Proceedings

of the

Fourth Biennial

Long Island Sound Research Conference

Robert B. Whitlatch
Joyce R. Wood-Martin
Editors

November 13-14, 1998
State University of New York
Purchase

The Editors would like to thank Barbara A. Mahoney from the Department of Marine Sciences, University of Connecticut, for her valuable contribution to the preparation of these Proceedings. We would also like to express our appreciation for the computer wizardry of Todd Fake, Department of Marine Sciences/Marine Sciences and Technology Center, University of Connecticut.

The Editors would like to thank Margaret S. Van Patten and Tessa Simlick from the Connecticut Sea Grant College Program, University of Connecticut, for their editorial comments.

ISBN 1-878301-05-5

The Connecticut Sea Grant College Program is sponsored by the National Sea Grant College Program, National Oceanic and Atmospheric Administration, United States Department of Commerce.

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ACKNOWLEDGEMENTS

We would like to thank Barbara Mahoney and Mellissa Thompson-Taylor for their assistance in making this conference a success, and to the Northeast Utilities Environmental Laboratory for the printing services they have provided for this program and the conference proceedings. Special thanks to the staff at the Performing Arts Center at SUNY/Purchase and their catering service and to the staff at The Castle at Manhattenville College.

PREFACE

This volume derives from the Fourth Biennial Long Island Sound Research Conference (LISRC) held at Purchase College, State University of New York, on November 13-14, 1998. The open nature of the conference brought together academic, non-profit and government researchers as well as a diverse set of research projects and viewpoints regarding the state of the Sound. The oral and poster presentations covered the gamut from biological to physico-chemical and geological studies of the Long Island Sound environment. The breadth of the presentations cannot fully be represented in this volume.

The intent of the biennial LISRC is to provide a sort of cross-fertilization among the different constituencies with the broader aim of assembling a synoptic view of the Long Island Sound ecosystem. This goal will be furthered during the next LISRC to be held November 17-18, 2000, at the University of Connecticut, Stamford, where the theme will be the Long Island Sound as a functioning whole.

As conference host, I must acknowledge the local support I received without which I would not have survived the November 1998 conference with sanity. Dr. Joseph Skrivanek (Chair, Division of Natural Sciences, Purchase College) helped "grease the skids" to accomplish a thousand little details. B. Ardohain, K. Baade, S. Browne, A. Eversley, H. Heiser, J.-S. Kolusniewski and M. Sellberg all helped organize and attend to last minute details facilitating the talks, the luncheon and Friday's banquet.

George P. Kraemer
Department of Environmental Sciences
Purchase College, State University of New York

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CONTAMINANTS

REGIONAL ACCUMULATION OF CONTAMINATED SEDIMENTS IN LONG ISLAND SOUND

Buchholtz ten Brink, M.R. and E.L. Mecray, U.S. Geological Survey, Coastal and Marine Geology Program, 384 Woods Hole Road, Woods Hole, MA 02543

ABSTRACT

The distribution of contaminants in surface sediments of Long Island Sound has been mapped as part of the United States Geological Survey (USGS) regional study on sediment quality and dynamics. Surface (0-2 cm) sediments and bottom photographs were collected at 130 stations throughout the Sound; sediment cores were collected at 58 of the stations. Concentrations of trace elements (cadmium (Cd), chromium (Cr), copper (Cu), zinc (Zn), lead (Pb), and others), major elements, *Clostridium perfringens* spores (a sewage tracer), and grain size distribution were determined in surface samples for each location and are currently being analyzed for down-core profiles. Focusing of contaminant-bearing particles in regions of deposition, and removal or non-deposition of these particles in other areas, results in high contaminant concentrations that parallel the distribution of fine-grained sediments. Chemical and lithological analysis of sediment cores has been used to delineate the distribution and accumulation of contaminated sediments in Long Island Sound. All profiles of *Clostridium perfringens* have concentrations above background levels to depths of at least 30 cm, with higher concentrations in the fine-grained sediments. The concentrations of metals (e.g., Cu, Cr, Zn) co-vary, indicating common sources and transport patterns within the basin. Recent sediments in the western basin typically have concentrations of metals up to three times greater than pre-industrial background values. In depositional regions, contaminant concentrations in surface sediment are generally within commonly cited sediment criteria values. Sedimentary environment and source proximity are the dominant factors controlling contaminant accumulation; however, accumulation patterns are modified in the sediments by mixing depths and transport mechanisms that vary greatly within Long Island Sound.

INTRODUCTION

The water and sediment quality of many coastal areas in the United States are impacted by proximity to urban centers, industrial activity, and agricultural activities. Semi-enclosed marine areas, such as Long Island Sound, are particularly sensitive to anthropogenic activity (O'Connor and Ehler, 1991) because pollutants may be less efficiently removed, dispersed, or diluted than on open coasts. Long Island Sound is bordered by New York City as well as the cities of Stamford, Bridgeport, Norwalk, New Haven, and New London, CT, and Huntington and Smithtown, NY. These urban areas are the source of direct input of pollutants in the form of sewage effluent, industrial discharge, dredging disposal, urban runoff, and atmospheric deposition (Buchholtz ten Brink et al. 1994). In addition, major rivers such as the Housatonic, Quinnipiac, Connecticut, Norwalk, and Thames drain extensive inland areas of Connecticut and Massachusetts into the Sound (Zimmerman et al. 1996). This potential for adverse environmental effects (Long et al. 1996 and references therein), plus management concerns (e.g., Robertson et al. 1991), prompted the USGS to undertake a multidisciplinary study (Poppe and Polloni 1998) of the Sound's environmental health and the influence that geologic processes have on its recovery. Objectives of the geochemical component of the study are to determine the regional distribution of contaminants in sediments of Long Island Sound, measure contaminant inventories and rates of sediment mixing and accumulation, and increase our understanding of the processes by which pollutants migrate in the marine environment in order to better predict he fate of existing and future contaminants in Long Island Sound sediments.

The fate of contaminants introduced into coastal water is affected by a number of processes (Figure 1) (Santschi et al. 1990; Salomons and Förstner 1984). Sedimentation, burial, resuspension, and winnowing are large-scale geological processes that effect the transport and accumulation of sediments, particulate material, and the particle-reactive contaminants that are associated with them. Processes of adsorption, chemical reaction, and diagenesis at the interface of solids and solutions can change the chemical speciation of a contaminant and determine what fraction of the material is dissolved vs. solid. Dissolved species are available for diffusion out of the sediments and can affect the health of biota. Solid species undergo physical transport processes such as burial, bioturbation, winnowing, or other types of sediment reworking. The input rates and the influence of various contaminant sources has changed over time as areas

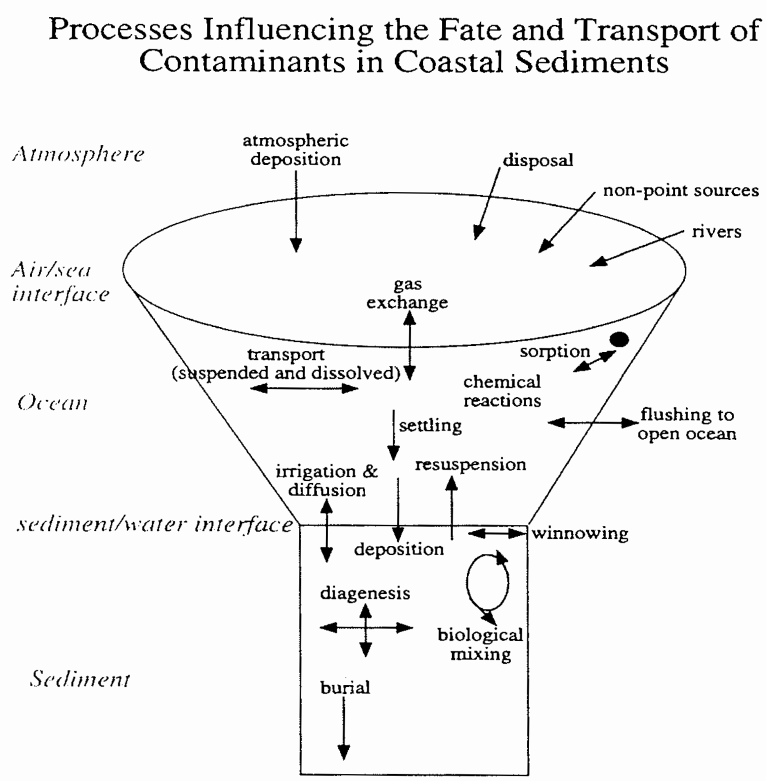


Figure 1. Processes that influence the fate and transport of particle-reactive contaminants in coastal water and sediment.

around the Sound experienced population growth and increasing industrial development. Within the last 20 years, efforts to reduce air and water pollution have resulted in decreased input of some contaminants, such as direct discharge of heavy metals and atmospheric input of lead. Physical processes, such as sediment accumulation or resuspension, can also vary over time on scales from days (e.g., storm action) to decades. The physical environment within Long Island Sound, which is created by the combination of bathymetry, current and wave action, and sediment sources, varies widely. The Sound consequently has great spatial diversity in its sedimentary environments (Knebel et al. 1999) and the current action that moves sediments (Signell et al. 1998). The patterns of contaminant distribution that are present in sediment cores and across the surface of the Sound provide both a record of past contaminant input and the means to measure the rates and magnitudes of many of the processes that transport them. Understanding the relative magnitudes and variability of transport processes in Long Island Sound enhances our ability to manage the marine environment and strive towards a healthy, sustainable ecosystem.

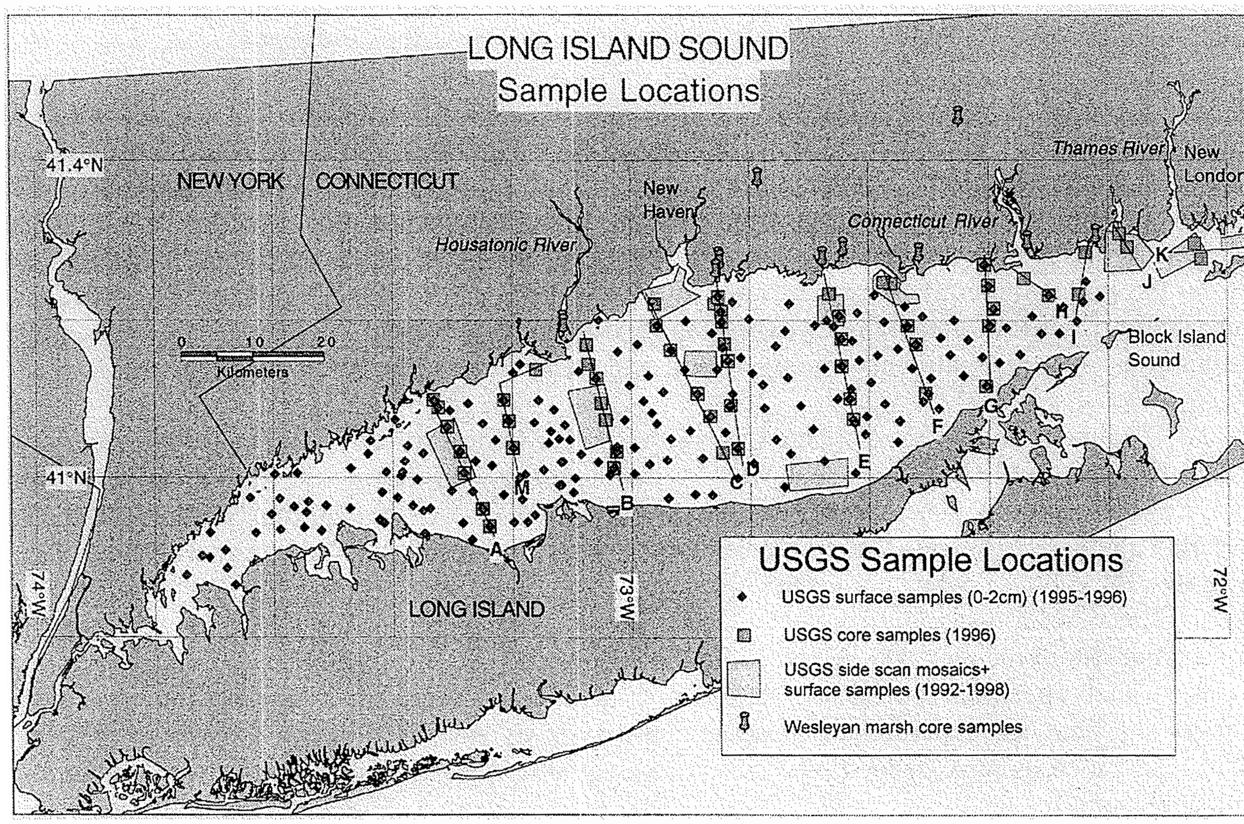


Figure 2. Location of sediment samples collected in Long Island Sound by the U.S. Geological Survey in 1996-1998.

In this study, geochemical and lithological analyses were performed on surface sediments and sediment cores, which were collected in 1996 throughout Long Island Sound (Figure 2). These data provide a baseline for contaminant distribution, identify sources, determine transport and dispersal paths, identify areas of extensive contaminant accumulation, and provide a basis for model predictions of burial and remobilization rates. Most studies prior to 1996 were geographically or topically restricted, or were two decades old and did not address current conditions in the Sound. The distributions of *Clostridium perfringens*, a tracer for sewage input, and a suite of other contaminants were measured in surface sediments and sediment cores. Ongoing studies of lithological features, sediment properties, and radiometric dating of cores provide temporal constraints.

METHODS

Study Area and Sampling. Sediment cores were collected for geochemical analysis in June 1996 at 58 stations along N-S transects (Figure 2). Sample sites were selected to include: 1) areas that are representative of particular sedimentary regimes (Knebel et al. 1999); 2) transects seaward of contaminant sources; 3) regions where sidescan mosaics and biological community data have been generated (Poppe and Polloni 1998); and 4) sites offshore from land-based sites of study (Varekamp and Scholand 1996; Varekamp 1996). Chemical analysis of grab samples collected in April and June 1996 along seismic track lines, and in March, 1997 in western Long Island Sound, provided additional coverage for a total of 219 surface samples.

The grab sampler was a modified Van Veen grab (Teflon-coated) that has a downward looking video and still camera attached in order to characterize bottom habitats and the quality of the sediment grab. After photographing and describing the sediment collected in the grab, the overlying water was removed and the upper 2 cm of sediment was sampled with a Teflon-coated shovel. Sediment was placed in a pre-cleaned (5% nitric acid, distilled water, and methanol rinse) plastic container, homogenized, and aliquots were separated for later analysis of grain size distribution, *Clostridium perfringens* spores, and chemical analysis of major components and contaminants. Sample aliquots were weighed, sealed, and refrigerated until analysis.

Sediment cores were collected using the USGS hydrostatically- dampened gravity corer (Bothner et al. 1997), which collects 11 cm diameter cores up to 70 cm in length in polycarbonate tubing. The corer is designed to minimize disturbance of the sediment surface so detailed chemical gradients can be measured in the sediments near the sediment-water interface. The overlying water collected in the corer was usually very clear and surface features, such as fecal pellets, were distinct. A bottom-facing video was mounted on the frame of the corer and focused to record the entry of the corer into the sediment. This allowed the core to be placed in a location that appeared typical for the station, avoided placement of the sampling gear in dangerous spots, and recorded any leakage that may have occurred from the bottom of the core. On deck, the cores were capped, described, and stored refrigerated or frozen for later sectioning and analysis.

Analysis. On shore, X-radiographs were taken of all cores (in an upright position) to document fine-scale lithological features and aid in selection of cores for sectioning. Immediately prior to sectioning, the overlying water was removed and profiles of magnetic susceptibility, bulk density, and P-wave velocity were determined on each whole core with a Multi-sensor Core logger using methods described in Boyce (1973). Cores were then vertically extruded and sectioned in 0.5 or 1.0 cm intervals, with edges trimmed to avoid sample smearing. Sediments were sectioned with titanium (Ti) spatulas to prevent contamination. All surfaces contacting sediment (core barrels, sampling spatulas, aliquot containers) were rinsed with 5%

nitric acid, distilled water, and methanol, and dried in a laminar-flow hood prior to use. Each core section was homogenized and an aliquot removed for analysis of *Clostridium perfringens*. The remainder of the sample was weighed, freeze-dried, and water content determined by weight loss. Sample analysis is in progress on the cores for major components (e.g., aluminum (Al), barium (Ba), iron (Fe), organic carbon (C_{org}), and manganese (Mn)), anthropogenic contaminants (e.g., silver (Ag), Pb, Cu, Zn, Cd, and mercury (Hg)), selected organic compounds, radioisotopes, grain size, and environmental indicators (e.g., foraminifera and pollen). Eventually, radiochemical and lithological data will allow historical horizons to be identified so that contaminant accumulation rates and mobility can be determined.

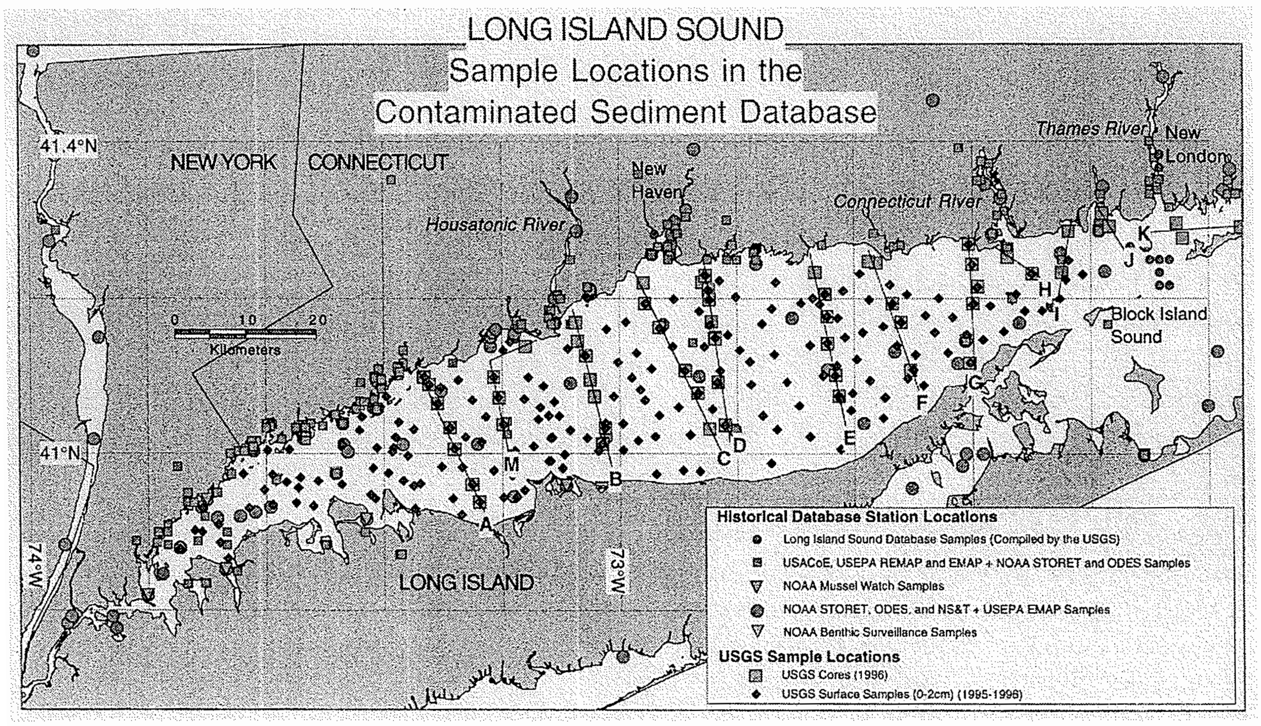


Figure 3. Locations of samples from the literature that are entered to date in the Contaminated-Sediment Database for Long Island Sound, an on-going multi-institutional effort to compile historical data.

Grain size was determined using laboratory methods described in detail by Folk (1974) and Poppe et al. (1985). *Clostridium perfringens* aliquots were kept refrigerated for analysis by membrane filtration and enumeration (United States Environmental Protection Agency 1995). A subsample (> 50 mg) was prepared for chemical analysis by total digestion using nitric, hydrochloric, perchloric, and hydrofluoric acids, a microwave, and hydrogen peroxide. The solubilized sediment was measured for Fe, Al, calcium (Ca), Ti, Mn, Zn, nickel (Ni), Cr, vanadium (V), Cu, Ba, and zirconium (Zr) using a Jobin-Yvon JY24 inductively-coupled plasma optical emission spectrometer (ICP-ES). Cadmium, lead, and silver were measured with a Perkin-Elmer 5100PC Graphite-furnace Atomic Adsorption Spectrometer (GFAAS). Digestion, ICP-ES and GFAAS analysis followed procedures used by Murray and Leinen (1996). Accuracy and precision was established by analyzing standard sediments and replicate samples. Analytical results and sample identification information for both grab and core samples were entered, organized, and reduced using standard spreadsheet and plotting software. Gamma emitting radioisotopes, including ^{210}Pb and ^{137}Cs, were identified on a low-background Princeton Gamma

Tech Germanium well detector and were used to quantify sediment accumulation and the depth of sediment mixing. Preliminary results for some cores are reported herein.

Contaminated Sediment Database. The USGS, and collaborators from numerous agencies and institutions, has compiled a Contaminated Sediment Database for the Gulf of Maine (Buchholtz ten Brink et al. 1997) and is extending the compilation to Long Island Sound (Figure 3) (Buchholtz ten Brink and Mecray 1998) to supplement the 1996 samples and provide a tool for research and environmental management use. The database contains original data on chemical constituents and sample identification from published and gray literature sources along with documentation about the quality of the data. As noted by Brownawell et al. (1992), documentation of analytical quality is often missing from the literature so batch validation techniques are used for this heterogeneous data set (Manheim et al. 1998).

RESULTS AND INTERPRETATION

Sedimentary environment. Long Island Sound sediments are generally muddy in the west with a transition to coarser textures in the east (Figure 4) (Knebel et al. 1998 and 1999; Poppe et al. 1998 and references therein). Knebel et al. (1999) has classified the sedimentary environments of the Sound into areas of fine-grained-deposition, sediment sorting and reworking, coarse-grained bedload transport, and erosion or nondeposition. These environments closely follow the patterns of mean bedload transport predicted by wave and current modeling for the Sound (Signell et al. 1998b). Sediments observed on bottom video, and in cores, range from soft, black odorous mud to well-sorted coarse sand and exhibit lithological properties that correspond to characteristics of the environment in which they were located. Cores along transects A, M, B, C, D and K (Figure 5) are muddy, are located in areas of fine-grained deposition, and often have layering preserved in nearshore cores. The seafloor is covered with polychaete tubes and evidence of other stationary infauna. Cores from transects E, F, G (except the most northern and southern nearshore samples), H, I, J, and L are in areas where strong current action has prevented accumulation of fine-grained sediment. Bottom photographs show rippled sands or stony deposits and cores from these areas generally are sandy throughout and frequently contain mollusk shells. The paucity of structures in the x-radiographs of sandy cores from areas of sorting or transport suggests that contaminant gradients are unlikely to persist in the sediments.

Regional contaminant distribution. The distribution of the *Clostridium perfringens* spore counts (Figure 6a) and the concentration of particle-reactive metals in surface sediments (Cu, Zn, and Pb; Figures 6b,c, and d, respectively) follow patterns that correlate with the sedimentary environments and the sediment texture (Figure 7). In depositional areas, the sediments consist primarily of silt and clay-sized particles. These particles have a large sorption capacity for contaminant metals because of their high surface area and surface reactivity. The predominance of clay minerals (which is also indicated by higher aluminum concentrations), high levels (1-2%) of C_{org} in the sediments, and the presence of Fe, Mn, and sulfur (S) bearing minerals contribute to strong interaction of the sediment surfaces with contaminants. In these

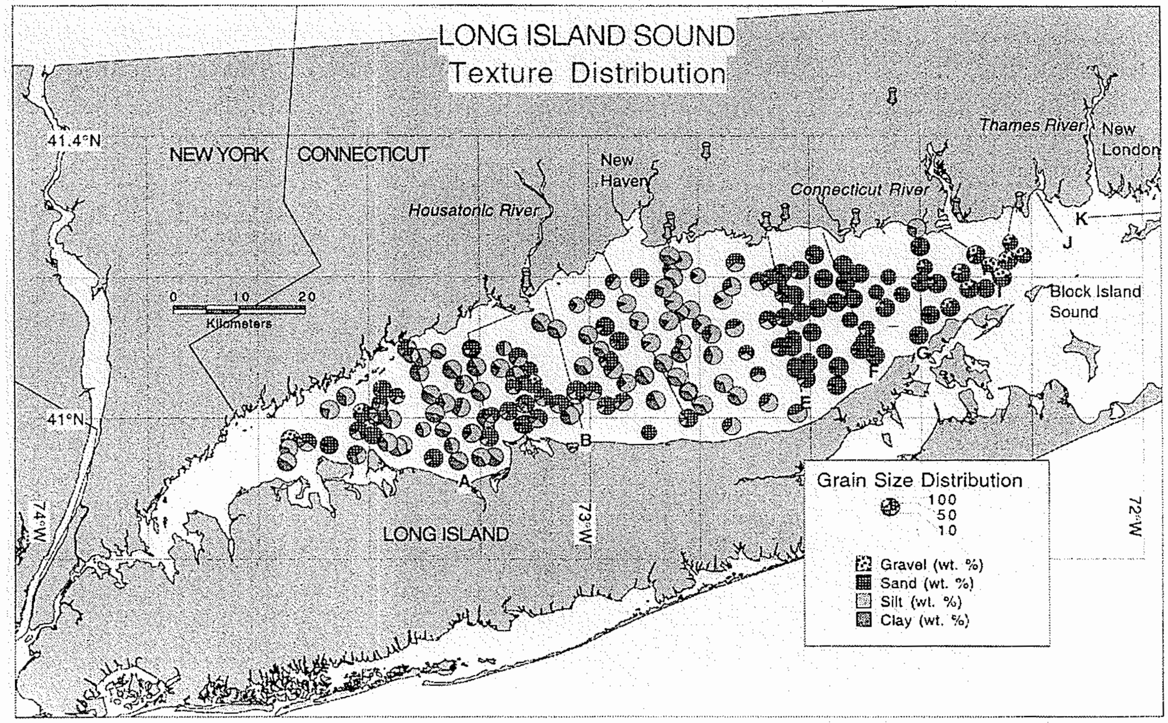


Figure 4. Sediment grain size in surface samples from Long Island Sound.

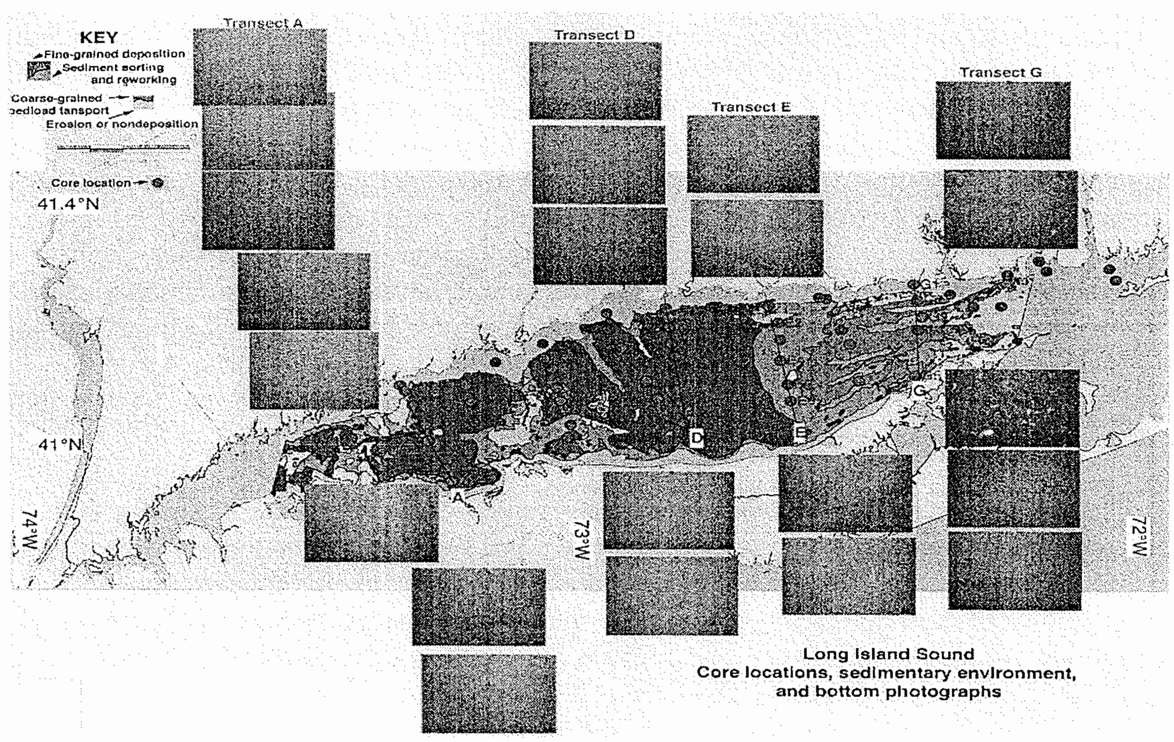
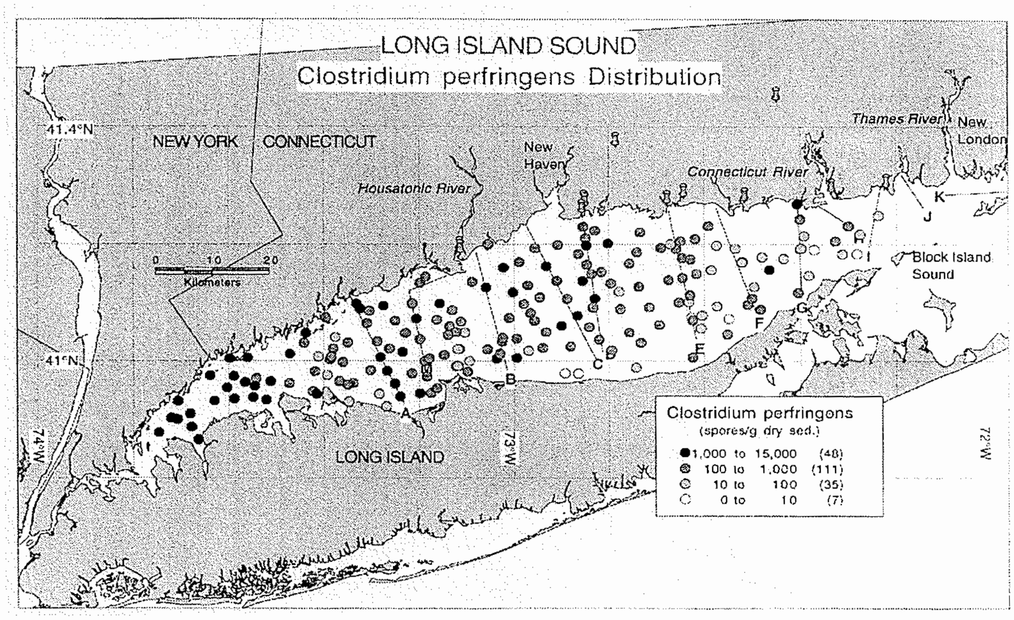


Figure 5. Locations of sediment core transects in characteristic regimes. The base map of sedimentary environments is modified from Knebel *et al.* (1998, 1999).

depositional areas, sediments are rarely remobilized by wave and current action, but may experience significant amounts of mixing due to bioturbation (Knebel et al. 1999). In the other sedimentary environments, the sediments consist primarily of sand-sized particles that are predominantly composed of silicate minerals. These sediment particles have a lower sorptive capacity for contaminants due to less reactive mineralogy, lower C_{org} (<0.5%) concentrations, and smaller surface areas than the sediment in depositional areas. Resuspension and bedload transport are the dominant mechanisms of sediment mixing in these areas (Knebel et al. 1999); mechanisms that cause sediments to be repeatedly exposed to overlying water and allow removal of small particles by winnowing. The strong correlation of contaminant concentrations with sedimentary environment and texture is a result of both the removal of these materials from areas of high bedload transport and the greater capacity of the sediments in depositional areas to chemically sequester and retain contaminants.

Clostridium perfringens is a bacterial spore that is present in the intestinal tract of mammals. Its spores are essentially inert in marine sediments and consequently are an excellent tracer of sewage input into an ecosystem (Hill et al. 1993). Since sewage is often a major source of other contaminants (e.g., Ag, Cu, and Hg) in coastal waters, these bacteria concentrations are a valuable screening tool for the magnitude and location of other contaminants in sediments. *Clostridium perfringens* concentrations measured in surface sediments and core samples indicate widespread and long-term addition of anthropogenic components to the sedimentary system in Long Island Sound. *Clostridium perfringens* concentrations range from 15,000 to 0 spores per g dry^{-1} sediment in the upper 2 cm. Surface sediments (0-2 cm) have the highest *Clostridium perfringens* concentrations in the western end of the Sound, very low or non-detectable values in the eastern region, and intermediate values in the central basins. The pattern of *Clostridium perfringens* concentrations seen across the Sound is attributed to factors of transport and source proximity. Unlike metal contaminants, the spores do not chemically interact with mineral surfaces, so the sorptive capacity of the sediments has little to do with the observed distribution; although it is possible that spore mobility could be reduced by aggregation in organic-rich environments. Like fine sediment, little accumulation of the small spores (1.0 to 1.5 μm wide by 4.0 to 8.0 μm long) occurs in sediments that have a long-term sandy character, such as the dynamic eastern end of the Sound. Intermediate and relatively uniform concentrations of *Clostridium perfringens* in the central basins result from the strong tidal flushing and wave action, which can widely disperse and homogenize suspended material prior to deposition. The highest concentrations of *Clostridium perfringens*, found in the western Sound, are probably associated with input from New York City sewage plants.

The widespread *Clostridium perfringens* values throughout the Sound predict the presence of other contaminants that are associated with sewage input. Copper, zinc, lead (Figure 6b, c, and d) and mercury (Kreulen et al. 1997 and 1998; not shown) concentrations in surface sediments show patterns that strongly resemble the *Clostridium perfringens* distribution. High concentrations of many other contaminants reflect the focusing of contaminant-bearing particles in regions of deposition and their removal or non-deposition in other areas. It is interesting to note that these contaminant metals have strong positive correlation with other indicators of depositional environments, such as fine grain size, high C_{org} content, and high Al concentrations. These correlations can be readily used to predict probable sediment loadings for other elements or locations, but should not be used to assign causality.



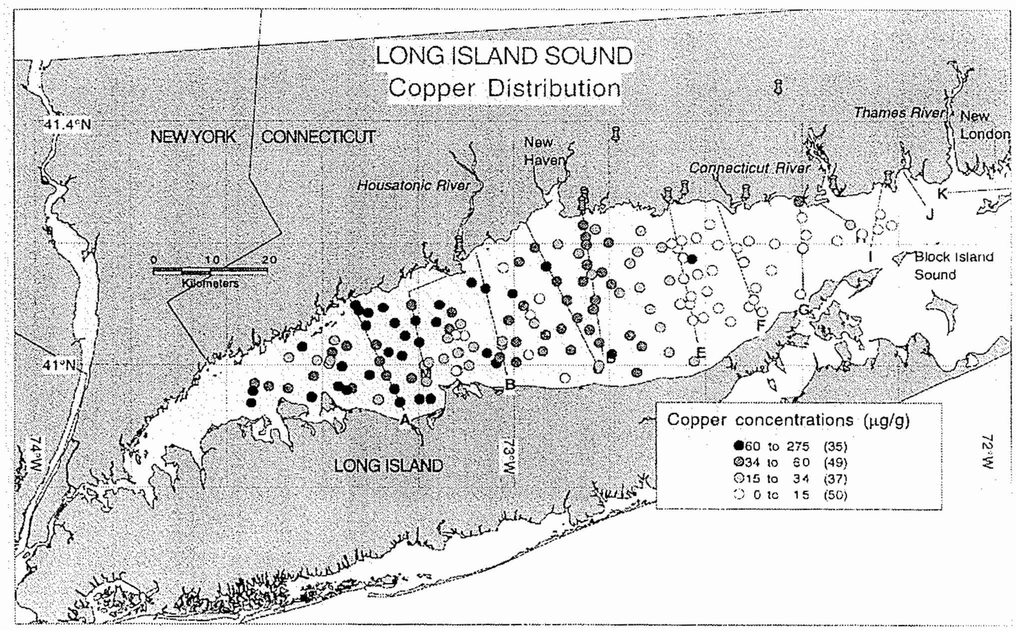


Figure 6. Concentration distribution of contaminants in Long Island Sound surface sediments: Upper panel -- *Clostridium perfringens*, Lower panel – Cu.

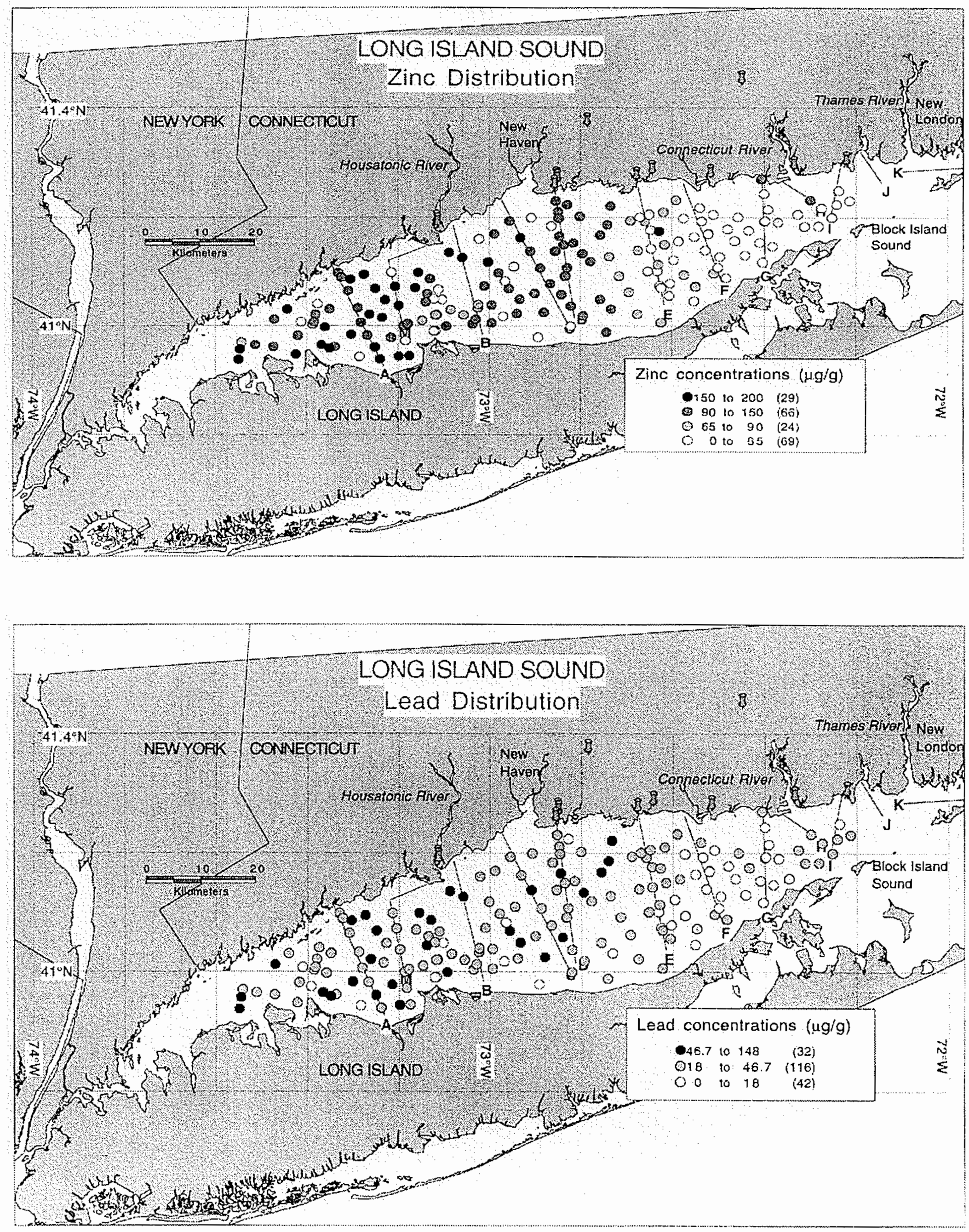


Figure 6 (continued). Concentration distribution of contaminants in Long Island Sound surface sediments: Upper panel – Zn; Lower panel – Pb. The concentration ranges indicate naturally occurring background, anthropogenically elevated values, and concentrations that have potential to induce detrimental effects on ecosystem health (Long *et al.*, 1998). Background values measured in cores: 15 µg g^{-1}, 65 µg g^{-1}, and 18 µg g^{-}1 for Cu, Zn and Pb respectively. Sediment ER-L and ER-M quality criteria are respectively Cu 34 and 270 µg g^{-1}; Zn 150 and 410 µg g^{-1}; and Pb 46.7 and 218 µg g^{-1}.

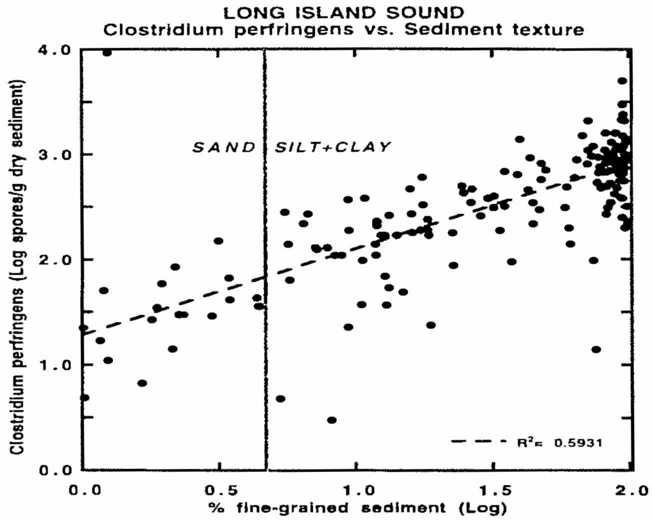


Figure 7. Correlation of *Clostridium perfringens* concentration and sediment texture for Long Island Sound surface sediments.

The spatial heterogeneity of contaminant concentrations for surface sediments is relatively small within a particular sedimentary environment. Contaminant concentrations in surface sediments are nearly uniform across the study area after normalization to % fines, Fe, Al, or C_{org}; all of which correct for dilution of the contaminant-bearing, fine-grained fraction with

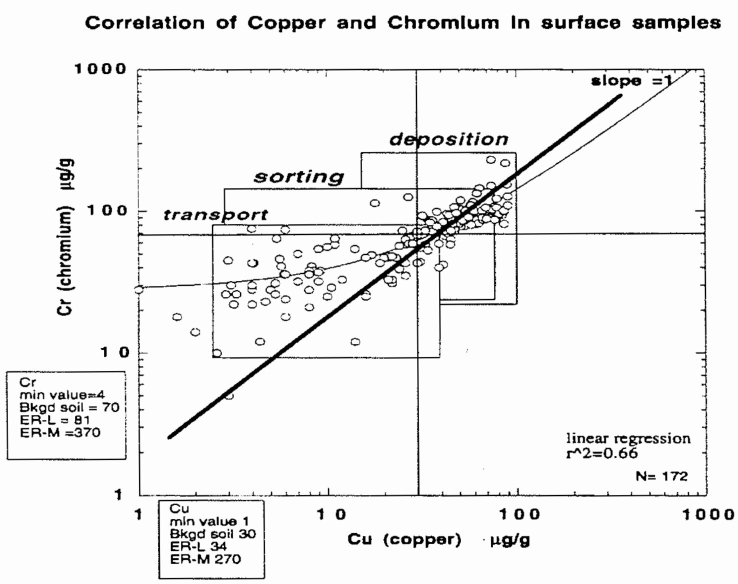


Figure 8. Correlation of contaminant metals Cu and Cr in Long Island Sound surface sediments. Boxes enclose samples from the indicated sedimentary environments.

variable amounts of coarse-grained, "cleaner" material. Metal contaminants in coastal sediments often co-vary because of their common sources and transport patterns. Deviation from typical ratios may indicate either changes in sources or differences of geochemical mobility *in situ*. For these sediments, both chromium (Figure 8) and zinc concentrations are highly correlated with copper concentrations, particularly in depositional environments.

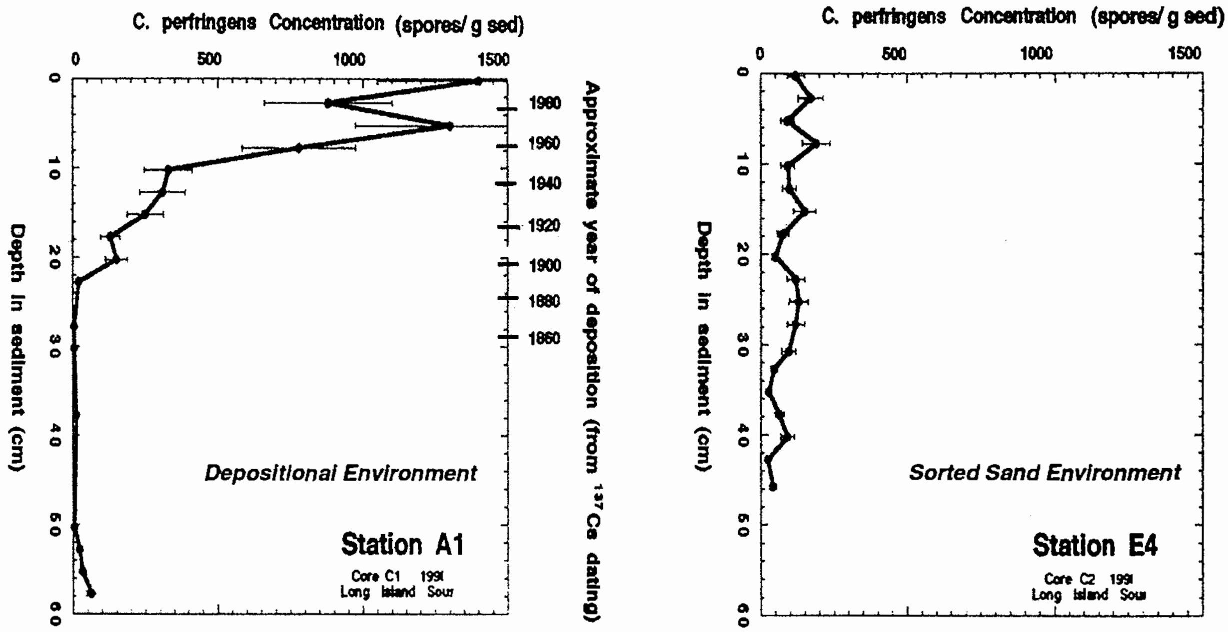


Figure 9 Characteristic *Clostridium perfringens* profile for sediment from (a) depositional areas and (b) areas of sediment sorting and reworking.

Toxicity and background. The concentration maps for Cu, Zn, and Pb (Figure 6b, c, d) depict samples having naturally-occurring background concentrations for these metals, enriched concentrations, and concentrations with the potential to induce detrimental effects on ecosystem health. In Long Island Sound, the surface sediments collected in 1996 have Cu and Zn concentrations that are greater than natural levels for all of the areas having depositional environments; i.e., nearly half of the area of the Sound. Sediment quality criteria for contaminants in sediments, determined by laboratory testing, provide a measure to estimate risk (Long and MacDonald 1998). Sediments with concentrations above the "effects-range medium" (ER-M) sediment quality criteria probably will induce some toxic effects in biota exposed to them, while sediments with contaminant concentrations between the "effects-range low" (ER-L) and ER-M values may do so. Sediment with concentrations below the ER-L values are not likely to cause any toxic effects, although the role of cumulative effects is not well known. In Long Island Sound, Cu concentrations in surface sediments exceed the ER-L values in the depositional regions as do Zn concentrations in the west-central region. Lead concentrations, which have a larger component of atmospheric input than Cu or Zn, are not as uniformly high in the depositional areas and fewer areas have concentrations that exceed toxicity criteria.

Sub-surface contaminant distribution. Profiles of *Clostridium perfringens* and metal contaminants in sediment cores indicate that contaminant inputs to the sediments are not only widespread, but have also impacted the Sound for more than a century. *Clostridium perfringens* profiles show concentrations above background levels to depths of approximately 30 cm in all regions; however, the bulk concentrations are much greater (more than 10x) in modern muddy sediments than in sandy ones. Steep gradients in contaminant profiles of the muddier cores, which are located in the depositional regions (Figure 9a), contrast with concentrations that are generally constant with depth in the sandier cores (Figure 9b). Despite the differing concentrations, the inventories of *Clostridium perfringens* in these two cores, A1 and E4, are both approximately 1050 spores cm^{-2}. This suggests that in areas of sorting and transport, newly deposited fine-grained material, and any contaminants associated with it, is both winnowed away and mixed down. The uniformity with depth in the sandy profiles observed for both *Clostridium perfringens* and contaminant metals is consistent with the active reworking of the seafloor that occurs due to bottom currents and wave action in the higher-energy environments (Signell et al. 1998). Assuming these two cores are representative of all depositional (core A1) and sorting (core E4) environments, nearly twice as much mass of contaminants has accumulated in depositional areas than in the sorted regions.

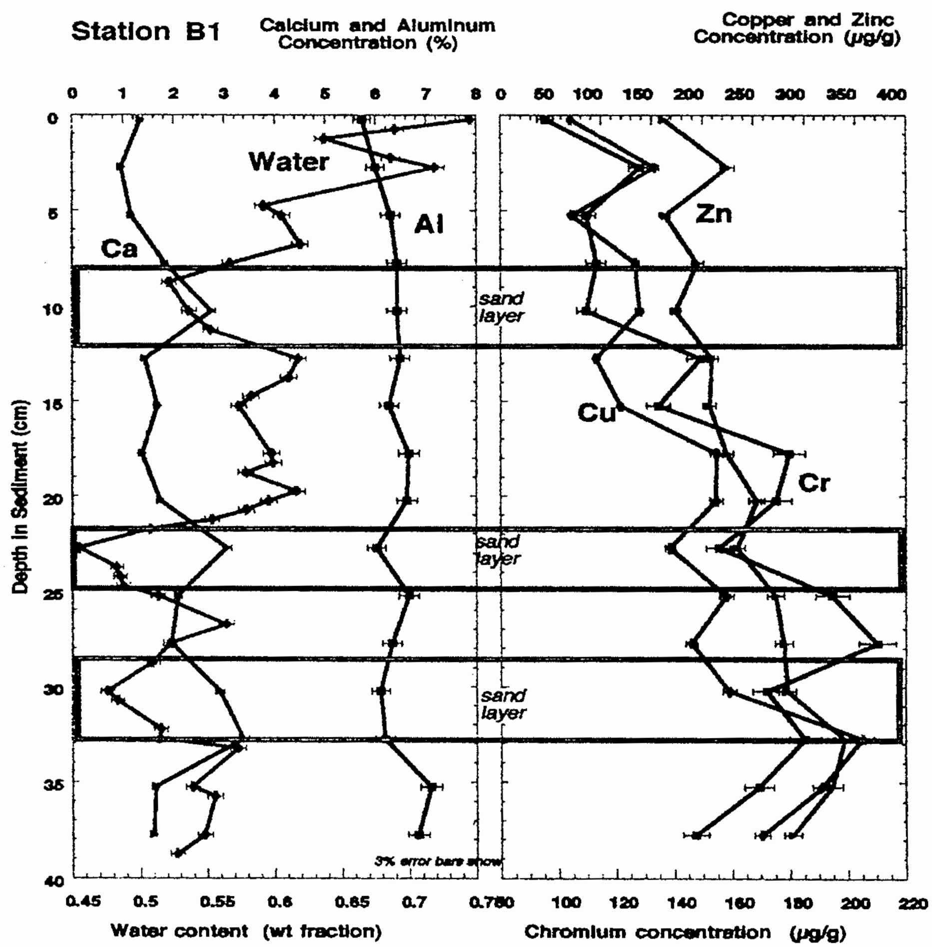


Figure 10. Concentration profiles for parameters indicative of lithological properties (Ca, Al, water content) and contaminant loadings (Cu, Zn, Cr).

Concentrations of metal contaminants in sediment cores also record the influence of sedimentary processes. Recent sediments in the western and central basins typically have concentrations of metals up to three times greater than pre-industrial background values. In most depositional locations, however, the metal concentrations in sediment cores decrease near the surface, reflecting reduction in contaminant sources during the 1980s and 1990s. X-radiographs (e.g., Figure 10) from some cores taken in muddy, depositional areas show the presence and preservation of sand layers. Additional indicators of lithological properties (e.g., high Ca, low Al, low water content, high grain density) identify sections in the core where contaminant concentrations will be lower because of associated winnowing, dilution, and lower sorption capacity. The record of historical contaminant input within the sandier intervals may be underestimated if concentrations are not normalized to texture. In core B1, the Cu, Cr, and Zn follow similar patterns, as was observed in the surface sediment in Long Island Sound. The two- to five-fold decrease in metal concentration towards the surface, seen in the upper 20 cm of the Stratford, CT core, can reflect both the recent source and the influence of diagenetic remobilization. Future work will include use of metal profiles, elemental ratios and differences between various metals to delineate the effects of *in situ* process and provide constraints for flux estimates.

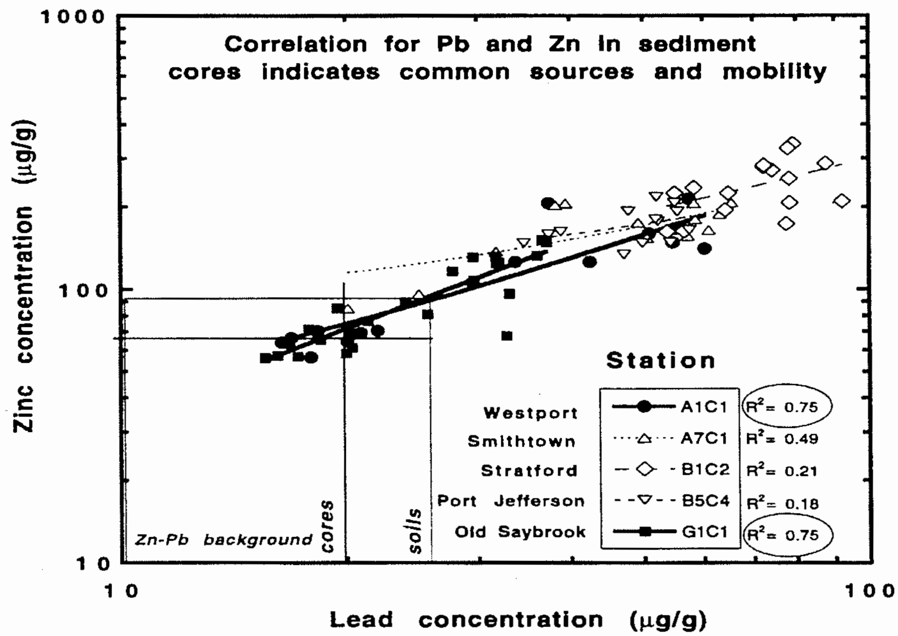


Figure 11. Correlation of contaminant metals Zn and Pb in subsurface sediment (core) samples.

The correlation of Pb and Zn in cores from depositional regions across the Sound (Figure 11) indicates common sources and mobility *in situ.* Lead has atmospheric and sewage-related sources, however its distribution in the sediment system is dominated by its interaction with particles. These pre-industrial estuarine values provide constraints on natural conditions and natural variability that are needed to be able to correctly identify where, and to what extent, pollution has cause elevated contaminant concentrations.

Accumulation rates. Profiles of fallout isotopes (^{137}Cs), naturally- occurring isotopes (^{210}Pb), and many metals profiles indicate that mixing depths vary greatly within Long Island Sound. Station A1, near Westport, meets boundary condition requirements for radiometric dating. It has preserved layering in x-radiographs of the core, a mixing depth of near one cm, a ^{210}Pb profile that indicates a constant accumulation rate over the upper 20 cm of the core, and a clear ^{137}Cs peak at 7 cm. Age assignments for this core (Figure 9a and 12) show the onset of a sewage record in the late 1800's with a marked increase of contaminant concentrations in the post WWII period. Profiles of Hg in well-dated, nearby marsh cores (Varekamp et al. 1998) have the same concentration features over this time period. Preliminary data for other cores in depositional regimes indicate that the depth of a well-mixed surface layer ranges from 5 cm to more than 20 cm and sediment accumulation rates, since the mid 1800's, are 0.2 to 0.5 cm yr^{-1}.

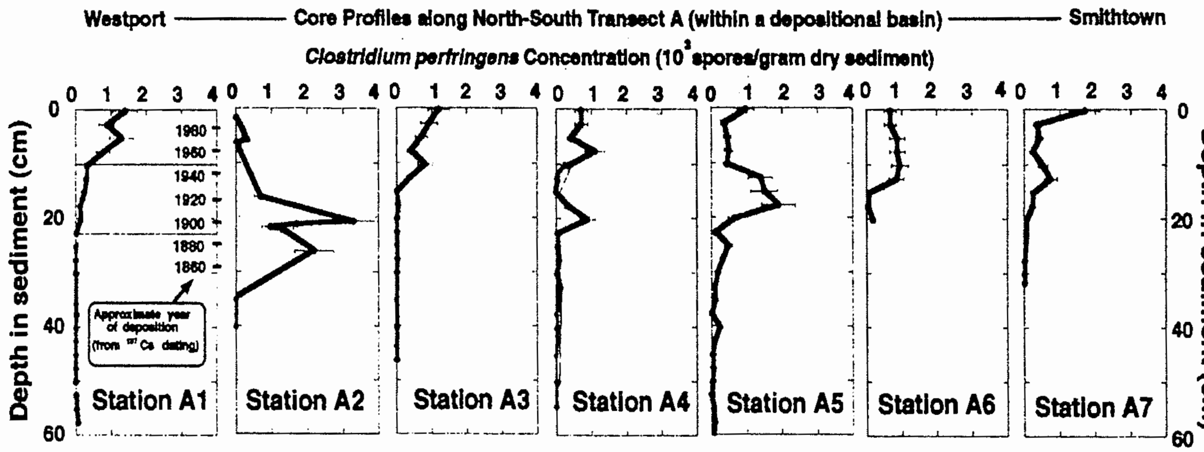


Figure 12. *Clostridium perfringens* profiles in sediment cores from a North-South transect in the western depositional basin (transect A).

The regional patterns identified in metal concentrations, transport processes, and chemical mobility offer a means to understand the broad scale behavior of contaminants in Long Island Sound. The extensive core collection allows study of smaller scale features and processes that create complexity within the broader picture. A transect of *Clostridium perfringens* profiles from north to south across the western depositional basin in Long Island Sound (Figure 12) illustrates the variation in sediment record that differing degrees of sediment winnowing, mixing, and source proximity can introduce into the sediment record. The mid-core peaks at stations A2 and A5 may result form nearby disposal events, the uniformity in station A6 probably reflects extensive biological mixing, and the steep gradients near the surface in cores at stations A1, A3 and A7 suggest that, unlike the metal input, the input of bacteria from sewage has not decreased in recent decades.

SUMMARY

• Results from chemical and lithological analysis of surficial samples and sediment cores collected in Long Island Sound delineate the distribution and accumulation of contaminated sediments in Long Island Sound.

• High concentrations and accumulation of contaminants follow the distribution of fine-grained sediments, reflecting widespread dispersal in the Sound, focusing of contaminant-bearing particles in regions of deposition, and removal or non-deposition of contaminants in other areas.

• Many contaminant metals (e.g., Cu, Cr, Zn) co-vary, indicating common sources and transport patterns within the basin. In depositional regions, surface concentrations generally are between ER-L and ER-M values for sediment quality criteria.

•*Clostridium perfringens* profiles show concentrations above background levels to depths of at least 30 cm in all regions. Contaminant accumulation patterns are predominantly influenced by features of the sedimentary environment and by source proximity; however, contaminant distribution is modified in the sediments by mixing depths and mechanisms that vary greatly within Long Island Sound.

• In depositional regions throughout the Sound, contaminant depth-profiles record the onset of the industrial era, a post-WWII period, and response to remediation activities in recent years.

• Sediments from surface grabs and sediment cores are currently being analyzed for many parameters; however, aliquots are available to interested collaborators. The data generated will complement existing data on contaminant distributions in LIS and allow a detailed reconstruction of the contaminant and sediment history of the Sound over the last hundred years.

ACKNOWLEDGMENTS

Many thanks and much appreciation is given to the numerous people who have contributed to various aspects of this work, and in many cases, continue to do so. Technical assistance in sample analysis and processing was provided by M. Casso, P. Gill, E. Galvin, S. McDaniel, and A. Jamil. R. Rendigs, E. Banks, B. Kreulen, A. Ouimette, K. Hall, J. Cookman, E. Young, R. Urbach, E. Duffy, A. Goss and Z. Cruz-Monserrate also contributed to sample collection and processing. *Clostridium perfringens* analysis was done by K. Feldman of Biological Analysis Laboratory/Thielsch Engineering, Inc., and trace and major element analysis was done in the laboratory of R. Murray at Boston University. Database collaborators include scientists at the United States Army Corps of Engineers, New England Division (USACE-NED), Science Applications International Corporation (SAIC), the National Oceanographic and Atmospheric Administration (NOAA), the United States Environmental Protection Agency (USEPA), Region I, the Long Island Sound Research Center, Wesleyan University, the State University of New York at Stony Brook, the University of Connecticut, and others. J. Reid and P. Hastings spearhead the database component of this work and its success owes much to the gracious collaboration of colleagues F. Manheim (USGS), T. Fredette (USACE-NED), P. Myre Murray (SAIC), E. Long (NOAA), M. Leibman (USEPA), and R. Lewis (Long Island Sound Research Center and State of CT). X-radiography was done with the generous assistance of the Radiology Department staff at Falmouth Hospital (B. Ekholm, M. Fishbein, J. Smith, and P. Nolan). The sediment geochemistry represented by this report is a part of an interdisciplinary, multi-institutional study of pollutant and sediment fate and transport in Long Island Sound. Our understanding of human impact on the sediment of this embayment would be lessened without the contributions of colleagues L. Poppe (USGS), H. Knebel (USGS), R. Signell (USGS), J. Latimer and D. Keith (both at USEPA), R. Murray (Boston University), and J. Varekamp (Wesleyan University). Funding for this work is provided by the USGS Coastal and Marine Program.

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INVESTIGATIONS OF BACKGROUND CONTAMINANT LEVELS IN WESTERN LONG ISLAND SOUND

Saffert, H.L., P.M. Murray and D.A. Carey, Science Application International Corporation, Newport, RI 02840; T.J. Fredette, U.S. Army Corps of Engineers, New England District, Concord, MA 01742

ABSTRACT

Over the last decade, sediments from western Long Island Sound (LIS) have been tested by the New England District of the U.S. Army Corps of Engineers (NAE) and the U.S. Environmental Protection Agency (EPA) for the purposes of dredged material permitting and monitoring. Background, or reference, contaminant levels are compared to measured values at the nearby open-water Western Long Island Sound Disposal Site (WLIS). Reference information is required for the dredging and disposal permit application process and for monitoring at the disposal site. We integrated the reference area data with regional data from the National Oceanic and Atmospheric Administration's National Status and Trends Program and EPA's Environmental Monitoring and Assessment Program. Because of long-term environmental impacts to the western LIS region, including the presence of historic dredged material and other contaminant sources, background sediment concentrations are variable and patchy. The data indicated that the highest concentrations in the study area commonly were measured near WLIS, partially due to the concentrated sample distribution near WLIS, as well as the proximity of the samples to historical dumping areas. We did not address any data collected within WLIS; the site is actively managed by NAE and, therefore, was not relevant to this study of background contaminant levels.

INTRODUCTION

The western Long Island Sound (LIS) coastline is densely populated and industrialized, resulting in various point and non-point sources releasing contaminants to the waters and sediments of the Sound over time (Turgeon et al. 1989). Although awareness of pollution has increased resulting in reduction of some contaminant sources, the sediments of industrialized harbors still retain accumulated historic contaminants. When dredging of these harbors is necessary, measurement of contaminant levels is required by current management practices of potentially contaminated dredged sediments.

History of Dredging and Disposal Activities in Long Island Sound. Sediments from shipping ports and waterways along the Connecticut and New York coasts have been dredged and disposed in LIS since the late 1800s, long before a developed management plan was in operation. In 1954, eight disposal sites were identified for western LIS (Figure 1). From 1954 to 1972, an estimated total of 22 million cubic yards of dredged material was disposed at these eight sites, with sixty percent (13.2 million cubic yards) at Eaton's Neck which is close to the Western Long Island Sound Disposal Site (Fredette et al. 1993). These historic sites continued to receive large volumes of dredged material until the late 1970's. In 1982, the New England

District of the U.S. Army Corps of Engineers (NAE) selected WLIS as the only active site of the region. Dredged material disposal, from approved and permitted dredging projects, has been directed to positioned buoys in the southwestern quadrant of WLIS and carefully monitored and managed under the auspices of the Disposal Area Monitoring System (DAMOS) Program (USACE 1998). Since 1986, a permit chemistry database has been compiled on the dredged sediments disposed at WLIS. Although this paper does not include sediments tested from within the active disposal site, the majority of the contaminant concentrations of permitted harbor sediments have been within or below acceptable reference values (Murray 1998).

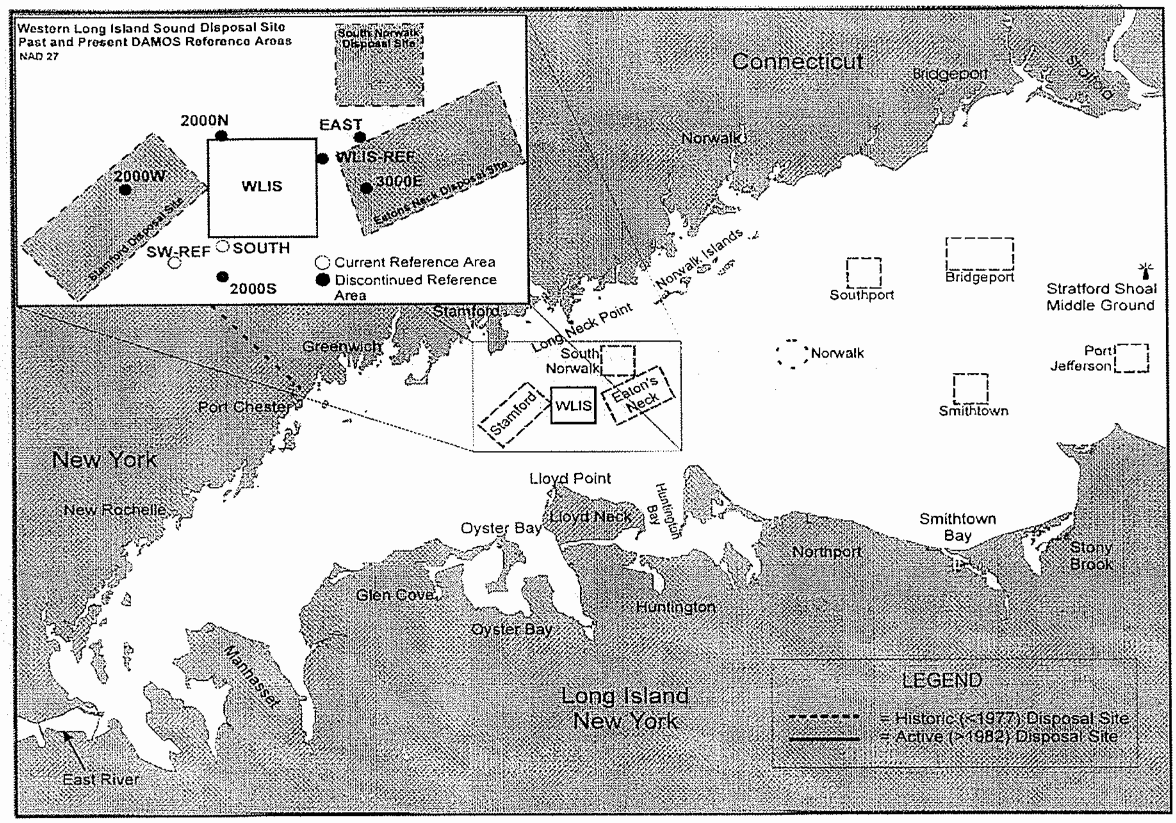

Figure 1. Locations of current and discontinued DAMOS WLIS reference areas relative to WLIS and discontinued historic disposal site boundaries.

Utility and Importance of Reference Areas. To assess the environmental impacts of dredged material disposal, DAMOS protocols utilize monitoring data collected from three reference areas surrounding WLIS (Germano et al. 1994). Reference areas provide, ideally, a control region of ambient sediments for comparison with results of monitoring surveys at the disposal sites. Selection of a reference area is based on proximity to the disposal site, sediment chemistry and similar water depths and grain size (EPA and USACE 1991). Reference sediments should be representative of the conditions that would exist in the vicinity of the disposal site had no dredged material disposal occurred.

The widespread presence of historic dredged material in the region surrounding WLIS has complicated the ongoing search for suitable reference areas for this disposal site. Two sites are currently used (SW-REF and SOUTH, Figure 1), but the search for an acceptable third reference area has been hampered by evidence of historic dredged material in the many locations investigated. Since 1991, dredged material has been detected using sediment profile imaging and other techniques at several former WLIS reference areas, including 3000E, EAST, WLIS-REF, 2000S and 2000W which were thus discontinued (Figure 2; Eller and Williams 1996; Charles and Tufts 1996; Morris 1998). Dredged material also was detected in sediment images collected to the north and northwest of WLIS and in a side-scan survey just south of Eaton's Neck Disposal Site. A region relatively free of dredged material deposits, called SE-REF, is currently being evaluated to serve as the third reference area for WLIS (Saffert and Murray 1998).

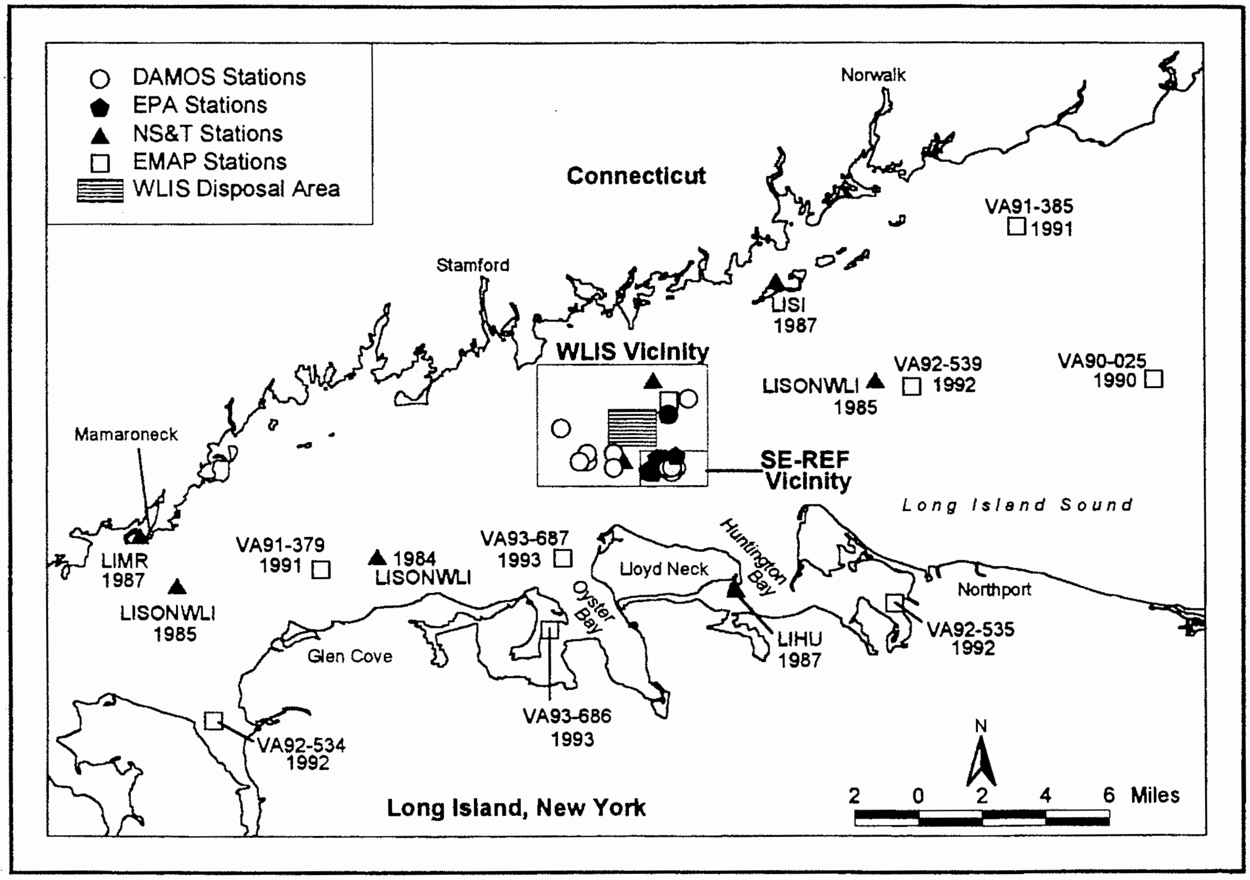

Figure 2. Sampling stations in the western Long Island Sound region.

In addition to serving monitoring purposes, reference areas are now used in determining the suitability of dredged material for disposal through the permit evaluation process. Classification of sediments is important for assessing disposal site selection for dredging projects. Only those sediments deemed suitable for open-water disposal are currently disposed at WLIS. In addition, dredging operations are restricted generally to the period from the end of September to May.

METHODS

Sediments from LIS have been collected and analyzed by a number of agencies and researchers. We have integrated selected data to provide a larger context to view the local reference area surrounding WLIS (Table 1, Figures 3 and 4). Detailed descriptions of sample collection, laboratory methods and quality control reviews may be found in the referenced literature. A brief review is provided for each program in addition to information on data qualification, normalization and statistical procedures.

Table 1 Sediment Chemistry Datasets

	WLIS Reference Areas		Regional Stations	
Data Source:	DAMOS*	EPA*	EMAP*[1]	NS&T*
Year				
1998	X			
1995		X		
1993	X		X	
1992	X	X	X	
1991	X		X	
1990	X≠		X	X≠
1984-89	X≠			X

*Acronyms explained in text. X≠ Sample data not included in analysis.

[1]Within the EMAP metadata, the EPA requires that the following statement be included when publishing any of the EMAP data. "Although the data described in this article has been funded wholly or in part by the U. S. Environmental Protection Agency through its EMAP-Estuaries Program, it has not been subjected to Agency review, and therefore, does not necessarily reflect the views of the Agency and no official endorsement should be inferred."

DAMOS - Disposal Area Monitoring System Program. During DAMOS surveys, three reference areas were sampled during monitoring surveys at the disposal sites, providing historic data sets. Available data collected prior to 1991 were not included to maintain consistency and reliability of the analysis. Chemistry data were collected in 1991 (Williams 1995), 1992 (Eller and Williams 1996), 1993 (Charles and Tufts 1996) and 1998 (Saffert and Murray 1998).

EPA. The EPA Region I office sampled reference areas from WLIS in June 1992 (Metcalf and Eddy 1992) and again in 1995 (Leibman, personal communication). DAMOS-designated reference areas were sampled as well as alternative areas.

EMAP - Environmental Monitoring and Assessment Program. EMAP, initiated by the EPA's Office of Research and Development, was a nationwide program developed to study pollution in the coastal environments and impacts on ecological resources (Strobel et al. 1995). The LIS sediment sample data in this report are from a four-year study (1990-1993) in the estuaries of the Virginia Province extending from Cape Cod to Chesapeake Bay.

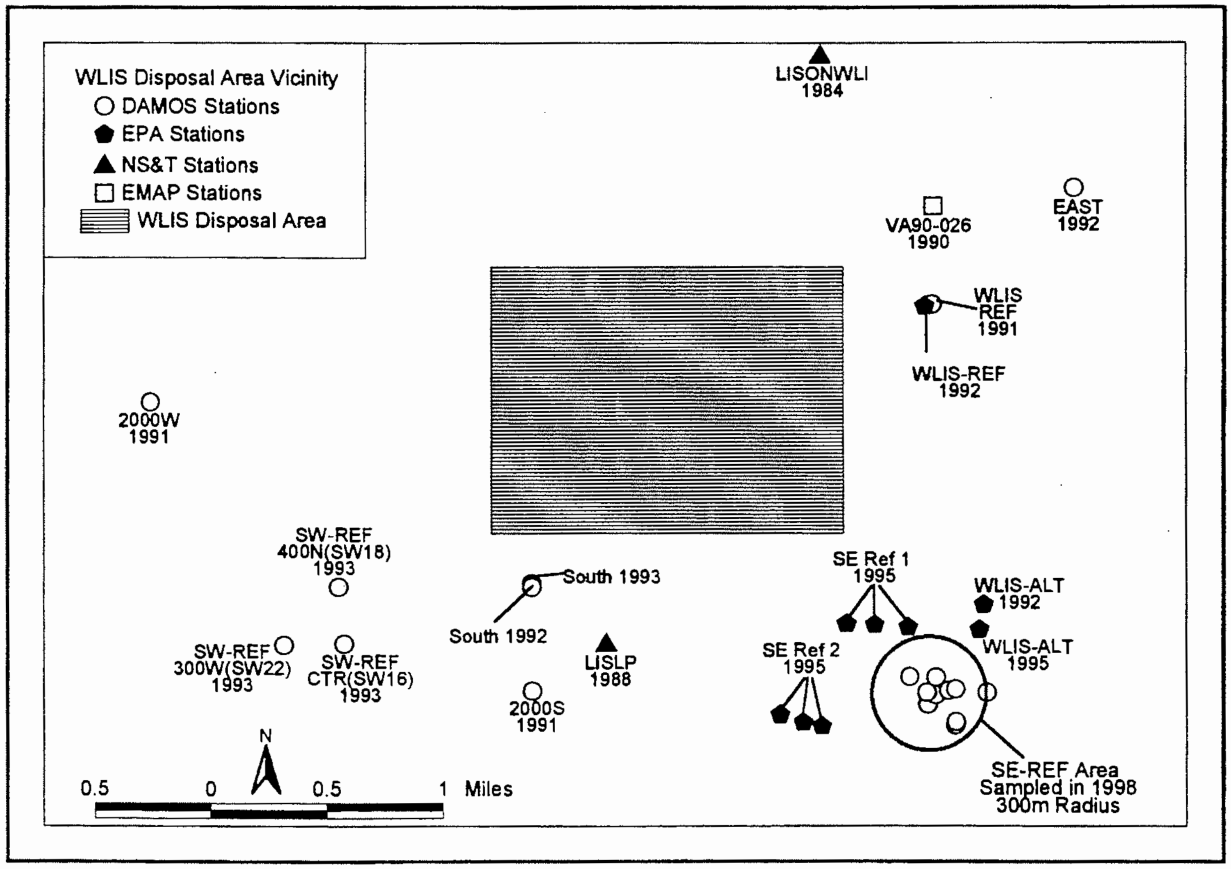

Figure 3. Sampling stations in the western Long Island Sound.

NS&T - National Status and Trend Program for Marine Environmental Quality. The National Oceanic Atmospheric Administration NS&T Program analyzed samples of surface sediment collected at almost 200 coastal and estuarine sites throughout the United States since 1984 (NOAA 1988, 1991). Similar to EMAP, the project was designed to evaluate the status of sediment contaminant levels and the biological effects of pollution in order to detect changes over time. NS&T sites were chosen to "quantify general, depositional areas of contamination and not to define 'hot spots'" (NOAA 1988). Major point sources were deliberately avoided so that the combined influence of pollution sources could be identified. We selected sites that were spatially appropriate for the region surrounding WLIS. At a few stations, the 1986 data overlapped with stations from earlier and later years and, therefore, were not shown on the maps.

Selected Contaminants Tested and Laboratory Methods. The contaminants of concern tested in the data collection efforts generally included trace metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel and zinc), polychlorinated biphenyls (PCBs), pesticides and polynuclear aromatic hydrocarbons (PAHs). In the DAMOS and EPA reference area data sets, sediment concentrations of PCBs and pesticides were generally low or below detection levels. Therefore, we focused our mapping efforts on selected trace metals and PAHs. The DAMOS and EPA programs used EPA SW-486 laboratory methods and quality control procedures (EPA 1997). The NS&T and EMAP program tested the same suite of analytes and followed program-specific methods and stringent quality control procedures. Generally, the NS&T and EMAP had lower detection limits for analytes than EPA and DAMOS.

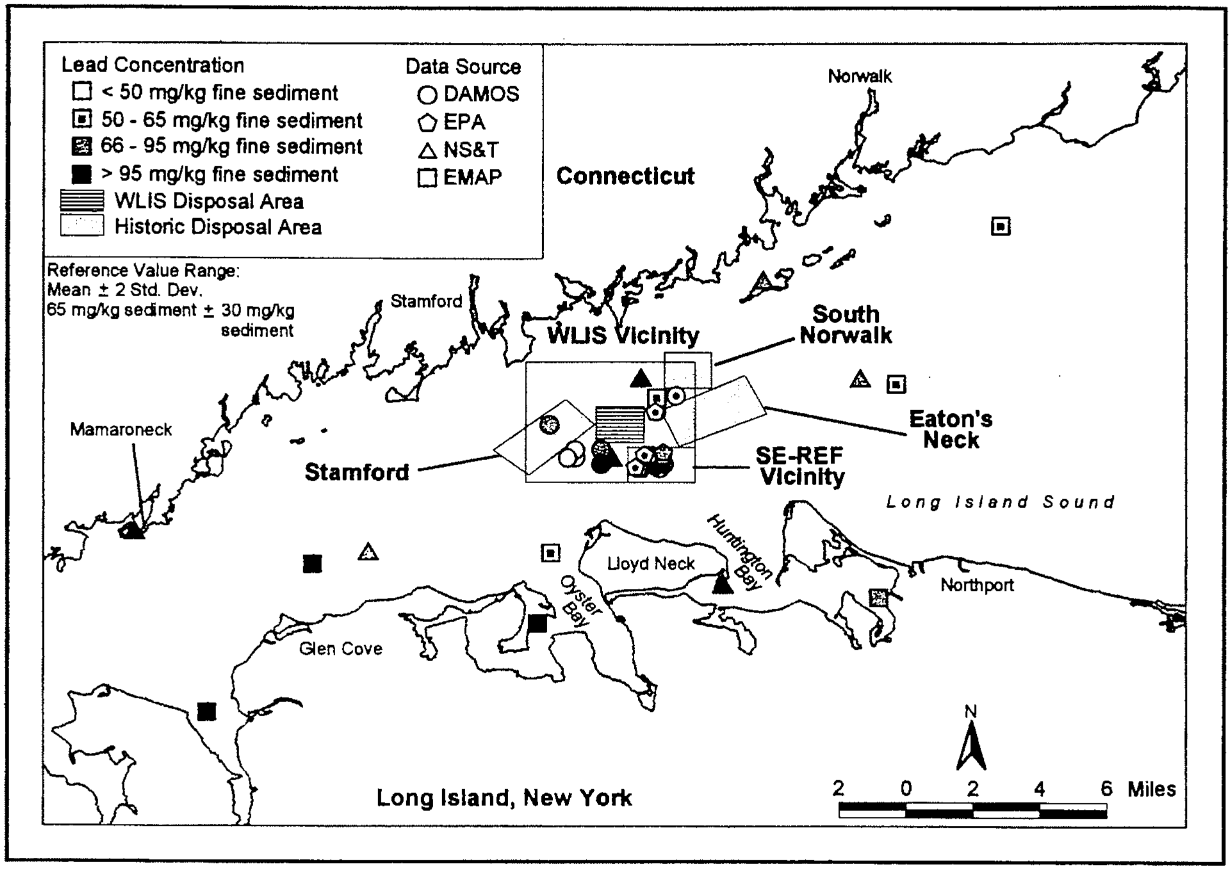

Figure 4. Lead concentrations normalized to fine-grained sediment in western Long Island Sound sediments.

Data Qualifications. A few qualifications of the data are necessary to clarify spatial distribution of contaminants. Some variability in the data is due to discrepancies in the sample techniques and laboratory analytical methods. All the data were reviewed for quality assurance. Because we did not have enough data from multiple years at the same stations, we did not perform a temporal analysis of the data. Some discrepancies also may be due to difference in surface sample depth. For instance, the EMAP program homogenized only the top 2 cm of sediment, where as the DAMOS program typically collected the top 10 cm of sediment.

Data Normalization. Data were normalized to allow for comparison among data sets. Metals were normalized to fine-grained sediments (total percent silt and clay) and reported in units of milligrams per kilogram of fine sediment (mg/kg-fine sediment), where as PAHs were normalized to total organic carbon (TOC) and reported in units of micrograms per kilogram of TOC (μg/kg-TOC). Normalization is a tool that is used to adjust for variation in sediment characteristics that may influence the contaminant distribution. For instance, a strong correlation exists between fine-grained sediments and trace metal concentrations (NOAA 1988). The percentage of TOC is also positively correlated with contaminants of concern, especially organic constituents. Particulate and organic matter, because of its fine grain size, surface charge, high surface area to volume ratio and microbial coatings serve to sorb or chelate organic contaminants and trace metals (e.g., Murray et al. 1995; Horowitz 1985). One EMAP sample, located near the CT shore between Greenwich and Stamford, had a fine sediment fraction below 20% and a very low percent TOC and, therefore, was not included. The majority of samples contained >40% silt and clay fraction and had relatively high TOC percentages.

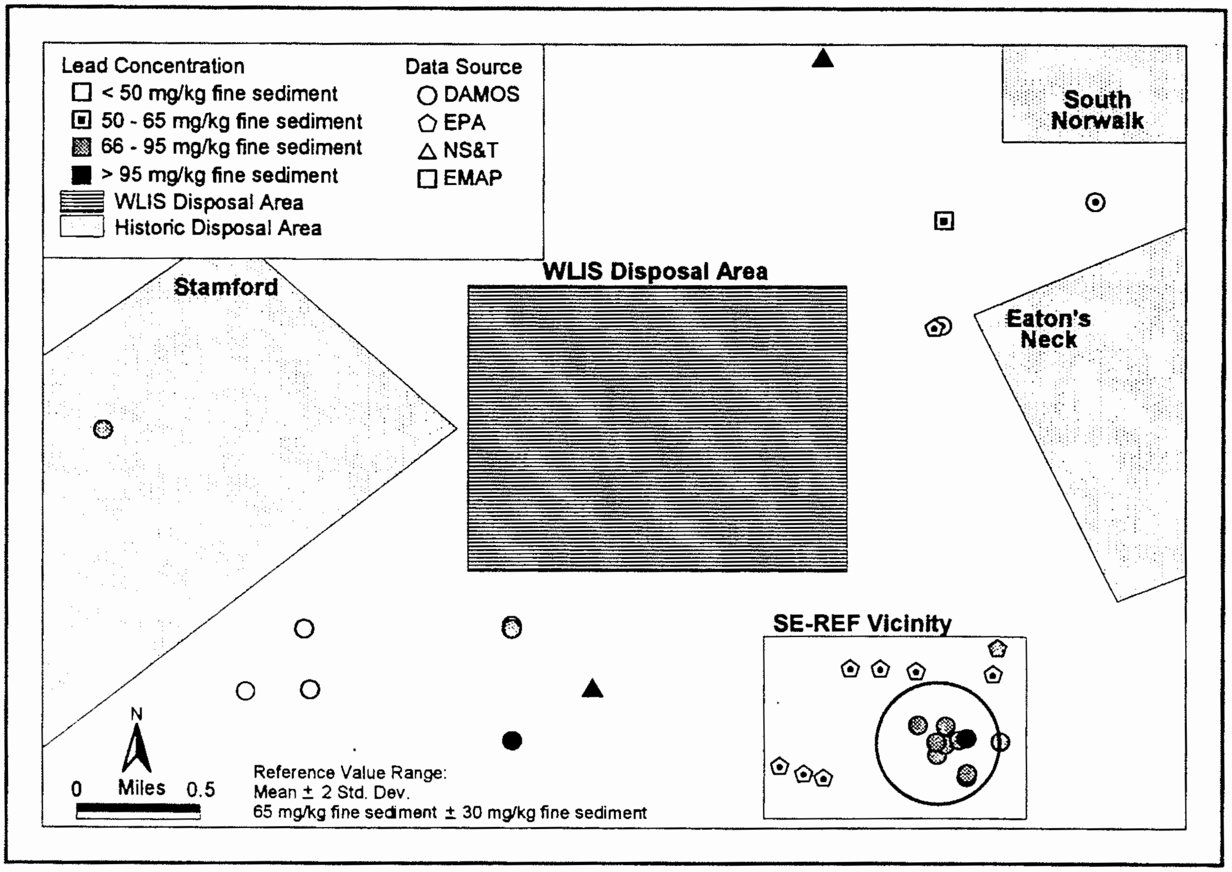

Figure 5. Lead concentrations normalized to fine-grained sediment within the WLIS vicinity.

Data Display and Statistical Calculations. Because of the difference in sampling protocols between EPA and NAE (DAMOS), replicate samples were handled differently for data summary and display. If replicate samples were collected at the same geographic coordinates, those data were averaged and a single value was used to represent that location. If replicates were collected at unique geographic coordinates, those samples were treated as individual values; and each value was displayed on the maps in the **RESULTS** section.

Following compilation of all of the sediment chemistry data in the western LIS region, data were scaled for mapping purposes relative to the Reference Area Database screening values (Murray 1995). The Reference Area Database was compiled using data collected at accepted DAMOS WLIS reference areas (SW-REF [1993, 1995] and SOUTH [1992, 1993]) and included data collected by EPA at other sites considered to be relatively free of dredged material (WLISREF [1992], WLIS-ALT [1992, 1995] and SE-REF 1 and 2 [1995]). These values currently are used for the purpose of permitting open-water dredged material disposal. Data collected at proposed dredging sites that exceed the mean reference screening value by more than two standard deviations (2sd) are flagged for further testing (Murray 1995). Using the WLIS reference area data for each contaminant, the values collected for this study were scaled relative to one and two standard deviations of the mean, lower and upper limits respectively.

RESULTS

Distribution of Trace Metals. Two metals were selected for presentation, lead (Pb) and zinc (Zn). Because the sampling density is higher near the vicinity of WLIS, metal distributions will be discussed both for regional western LIS, and for the area surrounding WLIS. Regionally, normalized Pb concentrations were greater than two standard deviations above the reference value mean (65 mg/kg-fine sediment) among stations along the shore and in the western end of the Sound (Figure 5). Exceptions to this pattern were several stations in the WLIS vicinity that exhibited high Pb concentrations, including the NS&T stations and two DAMOS areas (Figure 6). In fact, the maximum Pb concentration of the survey area was at 2000S, located south of WLIS (Figure 2), with a normalized concentration of 167 mg/kg-fine sediment. Lead concentrations surrounding WLIS exhibited a patchy distribution, with concentrations falling among all categories relative to the reference value (mean ± 2 sd), including samples greater than 2sd located south, southeast and north of WLIS (Figure 6).

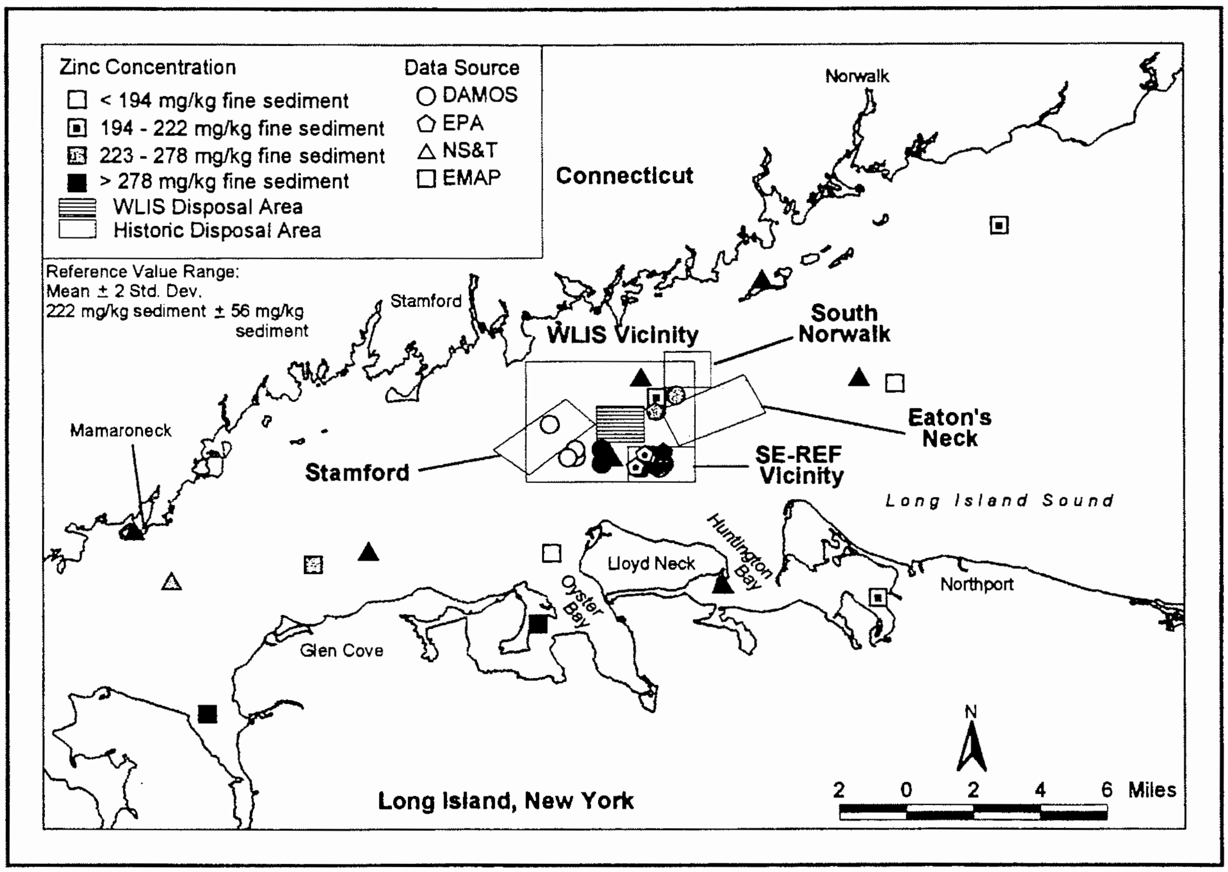

Figure 6. Zinc concentrations normalized to fine-grained sediment in western Long Island Sound sediments.

The regional distribution of normalized Zn concentrations was similar to that of Pb (Figure 7), although one NS&T sample in the farthest eastern edge of the region was in the highest Zn category (>278 mg/kg-fine sediment). The highest individual Zn value was again measured south of WLIS, at NS&T Station LISLP (1988) located near 2000S (Figure 4), with a concentration of 568 mg/kg-fine sediment (Figure 8). The lowest concentrations of both Pb (27 mg/kg-fine sediment) and Zn (104 mg/kg-fine sediment) were measured at the proposed DAMOS Reference Area SE-REF.

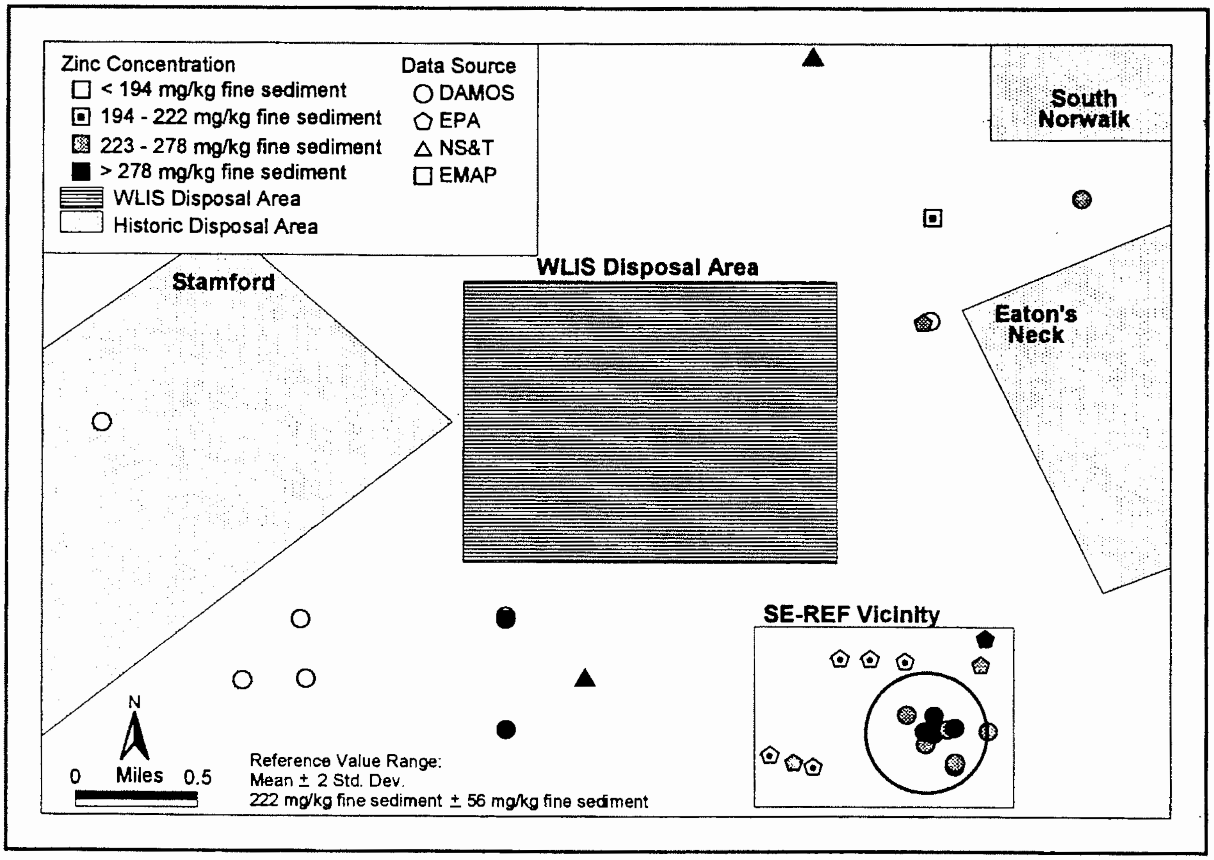

Figure 7. Zinc concentrations normalized to fine-grained sediment within the WLIS vicinity.

Distribution of Polynuclear Aromatic Hydrocarbons. Two PAH compounds were selected for presentation, one low molecular weight (LMW) PAH compound (naphthalene) and one high molecular weight (HMW) compound (pyrene). Inferences of contaminant sources can be drawn based on the different nature of the source of LMW (primarily petrogenic) and HMW (primarily pyrogenic) compounds. Petrogenic PAHs are typically associated with point sources, whereas pyrogenic compounds are usually derived from non-point sources (e.g., atmospheric deposition).

The normalized concentrations of naphthalene were within 2 sd of the mean (< 6,340 µg/kg-TOC) at almost all stations except one EMAP station located in the eastern edge of the study area (Figure 9), and two DAMOS stations near WLIS (2000S and WLIS-REF, Figure 10). Napthalene values ranged from 240 (SE-REF) to 36,889 µg/kg-TOC (2000S). In contrast to the metals, the near-shore naphthalene concentrations were commonly less than the mean reference value (2,660 µg/kg-TOC), and similar or less than those measured in the deeper basin of the Sound (Figure 9). Naphthalene values were generally greater than the mean reference value except in the vicinity around the proposed Reference Area SE-REF (Figure 10).

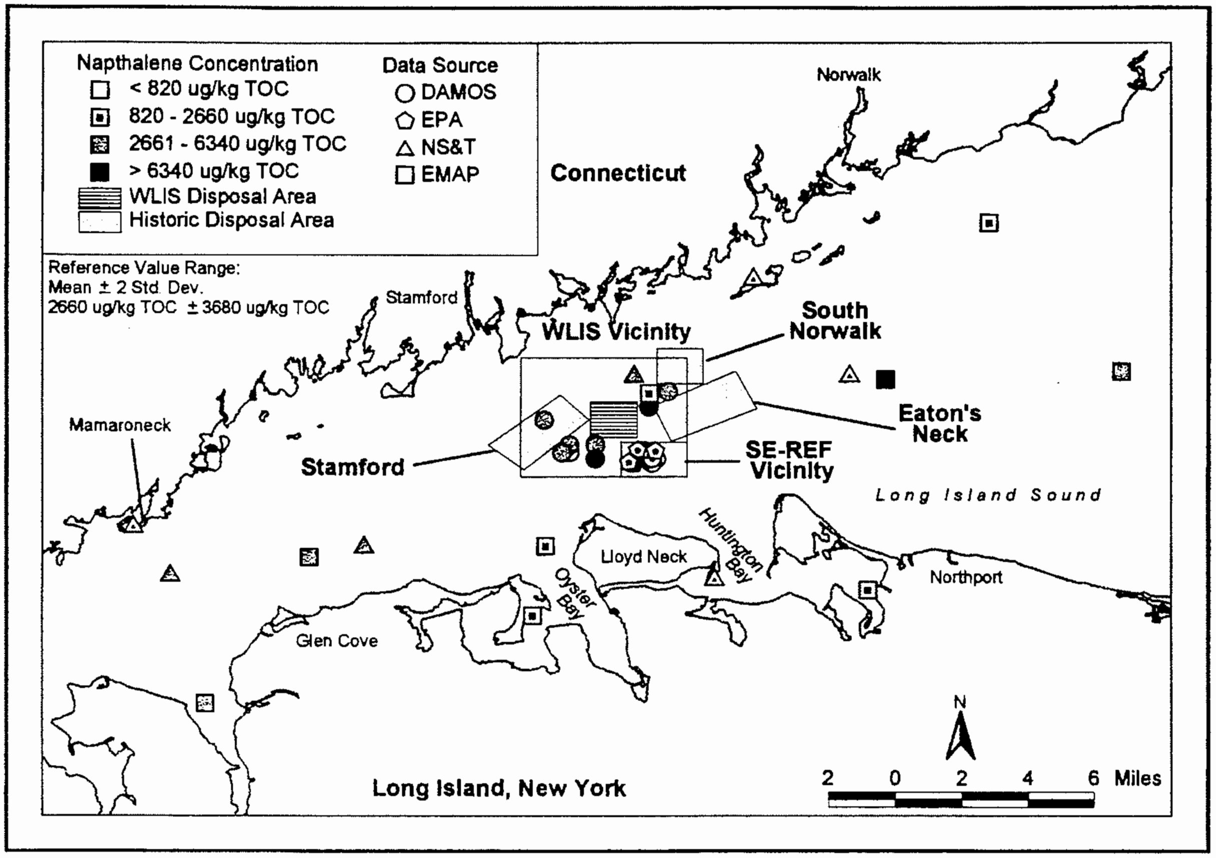

Figure 8. Napthalene concentrations normalized to total organic carbon in western Long Island Sound sediments.

The highest normalized concentrations of pyrene in the study area were around WLIS (Figure 11). Except for the WLIS area and one NS&T station located east of WLIS, pyrene values were less than the mean reference value (29,140 µg/kg-TOC). There was no apparent geographic pattern, neither near-shore and basin nor western and eastern variation, in the pyrene data, consistent with the predicted even distribution of pyrogenic compounds. The greatest range of normalized pyrene concentrations was near WLIS, ranging from 960 (SE-REF) to 213,674 µg/kg-TOC (2000S), indicating that historical dumping is also a source of pyrene to the area surrounding WLIS (Figure 11).

DISCUSSION

National Comparison. The western LIS data exhibited a wide range of contaminants, consistent with the results of national NS&T and EMAP studies of estuarine and coastal areas. Although in general, contaminant concentrations were consistent with observed increased concentrations towards the western end of LIS with proximity to New York and other industrialized harbors (Turgeon et al. 1989), the presence of local contaminant sources as sampled near WLIS masked this overall trend. NS&T compared regional background contaminant levels nationally and compiled a list of the highest 25 concentrations for each analyte. If "hot spots" near point sources had been sampled, these sites would exceed almost all of the values of the sites tested and would replace the top NS&T ranked sites (NOAA 1988). The current mean reference values used for comparison to WLIS were lower than measured among the top 25 NS&T stations for both Pb (concentrations of >100 mg/kg-fine sediment) and Zn (concentrations of >290 mg/kg-fine sediment; NOAA 1988). The highest Pb value at the discontinued reference area 2000S was comparable with data from a station located in Boston Harbor, MA, ranked fourteenth in NS&T's background contaminant study. Lead levels generally were comparable to those observed at stations in other east coast estuaries including Narragansett Bay, RI and Cape Ann, MA. The highest Zn concentration observed near WLIS (NS&T LISLP) was relative to Raritan Bay, New Jersey.

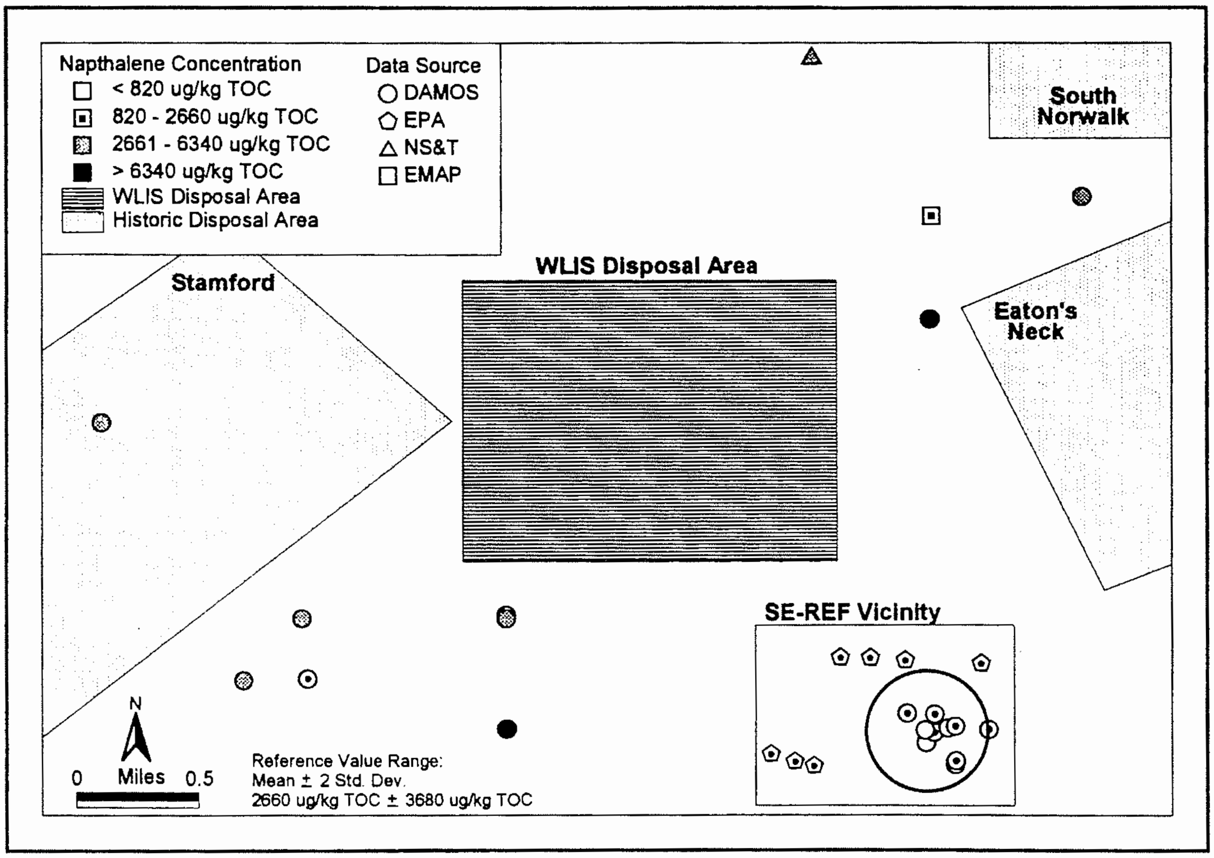

Figure 9. Napthalene concentrations normalized to total organic carbon within the WLIS vicinity.

Naphthalene and pyrene values also were highly variable in the selected region. Most of the mapped stations were below the WLIS reference area mean. The NS&T station with the highest pyrene value (69,133 µg/kg-TOC) was ranked as the 25th highest NS&T station (NOAA 1989). The discontinued DAMOS Reference Station 2000S was comparable to some of the highest measured values in the NS&T Program (Hudson River and Raritan Bay).

Contaminant Levels Near WLIS. The highest concentrations of all of the contaminants discussed in this paper were located near WLIS, most consistently at the discontinued 2000S (DAMOS) site for both metals and PAHs. The localized presence of contaminated sediment near WLIS was supported by similar results from an NS&T station located nearby (LISLP).

Because this area is not located specifically within a historical disposal site, the presence of contaminants at this station indicates that dredged material disposal may have been more widespread than within the named sites, or that there are other point sources of contaminants that are not associated with dredged material, or both. The historical dumping grounds may have been used for disposal of industrial waste and construction materials in addition to dredged material.

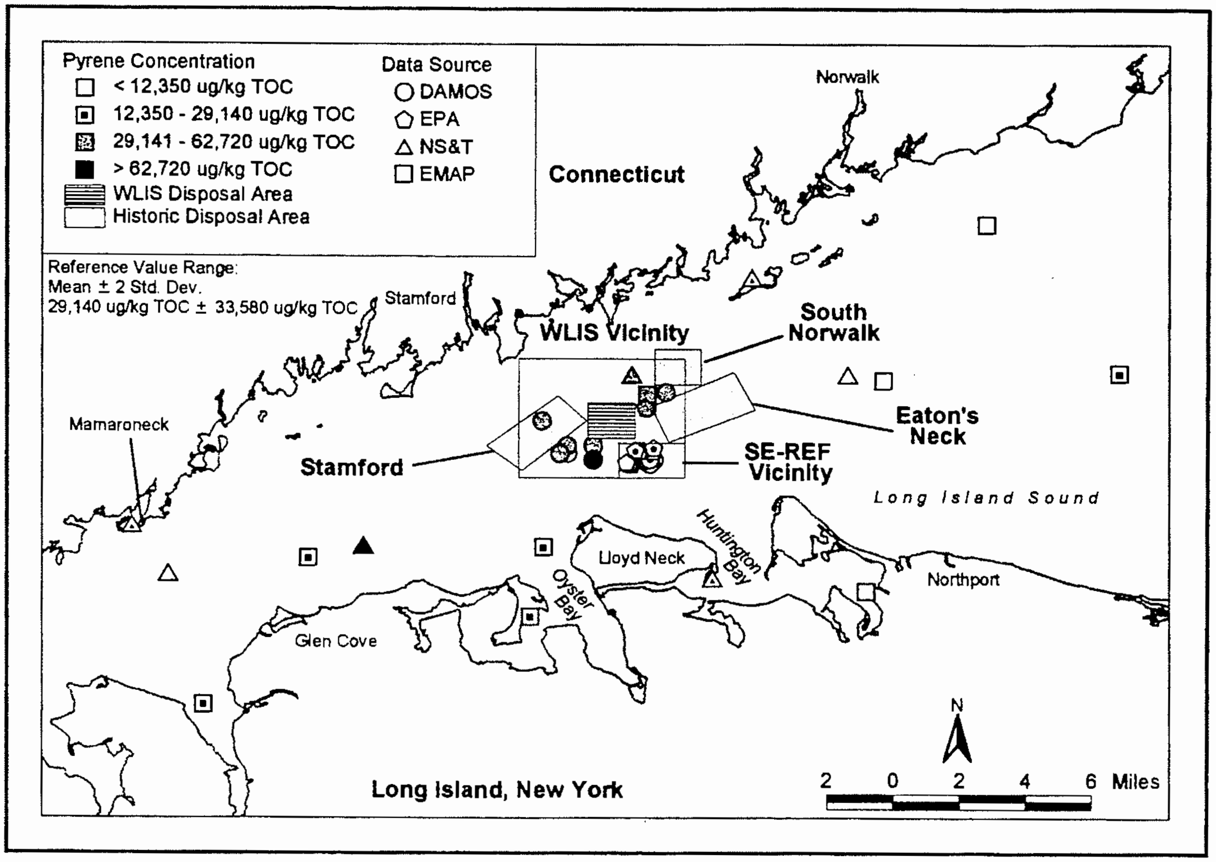

Figure 10. Pyrene concentrations nromalized to total organic carbon in western Long Island Sound sediments.

. Due to the growing concern for environmental protection in the 1970's, ocean disposal was restricted and is now carefully regulated; management and monitoring practices for dredged material have evolved. Since 1982, sediment permitted for disposal has been carefully tested and evaluated (Carey 1998). As a result, the active disposal site (WLIS), although not explicitly investigated in this study, is not likely to serve as a source of contaminated sediments.

Although the presence of higher concentrations of contaminants surrounding WLIS suggest that historic dredged material may be a source of contaminants to sediments of LIS, the selection of sample stations tends to bias the results. Because of the higher density of samples around WLIS, the variability is consequently higher. It is likely that if the same density of samples were collected at some other location within western LIS, a similarly large range and variability of contaminant concentrations would be measured. The variability of the few EMAP and NS&T samples collected in the region is consistent with this conclusion, as well as preliminary data from the U.S. Geological Survey for Pb and Zn levels which also show variability in western and central LIS (USGS 1998).

Implications for Use of the Reference Area Database. Compilation of sediment chemistry data from the region surrounding active dredged material disposal provides statistical data for establishing background levels required for dredging permits and the regulatory process. The Reference Area Database also provides a management tool for evaluating potential and current reference areas for the disposal site. Additional data will be included to the Reference Area Database only from reference areas that meet specific criteria to continue to refine background levels (EPA and USACE 1991). Because of the potential for alternative sources of contaminants other than dredged material, however, corroborative evidence for the presence of dredged material (i.e., acoustic, sediment profile images) should be used for determination of reference area suitability.

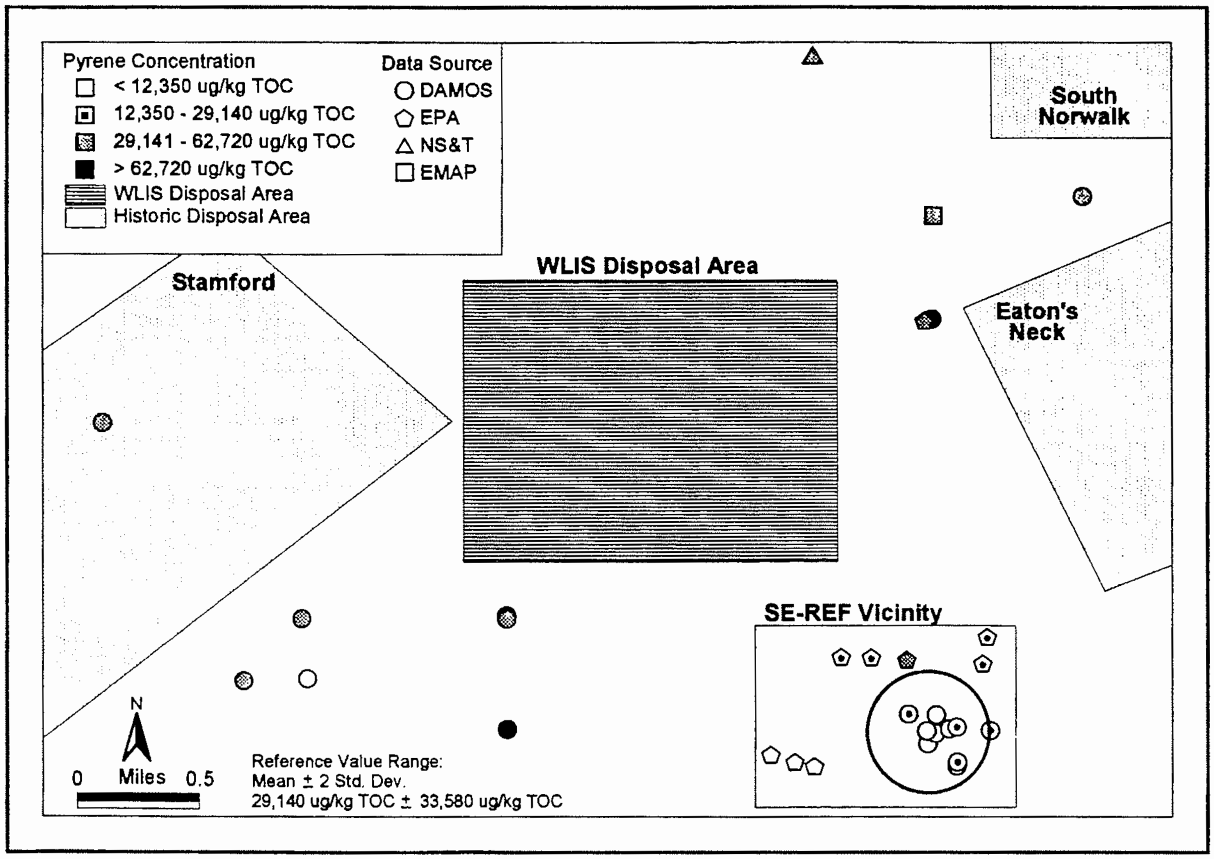

Figure 11. Pyrene concentrations normalized to total organic carbon within the WLIS vicinity.

Outlook. Historic dredged material, which has been present on the seafloor for at least the last twenty years, may still be a source of contaminants to the region; but the pollution signal over time will most likely be diluted with the continual accumulation of sediments in this depositional region (Rhoads et al. 1995). Dredged material management approaches since the early 1980's have resulted in controlled disposal operations and decreased contaminant levels of sediments disposed at WLIS. In the long-term, using the ongoing regulatory testing protocols and DAMOS management, dredged material will continue to have a decreasing impact on regional sediment quality.

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DISTRIBUTION OF TRACE METALS IN THE SEDIMENTS AND WATER COLUMN OF A SEMI-ENCLOSED BAY: PORT JEFFERSON HARBOR, NY

Breslin, V.T. and S. Sañudo-Wilhelmy, Marine Sciences Research Center, State University of New York, Stony Brook, NY 11794-5000

Eighteen sediment samples and six water column samples were collected in a small (6 km^2), coastal embayment (Port Jefferson Harbor, NY) to define high resolution spatial distributions of metals and to elucidate sources of contaminants to the harbor. Sediment metal (Ag, Cu, Fe, Ni, Pb, V and Zn) concentrations varied widely. Fe concentrations ranged from 0.44 to 3.55%, Cu from 0.1 to 86 $\mu g\ g^{-1}$, Pb from 3.8 to 58.9 $\mu g\ g^{-1}$, Zn from 14.3 to 191 $\mu g\ g^{-1}$, Mn from 85.9 to 542 $\mu g\ g^{-1}$, Ni from 0 to 29.7 $\mu g\ g^{-1}$, V from 11.6 to 89.9 $\mu g\ g^{-1}$ and Ag from 0.09 to 1.52 $\mu g\ g^{-1}$. The range of metal contents reflected differences in sediment grain size, with higher metal concentrations located in the fine-grain inner harbor sediments. Calculated enrichment factors for these sediment metals showed that Ag, Pb, Cu and Zn are elevated relative to both crustal abundances and their respective abundances in sediments in central Long Island Sound. Spatial variations in trace metals in surface waters within the bay paralleled the spatial variations of trace metals in sediments within the harbor. Water column Fe, Ni, Mn and Ag concentrations increased 1.2-10 fold in the inner harbor surface water in comparison to Long Island Sound water collected outside the mouth of the harbor. Elevated water column metal concentrations appeared to be partially derived from diagenic remobilization of metals from contaminated sediments within the southern portions of the harbor.

TEMPORAL AND SPATIAL VARIATION IN CONTAMINANT METALS IN LONG ISLAND SOUND SURFACE SEDIMENTS: 1972-1996

Brownawell, B.J. and V.T. Breslin, Marine Sciences Research Center, State University of New York, Stony Brook, NY 11794-5000

Long Island Sound (LIS) sediments are enriched in contaminant metals due to anthropogenic inputs via treated sewage discharges, atmospheric deposition of particulates and industrial wastewater discharged into rivers and harbors. As a result, metal contents in LIS sediments are higher than in many other comparable US coastal waters. In this study, we measured the Ag, Cd, Cu, Ni, Pb, Zn and total organic carbon (TOC) content of surface (0-3 cm) sediments from 35 stations along transects from western to central Long Island Sound. TOC-normalized sediment metal contents were then used to assess the spatial distribution of metals in LIS and to compare results of this study to previous studies to assess changes in the anthropogenic contribution of these metals over the past 25 years. Sediment metal contents varied for Ag (0.11-5.8 $\mu g\ g^{-1}$), Cd (0.2-2.3 $\mu g\ g^{-1}$), Cu (5.2-132 $\mu g\ g^{-1}$), Ni (2.0-28.4 $\mu g\ g^{-1}$), Pb (10.7-132 $\mu g\ g^{-1}$) and Zn (9.6-244 $\mu g\ g^{-1}$). In general, the highest metal concentrations were measured in western LIS sediments. West-to-east metal gradients were largely reduced when the metal concentrations were normalized to TOC. All metals were highly correlated to the TOC content of the surface sediments. Linear regression analysis showed that Ag (r^2=0.76), Cu (r^2=0.92), Zn (r^2=0.95) and Pb (r^2=0.93) can be predicted reasonably well by sediment TOC. A comparison with previous data sets shows that Zn, Ni and Cu concentrations in surface sediments have not significantly changed since the early 1970s. In contrast, increases in the Ag and Pb contents in surface sediments are apparent. Increases in Pb may reflect auto emissions while Ag increases in sediment may be associated with continued sewage inputs.

MERCURY POLLUTION IN AND AROUND LONG ISLAND SOUND

Kreulen, B. and J.C. Varekamp, Environmental and Earth Sciences, Wesleyan University, Middletown, CT 06459 and M.R. Buchholtz ten Brink, United States Geological Survey, Woods Hole, MA 02543

We have analyzed surface sediments from Long Island Sound (LIS) and sediment cores from the surrounding salt marshes for Hg and several major and trace elements (Pb, Zn, Cu). The marsh cores are dated with ^{210}Pb and ^{14}C as well as glitter horizons. LIS data show a a strong east-to-west gradient in Hg concentrations, which correlates with the abundance of fine-grained material. When normalized on a grainsize proxy, Hg levels are of the same order of magnitude throughout LIS. The marsh core data show strong Hg enrichments in the top 20-30 cm of the cores, commonly with declining levels in the top few cm. Common Hg abundance levels at peak pollution are about 300-500 ppb, whereas in the Housatonic estuary levels of 1500-2000 ppb Hg are encountered. The analytical data are converted into Hg accumulation rates with the sediment density and age models, leading to peak pollution fluxes in the marshes of 10-35 ng Hg/cm^2/yr. The onset of Hg pollution occurred mainly in the middle of last century, and peaked in the late 1960s-1970s. Degrees of pollution are largely related to the accumulation rate of fine-grained sediment, and muddy sequences have higher excess Hg inventories than organic rich peats. Excess Hg inventories range from 370-22,000 ng/cm^2, with the highest inventories in muddy freshwater swamps along the Connecticut River. The modern Hg accumulation rates re 30-50% lower than those during the peak period. These trends correlate very well with the Hg consumption pattern in the USA. Our current understanding is that Hg is deposited in the watersheds largely by atmospheric deposition, and polluted sediment grains are transported to the coast where they accumulate in the marshes and LIS. The *in situ* depositional flux in the marshes from the atmosphere is an order of magnitude lower than the particle-bound import. The very high Hg levels in the Housatonic River estuary may be related to local Hg sources (e.g., historic feltmaking industry of Danbury).

MERCURY EMISSIONS FROM LONG ISLAND SOUND

Rolfhus, K.R., C.H. Lamborg, W.F. Fitzgerald and P.H. Balcom, Department of Marine Sciences, University of Connecticut, Groton, CT 06340; G.M. Vandal and C.S. Russ, Pfizer, Inc., Groton, CT 06340

A preliminary mass balance (the framework for process and mechanistic studies) for total Hg in Long Island Sound (LIS) has revealed the presence of significantly large emissions of Hg° from the waters of LIS to the local/regional atmosphere. While the importance of Hg° cycling in natural waters in well known, its significance has not been documented in coastal waters. The scale of the Hg° emissions from LIS is evident in the Hg budget, where the initial estimates for seasonal Hg° fluxes vary between 310 and 483 pmol m^{-2} d^{-1} which corresponds to an annual Hg° emission of approximately 93 kg. Viewed biogeochemically, water to air Hg° volatilization is considerably larger than direct atmospheric Hg deposition (ca. 26 kg yr^{-1}). It represents (or is remobilizing) amounts of Hg comparable to the river input (ca. 120 kg yr^{-1}) or our preliminary values for annual Hg input from sewage (ca. 85 kg). Indeed, as much as 40% of the Hg input to LIS is transformed (reduced) and re-omitted to the atmosphere. Since most Hg entering LIS has an anthropogenic origin, a substantial pollution component is recycling. This flux is similar to estimates for Hg emissions from point sources such as coal fired power plants, sludge incinerators and commercial boilers in Connecticut (CT DEP, Air Resources Div.). Complementary laboratory studies indicate that (1) the reduction rates (ca. 0.5 to 3% d^{-1}) for both the biotic and abiotic reduction processes are more than sufficient to sustain the estimates for evasional fluxes of Hg° from LIS and (2) photochemical abiotic reduction of reactant Hg (e.g., labile species) in seawater is a major mechanism for the *in situ* production of Hg°. If Hg° emissions from shelf regions of the eastern US were comparable to LIS (ca. 28 g km^{-2} y^{-1}) the efflux might approach 15 Mg y^{-1} (15 tons). This potential mobilization of Hg is extraordinarily large, and environmentally significant. While we might anticipate that Hg° production and evasion from continental shelf waters would decline with distance from the near shore, this cycling of Hg has not been examined.

MONITORING

THE BENTHIC FAUNA OF THE LOWER QUINNIPIAC RIVER

Cuomo, C., Department of Earth and Environmental Sciences, Wesleyan University, Middletown, CT 06459 and Department of Geology and Geophysics, Yale University, New Haven, CT 06520

ABSTRACT

A systematic survey of the benthic fauna of the lower Quinnipiac River was undertaken as part of a larger study investigating nonpoint source pollution impacts on the Quinnipiac River watershed. A total of 17 sites between Wallingford and New Haven Harbor, CT, were chosen within the river and repeatedly sampled between March and September of 1997. Faunal analysis reveals that the lower Quinnipiac River is dominated by small, opportunistic polychaetes and chironomids depending upon salinity. Numerical abundances of all these organisms, however, were found to be very low. Additionally, the small sizes of all organisms found were also unusual. All study sites showed little variation over time, supporting the hypothesis that these organisms are the dominant members of the benthos throughout the year. These faunal assemblages reveal a system that is under relatively constant stress and contains a somewhat marginal fauna that has adapted to the stress over time.

INTRODUCTION

The Quinnipiac River is the fourth largest river entering Long Island Sound from the north. It originates in a red maple swamp in Farmington, CT, and flows south for 38 miles ultimately entering Long Island Sound at New Haven Harbor (Quinnipiac River Watershed Association 1995) (Figure 1a). Tidal influence reaches north to the North Haven-Wallingford border, a distance of approximately 14 miles from New Haven Harbor. The 170 square miles of the Quinnipiac River watershed is densely populated and significant areas of it are urbanized and/or industrialized, including the portion of the river under study. Prior to the 1960's when regulation of industrial and sewage effluent release took effect, significant amounts of industrial and municipal wastes were routinely released into the river (Quinnipiac River Watershed Association 1995). These wastes came from a variety of sources including, but not limited to, metal-working, pharmaceutical, and chemical industries and sewage treatment plants. At present, there are over twenty industries and municipalities with permits to release controlled amounts of effluent into the river (Quinnipiac River Watershed Association 1995). In addition to these point sources of pollution, the Quinnipiac River is severely impacted by various non-point pollution sources including urban and highway run-off, pesticide and fertilizer applications, and animal wastes.

The quality of the water and sediments of the Quinnipiac River has been impacted by both the historical and present sources of pollution. Numerous studies (Ginsberg and Vetrano 1992; Jop 1995; Pellegrino 1992; Tolonen and Wasielewski 1992) conducted on various environmental aspects of the Quinnipiac River and its tributaries reveal some of these impacts.

These studies report heavy metals and organic pollutants in both the sediments of the river and its tributaries and in the tissues of studied organisms.

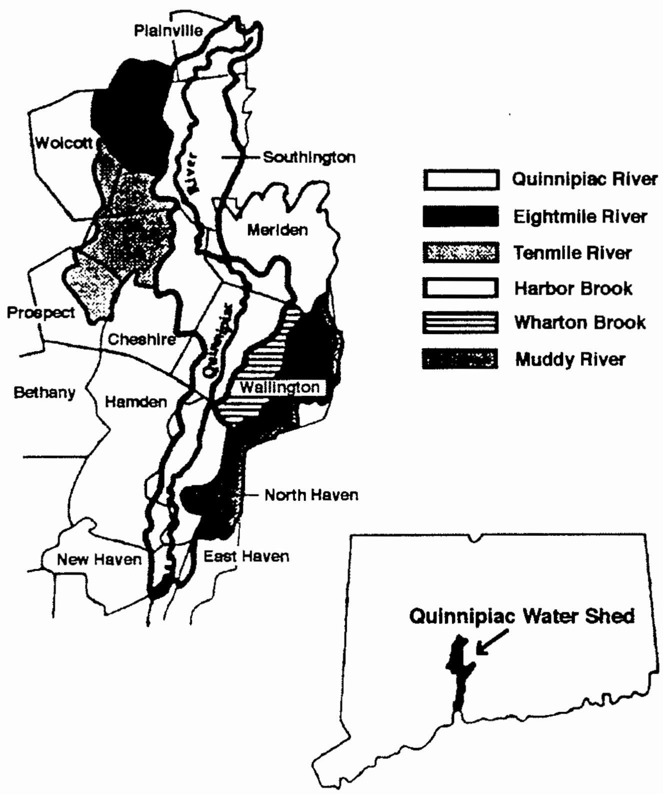

Figure 1a. Regional map showing the location and extent of the Quinnipiac River watershed (from Quinnipiac River Watershed Association 1995).

In 1997, a joint scientific investigation of the Quinnipiac River watershed was launched by researchers at the Yale University School of Forestry and Environmental Studies' Center for Coastal and Watershed Systems, the Peabody Museum of Natural History, the Quinnipiac River Watershed Association and the University of New Haven's Department of Biology and Environmental Sciences with funding from the Connecticut Department of Environmental Protection and the United States Environmental Protection Agency. One of the stated purposes of this study is to assess the biological health of the Quinnipiac River and its tributaries. The study reported on represents a small portion of this larger investigation and focuses specifically on the benthic organisms present in the lower reaches of the Quinnipiac River.

Benthic invertebrates are essential components of most healthy freshwater, estuarine, and marine soft sediment systems, although the organismal composition of the communities differs among the three environments (Bryce and Hobart 1972; McCall 1977; McCall and Tevesz 1982; Rhoads and Boyer 1982; Rhoads and Germano 1982; Rhoads and Young 1978; Rhoads et al. 1978). Common members of freshwater benthic invertebrate communities include oligochaete worms, chironomids, various insect larvae, leeches, freshwater gastropods, and freshwater bivalves whereas estuarine and marine benthic communities are typically composed of nematodes, polychaete worms, amphipods, crustaceans, gastropods, molluscs, and echinoderms.

Estuarine communities often contain stress-tolerant organisms from both end-member communities. In fact, in the lower reaches of a river, it is common to find shifts in benthic composition between freshwater and marine organisms over the course of a few years - or perhaps even seasonally - if there is significant variation in river discharge (Cuomo and Zinn 1997).

Studies of benthic organisms and communities have shown that benthic organisms can be employed as indicators of the health of aquatic systems (Rhoads and Germano 1982). The benthos - those organisms that live within and/or on sediments - are influenced by, and in turn influence, sediment and bottom water chemistry (Aller 1982; Fisher 1982), sediment organic content (Rhoads and Germano 1982), and sediment structure (McCall and Tevesz 1982; Rhoads and Boyer 1982; Rhoads and Young 1970). Additionally, benthic organisms serve as major prey species for higher level consumers within the local food web. Thus, anything that affects the benthos in a region has the potential to effect the entire larger biological system (Klerks and Bartholomew 1991; Botton et al. 1998). It was with this in mind that the present study of the Quinnipiac River benthos was undertaken.

The primary goals of this study are (a) to document the present benthic populations living within the lower regions of the Quinnipiac River and (b) to evaluate the health of the Quinnipiac River benthos based on the benthic community stages found. Both of these pieces of information are expected to be part of a longer term monitoring of the health of the Quinnipiac River benthic environment.

METHODS

Field collections were undertaken between the months of March and September, 1997. A total of 17 sites were chosen within the lower region of the Quinnipiac River (Figure 1b,c,d). Sites selected included those near to point sources of pollution, areas of suspected non-point source pollution, and hypothetically pristine sites. The northernmost site was located in Wallingford, and the southernmost site was located just north of Grand Avenue near the mouth of the Quinnipiac River.

Grab samples were taken at each site using a 0.25 m^2 Van Veen grab. Samples taken for biological study were placed in 2 liter Nalgene jars, fixed with formalin, stained with rose bengal, covered and labeled with their site number. Upon return to the laboratory, all samples were decanted and passed through a series of sieves (1000 μm, 500 μm, 250 μm, 125 μm and 63 μm). All materials retained on the sieves were examined under a dissecting microscope and all organisms were removed and placed in labeled glass vials until identified. Following identification, all samples were transferred to individual vials and covered with 10% ethanol.

Visual inspection of sediment characteristics, such as grain size, sorting, and degree of cohesion, was carried out in the field at the time of collection. Bottom water samples were taken for dissolved oxygen and sulfide analyses using a Niskan water sampler. Samples were fixed and analyzed using standard chemical titration methods. Salinity and bottom water temperatures were obtained for each site sampled.

RESULTS

The results of the benthic survey are presented in Tables 1-4. All stations sampled contained fauna that are typically associated with either freshwater or estuarine systems. Organic carbon content varied with sediment type - sandy sediments generally contained few organics (< 1% organic C by weight) whereas muddy sediments had over 5 % organic Carbon by weight. All water samples contained oxygen and no sulphides were measured in any bottom water samples.

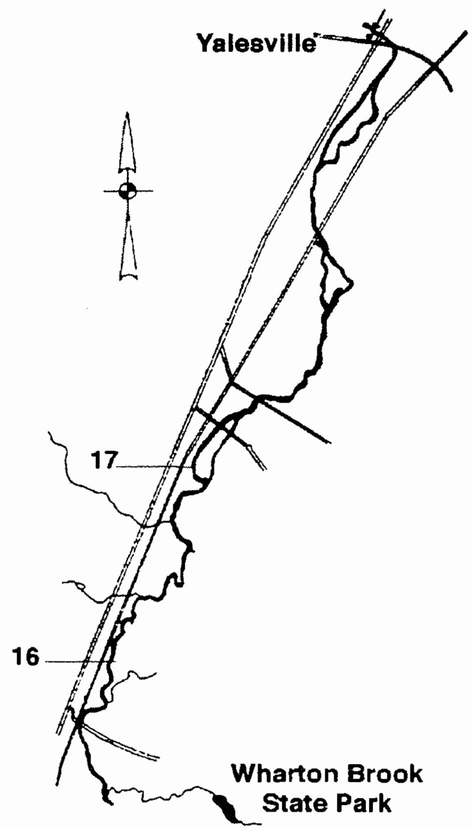

Figure 1b. Map of the lower Quinnipiac River showing sampling the northernmost sampling sites (from Quinnipiac River Watershed Association 1995).

The sites can be divided according to salinity and tidal influence into three groups - freshwater, brackish, and estuarine. Sites 11 - 17 have been classified as freshwater. The organisms at these sites are all typical inhabitants of freshwater benthic habitats. These sites are all located above the level of maximum tidal influence. Brackish water sites (Sites 6-10) are characterized by a mixture of both freshwater and estuarine organisms and salinities less than 15 ppt. Only estuarine benthic organisms were found at Sites 1-4 and salinities at these sites commonly ranged between 20 and 23 ppt.

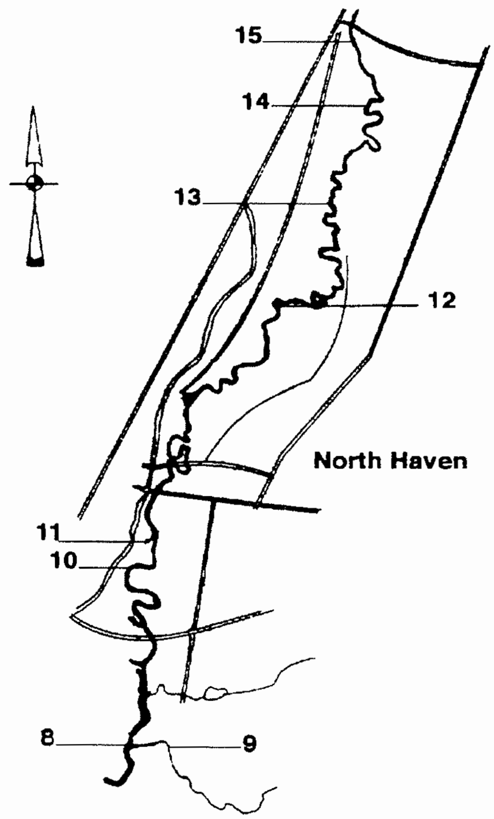

Figure 1c. Map of the lower Quinnipiac River showing the centrally located sampling sites (from Quinnipiac River Watershed Association 1995).

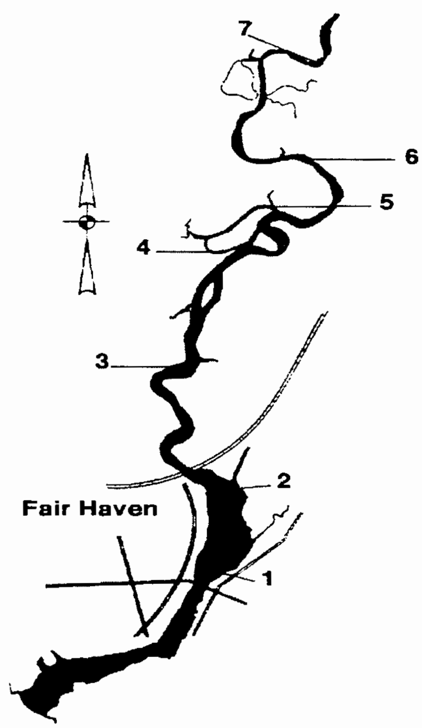

Figure 1d. Map of the lower Quinnipiac River depicting the southernmost sampling sites.

Chironomids and tubificid oligochaetes dominated the freshwater areas, small opportunistic polychaetes dominated the estuarine sites, and a mixture of chironomids, oligochaetes and small polychaetes were the numerical dominants of the brackish regions. Overall, species diversity was highest at the mouth of the river and decreased towards the north. Total species numbers at all sites except Site 2 never exceeded 100 organisms, regardless of sampling season. All organisms identified were $\leq$ 3 cm in length; many were $\leq$ 1.5 cm long.

DISCUSSION

Rhoads and Germano (1982) described a series of predictable stages that estuarine benthic communities undergo from their initial establishment onward. These successional sequences are characterized by particular functional types of organisms - small surface deposit feeders (Stage I) ultimately are displaced by deep-deposit-feeding organisms (Stage III), if conditions remain stable. Polychaetes, such as *Capitella sp., Polydora sp., and Streblospio benedicti*, are typical Stage I members. McCall and Tevesz (1982) have documented similar assemblages for freshwater systems. Early colonizers in these systems generally consist of chironomids and tubificid oligochaetes. According to Rhoads and Germano (1982), a benthic community can remain at an early successional stage if the system is subjected to frequent physical, chemical or biological disturbances. It follows that early successional stages can also be maintained within a system subjected to long-term cumulative environmental stress. This appears to be the situation for the lower Quinnipiac River.

Both chironomids and tubificid oligochaetes dominated the freshwater sites sampled in this study whereas nereid and orbinid polychaetes and chironomids dominated the brackish water sites. Orbinid polychaetes are common inhabitants of brackish water environments and nereids are errant polychaetes that range from the upper intertidal to fully marine conditions. The majority of these organisms fall within the Stage I assemblages of Rhoads and Germano (1982) and McCall and Tevesz (1982), as do those identified from the lowermost estuarine sites (Sites 1-4). All sites, regardless of proximity to a known pollution point source and/or month sampled, remained in a Stage I successional category, suggesting a stressor on the entire system rather than isolated, local stressors.

Additional support for this comes from the fact that all animals collected and identified were small in size. Either continuous recruitment by the benthos is occurring within the Quinnipiac River's lower reaches and organisms are not surviving past juvenile stages or the organisms that are there are experiencing some type of stunting.

Furthermore, Stage I organisms are usually found in great numbers. Data from this study, however, reveal a numerically depauperate fauna. This could be the result of limited larval availability. However, abundant larvae were observed in plankton tows taken at various sites within the river during the summer sampling. Salinity stress may be a factor affecting organism abundance. If this were the dominant factor, however, one would expect to see high numbers at the freshwater and estuarine sites with lower numbers at the more stressed brackish sites. A strong trend in this direction is not present in the data. Rather, the extremely low numbers of organisms collected per sample most likely reflects a system under extreme internal stress.

Table 1. Sample Site Sediment Characteristics.

Site #	Sediment Description	Site Type
17	Coarse sand;low org C[a]	Freshwater
16	Coarse-medium sand;low org C	Freshwater
15	Coarse-medium sand;low org C	Freshwater
14	Muddy sand;moderate org C[b]	Freshwater
13	Medium sand;low org C	Freshwater
12	Coarse sand;low org C	Freshwater
11	Muddy sand;moderate org C	Freshwater
10	Muddy silt;moderate org C	Brackish
9	Mud & leaf debris; high org C[c]	Brackish
8	Sands with bacterial mats; high org C	Brackish
7	Sandy mud;high org C	Brackish
6	Sandy mud;high org C	Brackish
5	Muddy sand;low org C	Estuarine
4	Fine sand with clay nodules; high org C	Estuarine
3	Peat; high org C	Estuarine
2	Fine mud & plant debris; high org C	Estuarine
1	Oyster shell hash;low org C	Estuarine

[a]Low Org C $\leq$ 1.00% by weight

[b]Moderate Org C > 1.00% and < 3.00% by weight

[c]High Org C $\geq$ 3.00% by weight

Table 2. Cumulative Organism Diversity and Abundance Date for Freshwater Sites.

Site #	Organism	Abundance/0.25 m^2
17	Chironomidae	24
	Tubificidae (oligochaeta)	2
	Nematoda	3
	Helius sp.	2
16	Tubificidae (oligochaeta)	15
	Nematoda	1
15	Chironomidae	9
14	Chironomidae	10
	Tubificidae (oligochaeta)	10
	Lymnaea sp.	1
13	Chironomidae	53
	Tubificidae (oligochaeta)	7
	Cathocampus minutus	1
12	Chironomidae	25
	Tubificidae (oligochaeta)	23
	Dipteran larvae	1
11	Chironomidae	57

Table 3. Cumulative Organism Diversity and Abundance Data for Brackish Water Sites

Site #	Organism	Abundance/0.25 m^2
10	Chironomidae	20
	Nematoda	10
	Crangon septemspinosa	3
	Miscellaneous fish larva	1
9	Chironomidae	3
	Trematoda	1
	Lumbrinereis fragilis	5
	Clymenella sp.	4
	Crangon septemspinosa	2
8	Chironomidae	9
	Nematoda	1
	Dipteran larva	1
	Lumbrinereis fragilis	11
	Orbiniidae	2
	Cyathura polita	1
	Crangon septemspinosa	12
7	Chironomidae	31
	Turbellaria sp.	3
	Lumbrinereis fragilis	25
	Cyathura polita	3
	Hypaniola grayi	2
	Sigambra tentaculata	1
	Crangon septemspinosa	13
	Miscellaneous fish larva	1
6	Chironomidae	4
	Orbiniidae	4
	Cyathura polita	4
	Hypaniola grayi	2
	Crangon septemspinosa	3

Table 4. Cumulative Organism Diversity and Abundance Data for Estuarine Sites

Site #	Organism	Abundance/0.25 m^2
5	Nematoda	2
	Orbiniidae	10
	Cyathura polita	4
	Hypaniola grayi	4
4	Nematoda	7
	Cyathura polita	23
	Hypaniola grayi	3
	Ampharetidae	12
	Notomastus sp.	1
	Crangon septemspinosa	7
3	*Nereis succinea*	1
	Nassarius obsoletus	2
	Gemma gemma	1
2	Nematoda	32
	Polydora ligni	20
	Streblospio benedicti	49
	Capitella capitata	5
	Gemma gemma	1
	Carcinus maenas	1
1	*Nereis succinea*	1
	Nassarius obsoletus	2
	Gemma gemma	1

The unexpectedly low organismal abundances and the small size of the organisms might also reflect low food availability. This is certainly a possibility for the freshwater sandy sites, as the organic carbon in these sediments was low (<1% organic C by weight); however, sites with muddy sediments and/or abundant decaying vegetation within the brackish and estuarine areas of the Quinnipiac River contained up to 5% organic C by weight - enough to support a thriving benthos.

In 1992, Pellegrino reported on the benthos present in the lowermost reaches of the Quinnipiac River. His sites overlapped with Sites 1-5 in the present study. He recorded a total of 50 invertebrate species, dominated by crustaceans, oligochaetes, and polychaetes. Interestingly, the recorded abundances per sample for the majority of these organisms from the intertidal sites, while reduced in number, are much higher than those obtained in this study. This suggests that the benthic communities within the lowermost Quinnipiac River have undergone further degradation over the past decade. Reasons for this might include sampling error, an anomalous recruitment year, a new stress upon the system (chemical or biological), or the accumulated stress of many years of pollution.

CONCLUSIONS

The distribution of benthic organisms living within the lower Quinnipiac River appears to be primarily controlled by salinity. The data suggests that the species composition and numerical abundances, however, are controlled, at least in part, by significant anthropogenic inputs - both historical ones and present day ones. The study indicates that the Quinnipiac River is in an unhealthy state as recorded by the benthos and further study and monitoring is recommended.

ACKNOWLEDGMENTS

This research was supported by the United State's Environmental Protection Agency and the state of Connecticut's Department of Environmental Protection. Field and/or laboratory assistance was provided by Mr. Timothy Keeney, Mr. Joel LaBella, Mr. Lee LeBlanc, Ms. Celia Lewis, Ms. Lena Maun, Dr. Ray Pupedis, and Mr. Greg Rutherford.

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SEAWATER TEMPERATURES IN LONG ISLAND SOUND: 1978-1998

Foertch, J., Northeast Utilities Environmental Laboratory, P.O. Box 128, Waterford CT 06385

ABSTRACT

Ambient seawater temperatures have been recorded at Millstone Nuclear Power Station in Waterford, CT, and Norwalk Harbor Station in Norwalk, CT (representing eastern and western Long Island Sound, respectively), for more than 20 years. Simple analyses of these data may be used to assess regional trends in seawater temperature, and may help interpret recent Sound-wide changes in finfish and shellfish populations. Although annual mean temperatures do not exhibit a significant long-term trend, there is a consistent pattern of warmer than average temperatures in late winter-early spring in recent years.

INTRODUCTION

Millstone Nuclear Power Station (MNPS) in Waterford, CT, and Norwalk Harbor Station (NHS) in Norwalk, CT, (Figure 1) are electricity-generating facilities within the Northeast Utilities System that use Long Island Sound (LIS) seawater for once-through condenser cooling in order to remove heat from steam produced as part of their power generation process. Measurement of ambient (intake) and effluent seawater temperature is used to calculate temperature rise across the condensers and total heat discharged to the marine environment. Temperature data are collected as part of the Environmental Data Acquisition Network (EDAN), a computerized information generation and storage system. The EDAN dataset includes water temperatures continuously sampled at 15-minutes intervals over a 20+ year timespan. Owing to the importance of water temperature as a factor in many physical and biological marine processes, and particularly with recent heightened interest in global warming and climate change, it seems appropriate to provide summaries of the temperature time-series to researchers studying LIS.

RESULTS AND DISCUSSION

For each station, daily means were calculated from the 15-minute observations of intake water temperature, and a 20-year (1978-97) mean was calculated for each day of the year (these long-term daily means are included in Appendix 1, and for each year, daily means are plotted with the long-term means in Appendix 2). Similarly, annuals means were calculated from daily means of each year, and an overall 20-year mean water temperature was calculated from the annual means. Table 1 presents the annual means at each station, with highest and lowest daily temperatures within each year. On the basis of annual means at both Millstone Point and Norwalk Harbor, 1991 was the warmest year in the study period (1978-1998), and 1996 was the coolest. Based on daily means, there was less consistency; the warmest day at Millstone occurred in 1979 (22.9°C) and at Norwalk in 1995 (27.5°C). The coldest day at Millstone occurred in 1981 (-0.8°C), and minimum daily means of -2.0°C occurred in 1978 and 1996 at Norwalk. In general, temperatures at Norwalk were more extreme than those at Millstone (i.e., warmer maxima, lower minima).

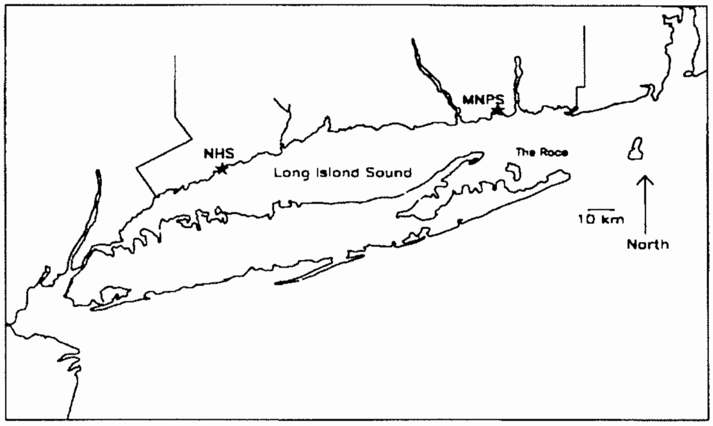


Figure 1. Map of Long Island Sound, showing location of temperature recording station; NHS = Norwalk Harbor Station in western LIS, MNPS = Millstone Nuclear Power Station in eastern LIS, approximately 100 km from NHS.

Table 1. Annual mean seawater temperatures at Millstone Nuclear Power Station and Norwalk Harbor Station, 1978-1998, including maximum and minimum daily mean temperatures within each year, and the differences between the annual means and the overall 20-yr mean at each station (11.55°C at Millstone, 11.76°C at Norwalk).

Millstone Station					Norwalk Harbor			
Year	Annual Mean	Max	Min	Diff	Annual Mean	Max	Min	Diff
1978	10.62	20.8	0.1	-0.92	10.92	23.9	-2.0	-0.85
1979	11.55	22.9	-0.1	0.00	11.36	25.6	-1.8	-0.41
1980	11.06	22.3	0.6	-0.49	11.49	24.7	-0.7	-0.27
1981	10.98	21.9	-0.8	-0.57	11.55	24.7	-1.7	-0.21
1982	11.12	22.4	0.3	-0.43	12.31	23.5	0.9	0.54
1983	11.96	21.5	2.8	0.41	12.12	23.8	-0.6	0.35
1984	11.95	21.4	2.7	0.40	11.61	24.1	-1.2	-0.15
1985	11.94	22.5	0.3	0.39	12.21	25.7	-0.8	0.45
1986	11.87	21.2	2.1	0.32	11.66	24.5	0.4	-0.11
1987	11.75	22.1	2.0	0.20	12.42	24.9	0.4	0.65
1988	11.10	22.0	1.2	-0.45	11.30	24.8	-0.8	-0.47
1989	11.32	21.0	1.2	-0.23	11.25	23.3	-0.6	-0.51
1990	12.05	22.1	2.6	0.50	12.44	24.5	0.8	0.68
1991	12.55	22.1	3.7	1.00	12.92	24.8	1.7	1.16
1992	11.42	20.4	2.3	-0.13	11.59	23.1	1.3	-0.18
1993	11.67	22.2	1.3	0.12	11.40	24.7	-0.7	-0.36
1994	11.58	22.2	0.5	0.03	11.25	24.0	-1.7	-0.51
1995	12.48	21.8	2.7	0.93	12.16	27.5	-0.6	0.40
1996	10.56	20.4	1.1	-0.99	10.64	22.6	-2.0	-1.12
1997	10.89	20.4	1.8	-0.66	11.56	23.6	0.6	-0.20
1998	12.13	22.0	3.5	0.58	12.87	25.1	2.5	1.11

Table 1 also presents the differences between the annual means and the 20-year means. These differences are illustrated in Figure 2. There are some discrepancies between Norwalk Harbor, near the western end of LIS, and Millstone, at the eastern end (e.g., in 1982, the mean temperature at NHS was more than 0.5°C above the 20-year average, while MNPS was almost 0.5°C below average); however, in general, the trends correspond closely, indicating Sound-wide conditions (e.g., as noted above, 1991 was a 'warm year' throughout LIS, 1996 was a 'cool year').

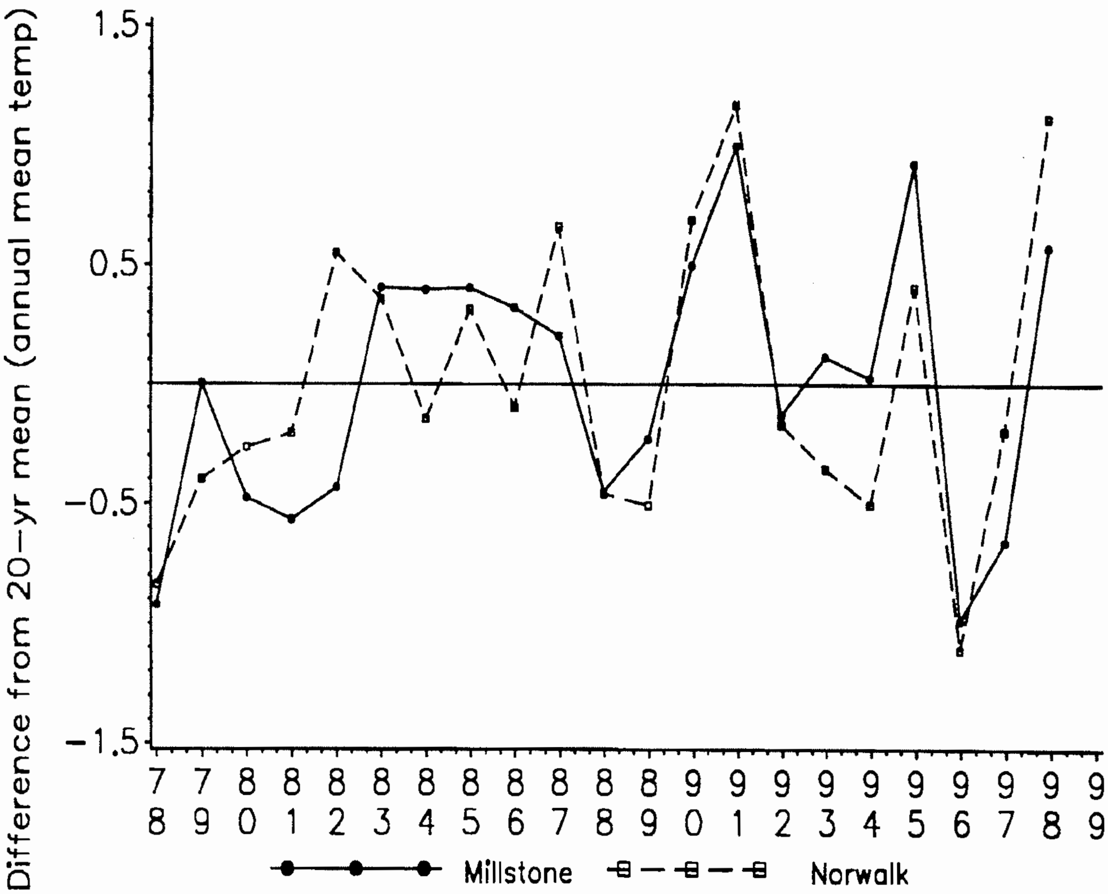


Figure 2. Differences (°C) between annual mean temperatures at Millstone Nuclear Power Station and Norwalk Harbor Station (1978-1998) and the 20-year mean calculated for each station (11.55°C at Millstone, 11.76°C at Norwalk).

No long-term trends were evident in annual means at either station. However, additional analyses were performed on quarterly data (i.e., Jan-Mar, Apr-Jun, Jul-Sep, Oct-Dec); these results are illustrated in Figure 3. It is apparent that water temperatures in the early part of the year (first quarter, and to a lesser extent, second quarter) have recently become warmer.

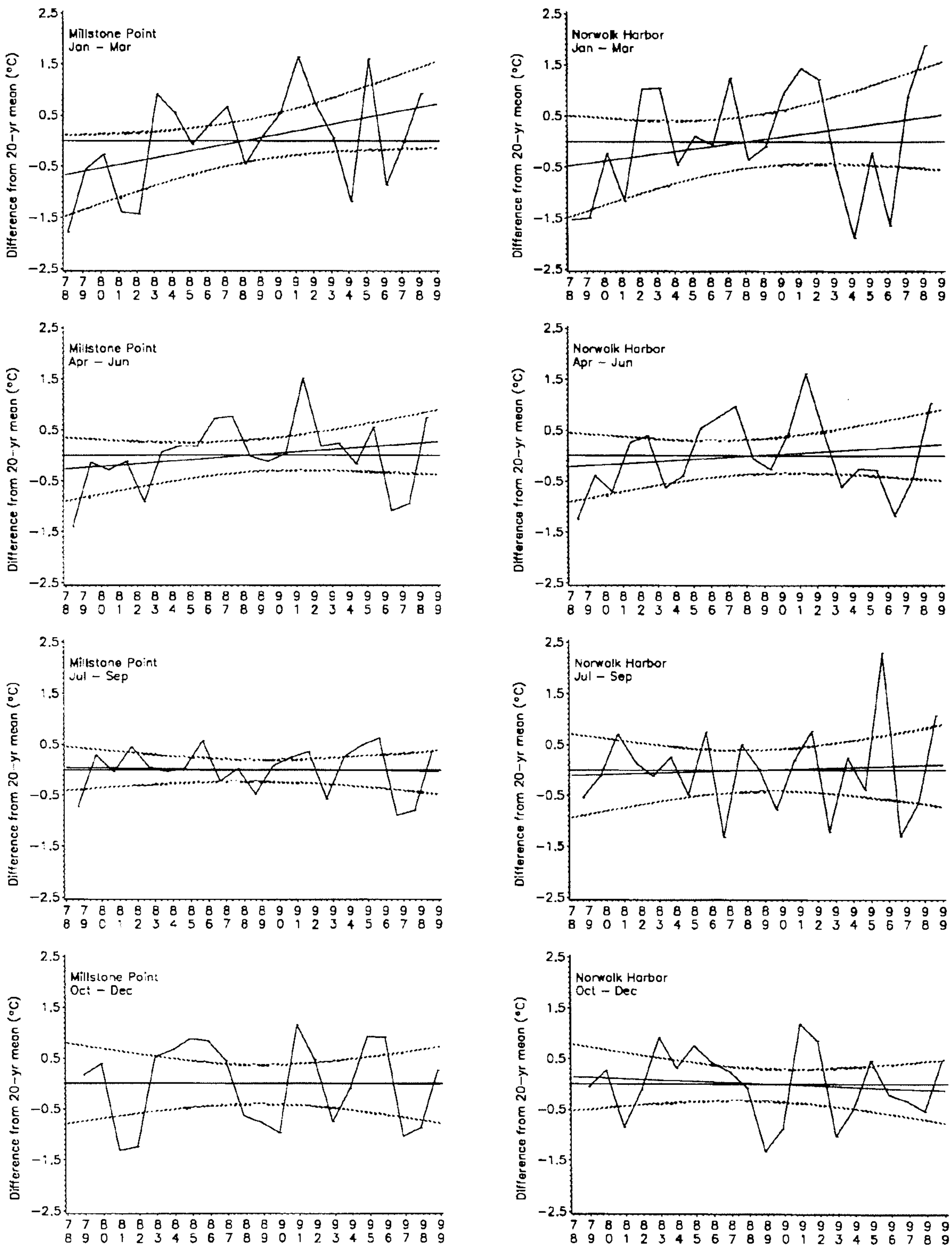


Figure 3. Differences (°C) between quarterly (Jan-Mar, Apr-Jun, Jul-Sep, Oct-Dec) mean temperatures at Millstone Nuclear Power Station and Norwalk Harbor Station (1978-1998) and the 20-year mean calculated for each quarter at each station.

APPENDIX 1. Mean daily water temperatures (°C) at Millstone Nuclear Power Station (MP) and Norwalk Harbor Station (NH) for the period 1978-1998, and the maximum and minimum daily mean temperatures that occurred on each day-of-the-year within that period.

OBS	DATE	MEAN_MP	MAX_MP	MIN_MP	MEAN_NH	MAX_NH	MIN_NH
1	01JAN	5.8	8.5	3.0	3.6	5.5	1.1
2	02JAN	5.7	8.5	2.6	3.6	5.8	1.4
3	03JAN	5.6	7.9	1.6	3.5	5.4	1.0
4	04JAN	5.4	7.8	0.7	3.3	5.5	0.1
5	05JAN	5.1	7.5	0.2	3.2	5.9	-0.2
6	06JAN	5.1	7.4	0.5	3.1	5.7	0.0
7	07JAN	5.1	7.4	1.4	2.9	5.9	-1.2
8	08JAN	4.9	7.1	0.6	2.7	5.4	-2.0
9	09JAN	4.6	6.9	0.6	2.7	5.4	-1.7
10	10JAN	4.6	7.4	0.8	2.4	5.3	-1.2
11	11JAN	4.4	8.3	-0.8	2.3	5.3	-1.1
12	12JAN	4.2	7.0	-0.4	2.1	5.3	-1.4
13	13JAN	4.3	6.6	-0.5	2.0	5.1	-1.1
14	14JAN	4.2	7.1	-0.4	2.0	6.3	-1.2
15	15JAN	4.1	7.6	0.0	2.0	5.6	-1.2
16	16JAN	3.9	7.7	0.1	2.0	5.7	-0.7
17	17JAN	3.9	7.4	-0.1	2.0	4.7	-0.6
18	18JAN	3.8	7.2	-0.7	1.9	4.1	-0.7
19	19JAN	3.8	7.2	-0.1	1.8	3.9	-0.8
20	20JAN	3.7	7.3	0.7	1.7	3.8	-1.0
21	21JAN	3.7	7.6	1.4	1.5	4.1	-1.0
22	22JAN	3.6	7.1	1.0	1.4	3.6	-0.7
23	23JAN	3.7	6.6	0.9	1.6	3.9	-1.0
24	24JAN	3.7	6.6	0.9	1.8	4.5	-0.5
25	25JAN	3.6	6.4	0.4	1.7	3.6	-0.1
26	26JAN	3.6	6.2	0.6	1.7	3.5	-1.0
27	27JAN	3.5	6.0	0.3	1.6	3.0	-1.6
28	28JAN	3.5	5.5	0.5	1.5	2.6	-0.9
29	29JAN	3.4	5.5	0.5	1.5	3.3	-0.8
30	30JAN	3.4	5.7	0.8	1.5	3.2	-0.8
31	31JAN	3.4	5.8	1.1	1.6	3.4	-0.7
32	01FEB	3.4	5.8	1.4	1.6	4.0	-0.7
33	02FEB	3.3	5.8	1.3	1.6	3.5	-0.7
34	03FEB	3.3	5.4	1.4	1.5	4.3	-0.7
35	04FEB	3.2	5.2	1.3	1.3	4.3	-1.2
36	05FEB	3.0	5.4	1.1	1.2	3.8	-1.7
37	06FEB	2.9	5.5	0.9	1.0	4.1	-2.0
38	07FEB	2.9	5.7	0.1	0.8	4.0	-2.0
39	08FEB	2.8	5.4	0.1	1.0	3.9	-1.9
40	09FEB	2.7	5.2	0.3	0.9	4.1	-1.8
41	10FEB	2.7	5.2	0.8	0.9	4.2	-1.7
42	11FEB	2.8	5.1	0.6	1.1	3.8	-1.7
43	12FEB	2.7	4.5	0.6	0.9	4.3	-1.6
44	13FEB	2.7	4.6	1.0	0.9	4.0	-1.6
45	14FEB	2.6	4.9	0.2	1.0	3.5	-1.2
46	15FEB	2.7	4.8	0.6	1.0	3.1	-1.8

OBS	DATE	MEAN_MP	MAX_MP	MIN_MP	MEAN_NH	MAX_NH	MIN_NH
47	16FEB	2.8	4.7	0.8	1.1	3.0	-1.7
48	17FEB	2.8	4.7	0.0	1.1	3.5	-1.5
49	18FEB	2.8	4.2	-0.1	1.2	4.2	-1.8
50	19FEB	2.9	4.6	0.2	1.3	4.3	-1.8
51	20FEB	3.1	4.7	0.5	1.4	3.5	-1.7
52	21FEB	3.1	4.6	0.8	1.7	4.0	-1.7
53	22FEB	3.2	5.0	0.9	1.8	3.9	-1.3
54	23FEB	3.3	5.0	0.9	1.9	4.3	-1.3
55	24FEB	3.3	5.1	1.0	1.8	3.8	-1.4
56	25FEB	3.1	5.1	1.3	1.7	3.8	-1.3
57	26FEB	2.9	4.8	1.0	1.6	4.0	-1.1
58	27FEB	3.0	4.6	0.5	1.6	4.1	-1.1
59	28FEB	3.1	4.6	0.5	1.6	4.4	-1.0
60	29FEB	3.1	5.0	1.0	1.8	4.7	-0.8
61	01MAR	3.1	5.4	0.6	2.0	4.6	-0.6
62	02MAR	3.2	5.6	0.7	1.9	4.7	-0.9
63	03MAR	3.2	5.8	0.6	1.9	5.0	-0.7
64	04MAR	3.2	5.7	0.3	2.0	4.9	-0.4
65	05MAR	3.3	5.8	0.2	2.3	5.0	0.0
66	06MAR	3.4	6.4	0.4	2.3	5.6	0.1
67	07MAR	3.4	5.3	0.6	2.2	5.0	-0.3
68	08MAR	3.4	5.3	1.0	2.3	6.5	-0.3
69	09MAR	3.4	5.5	1.1	2.2	5.7	-0.2
70	10MAR	3.5	5.2	1.3	2.4	5.3	0.0
71	11MAR	3.5	5.5	1.4	2.6	6.1	-0.3
72	12MAR	3.5	5.3	1.6	2.6	5.4	0.0
73	13MAR	3.6	5.2	1.6	2.7	5.2	0.1
74	14MAR	3.6	5.4	1.3	2.8	5.7	0.1
75	15MAR	3.6	5.4	1.4	3.0	6.2	-0.2
76	16MAR	3.7	5.9	1.3	3.0	7.3	0.0
77	17MAR	3.8	6.2	1.4	3.0	7.1	0.5
78	18MAR	3.9	5.8	1.6	3.1	5.6	0.4
79	19MAR	4.0	5.9	1.9	3.1	6.0	0.5
80	20MAR	4.1	5.9	2.0	3.3	5.9	0.9
81	21MAR	4.3	6.1	2.9	3.5	6.0	1.3
82	22MAR	4.2	6.0	2.5	3.6	6.1	1.5
83	23MAR	4.3	5.5	2.4	3.7	5.7	1.7
84	24MAR	4.3	6.0	2.4	4.0	6.4	1.7
85	25MAR	4.5	6.1	2.3	4.2	7.0	2.2
86	26MAR	4.7	6.3	2.3	4.4	6.9	2.2
87	27MAR	4.9	6.9	2.8	4.8	6.9	2.6
88	28MAR	5.1	6.8	3.4	5.0	6.9	3.1
89	29MAR	5.0	6.7	3.1	4.9	7.7	2.9
90	30MAR	5.2	6.7	3.2	5.1	9.1	3.0
91	31MAR	5.2	6.7	3.7	5.3	8.4	3.5
92	01APR	5.2	6.8	3.5	5.4	8.1	3.7

OBS	DATE	MEAN_MP	MAX_MP	MIN_MP	MEAN_NH	MAX_NH	MIN_NH
93	02APR	5.4	7.1	3.3	5.5	8.7	3.3
94	03APR	5.5	7.3	3.4	5.6	8.3	2.6
95	04APR	5.5	7.3	3.5	5.7	8.0	2.7
96	05APR	5.6	7.8	3.9	5.8	7.7	3.2
97	06APR	5.7	8.4	2.8	5.9	8.3	3.8
98	07APR	5.7	8.6	1.7	5.9	8.9	4.2
99	08APR	5.8	9.5	3.2	6.0	10.0	3.8
100	09APR	5.9	8.7	3.8	6.2	9.8	3.7
101	10APR	6.0	7.9	3.5	6.4	8.8	3.4
102	11APR	6.1	7.7	3.9	6.5	8.9	3.7
103	12APR	6.3	8.0	4.3	6.8	8.3	4.7
104	13APR	6.4	7.7	4.5	7.1	9.2	5.3
105	14APR	6.4	7.9	4.4	7.1	9.3	5.0
106	15APR	6.5	8.0	4.4	7.4	9.1	5.0
107	16APR	6.6	8.0	4.8	7.4	9.7	5.3
108	17APR	6.7	8.1	5.1	7.4	10.0	4.8
109	18APR	6.9	8.1	5.1	7.4	9.5	4.5
110	19APR	7.1	8.4	5.3	7.8	10.1	5.2
111	20APR	7.2	9.1	5.9	8.2	12.2	6.2
112	21APR	7.3	8.9	5.4	8.3	12.5	6.4
113	22APR	7.3	8.9	5.7	8.3	11.8	6.8
114	23APR	7.5	8.8	6.0	8.5	11.1	7.0
115	24APR	7.6	9.1	6.0	8.7	10.7	7.1
116	25APR	7.7	9.2	6.1	8.8	11.1	7.0
117	26APR	7.9	9.5	5.9	9.2	11.1	7.3
118	27APR	8.1	9.5	5.8	9.4	11.1	7.1
119	28APR	8.2	9.5	6.3	9.7	11.7	8.2
120	29APR	8.4	9.6	6.6	10.0	11.9	7.8
121	30APR	8.6	9.7	6.7	10.3	12.4	7.5
122	01MAY	8.7	10.0	6.9	10.3	12.3	7.4
123	02MAY	8.7	10.4	7.2	10.1	12.6	8.0
124	03MAY	8.8	10.2	7.2	10.2	12.0	8.7
125	04MAY	8.9	10.3	6.8	10.3	12.0	8.3
126	05MAY	9.1	10.5	7.0	10.4	12.0	7.9
127	06MAY	9.4	11.1	7.4	10.7	12.7	8.1
128	07MAY	9.5	11.1	7.7	11.0	13.1	8.8
129	08MAY	9.6	10.8	8.1	11.3	12.8	9.2
130	09MAY	9.7	11.0	8.1	11.4	13.3	8.6
131	10MAY	9.7	11.2	8.1	11.5	13.2	8.5
132	11MAY	9.9	11.6	8.5	11.7	13.6	8.4
133	12MAY	10.1	11.7	8.7	11.9	14.6	8.9
134	13MAY	10.3	11.6	8.4	12.0	14.5	8.8
135	14MAY	10.5	11.9	8.5	12.3	15.0	8.6
136	15MAY	10.6	12.6	8.2	12.5	15.5	9.5
137	16MAY	10.6	13.1	8.9	12.6	15.8	9.5
138	17MAY	10.7	12.4	8.9	12.7	15.9	9.8

OBS	DATE	MEAN_MP	MAX_MP	MIN_MP	MEAN_NH	MAX_NH	MIN_NH
139	18MAY	10.8	13.0	8.9	12.8	16.0	10.3
140	19MAY	10.9	13.0	9.2	13.1	16.4	10.5
141	20MAY	11.1	13.3	9.3	13.1	15.8	10.9
142	21MAY	11.2	13.8	9.1	13.2	15.5	10.9
143	22MAY	11.5	13.6	9.2	13.3	15.6	11.0
144	23MAY	11.8	14.2	9.8	13.7	16.3	11.8
145	24MAY	11.9	14.4	9.8	14.0	17.1	11.8
146	25MAY	12.0	13.9	9.6	14.1	17.4	12.0
147	26MAY	12.2	14.3	9.7	14.3	17.4	12.3
148	27MAY	12.4	14.6	10.4	14.5	18.0	12.2
149	28MAY	12.6	14.6	10.8	14.7	18.6	12.4
150	29MAY	12.7	14.6	10.8	15.1	18.9	12.2
151	30MAY	13.0	15.0	10.9	15.3	18.9	12.6
152	31MAY	13.2	15.1	10.9	15.2	19.2	12.4
153	01JUN	13.2	15.7	10.5	15.3	19.7	12.6
154	02JUN	13.3	15.8	10.3	15.4	20.7	12.6
155	03JUN	13.4	15.2	10.4	15.2	19.4	12.3
156	04JUN	13.4	15.4	10.8	15.2	18.1	12.5
157	05JUN	13.5	15.0	11.4	15.5	18.3	12.6
158	06JUN	13.7	15.5	11.4	15.6	19.4	13.0
159	07JUN	13.9	15.9	11.6	15.8	19.4	13.9
160	08JUN	14.1	15.8	12.1	16.1	19.1	14.2
161	09JUN	14.3	16.2	12.8	16.2	19.4	12.1
162	10JUN	14.4	16.4	13.0	16.3	19.1	11.9
163	11JUN	14.6	16.9	12.9	16.3	18.4	12.5
164	12JUN	14.8	16.3	13.7	16.3	17.8	13.2
165	13JUN	14.9	17.0	13.5	16.4	17.7	13.9
166	14JUN	15.1	16.6	13.9	16.7	18.4	13.5
167	15JUN	15.3	16.7	14.1	17.2	18.8	15.2
168	16JUN	15.6	17.1	14.3	17.4	19.3	15.1
169	17JUN	15.6	16.8	14.0	17.7	20.1	15.4
170	18JUN	15.8	16.9	14.4	18.0	20.5	16.2
171	19JUN	15.9	17.3	14.3	18.2	19.7	16.8
172	20JUN	15.9	17.4	14.4	18.2	20.0	16.7
173	21JUN	16.1	17.4	14.6	18.3	20.3	16.1
174	22JUN	16.1	16.8	14.7	18.1	21.0	14.7
175	23JUN	16.2	17.1	14.9	18.0	20.7	14.6
176	24JUN	16.3	17.4	15.1	18.1	20.7	15.1
177	25JUN	16.6	17.8	15.3	18.4	21.3	15.6
178	26JUN	16.6	18.3	15.2	18.5	21.2	16.1
179	27JUN	16.9	18.5	15.3	18.6	20.4	16.2
180	28JUN	17.0	18.8	15.2	18.8	21.6	16.7
181	29JUN	17.1	18.4	15.5	19.0	22.5	17.1
182	30JUN	17.2	18.9	15.5	19.2	22.8	17.3
183	01JUL	17.2	19.1	15.7	19.1	22.3	16.5
184	02JUL	17.4	18.9	15.3	19.2	21.8	16.2

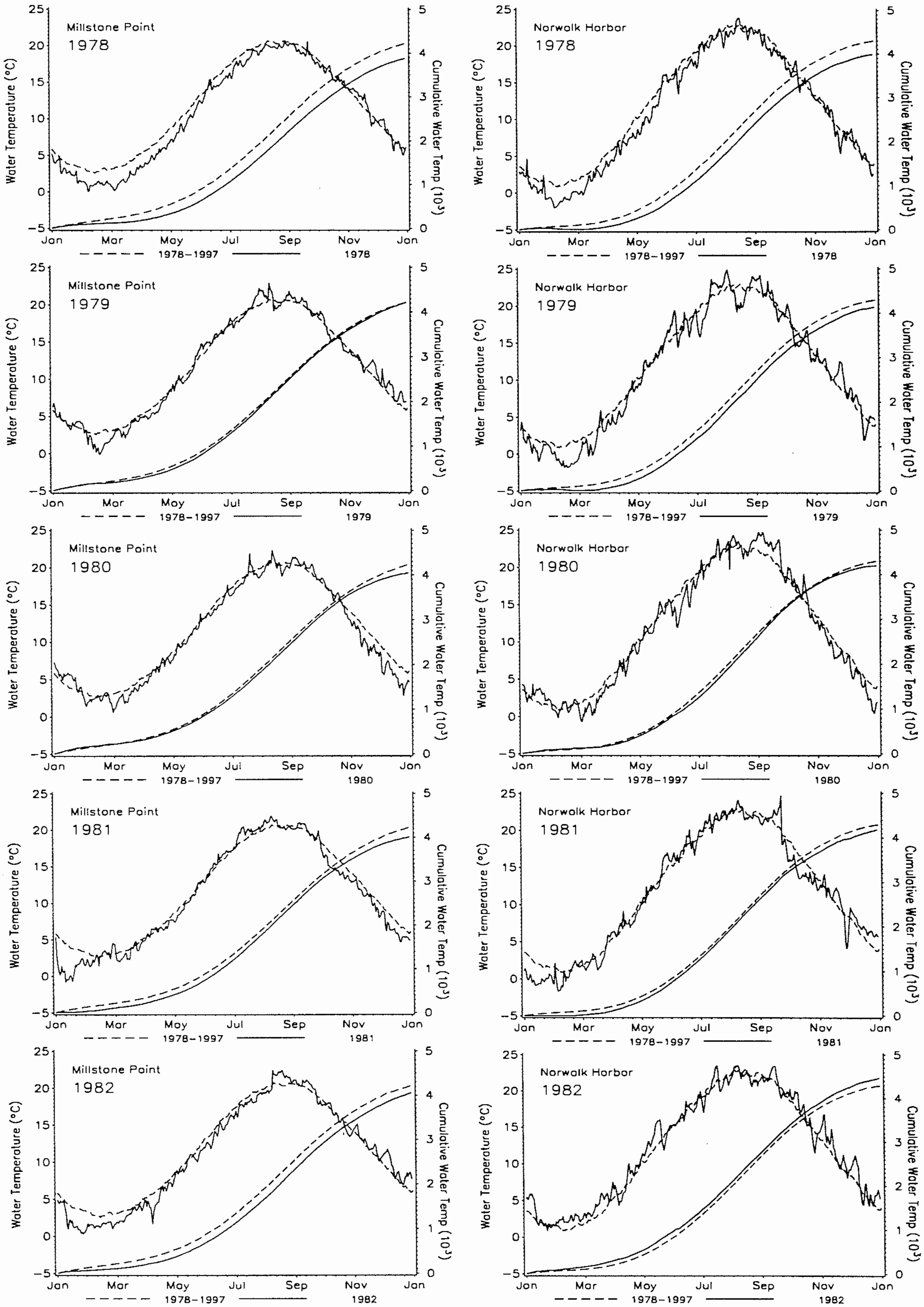
OBS	DATE	MEAN_MP	MAX_MP	MIN_MP	MEAN_NH	MAX_NH	MIN_NH
185	03JUL	17.5	19.0	15.6	19.2	22.0	16.9
186	04JUL	17.4	19.2	15.5	19.3	22.4	16.0
187	05JUL	17.8	19.9	16.1	19.7	22.7	15.8
188	06JUL	18.1	19.5	16.6	20.2	23.3	16.4
189	07JUL	18.2	19.6	16.9	20.3	23.1	17.4
190	08JUL	18.3	19.7	17.4	20.4	23.4	17.5
191	09JUL	18.4	19.9	17.6	20.4	23.5	17.8
192	10JUL	18.3	20.3	17.3	20.2	22.7	18.2
193	11JUL	18.5	20.3	17.3	20.2	22.9	18.4
194	12JUL	18.6	20.5	17.2	20.4	22.9	18.3
195	13JUL	18.6	19.9	17.6	20.5	23.0	18.3
196	14JUL	18.7	20.2	17.4	20.6	22.8	18.5
197	15JUL	18.8	19.8	17.2	21.0	22.9	19.4
198	16JUL	18.9	20.2	17.1	21.1	23.3	18.5
199	17JUL	19.1	20.7	17.4	21.5	23.6	19.5
200	18JUL	19.2	20.8	17.5	21.5	24.0	20.0
201	19JUL	19.3	21.3	17.8	21.6	24.5	18.9
202	20JUL	19.5	21.9	18.0	21.4	24.4	19.3
203	21JUL	19.4	21.5	17.6	21.4	24.4	17.9
204	22JUL	19.5	22.1	17.9	21.5	24.9	18.4
205	23JUL	19.5	21.5	17.6	21.6	25.9	18.5
206	24JUL	19.4	21.5	17.2	21.6	26.2	18.6
207	25JUL	19.6	21.5	17.5	21.9	26.6	19.3
208	26JUL	19.7	21.5	17.8	22.0	26.4	20.0
209	27JUL	19.8	21.5	18.0	22.2	26.1	20.0
210	28JUL	19.8	22.1	17.9	22.2	26.0	19.5
211	29JUL	19.9	22.1	18.2	22.0	24.7	19.9
212	30JUL	19.8	22.2	17.9	22.0	24.5	19.8
213	31JUL	19.9	22.2	17.6	22.2	25.3	19.8
214	01AUG	20.0	22.0	17.5	22.4	25.6	20.0
215	02AUG	20.1	22.0	17.8	22.6	25.5	20.0
216	03AUG	20.2	21.8	17.9	22.7	26.3	19.7
217	04AUG	20.2	21.9	18.3	23.0	26.3	20.8
218	05AUG	20.1	21.3	18.8	22.7	25.5	19.9
219	06AUG	20.2	21.3	18.8	22.7	24.7	20.2
220	07AUG	20.3	21.8	18.8	22.7	24.8	20.5
221	08AUG	20.3	21.8	18.6	22.8	24.8	20.3
222	09AUG	20.6	22.4	18.4	22.9	25.1	20.7
223	10AUG	20.7	22.9	18.3	22.7	25.6	20.1
224	11AUG	20.6	22.2	18.0	22.5	25.5	19.4
225	12AUG	20.7	22.3	17.6	22.5	25.2	19.2
226	13AUG	20.6	21.8	17.5	22.6	25.7	19.8
227	14AUG	20.6	21.8	17.9	22.8	26.2	19.9
228	15AUG	20.6	22.2	18.3	22.9	26.7	19.6
229	16AUG	20.6	22.2	18.2	22.9	27.5	19.7
230	17AUG	20.5	22.1	18.3	22.7	27.4	19.4

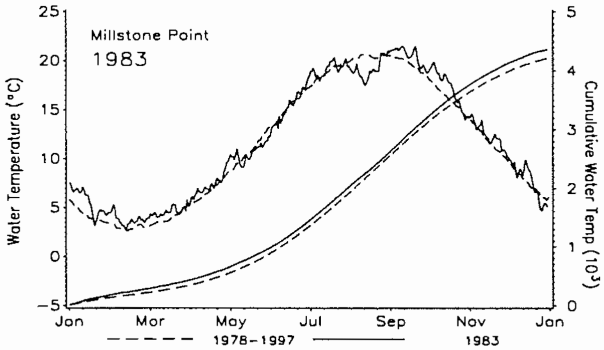
OBS	DATE	MEAN_MP	MAX_MP	MIN_MP	MEAN_NH	MAX_NH	MIN_NH
231	18AUG	20.4	22.3	18.5	22.4	26.5	19.7
232	19AUG	20.3	22.4	18.8	22.2	25.8	20.1
233	20AUG	20.3	21.9	18.9	22.1	26.4	19.9
234	21AUG	20.3	21.6	18.8	22.1	26.5	20.1
235	22AUG	20.3	21.5	18.8	22.1	25.7	20.4
236	23AUG	20.4	21.8	18.8	22.2	25.1	20.3
237	24AUG	20.4	21.9	18.9	22.0	24.2	20.0
238	25AUG	20.5	21.8	18.5	22.1	23.9	20.2
239	26AUG	20.7	22.1	19.3	22.5	24.5	20.8
240	27AUG	20.6	22.2	19.0	22.5	24.7	19.8
241	28AUG	20.6	21.8	18.7	22.5	24.6	18.8
242	29AUG	20.5	22.0	19.0	22.6	25.1	18.5
243	30AUG	20.5	22.0	19.2	22.4	24.9	19.5
244	31AUG	20.6	21.8	19.6	22.4	24.9	19.8
245	01SEP	20.5	22.1	19.3	22.3	24.6	20.2
246	02SEP	20.5	21.7	19.1	22.2	24.7	19.8
247	03SEP	20.3	21.5	18.7	22.0	24.4	18.7
248	04SEP	20.3	21.6	18.6	21.8	24.7	18.8
249	05SEP	20.4	21.8	18.8	22.0	25.1	18.7
250	06SEP	20.4	21.9	18.7	22.0	25.4	18.7
251	07SEP	20.3	22.4	18.6	21.8	24.7	18.8
252	08SEP	20.2	21.8	18.7	21.6	23.7	18.7
253	09SEP	20.3	21.7	19.0	21.6	23.2	19.3
254	10SEP	20.2	21.5	19.0	21.7	23.6	19.2
255	11SEP	20.2	21.4	18.9	21.7	23.7	19.1
256	12SEP	20.1	21.1	19.0	21.6	23.1	19.3
257	13SEP	20.1	21.4	18.9	21.6	23.5	19.1
258	14SEP	20.0	21.1	18.8	21.5	23.8	18.6
259	15SEP	19.8	20.8	18.4	21.3	23.4	18.5
260	16SEP	19.7	21.3	18.2	21.0	24.3	17.1
261	17SEP	19.6	21.0	18.5	20.8	24.0	17.1
262	18SEP	19.5	21.5	17.9	20.6	24.1	17.1
263	19SEP	19.4	21.2	17.8	20.4	22.6	17.4
264	20SEP	19.5	21.5	18.1	20.5	23.8	17.8
265	21SEP	19.4	20.8	18.1	20.5	23.1	17.6
266	22SEP	19.3	21.0	18.1	20.3	23.6	18.3
267	23SEP	19.0	20.3	18.0	20.0	24.7	17.7
268	24SEP	18.8	19.9	17.8	19.5	22.5	17.8
269	25SEP	18.8	19.8	17.6	19.3	21.5	17.3
270	26SEP	18.7	20.0	17.7	19.3	21.8	16.8
271	27SEP	18.7	19.9	17.5	19.2	21.7	17.2
272	28SEP	18.6	19.9	17.5	19.2	21.0	17.8
273	29SEP	18.4	19.7	17.6	18.8	21.0	17.0
274	30SEP	18.2	19.7	17.2	18.6	21.7	15.8
275	01OCT	18.1	19.7	15.8	18.6	20.8	15.8
276	02OCT	18.0	19.5	16.2	18.5	20.6	16.0

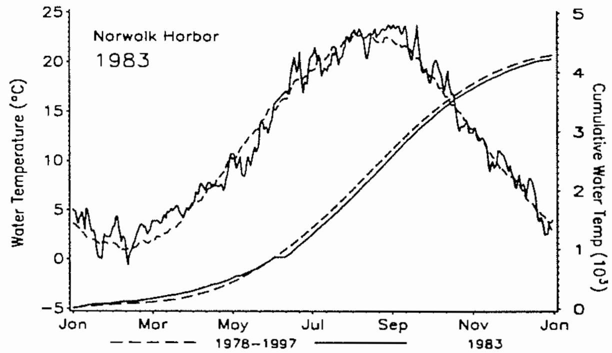
OBS	DATE	MEAN_MP	MAX_MP	MIN_MP	MEAN_NH	MAX_NH	MIN_NH
277	03OCT	17.9	19.7	16.2	18.4	21.0	15.9
278	04OCT	17.7	19.9	16.3	18.2	21.0	15.9
279	05OCT	17.6	19.5	16.1	17.9	20.9	15.3
280	06OCT	17.4	19.0	15.6	17.6	20.3	14.6
281	07OCT	17.2	18.8	15.1	17.4	19.9	14.6
282	08OCT	17.1	19.0	15.1	17.5	20.1	15.0
283	09OCT	16.9	18.9	14.9	17.1	20.4	14.6
284	10OCT	16.8	18.8	14.8	16.9	20.4	14.1
285	11OCT	16.6	18.9	14.8	16.6	20.5	13.5
286	12OCT	16.6	19.2	14.6	16.6	20.6	13.5
287	13OCT	16.5	19.3	14.4	16.6	20.6	13.6
288	14OCT	16.4	19.3	14.4	16.4	20.6	13.6
289	15OCT	16.2	18.3	14.6	16.0	19.5	14.0
290	16OCT	16.0	18.2	14.3	15.7	18.9	13.7
291	17OCT	16.0	18.6	14.5	15.7	19.5	13.6
292	18OCT	15.9	17.9	14.2	15.8	19.1	13.0
293	19OCT	15.8	17.7	13.8	15.4	18.0	13.0
294	20OCT	15.7	18.0	13.9	15.3	18.1	13.6
295	21OCT	15.4	17.6	13.7	15.1	17.7	12.8
296	22OCT	15.4	17.7	13.6	15.0	17.5	12.2
297	23OCT	15.3	18.3	13.3	14.9	18.0	12.2
298	24OCT	15.3	18.1	13.3	14.8	18.0	12.4
299	25OCT	15.0	17.6	13.3	14.4	17.3	11.5
300	26OCT	14.7	17.5	12.9	14.2	16.8	12.0
301	27OCT	14.6	17.9	12.8	14.2	17.3	11.8
302	28OCT	14.5	17.2	12.8	14.0	16.9	11.5
303	29OCT	14.3	16.5	12.6	13.7	16.7	11.5
304	30OCT	14.2	16.6	12.4	13.4	16.5	11.2
305	31OCT	14.1	16.7	12.3	13.3	16.3	10.6
306	01NOV	14.0	16.8	11.4	13.4	15.7	10.3
307	02NOV	13.9	16.9	12.0	13.2	16.0	10.3
308	03NOV	13.7	16.0	11.2	12.9	16.8	10.0
309	04NOV	13.6	15.3	11.6	12.8	16.8	10.2
310	05NOV	13.5	15.4	11.3	12.5	14.7	9.7
311	06NOV	13.2	15.2	11.8	12.4	14.2	10.6
312	07NOV	13.1	15.3	11.7	12.3	13.8	10.4
313	08NOV	13.0	14.7	11.3	12.2	16.1	9.7
314	09NOV	12.9	14.4	10.9	12.0	15.6	9.0
315	10NOV	12.8	14.5	11.0	11.8	13.7	8.7
316	11NOV	12.3	14.8	9.3	11.6	14.8	9.4
317	12NOV	12.1	14.7	9.3	11.2	14.0	9.7
318	13NOV	12.1	14.0	9.9	10.9	13.0	8.8
319	14NOV	12.0	13.6	10.5	10.9	13.0	8.3
320	15NOV	11.8	13.9	9.8	10.7	14.0	8.5
321	16NOV	11.5	13.1	9.4	10.4	12.8	8.4
322	17NOV	11.5	13.1	9.6	10.4	13.4	8.0

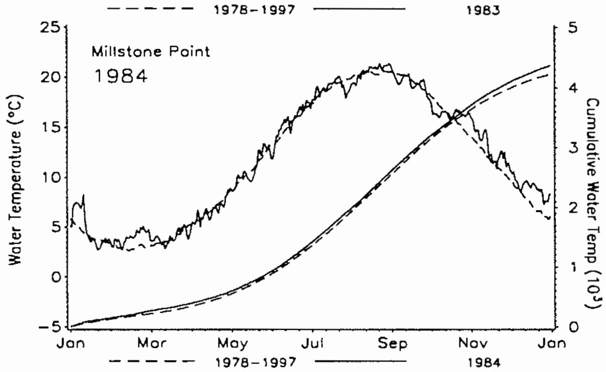
OBS	DATE	MEAN_MP	MAX_MP	MIN_MP	MEAN_NH	MAX_NH	MIN_NH
323	18NOV	11.5	13.4	9.5	10.0	12.5	7.7
324	19NOV	11.3	13.6	8.1	9.9	12.2	7.9
325	20NOV	11.1	13.3	9.0	9.7	12.4	7.8
326	21NOV	10.9	13.0	8.0	9.5	11.9	7.8
327	22NOV	10.9	12.4	8.7	9.1	11.7	7.5
328	23NOV	10.8	12.6	8.6	9.3	11.6	7.2
329	24NOV	10.6	12.7	8.1	9.1	11.5	7.4
330	25NOV	10.4	12.6	8.8	9.0	12.5	7.2
331	26NOV	10.4	12.6	8.3	8.8	12.4	6.3
332	27NOV	10.3	12.1	7.9	8.5	12.2	5.2
333	28NOV	10.1	11.7	7.6	8.4	11.1	4.1
334	29NOV	9.9	11.5	7.8	8.4	10.6	4.5
335	30NOV	9.8	10.8	7.8	8.4	10.7	6.5
336	01DEC	9.6	11.0	6.5	8.4	10.5	5.4
337	02DEC	9.6	11.1	7.0	8.2	9.9	5.2
338	03DEC	9.3	11.3	5.5	8.2	10.5	5.1
339	04DEC	9.1	11.3	5.9	7.9	10.6	4.5
340	05DEC	9.0	11.7	5.6	7.8	11.2	4.6
341	06DEC	8.8	11.3	6.0	7.7	10.7	5.1
342	07DEC	8.7	11.0	6.3	7.4	10.0	4.0
343	08DEC	8.6	10.3	6.2	7.2	9.6	3.6
344	09DEC	8.5	10.1	6.0	7.0	8.8	2.7
345	10DEC	8.3	9.9	6.3	6.7	9.0	3.4
346	11DEC	8.1	9.5	6.0	6.3	8.8	3.2
347	12DEC	8.0	9.8	5.9	6.1	9.2	2.7
348	13DEC	7.9	9.9	5.3	5.9	9.4	1.4
349	14DEC	7.7	9.9	5.3	5.8	8.7	1.2
350	15DEC	7.6	9.2	4.8	5.8	8.1	1.8
351	16DEC	7.4	9.0	4.0	5.7	7.7	2.3
352	17DEC	7.2	9.4	4.1	5.5	7.5	2.2
353	18DEC	7.0	9.5	4.4	5.2	7.5	1.9
354	19DEC	7.0	9.3	4.4	4.9	7.3	1.2
355	20DEC	6.8	9.1	3.8	4.9	7.0	1.1
356	21DEC	6.6	9.3	2.7	4.8	7.4	1.1
357	22DEC	6.6	9.7	2.2	4.7	7.4	0.4
358	23DEC	6.7	9.5	2.4	4.5	7.6	-0.5
359	24DEC	6.6	8.9	2.4	4.5	6.7	0.1
360	25DEC	6.3	8.5	2.8	4.3	7.0	1.2
361	26DEC	6.1	8.6	1.2	4.0	7.0	0.1
362	27DEC	6.1	8.6	2.3	3.9	6.7	0.0
363	28DEC	6.1	8.6	2.8	3.8	6.5	0.2
364	29DEC	5.8	8.2	2.5	3.8	6.4	-0.6
365	30DEC	5.9	8.3	2.9	3.7	6.6	-0.4
366	31DEC	6.1	8.4	4.6	4.0	5.9	1.9

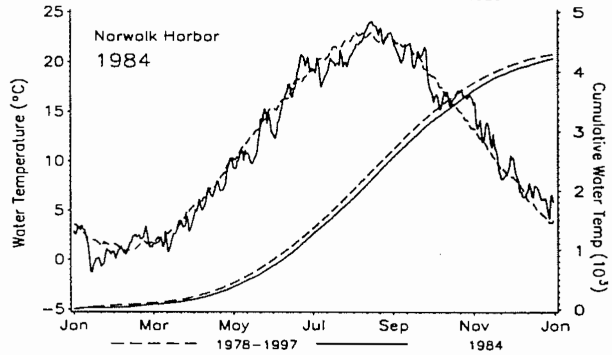
APPENDIX 2. Mean daily water temperatures (°C) at Millstone Point and Norwalk Harbor for each year from 1978 to 1998, plotted with the 20-year mean at each station. Also plotted are the corresponding cumulative water temperatures, i.e., the running sum of mean daily temperatures, intended as an index of conditions, and allowing rapid determination of whether a year is warmer or colder than average, and when during the year the deviation occurred.

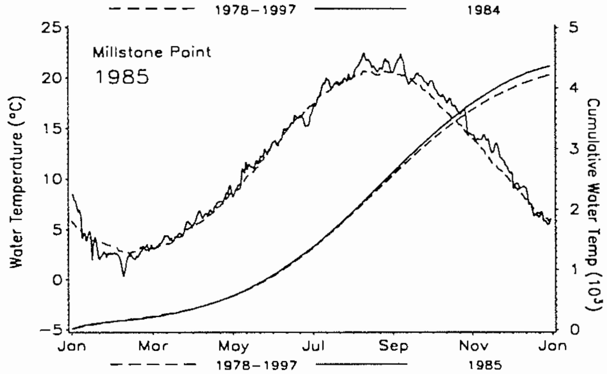

Millstone Point
1978
Norwalk Harbor
1978
Millstone Point
1979
Norwalk Harbor
1979
Millstone Point
1980
Norwalk Harbor
1980
Millstone Point
1981
Norwalk Harbor
1981
Millstone Point
1982
Norwalk Harbor
1982
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1978
1979
1980
1981
1982

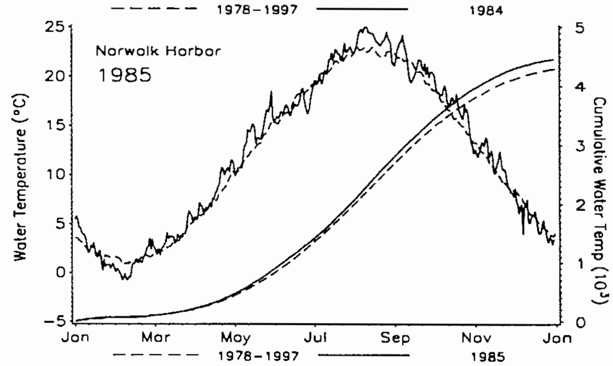

Millstone Point
1983
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1983

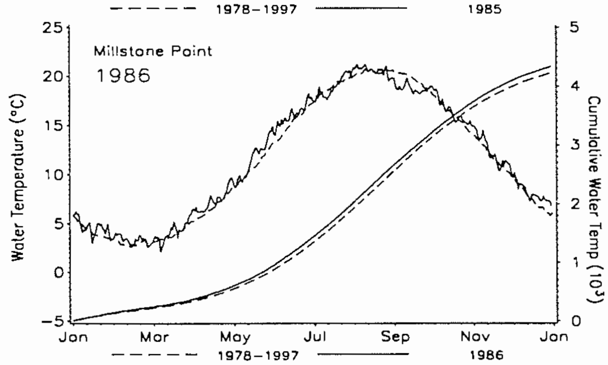

Norwalk Harbor
1983
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1983

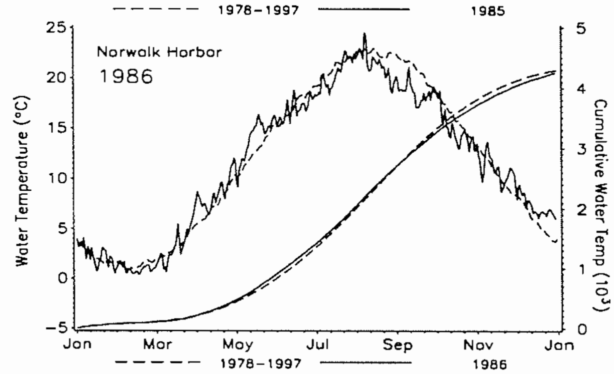

Millstone Point
1984
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1984


Norwalk Harbor
1984
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1984


Millstone Point
1985
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1985


Norwalk Harbor
1985
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1985


Millstone Point
1986
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1986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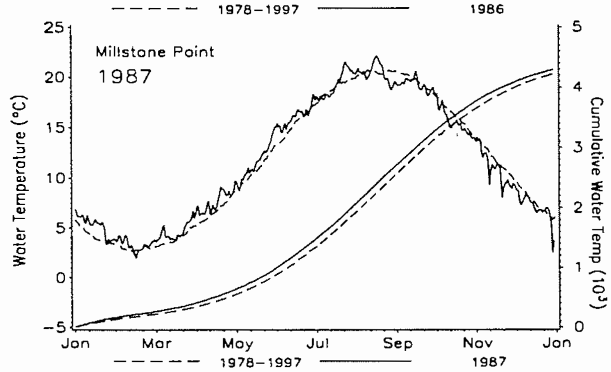
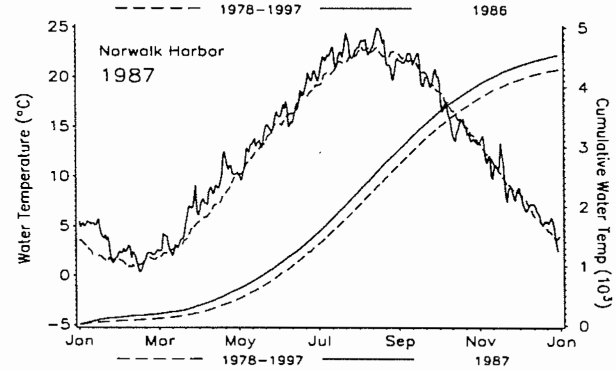
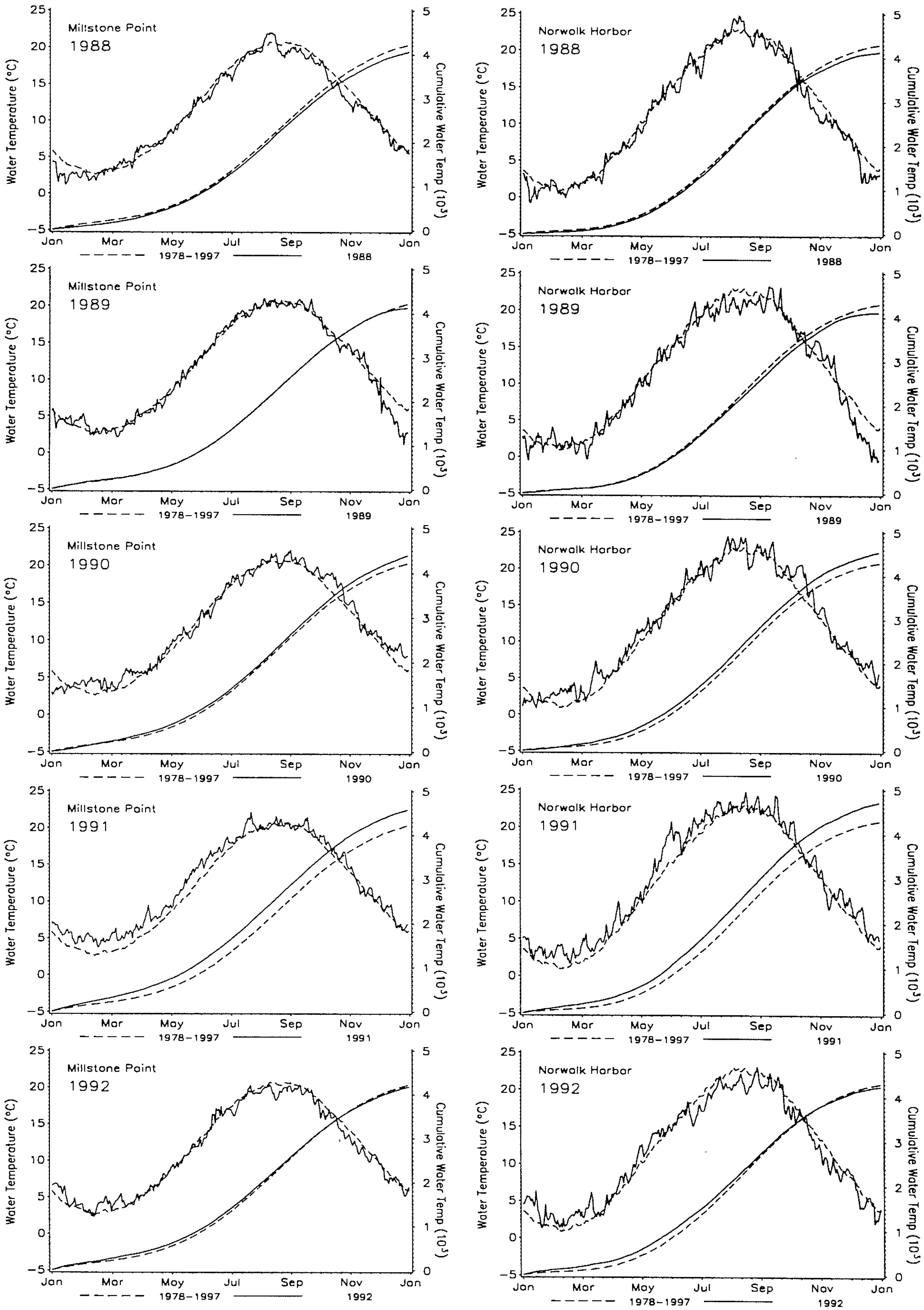
1986
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1986


Millstone Point
1987
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1987


Norwalk Harbor
1987
Water Temperature (°C)
Cumulative Water Temp (10³)
Jan Mar May Jul Sep Nov Jan
1978–1997
1987


Millstone Point
1988
Norwalk Harbor
1988
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Norwalk Harbor
1989
Millstone Point
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Millstone Point
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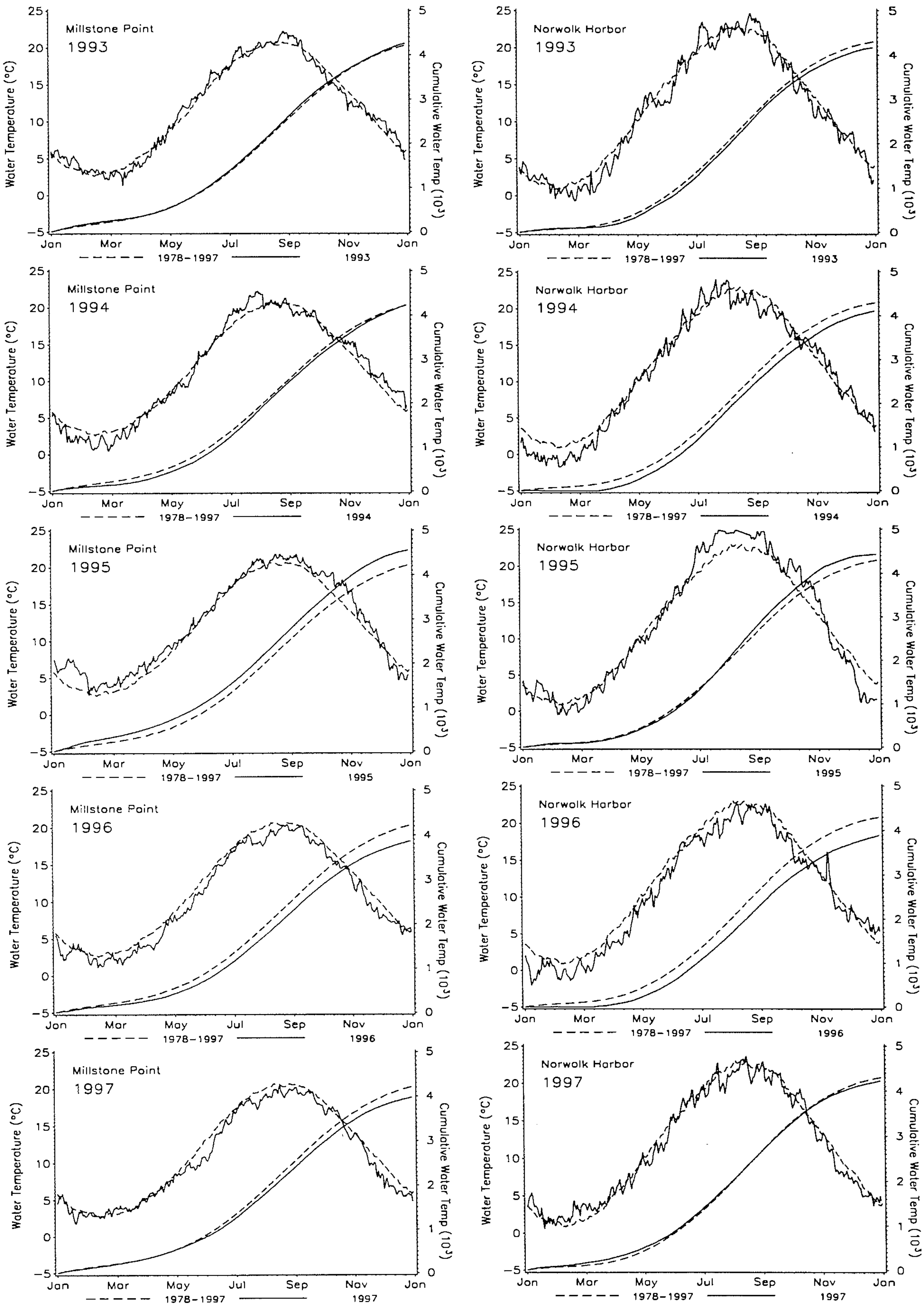
Millstone Point
1992
Norwalk Harbor
1992
Water Temperature (°C)
Cumulative Water Temp (10^3)
Jan Mar May Jul Sep Nov Jan
1978–1997
1988
1989
1990
1991
1992


Millstone Point
1993
Norwalk Harbor
1993
Millstone Point
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Norwalk Harbor
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Millstone Point
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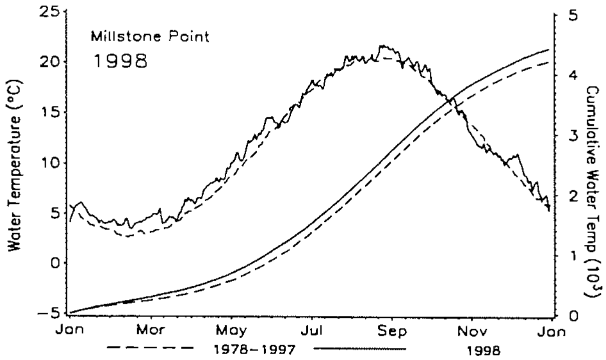
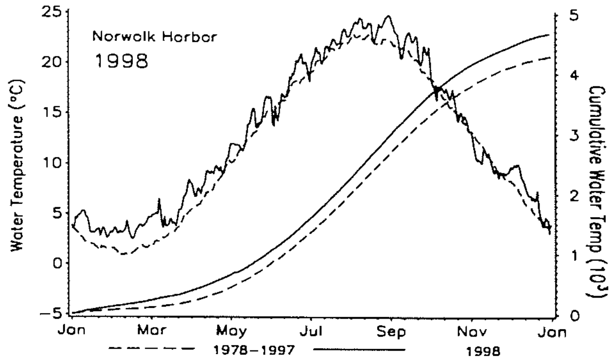
Millstone Point
1997
Norwalk Harbor
1997
Water Temperature (°C)
Cumulative Water Temp (10^3)
Jan Mar May Jul Sep Nov Jan
1978–1997
1993
1994
1995
1996
1997



Millstone Point
1998
Water Temperature (°C)
Cumulative Water Temp (10³)
25
20
15
10
5
0
−5
5
4
3
2
1
0
Jan
Mar
May
Jul
Sep
Nov
Jan
1978−1997
1998



Norwalk Harbor
1998
Water Temperature (°C)
Cumulative Water Temp (10³)
25
20
15
10
5
0
−5
5
4
3
2
1
0
Jan
Mar
May
Jul
Sep
Nov
Jan
1978−1997
1998

GROWTH, SURVIVAL AND CONDITION INDEX OF THE EASTERN OYSTER (*CRASSOSTREA VIRGINICA*) IN LONG ISLAND SOUND DURING 1997

Morgan, D.E., J.T. Swenarton, J.F. Foertch and M. Keser, Northeast Utilities Environmental Laboratory, P.O. Box 128, Waterford CT 06385; R.B. Whitlatch, Department of Marine Sciences, University of Connecticut, Groton CT 06340

ABSTRACT

Growth and survival of oysters on experimental substrates were monitored from May to November at four study sites: two at Norwalk Harbor (Sheffield Island and Manresa Island), one at Milford Harbor and one at Millstone Point, Waterford CT. Oysters and substrates were cleaned at approximately 2-week intervals to remove fouling organisms and photographed in May, June, July, September and November. Computer image analysis techniques were developed to assess oyster shell height. Substrates were removed in November and oysters were measured for comparison to image analysis data. Dry weights for shucked meat and shell for each oyster were also recorded so that a condition index (CI) could be calculated. Percentage survival was also determined in November. Results indicated that all study sites had favorable growing conditions for oysters with highest growth and survival at Milford Harbor and Sheffield Island. Temperature data indicate similar temperature regimes at Milford and Norwalk Harbors. Lower oyster growth and survival were observed at Manresa Island and Millstone Point. Heavy fouling on these plates likely increased interspecific competition (for food and space) which may have caused higher mortality. Additionally, lower water temperatures probably reduced growth rate at Millstone Point.

INTRODUCTION

The eastern oyster, *Crassostrea virginica* (Gmelin, 1791), is one of the most common bivalves of marine and estuarine nearshore habitats of the Atlantic and Gulf coasts of North America (Loosanoff and Nomejko 1949). This species supports important recreational and commercial shellfisheries throughout much of its range, including western Long Island Sound (LIS). Given its ecological and economic importance, considerable scientific research has been conducted to understand the many environmental factors that affect oyster growth and survival. Physical factors included in this research are temperature, salinity, dissolved oxygen and substrate characteristics (e.g., Loosanoff and Nomejko 1949; Haven et al. 1987; Crosby et al. 1991; Austin et al. 1993; Fisher et al. 1996). Much work has also focused on biological factors such as food quality and availability (Hofmann et al. 1992; Powell et al. 1992; Dekshenieks et al. 1993; Rheault and Rice 1996), disease (Abbe and Sanders 1988; Matthiessen and Davis 1992; Barber and Mann 1994), and interspecific competition (Osman et al. 1989; Zajac et al. 1989; Morales-Alamo and Mann 1990). Because oyster populations are often located near urban, industrialized areas, the impact of pollution on growth and survival has also been studied (e.g., Davis and Hidu 1969; Lawrence and Scott 1982; Scott and Lawrence 1982; Marcus et al. 1989).

One such area is Norwalk Harbor, CT, where concerns were raised regarding the potential effects of linear alkylbenzene dielectric fluid from underwater transmission cable leaks on productive shellfish resources. The objective of the study was to provide a baseline for assessing sublethal effects on oyster growth and survival from future leaks in this area, should they occur. This report summarizes the results of the baseline oyster growth studies conducted in 1997 in Norwalk Harbor and other sites.

MATERIALS AND METHODS

Approximately 1,000 oysters, 3-5 mm in shell height, were purchased from Aeros Cultured Oyster Company (41 Heathcote Court, Shirley, NY 11967) on June 4, 1997. These juveniles were maintained in a 60 x 90 cm floating grow-out tray with 1 mm mesh screen while the experimental substrate and rack assemblies were fabricated.

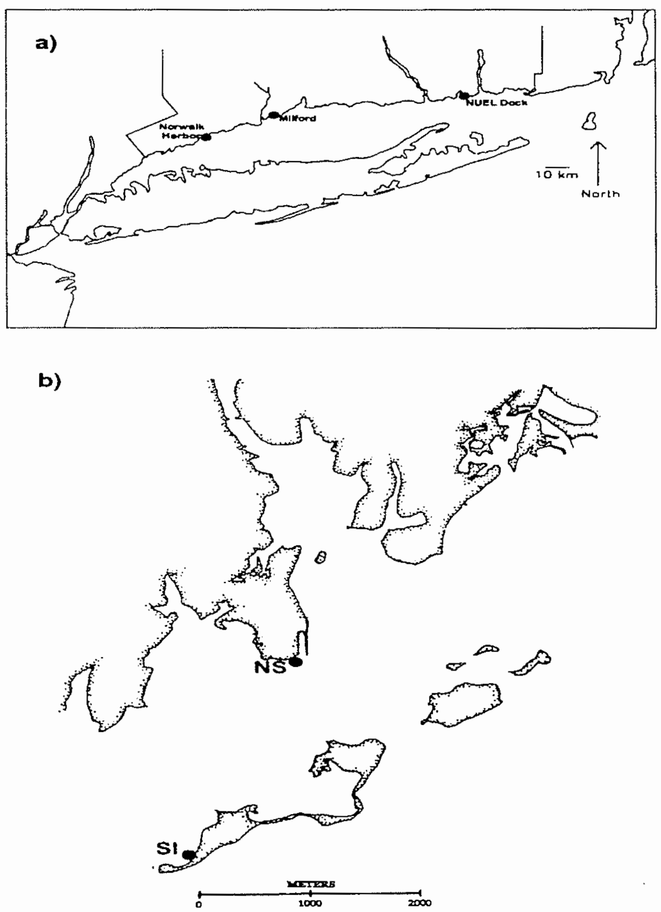


Figure 1. Location of oyster growth sites used during the 1997 study: (a) general location of sites in Long Island Sound and (b) specific locations of the two sites in Norwalk Harbor (NS=Norwalk Harbor Station, SI=Sheffield Island).

The substrate and rack assemblies used were similar to those described by Osman et al. (1989). Each substrate was a 10 x 10 cm PVC plastic square approximately 1 cm in thickness. Rack assemblies consisted of two PVC pipes (3.3 cm O.D. by 53.4 cm long) connected by a center mounted PVC pipe (2 cm O.D.) which held the two arms, each having four plates 13 cm apart. This "H" configured rack assembly held a total of 8 substrates of oysters and was suspended by a stainless steel cable (0.4 cm O.D.) which passed through a stainless steel sleeve in the center of the connecting arm. Two of these substrate and rack assembles for a total of 16 substrates were placed at each site monitored.

Four sites were selected to monitor oyster growth (Figure 1). Two were located in Norwalk Harbor near the cable array: the lighthouse dock on Sheffield Island (SI) and the outermost end of the barge dock at Norwalk Station (NS). One reference site (MD) was the dock at Connecticut Department of Agriculture, Bureau of Aquaculture, Milford. Another reference site was the Northeast Utilities Environmental Lab dock (ND) at Millstone Point, Waterford, CT. The substrates with oysters were positioned 1 meter from the bottom. Oysters were fastened to the bottom side of the plates using an underwater patching compound. Each oyster was positioned with its umbo (i.e., the beginning of its shell height growth vector) directed toward the center of the substrate. Oysters were arranged in two concentric circles of five individuals on each substrate. To track individual oyster growth, each oyster was sequentially numbered from 1 to 320 (Figure 2). After gluing 10 oysters to a substrate, the substrate was held in a flow-through seawater system overnight then attached to a rack and placed at the ND site from one to five days until all eight racks were ready for final site placement on May 28, 1997. A total of 160 oysters per site were initially deployed on that date. On June 20, 1997, each plate was thinned to 5 oysters where possible. Thinning minimized mortality caused by handling and prevented oysters from overgrowing each other. Oyster survival estimates were based on 76-80 oysters at each site starting on June 20, 1997, and ending on November 4, 1997.

Oysters were inspected every two weeks to remove fouling organisms that could affect their growth and survival. Photographs of each substrate of oysters were taken on May 28, June 20, July 21, September 17, and November 4. The 35 mm photographic data were processed using computer image analysis techniques. Incremental growth (based on shell height measurements), growth rates and survival were documented from the photographic data. In November, all surviving oysters were removed from their substrates to compile the following: shell height (measured to the nearest 0.1 mm using a vernier caliper), oyster shell wet weight, oyster meat wet weight, dry shell weight, and dry meat weight. Prior to recording wet weights, each oyster was thoroughly cleaned of surface fouling organisms and shucked. The shell was placed in one preweighed aluminum foil pan and the meat and fluids in a second pan. After recording wet weights, the shell and meat were placed in a drying oven at 85 ^{o}C for 48 hours and reweighed for dry weights. These latter data were used to calculate a condition index (dry weight of meat/dry weight of shell x 100 [Rainer and Mann 1992]).

Water temperature data were obtained using Northeast Utilities' Environmental Data Acquisition Network from continuous temperature recorders maintained in cooling water intakes at NS and ND. Daily average water temperatures were based on data recorded at 15 minute intervals. Water temperatures at MD were based on daily measurements made by National Marine Fisheries Service, Milford, CT.

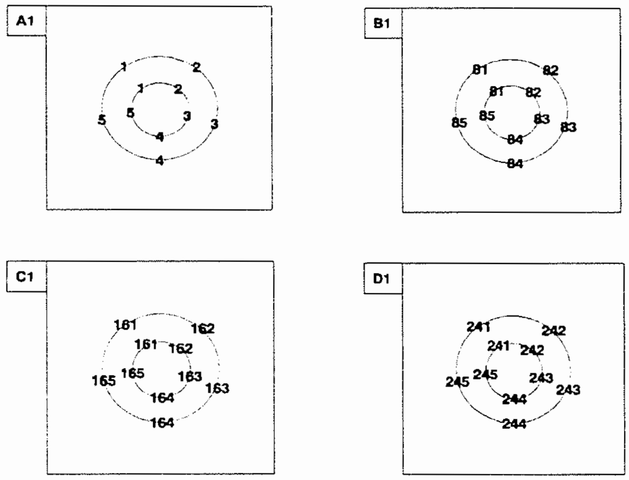


Figure 2. Diagrams of four oyster plates showing how oyster pairs were arranged in two concentric circles with each pair consisting of an inner and an outer location.

RESULTS

Water Temperature. Water temperatures at NS, MD and ND during the study period are summarized in Figure 3. Average daily water temperatures at Norwalk Harbor were 9-11°C in early May, reached a maximum of 24°C by mid-August, and declined to 7°C at the end of November. Average daily temperatures at ND were generally 1-4°C cooler than those measured at NS from May through September (maximum in August of 20°C), but were more similar later in the year. Measurements from MD indicated that water temperatures there were more similar to, or higher than, those measured at NS.

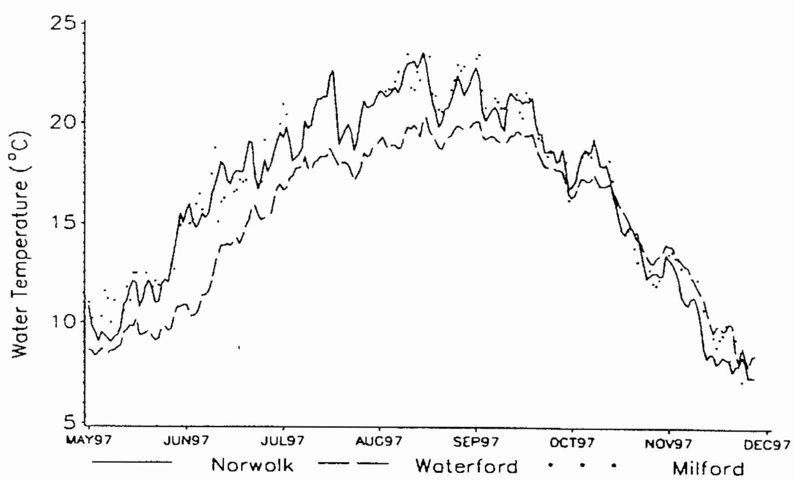


Figure 3. Water temperatures at oyster growth study sites. Norwalk and Waterford data based on daily averages of temperatures recorded at 15 min. intervals at cooling water intakes at Norwalk Harbor Station and Millstone Nuclear Power Station, respectively. Milford data are single measurements of Milford Harbor water provided by National Marine Fisheries Service, Milford, CT.

Oyster Growth, Condition, and Survival. Oyster growth, based on mean shell height in November, was greatest at SI (65.6 mm) and MD (65.1 mm); these mean shell height estimates were not significantly different (Figure 4a; Table 1). Growth was significantly lower at NS (60.7 mm) and ND (60.3), with no significant difference in growth between these two sites. Growth rates, defined as incremental growth (height increase in mm·mo^{-1}) also varied among stations (Figure 4b). Growth rates during the May-June period were lowest at ND (5 mm·mo^{-1}) and highest at MD (9 mm·mo^{-1}). Growth rates were highest at all sites during the June-July period, ranging from 15 mm/month at ND to 19 mm/month at MD. Growth rates were most similar among sites during the July-September and September-November periods.

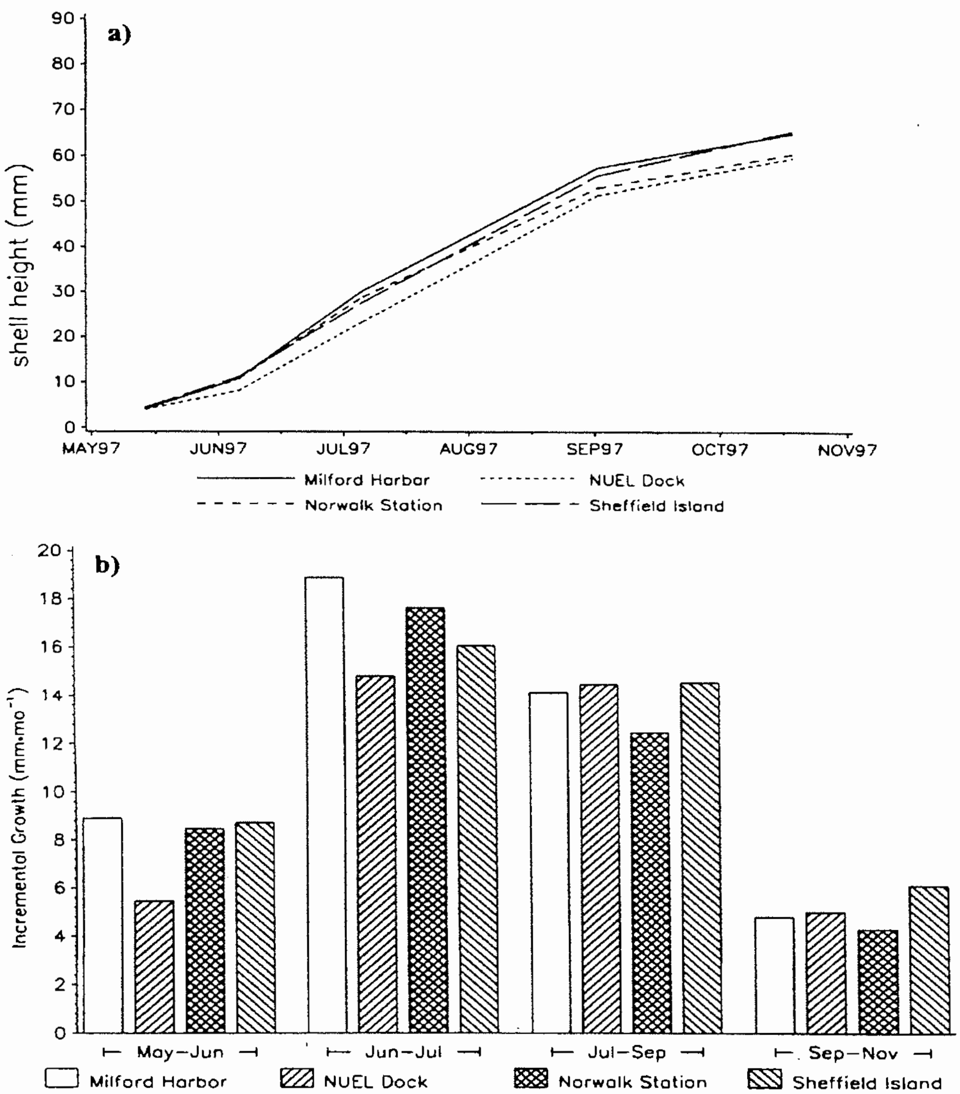


Figure 4. Oyster growth at the four study sites expressed as (a) shell height (mm) and (b) incremental growth rate (height increase in mm·mo^{-1}).

Oyster condition indices were similar to those observed for shell height for all the sampling sites (Table 1). Condition index was highest at MD (7.60) and SI (6.15); the difference between these indices was statistically significant. Significantly lower condition indices were noted at NS (5.28) and ND (5.15), with no significant difference between these sites.

Table 1. Survival, growth and condition index data for oysters at Sheffield Island (SI), Norwalk Station (NS), Milford Dock (MD) and NUEL Dock (ND).

Site	Oyster Survival[a]			Shell Height (mm)[b]				Shell Height (mm)[b] November[d]			Condition Index[c] November		
	I[#]	F[#]	%S	May Mean	June Mean	July Mean	Sept Mean	Mean	SE_{2x}	Tukey's	Mean	SE_{2x}	Tukey's
SI	80	75	93.7	4.4	11.1	27.7	55.8	65.6	2.08	A	6.15	0.28	B
NS	76	62	81.6	4.1	10.6	28.8	53.1	60.7	1.13	B	5.28	0.30	C
MD	78	78	100	4.0	10.7	30.2	57.5	65.1	1.23	A	7.60	0.22	A
ND	79	72	91.1	4.0	8.1	23.3	51.5	60.3	1.48	B	5.15	0.18	C

[a] Percentage survival (%S) of oysters based on the final number in November (F[#]) divided by the initial number (I[#]) in June, as recorded following the thinning of oysters from 10 to 5 individuals per plate.
[b] Shell heights (longest shell dimensions) obtained using computer assisted image analysis of oyster pictures.
[c] (Dried meat weight divided by the dried shell weight) multiplied by 100.
[d] November data for shell heights and condition index are compared using the standard error of the mean doubled (SE_{2x}), which is roughly equivalent to a 95% confidence interval, and Tukey's Studentized Ranged Test where the means having the same letter are not statistically (alpha=0.05, df=272, MSE=1.136916) different.

Oyster survival over the study period (Table 1; Figure 5).was highest at MD (100%), intermediate at SI (93.7%) and ND (91.1%), and lowest at NS (81.6%) .

DISCUSSION

The results of this study indicated that conditions for oyster growth and survival were suitable at all study sites during 1997, and growth and survival data were consistent with other research conducted in LIS (e.g., Loosanoff and Nomejko 1949; Matthiessen and Davis 1992). Spatial and temporal differences in growth and survival were apparent, with some of the variability attributed to differences in temperature regimes between the sites.

Water temperature has often been cited as an important environmental variable for oyster growth (e.g., Loosanoff and Nomejko 1949, 1951; Hofmann et al. 1992; Austin et al. 1993; Dekshenieks et al. 1993; Fisher et al. 1996). MD appeared to have the best conditions of the four study sites based on oyster growth, condition index and overall survival. More estuarine characteristics of upper Milford Harbor predominate at this site, where lower salinity, higher water temperatures, and perhaps greater food availability combined to create an ideal environment for the oysters.

Similar water temperature regimes and possibly food supply in NS resulted in oyster growth, condition indices and survival rates that were nearly as high as those observed at MD. Of the two sites in NS, environmental conditions were better at SI, where oyster shell height growth was comparable to that observed at MD. Lower growth, condition index and survival at NS was likely the result of heavy fouling on these plates, primarily from encrusting bryozoans, colonial tunicates, hydroids and barnacles. Fouling can negatively impact oysters by increasing competition for planktonic food, smothering from overgrowth and interspecific competition for space (Osman et al. 1989; Zajac et al. 1989; Morales-Alamo and Mann 1990). Also, the higher level of fouling necessitated more rigorous cleaning, which was observed to loosen oysters from the substrates in several instances.

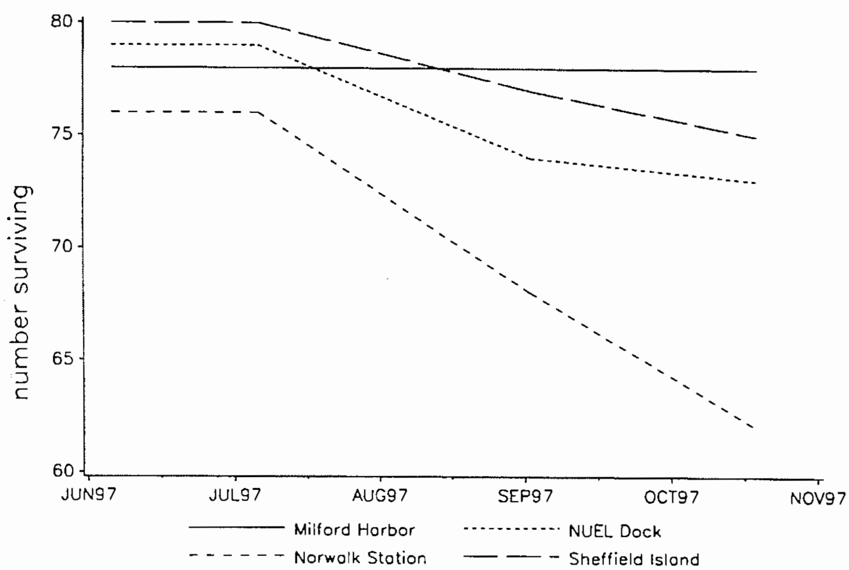


Figure 5. Oyster survival from June to November 1997 at the four study sites.

Given the importance of temperature and food supply, it is not surprising that growing conditions at ND in eastern LIS were not as favorable as those observed at the other sites. Oysters still grew well at the site, but cooler water temperatures delayed and likely shortened the growing season. Because nutrient levels are generally lower in eastern LIS (USEPA 1993), lower plankton densities may also have reduced the food supply for oysters to some degree. Some heavy fouling was also observed at ND, and may have also contributed to relatively lower growth and survival, as described above for NS.

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MONITORING OYSTER HEALTH IN LONG ISLAND SOUND

Sunila, I., State of Connecticut, Department of Agriculture, Bureau of Aquaculture, P.O. Box 97, Milford, CT 06460

ABSTRACT

The oyster health monitoring program in Long Island Sound (LIS) was started by State of Connecticut, Department of Agriculture, Bureau of Aquaculture, in 1997. Several sites representing both leased oyster growing areas and seed areas are monitored for the presence of pathogens and histopathological changes. Several factors cause pathological changes in oysters: variations and extremes in temperature and salinity, pollutants, oxygen deficiencies, age, predation, inappropriate substrate, nutrient availability, plant toxins and as the most important factor, infections caused by parasites, viruses, and bacteria. The oyster responds to pathological insults by an array of pathological changes: inflammatory responses (acute or chronic), degenerations (inclusions, vacuolization, ceroid), cell and tissue death (necrosis, apoptosis), growth derangements (hyperplasia, metaplasia), hemodynamic and fluid derangements (edema, hemorrhage), neoplasia (benign or malignant), teratological changes and genetic diseases. Isolated cases of all pathological categories were detected, but the most important factor affecting oyster health at the present time is MSX (multinucleated sphere X) disease. MSX is caused by a Protozoan parasite *Haplosporidium nelsoni*, which occurs at several sites in LIS at epizootic prevalences. Monitoring oyster health has significant applications: 1) determining safety of an animal used as a food product; 2) developing disease-resistant, highly productive strains for aquaculture; 3) using oysters as bioindicators of changes in water quality.

INTRODUCTION

The Eastern Oyster (*Crassostrea virginica*) is one of the most economically and ecologically important animals in Long Island Sound (LIS). The oyster aquaculture industry for the State of Connecticut ranks first in the entire U.S. for dollar value ($ 60,000,000 in 1995) and second for production (over 700,000 bushels in 1995, DEP 1995). The oyster industry leases grounds from the state. Over 46,000 acres of oysters are cultivated on large underwater farms. Seed areas, situated up the rivers or close to the shoreline, rely on natural spat setting on freshly distributed culch; but hatchery-raised seed is also used in some areas.

Oysters are major links in LIS's estuarine ecosystem; they transfer nutrients from the water column to sedimentary fauna and flora (Figure 1). One oyster may filter over 500 l of seawater daily in order to retain zoo- and phytoplankton for its nutrition (Haven and Burrell 1982). LIS's oyster population can filter the entire volume of 67 billion tons of seawater in LIS within a year. Oysters support populations of macroalgae by providing them nutrients and sedimentary animals, such as polychaetes and clams, through production of pseudofaeces. They help form hard bottom by binding calcium carbonate from the water column to their shells. Oyster eggs (15 to 115 million eggs/female during one spawn) together with oyster larvae provide food for several species of fish and crustaceans. Oysters are predated by gastropods (*Urosalpix cinerea* and *Eupleura caudata*) and starfish (*Asterias forbesi*) (Loosanoff 1956). Oysters can accumulate pollutants, such as heavy metals or polychlorinated biphenyls, which may be a hundredfold higher in concentration in the tissues in comparison to concentrations in seawater. This characteristic makes the oyster a useful bioindicator organism and is used by NOAA's National Status and Trends program to reflect environmental quality of seawater (O'Connor and Beliaeff 1995). The health of an oyster reflects the health of the entire water column.

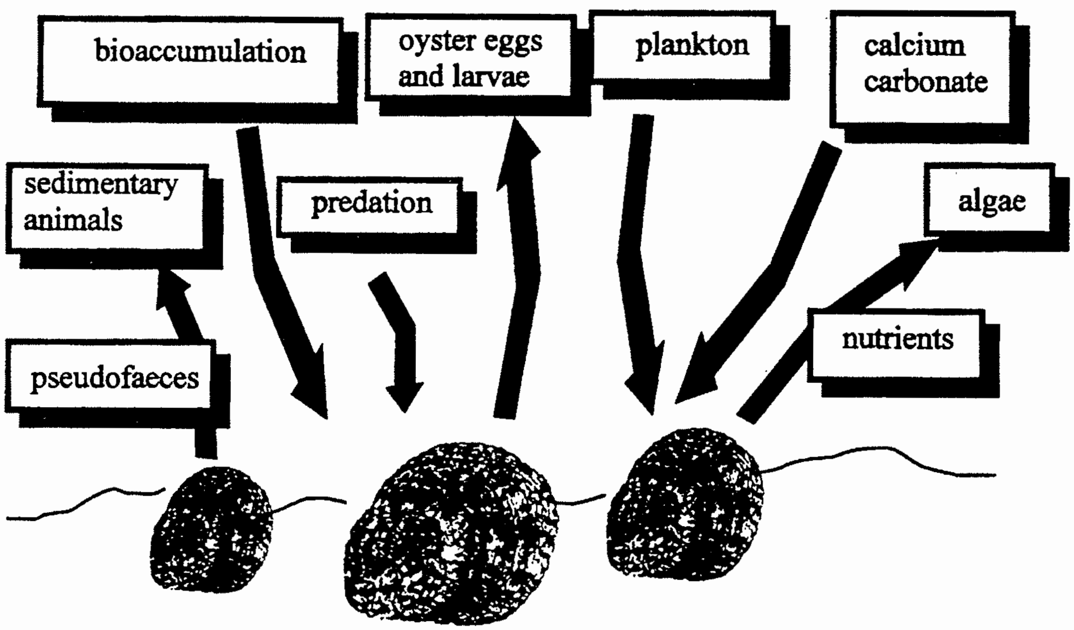

Figure 1. The eastern oyster and major links in the estuarine ecosystems of Long Island Sound.

Several factors affect the health of an oyster, Figure 2 (from Sindermann 1980). Predation by *Urosalpix cinerea*, *Eupleura caudata* and *Asterias forbesi* cause significant mortalities in LIS. Fouling organisms (slipper shells *Crepidula fornicata* and *C. plana*; barnacles *Balanus eburneus*, *Chthamalus fragilis*; bryozoa *Schizoporella unicornis*) compete for the same food source as the adult oyster, and also for setting space with oyster larvae on cultch. Malnutrition, which may cause poor growth and muscle atrophy, can be induced by trying to grow oysters in nontraditional oyster habitats. To provide appropriate substrate, clean oyster shell, cultch, has been spread by the State of Connecticut and private oyster companies since 1987. Siltation still causes problems in some areas, preventing filter feeding and causing malnutrition. Salinity variations and extremes facilitate the spreading of certain parasites such as MSX (multinucleated sphere X), *Haplosporidium nelsoni* (Farley 1975). Also, temperature variations and extremes facilitate spreading of certain parasites such as Dermo (*Perkinsus marinus*). Long-term warming was speculated to be a possible reason for the range extension of *Perkinsus marinus* to LIS and neighboring areas after 1991 (Ford 1996). Certain types of phytoplankton (*Nitzchia closterium*, *Prorocentrum triangulatum*, and *Chlorella sp.*), either the cells themselves or their metabolites, can have detrimental effects on an oysters' feeding

(Galtsoff 1964). Detrimental effects of toxic dinoflagellates on shellfish are reviewed by Shumway (1990).

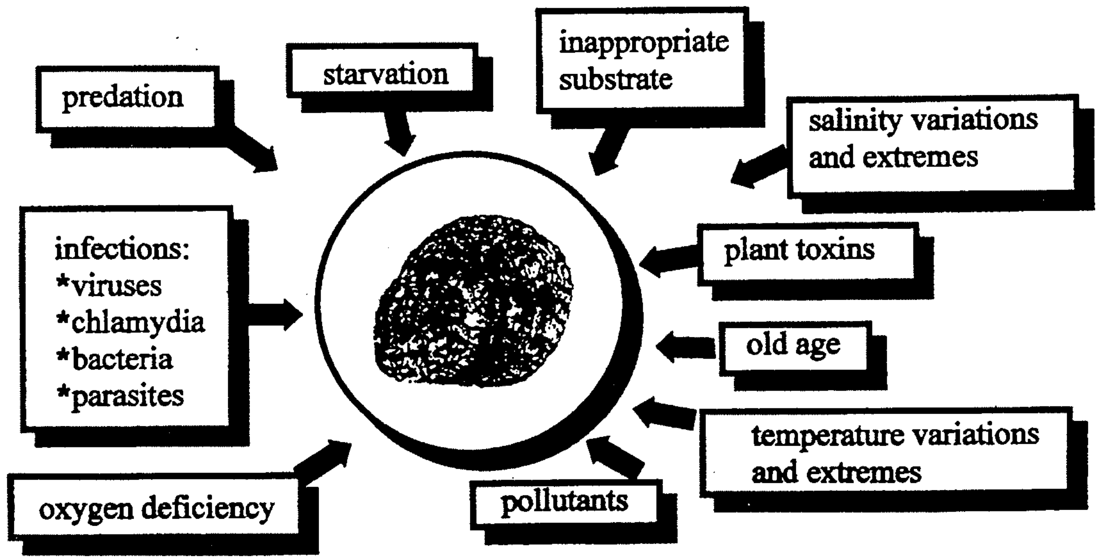

Figure 2. Factors causing pathological changes in the oyster.

Oysters are long-lived organisms, and aging can serve as a pathological irritant (Figure 2). Twenty- and thirty-year-old oysters are found regularly in LIS, and the number of layers in the shells of some unusually large specimens indicate that they may reach 40 years of age (Loosanoff 1965). Oysters have a high degree of tolerance over a wide range of environmental conditions, but high concentrations of pollutants can cause histopathological changes (Yevich and Barszcz 1983). Because of the oysters sessile lifestyle, accumulated concentrations and pathological changes reflect conditions at a specific sampling site. The following histological changes may be useful for monitoring pollution effects: neoplasia (Sindermann et al. 1980), atrophy of diverticular epithelium (Tripp et al. 1984) and formation of kidney stones (Rheinberger et al. 1979). Further, oxygen deficiency may serve as a pathological irritant (Figure 2). This is an especially important factor in LIS, which has a tendency to develop hypoxic conditions in late summer (Arnold 1988). The most important factors affecting the health of oysters are infectious agents such as parasites. Parasites have caused epizootics and mass mortalities, often due to transfer of susceptible animals into epizootic areas, or of individuals from such areas (Sindermann and Rosenfield 1967).

Any of the factors summarized in Figure 2 can serve as a pathological insult which may cause different types of pathological changes in the oyster. Categories of molluscan pathology are listed in Figure 3. (1) Inflammatory responses: acute - characterized by granular hemocyte infiltration and chronic - characterized by hyaline hemocyte infiltration; (2) Degenerations: ceroidosis (accumulation of lipofuchsin pigment), vacuolization, inclusions and atrophy; (3) Cell and tissue death: necrosis, pyknosis, apoptosis and lysis; (4) Growth derangements: hyperplasia, hypertrophy, metaplasia, xenoma; (5) Hemodynamic and fluid derangements: edema and hemorrhage; (6) Neoplasia: benign or malignant; (7) Genetic diseases; (8) Teratological changes.

The State of Connecticut, Department of Agriculture, Bureau of Aquaculture started an oyster health monitoring program in 1997. The approach is holistic: any factor causing pathological changes listed in Figure 2 may cause conditions listed in Figure 3. Different environmental stimuli and their manifestations in the oyster form a complex network; understanding that network is necessary to find out reasons for oyster mortalities and to develop management strategies to control them.

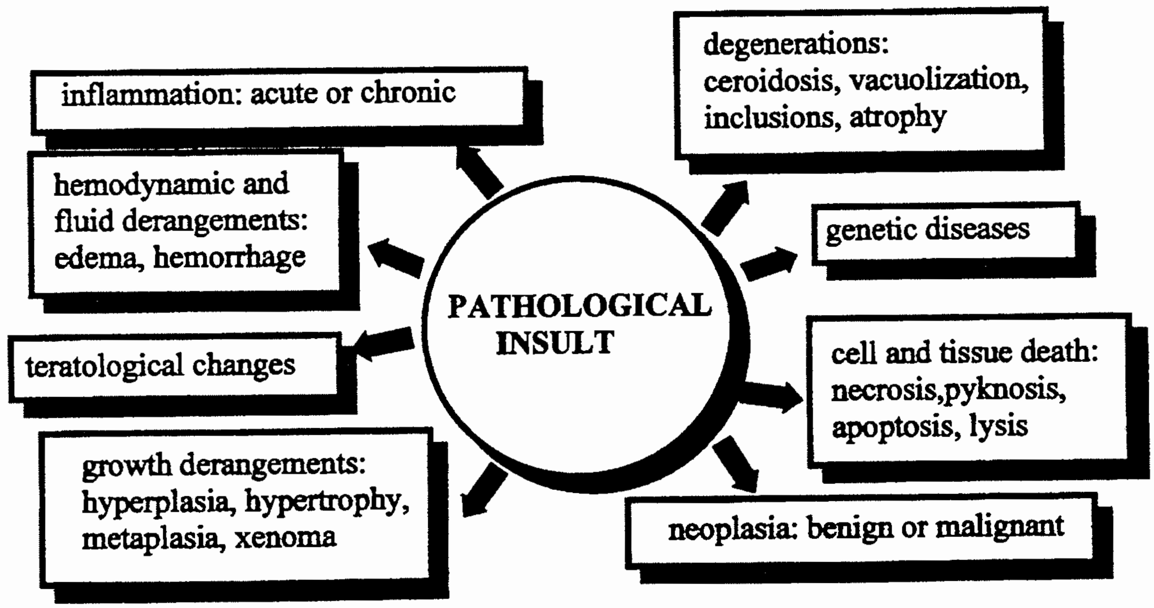

Figure 3. Categories of pathological changes in oysters.

MATERIALS AND METHODS

Oysters were collected all along the 250 mile shoreline of Connecticut, and in 1998 also from the south shore of LIS, New York. Sample sites represented either leased oyster growing areas or seed areas. More samples were collected from areas of great economic interest (e.g. Norwalk), but wild oyster populations were also sampled to gain an adequate geographical representation. Most of the samples originated from natural set, but hatchery-raised seed was also used in some areas, especially in the eastern part of Connecticut's shoreline, where natural set is scarce. Typically, each cultivated oyster is transplanted three or four times before it is marketed which exposes each oyster to parasitic infestation, chemical contamination or other factors listed in Figure 2 at several sites during its lifetime.

Results of this study were based on 22 sampling sites in 1997 and 29 sampling sites in 1998. Most of the sampling was accomplished during autumn when oyster mortalities and associated parasite prevalences were highest. Thirty oysters were collected from each sampling station. One section of each oyster, demonstrating the most important tissues, was fixed in

Davidson's fixative (in 20‰ artificial seawater). Samples were embedded in paraffin and 6 μm thick sections were stained with hematoxylin-eosin. Special stains were applied for further diagnosis of certain pathological conditions, such as Ziehl and Harris' Hematoxylin to detect acid-fast spores, Mallory's Trichrome for collagen, and Malt Periodic Acid-Schiff-Alcian Blue for differentiation between acid and neutral mucopolysaccharides in tumor tissues according to Howard and Smith (1983). Samples of anal-rectal tissues were cultured in Ray's Thioglycollate Medium according to Ray (1954) to detect infection by Dermo, *Perkinsus marinus*.

RESULTS AND DISCUSSION

The following pathological conditions were present in the oysters:

Infections caused by viruses. Viral gametocyte hypertrophy, *Papovaviridae* (Farley 1985), was detected in 2 out of 1530 oysters. However, a sample from Fishers Island, NY, had a 12% prevalence with a complete castration of one specimen. Infected cells were basophilic with fine granular material inside. Viruses infected either ova (ovacystis) or sperm, pushing the chromatin to the side and inducing massive hypertrophy of the host cell.

Infections caused by prokaryotes. Prokaryotic inclusions, Chlamydiae, Mycoplasms or Rickettsiae (Harshbarger et al. 1976), were detected in the stomach and intestinal epithelia or digestive tubules throughout the samples. They formed finely granular, basophilic masses inside the cells. Infections were light and were present in 4% of the oysters.

Infections caused by bacteria. No bacterial infections were detected except in association with post-mortem changes.

Infections caused by parasites. The most important factor affecting oyster health in LIS during the study period was infection by the protozoan parasite MSX, *Haplosporidium nelsoni*, which was associated with massive mortalities at several sites. Sampling started in the middle of epizootics with prevalences up to 87%. MSX-disease has caused heavy mortalities of oysters in Delaware Bay since 1957 (Haskin et al. 1966) and in Chesapeake Bay since 1959 (Andrews and Wood 1967), decimating oyster production in both areas ever since. Evidence of an earlier MSX-event in LIS in 1985 was presented by Haskin and Andrews (1988), but oyster production continued rising to reach 1995's peak values of over 700,000 bushels (DEP 1995). MSX is usually present as a multinucleated cell, plasmodium; but sporulation with acid-fast spores was also detected in several specimens. While major seed areas were uninfected when sampled in 1997, samples taken in 1998 showed infection had spread to these areas. MSX infection caused pathological changes in the oysters: inflammatory response in the mantle (perivascular hyaline hemocyte infiltration), hyaline hemocyte infiltration in the gills in the initial infections and systemic hyaline hemocyte infiltration in the terminal infections. Terminal cases represented areas of tissue lysis and necrotic cells, and sloughing of gill epithelia. Detailed data of LIS's MSX epizootics is presented by Sunila et al. (2000).

The apicomplexan parasite Dermo, *Perkinsus marinus*, has high prevalences in LIS (Brousseau et al. 1998). Dermo disease caused massive oyster mortalities in the Gulf of Mexico in the 1940s. It has caused chronic and occasionally high mortalities in the Chesapeake Bay. Since 1990, Dermo has been detected in Delaware Bay, LIS, Massachusetts, Rhode Island and Maine (Ford and Tripp 1996). Every sampling station of the present article was infected. Detailed data of LIS's Dermo prevalences are presented elsewhere (Karolus et al. 2000). Despite high prevalences and high intensities at some sampling sites, no histopathological changes indicative of terminal stages and approaching death were usually detected. Such changes could include sloughing of epithelia, atrophy of the epithelia of digestive tubules, occlusion of hemolymph vessels with parasites and hemocytes, tissue lysis and forming of abscesses containing parasites and hemocytes (Ford and Tripp 1976). This is in accordance with the observations of Ford and Tripp (1996), who stated: "In Long Island Sound and southern Massachusetts areas, mortalities have been relatively low or even absent despite infection prevalence and intensities that rival those in more southern regions." The histopathological marker associated with Dermo-infection was the accumulation of ceroid, lipofuchsin pigment, in the tissues of infected animals.

Nematopsis ostrearum, a gregarine parasite of the oyster, was present at several sites. Gymnospores were present in the tissues, especially in the mantle lobes, without any obvious health effects. For example, 100% of the oysters sampled were infected in the Guilford-Madison-Clinton area, while the parasite was absent from oysters sampled in Norwalk. The life cycle of *N. ostrearum* includes a mud crab host. *N. ostrearum* was thought to cause extensive oyster mortalities in Virginia and Louisiana, but further studies have failed to confirm that. Co-infection with Dermo might have caused the mortalities described in earlier reports (Sindermann and Rosenfield 1967). Another gregarine species (Sprague's Gregarine) was observed in the digestive ducts in less than 1% of the specimens sampled.

Several species of ciliates from the Family *Ancistrocomidae* were detected on the epithelium of the gills, mantles and palps with prevalences up to 53% (Groton). *Stegotricha*-like ciliates were frequently observed on the stomach epithelium and in the epibranchial chamber where they occurred in high numbers, but without any obvious health effects. Xenoma-forming ciliates, previous *Sphenophrya*-like ciliates (Otto et al. 1979), were observed in 1% of the oysters. Some of the xenomas on the gills were large enough to be visible to the naked eye. They were formed by single ciliates which invaded gill cells. They proliferated and caused massive hypertrophy of the host cell which could harbor hundreds of ciliates, each having one macro- and one micronucleus. Some of the xenomas were observed to erupt, releasing free ciliates between gill demibranches.

Metacercaria of the Family Bucephalidae infect gonads and other organs of the eastern oyster, occasionally causing castration (Lauckner 1983). Prevalence of trematodes averaged 2% with a few metacercacia or cercaria in the tissues, but exceeded 10% with heavy infiltration and sterilization of the gonad in a sample from Groton. While no samples contained nematode infections, flatworms with ciliated epithelium were regularly observed. The turbellarian flatworms had pigmented spots and often were bearing larvae inside. They were most frequently observed in the intestine, but also in the gills of heavily infected specimens. Prevalence of turbellarians was less than 1%, but exceeded 50% in a site from New Haven Harbor.

Pinnotheres ostreum was detected in less than 1% of the specimens examined. Their presence caused massive gill lesions. Unusually low, deformed gills had folded edges, fused gill filaments and excessive mucus production. This may interfere with feeding, since the food groove was situated in the middle of the lesions. Gill anomalies caused by *Pinnotheres ostreum* have been previously reported by Christensen (1958).

The following histopathological changes occurred in the osyters:

Inflammatory responses. Acute inflammatory responses included occlusion of hemolymph vessels and infiltration of granular hemocytes in 1% of the specimens. The most common reason for acute infection was the post-spawning status of the samples and the cleaning process of follicles and gonoducts which induced massive migration of hemocytes. Chronic inflammatory responses were commonly present in the samples with MSX-disease.

Degenerations. Degenerations are non-lethal and reversible changes. Accumulation of lipofucsin pigment (ceroidosis) was frequently present because of infection with Dermo. Degeneration of digestive tubule cells surrounded by a focal inflammatory reaction of unknown etiology was observed in one specimen.

Cell and tissue death. Necrotic cells have passed the "point of no return" and are on an irreversible path toward cell death. Necrotic and lysing cells were observed in terminal stages of MSX-disease. Hemocytes with morphological characteristics of apoptosis were observed in advanced Dermo-infections.

Growth derangements. Hyperplasia: Mucus cell hyperplasia was observed on the mantles in less than 1% of the animals sampled. *Hypertrophy:* Hypertrophy was observed in association with *Papilloma*-virus infections of the gametes (viral gametocyte hypertrophy) and in xenomas in the gills. *Metaplasia*: Dilatation of digestive diverticula beyond the normal variation from absorptive phases to reconstitutional phases (Langdon and Newell 1996) was observed in less than 1% of the specimens. In these cases, each digestive tubule epithelial cell in the section was abnormally replaced by flat squamous epithelium. *Sclerosis*: hardening of the tissues with increased collagen deposition, was observed in less than 1% of the specimens. Basement membranes around hemolymph vessels and ground substance around the stomach were thickened. In advanced cases, collagen fibers were deposited everywhere in the connective tissue.

Hemodynamic and fluid derangements. Edema of the mantle with unknown etiology was observed in less than 1% of the specimens. Fluid-filled, swollen mantle with loose remnants of connective tissue inside surrounded the entire specimen. Ulcers of the stomach or intestine epithelium were observed in 2% of the specimens. Ulcers had different stages of maturation starting from small gaps between epithelial cells with subsequent leaking of hemolymph to the stomach cavity to larger areas of missing epithelium surrounded by massive inflammatory responses and hemolymph-filled stomach cavities. Unexplained hemorrhage in the intestine or stomach may be due to ulcers which were not present on the section. Etiology of oyster ulcers is unknown, but Sunila (1984) reported the correlation of ulcers and hemorrhages of the digestive tract to polluted sampling sites in the common mussel, *Mytilus edulis*.

Neoplasia. Cases with disseminated neoplasia, the most common type of malignant neoplasia in bivalves (reviewed by Elston and Moore 1992), were not detected. Instead, several specimens with adenocarcinoma *in situ* were found at several different sampling stations. Well-differentiated lesions occurred at the intestine epithelium and in some cases at the stomach epithelium. There were glandular formations which were composed of cytologically benign looking, but mitotically active, cells. There was PAS-positive exudate in the lumen. Detailed description of these lesions will be presented elsewhere. This condition might be related to environmental carcinogenesis because the only previous description of this tumor was reported in eastern oysters exposed to Black Rock Harbor sediment (Gardner et al. 1991).

Genetic diseases or teratological changes. Pathological changes related to these categories were not observed in the samples.

CONCLUSIONS

Monitoring oyster health has several applications:

Public health aspects. This is the health status of an organism used as a food product and which is usually consumed raw. None of the conditions listed above is known to be harmful to humans.

Aquaculture aspects. Aquaculture ventures take disease status into account when developing management strategies and aim at high production figures even with existing oyster diseases. Oysters develop resistance for MSX and Dermo among native oyster stocks, and this process can be facilitated by selective breeding in a hatchery. The State of Connecticut, Department of Agriculture, Bureau of Aquaculture has taken the initiative to develop disease-resistant, hatchery-raised oysters for LIS. A proposal was made for the commercial oyster companies to not harvest 10% of their infected lots for a period of three years. This would establish broodstock sanctuaries in the field and permit survivors to produce resistant seed.

Biomonitoring. Histological sections are viable for at least one hundred years, giving them archival usefulness. An oyster health monitoring program will establish a permanent record of oyster health at several sampling sites in LIS at the time of sampling. These samples will serve as valuable baseline material in the event of an environmental insult, such as a contaminant spill, or discharge of polluted waters from a factory.

ACKNOWLEDGEMENTS

I warmly thank my co-workers in the laboratory of State of Connecticut, Department of Agriculture, Bureau of Aquaculture, John Karolus, Stacey Spear and Joe DeCrescenzo for the enormous amount of work they have given for the oyster health program. Thanks are due to Shannon Kelly for revising this manuscript.

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SATELLITE MONITORING OF PARTICULATE MATTER IN LONG ISLAND SOUND

Szekielda, K-H., Department of Geography, Hunter College, City University of New York, 695 Park Avenue, New York, NY

ABSTRACT

Research has been undertaken with the Sea-viewing Wide Field of view Sensor (SeaWiFS) to study the patterns and temporal changes of particulate matter in Long Island Sound from satellite altitudes. Spectral properties of plankton organisms and inorganic suspended matter have been taken into account to interpret the satellite data. A sequence of observations showed the fast changes in particle concentrations as a function of seasonal blooming in the offshore region and constant high particle load throughout the year in the Long Island Sound.

INTRODUCTION

The Long Island Sound is a large embayment open at both ends and is connected to the Atlantic Ocean at the northern part by the Race and at the western part by the East River and Hudson River Estuary. It differs from most estuaries in that it receives most of its freshwater input along its length rather than at its head (e.g., Housatonic River, Connecticut River, Thames River). The average daily freshwater delivered to the Sound is approximately 27,200 ft3/sec with loadings for total nitrogen from all sources of about 339,688 lb/day.

Long Island Sound interacts with the Hudson Estuary through the East River at an average of 385 cubic meters/second (Hudson River Foundation 1996), whereby a large amount of organic material, based on chlorophyll measurements, is transported out of the western basin of the Sound.

The fluxes from Long Island Sound, upwelling along the New Jersey coast and along the southern coast of Long Island, as well as the nutrient load from waste water treatment facilities (WTFs), are responsible for eutrophication, delivering an anthropogenic signal to the offshore region in the form of eutrophication. This has been documented with the Coastal Zone Color Scanner (CZCS) data for the Hudson River and tributaries draining into Long Island Sound. Satellite observations with the CZCS revealed eutrophication of the Hudson River Estuary and the Long Island Sound throughout the year and documented high concentrations of chlorophyll/particulate matter.

The geochemistry of Long Island Sound, in relation to nutrients and primary production, is strongly influenced by anthropogenic inputs through waters from WTFs, and consequently by nutrients loaded into the Sound's systems above naturally occurring levels. Such high nutrient levels dominate the budget of nutrient concentrations in the vicinity of New York. Clark et. al. (1996) showed that WTFs regulate the soluble reactive phosphate concentration in the Hudson River Estuary and that the atypical behavior in the estuary is due to the quasi-conservative behavior, which is based on relatively slow biological turnover rates and non-nutrient limited primary production.

The seasonal changes, ie., precipitation and resulting runoff, influence absolute concentrations. Taking the Hudson River as one river system to demonstrate the effect of eutrophication, some general conclusions can be drawn which may also be applicable for tributaries discharging into the Long Island Sound. Deck (1986) showed that heavy rain in March/April results in a conservative distribution of nitrate and silicate as a function of salinity with a maximum of phosphate, ammonia and nitrite concentrations at salinities indicative of anthropogenic input to relative low nutrient concentration in waters of the upper harbor. Chlorophyll observations for the Hudson River are sparse, although spectral data from Stross (1986) and Lonsdale et al. (1996) have been used to derive a very general distribution picture of the chlorophyll concentration. Keeping in mind that only a very limited number of actual measurements has been available, generally high concentrations of photosynthetically active

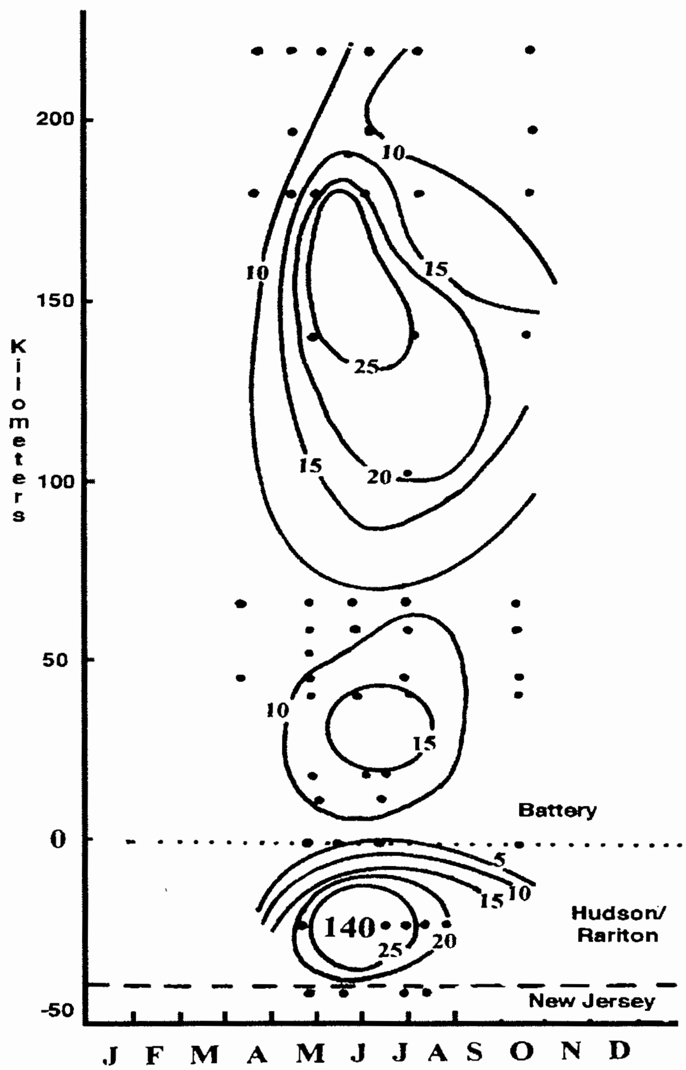

Figure 1. Generalized chlorophyll concentrations for the Hudson River and its estuary in a yearly cycle.

pigments in the upper Hudson River shows maxima at around 25 mg/l at the surface, with decreasing concentrations at around RKM 100 (Figure1). In the lower part of the Hudson River, low concentrations of chlorophyll have been identified. The data on chlorophyll are supported with actual cell counts of phytoplankton and its composition in relation to seasonal changes by Marshall (1988) for the Hudson River section between Poughkeepsie and Kingston.

The response of phytoplankton onto the high available nutrients discharge is primarily the result of sedimentation and clearing of the water column which allows deeper penetration of incidence radiance. As a consequence, higher chlorophyll concentrations are observed in those regions, where turbulence is low and where phytoplankton can take advantage of higher light intensities and high nutrient levels. In the case of the Hudson/Raritan estuary, this region is close to or out of the New York Harbor region where, as shown in Figure 1, chlorophyll concentrations may be in the neighborhood of 150 mg/l.

SATELLITE REMOTE SENSING OF PARTICULATE MATTER

Remote sensing has proven to be an important tool in oceanography and coastal research and monitoring. However, applications are limited to a certain degree as the interaction between parts of the electromagnetic spectrum; and the phenomena to be observed are in most cases restricted to the surface. Chlorophyll/ particulate matter in the Sound can be measured with data from the CZCS and more recently with the Sea viewing Wide Field of view Sensor (SeaWiFS), (Szekielda 1998). From the chlorophyll data obtained through conventional means and shown in Figure 1, it can be deduced that concentrations range over two orders of magnitude with values reaching close to 100 mg/l during the summer season in the vicinity of New York.

Blooming may occur within a few days during which chlorophyll concentrations may increase over orders of magnitude. It is not feasible to identify phytoplankton species from satellite observations, and the sensors cannot detect chlorophyll above 65 mg/l. To obtain this information, additional ship observations are required. On the other hand, using SeaWiFS data as a qualitative tool for detecting blooming, the unusual dynamics of phytoplankton in the waters in and around the Sound can be followed. Examples of pattern recognized in the anticipated "off season" for plankton blooming and/or high concentrations of particulate matter have been documented repeatedly. The Sound all year round showed the spectral response of plankton blooming and the impact of river discharge during the freshet in the satellite sensors.

The Coastal Zone Color Scanner (CZCS) and the Sea-viewing Wide Field of view Sensor (SeaWiFS) have been built to measure chlorophyll in the open ocean with an empirical algorithm that relates different spectral bands in the visible and near-infrared portion of the electromagnetic spectrum with ship measurements. In near coastal waters where concentrations of chlorophyll are high, and due to the presence of inorganic particulate matter, the algorithm is not valid anymore; and, at present, data have to be interpreted in qualitative terms (e.g., patch recognition and processes related to the dynamic field during tidal and current changes).

To demonstrate the role of inorganic particles on the spectral signatures, Figure 2 shows laboratory spectral measurements with algae cultures in connection with clay particles, both in varying concentrations. The chlorophyll concentrations and clay particle counts were chosen to demonstrate extreme conditions in near-coastal regions and to simulate bloom conditions in the presence of river discharge and erosion products. The relative reflectance demonstrates that at high concentrations of chlorophyll the typical absorption features recognizable at lower concentrations in the spectral region of the two chlorophyll absorption bands at around 0.44 and 0.66 mm are not recognized; rather, positive reflection is observed throughout the spectrum. This can be interpreted by additional reflectance of incident light at the phytoplankton organisms once they appear in high density. The reflectance spectrum 2 in Figure 2 has a concentration of about 64 mg/l and represents a concentration that can be frequently found in the Sound as well as in the nearshore region of Long Island and in the vicinity of the Hudson River discharge into New York Bight.

Laboratory Experiments on Spectral Response of Algae and Inorganic Particles			
NR	Chlorophyll *a* (mg m^{-3})	Phytoplankton (cells l^{-1})	Phytoplankton + clay (counts l^{-1})
1	0	0	0
2	64.18	$11.21x10^7$	$11.21x10^7$
3	118.76	$20.75x10^7$	$20.75x10^7$
4	118.31	$20.67x10^7$	$72.97x10^7$
5	117.44	$20.52x10^7$	$164.24x10^7$

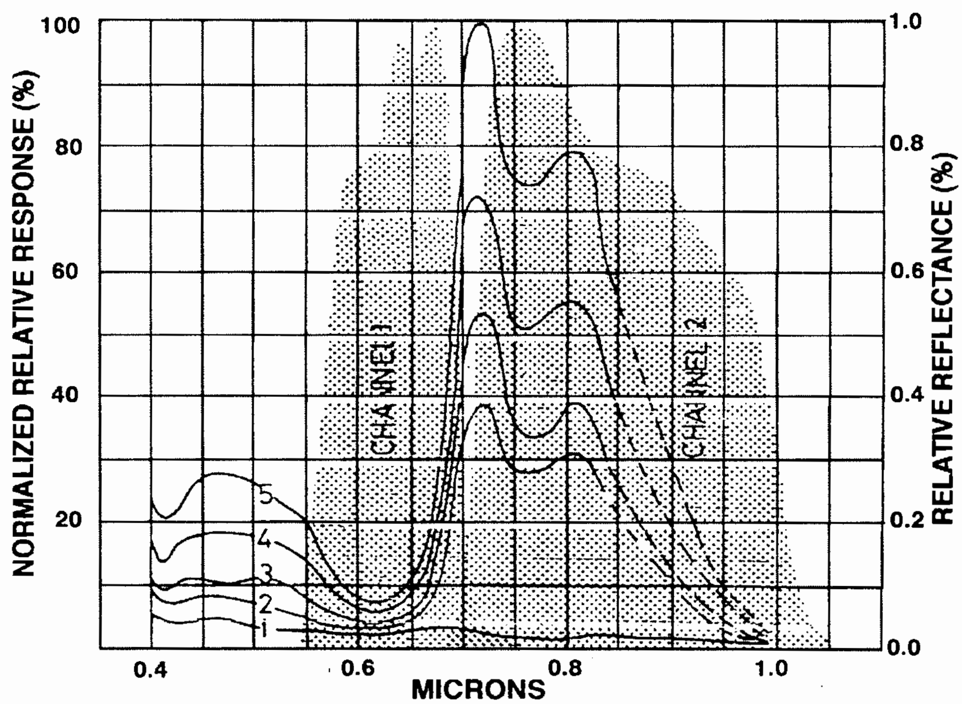

Figure 2. Laboratory experiments on spectral response of algae and inorganic particles. Modified after Szekielda et al. (1990). The shaded areas represent the normalized relative response of the AVHRR aboard NOAA satellites.

The main fact is, with increasing chlorophyll concentration, the reflectance increases; but no increase in absorption can be observed. If inorganic particles in the form of clay are added and the concentration of chlorophyll is kept constant, it can be recognized that the spectral response increases significantly, which demonstrates the problem one has when working with satellite data obtained through SeaWiFS or CZCS over coastal waters. An alternative for circumventing the interference of inorganic suspended matter is the use of other satellite systems such as the Advanced High Resolution Radiometer (AVHRR), which in the presence of high concentrations of suspended matter can bring useful information. For reference, the two normalized relative response curves are shown for two channels which have been used for identifying suspended matter and its patchiness in the Changjiang and the Amazon effluents as well as in the North Sea (Szekielda et al. 1988; Szekielda et al. 1990 and Szekielda et al. 1991).

The experimental work demonstrates that in the nearshore area satellite derived chlorophyll concentrations are overestimated. Typically, the Sound may reach 10 mg/l in its center basin; but in the nearshore location, where additional nutrient supply is injected from WWTFs, concentrations may go up to 80 mg/l (Figure 3). Nearshore harbors have been studied showing large algae blooms, defined as chlorophyll concentrations greater than 20 mg/l occur in all the harbors (S. Yergeau, personnel communication).

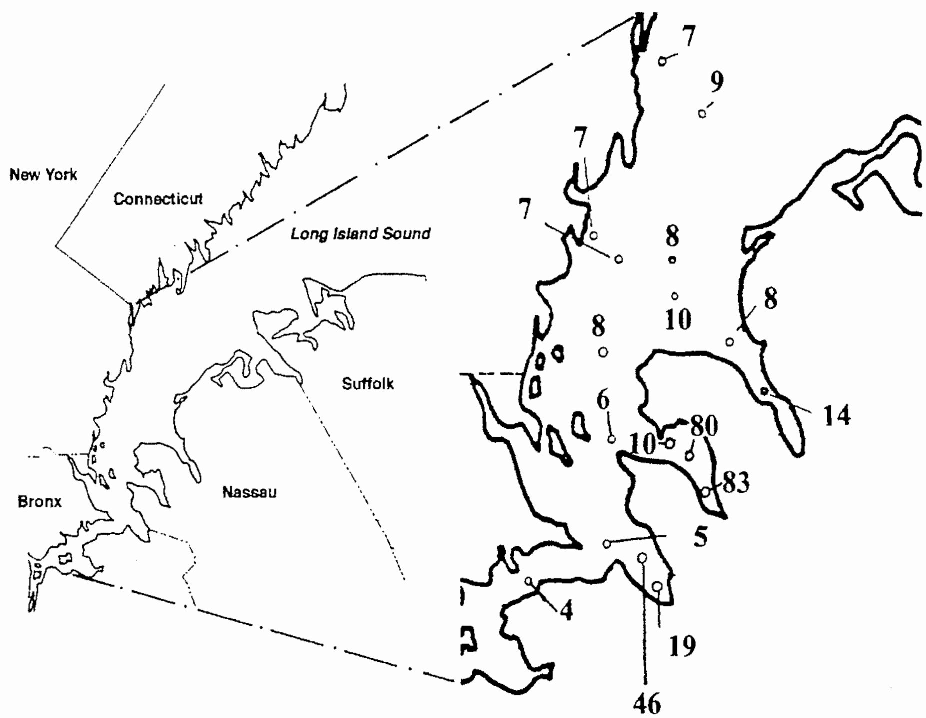

Figure 3. Chlorophyll concentration (mg/l) in the western basin of Long Island Sound in August 1997. Modified from Interstate Sanitation Commission Report.

Comparisons between selected targets with the same amount of picture elements for the offshore region and the Long Island Sound give a good comparison of the statistics related to plankton blooming and the distribution of chlorophyll/suspended inorganic material. Whereas the offshore bloom has a mean of about 6 mg/l, the Sound has a mean of about 32 mg/l. The latter is a value that is certainly an overestimate due to the presence of suspended inorganic material.

OBSERVATIONS

Figure 4 gives insight into the distribution pattern on 6 October 1997 of suspended material with plume structure off Long Island, in the vicinity of New York Harbor and in the upwelling regime off Cape Cod. The Long Island Sound compared to the southshore of Long

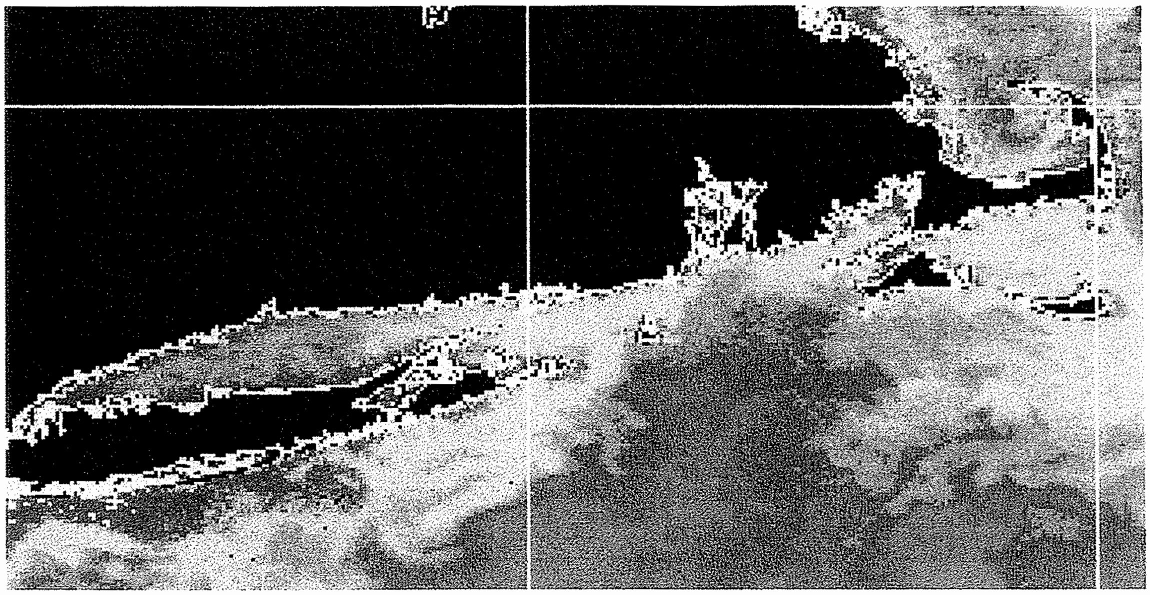

Figure 4. Chlorophyll concentrations from SeaWiFS on 6 October 1997 (Highest to lowest concentration: orange > yellow > green > blue).

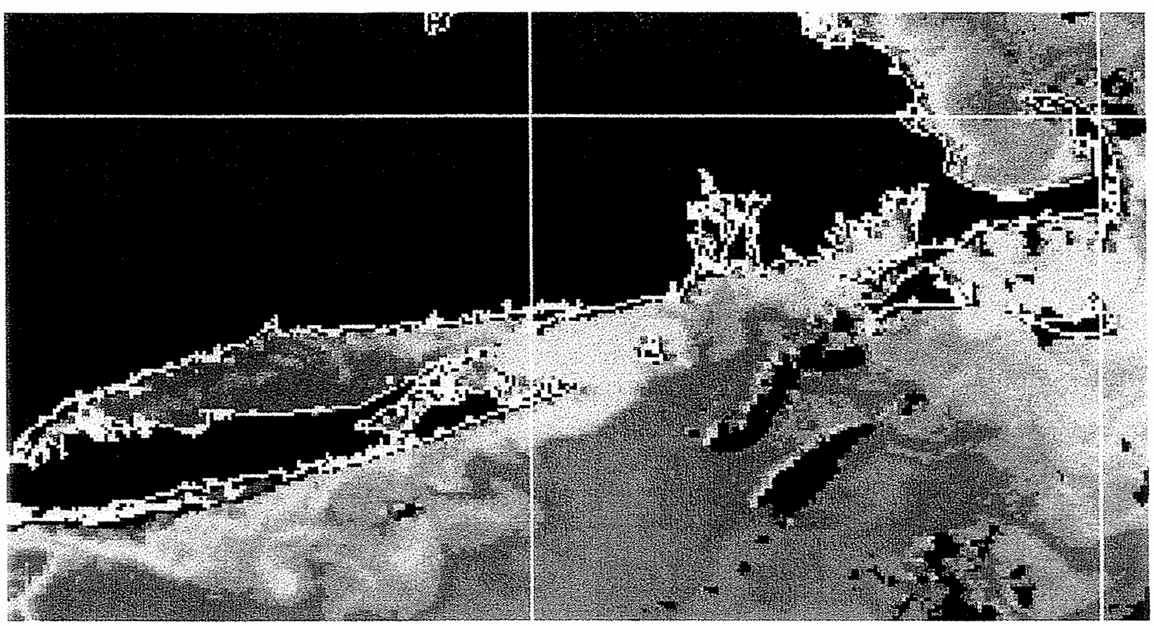

Figure 5. Chlorophyll concentrations from SeaWiFS on 10 October 1997 (see Figure 4 legend).

Island has relatively lower particle concentrations. Four days later (Figure 5), SeaWiFS indicates for the Sound increased concentrations of chlorophyll/particles, whereas the upwelling regime close to Cape Cod did not reveal any large-scale changes. This and the absence of river discharge are indications that the observed changes may be due to the effect of eutrophication. Temporal structural changes in the distribution patterns can be estimated with data sets taken at different time intervals. Figures 6 and 7 show that over the interval of one day only minor changes occur

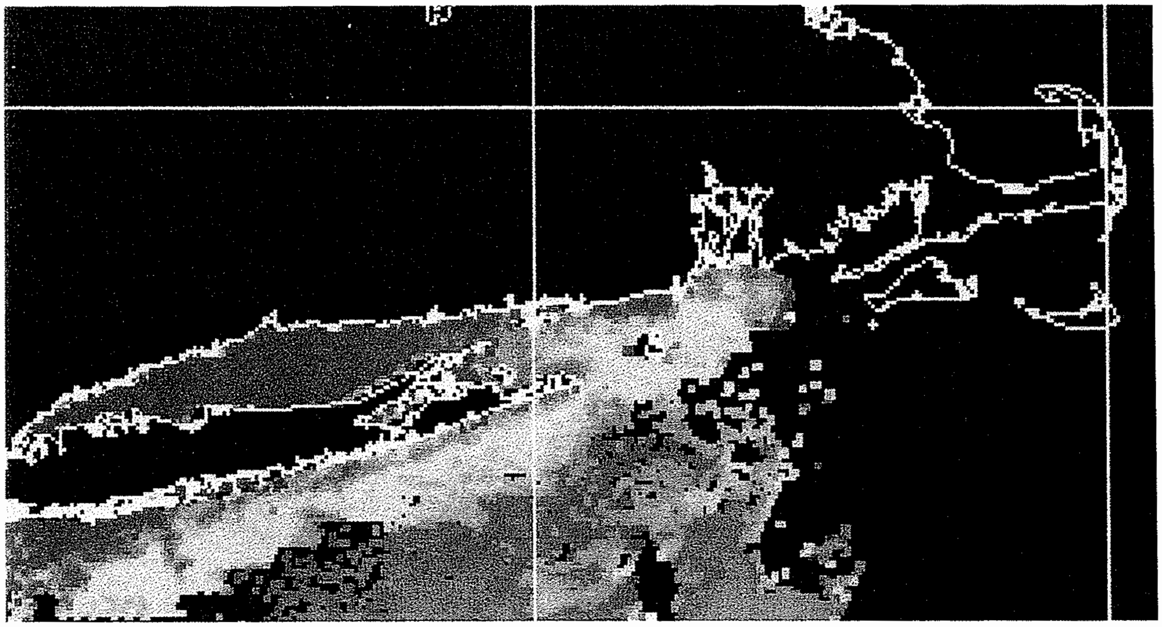

Figure 6. Chlorophyll concentrations from SeaWiFS on 20 October 1997 (see Figure 4 legend).

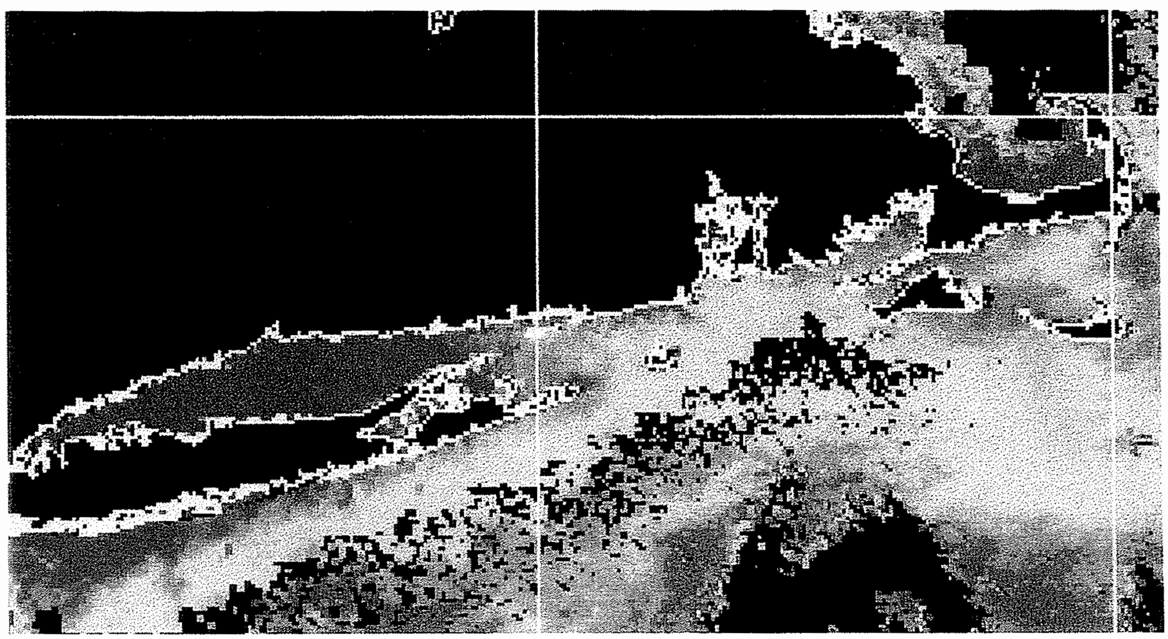

Figure 7. Chlorophyll concentrations from SeaWiFS on 21 October 1997 (see Figure 4 legend).

in the distribution of plankton and or particles. However, over longer time scales large area changes were observed as demonstrated in data in Figures 8 and 9. Taking into consideration the observations from the CZCS in previous years, plankton blooms may develop and break down within a few days.

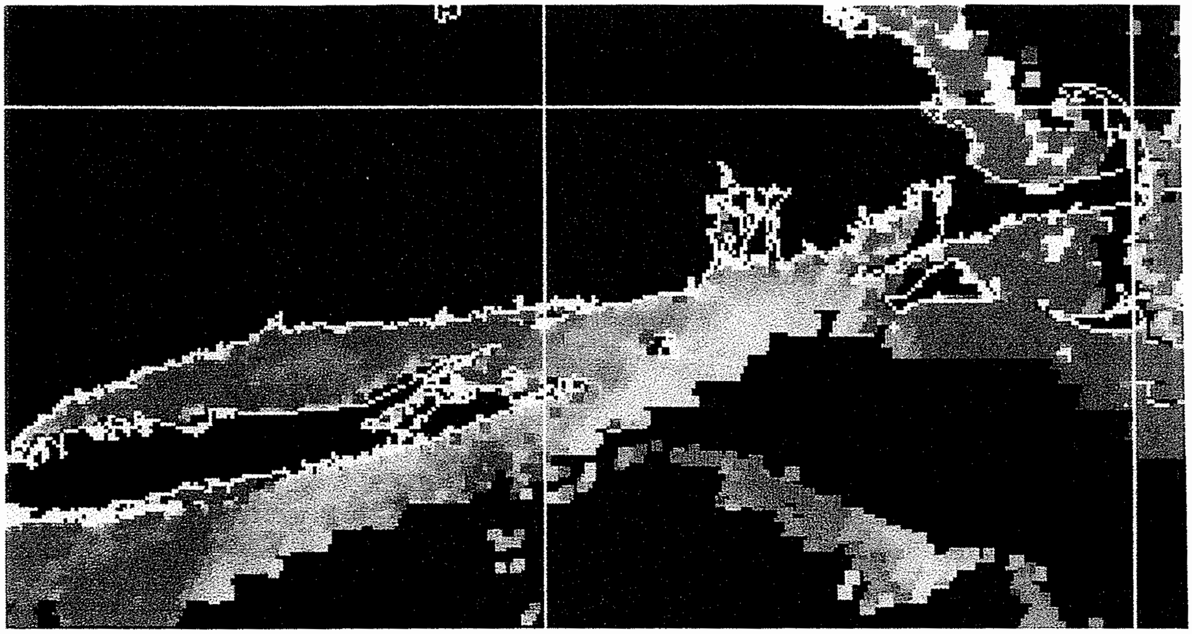

Figure 8. Chlorophyll concentrations from SeaWiFS on 12 January 1998 (see Figure 4 legend).

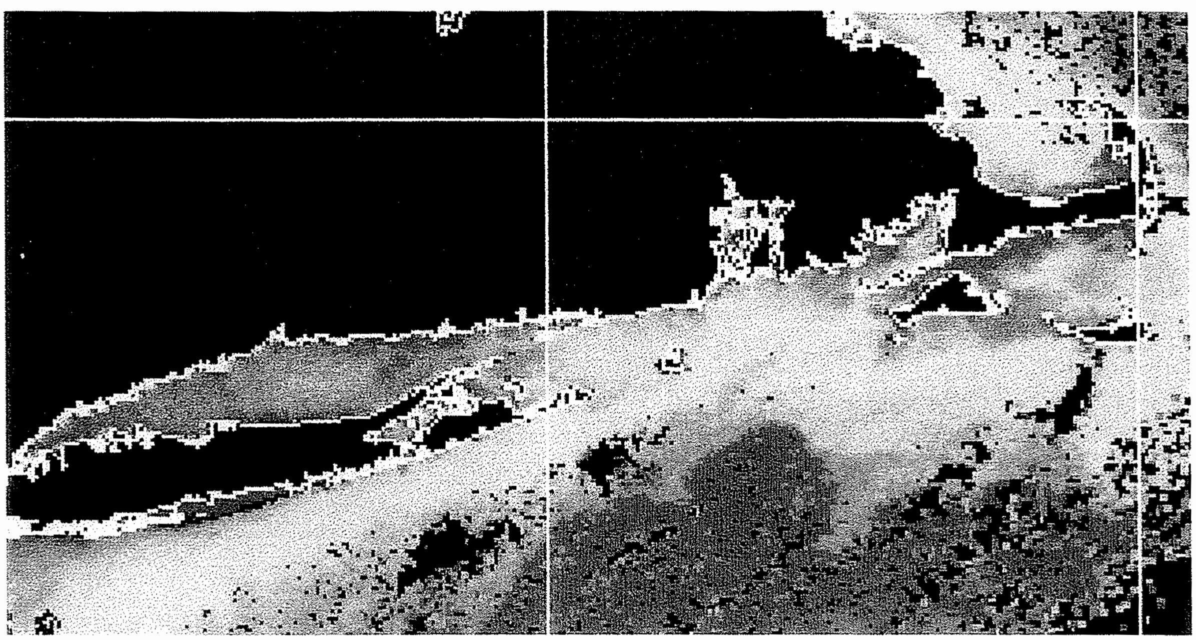

Figure 9. Chlorophyll concentrations from SeaWiFS on 9 February 1998 (see Figure 4 legend).

DISCUSSION AND CONCLUSIONS

Several conclusions derived from the satellite studies are issues which have high relevance to the environmental monitoring of the near coastal region around Long Island Sound and vicinity: plankton blooming and particulate material concentration vary considerably and appear within a very short time frame and may cover extensive areas. These events have also been successfully documented with repetitive satellite coverage of Long Island Sound.

At present, ship observations can resolve several parameters at the surface, at best in a quasi-synoptic way over a distance of 10-100 kilometers within a few hours. However, the time and space scale of particulate matter/phytoplankton as evidenced in Long Island Sound shows that the observed patchiness cannot be resolved according to their size and fluctuations in concentration.

In addition, advection and diffusion, which are responsible for the mixing of the outflowing effluent, are on a different time and spatial scale than primary producers. In response to changes in wind stress, turbulence and currents may also act on water and may modify the distribution pattern of particulate material in the vertical and horizontal directions.

The residence time of larger particles carried through the effluent is low due to their fast settlement, whereas smaller particles may be longer in suspension. Eventually, close to the boundary between the effluent and the adjacent ocean water, settling of particles clear the water column to a high degree which exposes lower water levels to higher illumination. In consequence, in an estuary and within the effluent of a river, the relationship between plankton and the dilution process is complimented by variable light conditions, sedimentation of particles and recycling of nutrients from WTFs. From data on relationship between patch size and lifespan (Szekielda and McGinnis 1991), it can be concluded that in a region with additional nutrient supply a certain patch size is required in order to develop the eutrophication in Long Island Sound . Within the effluent, it can be postulated that maximum concentrations of plankton should be observed at the outer part of the plume. If one considers the western basin as one outflow regime of the estuary, this would explain the high concentrations of chlorophyll observed by ship measurements. However, under extreme meteorological conditions during periods of low wind stress patches may be created based on low turbulence alone exposing the upper layer to high nutrient supply and sufficient light intensity.

Appearance of plankton patches and blooming and inorganic particulate matter are linked to each other not only through nutrients and light intensity but also to resuspension of sedimented particulate matter during the tidal cycle. The data from SeaWiFS indicated that the settling of particulate matter can be within a very short time frame during slack tide. Data from measurements in the German River Elbe confirmed satellite observations that the water column may clear or reload very quickly with particulate matter as a function of current speed (Figure 10). These measurements show that with decreasing current velocity the total suspended material decreases from about 200 mg/l to approximately one-fourth during slack tide. With the onset of the tidal current, the sedimented material is resuspended. Such mechanisms of resuspension and rapid settling of particulate matter could be the mechanism of explaining the lower reflectance of the Sound during slack tide.

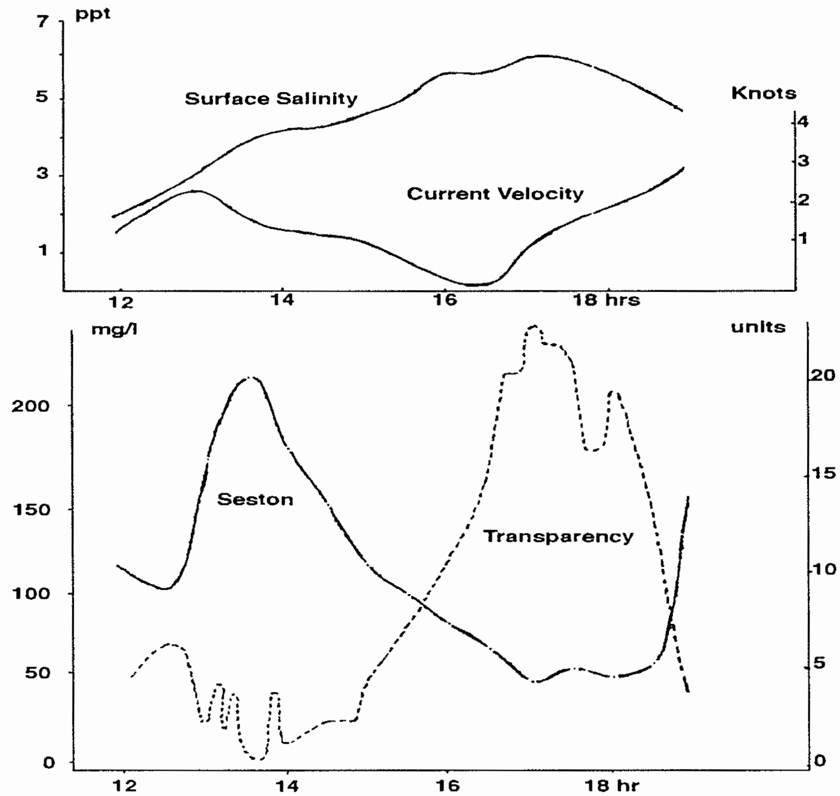

Figure 10. Relationship between tidal changes, suspended matter, salinity, current velocity and water transparency.

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DISTRIBUTIONAL SURVEYS OF EELGRASS (*ZOSTERA MARINA* L.) AT TWO LOCATIONS IN EASTERN LONG ISLAND SOUND FROM 1974-1997

Vozarik, J.M., M. Keser and J.T. Swenarton, Northeast Utilities Environmental Laboratory, PO Box 128, Waterford, CT 06385

ABSTRACT

The distribution of eelgrass (*Zostera marina* L.) in two coastal embayments in eastern Long Island Sound was surveyed periodically from 1985 to 1997, following local and regional (New Hampshire to the Chesapeake Bay) reports of declines in population. The Niantic River population fluctuated widely in distribution and density. For example, large declines occurred from 1987-1988, followed by a recovery from 1988-1989. The population in Jordan Cove was more stable. When compared to the high bottom coverage of eelgrass in the Niantic River noted in the late 1960s to early 1970s when areas were cleared of eelgrass with dynamite in an attempt to increase water flow, the present population is much reduced. However, it is important to note that this decline was not as drastic as the one that occurred in the 1930s, when most of the eelgrass disappeared from much of its geographical range, including our study area. The cause for that earlier devastation was never conclusively determined, nor are we certain of the reasons for the recent eelgrass declines. Pathogenic *Labyrinthula* spp. were present but not found in all plants. Water quality has also improved during our study of the area due to the expansion of the municipal sewage system to coastal residential areas. This may facilitate recovery of eelgrass in the Niantic River.

INTRODUCTION

Eelgrass (*Zostera marina* L.) is a marine angiosperm (flowering plant) found along temperate and sub-arctic coasts in the Northern Hemisphere (Setchell 1935). On the east coast of North America *Zostera* occurs from North Carolina to Greenland (Setchell 1929; Thayer et al. 1984). During the 1930s, widespread die-off of eelgrass was observed on both sides of the North Atlantic. This decline was attributed to the "wasting disease." Some researchers have associated wasting disease with a number of physical factors such as solar activity (Tutin 1938), precipitation (Martin 1954), salinity (Young 1937), or temperature (Rasmussen 1973). Others pointed to pathogenic microorganisms: bacteria (Lami 1935), fungi (Mounce and Diehl 1934), or the net slime mold *Labyrinthula* spp. (Renn 1934; Short et al. 1987; Short 1988). Regardless of the exact cause or causes, there were concerns about ecological implications associated with the eelgrass decline as it coincided with widespread declines in the abundance of many species of finfish, shellfish, lobsters, crabs and waterfowl (Stauffer 1937; Dexter 1947; Milne and Milne 1951; den Hartog 1977). Recovery of eelgrass and the associated fisheries throughout its range, has been gradual (Cottam 1945; Rasmussen 1977).

Historical data for eelgrass abundance in the vicinity of Millstone Point, Waterford, CT are mostly anecdotal. Marshall (1960) recalled that prior to the wasting disease, the Niantic River was "so choked by the blades (of eelgrass) that neither rowing nor outboard travel was possible in the shallows at low tide". Local eelgrass populations were almost completely destroyed by 1933 (Marshall 1960, 1994). By the late 1950s, considerable recovery had occurred, and by the early 1970s, eelgrass beds in Jordan Cove and the Niantic River were so extensive that explosives were used to clear some areas in an attempt to improve water circulation (Klotz and Knight 1973).

Due to the close proximity of eelgrass populations to the Millstone Nuclear Power Station (MNPS) and public concern with increased eelgrass accumulating on area beaches, Northeast Utilities initiated studies that included mapping of eelgrass beds in Jordan Cove (Klotz and Knight 1973; Knight and Lawton 1974). Declines of *Zostera* populations were again reported in the 1980s along the East Coast of North America (Orth and Moore 1983; Dexter 1985; Short 1988) including areas near MNPS (NUSCO 1989). As a result, a number of additional mapping studies in the Niantic River and Jordan Cove were conducted to document any changes in the distribution of those populations.

STUDY AREA

Jordan Cove and the Niantic River are located on the southeastern coast of Connecticut in the town of Waterford, approximately 8 km west of New London, between the mouths of the Connecticut and Thames Rivers (Figure 1). Eastern Long Island Sound water temperature follows a predictable seasonal cycle, with minimum daily average temperatures of about 0-2°C in early February, and maxima of 20-21°C in late August. Strong tidal currents minimize thermal stratification of the water column, and diurnal variability is typically on the order of 2-4°C.

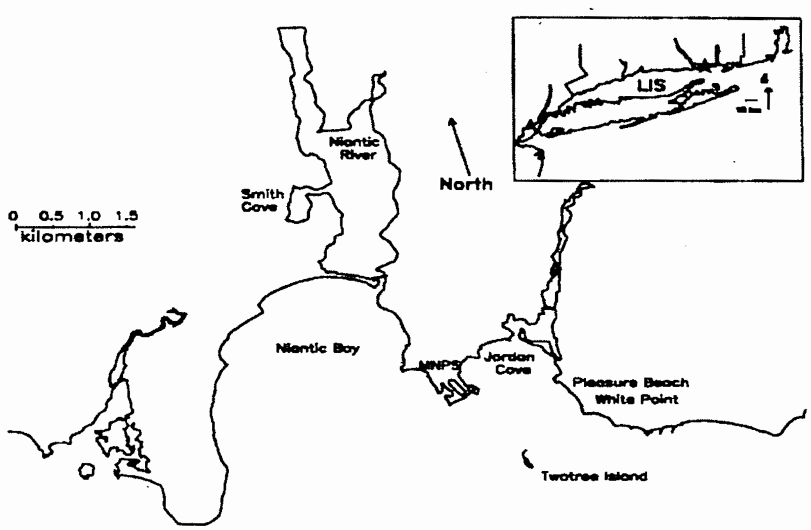

Figure 1. Map of the study area showing the location of Millstone Nuclear Power Station (MNPS) and eelgrass survey areas, Jordan Cove and Niantic River.

Jordan Cove is a relatively shallow (<6 m) embayment, on the east side of Millstone Point (Figure 1). Eelgrass habitats in Jordan Cove are exposed to elevated water temperatures resulting from solar warming of nearby sandflats, particularly during low tides in summer; this phenomenon can produce maximum afternoon water temperatures 5-7°C above nighttime lows and daily mean temperatures 2-3°C above 'ambient' conditions.

The Niantic River estuary is approximately 3 km northwest of Jordan Cove (Figure 1). The estuary is approximately 5 km long with a maximum width at the lower basin of almost 1 km, and depths up to 6 m. The inlet to the estuary, about 35 m wide and 3 m deep, is the only entrance and exit for tidal exchange. The southern portion of the lower basin consists primarily of sand flats that are partially exposed at low tides. A navigation channel traverses the sand flats to the river mouth. This area is occasionally dredged to allow boats access to the estuary. Freshwater input is minimal (Marshall 1960) and salinities are similar to those of LIS (mean of 30‰). However, during periods of heavy rain, salinities can be as low as 20‰. In the Niantic River estuary water temperatures are affected by insolation and tidal incursion.

MATERIALS AND METHODS

Qualitative mapping surveys of the Niantic River eelgrass beds were conducted in 1987-89, 1993, 1995 and 1997 and of Jordan Cove beds in 1974, 1985, 1993 and 1997; the 1974 study was conducted by Knight and Lawton (1974) (Figure 2; Figure 3). The 1985 survey of Jordan Cove was conducted by establishing shore-based benchmarks on the western edge of Jordan Cove. From these, a 100 m line marked at 5-m intervals was laid out to the eastern shoreline. Two SCUBA divers moved along the line, each placing a 0.25 x 0.25-m quadrat at the 5 m marks. A transect ended when no plants were recorded at four consecutive 5-m intervals. Visual observations were continued along the transect to mark smaller patches that occurred between the western and eastern shoreline. Subsequent surveys in Jordan Cove in 1993 and 1997, and all surveys in the Niantic River were done at low tide by measuring the length and width of the *Zostera* beds with marked line. The relative density (% cover) was subjectively estimated on the basis of the amount of sediment visible.

RESULTS

Surveys of eelgrass beds in the Niantic River showed that during 1987, major eelgrass beds were present in the lower river along the north and south sides of the navigation channel (Figure 2a). The southern bed was about 420 m long by 65-115 m wide; coverage ranged from 10 to 20% at the western end and from 50 to 80% at the eastern end. The bed on the north side of the lower channel was about 280 x 50 m; coverage ranged from 30 to 80%. Another area with extensive eelgrass cover in the late 1980s was along the west side of the navigation channel; this bed was about 500 m long by 25 m wide, and eelgrass cover was 10-20%.

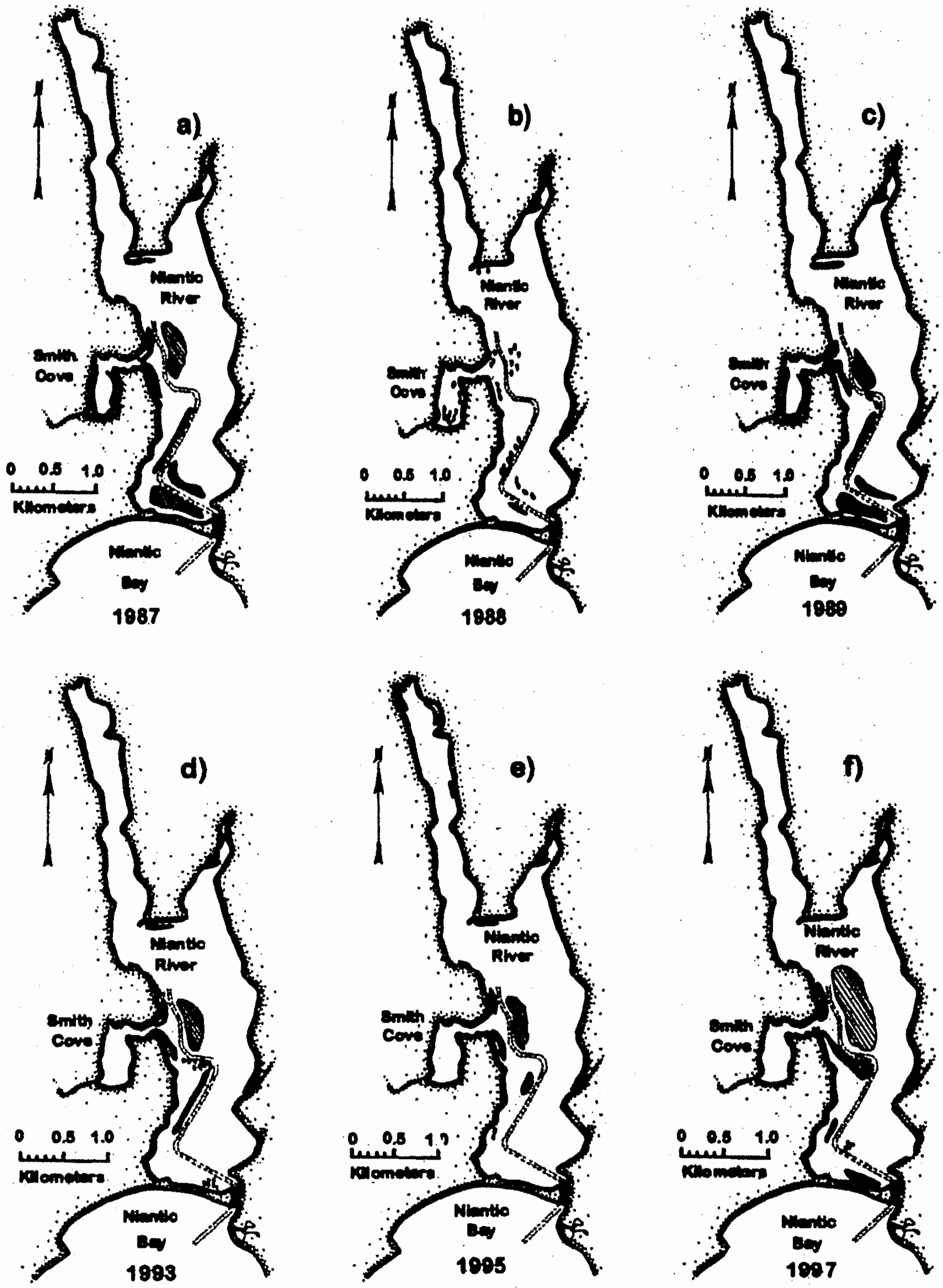

Figure 2. Maps of eelgrass (*Zostera marina*) distributions (hatched areas) in the Niantic River, CT, during surveys conducted in 1987, 1988, 1989, 1993, 1995 and 1997.

Eelgrass was also common on the east side of the navigation channel opposite to the entrance to Smith Cove, and on the north and south sides of the channel branch leading into the cove (Figure 2a). Maximum dimensions of the bed east of the channel were about 300 x 120 m; eelgrass coverage was 50-60%. On the west side of the channel, the northern segment was about 80 x 20 m and the southern segment was about 320 x 25 m; coverage in both areas was 80-90%. In Smith Cove, eelgrass was present in mixed stands with widgeon grass, *Ruppia maritima*. Plants of both species were scattered and highly epiphytized. Eelgrass coverage along the shoreline of the Cove in 1987 was estimated at 30-40%, and 10-20% in the center. Eelgrass was also found at the north end of the lower river basin between the upper arms. This dense bed (75-80% coverage) was about 200 m long, but owing to a sharp increase in depth to the south, was only 25 m wide.

In 1988 (Figure 2b), eelgrass was generally found in the same areas as in the previous year, but plants were scattered and bottom coverage was much lower. By 1989 (Figure 2c), abundance and distribution were again very similar to those in 1987. The 1993 survey (Figure 2d) showed the disappearance of beds in the lower river and Smith Cove. By 1995 (Figure 2e), the size of beds to the west of and adjacent to the navigation channel were reduced by almost 70%; and no plants were observed in the lower Niantic River. Eelgrass showed increases in 1997 (Figure 2f) in some of the beds in the Niantic River compared to 1993 and 1995, and beds near the entrance to Smith Cove and the river channel had expanded. Eelgrass was absent from Smith Cove in all three surveys conducted in the 1990s. At the southern end of the river, eelgrass appeared as patches along the southern and western edges of the navigation channel; and the previously extensive bed north of the navigation channel in the lower river was absent.

Eelgrass surveys in Jordan Cove (Figure 3) showed that these beds were more stable than those in the Niantic River with respect to location, but also varied in terms of plant density and areal coverage. In 1985 (Figure 3b), eelgrass beds formed an almost continuous band along the western and northwestern shore of Jordan Cove, representing a slight increase in length and considerable increase in width relative to the beds reported by Knight and Lawton (1974) (Figure 3a). Eelgrass beds along the western shoreline of Jordan Cove showed decreases in size since the 1985 survey. The once continuous beds along the western to northern shore had become a series of smaller, discontinuous beds in the 1993 and 1997 surveys (Figure 3c-d).

In all surveys of Jordan Cove, the northeastern portion of the cove was characterized by small, disjunct patches of eelgrass. An eelgrass bed that was located in the inner Jordan Cove (north of the sandbar) in 1974 did not exist in later surveys (1985, 1993, 1997). No eelgrass was reported at White Point in 1974 (Knight and Lawton 1974), but may not have been part of the survey area.

DISCUSSION

Periodic distribution surveys showed that *Zostera* populations in the Niantic River and Jordan Cove exhibited wide fluctuations in relatively short periods of time in both size and density of *Zostera* beds. This was particularly evident in the Niantic River, with large decreases from 1987 to 1988, and increases from 1998 to 1999. Similar fluctuations were reported by Marshall (1960) during the recovery of this population following widespread die-off. These patterns could not be explained by changes in physical parameters such as temperature and salinity, which are relatively stable from year to year.

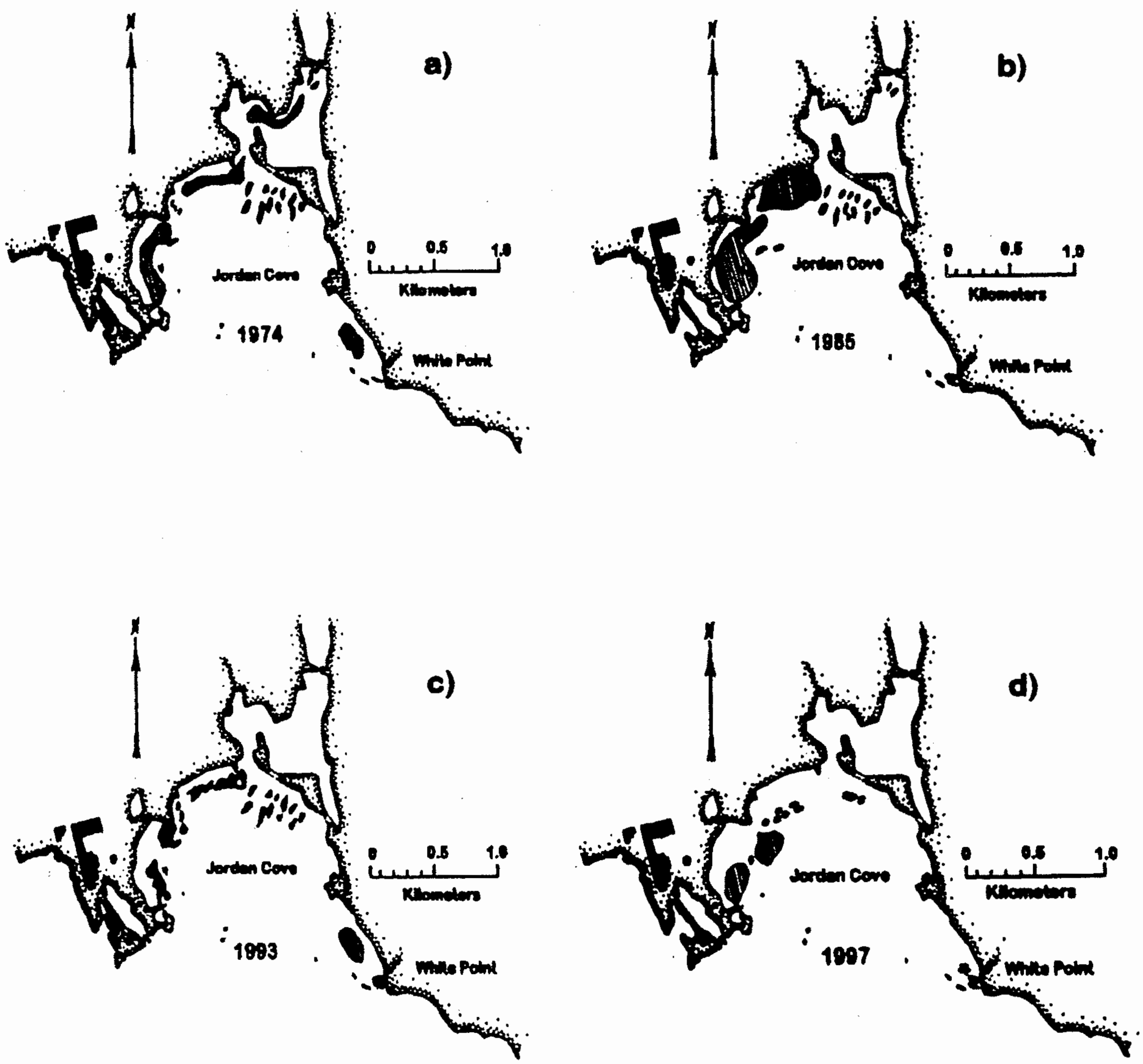

Figure 3. Maps of eelgrass (*Zostera marina*) distributions (hatched areas) in Jordan Cove, Waterford, CT, during surveys conducted in 1974, 1985, 1993 and 1997.

Short (1988) identified *Zostera* plants infested with *Labyrinthula* spp. in the Niantic River and speculated that the decline in these plants was due to both infection and degradation of water quality in the River. Our surveys indicate that the Niantic River *Zostera* population became most sparse in 1988. Some recovery was observed in subsequent years, but population distribution and density never reached levels observed in the mid 1970s (Klotz and Knight 1973). The Jordan Cove population was more stable. It is important to note that the Town of Waterford, Connecticut, has been expanding its municipal sewage system since 1993 to include areas along both the Niantic River and Jordan Cove, likely improving water quality in both habitats.

In summary, eelgrass decline observed in this study in the 1980s, and by other researchers from New Hampshire to Chesapeake Bay, was not as catastrophic as that observed in the 1930s when most populations disappeared in a 2-3 year time period. This loss of eelgrass in the 1930s corresponded with large declines in a number of commercial fisheries. Similarly, more recent eelgrass losses reported here correspond with marked declines of a number of commerically important species in the New England area. Particularly noteworthy for this study are substantial reductions in winter flounder, (*Pseudopleuronectes americanus*) in Long Island Sound and the bay scallop (*Argopecten irradians*) in the Niantic River. It remains to be seen whether recent eelgrass declines are related to similar factors responsible for widespread disappearance observed in the 1930. Additional research is needed to determine what effect eelgrass reductions is having on commercial fisheries and the local coastal marine ecosystem.

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WARM SEASON ALGAL BLOOMS IN FOUR LONG ISLAND SOUND HARBORS

Yergeau, S., Save the Sound, Inc., 185 Magee Avenue, Stamford, CT 06902

ABSTRACT

Near-shore water quality data was gathered in four harbors in western Long Island Sound to analyze the seasonal algal blooms and their relation to hypoxia and land-use. Measurements of dissolved oxygen, turbidity, chlorophyll *a* and phytoplankton taxa were taken weekly (bi-weekly for phytoplankton taxa) between May and October 1997, at 22 stations in the harbors. For each harbor, water quality impairment was observed at the stations closest to their major tributaries. Echo Bay had the overall best water quality of the four harbors tested. Dissolved oxygen in Echo bay averaged 9.56 $mg \cdot l^{-1}$ for the season. Milton Harbor averaged 7.51 $mg \cdot l^{-1}$, Cos Cob Harbor averaged 6.36 $mg \cdot l^{-1}$, and Stamford Harbor averaged 6.59 $mg \cdot l^{-1}$ for the season. Large algal blooms (defined as chlorophyll *a* concentrations greater than 20 $\mu g \cdot l^{-1}$) occurred in all the harbors studied at least once during the testing season. Echo Bay had an average chlorophyll *a* concentration of 11.4 $\mu g \cdot l^{-1}$ with a range of 1.1 $\mu g \cdot l^{-1}$ to 26.4 $\mu g \cdot l^{-1}$. Milton Harbor had an average chlorophyll concentration of 10.2 $\mu g \cdot l^{-1}$ and a range of 0.2 $\mu g \cdot l^{-1}$ to 23.8 $\mu g \cdot l^{-1}$. Cos Cob Harbor had an average chlorophyll *a* concentration of 7.0 $\mu g \cdot l^{-1}$ and a range of 0.9 $\mu g \cdot l^{-1}$ to 41.9 $\mu g \cdot l^{-1}$. Stamford Harbor had an average of 7.8 $\mu g \cdot l^{-1}$ and a range of 0.3 $\mu g \cdot l^{-1}$ to 43.7 $\mu g \cdot l^{-1}$. Storm-water runoff due to development in the surrounding watersheds was attributed to the degraded water quality and algal bloom formation at the study sites.

INTRODUCTION

Long Island Sound undergoes seasonal hypoxic events where levels of dissolved oxygen (DO) drop below 3.0 $mg \cdot l^{-1}$ (Yergeau and Ayala 1998). Oxygen enters the water from the churning action of the tides and wind and from photosynthesis of marine plants. In the marine environment, nitrogen, in the form of nitrates and nitrites, generally acts as the limiting nutrient for algal growth. As with many estuaries, there is an overabundance of nitrogen in Long Island Sound. Nitrogen enters the Sound through many sources such as sewage treatment plants, leaking septic systems, storm-water runoff, and acid rain, all of which lead to algal blooms (Long Island Sound Study 1994). When the algae die and sink to the bottom, oxygen is consumed during their decomposition by naturally occurring bacteria.

This increased demand for oxygen is in addition to the normal oxygen use from the metabolic activities of animals and plants. Hypoxia (DO levels below 3.0 $mg \cdot l^{-1}$) can be most severe during the summer when stratification prevents highly oxygenated surface water from mixing with the poorly oxygenated bottom water. Severe hypoxic events have occurred in the Sound that have resulted in large finfish and shellfish kills (Long Island Sound Study 1994; Brosnan and Stubin 1992; Miller et al. 1992; Poucher et al. 1992). This period of low DO in the Sound has been identified by the Long Island Sound Study as the highest priority upon which

New York, Connecticut, and the U.S. Environmental Protection Agency are focusing their efforts and resources (Long Island Sound Study 1994).

To determine the extent of hypoxic events in the coastal areas of Long Island Sound, water quality monitoring efforts are underway using chemical indicators. The development of a biological indicator for monitoring estuarine water quality to supplement DO concentration measurements can aid in expanding management efforts in large coastal bodies of water such as Long Island Sound. The measure being investigated by Save the Sound, Inc., is a diversity index based on phytoplankton presence or absence (from now on referred to as the phytoplankton diversity index, or PDI). The PDI is incorporated into Save the Sound's water quality monitoring program to supplement data on DO, pH, Secchi disk depth, temperature, salinity, and chlorophyll *a* concentration, and to help analyze and report on water quality data collected on a weekly basis from harbors in Long Island Sound. Phytoplankton were chosen as the indicators of water quality since microalgae have short generation times and respond quickly to changing water quality conditions (Yergeau et al. 1997).

MATERIALS AND METHODS

NOTE: For the monitoring performed during this study, trained volunteer citizens were used to gather data. For safety reasons or for lack of volunteers, some sampling was performed irregularly or not at all for some parameters. The data have, therefore, been simplified by examining at averages of the main water quality components as a means to perform analysis.

Sampling Locations. Two Connecticut and two New York harbors were tested weekly from May 17 to October 11, 1997. These four harbors were selected because two have highly developed waterfronts (Echo Bay in New Rochelle, NY and Stamford Harbor in Stamford, CT) and two are less densely developed (Milton Harbor in Rye, NY and Cos Cob Harbor in Greenwich, CT). The amount of development per harbor was related to the water quality data collected to determine the impact that land-use has on water quality.

Echo Bay in New Rochelle, NY, is a wide harbor with industrial development surrounding the waterfront. On the western bank are residential areas and Hudson Park, a multiple use outdoor facility. On the northern shore is a sewage treatment plant that receives waste from different towns within Westchester County with the majority coming from New Rochelle. In the center of Echo Bay is Five Islands Park, a recreation complex with areas for boating, fishing and swimming. In the northeast section is the Mill Pond fed by the Premium River. This area is filled with tidal flats and marsh areas. The Mill Pond is separated from Echo Bay by a dam that creates a waterfall into the harbor. In the northwest reaches, Echo Bay is fed by the Stephenson Brook, a culverted stream that runs underneath the city. Five sampling stations were monitored during the testing season.

Milton Harbor in Rye, NY, is a narrow harbor with primarily residential, marina and mooring areas. The dredged channel has approximately 2.0 m of water between the head of navigation and Milton Point and averages 2.5 m deep beyond Milton Point. There is a large tidal flat area on the northeastern quadrant. The harbor is fed by Blind Brook, which originates at the Westchester Airport. The Marshlands Conservancy and a golf course are on the western bank of the harbor. Hen Island is a residential island accessible by boat only. Six sampling stations were monitored during the testing season.

Cos Cob Harbor in Greenwich, CT, is an extension of the Mianus River. The harbor is divided into the inner and outer areas by the Metro North Railroad Bridge. The inner harbor has mudflats on the east bank which occupy more than half the width of the harbor. On the west side, there are several marinas running the entire length of the harbor just to the south of the Mianus River Dam. In the outer portion, there are large homes and Riverside Yacht Club on the east side of the harbor. The west side consists of mostly undeveloped land and the remains of the Cos Cob Power Plant. Six sampling stations were monitored during the testing season.

Stamford Harbor in Stamford, CT, is primarily industrial in its surrounding land use; however, there are also residential and mooring areas dotting its shores. The harbor is divided into east and west branches, forming a 'Y'. The Woodland Cemetery and Kosciusko Park peninsula divides the two branches. Both the east and west branches have small tidal flat areas along the shores of the peninsula. On the western bank of the west branch is Southfield Park, a public beach adjacent to the Hoffman fuel dock. The east branch of the harbor is separated and protected from the mouth of Long Island Sound by a hurricane barrier. Along the entire western bank of the east branch are tidal mudflats. A condominium complex is located just behind a series of boat slips and docks. To the north is the Stamford Water Pollution Control Facility and its freshwater outlet at the dead end of the East Branch. Five sampling stations were monitored during the testing season.

Sampling Procedures. Surface (0.5 m below water surface) and bottom (total water depth minus 0.5 m) measurements included DO, salinity, temperature, and Secchi disk depth. Photosynthetic pigment chlorophyll *a* was sampled weekly at the surface (1.0 m below water surface) and analyzed as a measure of algal biomass. Algal diversity was sampled bi-weekly at the surface (1.0 m below water surface). All measurements were taken from boats in the morning starting approximately at 7:00 a.m. and running until approximately to 9:00 a.m. (Yergeau 1997).

In Echo Bay and Milton Harbor, DO (measured in $mg \bullet l^{-1}$) and water temperature (recorded as °C) were measured using a Yellow Springs Instrument (YSI) Model 58 Dissolved Oxygen meter with digital display, stirring unit, and Model 5700 field probe. DO readings were not adjusted for salinity in the field, but were corrected using calculations in a computer database. Salinity (reported as ppt) was measured using a YSI model 33 Salinity-Conductivity-Temperature (SCT) meter with analog display. Salinity measurements were compensated for changes in temperature manually by direct dial (Yergeau 1997).

The DO and SCT meters were calibrated before the beginning of each testing session. Each day the DO probe was also checked for air bubbles and other membrane problems, with the membrane changed if necessary. The salinity and DO probes were attached to a platform and readings were taken at the surface (probes at 0.5 m below the water's surface), at one meter intervals, and at the bottom (probes 0.5 m above the bottom).

In Cos Cob Harbor and Stamford Harbor, a Hydrolab H20 Multiprobe was used to measure DO, salinity, and water temperature. The probes were calibrated before the beginning of each testing session. Each day the DO probe was also checked for air bubbles and other membrane problems. The Hydrolab instrument automatically adjusts DO readings for salinity and temperature, and automatically adjusts salinity readings for temperature; therefore, no additional calculations were used to correct these values (Yergeau 1997).

Water clarity was measured by using a Secchi disk. Secchi disk depth was determined by taking the average of the water depth that the disk disappeared from sight and the depth at which it reappeared into view. Two volunteers performed this measurement and these two averages are then averaged for the final reading at that sampling site (Yergeau 1997).

Water samples for chlorophyll *a* analysis were collected using a Van Dorn sampler at 1.0 m below the water surface. The mixed water sample was filtered on the boat, through a round 47 mm diameter glass fiber filter, using a Nalgene filter manifold and hand pump. The volume of water filtered varied at each sampling run, but total volume filtered was recorded for each station. The ending point for the filtration was determined by comparing the color on the filter to a color chart after a dark green or dark brown color was reached on the filter paper (Yergeau 1997). The filter was placed in a foil packet, labeled, and stored on ice until it was transferred to the laboratory freezer. The filter apparatus was rinsed three times with distilled water after each use to prevent cross-contamination. Any samples held longer than three weeks in the laboratory were noted in the sample log book as such, since there may be possible degradation of the chlorophyll in those samples (Greenberg et al. 1992).

Chlorophyll *a* extraction and analysis was performed at Save the Sound's water quality laboratory by a member of the research staff or by trained technicians following Greenberg et al. (1992). Pigments were extracted after grinding the filter with a Teflon pestle in a 55.0 ml grinding tube with a 90% aqueous acetone solution. The samples were clarified in a centrifuge for 20 minutes, then analyzed using a Perkin Elmer Lambda 11 UV/VIS spectrometer with a 2.0 nm band width. After being clarified, the samples were resuspended and centrifuged two more times to insure 99.1% retrieval of chlorophyll *a* (Kuntz 1995).

Chlorophyll concentrations were corrected for pheophytin *a*, so that chlorophyll *a* values were not overestimated (Greenberg et al. 1992). Simple correlations between DO and chlorophyll *a* were calculated and a correlation coefficient (r) was compared to critical values to determine statistical significance at the 1% level (Rohlf and Sokal 1981).

Water samples for phytoplankton identification were collected using a Van Dorn sampler at 1.0 m below the surface of the water. The mixed water sample was poured into a 500 ml opaque brown bottle containing 15.0 ml of Lugol's solution to preserve the sample (Yergeau 1997). The 500 ml samples are measured in a graduated cylinder and filtered to concentrate the phytoplankton and to facilitate identification. An amount of filtered water equal to 1/100th of the original sample size was used to wash the sample off the filter and to create a 100x concentrated sample. Three slide views from this 100x concentrated sample are then observed, with phytoplankton identified, and indication of presence or absence noted. Samples were analyzed within three weeks time to ensure there was no degradation of the sample and to coincide with the chlorophyll *a* analysis (Yergeau 1997).

The PDI is based simply on the presence or absence of the taxa within these three subsamples. For each of the subsamples, the phytoplankton groups present were counted for both individual groups present and for the total number of groups observed. The PDI has three main components which are as follows:

Standardized total taxa (tts). The number of groups identified is only slightly related to temperature ($r = 0.28$) by itself, but in combination with other variables mentioned below becomes highly significant. The total taxa (tt) is the number of different groups found in three separate subsamples of the concentrated main sample (100x) summed. A correction was made to find a total taxa standardized for temperature (te), and therefore season, by the following formula $tts = tt - (te/6)$.

Degree of taxa overlap (q). The number of groups found in only one of any of the three subsamples (a) is noted, as is the number found in two subsamples (b), and the number found in all three subsamples (c). These were combined to form a factor, q, through the following equation $q = a + 2b + 2c$.

Redundancy of populations in each sample (z). The number of different taxa in the first subsample (t1), the second subsample (t2), and third subsample (t3) were summed and divided by the number of total taxa (tt) $z = (t1 + t2 + t3)/tt$.

The latter two measures of taxa overlap and redundancy (q and z), empirically derived, reflect the skewness, or imbalance in the distribution, of the sample population. Average PDI for the entire testing season for all stations in the harbor was reported. The PDI ranges between 0.0 and 20.0, with a corresponding increase in diversity as PDI increases. Verification of the PDI is currently being studied through mathematical statistics sampling theory (Yergeau et al. 1997).

RESULTS

Water quality in Echo Bay was rated fair during the testing season since four out of the five stations experienced violations of the water quality standard (5.0 $mg \cdot l^{-1}$ DO) at least once (Figure 1). Station 5 was the only site to stay above the water quality. Generally, DO levels were the lowest at the stations closest to Stephenson Brook (Station 1). DO dropped in mid-July and rebounded in late July to mid-August, but dropped below the standard in the middle of August and did not recover until mid-September (Figure 1). DO averaged 9.56 $mg \cdot l^{-1}$ for the whole harbor (Table 1).

Table 1. Water quality measurement seasonal averages by harbor.				
Harbor Name	Dissolved Oxygen Level	Secchi Disk Depth	Chlorophyll *a* Concentration	Phytoplankton Diversity Index
Echo Bay	9.56 mg•l^{-1}	1.52 m	11.54 μg•l^{-1}	12.15
Milton Harbor	7.51 mg•l^{-1}	0.93 m	10.20 μg•l^{-1}	11.87
Cos Cob Harbor	6.36 mg•l^{-1}	1.17 m	7.00 μg•l^{-1}	11.86
Stamford Harbor	6.59 mg•l^{-1}	1.18 m	7.80 μg•l^{-1}	12.05

Surface levels of chlorophyll *a* were similar at stations 2, 3, 4, and 5. Chlorophyll levels at these stations were relatively low, with most of the values between 10.0 μg•l^{-1} and 15.0 μg•l^{-1} (Figure 1), indicating development of small algal blooms (North Carolina State University Water Quality Group 1998). Chlorophyll *a* concentrations ranged from 1.1 μg•l^{-1} to 26.4 μg•l^{-1}. A large algal bloom, defined as having a chlorophyll concentration greater than 20 μg•l^{-1} (National Oceanic and Atmospheric Administration 1997), was detected at Station 4 when chlorophyll concentration was 26.4 μg•l^{-1} on July 19, 1997 (Figure 1). The average chlorophyll *a* value during the study was 11.4 μg•l^{-1} (Table 1). Chlorophyll *a* and DO were positively correlated with statistical significance at the 1% level (r= 0.24). PDI averaged 12.15 for the season (Table 1).

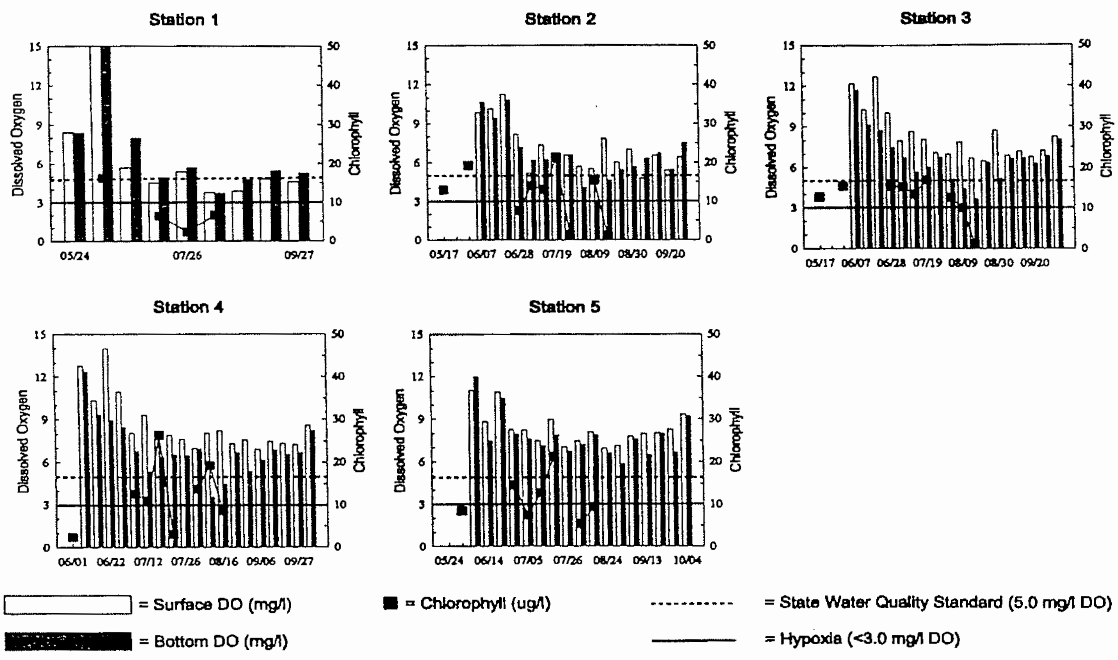

Figure 1. Echo Bay (New Rochelle, NY) dissolved oxygen and chlorophyll *a* by station.

The water quality in Milton Harbor was poor during the study with DO below the water quality standard (5.0 mg•l^{-1} DO) at every station at least once during the season (Figure 2). DO levels were lowest at the stations closest to Blind Brook. Stations 1 and 2 had the poorest water quality. The lowest oxygen level observed in this harbor (2.9 mg•l^{-1}) occurred at Station 1, the most inward site, on July 12, 1997. Stations 5 and 6, located towards the deeper portions of the harbor, had the highest DO levels (Figure 2). Oxygen levels were above or equal to the water quality standard throughout most of the study.

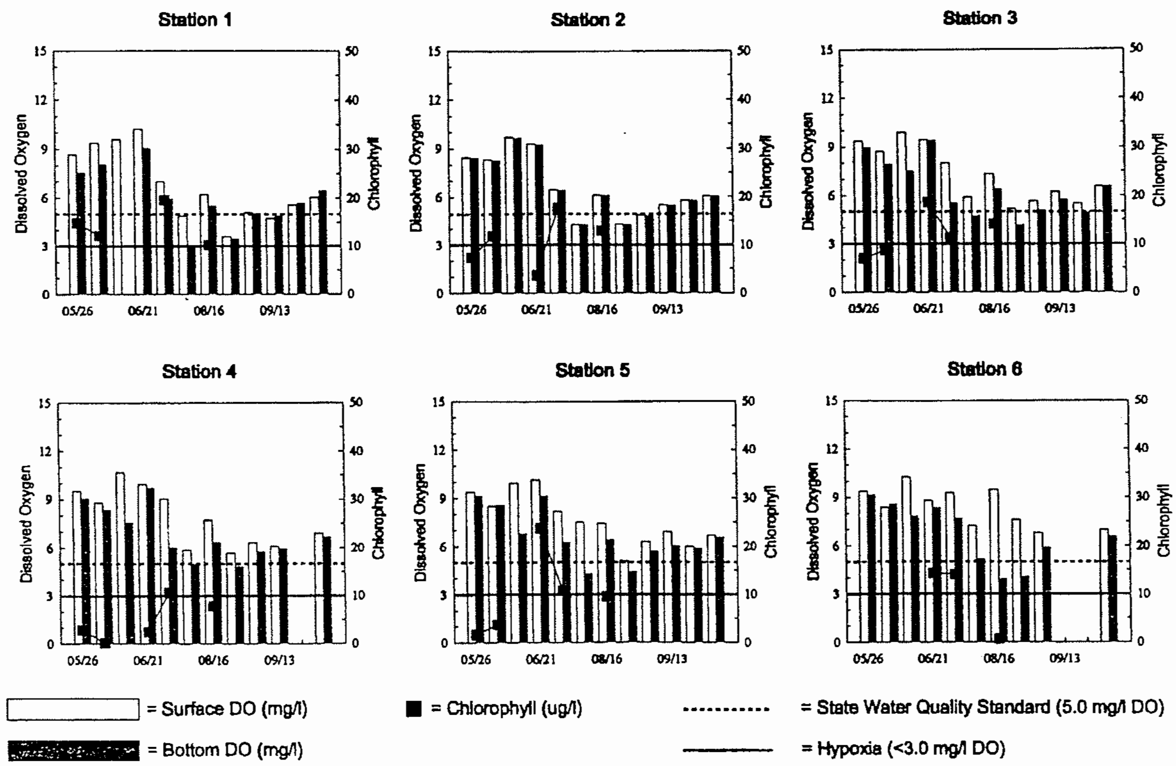

Figure 2. Milton Harbor (Rye, NY) dissolved oxygen and chlorophyll *a* by station.

Surface levels of chlorophyll were similar in stations 1, 2, 3 and 4 (Figure 2). Chlorophyll *a* values at these stations were moderate (10.0 μg•l^{-1} to 20.0 μg•l^{-1}), indicating some development of moderate algal blooms (North Carolina State University Water Quality Group 1998). Chlorophyll *a* concentrations ranged from a high of 23.8 μg•l^{-1} to a low of 0.2 μg•l^{-1} (Figure 2). The average chlorophyll value was 10.4 μg•l^{-1} for all stations in 1997 (Table 1). Chlorophyll *a* and DO were negatively correlated with statistical significance at the 1% level ($r= -0.59$). PDI averaged 11.87 for the season (Table 1).

Overall, the water quality in Cos Cob Harbor was poor during the sampling survey. DO levels were below the water quality standard (5.0 mg•l^{-1} DO) at every station at least once during the season (Figure 3). The stations closest to the Mianus River had the worst water quality. The duration that oxygen levels were below the water quality standard was longer and hypoxia (<3.0 mg•l^{-1} DO) occurred more frequently at these sites. Station 1, located just south of the Mianus River Dam and Route 1, had the lowest water quality compared to other stations in the harbor.

At Station 1, DO levels were below the water quality standard in the bottom water for most of the last half of the study and went hypoxic twice: in mid-July and early August (Figure 3). At Station 6, located at the mouth of the harbor, bottom water oxygen levels were below the water quality standard only four times: twice in late July, once at the end of August, and again in September (Figure 3).

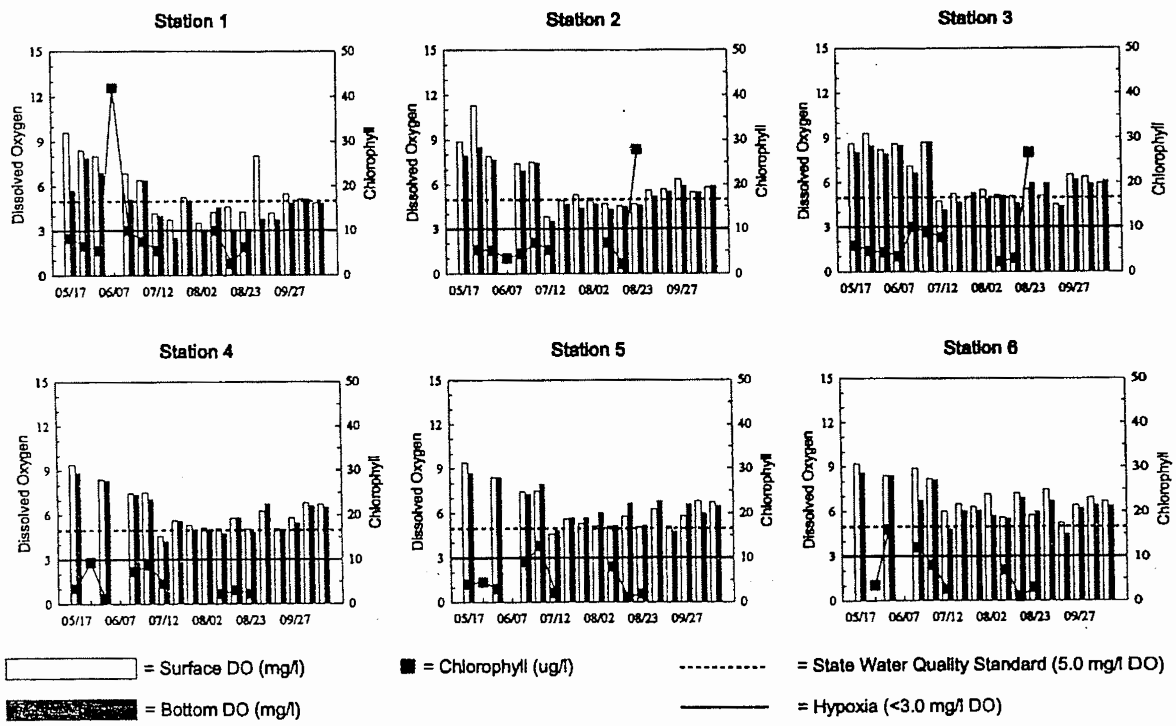

Figure 3. Cos Cob Harbor (Greenwich, CT) dissolved oxygen and chlorophyll *a* by station.

Surface chlorophyll was similar at Stations 2, 3, 4, 5 and 6 in Cos Cob Harbor. Station 1, however, had the lowest chlorophyll levels for most of the season but experienced the largest bloom in the harbor. Chlorophyll *a* concentration at Station 1 on June 7, 1997, was measured at 41.9 $\mu g \bullet l^{-1}$, indicating a very large bloom (Figure 3). At the other stations in Cos Cob Harbor, chlorophyll *a* values were low with most of the values between 5.0 $\mu g \bullet l^{-1}$ and 10.0 $ug \bullet l^{-1}$ for most of the season (Figure 3). The overall average chlorophyll value was 7.0 $\mu g \bullet l^{-1}$ (Table 1). Chlorophyll *a* and DO were negatively correlated with statistical significance at the 1% level (r= -0.06). PDI averaged 11.86 for the season (Table 1).

The water quality of Stamford Harbor was rated as poor during the season. DO was below the water quality standard (5.0 $mg \bullet l^{-1}$ DO) at every station at least once during the survey (Figure 4). The stations furthest from the mouth of the harbor had the poorest water quality. At Stations 1 and 5, in the East and West Branches, respectively, the oxygen levels were below or very close to the water quality standard for most of the season and were hypoxic (<3.0 $mg \bullet l^{-1}$ DO) one time at Station 1 and five times at Station 5 from August to September (Figure 4). Station 5, located in the West Branch, had the worst water quality, with oxygen levels that were above the water standard only three times during the last half of the survey.

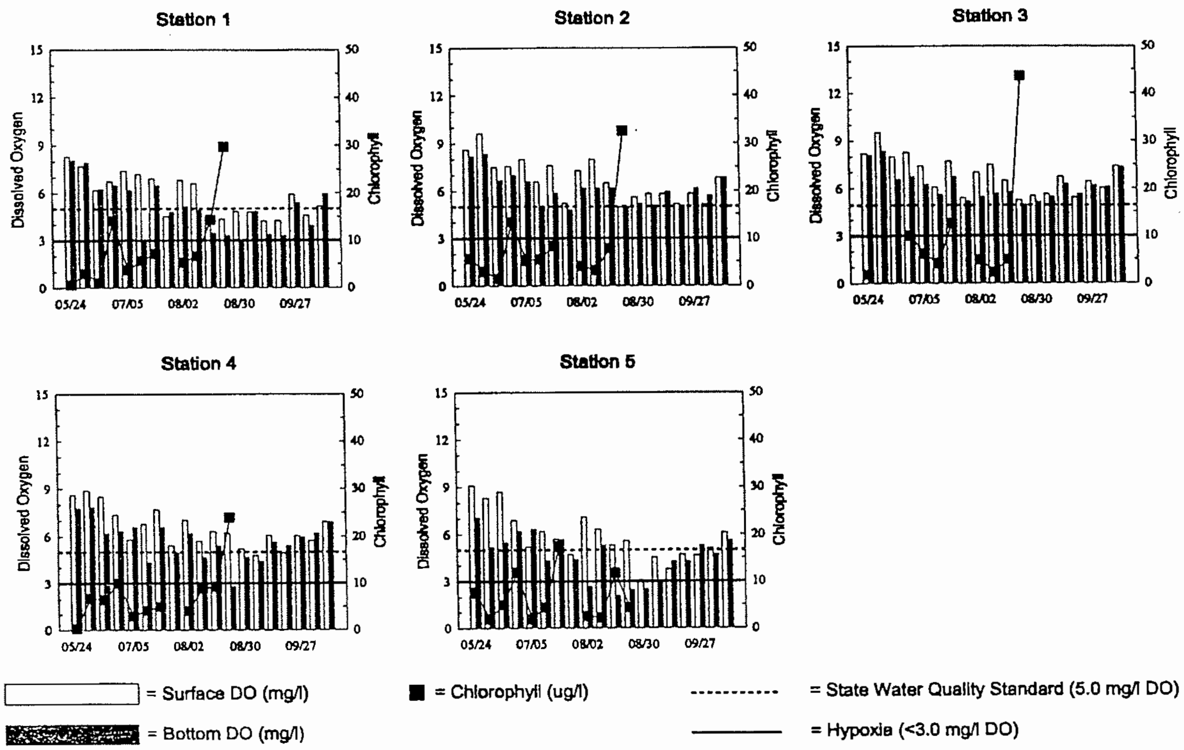

Figure 4. Stamford Harbor (Stamford, CT) dissolved oxygen and chlorophyll a by station.

At most of the stations (Stations 1, 2, 3 and 4), chlorophyll *a* levels were similar throughout the monitoring season. Chlorophyll *a* values at these stations were low, with most of the values between 5.0 $\mu g \bullet l^{-1}$ and 13.0 $\mu g \bullet l^{-1}$ (Figure 4). Station 3, however, experienced the largest level of chlorophyll detected during the 1997 season. On August 23, 1997, the level of chlorophyll *a* was measured at 43.7 $\mu g \bullet l^{-1}$ in the surface water, indicating a very large algal bloom (Figure 4). The overall average chlorophyll value for all stations in the harbor was 7.8 $\mu g \bullet l^{-1}$ (Table 1). Chlorophyll *a* and DO were negatively correlated with statistical significance at the 1% level (r= -0.23). PDI averaged 12.05 for the season (Table 1).

DISCUSSION

The results from our sampling survey showed that there is impairment of the harbors studied. Several factors, some not analyzed in this study, were influencing DO concentrations and algal bloom formation in embayments of Long Island Sound (including weather variability, nutrient loading and tidal influence). All of the harbors experienced large fluctuations in DO probably due to cyclical phytoplankton blooms, which are generally characterized by high DO levels followed by periods of low DO (Long Island Sound Study 1994). During a bloom, the number of phytoplankton (as well as the dissolved oxygen from photosynthesis) increased well beyond the amount consumed by animals. During this survey, surface water DO of Echo Bay, and Milton, Cos Cob and Stamford Harbors, was significantly higher whenever chlorophyll *a* was higher than 15.0 $mg \bullet l^{-1}$ (Figures 1-4).

A major source of non-point pollution at the study sites is storm-water runoff which carries contaminants, including nutrients, metals, oils and pesticides. The areas surrounding the harbors and the land-use within their watersheds are influencing the amount of storm-water runoff entering the harbors. In more developed areas, impervious paving materials prevent rainwater absorption by the soil, thereby increasing the amount of contaminants carried in the runoff. An analysis of rainfall data from 1997 is being undertaken to determine the atmospheric contribution to the hypoxia and algae dynamics in these harbors.

The rivers and creeks that drain into the harbors had a large influence on water quality. Those stations closest to these tributaries generally had the lowest DO levels. This is indicative of storm-water runoff influencing harbor water quality. Further study of nutrient inputs, particularly nitrogen, into these harbors would help to clarify this situation as well as aid in nonpoint pollution management. Human activities, such as discharges from sewage treatment plants and nonpoint source runoff, are responsible for 56% of the total annual nitrogen load in the Sound (Long Island Sound Study 1994).

In theory, the water quality in more developed harbors should be worse than the water quality in less developed harbors. Milton and Stamford Harbors both followed this pattern (Figure 2, Figure 4). Echo Bay and Cos Cob Harbor, however, did not seem to follow this scheme (Figure 1, Figure 3). Echo Bay had the highest rated water quality while Cos Cob Harbor had the poorest water quality. The shape of both Echo Bay and Cos Cob Harbor may influence the flushing of nutrients as well as plankton blooms. The widened mouth of Echo Bay allows for a large amount of tidal exchange as shown by Ferrand (1990). Part of this flushing is aided by the overall shallowness in Echo Bay that facilitates easier mixing by tide and wind. Cos Cob Harbor, on the other hand, has a narrow harbor mouth which experiences less tidal exchange and contains a deeper channel. Cos Cob Harbor also has mostly residential development surrounding its shores and these areas are currently not connected to waste water treatment facilities. Old, failing septic systems may be responsible for the low DO readings observed during this study. Milton Harbor has a shape similar to Cos Cob Harbor, but is bordered by a larger amount of undeveloped land surrounding its shores. Stamford Harbor, with the highest amount of impervious surfaces of the four harbors studied, had degraded water quality.

The results of the PDI survey showed that Echo Bay had the most diverse phytoplankton population (average PDI=12.15), and Cos Cob Harbor had the least diverse phytoplankton population (average PDI=11.86) (Table 1). This is also reflected in the average DO values for each of these embayments (Table 1). The relationship between phytoplankton diversity and DO concentration is under further investigation.

The effects of hypoxia on the living marine resources in the Sound depend upon the extent, duration, and intensity of the hypoxic period. It is likely that an increase in the sources and occurrences of coastal pollution, due to human activity, will result in more intense hypoxic events which severely stress and kill commercially and recreationally important fish and shellfish. The areas of concern identified in this study will be watched closely as they will be more severely impacted by poor water quality conditions in the future. Land use practices must be improved around these areas of concern to minimize non-point and point source pollution and their potential impact on marine life.

ACKNOWLEDGEMENTS

The author would like to thank Vic Pyle, III, Habitat Restoration Project Manager for Save the Sound, Inc., for producing the land-use maps used in this paper.

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MONITORING THE POPULATION ABUNDANCE OF WINTER FLOUNDER IN THE NIANTIC RIVER

Danila, D.J., Northeast Utilities Environmental Laboratory, P.O. Box 128, Waterford, CT 06385

The winter flounder is a coastal flatfish most abundant in the central portion of its range, which includes Long Island Sound. Its population is subdivided into a number of stocks associated with specific estuaries or coastal areas and adults tend to faithfully return to natal estuaries to spawn each winter. The abundance of adult flounder spawning in the Niantic River has been monitored during mid-February to early April each year since 1976. Winter flounder are captured using a 9.1-m otter trawl and all fish larger than 20 cm are marked with a brand chilled in liquid nitrogen before returning them to the river. Annually, relative abundance is characterized by trawl catch-per-unit-effort (CPUE) and absolute abundance estimates have been generated since 1984 using the mark and recapture data and the Jolly model for open populations. Information on sex ratio, length-frequency distribution and spawning condition is also recorded during each survey. Using a length-fecundity relationship, annual egg population estimates have also been calculated. Although the two abundance estimates are independent of one another, CPUE and absolute abundance estimates are highly correlated. Abundance of Niantic River winter flounder peaked in the early 1980s; but as fishing mortality increased to high levels in the late 1980s, abundance declined thereafter and presently is very low. Despite small numbers of spawners, winter flounder spawners remain capable of producing large numbers of larvae, which, under appropriate conditions, can result in a relatively strong year-class of juveniles and an increase in spawners some 3 to 4 years later.

DRAMATIC SUB-SURFACE CHLOROPHYLL MAXIMUM FORMATION IN THE UPPER THAMES RIVER

Kremer, J., Department of Marine Sciences, University of Connecticut, Groton, CT 06340; H.M. Weiss, Project Oceanology, Groton, CT 06340; E. Morales, Ecology and Evolutionary Biology, University of Connecticut, Storrs, CT 06269

Weekly water quality monitoring from June-September 1997 at six stations in the Thames River from Norwich Harbor to Long Island Sound documented the common appearance of a large, distinct maximum in photosynthetic pigments at about 1 m depth, coincident with a strong halocline. The shape and peak magnitude of the sub-surface pigment feature often changed dramatically within 1-2 km downriver of Norwich Harbor, the point where Shetucket and Yantic river water enters the estuary over dams. Large variations in the chlorophyll-fluorescence calibration factor were noted. CTD data were supported by direct counts of phytoplankton species. A similarity analysis indicated that the taxonomic composition of phytoplankton resembled source waters entering the rivers. However, the concentration of cells was higher in the mixing zone than either end member. These algae represent a potential contribution of organic oxygen demand to the deepwater hypoxia of Norwich Harbor. While the origin of the pigment peak seems likely to be riverborne phytoplankton cells, the mechanism by which the fluorescence is concentrated within the halocline is uncertain. Possible mechanisms include: (1) *in situ* growth, (2) inputs from coves or lateral tributaries below the Shetucket and Yantic rivers, (3) an artifact of changes in the fluorescence per cell of osmotically stressed phytoplankton and (4) physical concentration of algal cells in the shear zone associated with estuarine circulation.

LONG-TERM TRENDS OF AMERICAN LOBSTER (*HOMARUS AMERICANUS*) ABUNDANCE IN EASTERN LONG ISLAND SOUND, CONNECTICUT

Landers Jr., D.F. and M. Keser, Northeast Utilities Environmental Laboratory, P.O. Box 128, Waterford, CT 06385

From 1978-1997, abundance and catch-per-unit-effort (CPUE) of American lobster (*Homarus americanus*) have been examined from May through October in the nearshore Connecticut waters of eastern Long Island Sound (LIS). Catches were dominated by pre-recruit-sized lobsters. The average annual CPUE ranged from 0.904-2.457 lobsters/pot. While the abundance of pre-recruit lobsters increased over the study period, a significant decline was observed in legal-sized lobster abundance. Annual legal CPUE in 1978 was 0.173 compared to 0.099 in 1997. The decline in legal lobster CPUE can be attributed to increased fishing effort, which has more than doubled since 1978. Recognizing the economic importance of the lobster resource in LIS (ca. $5 million annually) and the intense exploitation rates, fishery managers implemented measures in the 1980s to improve lobster recruitment and survival, including requirements for escape vents to allow escapement of sublegal-sized lobsters and increases in the minimum legal size from 3 3/16" to 3 1/4" (carapace length). Several changes were observed in population characteristics of local lobsters following the implementation of these new fishery regulations. For instance, lower incidence of claw loss was attributed to the use of escape vents; and increases in the percentage of egg-bearing females corresponded to increases in the minimum legal size. While the status of the American lobster fishery has been characterized as overfished, the apparent stability of the LIS lobster population, despite current high exploitation rates, is most likely due to the fact that female lobsters in LIS become mature and spawn at smaller sizes than in more northern waters (e.g., Gulf of Maine), thereby providing a buffer against recruitment failure.

FLOW DYNAMICS AND WATER QUALITY IN A LONG ISLAND SOUND EMBAYMENT

Morton, B.L., Aqua Solutions, East Hartford, CT 06108; D.W. Gerwick, D.W. Gerrick Engineering, Waterford, CT 06385; J.R. Jadamec and C.P. Anderson, Coastal Environmental Laboratory, University of Connecticut, Groton, CT 06340

A study was initiated in late spring of 1997 to assess sediment conditions and water quality within Jordan Cove, a shallow estuary located in Waterford, CT. Two distinct reaches characterize the cove. One is a 1.5 mile narrow sinuous reach landward of a railroad causeway. It is fed by several freshwater channels. The other reach, seaward of the railroad causeway, is influenced predominately by tidal and wave action. Analysis of preliminary chemical and physical data collected monthly from May 1997 through January 1998 indicates that the landward reach is receiving nutrient enrichment from nonpoint sources resulting in seasonal eutrophication. The seaward reach is more thoroughly flushed, but suffers from heavy sediment inundation thus restricting its use for navigation and recreation. The results obtained during this ongoing study of water quality in Jordan Cove and the analysis of sediment distribution using a finite element model will be discussed.

CONNECTICUT DEPARTMENT OF ENVIRONMENTAL PROTECTION LONG ISLAND SOUND WATER QUALITY MONITORING PROGRAM: ANALYSIS TRENDS IN LONG ISLAND SOUND WATER QUALITY 1991-1997

Olsen, C.B., N.P. Kaputa and J.B. Flaherty, State of Connecticut Department of Environmental Protection, Bureau of Water Management, Hartford, CT 06106

Since 1991, the Connecticut Department of Environmental Protection (CT DEP) has been monitoring the water quality of Long Island Sound (LIS). The primary goal of this monitoring program is to develop a long-term data base from which the effectiveness of management actions to reduce nitrogen inputs to the Sound may be evaluated. Year-round monthly sampling at 18 stations from the western Narrows to Block Island Sound provides nutrient and chlorophyll *a* concentrations and water column profiles of temperature, salinity, irradiance and dissolved oxygen. Additional biweekly summer sampling at 40 stations provides data on the recurrent low dissolved oxygen condition known as hypoxia. The analysis of data from over seven years of monitoring has revealed some significant trends. The significant trends observed include decreasing surface water chlorophyll *a*, decreasing surface water dissolved organic carbon, decreasing bottom water ammonium, increasing surface water dissolved silicate and increasing surface and bottom dissolved oxygen. These trends are observed consistently throughout the Sound from stations in the western, central and eastern basins. Such trends may be evidence of improving water quality in LIS. The CT DEP encourages the research community to make use of the monitoring program and the resultant data base as an aid to complementary research efforts in LIS.

MONITORING STUDIES OF BENTHIC INFAUNAL COMMUNITIES NEAR A POWER PLANT COOLING WATER DISCHARGE FROM 1980-1997

Vozarik, J.M. and J.T. Swenarton, Northeast Utilities Environmental Laboratory, PO Box 128, Waterford, CT 06385

Monitoring studies of soft-bottom infauna near Millstone Nuclear Power Station (MNPS), Waterford, CT, have been conducted since 1980 to assess impacts associated with the condenser cooling water thermal discharge. Subtidal sediment core sampling was conducted annually in June and September to characterize the sedimentary environment and infaunal community composition. Startup of a third reactor at MNPS in 1986 modified sedimentary environments and infaunal community structure at two stations nearest to the discharge. Increased discharge velocity caused sediment scouring at the Effluent station (EF, 100 m seaward of the discharge), resulting in coarser sediment that supported a community with higher abundances of oligochaetes and fewer polychaetes (e.g., *Tharyx* spp. and *Polycirrus eximus*). Concomitant increases in sediment silt/clay content at the station in Jordan Cove (JC, 500 m east of MNPS) suggested that sediments scoured from EF were being deposited in this area. Rapid community change was observed at JC, as the previously dominant oligochaetes and polychaetes *Aricidea catherinae* and *Tharyx* spp. all decreased. This depositional event likely occurred over a short period (<1 yr) and yet only minor evidence of recovery has been noted after more than 10 years. The shutdown of MNPS since 1996 had observable effects only at EF (e.g., increases of *M. ambiseta* and *A. catherinae*) that were attributed to decreased water flow and possibly to the lack of elevated temperatures.

WATER COLUMN

AND REGIONAL

PROCESSES

A MODEL FOR LONG-TERM VEGETATION CHANGE AND POTENTIAL HABITAT LOSS IN CONNECTICUT TIDAL MARSHES

Bellet, L., Department of Botany, Connecticut College, New London, CT 06320

ABSTRACT

Quantitative data as well as qualitative observations at several tidal marshes along the coast of Connecticut reveal the occurrence of significant vegetation change. High marshes formerly dominated by *Spartina patens* and *Juncus gerardi* are being replaced by stunted *Spartina alterniflora* and a mix of forbs. Areas where this vegetation change is evident exhibit lower elevations (as much as 9.3 cm) relative to stable sites. This suggests that increased flooding frequency, likely driven by accelerated relative sea level rise (RSLR), is impacting high marsh vegetation. By re-surveying the elevation and vegetation at three 25-year-old gridded transects in southeastern CT, this study examines the rate of marsh accretion and vegetation changes which have occurred relative to SLR over this time period. Several edaphic factors associated with increased flooding frequency including sediment sulfide concentration, redox potential and salinity are followed throughout the growing season. The data suggest that these factors interact in a positive feedback loop which favors the stunted *S. alterniflora*/forb community over that of *S. patens* and *J. gerardi* on the high marsh. As the mechanisms of this change are better understood, attempts can be made to restore altered sites and possibly reverse this damaging process.

INTRODUCTION

This project sets out to test a multi-factor working model for tidal marsh vegetation change at three sites along the coast of southeastern Connecticut. Twenty-five years ago, permanent transects were established at the sites. Within these transects the exact elevation and vegetation patterns were determined. Continued monitoring at the sites has revealed that certain marshes are not maintaining their elevations relative to rising sea level, while others are. Sites on which the elevation has not kept up with relative sea-level rise (RSLR) have experienced a shift from the densely populated high marsh perennials, *Spartina patens* and *Juncus gerardi*, to stunted *Spartina alterniflora* and forbs. The latter exhibit a lower height and stem density than their floristic predecessors. The implications of this change include altered habitat for marsh-dependent organisms, a decreased coastal zone productivity and potential habitat loss.

The exact reasons for these changes are unclear. It is well documented that about 100-150 years ago RSLR increased from a rate of 1 mm yr^{-1} to 2.0-2.5 mm yr^{-1} (Niering et al. 1977; Orson et al. 1987). Warren and Niering (1993) propose that on changed marshes the increased hydroperiod leads to several edaphic changes and eventually to a shift in the vegetation. Over the last decade, such a shift in vegetation has been noted on many other Connecticut tidal marshes (Warren et al. 1991, personal observations); but the many factors possibly contributing to the change have not been tested at these sites.

With global warning and the likelihood that RSLR will continue to rise (Titus 1988), the prospect of Connecticut's tidal marshes being altered or submerged due to their inability to keep up with this rise is a cause for some alarm. The model proposed in this project (Figure 1) draws on over 20 years of research on community vegetation patterns in order to explain the nature of this recently observed and widespread vegetation change. The model is a potentially valuable tool for managers and scientists to understand the many interacting factors involved in this community change and to attempt to reverse these changes.

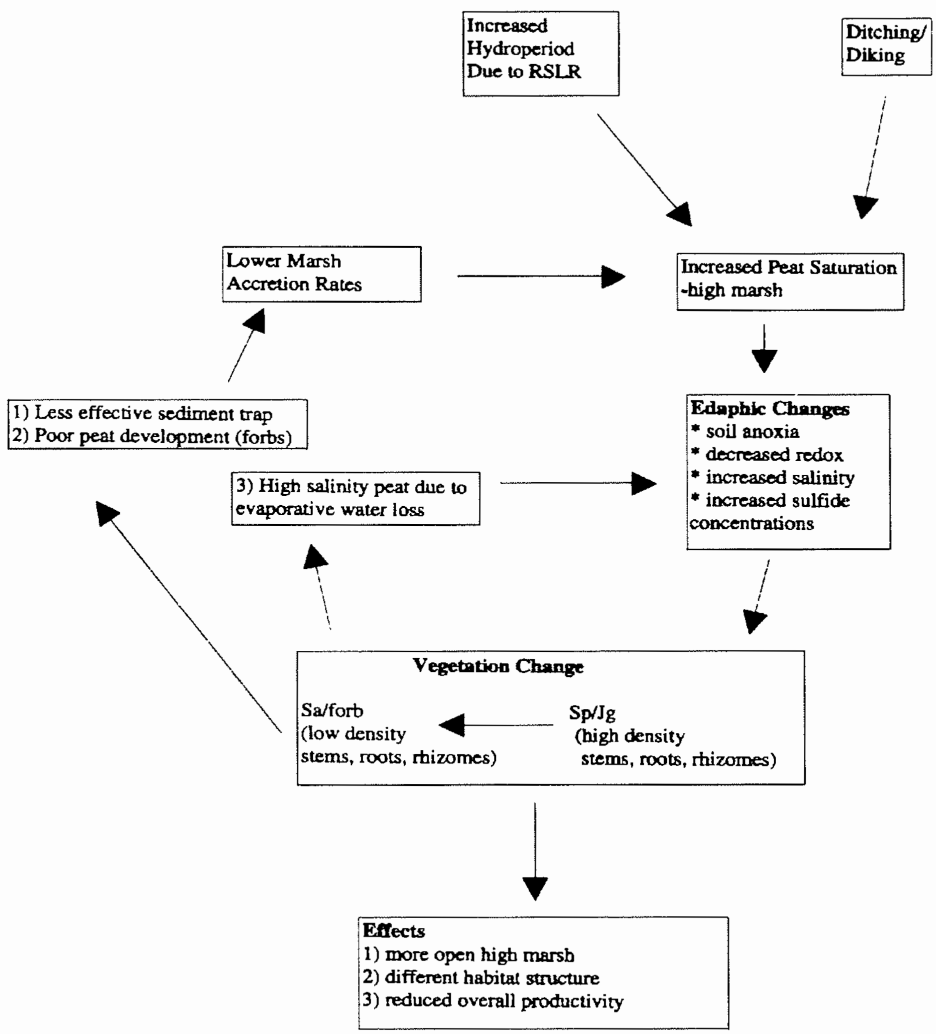

Figure 1. A model for the interaction of accelerated RSLR with marsh accretion and subsequent vegetation change. Adapted from Warren and Niering (1993).

THE MODEL

Driven primarily by accelerated RSLR, marsh surface elevations in some areas are lowering relative to sea level (Delaune et al. 1983; Warren et al. 1991). This circumstance initiates several positive feedback loops. Increased hydroperiod possibly accompanied by ditching and diking lead to greater peat saturation on the high marsh. A suite of edaphic changes will likely follow (Mendelssohn et al. 1981; King et al. 1982; Howes et al. 1986) including increased salinity, soil anoxia, decreased redox potential and increased sulfide concentrations due to the microbial reduction of SO_4^{2-} in the anoxic marsh peat. These changes directly affect the vegetation causing a decrease in species such as *S. patens* and *J. gerardi* which are less salt tolerant and not well adapted to extreme soil anoxia and sulfide phytotoxicity (Warren and Goslee 1995). As a result, there is an increase in abundance of species such as stunted *S. alterniflora* and forbs which are better adapted to these conditions. The latter species do not grow a tall and are characterized by low density stems, roots and rhizomes and subsequently reduced overall productivity.

Dacey and Howes (1984) found that stunted *S. alterniflora* has a lower transpiration rate than the tall form and were able to relate this to the decreased drainage rate in areas where the short form was abundant. Furthermore, the short form is less capable of oxidizing its sediments actively due to metabolic stress and passively due to the smaller size of its oxygen passageways (Howes et al. 1981). This combination of circumstances 'feeds back' to the edaphic changes which originally brought about vegetation change.

The lower above- and below-ground density of the stunted *S. alterniflora*-forb community contributes in other ways to the feedback loop. The decreased stem shading of the peat surface leads to greater evaporative water loss and a higher salinity peat than *S. patens* and *J. gerardi* can tolerate (Warren and Niering 1993). Additionally, lower density stems are a less effective sediment trap (Gleason et al. 1979); and forbs contribute minimally to peat development (Niering et al. 1977; Orson et al. 1987). These last two conditions may serve to bring about lower marsh accretion rates which ultimately lead to a lower marsh surface elevation relative to sea level. From this point, the interacting causes and effects which initiated the feedback loops begin once again.

The biodiversity which marshes support and the numerous roles that marshes play as transitional zones between terrestrial and marine ecosystems (Mitsch and Gosselink 1986; Day et al. 1989) lends a sense of urgency to our understanding of this recent and widespread threat. Utilization of a historical data set as well as experimentation with the proposed model will offer highly detailed insight as to the mechanisms of vegetation change. Once the causes are better understood, then attempts can be made to restore altered sites and possibly reverse this damaging process.

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1500 YEARS OF SEA LEVEL RISE IN LONG ISLAND SOUND

Thompson, W.G., E. Thomas and J. C. Varekamp, Department of Earth and Environmental Sciences, Wesleyan University, Middletown, CT 06459

ABSTRACT

Micropaleontological and lithological data were used to reconstruct the depositional environment of samples from peat cores, from which we infer the position of paleo mean sea level relative to the paleo marsh surface. These data were combined with ^{14}C ages to construct relative sea level rise curves for Connecticut marshes near Clinton, Guilford and Branford. The curves from all three marshes show a consistent overall pattern with superimposed abundant local variability. Relative sea level rose at about 1 mm/yr from about 0 AD until 1100 AD, at a slow rate of 0.5 mm/yr with frequent local fluctuations from 1100 to 1650 AD, and at a rapid rate of 3 mm/yr since about 1650 AD. We subtracted the local average rates of post-glacial crustal subsidence for the Long Island Sound region (~1 mm/yr) from the relative sea level rise curves to remove subsidence effects. The resulting estimates of variations in local eustatic sea level show undulations, with a drop during the period 1200 -1700 AD followed by a sudden rise from about 1700 -1800 AD on. The sea level data show a remarkable correlation with minor climate fluctuations on a time scale of centuries. The minimum in eustatic sea level occurred around 1500 -1750 AD, which coincides with the coldest part of the Little Ice Age. The following period of climate warming is characterized by a rise in sea level, but there is no evidence, for an acceleration in sea level rise during the 20^{th} century.

INTRODUCTION

Salt marsh peat sequences have long been used to estimate the position of paleo sea level (Redfield and Rubin 1962; Bloom and Stuiver 1963; Bloom 1964; Redfield 1967; Harrison and Bloom 1977; van de Plassche et al. 1987; Fletcher et al. 1993), because the growth of salt marshes in the intertidal zone keeps approximately pace with Relative Sea Level Rise (RSLR). In more detail, marsh accretion occurs in dynamic equilibrium with RSLR (Allen 1990a, b, c; 1991; 1994) and marsh deposits consist of stacked sequences that are related to relative drownings and emergences caused by variations in rates of accretion and RSLR (Shennan 1986; Scott et al. 1987; van de Plassche 1991). These small-scale variations in relative retreat and inundation by the sea are preserved in the lithology as well as in floral and faunal characteristics of salt marsh deposits. We use these subtle facies changes as indicators of variations in the rate of RSLR and quantify the rates as detailed below.

Modern coastal salt marshes exhibit a vertical zonation in many parameters in a complex response to ecological factors related to the local tidal frame (Bertness 1992; Niering et al. 1977; Nixon 1982). Benthic foraminiferal assemblage zones in modern marshes show a pattern of decreasing species diversity with increasing vertical distance from mean sea level, with no foraminifera above the highest high water line. Specific faunal assemblages have been defined for zones at different elevations parallel to mean sea level (Scott and Medioli 1980; Scott and

Leckie 1990; Gehrels 1994). Agglutinated benthic foraminiferal microfossil assemblages preserved in peats can thus be used to define paleoenvironmental facies that can be related to the tidal framework of the past by analogy with the modern marsh zonation.

From these paleo-environmental analyses we can make a relatively accurate estimate of the position of paleo sea level with respect to the depositional surface (Thomas and Varekamp 1991; Gehrels 1994; Gehrels et al. 1996). Radiocarbon ages of plant fragments grown *in situ* provide an accurate means of dating stratigraphic levels in cores older than a few hundred years. Numerical ages for the upper part of the cores have to be determined using ^{210}Pb dating methods, and possibly correlation to levels of pollutant metals (Nydick et al. 1995; Varekamp and Thomas 1998). The combination of a numerical age model with paleoenvironmental data from the same location thus permits the reconstruction of detailed RSLR curves.

STUDY SITES AND METHODS

We studied marshes of the Hammock River near Clinton, CT (Thomas and Varekamp 1991; van de Plassche 1991; Van de Plassche et al. 1998; Varekamp et al. 1992; 1999), the Guilford marshes in Connecticut (Nydick et al. 1995), and the Farm River and Kelsey Island marshes near Branford, CT (Figure 1). In addition, we studied marshes around Delaware Bay (NJ) with similar methods, to compare the LIS data with those from a different region (Varekamp and Thomas 1998). Initial field surveys of each marsh involved extensive coring and field description of core stratigraphies to map the subsurface marsh structure.

Each RSLR curve was determined from a single core location and the different records were established independently. Facies changes in a salt marsh occur over very short distances; so that combination of environmental data and age information from different cores, even within the same general marsh regions, increases the risk of substantial errors (van de Plassche et al. 1998; Varekamp et al. 1999).

Undisturbed sites with high-marsh peat sequences were selected for study, because that salt marsh facies has the highest resolution in foraminiferal assemblage zones (Scott and Medioli 1980). Continuous, undisturbed cores of approximately 2.5 m in length and 10 cm diameter were obtained, and sliced into 2.5 to 5 cm-thick samples for foraminiferal counts and floral analysis.

The whole core was thus analyzed. Portions of core sections from 5 to 10 cm in length were selected for bulk radiocarbon dates, and in some cases individual *in situ* plant fragments, were selected for AMS dates. In many cases, ^{210}Pb profiles were obtained to date the most recent parts of the cores.

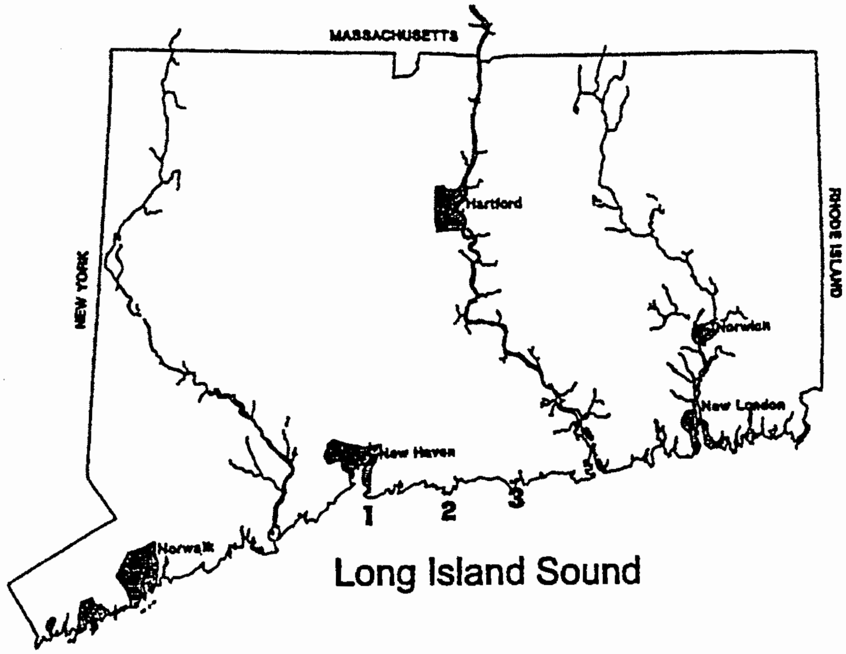

Figure 1. Salt marsh study sites in Connecticut. 1. Farm River marsh/Kelsey Island; 2. Hammock River marshes, Clinton; 3. Guilford marshes.

An age/depth model was developed for each core from the available radiometric ages, and an age was assigned to the midpoint of each sample slice through linear interpolation between age index points. The vertical distance between paleo mean high water and the dated paleo marsh surface was estimated from the benthic foraminiferal assemblages in each sampling interval. These data were combined and displayed graphically in a RSLR curve for each core location. The vertical precision is approximately ±15 cm and the age precision is ±75 years; both uncertainty estimates vary from sample to sample, with greater precision in the upper parts of the cores.

In earlier work, we divided the instantaneous rates of RSLR by the local average rate (rate ratios) to determine periods when the rate was anomalous as compared to the local average (Varekamp and Thomas 1998). This allowed us to compare SLR rate patterns from areas with different absolute rates of RSLR. In this paper, we aimed at estimating the regional eustatic component of sea level rise; and we subtracted the local long-term average rate from the rate for each sample. These patterns can be compared between regions as well, but this can only be meaningfully accomplished when a number of independently-dated RSLR curves from several marshes in a given region are available so that the local variability and "noise" level is well established.

RESULTS

The RSLR curves from Guilford (Nydick et al. 1995), Clinton (Varekamp et al. 1992), and the Farm River marsh show comparable patterns but have considerable superimposed local variability. The local variability may result from a combination of the compaction rate of the local marsh sequence and variations in tidal range. Tidal range variations within a marsh may be linked to the "openness of the marsh to the sea", which influences the local tidal range in marsh sections remote from the inlet (van der Molen 1997). The overall trends of RSLR in all 3 Long Island Sound marshes, however, are similar: approximately 1 mm/yr until 1100 AD, about 0.5 mm/yr between 1100 and 1650 AD, and generally 3 mm/yr since 1650 AD (Figure 2).

The RSLR curve determined on a core from Kelsey Island near the Farm River marsh shows faster rates and this curve has a very steep slope between 700-1300 AD. The section from 1600 AD to Recent is more similar to that of the other records, although the rates are still higher. The boundary between the two parts of the curve coincides with a lithological transition from gray-blue mudflat clays with mollusk shells to clay-rich low-marsh peat with *Spartina alterniflora* remains. The two different RSLR records from the Farm River marsh are separated by a faultzone, and syn-sedimentary movement along this fault might explain the unusual RSLR curve from Kelsey Island (research in progress).

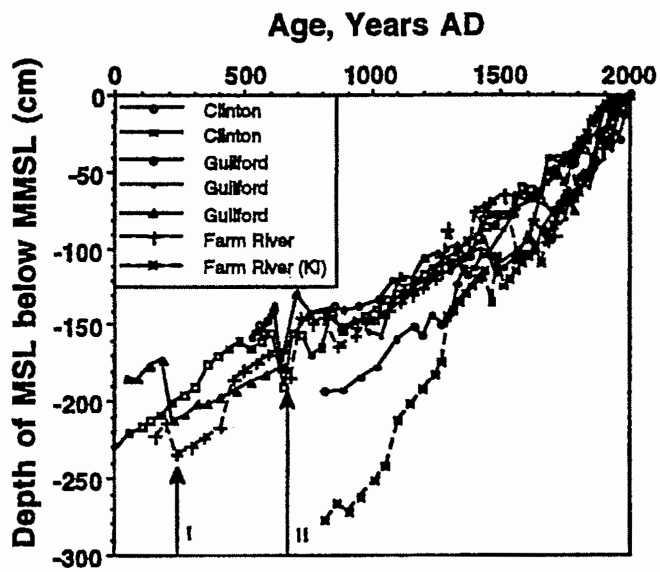

Figure 2. Relative Sea Level Rise Curves from Guilford, Clinton and Farm River marshes in Connecticut; KI denotes the Kelsey Island core as described in the text.

There is no evidence of a RSLR acceleration during the 20th century that could be associated with modern global warming. Many marshes were modified during the early part of this century through mosquito ditching and/or emplacement of tide gates, which modified their drainage characteristics. The last 100 years of these RSLR records may thus be compromised in some cases, and more work on undisturbed marshes is needed to confirm the recent RSLR record.

There are two periods of abrupt apparent SL drop in these curves at ~250 and 600 AD (I and II in Figures 2 and 3), and a sudden recovery of SL around 700 AD (II in Figures 2 and 3). These features occur in multiple cores from 3 marshes and thus represent a regional signal. Moreover, we re-interpreted faunal and age data from the Barnstable marshes on Cape Cod, MA (De Rijk 1995), and this record also shows the same "jumps" in SL at the same times (Figure 3).

These events could be related to a period of partial marsh closure, possibly the result of a true short-term regression. Most likely, several marshes became partially closed-off from Long Island Sound and experienced reduced tidal ventilation, which was restored again by 750 AD.

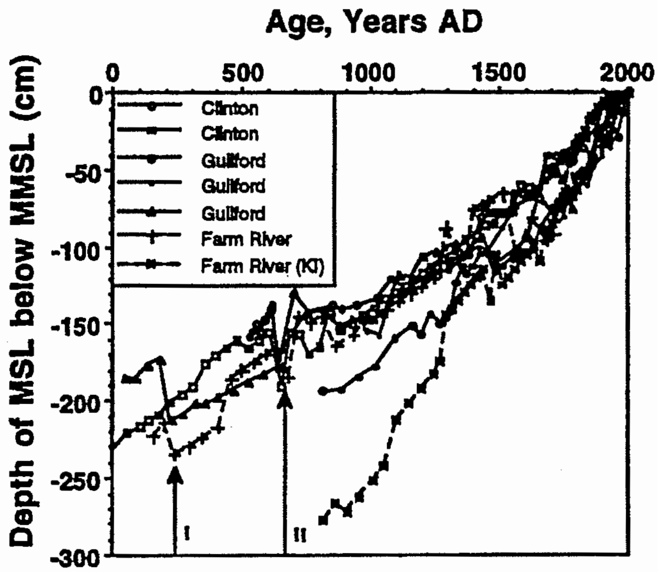

Figure 3. Detail of Figure 2 with the RSLR curve from the Barnstable Marshes (Cape Cod, MA) added. Note the drops and jumps in sea level (I and II) in these cores.

These effects may have been caused or enhanced by extensive freezing-over of the whole Sound because the period around 500 -700 AD was indeed an extremely cold period (Stuiver et al. 1997).

The crustal subsidence component in these RSLR curves can be removed through subtraction of the long-term RSLR trend over the last 2000 years (approximately 1 mm/yr for LIS; Peltier and Tushingham 1991; Patton and Horne 1992) or through subtraction of an averaged local long-term rate of RSLR. The latter includes local tectonic movements and possible compaction effects. Because the true rates of crustal subsidence are not exactly known, we chose to subtract local averaged rates of RSLR over the total available date range for each core. We used a best linear fit to the mean high water rise data to obtain the local averaged rate and subtracted these values from the observed mean high water rise data, setting the modern sea level at zero. The resulting curves (Figure 4) show the pattern of true eustatic sea level movement, but may still contain a local or regional crustal movement component. The absolute values of the differences in eustatic SL elevation with respect to the modern SL reference plane thus carry still some uncertainty.

All 3 LIS cores show a pattern of overall falling SL since about 1000 AD, with abundant super-imposed noise, with a minimum at about 1500-1700 AD. The subsequent rise in SL from about 1650 AD on brought SL up again, close to or somewhat higher than sea levels of about 1000 years ago. The "eustatic" SL minimum correlates well with the period of the Little Ice Age (Grove 1998; Bradley and Jones 1992; Stuiver et al. 1995). The "eustatic record" from Delaware

Bay is shown for comparison, and shows a very similar pattern, but at very different rates of RSLR (Varekamp and Thomas 1998).

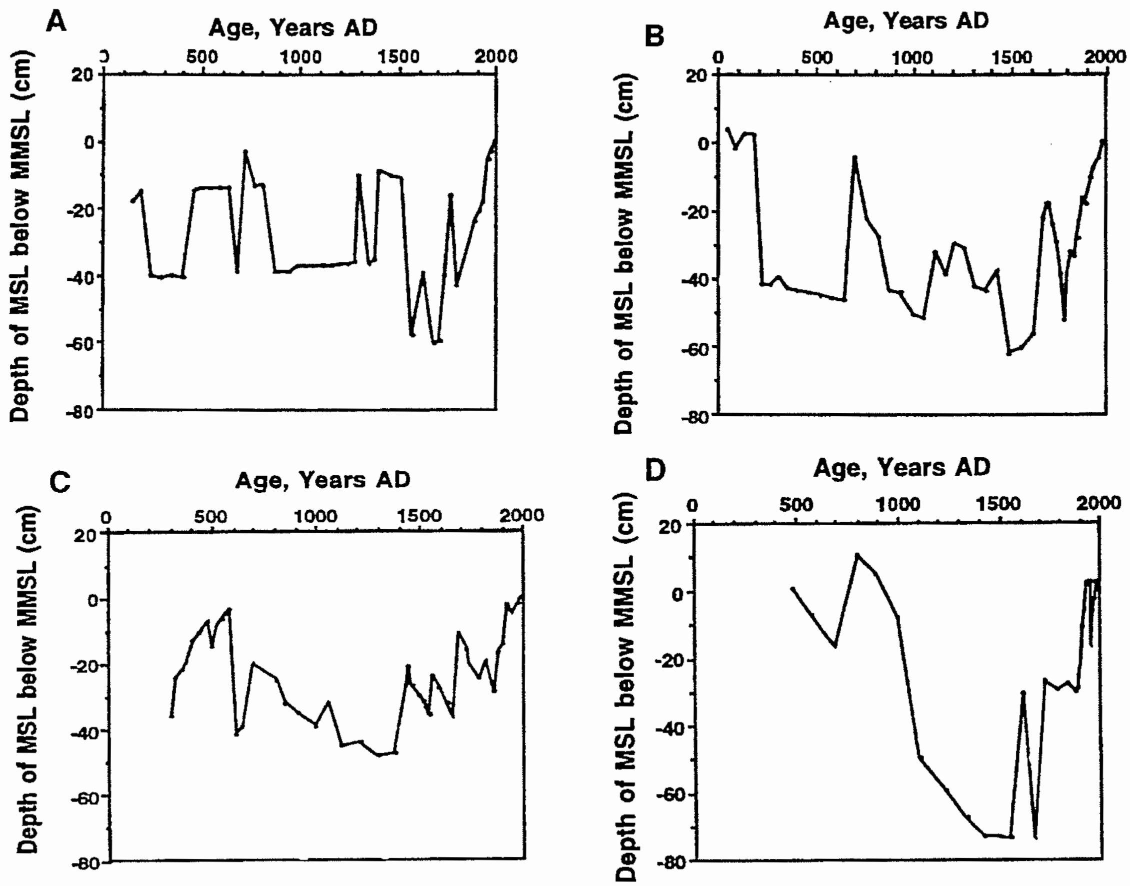

Figure 4. Eustatic sea level trends for Connecticut and Delaware Bay, NJ, marshes. A = Branford, CT, BFB Core; B = Guilford, CT, GK Core; C = Clinton, CT, F Core; Dennis Creek, NJ, DCE Core.

CONCLUSIONS

The RSLR curves from a series of marshes in Connecticut show a variety of structure in the data. The "eustatic components" can be approximated and show coherence. The Little Ice Age was a period of eustatic regression, and the last 300 years were characterized by a rapid recovery of eustatic sea level. The current sea level stand is probably higher than the level during the warm period of 1000 years ago. Sudden drops in SL at around 250 and 600 AD occur in several locations along Long Island Sound and on Cape Cod, and must be caused by some regional trigger. They occurred during two cold "snaps" in the first millennium, and may signify partial closure of the salt marshes for some 100 years. After reopening of the marshes, the former full tidal ventilation was re-established, and normal marsh accretion resumed.

Our data presented here and our earlier work strongly suggests a tight correlation between the cooling of the Little Ice Age, a slowing of the rates of RSLR, and a low stand of the eustatic sea level. A broadly similar pattern was observed by Van de Plassche et al. (1998), although we are somewhat critical of their approach of combining detailed age records with paleoenvironmental records from different core locations that contain hiatuses (Varekamp et al. 1999). The global warming of this century is not accompanied by a clear acceleration in the rate of RSLR; but overall, the rising temperatures of the last 300 years are accompanied by a rapid rise in eustatic sea level. We have no simple explanation for the correlation between RSLR and small climate fluctuations, and we speculate that apart from steric effects and Alpine glacier melting, the repositioning of ocean currents with changes in climate may induce regional trends in SLR of which we may register the effects along the North American eastern seaboard (Varekamp et al. 1992).

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UNDERSTANDING THE RELATIONSHIP BETWEEN LAND USE AND NITROGEN LOADING: THE IMPORTANCE OF SPATIAL AND TEMPORAL VARIABILITY

Anisfeld S., L. Fan and G. Benoit, Yale School of Forestry and Environmental Studies; R. Zajac, University of New Haven, New Haven, CT 06511

Export coefficients for different land uses (urban, agricultural, forested) are often used to estimate annual nonpoint source (NPS) nitrogen (N) loads Long Island Sound from various watersheds. While this approach provides a very useful starting point, the relationship between land use and N loading is not a straightforward one, and the predictive power of simple export coefficients is limited. The next step should involve incorporation of spatial, temporal and hydrologic variability into our models, which should allow us both to improve our ability to correctly predict real-world N loads, and to estimate the sources of N loading at specific points in time (e.g., spring bloom). For the last two years, we have measured nitrogen yields from 9 sub-basins of the Quinnipiac River watershed under different flow conditions. Using GIS land use/land cover data for these subbasins, we have attempted to understand the variation in N loads from these areas. The following principles have guided our analysis: (1) The number of land use categories utilized in a nitrogen loading model should be sufficient to allow the distinguishing of areas which contribute different N loads (e.g., suburbs vs. urban centers); (2) Land uses in riparian zones should be given greater weight than land uses farther away from the stream; and (3) The way in which N loading varies with flow must be understood and explicitly considered for each land use. These principles have been incorporated into a model describing the relationship between land use, hydrologic conditions and N loading.

GENERALIZED INVERSE ESTIMATES OF THE CIRCULATION IN LONG ISLAND SOUND

Bogden, P. and J. O'Donnell, Department of Marine Sciences, University of Connecticut, Groton, CT 06340

A generalized inverse of shipboard ADCP data distinguishes tidal and non-tidal flow. This can be accomplished with data from a single 10-hour survey. Data from several surveys were collected in Long Island Sound. The measurement program was designed with the intention of measuring the poorly understood general circulation. Support for the hypothesis that there is low-frequency non-tidal flow is obtained by showing that its converse (the null hypothesis of no such flow) is false. Rejection of the null hypothesis is necessary when non-tidal data residuals exhibit consistent patterns of westward flow along the northern boundary of Long Island Sound and eastward flow along the south. The inverse analysis incorporates a simple dynamical model for depth-averaged tidal currents and sea surface height. Errors in the prognostic calculation (the forward model) are too large to permit definitive conclusions from the data. The inverse calculation allows for errors in dynamics and boundary conditions. Once these model errors are accounted for, the improved tidal model allows straightforward estimation of the non-tidal component of the data. Predictability studies verify that inversion of a single 10-hour survey improves the tidal prediction. Experiments with the same data show that tuning a friction parameter in the forward model reduces model-data misfit for hindcasts. However, in contrast with inversion, tuning actually degrades forecasts.

REGIONAL SEAFLOOR ENVIRONMENTS IN LONG ISLAND SOUND

Knebel, H.J., R.P. Signell, R.R. Rendigs, L.J. Poppe, J.H. List and M.R. Buchholtz ten Brink, United States Geological Survey, Woods Hole, MA 02543

We have identified and mapped four categories of modern seafloor sedimentary environments across the topographically complex Long Island Sound (LIS) estuary from a regional set of sidescan sonographs, bottom samples and video-camera observations, supplemented by modeled physical-oceanographic data. (1) Environments of erosion or nondeposition consist of exposures of glacial drift, coarse lag deposits and possibly bedrock and include sediments which range from boulder fields to gravelly, coarse to medium sands. (2) Environments of coarse-grained bedload transport are mantled by sand ribbons and sand waves and contain mostly coarse to fine sands with only small amounts of mud. (3) Environments of sediment sorting and reworking comprise both uniform and heterogeneous sediment types and contain variable amounts of find sand and mud. (4) Environments of deposition are blanketed by homogeneous very fine sands and mud. The patchy distribution of sedimentary environments reflects both regional and local changes in bottom processes. Regional changes are primarily the result of a strong east-to-west-decreasing gradient in bottom tidal-current speeds caused by the westward widening of the basin. This gradient has produced a westward succession of environments beginning with erosion or nondeposition at the narrow eastern entrance changing to an extensive area of coarse--grained bedload transport, passing into a contiguous band of sediment sorting and ending with broad areas of fine-grained deposition on the flat basin floor in the central and western sound. Locally, the extensive fine-grained deposits in the central and western sound are discontinuous where the bottom flow is enhanced by and interacts with the irregular seafloor topography. In these areas, increased sediment movement has produced variable assemblages of environments of erosion, sorting and reworking.

CIRCULATION IN THE ESTUARINE REACH OF THE CONNECTICUT RIVER

O'Donnell, J., W.F. Bohlen, M.M. Howard-Strobel and D. Cohen, Department of Marine Sciences, University of Connecticut, Groton, CT 06340

We report the results of an analysis of observations of current meters in the lower Connecticut River and in the adjacent Long Island Sound. We find that the region has strong tides (~70 cm/s), strong residual motion (~10 cm/s), and large spatial variations in both the tidal and the residual velocity fields. We confirm that tidal currents are principally semidiurnal and report the amplitude and phase variations. The subtidal frequency variability is more energetic in the winter-spring period and we attribute this to meteorological forcing by local and remote winds. The anticyclonic pattern of the residual circulation around Long Sand Shoal is surprisingly consistent with previous inferences. Since the discharge of the Connecticut River is the dominant source of fresh water (and associated material) in LIS, its dispersion and fate is of critical importance to the understanding and management of the Sound. Recent theoretical work has demonstrated that the combination of strong tides and complex patterns of strong residual circulation result in vigorous stirring (Lagrangian dispersion). We argue that this may be the most important mechanism of effluent transport from the Connecticut River.

A DEMONSTRATION OF THE LONG ISLAND SOUND ENVIRONMENTAL STUDIES CD-ROM - U.S. GEOLOGICAL SURVEY OPEN-FILE REPORT 98-502

Poppe, L.J. and C.F. Polloni, Coastal and Marine Geology Program, United States Geological Survey, 384 Woods Hole Road, Woods Hole, MA 02543

ABSTRACT

The Coastal and Marine Geology Program of the U.S. Geological Survey has produced a CD-ROM that contains an archive of sidescan sonar, high-resolution seismic-reflection, bathymetric, sediment (texture and geochemistry), ecologic, and bibliographic data and interpretations from Long Island Sound. These data and interpretations were collected, compiled, and produced in cooperation with the State of Connecticut, Department of Environmental Protection, Geological and Natural History Survey; the University of New Haven, the U.S. Environmental Protection Agency; and the U.S. Army Corps of Engineers. The full-resolution continuous-coverage sidescan sonar mosaics, which are provided on the CD-ROM with geologic interpretations, can serve as base maps for subsequent sedimentological, geochemical and biological observations because precise information on environmental setting is important for selection of sampling sites and for accurate interpretation of point measurements. The sediment database contains over 14,600 records of texture and organic carbon; the geochemical database contains over 220 records of *Clostridium perfringens* and trace metal data; and the bibliographic database contains almost 2200 citations. Bathymetric data sets include a topographic grid, contours from NOAA charts, and a computer generated fly-over. ESRI's ArcExlorer, a supplied mapping tool, allows users to navigate the maps and data as GIS coverages. The goal of this project was to provide a regional synthesis of the offshore surficial geology and oceanography of Long Island Sound to support a wide variety of management decisions and to provide a basis for further process-oriented investigations. The data and interpretations presented on the CD-ROM serve many purposes including: (1) defining the geological and bathymetric variability of the sea floor, two of the primary controls of benthic habitat diversity; (2) improving our understanding of the processes that control the distribution of bottom sediments and benthic habitats; and (3) providing raw data and a detailed framework for future research, monitoring, and management activities.

MODELING TIDE, WIND, WAVE AND DENSITY-DRIVEN BOTTOM CURRENTS IN LONG ISLAND SOUND

Signell, R.P., J.H. List, H.J. Knebel and A.S. Farris, United States Geological Survey, Woods Hole, MA 02543

Currents at the bottom of Long Island Sound play a large role in determining the distribution of sedimentary environments. The response of the Sound to tidal forcing, typical winter storms, and the along-axis salinity gradient has been investigated using the Blumberg-Mellor 3D circulation model and the HISWA wind-wave model. Toward the eastern end of the Sound, the model shows strong tidal currents that correlate with large regions of sediment erosion and transport. The modeled tidal currents decrease rapidly toward the west away from the mouth, and areas of relatively weak currents correspond to broad depositional environments in the central and western parts of the Sound. In the axial depression that runs through the central and western Sound, estuarine circulation and circulation associated with the prevailing westerly wind locally enhance the incoming tide, providing an explanation for areas of relatively coarse sediment found in the depression. The wave model indicates that because the Sound is well protected from incoming sea swell and locally generated wind waves are limited by the short fetch, resuspension of fine-grained sediments due to surface waves should be limited to the shallow periphery of the Sound (water depths less than about 15-20 m). This provides an explanation for the relatively coarse material found along the margins of the Sound. The strong correlation between the distribution of modeled bottom currents and the distribution of sedimentary environments provides a framework for predicting the long-term effects of anthropogenic activities.

PLANKTONIC PRODUCTIVITY, RESPIRATION AND CARBON CYCLING IN WESTERN LONG ISLAND SOUND AND LOWER HUDSON RIVER ESTUARIES

Taylor, G.T., Y. Yu, J. Way and M.I. Scranton, Marine Sciences Research Center, State University of New York, Stony Brook, NY 11794-5000

Although Western Long Island Sound (WLIS) and the lower Hudson River Estuary (HRE) are two heavily urbanized adjoining estuaries, their planktonic community dynamics, carbon cycling and dissolved oxygen dynamics appear dissimilar. Differences are manifested, for example, by hypoxia, which is a predictable phenomenon in WLIS most summers and an infrequent problem in the HRE in the recent past. Much of the variance between these systems can be explained by circulation and nutrient loadings. Our study compares seasonal changes in plankton and nutrient dynamics in these estuaries and attempts to understand the mechanisms that control their variability. Four stations in WLIS and lower HRE were sampled at three depths on 12 cruises from October 1996 to September 1998. Hydrography, turbidity, inorganic and organic nutrients, primary production, chlorophyll concentrations, bacterial production and abundances, turnover of selected radiotracer substrates, hydrolytic exoenzyme activity and biological oxygen demand (BOD) were characterized at each station. As previously reported, primary productivity appears largely controlled by light attenuation. Controls of bacterial production and BOD are less apparent. Bacterial production (in mg C m^{-2} d^{-1}), while coherent with trends in primary production, often exceeds it; demonstrating importance of allochthonous input. BOD was highly correlated with water temperature.

SPATIAL AND TEMPORAL VARIABILITY OF HYPOXIA IN THE THAMES RIVER ESTUARY IN 1997

Weiss, H.M., and S. Yarish, Project Oceanology, Avery Point, Groton, CT 06340 and J. Kremer, Department of Marine Sciences, University of Connecticut, Groton, CT 06340

The onset of hypoxia (<3.0 ppm D.O.) in the Thames River occurred in early June at the bottom of Norwich Harbor and gradually spread upward in the water column and then down river. The hypoxic zone generally increased throughout July and August extending throughout all or most of the water column below the halocline from mid-August to mid-September south past Trading Cove. The extent of hypoxia declined at the end of September but Norwich Harbor bottom waters remained hypoxic through November. Fluctuations in the extent of hypoxia were relatively small compared with other years, probably due to low rainfall and river flow. Temperature and oxygen data suggest that the sill at the southern end of Norwich Harbor may prevent bottom water from flushing out of the basin except during high river flow or exceptionally high tide conditions. If so, the water beneath the sill may stagnate. The extent and severity of the depressed oxygen levels found in 1997 is comparable to that found in previous years, including surveys conducted prior to the installation of Pfizer's new waste water treatment facility. These data do not indicate any improvement in the hypoxic conditions occurring in Norwich Harbor which can be attributed to the substantial reduction of BOD from Pfizer. Thus, other contributors to the hypoxia may have been and continue to be more significant than Pfizer wastewater. Hypoxia and depressed oxygen levels in the upper Thames always occur below the halocline with the oxycline usually located at the halocline. This suggests that decomposition is occurring at this level, perhaps of phytoplankton from surface waters sinking to the top of the saltier water mass below. Processes occurring at and just below the halocline may make a significant contribution to the hypoxia and should be studied at a fine vertical scale.

DISSOLVED INORGANDIC NITROGEN AND HYPOXIA IN THE THAMES RIVER ESTUARY

Yarish, S., H.M. Weiss, Project Oceanology, Avery Point, Groton, CT 06340; J. Kremer, Department of Marine Sciences, University of Connecticut, Groton, CT; R. Jadamec and C. Anderson, Coastal Environmental Laboratory, University of Connecticut, Groton, CT 06340

Water quality surveys were conducted from May-December 1997 in the Thames River estuary from Long Island Sound to Norwich Harbor and in the Yantic and Shetucket Rivers which meet in Norwich to form the Thames. The lower Thames River had consistently low dissolved inorganic nitrogen concentrations while Norwich Harbor was enriched in inorganic nitrogen. The water column in Norwich Harbor was highly stratified. The well oxygenated, low salinity surface water above the halocline was enriched in nitrate and nitrite (NO_X-N) while the high salinity, low dissolved oxygen water located below the halocline was enriched in NH_3-N(>0.5 mg/l). The NH_3-N maximum and oxygen minimum often occurred simultaneously just below the halocline. High surface NO_X-N concentrations in Norwich Harbor are due in part to upstream sources since stations above Norwich Harbor had NO_X-N concentrations equal to or above those in the Harbor. Surface NO_X-N concentrations decreased consistently down river to low values (0.2 mg/l) in Long Island Sound. This trend could be caused by a combination of dilution and absorption by algae as the water moves down river. Surface and bottom NH_3-N concentrations were low at all stations other than Norwich Harbor. Enriched NH_3-N water is confined to the sub-halocline water in Norwich Harbor indicating that biological or chemical processes are forming the NH_3-N in the water column and/or sediment and may be linked to the formation of hypoxia.

BACTERIAL TURNOVER OF SELECTED LOW MOLECULAR WEIGHT ORGANIC MOLECULES IN WESTERN LONG ISLAND SOUND

Yu, Y., M.I. Scranton and G.T. Taylor, Marine Sciences Research Center, State University of New York, Stony Brook, NY 11794-5000

It is well known that DOC in aquatic ecosystems represents a large pool of energy for heterotrophic microorganisms. Since bacteria can only use Low Molecular Weight Organic Matter (LMWOM) directly, we decided to assess DOC cycling using labile monomers. For this study, the contribution of metabolism of LMWOM (acetate, glucose and glycolate) to heterotrophic bacterial activity and to the flux of DOC were measured in western Long Island Sound and at three stations in the Hudson River. Stations were occupied on 12 cruises between October 1996 and September 1998. Both concentrations and turnover rate constants were measured for the labile monomers. The acetate concentration in western Long Island Sound was generally in the range of 0 to 2 μM, with the highest concentration (8.4 μM, in bottom water) found in early summer when the water column became quite stratified and bottom water turned hypoxic. Acetate uptake rate constants varied from 0.6 to 3.6 d^{-1}, and uptake rates varied from 0-4.8 μM d^{-1} with the highest rates found in July coupled with the highest primary production, bacterial production and bacterial respiration. Glucose and glycolate concentrations were measured on fewer cruises, and were all in the nanomolar range. Uptake rate constants of those two monomers ranged fro 0.02 to 3.5 d^{-1} and were higher in summer. A comparison was made between western LIS and the lower Hudson River estuary. It showed that in western LIS the carbon source for bacteria was mainly from phytoplankton; and the uptake of monomers correlated well with PP, BP and total bacterial respiration, while in the Hudson, allochthonous carbon input was more important to heterotrophic bacterial activity.

POSTER

ABSTRACTS

POLLUTION OF THE MARINE SURFACE FILM (GELBSTOFF) IN LONG ISLAND SOUND AND THE HUDSON RIVER

Bush, B. and L. Karam, Wadsworth Center, New York State Department of Health, P.O. Box 509, Albany, NY 12201

The marine surface film stands as a transport barrier between the water and air phases of the environment. Pollutants which would be transported into the water phase slowly because of low affinity for water can be trapped by the film for subsequent entry into the aquatic environment. Using a 1 m^2 stainless steel gauze frame, the film has been sampled repeatedly at one site to determine the extent of variation of composition with atmospheric and tidal conditions. The organic extract was analyzed for polynuclear aromatic compounds and their derivatives, with gas chromatography linked to Fourier transform infrared and mass spectral detection. This technique allows 50 ng of a chromatographic zone to be identified by two spectroscopic methods simultaneously (Bush and Barnard, 1994) so that a clear picture of surface film composition may be achieved. The results of a brief survey of the Hudson River estuary and western Long Island Sound will be presented. Probably contributors to the film pollution are exhaust from diesel and two cycle motors, atmospheric fall out of urban pollution, and various industries such as aluminum smelting. The high chromatographic resolution to be employed for this analysis (Choudhury and Bush, 1981) will allow these different sources to be discriminated.

Choudhury, D.R. and B. Bush. 1981. Chromatographic-spectrometric identifications of airborne polynuclear aromatic hydrocarbons. *Analytical Chemistry 53*: 1351-1356.

Bush, B. and E.L. Barnard. 1994. Gas phase infrared spectra of 209 PCB congeners using gas chromatography with Fourier transform infrared detection: internal standardization with a ^{13}C-labelled congener. *Archives of Environmental Contaminant Toxicology 29*: 322-326.

POPULATION BIOLOGY OF DIAMONDBACK TERRAPINS IN THE LOWER HOUSATONIC RIVER

Gallowitsch, M. and R. Chambers, Biology Department, Fairfield University, Fairfield, CT 06430

The nesting habitats and adult population structure of diamondback terrapins in a marsh on the lower Housatonic River were determined during summer 1998 to establish baseline information on this important estuarine species. The primary method of sampling adult terrapins in the Nell's Island marsh was via net capture, using 50' and 75' trammel nets strung across sections of creeks where terrapins had been observed by canoe survey. Mark and recapture techniques were used to generate estimates of: (1) total population size, (2) terrapin age and size structure and (3) terrapin sex ratios. Terrapin nesting potential was based on behavioral observations of terrapins and measurements of habitat quality in the marsh complex, including the recently restored section of upland nesting habitat at Milford Point. Perhaps more than for any other estuarine species, the cumulative effects of habitat degradation have the potential for negatively impacting the success of diamondback terrapins. This study of diamondback terrapins of the lower Housatonic River system will provide a benchmark for population size and structure and will allow a fuller understanding of the potential for maintenance and growth of terrapin populations in the face of habitat alteration.

POPULATION DYNAMICS OF EELGRASS (*ZOSTERA MARINA L.*) IN EASTERN LONG ISLAND SOUND FROM 1985 TO 1996

Keser, M., J.M. Vozarik, J.F. Foertch and J.T. Swenarton, Northeast Utilities Environmental Laboratory, P.O. Box 128, Waterford, CT 06385

This long-term monitoring study, conducted from 1985 to 1996, was designed to investigate the effect of a thermal effluent from a nuclear power station on nearby eelgrass (*Zostera marina L.*) populations. Monthly population sampling, conducted primarily in summer (June-September), revealed reductions in size of individual beds as well as declines in shoot density, shoot length and standing stock at all three study sites. Also, percentage of reproductive shoots declined and the reproductive season became shorter over the study period. Physical environmental factors including sediment characteristics (mean grain size, silt/clay and organic content), water temperature and salinity could not account for the observed trends, nor could effects of power plant operation, since the most pronounced population changes occurred at the study site in the Niantic River, well beyond any influence of the discharge thermal plume. We were able to associate several short-term population declines with fouling by blue mussels (*Mytilus edulis*), shading from an algal bloom (*Cladophora* spp.), and possibly infection by the net slime mold *Labyrinthula* spp., waterfowl grazing, sediment freezing, sand movement and disturbance by a hurricane. *Zostera* declines observed in this study are similar to trends noted by other researchers from New Hampshire to Chesapeake Bay. Also noted in this study was the ability of *Zostera* to successfully recolonize areas through seed dispersal from nearby healthy populations.

INTEGRATING RESEARCH, MANAGEMENT AND EDUCATION: THE CONNECTICUT AUDUBON COASTAL CENTER (CACC) MODEL

Milton, B. and R. Julian, Connecticut Audubon Coastal Center, Milford, CT 06460 and J. Tait, Earth Science Department, Southern Connecticut State University, New Haven, CT 06415

It has become apparent to researchers that effective communication of their results and methods to the concerned public is a necessary part of their job, not only in terms of obtaining funding but in terms of promoting appropriate policy measures in response to research findings. Finding effective means of doing this can be problematic. Environmental advocacy groups and educators, on the other hand, work with land use managers and the public to facilitate informed decision making and to develop behaviors that conserve open space, wetlands, clean water and wildlife habitat. Communication of research results in accessible form is key to both enterprises. As part of its educational mission CACC seeks to disseminate up-to-date, multidisciplinary coastal research to advocacy groups, educators and environmental managers. CACC at Milford Point is one of 5 facilities of the CT Audubon Society whose primary mission is environmental education; CACC is located on an 84-acre spit of land at the mouth of the Housatonic River. A LIS barrier beach borders the south side of the spit. A major salt marsh (840 acres) is still forming on the other side. Pursuant to its mission of raising awareness of the Long Island Sound ecosystem and foster its preservation, CACC encourages research at Milford Point with the objective of integrating that research into its education programs and management plans. Thirty-six scientists from area universities and government agencies are working together on the Milford Point Ecosystem Project and are participating directly or indirectly in CACC's education programs. These programs include adult lectures and field studies, workshops for municipal officials, watershed workshops for high school teachers, and field and lab studies for middle and high school students. Research interns interact with staff and design exhibits and curriculum concerning CACC research.

UTILIZATION OF INTERTIDAL AND MARINA HABITATS BY JUVENILE WINTER FLOUNDER, *PLEURONECTES AMERICANUS*

Mroczka, M.E., Cedar Island Marine Research Laboratory, P.O. Box 181, Clinton, CT 06413; J.K. Carlson, National Marine Fisheries Service, Southeastern Fisheries Center, 3500 Delwood Beach Road, Panama City, FL 32408; T.A. Randall, Department of Biology, University of Mississippi, University MS 38677 and P.E. Pellegrino, Department of Biology, Southern Connecticut State University, New Haven, CT 06515

Creation of marinas involves removal of integral parts of the existing ecosystem such as salt marshes and intertidal flats which function as nursery habitat for juvenile fish. Yet, preliminary studies on marinas have suggested that they do not totally displace juvenile fish. To evaluate this suggestion, the relative abundance of juvenile winter flounder, *Pleuronectes americanus,* was compared in two areas: a marina basin and an adjacent intertidal habitat. Winter flounder were sampled with a 1-meter beam trawl monthly from March through November 1990-1995. Both habitats were dominated by young-of-the-year and age 1+ fish. We found no significant difference in the relative abundance of flounder among habitats for all combined years sampled. The average density was 0.04 ± 0.06 flounder/m^2 within the marina and 0.03 ± 0.04 flounder m^2 within the intertidal flat. We found seasonal variation in abundance with highest number caught during the summer months (June-August) and lowest during spring (March-May). The results of this study suggest that young-of-the-year winter flounder are equally abundant in both natural intertidal habitats and marina basins, indicating that both could serve as nurseries. However, more specific research is required to resolve the importance of marinas and the factors involved in the utilization of each habitat.

DISTRIBUTION AND DENSITY OF SUBMERGED AQUATIC VEGETATION BEDS IN A CONNECTICUT HARBOR

Mroczka, M.E., Cedar Island Marine Research Laboratory, P.O. Box 181, Clinton, CT 06413; T.A. Randall, Department of Biology, University of Mississippi, University, MS 38677; J.K. Carlson, National Marine Fisheries Service, Southeastern Fisheries Center, 3500 Delwood Beach Road, Panama City, FL 32408

Submerged aquatic vegetation (SAV), *Zostera marina* and *Ruppia marina*, was surveyed and mapped for an inventory of inner Clinton Harbor, Clinton, CT. Transects set at 30 meter intervals were established along the northern shoreline of the inner harbor, and SCUBA was utilized to count SAV short shoot densities. Surveys revealed that the majority of the inner harbor was dominated by low density grassbeds. Nine areas of high density and ten areas of medium density grassbeds were located. No SAV was found in the navigational channel or upon the mudflat along the northern shore of the inner harbor.

WATER QUALITY MONITORING: AN EDUCATION PROJECT IN NEW HAVEN HARBOR

Payne, D.L., Schooner, Inc., New Haven, CT 06519; S.E. Yergeau, Save The Sound, Inc., Stamford, CT 06092

The most pressing issue facing New Haven Harbor and Long Island Sound is the general lack of accurate knowledge regarding the environmental conditions of the waters. Reliable information on the state of New Haven Harbor waters is scarce, as water quality studies have been few and far between. The lack of knowledge and accessible data to change the existing perception are the leading factors contributing to the general misconception of health and lack of appreciation for the harbor's waters. In an effort to change the existing public perception of New Haven Harbor and promote research and long-term trend analysis, Schooner, Inc. and Save the Sound, Inc. have developed a model program to obtain credible water quality data and educate the people of greater New Haven about water quality. The project includes the implementation of Save the Sound's Adopt-A-Harbor program, a volunteer research program to monitor the water quality of harbors in Long Island Sound. Additional aspects of the program include teacher workshops on water quality monitoring, and education programs involving students in research, monitoring, and data preparation and presentation.

THE LIVING SALTMARSH: AN INTERACTIVE GUIDE TO THE INVERTEBRATES OF CONNECTICUT'S TIDAL WETLANDS

Pellegrino, P.E. and P. Festa, Southern Connecticut State University, New Haven, CT and M. Mroczka, Cedar Island Marine Laboratory, Clinton, CT

The Living Saltmarsh is a computer-driven, interactive multimedia display that incorporates live video footage of invertebrate organisms along with their associated saltmarsh habitats. It focuses on the environmental fragility of wetland habitats along with the importance of invertebrates in saltmarsh food chains. It allows students to learn in a dynamic and interesting way about the functional importance of wetland habitats and to appreciate the beauty and significance of invertebrate animals. Students will be able to view living saltmarsh animals over and over again without having to collect and kill additional specimens. This project allows students to jump quickly from marsh habitat to habitat and from species to species. Students will control the flow of information rather than being passive participants. Students feel like they are actually on a saltmarsh just by sitting at the computer. This project allows students to see magnified close-up views of living saltmarsh invertebrates with their natural colors rather than just looking at preserved specimens.

WATER QUALITY AND ASSOCIATED LAND USE IN TWO SMALL, ADJACENT COASTAL CONNECTICUT WATERSHEDS

Penniman, C.A., M.A. Costa and V.A. Kruse, Department of Biological Sciences, Central Connecticut State University, New Britain, CT 06050

Water quality conditions were assessed in two adjacent coastal Connecticut watersheds: the Lieutenant and Black Hall rivers (Old Lyme/Lyme, CT), over a ten-month period (May 1996 through February 1997). Chemical and physical parameters were measured at approximately biweekly intervals (*i.e.*, every two weeks) at a total of 15 stations in the tidal and freshwater segments of the two watersheds. Additionally, several transect studies were conducted for the tidal reaches of both rivers and the lake present in one watershed. Land use in the two watersheds was assessed using ArcView GIS. Dissolved oxygen concentrations were generally above state water quality criteria for Class A waters with several exceptions during the summer months. A major exception to the generally high DO concentrations was the development of extensive anoxic conditions in the hypolimnion (below 6-7 m depth) in Rogers Lake (part of the Lieutenant River Watershed) during the later summer to early fall of 1996. Bacterial concentrations (as fecal coliforms, mFC, and enterococci, mE) were generally low during the early spring and later fall-winter. However, very high counts for both indicators occurred consistently at most sampling stations during the warmer months (*i.e.*, June through September).

SEDIMENTARY ENVIRONMENTS OFF ROANOKE POINT, NY, IN SOUTHEASTERN LONG ISLAND SOUND

Poppe, L.J., H.J. Knebel, R.P. Signell, J.H. List and M.R. Buchholtz ten Brink, United States Geological Survey, Woods Hole, MA 02543; R.S. Lewis and M.L. DiGiacomo-Cohen, Long Island Sound Resource Center, Connecticut Department of Environmental Protection, University of Connecticut, Groton, CT 06340

Acoustic surveys (sidescan, subbottom and bathymetric), sediment sampling and bottom photography were used to map depositional environments around the Roanoke Point shoal, one of a series of shoreface-attached arcuate shoals located along the north shore of Long Island east of Port Jefferson, NY. This shoreline is characterized by bluffs, which are an expression of the Harbor Hill Moraine, and by gently curved beaches separated by headlands, which project slightly into the Sound. Sediments eroded from the bluffs are transported eastward by a strong wave-driven littoral drift to headlands where they are deflected offshore. Coarser grained sediments accumulate on the shoals; finer grained sediments are deposited in lower energy environments farther offshore and in the areas between shoals. Tidal constriction by the shoal results in a steep shoal face, and isolated bathymetric depressions and thinning of the Holocene marine section on the adjacent basin floor. A stronger flood tide seaward of the shoal results in a net westward transport, as evidenced by asymmetrical obstacle marks on sidescan images. This residual flow is strong enough to sweep fine sands off the shoal face, which are subsequently deposited on the the basin floor northwest of the shoal, and to permit transport by eddies typically associated with headlands. Unlike most other shoreface-attached shoals along the Atlantic coast, the Roanoke Point Shoal and other shoals along the north shore of Long Island have a hooked, asymmetrical topography that is steeper on the up-current flank (in the near-shore regime) and gentler and open on the eastern or down-current slope. This difference in morphology is probably related to the abundant sediment supply and to the narrow nearshore zone.

GEOMORPHIC EVOLUTION OF THE CHARLES E. WHEELER WILDLIFE SANCTUARY AT MILFORD POINT, CT

Tait, J., Earth Science Department, Southern Connecticut State University, New Haven, CT 06515 and S. Belden, Connecticut Audubon Coastal Center, Milford, CT 06460

The Charles E. Wheeler Wildlife Sanctuary in Milford is a relatively young, (100-150 yrs old), unditched salt marsh located at the mouth of the Housatonic River. It evolved from a shallow bay in the later part of the 19th century into one of the most extensive (840 acres) salt marshes on the Connecticut shore. Early maps and land use history records indicate that marsh development may have been fostered by dredging of the lower Housatonic and construction of a breakwater. This ongoing documents the marsh's geomorphic evolution and assesses contemporary geomorphic processes. Historic maps, digitized aerial photos and field measurements are used to obtain data on decadal geomorphic changes. We address several questions: Has a dynamic equilibrium been achieved? How long did it take for this state to be achieved? If the marsh is not geomorphically stable, what types of changes are taking place, at what rates and what are the driving mechanisms? Preliminary results indicate that there are regions of relative stability and regions that are still dynamic. The central marsh appears to have changed little since 1965. Portions of the marsh immediately adjacent to the Housatonic, however, are undergoing bank erosion while the sand spit that separates the marsh from the LIS appears to still be accreting. One of the most interesting developments is the formation of new marsh along the seaward edge of the Milford Point spit. Formation of extensive nearshore sand bars has created conditions favorable to deposition of fine-grained sediments and colonization by *Spartina alterniflora*.

PATTERNS OF BACTERIOPLANKTONIC EXOENZYME ACTIVITY IN WESTERN LONG ISLAND SOUND AND LOWER HUDSON RIVER ESTUARY

Way, J.L., M.I. Scranton, G.T. Taylor and Y. Yu, Marine Sciences Research Center, State University of New York, Stony Brook, NY 11794

Before macromolecular organic matter is assimilated by bacteria, it must be hydrolyzed by exoenzymes into monomers. Measurement of exoenzyme activity using fluorogenic substrate analogs provides information about which classes of organic polymers are metabolized by bacteria and about the diverse composition of organic matter pools in western Long Island Sound (WLIS) and lower Hudson River estuary (HRE). On 12 cruises between October 1996 and September 1998, hydrolytic activities of β-glucosidases, peptidases, chitinases, lipases and alkaline phosphatases were measured at four stations in WLIS and HRE. Fluorescence of the samples was measured with fluorogenic methylumbelliferyl (MUF)-compounds and a methylcumarinyl amide (MCA). Initial results suggest that lipase and peptidase activity were most prevalent throughout the year at the WLIS station and activity patterns among the three HRE stations varied seasonally. Experiments were conducted to determine whether competition occurred between fluorogenic substrate analogs and the naturally occurring substrates cellulose, chitin and serum albumin. Decreases of hydrolysis rates of β-glucoside in the presence of cellulose were most pronounced and were least pronounced for peptidase in the presence of serum albumin. Data from measurements of bacterial production, primary production, chlorophyll, turnover of selected monomers and various environmental parameters will be compared to the exoenzyme activity data. Maximum hydrolysis rates are expected to correspond to times of high bacterial biomass and production, as well as to chlorophyll-a content.