8-2016

Experimental Assessment of Floating Seagrass Wrack as Potential Habitat for Benthic Organisms

Joseph R. McGinnis

Abstract

During the summer, seagrass blades are frequently released into the water column as a method to reduce respiration demands of the plant and through physical disruption of the bed (e.g., crab foraging, boat propellers). This wrack has the potential to serve as both food and habitat for organisms dislodged or actively moving within the system. The purpose of this research was to determine how benthic organisms may use floating wrack as a food resource and shelter. Three experimental floating wrack bag treatments were assembled containing using artificial *Zostera marina* (i.e., poly-ribbon), *Z. marina* blades, and a wrack bag control in order to examine if organisms prefer to use wrack for food, shelter, or both. Wrack bags were placed in Barnegat Bay, NJ during the summer of 2015 with collection and replacement of experimental bags occurring bi-weekly. Benthic core samples were taken as well to determine if the benthic organisms matched that of the fauna found in floating wrack. The major organisms identified in this study include Peracarid crustacean dominated by Corophiidae, Aoridae, Caprellidae, *Idotea balthica*, *Erichsonella* spp., Gammaridae, and Melitidae. In general, the results demonstrate a clear preference by amphipods and isopods for *Z. marina* wrack over the artificial *Z. marina*. The same taxa found in treatment bags were also found in core samples, albeit their relative abundances differed among the different taxa. The stable isotope analysis showed carbon signatures for faunal taxa similar to algae and/or *Ruppia maritima*. However, it appears that isopods showed combined N and C signatures similar to *R. maritima*, while the amphipods showed signatures closer to algal food resources. When assessing the overall results of my research, results showed that there was a lack of solitary response to the artificial *Z. marina*. Therefore, I conclude that benthic organisms use wrack as refuge

and potential transport mechanism, but also may obtain trophic resources from them. However, the trophic resources are not coming from *Z. marina*, but most likely from associated epiphytic algae. As such, there may be a potential benthic-pelagic link occurring due to a clear distribution of organisms from the benthos into the pelagic zone via floating wrack.

MONTCLAIR STATE UNIVERSITY

EXPERIMENTAL ASSESSMENT OF FLOATING SEAGRASS WRACK AS POTENTIAL HABITAT FOR BENTHIC ORGANISMS

by

Joseph R. McGinnis

A Master's Thesis Submitted to the Faculty of

Montclair State University

In Partial Fulfillment of the Requirements
For the Degree of

Master of Science, August 2016

College of Science and Mathematics

Marine Biology and Coastal Science

Thesis Committee:

[redacted]

Thesis Sponsor Dr. Paul Bologna

[redacted]

Committee Member Dr. Scott Kight

[redacted]

Committee Member Dr. Jorge Trueba

EXPERIMENTAL ASSESSMENT OF FLOATING SEAGRASS WRACK AS POTENTIAL HABITAT FOR BENTHIC ORGANISMS

A THESIS

Submitted in partial fulfillment of the requirements

For the degree of Master of Science

by

Joseph R. McGinnis

Montclair State University

Montclair, NJ

2016

Acknowledgements

I would like to give a special thank you to my advisor Dr. Paul Bologna for sharing his advice, wisdom, and help whenever I needed guidance or assistance. I would also like to thank my committee members Dr. Jorge Trueba and Dr. Scott Kight for helping me with my thesis and being there for me whenever I needed help. Another thank you goes out to Island Beach State Park and the New Jersey Department of Environmental Protection that allowed me to conduct my research within the park. A special thanks goes to my co-workers/friends within the Montclair State University Marine Laboratory for always being there for help, advice, and making me feel welcomed. A final thanks to my friends and mother who would volunteer and help me with my research whether it was carpooling or sampling.

Table of Contents

List of Figures

List of Tables

Introduction

Seagrass wrack is floating vegetation found in estuaries and oceans around the world. Seagrass wrack can be beneficial in many aspects like when it is collected for fertilizer and as a soil improver as seen in South Australia (Kirkman and Kendrick, 1997). Seagrass wrack coming from *Zostera marina* (eelgrass) can promote diversity in saltmarshes by shading soil and reducing physical stress and/or by providing nutrients to nutrient-poor soil (Chapman et al., 2004). The grasses may come to the surface after catastrophic events or in some cases increases in temperature causes blade release in order minimize respiration demands of the plant. Waycott et al. (2009) notes natural disturbances that are most commonly responsible for seagrass loss include hurricanes, disease, and grazing by herbivores. Human activities most affecting seagrasses are those which alter water quality or clarity and include nutrient and sediment loading from runoff and sewage disposal, dredging and filling, pollution, upland development, and certain fishing practices (Waycott et al., 2009). However, recreational activities like boating often result in blades being cut or entire plants being uprooted. As such, seagrass wrack can be substantial at times, but little research has been done to identify the role seagrass wrack may play in estuarine systems. However, another group of estuarine vascular plants, salt marsh grasses, also produce large quantities of floating wrack and they have been studied extensively.

The salt marsh plant *Spartina alterniflora* has been extensively researched in many aspects such as production in estuarine systems and fate and transport of wrack biomass (Teal, 1962; Squires and Good, 1974). Marsh plant's biomass entering detrital pathways has been proposed to explain high secondary production of estuarine

consumers (Nixon 1988; Keller et al., 1990; Mallin and Paerl, 1994; Deegan et al., 1995). Vascular plants are hard to break down because they have lignin and cellulose, but fungi and bacteria can degrade these plants. In aerobic conditions, fungi may break down cord grass faster than bacteria in estuarine systems (May, 1974; Sieburth et al., 1974; Gessner, 1977, 1978; Rublee et al., 1978). *Spartina* spp. is generally shed during the fall and winter times when the plants normally die off and are then deposited into estuarine systems. During this time, bacteria and fungi do not have high metabolic activity and break down the marsh grass slowly. During this period and into the spring, the salt marsh plant biomass can be liberated and enter coastal waters.

Through previous unpublished experimentation I conducted, it was determined that when *Zostera marina* grass blades are at the surface, there is a presence of benthic organisms such as amphipods and isopods. Benthic organisms could utilize the grass blades for food, protection, or potentially as a transport mechanism which provides both. The presence of these organisms shows that benthic fauna are an option for pelagic predators. Food type may be considered as a universal mechanism that partly determines the presence of grazers in seagrass habitats and is, in the absence of a predator, more important than shelter (Bostrom and Mattlic, 1999). The possibility of this can link interactions between benthic and pelagic food webs, which can help scientists determine food web dynamics between pelagic and benthic individuals via seagrass wrack.

Stable Isotopes are used as integrators and tracers of both natural and experimental ecological processes (Robinson, 2001). Knowing where food sources derive from is important in understanding trophic linkages. δ15N values are related to inorganic Nitrogen incorporation by seagrasses, algae, sediment, and the water column (Fourqurean

et al., 1997). δ13C values are determined primarily from photosynthesis, but seagrasses obtain their high values from their ability to use bicarbonate as an inorganic carbon source (Beer et al., 2002). Further analysis into both Carbon and Nitrogen values from this research can determine food sources of different invertebrates. If organisms are deriving nutrition from the floating wrack, then their isotopic signature should match to some degree the potential food resources in the system. Consequently, analyzing the stable isotopes of Carbon and Nitrogen can identify which invertebrates are utilizing what food sources. The δ13C of seagrasses is often depth-related and shows variations according to season, location, and community structure (Rose and Dawes, 1999; Boyce et al., 2001; Anderson and Fourqurean, 2003; Lepoint et al., 2003; Vizzini et al., 2003). Essentially, herbivorous organisms should show similarities in δ13C content to their food sources and relative enrichment in δ15N as you increase trophic level (Middelburg, 2014). Coupling these types of data with density of organisms in floating wrack should help to distinguish the relative importance of floating wrack as food and habitat for associated organisms.

Site Analysis and Research Objectives

The study site was located in Barnegat Bay adjacent to Island Beach State Park, New Jersey. The exact coordinates were 39.79074 degrees north, 74.09881 degrees west. The bay is relatively shallow (average depth at mean low water 1.7 m; Durrand, 1984) with water temperatures ranging from −2 to 28 °C (Able et al., 1992). Experiments were located about 10 meters off the coast line and the core samples were taken within seagrass beds close to the wrack experiment.

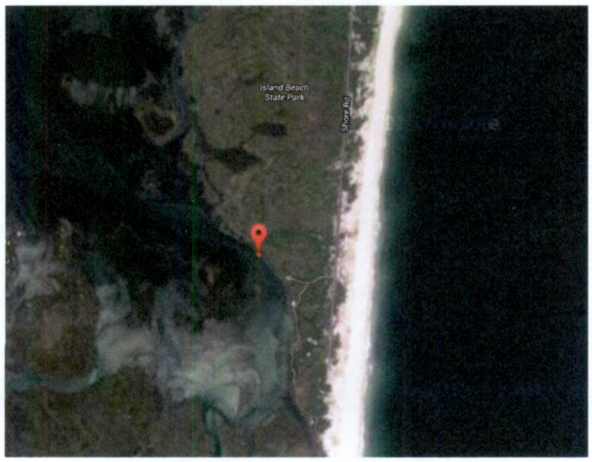

Figure 1. A google map representation of my study site. https://www.google.com/maps/place/39°47'25.9%22N+74°05'56.5%22W/@39.7899874,-74.0979248,306m/data=!3m1!1e3!4m2!3m1!1s0x0:0x0

Research Objective A: Floating seagrass wrack has the potential to support organisms seeking refuge, trophic resources, or both. If organisms are seeking refuge, there should be no difference between experimental treatments providing refuge with or without trophic resources.

Research Objective B: Do local populations of invertebrates act as the primary source of organisms occupying floating wrack. If they do, there should be a positive relationship between the abundance of these organisms in benthic cores and their relative abundance in floating wrack.

Methodology

Research Objective A

In order to analyze food and structure of seagrass wrack, a controlled experiment using a control leaf litter bag with nothing in it, a leaf litter bag with artificial *Z. marina* inside of it, and finally a leaf litter bag with natural *Z. marina* inside was conducted. Artificial plants have been used in past experiments to study ecological processes within seagrass ecosystems (Bell et al., 1985; Virnstein et al., 1986; Sogard, 1989). The blank litter bag was used as a structural control and consisted of a 400 cm^2 (20x20cm) plastic mesh 'envelope' (5 mm mesh). The other two treatments used the same envelope, but had either artificial *Z. marina* blades or field collected floating *Z. marina* blades. Artificial grass for *Z. marina* was constructed from green poly-ribbon. This ribbon was measured to 20 meters, which when compressed into a pitcher was measured to be approximately 500ml and placed into leaf litter bags. To construct the treatment using natural *Z. marina*, mesh litter bags were brought to the field where they were filled with ~500ml of *Z. marina* wrack to approximate the structural complexity provided by the artificial *Z. marina* treatments. Each litter bag was then attached to a buoy using a cable tie that was placed above a knot to prevent sinking of the bag and simulate floating wrack. Each individual litter bag treatment was then anchored to the bottom by tying the line to a submerged brick.

The in situ site of experimentation is located off of Island Beach State Park which contains *Z. marina* and *R. maritima* grass beds. Five empty litter bags (control), five litter bags with artificial *Z. marina*, and five litter bags with *Z. marina* were placed out into the

water (N=15). Experimental treatments were allowed to undergo colonization for approximately two weeks when treatments were retrieved and new, independent wrack bags were replaced (Table 1). When experimental treatments were collected, the wrack bag was placed directly into a Ziploc bag and then cut from the buoy assembly. Each treatment was labeled with date of collection and treatment and then transported to MSU for processing and evaluation. After return to MSU, samples were frozen until processed in the laboratory.

Table 1. Sampling Event timing and collection of experimental wrack bags. Due to extraneous events (e.g., storms, human curiosity), not all experimental treatment wrack bags placed were retrieved during the subsequent sampling time frame, resulting in unequal sample sizes for some events.

Date Placed in Field	Treatments Placed	Date Retrieved	Treatments Retrieved
June 17, 2015	5 Control 5 Artificial *Zostera* 5 Natural *Zostera*	June 30, 2015	5 Control 5 Artificial *Zostera* 5 Natural *Zostera*
June 30, 2015	5 Control 5 Artificial *Zostera* 5 Natural *Zostera*	July 14, 2015	5 Control 4 Artificial *Zostera* 4 Natural *Zostera*
July 14, 2015	5 Control 5 Artificial *Zostera* 5 Natural *Zostera*	July 29, 2015	4 Control 4 Artificial *Zostera* 5 Natural *Zostera*
July 29, 2015	5 Control	August 12, 2015	2 Control

	5 Artificial *Zostera* 5 Natural *Zostera*		3 Artificial *Zostera* 2 Natural *Zostera*
August 12, 2015	5 Control 5 Artificial *Zostera* 5 Natural *Zostera*	August 26, 2015	2 Control 4 Artificial *Zostera* 2 Natural *Zostera*
August 26, 2015	5 Control 5 Artificial *Zostera* 5 Natural *Zostera*	September 13,2015	4 Control 4 Artificial *Zostera* 4 Natural *Zostera*

Fauna Analysis

To determine the abundance of organisms associated with the floating wrack bags, samples were thawed and then processed in the following manner. Thawed samples were sieved through a graduated sieve series similar to the process outlined in Bologna (2006). Specifically, samples were sieved through 4mm, 2, mm, 1mm. 0.71mm and finally 0.5mm sieve. Each sieve size was then evaluated for each sample. Organisms were identified to lowest reasonable taxon and enumerated. All individuals from a given sample were then transferred into 95% ethanol for long-term storage.

Research Objective B

To assess the distribution and abundance of potential organisms which might be found in association with the floating wrack bag experiment, 6" benthic core samples were collected on the same dates that wrack bag experiments were placed/retrieved

(Table 1). Specifically, 5 core samples were collected by pushing the corer into *Z. marina* grass beds, capping and then removing the sample. In the field, the plant biomass was separated from the core and frozen until laboratory analysis. The remaining material (i.e., sediments and fauna) were field sieved through a 0.71mm sieve, placed into sample jars and preserved in 95% ethanol. Both types of samples were returned to MSU where they were either frozen (plant) or stained with Rose Bengal (fauna) prior to laboratory analysis.

Core Flora Analysis

The grass from experimental wrack bags was thawed and then placed on a tray. Spirorbids and other sessile objects attached to the grass were scraped off. After the grass was cleaned from the excess sessile objects and organisms, the grass was then placed into a large aluminum pan that was afterwards, dried in a drying oven at 80°C and weighed after a drying period of one week. Samples were then placed into a muffle furnace at 500°C for at least 8 hours which was afterwards removed and weighed to determine the ash weight. The ash weight was then recorded and the difference between dry weight and ashed weight provided the ash free dry weight (AFDW) of the sample. Grass from core samples followed the same cleaning process, but it was separated into above ground biomass (shoots), below ground biomass (roots and rhizome), algae, and detritus. The leaf shoot lengths and widths were recorded to the nearest millimeter before being placed into a pan and the total number of shoots in a sample was recorded. Each component of a grass core sample (i.e., above, below, algae, and detritus) followed the same drying and ashing process as the leaf litter bags.

Carbon and Nitrogen Isotope Analysis

Samples of *Z. marina* blades, *R. maritima* blades, green algae (*Ulva lactuca*), and several of the most common faunal organisms associated with the wrack were isolated and sent to Cornell University's Isotopic Analysis Laboratory. The fauna included *Idotea balthica*, *Erichsonella spp.*, Melitidae, Aoridae, and Caprellidae. At Cornell, samples underwent Isotopic analysis to determine their individual δ13C and δ15N content which could then show trophic linkages to a food resource. For the sample submitted for my research Cornell stated that they used, "the standard deviation for the internal Deer standard was 0.11 parts per thousand for Nitrogen 15 and 0.06 parts per thousand for Carbon 13" (Cornell Data Interpretation guide). Also according to the guide, the values obtained between "500mV and 14000mV for Nitrogen 15 had an error of 0.49 parts per thousand and 300mV and 12000mV for Carbon 13 had an error of 0.28 parts per thousand" (Cornell Data Interpretation Guide). Results from these analyses were then plotted to assess potential trophic linkages among the three primary producers and the content of the grazing organisms.

Statistical Analyses

To determine the spatial and temporal patterns of fauna using the experimental wrack bags and to assess these patterns in benthic cores, the following statistical analyses were conducted. A One-Way ANOVA (PROC GLM, SAS ®) with date of collection as the independent variable was used to assess differences among dates from core data related to plant biomass and animal densities. A Two-Way ANOVA with wrack bag treatment and date of collection as independent variables was used to assess differences

among treatments and date of collection for animals and determine if they demonstrate a preference among the treatments. Finally, relationships among organisms and floral biomass components within core samples were determined using a correlation analysis (PROC CORR, SAS ®) as well as relationships among organisms using the experimental wrack bags.

Results

Experimental Treatment Bags

Experimental Wrack Bags Results showed that significantly more organisms were collected from natural *Z. marina* compared to Artificial *Z. marina* and control treatments ($F_{2,49}$= 30.26, P< 0.0001; Figure 2). Additionally, there were significant differences among dates of collection ($F_{5,49}$= 23.5, P< 0.0001). Specifically, more animals were collected on June 30, August 26, and September 13 as compared to July 14 through August 12 (Figure 2). The most abundant organisms from samples were amphipod and isopod taxa including: Aoridae, Phoxocephalidae, Melitidae, Corophiidae, Gammaridae, Caprellidae, *Erichsonella spp.*, and *Idotea balthica*. Analyzing the temporal trends of wrack bags, natural *Z. marina* wrack bags are preferred over artificial *Z. marina* wrack bags and control bags from June until August, but in September there is an increase in organisms utilizing the artificial wrack bags. A full listing of all taxa identified during the experiment are listed in Appendix 2.

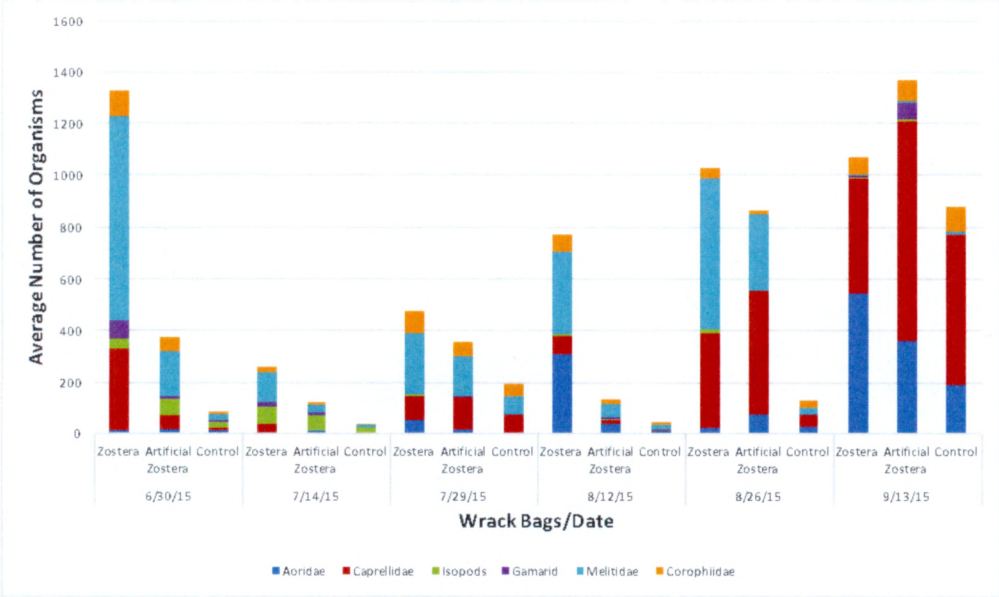

Figure 2. Temporal trends in the average abundance of dominant organisms within treatment bags for the dates of collection.

When assessing the organisms that caused the switch from *Z. marina* wrack bags to artificial wrack bags, it appears that Caprellidae populations cause the preference to alter. Without Caprellidae, as illustrated in Figure 3, the trend is normalized with *Z. marina* being the most preferred microhabitat as compared to artificial *Z. marina* and control treatment bags. Given that Caprellidae were equally abundant among all treatments, including the controls (Figure 2), their response is most likely a response to the mesh associated with the wrack bags and not indicative of any habitat preference.

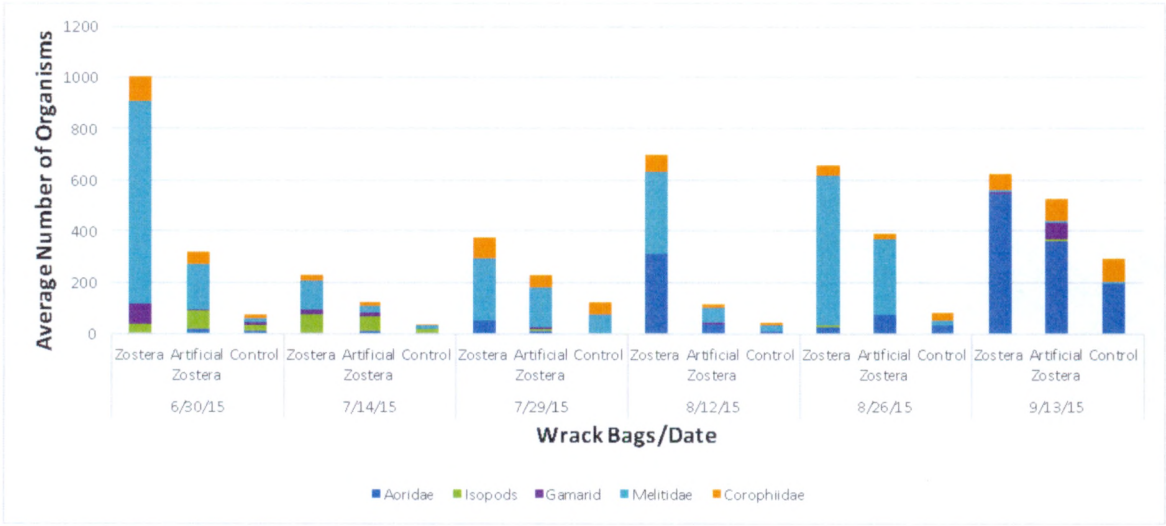

Figure 3. The average abundance of dominant organisms within treatment bags excluding Caprellidae.

Further analysis also shows that the most dominant organisms prefer *Z. marina* treatment bags over artificial and control bags. While overall results showed natural *Z. marina* treatment bags were favored over artificial and control bags, individual taxa also showed preferences associated with food and/or refuge (Table 2). Specifically, Aoridae were significantly more abundant in *Z. marina* bags and Artificial *Z. marina* bags as compared to controls ($F_{2,49}$=5.52, P< 0.0068; Table 2). Furthermore, Aoridae showed significant differences among collection dates ($F_{5,49}$=22.82, P< 0.0001) with significantly more collected in September compared to other dates (Figure 3). Melitidae were significantly more abundant in *Z. marina* and Artificial *Z. marina* treatments as compared to controls ($F_{2,49}$=64.07, P< 0.0001) (Table 2.). Melitidae also were significant in terms of collection dates ($F_{5,49}$=25.12, P< 0.0001) with the majority of Melitidae collected in every date except September (Figure 3). Corophiidae populations were collected in *Z. marina* and Artificial *Z. marina* treatments equally, but were found less or about at equally as much in control bags ($F_{2,49}$=64.07, P< 0.0001; Table 2). Corophiidae were significant in terms

of collection dates ($F_{5, 49}$=3.69, P< 0.0064) with significantly fewer being collected in September.

Table 2. Prevalent Taxa relative preference among experimental treatments. Significance levels provided as: * P< 0.05, ** P< 0.001, *** P< 0.0001; with significant differences distinguished as >. NS indicates no significant differences, but order of preference is based on total abundance.

ABUNDANT TAXA	Treatment Preference
Aoridae*	*Z. marina* > Artificial *Z. marina* = Control
Melitidae***	*Z. marina* > Artificial *Z. marina* = Control
Corophiidae*	*Z. marina* = Artificial *Z. marina* ≥ Control
Gammaridae NS	*Z. marina* = Artificial *Z. marina* = Control
Caprellidae NS	*Z. marina* = Artificial *Z. marina* = Control
Erichsonella spp. NS	*Z. marina* = Artificial *Z. marina* = Control
Idotea balthica NS	*Z. marina* = Artificial *Z. marina* = Control

The correlation analysis showed numerous relationships among the taxa identified (Table 3). Among all taxa, the only significant relationships were positive. Specifically Aoridae, Phoxocephalidae, Melitidae, and Gammaridae were all significantly related to one another. Melitidae also showed significance with Caprellidae. Corophiidae, *Erichsonella* spp., and *Idotea balthica* showed no significant relationships with any other organisms.

Table 3. Correlation analyses among taxa identified associated with the experimental wrack bags. Values in table represent R-values between individual taxa. Significant relationships designated as * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$. Taxa abbreviations: Aor=Aoridae, Phoxo=Phoxocephalidae, Melit=Melitidae, Coro=Corophiidae, Capre=Caprellidae, Gam=Gammaridae, Erich=*Erichsonella*, IdBal= *Idotea balthica*

	Aor	Phoxo	Melit	Coro	Capre	Gam	Erich	IdBal
Aor	1	0.68***	0.64***	-0.11	0.47	0.84***	-0.11	-0.2
Phoxo	0.68***	1	0.55**	-0.02	0.29	0.66***	0.33	0.14
Melit	0.64***	0.55**	1	0.04	0.91***	0.44**	0.1	-0.09
Coro	-0.11	-0.02	0.04	1	0.16	-0.14	-0.08	0.38
Capre	0.47	0.29	0.91***	0.16	1	0.27	-0.09	-0.05
Gam	0.84***	0.66***	0.44**	-0.14	0.27	1	-0.06	-0.07
Erich	-0.11	0.33	0.1	-0.08	-0.09	-0.06	1	0.16
IdBal	-0.2	0.14	-0.09	0.38	-0.05	-0.07	0.16	1

Temporal Trends for the Dominant Taxa

Melitidae Population Trends in Treatment Bags

The trend in Melitidae suggests a strong preference for *Z. marina* wrack (Figure 4), but also a seasonal signal. Melitidae showed significant differences among dates of collection and treatment type ($F_{5,49}=38.97$, $P< 0.0001$, $F_{2,49}=610.5$, $P< 0.0001$). Initial colonization was high in June with a rapid drop in early July. However, their abundance in the floating wrack continued to increase throughout the summer then declined dramatically in September. This suggests that they may be active movers during the summer, but as fall approaches they actively seek refuge in the benthos.

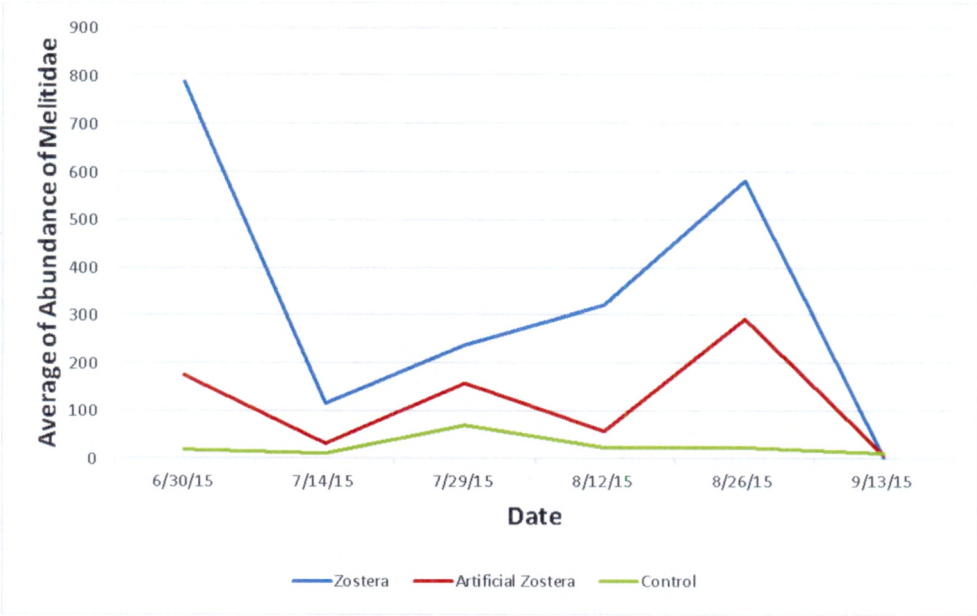

Figure 4. Temporal trends in Melitidae populations from the Experimental Wrack Bags.

Aoridae Population Trends in Treatment Bags

The Aoridae population within treatment bags follows the standard trend of *Z. marina* treatment bags being preferred over artificial *Z. marina* and control (Figure 5). Aoridae populations show significant differences among dates and treatment type ($F_{5,49}$=32.5, P< 0.0001, $F_{2,49}$=6.4, P< 0.0004) with significantly more occurring in September and in *Z. marina* experimental treatments. The abundance spike on August 12, 2015 in the *Z. marina* treatments is unexplainable, but might reflect a storm event during that time dislodging benthic individuals who colonized the treatment providing refuge and trophic resources.

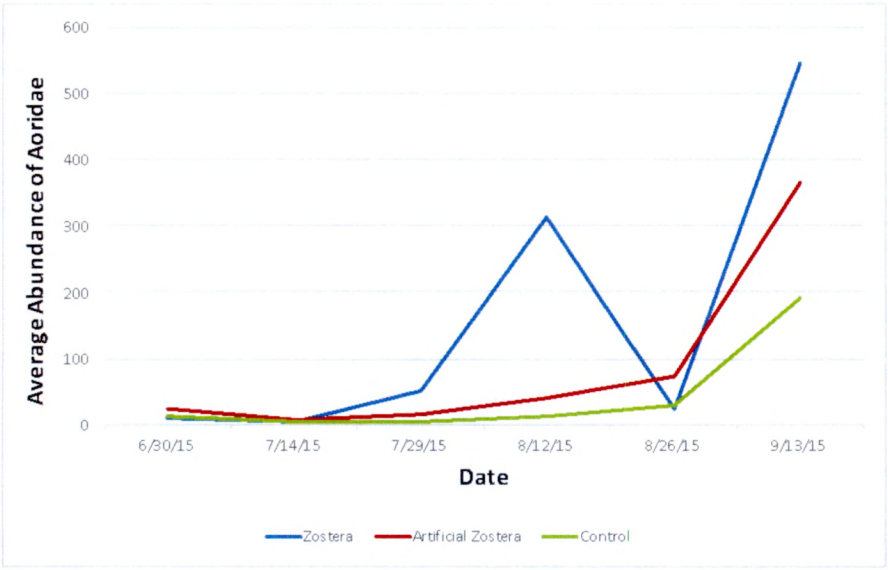

Figure 5. Temporal trends in Aoridae populations from the Experimental Wrack Bags.

Caprellidae Population Trends in Treatment Bags

Caprellidae population trends show initial colonization of wrack bags in June with a preference for *Z. marina*, but later in the experiment there is a massive and significant increase in their population in September ($F_{5,49}$=6.6, p<0.0001), with no apparent habitat choice and the lowest abundances occurring in the *Z. marina* treatments (Figure 6). This increase in abundance most likely is a result of seasonal reproduction and their lack of habitat preference may reflect their small size and the mesh associated with the mesh bag providing adequate refuge quality.

Figure 6. Temporal trends in Caprellidae populations from the Experimental Wrack Bags.

Core Samples

Floral Analyses

The biomass of above and below ground *Z. marina* from core samples appears to increase from June into July, but then decreases from August into September (Figure 7) and suggests a typical seasonal growth pattern. Along with this pattern is the increase in detrital material observed in August, suggesting the release of *Z. marina* leaves accumulating on the bottom. In terms of plant biomass, there was no significance difference between below/above ground biomass, detritus, or algae among all the dates. However, there was positive correlation between below and above ground biomass of *Z. marina* (r value= 0.4768 $P< 0.0067$).

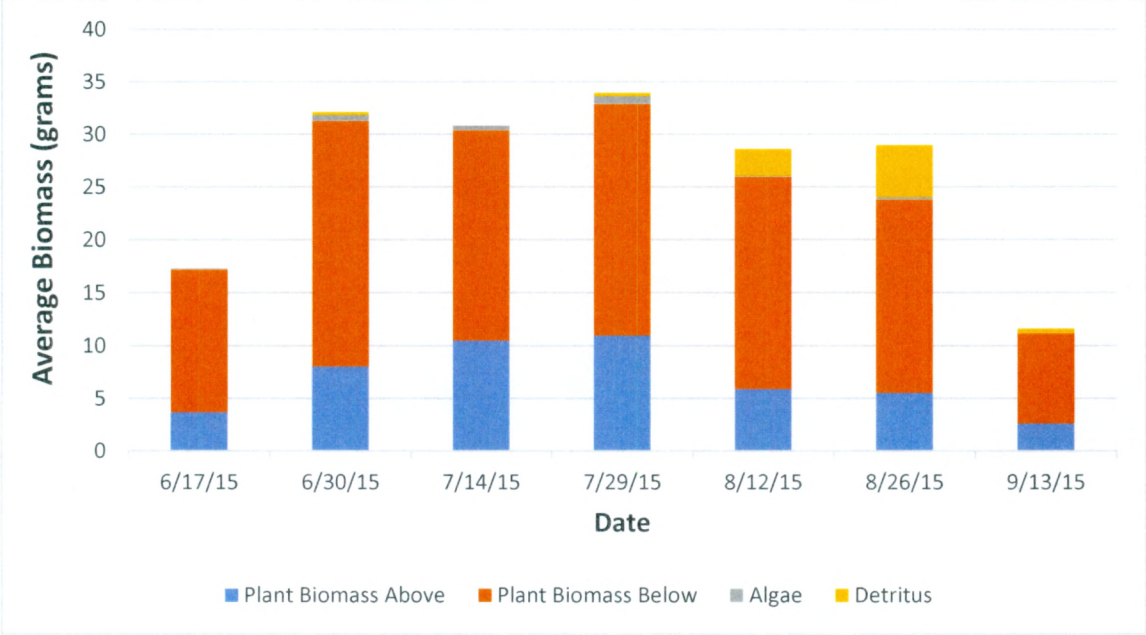

Figure 7. Illustrates the average above/ below ground AFDW biomass of *Z. marina*, algae, and detritus recovered from core samples against the date per meter square.

Core Faunal Analyses

The overall density of major taxa from benthic cores shows substantial variability from June through to September (Figure 8). This may reflect natural reproductive cycles, emigration, predation, or mobile organism leaving the floating wrack and seeking refuge in the benthos to over-winter. Aoridae were very abundant in core samples showing significant increases in September ($F_{6,31}$=3.28, $p<0.016$). Isopods (both *Idotea balthica* and *Erichsonella* spp.) were collected in cores in early summer, but were relatively absent in collection dates past June. *Erichsonella* spp. doesn't show any significance with date, but *Idotea balthica* does ($F_{6,31}$=7.37, $p<0.0001$). Corophiidae show significantly higher average abundance on June 30, 2015 and low abundance for other dates ($F_{6,31}$=4.24, $p<0.0044$). Gammaridae population sizes were minor within core samples

and showed no significance among dates. Other organisms found within core samples such as decapods, gastropods, Polychaeta, and bivalves were not as abundant as the amphipods and isopods found in the core samples and showed no significant temporal patterns. Appendix 1 provides a listing of all taxa collected during core sampling.

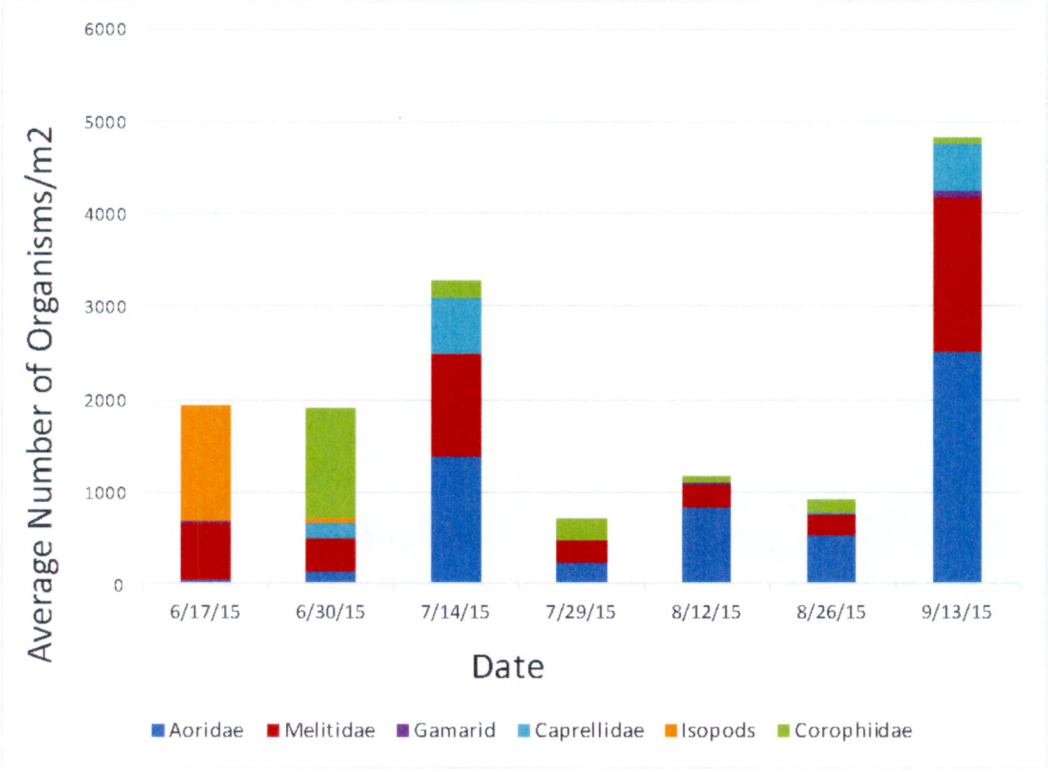

Figure 8. The average number of dominant organisms within core samples is represented by the average number of organisms against the date at which each organism was sampled.

Comparison of Core and Wrack populations

Three dominant taxa found within both treatment bags and core samples were compared to determine if there is a possible migration between treatment bag populations and the benthos. In the case of Aoridae, there appears to be similarities in both treatment

bags and core samples, with elevated abundances in September, but a spike in the *Z. marina* wrack on August 12, 2015 (Figure 9). This may reflect overall population increases and possible dispersal in the fall or competition for space and displacement from the benthos in August.

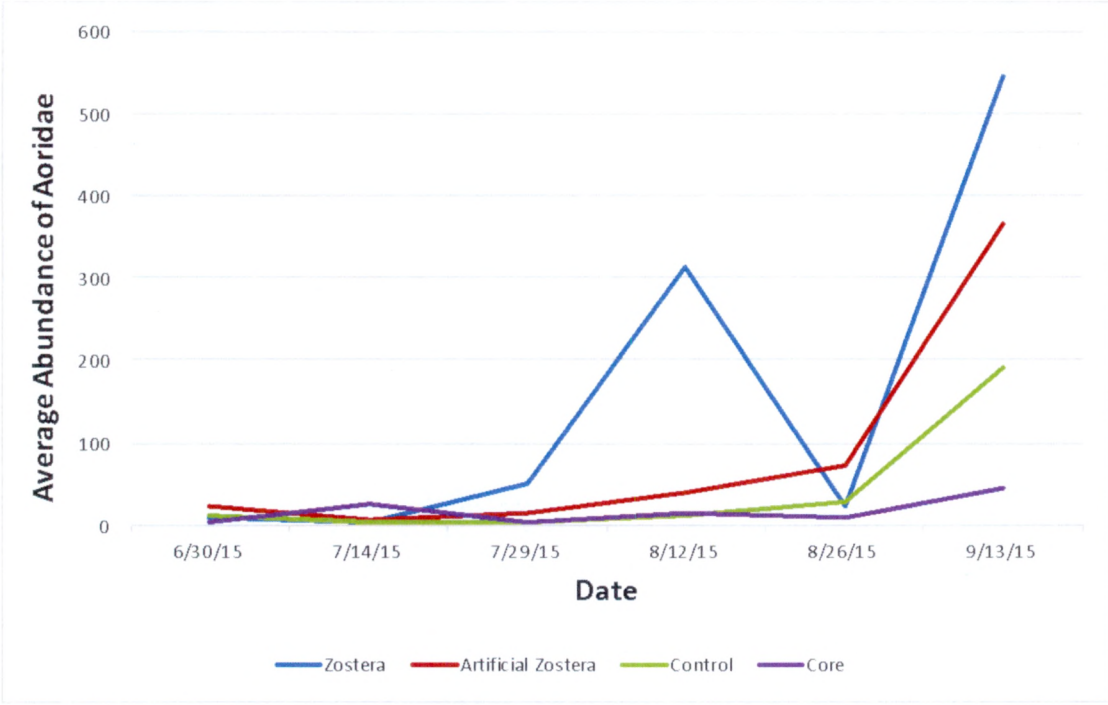

Figure 9. Average abundance of Aoridae from treatment bags and core samples.

Caprellidae were found in both core and treatment bag samples (Figure 10), but were abundant only in wrack bags. Additionally, they showed significant differences among dates in experimental wrack bags ($F_{5,49}$=5.24, p<0.0006), meaning that populations are primarily found there and dominated the September samples for all wrack bag results (see Figure 2). Caprellidae are epiphytic on seagrass blades, so cores may not reflect their distribution in the water, but floating structures seem to be a valuable habitat for these amphipods. Additionally, they were equally abundant on all treatments

including the controls so they may merely be reacting to the structure of the mesh envelope.

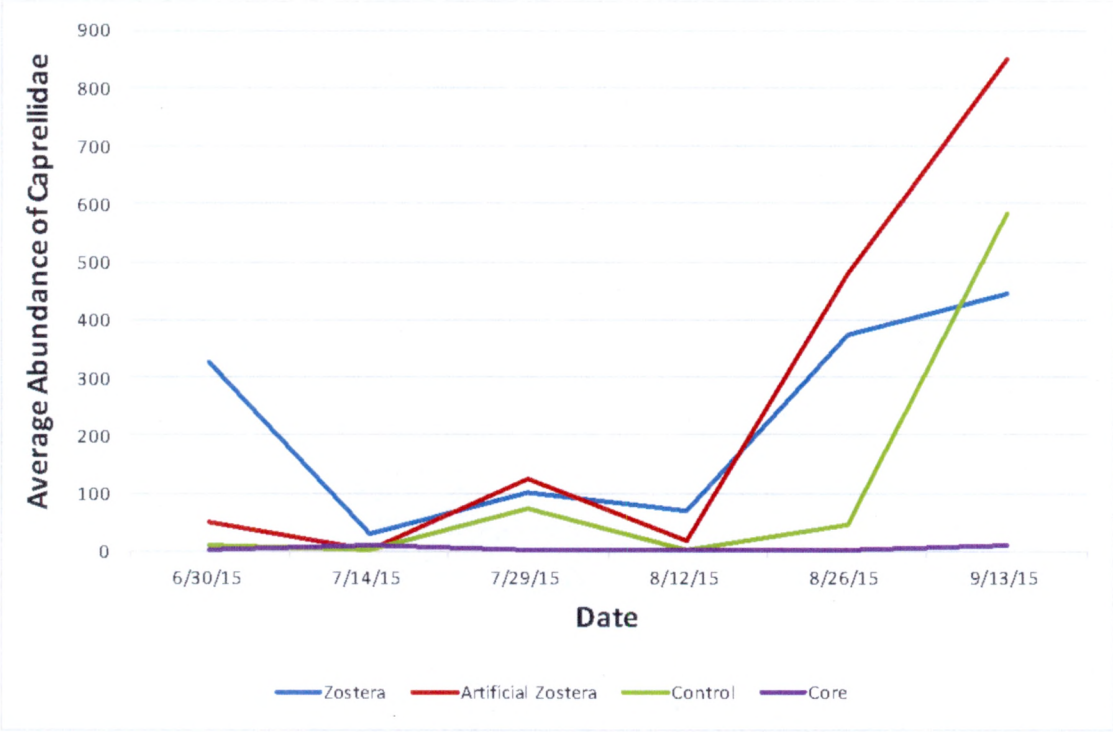

Figure 10. Average abundance of Caprellidae from treatment bags and core samples.

When examining Melitidae, they showed a significant preference for *Z. marina* wrack (Figure 11). Additionally, there may be active migration occurring between the water column and the benthos. Specifically, there is a general inverse relationship between abundance in wrack experiments and the benthos. This is especially prevalent in September when abundance in the wrack drops substantially, but increases dramatically in the core sample. This suggests active migration from the water column to the benthos which may signal a preparation strategy for over-wintering.

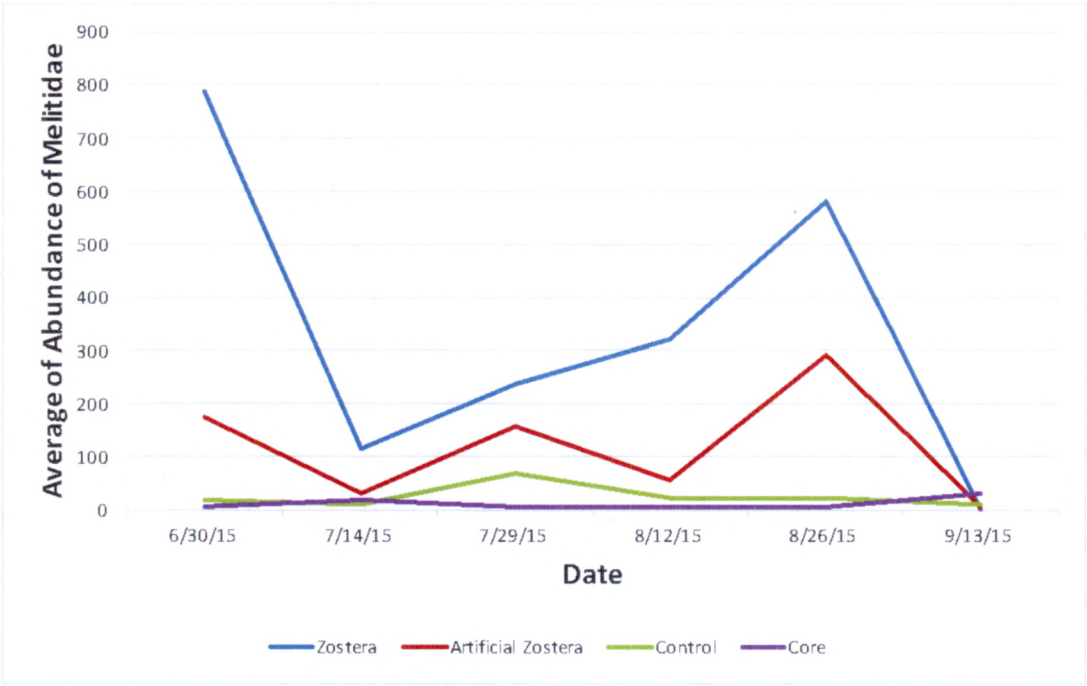

Figure 11. Average abundance of Melitidae from treatment bags and core samples.

Stable Isotope Analysis

Results from the isotopic analyses show variation among the fauna tested and the potential food resources (Figure 12). In terms of food resources, δ13C results show enrichment by *Z. marina*, followed by lower values for *R. maritima* and the lowest values for *Ulva*. For the fauna tested, the amphipods showed signatures similar to *Ulva* for Carbon (-15 to -20), while the isopods showed some similarities to *R. maritima*. (-11 to -15). In terms of Nitrogen enrichment and trophic placement, *Ulva* showed wide ranges of N content, but *R. maritima* was centered at ~5.5 and *Z. marina* was at ~8. Given the expectations of carbon similarities and nitrogen enrichment affiliated with stable isotopes, none of the fauna tested showed signatures suggesting that *Z. marina* was an important component in their diet. Rather, it appears that amphipod food resources were

dominated by algae and isopods showed signatures similar to expectations for feeding on *R. maritima* and possible mixed diet with algae (Figure 12). These results suggest that the response of organisms affiliated with the wrack bag experiment were most likely due to the potential presence of epiphytic algal food resources on natural *Z. marina*, rather than obtaining trophic resources directly.

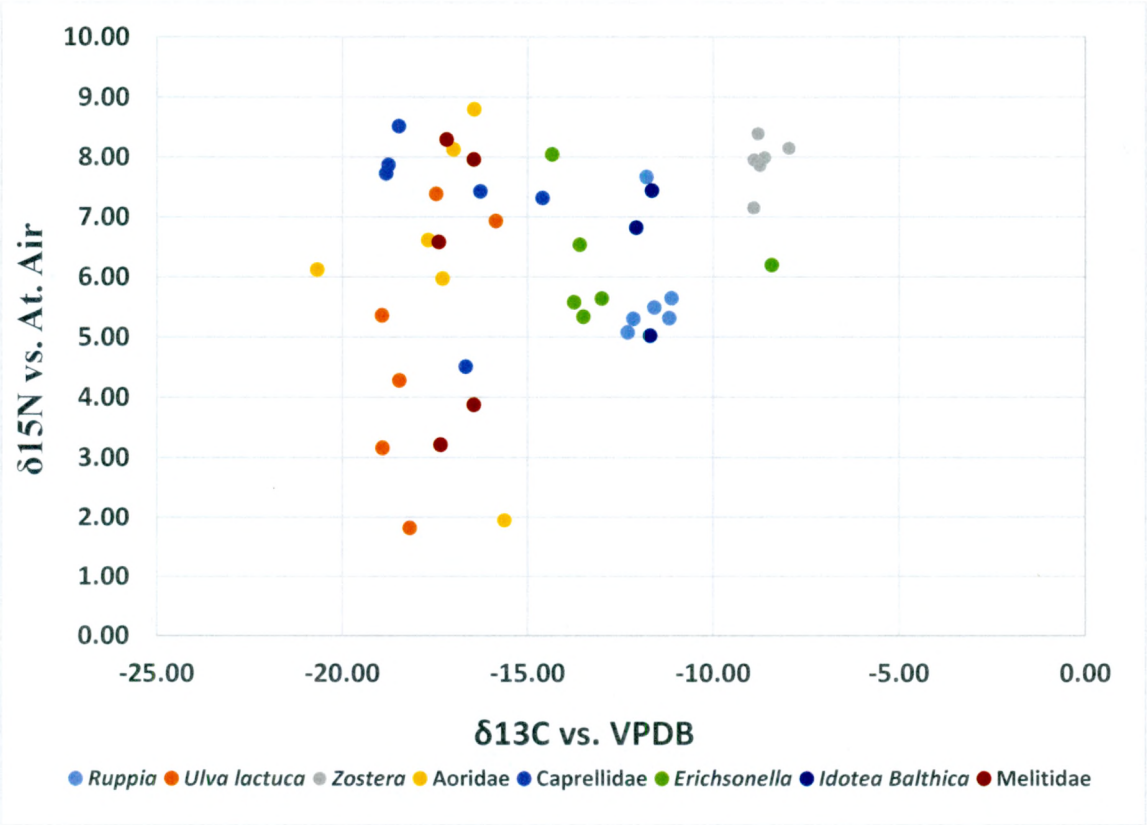

Figure 12. δ13C and δ15N values are represented on the x and y axis. Food sources and invertebrates were plotted along their corresponding δ13C and δ15N values.

Discussion

Zostera marina as a habitat supports large and diverse faunal assemblages (Thayer et al., 1984; Heck et al., 1995). Its biomass represents both refuge and trophic resources for many organisms (Best et al., 2013; Duffy et al., 2015), but when this biomass is released to the water column, its value is relatively unknown. Results from my core samples contained numerous taxa, but were dominated by several Peracarida taxa including Corophiidae, Caprellidae, Aoridae, Gammaridae, *Erichsonella* spp., and *Idotea balthica*. When analyzing the dominant taxa within core samples, the relative abundance of various organisms shifts during the summer. Initially, isopod abundance is high, but diminishes in July when Aoridae and Melitidae abundances increase. In August, all taxa density is reduced but recovers in September (Figure 8). These patterns are somewhat similar to plant biomass, which showed increases in the early summer, but reductions in August (Figure 7). Consequently, there may be some underlying structural or trophic interactions occurring in benthic cores driving faunal abundance.

Biomass from core samples did not correlate to any organism or show any significance in organism abundance. One may assume that the greater biomass may result in more places to hide from predators or more epiphytes that can be attached resulting in a better food resource (Orth and van Montfrans, 1984; van Montfrans et al., 1984). Based on the core data, below ground biomass did not change dramatically during the season, but large reductions in above ground biomass occurred in August, and this corresponds to the reduction in the densities of benthic organisms.

It is well recognized that organisms may not use *Z. marina* for a direct food resource, but for the epiphytes among the grass blades (Caine, 1980; van Montfrans et al.,

1982). Invertebrates are known to graze on seagrasses for epiphytic algae (Kitting et al., 1984) and the results from the stable isotope analysis support this (Figure 12). Studies in seagrass meadows have shown that epiphytic primary production may often exceed seagrass production in both weight specific (Pollard and Kogure, 1993) and total annual production (Morgan and Kitting, 1984; Moncreiff et al., 1992). Additionally, experimental work by Bologna and Heck (1999) demonstrated that the critical factor for mobile epifauna was the presence of a natural epiphytic algal community on artificial seagrass blades and not the structural resources that they provide. If the reduction in *Z. marina* biomass impacted the epiphytic algal resources, this might suggest that the fauna is merely responding to reduced food resources. The stable isotopic analyses suggest that most amphipods are receiving their nutrition from algal resources and not *Z. marina*. Consequently, this supports the reduction in faunal density in August relating to reductions in *Z. marina* above ground biomass as a corollary with potential epiphytic algal resources. A reduction in biomass may also reduce the refuge capacity of the grass bed as well. As such, multiple system factors may be influencing the patterns observed. In a similar *Z. marina* habitat in Barnegat Bay, Bologna (2006) showed trophic cascades in two difference *Z. marina* grass beds relating to the relative abundance of large predators. While large fish predators were not assessed in this research, the potential that the coupled lower trophic resources and reduced refuge quality could both explain the drop in faunal density in August. The reduction in plant biomass in August may also signal greater wrack volume in the system as plants compensate for respiration demands.

It is known that seagrass detritus can be transported by currents and waves (Robertson and Lucas, 1983; Hemminga et al., 1994). Organisms can be associated with

this floating wrack and can be added into the food chain. The production value of eelgrass does not come from the grass directly, but rather the epiphytes attached (Heck et al., 2008). Epiphytes can be eaten directly by small benthic organisms which increases the trophic interactions occurring (Orth et al., 1984; Greenway, 1995). My isotope analysis concurs with these findings. It appears that amphipods are generally receiving their nutrition through algal resources (Figure 12), but the trophic pathway for isopods is not quite as clear. According to their C13 content, both *Idotea balthica* and *Erichsonella* spp. appear to derive their carbon from a mixed diet of *R. maritima* and algae, but the N15 signatures point to more algae in their diet. While these results suggest a more robust diet for isopods, greater research is needed to fully evaluate the trophic interactions. Consequently, there is a great deal of research that is needed to fully understand the role of eelgrass biomass in coastal systems. In particular, there is little research to determine the full value of seagrass wrack production, but analysis of my experiment shows the potential value of seagrass wrack.

Results from the experiment show a generalized pattern where taxa appear to show a preference for *Z. marina* wrack over artificial *Z. marina* wrack bags and control bags. *Zostera marina* appeared to be preferred due to a probable food component (i.e., epiphytic algae), while artificial *Z. marina* may be utilized as a refuge. When evaluating individual groups, Melitidae populations in natural *Z. marina* treatment bags appeared to be supported and could reflect both refuge and trophic interactions, but could also signal potential emigration using wrack. This corresponds to higher abundances in the wrack during July and August, but their virtual disappearance in September (Figure 4). At the same time, Melitidae densities dramatically increase in core samples from September,

suggesting they have stopped emigrating and are returning to the benthos to over-winter (Figure 11).

Aoridae populations appear to follow a pattern that showed natural wrack treatment bags have more organisms than artificial bags and control but Aoridae correlation to Melitidae populations is an interesting component to this research. The shift of dominant taxa within a sample from Melitidae to Aoridae dominating may be represented by carrying capacity issues, competition for resources (MacArthur 1969), or other issues that may not involve any interaction. Another explanation may be the two species may follow niche partitioning (Schoener, 1982). When an organism disappears from a habitat, there may be a niche that can be filled. If Melitidae are top consumers in wrack, when they are not present, Aoridae may step in to fulfill their consumer role (Jackson et al., 2001).

The unusual patterns noted in the treatment bag portion of the experiment derive from Caprellidae. These organisms cause the abundance of organisms sampled within artificial treatment bags to peak. After removing skeleton shrimp, treatment bag trends then followed an expected pattern of greater individuals associated with *Z. marina* wrack then artificial *Z. marina*. An explanation for Caprellidae may explain why these dictated patterns in treatment bags might be due to abiotic factors. According to Virnstein et al. (1984), "Caprellidae like to live on top of seagrass blades". If seagrass is exposed to too much sunlight and temperatures fluctuate causing seagrass leaves to detach, Caprellidae may be swept to the surface which would force populations to look for structure within wrack. Organisms that may cut away seagrass (i.e., blue crabs) or any other organism that

may disturb seagrass meadows (i.e., waterfowl), may cause populations of Caprellidae to relocate into the pelagic zone via floating wrack.

The similarity in taxa between experimental treatments and core samples demonstrates that there is a probable link that the organisms utilizing wrack originate from nearby benthic sources. Organisms may be moving from seagrass habitats into wrack bags as a food resources or refuge structure if dislodged from the benthos. A study conducted by Hyndes and Lavery (2005) demonstrated that fish in the near shore waters consume mainly small crustacean and polychaete prey. The fishes rely on the consumption of invertebrates that graze either directly on fresh plant material or indirectly through detritus, which was determined through isotopic analysis. The Amphipods food source of algae was similar in this experiment to experimentation done by Smit et al. (2005). Isopods utilizing seagrass as food was also noted in research elsewhere (Bostrom and Mattila, 1999; Wootton and Bologna, 2004) and results from my work showed isopods had signatures similar to *R. maritima* mixed with algae (Figure 12). Other studies show support that fish rely on this benthic pelagic coupling for a food source in dislodged near shore seagrass wrack invertebrates (Robertson and Lenanton, 1984).

Experimentation on floating wrack demonstrated that *Z. marina* wrack is preferred by organisms that utilize the seagrass epiphytic food resources (e.g. Melitidae and Aoridae populations) and refuge (e.g. Caprellidae population, sensu Virnstein et al., 1986). The food source on wrack is associated epiphytes, as they are the usual food source for macrofauna (Kristensen, 1972; Kitting, 1984; van Montfrans et al., 1984;

Nielsen and Lethbridge, 1989). The composition of organisms within the wrack was the same as that found within core samples. Why these organisms may move between the benthos and the floating wrack suggests multiple driving factors which may be taxa specific. The populations moving from the benthic zone into the pelagic zone may be due to predation related issues (Shurin et al., 2002; Morin, 2011), a lack of food resources, a lack of space generated from competitive interaction, active dispersal, or stochastic events merely dislodging them from the bottom. However, organisms moving from the pelagic zone into the benthos may be due to colonization of new habitat (e.g., immigration), predation avoidance, or survival strategies for overwintering.

Most studies of fauna and wrack have examined the role of the fauna (i.e. isopods, amphipods, oligochaetes and insect larvae) in breaking down wrack, particularly on sandy beaches (Inglis, 1989; Columbini et al., 2000). Transportation of organisms can establish new populations in different parts of a coastal embayment, coastline, or other areas that wrack may accumulate. When new individuals are established, this can lead to increased genetic diversity and population robustness.

Assessing patterns at time and spatial scale in seagrass habitat should be addressed to understand how the communities function (Bologna et al., 2007). This research found that *Z. marina* wrack is utilized mostly by amphipods for the algae associated with *Z. marina,* but more research must be conducted to analyze wrack as a habitat and feeding area for organisms. Furthermore, additional research on seagrass wrack is needed to understand how it can function as a habitat for benthic-pelagic coupling. Specifically, assessment of the epiphytic algal communities on seagrass wrack

with regards to biomass and isotopic signatures would be useful in establishing trophic links. Future research should be able to use these results to also determine why seagrass wrack is a sufficient habitat for benthic organisms and its potential to act as a dispersal mechanism for benthic organisms.

REFERENCES

Able KW., Hoden R., Witting D., and Durand J. 1992. Physical parameters of the Great Bay-Mullica River estuary. *Technical Report* 92-06, pp. 38.

Anderson WT., Fourqurean JW. 2003. Intra- and inter annual variability on seagrass carbon and nitrogen stable isotopes from south Florida, a preliminary study. *Organic Geochemistry* 34, pp. 185–194.

Beer S., Bjork M., Hellblom F., Axelsson L. 2002. Inorganic carbon utilization in marine angiosperm (seagrasses). *Functional Plant Biology* 29, pp. 349-354.

Bell JD., Steffe AS., and Westoby M. 1985. Artificial seagrass: how useful is it for field experiments on fish and macroinvertebrates? *Journal of Experimental Marine Biology and Ecology* 90, pp. 171–177.

Best RJ., Caulk NC., Stachowicz JJ. 2013. Trait vs. phylogenetic diversity as predictors of competition and community composition in herbivorous marine amphipods. *Ecology Letter* 16, pp.72–80.

Bologna PAX. 2006. Assessing within habitat variability in plant demography, faunal density, and secondary production in an eelgrass (Zostera marina L.) bed. *Journal of Experimental Marine Biology and Ecology* 329, pp. 122–134.

Bologna PAX. 2007. Impact of differential predation potential on eelgrass (*Zostera marina*) faunal community structure. *Aquatic Ecology* 41, pp. 221-229.

Bologna PAX., Heck KL. Jr. 1999. Macrofaunal associations with seagrass epiphytes Relative importance of trophic and structural characteristics. *Journal of Experimental Marine Biology and Ecology* 242, pp. 21-39.

Bostrom C., Mattila M. 1999. The relative importance of food and shelter for seagrass associated invertebrates latitudinal comparison of habitat choice by isopod grazers. *Oecologia* 120, pp. 162-170.

Boyce MC., Lavery P., Bennett IJ., Horwitz P. 2001. Spatial variation in the $\delta\ ^{13}C$ signature of *Ruppia megacarpa* (Mason) in coastal lagoons of southwestern Australia and its implication for isotopic studies. *Aquatic Botany* 71, pp. 83–92.

Caine, EA., 1980. Ecology of two littoral species of caprellid amphipods (Crustacea) from Washington, U.S.A. *Marine Biology* 56, pp. 327-335.

Chapman MG., Roberts DE. 2004. Use of seagrass wrack in restoring disturbed Australian saltmarshes. *Ecological Management and Restoration* 5, pp. 183-190.

Colombini I., Aloia A., Fallaci M., Pezzoli G., Chelazzi L. 2000. Temporal and spatial use of stranded wrack by the macrofauna of a tropical sandy beach. *Marine Biology* 136, pp. 531–541.

Deegan LA., Finn J., Hopkinson CS., Giblin AE., Peterson BJ., Fry B., Hobble J. 1995. Flow model analysis of the effects of organic matter-nutrient interactions on estuarine trophic dynamics. *In*: Dyer KR, 01th RJ (eds) Changes influxes in estuaries. Olsen and Olsen, Fredensborg, pp. 273-281.

Duffy JE., Reynolds PL., Boström C., Coyer JA., Cusson M., Donadi S., Douglass JG., Eklöf JS., Engelen AH., Eriksson BK., Fredriksen S., Gamfeldt L., Gustafsson C., Hoarau G., Hori M., Hovel K., Iken K., Lefcheck JS., Moksnes PO., Nakaoka M., O'Connor MI., Olsen JL., Richardson JP., Ruesink JL., Sotka EE., Thormar J., Whalen MA., Stachowicz JJ. 2015. Biodiversity mediates top-down control in eelgrass ecosystems: A global comparative- experimental approach. *Ecology Letter* 8, pp. 1-10.

Durrand J. 1984. Nitrogen distribution in New Jersey coastal bays. *In*: Kennish M.J., Lutz R.A. (eds) Lecture Notes on Coastal and Estuarine Studies, *Ecology of Barnegat Bay*, New Jersey. *Springer*, New York, pp. 29-51.

Fourqurean JW., Moore TO., Fry B. & Hollibaugh, JT., 1997. Spatial and temporal variation in C:N:P ratios, $\delta^{15}N$, and $\delta^{13}C$ of eelgrass *Zostera marina* as indicators of ecosystem processes, Tomales Bay, California, USA. *Marine Ecology Progress Series* 157, pp. 147–157.

Gessner RV. 1977. Seasonal occurrence and distribution of fungi associated with *Spartina alterniflora* from a Rhode Island estuary. *Mycologia* 69, pp. 477-491.

Gessner RV. 1978. *Spartina alterniflora* seed fungi. *Canadian Journal of Botany* 56, pp. 2942–2947.

Greenway M. 1995. Trophic relationships of macrofauna within a Jamaican seagrass meadow and the role of the echinoid *Lytechinus varietatus* (Lamarck). *Bulletin of Marine Science* 56, pp. 719–736.

Heck KL. Jr., Able KW., Roman CT., Fahay MP., 1995. Composition, abundance, biomass and production of macro fauna in a New England estuary: Comparisons among eelgrass meadows and other nursery habitats. *Estuaries* 18, pp. 379 - 389.

Heck KL. Jr., Carruthers TJ., Duarte CM., Hughes AR., Kendrick GA., Orth, RJ., Williams SL. 2008. Trophic transfers from seagrass meadows subsidize diverse marine and terrestrial consumers. *Ecosystems* 11, pp. 1198-1210.

Hemminga MA, Slim FJ, Kazungu J, Ganssen GM, Nieuwenhuize J, Kruyt NM. 1994. Carbon outwelling from a mangrove forest with adjacent seagrass beds and coral reefs (Gazi Bay, Kenya). *Marine Ecology Progress Series* 106, pp. 291–301.

Hyndes GA., Lavery PS. 2005. Does transported seagrass provide an important trophic link in unvegetated, nearshore areas? *Estuarine and Coastal Shelf Science* 63, pp. 633–643.

Inglis G. 1989. The colonisation and degradation of stranded *Macrocystis pyrifera* (L.) C. Ag. by the macrofauna of a New Zealand sandy beach. *Journal of Experimental Marine Biology and Ecology* 125, pp. 203–217.

Jackson J., Kirby M., Berger W., Bjorndal K., Botsford L., Bourque B., Bradbury R., Cooke R., Erlandson J., Estes J., Hughes T., Kidwell S., Lange C., Lenihan H., Pandolfi J., Peterson C., Steneck R., Tegner M., Warner R. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293, pp. 629–63.

Keller PD., Kelly S., Sullivan B. 1990 Growth of juvenile Atlantic menhaden, *Brevoortia tyranus* in MERL mesocosms: effects of eutrophication. *Limnology and Oceanography* 35, pp. 109-122.

Kirkman H., Kendrick AG. 1997. Ecological significance and commercial harvesting of drifting and beach-cast macro-algae and seagrasses in Australia: a review. *Journal of Applied Phycology* 9, pp. 311–326.

Kitting, C., 1984. Selectivity by dense populations of small invertebrates foraging among seagrass blade surfaces. *Estuaries* 7, pp. 276–288.

Kitting CL., Fry B., Morgan MD. 1984. Detection of inconspicuous epiphytic algae supporting food webs in seagrass meadows. *Oecologia* 62, pp.145–149.

Kristensen JH. 1972. Carbohydrates of some marine invertebrates with notes on their food and on the natural occurrence of the carbohydrates studied. *Marine Biology* 14, pp. 130–142.

Lepoint G., Fontaine M., Dauby P., Gobert S., Bouquegneau JM. 2003. Carbon and nitrogen isotopic ratios of the seagrass *Posidonia oceanica*: depth related variations. *Botanica Marina* 46, pp. 555–561.

MacArthur RH. 1969. Species packing, or what competition minimizes. *Proceedings of the National Academy of Sciences* 64, pp. 1369-1375.

Mallin M., Paerl H. 1994. Planktonic trophic transfer in an estuary: seasonal diet and community effects. *Ecology* 75, pp. 2168-2184.

May MS. 1974. Probable agents for the formation of detritus from the halophyte, *Spartina alterniflora*. *In*: R. J. Reimold and W. H. Queen (eds.), Ecology of Halophytes. *Academic Press, N.Y.,* pp. 429–440.

Middelburg JJ. 2014. Stable isotopes dissect aquatic food webs from the top to the bottom. *Biogeosciences* 11, pp. 2357-2371.

Moncreiff CA., Sullivan M., Daehnick AE. 1992. Primary production dynamics in seagrass beds of Mississippi Sound: the contributions of seagrass, epiphytic algae, sand micro-flora, and phytoplankton. *Marine Ecology Progress Series* 87, pp. 161–171.

Morgan MD., Kitting CL. 1984. Productivity and utilization of the seagrass *Halodule wrightii* and its attached epiphytes. *Limnology and Oceanography* 29, pp. 1066–1076.

Morin PJ. 2011. Predation and communities: empirical patterns. *In*: Morin PJ (ed) Community Ecology, 2nd edn. *Wiley Blackwell, Oxford,* pp. 90–119.

Nixon SW. 1988 Physical energy inputs and the comparative ecology of lake and marine ecosystems. *Limnology and Oceanography* 33, pp. 1005-102.

Nielsen J., Lethbridge R. 1989. Feeding and epiphyte food resource of gastropods living on the leaves of the seagrass *Amphibolis griffithii* in southwestern Australia. *Journal of the Malacology Society of Australia* 10, pp. 47–58.

Orth RJ., and van Montfrans J. 1984. Epiphyte–seagrass relationships with an emphasis on the role of micrograzing: a review. *Aquatic Botany* 18, pp. 43–69.

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

Pollard PC., Kogure K. 1993. The role of epiphytic and epibenthic algal productivity in a tropical seagrass, *Syringodium isoetifolium* (Aschers.) Dandy, community. *Australia Journal of Marine and Freshwater Research* 44, pp. 141–154.

Robertson AI., Lucas JS. 1983. Food choice, feeding rates, and the turnover of macrophyte biomass by a surf-zone inhabiting amphipod. *Journal of Experimental Marine Biology and Ecology* 72, pp. 99-124.

Robertson AI., Lenanton RCJ. 1984. Fish community structure and food chain dynamics in the surf-zone of sandy beaches: the role of detached macrophytes detritus. *Journal of Experimental Marine Biology and Ecology* 84, pp. 265-283.

Robinson D. 2001. δ15N as an integrator of the nitrogen cycle. *Trends in Ecology and Evolution* 16, pp. 153–162.

Rose CD., Dawes CJ. 1999. Effects of community structure on the seagrass *Thalassia testudinum. Marine Ecology Progress Series* 184, pp. 83–95.

Rublee PA., Cammen LM., Hobbie JE. 1978. Bacteria in a North Carolina salt marsh: standing crop and importance in the decomposition of *Spartina alterniflora. University of North Carolina Sea Grant Publications 78-11.*, p. 88.

SAS ® Version 9.3 Copyright (c) 2002-2010 by SAS Institute Inc., Cary, NC, USA.

Schoener TW. 1982. The controversy over interspecific competition. *American Scientist* 70, pp. 586–595.

Squires ER., Good RE. 1974. Seasonal changes in the productivity, caloric content, and chemical composition of a population of salt-marsh cord-grass *(Spartina alterniflora). Chesapeake Science* 15, pp. 63-71.

Sieburth JM., Brooks RD., Gessner RV., Thomas C. D., Tootle J. L. 1974. Microbial colonization of marine plant surfaces as observed by scanning electron microscopy. *In*: R. R. Colwell and R. Y. Morita (eds.), Effect of the Ocean Environment on Microbial Activities. *University Park Press, Baltimore.*, pp. 418–432.

Shurin JB., Borer ET., Seabloom EW., Anderson K., Blanchette CA., Broitman BR., Cooper SD., Halpern BS. 2002. Across ecosystem comparison of the strength of trophic cascades. *Ecology Letter* 5 pp. 785–791.

Smit AJ., Brearley A., Hyndes GA., Lavery PS., Walker DI. 2005. Carbon and nitrogen stable isotope analysis of an *Amphibolis griffithii* seagrass bed. *Estuarine and Coastal Shelf Science* 65, pp. 545–556.

Sogard SM. 1989. Colonization of artificial seagrass by fishes and decapod crustaceans: importance of proximity to natural eelgrass. *Journal of Experimental Marine Biology and Ecology* 133, pp. 15–37.

Sparks K., Sparks JP. 2016. Data Interpretation Guide for Isotopic Analysis. *Cornell University*, pp.1

Teal JM. 1962. Energy flow in the salt marsh ecosystem of Georgia. *Ecology* 43, pp. 614-624.

Thayer GW., Kenworth WJ., Fonseca MS. 1984.The ecology of eelgrass meadows of the Atlantic coast: A community profile. *US Fish and Wildlife Service 84-248*, pp. 147.

van Montfrans J., Orth RJ., Vay SA. 1982. Preliminary studies of grazing by *Bittium varium* on eelgrass periphyton. *Aquatic Botany* 14, pp. 75-89.

van Montfrans J., Wetzel RL., and Orth RJ. 1984. Epiphyte-grazer relationships in seagrass meadows: Consequences for seagrass growth and production. *Estuaries* 7, pp. 298–309.

Virnstein RW., Nelson WG., Lewis FG., Howard RK. 1984 Latitudinal patterns in seagrass epifauna: do patterns exist, and can they be explained? *Estuaries* 7, pp. 310–330.

Virnstein RW., Curran MC. 1986. Colonization of artificial seagrass versus time and distance from source. *Marine Ecology Progress Series* 29, pp. 279–288.

Vizzini S., Sarà G., Mateo MA., Mazzola A. 2003. $\delta\ ^{13}C$ and $\delta\ ^{15}N$ variability in *Posidonia oceanica* associated with seasonality and plant fraction. *Aquatic Botany* 75, pp. 195–202.

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

Wootton L., Bologna P. 2004. Community Structure and Detrital Dynamics in *Zostera marina* and *Ruppia maritima* Beds in Barnegat Bay; Implications from Essential Fish Habitat. Project Report, Barnegat Bay Estuary Program.

Appendix 1. Average abudance of organisms (±SD) within Core Samples

	6/17/15	6/30/15	7/14/15	7/29/15	8/12/15	8/26/15	9/13/15
Aoridae	0.8±1.8	2.8±2.8	19.2±23.3	4.0±2.4	15.2±6.4	9.8±5.6	101.0±102.2
Ampelisca spp.	0.4±5.9	0.0±0.0	0.6±1.3	0.6±1.3	0.2±0.4	0.0±0.0	0.0±0.0
Phoxocephalidae	6.2±5.9	4.0±4.1	3.2±2.2	1.8±2.0	2.8±2.6	1.8±1.1	9.3±9.4
Melitidae	11.4±10.9	6.5±5.6	3.0±3.8	4.8±5.8	4.6±5.9	4.0±3.1	30.3±27.3
Ischyroceridae	0.0±0.0	0.0±0.0	0.2±0.4	0.2±0.4	0.2±0.4	0.0±0.0	0.3±0.4
Tanais spp.	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.2±0.4	0.0±0.0
Lillibjordae	0.2±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Corophiidae	0.0±0.0	21.8±22.0	3.6±3.6	4.0±6.2	1.4±2.2	2.4±1.7	1.0±0.9
Gammarus spp.	0.2±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.2±0.4	0.0±0.0	1.0±1.3
Caprellidae	0.2±0.4	3.0±4.3	0.0±0.0	0.0±0.0	0.0±0.0	0.6±1.3	9.3±9.4
Erichsonella	28.0±47.9	18.3±12.4	2.4±4.3	0.8±0.8	2.8±2.6	0.0±0.0	0.3±0.4
Idotea balthica	17.4±12.3	17.8±15.5	9.0±19.6	0.4±0.9	0.4±0.9	0.0±0.0	0.3±0.4
Cyathura polita	0.0±0.0	4.8±6.9	0.0±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Edotea	0.6±0.9	0.0±0.0	0.0±0.9	0.0±0.0	0.0±0.0	0.0±0.0	0.7±0.9
Pycnogonid	1.2±1.8	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Ostrocods	6.8±14.1	3.3±4.0	0.8±1.8	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Palomaenetes	0.0±0.0	0.5±0.5	0.2±0.4	1.0±1.4	0.2±0.4	0.0±0.0	0.0±0.0
Crangon septemspinosa	0.0±0.0	0.0±0.0	0.0±0.0	0.2±0.4	0.0±0.0	0.0±0.0	0.0±0.0
Polychaete	19.6±12.0	12.5±8.8	1.6±3.7	1.4±1.5	0.6±0.5	2.0±1.4	8.3±7.1
Mytilus edulis	2.2±3.3	0.8±0.5	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0

Gemma gemma	90.8±75.2	19.3±20.0	7.2±5.2	8.2±12.2	0.0±0.0	0.0±0.0	4.3±5.3
Argopecten irradians	0.0±0.0	0.5±0.9	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Hippolyte spp.	0.0±0.0	0.0±0.0	0.0±0.0	0.4±0.9	0.2±0.4	0.2±0.4	0.0±0.0
Panopeus herbstii	0.6±0.9	0.0±0.0	0.2±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Rhithropanopeus harrisii	0.0±0.0	0.5±0.9	0.6±1.3	0.0±0.0	0.4±0.5	0.0±0.0	0.3±0.4
Pinnotheres	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.4±0.9	0.0±0.0	0.0±0.0
Solemya velum	2.8±1.9	1.5±2.7	7.4±13.8	3.2±1.1	1.6±1.8	1.0±1.0	0.0±0.0
Syngnathus fuscus	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Apeltes quadracus	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Botryllus schlosseri	0.0±0.0	0.8±1.3	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Panopeus herbstii	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Dysanopeus sayi	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Euryanopeus depressus	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Illyanasa	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	1.0±1.3
Geukensia demissa	0.6±1.3	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Ensis spp.	0.2±0.4	0.0±0.0	0.0±0.0	0.2 ±0.4	0.0±0.0	0.4±0.5	0.0±0.0

Appendix 2a. Average abundance of organisms (±SD) within *Zostera marina* Treatment Bags.

	6/30/15	7/14/15	7/29/15	8/12/15	8/26/15	9/13/15
Aoridae	7.4±11.3	6.3±5.3	51.2±58.0	312.5±258.3	23.0±16.9	1543.8±307.2
Ampelisca spp.	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Phoxocephalidae	13.0±5.3	2.3±4.0	1.0±1.7	0.0±0.0	1.0±0.9	0.3±0.4
Melitidae	787.4±198.6	114.0±58.9	238.6±92.4	318.5±206.6	582.5±339.3	2.3±4.0
Ischyroceridae	0.0±0.0	0.0±0.0	1.4±3.1	2.5±1.7	9.5±5.5	0.0±0.0
Tanais spp.	0.0±0.0	0.0±0.0	0.0±0.0	0.5±0.4	0.0±0.0	0.0±0.0
Hausteridae	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Amphithoidae	0.0±0.0	26.5±47.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Corophiidae	101.0±106.7	20.3±16.4	81.8±37.8	68.0±37.2	44.5±25.3	68.0±63.0
Gammarus	72.6±98.7	19.5±18.8	0.8±1.3	0.0±0.0	0.0±0.0	9.5±17.0
Caprellidae	326.4±234.2	30.8±19.6	99.4±29.2	70.0±39.1	371.5±315.2	446.3±520.6
Erichsonella spp.	5.2±7.9	6.5±9.5	2.6±3.2	2.5±2.2	7.5±4.1	1.0±1.3
Idotea balthica	28.6±14.8	63.5±54.1	0.6±0.9	0.0±0.0	1.5±1.3	0.3±0.4
Cyathura polita	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Edotea spp.	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Pycnogonid	0.2±0.4	0.5±0.9	5.2±4.0	10.5±9.4	0.0±0.0	7.0±8.3
Palaemonetes	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0

Polychaete	1.8±0.8	0.5±0.9	1.2±1.3	0.0±0.0	0.0±0.0	0.3±0.4
Mytilus edulis	0.8±0.8	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Gemma gemma	2.0±2.7	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Mercenaria mercenaria	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.5±0.4	0.0±0.0
Tunicates	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0	0.5±0.4	0.0±0.0
Panopeus herbstii	0.0±0.0	0.3±0.4	0.4±0.5	1.5±1.3	0.0±0.0	0.0±0.0
Rhithropanopeus harrisii	5.0±2.6	0.5±0.5	0.0±0.0	0.0±0.0	0.0±0.0	0.3±0.4
Macoma	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Crepidula fornicata	1.0±1.0	0.8±0.9	0.2±0.4	0.0±0.0	0.0±0.0	0.0±0.0
Tunicate Larvae	0.0±0.0	2.0±2.3	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Megalopa	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0

Appendix 2b. Average abundance of organisms within(±SD) Artificial *Zostera marina* Treatment Bags

	6/30/15	7/14/15	7/29/15	8/12/15	8/26/15	9/13/15
Aoridae	17.8±11.7	8.5±7.0	14.8±8.7	38.3±22.0	73.0±50.5	365.8±222.8
Ampelisca spp.	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.8±1.3
Phoxocephalidae	0.0±0.0	0.0±0.0	0.3±0.4	0.0±0.0	0.3±0.4	0.3±0.4
Melitidae	173.2±119.0	31.3±29.5	155.3±92.3	57.7±39.3	289.8±151.6	7.5±12.3
Ischyroceridae	0.0±0.0	0.0±0.0	0.0±0.0	9.0±7.5	65.0±115.2	2.0±3.6
Tanais spp.	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Hausteridae	0.0±0.0	5.8±10.3	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Amphithoidae	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Corophiidae	54.0±38.8	16.0±14.4	51.3±28.2	16.7±9.5	21.0±16.4	84.5±42.7
Gammarus spp.	9.2±16.3	10.3±6.3	7.3±10.4	5.3±4.4	0.0±0.0	66.3±118.5
Caprellidae	50.8±45.3	0.5±0.9	124.8±82.1	16.0±12.1	480.3±739.2	846.8±835.7
Erichsonella spp.	11.6±16.9	14.0±17.0	2.5±2.8	0.7±0.5	4.8±3.6	0.8±1.3
Idotea balthica	58.6±31.2	46.3±27.5	0.5±0.5	0.7±0.9	0.3±0.4	0.8±0.5
Cyathura polita	0.2±0.4	0.0±0.0	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0
Edotea	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Pycnogonid	0.0±0.0	0.0±0.0	4.5±2.6	0.3±0.4	0.0±0.0	1.8±3.1
Palaemonetes	0.0±0.0	0.3±0.4	0.0±0.0	1.0±0.9	0.0±0.0	0.0±0.0
Polychaete	1.0±0.7	0.0±0.0	0.0±0.0	0.7±0.9	0.0±0.0	0.0±0.0

Mytilus edulis	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Gemma gemma	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.3±0.4	0.0±0.0
Mercenaria mercenaria	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Tunicates	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Panopeus herbstii	0.0±0.0	0.0±0.0	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0
Rhithropanopeus harrisii	0.8±1.1	0.0±0.0	0.3±0.4	0.0±0.0	0.5±0.9	0.0±0.0
Macoma	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Crepidula fornicata	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Tunicate Larvae	0.0±0.0	13.5±20.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Megalopa	0.0±0.0	0.0±0.0	0.5±0.5	0.0±0.0	0.0±0.0	0.0±0.0

Appendix 2c. Average abundance of organisms within (±SD) Control Treatment Bags.

	6/30/15	7/14/15	7/29/15	8/12/15	8/26/15	9/13/15
Aoridae	8.8±12.9	3.4±1.9	2.8±2.0	12.0±6.6	29.5±23.7	191.0±186.0
Ampelisca	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Phoxocephalidae	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	22.5±20.1	63.0±112.7
Melitidae	19.0±21.5	8.8±9.7	67.0±53.3	21.5±11.8	23.5±21.0	9.5±15.4
Ischyroceridae	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	1.5±1.3	0.8±1.3
Tanaid	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Hausteridae	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Amphithoidae	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Corophiidae	14.0±14.2	8.6±7.3	52.5±39.5	11.0±6.0	30.0±16.5	94.0±90.3
Gammarus spp.	9.6±15.8	1.2±2.2	1.5±2.7	0.0±0.0	0.0±0.0	0.0±0.0
Caprellidae	10.4±14.3	1.6±1.5	73.3±79.1	2.5±1.7	43.5±34.6	582.8±439.0
Erichsonella spp.	4.4±6.4	2.8±4.8	0.5±0.5	0.5±0.4	1.0±0.9	0.0±0.0
Idotea balthica	21.2±30.6	14.2±11.3	0.8±0.9	0.0±0.0	0.0±0.0	1.0±0.8
Cyathura polita	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Edotea	0.0±0.0	0.0±0.0	0.0±0.0	0.5±0.4	0.0±0.0	0.0±0.0
Pycnogonid	0.0±0.0	0.0±0.0	2.0±2.6	0.0±0.0	0.0±0.0	8.5±15.2
Palaemonetes	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Polychaete	0.2±0.4	0.2±0.4	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Mytilus edulis	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Gemma gemma	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Mercenaria mercenaria	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Tunicates	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.3±0.4

Panopeus herbstii	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Rhithropanopeus harrisii	0.0±0.0	0.0±0.0	0.0±0.0	0.5±0.4	0.0±0.0	0.0±0.0
Macoma	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Crepidula fornicata	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Tunicate Larvae	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0	0.0±0.0
Megalopa	0.0±0.0	0.0±0.0	0.3±0.4	0.0±0.0	0.0±0.0	0.0±0.0