

# A Re-assessment of Narragansett Bay Benthic Habitat Quality Between 1988 and 2008

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**Abstract** The first bay-wide synoptic survey of benthic habitat quality in Narragansett Bay, Rhode Island, USA, was conducted in August of 1988. Twenty years later, we revisited the same sampling locations as the original survey using similar sediment profile imagery technology and analysis tools. Like estuaries throughout the US, increased temperatures and reductions to anthropogenic nutrient inputs have cumulatively affected Narragansett Bay in the intervening 20 years. To understand how these changes may have influenced benthic organic enrichment and habitat quality, we compared the prevalence and spatial arrangement of benthic biotopes (i.e., biotic and abiotic benthic descriptions) between 1988 and 2008 surveys. Biotopes dominated by *Ampelisca* spp. tubiculous amphipods increased >fivefold between 1988 and 2008, and expanded into the more urban, anthropogenically stressed Providence River estuary. *Ampelisca* beds occurred at critical boundaries in organic enrichment and habitat quality in both years and indicated the quantity of organic matter reaching the benthos. In general, benthic biotopes reflect the degree of benthic-pelagic coupling and are an important link between

estuarine water quality and other marine life. As estuaries globally cope with the effects of increased warming and legislated anthropogenic nutrient reductions, rapid assessments of benthic biotopes will be critical for understanding changes to whole-estuary condition as a result of these cumulative stressors.

**Keywords** Benthic habitat · Biotope · Narragansett bay · Climate change · Organic enrichment

## Introduction

Estuaries are highly dynamic ecosystems, greatly affected by the large human populations that have settled in their watersheds. Over the last several decades, eutrophication (Nixon 1995) and related hypoxia (Diaz and Rosenberg 2008) have become common stressors to estuaries, with deleterious effects for the benthos and dependent fisheries (Levin et al. 2009 and references therein). However, some estuaries are now undergoing oligotrophication, a decline in the supply of organic matter (Nixon 2009), due to nutrient management policies (Mozetič et al. 2010; Riemann et al. 2016). These improvements, ushered in the US by the Clean Water Act, have also continued to lower the concentrations of chemical contaminants in sediments and the water column from maximum levels since the 1950s (Corbin 1989; Nixon and Fulweiler 2012). Climate changes, including warmer waters, altered phenology, and lower rates of primary productivity, may further reduce the supply of organic matter to estuaries (Chaalali et al. 2013), as is hypothesized to be the case in our study area, Narragansett Bay (Northeast USA) (Nixon et al. 2009). Management actions targeted to address only one of many stressors in coastal ecosystems can have unintended consequences (Cloern et al. 2016), as was the case in Danish

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coastal waters where higher water temperatures have counteracted the expected positive effects of nutrient reductions (Riemann et al. 2016). Water column temperature, light attenuation, anthropogenic nutrient inputs, and the timing of plankton blooms in Narragansett Bay have all changed dramatically over the last 15 years, affecting the quantity and quality of organic matter delivered from the pelagic zone to the benthos (Li and Smayda 1998; Oviatt et al. 2002; Borkman and Smayda 2009; Nixon et al. 2009). Reductions in organic matter delivery to the benthos may have led to changes to benthic abundance, community composition, and structure (Frithsen 1990; Oviatt 2008; Deacutis 2008). However, methodological differences in benthic surveys over the past 30–40 years confound any trends in the data (Frithsen 1990; Nixon et al. 2009). Decreases in sediment metals concentrations between 1988 and 1998 ([http://www.narrbay.org/d\\_projects/rised/data.html](http://www.narrbay.org/d_projects/rised/data.html)) were most pronounced near the industrial head of the Narragansett Bay estuary and could be responsible for changes to the benthos through time. Trends in benthic ecosystem status could reflect the cumulative effects of climate change and reduced anthropogenic nutrient inputs and may highlight other potentially dramatic ecosystem changes.

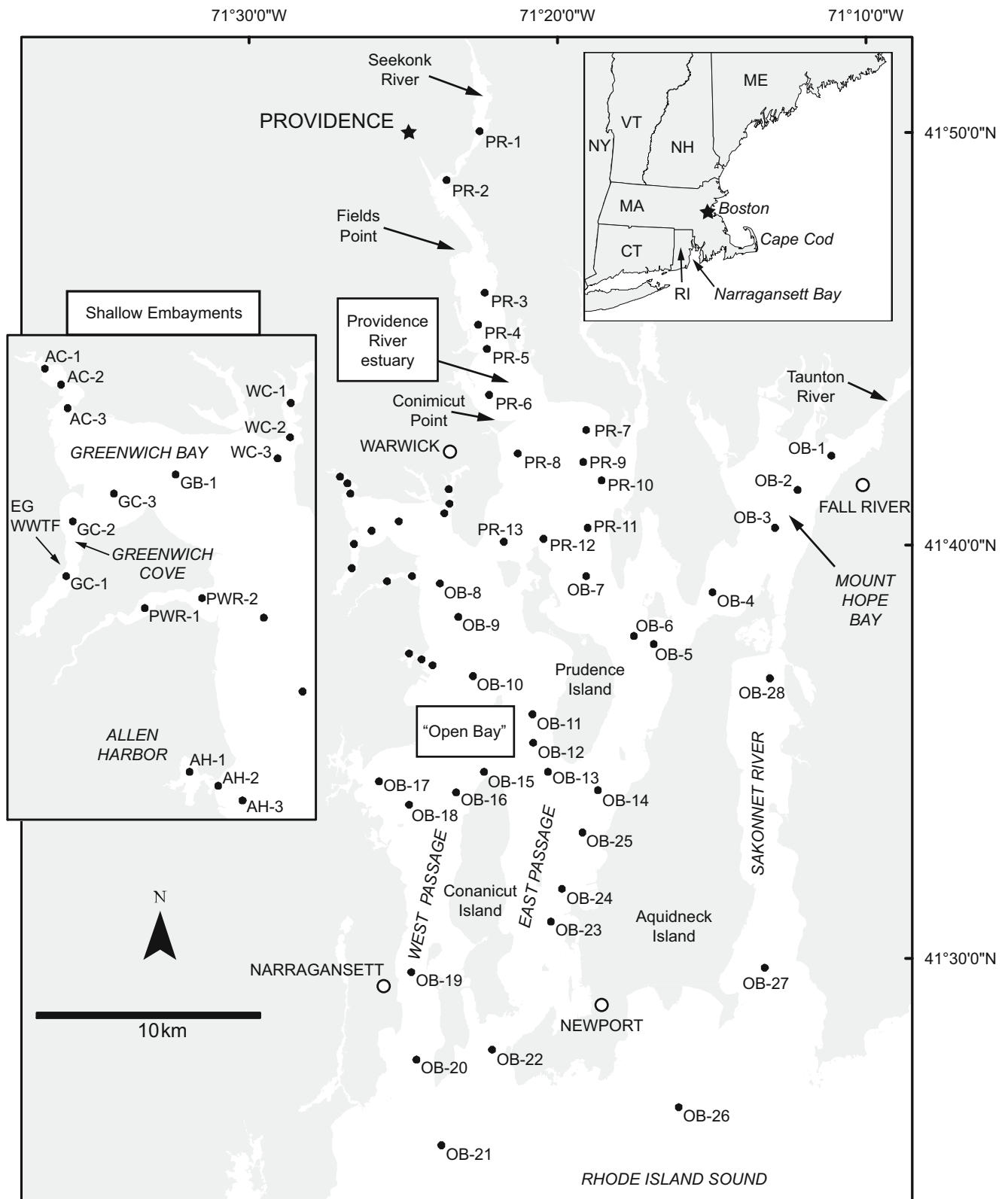
Benthic biotopes are units of the abiotic environment and associated assemblies of species (Connor et al. 2004; Costello 2009; Davies et al. 2004). The structure of surface sediments and the composition, or successional stage, of benthic communities is linked to the degree of organic loading to a water body (Pearson and Rosenberg 1978; Rosenberg 2001) and readily indicates recent (weeks to months) water quality conditions (Cicchetti et al. 2006; Shumchenia and King 2010). The composition of a biotope mosaic and how it changes over time may indicate degradation or recovery of an ecosystem (Dunning et al. 1992; Wiens et al. 1993; Pittman et al. 2007) and monitoring of biotope mosaics can help assess the effects of multiple stressors on coastal marine ecosystems (Cicchetti and Greening 2011). To date, studies to quantify change in benthic biotope mosaics have been mostly limited to aerial surveys of seagrass, mangrove, and saltmarsh ecosystems (Bostrom et al. 2011; Pittman et al. 2011; Cicchetti and Greening 2011; Zajac 2008). Benthic biotope characterization by traditional sampling methods is incredibly labor-intensive due to the collection, sorting, and identification of benthic samples. However, a recent example in the Milford Harbor estuary (Wales, UK) shows that utilizing the biotope approach in soft sediment environments provides a robust assessment of biological and physical characteristics and can lead to better management and monitoring (Carey et al. 2015).

One method that has increased the efficiency of benthic assessments is sediment profile imagery (SPI). SPI is a rapid reconnaissance technique that delivers clear images of benthic

biotopes regardless of water column turbidity (Germano et al. 2011). Ideally, SPI images capture an area including the sediment-water interface and up to 20 cm below—the most biologically active zone of the sediment column. SPI has long been used to characterize and map seabed patterns, sediment quality, and benthic biotopes (Germano et al. 1989; Valente et al. 1992; Solan et al. 2003). The depth of oxygenated sediment, depth of bioturbation, and benthic community successional stage visible in these images provide clues to recent incidences of hypoxia, organic loading rates, and the ability of the benthos to process organic carbon (Diaz et al. 2008). SPI is especially useful for long-term monitoring and has been used to measure the effects of multiple human activities on the benthic ecosystem including dredging and dredged material disposal, aquaculture, eutrophication, and low dissolved oxygen (Germano et al. 2011 and references therein).

In 1988, SPI was used in the first comprehensive survey of benthic habitat quality in Narragansett Bay (Fig. 1) (Valente et al. 1992). This study provided the first in situ snapshot of benthic processes in Narragansett Bay soft sediments. Most researchers were unaware and “surprised” by the proportion of the bottom that had been exposed to high levels of organic deposition and low concentrations of dissolved oxygen (Granger et al. 2000). Many of the sites identified as having excessive organic enrichment and degraded benthic habitat were in the Providence River estuary or shallow embayments; sites near waste water treatment facility (WWTF) outfalls, in coves, or other spatially constricted areas that received effluent (refer to Fig. 1; Valente et al. 1992).

Like most estuaries and coasts globally, there has been a great deal of human intervention and human-mediated change in Narragansett Bay and its watershed since the 1980s. Human population in the watershed has increased by about 200,000 to a total of about 2 million people (Nixon et al. 2008), increasing both impervious surfaces and WWTF loads. Between 1980 and 1995, the Field’s Point WWTF in Providence (responsible for ~55 % of total effluent discharged directly to the Bay) transitioned from being considered by the US Environmental Protection Agency one of the worst in the country to one of the best (Nixon and Fulweiler 2012). The Field’s Point plant initiated secondary treatment of its sewage in June of 1988, just months before the benthic habitat assessment took place. Although chemical contaminants were once one of the dominant ecological stressors in the Narragansett Bay watershed, they have both declined in concentration since the 1950s (Nixon and Fulweiler 2012) and declined in prominence relative to other stressors in the Bay (e.g., Shumchenia et al. 2015). In the 1990s, seasonal nutrient- and stratification-driven hypoxia was discovered in upper Narragansett Bay and has since been monitored by state and academic programs (Deacutis 2008; Codiga et al. 2009). In 2003, severe hypoxia



**Fig. 1** Locations within Narragansett Bay, Rhode Island USA, where sediment profile images were taken in 1988 and 2008. Three sub-regions used in the 1988 analyses are highlighted for comparison:

Providence River estuary stations (PR-); Open Bay (OB-); Shallow Embayments (inset). Note locations of Fields Point and East Greenwich (EG) Waste Water Treatment Facilities (WWTF) at labels

and anoxia cause a large fish kill and “nuisance” macroalgal concentrations in Greenwich Bay (RI DEM 2003). The fish kill resulted in media and political attention, and in 2004 the Rhode Island Department of Environmental Management (RI DEM) issued a statutory mandate to eleven WWTFs within the upper Narragansett Bay watershed to reduce summer season nitrogen discharges to the Bay between 48 and 65 % with respect to 1995–1996 levels (RI DEM 2005). Most monitoring efforts since the 2003 fish kill and 2004 nutrient reduction mandate have focused on dissolved oxygen data to evaluate compliance with water quality standards (RI DEM 2005) and highlight the summer recurrence of bottom water hypoxia in upper- and mid-Narragansett Bay (Bergondo et al. 2005; Deacutis et al. 2006; Melrose et al. 2007; Codiga et al. 2009). Impacts to mid-Narragansett Bay from climate-induced oligotrophication have been recently documented, and further changes Bay-wide are expected to result from mandated nutrient reductions (Fulweiler and Nixon 2009).

Exactly 20 years after the 1988 study (i.e., August 2008), and using the same SPI techniques, we revisited the same sites to reassess benthic biotope status. Using the same image analysis approach (Germano et al. 2011) on both 1988 and 2008 datasets, we classified benthic biotopes from the abiotic and biotic features of the surface and near-surface environment to compare biological and physical processes between surveys. We analyzed our results using a biotope mosaic approach (Cicchetti and Greening 2011) to assess benthic condition throughout the system. Here, we compare the spatial distribution, composition, and diversity of benthic biotopes throughout Narragansett Bay in 1988 and 2008 and relate any observed trends in biotope condition to changes in organic loading. This study provides a starting point for understanding these changes in Narragansett Bay and an approach that can be used to monitor changes to cumulative stressors in other estuaries.

## Methods

### Study Area

Narragansett Bay, a temperate, phytoplankton-based ecosystem, is the second-largest estuary on the east coast of the US and has the most densely populated watershed. It is generally well mixed with salinities ranging from ~20 psu at the head of the estuary to ~34 psu at the mouth, and a mean residence time of 26 days (Pilson 1985). In the north, the Providence River estuary is heavily populated, narrow, and opens to Narragansett Bay (i.e., Open Bay), which consists of the East and West Passages on either side of Conanicut Island (Fig. 1; Valente et al. 1992; Nixon et al. 2009; Raposa 2009). In addition, a few Shallow Embayments (e.g., Greenwich Bay and associated coves) form distinct regions of the mid-Bay (Fig. 1). Narragansett Bay lies near the

boundary between the boreal species of the Gulf of Maine/Bay of Fundy (also known as Acadian) ecoregion to the north and warmer water species of the Virginian ecoregion to the south (Spalding et al. 2007). Recent shifts in marine species distribution and abundance near this boundary are driven in part by climate change (Oviatt 2004; Collie et al. 2008; Pinksy et al. 2013). Because of its position on this boundary, Narragansett Bay provides an excellent case study for other estuaries affected by climate change and oligotrophication.

### Data Collection

To be most comparable with the 1988 study, we conducted our survey over the same length of time in mid-August and during the same neap tidal stage. Station locations were originally chosen to define Bay-wide trends in soft sediment benthic habitat quality and were located in areas 5 to 20 m deep (Fig. 1). Fifty-two of the 56 stations sampled in 1988 were revisited from August 12 through 15, 2008. We excluded stations in the Sakonnet River (OB-27, OB-28) and Potowamut River (PWR-1, PWR-2) for logistical reasons. We used Hypack software (Hypack, Inc., Middletown, CT) linked to a differential GPS with sub-meter accuracy (Trimble ProXH, Trimble Navigation Ltd., Sunnyvale, CA) for navigation in the field. At each station, we deployed an Ocean Imaging Systems Inc. (Falmouth, MA) digital sediment profile camera for three to five replicate drops, as in the 1988 study (Valente et al. 1992). We recorded coordinates of the first camera drop at each station to compare with the 1988 target location.

### Analysis of 2008 Images and Re-analysis of 1988 Images

Original, analog printed black and white images from the 1988 survey were obtained from archives at the Graduate School of Oceanography, University of Rhode Island, and scanned into digital format at a resolution of 300 pixels per inch. The 2008 digital images were uploaded to the analysis computer in .jpg format at 335 pixels per inch. The difference in image color compromised our ability to compare the depth of sediment oxygenation between surveys. Sediment oxygenation is detectable as a change in sediment color and is an important indicator of habitat quality. To overcome this limitation, we relied on the other abiotic and biotic surface and subsurface features visible in images to classify biotopes. To increase the comparability of these other features, we re-analyzed 1988 images using the same method as the 2008 images.

All SPI photos were imported into Adobe Photoshop CS6, and image brightness and contrast were adjusted manually to increase the detectability of habitat features such as tubes and burrows. For each station in each survey, we examined each available replicate and recorded information on sediment grain size, surface features, and subsurface features following the protocol described in Rhoads and Germano (1982; 1986).

*Sediment Grain Size*

We distinguished between organic-rich mud, mud, sandy mud, sand and gravel. Coarse-grained sediments were indicated by shallow prism penetration and suggested physically dominated habitats. Fine-grained sediments were indicated by deeper prism penetration and suggested that the seafloor environment was depositional and less-frequently physically disturbed. Organic-rich muds were characterized by fine-grained sediments and very deep prism penetration, little to no visible surface oxidation, and minimal surface disturbance or roughness.

*Surface Features*

Surface descriptors included both biogenic and physical features, such as amphipod and worm tubes, epifauna (e.g., snails, crabs), shells, macroalgae, bacterial mats (e.g., *Beggiatoa sp.*), feeding pits/mounds, bedforms and roughness. The presence of tolerant species (e.g., tubicolous polychaetes such as capitellids and spionids) associated with early infaunal succession was also noted (Rhoads and Germano 1986). The presence of these features was noted to indicate any recent disturbance and the degree and nature of biological activity at each station.

*Subsurface Features*

The presence of burrows, infaunal feeding voids, infauna, and gas voids were recorded in each image. Subsurface features such as burrows, feeding voids, and infauna indicated biologically active environments, whereas the presence of gas-filled voids at depth indicated high rates of methanogenesis and anaerobic respiration (Rhoads and Germano 1986) associated with high rates of organic matter decomposition.

*Benthic Biotope Classification*

To classify biotopes, we first assigned each image to a sediment grain size category. We then described the biota for all station replicates from the biotic surface and subsurface features. Following guidance from the Coastal and Marine Ecological Classification Standard (FGDC 2012), these elements were summarized into a short phrase using the sediment grain size descriptor and dominant biota such as “Burrowing fauna on mud with shell hash.” Non-dominant but conspicuous biota were included in the descriptions when present. Biotope descriptions were kept consistent among images, and any minor variation among replicates was integrated by grouping descriptions by dominant biota and/or sediment type into eight general biotopes (Table 1). Detailed biota and sediment descriptions and the designated biotope for each SPI image replicate are presented in the Supplementary Material

**Table 1** Descriptions of the eight biotopes based on observed sediment type and biota to which sediment profile images were assigned

Sediment category	Dominant biota	Other biota	Biotope	Code
Organic-rich mud; sandy mud; sand	<i>Ampelisca spp.</i>	Small tube-building fauna; Shallow-burrowing fauna; <i>Beggiatoa spp.</i> ; <i>Mulinia lateralis</i>	<i>Ampelisca</i> beds occasionally with other small tube-building and shallow-burrowing fauna on substrates ranging from organic-rich muds to sand. <i>Beggiatoa spp.</i> or <i>Mulinia lateralis</i> may be present.	AM
Organic-rich mud	None dominant	<i>Beggiatoa spp.</i> ; Tube-building polychaetes; Shallow-burrowing fauna	Organic-rich mud with tolerant species such as <i>Beggiatoa spp.</i> , tube-building polychaetes, and shallow burrowing fauna.	UN.SF
Mud with shell hash	Burrowing fauna	Tube-building polychaetes; Deep-burrowing fauna	Burrowing fauna on mud with shell hash. Tube-building polychaetes or deep burrowing fauna may be present.	UN.SH
Mud	Burrowing fauna	Tube-building polychaetes; <i>Chaetopterus</i> ;	Burrowing fauna on mud. Tube-building polychaetes, larger tube-builders such as <i>Chaetopterus</i> , or deep burrowing fauna may be present.	UN.SI
Sandy mud	None dominant	Burrowing fauna; Tube-building fauna	Burrowing and tube-building fauna on sandy mud.	UN.SS
Mud	<i>Crepidula fornicata</i>	Mobile crustaceans; Mobile gastropods; <i>Beggiatoa spp.</i>	<i>Crepidula</i> bed on mud. Mobile crustaceans, gastropods or <i>Beggiatoa spp.</i> may be present.	SH.SI
Very coarse sand with shell hash	None dominant	Rafting macroalgae; <i>Crepidula fornicata</i>	Very coarse sands with shell hash. Rafting macroalgae or <i>Crepidula</i> beds may be present.	SH.SA
Hard sand	None dominant	Epibenthic sponges; Rafting macroalgae; Attached macroalgae; Mobile gastropods	Hard sands with epibenthic sponges, rafting or attached macroalgae, and/or mobile gastropods.	SA

(S1). It was not surprising to find that some biota did not appear to have preference for a particular sediment category (i.e., were associated with multiple sediment grain sizes), as this has been previously noted in Narragansett Bay and elsewhere (Calabretta and Oviatt 2008; Snelgrove and Butman 1994).

### Comparison of 1988 and 2008 Surveys Using Biotope Mosaic Approach

To assess the influence of survey repositioning on interpreted biotope change, we checked for a relationship between biotope change and the 2008 field distance from the 1988 target location. We found that sites classified as “changed” were a mean 61 m from the target location ( $n=25$ ) and sites classified as “no change” were a mean 57 m from the target location ( $n=15$ ), indicating that repositioning likely had little to do with change detection. We address biotope change detection with regard to specific stations in [Benthic biotope change](#) section.

We assessed Bay-wide biotope diversity for each survey and within each Bay region each year (Providence River estuary, Shallow Embayments, and Open Bay) (Fig. 1). Using only the subset of stations that could be directly compared between years, we calculated the proportion of images for which biotope changed between 1988 and 2008 Bay-wide and among each Bay region. If a biotope changed between 1988 and 2008, we noted to which type of biotope it changed. We measured changes in the composition and spatial structure of the benthic biotope mosaic by calculating the observed proportion of each biotope for each survey Bay-wide and among Bay regions.

## Results

In 2008, we re-sampled 52 of the 56 stations from the 1988 survey. On average, we were able to sample each station within 59 m of the mapped location, with distances ranging from 9 to 218 m from the targeted station location (Table 2).

We examined 143 images from 50 stations in the 1988 archive and 152 images from 52 stations from the 2008 survey (Table 2). From these, 47 stations had at least one replicate image from each survey to compare, and 42 stations had at least three replicate images from each survey. Biotope variability among replicates was less than between surveys (see S1). We were able to classify 48 stations to the biotope level from the 1988 archive and 45 stations from the 2008 survey to the biotope level (Table 2). Between the two survey years, 38 total stations could be directly compared at the biotope level (Table 2).

## Benthic Biotope Diversity

Differences in Bay-wide environmental conditions between surveys were not large enough to generate novel biotopes but were sufficient to generate differences in sediment structure and movement of species and communities. Bay-wide biotope diversity was the same between 1988 and 2008 but differed among sub-regions. In 1988, the Providence River estuary contained five different biotope types and in 2008 it contained three. Biotope diversity rose from four to five biotope types in Shallow Embayments between 1988 and 2008. Finally, the Open Bay stations contained five biotope types in 1988 and six in 2008. “Burrowing fauna on mud” and “*Ampelisca* beds” were the only two biotopes observed in all of the three sub-regions between 1988 and 2008. “Organic rich mud with tolerant species” were never observed in the Open Bay, and “Burrowing and tube-building fauna on sandy mud” were only observed in the Open Bay.

## Benthic Biotope Change

We noticed examples of both apparent biotope fidelity (Fig. 2 a, b) and marked change (Fig. 2c, d). For sites showing a stark change in sediment type between years, apparent biotope change could actually reflect high degrees of small-scale benthic heterogeneity or a shift in sediment transport dynamics in the intervening 20 years (e.g., sites PR-9 and AC-2 in Table 2). Of the 38 stations that could be reliably compared between years, we observed biotope change in 22 (58 %) (Fig. 3). The Providence River estuary saw the highest degree of benthic biotope change (8/10, 80 %), followed by Shallow Embayments (6/10, 60 %) and Open Bay (10/18, 55 %).

## Changes in the Composition and Spatial Structure of the Benthic Biotope Mosaic

In 1988, the Bay-wide biotope mosaic was dominated by Burrowing fauna on mud with 13/38 (34 %) stations classified mainly in the Open Bay. *Ampelisca* beds comprised 4/38 (11 %) of the mosaic and were concentrated mainly in Shallow Embayments, with none in the Providence River estuary (Fig. 4a). Organic-rich mud with tolerant species (9/38; 24 %) were found in the Providence River estuary and Shallow Embayments only. Other biotopes comprised a small portion of the total benthic biotope mosaic.

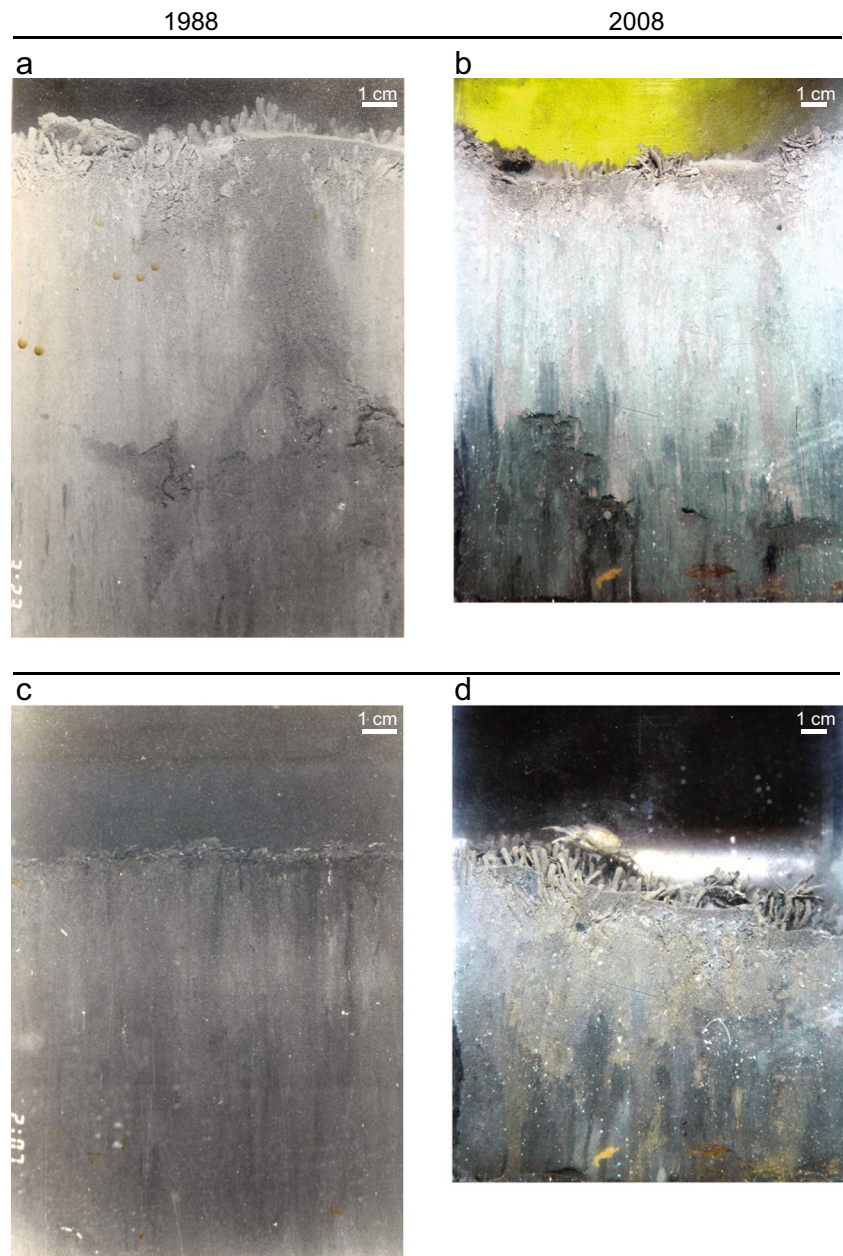
In 2008, *Ampelisca* beds dominated the Bay-wide biotope mosaic (22/38 or 58 %), representing a greater than fivefold increase from 1988. *Ampelisca* beds shifted distribution into the Providence River estuary (seven stations) and expanded within the Open Bay (nine stations) and Shallow Embayments (six stations) (Fig. 4b). Burrowing fauna on mud and organic-rich mud with tolerant species were much less abundant, each with 2/38 stations. Burrowing fauna on mud were no longer

**Table 2** Stations and replicates analyzed for 1988 archive and 2008 survey sediment profile images, with biotopes classified for 1988 and 2008

Bay region	Station	Distance between 2008 sample and 1988 mapped location (m)	Number of replicate images examined in this study		Classified biotope	
			1988	2008	1988	2008
Providence River estuary	PR1	16	1	3	UN.SF	UN.SF
	PR2	98	3	3	<i>UN.SI</i>	<i>UN.SF</i>
	PR3	41	3	3	<i>UN.SF</i>	AM
	PR4	42	3	3	<i>UN.SF</i>	AM
	PR5	60	3	3	<i>UN.SF</i>	AM
	PR6	20	3	3	UN.SH	-
	PR7	67	3	3	<i>UN.SI</i>	AM
	PR8	24	3	3	UN.SI	-
	PR9	63	3	3	<i>SH.SA</i>	AM
	PR10	115	3	3	<i>UN.SF</i>	AM
	PR11	45	0	3	-	AM
	PR12	20	3	3	<i>UN.SI</i>	AM
	PR13	54	3	3	SH.SI	SH.SI
Shallow Embayments	AC1	10	3	3	AM	-
	AC2	69	3	3	<i>UN.SF</i>	<i>SH.SA</i>
	AC3	32	3	3	AM	-
	AH1	22	3	3	<i>UN.SI</i>	AM
	AH2	48	3	2	<i>UN.SI</i>	AM
	AH3	53	3	3	SH.SI	SH.SI
	GB1	45	3	3	AM	AM
	GC1	66	3	3	<i>UN.SF</i>	AM
	GC2	30	3	3	<i>UN.SF</i>	AM
	GC3	40	3	3	<i>UN.SF</i>	AM
	WC1	63	3	3	AM	<i>UN.SI</i>
	WC2	85	3	3	AM	<i>SH.SI</i>
	WC3	218	3	2	-	SA
PWR1	NS	3	0	UN.SF	-	
PWR2	NS	3	0	AM	-	
Open Bay	OB1	67	3	3	<i>UN.SI</i>	AM
	OB2	171	1	3	AM	AM
	OB3	120	1	3	<i>UN.SH</i>	AM
	OB4	52	3	3	UN.SI	-
	OB5	75	3	3	<i>UN.SI</i>	AM
	OB6	15	3	3	UN.SH	UN.SH
	OB7	9	3	3	UN.SI	UN.SI
	OB8	34	0	3	-	SH.SA
	OB9	82	0	3	-	UN.SI
	OB10	37	0	3	-	AM
	OB11	55	0	3	-	AM
	OB12	65	3	3	<i>UN.SI</i>	AM
	OB13	65	0	3	-	AM
	OB14	10	3	3	<i>UN.SH</i>	AM
	OB15	26	3	3	UN.SI	SH.SI
	OB16	47	3	3	<i>UN.SI</i>	AM
	OB17	119	3	3	<i>UN.SI</i>	AM
	OB18	53	3	3	<i>UN.SI</i>	AM
	OB19	34	3	3	SA	<i>UN.SH</i>
	OB20	20	3	3	UN.SS	UN.SS
	OB21	58	3	3	UN.SS	UN.SS
	OB22	60	2	1	UN.SS	-
	OB23	111	3	3	UN.SS	UN.SS
	OB24	99	3	3	UN.SS	UN.SS
	OB25	39	3	3	UN.SS	UN.SS
	OB26	53	3	3	AM	-
	OB27	NS	3	0	-	-
	OB28	NS	3	0	UN.SI	-

See Table 1 in [Sediment grain size](#) section for code definitions. *NS* not sampled in 2008, - not classified, *italicized* change in biotope between 1988 and 2008

**Fig. 2** Example images of a site within Narragansett Bay (OB-2) where benthic biotope appeared to remain the same (*Ampelisca* beds) between 1988 (a) and 2008 (b) surveys; example images of a site (PR-3) where benthic biotope appeared to change from Organic rich muds with tolerant species to *Ampelisca* beds between 1988 (c) and 2008 (d) surveys



found within the Providence River estuary; there was one instance in Shallow Embayments and one in the Open Bay. Organic-rich mud with tolerant species were confined to the Providence River estuary.

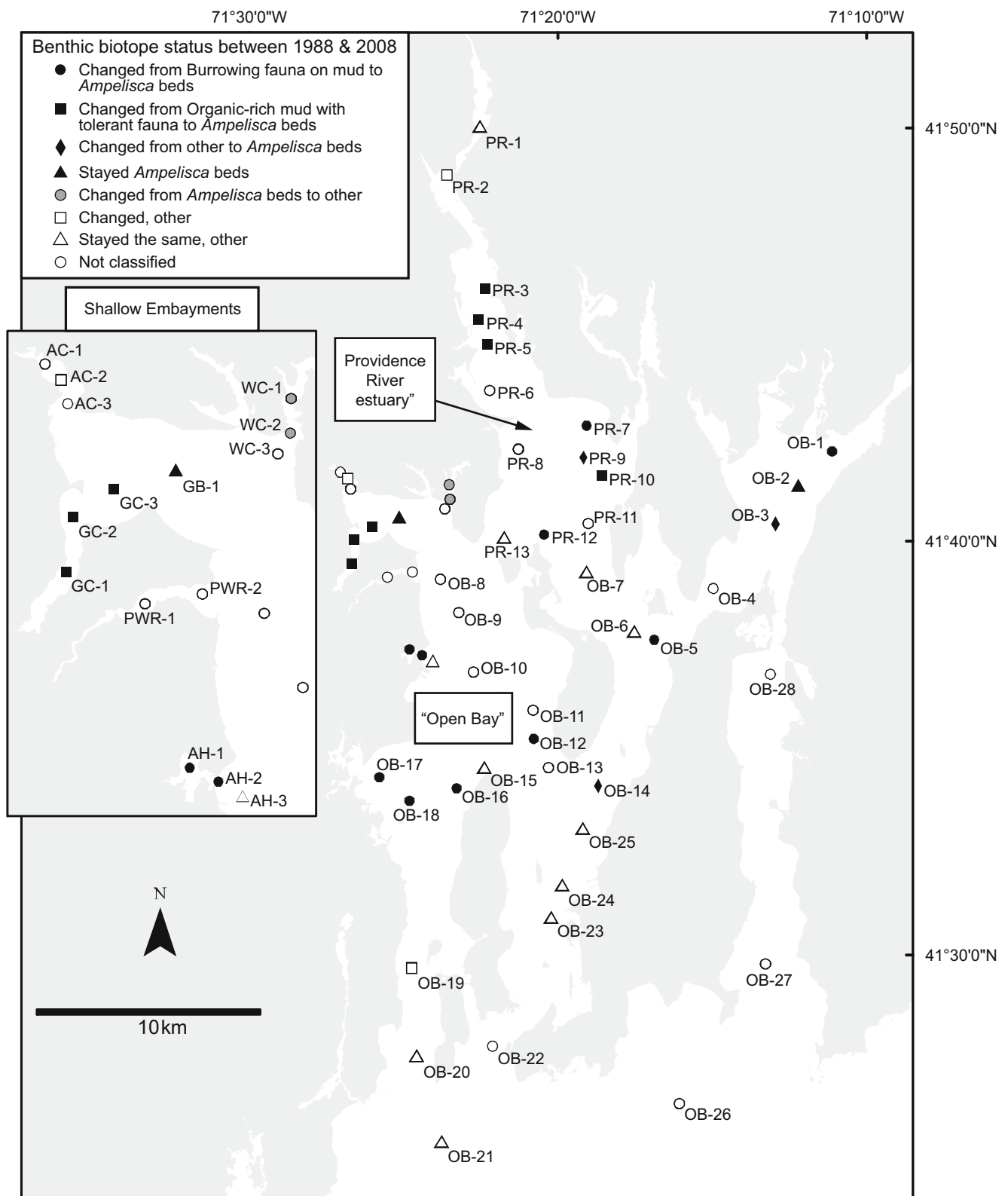
The most common type of benthic biotope change was a change to *Ampelisca* beds between 1988 and 2008 (Fig. 3). Ten stations changed from Burrowing fauna on mud in 1988 to *Ampelisca* beds in 2008. Seven additional stations changed from organic-rich mud with tolerant species in 1988 to *Ampelisca* beds in 2008. The increases in *Ampelisca* beds occurred in roughly equal proportions in both the Providence River estuary (7/9) and Open Bay (8/9). Most (5/8) of the biotope change in Shallow Embayments was also

to *Ampelisca* beds in 2008. However, two Shallow Embayment stations classified as *Ampelisca* beds in 1988 changed to other types in 2008, a shift not observed in other Bay regions.

## Discussion

We have observed a Bay-wide expansion of *Ampelisca* beds into the Providence River estuary and throughout Narragansett Bay between 1988 and 2008. Changes to the benthic biotope mosaic have occurred in the same time frame as oligotrophication due to climate change (Fulweiler and

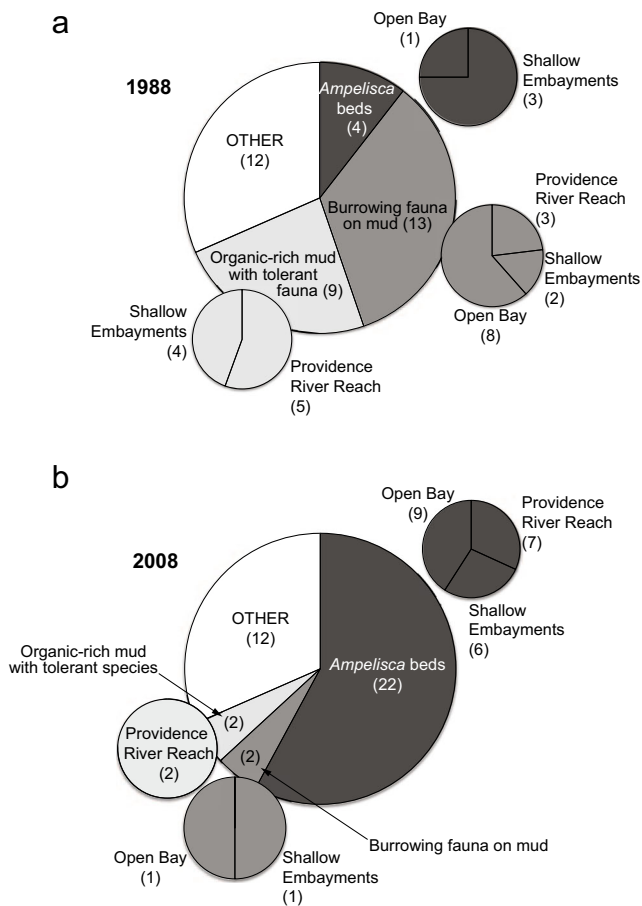




**Fig. 3** Benthic biotope change between the 1988 and 2008 surveys in Narragansett Bay was primarily from Burrowing fauna on mud or Organic-rich muds with tolerant species to *Ampelisca*. beds

Nixon 2009; Nixon et al. 2009) and initial reductions in anthropogenic nutrient inputs (i.e., initiation of biological

nitrogen removal at five small facilities as of January 2008; RI DEM 2008). This work is the first step toward



**Fig. 4** Composition and spatial arrangement of the Narragansett Bay benthic biotope mosaic in 1988 (a) and 2008 (b). The smaller bubbles show spatial distribution for each biotope

characterizing the responses of the Narragansett Bay benthic ecosystem to these stressors.

### Bay-wide Benthic Biotope Patterns

The Bay-wide gradient of benthic biotopes matched the known Bay-wide gradient in organic enrichment described by Valente et al. (1992), Oviatt (2008) and Nixon et al. (2008); organic enrichment is highest in constricted Shallow Embayments with high nutrient loads, at the head of the Bay in the Providence River estuary, and decreases towards the mouth of the Bay. As expected, biotopes composed of organic-rich sediments were confined to Shallow Embayments and the Providence River estuary and were not observed at Open Bay stations in either survey year. Biotopes composed of coarser-grained sediments, indicating lower organic deposition, higher benthic carbon mineralization, and higher flushing rates (e.g., “Burrowing and tube building fauna on sandy muds”), were observed further down the gradient, primarily south of Prudence Island (refer to Fig. 1). Calabretta and Oviatt (2008) describe the same gradient in benthic

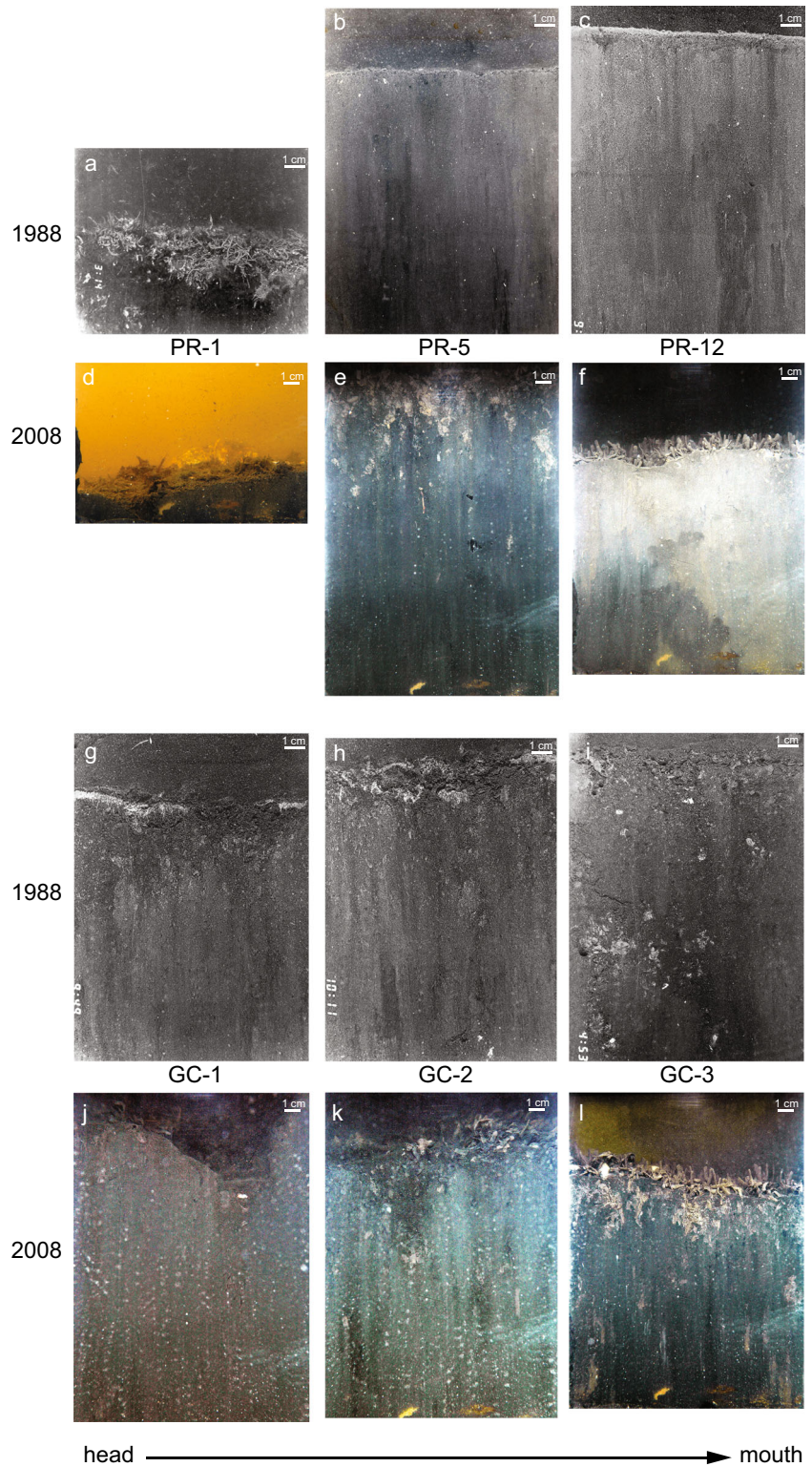
successional stage from north to south and similarly composed benthic communities. In their study, *Ampelisca abdita* was most numerous at their Providence River estuary and Mount Hope Bay stations between 2000 and 2004, with high densities appearing inconsistently at the Greenwich Bay station (Calabretta and Oviatt 2008).

The change from predominantly Burrowing fauna on mud and organic-rich mud with tolerant species in 1988 to *Ampelisca* beds in 2008 reflects a shift to a more “mature” benthic community (Pearson and Rosenberg 1978). Biotopes indicative of high organic loading decreased by 78 % between 1988 and 2008, entirely within the constricted areas of the Bay with high historical nutrient loads. These observations suggest that on average, there was less organic matter available to the benthos in August 2008 than in August 1988.

### Benthic Biotope Patterns in Bay Sub-regions

The largest area of organic enrichment in 1988 (Valente et al. 1992) contained biotopes classified as organic-rich muds with tolerant species (stations PR-1 through PR-5, PR-8, PR-11, and PR-12). The northernmost two stations in the Providence River estuary (PR-1 and PR-2) remained the same in 2008, evidence of continued organic enrichment. However, the other stations for which images were available (PR-3, PR-4, PR-5, PR-11, PR-12) all changed to *Ampelisca* beds. Images from three Shallow Embayment stations (Greenwich Cove: GC-1, GC-2, GC-3) in 1988 had “no visible macrofauna,” were classified as organic-rich muds with tolerant species, and showed signs of “excessive organic enrichment” likely from the nearby East Greenwich WWTF discharge located within the cove (Valente et al. 1992; Fig. 1). In 2008, all three of these Shallow Embayment stations were classified as *Ampelisca* beds (Fig. 5). Biotopes at the heads of the Providence River estuary and Shallow Embayments (i.e., in the most constricted portions of the water bodies) had organic-rich muds with little visible oxidized sediments and occasional *Ampelisca* spp. tubes in “life-position” (e.g., Fig. 5e, j, and k), whereas biotopes at the mouths of these water bodies were more robust *Ampelisca* beds with visibly oxidized sediments and evidence of burrowing fauna (e.g., Fig. 5f, l). Although portions of the Providence River estuary and Shallow Embayments still show signs of organic enrichment, it appears that the gradients have moved, and moved in similar ways between these two areas in Narragansett Bay: landward up the Providence River estuary and up Shallow Embayments. Within the Open Bay, some stations near places of apparent organic matter focusing (southwest of Prudence Island OB-11 and OB-12; Valente et al. 1992) and organic loading (the mouth of the Taunton River through Mount Hope Bay OB-1–OB-4) also changed to *Ampelisca* beds between 1988 and 2008.

**Fig. 5** Sediment profile images along gradients from the heads to the mouths of the Providence River Reach (a–f) and Greenwich Cove (g–l) in 1988 and 2008. In 1988, each of these constricted water bodies showed organic-rich sediments with few visible macrofauna. The gradient of habitat quality appears to have shifted up-bay and shoreward such that images show *Ampelisca* beds at the mouths of each water body in 2008. Note visible camera artifacts in the last row of images are not features of the sediment or water column



**The “*Ampelisca* beds” Benthic Biotope**

It has been suggested that *Ampelisca* spp. are organic enrichment opportunists (McCall 1977). There is also a debate as to whether ampeliscids serve as indicators of impending hypoxia

(Levin et al. 2009) or of improving conditions (Diaz et al. 2008; Rhoads and Germano 1986). A recent study of hypoxia in Greenwich Bay (i.e., one of the Shallow Embayments in this study) demonstrated that ampeliscids colonized degraded habitats soon after the resumption of normal oxygen

conditions and were reliable indicators of improving water quality (Shumchenia and King 2010). Ampeliscid tube structures have been associated with increased biogenic activity and oxygen penetration into the sediment (Diaz et al. 2008) and increased hard clam abundance (MacKenzie et al. 2006), but have also contributed to the exclusion of other tubedwelling species (Santos and Simon 1980). In Jamaica Bay, New York, amphipod productivity was so high that it was likely more than sufficient to support the entire local winter flounder (*Pleuronectes americanus*) population, with *Ampelisca abdita* making up 88 % of the diet of juveniles (Franz and Tanacredi 1992). In Narragansett Bay, winter flounder juveniles have been most abundant in the Providence River estuary and Shallow Embayments recently (Meng et al. 2005) and so the observed increases in *Ampelisca* beds in these Bay regions may have already benefited this important fish species. Ampeliscids do require large quantities of organic matter to sustain “mat” densities (Franz and Tanacredi 1992; McCall 1977), which signals eutrophic conditions. It was estimated that 500-g carbon m<sup>-2</sup> yr<sup>-1</sup> is required to maintain *Ampelisca spp.* tube mats in Boston Harbor (Massachusetts, USA), approximately 60 miles to the north of Narragansett Bay (Diaz et al. 2008). Assuming this relationship is relevant to Narragansett Bay, there has likely been enough carbon produced historically via primary productivity on an annual basis in the Providence River estuary (mean of 559-g C m<sup>-2</sup> yr<sup>-1</sup>), Greenwich Bay (mean 219–254-g C m<sup>-2</sup> yr<sup>-1</sup>) and the Open Bay north of Prudence Island (mean of 517-g C m<sup>-2</sup> yr<sup>-1</sup>) to support these communities (Oviatt et al. 2002). Given that primary productivity historically comprised an estimated 80 % of the total organic carbon input to the Bay (Nixon et al. 1995), it is also likely that there are sufficient amounts of organic matter stored in the sediments on which these ampeliscids thrive.

Dense *Ampelisca spp.* communities in areas with high organic input and good water quality have been previously observed within Narragansett Bay and in Boston Harbor (Stickney and Stringer 1957; Diaz et al. 2008). The cessation of primary sewage discharges to Boston Harbor in the early 1990s appears to have “set the stage” for the observed “widespread increases” in *Ampelisca spp.* throughout the harbor. Prior to 1992, organic loading was high but water quality may have been too poor to allow *Ampelisca spp.* to thrive (Diaz et al. 2008). A decade later, declines in *Ampelisca spp.* tubes were associated with the reductions in organic loadings to the harbor and the eventual depletion of sediment organic inventories (i.e., surface sediment total organic carbon) (Diaz et al. 2008). Reductions in organic matter and lower numbers of *Ampelisca spp.* have apparently had a positive effect on fish species, as the recreational fishing community has made note of significant recent increases in winter flounder populations in Boston Harbor (Powers 2015).

Narragansett Bay could be exhibiting a pattern similar to the Boston Harbor example. In 1988, we observed conditions that did not favor widespread *Ampelisca* beds: stations with high organic loading and surface sediments that indicated poor water quality conditions (Valente et al. 1992). Between 1988 and 2008, conditions theoretically became increasingly favorable for *Ampelisca* beds: climate changed and management strategies to reduce organic loadings and improve water quality were initiated. In 2008, we observed an increase in the proportion of *Ampelisca* beds Bay-wide and especially in areas where organic loading was known to be previously high. Water quality monitoring programs continue to record hypoxic events (see <http://www.dem.ri.gov/bart>), but it is possible that hypoxia occurs now over a smaller area, with less frequency and/or intensity than previous events (the first Bay-wide dissolved oxygen monitoring program did not begin until in 1999; Prell et al. 2004). Unfortunately, we did not continuously monitor benthic biotopes between 1988 and 2008, and we cannot determine where Narragansett Bay is “located” along the Boston Harbor trajectory. Assuming oligotrophication continues, future surveys would demonstrate whether benthic biotope quality could be:

1. Staying the same—*Ampelisca* beds are maintaining position via existing sediment organic matter inventories under good or improving water quality,
2. Improving—*Ampelisca* beds decrease and biotopes reflecting more “mature” successional stages increase under good or improving water quality conditions,
3. Declining—*Ampelisca* beds decrease; novel but more depauperate biotopes emerge.

As organic loading continues to decrease and sediment organic matter inventories are depleted, we would expect to see changes in benthic community composition and reductions in overall benthic biomass (e.g., Riemann et al. 2016). The nature of these changes Bay-wide could have a profound effect on the ecosystem services delivered by the benthos.

### Implications for Other Estuarine Biota

The benthic biotope mosaic of 2008 may represent improved conditions for the protection and growth of other organisms. *Ampelisca* tubes can exist in such dense aggregations that they are likely important prey sources for the demersal fish of Narragansett Bay. However, demersal fish species in Narragansett Bay have declined in number over the past 47 years and especially since 1980, concurrent with increases in water temperature and decreases in chlorophyll concentrations (Collie et al. 2008). When the critical boundaries of organic enrichment are in the more shallow, protected (constricted) regions of the Bay as in 2008, robust *Ampelisca* beds may serve as critical habitats for juvenile

demersal fish such as winter flounder. When the critical boundaries of organic enrichment existed in deeper, less protected waters as in 1988, we observed fewer *Ampelisca* beds. With future warming and decreasing anthropogenic nutrient inputs, *Ampelisca* beds should be monitored more frequently as potential indicators of patterns in organic enrichment and important fish habitat.

We have inferred that a zone of organic enrichment is necessary to support biotopes valuable to other organisms. Indeed, estuaries are naturally fertile environments that serve as nursery grounds for many species. As others have described (Calabretta and Oviatt 2008; Nixon et al. 2009; Nixon and Fulweiler 2012), the cumulative stress of climate change and decreasing anthropogenic nutrient inputs will influence estuarine primary production and the flux of organic matter to the benthos in unknown ways. Future changes to benthic biotope composition and spatial arrangement could help describe these impacts. In this case, benthic biotope monitoring has added to our understanding of ecosystem status in the context of climate-driven reductions in primary productivity and reductions to anthropogenic nutrient inputs.

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#### Compliance with Ethical Standards

**Conflict of Interest** The authors declare they have no conflict of interest.

## References

- Bergondo, D.L., D.R. Kester, H.E. Stoffel, and W.L. Woods. 2005. Time-series observations during the low sub-surface oxygen events in Narragansett Bay during summer 2001. *Marine Chemistry* 97: 90–103.
- Borkman, D.G., and T.J. Smayda. 2009. Multidecadal (1959–1997) changes in *Skeletonema* abundance and seasonal bloom patterns in Narragansett Bay, Rhode Island, USA. *Journal of Sea Research* 61: 84–94.
- Boström, C., S.J. Pittman, C. Simenstad, and R. Kneib. 2011. Seascape ecology of coastal biogenic habitats: advances, gaps, and challenges. *Marine Ecology Progress Series* 427: 191–217.
- Calabretta, C.J., and C.A. Oviatt. 2008. The response of benthic macrofauna to anthropogenic stress in Narragansett Bay, Rhode Island: a review of human stressors and assessment of community conditions. *Marine Pollution Bulletin* 56: 1680–1695.
- Carey, D.A., M. Hayn, J.D. Germano, D.I. Little, and B. Bullimore. 2015. Marine habitat mapping of the Milford Haven Waterway, Wales, UK: comparison of facies mapping and EUNIS classification for monitoring sediment habitats in an industrialized estuary. *Journal of Sea Research* 100: 99–119.
- Chaalali, A., G. Beaugrand, P. Boet, and B. Sautour. 2013. Climate-caused abrupt shifts in a European macrotidal estuary. *Estuaries and Coasts* 36: 1193–1205.
- Cicchetti, G., and H. Greening. 2011. Estuarine biotope mosaics and habitat management goals: an application in Tampa Bay, FL, USA. *Estuaries and Coasts* 34: 1278–1292.
- Cicchetti, G., S. Latimer, S. Rego, G. Nelson, J. Bergen, and L. Coiro. 2006. Relationships between near-bottom dissolved oxygen and sediment profile camera measures. *Journal of Marine Systems* 62: 124–141.
- Cloern, J.E., P.C. Abreu, J. Carstensen, L. Chauvaud, R. Elmgren, R., J. Grall, et al. 2016. Human activities and climate variability drive fast-paced change across the world's estuarine-coastal ecosystems. *Global Change Biology* 22: 513–529.
- Codiga, D.L., H.E. Stoffel, C.F. Deacutis, S. Kiernan, and C.A. Oviatt. 2009. Narragansett Bay hypoxic event characteristics based on fixed-site monitoring network time series: intermittency, geographic distribution, spatial synchronicity, and interannual variability. *Estuaries and Coasts* 32: 621–641.
- Collie, J.S., A.D. Wood, and H.P. Jeffries. 2008. Long-term shifts in the species composition of a coastal fish community. *Canadian Journal of Fisheries and Aquatic Sciences* 65: 1352–1365.
- Connor, D.W., J.H. Allen, N. Golding, L. Kerry, L.M. Lieberknecht, K.O. Northen, and J.B. Reker. 2004. The marine habitat classification for Britain and Ireland version 04.05. In *JNCC (2015) The Marine Habitat Classification for Britain and Ireland Version 15.03*. <http://jncc.defra.gov.uk/MarineHabitatClassification> Accessed 29 June 2015.
- Corbin, J.M. 1989. Recent and historical accumulation of trace metal contaminants in the sediment of Narragansett Bay, Rhode Island. MS thesis, oceanography, University of Rhode Island.
- Costello, M.J. 2009. Distinguishing marine habitat classification concepts for ecological data management. *Marine Ecology Progress Series* 397: 253–268.
- Davies, C.E., D. Moss, and M.O. Hill. 2004. EUNIS habitat classification revised 2004. Report to European Environment Agency, European Topic Center on Nature Protection and Biodiversity.
- Deacutis, C.F. 2008. Evidence of ecological impacts from excess nutrients in upper Narragansett Bay. In *Science for ecosystem-based management: Narragansett Bay in the 21st century*, ed. A. Desbonnet and B.A. Costa-Pierce, 349–382. New York: Springer.
- Deacutis, C.F., D.W. Murray, W. Prell, E. Saarmann, and L. Korhun. 2006. Hypoxia in the upper half of Narragansett Bay, RI, during August 2001 and 2002. *Northeastern Naturalist* 13: 173–198.
- Diaz, R.J., and R. Rosenberg. 2008. Spreading dead zones and consequences for marine ecosystems. *Science* 321: 926–929.
- Diaz, R.J., D.C. Rhoads, J.A. Blake, R.K. Kropp, and K.E. Keay. 2008. Long-term trends of benthic habitats related to reduction in wastewater discharge to Boston Harbor. *Estuaries and Coasts* 31: 1184–1197.
- Dunning, J.B., B.J. Danielson, and H.R. Pulliam. 1992. Ecological processes that affect populations in complex landscapes. *Oikos* 65: 169–175.

- FGDC (Federal Geographic Data Committee, Marine and Coastal Data Subcommittee). 2012. Coastal and marine ecological classification standard. FGDC-STD-018-2012. Washington DC.
- Franz, D.R., and J.T. Tanacredi. 1992. Secondary production of the amphipod *Ampelisca abdita* Mills and its importance in the diet of juvenile winter flounder (*Pleuronectes americanus*) in Jamaica Bay, New York. *Estuaries* 15: 193–203.
- Frithsen, J.B. 1990. The benthic communities within Narragansett Bay. Technical report #NBP-90-28 of the Narragansett Bay Estuary Program.
- Fulweiler, R.W., and S.W. Nixon. 2009. Responses of benthic-pelagic coupling to climate change in a temperate estuary. *Hydrobiologia* 629: 147–156.
- Germano, J.D., D.C. Rhoads, L.F. Boyer, C.A. Menzie, and J.A. Ryther Jr. 1989. REMOTS imaging and side-scan sonar: efficient tools for mapping sea floor topography, sediment type, bedforms, and biology. In *Oceanic Processes in Marine Pollution. Volume 4: Scientific Monitoring Strategies for Ocean Waste Disposal*, ed. D.W. Hood et al., 39–48. Malabar: Krieger.
- Germano, J.D., D.C. Rhoads, R.M. Valente, D.A. Carey, and M. Solan. 2011. The use of sediment profile imaging (SPI) for environmental impact assessments and monitoring studies: lessons learned from the past four decades. *Oceanography and Marine Biology: An Annual Review* 49: 235–298.
- Granger, S.L., M. Brush, B.A. Buckley, M. Traber, M. Richardson, and S.W. Nixon. 2000. An assessment of eutrophication in Greenwich Bay. In *Restoring water quality in Greenwich Bay: a whitepaper series*, ed. M. Schwartz. Narragansett: Rhode Island Sea Grant.
- Levin, L.A., W. Ekau, A.J. Gooday, F. Jorissen, J.J. Middelburg, W. Naqvi, C. Neira, N.N. Rabalais, and J. Zhang. 2009. Effects of natural and human-induced hypoxia on coastal benthos. *Biogeosciences Discussions* 6: 3563–3654.
- Li, Y., and T.J. Smayda. 1998. Temporal variability of chlorophyll in Narragansett Bay 1973–1990. *ICES Journal of Marine Science* 55: 661–667.
- MacKenzie Jr., C.L., R.A. Pikanowski, and D.G. McMillen. 2006. *Ampelisca* amphipod tube mats may enhance abundance of northern quahogs *Mercenaria mercenaria* in muddy sediments. *Journal of Shellfish Research* 25: 841–847.
- McCall, P.L. 1977. Community patterns and adaptive strategies of the infaunal benthos of Long Island Sound. *Journal of Marine Research* 35: 221–266.
- Melrose, D.C., C.A. Oviatt, and M.S. Berman. 2007. Hypoxic events in Narragansett Bay, Rhode Island, during the summer of 2001. *Estuaries and Coasts* 30: 47–53.
- Meng, L., G. Cicchetti, and S. Raciti. 2005. Relationships between juvenile winter flounder and multiple-scale habitat variation in Narragansett Bay, Rhode Island. *Transactions of the American Fisheries Society* 134: 1509–1519.
- Mozetič, P., C. Solidoro, G. Cossarini, G. Socal, R. Precali, J. Francé, F. Bianchi, C. De Vittor, N. Smoldaka, and S.F. Umani. 2010. Recent trends towards oligotrophication of the northern Adriatic: evidence from chlorophyll a time series. *Estuaries and Coasts* 33: 362–375.
- Nixon, S.W. 1995. *Metal inputs to Narragansett Bay: a history and assessment of recent conditions*. Narragansett RI: Rhode Island Sea Grant.
- Nixon, S.W. 2009. Eutrophication and the microscope. *Hydrobiologia* 629: 5–19.
- Nixon, S.W., and R.W. Fulweiler. 2012. Ecological footprints and shadows in an urban estuary, Narragansett Bay, RI (USA). *Regional Environmental Change* 12: 381–394.
- Nixon, S.W., S.L. Granger, and B.L. Nowicki. 1995. An assessment of the annual mass balance of carbon, nitrogen, and phosphorus in Narragansett Bay. *Biogeochemistry* 31: 15–61.
- Nixon, S.W., B.A. Buckley, S.L. Granger, L.A. Harris, A.J. Oczkowski, R.W. Fulweiler, and L.W. Cole. 2008. Nitrogen and phosphorus inputs to Narragansett Bay: past, present, and future. In *Science for ecosystem-based management: Narragansett Bay in the 21st century*, ed. A. Desbonnet and B.A. Costa-Pierce, 101–175. New York: Springer.
- Nixon, S.W., R.W. Fulweiler, B.A. Buckley, S.L. Granger, B.L. Nowicki, and K.M. Henry. 2009. The impact of changing climate on phenology, productivity, and benthic-pelagic coupling in Narragansett Bay. *Estuarine, Coastal and Shelf Science* 82: 1–18.
- Oviatt, C.A. 2004. The changing ecology of temperate coastal waters during a warming trend. *Estuaries* 27: 895–904.
- Oviatt, C.A. 2008. Impacts of nutrients on Narragansett Bay productivity: a gradient approach. In *Science for ecosystem-based management: Narragansett Bay in the 21st century*, ed. A. Desbonnet and B.A. Costa-Pierce, 523–544. New York: Springer.
- Oviatt, C.A., A. Keller, and L. Reed. 2002. Annual primary production in Narragansett Bay with no bay-wide winter–spring phytoplankton bloom. *Estuarine, Coastal and Shelf Science* 54: 1013–1026.
- Pearson, T.H., and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology: An Annual Review* 16: 229–311.
- Pilson, M.E.Q. 1985. On the residence time of water in Narragansett Bay. *Estuaries* 8: 2–14.
- Pinksky, M.L., B. Worm, M.J. Fogarty, J.L. Sarmiento, and S.A. Levin. 2013. Marine taxa track local climate velocities. *Science* 341: 1239–1242.
- Pittman, S.J., C. Caldwell, S.D. Hile, and M.E. Monaco. 2007. Using seascape types to explain the spatial patterns of fish in the mangroves of SW Puerto Rico. *Marine Ecology Progress Series* 348: 273–284.
- Pittman, S.J., R. Kneib, and C. Simenstad. 2011. Practicing coastal seascape ecology. *Marine Ecology Progress Series* 427: 187–190.
- Powers, R. 2015. Boston blackbacks. On the water New England edition, May 2015. pp 50–55.
- Prell, W., E. Saarman, D.W. Murray, and C.F. Deacutis. 2004. *Summer-season, nighttime surveys of dissolved oxygen in upper Narragansett Bay (1999–2003)*.
- Raposa, K.B. 2009. Ecological geography of Narragansett Bay. In *An ecological profile of the Narragansett Bay estuarine research reserve*, ed. K.B. Raposa and M.L. Schwartz, 77–88. Narragansett RI: Rhode Island Sea Grant.
- Rhoads, D.C., and J.D. Germano. 1982. Characterization of organism-sediment relationships using sediment profile imaging: an efficient method of remote ecological monitoring of the seafloor (REMOTS system). *Marine Ecology Progress Series* 8: 115–128.
- Rhoads, D.C., and J.D. Germano. 1986. Interpreting long-term changes in benthic community structure: a new protocol. *Hydrobiologia* 142: 291–308.
- RIDEM (Rhode Island Department of Environmental Management). 2003. *The Greenwich Bay fish kill—August 2003: causes, impacts and responses*. Providence RI: Rhode Island Department of Environmental Management.
- RIDEM (Rhode Island Department of Environmental Management). 2005. *Plan for managing nutrient loadings to Rhode Island waters*. Providence RI: Rhode Island Department of Environmental Management.
- RIDEM (Rhode Island Department of Environmental Management). 2008. *Rhode Island wastewater treatment facility information*. Providence RI: Rhode Island Department of Environmental Management.
- Riemann, B., J. Carstensen, K. Dahl, et al. 2016. Recovery of Danish coastal ecosystems after reductions in nutrient loading: a holistic ecosystem approach. *Estuaries and Coasts* 39: 82–97.
- Rosenberg, R. 2001. Marine benthic faunal successional stages and related sedimentary activity. *Scientia Marina* 65: 107–119.

- Santos, S.L., and J.L. Simon. 1980. Response of soft-bottom benthos to annual catastrophic disturbance in a south Florida estuary. *Marine Ecology Progress Series* 3: 347–355.
- Shumchenia, E.J., and J.W. King. 2010. Evaluation of sediment profile imagery as a tool for assessing water quality in Greenwich Bay, Rhode Island, USA. *Ecological Indicators* 10: 818–825.
- Shumchenia, E.J., M.C. Pelletier, G. Cicchetti, S. Davies, C.E. Pesch, C.F. Deacutis, and M. Pryor. 2015. A biological condition gradient model for historical assessment of estuarine habitat structure. *Environmental Management* 55: 143–58.
- Snelgrove, P., and C.A. Butman. 1994. Animal-sediment relationships revisited: cause versus effect. *Oceanography and Marine Biology: An Annual Review* 32: 111–177.
- Solan, M., J.D. Germano, D.C. Rhoads, C. Smith, E. Michaud, D. Parry, F. Wenzhöfer, R. Kennedy, C. Henriques, E. Battle, D.A. Carey, L. Iocco, R.M. Valente, J. Watson, and R. Rosenberg. 2003. Towards a greater understanding of pattern, scale and process in marine benthic systems: a picture is worth a thousand worms. *Journal of Experimental Marine Biology and Ecology* 285(286): 313–338.
- Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, et al. 2007. Marine ecoregions of the world: a bioregionalisation of coastal and shelf areas. *BioScience* 57: 573–583.
- Stickney, A.P., and L.D. Stringer. 1957. A study of the invertebrate fauna of Greenwich Bay, Rhode Island. *Ecology* 38: 111–122.
- Valente, R.M., D.C. Rhoads, J.D. Germano, and V.J. Cabelli. 1992. Mapping of benthic enrichment patterns in Narragansett Bay, Rhode Island. *Estuaries* 15: 1–17.
- Wiens, J.A., N.C. Stenseth, B. Van Horne, and R.A. Ims. 1993. Ecological mechanisms and landscape ecology. *Oikos* 66: 369–380.
- Zajac, R.N. 2008. Challenges in marine, soft-sediment benthoscape ecology. *Landscape Ecology* 23: 7–18.