Final Research Report to:

Natural Resources Department, Town of Nantucket, MA

Project Title:

Continuing a long-term benthic ecological monitoring survey for bay scallops, channeled whelk, knobbed whelk, and eelgrass in Nantucket Harbor, MA – September 2022

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Abstract

Nantucket Harbor in Nantucket, MA, is home to a diverse and productive marine ecosystem including expansive eelgrass (*Zostera marina*) beds that are home to one of the few remaining populations of bay scallops (*Argopecten irradians*) that supports both commercial and recreational fisheries. Furthermore, its populations of channeled (*Busycotypus canaliculatus*) and knobbed whelk (*Busycon carica*) also support a commercial fishery. A benthic ecological survey of Nantucket Harbor was founded by Dr. Robert Kennedy and Dr. Peter Boyce in 2006. This survey was conducted annually from 2006 through 2014. The survey was restarted and completed during 2019. During 2020 and 2021 there were funding and logistical challenges stemming from the Covid-19 pandemic. The ecological survey of Nantucket Harbor was completed during 2020, 2021, and 2022 with funding, in-kind support, and field work assistance from the Town of Nantucket's Natural Resources Department (NRD). An analysis of data from 2006-2014 and 2019- 2022 yielded the following results:

- An intensive bay scallop restoration project was begun by Tara Riley, Shellfish Biologist for the Town of Nantucket's Natural Resources Department, in 2010. For the following comparisons, 2006-2010 was considered before and 2011-2014 and 2019-2022 after restoration had started.
- The average abundance of total (adult + juvenile) bay scallops increased by 132% in the years following the initiation of an intensive bay scallop population restoration efforts. Furthermore, the average abundance of seed (juvenile) and adult bay scallops increased by 382% and 16%, respectively, after the bay scallop restoration work had begun. This provides strong evidence that the efforts of Tara Riley and the NRD have been very effective and essential to maintaining the population of bay scallops in Nantucket Harbor.
- The lower overall increase in adult versus juvenile scallop abundance reflects high cumulative mortality over time. A similar trend has been observed during bay scallop restoration work in the Peconic Bays of Long Island, NY. Since the high mortality of scallops between seed and adult life stages is a clear limitation to restoration efforts, this needs to be investigated further. Developing an effective strategy to increase the survival of bay scallops between the juvenile and adult life stages is essential for the long-term sustainability of the bay scallop population.
- Filamentous macroalgae peaked during 2012 with an average percent cover of 42% and declined dramatically through 2019 and 2020 with an average percent cover of only 2% and 6% during 2019 and 2020, respectively. An uptick was observed during 2021, with an average percent cover of 24% but dropped again in 2022 to 7%. Prior to 2021, filamentous macroalgae were primarily composed of an invasive mat-forming blue-green filamentous algae (*Hydrocoleum* sp.), however another filamentous algae, *Ectocarpus siliculosus*, was observed at relatively high abundances during 2021 and 2022.
- The density of knobbed whelk during 2022 was within the higher half of the range of observed values whereas the density of channeled whelk was within the lower half encountered over the time range of the survey.
- Overall eelgrass coverage declined within Nantucket Harbor from 2006 through 2014 and 2019 through 2022. The survey locations with relatively high eelgrass coverage during 2006 (>40%) saw a steady decline that resulted in a severe reduction in the average percent cover from 2006 through 2022 from 56% to 20%. The overall average percent cover of eelgrass in Nantucket Harbor during 2022 (16%) was less than half of that observed during 2006 (37%). The severe decline in eelgrass coverage between 2006 and 2022 is very concerning given the plethora of ecosystem services that eelgrass provides including serving as an essential habitat for bay scallops. Halting and reversing the decline of eelgrass in Nantucket Harbor is imperative to maintaining a healthy, functioning, productive coastal ecosystem.
- Continuing an annual benthic ecological survey of Nantucket Harbor allows for a sustained assessment of the health of bay scallop, channeled whelk, knobbed whelk, and eelgrass

populations. Since dramatic changes can occur in these populations between years, an annual assessment is essential to understanding the effectiveness of efforts to restore these important components of the ecosystem and respond to potential future decline before it is too late.

Report Summary

Coastal embayments throughout southern New England were once home to expansive eelgrass (*Zostera marina*) meadows as well as a diverse and abundant assemblage of fish and shellfish populations. These healthy ecosystems fueled vibrant coastal communities. Today, due to a range of human-induced environmental impacts including nutrient loading and overharvesting, many of these areas are now characterized by a loss of eelgrass and severe declines in fish and shellfish populations. The waters around Nantucket, Massachusetts have seen declines in bay scallops (*Argopecten irradians*), channeled whelk (*Busycotypus canaliculatus*), and eelgrass, among other species, over the years. However, in response, there have been concerted efforts directed at mitigating human-induced environmental issues driving these decreases, as well as intensive shellfish restoration work aimed at facilitating the recovery of bay scallop populations.

Long-term benthic ecological monitoring programs are critical to understanding how human impacts are altering coastal ecosystems and evaluating the efficacy of shellfish population restoration work and initiatives aimed at mitigating environmental issues. A benthic ecological survey had been conducted every September for nearly a decade (2006 through 2014) that included sites throughout Nantucket Harbor, Massachusetts. This survey had been conducted by the Maria Mitchell Association (MMA) (most recently led by Dr. Peter Boyce). The survey led by Dr. Peter Boyce (initially led by Dr. Bob Kennedy) of MMA included extensive dive surveys of bay scallop, channeled whelk, and knobbed whelk (*Busycon carica*) population density and size distribution, as well as eelgrass distribution, eelgrass percent cover, eelgrass canopy height, and macroalgae percent cover throughout Nantucket Harbor.

During 2019, the Great Harbor Yacht Club Foundation (GHYCF) funded an effort to restart this long-term survey. With the support of the GHYCF and expansive help from personnel of the Town of Nantucket's Natural Resources Department (NRD), a similar benthic ecological survey was conducted during September 2019 that built on the long-term ecological survey previously founded by Boyce and Kennedy. The survey during 2019 served to re-establish the aforementioned long-term annual benthic ecological monitoring program, which included the evaluation of eelgrass and macroalgae coverage as well as the abundance and age/size distribution of bay scallops, channeled whelk, and knobbed whelk. In total, 39 locations within Nantucket Harbor were surveyed during 2019. These were the same 39 locations surveyed during 2014. Since the survey methods used during 2019 emulated those used by Dr. Peter Boyce and the Maria Mitchell Association, this data was able to be used to evaluate whether these environmental parameters have changed over time from 2006 through 2014 to 2019. However, since location changes and site additions over the course of the past survey work had been made prior to 2014, only a subset of the 39 sites surveyed during 2019 could be directly compared across all years.

Funding and logistical challenges stemming from the Covid-19 pandemic arose during 2020 and 2021. However, with the expansive help of the NRD, we successfully navigated around those issues. The NRD fully funded the benthic ecological survey during 2020, 2021, and 2022. Furthermore, the staff from the NRD were able to shift a large amount of their efforts to help with the benthic ecological survey during September of 2020, 2021, and 2022. The survey was only possible during 2020, 2021, and 2022 due to funding and extensive in-kind support (field work assistance from NRD personnel, use of an NRD research vessel, and NRD dive tanks) from the NRD. The same survey sites and survey methods used during 2019 were used during 2020, 2021, and 2022. Therefore, data collected during 2006 through 2014 and 2019 through 2022 could be compared directly.

We believe that this data has valuable implications for gauging the status of eelgrass, macroalgae, bay scallops, channeled whelk, and knobbed whelk in Nantucket Harbor. Furthermore, this gives us insight into the effectiveness of ongoing efforts to restore bay scallop populations and reduce anthropogenic impacts that negatively impact eelgrass meadows. Additionally, it provides us with insight into understanding how channeled and knobbed whelk populations have changed over time.

When looking at eelgrass at survey locations throughout Nantucket Harbor, the overall average percent cover from 2006 through 2022 shows a drastic decline from an average percent cover of eelgrass of 37% in 2006 to 16% during 2022 (Figure A). While an uptick in the average percent coverage of eelgrass during 2014 and 2019 appeared to suggest the possibility of eelgrass recovery, the lowest 3 years of average percent coverage of eelgrass out of the 13 years of data were observed during 2020, 2021, and 2022 (Figure A). The average observed percent cover of eelgrass in the upper and lower areas of Nantucket Harbor had increased from 2014 to 2019 from 17% to 37% and 31% to 42%, respectively. With cautious optimism, during 2019 it was suspected that eelgrass might be recovering in these two regions of the harbor. However, 2020 had the third lowest average percent cover of eelgrass observed in the upper region of the harbor (13%) and the third lowest in the lower region (26%) that had been observed since the survey began in 2006. In 2021, eelgrass coverage in the upper region of the harbor was slightly higher (20%) but declined down to 14% during 2022. The lower region of the harbor did not see any recovery in eelgrass coverage during 2021 (26%) and 2022 (25%). The central region of the harbor had the lowest observed average eelgrass percent cover during 2021 and 2022 (9%) with 11% during 2020 – which is less than a third of the average coverage observed during 2006 (34%) and 2007 (35%).

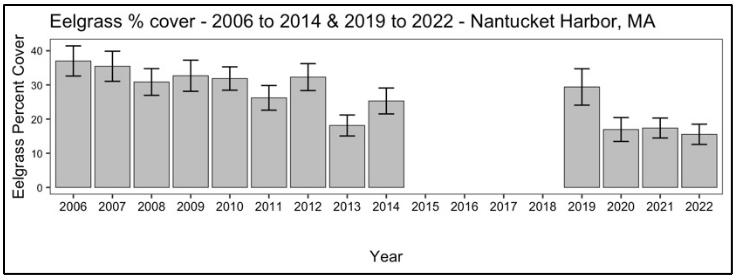

Figure A. Averages of eelgrass percent cover ± standard error of the mean (SE) of 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA.

Given the immense number of ecosystem services that eelgrass provides, including serving as essential nursery habitat for many important commercially and recreationally important fish and bivalves, the overall declining trajectory of eelgrass throughout the harbor is very concerning. Additionally, survey locations with high eelgrass percent cover during 2006 (>40%) experienced severe declines in eelgrass coverage from 2006 to 2022 from 56% to 20%. Since these survey locations with high eelgrass coverage are likely disproportionately beneficial as nursery areas for many species of fauna, including bay scallops, these declines are particularly worrisome. It appears likely that blooms of macroalgae (elaborated on below) and phytoplankton (fueled by nutrient loading from fertilizer and waste water input into the harbor) that restrict light availability to eelgrass are responsible (at least in part) for driving these declines in eelgrass percent cover. Efforts to curb the amount of nutrients that enter Nantucket Harbor have been underway with the intention of mitigating blooms of phytoplankton and macroalgae. These have come in the form of fertilizer reduction programs and expanding the sewer system network on the island. An increasing number of oysters (on farms and as part of wild oyster restoration work) in the harbor may also aid in improving the water clarity since oysters consume phytoplankton. Herbicide run-off via terrestrial application may also be contributing to the observed declines in eelgrass meadows. Increased turbidity (and associated reduced water clarity) from wind and boat traffic as well as eelgrass disease may also be potential factors contributing to eelgrass declines. Heightened efforts to mitigate anthropogenic impacts on eelgrass meadows are clearly needed to allow this important habitat-forming species to recover. More research concerning phytoplankton blooms, turbidity, herbicide run-off, disease, and other potential factors that could contribute to eelgrass declines would be of paramount importance to prioritizing human impact mitigation and eelgrass restoration efforts.

The average percent cover of branching macroalgae was markedly different between years from 2006 through 2022 when considering all sites throughout Nantucket Harbor that were evaluated consistently during all years of the survey. The highest average branching macroalgal percent cover was found during 2007 at 37%. The average percent cover of branching macroalgae during 2022 (7%) was the lowest observed over the course of the surveys. Within the lower region of Nantucket Harbor, the average percent cover of branching macroalgae during 2022 (15%) was the third lowest observed since the beginning of the survey in 2006. During 2022, the central region and upper regions of Nantucket Harbor had the lowest coverage of branching macroalgae in the history of the survey at 3% and 4%, respectively.

The filamentous macroalgae observed in the survey was defined as consisting of long visible filaments that are a single cell in thickness. Prior to 2021 (2006 – 2014 and 2019 – 2020), over 95% of the filamentous macroalgae observed in the survey were believed to be a species of blue-green macroalgae (cyanobacteria) that had previously been identified as *Lyngbya* sp. but is now considered to be most closely related to *Hydrocoleum lyngbyaceum.* The average percent cover of filamentous macroalgae gradually increased in Nantucket Harbor from 2006 through 2012, where it peaked at an average of 42% cover. After 2012, the average percent cover of filamentous macroalgae declined and was only 2% in the harbor during 2019, the lowest observed other than in 2006 and 2007. During 2020, percent cover of filamentous macroalgae increased to 6% however the coverage was still relatively low in comparison to the peak during 2012. During 2021, a large portion of the observed filamentous macroalgae also included *Ectocarpus siliculosus*, a filamentous brown macroalgae. The average percent cover of filamentous macroalgae increased to 24% during 2021 but dropped to 7% during 2022. This appears to suggest that other factors besides those mentioned above are contributing to declines in eelgrass coverage throughout the harbor. While branching macroalgae and filamentous macroalgae were both substantially lower during 2020 and 2022 in comparison to peak levels (except for branching macroalgae in the lower region of the harbor), continued eelgrass declines were observed throughout the harbor.

While the overall average densities of bay scallops observed throughout Nantucket Harbor have varied over the years that the survey has been conducted, the highest average densities of seed (juvenile) scallops were seen during 2011, 2012, 2019, 2020, and 2022 – all after intensive bay scallop population restoration efforts were initiated during 2010 by Tara Riley, Shellfish Biologist of the Town of Nantucket Natural Resources Department (NRD). The lower densities of seed bay scallops observed during 2013, 2014, and 2021 were similar to those observed in the years before restoration work began (prior to 2011). These levels may represent natural fluctuations but may also reflect low survival of seed bay scallops driven in part by harmful algal blooms during those years.

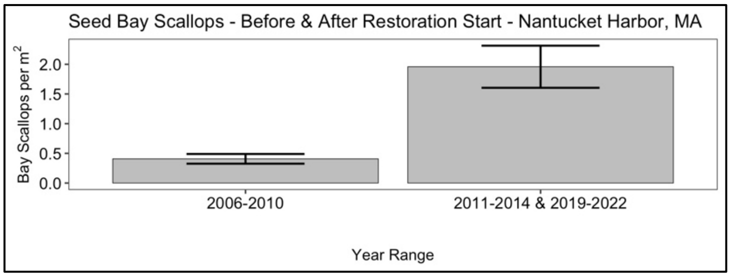

Figure B. Average seed bay scallop densities ± standard error of the mean (SE) before and after the beginning of bay scallop population restoration in 2010. Data were obtained at 28 sites throughout Nantucket Harbor, MA, that were surveyed consistently from 2006-2014 and 2019-2022.

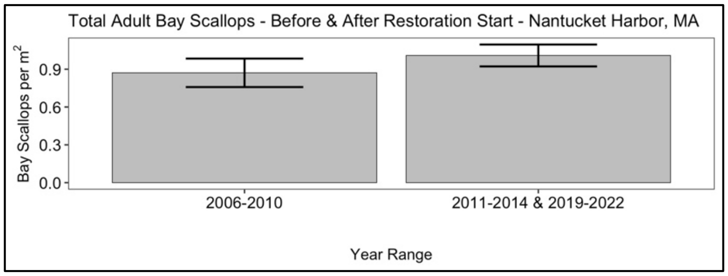

Figure C. Averages adult bay scallop densities ± standard error of the mean (SE) before and after the beginning of bay scallop population restoration in 2010. Data were obtained at 28 sites throughout Nantucket Harbor, MA, that were surveyed consistently from 2006-2014 and 2019-2022.

When all age classes of adult bay scallops were pooled, the five highest average densities of adult bay scallops were observed during 2007, 2009, 2011, 2012, and 2019. When comparing the five years of data that we have before the bay scallop population restoration began in 2010 to the eight years of data after the restoration work began, the average density of all bay scallops (including all ages) is seen to have increased by 132% after the restoration work was initiated. The average abundance of seed bay scallops and total adult bay scallops increased by 382% and 16%, respectively, after the bay scallop restoration work started (Figure B, C). While many environmental factors influence bay scallop populations, this data provides strong supporting evidence that the NRD restoration efforts contributed to a substantial increase in the population of bay scallops in Nantucket Harbor. However the low rate of survival between the seed and adult life stages of bay scallops is an issue that merits further study to evaluate the causes and whether survival can be improved.

While there were annual fluctuations in channeled and knobbed whelk densities from 2006 through 2014 and 2019 through 2020, densities of both species generally declined during this period. The average density of knobbed whelk was higher than that of channeled whelk during 8 of the years of the survey. This was expected to some extent since channeled whelk are targeted by local commercial fishermen. The density of knobbed whelk during 2021 was the lowest observed over the history of the survey whereas the density of channeled whelk in 2019-2021 increased and were within the higher half of the range of observed values from the survey. It is important to note that there was little, if any, commercial whelk harvesting within the harbor from 2019 through 2021. Between 2021 and 2022, the density of channeled whelk decreased while the density of knobbed whelk increased. During 2022, the average density of channeled whelk was within the lower half of values recorded whereas the average density of knobbed whelk was within the higher half.

Despite a range of human-induced impacts on its marine ecosystems, Nantucket contains one of the few remaining bay scallop populations that supports both commercial and recreational fisheries. The coastal waters of Nantucket have relatively healthy eelgrass meadows within southern New England in comparison to other locations, supporting a diverse and abundant assemblage of fish, crustaceans, whelk, and bivalves. With that said, there are clear indications that eelgrass health has declined severely since 2006 and that the status of the population of bay scallops is concerning and appears to be far from stable. Continuing to build on the long-term benthic ecological survey of Nantucket Harbor over the years allows for the pulse of this valuable ecosystem to be monitored. It allows for an assessment of the effectiveness of bay scallop restoration efforts, the health and distribution of eelgrass meadows, and the status of channeled and knobbed whelk. With this information on ecosystem parameters, prioritization of particular response efforts to curb human impacts can be better informed and potentially more effective.

Acknowledgements

We are sincerely thankful that the Town of Nantucket's Natural Resources Department (NRD) was willing to come up with creative ways to ensure that the benthic ecological survey of Nantucket Harbor happened during September of 2022. They worked incredibly hard to frontload their intensive shellfish hatchery work schedule to allow for all of their personnel to provide an enormous amount of assistance in the form of conducting dive surveys and all required fieldwork on the research vessel during the survey (especially Tara Riley, Jeff Carlson, Leah Hill, Joe Minella, David Berry, Griffin Harkins, and Josh Whitehead). While this is likely not possible in future years given the demanding work in the shellfish hatchery and field associated with their bay scallop, oyster, and hard clam restoration efforts, this help was absolutely essential to the completion of the survey this year. The NRD also provided their research vessel and SCUBA tanks to use for the dive survey. The help in the form of personnel and dive tank/research vessel usage allowed for a greatly reduced cost of the survey during 2022. The NRD also fully funded the cost of the project during 2020, 2021, and 2022. This project truly could not have happened during 2022 without all of the collaboration, support, and funding from the NRD.

We are immensely appreciative of the support of the Nantucket Land Bank, in particular Jesse Bell, for providing us with housing for the duration of the survey.

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Kennedy). And a special thank you to Dr. Val Hall for her field work assistance this year and for providing us with the hard copies of the previous years of data. None of this would have been possible without all of you and it has been an honor to be able to continue this important long-term monitoring dataset!

Introduction

Embayments along the coast of southern New England were historically characterized by expansive seagrass meadows whose complex structure supported an immense diversity and abundance of life including fish, crustaceans, and bivalves (Orth and Heck 1980, Jackson et al. 2001a, Hughes et al. 2002, Fonseca and Uhrin 2009). The ecosystem services provided by these highly productive ecological communities served as the foundation for many vibrant coastal cities and towns (Halpern et al. 2008, Mackenzie 2008, Cullen-Unsworth et al. 2014). Due to centuries of anthropogenic environmental impacts – including overharvesting, nutrient loading (from wastewater and fertilizers), a changing climate, and species introductions – many of these ecosystems have experienced severe reductions in seagrass coverage, declines in water quality, and a loss of biodiversity and species abundance of fish, crustaceans, and bivalves such as bay scallops (*Argopecten irradians*), hard clams (*Mercenaria mercenaria*), and eastern oysters (*Crassostrea virginica*) (Jackson et al. 2001b, Lotze 2006, Jackson 2008, Mackenzie 2008, Costello and Kenworthy 2011, Tettelbach et al. 2013).

Nantucket Island is located approximately 30 miles off the coast of mainland Massachusetts. The coastal ecosystems of Nantucket are undoubtedly subjected to similar human-induced environmental issues as the mainland. However, the relatively small area of the watershed in comparison to those found on the mainland, a relatively low human population density, and the lack of a developed industrial area have likely reduced the severity of many anthropogenic impacts on the marine ecosystem within Nantucket Harbor. An understanding of how these coastal ecosystems are changing over time is essential to developing ways to mitigate human-induced impacts and facilitate their restoration (Sukhotin and Berger 2013).

Bay scallop populations declined dramatically throughout estuaries of southern New England and Long Island, NY, during the late 1980s and early 1990s. This was primarily attributed to harmful algal blooms throughout the region (Mackenzie 2008, Tettelbach et al. 2013). Other potential factors responsible for declines in bay scallop populations include the loss of seagrass habitat due to nutrient loading that fuels phytoplankton and macroalgae blooms in turn restricting light availability (Serveiss et al. 2004). As both filter feeders and prey for many other organisms, bay scallops serve important ecological roles (Palmer and Williams 1980, Carroll et al. 2015). Historically, bay scallops have supported recreational and commercial fisheries throughout their range, with cascading economic effects throughout coastal communities (Mackenzie 2008). The population crashes of bay scallops, coupled with their ecological, economic, and cultural value, spurred the development of several large-scale bay scallop restoration projects (Mackenzie 2008, Tettelbach et al. 2015).

While the bay scallop population in Nantucket did not drop nearly as low as in other coastal ecosystems, the number of bushels of bay scallops that were landed between 1980 and 2000 dropped by 87 percent (NSMP 2012). Tara Riley, Shellfish Biologist of the Town of Nantucket's Natural Resources Department (NRD), began an intensive bay scallop population restoration project in 2010 in Nantucket and Madaket Harbors. Tara Riley, Leah Hill, Jeff Carlson, Thaïs Fournier, Joe Minella, and David Berry have since expanded that work to include eastern oyster and hard clam populations. Since eastern oysters, hard clams, and bay scallops are prolific filter feeders, the restoration of these species may increase the amount of phytoplankton that is consumed by bivalves in Nantucket Harbor and mitigate the risk of harmful algal blooms (Newell 2004, Li et al. 2009, Wall et al. 2011). In turn, this could indirectly enhance juvenile bivalve survival and improve seagrass health if water clarity improves. Evaluating the abundance and population structure of bay scallops throughout Nantucket Harbor over time allows for the determination of the bay scallop population status and the efficacy of bay scallop restoration efforts.

Seagrasses have declined throughout the globe (Orth et al. 2006, Waycott et al. 2009). Declines in seagrasses are particularly concerning given their high ecological importance (Jackson et al. 2001a). As foundation species, seagrasses are habitat-forming organisms that many other species depend on for refuge from predators, spawning, and foraging (Hughes et al. 2009). Eelgrass (*Zostera marina*) is the dominant species of seagrass found in the waters of Nantucket (NSMP 2012). Eelgrass provides important habitat for many ecologically and economically valuable species of fish, crustaceans, and bivalves, including bay scallops (Heck and Orth 1980, Orth and Heck 1980, Pohle et al. 1991, Hughes et al. 2002). Declines in eelgrass can be driven by many factors, including nutrient loading (from septic systems and fertilizers) that fuels blooms of phytoplankton (thereby increasing water turbidity) and macroalgal blooms (that cover over eelgrass), a changing climate, coastal development, and physical damage from vessels (Cosper et al. 1987, Short and Wyllie-Echeverria 1996, Hauxwell et al. 2001, McGlathery 2001, Anderson et al. 2002, Hughes et al. 2009). Reductions in eelgrass coverage have been shown to negatively impact the abundance of many eelgrass-associated species (Hughes et al. 2002, Fonseca and Uhrin 2009). Bay scallops are among the many species that can be adversely impacted by a loss of eelgrass habitat (Mackenzie 2008, Fonseca and Uhrin 2009, Carroll et al. 2015). For instance, during 1930s, vast die-offs of eelgrass meadows along the coastline of North America and Europe were driven by eelgrass "wasting disease" (Fonseca and Uhrin 2009). Severe declines in bay scallop populations followed suit throughout much of their range (Fonseca and Uhrin 2009). This demonstrated the importance of eelgrass to bay scallop survival on a large scale. Research has documented that eelgrass has declined regionally (Hughes et al. 2002, Costello and Kenworthy 2011). Between 2006 and 2014, many sites included in the Maria Mitchell Association's benthic ecological survey in Nantucket Harbor demonstrated declining coverage of eelgrass (Peter Boyce unpublished data). To mitigate the occurrence of harmful algal blooms, efforts to curb nutrient inputs into Nantucket Harbor have been underway, including the implementation of responsible fertilizer-use practices and installation of sewer lines in certain areas of the island. Since declines in eelgrass coverage can have detrimental consequences for the entire ecosystem, monitoring of the density, canopy height, and distribution of eelgrass throughout Nantucket Harbor is essential to understanding whether efforts to address the causes of declines in eelgrass have been effective and whether current levels of restoration efforts are sufficient to maintain/increase eelgrass populations. This is particularly important considering a seasonal blue-green macroalgae bloom that has occurred in certain areas of the harbor in recent years and appears to be smothering the eelgrass during periods of heightened abundance (Peter Boyce unpublished data). This blue-green macroalgae (cyanobacteria) had been previously identified as *Lyngbya* sp. but has since been corrected to be most closely related to *Hydrocoleum* sp. (Moisander 2015).

Channeled whelk (*Busycotypus canaliculatus*) and knobbed whelk (*Busycon carica*) are two other species that are harvested commercially in southern New England, including Nantucket Harbor (Kaplan 1998, NSMP 2012). The directed commercial harvesting is done entirely using pots; however, both species are caught as bycatch in trawl fisheries. Channeled whelk are the dominant of the two whelk species captured through the use of pots, with knobbed whelk making up a minor part of the total landings. While Massachusetts does have a minimum legal size in place, the most recent stock assessment indicates that the channeled whelk population is currently in decline and experiencing overfishing, posing an immediate threat to the longevity of this highly lucrative fishery (Edmundson 2016, Nelson et al. 2018). Given that undersized whelk that are captured and returned to the water through the directed pot fishery likely experience incredibly low, if any, post-release mortality, it appears that shifts in harvest regulations have the potential to help this fishery recover and become a long-term sustainable, profitable fishery. Only a few commercial fishermen are allowed to fish for these whelk species within Nantucket Harbor. Understanding whether the channeled and knobbed whelk populations have changed within the harbor over time and whether the limited amount of fishing pressure within

Nantucket Harbor has mitigated the regional declines observed in Nantucket Sound is useful to improving management measures to help sustain this lucrative fishery.

The Maria Mitchell Association (MMA) conducted a benthic ecological survey every September from 2006 through 2014 at sites throughout Nantucket Harbor. Founded by Dr. Bob Kennedy and Dr. Peter Boyce of MMA, this effort was most recently led by Dr. Peter Boyce. Forrest Kennedy and Dr. Valerie Hall also contributed substantially to these survey efforts, all former or current (Dr. Hall) associates of MMA. The survey was originally funded for two years by the United States Environmental Protection Agency (EPA) and afterwards, through 2014, was funded by MMA and several other local donors and businesses. With the generous financial support of the Great Harbor Yacht Club Foundation (GHYCF) and expansive help from personnel of the Town of Nantucket Natural Resources Department (NRD), a similar benthic ecological survey of Nantucket Harbor was conducted during September of 2019. The purpose of this dive survey work during 2019 was to build on the foundation of the long-term ecological survey that had been started by Boyce and Kennedy in 2006. The sites used during the 2019 survey were sites included during 2014 of MMA's prior research. The survey was conducted again during September 2020, 2021, and 2022, entirely funded by the NRD. The sites used during 2020, and 2021, and 2022 were the same as 2019. With the integral help of the NRD, we managed to come up with creative solutions to the issues associated with the Covid-19 pandemic that arose during 2020 and 2021. The benthic ecological survey would not have been possible during 2020, 2021, and 2022 without the financial support from the NRD which funded the entire project during these years. Funding and housing constraints, partially driven by concerns over Covid-19, reduced the number of researchers that could travel to Nantucket and work on the survey in comparison to 2019. Fortuitously, the NRD personnel had accomplished many of their restoration goals much earlier in the season than usual, which allowed time to shift a large amount of their efforts to shift to focus on helping with the benthic ecological survey during September of 2020, 2021, and 2022. While it is extremely unlikely to expect that this will be possible in future years, extensive in-kind support from the NRD in the form of field work help from the NRD personnel along with the use of an NRD research vessel and NRD dive tanks made the survey possible during 2020, 2021, and 2022.

During 2019, 2020, 2021, and 2022 each site was surveyed for bay scallop density and size/age distribution; eelgrass coverage, canopy height, and health; macroalgae coverage; and channeled and knobbed whelk density and size distribution. Since the methods used to survey for these parameters during 2019 through 2021 were effectively the same used by MMA from 2006 through 2014, we have used these data (2006-2014, 2019-2022) to evaluate whether such parameters have changed over time. We believe that these data shed light on the status of the bay scallop population over this time range, provides an assessment of the efficacy of bay scallop restoration efforts, and may be valuable in further enhancing the effectiveness of future bay scallop restoration efforts of the NRD. Further, the survey data illuminates long-term trends in eelgrass coverage and health as well as macroalgae coverage, all valuable to understanding and prioritizing restoration efforts. Lastly, the data have allowed us to investigate trends of channeled and knobbed whelk abundances and understand the relative health of their populations over time within Nantucket Harbor.

While human-induced impacts are not absent from coastal ecosystems of Nantucket, the eelgrass meadows of Nantucket are some of the healthiest in the region with a diverse and complex assemblage of species. Nantucket is home to one of the few remaining commercial and recreational bay scallop fisheries, as well as a lucrative whelk fishery. Excessive nutrients driving algal blooms continue to threaten eelgrass and bay scallop populations, while overharvesting likely poses the largest threat to the two whelk species. The overarching goal of this benthic ecological survey in Nantucket Harbor during September of 2019, 2020, 2021, and 2022 has been to re-establish and build upon a long-term survey of the status of several commercially important shellfish species and a foundation species, eelgrass, that make up major components of the benthic ecosystem in Nantucket Harbor. We believe that the data from the benthic ecological survey that we conducted serves to illuminate temporal changes in the abundance and distribution of these species within Nantucket Harbor and is beneficial to improving concerted efforts to restore these ecologically and economically valuable species.

Methods

Origins of the Long-Term Benthic Ecological Survey of Maria Mitchell Association

The benthic ecological survey conducted by the Maria Mitchell Association was started in 2006 within Nantucket Harbor of Nantucket Island, Massachusetts (Figure 1). The original goal of this survey was to understand what habitat characteristics were most suitable for bay scallops (Peter Boyce *pers. comm.*, Valerie Hall *pers. comm.*). The project was initially funded for two years by the United States Environmental Protection Agency (EPA). Personnel from the EPA Environmental Effects Research Laboratory in Narragansett, Rhode Island, developed the survey methodology and performed the original site selection used in the stratified random survey design (Valerie Hall and Peter Boyce *pers. comm.*). MMA also received input from local commercial fishermen in reviewing the selection of randomized sites that were used in the survey. From 2006-2010, the benthic ecological survey included 48 locations that were surveyed every September (Peter Boyce *pers. comm.*). During 2011, 40 sites were surveyed, including 32 of the original sites plus 8 new ones (Peter Boyce *pers. comm.*). The objective of these changes was to better characterize the structure of the bay scallop population within Nantucket Harbor by removing locations where no bay scallops had been observed for 5 consecutive years, and to add locations that were expected to have higher abundances of bay scallops. The elimination of sites was done based on data from the annual benthic ecological survey while the additional sites were chosen based on the knowledge of local commercial bay scallop fishermen (Peter Boyce *pers. comm.*). During September of 2019, 2020, 2021, and 2022 we surveyed the 39 sites that were used during 2014 (Figure 2). Three sites surveyed during 2019, 2020, 2021, and 2022 were excluded from the analyses: two of which had not been continuously sampled during prior years and another whose position had been moved substantially over the preceding 9 years of the survey. In total, 28 sites had been surveyed consistently from 2006 through 2014 as well as 2019 through 2022 (Figure 3). Further, 36 sites had been surveyed consistently from 2011 through 2014 as well as 2019 through 2022 (Figure 5). Comparisons of the ecosystem parameter data over both groups of years were made with the sites that had been surveyed across all of the years within those time spans. Comparisons for eelgrass coverage were slightly different and this is elaborated on below. During 2019, 2020, 2021, and 2022 we used methods that were comparable to those used in the survey conducted by the Maria Mitchell Association from 2006 through 2014. While some minor modifications were made to the methodology used during 2019 through 2022 to increase the time efficiency of the dive survey work, we believe that these changes do not impact the comparability of the data to previous years. These changes are elaborated on below.

Benthic Ecological Survey Data Collection

During 2022, the benthic ecological survey included a total of 39 sites within Nantucket Harbor (Figure 2) that were surveyed over the course of 8 field days, beginning on September 20th, 2021 and ending on September 28th, 2022. During this time period, high winds prevented us from conducting field work on September 23rd The field personnel included Stephen Heck, Kelley Coughlan, Tara Riley, Leah Hill, David Berry, Griffin Harkins, Joseph Minella, Josh Whitehead, and Dr. Valerie Hall. Depending on the day, survey work began between 7:00 AM and 8:00 AM and finished between 4:00 PM and 7:30 PM. The coordinates (latitude and longitude) for all survey locations were loaded onto a handheld Garmin GPSMap 78sc Waterproof Marine GPS unit. At each site, a weight with a line and a buoy attached was dropped on the coordinates of the site (within 2 meters (m) according to the GPS unit) to mark the beginning of one of the transect lines. The boat was anchored in close proximity to the deployed buoy.

Two divers entered the water. The first diver anchored the beginning of a 25-m transect tape directly adjacent to the location of the weight using a piece of ½-inch diameter rebar (with a bent end) that was pushed into the sediment, securing the tape under the bent end. The beginning of a 5-m line was also attached to the rebar. The first diver led their transect tape out in the approximate direction of the current flow or approximately parallel to the shoreline; this was typically one in the same. This direction was determined before divers entered the water. The compass heading of the transect direction was recorded. Two parallel 25-m transect tapes were deployed 5-m apart and anchored on either end using rebar stakes. The transect tapes were oriented such that the first transect tape was on the left-hand side and the second transect tape was on the right-hand side if looking in the direction that the transect tapes were both extended. This differed slightly from previous methods in that we used transect tapes with reels for the transect lines instead of coiled ropes that had been used in previous years (2006-2014) by MMA personnel. We made this switch in an effort to eliminate the possibility that the transect lines would become tangled, which happened occasionally with the previous use of rope.

Once the transect lines were anchored to the seafloor, both divers surfaced and swam back to the beginning of their respective transect tapes. Each diver swam down each transect line, collecting every bay scallop, channeled whelk, and knobbed whelk that they found within ¼ meter on each side of the line (½ meter width in total). This was accomplished by having the divers swim down the transect tape with the tape directly in the middle of their chest and a calibrated awareness (that was routinely checked) as to how far apart a ½ meter was between their hands. For all divers involved, this was approximately shoulder-width. All bay scallops, channeled whelk, and knobbed whelk that were found were placed in a mesh collection bag. To ensure that all bay scallops, channeled whelk, and knobbed whelk were collected along each transect tape, once each diver got to the end of their 25-m transect tape, they swam back in the opposite direction using the same methods to collect any individuals of the same species that may have been overlooked. A separate mesh collection bag was used for all animals collected on the second pass of the transect tape. Each diver also used a fine mesh spat bag to collect any bay scallops that were smaller than the mesh of our normal mesh collection bags. This had been unnecessary during 2019 and 2020, but essential during 2021 and 2022.

Two passes along each transect line (one pass in each direction) was a method established from the inception of the survey in 2006. This was deemed necessary since many of the survey locations contain high densities of eelgrass and macroalgae and it can be difficult to ensure that all bay scallops, channeled whelk, and knobbed whelk were collected in these environments on just one pass, especially when searching for them is sometimes conducted largely by feel. Prior to 2019, MMA personnel had split the 25-m transect line up into five 5-m, sections. Separate bags were used to collect the aforementioned animals in each of these 5 sections. The original purpose was to understand whether fine scale habitat parameters influence the density and distribution of these animals. We decided to eliminate this aspect of the previous methods to reduce the amount of time that each location took to survey. Further, the previous method of dividing the transect line into 5 sections did not use separate bags for the first and second pass along the transect line. Since bay scallops, channeled whelk, and knobbed whelk move, we used separate bags for each pass along each transect line in an attempt to understand whether movement of these animals was artificially elevating the perceived density of these species across the surveyed locations (we found no evidence that it did). Once each diver finished collecting all bay scallops, channeled whelk, and knobbed whelk in both directions along the transect line, all collected animals were returned to the research vessel.

The shell height and growth rings of individual bay scallops were then measured and recorded. Further, the age category of each individual bay scallop was recorded. Shell heights were measured along a tangential distance from the umbo to the opposite, ventral shell margin. Annual growth rings (which consist of a (usually) pronounced shell ridge that is produced when shell growth stops at the onset of winter) were also measured from the umbo to the opposite edge of the defined growth ring. During 2019, 2020, 2021 and 2022 we used three categories. The first category included juvenile bay scallops, or seed, that did not have any growth rings - indicating they had not yet overwintered (0+ age (or year) class). The second category included first year adult bay scallops that had 1 defined growth ring and had clearly overwintered (1+ age (or year) class). The third category included second year adult bay scallops that had two growth rings and had overwintered twice (2+ age (or year) class). Based on decades of previous experience conducting similar surveys for bay scallops in New York and Connecticut, if a minimum of 30 bay scallops in either the seed or adult category were collected in a single mesh collection bag, that was deemed sufficient enough to determine an accurate size and age distribution of seed or adult bay scallops at that individual location (Stephen Tettelbach, *pers. comm.*). Since all bay scallops were measured in any bag where measurements had been initiated; we could avoid any bias associated with selecting scallops for measurements (Stephen Tettelbach, *pers. comm.*). If at least 30 individuals in any of the seed, first year adult, or second year adult categories had been measured, only counts were done of any remaining individuals of the given age category during 2019. During 2020, 2021, and 2022 all bay scallops were measured.

Before 2019, all bay scallops were measured except in locations where exceptionally high numbers had been found. Again, prior to 2019, only the total shell height of bay scallops had been measured, but no growth rings had been measured; however, the age category of each bay scallop was recorded. Prior to 2019, there were a total of 5 age categories used. These included juvenile bay scallops or seed, first year nubs, second year nubs, first year classic adults, and second year classic adults. Juvenile bay scallops or seed were defined in the same manner that was previously mentioned. According to Dr. Peter Boyce and Dr. Valerie Hall, they generally adhered to the guideline that first year nubs (N1) had a single growth ring of less than 10mm from the hinge. This indicated that it was spawned in the fall and overwintered at a small size. First year classic adults (A1) were generally defined as bay scallop with a single growth ring that was equal to or greater than 10mm. This indicates that it was spawned in the spring/summer and overwintered at a relatively larger size. While Dr. Boyce and Dr. Hall generally followed these guidelines, if the bay scallop had what was deemed a “nub appearance”, but a growth ring up to 15mm, it was classified as a nub (Peter Boyce and Valerie Hall, *pers. comm.*). According to Dr. Boyce and Dr. Hall, if the shell height is equal to or greater than the shell length and the bay scallop has a large shell width, and a single growth ring, this generally indicates that the bay scallop was spawned during the previous fall (Peter Boyce and Valerie Hall, *pers. comm.*). While the term shell height was previously defined above, shell length and shell width merit definitions. Shell length is measured on the same plane as the shell height and is perpendicular to the shell height, i.e. the largest measurement between the posterior and anterior margins (Zheng et al. 2011). Shell width is the largest measurement between the left and right valve, or the thickness of the scallop (Zheng et al. 2011). If a bay scallop exhibited these qualities, it was classified as a first year nub (N1) if the growth ring was up to 15mm (Peter Boyce and Valerie Hall, *pers. comm.*). Furthermore, first year classic adults (A1) generally form a “bull-nose” shape at the hinge which makes them easily distinguished from N1 bay scallops (Peter Boyce and Valerie Hall, *pers. comm.*). Second year nubs (N2) were nubs with a second annual growth ring indicating that they had overwintered twice. Second year classic adults (A2) were adults with a second annual growth ring indicating that they had overwintered twice. Before 2019, no growth ring measurements had been recorded. Although we did not actively distinguish between bay scallops that were first year nubs (N1) and first year classic adults (A1) or second year nubs (N2) and second year classic adults (A2) during 2019, 2020, 2021, and 2022, the growth ring measurements that we recorded allowed us to determine the proportion of total bay scallops in these categories and respective densities of these age classes at each survey location. In practice, very few bay scallops had first growth rings that were between 10-15mm. According to Dr. Boyce, a maximum of 1 or 2 bay scallops per site was encountered with a first growth ring (annulus) within this range (Peter Boyce *pers. comm.*). During 2019, 2020, 2021, 2022 we defined bay scallops as nub (N1 or N2) bay scallops if the first growth ring was less than 10mm and classic adult (A1 or A2) bay scallops if the first growth ring was equal to or greater than 10mm. While this change may have presented a degree of discrepancy between the data collected from 2006 through 2014 and from 2019 through 2022, we believe that this presents an insignificant amount of variation in the data collected across years.

The total length of all channeled and knobbed whelk was recorded, as measured from the tip of the spire to the tip of the siphonal canal. All bay scallop and whelk measurements were recorded by survey personnel on the research vessel. After all whelk and bay scallops were measured, they were released at the same location from where they were collected.

Prior to 2019, all crustaceans observed within ¼ meter of each side of the transect line had also been collected and brought back to the research vessel. The species were identified and the carapaces were measured. This aspect of the survey is highly dependent on the visibility of the water and the ability of divers to sight and capture them (Peter Boyce *pers. comm.*). There can be a large amount of variability in water clarity based on weather conditions and location within Nantucket Harbor. Further, the ability of divers to sight and capture crustaceans along the transect line is inherently inconsistent (Peter Boyce *pers. comm.*). According to Boyce, green crabs are particularly wary of divers and often move out of the transect before the diver can capture them (Peter Boyce *pers. comm.*). For these reasons, there is a high amount of variability in this previously collected data since standardizing this method of collection of crustaceans along the transect line is very difficult. Subsequently, we decided to eliminate this aspect of the survey. Instead, during 2019, 2020, 2021, and 2022 divers recorded the presence of crustaceans and fish within ¼ meter on either side of the transect line on dive slates during each pass along the transect. This switch in methodology does not negate the above inconsistencies. However, the switch was made in order to reduce the amount of time required to survey each location.

Furthermore, before 2019, divers collected hard clams within ¼ meter of the transect line by inserting their fingers into the sediment to a depth of approximately 5 centimeters (cm). Given the variability in sediment composition, inconsistency of hard clam collection between divers, and the large amount of time that such collection added to the survey, we decided to eliminate this aspect of the survey during 2019, 2020, 2021, and 2022. We decided to prioritize the other parameters that were surveyed for. This was also done to ensure that we could accomplish the survey in the period of time that we had proposed.

While personnel on the research vessel began measuring all bay scallops, channeled whelk, and knobbed whelk that the two divers had collected along the transect lines, the two divers surveyed for submerged aquatic vegetation parameters. Each diver took a ¼-m^2 quadrat (½ m on each side) made from ½ inch PVC pipe, and at standardized 5-m intervals along each transect line, beginning at the 5-m mark along the transect line, placed one corner against the transect tape at each incremental distance (5, 10, 15, 20, and 25 m) along the transect line with the rest of the quadrat situated towards the end of the transect line. The quadrats were placed on the outside of each transect line (on the opposite side of the transect line from the other transect line). At each of the 5 locations, the quadrat was placed and then flipped away from the quadrat so that the quadrat rested ½ meter away from the transect line. This was done to minimize any disturbance of the submerged aquatic vegetation that may have previously occurred within ¼ meter of the transect lines as divers were collecting bay scallops and whelk. Prior to disturbing anything within the quadrat, depth at each quadrat was recorded and then the total percent cover of submerged aquatic vegetation (including eelgrass and macroalgae) was estimated. In addition, the total percent of bare space was also estimated without disturbing anything within the quadrat. After that, the percent cover of branching algae, filamentous algae, *Codium* algae, sheet algae, and boring sponge was estimated and recorded. The area inside of the quadrat was then gently swept by hand to remove the macroalgae. The percent cover of eelgrass was then estimated as well as the percent of bare space. The percent cover of all of the aforementioned parameters was done according to a method developed by EPA personnel (Peter Boyce, *pers. comm.*). For each of the percent cover parameters, each diver estimated to which of the following numbers the percent cover was closest to: 0, 1, 3, 7, 15, 25, 38, 55, 76, 94, 100. The intention of this method was to reduce the bias of individual divers in the estimation of percent cover of the aforementioned parameters. These methods were consistent across all years that these surveys were conducted, including 2019, 2020, 2021, and 2022. If eelgrass was present, the fouling on the eelgrass blades was estimated on a scale of 0 to 5 (0 = none, 1 = 1 – 25.5%, 2 = 26 – 50%, 3 = 51 – 75%, 4 = 76 – 95%, 5 = 100%). Epiphytes were included only in the fouling estimate, not in the percent cover of macroalgae. Again, if eelgrass was present, the 5 longest shoots within the quadrat were measured from the meristem to the longest blade on the shoot down to the nearest centimeter. If fewer than 5 shoots were present, only the shoots present were measured. The sediment type was categorized as cobble, pebbles, sand, sandy mud, muddy sand, mud, silt, and/or shell hash. (From 2006 through 2008, the bottom composition was estimated in a more quantitative manner by sending samples from each site to the aforementioned EPA facility for analysis.)

While divers were in the water, personnel on the boat recorded environmental parameters at each location. These included estimations of the percent of cloud cover, wind speed and direction, sea state, and Beaufort scale. Beaufort scale approximations were made on a scale ranging from 0 to 5 that incorporates both wind speed and sea conditions. Water depth was measured using a handheld depth sounder. Dissolved oxygen concentration and salinity were measured within 5 cm of the surface of the water column using a YSI ProDSS Multi-Parameter Water Quality Meter. During 2019, the temperature probe on the YSI ProDSS appeared to be reading inaccurately and so the water temperature readings were recorded from the temperature probe attached to the GPS and transducer system of the research vessel. During 2020, 2021, and 2022 the YSI ProDSS temperature probe was functioning properly and was used for all parameters. Prior to 2019, water clarity and pH were also measured in addition to the parameters that were measured during 2019, 2020, 2021, 2022, and all of the parameters were measured throughout the water column. We decided to reduce the measurement of environmental variables within the water column since prior measurement of the parameters changed drastically with the tide, making discrete measurements less useful than the continuous data collected throughout the season by the Town of Nantucket Natural Resources Department. Reducing the extent of the environmental parameter measurements in the water column reduced the time spent at each site - which helped to ensure that we were able to finish the survey in the proposed amount of time.

During 2019, 2020, 2021, and 2022, after divers returned to the research vessel, images of their dive slates were taken using an Olympus Tough TG-6 Waterproof Camera. Dive slates were then transcribed onto waterproof data sheets and then erased for use at the next survey location. During 2020, 2021, and 2022 representative photographs were taken at each site using an Olympus Tough TG-6 Waterproof Camera by one of the divers after all other parameters of the survey were completed.

At the end of every field day of the 2019, 2020, 2021, and 2022 benthic ecological surveys, all field sheets and marked pages of the field notebook were scanned and backed up electronically. All photographed images were also backed up electronically.

Statistical Analyses

Eelgrass Percent Cover, Eelgrass Blade Length, and Macroalgae Percent Cover Over Time

Measurements of eelgrass percent cover and eelgrass blade length within the ten ¼-m^2 quadrats surveyed per site were averaged by site. Sites that had no eelgrass during any year of the 13 years of data were excluded from the analysis of eelgrass percent cover since those sites were deemed too deep to ever contain eelgrass (Figure 4, 6, 9, 11). Further, the inclusion of those sites would artificially mask any changes in eelgrass coverage over time (Figure 4, 6, 9, 11). Eelgrass blade length was only evaluated for sites where actual measurements of eelgrass blades were made, subsequently only sites that contained eelgrass blades were included (Figure 4, 6, 9, 11). Although 28 sites were surveyed consistently from 2006 through 2014 and 2019 through 2022, two of these were excluded since they never contained any eelgrass and were at a depth stratum that made it extremely unlikely there would ever be eelgrass there (Figure 3, 4). Comparisons of average eelgrass percent cover and average eelgrass blade length were made between years using the 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 and had eelgrass at some point during the 13 years of collected data (Figure 4, 6). Comparisons were made between years that included 2011 through 2014 as well as 2019 through 2022 using a higher number of sites. Although 36 sites were surveyed consistently from 2011 through 2014 and 2019 through 2022, the same two sites mentioned above were excluded since no eelgrass was ever observed there and were at a depth that made it improbable that there would ever be eelgrass there (Figure 5, 6). These same types of comparisons, between years for the previously mentioned time ranges, were then made within each of the three areas of Nantucket Harbor (Upper Harbor, Central Harbor, Lower Harbor) (Figure 9, 11). These comparisons were also made for eelgrass percent cover between years including the 12 survey locations that had an average percent cover of eelgrass that was higher than 40% during 2006 (Figure 2). The 12 survey locations were: 08, 09, 34, 35, 20, 38, 05, 14, 19, 24, 26, 02 (Figure 2). This was done to evaluate whether there were difference between years in eelgrass percent cover of locations that had among the highest average values at the start of the survey in 2006. Comparisons were also made a select group of sites within Third Bend and the Head of the Harbor (Upper Harbor) over time.

Macroalgae Percent Cover Over Time

Two categories of macroalgae (branching and filamentous (unbranching)) predominated the macroalgae percent cover data. Percent cover data of branching macroalgae and filamentous macroalgae within the 10 ¼-m^2 quadrats surveyed per site was averaged by site. Comparisons of the average percent cover of both groups of macroalgae were made using the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 3). Comparisons were also made between years for the same parameters using the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 5). These comparisons were also made within three areas of Nantucket Harbor comparing between years for the previously described time ranges (Upper Harbor, Central Harbor, Lower Harbor) (Figure 8, 10).

Bay Scallop Density By Age Class Over Time

Overall densities of bay scallops (number per m^2), for all age classes, were averaged by site. Additionally, densities of seed (juvenile) bay scallops, first year nub bay scallops, first year adult bay scallops, second year nub bay scallops, and second year adult bay scallops were averaged by site. Comparisons were made between years of the densities of different age classes as well as the overall bay scallop density and total adult bay scallop (anything older than seed bay scallops) density. These comparisons were made for densities of the different age categories of bay scallops across the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 as well as between years using the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 3, 5). The age distributions of bay scallops were plotted by year using the sites surveyed consistently during all years of the survey.

In order to understand how bay scallop populations may have been influenced by intensive bay scallop population restoration efforts begun by Tara Riley of the Town of Nantucket Natural Resources Department in 2010, densities of total bay scallops, seed, and total adult bay scallops (anything older than seed bay scallops) were pooled for the five years of data prior to the start of the restoration work from 2006 through 2010 and for the eight years of data after the start of the restoration work from 2011 through 2014 and 2019 through 2022. Only the 28 sites that were surveyed consistently across this time span were included in the analysis (Figure 3). While Tara Riley began her work in 2010, most of that season was focused on getting the Brant Point Shellfish Hatchery up and running and so the number of bay scallop larvae released that year into the Harbor was a small fraction of what was done in subsequent years. Additionally, the release of bay scallop larvae as part of the directed bay scallop restoration effort during 2010 occurred late in the summer, so any seed bay scallops that resulted from this effort would have been too small to observe within the benthic survey that fall. Given that the adult bay scallop population during 2010 would not have been affected by Riley's efforts, 2010 was considered to be before bay scallop restoration efforts had begun. Since the data for total bay scallop density, seed bay scallop density, and total adult bay scallop density before and after the start of the restoration work was not normally distributed, non-parametric Kruskal-Wallis rank sum tests were used to evaluate whether differences that were observed between years before and after restoration efforts had begun were statistically significant.

Channeled and Knobbed Whelk Density Over Time

Channeled and knobbed whelk density were averaged by site within each year of data. Comparisons were made between years using the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 3). Comparisons were also made between years for the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 5).

Results

Eelgrass Percent Cover and Blade Length Over Time

While the average percent cover of eelgrass (*Zostera marina*) varied on an annual basis from 2006 through 2014 and 2019 through 2022 within the 26 sites that were surveyed consistently over this time-period and contained eelgrass at some point over the course of the survey, overall there was a strong harbor-wide decline in the average percent cover (Figure 4, 12). The highest average percent cover of eelgrass was observed during 2006 (during the first year that the survey took place) at 37% (Figure 12). While the average percent cover of eelgrass varied annually, there has been a declining trend over the 13 years of the survey (Figure 12). After a four-year gap in the survey, the data from 2019 appeared to suggest the possibility of eelgrass recovery with an average percent coverage of 29%, up from 18% during 2013 and 25% during 2014 (Figure 12). Unfortunately, the lowest average percent cover of eelgrass was observed during 2020 and 2021 at 17% and was less than half of that observed during 2006 (Figure 12). The average percent cover of eelgrass during 2022 was even lower than 2021 (16%). It is important to note that any differences (increases or declines) in the percent cover of eelgrass or macroalgae are presented as raw differences in the values of percent cover rather than a percent difference in the observed values.

Trends within each of the three different regions of Nantucket Harbor (upper harbor, central harbor, and lower harbor) were evaluated using the same 26 survey locations from 2006 through 2014 and 2019 through 2022 that were confined to each region (Figure 9, 13-15). While interannual differences in the average percent cover of eelgrass varied between these three regions, all three regions experienced declines in the average percent cover of eelgrass between 2006 and 2014 (Figure 13-15). Between 2006 and 2014, the observed average percent coverage of eelgrass declined from 33% to 17% in the upper harbor, 34% to 26% in the central harbor, and 44% to 31% in the lower harbor (Figure 9, 13-15). The upper and lower harbor experienced what appeared to be potential recovery from 2014 to 2019 with an increase in the observed average percent cover of eelgrass from 17% to 37% in the upper region and 31% to 42% in the lower region (Figure 9, 13, 15). Unfortunately, this increase in the average percent cover of eelgrass did not continue during 2020 in these two regions (Figure 13, 15). In fact, during 2020 the second lowest average percent cover of eelgrass was observed in the upper region of Nantucket Harbor (13%) which was less than half of that found in this area during 2006 (33%) (Figure 15). The average percent of eelgrass in the upper region had increased between 2020 and 2021 to 20%, but dropped again during 2022 to 14% (Figure 15). Furthermore, the absolute lowest average percent cover of eelgrass that had been observed in the lower region of the harbor was found during 2022 (25%) which was even lower than what was observed during 2020 (26%) and 2021 (26%) and drastically lower than that observed during 2006 (44%) (Figure 13). Additionally, the central region of the harbor had the lowest observed average eelgrass percent cover during 2022 (9%) which is less than a third of the average coverage observed during 2006 (34%) and 2007 (335%) (Figure 14). In short, all three regions of Nantucket Harbor saw severe declines in average eelgrass coverage between 2006 and 2022 (Figure 9, 13-15).

When the analysis was confined to years ranging from 2011 through 2014 and 2019 through 2022, 34 sites were included that were surveyed consistently across these years and contained eelgrass at some point over the course of the survey (Figure 6, 11). While additional sites could be included in comparisons between these years, the same general trends in the observed average eelgrass percent cover over time were observed both at the harbor-wide level as well as within the three different regions of Nantucket Harbor (Figure 6, 11, 16-19).

The 12 survey locations that had an average eelgrass percent cover of 40% during 2006 experienced a drastic decline in eelgrass coverage over the course of this study (Figure 4, 20). During 2006, these 12 sites had an average percent cover of eelgrass of 56% (Figure 20). However, in 2020, 2021, and 2022 the average percent cover was less than half of what it was during 2006, at 22%, 21%, and 20%, respectively (Figure 20). Looking at the sites within the Head of the Harbor (sites 34, 35, 33, and 38), there was a drastic amount of variation in the average percent cover of eelgrass between years, however, between 2006 and 2022 there was a concerning decline in average eelgrass percent cover from 47% to 12% (Figure 4, 22). Furthermore, the average eelgrass percent cover observed at two sites found within Third Bend (sites 24 and 20) dropped from 53% during 2006 to being virtually absent during 2020 (2%) and 2021 (1%) although increasing slightly to 5% during 2022 (Figure 21).

The average eelgrass blade length (cm) was relatively constant over the years from 2006 through 2014 and 2019 through 2022 within the 28 sites that were surveyed consistently during these years throughout Nantucket Harbor (Figure 3, 23). The average eelgrass blade length (cm) was relatively constant and ranged from 26.6 cm to 35.0 cm, with annual fluctuations but no apparent trends in terms of increases or decreases (Figure 23). The average eelgrass blade length during 2006 was similar to that found during 2022 at 28 cm and 27 cm, respectively (Figure 23). When considering the sites confined to the three regions of Nantucket Harbor (lower harbor, central harbor, and upper harbor), the variation in the average percent cover of eelgrass observed was much larger within each region between years in comparison to the harbor-wide assessment (Figure 24-26). Still, no apparent trends were observed over the time range that the survey has taken place (Figure 24-26). When the analysis for average eelgrass blade length between years was expanded to 36 sites that were surveyed consistently within Nantucket Harbor from 2011 through 2014 and 2019 through 2022, a similar absence of any trend was observed and the same relative differences between years were observed (Figure 27). This is also true for the three regions of Nantucket Harbor (Figure 28-30).

Macroalgae Percent Cover Over Time

There were notable differences between years in the average percent cover of branching macroalgae from 2006 through 2014 and 2019 through 2022 within the 28 sites that had been surveyed annually over this time span throughout Nantucket Harbor (Figure 3). The peak average branching macroalgae percent cover occurred in 2007 at 37% followed by 2011 with 34% (Figure 31). The average percent cover of branching macroalgae during 2022 (7%) was the lowest in all thirteen years of data (Figure 31). Furthermore, there has been a drop in the average percent cover of branching macroalgae during each year since 2019 (Figure 31).

When the average percent cover of branching macroalgae was evaluated for sites confined to the three regions of the harbor (upper harbor, central harbor, and lower harbor), a variety of trends were observed (Figure 3, 32-34). In 2022, the upper region of the harbor had the lowest average percent cover of branching macroalgae observed during the years that the survey took place at 4% (Figure 8, 34). This was less than a seventh of the highest average percent cover of branching macroalgae found during 2011 at 30% (Figure 8, 34). During 2022, the central region of the harbor had the lowest average percent cover of branching macroalgae observed over the course of the survey at 3% (Figure 8, 33). This was dramatically lower than the highest average percent cover of branching macroalgae found at the peak during 2007 at 51% (Figure 8, 33). The lower region of the harbor had the third lowest average percent cover of branching macroalgae during 2022 (15%) (Figure 8, 32). The data from 36 locations surveyed annually during 2011 through 2014 and 2019 through 2022 showed that the average percent cover of branching macroalgae had the same trends throughout Nantucket Harbor as were observed within the 28 sites surveyed across 2006 through 2014 and 2019 through 2022 (Figure 5, 35). While the values differed slightly, similar relative differences between years in the average percent cover of branching macroalgae from 2006 through 2014 and 2019 through 2022 within the three regions of the harbor were observed when evaluating data from the 36 sites surveyed from 2011 through 2014 and 2019 through 2022 (Figure 10, 36-38).

The other category of macroalgae that dominated the composition of macroalgae percent cover in many years was the filamentous macroalgae group. During all years prior to 2021 that it was encountered, it is estimated that over 95% of the filamentous macroalgae that was encountered was a blue-green macroalgae (cyanobacteria) that had previously been identified as *Lyngbya* sp. but has since been corrected to being most closely related to *Hydrocoleum lyngbyaceum* (Peter Boyce *pers. comm.*, (Moisander 2015). However, during 2021, the filamentous macroalgae group included a high proportion of *Ectocarpus siliculosus* as well. When considering the 28 sites surveyed consistently from 2006 through 2014 and 2019 through 2022, the average percent cover of filamentous macroalgae increased from 2006 through 2012, peaked in 2012 with an average percent cover of 42% harbor-wide, and declined through 2019 (Figure 3, 39). The average percent cover of filamentous macroalgae was 2% in the harbor during 2019 which was the lowest observed apart from 2006 and 2007 (Figure 39). The average percent cover of filamentous macroalgae was still relatively low during 2020 in comparison to peak years, but at an average percent cover of 6%, it was three times the amount observed during 2019 (Figure 39). During 2021, the observed average percent cover of filamentous macroalgae throughout the harbor rose to 24% (Figure 39). However, the average percent cover of filamentous macroalgae dropped to 7% during 2022 (Figure 39). It is important to note that the heightened coverage of filamentous macroalgae during 2021 appears to have been driven by both an increase in *Hydrocoleum* sp. as well as *Ectocarpus siliculosus.* All three regions of Nantucket Harbor had increases in the percent cover of filamentous macroalgae from 2020 and 2021, some more than others (Figure 40-42). Between 2020 and 2021, the average percent cover of filamentous macroalgae in the lower region increased from 9% to 33%, in the central region it went from 8% to 24%, and in the upper region it went from 1% to 15% (Figure 40-42). During 2022, all three regions had strong declines in the average percent cover of filamentous macroalgae with 15% in the lower region, 4% in the central region, and 1% in the upper region (Figure 40-42). Similar trends were observed in the 36 sites surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 36, 43-46).

Bay Scallop Density Over Time

The average density of total bay scallops (all age classes pooled together) varied substantially across years from 2006 through 2014 and 2019 through 2022 within the 28 sites that were surveyed consistently through this range of time (Figure 3, 47). Within these sites and years, there was an overall average density of 2.3 total bay scallops per m^2 and a maximum average density of 4.8 total bay scallops per m^2 found in a single year during 2011 (Figure 47). The maximum density of total bay scallops found at a single survey site was 44.4 per m^2 at site 35 during 2011 (Figure 3, 47). The highest average densities of total bay scallops were observed during 2011, 2012, 2019, 2020, and 2022 with 4.8, 4.1, 3.7, 3.9, and 4.2 bay scallops per m^2, respectively (Figure 47). During all five of these years, total bay scallop density was a minimum of 134% higher than any other year (Figure 47). All five of these years were after Tara Riley initiated her intensive bay scallop population restoration efforts. With that said, the density of total bay scallops observed during 2021 was on the lower end of the spectrum of densities observed throughout the history of the survey at 1.2 total bay scallops per m^2, which is similar to densities seen prior to the beginning of restoration efforts (Figure 47).

When the analysis was confined solely to seed (juvenile) bay scallops, peak densities of seed bay scallops were also observed during 2011, 2012, 2019, 2020, and 2022 with 3.7, 2.1, 2.2, 3.1 and 3.5 seed bay scallops per m^2 (Figure 3, 48). During these five years, seed bay scallop density was a minimum of 143% higher than in any of the other years within this time range (Figure 48). Again, all of these years were after Tara Riley began her focused bay scallop population restoration work. The density of seed bay scallops found during 2021 was within the lower half of the range of values observed during the 12 years of data at 0.6 seed bay scallops per m^2 (Figure 48). First year nub (N1) bay scallop densities were highest in 2011 followed by 2019 with an average of 0.9 and 0.7 N1 bay scallops per m^2, respectively (Figure 49). First year classic (A1) bay scallop density was highest during 2012, followed by 2007, 2009, and 2019, in that order (Figure 50). There was an average density of 1.3, 1.1, 1.0, and 0.7 A1 bay scallops per m^2, respectively (Figure 50). During some years, A1 bay scallops made up the majority of the adult bay scallops observed whereas in other years N1 bay scallops were the dominant age class of adult bay scallops observed (Figure 49-50, 62-73). Year 2 nub bay scallops (N2) composed a small minority of the observed bay scallops collected over the course of the surveys, although there were peaks in N2 bay scallop density during 2009 and 2012 (Figure 51, 62-72). Second year classic adult bay scallops (A2) were found at very low abundances but were included in total adult bay scallop densities and total adult bay scallop densities (Figure 47, 53). The peak number of total adult bay scallops was observed during 2012, following by 2019, then 2007, 2011, and 2009, in that order (Figure 52). While the density of total adult bay scallops fluctuated substantially between years over the course of the study between 2006 through 2022, the average density of total adult bay scallops observed during 2022 was on the lower end of the spectrum of observed values (Figure 52).

When the evaluations were confined to the 36 sites surveyed during each September from 2011 through 2014 and 2019 through 2022, the relative densities observed during those years for total bay scallops, seed bay scallops, N1 bay scallops, A1 bay scallops, N2 bay scallops, and total adult bay scallops follow similar relationships that were observed using the data from 28 sites surveyed from 2006 through 2014 and 2019 through 2022 (Figure 5, 53-58).

The most notable difference in bay scallop density was observed when comparing data from the five years prior to the commencement of Tara Riley's intensive bay scallop population restoration efforts in Nantucket Harbor (2006 through 2010) to the eight years of data that exist after these restoration efforts began (2011 through 2014 and 2019 through 2022). The 28 locations that were surveyed consistently during all years from 2006 through 2014 and 2019 through 2022 were used in these comparisons (Figure 3). The average density of total bay scallops (all age classes) was 132% higher during the eight years after the bay scallop population restoration work began than during the five years before (Figure 59). On average, during the five years prior to the commencement of bay scallop restoration efforts, there was an average of 1.3 total bay scallops per m^2 (Figure 59). During the eight years after the start of bay scallop restoration efforts, there was an average of 3.0 total bay scallops per m^2 (Figure 59). A Kruskal-Wallis rank sum test revealed that this difference was statistically significant ($p < 0.0001$) (Figure 59). Furthermore, there was a 382% increase in the average density of seed (juvenile) bay scallops during the eight years after restoration work began compared to the five years before (Figure 60). A Kruskal-Wallis rank sum test revealed that this difference was statistically significant ($p < 0.0001$) (Figure 60). Prior to the beginning of bay scallop population restoration efforts, there was an average of 0.4 seed bay scallops per m^2 whereas after there was an average of 2.0 seed bay scallops per m^2 (Figure 60). Moreover, when all adult bay scallop age classes were pooled, there was a 16% increase in the average density of adult bay scallops during the seven years after bay scallop restoration work had started when compared to the previous five years prior to restoration work (Figure 61). A Kruskal-Wallis rank sum test revealed that this difference was statistically significant ($p = 0.0498$). Prior to restoration work, there was an average of 0.9 adult bay scallops per m^2 versus 1.0 adult bay scallops per m^2 after restoration work had begun (Figure 61).

Channeled and Knobbed Whelk Density Over Time

The average density of channeled whelk and knobbed whelk varied markedly between years from 2006 through 2014 and 2019 through 2022 within the 28 sites that were surveyed annually during the time span that the survey was conducted (Figure 3, 75). In general, there appears to be a pattern of decline for both species of whelk from 2006 through 2019, with a continued decline in knobbed whelk through 2021 (Figure 75). However, there is an uptick in the relative density of channeled whelk in 2020, 2021, and 2022 in comparison to 2019 (Figure 75). Between 2021 and 2022, the average density of channeled whelk declined whereas the density of knobbed whelk increased. Knobbed whelk had higher densities than channeled whelk during 8 out of the 13 years of data (Figure 75). During 2022, the average density of channeled whelk was within the lower half of values recorded whereas the average density of knobbed whelk was within the higher half (Figure 75). When expanding the analysis to the 36 sites that were surveyed each year from 2011 through 2014 and 2019 through 2022, the average density of both channeled whelk and knobbed whelk during 2022 was within the top half of the values observed within this time period (Figure 76).

Discussion

The considerable 382% increase in the average number of seed (juvenile) bay scallops per m^2 and 16% increase in the average total adult bay scallops per m^2 between the five years of data before (2006-2010) and the eight years of data after the bay scallop population restoration work began (2011-2014 & 2019-2022) provides strong supporting evidence for the success of the late stage larval release bay scallop restoration strategy devised by Tara Riley of the Town of Nantucket Natural Resources Department. Bay scallop spawning and recruitment success is influenced by complex interactions between many environmental factors including the density of reproductively mature bay scallops during spawning events (Peterson et al. 1996, Tettelbach et al. 2011); water quality parameters such as temperature, dissolved oxygen, and acidity (Talmage and Gobler 2009, Gobler et al. 2017); harmful algal blooms (Summerson and Peterson 1990, Talmage and Gobler 2012); predation post-settlement (Carroll et al. 2015); and suitable settlement habitat (Bologna and Heck 1999, Carroll et al. 2012), among others. While these parameters may have changed over time, the most consistent difference between the five years of data before (2006-2010) and the eight years of data after (2011-2014 and 2019-2022) directed bay scallop population restoration efforts began in the waters of Nantucket, Massachusetts, appears to be the restoration work itself.

There were a substantial overall increase in the average density of seed bay scallops during the years of data from 2011 through 2014 and 2019 through 2022 in comparison to 2006 through 2010, whereas the density of seed bay scallops during 2013, 2014, and 2021 was much lower and close to the levels encountered prior to 2011. The lower abundance of seed bay scallops during 2013, 2014, and 2021 in comparison to the other years following the initiation of restoration work may be reflective of low survival of juvenile bay scallops due to blooms of *Cochlodinium polykrikoides* (recently changed to *Margalefidinium*) *polykrikoides* (Griffith et al. 2019). *Cochlodinium polykrikoides* is a species of dinoflagellate that causes harmful algal blooms and has been shown to induce high levels of mortality in juvenile bay scallops in the laboratory and in the field (Tang and Gobler 2009, Griffith et al. 2019). While blooms of this species did occur during 2013 and 2014, the exact timing and concentrations would be essential pieces of data to understanding whether this was a driving factor in the lower abundance of seed bay scallops during 2013 and 2014. However, it is currently unclear whether these data exist. There was no apparent bloom of *Cochlodinium* during 2021, but there was one of the lowest observed densities of seed bay scallops. While the continued declining trend of eelgrass is likely negatively impacting bay scallop recruitment and survival, bay scallops are also notorious for population fluctuations from year to year, which has been largely attributed to the short life history of the species. In short, reduced seed abundance can be influenced by a slew of many environmental factors including habitat degradation. With the highest densities of seed observed during 2011, 2012, 2019, 2020, and 2022, it appears that restoration efforts of the NRD are having a positive effect on seed abundance despite declining levels of eelgrass and periodic harmful algae blooms.

While first year nub (N1) and first year classic adult bay scallops (A1) were encountered at similar average densities over the 13 years of data, second year nub bay scallops were encountered at approximately an order of magnitude less in terms of density. Depending on the year, first year nubs or first year classic adults can predominate the adult bay scallops collected in the survey while in some years the relative abundance of the two age classes is close to equal. Interestingly, the observed density of first year nub bay scallops was higher than first year classic adult bay scallops every year that the survey took place from 2014 through 2022. Since fall-spawned bay scallops are incredibly small during September, sometimes only a few millimeters in shell height or less, it is very difficult to reliably find seed bay scallops at this time of year that ultimately become nubs (Hall et al. 2015). Subsequently, understanding whether the differences in abundance between years for first year nub bay scallops are driven by the intensity of fall spawning events or post-settlement survival is a challenging question to answer. However, the 382% increase in the average density of seed bay scallops after restoration work had begun, compared with only a 16% increase in the average density of all adult bay scallops (including first year nubs and all other age classes of adults), suggests two things. First, the work of Tara Riley and her NRD team (including Leah Hill, Joseph Minella, David Berry, and Griffin Harkins) has been incredibly successful at increasing the abundance of seed bay scallops. Second, that there is a high degree of mortality between the juvenile (seed) and adult life stages. These patterns of relative increases in juvenile and adult populations are similar to those seen in New York following successful restoration efforts (Tettelbach et al. 2015). Understanding the relative impacts of different sources of mortality could be highly beneficial to enhancing the number of bay scallops that ultimately survive to reproductive and harvest size.

Prior to 2021, the filamentous macroalgae category of this benthic ecological survey was estimated to be composed of more than 95% of a filamentous blue-green macroalgae (cyanobacteria), most closely related to *Hydrocoleum lyngbyaceum* (previously identified as *Lyngbya* sp.). The data demonstrated an increasing abundance of filamentous macroalgae from 2006 through 2012, with a peak in 2012 with an average percent cover of 42% harbor-wide, and a gradual decline thereafter (Boyce unpublished data, (Moisander 2015). Since 2007, the lowest average percent cover of this filamentous macroalgae was observed during 2019 at 2%. While the average percent cover of this filamentous macroalgae was still comparatively low during 2020 in relation to the coverage during peak years, it was still approximately three times the amount observed during 2019 at 6%. During 2021, another species of filamentous macroalgae, *Ectocarpus siliculosus*, was observed at increased densities in comparison to previous years. The overall average percent cover of filamentous macroalgae increased to 24% during 2021. This increase was driven by both an increase in the coverage of *Hydrocoleum lyngbyaceum* as well as *Ectocarpus siliculosus*. However, during 2022, the average percent cover of filamentous macroalgae dropped to 7%. Heightened coverage of filamentous macroalgae is concerning since it can restrict light availability to eelgrass, preventing the eelgrass from photosynthesizing and ultimately leading to eelgrass die-offs. However, the increase in cover of *Hydrocoleum lyngbyaceum*, specifically, is concerning for several reasons. In other regions of the world, the toxins produced by *Hydrocoleum lyngbyaceum* have been found in bivalve tissues (Mejean et al. 2010). Such toxins could potentially have direct, adverse consequences for larval, newly settled juvenile, and adult bay scallops and could have been a driving factor in the overall reduction of bay scallop abundance during 2013, 2014, and 2021. Understanding the direct effects of this filamentous blue-green algae on bay scallops is of paramount importance to evaluating its effects on the bay scallop population. In addition to its toxicity, *Hydrocoleum* is a mat-forming cyanobacterium (Mejean et al. 2010). During 2012, dense mats of *Hydrocoleum* that were more than 0.5 meters thick were observed covering eelgrass beds in several areas within Nantucket Harbor (Peter Boyce *pers. comm.,* Stephen Heck *pers. comm.*). Further, measurements of photosynthetically active radiation (PAR) during October of 2012 showed that under some the thickest *Hydrocoleum* mats, more than 99% of light was blocked (Peter Boyce *pers. comm.,* Stephen Heck *pers. comm.*). Blooms of macroalgae have been implicated in the loss of eelgrass in temperate estuaries due to the restriction of light availability to eelgrass (Hauxwell et al. 2001, Hauxwell et al. 2003). The rise in filamentous macroalgae between 2019 and 2021 is alarming and should be monitored closely, despite a decrease during 2022. Furthermore, an increase of *Hydrocoleum* may be directly detrimental to bay scallop abundance if the presence of *Hydrocoleum* directly causes bay scallop mortality through toxicity.

Overall, the average percent cover of filamentous macroalgae throughout Nantucket Harbor decreased dramatically from 42% at its peak in 2012 to 7% during 2022. The highest average percent cover of branching macroalgae occurred during 2007 (37%) followed by 2011 (34%). Since 2019 there has been an incremental decline in the average percent cover of branching macroalgae from 18% during 2019 to 7% during 2022. The lowest percent cover of branching macroalgae observed over the course of the 13 years of survey occurred during 2022. When focusing on the lower region of the harbor, there is no clear trend over the history of the survey, however there has been an annual decreased in the average percent cover of branching macroalgae from 2019 (36%) to 2022 (15%). During 2022, the central and upper areas of the harbor had the lowest average percent cover of branching macroalgae observed over the 13 years of the survey. The relatively low percent cover of branching macroalgae observed in the survey during 2022 may be reflective of reductions in nutrient inputs to the harbor. To further evaluate this, nutrient data over this time-period would be essential. Excessive nutrient inputs into estuaries fuel both macroalgal and phytoplankton blooms that can ultimately reduce eelgrass shoot density and lead to vast eelgrass die-offs (Hauxwell et al. 2001, Nelson and Lee 2001, Hauxwell et al. 2003). Water temperature may also play a role in these dynamics, but this merits further research (Valiela et al. 1997).

When considering the 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 within Nantucket Harbor that were included in eelgrass percent cover comparisons, a severe decline in the average percent cover of eelgrass was observed between 2006 and 2022 from 37% to 16%. Based on the data from 2014 and 2019, it had appeared that eelgrass was beginning to recover. However, the average percent cover of eelgrass observed during 2022 was less than half of that observed during 2006 and the lowest recorded to date since the start of the survey. Heightened macroalgae coverage does not appear to be consistently paired with the reductions in eelgrass throughout the harbor but it can certainly be a factor heavily influencing eelgrass meadow health and distribution (Hauxwell et al. 2001). However, macroalgae coverage is not the only factor controlling light availability to eelgrass. Eutrophication can drive increases in phytoplankton abundance in the water column and subsequently reduce light availability to eelgrass (Wall et al. 2008). It is suspected that the overall decline in eelgrass coverage during this time span is likely driven by a combination of decreased light availability due to blooms of phytoplankton and macroalgae, driven by nutrient loading. Further, it appears likely that the drastic drop in average percent cover during 2013 may have been driven by particularly excessive amounts of both filamentous and branching macroalgae during 2012 and 2013 as well as blooms of phytoplankton. In 2019, we had suspected that the increase in eelgrass percent cover during 2014 and 2019 (relative to 2013) may have been an indicator that the efforts to curb fertilizer use in Nantucket, that were instituted at around this time (2013), could have been effective at reducing the amount of nutrients making it into the harbor, subsequently reducing blooms of phytoplankton and macroalgae and allowing eelgrass to recover (Peter Boyce *pers. comm.*, Jeff Carlson *pers. comm.*). While the average percent cover of filamentous macroalgae had drastically declined by 2019, branching macroalgae coverage did not show this clear decreasing trend. This may suggest that mats of the filamentous macroalgae, *Hydrocoleum*, have a greater negative effect than branching macroalgae or that blooms of phytoplankton may be a more prominent factor in determining eelgrass health in this ecosystem. Despite a relative increase in the average percent cover of eelgrass within the harbor between 2014 and 2019, 2020 through 2022 had the lowest observed eelgrass coverage since the beginning of the survey, with the absolute lowest during 2022.

While nutrient concentration data for the harbor would be helpful to elucidate these ongoing dynamics, we suspect that the Covid-19 pandemic may have indirectly prompted an increase in certain anthropogenic impacts in Nantucket Harbor. First, it appeared that there was an influx of seasonal residents to the island much earlier than normal in response to the pandemic during 2020 and this appears to have continued during 2021 and 2022. Such an increase in the population of the island earlier in the spring may have caused an increase in the amount of nutrient loading in the harbor from septic systems. This may have driven phytoplankton blooms or possibly contributed to the relative increase of filamentous algae during 2021 and 2020 when compared to 2019. There also appeared to be an increased amount of boat traffic within the harbor during the summer of 2020 and 2021 (Tara Riley, anecdotal observation). A higher amount of boat traffic within the harbor could have caused an increase in the turbidity of the harbor water. While a larger human population present on the island earlier in the spring and increased boat traffic during the summer months is based on anecdotal observations, if present, these two factors could certainly have contributed indirectly to a loss of eelgrass during 2020 and a lack of recovery during 2021 and 2022. More data is needed to evaluate whether these could have been factors influencing eelgrass declines over the past two summers.

If the three regions of the harbor (Upper Harbor, Central Harbor, and Lower Harbor) are evaluated individually, the average percent cover of eelgrass declined in all three regions between 2006 and 2013/2014. In the lower region of the harbor, eelgrass declined from 44% during 2006 to 31% during 2014. In the central region of the harbor, eelgrass declined from 34% to 13% during 2013 and 26% during 2014. In the upper region of the harbor, eelgrass coverage declined from 33% during 2006 to 17% during 2014. While these declines between 2006 through 2014 in each of the three regions are reflective of the overall declines in eelgrass coverage, one of the most drastic declines in eelgrass percent cover within the upper region of Nantucket Harbor occurred between 2006 and 2007, dropping by nearly half. While there are many factors influencing eelgrass, a strong Nor'easter storm occurred during the winter of 2007, which is suspected to have caused shoaling at several sites in the upper region of the harbor (Peter Boyce *pers. comm.*). This may have buried eelgrass in some survey locations and contributed to the initial eelgrass declines in this area. The declines that occurred across the board in all three regions of Nantucket Harbor are certainly suspected to be caused primarily by restricted light availability driven by phytoplankton and macroalgae blooms, but there are many factors that need to be considered (Hauxwell et al. 2001, Wall et al. 2008).

Within the lower and upper harbor, we observed what appeared to be a possible degree of recovery between 2014 and 2019 in the average eelgrass percent cover. Between 2014 and 2019, the average eelgrass percent cover went from 31% to 42% in the lower region and 17% to 37% in the upper region. The increases in these two regions between 2014 and 2019 appear to be what drove the overall increase between these years harbor-wide. In the lower region of the harbor, the recovery of eelgrass coverage between 2014 and 2019 is hypothesized to be a result of a reduction in nutrient loading from the island's efforts to curb fertilizer use, resulting in a reduction in phytoplankton and macroalgae blooms, and associated enhanced water clarity. However, the one of highest observed percent cover of branching macroalgae occurred in the lower region of the harbor during 2019. This could suggest that improved water clarity (due to a reduction in blooms of phytoplankton) was the driving factor for allowing for eelgrass recovery in this region during this time period. If enhanced water clarity was driving this improvement in eelgrass coverage in the lower region of the harbor, this may also have been driven in part by heightened water exchange with Nantucket Sound as a result of the reconstruction of the jetties along the entrance to the harbor in recent years. This could have contributed to improved water clarity and allowed for increased eelgrass percent cover. Furthermore, solely increased flow may have had a positive impact on eelgrass health. As previously mentioned, there was a drastic increase in the average percent cover of eelgrass from 2014 to 2019 in the upper region of the harbor. While this could have been a result of a reduction in nutrient loading resulting from efforts to mitigate nutrient inputs into the harbor, the upper region of the harbor has also seen an increase in not only the number of oyster farms in recent years but also the overall number of oysters being grown by the farms. Since bivalves (including oysters) can increase light penetration through the water column by filter-feeding on phytoplankton, the growing number of oysters being farmed in the upper region of the harbor could be contributing to increased water clarity and, subsequently, recovery of eelgrass percent cover (Wall et al. 2008). Unfortunately, the increases in the average percent cover of eelgrass between 2014 and 2019 in the upper and lower regions of Nantucket Harbor were short-lived. Between 2019 and 2020, both regions experienced drastic reductions in the average percent cover of eelgrass that continued through 2021 and 2022. In the upper region of Nantucket Harbor, 2020 represented the second lowest average percent cover of eelgrass that had been observed since the beginning of the survey at 13%, which was less than half of what was found during 2006 (33%). While the average percent cover of eelgrass in the upper region increased to 20% during 2021, it dropped down to 14% during 2022. In the lower region of the harbor, the three lowest average percent cover values of eelgrass were observed during 2020, 2021, and 2022 at 26% and 26%, and 25%, respectively, which was substantially lower than what was observed during 2006 at 44%. While these two regions certainly drove the increase in the average percent cover observed during 2019 when averaging the data for the entire harbor, the status of eelgrass appeared to reverse course in both regions in a strong way during 2020, with continued declines in the lower region during 2021 through 2022. As previously discussed, we suspect that circumstances surrounding the Covid-19 pandemic may have contributed indirectly to these strong reductions in eelgrass coverage.

Within the central region of the harbor, the average percent cover of eelgrass during 2019 (15%) was less than half of what it was during 2006 (34%). In the central region of Nantucket Harbor, eelgrass coverage had declined between 2014 and 2019 from 26% to 15%. Additionally, the central region saw continued declines into 2020 with the average eelgrass coverage observed at 11%, was less than a third of that observed during 2006. The observed coverage dropped further to 9% in 2021 and remained there during 2022. In the absence of high levels of both filamentous and branching macroalgae cover during 2019 and 2020, it appears that other factors in the central region of the harbor may be driving this continued decline. Filamentous and branching macroalgae coverage increased between 2020 and 2021 in the central region which certainly work against the recovery of eelgrass beds but both dropped during 2022. While the most straightforward reason may be blooms of phytoplankton that may be restricting light availability to eelgrass, other factors may also be at play, including herbicide runoff, disease, and increased turbidity from boat traffic and wind events. With that said, these are speculations and more research would need to be done to determine the drivers of these changes in a definitive way.

When the analysis evaluating eelgrass coverage between years was confined to 12 survey locations that had an average eelgrass percent cover of over 40% during 2006, a gradual but steady decline in the average percent cover of eelgrass was observed between 2006 through 2014 and 2019 through 2022. By 2022, the average percent cover of eelgrass at these sites had dropped by more than half from 56% to 20% between 2006 and 2022, respectively. This steady rate of decline resulted in a considerable loss of eelgrass cover at high eelgrass coverage survey locations. Given that higher habitat complexities of eelgrass have a range of benefits, including the facilitation of reduced predation rates of juvenile bay scallops, the serious decline in these high eelgrass coverage sites is very alarming (Carroll et al. 2015).

Certain areas of the harbor have experienced particularly severe declines. Two sites within Third Bend (sites 24 and 20) experienced an average eelgrass percent cover drop from 53% during 2006 to being virtually absent at 2% during 2021. There was a slight increase in eelgrass coverage during 2022 but eelgrass coverage remained minimal at 5%. Furthermore, when looking at four sites in the head of the harbor (sites 34, 35, 33, 38), there was a drastic amount of variation in the average percent cover of eelgrass between years, but overall, there was a drastic decline from 2006 to 2022, decreasing from 47% to 12%, respectively. These severe reductions in eelgrass coverage are immensely concerning given the vast ecological value that eelgrass has in the ecosystem (Hughes et al. 2002, Carroll et al. 2015).

The average eelgrass blade length was relatively constant from 2006 through 2014 and 2019 through 2022. This was true for all three regions of the harbor. As with percent cover, increased blade length is typically associated with increased light penetration. However, the analysis for blade length was confined to quadrats that contained eelgrass, so no zeros were included. This indicates that where eelgrass still exists, blade length has remained relatively constant.

The complex biogenic habitat of eelgrass has been shown to provide a refuge from predation for many animals (Orth et al. 1984). Juvenile bay scallops attach into the eelgrass canopy to reduce their risk of being consumed (Ambrose and Irlandi 1992). Increased eelgrass shoot density has been shown to increase the survival rate of juvenile bay scallops (Carroll et al. 2015). Further, the higher bay scallops are attached in the eelgrass canopy, the higher their survival since it is more difficult for predators to find and consume them (Ambrose and Irlandi 1992). However, although their survival rates are elevated higher in the canopy, their feeding and growth rates are lower (Ambrose and Irlandi 1992). The attachment height of juvenile bay scallops varies between embayments and appears to be driven by a combination of the predator assemblage as well as food levels and water flow (Garcia-Esquivel and Bricelj 1993). Subsequently a change in eelgrass blade length may not necessarily be ecologically significant for them in this way since there is a trade-off between growth and survival. However, eelgrass stabilizes the sediment, buffers wave action, and can reduce turbidity (Orth et al. 2006, Carr et al. 2010). During high wind events, the stranding of bay scallops on the beach may serve as a potentially high source of mortality, especially in Nantucket where this is known to happen multiple times during some years (NSMP 2012) (NSMP 2012). Presumably, eelgrass of a higher percent cover and longer blade length attenuates wave energy to a greater degree. Subsequently higher density eelgrass with longer blades may reduce the tendency for bay scallops to strand on beaches during high wind events.

In regards to whelk, the number of commercial fishermen allowed to target channeled and knobbed whelk within Nantucket Harbor is very limited. However both species generally declined in density between 2006 and 2019. These decreases seem to be generally reflective of the regional declines in both species that have been particularly exacerbated due to drastic increases in their market price within the past 15 years, their slow rate of growth, and late age of maturity (Kaplan 1998, Edmundson 2016, Nelson et al. 2018). Knobbed whelk continued to experience a decline in density through 2021, however knobbed whelk are not generally targeted by the pot fishery for whelk. During 2022, there was an increase in knobbed whelk density compared to 2021. Interestingly, there was an increase in the relative density of channeled whelk in 2020, 2021, and 2022 in comparison to 2019. It is important to note that there was little, if any, commercial whelk harvesting activity between 2019 and 2021within Nantucket Harbor since given that there were no active commercial permits during that time (Tara Riley personal communication). Since the commercial pot fishery generally harvests channeled whelk, this may be a driving factor in this shift. Given that post-release mortality is remarkably low, if there is any at all, any undersized whelk should be able to be released unharmed. Subsequently, management measures to ensure the long-term sustainability of this lucrative fishery should be made.

In summary, monitoring the annual spatial distribution, population structure, and abundance of bay scallops in Nantucket Harbor is essential to evaluating and enhancing the efficacy of ongoing bay scallop population restoration efforts. Annual surveys for bay scallops are particularly important due to their short life spans which can drive drastic fluctuations in their populations in a short time span. Surveying for eelgrass coverage and health as well as macroalgae can allow for an assessment of the long-term effectiveness to reduce eutrophication of Nantucket Harbor to improve the health of the ecosystem. Mass mortalities of adult bay scallops were seen in Peconic Bay in eastern Long Island, NY, during 2019, 2020, 2021, and 2022 where respective estimated mortality rates between spring and fall were 90%-99% (Stephen Tettelbach, *pers. comm.*). Monitoring for environmental factors in association with bay scallop and eelgrass populations is critically important to understanding what is driving declines in these ecologically and economically valuable species and how best to respond. Understanding long-term trends in channeled and knobbed whelk abundance within Nantucket Harbor allows for an evaluation of whether any changes in management measures have had any effect or should be re-examined. The benthic ecological surveys conducted during 2019, 2020, 2021, and 2022 were able build on an incredibly comprehensive long-term dataset founded by the Maria Mitchell Association and most recently led by Dr. Peter Boyce.

Figures

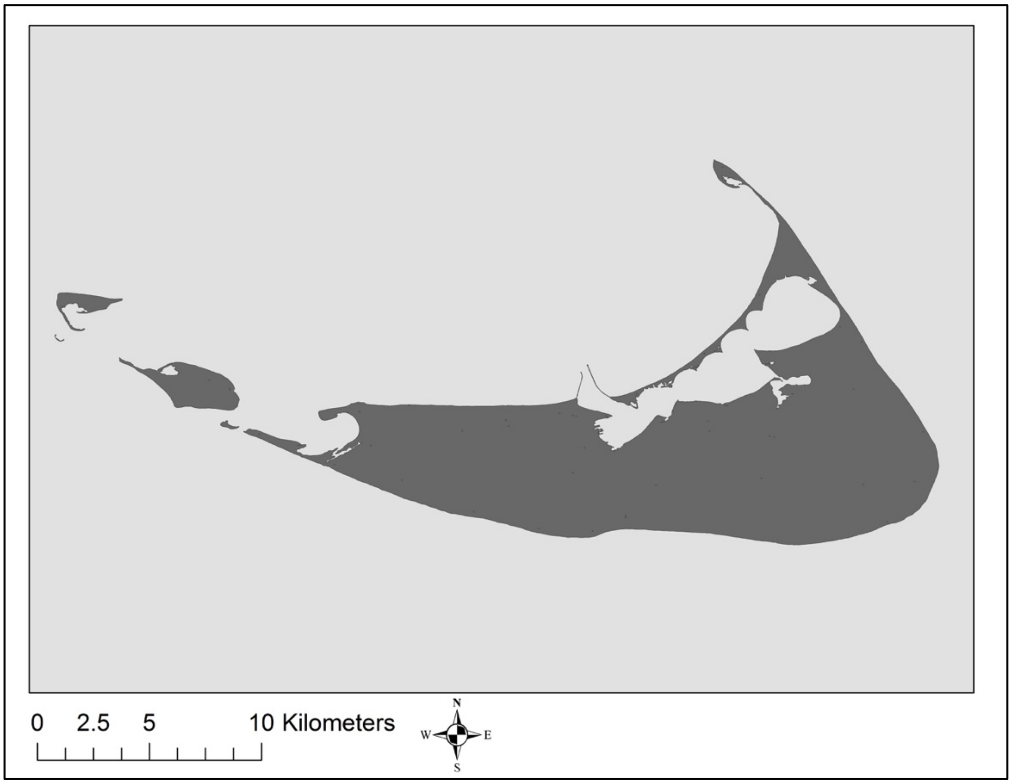

Figure 1. A map of Nantucket Island, Massachusetts.

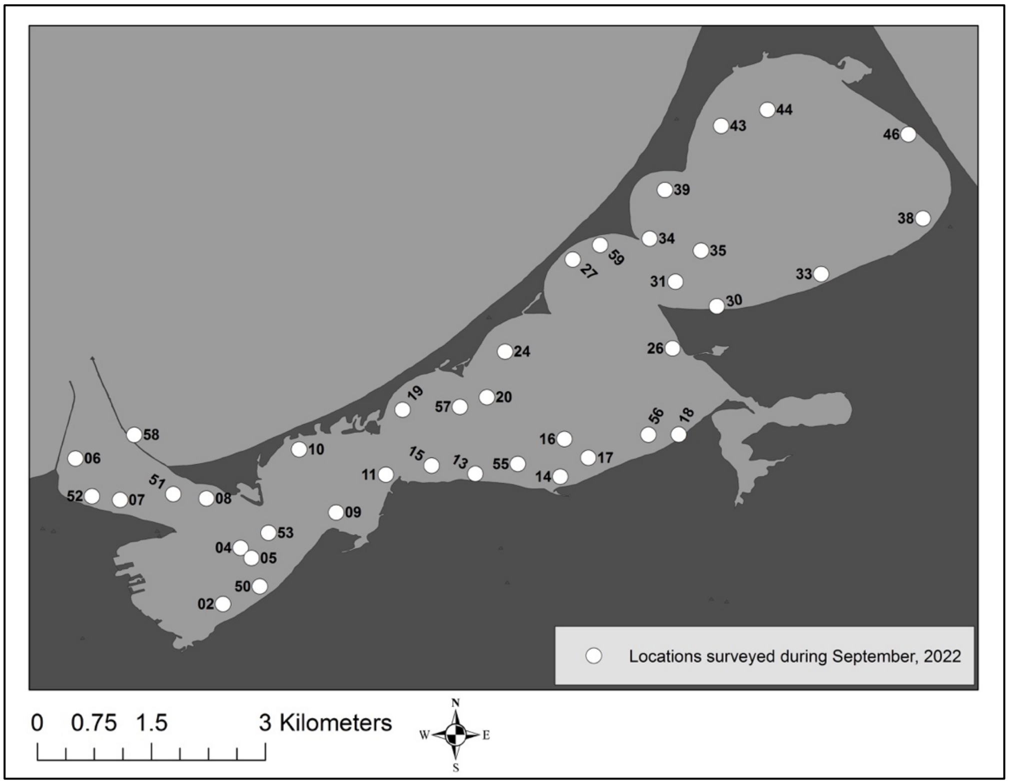

Figure 2. Depicts the 39 locations surveyed during September of 2019 through 2022 within Nantucket Harbor, MA along with the identifying numbers of each location.

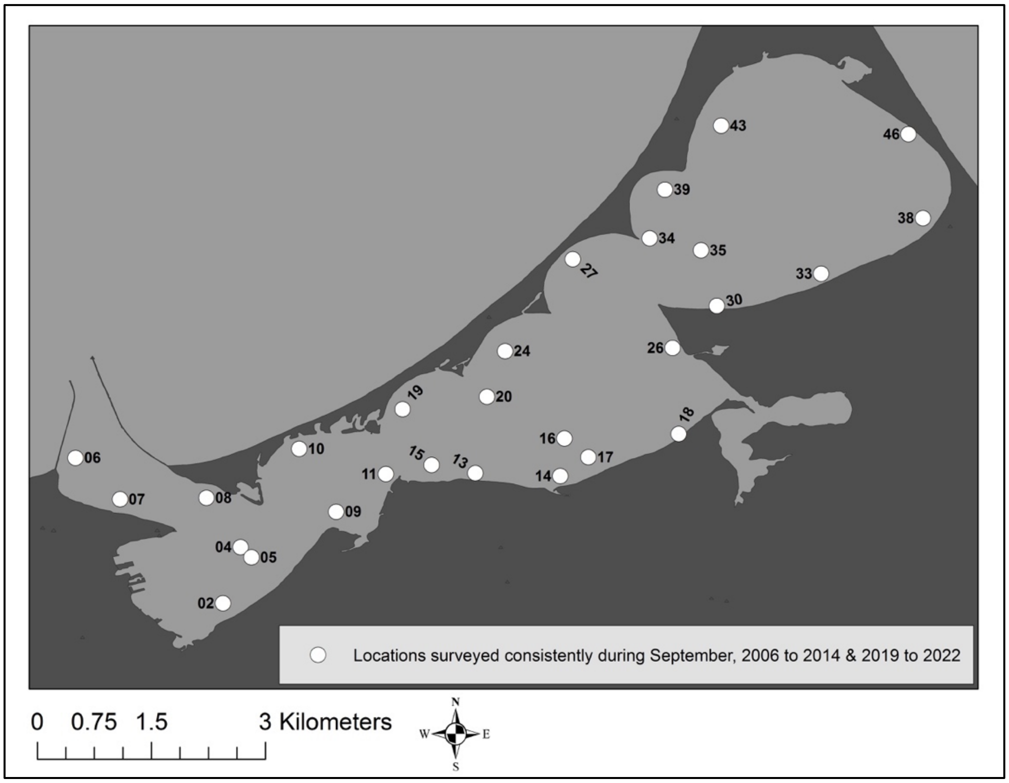

Figure 3. Depicts the 28 locations surveyed consistently from 2006 through 2014 and 2019 through 2022 within Nantucket Harbor, MA along with the identifying numbers of each location.

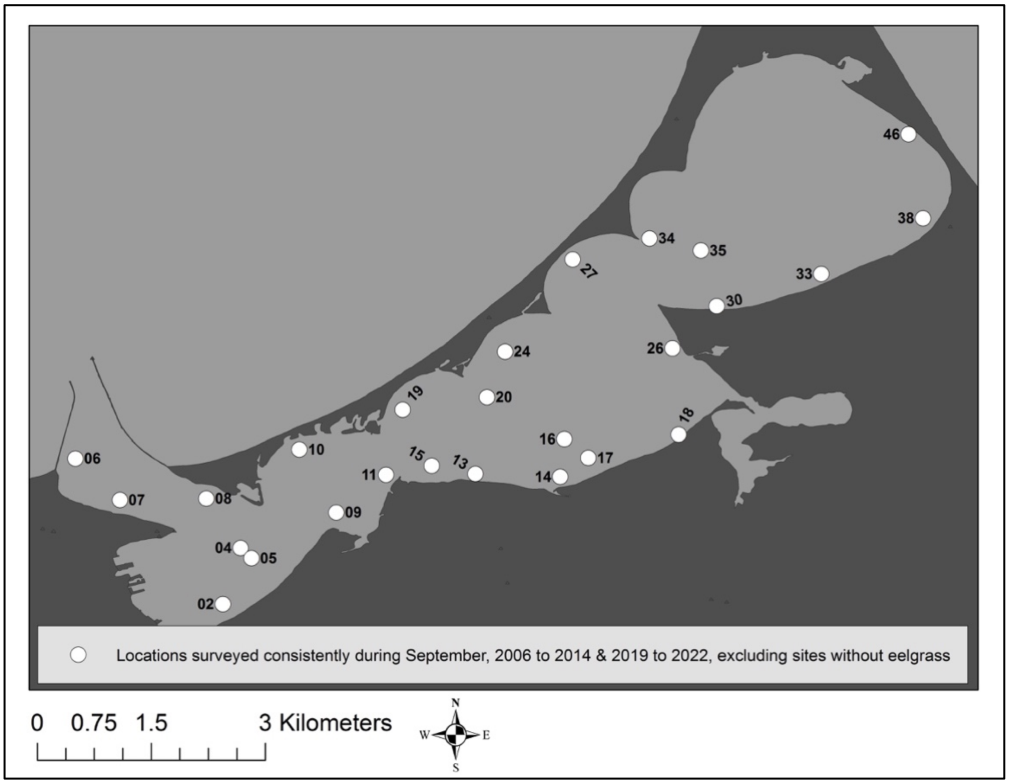

Figure 4. Depicts the 26 locations that had been surveyed consistently from 2006 through 2014 and 2019 through 2022 that contained eelgrass at some point over the course of the survey within Nantucket Harbor, MA along with the identifying numbers of each location.

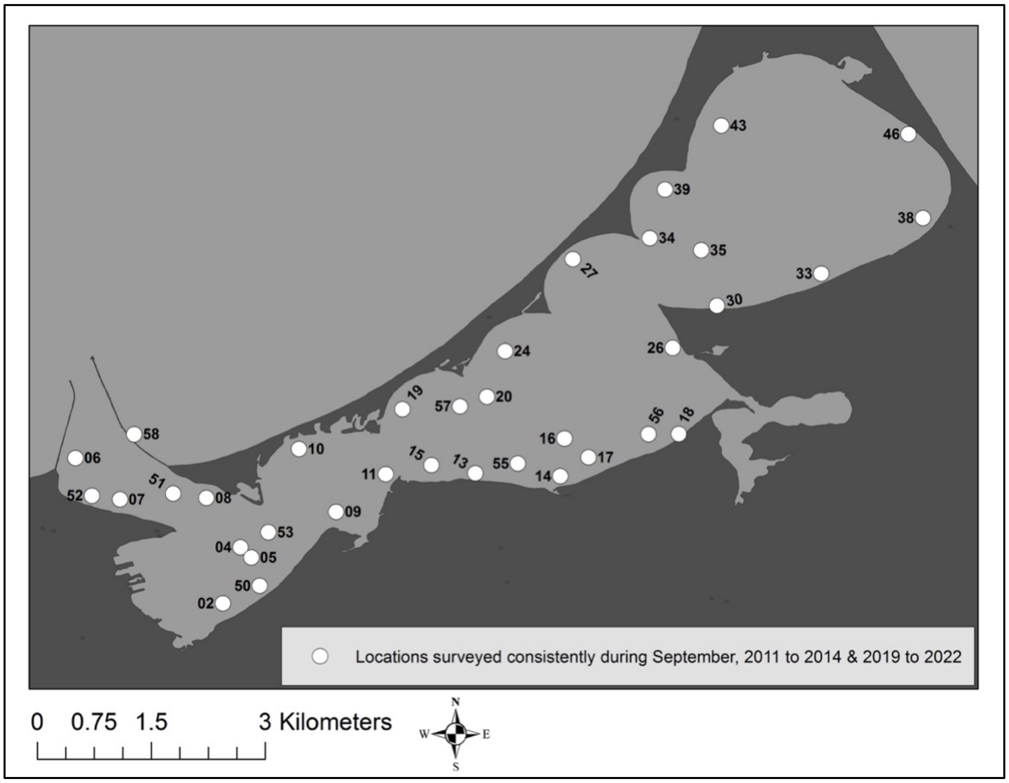

Figure 5. Depicts the 36 locations surveyed consistently from 2011 through 2014 and 2019 through 2022 within Nantucket Harbor, MA along with the identifying numbers of each location.

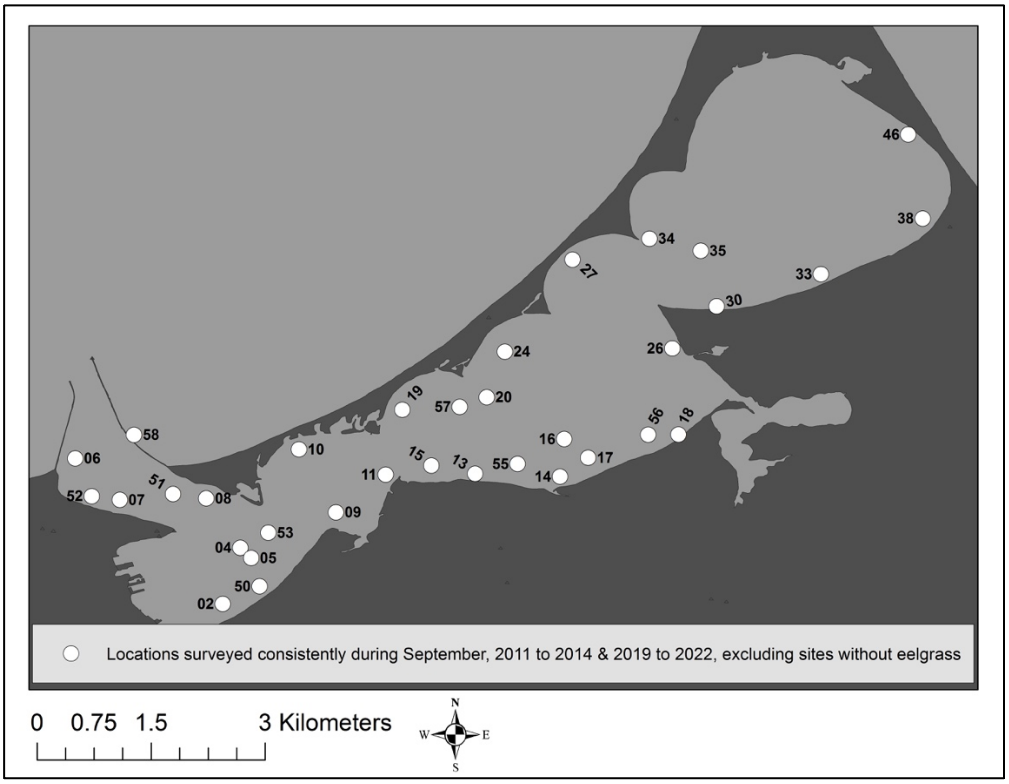

Figure 6. Depicts the 34 locations that had been surveyed consistently from 2011 through 2014 and 2019 through 2022 that contained eelgrass at some point over the course of the survey within Nantucket Harbor, MA along with the identifying numbers of each location.

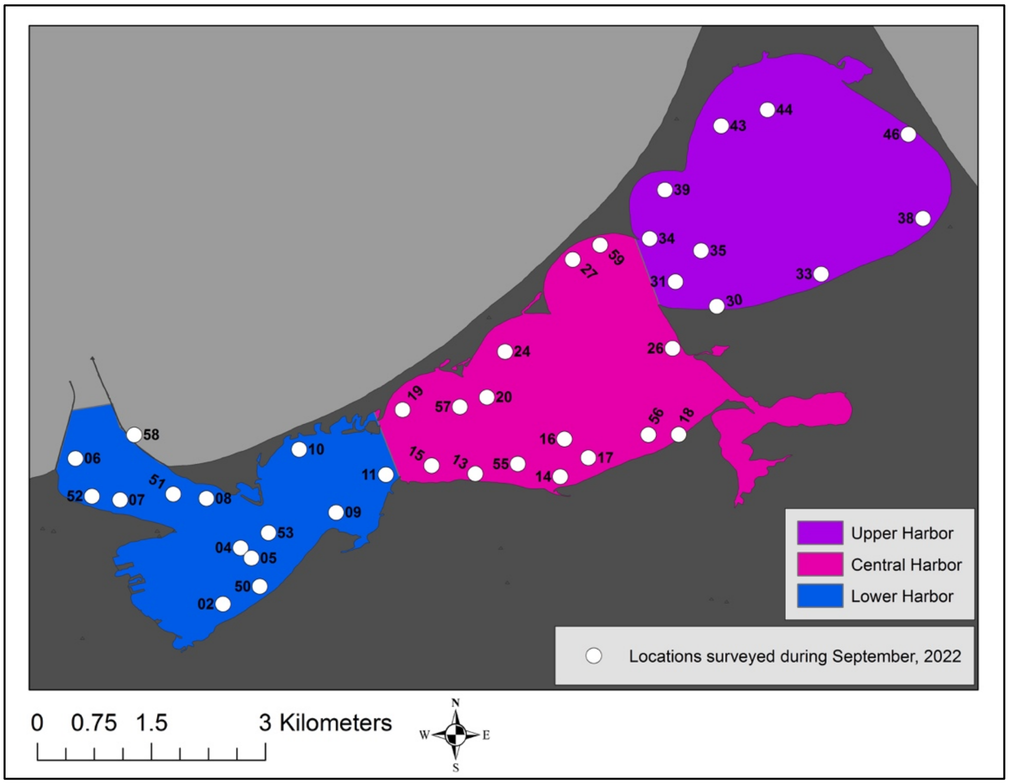

Figure 7. Depicts the 39 locations surveyed during September of 2019 through 2022 within Nantucket Harbor, MA along with the identifying numbers of each location and Nantucket Harbor divided into three regions.

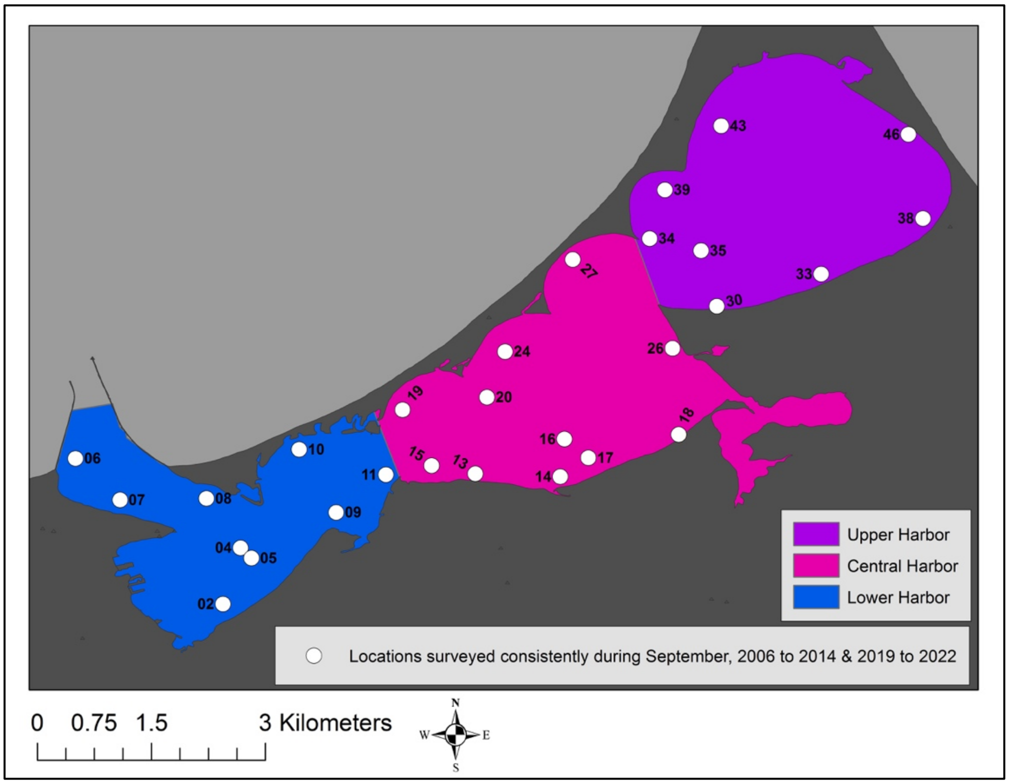

Figure 8. Depicts the 28 locations surveyed consistently from 2006 through 2014 and 2019 through 2022 within Nantucket Harbor, MA along with the identifying numbers of each location and Nantucket Harbor divided into three regions.

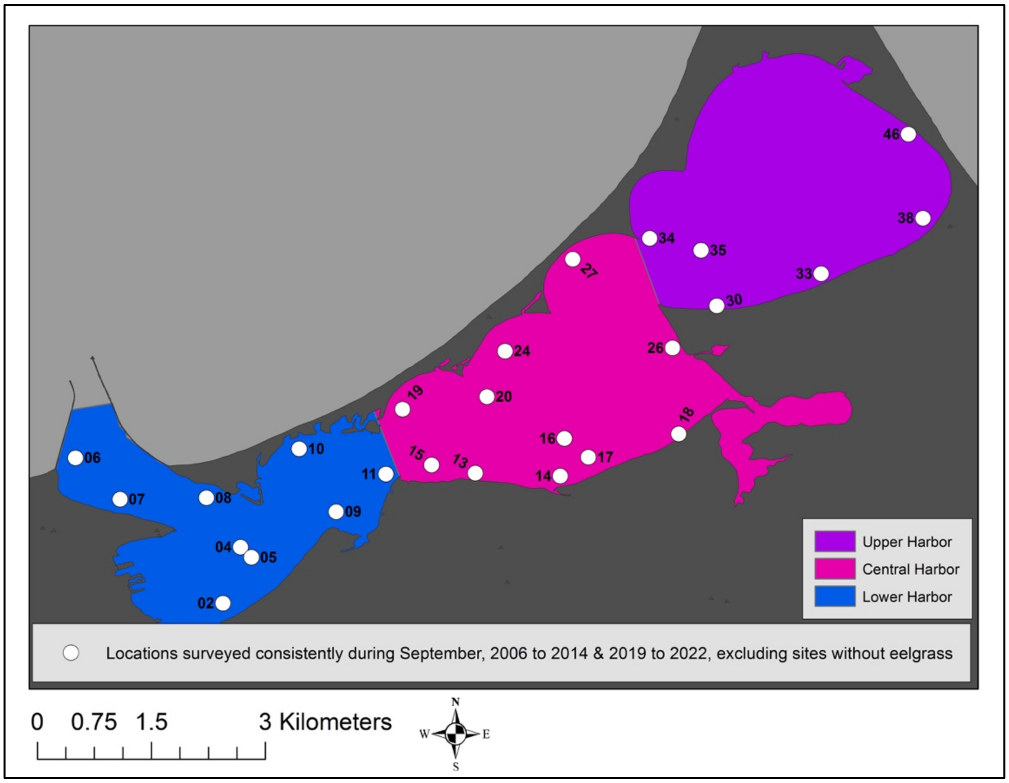

Figure 9. Depicts the 26 locations that had been surveyed consistently from 2006 through 2014 and 2019 through 2022 and contained eelgrass at some point over the course of the survey within Nantucket Harbor, MA, along with the identifying numbers of each location and Nantucket Harbor divided into three regions.

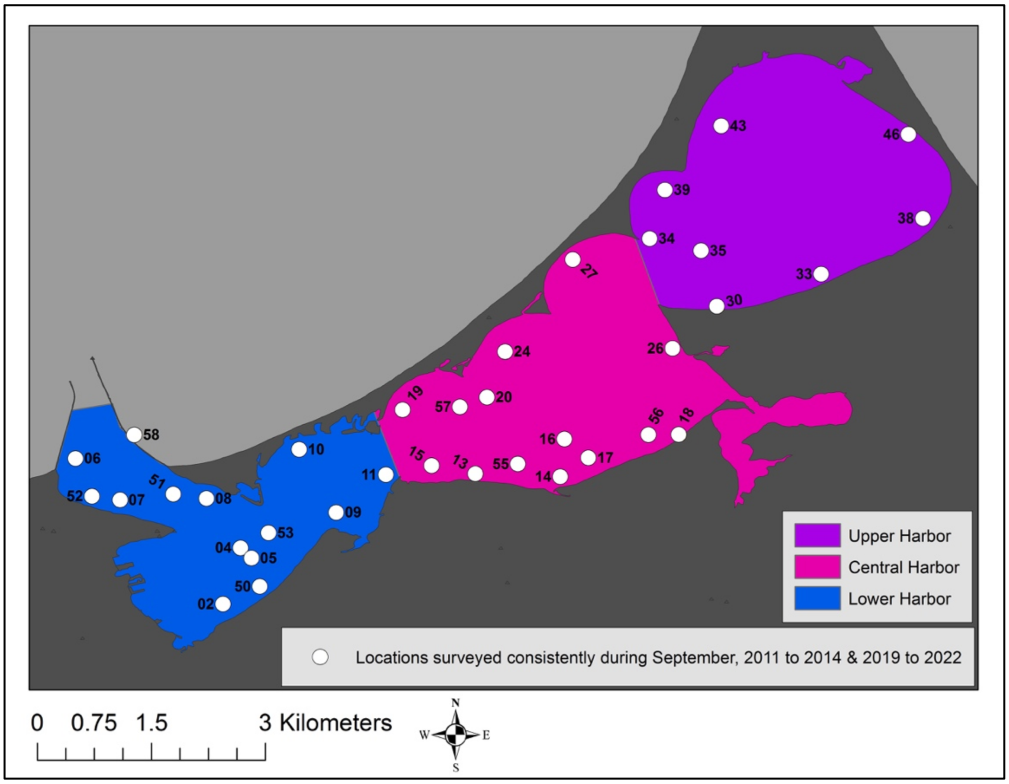

Figure 10. Depicts the 36 locations surveyed consistently from 2011 through 2014 and 2019 through 2022 within Nantucket Harbor, MA, along with the identifying numbers of each location and Nantucket Harbor divided into three regions.

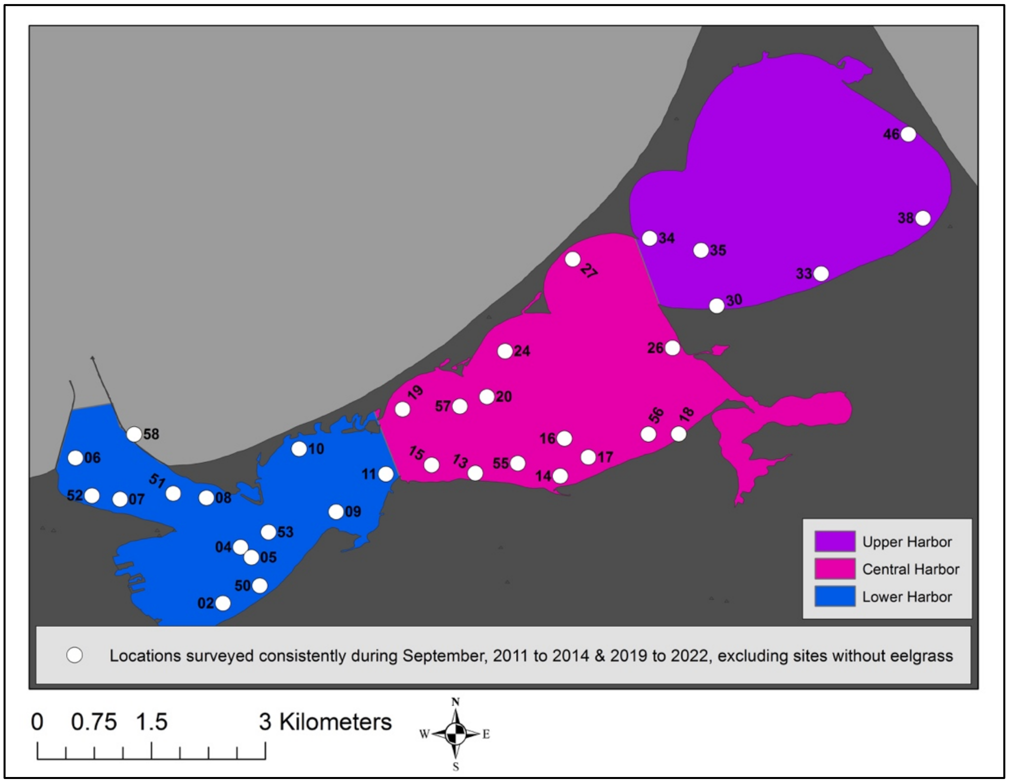

Figure 11. Depicts the 34 locations surveyed consistently from 2011 through 2014 and 2019 through 2022 and contained eelgrass at some point over the course of the survey within Nantucket Harbor, MA, along with the identifying numbers of each location and Nantucket Harbor divided into three regions.

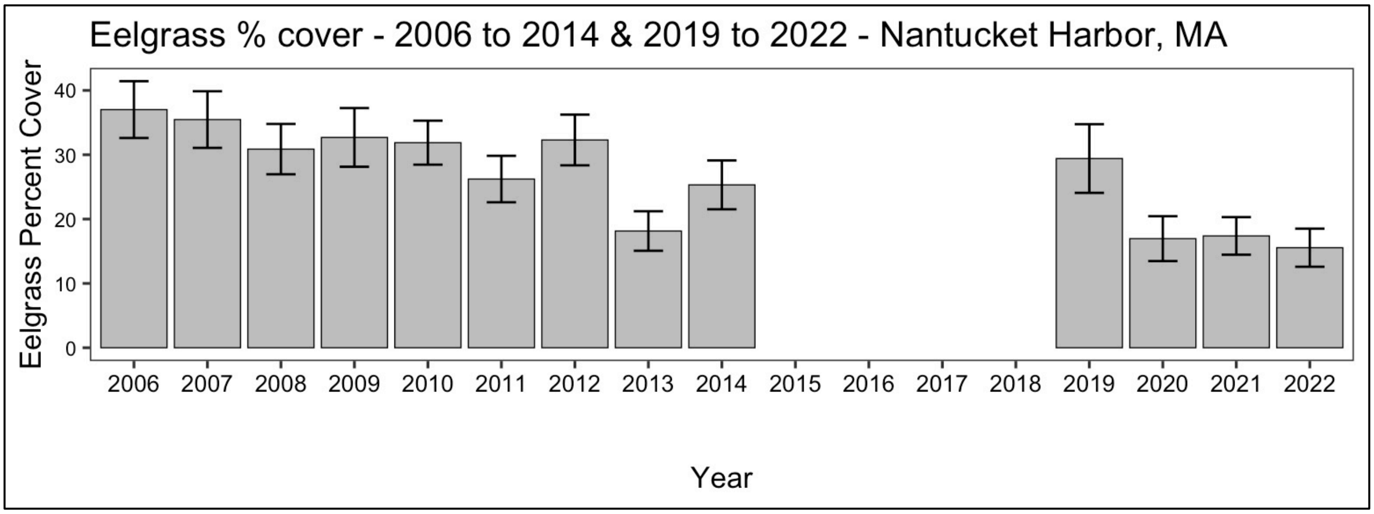

Figure 12. Averages of eelgrass percent cover by year ± standard error of the mean (SE) of the 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 and contained eelgrass during at least 1 year of the 13 years of data throughout Nantucket Harbor, MA (Figure 4).

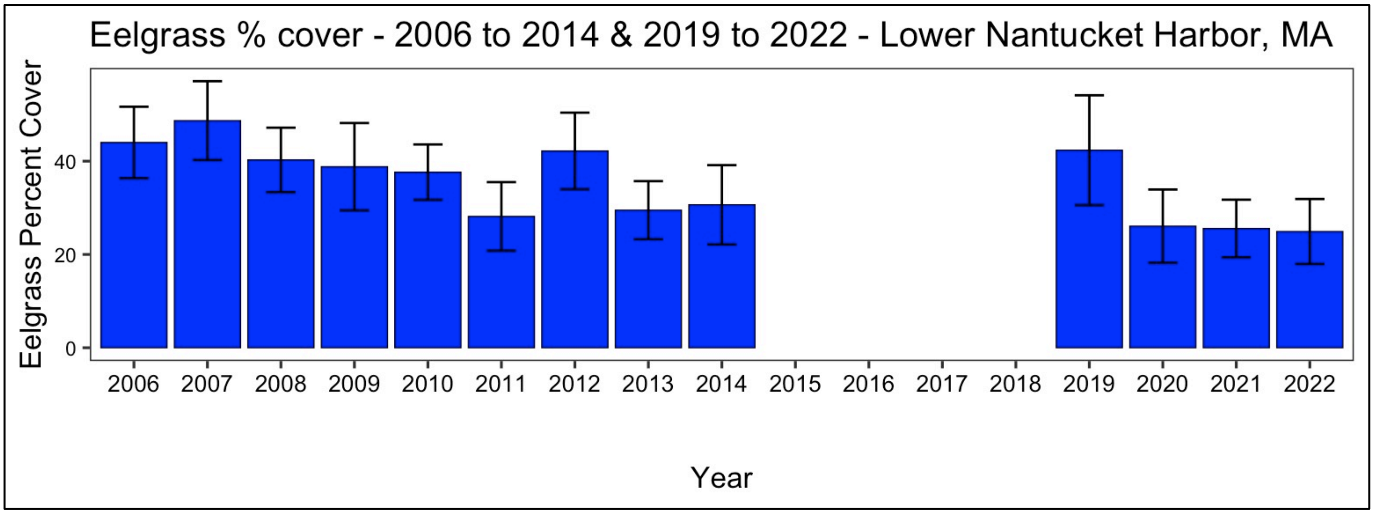

Figure 13. Averages of eelgrass percent cover by year ± standard error of the mean (SE) of the sites confined to the lower region of Nantucket Harbor, MA, out of the 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 and contained eelgrass during at least 1 year of the 13 years of data (Figure 9).

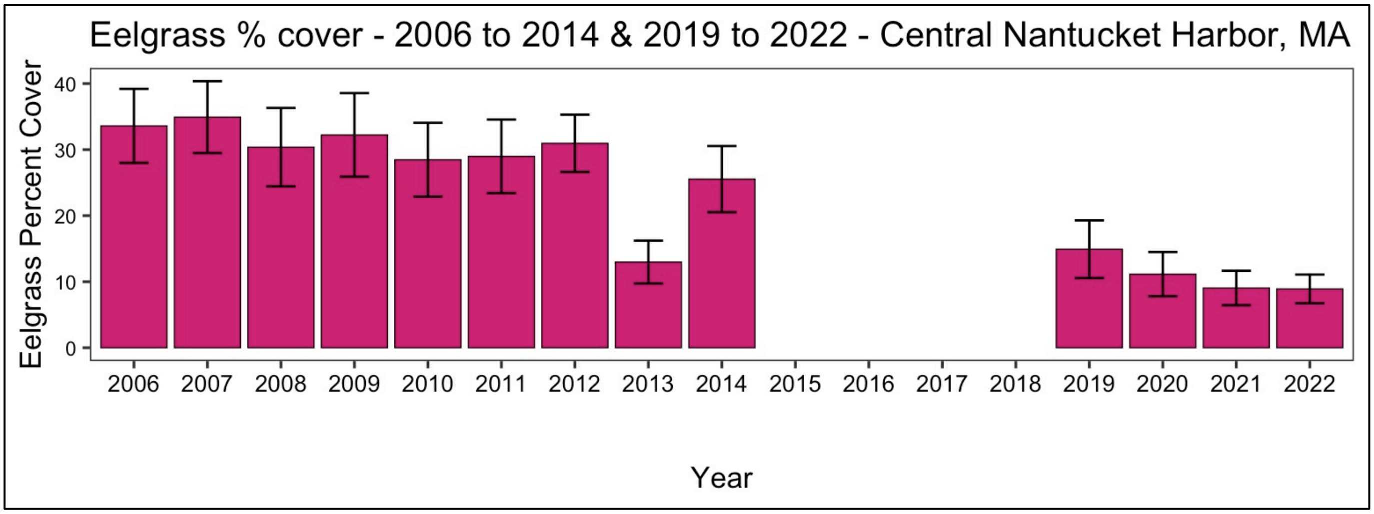

Figure 14. Averages of eelgrass percent cover by year ± standard error of the mean (SE) of the sites confined to the central region of Nantucket Harbor, MA, out of the 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 and contained eelgrass during at least 1 year of the 13 years of data (Figure 9).

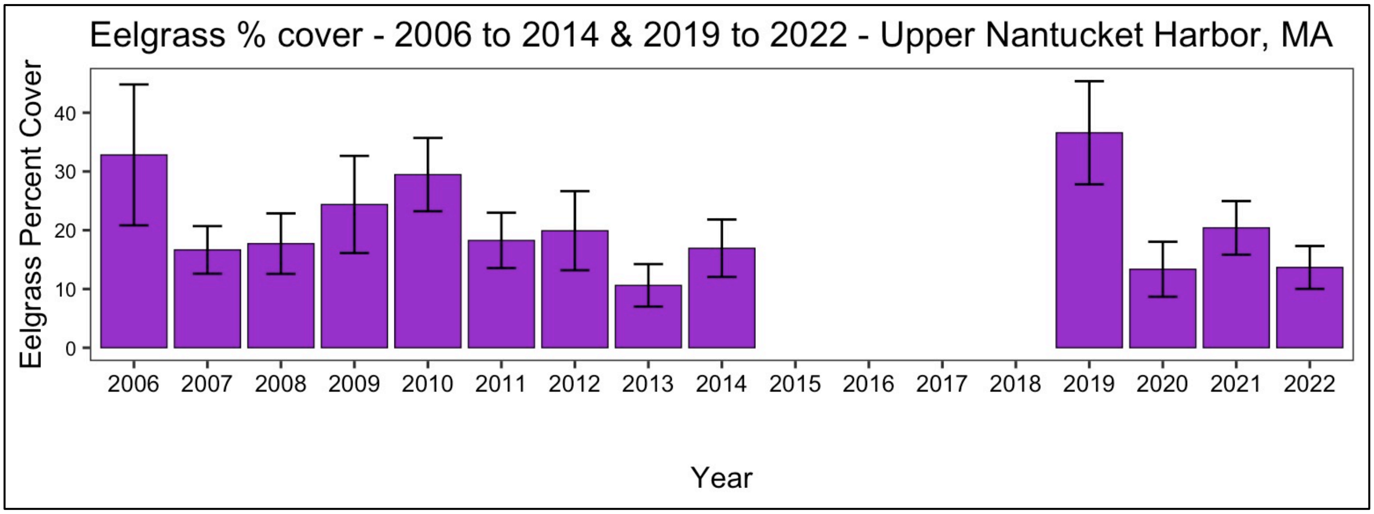

Figure 15. Averages of eelgrass percent cover by year ± standard error of the mean (SE) of the sites confined to the upper region of Nantucket Harbor, MA, out of the 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 and contained eelgrass during at least 1 year of the 13 years of data (Figure 9).

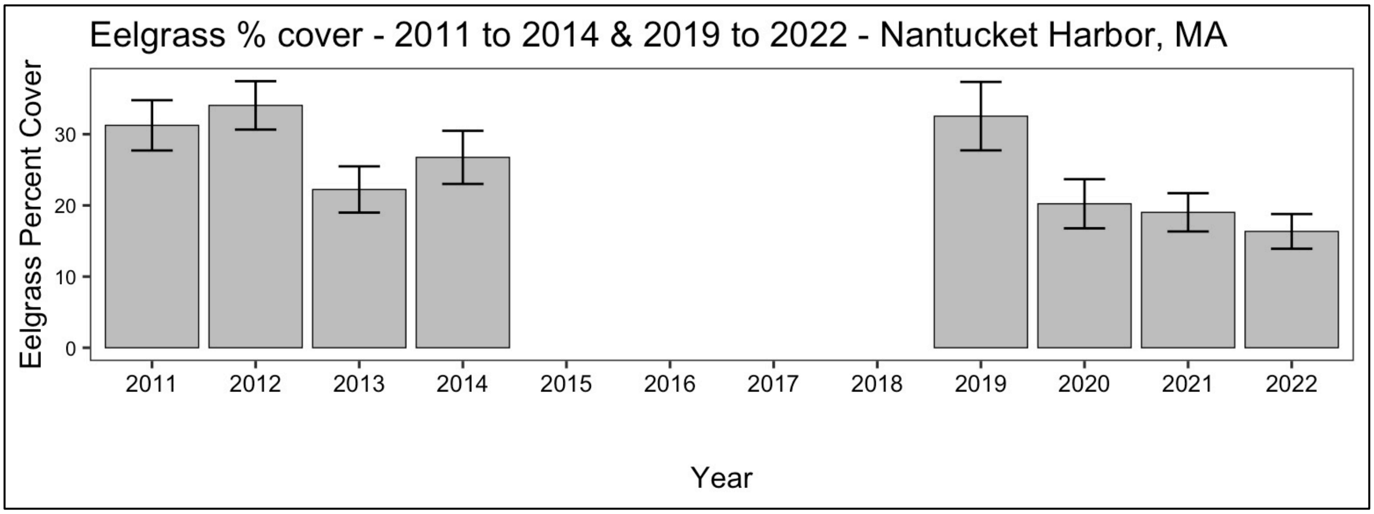

Figure 16. Averages of eelgrass percent cover by year ± standard error of the mean (SE) of the 34 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 and contained eelgrass during at least 1 year of the 13 years of data within Nantucket Harbor, MA (Figure 6).

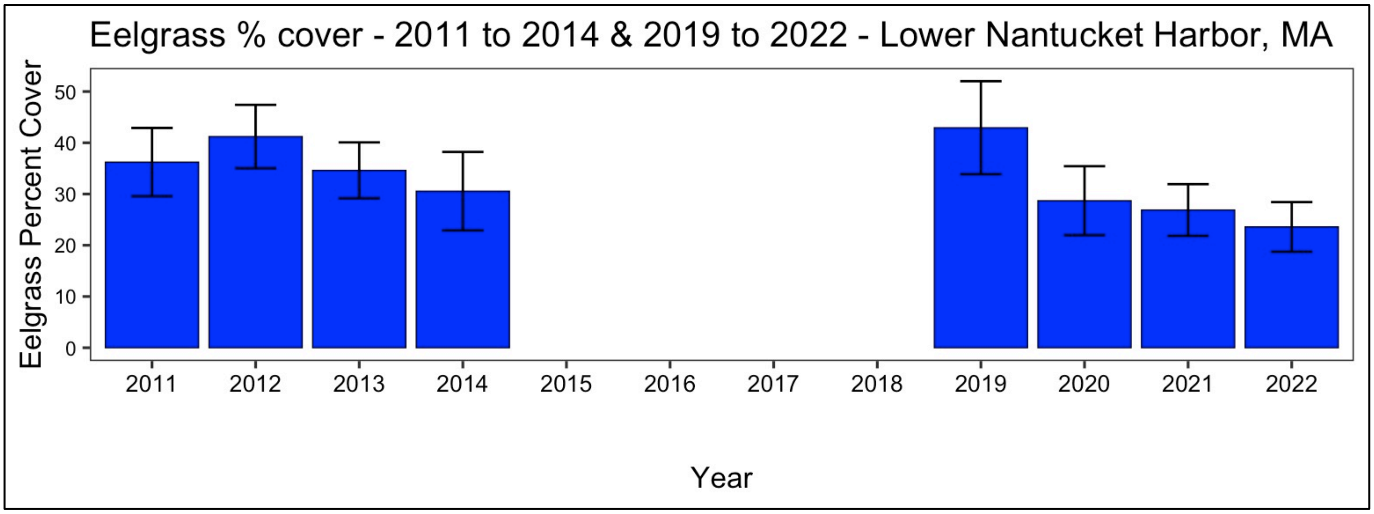

Figure 17. Averages of eelgrass percent cover by year ± standard error of the mean (SE) of the sites confined to the lower region of Nantucket Harbor out of the 34 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 contained eelgrass during at least 1 year of the 8 years of data (Figure 11).

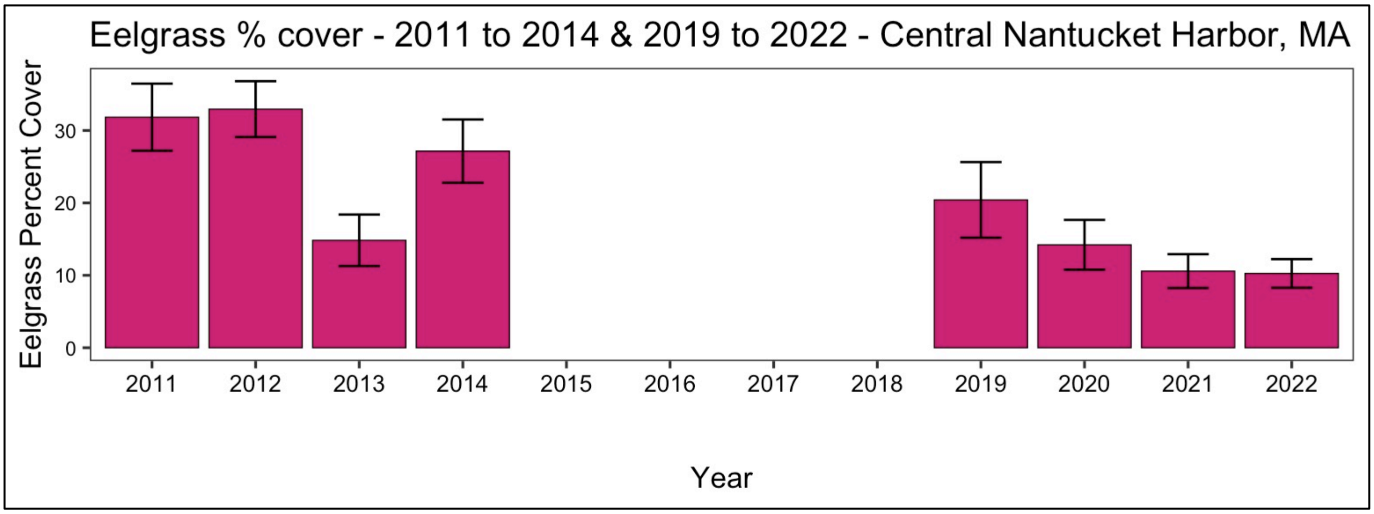

Figure 18. Averages of eelgrass percent cover by year ± standard error of the mean (SE) of the sites confined to the central region of Nantucket Harbor out of the 34 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 contained eelgrass during at least 1 year of the 8 years of data (Figure 11).

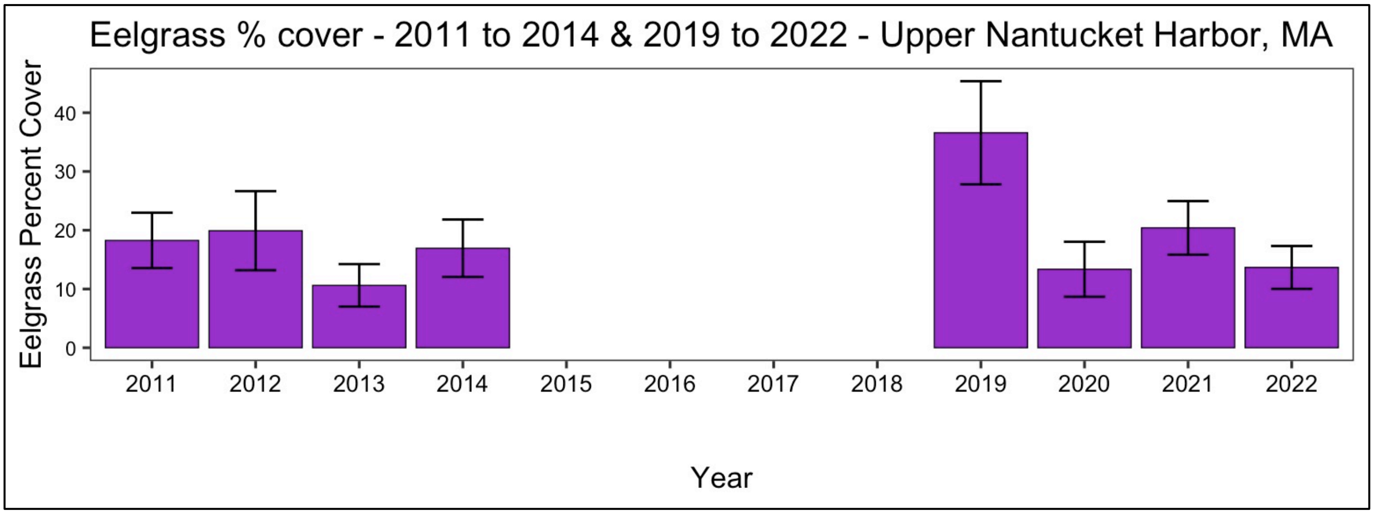

Figure 19. Averages of eelgrass percent cover by year ± standard error of the mean (SE) of the sites confined to the upper region of Nantucket Harbor out of the 34 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 contained eelgrass during at least 1 year of the 8 years of data (Figure 11).

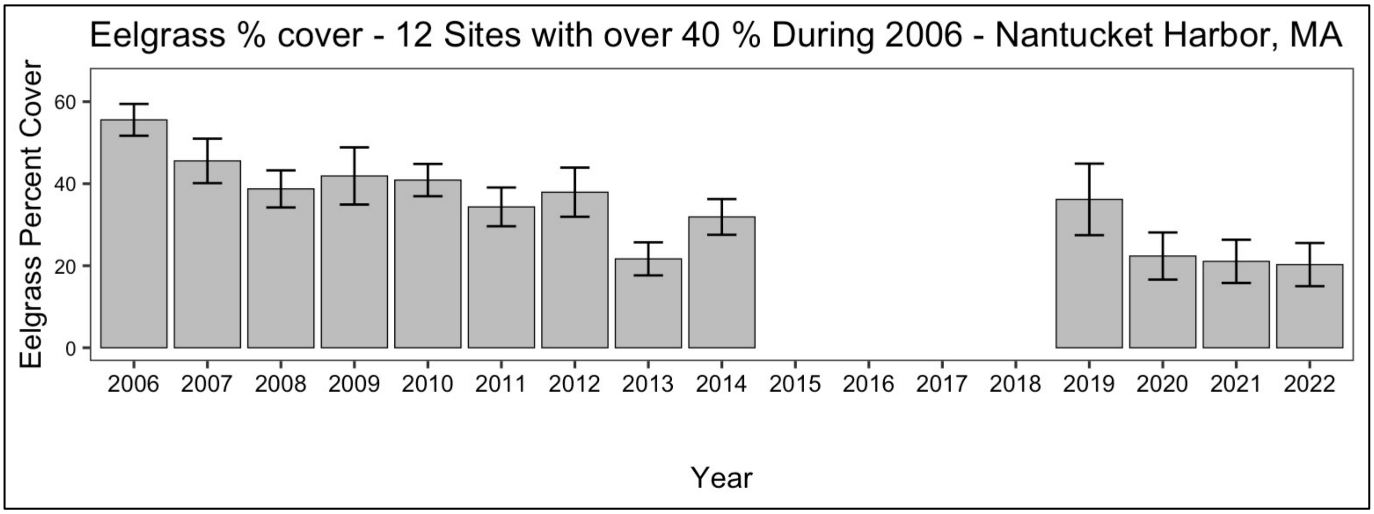

Figure 20. Averages of eelgrass percentage cover by year ± standard error of the mean (SE) of 12 sites that had over 40% eelgrass cover during 2006 throughout Nantucket Harbor, MA. This included the following sites: 08, 09, 34, 35, 20, 38, 05, 14, 19, 24, 26, 02.

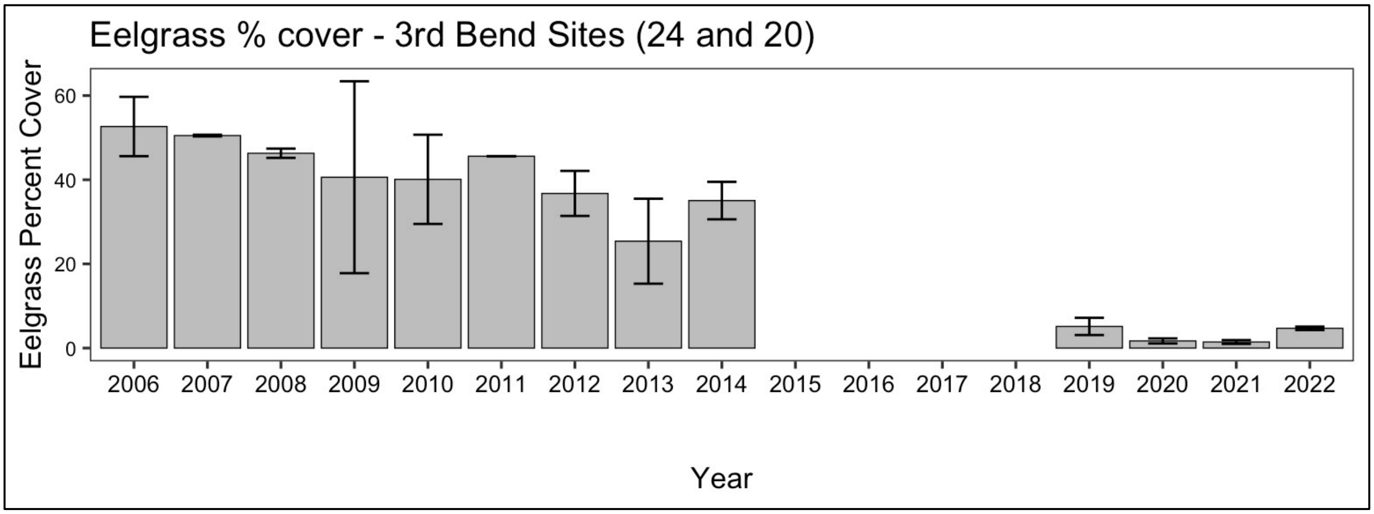

Figure 21. Averages of eelgrass percentage cover by year ± standard error of the mean (SE) for two sites in third bend within Nantucket Harbor, MA (Figure 4).

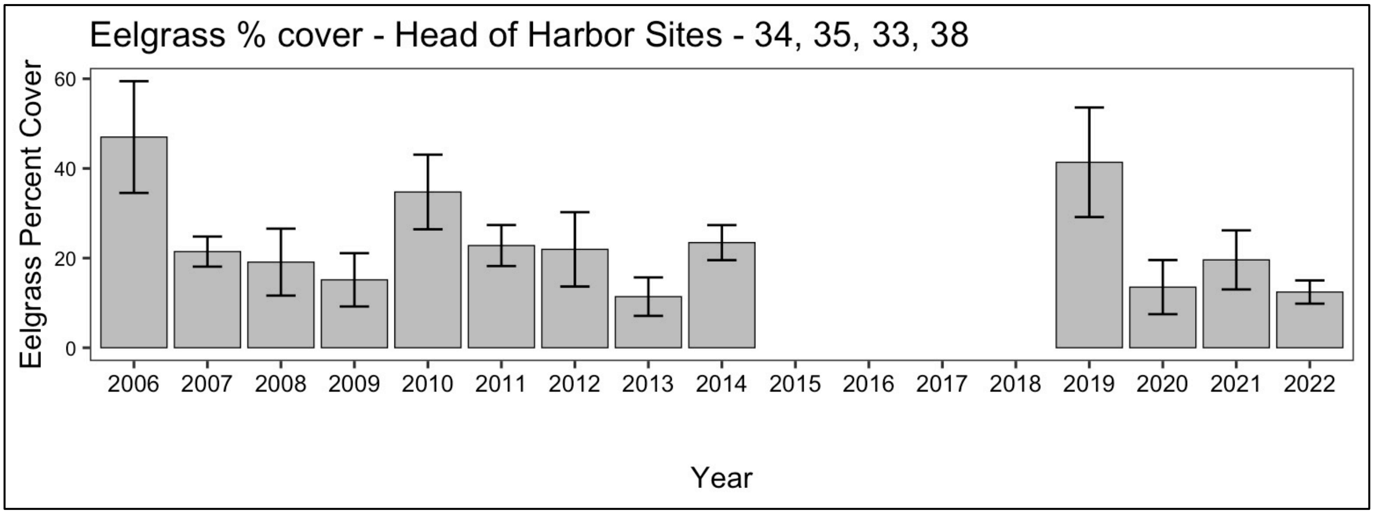

Figure 22. Averages of eelgrass percentage cover by year ± standard error of the mean (SE) for four sites in the Head of the Harbor (Upper Harbor) within Nantucket Harbor, MA (Figure 4).

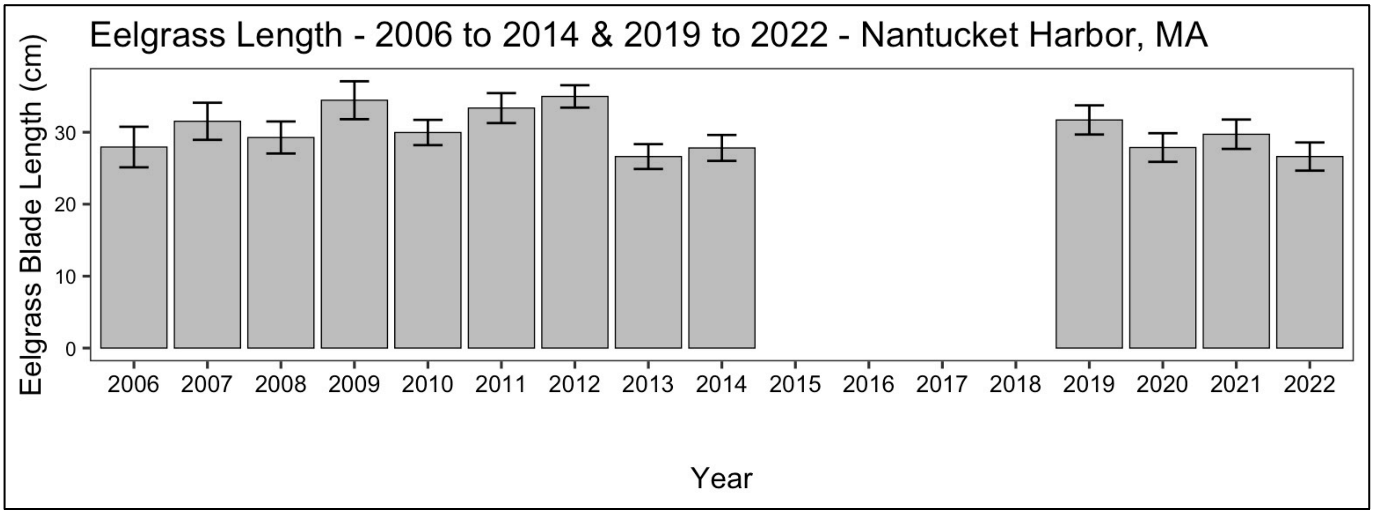

Figure 23. Averages of eelgrass blade length by year ± standard error of the mean (SE) of the 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 4).

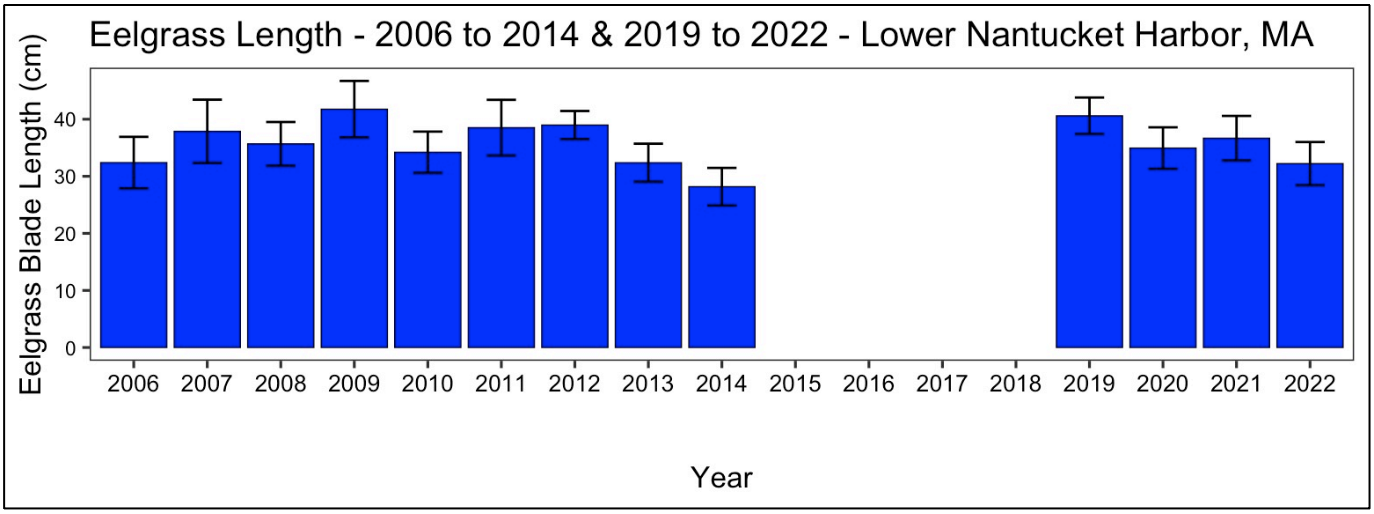

Figure 24. Averages of eelgrass blade length by year ± standard error of the mean (SE) of the sites confined to the lower region of Nantucket Harbor, MA, out of the of the 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 9).

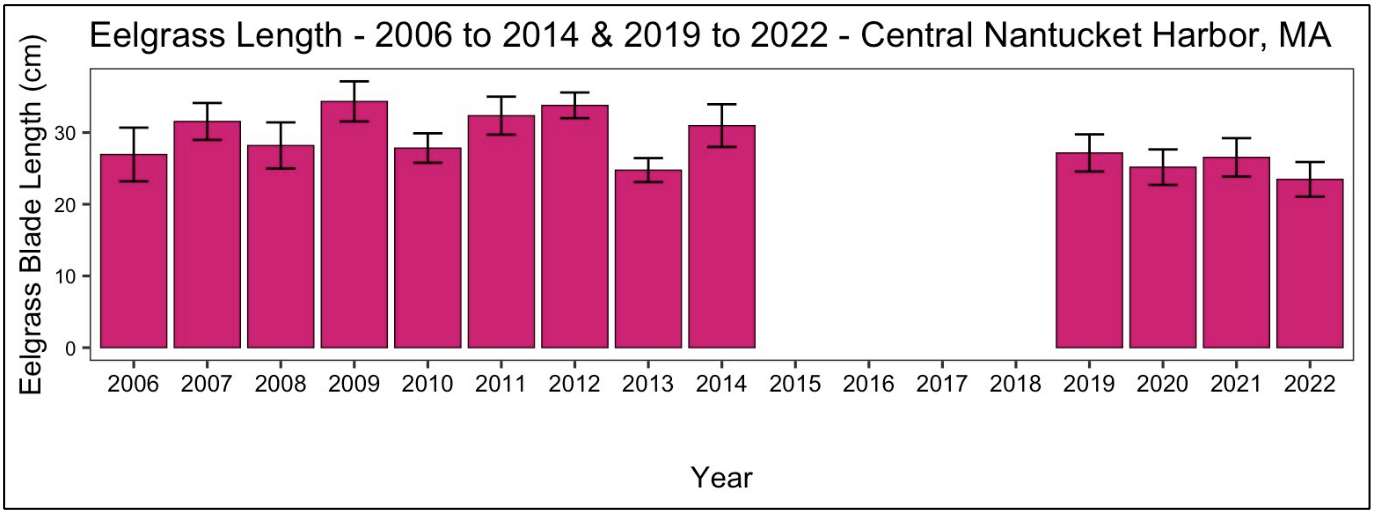

Figure 25. Averages of eelgrass blade length by year ± standard error of the mean (SE) of the sites confined to the central region of Nantucket Harbor, MA, out of the 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 9).

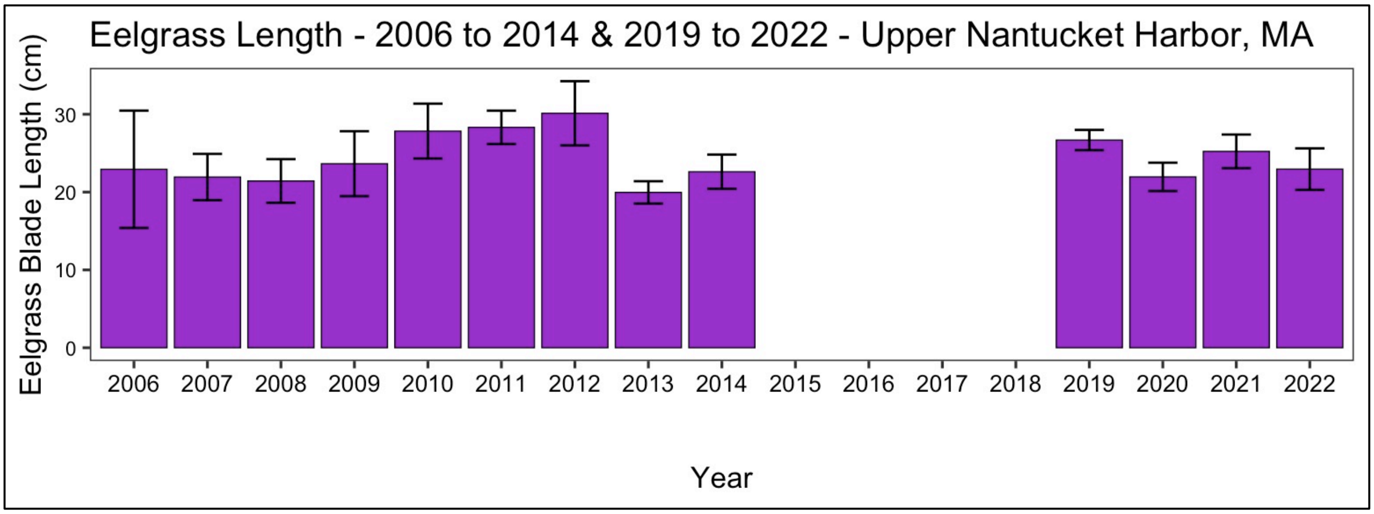

Figure 26. Averages of eelgrass blade length by year ± standard error of the mean (SE) of the sites confined to the upper region of Nantucket Harbor, MA, out of the 26 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 9).

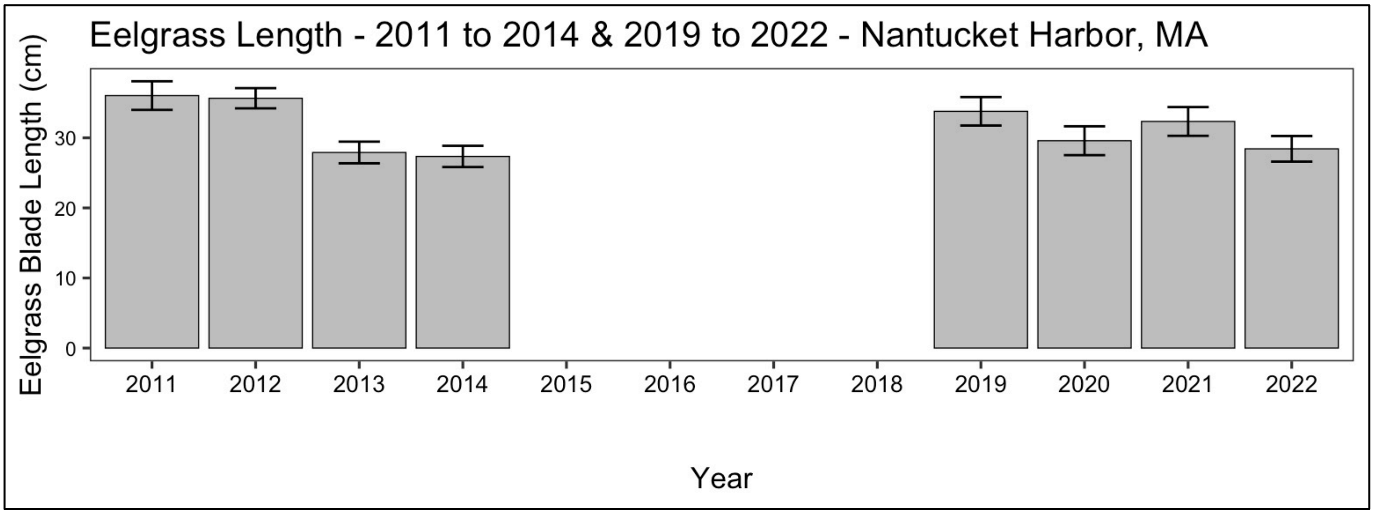

Figure 27. Averages of eelgrass blade length by year ± standard error of the mean (SE) of the 34 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 6).

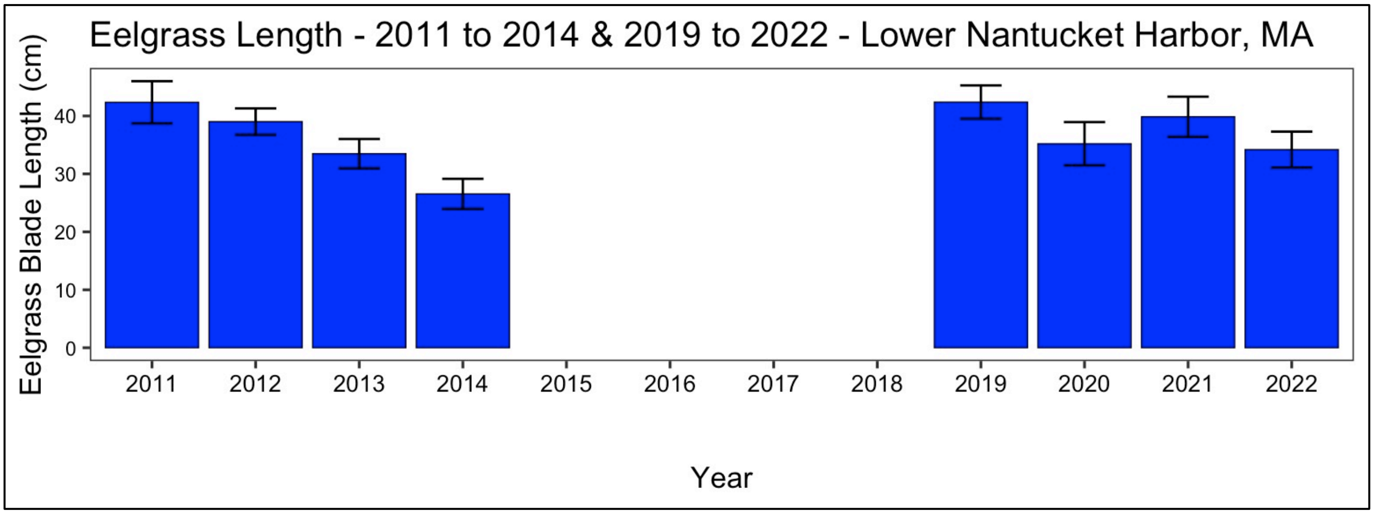

Figure 28. Averages of eelgrass blade length by year ± standard error of the mean (SE) of the sites confined to the lower region of Nantucket Harbor, MA, out of the 34 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 11).

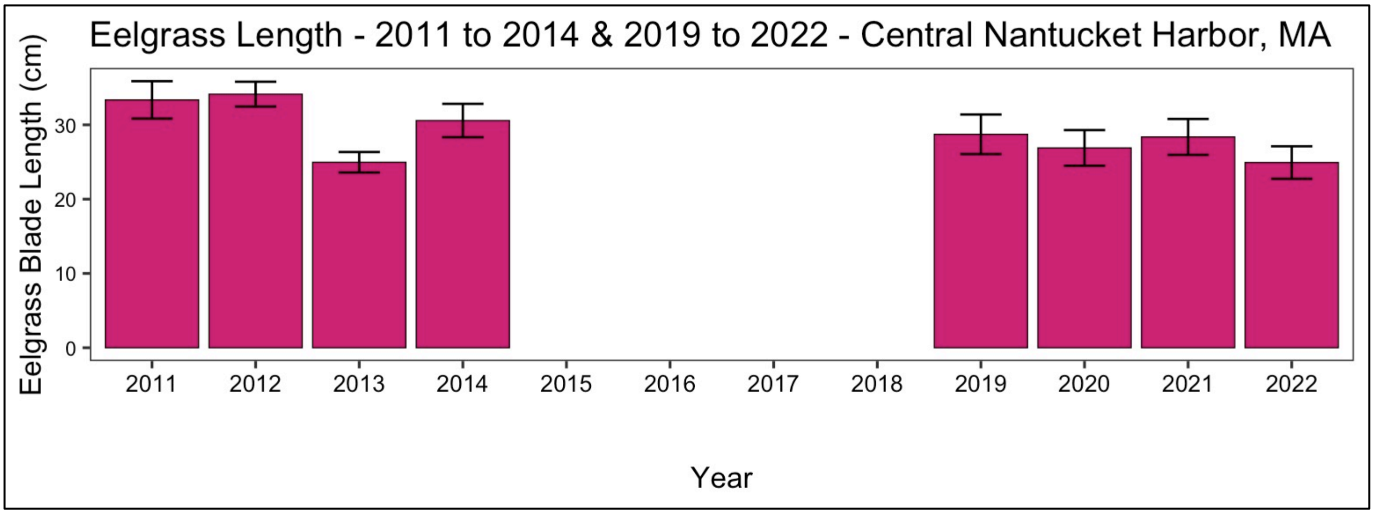

Figure 29. Averages of eelgrass blade length by year ± standard error of the mean (SE) of the sites confined to the central region of Nantucket Harbor, MA, out of the 34 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 11).

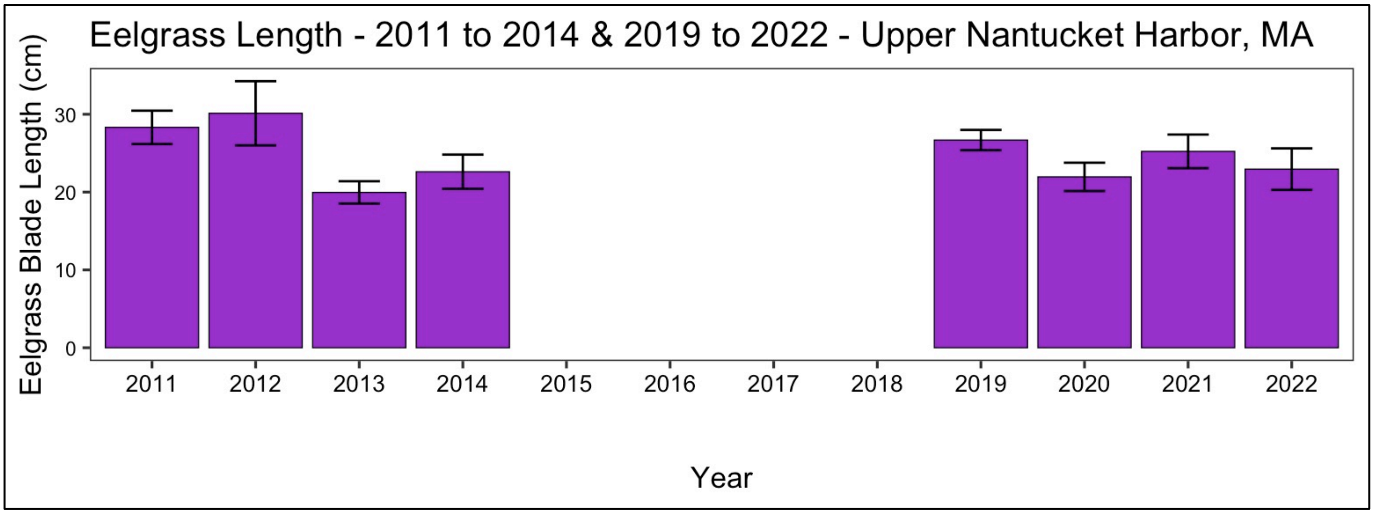

Figure 30. Averages of eelgrass blade length by year ± standard error of the mean (SE) of the sites confined to the upper region of Nantucket Harbor, MA, out of the 34 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 11).

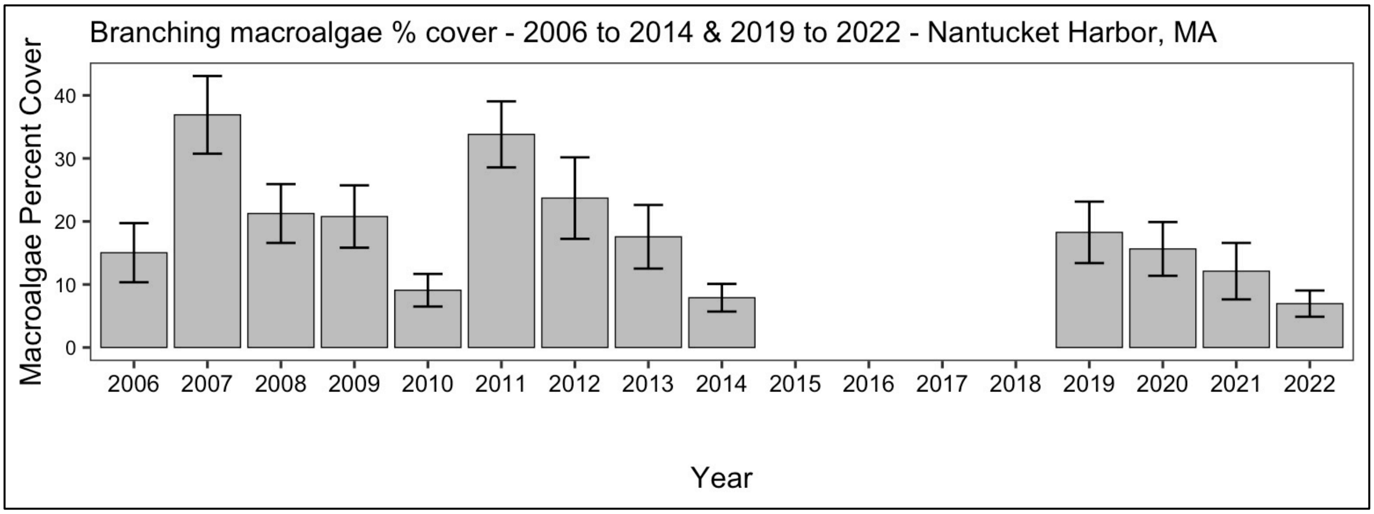

Figure 31. Averages of branching macroalgae percent cover by year standard error of the mean (SE) of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 3).

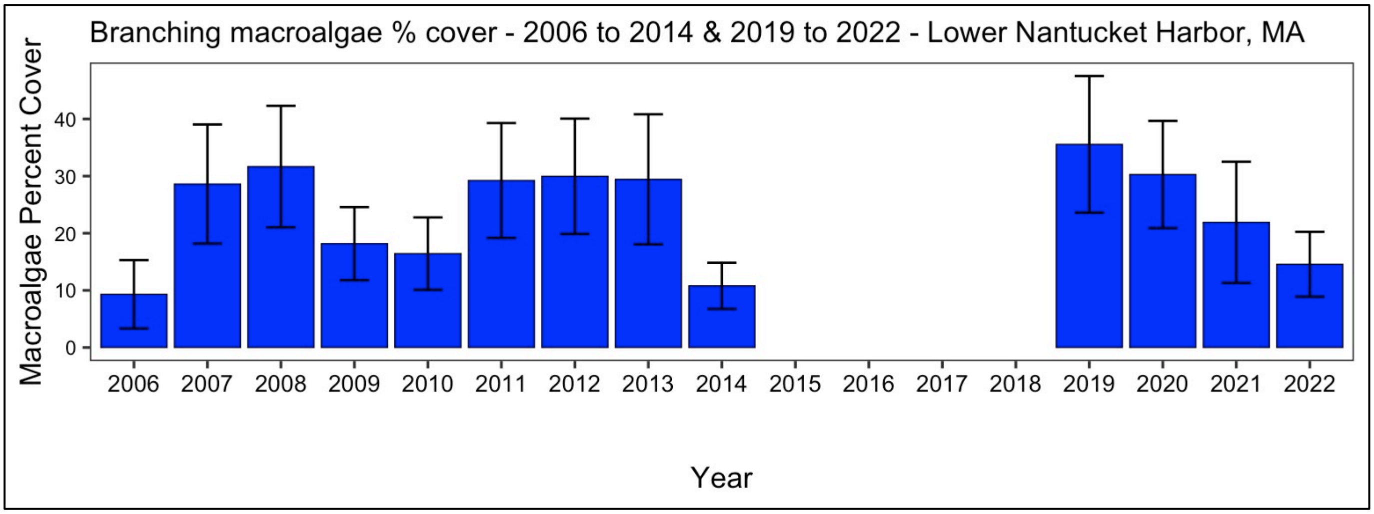

Figure 32. Averages of branching macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the lower region of Nantucket Harbor, MA, out of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 8).

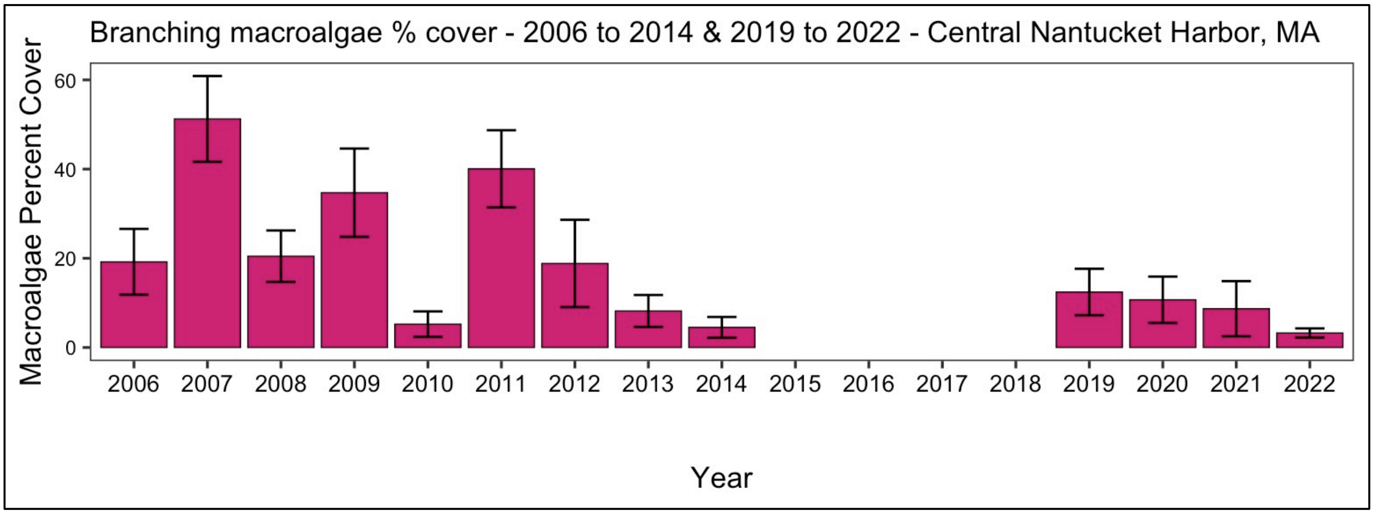

Figure 33. Averages of branching macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the central region of Nantucket Harbor, MA, out of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 8).

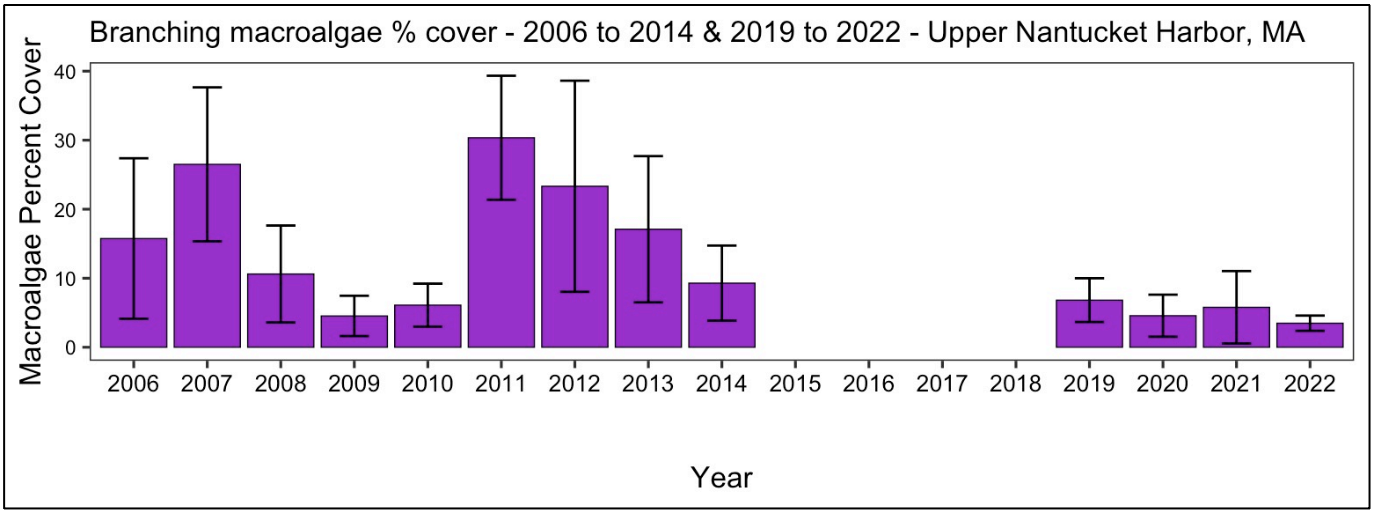

Figure 34. Averages of branching macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the upper region of Nantucket Harbor, MA, out of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 8).

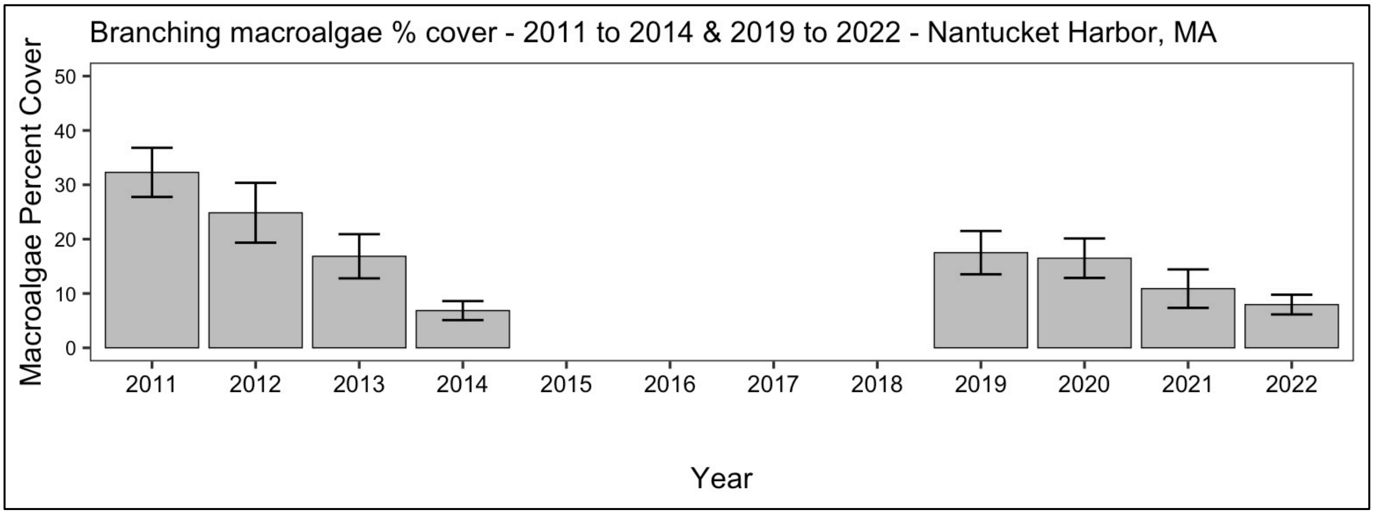

Figure 35. Averages of branching macroalgae percent cover by year ± standard error of the mean (SE) of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 within Nantucket Harbor, MA (Figure 5).

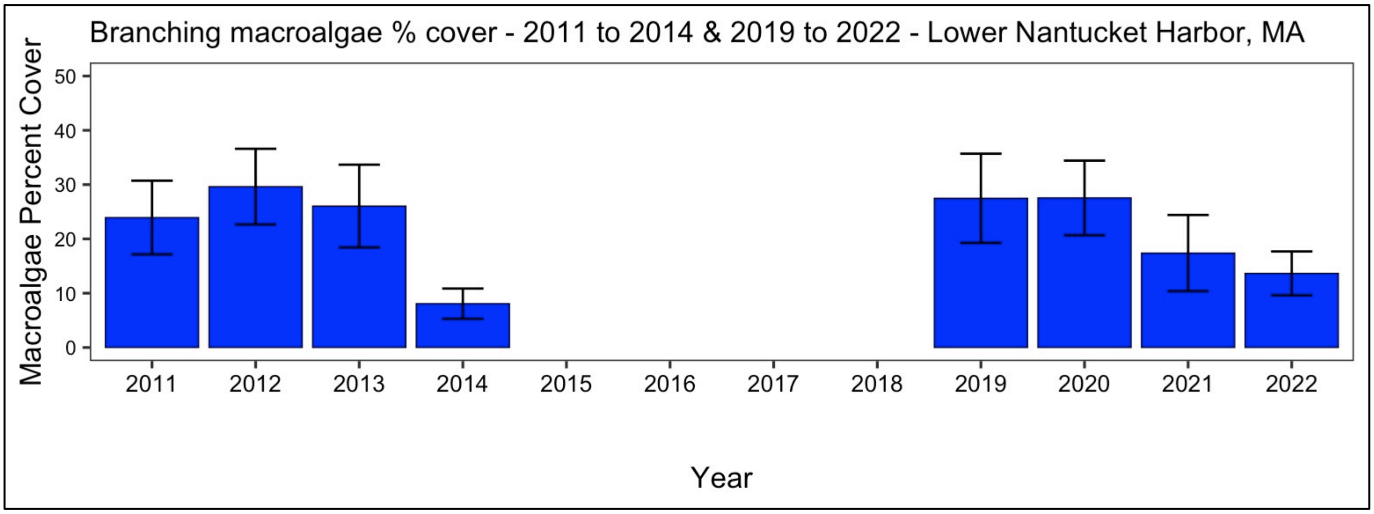

Figure 36. Averages of branching macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the lower region of Nantucket Harbor, MA, out of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 10).

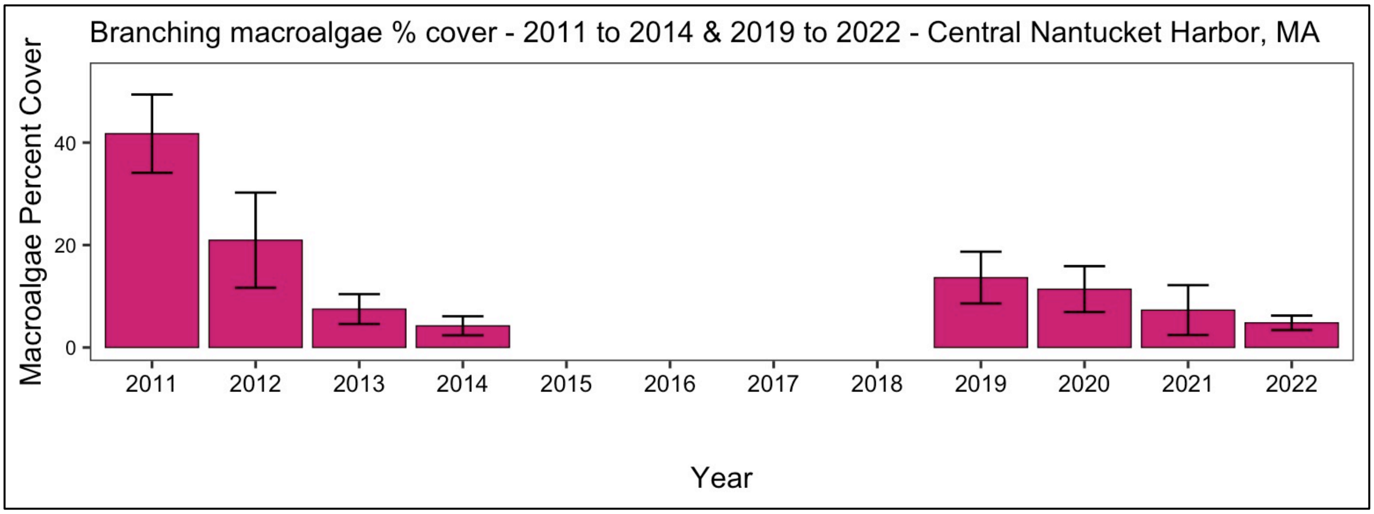

Figure 37. Averages of branching macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the central region of Nantucket Harbor, MA, out of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 10).

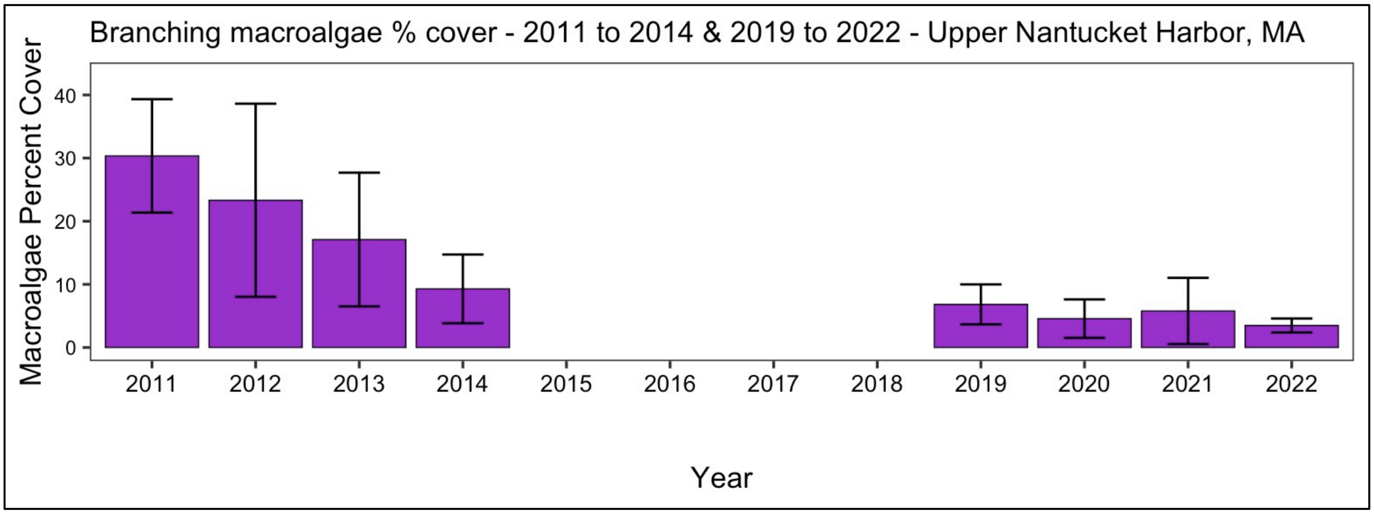

Figure 38. Averages of branching macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the upper region of Nantucket Harbor, MA, out of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 10).

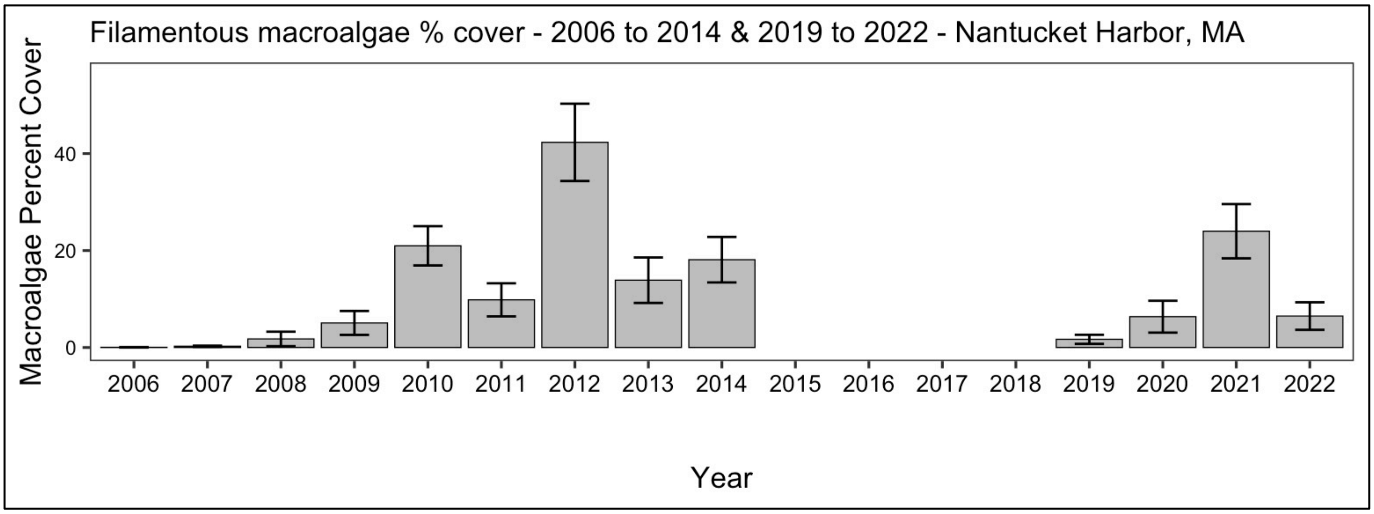

Figure 39. Averages of filamentous macroalgae percent cover by year standard error of the mean (SE) of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 3).

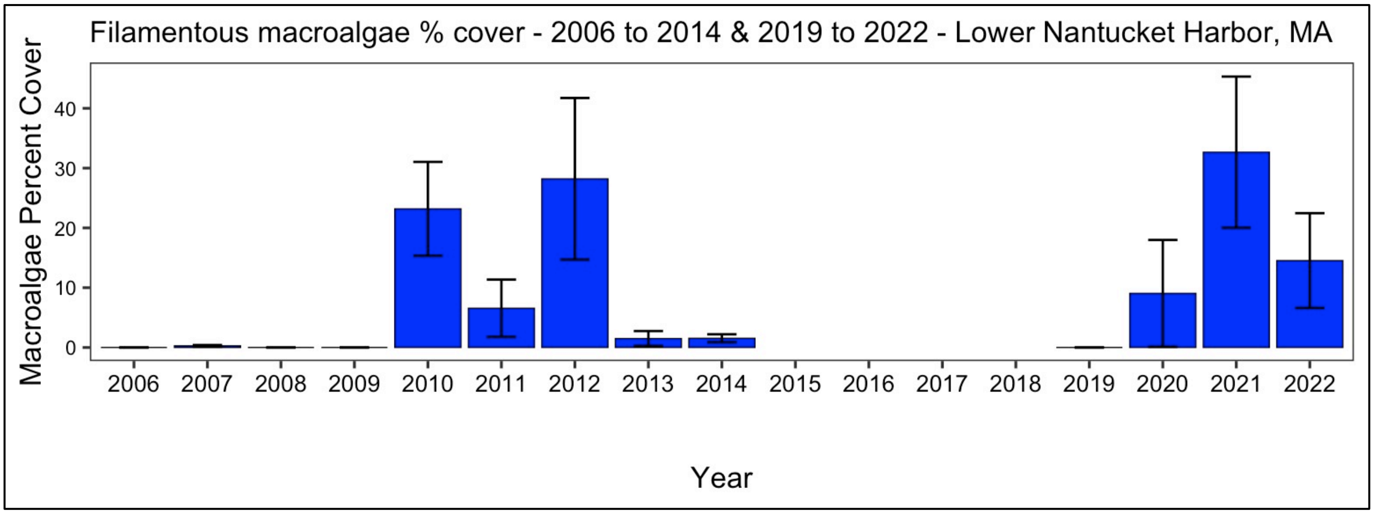

Figure 40. Averages of filamentous macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the lower region of Nantucket Harbor, MA, out of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 8).

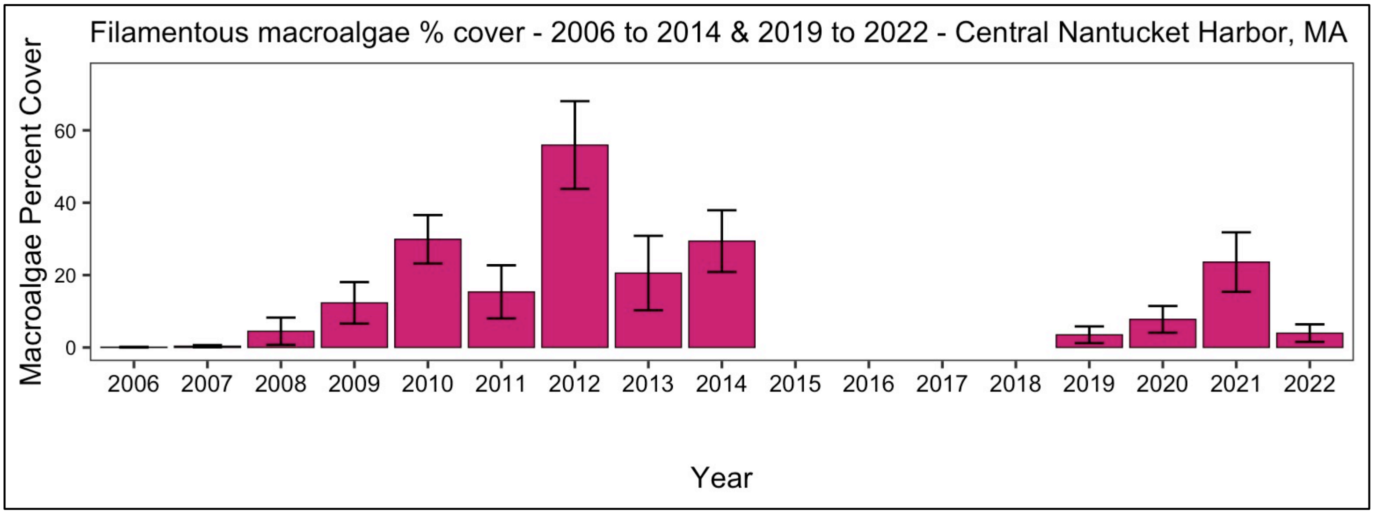

Figure 41. Averages of filamentous macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the central region of Nantucket Harbor, MA, out of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 8).

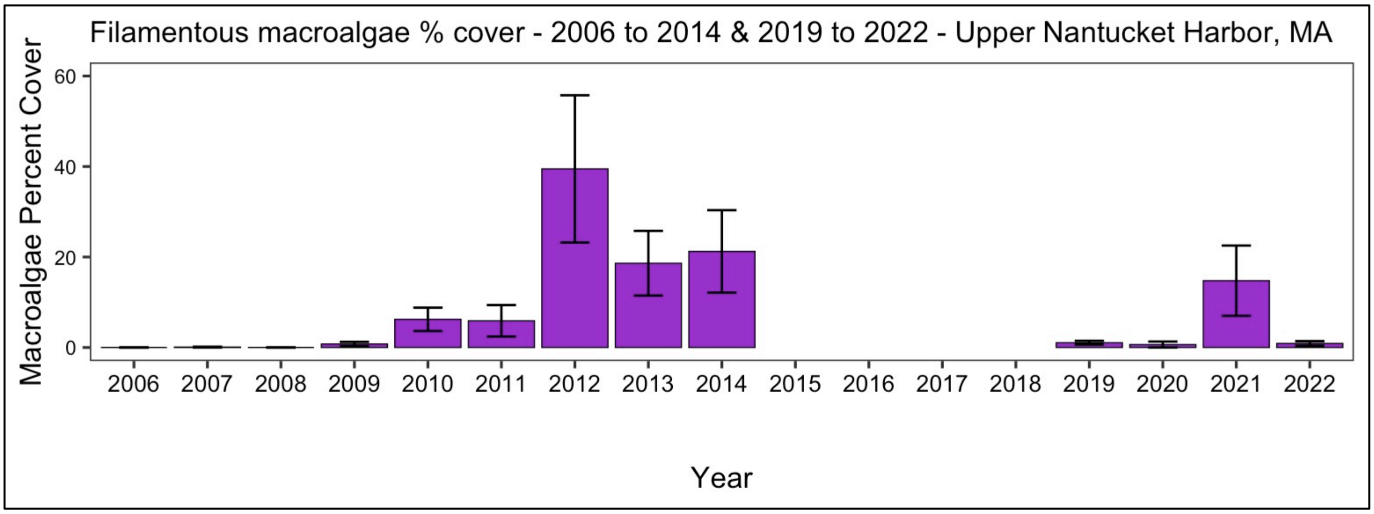

Figure 42. Averages of filamentous macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the upper region of Nantucket Harbor, MA, out of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 (Figure 8).

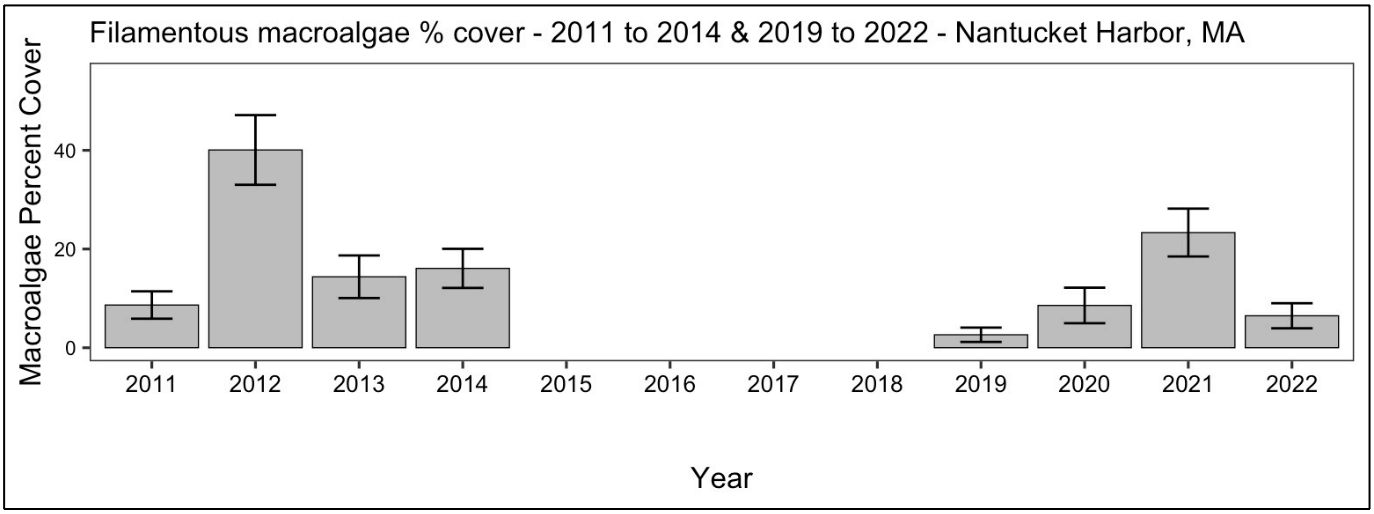

Figure 43. Averages of filamentous macroalgae percent cover by year ± standard error of the mean (SE) of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 within Nantucket Harbor, MA (Figure 5).

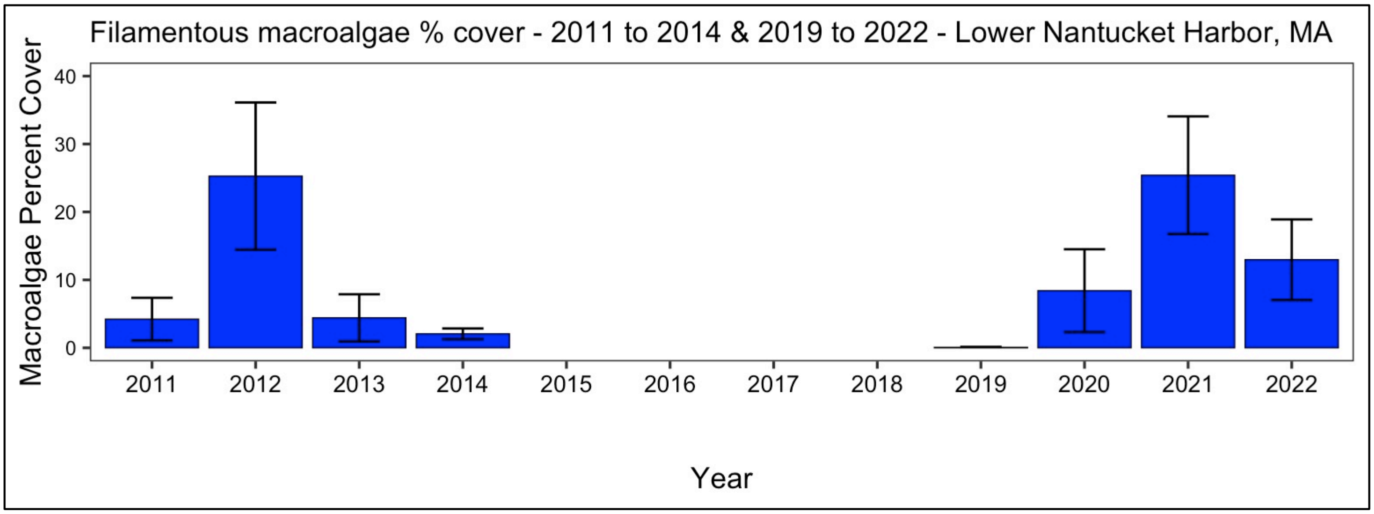

Figure 44. Averages of filamentous macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the lower region of Nantucket Harbor, MA, out of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 10).

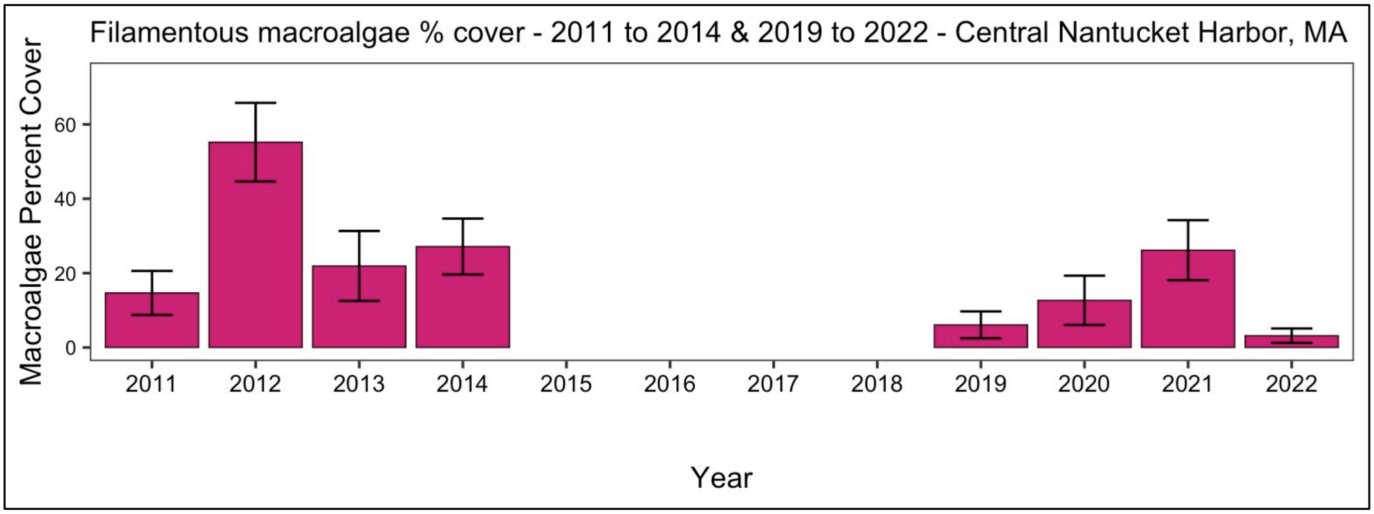

Figure 45. Averages of filamentous macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the central region of Nantucket Harbor, MA, out of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 10).

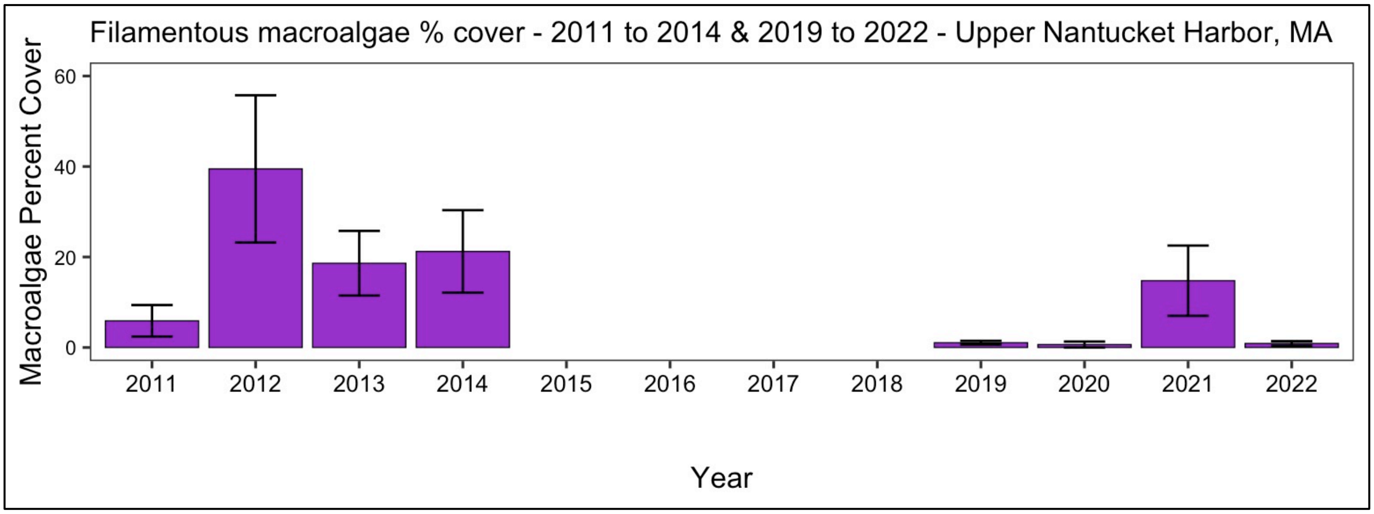

Figure 46. Averages of filamentous macroalgae percent cover by year ± standard error of the mean (SE) of the sites confined to the upper region of Nantucket Harbor, MA, out of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 (Figure 10).

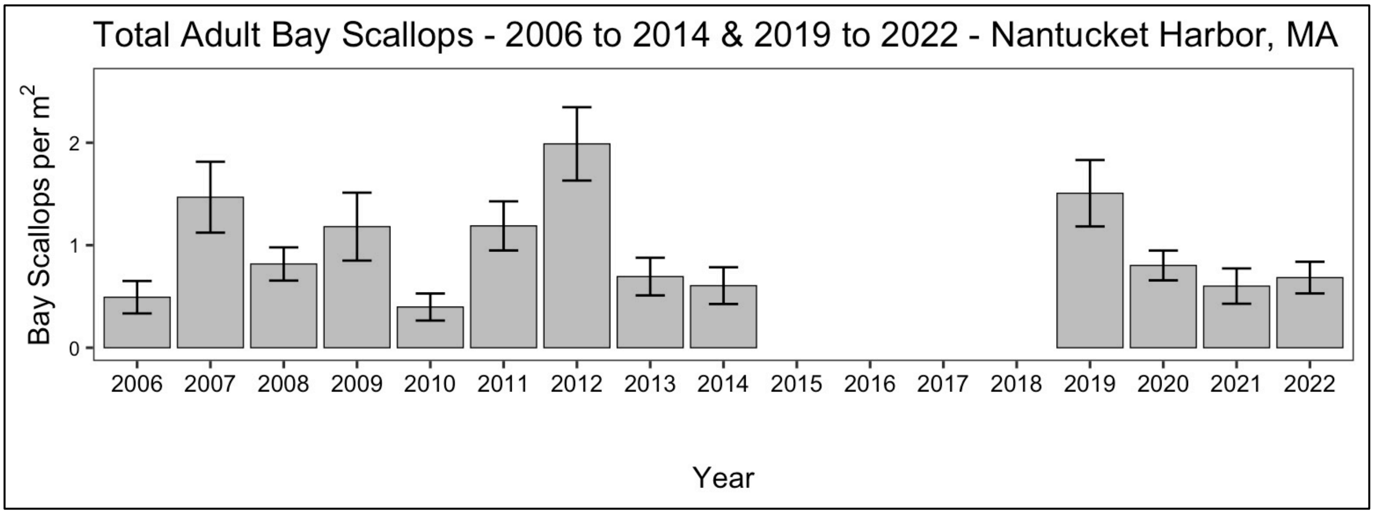

Figure 47. Average density of total bay scallops by year ± standard error of the mean (SE) of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 3). This total abundance pools seed (juvenile) bay scallops, first year nub (N1) bay scallops, classic adult (A1) bay scallops, second year nub (N2) bay scallops, and second year classic adult (A2) bay scallops.

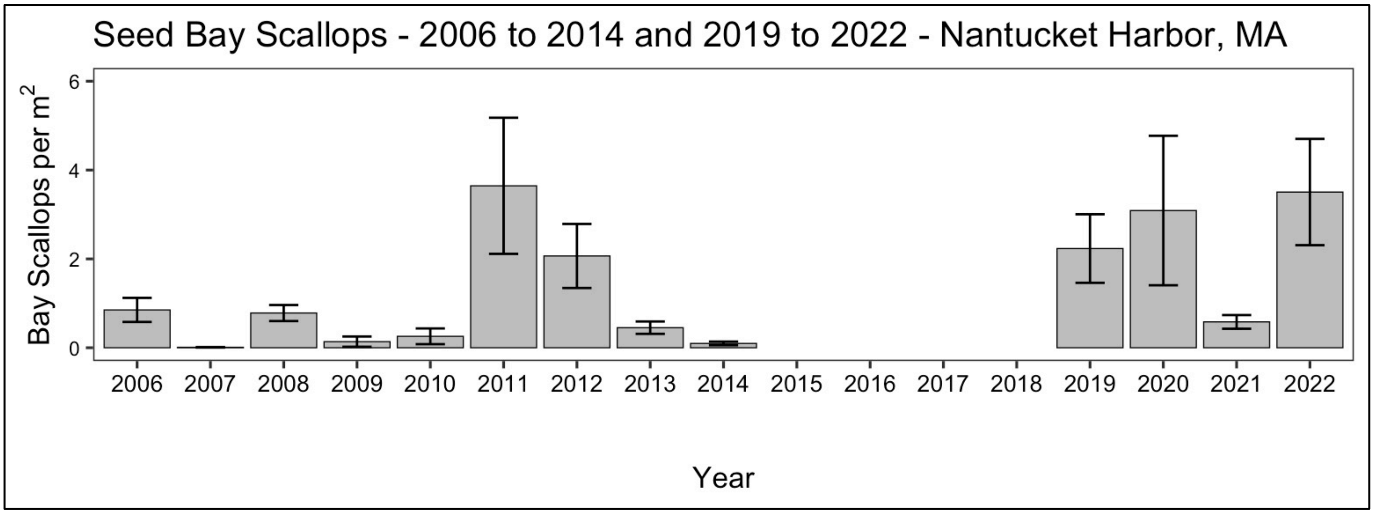

Figure 48. Average density of seed (juvenile) bay scallops by year ± standard error of the mean (SE) of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 3).

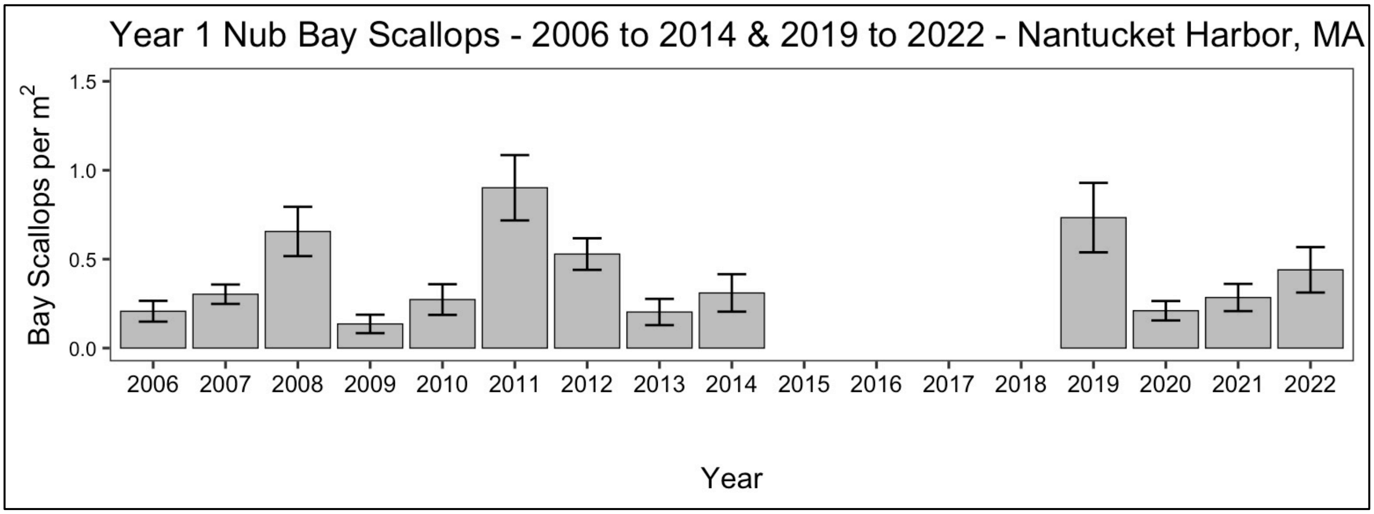

Figure 49. Average density of first year nub (N1) bay scallops by year ± standard error of the mean (SE) of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 3).

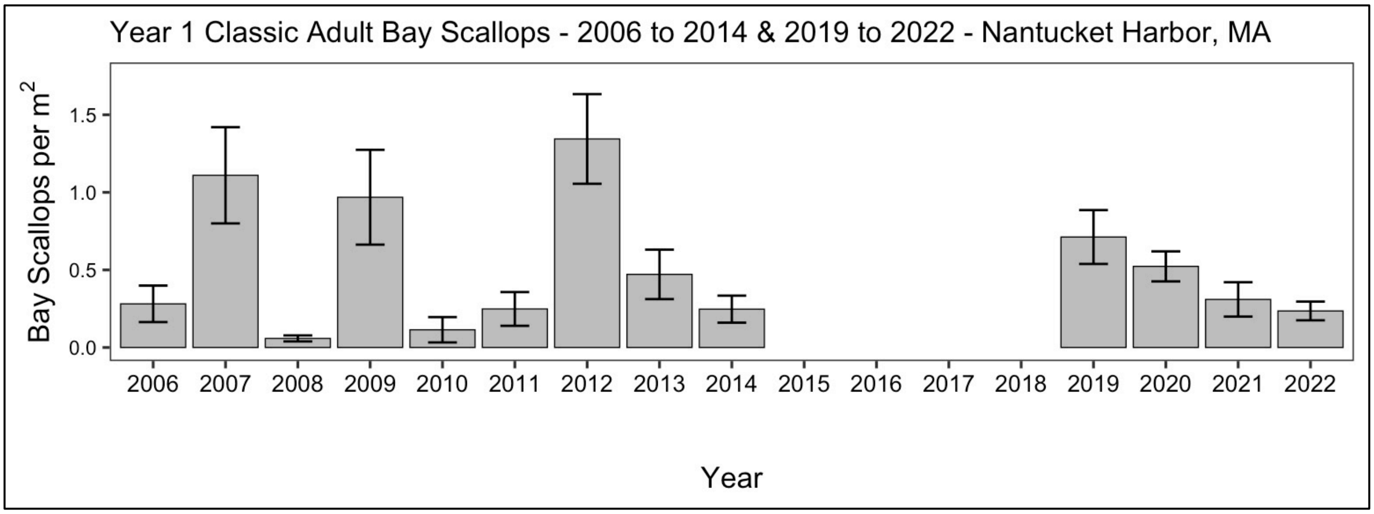

Figure 50. Average density of first year class adult (A1) bay scallops by year ± standard error of the mean (SE) of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 3).

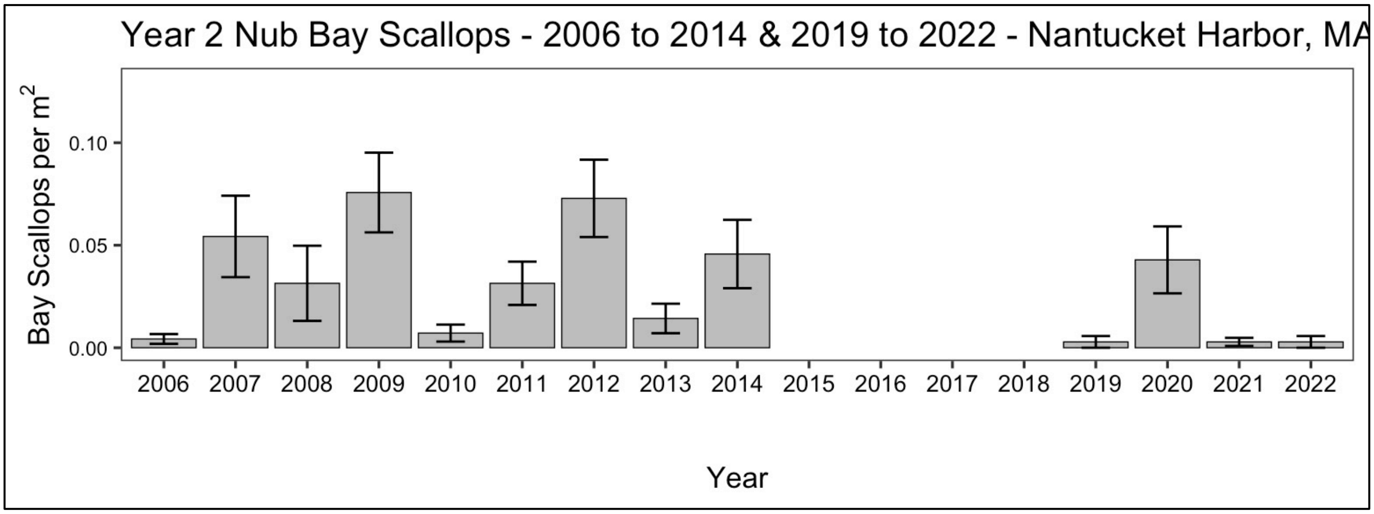

Figure 51. Average density of second year nub (N2) bay scallops by year ± standard error of the mean (SE) of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 3).

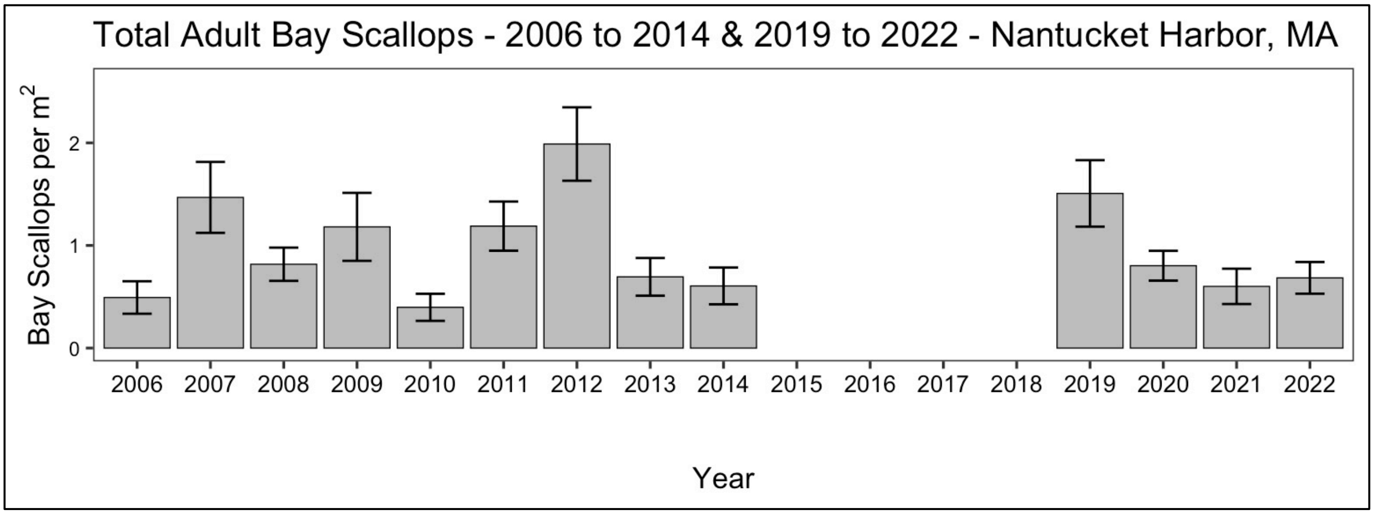

Figure 52. Average density of total adult bay scallops by year ± standard error of the mean (SE) of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 3). This total abundance pools all scallops that are older than seed (juvenile) bay scallops, including first year nub (N1) bay scallops, classic adult (A1) bay scallops, second year nub (N2) bay scallops, and second year classic adult (A2) bay scallops.

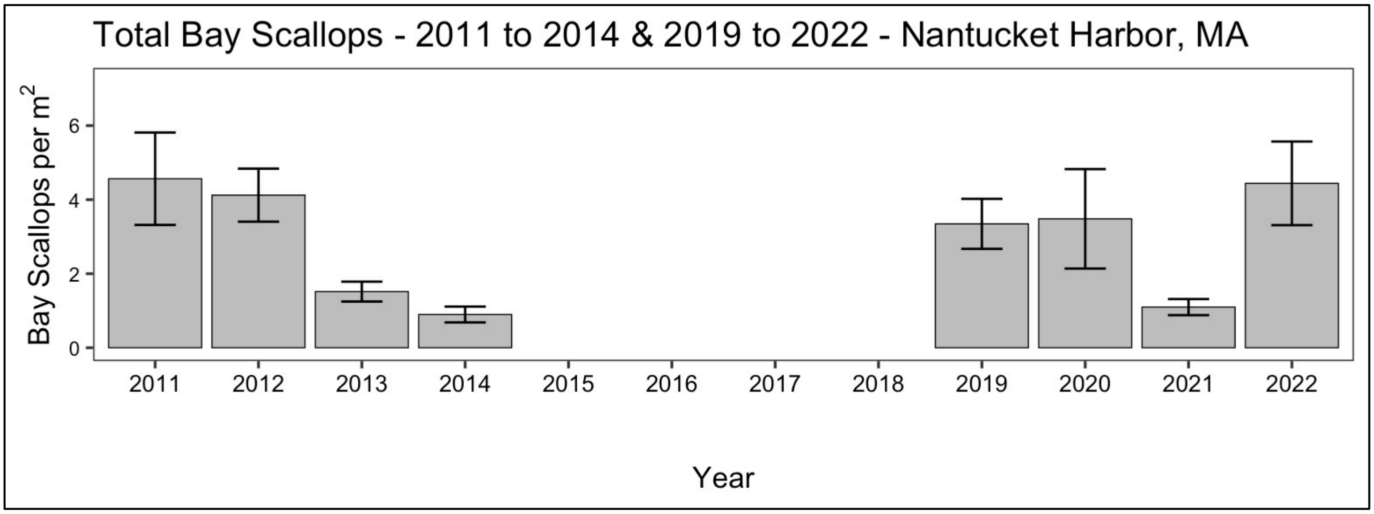

Figure 53. Average density of total bay scallops by year ± standard error of the mean (SE) of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 5). This total abundance pools seed (juvenile) bay scallops, first year nub (N1) bay scallops, classic adult (A1) bay scallops, second year nub (N2) bay scallops, and second year classic adult (A2) bay scallops.

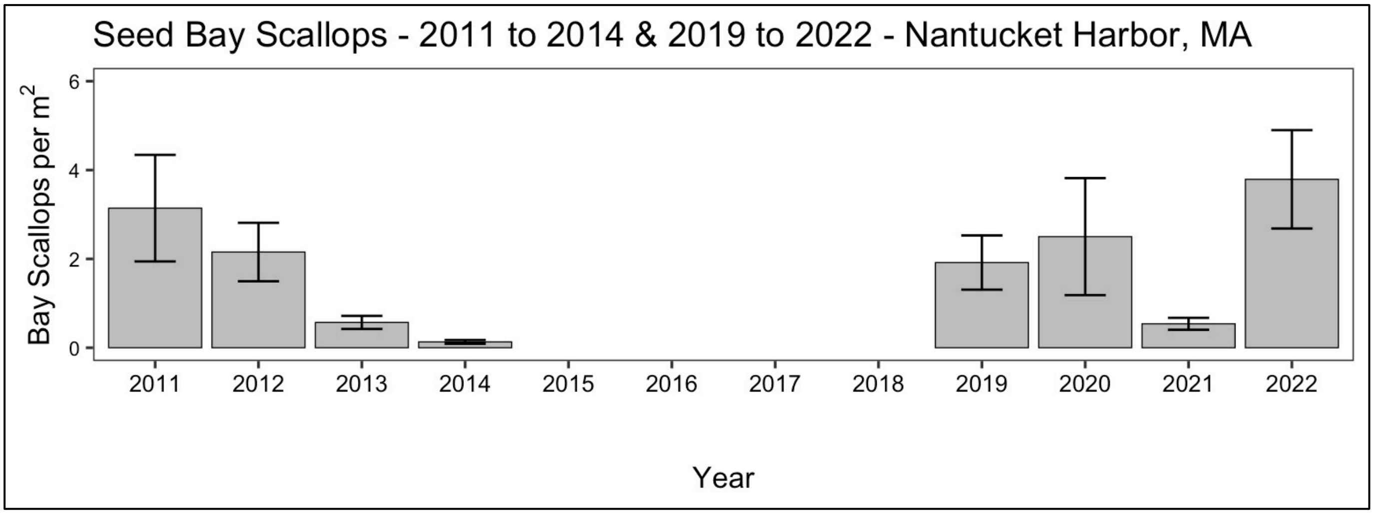

Figure 54. Average density of seed (juvenile) bay scallops by year ± standard error of the mean (SE) of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 5).

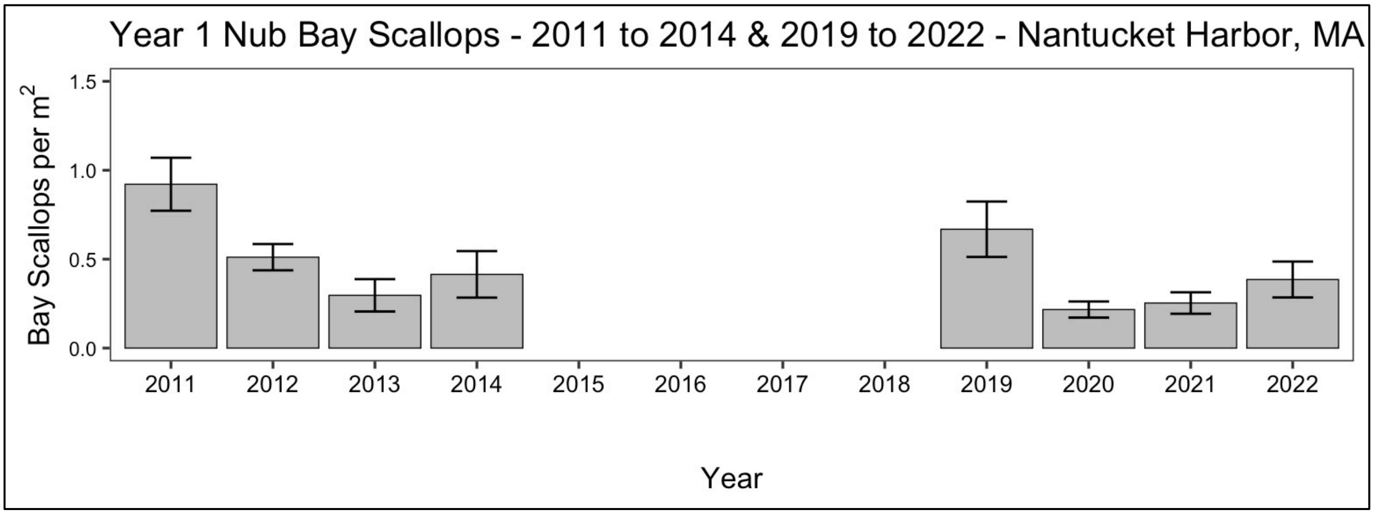

Figure 55. Average density of first year nub (N1) bay scallops by year ± standard error of the mean (SE) of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 5).

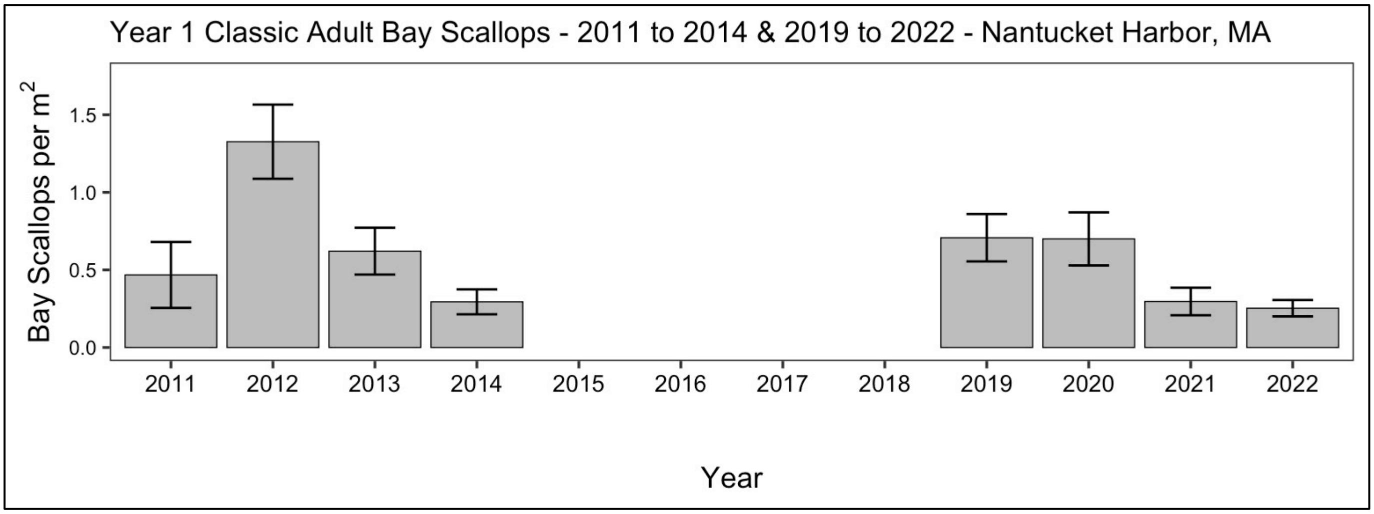

Figure 56. Average density of first year class adult (A1) bay scallops by year ± standard error of the mean (SE) of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 5).

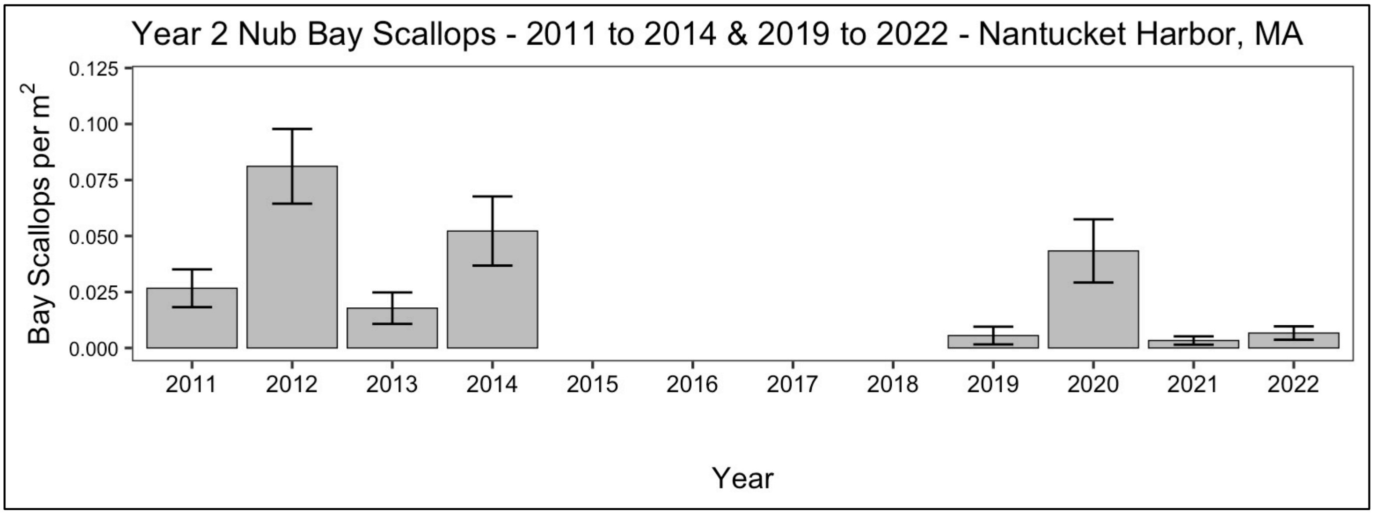

Figure 57. Average density of second year nub (N2) bay scallops by year ± standard error of the mean (SE) of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 5).

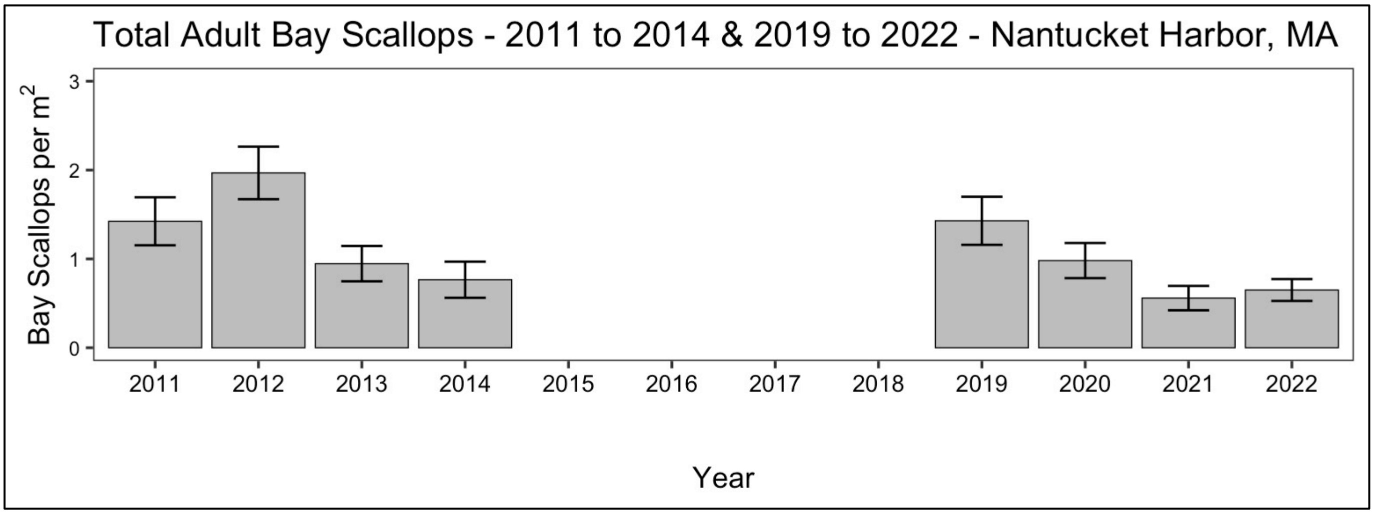

Figure 58. Average density of total adult bay scallops by year ± standard error of the mean (SE) of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 5). This total abundance pools all scallops that are older than seed (juvenile) bay scallops, including first year nub (N1) bay scallops, classic adult (A1) bay scallops, second year nub (N2) bay scallops, and second year classic adult (A2) bay scallops.

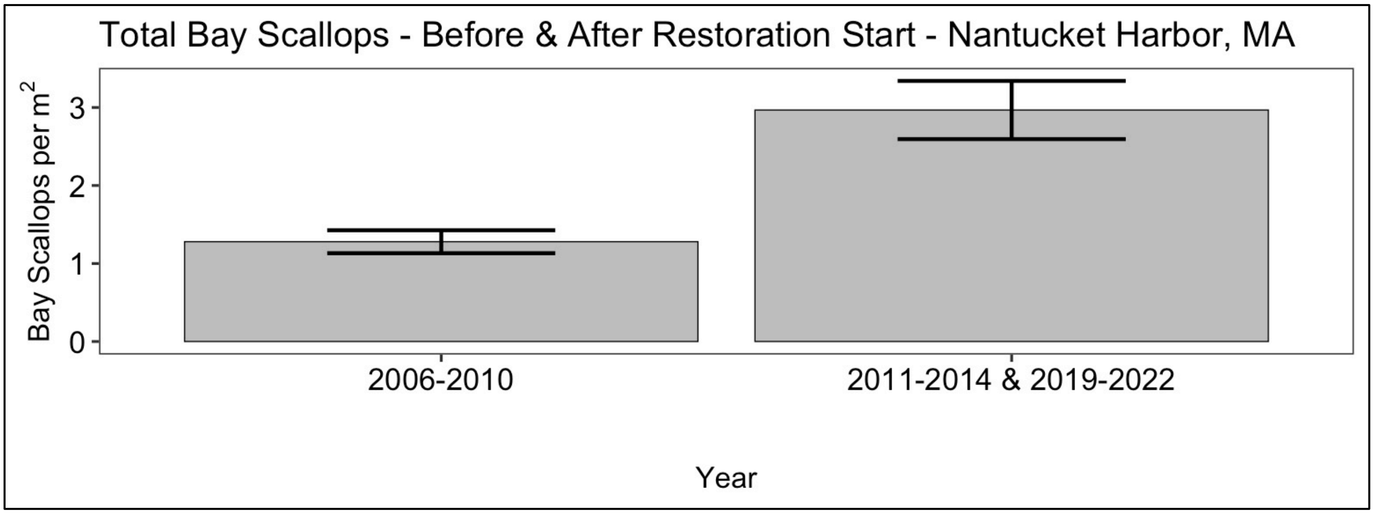

Figure 59. Averages of the density of total scallops pooled for the 5 years of data prior to the beginning of the start of bay scallop restoration efforts (2006-2010) and for the 7 years of data after bay scallop restoration efforts had been initiated by Tara Riley of the Town of Nantucket Natural Resources Department (2011-2014 and 2019-2022) using the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 ± standard error of the mean (SE) (Figure 3).

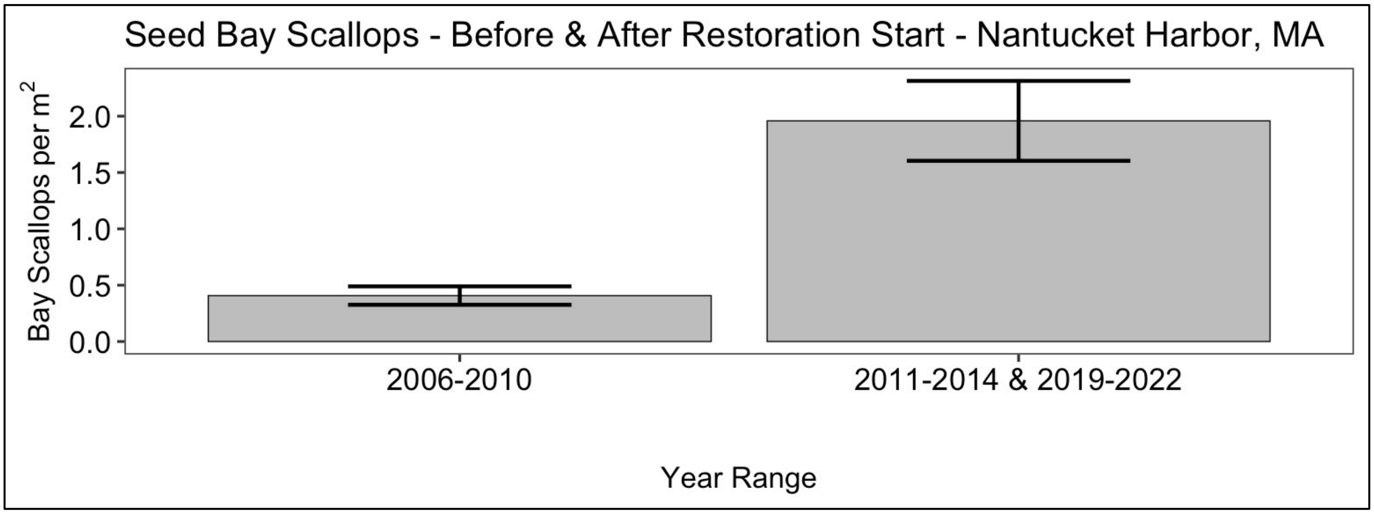

Figure 60. Averages of the density of seed scallops pooled for the 5 years of data prior to the beginning of the start of bay scallop restoration efforts (2006-2010) and for the 7 years of data after bay scallop restoration efforts had been initiated by Tara Riley of the Town of Nantucket Natural Resources Department (2011-2014 and 2019-2022) using the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 ± standard error of the mean (SE) (Figure 3).

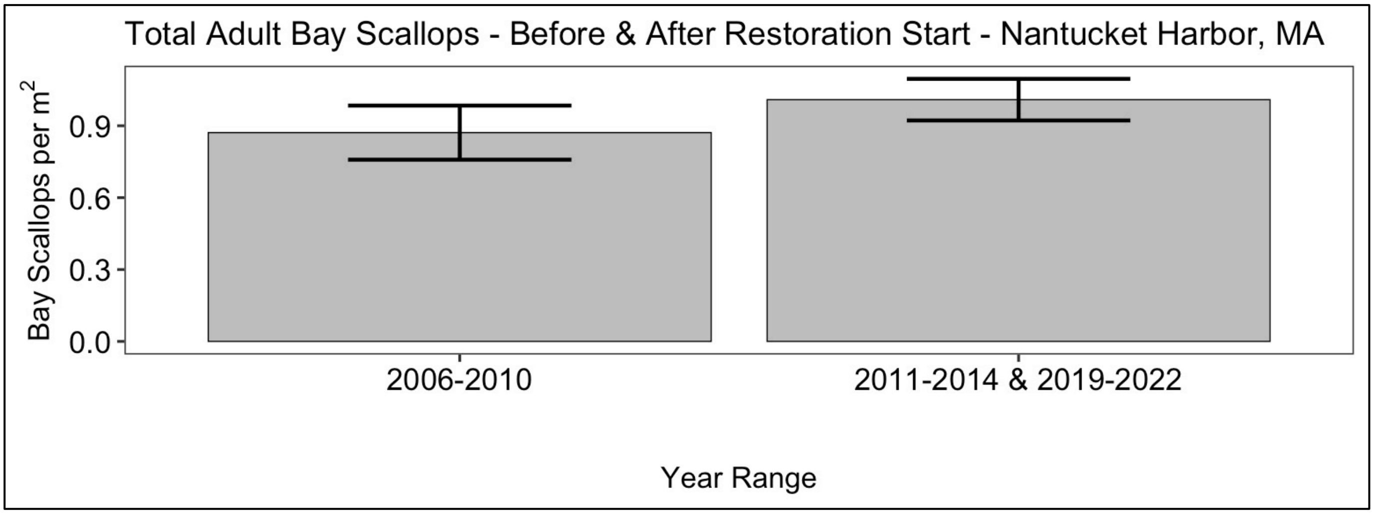

Figure 61. Averages of the density of adult scallops pooled for the 5 years of data prior to the beginning of the start of bay scallop restoration efforts (2006-2010) and for the 7 years of data after bay scallop restoration efforts had been initiated by Tara Riley of the Town of Nantucket Natural Resources Department (2011-2014 and 2019-2022) using the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 ± standard error of the mean (SE) (Figure 3). This total abundance pools all scallops that are older than seed (juvenile) bay scallops, including first year nub (N1) bay scallops, classic adult (A1) bay scallops, second year nub (N2) bay scallops, and second year classic adult (A2) bay scallops.

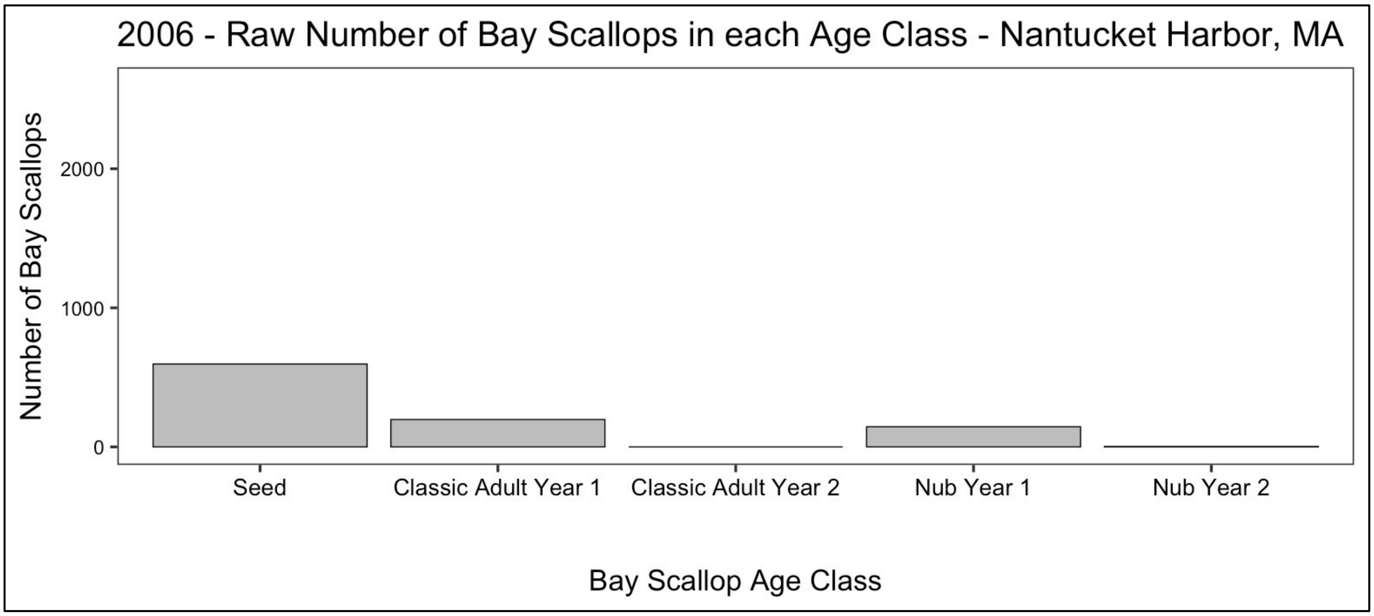

Figure 62. Age distribution of bay scallops during 2006 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

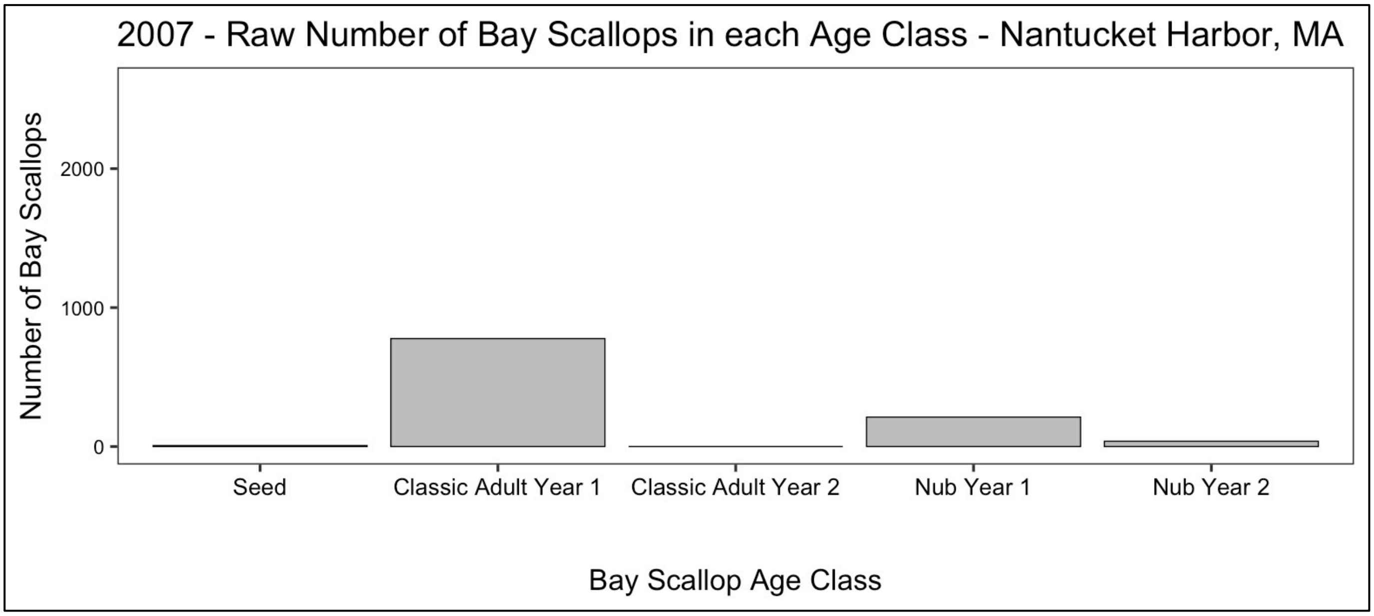

Figure 63. Age distribution of bay scallops during 2007 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

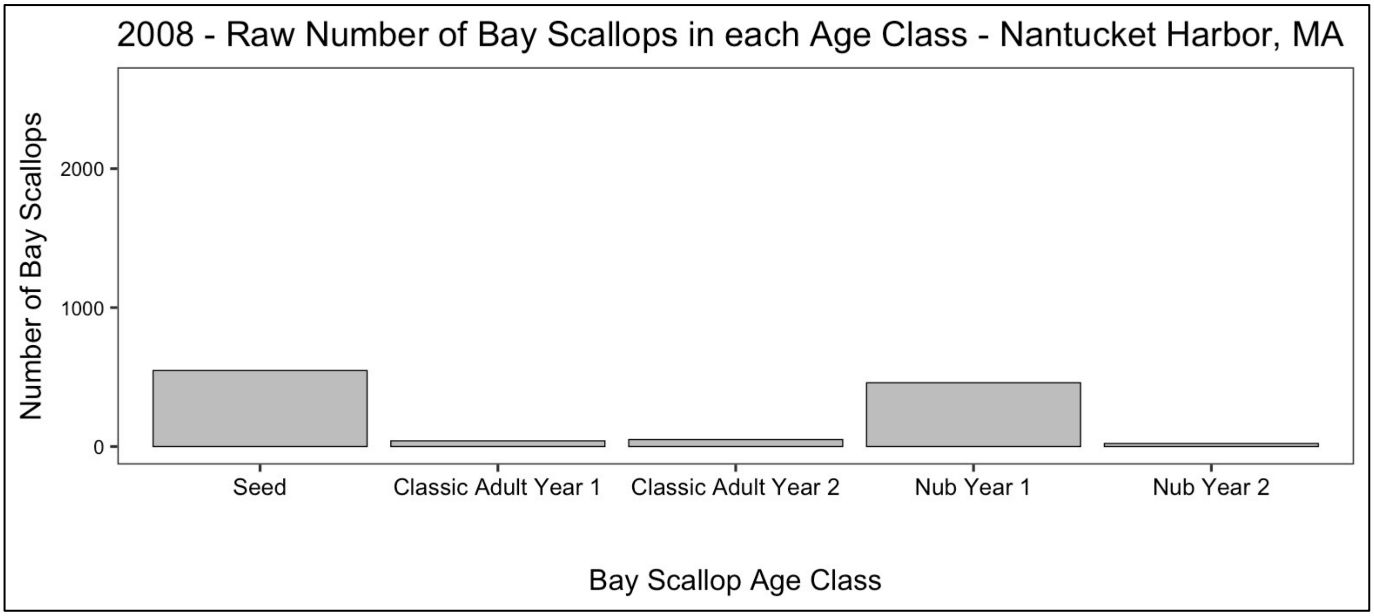

Figure 64. Age distribution of bay scallops during 2008 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

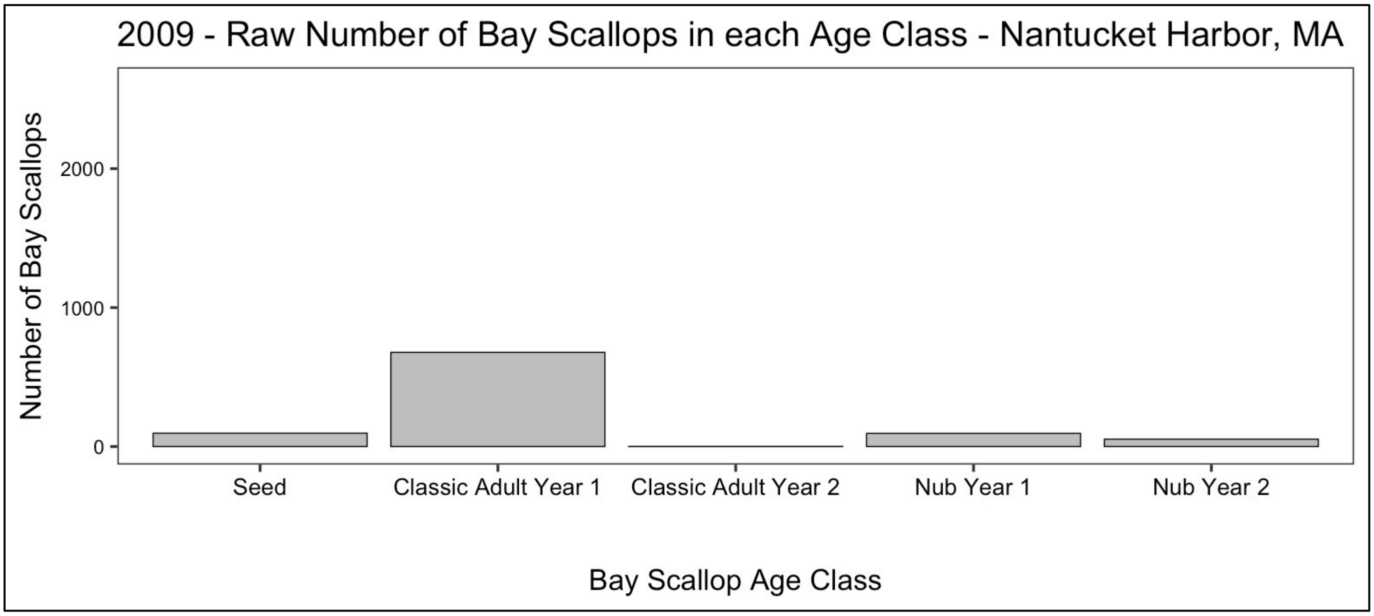

Figure 65. Age distribution of bay scallops during 2009 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

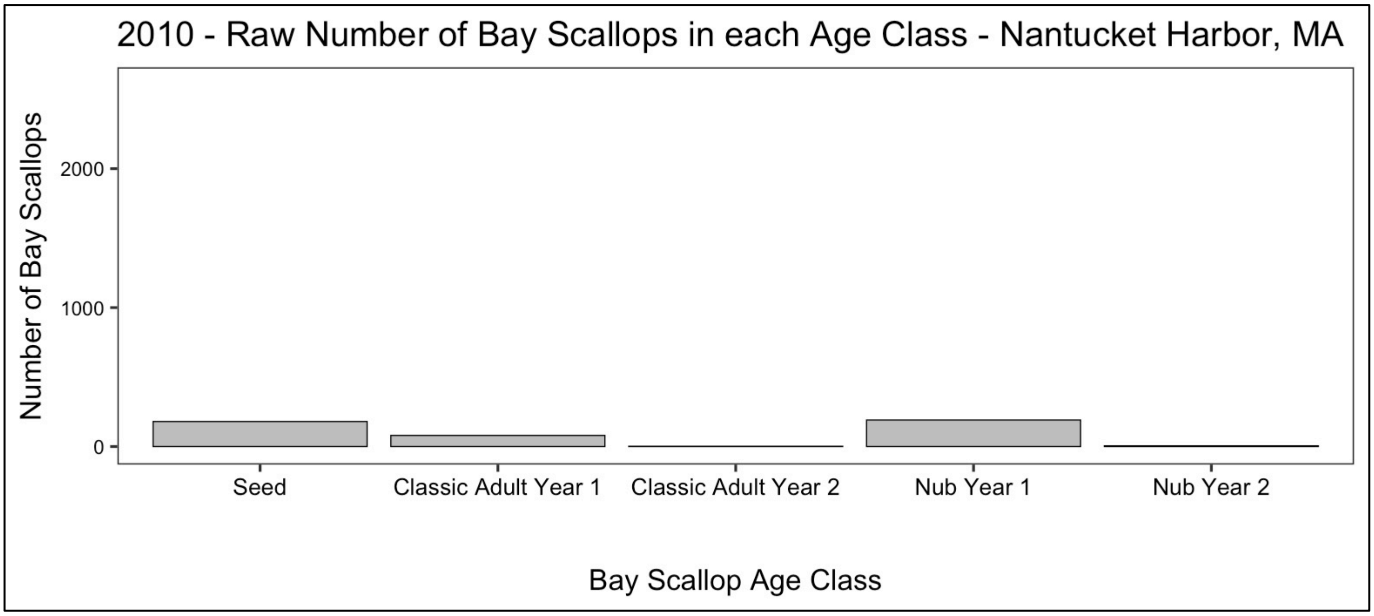

Figure 66. Age distribution of bay scallops during 2010 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

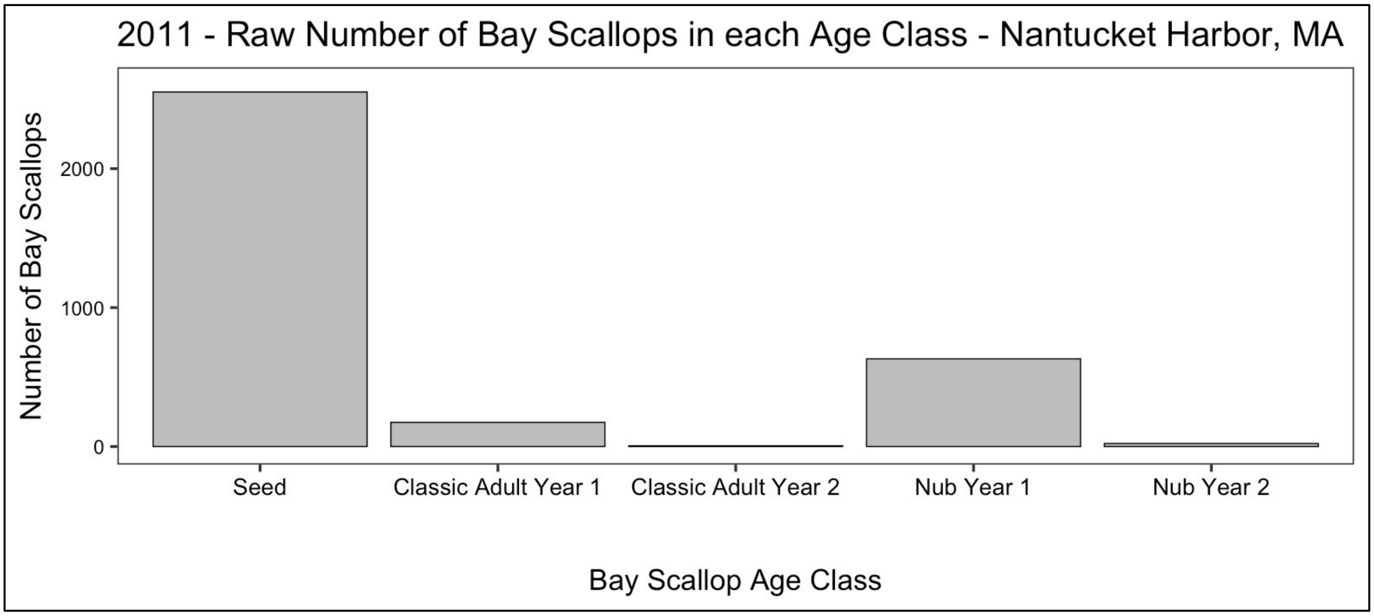

Figure 67. Age distribution of bay scallops during 2011 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

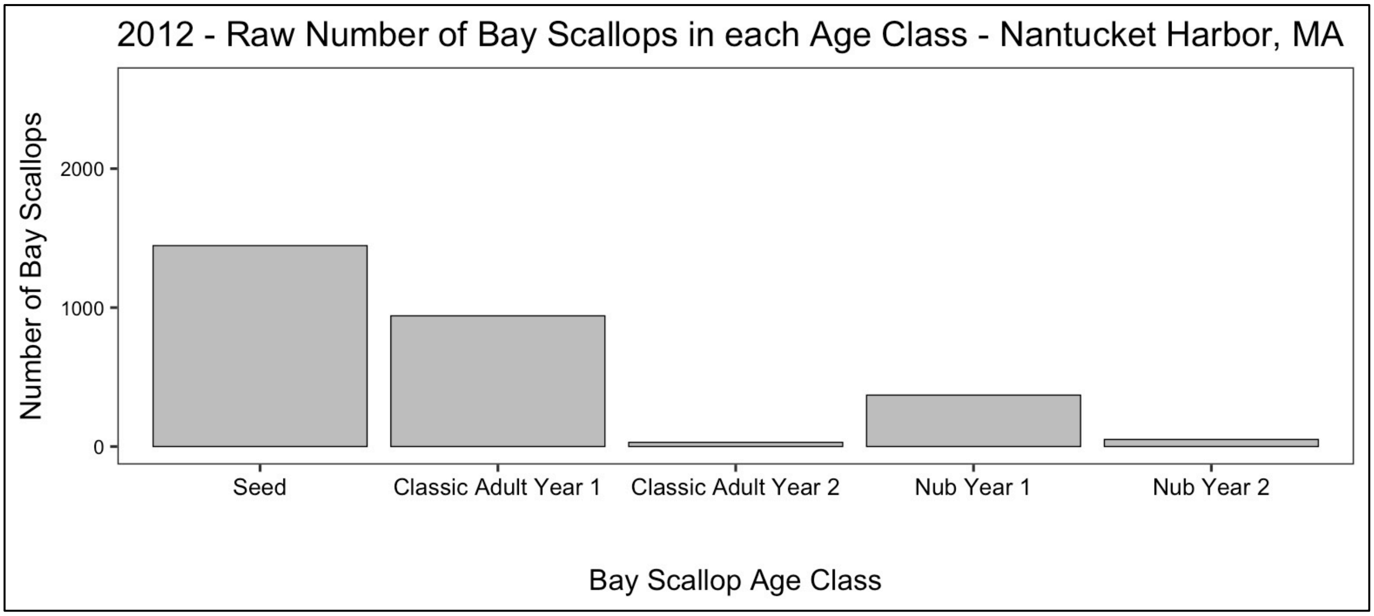

Figure 68. Age distribution of bay scallops during 2012 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

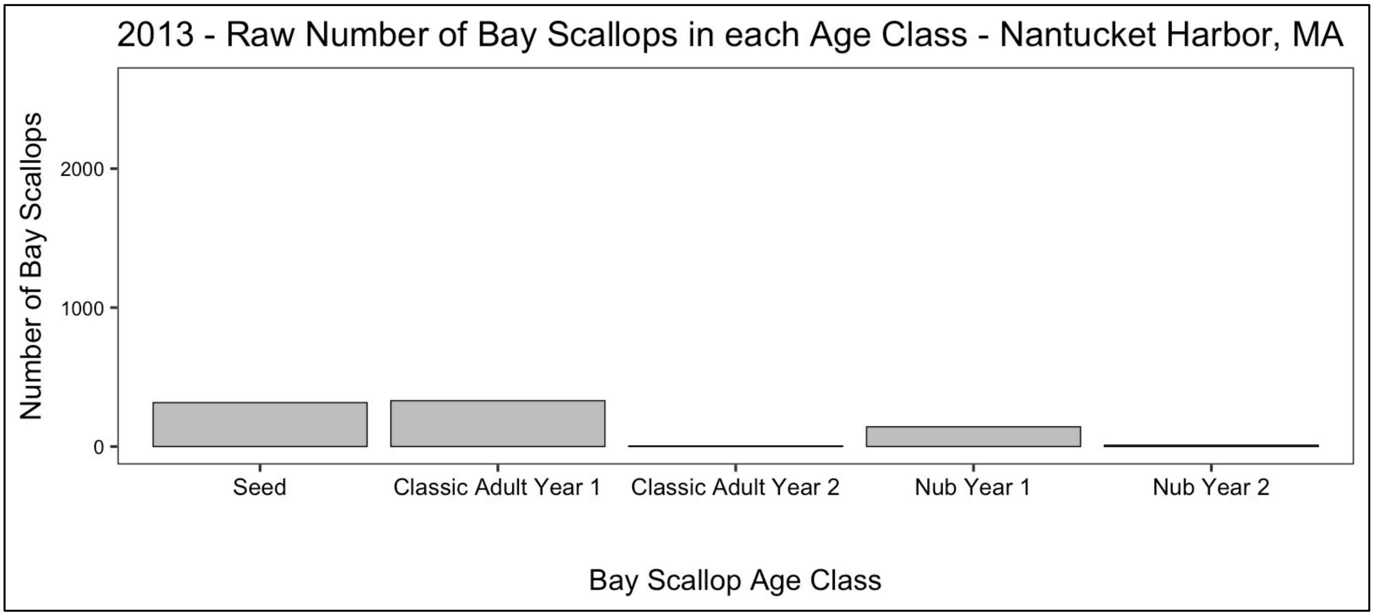

Figure 69. Age distribution of bay scallops during 2013 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

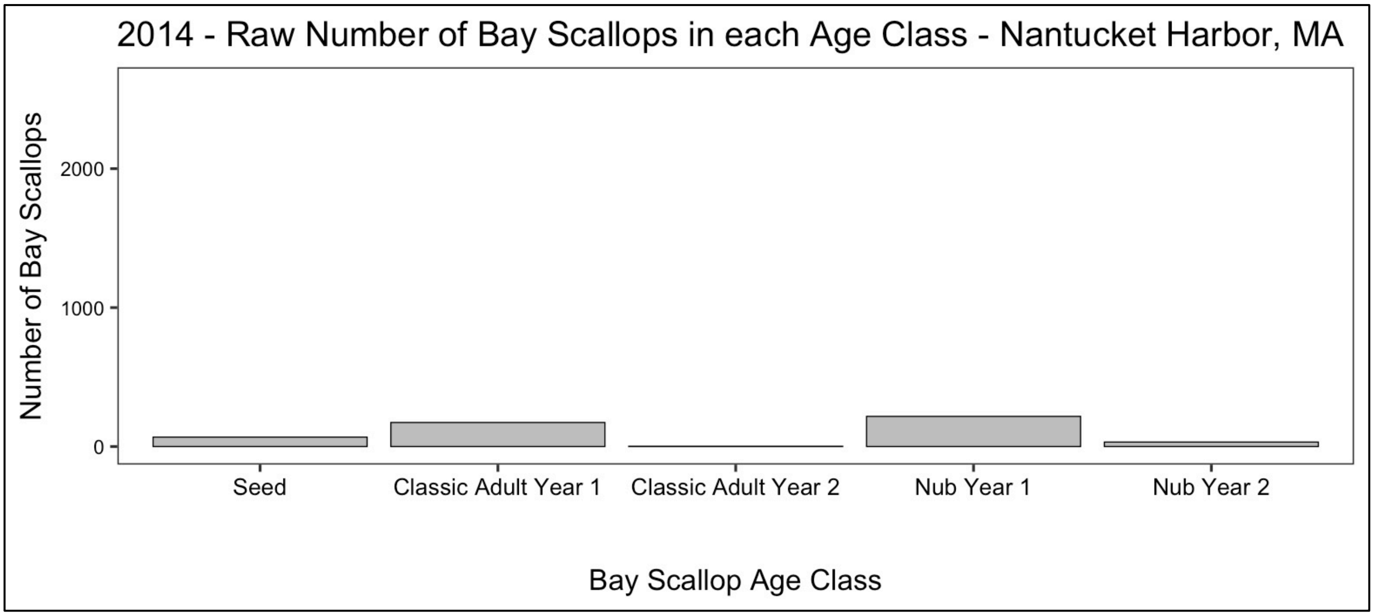

Figure 70. Age distribution of bay scallops during 2014 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

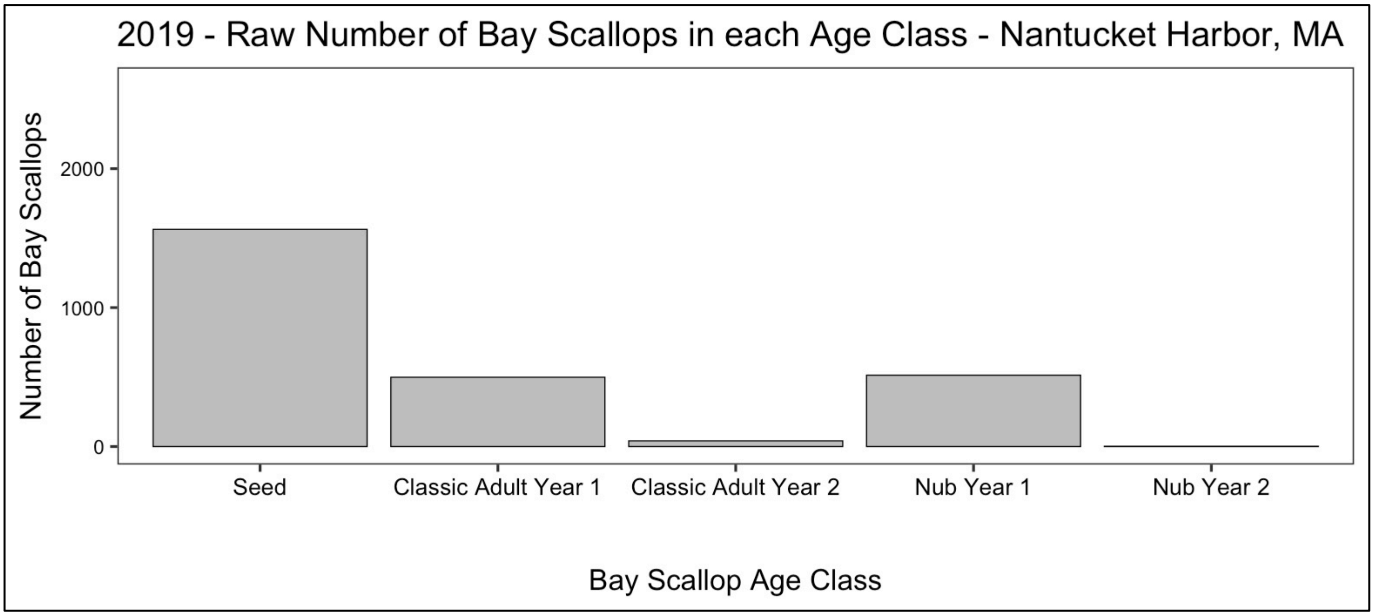

Figure 71. Age distribution of bay scallops during 2019 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

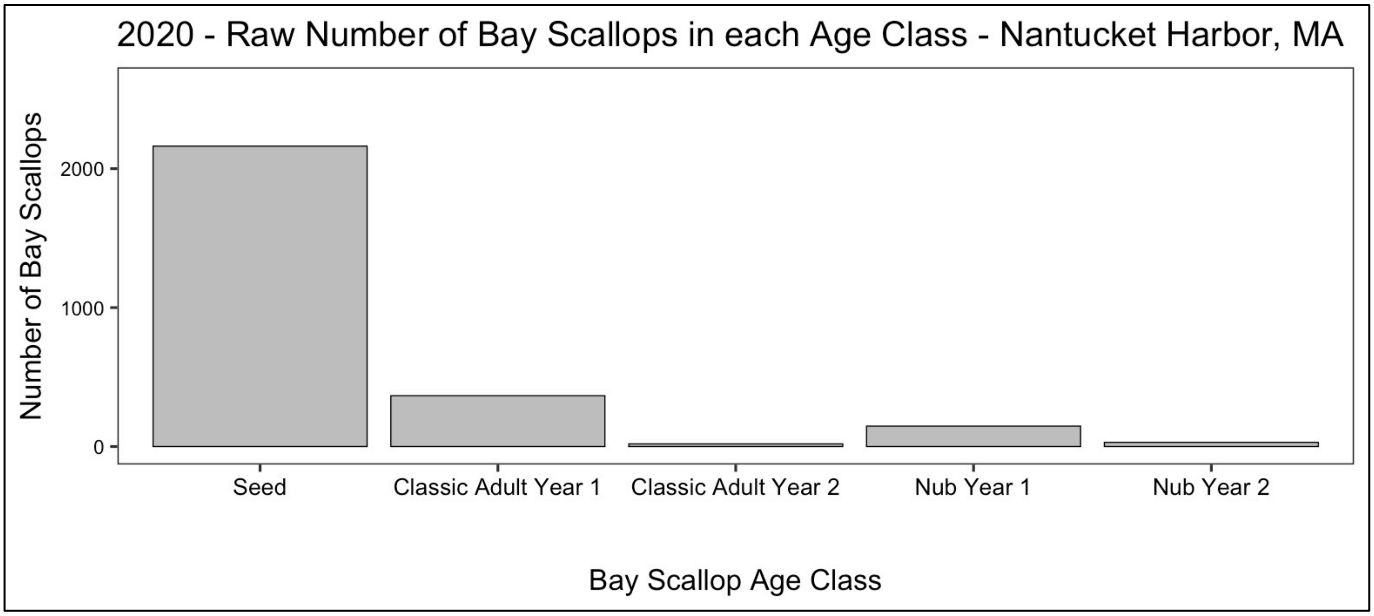

Figure 72. Age distribution of bay scallops during 2020 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

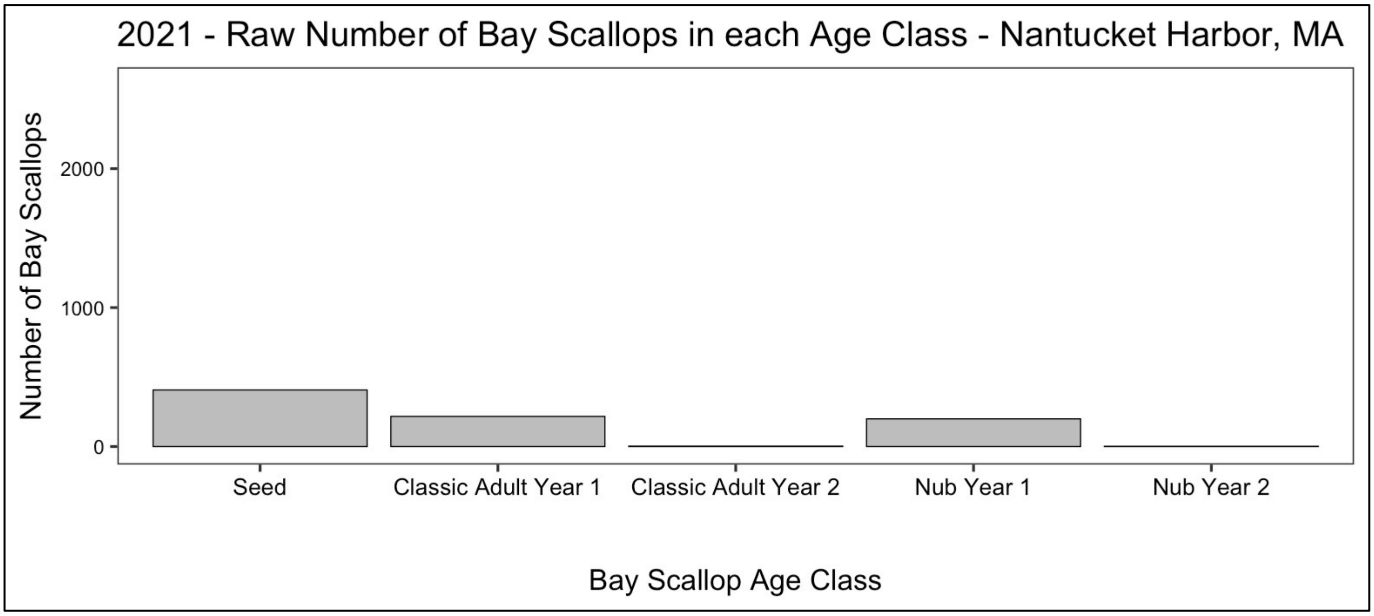

Figure 73. Age distribution of bay scallops during 2021 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

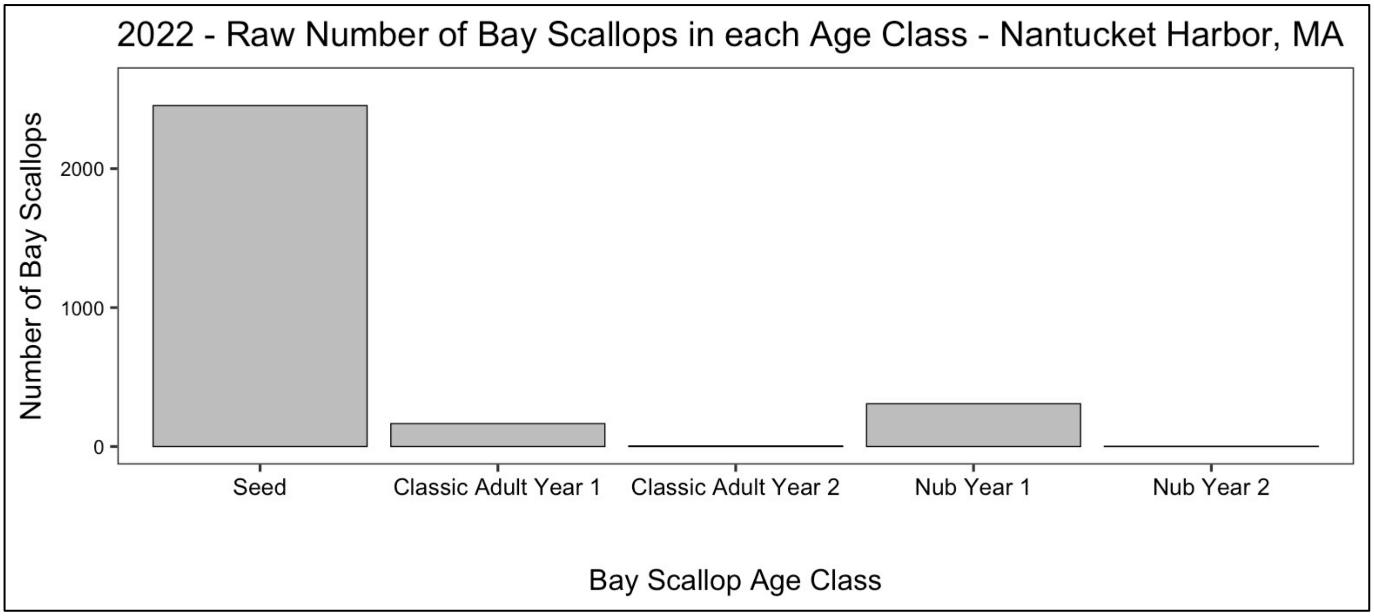

Figure 74. Age distribution of bay scallops during 2022 in terms of the raw number of bay scallops found at 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor (Figure 3).

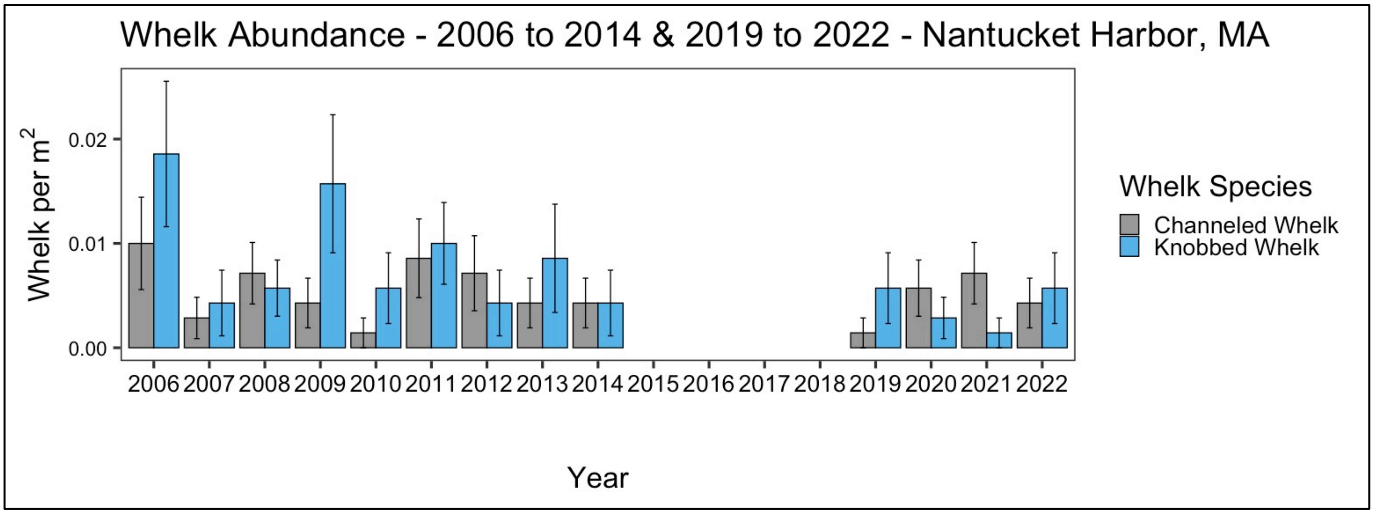

Figure 75. Average density of channeled whelk and knobbed whelk by year ± standard error of the mean (SE) of the 28 sites that were surveyed consistently from 2006 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 3).

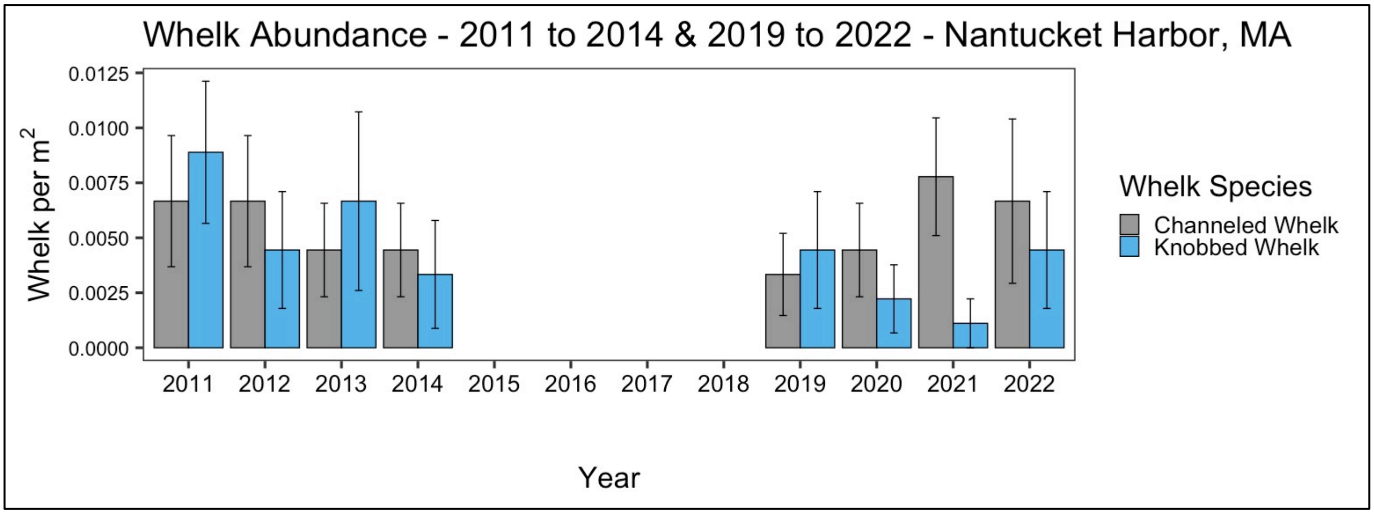

Figure 76. Average density of channeled whelk and knobbed whelk by year ± standard error of the mean (SE) of the 36 sites that were surveyed consistently from 2011 through 2014 and 2019 through 2022 throughout Nantucket Harbor, MA (Figure 5).

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