Estuaries and Coasts
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Comparison of Artificial and Natural Reef Productivity in Nantucket Sound, MA, USA

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Abstract

Artificial reef communities undergo long periods of succession prior to becoming stable, but funding challenges often prevent post-deployment monitoring from evaluating long-term successional changes. The present study employed baited remote underwater video surveillance (BRUVS) to compare the species richness, diversity, abundance, and age structure of fishes across a 4-year-old artificial reef, a 41-year-old artificial reef, a representative natural reef, and a bare control in Nantucket Sound, MA, USA, to address whether the perceived success of an artificial reef can be determined 5 years after deployment. Results indicated that, while species richness and diversity were largely uniform throughout the Sound, fish appeared on camera 93.4% faster at artificial reef sites than they did at the bare sand control. Reef-associated fish were 103.7% more abundant on the older artificial reef than on the younger artificial reef. Therefore, although the younger artificial reef is in its fifth year of monitoring, abundances of economically important fishes on the reef may continue to increase in future years and current numbers may not accurately reflect reef success or failure. Future management plans should consider extending monitoring programs longer than 5 years and implementing temporal fishing closures on newly deployed reefs to facilitate earlier post-deployment community stabilization.

Keywords Artificial reef · Attraction · BRUV · Fisheries management · Nantucket Sound · Production

Introduction

Reef fishes are a crucial component of the biodiversity of coastal systems and provide a large source of biomass for both recreational and commercial fisheries on a global scale (Malcolm et al. 2007; Harasti et al. 2015). Excessive extractive pressures on reef fishes have driven many countries to deploy artificial reefs within their territorial waters to promote fisheries protection and enhancement (Hunter and Sayer 2009; Folpp et al. 2013). Artificial reefs may be defined as approved structures that have intentionally been placed or constructed for the purpose of enhancing benthic relief (Rousseau 2008). Deployment of such reefs has substantially increased in recent decades, and they now form part of a broader strategy for marine ecosystem management in many countries (Baine 2001; Becker et al. 2017). While primarily focused on bolstering coastal fisheries, artificial reefs may provide compounding secondary benefits such as habitat and shoreline protection, sediment accretion, and mitigation for habitat loss resulting from coastal alterations (Rousseau 2008; Hunter and Sayer 2009).

Structured habitat enhancement of an otherwise featureless substrate fosters the development of rapidly growing, highly productive fouling communities that feed on plankton and detritus (Baine 2001). Hard reef substrate enhances surface area on which fouling organisms can settle and anchor to better withstand ocean currents and the destructive force of ocean storms. Over time, the resulting increase in biomass enriches the surrounding water column and sediments, promotes the growth of infauna and colonization by filter-feeding invertebrates (Svane and Petersen 2001), and provides food sources for reef-associated invertebrates such as *Cancer* sp. crabs, American lobster (*Homarus americanus*), and finfish such as scup (*Stenotomus chrysops*), black sea bass (*Centropristis striata*), and tautog (*Tautoga onitis*) (Rooker et al. 1997; Baine 2001). Additional factors such as the timing of reef installation, materials used to create structured habitat,

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and post-deployment use (e.g., species harvesting) may also influence community development (Carter et al. 1985).

Artificial reef designs are often structurally complex and provide vertical relief that offers more living area for the same relative unit of seafloor. Crevices and interstitial spaces provide refuge from predation and energy-depleting currents for demersal and pelagic fishes. Ultimately, the interspersion of reef, natural substrate, and open-water habitats creates greater environmental complexity that confers observable, positive impacts on reef fishes. Increases in species richness (Gratwicke and Speight 2005), diversity (Fabi and Fiorentini 1994), and abundance (Charbonnel et al. 2002; Brickhill et al. 2005; Hunter and Sayer 2009) have been attributed to the removal of environmental or ontogenetic bottlenecks by the deployment artificial reefs (Hunter and Sayer 2009). Such benefits are then returned to coastal populations in the form of ecotourism, recreational diving and fishing, and commercial fishing enhancement.

Whether the observed increases in richness, diversity, and abundance are due to fish production or simply attraction is of particular interest to fisheries managers. The attraction hypothesis suggests that artificial reefs attract fishes from surrounding habitat by providing food, shelter, and behavioral cues that exploit the thigmotactic tendencies (inclination to move towards structured rather than featureless habitat) of fishes (Bohnsack 1989; Powers et al. 2003; Brickhill et al. 2005; Smith et al. 2015). It has been largely assumed that attraction over production of fish by an artificial reef is a harmful process because the aggregation of existing fish biomass into a smaller area of habitat can make it easier to exploit fish stocks that have been previously cost-ineffective to pursue (Grossman et al. 1997; Brickhill et al. 2005; Smith et al. 2015). This attractive property has been demonstrated by higher catch rates yielded on artificial reefs compared with natural reefs and may potentially imply that initial increases in abundance are temporary (Gregg 1995; Smith et al. 2015). In contrast, the production hypothesis suggests that the additional habitat provided by artificial reefs may increase a region's carrying capacity by allowing a greater number of juveniles to settle, survive to adulthood, and contribute new individuals to local fish stocks by spawning (Bohnsack 1989; Powers et al. 2003; Brickhill et al. 2005). Such productivity of artificial reefs has been demonstrated by numerous studies and attributed to increased habitat connectivity (Cenci et al. 2011), increased production of organic matter (Cresson et al. 2014), and increased biomass and fecundity tied to larger structure (Granneman and Steele 2014).

There is a general understanding that attraction to and production of fishes by artificial reefs are not mutually exclusive processes, but rather a continuum (Bohnsack 1989; Smith et al. 2015). Management plans implementing artificial reefs should therefore attempt to favor increased fish production over harvest through appropriate siting and design. It has been suggested that fishes will be more attracted to lone artificial reefs deployed in oligotrophic environments, while increased reef complexity and connectivity will promote production (Brickhill et al. 2005). Furthermore, recent studies have increasingly suggested that attraction of fishes to artificial reefs may disperse existing fish biomass to multiple reefs rather than concentrating it, thus making fish stocks harder to exploit (Smith et al. 2015; Becker et al. 2017). Ecosystem management should therefore favor greater habitat connectivity for reef fishes via strategic deployment of additional artificial reefs to promote stock dispersion over aggregation (Smith et al. 2015).

Post-deployment monitoring of artificial reefs is necessary to distinguish between the qualities that promote attraction versus production (Relini et al. 2002; Becker et al. 2017). Such monitoring should be long-term, as communities on artificial reefs undergo long periods of succession prior to becoming stable (Becker et al. 2017). While young artificial reefs are quickly colonized by adult fishes, there are positive, linear relationships between reef age and species richness, diversity, and abundance that may extend dozens of years after a reef is deployed (Coll et al. 1998; Relini et al. 2002; Folpp et al. 2013). Therefore, younger artificial reefs may exhibit greater attractive properties, while older reefs may exhibit a stronger balance between attraction and production. However, research supporting this claim is limited as many monitoring efforts of artificial reef productivity are restricted to short periods of a year to a few years after reef deployment due to challenges of funding longer monitoring programs on artificial reefs (Relini et al. 2002; Barber et al. 2009; Becker et al. 2017).

A range of techniques have been employed in the post-deployment monitoring of artificial reefs, many of which rely on underwater visual census (UVC) by SCUBA divers or snorkelers, or fish capture by angling and trapping (Malcolm et al. 2007; Harasti et al. 2015). Such methods introduce biases associated with SCUBA diver presence or gear selectivity. Baited remote underwater video surveillance (BRUVS) have been increasingly employed in recent years to minimize these biases. BRUVS are valuable as non-extractive survey tools that do not cause major benthic disturbance, can be deployed in environments unsuitable for conventional UVC assessments, and may reveal shy or cryptic species (Cappo et al. 2004; Harvey et al. 2007; Gladstone et al. 2012; Harasti et al. 2015). Permanent visual records of surveys allow for impartial, repeatable measurements and enable standardized data collection (Cappo et al. 2004). Furthermore, the simultaneous deployment of multiple BRUVS greatly increases the potential replication and spatial range of sampling (Harvey et al. 2007). It should be noted that species attracted to BRUVS bait in large numbers may obstruct the camera view of less common or aggressive visitors; such species shading may impact species richness survey results. Furthermore, baited sampling methods are biased toward species attracted to bait, though this bias may be advantageous for fisheries management purposes as it focuses sampling on species that are targeted by recreational and commercial fishing.

The purpose of this study is to compare species richness, diversity, abundance, and age structure of economically important finfish species with regard to reef age across natural reef, artificial reef, and bare substrates in Nantucket Sound, MA, USA. Natural bedrock outcroppings, cobble shoals, and other hard substrate types are more common to Massachusetts than other coastal Atlantic states in the USA (Rousseau 2008); however, Nantucket Sound is largely comprised of featureless, sandy substrate and is home to the state's oldest and youngest artificial reefs. For this reason, the Sound provides a unique study system to examine variations in fish assemblages across different reef types. BRUV surveys were simultaneously conducted on the state's oldest artificial reef (Yarmouth, 1978), youngest artificial reef (Harwich, 2016), a representative natural reef, and a featureless bare control from June to September 2019. This study hypothesized that species richness, diversity, and abundance would be largely uniform across all reefs regardless of age and greater than the control, while fish age structure would vary according to reef age. Results from this study may contribute to the body of work helping to inform the role of artificial reefs in marine ecological management regimes.

Methods

Study System

The Massachusetts Division of Marine Fisheries (MA DMF) has deployed two artificial reefs in Nantucket Sound: Yarmouth reef in 1978 and Harwich reef in 2016. Yarmouth reef is a 127-acre site comprised of 2500 reef units, each consisting of three to five tires filled with concrete and bundled together. Harwich reef is a 10-acre site comprised of 1600 cubic yards of concrete rubble. Both reefs were deployed on featureless sandy substrate to provide 0.75–1.5 m of vertical structured habitat for structure-oriented finfish. A shallow, isolated rocky outcrop with a well-defined edge between structure and sand was selected to serve as a representative natural reef. A featureless sandy substrate 5 km away from the nearest reef structure served as a bare control. Mean low water (MLW) marks ranged from 7.1 to 10.6 m across all sites, and sites were located no closer than 5 km from one another (Fig. 1; NOAA 2004). At this distance, no overlap of bait plumes occurred during the deployment time, allowing for independent deployments to be made in a single day (Taylor et al. 2013). The present study examined the effect of reef age on reef community composition. The factor reef age contained four levels: the bare control (control site), the Harwich reef (4-year-old reef), the Yarmouth reef (41-year-old reef), and the natural reef (reference site). A total of 20 temporally independent replicates were sampled on each site.

Sampling Design

Five identical mono-horizontal baited remote underwater video (mono H-BRUV) units were constructed for this study; one BRUV was deployed per site and one was reserved as a backup. Each BRUV consisted of a weighted PVC frame, a solid state sports camera with underwater housing attached to an aluminum bait-pole, a bait box located 80 cm from the camera and suspended 49.5 cm above the benthos, and a rope and float system linking the BRUV to a surface buoy (Fig. 2a; Malcolm et al. 2007; Haggitt et al. 2013). All cameras were set to 960 video resolution, 60 frames per second, and a wide field of view to maximize battery life. Video focal width at the bait box was demarcated in 7.6-cm increments using alternating black and white colored tape under the bait box and extending along the bait-pole to accommodate potential image elongation associated with the wide camera setting; fish at the extreme edges of the camera frame were not binned until they approached the bait box (Fig. 2b). Each camera was bolted to the bait-pole such that fish could be viewed in a horizontal orientation to the benthos. The field of view was standardized to a distance of approximately 3 m in front of the camera for analysis to minimize the effects of water clarity on estimates of size and relative abundance.

Three to four pounds (1.4–1.8 kg) of Atlantic mackerel, *Scomber scombrus*, were inserted into the bait box for each deployment to attract piscivorous fish to a viewing area in front of the camera. Two-thirds of the bait was chopped and inserted into a mesh bag within the bait box to ensure fish could not completely consume the bait before the end of a recording period. Additional whole fish were added to the bait box for each deployment.

Between June and September 2019, a total of 20 BRUV deployments were conducted across the four sites. Sampling was restricted to calm days with ocean swells of 0.3 m or less; all units were deployed during daylight hours between 08:00 and 16:00 h to prevent any biases associated with diurnal behavior (Haggitt et al. 2013; Schultz et al. 2018). BRUVS were consecutively deployed from a small vessel and retrieved 45 min after the final deployment. This allowed for a 15-min soak period and at least 30 min of overlapping video across all four recordings for comparison (Folpp et al. 2013; Harasti et al. 2015). The desired products for analysis were four 30-min video samples with uniform field conditions across all sites (time of day, tide, swell, wind, cloud cover, etc.).

Video files were manually analyzed by a single reviewer using the open-source VLC™ (VideoLAN Client) media player to ensure video analysis and related bias was consistent across all replicates. Visibility was estimated directly from the

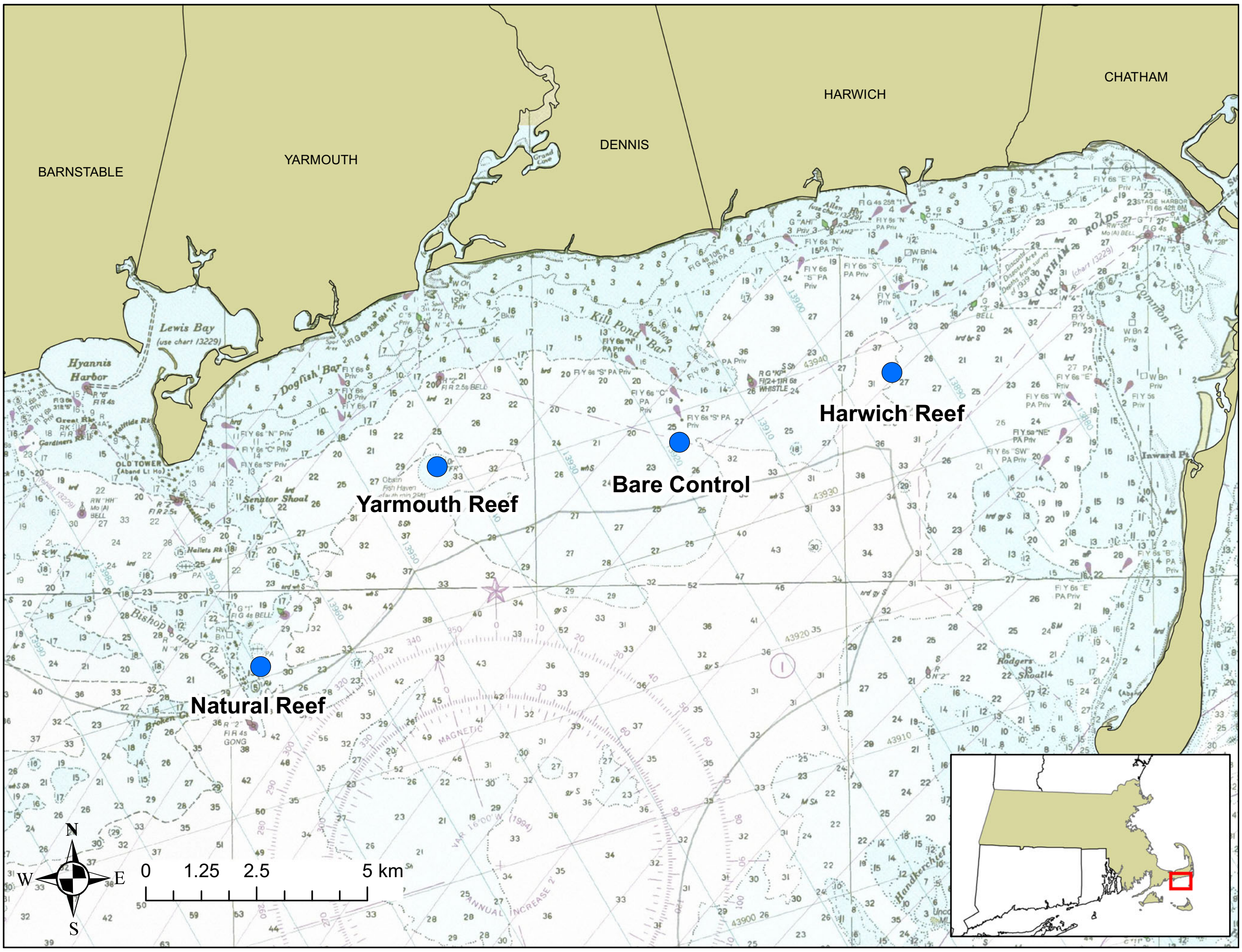


Fig. 1 Study sites within Nantucket Sound, MA. Left to right: natural reef (41.56829, −70.24202), Yarmouth reef (41.60832, −70.192077), bare control (41.61255, −70.12772), and Harwich reef (41.625972, −70.06944). Mean low water (MLW) marks for natural reef, Yarmouth reef, bare control, and Harwich reef measured 7.10 m, 10.63 m, 9.64 m, and 10.18 m, respectively. Sites were located at least 5 km from each other

video using the bait box (80 cm from camera) as a guide. Time to first fish sighting was documented. Still frames for analysis were captured from each 30-min recording in 30-s increments for a total of 60 analyzed frames per recording. Additional examination of up to 5 s before and/or after a given still frame was allotted to ensure the profile of each fish was visible for measurement or to allow any obstructions (e.g., floating algae) to pass. The identity of each species of fish, an index of its relative abundance (MaxN), and quantitative length estimates of two species of economic significance, black sea bass and scup, were documented within each frame. MaxN is the maximum number of a given species of fish within the field of view at any one frame during a 30-min recording; this index was employed to prevent double counts of individual fish (Cappo et al. 2004; Malcolm et al. 2007). Due to the documented error in estimating exact fish length measurements from mono-H-BRUV recordings (Cappo et al. 2004; Folpp et al. 2013), fish of interest were binned by species into size ranges to distinguish between juvenile, undersized adult, and legally fishable adults. Specifically, black sea bass were binned into 0–7.6, 7.6–38, and over 38 cm and scup were binned into 0–7.6, 7.6–23, and over 23 cm. ImageJ 1.52a NIH software was used to aid fish binning when necessary.

Statistical Analyses

R software V3.5.1 was used for all statistical analyses. To determine finfish aggregation patterns, species richness, and diversity, one-way ANOVAs were conducted on the average time to fish arrival on site, number of species, and Shannon-Wiener diversity index (*H*) across all sites. Prior to analysis, assumptions of normality and homogenous variances were tested for all data; species number means were square-root-transformed and time to fish arrival means were log-transformed. Tukey HSD post hoc tests were conducted to

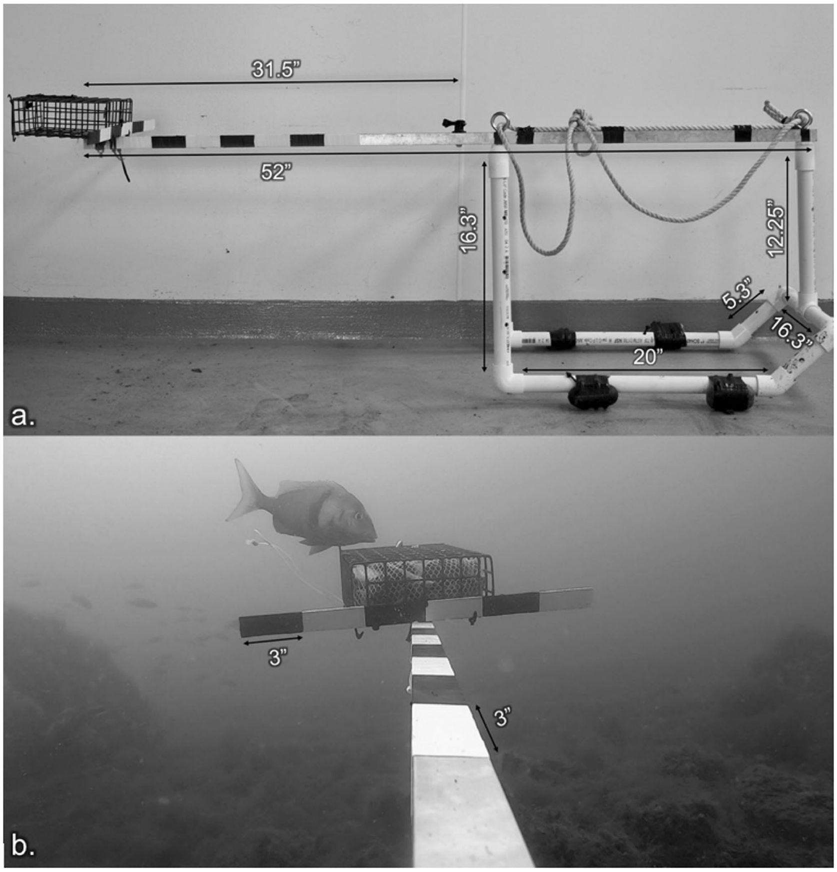


Fig. 2 **a** BRUV rig consisting of weighted PVC frame, aluminum bait-pole, and bait box located 0.8 m from GoPro housing attachment point. **b** Bait-pole and ruler attachment under bait box demarcated in 7.6-cm increments to aid fish length estimates

compare the means of all significant ANOVA results, and bar graphs were created to visualize the data.

A two-way permutational multivariate analysis of variance (PERMANOVA) was conducted to test for differences in relative abundances of fish as a function of reef age and seasonality using a Bray-Curtis dissimilarity measure. For each term in the analysis, 1999 permutations of the raw data units were computed to obtain *p* values. Non-metric multidimensional scaling (nMDS) was used to examine spatial patterns and SIMPER tests were conducted to determine the species most responsible for driving patterns.

Generalized linear models (GLMs) with negative binomial distributions were used to examine the relative abundances (total MaxN) of species identified by SIMPER as the drivers of abundance patters. Similarly, GLMs with negative binomial distributions were used to examine the age structures (MaxN of juveniles, undersized adults, and fishable adults) of these species. Date, temperature, tide, initial depth, and visibility were all tested as additive effects in the original models; the Akaike information criterion (AIC) was used for comparison to determine the most parsimonious models. One-way ANOVAs and subsequent pairwise comparisons were conducted on the most parsimonious models. Estimated marginal mean abundances were plotted and included date as a linear predictor to account for seasonal effects.

Results

A total of 14 finfish species were observed across the bare control, Harwich reef, Yarmouth reef, and natural reef from June to September 2019. Of these species, black sea bass, scup, bluefish (*Pomatomus saltatrix*), and dogfish (*Squalus acanthias*) were observed on all four sites. Finfish appeared on camera 93.4% faster (33 s) at reef sites than they did at the bare control (502 s) ($p < 0.0001$; Fig. 3a). Regarding species richness, 2.5 species were observed on the bare control, 3.4 species were observed on the Harwich reef, and 3.1 species were observed on both the Yarmouth and natural reefs on average ($p = 0.0777$; Fig. 3b). Regarding species diversity, the natural reef exhibited an average Shannon-Wiener value 26.7% higher than that of the bare control ($p = 0.0391$; Fig. 3c). The Yarmouth reef exhibited an average Shannon-Wiener value just 16.1% higher than that of the Harwich reef ($p = 0.3259$; Fig. 3c) and both values fell between those of the natural reef and bare control.

A two-way PERMANOVA revealed that the interaction of reef age and month impacted the relative abundance of species (MaxN) and resulted in distinct clustering of samples in the non-metric multidimensional scaling (nMDS) plot ($p = 0.0040$; Fig. 4). SIMPER identified scup and black sea bass as the most influential species in

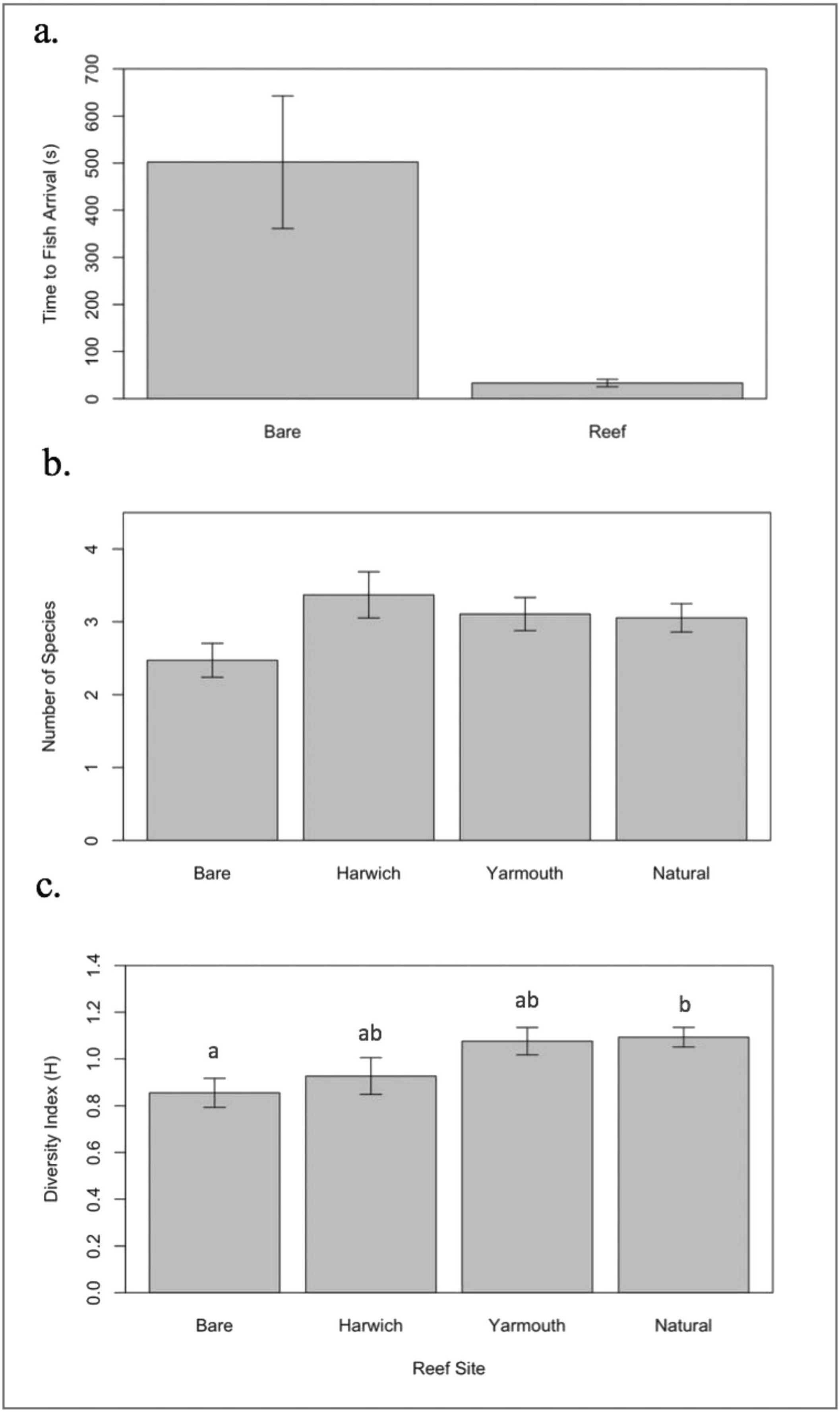

a.
Time to Fish Arrival (s)
0
100
200
300
400
500
600
700
Bare
Reef
b.
Number of Species
0
1
2
3
4
Bare
Harwich
Yarmouth
Natural
c.
Diversity Index (H)
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
a
ab
ab
b
Bare
Harwich
Yarmouth
Natural
Reef Site

Fig. 3 **a** Average time in seconds to finfish arrival at bare versus reef sites after BRUV deployment. **b** Average species richness across all sites. **c** Average Shannon-Wiener diversity index (*H*) across all sites. Sites are organized by increasing reef age from left to right. $N = 20$ and error bars represent one standard error of the mean

differentiating the fish assemblages across reef ages and months (Tables 1 and 2).

Black sea bass were 103.7% more abundant on the Yarmouth reef than on the Harwich reef ($p = 0.0257$; Fig. 5a). This corresponds to a 36.9% difference in total black sea bass abundance between the Yarmouth and natural reefs ($p = 0.3441$; Fig. 5a) compared with a 99.0% difference in total black sea bass abundance between the Harwich and natural reefs ($p < 0.0001$; Fig. 5a). Conversely, this corresponds to a 69.0% difference in total black sea bass abundance between the Yarmouth reef and bare control ($p = 0.0195$; Fig. 5a) compared with a 0.7% difference in total black sea bass abundance between the Harwich reef and bare control ($p = 1.000$; Fig. 5a). The model explained ~41% of the overall variation. This trend varied with black sea bass age structure, though there were too few juveniles observed on average to determine a trend ($p = 0.581$; Fig. 6a). There was a 62.9% difference in undersized adult black sea bass abundance between the Yarmouth and natural reefs ($p = 0.0077$; Fig. 6b) compared with a 103.0% difference between the Harwich and natural reefs ($p < 0.0001$; Fig. 6b). Finally, there was a 100.0% difference in fishable adult black sea bass abundance between the Yarmouth and natural reefs ($p = 0.0915$; Fig. 6c) compared with a 160.0% difference between the Harwich and natural reefs ($p = 0.0169$; Fig. 6c).

Scup did not exhibit the same trends in abundance. Scup were 23.4% more abundant on the Harwich reef than on the Yarmouth reef ($p = 0.9044$; Fig. 5b). This corresponds to a 79.6% difference in total scup abundance between the Yarmouth and natural reefs ($p = 0.0283$; Fig. 5b) compared with a 61.3% difference in total scup abundance between the Harwich and natural reefs ($p = 0.1534$; Fig. 5b). Conversely, this corresponds to a 21.2% difference in total scup abundance between the Yarmouth reef and bare control ($p = 0.9009$; Fig. 5b) compared with a 0.3% difference in total scup abundance between the Harwich reef and bare control ($p = 1.000$; Fig. 5a). The model explained ~21% of the overall variation. This trend varied with scup age structure, though there were too few juveniles observed on average to determine a trend ($p = 0.1274$; Fig. 7a). There was an 88.8% difference in undersized adult scup abundance between the Yarmouth and natural reefs ($p = 0.0400$; Fig. 7b) compared with a 55.7% difference between the Harwich and natural reefs ($p = 0.3724$; Fig. 7b). Finally, there was a 127.6% difference in fishable adult scup abundance between the Yarmouth and natural reefs ($p < 0.0001$; Fig. 7c) compared with a 144.3% difference between the Harwich and natural reefs ($p < 0.0001$; Fig. 7c).

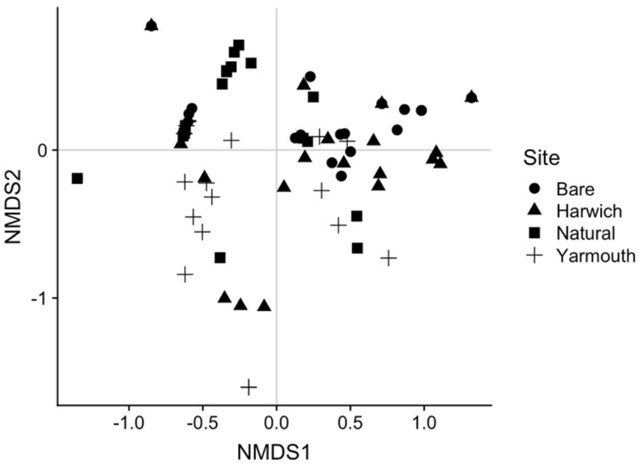


Fig. 4 nMDS ordination plot of finfish assemblages according to species MaxN averages across natural reef, artificial reefs, and bare control site sampled in Nantucket Sound from June to September 2019. Stress = 0.17

Discussion

The results of this study support the hypothesis that seasonal species richness and diversity are largely uniform across the sampled natural and artificial reefs in Nantucket Sound. There were negligible differences in species richness across all reefs and the bare control, while species diversity increased only slightly with reef age (Fig. 3). Therefore, community structure appears to be uniform throughout Nantucket Sound regardless of substrate type. This may imply that certain economically important fishes known for being reef-associated, such as black sea bass, are willing to cross sand barriers to move between reef types in the pursuit of food; this may also reflect the migratory nature of such fishes (Sedberry et al. 1998; Drohan et al. 2007). Future research into the distances black sea bass are willing to travel from reefs in the pursuit of food may help to inform how closely artificial reefs must be sited to boost habitat connectivity. Despite this uniformity of community structure throughout the Sound, fish arrival to bait was much faster at reef sites than it was at the bare control, implying that the species observed throughout the Sound tend to aggregate to or are produced on natural and artificial reef structure (Fig. 3). This supports assertions that artificial reefs increase the biomass of reef-associated prey species and provide refuge from predation and currents via structural complexity, thereby bolstering the region's fisheries stocks (Rooker et al. 1997; Baine 2001).

Contrary to our hypothesis, abundance varied with reef age. Black sea bass and scup were identified as the two greatest determinants of variation in total abundance across sites (Tables 1 and 2). Both species are fished commercially and recreationally and are subject to the U.S. Atlantic States

Table 1 Species contributions from SIMPER analysis (reduced dataset) for differences in relative abundance of species (MaxN) between sites. Average percent contributions to overall dissimilarity (δ_i), ratios of average to standard deviation of contribution (δ_i/SD (δ_i)), and ranks of most influential species (Rank) are included for all pairwise comparisons. Species regarded as most important contributors to the assemblage dissimilarity are shown in italics

Species	Harwich vs. bare			Harwich vs. natural			Harwich vs. Yarmouth		
	δ_i	δ_i/SD (δ_i)	Rank	δ_i	δ_i/SD (δ_i)	Rank	δ_i	δ_i/SD (δ_i)	Rank
Black sea bass	*9.96*	*0.82*	*2*	*13.48*	*1.17*	*2*	*11.05*	*1.12*	*2*
Scup	*38.78*	*1.59*	*1*	*39.96*	*1.88*	*1*	*29.92*	*1.43*	*1*
Tautog	0.52	0.43	9	0.51	0.49	9	0.99	0.56	14
Bluefish	2.54	0.56	6	0.98	0.54	6	1.69	0.51	9
Striped bass	0	0	4	0.26	0.21	10	0	0	6
Dogfish	3.46	0.7	10	2.5	0.97	12	3.67	0.75	10
Sea robin	0	0	3	0.32	0.22	4	0.43	0.31	4
Skate	0	0	12	0	0	3	0	0	3
Cunner	6.52	0.45	11	4.76	0.43	7	6.93	0.53	13
Butterfish	2.27	0.23	5	1.61	0.24	5	2.05	0.23	7
Winter flounder	0 .07	0.23	7	0.21	0.28	11	0.07	0.23	12
Summer flounder	0.24	0.32	8	1.01	0.71	13	1.15	0.23	11
Sand tiger shark	0	0	13	0.16	0.22	8	0.93	0.3	5
Blue runner	0	0	14	0	0	14	8.17	0.42	8
	Bare vs. natural			Bare vs. Yarmouth			Natural vs. Yarmouth		
	δ_i	δ_i/SD (δ_i)	Rank	δ_i	δ_i/SD (δ_i)	Rank	δ_i	δ_i/SD (δ_i)	Rank
Black sea bass	*14.77*	*1.27*	*2*	*11.22*	*1.18*	*2*	*11.01*	*1.18*	*2*
Scup	*36.92*	*1.57*	*1*	*33.35*	*1.46*	*1*	*31.97*	*1.51*	*1*
Tautog	0.16	0.23	6	0.68	0.41	14	0.54	0.48	14
Bluefish	0.63	0.39	12	0.86	0.4	6	0.19	0.32	6
Striped bass	28	0.21	4	0	0	9	0.24	0.21	12
Dogfish	2.48	1.13	7	3.54	0.79	13	1.49	0.73	9
Sea robin	0.34	0.23	5	45	0.32	4	0.55	0.36	13
Skate	0	0	11	0	0	3	0	0	7
Cunner	0	0	3	1.67	42	7	0.84	46	3
Butterfish	0	0	13	0	0	12	0	0	5
Winter flounder	0.17	0.23	8	0	0	5	0.15	0.22	4
Summer flounder	1.05	0.72	9	0.12	0.22	8	0.93	0.7	11
Sand tiger shark	0.16	0.23	10	1.02	0.3	10	0.56	0.4	8
Blue runner	0	0	14	8.41	0.43	11	7.24	0.43	10

Marine Fishery Commission's interstate fisheries management plan and seasonal, size, and bag limits regulated by the Massachusetts Division of Marine Fisheries. Black sea bass are strongly associated with structurally complex habitats including rocky reefs, cobble and rock fields, exposed stiff clay, and mussel beds; though migratory, they exhibit enduring habitat fidelity to their structure of choice (Sedberry et al. 1998; Drohan et al. 2007). A coastwide quota increase for black sea bass was issued in Massachusetts in 2017 and both recreational and commercial fisheries have achieved their extractive limits in the years since, perhaps calling for the development of additional habitat to support the habitat requirements of the stock in southern New England. Unlike black sea bass, scup do not exhibit strong association to structurally complex habitats and can be found schooling in open water or on sandy bottoms; however, they are frequently observed seeking food or shelter on or near structures such as rocky reefs and mussel beds (Steimle et al. 1999).

Total black sea bass abundance on the younger Harwich reef more closely resembled that of the bare control, while abundance on the older Yarmouth reef more closely resembled that of the natural reef (Fig. 5). This trend was reflected in age sub-categories, with both undersized and fishable adults increasing in abundance as reef age increased (Fig. 6). Large variances prevent strong conclusions from being drawn regarding juvenile abundance (Fig. 6). However, on average, juveniles were least abundant on the Harwich reef, whereas they were most abundant on the Yarmouth reef. This trend

Table 2 Species contributions from SIMPER analysis (reduced dataset) for differences in relative abundance of species (MaxN) between months. Average percent contributions to overall dissimilarity (δ_i), ratios of average to standard deviation of contribution (δ_i/SD (δ_i)), and ranks of most influential species (Rank) are included for all pairwise comparisons. Species regarded as being important contributors to the assemblage dissimilarity are shown in italics

Species	June vs. July			June vs. August			June vs. September		
	δ_i	δ_i/SD (δ_i)	Rank	δ_i	δ_i/SD (δ_i)	Rank	δ_i	δ_i/SD (δ_i)	Rank
Black sea bass	*9.35*	*1.21*	*2*	*11.18*	*1.1*	*2*	*10.53*	*1.17*	*2*
Scup	*25.49*	*1.34*	*1*	*34*	*1.73*	*1*	*34.51*	*1.8*	*1*
Tautog	0.55	0.39	9	0.33	0.34	10	0.14	0.37	14
Bluefish	0.27	0.19	10	1.32	0.45	6	0.48	0.35	10
Striped bass	0.61	0.35	6	0.6	0.33	4	0.38	0.31	6
Dogfish	3.31	1.01	7	3.23	0.93	7	2.35	81	7
Sea robin	1.38	0.57	13	125	0.52	5	0.87	0.48	4
Skate	0	0	5	0	0	11	0	0	5
Cunner	6.04	0.49	3	0.35	0.36	9	0.25	0.34	12
Butterfish	5.05	0.36	11	4.87	0.36	3	3.59	0.35	11
Winter flounder	0.43	0.4	12	0.37	0.35	12	0.27	0.34	9
Summer flounder	0.4	0.35	4	0.28	0.4	8	0.37	0.46	3
Sand tiger shark	0.72	0.3	8	0	0	13	0.08	0.26	13
Blue runner	0	0	14	0	0	14	9.92	0.47	8
	July vs. August			July vs. September			August vs. September		
	δ_i	δ_i/SD (δ_i)	Rank	δ_i	δ_i/SD (δ_i)	Rank	δ_i	δ_i/SD (δ_i)	Rank
Black sea bass	*12.02*	*1.08*	*2*	*10.55*	*1.07*	*2*	*10.93*	*1*	*2*
Scup	*35.9*	*1.62*	*1*	*35.3*	*1.54*	*1*	*35.48*	*1.39*	*1*
Tautog	0.8	0.48	9	0.47	0.43	14	0.35	0.44	14
Bluefish	1.55	0.46	6	0.61	0.36	9	1.12	0.48	6
Striped bass	0	0	4	0	0	6	0	0	4
Dogfish	3.02	0.71	3	2.3	0.66	4	2.31	0.64	12
Sea robin	0.15	0.19	13	0.11	0.18	12	0	0	3
Skate	0	0	12	0	0	13	0	0	10
Cunner	5.9	0.47	7	4.25	0.43	3	0	0	13
Butterfish	0	0	11	0.08	0.26	7	0.08	0.25	5
Winter flounder	0.05	0.21	5	0.04	0.2	10	0	0	7
Summer flounder	0.62	0.48	8	0.56	0.53	11	0.5	0.57	8
Sand tiger shark	0.77	0.28	10	0.51	0.3	5	0.08	0.25	9
Blue runner	0	0	14	9.87	0.47	8	9.63	0.47	11

may in part explain the greater number of adults present on the Yarmouth reef and support the claim that older artificial reefs exhibit greater balance between attraction and production of finfish than younger artificial reefs. Particularly in temperate environments, reefs may undergo years of community succession prior to approaching the diversity and stability of natural reefs (Becker et al. 2017). Older, more stable reefs are therefore more likely than younger reefs to provide the structurally complex protection and food biomass necessary to support juvenile finfish settlement and survival to adulthood (Coll et al. 1998; Relini et al. 2002; Barber et al. 2009; Folpp et al. 2013).

These abundance trends were not reflected in scup populations. As with black sea bass, total scup abundance at the Harwich reef more closely resembled that of the bare control (Fig. 5). However, scup abundance was lowest on the Yarmouth reef, a trend that was again reflected in both the undersized and fishable adult sub-categories (Fig. 7). Juveniles presented large variances across sites; yet, as with the black sea bass, the fewest number of juveniles was observed on the Harwich reef, again implying that young reefs may not yet be able to support juvenile production (Fig. 7). The relative absence of adults on the Yarmouth reef is interesting given the greater number of juveniles present. As black sea bass are known predators of scup (Bowman et al. 2000), their success on the Yarmouth reef may in part be explained by increases in predation on scup juveniles and young adults. Additionally, because scup life histories are not strictly tied to

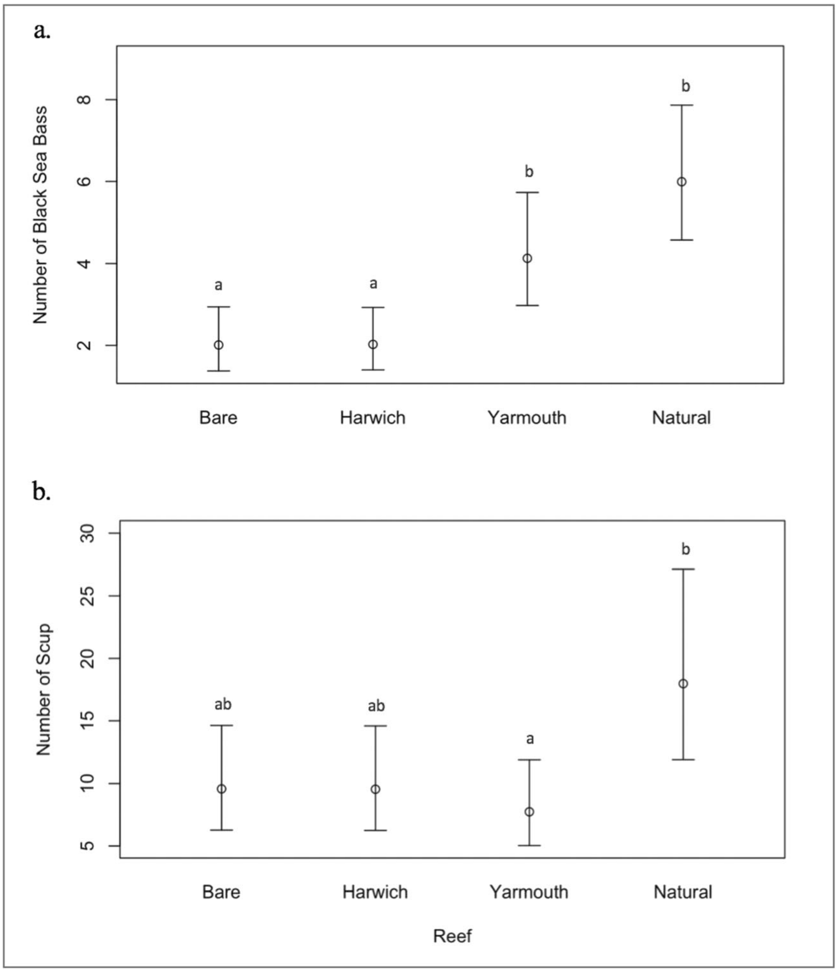
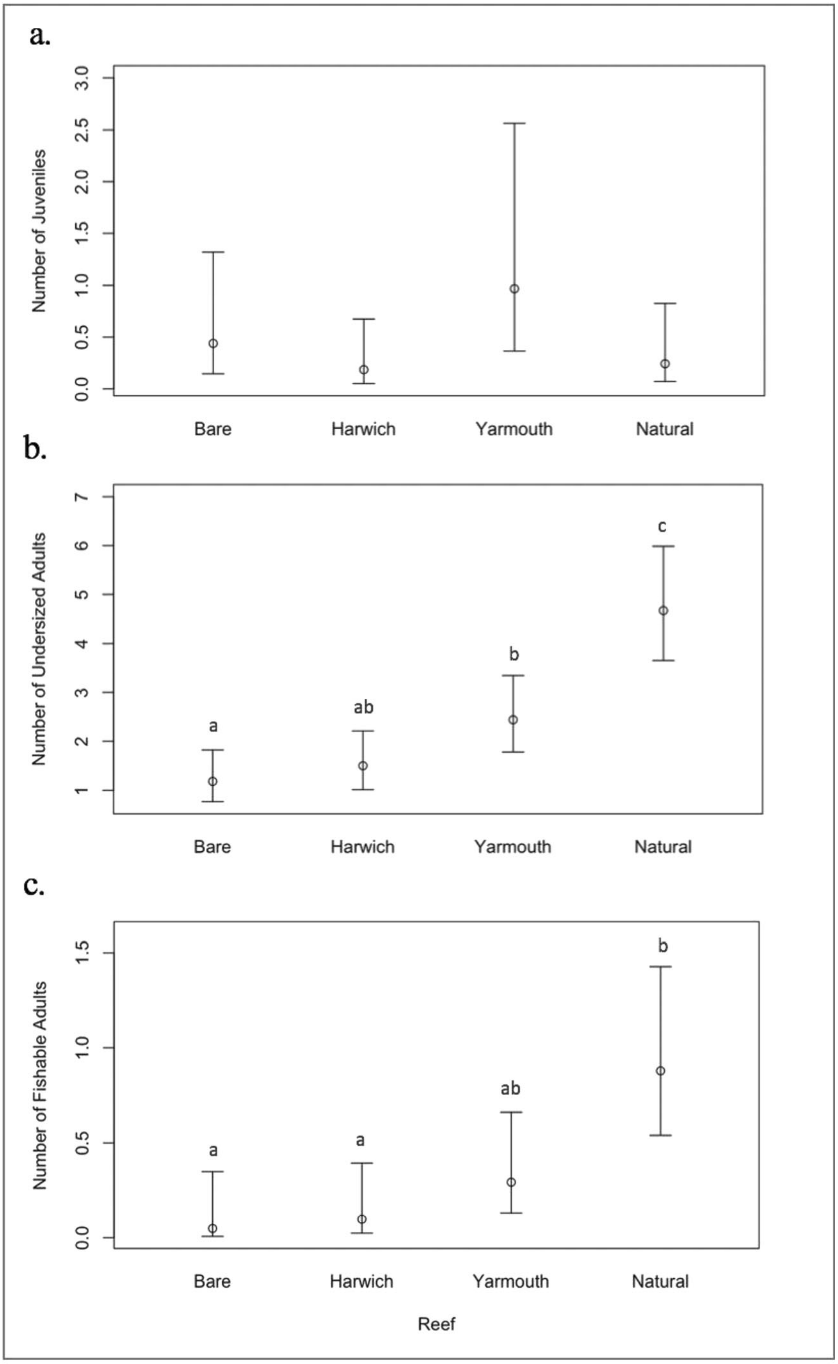


Fig. 5 Estimated marginal mean abundances of **a** black sea bass and **b** scup across natural reef, artificial reefs, and bare control site. Sites are organized by increasing reef age from left to right. $N = 20$ and error bars represent one standard error of the mean

presence of structure (Steimle et al. 1999), their abundances may vary more freely across bare and reef sites than strictly reef-associated fishes. Finally, sample error may have factored into these results. Visibility was consistently worse (reduced by 0.25–1 m) on the Yarmouth reef than on the other sites throughout the sample months; while this may not have severely affected the ability to discern darkly colored black sea bass during video analysis, it may have inhibited the view of lighter scup and impacted overall count data. Future research should be conducted on sites with more uniform visibilities to determine how greatly this may have impacted the present study.

Additional error may have factored into this study. Although BRUVS were deployed within 3 m of reef structure during any given sampling period, the cameras did not always capture reef structure within their frame. As such, species strictly affiliated with structure, such as cunner (*Tautogolabrus adspersus*), were only observed when reefs fell within the camera frame and when visibility permitted. This may have skewed species richness or diversity indices. Future research should attempt to more directly include structures of interest within the camera frame. Bait box elevation and camera orientation may have further impacted species counts by limiting view of groundfish or species unable to access bait elevated off bottom sediments, such as flounder. Finally, the artificial reefs monitored in this study were constructed of different materials: Yarmouth was constructed of ballasted tires and Harwich was constructed of repurposed concrete rubble. Folpp et al. (2013) noted that differences in artificial structures may yield associated differences in fish assemblages, implying that reef age may not have been the only factor contributing to the observed differences in assemblages across reefs in the present study. Tire reefs have historically exhibited less success than concrete reefs due to their propensity to shift during ocean storm events (AGSMFC 2004). However, the Yarmouth reef has withstood seven large hurricane and tropical cyclone events impacting Cape Cod since its deployment, including Hurricanes Gloria and Bob; monitoring did not reveal any tire shifting that may have been associated with these extreme weather events (Schwartz and Murray 2013). The Yarmouth reef's apparent success despite its tire composition may therefore lend more credence to the assertion

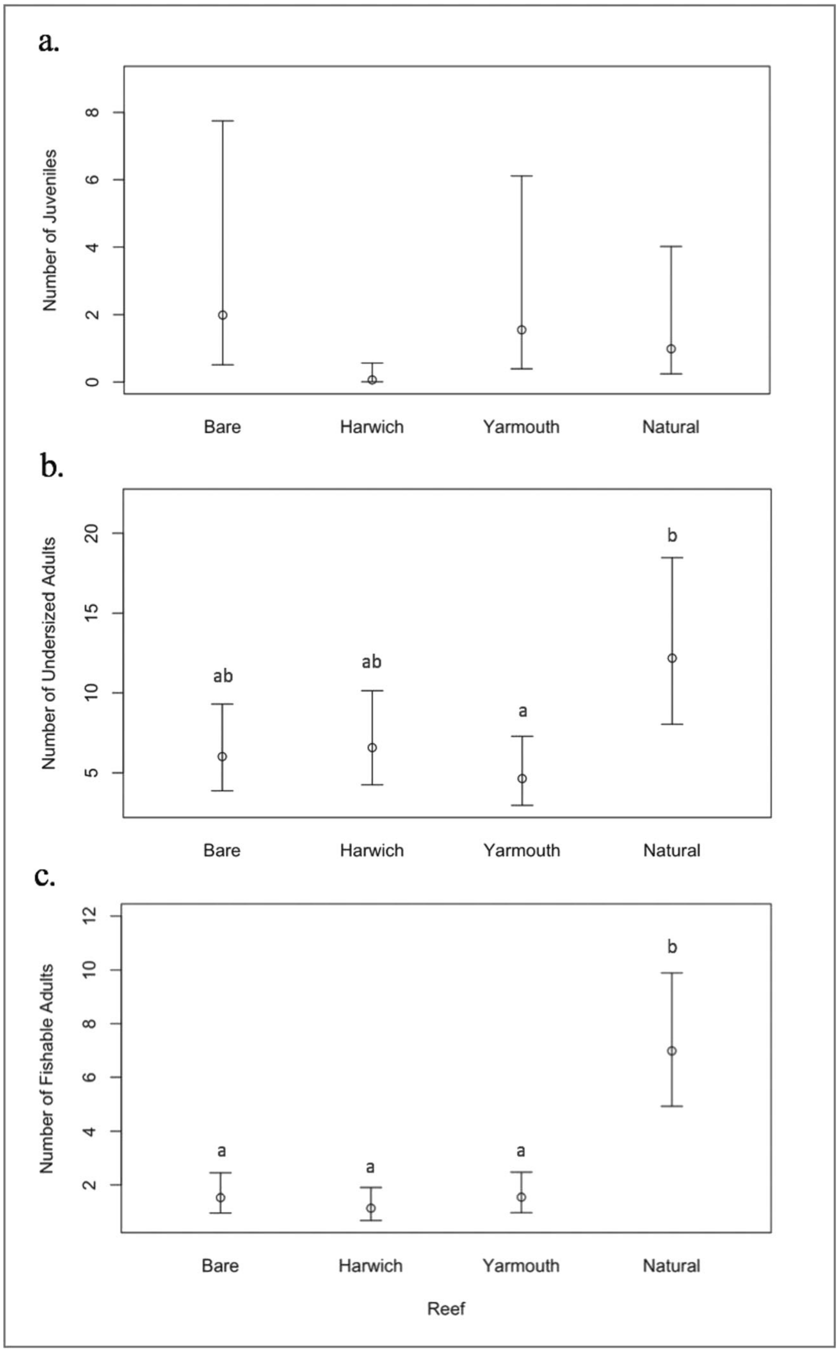
Fig. 6 Estimated marginal mean abundances of **a** juvenile, **b** undersized, and **c** fishable black sea bass across natural reef, artificial reefs, and bare control site. Sites are organized by increasing reef age from left to right. $N = 20$ and error bars represent one standard error of the mean



that increasing reef age subsequently increases community stability and fish abundance. Such stability may stem from the fact that the Yarmouth reef's tires were ballasted with concrete and are less likely to shift during storm events, thereby allowing for community stabilization.

While artificial reefs provide distinct benefits to reef-associated fishes, these benefits apparently continue to accrue over a long timescale. Though adult fishes may rapidly colonize young reefs, increasing reef age has been linked to increased species richness, diversity, and production of juveniles (Coll et al. 1998; Relini et al. 2002; Folpp et al. 2013). Many coastal management and mitigation monitoring plans require a minimum of 5 years of post-deployment monitoring to determine the effectiveness of artificial reefs, yet it may be prudent to continue monitoring reefs well beyond this timeframe to fully record the benefits conferred to a region's fisheries stocks (Barber et al. 2009). This is particularly relevant when the assessed fish stocks are migratory species with only seasonal presence. It is also important to note that neither

Fig. 7 Estimated marginal mean abundances of **a** juvenile, **b** undersized, and **c** fishable scup across natural reef, artificial reefs, and bare control site. Sites are organized by increasing reef age from left to right. *N* = 20 and error bars represent one standard error of the mean



artificial reef outperformed the natural reef in terms of species richness, diversity, or abundance in this study. This is ultimately a positive quality, as artificial reefs should be viewed as support for rather than direct surrogates of naturally occurring reefs and the coordinates of most artificial reefs are publicly available to fishing communities. The Massachusetts Division of Marine Fisheries has implemented regulatory prohibitions on commercial fishing activity on newly deployed artificial reefs in an effort to control early extractive pressure; perhaps these prohibitions may be extended to allow for communities to stabilize after fishes are initially attracted to the reefs both within the state and more broadly in national and international reef programs. Furthermore, such artificial reefs are most effective when they increase habitat connectivity, which serves to disperse rather than aggregate stocks (Smith et al. 2015; Becker et al. 2017). Future management plans may also consider increasing artificial reef cover in Nantucket Sound and similar regions largely characterized by sandy substrate and limited structured habitat. Finally, management plans should consider how artificial reefs may fit into greater climate change mitigation strategies by restoring ecological function and helping to mitigate species migrations in the wake of elevated sea surface temperatures.

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