HEALTH ASSESSMENT FOR EELGRASS IN NANTUCKET HARBOR, NANTUCKET MASSACHUSETTS

FINAL REPORT TO NANTUCKET LAND COUNCIL

submitted by

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Introduction

Eelgrass (*Zostera marina* L.) is a marine flowering plant that forms extensive meadows in the shallow coastal waters of Massachusetts. The value of eelgrass meadows is well documented and includes stabilizing sediments, improving water quality and clarity, mitigating for CO_2 emissions, and providing habitat to a number of commercially important and/or endangered species (Orth et al. 1984; Heck et al. 1989; Hughes et al. 2002; Lazarri and Tupper 2002). Nantucket Island, located 30 miles off the coast of Cape Cod, supports over 2,000 acres of eelgrass (Costello 2015) that serve as essential habitat to a number of different species including the last commercially viable "wild" bay scallop fishery in the U.S. The abundance of eelgrass, however, has diminished from historic levels in some areas, potentially threatening the future ecology and economy of this system.

Declines in eelgrass in Nantucket over the last decade have been mostly confined to Nantucket Harbor (Costello and Kenworthy 2011; Costello 2015). The loss in size and density of eelgrass in the harbor is likely due to an increase in nutrient loading (Curley 2002). In the Executive Summary of the Massachusetts Estuaries Project report, it was noted that to maintain or preserve eelgrass meadow health, a nitrogen threshold of 0.350 mg N L^{-1} should not be exceeded. Nitrogen levels in East Polpis Harbor in 2006 were 0.361 mg N L^{-1} and eelgrass had recently disappeared from most of the area (Shellfish Report 2012), indicating that this is an accurate threshold. At present, it is believed that Nantucket Harbor has reached its nitrogen loading threshold (Howes et al. 2006) and is eutrophied (over-enriched with nutrients; Conant et al., 2006).

Nitrogen loading to Nantucket Harbor results primarily from on-site disposal of wastewater. The Town has a centralized wastewater treatment facility, but there are a number of areas on septic that contribute nitrogen to the system both through transport in direct groundwater discharges to estuarine waters and through small surface water flows to the fresh and saltwater marshes that are located along the harbor shore (e.g. Mill Brook discharging to Polpis Harbor). In addition to residential septic systems, other nutrient sources include runoff from roads and lawn fertilizers, groundwater discharge, and natural areas such as salt marshes and ponds (Howes et al. 2006; Shellfish Report 2012). In 2013 a Town of Nantucket Board of Health regulation went into effect to control the content and application of fertilizer containing phosphorus and nitrogen into Nantucket's waters and wetlands through an organized educational program, licensure and regulations of practice (Nantucketlandcouncil.org). The effectiveness of these efforts on nitrogen loading in the system has yet to be determined.

Eutrophication can have negative impacts on seagrasses. As eutrophication progresses, macroalgae (in shallow waters) or phytoplankton (in deeper waters) dramatically increase and become dominant resulting in declines of seagrass. Direct underlying mechanisms for declines include competition for light/nitrogen, nitrate inhibition or ammonium toxicity, with light playing a more important role in advanced eutrophication stages (Orth and Moore, 1983; Twilley et al., 1985; Dennison et al., 1993; Harlin, 1993; Lapointe et al., 1994; Short et al., 1995; Hauxwell and Valiela, 2004; Ralph et al., 2006). Many indicators of seagrass plant health and environmental quality have been identified in previous monitoring studies and workshops to help assess the impacts of eutrophication on seagrass. Seagrass cover, above-ground biomass, leaf length and width have been shown to be affected by nutrient loading and shading (Erftemeijer 1994; Lee and Dunton 2000; Burkholder et al. 2007; van Katwijk 2010) along with epiphyte content on blades (Bohrer et al. 1995; Uku and Bjork 2001). In addition, stable isotope analysis is being increasingly used to monitor the health status and nutrient pollution sources of various ecosystems. For example, Cole et al. (2006) showed that water derived from sewage on Cape Cod typically has ^{15}N values of +10 to +20 ‰, while water influenced by atmospheric deposition has values of +2 to +8, and water loaded with fertilizer features values between -3 to +3. Thus, using these parameters, stable isotope analysis can be used to detect the presence of sewage-derived or agricultural nitrogen (N) in the tissues of eelgrass that continually uptake nutrients from their environment.

The purpose of our study was to assess the health of eelgrass meadows at six sites (i.e., Monomoy, Pimny's Point, Fulling Mill, Quaise, Pocomo, Wauwinet) in Nantucket Harbor influenced by nutrient input. Our objective was accomplished by collecting information on various plant and environmental parameters at each site between May and August 2019. In addition, environmental data was used to identify potential mechanisms responsible for reported declines of eelgrass in this system.

Methods

Eelgrass plant health was assessed by collecting information on meadow structure and nutrient content in leaf tissue and sediment.

Eelgrass morphology and meadow structure

In July 2018, when plants had reached peak biomass, information on meadow structure was collected at 6 sites in Nantucket Harbor (i.e., Monomoy, Pimny's Point, Fulling Mill, Quaise, Pocomo, Wauwinet) as well as at a reference location on Tuckernuck Island (Figure 1). At each site, one 50 m cross transect was laid parallel to the shore. Five 0.25 m^2 quadrats were then haphazardly tossed along the transect and information was collected on percent cover, canopy height, and shoot density. In addition, two representative shoots with roots and rhizomes were collected from each quadrat for morphological measurements (number of leaves, leaf width, above/below-ground weight, and internode length).

Nutrient Content in Leaf Tissue

The influence of nitrogen on eelgrass was assessed by measuring nitrogen (%N), carbon (%C) and stable isotopes of δ13C and δ15N in leaves, as well as calculating C:N ratios and a Nutrient Pollution Indicator (NPI) for eelgrass at each site (Lee et al. 2004). In May 2018, during a period of increased precipitation, ten representative eelgrass shoots were sampled at each of the 6 sites in Nantucket Harbor, with at least 1 m between any two sampled shoots. In July 2018, during a period of decreased precipitation, sampling was repeated at each of the 6 sites in Nantucket Harbor as well as the reference location on Nantucket (Figure 1.) After each sampling event, shoots were returned to the lab for measurements.

In the lab, leaf mass was determined on the second and/or third youngest leaves of each shoot. All epiphytes were removed from leaves. Six 10 cm long sections of constant width were then cut from each leaf to obtain samples of mature leaf tissue. The cleaned leaf sections were dried at 60 °C to a constant weight and leaf mass was quantified. Each leaf segment was then assessed for %C and %N content and stable isotopes of δ13C and δ15N using an Eurovector CN analyzer (see stable isotope section below).

The ratio of the leaf nitrogen content (%N) to area normalized leaf mass mg dry weight cm^{-2} was used to calculate a Nutrient Pollution Indicator value as developed by Lee et al. (2004)

$$NPI = \frac{\text{Leaf nitrogen content (\%N)}}{\text{Area normalized leaf mass (mg dry wt cm}^{-2}\text{)}}$$

Sediment Samples

One 5 cm sediment sample was taken from each site including the reference using a syringe for sediment grain-size analyses. In addition, sediment cores were taken at 3 sites (i.e., Monomoy, Fulling Mill, and Wauwinet) to assess sediment and nutrient characteristics. The corer (length: 50 cm, diameter: 70 mm) was manually driven to a depth of 25 cm or point of refusal, extracted, capped at both ends under water, and kept in a vertical position during transport to shore. Cores were divided into sections (1 cm sections for the first 10 cm and 5cm sections for the remaining core) and used to measure dry bulk density, %C, %N, analyze stable isotopes of δ13C and δ15N, and determine age of sediments (see methods below).

Grain Size

Grain size was determined for sediment samples taken at each site including the reference using the Malvern Mastersizer 2000 with the Hydro 2000S wet dispersion unit (Malvern Instruments, Malvern, UK) system. Sediment samples were homogenized and extruded through a 2 mm sieve into a beaker, then deionized water was added to the sample to create a suspension that was then analyzed.

Dry Bulk Density

Bulk density reflects the size, shape and arrangement of particles and voids (soil structure) and gives a good indication of the suitability for root growth and sediment permeability. Bulk density generally increases with compaction and tends to increase with depth. Sandy substrate is also more prone to high bulk density. Dry bulk density was determined for sediment core section take at Monomoy., Fulling Mill, and Wauwinet using the mass of sediments dried at 60°C for 7 days divided by the volume of the sediment section. Following bulk density measurements, the sample was sub-divided using a sediment splitter to obtain a smaller portion for stable isotope and ^{210}Pb analyses.

Sediment Accumulation and Core Age

Sedimentation rates for at three sites (i.e., Monomoy, Fulling Mill, Wauwinet) were obtained by analyzing core samples for ^{210}Pb radioisotopes using gamma spectroscopy. Samples were packed in Petri dishes and sealed with electrical tape and paraffin wax 30 days prior to analysis to allow for equilibration between ^{226}Ra and its daughter isotopes (^{214}Pb and ^{214}Bi). Radioisotopic concentrations were determined for all samples along each core using a Canberra GL 2020 low energy germanium detector (Virginia Institute of Marine Science, Gloucester Point, VA). The concentrations of excess ^{210}Pb used to obtain the age models were determined as the difference between total ^{210}Pb and ^{226}Ra (supported ^{210}Pb). The

Constant Rate of Supply (CRS) model was used to calculate mean sedimentation rates over the last 100 years at all sites (Appleby and Oldfield, 1978). These rates were calculated using the following formula:

$$A = A(0)e^{-\lambda t}$$

where A is the excess (unsupported) ^{210}Pb inventory below a given core section, A(0) is the excess ^{210}Pb inventory for the entire core profile, and λ is the ^{210}Pb decay constant. This was used to calculate t, the time a now-buried section of core was at the surface.

The formula from Kaste et al. (2011) was used to calculate error for the CRS model:

$$1\sigma=\sqrt{n}/n$$

where n = the number of detected counts.

Carbon, Nitrogen, and Stable Isotope Analyses

Carbon (%C), nitrogen (%N) and stable isotope analyses on plant and sediment samples was carried out in a Eurovector CN analyzer. During each sequence run by the mass spectrometer, each sample was flash combusted at 1800ºC and the combustion products (CO_2, N_2 and H_2O) were separated chromatographically and introduced into the mass spectrometer, with water removed in a chemical trap. The gases of interest were then introduced into the mass spectrometer for isotope analysis and the rest pumped away. The sample isotope ratio was compared to a secondary gas standard, whose isotope ratio has been calibrated to international standards. For 13CV-PDB the gas will be calibrated against NBS 20 (Solenhofen Limestone), NBS 21 (Spectrographic Graphite), and NBS 22 (Hydrocarbon Oil); for ^{15}Nair the gas was calibrated against atmospheric N_2 and IAEA standards N-1, N-2, and N-3 (all are ammonium sulfate standards). Elemental content of leaf tissue was calculated on a dry weight basis and elemental ratios on a molar basis (QA/QC BU Stable Isotope Lab 2013).

Environmental Conditions

Environmental conditions at each of the six sites were also assessed. Two Hobo light/temperature sensors (http://www.onsetcomp.com/ sensors) were deployed in an array at each site for 2-week intervals from mid-May to the end of August 2018 (peak growing season). Each array included a light sensor at the bottom and a second sensor 0.3 m higher to determine light available to eelgrass at the site and light attenuation due to the water column. One sensor was deployed on land, attached in an unobstructed location to a fence at Monomoy. The sensors measured and recorded temperature and light every 15 minutes. For comparison between sites, a subset of the light data was collected in a 4-hour period around solar noon (10:00 to 14:00) for two weeks each month for analyses. All the temperature data between May and August.

Statistics

To assess differences among sites in plant and environmental parameters, one way ANOVAs were performed. All data were tested for homogeneity of variances using Cochrans' test. Tukey's post-hoc tests were used to determine groupings in the analysis of stable isotope data. Differences among nitrogen data were not assessed for May and July due to unequal sample sizes.

Results

Eelgrass morphology and meadow structure

Morphological and structural characteristics of the meadow were significantly different among sites in July 2018 (Figures 2 & 3; one-factor ANOVA: leaves/shoot $F_{6,63}$= 4.4663, p<0.0008; leaf width $F_{6, 63}$ = 2.1807, p=0.0565; internode length $F_{6,63}$= 3.6630, p=0.0035; above-below ground weight $F_{6,63}$ = 3.6630, p=0.0035; canopy height $F_{6,63}$ = 30.6118, p<0.0001; percent cover $F_{6,20}$ = 5.6475, p=0.0014; shoot density $F_{6,20}$ = 2.4752, p=0.0591.) There were no consistent patterns in shoot morphology and/or meadow structure observed among sites in Nantucket Harbor. However, Tuckernuck, the reference site, had larger shoots with longer leaves than plants from Nantucket Harbor.

Influence of Nitrogen on Leaf Tissue

May

Analysis of eelgrass leaf N content for May samples revealed significant differences in mean leaf %N. (ANOVA: %N $F_{5,6}$= 4.1072, p=0.050) with %N in tissue ranging from 1.2% to 1.9% (Figure 4). There were no obvious patterns indicating a gradient towards higher or lower areas of nutrient enrichment or loading between the six sites. No differences among sites were observed for δ15N values, C:N or NPI (ANOVA: δ15N values $F_{5, 6}$ = 4.3232, p=0.0516; C:N $F_{5, 6}$ = 1.7658, p=0.2539; NPI $F_{5,6}$= 1.1683, p=0.4286). Mean δ15N for the system was 4.03 ± 0.39 while C:N for the system was 22.4 ± 0.54 and mean NPI for the system was 0.421 ± 0.028 (Figure 4).

July

Analysis of eelgrass leaf N content for July samples revealed significant differences in mean leaf N, C:N, NPI, and δ15N among sites (Figure 5; ANOVA: %N $F_{6,63}$= 4.9518, p=0.0003; C:N $F_{6, 63}$ = 3.6588, p=0.0035; NPI $F_{6,63}$= 3.8674, p=0.0024; δ15N $F_{6,66}$= 54.57, p<0.0001). Leaf N ranged from 0.849 % to 1.2%, C:N ranged from 32.3 to 45.8 and NPI ranged from 0.161 to 0.244. July δ15N values ranged from 3.08 to 5.08‰. Tuckernuck had an average value of 7.46 ‰, which was significantly higher than the other sites (Figure 5). There were no obvious patterns indicating a gradient towards higher or lower areas of nutrient enrichment and/or type of loading between the six sites and/or the reference site. C:N ratio differences among sites were driven by differences in N content as there was no difference in carbon content among sites.

Sediment Samples

Grain-Size

All sediment samples from Nantucket Harbor and Tuckernuck, consisted of 98 to 99 % sand-sized grains. The highest percentage of coarse sand was found at Fulling Mill (44%) while the highest portion of fine to very fine sand was found at Monomoy (20%) and Pimny's Point (22%). The predominant sediment type was medium to fine-grained sand with lesser amounts of medium and fine sand (Figure 6). Eelgrass grows well in sediment that consists of <70 percent silt to clay so the sediment grain-size distribution was not unexpected.

Bulk Density

The bulk density measurements are comparable to other eelgrass meadows in the region (Plaisted, pers. comm.). The density ranged from 1.25 to 2 g/cm^3 and slightly increased with depth at each site. Dry bulk density at Monomoy (1.6 ± 0.03 g cm^{-3}) was similar to Fulling Mill (1.7 ± 0.01 g cm^{-3}) and Wauwinet (1.5 ± 0.01 g cm^{-3}; Figure 7). The large fluctuation in bulk density values at the top 5 cm at each site can be attributed to the high mobility of substrate in this dynamic system.

Sedimentation Rates and Core Age

The three cores that were dated ranged in age from 64 to 98 years. The highest depth integrated sedimentation rate and youngest core were found in Wauwinet (7.1 ± 1.15 mm/yr; 64 years in the upper 25 cm). Monomoy had a depth integrated sedimentation rate of 6.4 ± 1.6 mm/yr and core age of 98 years in the upper 25 cm. In contrast, Fulling Mill had the lowest depth integrated sedimentation rate (3.95 ± 0.5 mm/yr) and a core age of 82 years in the upper 25 cm (Figure 8). The large variability in sedimentation rates in the upper 5-10 cm of each core can be attributed to high mobility of substrate within the system (e.g., shoaling).

Stable isotopes

Sediment core material $\delta^{15}N$ ranged from -3.08 ‰ to 6.81 ‰ while $\delta^{13}C$ ranged from -28.63 ‰ to -7.10 ‰ (Figures 9 & 10). Each core showed variability with depth for $\delta^{15}N$ and $\delta^{13}C$. Cole et al. (2006) showed that water derived from sewage typically has $\delta^{15}N$ values of +10 to +20 ‰, while water influenced by atmospheric deposition has values of +2 to +8, and water loaded with fertilizer features values between -3 to +3. Phytoplankton and particulate organic matter have $\delta^{13}C$ ranging from -15 ‰ to -28 ‰ while eelgrass have $\delta^{13}C$ ranging from -5 ‰ to -10 ‰ (Fry and Wainwright, 1991; Fry, 2006; Novak et al., in review).

Environmental Conditions

Light

The average daily light available to eelgrass relative to ambient land conditions between May and August ranged from 0.03% measured at the bottom of the canopy located at Pimny's Point up to 45.9% measured at the top of the canopy Fulling Mill. Large quantities of algae were consistently found in the eelgrass meadow at Pimny's Point. Algae was also observed at the other sites, but mats were not as dense. In June and July, the bottom light sensor at Fulling Mill and the top and bottom light sensors at Pocomo were lost/damaged (Figures 11 & 12).

Temperature

The average monthly temperature measured between May and August ranged from 17.7˚C measured at Monomoy in May and up to ~31˚C measured at Wauwinet in July. During the months of July all sites were above 25˚C more than 50% of the time. In August, Quaise, Pocomo and Wauwinet were above 25˚C more than 50% of the time (Figure 13).

Discussion

Recent studies show that the abundance of eelgrass has diminished from historic levels in Nantucket Harbor (Costello and Kenworthy 2011; Costello 2015). The loss in size and density of eelgrass in the Harbor is believed to be due to an increase in nutrient loading to the system (Curley 2002). In our study, we assessed the health of eelgrass meadows at six sites in Nantucket Harbor influenced by high nitrogen loadings by collecting information on various plant and environmental parameters and identified mechanisms of declines. Our results show that eelgrass meadows in Nantucket Harbor are light-limited and thermally stressed during the peak growing season, suggesting that long-term loss of eelgrass in this system is due to the joint effect of cultural eutrophication (high nutrient loadings) and warming waters.

Ochieng et al. (2010) demonstrated that eelgrass plants in New England require 58% surface irradiance (SI) and above to grow and expand and are light-limited at 34% SI and below (Ochieng et al. 2010). Kenworthy et al. (2014) further suggested that the threshold for survival is 13.9% based on measurements at the deep edge (2.56 m) of eelgrass meadows in Nantucket Harbor. In our study, all sites received less than 34% SI from May to August except for Fulling Mill, which received more than 45% during the month of August (Figures 11&12). These results indicate that eelgrass meadows in Nantucket Harbor are not receiving enough light throughout the peak growing season to maintain a positive carbon balance and allow growth and expansion of meadows. The causes of light-limitation can be attributed to large quantities of drift macro-algae collecting in eelgrass meadows and reducing the light available to eelgrass through shading. Large quantities of algae were especially prevalent at Pimny's Point from May thru August. In addition to algae, sediment resuspension due to the loss of eelgrass in the system, as well as moored boats may be further reducing light levels in eelgrass meadows and causing declines.

Temperatures above 25°C have previously been identified as another stressful threshold for eelgrass (Greve et al. 2003; Reusch et al. 2005). At 25 °C, water temperatures cause rates of respiration to exceed photosynthesis, resulting in a negative carbon balance (Marsh et al. 1986; Moore et al. 1997). At 28 °C, large scale declines in eelgrass cover have been observed at the southern range of this species distribution (Shields et al. 2019). During the month of July, all study sites in Nantucket Harbor had an average water temperature above 25 °C and were exposed to temperatures above 28 °C for ~13% of the time. In August, only Quaise, Pocomo, and Wauwinet had an average water temperature above 25°C. However, all sites spent more than 50% of the time above the 25°C and more than 3% of the time above 28 °C (Figure 13). The warm water temperatures in Nantucket Harbor during the summer months is higher than temperatures in nearby eelgrass meadows located in shallow subtidal waters on Cape Cod. Between 2003 and 2015, Pleasant Bay (Orleans) and Duck Harbor (Wellfleet) were exposed to temperatures above 25°C less than 16% of the time and above 28 °C less than 2% (NPS, SeagrassNet data). Based on the results of this study, it appears as though eelgrass meadows are also thermally stressed in this system possible due to climate change.

C:N, NPI, and % N in leaves have be used as indicators of nutrient enrichment in seagrass meadows. C:N ratios less than 20 in leaves, NPI values greater than 0.3, and/or leaf nitrogen values above 1.6 % have been found in nutrient enriched systems such as Great Bay (NH), Waquoit Bay (MA), and Narragansett Bay (RI; Heminga and Duarte 2000; Lee et al. 2004). In our study, Pimny's Point had %N values in leaves greater than 1.6% during May (Figure 4). However, all sites had higher C:N, and lower NPI, and %N values during the month of July than nutrient enriched systems (Figure 5). The lower values of %N in eelgrass during the July 2018 sampling could be a result of seasonal nitrogen availability and eelgrass growth rates (Duarte 1990, Fourqurean et al. 1997). Nitrogen is typically limited in the nearshore during summer growing seasons and growth rates are often elevated increasing plant biomass while reducing total nutrient concentrations. Moreover, the extensive algae mats found throughout the system may be reducing the amount of nitrogen available to eelgrass during the summer months; Alexandre et al. (2017) found that some species of algae have higher uptake capacities for nutrients than eelgrass.

Stable isotopes analysis of plant material offers the possibility of detecting the biological role of groundwater flow in the marine environment or the impact of sewage effluent before major ecological changes occur (Mac Clelland et al. 1997; Mac Clelland and Valiela 1998). It is particularly useful in areas where a small nutrient increase could have a significant impact on the ecosystem especially where this nutrient increase is undetectable in the water due to, for example, a low sewage load or rapid dilution in the surrounding environment (Gartner et al. 2002; Yamamuro et al. 2003). In Waquoit Bay (Massachusetts, USA), isotopic studies have permitted the attribution of an isotopic signature to nitrates from waste water, from fertilizer and from atmospheric deposition (MacClelland et al. 1997). In our study, July $\delta^{15}N$ values ranged from 3.08 to 5.08‰ in Nantucket Harbor and there was no obvious pattern indicating a gradient towards higher or lower areas of nutrient enrichment and/or type of loading among the six sites. Tuckernuck, a relatively preserved site, had a high $\delta^{15}N$ of 7.46‰ (Figure 5). The lack of differentiation between $\delta^{15}N$ sources in July in Nantuckets Harbor suggest multiple inputs (i.e., fertilizer and atmospheric deposition). Likewise, the high values in eelgrass tissues from Tuckernuck are not necessarily the reflection of sewage or ground water impacts. For example, Fourqurean et al. (1997) measured the increase of $\delta^{15}N$ values of eelgrass from the mouth to the head of Tomales Bay in California. In this relatively preserved bay, groundwater discharge is considered low. The high $\delta^{15}N$ values (+12‰) are attributed to the occurrence of denitrification processes in Tomales Bay marine waters, which may have resulted in the ^{15}N enrichment of the remaining inorganic N pool and, consequently, a ^{15}N enrichment of plants which incorporate inorganic N from the water column.

Recommendations

Our study provides baseline information on eelgrass health for Nantucket Harbor and identifies mechanisms for reported declines. As the climate continues to warm, eelgrass in Nantucket Harbor will continue to be exposed to increased water temperatures and periods of thermal stress. However, eelgrass can survive if other environmental parameters that promote growth and expansion are optimal. Below are some recommendations for future work, as well as management actions that will improve eelgrass health and facilitate recovery in the harbor

Restoration of eelgrass meadows in Nantucket Harbor is one suggested strategy to facilitate recovery in this system even in the face of climate change. Restoration involves improving environmental conditions (e.g. water quality) to encourage natural regeneration and/or seeding/transplanting plants from donor meadows. We recommend managers improve water quality within the harbor by reducing land-based pollution and decreasing nutrient and sediment run-off, reducing or eliminating the use of fertilizers and persistent pesticides and increasing filtration of effluent. The reduction in nutrients within the system will lead to a reduction in nuisance algae which limit the amount of light available to eelgrass for growth. Moreover, if plants are no longer light stressed they will be able to tolerate longer periods of thermal stress. In addition, to improving water quality, we recommend continuing transplanting efforts in well flushed areas with low quantities of algae. In September 2018, the Nantucket Land Council along with Boston University began transplanting 1/4 acre plots of eelgrass at Monomoy. This location was selected because it is well-flushed and has historically supported eelgrass (Shellfish Report, 2012). The establishment of eelgrass within this area should help “kick start” natural recovery within the Monomoy section of the harbor. In addition, newly transplanted eelgrass in this area is expected to further improve water quality and clarity through nutrient uptake and suspended sediment deposition (Duarte, 1995).

In addition to restoring eelgrass in Nantucket Harbor, we also recommend monitoring existing eelgrass meadows in the harbor using a hierarchical framework to detect and predict changes so that appropriate management strategies can be developed. The monitoring approach would include three tiers that are integrated across spatial scales and sampling intensities (see Neckles et al. 2012). Tier 1 monitoring would involve mapping eelgrass in Nantucket Harbor every three-five years to provide large-scale information on seagrass distribution and meadow size. Costello (2015) has already developed an appropriate mapping process for this system that involves the acquisition of high resolution digital imagery captured within strict environmental conditions. Tier 2 monitoring would involve bay-wide, quadrat-based assessments of eelgrass percent cover and canopy height at permanent sampling stations following a spatially distributed random design. The National Park Service on Cape Cod has a design that is used to monitor Pleasant Bay that could be adapted to this system. Tier 3 monitoring would involve high-resolution measurements of seagrass condition (percent cover, canopy height, total and reproductive shoot density, biomass, and seagrass depth limit) at a representative index site in the system. SeagrassNet is an example of a program that collects more detailed Tier 3 data and could be easily implemented in the harbor (http://www.seagrassnet.org/). If a hierarchical approach to monitoring is not feasible for Nantucket Harbor at this time, light and temperature data, as well as percent cover of algae and eelgrass should be monitored at multiple sites within the system.

Costello (2015) showed a slight decline in eelgrass in Madaket while eelgrass was stable to increasing on Tuckernuck. The decline in Madaket was along the deeper edges South of Eel Point and possibly due to storm and tidal current action. A complete loss of habitat in the inner Madaket Harbor area on both sides of the channel and in the upper reaches of the inner harbor was also noted. Water quality issues were suspected in those losses as the contributing watershed seems to have experienced increased development. To increase our understanding of the factors responsible for eelgrass declines and the conditions required to facilitate eelgrass growth and expansion on Nantucket we suggest conducting eelgrass plant health assessments in Madaket Harbor and Tuckernuck utilizing the methodology outlined in this study. Specifically, information on meadow structure and nutrient content in leaf tissue and sediment as well as light and temperature data should be collected at multiple sites in Madaket Harbor and Tuckernuck and compared to the data collected in Nantucket Harbor. We suggest collecting in both spring and summer months.

Lastly, we recommend raising awareness about the socio-economic and ecological values of eelgrass as it is critical in building support for seagrass conservation. Engaging local communities and stakeholders is essential in any conservation strategy. Volunteer monitoring programs can be effective in increasing public awareness of the value of eelgrass meadows and the threats to their survival. Community monitoring programs, such as SeagrassNet, successfully promote stewardship, reinforce the value of eelgrass habitats and collect data about the condition of this species. Public education programs should identify actions that individuals can take to reduce stresses on eelgrass in this system. For example, individuals can help reduce threats to water quality by preventing pollutants (e.g. fertilizers, paint, gasoline, solvents and garden chemicals) from entering storm-water drains. To reduce sediment and nutrient run-off into waterways, individuals can maintain vegetation on riverbanks and adjacent to the harbor, create retention ponds or ditches to reduce high-discharge flows or plant a buffer strip of plants in these areas. Boaters can also avoid anchoring and running their propellers through eelgrass meadows. Mooring in eelgrass meadows appears to be a serious problem across the island.

References

Alexandre A, Baeta A, Engelen AH, Santos R. Interactions between seagrasses and seaweeds during surge nitrogen acquisition determine interspecific competition. Scientific Reports. 2017;7(1). doi:10.1038/s41598-017-13962-4

Appleby, P.G., Oldfield, F., 1978. The calculation of lead-210 dates assuming a constant rate of supply of unsupported 210Pb to the sediment. Catena 5, 1–8. https://doi.org/10.1016/S0341-8162(78)80002-2

Burkholder, J. M., Tomasko, D.A., and Touchette, B. W. 2007. Seagrasses and eutrophication. Journal of Experimental Marine Biology and Ecology. 350: 46-72.

Bohrer, T., Wright, A., Hauxwell, J., and Valiela, I. 1995. Effect of epiphyte biomass on growth rate of Zostera marina in estuaries subject to different nutrient loading. Biol. Bull. 189: 260.

Cole, M. L., K. D. Kroeger, J. W. MCClelland, Valiela, I. 2006. Effects of watershed land use on nitrogen concentrations and d15N in groundwater. Biogeochem 77: 199– 215.

Conant, K. 2006. Nantucket Harbor Water Quality Annual Report 2005. Prepared for the Marine and Coastal Resources Department.

Costello C. T. and Kenworthy, W.J. 2011. Twelve-Year Mapping and Change Analysis of Eelgrass (Zostera marina) Areal Abundance in Massachusetts (USA) Identifies Statewide Declines. Estuar. Coasts 34(2): 232-242.

Costello, C.T. 2015. Nantucket eelgrass mapping project. Prepared for Nantucket Natural Resources Department, Nantucket, MA.

Curley, T. 2002. Nantucket Harbor Water Quality, Annual Report.

Dennison, W.C., Orth, R.J., Moore, K.A., Stevenson, J.C., Carter, V., Kollar, S., Bergstrom, P., Batuik, R.A., 1993. Assessing water quality with submerged aquatic vegetation. Biosci 43: 86–94.

Duarte. C. M. 1990. Seagrass nutrient content. Mar Ecol. Prog. Ser. 67: 201-207

Duarte, C.M. 1995. Submerged aquatic vegetation in relation to different nutrient regimes. Ophélia 41: 87–112.

Erftemeijer, P.L.A. 1994. Differences in nutrient concentrations and resources between seagarass communities on carbonate and terrigenous sediments in South Sulawesi, Indonesia. Bull. Mar. Sci. 54: 403–419.

Fourqurean, J.W., Moore, T.O., Fry, B., Hollibaugh, J.T. 1997. Spatial and temporal variation in C:N:P ratios, δ 15N and δ 13C of eelgrass Zostera marina as indicators of ecosystem processes, Tomales Bay, California, USA. Mar. Ecol. Prog. Ser. 157: 147–157

Gartner, A., Lavery, P., Smit, A.J. 2002. Use of δ 15N signatures of different functional forms of macroalgae and filter-feeders to reveal temporal and spatial patterns in sewage dispersal. Mar Ecol Prog Ser 235: 63-73

Greve, T.M., Borum, J., Pedersen, O., 2003. Meristematic oxygen variability in eelgrass (Zostera marina). Limnol. Oceanogr. 48: 210–216.

Hauxwell, J., Valiela, I., 2004. Effects of nutrient loading on shallow seagrass-dominated coastal systems: patterns and processes. In: Nielsen, S., Banta, G., Pedersen, M. (Eds.), Estuarine Nutrient Cycling: the Influence of Primary Producers. Kluwer Academic Publishers, the Netherlands, pp. 59–92.

Harlin, M.M. 1993. Changes in major plant groups following nutrient enrichment. In: McComb, J. (Ed.), Eutrophic Shallow Estuaries and Lagoons. CRC Press, Inc., Boca Raton (FL), pp. 173–187.

Heck Jr., K.L., K.W. Able, M.P. Fahay, and Roman, C. 1989. Fishes and decapod crustaceans of Cape Cod eelgrass meadows: species composition, seasonal abundance patterns and comparison with unvegetated substrates. Estuar. 12: 59–65.

Hemminga MA, Duarte CM. 2000. Seagrass ecology. Cambridge University Press, Cambridge (U.K.)

Howes B., S. W. Kelley, J. S. Ramsey, R. Samimy, D. Schlezinger, and Eichner, E. 2006. Linked Watershed Embayment Model to Determine Critical Nitrogen Loading Thresholds for Nantucket

Harbor, Town of Nantucket, Nantucket Island, MA. Massachusetts Estuaries Project, Massachusetts Department of Environmental Protection. Boston, MA.
Hughes, J.E., L.A. Deegan, J.C. Wyda, M.J. Weaver, and Wright, A. 2002. The effects of eelgrass habitat loss on estuarine fish communities of southern New England. Estuaries 25: 235–249.
Kaste, J.M., Bostick, B.C., Heimsath, A.M., Steinnes, E., Friedland, A.J., 2011. Using atmospheric fallout to date organic horizon layers and quantify metal dynamics during decomposition. Geochim. Cosmochim. Acta 75, 1642–1661. https://doi.org/10.1016/j.gca.2011.01.011
Lapointe, B.E., Tomasko, D.A., Matzie, W.R.. 1994. Eutrophication and trophic state classification of seagrass communities in the Florida Keys. Bull. Mar. Sci. 54: 696–717.
Lazarri, M.A., and Tupper, B. 2002. Importance of shallow water habitats of demersal fishes and decapod crustaceans in Penobscot Bay, Maine. Environ. Biol Fish. 63: 57–66.
Lee, K.S., and Dunton, K.H. 2000. Effects of nitrogen enrichment on biomass allocation, growth, and leaf morphology of the seagrass Thalassia testudinum. Mar. Ecol. Prog. Ser. 196, 39–48.
Lee, K.-S., Short, F.T., Burdick, D.M., 2004. Development of a nutrient pollution indicator using the seagrass, Zostera marina, along nutrient gradients in three New England estuaries. Aquat. Bot. 78:197–216.
Marsh, J. A., Dennison, W. C., and Alberte, R. S. 1986. Effects of temperature on photosynthesis and respiration in eelgrass (*Zostera marina* L.). *J. Exp. Mar. Biol. Ecol.* 101, 257–267. doi: 10.1016/0022-0981(86)90267-4
McClelland, J. W., Valiela, I., Michener, R. H. 1997. Nitrogen-stable isotope signatures in estuarine food webs: a record of increasing urbanization in coastal watersheds. *Limnol. Oceanogr.* 42: 930–937. doi: 10.4319/lo.1997.42.5.0930
McClelland, J. W., Valiela, I. 1998. Linking nitrogen in estuarine producers to land-derived sources. Limnol. Oceanogr. 43, 577–585. doi: 10.4319/lo.1998.43.4.0577
Moore KA, Wetzel RL, Orth RJ. 1997. Seasonal pulses of turbidity and their relations to eelgrass (Zostera marina L.) survival in an estuary. J Exp Mar Biol Ecol 215: 115−134
Neckles, H., Kopp, B., Peterson, B., Pooler P. 2012. Integrating Scales of Seagrass Monitoring to Meet Conservation Needs. Estuar. Coasts. 35: 23-46. 10.1007/s12237-011-9410-x.
Ralph, P.J., Tomasko, D., Moore, K., Seddon, S., Macinnis-Ng, C.M.O. 2006. Human impacts on seagrasses: eutrophication, sedimentation, and contamination. In: Larkum, A.W.D., Orth, R.J., Duarte, C.M. (Eds.), Seagrasses: Biology, Ecology and Conservation. Springer, the Netherlands, pp. 567–593.
Reusch TBH, Ehlers A, Hämmerli A, Worm B. 2005. Ecosystem recovery after climatic extremes enhanced by genotypic diversity. Proc Natl Acad Sci USA 102:2826–2831.
Shields, E. Parrish, D., Moore, K. 2019. Short-Term Temperature Stress Results in Seagrass Community Shift in a Temperate Estuary. Estuaries and Coasts. 10.1007/s12237-019-00517-1.
Short, F.T., Burdick, D.M., Kaldy, J.E., 1995. Mesocosm experiments quantify the effects of eutrophication on eelgrass, Zostera marina L. Limnol. Oceanogr. 40, 740–749.
Shellfish Report 2012. Prepared for the Marine and Coastal Resources Department Town of Nantucket, Massachusetts.
Ochieng, C. A., Short, F. T., Walker, D. I. 2010. Photosynthetic and morphological responses of eelgrass (Zostera marina L.) to a gradient of light conditions. J Exper. Mar. Biol. Ecol. 382:117-124.
Orth, R.J., K.L. Heck Jr., and van Montfrans, J. 1984. Faunal communities in seagrass beds: a review of the influence of plant structure and prey characteristics on predator-prey relationships. Estuar. 7: 339–350.
Orth, R.J., Moore, K. 1983. Submersed vascular plants: Techniques for analyzing their distribution and abundance. Mar. Tech. Soc. J. 17. 38-52.
Twilley, R. R., Kemp, W. M., Staver, K. W., Stevenson, J. C., Boynton, W. R. 1985. Nutrient enrichment of estuarine submersed vascular plant communities. Algal growth and effects on production of plants and associated communities. Mar. Ecol. Prog. Ser. 23: 179-191

Uku, J. and Bjork, M. 2001. The distribution of epiphytic algae on three Kenyan seagrass species. S. Afr. J. Bot. 67: 475–482.

van Katwijk, M.M., Bos, A.R., Kennis, P. and de Vries, R. 2010. Vulnerability to eutrophication of a semi-annual life history: a lessons learnt from an extinct eelgrass (Zostera marina) population. Biol. Conserv. 143: 248–254.

Yamamuro, M., Kayanne, H., Yamano, H. 2003. δ15N of seagrass leaves for monitoring anthropogenic nutrient increases in coral reef ecosystems. Marine pollution bulletin. 46: 452-8. 10.1016/S0025-326X(02)00463-0.

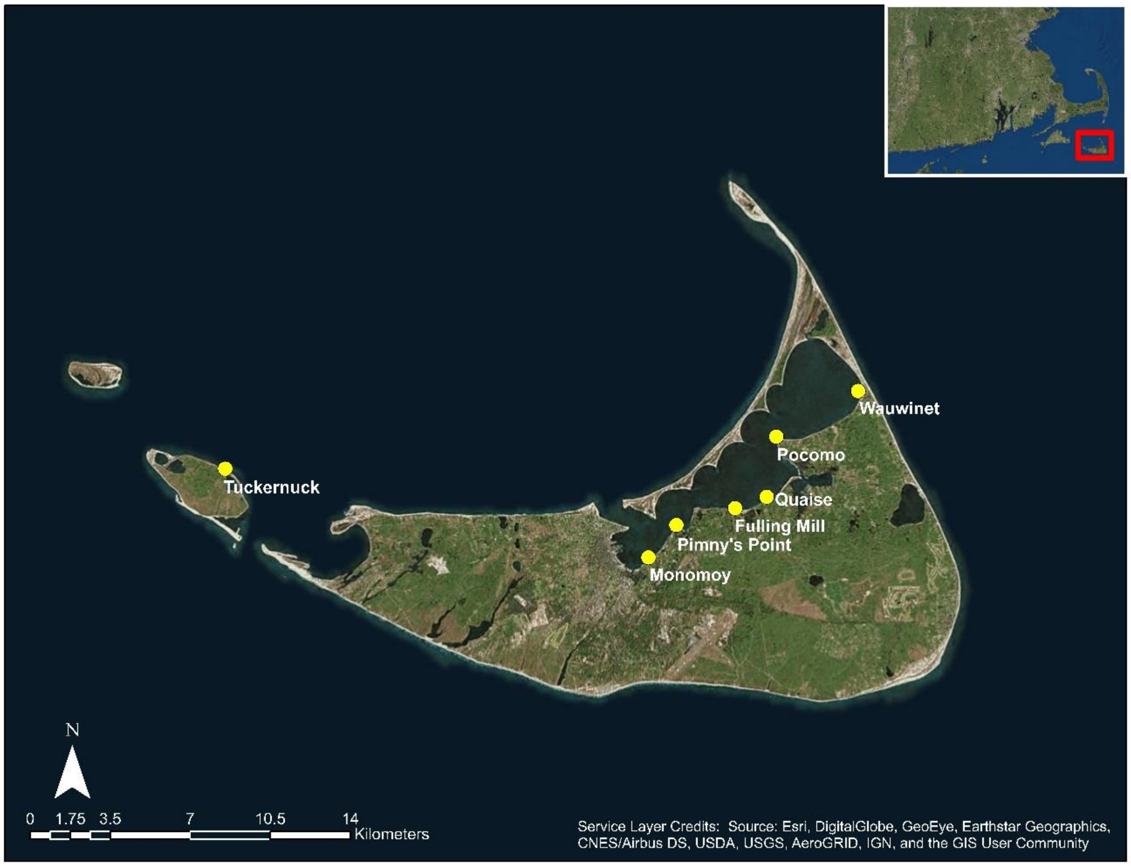

Figure 1. Map showing the location of sampling sites in Nantucket Harbor as well as a reference site near Tuckernuck Island.

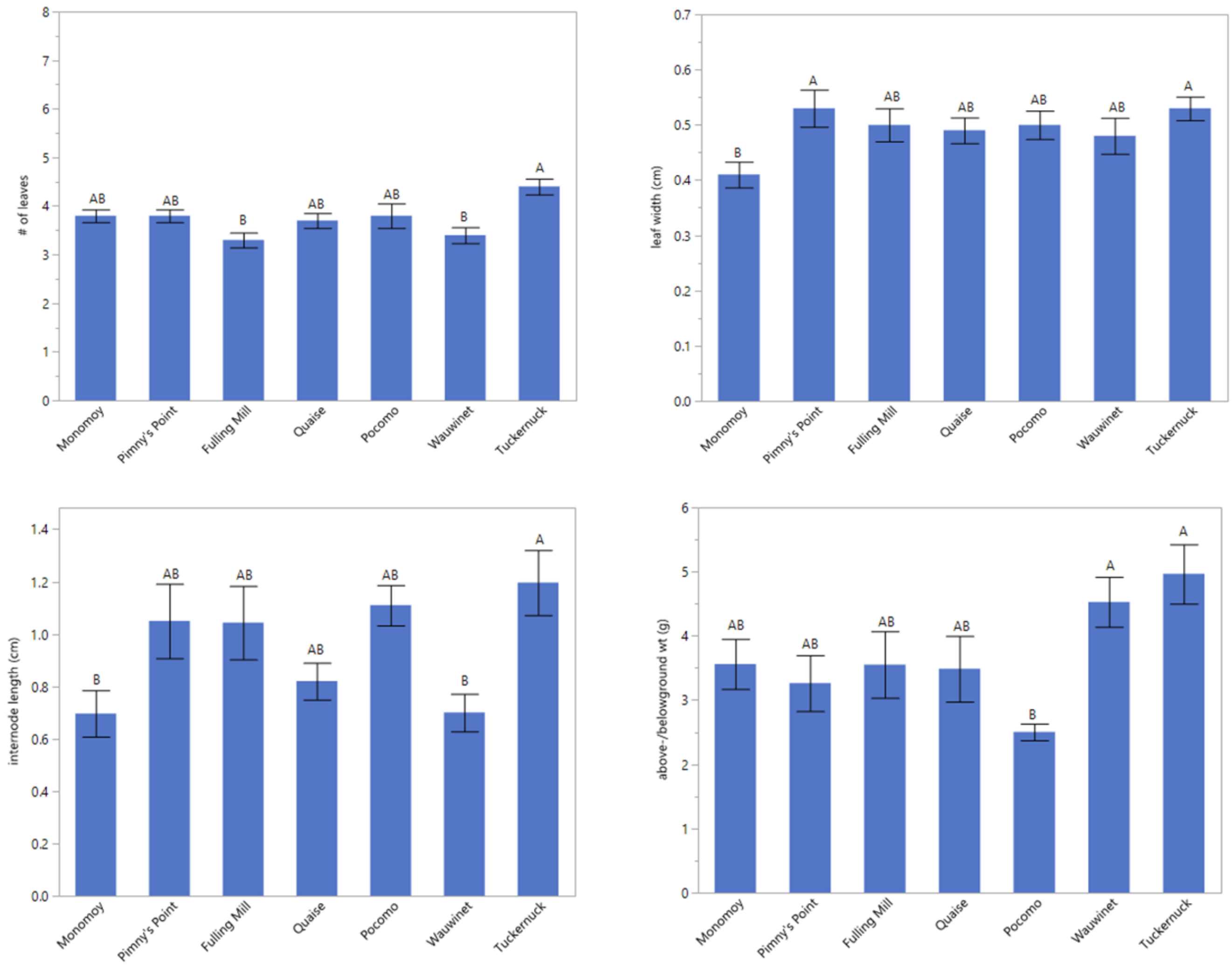

Figure 2. Significant differences in morphological characteristics were observed among sites in July 2018 (means ±SE). Different letters A-B denote Tukey's test results for significant differences among sites at P<0.05.

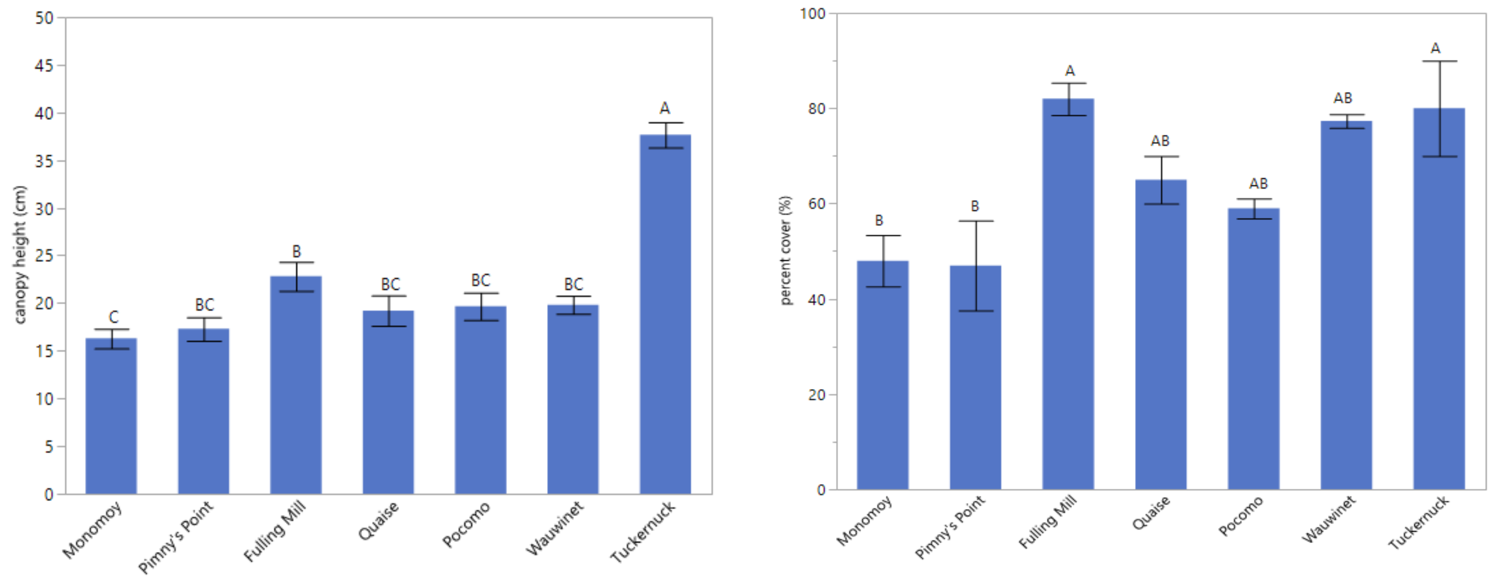

Figure 3. Significant differences in structural characteristics of the meadow were observed among sites for canopy height and percent cover in July 2018 (means ±SE). Different letters A-C denote Tukey's test results for significant differences among sites at $P<0.05$.

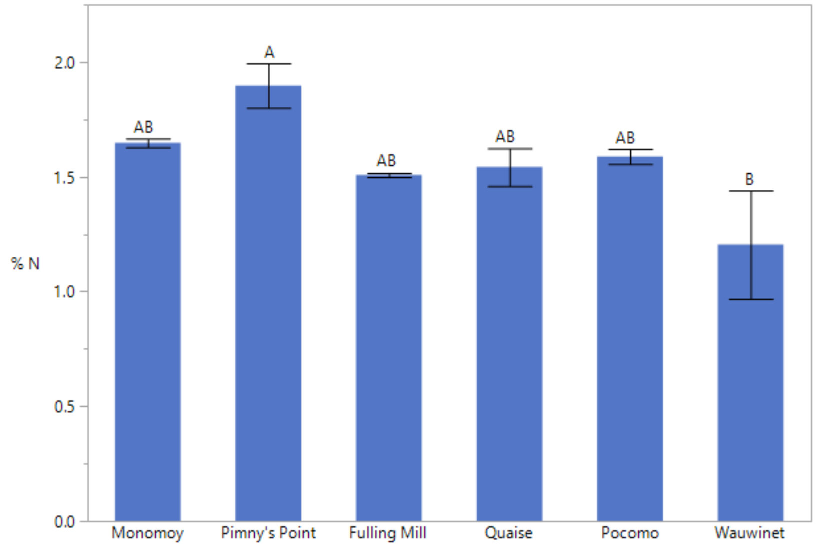

Figure 4. Significant differences in %N were observed among sites during the Spring of 2018 (means ±SE). Different letters A-B denote Tukey's test results for significant differences among sites at P<0.05.

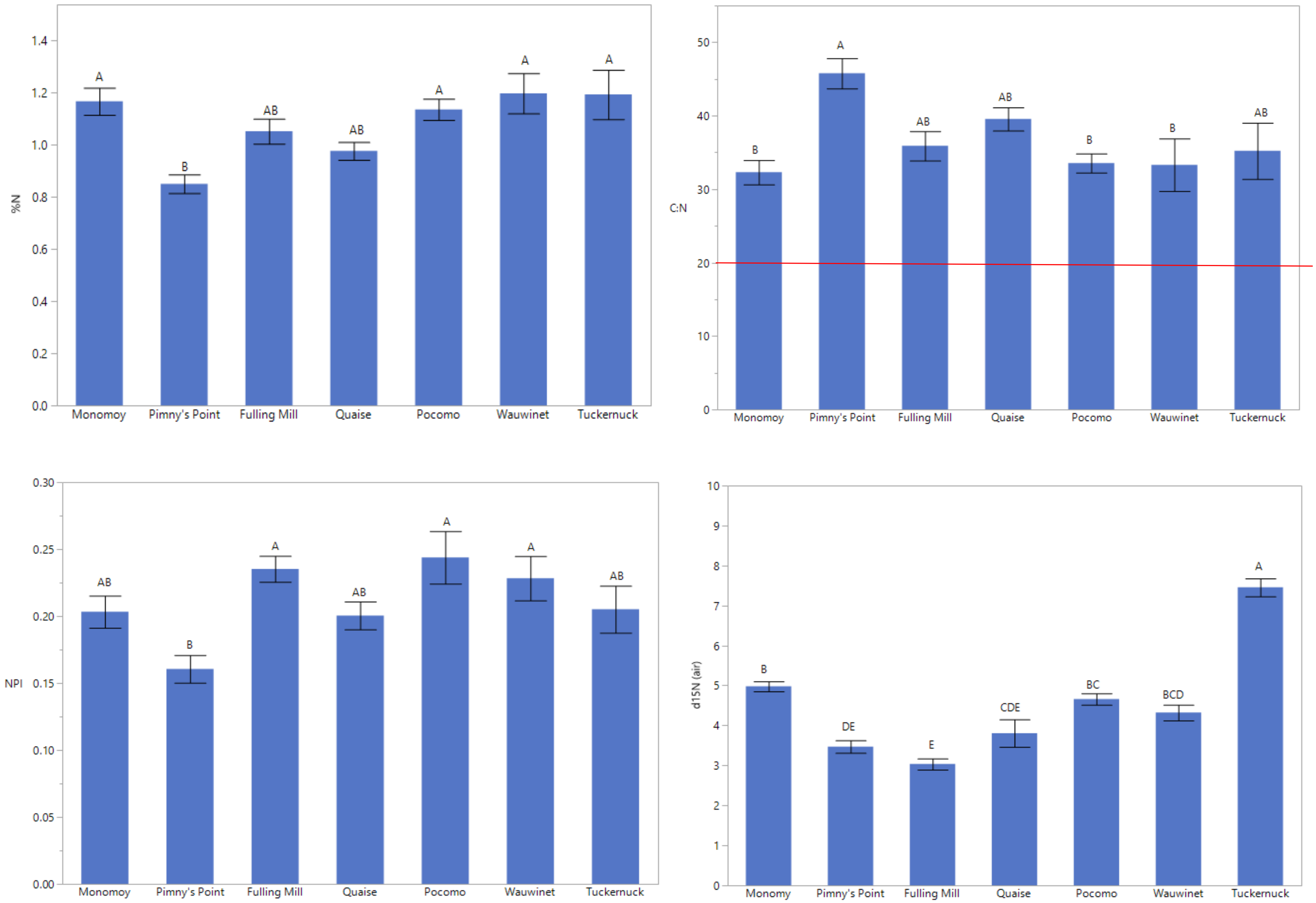

Figure 5. Significant differences in %N, C:N, NPI, and δ15N were observed among sites during the Summer of 2018 (means ±SE). Different letters A-E denote Tukey's test results for significant differences among sites at P<0.05. C:N values < 20 indicate nitrogen enrichment (red line).

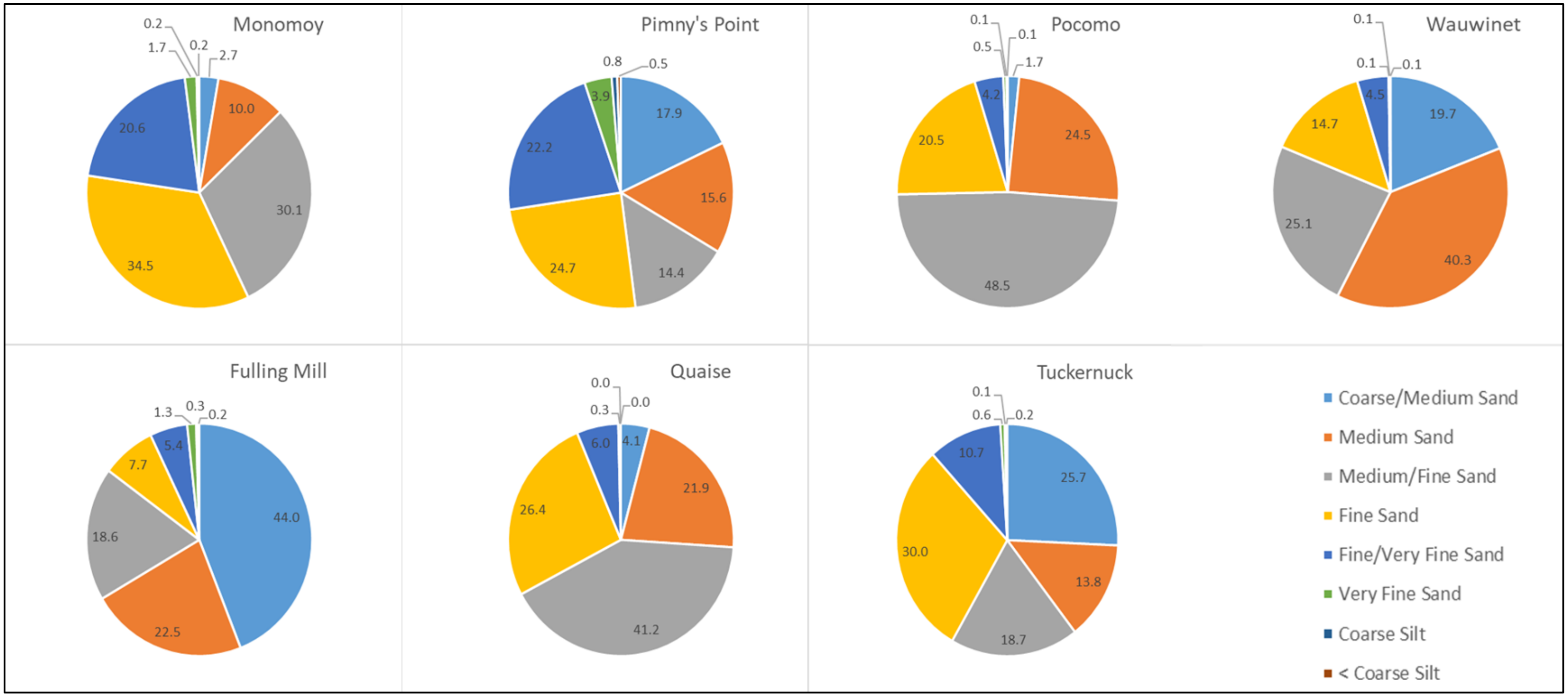

Figure 6. Charts showing sediment grain-size distribution at each of the six sites in Nantucket Harbor as well as at the sample site on Tuckernuck Island.

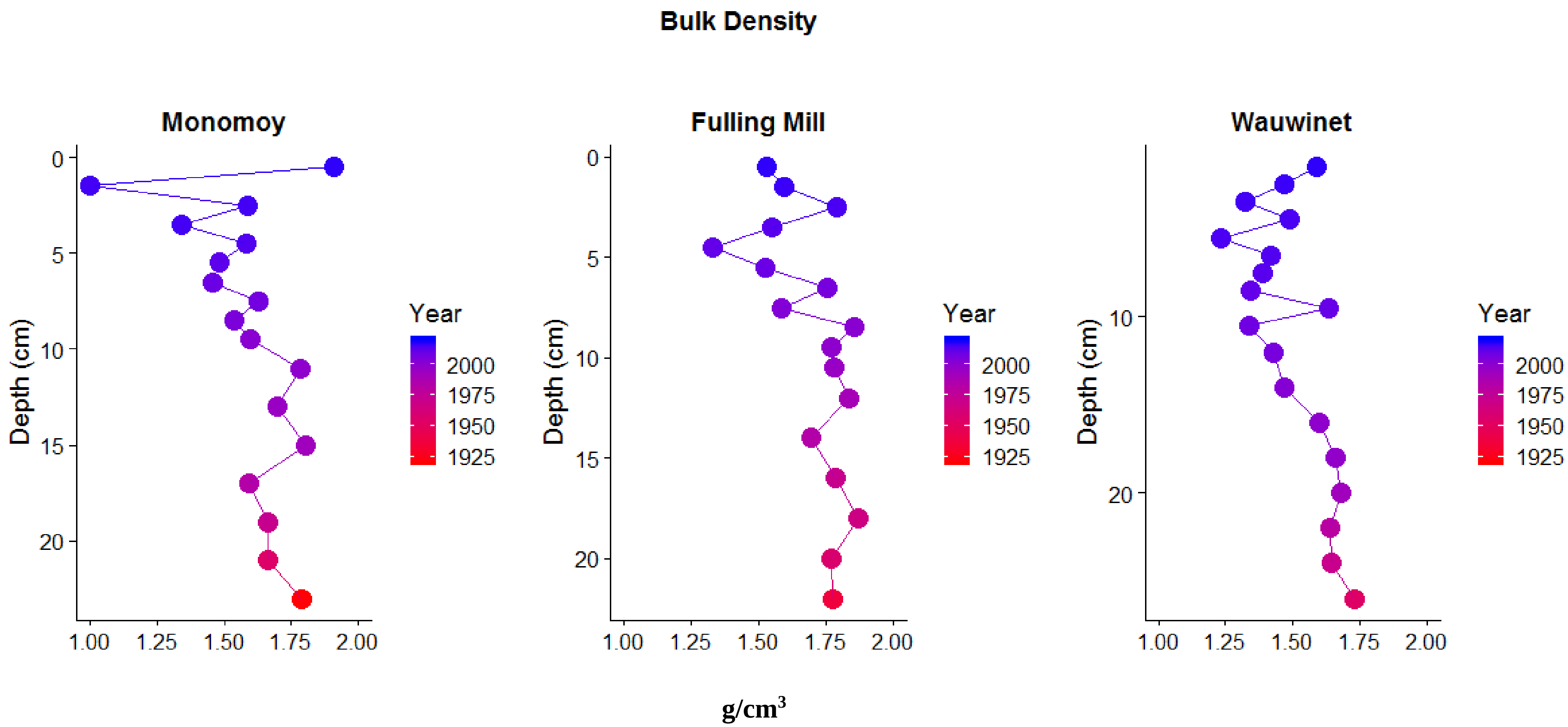

Figure 7. Bulk density (g/cm^3) of sediment cores collected from Monomoy, Fulling Mill, and Wauwinet.

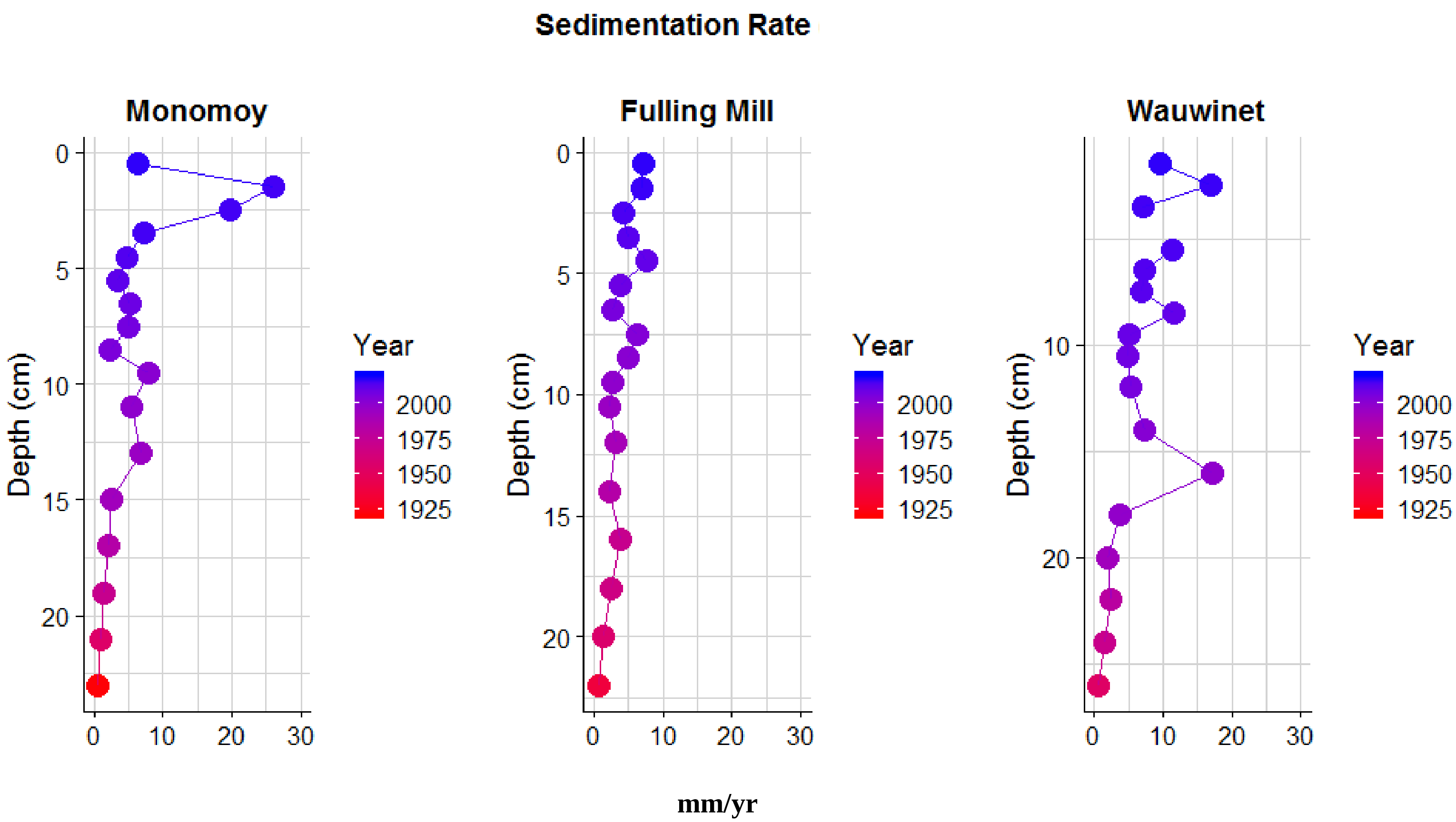

Figure 8. Sedimentation rates at Monomoy, Fulling Mill, and Wauwinet.

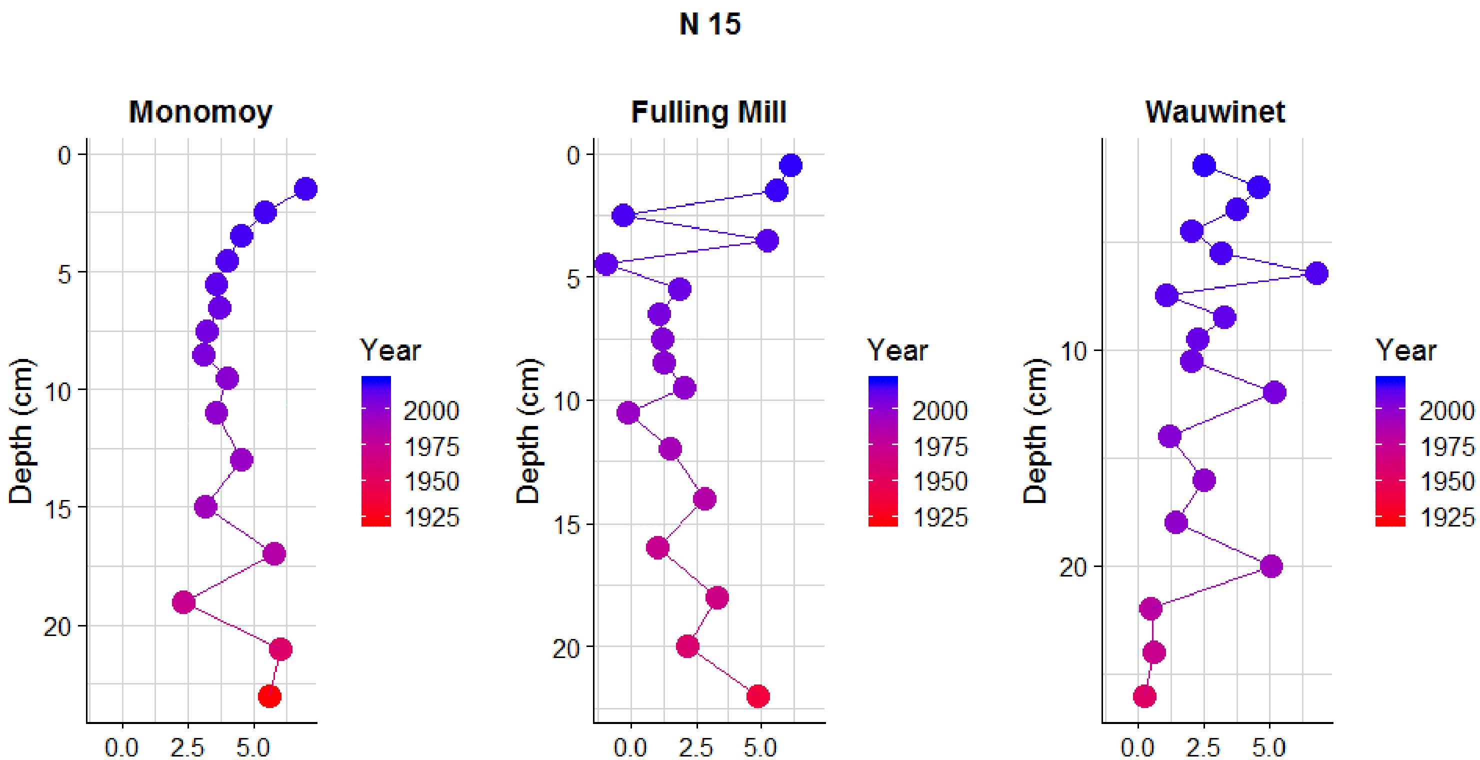

Figure 9. $\delta^{15}N$ values values at Monomoy, Fulling Mill, and Wauwinet. $\delta^{15}N$ can be used to identify nutrient pollution sources.

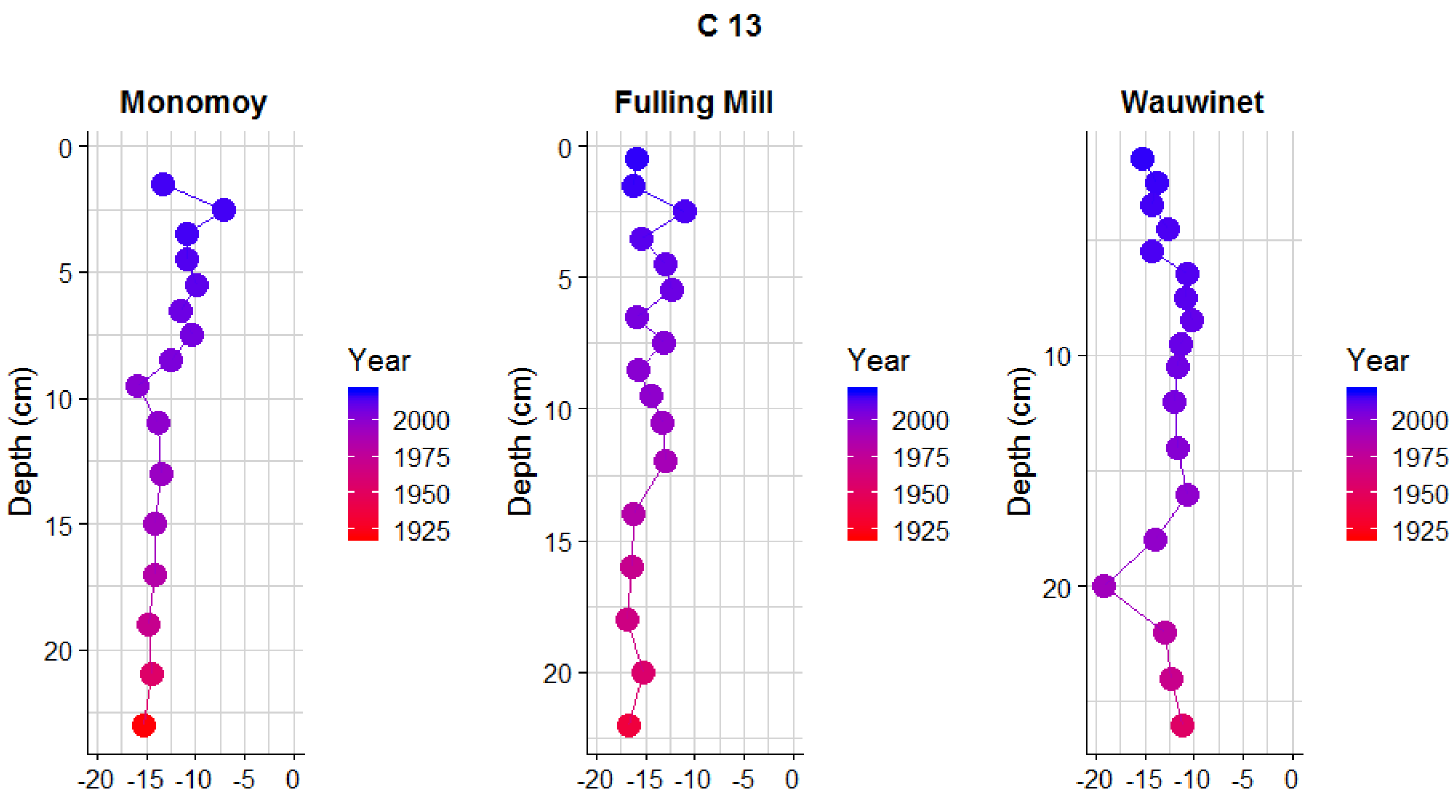

Figure 10. $\delta^{13}C$ values values at Monomoy, Fulling Mill, and Wauwinet. $\delta^{13}C$ can be used to identify sources of carbon.

MAY

Site	Avg. Intensity	% Surface Irradiance
Monomoy (Top)	3274	23.07
Monomoy (Bottom)	2409	16.97
Pimny's Point (Top)	2705	19.06
Pimny's Point (Bottom)	0	0.00
Fulling Mill (Top)	3615	25.46
Fulling Mill (Bottom)	2115	14.90
Quaise (Top)	3176	22.37
Quaise (Bottom)	1454	10.24
Pocomo (Top)	2825	19.90
Pocomo (Bottom)	1836	12.93
Wauwinet (Top)	2920	20.57
Wauwinet (Bottom)	2259	15.91

June

Site	Avg. Intensity	% Surface Irradiance
Monomoy (Top)	3155	28.76
Monomoy (Bottom)	1632	14.88
Pimny's Point (Top)	3217	29.32
Pimny's Point (Bottom)	2276	20.75
Fulling Mill (Top)	3063	27.92
Fulling Mill (Bottom)		
Quaise (Top)	2772	25.27
Quaise (Bottom)	1788	16.30
Pocomo (Top)		
Pocomo (Bottom)		
Wauwinet (Top)	2082	18.98
Wauwinet (Bottom)	1087	9.91

Figure 11. Average percent of light reaching the top and bottom of the eelgrass canopy between 10:00 and 14:00 for the months of May and June.

July

Site	Avg. Intensity	% Surface Irradiance
Monomoy (Top)	3654	24.46
Monomoy (Bottom)	2498	16.72
Pimny's Point (Top)	4694	31.43
Pimny's Point (Bottom)	1081	7.23
Fulling Mill (Top)	4304	28.81
Fulling Mill (Bottom)	2413	16.15
Quaise (Top)	3676	24.61
Quaise (Bottom)	2311	15.47
Pocomo (Top)	3281	21.96
Pocomo (Bottom)	1288	8.62
Wauwinet (Top)	3180	21.29
Wauwinet (Bottom)	2369	15.86

August

Site	Avg. Intensity	% Surface Irradiance
Monomoy (Top)	2858	26.49
Monomoy (Bottom)	1262	11.70
Pimny's Point (Top)	2299	21.32
Pimny's Point (Bottom)	358	3.32
Fulling Mill (Top)	4878	45.23
Fulling Mill (Bottom)	2514	23.30
Quaise (Top)	3031	28.10
Quaise (Bottom)	1657	15.36
Pocomo (Top)	2442	22.64
Pocomo (Bottom)	1811	16.79
Wauwinet (Top)	3107	28.80
Wauwinet (Bottom)	1939	17.98

Figure 12. Average percent of light reaching the top and bottom of the eelgrass canopy between 10:00 and 14:00 for the months of July and August.

May

Site	MIN (°C)	MAX (°C)	AVG (°C)	% Time > 25 °C	% Time > 28 °C
Monomoy (Top)	14.04	28.75	17.74	1.4	0.2
Monomoy (Bottom)	13.94	28.66	17.72	1.3	0.2
Pimny's Point (Top)	13.85	28.75	18.19	1.3	0.3
Pimny's Point (Bottom)	14.23	28.75	17.89	1.3	0.3
Fulling Mill (Top)	13.65	29.45	18.73	1.3	0.4
Fulling Mill (Bottom)	13.75	29.35	18.66	1.1	0.3
Quaise (Top)	14.52	29.45	18.67	1.2	0.3
Quaise (Bottom)	14.52	28.85	18.54	1.2	0.3
Pocomo (Top)	14.33	28.75	18.94	1.1	0.3
Pocomo (Bottom)	14.13	28.56	18.78	1.1	0.3
Wauwinet (Top)	16.05	29.65	19.11	1.0	0.3
Wauwinet (Bottom)	15.95	29.75	19.02	0.9	0.2

June

Site	MIN (°C)	MAX (°C)	AVG (°C)	% Time > 25 °C	% Time > 28 °C
Monomoy (Top)	18.14	29.15	22.19	10.6	0.7
Monomoy (Bottom)	18.24	27.86	22.02	9.3	0.0
Pimny's Point (Top)	18.33	30.76	22.72	19.4	1.6
Pimny's Point (Bottom)	18.43	30.05	22.57	15.3	1.1
Fulling Mill (Top)	18.90	30.46	23.20	27.2	2.8
Fulling Mill (Bottom)					
Quaise (Top)	18.90	29.35	23.14	25.3	1.6
Quaise (Bottom)	19.00	29.15	23.06	24.4	1.3
Pocomo (Top)					
Pocomo (Bottom)					
Wauwinet (Top)	19.28	30.56	23.66	24.8	3.7
Wauwinet (Bottom)	19.28	30.15	23.51	22.3	3.2

July

Site	MIN (°C)	MAX (°C)	AVG (°C)	% Time > 25 °C	% Time > 28 °C
Monomoy (Top)	21.86	29.65	25.02	52.7	5.1
Monomoy (Bottom)	21.76	29.65	24.90	49.2	4.5
Pimny's Point (Top)	21.57	30.56	25.41	58.4	7.3
Pimny's Point (Bottom)	21.76	29.65	25.15	54.1	4.3
Fulling Mill (Top)	21.76	30.66	25.75	64.9	9.9
Fulling Mill (Bottom)	21.95	30.36	25.74	65.8	8.2
Quaise (Top)	20.23	30.17	25.70	65.3	9.5
Quaise (Bottom)	20.04	30.95	25.65	64.7	9.0
Pocomo (Top)	20.33	30.17	26.16	77.6	13.0
Pocomo (Bottom)	19.95	30.44	25.90	73.3	9.3
Wauwinet (Top)	20.04	31.94	26.25	77.9	12.3
Wauwinet (Bottom)	19.57	30.84	26.14	67.0	10.8

August

Site	MIN (°C)	MAX (°C)	AVG (°C)	% Time > 25 °C	% Time > 28 °C
Monomoy (Top)	21.95	29.55	24.77	41.6	1.5
Monomoy (Bottom)	22.05	29.05	24.61	35.2	0.7
Pimny's Point (Top)	21.86	29.55	24.92	46.7	1.1
Pimny's Point (Bottom)	21.76	28.85	24.69	38.8	0.8
Fulling Mill (Top)	21.76	30.66	24.98	45.9	3.4
Fulling Mill (Bottom)	21.66	29.75	24.87	44.0	2.2
Quaise (Top)	22.33	29.05	25.03	51.2	1.7
Quaise (Bottom)	22.43	28.85	25.07	51.1	1.4
Pocomo (Top)	21.86	29.55	25.27	53.1	4.7
Pocomo (Bottom)	21.86	29.25	25.26	53.0	3.4
Wauwinet (Top)	22.91	29.95	25.54	59.9	5.0
Wauwinet (Bottom)	22.91	29.65	25.44	56.5	2.8

Figure 13. Descriptive statistics for temperature between the months of May and August. Continuous water temperature were collected at 15 minute intervals.