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Town of Essex

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Summary Conclusions:

Our review of the thermal habitat requirements of the eastern oyster and the blue mussel suggests that blue mussels, not surprisingly, are adapted to substantially colder environments. This difference in thermal performance, coupled with the warming that the Gulf of Maine has already experienced, could explain anecdotal observations suggesting a decline in historical mussel bed habitat in eastern Massachusetts and throughout the Gulf of Maine (Sorte et al. 2017) as well as observations that oysters are becoming more prevalent in this region. Indeed, the inshore water temperature monitoring at the Plum Island Sound LTER demonstrated that sea-surface temperature is already occasionally surpassing the 25 ºC upper threshold for blue mussels. Moreover, this phenomenon will likely increase in the coming years, if climate scenarios projecting warming waters in the Gulf of Maine are accurate. Thus, integrating blue mussels into living shorelines restoration designs may not be a viable long-term strategy.

The eastern oyster, which is at the northern end of its range in the Gulf of Maine, is more common in the Mid-Atlantic and South Atlantic Bights in warmer waters (Byers et al. 2014). The studies investigating the thermal habitat preferences of the eastern oyster suggested that seed oysters are slightly more vulnerable to colder waters than spat oysters, and much more vulnerable than adult oysters, which performed well above 15 ºC. However, given the trajectory of warming waters throughout the Northwest Atlantic, especially in the Gulf of Maine, the eastern oyster may offer a potential option for living shorelines and coastal habitat restoration opportunities in the region.

As the science and art of coastal zone protection and restoration advances in the wake of climate change and sea-level rise, the accumulated tools that develop will make up a set of resilient strategies to assist communities during these changing times. Based on the current literature, it is our opinion that constructed living shorelines should be included in this set of tools. Coastal shorelines, in particular, are increasingly susceptible to storm damage and flooding from rising seas. Essex Bay is an area that has become dramatically impacted by these changing conditions. Specifically, it is already experiencing infilling of tidal rivers with sediment from eroded marsh habitats, loss of shellfish habitat, and the loss of associated ecological services such as diminished flood storage and storm damage prevention. Furthermore, the loss of portions of the Great Marsh have resulted in significant impairment of the estuary and the loss of ecological services provided by this unique ecosystem. Utilizing green technology such as living shorelines in concert with thin layer deposition, eel grass, oyster and blue mussel beds, among other strategies, could serve to ameliorate the impacts of the adverse impacts of climate change and sea-level rise.

Introduction:

Coastal ecosystems have supported human societies for millennia and provide a diverse array of valuable ecosystem services (Beck et al. 2001, Jackson 2001, Barbier et al. 2011). These ecosystems are comprised of many valuable habitats, including salt marshes, shellfish reefs, coral reefs, mangroves, and seagrass meadows. Collectively, they provide nursery habitat for economically valuable fishes, remove excess nitrogen, bury carbon, reduce coastal flooding, and reduce shoreline erosion (Thayer et al. 1978, Peterson and Lipcius 2003, Shepard et al. 2011). Thus, these habitats are critical to the long-term health of coastal ecosystems and the human societies that they support.

Human population growth in coastal regions globally over the past century unfortunately has led to ecosystem degradation (Vitousek 1997, Halpern et al. 2008, Barbier et al. 2011). For instance, the extent of oyster reef, seagrass, and salt marsh habitats in temperate and tropical estuaries has been reduced to a small fraction of the area these ecosystems occupied historically, with much of the remaining habitat having been degraded (Lotze et al. 2006, Wilkinson 2008, Waycott et al. 2009, Beck et al. 2011, Grabowski et al. 2012, zu Ermgassen et al. 2012). Of further concern is that these impacts will be exacerbated by sea-level rise (SLR) and other climate-related hazards. Indeed, adopting climate adaptation strategies will likely be necessary to conserve and restore these critical habitats and ultimately reduce the vulnerability of coastal communities.

Salt marshes are critical because they protect coastal communities from coastal hazards. Shepherd et al. (2011) reviewed evidence to determine the conditions under which salt marshes attenuate waves, stabilize shorelines, and attenuate floodwaters. They found that vegetation density, biomass production, and marsh size were all important factors that were positively correlated with both wave attenuation and shoreline stabilization. Meanwhile, Narayan et al. (2017) found that coastal wetlands in the Northeastern U.S. reduced flood damage during Hurricane Sandy in 2015 and in Ocean County, NY annually. Estimates of flood reduction averaged around 16%, and these benefits were even greater at lower elevations. These studies indicate that salt marsh conservation and restoration are important climate adaptation strategies for coastal areas (EPA 2009).

In temperate estuaries, shellfish habitat and tidal salt marshes often coexist directly adjacent to each other, with oyster reefs and mussel beds lining the shoreward edge of marshes. Several studies have found that fringing shellfish habitat reduces erosion of adjacent salt marshes (Meyer et al. 1997, Piazza et al. 2005, Scyphers et al. 2011), perhaps explaining the emergence throughout the U.S. and elsewhere of “living shorelines” restoration projects that promote shellfish and salt marsh habitat restoration together (Gittman et al. 2016a). Given the many factors that are contributing to erosion and degradation of tidal salt marshes are projected to increase due to SLR, efforts to stabilize the edges may be necessary to avoid future losses and enhance the resilience of salt marshes in areas already experiencing salt marsh erosion. Shellfish offer a nature-based approach to promoting the stabilization and accretion of critically important salt marshes.

In this study, we evaluated the degree to which eastern oysters and blue mussels are viable options for use in living shoreline restoration projects in Essex, MA. First, we reviewed the thermal habitat requirements for both of these bivalve species throughout their life histories. Next, we evaluated available historical data on seawater temperature in coastal embayments and estuaries located in northern Massachusetts. In addition, we searched for available seawater temperature forecasts for this region so that we could examine whether the projected warming is still suitable for either bivalve species over the next several decades. Finally, we reviewed living shoreline designs and materials to explore potential options for use in coastal restoration projects in northern Massachusetts.

Methods:

a. Review of thermal habitat requirements for bivalve life-history phases

To determine the thermal habitat requirements for the eastern oyster and blue mussel, we conducted a literature review using Google Scholar and Northeastern University's Scholar OneSearch database. Specifically, we used the following search terms: organism ("*Crassostrea virginica*" OR "*Mytilus edulis*") AND temperature response (temperature limits OR thermal tolerance OR thermotolerance). We retained studies that manipulated temperature and examined the performance of either bivalve species. When possible, the temperature ranges for specific life stages were clarified. For eastern oysters, life-history stages were parsed to larval, spat (<25mm in shell height), seed (25 to 75mm in shell height), and adult (>75mm in shell height). For blue mussels, this included veligers and adults.

b. Description of seawater temperature history in coastal northern Massachusetts

Available sources of seawater temperature data in northeastern Massachusetts were identified and reviewed. The closest and most relevant source of seawater temperature data to Essex, Massachusetts has been collected at the Plum Island Ecosystems' Long Term Ecological Research Program. They established a sampling site at Ipswich Bay Yacht Club (42.708995, -70.796577) in 2000. Located at the mouth of Plum Island Sound, this inshore sampling location is in the town of Ipswich, 8.5 kilometers north of Essex Town Hall (Figure 1).

Water quality conditions were collected from 2000 to 2018 via a YSI 6600 sonde attached to the bottom of the club's pier. Data, including water temperature and depth, were collected at 15-minute intervals during this time period. Because the location is shallow enough to freeze over during the winter, the sonde is placed into the water in the spring, often around April, and is usually removed by mid- to late- November.

Because the sonde is attached to a fixed structure in the intertidal zone, the 19-year dataset was cleaned by removing any timestamps with depth readings of 0 meters or less. It was also cleaned of any timestamps that were missing temperature readings. If this resulted in a substantial portion of the day being deleted, the whole date was removed from the dataset using the following three scenarios: >10%, >25%, or >40% of the data for a particular day were missing.

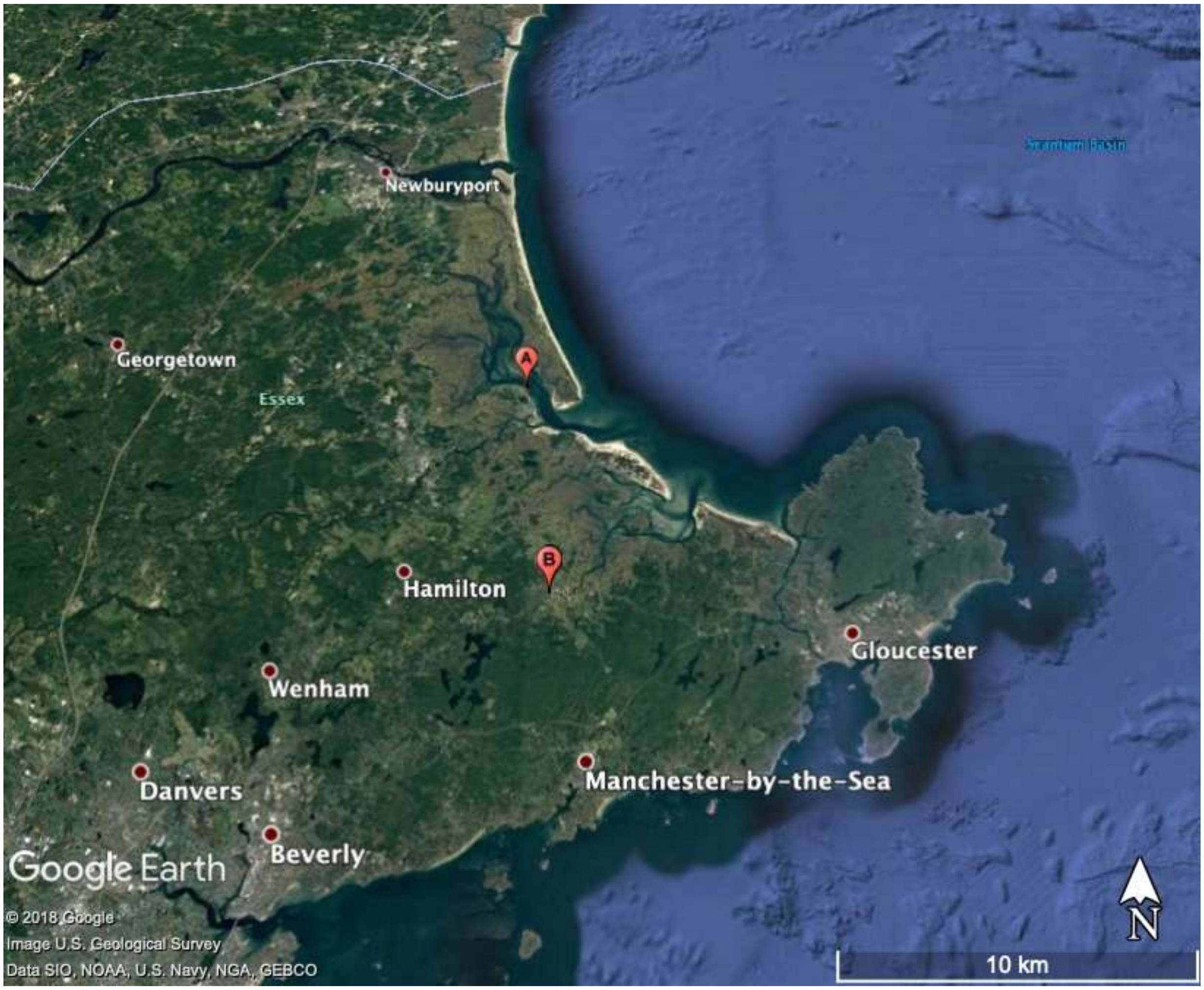


Figure 1: A) Ipswich Bay Yacht Club and B) Essex Town Hall

c. Seawater temperature projections for coastal northern Massachusetts under climate change

A second literature review was conducted to examine future sea surface temperature (SST) projections for the Gulf of Maine. Specifically, we were interested in identifying studies that examined changes to sea surface temperature over the next several decades under different CO_2 emissions scenarios to determine if and when coastal Massachusetts may become inhospitable for various life-history phases of either bivalve species.

d. Review of living shoreline design and materials

Finally, a third literature review was conducted using and combining the following search terms in google scholar: living shorelines, eastern oysters, shellfish, managing coastal risk, ecosystem functions of shellfish, rocky intertidal habitat, algal canopies, *Ascophyllum nodosum* (rockweed), and history of shellfish in Massachusetts. Once key journal articles and reports were identified, the references sections of these documents were reviewed, and any valuable references were identified, reviewed, and added to the list of citations. Citations were entered and organized in EndNote (endnote.com) and later reported in the attached list of citations.

Results:

a. Review of thermal habitat requirements for bivalve life-history phases

The literature review of the thermal preferences of oysters and blue mussels yielded 16 papers that studied the temperature thresholds for the various life-history stages of these two bivalve species. In general, these studies indicated that eastern oysters prefer water temperatures between 22 ºC and 30 ºC, but optimal ranges varied highly across life stages (Figure 2A). Larvae experienced optimal growth between 20 ºC to 30 ºC, but we found no other studies that investigated the effects of other temperature ranges on the performance of this life stage (Barnes et al. 2017). Spat, oysters less than 25 mm in shell height, grew the fastest in water temperatures ranging between 27 ºC and 30 ºC, but they also tolerated temperatures as low as 20 ºC (Lowe et al. 2017). Water temperatures below 20 ºC led to reduced growth rates, and temperatures over 30 ºC led to death (Southworth 2017). Seed oysters (25-75 mm in shell height) had maximum growth rates at 27.8 ºC but still grew in temperatures from 22 ºC to 30 ºC (Lowe et al. 2017). They survived in waters as low as 15 ºC and as high as 36 ºC but exhibited declining growth rates. Lastly, adult oysters (>75 mm in shell height) thrived in 15-27 ºC waters. Their growth rates decreased as waters warmed to 30 ºC, and they could not survive in anything warmer than that (Southworth 2017). They also stopped feeding below 5 ºC and suffered high mortality rates when water temperatures decreased beneath this threshold (Barnes et al. 2017).

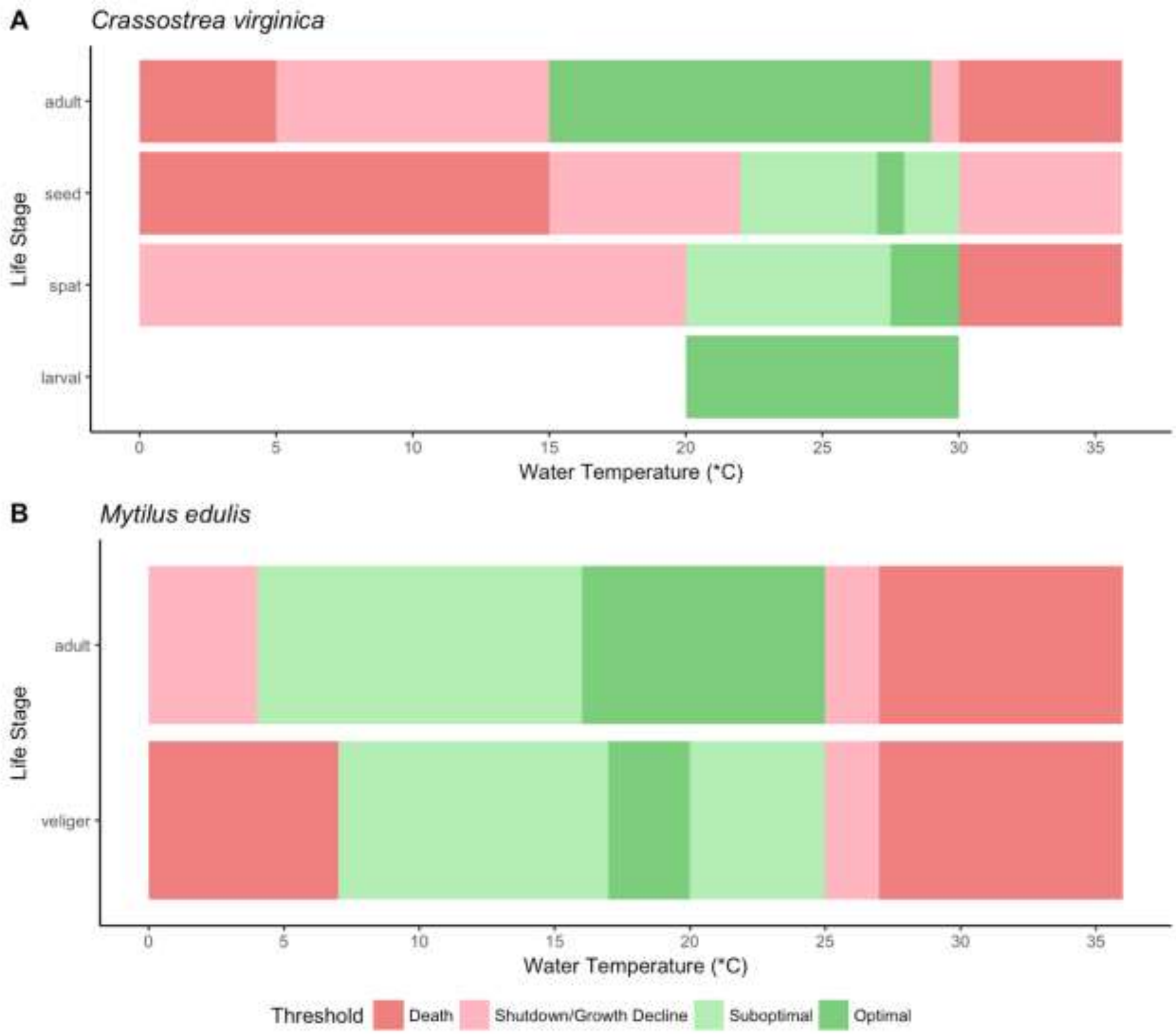


Figure 2: Review of temperature thresholds for A: Crassostrea virginica *and B:* Mytilus edulis *during different life stages.*

In general, blue mussels preferred seawater temperatures between 7 ºC and 25 ºC (Figure 2B). For veligers, optimal growth occurred when seawater was between 17 ºC and 20 ºC, though they could survive temperatures as low as 7 ºC (Rayssac et al. 2010, Beaumont et al. 2004). Optimal temperatures for adult blue mussels ranged from 16 ºC to 25 ºC (Schulte 1975). Adult mussels survived at temperatures as low as 4 ºC, but their growth rates were vastly stunted (Thomas and Bacher 2018). Additionally between 25-27 ºC growth rates of both veligers and adult blue mussels decreased, with death occurring at temperatures higher than 27 ºC (Hiebenthal et al. 2012, LeBlanc et al. 2005).

b. Description of seawater temperature history in coastal northern Massachusetts

A total of 4,327 sampling days occurred at the Ipswich Bay Yacht Club from 2000 to 2018. When utilizing the strictest removal threshold (10%), only 3,927 days remained. A 25% removal threshold left 4,034 days, whereas a 40% removal threshold increased the number of days that remained to 4,138. After qualitatively inspecting these and determining that they were functionally similar, we chose to use 25% for all further analyses.

After cleaning, the YSI sonde's depth fluctuated between depths of 0.001m and 4.195m. The average water temperature of the Plum Island Sound at the Ipswich Bay Yacht Club rose 2.08 ºC between 2000 and 2018 (Figure 3A). This equates to an increase of roughly 0.1 ºC per year. Much of this increase is driven by the summer ($F_{(1,17)} = 5.715$, $r^2 = 0.2516$, $p < 0.05$, $\beta = 0.5707$) and fall ($F_{(1,7)} = 20.04$, $r^2 = 0.541$, $p < 0.001$, $\beta = 0.9242$) seasons (Figure 4).

Currently, the average inshore water temperatures do not exceed the upper threshold of 30 ºC for the eastern oyster and the 25 ºC limit for the blue mussel (Figure 3A). However, when considering hourly temperatures in the past 19 years, there have been instances where water temperatures surpassed 25 ºC. Also of note, 16 of the 21 days that had daily temperatures averaging higher than 25 ºC occurred in the last five years.

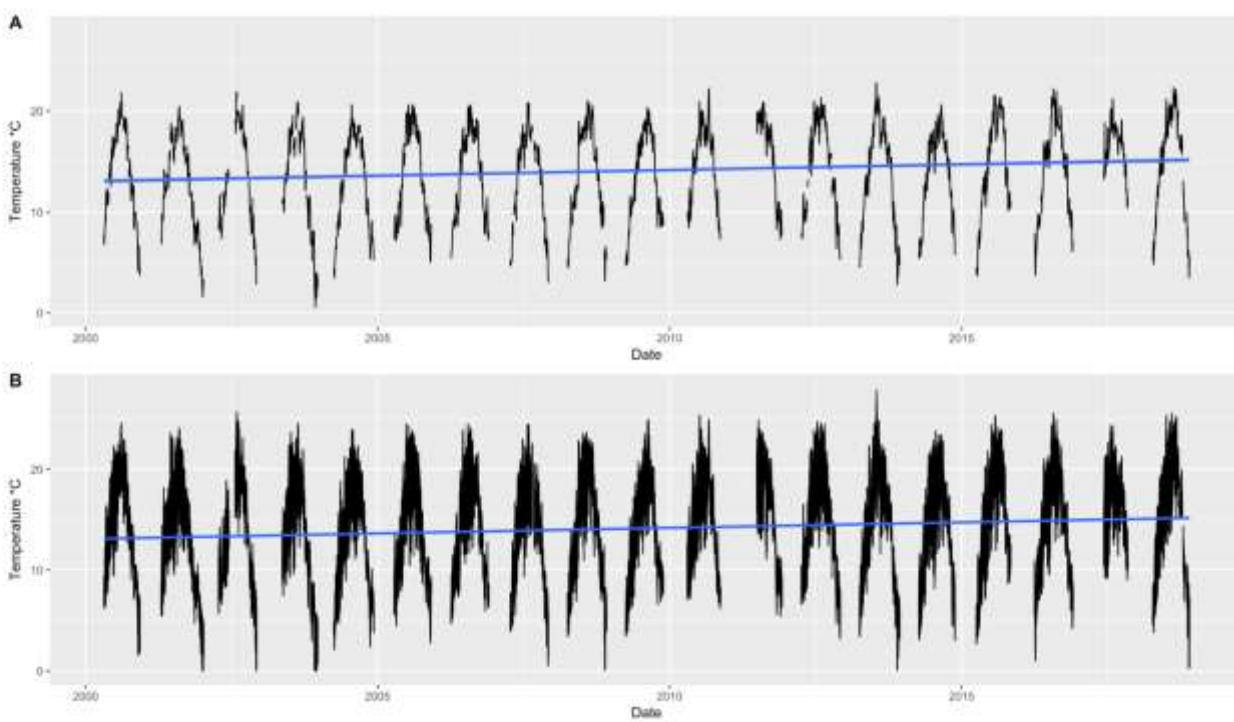


Figure 3: A) Daily average water temperature and B) daily water temperature for Plum Island Sound at Ipswich Bay Yacht Club.

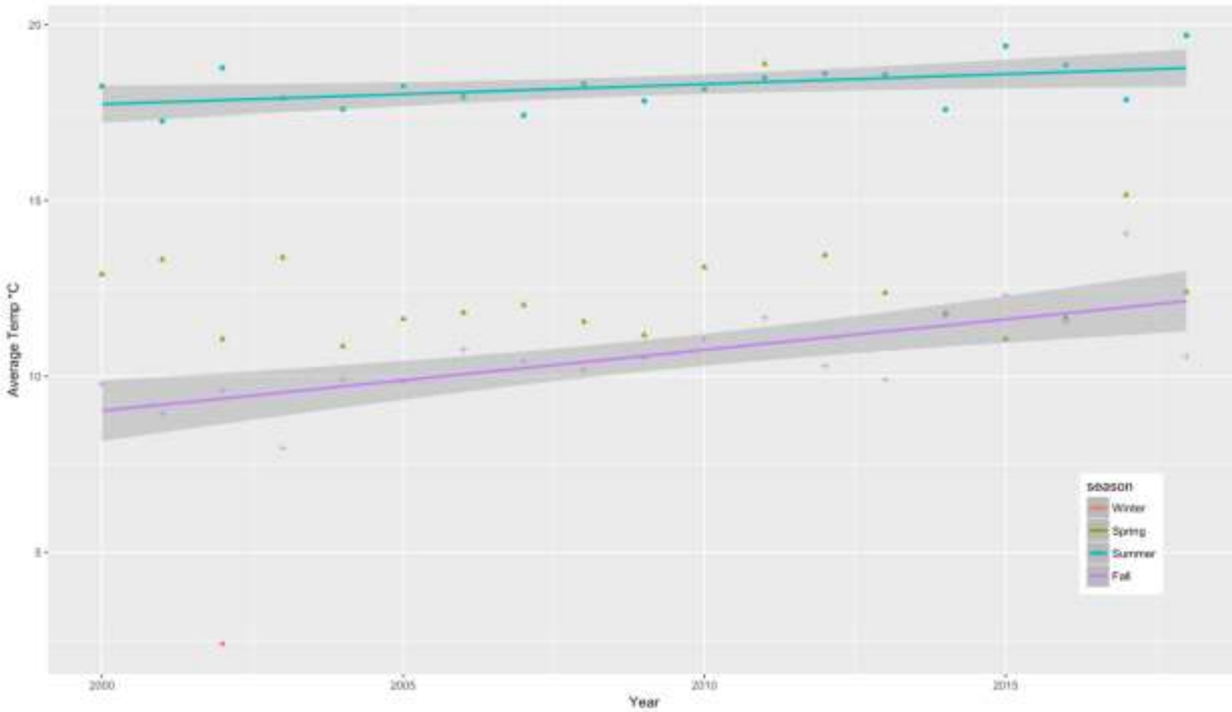


Figure 4: Average seasonal water temperature for Plum Island Sound at Ipswich Bay Yacht Club.

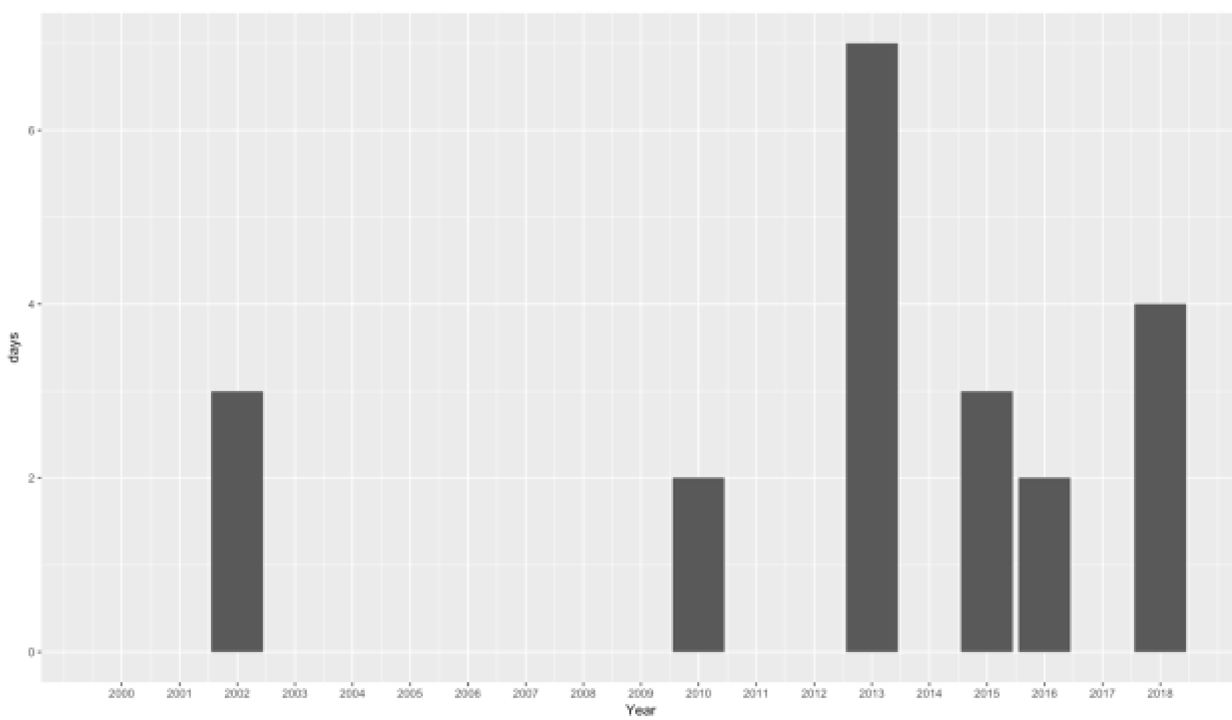


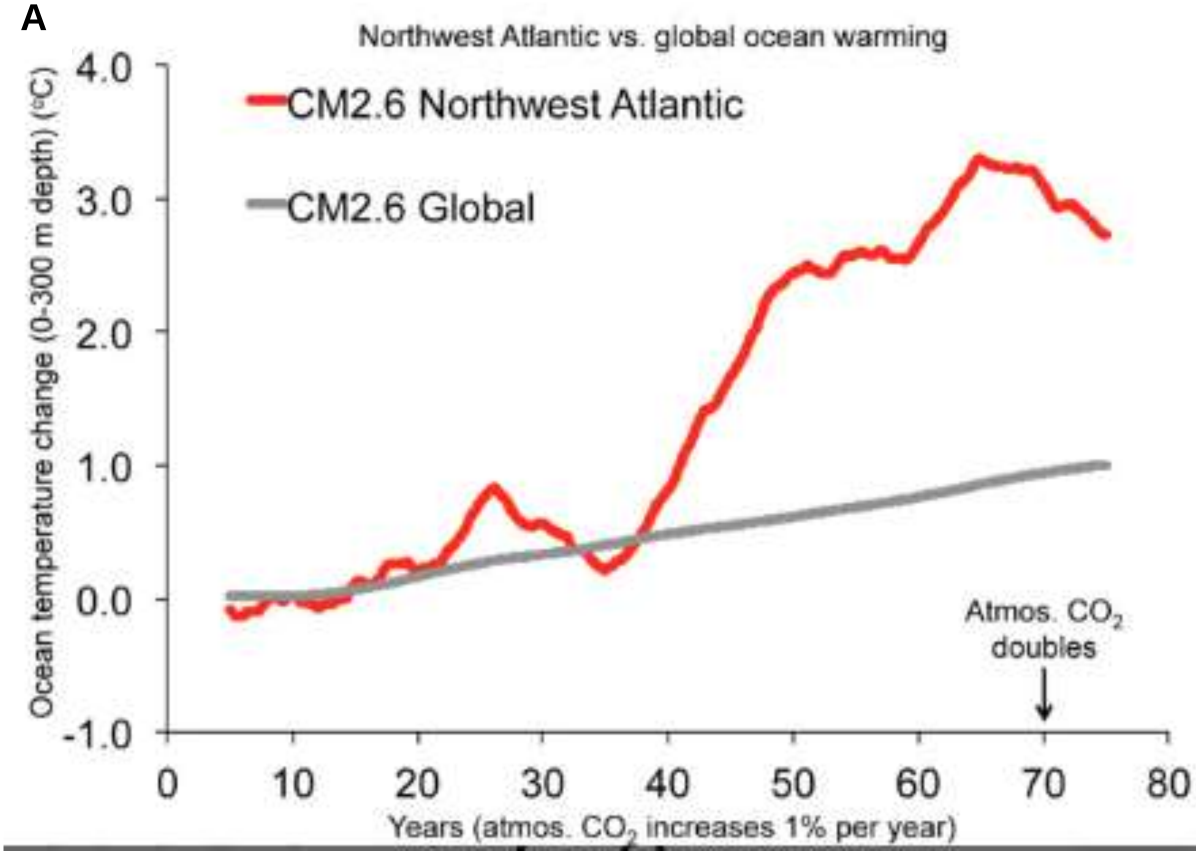
Figure 5: Number of days when water temperature at Ipswich Bay Yacht Club exceeded 25ºC.

c. Seawater temperature projections for coastal northern Massachusetts under climate change

No sea surface temperature projections could be found for the inshore waters of northeastern Massachusetts. However, climate models examining a wider scope of area, such as the Northwest Atlantic and Gulf of Maine, have been developed. In general, the consensus is that the Northwest Atlantic is warming at a faster rate than the global ocean (Saba et al. 2016, Figure 5A). The annual sea-surface temperature in the GOM has increased by an average of 0.23-0.26°C yr^{-1}, which far surpasses the global ocean average of 0.01 °C yr^{-1} (Mills et al. 2013; Pershing et al. 2015).

Warming projections suggest substantial temperature increases will occur in this region over the next century under most climate model scenarios. Within 80 years, assuming a 1% per year increase, atmospheric CO_2 will have doubled. By 2100, multiple models put forth by the NOAA Geophysical Fluid Dynamics Laboratory predict a potential increase of up to 3 ºC for the northwest Atlantic (Figure 5B). Sea-surface temperature for the Gulf of Maine is predicted to follow suit with an increase by roughly 1 ºC in the next 50 years and reaching 3 ºC in the next 80 (Brickman et al. 2016 and Saba et al. 2016, Figure 6 and 7). While it is challenging to predict what will occur within inshore waters, a 3 ºC seawater temperature increase in the Gulf of Maine would likely correspond with dramatic temperature increases in coastal embayments and estuaries.

A


Northwest Atlantic vs. global ocean warming
CM2.6 Northwest Atlantic
CM2.6 Global
Ocean temperature change (0-300 m depth) (°C)
4.0
3.0
2.0
1.0
0.0
-1.0
Atmos. CO_2 doubles
0
10
20
30
40
50
60
70
80
Years (atmos. CO_2 increases 1% per year)

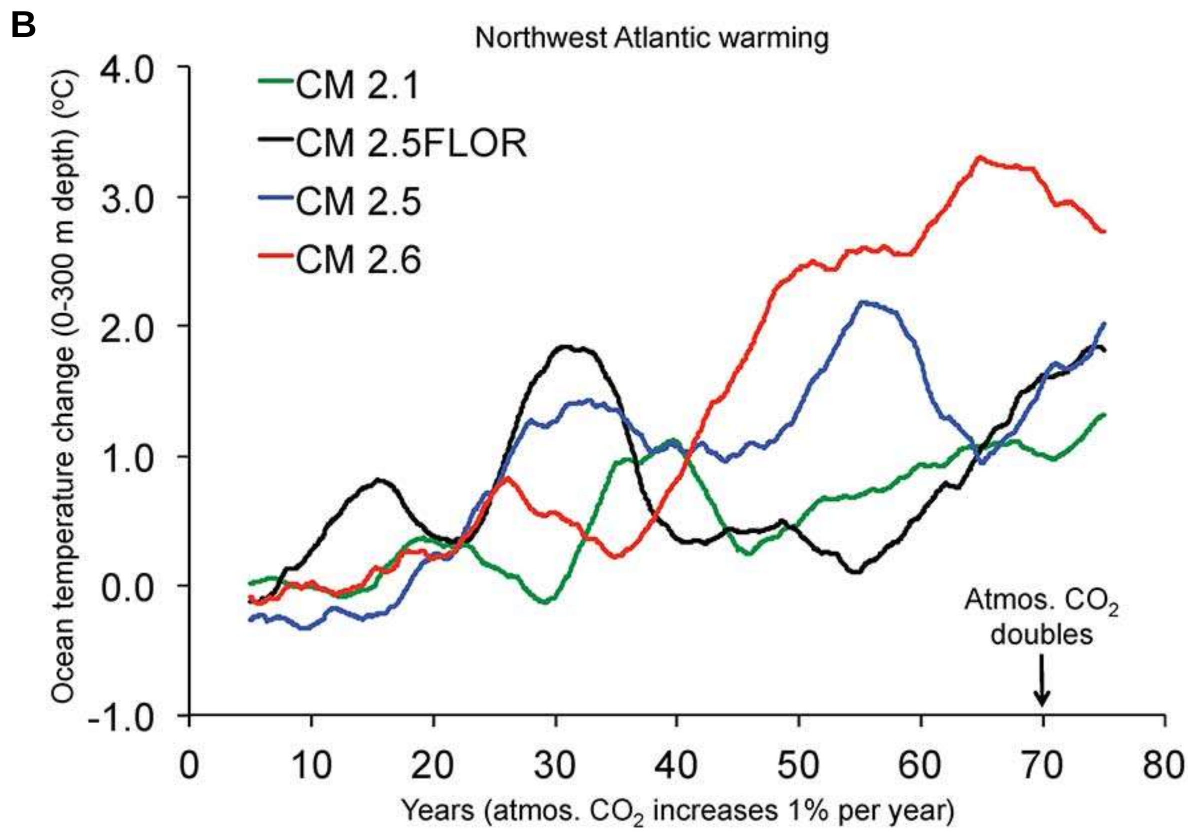


Figure 5: A) Surface of Northwest Atlantic versus global upper-ocean (0 to 300m) temperature change and B) Northwest Atlantic surface temperature (0 to 300m) change from four Geophysical Fluid Dynamics Laboratory climate models. Adapted from Saba et al. 2016.

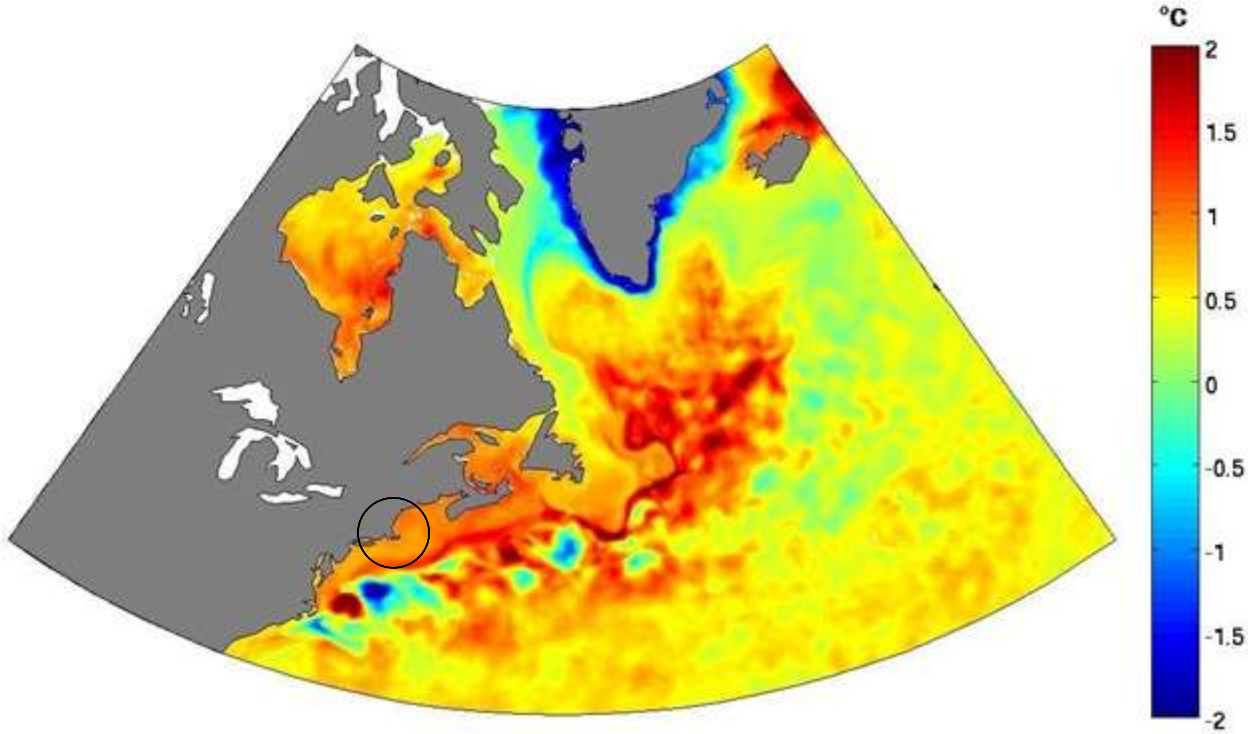


Figure 6: The predicted change in annually averaged sea-surface temperature between the current climate and a future climate simulation (2046-2065), with a focus on Massachusetts, USA. Adapted from Brickman et al. 2016.

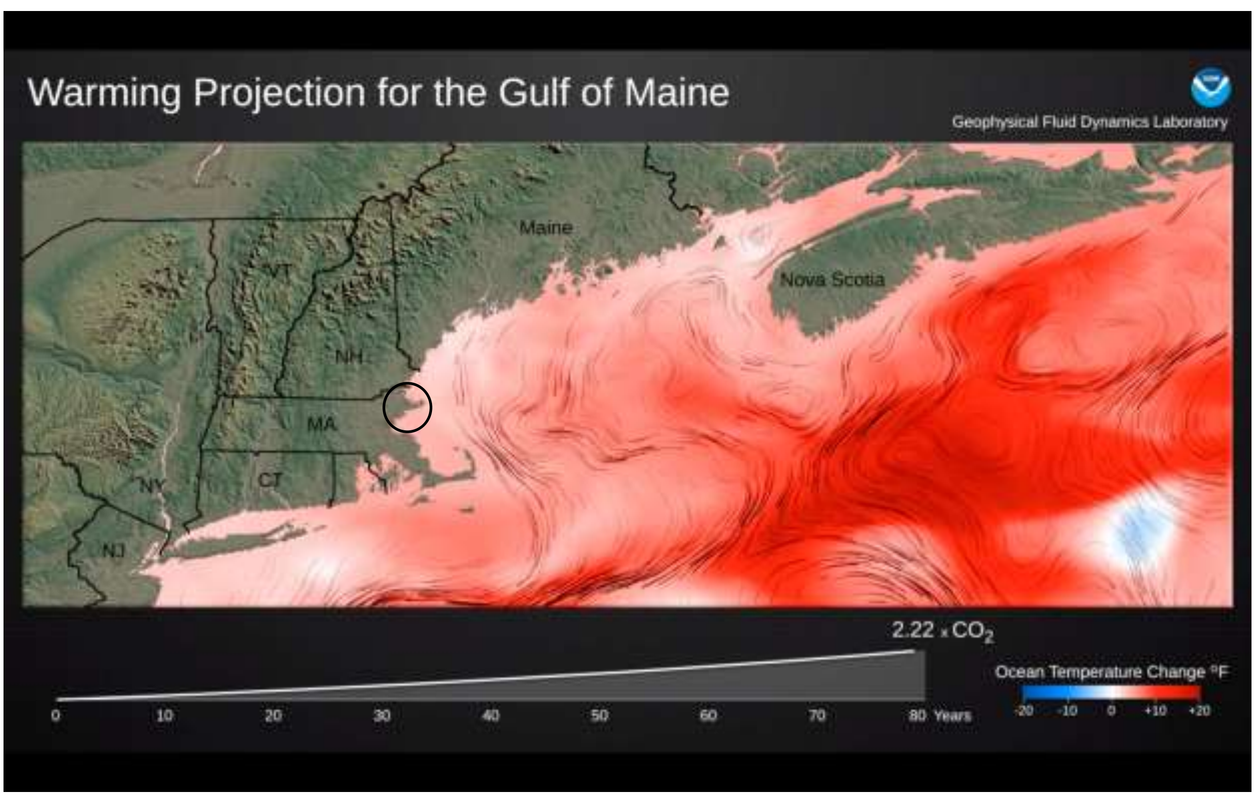


Figure 7: Surface ocean temperature (0 to 200m) under a transient climate response (1% per year increase in atmospheric CO_2, model GFDL CM2.6) with a focus on Ipswich Bay. Adapted from Saba et al. 2016.

d. Review of living shoreline materials

Living Shorelines. Use of living shorelines to stabilize eroding coastal zones is a recently developing strategy utilized by communities living within impacted areas. Living shorelines are constructed barriers that function to "... stabilize the shoreline, reduce erosion, and provide ecosystem services... (NOAA Habitat Blueprint, Living Shorelines: https://www.habitatblueprint.noaa.gov/living-shorelines/)."

With the loss of the extreme end of Castle Neck during recent winter storms, Essex Bay has become increasingly susceptible to storm damage, resulting in the sedimentation of river channels, erosion of existing salt marsh and adjacent shoreline, and loss of ecosystem services in the form of shellfish habitat, fish habitat, and foraging areas for higher vertebrate species. The intent of this literature review is to inform the thinking of engineers and designers as we move to develop a tool box of strategies to protect existing shoreline property and work to restore already impaired and damaged salt marsh and coastal zone resources.

Design and Construction Guidelines. Site specific planning is emphasized in the literature reviewed to ensure success of living shoreline projects. Several reports reviewed provide screening tools for planning (Balasubramanayam and Howard 2019, Woods Hole Group 2017) and regulatory, policy, and engineering considerations as a guide for decision makers and practitioners in coastal environments (Cunniff and Schwartz 2015, Miller et al. 2016, NOAA 2015, O'Donnell 2016, Spalding et al. 2014, USACOE 2013, and Woods Hole Group 2017).

New England presents specific challenges to the design and construction of living shorelines due to: (1) larger fetch and consequently larger coastal wave amplitude and period; (2) winter ice; (3) larger tidal range; (4) effects of storm surge; and (5) the highly variable coastal geomorphology. These conditions require greater site-specific study when selecting and designing a shoreline stabilization strategy (O'Donnell 2016 and Woods Hole Group 2017).

The Woods Hole Group (2017) provides a thorough and detailed document outlining living shoreline approaches appropriate for New England and a list of considerations on how to meet the unique challenges of this region. Two of these approaches are of particular interest when considering salt marsh restoration in the Essex Bay: (1) Marsh Creation/Enhancement with Toe Protection; and (2) Living Breakwaters. Some of the unique adaptations suggested for New England projects include using roughened surfaces to break up ice sheets, designing gentler slopes (6:1-10:1) with shrubs for marsh projects to respond better to ice; planting in the spring to allow vegetation to become established before it has to withstand ice; using hardy, salt-tolerant shrubs for shorelines that are affected by ice; and carefully considering where within the tidal range oysters will be placed, if they are used, as placement high in the intertidal zone may result in freezing. The reef balls installed in Stratford, Connecticut, for example and as outlined in the case studies herein, withstood significant icing during the 2014-2015 winter season.

Ecological Value. The literature review provided persuasive reasoning that one of the key bases for installing living shorelines is that protective shoreline structures can, and should, provide significant ecosystem services. In fact, many of the reports and articles encourage the consideration of these ecosystem services that benefit society, including habitat for fish and other living marine resources, food production, nutrient and sediment removal, and water quality improvement (NOAA 2015, Odell et al. 2006, O'Donnell 2016).

Gittman et al. (2016) evaluated the effectiveness of living shorelines in maintaining or improving ecosystem services compared to naturally vegetated shorelines and hardened shorelines. Three shoreline conditions were compared: constructed sills consisting of an offshore low-profile breakwater with landward marsh, natural salt marsh shorelines (control marsh), and unvegetated bulkheads. After quantifying and then comparing the effectiveness of the three shoreline conditions in providing habitat for fish and crustaceans, "…sills supported higher abundances and species diversity of fishes than unvegetated habitat adjacent to bulkheads, and even control marshes." Importantly, the sill ecosystem-service enhancements were only identified three or more years after construction. Gittman et al. (2016) concluded that "[s]ills provide added structure and may provide better refuges from predation and greater opportunity to use available food resources for nekton than unvegetated bulkheaded shores or control marshes."

O'Donnell (2016) noted that there is good reason to attempt intertidal restoration, which may or may not include shellfish, as "…traditional coastal armoring impacts or results in loss of the intertidal zone, which is critical to submerged aquatic vegetation and shallow water habitat vital to a diverse range of species." Integration of a structural barrier (e.g., a subtidal sill or reef) and development of salt marsh landward of the sill appear to be the standard practice at projects where living shorelines have been successful. The constructed sill, or reef, serves as a wave energy dissipater located seaward of the planted marsh areas. Indeed, living breakwaters (typically placed sub-tidally) have the capacity to break waves, dissipating wave energy, and help to reduce wave related erosion (Woods Hole Group 2017), while creating valuable habitat with significant ecosystem services.

Performance in Decreasing Coastal Risk. Prevalent throughout the results of the literature review is the analysis of living shoreline performance in decreasing coastal risk given changing coastal conditions due to sea-level rise and global warming (USACOE 2013, National Research Council 2014). Reports note that marshes protect estuarine shorelines from erosion better than bulkheads during a Category 1 hurricane (Gittman et al. 2014), and they also analyze existing living shorelines and their relative coastal protection performance (Cunniff and Schwartz 2015). Spalding et al. (2014), state that the role healthy coastal ecosystems play in shoreline protection has not been sufficiently accounted for in coastal planning and engineering in the past. They note that natural ecosystems have advantages over more traditional, engineered shoreline protection in their capacity for self-repair and recovery and the significant co-benefits they provide.

The National Research Council (2014) notes that "…the majority of coastal-storm-related federal investments are provided only after disasters," and it recommends advance planning and investment, at a local level, to help alleviate the enormous and rising costs of coastal disasters. Part of the strategy it recommends includes incorporating living shoreline approaches planned at a local level. As coastal communities consider potential shoreline protection approaches, more are choosing integrated living shoreline approaches (Matchar 2018) in combination with carefully designed construction (Caulfield 2017).

Incorporation of Shellfish Restoration into Living Shorelines. The incorporation of shellfish restoration (placed both inter-and sub-tidally) into a "hybrid" living shoreline approach to maximize ecological services benefits, both species diversity and water quality improvement, was reviewed. Substantial decreases in native oyster reefs have occurred worldwide, resulting in the loss of associated ecosystem services (Kirby 2004, Beck et al. 2011, zu Ermgassen et al. 2012). For example, zu Ermgassen et al. (2013) estimated the volume of water filtered by oyster populations in the past (c. 1880–1910) and present (c. 2000–2010) in 13 US estuaries; they concluded that the filtration capacity of oysters has declined almost universally (12 of the 13 estuaries examined) by a median of 85%.

Several studies have focused on the possibility of reversing, or at least lessening, the impacts of eutrophication through oyster restoration and have found significant improvements in the level of dissolved oxygen (DO), chlorophyll, light attenuation, and submerged aquatic vegetation in Chesapeake Bay (Cerco and Noel 2007). They recommend oyster restoration as a supplement to nutrient load reduction, not as a substitute. In general, oyster reef restoration (as well as other shellfish restoration) can significantly improve water quality by reducing excess nitrogen (McDermott et al. 2008, Piehler and Smyth 2011, Kellogg et al. 2013, Smyth et al. 2013 & 2015, Hoellein and Zarnoch 2014).

Pilot projects and studies in the southern United States have used oysters to construct breakwater reefs along stretches of eroding shoreline. At least one of these studies "…found that the corridor between intertidal marsh and oyster reef breakwaters supported higher abundances and different communities of fishes than control plots without oyster reef habitat. Among the fishes and mobile invertebrates that appeared to be strongly enhanced were several economically-important species" (Scyphers et al. 2011). This study also reported that "[a]lthough the vertical relief of the breakwater reefs was reduced over the course of our study and this compromised the shoreline protection capacity, the observed habitat value demonstrates ecological justification for future, more robust shoreline protection projects" (Scyphers et al. 2011). Because oysters create reefs by cementing together, they can help reinforce the structural robustness of the constructed shoreline.

Beck et al. (2011) state that native oysters provide valuable reef habitat, and an increasing number of examples show that reef recovery is feasible. They go on to note that "[w]e need new approaches within the regulatory and management communities to lead to shellfish habitat conservation and restoration designed not just for fisheries production but specifically to recover these critical ecosystems and their services." Adding oyster reef restoration to living shoreline design may greatly increase a project's habitat and ecosystem services value. Coastal managers will likely need to alter their methods of quantifying a project's overall value and benefits in their decision making to start including oyster restoration as a benefit in living shorelines.

Living shorelines projects often include oysters and other shellfish to increase biodiversity. For instance, in Great Bay Estuary, New Hampshire, an integrated ecosystem approach was developed to identify multi-habitat restoration opportunities. Restoration targets included oysters and softshell clams, salt marshes, eelgrass beds, and seven diadromous fish species. As part of developing this project, "…a matrix of habitat interactions was created to identify potential for synergy and subsequent restoration efficiency" (Odell et al. 2006). The projects in Great Bay Estuary were designed to enhance and a restore a complex ecosystem, not just a particular habitat.

Subtidal Oyster Restoration. Restoration of subtidal oyster (*Crassostrea virginica*) reefs has been conducted on the east coast of the United States (Kennedy and Sanford 1999, Kennedy et al. 2011). For instance, extensive restoration efforts have been conducted in the Chesapeake Bay, Maryland (Maryland Oyster Restoration Interagency Workgroup 2015, NOAA 2019), New York Harbor, New York, and the Great Bay Estuary, New Hampshire (Grizzle et al. 2006, Odell et al. 2006, Konisky et al. 2012, Grizzle and Ward 2016, Moeser et al. 2016). Much documentation exists that summarizes the efforts made in these regions to date, their level of success, and suggestions on future restoration and research efforts for subtidal reef habitat. In addition, Brumbaugh et al. (2006), in collaboration with The Nature Conservancy, offer a detailed guide to the technical aspects of shellfish restoration and reef ecosystem services.

In Massachusetts, one of the earlier oyster reef restoration efforts was conducted in the Massachusetts Audubon Wellfleet Bay Wildlife Sanctuary (Faherty et al. 2011). This project compared different substrates for constructing reefs on tidal flats in Cape Cod. The goal of the project was to restore an oyster reef on tidal flats, and some of its objectives were to use wild oysters, monitor the growth and survival of the oysters on three different types of substrates (oyster castles, shell cultch, and reef balls), and determine which treatment worked best to recruit and grow wild oysters and to increase the diversity of other organisms present. The study determined that the oyster castles "…were the only substrate to maintain their structural integrity and to show a net increase in their oyster population each year. Invertebrate abundance and diversity has measurably increased on the project site relative to control sites and shorebird use increased." Additional restoration efforts have been conducted more recently on Martha's Vineyard and Nantucket, as well as in southeastern Massachusetts. For instance, The Nature Conservancy coupled with Chillmark and West Tisbury to conduct successful restoration efforts in Tisbury Pond, Martha's Vineyard in 2012 and 2013.

After working on the Massachusetts Audubon Wellfleet Bay Wildlife Sanctuary project, Faherty (2011) noted that one of the most important results of their work was a better understanding of the permitting process for shellfish in Massachusetts. Current regulations do not allow shellfish sanctuaries, but instead require newly established reefs be harvested after three years. Thus, because it could not obtain total protection, Massachusetts Audubon "…negotiated an experimental harvest plan to study the effect of different harvest levels (0%, 50%, and 100% of legal oysters) on the reef." Harvest was scheduled for the summer of 2012. In recent personal communication with Faherty, we learned that no further reports exist of the site because two difficult winters with ice sheets and rough water demolished the remaining reefs (Faherty, pers. comm. 2019).

Intertidal Oyster Restoration. While ice sheets present unique challenges for oyster reef restoration in New England, oysters have been documented recently in the intertidal zones of this region. For instance, Capone et al. (2008) found dense populations of intertidal oysters "…at several estuarine sites within New Hampshire and mid-coastal Maine, with these growing under dense canopies of the long-lived *Ascophyllum nodosum* (fucoid alga, rockweed). The densities of these northern intertidal oysters rival subtidal populations in the same geography, and their sizes suggest a persistence of 5 or more years." The authors note that they were not aware of any previous reports on high densities of intertidal oysters in the northeast and concluded that either these intertidal oysters did not exist historically, or there were so few that they were not identified. They believe it is possible that these intertidal oysters were not noticed, since they are found under dense canopies of rockweed, but they also state that it would be valuable to investigate whether changing climate conditions have resulted in the new presence of these intertidal oysters. Capone et al. (2008) also note that "[i]ntertidal oysters provide a complex structure for the attachment of additional epifauna, such as ribbed mussels and barnacles. At low tide, the extensive rockweed cover in these geographies can completely cover oysters protecting them from environmental extreme[s]." The authors noted that they rarely found oysters on "bare rocky substrata" at all study sites and that "[b]are rock outcrops of equal size and tidal height compared to those covered by *Ascophyllum* typically had no attached oysters." It is likely that the presence of an algal canopy supports the recruitment, growth and/or survival of these intertidal oysters.

Bertness et al. (1999) studied the habitat modifications that result from the presence of seaweed canopies in intertidal habitat. They determined that the "…algal canopy greatly reduced potential physical stresses, particularly at high tidal heights" and "…at the high intertidal border of the canopy the recruitment, growth, and survival of understory organisms were enhanced by the canopy." It is possible, therefore, that a rich ecosystem of epifauna may exist under algal canopies and be at least partially protected from harsh environmental factors.

Oysters in Essex Bay. Frye (2017) conducted a study "…to inform the Town of Essex on solutions to marsh retreat in the Great Marsh and provide a methodology for selecting suitable sites for living shorelines aimed at reducing marsh retreat." Frye studied suitable sites for a subtidal oyster reef breakwater in Essex Bay by measuring sediment deposition, sediment organic matter (SOM), natural recruitment, and rate of marsh retreat. Results of the study indicated "…that sediment accumulation was the main factor in determining suitable sites for a successful reef, as oyster height and survival significantly decreased with sedimentation. Sites that had both a need for shoreline stabilization and replicate oysters' natural habitat, such as low energy, high SOM, and lower salinity within the protection of the estuary, were found to be the most suitable for a successful oyster reef breakwater." Several sites within Essex Bay with good potential for restoration were identified.

Living Shoreline Case Studies. Summarized below are living shoreline case studies determined to be applicable to the Essex Bay based on their location, surrounding conditions, and/or materials used.

The Cutts Cove project, located in Portsmouth, New Hampshire, commenced in 2015 and aimed to restore a salt marsh. A living shoreline marsh was constructed by placing an erosion control sill parallel to the vegetated shoreline to reduce wave energy and prevent erosion. The sill is a partially removed riprap wall designed to be "climate ready" to 2060. The construction sequence required clearing and grubbing, flattening the existing riprap wall and building a stone sill, and then backfilling with sandy silt to the determined elevation. Monitoring included evaluating erosion, plant establishment and growth, and animal use of habitat. Important conclusions from this project included (1) recognizing the limited growing season available for plantings, (2) acknowledging that project success can decrease with increased tidal range and physical exposure from shear stress due to waves and ice, (3) putting in irrigation for the new salt marsh plants was necessary, (4) shading of plants and trampling from people and other animals (geese, crabs, snails) can reduce the success of these projects, and (5) considering management at the landscape scale (Burdick et al. 2019).

The Senator Joseph Finnegan Park project, located in Dorchester, South Boston, Massachusetts, (Hagopian and Schwanof 2015), converted an old industrial site with an existing broken granite wall and dilapidated timber bulkhead into a living shoreline. Soil and loose debris were excavated to appropriate elevations, and a timber bulkhead was cut and removed after salt marsh creation was complete. Coir logs were placed at intervals with aluminum anchors to set proper grades, and salt marsh plantings were installed. The project is currently in its third growing season and both high (*Spartina patens*) and low (*Spartina alterniflora*) marsh grasses are well established. Portions of the restored high marsh utilized existing high marsh that had become established on pavement (former parking lot), which was harvested and used as "sod" (DeRosa pers. comm. 2019).

Clippership Wharf located within Boston Harbor, in East Boston, Massachusetts, used salvaged granite blocks excavated from the site's old sea walls to create terraced planting cells down to a tide pool habitat and rocky intertidal shore. High marsh (*Spartina patens*) and low marsh plantings (*Spartina alterniflora*) were installed within each cell and are entering their second growing season. The site has attracted wildlife while also acting as a buffer to storm water to the new development (Hagopian and Schwanof 2015). Although establishment of the low marsh was slower than at Finnegan Park due to the lower elevations of the planting cells, *Spartina alterniflora* is becoming established. Furthermore, the deeper cells are functioning as rocky intertidal shore and tide pool habitat, all of which provides substantial ecological services to the local habitat. Blue mussels, rock weed, barnacles, crabs, as well as sea stars, striped bass, and other fish species have all been observed occupying the newly created living shoreline (DeRosa, pers. comm. 2019).

The Stratford Point project located in Connecticut began in 2010 and involved the installation of 64 pre-cast, concrete reef balls, or a “living breakwater,” in the intertidal zone, in conjunction with restoration of low and high marshes and dune shoreward of the artificial reef. The project is designed to also restore upland shrub, coastal forest, and meadow mosaic to improve bird and pollinator habitat. The living breakwater acts as a fish and blue crab nursery and a hard substrate for shellfish settlement. To date, the project has stabilized the shoreline and restored the desired habitats, with increased species diversity (Connecticut Audubon Society 2013, Woods Hole Group 2017).

The Wagon Hill Farm project is located along the tidal Oyster River within the Town of Durham, New Hampshire. Grant funding ($250,000) was awarded to this project in November 2018 to create a 0.36-acre living shoreline. Loss of salt marsh vegetation and erosion of marsh sediments resulted in shoreline retreat at a rate of up to 1 foot per year along almost 2,000 linear feet of shoreline. The project includes shoreline stabilization, habitat enhancement, and flood damage protection by incorporating natural, green, “soft” infrastructure (University of New Hampshire and NOAA 2016). The Durham Conservation Commission is also considering the creation of an oyster reef to help decrease wave erosion at this site (Durham Conservation Commission 2009).

North Mill Pond Marsh Restoration project, located in Portsmouth, New Hampshire, involved restoration of low and high marsh along North Mill Pond, with about half of the area consisting of new marsh creation, and the other half consisting of restoration of degraded low and high marsh through sediment addition (i.e., thin layer deposition). The construction of toe protection was included (Burdick et al. 2019, Woods Hole Group 2017, DeRosa pers. comm. 2019).

Collectively, these projects demonstrate that a number of similar living shoreline projects have been conducted in northern Massachusetts and New Hampshire. Furthermore, they offer several insights that will enhance the success of future living shoreline restoration projects in the region.

Conclusions

Our review of the thermal habitat requirements of the eastern oyster and blue mussel suggest that blue mussels not surprisingly are adapted to substantially colder environments. This difference in thermal performance, coupled with the warming that the Gulf of Maine has already experienced, could explain anecdotal observations suggesting a decline in historical mussel bed habitat in eastern Massachusetts and throughout the Gulf of Maine (Sorte et al. 2017), and that oysters are becoming more prevalent in this region. Indeed, the temperature monitoring occurring at the Plum Island Sound LTER demonstrated that temperature is already occasionally surpassing the 25 ºC upper threshold for blue mussels, and this phenomenon will likely increase in the coming years if climate scenarios projecting warming waters in the Gulf of Maine are accurate. Thus, integrating blue mussels into living shorelines restoration designs may not be a viable long-term strategy.

The eastern oyster, which is at the northern end of its range in the Gulf of Maine, is more common in the Mid-Atlantic and South Atlantic Bights in warmer waters (Byers et al. 2014). The studies investigating the thermal habitat preferences of the eastern oyster suggested that seed oysters are slightly more vulnerable to colder waters than spat oysters, and much more vulnerable than adult oysters, which performed well above 15 ºC. However, given the trajectory of warming waters throughout the Northwest Atlantic and especially in the Gulf of Maine, the eastern oyster offers a potential option for living shorelines and coastal habitat restoration opportunities in the region.

As the science and art of coastal zone protection and restoration advances in the wake of climate change and sea-level rise, the accumulated tools that develop will make up a set of resilient strategies to assist communities during these changing times. Based on the current literature, constructed living shorelines should be included in this set of tools. Coastal shorelines, in particular, are increasingly susceptible to storm damage and flooding from rising seas. Essex Bay is an area that has become dramatically impacted by these changing conditions. Specifically, it is already experiencing infilling of tidal rivers with sediment from eroded marsh habitats, loss of shellfish habitat, and the loss of associated ecological services such as diminished flood storage and storm damage prevention. Furthermore, the loss of portions of the Great Marsh has resulted in significant impairment of the estuary and the loss of ecological services provided by this unique ecosystem. Utilizing green technology such as living shorelines in concert with thin layer deposition, eel grass beds, and oyster and blue mussel beds, among other strategies, could serve to reduce the impacts of the adverse impacts of climate change and sea-level rise.

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