2018 Boston Harbor Benthic Monitoring Report

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EXECUTIVE SUMMARY

Boston Harbor was once considered among the most degraded urban estuaries in the country. Direct discharge of wastewater with limited treatment into the harbor had affected both water quality and ecological conditions. Both the EPA and the Federal Court prevailed upon the Commonwealth of Massachusetts to take actions to improve these conditions substantially and the Massachusetts Water Resources Authority (MWRA) was created in response. In 1985, MWRA initiated a multi-faceted plan to meet these mandates, most significantly including an upgrade to secondary treatment, elimination of sludge disposal into the harbor, and relocation of wastewater discharge through an offshore diffuser located 9.5 miles off Deer Island in Massachusetts Bay. Subsequently, improvements have been made to numerous Combined Sewer Overflows (CSOs) reducing another source of water quality concern.

Conditions in the sediments and the associated benthic infaunal community reflect cumulative water quality exposures. In order to evaluate changes to the ecosystem, particularly the benthic community, resulting from reductions in contaminated discharges over time, MWRA has conducted benthic monitoring in Boston Harbor on an ongoing basis since 1991. This report provides a summary of the results of the benthic surveys that were conducted in 2018. These include sediment conditions, benthic infauna, and sediment profile imagery.

Overall, the improvements in wastewater treatment and moving the outfall offshore have led to improvements in water quality and benthic habitat quality within Boston Harbor by reducing organic and nutrient loading. These reductions have enabled processes that enhance bioturbation or the mixing of sediment by organisms to predominate. At one time, the sediment organic matter in Boston Harbor was derived primarily from sewage but once changes in wastewater treatment and disposal were initiated, marine-derived organic material has become prevalent in the sediment. The infaunal community has responded to this change and subsequent biological activity has led to more aerobic and ‘healthy’ sediment conditions. Physical and biological properties of the soft substrate in Boston Harbor in 2018 exhibited minor changes from recent years but were consistent with longer-term trends (Rutecki et al. 2019a). Concentrations of both total organic carbon (TOC) and *Clostridium perfringens*, indicators of organic enrichment and deposition from municipal wastewater discharge, have remained low compared to the levels occurring prior to the offshore diversion.

Community structure measures also continue to point to improving benthic conditions. As municipal wastewater and sludge discharges were eliminated from the harbor, total abundance has trended downward, particularly for opportunistic species such as *Capitella capitata*, reaching a fairly stable level for about the last decade. Species richness, while variable, has trended upward as did evenness (J’), Shannon diversity (H’), and log series alpha diversity. Although abundances of individual taxa, such as the dominants *Ampelisca* spp., *Aricidea catherinae,* and *Polydora cornuta* have varied over time, the infaunal community structure and evidence from sediment profile imaging suggest that these changes relate more to normal physical disturbances than organic stress in the years since wastewater impacts to the Harbor have been reduced. The trends observed in previous years, including reduction of indicators of organic enrichment and increases in species diversity that continued in the 2018 survey are consistent with the recovery of benthic habitats from decades of inadequately treated sewage discharges.

1. INTRODUCTION

Boston Harbor was once considered among the most degraded urban estuaries in the country. Direct discharge of wastewater with limited treatment into the harbor had affected both water quality and ecological conditions. Both the EPA and the Federal Court prevailed upon the Commonwealth of Massachusetts to take actions to improve these conditions substantially and the Massachusetts Water Resources Authority (MWRA) was created in response. In 1985, MWRA initiated a multi-faceted plan to meet these mandates, most significantly including an upgrade to secondary treatment, elimination of sludge disposal into the harbor, and relocation of wastewater discharge through an offshore diffuser located 9.3 mi off Deer Island in Massachusetts Bay. In 2015, MWRA and its member communities completed a series of 35 projects intended to eliminate or minimize the impacts of Combined Sewer Overflows (CSOs) to Boston Harbor and its tributaries (Kubiak et al. 2016).

MWRA has conducted benthic monitoring in Boston Harbor on an ongoing basis since 1991 to evaluate changes to the ecosystem, in particular the benthic community, as contaminated discharges have been reduced over time. This report provides a summary of the results of the benthic surveys that were conducted in 2018. These include sediment conditions, benthic infauna, and sediment profile imagery.

2. METHODS

Benthic monitoring in Boston Harbor continued at the same stations that have been surveyed since 1991. This survey comprises three components: sediment conditions (grain size, total organic carbon, and *Clostridium perfringens*), benthic infauna, and sediment profile imaging (SPI).

Methods used to collect, analyze, and evaluate all sample types remain largely consistent with those reported by Rutecki et al. (2019a) for previous monitoring years. Detailed descriptions of the methods are contained in the Quality Assurance Project Plan (QAPP) for Benthic Monitoring (Rutecki et al. 2017). A brief overview of methods, focused on information that is not included in the QAPP, is provided in Sections 2.1 to 2.3.

2.1 FIELD METHODS

Sediment and infauna sampling was conducted at 9 stations in August 2018 (Figure 2-1). Soft-bottom stations were sampled for grain size composition; total organic carbon (TOC); the sewage tracer *Clostridium perfringens*, an anaerobic bacterium found in the intestinal tract of mammals; and benthic infauna. One sediment sample and triplicate infauna samples were collected at each station on August 2-3, 2018.

SPI samples were collected in triplicate at 61 stations from August 5-7, 2018 (Figure 2-1).

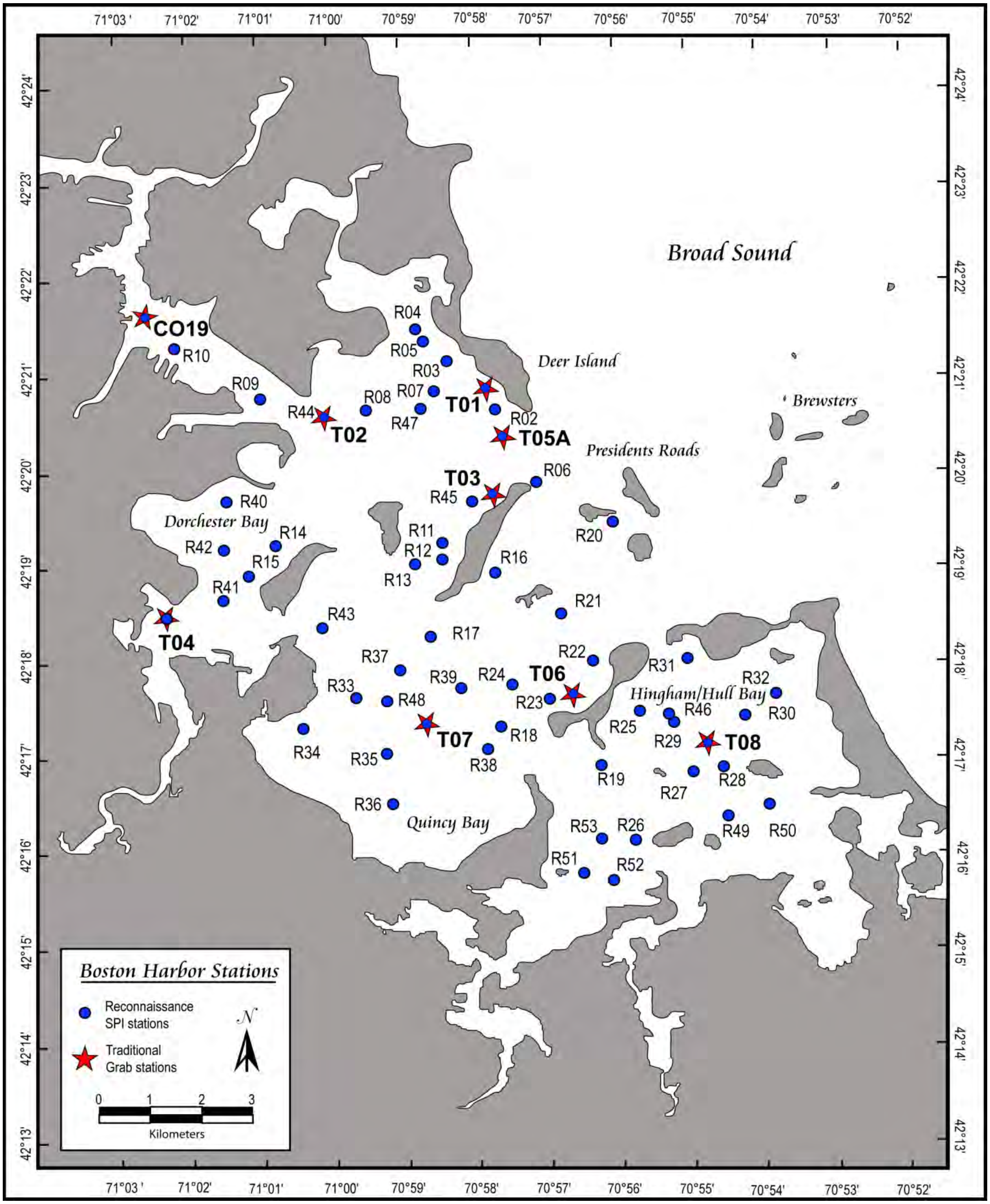

Figure 2-1. Locations of soft-bottom sampling and sediment profile imaging stations for 2018.

2.2 LABORATORY METHODS

Laboratory methods for benthic infauna and SPI image analyses were consistent with the QAPP (Rutecki et al. 2017). Two of the infauna samples were randomly selected for processing, while the third was archived. Analytical methods for grain size, TOC, and *Clostridium perfringens* are described in (Constantino et al. 2014).

2.3 DATA HANDLING, REDUCTION, AND ANALYSIS

All benthic data were extracted directly from the HOM database and imported into Excel. Data handling, reduction, and graphical presentations were performed in Excel or SAS (version 9.3), as described in the QAPP (Rutecki et al. 2017) or in Maciolek et al. (2008). Data are presented graphically as means (averages per station) unless otherwise noted. Shannon-Weiner (H') was calculated using log base 2. Multivariate analyses were performed using PRIMER v6 (Plymouth Routines in Multivariate Ecological Research) software to examine spatial patterns in the overall similarity of benthic assemblages in the survey area (Clarke 1993, Warwick 1993, Clarke and Green 1988). These analyses included classification (cluster analysis) by hierarchical agglomerative clustering with group average linking and ordination by non-metric multidimensional scaling (MDS). Bray-Curtis similarity was used as the basis for both classification and ordination. Prior to analyses, infaunal abundance data were fourth-root transformed to ensure that all taxa, not just the numerical dominants, would contribute to similarity measures.

Cluster analysis produces a dendrogram that represents discrete groupings of samples along a scale of similarity. This representation is most useful when delineating among sites with distinct community structure. MDS ordination produces a plot or "map" in which the distance between samples represents their rank ordered similarities, with closer proximity in the plot representing higher similarity. Ordination provides a more useful representation of patterns in community structure when assemblages vary along a steady gradation of differences among sites. Stress provides a measure of adequacy of the representation of similarities in the MDS ordination plot (Clarke 1993). Stress levels less than 0.05 indicate an excellent representation of relative similarities among samples with no prospect of misinterpretation. Stress less than 0.1 corresponds to a good ordination with no real prospect of a misleading interpretation. Stress less than 0.2 still provides a potentially useful two-dimensional picture, while stress greater than 0.3 indicates that points on the plot are close to being arbitrarily placed. Together, cluster analysis and MDS ordination provide a highly informative representation of patterns of community-level similarity among samples. The "similarity profile test" (SIMPROF) was used to provide statistical support for the identification of faunal assemblages (i.e., selection of cluster groups). SIMPROF is a permutation test of the null hypothesis that the groups identified by cluster analysis (samples included under each node in the dendrogram) do not differ from each other in multivariate structure.

Temporal patterns in the infaunal community and sediment chemistry were evaluated in the context of the four discharge periods described by Taylor (2006) offset by one year to allow for a lag in response time. These evaluations generally focused on Stations T01 to T08 (traditional stations, Periods I-IV) and excluded data from Station C019 which are only available since 2004 (Period IV).

Details on parameters and analysis of SPI images can be found in Diaz et al. (2008). For this report, quantitative SPI parameters (e.g. apparent redox potential discontinuity [aRPD] layer depth) were averaged from the three replicate images. For categorical parameters (e.g. presence of biogenic structures), the median value of the three replicate images was assigned to a station. As the selection of station locations in the Harbor was non-random, fixed-effect nominal logistic models, which treat each station measurement as a separate observation, were used to analyze patterns in categorical data (Agresti 1990). For continuous variables, general linear models were used to test for differences within and between quantitative parameters. When appropriate, normality was checked with the Shapiro–Wilk test and homogeneity of variance with Bartlett’s test. Trends in quantitative variables were tested using simple linear regression and segmented linear regression. Significance of odds was tested using logistic regression. All statistical tests were conducted with the statistical package R version 3.5.2 (2018-12-20) "Eggshell Igloo” (The R Foundation for Statistical Computing).

3. RESULTS AND DISCUSSION

3.1 SEDIMENT CONDITIONS

Sediment conditions in Boston Harbor were characterized in 2018 by measuring three parameters at each of the nine sampling stations: (1) grain size (gravel, sand, silt, and clay), (2) total organic carbon, and (3) *Clostridium perfringens* (Table 3-1).

Grain Size. Surface sediments at the nine stations sampled during 2018 included a wide range of sediment types (Table 3-1, Figure 3-1). Grain size profiles ranged from predominantly sand (e.g., T08, T05A) to almost entirely silt and clay (i.e., C019 and T04). Most stations had mixed sediments. The outer harbor stations (T01, T05A, T06, and T08) generally had more than 50% sand with varying fractions of silt-clay. Outer harbor stations T02 and T07 had nearly equal proportions of sand and silt-clay. Grain size has been variable at Station T03, in the lee of Long Island, having more fines than sand in 2017 and 2018 but more sand in 2016. The grain size composition at each station in 2018 generally remained within the ranges reported in prior years although two stations (T03 and T05A) exhibited an increase in percent fines of about 10% compared to 2017. T01 (just west of Deer Island) and T02 (at the mouth of the Inner Harbor) have exhibited the largest fluctuations in fine sediments over the years and T04 has occasionally exhibited fairly large drops in percent fines (Figure 3-2).

Total Organic Carbon. Concentrations of total organic carbon (TOC) in 2018 were generally similar to 2017, remaining within the ranges reported in prior years (Figure 3-3). Higher TOC values were generally associated with higher percent fine sediments (silt + clay; Figures 3-2 and 3-3). During 2018, Stations T03, T04, and C019 had among the highest concentrations of TOC and also had the highest proportions of fines (Table 3-1). T04 and C019 have typically had the highest TOC. TOC levels at T03 have been below 3% since 2005 and below 2.5% since 2011; levels in 2018 are consistent with this trend. The lowest TOC concentrations for 2018 were reported at Stations T01 and T08 which have predominantly sand sediments.

***Clostridium perfringens*.** Spores of *Clostridium perfringens*, an anaerobic bacterium found in the intestinal tract of mammals, provide a sensitive tracer of effluent solids. Abundances of *C. perfringens* (normalized to percent fines) during 2018 (Table 3-1) were generally low and comparable across all monitoring stations, especially when compared to historic data (Figure 3-4). In the Outer Harbor, abundances have exhibited the highest variability among years at T04, a depositional site in Savin Hill Cove. After a period of decline from 2008-2012, followed by an increase in 2013 and 2014, *C. perfringens* counts at T04 decreased again in 2015, remained steady in 2016, declined in 2017, and increased in 2018 (Figure 3-4). *C. perfringens* counts at Station T08 were considerably higher in 2016 than in the present year and most recent years; this was largely an artifact of normalizing the counts by percent fine sediments. Percent fine sediments were extremely low at T08 in 2016. Except at Inner Harbor station C019, *C. perfringens* counts at other Harbor stations have remained consistently below historical averages since around 1999 (Figure 3-4).

Table 3-1. Monitoring results for sediment condition parameters in 2018.

Parameter		C019	T01	T02	T03	T04	T05A	T06	T07	T08
Grain Size	**Gravel (%)**	**0**	**3.40**	**0**	**0**	**0**	**0.06**	**0**	**0**	**0.67**
	Sand (%)	**5.6**	**78.4**	**52.8**	**30.7**	**13.8**	**78.4**	**72.8**	**52.8**	**88.2**
	Silt (%)	**58.9**	**12.8**	**30.2**	**42.4**	**66.3**	**14.0**	**16.8**	**26.0**	**5.4**
	Clay (%)	**34.4**	**5.4**	**17.0**	**27.0**	**19.8**	**7.6**	**10.4**	**21.2**	**5.6**
	Percent Fines (Silt + Clay)	**94.4**	**18.2**	**47.2**	**69.3**	**86.2**	**21.6**	**27.2**	**47.2**	**11.0**
Total Organic Carbon	**TOC (%)**	**2.7**	**0.4**	**1.1**	**2.2**	**4.5**	**0.5**	**1.0**	**2.1**	**0.6**
Clostridium perfringens	**Not Normalized**	**3730**	**240**	**994**	**4060**	**4390**	**345**	**478**	**822**	**573**
	Normalized (cfu/g dry/%fines)	**39.5**	**13.2**	**21.1**	**58.6**	**51.0**	**16.0**	**17.6**	**17.4**	**52.0**

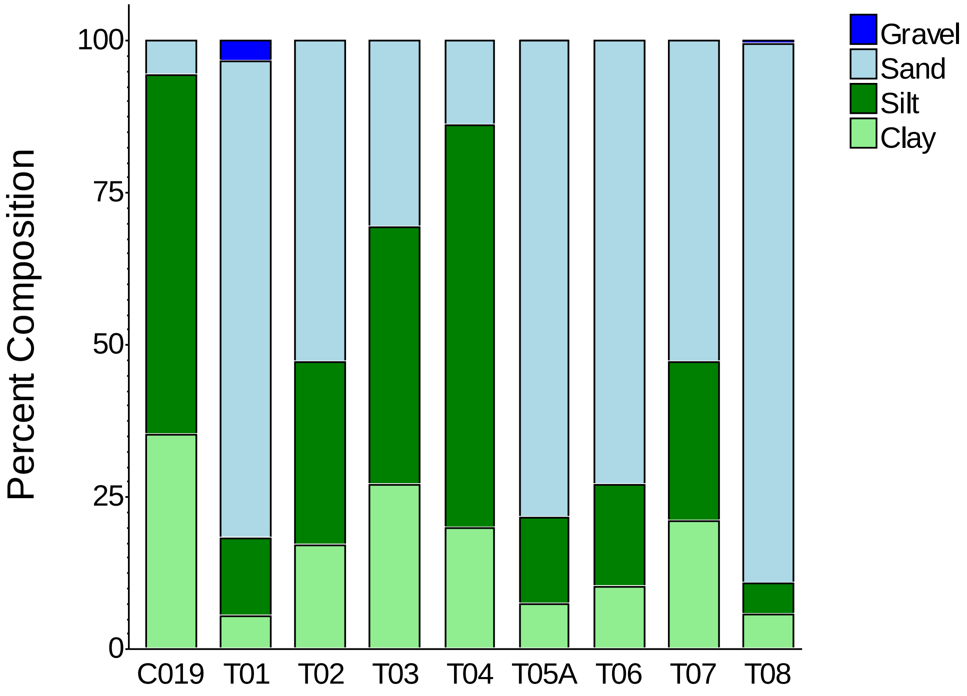

Figure 3-1. Monitoring results for 2018 sediment grain size in Boston Harbor.

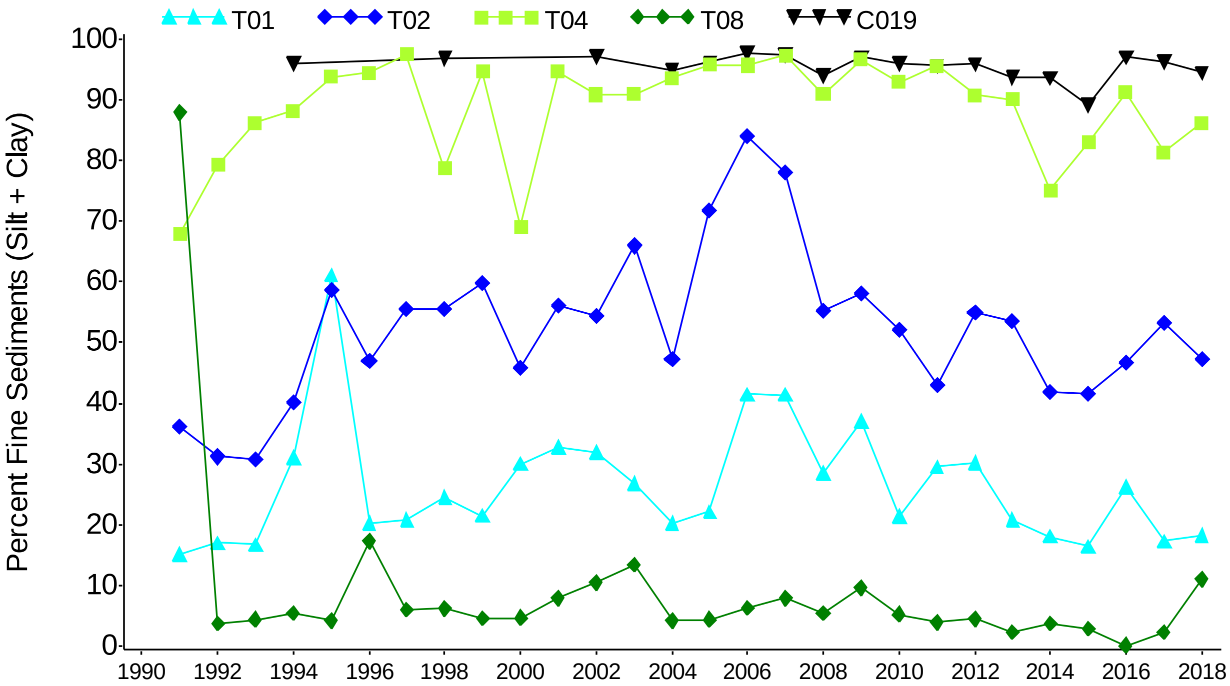

Figure 3-2. Mean percent fine sediments at five stations in Boston Harbor, 1991 to 2018.

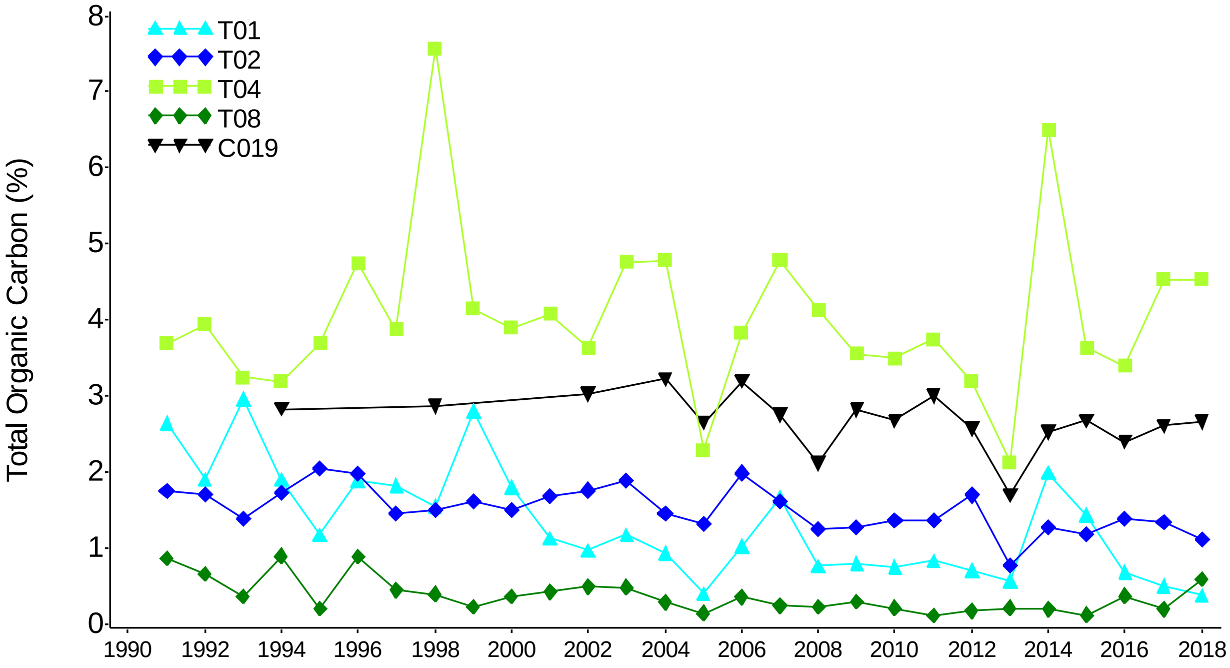

Figure 3-3. Mean concentrations of TOC at five stations in Boston Harbor, 1991 to 2018.

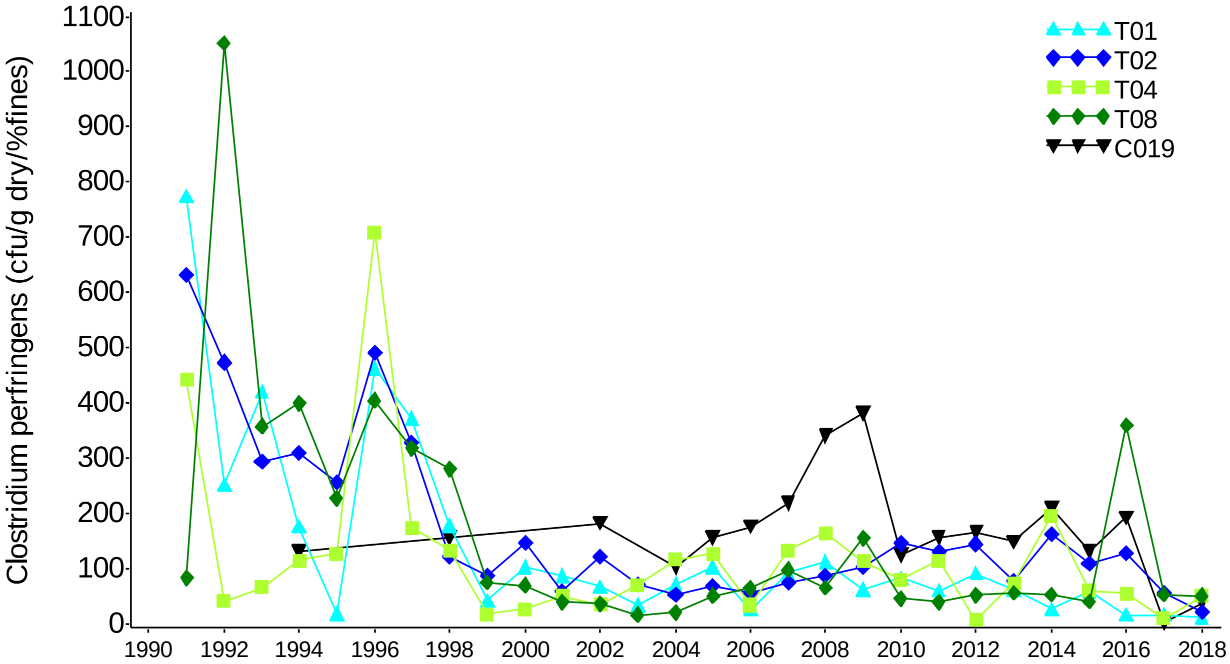

Figure 3-4. Mean normalized concentrations of *Clostridium perfringens* at five stations in Boston Harbor, 1991 to 2018.

Association with wastewater treatment improvements. Taylor (2006) defined time periods related to various changes in MWRA's management of wastewater and sludge treatment and discharges. Significant events that had a likelihood of affecting the harbor benthic community included ending the discharge of sludge in 1991 (end of Period I); opening of the new primary treatment plant at Deer Island (1995) and its upgrade to secondary treatment between 1997 and 2001 (Period II); implementation of the transfer of wastewater from Nut Island to Deer Island, ending the Nut Island discharge in 1998 (onset of Period III); and implementation of the offshore outfall in 2000 (onset of Period IV).

Results during 2018 for grain size, TOC, and *C. perfringens* in Boston Harbor sediments, are consistent with monitoring results from previous post-outfall implementation years (Maciolek et al. 2008, 2011; Rutecki et al. 2019a). Concentrations of both TOC and *C. perfringens* at the traditional harbor stations (T01 to T08) have remained lower during the past decade than those reported during the 1990s (Figures 3-5 and 3-6). In Figures 3-5 and 3-6, Period IV has been divided into five-year segments to show potential changes of TOC and *C. perfringens* concentrations over time in this period. These findings are consistent with other changes documented in the Harbor following improvements to the collection, treatment, and disposal of wastewater as part of the Boston Harbor Project (Taylor 2006, Maciolek et al. 2008).

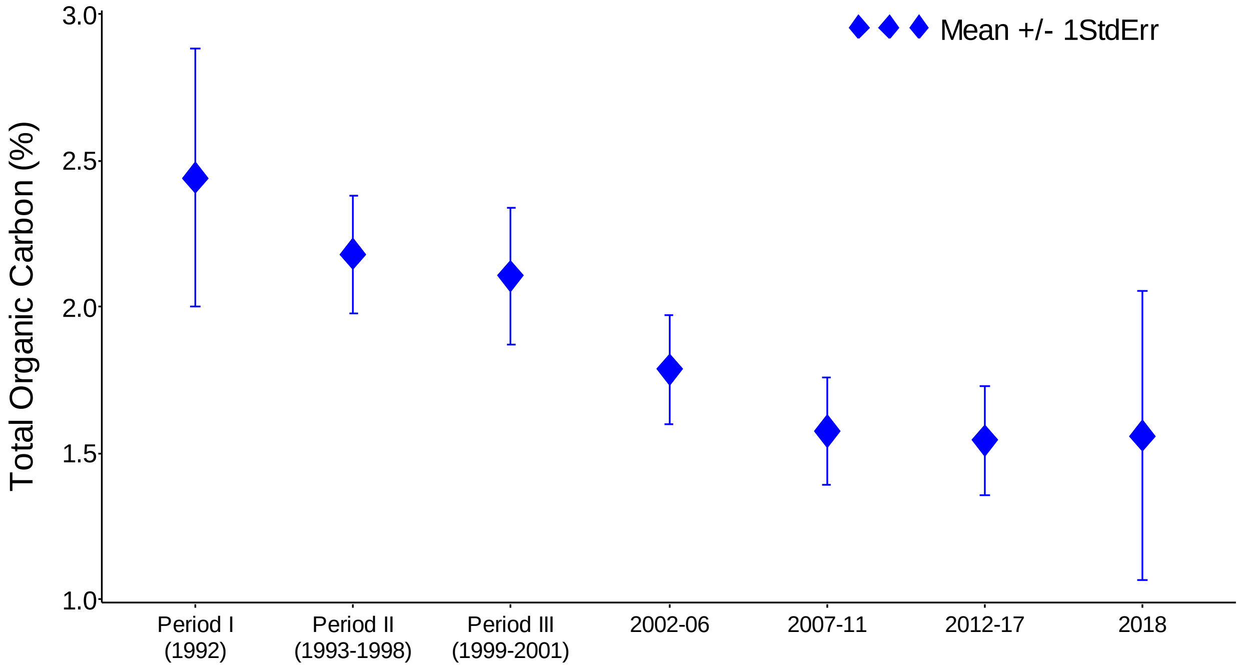

Figure 3-5. Comparison of TOC across time periods (offset by one year) at Boston Harbor stations (T01-T08) from 1992 to 2018 (1991 excluded).

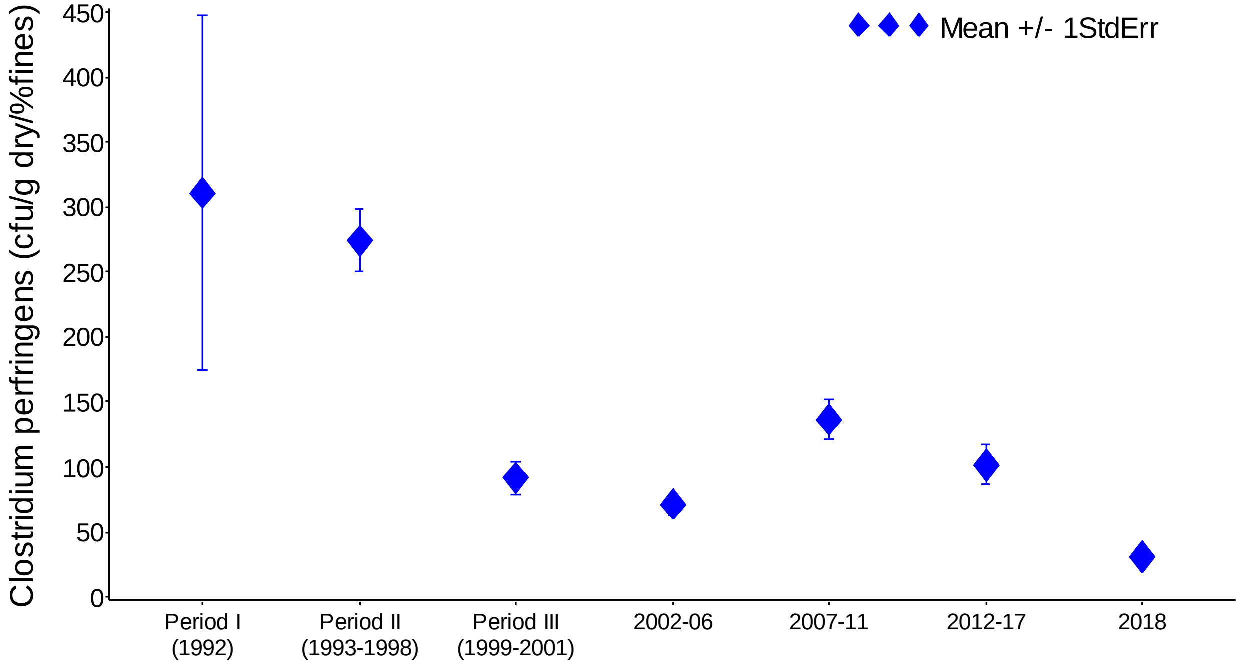

Figure 3-6. Comparison of *Clostridium perfringens* across time periods (offset by one year) at Boston Harbor stations (T01-T08) from 1992 to 2018 (1991 excluded).

3.2 BENTHIC INFAUNA

3.2.1 Community Parameters

A total of 39,032 infaunal organisms were counted from the 18 samples processed in 2018. Organisms were classified into 148 discrete taxa, 136 of which were species-level identifications. More than 99% of the individuals were identified to species; all remaining individuals were identified to genus or family. These species richness values are consistent with values observed in the harbor in recent years. Abundance values reported herein reflect the total counts from both species and higher taxonomic groups, while diversity measures are based on the species-level identifications only (Table 3-2). Data are presented in figures as means (averages per station) unless otherwise noted.

Table 3-2. Mean 2018 infaunal community parameters by station.

Station	Total Abundance	Number of Species	Pielou's Evenness (J')	Shannon-Wiener Diversity (H')	Log-series alpha
C019	1164.5	18.0	0.51	2.13	3.03
T01	2232.0	44.5	0.57	3.12	7.96
T02	2528.5	59.5	0.55	3.23	10.95
T03	3459.5	49.5	0.51	2.86	8.23
T04	1570.0	9.5	0.14	0.42	1.36
T05A	4192.0	48.5	0.40	2.22	7.72
T06	3337.5	51.0	0.54	3.07	8.55
T07	412.5	23.0	0.56	2.54	5.26
T08	655.0	38.5	0.58	3.04	8.96

At most stations mean total abundance values reported for 2018 were lower than values in 2017. Decreases in mean abundance were observed at six stations (T01, T02, T03, T04, T07, and T08) and ranged from 4 to 31% (Rutecki et al. 2019a). Station T07 typically has low infaunal abundance. Stations T06 and C019 exhibited increases in mean abundance of 9 and 15%, respectively. Station T05A increased by more than 3.4 times the mean abundance observed in 2017 (Rutecki et al. 2019a). Abundances in 2018 were within the range observed during the post-offshore diversion period but relatively low compared to pre-diversion values (Figure 3-7). The eight most abundant species in 2018 each contributed 2.5% or more of the animals counted, and together they provided nearly 72% of the total abundance across the eight traditional benthic stations. Most of the species that dominated the infauna at these stations in the past several years continued to do so in 2018 (Table 3-3) although the rank order changed. The tube-dwelling amphipod *Ampelisca* spp. decreased in 2018 compared to 2017 due to lower abundances at T03 and T07. Seven species of polychaetes were among the most abundant taxa in 2018. The five most abundant taxa in 2018 have frequently been among the most abundant in the harbor during previous years (Table 3-4). Certain spatial patterns of abundance also appeared to be

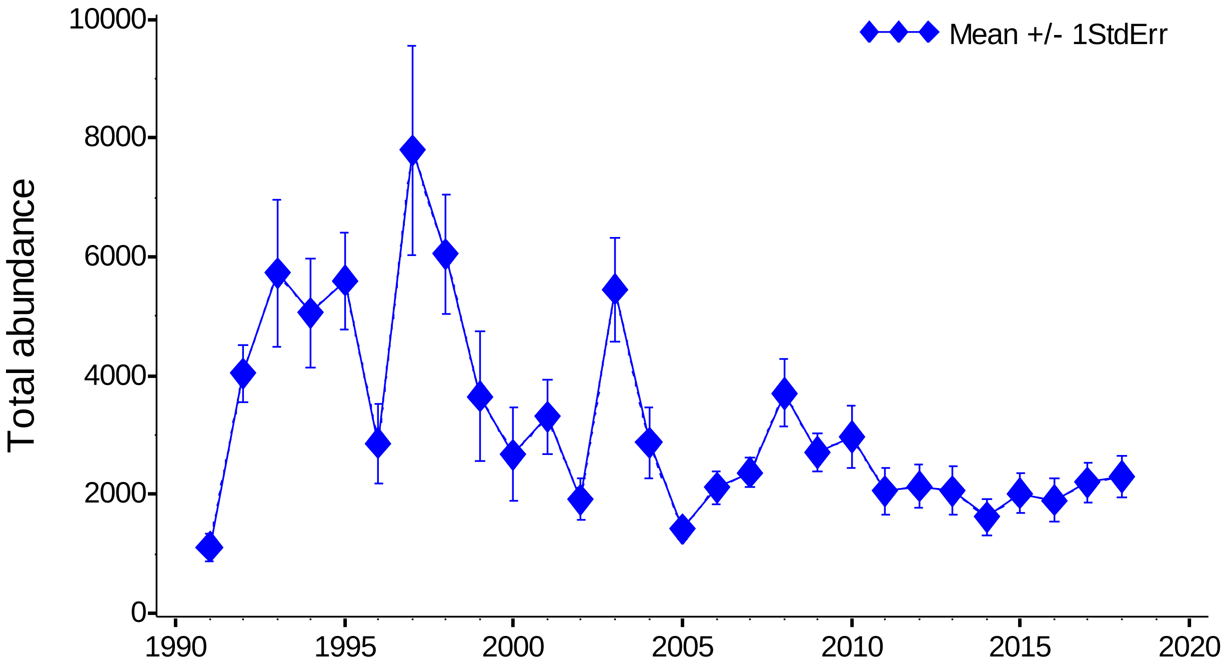

Figure 3-7. Mean abundance of benthic infauna at eight Boston Harbor stations (T01-T08), 1991-2018.

Table 3-3. Dominant taxa at eight grab stations (T01-T08) in Boston Harbor in August 2018.

Taxon	**Total 2018 Abundance (compared with 2017)[a]**
Polydora cornuta	10,750 (increase)
Aricidea catherinae	6,844 (increase)
Streblospio benedicti	3,295 (decrease)
Scoletoma hebes	2,209 (similar)
Ampelisca spp.	1,566 (decrease)
Polygordius jouinae	1,193 (increase)
Clymenella torquata	1,180 (decrease)
Tharyx acutus	1,018 (increase)

[a] increase or decrease indicates ≥25% change from previous year

consistent with previous years; T04, T07 and C019 continued to support low infaunal abundances (Table 3-2). As previously observed, Station T03 supported among the highest abundances among the harbor stations.

Temporally, benthic infaunal abundance in the harbor has been controlled by a limited number of species (Table 3-4). Although these dominants have varied somewhat over the course of the surveys, four taxa (*Ampelisca* spp., *Aricidea catherinae*, *Limnodriloides medioporus*, and *Polydora cornuta*) have been among the most abundant organisms during the baseline and subsequent periods. Annual abundances of these species are presented in Figure 3-8. *Aricidea catherinae* and *Limnodriloides medioporus* have exhibited little interannual variation in abundance whereas both *Polydora cornuta* and *Ampelisca* spp. have exhibited fluctuations of one to two orders of magnitude and abundances of both species are currently low. Both of these species reside in the upper substrate layer so variability could be related to many factors, including changes in organic enrichment, physical forces such as storm events, and reproductive success in a given year as was described in previous reports (Maciolek et al. 2006, 2011). It is noteworthy that abundances of the oligochaete *Tubificoides intermedius* (not shown) have steadily increased.

Table 3-4. Mean abundance per sample of dominant taxa during four discharge periods at eight Boston Harbor stations (T01-T08), 1992-2018.

Phylum	Higher taxon	Family	Species[a]	Period I	Period II	Period III	Period IV	2018
Annelida	Polychaeta	Capitellidae	*Capitella capitata* complex	65.2	88.8	3.4	6.1	7.8
		Cirratulidae	*Tharyx acutus*	50.6	111.8	52.4	53.7	63.62
		Lumbrineridae	*Scoletoma hebes*	3.4	10.4	4.2	75.9	138.1
		Maldanidae	*Clymenella torquata*	34.7	17.6	10.6	22.8	73.8
		Nephtyidae	*Bipalponephtys neotena*[b]	-	11.4	10.3	174.2	45.9
		Paraonidae	*Aricidea catherinae*	325.0	237.4	204.3	241.9	427.8
		Polygordiidae	*Polygordius jouinae*	19.2	22.4	44.0	18.1	74.6
		Spionidae	*Polydora cornuta*	525.8	1053.0	269.6	272.6	671.9
			Streblospio benedicti	236.0	298.6	27.7	87.4	205.9
	Oligochaeta	Tubificidae	*Limnodriloides medioporus*	484.6	297.9	315.2	231.2	54.0
			Tubificoides intermedius	42.6	101.4	231.2	238.3	42.4
Arthropoda	Amphipoda	Ampeliscidae	*Ampelisca* spp.	354.3	1698.3	1205.9	528.4	97.9
		Corophiidae	Corophiidae spp.	16.1	336.2	23.0	1.5	4.1
			Crassicorophium bonellii	7.9	217.3	37.3	7.1	2.2
			Leptocheirus pinguis	29.0	117.4	66.0	74.3	43.2
		Photidae	*Photis pollex*	11.4	77.0	86.8	29.1	1.4
		Phoxocephalidae	*Phoxocephalus holbolli*	28.0	116.9	125.9	6.1	4.7

[a] Dominants identified as taxa cumulatively composing 75% of total abundance in each period.

[b] previously identified as *Nepthys cornuta*.

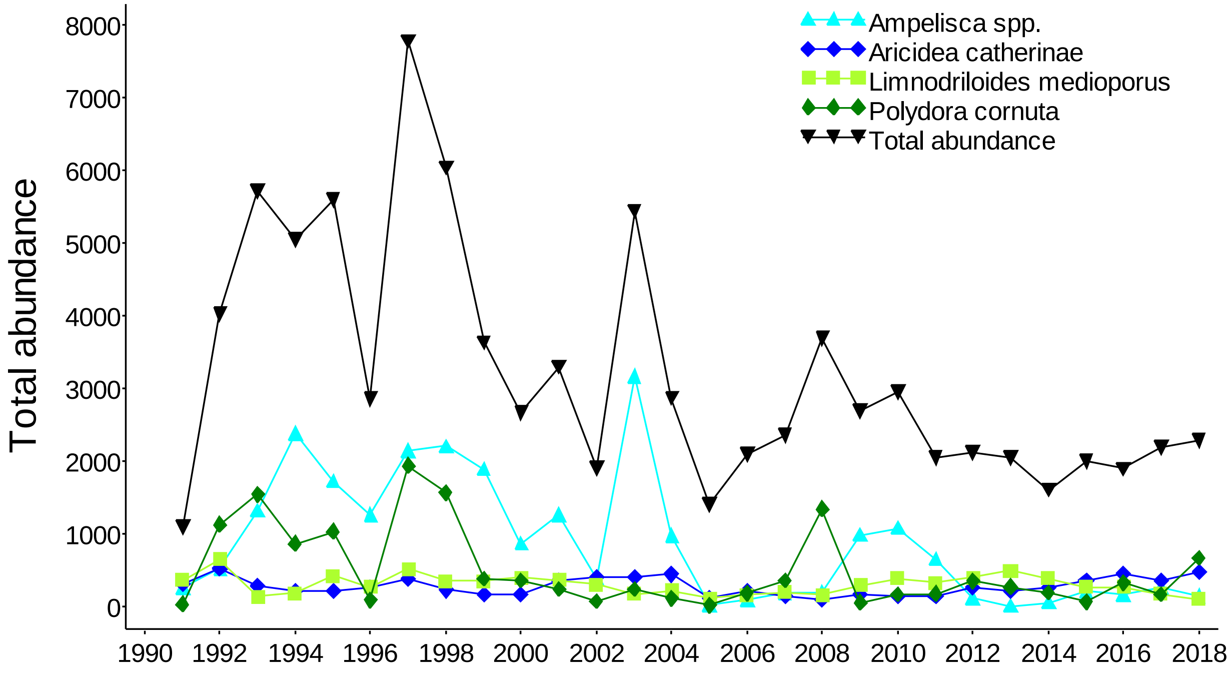

Figure 3-8. Total abundance of four dominant taxa at eight Boston Harbor stations (T01-T08), 1991-2018.

Patterns of abundance of *Ampelisca* spp. have been of particular interest throughout the history of the program due to the numerical dominance of this taxon in Harbor assemblages. Previous annual reports on the harbor surveys have related changes in the spatial extent of *Ampelisca* spp. beds and abundances of individual amphipods to changes in organic loading (the period from 1991 through about 1999) and storm events and subsequent recruitment (around 2005 through 2009; Maciolek et al. 2006, 2011). Following a period of low abundances (2005-2008), average abundances of *Ampelisca* at these stations reached moderately high levels from 2009 through 2011, declining in 2012-2014 to levels comparable to those seen in 2005-2008, and increase slightly in 2015 and 2017 (Figure 3-9). *Ampelisca* was more widespread in 2015 (seven stations) compared to 2014 (five stations) although in both years more than 90% of the population was found at Station T03 in the vicinity of the Main Ship Channel (Figure 3-10). Severe winter storms in 2004-2005 and 2012-2013 have been suggested as a major factor in disturbing *Ampelisca* mats. Maintenance dredging conducted in the nearby President Roads Anchorage and Main Ship Channel in 2004-2005 could also have resulted in disturbances (sedimentation or substrate contact from anchored vessels) to stations T03 and T05A. Abundances at Stations T03 and T06 decreased in 2018 compared to 2017. As observed in the past three years, Station T03 supported the largest population of this species.

The numbers of species reported for 2018 ranged from 9.5 to 59.5 per station and averaged 40 species per station. These values were higher than numbers reported in most years in the 1990s and about average for the 2000s, when species richness exhibited inter-annual variation (Table 3-2, Figure 3-11). Mean species richness was slightly lower in 2018 than in 2017 (Figure 3-11). Stations T02, T03, and T08 were among the most species-rich stations from 2011 through 2017 (Rutecki et al. 2019a); in 2018 species richness at Stations T02 and T03 was above the average for the Harbor, while below the average at Station T08. Station T04 has consistently supported the lowest species richness in this time frame. Species richness at T07 in Quincy Bay, which reached unusual peaks in 2013 and 2016 (Pembroke et al. 2014, 2017), exhibited typical species richness in 2018.

When averaged across the eight outer harbor stations, two measures of community structure, Pielou's evenness and Shannon-Weiner diversity, decreased in 2018 as they did in 2016 and 2017 (Figures 3-12 and 3-13, Table 3-2). Within each station, differences in these metrics between 2017 and 2018 were typically small and, in general, spatial patterns in these parameters were similar between the two years suggesting that the integrity of the benthic infaunal community had not changed. Average diversity for the harbor stations, as measured by log-series alpha, decreased in 2018 compared to 2017 (Rutecki et al. 2019a). Although the value in 2018 was below the relatively high values seen in 2012 and 2016 it was well above pre-diversion values (1991-2000; Figure 3-14). The highest log-series alpha diversity occurred at T02 where this measure has increased annually since 2016 (Pembroke et al. 2017, Rutecki et al. 2019a). Typically Stations T01 or T08 have the highest log-series alpha diversity in the Harbor. Consistent with recent patterns, log-series alpha diversity was lowest at T04 in 2018 (Table 3-2; Pembroke et al. 2017, Rutecki et al. 2019a). The largest change in log-series alpha compared to 2017 was the increase at T06.

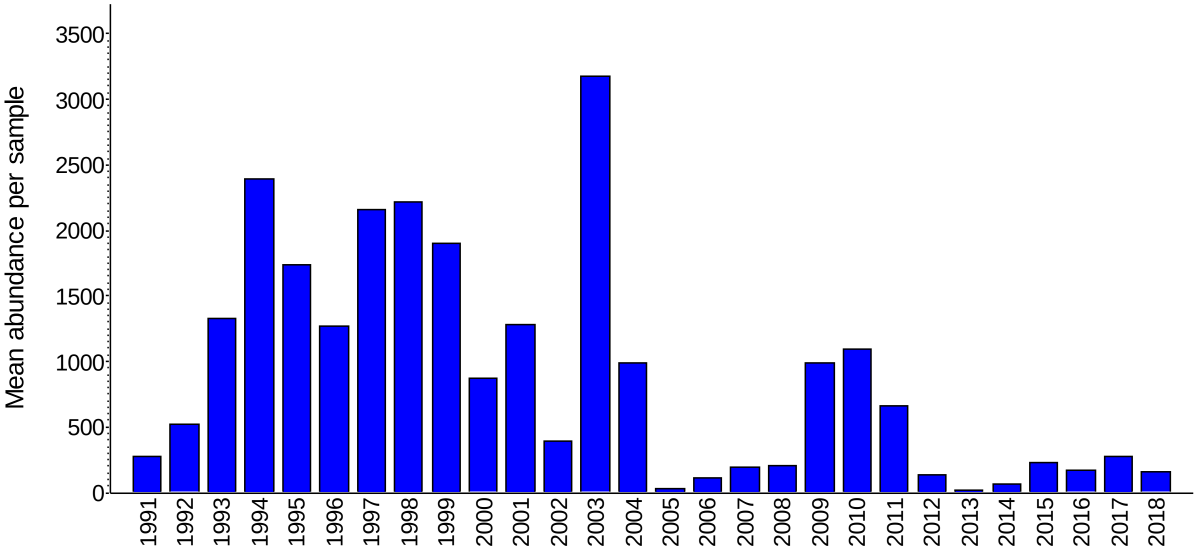

Figure 3-9. Mean annual abundance of *Ampelisca* spp. averaged over eight Boston Harbor stations (T01-T08), 1991-2018.

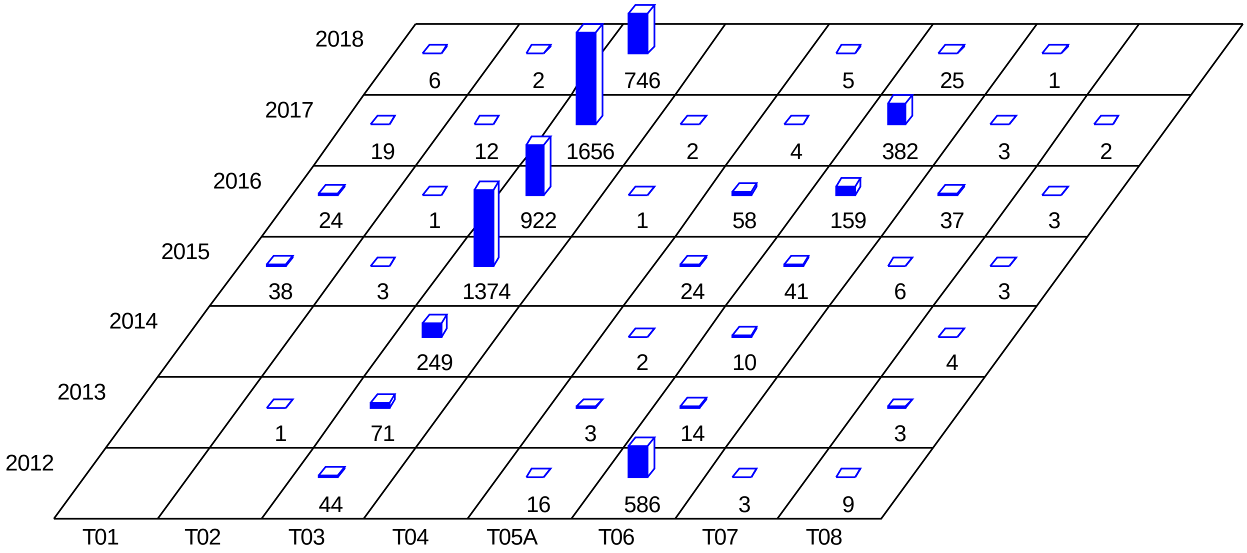

Figure 3-10. Spatial distribution of *Ampelisca* spp. at eight Boston Harbor stations (T01-T08), 2012-2018.

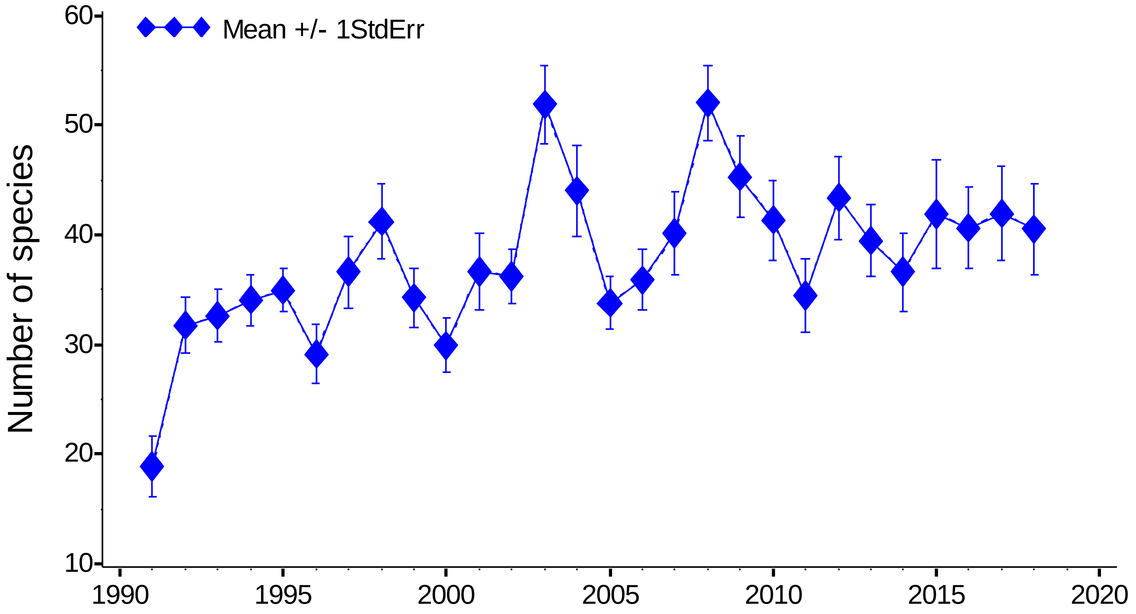

Figure 3-11. Mean species richness at eight Boston harbor stations (T01-T08), 1991-2018.

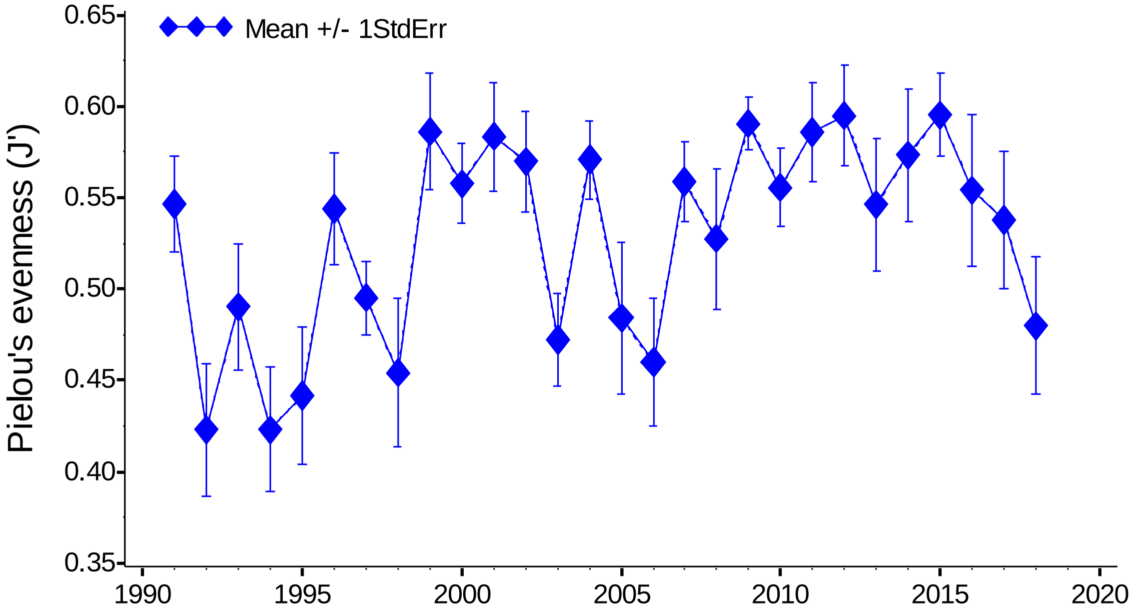

Figure 3-12. Mean community evenness at eight Boston Harbor stations (T01-T08), 1991-2018.

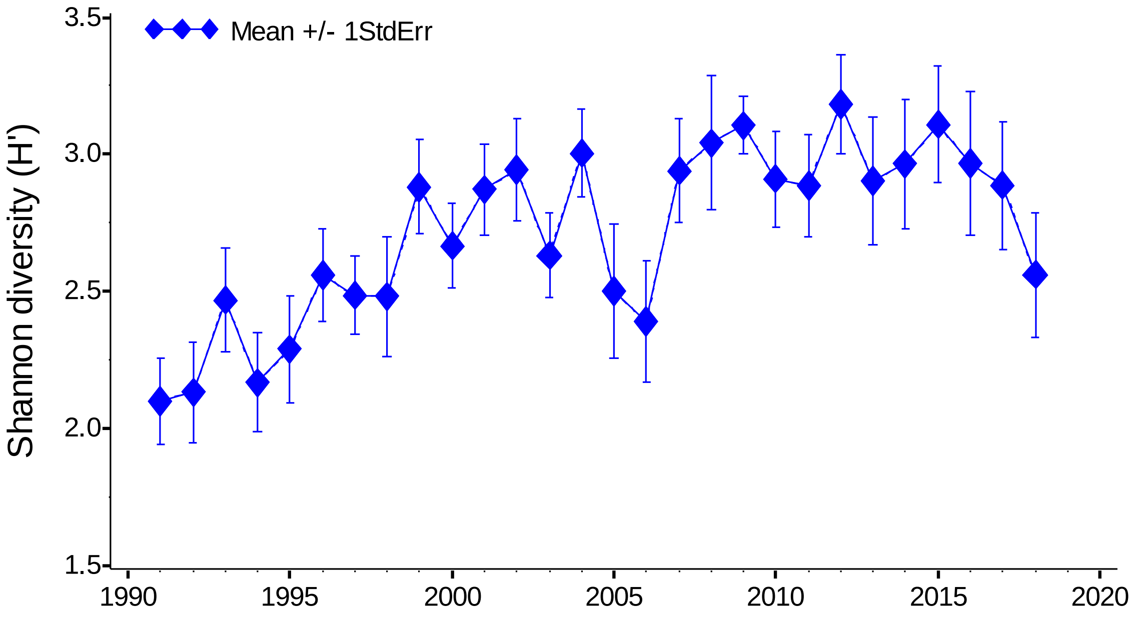

Figure 3-13. Mean Shannon-Weiner diversity at eight Boston harbor stations (T01-T08), 1991-2018.

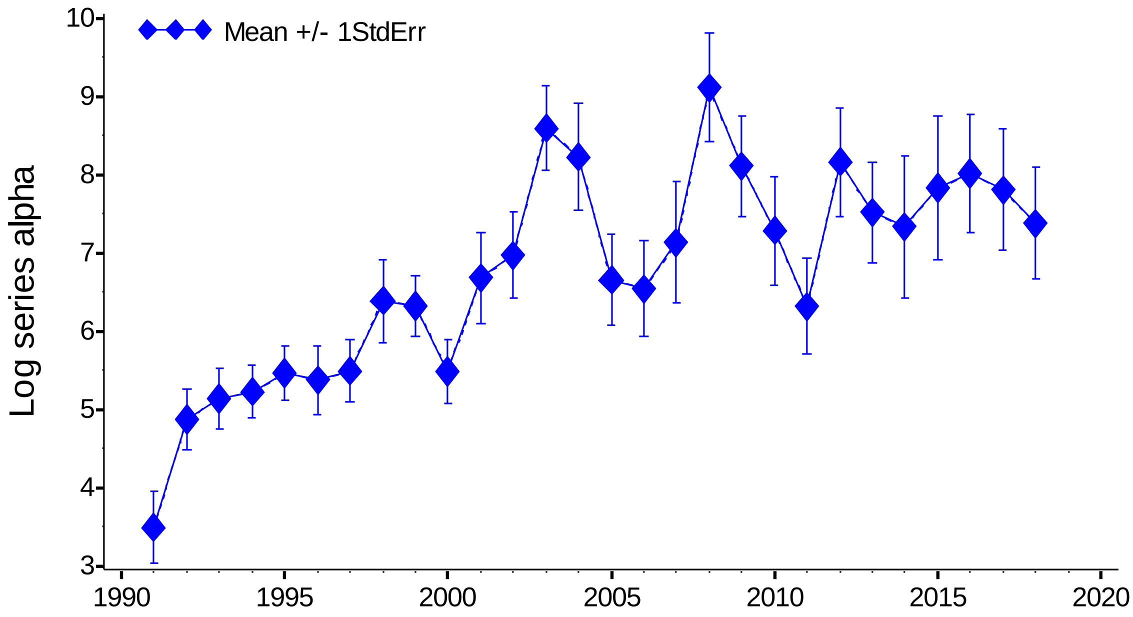

Figure 3-14. Mean log-series alpha diversity at eight Boston Harbor stations (T01-T08), 1991-2018.

3.2.2 Infaunal Assemblages

Multivariate analyses were used to assess spatial and temporal patterns in the faunal assemblages at the Boston Harbor sampling stations. Replicates within each station exhibited high similarities in community structure and grouped together. Four main assemblages were identified in a cluster analysis of the 18 samples from 2018 (Figure 3-15). Patterns identified through cluster analysis were confirmed in the MDS ordination plot for the 2018 Harbor samples (Figure 3-15). All assemblages were dominated by polychaetes or oligochaetes. Spatial patterns in the faunal assemblages of Boston Harbor reflect a habitat gradient from species poor stations C019, T04, and T07, to more diverse stations in the mid to outer Harbor. The Group IA assemblage was found at outer Harbor Stations T08 and T05A; the Group IB assemblage was found at outer Harbor to mid-Harbor stations (T01, T02, T03, and T06); the Group IIA assemblage was found at Stations T07 (Quincy Bay) and C019 (Inner Harbor); and the Group IIB assemblage was found at Station T04, at the mouth of Savin Hill Cove, in Dorchester Bay. The assemblage at Station T04 is typically the most different from all of the others. This gradient was also observed in the community parameters (Section 3.2.1) and likely reflects differences in tidal flushing in the Outer Harbor compared to the Inner Harbor along with differing proximity to sources of organic inputs, nutrients, and other possible contaminants.

Main Group I encompassed six stations (T01, T02, T03, T05A, T06, and T08) and abundances averaged 2,734 individuals across 48 species. Eight species (*Polydora cornuta, Aricidea catherinae, Scoletoma hebes, Ameplisca* spp., *Polygordius jouinae, Clymenella torquata, Tharyx acutus,* and *Limnodriloides medioporus*) contributed 83% of the total abundance (Table 3-5). The deeper-dwelling polychaete *C. torquata* is considered an indicator of a stable community. Within Group I, Stations T05A and T08 were distinct enough to form Subgroup IA dominated by *P. cornuta, P. jouinae, and Ameritella agilis,* and characterized by moderately high species richness (averaging 43 species per collection). Subgroup IB was comprised of Stations T01, T02, T03, and T06 characterized by high abundance and species richness (averaging 51 species per collection). Dominants included *A. catherinae*, *P. cornuta*, *S. hebes* and *Ameplisca* spp. The main Group II consisted of Stations C019, T04, and T07 dominated by three species (*S. benedicti*, *Bipalponephtys neotena*, and *Cossura* sp. 1) which contributed 85% of the total abundance (Table 3-5). Stations C019 and T07 formed Subgroup IIA and were dominated by the three species that dominated the main Group II. This subgroup was characterized by low abundance and species richness (averaging 20 species per collection). Station T04 was distinct enough to form Subgroup IIB dominated by *S. benedicti*, total abundance was moderate and species richness was low. Sediment conditions, particularly percent fines and TOC were likely factors in distinguishing the benthic communities among the Boston Harbor stations. Subgroup IA (T05A and T08) was predominantly sand and very low TOC. Sediments at stations in Subgroup IB ranged from about 31 to 78% sand and moderate to low TOC (0.4 to 2.2%). Sediments at stations C019 and T07 (Subgroup IIA) and T04 (Subgroup IIB) were predominantly fines and TOC was higher than at other locations (Table 3-1).

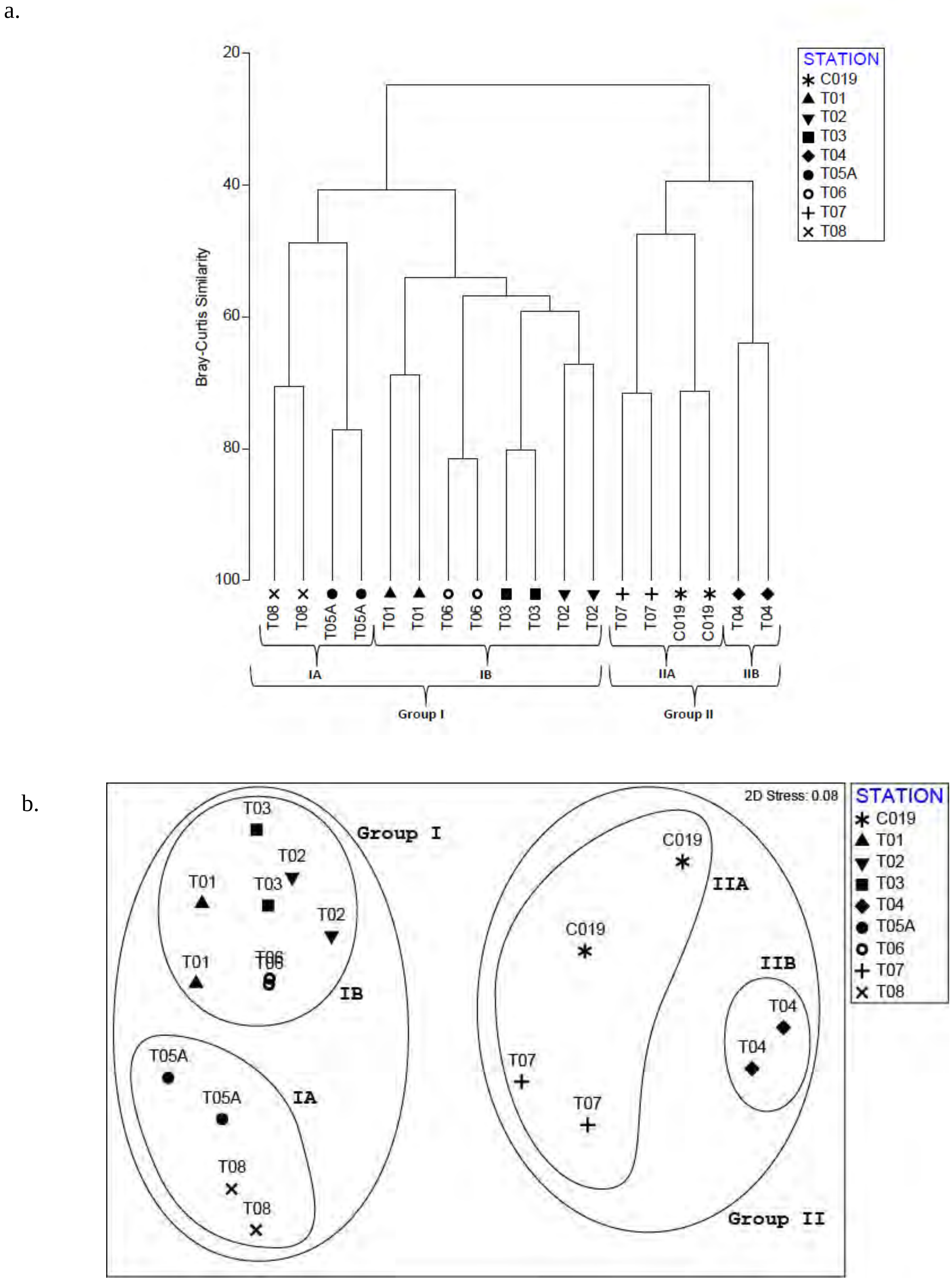

Figure 3-15. Results of (a) cluster analysis and (b) multidimensional scaling analysis of the 2018 infauna samples.

Table 3-5. Mean abundance of dominant taxa in 2018 Boston Harbor station groups defined by cluster analysis.

Major Taxon	Family	Species	I	IA[a]	IB[a]	II	IIA[b]	IIB[b]
Turbellaria		*Platyhelminthes* sp. 16	**-**	-	-	**-**	-	1.0
Bivalvia	Arcticidae	*Arctica islandica*	**1.7**	-	-	**0.8**	-	-
	Pandoridae	*Pandora gouldiana*	**0.8**	-	-	**0.8**	-	-
	Tellinidae	*Ameritella agilis*	**61.8**	124.8	30.4	**1.7**	1.0	3.0
Polychaeta	Ampharetidae	*Ampharete oculata*	**1.7**	-	-	**0.5**	-	-
	Capitellidae	*Capitella capitata complex*	**-**	9.0	10.9	**-**	0.3	1.0
		Mediomastus californiensis	**18.9**	-	-	**0.2**	-	-
	Cirratulidae	*Tharyx acutus*	**80.3**	148	46.5	**11.8**	8	19.5
	Cossuridae	*Cossura* sp. 1	**2.7**	-	4.0	**143.0**	214.5	-
	Hesionidae	*Microphthalmus pettiboneae*	**7.0**	2.0	9.5	**9.5**	13.8	1.0
	Lumbrineridae	*Scoletoma hebes*	**182.1**	1.5	272.4	**4.2**	6.0	0.5
	Maldanidae	*Clymenella torquata*	**98.3**	13.3	140.9	**-**	-	-
	Nephtyidae	*Bipalponephtys neotena*	**28.8**	-	43.1	**151.5**	226.5	1.5
		Nephtys incisa	**0.3**	-	-	**1.2**	-	-
	Orbiniidae	*Leitoscoloplos robustus*	**5.3**	-	-	**1.3**	-	-
	Paraonidae	*Aricidea catherinae*	**569.8**	22.3	843.5	**1.7**	2.5	-
	Pholoidae	*Pholoe* spp.	**6.8**	-	-	**0.7**	-	-
	Phyllodocidae	*Phyllodoce mucosa*	**33.6**	1.5	49.6	**-**	-	-
	Polygordiidae	*Polygordius jouinae*	**99.4**	296.3	1.0	**0.2**	0.3	-
	Spionidae	*Dipolydora quadrilobata*	**39.4**	-	59.1	**-**	-	-
		Polydora cornuta	**880.7**	1378.3	631.9	**86.8**	111.5	37.5
		Pygospio elegans	**24.2**	-	-	**-**	-	-
		Spiophanes bombyx	**18.0**	45.5	4.3	**-**	-	-
		Streblospio benedicti	**18.2**	2.8	25.9	**591.8**	139.3	1497.0
	Syllidae	*Exogone hebes*	**19.1**	30.8	13.3	**-**	-	-
	Terebellidae	*Polycirrus phosphoreus*	**4.3**	0.8	6.0	**7.2**	10.8	-
Oligochaeta	Tubificidae	*Limnodriloides medioporus*	**69.5**	23.8	92.4	**5.0**	7.5	-
		Naidinae sp. 1	**22.9**	61.5	3.6	**0.3**	0.5	-
		Tubificoides benedeni	**-**	25.5	-	**-**	0.25	-
		Tubificoides intermedius	**47.7**	65.8	38.6	**17.8**	26.8	-
Mysidacea	Mysidae	*Neomysis americana*	**-**	-	0.125	**-**	0.25	1
Amphipoda	Ampeliscidae	*Ampelisca* spp.	**130.4**	2.5	194.4	**0.2**	0.3	-
	Aoridae	*Grandidierella japonica*	**-**	-	-	**0.7**	-	2.0
	Corophiidae	*Leptocheirus pinguis*	**57.7**	14.8	79.1	**-**	-	-
Isopoda	Idoteidae	*Edotia triloba*	**-**	28.75	0.875	**-**	-	-
Decapoda	Paguridae	*Pagurus longicarpus*	**0.3**	-	-	**1.7**	-	-

[a] distinct subgroup of Group I

[b] distinct subgroup of Group II

3.2.3 Selected Stations

Station T01. Infaunal community conditions at Station T01, located near Deer Island Flats north of President Roads, have typically exemplified conditions at most stations in the Harbor throughout the survey period. In 2018, species richness, Shannon-Weiner diversity, evenness, and log-series alpha were high at T01. All of these community parameters decreased slightly compared to 2017 but remained within the ranges of values recorded in recent years (Rutecki et al. 2019a). Mean abundance in 2018 decreased compared to 2017 but was similar to the value observed in 2016 (Figure 3-16). Species richness, Shannon-Weiner diversity, Pielou’s evenness, and log-series alpha were about average for the period since the diversion (Figures 3-17 through 3-20). In 2018, all of these community parameters except Pielou’s evenness remained above the relatively low values observed in 2013 (Figures 3-16 through 3-20).

Station C019. Station C019 was initially included in the Harbor sampling program as part of the Sediment-Water Exchange (SWEX) study (Gallagher and Keay 1998). When first sampled in 1989, the benthic infauna consisted of only a few taxa and was overwhelmingly dominated by two polychaete species, *Streblospio benedicti* and *Chaetozone anasimus* (previously called *C. setosa*). Mean abundance declined from its 2013 peak in 2014 and 2015, but increased to above average (834) levels in 2017 (1,010) and 2018 (1,165; Figure 3-16). Species richness also peaked in 2013 and decreased in both 2014 and 2015, reached average levels in 2016, and continues to fluctuate with an increase to average levels in 2018 (Figure 3-17). Log series alpha and Shannon-Weiner diversity reached their peak values in 2014, declining from 2015 through 2017. In 2018, log series alpha increased slightly while Shannon-Weiner diversity continued to decline (Figures 3-18 and 3-19). Pielou’s evenness reached its peak value in 2014, declined in 2015 and 2016, increased in 2017, and then declined again in 2018 (Figure 3-20). Despite decreasing values in some recent years, the diversity measures remained among the highest levels observed to date. The polychaete *Bipalponephtys neotena* (formerly called *Nepthys cornuta*) had represented the majority of the infauna found at Station C019 from 2004-2011 (Figure 3-21) but the relative abundance of this species has been substantially lower since 2012 (Pembroke et al. 2013). The community structure has varied since then with *C. anasimus* dominating in 2013 (Pembroke et al. 2014), no numerical dominant in 2014 (Pembroke et al. 2015), the oligochaete *Tubificoides intermedius* dominating in 2015, and *Polydora cornuta* in 2016. *Cossura* sp 1 dominated the community structure in 2017 and 2018. *B. neotena* and *S. benedicti*, respectively, were the next two dominant taxa in the infaunal community in 2018 (Figure 3-21).

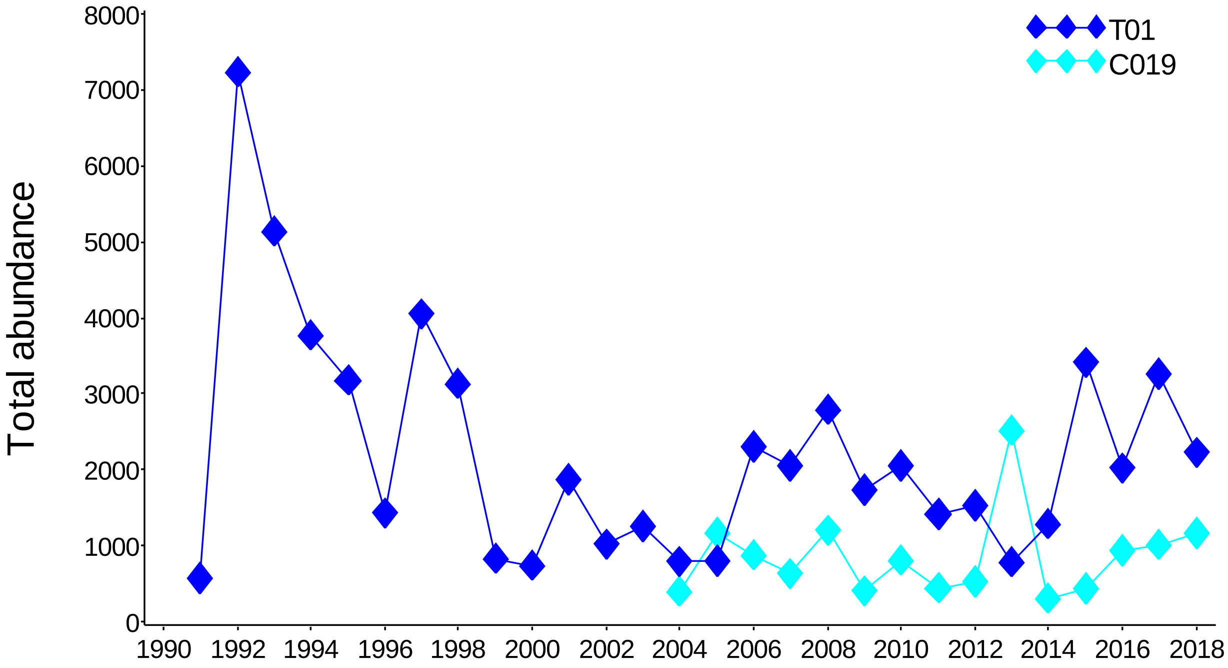

Figure 3-16. Mean total abundance at Boston Harbor Stations T01 and C019, 1991-2018.

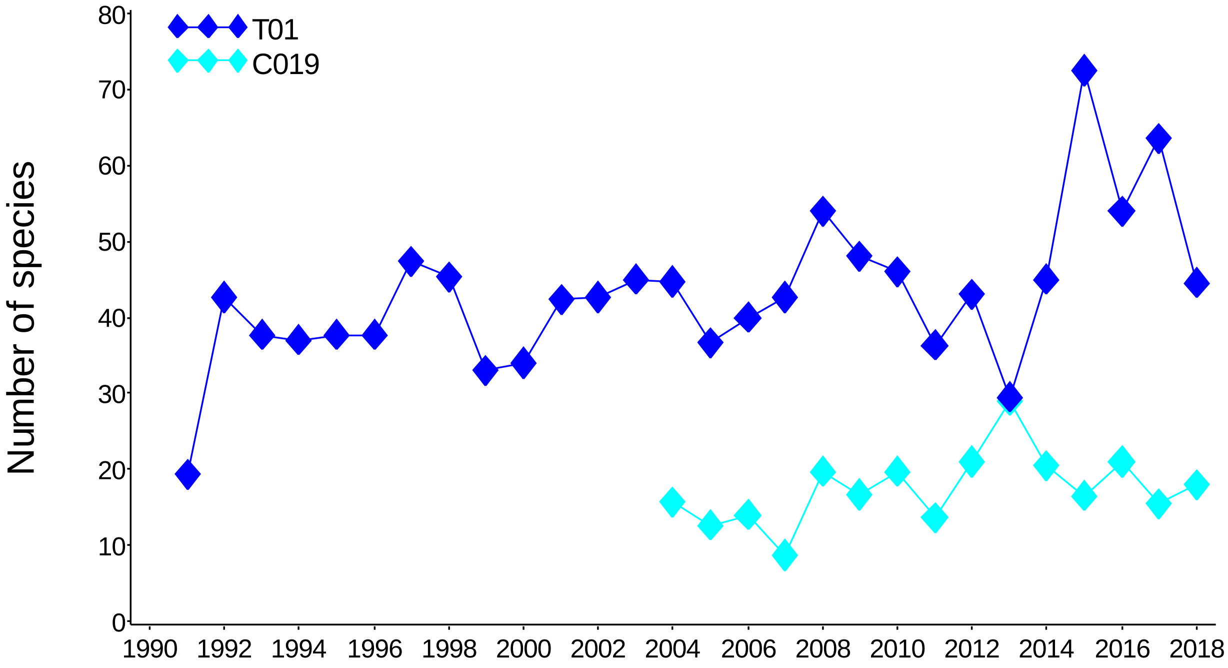

Figure 3-17. Mean species richness at Boston Harbor Stations T01 and C019, 1991-2018.

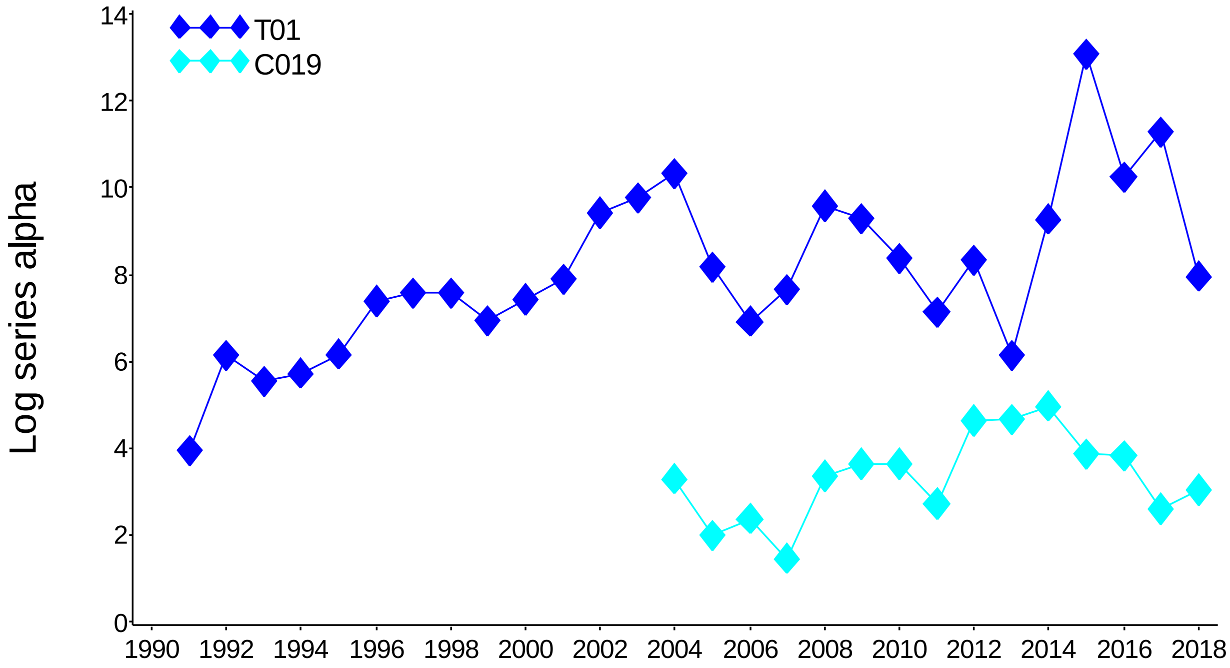

Figure 3-18. Mean log-series alpha diversity at Boston Harbor Stations T01 and C019, 1991-2018.

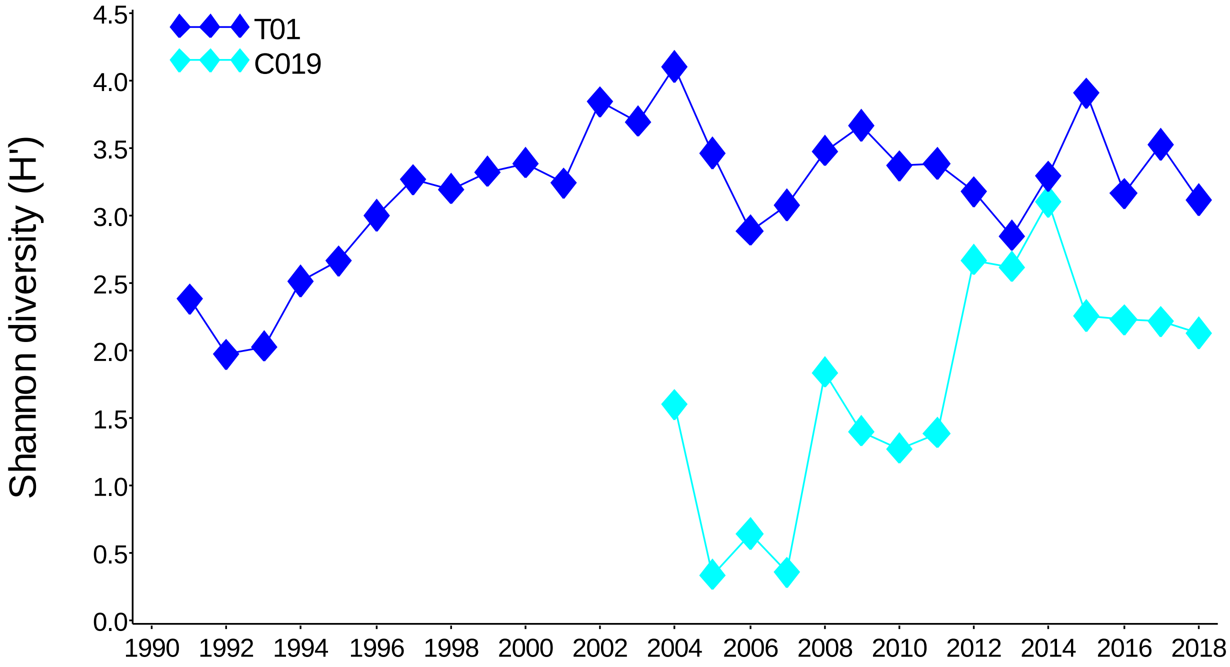

Figure 3-19. Mean Shannon-Weiner diversity at Boston Harbor Stations T01 and C019, 1991-2018.

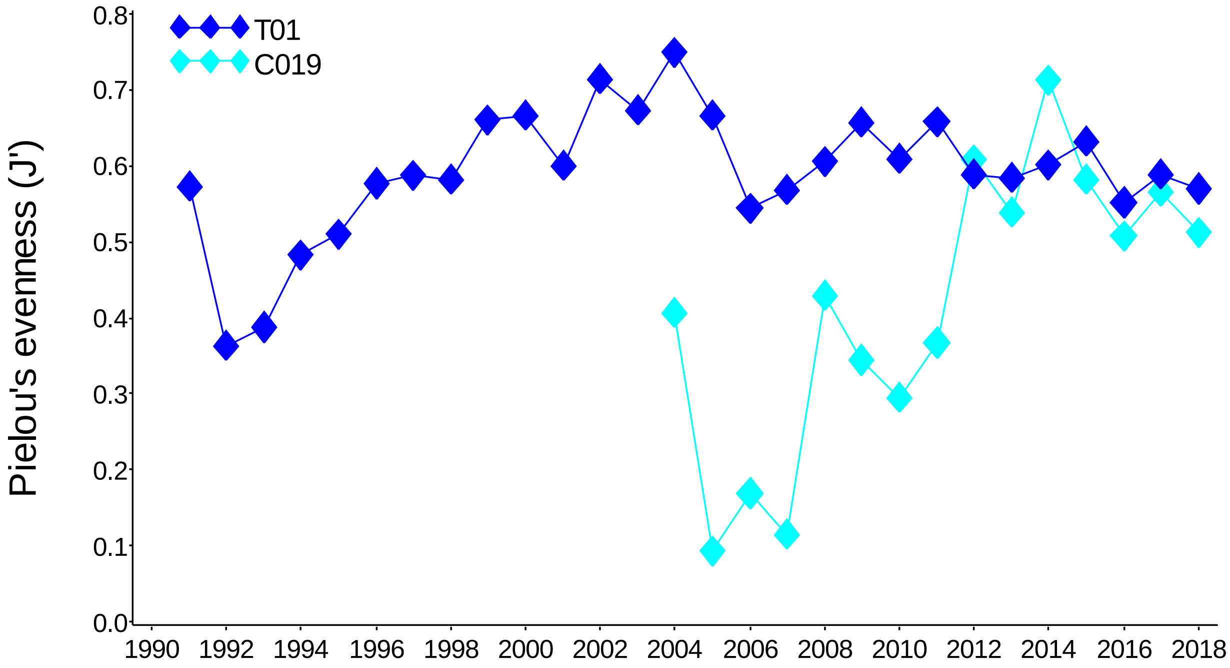

Figure 3-20. Mean evenness at Boston Harbor stations T01 and C019, 1991-2018.

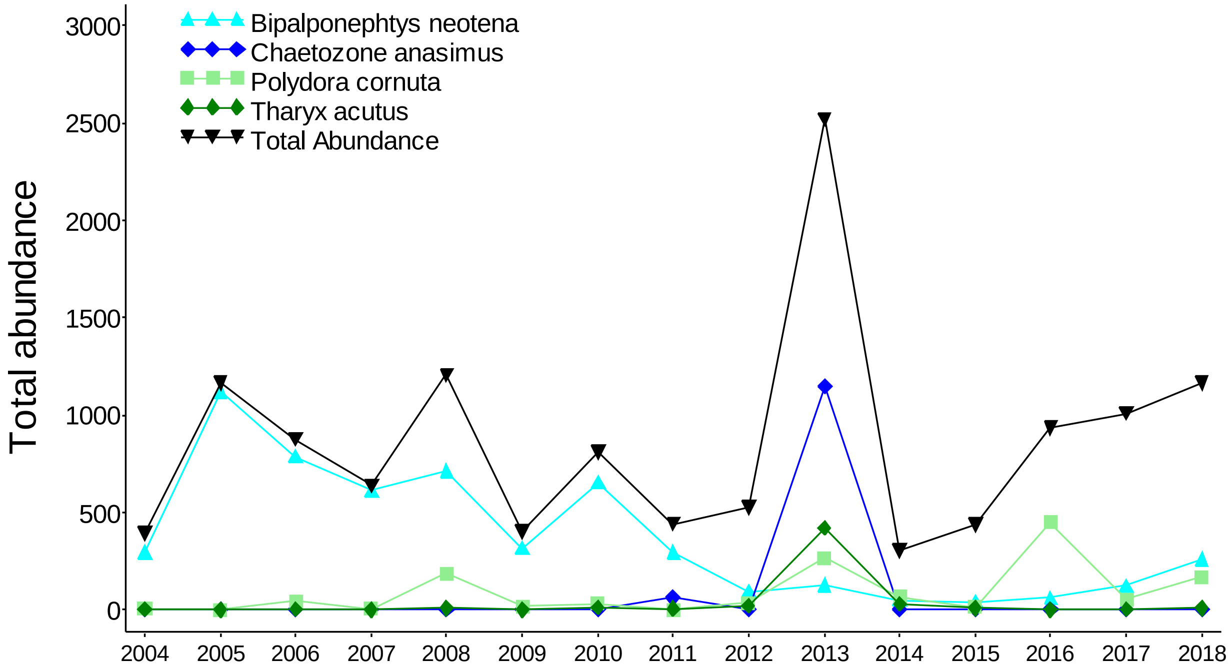

Figure 3-21. Mean abundance of *Bipalponephtys neotena* and other dominants and total community abundance at Station C019, 2004-2018.

3.2.4 Temporal Trends

Benthic community parameters for Boston Harbor have exhibited changes since 1991 that correspond well to the management changes defined by Taylor (2006) when a one-year offset is included to account for a lag in benthic community response (Table 3-6). While mean total abundance has not changed dramatically through these periods, increases in species richness, evenness, and diversity between Periods I and IV have been notable and the trend in all of these parameters suggests a response to the reduction in pollutant loading into the Harbor. Mean Period IV values for total abundance, number of species, evenness, Shannon-Weiner diversity, and log-series alpha diversity for 2002-2018 were virtually the same as for 2002-2017 (Rutecki et al. 2019a) so it is apparent that this trend has continued. It should be noted that the mean total abundance doubled in Period II largely as a result from sharp increases in *Ampelisca* spp. abundance in those years at some stations, especially T03, T06, and T08. For example, abundances of *Ampelisca* spp. at Station T03 were observed at > 6,000 individuals/sample during those years (Maciolek et al. 2008).

Table 3-6. Benthic community parameters for stations T01-T08, summarized by time periods defined by Taylor (2006).

Parameter	Period I (prior to Dec. 1991)		Period II (Dec. 1991-mid 1998)		Period III (mid-1998-Sept. 2000)		Period IV (after Sept. 2000)	
	Mean	Std Err	Mean	Std Err	Mean	Std Err	Mean	Std Err
Total Abundance	2,606	344	5,513	469	3,213	493	2,526	120
Log-series alpha	4.2	0.3	5.5	0.2	6.2	0.3	7.6	0.2
Shannon-Wiener Diversity (H')	2.1	0.1	2.4	0.1	2.8	0.1	2.9	<0.1
Pielou's Evenness (J')	0.48	0.02	0.47	0.01	0.58	0.02	0.55	0.01
Number of Species	25.4	2.1	34.7	1.1	33.6	1.7	41.2	0.9
Years included (with one year offset)	1991-1992		1993-1998		1999-2001		2002-2018	

3.3 SEDIMENT PROFILE IMAGING

Since the start of Sediment Profile Image (SPI) monitoring in 1989 (SAIC 1990, 1992) much of the improvement in benthic habitat quality documented within Boston Harbor can be directly related to upgrades in sewage treatments, relocation of outfalls to the mouth of the Harbor, and finally to operation of the ocean outfall (Taylor 2010, Taylor et al. 2010, Taylor et al. 2019). Key factors in benthic habitat quality improvement were large reductions in organic and nutrient loadings (Gallagher and Keay 1998, Oviatt et al. 2007, Diaz et al. 2008), biogeochemical processes that resulted in more aerobic sediments (Giblin et al. 1997, Tucker et al. 2014), and the underlying hydrodynamics and coastline geometry of the Harbor that continually contributes to quick exchange of water and materials with Massachusetts Bay (Signell and Butman 1992). Superimposed on these factors are tidal surge and storms (Butman et al. 2008, Talke et al. 2018) that also have the potential to affect the quality of benthic habitat within the Harbor. In the nearfield area around the MWRA outfall, storm driven bottom wave-orbital current and bottom-wave stress affected habitat quality by coarsening sediment grain size and increasing the dominance of physical processes (Rutecki et al. 2019b). To identify the extent to which SPI results for the Harbor may be related to storminess and associated sediment transport events we conducted a series of exploratory data analyses.

To assess how storm related factors affected benthic habitat, sediments, and benthic communities, within the Harbor we used integrated wind and wave stress calculated for all storms from January 1990 to 2019, based on winds and waves observed at the 44013 buoy in Massachusetts Bay 16 nautical miles east of the Harbor and following the methods of Butman et al. (2008). A storm based on wind stress was defined as a period when winds exceeded about 23 knots (kn), producing a stress of 0.2 Pascals (Pa), for at least 6 hours (h). Integrated wind stress (IWindS; units Pascal hours, Pa h) was the sum of the magnitudes of the hourly wind stress for the duration of the storm. Within the Harbor typical wind speeds of about 10 to 20 kn can drive surface currents of the order of 5-10 centimeters per second (cm s^{-1}). These currents are small relative to average tidal currents that are 20-40 cm s^{-1} over most of the Harbor and up to 80 cm s^{-1} in President and Nantasket Roads (Signell and Butman 1992). During storm events, wind-driven bottom currents and tidal surge would be larger, depending on storm intensity, direction, and timing relative to tidal stage (Butman et al. 2008, Talke et al. 2018).

Based on bottom-wave stress a storm was defined as a period when the bottom stress caused by waves was greater than 0.1[1] Pa for at least 6 h for a depth of 20 meters (m; the maximum depth for Harbor stations). Integrated excess bottom stress caused by waves (IWaveS) was defined as the sum of the magnitude of the bottom stress, minus the threshold of 0.1 Pa, for the duration of the storm. Storm parameters were summarized for the winter year or period prior to the August monitoring (the 8-months from October through May). For example, storm data related to August 1992 monitoring data would be October 1991 to May 1992. Storms can also cause surge within the Harbor that potentially could affect the bottom through altered currents. To assess this, the sum of storm surge for the year prior to August

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[1] A bottom stress of 0.1 Pa is used as the threshold to define a storm on the sea floor. This is the stress required to initiate motion of very-fine-sand (4 Phi) and is taken as an approximate threshold for resuspension of fine-grained sediments (Butman et al. 2008).

monitoring was calculated from skew surge (difference between the maximum measured storm tide and predicted astronomical tide) data of Talke et al. (2018).

3.3.1 Storms

From January 1990 to January 2019 there were a total of 832 IWindS and 943 IWaveS[2] storms. The year 2012 was not included as data were not available for 72% of the year. Most storms (84%) and the strongest storms occurred during the winter period (October through May). Storms that occurred from June through September were smaller and had lower energy (Figure 3-22). From winter periods 1991 to 2018, integrated wind stress increased over the Harbor and nearfield region (Rutecki et al. 2019b). This increase in wind stress was consistent with North Atlantic Ocean trends of increasing wind speed (Young and Ribal 2019). The direction of wind for the largest storms was northeast to northwest, which exposes the northern and central Harbor to longest wind fetch (Figure 3-23). There was also an increase in the total duration of IWindS storms by about 9 hours per year (linear regression of year with sum of IWindS duration in hours, $R^2 = 0.26$, df = 25, p = 0.007). The mean duration per storm was longer but not significantly.

The number of IWindS and 20 m IWaveS storms were correlated (r = 0.61, t = 3.8, df = 25, p = <0.001) but not all storms were classified as both IWindS and IWaveS events. For example, the ninth largest IWindS storm was not associated with an IWaveS storm and the sixth largest IWaveS storm was not associated with an IWindS storm (Table 3-7). Factors that go into generation of IWindS and IWaveS storms are complex and act at broad regional scales (Butman et al. 2008, Rutecki et al. 2019b). In addition, storms also produce storm surges that are an important factor for flooding hazards in Boston Harbor (Talke et al. 2018).

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[2] IWaveS calculated for a depth of 20 m was used for Harbor storm assessments. 30 m depth was used for assessing storms in nearfield (Rutecki et al. 2019b).

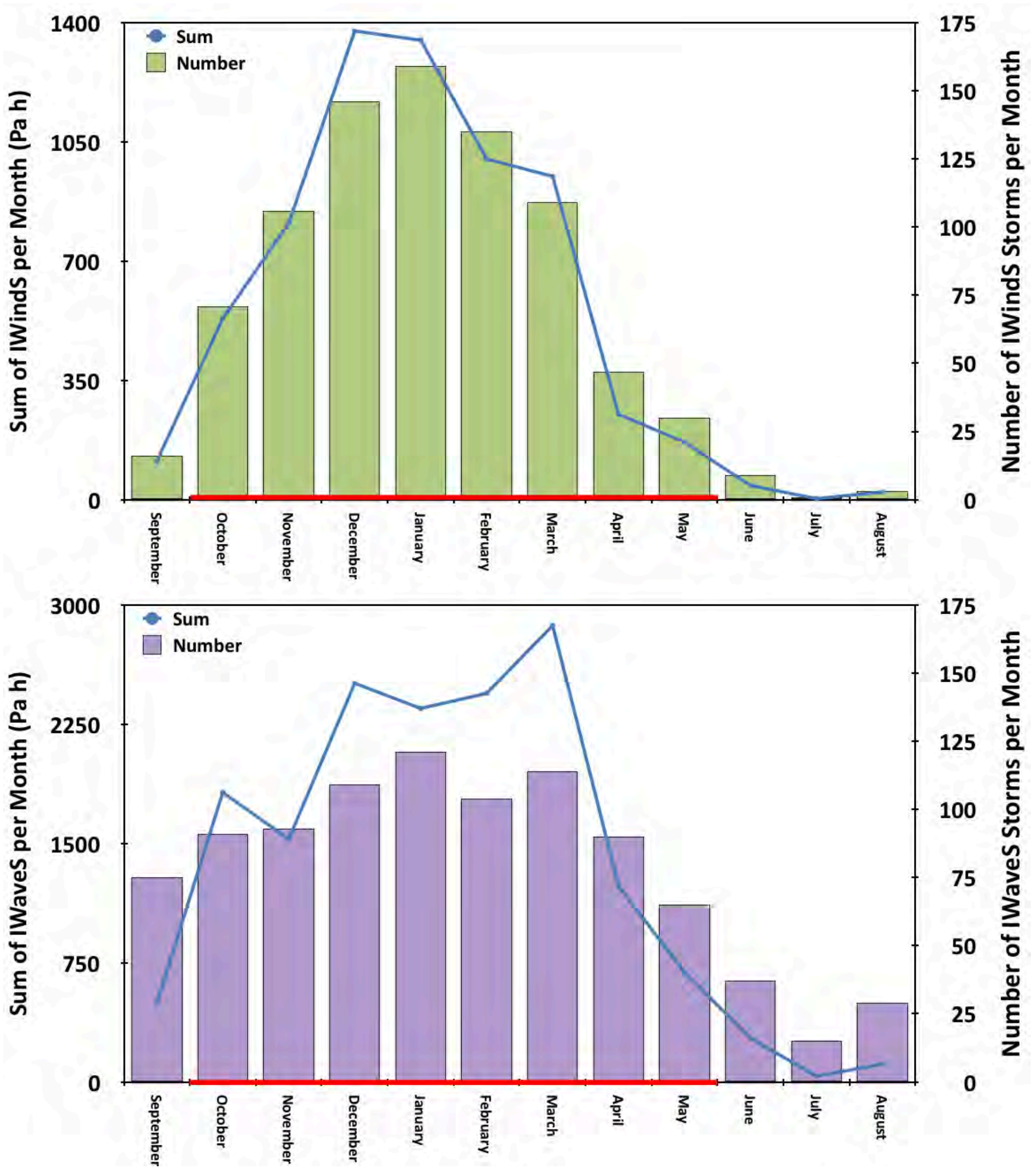

Figure 3-22. Monthly summary of IWindS and 20 m depth IWaveS storms from 1991 to 2018. Storm parameters were summarized by winter period (red line) prior to August monitoring.

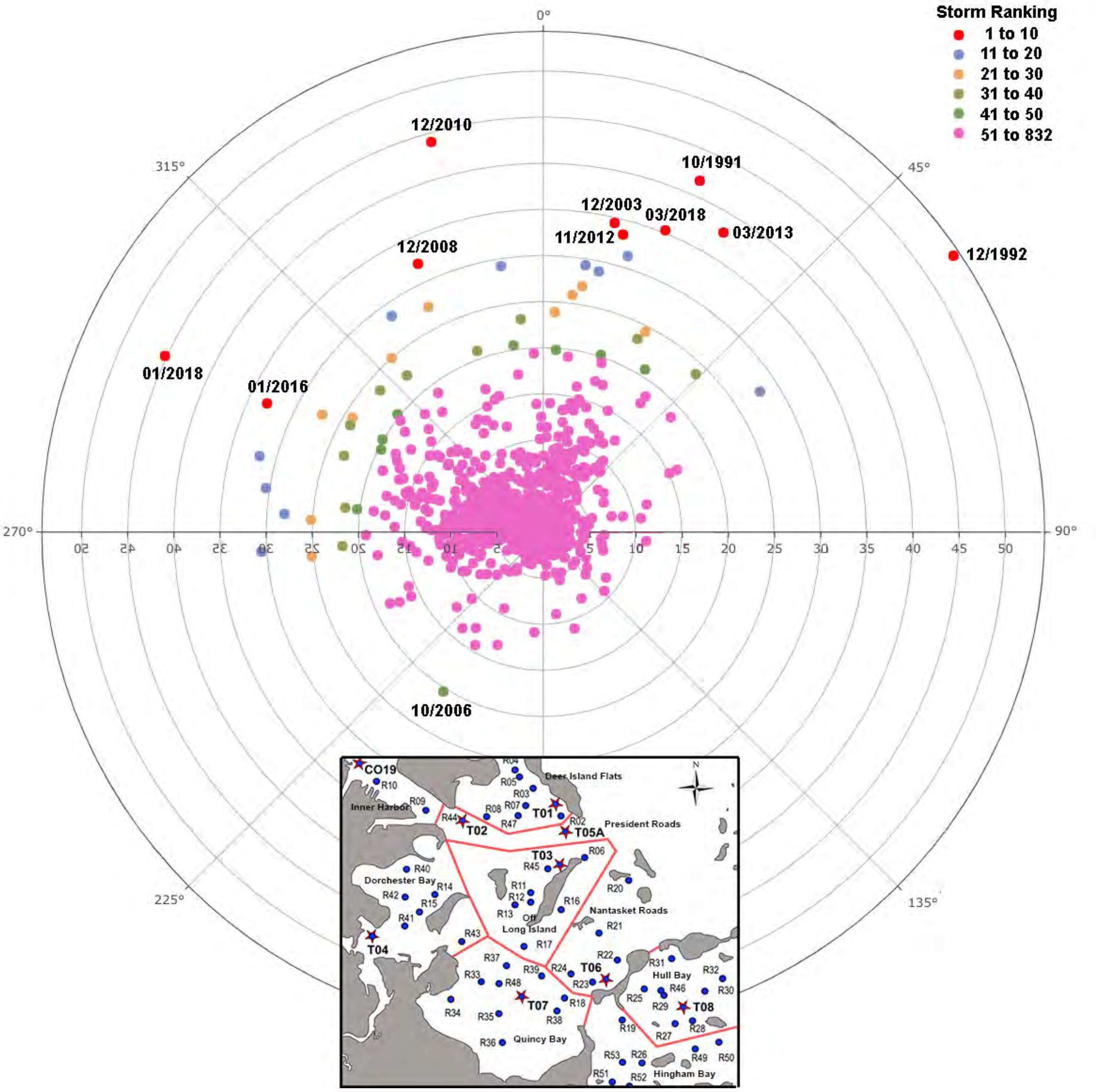

Figure 3-23. Summary of wind direction (from which the wind blows) for IWindS storms from 1991 to 2018. Map insert shows the orientation of the Harbor relative to winds. Data from January to June 2012 were missing.

Table 3-7. Comparison of strength and timing of the top ten IWaveS and IWindS storms for Boston region.

IWaveS Rank	IWaveS Duration (hr)	IWaveS Stress (Pa-hr)	IWindS Rank	IWindS Duration (hr)	IWindS Stress (Pa-hr)	Vector Averaged Wind Direction (Clockwise Deg Rel TN)	Storm Start Date	SPI Sampling Year
1	234	378	6	59	35	22	Feb 28, 2018	2018
2	166	319	1	78	54	56	Dec 11, 1992	1993
3	223	282	5	76	38	31	Mar 5, 2013	2013
4	178	244	7	62	34	13	Dec 6, 2003	2004
5	142	230	4	98	42	24	Oct 28, 1991	1992
6	136	226	None				Mar 5, 2001	2001
7	281	218	396	12	6	97	Feb 24, 2010	2010
8	164	203	171	19	11	95	Apr 15, 2007	2007
9	131	191	21	69	28	57	Mar 13, 2010	2010
10	320	184	12	103	31	17	Nov 4, 2010	2011
24	67	114	8	53	33	15	Nov 7, 2012	2013
43	43	80	3	65	44	344	Dec 26, 2010	2011
72	59	57	10	92	32	335	Dec 19, 2008	2009
127	54	38	2	77	45	295	Jan 4, 2018	2018
None			9	94	33	295	Jan 18, 2016	2016

The six stormiest years since 1990, based on winter-period integrated IWaveS stress, were 2013, 2005, 2010, 1993, 1998, and 2018, respectively (Figure 3-24). The highest skew surge[3] years were 2013, 2018, 2005, 1992, 2011, and 1993 (Figure 3-25). The year 2013 was the stormiest year for three parameters (IWindS, IWaveS, and skew surge). Major storms, defined as being in the top 40, occurred within 6-months of the August monitoring during these stormier years (Figures 3-24 and 3-25). Least stormy years were 2002 and 1991 followed by 1999, 1995, and 2014, all years with a total integrated bottom stress of less than 400 Pa hr, and also low skew surge years. In 1999, 2002 and 2014 at least 18-months passed without a major storm prior to August monitoring. In 1995, a major storm occurred 6-months before August (Figure 3-24). The ranking of storminess by year was different for IWindS and IWaveS (See Table 3-7) as each is a measure of different integrated stress (Butman et al. 2008). Top stormy years tended to be high for both integrated stress measures and lowest storm years tended to be low for both measures (Figure 3-26).

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[3] Skew surge is defined to be the difference between the maximum measured storm tide and the predicted astronomical tide (Talke et al. 2018).

It is worth noting that the winter-period of 1991-1992 included the October 1991 “perfect storm” (fifth top IWaveS, fourth top IWindS, and highest surge storm) that was ranked 20th overall. Even though there have been four stronger storms since 1991, the timing of the perfect storm was important. It occurred just before the December 1991 cessation of digested sewage sludge dumping that marked the end of highest organic loading (Taylor 2010). Both events played an important role in the early transition of the Harbor from the depauperate communities found in September 1991 to the rapidly changing fauna observed from 1992 on (Gallagher and Keay 1998).

3.3.2 Sediments

SPI monitoring stations were located in soft-sediment areas throughout Boston Harbor that would likely be depositional or at least low energy bottoms with a higher likelihood of responding to effects related to wastewater discharge. Over the entire Harbor about 50% of the bottom has heterogeneous sediments that are long-term depositional, about 30% has reworked sediments containing patches of fine-grained sediments, and about 20% has erosional–nondepositional with pebble-sized grains and larger (Knebel 1993, Knebel and Circé 1995). Geomorphology and hydrodynamics are the primary factors responsible for this pattern of sediment distribution (Signell and Butman 1992) with small-scale spatial variation and storm events responsible for much of the changes in sediment grain-size through time. Storms could affect sediment grain-size through resuspension events that exceed the critical erosion threshold, which represents stability/erodability of the bed (Thompson et al. 2017) that in turn would affect benthic habitat stability. The threshold for defining IWaveS storms was a critical stress of 0.1 Pa, which would resuspend very-fine-sand (4 Phi) and finer sediments. Fine-sand (3 Phi) would be resuspended by 0.14 Pa (Butman et al. 2008).

Sediments at most stations were silt-clays to sandy-silts and tended to be homogeneous through time (Figure 3-27). For example, Stations R10 and T04 consistently had silt-clay sediment for 26 of 27 years (Figures 3-28 and 3-29). Station CO19, which was sampled annually starting in 2004, was the only station that had the same estimated modal sediment through time. All other stations were more heterogeneous (R13 or R16 for example, Figures 3-30 and 3-31) and coarsened and/or fined, several times between 1992 and 2018 (Figure 3-27). Overall the odds of a station not changing modal grain-size from one year to the next (versus changing) was 2.7:1 (1052 remaining the same and 392 changing out of 1,444 station-years). Of the 392 station-year combinations that did change, 188 coarsened in estimated modal grain-size and 204 became finer. For example, Station R13 off the southwest side of Long Island went from a modal grain-size of fine-sand-silt in 1997 then coarsened to fine-sand in 1998 (the fifth stormiest year) and fined to silt-clay in 1999 (the third least stormy year). Sediments at Station R13 also coarsened from fine-sand-silt in 2009 (the sixth least stormy year) to fine-sand-pebble in 2010 (the third stormiest year) and fine-medium-sand in 2011 through 2014 (Figure 3-30).

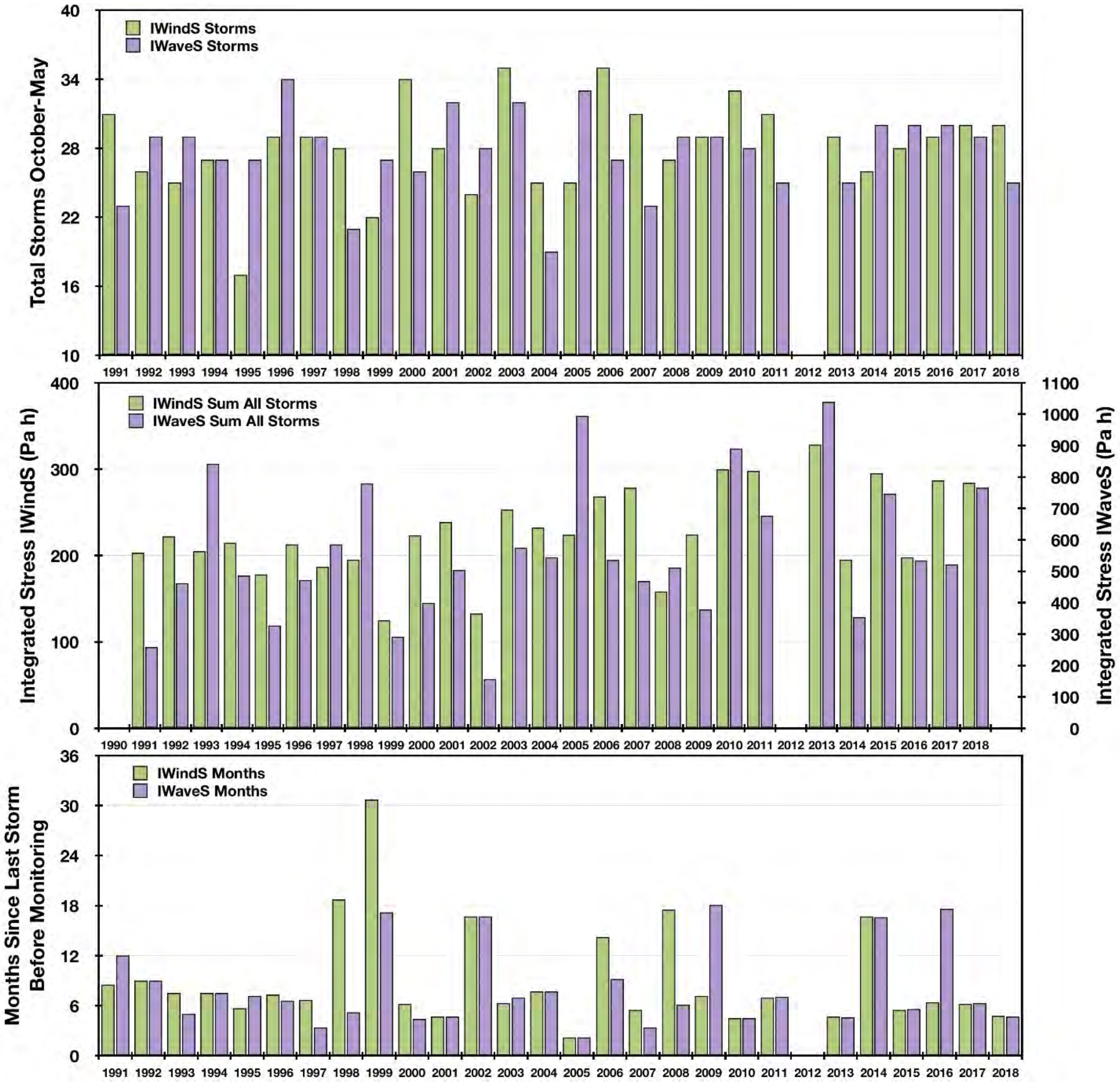


Figure 3-24. Sum and number of IWindS and 20 m depth IWaveS storms for the winter period (October through May) prior to August monitoring. The year 2012 was not included due to missing data.

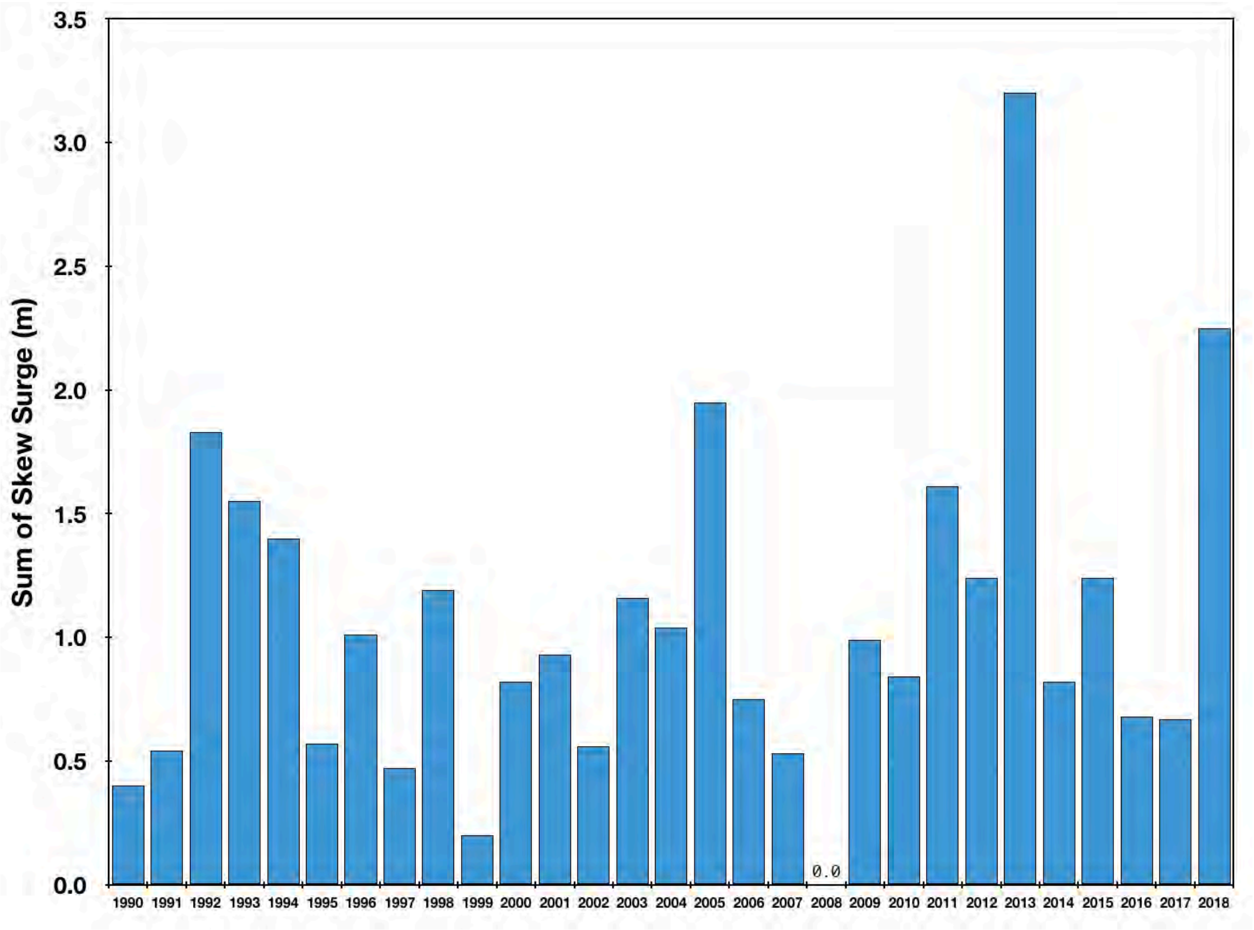


Figure 3-25. Sum of storm skew surge for the year prior to August monitoring. Data from Talke et al. (2018).

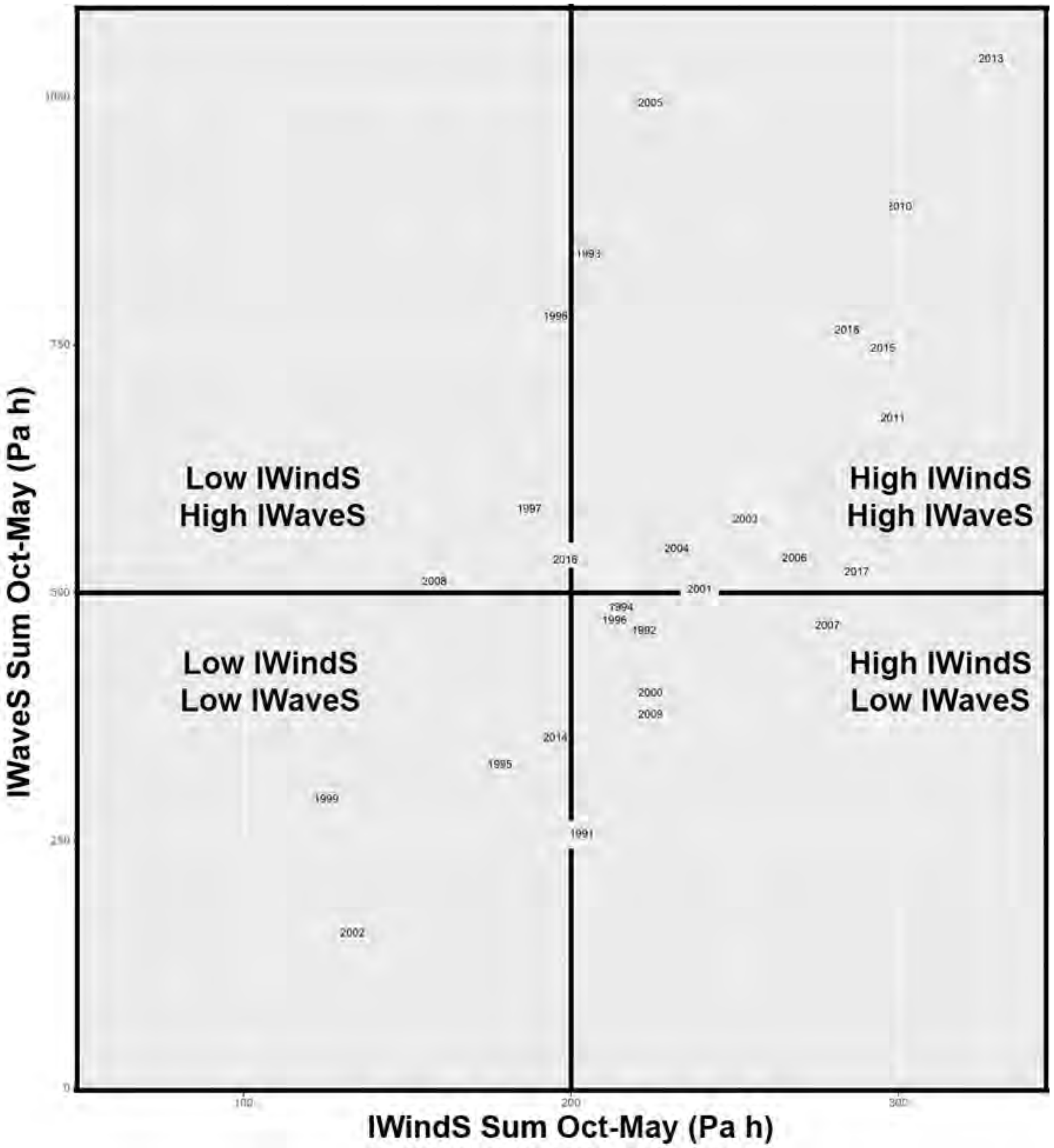
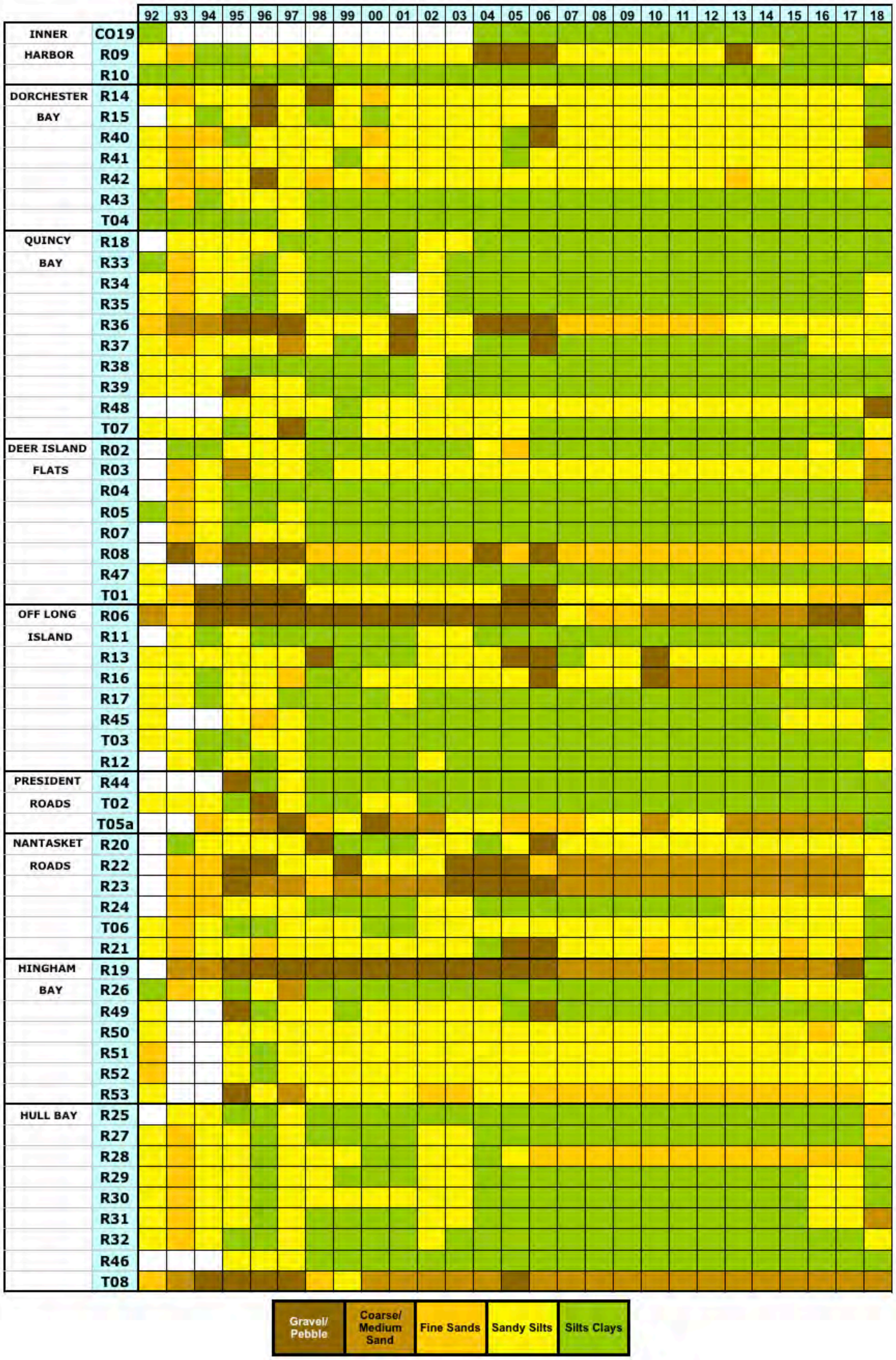


Figure 3-26. Relationship between sum of IWindS and 20 m depth IWaveS storms for the winter period (October through May) prior to August monitoring. The year 2012 was not included due missing data.



		92	93	94	95	96	97	98	99	00	01	02	03	04	05	06	07	08	09	10	11	12	13	14	15	16	17	18
INNER HARBOR	CO19	🟩												🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	R09	🟨	🟧	🟩	🟩	🟨	🟨	🟩	🟨	🟨	🟨	🟨	🟨	🟫	🟫	🟫	🟨	🟨	🟨	🟨	🟨	🟨	🟫	🟨	🟩	🟩	🟩	🟩
	R10	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨
DORCHESTER BAY	R14	🟨	🟧	🟨	🟨	🟫	🟨	🟫	🟨	🟧	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟩
	R15		🟨	🟩	🟨	🟫	🟨	🟩	🟨	🟩	🟨	🟨	🟨	🟨	🟨	🟫	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟩
	R40	🟨	🟧	🟧	🟩	🟨	🟨	🟨	🟨	🟧	🟨	🟨	🟨	🟨	🟩	🟫	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟫
	R41	🟨	🟧	🟨	🟨	🟨	🟨	🟨	🟩	🟨	🟨	🟨	🟨	🟨	🟩	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟩
	R42	🟨	🟧	🟧	🟨	🟫	🟨	🟧	🟨	🟧	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟧	🟨	🟨	🟨	🟨	🟧
	R43	🟩	🟧	🟩	🟨	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	T04	🟩	🟩	🟩	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
QUINCY BAY	R18		🟨	🟨	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	R33	🟩	🟧	🟨	🟨	🟩	🟨	🟩	🟩	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	R34	🟨	🟧	🟨	🟨	🟩	🟨	🟩	🟩	🟩		🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨
	R35	🟨	🟧	🟨	🟩	🟩	🟨	🟩	🟩	🟩		🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨
	R36	🟧	🟤	🟤	🟫	🟫	🟫	🟨	🟨	🟨	🟫	🟨	🟨	🟫	🟫	🟫	🟧	🟧	🟧	🟧	🟧	🟧	🟨	🟨	🟨	🟨	🟨	🟨
	R37	🟨	🟧	🟨	🟨	🟨	🟤	🟨	🟩	🟨	🟫	🟨	🟨	🟩	🟩	🟫	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟨	🟨
	R38	🟨	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	R39	🟨	🟨	🟨	🟫	🟨	🟨	🟩	🟩	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	R48				🟨	🟨	🟨	🟨	🟩	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟫
	T07	🟨	🟨	🟨	🟩	🟨	🟫	🟩	🟩	🟨	🟨	🟨	🟨	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨
DEER ISLAND FLATS	R02		🟩	🟩	🟨	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟧	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟩	🟧
	R03		🟧	🟨	🟤	🟨	🟨	🟩	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟤
	R04		🟧	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟤
	R05	🟩	🟧	🟨	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨
	R07		🟧	🟨	🟩	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	R08		🟫	🟧	🟫	🟫	🟫	🟧	🟧	🟧	🟧	🟧	🟧	🟫	🟧	🟫	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟨
	R47	🟨			🟩	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	T01	🟨	🟧	🟫	🟫	🟫	🟫	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟫	🟫	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟧	🟧	🟧
OFF LONG ISLAND	R06	🟤	🟧	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟨	🟧	🟧	🟤	🟤	🟤	🟤	🟤	🟤	🟫	🟫	🟨
	R11		🟨	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨
	R13	🟨	🟨	🟨	🟨	🟨	🟨	🟫	🟩	🟩	🟩	🟨	🟨	🟨	🟫	🟫	🟩	🟨	🟨	🟫	🟨	🟨	🟨	🟨	🟩	🟩	🟨	🟨
	R16	🟨	🟨	🟩	🟨	🟨	🟧	🟩	🟩	🟨	🟨	🟨	🟨	🟨	🟨	🟫	🟨	🟨	🟨	🟫	🟤	🟤	🟤	🟤	🟨	🟨	🟨	🟩
	R17	🟨	🟨	🟩	🟨	🟨	🟩	🟩	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	R45	🟨			🟨	🟧	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟨	🟨	🟩
	T03	🟨	🟨	🟩	🟩	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	R12		🟨	🟩	🟨	🟩	🟨	🟩	🟩	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨
PRESIDENT ROADS	R44				🟫	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	T02	🟨	🟨	🟨	🟩	🟫	🟨	🟩	🟩	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	T05a			🟧	🟨	🟤	🟫	🟧	🟨	🟫	🟤	🟤	🟨	🟨	🟧	🟧	🟧	🟨	🟨	🟤	🟨	🟨	🟤	🟤	🟤	🟤	🟤	🟩
NANTASKET ROADS	R20		🟩	🟨	🟨	🟨	🟨	🟫	🟩	🟩	🟩	🟨	🟨	🟩	🟨	🟫	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨
	R22		🟧	🟧	🟫	🟫	🟨	🟨	🟫	🟨	🟨	🟨	🟫	🟫	🟫	🟧	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟨
	R23		🟧	🟧	🟫	🟤	🟤	🟧	🟤	🟤	🟤	🟤	🟫	🟫	🟫	🟫	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟨
	R24		🟧	🟧	🟨	🟨	🟨	🟩	🟩	🟩	🟩	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟨	🟨	🟨	🟨	🟩
	T06	🟨	🟧	🟨	🟩	🟩	🟨	🟨	🟨	🟩	🟩	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟩
	R21	🟨	🟧	🟨	🟨	🟧	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟩	🟫	🟫	🟨	🟨	🟨	🟧	🟨	🟨	🟨	🟨	🟧	🟨	🟧	🟩
HINGHAM BAY	R19		🟤	🟤	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟫	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟫	🟩
	R26	🟩	🟧	🟨	🟩	🟨	🟤	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟨	🟨	🟩
	R49	🟨			🟫	🟩	🟨	🟨	🟩	🟨	🟨	🟨	🟨	🟨	🟩	🟫	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨
	R50	🟨			🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟧	🟨	🟩
	R51	🟧			🟨	🟩	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨
	R52	🟧			🟨	🟩	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟨
	R53	🟨			🟫	🟨	🟤	🟨	🟨	🟨	🟨	🟧	🟧	🟨	🟨	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟧	🟨
HULL BAY	R25		🟨	🟨	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟧
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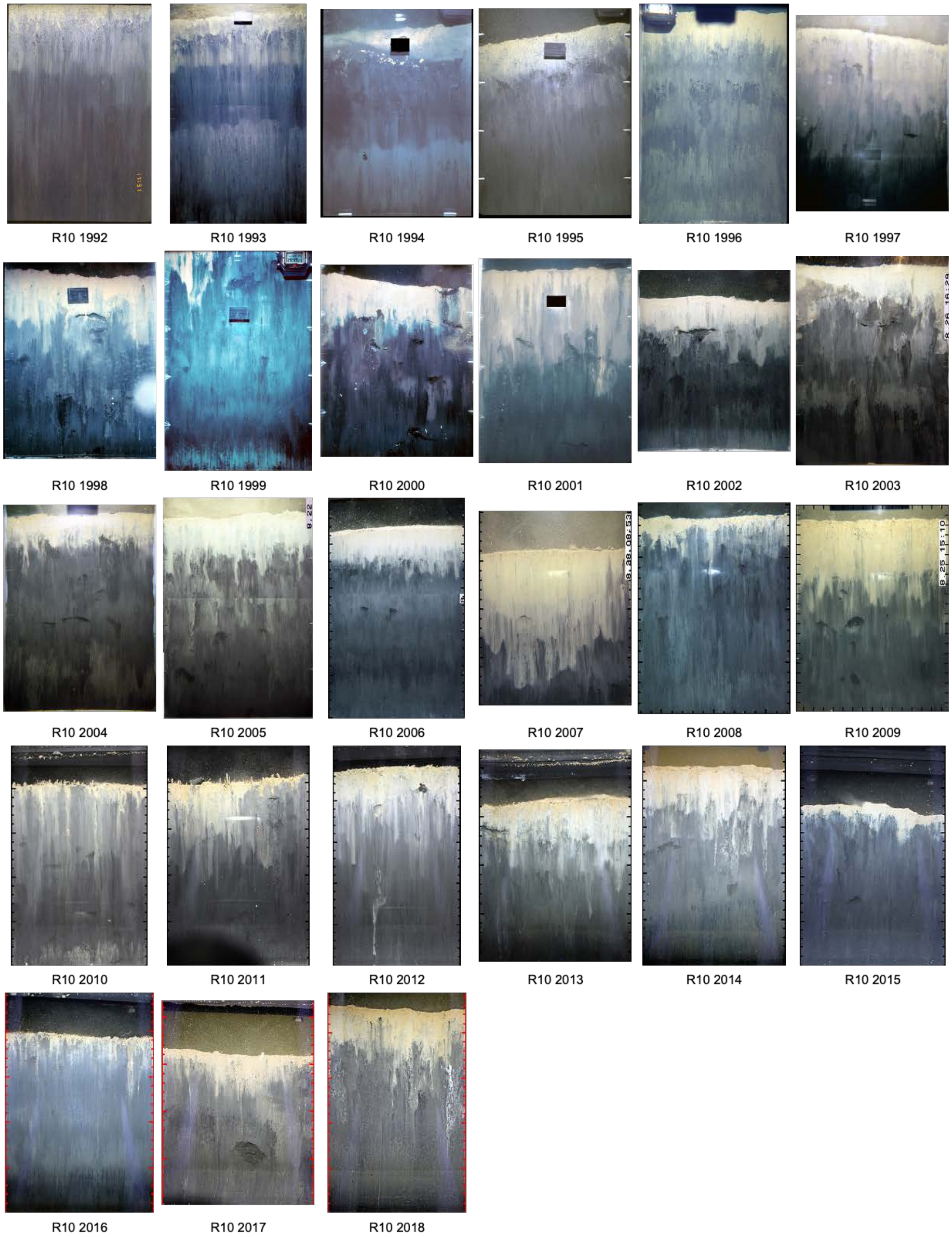
	R29	🟨	🟧	🟨	🟨	🟩	🟨	🟨	🟩	🟩	🟩	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟨	🟩
	R30	🟨	🟧	🟨	🟨	🟩	🟨	🟨	🟨	🟨	🟨	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟨	🟩
	R31	🟨	🟧	🟨	🟨	🟩	🟨	🟩	🟩	🟩	🟩	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨	🟨	🟤
	R32	🟨	🟧	🟨	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟨
	R46				🟨	🟨	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
	T08	🟧	🟤	🟫	🟫	🟫	🟫	🟧	🟨	🟤	🟤	🟤	🟤	🟤	🟫	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤	🟤

🟫	🟤	🟧	🟨	🟩
Gravel/ Pebble	Coarse/ Medium Sand	Fine Sands	Sandy Silts	Silts Clays

Figure 3-27. Matrix of modal grain-size from SPI with stations arranged by Harbor region (see Figure 3-42 for region boundaries). Blank cells indicate station was not sampled.



R10 1992
R10 1993
R10 1994
R10 1995
R10 1996
R10 1997
R10 1998
R10 1999
R10 2000
R10 2001
R10 2002
R10 2003
R10 2004
R10 2005
R10 2006
R10 2007
R10 2008
R10 2009
R10 2010
R10 2011
R10 2012
R10 2013
R10 2014
R10 2015
R10 2016
R10 2017
R10 2018

Figure 3-28. Mosaic of images from Station R10 in the Inner Harbor with homogeneous sediments through time. Scale on side of images is in cm.

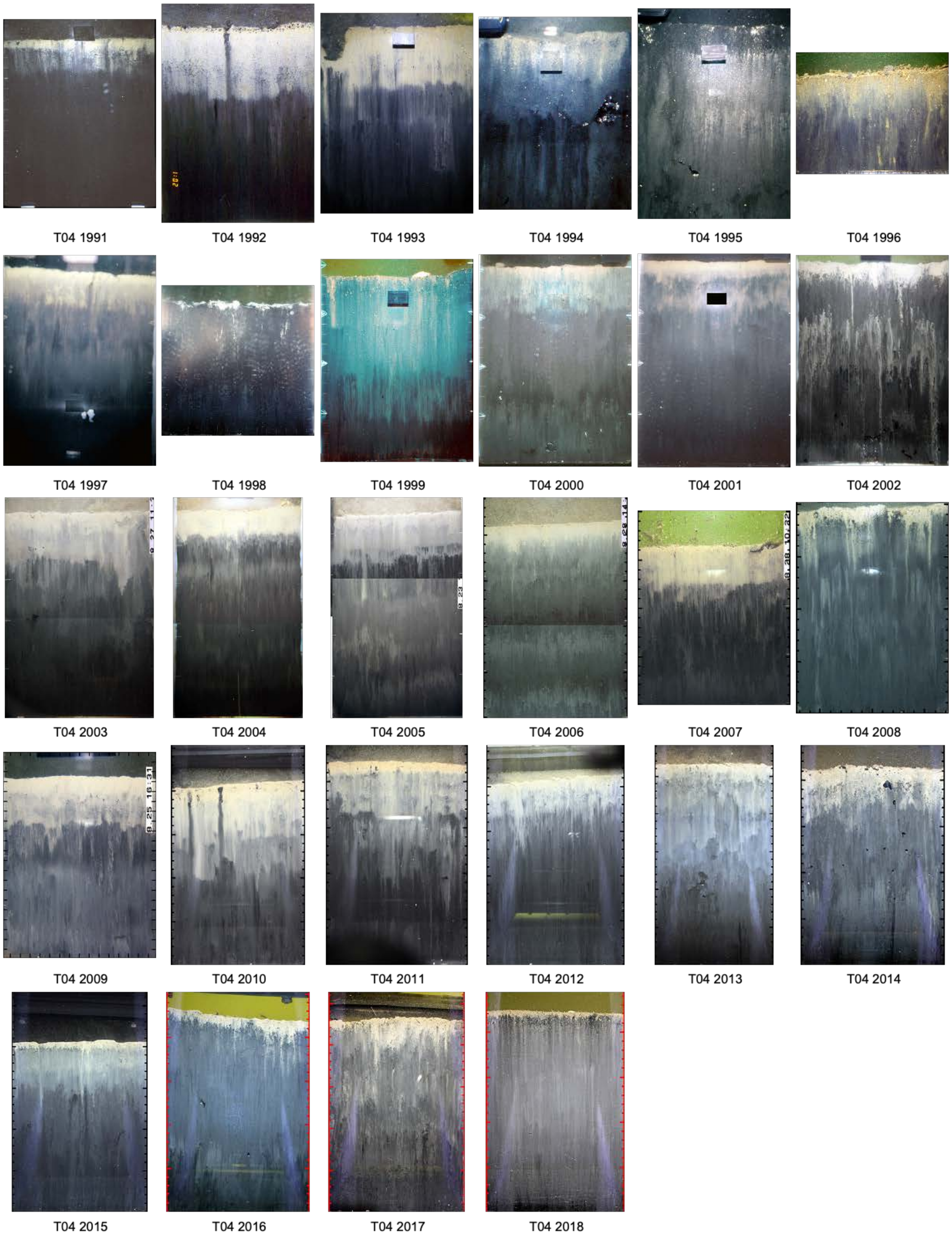
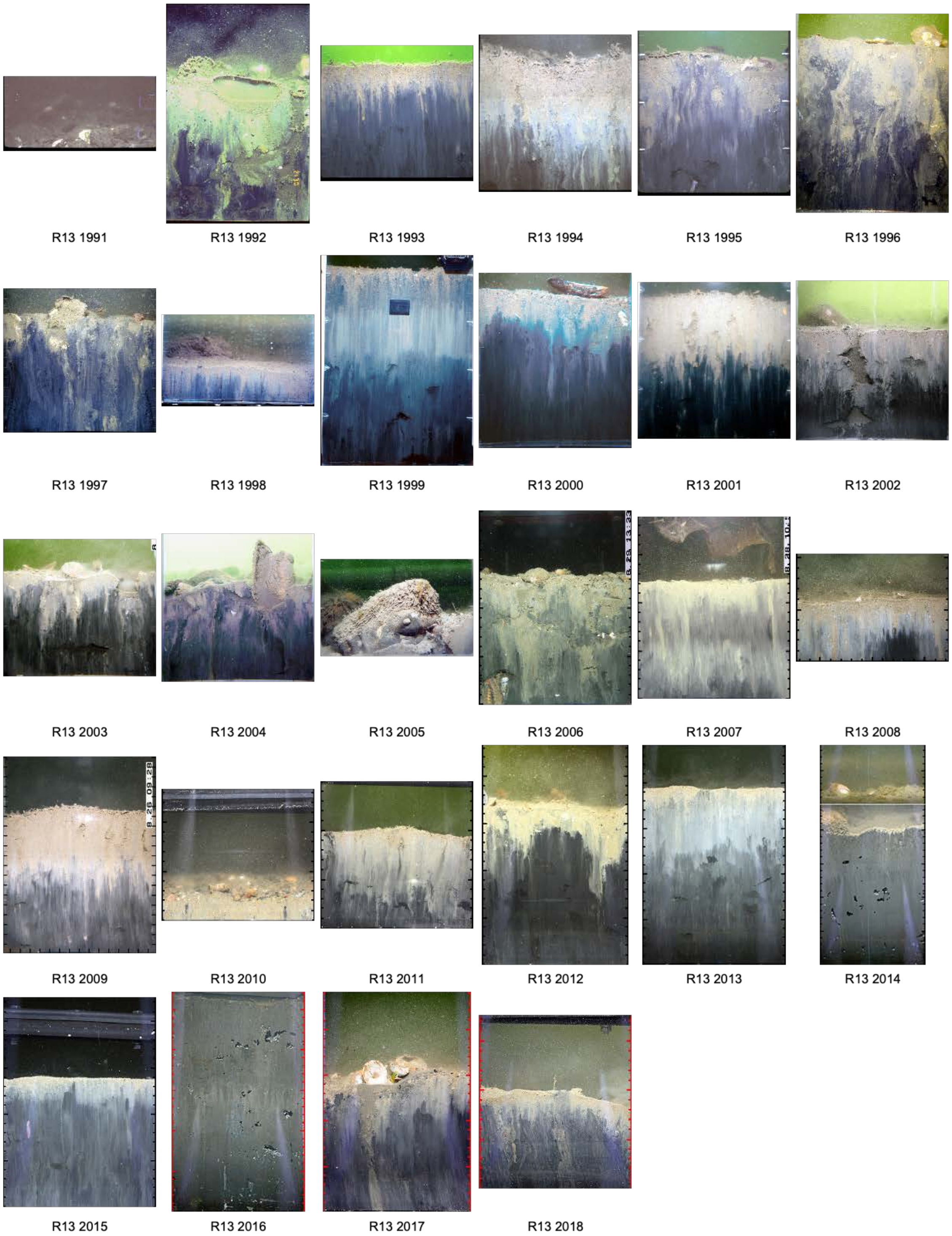


Figure 3-29. Mosaic of images from Station T04 in Dorchester Bay with homogeneous sediments through time. Scale on side of images is in cm.



R13 1991 R13 1992 R13 1993 R13 1994 R13 1995 R13 1996

R13 1997 R13 1998 R13 1999 R13 2000 R13 2001 R13 2002

R13 2003 R13 2004 R13 2005 R13 2006 R13 2007 R13 2008

R13 2009 R13 2010 R13 2011 R13 2012 R13 2013 R13 2014

R13 2015 R13 2016 R13 2017 R13 2018

Figure 3-30. Mosaic of images from Station R13 off the western side of Long Island with heterogeneous sediments through time. Scale on side of images is in cm.

R16 1992
R16 1993
R16 1994
R16 1995
R16 1996
R16 1997

R16 1998
R16 1999
R16 2000
R16 2001
R16 2002
R16 2003

R16 2004
R16 2005
R16 2006
R16 2007
R16 2008
R16 2009

R16 2010
R16 2011
R16 2012
R16 2013
R16 2014
R16 2015

R16 2016
R16 2017
R16 2018

Figure 3-31. Mosaic of images from Station R16 off the eastern side of Long Island with heterogeneous sediments through time. Scale on side of images is in cm.

When estimated modal grain-size at a station did change from year to year, it appeared to be related to storm parameters and time (logistic model based on the odds of stations that changed getting either coarser vs. finer, Table 3-8). Years with higher total winter period IWindS storm energy favored a change from coarser to finer grain-size. The effect was reversed for IWaveS storm energy, which favored change from finer to coarser. Storm skew surge was not a factor in changing sediment grain-size. Through time there was a marginally significant trend for an increase in the chance of coarsening. When estimated modal grain-size did changed, for every 10 Pa h increase in IWindS the odds of the change being finer increased by a factor of 1.08. For a 10 Pa h increase in IWaveS the odds of the change being coarser increased by a factor of 1.02. The differing response to IWindS and IWaveS may be related to the position of stations within the Harbor and exposure to wave and storm currents.

If the modal grain-size at a station changed from year to year, it tended to become coarser as opposed to finer (positive slope for IWaveS and Year, Figure 3-32). In 2005, the second stormiest year, modal grain-size at 8 of 12 stations that changed coarsened (Figure 3-27). The effect of 2005 storms may have carried into 2006 when modal grain-size at more stations coarsened than fined (10 of 14). By 2007 most stations had become finer and returned to pre-2005 modal grain-size sediments. In 2009 and 2012 none of the 61 stations changed modal grain-size. With the third biggest storm year in 2010, modal grain-size at all five stations that changed coarsened and in 2011 they returned to finer grain-size. In 2013, the top stormiest year, four out of five stations coarsened and the following year two returned to being finer. In 1993, the fourth stormiest year, modal grain-size at all 24 stations that changed coarsened. The biggest year for modal sediment change came in 2018, the sixth stormiest year, when over half of all stations changed with 18 becoming coarser and 20 finer (Figure 3-27). The high number of stations changing in 2018 could be related to transport of finer sediments from one area to another.

Median grain-size estimated from benthic grabs at the eight traditional stations (T-Stations) follow a similar pattern of change[4] as modal grain-size from SPI only for time (Table 3-9). Median grain-size tended to become coarser through time (Figure 3-33) but there was no relationship with storm parameters based on a linear regression model. The lack of relationship between median grain-size and storms could be related to the small sample size of eight T-Stations or missing the effect of the fourth stormiest year in 1993 (grain-size analysis for Phi intervals started in 1994). Analysis of modal grain-size from SPI for only T-Stations produced similar results.

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[4] Median grain-size was estimated graphically using cumulative percentage weights of Phi intervals. Median grain-size was considered to change when there was at least a 0.4 Phi unit change from one year to the next.

Table 3-8. Logistic model for the odds of a station that changed modal grain-size from year to year becoming coarser (positive parameter), as opposed to becoming finer (negative parameter) relative to storm parameters and time.

Factor	Parameter	SE	z-value	p
IWaveS per 10 Pa h	0.015	0.001	2.02	0.043
IWindS per 10 Pa h	-0.080	0.003	-2.36	0.018
Skew Surge per 1 m	-0.199	0.256	-0.78	0.437
Per Year	0.035	0.018	1.96	0.050

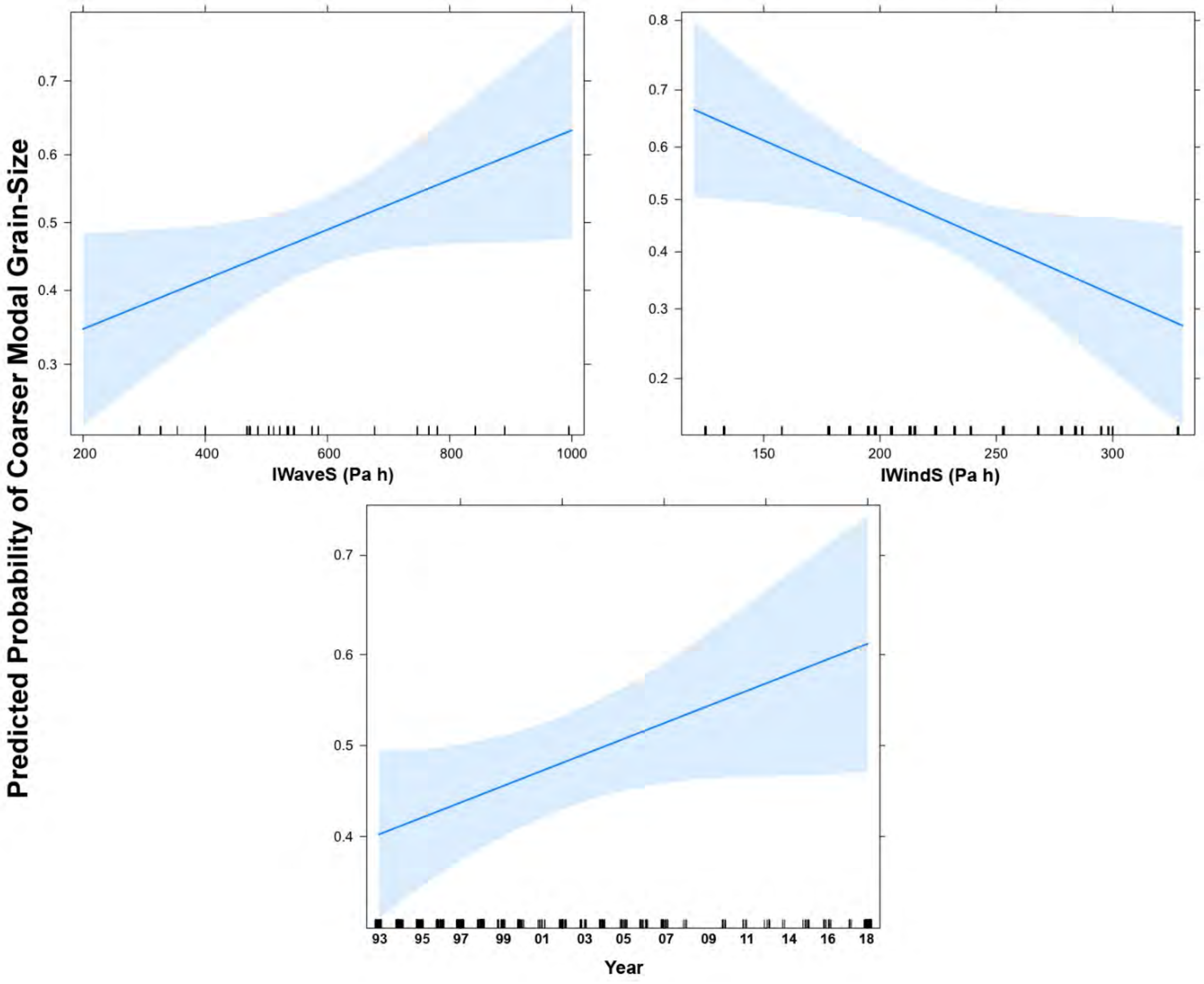

Figure 3-32. Predicted probability that if a station changed modal grain-size it would be in the coarser sediment direction. Based on logistic regression of storm parameters and grain-size change (finer vs. coarser). Parameter estimates are in Table 3-8. Shaded area is 95% confidence interval.

Table 3-9. General linear regression model of median grain-size at T-Stations with storms parameters and time. Negative parameter indicates coarsening of median Phi grain-size.

Factor	Parameter	SE	t-value	p
IWaveS per 10 Pa h	-0.00001	0.0005	-0.00	0.999
IWindS per 10 Pa h	0.00497	0.0018	0.27	0.787
Skew Surge per 1 m	0.08254	0.1436	0.58	0.566
Per Year	-0.04470	0.0119	-3.75	<0.001

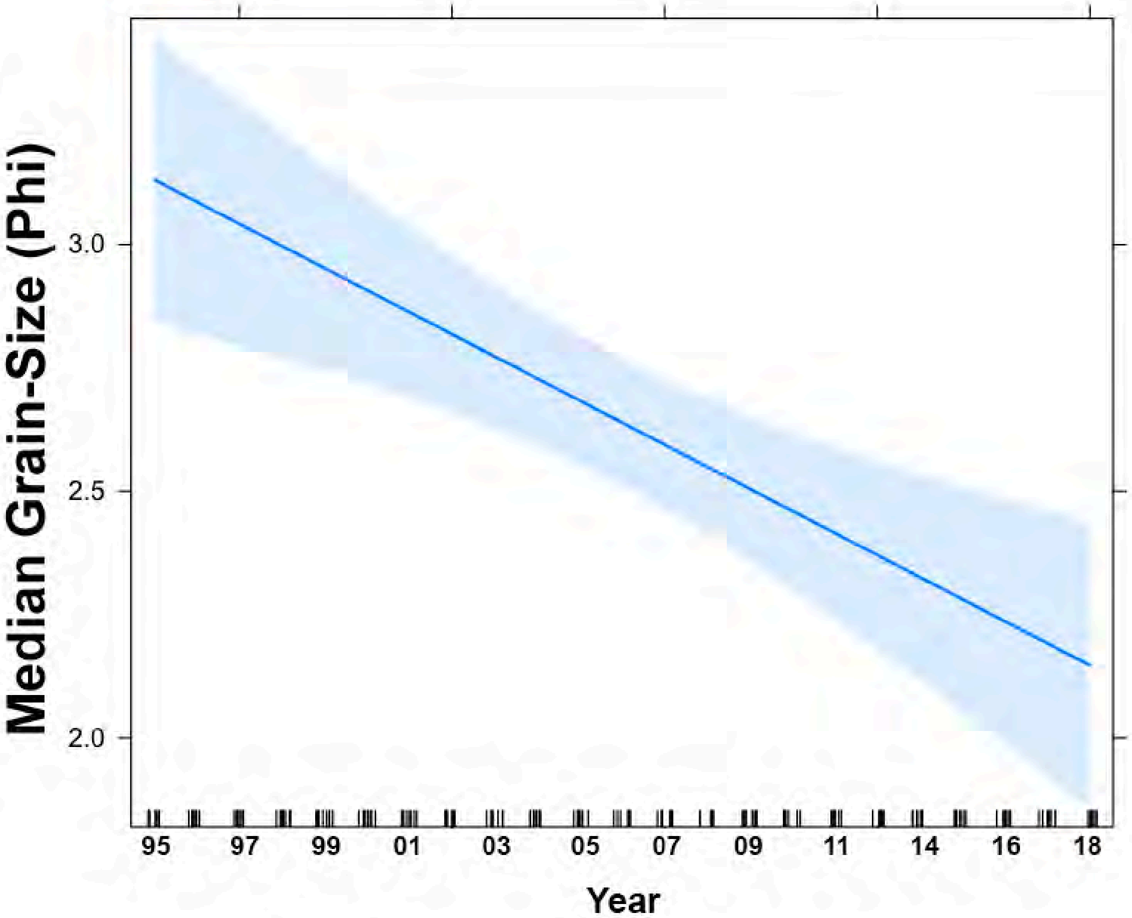

Figure 3-33. Relationship of median grain-size and time. Based on general linear model (see Table 3-9 for parameters).

3.3.3 Surficial Sediment Dynamics

Bed roughness estimated from images was categorized as being dominated by biological or physical processes, or a combination of both (Figures 3-34 and 3-35). The ratio, or odds (Agresti 1990), of stations that had physically dominated bed roughness relative to those that were biologically dominated was related to the Harbor's particulate organic carbon (POC) loading periods (Taylor 2010, 2018) and storm parameters, based on logistic regression (Table 3-10). The probability of bed roughness becoming more physically dominated increased as the sum of winter-period IWaveS storms increased. It also increased as the number of months before August monitoring without a major IWaveS storm increased (Figure 3-36), which likely reflects interaction of POC loading and storm parameters with time.

The influence of biological processes was strongest from 1998 to 2000 with implementation of full secondary treatment (Period III). Prior to this, during post sludge disposal and implementation of full primary treatment (Period II), biological processes were weakly dominant (Figure 3-37). There was dominance of physical processes after offshore diversion (Period IV) and continuing through to 2018. In 2018, the sixth stormiest year, only Stations R45 and T03 had biologically dominated surfaces (Figure 3-34). Another 12 stations had a combination of biological and physical processes dominating bed roughness, and 47 were primarily dominated by physical processes.

Declining POC loading, through its influence on biology (less organic matter as food), and increased winter-period storm strength, through its influence on surface and bottom-wave stress (higher IWindS and IWaveS, respectively), had the strongest influences on bed roughness. Biological processes dominated in 1999 when odds of biological dominance were 59:1 and sum of IWaveS was 292 Pa h, third lowest for the entire monitoring period (Figure 3-37). All but one Harbor station (T04) was biologically dominated (59 of 60 stations) in 1999. For 2013, the highest IWaveS year when the sum was about 1,000 Pa h, the odds of bed roughness being physically dominated was over 14:1 and remained high until 2016 when the odds dropped to less than 3:1. In 2015, 60 of 61 stations were physically dominated. This dramatic increase in odds was consistent with the stormy 2015 winter-period. Strong northeasters in October and February, a northwester in March, and a late northeaster in June all mixed the water column to depths greater than what are found in Boston Harbor. The last of these storms occurred two months prior to August 2015 sampling and likely affected surficial sediments by redistributing fine-grained sediments and destroying biogenic structures.

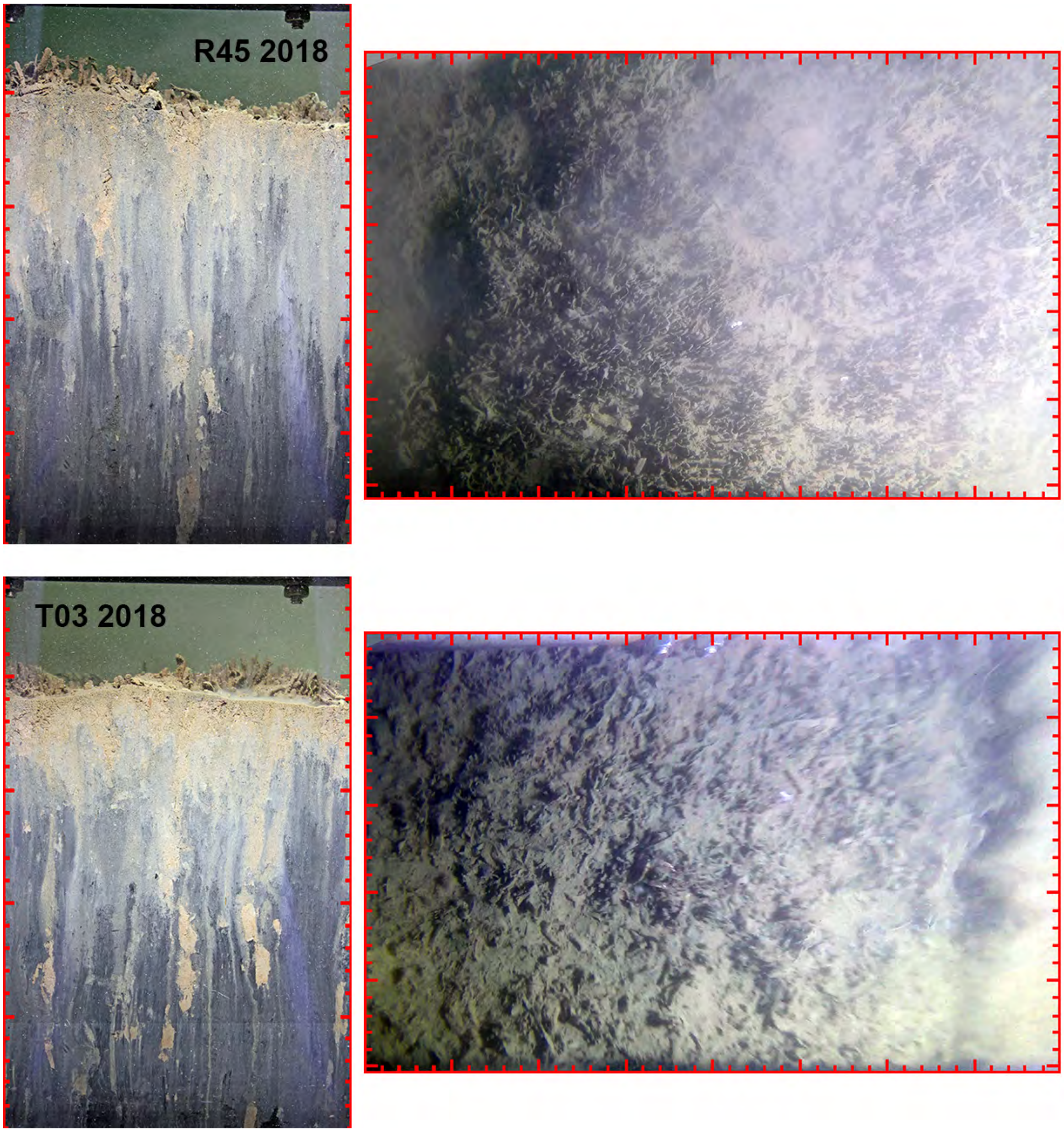

Figure 3-34. Examples of bed roughness being dominated by biological processes at Station R45 and Station T03 both off the northwest side of Long Island. SPI and surface image are from the same deployment. Scale on side of image is in cm.

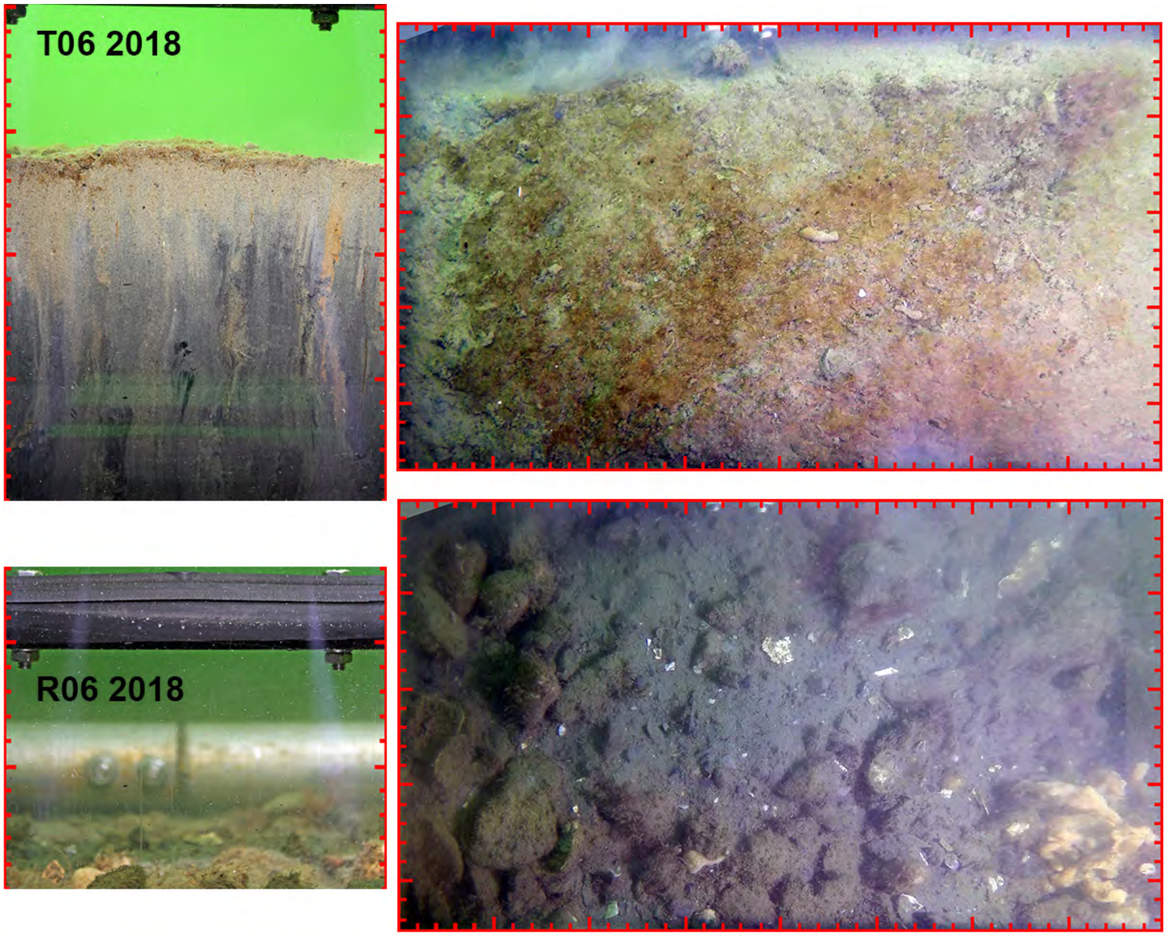

Figure 3-35. Examples of bed roughness being dominated by a combination of biological and physical processes at Station T06 in Nantasket Roads, and by physical processes at Station R06 off the northern tip of Long Island. Microalgal mat helps to bind sediment at T06. At R06 bottom wave stress has removed most fine sediments. SPI and surface image are from the same deployment. Scale on side of image is in cm.

Table 3-10. Logistic model for odds of bed roughness being physically dominated (positive parameter) verses biologically dominated (negative parameter) relative to storm parameters and POC loading periods of Taylor (2010).

	Parameter	SE	z-value	p
Number of IWaveS Storms	0.006	0.017	0.33	0.744
IWaveS Sum	0.002	<0.001	5.60	<0.001
Months Since Last IWaveS Storm Before August	0.055	0.016	3.41	<0.001
Period II to III	-1.248	0.269	-4.65	<0.001
Period II to IV	0.976	0.148	6.57	<0.001

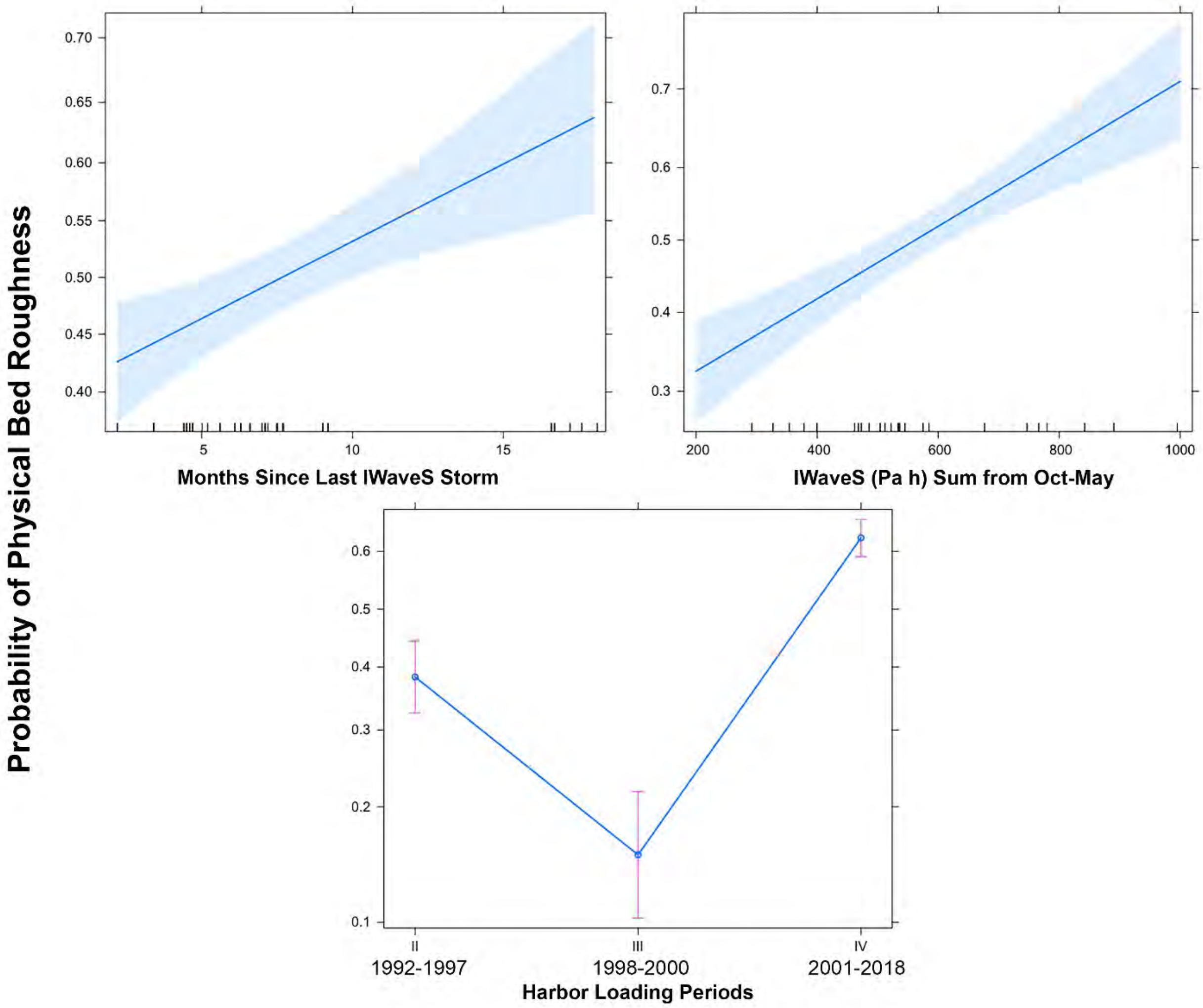

Figure 3-36. Predicted probability of physical processes dominating bed roughness for winter-period storm parameters, loading periods (Taylor 2010), and time. Parameter estimates are in Table 3-10. Shaded area and whiskers are 95% confidence interval.

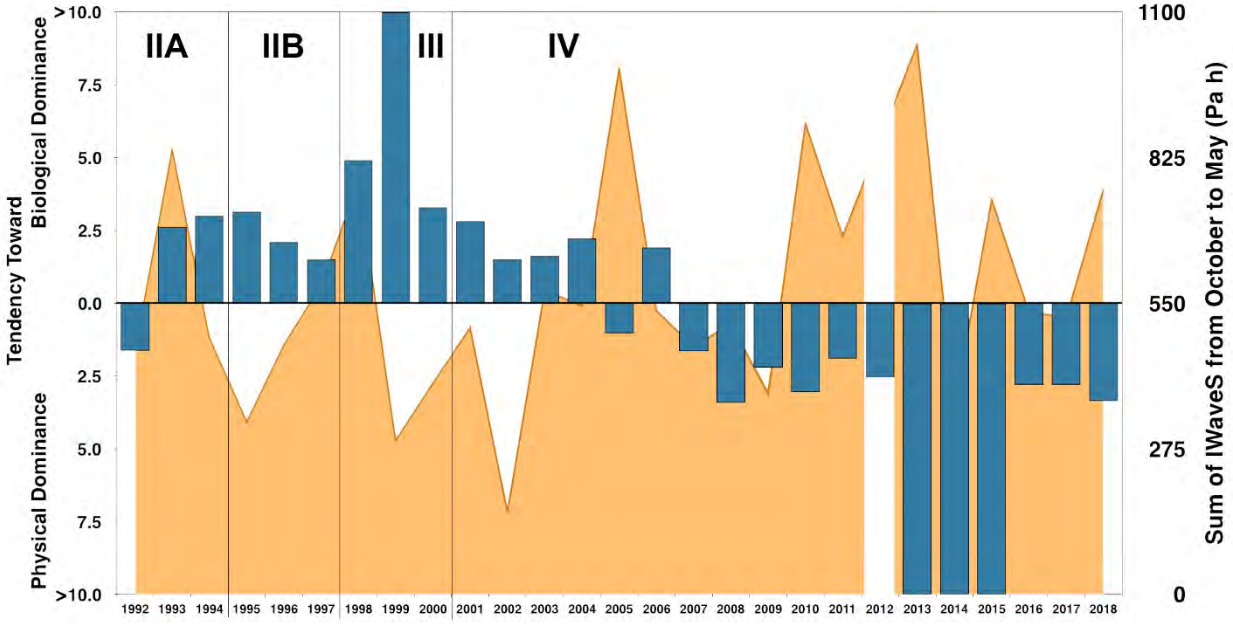

Figure 3-37. Trend in processes that control bed roughness at Harbor stations (express as odds) relative to winter-period IWaveS and loading periods. Vertical lines separate periods of change in wastewater loading (Taylor 2010). Period IIA is post sludge disposal, Period IIB is full primary treatment, Period III is full secondary treatment, and Period IV is offshore diversion.

3.3.4 General Benthic Habitat Conditions

The Organism Sediment Index (OSI) of Rhoads and Germano (1986) was used to assess general benthic habitat conditions within the Harbor. The OSI defines quality of benthic habitats by evaluating the depth of the aRPD layer, estimated benthic successional stage, the presence of gas bubbles in the sediment (an indication of high rates of methanogenesis that are associated with high carbon inputs to the sediment), and visual signs of the presence of low dissolved oxygen conditions (sulfide covered tubes, anaerobic sediment at the interface, bacterial mats) at the sediment-water interface. The OSI ranges from -10 for poorest quality habitats to +11 for highest quality habitats. For northeast coastal embayments OSI values >6 indicate good habitat conditions and are generally associated with bottoms that are not heavily influenced by stress, either physical or anthropogenic (Rhoads and Germano 1986).

Historically, the five lowest years for OSI were in the period from 1992 (the start of monitoring) to 2000 (Figure 3-38). Variation in OSI from 1992 to 2018 was related to the sequence of treatment upgrades prior to diversion in 2001 (Taylor 2010) but not to storm parameters (IWindS, IWaveS, or skew surge; Table 3-11). During post sludge disposal (1992 to 1994, Period IIA), OSI averaged <6, with implementation of full primary treatment (1995 to 1997, Period IIB) it was >6, and with implementation of secondary treatment (1998 to 2000, Period III) OSI was again <6. From offshore diversion in 2001, the start of Period IV, to 2009, OSI increased (breakpoint regression, df = 7, $R^2 = 0.58$, p = 0.017). The three highest years for OSI were 2007 to 2009 with an average of 7.8. From 2010 to 2018 there was a general trend for OSI to decline (Figure 3-38). The significant decline in OSI between 2017 and 2018 was harbor-wide (based on pair t-test, Table 3-12) with the average OSI for the nine harbor regions all lower in 2018 (Figures 3-39 and 3-40). 2018 was the sixth stormiest year and more stations changed modal grain-size in 2018 than any other year. Over half of all stations changed modal grain-size (38 of 61) with 18 becoming coarser and 20 finer (Figure 3-27).

The last year the Harbor grand mean OSI was <6 was 2002, but stressed benthic habitat conditions continued to predominate in Dorchester Bay and Inner Harbor in 2018. In those regions, both the grand mean and the mean for 2018 are less than 6.0 (Figure 3-40). Additionally the Deer Island Flats average OSI was <6, for the first time since 2000, due to channel dredging that removed good quality habitat at Station R02 in August 2018 just days before monitoring (Figure 3-41). If R02 is excluded, the mean OSI for Deer Island Flats would be 7.6. At the time of sampling other stations near R02 did not appear to be affected by the ongoing dredging operations, but it is expected that Stations T05A and R09 will be affected in August 2019 (Figure 3-42).

Better habitat conditions continued to characterize the mid and outer harbor areas of Deer Island Flats and President Roads with the highest OSI regions being off Long Island, Nantasket Roads, and Hull Bay. Over the 27 years of Harbor SPI monitoring, stations off Long Island, Nantasket Roads, and Hull Bay consistently had good benthic habitat conditions (Table 3-13, Figure 3-40).

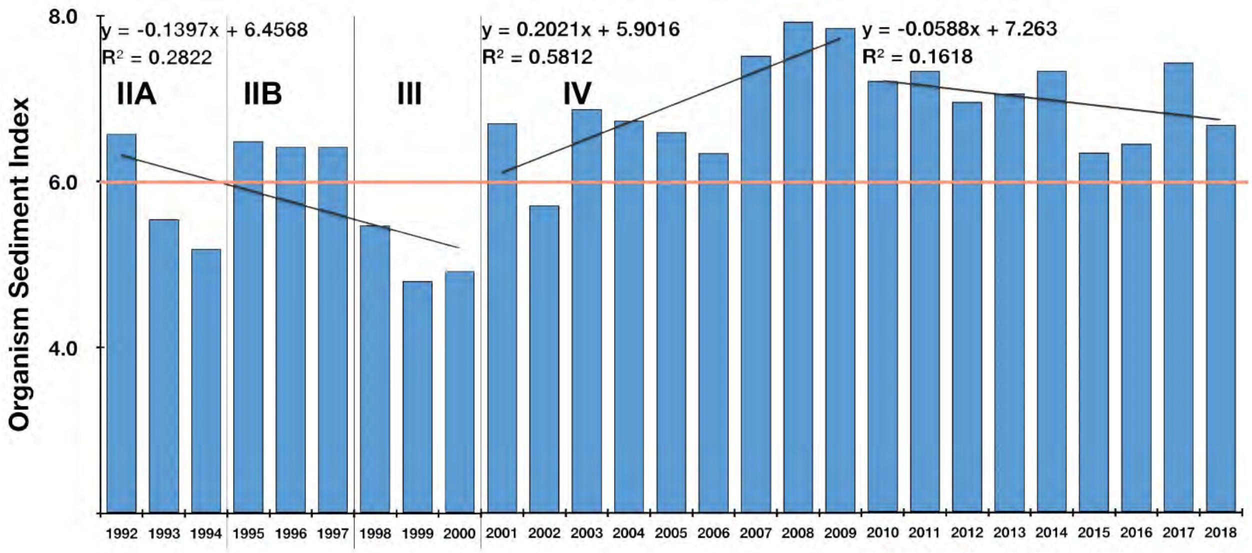

Figure 3-38. Annual Organism Sediment Index (OSI) for all Harbor stations. Regression lines are from breakpoint regression. Only the period from 2001 to 2009 showed significant change (p = 0.017). The horizontal line is OSI of 6, the breakpoint between good vs poor habitat. Vertical lines separate periods of change in wastewater treatment based on Taylor (2010). IIA is post sludge disposal, IIB is full primary treatment, III is full secondary treatment, and IV is offshore diversion. OSI <6 indicates stressed benthic habitat.

Table 3-11. General linear models of OSI for all Harbor stations with storm parameters and POC loading periods of Taylor (2010).

Factor	Parameter	SE	t-value	p
Number of IWaveS Storms	-0.006	0.038	-0.17	0.871
IWaveS Sum	-0.000	0.001	-0.56	0.585
Months Since Last IWaveS Storm Before August	-0.026	0.033	-0.79	0.440
Period II to III	-1.012	0.467	-2.17	0.043
Period II to IV	0.895	0.310	2.88	0.009
Number of IWindS Storms	0.002	0.040	0.06	0.956
IWindS Sum	0.002	0.004	0.52	0.612
Months Since Last IWindS Storm Before August	0.009	0.031	0.30	0.769
Period II to III	-1.097	0.544	-2.02	0.058
Period II to IV	0.720	0.341	2.12	0.047
Skew Surge	-0.123	0.168	-0.73	0.471
Period II to III	-1.085	0.405	-2.68	0.013
Period II to IV	0.821	0.266	3.09	0.005

Table 3-12. Comparison of annual change in OSI from year to year based on paired t-test. Storm ranking is based on sum of winter-period IWaveS.

Year Pair Compared	t	df	p	Change in OSI	Storm Year Rank
92-93	-2.68	33	**0.011**	-1.1	4
93-94	-0.91	44	0.369	-0.4	17
94-95	3.43	45	**0.001**	1.6	23
95-96	-1.15	54	0.255	-0.4	18
96-97	0.06	42	0.955	0.0	9
97-98	-1.03	43	0.308	-0.4	5
98-99	-2.00	53	0.051	-0.6	24
99-00	-0.21	56	0.835	0.1	20
00-01	5.90	54	**<0.001**	1.8	16
01-02	-3.00	53	**0.004**	-1.0	25
02-03	5.50	56	**<0.001**	1.1	10
03-04	-0.34	55	0.731	-0.1	11
04-05	-0.44	57	0.660	-0.1	2
05-06	-1.33	55	0.190	-0.3	12
06-07	5.70	56	**<0.001**	1.3	19
07-08	1.17	55	0.249	0.3	15
08-09	-0.29	56	0.773	-0.1	21
09-10	-2.30	54	0.025	-0.6	3
10-11	0.55	54	0.604	0.1	8
11-12	-1.97	56	0.054	-0.4	No Storm Data
12-13	0.39	55	0.700	0.1	1
13-14	1.57	53	0.121	0.3	22
14-15	-4.66	55	**<0.001**	-1.0	7
15-16	0.69	55	0.495	0.1	13
16-17	4.49	55	**<0.001**	0.9	14
17-18	-2.96	56	**0.004**	-0.8	6

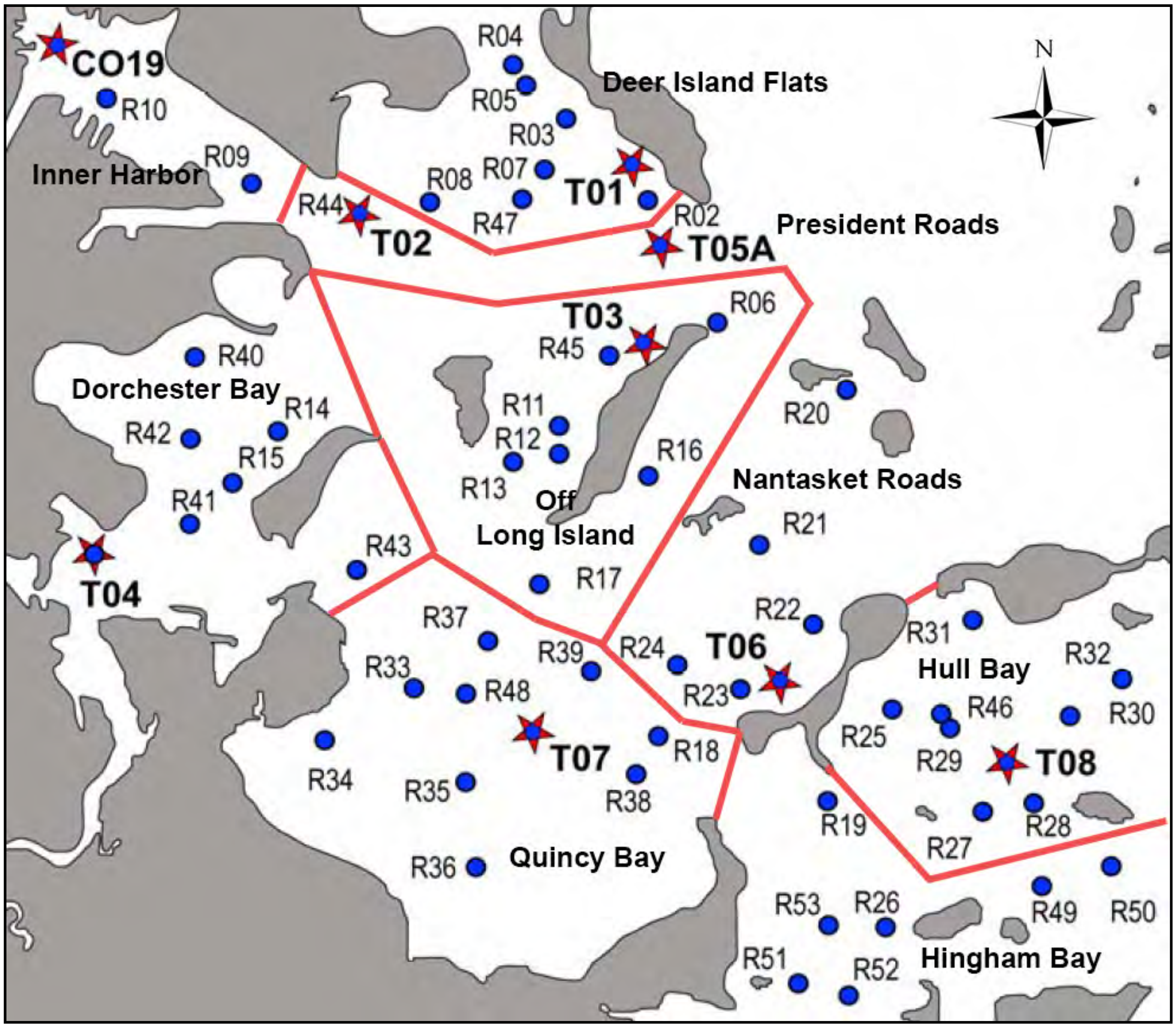

Figure 3-39. Boundaries for Boston Harbor regions. SPI (circles) sampled at 53 reconnaissance SPI-stations, and SPI and infauna (stars) at eight traditional grab stations.

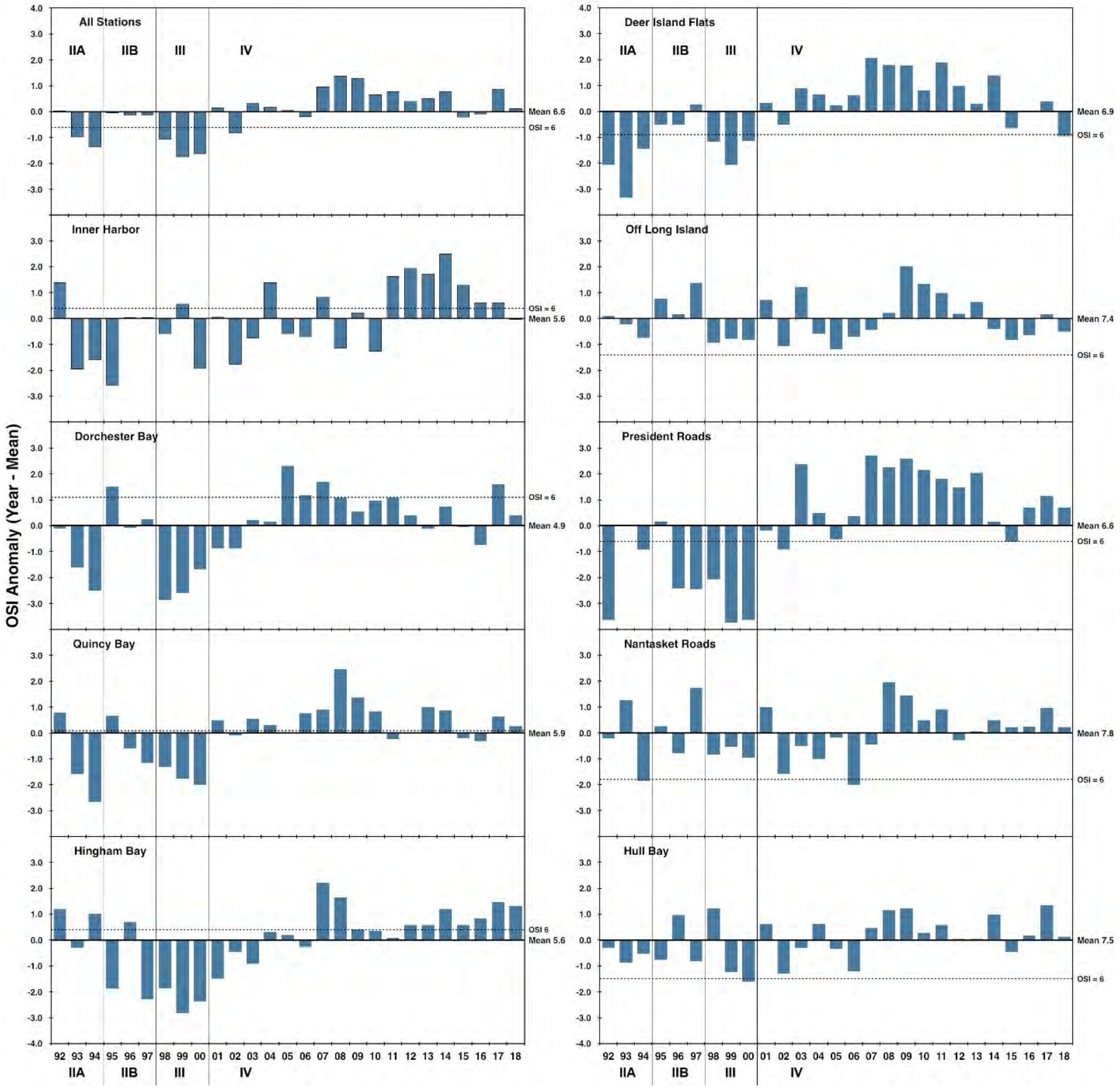

Figure 3-40. OSI anomaly (year mean - grand mean) for Boston Harbor and regions (see figure 3-23). Higher than grand mean OSI years are positive and lower are negative values. Shading separate periods of change in wastewater treatment based on Taylor (2010). Period IIA is post sludge disposal, Period IIB is full primary treatment, Period III is full secondary treatment, and Period IV is offshore diversion. OSI <6 indicates stressed benthic habitat.

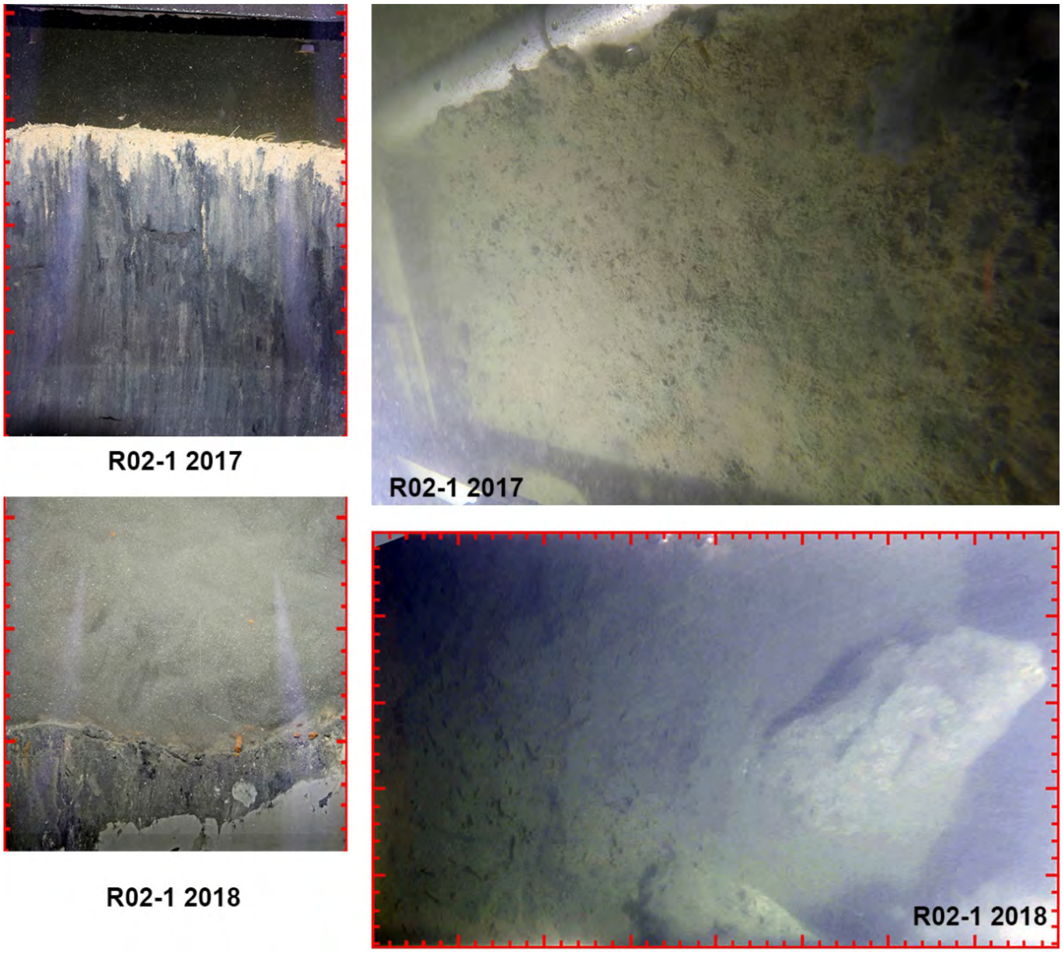

Figure 3-41. Benthic habitat conditions at Station R02 in August 2017 prior to dredging and in August 2018 during nearby dredging operations. OSI in 2017 was high at 7.6 and low at -4 in 2018. Scale on side of images is in cm. Surface image from 2017 is not scaled. Frame tubing is 5 cm diameter.

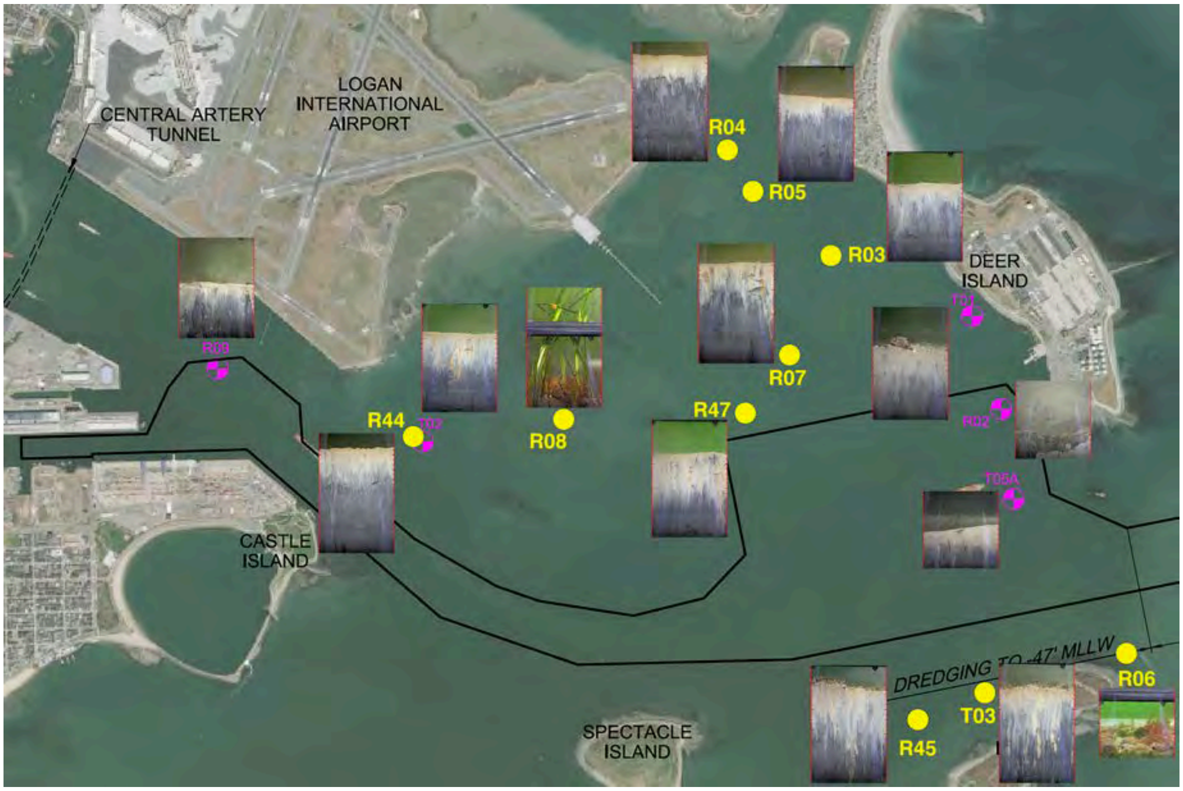

Figure 3-42. Extent of channel deepening within Boston Harbor 2018-2019 (solid lines). Images for nearby SPI stations indicate a diversity of habitats are present. Dredging impacts were observed at R02 in August 2018, and expected at T05A and R09 by August 2019. Project base map provided by Corps of Engineers. Scale on side of images is in cm.

Table 3-13. OSI summarized by Boston Harbor regions from 1992 to 2018 (see Figure 3-42 for boundaries). Regions are arranged from lowest to highest grand mean for all years. An OSI value of <6 indicates stressed benthic habitat.

Harbor Region	Minimum of All Years		Maximum of All Years		Grand Mean All Years	2017	2018	Delta (2017-2016)	N
Dorchester Bay	2.1	1998	7.2	2005	4.9	6.5	5.3	-1.2	7
Inner Harbor	3.0	1995	8.1	2014	5.6	6.2	5.6	-0.7	3
Hingham Bay	2.8	1999	7.9	2007	5.6	7.1	7.0	-0.2	7
Quincy Bay	3.3	1994	8.4	2008	5.9	6.6	6.2	-0.4	10
President Roads	2.9	1999	9.3	2007	6.6	7.8	7.3	-0.4	3
Deer Island Flats	3.6	1993	9.0	1997	6.9	7.3	6.0	-1.3	7
Off Long Island	6.2	2005	9.4	2009	7.4	7.5	6.9	-0.7	7
Hull Bay	5.9	1999	8.8	2017	7.5	8.9	7.7	-1.2	9
Nantasket Roads	5.8	2006	9.8	2008	7.8	8.8	8.1	-0.7	6

3.3.5 Apparent Color Redox-Potential Discontinuity (aRPD) Layer Depth

The apparent color redox potential discontinuity (aRPD) layer depth is an approximation of the oxidation-reduction state of sediments. The primary factors determining aRPD layer depth are sediment grain-size, organic content, and biogenic activity (microbial and metazoan). As grain-size increases, sediments should become more oxic through porewater pumping (Huettel and Webster 2001). As organic content declines, sediments should become more oxic through microbial metabolism and bioturbation (Yingst and Rhoads 1978, Tucker et al. 2014).

There were no significant relationships between storm factors and aRPD layer depth. This is not surprising given the complexities of biogeochemistry and animal-sediment interactions. However, the oscillating pattern in aRPD appeared loosely linked to biogenic activity. The depth of the aRPD layer increased as the number of oxic voids increased, which indicate high levels of bioturbation (GLM, F = 20.7, p = <0.001) (Figure 3-45). Most oxic voids were formed by feeding of larger head-down subsurface deposit feeders (for example the polychaete *Clymenella torquata*) or burrowing species that ventilate their burrows during feeding activities (for example the amphipod *Leptocheirus pinguis*). Biogenic structures of both species were seen at Station T02 where their abundance was 180 individuals (ind)/0.04m^2 for *Leptocheirus pinguis* and 260 ind/0.04m^2 for *Clymenella torquata* in 2018 (Figure 3-44). Harbor wide patterns in biogenic structures from 1998 to 2018 were not related to storm factors.

The numbers of *Ampelisca* spp. oscillated twice between 2007 and 2017 (See Section 3.3.6, *Ampelisca* spp. Trends) however its effect on the aRPD was not as clear as *Leptocheirus pinguis*. In 2007 *Leptocheirus pinguis* abundance started to increase and peaked at over 2,165 ind/0.04 m2 in 2008 at

Station T02, then crashed in 2009 to 51 ind/0.04 m2 (Figure 3-45). Annual mean aRPD layer depth for all Harbor stations and *Leptocheirus pinguis* abundance at the eight T-Stations were strongly correlated over the entire monitoring period (r = 0.66, p = <0.001, df = 24). This was not the case for *Ampelisca* spp. abundance (r = 0.15, p = 0.728, df = 24; Figure 3-46).

On a harbor-wide basis there were significant patterns in aRPD layer depth related to organic loading periods as defined by Taylor (2010; Table 3-14). The first was an increase in aRPD layer depth that occurred as post sludge disposal (Period IIA) ended in 1994 and full primary treatment started in 1995 (Period IIB; Figure 3-47). At this time, POC loading went from about 60 tons per year (ton/y) to 50 ton/y (Taylor 2010). aRPD increased one cm from 1.8 cm (SD = 1.22 cm) in 1994 to 2.8 cm (SD = 2.17 cm) in 1995. With the start of full primary treatment in 1998 (Period III), POC declined again to about 23 ton/y and the aRPD layer depth was 2.0 cm (SD = 1.58 cm) but not significantly different from Period IIA. In the post diversion period (Period IV), POC loading declined to <6 ton/y and aRPD layers were again significantly deeper than Period IIA. Between 1992 to 2018 the largest annual mean aRPD layer depth occurred at the start of Period IV in 2001 (3.4 cm, SD = 2.10 cm). When controlling for loading periods, there was still a significant effect of year resulting in the aRPD layer depth decreasing by a few mm per year. For 2018, the aRPD layer depth was 2.0 cm (SD = 1.35 cm), down from 2.4 cm (SD = 1.67 cm) in 2017 and below the long-term mean of 2.2 cm.

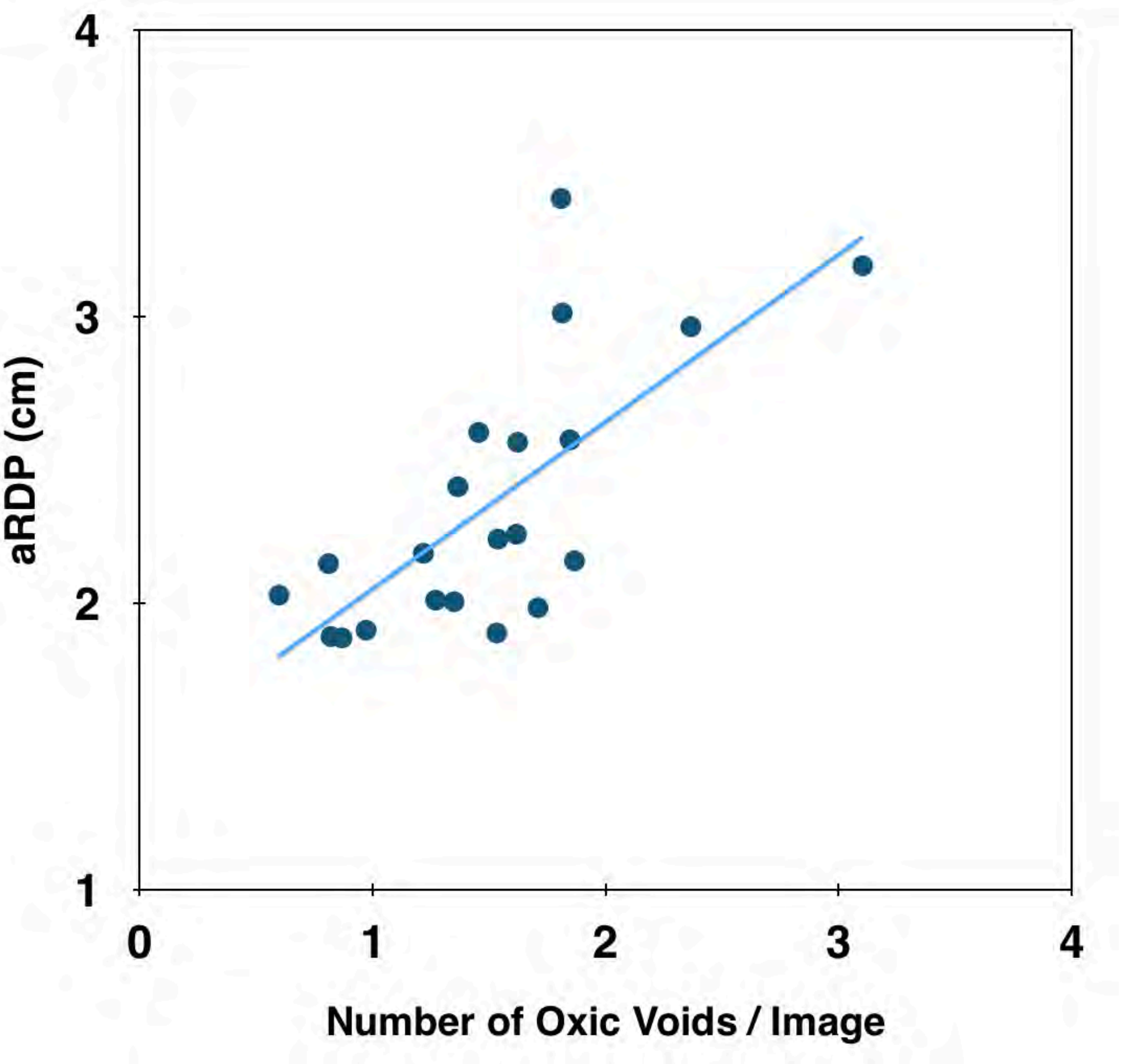

Figure 3-43. Relationship between annual mean aRPD layer depth and number of oxic voids/image.

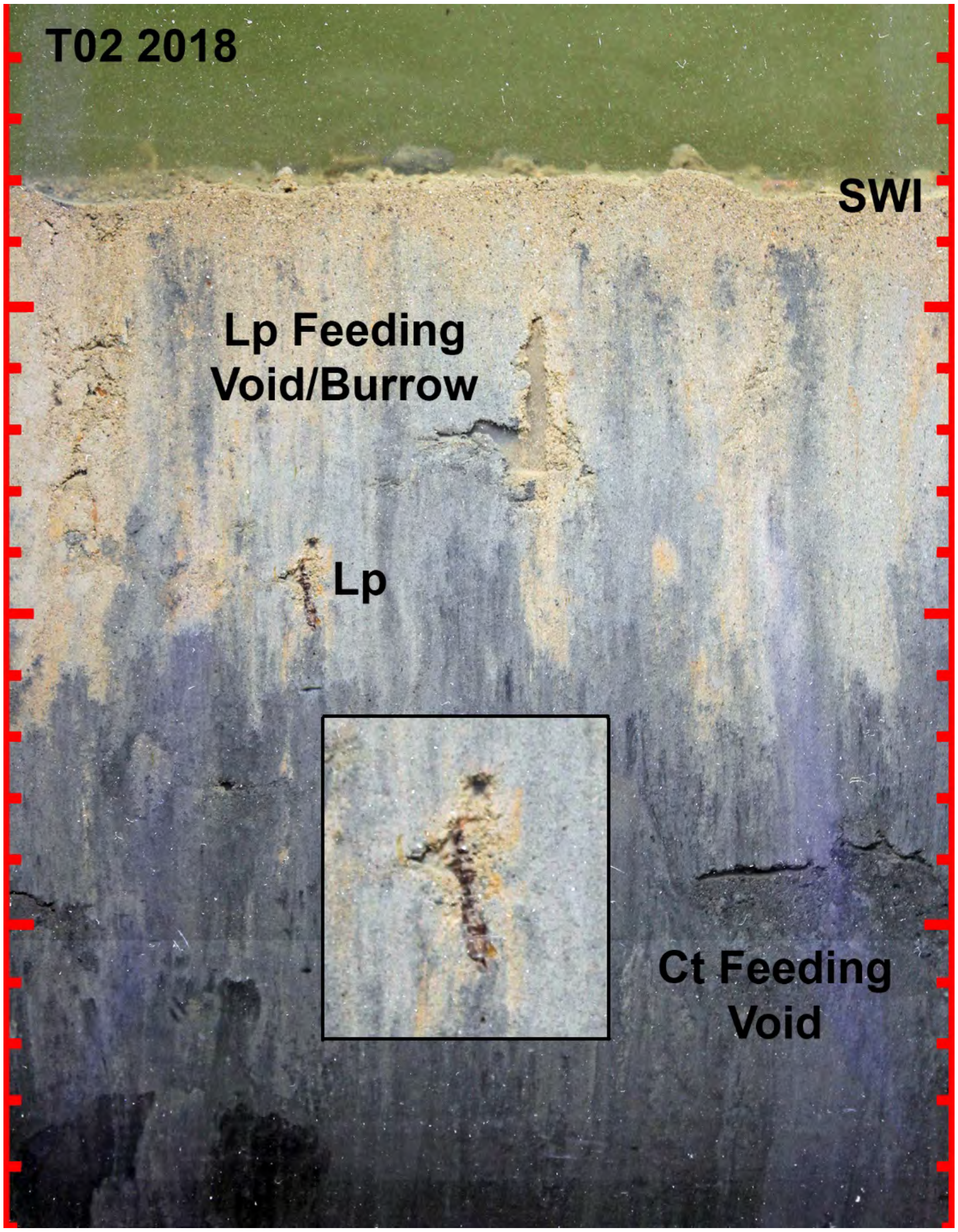

Figure 3-44. Examples of biogenic structures associated with feeding activities of the polychaete *Clymenella torquata* (Ct) and the amphipod *Leptocheirus pinguis* (Lp), Station T02 in President Roads, 2018. A *Leptocheirus pinguis* can be seen with its dorsal side pressed against the faceplate (insert). SWI is sediment-water-interface. Scale on side of image is in cm.

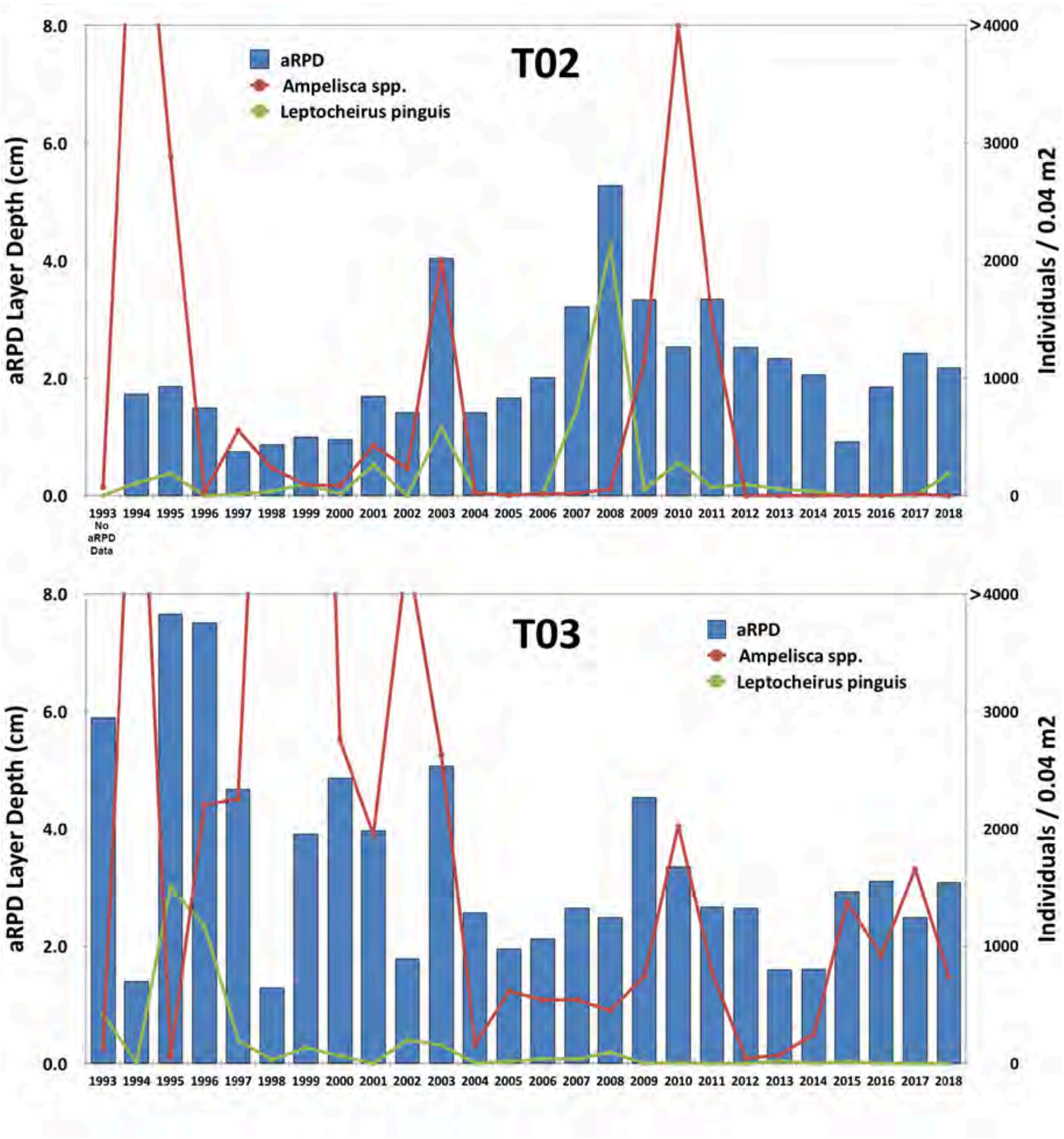

Figure 3-45. aRPD layer depth (cm) for Station T02 in President Roads and Station T03 off the western side of Long Island, and *Ampelisca* spp. and *Leptocheirus pinguis* abundance.

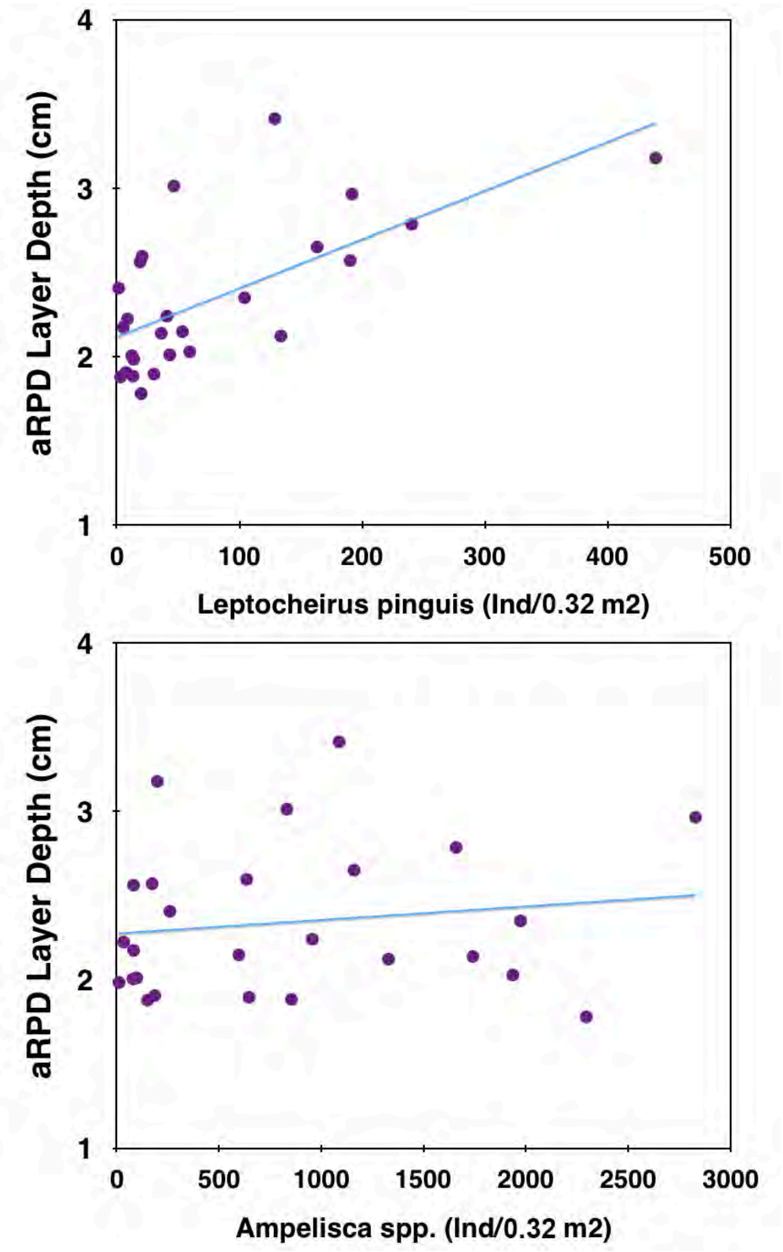

Figure 3-46. Scatter plot with linear regression line for mean annual aRPD layer depth (cm) for all 61 Harbor stations and the sum of abundance of *Leptocheirus pinguis* and *Ampelisca* spp. at the eight T-Stations. The mean annual aRPD layer depth and sum of abundance of *Ampelisca* spp. were not significantly correlated.

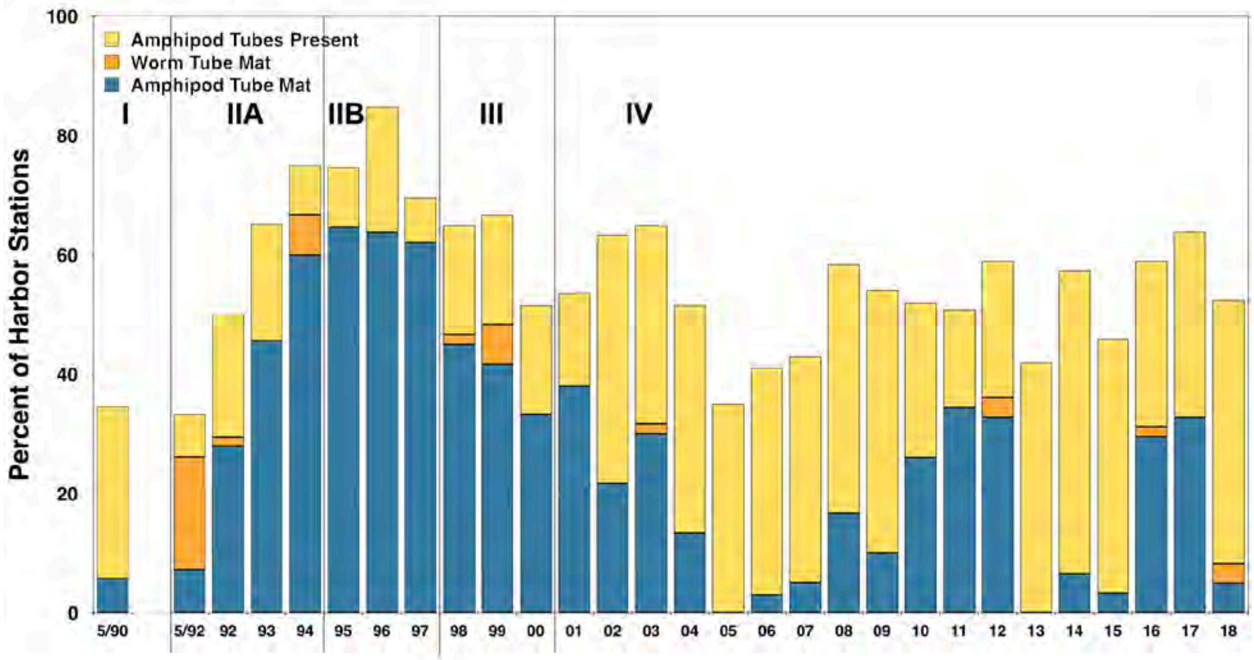

Figure 3-47. Annual mean aRPD layer depth (cm) for all Harbor stations. Shading separate periods of change in wastewater treatment based on Taylor (2010). Period IIA is post sludge disposal, Period IIB is full primary treatment, Period III is full secondary treatment, and Period IV is offshore diversion.

Table 3-14. Parameter estimates from general linear models of aRPD layer depth with organic loading periods of Taylor (2010) and year for the whole Harbor and regions (see Figure 3-42 for region boundaries).

Factor	Parameter	SE	t-value	p		Parameter	SE	t-value	p
	Whole Harbor					**Deer Island Flats**			
Period IIA to IIB	0.766	0.322	2.38	0.027		1.175	0.509	2.308	0.031
Period IIA to III	0.298	0.335	0.89	0.383		1.444	0.529	2.728	0.012
Period IIA to IV	1.074	0.379	2.84	0.010		2.978	0.598	4.981	<0.001
Year	-0.037	0.018	-2.13	0.045		-0.088	0.028	-3.184	0.004
	Inner Harbor					**Off Long Island**			
Period IIA to IIB	0.324	0.391	0.83	0.415		1.578	0.673	2.344	0.029
Period IIA to III	0.409	0.406	1.01	0.325		1.119	0.700	1.600	0.124
Period IIA to IV	0.530	0.459	1.16	0.260		1.645	0.790	2.081	0.049
Year	-0.011	0.021	-0.54	0.597		-0.093	-0.037	2.523	0.019
	Dorchester Bay					**President Roads**			
Period IIA to IIB	0.824	0.418	1.97	0.061		0.597	0.725	0.824	0.419
Period IIA to III	-0.389	0.434	-0.90	0.381		0.189	0.748	0.253	0.803
Period IIA to IV	0.479	0.491	0.98	0.340		1.998	0.828	2.414	0.025
Year	-0.015	0.023	-0.64	0.526		-0.022	-0.035	0.615	0.545
	Quincy Bay					**Nantasket Roads**			
Period IIA to IIB	0.944	0.368	2.57	0.018		0.511	0.814	0.628	0.536
Period IIA to III	0.575	0.382	1.51	0.146		-0.327	0.846	-0.387	0.703
Period IIA to IV	1.771	0.432	4.10	0.000		-0.298	0.956	-0.312	0.758
Year	-0.081	0.020	-4.06	0.001		0.022	0.044	0.490	0.629
	Hingham Bay					**Hull Bay**			
Period IIA to IIB	-0.083	0.271	-0.31	0.761		0.788	0.686	1.149	0.263
Period IIA to III	-0.900	0.281	-3.20	0.004		0.533	0.713	0.747	0.463
Period IIA to IV	-0.199	0.318	-0.63	0.537		0.620	0.805	0.769	0.450
Year	0.029	0.015	1.96	0.063		0.003	0.037	0.076	0.940

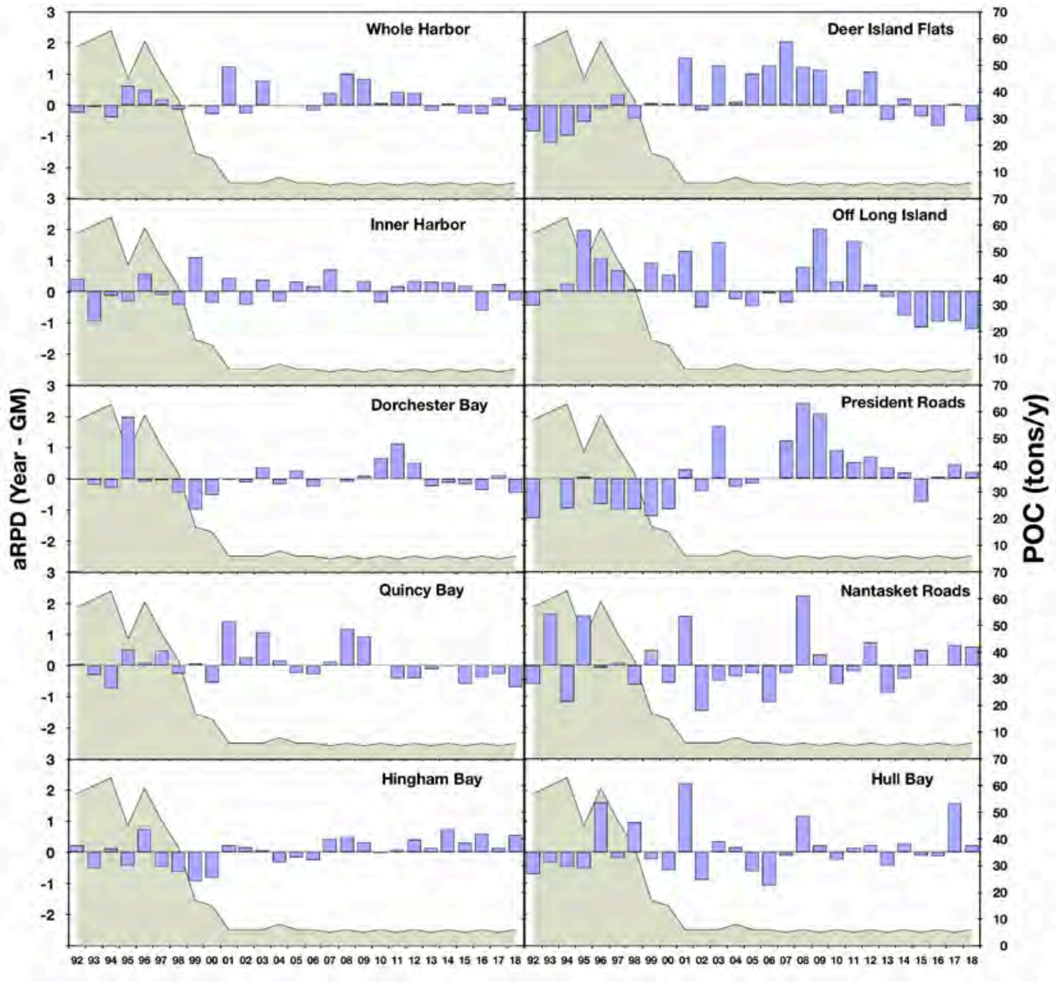

Figure 3-48. aRPD layer depth (blue bars) anomaly (year mean - grand mean in cm) for Boston Harbor and regions (see Figure 43 for region boundaries) compared with particulate organic carbon (POC) loading (shaded region) from Taylor (2010). POC loading from 2007 to 2018 are estimated base on 2003 to 2006 loading.

3.3.6 *Ampelisca* spp. Trends

Much of the change in benthic habitat quality can be seen in the dynamics of the tube building amphipod *Ampelisca* spp. Over the 27 years of SPI monitoring, biogenic activity associated with the presence of *Ampelisca* spp. and the burrowing amphipod *Leptocheirus pinguis* had the most influence on improving benthic habitat quality (Diaz et al. 2008, Tucker et al. 2014). *Ampelisca* spp. were also the numerically dominant suspension feeder and a key taxa in stabilizing sediments with their dense tube mats. They are not active bioturbators like the interface feeding *Leptocheirus pinguis*, but the decline in both of these species from 1995 to 2000 was a large factor in a shallowing of the aRPD layer depth. *Leptocheirus pinguis* construct low stability U-shaped burrows in muddy sediments, which can extend up to 10 cm below the sediment surface (Thiel 1999). It pumps water through its burrow to feed on suspended particles (Thiel 1997, Shull et al. 2009).

Benthic data collected prior to the start of MWRA long-term monitoring documented that *Ampelisca* spp. occurred over much of the Harbor (Blake et al. 1989). In 1978, they were at mat densities[5] at two stations in Quincy Bay. The following year mat densities occurred at three stations (one each in Dorchester Bay, Nantasket Roads, and Hull Bay). In 1982, *Ampelisca* spp. did occur but at lower densities (Blake et al. 1989). It is likely that *Ampelisca* spp. were, and continue to be, a consistent predominant taxa within Boston Harbor (Blake et al. 1989, Kropp and Diaz 1995, Gallagher and Keay 1998, Rutecki et al. 2019a).

The first SPI survey in Boston Harbor was June 1989 and blue mussels and polychaete tube mats were widely distributed within the Harbor but not *Ampelisca* spp. tube mats (SAIC 1990). By May 1990, *Ampelisca* spp. tube mats were present at 10% of SPI stations (3 of 30) close to current long-term monitoring stations (SAIC 1992). Another SPI survey in May 1992 also found *Ampelisca* spp. tube mats at about 10% of stations (4 of 43) close to current monitoring stations and polychaete tube mats at another 20% of stations (8 of 43; Figure 3-49). Considering all stations sampled in May 1990, 6% had amphipod mats (3 of 52) with no polychaete mat reported. Two years later in May 1992, 7% had amphipod mats (5 of 69) and 14% had polychaete mats (13 of 69; SAIC 1992). When annual August SPI monitoring started in 1992, three months later, there was an increase in the occurrence of *Ampelisca* spp. tube mats to about a third of stations (14 of 43) with polychaete mats at one station (Figure 3-50). Part of the pattern in *Ampelisca* spp. between spring (May) and summer (August) was likely related to its life-history and timing of SPI sampling. Breeding of the overwintering population occurs in April-May which gives rise to the summer population that breeds in June-July (Mills 1967). It is the tubes of the summer population that are seen in August.

Mats peaked from 1994 to 1997 likely in response to a combination of factors related to POC loading and storms (Figures 3-50 and 3-51). Highest POC loading years were 1990 and 1991 prior to end of sludge dumping (Period I). From 1992 to 1998 (Period II) POC loading continued to decline with improvements in treatment and relocation of Nut Island outfalls to Deer Island in 1998 (Taylor 2010). Of the top 14 winter-period storm years only one in 1993 occurred prior to offshore diversion in 2001 (Figure 3-51). Other factors involved with variation in *Ampelisca* spp. populations would include inter-annual variation in population size and composition of the *Ampelisca* species (Mills 1967). At least three species occurred at Harbor T-Stations (*A. abdita, A. macrocephala,* and *A. abdita*).

It is possible that prior to reductions in POC loading and improvements in treatment levels, either organic or pollutant loading was too high for *Ampelisca* spp. to thrive. The probability of tube mats being present was significantly related to loading periods but also to storms and time (Table 3-15). Higher POC loading in Periods II and III favored the presence of tube mats relative to post-diversion Period IV. The probability of mats declined in years with higher numbers of storms and with increasing IWaveS storm intensity (Figure 3-52). Months since the last major IWaveS storm before August was not a significant factor. Controlling for POC loading and storm factors, odds of mats still declined through time pointing to other factors being involved in long-term population dynamics. Possibilities include declining primary

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[5] To be consider an *Ampelisca* spp. tube mat, densities had to be >20,000 individuals/m^2, which is about the 80th percentile for abundance at stations where SPI identified tube mats.

productivity within the Harbor related to improvements in wastewater treatment (Oviatt et al. 2007) or climate related factors (Goberville et al. 2010).

The number of stations with mat densities of *Ampelisca* spp. started to decline in 1998 coincident with the declines in organic and pollutant loading, and primary production. *Ampelisca* spp. do have a life history that reflects a combination of opportunism in responding to organic matter (McCall 1977) but are also sensitive to pollutants (Wolfe et al. 1996) making them good indicators of habitat quality (Chang et al 1992). The year 1998 was also the fifth stormiest year from 1992 to 2018 based on winter-period IWaveS stress. By 2005, the second stormiest year and organic loading about 6 ton/yr, no tube mats were observed. That year integrated IWaveS stress was 995 Pa h from a series of strong storms (14th, 16th, and 20th largest for IWaveS), the 20th occurring in late May. The bottom turbulence from these storms may have disrupted tube mats and allowed recolonization by other species. Tube mats reappeared 2006 and increased to levels seen in the late 1990s by 2011 and 2012, despite 2010 being the third stormiest year (Figure 3-24). In 2013, the stormiest year with an integrated IWaveS of 1,039 Pa h, no tube mats were observed for the second time. In 2014 tube mats reappeared and increased to 20 stations in 2017. This is an indication there was sufficient food and habitat stability to support high population densities over a broad area of the harbor. *Ampelisca* spp. mats again declined in 2018 to three stations, likely in response to 2018 being the sixth stormiest year. An additional two stations had polychaete tube mats (Figures 3-50 and 3-51).

3.3.7 Eelgrass

Further evidence of improvement in benthic habitat quality is the eelgrass (*Zoestra marina*) bed that has persisted at Station R08 since 2008 (Figure 3-53). The presence of eelgrass at Station R08 was a result of successful restoration efforts by MA Division of Marine Fisheries in the Deer Island Flats (Governors Island Flats) area (Evans et al. 2018). Beds of eelgrass have also been reported in and around Nahant and Lynn Harbor, inside the northernmost breakwater in Winthrop, western and southwestern sides of Calf Island, off Long Island, and Whitehead Flats in Hull (P. Colarusso personal communication to K. Keay). In a statewide analysis of eelgrass trends between 1996 and 2007, Costello and Kenworthy (2011) found eelgrass area declined within Boston Harbor from about 81 ha to 27 ha between 1996 to 2001and then increased to 47 ha by 2007. The increase in Harbor eelgrass coincided with the startup of the offshore outfall. Costello and Kenworthy (2011) reported a similar decadal scale recovery of eelgrass in a small embayment on Long Island Sound following the removal of a municipal wastewater discharge source. They concluded that with improved management of water quality environmental conditions exist in Massachusetts where seagrasses can either thrive or expand.

Figure 3-49. Examples of tube mats present in April 1989 that were composed primarily of polychaetes. By May 1992 mats were composed primarily of *Ampelisca* spp. (SAIC 1990, 1992).

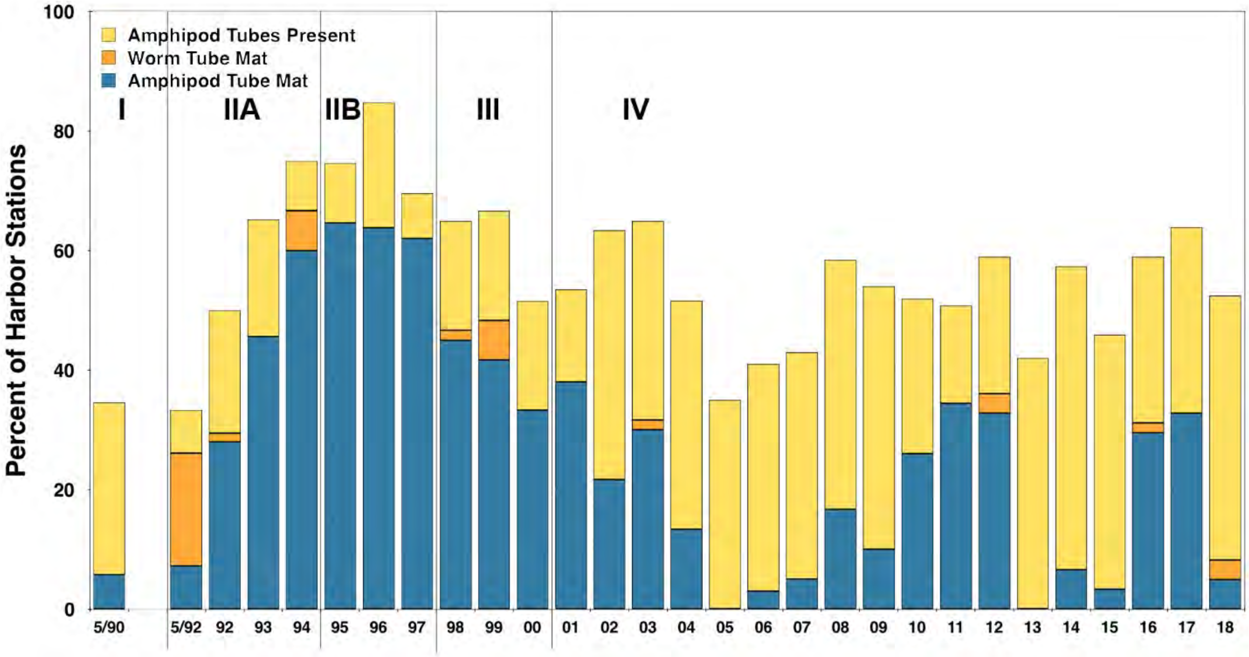


Figure 3-50. Histogram of *Ampelisca* spp. and polychaete worm tube mats present at Harbor stations. Annual monitoring started August 1992. Shading separate periods of change in POC loading based on Taylor (2010). Period I is primary treatment and sludge disposal, Period IIA is post sludge disposal, Period IIB is full primary treatment, Period III is full secondary treatment, and Period IV is offshore diversion.

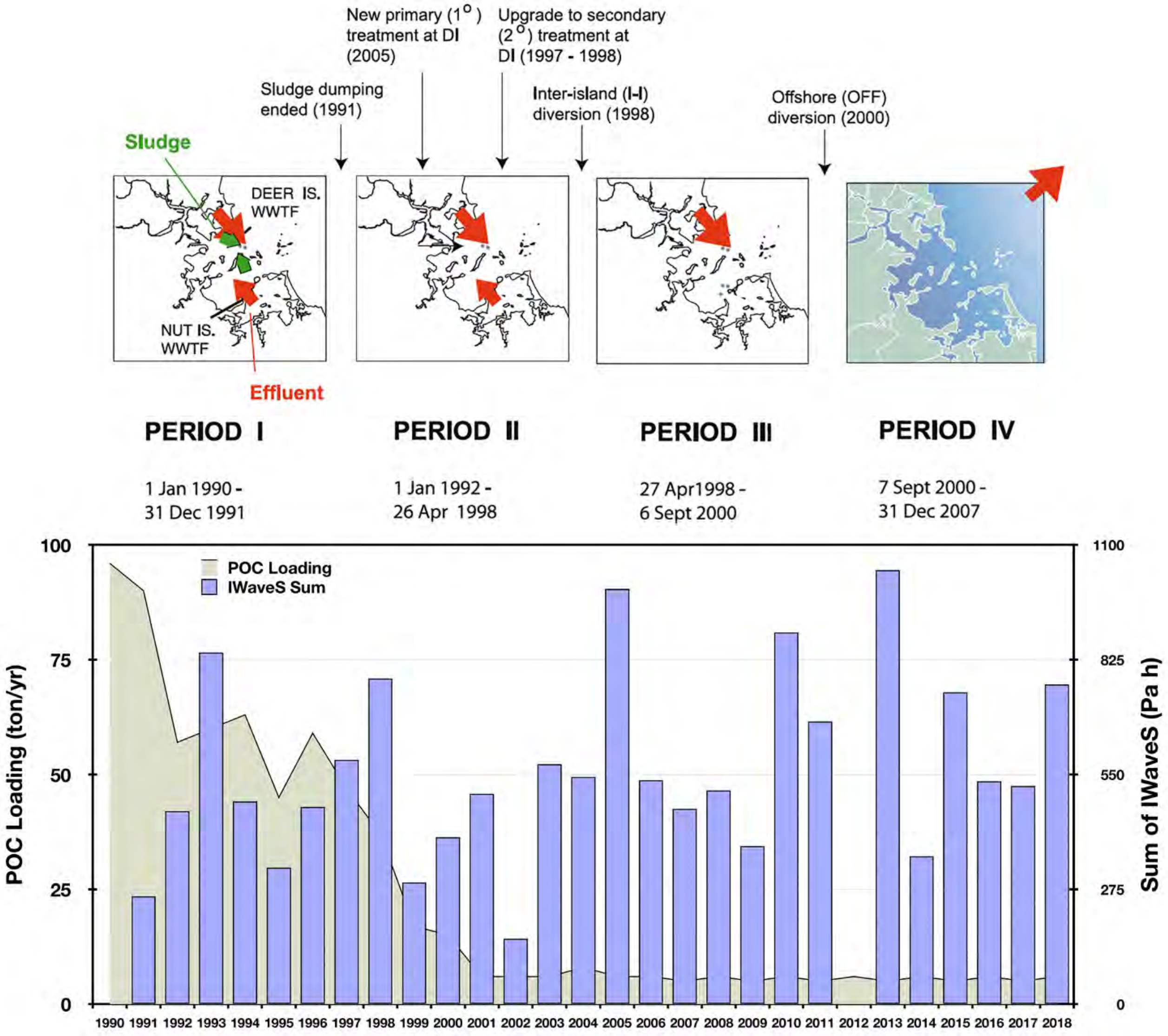


Figure 3-51. Change in POC loading based on Taylor (2010) loading periods and winter-period storm strength (sum IWaveS). POC loading from 2007 to 2018 are estimated base on 2003 to 2006 loading.

Table 3-15. Parameter estimates from logistic model of the odds of *Ampelisca* spp. tube mats being present with storm parameters, POC loading periods of Taylor (2010), and year.

Factor	Parameter	SE	z-value	p
Number of IWaveS Storms	-0.001	0.000	-3.38	0.001
Sum of IWaveS Storms	-0.037	0.018	-2.11	0.035
Months Since Last IWaveS Storm Before August	-0.021	0.224	-0.09	0.926
Period II to III	-1.241	0.256	-4.85	<0.001
Period II to IV	-0.018	0.016	-1.14	0.256
Year	0.050	0.021	2.44	0.015

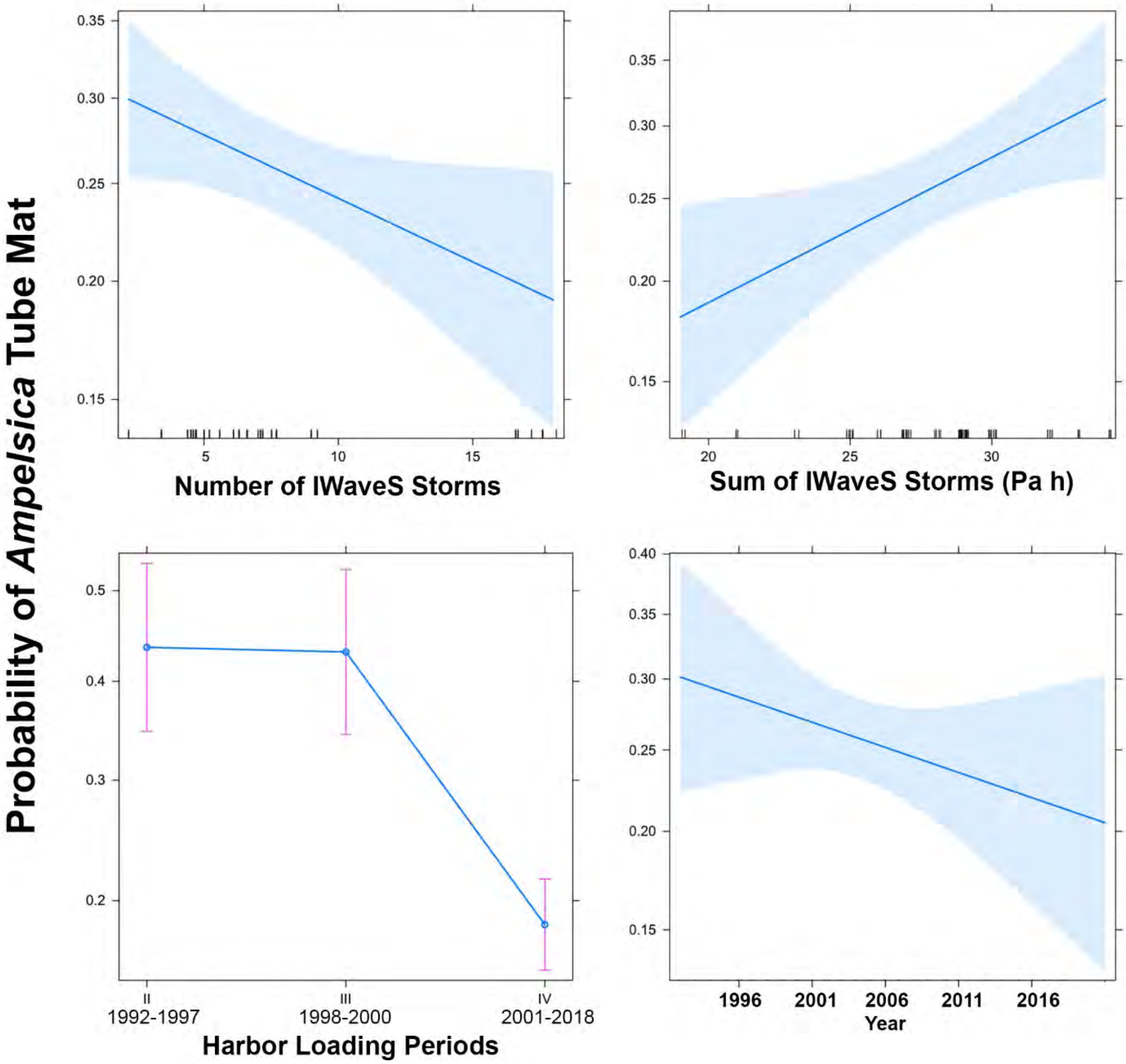
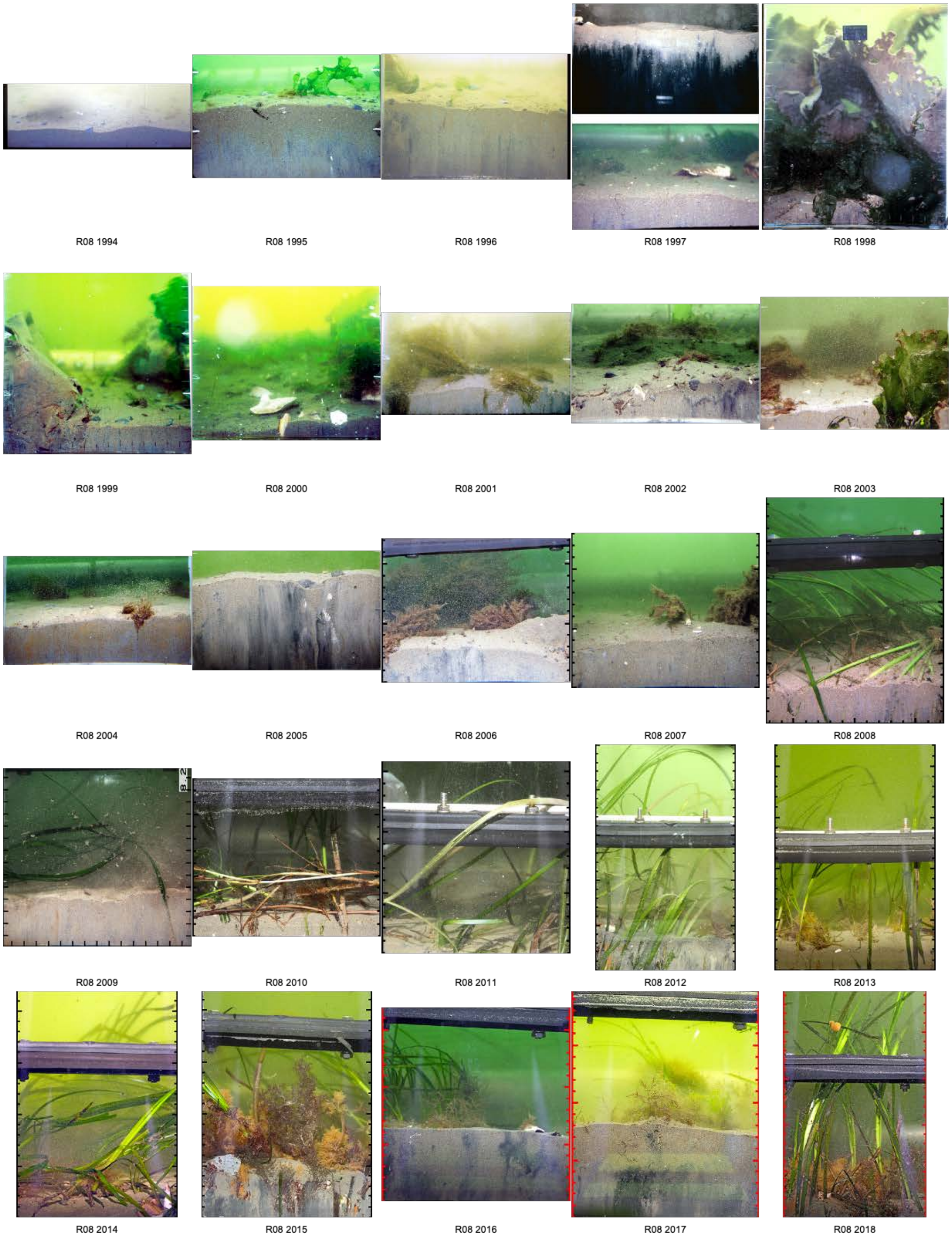


Figure 3-52. Predicted probability of *Ampelisca* spp. tube mats being present vs. absent based on logistic regression of storm parameters, POC loading periods, and year. Parameter estimates are in Table 3-15. Shaded area and whiskers are 95% confidence interval.



R08 1994 R08 1995 R08 1996 R08 1997 R08 1998

R08 1999 R08 2000 R08 2001 R08 2002 R08 2003

R08 2004 R08 2005 R08 2006 R08 2007 R08 2008

R08 2009 R08 2010 R08 2011 R08 2012 R08 2013

R08 2014 R08 2015 R08 2016 R08 2017 R08 2018

Figure 3-53. Eelgrass bed at Station R08 on Deer Island Flats. The bed first appeared in SPI in 2008. Prior to 2008, sediments were fine-sand with macroalgae. Scale on sediment profile image is in cm.

Summary

Since 1991 improvements in wastewater treatment and moving the outfall offshore did result in major improvements in benthic habitat quality within Boston Harbor by favoring processes that enhance bioturbation or the mixing of sediment by organisms, which has led to more aerobic sediment conditions, particularly for outer Harbor regions.

The biggest factors affecting benthic habitat quality in 2018 were dredging in the Deer Island Flats and President Roads areas, and winter-period storms. Station R02 was dredged just prior to August sampling resulting in poor benthic habitat quality (-4 OSI). It is expected that by the completion of dredging several more monitoring stations will be affected.

Since the start of monitoring in 1991, 2018 was the fourth stormiest year and likely the principal factor that put slight downward pressure on habitat quality across Harbor regions. The second strongest integrated wind-stress storm (IWindS) occurred in January, and was followed by the strongest integrated bottom-stress storm (IWaveS) and sixth strongest IWindS storm in February-March.

Over the monitoring period and superimposed on effects of wastewater treatment on benthic habitat conditions was the effect of storm events. Intensity and timing of winter period storms were related to change in sediments, *Ampelsica* spp. tube mat occurrence, and overall benthic habitat quality. The influence of storms was most pronounced after diversion (Period IV) when effects of high organic loading (Periods I and II) had diminished.

Benthic habitats within Boston Harbor appear to be sensitive to climate variability (long-term pattern of storm timing and intensity) and the impact of a changing climate, especially the increasing temperature trend documented in outer Harbor (Taylor 2018), may become more prominent with time.

The evaluations carried out between Harbor trends and storms strongly support a linkage. A review of these exploratory analyses identified alternative approaches, such as using more local winds to estimate IWindS or determining how well wave stress measured offshore translates into the Harbor, that may be addressed in future reporting.

4. CONCLUSION

Overall, the improvements in wastewater treatment and moving the outfall offshore have led to improvements in water quality and benthic habitat quality within Boston Harbor by reducing organic and nutrient loading and favoring processes that enhance bioturbation or the mixing of sediment by organisms. From 1990 to the present, the predominant origin of sediment organic matter in Boston Harbor has shifted from sewage to marine derived. Recovery of benthic communities has enhanced bioturbation rates by infaunal and epibenthic species and has led to more aerobic and 'healthier' sediment conditions by enhancing remineralization of legacy organics and nutrients stored in the sediments from years of sewage disposal. Physical and biological properties of the soft substrate in Boston Harbor in 2018 exhibited minor changes from recent years but were consistent with longer-term trends (Rutecki et al. 2019a). Concentrations of both TOC and *Clostridium perfringens*, indicators of organic enrichment and deposition from municipal wastewater discharge, have remained low at most stations compared to the levels occurring prior to the offshore diversion.

Community structure measures also continue to point to improving benthic conditions. Since municipal wastewater and sludge discharges have been eliminated from the Harbor, total abundance has trended downward, particularly for opportunistic species such as *Capitella capitata*. Species richness, while variable, has trended upward as did evenness (J'), Shannon diversity (H'), and log series alpha diversity with all measures appearing to reach static levels in recent years. Although abundances of individual taxa, such as the dominants *Ampelisca* spp., *Aricidea catherinae*, and *Polydora cornuta* have varied over time, the infaunal community structure and evidence from sediment profile imaging suggest that these changes relate more to normal physical disturbances than organic stress in the years since wastewater impacts to the Harbor have been reduced. The trends observed in previous years, including reduction of indicators of organic enrichment and increases in species diversity that persisted in the 2018 survey, are consistent with the recovery of benthic habitats from decades of inadequately treated sewage discharges.

5. REFERENCES

Agresti A. 1990. Categorical data analysis. New York: Wiley. 558 p.

Blake JA, Maciolek NJ, Rosen P. 1989. Benthic infaunal communities of Boston Harbor. Report to Massachusetts Water Resources Authority, Boston, MA. 25 pp.

Butman B, Sherwood CR, Dalyander PS. 2008. Northeast storms ranked by wind stress and wave-generated bottom stress observed in Massachusetts Bay, 1990–2006. Continental Shelf Research 28:1231-1245.

Chang S, Steimle FW, Reid RN, Fromm SA, Zdanowicz VS, Pikanowski RA. 1992. Association of benthic macrofauna with habitat types and quality in the New York Bight. Marine Ecology Progress Series 89:237-251.

Clarke KR. 1993. Non-parametric multivariate analyses of changes in community structure. Aust. J. Ecol., 18:117-143.

Clarke KR, Green RH. 1988. Statistical design and analysis for a ‘biological effects’ study. Mar. Ecol. Prog. Ser., 46:213-226.

Constantino J, Leo W, Delaney MF, Epelman P, Rhode S. 2014. Quality Assurance Project Plan (QAPP) for Sediment Chemistry Analyses for Harbor and Outfall Monitoring, Revision 4 (February 2014). Boston: Massachusetts Water Resources Authority. Report 2014-02. 53 p.

Costello CT, Kenworthy JD. 2011. Twelve-year mapping and change analysis of eelgrass (Zostera marina) areal abundance in Massachusetts (USA) identifies statewide declines. Estuaries and Coasts 34:232-242.

Diaz RJ, Rhoads DC, Blake JA, Kropp RK, Keay KE. 2008. Long-term trends in benthic habitats related to reduction in wastewater discharges to Boston Harbor. Estuaries and Coasts 31:1184–1197.

Evans T, Carr J, Frew K, Rousseau M, Ford K, Boeri A. 2018. Massachusetts Division of Marine Fisheries 2010 to 2016 HubLine eelgrass restoration. Final Report. Submitted to: Massachusetts Department of Environmental Protection. 25 p. plus appendices.

Gallagher ED, Keay KE. 1998. Organisms-Sediment-Contaminant Interactions in Boston Harbor. p. 89-132 In: *Contaminated Sediments in Boston Harbor*. K.D. Stolzenbach and EE. Adams (eds.) Marine Center for Coastal Processes, MIT Sea Grant College Program. Cambridge, MA 02139.

Giblin AE, Hopkinson CS, Tucker J. 1997. Benthic metabolism and nutrient cycling in Boston Harbor and Massachusetts Bay, Massachusetts. Estuaries 20:346–364.

Goberville E, Beaugrand G, Sautour B, Tréguer P, SOMLIT Team. 2010. Climate-driven changes in coastal marine systems of western Europe. Marine Ecology Progress Series 408:129-48.

Huettel M, Webster IT. 2001. Porewater flow in permeable sediments, pp. 144–179. In BP Boudreau and BB Jørgensen (eds.), The benthic boundary layer: Transport processes and biogeochemistry. Oxford Univ. Press.

Knebel, HJ. 1993. Sedimentary environments within a glaciated estuarine-inner shelf system: Boston Harbor and Massachusetts Bay. Marine Geology 110:7-30.

Knebel H, Circé RC. 1995. Seafloor environments within the Boston Harbor-Massachusetts Bay sedimentary system: A regional synthesis. Journal of Coastal Research 11:230–251.

Kubiak DA, Smoske NS, Coughlin K, Rando L, Wu D, Whittaker E. 2016. Combined Sewer Overflow Control Plan Annual Progress Report 2015. 2015 csoar-r4. 58 p.

Maciolek NJ, Dahlen DT, Diaz RJ. 2011. 2010 Boston Harbor Benthic Monitoring Report. Boston: Massachusetts Water Resources Authority. Report 2011-18. 20 p. plus appendices.

Maciolek NJ, Diaz RJ, Dahlen DT, Doner SA. 2008. 2007 Boston Harbor Benthic Monitoring Report. Boston: Massachusetts Water Resources Authority. Report ENQUAD 2008-22. 77 p. plus appendices.

Maciolek NJ, Diaz RJ, Dahlen DT, Williams IP. 2006. 2005 Boston Harbor Benthic Monitoring Report. Boston: Massachusetts Water Resources Authority. Report ENQUAD 2006-24. 87 p. plus appendices.

McCall PL. 1977. Community patterns and adaptive strategies of the infaunal benthos of Long Island Sound. Journal of Marine Research 35:221–266.

Mills EL. 1967. The biology of an ampeliscid amphipod crustacean sibling species pair. Journal of the Fisheries Research Board of Canada 24:305−355.

MWRA. 2010. Ambient monitoring plan for the Massachusetts Water Resources Authority effluent outfall revision 2. July 2010. Boston: Massachusetts Water Resources Authority. Report 2010-04. 107 p.

Nestler EC, Diaz RJ, Hecker B. 2018. Outfall benthic monitoring report: 2017 results. Boston: Massachusetts Water Resources Authority. Report 2018-xx. 57 p. plus appendices

Oviatt CA, Hyde KJW, Keller AA. 2007. Production patterns in Massachusetts Bay with outfall relocation. Estuaries and Coasts 30:35–46.

Pembroke AE, Diaz RJ, Nestler EC. 2013. Harbor Benthic Monitoring Report: 2012 Results. Boston: Massachusetts Water Resources Authority. Report 2013-13. 41 p.

Pembroke AE, Diaz RJ, Nestler EC. 2014. Harbor Benthic Monitoring Report: 2013 Results. Boston: Massachusetts Water Resources Authority. Report 2014-12. 43 p.

Pembroke AE, Diaz RJ, Nestler EC. 2015. Boston Harbor Benthic Monitoring Report: 2014 Results. Boston: Massachusetts Water Resources Authority. Report 2015-10. 37 p.

Pembroke AE, Diaz RJ, Nestler EC. 2016. Boston Harbor Benthic Monitoring Report: 2015 Results. Boston: Massachusetts Water Resources Authority. Report 2016-14. 38 p.

Pembroke AE, Diaz RJ, Nestler EC. 2017. Boston Harbor Benthic Monitoring Report: 2016 Results. Boston: Massachusetts Water Resources Authority. Report 2017-10. 45 p.

Rhoads DC, Germano JD. 1986. Interpreting long-term changes in benthic community structure: a new protocol. Hydrobiologia 142:291-308.

Rutecki D, Nestler E, Hasevlat R. 2017. Quality Assurance Project Plan for Benthic Monitoring 2017–2020. Boston: Massachusetts Water Resources Authority. Report 2017-06. 92 p. plus Appendices.

Rutecki DA, Nestler EC, Diaz RJ. 2019a. Boston Harbor benthic monitoring report: 2017 results. Boston: Massachusetts Water Resources Authority. Report 2019-01. 56 p.

Rutecki DA, Diaz RJ, Nestler EC, Codiga DL, Madray ME. 2019b. 2018 Outfall Benthic Monitoring Results. Boston: Massachusetts Water Resources Authority. Report 2019-06. 59 p., plus appendices.

SAIC. 1990. REMOTS® sediment-profile photography surveys of Boston Harbor, Dorchester, Quincy, Hingham, and Hull Bays: June 1989 and May 1990. SAIC Report No. SAIC-90/7578&236. Science Applications International Corporation, Woods Hole, MA 45 p.

SAIC. 1992. REMOTS® sediment-profile photography survey of Boston Harbor, Dorchester, Quincy, Hingham, and Hull Bays: May 1992. SAIC Report No. 266. Science Applications International Corporation, Newport, RI. 20 p.

Shull DH, Benoit JM, Wojcik C, Senning JR, 2009. Infaunal burrow ventilation and pore-water transport in muddy sediments. Estuarine, Coastal and Shelf Science 83:277-286.

Signell RP, Butman B. 1992. Modeling tidal exchange and dispersion in Boston Harbor. Journal of Geophysical Research 97:15591–16606.

Talke SA, Kemp AC, Woodruff J. 2018. Relative sea level, tides, and extreme water levels in Boston Harbor from 1825 to 2018. Journal of Geophysical Research: Oceans 123:3895-3914.

Taylor DI. 2006. Update of patterns of wastewater, river, and non-point source loadings to Boston Harbor (1990-2005). Boston: Massachusetts Water Resources Authority. Report 2006-22. 77p.

Taylor DI. 2010. The Boston Harbor Project, and large decreases in loadings of eutrophication-related materials to Boston Harbor. Marine Pollution Bulletin 60:609–619.

Taylor, DI. 2018. Boston Harbor Water Quality Update, 1995-2017. Boston: Massachusetts Water Resources Authority Report 2018-06. 16 p.

Taylor DI, Oviatt CA, Borkman DG. 2010. Non-linear responses of a coastal aquatic ecosystem to large decreases in nutrient and organic loadings. Estuaries and Coasts 34:745–757.

Taylor DI, Oviatt CA, Giblin AE, Tucker J, Diaz RJ, Keay K. 2019. Wastewater input reductions reverse historic hypereutrophication of Boston Harbor, USA. Ambio https://doi.org/10.1007/s13280-019-01174-1.

Thiel M. 1997. Reproductive biology of a filter-feeding amphipod, *Leptocheirus pinguis*, with extended parental care. Marine Biology 130:249-258.

Thiel M. 1999. Extended parental care in marine amphipods II. Maternal protection of juveniles from predation. Journal of Experimental Marine Biology and Ecology 234:235-253.

Thompson CEL, Williams ME, Amoudry L, Hull T, Reynolds S, Panton A, Fones GR. 2017. Benthic controls of resuspension in UK shelf seas: implications for resuspension frequency. Continental Shelf Research. (doi:10.1016/j.csr.2017.12.005).

Tucker J, Giblin AE, Hopkinson CS, Kelsey SW, Howes BL. 2014. Response of benthic metabolism and nutrient cycling to reductions in wastewater loading to Boston Harbor, USA. Estuarine, Coastal and Shelf Science 151:54-68.

Valente RM, Rhoads DC, Germano JD, Cabelli VJ. 1992. Mapping of benthic enrichment patterns in Narragansett Bay, Rhode Island. Estuaries 15:1-17.

Warwick RM. 1993. Environmental impact studies on marine communities: pragmatical considerations. Aust. J. Ecol., 18:63-80.

Wolfe DA, Long ER, Thursby GB. 1996. Sediment toxicity in the Hudson-Raritan Estuary: Distribution and correlations with chemical contamination. Estuaries 19:901–912.

Yingst JY, Rhoads DC. 1978. Seafloor stability in central Long Island Sound: Part II. Biological interactions and their potential importance for seafloor erodibility. pp. 245-260. In: Estuarine Interactions, Academic Press.

Young IR, Ribal A. 2019. Multiplatform evaluation of global trends in wind speed and wave height. Science 364:548-552.

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