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

Task 3 Literature Review
Book 9 Part 3: Natural & Managed Salt Marshes

Prepared for:

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January 2005

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

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List of Acronyms and Abbreviations

NPS	National Park Service
OMWM	Open Marsh Water Management
USFWS	US Fish and Wildlife Service
YOY	young of the year

Executive Summary

Salt marshes are found at the interface of land and sea. Although now highly valued by society, they have not always been so well regarded. For example, over half of the original salt marsh acreage on Long Island was lost prior to the establishment of current protective regulations, to filling and dredging as means of creating uses that were considered to be of greater value than were provided by the marshes themselves.

Salt marshes are very productive, yet relatively simple ecosystems, composed of relatively few species that appear to be well-ordered geographically. The most obvious attribute of salt marsh land is usually considered to be the clear zonation of plants. Low marsh, which is inundated on every tide, is typically covered by one grass, *Spartina alterniflora*. High marsh, the area that is flooded by tides each month, although not daily, is dominated by another grass, *Spartina patens*. A few salt-tolerant plant species can be found between the high marsh zone and the beginning of more typical uplands vegetation. Because of the relationship between elevation and the plant zonation, a visual contour is often created across the marsh that is instantaneously recognizable to even untutored eyes. These characteristic patterns are repeated at marsh after marsh, across much of Long Island and most of the northeast US. However, this simple pattern can be much more complex in certain situations, where the distribution of factors such as salinity, nutrients, and sediments become more important than tidal elevation. That these factors can override the dominant impact of tides shows there is an underlying complexity beneath the apparent simplicity of these systems.

Salt marshes support life beyond the obvious vascular plant growth. There are two important microscopic communities. One is the plankton and algae growing on the marsh surface and in the marsh waterways, turning sunlight into energy. The second is the decomposer community. This community, living in the peat and mud of the marsh, brings buried, indigestible detritus back into the food web.

The salt marsh is a harsh environment. There is strong sunlight in summer, ice in winter, and constant battering by water. There is salt everywhere, requiring special adaptations by plants and animals to cope with osmotic pressures. Much of the subsurface environment lacks oxygen, and, so, relies on metabolic processes that use pathways other than those used by aerobic organisms.

The production from photosynthesis and detrital decay is consumed by fish, invertebrates including crabs and mussels, and birds. The marsh-resident fish and invertebrates serve as prey for juvenile and adult estuarine fish, including some of the charismatic species of sport and commercial importance. Long Island salt marshes have been called the most important part of the North Atlantic Flyway, the pathway used by migratory birds, especially waterfowl. The marshes are seen as important controllers of local and areal water quality through physical and chemical transformations of potential contaminants. They also serve as buffers of land and sea, minimizing the impacts of one on the other.

However, it is also clear that important processes for marshes vary considerably from marsh to marsh, and over time and space within each marsh. While marshes have a consistent appearance, and are relatively stable environments in terms of features such as creeks and the distribution of vegetation types that can persist for hundreds of years, they are also very dynamic and variable environments with great dissimilarities. The differences between marshes are largely controlled by geography, with tidal regimes and the overall relationship to the accompanying estuarine system being of paramount importance in driving the other important determinants of marsh features, such as nutrient availability, sediment supplies, and salinity gradients. Similarly, annual variations in weather and the position within the marsh with respect to tidal impacts control the ways that variations occur within a particular marsh across time and space.

Salt marshes also include mosquitoes as part of their ecology. In the early part of the 20th Century, ditching salt marshes was seen as a means of controlling mosquito production. Ditches were believed to disrupt the hydrological processes that resulted in optimal mosquito breeding conditions, generally by draining water from the surface of the marsh, and by allowing more access to the interior of the marsh by insectivorous fish. By 1939, 90 percent of the salt marshes of the northeast US that still remained after other alterations were ditched for mosquito control. Approximately 95 percent of Suffolk County's salt marshes were ditched during this time period.

The literature is not clear as to whether ditching is effective over the long-term for mosquito control, and whether it constitutes a major alteration of salt marsh ecology. It is probable that these issues are resolved by marsh-specific factors, so that, for example, ditching has been effective as a means of mosquito control in the long-term for some salt marshes, yet it has resulted in major impacts for some marshes. Ditching impacts on the environment, per se, have not been a major focus of salt marsh research, however. This is partly because ditching and other water management tools used for mosquito control are usually addressed more as engineering problems, where project optimization is the goal, rather than as scientific questions where careful determinations of impacts are more likely to be measured. This may also be because ditching was essentially completed prior to the initiation of widespread salt marsh research, and therefore ditches are simply another physical feature of the existing marsh environment to most researchers.

It is clear that ditching is a rather blunt tool for mosquito control purposes. Many ditches were put in areas where mosquitoes do not breed. In addition, ditching may cause changes in marsh hydrology, vegetation patterns, and wildlife usage of the salt marsh, and may change how the marsh physically and chemically alters water that comes in contact with it. These concerns led marsh experts, even as the initial ditches were being constructed, to alter aspects of the standard ditching design to mitigate the potential impacts. The suite of changes came to be known as Open Marsh Water Management (OMWM). OMWM was originally conceived as a means of addressing mosquito breeding in salt marshes, but has also become a means of conducting general salt marsh ecological restoration, with or without respect to mosquito management.

There are four basic OMWM design types, three of which have widespread use in the northeast US. All three share the goal of improving insectivorous fish habitat on the marsh, and in improving access for the fish to mosquito breeding points. All three establish ponds in the marsh as fish refuges. Additionally, they exploit various aspects of the tidal forces affecting the marsh to achieve certain ends.

The three OMWM types are called open systems, semi-open (sill ditch) systems, and closed (full ditch blockage) systems. Open systems enhance tidal circulation through the marsh, often by maintaining and expanding existing ditches. Sill ditch systems retain some water in existing ditches by partially damming them, potentially increasing the water table height and creating additional fish refuges, while allowing for some daily tidal flushing. Full ditch blockages create the greatest degree of fish refuges by isolating all of the refuges from the daily tidal circulation; this OMWM technique is intended to most fully restore water tables that may have drained due to ditch construction. Most experts believe that each OMWM type has advantages in particular settings, and that, generally, an appropriately chosen OMWM can mitigate potential impacts to a salt marsh that may have resulted from the installation of ditches, without sacrificing larval control of mosquito breeding.

OMWM has, generally, been applied across the northeast US. Extensive OMWM programs are in place across Connecticut and New Jersey, in particular. These jurisdictions have found that OMWM is very successful at controlling mosquito breeding, and, especially in Connecticut, have achieved notable ecological restoration goals, such as improvements in waterfowl habitats and control of invasive *Phragmites*. Some ditch blocking projects have been undertaken in Long Island salt marshes, generally as part of overall marsh restoration efforts. However, OMWM as a mosquito control tool has not yet been incorporated into Long Island practices. Regulators at the New York State Department of Environmental Conservation, which has overall jurisdiction for salt marsh management in New York, have been slow to permit OMWM demonstration projects because of concerns that the project plans have not accounted for Long Island- and site-specific issues.

1. Introduction

This section of the literature search addresses the ecology and food chains of natural salt marshes, including the impacts and effects of traditional water management (grid or parallel ditching) and Open Marsh Water Management (OMWM). This discussion is limited by a lack of consensus in the scientific literature, especially concerning the overall ecological status of salt marshes and the implications that has for understanding marsh processes, and the impacts of traditional water management.

The conflict may be partially caused by the nature of salt marshes. Most analysts describe individual salt marshes as being essentially similar to one another, based on the obvious, shared, repeated pattern of zonation of marsh vegetation. Generally, since all wetlands share this feature, it is tempting to assume they share many other basic qualities (see, for example, Chapman, 1974; Teal and Teal, 1969; Nixon, 1982; Teal, 1986).

Careful research within and across sites shows the assumption of overall marsh process similarity to be an oversimplification for many of the important salt marsh attributes. Dale and Hulsman (1990) stressed that marshes are heterogeneous over distance and time, and discussed how this was a major constraint on the ability to establish replicates and controls. Among others, this general view is supported by Peck et al. (1994). Pomeroy and Imberger (1981) agreed, suggesting that marshes vary in terms of being sources or sinks for important features (i.e., nutrients, organic carbon, trace metals). They noted that due to the number of important variables affecting estuarine systems,

> "the resulting biological structure and function may vary substantially … although these variations may not be apparent on cursory examination."

Even within one marsh, for one particular attribute, tremendous variations have been found. For example, Odum (1988) found that productivity in salt marshes can vary 200 to 300 percent interannually, and 300 to 500 percent along transects from creek edges to the center of marshes, and so inquired as to the sense of generating "marsh productivity" data. Adding to this confusion is the finding of Linthurst and Reimold (1978) that there is an order of magnitude difference in results for productivity of marsh plants based on the methods used to measure this attribute. Regarding the general lines of variations in these systems, Walker (1973) denounced "stability" as a concept appropriate for wetlands management, finding they are prone to great fluctuations in measured conditions. Wiegert (1980) determined that modeling of salt marshes is a useful and productive exercise, but found that the work was challenged by many confounding factors, including temporal and spatial heterogeneity.

It is clear that many important generalities ascribed to salt marshes as a class may not apply to particular systems or places. The reader must understand that many of the statements made in this report contain an unstated qualifier because of the underlying heterogeneity of the systems. Where possible, the report will include discussions of concepts and results that seem to indicate disagreement or a lack of scientific consensus.

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2. Salt Marsh Ecology and Food Chains

Teal (1962), in a seminal study of the Sapelo Island marsh in Georgia, found that it produced more organic matter than it consumed. He concluded that the excess production was removed by tides to the surrounding estuary, where it supported a high level of secondary production. In retrospect, there were flaws in the study and its conclusions (see Nixon, 1980), but the idea that salt marshes have an ecological and economic impact beyond their borders has lead to a myriad of laws aimed at protecting wetland ecosystems. These laws were fostered by the continuing destruction of wetlands, which accelerated with the growth of suburban development following World War II into the 1970s (Porter, 1990). In Suffolk County, specifically, 47 percent of tidal wetlands were lost to human disturbance between 1954-1971, with degradation of much of the remaining marshland (Kavenaugh, 1980).

The salt marshes of the US east coast are highly productive, yet harsh environments. The diurnal tides are the primary influence on the development and function of the intertidal community in the marsh. The influx of saline waters produces a high osmotic gradient for plants to cope with. The upper intertidal zone is free from water during part of each day. When evapo-transpiration is high enough, interstitial water is removed from the soil at such a rate that soil is salinity may be higher than the salinity of the nearby waters (Pomeroy and Imberger, 1981).

Plants attempting to survive in these zones encounter a physiological perception of a scarcity of water. In this respect, the community has been compared to terrestrial salt deserts (Chapman, 1960; Weigert and Freeman, 1990).

Marshes are documented as having low species diversity (Weigert et al., 1981), perhaps a product of the stressful environment and relative to lack of niches resulting from the structural and productive dominance of *Spartina spp.* (Montague et al., 1981). The environmental extremes allow the limited number of adapted organisms to be relatively free from competing species and enemies. This lack of competition and low vegetation species diversity allow adapted organisms to occupy broader niches and become more abundant than would otherwise be possible (Teal, 1962; Ambrecht et al., 2004).

MacArthur (1965) theorized that community stability is increased whenever consumers in a low species diversity habitat have generalized diets. Omnivores should provide population stability in a low diversity habitat, because dominant species are not impacted by periodic oscillations in any one resource's availability. Since the major groups of salt marsh consumers are dominated by omnivores, the entire community is relatively stable in regard to shifts of resource availability (Kreeger and Newell, 2000).

Long Island salt marshes are on the southern border of what is known as the New England type of marsh (Redfield, 1965). They are characterized as being small, built on the glaciated coastal plain with marine sediment and marsh peat, with little transport of sediment from the uplands (Mitsch and Gosselink, 1993). Salt marshes of this type constitute less than 2 percent of the marsh area along the US Atlantic Coast (Reimold, 1977). Southern marshes are generally much larger, as a consequence of large supplies of mineral sediments provided by rivers that help to build the marshes outward (Frey and Basan, 1985).

The New England marsh typically contains three vegetative zones:

- a *Spartina alterniflora* low marsh
- a high marsh dominated by *S. patens*, with *Distichlis spicata*
- an upper border of *Juncus gerardi* with shrubby species at the territorial edge

(Nixon, 1982; Teal, 1986)

In contrast, southern marshes are dominated by *S. alterniflora*, with a stunted form of the grass covering the majority of the high marsh. This is the case as far south as Florida, where mangrove swamps gradually replace *Spartina spp.* marshes (Wiegert and Freeman, 1990).

2.1 Production

Research by Teal (1962) and Odum (1971) on vascular marsh plants led to the theory that salt marshes are among the most productive natural systems on Earth. Productivity in salt marshes varies greatly with latitude, with the highest values occurring in the south with longer growing seasons and higher solar input. There is approximately a threefold variation in productivity over the latitudes of the eastern US. There is a similar variation in productivity within any one marsh (Odum, 1988; Teal, 1986).

Teal (1962) estimated gross productivity in a Georgia marsh at 6.1 percent of incident light energy, a high value compared to the 0.1 percent to 3.0 percent reported for typical freshwater and marine areas (Odum, 1959). Teal found net productivity to be 1.4 percent. Nixon and Oviatt (1973) in a Rhode Island study found the net production for all grasses on the emergent marsh to be 0.24 percent of incident light. They theorized that this lower value could be attributed to the harsher climate and shorter growing season in the north.

A high proportion of grass production is metabolized by the plants themselves. Plants inundated by salt water, as all plants are on the salt marsh, grow in an osmotically stressful situation, having to obtain carbon dioxide (CO_2) without losing too much water vapor through transpiration. An increase in respiration is necessary for the plant to maintain the higher osmotic gradient, lowering production (Chapman, 1960).

Nixon (1982) notes that measuring production by harvesting the grass is an underestimate of the total energy (measured as carbon [C]) fixed by the plants. Missing will be the growth lost to feeding, leaf fall, seed dispersal, and any organic exudates. Attempts are made by researchers to account for these losses. In comparing commonly used techniques, Nixon concluded that the choice of estimation methodology has a large influence on results (citing Linthurst and Reimold, 1978).

2.1.1 Algae

There are three major groups of primary producers in the salt marsh; the most visible, and usually considered to be the most productive in terms of total fixed carbon (C), is the rooted plant community. Algae, present on the marsh surface (microphytobenthos) and on the stems of the macrophytes (epiphytes) comprise a second set of producers. The free-floating phytoplankton of the tidal waters within the marsh is the third group. The tidal flux connects waters in the marsh, and its associated phytoplankton, with the benthic algae, depositing the plankton up onto the marsh during ebbing tides, and suspending a portion of the algae during flooding tides. Pomeroy et al. (1981) describe the algae habitat as one

> "between a dark nutrient-rich anaerobic sediment and either an illuminated, aerobic, comparatively nutrient-poor water column, or at ebb tide, the atmosphere."

They found that approximately 75 percent of algal production occurs during ebb tide, with bare creek banks being the most productive areas. Little production occurs under the dense plant cover found in the high marsh (Blum, 1968).

Algal mats often cover unvegetated marsh surfaces in New England (Teal, 1986). The algae in this class include macroalgae, such as *Ascoplyllum rodosum* (knotted wrack) and *Fucus vesiculosus* (rock weed), which grow on the estuarine edge of the *S. alterniflora* zone. *Enteromorpha* (hollow green weeds) and *Ulva* (sea lettuce) may be dominant in early summer. Williams (1962) found pennate diatoms to be the most important microalgae group in a Georgia marsh, comprising 75 to 93 percent of the total algal biomass. This was also found to be the case in Barnstable, Massachusetts (Blum, 1968), and in Rhode Island for most of the year (Nixon and Oviatt, 1973). Filamentous cyanobacteria make up most of the remainder of benthic primary producers (Pomeroy et al., 1981), with green algae present in algae mats when light levels are high (Sullivan and Currin, 2000).

These algae have a high turnover rate, compared to the macrophytes, and respond more rapidly to changing environmental factors that influence production, such as light, pH, salinity, and nutrients. Microalgae are readily eaten by benthic and suspension feeding animals. They are more nutritious and digestible then *Spartina* detritus (Kreeger and Newell, 2000). When algae-detritus feeders utilize algae, there is negligible lag between production and primary consumption, unlike the consumers of *Spartina* detritus and its associated microfauna (Teal, 1962). Biomass may be low relative to the vascular plant community, but this may be due to high grazing pressure from primary consumers, making the microphytobenthos the "secret garden" (MacIntyre et al., 1996; Miller et al., 1996).

Phytoplankton are not as important in the marsh ecosystem as algae, but because of their high nutritive value, are important resources for those consumers that can access them. Most phytoplankton in marshes are diatoms or dinoflagellates, with cell diameters so small (2 to 5 millimeters [mm]) that only suspensions feeders can efficiently utilize them (Kreeger and Newell, 2000). Phytoplankton production occurs primarily in the adjacent estuary, but enters into the marsh ecosystem with the tides. Pomeroy et al (1981) found phytoplankton productivity in Sapelo Island marsh to be approximately 12 percent that of vascular plants.

Weigert et al. (1981) estimated that 80 percent of the primary production in the salt marsh is provided by rooted plants, with a 10 percent contribution by phytoplankton and 10 percent by benthic algae. Over half of the production of *Spartina* results in roots and rhizomes, which do not enter directly into the aboveground food web. If only aboveground production is considered, algal production values are reported to be 25 to 36 percent of the vascular plant production (Nixon, 1982; Weigert et al, 1981). Rates may vary greatly among marsh systems and seasons. Other estimates range from less than 10 percent of vascular plant production (Sullivan and Mantcreiff, 1988) to more than 100 percent (Zedler, 1980, in arid California settings).

TABLE 1. Conceptual framework for comparing relative food value of some of the main resources available to dominate primary consumers residing in a typical salt marsh. Approximate relative food vales are estimated by integrating the relative availability (i.e., resource abundance), consumer access (i.e., how may major consumers can ingest it), nutritional value (i.e., energy and nutrient content), and digestibility of each resource.

Resource		**Food Value Characteristics**				
Class	**Type**	**Availability**	**Accessibility**	**Nutritional Value**	**Digestibility**	**Relative Food Value**
Primary Producers	Vascular Plants	High	High	Low	Low	Moderately Low
	Surface Associated Algae	Low	Low[1]	High	High	Low
	Phytoplankton	Low	Low[1]	High	High	Low
Detritus Complex	Dead Vascular Plants	High	High	Low	Low	Moderately Low
	Surface-Associated Bacteria	Moderate	High	Moderate	Moderate	Moderate
	Suspended Free Bacteria	Low	Low[1]	Moderate	Moderate	Low
	Heterotrophic Protiststs	Moderately Low	Low[1]	Moderately High	High	Low[1]
	Fungi	Moderate	Moderately High	Unknown	Unknown	Moderately Low

[1]with the exception of suspension-feeding consumers, which could derive more from this resource.

2.1.2 Plants

The plant composition of a salt marsh is thought to be the result of a two-part process. Competitively superior plants dominate in physically mild habitats, relegating competitively inferior plants to physically harsh habitat areas. In salt marshes, the inferior competitor is *S. alterniflora*, which has been shown to thrive in the relatively physically benign upper marsh if competitors are removed (Bertness and Leonard, 1997). Thus, *S. alterniflora* is pushed to the physically stressful low marsh. The competitive dominants such as *S. patens* that inhabit the high marsh are unable to colonize the periodically flooded low marsh, as they cannot tolerate the physical stress. Bertness and Pennings (2000) believe that marsh plant zonation is influenced by climate. In northern, colder areas, the zonal limits on plant growth are set up by the tolerance of the plants to flooding. In southern sites, with high evaporation rates, especially at middle elevations, zonal limits are set by plant tolerance to increased salinities.

Characteristic of this vertical zonation is the vegetation pattern in the New England marsh, where the woody shrub *Iva frutescens* dominates the upper border and black rush (*Juncus gerardi*) is found at lower elevations along the upland fringe. Salt marsh hay (*S. patens*) is the characteristic plant of the irregularly flooded high marsh. *S. alterniflora* dominates the tidally-flooded low marsh, inundated usually twice each day (Niering and Warren, 1980).

The cause of salt marsh zonation has been extensively discussed. Johnson and York (1915), in a study at Cold Spring Harbor (Long Island), found the key was elevation in relation to mean high water. Adams (1963) carefully classified many North Carolina marshes and their vegetation. He found the distribution of plants was also explained by tidal elevations.

General zonation of plants in the marsh can be described as being controlled by the interplay of two factors (Bertness and Pennings, 2000). The lower bound of a plant's distribution is set by physical stress, where some combination of factors makes it impossible for one plant to thrive and yet allows the other to succeed. This physical stress may change depending on evaporation levels, so that flooding determines distributions in "New England" marshes, and soil salinity drives the distribution in southern marshes (although the results from Hester et al., 1996, where soil salinity was determined not to be the prime determinant of southern plant zonation, dispute this last assertion). The upper bound of a plant's distribution is determined by competition, in that plants unable to thrive closer to the water are able to out produce those that can thrive there (Figure 1).

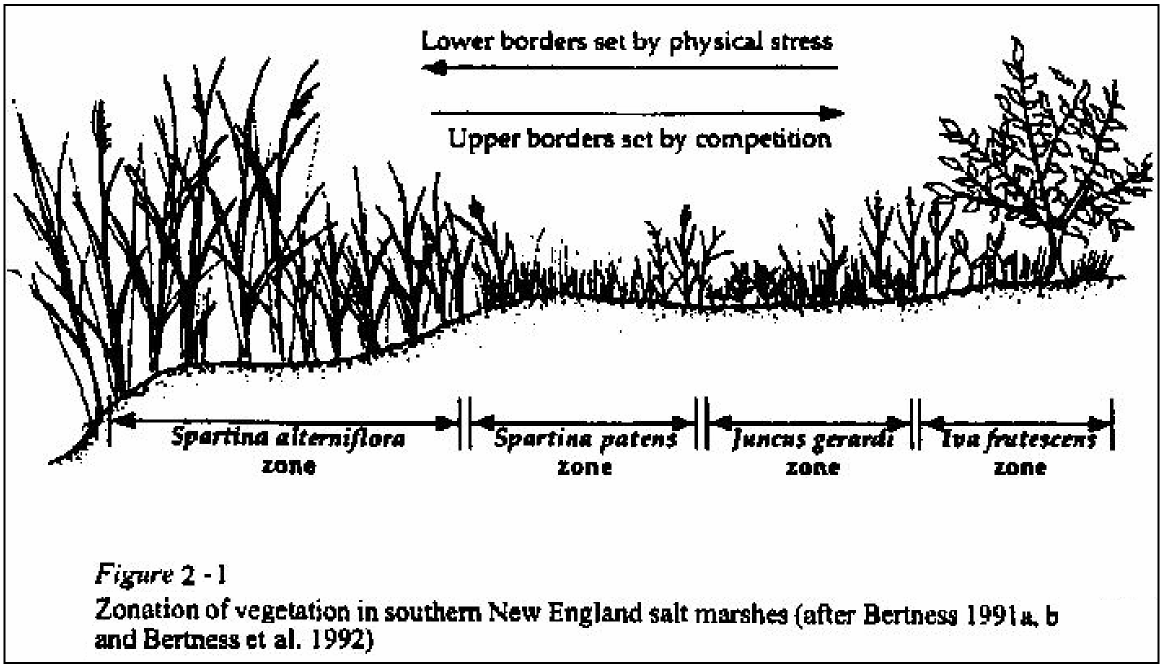

Figure 1. Zonation of vegetation in southern New England salt marshes (after Bertness, 1991a, 1991b, and Bertness et al., 1992)

Bertness has been constructing more complicated explanations of the distribution of plants across the marsh over the past decade or so. He found that *S. alterniflora* can be found in the high marsh, but only following disturbances (such as wrack smothering the pre-existing plants) and is soon displaced by *S. patens* and other high marsh plants (which cannot grow in the low marsh) (Bertness and Ellison, 1987). General zonation between *S. alterniflora* and *S. patens* was hypothesized to occur because of the interplay between several factors. One is that *S. patens* cannot oxygenate anoxic sediments, and so will not colonize bare patches in such environments. Secondly, *S. alterniflora*, in larger aggregations, can oxygenate such soils (but lone plants or small clumps cannot, or cannot do it well, and so are stunted). Finally, although *S. alterniflora* can grow in parts of the marsh where sediments are generally well oxygenated, it is displaced in those areas by *S. patens* after several seasons. Therefore, *S. patens* out competes *S. alterniflora*, but *S. alterniflora* can grow in more stressful environments where *S. patens* cannot (Bertness, 1991b). Fertilization by nitrogen from groundwater upwelling can alter the natural competitive arrangement of marsh plants (see, also, Levine et al., 1998). In fact, Pennings et al. (2002) described impacts from nutrient additions as being independent of specific marsh characteristics, in that *S. alterniflora* expansion, at the expense of high marsh plants, was apparently universal wherever nutrient inputs to a marsh are increased. Theodose and Roths (1999) might describe this depiction as a bit simplistic, as they found the zonation of plants in the high marsh to be a complicated interrelationship between nutrient availability (both nitrogen and phosphorus), and the actual absorption of nitrate by specific plants.

2.1.2.1 Low Marsh

The low marsh is the area flooded by all tides under normal conditions (Teal, 1986). In this zone, for the east coast of the US, the macrophyte community is typically a monoculture of *S. alterniflora*. Here the soil is usually muddy and saturated with water. This generally creates anoxic sediment conditions that can limit the ability of plants to colonize the substrate (Howes et al., 1981). *S. alterniflora* has aerenchyma tissue that supplies oxygen to the roots, thus aerating the soil in the near vicinity of its roots. The presence of already established soil oxygenating plants creates a less stressful environment, leading to denser growth and higher productivity (Witje and Gallagher, 1996a, 1996b).

S. alterniflora occurs in two forms, categorized as tall and short. Tall forms are found along banks and tidal creeks, and have thick, widely spaced stems. The short form is found in the remaining low marsh area and, in the south, throughout the high marsh. It is characterized by thinner, more densely packed stems of shorter stature. This disparity in growth is mainly a function of the environmental conditions under which the plants develop (Mendelssohn, 1979; Weigert and Freeman, 1990).

Ideal factors for growth along creek banks include the lack of competition for light and space, and a plentiful water supply. Adequate minor nutrients and potassium are present in the tidal waters, while major nutrients are available in the creek side mud. With these inputs, the tall form of *S. alterniflora* is as productive as any naturally growing plant (Teal, 1986). Odum (2000) believed that the energy inputs represented by the twice-daily tidal flushing were the ultimate sources of the high productivity. Productitivity values for *S. alterniflora* are listed in Table 2.

TABLE 2. Representative production values for *Spartina alterniflora* (all data above-ground growth only)

Location	Production (g m^{-2} yr^{-1})	Source	Notes
Central Gulf Coast	812	Mendelssohn and Morris, 2000	Marsh averages
Connecticut	699-80	Turner,1976	
Connecticut	920-1,250	Steever et al., 1976	
Georgia	3,700	Gallagher et al., 1980	Tall form
Georgia	1,300	Gallagher and Plumley, 1979	Short form
Georgia	2,840	Schubauer and Hopkinson, 1984	
Louisiana	2,895	White et al., 1978	
Maine	431-1,602	Linthurst and Reimold, 1978	5 methods
Massachusetts	1,320	Valiela et al., 1976	Tall form
Massachusetts	420	Valiela et al., 1976	Short form
New Jersey	500	Smith et al., 1979	
New York	827	Udell et al., 1969	Hempstead Bay
North Carolina	635-931	Hardisky, 1980	2 methods
North Carolina	214-1,038	Shew et al., 1981	3 methods
Northern Canada	60	Mendelssohn and Morris, 2000	Marsh averages
Rhode Island	480-832	Nixon and Oviatt, 1973	

As mentioned, a major portion of the low marsh productivity results in the formation of roots and rhizomes. The proportion of aboveground and belowground growth varies with the overall productivity of the area (Teal, 1986). In the most productive zones (tall *S. alterniflora*), nearly equal biomass is produced above and below the ground. In contrast, *Spartina* (generally) in areas of lower productivity directs considerably more energy to belowground growth. This is the case in both northern and southern marshes. This direction of production may be seasonal as well, with rhizomes storing energy as winter approaches to sustain rapid growth in the spring (Schubauer and Hokinson, 1984).

Steever et al. (1976) associated 90 percent of the variation in productivity in different Long Island salt marshes with tidal range, with higher tidal ranges corresponding to greater productivity. They also found this to be the case generally for the east coast. Evidently, tidal flux correlates with increased productivity, probably through the mechanisms of nutrient supply, waste removal, and salinity control (Teal, 1986).

2.1.2.2 High Marsh

Nixon (1982) considers the definition of a New England high marsh to be taxonomic. The high marsh includes the area dominated by salt marsh hay (*S. patens*) and spike grass (*D. spicata*) as well as its upland border, inhabited by black grass (*J. gerardi*) and switch grass (*Panicum virgatum*). At the upper elevations, the high marsh reaches a transition zone on the edge of the upland habitat. This fringe is dominated by shrubby species such as marsh elder (*Iva fructesans*) and groundsel bush (*Baccharis halimifolia*), along with *Phragmites australis* (*Phragmites*) and cattail (*Typha spp*.) where there is a fresh water influence. Table 3 lists productivity values for *S. patens*.

TABLE 3. Production values for *Spartina patens* (all data aboveground growth only)

Location	Production ($gm^{-2}yr^{-1}$)	Source	Notes
Connecticut	300	Steever, 1972 (cited in Nixon, 1982)	
Delaware	522-2,753	Linthurst and Reimold, 1978	5 methods
Georgia	705-3,925	Linthurst and Reimold, 1978	5 methods
Georgia	3,824	Pomeroy et al., 1981	Short form
Louisiana	1,428	White et al., 1978	
Maine	912-5,833	Linthurst and Reimold, 1978	5 methods
Massachusetts	1,100	Ruber et al., 1981	
New York	910	Harper, 1918	Cold Spring Harbor
New York	503	Udell et al., 1969	Hempstead Bay
Rhode Island	430	Nixon and Oviatt, 1973	

Nixon (1982) quantified the belowground production of roots and rhizomes as being four times the aboveground value. This dense subterranean growth drives the vertical growth of the marsh through its volume and sediment trapping ability. Maintenance of high marsh elevation prevents inundation by tides and so perpetuates the environmental conditions that foster the particular plants found there (Redfield, 1965; Nixon, 1982).

The number of plant species in a marsh increases with elevation, with greatest variety in the marsh border (Miller and Egler, 1950). Besides the marsh hay and spike grass, the high marsh is home to sea lavender (*Limonium carolinianum*), seaside plantago (*Plantago juncoides*), slender salt marsh aster (*Aster tenuifolius*), seaside goldenrod (*Solidago sempervirens*), salt bush (*Atriplex patula*), sealite (*Suaeda linearis*), and glassworts (*Salicornia spp.*) (Nixon, 1982).

Salt marshes of the New England type constitute less than two percent of the marshes along the US Atlantic coast and the high marsh may amount to only 25 to at most 50 percent of that amount (Nixon, 1982). The portion of marsh covered by high marsh species may be decreasing due to losses to development. In addition, nitrogen loading may lead to intrusions of *S. alterniflora* into the *S. patens/Distichlis* zone. Apparently, adding nitrogen tilts the competitive balance in favor of salt marsh cordgrass (Bertness et al., 2002).

Nixon (1982) noted that *S. patens* forms a more tussocky, uneven surface than *S. alterniflora*, and that "rotten spots" may form under high marsh cowlicks. Sediment deposition on the high marsh tends to keep pace or slightly exceed sea level increases (Redfield, 1965).

While *S. patens* largely dominates the New England high marsh, it becomes relatively uncommon in southern marshes. In fact, the ratio of high marsh to low marsh generally decreases with latitude, falling to 0.3 in Georgia (Spinner, 1969). In southern marshes, short-form *S. alterniflora* dominates the high marsh, intermingled with glasswort (*Salicornia bigelovii*), sampshires (*S. europ, S. virginica*), and spike grass. At higher, drier elevations, black needle rush (*Juncus roemerianus*) dominates, displacing the northern species, *J. gerardi*. The upper marsh fringe, where salinity drops to 2 parts per thousand (ppt) or less, is covered with big cordgrass (*Spartina cynosoroides*) and, to a lesser extent, salt marsh bulrush (*Scirpus robustus*) (Wiegert and Freeman, 1990).

Intra-marsh salinity profiles vary with latitude, and are driven largely by climate (Pennings and Bertness, 1999). In a New England marsh, salinity usually decreases from the waters edge to the terrestrial border. In contrast, with greater evapo-transpiration rates in the south, hypersaline soil is typically found at mid marsh elevations, even in undisturbed stands of vegetation (Weigert and Freeman, 1990). These areas are usually not affected by frequent tidal inundation or significant freshwater runoff from the uplands. In low spots or areas of poor soil drainage, evaporation increases interstitial salinities to levels where no vascular plants can survive. These salt pannes (salt barrens) may have a thin covering film of blue-green algae, and are often ringed by succulent *Salicornia spp*. Pannes that form on the high marsh in New England marshes are usually formed by different mechanisms, such as smothering of vegetation by wrack (Nixon, 1982).

Williams et al. (1994) discussed how high marsh water tables fluctuate in response to irregular flooding, noting that the degree of variation is a function of the frequency and duration of flooding, marsh elevation, proximity to and the number of creeks, depressions, pannes, and sediment type. Pomeroy and Imberger (1981) disagreed somewhat, describing how natural creeks drain little water from the marsh, and describing a consistent perched water table. The disassociation between the creek water and the perched water table is marked by differences in salinity. The salinity of the creek waters is usually the same as the water found in the bankside levees, but the marsh water table water is usually higher in salinity (due to high evaporation in this Georgia marsh).

Frey and Basan (1985) noted that the hydraulic head (water pressure) associated with tides controls percolation through the marsh surface. Percolation of estuarine water into sediments is believed to be important to many marsh processes, including oxidation of peat, aeration of root zones, redistribution of salts, and the transport of nutrients. Burke et al. (1980) measured infiltration into the marsh surface; infiltration during high tides was matched by discharge from the sides of tidal creeks during low tides, and the amount of water infiltrating into the marsh surface decreased with distance from the marsh creek. Harvey et al. (1987) found that the head in the marsh peat layers dictates horizontal flows to the creek bank following retreat of the tide off the marsh surface; their hydrological model indicated that two-thirds of the water infiltrating the marsh surface during a tide will drain out of the marsh during that same tidal cycle (note the study was made in a shallow, 20 meter [m] wide, *S. alterniflora* marsh that was completely flooded each tidal cycle). Howes and Goehringer (1994) found that flow out of the creek bank is non-linearly related to the height of the creek bank; flows from taller banks were much greater than those from lower banks.

Nixon (1982) indicated that there are seasonal cycles of mean sea level increases and decreases, and that tide heights are extremely variable through lunar and seasonal cycles, and are often greatly affected by meteorological conditions. This means the number of times a portion of the marsh is inundated by tidal flows and the height of those tides over the marsh varies according to a number of factors.

2.1.2.3 Phragmites

Phragmites australis (Cav.) Trin. ex Steudel (formerly *Phragmites communis*, simply called *Phragmites*) has been present in North America for at least 40,000 years (Salstonstall, 2002). It has recently become more aggressive and invasive. Salstonstall has determined that the more aggressive strain is the same genotype as is found in Europe, and therefore this is a non-native, invasive strain. It has the ability to grow more than four m in height with a dense underground rhizome system.

Where and when *Phragmites* became invasive an, and, therefore, a problem, is often disputed. Redfield (1972) found localized *Phragmites* presence at the upland edges of a ditched marsh, where freshwater inputs were notable. In 1984, Clarke et al. found a Massachusetts marsh was being encroached upon by *Typhus* (cattails) from the freshwater edges, and did not mention *Phragmites*. Orson et al. (1987) found *Phragmites* in cores ranging back thousands of years in a Connecticut marsh, but noted that there are "recent" increasing monospecific stands of *Phragmites* in the marsh. A more generalized study by Orson (1999) of cores from several Connecticut marshes, and one each in Rhode Island and Massachusetts, found *Phragmites* dating back thousands of years, but only in association with marsh edge or brackish marsh plants.

Monospecific stands and/or associations with *Spartina spp*. are mostly not discussed until 50 to 100 years ago. Generally, northeastern US salt marshes are now noted as being heavily invaded by *Phragmites* (Lathrop et al., 2003). However, there are also data suggesting not all invasive events are caused by European-stock *Phragmites* (Lynch and Salstonstall, 2002).

On Long Island, Lamont (1997) noted that *Phragmites* was collected in Jamaica Bay in 1864, and from Wading River in 1872. Harper (1918) reported *Phragmites* upland from *S. patens* at a marsh near Whitestone. At Cold Spring Harbor, it is clear that the salt marsh was free of *Phragmites* as late as 1920; however, by 1997, the high marsh was monoclonal *Phragmites*. Similarly the spread of *Phragmites* on the East End of Long Island can be traced from Orient in 1900 to Cutchogue by 1918 and the South Fork by 1920 (Lamont, 1997). Udell et al. (1969) did not mention *Phragmites* in a discussion of primary production in Hempstead Bay (albeit, only the four most common marsh plants were discussed). O'Connor and Terry (1972) generally found that *Phragmites* was restricted to areas impacted by dredge spoils or without much salt water influence, although south shore areas with higher salinities and *Phragmites* presence were noted. Cademartori (2000), in an unpublished thesis, found *Phragmites* increases in Stony Brook Harbor from the 1930s through 1999. She linked increased fresh water inputs from upland drainage to the increases in *Phragmites* abundance.

According to Penny (1977), local residents linked *Phragmites* expansion on the East End of Long Island to the Hurricane of 1938. Support for this theory comes from Bart and Hartman (2003), who thought storms could upend natural salinities enough to allow a *Phragmites* foothold to develop. Dreyer and Niering (1995) specify that *Phragmites* invasions are due to reductions in tidal flooding. Burdick and Konisky (2003) suggest the reaction of *Phragmites* to the stresses brought about by salt water flooding are not well-described, and so it is not clear whether or not there are fundamental reasons that restrict *Phragmites* from saltier waters. Many others note that salinities in excess of 18 to 20 ppt seem to inhibit or reduce *Phragmites* growth (Marks et al. 1994; Meyerson et al., 2000; Bart and Hartman, 2002; Chambers et al., 2003; Havens et al., 2003; Lathrop et al., 2003). Witje and Gallagher (1996a, 1996b) found that *S. alterniflora* seeds could germinate at higher salinities than could *Phragmites* seeds, and that the *S. alterniflora* seeds grew rapidly, especially under anoxic conditions that *Phragmites* did not tolerate. They suggested this ability to tolerate more stressful conditions may establish initial zonations between the plants on marshes. Havens et al. (2003) even suggest constructing subtidal ditches to convey saltier water into *Phragmites* stands to reduce their expansion. Hellings and Gallagher 1992 found that the combination of flooded conditions and 18 ppt salinity can prevent rhizomes from budding. Bart and Hartman (2003) suggested that the burial of larger rhizomes (perhaps through ditch maintenance, storms, or even duck blind construction) in well-drained areas such as ditch or creek banks is the way that *Phragmites* might overcome otherwise hypersaline marsh conditions.

The spread of *Phragmites* in a Connecticut salt marsh, noted by Orson et al. (1987), was attributed to uplands development (citing Roman et al., 1984). Bertness et al. (2000) also found that shoreline development precipitated *Phragmites* expansion (there was a positive, statistically significant correlation between marshes with developed fringes and *Phragmites* invasion of the marsh), and said the mechanism for the change was unbalanced competition due to nitrate inputs. In 2004, Bertness et al. modified this position slightly, suggesting that development of the marsh fringe reduced absorption and/or infiltration of precipitation, and that increased runoff over the marsh surface caused *Phragmites* expansion, through a lowering of salinities and increased inputs of nitrate. Meyerson et al. (2000) point to nitrogen inputs as the initiating event causing *Phragmites* expansion. Decreases in sulfide concentration were shown to allow *Phragmites* to better absorb ammonium and so meet the nitrogen conditions necessary for it to out compete *S. alterniflora* (Chambers et al., 1998). *Phragmites* appears to alter nitrogen flows within a marsh, increasing the amounts found in standing vegetation, which may change the nitrogen balance for an invaded marsh (Windham and Meyerson, 2003). Marks et al. (1994) summarized the conditions resulting in the spread of *Phragmites* as disturbances and stresses, such as pollution, hydrologic changes, dredging, increases in sedimentation and/or soil salinity (from fresh to brackish) and/or nitrate concentrations, all abetted by the potentially invasive European genotype. Another summary of the requirements for *Phragmites* expansion included salinity less than 10 parts per thousand, low sulfide concentrations, and inundation frequencies less than 10 percent (measured in terms of number of times flooded per number of high tides) (Chambers et al., 2003). These changes may impact only a small area on a particular marsh, allowing only a few plants to expand their range, and then this initial foothold can allow for expansion by *Phragmites* into otherwise less inviting habitat and eventual domination of the entire habitat. On the other hand, Burdick and Konisky (2003), although agreeing that greater drainage is important and nitrogen inputs may play a role, suggested that filling and road-building are greater contributors to *Phragmites* invasions, by compacting soils and increasing groundwater inflows (resulting in decreased salinities).

The rapid vertical and horizontal clonal growth of *Phragmites* allows it to overgrow other wetland plants by physical displacement. Its tall, dense aboveground growth alters environmental conditions such as light, space, and temperature (Meyerson et al., 2000). It has been shown to invade areas periodically flooded by full strength seawater through clonal integration (Amsberry et al, 2000), and by accessing freshwater lenses via deep taproots (Meyerson et al, 2000).

In a Rhode Island marsh, Nixon and Oviatt (1973) found aboveground *Phragmites* production to be greater than that of other vascular plants. It was measured at 900 gm^{-2} compared to 680 gm^{-2} for a *S. patens* and *D. spicata* mixed area. However, it is reported that in North America, *Phragmites* is not be consumed to any great extent by wildlife, nor is it considered an important nesting habitat for most marsh-resident birds (Buchsbaum et al., 1998). Wainright et al. (2000) did find that *Phragmites* may be contributing to the mummichog (*Fundulus heteroclitus*) food chain, and so to marsh food webs generally.

2.1.3 Limiting Factors for Production

The environmental factors of light, salinity, and nitrogen availability may act to limit productivity in salt marshes. Productivity varies with latitude, as well. Gross production is almost nonexistent in New England marshes in winter (Nixon and Oviatt, 1973), but in southern climates there is a continual, albeit retarded, plant growth through the cooler periods. Even in summer, the efficiency of net production of all grasses in the marsh is lower in the north, a result of the shorter, cooler growing season and lower solar input (Wiegert and Freeman, 1990).

2.1.3.1 Light as a Limiting Factor

Light is a major factor limiting algal production. The most productive areas of the marsh for algae are unvegetated creek banks (Pomeroy et al, 1981). Blum (1968) attributed the lack of algae cover in the high marsh to shading of the surface by dense *S. patens* growth. On a spring day he found only two to three percent of incident light reached the soil, while 50 to 55 percent did so under stunted *S. alterniflora*, where algal growth is greater. Thus, growth is greater where plant stems are more widely spaced (i.e., *S. alterniflora* stands) or in pannes and puddles (Gallagher and Daiber, 1974). Additionally, a large portion of annual benthic microalgal growth occurs when vascular plants are dormant in early spring and fall. This may represent the primary source of newly fixed C on the marsh during this time (Sullivan and Currin, 2000). Phytoplankton in marsh creeks can be extremely light limited by the turbidity of the water.

Phragmites may limit access to light by its competitors (see above). Tall-form *S. alterniflora* may out compete other marsh grasses for light – and, yet, still be competitively disadvantaged because of nutrient limitations (Bertness et al., 2004). Accumulation of wrack across the high marsh can lead to die-off of the smothered plants. This appears to be the mechanism by which pannes are created in New England marshes, and may be how "rotten spots" are formed there (Nixon, 1982).

2.1.3.2 Salinity as a Limiting Factor

Salinity is a limiting factor in production and in setting plant community boundaries. High soil salinities play a larger role in southern marshes, where higher temperatures and evapo-transpiration rates lead to salt accumulation. Permanent salt pannes, where salinity is so severe that no plants can grow are common in the south, but not in New England marshes (Chapman, 1960).

Although *S. alterniflora* is a salt obligate (Adams, 1963), it grows better under lower salinity conditions. The plant exudes salt out of stomata as a means of controlling osmotic pressures. Stressed at salt levels of 25 ppt or more, it is still able to grow better than its competitors. There is a limit to its tolerance. As salinity increases, plants exhibit higher respiration rates and reduced productivity (Witje and Gallagher, 1996a, 1996b). Above 45 ppt salinity, there is a dramatic increase in respiration and a decrease in growth, with survival times decreasing with length of exposure (Teal, 1986).

2.1.3.3 Nutrients as Limiting Factors

All the minor nutrients needed by plants, as well as the major nutrient, phosphorus, are present in seawater. Coastal marshes, as with most other coastal marine ecosystems, are nitrogen limited (Bertness, 1991a). Increasing nitrogen (N) input has been shown to increase primary productivity of both grasses and algae, with exception of the already highly productive tall from of *S alterniflora* (Chalmers, 1982). Even low N additions ($0.8 gN\ m^{-2} wk^{-1}$) during the growing seasons more than doubled the aboveground production of *S. patens* and *Distichlis* in the high marsh (Valiela and Teal, 1974). With very high N levels, the microbial denitrification pathway out competes plants for additional N uptake.

Levine (1988) showed that the addition of N alters the competitive balance in a New England marsh. Without exception, adding N led to the success of the usual competitive subordinate and the decreased success of the usual competitive dominant. Bertness et al. (2004) have also presented evidence that increasing N levels is a factor in changing species compositions. Greater inputs of N from the terrestrial border may allow *Phragmites* to invade the salt marsh, and allow *S. alterniflora* to displace high marsh species. The increase in nutrient availability may alleviate belowground competition for nutrients and lead to aboveground competition for light, thus favoring strong aboveground competitors.

2.1.4 Fate of Production

Sapelo Island Georgia was the center of a plethora of research on salt marsh productivity in the late 1950s and 1960s. The highlight was Teal's 1962 publication in *Ecology*, which concluded with:

> "…the tides moved 45% of the production before marsh consumers have a chance to use it and in doing so permit the estuaries to support an abundance of animals."

Odum (1961) used the term "outwelling" to describe the movement of nutrients and energy from shorelines to estuaries and coastal waters, as a parallel process to the delivery of nutrient by upwelling. He envisioned rivers and coastal marshes as being major contributors of allochthonous materials to support coastal productivity in the same way that deep waters enriched in nutrients can enrich particular coastal areas. Subsequent publications bolstering this hypothesis contained little in the way of data to support the selection of salt marshes as sources of nutrients and C (Odum and de la Cruz, 1967; Pomeroy et al, 1967). Nonetheless the idea that salt marshes act as productivity pumps that feed adjacent waters became dogma (Nixon, 1980).

Nixon's 1980 review of marsh-estuarine interaction studies determined that most generalized salt marsh scientific theory was grounded not in data, but in speculation. Nixon concluded that tidal marshes appear to export organic C, but that the available data available did not substantiate the outwelling hypothesis as defined at the time.

It is difficult to measure the flux of C accurately from marshes. Water flows through channels vary with tides and weather conditions. Measurements of steady state conditions may not capture episodic events such as the transport of wrack. Many researchers now focus on secondary consumers, especially transient, predacious fish, as the main export pathway from salt marshes. Others have focused on differentials between "old" and "young" marshes, as the degree to which peat is being formed may play a major role as to whether C is being retained within the marsh, or is in excess and is available for export from the system. Childers et al.

(2000) reviewed the studies performed since Nixon's criticism, and were not convinced that the major studies provided adequate proof of outwelling.

Odum (2000) has radically modified the hypothesis statement, in any case. Rather than being viewed as steady state exporters of productivity, marshes may export through episodic or "pulsing" events that are associated with heavy rainfall or unusually high tides. The degree of export appears to be a function of individual marsh productivity, maturity, tidal amplitude, and geomorphology. A mature marsh has filled its basin to the high tide level and acts as a sediment sink only in relation to the gradually rising sea level (Teal 1986). This is the case with the Great Sippewissett Marsh in Massachusetts, which exports organic C to Buzzards Bay. In contrast, a young marsh, such as Flax Pond on Long Island, may show a net import of organic C from surrounding waters (Valiela, 1982).

The physical structure of a marsh system affects the export/import role. Odum (1979) classified marshes into three types according to their flow and tidal exchange characteristics.

1) Where tidal exchange occurs through a restricted or long and narrow channel, export of production (if any) would be lessened.

2) Marshes located in restricted basins or basins newly opened to the sea that are not importers of sediment may only export to the adjacent basin.

3) Outwelling (steady export of material), at least to nearby waters, is most likely to occur in areas where mature, productive marshes are extensive, and are open to the sea, if tidal amplitudes sufficient to provide the energy to drive the export occur. These are the areas that Odum (2000) now terms "outwelling hot spots."

Further review of the basis of the theory has found many difficulties. Vascular salt marsh plants tend to create refractory detritus. Ribelin and Collier (1979) and Haines (1979) found algal derived organic matter to constitute the bulk of marsh detritus in surrounding waters. Studies employing stable isotopic analysis have concluded that benthic microalgae produce 50 percent or more of the C assimilated by marsh consumers (Sullivan and Currin 2000). The algal consumers, such as killifish, amphipods, snails, and fiddler crabs, in turn are eaten by transient fish and bird species.

Transient marine fish may directly graze on detritus, microbes, microflora, and algae as larvae or juveniles, in the warmer, protected marsh creeks and on the marsh at high tide. By exiting the marsh in the fall, they accomplish a trophic relay to coastal waters. This may be the dominant pathway for salt marshes to support off shore fisheries (Deegan et al., 2000). Smith et al. (2000) found that mummichogs, because they may consume detritus directly and in turn are preyed upon by non-resident fish, may represent an important transfer mechanism for marsh productivity to the estuary. Transient black ducks and Canadian and snow geese feed in marshes on vegetation (Nixon, 1982) and hence export a difficult-to-quantify level of production.

The primary production that is not exported will be accumulated as peat, decomposed within the marsh, or consumed directly. Grazers consume approximately 10 percent of vascular plant production (Weigert and Freeman, 1990). The marsh accumulates only a small portion, the majority decomposing on the marsh surface or washing away with tides. Some 60 percent of the net C fixed by *Spartina spp*. is deposited belowground as roots and rhizomes. Some is used as an energy source for new spring top growth. Most of the remainder is decomposed within the usually anoxic sediments through denitrification and sulfate reduction. After two years on a New England marsh, only five percent of the initial detritus remains (Teal, 1982). This residue resembles the resident organic marsh sediments and probably accumulates as marsh peat.

Valiela and Teal (1979) found that, because nitrogen demands exceed inputs for salt marshes, recycling of nutrients is essential, and that the nutrient exchanges between uplands and the open waters are important to structuring the salt marsh. Teal (1986) noted that nitrogen cycling in the marsh results in the conversion of nitrate to organic nitrogen (especially bound into particles) and ammonia, and that nitrogen tends to be pulsed from the marsh in the fall instead of being released more steadily with stormwater outflows. Woodwell et al. (1979) described essentially the same relationship for a Long Island marsh in the fall, but found that the marsh imported nutrients from Long Island Sound in winter. Wolaver et al. (1980) were not certain that the process was consistent from marsh to marsh, but tended to support a seasonal export model on the whole. Most denitrification (nitrate converted to nitrogen gas) in marshes occurs in the muddy bottoms of creeks (Kaplan et al., 1979), meaning that increases in creek bottomland may aid a marsh's ability to treat additional nitrate inputs.

2.2 Detrital Consumption

Most organic matter associated with the detritus complex is derived from vascular plants. Other dead producers and consumers can contribute to detritus, but most of this is rapidly recycled, unlike resistant plant lignocelluloses. Detritus from the plant community has low nutritional value. Experimentally, no consumers have been able to grow or produce when cultured on sterile detritus (Kreeger, 1988). Decomposers, in or on plant-derived detritus, initiate the transfer of C fixed by plants to forms that can be utilized by the fauna of the marsh and estuary. Fungal activity is the key agent for detrital decay with a standing crop calculated at three (summer) to 28 percent (winter) on a per square meter basis of the living cordgrass standing crop in a Georgia marsh (Newell and Porter, 2000).

Levels of productivity for bacterial decomposers in the same marsh were about two times that of fungus in the summer and 0.07 percent in the winter. Other consumers often consume the microbially coated detritus. The bacterial biomass is digested, leaving the macerated detrital particulate substrate for defecation and subsequent recolonization, a process called "microbial stripping" (Newell, 1965). Such deposit-feeding strippers, including amphipods, gastropods, and fiddler crabs, consume the substantial fungal and bacterial biomass, resulting in high levels of secondary production.

This microbial stripping process has also been proposed for aquatic consumers such as oysters and mussels (Newell, 1965). The bacterial food they seek can be removed from the water column more efficiently when consumed with larger particles. Mummichogs have also been found to ingest detritus, but can not gain weight without the supplemental protein the microbes provide (Prinslow et al, 1974). Large young of the year (YOY) consume detritus and its microbial coating and then transport marsh surface primary production to surrounding intertidal creeks when migrating with each tidal cycle.

Filter feeders commonly found in or near salt marshes include ribbed mussels (*Geukensia demissa*), oysters (*Crassostrea virginica*), hard clams (*Mercenaria mercenaria*), and, in southern marshes, the marsh clam (*Polymedosa caroliniana*). These invertebrates are adapted to consume large quantities of seston, containing particulate organic matter from a wide variety of sources. The seston contains detritus, phytoplankton, suspended surface-associated algae, bacteria, microheterophic protests, and unidentifiable organic aggregates. As the nutritional value of these items varies greatly, they are selectively sorted and utilized (Teal, 1986; Wigert and Freeman, 1990).

Ribbed mussels are common marsh inhabitants along the Atlantic and Gulf coats. In many cases, their biomass can exceed that of all other marsh metazoans combined (Jordan and Valiela, 1982). It has been estimated that the mussel population is capable of filtering the entire volume of water on the marsh per tidal cycle in the course of a year. Due to its dominance in both biomass and secondary production, it may be a keystone marsh species.

Despite ingesting copious quantities of suspended detritus, ribbed mussels utilize little directly. Using C^{14}-labeled micro-particulate detritus from *S. alterniflora,* Kreeger and Newell (2000) found it only supplies one to nine percent of the mussels' C requirements. They suspect lower rates of utilization for other bivalve suspension feeders. As with other marsh consumers, detrital consumption occurs indirectly via microheterotrophic intermediaries (detrital decomposers), with a high rate of efficiency. More than half of ribbed mussel C demands can be met through this pathway (Peterson et al., 1981; Langdon and Newell, 1990; Kreeger and Newell, 1996). This organism is an important conduit for moving detrital C to higher trophic levels.

As roots and rhizomes die, their organic compounds provide energy for oxidative and fermentative transformations. The nature and rate is determined by the oxidative state of the benthic environment. Aerobic sediments are found only in the top few mm, and in microzones around *S. alterniflora* roots and fiddler crab and other infaunal burrows. Anoxic processes use nitrate and sulfate as electronic acceptors in lieu of oxygen. Both pathways yield less energy to microbes performing them than does aerobic decomposition, with sulfate reduction being less efficient. Therefore where oxygen is available, breakdown is performed by aerobic organisms. At sediment depths where no oxygen is present, denitrifying organisms dominate. Decomposition via sulfate reduction occurs deeper in the sediments and at the slowest rates. The actual amount of decomposition that flows through these various pathways is the subject of much research.

2.3 Primary Consumers

Animals feeding at the bottom of the food web in salt marshes have a wide variety of foods from which to choose. Photosynthetic organisms include the vascular plant community, epiphytic algae, macroalgae, microphytobenthos, and phytoplankton. These are mostly autochthonous with the exception of the phytoplankton imported to the marsh with flooding tides. With the great productivity of the vascular plants and high level of secondary algal production, one would expect to find a flourishing community of grazers; however, in southern marshes only two species of *Spartina* grazers are present (Wiegert and Evans, 1967). This was attributed to low maintenance cost poikilothermic organisms dominating – as is not the case with terrestrial grasslands, for instance, where homeotherms are common (Humphreys, 1979).

Only a few species of marsh invertebrates consume living plant material as a sole nutritional resource, removing less than 10 percent of overall plant production (Teal, 1962). In southern marshes, this group includes only the grasshopper *Orchelimon fidicinium* and the plant hopper *Prokelisa marginata*. In New England marshes such as the Great Sippewissett marsh, insect herbivores include the chewers such as the longhorned grasshopper (*Conocepnaluss spartinae*) and the suckers such as plant bugs (*Miridae*), plant hoppers (*Delphacidaw* and *Cicadellidue*), aphids, and scale insects (Teal, 1986; Nixon, 1982). The only other invertebrate reported to directly feed on vascular plants were gastropod snails. They preferentially feed on epiphytic microalgae and fungi colonizing senescent plants and only consume living plant material when forced to by high population densities (Bertness et al., 2004).

Much has been made of the "detritus driven" food chain in marshes, yet as Haines (1979) points out,

> "in the purest sense, the only detritivores are the bacteria, fungi and perhaps polychaete worms which assimilate plant material directly."

Meiofaunal consumers of these organisms include protozoa and nematodes, the latter being very numerous (Kroczynski and Ruth, 1997). The feeders usually considered detritivores (fiddler crabs, snails, grass shrimp, mummichogs) actually should be classified as "opportunistic omnivores" (Haines, 1979); they defy easy classification in a classic trophic scheme. Fiddler crabs ingest algae, detritus, foraminiferans, nematodes, inorganic particles, and sometimes carrion (Teal, 1962). Mummichogs are often predatory, consuming snails, grass shrimp and other crustaceans, but also filter detrital particles and algae from the water, and feed on carrion when available (Valiela et al., 1977). The marsh snail (*Littorina irrorata*) grazes on the marsh surface at low tide, ascending the cordgrass to feed on standing dead shoots and its associated microbes at high tide. Omnivory is the rule where there is a scarcity of food and variability in food type and quality from place to place or through the season (Odum, 1971). Kreeger and Newell (2000) found that no single food source could meet both the C and N demands for most consumers in a salt marsh.

What the marsh lacks in species diversity it makes up for in numbers of individuals of select invertebrate species. The Sapelo Island marsh intertidal zone was found to have a macroconsumer biomass of 15 gC m^{-2} (Montague et al., 1981). This included 80 to 200 mud fiddler crabs (*Uca pugnax*), 400 to 700 mud snails (*Ilyanassa obsolete*) or marsh snails, and seven to eight ribbed mussels per square meter of marsh (Teal, 1962).

Salt marsh sediments contain high levels of organic C, making it a desirable habitat for deposits feeding invertebrates. Meiofaunal deposit feeders include nematodes, harpacticoid copepods, amphipods, polychaetes, oligochaetes, turbellanans, and ostracods. Macroinvertebrate deposit feeders include fiddler crabs, snails, grass shrimp (*Palaemonetes spp.*), annelids, and certain bivalves (Teal, 1962). Some are selective feeders, sorting sediments before ingestion to increase the food value. The mud snail feeds primarily on algae on the mud flats. Grass shrimp graze phytoplankton and benthic algae. Teal called fiddler crabs "moderately selective." They group sediment, remove large inorganic particles, and ingest the remainder, assimilating approximately 25 percent of it. Other macroinvertebrate deposit feeders have a similar strategy for balancing nutrition.

Fiddler crabs are major consumers of marsh production and greatly impact the intertidal zone where they reside. By burrowing 10 to 30 centimeters (cm) deep, they work over much of the top layer of the low marsh each season. This increases the soil drainage and oxygen content, and in turn may enhance plant growth. Diatom-production may increase as they are brought n closer contact with light and nutrients (Montague et al., 1981). Large quantities of living and recently dead biomass are brought to the surface and deposited, hastening decomposition (Weigert et al., 1981). Bertness (1992) called them the "earthworms of the marsh."

The marsh's dominant suspension feeder is likely to be the rubbed mussel, important for its biomass and productivity, water filtering, and deposition of nutrients in the marsh. An individual mussel may filter up to five liters of water per hour while feeding. This can decrease water turbidity, aiding phytoplankton production. The nutrient-rich feces and pseudofeces deposited can increase the growth of nearby *Spartina* by 50 percent in a season (Bertness, 1992). Byssal strands produced by the mussel serve to anchor it to the substrate and cordgrass roots, binding the marsh and decreasing erosion. The mussel shell provides a stable habitat for organisms like barnacles. Mussels have a varied diet. Phytoplankton are readily ingested and assimilated, but are only seasonally abundant. This is also the case for benthic algae that may be suspended by the tides. Detrital cellulose directly supplies little of its C needs, but associated microbes make up a large portion of a mussel's diet (Kreeger and Newell, 2000).

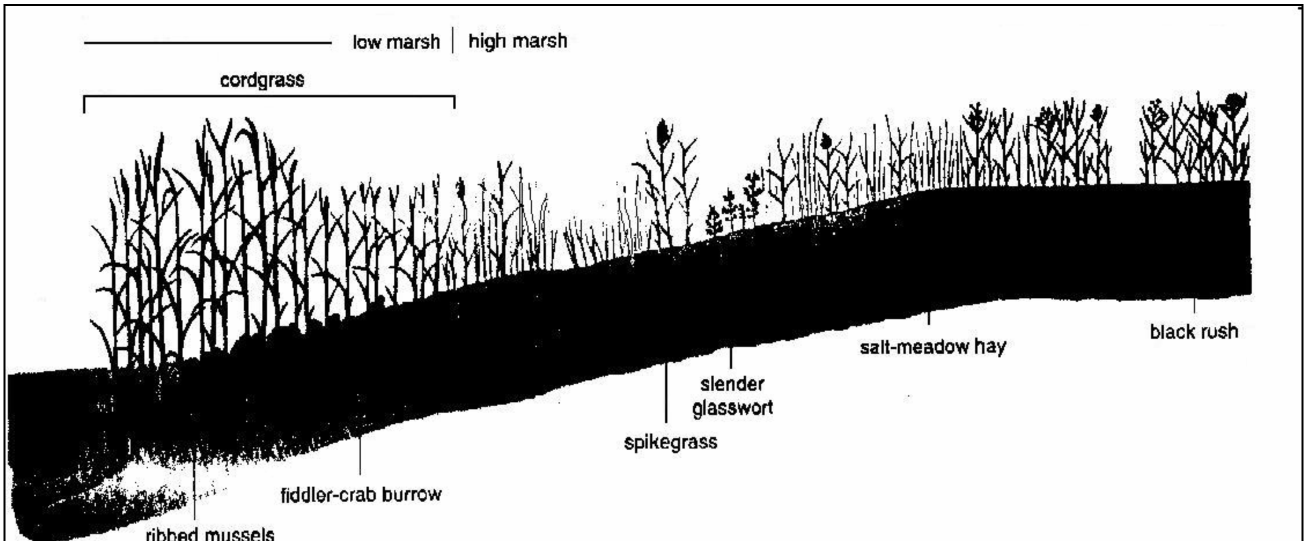

Figure 2-2
Distinct zones characterize a New England salt marsh, for animals as well as plants. On the coastal side of the low marsh, shown in cross section, thick beds of ribbed mussels are attached to the roots of cordgrass. The mussels decrease in abundance as one moves inland, and the marsh soil becomes dotted with small holes, the burrows of fiddler crabs. These too, however, are largely limited to the low marsh. Even the cordgrass of the low marsh is divided into two zones: The cordgrass closer to the sea is tall and the cordgrass farther from the sea is short because soil there is composed of compacted peat produced by the decay of the grass. The high-marsh zones of salt-meadow hay and black rush are primarily monocultures, but disturbed areas support small populations of spikegrass and slender glasswort.

(Bertness et al. 1992)

Figure 2. Zonation of a New England marsh (Bertness et al., 1992).

Craig and Crowder (2000) note there are six factors that influence fish choice of habitat:

- abiotic factors (dissolved oxygen, temperature, etc.)
- food resources
- bioenergetics (tradeoffs in energy associated with abiotic factors, food resources, predator presence or absence, etc.)
- competition
- predation
- the influence of ontogeny and habitat structure (how a particular habitat may affect or change an organism's growth)

Their review of 61 papers found that the earliest studies tended to focus on abiotic factors, but later studies were more wide-ranging.

Wiegert and Freeman (1990) noted that most fish in marine recreational fisheries require marshes for juvenile life stages. Knieb (1997) noted that most studies of marsh nekton (those creatures capable of self-propulsion horizontally) are biased towards fishes and towards species of commercial value – which tend to be transient species. Knieb (2000) also asserted that fish in salt marshes are drawn from the assemblages found in the estuary, although the marshes are said to have lower species richness. Deegan et al. (2000) called the non-resident species "marine transient" species, and identified menhaden, mullet, croaker, spot and flounder as typical examples. They also cited a report (Day et al., 1989) that found over 55 species of fish from estuaries along the eastern and Gulf coasts of the United States could be classified as marine transients. Deegan et al. do point out that it is difficult to say whether or not a fish "requires" salt marsh habitat, since it may only spend a few weeks a year in or near one. Miller and Dunn (1980) point out that transient juvenile fish may feed in estuarine environments for one of two reasons:

1. food is concentrated there due to high productivity rates; or

2. there are so few predacious native estuarine fish (often only two or three species) to compete against.

Craig and Crowder (2000) cited many studies that found estuarine fish had fuller stomachs on ebb tides as compared to flood tides, which supports the notion that they move into creeks and ditches to feed. Bertness et al. (2004) limit the use of salt marshes by fish to many commercially important species (specifying shrimp, oysters, and crab) that use the edge of the marsh as nursery areas. Nixon (1980), on the other hand, found no relationship between fisheries data and the amount of marshes near a particular estuarine system. Therefore, he thought the entire thesis of marshes serving as important nursery areas for estuarine and coastal fishes unfounded (as part of his general discussion of whether marshes serve as "outwelling" sources). Weinstein et al. (2000) did show that bay anchovies captured several kilometers offshore from a marsh had isotopic signatures similar to salt marsh microalgae, however.

Haines (1979) noted that the true nursery area of an estuary is not the open water of the sounds and rivers, but the wetlands themselves and their creeks, because small fish forage on and use the marsh for protection. Deegan (2002) noted that it appears stem density is the true cause of predator aversion from marshes.

In a study of a tidal, but fresh, marsh, McIvor and Odum (1988) found that the small resident fishes used the creek channels at low tide, but then moved into the adjacent marsh area as the surface was inundated. They showed that fish entering the marsh surface preferred gently sloping creek sides to more sharply sloped banks. Rozas et al. (1988) further showed that although rivulets can be important pathways for fish to reach the marsh surface, by far most fish used the creek banks as the means to access the surface of the marsh. Knieb (1997) noted that increasing complexity of the drainage systems increased fish use of the channels. Intertidal creeks rarely drain completely, and the remnant ponds and rivulets can serve as fish habitat for very small, marsh-resident fish. In addition, Knieb found that "microhabitat" (small impermanent pockets of water on the marsh surface) was very important to larval and juvenile marsh fish.

Several papers noted that the duration of tidal flooding (percent flooded on a monthly basis) is probably the greatest factor in determining habitat use by different species (Knieb and Wagner, 1994; Rozas, 1995; Knieb, 1997; West and Zedlar, 2000). Knieb (1984) found that *Fundulus spp.* appeared to use the marsh surface as a nursery and to reside there using puddles and pannes as habitat, although larger juvenile and adult mummichogs appear to retreat to marsh creeks in between high tides. Yozzo and Smith (1998) found grass shrimp to be the most common nekton found in tidally flooded marsh surfaces, although there were only two other common species caught, mummichogs and blue claw crabs. Some 40 percent of the captured nekton were adults, and the numbers caught correlated with the depth of inundation. They thought the data suggested seasonal shifts in habitat usage, as species that prefer submerged aquatic vegetation (SAV) may use the marsh surface for refuge as SAV dies off in autumn. On the other hand, along with mummichogs, Hettler (1989) found that spot and pinfish (*Lagodon* rhomboids) were commonly captured from the surface of North Carolina marshes. Halpin (1997) found that mummichogs preferred marsh surface habitats to more open water, and further preferred shallower environments to those with greater flooding. They are seasonal, with more fish being found in summer.

Teal and Howes (2000) suggested that the piscivorous fish found in creeks at high tides should be considered to be estuarine rather than marsh fish. Although non-resident fish are found in the marsh during summer and gut surveys of striped bass caught in marsh creeks show them to be full of mummichogs, a low level of correlation was found in a comparison of 1880s acres of marsh area and fish landings for nearby ports. This was suggestive that marshes do not support local fisheries as almost all commercial fishers in the 1880s caught fish locally. However, there was a positive correlation between the length of the marsh edge with the estuary and fish catches. Teal and Howes thus proposed that marsh edge serve as a surrogate for the amount of marsh productivity exported to the estuary. Conversely, Deegan et al. (2000) found that the warmer temperatures, shelter, and food sources found in marsh creeks made them important for larval fish of many species. West and Zedlar (2000) noted that the intermittent access to the marsh surface may mean that food resources accumulate on the marsh, and thus fish may actively seek to forage on the marsh surface in comparison to creek or open estuary areas. Rozas and LaSalle (1990) found that Gulf killifish (*F. girardis*) had fuller guts leaving the marsh surface as compared to when they entered it, which supports the hypothesis that it is an enriched food source.

At least 50 percent of the carbon used by fish and other larger organisms in the marsh comes from benthic macroalgae production (Sullivan and Currin, 2000); Pomeroy et al. (1981) found that nearly all epi-benthic algae was grazed, mostly by fish. Wiegert et al. (1981) suggested that production of primary and secondary consumers in the marsh is limited by the amounts of usable carbon generated in the marsh (algal carbon plus grazed and decomposed *Spartina* carbon). Currin et al. (2003) found evidence to support these positions, as larval mummichogs were found to consume benthic microalgae. However, adult mummichogs were shown to be at least two trophic levels removed from algae consumption, on average. This allowed them to thrive in *Phragmites*-dominated areas where algae biomass is considerably reduced. Knieb (1997) found that most research seemed to indicate that resident fish consumed detritivores, and so were secondary consumers at best.

Teal (1986) found Atlantic silverside (*Menidia menidia*), mummichog, striped killifish (*Fundulus majalis*), sheepshead minnow (*Cyprinodon variegates*), four-spined stickleback (*Apeltes quadraucus*), three-spined stickleback (*Gasterosteus aculeatus*), and American (common) eel (*Anguilla rostrata*) as the resident fish of New England low marshes, which penetrate into the grasses of the marsh when water levels would allow. Fish using the marsh as a nursery were specified as winter flounder (*Pleuronectes americanus*), tautog (*Tautoga onitis*), sea bass (*Centropristes striata*), alewife (*Alosa Pseudoharenges*), menhaden (*Brevoortia tyrannus*), bluefish (*Pomatus saltatrix*), mullet (*Mugil spp.*), sand lance (*Ammodytes americanus*), and striped bass (*Morone saxatilis*). These fish were said to be restricted to the creeks. Dreyer and Niering (1995) specified that the fish in creeks and ditches were comprised of common mummichog, striped killifish, sheepshead minnow, and Atlantic silverside, and that young-of-the-year winter flounder may be found there. Blue crab (*Callinectes sapidus*), green crab (*Carcinus maenas*), shore shrimp (*Palaemontes* spp.), and sand shrimp (*Crangon septemspinosa*) are also resident in the creeks and ditches. Other fish, especially bluefish, fluke (*Paralichthys dentatus*) and striped bass forage on these and other fish in the nearby shallow, estuarine waters. Briggs and O'Connor (1971) found 40 species of fish adjacent to island marshes in Great South Bay. The most common were Atlantic silverside, fourspine stickleback, striped killifish, mummichog, sheepshead minnow, northern puffer (*Sphoeroides maculates*), northern pipefish (*Syngnathus fuscus*), Atlantic needlefish (*Stongylura marina*), white mullet (*M. curema*), and threespine stickleback. O'Connor and Terry (1972) reported on an unpublished study at Flax Pond that found 24 species of fish, with a very different distribution. The Flax Pond study found dominance by winter flounder, and large numbers of American eels and grubbies (*Myoxocephalus aeneus*). Able et al. (2001) conducted extensive trawl surveys of five New Jersey (Delaware Bay) deep, but still intertidal, salt marsh creeks. 40 species of fish were collected (815 tows); nearly 60 percent were classified as transient species, with 40.5 percent classified as resident fish. They noted that the use of seines might not have accurately sampled fish that do not commonly stray far from the marsh surface (a result found by Fulling et al., 1999). Species composition did not vary much across the sites, although relative abundances did. The study design was intended to determine differences between *S. alterniflora*-dominated marshes and those invaded by *Phragmites*, but no such result occurred. A similar effort by Grothues and Able (2003a), designed to determine the impacts of *Phragmites* removal, did not find a difference in adult assemblages; the authors attributed this to the relative environmental insensitivity of the more numerous transient fish species that controlled the statistical results. A small difference was determined for juvenile assemblages, but the relationship was described as "weak" (Grothues and Able, 2003b). Cooper (1974) noted that juvenile and mature fish such as flounder, bluefish, menhaden, croaker (*Micropogonias undulates*), and tarpon (*Megalops atlanticus*) had been found in Georgia marsh creeks.

Large grazing mammals, common to interior grasslands, are not found in salt marshes, but smaller ones feed and find shelter here. Dense vegetation in the high marsh provides habitat for the field mouse (*Microtonus pennsylvanicus*). It feeds on *S. alterniflora* by cutting the plant at soil level while consuming only the tender basal portion. It was found to damage 2.5 percent of the *S. alterniflora* in Great Sippewissitt marsh in this manner (Teal, 1986). The seed feeding white footed mouse (*Peromysus leucopus*), as well as other small rodents, may be as common in northern marshes as the rice rat (*Oryzomys palustris*) is in the south. One of the most conspicuous marsh residents is the muskrat (*Ondatra zibethica*), whose diet consists entirely of roots and tubers. It favors low salinity marshes with low tidal variation. Larger mammals such as rabbits and white tailed deer may occasionally feed on the marsh fringes, but are not residents (Nixon, 1982).

Most waterfowl and shorebirds eat a great variety of plant or animal matter, or both. This may be a reflection of relative food abundance at a particular time, rather than a requirement.

Mallards (*Anas platyrynchos*) capture shrimp and mummichogs to add to their diet of the macroalgae *Ruppia* and *Ulva*, which are also the mainstay of the black duck (*A. rubrides*). Canada geese may feed on *Spartina* leaves, and, in the winter, snow geese may consume large quantities of the rhizomes (Teal, 1986).

Burger (1991) stated that, generally, birds in New York salt marshes nest in different parts of the marsh: laughing gulls select *S. alterniflora*; common and Forsters terns nest in wrack-filled areas in the high marsh (as do skimmers, sometimes nesting with the terns); herring gulls nest from there up into the *Iva* layer; and herons, egrets, and ibises prefer *Phragmites* or shrubs. She did not discuss ditching as an anthropogenic problem for coastal birds in her review. Reinert and Mello (1995) generally divided habitat use between assemblages as waterfowl, gulls, and shorebirds in tidal ponds and mudflats, wading birds more abundant on the marsh than in pool habitats, and songbirds exclusively on the marsh. They further suggested that, because multiple, overlapping habitats are used by more than one assemblage, the loss of any part of the whole system could result in substantial population loss for many of the assemblages. This was asserted although the lost habitat might not constitute the primary or even secondary habitat used by any one of the bird assemblages.

Seaside and sharp-tailed sparrows (*Ammodramus spp.*) are species of concern in the northeast US due to dwindling numbers. These birds are generally considered salt marsh residents, although seaside sparrows have colonized some fresh water marshes, especially in the Hudson River valley (Post and Greenlaw, 2000). These sparrows are omnivores, with perhaps 80 percent of their diets coming from small marsh invertebrates, larger flying insects, and spiders, and the reminder from marsh grass seeds. They prefer to forage in the wetter portions of the salt marsh, although they may form loose colonies for nesting in the drier portions of the high marsh (Austin, 1983). They catch their prey by walking on the marsh substrate, climbing through the vegetation, or wading through shallow pools and pannes (Greenlaw, 1992). It is reported that changes to marsh habitats due both to impounding and ditching have resulted in decreases in numbers; however, the heavy use of DDT in the period of 1945 to 1970 nearly eliminated most seaside sparrows from East Coast salt marshes (Austin, 1983). The distribution of seaside sparrows in south shore Long Island marshes has been described as patchy. Territories are defined in terms of reproductive behaviors, such as siting nests, caring for young, and singing.

Singing requires stiff, raised clumps of plants, whereas nests need to be in or near feeding areas, have cover, and been sufficiently raised to reduce the chance of flooding. While some *Iva* or *Spartina* areas may meet both requirements, often the area used in the marsh was based on "reasonable commuting distances," in a habitat that contained all the necessary components in general proximity. This, rather than any colonial urges, is what causes the sparrows to found in loose associations. Sharp-tailed and seaside sparrows can share habitats, but the seaside is dominant when this occurs. The dominance does not affect the sharing of the overall habitat space, however (Greenlaw, 1983), although the sharp-tailed sparrow may be forced to nest in less desirable areas (Post, 1970). Breeding densities for seaside sparrows can be as high as 30 pairs per hectare (Post and Greenlaw, 1975). Seaside sparrows are unusual in north shore marshes and on the East End of Long Island (Greenlaw, 1983), and their range is from Jamaica Bay east along the south shore to Mecox Bay. They are "regularly but sparingly" sighted during mild winters, but are generally characterized as summer birds on Long Island. The population on Long Island was described as "secure" in 1992 (Greenlaw, 1992). Sharp-tailed sparrows are characterized as being common in north shore salt marshes in the summer (Greenlaw, 1983).

2.4 Secondary Consumers

Due to widespread omnivory, most marsh consumers don't fit neatly into primary and secondary consumer categories. The major fish (mummichogs), bivalve (ribbed mussel), crab (fiddler crab) and gastropod (marsh snail) species feed on both auto- and heterotrophs. Even the blue crab, a summer resident of marsh creeks, consumes submerged aquatic plants, macroalgae, and organic detritus in addition to preying upon grass shrimp, minnows, snails, and bivalves (Virstein, 1977).

By far, the most numerous predators of marsh insects are the spiders (Pfeifer and Weigert, 1981). This includes web spinners such as *Grammonata spp.*, and smaller members of the family *Clubionidae*. Wolf spiders (*Lycosidae*) are also common. They hunt using visual and tactile means, even seeking large prey such as amphipods. Mites are the dominant predators of the macroarthropod community on marsh vegetation. Pfeiffer and Weigert (1981) list predation and food scarcity as the major factors regulating population densities of these Arachnids.

Very high tides drive insects and spiders out of their cover, attracting bird species like sparrow, warblers, and wrens. Kale (1965) found marsh wrens (*Cistothorus palustries*) to exert substantial control on spider and wasp populations, consuming up to 5 percent of the mean standing crop of spiders from April to August. Nixon (1982) lists more than 20 species of birds associated with the high marsh, attributing the high diversity to the "edge effect." This comprises the convergence of the marsh to the shore ecotone, and thus shore and wading birds with the field and forest species. They may reside permanently or seasonally, attracted to fish or other aquatic species (osprey, kingfisher, herons, egrets, ibis, gulls) to flying insects (swallows, chimney swifts) or to small mammals (owls, harriers, hawks). O'Connor and Terry (1972) quoted a Department of Interior report as saying that the south shore marshes of Long Island are "the most important coastal waterfowl area in the North Atlantic states," and found studies enumerating at least 25 species of migratory waterfowl using Long Island marshes. Furthermore, 41 species of shorebirds were found to use the marshes.

The only reptile classified as a marsh resident in New England salt marshes is the diamondback terrapin (*Malaclemys terrapin*), which feeds on fish, mollusks, and crustaceans in marsh creeks (Teal, 1986). However, there have been reports that other turtles do use Long Island salt marshes as habitat, especially during winter (K. McAllister, Peconic BayKeeper, personal communication, 2004). Therefore, a special research effort was made concerning the spotted turtle (*Clemmys guttata*), Eastern mud turtle (*Kinosternon subrubrum*), and Northern diamondback terrapin (*M. terrapin terrapin*).

Spotted turtles are diurnal, semi-aquatic, primarily freshwater reptiles, which are active during the spring (Stewart and Springer-Rushia, 1998), specifically, between the months of March and May (NYSDEC, 2003a). Haxton and Berrill (2001) report that spotted turtles are active for a relatively short period of time per year as compared to most other North American turtles.

Their shells range from 3.5 to 5 inches in length (NYSDEC, 2003a). The upper shell (carapace) is black with a series of small, round, yellow spots or "polka-dots." Hatchlings have one dot per scute; however, mature turtles may have a hundred or more spots in total. The lower shell (plastron) is yellow and black and the head, legs, and tail of spotted turtles exhibit small yellow and orange spots. The male's jaws are dark in color and their eyes are brown. The female's jaws and eyes are both yellow (Stewart and Springer-Rushia, 1998).

Spotted turtles are carnivores. Their diet consists primarily of insects, snails, worms, and slugs (NYSDEC, 2003a). They commonly bask in the sun during the day and will enter the water slowly when startled. During evenings and nights, they dive to the bottom of a pond and stay there until the next day. Preferred habitats include marshes, swamps, bogs, fens, muddy streams, wet meadows, sedge meadows, ponds, and ditches that contain freshwater (Stewart and Springer-Rushia, 1998; Harding, 2004). Spotted turtles like shallow water bodies with soft bottoms that are supportive of emergent and submergent vegetation (Harding, 2004). They are known to hibernate within the water beneath mud and debris (Western New York Herpetological Society, 2004). Streams that are inhabited by spotted turtles are characteristically muddy and slow-flowing (Harding, 2004).

Spotted turtles are sexually mature between eight and 10 years of age and have life spans that range between 25 and 50 years. Breeding occurs between March and May, when the turtles are active. In May, females seek nesting areas to oviposit (lay her eggs). Nesting areas are often established in meadows, fields, or along roadsides. When a suitable nesting area is found, the female will dig a hole in the ground that is about two inches deep. She then lays three to four eggs in the hole and buries them. The eggs hatch in about 11 weeks (NYSDEC, 2003a).

Millam and Melvin (2001) found that a spotted turtle's habitat is generally 3.5 hectares (8.6 acres) with a home range length of 313 m (1,027 feet). Spotted turtle movement generally includes travel to and from vernal pools, and movement between aestivation, over-wintering, and nesting sites. They found that 25 of 26 turtles spent between 20 and 150 cumulative days per year (with a mean of 80 days) basking, foraging, and mating in seasonal pools.

Spotted turtles were once very common. However, the New York State Department of Environmental Conservation has now classified the spotted turtle as a "species of concern" in the State of New York (NYSDEC, 2003a; NYSDEC, 2003b). Their status as a species of concern was established largely as a result of ongoing widespread disturbance to their habitat, which has adversely affected their overall numbers. NYSDEC has reported occurrences of spotted turtles on eastern and east-central Long Island (NYSDEC, 2003c).

Decreases in the number of spotted turtles on Long Island can be attributed to a variety of other causes, including being killed by predators or automobile traffic. They are also vulnerable to polluted water, particularly toxic chemicals, and are sometimes taken from the wild to be kept as pets (NYSDEC, 2003a).

The carapace of the eastern mud turtle ranges from olive to dark brown to nearly black. It has eleven marginal scutes as opposed to the twelve most turtles have. The plastron ranges in color from yellow to brown. It is double hinged and also has eleven scutes. The mud turtle grows to a length of three to four inches (NYSDEC, 2003d).

Mud turtles are semi-aquatic omnivores which, in the United States, range in aerial extent from Long Island (its northernmost habitat) south to Florida, and throughout the southeastern states which border the Gulf of Mexico. The mud turtle's diet consists primarily of insects, mollusks, crustaceans, amphibians, carrion, and, occasionally, fish (USGS, 2004). It prefers shallow, still, and lushly-vegetated waters with soft beds of sand or mud, small ponds, the edges of marshes, streams, bogs, forested wetlands, ditches, wet meadows, sedge meadows, off-shore islands, and seasonal pools (NYSDEC, 2003d; Carr 1952; Ernst 1976; Ward et al., 1976; USGS, 2004). Niering (1997) also points-out that mud turtles commonly inhabit muskrat lodges and are tolerant of brackish waters. They also do well on land and may be found a great distance from water.

Mud turtles are active between the months of April and October. When a water body in its habitat dries up, mud turtles will either migrate to another area that has water or will burrow themselves into the mud to a depth of one to three feet (NYSDEC, 2003d) where they remain in a state of torpor or dormancy (Niering, 1997).

The mud turtle reaches sexual maturity between the ages of five and seven, breeds from mid-March to May, and typically nests in June (Niering, 1997). Females usually lay between one and three clutches per year. Each clutch contains between one and six hard-shelled eggs which are deposited in a well-drained soil at a depth of three to five inches, where they are typically concealed beneath some plant life or vegetative debris (USGS, 2004).

The mud turtle is an endangered species in the State of New York (NYSDEC, 2003b) and is considered the rarest of New York's turtles (NYSDEC, 2003d). On Long Island, NYSDEC has identified mud turtles in and around the lower Carmans River, as well as the lower Peconic River and Flanders Bay areas (NYSDEC, 2003c). In 1995, mud turtles were known to be in just four locations in New York, three of which were at least partially located within Suffolk County's Central Pine Barrens (CPBJPPC, 1995). The existence of predators and disturbance to localized habitats are specific factors affecting the mud turtle's overall numbers and occurrence. They are commonly seen in terrestrial environments and are often killed while crossing streets (Niering, 1997). NYSDEC (2003d) suggests that road-kills, the illegal pet trade, and the draining of wetlands and clearing for development are among the most significant causes of mud turtle mortality in the State.

The northern diamondback terrapin has a black to light-brown carapace with a slight dorsal keel and concentric, grooved markings on its diamond-shaped scutes. Its skin ranges from dark gray to white with light specks and/or streaks. Female diamondback terrapins are nearly twice as long as males, with the female's carapace reaching a length of about nine inches. The male's shell grows to a length of approximately five inches. Females also have larger and wider heads and are more ruggedly built. Additionally, diamondback terrapins are known to have proportionately larger back feet than other turtle species. Their large back feet are effective in propelling them through strongly flowing or choppy waters. The diamondback terrapin is a carnivore positioned near the top of the food chain. It eats fish, bivalves, gastropods, and crustaceans (NYSDEC and USFWS, 1998).

The northern diamondback terrapin's home range is restricted to salt and brackish waters between Cape Cod, Massachusetts, and Cape Hatteras, North Carolina, although similar varieties of diamondback terrapins are found in other parts of the country. In New York, year-round populations of northern diamondback terrapins have been identified in the estuaries and tidal marshes of Long Island, Staten Island, and along the lower Hudson River into Rockland, Putnam, and Orange Counties (Feinberg and Burke, 2003). Morreale (1992) identified 993 diamondback terrapins in various estuarine environments including bays, harbors, salt marshes, and tidal creeks. 45 sites were found in the Peconic Estuary, 10 sites were found along Long Island Sound, and 18 sites were noted along Long Island's Atlantic beaches. Population densities were highest in areas containing large salt marshes with associated tidal creeks and channels. The highest densities were found in the Cedar Creek complex, the Hubbard Creek area, Scallop Pond and West Cove Creek, and the Sag Harbor complex. Morreale also noted that younger terrapins appeared to occupy slightly different habitats than older individuals.

According to the US Fish and Wildlife Service (USFWS) (1998), diamondback terrapins have a very specific habitat requirement, which includes brackish waters that have salinity concentrations ranging between nine and 21 ppt. A tear gland which releases excess sodium from its body helps the diamondback terrapin to regulate internal salt concentrations and, thus, remain stable and survive over this range of salinity.

The number of diamondback terrapins in the area has been increasing over time since the 1960s, after having plummeted during the early 20th Century due to over-harvesting (Morreale, 1992). In 1990, the NYSDEC began requiring that persons wishing to harvest the diamondback terrapin first secure a permit to do so, although earlier reports indicate that there is generally little activity in this regard (NYSDEC and USFWS, 1998). Feinberg and Burke (2003), however, report that diamondback terrapins are still sold illegally in many major cities.

The diamondback terrapin hibernates during the winter but is active between the months of April and October. For about six weeks of the year, from mid-June to mid-July, females will come to shore to nest (NYSDEC and USFWS, 1998). Feinberg and Burke's (2003) research indicate that nesting may occur from early June through early August. Terrapins mate in the water and females nest and oviposit onshore above the high tide line (Morreale 1992). Nests are often established in areas that are within 250 m of water, are exposed to sunlight, and have sandy soils and sparse vegetation. Nests were most often established in shrubland, dune, and mixed-grassland habitats; the highest densities were found on beaches and man-made trails, which are frequently traveled and more greatly affected by human activity (Feinberg and Burke, 2003).

The eggs and newborn terrapins are vulnerable to raccoons, red foxes, gulls, crows, and ghost crabs (Feinberg and Burke, 2003; Morreale 1992). Eggs and hatchlings can also succumb to fungus, maggots, inundation by water, wind erosion, land development, motor boats and automobiles, and harvesting by humans (Feinberg and Burke, 2003; Morreale, 1992; NYSDEC and USFWS, 1998).

2.5 Mosquitoes as Prey

It should be understood that marshes are very productive sources of insects. Davis and Gray (1966) pointed out that not only are marshes sources of "noxious" insects such as mosquitoes, midges, and biting flies, but they produce a great variety and abundance of other insects, as well.

Mosquitoes serve as prey for a wide variety of predatory organisms. The adults are consumed primarily by birds, bats and dragonflies. Generally, mosquito larvae are consumed by a number of predacious aquatic insects, a wide variety of predatory fishes, and a few species of predacious mosquito larvae. Larval habitats regulate the scope of predators that prey on the immature stages of individual mosquito species; in salt water habitats, for example, insects are negligible predators on mosquito larvae, and less important for adult predation. Predators that attack the adult stage of the mosquito are usually more generalized because mosquitoes tend to disperse from the breeding habitat once the terrestrial stage is reached, and saline conditions are less of a factor.

2.5.1 Organisms that Feed on Adult Mosquitoes

A wide variety of dragonflies (Order Odonata) feed on adult mosquitoes that have recently emerged from swamps, bogs and marshes (Silsby, 2001), but only one species preys on salt water mosquitoes. The sole salt marsh dragonfly in the northeast US is the libellulid species, *Erythrodiplaz bernice* (Frank W. Carle, Rutgers University, personal communication). Dragonflies catch their prey on the wing using a basket-like apparatus that they form with their legs (Borrer et al., 1981). The prey of most species includes small flying insects, like midges and mosquitoes, but larger dragonflies often capture and consume bees, butterflies and, occasionally, other dragonflies. Small insects are consumed while in flight; most species land on vegetation before they attempt to consume larger insects. Salt marsh mosquitoes often attract large numbers of dragonflies immediately after a lunar tide has produced a fresh emergence. Foraging behavior is most intense at twilight because floodwater mosquitoes rest in vegetation during daylight hours, but exhibit increased activity during the dusk and dawn photoperiods. Predation by dragonflies in saltmarsh areas is more difficult to observe after the brood has dispersed. The absence of large concentrations of mosquito prey force resident dragonfly populations to shift to other food sources between the twice-monthly broods created by the lunar cycle, and/or vacate areas that previously provided excellent foraging opportunities. Most studies of dragonfly predation on mosquitoes have been with permanent swamp mosquitoes and the dragonfly species found around swamps and bogs (Kumar, 1984; Chowdhury and Rahman, 1984), and so may not be applicable to salt marsh conditions.

A variety of insectivorous birds also feed on adult mosquitoes and often take insects on the wing. Phoebes, flycatchers, and related insectivorous species feed by "hawking" (Sibley, 2001), which limits the numbers of mosquitoes in their diets. Hawking involves swooping down from a perch to glean individual insects as they are spotted, and rarely includes large numbers of insects as small as mosquitoes. The swallows and martins employ an opportunistic feeding strategy and forage far and wide for ephemeral food sources, which they take directly on the wing. When mosquito populations are numerous (*i.e.*, after a major floodwater emergence) these birds are frequently seen flying back and forth over open areas taking in large numbers of aerial insects. Crepuscular mosquito behavior severely limits the time frame for energy efficient foraging with these birds. In addition, these opportunistic insect feeders must switch to a more available food source as soon as the brood has dispersed.

Purple martins (*Progne subis*) have been credited with exaggerated claims on the numbers of mosquitoes they consume. Kale (1968) provides strong arguments that refute popular claims for using purple martins for mosquito control. Almost all stomach content data show the birds eat other, larger insects; in addition, the standard foraging behavior is to fly above open fields from 50 to 200 feet above the ground; mosquitoes stay low in open country and almost never fly above tree canopies. Almost all statements concerning purple martin predation on mosquitoes prior to 1968 were either unverified reports of others' observations, or speculation based upon general behaviors. For example, the claim that martins eat 2,000 mosquitoes a day was based upon speculation by Wade (1966, cited in Kale) that a high metabolism bird such as a martin eats its own body weight in insects each day, and that the average martin weighs 4 ounces, which is the equivalent of approximately 14,000 mosquitoes. Therefore, on the assumption that a martin in an area infested by mosquitoes might feed solely on mosquitoes, the claimant felt using an estimate of 2,000 consumed per day would be conservative. Kale refuted each assumption, based on actual data. Therefore, it is not credible to assert the purple martin is a major predator of mosquitoes.

Bats, insectivorous mammals that include mosquitoes in their diet, are another cause for controversy regarding exaggerated potential for mosquito control. The original argument dates back to investigations using bat towers by Campbell (1907), Storer (1926), and Allen (1939). In all cases, the hypotheses appeared sound, but failed to produce the desired results. Numerous claims regarding the numbers of mosquitoes that one bat can eat in an hour routinely appear in popular articles and web sites (e.g., McAvoy, 2004). Claims vary from 300 to 600 per hour, and are usually then re-calculated to reveal the number of mosquitoes each bat consumes every night, often from restricted areas (such as one backyard). Most figures are based on sonar studies where bats were purposely released in a room filled with mosquitoes to compare the efficiency of sonar-directed foraging versus random catch. The stomachs of these experimental animals provide the data cited by bat enthusiasts on the numbers of mosquitoes consumed by bats per hour. Corrigan (1997), following Masters Thesis work on the subject, has written a number of non-technical articles on the feeding behavior of bats as well as the mosquito-bat controversy. His research indicates that the little brown bat (*Myotis lucifragus*) prefers small, soft-bodied insects and does include mosquitoes in its diet. Larger bats such as the big brown bat (*Eptesicus fuscus*) are more opportunistic and prefer beetles and moths because of their size. Bat feeding strategies, however, are to consume as much food as they can in the first hour and either rest or return to the roost. Most of the figures for bat predation are based on the mistaken assumption that bats continue peak feeding efficiencies throughout the night. Corrigan's research also indicated that bats often tend to feed in areas where electric lights attract large numbers of insects. As a result, they frequently feed selectively on moths and beetles at a single focal point rather than combing a wide area, or away from lights where mosquitoes are more likely to be.

Most of the work conducted on animals that prey upon adult mosquitoes indicates that mosquitoes can be valuable for some. Each of these animals tends to only eliminate part of the usually large number of organisms associated with a mosquito brood, however. Managing one or more mosquitovorous animals as a mechanism to eliminate mosquitoes is improbable. However, increasing numbers of these predators can contribute to the overall control of any pest problem.

2.5.2 Organisms that Feed on Mosquito Larvae

A wide variety of predacious insects feed on mosquito larvae during their aquatic developmental stages in fresh water environments. However, floodwater mosquito species, in general, live in transient water habitats and undergo rapid development which significantly reduces opportunities for insect predators to utilize them as a food source. Salt marsh floodwater supports even fewer insect predators because of the limiting factors that high salinity poses.

Predacious fish are probably the most efficient predators of mosquito larvae and have the ability to completely control mosquito larvae if managed properly (Gerberich, 1985, Haas and Pal, 1984). Salt marsh mosquito producing habitats in the northeastern United States have large populations of mummichogs, which can be managed to function as a voracious predator of *Ochlerotatus sollicitans* and other salt marsh mosquito species. Salt marsh mosquitoes generally lay their eggs on areas of high marsh dominated by *S. patens*, which can grow so closely that it tends to screen mummichogs from the larvae that develop in depressions on the high marsh. The fish can gain access to breeding depressions during lunar floodings by swimming over the grasses, but become stranded and die when the tide subsides and the pools dry down. This allows the eggs to develop into larvae during the next flooding period if the fish cannot reach the pools again. The practice of OMWM was devised to minimize mosquito breeding and provide refuge for predacious fish as a supplement to water management for mosquito control purposes (Ferrigno and Jobbins, 1968). OMWM, in all of its many variations, is intended to improve fish habitat, especially by creating refuges for low water periods, and to improve access for mummichogs and other marsh-resident mosquitovores to breeding sites.

However, mosquitovorous fish are not dependent on mosquitoes for sustenance. A study in Mississippi examined gut contents of killifish before and after the fish went onto the marsh surface. Rozas and LaSalle (1990) did not find enough mosquito larvae in the fish for larvae to make the list of primary prey. Fiddler crabs, and amphipods were the dominant prey, along with tanaidaceans and hydrobiids (polychaetes).

However, under circumstances where mosquito larvae are extremely abundant, certain marsh surface fish were found to feed exclusively on mosquitoe larvae. A high marsh in Florida was flooded at the time that a brood of mosquitoes was almost ready to hatch, providing general access for fish to the breeding points. Harrington and Harrington (1961) found that mosquito larvae can constitute between 50 and 90 percent of some species' diets when abundant. The fishes revert to other, various food sources that include vegetative matter and detritus, copepods, other fish, or other insects and insect larvae, depending on the species, when mosquitoes are not available. Because mosquitoes tended not to be abundant most of the time in this marsh, insects, in general, constituted only two percent of the fish prey by mass for the overall study, in which a total of 16 fish species were sampled.

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3. Impacts of Historic Ditching and Standard Water Management

Salt marshes have been altered since pre-Columbian time in Connecticut, at least, when Native Americans may have burned wetlands at frequent intervals-+*. Salt marshes have been mown for cattle fodder from the 1600s (Miller and Egler, 1950). From the 1700s onwards, ditches have been installed in salt marshes in the US (Shisler, 1990). Originally this practice was adopted by farmers to increase the acreage of *S. patens* (used as salt hay) produced by the marsh (Daiber, 1986), although the acreage used in these efforts was extremely limited (Bourn and Cottam, 1950).

It is commonly asserted that ditches promote drainage of tidal flooding. They are thought to decrease the water table in the marsh, and to increase soil salinities. Most observers find an expansion of *S. patens* acreage, usually at the expense of low marsh *S. alterniflora*. By 1900, some 50 percent of Connecticut's salt marshes had been ditched (Dreyer and Niering, 1995).

Discovery of the mosquito as the vector for malaria and yellow fever allowed for control of those diseases through aggressive mosquito control, beginning about the turn of the century (Glasgow, 1938). The lessons learned in Panama and Cuba concerning water management and chemical control of mosquitoes were imported into the US. The first ditches constructed on Long Island for mosquito control were installed in 1900 in Lloyd Harbor, and the effort was expanded there and on Centre Island in 1901. The results at these locations led New York City to begin ditching in 1903 (Richards, 1938), although the first extensive use of ditching for mosquito control is often said to have been in New Jersey (Provost, 1977). Between 1912 and 1938, 90 percent of the salt marshes between Maine and Virginia were ditched, with most of the effort occurring after 1932 (Bourn and Cottam, 1950). For Long Island, 1880s-era maps accessed by the New York State Department of State do show ditches in salt water wetlands along the south shore, possibly for hay production. After the turn of the century, large estates on the south shore apparently were the first areas to be routinely ditched, with the Seatuck National Wildlife Refuge marsh being ditched between 1915 and 1920 (Cowan et al., 1986). In the 1930s, the Works Progress Agency (WPA), the Civilian Conservation Corps (CCC), and other federal, state, and local public works successors employed thousands on Long Island to expand and enhance the ditch network in Long Island's salt and fresh water wetlands. Glasgow (1938) noted that concerns were raised at the time that

> "harm to wildlife habitats might result from hastily organized or inadequately supervised mosquito control work."

The systems of parallel, and sometimes grid, ditching that resulted affected nearly all of Suffolk County's salt water wetlands.

Ditching and maintenance of existing ditches have become known as "standard water management" in mosquito control terminology. Ditch maintenance is the practice of excavating slumps and in-filled material. This is done for the expressed purpose of allowing the ditches to continue to drain the tidal inflows. Anecdotal information suggests that when ditches are not maintained, mosquito populations burgeon. This, together with other anecdotal information from when ditches were first installed, suggests that ditching can be effective at reducing mosquito breeding.

Ditches are thought to control mosquitoes for two contradictory reasons. The most common statement is that they draw down the water table under the marsh, thus preventing standing water from persisting on the surface of the marsh long enough for salt marsh mosquitoes to complete its maturation from egg to larvae to adult, which must occur in an aqueous environment (Dale and Hulsman, 1990; Kennish, 2001). The contradictory assertion is that ditches propagate tidal flows into the marsh, which means the high marsh floods too often to produce broods (Provost, 1977). This reduces mosquito development by preventing the curing of eggs, and, also, perhaps, by preventing the formation of isolated pools. Under both theories, ditches are also said to allow access for fish to the center of marshes they might otherwise not have been able to reach, and supporting additional fish predation on larvae.

Marsh ditches are believed to have environmental consequences. Most of the changes that are associated with ditches are now evaluated as being negative in character. As a result, most environmental planning efforts that address coastlines call for "marsh restoration," and many of these plans intend that ditching be undone, in one fashion or another (Niedowski, 2000). On the other hand, major explorations of modern marsh issues (Weinstein and Kreeger, 2000; Mitsch and Grosselink, 2000) do not specifically explore ditching and its impacts, and only discuss ditching briefly in the context of other topics; one major ecological study of low marshes in the northeast US never specifically discussed ditching (Teal, 1986); and its companion study of high marshes covered ditching rather cursorily (Nixon, 1982).

The literature search on these issues found conflicting information. Solid, basic studies were not uncovered on many topics of interest, including comprehensive assessments of the effectiveness of traditional water management for mosquito control, and descriptions of the environmental effects of installing ditches in a marsh. Nixon (1982) dismissed most studies of ecological impacts of ditching as containing only "casual impressions and anecdotal information … reflect[ing] the biases of 'mosquito controllers' or conservationists." These factors limit the utility of the discussion that follows.

3.1 Basis of Comparison

The impacts of ditching can be described in a comparative way, but the basis of comparison can be difficult to determine.

3.1.1 Unditched Wetlands

The clearest means of determining impacts is to compare a ditched wetland and a pristine, or unditched, wetland. Differences between the two could then be ascribed to the impacts of ditching. However, there are few wetlands in the northeast US that have not been managed in some way or another. Direct comparisons have been made in some studies:

- Bourn and Cottam (1950) in Delaware
- Redfield (1972) in Massachusetts
- Kuenzler and Marshall (1973) in North Carolina
- Lesser et al. (1976) in Delaware
- Clarke et al. (1984) in Massachusetts
- Lathrop et al. (2000) in New Jersey
- Markus (2003) in Maine.

However, even in these cases, there are likely to be complicating factors such as differences in tidal regimes, surrounding land use or settings, etc., that might also play a role in the comparison (see Provost, 1977, for many such examples). There are a few studies that directly describe the impacts of installing ditches (e.g., Bourn and Cottam, 1950); these tend to either be old, limited in the timeframe used to determine impacts, or may only be of local interest due to specific geographic factors that appeared to drive the results. The Bourn and Cottam paper, in particular, has been the subject of much discussion as to its accuracy and relevance.

There have been some attempts to describe a generalized northeast US saltwater wetland (e.g., Nixon, 1982; Teal, 1986; also, see Wiegert and Freeman, 1990). These descriptions create an ideal to which existing wetlands can be compared. Ways that ditched wetlands vary from these models may illustrate the ways that the ditches have impacted the wetlands.

3.1.2 Comparative Studies

Comparisons can be made between ditched marshes and those that have been restored to one degree or another. These are probably the most common discussions of the impacts of ditched wetlands (examples include Wolfe, 1996; Lathrop et al., 2000 [three-way comparison of ditched, pristine, and OMWM marsh segments]; Dale and Knight, unpublished). However, most of these describe the impacts of ditching in a reverse sense, as they tend to describe the salt marshes in terms of factors that change through the removal of ditches. "BACI" (before-after, control-impact) experimental designs, where pre-alteration data and control site information can be compared to the treatment site data, are somewhat rare in marsh water management literature (Dale and Hulsman, 1990; Crosland, 1974). This is because much of the field is engineering driven, where projects are judged on a success-failure basis rather than being science-driven, where the intent is to describe the effects of the manipulation.

3.2 Grid Ditching vs. Parallel Ditching

Two kinds of traditional water management are possible, parallel ditching and grid ditching. Parallel ditching is characterized by the ditches running in one direction, generally, from the upland to the shoreline, with relatively constant distances between the individual excavations. This creates panels of vegetation separated by the waterways. Grid ditching requires crosscutting the ditches, creating a grid of vegetation islands, and is sometimes called "checkerboarding". This second method was used initially in larger marshes until it was found that parallel ditching was as effective as grid ditching, and required less maintenance (Richards, 1938). Grid ditching is assumed to have greater environmental impacts than parallel ditching. Most comparative studies do not differentiate between grid and parallel ditching.

In either case, typically, steep-sided ditches were installed, that were up to three feet deep and two to eight feet wide. Distances between ditches were commonly 100 to 300 feet, with the choice being made on the basis of soil permeability (Dale and Hulsman, 1990). It has been asserted that the massive ditching effort of the 1930s, although putatively intended to control mosquitoes, was not focused on areas of high mosquito productivity but was simply intended as a "make-work" program (Bourn and Cottam, 1950; Provost, 1977; Daiber, 1986). Others, especially those writing about and involved in ditching efforts in the 1930s, believed the work was justified (Corkran, 1938; Richards, 1938; Taylor, 1938).

3.3 Gross Changes Theses

Ditching is generally assumed to alter the marsh in four easily detectable ways:

1) to reduce the amount of mosquito breeding;

2) to reduce the water table within the marsh;

3) to change the vegetation distribution in the marsh

4) to change the use of the marsh by important species guilds.

3.3.1 Ditches are Effective Means of Mosquito Control

Generally, it is though mosquitoes are, or would be, produced in large numbers in most unmanaged, East Coast salt marshes. Chapman (1974) states that "wild" salt marshes support large populations of mosquitoes. Daiber (1986) cited turn of the 20th Century authors who claimed that salt marsh mosquitoes caused so much discomfort they hindered the development of areas near breeding locations in the absence of control measures.

Bourn and Cottam (1950) found that draining marshes can have impacts on mosquitoes, but that grid ditching is too indiscriminant. Impacts include affecting *S. alterniflora*, where no mosquitoes hatch because tidal inundations are too frequent, and draining ponds where fish formerly had controlled mosquito breeding. In Florida, where evenly-spaced parallel ditches were installed beginning in the 1920s, mosquito breeding was found to continue unabated between the ditches (Carlson et al., 1991). Portnoy (1984) found the combination of diking and ditching for drainage purposes appeared to result in more mosquito production in salt marshes in Cape Cod, Massachusetts. Marshes ditched in North Carolina were found to breed as many *Anopheles* and *Culex* mosquitoes as unditched marshes despite having somewhat longer dry periods, probably because the frequency of flooding (14 to 21 times per month) was unchanged after ditching. However, where the marsh was steep enough, ditching seemed to promote drainage, and so *Aedes* breeding was reduced in some cases (LaSalle and Knight, 1973). Shisler (1973) and Kuenzler and Marshall (1973) both noted that allowing ditch spoils piles to remain could allow mosquito breeding pools to develop due to decreased percolation caused by marsh compression and ponding behind the elevated piles. Nixon's (1982) judgment was that ditching was of "questionable value" for mosquito control. Cowan (1985), in an unpublished thesis, found intense mosquito breeding in a ditched marsh with restricted tidal access. He measured six environmental features, and found that marsh elevation and distance from the culvert allowing tidal ingress were the best predictors of breeding. He suggested these two factors are surrogates for a direct measurement of the number of flooding events experienced at those plots. Cowan et al. (1986) note that ditching may have been effective at first by removing surface waters from the Seatuck marsh. However, the non-specific placement of the ditches with regard to breeding areas resulted in only patchy effectiveness, at best. Additionally, due to spoils placement on the marsh with subsequent ponding and vegetation changes, and poor maintenance of the ditches and culverts leading to decreased circulation and drainage, the ditching process actually resulted in increases in breeding areas. Daiber (1986) cited numerous reports from Delaware in the 1940s and 1950s that found mosquito breeding was unaffected by ditching, but also found several reports from the same areas discussing how ditching reduced mosquito numbers. Wolfe (1996) noted that much indiscriminate ditching was done in areas that probably did not support mosquito production. Richards (1938), while generally not finding many breeding locations in ditched salt marshes, did report some; and he also found extensive breeding at the edge of ditched marshes where fill had been poorly graded onto the marsh.

Nonetheless, the greater consensus appears to be that ditching can be highly effective. Taylor (1938) noted that the use of filled marshes for residential development would have been impossible prior to the construction of "millions of feet" of ditches in Nassau County, leading to "the very ancient curse of Long Island [being] now well under control." He continued,

> "the effectiveness of the ditches in controlling mosquitoes is so overwhelming … there seems to be no reason to oppose the ditching of all salt marshes."

He attributed most of the impact to greater access by killifish to areas of the marsh that breed mosquitoes. Daigh et al. (1938) reported there had been a "substantial reduction in the prevalence" of mosquitoes five years after ditching had begun in Delaware. Provost (1977) notes that salt marsh mosquito species thrive under specific tidal flooding circumstances – no more than four tidal floods per month. This, he asserts, means that ditching specific habitat areas of the marsh can be extremely effective in reducing breeding by allowing propagation of tides up into these habitats. Glasgow (1938), Cooper (1974), and Dale and Hulsman (1990) noted that ditching should impact mosquito breeding negatively by draining the wetlands, since that removes three essential features required by salt marsh mosquitoes:

1) damp conditions that support oviposition
2) wet conditions that are required for hatching
3) the small, isolated, sustained ponds required for larval development

Glasgow further noted that ditching provides additional fish access to the interior of marshes, but cautioned that the effectiveness of the technique in high marshes may require additional study. Dale and Hulsman also noted that ditching should enhance dry conditions required for egg conditioning, which could increase mosquito breeding. Dreyer and Niering (1995) reported that ditching in Connecticut removed the intermittent pools and salt pannes that were preferred mosquito breeding habitats. A study in wetlands in Petaluma, California, found that constructing ditches to mosquito-producing ponds eliminated breeding. This was not believed to be the result of increased fish access, but rather due to increased tidal flows to the ponds along the ditches, preventing egg-laying on dry shoreline by the species of interest (Balling and Resh, 1983a). Decreases in oviposition sites due to increased flooding of isolated pools is also cited as a benefit of runnelling (runnels are ditches that are 30 cm or less in depth that connect open water to mosquito breeding pools within a marsh) (Hulsman et al., 1989; Dale et al., 1993). General, overall inundation of marshes through impoundments also decrease oviposition site availability (Clements and Rogers, 1964). A comparison of a runnelled site and an unditched site found that the criteria for nuisance mosquito control spraying was exceeded 38 times in the unditched area, but only four times in the runnelled area (Dale et al., 1993). A comparison between a 13-year old runnelled site, a six-year old OMWM site, and a two-year old grid-ditched site in Australia found few larvae at any of the three sites (Dale and Knight, unpublished). An analysis of mosquito habitat formation in California found that tall forms of vegetation grow along natural channels, fall over and bridge the channels at narrower sections. The vegetation bridges trap sediments and debris, and fill from beneath. This creates isolated puddles and ponds, also called "potholes" that can seasonally dry out, forming "channel pans." Channel pans are described as the predominant form of mosquito habitat in the Bay area. Mosquito ditches are too narrow to maintain pothole formation, but fill more evenly throughout their length, and so do not generate mosquito habitat as they fill (Collins et al., 1986). Redfield (1972) also described pothole development at the heads of marsh creeks, although not in as much detail.

On the other hand, ditching may expand the extent of high marsh (see below). High marsh is generally a preferred mosquito breeding habitat. High marsh tends to be tussocky and uneven, and may be pocked with rotten spots (Nixon, 1982). These depressions and uneven surfaces can create intermittent or unconnected pools to support mosquito breeding. Chapman (1974) states that mosquito ditches promote rotten spots by inducing waterlogging of vegetation in areas that have few natural creeks.

Ditching, even if effective for mosquito control, appears to increase biting flies populations (Daiber, 1986).

3.3.2 Ditches Result in Water Table Changes

Daigh et al. (1938) attributed the decrease in mosquitoes following ditching to "draining of the marshes." However, they note the actual decline in the water table depends on the rate of water movement associated with each tidal recession and the rate of absorption or permeability of the substrate. They also noted that various experts disagreed, with some determining ditching lowered water tables, while others determined it did not. Their research showed that water levels tend not to vary with tides, except near streams and ditches. They determined that ditches, generally, lower water tables and so change vegetation patterns, since *S. patens* does very well with lowered water tables. Cory and Crosthwait (1939) found that water levels declined in high marsh areas described as muskrat fodder following ditching, and attributed it to the high porosity of the soils. Bourn and Cottam (1950) reported reductions in the water table height when comparing a ditched marsh to an unditched marsh, noting the unditched marsh froze in winter but the ditched areas did not, (presumably because the water had all drained away. Dowhan and Lagna (1971) asserted that ditches on the bayside of Fire Island drained the freshwater table on the barrier island. Redfield (1972) said that ditches drain the marsh, but did not find differences in the number of ponds in ditched and unditched areas of the marsh. LaSalle and Knight (1973) found ditched marshes had longer dry periods than unditched marshes, although the frequency of flooding remained the same, suggesting that the ditches remove water off the marsh with greater efficiency than would otherwise occur. Cooper (1974), citing Bourn and Cottam, noted that ditches dry out high marsh, and Cowan et al. (1986) echo the claim. A study of two geographically-close salt marshes on Long Island found that the unditched marsh had more than 16 percent "deep pools and ditches," and the ditched marsh had less than 11 percent (Merriam, 1983).

Daiber (1986), in a review, states the reason ditches were installed is that they dry out breeding areas. The rate of water table depression is said to be controlled by soil type and ditch spacing. A study in Petaluma, California, found that ditches there (60 cm deep) depressed the water table for a distance of approximately three m from the edge of the ditch. This effect was comparable to that caused by a natural channel (Balling and Resh, 1982). Chapman (1974) also found that natural channels depress the water table locally, with the effect being dependent on the size of the creek. However, he thought the American concept that mosquito ditches spaced 100 feet apart could "drain" a marsh "optimistic." Daiber (1986), Wolfe (1996), and Niedowski (2000) all state in their reviews that ditching reduces water tables in the marsh.

A longer-term, 6.5 years, study of runnelling found that it seemed to initially increase the water table, but that after year four the water table appeared to undergo fluctuations up and down. It was hypothesized that increased tidal flooding into the marsh made the areas affected by runnelling more similar to marsh areas closer to the tidal source (Dale et al., 1993). A study of a recently grid-ditched marsh in Australia found the water table height was higher in the treated portions of the marsh than the untreated portions (Dale and Knight, unpublished), apparently an artifact of a nearby upland water source.

Other studies find no such impacts, or at least limit the effect of ditching on the water table. Corkran (1938) reported no change in the number of ponds on the marsh surface following ditching, although groundwater was less salty. After an extended droughty period, water table heights were greater near the ditches than away from them. Taylor (1938) found ditches drain sheet water from the marsh surface, but have only slight water table impacts. He called it "concentrating the water of the marsh in the ditches," and denied it was "draining" the marshes. The data were generated through open pits, however, and Chapman (1974) criticizes this method because flooding tides can fill the pits, and, so, they may not reflect the actual water table. An unpublished thesis by Crosland (1974) found no variation in water table heights in areas with open ditches or dammed ditches on Fire Island. Provost (1977) stated that water table impacts were limited to within a few feet of the actual ditch, especially for the high marsh, as percolation is related to porosity, and as the peaty high marsh does not drain very well. However, impacts in low marshes, where the substrate is sandier, may be greater. Hemond and Fifield (1982) thought that seepage in the marsh peat is negligible except in the close vicinity of creeks. Nuttle (1988) found that water loss to creeks from the marsh was limited to the area within 10 to 15 m of the bank.

3.3.3 Ditches Result in Vegetation Changes

3.3.3.1 High Marsh

Many studies have found that ditching increased the amount of high marsh plants, particularly woody plants along the margins of the ditches. This generally has been attributed to the spoils piled along the ditches, although some also attribute the impact to decreases in water table height (Daigh et al., 1938; Daigh and Stearns, 1939; Miller and Egler, 1950; Kuenzler and Marshall, 1973; Chapman, 1974; Cooper, 1974; Clarke et al., 1984). Note that Dunton et al. (2001) found a similar change in vegetation zonation in a Texas marsh in response to elevated precipitation, which decreased water table salinities substantially.

Daigh et al. (1938) found that *S. patens* replaced *S. alterniflora*, and attributed the change to the lowered water table. Bourn and Cottam (1950), after mapping plants in relation to height above sea level pre-ditching, found that a marsh that was 90 percent *S. alterniflora* was replaced by saltmarsh fleabane (*Pluchea camphorata*), which in turn was replaced by groundsel bush (*Baccharis halimifolia*) with some *Iva frutescens* as the ditches deteriorated, although some *S. alterniflora* reestablished itself at the upper ends of ditches where water pooled. Miller and Egler (1950) cite Taylor (1938), who found no change in vegetation or water table height due to ditching, but disagree, suggesting that the drainage of surface water leads to vegetation changes. They note the presence of a “turf line” from ditch construction and the development of natural berms along the ditches leads to the pooled water on the marsh, and that these are in conflict with the intention of ditching, which was to drain standing water. They note that *Iva* is essentially unknown in the open marsh except where ditches have been installed. Redfield (1972) found *S.*

patens replacing *S. alterniflora* along creek and ditch banks and "where the marsh is sufficiently well drained." However, this Massachusetts marsh was still 60 percent *S. alterniflora*. Chapman (1974) found *S. alterniflora* replaced by *S. patens*, and attributed this to a lack of waterlogging following ditch construction. He noted that ditching seems to promote the growth of high marsh plants lower in the marsh, and so concluded that tidal inundations do not entirely control plant distributions. A study of two salt marshes near Fire Island Inlet (Long Island) found that the ditched marsh had less than 25 percent unmixed *S. alterniflora* stands, but that a nearby unditched marsh was nearly 50 percent unmixed *S. alterniflora*. *S. patens* and mixed grass stands were much more common in the ditched marsh (Merriam, 1983). Clarke et al. (1984) found that ditching changed the characteristic plants of panne areas from short-form *S. alterniflora* to *S. patens*. Daiber (1986), in a review, assessed the literature as generally finding that ditching caused the replacement of *S. alterniflora* with "less desirable" plants of the high marsh. Weinstein et al. (2000) also claim after "extended" time periods of water management, high marsh species are artificially sustained at low marsh elevations. In an unpublished thesis, Markus (2003) explored the dynamics of vegetation patterns in two Maine marshes. She compared ditched areas with natural channels, and found the most striking difference between ditch and unditched areas, when elevation above sea level was included as a factor of the analysis, was the presence of many more minor species along the ditches than along natural channels. The vegetation diversity was statistically significantly different. She also found a correlation in vegetation types between areas where ditches were enlarging and those that were infilling. The vegetation in the enlarging ditches was similar to that found in natural creeks. Portnoy (1984) found that the combination of diking and ditching succeeded in draining the marshland, and the impact was best demonstrated by the succession of woody shrubs and upland trees out onto former marsh areas. Provost (1977) found that ditching increased productivity in the high marsh, possibly through greater transfer of off-shore energy by greater tidal inflows. He noted that this is the reason for the extraordinary productivity of low marshes, according to Odum (originally, 1961; updated in 2000).

On the contrary, in a southern marsh, ditching seemed to expand low marsh at the expense of dominant high marsh vegetation, perhaps due to greater tidal circulation (Travis et al., 1954). Miller and Egler (1950) also note that *S. alterniflora* will colonize ditch borders if maintenance does not occur. Rockel (1969), because of slumping in the near vicinity of ditches, found that tall form *S. alterniflora* should increase its area. O'Connor and Terry (1972) noted that the overall composition of marshes studied in 1938 by Taylor shifted from *S. patens* dominance to a much higher percentage of *S. alterniflora* in their surveys. This was attributed to preferential filling leading to loss of high marsh, rather than long-term changes brought about by mosquito ditching. In fact, they did not mention the ditches. Niering and Warren (1980) mapped an expansion of tall-form *S. alterniflora* at ditches nearer to the estuary, but also found more woody plants dominating ditch edges in the high marsh. Kennish (2001) asserted that ditching increases tidal penetration into the marsh, and, therefore, increases the amount of low marsh forms found in the interior of the marsh. Bertness et al. (2002) found an expansion of *S. alterniflora* in general in salt marshes, but thought this was due to not to ditching but nitrate inputs associated with development. These conditions made *S. alterniflora* competitive with *S. patens*, which under normal nutrient conditions is competitively favored.

Collins et al. (1986) found in California that ditching promoted the growth of tall vegetative forms along the ditches, although these were not described further. Resh and Balling (1983) found greater productivity, but unchanged plant forms, in a California marsh with shallow ditches, which they attributed to greater flushing, said to reduce negative impacts of high soil salinities and an elevated water table. Steever et al. (1976) found that productivity of *S. alterniflora* correlated with tidal ranges on Long Island Sound, which may support this notion.

Nixon (1982) quoted a mid-1700s report on an "unprofitable" attempt to convert a fresh water marsh to high marsh by installing a four-foot wide ditch from the bay to the swamp. In contrast, Heuser et al. (1975) report ditching in Flax Pond, Long Island, helped to maintain salt marsh vegetation in an artificially-created salt marsh by keeping salinities near estuarine levels. Taylor (1938) found no impacts to "the fundamental distribution" of the four major marsh plant species: *S. alterniflora*, *S. patens*, *J. gerardi*, and *Distichlis*. Taylor thought this was due to the lack of change in the height or salinity of the water table under the marsh. He found tat some of the minor species of high marsh plants appearedo expand their ranges on spoil piles near the ditch edges, but, with the exception of *Iva*, the changes appeared to be "seasonal." A two-year old grid ditched site in Australia had no vegetation changes (Dale and Knight, unpublished).

Bourn and Cottam (1950) may be the most widely cited source for the claim that ditching has major impacts on vegetation distributions. Provost (1977) asserts that the impacts measured by Bourn and Cottam were due to dredging of the nearby tidal creek, not the installation of ditches. Lesser et al. (1976) found that the marsh had reverted by 1974 back to a pre-ditch vegetation pattern, which was hypothesized to have occurred because the ditches provide more nutrient transport than would have otherwise occurred, due to greater tidal flooding. They noted that any impacts assigned to ditching were clearly confounded by the contemporaneous dredging of the river channel.

Wolfe (1996) summarized the effects as resulting in initial changes in vegetation, and that productivity of the marsh as a whole may increase over a long period of time following the installation of ditches.

Other controls on vegetation patterns and productivity, as tempered by geography, may be the reason for the differences in impacts associated with ditching. Clark (1986), in a study of several centuries of vegetation history at William Floyd marsh, Long Island, by analysis of pollen from cores, found that physical changes to the environment caused fluctuations in vegetation on the high marsh. The primary cause was the opening and closing of Moriches Inlet, but also some shorter time scale disturbances, assumed to be storm events, had impacts. Although ditching in the 1930s was mentioned, no impacts from the ditches were reported. Similarly, Orson and Howes (1992) reported that vegetation changes in marshes with restricted estuarine exchange appeared to be driven by dramatic physical forces such as major storms, floods, and the like, that perturbed the system. Contrary to the finding for the restricted marshes, they found marshes that were open to the estuary to be controlled by biotic factors (i.e., ecological competition). In either case, vegetation changes generally appeared to be persistent, on the order of hundreds to thousands of years). The vegetation persistence was affected by the factors influencing competition, or the time between perturbation events, depending on the connection to the surrounding estuary.

3.3.3.2 More Phragmites

There is some evidence that ditching can lead to increased opportunities for *Phragmites* invasion. Orson et al. (1987) attributed uplands development as the general cause of *Phragmites* spread in a Connecticut salt marsh. Although it was noted that the marsh was ditched, no timeframe for the ditching was discussed, nor was the ditching linked to the *Phragmites* spread. Taylor (1938) classified *Phragmites* as a "secondary species," and noted that it had "captured" a few ditches near Merrick in his study of Long Island ditching. Taylor noted that it preferred dry sands, but grew in water ranging from fresh to nearly sea water salinities. Bart and Hartman (2000) mapped *Phragmites* in a New Jersey marsh and found a correlation between ditches and the areas invaded by *Phragmites*, due, in part, to the drier nature of ditch banks, but also to the lack of soil sulfides. Sometimes, this relationship was fostered: Bart and Hartman (2002) quoted Headlee (1945), who noted that *Phragmites* was planted along ditches in the New Jersey Meadowlands to stabilize the ditch banks. Phillips (1987), discussing a seemingly inexplicable *Phragmites* invasion of what was described as relatively undisturbed (e.g., lower number of habitations) Delaware shoreline, also described extensive mosquito control ditches and dikes there to support salt haying, and did not account for these potential causes. At two marshes on Long Island, one ditched and the other not, *Phragmites* was measured as accounting for 15 percent of cover at the ditched marsh and 10 percent at the unditched marsh (Merriam, 1983). Along the brackish Connecticut River, Warren et al. (2001) described how, at higher salinities, creek and ditch banks provided locations for *Phragmites* to establish a foothold, and that it was then able to expand away from the edge habitats. The time period of inundation distinguished *Spartina* and *Typha* stands from *Phragmites* areas, as both the wettest and driest areas resisted invasion.

Balling and Resh (1983b) found that ditching Petaluma, California, marshes enhanced the growth of plants near the ditches primarily by reducing salinities of groundwater and soils by enhancing flushing. Dale et al (1993) found that, overall, soil salinities were reduced following runnelling. Collins et al. (1986) described how the reduction in tidal heights in channels, generally due to ditching, can lead to freshening of environments furthest from the estuary. The theory is that the tidal heights decrease because the area of channels increases, but the volume of water transmitted to the marsh by similar tides remains constant. They noted that since species are often very sensitive to small changes in salinity where salinities are the lowest, these differences could lead to dramatic changes in plant distribution. Any excavations in *Phragmites* stands, as might occur through ditch construction of installation, has the potential to spread

Phragmites through rhizome dispersal (Bart and Hartman, 2003), depending how the spoils are managed.

On the other hand, it has also been thought that higher salinities can control *Phragmites*, and the ditches may serve as conduits for higher salinity waters into back areas of the marsh (Havens et al., 2003).

Windham and Lathrop (1999) noted that in *Phragmites* stands, the soil salinities are lower, there is a lower water table, the micro-surface topography is smoother, and the soils are more oxygenated. Generally, at least some of these factors bear on the suggestion that *Phragmites* stands do not breed mosquitoes (especially compared to high marsh areas).

3.3.3.3 Impacts on Marsh Vegetation Diversity

Predominantly, the literature supports the notion that ditching increases high marsh at the expense of low marsh, although there is not true consensus on this point. Generally, high marsh is thought to be a more diverse ecosystem than low marsh.

Bertness (see Hacker and Bertness, 1999; Bertness et al., 2002) believes that high marsh is more diverse than either low marsh or monotypic stands of invasive *Phragmites*, because along with the dominant species *S. patens* and *J. gerardi*, high marsh supports more co-existing plants. Therefore, he believes management plans should favor high marsh to increase diversity. He also notes that, because anthropogenic nutrient inputs appear to be driving the loss of high marsh, the succession is "unnatural." High marsh also supports a greater diversity of benthic diatoms than does low marsh, apparently because of the variety of salinities (Sullivan, 1997). Shisler (1990) found that ditching increased diversity of the marsh, primarily through colonization of the dredge piles. He noted that OMWM minimizes vegetation changes. Kuenzler and Marshall (1973) explicitly stated that the colonization of ditching spoil piles by bushes and other woody vegetation increased habitat diversity for animals using the marsh.

3.3.4 Ditches Result in Associated Ecological Impacts (Habitat Changes)

3.3.4.1 Losses in Waterfowl Habitat

It is commonly reported that ditching decreases waterfowl usage of the marsh; the presumptive cause of the loss of habitat is a reduction in open water in the interior of the wetlands (Clarke et al., 1984). Studies supporting such conclusions include Urner (1935), Ferrigno (1970), Ferrigno et al. (1975), Nixon (1982), and Cowan et al. (1986). Bourn and Cottam (1950) report 1930s-era complaints that ditching drained ponds where waterfowl fed, and that the ditches were "death traps" for young birds. Bourn and Cottam, in their own research at the urging of the National Association of Audubon Societies, tested a thesis that changes in plant life were responsible for the waterfowl population changes in a twelve year study in Kent County, Delaware. The loss of wigeon grass (*Ruppia maritima*) from draining ponds was believed to be important, but their paper did not quantify any change in waterfowl usage, although it was stated in the paper to have occurred. Nixon (1982) also believed the loss of submerged aquatic vegetation from the high marsh was key, and led to less use of the marsh by waterfowl. Clarke et al. (1984) reported greater foraging by shorebirds, ibises, herons, terns, and swallows in unditched areas, with aerial insectivores concentrating on the areas near pools. The steep sides of the ditches and constantly fluctuating water levels reduced foraging opportunities for small wading birds in the ditched areas. Passerine foraging appeared to be unaffected by ditching. Daiber (1986), Dreyer and Niering (1995), and Wolfe (1996) also reported decreases in waterfowl usage of the marsh after ditching. Daiber stated that the cause of the change was the replacement of vegetation suited for waterfowl by a "colorful but useless expanse of greenery."

Dreyer and Niering (1995) also noted that ditching seemed to have affected the abundance of the seaside sparrow. Breeding pair density was as low as 0.5 pairs per hectare in ditched marshes, as compared to 30 pairs per hectare in unditched marshes on Long Island, and singing and other display behavior were noticeably different. This was partly ascribed to the presence of predators such as weasels and raccoons on the drier ditched marsh, which impacted the birds' behavior (Post and Greenlaw, 1975). Another set of Long Island data showed that beeding success was much less at a ditched marsh, where no young were fledged from five nests in one survey (Post, 1970a), compared to an nearby unditched marsh where 10 fledglings were produced by four nests (Post, 1970b). This second marsh also had evidence of weasel predation, however, despite being unditched. In yet another Long Island study, because there was a lack of pool habitat at the ditched marsh, Merriam (1983) measured a statistically different diet for sparrows there compared to another nearby unditched marsh. The difference in use of ditched marshes was ascribed to seaside sparrows occupying an intermediate position in the moisture gradient of the marsh, where drying or wetting can lead to adverse impacts (Greenlaw, 1992). Post (1970a, b) also found less breeding success for sharp-tailed sparrows in the ditched marshes, and found clear evidence of predation on eggs in the ditched marsh.

It has been noted that ditching may encourage *Phragmites* invasions of the marsh. This may impact waterfowl populations. Benoit and Askins (1999) found that, overall, bird populations declined if *Phragmites* replaced *Spartina spp.*, although the effect was more readily measured as species shifts rather than an overall population drop. In *Phragmites* areas, seaside sparrow, sharp-tailed sparrow, and willet were replaced by marsh wren and swamp sparrow. Shorebirds and waterfowl are less abundant in a *Phragmites*-dominated marsh, as compared to a marsh dominated by short-grass such as *Spartina*, *Juncus*, and *Distichlis* (Fell et al., 2000). However, Parsons (2003) recommended that management plans for marsh islands retain stands of *Phragmites*, as these were important nesting habitats for certain long-legged wading birds in Delaware, including little blue heron, snowy egret, black-crowned night-heron, and, especially, cattle egret and glossy ibis.

Lathrop et al. (2000), while not specifying a difference in waterfowl usage of the marsh areas, quantified more pond acreage, pond density, and greater sizes of ponds in comparing a pristine marsh segment to a ditched marsh area. They noted that it could not be confirmed if this difference existed pre-ditching, but they identified what appeared to relict creeks in the ditched area; however, no former ponds could be found. Redfield (1972) has hypothesized that older, more mature marshes have fewer creeks and ponds than younger marshes or marsh areas, and Lathrop et al. acknowledged that the ditched area might be more mature. It should be noted that Cory and Crosthwait (1939) reported that the installation of shallow ditches connecting deeper ditches to pools supported bird populations in Maryland wetlands, as the shallow ditches prevented the pools from drying in the summer. Daiber (1986) noted that birds requiring a fairly constant water supply, including American bittern, pied-billed grebe, and American coot, will be most affected by losses of open water, and that the increase in brushy areas can cause herring gulls to abandon the marshes. Low marsh birds (e.g., clapper rail) and birds that prefer bushes (e.g., willets) may thrive after ditching. However, Corkran (1938) reported that ditching in Delaware did not drain duck ponds on the marsh surface, using aerial photographs to make the point. Additionally, he noted no loss in muskrats. Cottam (1938) strongly disagreed, noting that when tidal amplitudes were small, effects from ditching were slight, but that, generally, "indiscriminate" ditching resulted in losses in duck ponds and, therefore, duck habitat.

3.3.4.2 Increases in Fish Habitat

Lathrop et al. (2000) noted that a grid-ditched marsh had more tidal edges and decreased average proximity for the marsh interior to open waters as compared to a pristine marsh. They used the metaphor of "commuting distance" to suggest this increased fish access to more of the marsh surface under ditched conditions, but noted that spoil piles on the marsh surface might serve as an impediment to access. Also, the measured decrease in ponds reduces low tide and over-wintering refuge areas for marsh-resident fish. Generally, it has been noted that increasing channels and edge areas probably increases fish use of the marsh area (see Section 2.3, Primary Consumers, above).

Kuenzler and Marshall (1973) found that ditching increased fish habitat in a marsh by a factor of five. They also compared fish caught by seine from the open estuary, natural creeks, and mosquito ditches, and found the species composition for the three areas to be similar; however, the ditches and creeks contained more juvenile fishes, suggesting that they serve as nursery grounds for estuarine species. Clarke et al. (1984) found greater predation by birds on minnows in unditched marshes with pools as compared to ditched marshes, and consequently reported greater fish abundances in the ditched marshes. It should be noted that Knieb (1997) found that crabs may actually be the most efficient marsh-resident fish predator, especially in isolated pools. Balling and Resh (1980) compared fish densities and species compositions between ditched areas and shallow ponds and blocked channels in an Alameda County, California, marsh. The ditched areas had more fish and greater species diversities, apparently because they provided habitat and foraging opportunities for open water fish. It should be noted that the ditches in this study did not drain at low tides. Daiber (1986), in his conclusions regarding the impacts of ditching, found it benefited fish by increasing available habitat.

Deegan (2002) deduced that *S. patens* is much better for smaller fish than *S. alterniflora*, because predators can follow mummichogs and juvenile fish between the more widely-spaced plants. If ditching increases high marsh at the expense of low marsh, it could be better at protecting juvenile fishes. However, the ditches themselves are often lower in dissolved oxygen with higher temperatures compared to than natural creeks (Kuenzler and Marshall, 1973). While some marsh fish, mummichogs, in particular, are highly tolerant of such conditions (Knieb, 1997), other juvenile fish may require better conditions.

3.3.4.3 Increases in Edge Habitats/Improvements in Diversity

There is some controversy regarding the ecological value of edge environments. In some instances, edges promote habitat fragmentation and generalist species over more specialized creatures. Others see edges (ecotones) as highly productive environments that mix the resources of the abutting systems. Edge systems are generally characterized as having more species diversity.

Lathrop et al. (2000) found an increase in channel edges associated with ditching by approximately 100 percent as compared to a pristine marsh. However, they also determined there was much less open water on the ditched marsh, with the marsh surface area to open water ratio of 6.6:1 for the ditched marsh, as compared to 2.8:1 for the pristine marsh. The study used a habitat definition based on the kind of waterway, called “aquatic cover habitats.” The ditched marsh had less of six of seven aquatic cover habitats, with the exception being “intertidal ditch,” defined as the mosquito ditches themselves. Markus (2003) noted that many ditched areas had greater diversity in vegetation than unditched areas, perhaps due to greater disturbance factors. These factors included greater soil draining immediately at the bankside, with associated potential changes in salinity and redox conditions. Cooper (1974) noted that for the irregularly flooded areas of a marsh, ditching is likely to increase edge habitat and increase tidal penetration into the marsh, increasing overall productivity.

Rockel (1969) found increases in crabs due to the increased edge habitat to support their burrows. Dale and Knight (unpublished) did not find increased crabs at a two-year old grid-ditched site in Australia, but did find increases in crab burrows associated with OMWM and runnelling sites. Lesser et al. (1976) found positive impacts on invertebrate communities from grid ditching in Delaware marshes, in contrast to earlier studies of the same areas by Bourn and Cottam (1950). Lesser et al. found a greater density of fiddler crabs (*Uca* spp.) and salt marsh snails (*Melampus bidentatus*) when comparing a ditched marsh to an unditched marsh. They did note that other work found salt marsh snails in considerably lower densities in ditched areas, and that the snails are associated with high marsh vegetation. Clarke et al. (1984) studied maintained, neglected, and unditched areas in a single marsh. They found that marsh surface invertebrate diversity might be greater in the maintained ditch area, but that overall invertebrate diversity, including below-marsh surface, water column, and benthic communities, were not statistically significantly different. This was true although they did find greater avian predation in the unditched areas. The lack of avian predator impact on invertebrate populations and species composition was confirmed by Ashley et al. (2000) and Sherfy and Kirkpatrick (2003). Whaley and Minello (2002) confirmed that the marsh edge serves as a tremendous source of invertebrate prey for nekton, with seasonal populations of infaunal invertebrates inversely related to nekton presence. Balling and Resh (1982) found in Petaluma, California, that installing ditches apparently altered arthropod community structures in the short term. However, over longer time periods, the community structure was the same for 50-year old ditches and natural channels. In addition, generally there was no significant difference between community structure near the ditches and in the center of the marsh. An exception was found in the dry season, when diversity was greater near water courses in the marsh. The difference between new and older ditches was attributed to the development of different plant communities, and the greater dry season diversity was attributed to improved food quality near the ditches as compared to the interior of the marsh.

Travis et al. (1954) found that ditching caused vegetation changes, but absolutely no change in wildlife composition. This may be because the prevalence of omnivores in the invertebrate populations in marshes may make those populations more resilient to environmental perturbations (Kreeger and Newell, 2000), and therefore the base of the food chain may stay similar despite changes in vegetation. In fact, Teo and Able (2003) found, through a capture and recapture program, that mummichogs preferred the marsh surface together with smaller creeks and ditches as habitat, compared to the larger, natural main marsh creek in a New Jersey marsh. Peterson and Turner (1994) found that transient species appeared only to use the edge of the marsh, as they were not captured on the marsh surface although resident species were found there. Resident species also tended to use the edge of the marsh, and, in fact, the greatest biomass of nekton is found within three m of the edge of the marsh. These finding imply that ditching has a potential to increase transient fish use of the marsh.

On the other hand, using quadrats for sample collection, and comparing results to an unditched marsh, Bourn and Cottam (1950) found reduced invertebrate populations in the upper inch of ditched marsh areas. Four vegetative associations, *S. alterniflora*, *Distichlis*, *S. patens*, and *Scirpus robustus*, were studied, and the reductions were described as ranging from 50 percent to as much as one to two orders of magnitude. The number of species was reduced by up to 50 percent for most samples. Much of the impact is attributed to drying of the sediments. Ditching was also said to destroy muskrat habitat, which was measured indirectly through changes in muskrat trapping revenues when ditching in Delaware was essentially completed, after 1938. Daiber (1986) and Wolfe (1996) also noted decreases in muskrats following ditching, but it is not clear if these observations are independent or are citations of the earlier Delaware research.

Teal (1986) noted that *S. alterniflora*, when it grows on stream banks, is more productive, but also has wider spacing between stems, meaning there is more bare ground when compared to *S. alterniflora* growing on lower low marsh surfaces. Fischer et al. (2000) found that wrack caused breaks in the *S. alterniflora* monoculture, at points along channels where flows slowed, such as creek bends and drainage channels. Straight-line ditches, absent these accumulation points, could tend to support the monoculture. On the other hand, Hacker and Bertness (1999) found the "upper middle intertidal," which generally accords with the *S. patens* high marsh, is much more diverse than the other areas of the marsh for New England marshes. They attribute this to facultative as opposed to competitive relationships among the dominant species.

Warren et al. (2001) noted that, generally, fish, invertebrate, and plankton diversity were less in ditches and creeks within *Phragmites* stands than in areas not invaded by *Phragmites*. Able and Hagan (2003) found that, although mummichogs will lay eggs, and the eggs will hatch, with approximately the same frequency in *Phragmites* as compared to *S. alterniflora* stands, larval and juvenile abundances were significantly lower. This research suggests that *Phragmites* invasions may have a deleterious impact on marsh mummichog populations. The change in adult fish use of marsh surfaces was also noticeable, but not as great as the difference for the younger fish (Able and Hagan, 2000).

Of some interest in the notion that younger, less mature marshes have more open water and simpler connections to the estuary (Redfield, 1972; Frey and Basan, 1985). By this measure, ditching tends to "age" or "mature" a marsh, as older and more mature marshes have more connectivity to the estuary and less surface water. However, the relatively increased complexity associated with ditching, if such ditching were comparable to natural channel patterns, would be a characteristic of younger, less mature marshes (Odum et al., 1979). Valiela et al. (2000) found mature marshes export more material to estuaries then do younger marshes, suggesting that ditched marshes may be ecologically more connected to the estuarine system.

3.3.5 Changes to Water Flows Theses

Collins et al. (1986) constructed a description of how the installation of mosquito ditches fundamentally alters the flow of water through a marsh system. Because the tidal prism transmitted to the marsh is unchanged, as it is controlled by offshore bathymetry, the construction of ditches increases the capacity of the tidal drainage system, but only for those tides that do not overwash the surface of the marsh. This means that the amount and velocity of water transmitted up natural channels is less, and the overall height of the water transmitted through the ditch-channel system will be less. This means there will be less frequent inundation of the marsh surface; less frequent inundation and slower tidal velocities suggest that there will be less transmission of sediments into the marsh from off-shore sources following ditching.

Boon (1975) found that the particular stream morphologies of particular marshes generate specific tidal discharge asymmetries. This is the relationship between the time spent flooding the marsh and the time spent draining the marsh. Greater velocities are associated with shorter time periods. Modeling of the Satilla River estuary, Georgia, by Zheng et al. (2003) also found varying tidal asymmetries for different salt marshes. These studies, in opposition to Collins et al.

(1986), suggest that changing the stream morphology can impact tidal flows and velocities in difficult-to-forecast ways.

Taylor (1938) and Provost (1977) found that ditches allowed for more effective transmission of tidal flows into the marsh. Heuser et al. (1975), discussing Flax Pond, New York, where salt marsh was created by breaching a barrier bar, also imply that ditching effectively transmits salt water, as they claim higher salinities necessary to support salt marsh vegetation are enabled by ditches in the marsh.

3.3.5.1 Ditches Convey Upland Pollutants to Offshore Waters

3.3.5.1.1 Via Stormwater

It has become commonplace to state that marshes can serve as filters for land-based pollutants. This general precept is fixed in many peoples' minds, despite uncertainties such as those discussed by Nixon (1980), showing that different researchers had found both nutrient absorption (Gosselink et al., 1972) and release to the surrounding estuary (Reimold, 1972) as fundamental marsh attributes. Nonetheless, this precept, together with the successful, designed use of artificial wetlands for sewage treatment, leads many people to assume all marshes serve as water treatment locales and remove contaminants that may enter them via water flows. If this is assumed to be true, it follows that ditches, therefore, upset this natural function by altering the underlying hydrology, especially as regards stormwater flows that enter the marsh.

Kuenzler and Marshall (1973) believed their work showed that the marsh surface reduced particulates in run-off, and so decreased overall water turbidity. They concluded that ditches convey silt and decrease salinities offshore from marshes due to rapid transport of run-off. This was especially true when they were connected with upland drainage systems. The impacts were expected to be worse where ditches drained into the headwaters of natural creeks as opposed to emptying closer to their mouths. However, they also noted that large amounts of run-off were conveyed into natural creeks via overland flows.

Crosland (1974), in an unpublished thesis, found that open ditches had higher salinities than dammed ditches, suggesting that flows in such systems were greater, and that blocked ditches would retain freshwater runoff. Fultz (1978), in discussing OMWM ditching installed in

Georgia, noted that stormwater is conveyed more rapidly through the marsh via ditches than it had been prior to ditching. This was noted with approval, as the rapid draining of surface water from the marsh prevented optimal mosquito breeding conditions from occurring. Cory and Crosthwait (1939) also found that ditched marshes conveyed stormwater more quickly than unditched marshes; however, their focus was on the run-off of stormwaters from increased tides that are generated in the estuary and then conveyed into the marsh, not terrestrial flows moving through the marsh. Reimold (1969), in an unpublished thesis, found that ditches conveyed less phosphorous to the estuary than did natural creeks.

Metals are important stormwater contaminants. Gambrell (1994) reported that run-off from metals contaminated uplands into shoreline wetlands is enriched in copper, nickel, zinc, lead, and manganese as compared to offshore waters, supporting the general contention that marshes filter stormwater contaminants. He found that chromium concentrations did not follow the broader trend. Turner et al. (1985) reported that for lead, at least, contact with fine textured soils and mucks resulted in 98 percent immobilization of contamination in run-off entering stream systems or wetlands. Williams et al. (1994) noted that organic content is key in determining the sorption of metals to surface sediments. Plant detritus appears to be the most effective media, although it is continually being degraded, meaning the storage of the metals may not be long-term. In addition, increasing salinity can cause immobile cadmium, mercury, and zinc to become labile, by forming complexes with chloride ions. Therefore, if these metals are retained on the marsh during stormwater flows, they may be released from the surficial sediments in subsequent tidal flushings. Generally, salt marsh plants tend to not be impacted by the toxicity of environmental metals.

Teal (1986) found that marshes absorb heavy metals as insoluble sulfides. Lead appears to be essentially permanently bound to sediments. On the other hand, the most loosely bound metal was cadmium, which appeared to have, on average, a two-year residence time in marsh sediments.

Nixon (1980), despite overall reservations regarding general statements pertaining to marsh processes, concluded that the accumulation of sediment in marshes generally indicates that nutrients and particle-associated contaminants will also accumulate in a marsh. This includes contaminants transported via stormwater, atmospheric deposition, and tidal flows. Marshes may remove different contaminants preferentially from different sources. For example, atmospheric deposition were cited as resulting in lead and nitrogen accumulations, and tidal inputs were belived to cause manganese accumulations in marsh sediments. Nixon did not identify stormwater as a particular source of any accumulating species in marsh sediments.

Banus et al. (1975) found that lead was retained more than zinc which was retained more than cadmium in plots on the salt marsh that had been fertilized by sewage sludge. Giblin et al. (1983) found marsh sediments retained heavy metals to different degrees. Less cadmium, 15 percent, was retained, and more lead, 60 percent, was retained, from starting concentrations contained in the sludge. Giblin et al. (1986) summarized the findings of the metals enrichment experiments by determining that iron hydroxides found in the upper few cm of marsh soils can effectively immobilize dissolved and particulate-associated metals in a sludge, with certain metals being more effectively retained than others. In addition, high marsh was more effective at retaining the metals than low marsh (Giblin et al, 1980).

Nixon (1982) noted that metals in marsh sediments are stable due to existing redox and organic matter conditions; changes in these could mobilize the metals. For example, Kerner and Waldman (1992) found that zinc and cadmium can be mobilized from marsh sediments and mud flats under aerobic conditions. Remobilization of metals is greatest during colder seasons when microbial activity decreases, and the aerobic zone reaches deeper into the sediments (Hines et al., 1984). The micro-aerobic zone that forms around *S. alterniflora* roots is enriched in dissolved metals, because of the changed redox chemistry associated with oxic conditions (Williams et al., 1994). The effect is greater during the day than at night, due to oxygen production by the plant through photosynthesis (Howes et al., 1981).

Salt marsh plants also take up metals from the sediments, and tend to become metals-enriched in their vegetation. Many salt marsh plants excrete metals through their stomata, transferring sediment metals back into the aquatic system (Williams et al., 1994). This means that salt marsh sediments, through the uptake by plants and their subsequent redistribution of the metals back to the aquatic environment as the plant material becomes detritus, will continue to be sources of metals contamination long after the pollution has ceased (Wiegert and Pomeroy, 1981).

Although many salt marsh plant metals concentrations correlate well to sediment concentrations, this tends not to be the case for *S. alterniflora*. Its roots tend to be where metals concentrate, so that the overall concentrations measured for *S. alterniflora* depend on the parts of the plant that are tested (Breteler et al., 1981). And, *S. alterniflora* was also shown to excrete concentrated salts of mercury, cadmium, and zinc through the stomata of its leaves (Kraus et al., 1986). Although *Phragmites* takes up metals from sediments approximately as readily as *S. alterniflora* does, it does not excrete it through salt stomata (Weis et al., 2002; Windham et al., 2001). Lee et al. (2000) found that metals concentrations in water column organisms generally relates more closely to concentrations measured in bulk sediment rather than those in pore waters, suggesting the dissolved concentration of the metals may not be the key factor in determining environmental impact.

Soil cores in marshes find metals concentrations reflect releases of metals in surrounding environments, although factors such as the source of sediment material, particle size, organic content, and sediment type affect the concentrations measured in the sediments (Williams et al., 1994). McCaffrey and Thomson (1980) found long-term increases in copper in a Connecticut marsh sediment record that matched US copper production records. Several of these studies relate distributions of metals in sediments to atmospheric deposition records or sources (Griffin et al., 1989; Bricker, 1993; Weis et al., 2001). Using measurements of local sediment characteristics and metals' affinities for organic matter (Morrisey et al., 2000), Williamson and Morrisey (2000) developed a simple model to predict metals concentrations and distributions in estuaries. It did not include a term for salt marsh filtering, but nonetheless did well at predicting the actual concentrations and distributions of metals in an urban estuary and also a rural estuary, in New Zealand.

Generally, increases in stormwater quantities or contamination in stormwater appear to impact marshes. Lerberg et al. (2000) did not find increased metals concentrations in sediments for marshes in suburban or industrial watersheds as compared to control sites, and they did find invertebrate diversity and abundance was greater in the impacted marshes. However, the contaminated sites were dominated by pollution-intolerant species as compared to the control sites, suggesting that the food web affected by pollution might be becoming simpler. Swales et al. (2002) noted that increases in urbanization in stormwater catchments leads to increased sedimentation in the associated downstream tidal waterways, with subsequent increases in contaminant concentrations.

3.3.5.1.2 Via Groundwater

Groundwater inflows into marshes are not well-studied. Generally, aquifers are thought to discharge off-shore. It is clear that most discharges into estuarine waters occur where the land surface falls below mean high water. Most marsh creeks are at or below mean high water, or, since they tend to nearly dry out at low tide, are above mean low water. Groundwater discharges from the aquifer at increasing rates, generally, as the tide falls. This is because a falling tide means that the head pressure of the estuary is becoming less, while the groundwater head pressure stays constant. Conversely, at high tides, the saline water has a greater head than fresh water at the salt water-fresh water interface, and so the saline water will push the fresh water back. The greater the tidal range, the larger the mixing is. This means that more water will flow into and out of the sediments for an equal amount of aquifer discharge, if the tidal range is larger (Paulsen, 2001).

However, an aquifer that experiences different tidal ranges over its discharge points almost certainly discharges equal volumes per linear frontage on the estuary over the course of the year. The greater discharge rates for the higher tidal ranges balance the greater saline inflows at high tides.

If a ditch intercepts the fresh water table, it will serve as a discharge point for the fresh water aquifer. If this is the case, then general hydrology would lead to the fresh water head to be greater slightly away from the discharge point. Inflows into the creek would therefore occur from the banks and creek bottom. However, the bed of the creek or ditch is probably comprised of material with a lower conductivity than the banks are, if the marsh waterways are similar to other kinds of waterways. Ditches and creeks also serve as discharge points for the saline water table, which is comprised of tidal waters stored in the marsh peat. Discharges coming from the banks of a creek or ditch at low tide could very well be from the saline water table, or, if the fresh water table has been intercepted, from the fresh water table as well. Visual inspection of a bank will not determine if fresh groundwater is discharging into a ditch or creek, therefore.

Teal and Howarth (1980) diagramed water flows in a marsh stream bank, suggesting that at low tide, saline pore waters from the marsh surface drain down into the stream, but that fresh groundwaters discharge up into the stream. Valiela et al. (1978) found that groundwater was a major source of inorganic nitrogen to the marsh, where it is converted to organic nitrogen, which may be more available to microorganisms. Theoretically, ditches may be low pressure zones where groundwater would discharge, and not enter into the marsh sediments where this chemical reaction occurs.

Hemond and Fifield (1982), believing that seepage in the marsh peat is negligible except near creeks, theorized that evapo-transpiration is the primary means for removing water from marsh peat away from creeks. Then, due to the loss of head, groundwater inflows would ensue. Nuttle and Harvey (1995) expanded this argument by constructing a water balance based on head measurements that accounted for evapo-transpiration rates. Assuming no loss of water to the creek from the interior of the marsh, they determined that groundwater upflows were twice as great as tidal inflows for an irregularly flooded high marsh.

Harvey and Odum (1990), working in a fringing marsh with a “hillslope” aquifer (the hills were six to 20 m tall), found that maximal discharge into the wetlands was greatest at the upland fringe, and decreased with distance towards the open estuary. The pore water flows in the marsh were dominated by tidal flows, meaning that groundwater had long residence time in the marsh peats and thoroughly mixed with saline waters prior to discharge through the marsh. The thesis allowing for discharge to the marsh peat is that it lies deeper than the zero head height for the fresh water aquifer.

Discharge of groundwater through the sulfidic, organic marsh peats may result in transformations and absorption reactions for groundwater contaminants, probably to a greater degree than occurs with off-shore groundwater discharge. Therefore, if groundwater contaminants are a concern and ditches do encourage groundwater discharges in the marsh, the presence of ditches may serve as a minor mitigation for estuarine water quality.

3.3.5.2 Ditches Convey "Marsh-generated" Pollutants to Offshore Waters

It has been shown that stream-side marsh areas have less detrital material on them than interior areas of the marsh. Therefore, since ditched wetlands appear to have smaller distances from the interior of the marsh to waterways than unditched marshes (Lathrop et al., 2000), ditched marshes may be expected to be better at exporting detritus than unditched marshes (Odum et al., 1979). Montague et al. (1987) point out that since run-off from the marsh surface may merely collect in channels at low tide and be recycled back into the marsh interior on the next high tide, increasing groundwater discharge rates and channel connectivity helps ensure export of the material from the interior of the marsh.

Gardner (1975) and Childers et al. (2000) discussed pore water seepage to channels as an important mechanism for the transport of nutrients to the estuary from the marsh. If ditching increases drainage of the water table, this may increase the export of pore-water constituents.

Ribelin and Collier (1979) noted it is quite common for ebbing tides during the daylight hours to transport films of organic material off the marsh surface into creeks and, presumably, ditches. The films were described as "brown, oily to the touch, and easily disrupted," and were composed of benthic and filamentous algae detritus. These films might be mistaken for contamination releases. Grant and Bathman (1987) noted that "white sulfur mats" of filamentous sulfur bacteria are easily dislodged by currents from benthic settings, and are a significant flux of sulfur from the environments they leave to the settings they are transported to. Harriss et al. (1980) discussed how most of the particulates exported from southern marshes are composed of amorphous organic films. Increasing transport rates from the interior of marshes may result in greater releases of all of these materials.

3.3.5.2.1 Coliform

Stevenson et al. (1979) found heterogeneous fluxes of bacteria at three different channels from a marsh in Georgia, and cautioned that determinations of bacterial inwelling or outwelling from a marsh system need to be carefully made. Jensen et al. (1980) found strong correlations between wetlands and high coliform readings. They were not able to link a cause to the readings, but found coliform typically varied inversely with light intensity and directly with nitrate concentrations. Zdragas et al. (2002) found the ability of a wetland to remove coliform from inflows depended directly on temperature and solar insolation. If coliform have marsh surface and/or sediment sources, then increasing transport rates from the interior of the marsh may increase releases of coliform to the estuary.

3.3.5.2.2 Pesticides

The first flush of stormwater is often measured to be more contaminanted than other run-off. In particular, the first flush to San Francisco Bay was found to be particularly enriched in pesticides. The pesticides were associated with suspended matter, not particulates (Bergamashi et al., 2001). If ditches encourage stormwater to bypass absorptive and reactive surfaces on the marsh surface, they may lead to increased inputs of stormwater associated pesticides.

The pesticides applied to marshes for mosquito control have not received much testing in terms of transport to the neighboring estuary. Sampling by the USGS associated with the Long-Term Plan has found little, if any, transport of Vector Control larvicides and adulticides to estuarine waters, even when testing in creeks in close proximity to the treated marsh. Nearly all salt marshes in Suffolk County have been ditched; therefore, the marshes that were sampled were ditched marshes (S. Terraciano, USGS, personal communication, 2004).

3.3.5.2.3 Other

The reducing conditions in anoxic soils results in a soil that is buffered with a pH of approximately 7 (neutral) (Gambrell, 1994). If these sulfidic-soils are uncovered, sulfuric acid may be generated through oxidation. Bourn and Cottam (1950) report complaints from sportsmen that ditching killed fish by releasing “marsh gasses,” and Kuenzler and Marshall (1973) reported boating in mosquito ditches caused releases of hydrogen sulfide.

The depth of the ditches is important with regard to sulfuric acid production. Portnoy (1984) reported that the exposure of pyrite sediments, due to the excavation of ditches one m deep, resulted in streams impacted by sulfuric acid, with pH values of 3. Frey and Basan (1985) also reported oxidation of sulfidic soils leading to highly acidic conditions following marsh disturbances, specifically, an impoundment. Soukup and Portnoy (1986) reported on fish kills which were apparently caused by the acidification of fresh water streams following the diking and ditching of saltwater marshes in Massachusetts, where sulfidic soils were uncovered. Daiber (1986) also cited reports discussing the development of acidic conditions after ditching. Dale and Knight (unpublished) discussed decreases in pH values associated with a newly grid-ditched marsh, when compared to control sites and older OMWM and runnelling sites. Howarth and Teal (1980) indicated that only 10 percent of reduced sulfur compounds produced by the degradation of organic matter in the sediments are actually buried in marsh sediments. The remainder is oxidized, and likely enters the marsh pore waters as sulfate. This natural process may result in the production of acidic pore waters, therefore, which is supported by pore water measurements made by Lesser (undated). Generally, however, the construction and/or maintenance of ditches may result in the uncovering of reduced sediments, and lead to acidified drainage until conditions again reach equilibrium.

3.3.5.3 Ditches Affect Marsh Accretion Theses

In terms of an overall sediment budget, Frey and Basan (1985) suggest that sediments are more likely to be lost from channels, when compared to the marsh surface, due to greater velocities experienced under confined flow conditions as compared to sheet flows. Conversely, this means that sediment is more likely to be deposited on the marsh surface than in a channel. Pomeroy and Imberger (1981) note that physical processes dominate sediment reworking in channels, but that biological forces dominate on the marsh itself. Reed (1995) noted that local interrelationships between vegetation, soil, and hydrologic processes that combine to create accretion on a marsh surface make it nearly impossible to predict what changes may occur to a marsh due to sea level rise.

Boon (1975) found that the stream morphologies of particular marshes will determine a specific tidal discharge asymmetry. He cited work from the Netherlands showing that these asymmetries can determine if sediments are predominantly conveyed into a tidal system or out from a tidal system. Therefore, changes to the stream network may impact water flows in and on the marsh.

3.3.5.3.1 Transmission of Sediments to the Marsh

3.3.5.3.1.1 From Off-shore

Cahoon and Reed (1995) described sedimentation in Louisiana as being driven by the resuspension of sediments present in nearby bodies of water. Therefore, storms are important in when considering long-term sedimentation processes, as they mobilize the largest volumes material from the bottom of these bodies of water. Stumpf (1983) described an unditched marsh in Delaware where tidal influxes of sediment were insufficient to maintain the marsh surface. However, inputs from severe storms were sufficient to maintain the marsh, especially maintaining the sediment supply to the high marsh. In a retreating marsh in a high tidal range area, Reed (1988) found that repositioning of sediment from the eroding marsh front to the interior of the marsh, creating accretion in the interior, was more important than the importation of material via creeks.

Collins et al. (1986) suggest that because of tidal dispersal in the increased channel areas, there will be reduced transmission of sediments into interior marshes within a ditched system. The deposition of spoils along ditch banks can exacerbate this. Kennish (2001) also suggested that spoils deposition along ditches impedes the delivery of sediment to the interior of the marsh. Anisfield et al. (1999) found that sedimentation rates increased in marshes that, as part of restoration projects, had increased water flows and water table heights, and attributed the difference, primarily, to increased pore spaces and greater organic matter inputs. Harrison and Bloom (1977) found that sedimentation rates in high marshes vary with tidal ranges, and that years with more storms have more deposited sediment than years with fewer storms. Morris et al. (2002) found that in systems with high rates of sediment loadings, such as the southeastern US, the marshes should be able to sustain themselves better against sea level rises than those with smaller sediment inputs such as the northeast US. This is because peat formation, which is more important for northeast US marshes, may not be able to ensure sufficient material inputs.

Frey and Basan (1985) note that marshes tend to grade from finer to coarser grained sediments moving from low marsh to high marsh. This may be related to stem density or other trapping mechanisms.

3.3.5.3.1.2 From Uplands

Collins et al. (1986) found that natural channels may infill quicker due to erosion in the presence of mosquito ditching. This is because the level of water associated with tidal inflows will decrease due to tidal dispersion, and so velocities of ebb tides may not be great enough to remove deposited material, such as material from banks, vegetation trapping, or from upland sources. However, Frey and Basan (1985) classify terrestrial uplands as “lesser” sediment sources, especially in the northeast US.

3.3.5.3.2 Affect Peat Accretion

A review of sedimentation rates for east coast and Gulf marshes found that organic inputs are five times more important than inorganic inputs in determining sedimentation rates for east coast marshes (Turner et al., 2000). This suggests that, generally, plant community changes may be more important in determining accretion rates for ditched marshes than sediment transmission. Peat formation is enhanced when winter ice shears plants off, creating greater detritus loadings (Frey and Basan, 1985). Marshes with more peat generally have less developed creek structures (Chapman, 1974). Allen (2000) theorized, for European marshes which generally have higher tidal ranges, that mineral sediment inputs dominate sedimentation processes when sea level is rising, but when sea levels are more stable or even falling, organic matter dominates the sediment deposition process.

Armentano and Woodwell (1975), using lead-210 dating techniques at Flax Pond, Long Island, found that sedimentation rates were lower by nearly one third in close proximity to the creek than further away, although sediment deposition was outpacing sea level rise throughout the marsh. Roman et al. (1997), using marker horizons, found distance from the estuary inlet to be the telling factor, with increasing distances leading to lesser accumulations. Leonard et al. (2002), using sediment traps, found sediment rates to be greater near marsh creeks in the Chesapeake Bay. They also found that sedimentation rates in *Phragmites* stands were similar to those in *S. alterniflora* stands. However, Rooth et al. (2003), also working in the Chesapeake Bay area, found increased sedimentation rates in *Phragmites* stands, which they concluded came from increased litter generation.

3.3.5.3.2.1 Low Marsh vs. High Marsh

Redfield (1972) noted that peat accumulation rates in the high marsh were greater than in the low marsh. Nixon (1982) noted that high marsh develops tussocks due to uneven peat accretion. Conversely, Jordan and Valiela (1983) found sedimentation rates were greatest in the tall-form *S. alterniflora* zones, and lowest in *S patens* and short-form *S. alterniflora* areas. Richard (1978) and Bricker-Urso et al. (1989) found that overall sedimentation rates were greater in low marshes than in high marshes. Chapman (1974) noted that sediment accumulation is usually greater in low marshes than high marshes, as the low marshes flood more often.

Warren and Niering (1993) described changes in vegetation of the high marsh at the Wequetequock-Pawcatuck marshes, Connecticut, over 40 years, and ascribed them to a failure of the marsh to maintain its sedimentation against local sea level rise. The losses of monospecific stands of *S. patens* appeared to be primarily where sedimentation was least; the difference in sedimentation appeared to be linked to a lack of accretion of peat and lesser sediment trapping by *S. alterniflora*. Saturation of the high marsh by mosquito ditches was cited as a potential cause, due to spoil bank impoundments and also because they serve as a novel source of water to the central marsh.

3.3.5.3.2.2 Oxidation of Peat

Daigh and Stearns (1939) reported that reductions in the water table associated with ditching lead to peat oxidation in the uncovered area. Rockel (1969) found that ditching results in a slumping of what formerly had been a level marsh surface; typically this occurs within seven m of the ditches. This may be due to the lowered water table leading to oxidation of peat or loss of permeability associated with collapsing void spaces because of the withdrawal of water. Rockel suggested that 42 percent of a marsh would be so affected by 100 foot spacing of ditches. Bourn and Cottam (1950) also reported extensive slumping throughout ditched areas of the marsh. Redfield (1972) found similar slumping along natural creeks. Weinstein et al. (2000) also found subsidence in marsh surfaces in ditched wetlands, and attributed it to oxidation of peat.

3.3.5.4 Ditches Widen with Time

Pethick (1992) provided a theoretical reason for the general widening of ditches in his discussion of English marsh channel morphology. He found that natural marsh channels have wide, but rapidly narrowing, mouths, caused by the dissipation of wave and tidal energy onto the shoreline, analgous to the forces acting in estuary mouths. Natural marsh channels then have a long, constant width, unbranching segment, which is maintained by the focusing of the remaining wave and tidal energy. At the head of the marsh, where energy dissipates, a natural drainage morphology in the form of a multibranching structure overcomes the wave energy, which is largely lost to friction. Channels and, by extension, ditches are thus controlled by the open water system, and so are morphologically part of open water rather than being marsh surface features.

On the contrary, a study of Italian wetlands systems by Marani et al. (2003) found that total channel lengths in a wetland were a function of watershed area rather than the tidal prism, suggesting the channels are a feature associated more with drainage than tidal propagation.

Miller and Egler (1950) found a regular progression of ditches widening and becoming more bowl-shaped in profile. The enlarging-type ditch will support a *S. alterniflora* fringe, and create a natural looking levee with panne formations lying behind. Dale and Hulsman (1990) observed that ditches in Australia dug “a spade deep and wide” are now 20 m wide after 65 years. Bourn and Cottam (1950) found 20 inch wide and deep channels eroded to several feet wide and deep over a decade. Lateral erosion at ditch mouths is rather commonly found, although it is noted that such erosion further up a ditch generally results in bank slumping and, thus, ditch blockages (Dale and Hulsman, 1990). Aerials provided by the New York State Department of State show widening of many ditch mouths over time in the South Shore estuary (J. Zappieri, NYSDOS, personal communication, 2004). Frey and Basan (1985) note that the greater the percentage of peat in the substrate, the more stable channel structures are. For example, Redfield (1972) showed that channels at Barnstable, Massachusetts, had been stable for approximately 100 years, neither growing, shrinking, nor changing position appreciably.

Odum (1988) notes that salt water marshes are generally resistant to erosion for several reasons:

- high biomass of root materials per unit area

- large amounts of plant litter on the sediment surface

- relatively coarse particle sizes when compared to other wetland environments

Pethick (1992) and Odum et al. (1979) note that long-period storms that cause erosion of the face of the marsh are, probably, the greatest sediment removal mechanism for tidal marshes, especially those marshes that face open water. Odum et al. also cite intense rainstorms occurring at or near low tide that may loosen and wash fine particles off the marsh surface as potential erosive forces.

3.3.5.5 Ditches Infill

Generally, sedimentation is not a problem for ditches, according to a review paper by Dale and Hulsman (1990). They found ditches tend to persist over time. However, Bourn and Cottam (1950) found that ditches needed maintenance, especially at the upper ends, after only a couple of years. Kuenzler and Marshall (1973) reported that ditches in silty soils require maintenance in less than 10 years. In Florida, ditches were characterized as filling at the estuarine mouth (Carlson et al., 1991). Lathrop et al. (2000) discussed slumping as a mechanism for reducing the steep sides of ditches, but did not suggest they fill in entirely. Pomeroy and Imberger (1981) reported that, for natural creeks in three m tidal regimes, levees form on the banks of the creeks, then migrate creekward, helping to cause slumping. Slumping also occurs during intense rains at low tides, due to loss of binding strength of uncovered clays. Collins et al. (1986) and Chapman (1974) believed that the presence of plant forms tall enough to bridge the ditches would result in infilling of the ditches due to sediment trapping below fallen vegetative matter. Teal (1986) noted that ice is a powerful erosive device in New England marshes, and often removes vegetation that became frozen into shoreline ice floes. Marshes that are often frozen may thus have a means for clearing ditches of overbridging vegetation.

Taylor (1938) found there is an optimal length to ditches, which should vary with the tidal range. Ditches could be constructed longer with greater tides. Ditches that are too long will not be able to maintain themselves through tidal flows. Taylor suggested the maximum length should be a quarter mile. Miller and Egler (1950) found a series of ditches that "aggrade," so that the ditch infills and is covered by *S. alterniflora*. A remnant "turf-line," or raised elevation from the ditch spoils, is indicative of the former ditch. They provided no rationale for why some ditches fill and others widen, although it is clear those ditches subjected to greater tidal forces tend to widen at the mouth. Redfield (1972) found that ditches "overdrain" the marsh and so collect sediments, creating a need for maintenance. In a study in Maine, Markus (2003) found ditches both filling and widening at the same time, in the same marsh, and even for different stretches along the same ditch.

3.3.6 Impacts from Ditch Maintenance Theses

Ditch maintenance, it should be noted, was required more frequently when cattle were allowed to graze in the marshes. Grid ditching also required more maintenance than parallel ditching. New York City ditches, installed primarily from 1916 to 1922, required a great deal of maintenance by the onset of the Great Depression, circa 1933 (Richards, 1938). Taylor (1938) suggested that ditches along the south shore of Long Island did not need any maintenance in 1936-1937, three to four years after construction.

3.3.6.1 Spoils Disposal

Miller and Egler (1950) noted that maintained ditches will have an artificial levee formed from the discarded spoils, and, if complete enough, this barrier may prevent the marsh from draining and, thus, undo the intent of the ditching. Shisler (1973) studied the impacts of spoils piles left on the marsh over the short-term. He found that the piles attracted predominantly *S. alterniflora* in the low marsh and *S. patens* and other high marsh species, including *Atriplex patula*, *Distichlis*, *Salicornia*, and *Iva*, in the high marsh, if the piles were "mashed" down to the level of the open marsh. Others (see Section 3.3.3.1, High Marsh, above) have found spoils piles can become sites of woody, high marsh species. Wolfe (1996) notes that the increase in topography, coupled with drainage of the marshes from the ditch installation, caused great changes in marsh hydrodynamics. However, Wiegert and Freeman (1990) emphasize that high tidal amplitude marshes seem to construct natural "levees" at creek edges from the process of sediment laden water spilling out onto the marsh, as the increase in area would cause the water to slow and, therefore, tend to drop its sediment right on the banks.

3.3.6.2 Impacts to Ditch Dwellers

Concerns have been raised that ditch maintenance might harm diamond-backed terrapins, in particular. Teal (1986) states that terrapins feed in the low marsh during low tides, but do not live there. CA spoke with Andy Sabin, President of the South Fork Natural History Museum, who had involvement in a study of spotted turtles inhabiting ditched wetlands in southeastern Long Island with John Behler, Curator of Herpetology from the Bronx Zoo. He indicated that it appeared that 25 percent of the radio-tagged turtles were lost over one winter. Ditch maintenance is believed to be involved, as extensive work was conducted in the study area at that time (Andy Sabin, South Fork Natural History Museum, personal communication, 2004). However, Michael Bottini, in an application to the Long-Term Plan to study spotted turtles at Napeague marsh, indicated that the radio tagging experiment suffered equipment failure, based on e-mail communication with Behler, and no reliable information had been generated by it. To Bottini's knowledge, it was the failure to find approximately one-quarter of previously captured turtles marked with painted or notched scutes following ditch maintenance that caused the suggestion that ditch maintenance impacted the population. Bottini also notes that the turtles use the fresh water wetlands as habitat, not the adjacent salt marsh (Michael Bottini, independent researcher, personal communication, 2004).

3.4 Summary

Salt marsh mosquitoes have precise requirements for successful breeding. Richards (1938) underscored that it is not a matter of salinity. Rather, as shown in the early 1900s and echoed in Dale and Hulsman (1990), their habitat is controlled by infrequent inundations. Daily inundation is too frequent to support breeding, but four to eight times a month seems to be optimal. LaSalle and Knight (1973) explicitly note that ditching, as a mosquito-control water management technique, cannot succeed if the frequency of tidal inundations is unchanged, and water does not drain quickly enough to preclude larval development. However, over the short-term, ditching did seem to decrease mosquito populations substantially in most cases. It must, therefore, either impact water flows or allow greater access of fish to breeding points, or both. The effect must be substantial to be so effective. It is far from clear whether marsh processes allow this effectiveness to be maintained for the long term. It is not apparent whether merely clearing ditches of accumulated material is sufficient for ditches to regain their effectiveness for mosquito control. The creation of berms and long-term alteration of marsh vegetation regimes may establish increased mosquito breeding opportunities that undo ditch installation impacts.

New York State has stated unequivocally that ditching decreases the “functions and habitat value” of a salt marsh (Niedowski, 2000). While avian impacts were mentioned more, the loss of submerged aquatic vegetation due to the disappearance of upland pools was also said to be important. Chapman (1974), especially citing the work of Bourn and Cottam (1950), noted that ditching cost a great deal of money, may not be effective at mosquito control, and destroyed the economic value of the marshes. Dale and Knight (unpublished) found more impacts from ditching than from runnelling or OMWM. However, it was their opinion that these impacts were insignificant for these particular marshes.

Provost (1977) found that many discussions of ditch impacts were flawed because of poor consideration of all the relevant factors, such as concurrent dredging of nearby channels. He found that such poorly considered claims are then repeated in standard texts. Provost also was concerned that the impacts cited about ditching were parochial in nature, being “waterfowl effects and sportsmen’s concerns.” He noted that the studies did not consider, given the necessity for management of these environments, whether managing for waterfowl or muskrat production is any more "natural" than ditching for mosquito control. Provost further noted that, generally, since "natural" conditions may be considered to be the equilibrium state of the salt marsh, any alteration of the system to favor or discourage something in that system will necessarily impact other elements, perhaps in unpredictable ways. The alterations could be to remove mosquitoes, or to foster waterfowl usage. Daiber (1986), too, noted that any management practices in the wetlands will result in some form of habitat change, leading to further alterations in speciation, population densities, and distributions of flora and fauna.

Cooper (1974) described ecological losses, due to ditching, for the regularly inundated portion of the marsh, as the drying out process would change the overall vegetation to be more like the upland fringe, comprised of much less productive plants that do not export detritus to the estuary. However, the irregularly flooded parts of a marsh would benefit from increases in edge habitats and tidal influxes, increasing productivity. In addition, there is some evidence that ditching is a potential habitat improvement for transient fish, especially juveniles. However, it is far from clear whether ditching results in greater production of the forage fish and other potential prey for these juveniles, and so the overall production of fish may not be impacted by ditch installation.

Since 90 percent of the salt marshes in the northeast US were ditched by the late 1930s, most marsh research after that time has accepted ditches as a given feature of the marsh. Therefore, modern research has tended to avoid taking a hard look at the impact of ditches — except in the mosquito control portion of marsh research. As there are options to ditch maintenance such as impoundments and OMWM for modern mosquito program managers, the technical literature aimed at this audeince has included an interest in determining what impacts result from ditches and whether other management means have fewer impacts. This technical literature has been expanded recently due to increases in opportunities for "restoration" of ditched salt marshes. However, it is clear that the issue lacks factual, disinterested research on key elements of concern; this makes informed speculation often the only means of making choices regarding important issues regarding the impact of ditching on salt marshes.

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4 Open Marsh Water Management Practices and Impacts

4.1 Definitions

Open Marsh Water Management (OMWM) is a guild of techniques that has been developed with the intention of controlling mosquito production, while at the same time avoiding environmental impacts associated with ditching. OMWM accomplishes this through physical and biological manipulation of marsh conditions. OMWM is intended to enhance habitat for fish, such as the mummichog, that consume mosquito larvae, and increase access for the fish to potential mosquito breeding sites. The implementation of an OMWM system may also partially restore water levels to pre-ditching variability. OMWM is, therefore, often classified as a means of salt marsh restoration (Wolfe, 1996).

Prior to implementing an OMWM system, information from the marsh in question must be collected in order to effectively develop a successful design. An OMWM design depends on the baseline physical, chemical and biological conditions present on the salt marsh. Key variables include: the number and location of mosquito breeding locations on the marsh; the water depth over the marsh during flooding events and the number of these each month; the distribution of dominant plants on the marsh and along the upland edge; the levels of dissolved oxygen, temperature and salinity in ditches and pools; and a qualitative understanding of the freshwater and saltwater flow patterns. These data also support post-project assessments of impacts to the marsh subsequent to the implementation of the OMWM (Wolfe, 1996).

When properly designed, a typical OMWM system has a life expectancy of 20 years or longer (Lent et al., 1990). OMWM ditches and ponds are not expected to deteriorate as rapidly as grid ditch systems, because they tend to remain flooded and are less subject to fluctuating water levels which lead to erosion (Lent et al., 1990). An OMWM system may only require periodic maintenance once the system is in place.

4.1.1 Aims and Purposes

OMWM is based on the following assumptions:

- not all parts of a tidal marsh breed mosquitoes

- mosquitoes are greatly reduced or absent from portions of the marsh where tidal action circulates water over the surface and removes excess water

- biological control in the form of predation by marsh fishes will reduce mosquito populations

- permanent pools of water on the marsh surface serve as reservoirs for mosquito-eating fish

(Daiber, 1986)

States adjacent to New York (New Jersey, Connecticut, and Massachusetts) rely on OMWM or OMWM-like methods as primary means of water management for controlling mosquito populations (Wolfe, 1996).

Several projects on certain Long Island wetlands have used OMWM principles or OMWM-like techniques, such as the plugging of drainage ditches (see, Lent et al., 1990). However, none of these projects have constituted a comprehensive demonstration project of standard OMWM, involving the excavation of fish reservoirs or establishment of shallow spur ditches, and none of the projects were implemented with the overt intention of demonstrating mosquito control feasibility.

4.1.2 OMWM Engineering Options

OMWM installations typically utilize some existing ditches either in whole or in part, completely fill in other ditches, and create small ponds to provide a valuable habitat for larvivorous fish and a variety of aquatic birds. Three types of OMWM systems have been generally classified:

- open systems
- semi-open systems
- closed systems.

The classes are determined by the degree of connection to the estuary. An open system consists of tidal ditches connected to relatively deep tidal outlets which, when combined with spur ditches, permit daily tidal exchange. A semi-open system consists of full-depth ditches with a shallow tidal outlet or sill combined with lateral spurs, creating a system with some, but not complete, tidal exchange for each tidal cycle. Closed systems contain shallow pools, deeper reservoirs, and pond radial ditches with no associated tidal outlet (Dale and Hulsman, 1990). Tidal exchange in a closed system occurs only during spring or storm tides.

4.1.2.1 Ponds and Spurs

4.1.2.1.1 New

OMWM involves the selective excavation of shallow ponds in mosquito breeding areas to provide habitat for small, resident marsh fish (typically, mummichogs) that prey on mosquito larvae. The construction of artificial ponds provides permanent open water habitat on high marsh areas. These areas typically lost open water features when grid ditching was installed. Therefore, this part of an OMWM installation is often described as an ecological restoration effort. A typical OMWM pond design has a maximum depth of 36 inches, with gently sloping edges. This depth acts as a reservoir, so as to keep the larvivorous fish on the marsh and alive during low tide. OMWM ponds often are not connected directly to tidal channels to prevent drainage during low tides. In addition to providing habitat for fish that consume mosquito larvae on the surrounding marsh, OMWM ponds provide a valuable habitat for ducks, shorebirds, and wading birds (USFWS, 1998).

The intent of an OMWM design is for fish to leave constructed ponds during high spring tides to feed on mosquito larvae in high marsh areas. In order to ensure fish access to breeding areas on the high marsh, shallow ditches, known as spurs, are constructed off main ditches and/or fish reservoirs. Spurs are constructed at least 18 inches deep and 24 inches wide (Hruby and Montgomery, 1986); they are designed to provide reservoir dwelling fish access to all potential breeding sites within the marsh, while not significantly altering the height of the water table. They are a key design feature for mosquito control. Spurs are never connected to each other or to another ditch, besides the main ditch; this avoids complete drainage of a panel should a plug fail.

Ponds are sized differently in various jurisdictions. Ponds in New Jersey tend to be smaller than those installed in Connecticut. Larger ponds tend to provide more waterfowl features than smaller ponds do. In Connecticut, marsh restoration is an explicit part of OMWM installations; in New Jersey, mosquito control commissions are explicitly required to only address mosquito control issues (Warren et al., 2002; NJDEP, 1980).

Figure 3. OMWM pond on a Connecticut marsh

4.1.2.1.2 Sills

OMWMs are often installed in marshes that have been grid ditched. Often, it is the intent of the OMWM to undo some of the environmental impacts associated with grid ditching. One of those impacts is a lowering of the marsh water table because of drainage of the water table. OMWMs can address this change by including sills in the design. Sills are plugs in ditches that are completed short of the marsh surface. They are intended to retain water in the ditch, and thereby prevent dewatering. Sills are often used in areas of large shallow salt pannes to create a semi-tidal OMWM system, where a small four to six inch rise and fall in accordance with daily tides is created within the ponds and spurs. Sills are usually 50 to 100 feet long and are placed at or near tidal ditch outlets to a depth of approximately four to eight inches below the high marsh surface (USFWS, 1998). Sills allow excess ephemeral sheetwater to be removed from the marsh surface during ebb tides, while maintaining the subsurface water table level (Lesser, undated). This method supports fish habitat on or near the surface of the marsh while still allowing for more water and nutrient exchange with the estuary compared to a closed or non-tidal system. Sill ditches that have a gradual slope are less desirable in high marsh areas because they will result in the lowering of the marsh's tidal and semi-permanent water levels (USFWS, 1998).

4.1.2.1.3 Full Ditch Blockages

Full ditch blockages are created by damming ditches completely to the level of the existing marsh. These dams are usually installed at the mouth of the ditches, and are most effective if constructed 50 to 100 feet long. Full ditch plugs are intended to maintain salt water behind the dams because salt water displaces fresh water in the ditches due to greater density when the tide overruns the plug. Full ditch plug systems are intended to raise water table levels as high as possible. The saline water and increasing water table are thought to be effective for *Phragmites* control. The ditch itself will serve as a fish reservoir (James-Pirri et al., 2001). The technique has been criticized for isolating resident fish populations from their predators.

In certain cases, old parallel grid ditches that are not incorporated into OMWM systems will no longer be routinely cleaned, but instead will be allowed to naturally fill in with tidally-borne sediments. In other OMWM scenarios, existing mosquito ditches are completely filled in with spoil to effectively achieve areas of open water on a marsh surface (USFWS, 1998).

On Mosquito Beach in Rhode Island, OMWM techniques that were put into practice in an effort to reduce mosquito populations consisted of filling unnecessary mosquito ditches with excavated spoil, combined with deepening and damming an existing fish reservoir (James-Pirri, 1998).

On Little Pine Island in Florida, the filling of approximately six miles of mosquito ditches was proposed in an effort to restore the native vegetation in the marsh. As of 2001, three miles of mosquito ditches were filled in and the marsh restoration was classified as a success (Rathbun, 2001). Sections along the ditches where native vegetation did not naturally take hold were planted with native species.

OMWM has also been implemented in fresh water settings. Wetland restoration projects in Wisconsin are also implementing OMWM techniques that include the filling of old mosquito ditches. The Compton Wetland Restoration Project and the Jefferson Marsh will undergo OMWM alterations that include full ditch blockages in 2004 (NEPAC, 2004). In Columbia

County, a wetland restoration project took place on a marsh consisting of a 90-acre basin crisscrossed by a series of drainage ditches that were constructed in the 1930s. Beginning in 1996, the most critical portions of these ditches were filled over a four-year period. Water levels on the marsh rose with the combination of ditch filling and a temporary water control structure, but began to stabilize three years after construction. The marsh became covered with open water and as a result, lake sedge (*Carex lacustris*) and tussock sedge (*Carex stricta*) flourished in the saturated soils (Thompson and Luthin, 2004).

4.1.2.1.4 Runnels

Runnelling is a mosquito management tool that consists of a network of very shallow spoon-shaped channels that connect pools to each other and to the tidal source (Hulsman et al., 1989). Runnelling alters the marsh as little as possible while causing significant reductions in mosquito numbers. Whenever possible, runnels are designed to follow natural patterns of water flow. This OMWM technique facilitates fish predation on larvae and, in some cases, physically transports larvae off the marsh. The flushing of larvae more likely occurs if water flow is relatively slow, since larvae avoid rapidly flowing water.

Runnelling is a simple form of water management whose long-term effects appear to be minimal (Dale et al., 1993). It is best suited for marshes with simple and defined water movement patterns, and where the length of the runnel is relatively short (Dale and Knight, unpublished).

4.2 Manuals

4.2.1 Ferrigno & Jobbins (1968)

Modifications to ditching to minimize environmental impacts were being described even as ditching was first being widely utilized (Smith, 1904 [out-of-print and unavailable; as quoted by Wolfe, 1996, and confirmed by Crans, Rutgers University, personal communication, 2004]). In Delaware, the practice of "quality ditching" was described in the late 1930s, and was used in many of the nature preserves in that state (Cottam, 1938). However, "Open Marsh Water Management" as a method for controlling mosquitoes was codified by Ferrigno and Jobbins in the late 1960s; their work has since been used as the basis for subsequent OMWM and marsh restoration projects. This is because Ferrigno and Jobbins established a state-wide program through their efforts, and, through the process of renaming the technique, spurred its widespread adoption and adaptation throughout the northeast US.

They proposed that instead of draining a marsh through the use of parallel and grid ditches, "quality ditching" and water management on an open marsh environment would effectively reduce mosquito populations for a longer amount of time and would be beneficial to the entire marsh ecosystem.

According to Ferrigno and Jobbins, in order to obtain complete mosquito control for longer periods of time, every breeding and potential breeding depression on the marsh has to be identified prior to implementing any OMWM techniques. Each depression must be connected to a tidal ditch to allow tidal circulation, or to some other kind of permanent body of water, to insure access for larvivorous fish. Deeper ditches were recommended because they are more efficient at transmitting water; therefore, they provide better circulation and greater degrees of tidal inundation, and tend to be more persistent marsh features. If any permanent water areas were apparent, such as ponds, it was recommended that they should be preserved and isolated from the rest of the ditching system to ensure they maintained water levels and served as effective fish reservoirs.

In order to achieve an effective OMWM system, Ferrigno and Jobbins advised the adherence of the following precautions:

- Quality ditches should be constructed at least two feet deep in order to have water flows reach low marsh areas. Deeper ditches (more than three feet deep) are preferred when reaching high marsh areas.

- Mains should be connected on both ends to tidal ditches or band ditches. Band ditches are recommended along the upland edges with spoil placed on the upland side at irregular intervals.

- Lateral ditches should be straight and connected at both ends, to prevent silt deposition.

- Ditches with a gradual decrease in elevation will lead to revegetation of the ditch bottom, leading to re-isolation of breeding depressions, and so lead to renewed mosquito breeding.

- Spoil should be graded with the marsh surface to provide the least interference of water moving over the surface of the marsh.

4.2.2 Audubon - Massachusetts Manual

In Massachusetts, OMWM design selection depends on the specific physical and biological characteristics of the marsh. The main characteristic in determining the need for OMWM alterations is the number of mosquito larvae present on the marsh during the breeding season.

For example, in Essex County, OMWM is implemented only if at least three broods are observed during the summer, and the average larvae dip count is greater than five. If two broods are observed with this high average number, another season of monitoring takes place before a final decision is made. Before a marsh in Essex County is considered for OMWM, additional information regarding the level of spring tides in breeding areas, as well as the distribution of the existing vegetative communities on the marsh and upland edge are well documented. A qualitative understanding of fresh and salt water flow patterns, levels of dissolved oxygen in existing ditches and pools, and salinity and temperature measurements in major bodies of water are also components of pre-monitoring efforts.

According to Hruby and Montgomery's OMWM manual (1986), OMWM reservoirs on Massachusetts marshes must be constructed to a depth of three feet and a width of two feet. Vertical sides for reservoirs are preferred in order to eliminate shorebird predation. Reservoirs are placed within 55 yards of breeding areas and are at least 100 square feet in surface area. Preferably, reservoirs are constructed from existing tidal pools, existing perimeter ditches, or existing ditches. If none of these exist, new reservoirs are dug at the edge of shallow permanent or temporary pools, or existing depressions.

Spurs are constructed 18 inches deep and 18 inches wide and extend from the middle of a reservoir to an edge of a breeding area. Shallower spurs are not recommended. A 95 percent reduction in larval numbers was achieved when spurs were within 75 feet of each other in large mosquito breeding areas (Hruby and Montgomery, 1986).

The Massachusetts OMWM manual further recommends that ditch plugs be constructed from dredged spoil to a length of at least 50 feet long, and should be four inches to six inches above the marsh surface (due to eventual subsidence of the emplaced material). In areas where muskrats are active, plugs are constructed 100 feet long to prevent the animals from burrowing through the plug to the tidal channel. Excavated spoil from digging or cleaning operations are not set on the marsh surface if it will raise the surface above the level flooded by spring tides. No more than three inches of spoil on the marsh surface is permitted during the disposal of spoil.

4.2.3 Rhode Island Manual (Christie, 1990)

This succinct manual addressed salt marsh geology and ecology, mosquito ecology, and OMWM theory, preparation, permitting, construction, and monitoring in 30 pages (including appendices). Despite its brevity, it did contain some at least one unique point. It included an OMWM justification model attributed to Sjogren and Genereux (1987). This is a formulation of an OMWM index:

I = MA x SC x AC

Where

MA = percent of the marsh generally capable of breeding mosquitoes

SC = number of field visits where dip counts exceeded five per dip (sufficient count)

AC = average mosquitoes per dip in the sufficient counts

If the index (I) exceeded 100, an OMWM may be justified.

The Rhode Island manual supported the use of full ditch plugs (closed systems), with an edging ditch installed to minimize freshwater intrusion. Ponds tended to be large, up to 100,000 square feet, which is more than two acres.

4.2.4 Long Island Region Tidal Wetlands Management Manual (Hruby, 1990)

This report was designed to provide general guidelines towards best management practices in tidal wetlands for Long Island. Although generic in nature, and intended to address all forms of wetlands management, the manual was pointed at OMWM implementation as a general mosquito management and preferred wetlands restoration tool. It defined specific conditions that made a salt marsh a good candidate for OMWM:

- More than 80 percent vegetated
- Excessive mosquito breeding
- Salinity in surface waters above 15 ppt
- Marsh surface flooding more than three times per summer

The manual also laid out pre-implementation monitoring to address these criteria. The preferred OMWM implementation, as the manual was based on the Seatuck National Wildlife Refuge demonstration project, was to create fish reservoirs and use ditch plugs, and so was a closed system. Post-project monitoring, including mosquito larvae surveys, ditch and pool dissolved oxygen, temperature, and salinity measurements, and a vegetation survey, was also described.

4.2.5 NYSDOS/NYSDEC (Niedowski, 2000)

The *Salt Marsh Restoration and Monitoring Guidelines* Report was compiled in December 2000 through a joint effort between NYSDOS and NYSDEC. The document serves as a framework for New York salt marsh restoration activities, including planning, design, implementation, and monitoring for restoration projects sponsored by municipalities. The goal statements for habitat restoration in New York State are summarized as follows:

- To the greatest extent practicable, achieve functional, community, and/or ecosystem equivalence with reference sites when undertaking restoration.
- Restore critical habitats for priority fish, wildlife, and plant species, including those listed as threatened, endangered, and of special concern by Federal and State governments, and species of historical or current commercial and/or recreational importance in New York State.
- Plan and implement restoration initiatives using a regional perspective to integrate and prioritize individual restoration projects and programs.
- To the extent practical, use historical acreages, proportions, and/or spatial distributions to prioritize habitats from a state or regional perspective.
- To the extent practical, ensure where appropriate that historical acreages, proportions, and/or spatial distributions of priority habitats are restored and preserved.

Two desirable OMWM techniques described in the manual are closed systems and semi-tidal systems. According to the guidelines, closed systems should consist of shallow ponds and pannes ranging from two to 18 inches deep, sump ponds ranging from 30 to 36 inches deep, and pond radial, spur ditches approximately 30 inches deep. Ponds with gentle slopes are recommended in areas where mosquito breeding is evident. More shallow areas may be constructed in a pond for shorebird foraging areas. Excavated spoil resulting from pool and ditch creation is recommended to be used to raise the bottom of ditches, and for plugging ditches. The use of rotary ditching equipment is advised to minimize the impacts of spoil disposal. The semi-tidal systems are described as consisting of 30 inch deep ditches with sills that are only partially tidal. A sump pond and connector ditch system is recommended for semi-tidal systems as well.

When using biological methods as part of OMWM techniques for mosquito control, the introduction of non-native mosquito fish into New York salt marshes is not discouraged in the *Salt Marsh Restoration and Monitoring Guidelines.*

4.3 OMWM Examples: Salt Marshes Outside the Northeast US

4.3.1 Australia

Runnelling was implemented on a 0.5 hectare section of an eight hectare tidal salt marsh on Coomera Island, Australia, in an effort to reduce mosquito larvae. Runnels were connected to isolated pools, to other runnels, and to the tidal source. When possible, runnels were constructed to follow natural water movement. Although not immediately effective, this method successfully reduced larval numbers to below nuisance levels for the 6.5 year study period. Runnel depths were observed to vary due to erosion and deposition, resulting from the slope of the runnel. For the first four to five years of the study, water table height and substrate moisture increased, and salinity decreased (Dale et al., 1993).

On Kooragang Island, tidal flushing was restored as part of the Kooragang Wetland Rehabilitation Project in 1995. The primary objective of this project involved the removal of culverts to improve fisheries habitat. Mosquito eggshell densities, a measure of mosquito breeding, decreased in areas affected by the culvert removal, as compared to reference marsh areas. The increase of tidal flushing resulted in vegetation patterns and mosquito eggshell densities that typically occur in a more frequently inundated saltmarsh-mangrove complex (Turner and Streever, 1999).

Alterations to three salt marsh sites in Queensland involved runnelling, OMWM, and grid ditching. Results over a three-year post-monitoring period identified impacts on the water table, substrate and vegetation, and crab activity. The runnelled site demonstrated the fewest number of environmental impacts, while the grid-ditched marsh showed the most impacts. The type of OMWM techniques used in Queensland involved a ditch, approximately 24 inches deep, which linked mosquito breeding areas to an old channel that connected to the Noose River. OMWM alterations resulted in higher pH levels, slightly higher substrate salinity, and a decrease in the density of *Sporobolus*, which is an invasive grass in Australia. The OMWM site did not significantly effect the water table salinity, and crab activity increased (Dale and Knight, unpublished).

4.3.2 Florida

For the past 20 years, the Florida Coordinating Council on Mosquito Control (FCCMC) has been practicing ditching as part of its source reduction techniques. Ditching networks in Florida are constructed to connect shallow ditches to permanent water habitats. Permanent ponds are also constructed in areas where it is impossible or impractical to connect ditches to major waterways. Dewatering of the marsh has been avoided through the implementation of ditch sills installed at mean high water. Installation of the ditch-pond networks has decreased the need to larvicide these areas of the marsh (FCCMC, 1998).

OMWM methods in Florida have been modified to suit the low tidal amplitudes of the Indian River. The use of both open and closed systems, variations in the placement and height of sills, and the use of graded, meandering ditches, have been adopted to accomplish specific management needs throughout the state. OMWM techniques involving the creation of a fish reservoir with associated spurs have been applied in Hillsborough County (Carlson, 1991).

4.3.3 Maryland

Water management for mosquito control in Maryland began in 1933, with concentration on tidewater marshes (Lesser et al., 1978). Early wetland restoration efforts in Maryland focused on restoring hydrology on previously grid-ditched marshes. The installation of ditch plugs, small berms, and water control structures has been used with much success (USFWS, 2001).

A large-scale study was conducted in 1978 in order to determine the effects of three various OMWM techniques on high marsh wetlands in the Maryland portion of Chesapeake Bay. Significant changes in vegetation, nutrient concentrations, and water table levels had occurred at the three manipulated sites (Whigham et al., 1982). Two high marsh areas in the Chesapeake Bay area were treated with tidal (open), semi-tidal (sill), and non-tidal systems (closed). The closed system was reported to have had the least change in plant community structure when compared to open and semi-tidal systems. One year after the excavations, *Iva frutescens* was observed invading some of the sill and open systems of the marsh (Wolfe, 1996).

Since 1998, Maryland has focused on wetland restoration through the recreation of microtopography, such as small ridges and swales on the marsh surface, to create a more diverse soil moisture regime (USFWS, 2001). In some cases, straw and hay have been extensively used to stimulate the denitrification process and to provide optimum substrates for aquatic invertebrates. Presently, wetland management, or source reduction, is not a significant program component for Maryland's marshes due to wetland protection regulations and opposition by the Maryland Department of Natural Resources (DNR) and USFWS (Maryland Department of Agriculture, 2003).

4.3.4 Delaware

Modified ditching (quality ditching, as it was called) has been utilized in some wetlands in Delaware since the 1930s, to address conservation officials' concerns that standard water management could reduce waterfowl habitat quality (Cottam, 1939). The state, through the Delaware Mosquito Control Section (Division of Fish and Wildlife), has been applying OMWM techniques since 1979 to mosquito breeding marshes throughout the state with much success (Wolfe, undated). By the mid-1980s, 6,000 acres of previously parallel grid ditched marshes in Delaware had been treated with OMWM (Meredith et al., 1985). More than 28 percent (4,200 acres) of Delaware's salt marsh mosquito breeding habitats have been eliminated from aerial chemical insecticide treatment as a result of OMWM activities (Wolfe, 1996). Lesser (undated) showed that functioning OMWM systems in Delaware resulted in a 95 to 98 percent reduction in mosquitoes.

OMWM practices used on Delaware marshes include open tidal systems with restricted tidal exchange, and closed nontidal systems. The type of OMWM technique used is largely based on the type of mosquito breeding being addressed and concerns regarding long-term water quality within OMWM ponds and ditches. The most common OMWM technique implemented in Delaware includes infrequently flooded or semi-tidal permanent bodies of water in high marsh vegetation (Lesser, undated). Open tidal ditches are used in a very limited capacity due to the undesirable effects on hydrology and vegetation that may result from excessive drainage. Mosquito breeding areas found in large shallow pannes are treated with a sill outlet to allow the surface sheetwater to drain during ebb tides, while still maintaining groundwater levels. Excavated spoil material is deposited on-site to fill adjacent mosquito breeding potholes, or is thinly spread across the marsh surface so as to not impact existing vegetation (Wolfe, 1996).

By 1994, approximately 1,260 of 1,350 potential breeding areas at Prime Hook National Wildlife Refuge had been treated with OMWM. A total of 234 ponds were created, providing over 19 acres of open water habitat, with the intent of reducing or potentially eliminating the use of insecticides for the next 20 years (Wolfe, 1996).

In 2001, the USFWS initiated a three-year study of OMWM throughout Region 5, the northeast US. The study was intended to use a rigorous BACI (Before-After, Control-Impact) study design to determine the ecological impacts of ditch plugging, which was the predominant form of OMWM for the sites selected. The study was somewhat impacted because almost all of the selected sites had, in fact, been plugged prior to the study start. However, it is still a comprehensive, multi-site, multi-parameter assessment of the effects of ditch-plugging, one that uses control sites and multiple years of data collection to offset some of the variability among individual marshes. In Delaware, the study was set in Prime Hook National Wildlife Refuge, using sites in Petersfield (6.5 hectares, originally sill ditched in 1989, but now extended to full ditch plugs in 2002) and Slaughter (a 6.5 hectare site, with ditch plugs installed in 1992 – those that failed were replaced by sill ditches in 2002) (James-Pirri et al., 2001; James-Pirri et al., 2002).

Figure 4. Plugged ditch

4.3.5 New Jersey

OMWM is the major source reduction technique used by coastal mosquito control agencies in New Jersey. OMWM techniques were initially developed in New Jersey through the cooperative efforts of the coastal County Mosquito Commissions, the New Jersey Division of Fish, Game and Wildlife, and Rutgers University (Ferrigno and Jobbins, 1968). In 1980, New Jersey published *Standards for Open Marsh Water Management*, which was adopted by both state and federal regulatory agencies for use when evaluating applications for water management projects on salt marshes (NJDEP, 1980).

Three basic OWMW types used in New Jersey involve the construction of tidal ditches, ponds, and pond radials, also called spurs. These techniques are confined to high marsh areas vegetated by *S. patens* and *S. alterniflora*. Since 1970, several thousand hectares of salt marsh have been treated with OMWM techniques and larvicide applications have been eliminated (Barnegat Bay National Estuary Program, 2001).

Egg Island Fish and Wildlife Management Area located in Cumberland County was chosen to determine the effects of OMWM on mosquitoes in the late 1960s. The marsh vegetation was primarily *S. patens* and was riddled with thousands of depressions created by large populations of wintering snow geese. Following a three-year, post-OMWM monitoring period, it was estimated that for every 1,000 acres altered, 40 to 60 billion mosquitoes would be eliminated annually without the use of larvicides for as long as the ditching remained effective (Ferrigno 1970).

The Bombay Hook Wildlife Refuge marsh project had a signature design of “natural” pools. These are not intended to entirely drain the surrounding area and generally have irregular shorelines. The OMWM also included the construction of “blind sumps,” which are deepened potholes with radiating ditches dug to low portions of the surrounding area to facilitate drainage, and “champagne pools,” which are similar to blind sumps, but with a controlled outlet to the estuary. Bodola (1969) studied the effectiveness of these ponds on mosquito control and reported that all of the pools studied were effective in reducing the number of mosquitoes produced.

CA personnel visited OMWM sites in Ocean County in May, 2004. Ocean County has approximately 27,000 acres of tidal wetlands, much of which are managed by USFWS. Ocean County Mosquito Control believes most of its breeding problems come from the inundation of high marsh, where water is not completely transported away at the end of the tidal cycle. This could be due to hummocky terrain, clogged ditches, berms, and tidal restrictions. When a trouble spot is identified, a standardized approach is used to address it via OMWM.

Stakes are arranged in various ways at the site by supervisory personnel to indicate pond areas, spoils deposition areas, and ditch cleaning/construction areas. The operator of the machine, typically a rotary ditcher, has a great deal of latitude in following these broader guidances. It may be that too much material is produced from pond construction to follow the spoils plan, or that the construction of the pond requires alteration due to on-the-ground conditions. The experience of the operator and the continuity of supervision allow the operator to meet the intent of the plan without following instructions exactly. The general plan of action is to excavate ponds in the densest areas of mosquito breeding, fill in hummocky areas with spoils, and, through ditch construction and maintenance, ensure there is tidal flow to the region following the work. Ponds tend not to be connected directly to the ditches. Ponds tend to be small, generally, room sized rather than substantial portions of acres. The ponds are sinuous, multiple-pass ditches that close back in on themselves, creating and island or islands. A lip is created along the outer edge of the pond, and otherwise depths are on the order of three feet.

Figure 5. New Jersey marsh after OMWM

OMWM sites that were five or more years old had natural-appearing features, supported fish, and seemed to have persistent features that needed no maintenance. The sites tended to revegetate with the surrounding vegetative community, although there were some transitions from high marsh to low marsh in areas where tidal circulation was increased. The use of spoils to fill the hummocky areas meant that many sites, even those three years old, had very extensive bare spots. The degree of barrenness is a function of whether the site supported vegetation prior to the work, and the depth of the spoils placement. Improvements in tidal circulation, together with aggressive mowing in places, appeared to keep *Phragmites* in check, and sometimes to cause retreats.

Project success is measured in terms of larvicide application reductions. Each mosquito season, Ocean County maps the number of times each marsh tract is larvicided. Areas that have had OMWM installations show large reductions in applications each year, although elimination of larvicides is generally not achieved.

4.4 OMWM Examples: Salt Marshes of the Northeast US

4.4.1 Connecticut

In 1985, Connecticut determined that its practice of ditch maintenance should be gradually replaced by OMWM installations. This was adopted, not only as a mosquito control practice, but as part of an overall salt marsh restoration program. In fact, many Connecticut OMWMs are installed primarily for wetlands reclamation or restoration purposes, rather than as mosquito source control. Connecticut now refers to its efforts in tidal wetlands as Integrated Marsh Management. It is comprised of four major, interwoven components:

- mosquito management (mostly, OMWM replacing ditches)
- marsh restoration (tidal connection improvement and marsh re-creation)
- vegetation management (predominantly, *Phragmites* control)
- public education

(Wolfe et al., undated)

Connecticut has a rigorous site approval process, albeit one where the structure and content of the site review has been optimized over 15 years of experience. Although largely internal to the Connecticut Department of Environmental Protection, other involved stakeholders including those from interested federal parties (e.g., USFWS and Army Corps of Engineers) are included. Designers attempt to reconcile potential conflicts between technical experts; common sources of disagreement are the views of bird and marsh vegetation natural resource specialists, as gains in bird habitat often occur at the expense of wetlands plant acreage. Following a preliminary design of a project, at least one extensive site visit is made by all of the participants in the review process. The design is then altered, using consensus as the means to ensure optimization (P. Capotosto, CDEP, personal communication, 2004).

Sites where OMWM has been implemented do not require larviciding, and maintenance of the installed structures has not been necessary. Connecticut's preferred OMWM technique is the use of full ditch plugs coupled with constructed open water areas. Sill ditches may be used to connect ponded areas to breeding sites. Improvements in waterfowl habitat have been the most notable environmental impact, although, as part of the Integrated Marsh Management program, Connecticut does not like to single out particular aspects as having primacy over others (Wolfe et al., undated). Paul Capotosto (CDEP, personal communication, 2003) notes that none of the projects completed since 1985 have required maintenance to date. Also, none of the OMWM sites requires regular larviciding, and, except for instances associated with unusual environmental conditions such as exceptional rains or tides, none of the sites requires any larvicide applications.

Figure 6. Aerial view of an OMWM marsh in Connecticut

4.4.2 Rhode Island

The salt marshes in Rhode Island are not as extensive as New Jersey or Delaware marshes, ranging from two to 150 acres in size). As early as 1937, it was recognized that standard ditching should be modified as the primary means of mosquito reduction. A new focus was initiated to bring more water onto the entire marsh surface instead of draining the marshes through ditches (Price, 1938).

All ponds and pot holes throughout the marshes on Prudence Island were connected by shallow, 15 to 18 inch-wide ditches in 1937. One main outlet was cut in each area to Narragansett Bay. Each tide completely flushed all the ponds and pot holes, and delivered a new supply of minnows. These ponds and pot holes were free of mosquito larvae within a year after the alterations to the marsh; this was not the case in a marsh where no OMWM had been applied. Additionally, the water table appeared to be restored to its pre-ditching levels, and ponds that formerly were stagnant and dried out were supplied with water on every tide.

Christie (1990) generated a manual for Rhode Island marshes, based on experience at one site in-state, the Seapowet Management Area, Tiverton, and the 1986 Massachusetts-Audubon manual. The manual called for ditch plugging with pond creation – albeit, ponds were to be of modest sizes, befitting the generally smaller size of most Rhode Island salt marshes.

4.4.3 Massachusetts

Marshes in Massachusetts that breed mosquitoes are rather small in scale and, therefore, are not best suited for the construction of large OMWM ponds. Instead, small reservoirs are created by digging ditches that are approximately three feet deep, and 18 inches wide. During the reservoir construction, old ditches that are open to tidal flow are cleaned out and remain open. This allows the sediments in the ditch to settle, allowing the ditch to become oxygenated. After a month, the seaward end of the ditch is plugged to the level of the marsh surface with a spoil plug. Old upland perimeter ditches are preferred for reservoirs due to their proximity to major mosquito breeding areas (Hruby, 1985). Radial ditches are constructed, 18 inches deep by one foot wide, to connect mosquito breeding sites to the reservoir. The radial ditches are not connected to the tidal channels, reducing the potential to drain the water table (Hruby, 1985).

Three salt marshes on Nantucket were treated with water management techniques in the winter of 1992-1993 for mosquito control. These marsh sites consisted of Eel Point, Warrens Landing, and Madaket Ditch. Eel Point was treated with OMWM techniques that involved the transformation of an overgrown ditch into a reservoir and a radial ditch, combined with the backfilling of the remainder of the ditch with spoil. Existing mosquito ditches were re-opened at Warren's Landing and a spoil ridge blocking the marsh from tidal channels was cut. At the Madaket Ditch marsh, existing ditches were re-opened to channel freshwater through the salt marsh, an OMWM system was created throughout the marsh, and a spoil ridge from original ditching was cut. All three installations appeared to be successful, as mosquito breeding was virtually eliminated within one year (Christie, 1993).

Parker River National Wildlife Refuge had a 3.5 hectare area plugged and radially-ditched in 1994. An additional two sites, comprising 16 hectares in total, were similarly treated in 2002, as part of the USFWS Region 5 study of OMWM impacts (James-Pirri et al., 2001; James-Pirri et al., 2002).

4.4.4 Maine

The tidal marshes of the Gulf of Maine consist of saltmarshes that are periodically exposed and flooded by salt water through tides and storms. Hundreds of restoration projects have been completed there. However, historically, sufficient information has not been compiled to adequately track these projects (Cornelisen, 1998). Long-term evaluation of the state's restoration projects is inhibited due to the absence of baseline data and inconsistencies in data collection.

A study at Rachel Carson National Wildlife Refuge evaluated the response of three salt marshes, Granite Point Marsh, Moody Marsh, and Marshall Point Marsh, to the practice of ditch plugging. The study focused on the effects of ditch plugging on marsh hydrology, sedimentation and marsh development processes, vegetation patterns, and utilization by nekton and birds. As a result of the ditch plugging, water table levels and standing water increased. Vegetation observed in this study shifted from *S. patens* to *S. alterniflora* at Granite Point and Marshall Point. No significant vegetation change was noted at Moody Marsh. Nekton species richness, total fish density, total decapod density, and nekton community structure were unaltered following ditch plugging at both Moody and Granite Point marshes. When compared to the associated control marshes, nekton richness and density were greater at Marshall Point, and total fish abundance and bird species richness were greater at Granite Point (Adamowicz and Roman, 2002).

4.4.5 Long Island

In the early 1980s, OMWM pilot studies were conducted on the salt marsh at Seatuck National Wildlife Refuge aimed to reduce mosquito numbers with minimal damage to the marsh community and to reduce dependency on chemical pesticides.

A baseline study was conducted by Cowan et al. to establish baseline data for selected ecological and hydrological parameters. The parameters included:

- the timing of mosquito breeding and larval densities
- resident and migratory bird usage
- vegetative composition and distribution
- soil invertebrate analysis
- nutrient levels relative to tidal cycles
- distribution and abundance of fish species.

Marsh hydrology and topography were also studied to define the primary inputs, outputs, and pathways of water in the Seatuck marsh system. The objectives of this baseline study were:

- to develop a general hydrological model of the marsh
- conduct experimental OMWM alterations suited to local physical and biological conditions
- evaluate the effectiveness of marsh alterations for mosquito control
- assess any environmental impacts resulting from experimental marsh alterations through comparison of baseline data to post-alteration data

(Cowan et al., 1986).

After two years of pre-project monitoring, OMWM alterations were implemented on a test plot at Seatuck and monitored for an additional two years. In addition to mosquito control, the OMWM design focused on redirecting freshwater inputs to the marsh from upland edges, improving tidal circulation between the marsh and Great South Bay, and providing a permanent habitat for native larvivorous fish. The results of OMWM at Seatuck determined that mosquito production was reduced, but not eliminated, without any significant adverse impacts on the marsh.

As a result of this study, NYSDEC produced the manual of methods for OMWM (Niedowski 2000). Afterward, the USFWS constructed a tidal creek to further restore this wetland. Most of this marsh no longer breeds significant numbers of mosquitoes, but one section, known as IS-74, continues to require regular larvicide applications. In addition, the reduction of *Phragmites* combined with OMWM techniques at the Seatuck NWR has resulted in a fivefold increase in shorebird use (R. Parris, LI NWR, personal communication, 2004).

As part of the USFWS Region 5 study, the following sites were selected for study:

- Flanders (two sites, 6.4 hectares total size), plugged in 2001
- the western part of Wertheim National Wildlife Refuge (8.5 hectares), plugged in 1997
- the eastern part of Wertheim National Wildlife Refuge (8.5 hectares), plugged in 1998
- Sayville (9.4 hectares), plugged in 1998

(James-Pirri et al., 2001; James-Pirri et al., 2002)

As part of the development of the Management Plan, CA personnel observed some of the local OMWM sites this spring. Sites visited were West Sayville County Gold Course, Fireplace Neck in Islip, Seatuck National Wildlife Refuge, and the William Floyd Estate in Shirley.

At West Sayville, ditch plugging was done under the direction of Robert Parrish (USFWS). The typical ditch plug was constructed with a small piece of plywood, about three feet long, placed in the ditch with marsh material placed behind it. A small fish reservoir was a common feature just behind the plug. Over time, the plugs became vegetated, and the plywood was no longer visible. Many of the plugs were intact, and caused saltwater impacts to the marsh. The most noticeable impact was an increase in vegetation diversity, including *S. patens* grasslands, especially towards the uplands, and many large ponds. especially towards the bay. Many killifish were visible in the ditches and ponds. There were many birds using the marsh, especially in the areas of standing pools. These pools tended to be only a few inches deep. *Phragmites* are abundant in the upper marsh, but appear to be dying back where ponded water has been maintained. A comparison of older aerial photographs to current conditions showed no major changes in the marsh. However, because the marsh no longer drains at low tides, mudflats have become standing pools. Birds observd during the visit included:

- black duck (in shallow panne)
- green-winged teal (in shallow panne)
- Canada geese (in shallow panne)
- gulls (in shallow panne)
- greater yellowneck (in shallow panne)
- great egret
- snowy egret
- osprey

As part of a multi-agency cooperative venture, in the winter of 1999, a pilot OMWM project was conducted at the William Floyd Estate, which is managed by the National Park Service (NPS). The OMWM consisted of using a series of plywood sheets and plugs consisting of organic matter from the grid ditch substrate. This project's initial activities took place over the course of 10 days, on 200 acres. Pannes or ditches that largely drained at low tide are now linear ponds, and fish, crabs, and invertebrates are now observed in areas that once were breeding mosquitoes.

There were two “hot spots” on the marsh, formerly, where an appreciable amount of mosquito breeding was taking place, that no longer exist.

According to Richard Stavdal (Unit Manager, NPS), the impact on the bird population has been noticeable. Migratory wading birds now find more food sources. Waterfowl can use the flooded salt marsh for brooding, feeding and resting. In addition to the formation of linear ponds, salt pannes began to form in lower elevations of the marsh. The impounded, high salinity waters associated with the OMWM have caused a noticeable decrease in *Phragmites* stands. The ponds created by this project stopped increasing in size after four years, and *Phragmites* are not found in or around these ponds.

4.5 Impacts to Mosquito Control

According to Ferrigno et al. (1975), when properly designed, OMWM should achieve greater than 95 percent reduction in mosquitoes. In a comparison of mosquito emergence in an unaltered marsh and an OMWM-treated marsh, significantly fewer mosquitoes were observed emerging from the OMWM-treated marsh. Prior to that study, Ferrigno (1970) had found that mosquito production fell from 10,000 mosquitoes per square foot to less than a thousand in the first year after OMWM, to zero at the same site in the second year.

4.5.1 Breeding Locations

In addition to allowing fish predation on mosquito larvae, OMWM is likely to interfere with the hatching cycle of mosquito eggs. Water management for mosquito control is based upon three fundamental principles: removal of excess surface water; increasing the amount of standing water; and increasing the movement of water (Shisler, 1978). The numbers of mosquito larvae that survive to pupate as adults on the marsh surface are negatively correlated with both tidal inundation and the number of larvivorous fish (Buchsbaum, 2001).

Numerous studies have shown that OMWM alterations resulted in a decrease in the amount of mosquito breeding locations. Marshes in Delaware, Florida, Rhode Island, New Jersey and Maryland subsequent to OMWM reported success in the reduction of mosquitoes (Wolfe, 1996; Ferrigno, 1970; Daiber, 1974; Lesser and Saveikis, 1979; Hruby et al., 1985). Dale and Hulsman (1990) noted that one impact of OMWM is to reduce the drying out of potential mosquito breeding locations, which disrupts the cycle events needed for successful breeding.

During the first season of OMWM, a Massachusetts marsh had significantly lower numbers of mosquito larvae and pupae when compared to adjacent control sites (Hruby et al., 1985). In New Jersey, for every 1,000 acres of the marsh treated with OMWM, it is estimated that 40 to 60 billion mosquitoes will be eliminated annually for the life of the OMWM system (Ferrigno and Jobbins, 1968). According to Dale et al. (1993), a runnelled marsh in Australia exhibited a decrease in the number of mosquito larvae within the first three months of alterations. Continuing for over six years, they reported a reduction in mosquito larvae to be "below nuisance levels." Similar reductions were reported in North Carolina, Massachusetts and

California marshes subject to OMWM (Wolfe, 1996). After the first year of OMWM alterations at two New Jersey marshes, mosquito breeding was eliminated for five years at one marsh, and two years at the other marsh (Ferrigno, 1970). OMWM alterations to a Connecticut marsh in 2001 resulted in the elimination of mosquito breeding in the trenches and ponds; however, larviciding was required in surrounding areas (Wrenn, 2002). A ninety-five percent reduction in mosquito larvae and pupal population was observed at Fairhill marsh in New Hampshire following the re-designing of existing pannes as OMWM pools to increase the amount of permanent water on the marsh (New Hampshire Coastal Program, 2004).

Freshwater mosquito breeding occurred in lower marsh areas at Seatuck Refuge subsequent to OMWM alterations. It was thought that the alterations, possibly, were preventing rain water from draining off the marsh surface, and that salinity may have decreased in marsh depressions enough for them to become a favorable habitat for freshwater mosquitoes (Guirgis, undated).

Adult mosquito community structure can be studied by analyzing trap collections (Zhong et al., 2003). The mosquito light trap was designed by the New Jersey Mosquito Control Association more than 50 years ago as a surveillance device to monitor mosquito populations. Most mosquito control agencies use light traps in their programs, but have additional tools that provide data to guide their activities (NJMCA, 1997). At the Seatuck Refuge, OMWM alterations did not reduce the average number of female mosquitoes collected in the refuge light trap. Prior to OMWM alterations between 1986 and 1988, the average females per night varied between 28.8 and 34.5. After OMWM alterations in 1989 and 1990, the average number reached 41.7 and 32.2 each year, respectively (Guirgis, undated).

4.5.2 Pesticide Applications

4.5.2.1 Larviciding

Hansen et al. (1976) indicated that larviciding should not be used as a primary mosquito control technique. Problems include:

- The necessity for routine marsh inspections to determine breeding locations, to be made both before and after larvicide applications

- the limited time available to inspect and treat breeding locations after flooding
- coinciding larvicide applications with permissible weather conditions
- accurately applying larvicide to small scattered breeding depressions
- the potential for a mosquito population to develop a resistance to the applied larvicide

Thus, the combination of OMWM with limited larvicide applications is a much preferred means of mosquito control.

A study conducted in Ocean County, New Jersey, demonstrated that water management can result in significantly lower required larvicide applications. Water management alterations conducted on three separate marshes eliminated over 93 percent of the acreage of mosquito breeding (Shisler et al., 1979). As discussed above, larviciding has been eliminated on marshes treated with OMWM in Connecticut, and decreased on OMWM-treated marshes in Florida. The experience in Ocean County, New Jersey (as related to CA), has also been very positive, although the need for some larviciding is usually not eliminated through OMWM.

However, in Seatuck marsh, Long Island, ditch-plugging in a tidally-restricted area did not result in any decrease in larvicide applications (Guirgis, undated).

Overall, the major benefit cited for OMWM, beyond reducing ecological effects associated with standard water management, is to substantially reduce the need for larviciding. In fact, Wolfe (1996) spends some time marshalling evidence that OMWM can be justified economically merely in terms of the savings associated with less frequent larvicide applications. This appears to be the case for the very large marshes of New Jersey, especially, where large amounts of chemicals need to be applied in the absence of water management, but the single capital investment in water management may reduce the amounts used by 90 percent or more.

4.5.2.2 Adulticiding

The primary goal of OMWM is to reduce the frequency of larviciding, by encouraging the consumption of larval mosquitoes by fish. This is also intended to reduce the number of adulticide applications in surrounding areas. However, there is little in the literature that documents a reduction in adulticide use. It should be noted that most mosquito control programs now restrict adulticide applications for salt marsh mosquitoes control to the prevention of disease transmission, especially because of West Nile virus threats.

In one example where adulticiding was explicitly discussed, Montgomery (1998) reviewed the impact of OMWM at Rumney Marsh in Massachusetts. Prior to its implementation, mosquito abatement focused primarily on the use of adulticide. In the 1990s, OMWM techniques were applied to restore the degraded state of the marsh. As a result, mosquito populations decreased and the need for adulticide treatments became rare. The remaining mosquito breeding areas were managed by hand larviciding.

4.6 Impacts to Vegetation

Dominant plants characteristic of high salt marsh areas include *S. patens*, *Distichlis*, and short-form *S. alterniflora*. *I. fructesans* and *B. halimifolia*, *Solidago spp.* and *Phragmites* are typical plants found along the perimeter of the high salt marsh (Nixon, 1982).

4.6.1 Phragmites Control

Sulfides, anoxia, and salinity are known stressors to *Phragmites* growth (Bart and Hartman 2002). Because of the presumed impacts on *Phragmites* by increased salinity, one method of remediating a *Phragmites* invasion is to increase tidal flushing to impacted marshes. Another is to intercept freshwater inflows that decrease salinities. The implementation of OMWM perimeter ditches in the upper edge of a salt marsh has been used to prevent further *Phragmites* encroachment (Buchsbaum et al., 1998). Herbicides are often used in conjunction with other efforts, such as controlled burning, to remove *Phragmites*. However, most marsh managers believe these practices will not be successful as long-term strategies unless the underlying site hydrology is changed at the same time that spraying and burning occur (Mitsch, 2000).

On-going *Phragmites* management efforts in Connecticut have focused on changing the environmental conditions favoring *Phragmites* through OMWM techniques. In 1985, in Clinton, Connecticut, a cooperative program was begun between the Department of Environmental Protection and the Mosquito Control Unit in an effort to restore degraded wetlands on the Hammock River by implementing OMWM. The plan focused on restoring tidal flushing during the summer to maximize the emergent vegetation and minimize the conversion of salt marsh to open water. After the first three years of the program, the annual height reduction averaged one foot. By the fifth and sixth year, *Phragmites* stopped growing, dead shoots no longer persisted, and exposed peat was colonized by salt marsh grasses. Targeted birds, such as egrets and waterfowl, increased as a result of the program (Dreyer and Niering, 1995).

In 2000, Connecticut Wetlands Habitat and Mosquito Management (WHAMM) Program installed OMWM ponds on a marsh dominated by *Phragmites* in an effort to restore marsh vegetation. The *Phragmites* stands were initially sprayed with herbicide and then mulched. Five OMWM ponds were installed and several old mosquito ditches were plugged (Capotosto, 2000).

The results have been favorable, in that *Phragmites* has not been able to re-infest the marsh (Paul Capotosto, CDEP, personal communication, 2004).

Observations of Long Island marshes where ditch plugs were installed suggest they can be effective against *Phragmites*. Personal communications from Susan Adamowicz (USFWS, 2004) indicate that has also been the case at Rachel Carson National Wildlife Refuge in Maine.

4.6.2 High Marsh/Low Marsh Shifts

Most OMWM implementations will not substantially alter the marsh surface elevation or restrict surface water movements. Therefore, there should be no shift in the overall distribution of wetlands vegetation (Wolfe, 1996). Changes in marsh resources are most affected by altered hydrologic patterns and spoil deposition. If the water table of a marsh is excessively lowered, marsh shrubs will likely inhabit the area because of the drier habitat. In addition, if spoil piles are placed on the marsh surface, higher successional plants (i.e., *Iva*) are likely to cultivate on the piles (USFWS, 1998).

According to Mitsch (2000), salt marshes that have been altered to reestablish the hydrologic connections of coastal ecosystems to adjacent bodies of water will reestablish salt-tolerant vegetation, such as *Spartina spp*. Lesser (undated) reached the same conclusions. In addition, Lesser noted that a single tidal ditch transversing a low *S. alterniflora* salt marsh will have no adverse effect on marsh vegetation, and in fact, marsh faunal diversity can increase. However, when a network of open tidal ditches passes through a high (*S. patens*) marsh, this can lead to changes in vegetation.

Marshes on Maryland's eastern shore experienced a vegetation shift toward a high marsh after the installation of open ditches. The excessive drainage associated with the open ditches may account for Maryland's high marsh vegetation shift (Daiber, 1986). In Delaware, wherever OMWM techniques were implemented and the water table dropped five inches or fluctuated widely as in open-ditched high-marsh areas, *Baccharis*, *Iva* and other drier-soil plants such as *Pluchea purpurascens* invaded the ditched area (Daiber, 1986).

However, vegetation changes that do occur with OMWMs may not extend throughout the marsh. A Maryland study showed that *I. frutescens* rapidly colonized a marsh that had been treated with an open ditch system, but did not occur in adjacent closed or water controlled sites (Whigham et al., 1982). When high marsh grid ditches are kept open to daily tidal exchange, *S. patens* is often converted to a mixture of *S. alterniflora* and *S. patens* along the edge of new open ditches (USFWS, 1998; Shisler and Jobbins, 1977a).

Ferrigno (1970) concluded that the standard New Jersey OMWM technique encouraged a shift in vegetation to that of a low-marsh. For example, a mosquito ditched marsh in Tuckerton, New Jersey shifted toward a low marsh community after OMWM implementation (Shisler and Jobbins, 1977a). This shift was attributed to the increase of tidal circulation, and possible nitrogen fixation in the ditched marsh. On the other hand, CA's tours of OMWM sites in Ocean County, New Jersey, generally found no shift in overall vegetation communities from the pre-operational vegetation conditions.

A four-year study at the Seatuck NWR, Long Island, concluded that while vegetation composition in some plots within the altered marsh changed from year to year, there was no clear relationship between observed vegetation changes and OMWM alterations. The analysis was not able to associate the vegetation shifts to any contemporaneous hydrological and salinity changes induced by OMWM alterations to the marsh (Lent et al., 1990).

4.6.3 Aerial Losses/Gains of Vegetation

4.6.3.1 Initially

Ferrigno (1970) reported a reduction in the amount of short-form *S. alterniflora*, as well as *Salicornia* and *Cladophora spp.*, on a New Jersey marsh immediately following OMWM alterations. Increased tidal circulation and the removal of stagnant surface sheet water, which sometimes have been found to promote the growth of these types of vegetation, were thought to be the reason for this vegetation change. No changes in the amount of salt hay grasses *Distichlis* and *S. patens* were reported; increases in the area of widgeon grass (*Ruppia maritime L.*) and sea lavender (*Limonium caroliniamm Walt.*), both beneficial food sources for waterfowl, were reported. An increase in the occurrence of tall-form *S. alterniflora*, bassia (*Bassia hirsuta L.*), sea-blite (*Suaeda linearis Ell.*), sea rocket (*Cakile edentula Bigel*), slender leave aster (*Aster* *tenuifoluis L.*), saltmarsh aster (*Aster subulatus Michx.*), smartweed (*Polygonum aviculare L.*), saltwort (*Salsola pali L.*), and saltbush (*Atriplex harta L.*) were noted near the ditch edges.

Upon restoring tidal flow to a New England salt marsh, the amount of bare area increased as a result of the removal of a small berm, and from the mechanical equipment used to create OMWM pools (Roman et al., 2002). As vegetation (*S. patens, S. alterniflora,* and *Salicornia*) colonized the marsh, the relative cover of bare areas decreased during the second year of restoration.

Following the plugging of ditches at Granite Point Marsh in Maine, high marsh *S. patens* declined after one growing season due to the increase in water cover on the marsh (Adamowicz and Roman, 2002). However, no initial change in vegetation was observed at Moody Marsh, Maine, subsequent to ditch plugging.

CDEP reported good recovery of vegetation following OMWM, with revegetation usually occurring by the end of the first year. The Connecticut intent is (generally) to increase surface water areas in grid-ditched marshes, and so CDEP expects there will be some decreases in the overall number of acres covered by plants (Wolfe et al. undated).

CA's observations in Ocean County, New Jersey, where spoils are cast out over the marsh surface, are that bare areas can persist for several years. However, installations past the initial stage of recovery appear to be thickly vegetated. Ocean County personnel reported that there was no marsh retreat due to the construction activities. Observations of Long Island marshes treated by ditch plugs show that some vegetation can be lost due to the expansion of surface water area; however, the expansion of the surface water area appears to stop after several years.

4.6.3.2 Long-term Trends

The re-establishment of water in the interior marsh areas does not appear to lead to trends of increasing erosion of the marsh surface or other kinds of losses of marsh vegetated areas. In fact, after two years of restored tidal exchange on the New England marsh, vegetation was noted to be developing toward the typical pattern of a southern New England marsh (Roman et al., 2002). An impounded freshwater marsh in New Hampshire showed subtle changes in vegetation only two years after tidal restoration was implemented, with expectations that changes in vegetation will continue (Burdick et al., 1997).

In Connecticut, where tidal restrictions are often addressed together with the installation of OMWM, vegetation recovery was noted to be an on-going process after tidal flow was reintroduced to the marsh (Sinicrope et al., 1990). A 40-year process of vegetation change was observed by Rozsa (1995), where areas of intertidal flats became a low marsh *S. alterniflora* community on a Long Island Sound marsh after the removal of tide-restricting gates.

The Seatuck OMWM on Long Island will be the subject of an intensive retrospective investigation as part of this project. However, anecdotal observations indicate that no major losses in vegetated areas have occurred from ditch plugging 15 years ago. Similarly, none of the other Long Island sites that have had ditch plugs installed appear to have suffered vegetation losses over the past five to 10 years.

4.7 Impacts to Biota

4.7.1 Birds

OMWM has little or no adverse impact on waterfowl habitat, and is generally thought to have positive effects. OMWM ponds are expected to provide a feeding and resting area for migrating waterfowl. Submerged vegetation found in ponds offers an important food supply for wintering ducks (Widjeskog, 1994). Most reports find that marshes altered with extensive networks of pools are utilized by larger bird populations than grid-ditched marshes that have few pools (Reinert et al., 1981; Clarke et al. 1984, Brush et al., 1986, Adamowicz and Roman 2002). Thus, OMWM ponds can improve or restore waterfowl habitat. Montgomery (1998) concluded that the OMWM alterations at Rumney Marsh in Massachusetts, which included the construction of ponds, dramatically enhanced or restored wading shore bird and waterfowl habitat.

Erwin et al. (1994) recommended that the emphasis should be on fewer numbers of large OMWM ponds, defined as being larger than 0.10 hectare. They should be constructed with shallow basins, defined as being less than 15 cm. They should have sloping sides. This design is preferred over a larger number of small, deeper ponds to maximize waterfowl marsh use. Erwin et al. showed that, one year after construction, most water-bird species used the OMWM ponds more often than other water bodies on the marsh, such as natural tidal ponds, creeks, and old ditches. However, when OMWM ponds were located near impoundments, black ducks (*Anas rubripes*) and other waterfowl such as American wigeon, gadwall, and northern pintails, were more likely to utilize the impoundment for nesting, and during the autumn and winter, compared to the OMWM ponds. The large open water areas and submerged aquatic vegetation were thought to be the reason why the impoundments were favored by waterfowl.

At the Egg Island marsh in Cumberland, New Jersey, waterfowl use did not differ from the control marsh, with the exception of greater snow goose (*Chen hyperborean*) and Wilson snipe. Snow goose and snipe numbers were considerably less at the OMWM treated marsh than at the poorly-drained control area (Ferrigno, 1970).

It has been suggested that OMWM does not significantly impact invertebrate populations (Wolfe, 1996). This is perhaps the greatest food source for non-waterfowl birds, and so suggests that bird populations should not be significantly impacted. However, if vegetation patterns are altered, including the loss of woody plants from the high marsh and banksides, birds that rely on those plants for cover may reduce their use of the marsh. Ferrigno (1966) noted that when the marsh ecology is changed by removing the influence of tides, or by blocking the tidal influx by dikes, numbers of clapper rails (*Rallus longirostris crepitans*) and their major food, fiddler crabs (*Uca spp.*), may decline.

OMWM has few immediate adverse or beneficial impacts on salt marsh birds in areas that formerly were ditched (Brush et al., 1986; Grant and Smith, 1998). Although the study conducted by Brush et al. (1986) concluded that OMWM had little impact on bird numbers on a marsh that was previously ditched and converted to an OMWM system, the data were a little more ambiguous. During the first year of monitoring, shorebird numbers increased, but then declined in subsequent years. This decline was thought to be the result of vegetation growth on spoils. The spoils initially provided accessible and plentiful foraging for invertebrates by the birds. However, as the vegetation grew through the spoils, invertebrates were harder to obtain. Brush et al. suggested that bird numbers were more closely related to the number of pannes on a marsh rather than whether it was altered by OMWM, ditched, or remained natural.

OMWM techniques at the Seatuck NWR resulted in a fivefold increase in shorebird use (R. Parris, LI NWR, personal communication, 2004). Red-winged blackbird numbers, however, showed a decrease from 55 before OMWM, to less than 10 after alterations to the marsh (Lent et al., 1990).

Negative impacts to migratory birds were observed in a Massachusetts ditched marsh resulting from vegetation changes. Shrubs or exotic species invasion dominated the marsh vegetation, decreasing habitat use by shorebirds, wading birds, and aerial insectivores (USFWS, 1998). Although prey population is not reduced by ditches, Clarke et al. (1984) concluded that ditching can adversely impact bird populations by draining pools that are used for foraging.

Foraging areas within ditches are further limited by their narrow width. OMWM, by restoring open waters on the marsh, should not have these kinds of negative impacts (although ditches are not always eliminated in OMWM applications).

4.7.2 Juvenile Fish

Juvenile fish often utilize salt marshes for the abundant food supply and to seek refuge from predation (Deegan, 2000). Wolfe (1996) demonstrated that tidal circulation, enhanced by ditches, replenishes the larvivorous fish in the high marsh pools. At a previously severely ditched marsh in New Hampshire, the ditches drained the marsh surface of deep, permanent pools of water. The amount of permanent open water on the marsh was increased in 1999 due to restoration efforts, resulting in an increase of mummichog and stickleback populations, fish that accessed over ninety percent of the restored marsh (New Hampshire Coastal Program, 2004).

Fish responded immediately to a New Jersey marsh restoration project which involved the creation of fairly large subtidal creeks. Most population structural parameters, such as seasonal occurrence, average size, and size frequency distribution, were similar to those of the reference marsh creeks. The abundance of fishes was invariably greater in the creeks of the restored marsh. This may be related to greater food availability, which may be a short-term response by selected prey species and result in an influx of fish to the creeks. Considerable variation in the abundance of some fish species resulted from the OMWM alterations over a period of several months. Significant decreases in the mean number of fish per sample and the percent frequency of occurrence were observed for *F. luciae* and *L. parva,* and an absence of *M. beryllina* (Able et al., 2000).

Other OMWM-treated marshes in New Jersey had tidal flows and fish assemblages similar to those of unaltered marshes (Talbot et al., 1986). This study showed that if shallow, non-vegetated potholes are deepened or enlarged to create a permanent vegetated pond or system of ditches, relative and absolute abundances of mummichogs and spotkin killifish will likely decrease, and sheepshead minnows, inland silversides, and rainwater killifish will increase. Although these four typical killifishes all prey on mosquito larvae, mummichogs and spotfin killifish will occur in greater abundance in shallow areas and the larvae and juveniles will move about on the top of the marsh more readily (Talbot and Able, 1984). Therefore, the fishes that prefer the top of the marsh are potentially more important mosquito predators than the fishes that favor deeper pond habitats (Talbot et al., 1986).

Changes in fish species composition occurred in the Seatuck marsh subsequent to OMWM implementation. Salt marsh fish species increased significantly, and freshwater fish species decreased two years following OMWM completion as a result of the increase in marsh salinity (Lent et al., 1990).

4.7.3 Ditch Dwellers

Marsh alterations, such as ditching, do not have marked effects on soil invertebrates (Rockel, 1969; Shisler and Jobbins, 1975; Lesser et al. 1976; Chick 1979). However, the ditching of a marsh will impact other species. In a ditched marsh where the water table level dropped five inches, muskrats were observed departing the area (Daiber, 1986). The same observation was made by Stearns et al. (1939). Stearns et al. observed that effective ditching of a Delaware marsh for mosquito control lowered the water table level, changed vegetation, and, as a result, adversely impacted the welfare of the muskrat populations that previously inhabited the area.

At two marshes treated with OMWM techniques, Ferrigno (1970) reported increased numbers in fiddler crabs, ribbed mussels, and blue claw crabs. Salt marsh snails were found in fewer numbers when compared to control sites. An increase in amphipods was noted on the lower cordgrass at one marsh, but not at the other marsh.

Romanowski (1991) conducted a study pertaining to the use of an altered marsh by the Meadow Vole (*Microtus pennsylvanicus*) in the months following management. Romanowski's study concluded that with respect towards OMWM, the size of the *Microtus* populations seemed to have been a function of the revegetation process following marsh management. This study showed that in a quickly revegetated marsh, *Microtus* populations increased more rapidly then compared to a slower revegetating marsh.

4.8 Off-shore Impacts

4.8.1 Vegetation/energy export

Typically, production taking place on a marsh may either accumulate in sediments as peat, decompose within the marsh, or be exported by the tides to more open estuarine and coastal waters (Nixon, 1982). Many salt marshes export materials to deeper waters as shown by mass balance and stable isotopic studies (Valiela et al., 2000). Intertidal habitats, such as the marsh surface, depositional marsh edge, erosional marsh edge, and adjacent unvegetated intertidal flats, can serve as important sources of energy through exports to deeper water ecosystems, especially via predation by transient fish on marsh resident species (Cicchetti and Diaz, 2000). The edges of a tidal marsh tend to support a higher biomass and diversity of fishes and crustaceans than the marsh interior (Minello and Zimmerman, 1992; Baltz et al. 1993, Minello et al. 1994, Peterson and Turner 1994).

Shisler and Jobbins (1977b) demonstrated that ditched marshes release significantly lower levels of total organic carbon and particulate organic carbon than natural marshes. However, a study conducted by Cicchetti and Diaz (2000) concluded that trophic export from the depositional edge of a marsh has a significant contribution to deeper waters. Cicchetti and Diaz reported that blue crab use of depositional marsh edges was an important mechanism for movement of trophic energy off the marsh surface. OMWM, because it maintains many ditch surfaces, allows significant crab habitat to remain.

4.8.2 Coliform/Shellfisheries

The presence of coliform in aquatic environments is an indicator of contamination with fecal material and other possible pollutants. Coliform contaminants may occur in ambient waters as a result of overflow of domestic sewage or non-point sources of human or animal waste. Coliform pathogens impair coastal water quality, which may lead to the closure of shellfish harvesting areas.

It has been suggested that, by increasing retention time for water on the marsh, ditch plugging may reduce the export of coliform bacteria caused by wildlife from a marsh. Wetlands where

OMWM techniques are implemented to repair the damage caused by grid ditching could help improve water quality in an area where shellfish beds have been closed as a result of fecal coliform contamination (SCDHS, 1999).

4.8.3 Loss of Tidal Creek Functionalities

Tidal restrictions negatively impact salt marsh ecosystems (Burdick et al., 1997). According to Ferrigno et al. (1975), when daily tidal action is blocked, organisms important to the tidal marsh nutritional web are considerably impacted. This is why almost all OMWM installations use tidal flows as part of the water management regime. Full ditch plugs do not emphasize daily tidal flows as part of the water management efforts. OMWMs using full ditch plugs do require intermittent inundations, through spring tides and/or storms. A trade-off is created. There are water table increases and the retention of water in the ditches, which should create ponded areas, for example. These are deemed to be more beneficial to the overall health of the marsh, and to meet the aim of the restoration effort, than the benefits associated with tidal flows.

Sills also restrict some tidal flows in the ditches. Again, the judgment made with a sill ditch OMWM is that retention of water and the potential water table restorations provide greater benefits than would be received if full tidal circulation occurred.

Water retention is expected to increase water tables. This can result in expansion of low marsh into formerly high marsh areas, reduce woody plant and *Phragmites* vigor, and restore drained ponds and pannes. In addition, retention of water in the ditches creates refuges for insectivorous fishes between high tides, and may increase waterfowl habitat. These benefits need to be weighed against the impacts of tidal influxes. The consensus of opinion is that it is the importation of energy, nutrients, sediment, and biota on the tides that supports the vigorous marsh ecosystem. Limiting the tides, therefore, will have an overall impact on the health of the salt marsh, although that impact may not be significant.

4.9 Summary

OMWM has four major variants. Only three are extensively used on the east coast of the US. All kinds of OMWM develop better habitat for insectivorous fish, and provide better access for such fish into the marsh areas where mosquitoes breed. The differences in OMWM classes have to do with connections to the estuary.

Open OMWM systems have a direct and full connection to the estuary. These systems address mosquito breeding that is caused by a lack of tidal circulation. Ponds created for such systems are often not directly connected to the ditches, or are connected by shallower sills. This prevents the ponds from emptying through tidal cycles, so that they retain their fish reservoir characteristics. Most New Jersey OMWMs are open systems.

The other two systems involve blockages of existing ditches. Blocking the ditches may restore water tables that were drained by the ditch system. It is also thought that the ditch plugs retain saline water on the marsh, and, so, may control *Phragmites* infestations. Plugs make the existing ditches fish reservoirs.

Sills only partially block the ditches. The resultant sill ditches allow a measure of tidal exchange each tidal cycle, and so keep the OMWM ponds from becoming isolated from the estuary. Sill ditches may also avoid creating conditions that favor *S. alterniflora* over *S. patens*. Sill ditches are often used as a compromise OMWM, when neither a full ditch plug nor an open system is appropriate for the environmental setting.

Full ditch plugs create the highest water table. The hydraulics of the system may make the water retained in the ditches more saline. Full ditch plugs create isolated pools that only receive tidal inputs during spring tides or storm conditions. Many of the marsh restorations in Connecticut rely on full ditch plugs as their basic infrastructure. Generally speaking, ditch plugs are the most common form of OMWM in the northeast US.

OMWM was developed as a more nuanced approach to water management for mosquito control. It is believed to have fewer environmental impacts than grid ditching, as it is less likely to lower the marsh water table. It results in more surface water areas on the marsh, creating conditions favorable to waterfowl. It is less likely to result in a shift of vegetation patterns from what already exists on the marsh, although OMWM is thought to be effective at controlling the spread of *Phragmites*.

OMWM installations are generally thought to require less maintenance than grid ditching, and to be as effective at mosquito control. Some adherents believe OMWMs are more effective than ditching marshes. They have been adopted throughout the northeast US as an alternative to grid ditching and grid ditch maintenance. The sole exception to this general adoption has been New York. Here, regulators have concerns regarding the impact of OMWM to overall marsh health and functionalities.

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