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Dinoflagellate cyst records and human disturbance in two neighboring estuaries, New Bedford Harbor and Apponagansett Bay, Massachusetts (USA)

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Abstract

The dinoflagellate cyst records in sediments from New Bedford Harbor and Apponagansett Bay demonstrate sensitivity to environmental change caused by human activity in the watersheds over the last 500 years. Changes in the species richness, as well as absolute and relative abundance of dinoflagellate cyst taxa reflect recent periods of development around the estuaries. Cyst taxa sensitive to these changes include *Dubridinium* spp., *Polykrikos schwartzii*, *Lingulodinium machaerophorum*, *Operculodinium israelianum* and *Selenopemphix quanta*. The greatest changes in the dinoflagellate cyst record occur during the 20th century, when New Bedford Harbor was exposed to both toxic pollution and heavy nutrient loading from point and non-point sources. Apponagansett Bay was not subject to industrial pollution and nutrient enrichment has been lower (from non-point sources). In Apponagansett Bay there is an increase in the dinoflagellate cyst species richness while species richness first increased, then declined in New Bedford Harbor. During the same period, the total dinoflagellate cyst concentration in New Bedford Harbor fluctuated over a wide range. The decline of species richness and the large fluctuations in the total cyst abundances signal the intensified anthropogenic disturbance in the watershed, notably a high degree of eutrophication and toxic pollution.

Keywords: Buzzards Bay; Eutrophication; Heavy metals; Marine pollution; Nutrient loading; Organic carbon; PCBs; Species richness

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1. Introduction

As home to many of the world's major ports, estuaries have a significant economic importance. Not surprisingly, they often suffer from inputs of sewage and other contamination. Over history, estuarine watersheds have been subjected to veg-

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etation clearance, agricultural development, urbanization and industrialization, and their waters to dredging and other physical alterations. These factors have led to problematic changes in hydrological regimes and water quality.

One of the most acute problems in estuarine systems is nutrient over-enrichment, resulting in eutrophication. Human activity has accelerated nutrient enrichment during the last century with increased inputs of mostly nitrogen and phosphorus, stimulating the greater production and standing crops of phytoplankton. Comparative studies on Waquoit Bay, Massachusetts have shown that the size of the total dinoflagellate population increased with the level of nitrogen loading in an estuary (Evgenidou et al., 1999). High nitrogen loading, accompanied by other factors, such as water column stratification and high allochthonous organic matter loading, may lead to nuisance blooms of dinoflagellate species (Paerl, 1988). High levels of nutrient loading can also change the nature of the estuarine ecosystem as shifts in phytoplankton communities from diatoms to greater importance of nanoplankton (Eppley and Weiler, 1979). A decrease of species diversity (Cooper and Brush, 1993; Sullivan, 1999) also has been observed in highly eutrophic systems.

Within the last few decades a concern has emerged about eutrophication caused by continued human population growth and urbanization around North American estuaries (Neilson and Cronin, 1981; Nixon, 1995). However, the character and extent of eutrophication can vary from one estuary to another depending on the intensity of the anthropogenic activity within the watershed, the basic nutrient level of the system and the characteristics of the system itself.

Although there are many qualitative observations, long-term, empirical measurements (decades and beyond) of estuarine water quality are rarely available. Historical photographs can be used to document changes in indicators of eutrophication, such as the loss of eelgrass, *Zostera marina* (McCleland and Valiela, 1996), but are limited to the availability of the photographic record.

Paleoecological studies can provide critical information on the timing and magnitude of ecological change in estuaries caused by anthropogenic activity. Microfossils, those of phytoplankton in particular, are useful tools in paleoecological studies due to their ubiquity and abundance. For example, sedimentary records of diatoms showed changes in their populations coincident with human disturbance in Chesapeake Bay, particularly nutrient enrichment of the estuary (Brush and Davis, 1984; Cooper and Brush, 1993; Cooper, 1995).

Dinoflagellate cysts are an important group of microfossils with potential as biological indicators of the timing and degree of environmental change in estuaries. Approximately 13–16% of living dinoflagellates produce resting cysts during their life cycle (Head, 1996). The organic (dinosporin) nature of walls of many cysts makes them resistant to degradation (Fensome et al., 1993), providing a good fossil record. The assemblages of dinoflagellate cysts accumulated in sediments can encode information about dinoflagellate populations in the upper water column (Dale, 1976). Thus, the assemblage of dinoflagellate cysts reflects the ecology of living dinoflagellates, the latter influenced by environmental factors such as temperature, salinity, nutrients, diatom availability, and turbidity (Taylor and Pallingher, 1987). Approximately half of dinoflagellates are heterotrophic and half are autotrophic (Dale, 1996; Jacobson and Anderson, 1996). Heterotrophic dinoflagellates are mostly affected by the distribution of their preferred prey (diatoms), whereas the distribution of autotrophic species depends on the availability of light and dissolved nutrients.

Paleoecological studies of Norwegian fjords (Dale and Fjellså, 1994; Saetre et al., 1997; Thorsen and Dale, 1997; Dale et al., 1999; Dale, 2001) have examined dinoflagellate cysts as indicators of cultural eutrophication and industrial pollution, investigating the record from several fjords, which differ in the degree of industrial and nutrient pollution (for the most recent discussion, see Dale, 2001). Investigators concluded that increased concentrations of cysts were a consequence of the increase of dinoflagellate production due to anthropogenic nutrient enrichment, but suggested that where nutrient enrichment was accompanied by industrial pollution cyst production might actually decrease.

The Norwegian studies have provided evidence that some cyst taxa may show a positive response to cultural eutrophication. For example, in some fjords the number of *Lingulodinium machaerophorum* increased with increasing nutrient enrichment. In other fjords an increase of the relative proportion of cysts of the heterotrophic species *Selenopemphix quanta* (the paleontological name for the cyst of *Protoperidinium conicum*) also was reported. Thus Thorsen and Dale (1997) have proposed that the increase in the cysts of heterotrophic dinoflagellates may represent a eutrophication signal.

Matsuoka (1999, 2001) examined the cyst record over a period of increasing nutrient enrichment and industrial pollution in Tokyo Bay. He observed an increased number of cysts of heterotrophic dinoflagellates and an isolated peak in the abundance of *Gyrodinium instriatum* cysts that he assumed was caused by a bloom of this species of dinoflagellate. However, Matsuoka reported neither large increase in absolute cyst abundance nor an increase in *Lingulodinium machaerophorum* and *Selenopemphix quanta*, as suggested by researchers on Oslofjord (Dale and Fjellså, 1994; Dale et al., 1999).

Although studies in Japan and Norway show that the dinoflagellate cyst record reflects anthropogenic activity in an estuary, the response was encoded in different ways. Presently, there is no universally accepted cyst assemblage parameter or set of parameters that reflect nutrient over-enrichment and/or industrial pollution in an estuary. Because the primary mechanisms that influence phytoplankton production such as flushing rate, salinity and light regime, vary with estuarine hydrography, dinoflagellate cyst distributions may differ with the type of estuary [based on the classification of fjords, embayments, lagoons, river-dominated estuaries by Boynton et al. (1982)]. The signs of eutrophication encoded in dinoflagellate cyst records may vary with estuarine type but more research is needed to establish this.

The geographical application of dinoflagellate cyst studies is limited and there is no literature documenting the utility of dinoflagellate cysts as bio-indicators of anthropogenic changes in North American estuaries. We present the first such study for North American estuaries. Our research examines the sedimentary record of dinoflagellate cysts from two shallow embayments, New Bedford Harbor and Apponagansett Bay (Massachusetts), and compares it with detailed historical records of anthropogenic activity over the past 450 years. These embayments are located within 10 km of each other, minimizing climatic variation, but have distinct watersheds and different histories. Intensive urban and industrial development in the watershed of New Bedford Harbor created a number of stimulating (nutrient enrichment) and suppressing (toxic pollution) influences, which we expect to be reflected in dinoflagellate cyst records. Apponagansett Bay serves as our control site as it is subject to lower levels of nutrient loading and no industrial pollution. We compare our results to those from dinoflagellate cyst studies in estuaries outside North America to determine if there are common signals of eutrophication and industrial pollution and how they might vary with differences in estuarine hydrography.

2. Study area

2.1. History

New Bedford Harbor, also known as the Acushnet River estuary, and Apponagansett Bay are side embayments of the northwestern part of Buzzards Bay, Massachusetts (Fig. 1). Europeans first settled the estuarine watersheds approximately 350 years ago and from that time until the present day this region has been extensively exploited (Voyer et al., 2000).

Development and urbanization of the New Bedford area has had dramatic effects on the estuarine environment. New Bedford's history can be divided into four main periods (Pesch and Garber, 2001). The agricultural period spans the 1670s to the 1780s (Table 1). Forest clearance and land cultivation characterize this period. The second period is marked by the rise of the whaling industry and in the mid-19th century, whaling vessels from New Bedford Harbor numbered over 300 (Nelson et al., 1996). The boom in the textile industry, which began around the 1880s, marks the third period that lasted until the 1940s. This period

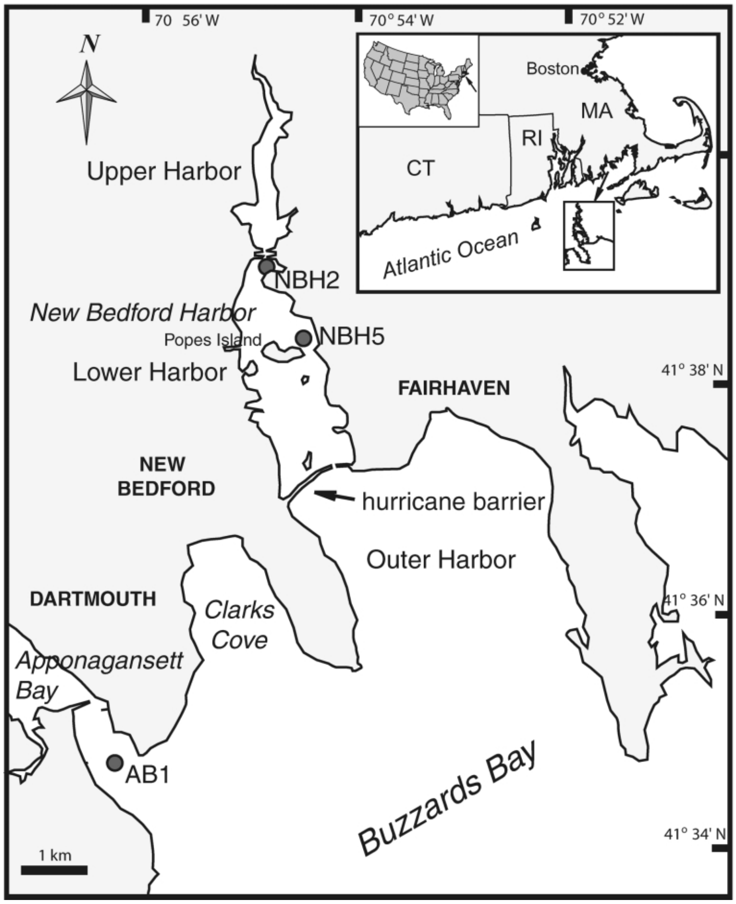

Fig. 1. Map of New Bedford Harbor and Apponagansett Bay showing core locations.

Table 1
Summary of assumed ecological effects of development around New Bedford Harbor (adopted from Pesch and Garber, 2001)

Time	Development	Consequence
Agricultural (1676–1780)	Cleared land, farmed	Minimal effect
Whaling (1750–1900)	Built wharfs New Bedford-Fairhaven Bridge Industries Cleared more land for building	Altered currents and sedimentation Altered currents and sedimentation Contaminated sediments in harbor Erosion, input sediment and nutrient to estuary
Textile (1880–1940)	Built mills on wetlands Dramatic population increase led to increased sewage input industries	Loss of habitat and filtering capability Increased organic matter, low species diversity, closed shellfish beds, Typhoid fever Contaminated sediments in barber
Post-textile (1940–present)	Electronics industries Other industries Hurricane barrier 1964	PCB contamination in harbor Contaminated sediments in harbor Altered circulation patterns

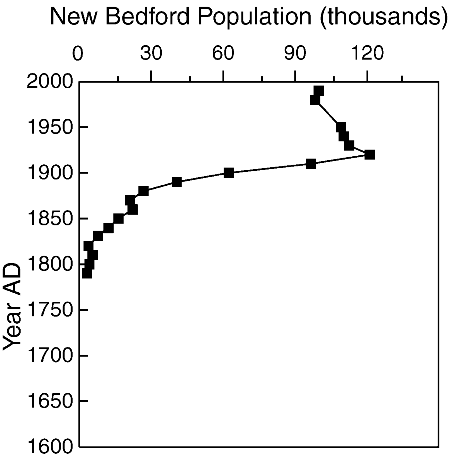

Fig. 2. Historical data on population grow in New Bedford Harbor (modified from Latimer et al., in preparation).

was related with a significant increase of the population in the watershed (Fig. 2). Municipal sewage from the City of New Bedford was directed into the harbor, and in the early 1900s sewage discharged into the harbor from ~30 sewage outfalls (Voyer et al., 2000). Voyer et al. (2000) propose that nutrient enrichment resulted in an increase of algal production to 320 g C m^{-2} in the 1910s, suggesting a shift in the trophic status of the estuary from mesotrophic to eutrophic. During the fourth, post-textile period, industry shifted towards production of electronic components and xenobiotic chemicals were discharged into the harbor (Summerhayes et al., 1977; Pruell et al., 1990). Finally, a hurricane barrier across the mouth of the harbor constructed in the 1960s, reduced flushing rates in the inner part of New Bedford Harbor by as much as 30% (Abdelrhman, in preparation), further degrading water quality. Eelgrass began disappearing during the 1960s and now is largely absent in New Bedford Harbor. Loss of eelgrass in estuaries is considered a signal of deteriorating water quality due to nutrient over-enrichment (Costa et al., 1996). In 1982, New Bedford Harbor was added to the Environmental Protection Agency (EPA) National Priority List for cleanup due to extreme levels of PCB sediment contamination (Nelson et al., 1996).

In contrast, Apponagansett Bay has experienced much lower levels of industrialization and population growth in its watershed. This bay can be considered as an example of an estuary affected mostly by nutrient enrichment, thus a control to which we contrast the impact of industrial pollution.

2.2. Modern and historical climatic conditions

The 30-year (1951–1980) average annual monthly air temperature at New Bedford is 11.3 °C (Environmental Data and Information Service, 1983). Mean monthly temperatures are above 0 °C 12 months of the year, but minimum temperatures are below 0 °C from December to February, with the lowest monthly mean in January (−4.0 °C).

Baron and Gordon (1985) used a combination of historical and instrumental records to reconstruct climate parameters in eastern Massachusetts over the period 1600–1980. They provide a reconstruction of winter air temperatures for 1742–1980, but the record is incomplete, missing the period 1780–1825. From 1742 to 1895 the average winter temperatures were below 0 °C. After this period winter temperatures were more variable, but remained above 0 °C for 19 years of their record. Baron and Gordon (1985) also reconstructed the length of the growing season in eastern Massachusetts for the years 1750–1980. Growing season length shows a different pattern than average winter temperatures. Periods of the longest growing seasons occurred from 1800 to 1830, 1865 to 1895, and 1945 to 1980.

2.3. Modern estuarine conditions

The Acushnet River flows into the northern part of New Bedford Harbor providing a small freshwater input (Nelson et al., 1996). Average salinity in the estuary ranges from 28 to 31 ppt and mean summer water temperature is approximately 21–23 °C (Howes et al., 1999). Water depth varies from 1 to 12 m with a tidal range of 1.2 m. New Bedford Harbor is highly industrialized and has the heaviest nitrogen-loading in Buzzards Bay (Howes et al., 1999). Modern sediment concentrations are as high as 100 μg g^{-1} PCBs, 1500 μg g^{-1} Cu, 600 μg g^{-1} Pb, and 1200 μg g^{-1} Zn (Latimer et al., in preparation). Mean summer concentrations of nitrate and phosphorus in harbor waters are 11 and 1.8 μM, respectively, resulting in average summer concentrations of chlorophyll *a* of 8.5 μg l^{-1} (ranging from 1.1 to 67.8 μg l^{-1}). The trophic status of New Bedford Harbor is considered to be eutrophic to hypertrophic (Howes et al., 1999).

As Apponagansett Bay has not been subjected to industrial pollution sediment concentrations of xenobiotics are relatively low: 0.6 μg g^{-1} PCBs, 50 μg g^{-1} Cu, 30 μg g^{-1} Pb, and 100 μg g^{-1} Zn (Latimer et al., personnal communication). Absence of intense urbanization of its watershed means that nutrients come from non-point sources and concentrations in its waters are also lower. Nitrate and phosphorus concentrations are 5.2 and 1.7 μM, respectively, resulting in 3.5 μg l^{-1} of chlorophyll *a*. Nitrate and chlorophyll *a* concentrations are less than half that recorded in New Bedford Harbor and the trophic status of Apponagansett Bay is considered to be only mesotrophic to eutrophic (Howes et al., 1999).

3. Materials and methods

Two sediment cores were collected in the fall of 1996 and one in the summer of 1998. Cores were collected at sites not affected by dredging or reclamation work. Two hand piston cores, NBH5 and NBH2, were obtained from the lower part of New Bedford Harbor in water depths of 2.3 and 3 m, respectively (Fig. 1). One gravity core, AB1, was collected from neighboring Apponagansett Bay, where the water depth was 4.6 m (Fig. 1). Sediments in all three cores are generally characterized as silt and sandy silt (Table 2) reflecting the generally coarse-grained character of the watershed's soils (Roffinoli and Fletcher, 1981). The mean grain size is shown in Figs. 3–5.

Sediments were dated using radiometric (^{210}Pb and ^{137}Cs) and palynological methods. Radiometric dating of two cores, AB1 (Apponagansett Bay) and NBH2 (New Bedford Harbor), was performed at the University of Liverpool's Environmental Radioactivity Centre, and core NBH5 (New Bedford Harbor) by OceanChem, Narragansett, Rhode Island. The palynological analysis was performed in the Paleoenvironmental Laboratory, Department of Geography, McGill University. Sedimentary concentrations of heavy metals and PCBs as well as the organic carbon content and δ^{13}C of sediments were measured by EPA and are reported by Latimer et al. (in preparation).

Samples for dinoflagellate cyst analyses were taken at 10-cm intervals, and at additional depths where required. Sampling intervals correspond to 4–40 years of deposition, with an average of 15 years during the 20th century.

Samples were treated according to a standardized protocol: dried at 40 °C, weighed, sieved through a 125-μm and retained on a 10-μm mesh to eliminate coarse and fine material, treated with warm HF (40%) to dissolve silicates, and HCl (10%) to remove carbonates and silicoflourides. Calibrated tablets of *Lycopodium* spores (Stockmarr, 1977), added during processing, allowed for calculation of dinoflagellate cyst concentrations based on the dry weight of sediments.

Aliquots were mounted on microscope slides with glycerin jelly. Identification of dinoflagellate cysts was made on the basis of published descriptions in accordance with taxonomy given in Lentin and Williams (1993). However, when species-level identification was not possible, identification was done at the genus level. There are different taxonomic systematics for cysts and thecal stages of dinoflagellates because paleontological studies of cysts originally were carried out independently from biological studies of the motile forms. Here we use the paleontological nomenclature according to Head (1996), Head et al. (2001), and Pospelova and Head (2002). A list of the dinoflagellate cysts counted and their known biological name or thecal equivalent is provided in Table 3.

Temporal changes in dinoflagellate cyst assemblages were analyzed by determining the total cyst concentrations, total cyst fluxes (or cyst accumulation rates), species richness, the proportions of cysts of heterotrophic and autotrophic dinoflagellates, as well as certain species-indicators. Species richness (or number of taxa) was preferred over other diversity indices as it has been found to be the most sensitive indicator of the response of

Table 2
General sediment characteristics, as well as dinoflagellate cyst total counts, relative abundance, and species richness in cores from Apponagansett Bay (AB1) and New Bedford Harbor (NBH2 and NBH5). Asterisk denote taxa not counted for cyst richness, including the freshwater *Protoperidinium wisconsinense*.

Core depth (cm)	Date (year AD)	Sediment color	Lithology	Numbers of counted cysts	Species richness	*Alexandrium tamarense*	*Ataxiodinium choane*	*Brigantedinium cariacoense*	*Brigantedinium simplex*	*Brigantedinium* spp.	*Dubridinium* spp.	*Impagidinium* spp.	*Islandinium brevispinosum*	*Islandinium? cezare*	*Islandinium minutum*	*Lejeunacysta sabrina*	*Lingulodinium machaerophorum*	*Nematosphaeropsis* spp.	*Operculodinium centrocarpum* sensu Wall & Dale 1966	*Operculodinium centrocarpum* var. *truncatum*	*Operculodinium israelianum*	*Pentapharsodinium dalei*	*Polykrikos schwartzii* & *P. kofoidii*	*Protoperidinium oblongum*	*Protoperidinium type*	*Protoperidinium wisconsinense**	*Quinquecuspis concreta*	*Selenopemphix nephroides*	*Selenopemphix quanta*	*Spiniferites* cf. *bentorii*	*Spiniferites elongatus*	*Spiniferites* spp.	*Stelladinium stellatum*	*Tectatodinium pellitum*	*Votadinium calvum*	*Votadinium spinosum*	Cyst type E	Cyst type X	Unknown cysts*
Core AB1																																							
0-1	1993	brownish-black	sandy silt	309	23	0.0	0.0	0.0	1.6	1.9	2.3	0.3	5.5	1.3	2.6	0.3	0.3	1.3	33.3	0.3	1.0	1.6	5.8	0.0	4.5	0.0	0.3	0.6	0.3	1.9	2.3	28.5	0.3	0.0	0.0	0.0	0.0	0.0	1.6
5-6	1976	brownish-black	sandy silt	220	21	0.0	0.0	0.0	0.9	0.5	6.4	0.5	1.8	1.4	2.3	0.0	0.5	0.0	30.0	0.5	0.5	0.9	2.7	0.0	8.2	0.0	0.0	0.0	1.4	3.2	1.8	34.1	0.5	0.0	0.0	0.9	0.0	0.5	0.9
10-11	1957	brownish-black	sandy silt	267	20	0.0	0.0	0.0	1.5	3.4	6.0	0.0	0.0	1.5	3.4	0.4	0.0	0.7	18.4	0.4	1.9	1.1	0.4	0.0	13.1	0.0	1.1	0.0	0.4	4.1	2.2	37.1	0.7	0.7	0.0	0.0	0.0	0.0	1.5
14-15	1939	olive gray	sandy silt	241	19	0.0	0.0	0.0	0.4	0.8	8.7	0.0	1.2	0.4	1.7	0.0	0.0	0.4	30.3	0.8	1.2	0.4	0.8	0.0	7.5	0.0	0.4	0.0	0.0	4.6	2.9	34.9	0.4	0.0	0.0	0.4	0.0	0.0	1.7
20-21	1901	olive gray	sandy silt	308	19	0.0	0.0	0.0	0.6	1.9	10.4	0.0	1.9	0.3	0.3	0.0	0.3	0.3	32.5	0.3	1.3	0.0	0.0	0.0	6.8	0.3	0.3	0.0	0.3	6.2	1.6	32.1	0.0	0.3	0.3	0.0	0.0	0.0	1.3
30-31	1826	olive gray	sandy silt	354	19	0.0	0.0	0.0	1.7	0.6	4.0	0.6	0.0	0.0	0.8	0.3	1.1	0.8	43.2	0.3	0.6	1.1	0.0	0.0	7.3	0.0	0.6	0.6	0.3	2.3	2.3	30.2	0.0	0.0	0.0	0.0	0.0	0.0	1.4
40-41	1750	olive gray	sandy silt	321	19	0.0	0.0	0.3	0.6	1.2	3.1	0.0	0.9	0.3	0.3	0.0	0.0	0.6	40.2	0.3	0.0	1.6	0.0	0.0	9.3	0.0	0.3	0.3	1.2	2.2	0.9	35.2	0.3	0.0	0.0	0.0	0.0	0.0	0.6
50-51	1675	olive gray	sandy silt	329	17	0.0	0.0	0.0	0.6	0.6	2.1	0.6	0.0	0.0	0.6	0.3	0.9	1.8	33.4	0.3	3.0	2.1	0.0	0.0	6.1	0.0	0.3	0.0	1.2	0.0	0.0	45.3	0.0	0.0	0.0	0.0	0.3	0.0	0.3
60-61	1600	olive gray	sandy silt	392	19	0.0	0.0	0.0	1.0	2.3	4.3	0.3	0.0	0.3	0.3	0.5	1.8	1.3	34.7	0.0	2.0	1.0	0.0	0.0	8.9	0.0	0.5	0.0	2.8	1.3	0.8	33.9	0.3	0.0	0.0	0.0	0.0	0.0	1.8
70-71	1525	olive gray	sandy silt	286	17	0.0	0.0	0.0	0.0	2.4	3.1	0.0	0.0	0.0	1.0	1.4	1.4	0.3	30.4	0.3	1.4	1.4	0.0	0.0	11.2	0.0	0.0	0.3	2.1	1.4	0.3	39.5	0.3	0.0	0.0	0.0	0.0	0.0	1.4
75-76	1487	olive gray	sandy silt	329	20	0.0	0.0	0.0	0.3	0.0	1.8	0.0	0.6	0.3	0.3	0.3	0.9	0.3	39.8	0.3	3.0	1.2	0.0	0.0	3.6	0.0	0.9	0.0	2.1	1.5	0.9	40.4	0.3	0.0	0.3	0.0	0.0	0.0	0.6
Core NBH2																																							
1-2	1996	olive black	sandy silt	106	17	0.9	0.0	0.0	0.9	2.8	0.9	0.9	0.9	1.9	3.8	0.0	2.8	0.9	15.1	0.0	0.0	3.8	0.0	0.0	15.1	0.0	0.0	0.0	0.0	0.9	0.9	37.7	0.9	0.0	0.0	0.0	0.0	0.0	8.5
10-11	1992	olive black	sandy silt	209	15	0.0	0.0	0.5	1.0	5.7	7.2	0.0	0.0	0.5	2.4	0.0	0.5	2.4	12.9	0.0	0.0	0.5	0.5	0.0	14.4	0.5	0.0	0.0	0.0	1.4	1.4	45.5	0.0	0.0	0.0	0.0	0.0	0.0	2.9
20-21	1983	olive black	sandy silt	222	15	0.0	0.0	0.0	0.9	2.3	11.3	0.0	1.8	0.0	0.5	0.0	0.9	1.4	13.1	0.0	0.0	0.5	0.9	0.0	15.8	0.0	0.5	0.0	0.0	2.3	0.0	46.4	0.0	0.0	0.0	0.0	0.0	0.5	1.4
30-31	1972	olive black	sandy silt	221	15	0.0	0.0	0.0	1.4	4.1	23.5	0.0	0.0	1.8	1.8	0.0	1.4	1.4	6.8	0.5	0.0	0.0	1.8	0.0	15.8	0.0	0.5	0.0	0.0	2.3	1.4	33.5	0.0	0.0	0.0	0.0	0.0	0.0	2.3
40-41	1959	grayish black	sandy silt	217	15	0.0	0.0	0.0	0.5	3.2	14.7	0.0	1.4	0.9	0.0	0.0	0.0	2.3	9.2	0.0	0.0	0.9	4.1	0.0	15.2	0.0	2.8	0.0	0.0	3.2	0.5	38.7	0.0	0.5	0.0	0.0	0.0	0.0	1.8
Core NBH5																																							
0-1	1972	olive black	silt	149	15	0.7	0.0	0.0	0.0	2.0	7.4	0.0	2.7	0.0	0.7	0.0	0.7	2.0	6.7	0.7	0.0	3.4	0.0	0.0	8.1	0.0	0.0	0.0	0.0	4.0	2.7	55.0	0.0	0.0	0.0	0.0	2.7	0.0	0.7
5-6	1957	olive black	silt	210	15	0.0	0.0	0.0	0.0	1.4	3.3	0.0	0.5	0.5	1.9	0.0	0.5	2.4	14.8	0.0	0.0	2.4	0.0	0.0	9.0	0.0	0.0	0.0	0.0	3.3	0.5	55.2	1.0	0.0	0.0	0.0	0.5	0.0	2.9
10-11	1943	olive black	silt	385	16	0.3	0.0	0.0	0.3	0.8	1.0	0.3	0.3	0.8	1.6	0.0	0.3	1.0	7.0	0.0	0.0	1.8	0.0	0.0	3.4	0.0	0.0	0.0	0.0	0.8	0.3	79.7	0.0	0.0	0.0	0.0	0.0	0.0	0.5
15-16	1928	olive black	silt	235	15	0.0	0.0	0.0	0.9	0.4	0.9	0.4	0.0	0.4	0.9	0.0	1.3	0.4	10.6	0.4	0.0	0.9	0.0	0.0	5.1	0.0	0.0	0.0	0.0	2.1	0.9	71.5	0.0	0.0	0.0	0.0	0.0	0.0	3.0
20-21	1911	olive gray	silt	349	18	0.0	0.0	0.0	2.0	2.9	2.6	0.3	0.3	1.4	3.7	0.3	1.4	1.4	16.6	0.0	0.0	2.6	0.9	0.0	10.0	0.3	0.3	0.0	0.0	0.6	2.0	49.3	0.0	0.0	0.0	0.0	0.0	0.0	1.1
30-31	1877	olive gray	silt	305	21	0.0	0.0	0.0	3.3	4.3	3.0	0.3	1.0	0.7	3.3	1.0	0.7	2.6	15.4	0.0	0.0	2.3	0.0	0.3	6.9	0.3	0.0	0.0	0.3	1.3	0.3	50.8	0.3	0.3	0.0	0.3	0.0	0.0	1.0
40-41	1843	olive gray	silt	304	20	0.7	0.0	0.0	0.3	3.9	2.0	0.3	3.9	1.0	3.0	0.0	0.7	3.6	21.4	0.0	0.3	3.6	0.0	0.0	8.2	0.3	0.3	0.0	0.0	1.6	1.6	41.4	0.7	0.7	0.0	0.0	0.0	0.0	0.3
50-51	1808	olive gray	silt	345	22	0.0	0.0	0.0	0.3	4.9	4.6	0.6	3.8	0.3	2.6	0.6	1.4	2.9	14.8	0.3	0.0	1.7	0.0	0.0	9.3	0.0	0.3	0.0	0.6	1.7	0.6	47.0	0.6	0.0	0.0	0.3	0.3	0.0	0.6
60-61	1774	olive gray	silt	264	19	0.0	0.0	0.0	0.0	5.3	3.8	0.8	1.1	0.8	1.5	0.4	1.1	3.4	18.9	1.5	0.0	4.9	0.0	0.0	7.6	0.4	0.8	0.4	0.4	3.0	1.9	42.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
70-71	1740	olive gray	silt	344	22	0.0	0.0	0.0	2.9	1.7	4.4	0.3	0.9	0.0	0.6	0.6	1.7	3.2	11.6	0.3	0.6	1.5	0.6	0.0	5.5	0.0	2.0	0.3	0.0	0.6	0.3	56.7	0.6	1.2	0.0	0.0	0.0	0.0	2.0
80-81	1706	olive gray	silt	374	21	0.3	0.0	0.0	1.6	4.0	5.9	0.3	1.3	0.3	1.9	0.5	0.5	3.2	14.7	0.3	0.0	1.6	0.0	0.0	7.8	0.3	1.1	0.0	0.3	1.3	0.5	51.1	0.5	0.0	0.0	0.0	0.0	0.0	0.8
90-91	1672	olive gray	silt	375	18	0.0	0.0	0.0	0.5	2.4	5.1	0.5	0.3	0.0	0.5	0.8	1.3	2.7	19.2	0.0	0.3	1.3	0.0	0.0	6.1	0.8	0.8	0.0	1.1	1.3	0.8	54.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
100-101	1638	olive gray	silt	450	22	0.4	0.0	0.0	1.6	2.9	2.7	0.4	1.1	0.7	1.6	0.9	2.0	2.4	17.3	0.0	0.2	2.9	0.0	0.0	8.9	0.0	0.4	0.0	0.9	1.3	0.4	50.0	0.4	0.2	0.0	0.0	0.0	0.0	0.2
110-111	1605	olive gray	silt	341	19	0.0	0.0	0.0	0.3	2.3	2.3	0.0	0.3	0.0	0.3	1.5	4.4	5.3	16.1	0.3	0.0	1.8	0.0	0.0	5.3	0.9	0.3	0.0	0.9	0.6	0.6	53.1	1.8	0.0	0.0	0.0	0.3	0.0	1.5
120-121	1570	olive gray	silt	324	18	0.0	0.3	0.0	0.3	2.8	0.6	0.3	0.3	0.3	0.6	0.9	2.8	5.6	16.0	0.0	0.0	1.5	0.0	0.0	6.5	0.6	0.0	0.0	0.0	0.0	0.9	57.7	0.0	0.9	0.3	0.0	0.0	0.0	0.6

Table 3
Taxonomic citation of dinoflagellate cysts used in this study (*continued*)

	Cyst species (paleontological name)	Dinoflagellate thecate name or affinity (biological name)
Autotrophic		
Gonyaulacaceae	–	*Alexandrium tamarense*
	Ataxiodinium choane	*Gonyaulax spinifera camp*
	Impagidinium spp.	*?Gonyaulax* sp. *indet.*
	Linguladinium machaerophorum	*Lingulodinium polyedrum*
	Nematosphaeropsis spp.	*Gonyaulax spinifera complex*
	Operculodinium centrocarpum sensu Wall and Dale (1966)	*Protoceratium reticulatum*
	Operculodinium israelianum	*?Protoceratium reticulatum*
	Spinifierites bentorii	*Gonyaulax digitalis*
	Spiniferites elongatus	*Gonyaulax spinifera complex*
	Spinifierites spp.	*Gonyaulax spinifera complex*
	Tectatodinium pellitum	*Gonyaulax spinifera complex*
Protoperidiniaceae	–	*Pentapharsodinium dalei*
Heterotrophic		
Protoperidiniaceae	*Brigantedinium cariacoense*	*Protoperidinium avellanum*
	Brigantedinium simplex	*Protoperidinium conicoides*
	Brigantedinium spp.	*?Protoperidinium spp.*
	Islandinium brevispinosum[a]	*Protoperidinium* sp. *indet.*
	Islandinium? cezare[b]	*Protoperidinium* sp. *indet.*
	Islandinium minutum[b]	*Protoperidinium* sp. *indet.*
	Lejeunecysta sabrina	*Protaperidinium leonis*
	Protoperidinium oblongum	*Protoperidinium oblongum*
	Protoperidinium type	*Protoperidinium group*
	Protoperidinium wisconsinense	*Protoperidinium wisconsinense*
	Quinquecuspis concreta	*Protaperidinium leonis*
	Selenopemphix nephroides	*Protoperidinium subinerme*
	Selenapemphix quanta	*Protoperidinium conicum*
	Stelladinium stellatum	*Protoperidinium stellatum*
	Votadinium calvum	*Protoperidinium oblongum*
	Votadinium spinosun	*Protoperidinium claudicans*
Diplopsalidaceae	*Dubridinium* spp.	*Diplopsalid group*
Polykrikaceae	–	*Polykrikos schwartzii*
	–	*Polykrikos kofoidii*

Thecal equivalents are taken from Head (1996).

[a] Thecal equivalents are taken from Pospelova and Head (2002)

[b] Thecal equivalents are taken from Head et al. (2001).

phytoplankton to changes in aquatic ecosystems induced by nutrient enrichment, pollution or environmental stress (Sommer, 1995; Tsirtsis and Karydis, 1998).

An average of 293 cysts and a minimum of 106 were counted in each sample. More than 34 dinoflagellate cyst taxa were identified and counted in sediments from the three cores (Tables 2 and 4). We observed no reworked pre-Quaternary dinoflagellate cysts or pollen in this study. Microphotographs of selected taxa are shown in Fig. 6.

Total cyst flux, or accumulation rate of cysts (cysts cm^{-2} $year^{-1}$), is the product of the total cyst concentration (cysts g^{-1}), the sedimentation rate (cm $year^{-1}$), and the dry bulk density (g cm^{-3}). Since it accounts for variable sedimentation rates, cyst flux is the preferable indicator of changes in cyst production (Dale, 2001). The total cyst fluxes were calculated for the upper parts of cores AB1, NBH5 and for the whole section of NBH2, that is for the sediment samples deposited during the 20th century where the sedimentation

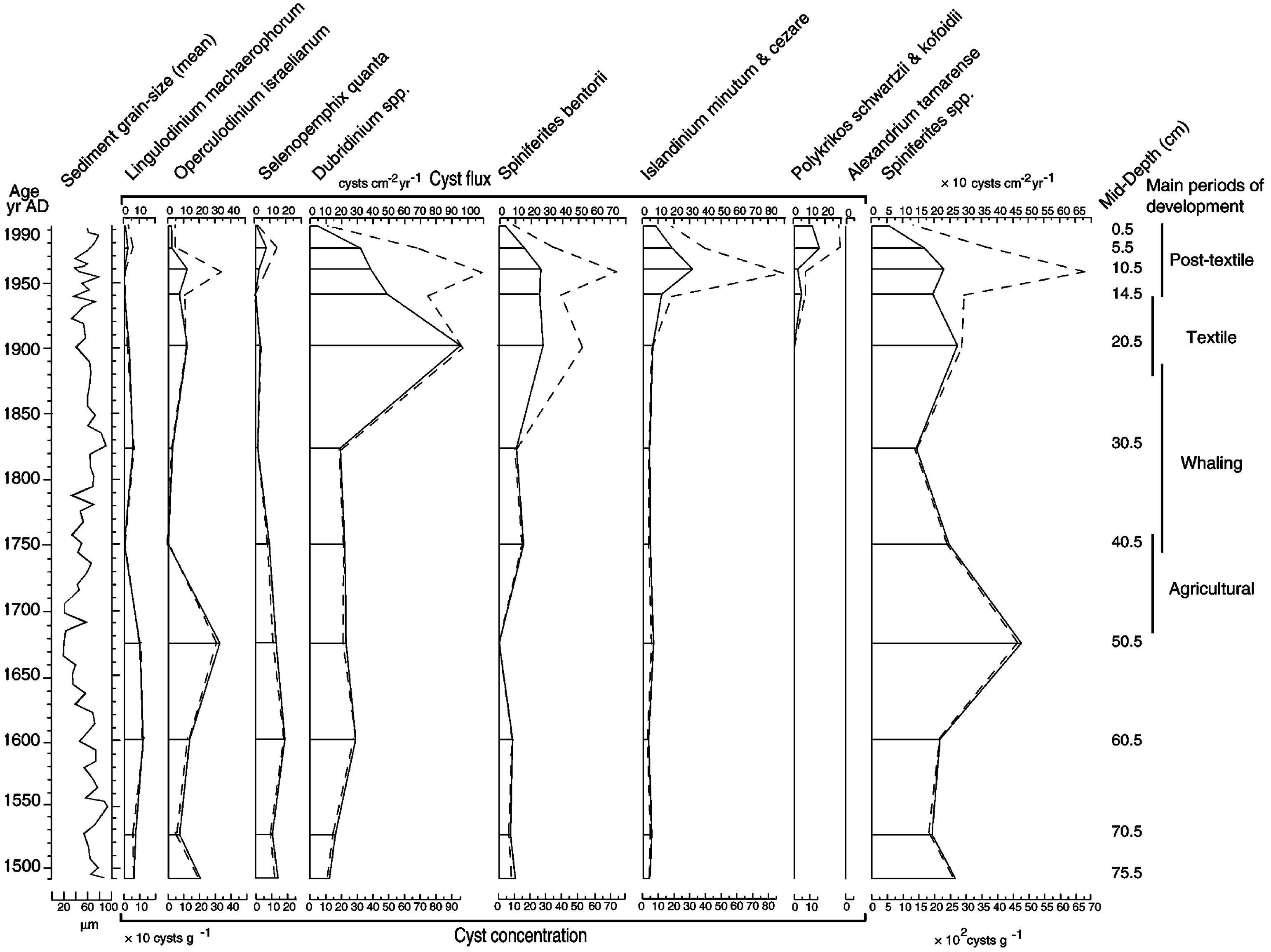

Fig. 3. Concentrations and estimated fluxes of selected dinoflagellate cyst taxa in core AB1, Apponagansett Bay.

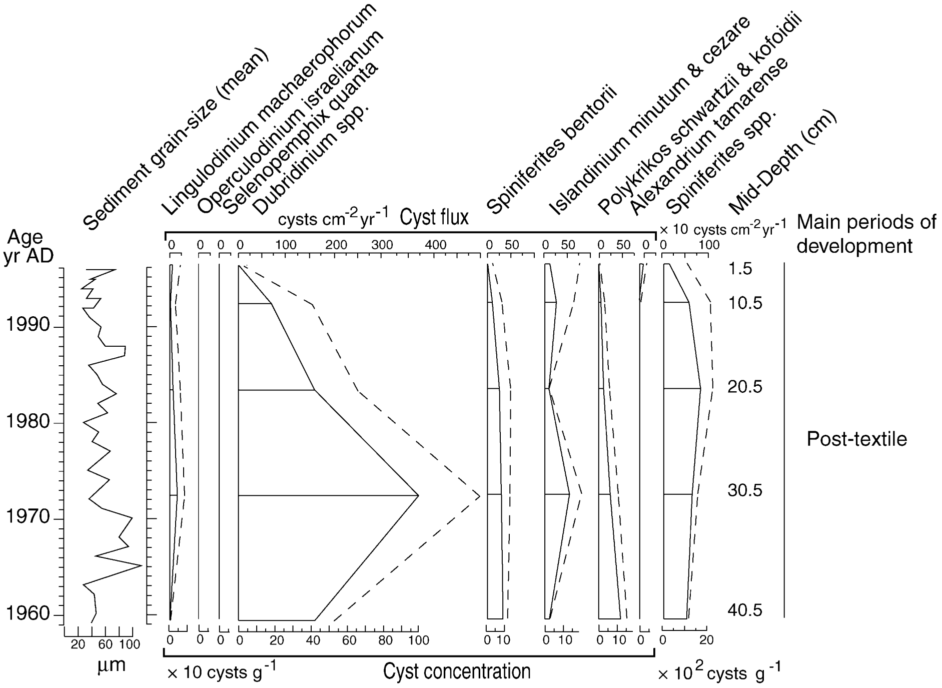

Fig. 4. Concentrations and estimated fluxes of selected dinoflagellate cyst taxa in core NBH2, New Bedford Harbor.

rates were determined directly from the radiometric dating. For lower parts of cores AB1 and NBH5 only approximate dinoflagellate cyst flux can be calculated. As bulk densities were not available for all samples, some values were estimated based upon the available data. Our minimum values of total dinoflagellate cyst flux are of the order of 100 times higher that those reported from oceanic sediments (e.g. Höll et al., 2000).

4. Results

4.1. Core chronologies

The combined results of the radiometric and palynological dating of the three cores are presented in Fig. 7. A detailed account of the dating will be reported elsewhere (Latimer et al., in preparation).

Core AB1, from Apponagansett Bay, is 0.76 m deep and represents more than 500 years of deposition. At 11.5±2 cm traces of ^{137}Cs signal the onset of fallout of radionuclide due to the nuclear weapon testing in 1954. The peak concentration of ^{137}Cs at 5.5±1.5 cm indicates the maximum fallout in 1963. The ^{210}Pb profile interpreted with the constant rate of supply (CRS) model (Appleby and Oldfield, 1978) dates the upper 16.5 cm of the core to 1931–1996. The sedimentation rate increases from ~0.23 cm year^{-1} in the 1930–1950s to 0.35 cm year^{-1} and greater in the following 20 years. Palynological analysis shows that ragweed pollen increases at ~50.5 cm indicating initial land clearance by European settlers (Brugam, 1978).

The sedimentation rate in core NBH2, from Lower New Bedford Harbor, was approximately three times higher than in Apponagansett Bay, and the upper 45 cm of this core represents approximately 40 years of deposition. The dating based on ^{137}Cs activity is in a broad agreement with the CRS interpretation of the ^{210}Pb profile. The sedimentation rate varies from 0.59 cm year^{-1} at the bottom of the core (1956) to 1.8 cm year^{-1} at the top.

The dating of core NBH5, from Lower New Bedford Harbor, was more problematic. The maximum of ^{137}Cs activity occurs within the top 6 cm

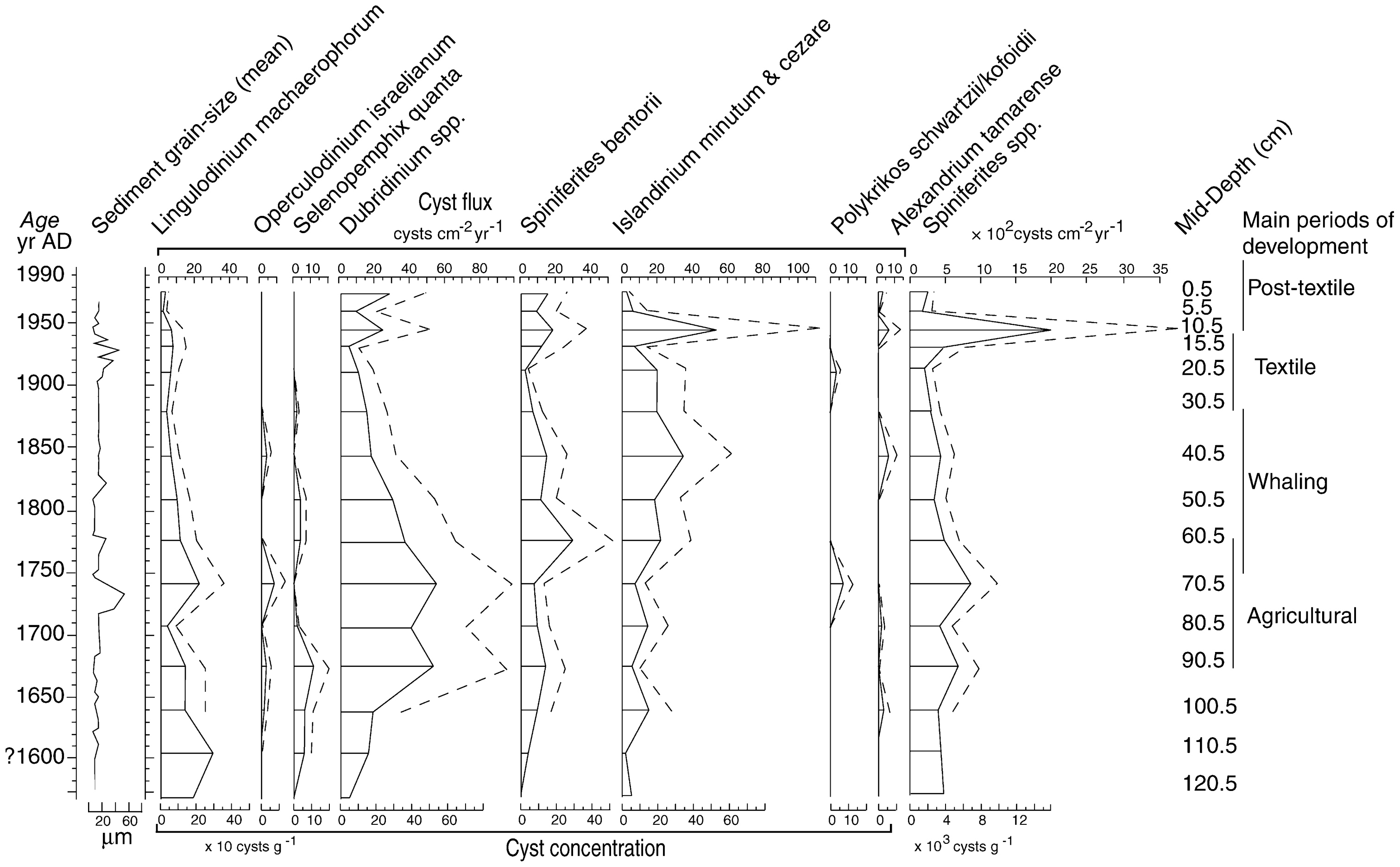

Fig. 5. Concentrations and estimated fluxes of selected dinoflagellate cyst taxa in core NBH5, New Bedford Harbor.

of the core. Together with sedimentation rates determined by the CRS ^{210}Pb method, it dates the top of the core as ~1973, suggesting that the part of this core was lost during core collection. The ^{210}Pb-dating is supported by PCB concentrations that first occur in trace amounts at 11 cm. This should correspond to the 1930–1940s, when PCBs were first synthesized. A mean sedimentation rate of 0.29 cm year^{-1} was extrapolated to the depth of 90.5 cm. This extrapolation was confirmed by pollen analysis. At 90–91 cm ragweed comprises $>1\%$ of the total pollen assemblages, corresponding to the time of the first European settlement in the area around the 1670s (Brugam, 1978). Dating below this point is uncertain.

4.2. Dinoflagellate cyst assemblages

4.2.1. Apponagansett Bay, Core AB1

The dinoflagellate cyst record in core AB1 generally can be divided into two stages. The first stage spans the 1480s to the 1900s. This stage is characterized by dinoflagellate cyst assemblages with $<22\%$ heterotrophic (Protoperidiniaceae, Diplopsalidaceae, and Polykrikaceae) taxa (Fig. 8), while the group of Diplopsalidaceae and Polykrikaceae comprises $<5\%$ of total assemblages. The total cyst concentration increases from 5120 cysts g^{-1} in the 16th century to a maximum of 10 774 cysts g^{-1} in the 1670s, then decreases to a minimum 4866 cysts g^{-1} at the end of 1820s (Fig. 9a). The estimated average cyst flux during this stage is ~650 cysts cm^{-2} year^{-1} (Fig. 9b). The number of dinoflagellate cyst taxa during the first stage ranges from 17 to 20 (Fig. 10). In the 17th century the maximum concentration and flux of all dinoflagellate cysts is coincident with increases in autotrophic *Operculodinium israelianum* and *Spiniferites* spp. (Fig. 3). The 17th century is also characterized by relatively high levels of the autotrophic *Lingulodinium machaerophorum* and heterotrophic *Selenopemphix quanta* (Fig. 3).

The second stage in Apponagansett Bay extends from 1900 to the present. The total cyst flux varies from ~500 to 1800 cysts cm^{-2} year^{-1} (Fig. 9b). Total concentrations of dinoflagellate cysts reach a maximum of 9204 cysts g^{-1} in ~1901, while total flux peaks in the 1950s (Fig. 9a).

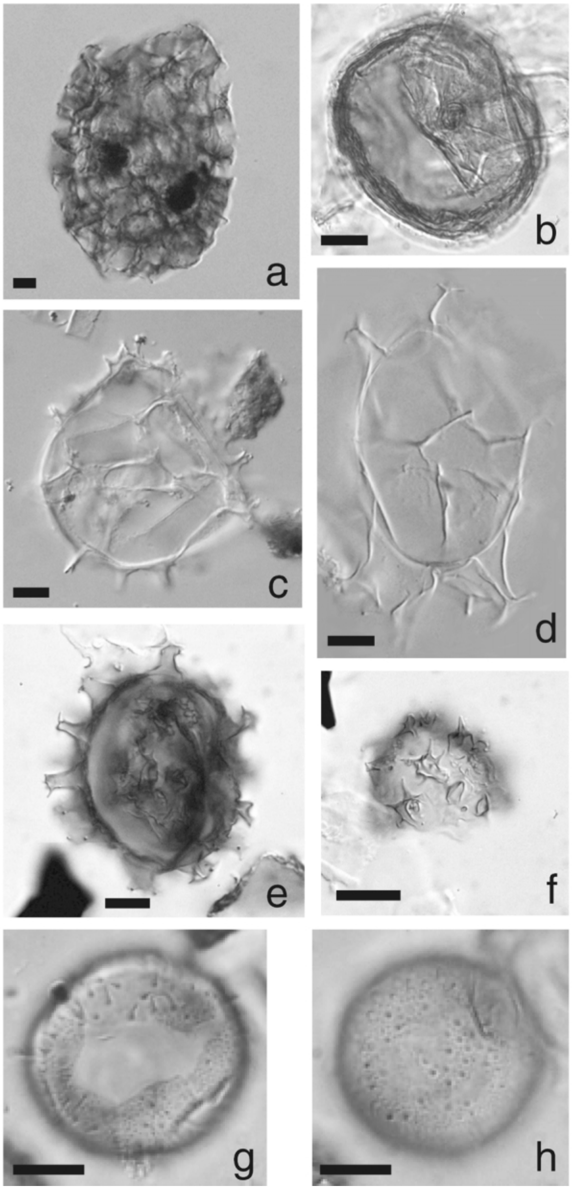

Fig. 6. Microphotographs of selected dinoflagellate cysts found in sediments of New Bedford Harbor and Apponagansett Bay: (a) *Polykrikos schwartzii*, (b) *Dubridinium* spp., (c) *Spiniferites bentorii*, (d) *Spiniferites elongatus*, (e,f) cyst type X, (g,h) cyst type E. Scale bar = 10 μm.

The 20th century assemblages are characterized by higher proportions of cysts of the heterotrophic taxa Diplopsalidaceae and Polykrikaceae (Fig. 8). In approximately 1901, concentrations of both

Table 4
Concentrations of dinoflagellate cysts (cysts per gram of dry sediments) in cores from Apponagansett Bay (AB1) and New Bedford Harbor (NBH5 and NBH2)

Core depth (cm)	Date (year AD)	Total cyst concentrations (cysts gr^{-1})	*Alexandrium tamarense*	*Ataxiodinium choane*	*Brigantedinium cariacoense*	*Brigantedinium simplex*	*Brigantedinium* spp.	*Dubridinium* spp.	*Impagidinium* spp.	*Islandinium brevispinosum*	*Islandinium? cezare*	*Islandinium minutum*	*Lejeunacysta sabrina*	*Lingulodinium machaerophorum*	*Nematosphaeropsis* spp.	*Operculodinium centrocarpum* sensu Wall & Dale 1966	*Operculodinium centrocarpum* var. *truncatum*	*Operculodinium israelianum*	*Pentapharsodinium dalei*	*Polykrikos schwartzii & P. kofoidii*	*Protoperidinium oblongum*	*Protoperidinium type*	*Protoperidinium wisconsinense**	*Quinquecuspis concreta*	*Selenopemphix nephroides*	*Selenopemphix quanta*	*Spiniferites* cf. *bentorii*	*Spiniferites elongatus*	*Spiniferites* spp.	*Stelladinium stellatum*	*Tectatodinium pellitium*	*Votadinium calvum*	*Votadinium spinosum*	Cyst type E	Cyst type X	Unknown cysts*
Core AB1																																				
0-1	1993	1973	0	0	0	32	38	45	6	109	26	51	6	6	26	658	6	19	32	115	0	89	0	6	13	6	38	45	562	6	0	0	0	0	0	32
5-6	1976	5051	0	0	0	46	23	321	23	92	69	115	0	23	0	1515	23	23	46	138	0	413	0	0	0	69	161	92	1722	23	0	0	46	0	23	46
10-11	1957	6427	0	0	0	96	217	385	0	0	96	217	24	0	48	1179	24	120	72	24	0	842	0	72	0	24	265	144	2383	48	48	0	0	0	0	96
14-15	1939	5627	0	0	0	23	47	490	0	70	23	93	0	0	23	1704	47	70	23	47	0	420	0	23	0	0	257	163	1961	23	0	0	23	0	0	93
20-21	1901	9204	0	0	0	60	179	956	0	179	30	30	0	30	30	2988	30	120	0	0	0	628	30	30	0	30	568	149	2959	0	30	30	0	0	0	120
30-31	1826	4866	0	0	0	82	27	192	27	0	0	41	14	55	41	2103	14	27	55	0	0	357	0	27	27	14	110	110	1471	0	0	0	0	0	0	69
40-41	1750	7102	0	0	22	44	88	221	0	66	22	22	0	0	44	2854	22	0	111	0	0	664	0	22	22	88	155	66	2500	22	0	0	0	0	0	44
50-51	1675	10774	0	0	0	65	65	229	65	0	0	65	33	98	196	3602	33	327	229	0	0	655	0	33	0	131	0	0	4879	0	0	0	0	33	0	33
60-61	1600	6681	0	0	0	68	153	290	17	0	17	17	34	119	85	2318	0	136	68	0	0	597	0	34	0	187	85	51	2267	17	0	0	0	0	0	119
70-71	1525	5120	0	0	0	0	125	161	0	0	0	54	72	72	18	1557	18	72	72	0	0	573	0	0	18	107	72	18	2023	18	0	0	0	0	0	72
75-76	1487	6720	0	0	0	20	0	123	0	41	20	20	20	61	20	2676	20	204	82	0	0	245	0	61	0	143	102	61	2717	20	0	20	0	0	0	41
Core NBH2																																				
1-2	1996	703	25	0	0	6	19	6	6	6	13	26	0	19	6	103	0	0	26	0	0	103	0	0	0	0	6	6	258	6	0	0	0	0	0	58
10-11	1992	2629	0	0	13	25	151	189	0	0	13	63	0	13	63	340	0	0	13	13	0	377	13	0	0	0	38	38	1195	0	0	0	0	0	0	75
20-21	1983	3787	0	0	0	34	85	426	0	68	0	17	0	34	51	495	0	0	17	34	0	597	0	17	0	0	85	0	1757	0	0	0	0	0	17	51
30-31	1972	4075	0	0	0	55	166	959	0	0	74	74	0	55	55	277	18	0	0	74	0	645	0	18	0	0	92	55	1364	0	0	0	0	0	0	92
40-41	1959	2924	0	0	0	13	94	431	0	40	27	0	0	0	67	270	0	0	27	121	0	445	0	81	0	0	94	13	1132	0	13	0	0	0	0	54
Core NBH5																																				
0-1	1972	3712	25	0	0	0	75	274	0	100	0	25	0	25	75	249	25	0	125	0	0	299	0	0	0	0	149	100	2043	0	0	0	0	100	0	25
5-6	1957	2572	0	0	0	0	37	86	0	12	12	49	0	12	61	380	0	0	61	0	0	233	0	0	0	0	86	12	1421	24	0	0	0	12	0	73
10-11	1943	22678	59	0	0	59	177	236	59	59	177	353	0	59	236	1590	0	0	412	0	0	766	0	0	0	0	177	59	18084	0	0	0	0	0	0	118
15-16	1928	5349	0	0	0	46	23	46	23	0	23	46	0	68	23	569	23	0	46	0	0	273	0	0	0	0	114	46	3824	0	0	0	0	0	0	159
20-21	1911	3828	0	0	0	77	110	99	11	11	55	143	11	55	55	636	0	0	99	33	0	384	11	11	0	0	22	77	1887	0	0	0	0	0	0	44
30-31	1877	4952	0	0	0	162	211	146	16	49	32	162	49	32	130	763	0	0	114	0	16	341	16	0	0	16	65	16	2516	16	16	0	16	0	0	49
40-41	1843	8680	57	0	0	29	343	171	29	343	86	257	0	57	314	1856	0	29	314	0	0	714	29	29	0	0	143	143	3598	57	57	0	0	0	0	29
50-51	1808	6312	0	0	0	18	310	292	36	237	18	164	36	91	182	930	18	0	109	0	0	584	0	18	0	36	109	36	2955	36	0	0	18	18	0	36
60-61	1774	9526	0	0	0	0	505	361	72	108	72	144	36	108	325	1804	144	0	469	0	0	722	36	72	36	36	289	180	4005	0	0	0	0	0	0	0
70-71	1740	12310	0	0	0	358	215	537	36	107	0	72	72	215	394	1431	36	72	179	72	0	680	0	250	36	0	72	36	6978	72	143	0	0	0	0	250
80-81	1706	6722	18	0	0	108	270	395	18	90	18	126	36	36	216	988	18	0	108	0	0	521	18	72	0	18	90	36	3433	36	0	0	0	0	0	54
90-91	1672	10269	0	0	0	55	246	520	55	27	0	55	82	137	274	1972	0	27	137	0	0	630	82	82	0	110	137	82	5559	0	0	0	0	0	0	0
100-101	1638	6769	30	0	0	105	196	181	30	75	45	105	60	135	165	1173	0	15	196	0	0	602	0	30	0	60	90	30	3385	30	15	0	0	0	0	15
110-111	1605	6640	0	0	0	19	156	156	0	19	0	19	97	292	350	1071	19	0	117	0	0	350	58	19	0	58	39	39	3524	117	0	0	0	19	0	97
120-121	1570	7112	0	22	0	22	198	44	22	22	22	44	66	198	395	1141	0	0	110	0	0	461	44	0	0	0	0	66	4105	0	66	22	0	0	0	44

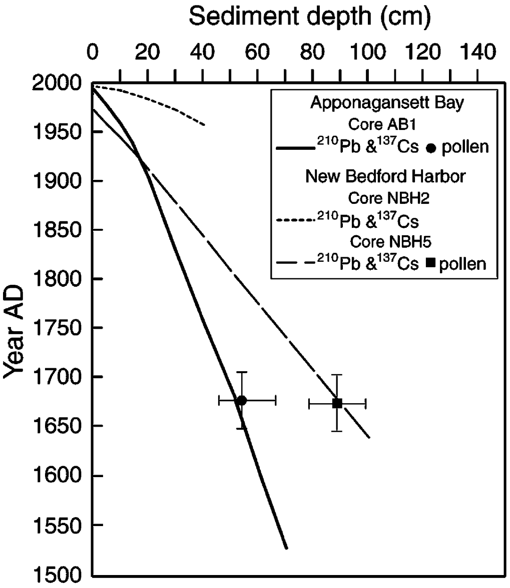

Fig. 7. Age/depth plots for sediments from Apponagansett Bay and New Bedford Harbor based on the radionuclide (^{210}Pb and ^{137}Cs) and palynological analyses.

Dubridinium spp. (956 cysts g^{-1}, 86 cysts cm^{-2} $year^{-1}$) and *Spiniferites bentorii* (568 cysts g^{-1}, 51 cysts cm^{-2} $year^{-1}$) reach a maximum (Fig. 3). As *Dubridinium* spp. and *Spiniferites bentorii* gradually decrease to the surface, cysts of Polykrikaceae appear for the first time in the record and persist to the surface. The abundances of *Islandinium minutum* & *cezare* are <65 cysts g^{-1} and 6 cysts cm^{-2} $year^{-1}$ until ~1901 and increases to 313 cysts g^{-1} or 88 cysts cm^{-2} $year^{-1}$ in the late 1950s. The second half of the 20th century is characterized by an increase in the number of dinoflagellate cyst taxa, from 19 to 23 (Fig. 10).

4.2.2. New Bedford Harbor, Core NBH2

The total concentration of dinoflagellate cysts ranges from 685 to 4075 cysts g^{-1} (Fig. 9a). The total flux of dinoflagellate cysts in this core ranges from ~1300 to 2200 cysts cm^{-2} $year^{-1}$ (Fig. 9b). Both parameters reach a maximum in the 1970s.

The main characteristics of core NBH2 are the low number of dinoflagellate cyst taxa in assemblages, which ranges from 15 to 17 (Fig. 10), and the high proportion (27–51%) of cysts of heterotrophic dinoflagellates (Fig. 8). The proportion of Diplopsalidaceae and Polykrikaceae ranges from 25% to 1%, and reaches its maximum in the early 1970s (Fig. 8). The maximum abundances of *Dubridinium* spp. (984 cysts g^{-1}, 499 cysts cm^{-2} $year^{-1}$) occur in the early 1970s (Fig. 4). The abundances of *Lingulodinium machaerophorum* are low (≤55 cysts g^{-1}, 37 cysts cm^{-2} $year^{-1}$) (Fig. 4). *Operculodinium israelianum* and *Selenopemphix quanta* were not observed in this core.

4.2.3. New Bedford Harbor, Core NBH5

The dinoflagellate cyst record in core NBH5 reflects changes in the history of the Harbor, but the most distinctive changes occur in the 20th century. In the 1940s both concentration and flux show an isolated peak (22 678 cysts g^{-1} and 4786 cysts cm^{-2} $year^{-1}$), approximately twice that of the next highest value. Otherwise, the total concentration of dinoflagellate cysts ranges from 12 310 to 2572 cysts g^{-1} and flux from 2219 to 601 cm^{-2} $year^{-1}$ (Fig. 9a). The species richness ranges from 18 to 22 until the 1880s, when it declines to 15 in the 1990s (Fig. 10). In the 16th century the maximum % of heterotrophic taxa is 15%, then ranges from 18 to 28% until the present, with an exception of the late 1920–1940s when relative abundance of cysts of heterotrophic dinoflagellates in assemblages is ~8% (Fig. 8). Higher abundances of *Dubridinium* spp. and *Lingulodinium machaerophorum*, as well as the presence of *Selenopemphix quanta* and *Operculodinium israelianum*, characterize the cyst record from the late 17th until the mid-19th century (Fig. 5). In contrast, abrupt, but distinctively brief increases in abundance of *Spiniferites* spp. (18 084 cysts g^{-1} and 3817 cysts cm^{-2} $year^{-1}$) and *Islandinium minutum* & *cezare* (530 cysts g^{-1}, 112 cysts cm^{-2} $year^{-1}$) occur in the early 1940s (Fig. 5). Cysts of *Operculodinium israelianum* and *Selenopemphix quanta* were not observed in the sediment record of the last century.

5. Discussion

5.1. General observations

Dinoflagellate cyst records show no detectable response to climate warming as indicated by winter

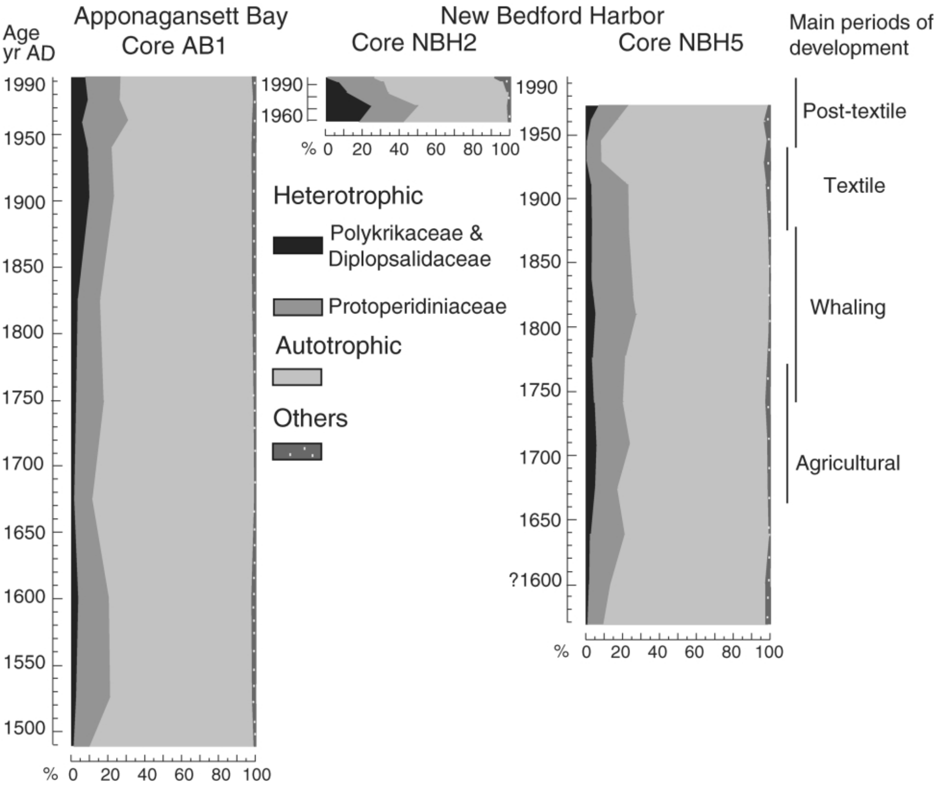

Fig. 8. Relative abundance (%) of heterotrophic (Protoperidiniaceae, Polykrikaceae and Diplopsalidaceae) and autotrophic taxa in dinoflagellate cyst assemblages in cores from Apponagansett Bay and New Bedford Harbor.

low temperatures or extended growing seasons reported by Baron and Gordon (1985). The greatest change occurs with the intense human activity of the 20th century. Our results support the conclusion of Pesch and Garber (2001) that the intensification of human activity and population growth in the watershed during the textile and post-textile periods caused the major degradation of New Bedford Harbor in the 20th century.

5.2. Total cyst concentrations and fluxes

Dale and Fjellså (1994), studying Norwegian fjords, suggested that an increase in total cyst concentration is an informative signal of eutrophication within an estuary. Our results (Fig. 9a) show no overall increase in total cyst concentration either in New Bedford Harbor or Apponagansett Bay. One could use the argument of Saetre et al. (1997) that industrial pollution can counterbalance the stimulating effects of nutrient enrichment so that the total cyst concentration remains constant or declines. We can exclude this explanation, as Apponagansett Bay was subject only to nutrient enrichment during the last century.

We suspect that increasing amplitude in the fluctuation of total cyst abundances may be a strong indicator of stressed environments. In the 16th–19th centuries, the total abundance of cysts in sediments from New Bedford Harbor varies by a factor of 2 (6000–12 000 cysts g^{-1}, 900–2200 cysts cm^{-2} $year^{-1}$) compared with a much larger range, by a factor of 9 (2600–23 000 cysts g^{-1}, 600–4800 cysts cm^{-2} $year^{-1}$) in the 20th century. This large magnitude of change in absolute cyst abundance occurs over a relatively short time interval. To some extent, this effect is caused by an abrupt increase of *Spiniferites* spp. around the

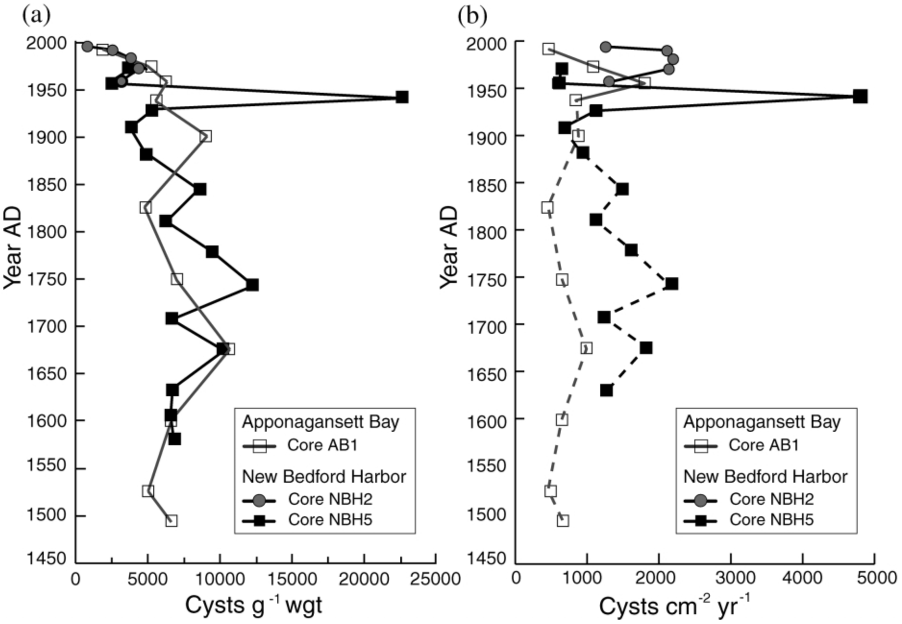

Fig. 9. Changes in the total dinoflagellate cyst concentrations (a) and fluxes (b) over time in cores from Apponagansett Bay and New Bedford Harbor. Dashed lines show the approximate dinoflagellate cyst fluxes in cores AB1 (gray) and NBH5 (black) prior to 20th century.

1940s and by the increase in the sedimentation rate during the past 70 years. In the 20th century New Bedford Harbor received the direct discharge of municipal sewage and organic pollutants (PAHs and PCBs) released from textile (during the third period) and electrical (during post-textile or the fourth period) industries (Latimer et al., in preparation). Changes in the production of dinoflagellate cysts in the 20th century in New Bedford Harbor is likely a response to this variety of new environmental influences that both suppress and stimulate dinoflagellate production. We assume that absence of large fluctuations in Apponagansett Bay total cyst abundances reflects lower environmental stress than in New Bedford Harbor.

5.2.1. Species richness

Although the lower level nutrient enrichment associated with early urbanization and population growth in New Bedford Harbor, and more recently in Apponagansett Bay, may have had positive effects on species richness, industrialization and final stages of urbanization had detrimental effects on species richness of dinoflagellates in New Bedford Harbor. Dinoflagellate cyst species richness in New Bedford Harbor is highest from the 17–19th centuries, when the system was shifting from an oligotrophic to mesotrophic condition. Through the same period, the species richness in Apponagansett Bay was generally lower than in New Bedford Harbor. However, during the 20th century, a period of population growth (and presumably, nutrient enrichment) around Apponagansett Bay, species richness increased from 19 to 23. In New Bedford Harbor this period was characterized by far greater population growth in addition to point source discharge of sewage and industrial pollutants. Species richness declined from 21 to 15. Such a magnitude of change (∼5 taxa) at this level of diversity is notable, as similar changes in phytoplankton species richness indicate major changes in trophic status (Tsirtsis and Karydis, 1998). The differences in timing of the responses in New Bedford Harbor and Apponagansett Bay allow us to exclude impacts of external factors such as climate change on the cyst record.

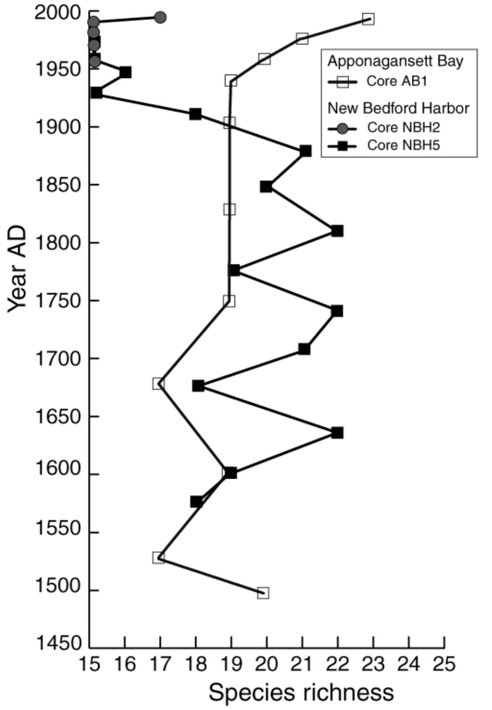

Fig. 10. Changes in the number of dinoflagellate cyst taxa over time in cores from Apponagansett Bay and New Bedford Harbor.

Our observations from Apponagansett Bay and the early history of New Bedford Harbor are consistent with those from studies reporting increasing species richness of estuarine phytoplankton subject to moderate nutrient enrichment, that is when the system shifts from oligotrophic to eutrophic conditions (Tsirtsis and Karydis, 1998). The record from the last century of New Bedford Harbor (decreasing cyst species richness) is consistent with those from highly eutrophic conditions. Studies of diatoms have shown that when the system is highly polluted with toxins or over-enriched with nutrients, diversity declines (Cooper and Brush, 1993; Sullivan, 1999). It seems appropriate to link the decline in dinoflagellate cyst species richness with the onset of hypertrophic and industrially polluted conditions in New Bedford Harbor in the 20th century.

We can compare this change in species richness, with two indicators of industrialization and urbanization, the latter being a major cause of eutrophication in an estuary. Heavy metals come from non-point as well as point sources of industrial and municipal sewage. As the toxicity of Cu to phytoplankton is well established (Taylor and Pallingher, 1987), we use its concentration in the sediments as a proxy for the stress of industrialization. Our second indicator is sedimentary organic carbon, which increases in both Apponagansett Bay and New Bedford Harbor after 1850. The carbon could come from increased pelagic production stimulated by nutrient inputs or directly from sewage outfalls in the case of New Bedford Harbor. (As the input of freshwater to these systems is minor it is unlikely that natural inputs from the watershed are detectable.) Both higher water column production and inputs of allochthonous carbon can be a stress by reducing light available to phytoplankton.

The source of sedimentary carbon can be deduced by examination of the δ^{13}C signature (e.g. Chmura and Aharon, 1995) which can indicate terrestrial (−26‰, PDB), saltmarsh (−16‰, PDB), eelgrass (−10‰, PDB), or marine (phytoplankton) sources (−20‰, PDB). We assume that human sewage carries the signal of terrestrial carbon. From approximately 1850 onwards there is a decline (from −18 to −24‰) in the δ^{13}C of sedimentary carbon in New Bedford Harbor (Fig. 11). The decline is probably due to the saltmarsh loss documented during this period, as well as the loss of eelgrass beds, presumed to have been once extensive in the Harbor. During the same period carbon concentration increases, probably due in part to increased phytoplankton production, but the low δ^{13}C indicates that terrestrial carbon sources are clearly increasing. The terrestrial component likely is from human sewage, a problem recognized throughout the history of the City (Pesch and Garber, 2001). The higher δ^{13}C (≥ -20‰) of sedimentary carbon in Apponagansett Bay indicates that terrestrial carbon sources are minor. (There are no municipal sewage outfalls in the Bay.) The minor decrease in the δ^{13}C over that last ~150 years probably reflects declines in eelgrass beds and increased inputs of carbon from phytoplankton production.

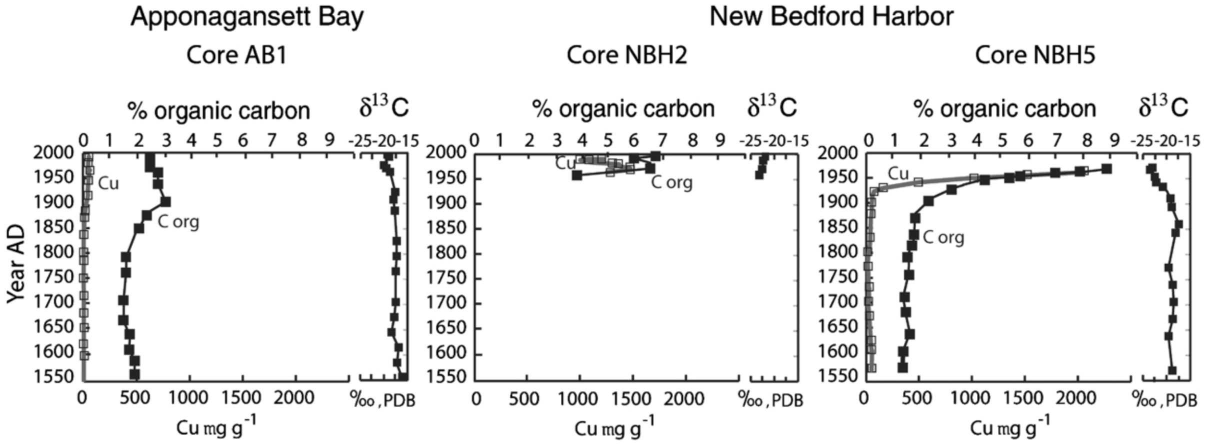

Fig. 11. Changes in % organic carbon, $\delta^{13}C$ and copper concentrations over time in Apponagansett Bay and New Bedford Harbor.

The timing of the greatest changes in dinoflagellate cyst assemblages, species richness, total cyst concentrations and flux in New Bedford Harbor coincide with the increase in the concentrations of sedimentary carbon and copper (Fig. 11). In fact the decline of cyst species richness can be predicted by the two parameters (Fig. 12a,b). Linear regression analysis shows that sedimentary carbon concentration alone explains 70% of the variability in species richness, copper 72%—and both together (as independent variables in a linear regression) explain 73% of the variability in cyst species richness.

A comparable study, where both parameters are available, is that from Frierfjord and Brevikfjord reported by Saetre et al. (1997). Extracting the species richness from their Appendix 1 and 2 and the organic carbon content and copper concentrations from Figs. 2 and 4, we observe a strikingly similar inverse relationships between both sedimentary carbon ($R=0.78$) and Cu ($R=0.73$) with species richness (Fig. 12c,d). Together these two variables explain 79% of the variability in cyst species richness of the Norwegian fjords.

5.3. Proportion of heterotrophic taxa

An increase in the proportion of cysts of heterotrophic dinoflagellates has been suggested (Saetre et al., 1997; Matsuoka, 1999, 2001) as a signal of eutrophication and industrial pollution. The cause of this increase is disputed. Matsuoka (1999) suggests increasing diatom production as the main cause, whereas Dale (2001) links the increase to the reduced production of autotrophic dinoflagellates due to reduced light penetration.

Our results show a small increase in the proportion of cysts of heterotrophic taxa (particularly the Polykridaceae and Diplopsalidaceae) as nutrient enrichment increased in the 20th century at our control site, Apponagansett Bay. However, in New Bedford Harbor where both nutrient and inorganic pollutants increased over the 19th and 20th century, the proportion of cysts of heterotrophic dinoflagellates is variable (Fig. 8).

5.4. Individual species-indicators

Individual taxa respond to water quality changes in different ways. The cysts of *Spiniferites* spp. dominate the cyst assemblages. This group seems to be tolerant of extreme environmental conditions as they are the most abundant in cyst assemblages from low salinity (5–15 ppt) environments (Pospelova and Chmura, 1998) as well as in highly eutrophic conditions in New Bedford Harbor. Abundances of subdominant taxa, such as cysts of *Dubridinium* spp., *Islandinium minutum* & *cezare*, and *Polykrikos schwartzii* produced by heterotrophic dinoflagellates, as well as the cysts of *Spini-*

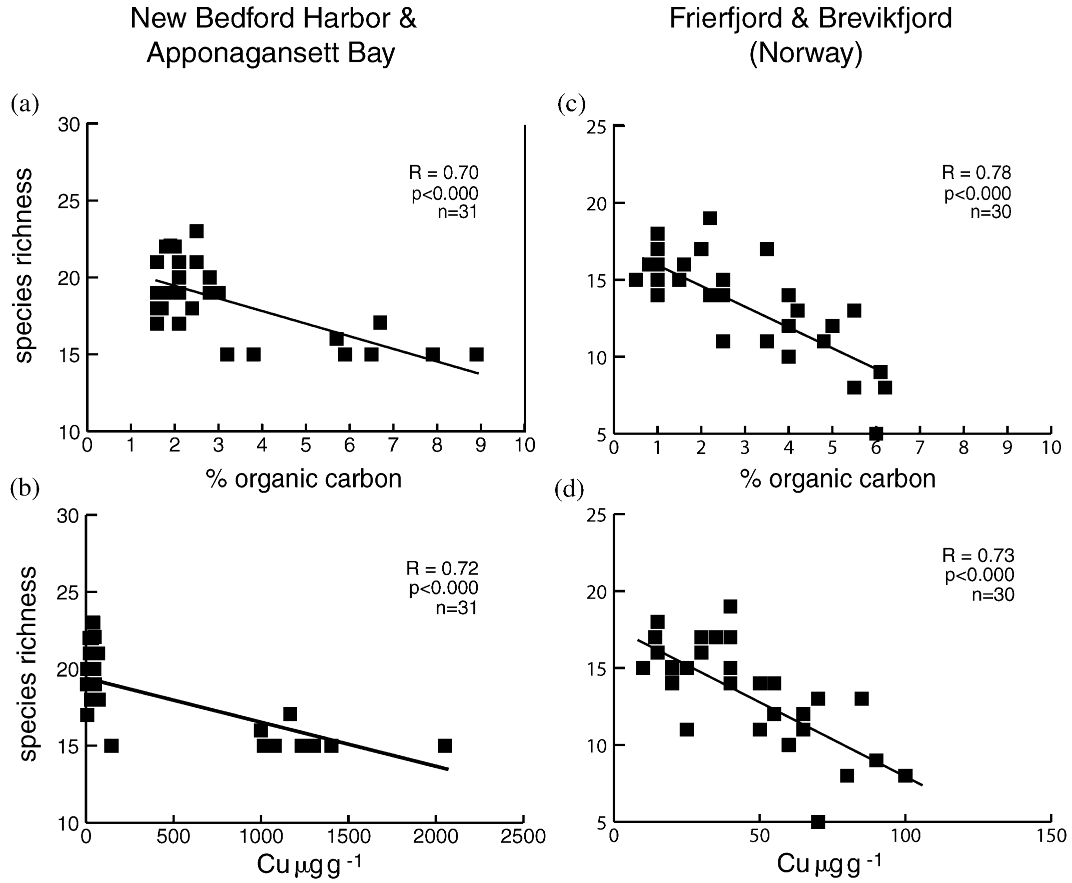

Fig. 12. Relationship between dinoflagellate cyst species richness and % organic carbon and copper concentrations based on samples from cores from Apponagansett Bay and New Bedford Harbor (a,b) and Norwegian data (c,d) published in Appendix 1 and 2 and Figs.1 and 2 by Saetre et al. (1997).

ferites bentorii, produced by autotrophic dinoflagellates, appear to increase with nutrient enrichment. In fact, the cysts of *Dubridinium* spp. and *Polykrikos schwartzii* are commonly recorded in nutrient-rich waters (Dale 1996; Matsuoka, personal communication). However, cysts of autotrophic dinoflagellates such as *Lingulodinium machaerophorum*, *Operculodinium israelianum* and heterotrophic *Selenopemphix quanta* tend to have negative response to eutrophication and inorganic pollution.

There are commonalties between species-indicators in the Massachusetts Bay and Tokyo Bay (Matsuoka, 1999, 2001). At Matsuoka's station 1 [his study site most subject to eutrophication Matsuoka (2001)] cysts of *Selenopemphix quanta* and *Lingulodinium machaerophorum* decrease over a period of increasing nutrient enrichment. The group of Diplopsalidaceae and Polykrikaceae increases in concentration during the same period, mostly because of increase in concentrations of *Polykrikos schwartzii* and *P.* cf. *kofoidii.*

The decreases in concentrations (and fluxes, in the case of Massachusetts estuaries) of *Lingulodinium machaerophorum* associated with nutrient enrichment in these shallow estuaries are not consistent with the observations from Norwegian fjords (Dale and Fjellså, 1994; Saetre et al., 1997; Dale et al., 1999). In the fjords *Lingulodinium machaerophorum* may not be responding to increases in nutrients, but to conditions associated with stratification and low oxygen concentrations characteristic of eutrophication in deep, stratified waters, such as Norwegian fjords. The shallow depths of Apponagansett Bay, New Bedford Harbor and Tokyo Bay mean they are generally unstratified and consequently do not suffer from severe oxygen depletion.

6. Conclusions

The study of dinoflagellate cysts from New Bedford Harbor and Apponagansett Bay over the last 500 years provides evidence that cyst assemblages contain a record of environmental changes induced by urbanization and industrial development. A number of parameters in the cyst record (species richness, concentrations, flux, and species abundances) can be used as indicators of environmental change.

Cyst species richness initially increases with nutrient level in an estuary, as observed in the 17–19th centuries in New Bedford Harbor and in 20th century in Apponagansett Bay. However, in environments with extremely high levels of nutrients (hypertrophic conditions) and inorganic toxins, this trend is reversed and the number of dinoflagellate cyst taxa declines, as seen in the 20th century in New Bedford Harbor.

Another parameter possibly indicating human disturbance in an estuary is the variability in total dinoflagellate cyst concentrations and fluxes. We find that it is not a unidirectional change, but rather a great fluctuation in the cyst production that characterizes heavily disturbed estuarine systems. These fluctuations are amplified by abrupt increases of certain taxa, such as *Spiniferites* spp. in the 1940s in New Bedford Harbor.

The effects of recent pollution are clearest when species richness, abundances, and composition of the dinoflagellate cyst assemblages are considered together. The major changes in the dinoflagellate cyst record occur within the 20th century in New Bedford Harbor. These changes are coincident with an increase of organic carbon input, PAHs, PCBs, and heavy metal concentrations that may have antagonistic or cumulative effects.

We believe that the eutrophication signal seen in dinoflagellate cyst assemblages in fjords and shallow estuaries can be distinctly different. Under simple nutrient enrichment, increased concentrations of cysts may not be expected in shallow, well-mixed estuaries, yet appear to be a response in fjords. Species proposed as indicators of eutrophication in Norwegian fjords (Dale and Fjellså, 1994; Saetre et al., 1997; Dale et al., 1999) show different responses in the shallow estuaries of Massachusetts, the latter more similar to Tokyo Bay.

More research is needed to refine the use of dinoflagellate cysts as record of paleoenvironmental changes in estuaries. Some important research directions should include separation of the role of individual stressors such as nutrient over-enrichment and toxic pollution, as well as studies of dinoflagellate cyst assemblages from different types of estuaries with different degrees of human disturbance.

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References

Abdelrhman MA. Modeling how a hurricane barrier in New Bedford Harbor, Massachusetts has affected the hydrodynamics and residence times (in preparation).

Appleby PG, Oldfield F. The calculation of lead-210 dates assuming a constant rate of supply of unsupported 210Pb to the sediment. Catena 1978;5:1–8.

Baron WR, Gordon GA. A reconstruction of New England climate using historical materials, 1620–1980. In: Harington CR, editor. Climatic change in Canada 5, critical periods in the Quaternary climatic history of northern North America. Syllogeus No. 55. Ottawa: National Museums of Canada, 1985, 1985. p. 229–245.

Boynton WR, Kemp WM, Keefe CW. A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. In: Kennedy VS, editor. Estuarine comparisons. New York: Academic Press, 1982. p. 69–90.

Brugam RB. Pollen indicators of land-use change in southern Connecticut. Quaternary Res 1978;9:349–362.

Brush GS, Davis FW. Stratigraphic evidence of human disturbance in an estuary. Quaternary Res 1984;22:91–108.

Chmura GL, Aharon P. Stable carbon isotope signatures of sedimentary carbon in coastal wetlands as indicators of salinity regime. J Coastal Res 1995;11(1):124–135.

Cooper SR. Chesapeake Bay watershed historical land use impact on water quality and diatom communities. Ecol Appl 1995;5(3):703–723.

Cooper SR, Brush GS. A 2500-year history of anoxia and eutrophication in Chesapeake Bay. Estuaries 1993;16(3b):617–626.

Costa JE, Howes BL, Gunn E. Report of the Buzzards Bay citizens. water quality monitoring program 1992–1995, Marion, MA: Buzzards Bay, National Estuary Program, 1996.

Dale B. Cyst formation, sedimentation, and preservation: factors affecting dinoflagellate assemblages in recent sediments from Trondheimsfjord, Norway. Rev Palaeobotany Palynol 1976;22:39–60.

Dale B. Dinoflagellate cyst ecology: modeling and geological applications. In: Jansonius J, McGregor DC, editors. Palynology: principles and applications, vol. 3. College Station, TX: American Association of Stratigraphic Palynologists Foundation, 1996. p. 1249–1275.

Dale B. Marine dinoflagellate cysts as indicators of eutrophication and industrial pollution: a discussion. Sci Total Environ 2001;264:235–240.

Dale B, Fjellså A. Dinoflagellate cysts as productivity indicators: State of the art, potential and limits. In: Zahn R, editor. Carbon cycling in the glacial ocean: constraints in the ocean's role in global change. Berlin: Springer, 1994. p. 521–537.

Dale B, Thorsen TA, Fjellså A. Dinoflagellate cysts as indicators of cultural eutrophication in the Oslofjord, Norway. Estuarine Coastal Shelf Sci 1999;48(3):371–382.

Environmental Data and Information Service. Climate normals for the US (base, 1951–80). National Oceanic and Atmospheric Administration, US National Climatic Center, 1983.

Eppley RW, Weiler CS. The dominance of nanoplankton as an indicator of marine pollution: a critique. Oceanol Acta 1979;2(2):241–245.

Evgenidou A, Konkle A, D'Ambrosio A, Corcoran A, Bowen J, Brown E, Corcoran D, Dearholt C, Fern S, Lamb A, Michalowski J, Ruegg I, Cebrián J. Effects of increased nitrogen loading on the abundance of diatoms and dinoflagellates in estuarine phytoplankton communities. Biol Bull 1999;197:292–294.

Fensome RA, Taylor FJR, Norris G, Sarjeant WAS, Wharton DI, Williams GL. A classification of living and fossil dinoflagellates. Micropaleontol Special Publ 1993;7:1–351.

Head MJ. Modern dinoflagellate cysts and their biological affinities. In: Jansonius J, McGregor DC, editors. Palynology: principles and applications, vol. 3. College Station, TX: American Association of Stratigraphic Palynologists Foundation, 1996. p. 1197–1248.

Head MJ, Harland R, Matthiessen J. Cold marine indicators of the late Quaternary: the new dinoflagellate cyst genus *Islandinium* and related morphotypes. J Quaternary Sci 2001;16(7):621–636.

Howes BL, Williams TW, Rasmussen M. Baywatchers II, Nutrient related water quality of Buzzards Bay embayments: a synthesis of baywatchers monitoring 1992–1998. The Coalition for Buzzards Bay, 1999.

Höll C, Zonneveld KAF, Willems H. Organic-walled dinoflagellate cyst assemblages in the tropical Atlantic Ocean and oceanographical changes over the last 140 ka. Palaeogeogr Palaeoclimatol Palaeocol 2000;160:69–90.

Jacobson DM, Anderson DM. Widespread phagocytosis of ciliates and other protists by marine mixotrophic and heterotrophic thecate dinoflagellates. J Phycol 1996;32:279–285.

Latimer JS, Boothman WS, Jayaraman S. Environmental stress and recovery: the geochemical record of human disturbance in New Bedford Harbor and Apponagansett Bay (in preparation).

Lentin JK, Williams GL. Fossil dinoflagellates: index to genera and species, 1993 ed. American Association of Stratigraphic Palynologists, Contribution Series, vol. 28 1993. p. 1–856.

Matsuoka K. Eutrophication process recorded in dinoflagellate cyst assemblages—a case of Yokohama Port, Tokyo Bay, Japan. Sci Total Environ 1999;231:17–35.

Matsuoka K. Further evidence for a marine dinoflagellate cyst as an indicator of eutrophication in Yokohama Port, Tokyo Bay, Japan. Comments on a discussion by B. Dale. Sci Total Environ 2001;264:221–233.

McCleland JW, Valiela I. Changes in food web structure under the influence of increased anthropogenic nitrogen inputs to estuaries. Marine Ecol Progr Ser 1996;68:259–273.

Neilson BJ, Cronin LE, editors. Estuaries and nutrients, Proceedings of an International Symposium on Nutrient Enrichment in Estuaries. Contemporary issues in Science and Society, Chesapeake Research Consortium Publication No 90 1981.

Nelson WG, Bergen BJ, Benyi SJ, Morrison G, Voyer RA, Strobel CJ, Rego S, Thursby G, Pesch CE. New Bedford Harbor long-term monitoring Assessment Report: Baseline sampling. Technical Report EPA/600/R-96/097. Narragansett, RI: US Environmental Protection Agency, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, 1996.

Nixon SW. Coastal marine eutrophication: a definition, social causes, and future concerns. Ophelia 1995;44:199–219.

Paerl HW. Nuisance phytoplankton blooms in coastal, estuarine, and inland waters. Limnol Oceanogr 1988;33:823–847.

Pesch CE, Garber J. Historical analysis: a valuable tool in community-based environmental protection. Marine Pollut Bull 2001;42(5):339–349.

Pruell RJ, Norwood CB, Bowen RD, Boothman WS, Rogerson PF, Hackett M, Butterworth BC. Geochemical study of sedimentary contamination in New Bedford Harbor, Massachusetts. Marine Environ Res 1990;29:77–101.

Pospelova V, Chmura GL. Modern distribution of dinoflagellate cysts in coastal lagoons of Rhode Island, USA. Norges teknisk-naturvitenskapelige universitet Vitenskapsmuseet. Rapport Botanisk Serie 1998;1:122–123.

Pospelova V, Head MJ. *Islandinium brevispinosum* sp. nov. (Dinoflagellata), a new organic-walled dinoflagellate cyst from modern estuarine sediments of New England (USA). J Phycology 2002;38(3):593–601.

Roffinoli RJ, Fletcher PD. Soil survey of Bristol County, Massachusetts, southern part. US Department of Agriculture Soil Conservation Service, 1981.

Saetre MML, Dale B, Abdullah MI, Saetre GPO. Dinoflagellate cysts as potential indicators of industrial pollution in a Norwegian fjord. Marine Environ Res 1997;44(2):167–189.

Sommer U. An experimental test of the intermediate disturbance hypothesis using cultures of marine phytoplankton. Limnol Oceanogr 1995;40(7):1272–1277.

Sullivan MJ. Applied diatom studies in estuaries and shallow coastal environments. In: Stoermer IF, Smol JP, editors. The diatoms: applications for the environmental and earth scienceCambridge University Press, 1999. .

Stockmarr J. Tablets with spores used in absolute pollen analysis. Pollen Spores 1977;13(4):615–621.

Summerhayes CP, Ellis JP, Stoffers P, Briggs SR, Fitzgerald MG. Fine-grained sediment and industrial waste distribution and dispersal in New Bedford Harbor and western Buzzards Bay, Massachusetts. Technical Report WHOI-76-115. Massachusetts: Woods Hole Oceanographic Institution, 1977.

Taylor FJR, Pallingher U. Ecology of dinoflagellates. In: Taylor FJR, editor. The biology of dinoflagellates Botanical Monographs, vol. 21. Oxford: Blackwell Scientific, 1987. p. 399–529.

Thorsen TA, Dale B. Dinoflagellate cysts as indicators of pollution and past climate in a Norwegian fjord. Holocene 1997;7(4):433–446.

Tsirtsis G, Karydis M. Evaluation of phytoplankton community indices for determining eutrophic trends in the marine environment. Environ Monitoring Assess 1998;50:255–269.

Voyer RA, Pesch CE, Garber J, Copeland J, Comeleo R. New Bedford Harbor, Massachusetts: a story of urbanization and ecological connections. Environ History 2000;5(3):352–377.