



Six decades of change in pollution and benthic invertebrate biodiversity in a southern New England estuary

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ABSTRACT

Pollution has led to a decline of benthic invertebrate biodiversity of Narragansett Bay, raising questions about effects on ecosystem functions and services including shellfish production, energy flow to fishes, and biogeochemical cycles. Changes in community composition and taxonomic distinctness (biodiversity) were calculated from the 1950s—when quantitative benthic invertebrate data first became available—to 2015. Change in community composition of the bay was correlated with changes in dissolved inorganic nitrogen, dissolved oxygen, and sediment contaminants. A mid-bay reference site showed moderate changes in community composition but no change in biodiversity. In contrast, a more impacted site in the upper bay showed substantial differences in community composition over time and a decline in taxonomic distinctness. Bay-wide, as inputs of some stressors such as nutrients and sediment contaminants have declined, there are signs of recovery of benthic biodiversity but other stressors such as temperature and watershed development are increasing.

1. Introduction

Biodiversity supports the functioning of ecosystems and the services they provide to people (Worm et al., 2006; Palumbi et al., 2009; Fautin et al., 2010; Solan et al., 2012; Gouletquer et al., 2014; Dornelas et al., *in press*). Ecosystem functions and services driven by benthic biodiversity include seafood for human consumption, water filtration (water quality), bioturbation and bio-irrigation (supporting nutrient cycling), shoreline protection, habitat for other species, and cultural services such as recreation (Snelgrove et al., 1997; Snelgrove, 1998, 1999; Levin et al., 2001; Weslawski et al., 2004; Gouletquer et al., 2014).

Marine ecosystems around the world have experienced rapid declines in biodiversity as a result of multiple stressors (Snelgrove et al., 2004; Jackson, 2008; Worm et al., 2006; Fautin et al., 2010; McCauley et al., 2015) and quantifying these changes has been recognized as a crucial research need (Fautin et al., 2010; Dornelas et al., *in press*). Estuarine functions are affected when benthic species losses lead to less food available for fishes, fewer large bioturbators, fewer suspension feeders, loss of reef or mat habitat, or collapse of biological interactions (Gouletquer et al., 2014). Loss of rare species reduces ecosystem functioning, productivity, and the ability to respond to environmental perturbations (Micheli and Halpern, 2005; Mouillat et al., 2013; Obst et al., 2017). Long-term benthic community studies have contributed to

an understanding of how the sum of multiple anthropogenic factors over long periods of time has adversely affected biodiversity. Factors implicated include eutrophication and hypoxia (Kemp et al., 2005; Pranovi et al., 2008; Reise et al., 2008; Krann et al., 2011), warming waters (Callaway et al., 2007; Shojaei et al., 2016), commercial fishing (Callaway et al., 2007; Trott, 2016), contaminants and combinations of these (Obst et al., 2017). Benthic communities are good integrators of these cumulative stressors (Obst et al., 2017).

This article describes how species biodiversity and community composition of the soft-bottom benthic invertebrate macrofaunal community of Narragansett Bay has changed over the past six decades and relates changes, where possible, to anthropogenic drivers. Sampling methods that allowed such quantitative comparisons of benthic invertebrates in the bay began in the 1950s. Recently, in an effort to reduce eutrophication and hypoxia, nutrient loads to the bay have been reduced, including a 50% reduction of total nitrogen input from wastewater treatment facilities (WWTF) that occurred 2005–2013 (NBEP, 2017). Additionally, inputs of metals, petroleum hydrocarbons, and synthetic organic contaminants have declined in recent years while other stressors (e.g., water temperature, watershed development) are increasing (NBEP, 2017). Questions arise as to what the effect will be on the estuarine ecosystem. This article explores whether it is possible to detect a benthic response to these changes.

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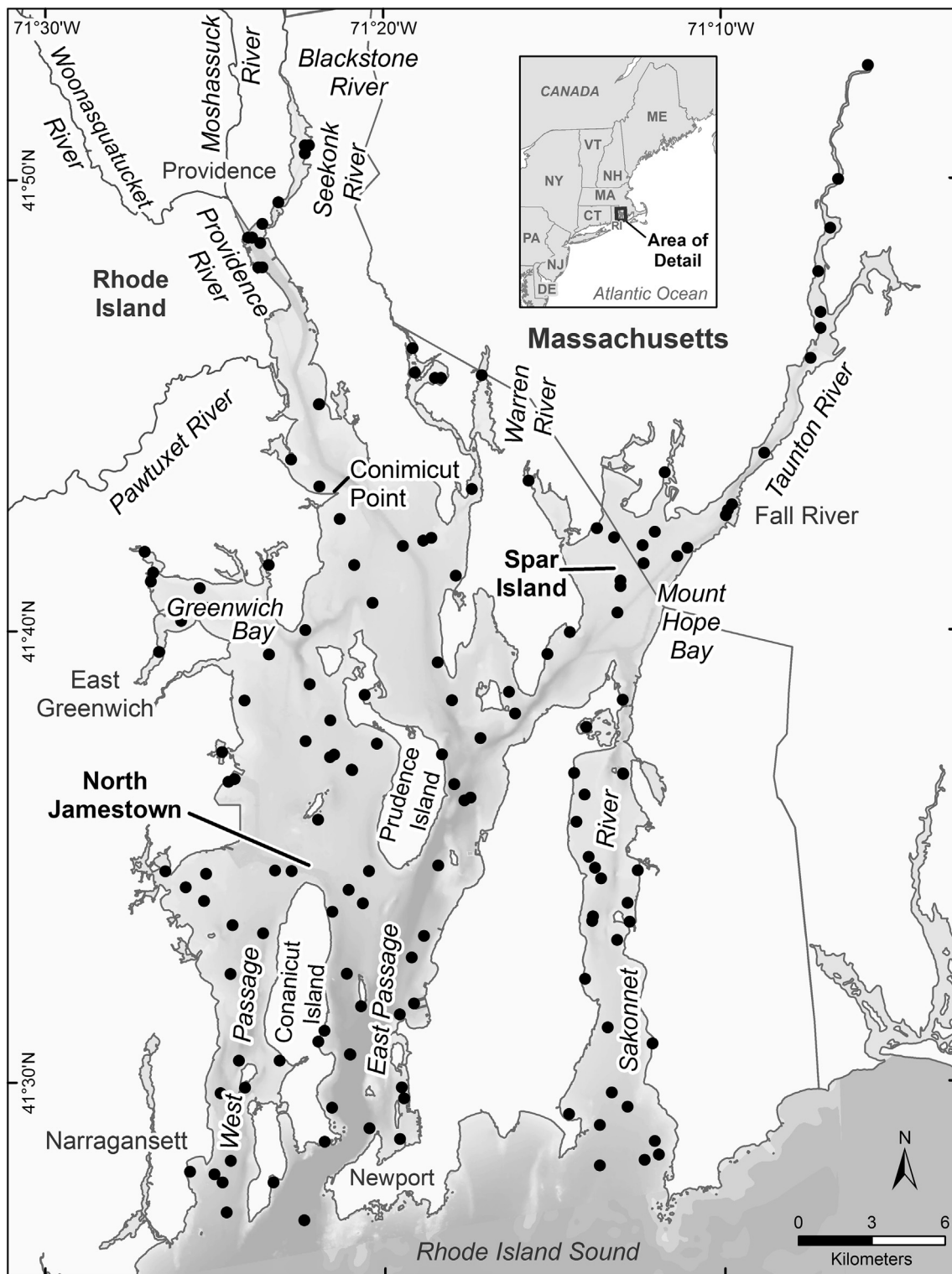


Fig. 1. Map of 166 USEPA stations in Narragansett Bay, Rhode Island and Massachusetts, 1990–2015, showing the location of the North Jamestown and Spar Island sites.

2. Methods

2.1. Study area

Narragansett Bay is a temperate northeastern U.S. estuary in Rhode Island and Massachusetts located at the northern end of the Virginian Biogeographic Province (Fig. 1). Benthic invertebrate biodiversity in

the bay stems from a mix of warm temperate species of the Virginian Province and Arctic-boreal species more common in the Acadian Biogeographic Province to the north (Hale, 2010), continental shelf species that extend up into the deep East Passage (Pratt, 1992), and rocky shore habitats at the mouth of the bay that are not common along the southern New England coast (French et al., 1992). The benthic system is closely coupled with the pelagic system and is an important part of the

bay ecosystem (Nixon et al., 2009; NBEP, 2017). Benthic communities support ecosystem functions and services such as shellfish production and shoreline protection that are important to people using the bay and living on the surrounding shores (Snelgrove, 1998, 1999; Weslawski et al., 2004). Over time, natural and human-driven forces have caused changes in biodiversity, resulting in changes to those functions such as trophic transfer to demersal fishes (Collie et al., 2008) and sediment bioturbation (Hale et al., 2016), and nutrient cycling (Nixon et al., 2009).

The 379 km² estuary has a 4421-km² watershed with (in 2010) over 1.9 million people, with a mean population density of 434 people per km² (NBEP, 2017). In 2014, 14% of the watershed had impervious surfaces, 37.7% was urbanized, 39.6% forested, and only 4.7% agricultural (NBEP, 2017). Mean depth of the bay is 8.3 m and mean tidal range is 1.1–1.4 m (Pilson, 1985). Freshwater inputs are mainly in the upper, northern end of the bay. The bay has a mean salinity of 27–33 (Pilson, 1985) with relatively small areas of oligohaline (salinity 0–5) or mesohaline (salinity 5–18) waters. Flushing time of the bay varies from 10 to 40 days with a mean of 26 days (Pilson, 1985). The bottom is mostly glacially-derived sediments, with clayey silt and sand-silt-clay predominant (McMaster, 1960). Bottom sediments with > 50% silt-clay occur primarily in the upper and mid-bay area (Murray et al., 2007). The benthos of the bay constitutes the largest stock of living organic carbon and the bottom sediments are the largest reservoir of total carbon (Frithsen, 1989). The distribution of benthic animals and habitats in the bay were described by Pratt (1992) and French et al. (1992). The bay has had a wide variety and long history of human-induced stressors (Nixon and Fulweiler, 2012; NBEP, 2017; Supplemental Table S1).

2.2. Data

2.2.1. National coastal monitoring programs

We used the U.S. Environmental Protection Agency's (USEPA) national coastal assessment programs, 1990–2015 for analyses of the bay. These surveys used a random sampling design that covered the entire bay, used standardized protocols for sample collection and analysis, collected concurrent physical and chemical data from the water column and the sediments, and have continued over 25 years. These features made it possible to conduct statistical inferences about the condition of the benthos of the entire bay over time. We extracted data for 166 Narragansett Bay stations from three monitoring programs: the Environmental Monitoring and Assessment Program (sampled annually from 1990 to 1993; EMAP, 2017), the National Coastal Assessment (sampled annually from 2000 to 2006; NCA, 2017), and the National Coastal Condition Assessment (launched in 2010, sampled again in 2015, and is scheduled to repeat every five years; NCCA, 2017). Henceforth, we refer to these three combined datasets as the “USEPA dataset” (Fig. 1). Samples were collected during a summer index period (July through September) when certain stresses on the benthic communities (e.g., hypoxia, temperature) were at their seasonal maximums (EMAP, 2017; NCA, 2017). Benthic macroinvertebrate assemblages were sampled using a 0.04-m² Young-modified van Veen grab and samples were sieved with a 0.5-mm mesh screen (USEPA, 1995, 2010). Concurrent sediment samples were collected and analyzed for grain size, organic matter content, chemical contaminants (metals, petroleum hydrocarbons, synthetic organic compounds), and toxicity (EMAP, 2017; NCA, 2017). Surface (0.5 m) and bottom (0.5 m above the bottom) water column samples were analyzed for physical-chemical properties (e.g., temperature, salinity, pH, dissolved oxygen, nutrients). Data from the three programs are available online: 1990–1993 (USEPA, 2017a or Hale et al., 2002); 2000–2006 (USEPA, 2017a or Kiddon et al., 2018); and 2010–2015 (USEPA, 2017b); subsets are available in the Ocean Biogeographic Information System (OBIS, 2017) and BioTIME database (Dornelas et al., in press).

2.2.2. North Jamestown and Spar Island

To extend trends back four decades before the USEPA monitoring started in 1990, we supplemented the USEPA dataset with data from two sites that had comparable data from other studies (Hale et al., 2018b). We compared biodiversity trends at a control site in mid-bay (North Jamestown, 1957–2010), and at a more impacted one in Mount Hope Bay (Spar Island, 1975–2008). The North Jamestown site (Fig. 1) is a characteristic Narragansett Bay benthic site representing the typical mid-bay, silt-clay bottom *Nephtys incisa*-*Nucula annulata* community. This community type was first described in Long Island Sound and Buzzards Bay by Sanders (1956, 1958). North Jamestown has been sampled by most of the benthic studies ever done in Narragansett Bay since Phelps (1958) and was used as the control field station for the University of Rhode Island's Marine Ecosystems Research Laboratory large mesocosm studies in the 1970s and 1980s (MERL, 2017). We compiled data from 12 studies (Phelps, 1958; Hale, 1974; Myers and Phelps, 1978; Rudnick, 1984; French et al., 1992; Shull, 2000; Schult, 2010; USEPA dataset [EMAP, 2017; NCA, 2017; NCCA, 2017]; MERL, 2017; K. Perez unpubl.) where quantitative abundance data were comparable with the USEPA dataset (same 0.5 mm sieve mesh size), sampled within 1 km of the site, and on a similar silt-clay, 7–9 m deep, bottom habitat type (French et al., 1992). The sampling window was generally the summer months (June through September). Taxonomic names were updated with, first ITIS (2017), and secondly, WoRMS (2017).

For a comparison with the reference site, we chose Spar Island in Mount Hope Bay because it has one of longest time series of benthic occurrence and abundance data in Narragansett Bay (MRI, 1994). Mount Hope Bay has experienced a wide variety and duration of anthropogenic changes from watershed development, the city of Fall River WWTF and combined sewer overflows (CSOs), and changes in the intake of cooling water and discharge of heated water from the Brayton Point Power Plant (Supplemental Table S1). It has higher levels of sediment TOC and metals than North Jamestown (Calabretta and Oviatt, 2008; USEPA, 2017a). MRI (1994) used Station F near Spar Island as a reference site for other stations around the power plant at the head of the bay. This station has the same clay-silt sediment (McMaster, 1960) and about the same depth (5–7 m) as the North Jamestown station. We merged the MRI species abundance data of Station F from July, August, and September with data from five other studies in Mount Hope Bay (French et al., 1992; USACOE, 1998; McCay et al., 2009; USEPA dataset [EMAP, 2017; NCA, 2017]) that had comparable quantitative abundance data, that sampled within 1 km, and had a similar bottom habitat type (French et al., 1992) and 5–7 m depth. We used only the summer months of the MRI data to be comparable with the sampling window of the other studies.

2.3. Statistical analyses

To characterize benthic communities of the bay, we plotted the rank order frequency diagram of the dominant species (Dominance Plot, PRIMER v.7). To examine species rareness, we calculated the number of occurrences of each taxon identified to species level in the entire Narragansett Bay USEPA dataset spanning 25 years and 166 stations and determined the number of species with only one occurrence (singletons) and only two occurrences (doubletons).

We ran a multidimensional scaling (NMDS; PRIMER v.7; Clarke et al., 2014) on the USEPA dataset to examine trends in community composition, 1990–2015. Abundance data were averaged by year, square root-transformed, and put into a Bray-Curtis similarity matrix. An ANOSIM was run on decadal differences and SIMPER (PRIMER v.7) was used to identify which species contributed the most to any significant differences found. To examine the influence of stressor variables on the variance of community composition in the USEPA stations, we ran a PRIMER BEST routine to compare the multidimensional pattern of species composition over the 25 years with the multidimensional

pattern of normalized stressor variables. The abiotic variables were: surface water dissolved inorganic nitrogen (DIN), surface water chlorophyll *a*, bottom water temperature, bottom water dissolved oxygen, sediment total organic carbon (TOC), number of sediment effects range medians (ERM, a measure of biological effects of sediment contaminants such as metals, pesticides, petroleum hydrocarbons; Long et al., 1995) exceeded, number of sediment effects range lows (ERL) exceeded, and human population of the watershed as estimated by NBEP (2017). Except for watershed population data, all these variables were measured concurrently with the biological samples.

Biodiversity includes species, genetic, functional, habitat, and ecosystem diversity (Gouletquer et al., 2014); unless otherwise noted, we use the term to refer to species biodiversity. To examine changes in species biodiversity, we calculated taxonomic distinctness (Δ^*) using abundance data (DIVERSE, PRIMER v.7; Clarke et al., 2014). Taxonomic distinctness uses the path length from each species to every other species in a sample through the taxonomic tree (species, genus, family, class, order, phyla) and measures the degree to which species in a sample are related to one another. A lowered Δ^* is indicative of deteriorating benthic community health, with fewer genera, families, classes, and phyla represented (Warwick and Clarke, 2001; Pranovi et al., 2008; Tweedley et al., 2015; Trott, 2016).

$$\Delta^* = \left[\sum_{i < j} \sum_{i < j} w_{ij} x_i x_j \right] / \left[\sum_{i < j} \sum_{i < j} x_i x_j \right]$$

where: w_{ij} = taxonomic distance from species *i* to species *j*

x_i = abundance of species *i*

x_j = abundance of species *j*

The lack of dependence of taxonomic distinctness on the number of species in the sample makes it useful for comparing studies that used different sampling methods (Clarke et al., 2014). Although this has been demonstrated by others, primarily in Europe (e.g., North Sea benthos by Callaway et al., 2007, the coast of the United Kingdom by Tweedley et al., 2015), we tested the assumption of independence in our three North American datasets by correlating taxonomic distinctness with the number of species in a sample (SAS v.9.4). Taxonomic distinctness has been used for assessing environmental quality (e.g., Tweedley et al., 2015), biogeography (e.g., Ronowicz et al., 2015) and conservation planning (e.g., Louzao et al., 2010).

Because the relationship of Δ^* to year did not meet the assumptions for an ANOVA, we used a nonparametric Kruskal-Wallis test (SAS, v9.4) to test for significant differences over time, 2000–2015. For this analysis, the 1990s were dropped because of low sample size. To determine the relationship between Δ^* and indicators of pollution, 1990–2015, we ran a nonparametric Kruskal-Wallis test (SAS, v9.4) with independent variables collected concurrently with the biological data. These variables (normalized) were surface water dissolved inorganic nitrogen and chlorophyll *a*, bottom water dissolved oxygen, sediment total organic carbon, and number of sediment ERLs and ERMs exceeded.

The same NMDS and taxonomic distinctness techniques that were used on the USEPA dataset were applied to the North Jamestown and Spar Island datasets. We ran the PRIMER BEST routine to compare the multidimensional pattern of species composition with the multidimensional pattern of two stressor variables: watershed population (NBEP, 2017) and water temperature (Fulweiler et al., 2015). For North Jamestown, we used the entire Narragansett Bay watershed and for Spar Island, just the Mount Hope Bay watershed.

3. Results

3.1. Biodiversity status and trends for Narragansett Bay

The USEPA dataset spanning 25 years and 166 stations included 561 taxa. Five species dominated numerical abundance (Table 1, Fig. 2). Like other soft-bottom benthic studies (Gray and Elliott, 2009; Trott, 2016), most of the taxa in the USEPA study were rare, as shown by the

Table 1

Twenty most prevalent (occurred at the most stations) and twenty most abundant benthic species (> 0.5 mm) in Narragansett Bay, USEPA dataset, 1990–2015.

Species	Occurrences ^a	Species	Abundance ^b
<i>Mediomastus ambiseta</i>	94	<i>Ampelisca abdita</i>	715,482
<i>Nucula proxima</i>	88	<i>Streblospio benedicti</i>	284,763
<i>Polydora cornuta</i>	86	<i>Mediomastus ambiseta</i>	267,445
<i>Nephtys incisa</i>	74	<i>Nucula proxima</i>	175,432
<i>Tharyx acutus</i>	73	<i>Spiochaetopterus costarum</i>	145,441
<i>Streblospio benedicti</i>	66	<i>Mulinia lateralis</i>	96,063
<i>Nassarius trivittatus</i>	66	<i>Mya arenaria</i>	85,400
<i>Ampelisca abdita</i>	52	<i>Mytilus edulis</i>	84,775
<i>Glycera americana</i>	48	<i>Tharyx acutus</i>	79,550
<i>Pectinaria gouldi</i>	45	<i>Crepidula fornicata</i>	75,025
<i>Acteocina canaliculata</i>	45	<i>Nucula annulata</i>	64,250
<i>Eumida sanguinea</i>	44	<i>Leptocheirus pinguis</i>	51,125
<i>Spiochaetopterus oculatus</i>	43	<i>Polydora cornuta</i>	50,400
<i>Scoloplos robustus</i>	43	<i>Spiochaetopterus oculatus</i>	48,900
<i>Crepidula fornicata</i>	42	<i>Ampelisca vadorum</i>	35,000
<i>Ninoe nigripes</i>	40	<i>Schizobranchia insignis</i>	31,500
<i>Tellina agilis</i>	40	<i>Nephtys incisa</i>	27,338
<i>Eusarsiella zostericola</i>	37	<i>Ennucula tenuis</i>	27,188
<i>Pholoe minuta</i>	37	<i>Crepidula plana</i>	25,100
<i>Unciola irrorata</i>	37	<i>Microdeutopus gryllotalpa</i>	22,374

^a At 166 stations.

^b At 166 stations, mean abundance in a 0.04 m² grab, 0.5 mm mesh sieve.

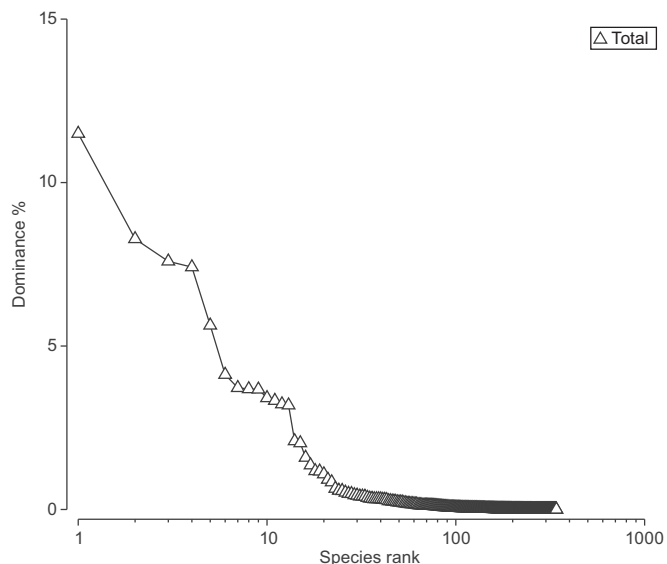


Fig. 2. Percent dominance for Narragansett Bay benthic species in the USEPA dataset, 1990–2015.

long tail to the right in the species dominance plot (Fig. 2). Rare species (< 20 occurrences in the 25-year dataset) comprised 88% of all taxa. Some of the species are rare because they were not efficiently captured by the bottom grabs used (e.g., hydroids attached to rocky bottoms; USEPA, 1995). There were 192 taxa (34%) that occurred only once (singletons) and 97 taxa (17%) that occurred only twice (doubletons). Three species that were rare in the USEPA dataset have not been found by other studies in the bay. *Imogene oculifera* and *Eurytellina nitens* are from the Gulf of Mexico and Caribbean and may represent taxa that have expanded their range northward; *Nephasoma pellucidum* is known from New Zealand (WoRMS, 2017).

The ANOSIM indicated significant overall changes in community composition ($R = 0.93$, $p = 0.001$; Table 2). The R statistic (range 0–1) is a measure of community dissimilarity where a low value indicates

Table 2

Analysis of similarity and community dissimilarity of Narragansett Bay benthic invertebrates in succeeding decades: USEPA dataset, North Jamestown, and Spar Island.

Dataset	Decade Groups	R	p	Species contributing the most to dissimilarity
USEPA	Overall	0.93	0.001	
	1990s–2000s	0.99	0.005	<i>Nucula annulata</i> , <i>Ampelisca abdita</i> , <i>Mediomastus ambiseta</i>
North Jamestown	2000s–2010s	0.93	0.036	<i>A. abdita</i> , <i>Spiochaetopterus</i> spp.
	Overall	0.23	0.04	
	1950s–1970s	NS	NS	
	1970s–1980s	NS	NS	
	1980s–1990s	NS	NS	
	1990s–2000s	NS	NS	
Spar Island	2000s–2010s	NS	NS	
	Overall	0.69	< 0.001	
	1970s–1980s	0.55	0.001	<i>A. abdita</i> , <i>M. ambiseta</i> , <i>N. annulata</i>
	1970s–1990s	0.77	0.018	<i>N. annulata</i> , <i>Spiochaetopterus</i> spp.
	1970s–2000s	0.69	0.018	<i>M. ambiseta</i> , <i>A. abdita</i>
	1980s–1990s	0.54	0.017	<i>A. abdita</i> , <i>M. ambiseta</i> , <i>Spiochaetopterus</i> spp.
	1980s–2000s	0.92	0.003	<i>M. ambiseta</i> , <i>A. abdita</i>
	1990s–2000s	NS	NS	

R = degree of dissimilarity. 0 = complete similarity, 1 = complete dissimilarity

p = probability

NS = not significant at $p < 0.05$.

Table 3

Results of BEST analysis (PRIMER v.7) linking multidimensional scaling plot of species abundance with multidimensional scaling plot of abiotic variables, Narragansett Bay.

Dataset	Stressor variables	r	p
USEPA	Surface water dissolved inorganic nitrogen, bottom water dissolved oxygen, and number of sediment effects range medians	0.57	0.03
North Jamestown	Watershed population, water temperature	0.42	0.01
Spar Island	Mount Hope Bay watershed population, water temperature	0.72	0.01

strong similarity between communities in different time periods and a high value indicates strong dissimilarity. Both succeeding decades were strongly different from the previous one. Species that contributed > 5% to the decadal differences were the bivalve *Nucula annulata*, the amphipod *Ampelisca abdita*, and the polychaetes *Mediomastus ambiseta* and *Spiochaetopterus* spp. The BEST routine identified a subset of stressor variables that led to the strongest correlations (Table 3) between the multidimensional pattern of species composition (Fig. 3a) and the multidimensional pattern of stressor variables (Fig. 3b).

The Kruskal-Wallis test to examine change in Δ^* over the 15-year period found no significant difference ($p = 0.08$, Fig. 4, Table 4). The Kruskal-Wallis test of the relationship of Δ^* to pollution factors (Table 5) was significantly correlated with a subset of pollution factors, primarily with indicators of eutrophication (water column chlorophyll *a*, sediment TOC) and sediment contaminants (#ERLs).

3.2. Comparison of temporal biodiversity trends at a reference site and an impacted site

3.2.1. North Jamestown control site

The NMDS (Fig. 5) and ANOSIM (Table 2) on the abundance data at the North Jamestown site, 1974–2010, showed moderate significant changes over all decades (global $R = 0.23$; $p = 0.04$). However, there were no year-to-succeeding year and no decade-to-succeeding decade

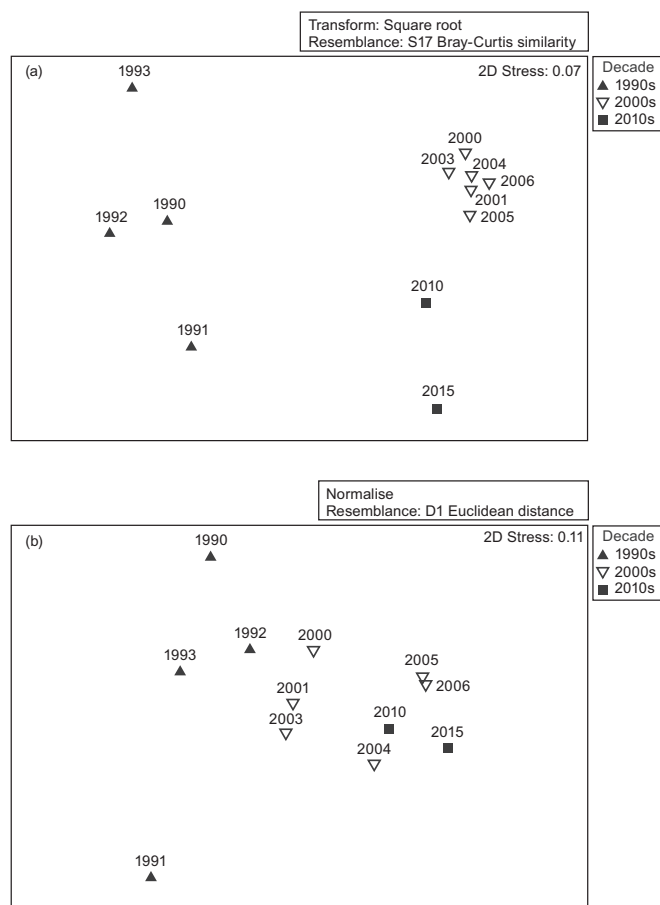


Fig. 3. Multidimensional scaling plots of USEPA data by decade, 1990–2015. (a) species abundance, (b) abiotic variables. Points that are closer together have a more similar species composition than points more distant. Decades were significantly different.

significant changes at $p < 0.05$. Community composition was correlated with watershed population and water temperature (Table 3). There were no significant differences ($p = 0.77$) in taxonomic distinctness (Δ^*) over decades (Fig. 6; Table 4) and none of the year-to-succeeding year changes were significant ($p = 0.51$).

3.2.2. Spar Island impacted site

In contrast, the NMDS (Fig. 7) and the ANOSIM of abundance data at Spar Island, 1975–2008, showed a large, significant change in community composition ($R = 0.69$; $p < 0.001$; Table 2). None of the year-to-succeeding year differences were significantly different ($p = 0.51$). However, there were significant decade-to-succeeding decade changes from the 1970s to the 1980s and from the 1980s to the 1990s (Table 2). The species contributing the most to the changes were *A. abdita*, *M. ambiseta*, *N. annulata*, and *Spiochaetopterus* spp. (Table 2). The average dissimilarity between decade pairs ranged from 50 to 79%. Community composition was correlated with watershed population and water temperature (Table 3). In contrast with the USEPA baywide dataset and the North Jamestown data, the Kruskal-Wallis test at Spar Island over five decades showed a significant ($p < 0.001$) decline in taxonomic distinctness (Fig. 8; Table 4).

For all three study areas, taxonomic distinctness was independent of number of species in the sample (USEPA dataset $r = -0.05$; $p = 0.50$; North Jamestown $r = -0.06$; $p = 0.80$; Spar Island $r = -0.01$, $p = 0.92$). This helps to validate comparisons among the 16 different studies used.

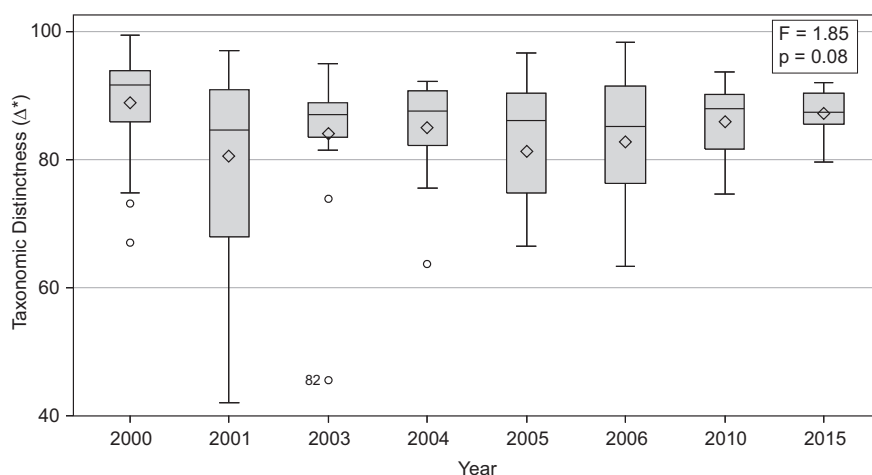


Fig. 4. Distribution of taxonomic distinctness, USEPA data, 2000–2015. The diamond is the mean, the line in the box of quartiles is the median, and the whiskers are the range. Changes in taxonomic distinctness (Δ^*) were not significant ($p = 0.08$).

4. Discussion

4.1. Current status of Narragansett Bay benthic communities

The mid-bay area has been characterized as a *Nephtys-Nucula* community (Pratt, 1992). In the USEPA 25-yr dataset for the entire bay, the bivalve *Nucula proxima* was in the top five for both prevalence and abundance but the polychaete *Nephtys incisa* was so only for prevalence. The polychaete *Mediomastus ambiseta* was more prevalent and abundant than both *Nucula* spp. and *M. ambiseta* were the species contributing the most to community similarity in the entire bay in the 1990s and the 2000s. Thus, the dominant benthic macrofaunal community type of the bay could be called a *Mediomastus-Nucula* community, as suggested by Frithsen (1989) and verified by Calabretta and Oviatt (2008) for the North Jamestown site. The small, short-lived *M. ambiseta* is better adapted to increased loading of organic matter than the larger *N. incisa* (Frithsen, 1989). However, in the 2010s, the polychaete *Spiochaetopterus costarum* was the largest contributor to community similarity, along with *M. ambiseta*. Species that contributed the most to the decadal differences (*Nucula* spp., the amphipod *Ampelisca abdita*, and *M. ambiseta*) have small body sizes and short life cycles and they expand and contract population sizes from year-to-year more readily than larger animals. *Ampelisca* can have a patchy distribution as a result of summer hypoxia and scouring of tube mats by winter storms and can quickly occupy new areas rich in organic matter if the level of dissolved oxygen is adequate (Diaz et al., 2008). There is some evidence that they are expanding their population into the lower Providence River in concordance with reductions in nitrogen loading (Shumchenia et al., 2016).

4.2. Temporal trends of Narragansett Bay benthic biodiversity

4.2.1. USEPA dataset

While community composition showed change over time in the USEPA dataset, taxonomic distinctness did not to a significant extent. The steady increase in taxonomic distinctness in the USEPA dataset after 2005 could potentially reflect a benthic response to the 50%

Table 5

Kruskal-Wallis test of relationship between taxonomic distinctness and independent pollution variables collected concurrently, USEPA dataset, Narragansett Bay, 1990–2010.

Independent variables	df	Chi-square	p
Water column chlorophyll <i>a</i>	3	13.11	0.004
Sediment TOC	3	10.63	0.014
# Sediment metal ERLs	3	9.77	0.021
# Sediment organic ERLs	3	9.89	0.011

TOC = Total organic carbon.

ERLs = effects range lows.

reduction of total nitrogen input from WWTFs that occurred 2005–2013; however, this trend was not statistically significant. More years of monitoring will be needed to see whether this trend continues and to determine if it is significant.

Human-caused stressors can account for many of the changes in community composition (Supplemental Table S1 gives a timeline of stressors on the bay ecosystem since 1950). The BEST routine on the USEPA dataset showed that the multidimensional pattern of species composition data was correlated with the pattern of eutrophication, hypoxia, and sediment contaminants. Additionally, the Kruskal-Wallis test indicated that variance of Δ^* was influenced by eutrophication (water column chlorophyll *a*, sediment TOC) and by sediment contaminants. Although correlations do not prove causation, these stressors are known to affect community composition (Jeon and Oviatt, 1991; Snelgrove et al., 2004; Diaz et al., 2008; Tweedley et al., 2015; Hale et al., 2016; MERL, 2017; Pelletier et al., 2017). Although moderate levels of nitrogen and sediment TOC support secondary production, too-high levels can lead to seasonal hypoxia and degraded sediment habitat, as exists in much of the upper third of the bay (Saarman et al., 2008). There, eutrophication and hypoxia are strong enough to overcome the effect of sediment grain size on the distribution of benthic animals (Pelletier et al., 2017). Using data from the UK National Marine Monitoring Program of estuaries and coasts, Tweedley et al. (2015) found that taxonomic distinctness was unaffected by natural differences

Table 4

Kruskal-Wallis test of change in taxonomic distinctness (Δ^*) by year for three data sets from Narragansett Bay.

Data set	Period	# Years	# Studies	# Stations	df	F	p
US EPA	2000–2015	15	3	151	7	1.85	0.08
North Jamestown	1957–2010	53	12	27	4	0.46	0.77
Spar Island	1975–2008	35	6	67	3	6.86	< 0.001

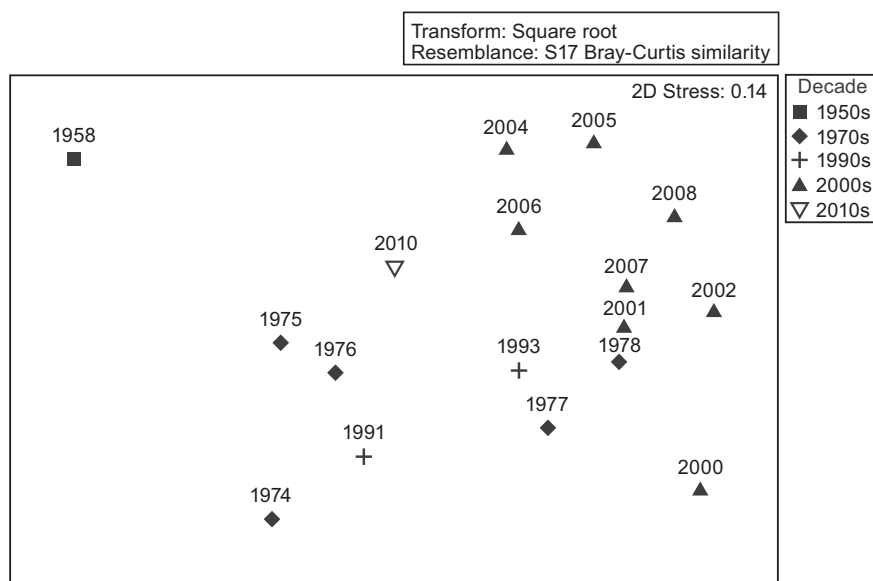


Fig. 5. Multidimensional scaling plot of benthic abundance data at the North Jamestown site, 1957–2010. Points that are closer together have a more similar species composition than points more distant. There was a significant change over the time series, but none of the decade-to-decade changes were different.

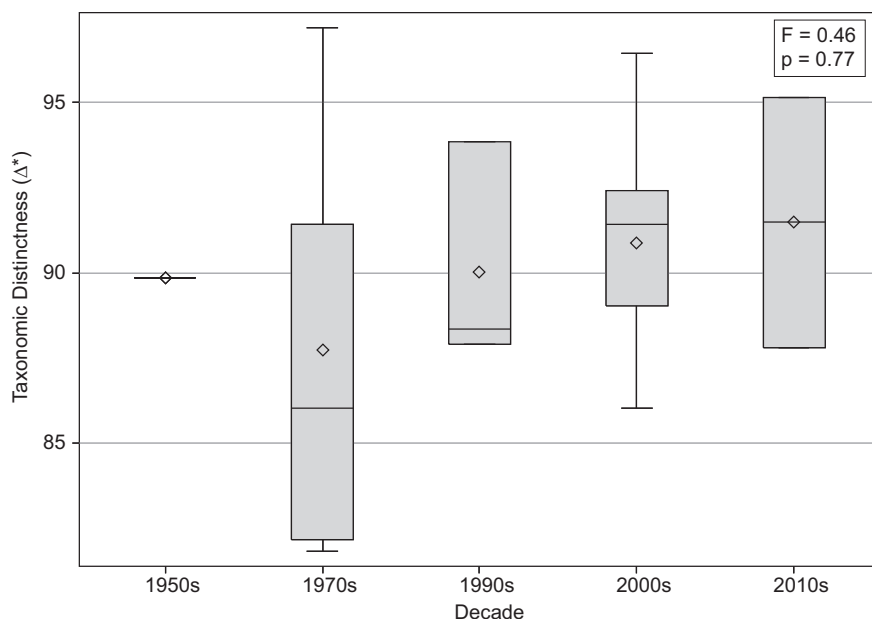


Fig. 6. Distribution of taxonomic distinctness, North Jamestown site, 1957–2010. The diamond is the mean, the line in the box of quartiles is the median, and the whiskers are the range. Changes in taxonomic distinctness (Δ^*) were not significant ($p = 0.77$).

between the two environments, but was significantly correlated with heavy metal concentrations in sediments.

4.2.2. North Jamestown control site

The North Jamestown benthic community in mid-bay was relatively stable over the 52-year period (1958–2010), with a moderate change in community composition and no significant change in biodiversity. Thus, this site provides a reasonable reference site for the classic southern New England *Nephtys/Mediomastus-Nucula* soft-bottom community. This is useful for comparisons with more stressed or more changeable communities of the upper Narragansett Bay, as well as portions of Buzzards Bay and Long Island Sound. The increase in taxonomic distinctness in every decade since the 1970s (Fig. 6) may reflect a response to better water quality. However, this trend was not statistically significant; more years of data are needed to verify whether

this trend will continue.

This site has been relatively isolated from local stressors affecting the upper bay, apart from a small WWTF at Quonset Point to the west (where several metals in sediments exceeded EPA standards from 1942 to 1973), dumping of Field's Point sewage sludge to the northeast, 1908–1949, and dredging of the channel from the East Passage to Quonset Point in early 1940s (Supplemental Table S1). The USEPA dataset showed exceedances of one metal and two synthetic organic ERLs in 1993; by 2010, there was an exceedance of only one metal ERL. The water column here is well-mixed and hypoxia is rarely a problem (Saarman et al., 2008; USEPA, 2017a). Sediment cores at a station near the Quonset Point WWTF showed an increase in hypoxia in the 1970s, which has since decreased (Boothman and Coiro, 2017).

The significant correlation of community composition with watershed population and water temperature likely reflects the influence

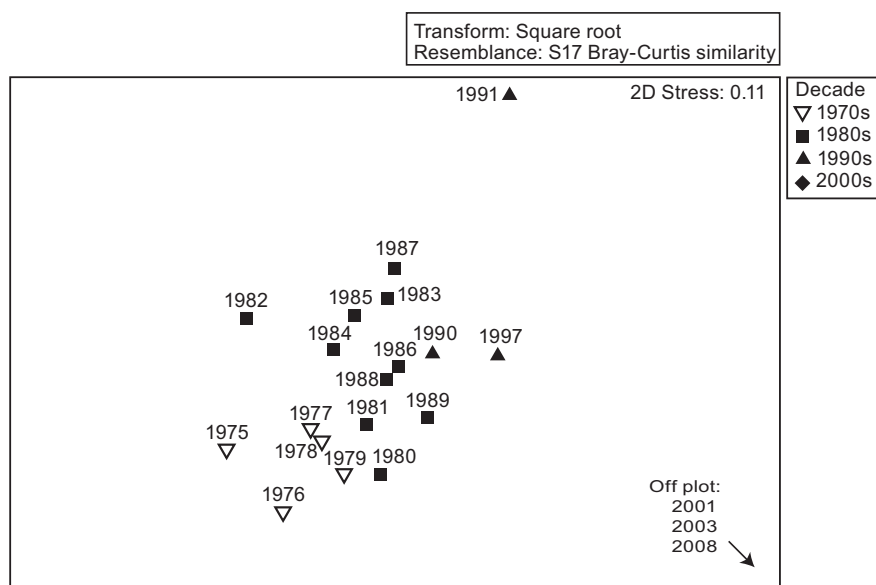


Fig. 7. Multidimensional scaling plot of benthic abundance data at Spar Island in Mount Hope Bay, 1975–2008, showing the significant changes in community composition over the 35 years. Points that are closer together have a more similar species composition than points more distant. Points from the 2000s were off the plot to the right. There were significant decadal differences.

of these two variables on benthic communities (e.g., Hale et al., 2004; Oviatt, 2004; Collie et al., 2008; Hale et al., 2017). We suggest that the significant difference in community composition over the six-decade time series may be related to reduction of phytoplankton biomass by rising water temperatures, which reduces the input of organic matter to the benthos (Nixon et al., 2009) and changes the demersal fish predators (Collie et al., 2008). Phytoplankton are the main contributors to primary production in the bay and exert a bottom-up control on benthic invertebrate populations, being food for filter-feeders and the source of organic matter that settles to the bottom and is processed by deposit-feeders (e.g., Rudnick and Oviatt, 1986). Demersal fish preying upon benthic invertebrates exert a top-down control (Collie et al., 2008). Eutrophication and rising water temperatures would affect all three trophic groups simultaneously and changes in any one would ripple through the other two.

Two of the longest time series of biological monitoring in Narragansett Bay are a phytoplankton station (Station 2), 1959–1997

(Smayda et al., 2004), and a bottom fish trawl station (Fox Island), 1959–2018 (Collie et al., 2008; GSO, 2017). Both sites are 5–7 km to the west of the North Jamestown benthic site. Phytoplankton and demersal fish communities experienced sharp changes related to warming waters and other factors (Supplemental Table S2). We speculate that the change in benthic community composition is related to the 20-fold decline in winter diatom abundance, the change in the dominant epifauna from *Asterias* starfish to *Cancer* crabs, and the regime shift from primarily demersal fishes to pelagic fishes that happened during those years (Supplemental Table S2). However, the benthic community appears to have been more stable than the phytoplankton and demersal fish communities. Regional factors (such as warming waters that affected the benthos, their food, and their predators) may be more important than local factors in driving changes at this site.

4.2.3. Spar Island impacted site

In contrast with North Jamestown, the Spar Island site showed

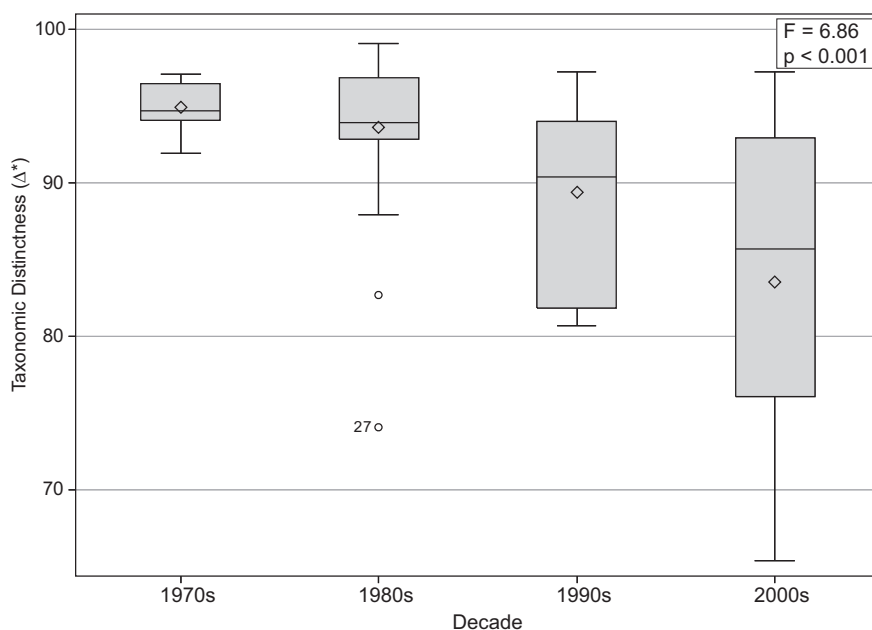


Fig. 8. Distribution of taxonomic distinctness at Spar Island in Mount Hope Bay over four decades, 1970s–2000s. The diamond is the mean, the line in the box of quartiles is the median, and the whiskers are the range. Taxonomic distinctness (Δ^*) declined significantly ($p < 0.001$).

significant changes in community composition and a decline in taxonomic distinctness. A decrease in Δ^* indicates a more compressed taxonomic tree and is indicative of increasingly stressed benthic communities that support a more restricted range of functional diversity (Dimitriadis and Koutsoubas, 2011; Clarke et al., 2014). The primary species contributing more than 5% to the observed inter-decadal changes in benthic community composition were relatively small opportunistic species with rapid development and high fecundity that can respond quickly to a changing environment. Other long-term benthic studies have shown similar declines in biodiversity as a result of eutrophication and hypoxia, contaminants, over-fishing, and warming waters (Callaway et al., 2007; Pranovi et al., 2008; Krann et al., 2011; Shojaei et al., 2016; Trott, 2016; Obst et al., 2017).

The history of increasing stressors on Mount Hope Bay suggests eutrophication, sediment contaminants, and water temperature as the likely causes for the decline in biodiversity after 1996 (Supplementary Table S1). Water discharge from the Taunton River is slightly higher than that of the Blackstone (Nixon et al., 2008). The WWTFs contribute about 70% of the total nitrogen load to this sub-embayment (Krahforst and Carullo, 2008). The Taunton River and the Fall River WWTF together accounted for more than one-third of the entire sewage loading to the Narragansett Bay (NBEP, 2017). While the Providence area WWTFs showed major reductions of nitrogen loading from 2000–2004 to 2013–2015, the Fall River WWTF showed only a very slight reduction during that period and the CSO remediation project is not yet completed (NBEP, 2017). The excess load of nitrogen to Mount Hope Bay has led to eutrophication and periodic summer hypoxia (Calabretta and Oviatt, 2008; Krahforst and Carullo, 2008). The water column is often well-stratified (MacDonald and Rountree, 2006), a situation that could lead to worsening hypoxia. In the early 1990s, Mount Hope Bay had greater inputs of Cd, Cr, Cu, and Pb than the rest of Narragansett Bay (Nixon, 1991). In the early 2000s, concentrations of seven metals all exceeded their ERLs and mercury was above its ERM (Calabretta and Oviatt, 2008). The level of mercury in surficial sediments is still high (NBEP, 2017).

In 1984, the Brayton Point Power Plant, which had opened in 1963, increased the amount of once-through cooling water drawn from Mount Hope Bay by 45% (USEPA, 2003). Discharges of heated effluent are known to reduce benthic biodiversity (D'Agostino and Colgate, 1973). A station in the thermal outfall canal, 2000–2001, had relatively high abundance of benthic invertebrates, which may have been a result of reduced predation by declining populations of demersal fishes (Ellis, 2002). The monitoring program associated with the power plant (MRI, 1994) showed that from 1975 to 1982, biodiversity at the Spar Island station declined nearly 60% (Desbonnet and Lee, 1991). A partial resurgence, 1982 to 1986, may have been related to the Fall River WWTF upgrade to secondary treatment in 1982 (Desbonnet and Lee, 1991). However, we found that by the following decade, Δ^* dropped significantly (Fig. 8), perhaps as a result of the 1984 increased intake of cooling water and return of heated water to the bay (Supplemental Table S1).

From 1985 to 1999, medium-density residential land use in the Taunton River watershed jumped 49% and urban lands 19% (Krahforst and Carullo, 2008). The sub-watershed 1990–2010 had the largest percent increase in population (22%) of all the Narragansett Bay sub-watersheds and during 2000–2010, the largest gain in urban (9–18%) and largest loss of forest (4–9%) (NBEP, 2017). Current impervious surface is 12%, a level where adverse effects are expected (NBEP, 2017). It appears that local factors (watershed population, nutrient enrichment, metals loading, warming from the power plant) have been a stronger influence than regional factors (climate-driven warming waters) in driving changes at this site.

5. Conclusions

For the entire bay (USEPA dataset) and for the North Jamestown

site, Δ^* showed a rising trend toward the end of each time series. Although not statistically significant, these trends may be indicative of a recovery of biodiversity in response to decreased anthropogenic stressors, as suggested by an analysis of historical data collected over 182 years (Hale et al., 2018a). Whether these trends will continue and become significant will require further years of monitoring. It was too early to statistically detect an improvement in benthic biodiversity following the recent nitrogen reductions from the WWTFs at the head of the bay. Recovery of benthic communities after nitrogen reductions can take years because of the need to use up existing sediment carbon stocks and the successional dynamics of benthic communities (Diaz et al., 2008).

Despite historical declines in biodiversity of the benthic invertebrate communities, primarily in the upper third of the bay, overall biodiversity is comparable with other systems of the northeast US (Trott, 2016). Our taxonomic distinctness results are comparable with those from other benthic community studies from Europe (Ellingsen et al., 2007; Louzao et al., 2010; Tweedley et al., 2015) and Maine (Trott, 2016). At the same time, there have been significant changes in community composition. Much of the year-to-year variance in community composition is driven by small, opportunistic, short life cycle species like *Mediomastus*, *Nucula*, and *Ampelisca*, while longer-term changes include larger fauna (Hale et al., 2018a). Variance of biodiversity in the bay was correlated with eutrophication, hypoxia, and sediment contamination by metals and synthetic organic compounds.

With the USEPA dataset, spatial differences in taxonomic distinctness due to the north-south declining pollution gradient showed stronger differences than temporal changes over the 20 years. The strength of the north-south pollution gradient has declined over the years as nutrient and sediment contaminant inputs at the head of the bay have been reduced. The North Jamestown site, with fewer human impacts than the upper bay, showed few significant changes and remains a good reference site. In contrast, at Spar Island since the 1970s, several local, human-induced stressors forced decade-to-decade changes in both community composition and loss of taxonomic distinctness. Regional factors may be more important than local factors in driving changes at North Jamestown while local pollution factors may be a stronger influence in driving changes at Spar Island. The two sites we used complement studies of benthic community trends in two other impacted areas of upper Narragansett Bay—Greenwich Bay and the Providence and Seekonk Rivers (Calabretta and Oviatt, 2008; Pelletier et al., 2017). Like Mount Hope Bay, Greenwich Bay is another sub-embayment that has shown few signs of ecosystem recovery (Pesch et al., 2012; Pelletier et al., 2017).

In general, the ANOSIM measure of community composition differences appeared to be more sensitive to temporal environmental changes than was the taxonomic distinctness measure. Similarly, in a review of worldwide ecological communities, Dornelas et al. (in press) found that variation in community composition, including losses and gains of species, varied more than species richness across local scales. Nevertheless, the taxonomic distinctness measure in our study was able to detect spatial differences between the North Jamestown site and the more impacted Spar Island site. It was highly effective at detecting strongly polluted sites, such as ones in the Providence/Seekonk Rivers with few taxa present (Pratt, 1972; NCA, 2017) where taxonomic distinctness values were at the low end ($\Delta^* = 72.9$) of the USEPA dataset. The finding that taxonomic distinctness (Δ^*) was statistically independent of the number of species in all three of our datasets lent support to the use of taxonomic distinctness as a biodiversity metric when combining data from multiple studies.

Historical studies are valuable for identifying ecological patterns and linking them to anthropogenic stressors (Reise et al., 1989; Pesch et al., 2012; Thurstan et al., 2015; Shojaei et al., 2016). Some stressors in the bay are declining (e.g., inputs of nutrients and sediment contaminants) and others are increasing (e.g., watershed development, water temperature, new synthetic compounds) (NBEP, 2017), thereby

making it difficult to predict future biodiversity. Biodiversity can be a useful tool for managing a complex system like Narragansett Bay (Palumbi et al., 2009). In the face of changing stressors, continued benthic monitoring is needed for effective adaptive management.

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.marpolbul.2018.05.019>.

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