **2.1.1 Overview of vertical and lateral marsh processes**

A critical assessment of salt-marsh health may be made in terms of its physical stability, in which the interaction of waves, tides, and storms with local sediment sources controls the balance of erosion and deposition (Leonard et al., 1995b). In this context, however, both vertical and horizontal processes of marsh gain and/or loss must be considered as they have different rates and controls.

In the vertical dimension, the key aspect of stability can be expressed by marsh-surface elevation relative to mean sea level (MSL), whereby the marsh surface ranges from MSL to mean high higher water (MHHW) across a range of floral communities. Marsh-surface elevation is a function of sediment deposition, below-ground organic production, surface erosion, and shallow compaction (Cahoon et al., 1995). The competing process, relative sea-level change, is a function of global (eustatic) changes, and regional temperature (steric) and land-surface (isostatic) contributions (Gornitz, 1995). In the case of Long Island, the mean rate of relative sea-level rise over the past century has been 2.7 mm/yr as determined by tide-gauge records from The Battery, New York Harbor (Gornitz, 2001). Estimates of future sea-level change vary widely based on different global climate change scenarios, but a 30 cm rise in sea level by 2050 is considered very plausible (Rosenzweig and Solecki, 2001). Across the US, present rates of marsh accretion appear to be keeping pace with sea-level rise, except in the Mississippi delta, Chesapeake Bay, and locally along the Atlantic coast (Bricker-Urso et al., 1989; Hartig et al., 2002; Reed, 1995). However, measured accretion rates do not always correlate well with marsh loss, except in the most severe cases, suggesting that other factors are also important in marsh stability (Kolker et al., 2004). Regardless, the rate of sea-level rise is increasing as a result of both environmental and anthropogenic changes, and many questions remain as to whether salt-marsh accretion will keep pace with the increasing sea-level rise, or if they will eventually drown in place (Reed, 1995).

In the horizontal dimension, physical marsh stability is also affected by perimeter loss as a consequence of lateral erosion. These effects are generally localized, but the problem is widespread, and can be severe (Nyman et al., 1994; Schwimmer, 2001). Salt marshes naturally develop in low wave-energy settings. This is because higher energy waves are very efficient at sediment erosion via shoreface scour, subtidal resuspension, and bedload transport (Allen, 2000). Where sediment delivery to the coast from upland or offshore sources is sufficient, marsh edges can remain relatively stable under conditions of moderate wave-energy. However, anthropogenic modifications of the coastal zone have frequently altered sediment and/or wave regimes. In general, localized delivery of sediment to the coast has declined in recent decades due to tributary damming, shoreline hardening, and expansion of the buffer zone of non-tilled and non-impervious upland, commonly forest, adjacent to the shoreline (Meade, 1982; Milliman, 1992; Milliman and Meade, 1983). The impact of such changes on Long Island sediment delivery has not been determined, but it is tentatively considered not to be a major factor in local marsh stability. In contrast, dredging activities have been extensive on Long Island, and in many cases have encroached or crossed the critical marsh-estuary boundary. This is the boundary along a stable marsh perimeter that consists of some distance, on the order of tens of meters, of shallow water that buffers the marsh edge from wave attack. In this case, even moderate dredging can dramatically increase local wave impacts, because wave energy is a function of wave height squared, and wave height is a function of water depth (Demir et al., 2004). Therefore, impacts on salt marsh perimeters increase with the depth, proximity, and fetch of dredged sections of the estuary (Cox et al., 2003; Schwimmer, 2001).

### **2.1.2 Patterns of marsh-surface accretion**

Accretion rates reported in the literature typically represent the time-averaged results of shorter-term sedimentation and erosion processes. Since accretion rates are reported as a unit thickness per time, generally in cm/yr, it is often understood that sediments and organic matter accumulate at a rate close to this long-term average. In reality, sedimentation is an episodic, non-steady-state process. The marsh surface may only receive significant sediment input on a small number of days during the year (Goodbred and Hine, 1995; Reed, 1989; Roman et al., 1997). This is particularly true for the marsh interior and high marsh, because of the distance from the sediment source and infrequent flooding, respectively.

For example, a typical marsh-surface accretion rate of 3 mm/yr converts to a mass flux of 0.5 g/cm<sup>2</sup>, or about 0.35 g of mineral sediment annually, assuming a 30 percent organic content. Using values of 50 mg/l for suspended sediment concentrations, typical Long Island tidal waters, a marsh-surface flooding depth of 10 cm, and an inundation frequency of 20 times per month, 0.12 g of sediment would be delivered annually by normal tidal processes. This amount is less than half the average rate of sediment accumulation in a marsh, demonstrating that episodic events, such as the passage of strong weather fronts, must play an important role with their high winds, wave-induced sediment resuspension, and possible coastal set-up (i.e., wind tide). Indeed, this has been shown by numerous studies comparing daily to weekly sedimentation rates under fair- and foul-weather conditions (Childers and Day, 1990; Leonard et al., 1995b; Nyman et al., 1995a; Reed, 1988; Wood and Hine, 2003). Furthermore, most tidally delivered sediment is deposited within several meters of a creek edge, which emphasizes the importance of infrequent, high-energy wind tides for marsh-surface accretion in the marsh interior and to the high marsh. For microtidal regimes, such as is found along much of Long Island's south shore, sediment delivery to marshes is even more event driven, because of the limited capacity for weak tidal currents to erode and transport sediment.

### **2.1.3 Sediment requirements**

Salt-marsh-surface elevation can be maintained despite rising sea level by both inorganic sediment accumulation and the production of above- and below-ground biomass (Nyman et al., 1995b; Nyman et al., 1993). This is in contrast with freshwater marsh systems, where accretion is sometimes solely through organic production. This is borne out by comparing the average organic content of freshwater, brackish, and salt marshes. Fresh marshes typically have an organic content of 70 to 100 percent, brackish marshes 50 to 70 percent, and salt marshes 10 to 50 percent (Mitsch and Gosselink, 2000). Within salt marshes, organic contents generally increase from 10 to 30 percent in low marsh to 30 to 50 percent in higher-marsh settings. The main reason for this variation is that, unlike fresh water, sea water is high in sulfate (SO<sub>4</sub><sup>2-</sup>), which generally becomes reduced to various sulfide species in marsh porewaters (Lord and Church, 1983b). At least one of these reduced species, hydrogen sulfide (H<sub>2</sub>S), is toxic to marsh plants (Howarth and Teal, 1979). Some amount of inorganic mineral input is required to buffer against the buildup of H<sub>2</sub>S in the marsh soils (DeLaune et al., 1983a, b). This buffering occurs

because sulfide precipitates as an inert mineral phase, pyrite ( $\text{FeS}_2$ ), in the presence of dissolved iron (Fe). However, the only adequate supply of iron to a marsh is through the delivery of inorganic sediments to the marsh surface during flooding (Lord and Church, 1983a, b). Thus, inorganic sediments serve as a geochemical buffer in addition to contributing to marsh-surface elevation.

These edaphic factors are critical to marsh health. They are also very complicated. In the case of *Spartina alterniflora*, which colonizes the frequently inundated low-marsh zone, the presence of special aerenchyma tissue allows it to oxygenate its root zone, thereby forming a local redox barrier to toxic sulfide (Linthurst and Seneca, 1980). In the high marsh, for *S. patens*, less frequent inundation and better drainage lead to a smaller flux of  $\text{SO}_4^{2-}$ , and limit the buildup of  $\text{H}_2\text{S}$ . This permits this plant zone to accrete largely through organic production as mineral inputs are not as necessary. Overall, the salt-marsh sulfur cycle is affected by:

- the flux of sulfate (a function  $f$  of dissolved concentration and inundation period),
- its rate of reduction ( $f$ [temperature, microbial activity, organic loading]), and,
- its complexation with mineral matter ( $f$ [sediment input, Fe dissolution])

(Luther et al., 1986a, b)

Impacts of sulfide on salt-marsh health, however, are further complicated by initial plant conditions, which influence their ability to withstand stresses either from high sulfide concentrations and/or long exposure times to it. Despite a lack of full understanding of these processes, and their great temporal and spatial variability (Luther et al., 1991), sulfide toxicity is recognized as the main reason for losses of vegetated marsh interior, known as marsh drowning (Howarth, 1984). In general, increased inundation, decreased sediment input, and decreased water quality and drainage are primary candidates for increasing sulfide stress and degradation of marsh health.

#### **2.1.4 Monitoring sedimentation, erosion and marsh-surface elevation**

To understand the role of sediment patterns on marsh health, it is important to know how accretion rates are determined, and what exactly they reflect in terms of marsh processes. In

general, accretion rates are dependent upon multiple factors, including:

- sediment input
- distance from channel
- flooding regime
- microtopography
- plant community structure
- substrate autocompaction

(Orson et al., 1998; Stumpf, 1983)

Common techniques used to determine accretion rates include marker horizons and sedimentation-erosion tables (SET), for short-term measures (sub-annual to annual), and measurements of sediment concentrations of radioisotopes such as  $^{137}\text{Cs}$ ,  $^{210}\text{Pb}$  and  $^{14}\text{C}$  for long-term measures, on the order of decades to millennia (Armentano and Woodwell, 1975; Cahoon et al., 2000; DeLaune et al., 1978; Roman et al., 1997). In addition, Orson et al. (1998) recommend that, to best estimate rates of accretion, evidence should be taken from multiple dating techniques, including:

- radioisotope analysis
- storm lenses
- metal and other contaminant profiles
- artificial surface markers
- historic records
- foraminifera and pollen indicators.

Despite the recognition that the use of multiple techniques is preferable, the most widely used method of assessing how well a marsh is accreting is to compare radioisotope-derived accumulation rates with tide-gauge derived sea-level records (Orson et al., 1998). SETs are arguably the most effective method for measuring actual elevation changes, which when used in

conjunction with marker horizons are more relevant than accretion rates alone (Cahoon et al., 2000).

One limitation of both tide gauges and short-term accretion measures is the difficulty of quantifying long-term trends, and, therefore, the stability of the marsh, within the strong interannual variability of sea level and sedimentation. For example, storm and wind events are an important factors in defining variability in sedimentation rates because accumulation events occur episodically. They are often the dominant process by which marsh-surface elevation keeps pace with sea-level rise (Goodbred and Hine, 1995; Reed, 1989; Roman et al., 1997). However, establishing a long-term accretion trend relative to sea-level rise is difficult because these events are episodic and not regular. The variability even is seasonal. At a Long Island salt marsh, it was found that maximum accretion rates occurred during the growing season from March to September, and that erosion or reduced accretion occurred from October to February during plant senescence (Richard, 1978). Thus, annual accretion rates should be calculated based on data representing several years in order to account for seasonal differences and short-term variations, and to allow long-term rates to smooth out the short-term variables (Bricker-Urso et al., 1989; van Wijnen and Bakker, 2001).

### **2.1.5 Role of plants in habitat stability**

Plants play an important role in physical marsh stability, and thus are a useful tool in understanding the fate of salt marshes in respect to sea-level rise. Reed (2000) identified several key ways in which plants enhance the stability of a coastal marsh system, including:

- colonization of unvegetated, erodable substrates
- vegetative binding of creek-bank and shore-edge sediments
- enhanced marsh-surface accretion via the baffling of flow by the grass canopy
- in-situ, above- and below-ground production of organic matter.

First, colonizing marsh plants such as *S. alterniflora* will grow to the limit of their physiological tolerances along the boundaries of a salt marsh, and thereby serve to buffer adjacent plants in the marsh interior. These boundary plants baffle wave energy, trap and bind sediments, and limit the effects of lateral erosion (Reed, 2000). This important colonizing role can commonly be observed where *S. alterniflora* revegetates tidal flats adjacent to an eroded marsh edge (Handa

and Jefferies, 2000; Vince, 1985). In general, a marsh system in a healthier state will commonly have a fringe of *S. alterniflora* colonizing the lowermost edges of a marsh system, reflecting that the marsh edge is stable or aggrading. However, cyclical advances and retreats of a marsh edge occur under the control of climatic variability, and so care should be taken in assessing absolute marsh health based the presence or lack of a fringe of *S. alterniflora*. Marsh plants also serve to stabilize creek banks (Garofalo, 1980), which can be stable over very long periods, hundreds to thousands of years (Ashley and Zeff, 1988).

Within the marsh, plants play key roles in both surficial and below-ground processes. At the marsh surface, Leonard and Luther (1995) demonstrated the significance of the dense grass-stem canopy in baffling overmarsh flow, which enables fine-grained sediment to settle to the surface and contribute to overall vertical accretion processes. In addition, a significant amount of sediment adheres to the plant stems themselves while inundated by tidal waters; these adhered sediments fall to the marsh surface after drying during low tide. Stems at the highest densities were the most efficient at sediment trapping sediment, but high stem densities at the marsh edge can limit the distance to which sediment is transported into the marsh. However, plant stems at any density significantly increase entrapment and retention of sediment over non-vegetated settings (Leonard et al., 1995a; Leonard and Luther, 1995). Below the marsh surface, research by DeLaune et al. (1990) suggests that the root-zone growth of *S. alterniflora* is a major influence on marsh-surface elevation via the production of below-ground biomass. Potential deficits of sediment accumulation under increasing rates of sea-level rise may be compensated by below-ground plant production. However, changes in soil conditions as a result of sea-level rise can have potentially negative effects on plant health and thus their ability for increasing below-ground biomass (Benner et al., 1991; DeLeeuw et al., 1990; Groenendijk and Vinklievaart, 1987; Turner et al., 2004). There have been no studies that identify whether plants of a particular marsh will be able to increase below ground biomass to compensate for increased sea-level rise, or whether the stress associated with sea-level rise instead will lead to increased inundation and declines in health . It is logical to assume that plants at or near their environmental tolerances will be most susceptible to loss. Based on these findings, Reed (2000) suggested that essential questions for coastal managers are:

- 1) which vegetation is best to plant in degrading marshes, and

- 2) what is the necessary stem density to enhance deposition.

No specific answers to these questions have been identified in recent research, particularly given the broad morphological and physical variability of marsh systems.

In contrast to the stabilization provided by live plants, the death of marsh plants leads to positive feedbacks that result in the catastrophic loss of vegetated wetland. DeLaune et al. (1994) documented an eight to 10 cm decrease in marsh-surface elevation within one year of plant death. Such loss of elevation is sufficient to inhibit, or even preclude, the regeneration and recolonization of plants there. The rapid loss in elevation was attributed to collapse of the living root-rhizome matrix, whereby turgid pressure of vascular gases and fluids was lost as the plant membranes degraded. Such gases and fluids can comprise 90 percent of the volume of the below-ground biomass. Thus, it would account for significant compaction if dispersed. Day et al. (1994) also documented a rapid elevation loss within one winter after a plant die-off. The elevation losses were attributed to microbial decomposition of the root-rhizome matrix. A white film from a sulfide-based chemoautotrophic bacteria, *Beggiatoa*, had been observed on the roots of the dying plants. The presence of H<sub>2</sub>S, indicated by the presence of the bacteria, suggests that marsh soils were already in poor condition to support a healthy plant community (see Section 2.4, Nutrient Regime). Such acute responses caution that the loss of marsh vegetation can have long-term, or irreversible, consequences under some conditions. However, it is not difficult to find previously vegetated panne or pond habitats at particular marshes where the underlying rooted substrate does not become seriously degraded or reduced in elevation over periods of several years.

## **2.2 Marsh Connectivity**

The vegetated portion of salt marshes are not isolated ecosystems; rather they are intimately tied to the adjacent estuarine system, largely via tidal creeks (French and Stoddart, 1992; Weinstein et al., 2000). The exchange of water, sediment, nutrients, and nekton are primary functions of tidal creeks in a healthy salt marsh (Hampel et al., 2003; Spurrier and Kjerfve, 1988; Ward, 1981; West and Zedler, 2000). Thus, the degree of connectivity, both within the marsh as well as to its neighboring estuary, is an important precursor for determining distinct parameters of a marsh system's health. Connectivity influences the hydrological regime and thus determines the amount of water delivered to the marsh each day (Kjerfve et al., 1991). The amount of water



delivered to the marsh ultimately affects the edaphic conditions for plants, as well as impacting the suitability of habitat for fish, birds, and other fauna (Reed et al., 1999).

### **2.2.1 Tidal creeks**

Tidal creeks are prominent features in most salt marsh systems. They serve as the primary conduits for material fluxes across the marsh surface (Baylissmith et al., 1979; Spurrier and Kjerfve, 1988; Weinstein et al., 2000). Understanding the evolution and behavior of tidal creeks is important because of the vital role that they play in marsh-estuary exchange. The morphology and hydrology of tidal creeks have been well studied (Ashley and Zeff, 1988; Fagherazzi and Furbish, 2001; Lawrence et al., 2004; Leopold et al., 1993; Novakowski et al., 2004; Zeff, 1999), yet a broadly accepted model of their evolution and behavior has not emerged. Most researchers note that marsh tidal creeks are more stable over time than their fluvial counterparts. This has been attributed variously to tidal creeks' bidirectional flow (Mitsch and Gosselink, 2000), plant stabilization (Garofalo, 1980), and hydraulic velocity geometry (Leopold et al., 1964). Numerous morphometric studies of tidal-creek networks have yielded several competing models of evolution. Weigert and Freeman (1990) described portions of marsh with low drainage density due to few creeks as being in a mature stage. Maturity is defined such that sediment has infilled small, low-order creeks to form a contiguous vegetated region. In this model, more nascent areas of marsh are suggested to have a high-density network of creeks. In contrast, Dame et al. (1992) discussed a model in which the most mature reaches of a marsh are those along the seaward edge where tidal creeks are large and stable, and connect to a progressively denser network of small, young, tidal creeks. Such discussions, although largely academic, are important because they establish a framework in which management and restoration decisions will be made (Coats et al., 1995). For instance, an important decision concerns the stage of maturity or youth that is appropriate for managed marshes. A driver behind this issue is the potential for inherent differences in health and stability between these stages of development.

It is important to note that some researchers use the terms maturity and youth to describe the various tidal networks within salt marshes (Pethick, 1980). These terms imply a relative age and unidirectional evolution of such systems, which is a misleading notion. In fact, distinct high- and low-density creek networks are more aptly considered as end-members of a marsh morphology continuum, across which any given marsh system might move freely as sediment supply and

tidal prism vary (Fagherazzi et al., 1999; Lawrence et al., 2004). This is especially relevant for marsh systems where changes in such parameters are large or frequent, such as river deltas and backbarrier lagoons. In particular for the lagoonal systems of Great South Bay, Long Island, episodic inlet breaches are likely to generate significant changes in tidal height, tidal prism, and sediment loading. These are the primary hydraulic factors influencing the morphology and behavior of tidal creeks (Myrick and Leopold, 1963). However, in deference to the at most decadal timescale associated with management efforts, tidal creeks can be considered relatively stable. This assumption allows emphasis to be placed on their initial geometry and hydraulics. Coats et al. (1995) suggest that appropriate geometries for constructed channels can be derived from natural examples. This should be considered with the caution that basic controls, such as tidal prism and sediment loading, need to be comparable between the natural and managed sites.

### **2.2.2 Connectivity with the estuary**

The connectivity via tidal creeks of a marsh with its adjacent estuary is the primary avenue for sediment and nutrient exchanges. In many salt marshes, particularly lagoonal systems that are not fed by longshore transport or direct fluvial inputs, sediment reaches the marsh surface via numerous substeps of subtidal transport and storage within nearshore mud bars and creek channels (Reed et al., 1999). If typical transport rates are considered for sediment in a tidal channel, the decrease in flow velocity with distance along the creeks requires that most sediment be stored in the creek on any given tidal cycle (Pillay et al., 1992). Thus, most sediment ultimately reaching the marsh surface has been stored in tidal bars and channels for periods of weeks to many months. Often, such stored sediments are only reworked onto the marsh in significant quantities during episodic wind or storm events (Leonard et al., 1995b; Reed, 1989). Thus, creek morphology and tidal hydrodynamics are key controls on the effective transport of sediment into a marsh system. Seminal studies of stream morphology show that creeks will evolve to evenly dissipate energy along its course, requiring that channels or thalwegs meander and that cross-sectional area decreases with discharge or tidal prism (Leopold and Maddock, 1953). These channel features of meandering and tapering have the effect of maintaining tidal velocities and a fully turbulent water column, thus evenly distributing energy, and, as a consequence, evenly distributing suspended sediment.

In the case of deep, non-tapering ditches, geometries and resulting flow dynamics are entirely inappropriate for sediment transport into a marsh system. Even if the total volume of water transported through a ditch had the potential energy to do so, this energy and any suspended sediment would be lost rapidly near the mouth of the ditch.

Irrigation and flushing with water has been shown to be strongly correlated with plant growth and biomass, suggesting that efficient water exchange across the marsh surface is a primary control on plant health (Hardisky et al., 1983; Padgett and Brown, 1999). In drained, or otherwise tidally restricted marshes, however, several ecological features are strongly affected by the loss of connectivity within and among the marsh and estuarine systems. When a marsh is drained, functions such as nutrient processing, sediment trapping, and nursery habitat are decreased or lost due to their dependence on tidal linkage between the marshland and neighboring body of water. In other situations, tidal restriction via dams or culverts associated with bridges and roads may have serious impacts on salt marsh functioning by freshening the salinity regime, with resultant changes in:

- plant community, often including *Phragmites* colonization
- nutrient cycling
- exchange with the estuary

(Anisfeld et al., 1999; Portnoy and Giblin, 1997; Raposa and Roman, 2001)

In terms of geochemical cycling, some of these restricted systems, often with ponded waters, may be more efficient at filtering upland-derived nutrients and pollutants due to the increased residence time of any runoff, and enhanced uptake of nutrients by reeds such as *Phragmites* and *Typha* (Findlay et al., 2002). However, others state that restored hydraulic exchange increases the frequency of marsh-surface inundation, which maintains the filtering capacity of the marsh while simultaneously being more beneficial to plant health and vigor (Lopez-Flores et al., 2003; Verity, 2002). Degraded hydrodynamics may increase residence time, but the system is not in contact with the adjacent estuary as frequently, thereby limiting its capacity for nutrient uptake. In a more open, natural system, the residence time of water on the marsh surface is reduced, but the frequency is enhanced, allowing regular flushing of soils and access of nekton into the marsh interior.

As the structure of salt marshes is largely organized by tides, returning tidal exchange and inundation to altered systems can be successful in reconnecting the wetland to the estuary, and, with time, return it to a self-maintaining tidal marsh (Warren et al., 2002). In general, improved hydrological functioning can be indicated by changes in tidal flooding, soil water depths, porewater salinities, and fish use (Burdick et al., 1997). The latter is often a target response for restoring wetlands' tidal exchange and overall connectivity between marsh and estuary. Conversely, changes in these indicators can reflect stress on a marsh, possibly leading to deterioration of the floral community that is most sensitive to edaphic conditions. However, one difficulty in monitoring such factors is that they have large natural ranges over shorter timescales of days to years, and thus require several years or more worth of data to recognize trends.

Another poorly quantified impact on marsh connectivity is the dredging of adjacent estuaries and waterways. Salt marshes on Long Island and other populated coasts are frequently bordered by channels dredged for marine navigation or construction. In terms of sediment exchange, such deep waterways tend to act as basins for sediment, by which suspended material is trapped in the deep dredged areas, and are subsequently unavailable for export to the marsh. This effect is a leading hypothesis for the extensive marsh losses occurring in Jamaica Bay (Jamaica Bay Blue Ribbon Panel, 2001). Typically, sediments reaching the marsh surface are derived from the wave-induced resuspension of nearby creek and estuarine deposits (Reed et al., 1999). Overdeepening of these source areas generally limits the effect of waves in both the amount and height of sediment resuspension. Slumping is another common impact of dredging along marsh edges, whereby oversteepened slopes become unstable and release blocks of marsh sediment to deep water.

Many juvenile fish feed along marsh edges and take refuge in shallow waters, thus requiring a healthy marsh ecosystem that supports creeks with distinct thalwegs and shoal reaches. These features provide sufficient habitat complexity for fauna of various size, life stage, and feeding strategy (Kneib, 1997a; Weinstein et al., 2000; West and Zedler, 2000).

### **2.2.3 Connectivity within the marsh**

Tidal creeks, pools and other open-water settings generally enhance within-marsh connectivity, in that these transitional edge habitats contribute to trophic exchange and provide habitat for crustaceans and transient and resident nekton at various life stages (Kneib, 1997b; Weinstein,

1979; Weinstein et al., 1980). Thus, wetlands with significant water edge, either as creeks or pools, are likely to support greater faunal abundance and biodiversity. However, such systems may be more susceptible to edge erosion and creek widening (Fagherazzi and Furbish, 2001); marshes with less edge habitat can suffer from limited hydrological exchange, and lack associated sediments and nutrients.

Within-marsh connectivity can involve issues associated with various mosquito management techniques, such as the traditional and widely used parallel grid ditching, and more recent concepts of Open Marsh Water Management (OMWM). OMWM was developed in the 1960s due to recognition of the negative impacts that ditching may cause to marshes, such as a loss of connectivity. It is also associated with the increasing recognition of health and environmental impacts from pesticide and larvicide use (OMWM) (Wolfe, 1996). OMWM often consists of creating small ponds and deep-water reservoirs that host predatory fish such as killifish (*Fundulus spp.*). The areas of open water created by OMWM are frequently well connected to the estuary, thus allowing the continuous movement of fish in and out of the system. Other purported benefits of such OMWM techniques are enhanced edge habitat associated with greater marsh-estuary connectivity, that is generally regarded as the major zone of use for marsh-utilizing macrofauna. In addition to traditional OMWM approaches, newer alternative techniques being used also create open-water areas that are *unconnected* with the estuary, but which remain permanently flooded and thus can support fish populations.

At several sites in New England and mid-Atlantic states, various OMWM strategies have been implemented with some success (e.g., Cowan et al., 1986). Even on Long Island, OMWM demonstration studies have conducted in parts of Seatuck National Wildlife Refuge (Lent et al., 1990) and the Long Island Complex Refuge (James-Pirri et al., 2002), and Fireplace Marsh. However, these efforts are either too new or not sufficiently monitored to assess overall successes and impacts. Currently, US Fish and Wildlife in conjunction with the Suffolk County Vector Control and Wetlands Management Long-Term Plan have proposed a major OMWM demonstration study for Wertheim National Wildlife Refuge (D. Tonjes, Cashin Associates, personal communication, 2004).

## 2.3 Ecological Communities

Fundamental to understanding the health of a marsh is recognizing its ability to support a variety of living things. If a marsh is healthy, then the plant and animal communities that define it should be flourishing. This section describes typical plant and nekton assemblages found in a Long Island salt marsh and will further develop the idea of how these communities can be used to assess salt-marsh health.

### 2.3.1 *Spartina* grasses

The defining plant species in any salt marsh in the northeast US are *S. alterniflora* and *S. patens*. *S. alterniflora* is found at the seaward edge of the salt marsh, and is replaced by *S. patens* with increasing elevation of the marsh. Studies by Bertness and colleagues (1991a; 1991b; 1987; 2002) helped elucidate the primary causes of zonation in the *Spartina* grasses. Unlike *S. patens*, *S. alterniflora* is able to oxygenate its root zone and rhizosphere due to the presence of aerenchyma tissue (Mitsch and Gosselink, 2000). This specialized tissue permits *S. alterniflora* to colonize more frequently inundated, and thus more chemically reducing, habitats. At the same time, *S. alterniflora* is excluded from less-frequently flooded high marsh due to competitive displacement by *S. patens*. Thus, physical conditions determine the lower zonation and competition decides the upper zonation in a salt marsh (Bertness and Shumway, 1993). Bertness (1988; 1991a) also noted that the limited success of *S. alterniflora* in the high marsh was partly a result of the heavy peat density and low nutrient levels present in this landward part of the marsh. In addition, high marsh soils contain half the amount of pore water ammonium, a primary nitrogen source, as low marsh soils do. In a 1987 study, Bertness and Ellison reconfirmed that the vegetation zones of a salt marsh correspond with differences in flooding frequency across the marsh. That is, in the lower marsh, which is flooded by all but the weakest neap tides, *S. alterniflora* dominates; at the upper most part of the marsh, flooded by only high spring tides, *Juncus gerardi* dominates; and, in between these two extreme zones, one finds *S. patens*.

It is important to note that salt marshes typically have large areas in which environmental conditions are at or near the physiological or stress limit of *S. alterniflora*, most notably along marsh edges and poorly drained interior settings. Therefore, it is expected that even a healthy marsh will have areas in which plants are at a less than optimal stem density or have a less than optimal growth rate. In this case, the salient characteristic of declining marsh health would be a

notable increase in the ratio of low stem density area to that of more robust stem densities. A characteristic range between low and normal stem densities for *S. alterniflora* is 100-150/m<sup>2</sup>. Furthermore, it is the ratio of low to normal density areas that is important, because sometimes vegetative loss in one place is balanced by gains in other portions of a marsh system. Again, any group of typical marshes is likely to show a large variability in the presence, extent, and distribution of vegetation and habitats, and so an assessment of health requires measurements made over time to recognize steady state, positive, or negative trends.

### **2.3.2 *Phragmites australis***

*Phragmites australis* (*Phragmites*) is found in freshwater and mesohaline water, defined as 0 to 18 parts per thousand (ppt) salinity, more often than in polyhaline water, defined as 18 to 35 ppt. Therefore, growth of *Phragmites*, in general, is stunted when flooded by sea water (Chambers et al., 1998). In areas that were once diked or drained, a decline in *Phragmites* cover and its average height were seen following restored tidal conditions after only one growing season (Roman et al., 2002). *Phragmites* is often found in large monocultures, especially where the hydrology of system has been altered (Orson, 1999). These findings suggest that restoring tidal wetlands to a proper salinity and hydroperiod may be an essential step leading to the control of *Phragmites*.

An increase in *Phragmites* during the past 100 years has led to the perception that it is an introduced species. However, as Orson (1999) pointed out, *Phragmites* has been a member of the New England plant community for thousands of years; it is only because of recent environmental changes and anthropogenic impacts that it is now found in abundance. Bertness et al. (2002) attribute invasions of *Phragmites* to nitrogen eutrophication and suggest that over 90 percent of *Phragmites* expansion in Narragansett Bay, Rhode Island, is due to shoreline development. Salt marshes without developed shorelines have less than 15 percent of their terrestrial borders dominated by *Phragmites*. It has been proposed that 50 percent developed border may serve as an indicator that a majority of the terrestrial border of the marsh is or will be invaded by *Phragmites*. However, *Phragmites* has also invaded many naturally bordered marshes, suggesting that multiple influences likely play a role in its distribution.

Such environmental factors almost certainly play a role in the continued replacement of *Spartina*

and mixed-flora marshes by monotypic stands of *Phragmites*. However, recent genetic research strongly supports the hypothesis that the recent, highly invasive character of *Phragmites* is a function of an introduced genotype. Saltonstall (2003) showed that up to 11 native haplotypes of *Phragmites* historically found along the Atlantic coast have been displaced by a unique, and introduced, Eurasian genotype. This same genotype is also recognized as competitively displacing other native haplotypes in the central and western US.

However, it should be noted that as more is learned about the ecosystem structure in *Phragmites*-dominated marshes and some potential benefits in terms of water quality are recognized, the prevailing assessment of its very low to negative ecological value has softened (Ludwig et al., 2003).

### **2.3.3 Plant response to environmental change**

Several studies suggest the use of plants as indicators of sea-level rise and marsh inundation; and, conversely, these forcings are suggested as indicators of the health of marsh floral communities. From their study in Nauset Bay, Massachusetts, Roman et al. (1997) noted that, over the past 40 years, the ecologically tolerant plant *Distichlis spicata* appeared in areas that were once occupied by *S. patens*, suggesting this site may be wetter as a result of marsh accretion not keeping pace with sea-level rise. However, interannual and longer-term variability in rainfall can also affect wetness and the relative abundance of *D. spicata* and *S. patens* within mixed plant communities (Hansen et al., 1976). In this instance, the replacement of *S. patens* by *D. spicata* does not necessarily implicate the change in salinity associated with sea-level rise. In other words, replacement of one high-marsh species by another may occur for competitive reasons not connected with a decline in marsh health. In contrast, replacement of a high-marsh species with low-marsh *S. alterniflora* may be a truer indicator of sea-level impacts, because *S. alterniflora* can only compete with high-marsh plants under the condition of prolonged inundation (Bertness, 1991a; 1991b). For example, research by Orson et al. (1998) and Warren and Niering (1993) suggest that *J. gerardi* is declining in certain New England high marshes and being replaced by stunted *S. alterniflora* under the impact of rising sea level. This is based on the assumption that salinity must be high enough to preclude succession to *D. spicata*, which is not especially flood-tolerant in the presence of salt.



Pursuant to these complexities of plant response, Warren and Niering (1993) suggest a scheme for understanding how plants can be used as indicators of sea-level rise and marsh inundation. An imbalance between marsh accretion and sea-level rise leads to increased hydroperiod over the high marsh, resulting in:

- a reduction in soil redox potential
- greater sulfide levels
- altered porewater salinities

each of which affects the health of marsh angiosperms. Often, the net effect is plant species replacement and plant community change. However, it is important to stress that such changes do not imply decreasing marsh health in and of themselves, but need to be considered within the period, frequency, and magnitude of the environmental forcings. For example, measurable, but non-permanent, changes in plant species composition are typical of heterogeneous marshes found in microtidal and/or brackish settings. Marshes in these settings typically have a lower marsh-surface gradient, in which a hummocky microtopography creates local subenvironments that, during periods of high precipitation, may favor brackish species like *Scirpus spp.* More halophytic species will occur during drier periods when evaporation raises surface salinities. Unfortunately, distinguishing the difference between long-term changes in plant health and shorter-term environmental variability is not straightforward. The amplitude of sub-decadal cycling of both climate and sea level are an order of magnitude greater than that of long-term sea-level trends. For example, tide-gauge records from The Battery, New York City, show patterns of sub-decadal changes on the order of five to 10 cm sea-level change, which in any given year overwhelms the long-term rate of rise of less than 0.5 cm/yr. Marsh plants respond to such shorter-term variability, and so trends data should ideally be collected across several cycles of variability to assess the significance of underlying sea-level rise. This is further complicated by interannual to decadal-scale variations in climate, which strongly impact marsh-plant communities via:

- summer rainfall control on inundation and salinity
- wind forcing of floodwater and sedimentation
- temperature control on evaporation, evapotranspiration, and porewater salinity,

- influence of storms on erosion and sedimentation
- winter ice effects.

For these reasons, changes in plant community structure determined from short-term datasets or discreet observations should only be attributed to sea-level rise or other factors with great care. This is especially true in low-gradient, brackish or microtidal systems that have complex marsh-surface topography and heterogeneous habitat distribution. In contrast, marsh plants in full-salinity or mesotidal settings are more strongly zoned with limited intermixing of species. It is suggested that in such settings, changes in vegetation are less susceptible to short-term variability because the environment is homogenous and well mixed under strong marine influence. Thus, such systems are more likely to track longer-term trends in climate or sea-level change or environmental degradation.

#### **2.3.4 Nekton**

Salt marshes are important habitats for both resident and transient nekton species. Kneib (1997b) reviewed the importance of tidal marshes to estuarine nekton. These nekton are typically less than 150 mm in length, and are comprised mostly of finfish, although crustaceans can be found in abundance. The frequency and duration of tidal flooding, along with salinity, ultimately control potential habitat of estuarine nekton (Kneib, 1997a). A combination of creeks, ditches, pannes, and aquatic microhabitats become important homes for any number of nekton. In particular, several studies emphasize that aquatic microhabitats that can be only a few millimeters deep and centimeters across are often overlooked despite their importance as intertidal low-tide refuge for larvae and juveniles of marsh resident nekton (Baltz et al., 1998; Baltz et al., 1993; Kneib, 1997b; McIvor and Odum, 1988).

Among finfish, the killifish group (*Cyprinodontidae*) is the dominant family in most intertidal, saline marsh habitats (Kneib and Stiven, 1978; Valiela et al., 1977). On Long Island, specifically, mummichog (*Fundulus heteroclitus*) and banded killifish (*F. majalis*) are the characteristic species of saltier and fresher intertidal marsh habitats, respectively. Physiologically tolerant to rapid salinity changes and low dissolved oxygen concentrations, killifish have a little competition and can be expected in significant numbers in most creek, pond, and even small pool habitats (Halpin, 1997; Smith and Able, 1994). Therefore, among the wide range of typical marsh settings, killifish can be expected in any of these and thus may serve as a

first-order indicator of marsh ecological health. Where absent, the implication is that a marsh health is reduced, most likely as a function of poor water quality or restricted tidal-exchange or estuary-connectivity, which could arise from a great number of causes.

Another characteristic family of finfish in coastal marsh systems are the sticklebacks (*Gasterosteidae*), which are often voracious predators of mosquito larvae (Worgan and Fitzgerald, 1981). In contrast with omnivorous killifish, sticklebacks are purely carnivorous and generally less tolerant of extreme salinity, temperature, and oxygen variations. Therefore, their presence across a range of marshes cannot be expected, and their absence does not necessarily reflect poor marsh health. However, marsh systems that support a stable stickleback population clearly indicate sufficient food prey such as insects and juvenile fish, and healthy water circulation patterns (Poulin and Fitzgerald, 1989; Walsh and Fitzgerald, 1984).

The killifish and sticklebacks are largely resident species that dominate vegetated portions of the salt marsh, as well as high marsh pools (Halpin, 1997; Komarow et al., 1999; Smith and Able, 1994). In subtidal habitats, larger, mainly transient, nekton become prevalent in many marsh systems (Rountree and Able, 1997; Weinstein et al., 2000; West and Zedler, 2000). Common predatory examples include young-of-the-year bluefish (*Pomatomus saltatrix*), striped bass (*Morone saxatilis*), and summer flounder (*Paralichthys dentatus*), whereas common omnivorous species include the silverside (*Menidia menidia*) and striped mullet (*Mugil cephalus*) (Able et al., 2003; Cadigan and Fell, 1985; Rountree and Able, 1992; Tupper and Able, 2000). Because of their transitory use of salt-marsh and tidal-creek habitats, the absence of such species is not indicative of poor ecological health. However, their regular presence is suggestive of a relatively healthy ecosystem capable of supporting larger grazers and upper-trophic-level species (Weinstein et al., 2000).

## **2.4 Nutrient Regime**

Nitrogen levels are often used as an indicator of marsh health. Nitrogen eutrophication of a marsh can be generated by shoreline development and population pressure. The result of eutrophication tends to be a shift in the competitive balance among marsh plants, by reducing nutrient competition (Bertness et al., 2002). It is predicted that increased nitrogen levels will allow low marsh plants, which are traditionally restricted by interspecific competition, to invade the high marsh, where they no longer will be poor competitors. In Narragansett Bay, Rhode

Island, Bertness and colleagues (2002) tested the hypothesis that increased nitrogen levels would favor invasion of *S. alterniflora*, typically restricted to the low marsh, into the high marsh, resulting in displacement of *S. patens*. The data showed that localized near-shore human activities were responsible for local increases in marsh nitrogen, therefore resulting in the expansion of *S. alterniflora* from low marsh to high marsh. This is an important finding, because observed shifts from *S. patens* to *S. alterniflora* are generally taken to indicate increased marsh inundation due to sea-level rise.

Under ambient nutrient loads, a typical New England salt marsh shows the pattern of *J. gerardi* and *S. patens* in the high marsh, with *D. spicata* and *S. alterniflora* in the low marsh. However under elevated nutrient conditions, a reverse hierarchy may occur (Levine et al., 1998). This change in hierarchy suggests the dramatic possibility that highly eutrophied conditions would allow *S. alterniflora* to be both the best competitor and most tolerant to physical stress, resulting in a monoculture of this single plant species. Whether such a response occurs in nature is not known, but intermediate responses to altered nutrient regimes may play a role in historical observations of *S. patens* to *S. alterniflora* transitions. Overall, the interplay of hydrology, nutrient load, and plant community structure is far from determined, based on existing research.

Ultimately, however, it is difficult to place a health assessment on such changes as they

- 1) may be a function of multiple factors, and,
- 2) the resulting community remains within the ranges of normal, healthy salt marsh.

However, eutrophication, or, more specifically, nitrogen loading, becomes a clear concern when it favors the establishment and propagation of invasive *Phragmites*. *Phragmites* is well known for replacing *Spartina* communities under increased nitrogen loading (Bertness et al., 2002; Chambers, 1997), most typically in somewhat reduced salinity environments (Chambers et al., 1998, 2003). However, *Phragmites* produces sufficient physical structure such that it modifies its local environment in favor of further propagation. (Bart and Hartman, 2003; Lathrop et al., 2003).

Whereas nitrogen is a limiting nutrient for marsh plants and its variability is a strong control on growth rates and competitive displacement, sulfur in its reduced state as sulfide is a major stressor of marsh plants (Teal and Teal, 1969; Teal and Kanwishe, 1966). Sea water contains

relatively high concentrations of sulfate, and serves as the primary source of sulfur to the marsh surface during tidal inundation. Thus, fluxes of sulfur to the marsh are highest in the frequently flooded low marsh, which is consequently where plants are most likely to be subject to sulfide toxicity. The basic geochemical transformation is that sulfate delivered to marsh is subsequently reduced to sulfide by sulfur-reducing bacteria as they respire organic matter in the marsh soils (Howarth and Teal, 1979; Shen, 1979). There are several possible fates for the resulting sulfide anion (King et al., 1985), the dominant pools for which are complexation with reduced iron to form pyrite or complexation with hydrogen ions to form hydrogen sulfide (Giblin et al., 1983; Howarth et al., 1983; Peterson et al., 1983). The latter is the form most toxic to plants (DeLaune et al., 1983b; King et al., 1982). Where *S. alterniflora* releases oxygen to its root zone, this oxidative layer buffers the plant from the reduced sulfide. However, the plant expends energy in this process, and if the redox demand of the soils is too high then growth, biomass, and health of the plant may decline or lead to death (Teal and Teal, 1969).

DeLaune and others (1983a; 1992) showed that marshes frequently flooded by marine waters and so with high sulfate input required a certain amount of inorganic sediment as an iron source to complex with sulfide produced in the soils. Hence, this is one reason that fresh water systems, and even high salt marshes, can have accretion dominated by organic production, whereas low salt marsh cannot keep pace with sea-level rise by organic production alone, as some mineral matter is required for healthy cycling of marine-sourced sulfur. In the case of drowning marshes, enhanced sulfide production under more frequent inundation is ultimately the reason for loss of vegetation, not the actual inundation process. Increased inundation, to a point, benefits low marsh plants by increasing nitrogen fluxes (c.f. Chalmers, 1979; Dai and Wiegert, 1997; Teal et al., 1979).

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