12-6-2018

Potential Impacts of the Purple Salt Marsh Crab, Sesarma Reticulatum, on Long Island Sound Salt Marshes

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POTENTIAL IMPACTS OF THE PURPLE SALT MARSH CRAB, *SESARMA RETICULATUM*, ON LONG ISLAND SOUND SALT MARSHES

A THESIS

submitted in partial fulfillment

of the requirements for the degree of

MASTER OF SCIENCE IN ENVIRONMENTAL SCIENCE

BY

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West Haven, Connecticut

December 6, 2018

POTENTIAL IMPACTS OF THE PURPLE SALT MARSH CRAB,

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ACKNOWLEDGMENTS

I would like to express my sincere gratitude to my advisor, Professor Roman Zajac, for his guidance, knowledge and patience.

Special thanks to Deena Chadi, Andrea McCarthy, Shannon Penna and all others for helping to collect data in the field; and to the University of New Haven for making this research possible.

ABSTRACT

Salt marshes are important, highly productive ecosystems that provide many ecological benefits, including shoreline protection. Salt marshes were long believed to be driven by bottom up forces, however, recent research has suggested that top down influences may impact the foundation species *Spartina alterniflora*, and consequently marsh structure. This study looked at patterns of *Spartina alterniflora* dieback along creek bank areas and assessed *Sesarma reticulatum* abundance spatially throughout the marsh to improve understanding as to whether *Sesarma* may be influencing the dieback phenomenon. Dieback was occurring in all three salt marshes studied, at two of those marsh sites the rate of dieback was greater during the 2010-2012 timespan than the 2004-2010 timespan. Rate of dieback was greatest in the marsh site where no field studies were conducted. There were no consistent relationships between dieback occurrence and geomorphological traits of the creek section or the creek bank heading. Spatially, dieback and *Sesarma* presence were found to overlap at both of the marsh sites where field studies were carried out. *Sesarma* abundance was greater at bare creek bank sites than creek bank sites with typical vegetation growth. Additionally, abundance was greater during predominantly low tide deployments compared to high tide deployments. *Sesarma* were found to utilize the high marsh bare areas studied. Transplant studies indicated that grazing pressure is present in the marsh sites studied, and that *Sesarma* could likely be contributing to this pressure. There are predators of *Sesarma* within the marsh study sites, further investigation is warranted to determine whether there is an inadequate balance that may be leading to overgrazing of *S. alterniflora*. The results of this study indicate that further investigation into *Sesarma reticulatum* dynamics within the marsh sites studied relative to *Spartina alterniflora* dieback areas could help increase understanding of potential interactions between these species impacting salt marsh structure and

potentially long term sustainability. Understanding these potential interactions, and how they may be complicated by events such as climate change, are important for making management decisions to help ensure that these systems continue to persist and provide the services that we rely upon.

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INTRODUCTION

Salt marshes are one of the most productive environments on the planet (Bertness, 1992; Gedan et al., 2009). Salt marshes are dynamic ecosystems that have displayed significant changes in the last few decades due to changes in climate (Bertness et al., 2004; Simas, 2001), sea level rise (Gedan et al., 2011a; Michener et al. 1997) and anthropogenic impacts (Bertness et al., 2002; Bertness and Silliman, 2008). Direct anthropogenic impacts, such as urban growth have resulted in habitat destruction and significant wetland loss (Bromberg and Bertness, 2005; Gedan et al., 2011a; Coverdale et al., 2013a; Pat, 2004; Mudd, 2011). While salt marsh ecosystems were generally believed to be highly resilient to indirect anthropogenic impacts (Gedan et al., 2011a and references therein), it has recently been suggested that this may not be the case (Bertness and Silliman, 2008; Gedan et al, 2011a). Current research is questioning whether or not salt marshes will be able to maintain their resilience to indirect anthropogenic impacts after decades of disturbance and in light of changes in climate and sea-level rise (Gedan et al., 2011a; Bertness and Silliman, 2008; Simas, 2001).

Observations of extensive dieback occurrence in the Cape Cod, MA, region have focused on biotic causes of marsh change, specifically whether the native Purple Marsh Crab, *Sesarma reticulatum* may be impacting salt marsh vegetation communities (Holdredge et al., 2009; Altieri et al., 2012). Some evidence of *Sesarma* activity in Connecticut salt marshes has been found (Coverdale et al., 2013b), and creek bank dieback (Figure 1) continues to be observed in an increasing number of LIS salt marshes. The objective of this thesis was to assess the historical aspects and spatial extent of salt marsh dieback that may be related to *Sesarma reticulatum* activity, and study in detail their populations and activity patterns on several central Long Island Sound salt marshes.

Importance of Salt Marsh Environments

As one of the most valuable ecosystem service providing environments in the world (Gedan et al., 2009; Levin et al., 2001), it is important to determine how salt marshes are responding to global change and anthropogenic impacts, so that we can understand and assess how these ecological services may be impacted. Some of the valuable ecological services salt marshes provide include filtering anthropogenic pollutants (Gedan et al., 2011a; Holdredge et al., 2009), buffering shorelines (Bertness and Silliman, 2008; Pennings and Bertness, 2001), providing nursery grounds for many fish species, including economically important species (Minello et al., 2003; Bertness and Silliman, 2008; Holdredge et al., 2009) as well as habitat for many other species and sequestering carbon which helps slow the greenhouse effect (Chmura, 2003; Bertness and Silliman, 2008).

Salt marshes are an excellent natural buffer, protecting coastal areas from storm surges through the reduction of wave energy (Gedan et al., 2011b). This will be of particular importance as storm intensity increases with rising global temperatures (Scavia et al., 2002; Michener et al. 1997). Salt marshes filter chemical pollutants and consequently are a valuable geochemical sink, and can reduce terrestrial anthropogenic impacts on coastal waters (Bertness and Silliman, 2008; Gedan et al., 2011b). This biological and mechanical filtering process is achieved through salt marsh vegetation and accumulated peat, which help to filter out anthropogenic pollutants such as pesticides and heavy metals, greatly reducing the amount that may enter coastal waters (Gedan et al., 2011a; Weis and Weis, 2004).

Another important service that salt marshes provide, which is of great importance with increased anthropogenic carbon outputs, is the role they play in the carbon cycle. Wetlands in general are the largest component of the terrestrial biological carbon pool (Dixon and Krankina,

1995, as cited in Chmura et al., 2003) which makes them significant in global carbon cycles (Chmura et al., 2003). It is suggested that salt marshes in particular may be the most beneficial in sequestering carbon due to higher burial rates and lower CH_4 emission rates (Chmura et al., 2006).

Typical New England Salt Marsh Structure

While accounting for less than two percent of the total Atlantic coast salt marsh area (Niering, 1982), New England salt marshes are some of the best studied marsh systems. The general structure of New England salt marsh vegetation composition (Niering and Warren, 1980; Bertness, 1992), formation (Redfield, 1972; Orson et al., 1987; Nixon and Oviatt, 1973; Niering and Warren, 1980), and biota (Nixon and Oviatt, 1973; Bertness, 1992) have been well described. Salt marsh vegetation consists of distinct bands from the low marsh, which is located at the water's edge, up to upland edge of the marsh (Figure 2). It is typical in New England salt marshes for *Spartina alternifora*, or cordgrass, to grow in the low marsh, followed by *Spartina patens*, or salt-meadow hay, in the high marsh with *Distichlis spicata* (spikegrass) and *Salicornia* (glasswort) also common, and then a band of *Juncus gerardii* (black rush) at the upper edge of the high marsh and into the transition zone which typically consists of *Iva frutescens* and/or *Phragmites austalis* (Berness, 1992; Bertness et al., 2002). It is also common for a stunted growth form of *S. alterniflora* to grow in the high marsh zone (Bertness, 1992). *S. alterniflora* in the high marsh zone is stunted due to the high peat content, and consequently low oxygen content, of soils caused by belowground composition of the highly productive plant (Bertness, 1992). Salt marsh vegetation zonation is determined by the ability of vegetation to adapt and compete within varying tidal conditions (Bertness, 1992).

Salt marshes are formed, and sustained, through the accumulation of sediment and the production of peat from dead aboveground organic material being incorporated into the substrate (Redfield, 1972; Titus et al., 2000; Kennish, 2001). Sediment reaches the system through tidal flushing (Reed, 1995) and river flow (Titus et al., 2000). New England salt marshes differ from those to the south along the eastern US coast (Bertness, 1999, as cited in Charles and Dukes, 2009). In New England, the accumulation of organic matter from salt marsh vegetation plays a large role in accretion compared to the delivery of sedimentation (Bertness, 1999 as cited in Charles and Dukes, 2009). Ultimately, a marsh's accretion rate is increased by sediment delivery to the marsh surface, above and below ground productivity, and the accumulation of dead organic matter; reversely, accretion is decreased by removal of sediment (i.e. through erosion) and the decomposition of below ground organic matter (Reed, 1995).

Recent Changes

New England salt marshes are changing rapidly in response to human generated changes such as eutrophication (Deegan et al., 2007; Bertness et al., 2002) and restriction (Gedan et al., 2009; Pat, 2004), and possibly human influenced changes in climate (Simas et al., 2001; Gedan et al., 2009) and sea level rise (Warren and Niering, 1993; Simas et al., 2001). These changes are further intensified by changes in biota present in the environment such as increased populations of some salt marsh crabs which have the ability to greatly alter the physical environment (Holdredge et al., 2009), its chemistry (McCraith et al., 2003), and the survivability of ecosystem foundation species (Bertness et al., 2009).

Impacts of Climate Change

Increased temperatures are likely to impact salt marshes through northward shifts in the range of biotas (Najjar et al., 2000), changes in salt marsh chemistry and hydrology (Charles and Dukes, 2009), and changing the general plant composition of New England salt marshes (Najjar et al., 2000; Gedan et al., 2011a; Charles and Dukes, 2009). Perhaps one of the most obvious impacts of global warming on salt marshes are increases in salt marsh surface temperatures, which consequently could lead to increased rates of evaporation and higher salinity levels (Bertness et al., 2002; Charles and Dukes, 2009). Global climate change will also alter precipitation patterns for areas which could result in increased sediment delivery rates due to erosion caused by increased stream flows (Najjar et al., 2000). The hypothesis that increased precipitation levels would reduce pore water salinity levels was experimentally tested by Charles and Dukes (2009) and was determined to not be the case.

Increased temperatures, related to global warming, allow for increased productivity of *Spartina* species which will allow them to outcompete salt marsh forbs (Gedan and Bertness, 2009). Changes in salt marsh vegetation communities will have an impact on the accretion rates of the salt marsh (Donnelly and Bertness, 2001; Reed, 1995). While plants are typically aided by increased temperatures, one negative impact is the increase of microbial activity which results in increased decay and decomposition rates of below ground material (Charles and Dukes, 2009). With an expected increase of 2-3°C in the average temperature from the late 20th century to the middle of the 21st century, it can only be expected that climate change will continue to affect salt marshes (Hayhoe et al., 2006).

Continued human activities are likely accelerating the rate of global warming through increased carbon dioxide outputs and other greenhouse gasses (Simas et al., 2001; Najjar et al.,

2000). Concurrently, the increased temperatures drive increased rates of sea-level rise (SLR) through ocean expansion and increased rates of ice melt (Chmura et al., 2006; Najjar et al., 2000; Bertness et al., 2004; Charles and Dukes, 2009). Despite the driving force climate change has on SLR, Najjar et al. (2000) suggests that increased temperatures may help accretion rates keep up with SLR in some cases through increased levels of precipitation which would increase river flow and sediment deposition on salt marshes, and in concert with increased levels of CO_2 levels increase plant productivity.

Impacts of Sea-Level Rise

It is estimated that sea level will rise 50 to 200 cm in the next one to two hundred years due to global warming (Titus et al., 1991). Because salt marshes are coastal habitats, even small changes in SLR can impact salt marsh environments (Simas, 2001). The accretion rate of soils and the accumulation of below ground organic matter, or peat, which forms the salt marsh table is a key determinant of a marsh's ability to keep up with SLR (Stevenson et al., 1986 and Reed, 1990, as cited in Reed 1995). Accretion rates in New England, and likely also along the eastern US coast, have increased in the last 150 years due to elevated sedimentation delivery from anthropogenic deforestation for agriculture and development (Kirwan et al., 2011). The levels of sediment in waters entering coastal environments have significantly declined in recent years and may reduce the ability of New England salt marshes to keep up with SLR rates (Kirwan et al., 2011). SLR itself affects accretion rates through the impacts associated with increased salinity and inundation on the salt marsh surface (Reed, 1995). Studies have suggested that SLR can have negative impacts on above ground vegetation production and below ground decomposition rates, which makes it difficult to fully understand how SLR affects salt marsh accretion rates (Reed, 1995).

Marshes typically move inland in response to SLR (Donnelly and Bertness, 2001; Bertness et al., 2004). This movement of vegetation toward the upland allows for the marsh to keep up with SLR and prevent it from disappearing. This process can however be blocked by anthropogenic buildup at the marsh/terrestrial upland boundary, and is often referred to as coastal squeeze (Bertness et al., 2004; Pat, 2004).

Apart from the salt marsh surface becoming flooded due to SLR, the loss of salt marsh also occurs through the expansion of the creek system, resulting in an increase to the tidal prism (Hughes et al., 2009). This is a natural process that has been observed over at least the last fifty years through aerial photography of some creek systems (Hughes et al., 2009). Creeks will branch out and extend at their head to increase the amount of water the system can hold and reduce the amount of water that floods the marsh surface (Hughes et al., 2009). While this process prevents the entire marsh surface from being inundated regularly with excess salt water, valuable marsh surface is being lost and will remain lost under current conditions.

Impacts of Marsh Crabs

Recent observation of high densities of *Sesarma reticulatum* (purple marsh crab) (Holdredge et al., 2009; Altieri et al., 2012) have been noted in some areas in the Cape Cod Region of Massachusetts. Salt marsh crabs have the potential to be ecosystem engineers and change their environment significantly. Increased abundance of crabs and crab burrows of *Sesarma* have been related to die-off events of *S. alternifora* (Holdredge et al., 2009). Other burrowing crab impacts include changes in accretion rates which have been linked to *Uca* (fiddler crabs) (Holdredge et al., 2010), and affecting marsh chemistry (McCraith et al., 2003) and water movement (Bertness, 1985).

Crab populations and burrows are most common near creek banks and become less frequent as you move toward the high marsh and short-form *S. alterniflora* (Bertness, 1985; McCraith et al., 2003). This distribution is likely related to root density, which in New England marshes is related to tidal height, and the softer sediment of tidal creek banks allows for easier burrowing compared to the hard high marsh surface (Bertness and Miller, 1984). However, in recent years fiddler crabs have been shown to be occupying more of the high marsh zone (Luk and Zajac, 2013).

Relatively little is known about the distribution of *Sesarma* within its range and what factors may influence its survival (Holdredge et al., 2009; Bertness et al., 2009). While herbivory was not believed to play a significant role in salt marsh productivity (Bertness, 1985), Holdredge et al. (2009) found that *Sesarma* herbivory has caused extensive die-off areas of *S. alterniflora* along creek banks on Cape Cod (Holdredge et al., 2009), based on grazing markings on grass blades and experimental manipulations. These die-off events have resulted in increased erosion, and consequently potential permanent marsh loss (Holdredge et al., 2009). Holdredge et al. (2009) found a positive correlation with crab density and vegetation loss. Tethering experiments were performed by Holdredge et al. (2009) to determine whether release from predation pressures was a likely cause of population increases and found it to be a likely cause.

One hypothesis for the localization of increased *Sesarma* populations is the release from predation pressures caused by overfishing (Holdredge et al., 2009; Bertness, Holdredge, and Altieri, 2009; Aleteri et al., 2012). A recent study by Altieri et al. (2012) looked at the possibility of a trophic cascade releasing *Sesarma* from predation pressure, which results in *S. alterniflora* die-off events. Altieri et al. (2012) found 50% fewer top level predators of *Sesarma* in areas of *S. alterniflora* die-off than in areas where *S. alterniflora* was doing well. Interestingly these

localized events of predator removal appeared to be related to recreational fishing activity (Altieri et al., 2012). While removal of predation pressure due to recreational activities provides one possible explanation for the localization of increased *Sesarma* populations and *S. alterniflora* die-off events, this potential relationship was noted only in this one study. Additional influences on *Sesarma* population size and impacts to *S. alterniflora*, such as increased habitat availability due to changes in vegetation composition, climate change effects on *Sesarma* biology and SLR, should also be considered.

In contrast to *Sesarma*, *Uca* species, one of the most common macroinvertebrates in salt marshes on the western Atlantic coastline (Bertness, 1985; Nomann and Pennings, 1998) are relatively well studied (Holdredge, 2010; McCraith et al., 2003; Montague, 1980; Ringold, 1979). Known predators of *Uca* include birds, channel bass, blue crabs, mud crabs, and raccoons (Nomann and Pennings, 1998). Fiddler crabs are detritivores, feeding along tidal creeks in the low marsh and in salt marsh pannes, turning over as much as 18% of the upper 15cm of the marsh surface (Katz, 1980). They build their burrows primarily in the low marsh along creek banks where it is easier to burrow and *S. alterniflora* offers soil stability and protection (Bertness and Miller 1984; Nomann and Pennings; 1998, Smith and Tyrrell, 2012).

The primary concern of increased *Sesarma* populations is the impact that they have on the foundation species *S. alterniflora* (Bertness, 2009), particularly along creek banks (Holdredge et al., 2009; Altieri et al., 2012). The intense grazing of *Sesarma* leaves bare areas which may be difficult to subsequently be recolonized (Smith and Tyrrell, 2012; Gedan et al., 2009). Herbivory and burrowing by *Sesarma* has negative impacts on plant growth along creek banks and the low to high marsh transition due to: increased erosion as peat stability decreases, changes in soil

chemistry, and increased pressures of SLR which result in faster tidal creek expansion and loss of vegetated salt marsh habitat (Hughes et al., 2009).

Despite negative feedbacks of crab activity in the low marsh, bioturbation due to burrowing may have positive impacts on vegetation production (Bertness, 1985; Smith and Tyrrell, 2012). Bertness (1985) found that *Uca* burrows increased above ground growth of tall form *S. alterniflora* by 47% and decreased belowground root mass by 35%. These findings support that *Uca* burrows are beneficial to *S. alterniflora* plant growth through increased aeration, drainage, and decomposition of below ground organic material (Bertness, 1985). In contrast, Nomann and Pennings (1998) found that *Uca* burrows did not positively affect plant growth, but rather decreased plant growth. Nomann and Peenings (1998) believe that it is more likely that salinity controls plant growth, however, they did note that their study may not have contained a large enough sample size and the crab removal process may not have been effective enough to determine potential crab impacts.

Another study on impacts of bioturbation of *Uca* activity, Smith and Tyrrell (2012) proposed that the positive effects of bioturbation due to crab activity is density dependent, and that at high crab densities bioturbation can have negative impacts on vegetative growth. Their study focused on the compounded effects of *Sesarma* and *Uca* on the generation and longevity of *S. alterniflora* bare spots, and found that *Uca* bioturbation through feeding and burrowing in *Sesarma* generated bare regions was extensive, negatively impacting new *Spartina* seedling establishment and growth, and thus preventing recolonization of the bare areas (Smith and Tyrrell, 2012). In addition to physical disturbance of the soil by *Sesarma*, removal of microbes typically associated with salt marsh vegetation that aid plant growth may have detrimental impacts on plant growth (Montague, 1980).

Aside from affecting vegetation growth, crabs also impact the accumulation of organic matter. Thomas and Blum (2010) found that *Uca* burrows positively affect decomposition rates, and consequently will usually have a negative impact on marsh accretion rates which is significant in respect to changes in SLR. Belowground herbivory is an often overlooked aspect in all types of plant communities (Coverdale et al. 2012). As a burrowing crab that consumes plant material, it is important to consider how *Sesarma* belowground grazing may affect salt marsh vegetation. Coverdale et al. (2012) found in a laboratory feeding preference experiment that while there was no significant difference in the consumption of below ground and above ground plant material, more below ground plant material was consumed (19.5 and 15.6% respectively). This is important to keep in mind because below ground grazing often goes unnoted. *Sesarma* are extensive burrowers; often there are many branches and entrances existing for one burrow and burrows are utilized by multiple *Sesarma* (Bertness et al., 2009). Coverdale et al. (2012) also tested the impact of aboveground and belowground herbivory independently and in conjunction on marsh vegetation. They found that both above and belowground herbivory independently affected the biomass of vegetation in addition to both above and belowground vegetation. Impacts were the greatest when both above and belowground herbivory occurred.

The possible impacts burrowing salt marsh crabs have on salt marsh vegetation, and consequently salt marsh structure, drives the importance of understanding why crab populations and their influence on salt marshes are changing. As previously stated, one suggested cause of *Sesarma* population changes is a reduction of predators. This line of thinking is supported through direct evidence from an experimental field study conducted by Bertness et al. (2014b) on Cape Cod, MA. Other research has shown that the invasive green crab, *Carcinus maenas*, may influence *Sesarma* activity levels and as such may aid in the recovery of dieback areas (Bertness

and Coverdale, 2013; Coverdale et al., 2013c). It has been observed that *Carcinus* will utilize *Sesarma* burrows and that particularly large *Carcinus* will prey upon *Sesarma* (Coverdale et al., 2013c). Recent work (Bertness et al. 2014a) has addressed the fact that lower levels of *Sesarma* predators are also being observed in the Narragansett Bay region. Many questions remain about how *Sesarma* populations are changing, the impacts they are causing, and how widespread this phenomenon can be expected.

Objectives

Most of the research conducted on *Sesarma* populations and their influences on the salt marsh environment have been performed in only several areas of southeastern New England. Research needs to be conducted in other regions to obtain a general understanding of what impacts *Sesarma* populations may have on salt marsh environments. The specific goal of this study was to understand in more detail *Sesarma* populations along the northern coast of Long Island Sound (LIS) and how they may be affecting salt marsh vegetation and habitat structure. The specific questions asked included:

1. Is the evidence of long-term change in low marsh habitats/vegetation consistent with *Sesarma* activity?
2. What are the patterns and rates of vegetation loss due to *Sesarma* on LIS marshes?
3. What are the population characteristics of *Sesarma* on the study marshes and what role does predation play in potentially controlling their abundances?
4. Are there any relationships in their apparent impacts with the activities of other marsh crab species?

In light of all the challenges salt marshes are facing today with anthropogenic impacts and changes in climate and SLR, the impacts of salt marsh crabs needs to be studied in more detail in order to further our understanding of these systems. By understanding crab impacts and what

drives changes in their population sizes, we can predict the impacts that they may have and possibly determine management plans that consider these phenomena.

Figure 1. Creek bank dieback area with *Sesarma* burrows located in Pleasant Point salt marsh (Branford, CT) in Long Island Sound. Photo was taken near high tide.

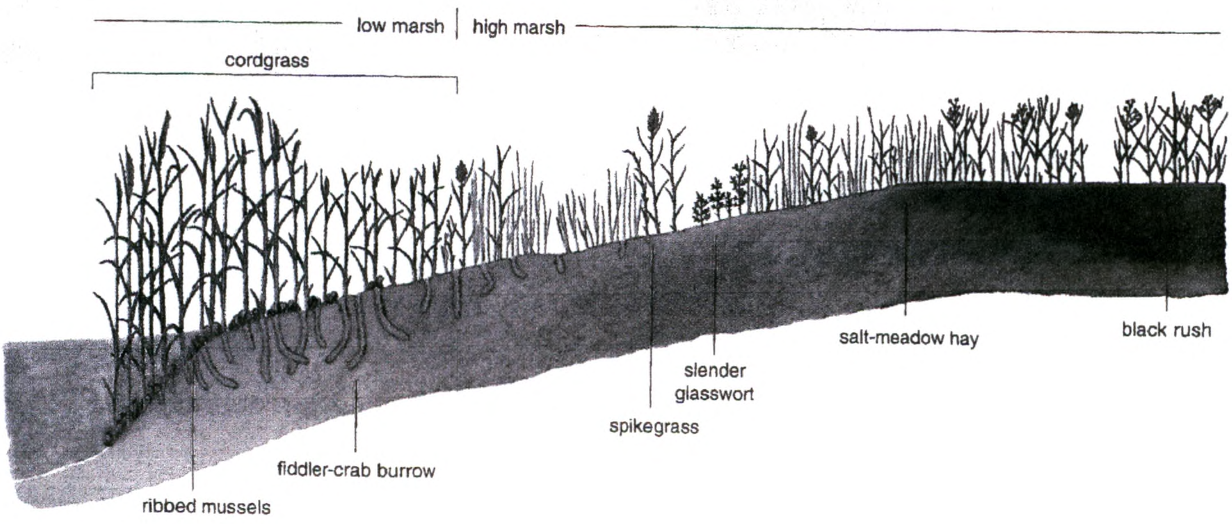

Figure 2. Bertness (1992) figure illustrating the typical zonation of New England salt marsh vegetation.

MATERIALS AND METHODS

Historical and Spatial Analysis of Low Marsh Dieback

In order to assess historical trends in dieback patterns, and whether they might be associated with *Sesarma reticulatum* (*Sesarma*) activity, a spatial and temporal analysis of historical image data was conducted. Dieback was assessed in three Connecticut salt marshes and assessed for patterns relative to *Sesarma* activity and other factors that could possibly be contributing to the dieback phenomenon such as location within the marsh and creek morphology.

Salt marsh creek bank dieback within Long Island Sound was assessed using aerial photographs of three salt marshes along the Connecticut coast of the Sound: Banca Marsh (41°16'12" N, 72°45'20" W), Pleasant Point Marsh (41°16'10" N, 72°45'42" W) and Chaffinch Marsh (41°15'51" N, 72°40'45" W) (Fig. 3). Aerial imagery was obtained from the University of Connecticut MAGIC website (http://magic.lib.uconn.edu/connecticut_data.html). Geo-referenced imagery of sufficient resolution for quantitative analysis was available for 2004, 2010 and 2012. This imagery was analyzed using ArcMap software (ESRI, Redlands, California; Version 10.1). Imagery was also obtained for 1974 (Banca and Pleasant Point only), 1980, 1986, 1990, 1995, and 2000 that was not geo-referenced, but was analyzed for general patterns in dieback occurrence to establish a baseline of when dieback began. These images were analyzed using ImageJ software (Rasband, W., National Institutes of Health, USA; Version 1.48s).

Both non and geo-referenced imagery was visually assessed for dieback using transects. The use of non-geo-referenced images extended the historic baseline for comparing dieback occurrence. Non-geo-referenced images where only used in the qualitative assessment, whereas the geo-referenced images were also assessed quantitatively.

Transects were located throughout the marsh to create a spatially comprehensive assessment at each site. Transects were classified based on creek type: areas along the main tidal channel of the marsh were classified as "main", areas along man-made ditches "ditch", and areas along natural creeks that branch off of the main channel "branch". Transects were placed so that no creek branching occurred within the extent of a transect to limit variability of transects within a creek type. This, in combination with the effort to sample as much of the marsh as possible and consistently identifying transect start and end points in non-referenced imagery, resulted in transects of varying lengths. Referenced images shared start and end points with the same coordinates, in non-reference imagery these points were best estimated based on characteristics of the marsh creek structure.

For the qualitative analysis, dieback occurrence along the length of each transect was classified as none, possible, light, moderate or high. Transects along which no dieback was evident were classified as none. While all images assessed were taken during low tide, at times it was difficult to determine whether exposed creek bank was exhibiting dieback or if there was just an exposed, naturally un-vegetated portion of the creek channel. In these instances where it was thought dieback could be occurring but confidence level was low, the transect was classified as possible dieback. Small patches of dieback that made up less than 25% of the creek bank extent were classified as light. Moderate dieback was classified as anything in-between 25% and 50% dieback along the creek bank extent. If greater than 50% of the creek bank extent exhibited dieback, then the dieback was classified as high. Imagery was assessed to understand when dieback may have become an issue in each of the marshes, and to assess whether there were any trends within or across the three marshes over time.

For the quantitative analysis, unique creek centerlines were drawn connecting the start and end points of transects for each aerial image to account for any creek morphological changes that occurred over time. Along each transect ten random points were selected using a random number generator. At each point the distance from the creek centerline to the vegetation was measured at a right angle from the creek centerline to the creek bank in both directions. The creek bank geomorphology at each measurement point (straight, inside curve, or outside curve) and general heading of where the bank was located relative to the creek centerline (N, S, E, or W) was noted.

Differences in the distance from the creek centerline to vegetation were assessed based on creek types, geomorphological characteristics and creek bank heading separately for each marsh using Kruskal-Wallis tests and Dunn's post hoc test. In the Pleasant Point marsh distance from the marsh inlet was also assessed to see whether dieback occurrence was related to distance from the creek mouth on Long Island Sound and due to factors such as tidal gradients and inundation levels which could be influenced by a partial tidal restriction. This was done by comparing the front marsh, closest to Long Island Sound (seaward side of Historic Trolley Trail, Fig. 3), to the back marsh (upland side of Historic Trolley Trail) using a Kruskal Wallis test. I also assessed the average rate of dieback estimated for the time periods between the 2002 and 2010 imagery and the 2010 and 2012 imagery using Kruskal Wallis analyses and Dunn's post hoc tests.

One focal site was selected within each marsh for a more in-depth analysis. For the Pleasant Point and Banca marshes, focal sites were selected based on the locations where *Sesarma reticulatum* abundance and experimental studies were conducted (Fig. 4, and see below). The creek sections studied at focal sites were classified in the analysis above as

"branch". In the Chaffinch marsh, the location of the only creek classified as "branch" was selected, as it was the only location representing the creek type and no experimental studies were conducted in that marsh. For the focal sites, the change in distance from creek centerline to vegetation was extended to include the 1995 and 2000 aerial imagery to assess longer-term changes. Distance to vegetation was assessed in the 1995 and 2000 imagery the same way as geo-referenced images, using ten random points along each transect using ImageJ software.

Assessing Crab Abundance in Dieback and Vegetated Areas

Field studies were conducted from late April through mid-November, 2013, at two salt marshes located in Branford, CT: Banca Marsh and Pleasant Point Marsh (Fig. 3). These marshes are *Spartina alterniflora* dominated systems with tall form *Spartina alterniflora* in the low marsh and along creek banks (natural and manmade mosquito ditches) and short form *Spartina alterniflora* dominating much of the high marsh. Also present in the high marsh are patches of *Spartina patens* and *Distichlis spicata* and salt marsh forbes such as *Limonium carolinianum* and *Salnicornia spp.*. The area of Banca marsh that was studied is owned by the Town of Branford; the southern area of the marsh that lies adjacent to Long Island Sound, south of the trolley trail, is owned by the University of New Haven for research purposes. The potion of Pleasant Point marsh where studies were conducted is owned by the Branford Land Trust; other portions of the marsh are owned by the Town of Branford, the State of Connecticut and private land owners.

To assess the relative abundance of *Sesarma reticulatum* at the Pleasant Point and Banca marsh sites, passive pitfall traps were constructed and placed along creek bank areas that 1) exhibited typical growth and density of *S. alterniflora* for a New England salt marsh (classified "vegetated") and 2) creek bank areas that exhibited a noticeable amount of bare spots or die back

(classified "bare"). Traps were made of plastic cylindrical containers (16 cm h x 8.5 cm d) with five drainage holes located in the bottom of the container. Traps were placed in 6 different locations within the marshes; 4 locations in Banca and 2 locations in Pleasant Point (Fig. 4). Areas designated as "bare" had a few or no *S. alterniflora* plants. Differences in abundance of *Sesarma* were assessed statistically based on date of collection, trap location ("vegetated" or "bare"), and marsh site using a three-way analysis of variance (ANOVA) and Tukey-Kramer multiple-comparison tests.

Vegetation at each creek bank trapping site was sampled over the growing season to quantify differences among vegetated and bare areas. Height of vegetation growth was quantified in five 0.25m^2 quadrats monthly throughout the growing season at each of the trapping sites. Locations of quadrats were marked with flags so that the area sampled remained consistent throughout the sampling period. Flags were sunk low to the marsh surface to prevent drawing attention to the sampling area. Sampling consisted of haphazardly measuring the stem height of five *S. alterniflora* plants within each quadrat intermittently over the study period. Observations of stem stubble present within a quadrat were noted. Differences in stem heights between Banca and Pleasant Point vegetated and bare sites were determined using Kruskal Wallis and Dunn's post hoc tests.

For each *Sesarma* sampling date, environmental conditions were noted including tidal phase, lunar phase, and maximum air and ponding water temperature. Air temperature was obtained from Tweed Airport, New Haven, CT and water temperature was collected from a HOBO sensor deployed at Banca marsh. Environmental factors were assessed for trends that might indicate relationships with *Sesarma* abundance or activity levels. Patterns relative to tidal phase were assess with a one-way ANOVA, lunar cycle patterns were assess with a Kruskal

Wallis analysis, and relationship between marsh surface temperature and *Sesarma* abundance was assessed using linear regression.

Crab size (carapace width) was measured and analyzed for any differences among date, vegetation characteristics of the trap location, marsh site and gender (which included a separate category for gravid females). A one-way ANOVA was used to determine the significance of differences in crab size over the study period; crab size relative to gender, marsh site and vegetation characteristics were each assessed using Kruskal Wallis analyses.

The trapping experiment was expanded in late summer to mid-fall to include a second grouping of traps located in high marsh die-off areas to assess *Sesarma* abundance in these locations. Each die-off area was sampled with four traps located at 1) the center of the patch, 2) the edge of the patch closest to a creek, 3) the edge farthest from a creek and 4) a control trap located outside of the bare patch within an area of short form *S. alterniflora* placed at an equal distance from the creek as the trap located in the center of bare patch (Fig. 5). The control trap was designed to determine to what extent *Sesarma* was found in areas away from bare patches.

A total of five die-off areas were selected: 2 in Banca and 3 in Pleasant Point (Fig. 4). Analyses were conducted to determine if there were significant differences in abundance among and within sampling dates, marsh sites (Pleasant Point vs. Banca) and location within the die-off patches using a three-way ANOVA. In addition to assessing whether there were any trends within the high marsh bare patches, crab abundance in the high marsh patch locations were compared with crab abundance along creek banks using a Kruskal Wallis test. Additionally, visual spatial analysis was conducted to see if there were any patterns in abundance relative to geomorphological traits of the marsh such as creek type, creek curvature, and location within the marsh relative to the marsh inlet.

Sesarma Herbivory Field Experiment

Vegetation transplant experiments were carried out at the Pleasant Point and Banca marshes to assess whether bare areas along the creek banks at the study sites were due to *Sesarma* herbivory. Nine replicate sets of transplants were deployed: 6 in Banca marsh, with 3 sets in bare areas and 3 sets in vegetated areas, and 3 in Pleasant Point marsh, all of which were located in bare areas (Fig. 4). Each replicate consisted of three treatments: uncaged culms of *S. alterniflora*, caged culms and culms located within a cage control structure (partially caged culms which allowed access for crabs) (Fig. 6). The caged transplant excluded *Sesarma* from the transplanted culm, eliminating the possibility of herbivory. Cages were constructed of 7.6 cm mesh galvanized wire fencing. Cages were 91cm high and 10 cm in diameter; 20 cm of the cage bottom was placed below ground level to exclude actively burrowing crabs in addition to crabs on the surface. The cage control tested for whether or not the cage itself had an effect on the survivorship of the transplant. Cage control treatments had the same dimensions as the full cage treatment except for a 10 x 10 cm opening was created at ground level to allow crabs access to the culm.

Each transplant consisted of 5-13 stems of tall form *S. alterniflora*. The number of stems was noted for each transplant at the start of the experiment. Monitoring began in mid to late July and was conducted on a weekly basis for six weeks after transplanting to quantify grazing and any plant growth based on the number of stems. The general overall health of the plants was also noted. To measure the condition of the transplants, the number of living stems were counted at each observation. Stems grazed to stubble, dried out and browned, or negatively impacted in some way that stem recovery would prevent be unlikely were not included in the count. Observations of each transplant were standardized relative to the original number of stems within

that particular transplant at the time of installing the transplant. Differences in the original stem count and the final stem count were assessed relative to the patch type and treatment group using a Friedman's test. Positively identified impacts due to grazing were assessed for any relationships with transplant site vegetation characteristics and spatial location within the marsh site.

Predation on *Sesarma* Field Experiment

To determine levels of predation on *Sesarma* at the study sites I performed a crab tethering experiment. The experiment was conducted with ten *Sesarma* that had a carapace width >15 cm. Experimental sample size was based on the number of crabs available from the pitfall traps sampled prior to the experimental run. There were two treatments: caged and uncaged crabs. Due to the small sample size, the experiment was only conducted at the Banca marsh. Crabs were deployed in sets consisting of a caged and uncaged crab roughly two meters apart (Fig. 7). One set was placed near the edge of a vegetated area and the other four were placed in areas exhibiting high levels of dieback. All crabs were tethered at a similar tidal height along the creek bank.

Tethering was achieved by tying 15 cm of braided fishing line around the carapace between the second and third pair of walking legs and securing the line with a dab of super glue (Holdredge et al. 2009). The other end of the line was attached to a 10 cm stake which was sunk flush with the marsh surface to anchor the tethered crab. The exclusion cages were constructed of galvanized hardware mesh (7.6 cm size holes) and were 35 cm in diameter and 60 cm in height and served to exclude potential predators (such as blue crabs, green crabs, fish, birds and raccoons). Crabs in the exclusion cages were tethered using the same method as the exposed crabs. The tethering experiment was run overnight for 12 hours at which point mortality was

assessed. Remaining pairs of crabs were left in the field under the conditions of their treatment for an additional 24 hours (36 hours total) at which point mortality was assessed again. No statistical analyses were conducted due to small sample size.

Carcinus Populations

Carcinus meanas presence was noted within the pitfall traps during the *Sesarma* abundance assessments. *Carcinus* abundance relative to marsh site and vegetation characteristics were assessed using Kruskal Wallis tests. The relationship between *Sesarma* and *Cracinus* abundance was assessed using linear regression based on crab abundance data from the creek bank traps from both marshes.

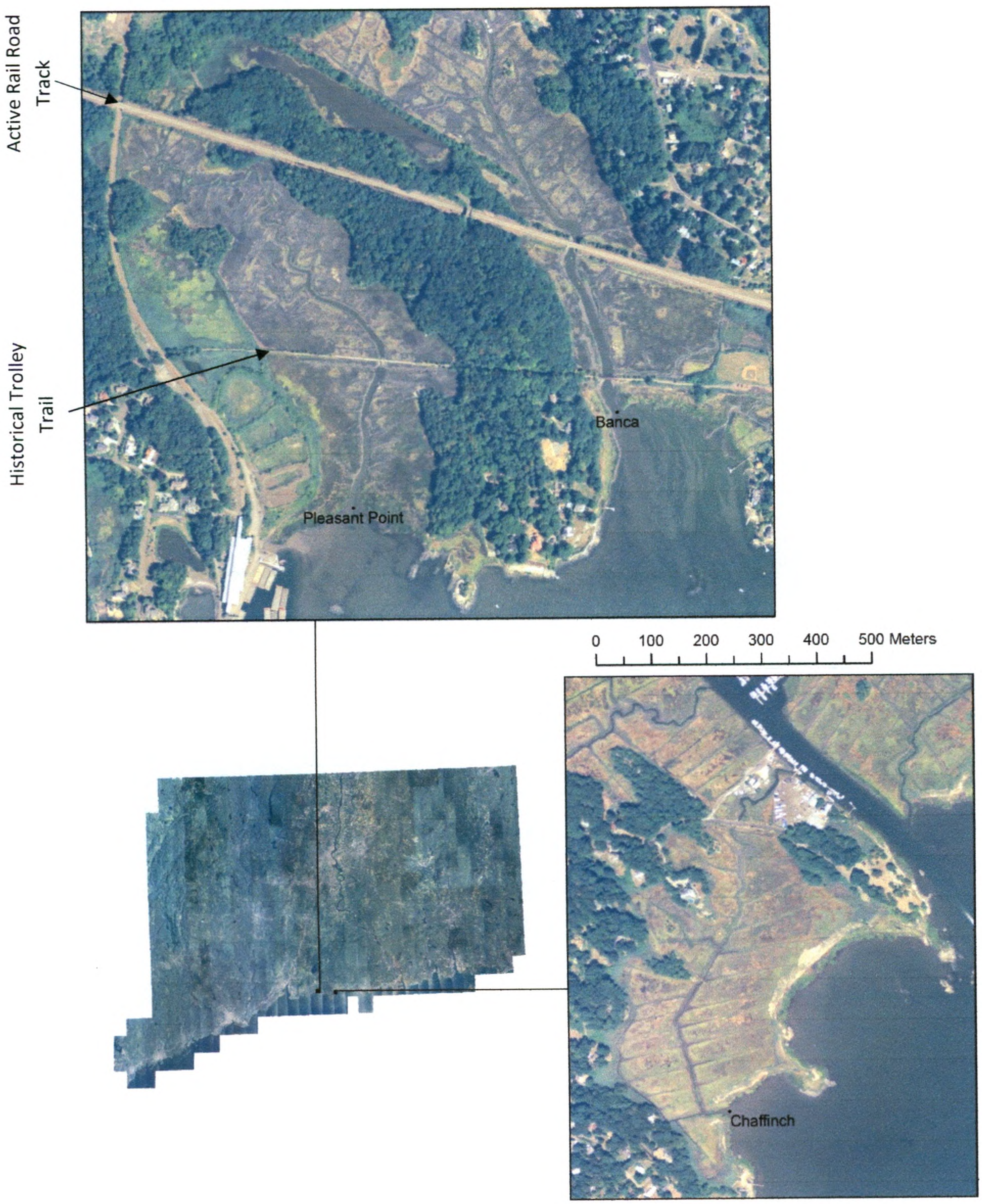

Figure 3. Locations of salt marsh study areas in Connecticut and detailed overall marsh landscapes. Imagery is from University of Connecticut MAGIC website, imagery date 2010.

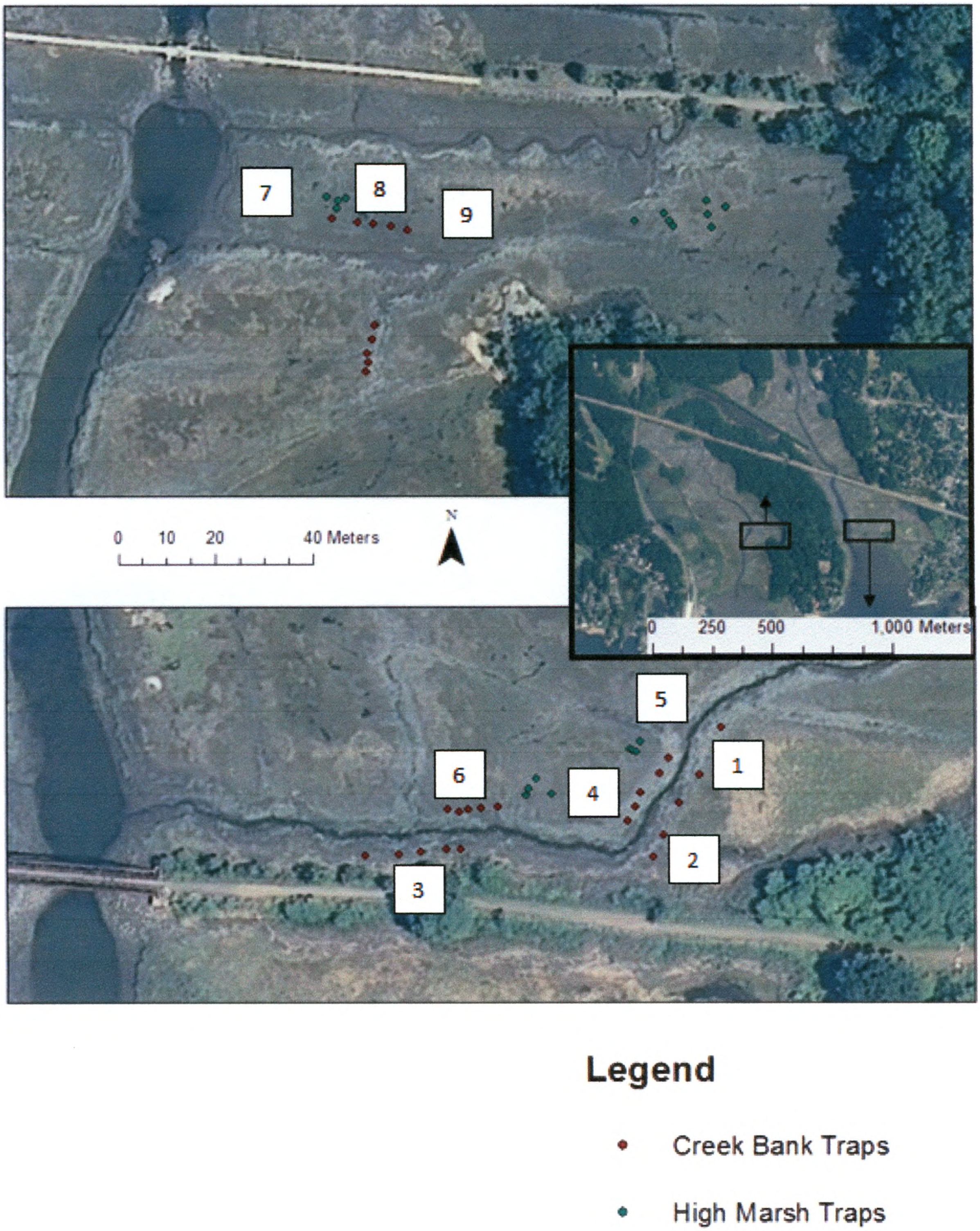

Figure 4. General locations of creek bank and high marsh pitfall traps used for assessing *Sesarma* abundance and general area of *Spartina alterniflora* transplants labeled 1-9.

Figure 5. Example of high marsh bare areas assessed for *Sesarma* activity using pitfall traps. Center trap can be seen in the photo (location indicated with arrow).

Figure 6. Herbivory experiment site located along a bare creek bank in Pleasant Point marsh. All three treatments are shown (left to right: cage control, uncaged, and exclusion cage).

Figure 7. Set of treatments for the predation experiment: predator exclusion (top), and a close up of the exposed crab treatment (bottom).

RESULTS

Historical Assessment of Vegetation Dieback

Dieback occurrence was assessed qualitatively between 1974 (1989 for Chaffinch) and 2012 within each marsh using aerial imagery. This initial analysis of the historical aerial imagery assessed, revealed that some level of dieback has been occurring since the earliest images used in this analysis in all three marshes (Figures 8-10). The first occurrences of high levels of dieback (>50%) were observed in 2000 for both Pleasant Point and Banca marshes, while in Chaffinch a high level was noted in 1990 (Fig. 11). There was a noticeable increase in dieback extent between 1986 and 1990 in all three marshes, and again between 1995 and 2000 in Pleasant Point and Chaffinch marshes (Fig. 8 and Fig. 10). The later time period corresponds with a sudden dieback event that has been documented in the literature that may have been caused by drought conditions (Alber et al., 2008). In all marshes once a high level of dieback was observed, the marsh continued to contain at least one area that exhibited a high level of dieback in subsequent years assessed.

There were significant differences (Kruskal Wallis test: $p<0.05$) in the distance to vegetation among the years analyzed (Table 1). The distance to vegetation was consistently significantly greater in 2012 than in 2004 for all creek types within all three of the salt marshes studied. Additionally, there were significant differences (Kruskal Wallis test: $p<0.05$) in the distances measured between the different creek types assessed (Table 2). The pattern and extent of dieback may vary depending on position in the marsh relative to levels of tidal inundation, distance from the coastal waterbody or hydrological alterations such as manmade constrictions. Difference in the mean distance from the creek centerline to vegetation was significantly greater

(Kruskal Wallis test: $p<0.05$) in front portion of Pleasant Point than the back portion for main and ditch creek types, both independently and combined (Table 3).

Trends in geomorphological patterns relative to distance measured to vegetation were not consistent across the three marshes studied except in 2004 when no significant differences were identified in any of the marshes (Table 4). In the 2010 and 2012 images, Pleasant Point and Banca marshes shared similar trends, with straight creek sections exhibiting significantly greater distances. In Chaffinch marsh in the years 2010 and 2012, distances along straight sections of the marsh were significantly lower than other creek sections (Table 4). Distance to vegetation was consistently greater (Kruskal Wallis test: $p<0.05$) along east and west facing creek banks than north and south facing for all creek types within all three marshes (Dunn's tests post hoc; Table 5).

The rate of change in distance to vegetation accelerated in Pleasant Point and Chaffinch marshes over the time periods assessed; no significant change in rate occurred in Banca marsh (Table 6). There were no consistent trends in rate of change based on creek type (Table 7). The rate of change in the distance to vegetation was significantly greater in the back than the front of Pleasant Point marsh during the 2004-2010 time period (Table 8). During the 2010-2012 time period the opposite was true for the ditch creek type banks, while there was no significant difference in the rate of change for main creek type banks or for the marsh site overall (Table 8). There were no consistent trends, either within or among marshes, as to whether the inside of curves, outside of curves, or straight creek bank sections had the greatest mean change in distance per year (Table 9). While there were significant differences in rate of change relative to creek heading, there were no consistent trends either across the three marsh sites or within a specific marsh (Table 10). Chaffinch marsh had a significantly higher rate of change in distance

to vegetation than Banca and Pleasant Point marshes for both time periods assessed (Table 11 and Fig. 12).

The focal sites in Pleasant Point and Banca marshes corresponded with the areas sampled for *Sesarma* abundance and seasonal vegetation growth characteristics. Aerial imagery of the focal sites from 1995, 2000, 2004 and 2010 are shown in Figures 13-15. The focal sites in all three marshes showed similar trends in variation in mean distance measured from creek centerline to vegetation in the aerial images assessed (Fig. 16).

Sesarma Population Characteristics and Activity

The mean abundance of *Sesarma* varied significantly among sampling dates over the study period (Three-way ANOVA: $F_{(13, 330)} = 4.22$, $p= 0.0000$). When considering data from all the creek bank sites sampling date differences were significant (Kruskal Wallis test $H_{13\ df}= 42.06$, $p=0.0001$). In general, abundances were higher in the late spring/early summer than in late summer/fall (Fig. 17). Crab abundance was significantly higher along bare creek banks than vegetated creek banks (Three-way ANOVA: $F_{1, 330} = 21.33$, $p= 0.0000$) (Fig. 18). There was no significant difference in *Sesarma* abundance between Banca and Pleasant Point marshes (Three-way ANOVA: $F_{1, 330} = 0.36$, $p= 0.5477$) (Fig. 19). There was however a significant interaction of marsh site and the vegetation characteristics on *Sesarma* abundance (Three-Way ANOVA: $F_{1, 330} =10.30$, $p=0.0015$). The significance of these differences were further analyzed (Kruskal Wallis: $H_{(3)\ df}=36.06$, $p=0.0000$; Dunn's test post hoc) (Fig. 20). There was no significant difference between the bare sites at either marsh, however, the vegetated sites at Pleasant Point marsh had a significantly greater abundance than the vegetated sites at Banca marsh. All other interactions among factors did not have a significant effect on *Sesarma* abundance.

Relationships with Habitat Conditions

S. alterniflora grew on average to over twice the height at vegetated sites than at bare sites by the end of the growing season (Fig. 21). Vegetated sites at both marshes exhibited significantly taller vegetation than the bare sites (Kruskal Wallis: $H_{3\ df}$= 24.81, p=0.0000; Dunn's test post hoc). In comparing the height of vegetation at the marsh sites based on vegetation characteristics, there were no significant differences in height at the Banca and Pleasant Point vegetated sites, nor the bares sites respectively.

For each *Sesarma* sampling date, I assessed several environmental factors including tidal phase, lunar phase, and maximum air and ponding water temperature (Fig. 22). Mean *Sesarma* abundance was significantly greater during collections that occurred during timeframes that were predominately low tide deployments compared to collections that occurred during predominately high tide periods (one-way ANOVA, $F_{(1,14)}$=8.47, p=0.013). The mean abundance of *Sesarma* was significantly higher on overnight sampling deployments that were greater than three days away from either the full or new moon than deployments within three days of either of these lunar phases ($H_{(3)\ df}$=10.30, p =0.02; Dunn's post hoc). There were no other significant differences related to lunar phase. There was no correlation between max air temperature and mean *Sesarma* abundance (p=0.18, R=0.14). *Sesarma* were caught during the deployment that reached the highest max air temperature as well as the lowest max temperature.

The high marsh bare patch analysis of *Sesarma* abundance indicated that there was no significant difference in abundance by date ($F_{(3, 30)}$=2.59, p=0.0710), marsh site ($F_{(1, 30)}$=0.44, p=0.5113), location within the bare patch ($F_{(3, 30)}$=0.36, p=0.7955), or any combination of the variables (Fig. 23). While on average more *Sesarma* were trapped on creek banks (mean=0.24 ±0.03) than the high marsh (M=0.17 ±0.10), the difference was not significant (Kruskal Wallis:

$H_{(1)\ df}$=0.01, p=0.94). Mean abundance of *Sesarma* within pitfall traps along creek banks and the high marsh bare patches were assessed visually to assess whether there were any potential patterns relative to geomorphological characteristics of the marsh (Fig. 24). It appears that in both the creek bank and high marsh sampling, traps located closer to the main creek of the marsh generally had a higher density of crabs than traps located further away.

Crab Size Patterns

Differences in crab size (carapace width) were analyzed in relation to date, vegetation characteristics, marsh site, and gender (including a separate category for gravid females). There was no significant difference in size relative to date (one way ANOVA, $F_{(10, 77)}$ =1.12, p=0.3603). There was no significant difference in crab size between males, females and gravid females ($H_{(2)\ df}$=5.76, p =0.0562) (Fig. 25), between Pleasant Point and Banca marsh sites ($H_{(1)}$ =2.05, p=0.1525), bare and vegetated sites ($H_{(1)}$ =0.11, p=0.7429).

***Sesarma* Herbivory Field Experiment**

Mean percent change in stem count in the exclusion cage treatment was positive (M =52.1%, SE±14.8%), which was significantly different (Friedman, $F_{(2,26)}$= 2.50, p=0.0200) from the negative change observed in both the cage control (M= -20.2%, SE±14.8%) and uncaged (M= -5.3%, SE±14.8%) treatments (Bonferroni post hoc test) (Fig. 26). *Sesarma* grazing was found in some of the uncaged and cage-control transplants. No grazing was noted in transplants located on vegetated creek banks. There was no significant difference in the percent change of stem count within a transplant culm over the study period based on whether it was located in a vegetated or bare creek bank area (Friedman, $F_{(2,26)}$= 4.74, p=0.1287). Comparing the observed impact to transplants spatially, it was noted that observed herbivory was most extreme in both marshes on bare creek banks close to the main creek channel.

Predation on *Sesarma* Field Experiment

There were not enough crabs available for the predation study to conduct a statistical analysis. After the initial 12-hour overnight deployment, one of the exposed *Sesarma* crabs was gone from its tether (Fig. 27A). After 36 hours from the initial deployment, two of the remaining four exposed crabs were also gone. In the caged treatment at the 36 hour observation one of the crabs had died from stress. Evidence that *Sesarma* predation occurs within Pleasant Point and Banca marshes was also made on the level of personal observation; one example of such observation in Pleasant Point marsh is shown in Figure 27B.

Crab Relationships

Carcinus meanus, which has been recorded in the literature as a predator of *Sesarma* (Bertness and Coverdale, 2013), was more abundant (Kruskal Wallis, $H_{(1)\ df}$=10.92; p=0.0009) at the Pleasant Point marsh (M=0.26 ±0.03) than the Banca marsh (M=0.06 ±0.02). While *Sesarma* were trapped more often at bare sites, there was no significance relative to site vegetation characteristics on *Carcinus* abundance (Kruskal Wallis, $H_{(1)\ df}$=0.59; p=0.4406). There is no relationship (R^2 =0.0007, p =0.7902) between *Sesarma* abundance and *Carcinus* abundance among the various collection points within both marshes (Fig. 28).

able 1. Mean distance (meters) from creek centerline to vegetation for 2004, 2010 and 2012 nagery at each of the salt marsh study sites for each creek type and all creek types combined for ach marsh site. Kruskal-Wallis analyses with Dunn's test post hoc performed to assess ifferences in distance over time for each Creek Type within each marsh.

	Creek Type	$\chi2$	P value	Results of post hoc test	Mean Distance (SE) 2004	2010	2012
Banca	Branch	124.53	0.0000	'12 > '10 > '04	0.34 (0.10)	1.24 (0.10)	1.36 (0.10)
	Ditch	79.73	0.0000	'12 > '10 > '04	0.18 (0.10)	0.85 (0.10)	1.01 (0.10)
	Main	11.84	0.0027	'12 > '04	11.30 (0.26)	11.88 (0.26)	12.37 (0.26)
	All Creeks	96.93	0.0000	'12 > '10 > '04	2.73 (0.25)	3.50 (0.25)	3.71 (0.25)
Chaffinch	Branch	42.35	0.0000	'10 & '12 > '04	0.00 (0.53)	4.18 (0.53)	3.36 (0.53)
	Ditch	486.42	0.0000	'12 > '10 > '04	0.12 (0.06)	0.65 (0.05)	1.67 (0.05)
	Main	268.26	0.0000	'12 > '10 > '04	0.60 (0.12)	2.23 (0.12)	3.80 (0.12)
	All Creeks	669.57	0.0000	'12 > '10 > '04	0.29 (0.07)	1.32 (0.07)	2.48 (0.07)
Pleasant Point	Branch	51.68	0.0000	'12 > '10 > '04	0.17 (0.05)	0.35 (0.05)	0.73 (0.05)
	Ditch	146.90	0.0000	'12 > '10 > '04	0.73 (0.10)	0.98 (0.10)	1.80 (0.10)
	Main	30.90	0.0000	'12 > '10 & '04	5.04 (0.17)	4.84 (0.17)	5.93 (0.17)
	All Creeks	62.25	0.0000	'12 > '10 > '04	2.99 (0.13)	2.99 (0.13)	3.93 (0.13)

Table 2. Differences in mean distance (meters) from creek centerline to vegetation by creek type and year for each marsh. Kruskal-Wallis analyses with Dunn's test post hoc analysis performed to assess differences in distance measured based on creek type within each marsh and imagery year analyzed.

Marsh	Year	$\chi2$	P value	Results of post hoc test	Mean Distance (SE) Main	Mean Distance (SE) Branch	Mean Distance (SE) Ditch
Banca	2004	209.31	0.0000	M > B > D	11.30 (0.12)	0.34 (0.08)	0.18 (0.11)
	2010	196.50	0.0000	M > B > D	11.88 (0.21)	1.24 (0.14)	0.85 (0.18)
	2012	190.47	0.0000	M > B > D	12.37 (0.17)	1.36 (0.11)	1.01 (0.15)
Chaffinch	2004	148.38	0.0000	M > D & B	0.60 (0.03)	0.00 (0.11)	0.12 (0.03)
	2010	130.19	0.0000	B > M > D	2.23 (0.11)	4.18 (0.37)	0.65 (0.08)
	2012	163.81	0.0000	M & B > D	3.80 (0.12)	3.36 (0.39)	1.67 (0.09)
Pleasant Point	2004	381.51	0.0000	M > D & B	5.04 (0.14)	0.17 (0.40)	0.73 (0.15)
	2010	380.39	0.0000	M > D > B	4.84 (0.12)	0.35 (0.36)	0.98 (0.14)
	2012	334.41	0.0000	M > D > B	5.93 (0.15)	0.73 (0.44)	1.80 (0.17)

Table 3. Differences in mean distance (meters) from creek centerline to vegetation by creek type and year for Pleasant Point front and Pleasant Point back. Kruskal-Wallis analyses with Dunn's test post hoc analyses performed to assess differences in distance measured in Pleasant Point front marsh compared to Pleasant Point back marsh by creek type. Distances in bold indicate that they are significantly greater than the other marsh area.

Year	Creek Type	$\chi 2$	P value	Mean Distance (SE) Front	Mean Distance (SE) Back
2004	Ditch	75.56	0.0000	**1.04** (0.10)	0.05 (0.15)
	Main	163.01	0.0000	**8.04** (0.21)	3.41 (0.16)
	Ditch & Main	7.81	0.0052	**3.84** (0.19)	2.51 (0.19)
2010	Ditch	1. 08	0.1802	1.13 (0.09)	0.66 (0.13)
	Main	165.92	0.0000	**7.48** (0.21)	3.39 (0.15)
	Ditch & Main	0.78	0.3784	3.67 (0.17)	2.67 (0.17)
2012	Ditch	45.34	0.0000	**2.29** (0.16)	0.69 (0.23)
	Main	159.82	0.0000	**8.55** (0.24)	4.50 (0.18)
	Ditch & Main	10.67	0.0011	**4.79** (0.20)	3.49 (0.20)

Table 4. Differences in mean distance (meters) from creek centerline to vegetation by creek geomorphology (S=straight, O=outside bend, I=inside bend) for each year analyzed for each marsh. Kruskal-Wallis analyses with Dunn's test post hoc analyses performed to assess differences in distance measured based on creek geomorphology within each marsh for each year imagery was analyzed.

Marsh	Year	$\chi 2$	P value	Results of post hoc test	Mean Distance (SE)		
					Inside	Outside	Straight
Banca	2004	4.37	0.1122	None	0.23 (0.84)	0.38 (0.84)	3.22 (0.27)
	2010	12.56	0.0019	S > O & I	0.99 (1.02)	1.56 (1.02)	3.80 (0.27)
	2012	21.77	0.0000	S > I & O	1.31 (0.84)	0.82 (0.84)	4.28 (0.27)
Chaffinch	2004	2.54	0.2810	None	0.51 (0.17)	0.27 (0.17)	0.30 (0.02)
	2010	22.76	0.0000	O & I > S	3.56 (0.45)	3.93 (0.45)	1.19 (0.07)
	2012	11.88	0.0026	I > O & S	3.98 (0.45)	2.5 (0.45)	2.43 (0.08)
Pleasant Point	2004	1.22	0.5427	None	1.77 (0.44)	2.00 (0.44)	3.35 (0.15)
	2010	13.68	0.0011	S > O & I	1.85 (0.33)	1.89 (0.33)	3.36 (0.14)
	2012	21.85	0.0000	S > O & I	2.44 (0.41)	2.82 (0.41)	4.30 (0.15)

Table 5. Differences in mean distance (meters) from creek centerline to vegetation by creek bank heading (N, S, E or W) and year for each marsh. Kruskal-Wallis analyses with post hoc Dunn's test performed to assess differences in distance measured based on creek bank heading within each marsh for each year analyzed.

Marsh	Year	χ2	P value	Results of post hoc test	Mean Distance (SE) N	S	E	W
Banca	2004	79.23	0.0000	W & E > N & S	0.36 (0.36)	0.21 (0.36)	5.93 (0.43)	6.53 (0.43)
	2010	134.22	0.0000	W & E > S > N	0.62 (0.34)	1.37 (0.34)	7.11 (0.44)	8.27 (0.44)
	2012	98.54	0.0000	W > E > S > N	0.92 (0.36)	1.56 (0.36)	6.79 (0.45)	8.22 (0.45)
Chaffinch	2004	261.28	0.0000	E & W > N & S	0.04 (0.04)	0.00 (0.04)	0.60 (0.04)	0.56 (0.04)
	2010	206.74	0.0000	E & W > S & N	0.37 (0.14)	0.50 (0.14)	2.12 (0.13)	2.07 (0.13)
	2012	136.51	0.0000	E & W > N > S	1.76 (0.15)	1.34 (0.15)	3.53 (0.14)	3.10 (0.14)
Pleasant Point	2004	213.14	0.0000	E & W > N > S	2.06 (0.19)	1.29 (0.19)	6.34 (0.26)	5.09 (0.26)
	2010	170.93	0.0000	E & W > N & S	2.02 (0.18)	1.62 (0.18)	5.42 (0.24)	5.00 (0.24)
	2012	156.27	0.0000	E > W > N & S	3.02 (0.21)	2.37 (0.21)	6.98 (0.29)	5.32 (0.29)

Table 6. Differences in mean rate of change in distance (meters) from creek centerline to vegetation between 2004-2010 and 2010-2012. Bold numbers indicate that the rate of change was significantly greater for that time span than the corresponding time span for that creek type within that marsh. Kruskal-Wallis analyses with post hoc Dunn's Test analyses performed to assess differences between the time spans in the rate of change in distance measured for each marsh by creek type.

	Creek Type	χ2	P value	Mean Change/Year (SE) 2004-2010	Mean Change/Year (SE) 2010-2012
Banca	Branch	0.02	0.8964	0.15 (0.06)	0.06 (0.06)
	Ditch	3.00	0.0835	0.11 (0.06)	0.08 (0.06)
	Main	3.57	0.0590	0.10 (0.07)	0.25 (0.07)
	All Creek Types	3.35	0.0674	0.13 (0.04)	0.11 (0.04)
Chaffinch	Branch	23.24	0.0000	**4.18** (0.61)	-0.82 (0.61)
	Ditch	56.16	0.0000	0.53 (0.06)	**0.99** (0.06)
	Main	1.57	0.2100	1.63 (0.15)	1.56 (0.15)
	All Creek Types	23.11	0.0000	1.04 (0.07)	**1.14** (0.07)
Pleasant Point	Branch	27.02	0.0000	0.03 (0.02)	**0.19** (0.02)
	Ditch	36.00	0.0000	0.04 (0.04)	**0.41** (0.04)
	Main	134.20	0.0000	-0.03 (0.03)	**0.55** (0.03)
	All Creek Types	190.84	0.0000	0.00 (0.02)	**0.47** (0.02)

Table 7. Differences in mean rate of change in distance (meters) from creek centerline to vegetation by creek type between 2004-2010 and 2010-2012. Kruskal-Wallis analyses with Dunn's Test post hoc analyses performed to assess differences among creek types for each marsh and time span.

Marsh	Time Span	$\chi 2$	P value	Results of post hoc test	Mean Change/Year (SE)		
					Main	Branch	Ditch
Banca	2004-2010	1.54	0.4630	No significance	0.10 (0.03)	0.15 (0.02)	0.11 (0.03)
	2010-2012	1.69	0.4294	No significance	0.25 (0.11)	0.06 (0.07)	0.08 (0.09)
Chaffinch	2004-2010	70.35	0.0000	B > M > D	1.63 (0.11)	4.18 (0.35)	0.53 (0.08)
	2010-2012	41.10	0.0000	M > D > B	1.56 (0.12)	-0.82 (0.40)	0.99 (0.09)
Pleasant Point	2004-2010	33.63	0.0000	D > M	-0.03 (0.01)	0.03 (0.04)	0.04 (0.02)
	2010-2012	20.05	0.0000	M > D & B	0.55 (0.04)	0.19 (0.13)	0.41 (0.05)

Table 8. Differences in mean rate of change in distance (meters) from creek centerline to vegetation by creek type and year span for Pleasant Point front and Pleasant Point back. Kruskal-Wallis analyses with Dunn's Test analyses performed to assess differences over time in the rate of change in distance measured between Pleasant Point front marsh and Pleasant Point back marsh by creek type. Numbers in bold indicate that the mean change per year was significantly greater than the other marsh area.

				Mean Change/Year (SE)	
Time Period	Creek Type	$\chi 2$	P value	Front	Back
2004-2010	Ditch	17.84	0.0000	0.02 (0.02)	**0.10** (0.03)
	Main	15.12	0.0001	-0.09 (0.02)	**-0.00** (0.02)
	Ditch & Main	9.57	0.0020	-0.03 (0.01)	**0.03** (0.01)
2010-2012	Ditch	28.45	0.0000	**0.58** (0.06)	0.02 (0.10)
	Main	1.13	0.2880	0.53 (0.07)	0.55 (0.05)
	Ditch & Main	0.63	0.4258	0.56 (0.05)	0.41 (0.05)

Table 9. Differences in mean rate of change in distance (meters) from creek centerline to vegetation based on creek geomorphology (S=straight, O=outside bend, I=inside bend). Kruskal-Wallis analyses with Dunn's Test post hoc analyses performed to assess differences in rates based on creek geomorphology within each marsh for each of the two time periods.

Marsh	Time Period	χ2	P value	Results of post hoc test	Mean Change/Year (SE) Inside	Outside	Straight
Banca	2004-2010	5.18	0.0751	S is different from I	0.12 (0.05)	0.23 (0.05)	0.12 (0.01)
Banca	2010-2012	2.49	0.2875	none	-0.09 (0.17)	-0.23 (0.17)	0.16 (0.05)
Chaffinch	2004-2010	25.39	0.0000	O & I > S	3.21 (0.45)	3.9 (0.45)	0.92 (0.07)
Chaffinch	2010-2012	11.19	0.0037	S > O	0.44 (0.48)	-1.36 (0.48)	1.21 (0.07)
Pleasant Point	2004-2010	23.01	0.0000	I & O > S	0.09 (0.03)	0.07 (0.03)	-0.03 (0.01)
Pleasant Point	2010-2012	9.87	0.0072	S > O & I	0.26 (0.09)	0.36 (0.09)	0.52 (0.04)

Table 10. Differences in mean rate of change in distance (meters) from creek centerline to vegetation based creek bank heading (N, S, E or W). Kruskal-Wallis analyses with Dunn's Test post hoc analyses performed to assess differences rate based on creek bank heading within each marsh for both time periods analyzed.

Marsh	Time Period	$\chi2$	P value	Results of post hoc test	Mean Change/Year (SE) N	S	E	W
Banca	2004-2010	23.25	0.0000	S & W > E & N	0.05 (0.02)	0.18 (0.02)	0.10 (0.03)	0.18 (0.03)
	2010-2012	1.01	0.7986	None	0.13 (0.09)	0.12 (0.09)	0.10 (0.11)	0.07 (0.11)
Chaffinch	2004-2010	63.63	0.0000	E & W > S & N	0.33 (0.14)	0.49 (0.14)	1.59 (0.13)	1.58 (0.13)
	2010-2012	12.23	0.0066	E & N > W & S	1.40 (0.15)	0.83 (0.15)	1.40 (0.14)	0.91 (0.14)
Pleasant Point	2004-2010	30.79	0.0000	S W & N > E W is different from S	-0.00 (0.02)	0.06 (0.02)	-0.12 (0.02)	0.02 (0.02)
	2010-2012	49.24	0.0000	E > N & S > W	0.51 (0.06)	0.38 (0.06)	0.82 (0.08)	0.19 (0.08)

Table 11. Differences in mean rate of change in distance (meters) from creek centerline to vegetation by marsh site. Kruskal-Wallis analyses with Dunn's Test post hoc analyses performed to assess differences in rate based on marsh site for both time periods analyzed.

Time Period	$\chi 2$	P value	Results of post hoc test	Mean Change/Year (SE) Banca	Chaffinch	Pleasant Point
2004-2010	220.12	0.0000	Chaffinch > Banca & Pleasant Point	0.1274	1.0387	0.0007
2010-2012	162.13	0.0000	Chaffinch > Pleasant Point > Banca	0.1069	1.1361	0.4678

Table 12. Survivorship of *Sesarma* from the caged and uncaged treatments of the predation experiment. The 36 hour deployment is a continuation (additional 24 hours) of the 12 hour deployment. Asterisk denotes incidences in which means other than predation affected survival (i.e. stress).

	Caged Treatment		Uncaged Treatment	
Deployment Length	Crabs Deployed	# Survived	Crabs Deployed	# Survived
12 hours	5	5	5	4
36 hours	4	3*	4	2

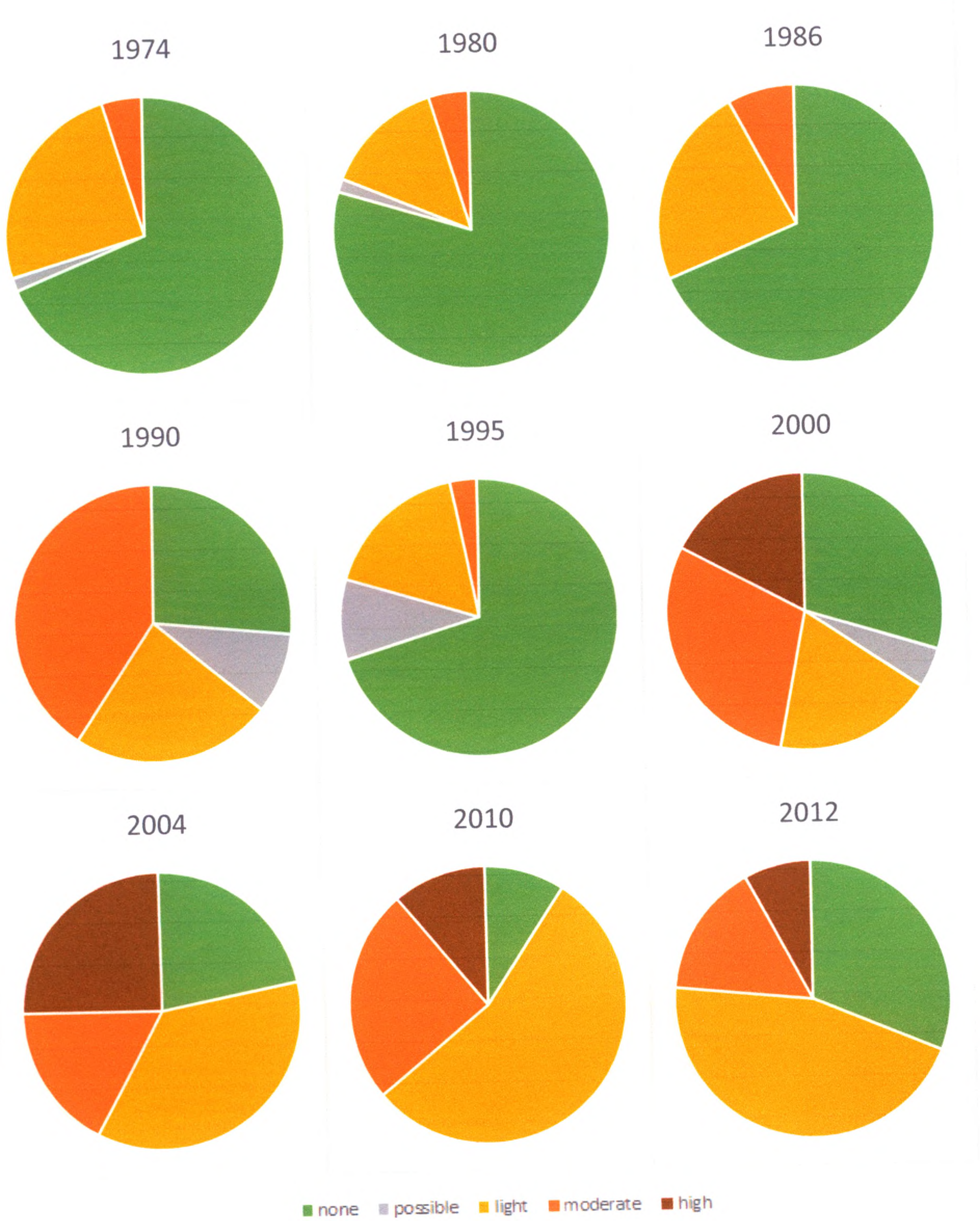

Figure 8. General trend of dieback occurrence for Pleasant Point marsh.

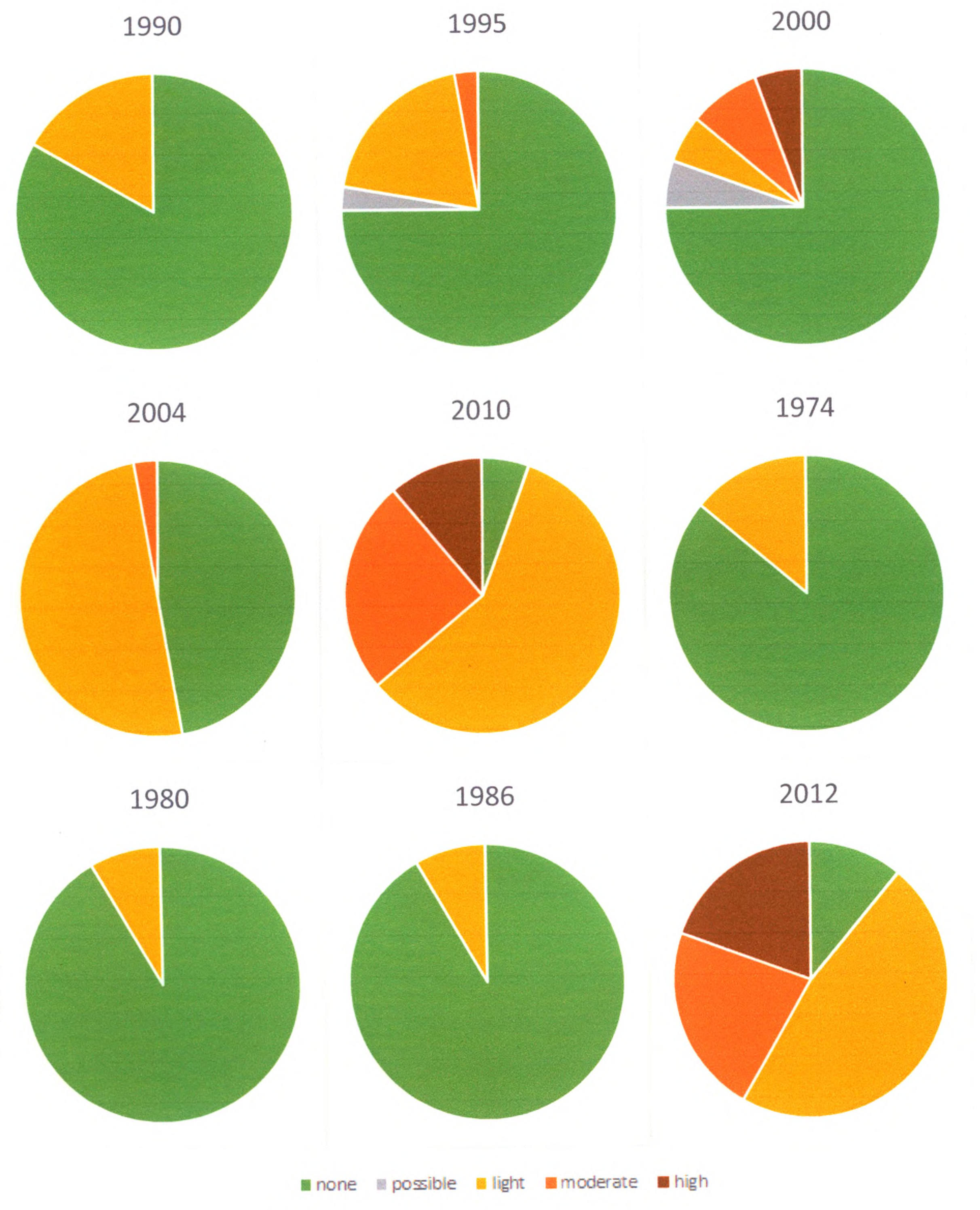

Figure 9. General trend of dieback occurrence for Banca marsh.

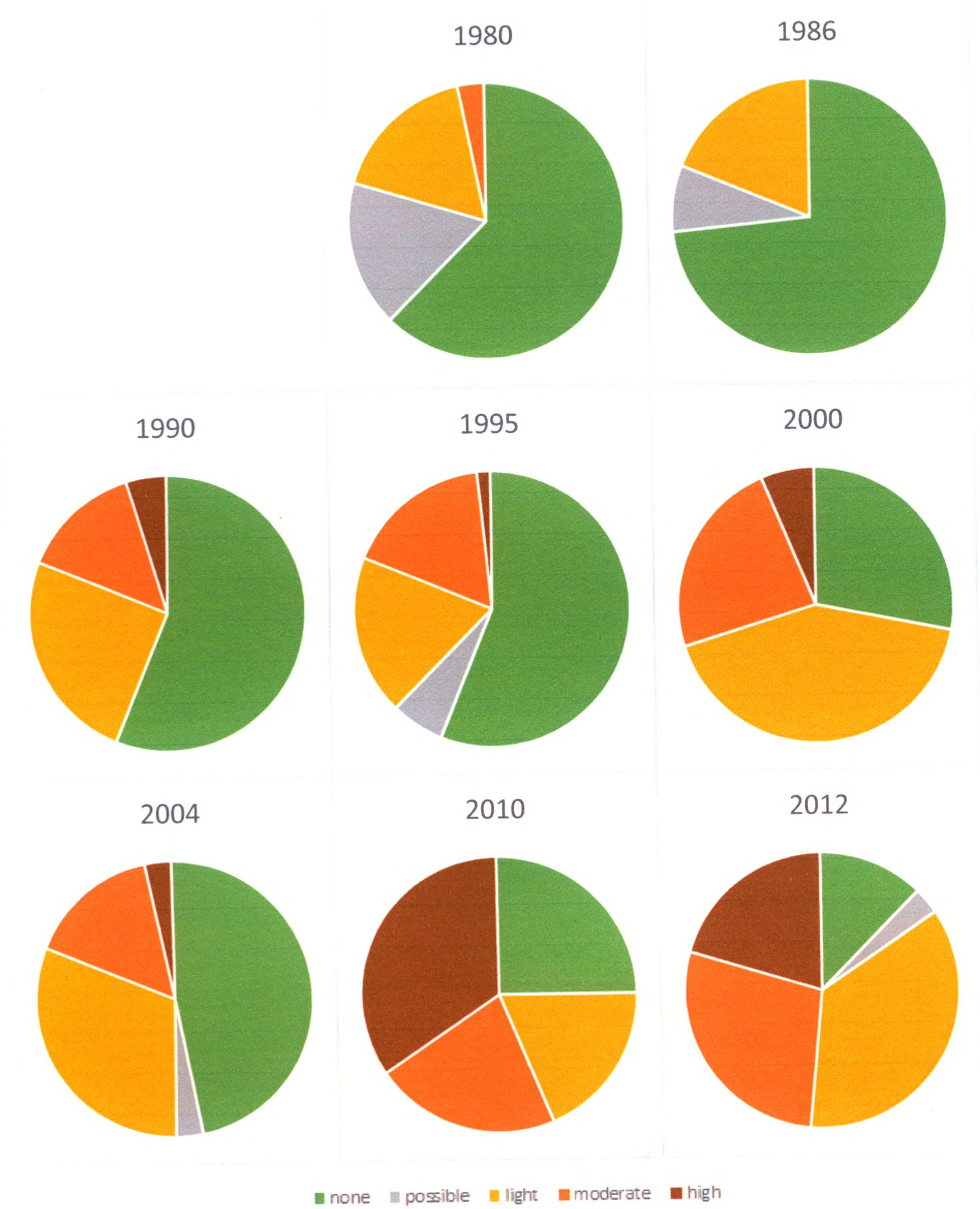

Figure 10. General trend of dieback occurrence for Chaffinch marsh.

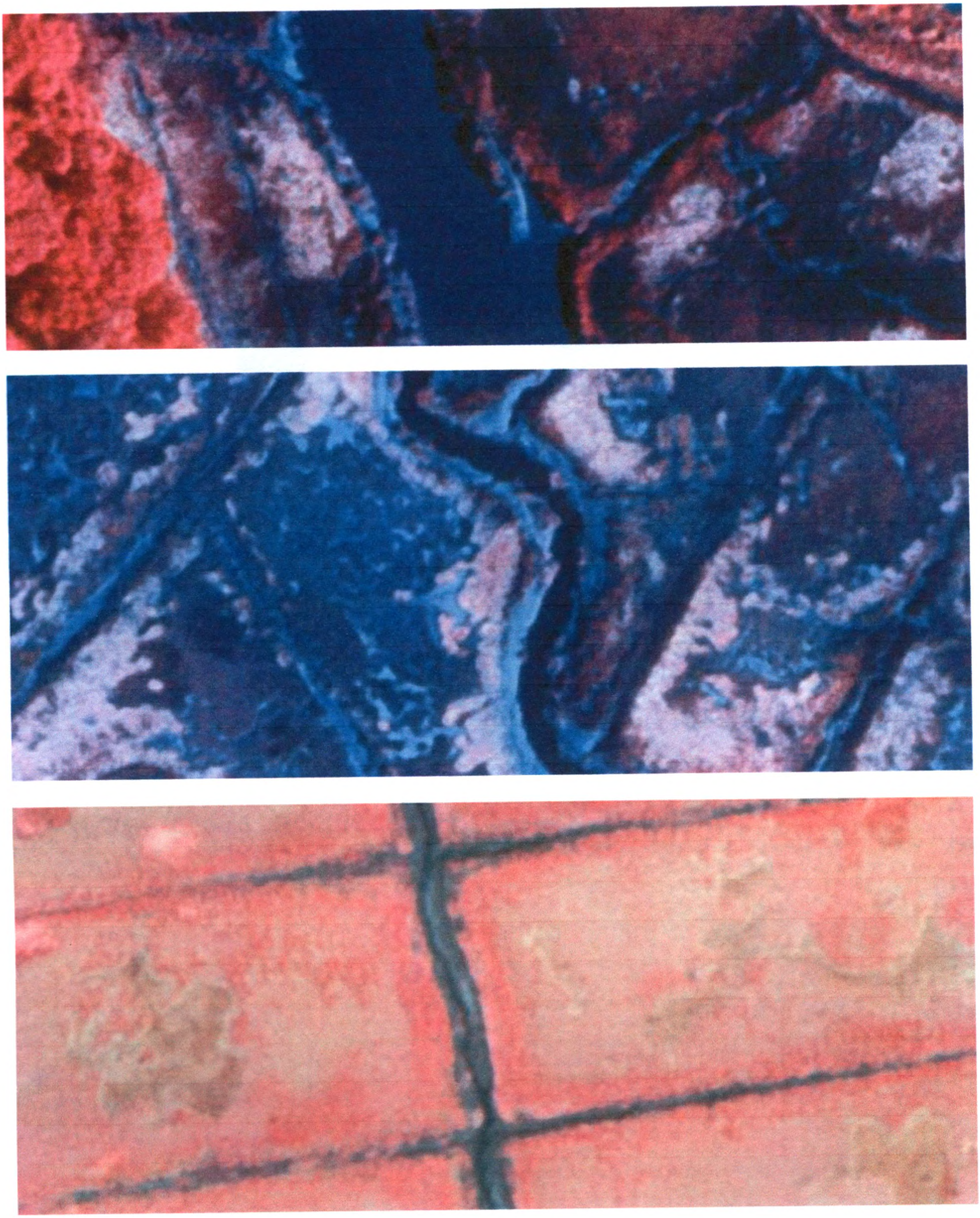

Figure 11. Example of early marsh dieback that was classified as high, greater than 50% of the creek bank, in: Banca marsh from 2000 (top), Pleasant Point marsh from 2000 (middle) and Chaffinch marsh 1990 (bottom).

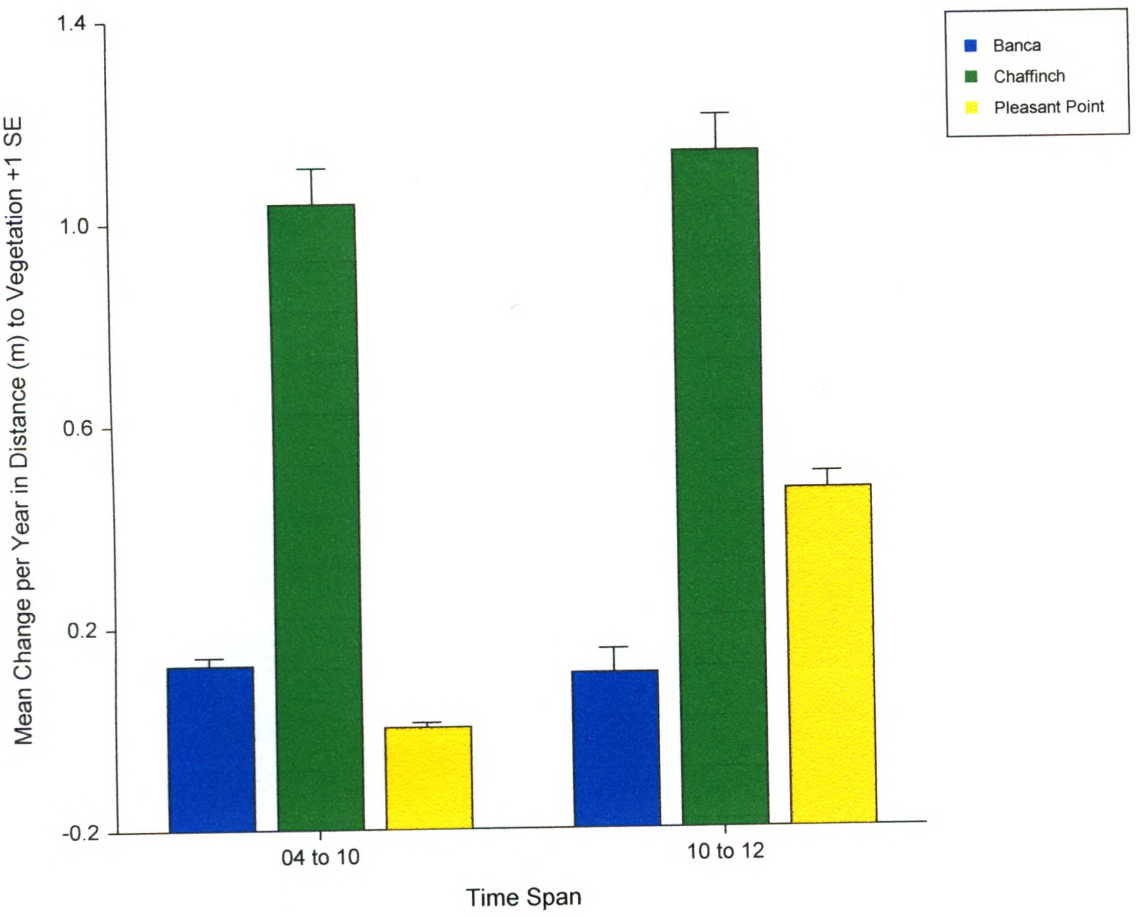

Figure 12. Mean change per year in distance (meters) from creek centerline to vegetation between 2004-2010 and 2010-2012 for each of the marsh sites assessed.

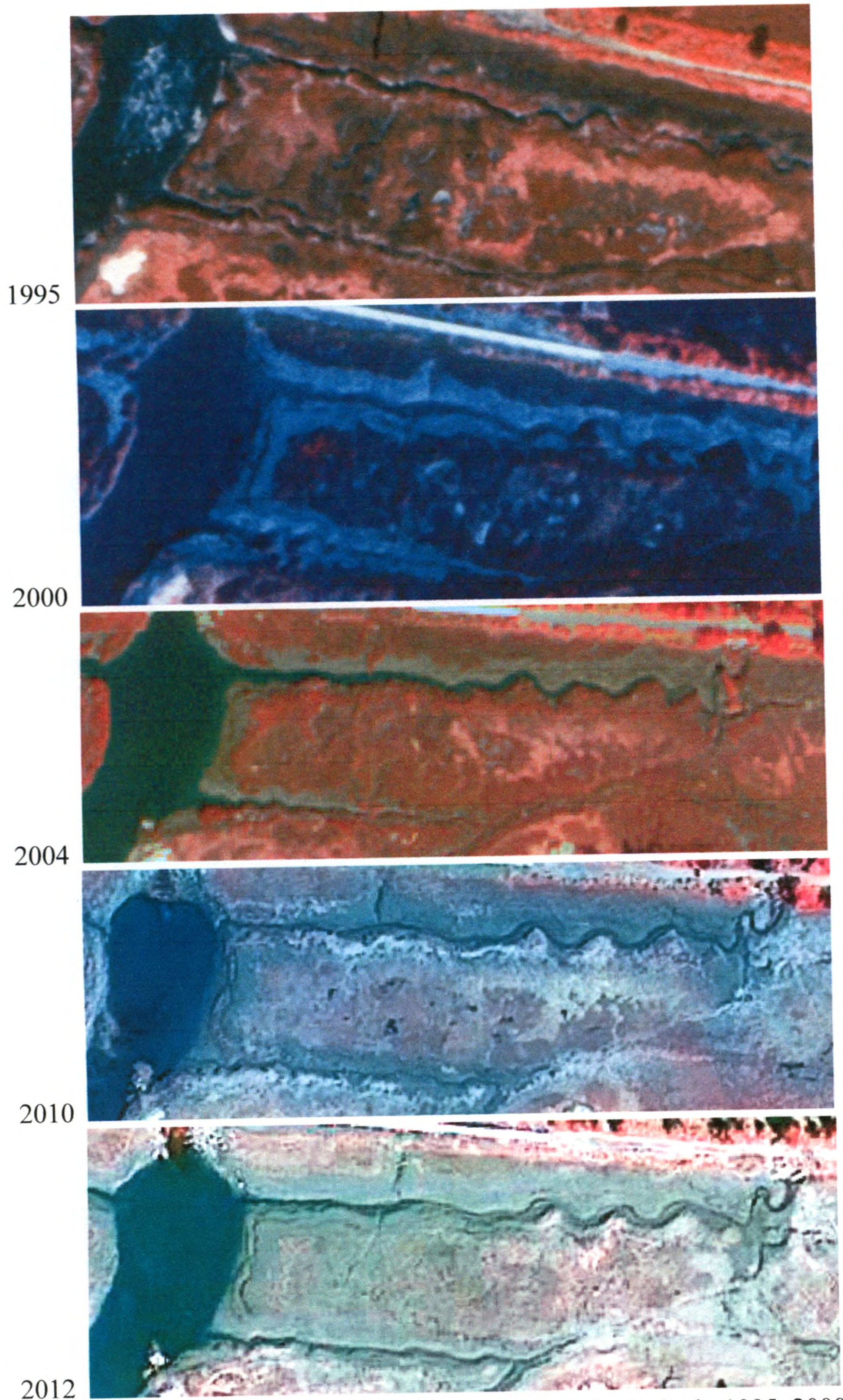

Figure 13. Top to bottom aerial imagery of a Pleasant Point dieback site in 1995, 2000, 2004, and 2010. The high marsh is not flooded in any of the imagery; however, tidal stage does vary slightly between the years (2010 imagery appears to have been taken at the lowest tidal stage).

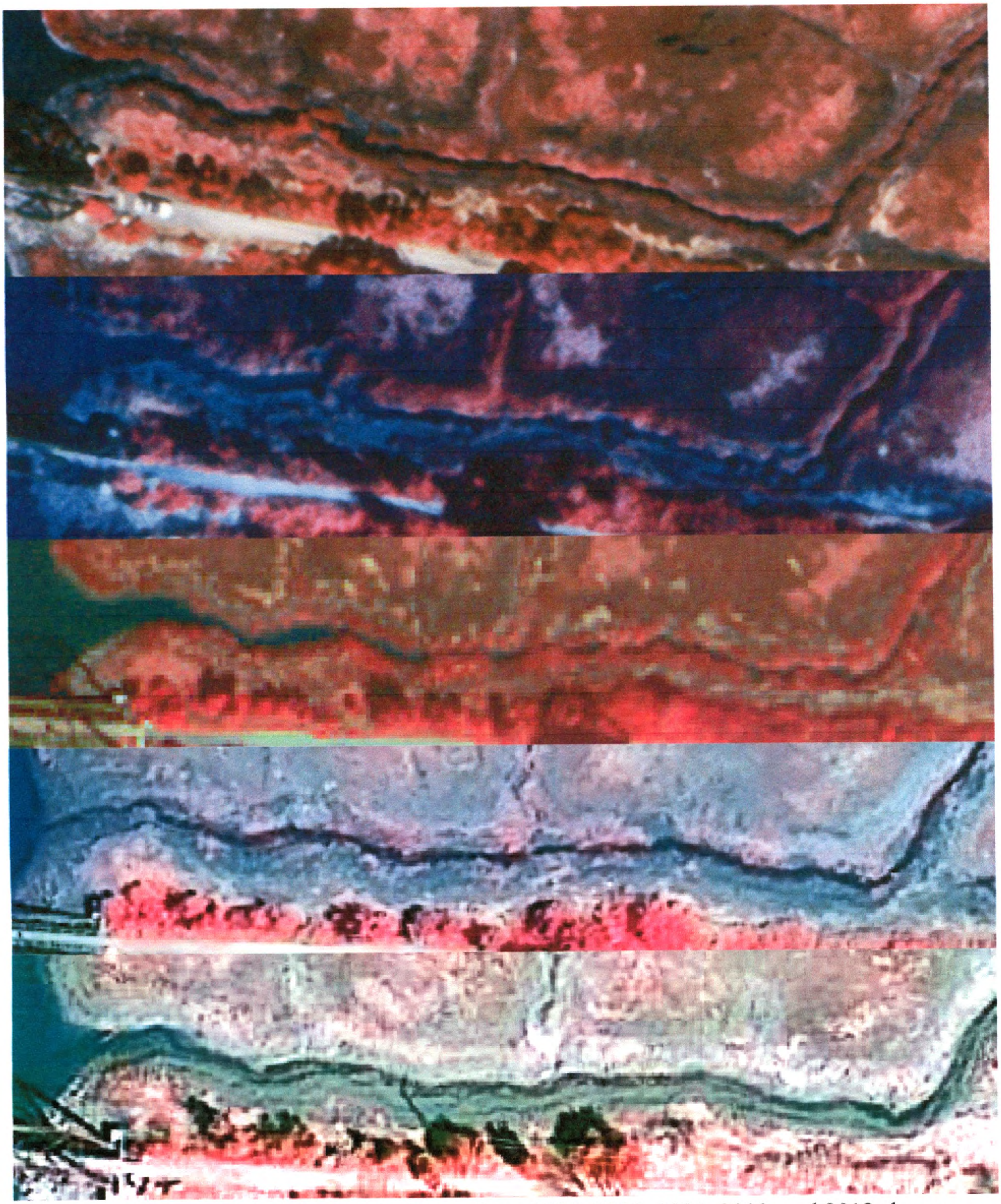

Figure 14. Imagery of creek branch in Banca from 1995, 2000, 2004, 2010, and 2012 shows changes in salt marsh vegetation and creek characteristics.

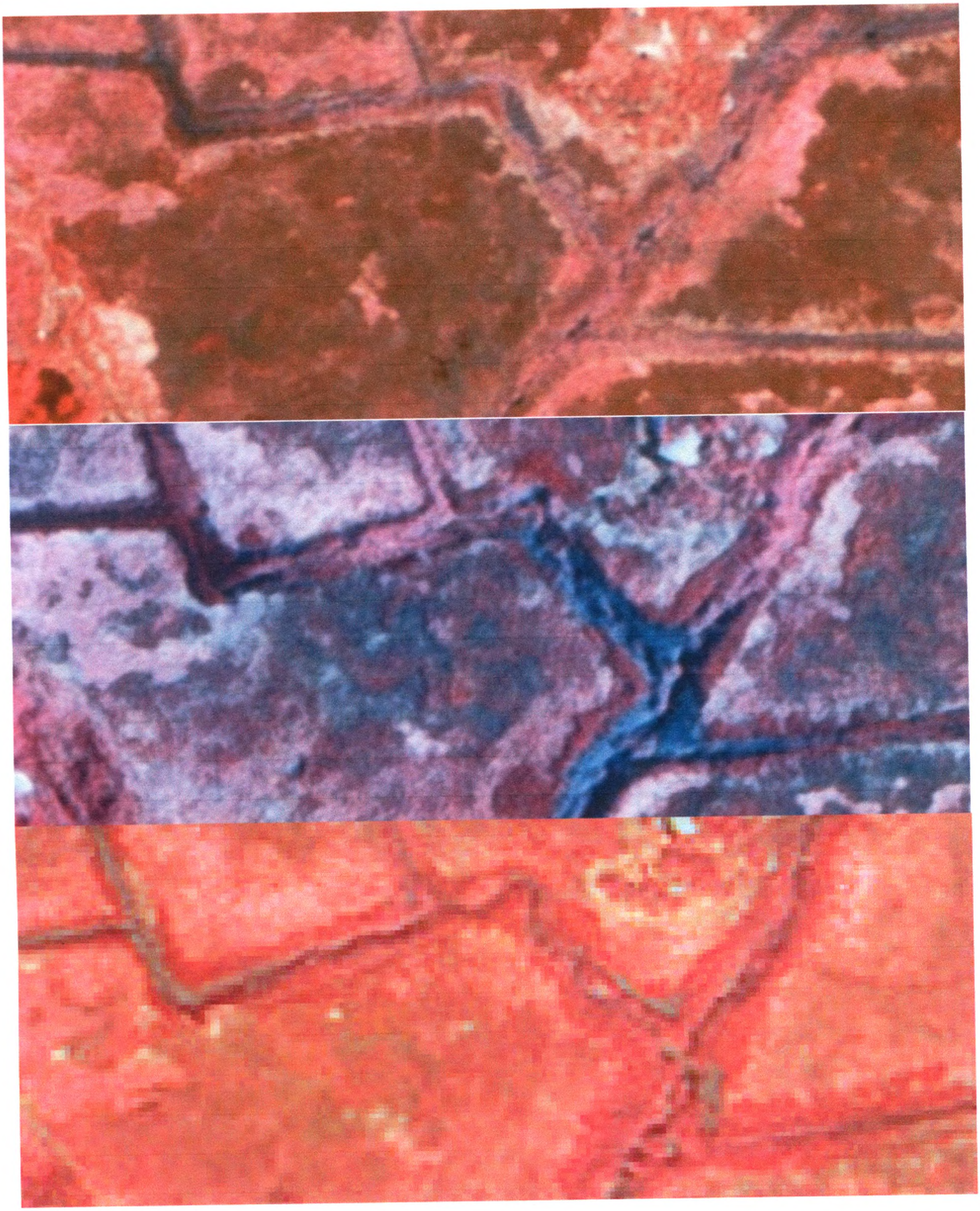

Figure 15 continued onto next page

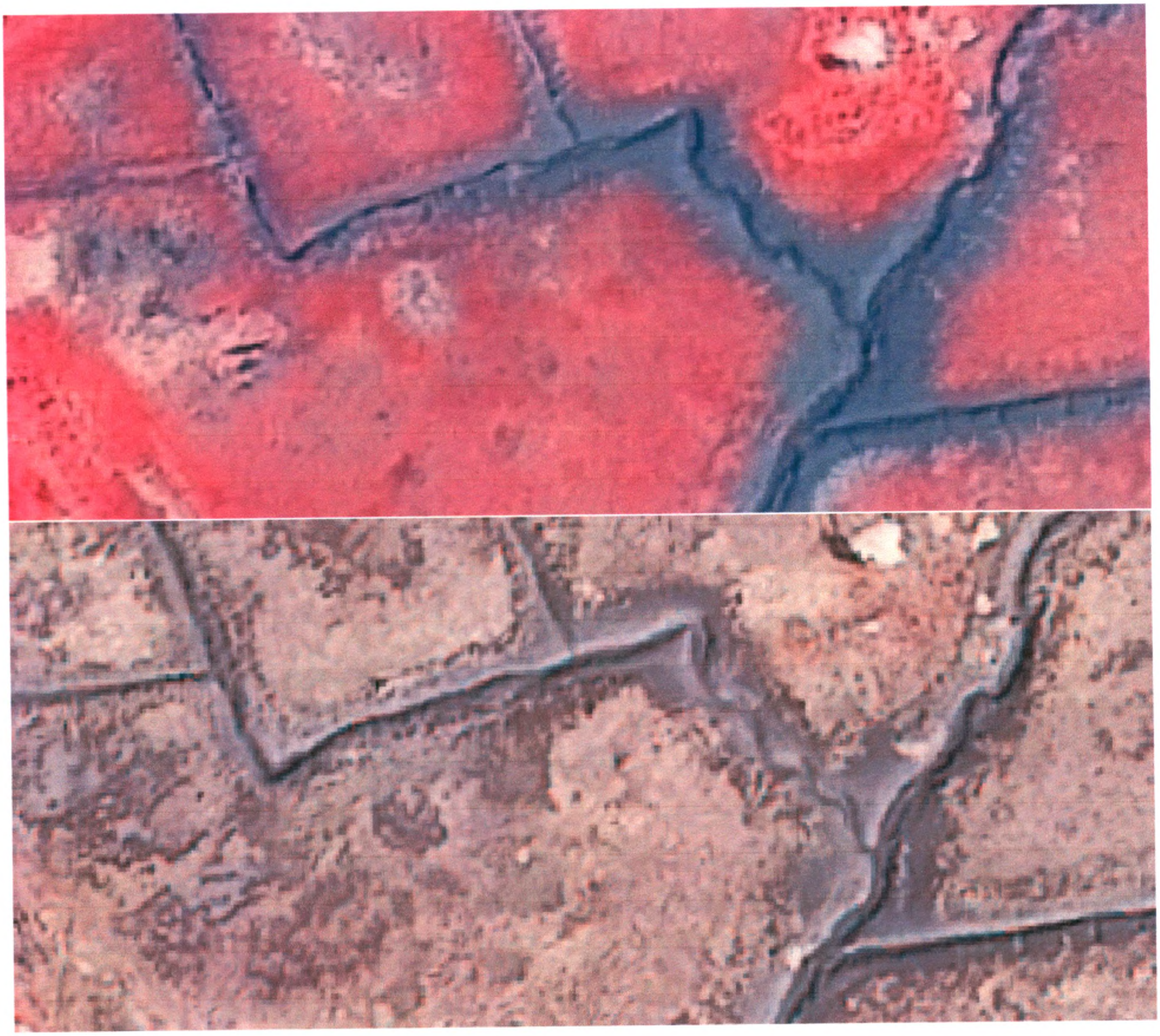

Figure 15. The branch site in chaffinch marsh (top to bottom page 1: 1995, 2000 and 2004, page 2 top to bottom: 2010 and 2012).

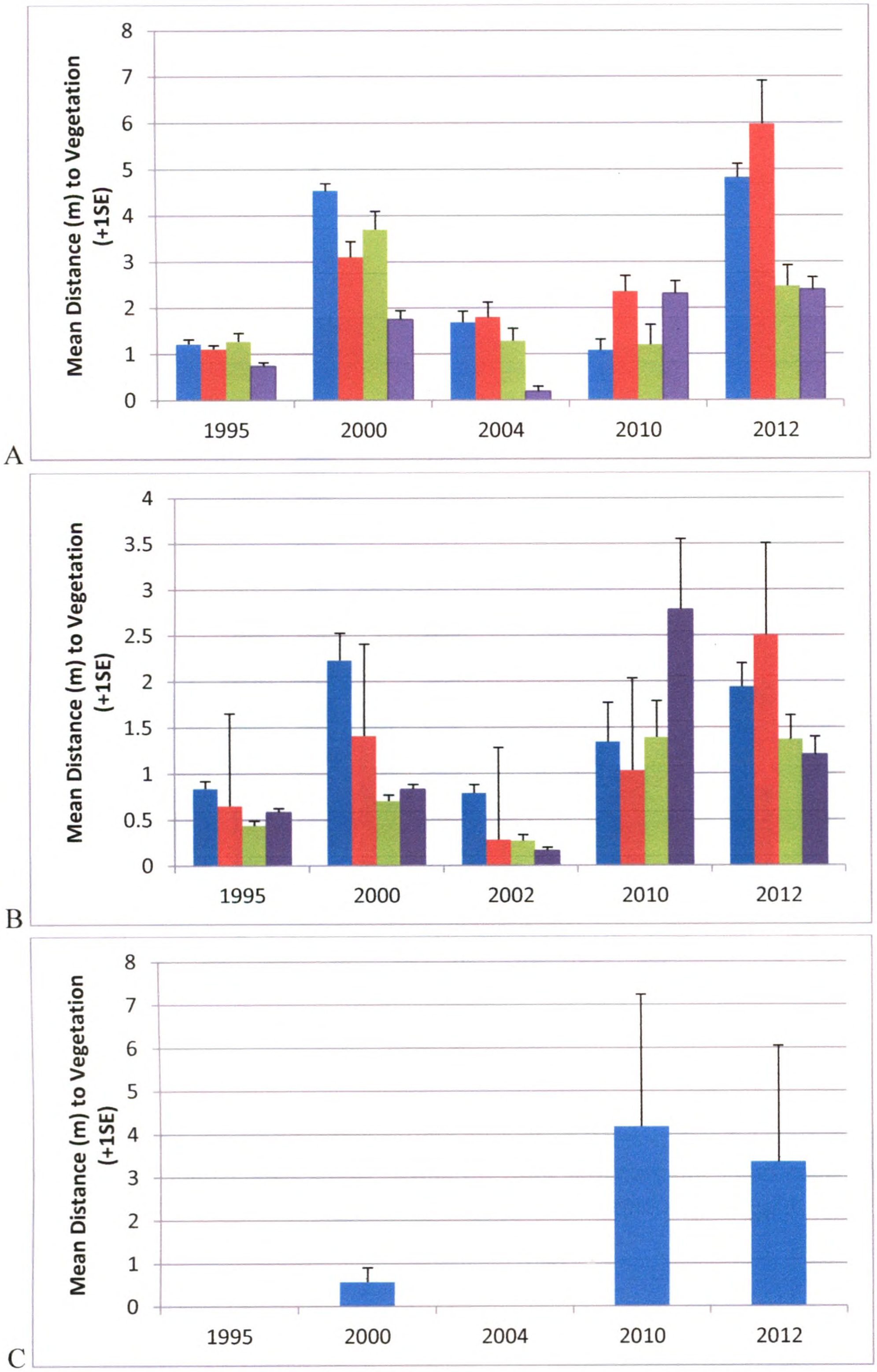

Figure 16. Mean distance (+1 SE) from ditch centerline to nearest vegetation for each marsh site: Pleasant Point marsh is displayed at top, Banca marsh in the middle and Chaffinch at the bottom. Each bar indicates a transect area studied. Transects were of the focal sites and are located within the areas shown in Figures 12-14.

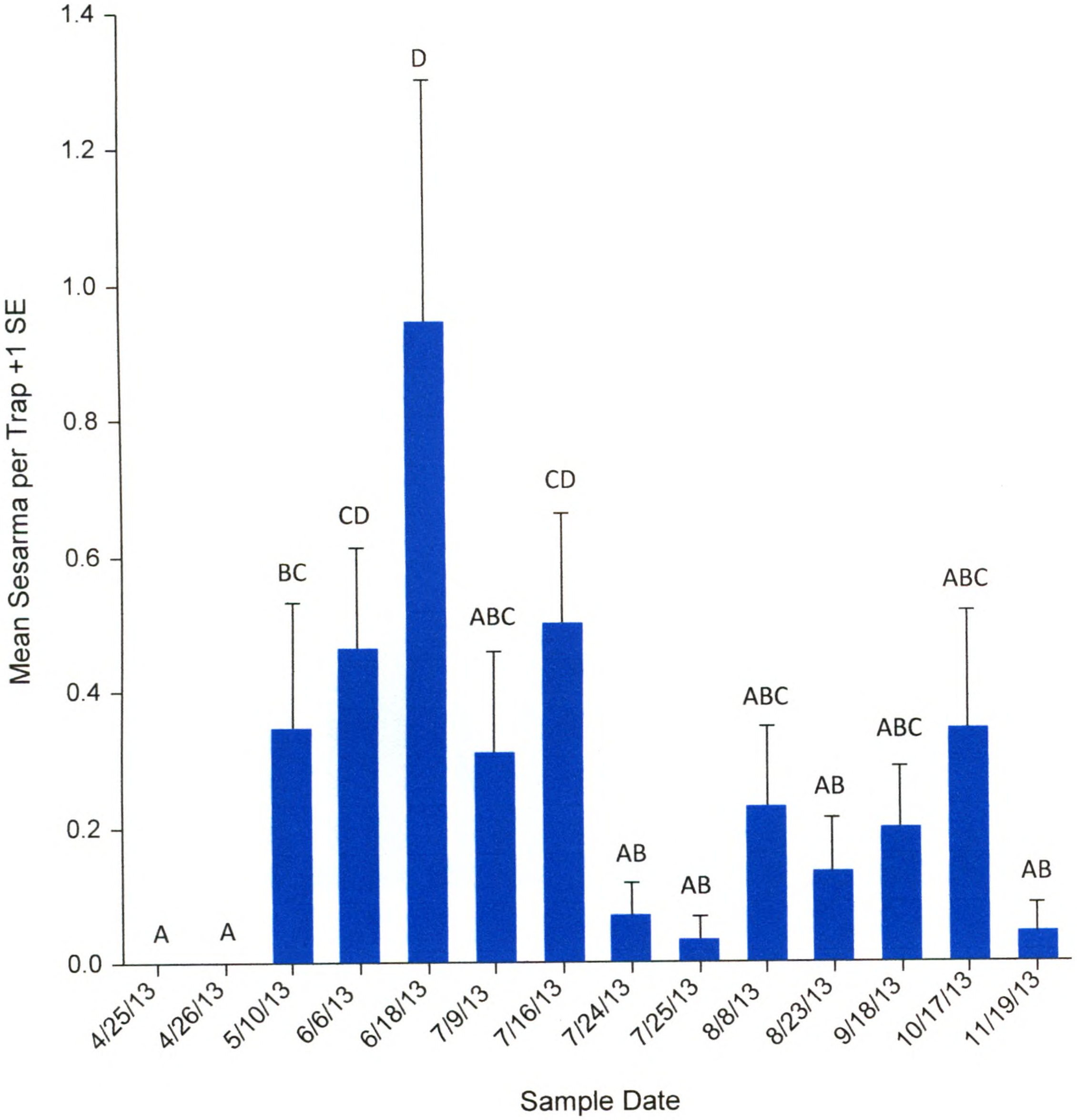

Figure 17. Mean *Sesarma* (+1 SE) per trap caught on each collection date from all creek bank traps at both field study marsh sites. Dates having the same letter are not significantly different based on Dunn's post hoc tests. No bars indicate that zero *Sesarma* were caught.

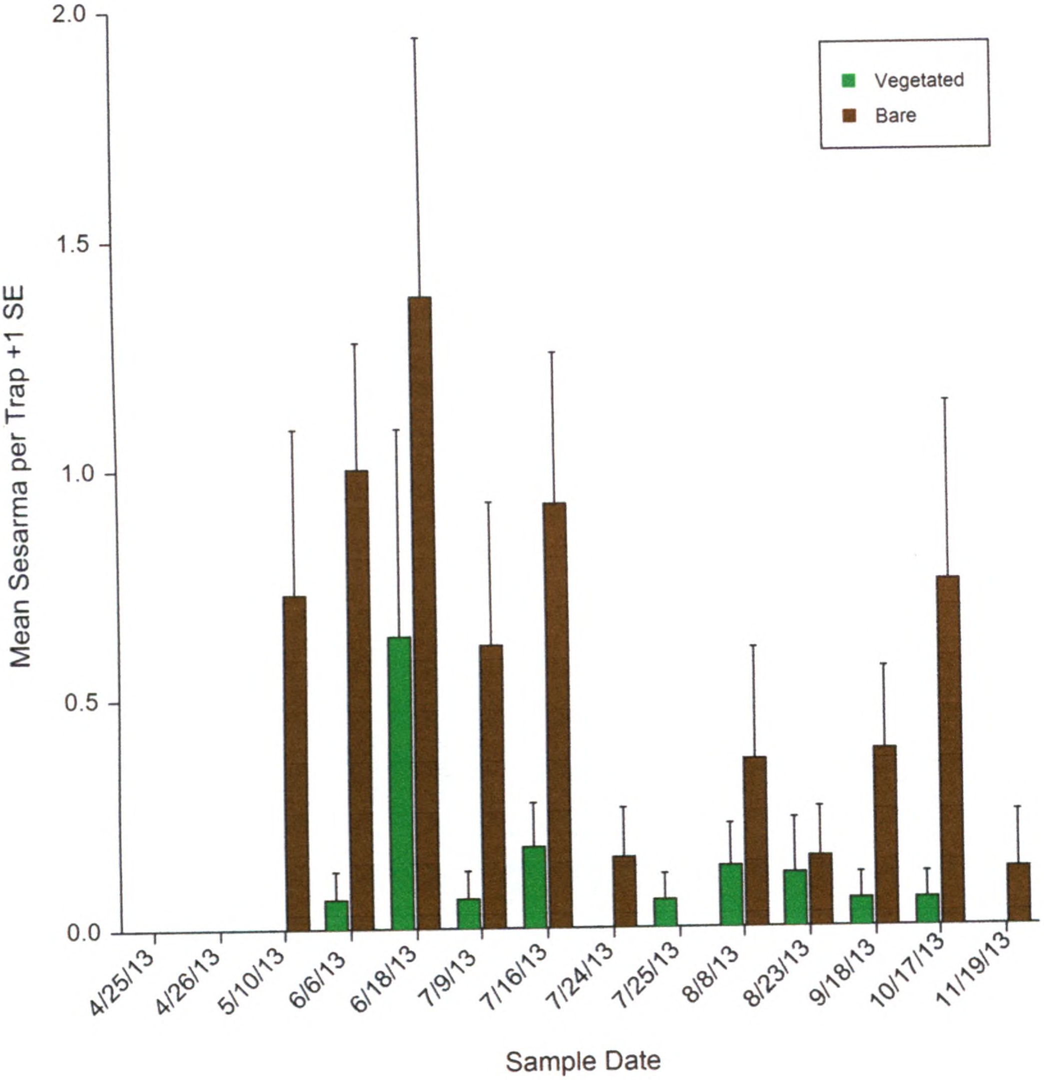

Figure 18. Mean *Sesarma* abundance (+1 SE) by collection date and vegetation characteristics of the trap location. No bars indicate that zero *Sesarma* were caught.

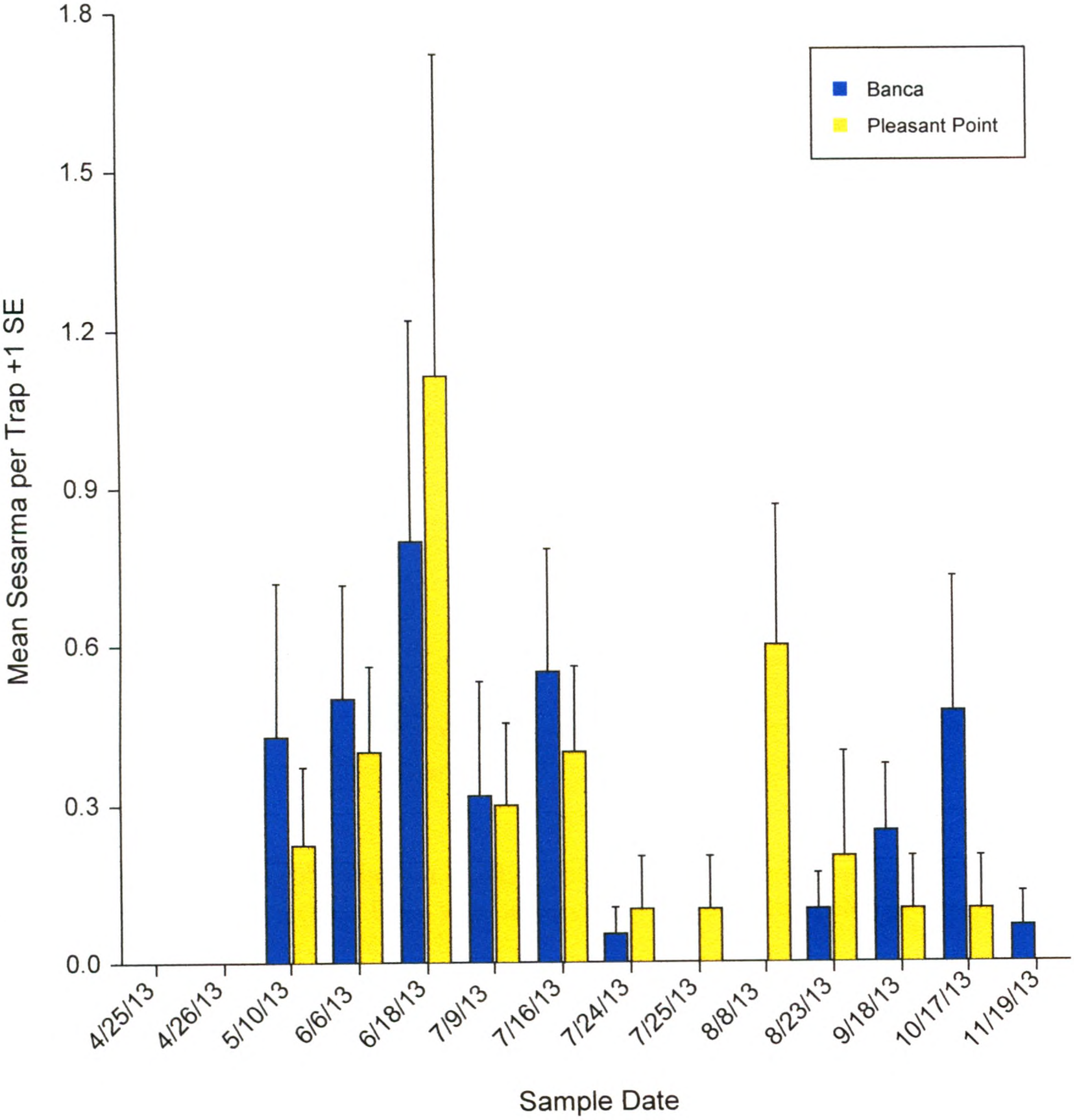

Figure 19. Mean *Sesarma* abundance (+1 SE) by collection date for Banca and Pleasant Point marshes. No error bars indicates that no *Sesarma* were caught.

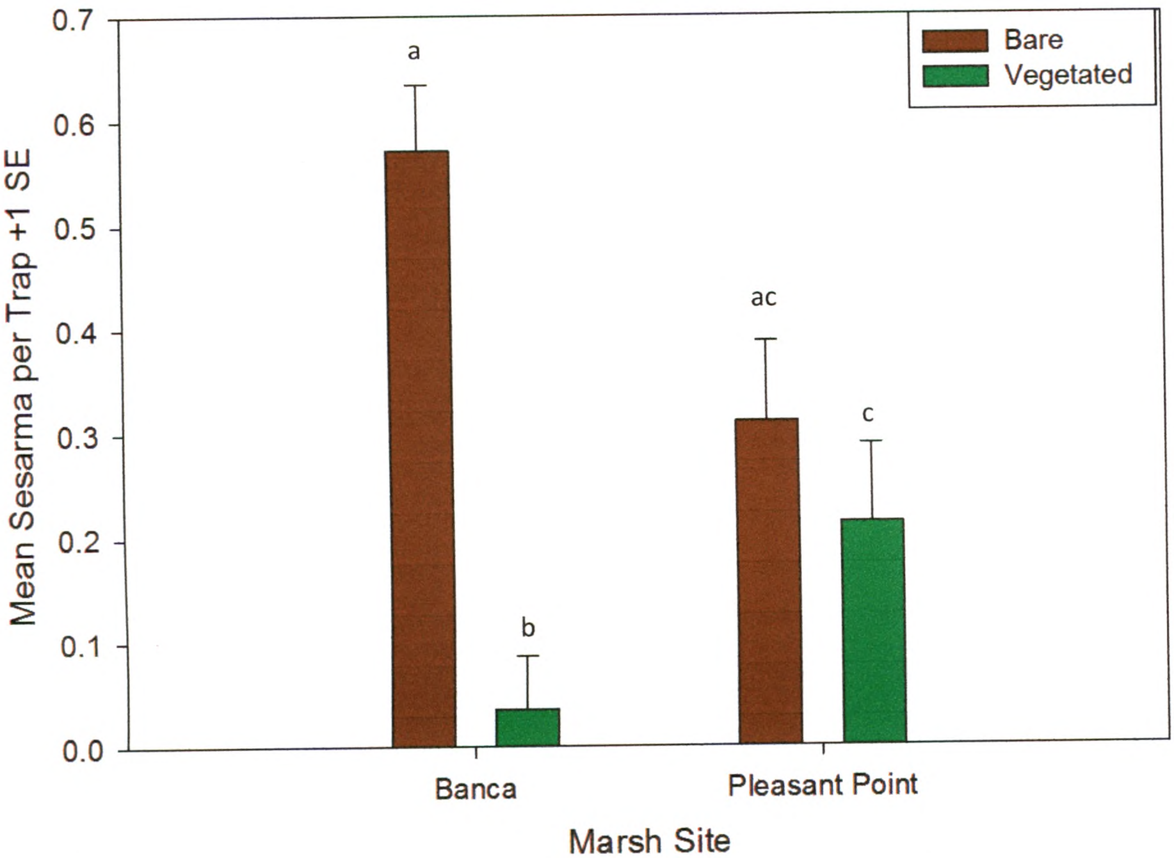

Figure 20. Mean *Sesarma* abundance (+1 SE) over the study period are shown for the Banca and Pleasant Point marshes by creek bank vegetation characteristics (vegetated or bare). Results of the post hoc Dunn's Test are given as letters.

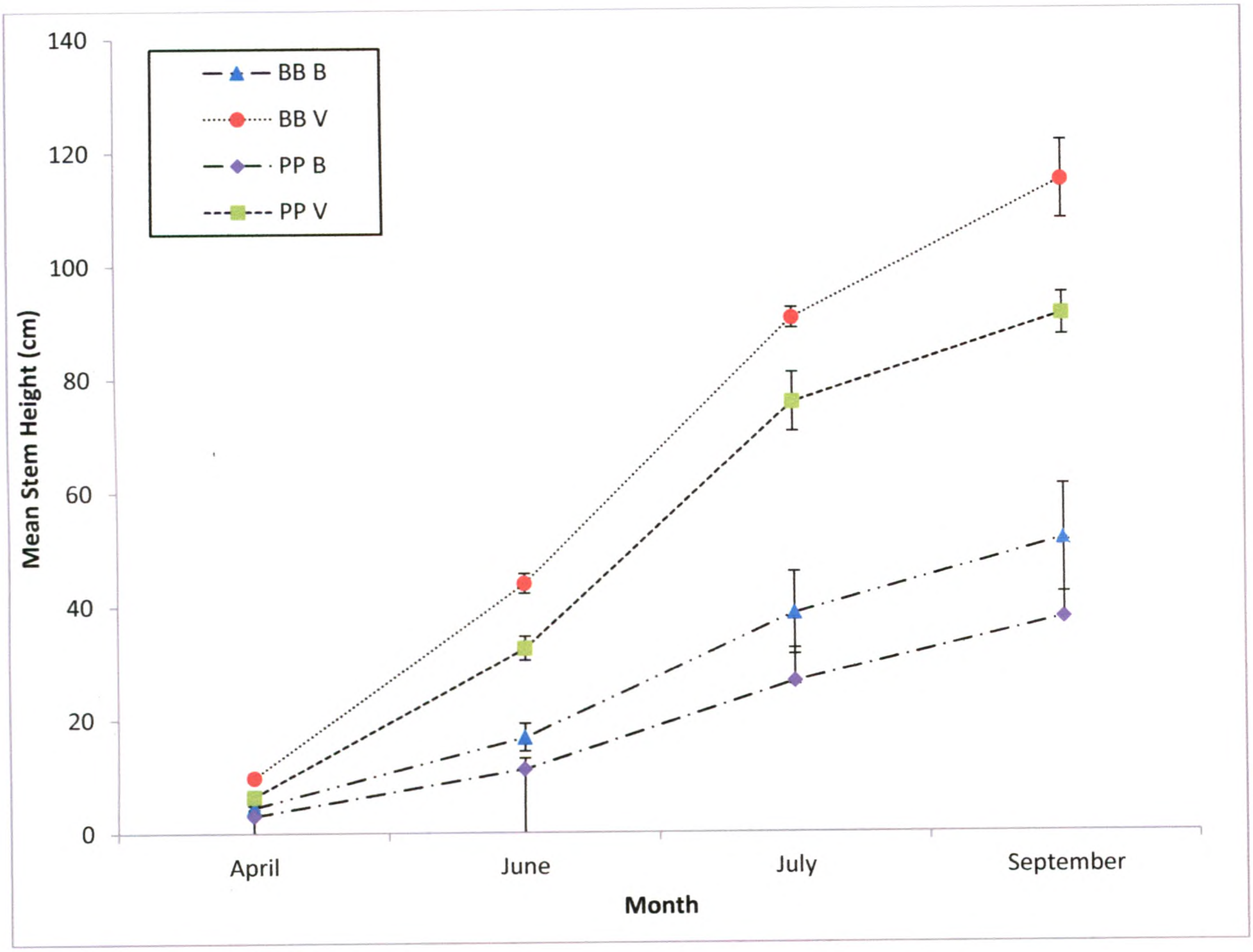

Figure 21. Mean stem *S. alterniflora* height in Banca and Pleasant point marshes in areas that are normally vegetated (BB V= Banca, PP V= Pleasant Point) and areas exhibiting dieback (BB B= Banca, PP B= Pleasant Point) over the growing season.

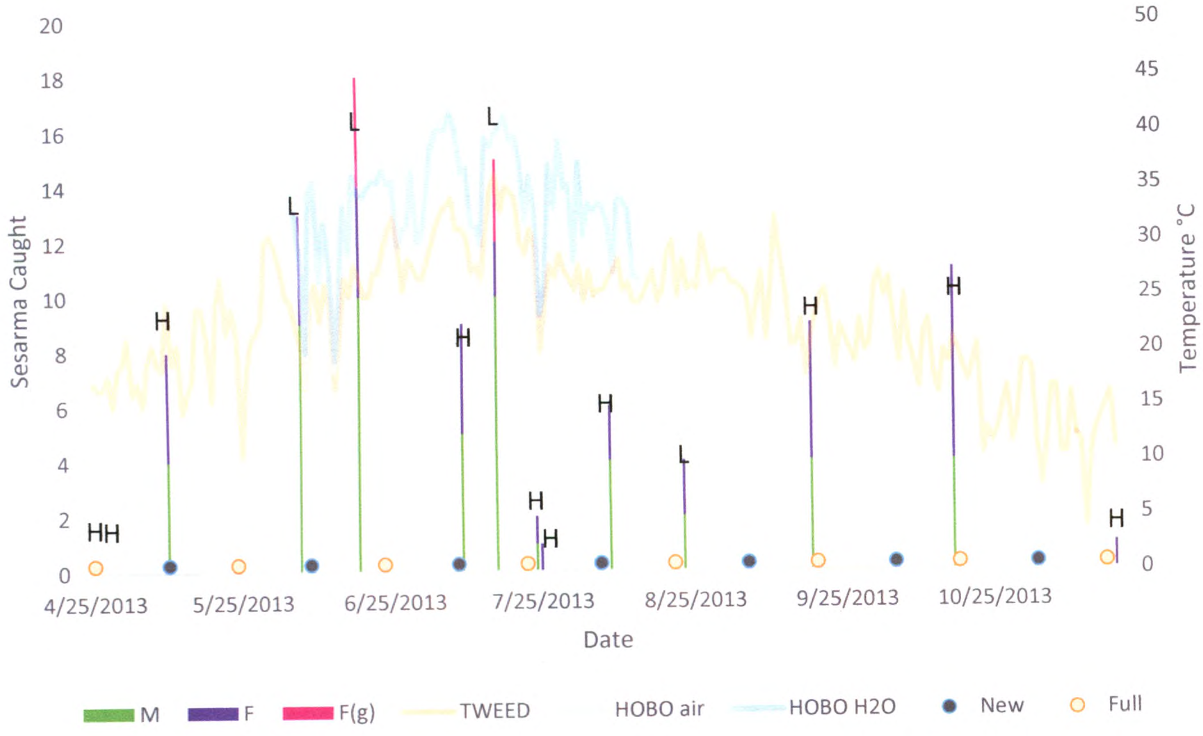

Figure 22. Mixed plot displaying total *Sesarma* caught per observation date by gender. Daily maximum air (Tweed Airport, New Haven, CT), marsh surface (HOBO air) and pond temperatures (HOBO H_20) and full and new moons are shown. Letters indicate whether a deployment was predominantly over a low tide (L) or high tide (H).

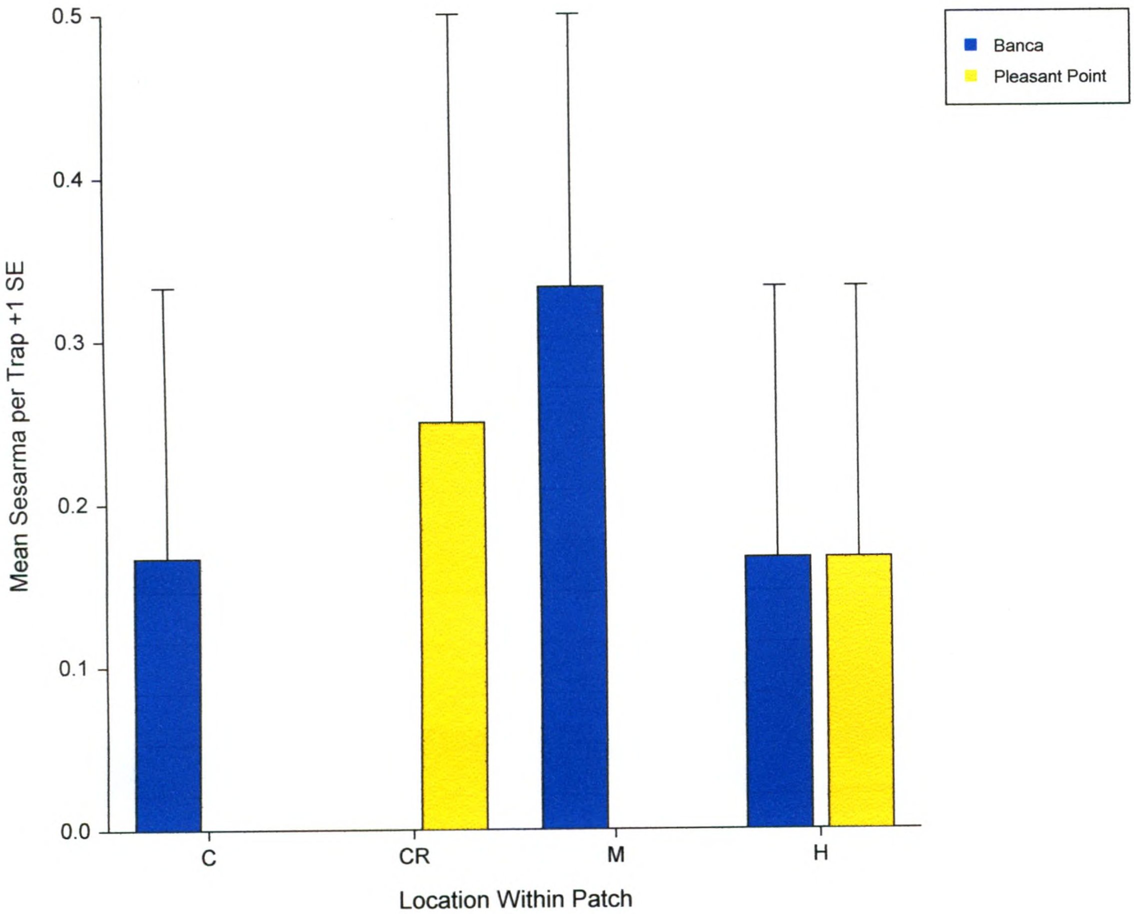

Figure 23. Mean (+1 SE) *Sesarma* abundance per trap by marsh site and location within high marsh bare patch (C=control, CR=closest to creek, H=high marsh furthest from creek, M=middle of bare patch).

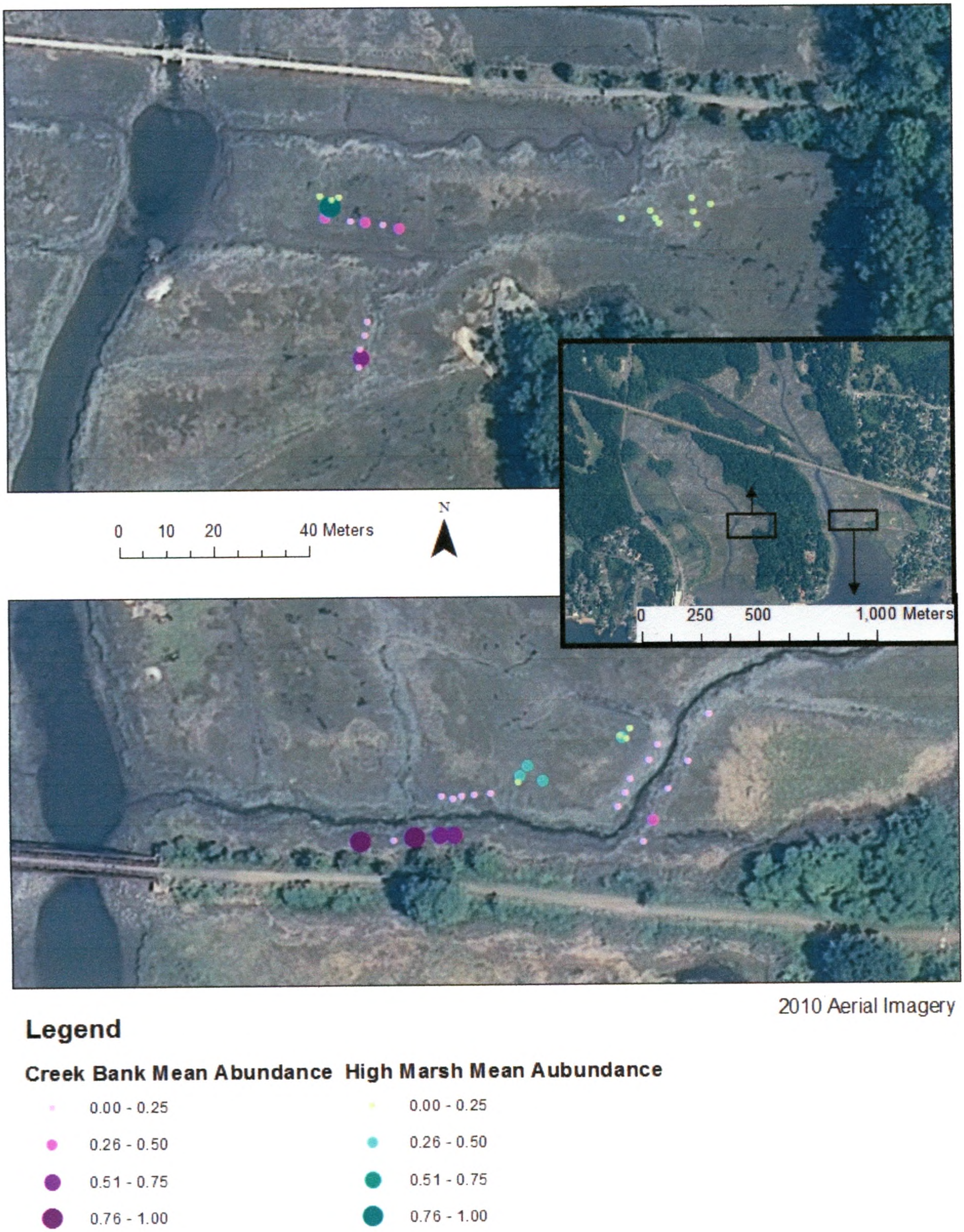

Figure 24. Mean trap abundance of *Sesarma* over all collection periods shown using weighted symbols for each pit fall trap location. Background imagery is from 2010 and was obtained from MAGIC GIS data website managed by the University of Connecticut.

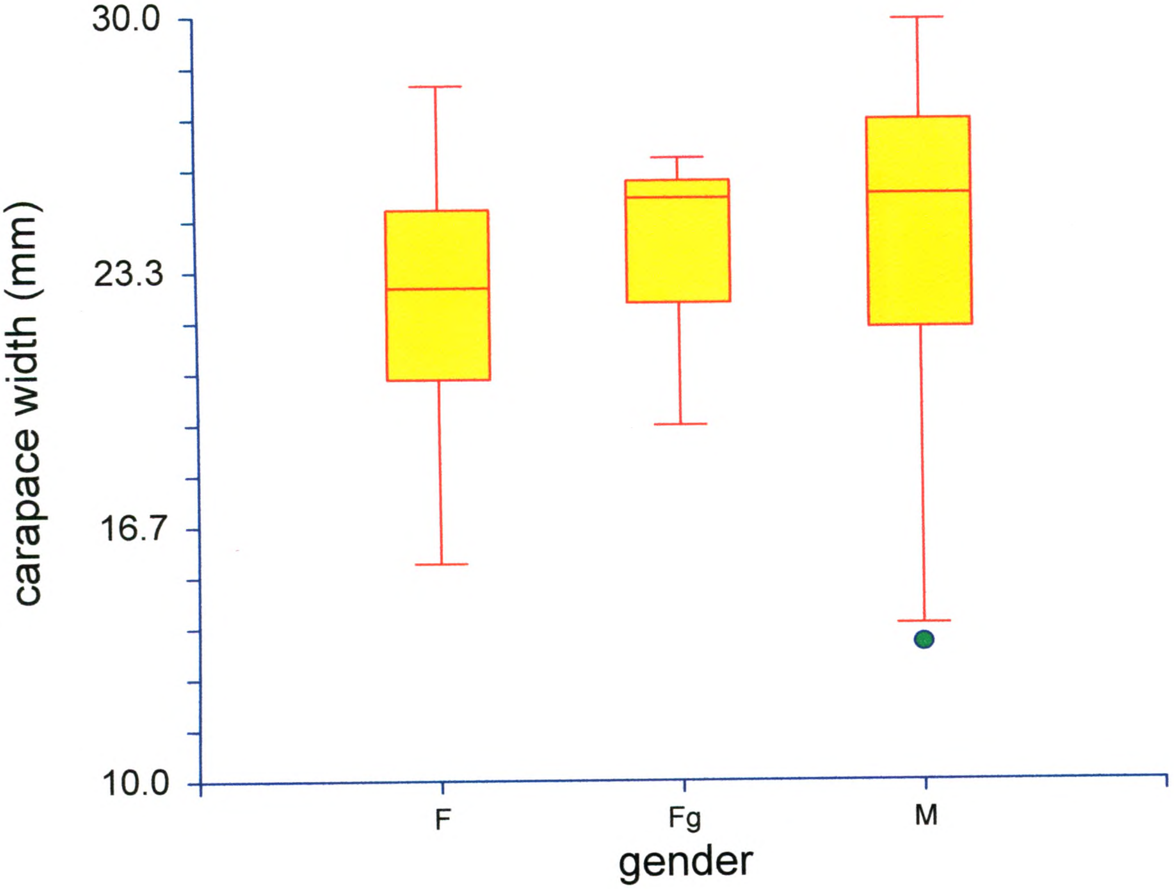

Figure 25. Box plot displaying *Sesarma* carapace width for females (F), gravid females (Fg, not included in F), and Males (M). There was no significant difference in carapace width between the genders (F/Fg/M).

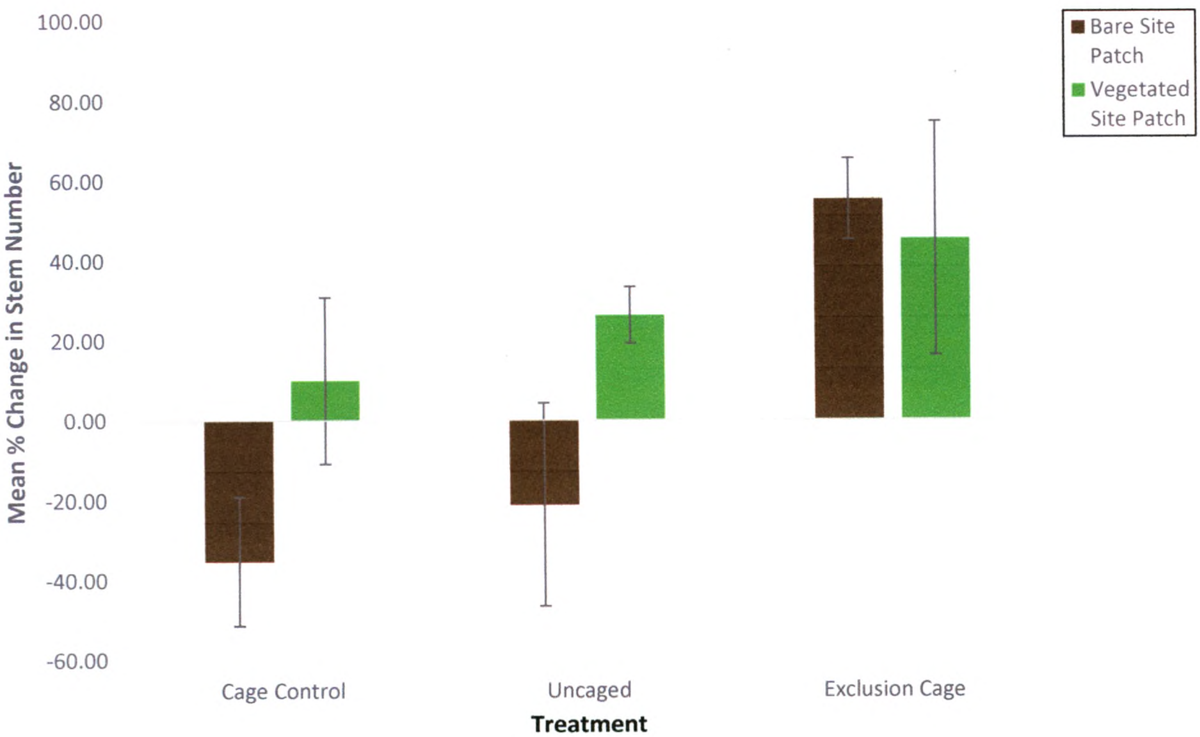

Figure 26. Mean percent change in stem count of transplants (±1SE) by treatment and creek bank vegetation characteristics. Growth was calculated as a deviance in stem count from the original stem count at time of transplant. A net loss in stem count results in a negative mean percentage of growth.

A

B

Figure 27. Remains of *Sesarma* (A) from the predation experiment and (B) general observation while collecting field data in Pleasant Point marsh.

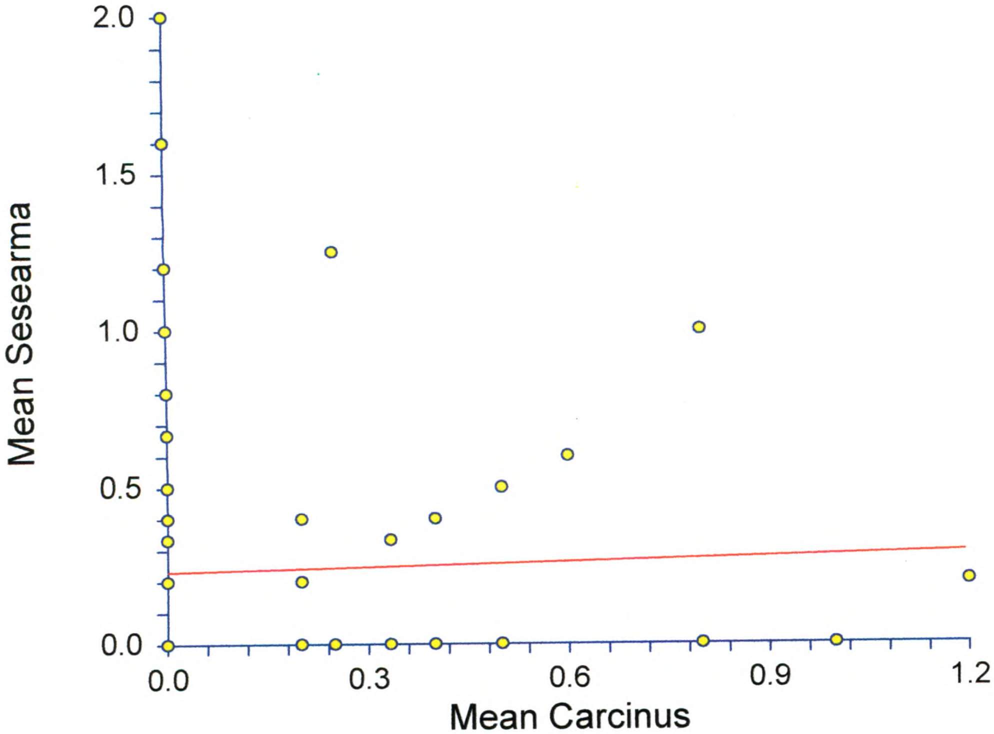

Figure 28. Relationship between *Carcinus* and *Sesarma* abundance (R^2=0.0007).

DISCUSSION

This study consisted of an analysis of: historical salt marsh creek bank dieback; seasonal *Sesarma* abundance and growth of *Spartina alterniflora* in different marsh habitats; and experimental studies of *Sesarma* herbivory and predation pressure on *Sesarma.* The overall objective was to better understand the role *Sesarma* plays in *S. alterniflora* dieback, as well as to obtain more information about its ecology. The foundation for this research was observation of *Sesarma* burrows and *S. alterniflora* dieback within the Connecticut marsh study sites. These observations led to question whether these sites may be experiencing similar phenomenon as that observed in research conducted in the Cape Cod region of Massachusetts that suggests there may be a connection between release of *Sesarma* from predation pressure and *S. alterniflora* dieback.

As there was no existing research on the potential link between *Sesarma* activity and the *S. alterniflora* dieback occurring in the CT marsh sites studied, this study assessed spatial patterns in where dieback was occurring relative to *Sesarma* abundance and geomorphological characteristics of the marsh sites to help identify possible contributing factors of the dieback that should be further assessed. The results of this study do not definitively identify the cause(s) of dieback within the marshes studied. However they do suggest that a) variable patterns of marsh dieback have been occurring for at least the last two decades at the study sites, b) at the time and locations studied, *Sesarma* was present in dieback areas and consuming *S. alterniflora*, and c) while the sample size was low, crabs were being preyed on in un-vegetated creek bank habitats. The results therefore provide insights as to *Sesarma* activity within CT salt marshes and how they may be affecting marsh plant communities and may help identify areas where further study may help increase the understanding of these interactions and their impact on the overall salt marsh structure.

Historical and Spatial Analysis of Low Marsh Dieback

Analysis of historical imagery indicated that some level of *Spartina alterniflora* dieback along creek banks can be observed since the beginning of the study period (1974) and that high levels of dieback started to occur in 1990 in Chaffinch marsh, and 2000 in Pleasant Point and Banca marshes. Dieback trends were not consistent over time, however, within all three marshes once a high level of dieback was observed in a particular marsh, at least one area with a high level of dieback was found through the remainder of the period analyzed. A documented dieback event occurred in creek bank areas at various sites in Connecticut in 1999 (Alber et al., 2008). My analysis supports that this phenomenon likely occurred, to varying degrees, in all three salt marshes studied based on the 2000 imagery which showed an increase in the severity of dieback since 1995.

While there are many factors that can cause a dieback event, such as, sea level rise, drought conditions, salinity, herbivory and pathogens (Alber et al., 2008), it is difficult to determine the cause(s) after the dieback has occurred. This study looked at the spatial patterns of dieback to see if they corresponded with any common factors such as type of creek bank, geomorphology of creek bank, creek bank verses high marsh areas, hydrological proximity of location to tidal influence and whether the location had tidal restrictions.

Dieback itself was not measured, for reasons that will be further discussed later in this chapter. Instead, distance from the creek centerline to vegetation was measured as a potential indicator of dieback, and then the average rate of dieback was calculated for the time periods between the images analyzed. For all three marshes, and all three creek types within each marsh, the distance measured to vegetation was significantly higher in 2012 than in 2004. In some cases 2010 imagery had a greater distance than the 2012 imagery and in other instances it was not

significantly different from the 2004 and/or the 2012 imagery. This finding indicates that there is variability in the influencing processes over time and that a simple consistent trend may not be occurring. While the overall trend appears to lean towards increased dieback, the variability in distances measured in the 2010 imagery relative to 2004 and 2012 is potentially good news for recovery of impacted areas, as some of these differences may be due to revegetation in affected areas.

The distance from creek centerline to vegetation was effective at comparing the same locations within specific marshes between time periods, and whether the overall trends of different areas were similar. This method of comparison would not have been effective at comparing different creek types and the different marshes to one another due to the characteristic differences in creek width among creek types (such as between main channels and mosquito ditches) and the variability in width of creek types across the different marsh sites (particularly in reference to the main channel creek type). Assessing the rate of change essentially eliminates the problem of creek width variability and allows for an increased understanding of whether one of the creek types or marshes is being impacted more than another. This study found no consistent trends in rate of change based on creek type; however, the Chaffinch marsh site as a whole had a significantly higher rate of change then the other two marsh sites. While this finding could be a result of the smaller creek widths of Chaffinch marsh translating into more error in the measurement and calculation of rate of change; it is recommended that field studies be carried out in Chaffinch marsh to understand if this is a shortfall of this study's methodology or a discernable difference that could help increase the understanding of Long Island Sound *S. alterniflora* dieback. Additional support for field studies at the Chaffinch marsh site is that the qualitative analysis of dieback identified high levels of dieback earlier at the Chaffinch marsh

site than the other two marsh sites; suggesting that this phenomenon may have a longer history at this marsh site. This variability among marsh sites, located within less than 10 miles of each other, indicates that some local factor(s) are causing and/or shaping the marsh specific patterns and extent of dieback.

Pleasant Point and Chaffinch marsh sites as a whole exhibited a significant increase in mean change in distance per year during the 2010-2012 time span relative to the 2004-2010 time span; however Banca marsh site showed no significant difference in rate. In assessing why Banca marsh may stand apart from the other two marshes, two possible influencing factors are noted: 1) the main creek channel is much larger in Banca marsh which may have an influence on the influencing factors and 2) Banca marsh had the least amount of transects sampled of the three marshes. In reference to the larger main creek channel, it is possible that this larger open water area may affect the way that the Banca marsh surface is flooded; the channel's ability to hold a larger volume of water may affect the flooding periods of the marsh surface. Relative to transect sampling, the variability in number of transects and the length of the transects could have influenced the final results. It is also noted that there was no significant difference is *Sesarma* abundance between the Pleasant Point and Banca marsh sites, so unless there was a recent change in the *Sesarma* population dynamics within one of these marshes it is unlikely to be a leading cause in the difference in whether dieback was accelerating or not.

The comparison of dieback in Pleasant Point front and back is better assessed by the rate of change than the distance measured to creek bank vegetation as the front of the marsh contains wider creeks than the rear portion. The main and ditch creek types were assessed individually and grouped together, branch creek type was not assessed in this analysis due to the absence of branch creek type assessment in the rear portion of the marsh. During the 2004-2010 time period

the marsh overall and ditch and main creek types individually, exhibited a greater change in rate in the back portion of the marsh compared with the front portion. During the 2010-2012 timeframe the opposite was true for the ditch creek type and the main creek type and the overall analysis indicated no significant difference in the rate of change. These findings don't indicate any identifiable relationship between the location within in the marsh relative to the trail interruption and consequent tidal restriction. This analysis further displays the variability in spatial and temporal dieback, indicating that the influencing factors are not static.

Basic creek bank geomorphological traits: whether the creek section was straight, located on the inside of a curve or the outside of a curve, was assessed for significance in relationship to dieback occurrence. The distance from centerline to vegetation analysis revealed that there was no significant difference in any of the three marshes in 2004. In both Banca and Pleasant Point marshes the straight sections were significantly greater distances than the inside and often outside of curves in both 2010 and 2012. This seems likely due to the fact that straight sections are often much wider in these two marsh systems than areas that exhibit curves. In Chaffinch marsh the results were the opposite in 2010 with the inside and outside of curves exhibiting greater distance than the straight sections. In Chaffinch marsh the channels are more uniform in size which could have been a leading factor in this marsh exhibiting different trends in this assessment. When assessing the rate of change in distance to vegetation, no trends exist relative to creek bank geomorphology. The reasoning behind assessing this information, beyond generally increasing the spatial understanding of dieback, was to assess whether potentially increased velocity of hydrological flows on the outside of a curve may lead to increased erosion and/or the slower velocity of the inside of the curve may lead to more deposition of suspended sediment and may influence suitability for *S. alterniflora* growth. Indirectly these differences

could also affect habitat preference of organisms that graze on *S. alterniflora*. Creek bank geomorphology was not accounted for in the selection of pitfall trap locations in the *Sesarma* population characteristics portion of this study, so it is not known whether *Sesarma* abundance varies based on the geomorphology of the creek bank. Given the lack of significance of geomorphological traits relative to dieback, this assessment of *Sesarma* habitat use would not be a priority for future investigations.

In assessing the influence creek bank heading might have on dieback, once again distance to vegetation does not appear to be the best metric for comparison as the wider creeks often run north/south in all three marshes. As such, the distance measured to east/west creek banks of these north/south creeks were consistently significantly greater than the measurements to banks facing the north and south along east/west flowing creeks. While significant differences in rate of change were identified within marshes for both time periods assessed, the differences did not exhibit any constant trends across the three marshes or within any of the marshes. The rational for assessing creek bank heading was to see whether there might be a relationship with sunlight aspect (unlikely given the relative flatness of the sites) or predominant wind direction and dieback occurrence. Wind direction could influence hydrology and most notably ice buildup which could impact bank stability and suitability for vegetation growth and habitat. Based on the high variability in the results, it seems unlikely that these are leading factors in the dieback that is occurring.

This study affirms that dieback is occurring within the marshes studied and that there is spatial and temporal variability in that dieback. Later in this chapter these patterns relative to *Sesarma* will be discussed. Dieback may be occurring across marshes in a region such as New England due to large scale impacts from climate change such as temperature changes, increased

sea level rise and regional changes in populations of species such as *Sesarma*. This study suggests however, that if that is the case then there is local variability in how these large scale factors are influencing salt marshes. This study also suggests that it may be possible for impacted areas to recover.

The methods applied to assessing historical dieback, while providing an overview of what has happened historically, are extremely limited and do not fully identify the variability that has likely occurred over the full time period from the earliest to latest aerial imagery assessed. The mean distance measured to vegetation methodology is a straight forward measurement taken at a very specific time. Measurements were only taken at three instances in time over a ten year period. Based on the variability observed in those three years it seems likely that there is more variability occurring on a year to year basis that might lend insight as to the influencing factors. The mean change in distance per year methodology consisted of two time spans, one calculated for a 2 year period, the other 6 years. The differences seen between theses time spans indicate that there could have been changes which may not have been identified. During the time period between images many things could happen which would allow vegetation to recover and dieback again and those intermittent events would not be identified. The fact that the results observed are so variable indicates that it is likely more variability occurred during the study period that was not accounted for due to lack of quality imagery available to assess.

Comparing the images that were available has its own challenges. While all imagery assessed were taken at low tide, it can be difficult to determine if an area is exhibiting dieback or whether it is just a shallow portion of the creek channel. This is why the dieback category "possible" was included in the qualitative analysis. This inability to truly identify where the creek bank is located in the imagery assessed is why measurements were made from the creek

centerline rather than the creek bank. One could hypothesize that these areas that are hard to identify as dieback or shallow creek area may be a more advanced stage of creek bank dieback that has resulted in, or perhaps been caused by, erosion of the upper portion of the bank. These areas may be important areas to focus on in future studies to determine if this is the case and what insight that may have on creek bank dieback within a marsh system.

As this is an assessment of vegetation growth, it is important that the imagery used was taken during the growing season to get the best assessment of the vegetation extent. Figure 29 compares 2016 imagery, released after this analysis was conducted, that highlights the importance of the imagery being taken during the growing season. Despite the leaf off imagery being a higher resolution than the leaf on, the leaf on imagery provides a much more crisp demarcation between the vegetation and the rest of the marsh surface. The figure also shows how changing the method in which the imagery bands are displayed can help highlight where the vegetation is, simplifying the analysis. Ideally the imagery used in the assessment would be from the same point within the growing season each year to ensure that early and late season differences are not influencing the results.

While more imagery was available for the study site areas, resolution limitations ruled out the use of some imagery. A 3.5 meter resolution is not sufficient to track trends in vegetation changes along creek banks. In this study the imagery assessed ranged from one to one third of a meter resolution in the georeferenced images. Aside from the resolution, aerial images have other quality limitations such as: minor distortion moving away from the image focal point and the level of post processing the image goes through. Aerial imagery is an expensive product that has many interest parties driving demand of what factors are important for data collection. The importance and use of aerial imagery has increased significantly over recent years as the

technology has advanced and the processing has become fiscally more achievable and timely. The progression in both imagery quality and availability over time will help increase the usefulness of historical imagery assessments moving forward.

An alternative method for dieback assessment may be outlining the vegetation edge along creek banks and tracking changes over time. This type of data collection would allow for increase understanding in where variability is occurring and in what direction. This analysis could be done using aerial imagery or by walking the edge of vegetation to collect the spatial data in the field. Accuracy of collecting data in the field would be dependent upon the unit being used to collect the data, however, this method provides the opportunity to 1) make sure that the post processing is conducted in the same manor across all time periods assessed, 2) control when data is collected within the growing season and 3) may allow for samples to be taken on a more regular basis as aerial imagery is expensive and there is often a couple of years spanning between capture of imagery of sufficient quality for analysis. Using aerials for analysis may allow for the assessment of a larger area since it does not require field work; however, it is limited by the imagery that is available. As the technology advances, high resolution aerial imagery is becoming more available; however it is still very expensive and often takes a while to post process and become available to the end user. Perhaps the use of drones equipped with GPS, altitude control and high resolution cameras may be an alternative way of moving forward with any form of dieback assessment. This method of data collection would allow for collection of imagery on a more regular and consistent basis than aerial imagery that is collected on a large scale. Collection of imagery in the field and post processing in the lab results in a more uniform timeframe for data collection relative to season and tidal stage and would result in less impact to the marsh sites than physically walking the extent of the dieback.

Sesarma Population Characteristics, Activity and Relationship to Vegetation Presence

As may be expected if increased *Sesarma reticulatum* grazing pressure was impacting creek bank vegetation, *Sesarma* abundance was greater along creek banks exhibiting creek bank dieback than creek banks exhibiting typical *S. alterniflora* growth. The higher abundance of *Sesarma* in areas of the marsh exhibiting dieback could be an indicator of a causative relationship. The vegetation growth measured at both Banca and Pleasant Point marsh did support that, in addition to growing more densely, vegetation grew significantly taller at vegetated sites than bare sites. It is interesting to note that when analysis was conducted based on grouping the vegetation characteristics with the marsh site, the Pleasant Point bare and vegetated sites did not exhibit a significant difference in *Sesarma* abundance. The Banca marsh bare site was not significantly different from the Pleasant Point bare site, however, the Banca marsh vegetated sites had a significantly lower abundance than the Banca bares sites and both the Pleasant Point vegetated and bare sites. This could indicate that vegetation characteristics may have more of a relationship with *Sesarma* abundance in Banca marsh than Pleasant Point marsh.

While there were no significant differences in the growth of *S. alterniflora* at vegetated or bare sites respectively from one marsh to the other, it is noted that vegetation at Banca marsh was taller than the vegetation sampled at Pleasant Point marsh (Fig. 21). This difference in vegetation height could be a result of many factors, such as: tidal influence, relative elevation within the marsh or a result of a small sample size. If *S. alterniflora* growth characteristics alone controlled *Sesarma* abundance, one may have expected there to have been more *Sesarma* caught on average at the Pleasant Point bare sites than the Banca marsh bare sites based on the measured growth characteristics. Alternatively, the greater difference in vegetation characteristics in Banca

marsh may have driven a greater difference in the distribution of *Sesarma* within that marsh, resulting in more *Sesarma* in the less vegetated areas.

Sesarma were first collected on the 5/10/13 sample date and continued to be caught through the end of the sampling on 11/19/13. Peak abundance was collected on 6/18 with the collections on 6/6 and 7/26 not being statistically different. It seems likely that these samples span the period in which *Sesarma* are most active in the marshes that were studied. The lower abundance collected on 7/9, which occurred within this window of time could be for a number of reasons. It is noted that the 7/9 sample occurred during a predominantly high tide period and that the sample occurred one day after the new moon. A closer look at the environmental data that was collected shows an interesting relationship between abundance and tidal and lunar phases. A significantly higher abundance of *Sesarma* were observed during predominantly low tide deployments compared to predominantly high tide deployments. Similarly, deployments that were more than three days away from a full or new moon also had a significantly higher abundance of *Sesarma* than within three days of a full or new moon. The fact that a similar trend was observed between high tides and spring tides relative to *Sesarma* abundance could either indicate that 1) *Sesarma* are less active during periods of marsh flooding or 2) that during these periods crabs were able to escape the traps due to elevated water levels providing a means of escape. The only two sample dates that coincided with predominately low tides and that were greater than three days away from a full or new moon were indeed the two samples where the highest *Sesarma* abundance was observed.

Temperature data was reviewed relative to the sampling period, but no meaningful connections could be identified relative to *Sesarma* activity. The first two sampling collections, both in April, yielded no *Sesarma* trapped. Perhaps it was too early in the season for *Sesarma* to

be active. The max air temperatures during these deployments were 17.2 and 16.7°C. It seems unlikely that low temperature alone has a strong impact on *Sesarma* activity as the November 19th deployment yielded a catch of one *Sesarma* despite the max air temperature only reaching 11.1°C. This indicates that the temperatures observed in April were probably not a restricting factor in *Sesarma* activity, but potentially rather the persistent cooler temperatures that would have occurred leading up to that time along with the colder water temperatures. It also seems likely that as *Sesarma* are herbivores, they would have reduced activity until vegetation starts to become productive with the start of the growing season. While temperatures may dip down lower in the early fall, the warmer waters and increased availability of a food source would likely provide a better tradeoff for activity at the close of the season than the onset.

Crab size did not differ significantly over the study period or across any of the variables assessed. This finding suggests that there is a typical population that is uniform across the studied areas that *Sesarma* inhabit and that there is no significant variability in the size of the *Sesarma* population over the study period. It is noted that only 18 of the 89 *Sesarma* collected were under 20mm in size. This indicates that the younger portion of the population was not well represented by the sample methods used; as such it would be difficult to identify how the presence of different age groups of *Sesarma* may vary within the marsh both spatially and temporally.

When comparing the high marsh die off areas with the creek bank areas, on average more *Sesarma* were caught along creek banks than in the high marsh; however, this difference was not significant. One possible explanation for this is that the high marsh traps were primarily deployed in bare areas with only the control trap residing in areas with typical vegetation growth. As such it is possible that only the more active areas in the high marsh were being sampled

compared with a more comprehensive sampling of creek bank areas. Traps along the creek banks were fairly evenly distributed between vegetated and non-vegetated areas: 43% of traps successfully sampled were in vegetated areas, the remaining 57% were in bare areas. Of the high marsh traps that were successfully sampled, only 21% were located in vegetated areas, 45% were on the edge of a bare patch and the remaining 34% were located in the center of a bare patch. Deploying more traps in vegetated portions of the high marsh could increase understanding as to whether *Sesarma* are more active in the high marsh dieback areas than in other high marsh areas and how these trends relate to creek bank areas.

***Sesarma* Herbivory**

The transplant study assessed overall impacts, both negative and positive, to the count of healthy stems of S. alterniflora in the culm from start to end of the experiment. This study indicated that there was no significant difference in the change of a culm's healthy stem count over the study period based on whether the site was vegetated or bare. It is noted that none of the observations of impact to the culms at vegetated sites could positively be identified as a result of grazing, while at bare sites this distinction could be identified at times. The fact that the grazing observations that were made correlated with the areas of higher Sesarma abundance suggests that further investigation into grazing impacts could be warranted. Culms were checked once a week for impacts, this meant there was time for evidence of the cause of impact to deteriorate between observations. Perhaps a more robust observation schedule would provide for a better understanding of what is influencing the vegetation stem health. Another potential shortcoming of this study could have been an insufficient understanding of what grazing impacts specific to Sesarma look like. It is also noted that impacts to above ground stem health could potentially be driven by underground burrowing and/or grazing activities that would have gone unobserved.

The fact that the greatest mean increase in S. alterniflora stem counts were observed in the exclusion treatment located in bare areas (Fig. 26) indicates that the plant is capable of growing in the bare areas despite its lack of presence. That the cage control and uncaged treatment did not fare well along bare creek bank areas further suggests that access to the vegetation influences its survivability rather than factors such as inundation, salinity or soil microbes which would not have been restricted by the exclusion measures. Since there was a significant difference in the mean change in stem count between the control treatment and the exclusion treatment, it can be concluded that the cage structure itself does not influence the culm's health; either as a support aid or a deterrence. A possible explanation for the higher mean increase in stem count in exclusion cages located on bare creek banks could be that the vegetation did not have to compete with other vegetation for resources to aid in growth. One could hypothesize based on the above observations that if negative factors are excluded in bare creek bank areas vegetation could be able to recover, at least initially.

The greatest negative change in stem count was observed in the transplant grouping located closest to the marsh main creek channel in both Banca and Pleasant Point marsh. This trend corresponds with the general spatial trend of where higher Sesarma abundance was observed. This could indicate an increase herbivory opportunity that would result in greater impact on S. alterniflora. Alternatively, another possible influence could be more tidal flooding that would likely occur closer to the marsh main creek, which would result in increased hydrological impacts and/or salinity stress on the transplants. As previously pointed out however, the caged transplants fared well in these locations, so it seems likely that the impacting force is something that requires access to the vegetation that was excluded by the cage.

Another possible influence on transplant stem health could be the initial health of the transplant and the changes in conditions it was subject to once moved to a different location within the marsh. All transplants were relocated within the same marsh from a nearby vegetated creek bank area to potentially limit stress and variability of conditions at the transplanted site from the plug's original location. While these measures were taken to reduce the variability in environmental conditions, as the influences on *S. alterniflora* growth are dynamic and the underlying cause of the dieback is unknown, it is hard to say just how much factors that were unaccounted for, such as soil biome or minor changes in relative elevation, may have influenced *S. alterniflora* growth.

Predation on *Sesarma*

The predation experiment was limited by *Sesarma* availability; as such only general observations could be made. Predation was noted within Pleasant Point marsh through observation of remains and Banca marsh through the field predation experiment; indicating that *Sesarma* have not been fully released from predation pressure in these salt marshes. The observed predation pressure could explain why *Sesarma* abundance and grazing impact appears to be lower than that observed on the cape by Holdredge et al. (2009) and explained by Altieri et al. (2012) to be caused by over fishing by recreational fishers removing *Sesarma* predators. The protected caged crab that died from stress was the one located in a transitional area between a vegetated and bare area. This could be an indicator that vegetated areas do not offer additional protections and/or that perhaps the lower abundance of *Sesarma* found in these areas correlates with less favorable conditions which resulted in elevated stress for the individual placed there.

It is noted that the three exposed crabs located closest to the main creek were the ones that were gone from their tethers. This could suggest that predation pressure is from a source

within the tidal waters and that the potentially increased flooding of this area of the marsh may have made these individuals easier targets. These individuals however, were also the ones that were located closest to the trail which may have made access by terrestrial predators easier. It is interesting that this area where the *Sesarma* seemed to be most influenced by predation pressure corresponds with where *Seasarma* were found in higher abundance and *S. alterniflora* was visibly impacted.

Perhaps replicating this experiment with the use of a waterproof video camera could be useful in identifying what predator consumed the crabs. Knowledge of the predators would help shape the cascade relationship that could be occurring down to the vegetation and overall salt marsh structure level. Further investigation is necessary to look at the significance of these possible spatial patterns and whether the overlap of predators, grazers and grazing impact may help balance the impacts that may occur at the site. Future questions would include whether *Sesarma* will spread to other areas of the marsh to avoid predation pressure and if so what impact they will have on the vegetation. Despite the small dataset, it can be concluded that there is predation occurring within the marsh systems and that while this pressure is not preventing all grazing, it may be helping to prevent catastrophic grazing impacts within the marsh sites.

Relationship between *Carcinus* and *Sesarma* Abundance

The literature cites *Carcinus maenas* as a predator of *Sesarma* (Coverdale et al., 2013c) so the abundance of the two species were assessed for relationships within the study sites. This study found no direct relationship between the abundance the two crab species within the marsh sites assessed. Based on this evidence, it seems unlikely that *Carcinus* is significantly impacting *Sesarma* activity and consequently aiding in vegetation recovery like Bertness and Coverdale (2013) and Coverdale et al. (2013) observed in the Cape Cod region of Massachusetts. Coverdale

et al. (2013) noted that the largest *Carcinus* populations were at bare sites with *Sesarma* burrows. In this study when assessing *Carcinus* abundance relative to vegetation characteristics, there was no relationship between *Carcinus* abundance and whether the site was vegetated or not. It is noted however, that the highest abundance of *Carcinus* in this study was observed in a bare site area with *Sesarma* burrows, and that the area did not exhibit vegetation recovery over the study period. This study also found that predation pressure is present in bare areas where *Sesarma* abundance is relatively high compared to other areas of the marsh. Perhaps these findings suggest that further investigation is warranted to better understand the relationships with potential predator species. It could be that the traps were not large enough to capture some *Carcinus* individuals. It was found that there was a significantly higher abundance of *Carcinus* in Pleasant Point marsh than Banca marsh; this may indicate that there is a higher predation pressure within Pleasant Point which may account for the lack of difference in *Sesarma* abundance based on vegetation characteristics at that marsh site.

Suggestions for Future Studies

This study assessed *Sesarma* abundance and relationship to above ground vegetation characteristics and possible impacts to *S. alterniflora*. It was found that *Sesarma* are more abundant on creek banks exhibiting dieback than creek banks exhibiting normal *S. alterniflora* growth and the transplant experiment suggests that there is *Sesarma* grazing pressure on *S. alterniflora,* however, the low level of observations of what was believed to be *Sesarma* herbivory among the transplants indicates that either there was a flaw in the study methods or this is not the only factor, or possibly not the most significant factor, in *S. alterniflora* dieback. Improvements to the study methodology could include 1) more routine inspection to improve

ability to identify grazing evidence, placing more transplants, 2) planning their spatial location within the marsh to compare with other possible influencing factors.

Below ground growth is often forgotten about as it is not observed from the marsh surface. The intense burrowing network *Sesarma* create and the high nutritional value below ground growth offers demands that further research is done on this type of impact to plant health. Belowground impacts provide less opportunity for recovery than above ground impacts and increase the soil disturbance and structure impacts to the marsh. This study did not assess below ground impacts, however it is suggested that information on burrowing extent of *Sesarma* relative to vegetation abundance be analyzed in future studies. This could be done through assessing burrow entry density in areas exhibiting typical *S. alterniflora* growth and areas exhibiting dieback. To get a better understanding of the below ground disturbance associated with various levels of burrow entrance densities, burrow castings could be conducted to approximate what the below ground disturbance is based on the aboveground evidence.

It was noticed was that *Sesarma* abundance was greater during predominantly low tide periods not near a spring tide. Whether this is an indication of reduced activity during high and spring tides or a limitation of the sampling method caused by excessive flooding could be helpful in understanding the ecology of the species. Interestingly, when looking at the observed abundance by spatial location of traps (Fig. 24), it appears that more *Sesarma* were caught closer to the main channel of the marsh; areas that would flood soonest and drain latest with each tide. Level of flooding within the traps was not noted during the span of deployment periods. It was noted during assessment however that at times traps were fully flooded, even with the surrounding marsh area had drained off despite the drainage holes located in the bottom of the trap. Further assessment as to dynamics of flooding duration, *Sesarma* abundance and *S.*

alterniflora growth could provide useful information on understanding changes to creek bank areas. Excess flooding could provide exposure to additional factors that could impact *Sesarma* activity and/or *S. alterniflora* health, such as salinity and access by aquatic consumers.

The assessment of high marsh bare patch areas revealed that *Sesarma* are active in these areas. This analysis was a first step to see if further high marsh study was warranted; the fact that *Sesarma* abundance was not different from creek bank abundance certain suggests that further investigation would be warranted. Most of the high marsh area samples were either in or adjacent to bare areas. A more balanced high marsh assessment of *Sesarma* abundance would help establish whether there is any significant difference in high marsh abundance of *Sesarma* and creek bank abundance, or establish whether the trend noted in this study is consistent across high marsh areas. In addition to increasing the replicates and spatial comprehensiveness of the high marsh extent studied, assessing the relative elevation and flooding of both the high marsh and creek bank areas sampled for relationships with abundance could help explain spatial utilization of the marsh sites and may lead to identification of other factors that have an influence on *Sesarma* ecology and/or *S. alterniflora* dieback.

Conclusions

Sesarma are present and active members of the Banca and Pleasant Point marsh communities. They have extensive burrowing networks and have demonstrated grazing impacts on *Spartina alterniflora*. Personal observations and field experiment has proven that predation pressure on *Sesarma* is present in these marsh systems. No hard conclusions can be drawn as a result of this study as to what impact *Sesarma* may be having on the salt marsh structure in Pleasant Point and Banca marsh; however, the possibility that *Sesarma* are having an impact on vegetation dieback and marsh structure cannot be ruled out. Further investigation into

belowground disturbance and local flooding influences on *Sesarma* populations and *S. alterniflora* are encouraged to increase the understanding of *Sesarma* ecology, *S. alterniflora* dieback occurrence and the possible link between the two.

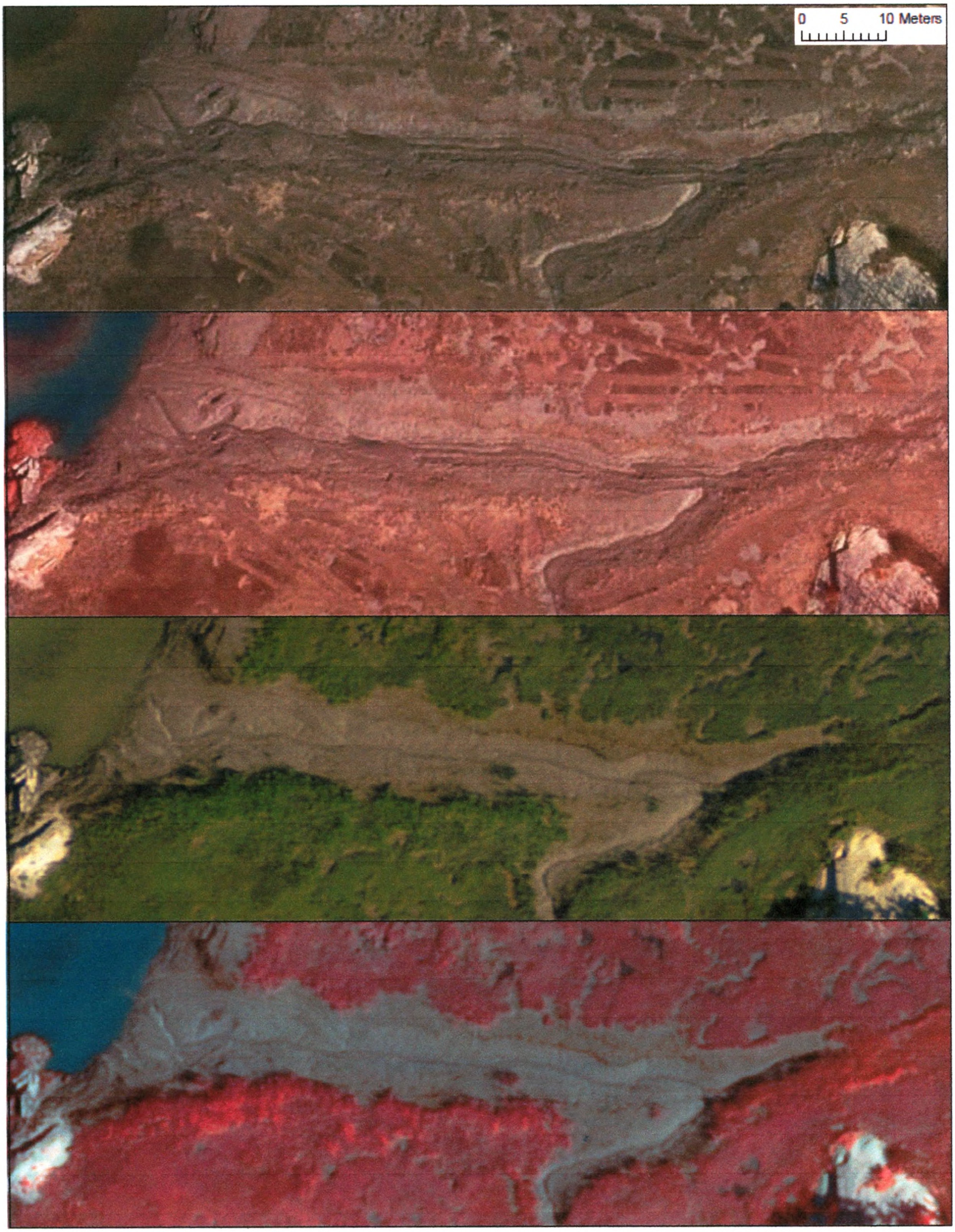

Figure 29. Comparison of 2016 imagery available post analysis using different display methods. From top to bottom: 0.25ft leaf off imagery, default display; 0.25ft leaf off imagery, 4, 1, 2 band display; 0.5ft leaf on imagery, default display; 0.5ft leaf on imagery, 4, 1, 2 band display.

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