Effects of Hurricane Sandy on Great South Bay, Long Island: Assessing water quality, seagrass and associated nekton communities

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Abstract of the Thesis

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On October 29, 2012 Hurricane Sandy hit the south shore of Long Island (NY) causing a major breach on Fire Island. The new influx of ocean water has changed various physical and ecological aspects of Great South Bay (GSB), particularly those areas near the breach. In this study, we measure several environmental variables (salinity, temperature, dissolved oxygen, turbidity, light attenuation, and chlorophyll *a*) that heavily influence the two seagrass species found in GSB, *Zostera marina* and *Ruppia maritima*. Through seagrass and nekton monitoring programs, we quantified the impact of Hurricane Sandy on the flora within the estuarine lagoon, as well as examined temporal and spatial differences in seagrass-associated fauna as a function of a major disturbance event and the proximity to a new oceanic inlet. The most evident water column changes since 2012 have been a drastic bay-wide increase in salinity and a decline in water temperature surrounding the breach area. Dissolved oxygen appears to remain stable after the breach while turbidity, light attenuation, and chlorophyll *a* varied highly interannually. Loss of *Z. marina* is evident bay-wide as a direct result of the breach, and data suggests the slow growing species remains at very low percent cover, only able to recover at selective sites across the bay. *Ruppia maritima*, on the other hand, was not directly impacted by Hurricane Sandy. Although a drastic decline in 2014 was observed, *R. maritima* has been able to successfully recover. Fauna abundances were significantly higher in 2016 (four years post-Sandy) as compared to 2013, immediately following the breach. Seagrass associated species, such as the pipefish (*Syngnathus fuscus*), increased in abundance following an increase in seagrass near the breach. Throw trap data suggest that predators presumably moving in and out of the new inlet are impacting the abundance of juvenile blue crabs (*Callinectes sapidus*) at the sampling site near the inlet. Further monitoring of water quality parameters, flora, and fauna is critical in determining how GSB will ultimately respond to this new influx of ocean water. The value of seagrass as a foundation species is very clear, and effort toward restoration and conservation, while difficult, are of utmost importance.

Dedication Page

To my parents Maria Luisa and Bill, my brothers Guillermo and Adrian, and my partner Lindsey, who's endless love and support make anything possible

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Chapter 1

INTRODUCTION

Coastal barrier island systems provide extensive economic and environmental functions. They protect the coastline from erosion and reduce wave damage by absorbing the energy associated with waves and storm surge (Dolan & Godfrey 1973; Stone & McBride 1998). Barrier islands provide habitat worldwide for various species of wildlife including birds and invertebrates, while allowing estuarine conditions to persist in coastal lagoons behind the barrier island that sustain many environmentally and economically important marine species, including seagrass beds.

Seagrass beds are recognized worldwide for their extensive ecosystem services. These ‘ecosystem engineers’ stabilize sediments, sequester nutrients, provide above and belowground habitat for a multitude of taxa, and fuel intense secondary productivity, among other services (Heck et al. 2008; Hemminga & Duarte 2000; Jones et al. 1994; Kitting et al. 1984). Unfortunately, the decline of seagrass worldwide is also well recognized. It is estimated that since the 1990’s seagrass has been lost at an average rate of 7% per year (Waycott et al. 2009). Seagrass is especially vulnerable in highly developed estuaries and coastlines due to added anthropogenic stressors such as eutrophication caused by nutrient loading and the urbanization of surrounding watersheds (Burkholder et al. 2007; Duarte 1995; Kemp et al. 2005). Human activities such as aquaculture, dredging, and boating might also interact with nutrient enrichment and algal overgrowth to exacerbate seagrass decline (Burkholder et al. 2007; Orth et al. 2006). In the Gulf of Mexico and the Atlantic coast of the United States, many seagrass beds found within the back bay region of coastal barrier islands such as Barnegat Bay (NJ), Pamlico Sound (NC), and Great South Bay (NY) are at direct threat from these anthropogenic influences (Kennish et al. 2007). The threats to these estuarine ecosystems may be exacerbated due to decreased tidal flushing, increased water temperatures, and reduced light penetration through the water column (Walker et al. 2006).

Due to the fragile estuarine nature of many back bay systems, tidal inlets exert a large control over various physical and subsequently biological characteristics of the ecosystem (Reid 1957). Therefore, the formation of a breach, or a channel created from a bisected barrier island due to strong currents and storm surge, is of major interest. Barrier islands are dynamic systems, constantly affected by wind, waves, tides, currents, and storms. For example, the south shore of Long Island has been prone to the creation of various inlets, with over 85% of the area historically affected by inlet activity (Leatherman 1985). Through geomorphic analysis, Leatherman and Allen

(1985) found evidence of breaching reoccurring at similar locations along the barrier island. These natural events not only alter direct island habitats and water quality parameters (Conley 1999), but indirectly affect the organisms living there. Whether this effect is beneficial or detrimental is typically unpredictable and is highly variable depending on the resource. For example, following the creation of Moriches Inlet (NY), Glancy (1956) observed increases in predators, leading to a decline in oyster sets within Moriches Bay, while at the same time observing an increase in flushing rates and improved water quality. Flora and fauna impacted by breaches must be monitored closely to determine their ultimate impact on ecosystem function.

The landfall of Hurricane Sandy on Long Island, NY (October 2012) led to the creation of three new inlets through Fire Island. Two of these breaches were immediately closed by the Army Corp of Engineers, following a mandatory Breach Contingency Plan (US Army Corps of Engineers 1996). However, the third inlet was located on federal land, within the Fire Island National Seashore wilderness area and remains open and subject to natural processes (Fig. 2). The breach occurred at a location historically known to have been an inlet site until 1857 (Leatherman & Joneja 1980), named Old Inlet. This breach has been very dynamic, migrating west (~200 m**)** since its creation over four years ago. The cross-sectional area of the inlet has remained relatively stable at about 400 square meters and six meters deep, despite occasional fluctuations in 2014 (Flagg et al. 2015). Tidal forcing models have estimated that flushing rates in east Great South Bay (GSB) have declined from 25 to 10 days (Gurdon et al. 2015). This change in residence time can have a dramatic impact on the natural resources within GSB. This study presents a natural experiment in which the formation of a barrier island breach creates an avenue for significant increases in tidal flushing, potentially alleviating seagrass beds from anthropogenic stressors and allowing these foundation species to respond to changing water quality characteristics.

Coastal New York seagrasses have undergone steady declines over the past 80 years. *Zostera marina* Linnaeus once dominated the benthos of the Long Island South Shore Estuarine system, in which GSB is found (Cosper et al. 1987; Dennison et al. 1989). It was estimated that from 1967 to 1977 *Z. marina* coverage declined by 23% in GSB and an additional 40% decline occurred from 1985 to 1988 (Dennison et al. 1989). These losses have been attributed to a mix of ecological disturbances and primarily anthropogenic stressors. Currently, two environmental parameters appear to be limiting the distribution and abundance of *Z. marina* within the south shore estuaries: (1) frequent occurrences of 'Brown Tide' events (*Aureococcus anophagefferens)* which limits light availability and (2) warm summer water temperatures which reduce seagrass productivity and can lead to mortality. The focus of this study was to assess the impact of a new inlet on various water quality parameters and determine the response of seagrass beds to changing water quality conditions in a nearshore community. It is expected that potentially higher salinities, cooler water temperatures, and increased water clarity from the new inlet would favor *Z. marina* growth and expansion.

METHODS

This study includes data collected for the National Park Service's Northeast Coastal and Barrier Network Vital Signs Monitoring Program (NCBN, 2006-2016) and the Hurricane Sandy Assessment Study (2013-2016) as well as data collected for the Suffolk County Office of Ecology. For further details on water quality and seagrass assessment monitoring protocols refer to the NCBN Protocol for Monitoring Estuarine Nutrient Enrichment (Kopp & Neckles 2009) and Neckles et al. (2012) respectively.

Site Description:

Fire Island is a 50-km barrier island on the south shore of Long Island (NY), spanning Great South Bay (GSB) and Moriches Bay (MB). The boundary of Fire Island National Seashore includes ~3,500 hectares of submerged estuarine land which extends 1.2 km north from the bay shoreline of the barrier island into the bays (Fig. 1). The average depth of GSB and MB are 1.3 m and 0.9 m respectively (Wilson et al. 1991).

Great South Bay and Moriches Bay contain the two species of seagrass found on the south shore of Long Island, eelgrass (*Z. marina marina*) and widgeon grass (*Ruppia maritima*). Historically, the temperate species *Z. marina* has been favored over *R. maritima*. The deteriorated water quality and warming summer water temperatures in GSB/MB has restricted *Z. marina* to depths (< 2 m) that allow sufficient light penetration but not so shallow that warm water temperatures cause mortality. The extremely shallow water shoals within GSB/MB are dominated by *R. maritima* due to the ability of this species to cope with warmer temperatures (Kantrud 1991). Phytoplankton blooms have restricted light availability to the benthos (Cosper et al. 1987; Dennison et al. 1989), limiting the growth of seagrass beds throughout GSB/MB (Dennison et al. 1989). This system, once dominated by benthic production, in which seagrasses were the dominant primary producer, has since shifted to a pelagic system, dominated primarily by phytoplankton production in the water column.

Water Quality Monitoring

Water quality data were collected during a specified index period, consisting of four consecutive weeks during the months of July and early August for six years (2009, 2011, 2013-2016). We used a restricted random sampling design to generate sampling locations. This method was chosen due to its ability to generate random sampling sites while ensuring good dispersion (Elzinga et al. 2001). The GSB/MB sampling area was divided into a grid of 30 stratified tessellated hexagons; within each hexagon, a random sampling site was selected each year. Six stations, referred to as trend stations, were measured each week during the index period to account for variability associated with phytoplankton blooms. These six locations remained the same for all years. The remaining 24 stations were divided evenly into four groups so that 6 stations, which spanned the length of our sampling area, were visited each week along with the trend stations. This method ensured both spatial and temporal balance. Water quality data from another five sites were obtained from the Suffolk County Office of Ecology, Marine Water Quality Monitoring Program.

Salinity, temperature, dissolved oxygen and turbidity were measured at the surface, mid-depth and bottom using a YSI 6600 Multi-parameter water quality monitoring sonde. Chlorophyll *a* was measured by taking replicate grab samples from 10 cm beneath the water surface using amber polyethelene bottles. Samples were transported in the dark on ice to the lab for filtration. Filters were stored at -18°C and shipped on dry ice within two weeks of collection for analysis using the USEPA fluorometric method 445.0 (Arar & Collins 1997) to the Atlantic Research Center at Cape Cod National Seashore, North Truro, Massachusetts. Photosynthetically active radiation (PAR) was measured at increasing depths using a Li-Cor LI-1400 datalogger with LI-192 underwater quantum sensors. The light attenuation coefficient, k_d (m^{-1}), was quantified from resulting depth profiles according to the Lambert-Beer law. Measurements for all water quality parameters were restricted to within three hours of apparent noon due to the influence of solar angle on light attenuation.

Rapid Seagrass Assessment

Seagrass sampling sites were visited during the month of August for six years (2007, 2009, 2013-2016), replicating the tier 2 sampling protocol described in Neckles et al. (2012). Once again, a restricted random sampling design was used to generate sampling locations in GSB/MB, this time dividing GSB/MB into a grid of 199 stratified tessellated hexagons, each 925-m wide. Unlike the water quality stations, these 199 sampling sites remained fixed each year.

At each site, four 0.25 m^2 quadrats were subsampled by divers by selecting random sites haphazardly located off the four corners of the boat. Percent cover for each species of seagrass within each 0.25 m^2 quadrat was estimated visually. Overall canopy height was measured at each quadrat. Three shoots per seagrass species were haphazardly selected within each quadrat and the total length was measured (max n=12 per species per site). Average *Z. marina* biomass per site was estimated from total shoot length, and average *R. maritima* biomass was estimated from core sampling, described below.

Biomass Estimation

One hundred random *Z. marina* shoots were collected throughout GSB/MB to develop a relationship between shoot length and weight (biomass). The total length of each individual shoot was measured, dried at 60°C for a minimum of 48 hours, and then weighed to obtain a dried weight. A nonlinear least squares regression was employed to derive a relationship between total shoot length and weight. The resulting regression model was used to predict the weight of the individual shoots collected at the 199 rapid seagrass sampling locations in which total lengths were measured ($W = 3.3 \times 10^{-6} * L^{1.87}$, $r^2 = 0.799$; $p < 0.001$). To estimate the average shoot count per 0.25 m^2, a 20.3 cm diameter PVC core was used to sample the aboveground biomass including all intact shoots at 4 sites across GSB/MB (n=16). The average shoot density per core was employed to estimate the average number of shoots per 0.25 m^2. Using the average shoot weight (estimates from the non-linear regression), average shoot density per 0.25 m^2, and the mean percent cover, *Z. marina* aboveground biomass was estimated per rapid seagrass sampling site. Aboveground biomass for *R. maritima* was estimated by using a 20.3 cm diameter PVC core to sample the aboveground biomass at 4 sites across GSB/MB (n=16). The average aboveground biomass from the cores (100% cover per core) and the percent cover per site from the 0.25 m^2 quadrats were used to estimate *R. maritima* biomass per site.

SeagrassNET Transect Sampling

Following the protocols described in the global monitoring program, SeagrassNET, three 50-m permanent transects were placed within a relatively homogenous seagrass bed and surveyed every year from 2007 to 2016, except 2008. The transects were placed parallel to shore along a depth gradient; one transect at the shallow edge of the seagrass bed (site closest to the barrier island), one at a mid-depth, and one at the deep edge (site farthest from the barrier island; Fig. 1). Transects were delineated by inserting screw anchors into the sediment at the middle and ends of each transect line (0 m, 25 m, and 50 m). Twelve permanent 0.25 m^2 quadrats were selected at random distances along each transect line. Percent cover, shoot density, canopy height, and the number of reproductive shoots at each quadrat were sampled once a year at peak biomass (mid-summer). Water temperature and light intensity were recorded continuously using a HOBO Onset light and temperature data logger, for one week prior to the seagrass sampling day. Data loggers were attached to the 0 m screw anchors ~10 cm from the sediment-water interface at the deep and shallow transects and recorded measurements every 15 minutes.

Statistical Analysis

Water quality and seagrass sites were divided into three different zones based on distance from the disturbance (breach site; Fig. 1); zone 1 includes any stations within a 4.5 km radius and zone 2 includes stations within a 9 km radius. Zone 2 was further divided into east (2E) and west (2W) due to extreme differences in site characteristics.

For water quality analysis, effect size (mean treatment – mean control) was estimated by combining all water quality years before the breach (2007-2012) as the control group and each year after the breach (2013-2016) as the treatment groups, subdivided by distance zone. A resampling method on 2,000 bootstrap samples (with replacement) was used to generate the effect size and 95% confidence intervals (CI). For the bay-wide rapid seagrass assessment, effect size (mean treatment – mean control) was estimated by combining both years before the breach (2007 and 2009), to account for inter annual variability, as the control group and each year after the breach (2013-2016) as the treatment groups, again subdividing by distance zone. The same resampling technique was used as described above for water quality. SeagrassNET data were analyzed using a repeated measures analysis of variance to determine if there was a difference in seagrass percent cover and biomass over time within transect depths, with year as a fixed factor and quadrat as a random factor nested within year.

RESULTS

The average salinity in zone 1 for 2007-2012 (pre-breach/control) was 24.04 PSU, while the average salinity for 2013-2016 (post-breach/treatment groups) was 29.81 PSU, an increase of 5.77 PSU (Fig. 3A). All the mean effect sizes and 95% confidence intervals (2.25 to 6.73 PSU) are positive, indicating that all treatment groups (years after the breach) saw increases in salinity as compared to pre-breach years (Fig. 4A). Post-breach salinity values ranged from 26.6 to 31.97 PSU while pre-breach values ranged from 19.57 to 26.62 PSU. Prior to Hurricane Sandy, the data revealed two different salinity regions; low salinity in zone 1 (range: 19.57 – 26.6 PSU), and high salinity in zone 2E (range: 22.9 – 31. 3 PSU). After the breach, the salinity regions converge on a much narrower and higher salinity range between 26.60 PSU and 31.97 PSU (Fig. 3A). Water temperatures in zone 1 declined by a mean of nearly 2°C (Fig. 3B), decreasing from 26°C pre-breach to 24.1°C post-breach; effect sizes and 95% confidence intervals (-3.88 °C to 0.95 °C; Fig. 4B) are primarily negative every year after the breach in zone 1 indicating a decline. The strongest change in temperature occurred in 2014 with a mean effect size of -3.88 (-4.82 °C to -2.74 °C). Temperature in zones 2E and 2W did not change post-Hurricane Sandy, although there was an increase in temperature in zone 2W in 2013 with a mean effect size of 2.19 (0.82 °C to 3.64 °C) and an increase in zone 2E in 2016, with a mean effect size of 2.84 (1.50 °C to 5.02 °C). Dissolved oxygen in zone 1 increased in 2013 as compared to post breach with a mean effect size of 0.95 (0.10 mg/L to 1.64 mg/L) and remained stable annually at around 7.34 mg/L (Fig. 3C). Pre-breach values were much more variable, fluctuating yearly between 5.15 mg/L and 10.47 mg/L (Fig. 3C). Turbidity, light attenuation, and chlorophyll a measurements varied greatly interannually throughout the entirety of the study, with increases in 2013 and 2015, and declines in 2014 and 2016 (Figs. 3D-F and 4D-F).

Rapid Seagrass Assessment

Mean (± SE) percent cover of seagrass within each zone was highest in 2009 for both species of seagrass, with the exception of Z. marina in zone 2E (Fig. 5A). Immediately following the breach, Z. marina percent cover in zones 1 and 2W declined to zero, remaining below 5% through 2016. The significant decline from pre-breach to post-breach percent cover is evidenced by the predominantly negative mean effect sizes (mean: -7.30%) and 95% confidence intervals (range: -16.9% to 0.3%) in all years post-breach (Fig. 6). Z. marina biomass presented similar negative trends (mean: -2.6 g; range: -5.94 g to 0.03 g). Although the number of sites with seagrass present in zone 1 steadily increased from 2013 to 2016, percent cover and biomass remained extremely low. Hurricane Sandy did not seem to impact Z. marina in zone 2E, in which percent cover and biomass remain steady for all four years after the breach (Fig. 5 and Fig. 6).

It appears that R. maritima was not impacted immediately following Hurricane Sandy, however a drastic decline in 2014 in zone 1 and 2W was observed, with a mean effect size of -12.2% (range: -21.4 % to -0.16 %; Fig. 6). Although R. maritima cover has recovered in zone 1 and 2W, the number of sites with R. maritima present has decreased two-fold (Fig. 5F). Zone 2 saw a recovery of R. maritima and in some instances percent cover that exceeded pre-breach conditions, however, the number of sites with seagrass present also declined two-fold (Fig. 5F).

SeagrassNET

In 2007, the deep transect had the highest percent cover of Z. marina, followed by the shallow transect, and then the mid-depth transect, with no Z. marina present. From 2007 to 2009, a significant decline in Z. marina percent cover was observed at the shallow and deep transects and since 2011 there has been no Z. marina present at any transect ($p < 0.001$; Fig. 7A, Fig. 8A, Table 1). Z. marina biomass declined significantly at the deep transect from 2009 to 2011 ($p < 0.001$; Fig. 7B, Fig. 8B, Table 1). The deep edge of the bed has receded about 250 m south toward the shoreline of Fire Island while the shallow edge has receded north a minimum of 100 m (Fig. 9).

R. maritima has been much more variable at each transect and from year to year. The mid-depth and shallow transects saw a very similar trend to that of zone 1 in the bay-wide rapid seagrass assessment, with a significant decline occurring in 2014 ($p < 0.001$; Table 1). However, unlike the bay-wide assessment, a recovery of R. maritima was not observed, but instead further declined in 2016 ($p < 0.001$; Table 1), with no R. maritima present in the deep transect, 1% in one quadrat in the mid-depth transect and only an average of 25% cover remaining in 7 out of the 12 quadrats at the shallow transect.

DISCUSSION

Nearshore marine communities can be heavily altered by hurricanes and cyclones through high winds, large wave action, high amounts of rainfall, and storm surge. The impacts of these weather events on seagrass beds can be variable, depending on the strength of the storm itself, the time of year, and the location of impact. Hurricanes can completely devastate a seagrass bed as in the case of Cyclone Sandy in the Gulf of Carpinteria, Australia (Poiner et al. 1989) or have almost no impact on the system, such as Hurricane Katrina in the northern Gulf of Mexico (Anton et al. 2009). In our study system, the barrier island parallel to the south shore of Long Island was heavily impacted by storm surge produced by Hurricane Sandy, leading to the creation of a new ocean inlet to GSB, a back-bay estuarine lagoon. The impact of this new inlet on water quality and seagrass was both site- and species-specific.

Among the factors driving seagrass loss were the intense sediment dynamics (burial and erosion) associated with the flood-tide delta formed in the estuary (Fig. 2). Using experimental manipulations, Mills & Fonseca (2003) found that 4 cm of sediment burial caused 50% mortality in *Z. marina* while 12 cm resulted in 100% mortality. Hurricanes and storms can cause large alterations to a seagrass habitat through sediment movement and deposition such as the impact of Hurricane George in south Florida (Fourqurean & Rutten 2004). Inlet formation, migration, and alteration is another factor which can directly impact seagrasses such as the *Zostera noltii* decline observed in Ria Formosa, Southern Portugal (Cunha et al. 2005). Even though sediment transport and deposition was not recorded in our study, it is possible that the flood shoals created during initial breach formation (Deltares pers. comm. 2016; Methratta et al. 2016) is a direct cause for the majority of *Z. marina* lost in zone 1 (pers. obs.). Several sites with seagrass beds present before Hurricane Sandy are now completely covered with intertidal sand and the flood shoal delta remains very dynamic, with continuously shifting bathymetry (Fig. 2; Deltares & Nelson 2016; Nelson et al. 2016a).

The effect of Hurricane Sandy on water quality was site-specific, understandably having the most pronounced impact on zone 1 where the barrier island breach occurred. Prior to the breach, salinity levels in zone 1 were mainly driven by riverine outflow from the Carman's River directly to the north of the breach site as well as groundwater seepage. The new oceanic inflow now allows for increased flushing rates in this back-bay region. Residence times near zone 1 (east Great South Bay) were estimated to have declined from 25 to 10 days (Flagg et al. 2015; Gurdon et al. 2015). Zone 2E is affected by Moriches Inlet to the east which contributed to its overall higher salinity before the breach. Although this resulted in a relatively smaller effect size than zone 1 or 2W, there was still a significant difference from pre-breach to post-breach salinity in zone 2E (Figs. 3A and 4A). Since this study was temporally and spatially limited, it is important to consider results from other studies that have examined water quality in this region, both annually and on a more finite scale. Gobler et al. (2014) found that the region designated as zone 1 saw somewhat cooler summer temperatures, and in some cases, declines by as much as 3°C, which correspond to results from this study.

Light limitation is a major environmental factor affecting the growth, photosynthesis, and distribution of seagrasses (Dennison 1987; Dennison & Alberte 1985). Several factors affect light availability in Great South Bay, including wind driven resuspension, harmful algal blooms, and eutrophication from anthropogenic sources (Schubel et al. 1991). Turbidity, light attenuation, and chlorophyll *a* were all extremely variable after the breach (Fig. 3 and Fig. 4). This high inter-annual variability in turbidity and light attenuation can be partially explained by levels of chlorophyll *a* concentrations (Fig 3). Water clarity is also heavily influenced by wind-driven sediment resuspension due to the overall shallow nature of Great South Bay, thereby contributing to the reduction in light penetration through the water column. It is possible that this sediment resuspension may have caused the large differences in turbidity and light attenuation present in zone 2W in 2013 and 2015 as compared to zone 1 and 2E where flushing rates are presumably higher due to a closer proximity to an inlet. Although there was no evidence in this study for the improvement in water clarity in zone 1, Gobler et al. (2014) saw a decline in the frequency and intensity of harmful brown tides, which began during the summer of 1985 and have continued to impact Long Island waters (Cosper et al. 1987; Gobler & Sunda 2012). The high inter-annual variability in turbidity and light attenuation, sediment resuspension, and the annual reoccurrence of brown tide events (even though less intense and less frequent) (Gobler & Sunda 2012) are limiting light availability to the benthos and therefore may be prolonging the recovery of *Z. marina* since Hurricane Sandy.

The autecology of the two seagrasses could assist in explaining the species-specific response to the creation of the new inlet. *R. maritima* is known to have a wide salinity tolerance, with a range that can extend from 0 to 70 PSU and in comparison to other species, can thrive in much higher water temperatures (Kantrud 1991). *R. maritima* is known as a "high-light" plant

(Wetzel 1981), which can explain the distribution of *R. maritima* in the shallowest areas of GSB/MB. *Z. marina*, on the other hand, is a slow growing species, with much narrower salinity and temperature thresholds (Bulthuis 1987; Marsh et al. 1986).

Unlike *Z. marina, R. maritima* was not immediately impacted by the breach. The pioneer species high fecundity (Kantrud 1991) could have facilitated its quick re-colonization into the disturbed sites and therefore no direct impact in 2013 was observed. The largest *R. maritima* decline in zone 1 (including seagrassNET data) and 2W occurred in 2014, which also corresponds with the largest change in water temperatures (decrease of nearly 4°C; Fig. 3 and Fig. 4) from pre-breach to post-breach conditions. Although the thermal range for *R. maritima* can be large (Kantrud 1991), the optimum temperature is at or near 30°C (Wetzel 1981). Common explanations for *R. maritima* declines are increases in turbidity, competition (usually for light) by algae or other angiosperms, or increased wave action or water depth (Kantrud 1991). While it is possible that there may have been an increase in turbidity at sites with *R. maritima,* neither this study, nor Gobler et al. (2014) saw evidence of this. On the contrary, water clarity was significantly lower than pre-breach years (Fig. 4). In GSB/MB, *Z. marina* and *R. maritima* have depth-dependent distribution patterns, the former typically inhabiting deeper depths due to thermal stressors in shallower water. Although competition may be apparent at certain sites where the species occur simultaneously, it is more typical that the two species do not coincide. Although no specific studies have been conducted to evaluate the effect of the breach on wave activity in the bay, the general consensus is that wave activity has not increased inside the bay (Methratta et al. 2016). If wave action or water depth were important factors in the decline in *R. maritima*, it would have been noted in 2013 immediately following the opening of the inlet. Although the cause of the *R. maritima* decline in 2014 is still unclear, *R. maritima* was able to recover at most sites across GSB/MB.

The re-colonization and formation of new seagrass beds following major disturbance events requires the joint effort of two major reproductive strategies (vegetative growth and sexual reproduction). The die-off of *Z. marina* in zones 1 and 2W as a result of the breach, and the subsequent years without substantial recovery, may have led to a large decline in the seed bank supplying the region, reducing the efficiency of *Z. marina* to recover through sexual reproduction. While the viability of *Z. marina* seeds in the field is unclear, experimental studies have found that seeds typically do not remain viable for more than a year (Hootsmans et al. 1987; Moore et al. 1993). The long distance dispersal and importance of sexual reproduction in slow-growing seagrass species, such as *Thalassia hemprichii*, for recolonization at far distances from existing meadows is clear (Olesen et al. 2004). Their high reproductive output as well as high seed dispersal can allow for the recolonization of large bare patches where rhizome extension could take many years (Olesen et al. 2004). The distance of dispersal for seeds and reproductive shoots range anywhere from 1 m to 1000 m (Kendrick et al. 2012; Waycott et al. 2006). However, several studies have found that naked seed dispersal distances are typically within 5 m of release sites (Billingham et al. 2007; Orth et al. 1994). For example, Furman et al. (2015) found that seed dispersal distances for naked *Z. marina* seeds ranged from ~2 m to ~5.3 m from the source, and the range for *Z. marina* seed dispersal released by floating reproductive shoots was ~0.17 m to ~35 m, although it is likely much farther.

The difference between the efficiency of asexual reproduction vs sexual reproduction can be variable post disturbance, depending on the initial intensity of the disturbance. Macreadie et al. (2014) showed that although *Zostera muelleri* is resilient to high intensity disturbance events through asexual reproduction (vegetative encroachment), recovery through sexual reproduction (seed dispersal) was absent. In Great South Bay, it appears that *Z. marina* has been unable to fully recover from Hurricane Sandy. However, the appearance of seedlings have been observed throughout several sites. This is evidenced by the increase in number of sites with grass present in zone 1 (Fig. 5E) with very low percent cover post-breach (Fig. 4A). This initial colonization into bare areas may lead to expansion through vegetative growth, and the future recolonization of these once dense seagrass beds; however, the time frame for recovery may be prolonged due to continued light limitation.

Due to the methodology in which the same site was revisited every year, the impact of Hurricane Sandy on specific seagrass beds not included by the randomly selected sampling sites was unable to be assessed. For example, one seagrass bed closest to the inlet sampled every year for fauna had rapidly expanded (per. obs.). In 2013, this sampling site within zone 1 was completely covered with sand, with no seagrass present. In 2014, the appearance of mainly *R. maritima*, with sparse *Z. marina,* was observed. In 2015, the seagrass bed was comprised of an even mix of both seagrass species, and in the most recent year of sampling a clear separation in the seagrass bed was observed, in which the shallows were dominated by *R. maritima* and the deep edge was dominated by *Z. marina.*

Tiered monitoring is a very effective tool for the evaluation of an ecosystem's condition. As described by Neckles et al. (2012), bay-wide rapid seagrass assessment is useful for recognizing environmental issues at large geographic scales as evidenced by the zone-specific impact of Hurricane Sandy on seagrass beds. Intensive fine scale monitoring at a representative site can help explain general changes occurring throughout the entire system. This provides higher resolution information, as evidenced by the steady decline in *Z. marina* at the seagrassNET site well before Hurricane Sandy (Figs. 7 and 8). This might suggest that *Z. marina* was already stressed prior to Hurricane Sandy, representing another factor contributing to its slow recovery. *R. maritima,* on the other hand, showed an increasing trend (although not significant) in percent cover and biomass prior to Hurricane Sandy.

Continuous environmental monitoring, although difficult, is essential in providing valuable information for decision-making. The long-term monitoring of seagrass, for example, provides very useful insight into the ability of seagrass beds to respond to climate change, anthropogenic stressors, and extreme environmental events. The relationship between seagrass beds and water quality has received considerable attention due to the impacts of deteriorating water quality on seagrass loss (Orth et al. 2006; Waycott et al. 2009). This is particularly true in coastal systems such as Great South Bay. Once New York State's most productive water body, Great South Bay has lost more than 90% of seagrass cover (NYS Seagrass Taskforce 2009) and has been impacted by nutrient loading from aging wastewater systems (Kinney & Valiela 2011) and the overfishing of bivalves such as the hard clam, *Mercenaria mercenaria* (Kraeuter et al. 2008). Alleviating anthropogenic stressors on seagrass beds is of primary concern among conservation groups, local policy makers, and governmental agencies. Understanding how seagrass beds respond to weather events and changes in water quality will be an important step in moving forward through a changing climate and an increasing human population. While the breach has had many impacts on Great South Bay such as explicit improvements in water quality and more ambiguous species-specific responses to changing water quality characteristics, further monitoring may reveal trends which still may be unclear four years after Hurricane Sandy.

LITERATURE CITED

Anton A, Cebrian J, Duarte C, Heck Jr KL, and Goff J. 2009. Low impact of Hurricane Katrina on seagrass community structure and functioning in the northern Gulf of Mexico. *Bulletin of Marine Science* 85:45-59.

Arar EJ, and Collins GB. 1997. U.S. EPA Method 445.0 In vitro determination of chlorophyll a and pheophytin a in marine and freshwater phytoplankton by fluorescence. Revision 1.2 .Environmental Monitoring Systems Laboratory, Office of Research and Development, U.S.E.P.A., Cincinnati, Ohio (EPA/600/R-92/121, Nov. 1992). .

Billingham MR, Simões T, Reusch TBH, and Serrão EA. 2007. Genetic sub-structure and intermediate optimal outcrossing distance in the marine angiosperm Zostera marina. *Marine Biology* 152:793-801.

Bulthuis DA. 1987. Effects of temperature on photosynthesis and growth of seagrasses. *Aquatic Botany* 27:27-40.

Burkholder JM, Tomasko DA, and Touchette BW. 2007. Seagrasses and eutrophication. *Journal of Experimental Marine Biology and Ecology* 350:46-72.

Cosper EM, Dennison WC, Carpenter EJ, Bricelj VM, Mitchell JG, Kuenstner SH, Colflesh D, and Dewey M. 1987. Recurrent and Persistent Brown Tide Blooms Perturb Coastal Marine Ecosystem. *Estuaries* 10:284-290.

Cunha AH, Santos RP, Gaspar AP, and Bairros MF. 2005. Seagrass landscape-scale changes in response to disturbance created by the dynamics of barrier-islands: A case study from Ria Formosa (Southern Portugal). *Estuarine, Coastal and Shelf Science* 64:636-644.

Deltares MvO, and Nelson T. 2016. Wilderness breach modeling study. http://cosmosdeltaresnl/FireIsland/ *Assessed January 2016*.

Dennison WC. 1987. Effects of light on seagrass photosynthesis, growth and depth distribution. *Aquatic Botany* 27:15-26.

Dennison WC, and Alberte RS. 1985. Role of daily light period in the depth distribution of *Zostera marina* (eelgrass) *Marine Ecology Progress Series* 25:51-61.

Dennison WC, Marshall GJ, and Wigand C. 1989. Effect of "Brown Tide" Shading on Eelgrass (Zostera Marina L.) Distributions. In: Cosper EM, Bricelj VM, and Carpenter EJ, eds.

Novel Phytoplankton Blooms: Causes and Impacts of Recurrent Brown Tides and Other Unusual Blooms. Berlin, Heidelberg: Springer Berlin Heidelberg, 675-692.

Dolan R, and Godfrey P. 1973. Effect of Hurricane Ginger on the Barrier Islands of North Carolina. *Geological Society of America Bulletin* 84:1329-1334.

Duarte CM. 1995. Submerged aquatic vegetation in relation to different nutrient regimes. *Ophelia* 41:87-112.

Elzinga CL, Salzer DW, Willoughby JW, and Gibbs JP. 2001. *Monitoring plant and animal populations*. Malden, Mass: Blackwell Science.

Flagg CN, Flood R, Wilson R, and Bokuniewicz H. 2015. Update on the Old Inlet Breach and Great South Bay. School of Marine and Atmospheric Sciences, Stony Brook University. August 7, 2014.

Fourqurean JW, and Rutten LM. 2004. The impact of Hurricane Georges on soft-bottom, backreef communities: site- and species-specific effects in south Florida seagrass beds. *Bulletin of Marine Science* 72:239-257.

Furman BT, Jackson LJ, Bricker E, and Peterson BJ. 2015. Sexual recruitment in Zostera marina: A patch to landscape-scale investigation. *Limnology and Oceanography* 60:584-599.

Glancy JB. 1956. Biological Benefits of the Moriches and Shinnecock Inlets with Particular Reference to Pollution and the Shellfisheries. Report to the District Engineer, U.S. Army Corp of Engineers, New York District, 18 pp.

Gobler CJ, Collier J, and Lonsdale D. 2014. Assessing the response of the Great South Bay plankton community to Hurricane Sandy. SUNY Stony Brook Powerpoint presentation.

Gobler CJ, and Sunda WG. 2012. Ecosystem disruptive algal blooms of the brown tide species, Aureococcus anophagefferens and Aureoumbra lagunensis. *Harmful Algae* 14:36-45.

Gurdon C, Clyde T, Wilson R, and Flagg C. 2015. Residence time changes in a coastal lagoon associated with a barrier island breach during Hurricane Sandy. *Estuaries and Coast (Submitted)*.

Heck KL, Carruthers TJB, Duarte CM, Hughes AR, Kendrick G, Orth RJ, and Williams SW. 2008. Trophic Transfers from Seagrass Meadows Subsidize Diverse Marine and Terrestrial Consumers. . *Ecosystems* 11:1198-1210.

Hemminga M, and Duarte CM. 2000. *Seagrass Ecology*. Cambridge, United Kigndom: Cambridge University Press.

Hootsmans MJM, Vermaat JE, and Van Vierssen W. 1987. Seed-bank development, germination and early seedling survival of two seagrass species from The Netherlands: Zostera marina L. and Zostera noltii hornem. *Aquatic Botany* 28:275-285.

Jones CG, Lawton JH, and Shachak M. 1994. Organisms as Ecosystem Engineers. *Oikos* 69:373-386.

Kantrud HA. 1991. Wigeongrass (*Ruppia maritima* L.): a literature review. *US Fish and Wildlife Service Fish and Wildlife Research* 10:58.

Kemp WM, Boynton WR, Adolf JE, Boesch DF, Boicourt WC, Brush G, Cornwell JC, Fisher TR, Glibert PM, Hagy JD, Harding LW, Houde ED, Kimmel DG, Miller WD, Newell RIE, Roman MR, Smith EM, and Stevenson JC. 2005. Eutrophication of Chesapeake Bay: historical trends and ecological interactions. *Marine Ecology Progress Series* 303:1-29.

Kendrick GA, Waycott M, Carruthers TJB, Cambridge ML, Hovey R, Krauss SL, Lavery PS, Les DH, Lowe RJ, Vidal OMi, Ooi JLS, Orth RJ, Rivers DO, Ruiz-Montoya L, Sinclair EA, Statton J, van Dijk JK, and Verduin JJ. 2012. The Central Role of Dispersal in the Maintenance and Persistence of Seagrass Populations. *Bioscience* 62:56-65.

Kennish MJ, Bricker SB, Dennison WC, Glibert PM, Livingston RJ, Moore KA, Noble RT, Paerl HW, Ramstack JM, Seitzinger S, Tomasko DA, and Valiela I. 2007. Barnegat Bay–Little Egg Harbor Estuary: Case Study of a Highly Eutrophic Coastal Bay System *Ecological Applications* 17:S3-S16.

Kinney EL, and Valiela I. 2011. Nitrogen Loading to Great South Bay: Land Use, Sources, Retention, and Transport from Land to Bay. *Journal of Coastal Research*:672-686.

Kitting CL, Fry B, and Morgan MD. 1984. Detection of Inconspicuous Epiphytic Algae Supporting Food Webs in Seagrass Meadows. *Oecologia* 62:145-149.

Kopp BS, and Neckles HA. 2009. A Protocol for Monitoring Estuarine Nutrient Enrichment in Coastal Parks of the National Park Service Northeast Region. Natural Resource Report. NPS/NCBN/NRR—2009/110. National Park Service. Fort Collins, Colorado.

Kraeuter JN, Klinck JM, Powell EN, Hofmann EE, Buckner SC, Grizzle RE, and Bricelj VM. 2008. Effects of the Fishery on the Northern Quahog (=Hard Clam, Mercenaria

mercenaria L.) Population in Great South Bay, New York: A Modeling Study. *Journal of Shellfish Research* 27:653-666.

Leatherman SP. 1985. Geomorphic and stratigraphic analysis of Fire Island, New York. *Marine Geology* 63:173-195.

Leatherman SP, and Allen JR. 1985. Geomorphic Analysis: Fire Island Inlet to Montauk Point, Long Island, New York.: National Park Service technical report. Boston, Massachusetts. . p 350 pp.

Macreadie PI, York PH, and Sherman CD. 2014. Resilience of Zostera muelleri seagrass to small-scale disturbances: the relative importance of asexual versus sexual recovery. *Ecol Evol* 4:450-461.

Marsh JA, Dennison WC, and Alberte RS. 1986. Effects of temperature on photosynthesis and respiration in eelgrass (*Zostera marina*). *Journal of Experimental Marine Biology and Ecology* 101:257-267.

Methratta ET, Pacelli CL, Fields L, Bosma K, and Clark HJ. 2016. Technical synthesis report for 24 physical and ecological resources at Fire Island National Seashore: . Natural 25 Resource Report NPS/XXXX/NRR—2016/XXX. Fort Collins, Colorado: National Park Service.

Mills KE, and Fonseca MS. 2003. Mortality and productivity of eelgrass Zostera marina under conditions of experimental burial with two sediment types. *Marine Ecology Progress Series* 255:127-134.

Moore KA, Orth RJ, and Nowak JF. 1993. Environmental regulation of seed germination in Zostera marina L. (eelgrass) in Chesapeake Bay: effects of light, oxygen and sediment burial. *Aquatic Botany* 45:79-91.

Neckles HA, Kopp BS, Peterson BJ, and Pooler PS. 2012. Integrating Scales of Seagrass Monitoring to Meet Conservation Needs. *Estuaries and Coasts* 35:23-46.

Nelson TR, Miselis JL, Hapke CJ, Wilson KE, Henderson RE, Brenner OT, Reynolds BJ, and Hansen ME. 2016a. Coastal bathymetry data collected in June 2014 from Fire Island, New York—The Wilderness Breach and Shoreface: U.S. Geological Survey Data Series 1007.

Olesen B, Marb, xe, N., Duarte CM, Savela RS, and Fortes MD. 2004. Recolonization Dynamics in a Mixed Seagrass Meadow: The Role of Clonal versus Sexual Processes. *Estuaries* 27:770-780.

Orth RJ, Carruthers TJB, Dennison WC, Duarte CM, Fourqurean JW, Heck KL, Hughes AR, Kendrick GA, Kenworthy WJ, Olyarnik S, Short FT, Waycott M, and Williams SL. 2006. A Global Crisis for Seagrass Ecosystems. *Bioscience* 56:987-996.

Orth RJ, Luckenbach M, and Moore KA. 1994. Seed Dispersal in a Marine Macrophyte: Implications for Colonization and Restoration. *Ecology* 75:1927-1939.

Poiner IR, Walker DI, and Coles RG. 1989. Regional studies: seagrasses of tropical Australia. In: A. W. D. Larkum, J. A. McComb, and Shepherd SA, eds. *Biology of seagrasses: a treatise on the biology of seagrasses with special reference to the Australian region*. Amsterdam: Elsevier, 279-303.

Reid GK. 1957. Biologic and Hydrographic Adjustment in a Disturbed Gulf Coast Estuary1. *Limnology and Oceanography* 2:198-212.

Schubel JR, Bell TM, and Carter HH. 1991. *The Great South Bay*. Stony Brook, NY: State University of New York Press, Albany.

Stone GW, and McBride RA. 1998. Louisiana Barrier Islands and Their Importance in Wetland Protection: Forecasting Shoreline Change and Subsequent Response of Wave Climate. *Journal of Coastal Research* 14:900-915.

Walker DI, Kendrick GA, and McComb AJ. 2006. Decline and Recovery of Seagrass Ecosystems— The Dynamics of Change. In: Larkum A, Orth JR, and Duarte CM, eds. *Seagrasses: Biology, Ecology and Conservation*. Dordrecht: Springer Netherlands, 551-565.

Waycott M, Duarte CM, Carruthers TJB, Orth RJ, Dennison WC, Olyarnik S, Calladine A, Fourqurean JW, Heck KL, Hughes AR, Kendrick GA, Kenworthy WJ, Short FT, and Williams SL. 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences* 106:12377-12381.

Waycott M, Procaccini G, Les DH, and Reusch TBH. 2006. Seagrass evolution, ecology and conservation: A genetic perspective. In. In: Larkum AWD, Orth RJ, and Duarte (Eds.) CM, eds. *Seagrasses: Biology, Ecology and Conservation (25 - 50)*. Dordrecht: Springer.

Wetzel RL. 1981. Structural and functional aspects of the ecology of submerged aquatic macrophyte communities in the lower Chesapeake Bay,U.S. In: Wetzel RL, editor. EPA Final Grant Report No R 805974 and X003245-01 and Virginia Institute of Marine Science SRAMSOE 267. Gloucester Point, Virginia, USA.

Wilson R, Wong K-C, and Carter HH. 1991. Aspects of circulation and exchange in Great Sout Bay. In: *In:* Schubel JR, Bell TM, and Carter (eds.) HH, eds. *The Great South Bay*. Stony Brook, New York: Marine Sciences Research Center, 9-21.

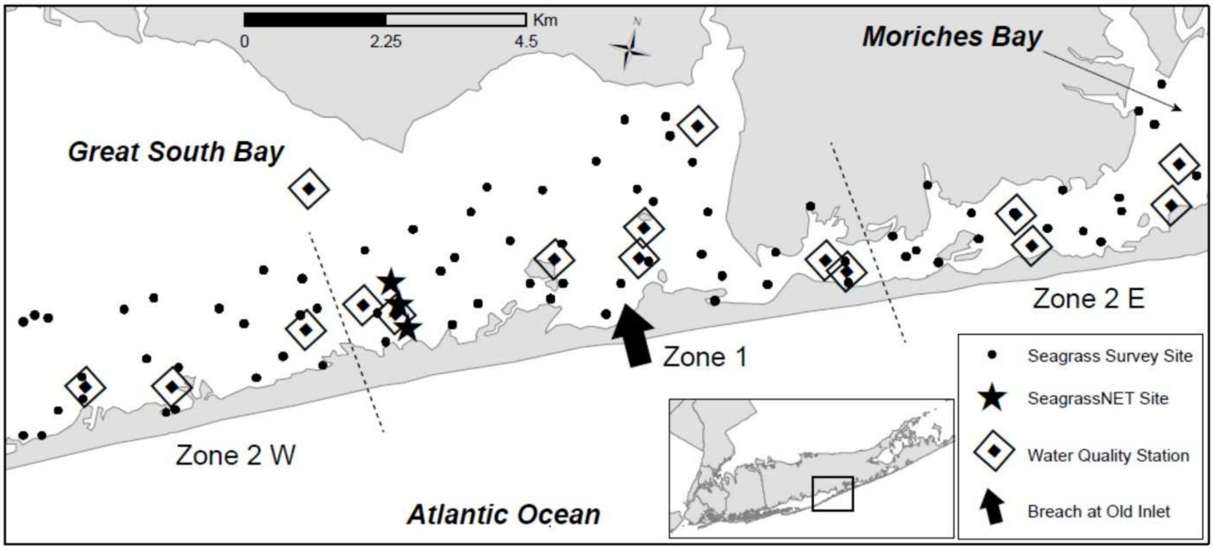


Figure 1. Location of water quality stations and seagrass sampling sites in Great South Bay, NY.

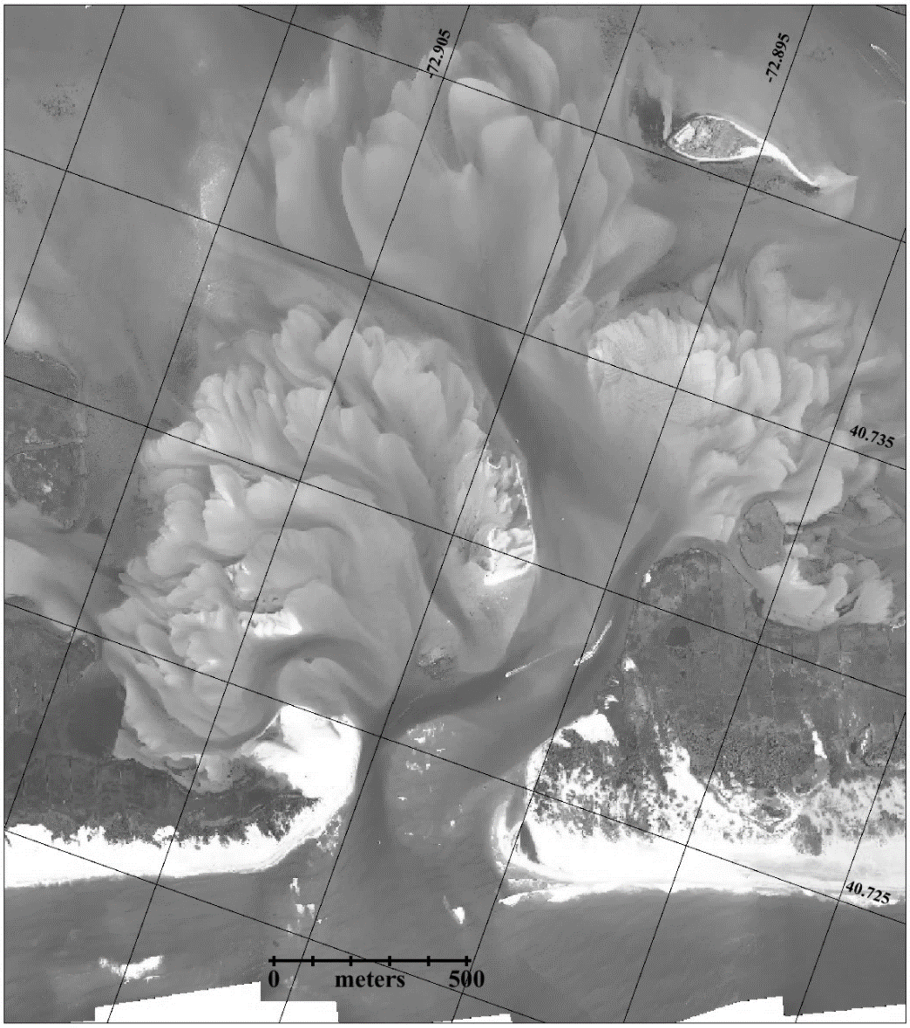


Figure 2. Aerial image of the breach at Old Inlet as of August 5, 2016. Source: Charlie Flagg

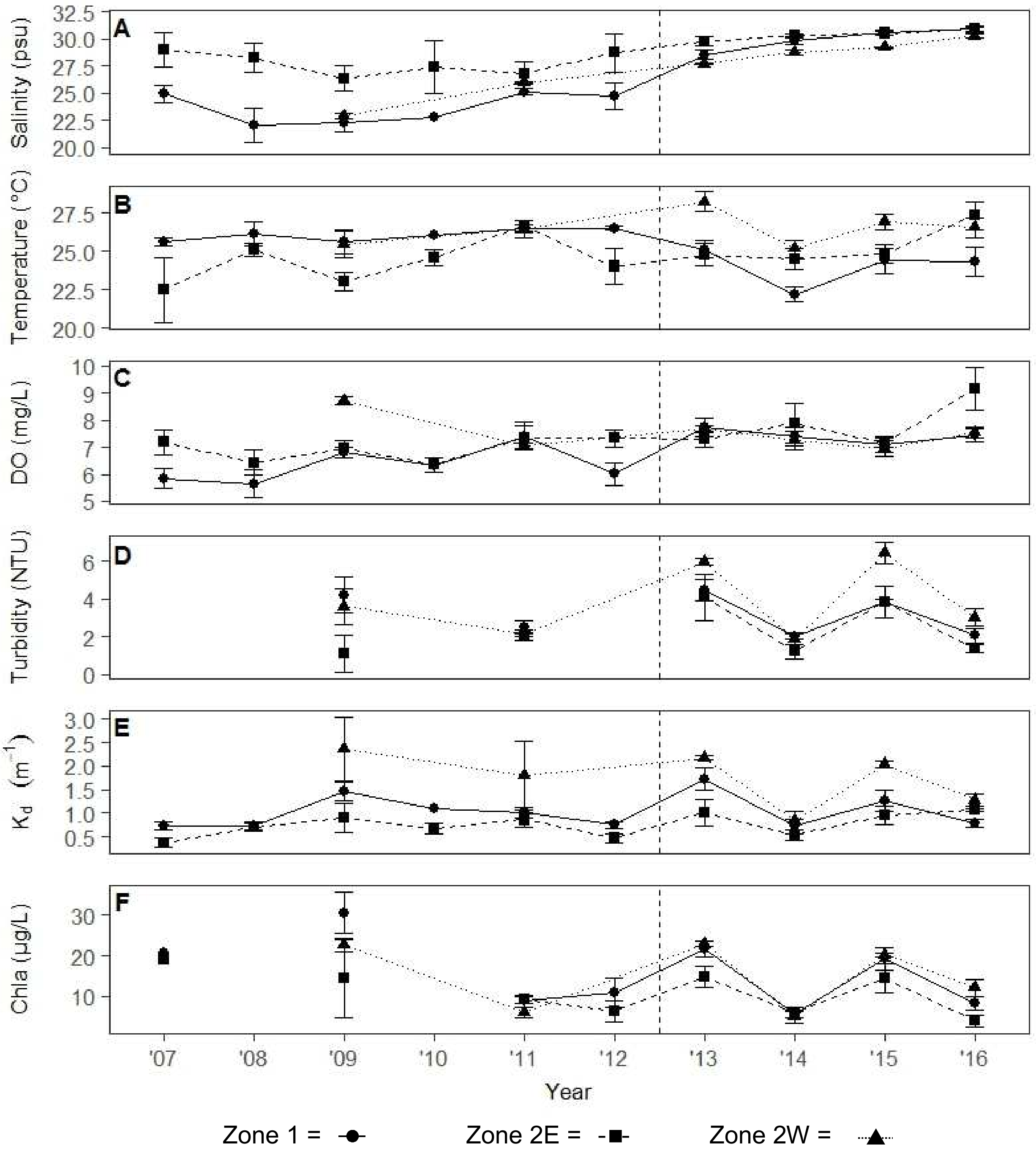


Figure 3. Mean (± SE) water quality data (A) salinity, (B) temperature, (C) dissolved oxygen, (D) turbidity, (E) light attenuation, (F) chlorophyll *a,* at all sites located within each sampling zone.

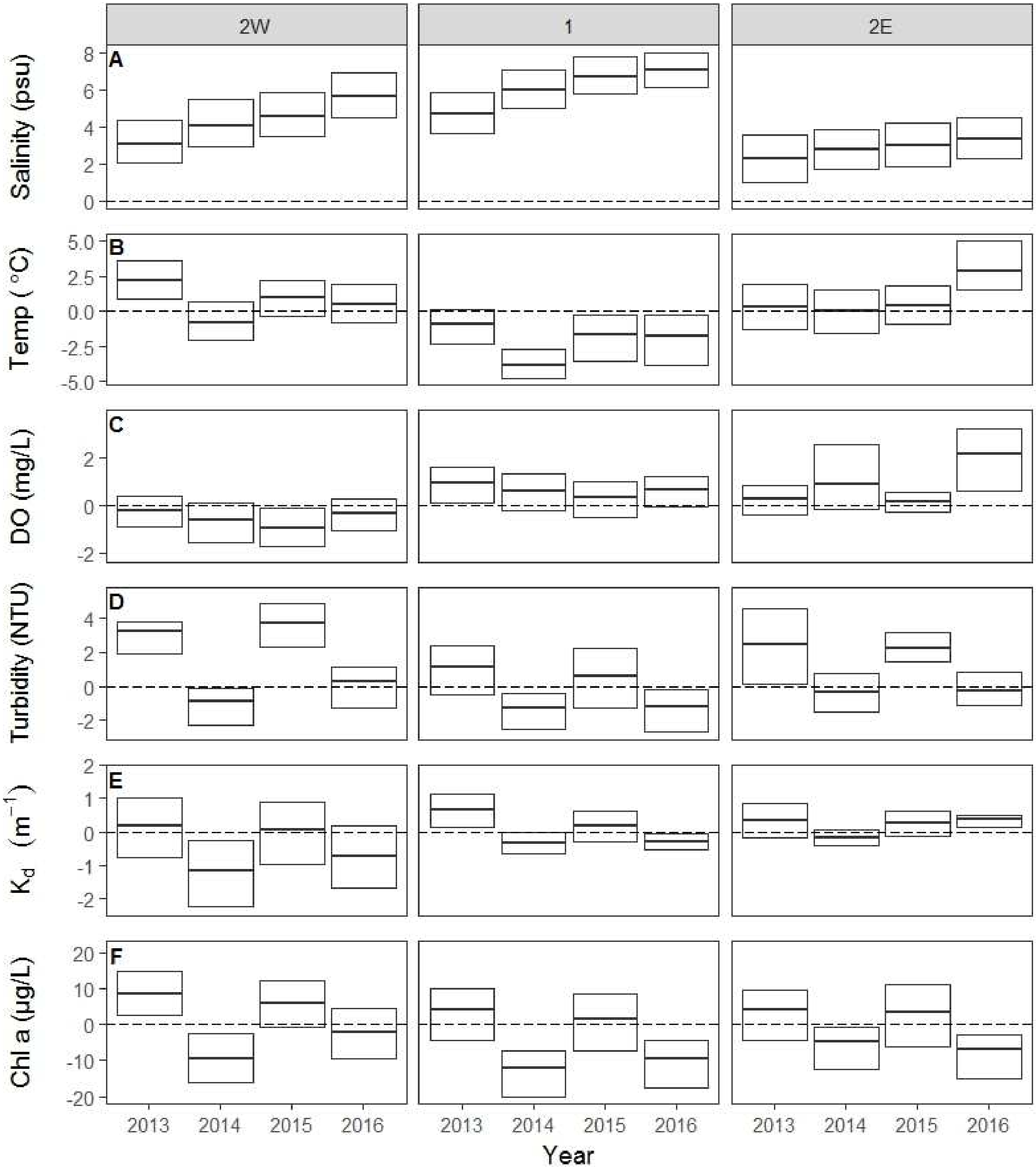


Figure 4. Mean effect size (95% confidence intervals) for water quality data (A) salinity, (B) temperature, (C) dissolved oxygen, (D) turbidity, (E) light attenuation, (F) chlorophyll *a,* at all sites located within each sampling zone.

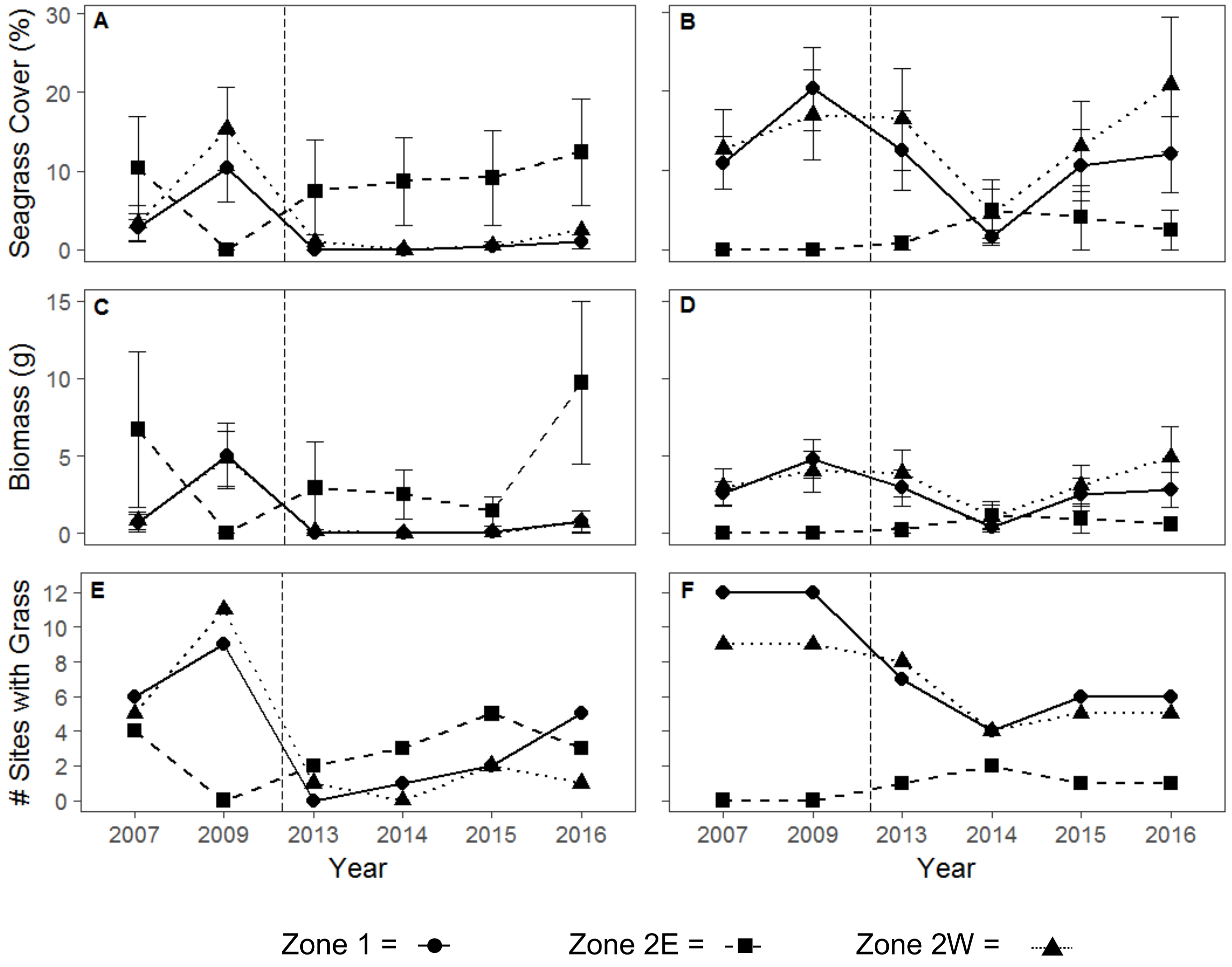


Figure 5. Mean (± SE) percent cover (AB), biomass (CD), and number of sites (EF) with seagrass present for bay-wide rapid seagrass sampling for sites located within each sampling zone. Panels ACE are *Z. marina*, panels BDF are *R. maritima*.

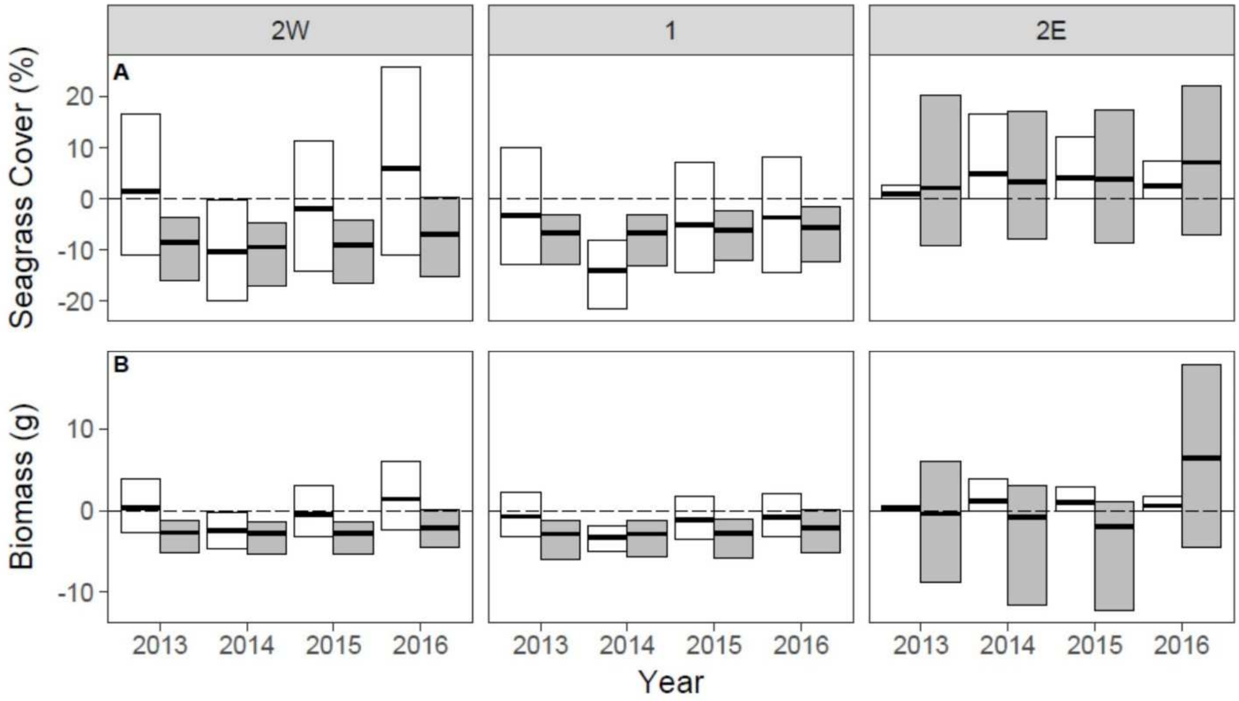


Figure 6. Mean effect size (95% confidence intervals) for rapid seagrass percent cover (A) and biomass data (B). White bars are *R. maritima*, grey bars are *Z. marina*

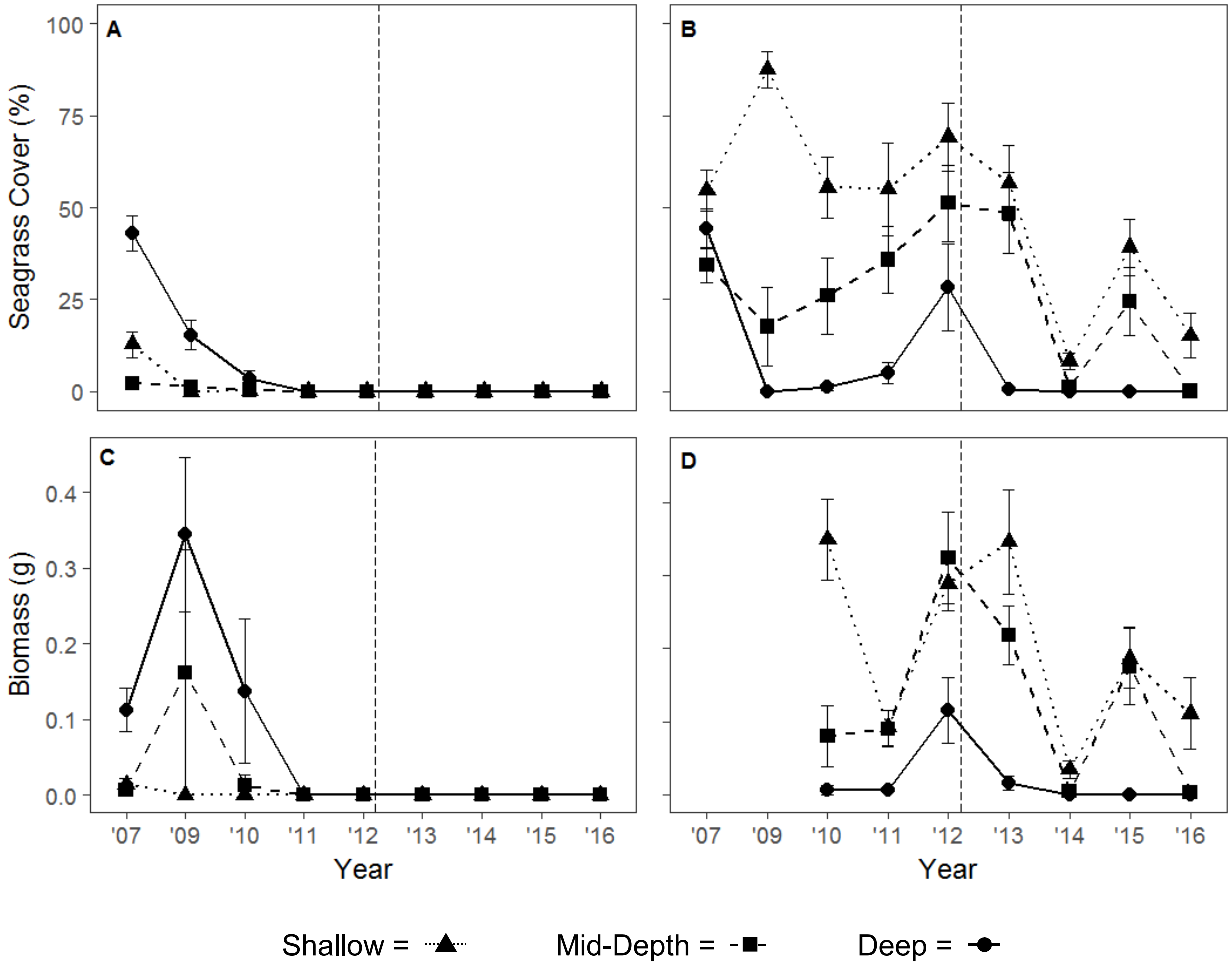


Figure 7. Mean (± SE) percent cover (AB) and biomass (CD) for seagrassNET data at all quadrats located within each transect depth. Panels AC are *Z. marina*, panels BD are *R. maritima*.

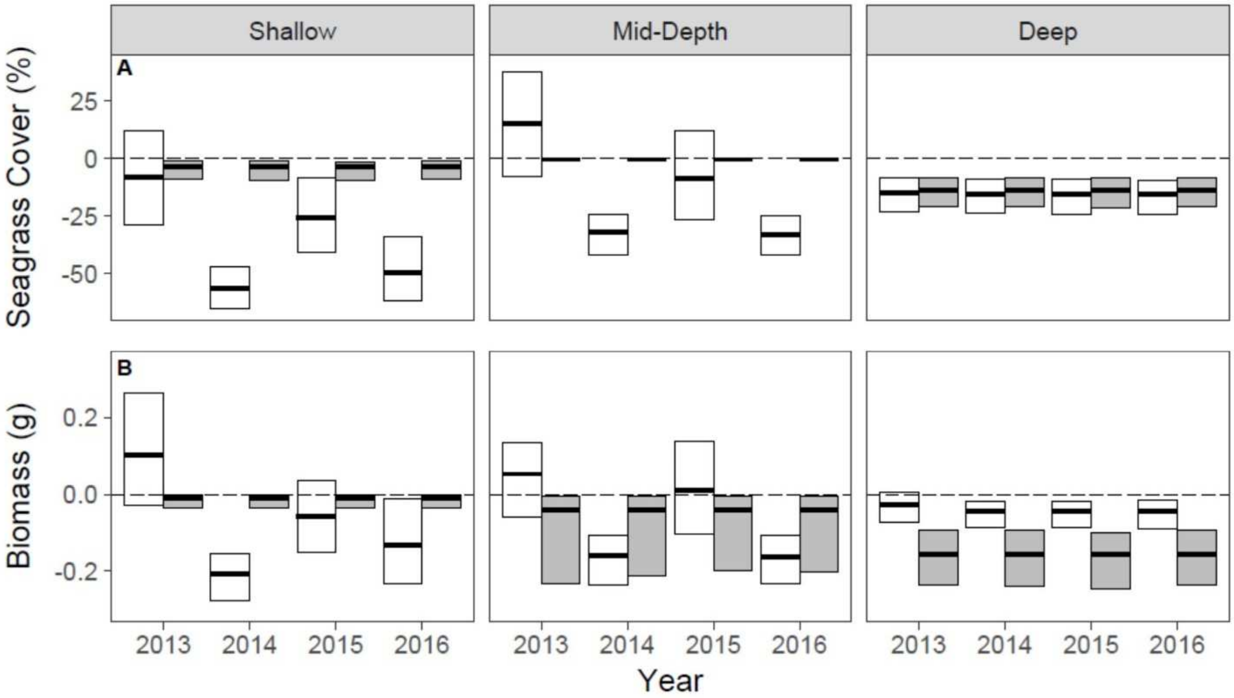


Figure 8. Mean effect size (95% confidence intervals) for seagrassNET percent cover (A) and biomass data (B). White bars are *R. maritima*, grey bars are *Z. marina*

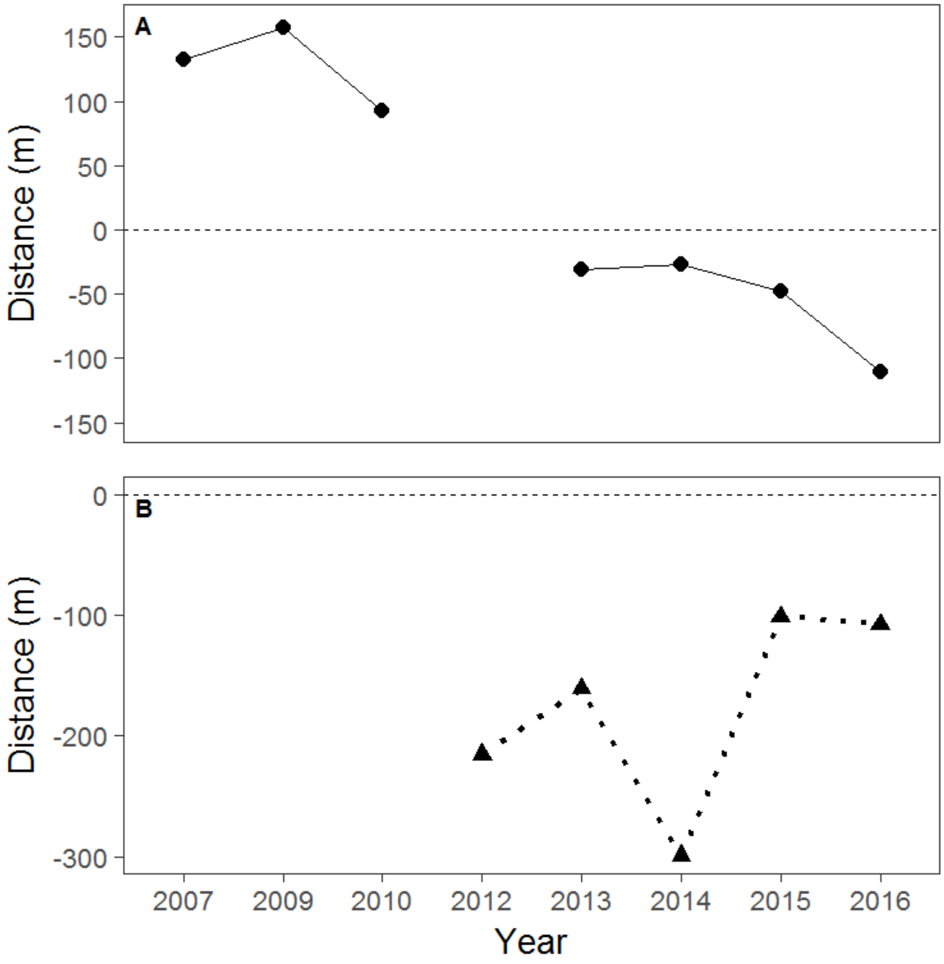


Figure 9. Distance from transect (dashed line) to edge of bed at seagrassNET sampling site. Panel A is the deep transect and panel B is the shallow transect

Table 1. Differences in *Z. marina* and *R. maritima* attributes over time as determined by repeated measures analysis of variance at the seagrassNET sampling site. Similar letters indicate no significant difference between years within species of seagrass and transect depth.

R. maritima

		2007	2009	2010	2011	2012	2013	2014	2015	2016
% Cover	Shallow	a	b	a	a	a	a	c	ad	cd
	Mid-Depth	a	a	a	a	ac	ac	b	bc	b
	Deep	a	b	b	b	a	b	b	b	b
Biomass	Shallow	-	-	a	bc	a	a	b	acd	bd
	Mid-Depth	-	-	a	a	bc	bc	d	bc	d
	Deep	-	-	a	a	b	a	a	a	a

Z.marina

		2007	2009	2010	2011	2012	2013	2014	2015	2016
% Cover	Shallow	a	b	b	b	b	b	b	b	b
	Mid-Depth	a	a	a	a	a	a	a	a	a
	Deep	a	b	b	b	b	b	b	b	b
Biomass	Shallow	a	b	b	b	b	b	b	b	b
	Mid-Depth	a	a	a	a	a	a	a	a	a
	Deep	a	a	ab	b	b	b	b	b	b

Chapter 2

INTRODUCTION

Coastal marine seagrasses are experiencing decline worldwide as a result of human related activities and climate change (Orth et al. 2006; Waycott et al. 2009) Threshold levels of seagrass loss have been associated with declines in the abundance and species composition of faunal communities, particularly with respect to the distribution and abundance of juvenile finfish and shellfish (Anderson 1989; Brendan & Kevin 2006; Heck et al. 2003; McCloskey & Unsworth 2015). In coastal estuarine systems, many species rely on seagrass beds as a nursery habitat and refuge, using the foundation species for recruitment, settlement, foraging, and protection from predation. It is widely known that these structurally complex ecosystem engineers consistently have higher abundances and diversity of organisms than nearby un-vegetated habitats (Orth et al. 1984a). Seagrass beds support higher trophic food webs through intense and diverse secondary productivity and can potentially act as a determinant of fish yield (Duffy et al. 2003).

Fauna assemblages (e.g community composition) and the nature of these communities (e.g. predator-prey relationships) can differ depending on a multitude of factors including, but not limited to, the source of recruitment populations, water quality characteristics, food abundance, protection from predation, and habitat complexity (Martin & Cooper 1981; Orth et al. 1984b; Stoner 1983). The species of seagrass present can play a large role in determining the faunal community assemblage, although the mechanism for these differences can be variable by location. For example, Martin & Cooper (1981) examined the fish fauna communities inhabiting two structurally distinct seagrass species, *Thalasia testudinum* and *Syringodium filliforme,* and found that the fauna associated with each seagrass species were highly dissimilar, although the mechanisms driving these differences were only speculated upon. Contrary to these results, Ray et al. (2014) found that different morphologies (between *Syringodium filiforme, Thalassia testudinum, and Halodule wrightii)* accounted for very little variation in fauna abundances, while epiphyte volume had a significant impact on the abundance of organisms such as fish and blue crabs. The supply of larvae for nursery and marine species in seagrass beds as well as temperature and salinity tolerances, however, are also critical in determining fauna abundances and distribution (Bell et al. 1988; Eckman 1987).

There are two seagrass species found in Great South Bay (GSB), NY, *Zostera marina* and *Ruppia maritima*, which differ in their morphology, structural complexity, growth dynamics, and physical tolerances. *Z. marina* is characterized by ribbon-shaped leaves up to 1 cm wide that can extend over 2 m into the water column. *R. maritima* vegetative shoots are much smaller, generally 1 mm wide and typically do not extend more than 20 cm into the water column (Kantrud 1991). During the warm summer reproductive season however, *R. maritima* reproductive shoots can be up to 3 m long, creating a dense canopy (Kantrud 1991). While these are general characteristics of both species, light limitation in GSB restricts the stature of these species. While *Z. marina* can have a very extensive and dense root mat, almost 90% of *R. maritima's* belowground biomass lies within the upper 5 cm of the bottom substrate. The two species generally inhabit distinct depth profiles in GSB, and typically do not directly compete for light and resources, although there are a few sites where they simultaneously occur.

Great South Bay once played a large role in commercially and recreationally important fisheries such as the hard clam (*Mercenaria mercenaria*), the eastern oyster (*Crassostrea virginica*), striped bass (*Morone saxatilis*) and winter flounder (*Pleuronectes americanus*). The annual harvest of hard clams plummeted during the 1970's and 1980's, leaving only 1% of historical populations today. Studies show that *M. mercenaria* can exert control over phytoplankton densities in the water column, suggesting that GSB may have been more resistant to brown tide (*Aureococcus anophagefferens*) events when populations of the filter feeding shellfish were at their historical abundances (Cerrato et al. 2004). This once benthic production dominated system, in which seagrasses were the dominant primary producer, has long been altered by stressors such as overfishing, deteriorating water quality due to wastewater systems, and urbanization of surrounding watersheds.

Since GSB is enclosed by a barrier island to the south (known as Fire Island), tidal inlets exert a large control over this fragile estuarine system. Barrier islands are heavily influenced by tides, currents, waves, and winds. Therefore, intense stochastic events such as hurricanes can have a major impact on not only the barrier island, but the back-bay systems as well, in which the majority of seagrass beds are found. Hurricane Sandy hit the south shore of Long Island, NY in October of 2012, creating a new inlet through Fire Island (see chapter 1; Fig. 2). Although certain water quality characteristics around the breach area have improved since the breach occurred (predominantly an increase in salinity, and a decrease in temperature), seagrass beds overall underwent large declines in the region closest to the breach (within a 4.5 km radius from the breach, zone 1; see chapter 1).

The primary focus of this study was to quantify the impact of a new inlet on faunal utilization within *R. maritima* and *Z. marina* habitats in GSB. The main objective was to identify patterns in seagrass-associated organisms within the context of changing habitat complexity and changing water quality (i.e. to investigate potential species relationships across a range of taxa and organism sizes). Specifically, the faunal utilization of *R. maritima* and *Z. marina* habitats in close proximity and at distance from the breach will be assessed. The community composition of seagrass associated fauna at a sampling site directly impacted by the breach as a function of time will also be examined. It is expected that declines in seagrass coverage near the breach will heavily impact the abundance and diversity of resident faunal communities. Given the significantly greater habitat complexity provided by *Z. marina* and the impact of the breach on seagrass coverage, it is assumed that there will be a dramatic effect on fauna abundance, composition, and utilization of seagrass species.

METHODS

This study includes data from the Northeast Coastal Barrier Network Fire Island National Seashore Hurricane Sandy Assessment Study as well as data obtained from a 1996 fauna sampling study conducted in GSB by Raposa & Oviatt (2000).

Site description

Great South Bay consists of 42 km of the barrier island known as Fire Island, including ~35 km^2 of submerged estuarine land which extends 1.2 km north from the bay shoreline of the barrier island. This study was concentrated predominantly in the eastern region of Great South Bay (GSB) and western region of Moriches Bay. The average depth of GSB and MB are 1.3 and 0.9 meters respectively (Wilson et al. 1991).

Four sites bracketing the breach, were chosen based on the locations of previous fauna sampling completed in 1996, in which similar methods were used to determine the effects of shoreline type on nekton communities in seagrass beds (Raposa & Oviatt 2000). The four sites, from west to east, are Watch Hill (WH), Robinson Cove (RC), Old Inlet (OI), and Pattersquash (PS) (Fig. 1). Prior to Hurricane Sandy, WH was characterized by a continuous ~0.23 km^2 *R. maritima* bed, RC was characterized by a large but patchy *R. maritima* bed, OI was dominated by patchy *Z. marina* and PS was characterized by an extensive continuous *Z. marina* bed, with *R. maritima* beds extending into the shallows (pers. obs.).

Field sampling

A large-scale nekton sampling study was conducted in proximity and at distance to the breach in the summer (July-Sept) of 2013 through 2016. Two nekton sampling techniques (otter trawls and throw traps) were employed during the sampling index period at the four locations bracketing the breach in GSB/MB (Table 1). Old Inlet is located within 4.5 km from the breach (zone 1; see chapter 1), RC and WH are located within 9 km to the west of the breach (zone 2W), and PS is located within 9 km to the east of the breach (zone 2E; see chapter 1). The throw trapping technique followed protocols used by Raposa & Oviatt (2000) in which they sampled nearly identical sites with identical sampling gear examining the influences of shoreline type and vegetative biomass on the nekton community in GSB/MB (Raposa & Oviatt 2000).

Replicate benthic otter trawling using a 4.9-m (16-ft) trawl with 30.48-m (100-ft) bridals and 0.61 x 0.91 m (2 x 3 ft) otter doors was conducted for 5 min tows in each seagrass habitat where water depths allowed (> 1 m; n=4). Precise track lines from each trawl were recorded using a Garmin 76CSx. Since trawl speeds and gape dimensions are a function of bottom type and bathymetry and often change significantly within any single trawling attempt, catch per unit effort (CPUE) was calculated as the number of organisms per unit trawl length, as measured in m. The dominant habitat type (*R. maritima*, *Z. marina*, Mixed, and Unvegetated) was visually identified at each site during each year of sampling. Due to poor visibility and a lack of aerial imagery, overall seagrass bed area was not quantified.

Replicate sampling and dip netting of a 1 m^2 area throw trap within a given habitat (n=4 for each seagrass species where present) was performed. The throw trap was made from aluminum bars with 6 mm diameter wire mesh surrounding the sides, leaving the top and bottom open. After a few moments of standing quietly, the trap was thrown into a seagrass bed and quickly pushed into the sediment. A dip net was used to remove all fauna from the trap, and the trap was considered empty after three consecutive sweeps with no organisms. The GPS coordinates of each throw trap location were recorded using a Garmin 76CSx. Before collection, percent seagrass cover within the throw trap was recorded. All macroalgae present in the dip net was taken to the lab to determine biomass along with a biocore (23 cm diameter) of the seagrass from outside, but adjacent to, the trap. The above- and below-ground plant biomass was washed of sediment inside of a 1 mm sieve and the retained material was placed into a plastic bag, transported on ice to the lab where shoot density and biomass were determined.

Captured organisms from each sampling strategy were separated by species and identified to the lowest practical taxa and enumerated. Thirty individuals of each species were processed on site for (1) total length and biomass (using Pesola® spring scales) for fish species or (2) sex (‘male’, ‘female’, ‘juvenile’) and carapace length for crab species. Very small species that occur in abundances beyond the capacity of the crew to enumerate on site were estimated by selecting a random subset of 30 individuals that were individually measured and weighed in aggregate – individual weights were then determined by dividing the 30 individuals by the overall weight. All remaining specimens were enumerated and recorded without total length or biomass estimates.

Statistical analysis

To determine whether there were spatial or temporal differences in fauna abundance and total biomass, multiple one-way analyses of variance (ANOVA) and Tukey's HSD tests for multiple comparisons were performed. The assumption of normality was tested using the Shapiro Wilk test and homogeneity of variances was tested using the Levene's test. When data failed to meet these assumptions, appropriate transformations were made. Shannon diversity (H') was calculated for throw trap fauna abundance at each site. A one-way ANOVA of diversity as a function of site and species of seagrass was used to understand differences in diversity between years and sites; data were transformed where necessary to meet assumptions of normality. A non-parametric distribution free method was used to analyze changes in community composition at OI over time. Fauna abundance data were fourth root transformed and a Bray-Curtis dissimilarity matrix was created for differences among species of seagrass between sampling years using the software PRIMER-6 (Clarke & Gorley 2006). The effect of year was tested using a one-way analysis of similarity (ANOSIM) along with pairwise comparisons (Clarke & Gorley 2006). Due to small sample sizes, Clarke & Gorley (2006) suggest interpreting the ANOSIM test statistic R (the absolute measure of difference between groups in the 2-dimentional space) instead of *p* values (as long as the global *p* value is significant). An R value close to 1 indicates a high amount of separation between the community composition, while a number close to 0 indicates there are no differences in community composition between test groups. ANOSIM was followed by similarity percentage analyses (SIMPER) to identify the percent contribution of individual species that lead to differences in nekton communities. Community composition in *Z. marina* from trawls was also examined as a function of site in 2016 using the same method as described above.

RESULTS

Site Characteristic Over Time

The ability to sample the fauna present within each species of seagrass was heavily dependent on depth and year (refer to Table 1 for the sampling techniques employed at each site and year). The seagrass bed at PS remained relatively constant over the course of the sampling period, with a *Z. marina* bed located in deeper water (>1m) and a *R. maritima* bed in shallower water (<1m). The most dramatic change in seagrass over time occurred at OI. In 2013, the trawling location consisted of sand. In 2014 *R. maritima* appeared with very few *Z. marina* shoots interspersed in between. In 2015, an even mix of *R. maritima* and *Z. marina* was apparent, and in 2016 a clear distinction between species within the bed was observed, with *Z. marina* displacing *R. maritima* towards the shallower, warmer edge of the bed. The *R. maritima* bed at RC remained constant in terms of coverage and patchiness over the entirety of the sampling period. In 2016, a newly colonized *Z. marina* bed was observed in >1-m deep. The *R. maritima* bed at WH changed over the course of the study. In 2013, the seagrass bed covering 0.23 km^2 was dominated by *R. maritima*. In 2016, the seagrass bed was > 90% *Z. marina* (pers. obs.). It is unclear when this change occurred.

Fauna Abundances and Biomass (Throw Traps)

Mean fauna abundance was significantly different at OI in 2016 as compared to 2013 ($F_{(3)} = 4.964$, p=0.007; Fig. 2A). This difference was mainly driven by an increase in shrimp abundances (primarily *Palaemonetes pugio, Hippolyte spp., and Crangon septemspinosa*) and by an increase in the abundance of blue crabs (*Callinectes sapidus*). Shrimp abundances in 2013 averaged 1.6 individuals/m^2 (hereafter ind/m^2), annually increasing to ~20 ind/m^2 in 2016 (see Appendix 1A). Of those species, *P. pugio* saw the largest increase in abundance. Changes in the mean abundance of *C. sapidus,* were also apparent, increasing from ~1 ind/m^2 in 2013 to 5 ind/m^2 in 2016. Fauna abundances at PS were significantly lower in 2014 than in any other year ($F_{(3)}$ = 8.522, $p < 0.001$; Fig. 2A). This decline was driven by the loss of *P. pugio* with only an average of 7 ind/m^2 in 2014 as compared to 82 ind/m^2, 64 ind/m^2, and 154 ind/m^2 in 2013, 2015, and 2016 respectively (see Appendix 1A). A significant increase in juvenile *C. sapidus* abundance (< 50mm CW) in 2016 was observed at all sites except OI. The mean abundance of juveniles at WH, RC, and PS was 22.8, 15.0, and 19.9 ind/m^2 respectively, while the mean abundance of juveniles at OI was 3 ind/m^2 (Fig. 3). The distribution of juvenile *C. sapidus* in seagrass was species-specific, with 99.5%, 91%, and 73% of juveniles inhabiting *R. maritima* as opposed to *Z. marina* at WH, RC, and PS respectively.

Mean biomass indicate no changes at RC over time. Biomass at WH was significantly different in 2016 as compared to 2013 and 2014 ($F_{(3)} = 5.421, p = 0.007$; Fig. 2C), driven not only by an overall doubling of species richness, but also in large part to a significant increase in *C. sapidus* abundances. Biomass was significantly different in 2016 at PS than every year prior ($F_{(3)} = 9.154, p < 0.001$; Fig. 2C), due to increases in *C. sapidus* and *P. pugio.* The only significant difference in biomass at OI occurred between 2015 and 2016 due to increases in *C. sapidus* ($F_{(3)} = 3.959, p = 0.018$; Fig. 2C).

Fauna Abundances and Biomass (Trawls)

Fauna abundances at WH, RC, and PS, did not change over time. At OI overall mean fauna abundances were significantly higher in 2014, 2015, and 2016 as compared to 2013 ($F_{(3)} = 9.499$, $p < 0.001$; Fig. 2B). This increase was primarily driven by an increase in the overall fauna community, with more than twice the species richness from 2014 to 2016 (species counts = 13, 27, and 19 in 2014, 2015, and 2016 respectively) as compared to 2013 (count = 6). This change in fauna was further intensified by annual increases in the Atlantic silverside (*Menidia menidia),* the blue crab *(C. sapidus),* and northern pipefish (*Syngnathus fuscus*).

Fauna biomass was not significantly different by year at PS. Significantly higher biomass at WH in 2016 ($F_{(3)} = 5.631, p = 0.007$; Fig. 2D) was mainly driven by *C. sapidus.* Biomass was significantly different in 2013 compared to 2016 at RC, driven primarily by increases in *C. sapidus* ($F_{(3)} = 3.397, p = 0.036$; Fig. 2D). Biomass at OI in 2013 was significantly different than every year after ($F_{(3)} = 9.007, p < 0.001$; Fig. 2D). This difference was driven primarily by increases in the entire fauna community, and secondarily by increases in *C. sapidus.*

Comparison of Fauna Abundance to 1996 data

Abundance data from throw traps completed in *Z. marina* in 1996 (Raposa & Oviatt 2000) were compared to throw traps at RC (2016) and OI (2013-2016). Due to missing raw data, values from 1996 are means of 5 to 10 replicates. Overall fauna abundance was significantly higher at RC in 1996 than in 2016 ($F_{(4)} = 33.16, p < 0.001$; Fig. 4) due to higher abundances of three shrimp species *C. septemspinosa* (52% of total abundance), *Hippolyte spp.* (17%), and *P.pugio* (11%). In 2016, the dominant species were *Hippolyte spp.* (26%), *C. sapidus* (20%), *S. fuscus* (16%), and *M. menidia* (16%). Fauna abundance in 1996 at OI was significantly higher than 2013, 2014, and 2015, but not significantly different than 2016. Fauna abundance in 1996 was dominated by *Apeltes quadracus* (33%), *C. septemspinosa* (26%), and *Hippolyte spp.* (11%). In 2016, *Hippolyte spp.* and *C. septemspinosa* were also two of the most dominant species, however the third most dominant species in 2016 was *C. sapidus.*

Multivariate Community Analysis and Diversity

The community assemblages of throw traps at OI by species of seagrass and year (including data from Raposa & Oviatt (2000)) were compared using nMDS ordination (stress 0.17; Fig. 5). ANOSIM revealed that the community composition by species of seagrass and year were significantly different (global R= 0.558, global $p = 0.001$). Community composition in *R. maritima* differed significantly between years (one-way ANOSIM, R > 0.78 for all pairwise comparisons) and the similarity between replicates was high every year (SIMPER average similarity between replicates > 70%). This similarity between replicates was not evident in the fauna communities inhabiting *Z. marina* however. The community composition was extremely variable in 2013 (< 20% similarity between replicates), and this variability decreased annually until 2016, in which variability between replicate throw traps was very low (~80% similarity between replicates). The nMDS ordination revealed that the community composition in *Z. marina* in 1996 was significantly different than the *Z. marina* community composition in 2016 at OI (one-way ANOSIM, R=1, Fig. 5). These differences were largely driven by the absence of the four-spined stickleback (*Apeltes quadracus*), the rainwater killifish (*Lucania parva*), and the mummichog (*Fundulus hetercolitus*) in 2016. Other differences include the presence of scup (*Stenotomus chrysops*) and black sea bass (*Centropristis striata*), both of which were not present in 1996, and increased abundances of blue crabs (*C. sapidus*) in 2016 as compared to 1996.

Fauna community composition using trawl data was only examined for 2016 (stress 0.18; Fig.6) since it is the only year in which *Z. marina* was present and sampled at every site. ANOSIM reveals that the community composition at PS is significantly different than RC, WH, and OI (one-way ANOSIM, R > 0.82 for all pairwise comparisons; Fig. 6). These differences were partially driven by an order of magnitude higher abundances of the three-spined stickleback (*Gasterosteus* *aculeatus),* the spider crab (*Libinia sp.*), the northern pipefish (*S. fuscus),* and the Atlantic silverside (*M. menidia*) at PS as compared to the other three sites*,* as well as the presence of the four-spined stickleback (*A. quadracus*) only in otter trawls completed at PS. It is also apparent that the community composition at RC in *R. maritima* is significantly different as compared to *Z. marina* (one-way ANOSIM, R = 0.89; Fig. 6). There appears to be no significant differences in community composition between OI and the sites to the west of the new inlet in 2016 (WH and RC; one-way ANOSIM, $R < 0.38$ for pairwise comparisons; Fig. 6).

Overall, there was no difference in Shannon diversity in *R. maritima* beds over time at WH, OI, or PS. An increase in diversity was observed in *R. maritima* in 2016 at RC, with diversity indices similar to those found at the other three sites that year (one-way ANOVA, $F_{(3)}= 13.05$, $p < 0.001$; Table 2). Although fauna abundance in RC in 1996 was significantly higher than in 2016 and there is a significant difference in community composition, there is no difference in fauna diversity between the two years (H' = 1.32 ± 0.36 (1996), 1.33 ± 0.20 (2016)). Diversity at OI in 1996 was not different than 2014, 2015 nor 2016, but it was significantly higher than 2013 (one-way ANOVA, $F_{(3)}= 5.024$, $p = 0.010$; Table 2). The community composition at OI in 2013 had not only the lowest measure of diversity but also the highest variability in diversity than at any other site or any other year sampled (H' = 0.61 ± 0.72; Table 2A). The diversity of the community at PS in *Z. marina* was the most variable between years.

DISCUSSION

Coastal marine ecosystems are experiencing deleterious impacts from many different anthropogenic and climatic sources. Hurricanes and tropical storms are common on the east coast of the United States and often have large impacts on dynamic barrier islands for example Hurricane Isabel on the Outer Banks of North Carolina and Hurricane Katrina on Dauphin Island, Alabama. The effects of habitat loss on fauna communities as a result of disturbances has been previously examined (Brendan & Kevin 2006; Hughes et al. 2002). In this study, we examined temporal and spatial differences in seagrass-associated faunal species as a function of a major disturbance event and the proximity to a new oceanic inlet.

The ecological succession of species after a disturbance event has been widely studied across a multitude of environments and taxa. After a disturbance event, bare space is typically colonized by vegetation that is rapidly growing, opportunistic, and highly reproductive, followed by sequential replacements of species with slower growth rates and lower dispersal capabilities (see McCook et al. (1994) for a review). This study presents an example of seagrass recovery and community succession after Hurricane Sandy, a major disturbance event. In 2014, two years post-Sandy, *R. maritima* was the first to re-colonize an unvegetated area near a disturbance site (OI). Sexual reproduction is very important for the colonization and expansion of this opportunistic species (Kantrud 1991). In 2015, the appearance of *Z. marina* was observed, however there was an even mix between the fast-growing *R. maritima* and the slow-growing *Z. marina.* In 2016, *Z. marina* seems to have displaced the pioneer species *R. maritima* towards the shallows (<1 m), establishing a monospecific bed at the deep edge. The future monitoring of this seagrass bed, which has been highly impacted by changing water quality conditions and sediment movement (see chapter 1), will be important to determine how fauna will respond to changing seagrass distributions.

Fauna sampling results with regard to the increasing seagrass bed at OI are consistent with the literature, in which a significantly lower fauna abundance was observed in 2013 (year with little to no seagrass cover). Abundance increased significantly from 2013 to 2016 (Fig. 2), in which beds of *Z. marina* and *R. maritima* were established. Community composition in 2013 at OI was dominated by highly mobile predators (e.g. scup (*Stenotomus* chrysops), bluefish (*Pomatomus saltatrix*), and gag grouper (*Mycteroperca microlepis*)), presumably moving in and out of the new inlet as opposed to resident species. Increases in nursery and resident species, such as the pipefish, tautog (*Tautoga onitis*), and blue crab, were evident during the following years. Raposa & Oviatt (2000) found similar results, in which abundances of the resident pipefish (*S. fuscus*), a species heavily dependent on seagrass for camouflage and foraging, were positively correlated to increases in *Z. marina* biomass.

Site-specific characteristics may influence the abundance and biomass of the fauna community inhabiting seagrass beds. For example, the highest mean abundance of *P. pugio* (135 ind/m^2 was observed in *R. maritima* beds closest to the salt marsh shoreline (< 40 m; PS). Mean abundances at OI were much lower in comparison (22.625 ind/m^2), in which *R. maritima* beds are much farther from salt marsh shorelines (>270 m). These results were consistent with data from 1996 in GSB in which *P. pugio* were more abundant in *Z. marina* beds near the salt marsh shoreline (10 m) than at a distance (300 m). Another potential explanation for these differences may include differences in predation as a result of the new inlet.

While pre-breach data were limited, a change in the fauna community composition was observed after a 20-year period (Fig. 4; 1996 to 2016). Interestingly, in 1996 when large *Z. marina* meadows were present in GSB, the species richness from throw trapping was half that of what it is today. The 16 species caught between 2013 and 2016 (not found in 1996) were comprised of ~70 percent resident and nursery species, while the other 30% are marine or tropical species. The change in the fauna community composition could potentially be a response to overall climate changes, particularly poleward distribution shifts in response to increased temperatures (e.g. back sea bass and scup (Bell et al. 2015; Nye et al. 2009; Parmesan & Yohe 2003; Walther et al. 2002). Other factors could include changes associated with the inter-annual variability of warm Gulf Stream eddies (in which many tropical species are transported), changes to the seagrass landscape (in which GSB has lost a majority of the seagrass once present over 20 years ago), and a new oceanic source of fauna. While it is not possible to directly correlate the appearance of marine and tropical species to the formation of the inlet (no recent pre-breach data are available), it is likely that this new source of offshore fauna can be influencing the community composition, structure, and predator-prey relationships in estuarine seagrass beds.

Larger mobile predators such as striped bass (*Morone saxatilis*), tautog (*Tautoga onitis*), bluefish (*P. saltatrix*), scup (*S.* chrysops), and black sea bass (*Centropristis striata*) entering GSB through the new inlet can have potential influences on the smaller prey species inhabiting these seagrass beds, such as shrimp and juvenile blue crabs (Orth et al. 1999). From the throw trap fauna data, it is clear that a significantly lower abundance of juvenile blue crabs was present in *R. maritima* beds at OI as compared to the other three sites, which are each at least 4.5 km from the nearest inlet ($p < 0.001$, Fig. 3). It is possible that predators entering GSB through the breach are the cause of this decline; however, there are insufficient data to provide evidence of this relationship. Not only was a spatial relationship apparent with respect to the site distribution of juvenile blue crabs, there was also a habitat specific relationship, in which juveniles were more significantly abundant within *R. maritima* beds, as opposed to *Z. marina* ($p < 0.001$; with the exception of OI, $p = 0.900$; Fig 3). Increased abundances in *R. maritima* could be due to higher shoot densities within *R. maritima*, increased predation in *Z. marina* (perhaps related to lower shoot density), or selective settlement. These results correlate with previous studies in which juvenile blue crabs (> 9 mm CW) were significantly more abundant in *R. maritima* as compared to *Z. marina* in Chesapeake Bay (Pardieck et al. 1999).

Overall fauna abundance data for both throw traps and otter trawls suggest that the influence of the new inlet on GSB was restricted to the nearest sampling site (OI; Fig 2A and B). The significantly lower abundance of juvenile blue crabs at OI as compared to WH, RC, and PS also suggests a restricted influence of incoming predators on resident nekton communities. Overall fauna biomass data shows an increase in 2016 across all sites, however, this difference was primarily driven by an increase in blue crab abundance and therefore biomass. While trends in abundance data and biomass data of rarer species are drowned out by high abundances of shrimp and high biomass of blue crabs, trends in other species away from the breach were apparent. For example, the tautog (*T. onitis*) increased in mean biomass at WH over time (0.15 g/m^2 in 2013 to 4.58 g/m^2 in 2016), while the northern pipefish increased in abundance over time at every site across GSB except for PS. While abundances of tropical and marine species (e.g the spotfin butterflyfish (*Chaetodon ocellatus*), the striped burrfish (*Chilomycterus schoepfi*), and the gag grouper (*Mycteroperca microlepis*)) were low and highly variable, their presence was nonetheless evident at even the site furthest from the breach (WH; ~ 9 km). Due to replication limitations, however, there is insufficient data to determine the spatial extent of the influence of the breach on these mobile predators.

The overall understanding of fauna abundance, biomass, and habitat utilization is influenced by sampling limitations. Our study was limited by depth, in which shallow (<1 m) seagrass beds (typically *R. maritima*) were unable to be sampled using an otter trawl. Some sites, including the *R. maritima* bed at OI and RC, were only able to be trawled at high tide. Due to the strong influence of tidal inlets on physical and biological aspects of this estuarine system, the tide state (ebb or flood) may play a large role in controlling the fauna present in seagrass beds at proximity to the breach. Marine species, such as scup, are considered 'visitors' to estuarine systems, in which spawning occurs offshore and juveniles and adults visit estuaries while foraging. To gain a more complete understanding of the influence of the offshore fauna community on this estuarine system and how extensive the influence of the breach is, more replicates would be needed at both tide states, ebb and flood, and at different times of the day, as these two factors may play a large role in the movement, direction, and schooling behavior of highly mobile predators (Becker et al. 2016). Despite the temporal and spatial sampling limitations in this study, we detected overall patterns in fauna abundance over time as a function of the proximity to a new oceanic inlet, as well as a change in the fauna community composition over a 20-year period.

Across the globe, there is a vast amount of literature depicting the decline of seagrass beds, and the subsequent effects on ecosystem function. The consequences of the loss of seagrass is evidenced by changes in fauna utilization over time, particularly those species that are heavily dependent on the habitat as a refuge from predation, as a nursery, and as a forging ground. This study presented a reversal in that declining trend, a recovery scenario, in which a major disturbance event lead to the decline in seagrass, a subsequent recovery, and the response of fauna as a result. The value of this foundation species is very clear, and effort toward restoration and conservation, while difficult, are of utmost importance.

LITERATURE CITED

Anderson EE. 1989. Economic Benefits of Habitat Restoration: Seagrass and the Virginia Hard-Shell Blue Crab Fishery. *North American Journal of Fisheries Management* 9:140-149.

Becker A, Holland M, Smith JA, and Suthers IM. 2016. Fish Movement Through an Estuary Mouth Is Related to Tidal Flow. *Estuaries and Coasts* 39:1199-1207.

Bell JD, Steffe AS, and Westoby M. 1988. Location of seagrass beds in estuaries: effects on associated fish and decapods. *Journal of Experimental Marine Biology and Ecology* 122:127-146.

Bell RJ, Richardson DE, Hare JA, Lynch PD, and Fratantoni PS. 2015. Disentangling the effects of climate, abundance, and size on the distribution of marine fish: an example based on four stocks from the Northeast US shelf. *ICES Journal of Marine Science* 72:1311-1322.

Brendan JR, and Kevin AH. 2006. Seagrass habitat disturbance: how loss and fragmentation of eelgrass Zostera marina influences epifaunal abundance and diversity. *Marine Ecology Progress Series* 326:133-143.

Cerrato RM, Caron DA, Lonsdale DJ, Rose JM, and Schaffner RA. 2004. Effect of the northern quahog Mercenaria mercenaria on the development of blooms of the brown tide alga Aureococcus anophagefferens. *Marine Ecology Progress Series* 281:93-108.

Clarke KR, and Gorley RN. 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E. Plymouth

Duffy JE, Richardson JP, and Canuel EA. 2003. Grazer diversity effects on ecosystem functioning in seagrass beds. . *Ecology Letters* 6:637-645.

Eckman JE. 1987. The role of hydrodynamics in recruitment, growth, and survival of Argopecten irradians (L.) and Anomia simplex (D'Orbigny) within eelgrass meadows. *Journal of Experimental Marine Biology and Ecology* 106:165-191.

Heck KL, Hays G, and Orth RJ. 2003. Critical evaluation of the nursery role hypothesis for seagrass meadows. *Marine Ecology Progress Series* 253:123-136.

Hughes JE, Deegan LA, Wyda JC, Weaver MJ, and Wright A. 2002. The Effects of Eelgrass Habitat Loss on Estuarine Fish Communities of Southern New England. *Estuaries* 25:235-249.

Kantrud HA. 1991. Wigeongrass (*Ruppia maritima* L.): a literature review. *US Fish and Wildlife Service Fish and Wildlife Research* 10:58.

Martin DF, and Cooper M. 1981. A comparison of fish faunas found in pure stands of two tropical atlantic seagrasses, *Thalassia testudinum* and *Syringodium filiforme Northeast Gulf Science* 5:31-37.

McCloskey RM, and Unsworth RK. 2015. Decreasing seagrass density negatively influences associated fauna. *PeerJ* 3:e1053.

McCook LJ. 1994. Understanding Ecological Community Succession: Causal Models and Theories, a Review. *Vegetatio* 110:115-147.

Nye JA, Link JS, Hare JA, and Overholtz WJ. 2009. Changing spatial distribution of fish stocks in relation to climate and population size on the Northeast United States continental shelf. *Marine Ecology Progress Series* 393:111-129.

Orth R, Heck K, and van Montfrans J. 1984a. Faunal communities in seagrass beds: A review of the influence of plant structure and prey characteristics on predator-prey relationships. *Estuaries* 7:339-350.

Orth RJ, Carruthers TJB, Dennison WC, Duarte CM, Fourqurean JW, Heck KL, Hughes AR, Kendrick GA, Kenworthy WJ, Olyarnik S, Short FT, Waycott M, and Williams SL. 2006. A Global Crisis for Seagrass Ecosystems. *Bioscience* 56:987-996.

Orth RJ, Heck KL, and van Montfrans J. 1984b. Faunal communities in seagrass beds: A review of the influence of plant structure and prey characteristics on predator-prey relationships. *Estuaries* 7:339-350.

Orth RJ, Montfrans Jv, and Fishman J. 1999. A preliminary study of predation on blue crabs by three fish predators in a seagrass bed. Virginia Marine Resources Commission. Report to the Virginia Marine Resources Commission. p 5.

Pardieck RA, Orth RJ, Diaz RJ, and Lipcius RN. 1999. Ontogenetic changes in habitat use by postlarvae and young juveniles of the blue crab. *Marine Ecology Progress Series* 186:227-238.

Parmesan C, and Yohe G. 2003. A globally coherent fingerprint of climate change impacts across natural systems. *Nature* 421:37-42.

Raposa KB, and Oviatt CA. 2000. The influence of contiguous shoreline type, distance from shore, and vegetation biomass on nekton community structure in eelgrass beds. *Estuaries* 23:46-55.

Ray BR, Johnson MW, Cammarata K, and Smee DL. 2014. Changes in Seagrass Species

Composition in Northwestern Gulf of Mexico Estuaries: Effects on Associated Seagrass Fauna. *PLOS ONE* 9:e107751.

Stoner AW. 1983. Distribution of fishes in seagrass meadows: role of macrophyte biomass and species composition. *Fisheries Bulletin* 84:837-846.

Walther G-R, Post E, Convey P, Menzel A, Parmesan C, Beebee TJC, Fromentin J-M, Hoegh-Guldberg O, and Bairlein F. 2002. Ecological responses to recent climate change. *Nature* 416:389-395.

Waycott M, Duarte CM, Carruthers TJB, Orth RJ, Dennison WC, Olyarnik S, Calladine A, Fourqurean JW, Heck KL, Hughes AR, Kendrick GA, Kenworthy WJ, Short FT, and Williams SL. 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences* 106:12377-12381.

Wilson R, Wong K-C, and Carter HH. 1991. Aspects of circulation and exchange in Great Sout Bay. In: *In:* Schubel JR, Bell TM, and Carter (eds.) HH, eds. *The Great South Bay*. Stony Brook, New York: Marine Sciences Research Center, 9-21.

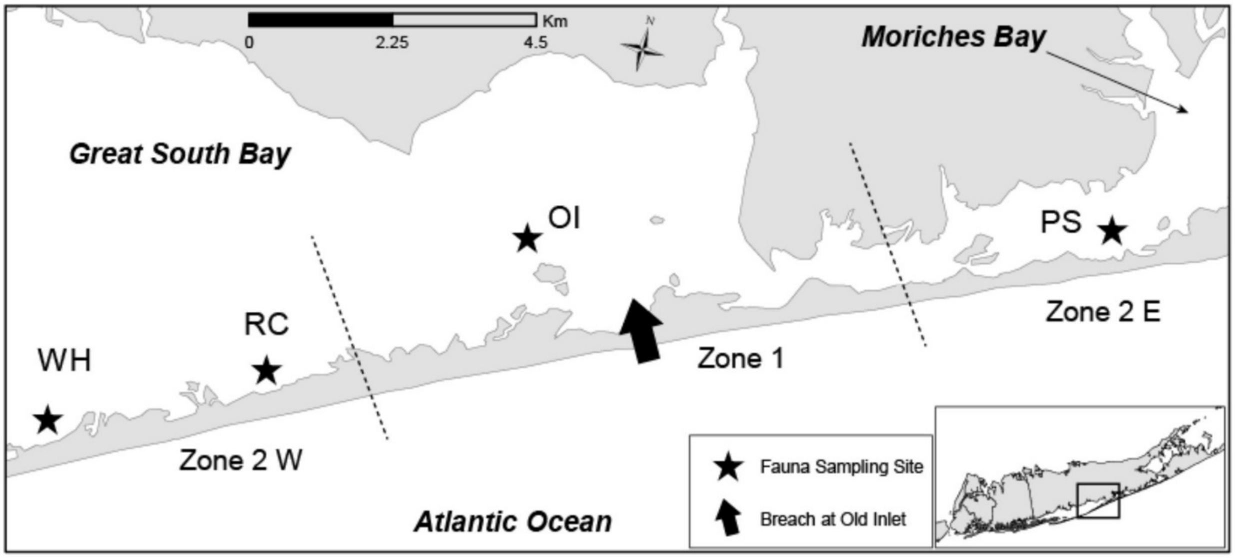


Figure 1. Location of fauna sampling stations in Great South Bay and Moriches Bay, NY (WH = Watch Hill, RC = Robinson Cove, OI = Old Inlet, PS= Pattersquash)

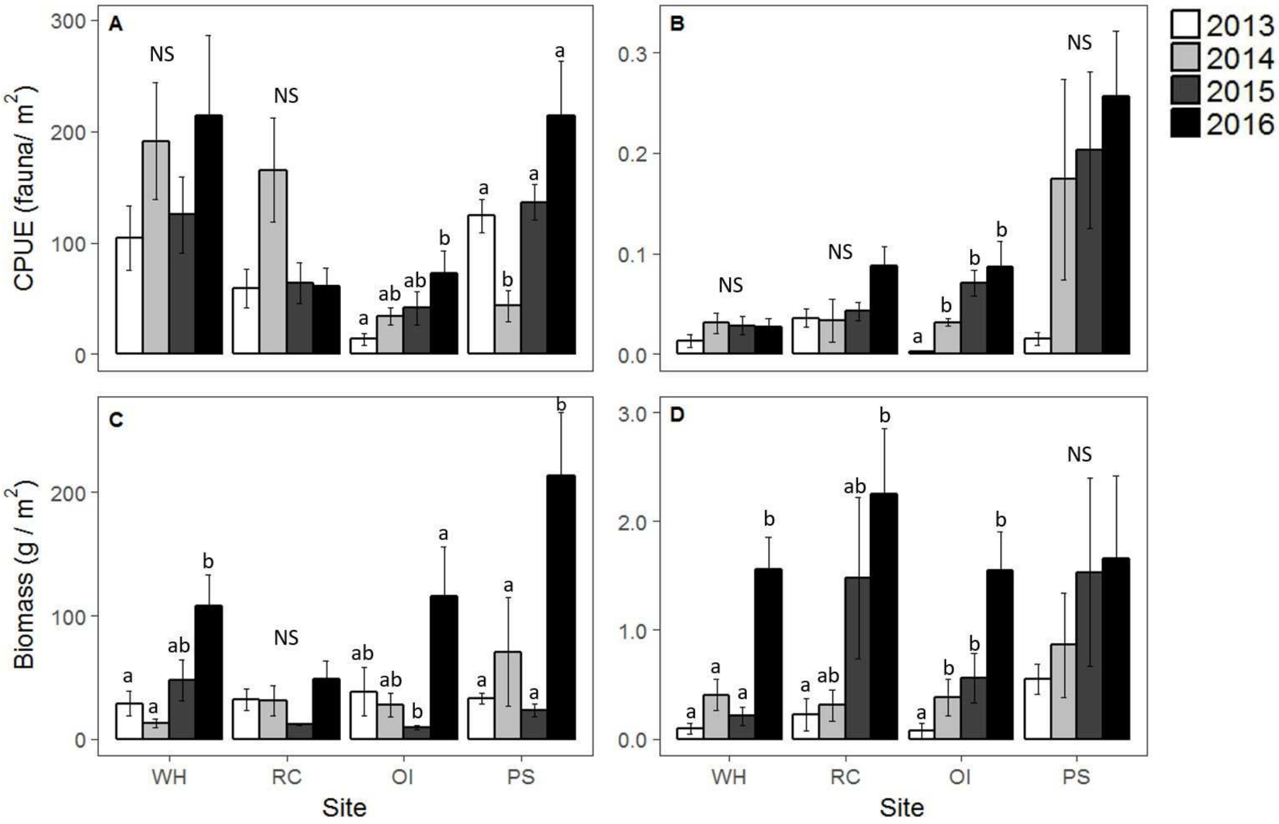


Figure 2. Mean abundance (±SE) and biomass (±SE) of fauna captured in either species of seagrass over time using throw traps (AC) and otter trawls (BD).

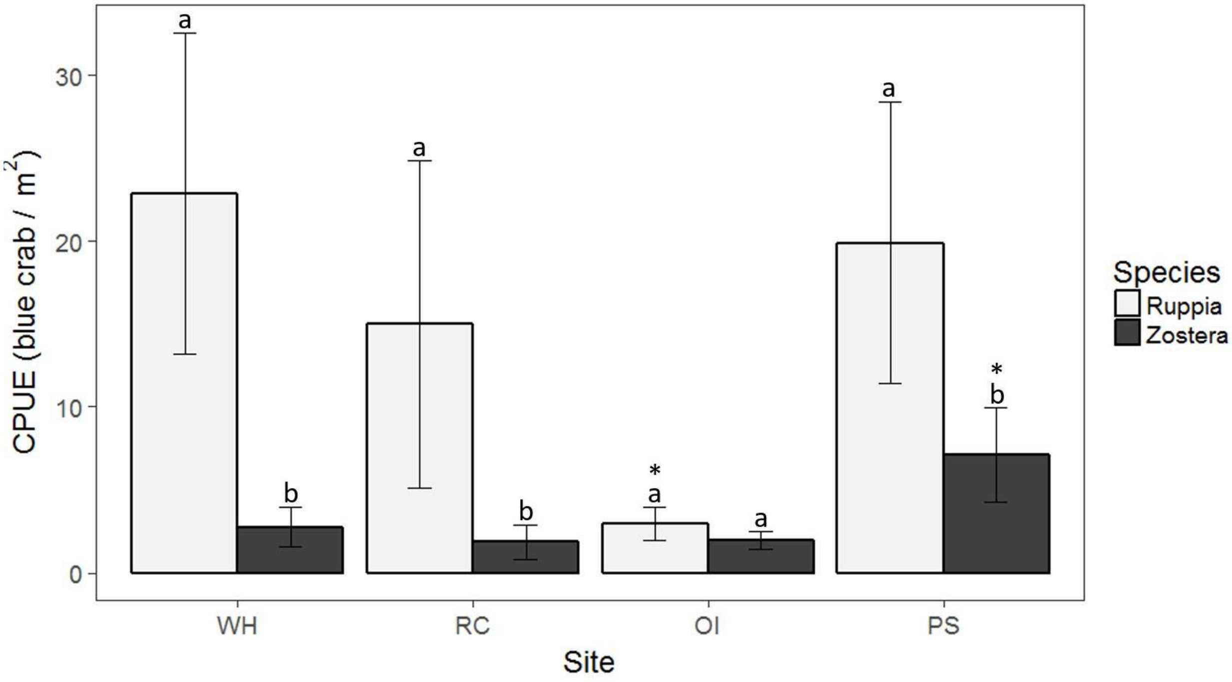


Figure 3. Mean abundance (±SE) of the blue crab (*C. sapidus*) as a function of site and species of seagrass in 2016 throw trap samples. Similar letters indicate no difference in the number of blue crabs /m^2 between species of seagrass within a single site. Asterisks indicate a significant difference in the number of blue crabs /m^2 between sites within a species of seagrass.

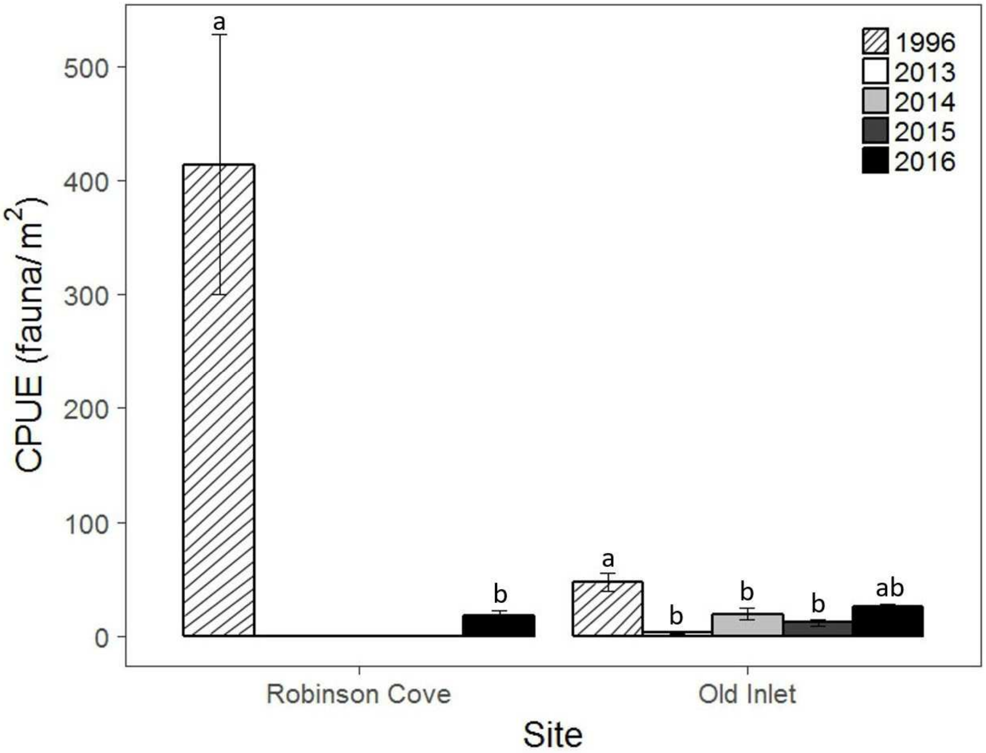


Figure 4. Mean abundance (±SE) of fauna as a function of site and year in *Z. marina* using throw traps. Similar letters indicate no significant difference in the fauna/m^2 between years within a site

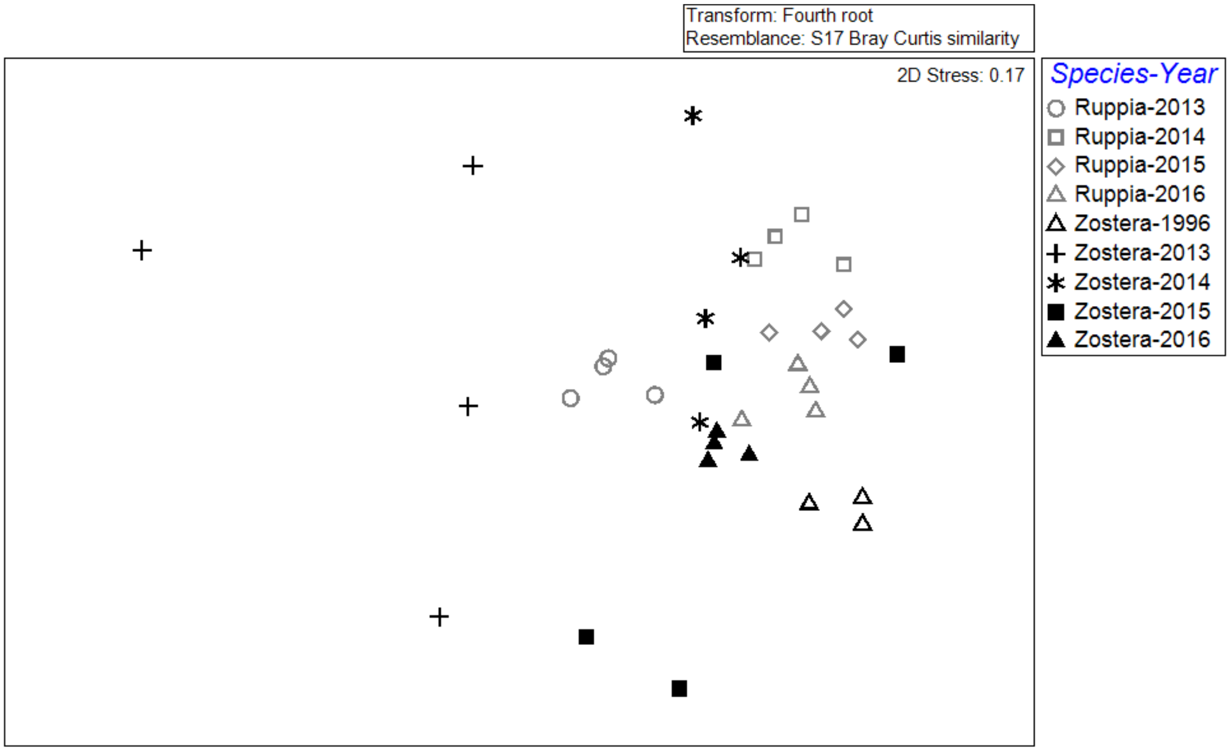


Figure 5. nMDS ordination of the fauna community composition at Old Inlet as a function of year and species of seagrass.

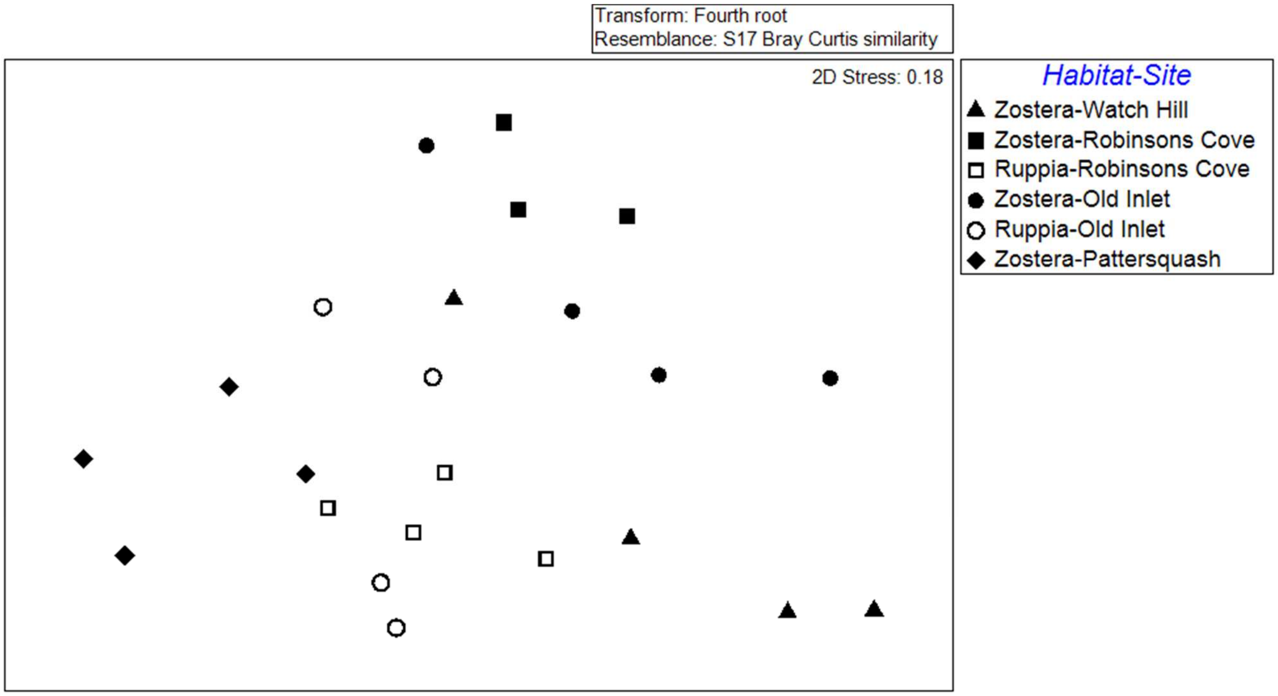


Figure 6. nMDS ordination of the fauna community composition in 2016 as a function of site and species of seagrass.

Table 1. Fauna sampling techniques (throw traps and otter trawls) employed at each site each year. Presence of a letter indicates the sampling technique was completed for that combination of year and species of seagrass. U = unvegetated, M= Mixed bed, R= *R. maritima*, Z= *Z. marina*. Years where both R and Z are present indicates the presence of two monospecific beds, one of each species, at that site.

Throw Trap	2013	2014	2015	2016
Watch Hill	R	R	R	RZ
Robinson Cove	R	R	R	RZ
Old Inlet	RZ	RZ	RZ	RZ
Pattersquash	RZ	RZ	RZ	RZ

Otter Trawl	2013	2014	2015	2016
Watch Hill	R	M	M	Z
Robinson Cove	R	M	M	Z
Old Inlet	U	R	M	RZ
Pattersquash	Z	Z	Z	Z

Table 2. Mean (±SE) Shannon diversity index (H′) values for each site, year, and species of seagrass combination (A) for throw traps. Similar letters (B) indicate no significant difference in diversity between years within site and species of seagrass. (-) indicates no throw traps were completed

A

Location	Species	1996	2013	2014	2015	2016
Watch Hill	*R. maritima*	-	1.00 ± 0.03	1.04 ± 0.12	1.12 ± 0.32	1.24 ± 0.26
	Z. marina	-	-	-	-	1.33 ± 0.15
Robinson Cove	*R. maritima*	-	0.93 ± 0.11	0.77 ± 0.11	0.63 ± 0.13	1.34 ± 0.27
	Z. marina	1.32 ± 0.36	-	-	-	1.33 ± 0.20
Old Inlet	*R. maritima*	-	1.5 ± 0.21	1.1 ± 0.38	1.28 ± 0.30	1.48 ± 0.17
	Z. marina	1.72 ± 0.19	0.61 ± 0.72	1.80 ± 0.27	1.22 ± 0.52	1.31 ± 0.11
Pattersquash	*R. maritima*	-	1.37 ± 0.10	1.15 ± 0.39	0.71 ± 0.41	0.83 ± 0.30
	Z. marina	-	0.81 ± 0.11	1.35 ± 0.22	0.67 ± 0.26	1.19 ± 0.37

B

Location	Species	1996	2013	2014	2015	2016
Watch Hill	*R. maritima*	-	a	a	a	a
	Z. marina	-	-	-	-	-
Robinson Cove	*R. maritima*	-	a	a	a	b
	Z. marina	a	-	-	-	a
Old Inlet	*R. maritima*	-	a	a	a	a
	Z. marina	a	b	a	ab	ab
Pattersquash	*R. maritima*	-	a	a	a	a
	Z. marina	-	a	b	ac	abc

Appendix I

Species captured in *R. maritima* and *Z. marina* using throw traps. Species listed totaled at least 1% of the catch at a site during any year of study.

Appendix I-A. Mean abundance (individuals / m^2)

Site	Species	1996	2013	2014	2015	2016
Watch Hill	*Anchoa mitchilli*	18.00				0.13
	Apeltes quadracus	16.50				0.25
	Callinectes sapidus	6.50	1.75	2.00	7.25	25.63
	Crangon septemspinosa	300.00	57.50	157.75	22.00	7.25
	Dyspanopeus sayi		0.50	0.50	5.50	1.00
	Hippolyte sp.	144.00	1.00	6.25	4.00	128.13
	Menidia menidia	6.50			0.25	0.13
	Pagurus longicarpus		1.75	7.00	3.75	3.63
	Palaemonetes pugio	46.50	67.00	62.25	110.00	42.13
	Syngnathus fuscus	2.00	0.75	3.25	3.00	3.75
Robinson Cove	*Anchoa mitchilli*	18.55	4.50			
	Apeltes quadracus	27.47				0.13
	Callinectes sapidus	3.05	49.25	2.25		16.88
	Crangon septemspinosa	182.97	0.25	189.75	2.50	10.00
	Dyspanopeus sayi			0.50	0.25	0.88
	Gobiosoma sp.	5.24	4.50			
	Hippolyte sp.	60.11		0.25	8.25	14.75
	Menidia menidia	2.52	14.75			1.63
	Palaemonetes pugio	42.12		4.00	67.00	12.75
	Pagurus longicarpus		0.25	8.25		1.00
	Syngnathus fuscus	5.48		1.50	2.25	2.13
Old Inlet	*Anchoa mitchilli*	3.83			0.13	
	Apeltes quadracus	15.77	1.13			0.13
	Callinectes sapidus	0.93		3.63	1.25	5.00
	Centropristis.striata		3.63	2.13	0.13	0.13
	Crangon septemspinosa	12.13		9.75	16.56	16.63
	Dyspanopeus sayi			3.88	3.38	0.75
	Fundulus.heteroclitus	1.60				
	Hippolyte sp.	5.17		0.38	4.81	17.13
	Libinia sp.			0.63	0.13	0.25
	Menidia menidia	1.97	0.13	0.88	0.13	0.88
	Pagurus longicarpus			2.50	4.00	1.00
	Palaemonetes pugio	4.67	4.75	6.38	9.00	26.63
	Syngnathus fuscus	0.63	2.25	3.25	1.51	3.38
	Tautoga onitis		0.88		0.25	
	Tautogolabrus adspersus		0.63	0.25		
Pattersquash	*Callinectes sapidus*		1.13	1.38	0.75	27.00
	Crangon septemspinosa		21.50	5.63	42.50	4.13
	Dyspanopeus sayi		7.50	2.13	5.63	17.00
	Gasterosteus aculeatus			0.13	2.50	1.13
	Hippolyte sp.		6.88	0.63	16.75	3.50
	Menidia menidia		0.13	20.38	1.63	0.63

Pagurus longicarpus	1.88	2.38	1.13	0.75
Palaemonetes pugio	82.50	7.38	63.88	154.00
Syngnathus fuscus	1.88	2.75	1.88	3.63

Appendix I-B. Mean biomass (g / m^2)

Site	Species	2013	2014	2015	2016
Watch Hill	*Callinectes sapidus*	7.25	3.88	24.34	88.53
	Crangon septemspinosa	14.18	7.00	1.63	0.50
	Hippolyte sp.	0.05	0.33	0.28	2.56
	Menidia menidia			0.03	0.06
	Opsanus tau				1.54
	Palaemonetes pugio	14.35	3.90	25.84	8.20
	Penaeus sp.			1.63	
	Syngnathus fuscus	0.28	0.79	2.85	3.50
	Tautoga onitis			2.03	1.88
Robinson Cove	*Callinectes sapidus*		14.21		38.02
	Crangon septemspinosa	14.96	15.75	0.18	0.52
	Hippolyte sp.	2.13	0.03	0.68	0.61
	Menidia menidia				3.19
	Opsanus tau				1.73
	Palaemonetes pugio	7.07	0.75	9.03	1.80
	Syngnathus fuscus	0.88	0.30	2.28	2.10
Old Inlet	*Callinectes sapidus*	18.15		3.27	98.76
	Centropristis.striata	0.53	1.66	0.11	0.19
	Crangon septemspinosa	2.53	0.88	0.98	1.39
	Hippolyte sp.		0.02	0.30	0.80
	Menidia menidia	0.48	0.50	0.08	2.82
	Palaemonetes pugio	3.48	1.18	0.88	8.05
	Stenotomus chrysops		0.28		
	Syngnathus fuscus	1.15	1.41	1.78	3.19
	Tautoga onitis	9.25		1.73	
	Tautogolabrus adspersus	1.74	0.20		
Pattersquash	*Callinectes sapidus*	2.24	9.61	4.30	175.74
	Crangon septemspinosa	6.47	1.19	3.80	0.56
	Gasterosteus aculeatus		0.05	0.63	0.68
	Hippolyte sp.	0.53	0.04	4.18	0.22
	Menidia menidia	0.06	6.53	0.30	0.83
	Palaemonetes pugio	19.55	2.23	7.54	26.86
	Paralichthys dentatus		0.74		
	Pseudopleuronectes americanus		3.98		
	Syngnathus fuscus	1.83	3.61	1.45	4.89
	Tautoga onitis		42.50		0.38

Appendix II

Species captured in *R. maritima* and *Z. marina* using otter trawls. Species listed totaled at least 1% of the catch at a site during any year of study.

Appendix II-A. Mean abundance (individuals / 100 m^2)

Site	Species	2013	2014	2015	2016
Watch Hill	*Anchoa mitchilli*	1.08	0.72	0.71	
	Callinectes sapidus	0.13	0.22	0.28	1.55
	Chilomycterus schoepfi		0.01		0.03
	Gasterosteus aculeatus			0.52	0.03
	Lagodon rhomboides				0.05
	Libinia sp.			0.83	0.02
	Menidia menidia	0.04		0.09	0.48
	Menticirrhus saxatilis	0.01	0.06	0.03	
	Ovalipes ocellatus	0.01	1.79	0.05	
	Paralichthys dentatus	0.01	0.13		
	Pomatomus saltatrix	0.01	0.07		0.03
	Pseudopleuronectes americanus				0.05
	Stenotomus chrysops	0.01	0.04		
	Syngnathus fuscus		0.03	0.27	0.37
	Tautoga onitis	0.01	0.01	0.02	0.03
Robinson Cove	*Anchoa mitchilli*	3.17	2.74	0.75	1.15
	Callinectes sapidus	0.22	0.23	0.86	1.59
	Chaetodon ocellatus			0.11	
	Gasterosteus aculeatus			0.05	0.04
	Libinia sp.	0.01		0.18	0.07
	Menidia menidia			0.84	4.40
	Menticirrhus saxatilis	0.06	0.07		
	Morone americana			0.08	
	Mycteroperca microlepis			0.05	
	Ovalipes ocellatus	0.02	0.17		
	Paralichthys dentatus	0.02	0.04		
	Sphoeroides maculatus		0.01	0.04	0.11
	Sphyraena borealis				0.16
	Syngnathus fuscus	0.05	0.08	0.46	0.89
	Tautoga onitis		0.01	0.79	0.10
Old Inlet	*Anchoa mitchilli*			1.54	0.02
	Callinectes sapidus	0.02	0.45	0.50	1.38
	Cancer irroratus			0.29	
	Carcinus maenas			0.12	0.02
	Fistularia tabacaria	0.02	0.13		0.01
	Gasterosteus aculeatus			0.19	0.15
	Gerreidae		0.04	0.01	
	Lagodon rhomboides			0.06	0.11
	Libinia sp.	0.02	0.05	0.28	0.02
	Menidia menidia		1.84	2.68	5.94
	Mycteroperca microlepis	0.02		0.07	

	Ovalipes ocellatus		0.01	0.10	0.07
	Pomatomus saltatrix	0.02			0.01
	Stenotomus chrysops	0.11	0.09	0.02	
	Strongylura marina			0.28	
	Syngnathus fuscus		0.35	0.51	0.67
	Tautoga onitis		0.13	0.19	0.08
Pattersquash	*Anchoa mitchilli*	0.86		0.02	
	Apeltes quadracus				0.67
	Callinectes sapidus	0.34	0.33	0.65	2.48
	Gasterosteus aculeatus		3.17	3.34	3.92
	Libinia sp.		0.19	0.27	2.68
	Lucania parva				0.73
	Menidia menidia	0.05	5.71	13.60	6.83
	Menticirrhus saxatilis	0.07			
	Pseudopleuronectes americanus	0.02	0.02	0.07	
	Sphoeroides maculatus		0.67	0.02	
	Syngnathus fuscus	0.07	4.40	1.25	7.50
	Tautoga onitis	0.07	1.85	0.30	0.06
	Tautogolabrus adspersus	0.02	0.54		

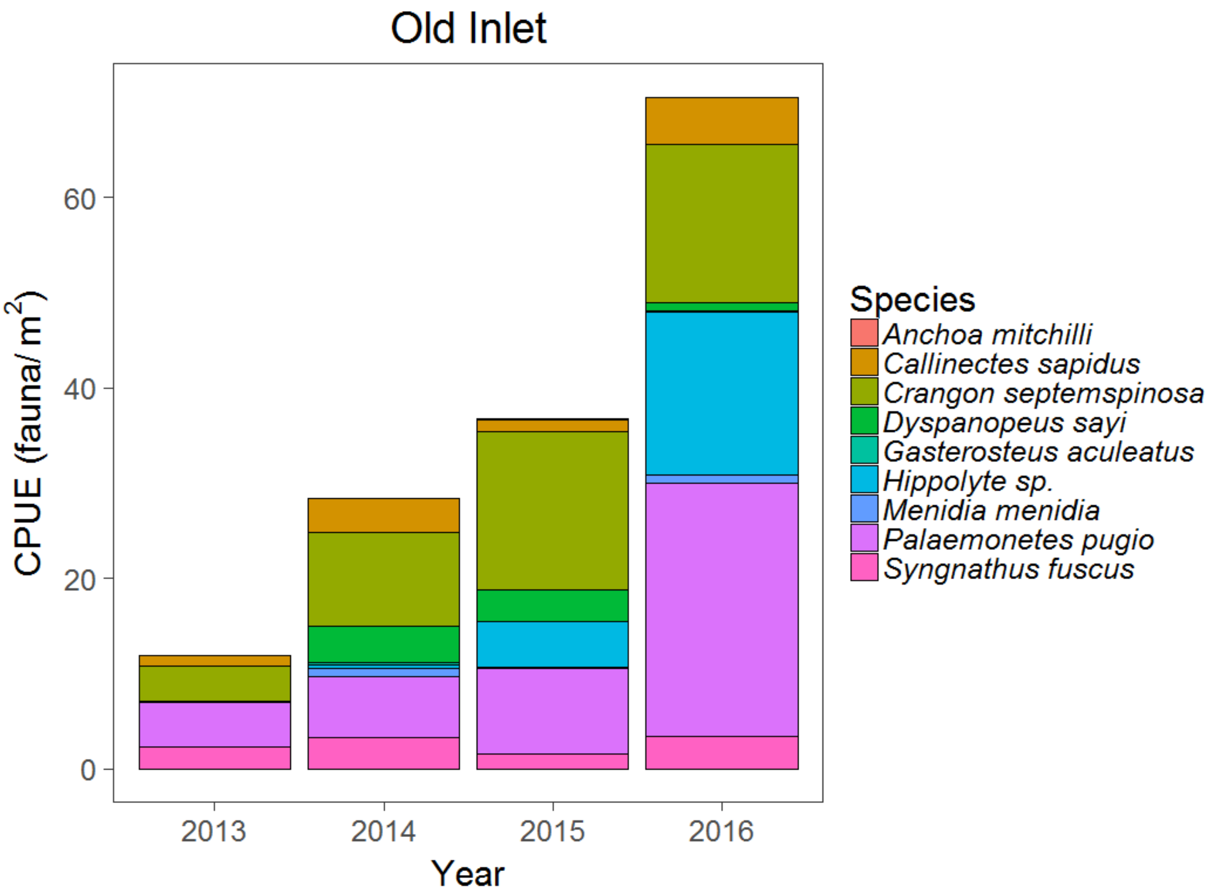
Appendix II-B. Mean biomass (g / 100 m^2)

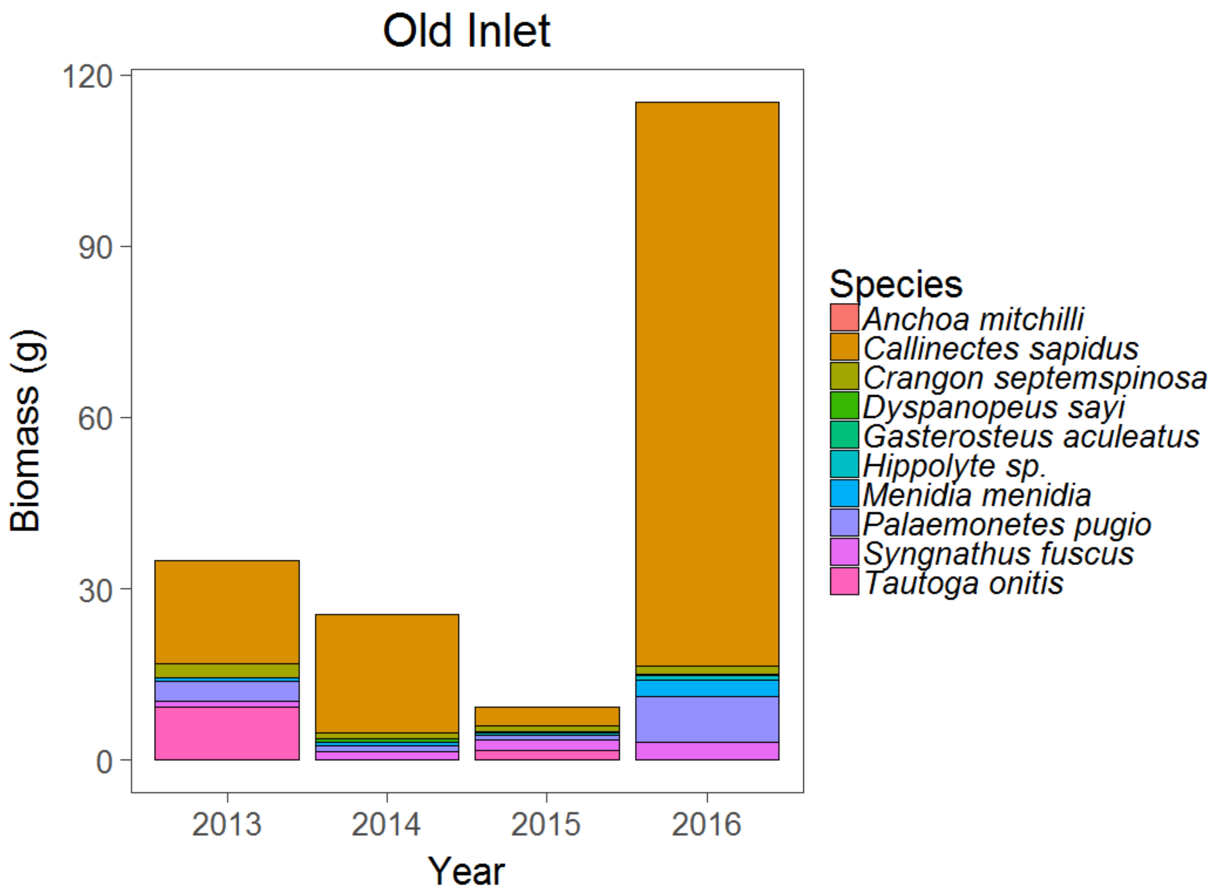
Site	Species	2013	2014	2015	2016
Watch Hill	*Anchoa mitchilli*	0.43	0.31	0.07	142.66
	Callinectes sapidus	6.75	21.56	19.70	
	Chilomycterus schoepfi		0.71		2.51
	Gasterosteus aculeatus			0.27	0.02
	Menidia menidia	0.12		0.01	0.78
	Menticirrhus saxatilis	0.67	3.36	0.12	
	Paralichthys dentatus	0.75	8.74		
	Pomatomus saltatrix	0.41	4.52		0.29
	Sphoeroides maculatus				2.33
	Stenotomus chrysops	0.29	0.15		1.80
	Syngnathus fuscus		0.04	0.44	0.51
	Synodus saurus		0.43		
	Tautoga onitis	0.15	0.43	0.38	4.58
Robinson Cove	*Anchoa mitchilli*	0.83	1.34	0.03	0.30
	Callinectes sapidus	13.85	21.35	108.62	188.96
	Chilomycterus schoepfi			8.14	3.99
	Leiostomus xanthurus	1.43			
	Menidia menidia			0.30	10.63
	Menticirrhus saxatilis	3.27	4.94		0.27
	Morone saxatilis				2.71
	Mycteroperca.microlepis			2.47	
	Opsanus tau	0.11			0.10
	Paralichthys dentatus	2.68	1.58		3.90
	Pomatomus saltatrix		0.82		0.13
	Sphoeroides maculatus			4.26	4.21
	Stenotomus chrysops		0.41	1.17	0.52
	Syngnathus fuscus	0.03	0.13	0.53	1.14
	Tautoga onitis		0.03	20.54	4.76
Old Inlet	*Callinectes sapidus*	0.01	31.47	38.28	134.42
	Centropristis striata		0.00		0.91
	Chilomycterus schoepfi		1.53	0.19	4.81
	Fistularia tabacaria	0.01	0.51		0.02
	Gerreidae		0.03	0.01	
	Lagodon rhomboides			1.88	
	Menidia menidia		2.85	0.43	9.06
	Menticirrhus saxatilis			0.74	
	Mycteroperca microlepis	0.57		1.67	
	Pomatomus saltatrix	2.95			
	Sphoeroides maculatus		0.46	5.25	0.87
	Stenotomus chrysops	3.90	0.98	1.17	2.45
	Syngnathus fuscus		0.25	0.48	0.77

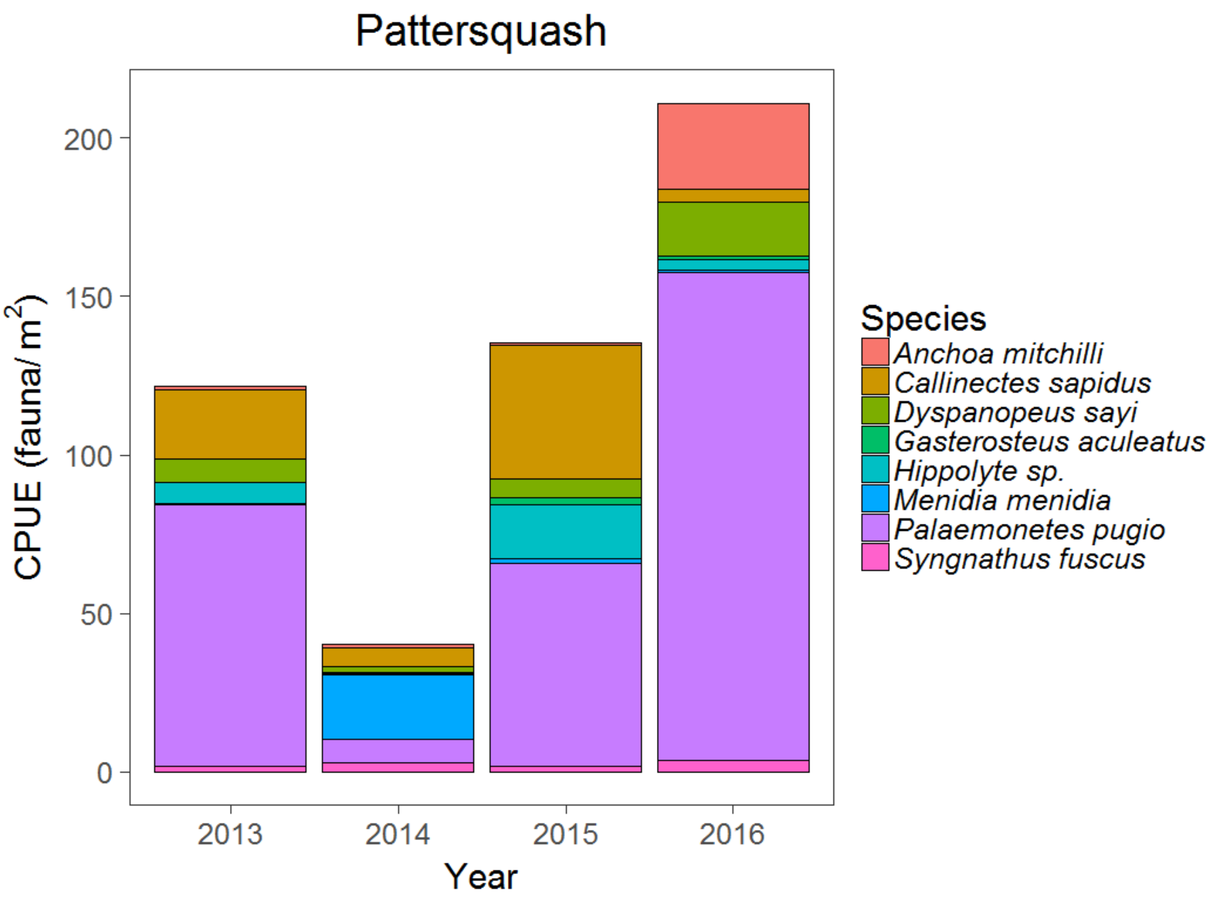
	Tautoga onitis		0.31	2.24	1.14
Pattersquash	*Anguilla rostrata*				12.35
	Anchoa mitchilli	0.47			
	Callinectes sapidus	38.01	27.95	59.04	122.22
	Chilomycterus schoepfi			25.99	0.40
	Gasterosteus aculeatus		1.40	2.36	1.47
	Menidia menidia	0.09	9.36	1.27	11.50
	Menticirrhus saxatilis	3.96			
	Morone saxatilis			21.37	
	Mycteroperca microlepis		2.07		
	Opsanus tau				4.72
	Paralichthys dentatus		4.82	5.18	
	Pseudopleuronectes americanus	12.10	0.16	0.84	
	Sphoeroides maculatus		20.14	4.24	
	Syngnathus fuscus	0.16	8.87	2.03	8.62
	Tautoga onitis	0.23	8.70	29.34	1.29
	Tautogolabrus adspersus	0.04	1.43		

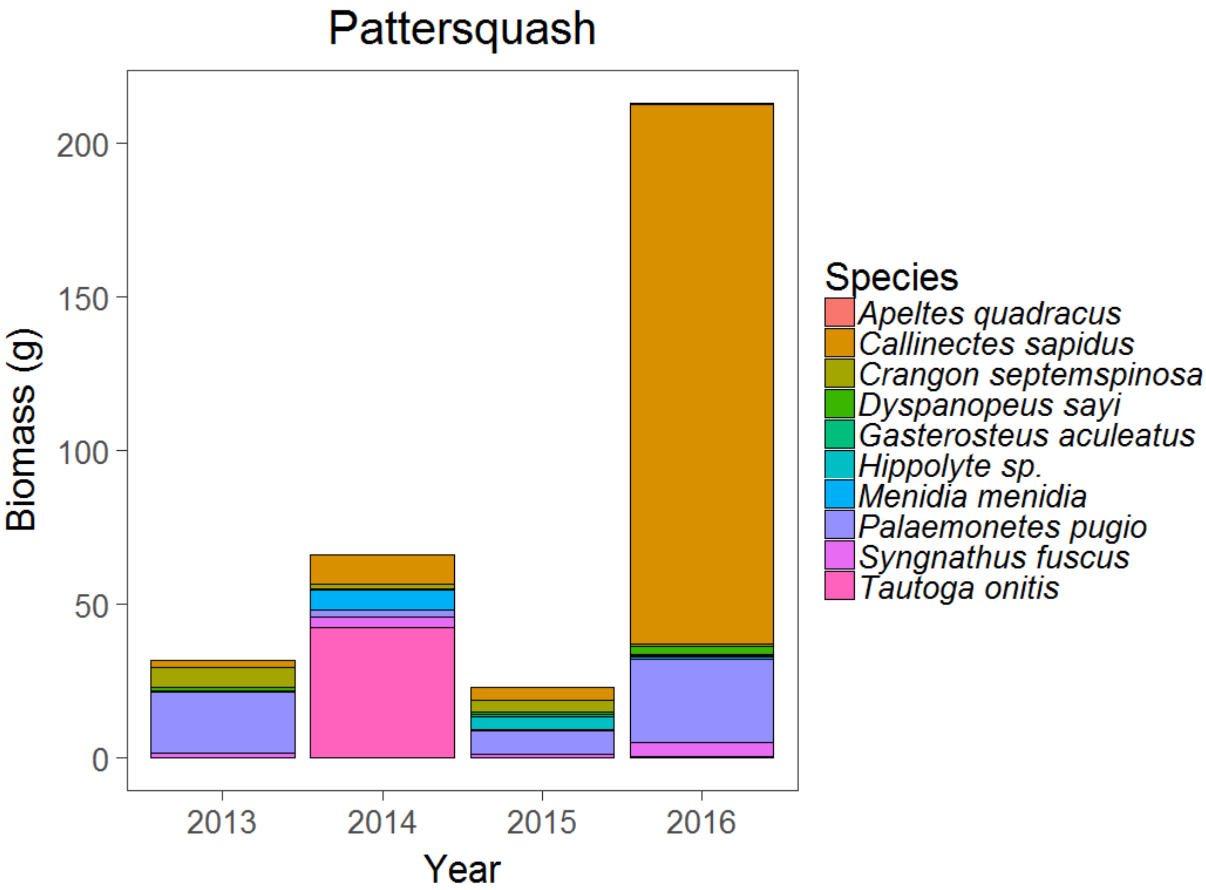
Appendix III

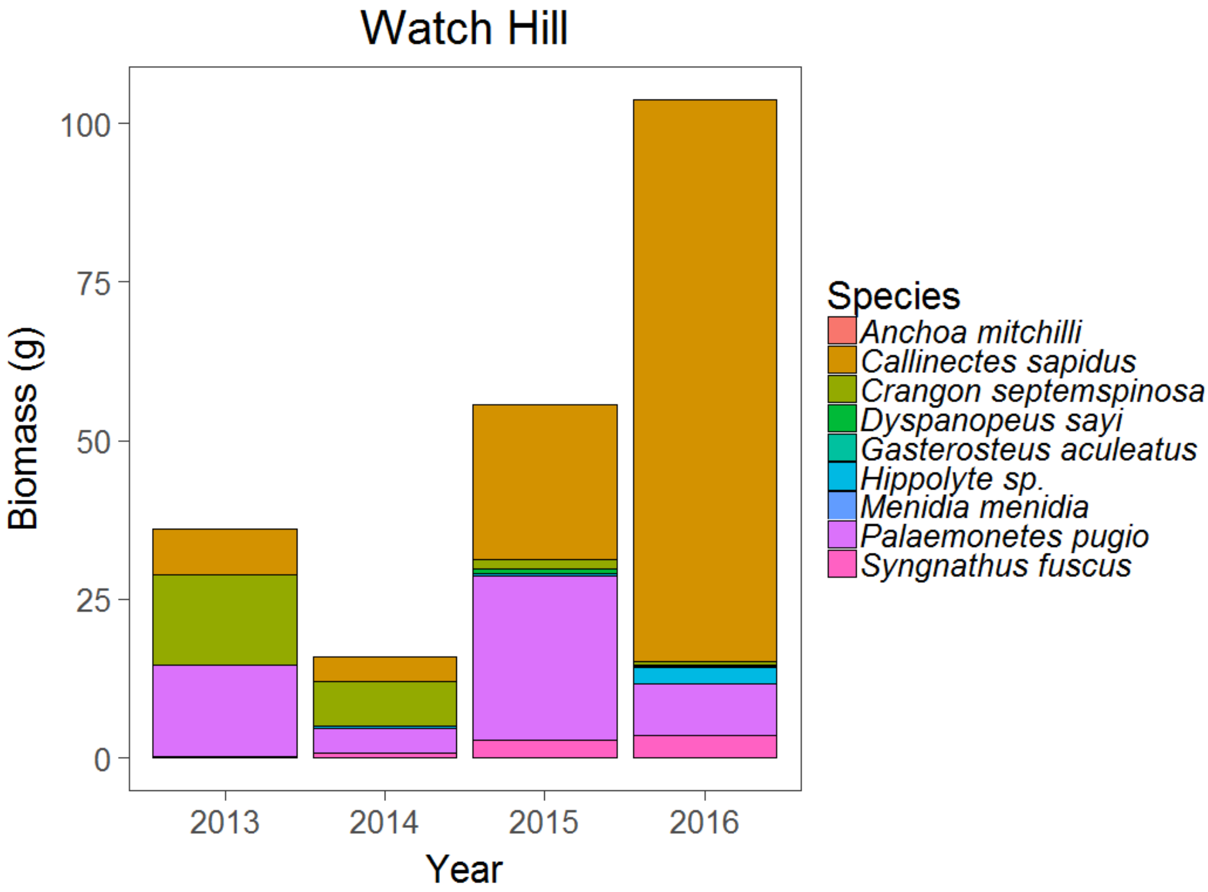
Community composition stacked bar plots of mean abundance (fauna/m^2) and biomass (g/m^2) of species captured using throw traps. Plots are only those in which there was a significance difference at a site over time (see Figure 2A and 2C).


Old Inlet
CPUE (fauna/ m^2)
0
20
40
60
2013
2014
2015
2016
Year
Species
Anchoa mitchilli
Callinectes sapidus
Crangon septemspinosa
Dyspanopeus sayi
Gasterosteus aculeatus
Hippolyte sp.
Menidia menidia
Palaemonetes pugio
Syngnathus fuscus


Old Inlet
Biomass (g)
120
90
60
30
0
2013
2014
2015
2016
Year
Species
Anchoa mitchilli
Callinectes sapidus
Crangon septemspinosa
Dyspanopeus sayi
Gasterosteus aculeatus
Hippolyte sp.
Menidia menidia
Palaemonetes pugio
Syngnathus fuscus
Tautoga onitis

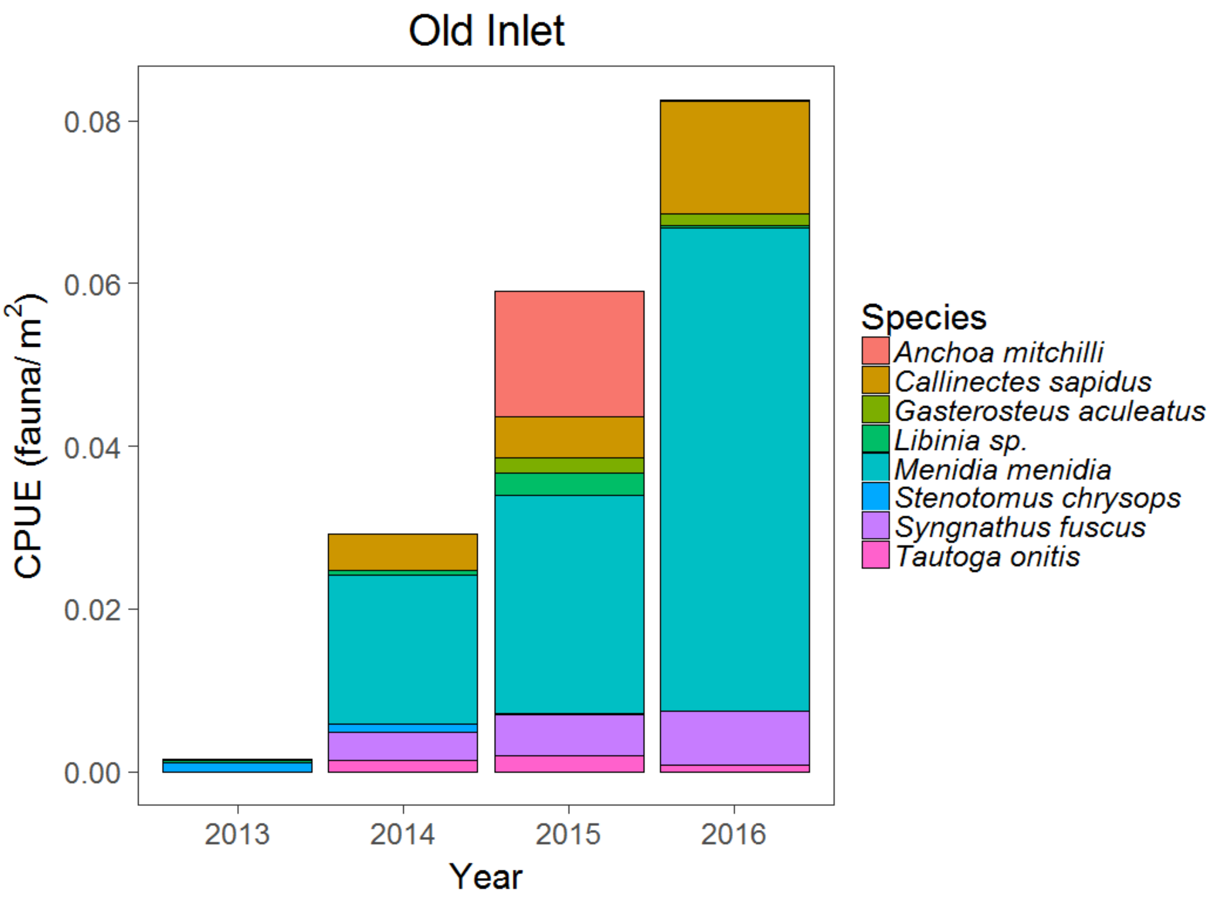

Pattersquash
CPUE (fauna/ m^2)
0
50
100
150
200
2013
2014
2015
2016
Year
Species
Anchoa mitchilli
Callinectes sapidus
Dyspanopeus sayi
Gasterosteus aculeatus
Hippolyte sp.
Menidia menidia
Palaemonetes pugio
Syngnathus fuscus

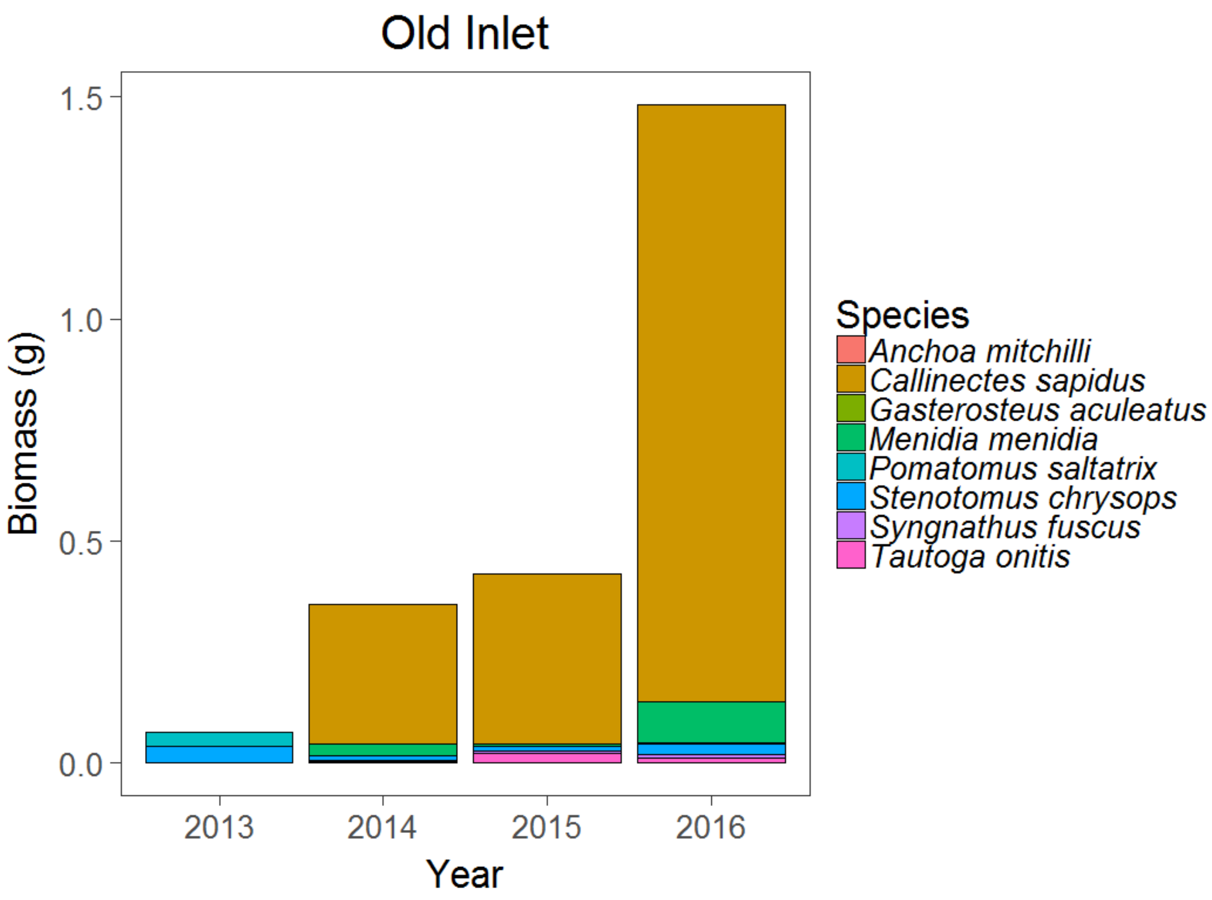

Pattersquash
Biomass (g)
200
150
100
50
0
2013
2014
2015
2016
Year
Species
Apeltes quadracus
Callinectes sapidus
Crangon septemspinosa
Dyspanopeus sayi
Gasterosteus aculeatus
Hippolyte sp.
Menidia menidia
Palaemonetes pugio
Syngnathus fuscus
Tautoga onitis

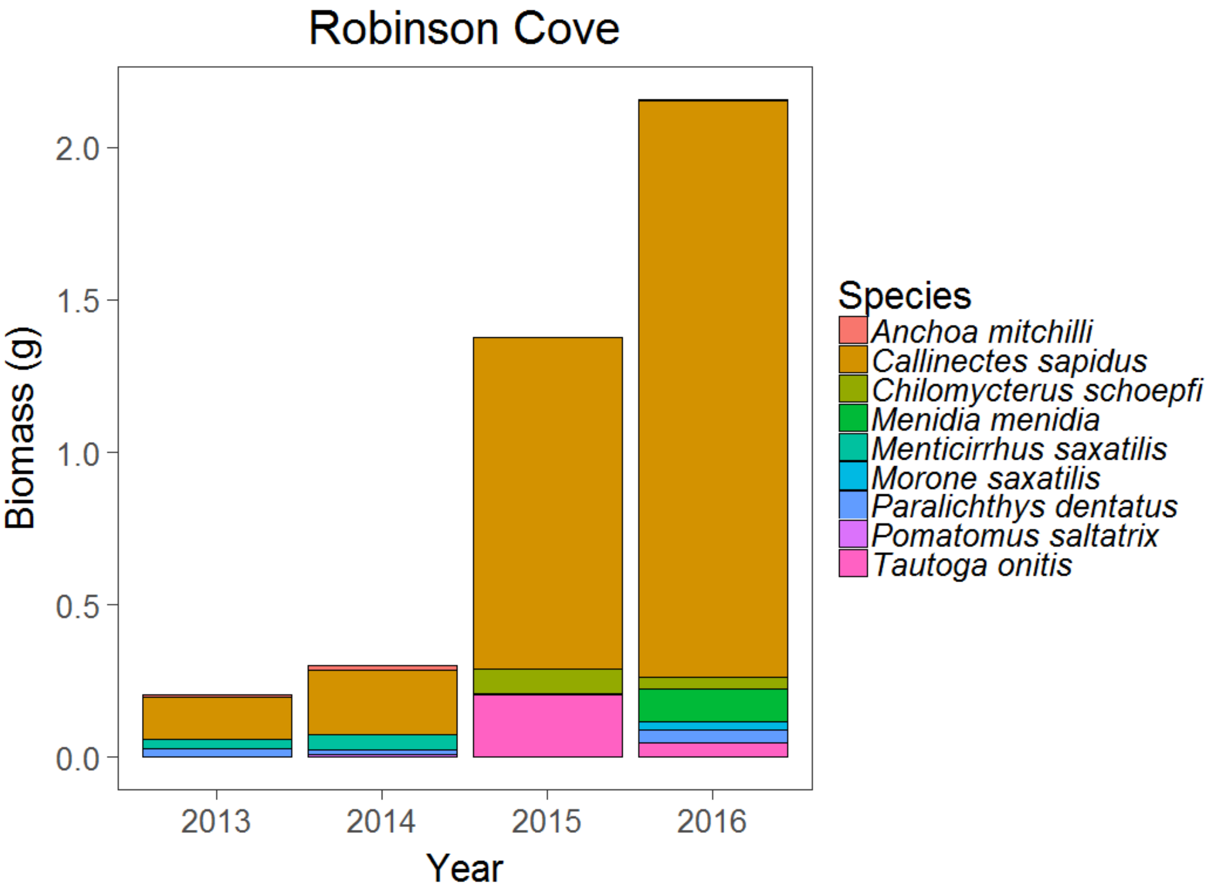

Watch Hill
Biomass (g)
100
75
50
25
0
2013
2014
2015
2016
Year
Species
Anchoa mitchilli
Callinectes sapidus
Crangon septemspinosa
Dyspanopeus sayi
Gasterosteus aculeatus
Hippolyte sp.
Menidia menidia
Palaemonetes pugio
Syngnathus fuscus

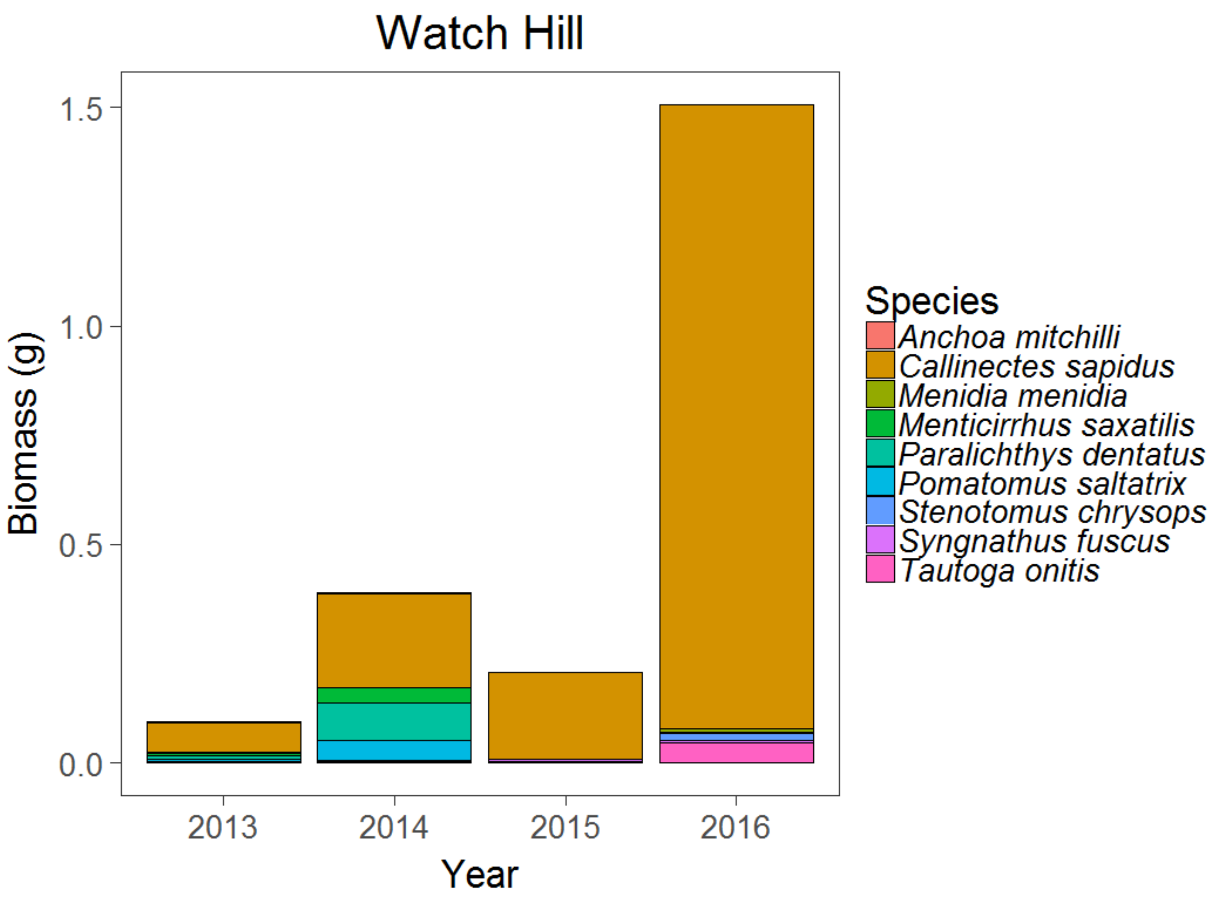
Appendix IV

Community composition stacked bar plots of mean abundance (fauna/m^2) and biomass (g/m^2) of species captured using otter trawls. Plots are only those in which there was a significance difference at a site over time (see Figure 2B and 2D).


Old Inlet
CPUE (fauna/ m^2)
0.08
0.06
0.04
0.02
0.00
2013
2014
2015
2016
Year
Species
Anchoa mitchilli
Callinectes sapidus
Gasterosteus aculeatus
Libinia sp.
Menidia menidia
Stenotomus chrysops
Syngnathus fuscus
Tautoga onitis


Old Inlet
1.5
1.0
0.5
0.0
Biomass (g)
2013
2014
2015
2016
Year
Species
Anchoa mitchilli
Callinectes sapidus
Gasterosteus aculeatus
Menidia menidia
Pomatomus saltatrix
Stenotomus chrysops
Syngnathus fuscus
Tautoga onitis


Robinson Cove
Biomass (g)
2.0
1.5
1.0
0.5
0.0
2013
2014
2015
2016
Year
Species
Anchoa mitchilli
Callinectes sapidus
Chilomycterus schoepfi
Menidia menidia
Menticirrhus saxatilis
Morone saxatilis
Paralichthys dentatus
Pomatomus saltatrix
Tautoga onitis


Watch Hill
Biomass (g)
1.5
1.0
0.5
0.0
2013
2014
2015
2016
Year
Species
Anchoa mitchilli
Callinectes sapidus
Menidia menidia
Menticirrhus saxatilis
Paralichthys dentatus
Pomatomus saltatrix
Stenotomus chrysops
Syngnathus fuscus
Tautoga onitis