



Effects of Hurricane Sandy on Fire Island National Seashore (NY)

Assessing water quality and seagrass

Natural Resource Report NPS/NCBN/NRR—2017/1467





ON THIS PAGE

Zostera marina (left) and *Ruppia maritima* (right)

Photograph by A. TINOCO

ON THE COVER

Aerial Image of wilderness Breach, Fire Island National Seashore, June 24, 2014.

Photograph by C. FLAGG

Effects of Hurricane Sandy on Fire Island National Seashore (NY)

Assessing water quality and seagrass

Natural Resource Report NPS/NCBN/NRR—2017/1467

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Abstract

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 bay-wide seagrass monitoring, we quantified the impact of Hurricane Sandy on the flora within GSB. The most evident water column changes were a drastic bay-wide increase in salinity and a decline in water temperature near the breach. 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. Further monitoring of water quality and flora is critical in determining how GSB will ultimately respond to this new influx of ocean water.

Acknowledgments

We thank Hilary Neckles, Charles Roman, and Robert Cerrato for providing invaluable comments and edits. We would like to acknowledge the following people for assistance with field work: Lisa Jackson, Jamie Brisbane, Brooke Rodgers, Bradley Furman, Stephen Heck, Amber Stubler, Kaitlyn O'Toole, Elizabeth Gomez, Will Weid, Steve Milea, Alyson Lowell, Niko Floros, Tracy Vlasek, Horia Papa, Mike Cashin, Taylor Aptner, Frankie Karnes, and Katie Lankowicz.

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. 1). 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.

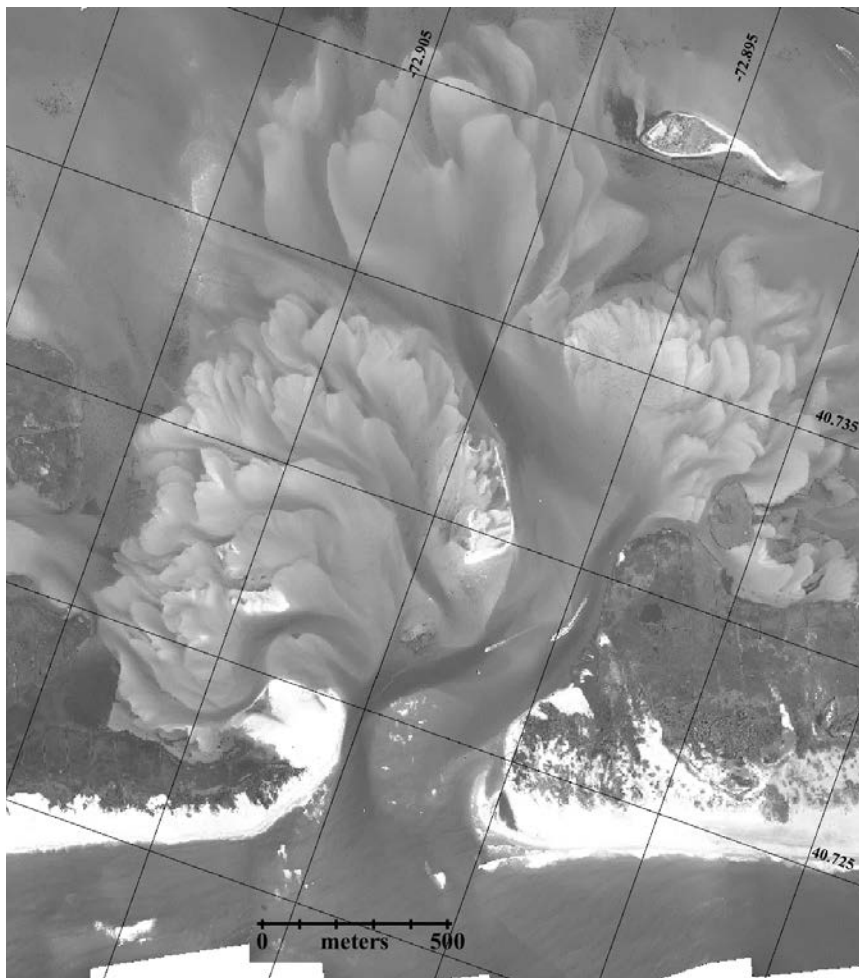


Figure 1. Aerial image of the breach at Old Inlet as of August 5, 2016. Source: Charlie Flagg.

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. 2). 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 (Casper 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.

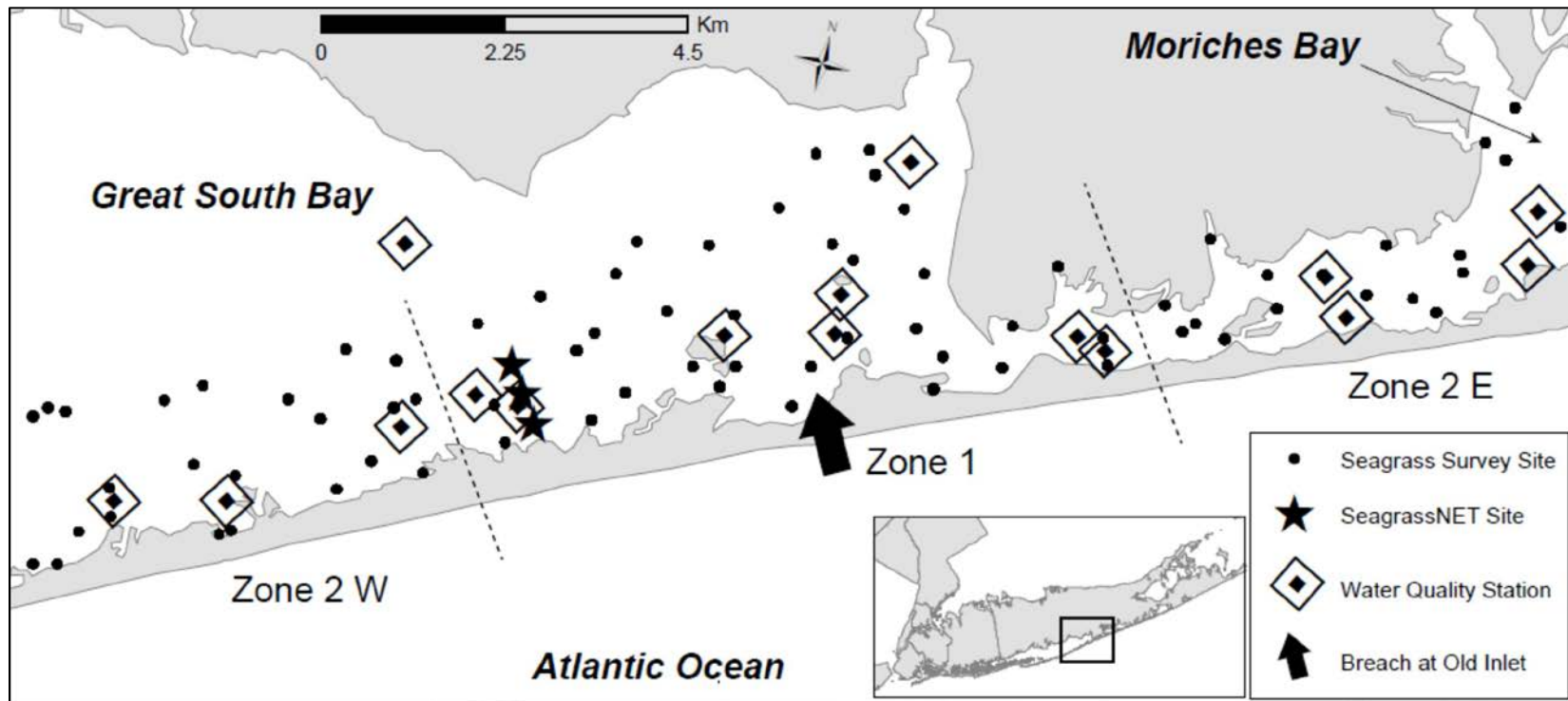


Figure 2. Location of water quality stations and seagrass sampling sites in Great South Bay, NY.

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 polyethylene 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 m² 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 m² 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 m², 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 m². Using the average shoot weight (estimates from the non-linear regression), average shoot density per 0.25 m², 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 m² 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. 2). 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 m² 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. 2); 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).

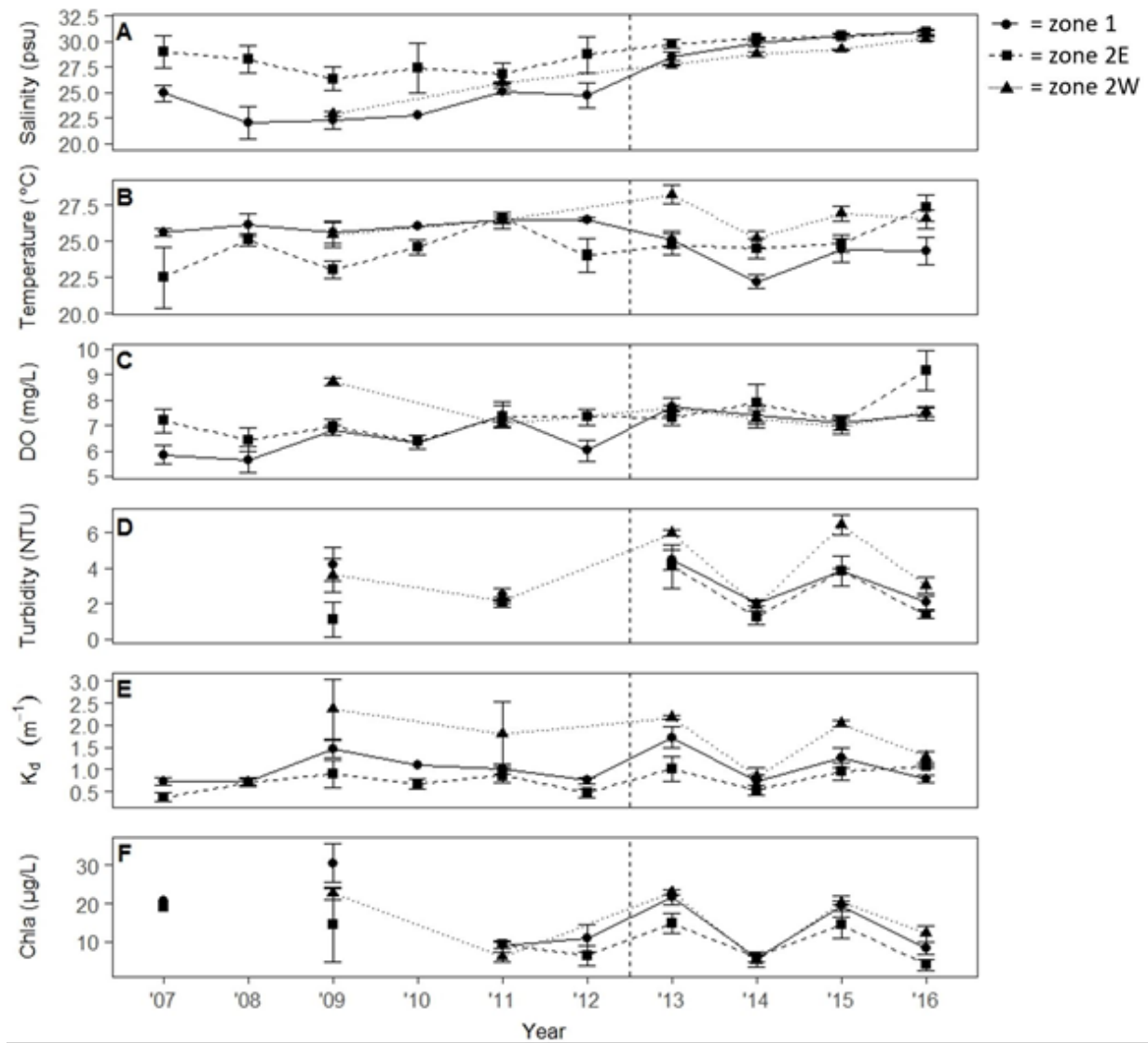


Figure 3. Mean ($\pm$ SE) water quality data (A) salinity, (B) temperature, (C) dissolved oxygen, (D) turbidity, (E) light attenuation, (F) chlorophyll a, at all sites located within each sampling zone.

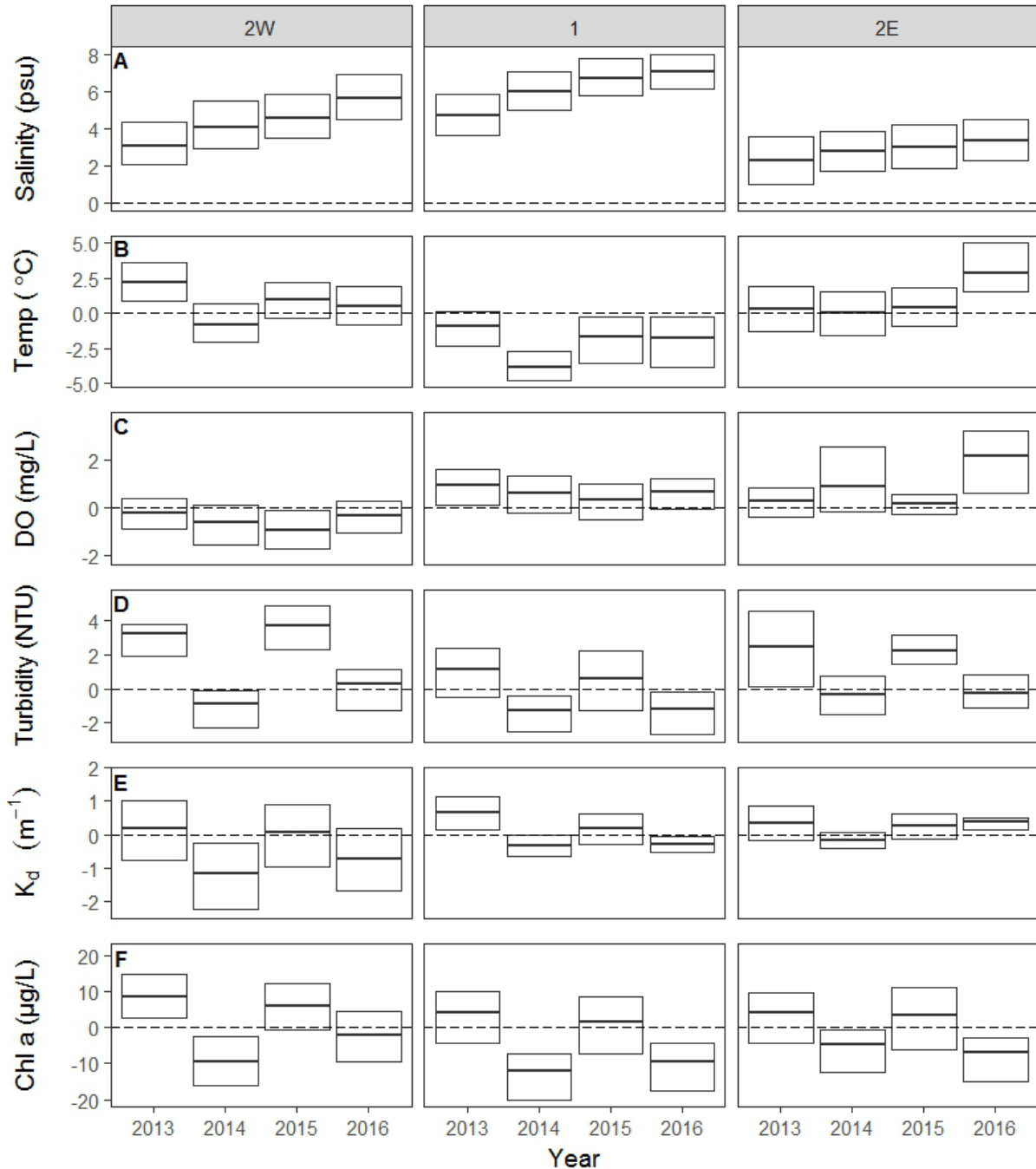


Figure 4. Mean effect size (95% confidence intervals) for water quality data (A) salinity, (B) temperature, (C) dissolved oxygen, (D) turbidity, (E) light attenuation, (F) chlorophyll a, at all sites located within each sampling zone.

Rapid Seagrass Assessment

Mean ($\pm$ 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).

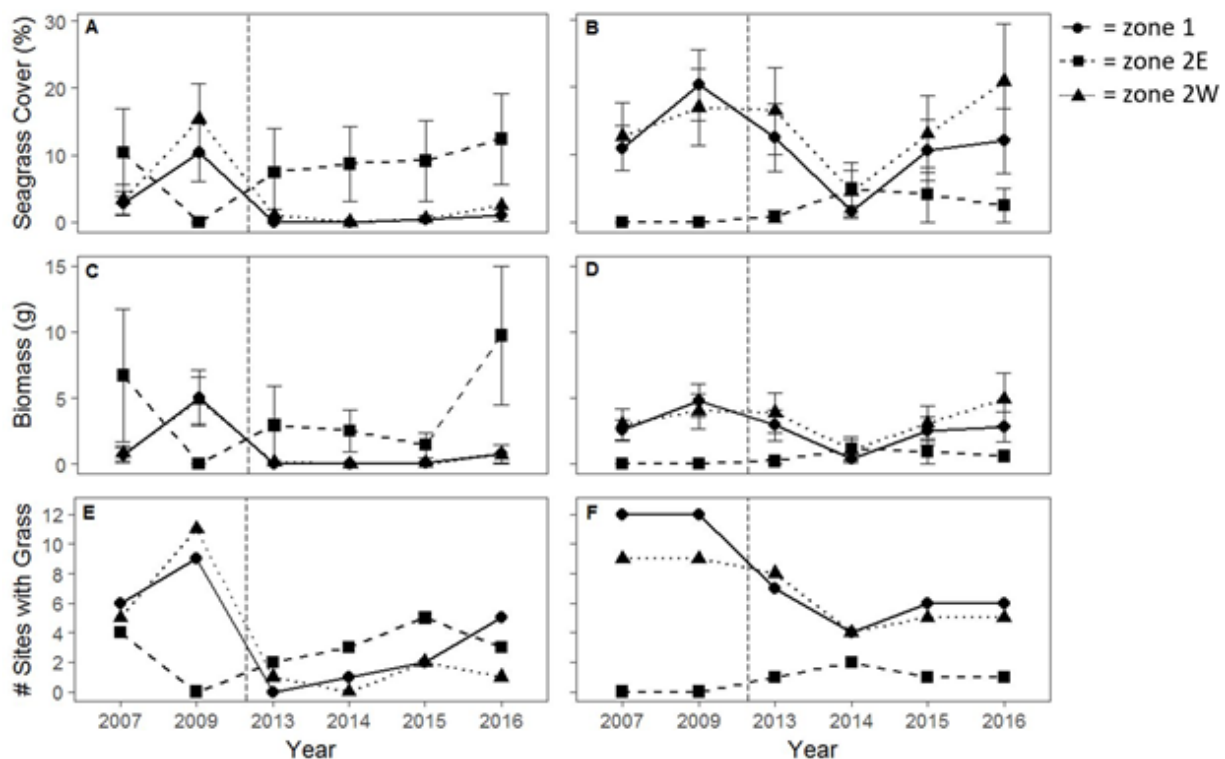


Figure 5. Mean ($\pm$ SE) percent cover and biomass, and number of sites with seagrass present for bay-wide rapid seagrass sampling for sites located within each sampling zone. Panels ACE are *Z. marina*, panels BDF are *R. maritima*.

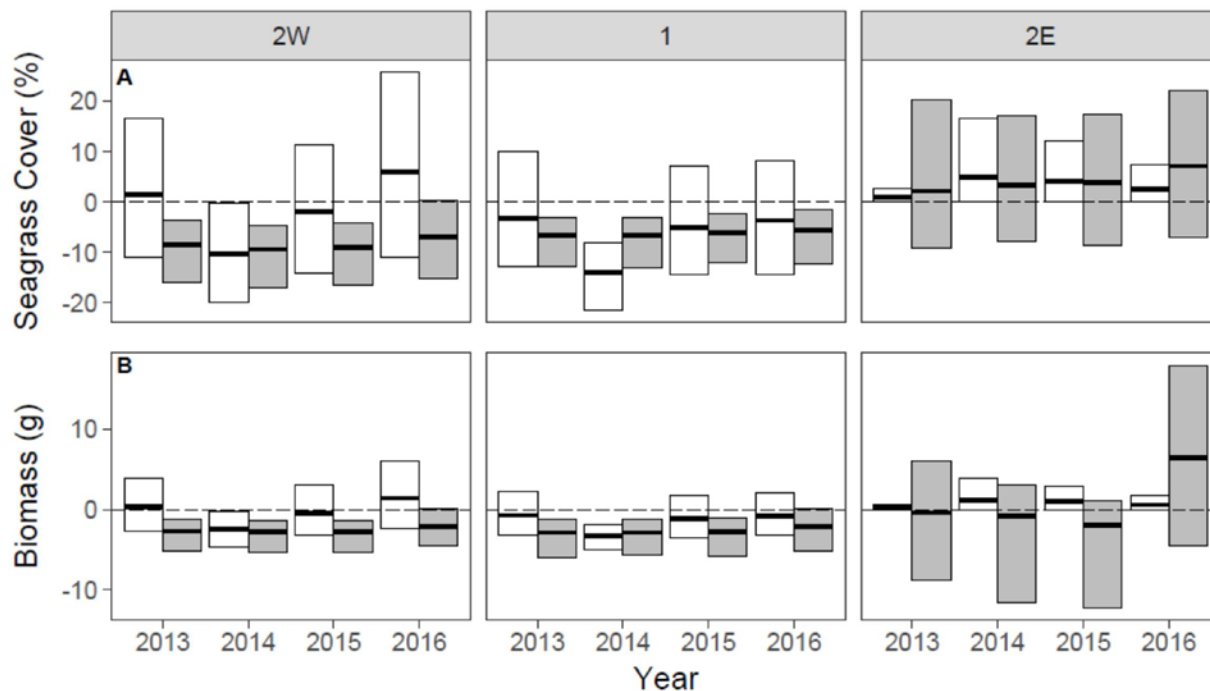


Figure 6. Mean effect size (95% confidence intervals) for rapid seagrass percent cover and biomass data. White bars are *R. maritima*, grey bars are *Z. marina*.

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.

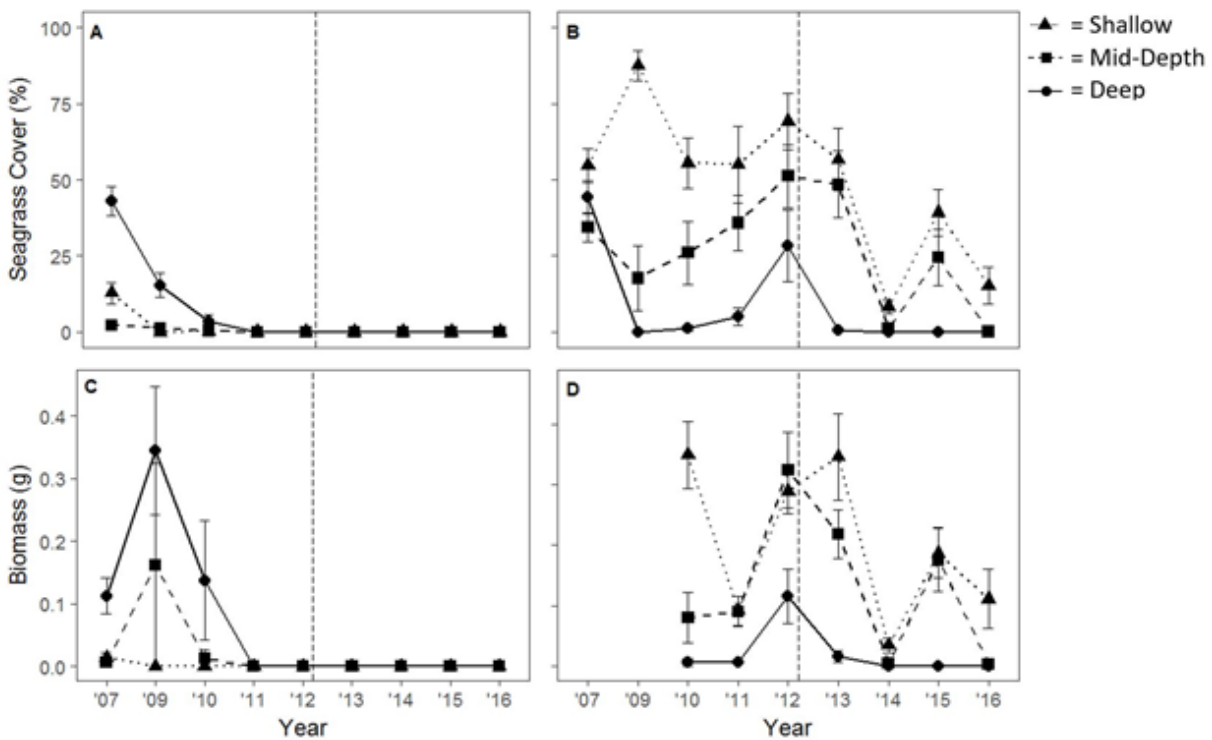


Figure 7. Mean (± SE) percent cover and biomass for seagrassNET data at all quadrats located within each transect depth. Panels AC are *Z. marina*, panels BD are *R. maritima*.

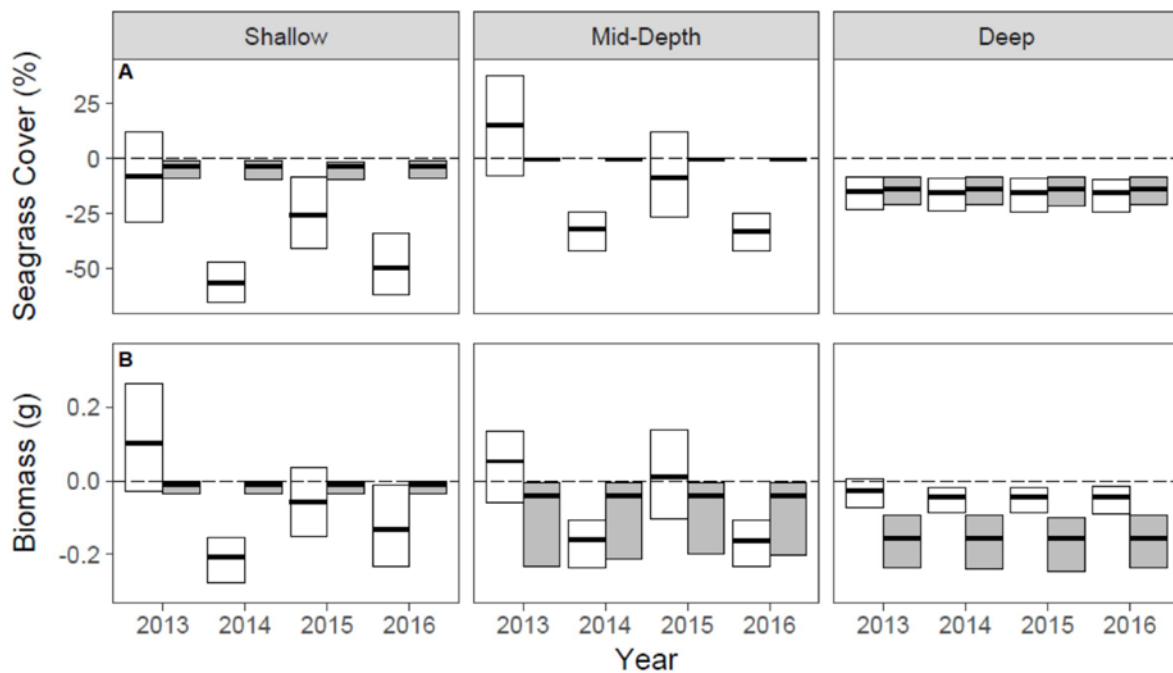


Figure 8. Mean effect size (95% confidence intervals) for seagrassNET percent cover and biomass data. White bars are *R. maritima*, grey bars are *Z. marina*.

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.

Species	Parameter		2007	2009	2010	2011	2012	2013	2014	2015	2016
<i>R. maritima</i>	% 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
<i>Z. marina</i>	% 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

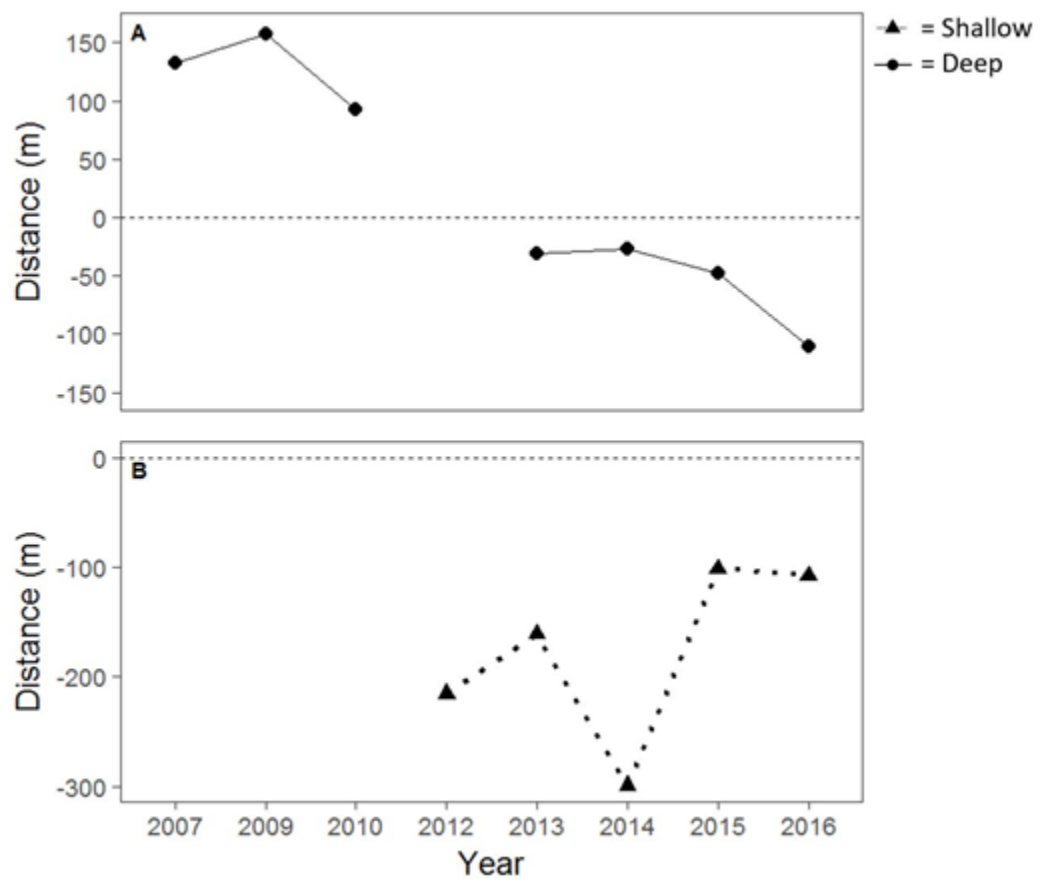


Figure 9. Distance from transect (dashed line) to edge of bed at seagrassNET sampling site. Panel A is the deep transect and panel B is the shallow transect.

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 Carpentaria, 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 (Deltare 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; Deltare & 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.

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