

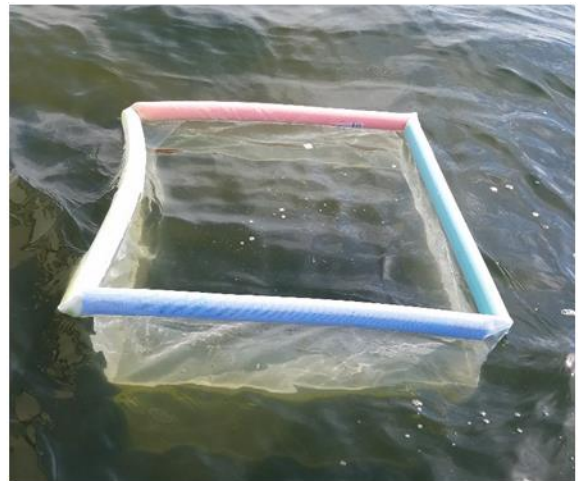


Effects of Hurricane Sandy on Fire Island National Seashore (NY)

Assessing Seagrass-associated Nekton Communities (July 2018 Update)

Natural Resource Report NPS/NCBN/NRR—2018/





ON THIS PAGE

Otter trawl (top), seining and throw trap (bottom) sampling techniques

Photograph by S. GULICK (bottom left) and A. TINOCO (top left, top right and bottom right)

ON THE COVER

Aerial image of Wilderness breach, Fire Island National Seashore, June 1, 2016

Photograph by C. FLAGG

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

*Assessing Seagrass-associated Nekton Communities (July
2018 Update)*

Natural Resource Report NPS/NCBN/NRR—2018/ XXXX

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July 2018

U.S. Department of the Interior
National Park Service
Natural Resource Stewardship and Science
Fort Collins, Colorado

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Contents

	Page
Figures.....	v
Tables.....	vii
Appendices.....	ix
Abstract.....	xi
Update Summary	xi
Acknowledgments.....	xiii
Introduction.....	1
Methods.....	5
Site Description	5
Field Sampling.....	6
Statistical Analysis	7
Results.....	9
Site Characteristic over Time	9
Fauna Abundances and Biomass (Throw Traps).....	9
Fauna Abundances and Biomass (Trawls)	11
Comparison of Fauna Abundance to 1996 data	11
Multivariate Community Analysis and Diversity.....	12
Discussion.....	17
Literature Cited	21

Figures

	Page
Figure 1. Aerial image of the breach at Old Inlet as of August 5, 2016. Source: Charlie Flagg.	3
Figure 2. Location of fauna sampling stations in Great South Bay and Moriches Bay, NY (WH = Watch Hill, RC = Robinson Cove, OI = Old Inlet, PS= Pattersquash).	5
Figure 3. Mean abundance ($\pm$ SE) and biomass ($\pm$ SE) of fauna captured in either species of seagrass over time using throw traps (A, C) and otter trawls (B, D).	10
Figure 4. Mean abundance ($\pm$ SE) of the blue crab (<i>C. sapidus</i>) 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.	10
Figure 5. Mean abundance ($\pm$ SE) of fauna as a function of site and year in <i>Z. marina</i> using throw traps. Similar letters indicate no significant difference in the fauna/m ² between years within a site.	12
Figure 6. nMDS ordination of the fauna community composition at Old Inlet as a function of year and species of seagrass.	13
Figure 7. nMDS ordination of the fauna community composition in 2016 as a function of site and species of seagrass.	14
Appendix C Figure 1. Community composition (abundance) at Old Inlet using throw traps.	33
Appendix C Figure 2. Community composition (biomass) at Old Inlet using throw traps.	34
Appendix C Figure 3. Community composition (abundance) at Pattersquash using throw traps.	35
Appendix C Figure 4. Community composition (biomass) at Pattersquash using throw traps.	36
Appendix C Figure 5. Community composition (biomass) at Watch Hill using throw traps.	37
Appendix D Figure 1. Community composition (abundance) at Old Inlet using otter trawls.	39
Appendix D Figure 2. Community composition (biomass) at Old Inlet using otter trawls.	40
Appendix D Figure 3. Community composition (biomass) at Robinson Cove using otter trawls.	41

Tables

Page

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= <i>R. maritima</i> , Z= <i>Z. marina</i> . Years where both R and Z are present indicates the presence of two monospecific beds, one of each species, at that site.	6
Table 2a. Mean ($\pm$ SE) Shannon diversity index (H') values for each site, year, and species of seagrass combination 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	15
Table 2b. Similar letters (b) indicate no significant difference in diversity between years within site and species of seagrass. (-) indicates no throw traps were completed	15
Appendix A Table 1. Mean abundance (individuals / m ²).	25
Appendix A Table 2. Mean biomass (g / m ²).	27
Appendix B Table 1. Mean abundance (individuals / 100 m ²).	29
Appendix B Table 2. Mean biomass (g / 100 m ²).	31

Appendices

	Page
Appendix A. Species captured using throw traps. Species listed totaled at least 1% of the catch at a site during any year of study.	25
Appendix B. Species captured using otter trawls. Species listed totaled at least 1% of the catch at a site during any year of study.	29
Appendix C. Community composition of mean abundance (fauna/m ²) and biomass (g/m ²) 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).	33
Appendix D. Community composition of mean abundance (fauna/m ²) and biomass (g/m ²) 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).	39

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 monitor fauna communities inhabiting the two seagrass species found in GSB, *Zostera marina* and *Ruppia maritima*. We quantified the impact of Hurricane Sandy on the fauna within the estuarine lagoon, examining temporal and spatial differences in seagrass-associated nekton as a function of a major disturbance event and the proximity to a new oceanic inlet. Fauna abundances were significantly higher in 2016 (four years post-Hurricane 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 breach. Further monitoring of 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.

Update Summary

The June 2017 version of this report has been updated. Figure 5 and Appendix A Table 1 were edited to reflect the correct data from 1996. The mistake made in the original submission was using the raw data from the 1996 report while not standardizing to replicate and m² in order to accurately compare to the data collected post-Sandy. The statistics corresponding to the data corrections were also edited within the text.

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 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, *Thalassia testudinum* and *Syringodium filiforme*, 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 (Fig. 1). 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; Tinoco & Peterson (2017)).

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.

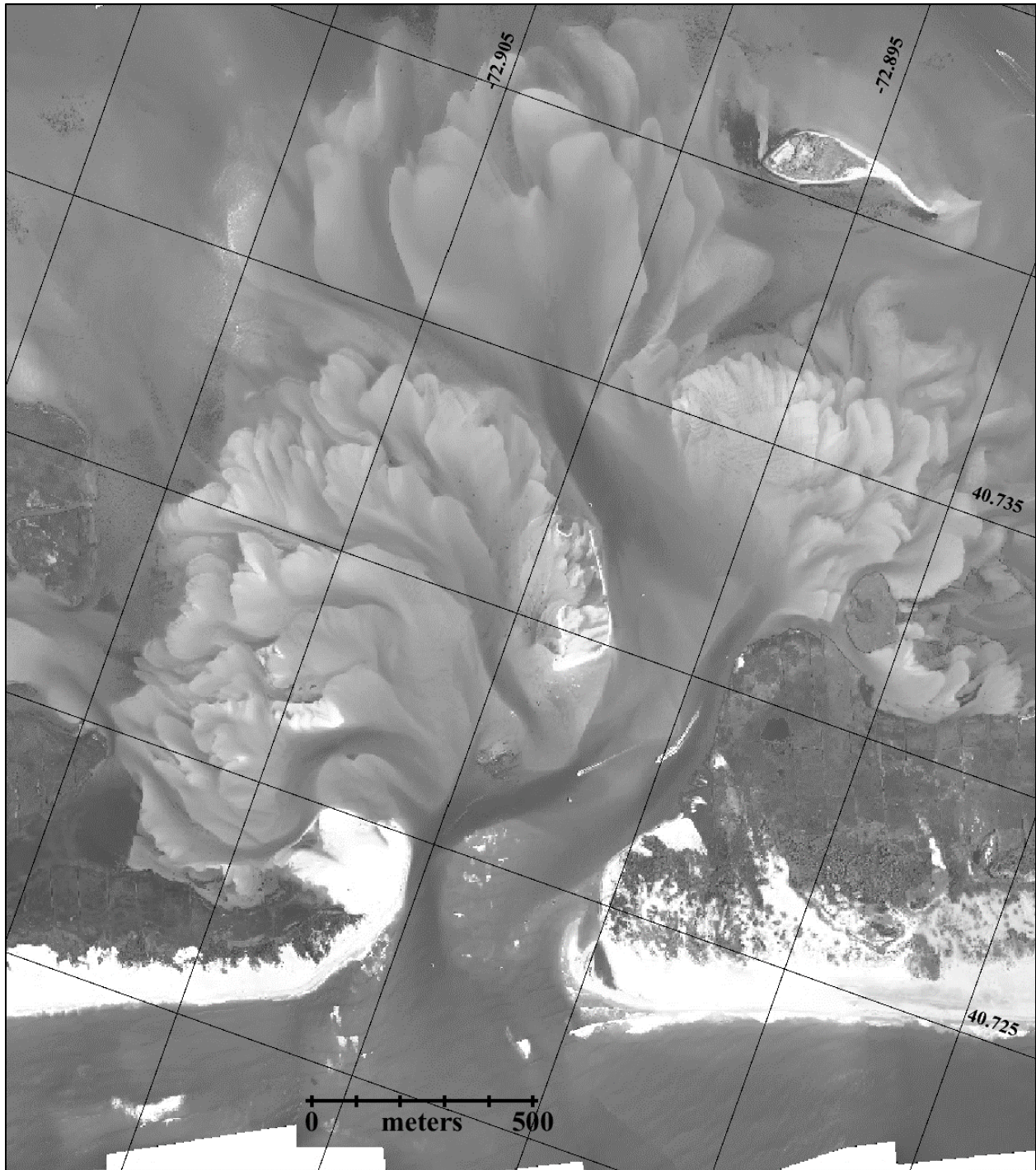


Figure 1. Aerial image of the breach at Old Inlet as of August 5, 2016. Source: Charlie Flagg.

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 km² 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. 2). Prior to Hurricane Sandy, WH was characterized by a continuous ~0.23 km² *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.).

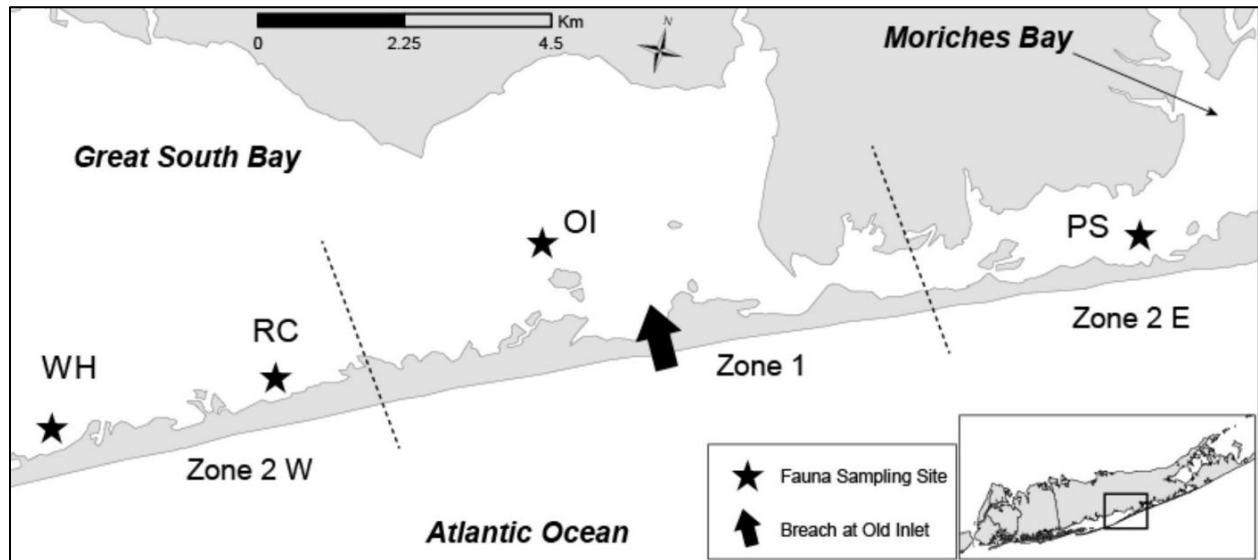


Figure 2. Location of fauna sampling stations in Great South Bay and Moriches Bay, NY (WH = Watch Hill, RC = Robinson Cove, OI = Old Inlet, PS= Pattersquash).

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 Fig. 2), 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 Fig. 2). 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).

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.

Sampling Technique	Sample Site	Sample Year			
		2013	2014	2015	2016
Throw Trap	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	Watch Hill	R	M	M	Z
	Robinson Cove	R	M	M	Z
	Old Inlet	U	R	M	RZ
	Pattersquash	Z	Z	Z	Z

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 m² 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-dimensional 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 km² 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. 3A). 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/m² (hereafter ind/m²), annually increasing to ~20 ind/m² in 2016 (see Appendix A Table 1). 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/m² in 2013 to 5 ind/m² in 2016. Fauna abundances at PS were significantly lower in 2014 than in any other year ($F_{(3)} = 8.522$, $p < 0.001$; Fig. 3A). This decline was driven by the loss of *P. pugio* with only an average of 7 ind/m² in 2014 as compared to 82 ind/m², 64 ind/m², and 154 ind/m² in 2013, 2015, and 2016 respectively (see Appendix A, Table 1). 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/m² respectively, while the mean abundance of juveniles at OI was 3 ind/m² (Fig. 4). 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. 3C), 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. 3C), due to increases in *C. sapidus* and *P. pugio* (Appendix C Figure 2). 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. 3C).

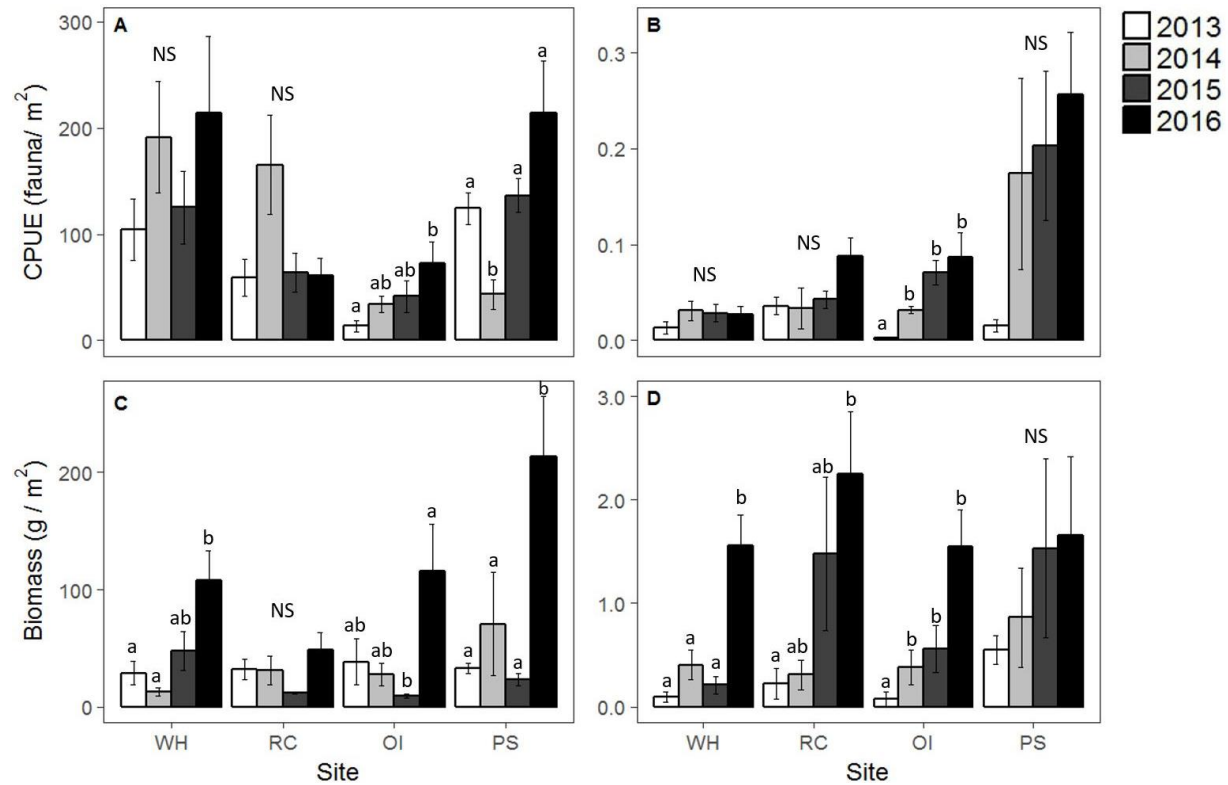


Figure 3. Mean abundance ($\pm$ SE) and biomass ($\pm$ SE) of fauna captured in either species of seagrass over time using throw traps (A, C) and otter trawls (B, D).

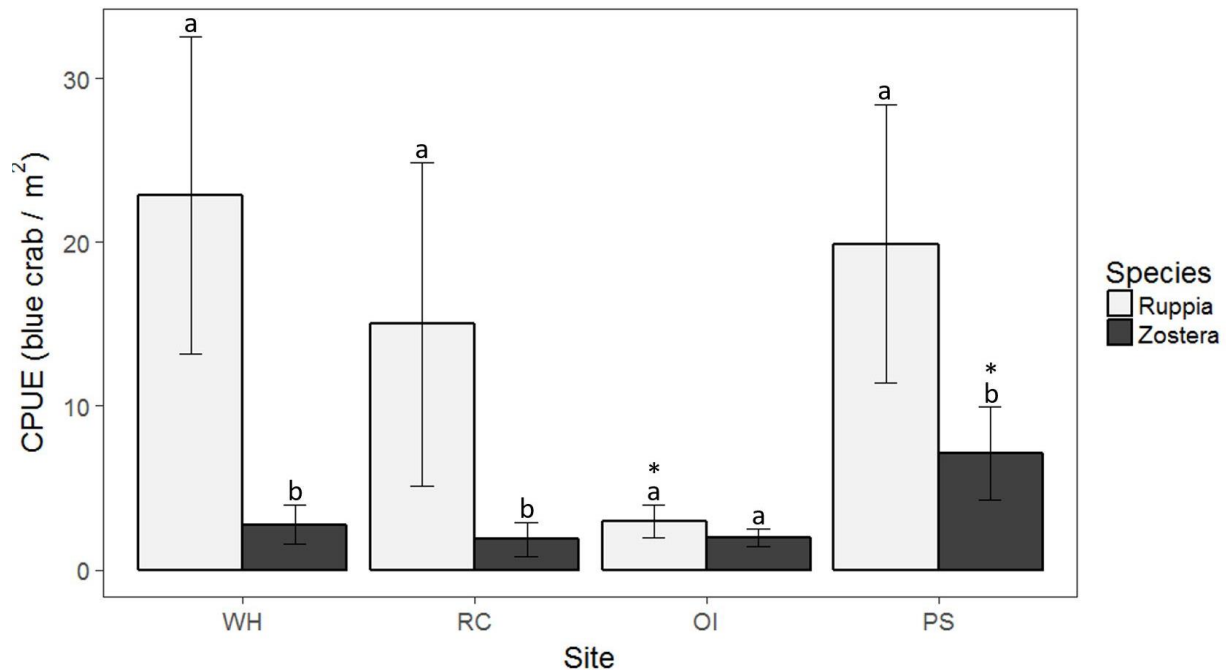


Figure 4. Mean abundance ($\pm$ SE) of the blue crab (*C. sapidus*) as a function of site and species of seagrass in 2016 throw trap samples. Similar letters indicate no difference in the number of blue crabs /m² between species of seagrass within a single site. Asterisks indicate a significant difference in the number of blue crabs /m² between sites within a species of seagrass.

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. 3B, Appendix D). 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; Appendix B) 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. 3D) 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. 3D). Biomass at OI in 2013 was significantly different than every year after ($F_{(3)} = 9.007$, $p < 0.001$; Fig. 3D). This difference was driven primarily by increases in the entire fauna community, and secondarily by increases in *C. sapidus* (Appendix D Figure 2).

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 ($F_{(1)} = 24.26$, $p < 0.0017$; Fig. 5) 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 ($F_{(4)} = 19.93$, $p < 0.001$; Fig 5). 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*.

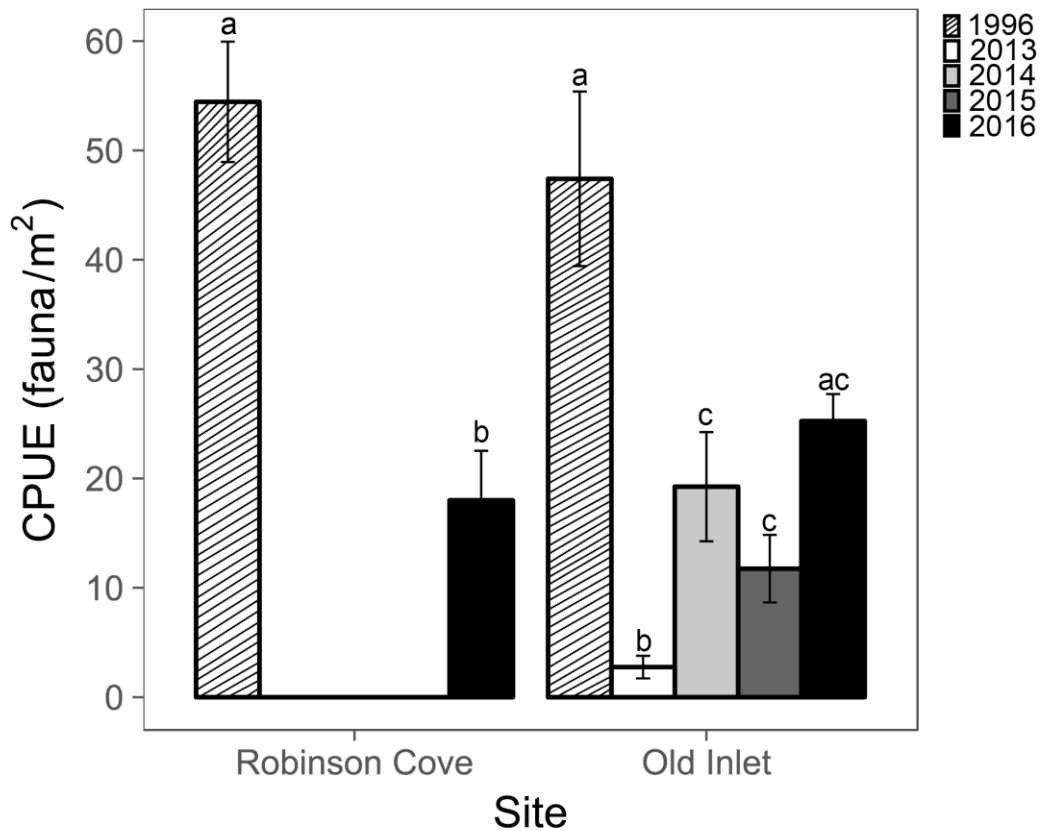


Figure 5. Mean abundance ($\pm$ SE) of fauna as a function of site and year in *Z. marina* using throw traps. Similar letters indicate no significant difference in the fauna/m² between years within a site.

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. 6). 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 ($\sim 80\%$ 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. 6). These differences were largely driven by the absence of the four-spined stickleback (*Apeltes quadracus*), the rainwater killifish (*Lucania parva*), and the mummichog (*Fundulus heteroclitus*) 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.

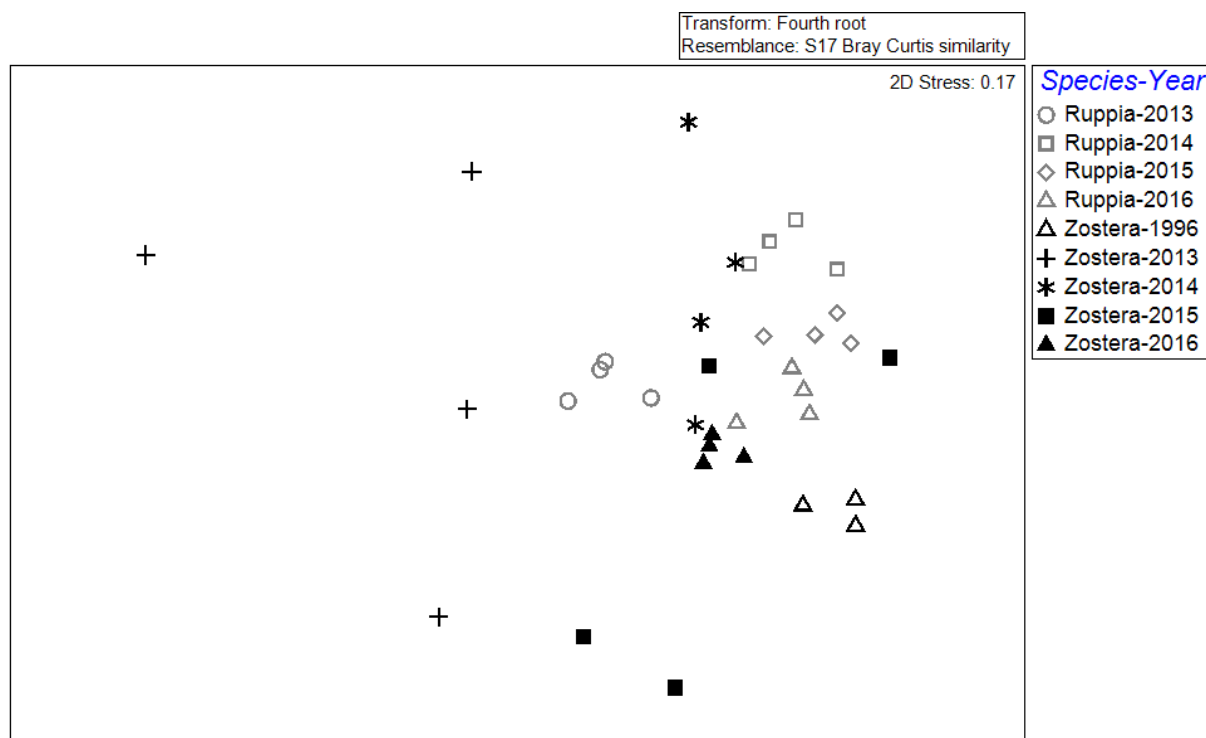


Figure 6. nMDS ordination of the fauna community composition at Old Inlet as a function of year and species of seagrass.

Fauna community composition using trawl data was only examined for 2016 (stress 0.18; Fig. 7) 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. 7). 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. 7). 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. 7).

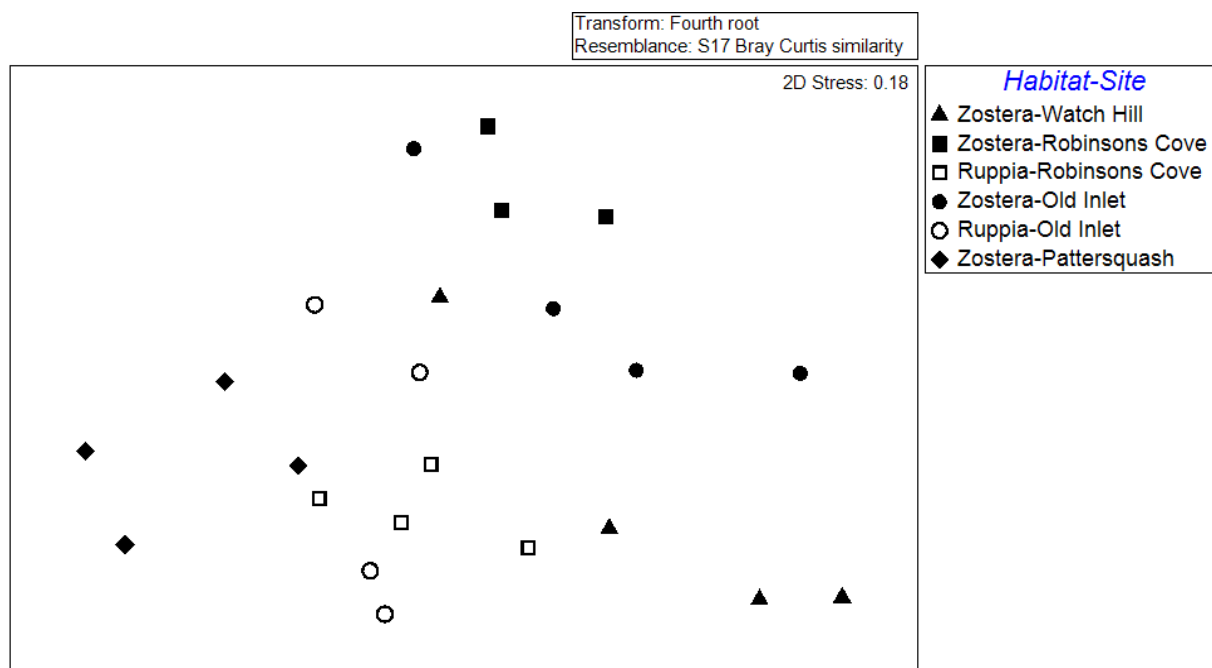


Figure 7. nMDS ordination of the fauna community composition in 2016 as a function of site and species of seagrass.

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 2a). 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 ($H' = 1.32 \pm 0.36$ (1996), 1.33 ± 0.20 (2016)) (Table 2b). Diversity at OI in 1996 was not different than 2014, 2015 nor 2016, but it was significantly higher than 2013 (one-way ANOVA, $F_{(3)} = 5.024$, $p = 0.010$). 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 ($H' = 0.61 \pm 0.72$). The diversity of the community at PS in *Z. marina* was the most variable between years.

Table 2a. Mean ($\pm$ SE) Shannon diversity index (H') values for each site, year, and species of seagrass combination 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

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

Table 2b. Similar letters (b) indicate no significant difference in diversity between years within site and species of seagrass. (-) indicates no throw traps were completed

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

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 (Tinoco & Peterson 2017), 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. 3A and 3B), 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/m² 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/m²), 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. 5; 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. 4). 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 4). 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 3A and 3B). 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/m² in 2013 to 4.58 g/m² 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.

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Appendix A. Species captured using throw traps. Species listed totaled at least 1% of the catch at a site during any year of study.

Appendix A Table 1. Mean abundance (individuals / m²).

Site	Species	1996	2013	2014	2015	2016
Watch Hill	<i>Anchoa mitchilli</i>	1.80	–	–	–	0.13
	<i>Apeltes quadracus</i>	2.20	–	–	–	0.25
	<i>Callinectes sapidus</i>	0.85	1.75	2.00	7.25	25.63
	<i>Crangon septemspinosa</i>	41.00	57.50	157.75	22.00	7.25
	<i>Dyspanopeus sayi</i>	–	0.50	0.50	5.50	1.00
	<i>Hippolyte sp.</i>	23.90	1.00	6.25	4.00	128.13
	<i>Menidia menidia</i>	0.65	–	–	0.25	0.13
	<i>Pagurus longicarpus</i>	–	1.75	7.00	3.75	3.63
	<i>Palaemonetes pugio</i>	7.65	67.00	62.25	110.00	42.13
	<i>Syngnathus fuscus</i>	0.35	0.75	3.25	3.00	3.75
Robinson Cove	<i>Anchoa mitchilli</i>	2.46	4.50	–	–	–
	<i>Apeltes quadracus</i>	4.18	–	–	–	0.13
	<i>Callinectes sapidus</i>	0.45	49.25	2.25	–	16.88
	<i>Crangon septemspinosa</i>	33.50	0.25	189.75	2.50	10.00
	<i>Dyspanopeus sayi</i>	–	–	0.50	0.25	0.88
	<i>Gobiosoma sp.</i>	0.92	4.50	–	–	–
	<i>Hippolyte sp.</i>	10.79	–	0.25	8.25	14.75
	<i>Menidia menidia</i>	0.54	14.75	–	–	1.63
	<i>Palaemonetes pugio</i>	8.47	–	4.00	67.00	12.75
	<i>Pagurus longicarpus</i>	–	0.25	8.25	–	1.00
Old Inlet	<i>Syngnathus fuscus</i>	0.89	–	1.50	2.25	2.13
	<i>Anchoa mitchilli</i>	3.83	–	–	0.13	–
	<i>Apeltes quadracus</i>	15.77	1.13	–	–	0.13
	<i>Callinectes sapidus</i>	0.93	–	3.63	1.25	5.00
	<i>Centropristis striata</i>	–	3.63	2.13	0.13	0.13
	<i>Crangon septemspinosa</i>	12.13	–	9.75	16.56	16.63
	<i>Dyspanopeus sayi</i>	–	–	3.88	3.38	0.75

Site	Species	1996	2013	2014	2015	2016
Pattersquash	<i>Fundulus.heteroclitus</i>	1.60	–	–	–	–
	<i>Hippolyte sp.</i>	5.17	–	0.38	4.81	17.13
	<i>Libinia sp.</i>	–	–	0.63	0.13	0.25
	<i>Menidia menidia</i>	1.97	0.13	0.88	0.13	0.88
	<i>Pagurus longicarpus</i>	–	–	2.50	4.00	1.00
	<i>Palaemonetes pugio</i>	4.67	4.75	6.38	9.00	26.63
	<i>Syngnathus fuscus</i>	0.63	2.25	3.25	1.51	3.38
	<i>Tautoga onitis</i>	–	0.88	–	0.25	–
	<i>Tautogolabrus adspersus</i>	–	0.63	0.25	–	–
	<i>Callinectes sapidus</i>	–	1.13	1.38	0.75	27.00
	<i>Crangon septemspinosa</i>	–	21.50	5.63	42.50	4.13
	<i>Dyspanopeus sayi</i>	–	7.50	2.13	5.63	17.00
	<i>Gasterosteus aculeatus</i>	–	–	0.13	2.50	1.13
	<i>Hippolyte sp.</i>	–	6.88	0.63	16.75	3.50
	<i>Menidia menidia</i>	–	0.13	20.38	1.63	0.63
	<i>Pagurus longicarpus</i>	–	1.88	2.38	1.13	0.75
	<i>Palaemonetes pugio</i>	–	82.50	7.38	63.88	154.00
	<i>Syngnathus fuscus</i>	–	1.88	2.75	1.88	3.63

Appendix A Table 2. Mean biomass (g / m²).

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

Site	Species	2013	2014	2015	2016
	<i>Palaemonetes pugio</i>	19.55	2.23	7.54	26.86
	<i>Paralichthys dentatus</i>	–	0.74	–	–
	<i>Pseudopleuronectes americanus</i>	–	3.98	–	–
	<i>Syngnathus fuscus</i>	1.83	3.61	1.45	4.89
	<i>Tautoga onitis</i>	–	42.50	–	0.38

Appendix B. Species captured using otter trawls. Species listed totaled at least 1% of the catch at a site during any year of study.

Appendix B Table 1. Mean abundance (individuals / 100 m²).

Site	Species	2013	2014	2015	2016
Watch Hill	<i>Anchoa mitchilli</i>	1.08	0.72	0.71	–
	<i>Callinectes sapidus</i>	0.13	0.22	0.28	1.55
	<i>Chilomycterus schoepfi</i>	–	0.01	–	0.03
	<i>Gasterosteus aculeatus</i>	–	–	0.52	0.03
	<i>Lagodon rhomboides</i>	–	–	–	0.05
	<i>Libinia sp.</i>	–	–	0.83	0.02
	<i>Menidia menidia</i>	0.04	–	0.09	0.48
	<i>Menticirrhus saxatilis</i>	0.01	0.06	0.03	–
	<i>Ovalipes ocellatus</i>	0.01	1.79	0.05	–
	<i>Paralichthys dentatus</i>	0.01	0.13	–	–
	<i>Pomatomus saltatrix</i>	0.01	0.07	–	0.03
	<i>Pseudopleuronectes americanus</i>	–	–	–	0.05
	<i>Stenotomus chrysops</i>	0.01	0.04	–	–
	<i>Syngnathus fuscus</i>	–	0.03	0.27	0.37
	<i>Tautoga onitis</i>	0.01	0.01	0.02	0.03
Robinson Cove	<i>Anchoa mitchilli</i>	3.17	2.74	0.75	1.15
	<i>Callinectes sapidus</i>	0.22	0.23	0.86	1.59
	<i>Chaetodon ocellatus</i>	–	–	0.11	–
	<i>Gasterosteus aculeatus</i>	–	–	0.05	0.04
	<i>Libinia sp.</i>	0.01	–	0.18	0.07
	<i>Menidia menidia</i>	–	–	0.84	4.40
	<i>Menticirrhus saxatilis</i>	0.06	0.07	–	–
	<i>Morone americana</i>	–	–	0.08	–
	<i>Mycteroperca microlepis</i>	–	–	0.05	–
	<i>Ovalipes ocellatus</i>	0.02	0.17	–	–
	<i>Paralichthys dentatus</i>	0.02	0.04	–	–
	<i>Sphoeroides maculatus</i>	–	0.01	0.04	0.11
	<i>Sphyræna borealis</i>	–	–	–	0.16
	<i>Syngnathus fuscus</i>	0.05	0.08	0.46	0.89

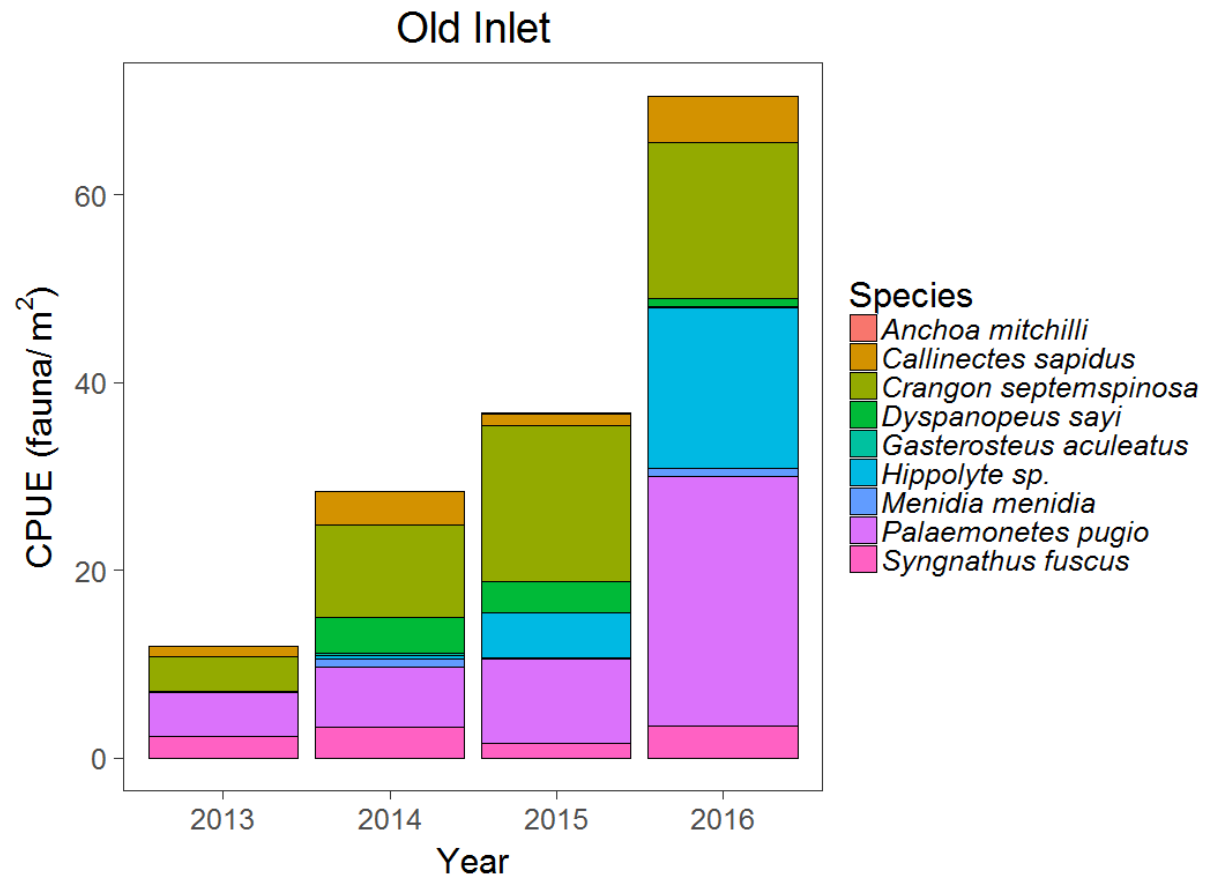
Site	Species	2013	2014	2015	2016
Old Inlet	<i>Tautoga onitis</i>	–	0.01	0.79	0.10
	<i>Anchoa mitchilli</i>	–	–	1.54	0.02
	<i>Callinectes sapidus</i>	0.02	0.45	0.50	1.38
	<i>Cancer irroratus</i>	–	–	0.29	–
	<i>Carcinus maenas</i>	–	–	0.12	0.02
	<i>Fistularia tabacaria</i>	0.02	0.13	–	0.01
	<i>Gasterosteus aculeatus</i>	–	–	0.19	0.15
	<i>Gerreidae</i>	–	0.04	0.01	–
	<i>Lagodon rhomboides</i>	–	–	0.06	0.11
	<i>Libinia sp.</i>	0.02	0.05	0.28	0.02
	<i>Menidia menidia</i>	–	1.84	2.68	5.94
	<i>Mycteroperca microlepis</i>	0.02	–	0.07	–
	<i>Ovalipes ocellatus</i>	–	0.01	0.10	0.07
	<i>Pomatomus saltatrix</i>	0.02	–	–	0.01
	<i>Stenotomus chrysops</i>	0.11	0.09	0.02	–
	<i>Strongylura marina</i>	–	–	0.28	–
	<i>Syngnathus fuscus</i>	–	0.35	0.51	0.67
	<i>Tautoga onitis</i>	–	0.13	0.19	0.08
Pattersquash	<i>Anchoa mitchilli</i>	0.86	–	0.02	–
	<i>Apeltes quadracus</i>	–	–	–	0.67
	<i>Callinectes sapidus</i>	0.34	0.33	0.65	2.48
	<i>Gasterosteus aculeatus</i>	–	3.17	3.34	3.92
	<i>Libinia sp.</i>	–	0.19	0.27	2.68
	<i>Lucania parva</i>	–	–	–	0.73
	<i>Menidia menidia</i>	0.05	5.71	13.60	6.83
	<i>Menticirrhus saxatilis</i>	0.07	–	–	–
	<i>Pseudopleuronectes americanus</i>	0.02	0.02	0.07	–
	<i>Sphoeroides maculatus</i>	–	0.67	0.02	–
	<i>Syngnathus fuscus</i>	0.07	4.40	1.25	7.50
	<i>Tautoga onitis</i>	0.07	1.85	0.30	0.06
	<i>Tautogolabrus adspersus</i>	0.02	0.54	–	–

Appendix B Table 2. Mean biomass (g / 100 m²).

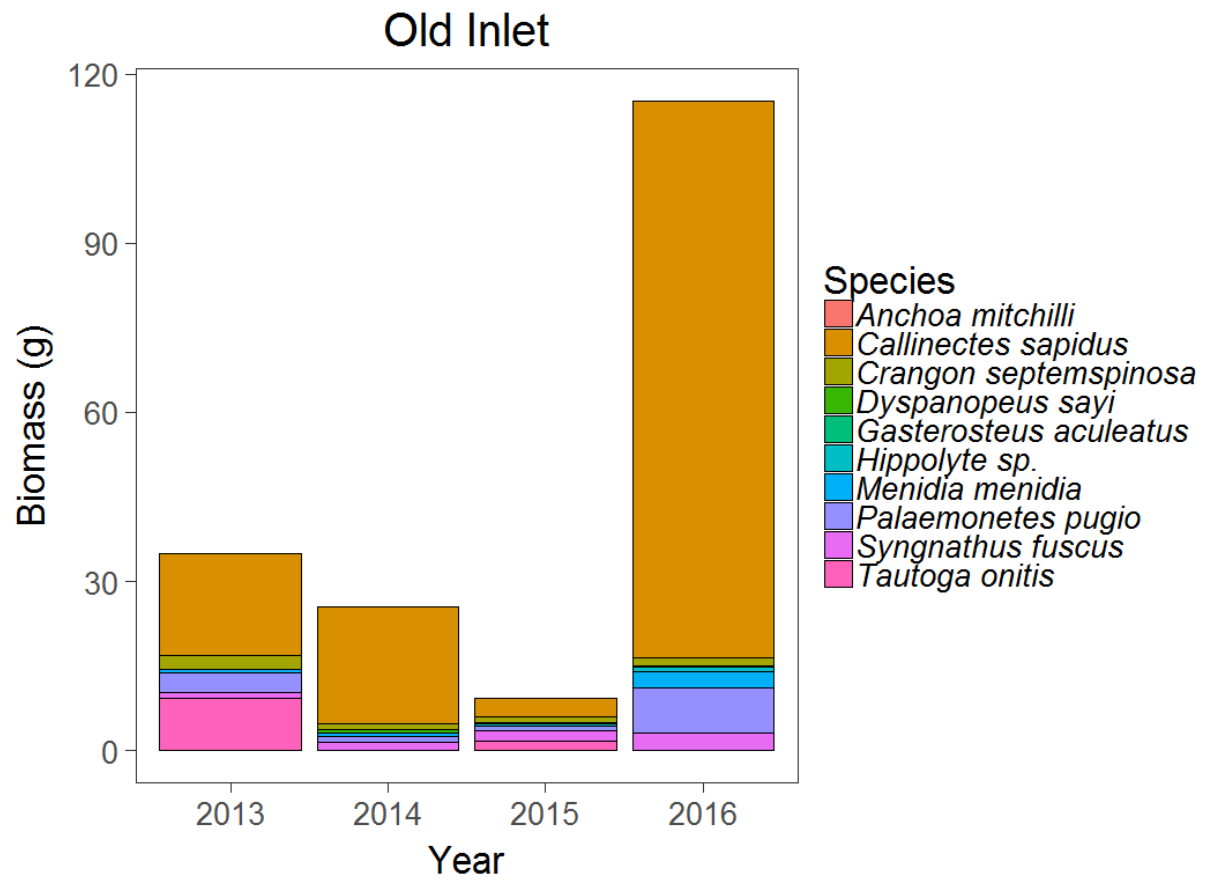
Site	Species	2013	2014	2015	2016
Watch Hill	<i>Anchoa mitchilli</i>	0.43	0.31	0.07	142.66
	<i>Callinectes sapidus</i>	6.75	21.56	19.70	–
	<i>Chilomycterus schoepfi</i>	–	0.71	–	2.51
	<i>Gasterosteus aculeatus</i>	–	–	0.27	0.02
	<i>Menidia menidia</i>	0.12	–	0.01	0.78
	<i>Menticirrhus saxatilis</i>	0.67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

Site	Species	2013	2014	2015	2016
Pattersquash	<i>Fistularia tabacaria</i>	0.01	0.51	–	0.02
	<i>Gerreidae</i>	–	0.03	0.01	–
	<i>Lagodon rhomboides</i>	–	–	1.88	–
	<i>Menidia menidia</i>	–	2.85	0.43	9.06
	<i>Menticirrhus saxatilis</i>	–	–	0.74	–
	<i>Mycteroperca microlepis</i>	0.57	–	1.67	–
	<i>Pomatomus saltatrix</i>	2.95	–	–	–
	<i>Sphoeroides maculatus</i>	–	0.46	5.25	0.87
	<i>Stenotomus chrysops</i>	3.90	0.98	1.17	2.45
	<i>Syngnathus fuscus</i>	–	0.25	0.48	0.77
	<i>Tautoga onitis</i>	–	0.31	2.24	1.14
	<i>Anguilla rostrata</i>	–	–	–	12.35
	<i>Anchoa mitchilli</i>	0.47	–	–	–
	<i>Callinectes sapidus</i>	38.01	27.95	59.04	122.22
	<i>Chilomycterus schoepfi</i>	–	–	25.99	0.40
	<i>Gasterosteus aculeatus</i>	–	1.40	2.36	1.47
	<i>Menidia menidia</i>	0.09	9.36	1.27	11.50
	<i>Menticirrhus saxatilis</i>	3.96	–	–	–
	<i>Morone saxatilis</i>	–	–	21.37	–
	<i>Mycteroperca microlepis</i>	–	2.07	–	–
	<i>Opsanus tau</i>	–	–	–	4.72
	<i>Paralichthys dentatus</i>	–	4.82	5.18	–
	<i>Pseudopleuronectes americanus</i>	12.10	0.16	0.84	–
	<i>Sphoeroides maculatus</i>	–	20.14	4.24	–
	<i>Syngnathus fuscus</i>	0.16	8.87	2.03	8.62
	<i>Tautoga onitis</i>	0.23	8.70	29.34	1.29
	<i>Tautoglabrus adspersus</i>	0.04	1.43	–	–

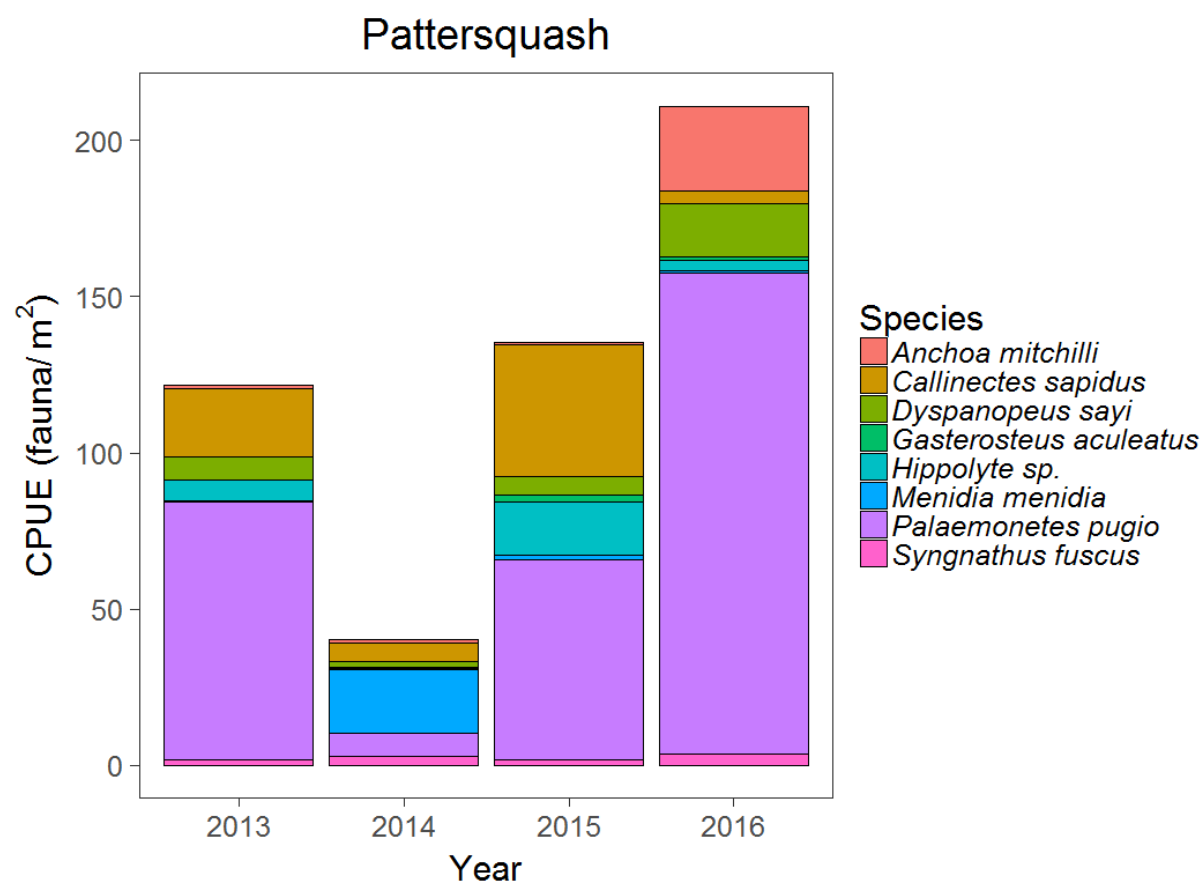
Appendix C. Community composition of mean abundance (fauna/m²) and biomass (g/m²) captured using throw traps. Plots are only those in which there was a significance difference at a site over time (see Figure 2A and 2C).



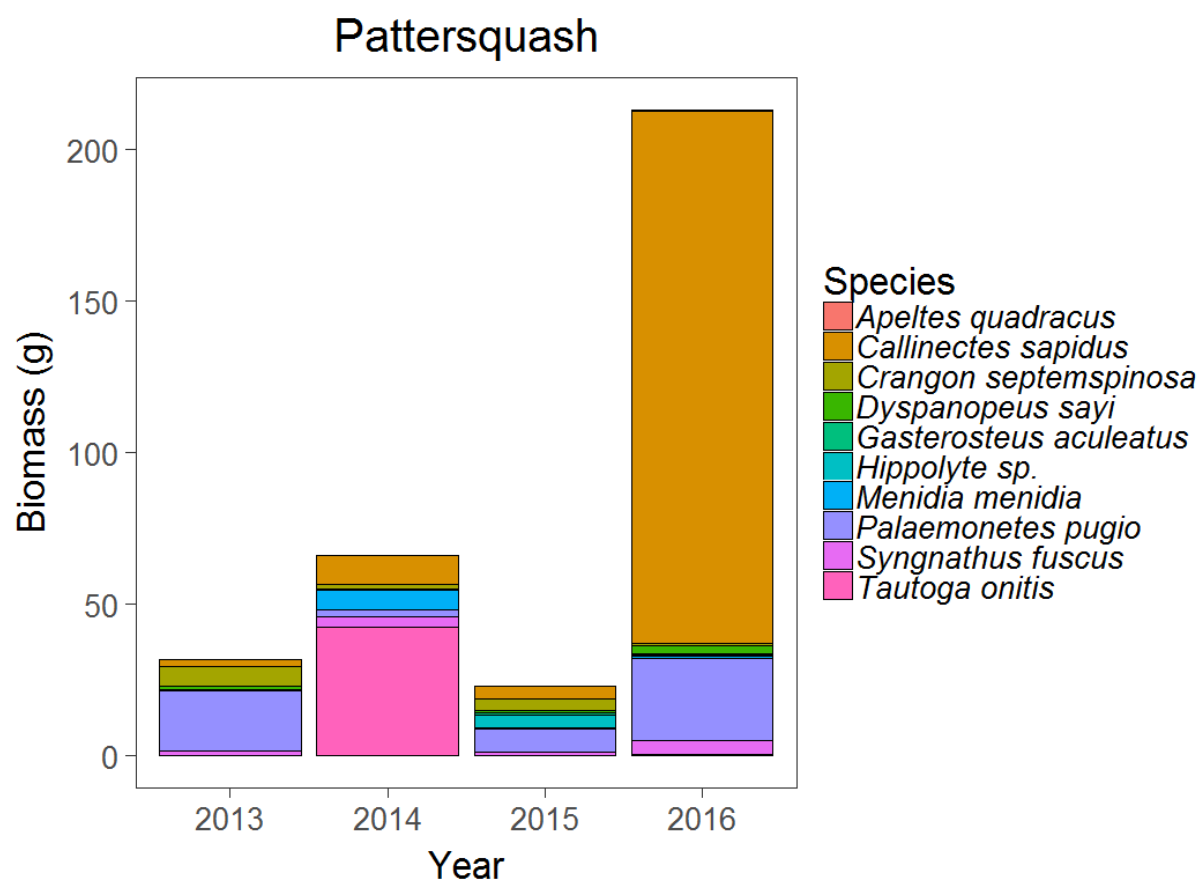
Appendix C Figure 1. Community composition (abundance) at Old Inlet using throw traps.



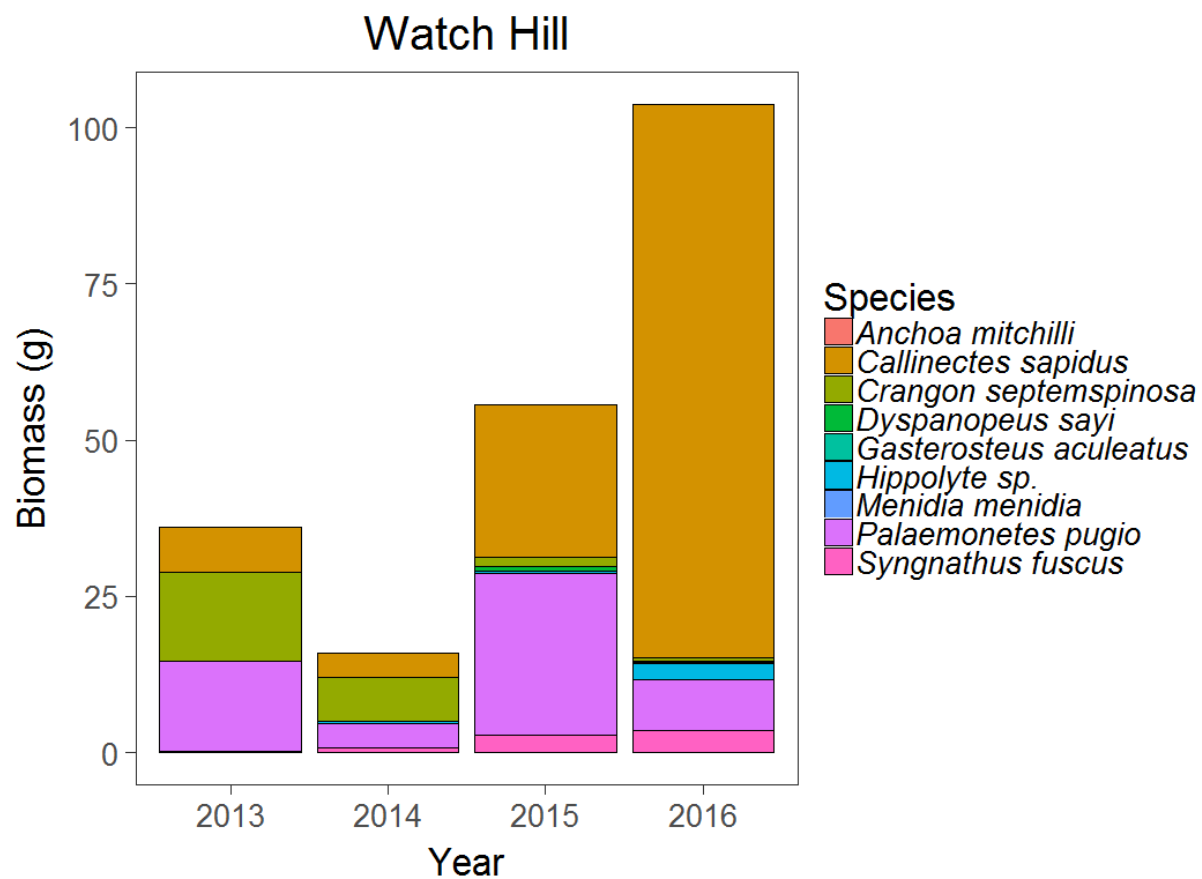
Appendix C Figure 2. Community composition (biomass) at Old Inlet using throw traps.



Appendix C Figure 3. Community composition (abundance) at Pattersquash using throw traps.

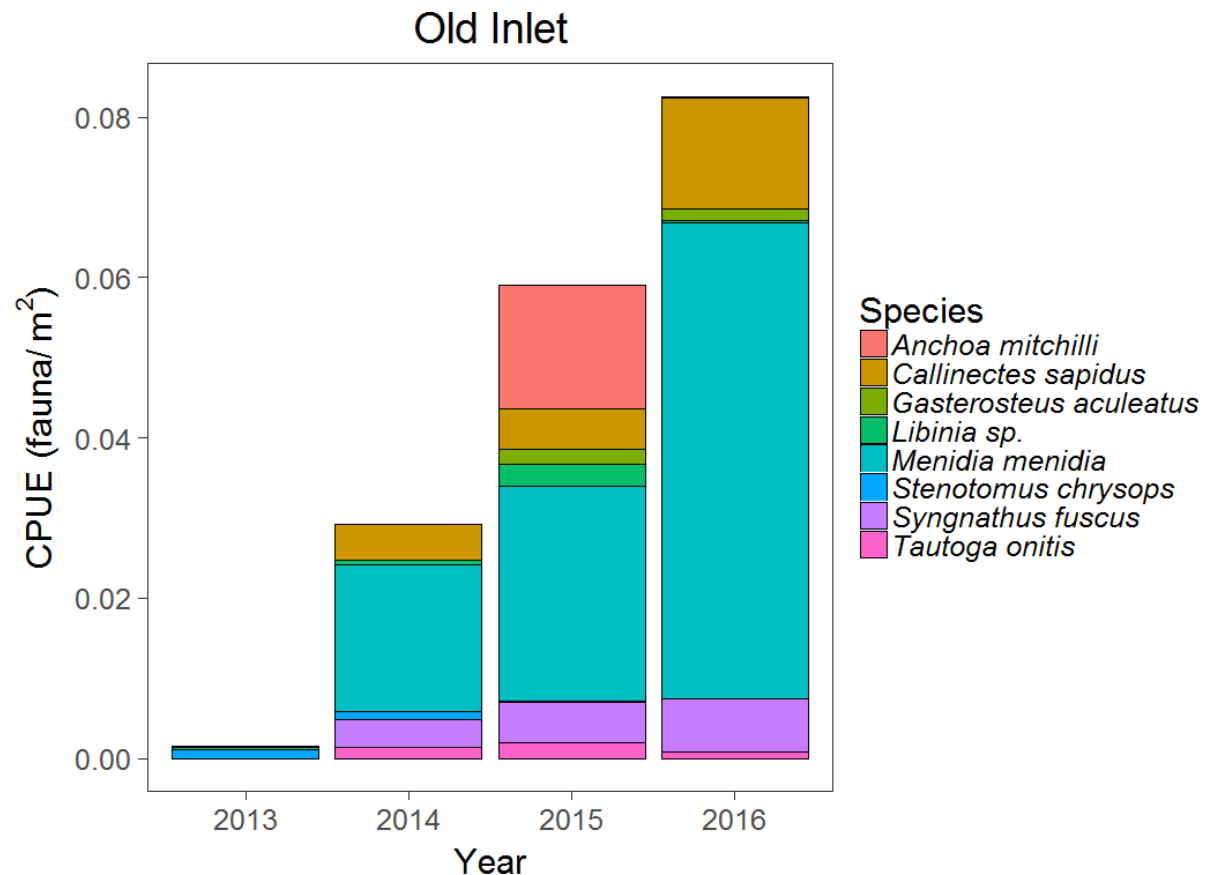


Appendix C Figure 4. Community composition (biomass) at Pattersquash using throw traps.

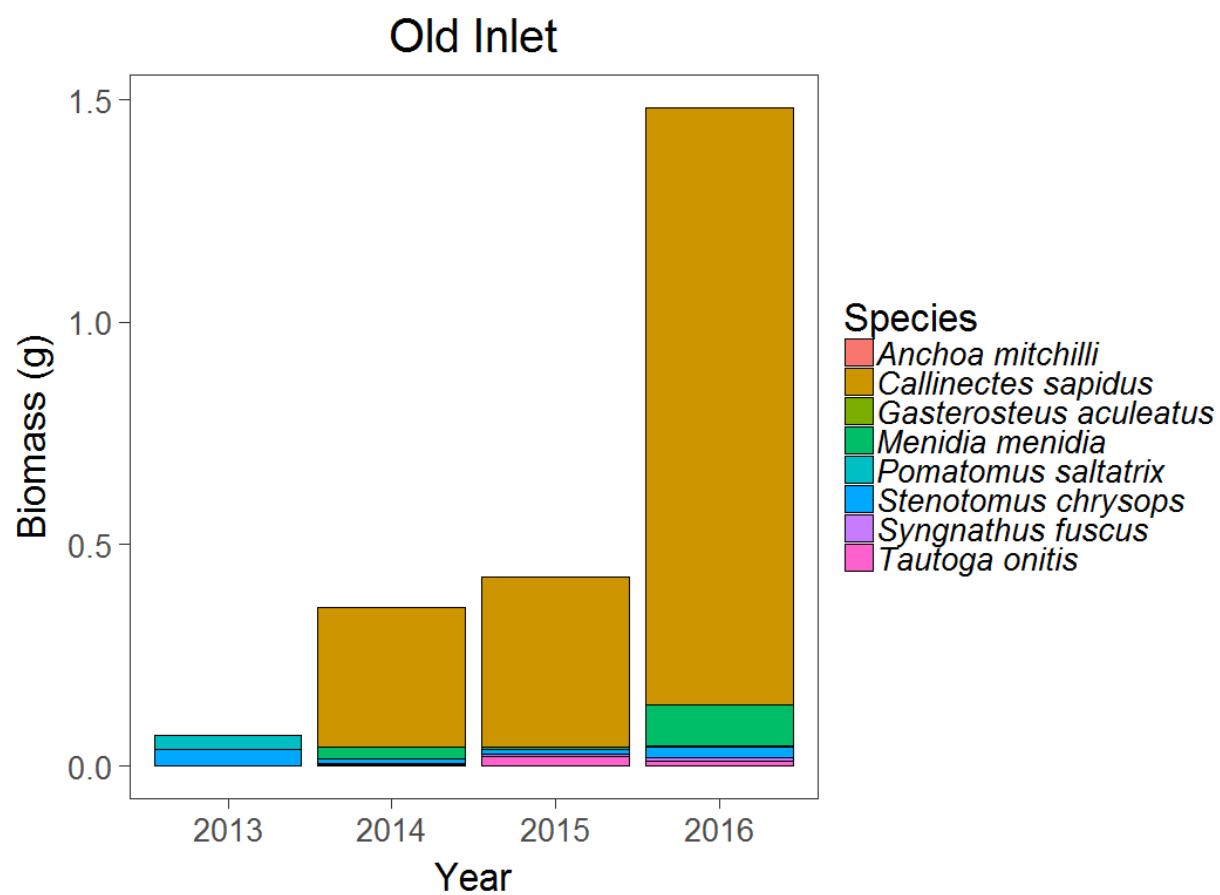


Appendix C Figure 5. Community composition (biomass) at Watch Hill using throw traps.

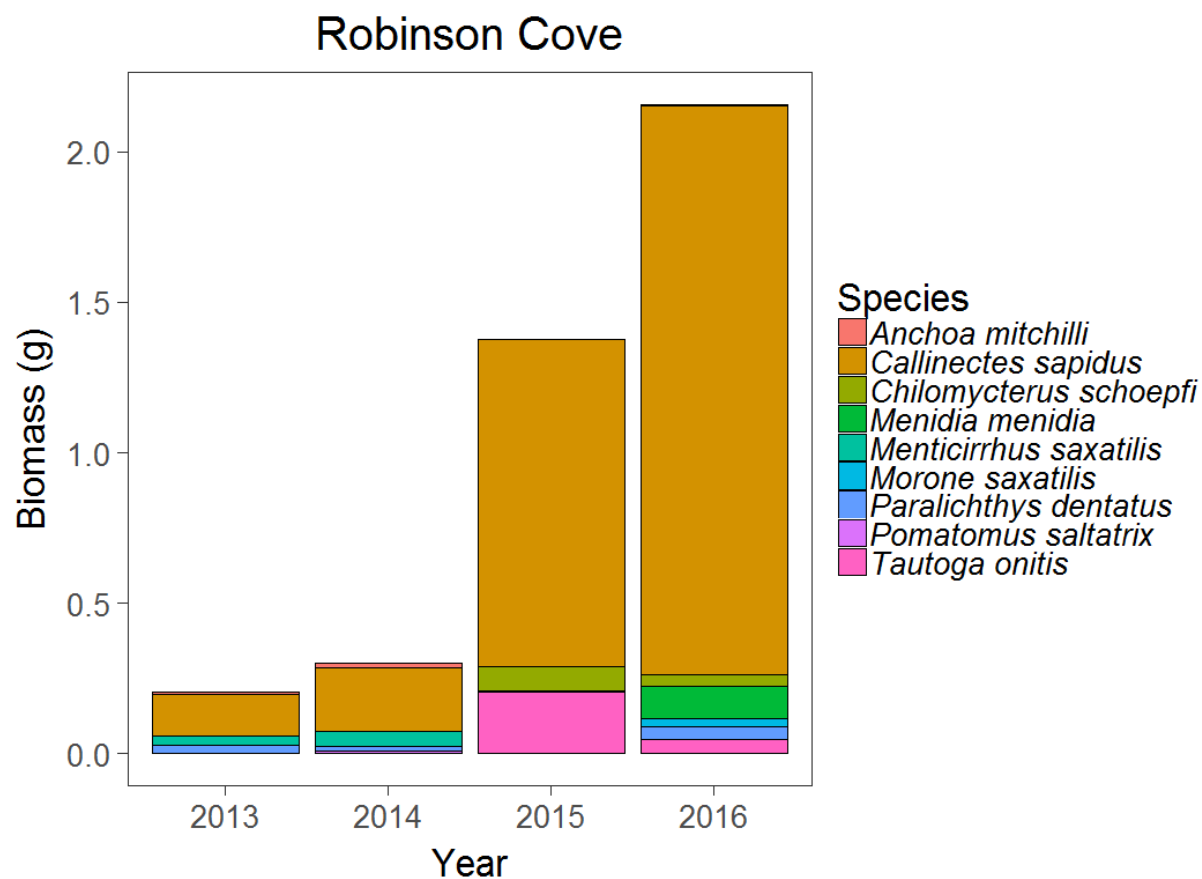
Appendix D. Community composition of mean abundance (fauna/m²) and biomass (g/m²) captured using otter trawls. Plots are only those in which there was a significance difference at a site over time (see Figure 2B and 2D).



Appendix D Figure 1. Community composition (abundance) at Old Inlet using otter trawls.



Appendix D Figure 2. Community composition (biomass) at Old Inlet using otter trawls.



Appendix D Figure 3. Community composition (biomass) at Robinson Cove using otter trawls.

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